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ENVIRONMENTAL SUSTAINABILITY ASSESSMENT OF SOYBEAN AND PALM BIODIESEL SYSTEMS: A LIFE-CYCLE APPROACH

PhD thesis in Sustainable Energy Systems, supervised by Professor Fausto Miguel Cereja Seixas Freire, presented to the Department of Mechanical Engineering of Faculty of Sciences and Technology of the University of Coimbra

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ENVIRONMENTAL SUSTAINABILITY ASSESSMENT OF SOYBEAN AND PALM BIODIESEL SYSTEMS: A LIFE-CYCLE APPROACH

Érica Geraldes Castanheira

PhD Thesis in Sustainable Energy Systems

In the context of the Energy for Sustainability Initiative of the University of Coimbra and MIT-Portugal Program

Supervisor: Professor Fausto Miguel Cereja Seixas Freire

Coimbra, 2014

É FOGO

Éramos uma pá de apocalípticos,
De meros hippies, com um falso alarme...
Economistas, médicos, políticos
Apenas nos tratavam com escárnio.
Nossas visões se revelaram válidas,
E eles se calaram mas é tarde.
As noites tão ficando meio cálidas...
E um Mato Grosso em chamas longe arde
O verde em cinzas se converte logo, logo...
É fogo! É fogo!
Éramos uns poetas loucos, místicos
Éramos tudo o que não era são;
Agora são com dados estatísticos
Os cientistas que nos dão razão.
De que valeu, em suma, a suma lógica
Do máximo consumo de hoje em dia,
Duma bárbara marcha tecnológica
E da fé cega na tecnologia?
Há só um sentimento que é de dó e de
Malogro...
É fogo... É fogo...
Doce morada bela, rica e única,
Dilapidada só como se fôsseis
A mina da fortuna econômica,
A fonte eterna de energias fósseis,
O que será, com mais alguns graus celsius,
De um rio, uma baía ou um recife,
Ou um ilhéu ao léu clamando aos céus, se os
Mares subirem muito, em Tenerife?
E dos sem-água, o que será de cada súplica,
De cada rogo
É fogo... é fogo...
Em tanta parte, do ártico à Antártida
Deixamos nossa marca no planeta:
Aliviemos já a pior parte da
Tragédia anunciada com trombeta.

Lenine, Brazilian musician

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ABSTRACT

In Portugal more than 50% of biodiesel is produced from imported soybean and palm oil. The increasing global consumption of these commodities for bioenergy purposes has been accompanied by a growing concern about their impacts, including the potentially high environmental impacts associated with intensive land-use practices and land-use change (LUC). This thesis presents an environmental sustainability assessment of biodiesel systems. A framework was developed and implemented for various biodiesel chains, which aims to contribute to the Life-Cycle (LC) modeling of multifunctional bioenergy systems. Critical modeling issues were addressed and assessed through LC models applied to different chains, pathways and scenarios for biodiesel produced from soybean and palm cultivated in South America.

Detailed LC modeling and inventories were implemented for three biodiesel chains: A) Biodiesel produced in Portugal based on palm oil imported from Colombia, B) Biodiesel produced in Portugal based on soybean imported from Brazil and Argentina and C) Biodiesel produced in Brazil and Portugal based on soybean produced in four Brazilian states (Mato Grosso, Goiás, Paraná and Rio Grande do Sul). The influence of the location of the oil extraction and biodiesel production mills was also assessed in chain C. The LC phases included were: LUC, cultivation, oil extraction and refining, biodiesel production and transportation. A sensitivity analysis of alternative multifunctionality procedures for dealing with co-products (allocation and substitution) was performed. Two LCIA methods (ReCiPe and CML) were adopted. The toxicity impacts of soybean biodiesel were also calculated based on ReCiPe and USEtox methods.

A comprehensive evaluation was carried out of the implications of alternative scenarios, namely LUC scenarios established on the basis of a combination of alternative previous land-uses and actual land-use (palm and soybean plantations), palm fertilization schemes (mineral and organic nitrogen fertilizers), soybean cultivation systems (tillage, reduced-tillage and no-tillage), biogas management at palm oil extraction mill (biogas released or captured and flared) and soybean transportation (road distances and types of lorry). LUC emissions were also calculated based on the expansion of the actual Colombian palm area from 1990 to 2010 and on the expansion of the actual soybean area from 1985 to 2006 in four Brazilian states. Nitrogen field emissions (N_2O, NH_3 and NO_3^-) were calculated based on

two approaches (IPCC Tier 1 and site-specific models) and a sensitivity analysis of field nitrous oxide (N_2O) emissions was performed.

The results demonstrate the importance of LUC in the GHG intensity of biodiesel, although the range is significant: the highest results were calculated for the scenarios in which tropical forest is converted into palm or soybean plantations, whereas the lowest (or negative values) were for the conversion of annual cropland (in palm plantation) and degraded grassland/savanna (in soybean plantation). Different results were obtained when expansion of the actual palm and soybean area was used in the calculation instead of generalized LUC scenarios. The GHG intensity of palm biodiesel is lower than soybean biodiesel when actual area expansion is adopted, since palm is a perennial crop (with a high carbon stock in the vegetation).

The environmental impacts of biodiesel are also greatly influenced by land-use practices, nitrogen field emission calculations, production schemes and pathways, the multifunctionality approach and LCIA method adopted. The lowest impacts of soybean cultivation were obtained when no- and reduced-tillage systems were adopted. The lowest impacts of palm cultivation depend greatly on the type of fertilizer used. Different results were obtained with the two nitrogen field emission calculation approaches. Field N_2O emissions play a major role in the GHG intensity of palm and soybean cultivation, which is very sensitive to the parameters adopted for the calculations. A huge variation in the environmental impacts from the two biogas management scenarios was obtained. The environmental impacts of soybean biodiesel chains are greatly influenced by the transportation phase. The effect of multifunctionality on the results is considerably more significant for soybean biodiesel than palm biodiesel. The impacts calculated with energy and price-based allocation were similar but higher than those obtained with mass allocation for both biodiesel chains. Adopting substitution approach led to the highest and lowest impacts in almost all categories, depending on the substitution scenario considered.

The GHG savings from replacing diesel with palm and soybean biodiesel were also assessed, with the aim of providing support for Portuguese companies in calculating and meeting the GHG saving criteria for biodiesel presented in the European Renewable Energy Directive (RED). The GHG savings calculated vary significantly and in most of the scenarios the results are different from the default GHG savings presented in the RED. The wide range of results presented in this thesis demonstrate that producing general figures for the environmental impacts of biodiesel systems is problematic and each case should be addressed individually.

Keywords: biodiesel, cultivation, fertilization, GHG intensity, land-use, land-use change, life-cycle assessment, multifunctionality, palm oil, soybean.

RESUMO

Em Portugal mais de 50% do biodiesel é produzido a partir de óleo de palma e soja importados. O crescimento mundial do consumo destes óleos vegetais para a produção de biocombustíveis, tem vindo a ser acompanhado por uma crescente preocupação relativa aos seus potenciais impactes ambientais, nomeadamente os impactes associados às práticas agrícolas e aos efeitos da expansão das áreas de cultivo. Esta tese tem como objetivo avaliar a sustentabilidade ambiental do biodiesel produzido a partir de soja e palma cultivados na América do Sul. A investigação é desenvolvida e implementada para várias cadeias de produção de biodiesel contribuindo assim para a modelação de ciclo de vida (CV) de cadeias multifuncionais de sistemas de bioenergia. Os aspetos críticos relacionados com a modelação de CV são avaliados através de uma análise de cenários.

O modelo e inventário de CV são desenvolvidos para três cadeias de produção: A) biodiesel produzido em Portugal a partir de óleo de palma importado da Colômbia, B) biodiesel produzido em Portugal a partir de soja importada do Brasil e Argentina e C) biodiesel produzido no Brasil e em Portugal a partir de soja cultivada em quatro estados brasileiros (Mato Grosso, Goiás, Paraná e Rio Grande do Sul). A influência do local onde ocorre a extração do óleo e a produção do biodiesel de soja é também analisada. As fases de CV do biodiesel incluídas nesta investigação são: as alterações do uso do solo (AUS), o cultivo, a extração e tratamento do óleo, a produção do biodiesel e o transporte de produtos. É realizada uma análise de sensibilidade a diferentes abordagens para lidar com a multifuncionalidade na cadeia de produção de biodiesel. Dois métodos (ReCiPe e CML) são adotados na avaliação de impactes de CV do biodiesel, sendo que os impactes relacionados com a toxicidade são avaliados através da utilização dos métodos ReCiPe e USEtox.

Os efeitos da utilização de diferentes cenários de produção de biodiesel ao longo do seu CV são avaliados: AUS definidas com base na conversão de diferentes usos do solo em plantações de soja ou palma, utilização de vários fertilizantes azotados no cultivo de palma, diferentes sistemas de cultivo de soja (mobilização completa, mobilização reduzida e sementeira direta), gestão do biogás produzido no tratamento dos efluentes na unidade de extração de óleo de palma (libertado para a atmosfera ou capturado) e transporte de soja (diferentes distâncias e tipos de veículos). As emissões relacionadas com as AUS são também calculadas com base em dados históricos da expansão da área cultivada com palma na Colômbia (1990-2010) e com soja nos quatro estados brasileiros analisados (1985-

2006). As emissões de azoto (N_2O, NH_3 e NO_3^-) decorrentes do cultivo são calculadas através de duas abordagens (IPCC Tier 1 e modelos baseados em dados específicos dos locais).

Os resultados demonstram a importância das AUS na intensidade de GEE do biodiesel, verificando-se porém uma grande variabilidade nos resultados: a intensidade de GEE é elevada quando florestas são convertidas em plantações de soja ou palma e é bastante baixa ou negativa quando culturas anuais são convertidas em plantações de palma e pastagens/savanas degradadas são convertidas em plantações de soja. Diferentes resultados são obtidos quando se utilizam dados históricos relativos à expansão das áreas de soja e palma, sendo que neste caso a intensidade de GEE calculada para o biodiesel de palma é mais baixa do que para o biodiesel de soja (a palma é uma cultura perene, com elevado potencial para armazenar carbono).

Os impactes ambientais do biodiesel são altamente influenciados pelas práticas agrícolas e de produção adotadas ao longo do seu CV. O método adotado na avaliação dos impactes ambientais também influencia significativamente os resultados. Relativamente ao cultivo da soja, os impactes mais baixos foram obtidos para os sistemas de cultivo com mobilização reduzida ou sementeira direta, sendo que no cultivo da palma os resultados variam para os diferentes fertilizantes utilizados dependendo das categorias de impacte. Diferentes resultados foram obtidos com as duas abordagens de cálculo das emissões de azoto decorrentes do cultivo. A intensidade de GEE do cultivo da palma e soja é altamente influenciada pelas emissões de N_2O, que por sua vez variam significativamente dependendo dos parâmetros utilizados no seu cálculo. Os impactes ambientais da extração de óleo de palma variam bastante e dependem da gestão do biogás produzido no tratamento dos efluentes. Verificou-se que os impactes ambientais do biodiesel de soja são altamente influenciados pelas emissões decorrentes do transporte. Relativamente ao efeito da multifuncionalidade nos resultados, verificou-se que este é consideravelmente maior para o biodiesel de soja do que para o biodiesel de palma. Os impactes ambientais calculados com alocação baseada no teor energético e nos preços dos coprodutos são similares, sendo ambos superiores aos resultados calculados com base em alocação mássica. O método da substituição é também adotado e demonstra que os resultados dependem muito do cenário considerado (qual o produto evitado).

A redução das emissões de GEE do biodiesel de soja e palma relativamente ao combustível fóssil de referência é avaliada de forma a apoiar as empresas portuguesas no cálculo e cumprimento dos critérios de sustentabilidade estabelecidos na Directiva Europeia das Energias Renováveis (RED). Os resultados variam significativamente e na maioria dos cenários analisados são diferentes da redução das emissões de GEE apresentadas na RED. A grande variabilidade de resultados apresentada nesta tese demonstra que a avaliação dos impactes ambientais do biodiesel é problemática e que cada caso deve ser analisado individualmente.

Palavras-chave: alteração do uso do solo, avaliação de ciclo de vida, biodiesel, cultivo de oleaginosas, fertilização, intensidade de GEE, multifuncionalidade, óleo de palma, soja.

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ABBREVIATIONS AND NOTATION

AGB: Above-ground biomass

AN: Ammonium nitrate

BGB: Below-ground biomass

CAN: Calcium ammonium nitrate

CFC-11: Trichlorofluoromethane

CH_4: Methane

CO_2: Carbon dioxide

CPO: Crude palm oil

CTU: Comparative Toxic Units

DAP: Diammonium phosphate

DB: Dichlorobenzene

EFB: Empty fruit bunches

EU: European Union

FU: Functional unit

FFB: Fresh fruit bunches

GHG: Greenhouse gas

GO: Goiás

GWP: Global warming potential

IPCC: Intergovernmental Panel on Climate Change

ISO: International Standardization Organization

K_2O: Potassium oxide

LCA: Life-cycle assessment

LCI: Life-cycle inventory

LCIA: Life-cycle impact assessment

LHV: Lower heating value

LUC: Land-use change

MT: Mato Grosso

N: Nitrogen

N_2O: Nitrous oxide

NH_3: Ammonia

NMVOC: Non-methane volatile organic compound

NO_3: Nitrate

NO_x: Nitrogen oxides (NO + NO_2)

NT: No-tillage

PKM: Palm kernel meal

PKO: Palm kernel oil

POME: Palm oil mill effluent

PR: Paraná

P_2O_5: Phosphorus pentoxide

RED: Renewable Energy Directive

RS: Rio Grande do Sul

RT: Reduced-tillage

T: Tillage

1 INTRODUCTION

1.1 RESEARCH RATIONALE

In order to reduce depletion of fossil fuels and the environmental impacts associated with energy services, particularly transportation (which is heavily dependent on petroleum-based fuels), it is crucial to implement more sustainable energy systems. Biofuels have been emerging as an alternative to meet the demand for transport fuel worldwide (REN21, 2013). Biodiesel production has been growing in the last decade and accounted for approximately 21% of the world's biofuel production in 2012, mostly produced from vegetable oils (Eisentraut, 2010; REN21, 2013; OECD/FAO, 2013). The European Union (EU) is the world's largest biodiesel producer. However, in order to meet domestic policy goals, EU imports of biodiesel and feedstock have increased significantly since 2006 (Flach et al., 2012). Approximately 20% of EU biodiesel has been produced from imported feedstock: soybean (imported as oil or grain from Argentina, Brazil and the United States of America) and palm oil (imported from Asia).

There is increasing recognition that while growth in biodiesel production offers new opportunities it also bears significant risks. In fact, the increasing global trade and consumption of biodiesel has been accompanied by a growing concern about their impacts, including the potentially high environmental impacts associated with intensive land-use practices and changes in the present land-use configurations (land-use changes) (Castanheira et al., 2014a; Castanheira and Freire, 2013; Hokazono and Hayashi, 2012; Chamberlain et al., 2011; Knudsen et al., 2010; Malça and Freire, 2009; Searchinger and Heimlich, 2009; van Dam et al., 2009; Panichelli et al., 2009; Reinhard and Zah, 2009; Scharlemann and Laurance, 2008; Fargione et al., 2008; Reijnders and Huijbregts, 2008a,b).

The Life-Cycle Assessment (LCA) methodology has been applied to investigate the environmental impacts of biodiesel chains (e.g. Castanheira et al., 2014b, Castanheira and Freire, 2013; Harsono et al., 2012; Silalertruksa and Gheewala, 2012a; Hou et al., 2011; Malça and Freire, 2006, 2009, 2010; Panichelli et al., 2009; Reinhard and Zah, 2009). However, there are substantial disagreements in current LCA studies due to differences in feedstock, land-use change and land-use practices, field emission calculation approaches and feedstock processing, as well as in the use of different multifunctionality approaches and impact assessment methods (Manik and Halog, 2012; Malça and Freire, 2011; van der Voet et al., 2010), namely:

i) The uncertainty of carbon dioxide (CO_2) soil emissions due to **land-use change** (LUC) (Siangjaeo et al., 2011; Schmidt, 2010; Smeets et al., 2009; Kendall and Chang, 2009; Fargione et al., 2008). Important environmental concerns have emerged regarding the carbon stock changes due to the LUC needed for the expansion of oil crop areas. Some studies have accounted for carbon emissions from direct LUC but a wide range of results was reported. The differences in the LUC emissions calculated are mostly related to modeling assumptions, including: a) the LUC area, b) previous land-use (climate, vegetation and soil regions), c) the time-span of oil crop plantation (e.g. 1 or 25 years), d) LUC location (Ponsioen and Blonk, 2012).

ii) The complex and controversial calculation of **nitrogen (N) field emissions** from oil crop cultivation, which is highly site specific (Reijnders and Huijbregts, 2011; Del Grosso et al., 2009; Smeets et al., 2009; Snyder et al., 2009; Panichelli et al., 2009; Reinhard and Zah, 2009; Smaling et al., 2008; Miller, 2010; Miller et al., 2006). The variation in N emissions associated with system definition and modeling choices (e.g. the cropping system, type of fertilizer applied and residues in the soil, climate conditions) have not been comprehensively addressed. Nitrous oxide emissions have been assessed in various studies but only a few studies have assessed the contribution of ammonia and nitrates emissions to the eutrophication and acidification impacts of oil crop production (Smaling et al., 2008; Achten et al., 2010; Payraudeau et al., 2007).

iii) The influence of **agricultural management practices** adopted for oil crop production (Flysjö et al., 2012; Hokazono and Hayashi, 2012; Chamberlain et al., 2011; Knudsen et al., 2010; Basset-Mens et al., 2007). Only a small number of studies have addressed alternative agricultural management systems (e.g. tillage, reduced tillage, no-tillage), material inputs and yields (Kim and Dale, 2009). Likewise, **different feedstock processing technologies, residues and wastewater management practices** should be assessed since they influence life-cycle impact assessment (LCIA) results (Lam and Lee, 2011; Stichnothe and Schuchardt, 2010; Hansen et al., 2012; Suppalakpanya et al., 2010).

iv) The place (e.g., country, region) of **origin of biodiesel, oil or crops**. Different distances, types of transport or pathways can greatly affect the life-cycle environmental impacts of biodiesel and should be comprehensively assessed (Castanheira and Freire, 2013; Prudêncio da Silva et al., 2010; Thamsiriroj and Murphy, 2009). Due to the importance of imports for the EU biofuels market, it is crucial to compare the environmental impacts of imported biodiesel and imported oil or crops used in the domestic production of biodiesel.

v) The **multifunctionality approach** adopted to deal with biofuel chain co-products. As the production of biodiesel involves the generation of several co-products (e.g., meal, glycerin), it is necessary to distribute the environmental impacts between these co-products in a meaningful and justifiable manner (The Royal Society, 2008). There are several possible multifunctionality procedures

(e.g. system expansion, substitution, allocation), which may lead to the conclusion that the choice and subsequent LCA results are arbitrary and potentially subjective (The Royal Society, 2008). A sensitivity analysis of alternative multifunctionality procedures should be conducted to evaluate the influence on the results for the various impact categories (Castanheira et al., 2014b; Malça and Freire, 2011).

vi) The **LCIA method** (Cavalett et al., 2012; Buchgeister, 2012; Landis and Theis, 2008; Dreyer et al., 2003). Although the objective of LCIA is to evaluate the potential impact of the substances emitted, different LCIA methods can lead to different results (Schmidt, 2007).

The life-cycle environmental impacts of biodiesel vary widely due to the modeling and methodological issues described previously, highlighting the need for further research on the environmental LCA of biodiesel. Additionally, the majority of LCA studies have focused on greenhouse gas (GHG) intensity, together with energy and fossil fuel use, and only a few studies have addressed a wider set of environmental impacts. This research also clarifies the role of LCA in legislation and regulation regarding the environmental sustainability of biofuels, in particular in the specific case of Portugal, where biodiesel is the only biofuel consumed in the transport sector (EurObserv'er, 2013).

1.2 OBJECTIVES AND RESEARCH QUESTIONS

The main aim of this PhD research is to present an environmental sustainability assessment of biodiesel systems. A framework was developed and implemented for various biodiesel chains, with the aim of contributing to the Life-Cycle (LC) modeling of multifunctional bioenergy systems. Critical modeling issues were addressed and assessed through LC models applied to different chains, pathways and scenarios for biodiesel produced from soybean and palm cultivated in South America. This research is based on an "applications-driven" approach and the generalization is achieved through its application to different biodiesel chains and production scenarios. A sensitivity analysis for nitrogen field emission calculations, multifunctionality approaches and life-cycle impact assessment (LCIA) methods was also carried out. A comparison was made with fossil diesel to quantify the potential GHG savings due to replacing diesel with biodiesel. To this end, six research questions were formulated and specific objectives were defined to respond to these research questions, as presented in **Table 1.1**.

Table 1.1. Deriving specific objectives from research questions.

Research question	Specific objectives
1. How can we account for the effects associated with direct LUC in the LCA of biodiesel?	1.1. Model and calculate the carbon stock changes from direct LUC resulting from the expansion of soybean and oil palm areas in South America. 1.2. Calculate N_2O emissions due to the nitrogen released by the mineralization of soil organic matter, as a result of land-use change. 1.3. Determine the influence of LUC in the GHG intensity of various scenarios for palm and soybean biodiesel production.
2. What are the land-use practices, production schemes and pathways that lead to lower impacts?	2.1. Extend the standard LCA methodology to address local aspects associated with land-use, including oil crop production in different countries, states and climate regions. 2.2. Determine the environmental impacts of alternative cultivation systems, fertilization schemes and production options. 2.3. Assess the environmental impacts of different pathways for biodiesel consumed in Portugal. 2.4. Provide a better knowledge and understanding of agricultural systems and their environmental hotspots.
3. Are the environmental impacts of biodiesel influenced by the emission calculation approach and LCIA method adopted?	3.1. Perform a sensitivity analysis for nitrogen field emission calculations. 3.2. Assess the influence of the time horizon considered for the GHG intensity calculation. 3.3. Determine which LC stages and processes contribute most to the environmental impacts of biodiesel. 3.4. Compare the LCIA results calculated using different LCIA methods and determine the extent to which the results are influenced by the method applied.
4. How does the selected multifunctionality approach influence biodiesel environmental impacts?	4.1. Perform a sensitivity analysis for alternative multifunctionality approaches. 4.2. Evaluate the influence of various multifunctionality approaches on LCA results.
5. What are the GHG emission savings when palm and soybean biodiesel replace diesel?	5.1. Assess the GHG emission savings when palm and soybean biodiesel replace fossil diesel.
6. How can the environmental sustainability of biodiesel be improved by applying the LCA methodology?	6.1. Suggest improvements for the environmental sustainability of soybean and palm biodiesel and for legislation and regulation regarding the sustainability of biofuels.

1.3 CONTRIBUTION

This PhD research contributes to advances in the environmental sustainability assessment of biodiesel produced from soybean and palm cultivated in South America. Soybean and palm oil together represent more than 60% of the world's vegetable oil consumption and they are the main biodiesel feedstock in Portugal (approximately 50-60%) (USDA, 2013a; DGEG, 2012). The research presented in this PhD thesis aims to innovate the state of the art along different interrelated lines and contribute towards:

1. Advancing LCA modeling of complex and multifunctional biodiesel systems, accounting for spatial differentiation, direct LUC, different land-use practices, production options and pathways.

2. Increasing knowledge of the major sources of uncertainty in the environmental sustainability assessment of biodiesel systems.

3. Increasing awareness of the GHG savings created by replacing fossil diesel with palm and soybean biodiesel in Europe and Portugal.

4. Supporting Portuguese companies in calculating and meeting sustainability criteria for oil crops and biodiesel.

Most of this PhD thesis is based on the following five core articles published or in review at ISI-indexed journals (abstracts, highlights and keywords for the articles are presented in **Appendix I**):

1. Castanheira, É.G., Grisoli, R., Freire, F., Garcilasso, V., Coelho, S., 2014. Environmental sustainability of biodiesel in Brazil. *Energy Policy* 65, 680–691.
http://dx.doi.org/10.1016/j.enpol.2013.09.062 **JCR® impact factor (2012): 2.743**

2. Castanheira, É.G., Acevedo, H., Freire, F., 2014. Greenhouse gas intensity of palm oil produced in Colombia addressing alternative land use change and fertilization scenarios. *Applied Energy* 114, 958-967.
http://dx.doi.org/10.1016/j.apenergy.2013.09.010 **JCR® impact factor (2012): 4.781**

3. Castanheira, É.G., Freire, F., 2013. Greenhouse gas assessment of soybean: implications of land use change and different cultivation systems. *Journal of Cleaner Production* 54, 49-60.
http://dx.doi.org/10.1016/j.jclepro.2013.05.026 **JCR® impact factor (2012): 3.398**

4. Castanheira, É.G., Grisoli, R., Coelho, S., da Silva, G.A., Freire, F., 2014. Life-cycle assessment of soybean-based biodiesel in Europe: comparing grain, oil and biodiesel import from Brazil (submitted).

5. Castanheira, É.G., Freire, F., 2014. Environmental assessment of palm oil produced in Colombia (submitted).

This PhD research also contributed to the following two articles:

6. Gülşen, E., Olivetti, E., Malça, J., Castanheira, É.G., Freire, F., Dias, L., Kirchain, R., 2014. Impact of Policy on Greenhouse Gas Emissions and Economics of Biodiesel Production. *Environmental Science & Technology* 48 (13), 7642–7650.
http://dx.doi.org/10.1021/es405410u **JCR® impact factor (2012): 5.257**

7. Figueiredo, F., Castanheira, É.G., Freire, F., 2014. Life-cycle assessment of irrigated and rainfed sunflower: implications of alternative land use change scenarios (submitted).

In addition, more than fifteen articles related to the PhD research were published in conference proceedings with scientific refereeing and nine technical (and confidential) reports were produced for five Portuguese biodiesel companies as part of a cooperation project with the Portuguese Association of Biofuel Producers (Life-Cycle Assessment of GHG emissions for soybean-based biodiesel in Portugal). The full list of publications is presented in **Appendix II**.

1.4 THESIS OUTLINE

This thesis consists of five chapters and is structured according to the research questions and objectives stated in this chapter. **Chapter 2** presents the state of the art regarding the environmental sustainability assessment of soybean and palm biodiesel, including an overview of soybean and palm biodiesel production, an introduction to the LCA framework and a description of key modeling and methodological issues. EU legislation on the sustainability of biodiesel and the main findings from a literature review of the LCA of palm and soybean biodiesel are also presented in this chapter. **Chapter 3** describes the main aspects of the methodology implemented for biodiesel chains, including the life-cycle modeling and inventories, as well as the different scenarios and modeling choices. **Chapter 4** presents the LCIA results for soybean and palm biodiesel systems and discusses the major sources of uncertainty. The environmental hotspots of soybean and palm biodiesel are identified and discussed. **Chapter 5** draws the conclusions together and presents recommendations and suggestions for further research. Finally, different options for improving the environmental sustainability of biodiesel systems are also presented.

2 ENVIRONMENTAL SUSTAINABILITY ASSESSMENT OF SOYBEAN AND PALM BIODIESEL: STATE OF THE ART

2.1 SOYBEAN AND PALM BIODIESEL PRODUCTION: CONTEXT

Biodiesel is a fuel produced from vegetable oils, waste cooking oil or animal fats (REN21, 2013; Eisentraut, 2010). About 83% of global biodiesel production comes from oilseed crops (OECD/FAO, 2013). **Figure 2.1** shows that the predominant use of vegetable oils is for food with over 80% of the market, with the industrial and biodiesel markets far behind. According to Rosillo-Calle et al. (2009), the rapid demand for vegetable oils has been sparked off by the food market rather than the industrial or biodiesel sectors.

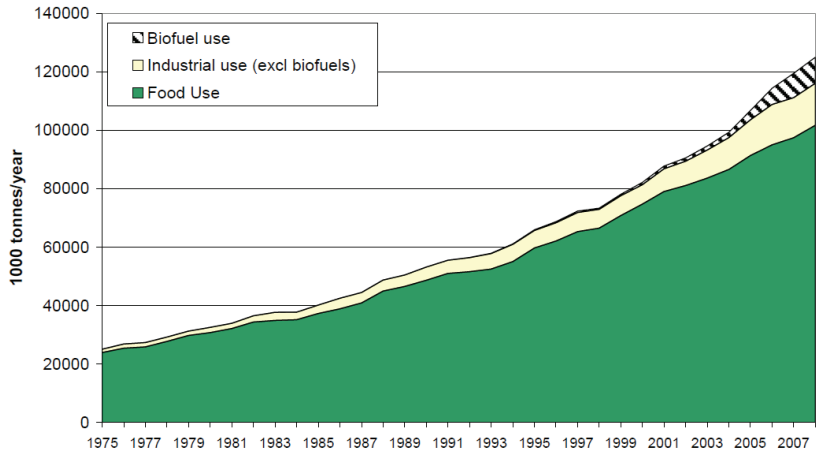


Figure 2.1. Global consumption of vegetable oils by major applications (Source: Rosillo-Calle et al., 2009).

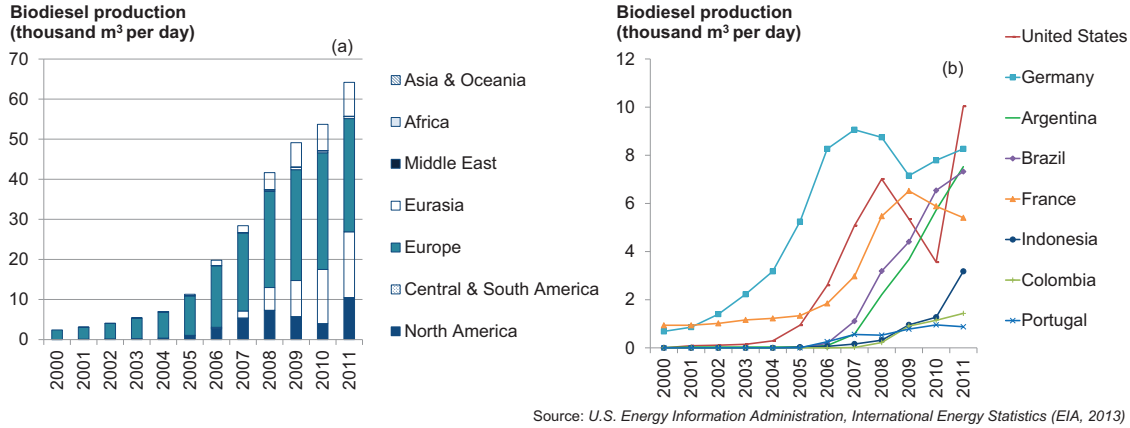
In the year 2012, the world's production of palm oil (55 million tonnes) and soybean oil (42 million tonnes) together represent more than 60% of the world's total vegetable oil production (USDA, 2013a). Palm oil is derived from the palm fruit (fresh fruit bunches, FFB) grown on the African oil palm tree (*Elaeis guineensis*), an important perennial oil crop. Other products are obtained from palm fruit milling: palm kernel oil (PKO), palm kernel meal (PKM) and byproducts or waste (e.g. shells, fibers, empty

fruits, palm oil mill effluent). Since palm fruits must be processed as soon as possible after harvesting, the distance from plantation to extraction mill must be short (the quality of the oil produced depends on the time interval between harvesting and palm oil extraction) (Lee and Ofori-Boateng, 2013).

The increase in palm oil production is being stimulated by the growing demand for food (e.g. margarine, ice cream, cooking oil) and numerous other non-food applications (e.g. biodiesel, plasticizers, paint, surface coatings) (FAO, 2006). Malaysia and Indonesia are responsible for 85% of the world's palm oil production (42 million tonnes), whereas Colombia has become the world's fourth largest producer in 2012 and the leading producer in South America (USDA, 2013b). Because of the high yield of palm oil per hectare (which exceeds the other vegetable oils) and its price competitiveness is forecast to see increased food and biofuel use (Krautgartner et al., 2013; Tan et al., 2009).

In turn, soybean oil and meal are both obtained from soybean (*Glycine max (L.) Merrill*). Soybean is an annual crop that belongs to the nitrogen-fixing leguminous plants and exists in a large number of forms and varieties. The increase in soybean production is being stimulated by the growing demand for livestock feed (soybean meal is the world's largest source of animal protein feed), food and biodiesel production (Castanheira and Freire, 2013). The United States of America (82 million tonnes), Brazil (66 million tonnes) and Argentina (51 million tonnes) were the major world's soybean producers in 2012 (FAO, 2013a).

Over the past five years, biodiesel production increased at an average annual rate of 27%, reaching over 64 thousand m^3 in 2011 (EIA, 2013). **Figure 2.2** shows the growth in biodiesel production since 2000. The columns in the graph (a) represent the contribution of the different world regions for biodiesel production and the lines in graph (b) represent the biodiesel production in the six most important countries (65% of the world's production in 2010 were in the United States, Germany, Argentina, Brazil, France, Indonesia), as well as in Colombia and Portugal.



Source: *U.S. Energy Information Administration, International Energy Statistics (EIA, 2013)*

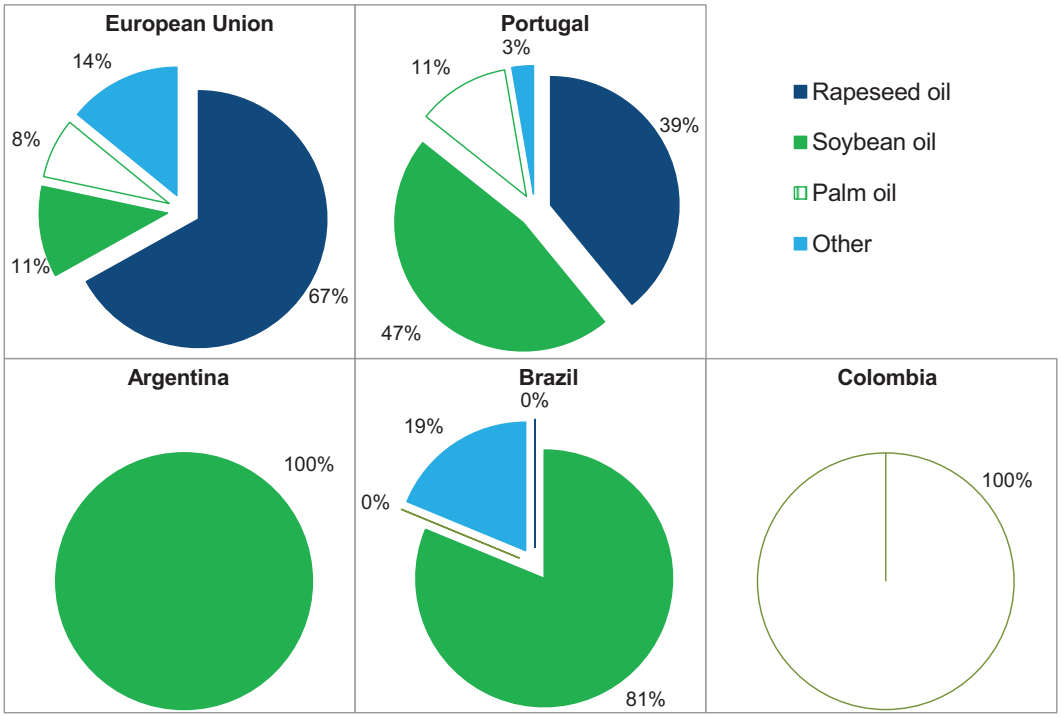
Figure 2.2. Global growth in biodiesel production (2000-2011): (a) contribution of world regions and (b) production in different countries.

Europe was the dominant region with increasing biodiesel production since 2005. As demand for biodiesel continues to increase, new plants have begun to open around the world and the representativeness of biodiesel production in Europe has been decreased, from more than 85% of the total world production to less than 50% in 2011. North America was a distant second producer led by the United States of America (USA) until 2009 when the production fell (mainly due to the economic downturn, incentives changes for biodiesel and foreign trade policies), while growth continued in Central & South America and Asia & Oceania. Central & South America has already become the second largest producing region in the world, mainly due to the biodiesel production in Argentina, Brazil and Colombia.

Figure 2.2 shows that USA was the world's leading producer in 2011, followed by Germany, Argentina and Brazil. The growth in biofuels markets has slowed in several countries in Europe (e.g. France, Portugal) in response to a number of factors, namely the economic crisis which prompted certain importing countries to reduce their incorporation level and the uncertainties surrounding forthcoming European legislation related to the sustainability criteria apply to the whole biofuel production and distribution chain (see **sub-chapter 2.4**). However, there was a striking growth in biodiesel production in Argentina, Brazil and Indonesia since biofuel blend mandates continue to drive demand (in 2011 about 23% of biodiesel consumed in EU were imported, Flach et al., 2012).

Biodiesel trade in South America countries is distinctly different. Argentina is a net exporter of biodiesel (since 2007 more than 70% of biodiesel produced in Argentina was exported), while almost all of biodiesel produced in Brazil is consumed domestically (Barros, 2013). Currently, Colombia neither imports nor exports biodiesel and in the short term, given the lack of biodiesel supply for covering the local demand, it is unlikely that exports will occur. However, in the medium term, it is expected that Colombia become an exporter of biodiesel (Pinzon, 2012).

Figure 2.3 shows the feedstock used for biodiesel production in European Union, Portugal and in the main biodiesel producers in South America (Argentina, Brazil and Colombia). Rapeseed oil is the major feedstock in the EU-27 and accounts for two thirds of total input in biodiesel production, whereas about 20% of EU-27 biodiesel have been produced from soybean (imported as oil or grain from America and mostly used in Spain, France, Italy and Portugal) and palm oil (imported from Asia) (Flach et al., 2012; Krautgartner et al., 2013). In Portugal more than 50% of biodiesel was produced from imported soybean and palm oil.



Sources: *ANP, 2013; DGEG, 2012; Flach et al., 2012; Pinzon, 2012; Joseph, 2013*

Figure 2.3. Biodiesel feedstock (2011) in European Union, Portugal and in the main biodiesel producers in South America: Argentina, Brazil and Colombia.

The EU-27 is highly dependent on imports of oilseeds and oilseeds products (protein meals and vegetable oils) to meet the demand for food, feed and biofuel production, especially oilseeds with no or limited domestic production, such as palm and soybean oil (Krautgartner et al., 2013). In the EU-27 palm oil consumed is totally imported, while soybean (more than 90% was imported from 2006 to 2009) is predominately used to produce soybean meal for the livestock feed industry. Without the protein provided by soybean, Europe would not be able to maintain its current level of livestock productivity (Krautgartner et al., 2012). In addition, more soybean meal must be imported to meet the EU-27 demand. Until 2010/11, the EU was also a net importer of soybean oil, mainly for biodiesel production; however, since 2011/12 the EU has become a net exporter of soybean oil, with exports about twice as high as imports. As a result of the implementation of the Renewable Energy Directive (see **sub-chapter 2.4**), soybean oil became more difficult to use as a feedstock for the biodiesel industry, and the EU has imported biodiesel from Argentina and Indonesia rather than imported soybean oil and palm oil (Krautgartner et al., 2013).

In year 2011, soybean oil was used for more than 80% of Brazil and Argentina biodiesel production, while in Colombia all biodiesel was produced from palm oil. There was an impressive growth in soybean production in Brazil and Argentina, mainly associated with an expansion in cultivation areas of 114% and 226% respectively during the period 1995-2012, but also due to an increase of 20% and 30% in the soybean yield associated with technological advances, management and efficiency aspects (FAO,

2013a). Colombian palm oil production in 1995 was about 388 thousand tonnes, almost doubling to 967 thousand tonnes in 2012 (FAO, 2013a). There was a production growth due to the area expansion in Colombia (area increased 46% since 1995) and the annual average yield, which increased from 16 tonnes of oil palm fruits per ha in 1995 to 23 tonnes of oil palm fruits per ha in 2012 (FAO, 2013a).

Biodiesel production is expected to steady increase in the coming years in the EU but a substantial growth is expected for South America (Brazil, Argentina and Colombia) and Asia (Rosillo-Calle et al., 2009). This motivated the investigation of the environmental impacts related to the increase production of palm oil and soybean in South America and to study the various pathways for their use as biodiesel in EU. The impacts of biodiesel produced with palm oil from Asia (see **sub-section 2.3.1**) and with rapeseed in EU have been extensively studied (Malça and Freire, 2006, 2009, 2010), whereas less published articles were found assessing a wider set of environmental impacts of biodiesel based on palm oil from Colombia and soybean from Argentina and Brazil.

2.2 LIFE-CYCLE ASSESSMENT OF BIODIESEL

The fast growing interest and production of palm and soybean oil as food and as a source for biodiesel worldwide has led to the increasing concern about the environmental impacts, especially regarding the land competition, air and water emissions (e.g. Padula et al., 2012; GEA, 2012; Janssen and Rutz, 2011; Diaz-Chavez, 2011; Lange, 2011; Lynd et al., 2011; Schaffel and La Rovere, 2010; Santos and Rathmann, 2009). In this context, it is crucial that the environmental impacts are evaluated in order to provide a rational basis for assessing the long-term viability and acceptability of individual biodiesel supply chain options (Castanheira et al., 2014a).

The environmental sustainability assessment of biodiesel systems has become an important focus of research and controversy within the scientific community, since this assessment is complex and challenging at a methodological and practical level because of many critical issues and difficulties:

i) the energy balance issue (de Souza et al., 2010; Malça and Freire, 2006);

ii) the high potential environmental impacts associated with agricultural practices (use of fertilizers and pest control techniques, full-tillage versus no-tillage systems, intensive versus extensive farming, material inputs, locations and yields) (Stichnothe and Schuchardt, 2010, 2011; Snyder et al., 2009; Malça and Freire, 2009; Kim and Dale, 2009);

iii) the uncertainties resulting from soil emissions, in particular nitrous oxide (N_2O) and carbon dioxide (CO_2) emissions due to land-use change (LUC) (Wang and Chen, 2012; Siangjaeo et al., 2011;

Erisman et al., 2010; Schmidt, 2010; Smeets et al., 2009; Crutzen et al., 2008; Kendall and Chang, 2009; Fargione et al., 2008; Malça and Freire, 2011, 2010; Soimakallio et al., 2009);

iv) the logistics and distribution networks (including biomass transport);

v) the approaches to deal with co-products of the palm and soybean biodiesel chains (Patthanaissaranukool et al., 2013; Kaewmai et al., 2012; Hansen et al., 2012; Harsono et al., 2012; van Dam et al., 2009; Huo et al., 2009).

In this context, a country-specific approach is crucial to assess the environmental impacts of biodiesel systems, since local conditions, such as agricultural practices, LUC and transport infrastructures, will have a major influence on the results (Panichelli et al., 2009).

Environmental Life-Cycle Assessment (LCA) is an internationally renowned methodology for evaluating the environmental impacts of different energy systems along its life-cycle (LC). The LCA methodology has been applied to investigate the energy and carbon balances of biodiesel chains and, in a smaller number of cases, has been used to look at wider environmental impacts (Larson, 2006). The chain modeling of the production of biomass and its use as an energy carrier must consider the whole LC: i) LUC, ii) cultivation and harvesting, iii) transport and iv) conversion of the biomass feedstock's to biodiesel and co-products. The disposal/treatment of residues and the production and use of any subsidiary inputs (such as agrochemicals, transport fuels and equipment) should also be considered.

According to ISO 14040 and 14044 standards (ISO, 2006a,b), LCA consists of four distinct phases: (1) goal and scope definition, (2) inventory analysis, (3) impact assessment and (4) interpretation. The first stage should include: the definition of goal, functional unit, system boundaries, multifunctionality procedures, assumptions and limitations, among others. In the inventory analysis, a flow model of the technical system is constructed using data on inputs and outputs of resources, energy and emissions to air and water for all activities within the system boundaries. The inventory analysis is followed by impact assessment, in which the inventory data are processed in terms of their environmental impact. Interpretation is the phase of LCA in which the findings from the inventory analysis and the impact assessment are combined together in order to reach conclusions and make recommendations (Castanheira et al., 2010).

Menichetti and Otto (2009) presented a review of the most relevant existing LCA in the area of biofuels and other environmental impact studies and indicates that: i) the majority of studies is limited to European or United States conditions, and is based on western agricultural processes and average conversion technologies; ii) few studies take into account LUC impacts driven by biofuel crop production; and iii) the transparency level of reports is quite heterogeneous with respect to data quality review and to treatment of co-products and multifunctionality approaches followed. Malça and Freire (2011) also demonstrated that LCA results of bioenergy vary quite widely, not only due to differences in data and scenarios, but also due to different normative choices in the modeling procedures. The key

modeling and methodological issues that contribute to the variability of LCA results are discussed in the following sub-chapter.

2.2.1 MODELING AND METHODOLOGICAL ISSUES

2.2.1.1 System boundaries and functional unit

The definition of system boundary is an important step in LCA. The ‘‘well-to-tank” (WtT) assessment of biofuels considers the steps required to deliver the final biodiesel into the on-board tank of a vehicle, namely biomass cultivation, processing, transportation and storage followed by biodiesel production and distribution. The “well-to-gate” (WtG) is similar to the ‘‘well-to-tank” assessment but not include biodiesel distribution. The ‘‘well-to-wheels’’ (WtW) modeling boundary includes both the ‘‘well-to-tank’’ (WtT) and ‘‘tank-to-wheels’’ (TtW) stages. The TtW assessment covers only the vehicle operation activities and can be based on data from vehicle simulation models, on-road testing, engine dynamometer experiments or fleet operation data (Malça and Freire, 2010). Cherubini et al. (2009) and Gnansounou et al. (2009) argue that WtW approach should be the first-choice in LC studies of biofuels, since different fuels may have different engine energy efficiencies. However, the ‘‘well-to-tank’’ (WtT) assessment is particularly appropriate if the goal and scope is concerned with biodiesel use as a generic energy carrier, without a particular transportation or energy conversion system being considered (Malça and Freire, 2011).

Different system boundaries can be defined depending on the scope of the study, which may also influence the choice of the functional unit. Functional unit 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 (Matheys et al., 2007; Weidema et al., 2004; ISO, 2006a). The choice of the functional unit requires special attention in order to allow comparisons between products without bias (Dias and Arroja, 2012; van der Voet et al., 2010). For instance, González-García et al. (2013) demonstrated that the choice of the best source of biomass to biogas production purposes from an environmental point of view depends on the functional unit assumed for the calculations. In the literature, we find the following functional units related to biofuels (van der Voet et al., 2010):

- Service-oriented: a specified transport distance, e.g. 1 km using the fuel in a certain type of car;
- Energy-oriented: a specified amount of energy contained in the fuel, e.g. 1 MJ;
- Mass-oriented: a specified amount of fuel produced;
- Land-area oriented: the amount produced from a certain surface of agricultural land, e.g. 1 ha.

The functional unit chosen for the application reported in this thesis is 1 MJ of biodiesel, measured in terms of the lower heating value (LHV). This functional unit is consistent with the goal and scope, which is to assess the life-cycle environmental impacts of biodiesel, used as a generic energy carrier, without a particular transportation or energy conversion system being considered.

2.2.1.2 Spatial variation: land-use change and field emissions

Biofuel feedstock cultivation currently occupies approximately 1% of arable land (Berndes et al., 2010) and the increase in land-use for biofuel production initiated a widespread debate among policy makers and researchers (Witcover et al., 2013; Ponsioen and Blonk, 2012; Lange, 2011; Walter et al., 2011; Nassar et al., 2010; Yang et al., 2009; Gibbs et al., 2008; Fargione et al., 2008). The contribution of biofuels to climate change mitigation can only be assessed if the GHG balance included LUC emissions from feedstock production (Lange, 2011). However, LUC has only been addressed recently and with limitations, since accounting for land-use in LCA is inherently problematic (Malça and Freire, 2011; Castanheira and Freire, 2013; Castanheira et al., 2014b; Milà i Canals et al., 2006; Larson, 2006).

LUC comprise both direct and indirect changes. Direct land-use change (dLUC) occurs when bioenergy crops displace a prior land-use (e.g., forest, grassland, other croplands) and indirect land-use change (iLUC) is the consequential effect from displacement of land currently used for food to fuel production (Fritsche et al., 2010). When quantifying the environmental impact of land-use, it is rather common to evaluate the GHG emissions associated with dLUC, whereas impacts associated with iLUC are less frequently assessed (Pawelzik et al., 2013). This is primarily related to the multiplicity of drivers behind iLUC, the uncertainty related to their assessment and the disagreement among experts about how to allocate the resulting impacts (Pawelzik et al., 2013). In this context, two articles published in *Science* brought the topic of iLUC caused by bioenergy to widespread attention (Fargione et al., 2008; Searchinger et al., 2008). However, there is still no sound and consensual methodology to take iLUC into account and different authors have emphasized the need for further research (Gawel and Ludwig, 2011; Kløverpris et al., 2008).

The GHG emissions due to dLUC can be determined from a comparison of the carbon balances of the previous land-use with those after the land has been used to produce biomass crops. This relates to the above-ground carbon content of the existing vegetation (if any), as well as the below-ground carbon levels, including soil carbon. Each balance might be negative or positive, so that the total direct carbon balance could also be negative or positive. Biofuel GHG emissions increase if carbon-rich land (such as peat under, rainforest) is converted for cultivation of the biomass crop; however, if feedstock are grown on low-carbon soils, the impact can be positive (Fritsche et al., 2010). For example, perennial plants such as oil palm, store carbon in their root system so that biological sequestration takes place and total GHG missions are usually reduced when dLUC is factored in and cultivation takes place on former

arable land (Brandão et al., 2011; Brandão et al., 2010; Fritsche et al., 2010). Due to the changes in the soil carbon stock (depending of the type and quality of soil), LUC also has environmental effects in the nitrogen dynamics (Reap et al., 2008).

Agriculture practices, such as nitrogen (N) amendments (e.g., fertilizer, manure), cultivation and nitrogen fixed from legume cropping (such as soybean) can increase N emissions (Del Grosso et al., 2006). Fertilizer application is one of the major components in this process leading to direct and indirect losses of reactive nitrogen in to the environment (Erisman et al., 2010). Losses can be in the form of nitrates (NO_3^-) to the groundwater, ammonia (NH_3), nitrous oxide (N_2O) and nitrogen oxide (NO_x) emissions to the air. However, the nitrogen balance is difficult to quantify because of the large variations in soil, cropping systems, plant N demand, climatic and environmental conditions and management of the fields (Reijnders and Huijbregts, 2011; Erisman et al., 2010).

N_2O emissions linked to the crop cultivation can be distinguished in direct and indirect emissions (Crutzen et al., 2008; IPCC, 2006). Direct N_2O emissions depend on the application of N fertilizers, the decomposition of crop residues and the mineralization of soil N through LUC. Indirect N_2O emissions are a function of volatilization of NH_3 and NO_x that is deposited on soils and leaching of NO_3^- that enters aquatic systems. The IPCC Tier 1 methodology (IPCC, 2006) has been adopted to calculate N emissions from palm and soybean cultivation, including fixed fractions of N that is volatilized and leached instead of site-specific data of N volatilization and leaching. Estimated life-cycle N_2O emissions can contribute substantially to the LC GHG emissions of biofuels (up to 80%) but may, however, vary by about two orders of magnitude (Reijnders and Huijbregts, 2011). Variation in N_2O emission associated with uncertainty in direct and indirect N_2O emission calculation (particularly emissions originating in the fraction of nitrogen lost via runoff, leaching and volatilization) have not been comprehensively addressed.

2.2.1.3 Multifunctionality

As the production of many biofuels involves the generation of other products (e.g., oilseed meals, glycerin), it is necessary to distribute the environmental impacts between such products in a meaningful and justifiable manner (The Royal Society, 2008). The LCA ISO standard (ISO, 2006b) presents a hierarchy of procedures to deal with co-production. Wherever possible, allocation should be avoided by i) dividing the unit process into two or more sub-processes or ii) expanding the product system to include the additional functions related to the co-products. Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the iii) physical relationships (e.g. mass, energy or carbon content) or iv) other relationships between them, such as the economic value of the products. ISO standards (ISO, 2006b)

also suggested that whenever several alternative multifunctionality procedures seem applicable a sensitivity analysis of alternative procedures shall be conducted to evaluate the influence on the results.

In the system expansion approach, the system boundaries are broaden and the function of the co-products are included. This means that a new functional unit is introduced and the LCA will no longer be about the original product, but about the original product plus a co-product (Heijungs, 2013; Azapagic, 1996). Guinée et al. (2002) added to this definition, based on Tillman et al. (1994), not to add functions but to subtract them from those alternatives providing additional functions, the so-called "substitution" or "avoided-burden" method. Several authors have argued that substitution is conceptually equivalent to system expansion (e.g. Heijungs, 2013; Ekvall and Tillman, 1997; Finnveden and Lindfors, 1998), since adding a function to a system is in some way equivalent to subtracting this function from the system. However, "equivalent" is not the same as "equal" and it does not mean that they provide the same results (Heijungs, 2013; Wardenaar et al., 2012). Nevertheless, various articles used the substitution method, claiming that ISO would recommend it and other articles which correctly claim that ISO recommends "system expansion," but then do a substitution method, putting the label of system expansion on it (Heijungs, 2013).

All multifunctionality approaches have advantages and drawbacks (Menichetti and Otto, 2009) but there is a general consensus that system expansion is more appropriate than allocation (Ekvall and Finnveden, 2001). However, system expansion (and also substitution method) have a high level of complexity since add extra processes and requires knowledge about the substituted product, as it implicitly assumes that co-products are sold on the market (Menichetti and Otto, 2009). Also, some challenges still remain since changes in product substitution are likely to occur as markets and prices for different products and co-products fluctuate. Thus, there will be inevitable disagreements on the assumptions made with regard to avoided chains and processes as well as the quantity of substitution that occurs (van der Voet et al., 2010; Kendall and Chang, 2009). Economic allocation reflects more properly the actual market conditions, but it also significantly increases the volatility of results and therefore their uncertainty (Menichetti and Otto, 2009).

Depending on the multifunctionality approach adopted, LCA results of palm and soybean biodiesel can vary widely; however, the influence of multifunctionality approach on the various environmental impact categories, other than global warming, has not been comprehensively addressed. Even though some attention has been paid to this issue, no agreement has been reached on which method should be used in biofuel policies and legislation. For example, the substitution method was advocated by the USA Renewable Fuels Standard 2010, while the energy-based allocation method was adopted in the Renewable Energy Directive (EC, 2009).

2.2.1.4 Type of LCA

LCA studies can be categorized into two general types: attributional (ALCA) and consequential (CLCA). A key decision at the goal and scope definition stage is whether an attributional or consequential approach is used (Kendall and Yuan, 2013). ALCA methodology accounts for immediate physical flows involved across the LC of a product (i.e., resources, material, energy and emissions) and uses average data for each unit process within the LC (Earles and Halog, 2011). On the other hand, CLCA aims to describe how physical flows can change as a consequence of an increase or decrease in the demand for the product system under study (Earles and Halog, 2011). Unlike ALCA, CLCA includes unit processes inside and outside of the product's immediate system boundaries and marginal data is used instead of average data (Reinhard and Zah, 2009). It utilizes economic data to measure physical flows of indirectly affected processes and allocation is avoided by expanding the system boundary (Weidema, 2003). The debate on how and when to perform ALCA versus CLCA is not yet resolved (Zamagni et al. 2008). The identification of affected technologies, collection of marginal data (i.e., which technologies will be affected and how much) and associated uncertainties are at the center of this controversy (Earles and Halog, 2011).

2.2.1.5 Environmental impacts and LCIA methods

The life-cycle impact assessment (LCIA) phase includes the following mandatory elements: i) selection of impact categories and corresponding category indicators and models, ii) classification (assignment of inventory parameters to the impact categories) and iii) characterization (the inventory results are multiplied by equivalency factors which are specific to each parameter and impact category; thereafter all parameters included in each impact category are added and the result of the impact category is obtained). In addition to these mandatory elements, the LCIA may include the following optional elements: normalization (calculation of the magnitude of the characterization results relative to reference information), grouping (sorting and possibly ranking the impact categories) and weighing (conversion and possibly aggregation of the characterization or normalization results across impact categories) (ISO, 2006a,b; Castanheira et al., 2010).

The main problems faced during LCIA result from the need to connect the right burdens with the right impacts at the correct time and place (Reap et al., 2008). There are also various practical difficulties currently associated with impact category selection. These difficulties spring from a lack of current standardization in several impact categories present in the LCA literature (Udo de Haes et al., 2002). In addition, different results (values, impact categories, units) can be obtained depending on the LCIA method adopted (Buchgeister, 2012; Cavalett et al., 2012; Dreyer et al., 2003; Landis and Theis, 2008). Despite of most of the LCA studies of biodiesel considered the climate change impact, only a few have accounted for other environmental impact categories such as eutrophication or acidification. Also, the

toxicity impacts originated from pesticides and fertilizers application (heavy metals emissions) are not typically addressed (Rosenbaum et al., 2008).

2.3 REVIEW OF PALM AND SOYBEAN BIODIESEL LCA STUDIES

This sub-chapter presents the main findings from a literature review of the LCA of palm and soybean-based biodiesel systems along its partial or whole LC. An online search of publicly available articles was conducted to find studies that have been published in recent years (since 2007) with detailed information on the methodology, assumptions and data used. The studies with lack of transparency or sufficient quantitative information were not included. It should be noted that in most of studies it is not clear if the authors adopted system expansion or substitution method since there is some misunderstood regarding the definition of both approaches.

2.3.1 PALM BIODIESEL

A total of more than 30 LCA studies of palm oil and palm biodiesel were assessed, of which a selection of 24 is presented in **Table 2.1**. The **geographical scope** of the studies is representative of the current world supply share: among the 24 studies 19 are located in Southeast Asia, the main world supply region, while only 3 studies were set in South America (Castanheira et al., 2014b; Angarita et al., 2009; de Souza et al., 2010) and 1 in Africa (Achten et al., 2010). Most of the studies conducted in Southeast Asia were set in Malaysia. Even though Indonesia is the world's largest palm oil producer, only few studies (3) were set in this country. In contrast, although palm oil production in Thailand represents only 3% of the world's palm oil production (USDA, 2013b), 7 studies were set in this country.

Table 2.1. Surveyed LC studies of palm biodiesel production (from 2007 to 2014): relevant data and assumptions, methodological choices and key results.

Relevant data, choices and assumptions	**Castanheira et al., 2014b**	**Harsono et al., 2012**	**Kaewmai et al., 2012**	**Silalertruksa and Gheewala, 2012a**	**Silalertruksa and Gheewala, 2012b**
Geographical scale	Colombia	Indonesia	Thailand	Thailand	Thailand
System boundaries	WtG (extraction mill)	WtG	WtG (extraction mill)	WtW	WtW
Functional Unit	1 kg PO	1 t PME	1 t CPO	1 MJ PME	1000 L PME
Multifunctionality approach	Mass, energy and economic allocation	Mass allocation (only on extraction phase)	Energy allocation	Economic allocation	Economic allocation
Type of LCA	Attributional	Attributional	Attributional	Attributional	Attributional
Land-use practices / production options	4 fertilization schemes and 2 biogas management options (captured and flared or released into the atmosphere)	Regional differentiation. POME & biogas emissions not included	Extraction mill with and without biogas capture	Four scenarios for EFB and POME management	Four scenarios for EFB and POME management
Nitrogen field emissions	Yes.	Yes (N_2O: 134–309 kg CO_2eq ha y^{-1})	Yes	Yes	Yes
LUC	Yes, dLUC: tropical forest, savanna, shrubland and cropland conversion	Yes, dLUC (5296–21255 kg CO_2eq hay^{-1})	No	Yes, dLUC & iLUC: rubber, cassava, paddy field, set-aside and forest conversion	Yes, dLUC from conversion of tropical forest
LCIA method	IPCC (2006)	-	-	-	CML 2 (2007)
Environmental impacts	GHG intensity (GWP100, GWP500, GWP20)	Energy and GHG balance (GWP100), carbon payback time	GHG intensity (GWP100)	GHG intensity (GWP100)	Net Energy Ratio (NER) and Renewability, land-use, global warming (GWP100), photochemical oxidation, acidification and eutrophication
Selected results					
GHG intensity	-3.0–5.3 (economic all.), -2.6– 4.5 (mass all.), -2.9– 5.0 (energy all.) kg CO_2eq FU^{-1}	1.97–5.63 t CO_2eq FU^{-1} (LUC) 0.52–75 t CO_2eq FU^{-1} (no LUC)	750 (biogas capture)–1087 (biogas without capture), 871 (average 14 mills) and 440 (best observed case) kg CO_2eq FU^{-1}	18–38 (no LUC), -5–95 (rubber), 16–116 (cassava), 8–27(paddy field), 9–28 (set-aside), 218–248 (forest) g CO_2eq FU^{-1}	1398 (no LUC) to 6620 (LUC) kg CO_2eq FU^{-1}
Carbon payback time		11–42 years			
Energy		Net energy yields: 43.6–49.2 GJ FU^{-1}			NER and Renewability: 2.07 and 2.12 per FU
Photochemical oxidation					0.26 (0.10–0.11) kg C_2H_4eq FU^{-1}
Acidification					2.14 (1.85–2.02) kg SO_2eq FU^{-1}
Human toxicity					2.67 (2.41–2.59) kg 1,4 DBeq FU^{-1}
Eutrophication					0.55 (0.52–0.53) kg PO_4^{3-}eq FU^{-1}
Abiotic depletion					
Ozone depletion					

Table 2.1. (Continued)

Relevant data, choices and assumptions	**Choo et al., 2011**	**Hassan et al., 2011**	**Siangjaeo et al., 2011**	**Stichnothe and Schuchardt, 2011**
Geographical scale	Malaysia	Malaysia	Thailand	Malaysia
System boundaries	WtW	WtW	WtG	WtG (extraction mill)
Functional Unit	1 t FFB, CPO, RPO & 1 MJ PME	1 MJ PME	1 million L PME day^{-1}	1000 kg FFB
Multifunctionality approach	Mass allocation	No	Economic allocation	System expansion
Type of LCA	Attributional	Attributional	Attributional	Attributional
Land-use practices / production options	Extraction mill with and without biogas capture	Biogas is released	Regional differentiation of LCI (Krabi, Chonburi and Pathumthani)	2 scenarios: worst (EFB is dumped and POME stored in ponds) & best (residues are treated by co-composting and returned to the plantation)
Nitrogen field emissions	Yes (N_2O: 58 g CO_2eq t^{-1} FFB)	Yes (N_2O: 15 g CO_2eq FU^{-1})	Yes	Yes
LUC	No	Yes, dLUC: peat, primary and secondary forest, grassland conversion	Yes, dLUC based on type of land-use in 3 regions: Krabi (rubber conversion), Chonburi (cassava) and Pathumthani (abandoned tangerine orchard)	No
LCIA method	IPCC (2001)	IPCC (2006)	IPCC (2006)	CML 2 (2007)
Environmental impacts	GHG intensity (GWP100)	GHG intensity (GWP100)	GHG intensity (GWP100)	Abiotic depletion, acidification, global warming, eutrophication and human toxicity
Selected results				
GHG intensity	119 kg CO_2eq t^{-1} FFB With CH_4 capture: 506 kg CO_2eq t^{-1} CPO, 626 kg CO_2eq t^{-1} RPO, 21.20 g CO_2eq MJ^{-1} PME Without CH_4 capture: 971 kg CO_2eq t^{-1} CPO, 1113 kg CO_2eq t^{-1} RPO, 33.19 g CO_2eq MJ^{-1} PME	56 (average), 225–3300 (peat forest), 270–530 (primary forest), 120–190 (secondary forest), 26–77 (grassland), -85–- -23 (degraded grassland) g CO_2eq FU^{-1}	750–1052 (no LUC), -709 (Krabi), -748 (Chonburi) and -600 (Pathumthani) t CO_2eq FU^{-1}	109.62 (best)–460.98 (worst) kg CO_2eq FU^{-1}
Carbon payback time				
Energy				
Photochemical oxidation				
Acidification				2.86 (worst)–2.89 (best) kg SO_2eq FU^{-1}
Human toxicity				34.86 (best)–48.16 (worst) kg DBeq FU^{-1}
Eutrophication				0.89 (best)–2.05 (worst) kg PO_4^{3-}eq FU^{-1}
Abiotic depletion				0.42 (best)–0.46 (worst) kg Sbeq FU^{-1}
Ozone depletion				

Table 2.1. (Continued)

Relevant data, choices and assumptions	**Achten et al., 2010**	**de Souza et al., 2010**	**Papong et al., 2010**	**Schmidt, 2010**	**Angarita et al., 2009**	**Lam et al., 2009**
Geographical scale	Cameroon	Brazil	Thailand	Malaysia and Indonesia	Colombia, Brazil	Malaysia
System boundaries	WtW	WtW	WtG	WtG (extraction mill)	WtG	WtG
Functional Unit	1 MJ PME	1 ha year^{-1} (4 t PME)	1 kg PME	1 t RPO	1 kg PME	1 t PME
Multifunctionality approach	System expansion (PKM used as feed)	Mass allocation	Mass, energy and economic allocation (in the different LC phases)	System expansion (in agricultural and mill stage and only in mill) & economic allocation	n/d	n/d
Type of LCA	Attributional	Attributional	Attributional	Scenarios: Consequential (1, 2 and 3), semi-consequential (4) and attributional (5)	Attributional	-
Land-use practices / production options	3 POME treatment scenarios: 1) untreated, 2) treated (no CH_4 recovering), 3) treated (CH_4 recovering and used)	No	No	No	Regional differentiation of life-cycle inventory	-
Nitrogen field emissions	Yes (IPCC, 2006)	Yes	No	Yes, based on nutrient balances	No	Yes
LUC	Yes (carbon payback time)	Yes (carbon payback time)	No	Yes, dLUC & iLUC: set-aside and secondary forest conversion	No	No
LCIA method	IPCC (2006), Nordic Guidelines on LCA (Lindfors et al.1995)	IPCC (2001)	-	Danish EDIP97 method (comparison with Impact 2002+ and EcoIndicator)	-	-
Environmental impacts	Acidification, global warming, eutrophication, land-use	Energy and GHG balance	Energy balance	Acidification, global warming, eutrophication, photochemical smog, ozone depletion, land-use and biodiversity	Energy balance	Land-use, energy and GHG balance
Selected results						
GHG intensity	60.5–87.5 g CO_2eq FU^{-1}	1437 kg CO_2eq FU^{-1} (no LUC)		2.60–3.45 t CO_2eq FU^{-1} (scen. 3 & 5)		-6.9 t CO_2eq FU^{-1} (include carbon sequestration)
Carbon payback time	47–49 years	39 years				
Energy	NER: 2.59–2.63	NER ratio: 5.4	NEV: 24.03 MJ FU^{-1} NER: 2.48		NER: 4.7 (Colombia) & 4.92 (Brazil) Energy consumption: 7.3 (Colombia) & 7.7 (Brazil) MJ FU^{-1}	NER: 2.27
Photochemical oxidation				0.509–0.617 kg C_2H_4eq FU^{-1} (scen. 2 & 3)		
Acidification	1.35–1.57 g SO_2eq FU^{-1}			13.0–23.5 kg SO_2eq FU^{-1} (scen. 2 & 3)		
Human toxicity						
Eutrophication	31.9–39.6 g O_2eq FU^{-1}			81–337 t NO_3eq FU^{-1} (scen. 1 & 3)		
Abiotic depletion						
Ozone depletion				43.8–77.8 mg CFC11eq FU^{-1} (scen. 1 & 3)		

Table 2.1. (Continued)

Relevant data, choices and assumptions	**Pleanjai and Gheewala, 2009**	**Reinhard and Zah, 2009**	**Thamsiriroj and Murphy, 2009**	**Yee et al., 2009**	**Fargione et al., 2008**
Geographical scale	Thailand	Malaysia (PME exported to Switzerland)	Thailand	Malaysia	Indonesia & Malaysia
System boundaries	WtG	WtT	WtT	WtG	WtG
Functional Unit	1 t PME	1 MJ PME	1 GJ PME	1 t PME	1 ha
Multifunctionality approach	n/d	Economic allocation & system expansion (2 scenarios)	No	No	Economic allocation
Type of LCA	-	Attributional & consequential	-	-	Attributional
Land-use practices / production options	-	Import PME from Malaysia	Import PME from Thailand or produce biodiesel from indigenous Irish rape seed	-	n/d
Nitrogen field emissions	-	Yes	Yes (9.78 kg CO_2eq MJ^{-1})	Yes	n/d
LUC	No	Yes, dLUC from rain forest on peatland, rain forest and grassland	No	Yes, dLUC from peatland conversion	Yes, dLUC: conversion of tropical and peatland rainforest
LCIA method	-	CML (midpoint) and Swiss ecological scarcity method 2006 (endpoint)	n/d	IPCC (2006)	IPCC (2006)
Environmental impacts	Energy balance	Abiotic depletion, acidification, eutrophication, global warming (GWP100), ozone depletion, photochemical oxidation, toxicity, land occupation.	Energy and GHG balance	Energy and GHG balance	GHG intensity
Selected results					
GHG intensity		24 (Attrib.), 208–211 (Conseq.) g CO_2eq MJ^{-1}	35.21 kg CO_2eq MJ^{-1}	-1797 t CO_2eq FU^{-1} (include carbon sequestration)	611 (tropical rainforest)& 3003 (peatland) t CO_2eq FU^{-1}
Carbon payback time					86 (tropical rainforest) & 423 (peatland) years
Energy	NER: 2.42 NEB: 55.51 GJ ha^{-1}		Net energy: 74.23 GJ ha^{-1}	NER: 3.53	
Photochemical oxidation		35 (Attrib.), 39 (Conseq.) mg C_2H_4eq MJ^{-1}			
Acidification		207 (Attrib.), 179–183 (Conseq.) mg SO_2eq MJ^{-1}			
Human toxicity		12 (Attrib.), 10 (Conseq.) g 1,4-DBeq MJ^{-1}			
Eutrophication		102 (Attrib.), 111–113 (Conseq.) mg PO_4eq MJ^{-1}			
Abiotic depletion		168 (Attrib.), 76–79 (Conseq.) mg SBeq MJ^{-1}			
Ozone depletion		0.0022 (Attrib.), -0.0101– -0.0100 (Conseq.) mg CFC-11eq MJ^{-1}			
Terrestrial ecotoxicity		23561 (Attrib.), 29772–31696 (Conseq.) mg 1,4-DB eq MJ^{-1}			

Table 2.1. (Continued)

Relevant data, choices and assumptions	**Germer and Sauerborn, 2008**	**Reijnders and Huijbregts, 2008a**	**Wicke et al., 2008**	**Yusoff and Hansen, 2007**
Geographical scale	n/d	South Asia	Malaysia (northern Borneo)	Malaysia
System boundaries	Palm plantation	WtG (extraction mill)	WtW	WtG (extraction mill)
Functional Unit	1 ha (25 years)	1 kg palm oil	1 MJ CPO	1 t CPO
Multifunctionality approach	-	Economic allocation	System expansion	n/d
Type of LCA	-	Attributional	Attributional	-
Land-use practices / production options	-	-	Yes. CH_4 collection and electricity production, improved yields, increased organic fertilizer (improvement scenario).	Yes. Alternative scenario: fertilizer production, reuse of POME and EFB were optimized, transportation was done by rail, emissions from boiler was reduced and anaerobic POME digestion was done (with CH_4 collection)
Nitrogen field emissions	Yes	Yes	Yes	Yes
LUC	Yes, dLUC from grassland and forest (mineral soil and peatland) conversion	Yes, dLUC from forest (peaty and non-peaty soils) conversion	Yes, dLUC: conversion of logged-over rainforest, natural rainforest, degraded grassland and peatland (forest cover).	No
LCIA method	IPCC (2001)	IPCC (2001)	IPCC (2006)	Eco-indicator 99 (weighted results)
Environmental impacts	GHG intensity (GWP100)	GHG intensity	GHG intensity	Eco-indicator 99 impact categories
Selected results				
GHG intensity	Grassland: -136 (zero burning) & -134 (burning) t CO_2eq FU^{-1} Forest conv., mineral soil: 647 (zero burning) & 668 (burning) t CO_2eq FU^{-1} Forest conv., peat: 1314 (zero burning) & 1335 (burning) t CO_2eq FU^{-1}	2.6–7.0 (non-peaty soils, 100&50y) to 10.2–18.2 (peaty soils, 100&50y) kg CO_2eq FU^{-1}	Biodiesel (g CO_2eq FU^{-1}): -51 (degraded land, -53 in improve. scenario), 42 (logged-over rainforest), 107 (natural rainforest), 391 (peatland). Electricity (g CO_2eq FU^{-1}): -35 (degraded land, -37 in improve. scenario), 48 (logged-over rainforest), 123 (natural rainforest), 407 (peatland).	3.5 Pt (alt. scenario: 1.9)
Carbon payback time			Biodiesel: 8 (logged-over rainforest)–169 (peatland) years. Electricity: 16 (logged-over rainforest)–320 (peatland) years	
Acidification/Eutrophication				2.7 Pt (alt. scenario: 1.5)
Ecotoxicity				1.0 Pt (alt. scenario: 0.3)
Ozone depletion				0.01 Pt (alt. scenario: 0.006)

CPBT: carbon payback time; CPO: crude palm oil; dLUC: direct land-use change; EFB: empty fruit bunches; FFB: fresh fruit bunches; FU: functional unit; n/d: not-distinguishable; GWP: global warming potential; iLUC: indirect land-use change; NEB: net energy balance; NER: net energy ratio; NEV: net energy value; PME: palm methyl-ester or palm biodiesel; PO: palm oil; POME: palm oil mill effluent; RPO: refined palm oil; WtG: well-to-gate; WtT: well-to-tank; WtW: well-to-wheel.

Concerning the **system boundaries**, different LC approaches have been adopted in the reviewed studies. The majority of studies (14 out of 24) were carried out using the "well-to-gate" approach, including 8 that considered the transesterification plant gate and 6 the oil extraction mill gate. Seven of the reviewed LCAs are "well-to-wheel" assessments including the transformation of palm oil into biodiesel and its consumption, although only some studies encompass a specific assessment of emissions from its consumption: Wicke et al. (2008) compared the GHG emissions from the use of crude palm oil for biodiesel and electricity production, Choo et al. (2011) evaluated the palm biodiesel from Malaysia in bench endurance tests and Silalertruksa and Gheewala (2012a) adopted the chassis dynamometer studies for a pickup truck (or light-duty diesel vehicle) given by Pleanjai et al. (2009). Two studies are "well-to-tank" assessments (Reinhard and Zah, 2009; Thamsiriroj and Murphy, 2009) and only one focused on the single LC phase of palm plantation (Germer and Sauerborn, 2008).

Various **functional units** were adopted in the reviewed studies. In the majority of studies (14 out of 24) the choice of functional unit was based on the mass or volume of product (e.g. kg, t, L): 8 LCAs of palm biodiesel and 6 of intermediate products (palm fresh fruit bunches: FFB, crude and refined palm oil: CPO and RPO). Seven studies used 1 MJ or 1 GJ of biodiesel or palm oil energy content (measured in terms of the lower heating value), as this is an appropriate basis for comparison of the energy delivered by the biodiesel or palm oil to the end user. Other studies (3 out of 24) adopt a measure of agricultural surface area (usually the hectare), emphasizing the importance of land-use impacts and the land-use expansion for growing energy crops. Choo et al. (2011) used more than one functional unit in order to analyze each LC phase individually. None of the reviewed studies use distance traveled (km) as the functional unit, even the WtW studies.

Regarding the **multifunctionality**, the handling of co-products (e.g., palm kernel oil, palm kernel meal, glycerin) is diverging and not always clearly stated (Bessou et al., 2013). It can be observed that in 3 out of 24 LCAs a sensitivity analysis for alternative approaches was performed to evaluate the influence on the results, as suggested by ISO 14044:2006 (ISO, 2006b). The sensitivity analysis implemented in these studies include a comparison of different allocation procedures (according to mass balance, energy content and price of products) and a comparison of allocation and system expansion or substitution approaches (Schmidt, 2010; Reinhard and Zah, 2009). Nine studies used a single allocation approach (5 based on prices, 3 based on mass balance and 1 on energy content of products) and three avoided allocation by using system boundary expansion (Stichnothe and Schuchardt, 2011; Achten et al. 2010; Wicke et al., 2008). In 7 LCAs multifunctionality is either not considered or no clear procedures are specified.

With respect to the **type of study**, all reviewed LCA studies in **Table 2.1** are attributional, except the consequential studies of Schmidt (2010) and Reinhard and Zah (2009). The different multifunctionality approaches adopted and types of LCA make it very difficult to compare the results obtained in the reviewed studies. However, Castanheira et al. (2014b) showed that comparable results were obtained

with the three attributional allocation methods (energy, mass and market value-based allocation), whereas Schmidt (2010) and Reinhard and Zah (2009) demonstrated that significant differences on results are obtained when system expansion and allocation were adopted. Also, the extent of these differences varies depending on the environmental impacts assessed.

In palm biodiesel system the farm is the LC phase that contribute most to the LC environmental impacts (mainly due to the production and use of fertilizers), whereas palm oil milling is the most complex stage associated with various residues (e.g. empty fruit bunches: EFB, palm oil mill effluent: POME, shell, fibers) for which there are many options for treatment (Bessou et al., 2013; Silalertruksa and Gheewala, 2012b). The influence of different **mill management practices** on the environmental impacts of palm oil or palm biodiesel were investigated in 9 of the reviewed studies, from which 8 compared the POME treatment systems, including management options of biogas (e.g., biogas released into the atmosphere, recovered and flared or collected and use for electricity production). Castanheira et al. (2014b) showed that if biogas was captured and flared could reduce the palm oil GHG intensity by 50-60%.

Six studies also compared different **land-use practices,** namely the use of different types and quantities of fertilizers (mineral and organic), the effect of optimization of fertilizers production and the alternative uses of EFB as fertilizer (directly dumped in the plantation or after co-composting with POME). Hansen et al. (2012) showed that the use of residues in an optimized manner can reduce 95% of the emissions from palm biodiesel production, while the results from Silalertruksa and Gheewala (2012a) revealed that the various uses of EFB and POME could help improve the GHG performance of palm biodiesel by around 48% to 57%, compared to the cases in which EFB and POME treatment were not included. It should also be emphasized that in 3 out of 24 studies there were implemented site-specific LC inventories, i.e. considering regional differentiation.

The majority of the studies reviewed (15) included the emissions related to the **land-use change** (LUC) due to the expansion of palm oil area and concluded that LUC emissions are an important aspect for the GHG intensity of palm biodiesel. However, a wide range of results was reported demonstrating that the estimation of carbon stock changes due to LUC has a high level of uncertainty associated. For example, palm biodiesel results (in g CO_2eq MJ^{-1}) vary from: i) -85 to 3300 (Hassan et al., 2011) and 24 to 211 (Reinhard and Zah, 2009) in Malaysia, ii) 53 and 150 in Indonesia[a] (Harsono et al. 2012), iii) -5 to 248 in Thailand (Silalertruksa and Gheewala, 2012a). Also, the GHG intensity of palm oil (kg CO_2eq kg^{-1} palm oil) greatly varies among the studies: 2.6 to 3.45 (Schmidt, 2010), -3.0 to 5.3 (Castanheira et al., 2014b), 2.6 to 10.2 (Reijnders and Huijbregts, 2008a). It should be noted that among the studies considering LUC, only two assessed the indirect impacts (Schmidt, 2010; Silalertruksa and Gheewala, 2012a).

[a]adopting a LHV of 37 MJ kg^{-1} palm biodiesel

The differences in the carbon emissions due to LUC are mostly related to the area that is converted, the type of previous land-use (reference land-use) and the climate region and soil type. Also, different approaches were undertaken by the authors in relation to incorporation of LUC in the LCIA: IPCC guidelines (IPCC, 2006, 2001), carbon payback time (the time that a biofuel system needs to repay the initial C emission caused by LUC) or other methods (e.g. Achten et al., 2010; Fargione et al., 2008). The wide range of results shows that producing general figures for the quantification of direct LUC in GHG intensity is difficult and each case should be addressed individually (Cherubini, 2010). However, all reviewed studies that considered LUC demonstrated that the LUC emissions are the highest when forests or peat lands are converted (Schmidt, 2010; Reijnders and Huijbregts, 2008a; Wicke et al., 2008). On the opposite the lowest LUC emissions occurred when grassland or cropland were converted in palm plantations (e.g., Castanheira et al., 2014b; Siangjaeo et al., 2011).

Other critical issue is the calculation of **nitrogen field emissions** (nitrous oxide N_2O, nitrates NO_3^-, ammonia NH_3 and nitrogen oxides NO_x) from palm cultivation, which contribute to several environmental impacts, such as eutrophication, acidification and global warming (Achten et al., 2010; Reijnders and Huijbregts, 2011). Several authors showed that N_2O field emissions can contribute 31-69% for the GHG intensity of palm plantations (Achten et al., 2010; Choo et al., 2011; Harsono et al., 2012; de Souza et al., 2010).

Even though N_2O emissions from soil were taken into account in the majority of reviewed studies (20 out of 24), in most cases it is not clear if both direct and indirect N_2O emissions (particularly NH_3 and NO_x emissions due to nitrogen volatilization and NO_3^- emissions originating in the fraction of nitrogen lost via runoff and leaching) were included and if direct N_2O emissions due to the soil carbon stock changes as a result of LUC were considered. Additionally, in some studies N_2O emissions were assessed based on IPCC approach, whereas other authors estimated N_2O emissions using single figures which were calculated as a percentage of the N fertilizer input to soil and ignoring the local environmental (e.g. soil clay content, precipitation, root depth) and technical (e.g. differences in fertilizers used) uniqueness. The variation in N_2O emissions calculated in the reviewed studies is associated with variability in system definition and modeling choices (fertilizer type and rates, other nitrogen inputs, soil type, climate), as well as with uncertainty in direct and indirect N_2O emission calculation (Reijnders and Huijbregts, 2011; IPCC, 2006).

The majority of reviewed LCA studies (18 out of 24) focused on the GHG intensity or global warming potential (or climate change), together with energy and fossil fuel use, without considering any further **environmental impact categories**. This approach is usually supported by motivations of energy efficiency and climate change mitigation of the development of renewable fuels. The IPCC guidelines are used in quantification of the global warming potential (GWP) in almost all studies; however, not all studies refer to the same version of the IPCC global warming equivalent factors. Also, Castanheira et al. (2014b) demonstrated that the time horizon considered for the GHG intensity calculation can also

influence the results (GWP of CH_4 and N_2O for time horizons of 20, 100 and 500 years vary significantly). Although in practice a time horizon of 100 years is often chosen, a time horizon of 500 years would reduce the importance of CH_4 emissions almost three times and N_2O emissions almost by half (IPCC, 2007; Reijnders and Huijbregts, 2011).

Only 6 LCA studies addressed a wider set of environmental impacts of palm oil or palm biodiesel. Other impact categories that were addressed include toxicity, eutrophication and acidification. Various LCIA methods were used to categorize the environmental impacts as well as to quantify the physical flow into each characterized impact: Nordic Guidelines on LCA and IPCC (2006) by Achten et al. (2010), CML 2 (2007) by Silalertruksa and Gheewala (2012b) and Stichnothe and Schuchardt (2011), whereas Schmidt (2010) adopted the Danish EDIP97 method and compared with Impact 2002+ and EcoIndicator. In two LCAs the results of end-point assessments using EcoIndicator 99 (Yusoff and Hansen, 2007) and Swiss ecological scarcity method 2006 (Reinhard and Zah, 2009) were presented. No articles were found using the recent ReCiPe and USEtox methods.

The LCIA results vary depending on the chosen method, which jeopardizes the consistency across these methods and the comparison between studies; however, all studies that included LUC demonstrated that it is the most decisive factor in determining the GHG intensity of palm oil biodiesel. In studies that exclude LUC (Kaewmai et al., 2012; Choo et al., 2011; Thamsiriroj and Murphy, 2009), the LC phases that contribute most to the GHG intensity are cultivation (due to chemical and energy inputs and subsequent field emissions) and oil extraction mill due to the methane emissions from POME treatment, which can be drastically reduced if the biogas was captured (Choo et al., 2011). In addition, Thamsiriroj and Murphy (2009) demonstrated that GHG emissions from transport also make an important contribution to the GHG intensity of palm oil biodiesel produced in Europe based on imported palm oil.

Because eutrophication, acidification, ozone depletion, photochemical oxidation and toxicity impacts are reported less uniformly, it is more hardly to discuss and compare the results. Studies state that the eutrophication and acidification impacts of palm biodiesel are also mainly caused by agricultural phase, namely associated with NH_3 emissions, as well as PO_4^{3-} and NO_3^- leaching (e.g. Achten et al., 2010; Stichnothe and Schuchardt, 2011). Reinhardt and Zah (2009) also showed that intensification driven by demand (i.e. driven by an additional input of fertilizer and pesticides) may reduce the GHG emissions related to LUC but it may enhance the scores of midpoint indicators such as acidification and eutrophication.

2.3.2 SOYBEAN BIODIESEL

A total of more than 30 LCA studies of soybean, soybean oil/meal and soybean biodiesel were assessed, of which a selection of 21 is presented in **Table 2.2**. The **geographical scope** of the reviewed LCA studies of soybean-based products was the main world supplier countries of soybean: USA (6), Brazil (10), Argentina (4) and China (2). Only one study was conducted in Europe (Buratti et al., 2012). However, various studies (9 out of 21) evaluated the impact of produced soybean or soybean-based products overseas (USA, Brazil, Argentina and China) and utilized in EU countries (Denmark, France, Netherlands and Portugal), demonstrating the EU dependence on imported soybean complex (Milazzo et al., 2013). The diversity of soybean cultivation and processing conditions would impose various LCAs; however, only few studies (2) described the environmental assessments of different regions within a country (Prudêncio da Silva et al., 2010; Kim and Dale, 2009).

Different **system boundaries** and **functional units** were defined because the scope varies greatly among the reviewed studies. The majority of studies (13 out of 21) were carried out using the "well-to-tank" approach (8 out of 13 considered "well-to-wheel" approach). In these studies the choice of functional unit was mostly based on the energy content (e.g., MJ, Btu), mass or volume (e.g. kg, L) of soybean biodiesel, whereas only two studies use distance traveled (e.g., mile, km) as the functional unit (Panichelli et al., 2009; Searchinger and Heimlich, 2009).

Six reviewed LCAs have focused on the production of soybean (Castanheira and Freire, 2013; Knudsen et al., 2010; Prudêncio da Silva et al., 2010) and co-products from soybean processing (Middelaar et al., 2013; Dalgaard et al., 2008; Kim and Dale, 2009) and adopted the mass of product as functional unit. Lehuger et al. (2009) assessed a feeding ration based on soy meal as concentrates, while Miller et al. (2007) analyzed the environmental impacts of a soybean lubricant used in an aluminum rolling manufacturing facility. These findings demonstrate that the importance of soybean goes far beyond its use as a feedstock for biodiesel.

Different **multifunctionality approaches** were adopted in the reviewed studies: system expansion or substitution approach (Huo et al., 2009; Kim and Dale, 2009; Reinhard and Zah, 2009; Dalgaard et al., 2008), allocation based on mass (Mourad and Walter, 2011; Lehuger et al., 2009; Searchinger and Heimlich, 2009; Miller et al., 2007), energy (Buratti et al., 2012; Cavalett and Ortega, 2010) and market value (Middelaar et al., 2013; Xue et al., 2012; Morais et al., 2010; Panichelli et al., 2009; Fargione et al., 2008; Reijnders and Huijbregts, 2008b). Even though system expansion and substitution are the preferred methods (Pradhan et al., 2008), the difficulty with these methods is to find an exact substitute for soybean meal. For example, dried distillers grain or canola meals are animal feed products similar to soybean meal, but they are not exact substitutes (Pradhan et al., 2008). In addition, since many displaced products are themselves co-products of other production systems, at some point a value based allocation may have to occur (Kendall and Chang, 2009).

Table 2.2. Surveyed LC studies of soybean biodiesel production (from 2007 to 2014): relevant data and assumptions, methodological choices and key results.

Relevant data, choices and assumptions	**Castanheira and Freire, 2013**	**Middelaar et al., 2013**	**Buratti et al., 2012**	**Xue et al., 2012**
Geographical scale	Argentina and Brazil (exported to EU)	Brazil	Italy (Veneto region)	USA (Pennsylvania)
System boundaries	Cultivation and transport to EU	CtG	WtT	WtW
Functional Unit	1 kg soybean	1 t soybean meal	1 MJ SME	1 ULSD equivalent L fuel[a]
Multifunctionality approach	-	Economic allocation	Energy allocation	Economic allocation
Type of LCA	Attributional	Attributional	Attributional	Attributional
Land-use practices / production options	Yes. No-tillage (NT), reduced-tillage (RT) vs tillage (T) cultivation systems. Regional differentiation (climate region). Various transport distances	Yes. No-tillage (NT), reduced-tillage (RT) vs tillage (T) cultivation systems	Yes. Physical vs chemical refining	B5, B20 and B100 3 transportation scenarios
Nitrogen field emissions	Yes, direct and indirect emissions and sensitivity analysis	Yes. NH_3 and NO_3^- calculated based on alternative methods	Yes, direct and indirect emissions (IPCC, 2006)	Yes.
LUC	Yes, dLUC (45 scenarios)	Yes, 3 methods: 1) dLUC (IPCC, 2006), 2) d+iLUC (Leip et al., 2010) and 3) d+iLUC (Audsley et al., 2009)	No	No
LCIA method	IPCC (2006), RED (EC, 2009)		RED (EC, 2009)	TRACI 1 v2.00
Environmental impacts	GHG intensity	GHG intensity or carbon footprint (CFP)	GHG intensity	Global warming, ozone depletion, toxicity, acidification, eutrophication, smog and respiratory impacts
Selected results				
GHG intensity	Including LUC (kg CO_2eq FU^{-1}): 0.06 (severely degraded grassland conversion in warm temperate dry region) to 17.8 (rainforest conversion in tropical region) No LUC (kg CO_2eq FU^{-1}): 0.30 (RT in warm temperate dry region) to 0.6 (T in tropical and warm temperate moist region)	No LUC (kg CO_2eq FU^{-1}): 483 (NT), 749 (RT), 874 (T) No LUC, NT (kg CO_2eq FU^{-1}): 483, 488, 541 (alternative NO_3^- emission calculation methods) LUC (kg CO_2eq FU^{-1}): 652–667 (1), 3536 (2), 879 (3)	30–31 g CO_2eq FU^{-1}	(normalized results)
Carbon payback time				
Energy				
Photochemical oxidation				
Acidification				
Human toxicity				
Eutrophication				
Abiotic depletion				
Ozone depletion				
Terrestrial ecotoxicity				
Freshwater ecotoxicity				
Marine ecotoxicity				

Table 2.2. (Continued)

Relevant data, choices and assumptions	**Hou et al., 2011**	**Mourad and Walter, 2011**	**Cavalett and Ortega, 2010**	**Knudsen et al., 2010**
Geographical scale	China	Brazil	Brazil	China (Jilin province, exported to Denmark)
System boundaries	WtW	WtT	WtT	Cultivation and transport to Denmark
Functional Unit	1 MJ SME	1000 kg SME	1 L SME	1 t organic soybean (protein content of min. 36%)
Multifunctionality approach	Mass and energy allocation (Ma &Ea)	Mass allocation	Energy allocation (no allocation in biodiesel production phase)	-
Type of LCA	Attributional	Attributional	-	Attributional
Land-use practices / production options	Transport distances	-	-	Farm gate: Organic (20 farms) vs conventional (15 farms).
Nitrogen field emissions	Yes	-	Yes	Yes. Sensitivity analysis of N_2O calculation methods
LUC	No	No	No	No
LCIA method	CML 2001	-	-	EDIP97
Environmental impacts	All included in CML method	Energy balance	Material, energy and emergy flows (emergy accounting, embodied energy analysis, material flow accounting)	Non-renewable energy use, global warming, eutrophication and acidification.
Selected results				
GHG intensity	34.9 (Ma)–75.7 (Ea) g CO_2eq FU^{-1}		0.86 kg CO_2 FU^{-1}	429 kg CO_2eq FU^{-1}; Organic farm gate: 156 (144–523) kg CO_2eq FU^{-1}; Conventional farm gate: 263 (232–609) kg CO_2eq FU^{-1}
Carbon payback time				
Energy		Renewability factor: 4.27	NER: 2.48 Emergy yield ratio: 1.62	Non-renewable energy use: 4377 MJ FU^{-1}
Photochemical oxidation	0.079 (Ma)–0.14 (Ea) g C_2H_4eq FU^{-1}			
Acidification	1.4 (Ma)–2.7 (Ea) g SO_2eq FU^{-1}			8.1 kg SO_2eq FU^{-1}
Human toxicity	12.8 (Ma)–24.9 (Ea) g 1,4DCBeq FU^{-1}			
Eutrophication	0.31 (Ma)–0.64 (Ea) g PO_4eq FU^{-1}			8.8 kg NO_3^-eq FU^{-1}
Abiotic depletion	0.17 (Ma)–0.34 (Ea) g Sbeq FU^{-1}			
Ozone depletion	$1.2x10^{-8}$ (Ma)–$2.9x10^{-8}$ (Ea) g CFC-11eq FU^{-1}			
Terrestrial ecotoxicity	0.34 (Ma)–0.82 (Ea) g 1,4DCBeq FU^{-1}			
Freshwater ecotoxicity	19 (Ma)–47 (Ea) g 1,4DCBeq FU^{-1}			
Marine ecotoxicity	40000 (Ma)–86000 (Ea) g 1,4DCBeq FU^{-1}			

Table 2.2. (Continued)

Relevant data, choices and assumptions	**Morais et al., 2010**	**Prudêncio da Silva et al., 2010**	**Huo et al., 2009**	**Kim and Dale, 2009**
Geographical scale	Brazil or USA (soybean oil exported to Portugal)	Brazil (South-SO and Center West-CW, exported to Netherlands)	USA	USA
System boundaries	WtW	Cultivation and transport to Netherlands	WtW	WtG
Functional Unit	1 MJ use in heavy-duty vehicles	1000 kg soybean	1 million Btu	1 kg soybean oil
Multifunctionality approach	Economic allocation	-	(1) Displacement approach[b]; (2) energy allocation; (3) economic allocation; (4) hybrid approach I (displacement for soy meal and glycerin and allocation for other co-products); (5) hybrid approach II (like (4) except that it address soy meal with economic allocation).	Displacement approach[b]
Type of LCA	Attributional	Attributional	Attributional	Attributional
Land-use practices / production options	-	Regional differentiation	Soybean biodiesel (BD), soybean renewable diesel I ("SuperCetane," RD-I), soybean renewable diesel II ("green diesel," RD-II)	Regional differentiation (40 counties)
Nitrogen field emissions	n/d	Yes	Yes (IPCC, 2006)	Yes
LUC	No	Yes, dLUC in Central-West (1% tropical forest and 3.4 % scrubland conversion).	No	Yes, dLUC (only soil organic carbon).
LCIA method	Characterization and normalization models proposed by Pennington et al. (2004)	CML 2001	IPCC, 2006	
Environmental impacts	Global warming, acidification, eutrophication, abiotic depletion, ecotoxicity, land-use and others	CED, acidification, eutrophication, climate change, terrestrial ecotoxicity, land occupation	GHG intensity, total energy use and fossil energy use	GHG intensity
Selected results				
GHG intensity	240 g CO_2eq MJ^{-1}	960 (CW) & 510 (SO) kg CO_2eq FU^{-1}	BD: ~ 0.0 (approach5)–30 (approach2&3) kgCO_2eq FU^{-1} RD-I: ~ -30.0 (approach1)–35 (approach3) kgCO_2eq FU^{-1} RD-II: ~ 22.0 (approach2&3)–40 (approach4) kgCO_2eq FU^{-1}	0.4–2.5 kgCO_2eq FU^{-1}
Carbon payback time				
Energy		CED: 12634 (CW) & 699 (SO) MJ FU^{-1}	Total energy use (million Btu FU^{-1}): BD ~ 1.0 (approach5)–1.4 (approach3); RD-I ~1.1 (approach1)–1.5 (approach4); RD-II ~1.4 (approach2)–1.6 (approach4)	
Photochemical oxidation	3.2x10^{-3} YOLL FU^{-1}			
Acidification	5.6 g 1,4DCBeq FU^{-1}	7.7 (CW) & 5.3 (SO) kg SO_2eq FU^{-1}		
Human toxicity				
Eutrophication	6.4x10^{-4} g PO_4eq FU^{-1}	6.8 (CW) & 6.9 (SO) kg PO_4eq FU^{-1}		
Abiotic depletion	0.6 g Sbeq FU^{-1}			
Ozone depletion	4.4x10^{-5} g CFC-11eq FU^{-1}			
Terrestrial ecotoxicity	0.5 g 1,4DCBeq FU^{-1}	4.9 (CW) & 3.1 (SO) kg 1,4-DCBeq FU^{-1}		

Table 2.2. (Continued)

Relevant data, choices and assumptions	**Lehuger et al., 2009**	**Panichelli et al., 2009**	**Reinhard and Zah, 2009**
Geographical scale	Brazil (soybean meal exported to France)	Argentina (SME exported to Switzerland)	Brazil (SME exported to Switzerland)
System boundaries	CtG	WtW	WtT
Functional Unit	1000 kg of feed (13.8% DM soybean meal)	1 km driven in a 28 t truck in Switzerland with B100	1 MJ SME
Multifunctionality approach	Mass allocation	Economic allocation	Economic allocation & system expansion (2 scenarios)
Type of LCA	Attributional	Attributional	Attributional & consequential
Land-use practices / production options	Tillage and no-tillage (NT)	7 scenarios: 1) yield increment (10%), 2) avoiding deforestation, 3) reduced or conventional tillage, 4) soybean inoculation with bacteria and BNF, 5) avoiding cypermethrin, 6) avoiding methanol, 7) glycerin price	Import soybean biodiesel from Brazil
Nitrogen field emissions	Yes	Yes	Yes
LUC	No	Yes	Yes, dLUC from savanna, grassland and rainforest
LCIA method	CML 2	CED, CML 2001	CML (midpoint) and Swiss ecological scarcity method 2006 (endpoint).
Environmental impacts	Abiotic depletion, climate change, ozone depletion, toxicity impacts, photo-oxidant formation, acidification, eutrophication and land-use	Global warming, CED, toxicity impacts, acidification, eutrophication and land-use	Abiotic depletion, acidification, eutrophication, global warming (GWP100), ozone depletion, photochemical oxidation, toxicity, land occupation.
Selected results			
GHG intensity	391 (NT: 307) kg CO_2eq FU^{-1}	1.2; 0.4(scen.2)–1.2(scen.4) kg CO_2eq FU^{-1}	39 (Attrib.), -40–210 (Conseq.) g CO_2eq MJ^{-1}
Carbon payback time			
Energy		CED: 9.6; 4.7(scen.2)–9.8(scen.3, convent. tillage) MJ FU^{-1}	
Photochemical oxidation	$5.13x10^{-2}$ kg C_2H_2eq FU^{-1}		65 (Attrib.), -2–39 (Conseq.) mg C_2H_4eq MJ^{-1}
Acidification	2.13 kg SO_2eq FU^{-1}	9; 8.0(scen.2)–9.7(scen.4) g SO_2eq FU^{-1}	245 (Attrib.), 188–496(Conseq.) mg SO_2eq MJ^{-1}
Human toxicity	203 kg 1.4-DCBeq FU^{-1}	0.4; 0.2(scen.2)–0.5(scen.3, convent. tillage) kg 1,4-DCBeq FU^{-1}	11 (Attrib.),10–45(Conseq.) g 1,4-DBeq MJ^{-1}
Eutrophication	3.89 kg PO_4eq FU^{-1}	4.0; 3.7(scen.3, reduced tillage)–5.1(scen.4) g PO_4eq FU^{-1}	348 (Attrib.), 115–282(Conseq.) mg PO_4eq MJ^{-1}
Abiotic depletion	1.06 kg Sbeq FU^{-1}		186 (Attrib.), 70–312 (Conseq.) mg Sbeq MJ^{-1}
Ozone depletion	$1.92x10^{-5}$ CFC-11eq FU^{-1}		$2.7x10^{-3}$ (Attrib.), $-1.0x10^{-2}$– $-6.2x10^{-3}$ (Conseq.) mg CFC-11eq MJ^{-1}
Terrestrial ecotoxicity	1.82 kg 1.4-DCBeq FU^{-1}	1.0 ; 0.0(scen.5)–2.5(scen.3, convent. tillage) kg 1,4-DCBeq FU^{-1}	23 (Attrib.), 29767–210806 (Conseq.) mg 1,4-DBeq MJ^{-1}
Freshwater ecotoxicity	19.0 kg 1.4-DCBeq FU^{-1}		
Marine ecotoxicity	$4.84x10^{4}$ kg 1.4-DCBeq FU^{-1}		

Table 2.2. (Continued)

Relevant data, choices and assumptions	**Searchinger and Heimlich, 2009**	**van Dam et al., 2009**	**Dalgaard et al., 2008**	**Fargione et al., 2008**
Geographical scale	USA	Argentina (exported to Netherlands)	Argentina (exported to Netherlands)	Brazil
System boundaries	WtW	WtW	WtG	WtT
Functional Unit	1 mile driven	1 L SME	1 kg soybean meal produced in Argentina and exported to Netherlands	1 ha
Multifunctionality approach	Mass allocation	Energy allocation (sensitivity analyses to various approaches)	System expansion	Economic allocation
Type of LCA	Attributional	Attributional	Consequential	Attributional
Land-use practices / production options	1) assuming no demand reduction or price-induced yield increase and 2) assuming demand reductions and price-induced yield increases replace 40% of diverted soybean oil	4 scenarios: current situation (CUR), a continuation of the current economic development (A), a stronger economic development more export oriented (C) and more environmental friendly (B). Biodiesel processing in Argentina vs processing in Netherlands. Transport by truck or train (inland).	Increased demand for soybean meal affects the agricultural production using the 2 soybean loops: soybean meal (PO), soybean meal (RSO)	n/d
Nitrogen field emissions	n/d	Yes	Yes	n/d
LUC	Yes, dLUC and iLUC	Yes	Yes	Yes. Amazonian rainforest and *Cerrado* biome (grassland)
LCIA method	GREET model	IPCC, 1995	EDIP97	IPCC, 2006
Environmental impacts	GHG intensity	GHG intensity	Global warming, eutrophication, acidification, ozone depletion and photochemical smog	GHG balance
Selected results				
GHG intensity	No LUC (g CO_2eq FU^{-1}): 139 LUC (g CO_2eq FU^{-1}): 795–1213 (assuming 1) and 533–784 (assuming 2)	0.3 (CUR, train, Argentina) to 3.1 (B, truck, Netherlands) kg CO_2eq FU^{-1} CUR, train, Argentina (kg CO_2eq FU^{-1}): 0.3–0.4 (mass, energy & price allocation) & -4.7–9.0 (substitution approaches)	344(RSO)–721 (PO) g CO_2eq FU^{-1} Including LUC: 5700 (PO) g CO_2eq FU^{-1}	33 (Cerrado) & 287(rainforest) t CO_2eq ha^{-1}
Carbon payback time				37 (Cerrado) & 319 (rainforest) years
Energy				
Photochemical oxidation			0.4(RSO)–0.4(PO) g ethaneeq FU^{-1}	
Acidification			-1.2(RSO)–3.1(PO) g SO_2eq FU^{-1}	
Human toxicity				
Eutrophication			-2(RSO)– -81(PO) g NO_3eq FU^{-1}	
Abiotic depletion				
Ozone depletion			0.20(RSO)–0.27(PO) mgCFC11eq FU^{-1}	

Table 2.2. (Continued)

Relevant data, choices and assumptions	**Reijnders and Huijbregts, 2008b**	**Miller et al., 2007**
Geographical scale	Brazil	USA
System boundaries	WtW	WtG
Functional Unit	1 kg SME	100000 $m^2$of rolled aluminum (soybean-based lubricant)
Multifunctionality approach	Economic allocation	Mass allocation
Type of LCA	Attributional	Attributional
Land-use practices / production options	Tillage vs no-tillage.	-
Nitrogen field emissions	Yes	Yes
LUC	Yes, dLUC due to rainforest and *Cerrado* conversion (10 & 25years)	No
LCIA method	IPCC, 2006	TRACI
Environmental impacts	GHG intensity	Global warming, acidification, eutrophication, photochemical smog formation, human health, fossil fuel use
Selected results		
GHG intensity	13.9–35.2 (rainforest) & 5.4–10.7 (*Cerrado*) kg CO_2eq FU^{-1}	-538 (-889–-288) kg CO_2eq FU^{-1} (including carbon sequestration)
Carbon payback time		
Energy		
Photochemical oxidation		3.42 (1.95–5.88) kg NO_xeq FU^{-1}
Acidification		166 (96–285) moles H^+eq FU^{-1}
Human toxicity		
Eutrophication		27.9 (10.7–73.3) kg Neq FU^{-1}
Abiotic depletion		
Ozone depletion		
Terrestrial ecotoxicity		
Freshwater ecotoxicity		
Marine ecotoxicity		

B5, B20, B100: 5%, 20% and 100% of biodiesel blended with fossil diesel; CED: cumulative energy demand; dLUC: direct land-use change; DM: dry matter; EU: European Union; FU: functional unit; n/d: not-distinguishable; GWP: global warming potential; iLUC: indirect land-use change; NER: net energy ratio; NT: no-tillage; RED: renewable energy directive; RT: reduced-tillage; SME: soybean methyl-ester; T: tillage; WtG: well-to-gate; WtT: well-to-tank; WtW: well-to-wheel; YOLL: years of life lost.

[a]ULSD: ultra-low sulfur diesel equivalent liter of fuel. To obtain ULSD equivalent volumes, the volume of any biodiesel blend was multiplied by the ratio of its energy content to the average energy content of the same volume of ULSD.

[b] Displacement approach is also known as the “substitution” or “replacement” approach.

In 5 of the reviewed studies (Hou et al., 2011; Huo et al., 2009; Reinhard and Zah, 2009; van Dam et al., 2009; Dalgaard et al., 2008) a sensitivity analysis of alternative multifunctionality approaches was performed, indicating that the choice of approaches has an important influence on the LCA results. The sensitivity analysis included a comparison of different allocation procedures (according to mass balance, energy content and price of products) and a comparison of allocation and system expansion or substitution approaches. van Dam et al. (2009) showed that price allocation is quite sensitive to price fluctuations while results in substitution can vary according to the product replaced and the data sources used for that product. Hou et al. (2011) showed that very different results for the various impact categories were obtained with energy and mass allocation: for instance, they found that abiotic depletion in the LC of soybean biodiesel increases approximately 102% when applying energy content-based allocation. In turn, Huo et al. (2009) demonstrated that similar GHG intensities of soybean biodiesel were obtained when energy and economic allocation were adopted.

Concerning the **type of study** all reviewed LCAs were attributional, except the consequential studies of Dalgaard et al. (2008) and Reinhard and Zah (2009). These studies demonstrate that identifying the correct products that are substituted by co-products can have a large impact on the results of consequential LCA. Identifying the correct marginal product is often dependent on the judgment of the LCA practitioner, and therefore reduces the certainty of the results. Dalgaard et al. (2008) and Reinhard and Zah (2009) also showed that significant differences on results are obtained when system expansion and allocation were adopted.

The influence of **land-use practices** on the environmental impacts of agricultural products is a challenging issue (Flysjö et al., 2012; Hokazono and Hayashi, 2012; Chamberlain et al., 2011; Knudsen et al., 2010; Basset-Mens et al., 2007). LCA has been applied to assess the environmental impacts of soybean biodiesel but the wide variety of soybean cultivation and processing conditions, as well as the influence of different climate vegetation and soil regions on the results have not been comprehensively addressed in previous researches. As these aspects are greatly dependent on the local where soybean was produced/processed, the regional differentiation (world, country or state level) should be considered in order to achieve more reliable LCA results.

A small number of studies (6 out of 21) addressed alternative land-use practices in order to assess the effects on the results, namely: organic vs conventional (Knudsen et al., 2010) and the use of different tillage systems (no-tillage, reduced-tillage and full-tillage) (Castanheira and Freire, 2013; Middelaar et al., 2013; Lehuger et al., 2009; Panichelli et al., 2009; Reijnders and Huijbregts, 2008b), as well as soybean inoculation with bacteria and avoiding cypermethrin (insecticide) use (Panichelli et al., 2009). In the remaining studies, it is not clear the management practice that was adopted at soybean farm.

Only a few studies (4 out of 21) analyzed the influence of regional differentiation on the LCA results. Kim and Dale (2009) demonstrated that the regional variations in GHG emissions of soybean oil are

significant, showing that farming sites with higher biomass yield, lower nitrogen fertilizer rate and less tillage are more favorable in terms of global warming. Also, van Dam et al. (2009) recommended an improvement of the analysis of the environmental principles for bioenergy chains on a regional level by field data collection, methodology improvement and better insight in relation carbon stock changes with management system and land-use changes.

Castanheira and Freire (2013) showed that GHG intensity of soybean in Brazil vary widely depending on the climate region were soybean was produced (tropical and warm temperate moist), while Prudêncio da Silva et al. (2010) demonstrated that GHG intensity of soybean produced in the Central-West region of Brazil is almost the double that obtained for soybean from South region. These findings confirmed that LCA studies involving soybean from countries as Brazil, with a wide range of production practices, soil and climatic conditions that affect agricultural production and its environmental impacts, should take into account the region of origin.

Although the intensity of input use contributes by itself to different LCA results, other factors, such as **land-use change** (LUC) and **transportation**, contribute even more (Prudêncio da Silva et al., 2010). Eleven reviewed studies accounted for the carbon stock changes due to LUC and a wide range of results was reported (e.g. Castanheira and Freire, 2013; Middelaar et al., 2013; Prudêncio da Silva et al., 2010; Kim and Dale, 2009; Panichelli et al., 2009; Reinhard and Zah, 2009; Searchinger and Heimlich, 2009; van Dam et al., 2009; Dalgaard et al., 2008; Fargione et al., 2008; Reijnders and Huijbregts, 2008b). The differences in the results are mostly related to LUC modeling assumptions, namely: i) the LUC area, ii) previous land-use, iii) the duration of land-use for soybean production (e.g. 10 or 25 years) and iv) LUC location. Although the wide range of results, these studies showed that the LUC emissions are the highest when forests are converted and the lowest when grassland (or Cerrado/savanna) was converted in soybean plantations.

Four studies assessed alternative scenarios for soybean or soybean-based products transportation, demonstrating that both the mode of transport chosen and the distance to be traveled greatly influence the environmental impacts. The importance of transportation phase is related to the large demand of these products of European countries. These studies showed that the transportation phase makes an important contribution for the LC environmental impacts of soybean-based products; for instance Prudêncio da Silva et al. (2010) showed that transportation is responsible for 30-40% of climate change, acidification and cumulative energy demand impacts calculated for soybean. Castanheira and Freire (2013) also demonstrated that 40-60% of GHG intensity of soybean was due to transportation emissions, which are highest in Brazil than in Argentina due to the greater road transport distances in Brazil.

The nitrogen sources for soybean cultivation include fertilizer (mineral or manure), crop residues, biological fixation of atmospheric N_2 by *Rhizobium sp*. bacteria associated with soybean (legume crop)

and mineralization associated with loss of soil organic matter resulting from land-use change (IPCC, 2006). These N sources can result in nitrous oxide (N_2O), nitrates (NO_3^-), ammonia (NH_3) and nitrogen oxides (NO_x) emissions that contribute to several environmental impacts. However, **nitrogen field emissions** from soybean cultivation are very site-specific and the calculation is complex and controversial (e.g. Castanheira and Freire, 2013; Reijnders and Huijbregts, 2011; Erisman et al., 2010; Del Grosso et al., 2009; Smeets et al., 2009; Snyder et al., 2009; Smaling et al., 2008; Miller et al., 2006).

Almost all reviewed studies calculated nitrogen emissions from soybean cultivation and recognized that nutrient fluxes are a significant issue; however, they focus primarily on air emissions and have not quantified aqueous emissions in a comprehensive manner. Only one study performed a sensitivity analysis of nitrogen calculation methods (Middelaar et al., 2013) and two have assessed how the uncertainties in N_2O emission calculations based on IPCC Tier 1 methodology influences the soybean GHG balance (Castanheira and Freire, 2013; Knudsen et al., 2010). Results showed that using more detailed methods to compute nitrogen emissions from cultivation hardly affected the environmental impacts of soybean, since those methods consider the local environmental and technical uniqueness (e.g. soil clay content, precipitation, root depth, differences in fertilizers used).

Despite of most of the reviewed LCAs of soybean-based products focused on the GHG intensity, together with energy and fossil fuel use, ten studies have accounted for other **environmental impact categories**, such as eutrophication, acidification and toxicity originate from pesticides and fertilizers application (Rosenbaum et al., 2008). The IPCC guidelines are used in quantification of the global warming potential (GWP) in almost all studies; however, not all studies refer to the same version of the IPCC global warming equivalent factors (IPCC, 2001, 2006).

Various LCIA methods were used in the reviewed studies: i) CML (2001, 2007) by Hou et al. (2011), Prudêncio da Silva et al. (2010), Lehuger et al. (2009), Panichelli et al. (2009), Reinhard and Zah (2009) (also presented the results of end-point assessments using the Swiss ecological scarcity method 2006); ii) TRACI by Miller et al. (2007) and Xue et al. (2012); iii) EDIP97 by Dalgaard et al. (2008) and Knudsen et al. (2010); iv) Morais et al. (2010) adopted the characterization and normalization models proposed by Pennington et al. (2004). No articles were found using the recent ReCiPe and USEtox methods.

As shown in **sub-chapter 2.3.1** for palm biodiesel, the LCIA results vary depending on the chosen method making it difficult to compare the results from the reviewed studies. The results demonstrate the importance of LUC to the GHG intensity of soybean-based products. When LUC is not considered, soybean cultivation is the LC phase that contributes most to the GHG intensity of soybean biodiesel, mostly due to diesel consumption and N_2O emissions (e.g. Buratti et al., 2012; Hou et al., 2011). Nevertheless, in the LCA studies for soybean (Castanheira and Freire, 2013; Knudsen et al., 2010; Prudêncio da Silva et al., 2010) and when LUC is not considered, the main contribution to the GHG

intensity came from the transport stage: about 51% (Knudsen et al., 2010), 30-40% (Prudêncio da Silva et al., 2010) and 47-60% (Castanheira and Freire, 2013) of the total result.

Acidification increased 40-53% when the emissions related to soybean transportation from Brazil to Netherlands were included (Prudêncio da Silva et al., 2010), while the transportation phase was also the main contributor (65%) to the acidification potential of soybean imported by Denmark from China (Knudsen et al., 2010). These findings demonstrated that transportation has a strong influence on various environmental impacts of soybean imported by European countries (from Brazil, Argentina or China). For this reason, a comprehensive assessment of the effects of importing soybean as grain, soybean oil or soybean biodiesel should be performed.

2.4 SUSTAINABILITY OF BIOFUEL: EU LEGISLATION

Although the quantification of the environmental sustainability of biofuel is complex, several initiatives have been started by governments, industry players and civil society to define sustainability criteria for biofuels. In June 2009 the European Parliament adopted the Directive 2009/28/EC on the Promotion of Renewable Energy Sources (Renewable Energy Directive, RED) (EC, 2009). The 2009 Directive establish that the share of energy from renewable sources in all forms of transport has to be at least 10% of the final consumption of energy in transport by 2020 and sets out sustainability criteria for biofuels, in particular concerning GHG savings and the impacts of their production on biodiversity, water resources, water quality and soil quality (EC, 2009).

Among the most important sustainability criteria, the GHG emission savings from the use of biofuels compared to fossil fuels have to be at least 35% (until the end of 2016), taking into account emissions from whole biofuel production and distribution chain, including the emissions from carbon stock changes caused by direct LUC. From 2017 onwards these savings have to be at least 50% and from 2018 onwards 60% for biofuels produced in installations that start their production in 2017 or later. Biofuels not fulfilling these newly formulated sustainability criteria may not be taken into account for: i) calculating the shares of energy from renewable sources; ii) measuring compliance with the targets set in the Directives; iii) the eligibility for financial support for biofuels.

The RED indicates the reference GHG emission values (typical and default values) for each LC phase (except LUC) and for the entire biofuel production chains. Typical value is an estimate of the representative GHG emission for a particular biofuel production pathway, while default value is calculated by multiplying typical value for processing emissions by 1.4 (EC, 2010a). Economic operators can use default values or own actual data to demonstrate that their products satisfy GHG thresholds by applying the GHG saving calculation methodology described by RED.

The relative reduction in the GHG emissions by replacing fossil fuel by certain biofuel is calculated based on the total GHG emissions of biofuel (considering all the different production phases and the savings from soil carbon accumulation via improved agricultural management, from carbon capture and replacement or from excess electricity from cogeneration) and the total emissions from the fossil fuel comparator (83.8 g CO_2eq MJ^{-1}) (EC, 2009). The RED also states that the allocation of emissions between the products inside the system boundary shall be carried out in proportion to the energy content of the products (determined by lower heating value in the case of co-products other than electricity).

The typical and default GHG emissions of each LC phase and GHG savings from replacing fossil diesel with palm (process not specified) and soybean biodiesel defined in the RED are presented in **Table 2.3**. It can be seen that the default GHG savings presented in the RED for soybean and palm (process not specified) biodiesel do not meet the GHG saving criteria for biodiesel stipulated by the RED (35%). In order to verify the consistency of the GHG emissions and GHG savings presented in the RED, they should be compared with own actual GHG emissions of palm and soybean biodiesel production in Europe (Portugal), which is the aim of this thesis.

Table 2.3. Greenhouse gas emissions and savings of soybean and palm biodiesel: typical and default values from the RED.

		Greenhouse gas emissions (g CO_2eq MJ^{-1})	
		Typical	**Default**
Soybean biodiesel	Cultivation	19	19
	Processing	18	26
	Transportation	13	13
	Total	***50***	***58***
Palm oil biodiesel	Cultivation	14	14
	Processing, not specified (with methane capture at oil mill)	35 (13)	49 (18)
	Transportation	5	5
	Total	***54 (32)***	***68 (37)***
		Greenhouse gas emission saving	
		Typical	**Default**
Soybean biodiesel		40%	31%
Palm oil biodiesel	Process not specified	36%	19%
	Process with methane capture at oil mill	62%	56%

2.5 CONCLUDING REMARKS

The state of the art of the environmental sustainability assessment of soybean and palm biodiesel was presented in this chapter and included: i) an overview of soybean and palm biodiesel production, ii) an introduction to the LCA framework and a description of key modeling and methodological issues, iii) the main findings from a literature review of the LCA of palm and soybean biodiesel systems and iv) a brief presentation of the EU legislation on the sustainability criteria for biodiesel.

The literature review of 45 studies demonstrated that LCA results of soybean and palm biodiesel systems vary widely due to several issues, namely: i) the uncertainty of soil emissions, in particular carbon dioxide (CO_2) and nitrous oxide (N_2O) emissions from LUC and cultivation; ii) the diversity of land-use and processing practices; iii) dealing with co-products of biodiesel chains and iv) LCIA methods adopted. These results highlighted the need for further research on the life-cycle modeling of palm and soybean biodiesel systems considering the main critical issues based on a scenarios analysis and on different modeling choices.

3 MODELING SOYBEAN AND PALM BIODIESEL SYSTEMS

3.1 INTRODUCTION

This chapter describes the life-cycle (LC) modeling and inventories of biodiesel produced with soybean and palm oil from South America, as well as the scenarios and modeling choices. **Sub-chapter 3.2** presents an overview of the biodiesel chains adopted in this research, including a description of the systems, an outline of the criteria analyzed in each biodiesel chain, the model adopted for data collection, the major data sources and the system boundaries and functional unit. The methodologies adopted in the calculation of nitrogen, phosphorus, heavy metals and pesticides field emissions from crops cultivation, as well as the calculation approach adopted for carbon stock changes due to land-use change (LUC), are described in **sub-chapter 3.3**. The main inputs and outputs of each production phase and biodiesel chains are presented in **sub-chapters 3.4 to 3.9**. In **sub-chapter 3.10** the multifunctionality approaches adopted are presented and explained.

3.2 OVERVIEW OF BIODIESEL CHAINS

In order to study the main environmental impacts of complex and multifunctional biodiesel systems, accounting for spatial differentiation and alternative LUC, land-use practices, production options and pathways, three biodiesel chains were defined: A) Biodiesel produced in Portugal based on palm oil imported from Colombia, B) Biodiesel produced in Portugal based on soybean imported from Brazil and Argentina and C) Biodiesel produced in Brazil and Portugal based on soybean cultivated in four Brazilian states (Mato Grosso, Goiás, Paraná and Rio Grande do Sul). In biodiesel chain C the influence of the location of the oil extraction and biodiesel production mills was assessed based on three alternative pathways: C1) biodiesel totally produced in Brazil and exported to Portugal (BR-BR-BR); C2) biodiesel production (transesterification) in Portugal using soybean oil imported from Brazil (BR-BR-PT); C3) biodiesel production and oil extraction and refining in Portugal using soybean imported from Brazil (BR-PT-PT).

Figure 3.1 shows the LC phases of the three biodiesel chains: LUC, soybean and palm cultivation, oil extraction and refining, biodiesel production (methyl transesterification) and final distribution to the fuel blending facility. Indirect LUC emissions were not addressed since they were out of the scope of this research and there is no consensus on how to account for this (EC, 2010b), as described in **sub-chapter 2.2.1.2**. The basic function of the biodiesel systems is providing energy, which means that the functional unit (FU) adopted was one megajoule (MJ) of biodiesel energy content (measured in terms of the lower heating value, LHV, 37 MJ kg^{-1}) delivered to a fuel blending facility in Portugal. The FU corresponds to 27.1 g of biodiesel with a density of 0.875 kg L^{-1} and does not distinguish between the different types of biodiesel (e.g., soybean or palm biodiesel).

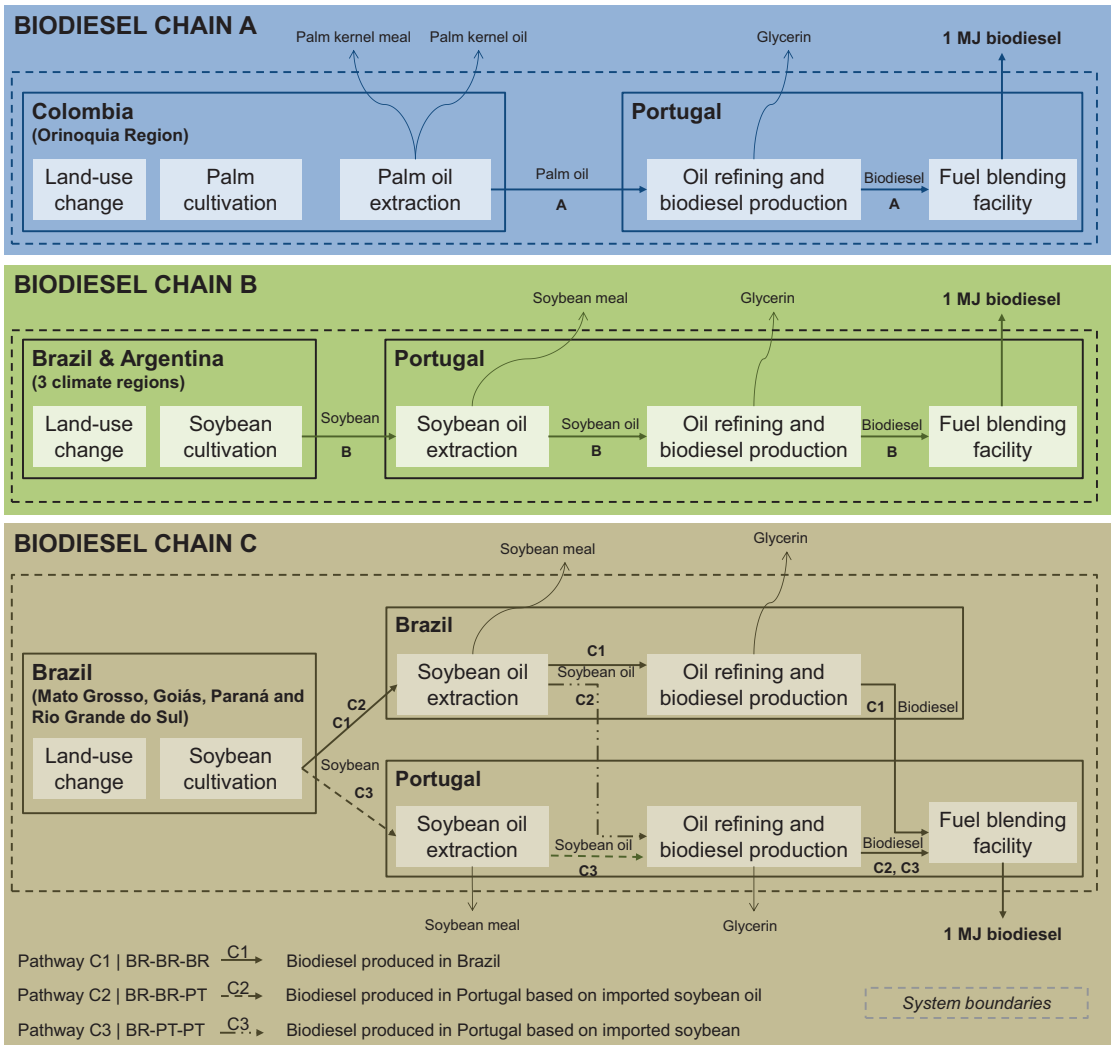


Figure 3.1. Overview of the three biodiesel chains.

The three biodiesel chains were selected in order to achieve the objectives defined for each research question. **Table 3.1** shows how the research questions will be answered by the adoption of these

chains. The rationales for studying Colombian palm oil-to-biodiesel and Brazilian and Argentinean soybean-to-biodiesel were:

– The relevance in terms of oil world production, since soybean and palm oil together represent more than 60% of the world vegetable oil production (USDA, 2013a).

– The importance of these countries. Approximately 1 million tonnes of palm oil were produced in Colombia in 2013, became the largest palm oil producer in the Americas and the fourth producer in the world. Brazil and Argentina were the second and third world producers of soybean in 2013, producing 88 and 53.5 million tonnes of soybean, respectively (USDA, 2013b).

– The relevance of these feedstocks for European Union (EU) and Portugal. Soybean and palm oil are the second and the third biodiesel feedstocks in EU (Flach et al., 2012). Soybean oil is the main biodiesel feedstock in Portugal (more than 45% of Portuguese biodiesel was produced from soybean oil in 2011 and 2012) and palm oil represented 12-14% of Portuguese biodiesel production (DGEG, 2012).

– Land-use change occurs and multifunctionality problems exist.

– Spatial differentiation (different climate regions and soil types) at a country level and a state level (for Brazil).

– Different types of crops and land-use practices. Soybean is an annual leguminous crop (cultivated under different tillage systems and in rotation with other crops), which fixes nitrogen (N_2) from the atmosphere (N-fixing crop) and thus there is no relevant nitrogen input. Also, the herbicides used in soybean plantations have been increased due to the broadening use of Roundup Ready seeds (genetically engineered crops that are resistant to their herbicide Roundup) (Meyer and Cederberg, 2010). On the other hand, palm is a perennial crop, with a life time of about 20-25 years, which require more nitrogen input (Zimmer, 2010), partly provided by different mineral fertilizers but also by the recycling of biomass (e.g. cutted fronds, empty fruit brunches and palm oil mill effluent).

– Different production options and pathways. Soybean oil is usually obtained by chemical extraction, also called solvent extraction (Schmidt, 2007), while most of the palm oil mills use the mechanical or physical milling processes which do not involve the use of chemicals (Lee and Ofori-Boateng, 2013). Additionally, since the quality of the palm oil produced depends on the time interval between harvesting and sterilization (the first stage of extraction), palm fruit (fresh fruit bunches) must be transported as soon as possible after harvesting and the distance from plantation to milling site must be short (Lee and Ofori-Boateng, 2013). On the opposite, soybean is widely traded as grain or as oil.

Table 3.1. How the research questions are answered by the application of LCA methodology to the selected biodiesel chains (A, B and C).

RESEARCH QUESTION	BIODIESEL CHAIN		
	A **Biodiesel produced in Portugal based on palm oil imported from Colombia**	**B** **Biodiesel produced in Portugal based on soybean imported from Brazil and Argentina**	**C** **Biodiesel produced in Brazil and Portugal based on soybean cultivated in four Brazilian states**
1. How can we account for the effects associated with direct LUC in the LCA of biodiesel?	Carbon stock changes due to LUC were calculated based on: – 12 scenarios (it was assumed that 100% of palm area was converted from a previous land-use) – The expansion of the Colombian palm area from 1990 to 2010 and historical data of vegetation cleared	Carbon stock changes due to LUC were calculated based on 45 scenarios. It was assumed that 100% of soybean area in 3 different climate regions was converted from a previous land-use	Carbon stock changes due to LUC were calculated based on the area variation of the different land-uses and the soybean from 1985 to 2006, in four states in Brazil (Mato Grosso, Goiás, Paraná and Rio Grande do Sul)
2. What are the land-use practices, production schemes and pathways that lead to lower impacts?	– 4 alternative fertilization schemes (3 mineral and 1 organic nitrogen fertilizer) – 2 biogas management options at oil extraction mill (captured and flared or released into the atmosphere) were assessed	– 3 land-use practices (no-tillage, reduced-tillage and tillage) in Brazil and Argentina – Alternative soybean transportation systems were evaluated	– 4 specific inventory data for different Brazilian states (Mato Grosso, Goiás, Paraná and Rio Grande do Sul) – Different pathways to produce biodiesel in Portugal (import feedstock, oil or biodiesel from Brazil?) were assessed
3. Are the environmental impacts of biodiesel influenced by the emission calculation approach and LCIA method adopted?	– 2 LCIA methods were adopted: CML and ReCiPe – Field nitrogen emissions were calculated based on two approaches (IPCC Tier 1 and site-specific models) – A sensitivity analysis of N_2O field emissions calculation was performed – The influence of time horizon considered for climate change was assessed	– 2 LCIA methods were adopted: CML and ReCiPe – A sensitivity analysis of N_2O field emissions calculation was performed	– LCIA method ReCiPe was adopted – Toxicity impacts calculated using ReCiPe method were compared with USEtox method results – Field nitrogen emissions were calculated based on two approaches (IPCC Tier 1and site-specific models)
4. How does the selected multifunctionality approach influence biodiesel environmental impacts?	Energy allocation was adopted to present the results, but a sensitivity analysis of alternative allocation procedures (mass, price and energy content) and substitution scenarios was performed	Energy allocation was adopted to present the results	Energy allocation was adopted to present the results, but a sensitivity analysis of alternative allocation procedures (mass, price and energy content) and substitution scenarios was performed

In **biodiesel chain A** a LC modeling and inventory was performed for biodiesel produced in Portugal from palm oil imported from Colombia. A comprehensive evaluation was carried out of the implications of alternative LUC scenarios (forest, shrubland, savanna and cropland conversion) and fertilization schemes (three synthetic and one organic nitrogen-fertilizer). The carbon emissions from LUC were also calculated based on the expansion of the Colombian palm area from 1990 to 2010 and on historical data of vegetation cleared for planting new oil palm. Nitrogen field emissions were calculated based on two approaches (IPCC Tier 1 and site-specific models). A sensitivity analysis of field nitrous oxide (N_2O) emission calculation, biogas management options at oil extraction mill and multifunctionality approach (allocation and substitution scenarios) was performed. Different time horizons for GHG intensity calculation were considered (IPCC, 2007): global warming potential (GWP) for a time horizon of 100, 20 and 500 years. Two life-cycle impact assessment (LCIA) methods (ReCiPe and CML) were adopted to determine the extent to which the results are influenced by the method applied.

The purpose of **biodiesel chain B** is to investigate the environmental impacts of biodiesel produced in Portugal, from oil extraction and refining also taking in place in Portugal, based on soybean cultivated in three different climate regions in Brazil and Argentina. A comprehensive evaluation was carried out of alternative LUC scenarios (conversion of tropical forest, forest plantation, perennial crops plantations, savannas and grasslands), cultivation systems (tillage, reduced-tillage and no-tillage) and soybean transportation (from plantations to ports and from ports to Portugal). A sensitivity analysis of field N_2O emissions was also performed. Energy allocation approach and the ReCiPe and CML methods were adopted.

A LC model and inventory of biodiesel produced in Brazil and Portugal based on soybean cultivated in four states in Brazil was implemented in **biodiesel chain C**. The LUC emissions were calculated based on the expansion of the soybean area from 1985 to 2006 in the each state and nitrogen field emissions were calculated based on two approaches (IPCC Tier 1 and site-specific models). The ReCiPe method was adopted for the environmental impact assessment. Results were also calculated using the USEtox method was adopted to determine the extent to which the toxicity impacts are influenced by the method applied. A sensitivity analysis of alternative allocation procedures (mass, price and energy content) and substitution scenarios were performed to evaluate the influence on the results.

The main inputs and yields for palm cultivation and palm oil extraction presented in **sub-chapter 3.4** were obtained from a representative plantation and mill in the Orinoquía Region of Colombia (based on data collected in a joint project between the National University of Colombia and the Center for Industrial Ecology at the University of Coimbra) (Pardo et al., 2006; Santos, 2006). The data for soybean cultivation in Brazil and Argentina was collected from transparent studies providing important quantitative information for the three cultivation systems (tillage, reduced-tillage, no-tillage) in both countries (FNP, 2011; Cavalett and Ortega, 2009, 2010; Ortega et al., 2005; Dalgaard et al., 2008; Panichelli et al., 2009) (**sub-chapter 3.5**). The inputs and yields from soybean cultivation in Mato

Grosso, Goiás, Paraná and Rio Grande do Sul were calculated based on the average production costs (2010-2011 period) given by FNP, 2010, 2011 ("Brazil Agrianual") (**sub-chapter 3.6**). The life-cycle inventories (LCI) of soybean oil extraction were implemented based on data collected from two Portuguese mills (2009 to 2010) and one Brazilian mill (2010) (**sub-chapter 3.7**). The LCI of biodiesel production was developed based on specific data collected in two mills in Brazil and five in Portugal (**sub-chapter 3.8**).

The inventory data for background processes were largely based on the Ecoinvent database (v2.0, Frischknecht et al., 2007a). Emissions associated with the production of agricultural inputs were accounted for using emission factors for limestone (Kellenberger et al., 2007), fertilizers and pesticides (Patyk and Reinhardt, 1997; Nemecek and Kägi, 2007) and diesel (production and combustion) (Nemecek and Kägi, 2007; Jungbluth, 2007). The emissions from chemicals production were calculated based on the emission factors obtained from Jungbluth et al. (2007), Althaus et al. (2007) and Sutter (2007). Emission factors for heat production based on natural gas (Faist Emmenegger et al., 2007), heavy fuel oil (Jungbluth, 2007) and biomass (Bauer, 2007) were adopted for the emission calculations. Emissions from cogeneration were calculated based on the approach presented on EC (2009)[a]. Emissions due to electricity consumption from the grid were calculated based on Frischknecht et al. (2007b) and on the national electricity mixes of:

- Colombia (IEA, 2008): 83% hydro, 10.5% natural gas, 5.4% coal and 1.1% biomass.
- Portugal (ERSE, 2013; REN, 2013) in 2012: 29% coal, 24% wind, 15% hydro, 24% natural gas, 7% biomass, 1% others (e.g., biogas, photovoltaic).
- Brazil (EPE, 2013): 77% hydro, 8% natural gas, 7% bagasse, 3% oil, 3% nuclear, 2% coal.

The emissions from the different transportation modes were calculated based on emissions factors given by Spielmann et al. (2007). Regarding the transport of fertilizers and pesticides from the storehouse to the farms it was considered an average distance of 350 km (by lorry "16-32t", EURO 3[b]). Pesticides and fertilizers are converted into the product weight in order to calculate the requirements for transports: for pesticides a mean active-ingredient content of 50% is used and for fertilizers the average nutrient contents are used, according to Nemecek et al. (2004).

[a]Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009, ANNEX V, Part C, Point 16: "Emission saving from excess electricity from cogeneration shall be taken into account in relation to the excess electricity produced by fuel production systems that use cogeneration except where the fuel used for the cogeneration is a co-product other than an agricultural crop residue. In accounting for that excess electricity, the size of the cogeneration unit shall be assumed to be the minimum necessary for the cogeneration unit to supply the heat that is needed to produce the fuel. The greenhouse gas emission saving associated with that excess electricity shall be taken to be equal to the amount of greenhouse gas that would be emitted when an equal amount of electricity was generated in a power plant using the same fuel as the cogeneration unit."

[b]EURO is the European Union emission standards for vehicles (Directive 98/69/EC).

3.3 FIELD EMISSIONS FROM CROPS CULTIVATION: CALCULATION APPROACHES

3.3.1 NITROGEN EMISSIONS

Four types of field nitrogen (N) emissions due to soybean and palm cultivation were calculated in this thesis: i) ammonia (NH_3) and nitrogen oxides (NO_x) volatilization to air; ii) nitrate (NO_3^-) leaching/runoff to water; iii) nitrous oxide (N_2O) emissions to air. Ammonia emissions from applied fertilizers were calculated based on emission factors for each group of fertilizer (Asman, 1992; Erisman et al., 2010), as presented in **Table 3.2**. NO_x emissions were estimated based on the percentage of NO_x lost for each type fertilizer applied (FAO and IFA, 2001), also presented in **Table 3.2**.

Table 3.2. NH_3 and NO_x emissions from fertilizers application (% N emitted in form of NH_3 and NO_x).

Type of fertilizer	Emission factor (%)	
	NH_3-N (Asman,1992; Erisman et al., 2010)	**NO_x-N** (FAO and IFA, 2001)
Ammonium nitrate	2	0.5
Calcium ammonium nitrate	2	0.6
Urea	15	0.6
Ammonium sulphate	8	0.6
Multi-nutrient fertilizers (e.g. NPK fertilizers)	4	0.5
Organic fertilizers	25	0.4

Nitrate (NO_3^-) emissions referred to the nitrate losses through leaching. NO_3^- emissions were calculated based on the **Equation 3.1**, according to the SQCB-NO3 model (Faist Emmenegger et al., 2009). The regression model adopted relates the NO_3^- leaching to the following parameters: amount of nitrogen fertilizer, amount of nitrogen taken up by the crop, rooting depth and specific values of soil clay content, nitrogen content in soil organic matter and precipitation.

Equation 3.1

$$NO_3^- = 21.37 + \frac{P}{c \times L}\left[(0.0037 \times F_{SN}) + (0.0000601 \times N_{org}) - (0.00362 \times N_{uptake})\right]$$

in which NO_3^- are the leached NO_3-N (kg N (ha*year)$^{-1}$), P is the precipitation plus irrigation (mm year^{-1}), c is the clay content in the soil (%), L is the rooting depth (m), F_{SN} is the synthetic fertilizer N applied to soil (kg N ha^{-1}), N_{org} is the nitrogen in organic matter (kg N ha^{-1}) and N_{uptake} is the nitrogen uptake by

crop (kg N ha^{-1}). Clay content in the soil, rooting depth and N_{uptake} were adopted from Faist Emmenegger et al. (2009) for each crop. Nitrogen in organic matter was calculated based on soil organic carbon and on C/N ratio (a default value of 15 from IPCC, 2006).

The IPCC Tier 1 methodology (IPCC, 2006) was used to calculate direct and indirect N_2O emissions. Direct N_2O emissions occur directly from the soils to which the N is added/released (from anthropogenic N inputs or N mineralization). Indirect N_2O emissions occur through two pathways (IPCC, 2006): i) following volatilization of NH_3 and NO_x from the soil and the subsequent deposition of these gases and their products (NH_4^+ and NO_3^-) to soils and waters and ii) after leaching and runoff of N, mainly as NO_3^-. Direct and indirect N_2O emissions (kg N_2O ha^{-1}) were calculated using **Equation 3.2** and **3.3** (IPCC, 2006):

Equation 3.2

$$N_2O_{Direct} = (F_{SN} + F_{ON} + F_{CR} + F_{SOM}) \times EF_1 \times 44/28$$

Equation 3.3

$$N_2O_{Indirect} = \left[((F_{SN} \times Frac_{GASF}) + (F_{ON} \times Frac_{GASM})) \times EF_4 + ((F_{SN} + F_{ON} + F_{CR} + F_{SOM}) \times Frac_{LEACH} \times EF_5)\right] \times 44/28$$

in which F_{SN} is the annual amount of synthetic fertilizer N applied to soils (kg N ha^{-1}), F_{ON} is the annual amount of organic N-fertilizer applied (kg N ha^{-1}), F_{CR} is the annual amount of N in crop residues (above-ground and below-ground) returned to soils (kg N ha^{-1}), F_{SOM} is the annual amount of N in mineral soils that is mineralized (the process by which organic N in soil organic matter is converted to inorganic forms: NH_4^+ and NO_3^-), in association with loss of soil C from soil organic matter as a result of changes to land-use or management (kg N ha^{-1}). Organic C and N are closely linked in soil organic matter and when soil C is lost through oxidation as a result of LUC, this loss will be accompanied by a simultaneous mineralization of N (IPCC, 2006).

EF_1, EF_4 and EF_5 are the emission factors adopted for N_2O emissions from N additions (kg N_2O-N kg^{-1} N input), from atmospheric deposition of N on soils and water surfaces (kg N_2O-N (kg NH_3–N+NO_x-N volatilized)$^{-1}$) and from N leaching and runoff (kg N_2O–N (kg N leached and runoff)$^{-1}$), respectively. $Frac_{GASF}$ and $Frac_{GASM}$ are the fraction of F_{SN} and F_{ON} that volatilizes as NH_3 and NO_x (kg N volatilized kg^{-1} N applied), respectively. $Frac_{LEACH}$ is the fraction of all N added to/mineralized in managed soils in regions where leaching/runoff occurs that is lost through leaching and runoff (kg N kg^{-1} N additions). Default values and uncertainty ranges (inside brackets) for the emission factors (EF_1, EF_4, EF_5) and the fractions of N that are lost through volatilization ($Frac_{GASF}$ and $Frac_{GASM}$) or leaching/runoff ($Frac_{LEACH}$) given by IPCC (2006) are presented in **Table 3.3**.

Table 3.3. Default values and uncertainty ranges (inside brackets) for the emission factors and the fractions of N that are lost through volatilization or leaching/runoff (IPCC, 2006).

	Value	Units
Fractions that volatilizes and lost through leaching and runoff		
$Frac_{GASF}$	0.1 [0.03-0.3]	kg NH_3-N+NO_x-N kg^{-1} N applied
$Frac_{GASM}$	0.2 [0.05-0.5]	
$Frac_{LEACH}$	0.3 [0.1-0.8]	kg N kg^{-1} N additions
Emission factors		
EF_1	0.01 [0.003-0.03]	kg N_2O-N kg^{-1} N
EF_4	0.01 [0.002-0.05]	kg N_2O-N (kg NH_3-N+kg NO_x-N volatilized)$^{-1}$
EF_5	0.0075 [0.0005-0.025]	kg N_2O-N kg^{-1} N leaching/runoff

It should be noted that the 2006 IPCC guidelines described in the previous paragraphs included significant adjustments to the methodology previously described in the 1996 IPCC guidelines, namely: i) biological nitrogen fixation was removed as a direct source of N_2O (after Rochette and Janzen (2005) concluded that N_2O emissions induced by the growth of legume crops may be estimated solely as a function of the above-ground and below-ground nitrogen inputs from crop residue) and ii) the release of N by the mineralization of soil organic matter as a result of change of land-use or management was included as an additional source.

Table 3.4 presents the different methodologies adopted for N emission calculations. NO_x emissions were calculated based on the emission factors for each group of fertilizer in all biodiesel chains. Direct and indirect N_2O emissions, NH_3 and NO_3^- emissions were calculated based on the IPCC tier 1 methodology (IPCC, 2006) for biodiesel chains A, B and C. It was considered that NH_3 emissions are equal to the difference between the fraction of N that are lost through volatilization ($Frac_{GASF}$ and $Frac_{GASM}$ from IPCC, 2006) and the NO_x emissions, whereas NO_3^- emissions are equal to the fraction that are leaching/runoff ($Frac_{LEACH}$ from IPCC, 2006).

For biodiesel chains A and C, N_2O emissions were also calculated based on the IPCC tier 1 methodology (IPCC, 2006), NO_3^- emissions on the SQCB-NO3 model (Faist Emmenegger et al., 2009), whereas NH_3 emissions on the emission factors for each group of fertilizer (Asman, 1992; Erisman et al., 2010). In these biodiesel chains, indirect N_2O emissions were calculated considering the NO_3^-, NH_3 and NO_x emissions calculated as previously described and using emission factors (EF_4 and EF_5) given by IPCC (2006). It is important to highlight that NH_3 and NO_3^- emissions calculated using specific emission factors for each type of fertilizer and SQCB-NO3 model are more site-specific because the calculation depends on data related to the cultivation (e.g. fertilizer applied, precipitation, soil clay content, rooting depth).

Table 3.4. Methodologies adopted for N field emission calculations in biodiesel chains A, B and C.

Biodiesel chain	A[a]		B[a]	C	
Approach	IPCC	Site-specific models	IPCC	IPCC	Site-specific models
Nitrous oxide (N_2O)	IPCC tier 1 methodology (IPCC, 2006)	IPCC tier 1 methodology (IPCC, 2006)	IPCC tier 1 methodology (IPCC, 2006)	IPCC tier 1 methodology (IPCC, 2006)	IPCC tier 1 methodology (IPCC, 2006)
Ammonia (NH_3)		Specific emission factor for each type of fertilizer (Asman, 1992; Erisman et al., 2010)			Specific emission factor for each type of fertilizer (Asman, 1992; Erisman et al., 2010)
Nitrate (NO_3^-)		SQCB-NO3 model (Faist Emmenegger et al., 2009)			SQCB-NO3 model (Faist Emmenegger et al., 2009)
Nitrogen oxides (NO_x)	FAO and IFA (2001)				

[a] In these chains a sensitivity analysis of the influence of adopting maximum and minimum values for parameters and emission factors in the calculation of field N_2O emissions was performed.

3.3.2 PHOSPHORUS EMISSIONS

The emission model SALCA-P (Prasuhn, 2006) was adopted to calculate the phosphorus emissions from soybean and palm cultivation. Two phosphorus emissions were considered: i) leaching and runoff of soluble phosphate (PO_4) to ground and surface water and ii) erosion of soil particles containing phosphorus (P). **Equation 3.4** (leaching to ground water) and **3.5** (runoff to surface water) were adopted for the phosphate emission calculations. Phosphorus emissions were calculated based on **Equation 3.6**.

Equation 3.4

$$PO_{4\,leach} = P_{gwl} \times F_{gw}$$

Equation 3.5

$$PO_{4\,runoff} = P_{rol} \times F_{ro} = P_{rol} \times \left(1 + \frac{0.2}{0.8} \times P_2O_{5\,\min} + \frac{0.7}{80} \times P_2O_{5sl} + \frac{0.4}{80} \times P_2O_{5man}\right)$$

Equation 3.6

$$P = S_{er} \times P_{cs} \times F_r \times F_{erw} = (R \times k \times LS \times c_1 \times c_2 \times P) \times P_{cs} \times F_r \times F_{erw}$$

in which PO_{4leach} is the quantity of P leached to ground water (kg PO_4–P ha^{-1}), P_{gwl} is the average quantity of P leached to ground water (0.07 kg P ha^{-1}) (Nemecek et al., 2004), F_{gw} is the correction factor for fertilization by slurry (it is assumed to be one because no slurry was applied). $PO_{4runoff}$ is the quantity of P lost through run-off to rivers (kg PO_4–P ha^{-1}), P_{rol} is the average quantity of P lost through runoff (0.175 kg P ha^{-1}) (Nemecek et al., 2004), F_{ro} is the correction factor for fertilization with P (kg P ha^{-1}) and P_2O_{5min}, P_2O_{5sl}, P_2O_{5man} is the quantity of P_2O_5 (kg ha^{-1}) contained in mineral fertilizers, slurry and manure, respectively.

P is the quantity of P emitted through erosion to rivers (kg P $(ha*y)^{-1}$), S_{er} is the quantity of soil eroded (kg $(ha*year)^{-1}$), P_{cs} is the P content in the top of the soil (the value of 0.95 kg P t^{-1}_{soil} was used) (Prasuhn, 2006), F_r is the enrichment factor for P (the value of 1.86 was used) (Wilke and Schaub, 1996) and F_{erw} is the fraction of the eroded soil that reaches the river (the value of 0.2 was used) (Oberholzer et al., 2006). R, k, LS, c_1, c_2 and P are the erosivity (MJ mm ha^{-1} $h^{-1}yr^{-1}$), erodibility (kg h MJ^{-1} mm^{-1}), slope, crop factor, tillage and practice site-specific factors, respectively. R and LS factors were calculated based on Renard and Freimund (1994) and Wischmeier and Smith (1978). Erodibility factor is presented by USDA (1999) for each USDA soil order and c_1, c_2 and P factors were adopted from Faist Emmenegger et al. (2009).

3.3.3 HEAVY METALS AND PESTICIDES

Heavy metals and pesticides soil emissions due to fertilizers and pesticides application were considered only for biodiesel chain C because it was the chain for which the toxicity impacts were assessed. Heavy metals emissions were estimated based on the difference between the inputs of heavy metals contained in the fertilizers (Gabe and Rodella, 1999) and the outputs through harvested soybean (Embrapa, 2010), as suggested by Nemecek and Kägi (2007). Some heavy metals were calculated as heavy metal uptake because the outputs were higher than the inputs. Pesticides applied were assumed to end up as emissions to soil. The amount of pesticides used as inputs were thus considered as emissions to agricultural soil (Nemecek and Kägi, 2007).

3.3.4 LAND-USE CHANGE: CARBON STOCK CHANGES

Carbon stock changes caused by LUC were calculated using **Equations 3.7** to **3.9**, following IPCC Tier 1 methodology, the European Directive 2009/28/EC and the guidelines for the calculation of land carbon stocks (IPCC, 2006; EC, 2009, 2010b). The carbon stock changes were calculated based on the difference between the carbon stock associated with Reference (previous) and Actual land-use

(soybean or palm plantation). Annualized emissions from carbon stock change due to LUC were found after dividing by the time period in which carbon pools are expected to reach equilibrium after land-use conversion (IPCC default: 20 years).

Equation 3.7

$$e_l = \Delta CS \times 44/12 \times 1/20 \times 1/P = (CS_R - CS_A) \times 44/12 \times 1/20 \times 1/P$$

Equation 3.8

$$CS_i = (SOC_i + C_{vegi}) = (SOC_{ST} \times F_{LUi} \times F_{MGi} \times F_{I_i}) + (C_{BM_i} + C_{DOMi})$$

Equation 3.9

$$C_{BMi} = (B_{AGBi} \times CF_{Bi}) + (B_{AGBi} \times CF_{Bi}) \times R$$

in which e_l (t CO_2eq t^{-1} soybean or palm fruit) are the annualized GHG emissions from carbon stock change due to LUC; CS_R (t C ha^{-1}) is the carbon stock associated with the reference (previous) land-use; CS_A (t C ha^{-1}) is the carbon stock associated with the actual land-use (soybean or palm plantation) and P (t soybean or palm fruit ha^{-1} $year^{-1}$) is the productivity. SOC_i (t C ha^{-1}) is the soil organic carbon in the reference (SOC_R) and actual land-use (SOC_A), SOC_{ST} (t C ha^{-1}) is the standard soil organic carbon based on the appropriate climate region and soil type, F_{LU}, F_{MG} and F_I are factors that reflect the difference in SOC_{ST} associated with the type of land-use (F_{LU}), principle management practice (F_{MG}) and different levels of carbon input to soil (F_I).

C_{vegi} (t C ha^{-1}) is the above and below ground vegetation carbon stock in living biomass (C_{BMi}) and in dead organic matter (C_{DOMi}) in the reference (C_{vegR}) and actual land-use (C_{vegA}). B_{AGBi} is the above ground living biomass (t dry matter ha^{-1}), CF_{Bi} is the carbon fraction of dry matter in living biomass (0.47 tonnes of carbon per tonne of dry matter, dm) and R is the ratio of below ground carbon stock in living biomass to aboveground carbon stock in living biomass. C_{DOM} is the sum of carbon stock in dead wood pool and carbon stock in litter and is usually low significance in land conversion for the establishment of crops for the production of biofuels, but should be taken into account at least for closed forests.

3.4 PALM CULTIVATION AND OIL EXTRACTION IN COLOMBIA

A LC model for palm oil addressing LUC, palm cultivation (planting and harvesting) and oil extraction was implemented. A flowchart of crude palm oil (CPO) production is presented in **Figure 3.2**. The main system inputs, products and yields obtained from a representative plantation, equipped with its own mill, in the Orinoquía Region of Colombia are also represented in the flowchart. Some processes were found to be not significant, as shown in other studies (Choo et al., 2011; Wicke et al., 2008), and were not included: oil palm nursery (until 9 month old) and immature plantation (first 2 years after planting the palms), fuel used for land clearing, emission embedded in infrastructure and machinery.

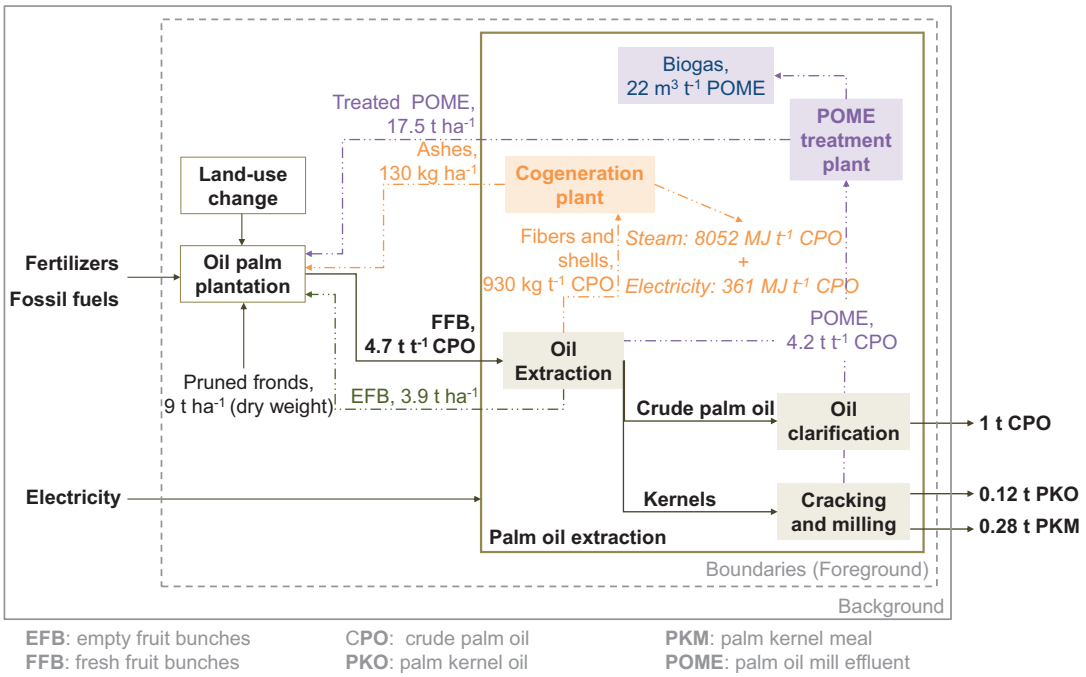


Figure 3.2. Crude palm oil production chain and system boundaries.

The palm plantation had 14000 ha and an average annual yield of 19.5 t ha^{-1} of fresh fruit bunches (FFB). The FFB harvested were transferred to the mill where they were sterilized, stripped, digested into a homogeneous oily mash and pressed to extract most of the crude palm oil. FFB contains 21.5% palm oil, 8.5% kernels, 14% fibers, 6% shells, 20% empty fruit bunches (EFB) and 30% palm oil mill effluent (POME). Kernels are cracked and milled to produce palm kernel oil (PKO) and palm kernel meal (PKM). Fibers and shells were separated from the kernel and used as a fuel in the boiler of cogeneration plant to produce both electricity and steam. POME generated from these processes is treated in anaerobic and stabilization lagoons. Treated POME and EFB were used as a fertilizer in the plantation.

The mature oil palm (20 years) requirements for nutrients were met by the application of fertilizers, pruned fronds and PO mill residues (EFB, treated POME and ashes). About 140 kg N ha^{-1} were applied as fertilizer. To assess the influence of alternative fertilization schemes on the results, it was assumed that different mineral N-fertilizers (#AS ammonium sulphate, #CAN calcium ammonium nitrate and #U urea) or organic N-fertilizer (#Poultry poultry manure) were applied, while yield remained constant. The mineral N-fertilization schemes were adopted because the preferred nitrogen source for palm are ammonium sulphate and urea (Gerendas and Heng, 2010; Von Uexkull, n.d.) and the main N straight fertilizers consumed in Colombia in 2010 were urea (81%), AS (8%), CAN (2%) and others unspecified (9%) (IFA, 2013). Organic fertilization scheme was considered to compare the impact of applying an organic fertilizer and a mineral fertilizer, but also because manure has a lower availability of mineral N (an efficiency of 70% for poultry manure was adopted, i.e., it was considered that 70% of N in poultry manure was available to the palm) (Isitekhale et al., 2013; Gutser et al., 2005).

The main inputs of palm cultivation (fertilizers, residues and fossil fuels for agricultural operations) and emissions from fertilizers and residues application are presented in **Table 3.5** for the four fertilization schemes. Plantation control was made by biological methods, avoiding pesticide use. CO_2 fixed in the urea production process that is released when urea is applied to soil was also calculated (IPCC, 2006).

NO_x emissions were calculated based on the emission factors for each group of fertilizer (FAO and IFA, 2001) and remain N emissions were calculated based on two approaches (**sub-chapter 3.3**):

- Site-specific models: NO_3^- emissions were calculated based on SQCB-NO3 model (Faist Emmenegger et al., 2009), NH_3 emissions on the emission factors for each fertilizer (Asman, 1992; Erisman et al., 2010). N_2O emissions were calculated based on the IPCC tier 1 methodology (IPCC, 2006), but indirect N_2O emissions were calculated considering the NO_3^-, NH_3 and NO_x emissions calculated as previously described instead of the default fractions given by IPCC (2006).
- IPCC: Direct and indirect N_2O emissions, NH_3 and NO_3^- emissions were calculated based on the IPCC tier 1 methodology (IPCC, 2006). NH_3 emissions are equal to the difference between the fraction of N that are lost through volatilization and the NO_x emissions, whereas NO_3^- emissions are equal to the fraction that are leaching/runoff (IPCC, 2006).

A sensitivity analysis of the influence of adopting maximum and minimum values for fractions and emission factors (**Table 3.3**) in the calculation of field N_2O emissions was also performed (the maximum and minimum N_2O emissions are presented inside brackets in **Table 3.5**). The models reported in **sub-chapter 3.3.2** were adopted in the calculation of phosphate (PO_4) and phosphorus (P) emissions. Regarding the nitrate and phosphorous emissions, it was considered an annual precipitation of 2500 mm $year^{-1}$ and clay content in the soil of 53.9% (Oxisol order taken from USDA, 1999). It was also considered a nitrogen uptake of 6 kg N per tonne of FFB harvested (Corley and Tinker, 2003).

Table 3.5. Oil palm cultivation: main inputs and direct emissions of four fertilization schemes.

		Fertilization scheme			
		#AS	#CAN	#U	#Poultry
Inputs					
Nitrogen fertilizer	*Type*	Ammonium sulphate	Calcium ammonium nitrate	Urea	Poultry manure
	Amount (kg N ha^{-1})	140			
Single superphosphate[a] (as kg P_2O_5 ha^{-1})		60			-
Potassium chloride[a] (as kg K_2O ha^{-1})		250			174
Type (amount) of residues applied		EFB[b] (N-content: 0.2%, P-content: 0.024%)			
		Treated POME[c] (N-content: 0.09%, P-content: 0.012%)			
		Pruned fronds[d] (12.2 kg N t^{-1} and 0.9 kg P t^{-1} of fronds, dry weight)			
Diesel (kg ha^{-1})		54.7			
Gasoline (kg ha^{-1})		1.6			
Products					
Fresh fruit bunches, FFB (kg ha^{-1})		19500			
Air emissions					
Ammonia (kg NH_3 ha^{-1})	*Site-specific model*	13.60	3.40	25.50	42.50
	IPCC	19.81	19.81	19.81	37.83
Carbon dioxide (kg CO_2 ha^{-1})		-	-	223.19	-
Nitrous oxide[e] (kg N_2O ha^{-1})	*Site-specific model*	5.60	5.47	5.75	6.03
	IPCC	5.50 [1.33-25.46]	5.50 [1.33-25.46]	5.50 [1.33-25.46]	5.77 [1.34-27.66]
Nitrogen oxides (kg NO_x ha^{-1})		1.80	1.80	1.80	1.20
Water emissions					
Phosphorus (kg P ha^{-1})		1.01	1.01	1.01	1.01
Phosphate (kg PO_4 ha^{-1})		0.91	0.91	0.91	1.15
Nitrate (kg NO_3^- ha^{-1})	*Site-specific model*	235.73	235.73	235.73	235.73
	IPCC	364.20	364.20	364.20	364.20

[a] Single superphosphate and potassium chloride were applied in reduced quantities in #Poultry scheme because phosphorus and potassium needs were partially fulfilled by applying poultry manure.

[b] Heriansyah (2008),[c] Schmidt (2007), [d] Corley and Tinker (2003) and Khalid et al. (2000).

[e]Not include the emissions associated with the amount of N that is mineralized in association with loss of soil C from soil organic matter as a result of land-use changes (F_{SOM}). N_2O emissions related to LUC are presented in **sub-chapter 3.4.1**.

Crude palm oil extraction emissions arise from the production of energy and from POME treatment. Total energy use at the mill was 3547 kg of steam and 186 kWh of electricity per tonne of CPO. Steam was totally produced onsite from the combustion of fibers and shells at the cogeneration plant. Electricity was obtained from the grid (46%) and from the combustion of fibers and shells (54%). The emissions from fibres and shells combustion at the cogeneration plant to produce steam and electricity were adopted from a similar system (wood chips burned in cogeneration plant, Bauer, 2007), but adjustments were implemented according to the dry matter content and low heating value of fibres and shells compared with wood chips.

Regarding the POME treatment, nitrous oxide emissions were estimated based on the nitrogen content of raw POME (0.95 kg N t^{-1} POME) and assuming that 0.1% of the N in POME in the anaerobic lagoons

denitrifies as N_2O (Schmidt, 2007). Biogas produced from POME treatment (22 m^3 t^{-1} POME) is captured and flared; however, before year 2005, biogas was released into the atmosphere (nowadays also occurring in some other mills). Thus, both situations were assessed.

Methane emissions from biogas released into the atmosphere were 9.6 kg CH_4 t^{-1} POME (0.72 kg per m^3), calculated based on: i) the methane content in biogas (60%), ii) the chemical oxygen demand (COD) of untreated POME (60 g L^{-1}, average of measurements at the mill), iii) the efficiency of treatment (COD removal efficiency: 97%) (Lam and Lee, 2011), iv) the average methane emission rate of 0.23 m^3 CH_4 kg^{-1} COD removed (literature values ranged from 0.1 to 0.35 m^3 CH_4 per kg of COD removed, e.g. Siangjaeo et al., 2011; Lam and Lee, 2011). Hydrogen sulphide, N_2O and ammonia emissions from anaerobic digestion of POME were adopted from Schmidt (2007). Methane emissions from biogas flared were calculated considering a flare efficiency of 90% (enclosed flares). Carbon monoxide, sulfur dioxide, N_2O, nitrogen oxides and particulates emissions from biogas flared were also calculated (Environment Canada, 2009; Schmidt, 2007).

3.4.1 LUC EMISSIONS: SCENARIOS AND PALM AREA EXPANSION IN COLOMBIA

Information on LUC in Colombia as a result of oil palm expansion is sparse (Henson et al., 2012). Most of the studies reported that the majority of land converted to oil palm was previously occupied by pastures, savanna, herbaceous vegetation, annual crops, while very little land with high biomass such as forests was used (Rincón, 2009; Rodríguez and van Hoof, 2004). For these reasons, a comprehensive evaluation was carried out of carbon stock changes caused by alternative LUC scenarios. Twelve LUC scenarios (plus no LUC) were defined based on the use of different Reference land-uses (forest, savanna, shrubland and cropland) with different input and management practices.

Table 3.6 describes the LUC scenarios (L0 to L12) and presents the values for the parameters used in the calculation of annualized CO_2 emissions from carbon stock change. It should be noted that for scenarios L2, L3, L5, L6 and L10, C_{vegR} was calculated based on specific B_{AGB} values for Colombian natural regions (Colombian amazon and Orinoquía region, also known as *Llanos Orientales*, characterized by savannas and shrublands) and vegetation indexes (primary forest and savanna) estimated by Anaya et al., 2009.

Table 3.6. Carbon stocks of previous (Reference) land-use and oil palm plantation (Actual): SOC_R, SOC_A, C_{vegR} and C_{vegA}.

LUC scenario	Reference land-use, *R*: description and data sources		SOC[a]					C_{veg}			
			SOC_{ST} (t C ha^{-1})[a]	F_{LU}[a]	F_{MG}[a]	F_I[a]	SOC (t C ha^{-1})	B_{AGB} (t dm ha^{-1})	R[b] (t root dm t^{-1} shoot dm)	C_{DOM}[d] (t C ha^{-1})	C_{veg} (t C ha^{-1})
L0	No LUC										
L1	Forest	Tropical rainforest	60	1	n/a	n/a	60	300[b]	0.37	2.1	195.3
L2		Colombian amazon region	60	1	n/a	n/a	60	291[c]	0.24	2.1	171.7[f]
L3		Primary forest in Colombia	60	1	n/a	n/a	60	264[d]	0.20	2.1	151.0[f]
L4		Forest plantation (Americas *Pinus sp.*)	60	1	1	1	60	-	-	-	87.0[a]
L5	Savanna	Nominally managed savanna in Orinoquía region in Colombia	60	1	1	1	60	37[e]	1.6	-	45.2[f]
L6		Nominally managed savanna in Colombia	60	1	1	1	60	21[d]	0.5	-	14.8[f]
L7		Nominally managed savanna	60	1	1	1	60	-	-	-	8.1[a]
L8		Improved savanna with high input	60	1	1.17	1.1	77.9	-	-	-	8.1[a]
L9		Severally degraded savanna	60	1	0.7	1	42.0	-	-	-	8.1[a]
L10	Shrubland	Orinoquía region in Colombia	60	1	1	1	60	37[e]	2.8	-	66.1[f]
L11		Shrubland	60	1	1	1	60	-	-	-	53.0 [a]
L12	Cropland	Annual cropland (reduced-tillage, medium input)	60	0.48	1.15	1	33.1	-	-	-	0.0[a]
	Actual land-use, *A*										
	Oil palm plantation (reduced-tillage, medium input)		60	1	1.15	1	69	-	-	-	60.0[a]

n/a = not applicable; in these cases F_{MG} and F_I shall not apply and for the calculation of SOC the following rule may be used: $SOC = SOC_{ST} \times F_{LU}$.

[a] Values adopted from EC (2010b)

[b] Data from IPCC (2006)

[c] B_{AGB} calculated by Anaya et al., 2009 for the specific Amazon region Colombia.

[d] B_{AGB} obtained by Anaya et al., 2009 from a model statistics by vegetation class.

[e] B_{AGB} calculated by Anaya et al., 2009 for the specific Orinoquía region in Colombia.

[f] Scenarios L2, L3, L5, L6 and L10: C_{vegR} was calculated based on specific values for Colombia.

Organic carbon (C) and nitrogen (N) are closely linked to soil organic matter. When annualized soil organic carbon stock changes (SOC_R minus SOC_A divided by 20 years) are positive means that carbon is lost through oxidation as a result of LUC and this loss will be accompanied by a simultaneous mineralization of nitrogen. For this reason, N_2O emissions occur when improved savanna with high input (scenario L8) is converted in palm plantation. These direct N_2O emissions were calculated on the basis of the annualized soil organic carbon stock changes of each LUC scenario and a default C:N ratio of 15 (IPCC, 2006).

In addition, the carbon stock change (∆CS) associated with LUC was calculated following the methodology described in **sub-chapter 3.3** and considering that Colombian oil palm area expanded by 84% from 1990 to 2010 (FAO, 2013a), mainly from shrubland (50.7% of the total LUC area), savanna/grassland (41.5%), cropland (6.8%) and forest (1%) (Fedepalma, 2009). The carbon stock of these four reference land-uses are the average of values presented in **Table 3.6**. **Figure 3.3** presents the ∆CS due to the expansion of the palm area in Colombia from 1990 to 2010. In this case, no N_2O emissions occur because the soil organic carbon stock changes are negative as a result of LUC (no N was mineralized).

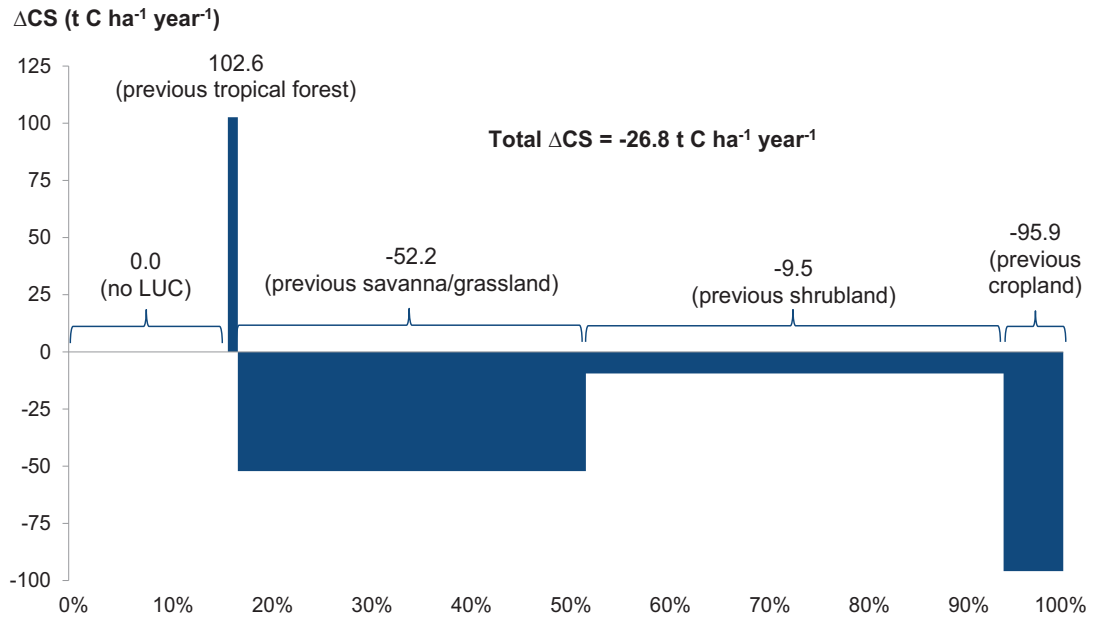


Figure 3.3. Carbon stock change (∆CS) associated with the expansion of palm area in Colombia.

3.5 SOYBEAN CULTIVATION IN BRAZIL AND ARGENTINA

Table 3.7 shows the annual production, main inputs and emissions of three soybean cultivation systems in Brazil and Argentina: no-tillage (NT), reduced tillage (RT) and tillage (T). Tillage (conventional or full tillage) is defined as the soil-related actions necessary for crop production (Boone, 1988), including all operations of seedbed preparation that optimize soil and environmental conditions for seed germination, seedling establishment and crop growth (Lal, 1983). No-tillage is a practice of growing crops from year to year without disturbing the soil through tillage and the crop residues are retained on the soil surface. Reduced tillage is a practice which lies between no-tillage and tillage (Baker et al., 2007). In the last 10 years the no-tillage technology has been expanded worldwide, but the growth of the area under no-till has been especially rapid in South America where the MERCOSUR countries (Argentina, Brazil, Paraguay and Uruguay) are using the system on about 70% of the total cultivated area (Derpsch et al., 2010).

The inputs for NT soybean cultivation in Brazil were based on official data for agricultural operations and inputs for transgenic Roundup Ready (RR) soybean production in Paraná state (FNP, 2011). In Paraná, more than 90% of soybean is RR produced under NT. Regarding the RT system the inputs were adopted from Cavalett and Ortega (2009, 2010) and for soybean cultivation under tillage (intensive system) in Brazil from Ortega et al. (2005). Pesticides use was calculated based on the input data and information on individual trade products, doses and main active ingredients. The type of fertilizers used in soybean plantations was adopted from Brazilian statistics for the fertilizers sector. In all cultivation systems, a residual effect of lime application for 5 years was considered (the values shown in **Table 3.7** are the corresponding annual values).

The main inputs of NT soybean production in Argentina were based on the LCI presented by Dalgaard et al. (2008). Concerning RT and T soybean production in Argentina, the LCI data was adopted from Panichelli et al. (2009), but adjustments were made for soybean yields and pesticides. The yields were calculated for RT (2677 kg ha^{-1}) and T (2248 kg ha^{-1}) based on the average yield of 2591 kg ha^{-1} and the respective RT and T shares in national production (79.9% and 20.1%) (Panichelli et al., 2009). It was also considered that the soybean yield is about 17%-20% higher under RT than T systems, based on information for cultivation in other countries (Opara-Nadi, 1993). Regarding the pesticides, it was considered that pesticide use is higher in RT systems (Deike et al., 2008; Friedrich, 2005), in particular the use of herbicides (2,4D is typically consumed in RT) (Tosi et al., 2005). The use of glyphosate was calculated as the weighted quantity of glyphosate for both systems, considering the national shares of RT and T production systems (79.9% and 20.1%).

Table 3.7. Soybean yield, main inputs and emissions of soybean cultivation systems in 3 climate regions: no-tillage (NT), reduced tillage (RT) and tillage (T).

	Brazil(tropical and warm temperate moist regions)			Argentina(warm temperate dry region)		
	NT[a]	RT[a]	T[a]	NT[a]	RT[a]	T[a]
Inputs						
Pesticides (kg ha^{-1})						
Pesticides, unspecified	0.2	1.1	1.0		0.13	0.13
Sulfonyl [urea-compounds]					0.003	0.003
Organophosphorus-compounds	1.4	1.0	1.2	0.8	0.42	0.42
Pyretroid-compounds	0.01	0.01	0.01	0.02	0.11	0.11
Glyphosate solution	1.0	1.4	1.2	2.6	2.6	1.1
2,4 D	1.2	1.6	1.4	0.3	0.3	
Triazine-compounds					0.01	0.01
Cyclic N-compounds	0.1	0.02	0.02		0.01	0.01
Benzimidazole-compound	0.1	0.01	0.01			
[Thio]carbamate-compound	0.03	0.01	0.01			
Limestone (kg ha^{-1})	40	75	200			
Fertilizers (kg ha^{-1})						
Single super phosphate, as P_2O_5	30	79	30			
Triple super phosphate, as P_2O_5	30			37	5.0	5.0
Monoammonium phosphate, as P_2O_5					5.2	5.2
Potassium chloride, as K_2O	60	79	30			
Potassium sulphate, as K_2O			75			
Diesel (L ha^{-1})	51	54	94	35	35	62
Products						
Soybean (kg ha^{-1})	2940	2830	2400	2630	2677	2248
Air emissions						
Ammonia (kg NH_3 ha^{-1})					0.13	0.13
Carbon dioxide (kg CO_2 ha^{-1})	19.1	35.8	95.3			
Nitrous oxide[b] (kg N_2O ha^{-1})	0.76 [0.19-3.12]	0.74 [0.19-3.04]	0.67 [0.17-2.73]	0.70 [0.18-2.87]	0.71 [0.18-2.93]	0.71 [0.18-2.93]
Nitrogen oxides (kg NO_x ha^{-1})					0.012	0.012
Water emissions						
Phosphorus (kg P ha^{-1})	0.57	1.36	2.26	0.23	0.55	0.92
Phosphate (kg PO_4 ha^{-1})	0.83	0.86	0.79	0.80	0.76	0.76
Nitrate (kg NO_3^- ha^{-1})	52.73	51.40	46.18	48.61	49.07	49.07

[a]NT: No-tillage, RT: Reduced tillage, T: Tillage (conventional or full tillage).

[b] N_2O emissions related to LUC are presented in **sub-chapter 3.5.1**.

Direct and indirect N_2O emissions were calculated based on the IPCC tier 1 methodology (IPCC, 2006) and NO_x emissions on the percentage of NO_x lost due to monoammonium phosphate applied in RT and T systems in Argentina (FAO and IFA, 2001). It was considered that NH_3 emissions are equal to the difference between the fraction of N that are lost through volatilization ($Frac_{GASF}$ from IPCC, 2006) and the NO_x emissions, whereas NO_3^- emissions are equal to the fraction that are leaching/runoff ($Frac_{LEACH}$ from IPCC, 2006). It should be emphasize that synthetic N-fertilizer is applied as monoammonium phosphate only in RT and T systems in Argentina and the amount of N in crop residues was estimated for all systems on the basis of the soybean yield and default factors for above-/below-ground residue

(IPCC, 2006). A sensitivity analysis of field N_2O emissions was implemented (maximum and minimum N_2O emissions presented inside brackets), since there is significant uncertainty regarding the emission factors and partitioning fractions adopted in calculations (IPCC, 2006).

The models reported in **sub-chapter 3.3.2** were adopted in the calculation of phosphate and phosphorus emissions and field CO_2 emissions from limestone application were calculated based on IPCC (2006). Regarding the phosphorous emissions, an average annual precipitation of 1761 mm $year^{-1}$ for Brazil and 940 mm $year^{-1}$ for Argentina was adopted. Oxisols and Mollisols, with a clay content of 53.9% and 21.1% respectively, are the soil orders adopted for Brazil and Argentina (USDA, 1999).

3.5.1 LUC EMISSIONS: SCENARIOS IN BRAZIL AND ARGENTINA

Forty five LUC scenarios were established on the basis of a combination of alternative previous land-uses (conversion of tropical forest land, forest plantations, perennial crop plantations, savanna and grasslands), different cultivation systems (tillage, reduced tillage and no-tillage), climate (tropical moist, and warm temperate, moist and dry) and soil characteristics (low and high activity clay soils). **Figure 3.4** shows the 45 LUC scenarios.

Three climate regions and two soil types were selected because they represent the most important area where soybean is produced in Brazil and Argentina. In Brazil (2009/2010) about 83% of soybean was produced in the Central-West (tropical moist climate) and South (warm temperate moist climate) regions, which are characterized by low activity clay soils (IBGE, 2012; EC, 2010c). In Argentina, approximately 76% of soybean (2009/2010) was produced in the provinces of Buenos Aires, Córdoba and Santa Fé in the Las Pampas region, characterized by a warm temperate dry climate and high activity clay soils (Product Board MVO, 2011; EC, 2010c). Concerning savannas and grasslands conversion, different management options were also included, namely improved management (IM), moderately degraded (MD) and severely degraded (SD).

These scenarios were considered since the soybean area increased significantly during the period 1991-2011 in Brazil (9.6 to 23.9 Mha) and Argentina (4.8 to 18.8 Mha) (FAO, 2013a). Panichelli et al. (2009) showed that in Argentina the expansion of the soybean area from 2000 to 2005 occurred in former cropland (32%), pasture land (27%), savannas (19%) and forests (22%). Regarding soybean expansion in Brazil, Macedo et al. (2012) showed that from 2001 to 2005 this took place in rainforest land (26%) and shrubland (74%) and from 2005 to 2009 mainly in shrubland (91%). Moreover, Dros (2004) forecasted the expansion of the soybean area in Brazil and Argentina up to 2020 as 13.2 Mha in Brazil and 5.4 Mha in Argentina.

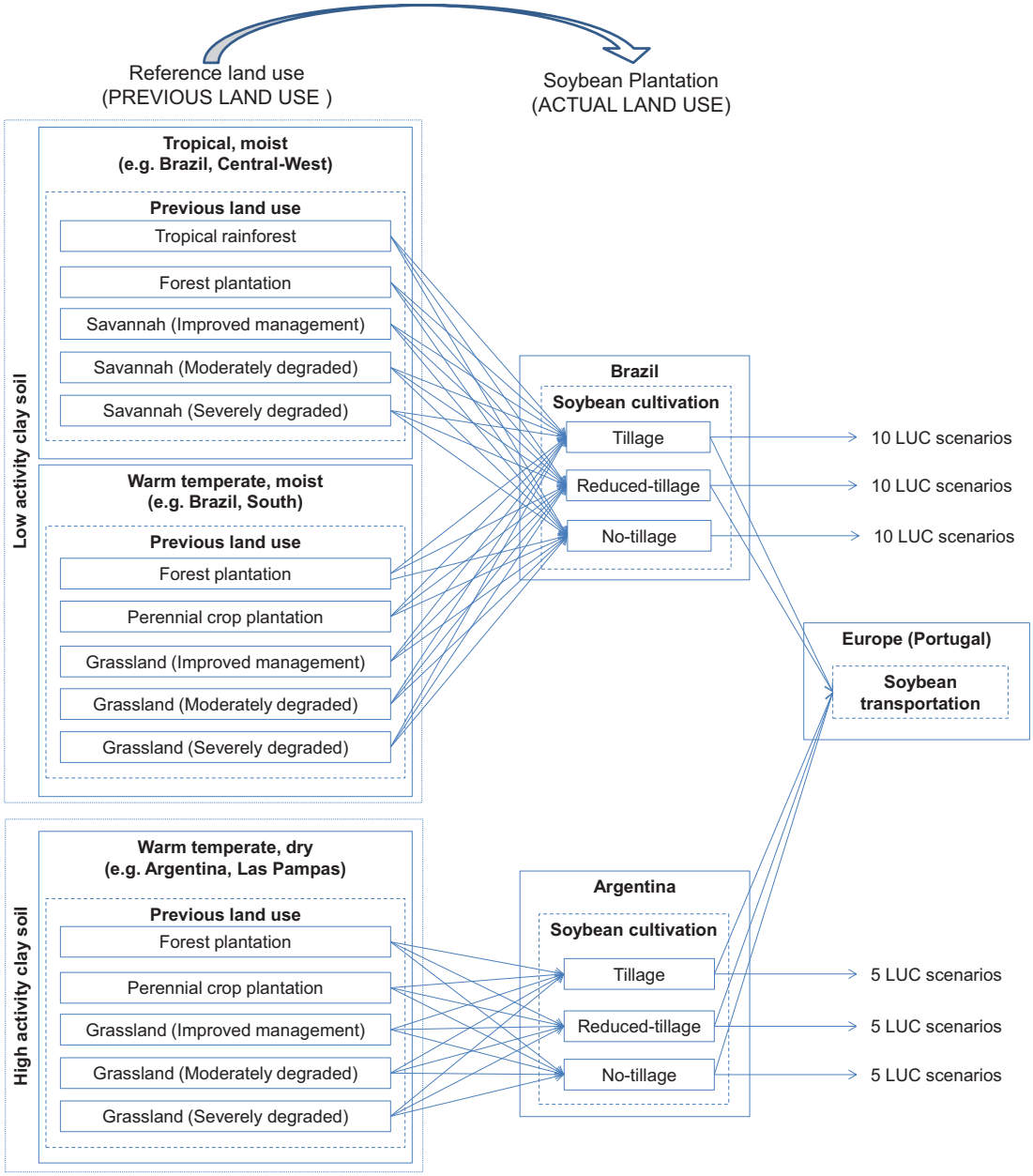


Figure 3.4. Forty five land-use change scenarios due to the expansion of soybean area in 3 climate regions in Brazil and Argentina.

GHG emissions from carbon stock changes caused by LUC were calculated following the IPCC Tier 1 methodology and Renewable Energy Directive (IPCC, 2006; EC, 2009, 2010b), presented in **sub-chapter 3.3.4**. **Table 3.8** presents the SOC_R and SOC_A calculated, as well as the C_{vegR}, C_{vegA} (equal to zero since soybean is harvested annually) and F_{LU}, F_{MG}, F_I factors from the EC (2010b) adopted for the calculations. SOC_{ST} values were selected for the aforementioned climate regions and types of soils. Direct N_2O emissions due to LUC, which were calculated on the basis of the annualized soil organic carbon stock changes of each LUC scenario and on a default C:N ratio of 15, are presented in **Table 3.9** (IPCC, 2006).

Table 3.8. Soil organic carbon and vegetation carbon stocks on previous (reference) and actual (soybean plantation) land-use for 3 climate regions.

Carbon stock of Reference land-uses									
Soil type	**Climate region**	**R: Reference land-use**		**SOC_{ST} (t Cha^{-1})**	**F_{LU}^{h}**	**F_{MG}^{h}**	**F_{I}^{h}**	**SOC_{R} (t Cha^{-1})**	**C_{vegR} (t Cha^{-1})**
Low activity clay soils	*Tropical, moist (Brazil, Central-West)*	*Tropical rainforest*				-	-	47	198.0
		Forest plantation (Eucalyptus sp.)				1.0	1.0	47	58.0
		Savanna (shrubland)	*Improved management*	47	1	1.17	1.11	61	
			Moderately degraded			0.97	1.0	46	53.0
			Severely degraded			0.7	1.0	33	
	Warm temperate, moist (Brazil, South)	*Forest plantation*				1.0	1.0	63	31.0
		Perennial crop (reduced tillage)				1.08	1.0	68	43.2
		Grassland	*Improved management*	63	1	1.14	1.11	80	
			Moderately degraded			0.95	1.0	60	6.8
			Severely degraded			0.7	1.0	44	
High activity clay soils	*Warm temperate, dry (Argentina, Las Pampas)*	*Forest plantation*				1.0	1.0	38	31.0
		Perennial crop (reduced tillage)				1.02	1.0	39	43.2
		Grassland	*Improved management*	38	1	1.14	1.11	48	
			Moderately degraded			0.95	1.0	36	3.1
			Severely degraded			0.7	1.0	27	

Carbon stock of Actual land-use								
Soil type	**Climate region**	**Soybean cultivation system**	**SOC_{ST} (t Cha^{-1})**	**F_{LU}**	**F_{MG}**	**F_{I}**	**SOC_{A} (t Cha^{-1})**	**C_{vegA} (t Cha^{-1})**
Low activity clay soils	*Tropical, moist (Brazil, Central-West)*	*No-tillage*	47	0.48	1.22	1	28	0
		Reduced tillage		0.48	1.15	1	26	
		Tillage		0.48	1.0	1	23	
	Warm temperate, moist (Brazil, South)	*No-tillage*	63	0.69	1.15	1	50	
		Reduced tillage		0.69	1.08	1	47	
		Tillage		0.69	1.0	1	43	
High activity clay soils	*Warm temperate, dry (Argentina, Las Pampas)*	*No-tillage*	38	0.8	1.1	1	33	
		Reduced tillage		0.8	1.02	1	31	
		Tillage		0.8	1.0	1	30	

Table 3.9. Direct N_2O emissions (kg N_2O ha^{-1}) due to N mineralization in the forty five LUC scenarios.

				A: Actual land-use					
				Brazil			Argentina		
				NT	**RT**	**T**	**NT**	**RT**	**T**
R: Reference land-use	*Tropical, moist (Brazil, Central-West)*	*Tropical rainforest*		1.64	1.71	1.83			
		Forest plantation (Eucalyptus sp.)		1.64	1.71	1.83			
		Savanna (shrubland)	*Improved management*	2.38	2.45	2.56			
			Moderately degraded	1.57	1.64	1.75			
			Severely degraded	0.91	0.97	1.09			
	Warm temperate, moist (Brazil, South and Argentina, Las Pampas)	*Forest plantation*		1.31	1.45	1.57	0.81	0.95	0.98
		Perennial crop (reduced tillage)		1.57	1.71	1.83	0.85	0.99	1.02
		Grassland	*Improved management*	2.18	2.32	2.45	1.34	1.47	1.51
			Moderately degraded	1.14	1.28	1.40	0.71	0.85	0.88
			Severely degraded	0.62	0.61	0.58	0.57	0.58	0.58

3.6 SOYBEAN CULTIVATION IN FOUR BRAZILIAN STATES

Table 3.10 presents the inventory for soybean cultivation in the Brazilian states of Mato Grosso (MT), Goiás (GO), Paraná (PR) and Rio Grande do Sul (RS). More than 70% of the total Brazilian soybean was produced in these states between 2009 and 2011 (IBGE, 2012). The inventories were implemented for transgenic Roundup Ready® (RR) soybean, which represents approximately 70% of the Brazilian soybean production in 2009 (James, 2003-2009). The inputs and soybean yields for each state were calculated based on the average production costs (2010-2011 period) given by FNP, 2010, 2011 ("Brazil Agrianual").

Regarding fertilization, "Brazil Agrianual" gives the quantity of N-P-K formulations and following Brazilian fertilizers statistics (ANDA, 2011) it was considered that nitrogen was applied as monoammonium phosphate (MAP), phosphorus as MAP, single superphosphate (SSP) and triple superphosphate (TSP) and potassium chloride (KCl) as potassium oxide (K_2O). Limestone was also added to soils to lower their acidity. A residual effect of lime application for 5 years was assumed and the values shown in **Table 3.10** are the corresponding annual values.

Pesticides inputs were calculated based on the quantity of herbicides, insecticides, fungicides and formicides applied (data from "Brazil Agrianual", which does not specify chemical group or active ingredient) and on data for individual trade products (doses and main active ingredients (a.i.)) of pesticides used in Brazil (MAPA, 2012; IBAMA, 2010; Andrei, 2005). The main herbicides used were glyphosate solution and 2,4-D. Insecticides applied were organophosphorus-compounds (a.i. methamidophos and acephate) and pyrethroids (cypermethrin). Fungicides were benzimidazole-compound (a.i. carbendazim), [Thio]carbamate-compound (a.i. pyraclostrobin) and cyclic N-compounds (a.i. tebuconazole and epoxiconazole). Diesel consumption was calculated based on data for agriculture operations from FNP (2010, 2011) and the specific consumption of machines (Romanelli and Milan, 2010).

Nitrogen oxides (NO_x) emissions were calculated based on FAO and IFA (2001). Ammonia (NH_3), nitrates (NO_3^-) and nitrous oxide (N_2O) emissions were calculated based on two approaches: site-specific models and IPCC. Direct and indirect nitrous oxide (N_2O) emissions were calculated according to IPCC Tier 1 methodology (IPCC, 2006). Using site-specific models, NH_3 emissions were calculated based on the rate of N-volatilization for MAP application (Asman, 1992) and NO_3^- emissions based on Faist Emmenegger et al. (2009). In IPCC approach, NH_3 and NO_3^- emissions were calculated based on the IPCC tier 1 methodology (IPCC, 2006): NH_3 emissions are equal to the difference between the fraction of N that are lost through volatilization and the NO_x emissions, whereas NO_3^- emissions are equal to the fraction that are leaching/runoff.

Table 3.10. Soybean cultivation in four states in Brazil: inputs, products and emissions.

		Mato Grosso (MT)	Goiás (GO)	Paraná (PR)	Rio Grande do Sul (RS)
Inputs					
Fertilizers					
N-fertilizer (kg N ha^{-1})		8	9	0	0
P-fertilizer (kg P_2O_5 ha^{-1})		80	90	60	50
K-fertilizer (kg K_2O ha^{-1})		80	68	60	50
Limestone (kg ha^{-1})		112	112	40	50
Pesticides (kg ha^{-1})					
Pyretroid-compounds		0.003	0.005	0.010	0.003
Benzimidazole-compounds		0.05	0.06	0.06	0.05
[Thio]carbamate-compounds		0.03	0.03	0.03	0.03
Glyphosate		0.99	0.99	0.99	1.13
2,4 D		1.16	1.16	1.16	1.33
Fipronil		0.31	0.20	0.20	0.50
Organophosphorus-compounds		0.40	0.69	1.37	0.39
Cyclic N-compounds		0.10	0.11	0.11	0.09
Diesel (L ha^{-1})		47.6	49.4	55.1	62.8
Products					
Soybean (kg ha^{-1})		2930	2970	2940	2340
Air emissions (per ha)					
Ammonia, kg NH_3	*Site-specific model*	0.39	0.44	0.00	0.00
	IPCC	0.92	1.04	0.00	0.00
Carbon dioxide, kg CO_2		53.4	53.4	19.1	23.8
Nitrous oxide [b], kg N_2O	*Site-specific model*	1.05	1.08	0.93	0.81
	IPCC	1.01	1.04	0.85	0.68
Nitrogen dioxide, kg NO_x		0.09	0.10	0.00	0.00
Water emissions (per ha)					
Nitrate, kg NO_3	*Site-specific model*	86.08	86.17	89.16	96.19
	IPCC	68.90	71.15	58.59	46.63
Phosphate, kg PO_4		0.86	0.87	0.83	0.82
Phosphorus, kg P		0.49	0.48	0.45	0.27
Soil emissions (per ha) [a]					
Cadmium, g Cd		0.14	0.16	0.08	0.07
Lead, g Pb		1.46	2.20	1.33	1.11
Nickel, g Ni		1.07	1.46	0.63	0.52
Cooper, g Cu		-26.96	-27.24	-28.24	-22.25
Zinc, g Zn		-111.0	-111.5	-114.7	-90.7
Chromium, g Cr		6.93	7.12	2.80	3.23
Strontium, g Sr		347.0	379.3	205.0	210.4

[a] The amount of pesticides used as inputs were assumed to end up as emissions to soil and are not shown in table.

[b] N_2O emissions related to LUC are presented in **sub-chapter 3.6.1**.

For the calculation of phosphate (PO_4) leaching and phosphorus (P) emissions through water erosion, the model reported in Faist Emmenegger et al. (2009) was adopted. The soil order of Oxisols (USDA, 1999) and an average annual precipitation of 1610, 1590, 1619 and 1554 mm $year^{-1}$ for MT, GO, PR and RS, respectively, were considered for nitrate and phosphorous emission calculations. Only 30% of nitrogen uptake was considered in order to reflect the fact that, for soybean, 70% of nitrogen are fixed

from the air and are not directly relevant to the balance of nitrogen supplied through fertilizers. Adding carbonates to soils in the form of lime leads to CO_2 emissions as the carbonate limes dissolve and release bicarbonate ($2HCO_3^-$), which evolves into CO_2 and water (H_2O). An emission factor of 0.13 for limestone was considered (IPCC, 2006). Heavy metals and pesticides emissions were estimated based on the approach described in **sub-chapter 3.3.3**.

3.6.1 LUC EMISSIONS DUE TO SOYBEAN AREA EXPANSION

A comprehensive quantification of LUC emissions caused by carbon stock changes due to the expansion of the soybean area was performed for the four Brazilian states, following the methodology presented in **sub-chapter 3.3.4**. Reference land-uses were selected based on the area variation of the different land-uses and soybean from 1985 to 2006, in the each state, as shown in **Table 3.11**. **Table 3.11** shows that soybean area increased significantly in the states of MT, GO and PR (respectively, 80%, 71% and 40% of the soybean area in 2006 were due to LUC) and reduced in RS (no LUC). It was assumed that the Reference land-use in each state corresponds to the land that reduced area in the same period (1985 to 2006): in MT and GO soybean area expanded totally from savanna/shrubland (100%) and in PR from grassland (8% natural and 79% planted) and forest plantations (14%).

Table 3.12 presents the carbon stock changes (ΔCS) due to LUC, calculated based on carbon stocks of Actual (CS_A) and Reference (CS_R) land-use and on the LUC area in each state. The standard soil organic carbon (SOC_{ST}), F_{LU}, F_{MG}, F_I, C_{vegi} were adopted from the EC (2010b) and are also presented in **Table 3.12**. Direct N_2O emissions due to LUC were calculated following IPCC (2006), on the basis of the soil organic carbon stock changes in each state and on a default C:N ratio of 15 (0.51 kg N_2O $ha^{-1}y^{-1}$ in MT and GO, 0.68 kg N_2O ha^{-1} y^{-1} in PR).

As a seasonal crop, soybean provides the farmer the opportunity to plant a second crop each year and, in Brazil, this second crop is generally maize (Achten and Verchot, 2011). For this reason it was assumed that maize and soybean grow in rotation on land that was cleared and half of the yearly carbon loss from LUC should be allocated to the soybean crop. The latter assumption is conservative since soybean is a low yielding crop if compared with corn, contributing relatively little to soil organic carbon sequestration (Reijnders and Huijbregts, 2008b).

Table 3.11. Soybean and land-use area (10^6 ha): years 1985, 2006 and difference (∆), adapted from IBGE (2012).

Land-use	Mato Grosso (MT)			Goiás (GO)			Paraná (PR)			Rio Grande do Sul (RS)		
	1985	2006	∆	1985	2006	∆	1985	2006	∆	1985	2006	∆
Soybean	0,82	4,19	3,36 *+80%*	0,60	2,04	1,44 *+71%*	2,08	3,49	1,41 *+40%*	3,61	3,49	-0,12 *-3%*
Cropland - perennial	0,14	0,40	0,26	0,06	0,25	0,18	0,63	0,98	0,35	0,18	0,29	0,11
Cropland - annual	1,99	5,93	3,93	2,87	3,36	0,49	5,43	5,49	0,06	6,41	6,61	0,20
Grassland – natural [a]	9,69	4,37	-5,32	9,57	3,13	-6,44	1,42	1,31	-0,12	11,94	8,25	-3,69
Grassland - planted	6,72	17,42	10,70	11,32	12,58	1,25	4,58	3,40	-1,18	1,02	0,95	-0,07
Forest - natural	14,13	18,62	4,49	2,83	5,56	2,73	2,01	2,79	0,78	1,66	2,27	0,60
Forest - planted	0,03	0,07	0,04	0,08	0,08	0,00	0,82	0,62	-0,20	0,57	0,78	0,21

[a] include savanna/shrubland

Table 3.12. Carbon stock changes (∆CS) due to LUC in each state: soil organic carbon (SOC_i), carbon in vegetation (C_{vegi}) and carbon stocks (CS_i).

State	Actual and Reference land-use		SOC_{ST} (t Cha^{-1})	F_{LU}	F_{MG}	F_I	SOC_i (t Cha^{-1})	C_{vegi} (t Cha^{-1})	Soybean area expansion (%)	CS_i (t Cha^{-1})	∆CS (t Cha^{-1})
MT[a]	*Actual LU*	*Soybean*	47	0.48	1.22	1	28	0	0.80	22.1	58.2
	Reference LU	*Savanna/shrubland*	47	1	1	1	47	53	0.80	80.3	
GO[a]	*Actual LU*	*Soybean*	47	0.48	1.22	1	28	0	0.71	19.4	51.1
	Reference LU	*Savanna/shrubland*	47	1	1	1	47	53	0.71	70.5	
PR[b]	*Actual LU*	*Soybean*	63	0.48	1.22	1	37	0	0.40	14.9	20.0
		Forest (planted)	63	1	1	1	63	31	0.05	5.2	
	Reference LU	*Grassland (natural)*	63	1	1	1	63	6.8	0.03	2.2	
		Grassland (planted)	63	1	1.14	1.11	80	6.8	0.32	27.6	

[a] Climate region and soil type: tropical, moist and low activity clay soils.

[b] Climate region and soil type: warm temperate, moist and low activity clay soils.

3.7 SOYBEAN OIL EXTRACTION

To produce soybean oil, soybeans are dehulled, cracked, heated, rolled into flakes and solvent-extracted with hexane. The extraction process consists of "washing" the oil from the soybean flakes with hexane solvent. The hexane/soybean oil mix (miscella) is separated from the flakes and transferred to evaporators where the oil and hexane are separated. The flakes leaving the extractor are desolventized, ground to a uniform size and blended with the hulls to produce the soybean meal. The evaporated hexane is recovered and reused in future extraction processes, while the hexane free crude soybean oil is taken for refining. Process flow diagram for crude soybean oil extraction is presented in **Figure 3.5**.

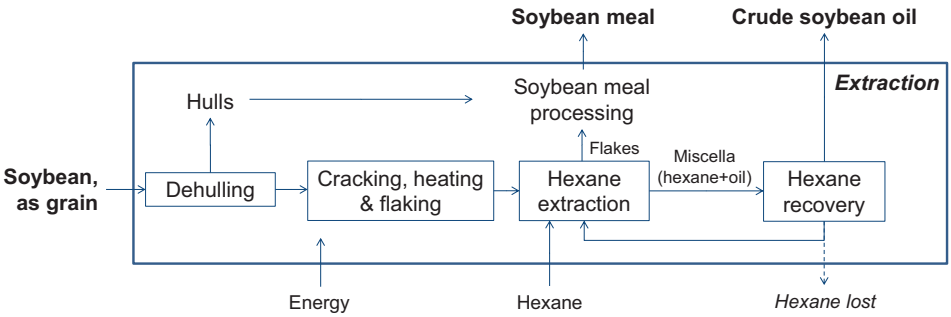


Figure 3.5. Process flow diagram for soybean oil production.

The LCI of soybean oil extraction was developed based on 2009 to 2010 data, collected in two Portuguese mills (Castanheira and Freire, 2011). **Table 3.13** presents the main inputs and outputs of soybean oil extraction in Portugal, which are the weighted average values calculated on the basis of the specific LCI of each mill. Oil extraction data was also collected in a Brazilian mill (2010) (CENBIO, 2013) and is presented in **Table 3.13**. Most of the hexane used in oil extraction is recovered and recycled, with some inevitable loss. Because hexane is volatile, it was assumed that the annual hexane usage is the same that the amount of hexane lost during oil extraction, in the form of hexane emissions to the atmosphere.

Natural gas and heavy fuel oil were used to produce heat in the Portuguese mills, whereas electricity was obtained from the national grid and produced onsite from a natural gas combined heat and power (cogeneration) plant. In Brazilian mill, electricity was obtained from the national grid and biomass (wood) was used to produce heat. Emissions from heat production were adopted from Bauer (2007), but a specific inventory for Eucalyptus forest plantation in Brazil (Silva et al., 2013) was adopted for the calculations.

Table 3.13. Soybean oil extraction in Brazil and Portugal: main inputs and products.

	Extraction (per kg crude soybean oil)	
	Brazil (chain C1 and C2)	**Portugal (chain B and C3)**
Inputs		
Soybean grain (kg)	5.21	5.13
Electricity, from grid (kWh)	0.18	0.16
Heat (natural gas) (MJ)	-	2.61
Heat (fuel oil) (MJ)	-	0.48
Heat (wood, as logs) (MJ)	1.08	-
Hexane [a] (g)	5.73	7.88
Products		
Crude soybean oil (kg)	1.00	1.00
Soybean meal (kg)	4.17	4.06

[a] *It was assumed that the input of hexane was simultaneously an output (emission to air).*

The emissions from transport of hexane, wood and fuel oil to the mills were calculated based on emissions factors for the different transportation modes (Spielmann et al., 2007) and the average distances presented in **Table 3.14**.

Table 3.14. Distances for hexane and fuels transportation to the extraction mills.

	Distances (km)	
	Brazil Lorry "16-32t", EURO 3	**Portugal** Lorry "16-32t", EURO 4
Wood	150	-
Fuel oil	-	100
Hexane	1021	271

3.8 OIL REFINING AND BIODIESEL PRODUCTION

Process flow diagram for crude vegetable oil refining and biodiesel production is presented in **Figure 3.6**. It was assumed that oil refining and biodiesel production occur in same plant. The main processes for refining crude oil when it will be used to manufacture biofuel are neutralization and degumming. The crude oil is degummed to remove phosphatides and is neutralized by treating the oil with aqueous alkaline solution to neutralize the free fatty acids. The soapstock formed in the reaction also adsorbs natural pigments, the gum and mucilaginous substances not removed by degumming. Biodiesel production consists on the transesterification reaction of the triglyceride of the fatty acid in the oil with methanol, catalyzed by a base or acid to produce methyl ester (biodiesel) as main product and glycerin as co-product. The mixture is allowed to separate by gravity and the methyl ester is separated from the glycerin and washed with water until the washing water is neutral. Residual glycerin and unreacted methanol are removed and recovered from the methyl ester, and then the ester is dried.

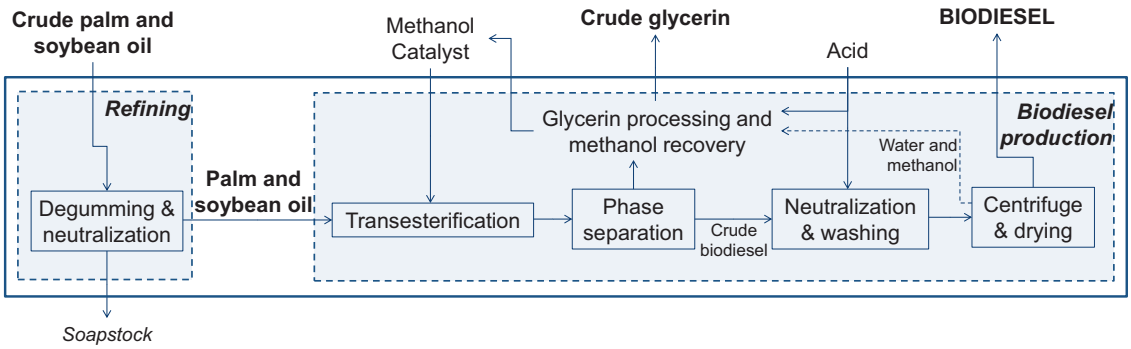


Figure 3.6. Process flow diagram for oil refining and biodiesel production.

Table 3.15 presents the LCI of crude vegetable oil refining, which are the weighted average values calculated on the basis of 2009 to 2010 data, collected in four Portuguese plants. It should be highlighted that in three mills crude oil is neutralized and degummed but in one of the mills the refining process is complete and also includes the bleaching and dewaxing of crude oil. The inventory of biodiesel production was developed based on the data collected in two mills in Brazil (CENBIO, 2013) and five in Portugal (Castanheira and Freire, 2011). The weighted average inputs and products of biodiesel production in both countries are presented in **Table 3.16**. The emissions from chemicals and fuels transportation to the plants were calculated based on emissions factors for the different transportation modes (Spielmann et al., 2007) and the average distances presented in **Table 3.17**.

Table 3.15. Crude vegetable oil refining: main inputs and products.

	Refining (per kg vegetable oil)
Inputs	
Crude vegetable oil (kg)	1.03
Electricity (kWh)	0.01
Heat (natural gas) (MJ)	0.27
Phosphoric acid, 85% in water (g)	1.60
Sodium hydroxide, 50% in water (g)	4.55
Citric acid (g)	0.4
Bleaching earth [a] (also called fuller's earth) (g)	1.2
Products	
Vegetable oil (kg)	1.00

[a] It was not considering in the LCIA

Table 3.16. Biodiesel production in Brazil and Portugal: main inputs and products.

	Biodiesel production (per kg biodiesel)	
	Brazil (chain C1)	Portugal (chain A, B, C2, C3)
Inputs		
Vegetable oil (soybean or palm oil) (kg)	1.00	1.00
Electricity (kWh)	0.03	0.04
Heat (natural gas) (MJ)	-	0.76
Heat (fuel oil) (MJ)	0.11	-
Sodium hydroxide, 50% in water (g)	1.50	0.88
Methanol (g)	109.90	105.47
Sodium methoxide (g)	6.99	5.16
Hydrochloric acid, 30% in water (g)	13.90	10.22
Citric acid (g)	1.71	0.77
Products		
Biodiesel (kg)	1.00	1.00
Glycerin (kg)	0.13	0.12

Table 3.17. Distances for chemicals transportation to the refining and biodiesel plants.

	Distances (km)			
	Brazil		Portugal	
	Lorry "16-32t", EURO 3	Transoceanic freight ship	Lorry "16-32t", EURO 4	Transoceanic freight ship
Fuel oil	100		100	
Sodium hydroxide	250		76	
Methanol	589	6445	182	
Sodium methoxide	250		717	1553
Hydrochloric acid	250		61	
Phosphoric acid			364	
Citric acid	160		148	
Bleaching earth			80	

3.9 TRANSPORTATION OF CROP, OIL AND BIODIESEL

BIODIESEL CHAIN A

The palm oil mill is surrounded by the palm plantation and for this reason it was considered that there is no emissions associated with the transport of fresh fruit bunches from plantation to the mill. It was assumed that crude palm oil (CPO) is transported from the mill to the port of Santa Marta by lorry "16-32t" EURO3 (1300 km) and by transoceanic freighter to the port of Lisbon, in Portugal (7077 km). Regarding transport of CPO from port of Lisbon to refining and biodiesel production plant, a distance of 100 km (lorry "16-32t" EURO4) was assumed. For biodiesel distribution to the fuel blending facility, an average distance of 330 km was calculated (based on data given by five Portuguese companies), using different transport modes: ship (60 km), lorry "16-32t" EURO4 (117 km), train (135 km) and pipeline (18 km).

BIODIESEL CHAIN B

The transportation of soybean from the plantations in Brazil and Argentina to Portugal encompasses transport by lorry ("16-32t") to the ports and by transoceanic freighter to the port of Lisbon (Portugal). The emissions from transoceanic and road transportation were calculated based on emissions factors (Spielmann et al., 2007) and distances between the different places of origin of the soybean and the port of Lisbon. It was assumed that the type of lorry complies with EURO 3. Regarding the transport of soybean from the plantations to the ports, the distances of 1456 km and 403 km were adopted for Brazil and Argentina, respectively. These weighted average distances were calculated based on the distances between the main ports and the main soybean producing locations (IBGE, 2012; SIIA, 2012) presented in **Tables 3.18** (Brazil) and **3.19** (Argentina), as well as the percentage of soybean production and exportation (shown in brackets in **Tables 3.18** and **3.19**) in relation to national production.

The influence of locations on results was assessed based on the use of maximum and minimum distances between plantations and ports. The effect of the type of lorry was analyzed based on the emission factors for eleven types of lorry, using a combination of different capacities (in tonnes) and standards for vehicles (EURO 3, 4, 5 and fleet average): >16t (fleet average), >32t (EURO3, 4, 5), 16-32t (EURO3, 4, 5), 3,5-16t (fleet average), 7,5-16t (EURO3, 4, 5).

The distances from Brazil and Argentina to the port in Portugal were 8447 km and 10244 km, respectively. The distances were estimated on the basis of the distances presented in **Table 3.20** and the quantity exported from each port (the weighted average distance). In Brazil (in 2010), about 85% of soybean was exported from the ports of Santos (25%), Paranaguá (36%), Rio Grande (16%) and Vitória (8%) (Silva, 2010). In Argentina, 75% of the soybean was exported (the average for 2009-2010) from Bahia Blanca (30%), Rosario (24%) and San Lorenzo/San Martin (21%) (MAGyP, 2012).

It was assumed that oil extraction mills in Portugal are located near the ports thus no emissions were considered for soybean transport between the ports and mills. Regarding the transport of crude oil from extraction mill to refining and biodiesel production plant, an average distance of 184 km (95 km by lorry "16-32t" and 89 km by rail) was calculated based on data given by Portuguese companies. For biodiesel distribution to the fuel blending facility, the average distances and transport modes adopted for biodiesel chain A were considered.

Table 3.18. Distances between the main soybean plantation regions and ports in Brazil.

Distances (km)	Mato Grosso - MT (26%)						Goiás - GO (11%)					Paraná - PR (20%)					Rio Grande do Sul - RS (15%)				Weighted average for each port
	Campo Novo do Parecis (5%)	Diamantino (5%)	Nova Mutum (6%)	Sapezal (6%)	Sorriso (10%)	Weighted average for MT to port	Chapadão do Céu (5%)	Cristalina (7%)	Rio Verde (11%)	Jataí (9%)	Weighted average for GO to port	Cascavel (8%)	Goioerê (7%)	Campo Mourão (7%)	Toledo (10%)	Weighted average for PR to port	Santo Ângelo (9%)	Passo Fundo (10%)	Cruz Alta (13%	Weighted average for RS to port	
Santos, São Paulo (25%)	2034	1826	1878	2119	1972	1974	1019	953	986	1034	997	923	859	788	965	892	1178	966	1125	1093	*1340*
Paranaguá, Paraná (36%)	2206	1998	2049	2290	2161	2151	1205	1298	1312	1239	1271	596	627	557	638	607	851	639	797	765	*1299*
Rio Grande, Rio Grande do Sul (16%)	2748	2540	2592	2832	2686	2688	2011	2199	2087	2098	2103	1027	1146	1202	1069	1106	565	574	479	534	*1710*
Vitória, Espírito Santo (8%)	2511	2303	2354	2595	2448	2450	1596	1123	1384	1473	1382	1837	1732	1661	1852	1779	2125	1913	2071	2040	*2015*
						2228					***1358***					***895***				***938***	***1456***

Table 3.19. Distances between the main soybean plantation regions and ports in Argentina.

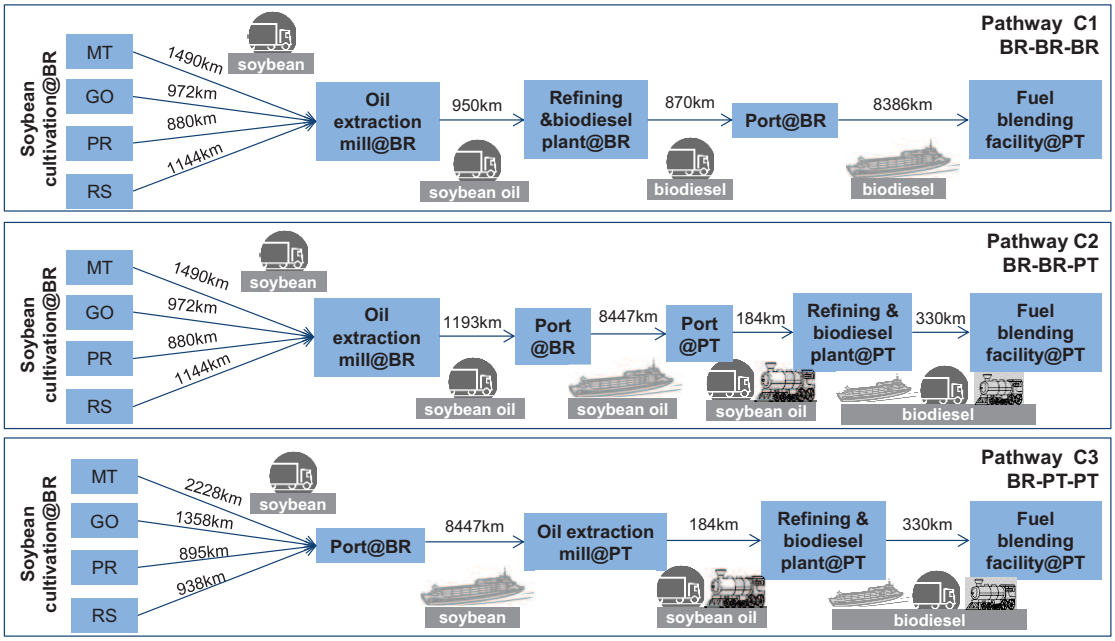
Distances (km)	Buenos Aires (32%)			Córdoba (25%)			Santa Fé (20%)	Weighted average
	General Villegas	Pergamino	Average	Union	Marcos Juarez	Average	General López	
Bahia Blanca (30%)	539	640	590	869	790	830	638	*680*
Rosario (24%)	357	114	236	240	143	192	186	*208*
San Lorenzo/San Martin (21%)	381	143	262	249	152	201	211	*229*
								403

Table 3.20. Distances of transportation of soybean to Portugal from Brazilian and Argentinean ports.

Country	Port	Distance (km) to port of Lisbon (Portugal)
Brazil	*Santos (São Paulo)*	8216
	Paranaguá (Paraná)	8522
	Rio Grande (Rio Grande do Sul)	9185
	Vitória (Espírito Santo)	7357
	Weighted average	**8371**
Argentina	*Bahia Blanca*	10366
	Rosario	10147
	San Lorenzo/San Martin	10179
	Weighted average	**10244**

BIODIESEL CHAIN C

Three alternative pathways were performed based on the location of the main plantations in each Brazilian state and on where oil extraction and biodiesel production take place: C1) biodiesel totally produced in Brazil and exported to Portugal (BR-BR-BR); C2) biodiesel production (transesterification) in Portugal using soybean oil imported from Brazil (BR-BR-PT); C3) biodiesel production and oil extraction and refining in Portugal using soybean imported from Brazil (BR-PT-PT). The distances and types of transport used in each pathway are presented in **Figure 3.7**.



BR: Brazil, PT: Portugal, MT: Mato Grosso, GO: Goiás, PR: Paraná, RS: Rio Grande do Sul

Figure 3.7. Distances and types of transport used in the three alternative pathways for biodiesel chain C.

The transportation of soybean from the plantations in each state to the mills and ports in Brazil encompassed transport by lorry ("16-32t", EURO 3). The weighted average distances presented in **Figure 3.7** were calculated based on the distances between the main soybean producing locations, the

municipalities with higher crushing capacity and the most important ports, as well as the percentage of soybean production, oil extraction and exportation in relation to national production. The distances regarding the transport by lorry ("16-32t", EURO 3) of soybean oil to biodiesel plant and to ports in Brazil, as well as biodiesel to ports, were calculated based on the distances between the municipalities with higher crushing capacity and the two biodiesel plants under study and the most important ports.

The transport of soybean and soybean oil export to Portugal (C3 and C2) encompasses transport by transoceanic freighter to port of Lisbon (Portugal), while in C1 it was assumed that biodiesel was export to port of Sines (Portugal), where the most important fuel blending facility is located. It was assumed that oil extraction mills in Portugal are located near the ports thus no emissions were considered for soybean transport between the port of Lisbon and mills (C3). Regarding the transport of soybean oil from port and extraction mill to refining and biodiesel plant (C2 and C3), an average distance 184 km (95 km by lorry "16-32t" and 89 km by rail) was calculated based on data given by Portuguese companies. For biodiesel distribution to the fuel blending facility, the average distances and transport modes adopted for biodiesel chain A and B were considered.

3.10 MULTIFUNCTIONALITY

As the production of biodiesel involves the generation of other products (e.g., meals, glycerin, soapstock[a], etc.), it is necessary to distribute the environmental impacts between the various co-products. Energy allocation was adopted in the three biodiesel chains (A, B and C), but a sensitivity analysis of alternative allocation procedures and substitution scenarios were performed to chains A and C to evaluate the influence of multifunctionality approaches on the results. Allocation and substitution approaches are explained in the following sub-chapters.

3.10.1 ALLOCATION

Soybean and palm biodiesel systems are multifunctional, with oil and meal produced in the oil extraction mill and glycerin and biodiesel produced in the biodiesel plant. Three allocation procedures were adopted based on physical properties (mass and energy content) and price of products. **Table 3.21** presents the physical properties and prices of products, as well as the allocation factors. Mass allocation factors were calculated based on the mass balance performed for the oil extraction mills in

[a]Soapstock produced in the refining process was not considered in this research because it represents less than 4% of production (in mass) and has a low price.

Colombia, Portugal and Brazil and for the biodiesel plants in Portugal (average value obtained from five companies).

Energy allocation factors were calculated based on the lower heating value (LHV) of products. The LHV was calculated based on the dry matter (given by Fehrenbach et al., 2007), the latent heat of vaporization of water at 25°C and the wet basis moisture content of products. The wet content of the palm kernel meal (11%) were obtained from Fehrenbach et al. (2007), whereas wet content of soybean meal and glycerin (13% and 9%) are average values calculated based on the specific data of Portuguese companies. The wet content of remain products were considered to be zero (Fehrenbach et al., 2007).

Price allocation factors were obtained based on the world average annual prices (US$) of oil and meal (2009-2013 period) (World Bank, 2013; FAO, 2013b). The average annual price of biodiesel (2009-2013 period), in euros per tonne (€ t^{-1}), are fixed by the Portuguese Government (DGEG, 2013) and were adopted in this research. A price of 100 € t^{-1} were adopted for glycerin based on information given by Portuguese companies. The prices of products are presented in **Table 3.22**. To account for price variability, two scenarios were implemented based on the ratio of oil and co-products prices: i) allocation factors calculated based on 2009 prices of palm oil and palm kernel meal and on 2011 prices of soybean oil and meal, when the ratio oil/co-products were the maximum **(Max ratio)** and ii) allocation factors calculated based on 2013 prices of palm oil, palm kernel meal, soybean oil and meal, when the ratio oil/co-products were the minimum **(Min ratio)**.

Table 3.21. Physical properties, price of products and allocation factors for soybean and palm oil biodiesel systems.

Process		Products	Allocation approach								
			Mass	Energy		Market price					
			Allocation factor (%)	LHV (MJ kg^{-1})	Allocation factor (%)	Average (US$ t^{-1}, € t^{-1})	Allocation factor (%)	Max ratio (US$ t^{-1})	Allocation factor (%)	Min ratio (US$ t^{-1})	Allocation factor (%)
Palm oil extraction (CO)		*Crude palm oil*	72	36.5	81	856	83	644	85	761	83
		Palm kernel oil	8	39.0	10	1104	12	700	11	876	11
		Palm kernel meal	20	15.1	9	170	5	102	4	187	6
Soybean oil extraction	*BR*	*Crude soybean oil*	19	36.6	35	1021	38	1216	44	1024	34
		Soybean meal	81	16.3	65	404	62	379	56	476	66
	PT	*Crude soybean oil*	20	36.6	36	1021	38	1216	44	1024	35
		Soybean meal	80	16.3	64	404	62	379	56	476	65
Transesterification (BR & PT)		*Biodiesel*	89	37.0	95	1078	99	-	-	-	-
		Glycerin	11	15.2	5	100	1	-	-	-	-

CO: Colombia; BR: Brazil; PT: Portugal; LHV: lower heating value.

Table 3.22. Prices of products for soybean and palm oil biodiesel systems (from 2009 to 2013).

Year	World prices[a] (US$ t^{-1})					Prices[b], in Portugal (€ t^{-1})	
	Soybean oil	Soybean meal	Palm oil	Palm kernel oil	Palm kernel meal	Biodiesel	Glycerin
2009	787	359	644	700	102	872	
2010	925	331	860	1186	166	955	
2011	1216	379	1077	1649	192	1226	100
2012	1152	473	940	1111	184	1245	
2013	1024	476	761	876	184	1091	

[a] World Bank, 2013; FAO, 2013b

[b] DGEG, 2013

3.10.2 SUBSTITUTION

In the substitution approach soybean and palm oil systems are enlarged to include the additional functions provided by co-production of oil meal. Glycerin is also co-produced in transesterification process, but substitution was not adopted since previous studies showed that different methods have a low influence on the LCA results for biodiesel production (Reinhard and Zah, 2011). In addition, glycerin has several potential uses (e.g. displacing synthetic glycerin, replacing grain as animal feed, processed to generate biogas and electricity), which would make this assessment complex without adding significant information for decision support. For this reason, allocation approach based on biodiesel and glycerin prices (average) was adopted for transesterification process.

BIODIESEL CHAIN A

Palm oil is the product determining the production of palm kernel meal (Schmidt and Weidema, 2008). An increased demand for palm oil biodiesel in Portugal should cause an increased production of palm oil (palm oil denotes further both crude palm oil and palm kernel oil, since they jointly can be considered as the marginal oil on the global market (Schmidt and Weidema, 2008)). This additional oil, in turn, should induce an increased production of palm kernel meal, which was assumed that will substitute soybean meal on the market. Also, the co-produced soybean oil induces an increase in the production of palm oil. Palm meal-soybean meal loop caused by the additional production of 1 MJ of palm oil biodiesel is presented in **Figure 3.8**.

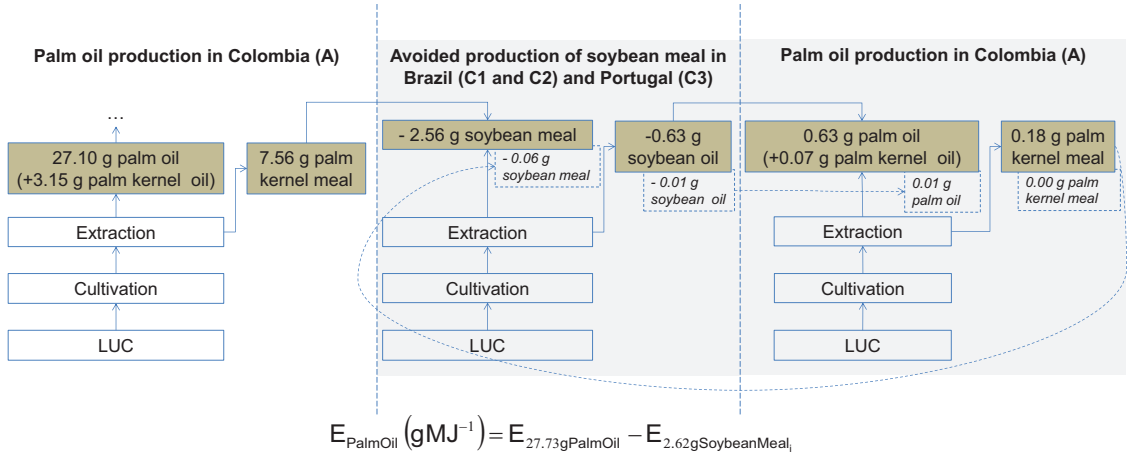


Figure 3.8. Palm meal-soybean meal loop associated to the additional production of 1 MJ of palm oil biodiesel (shadowed boxes show the beginning of second loop).

By iteration, an increased demand for 27.10 g of palm oil (the amount required to produce 1 MJ of palm biodiesel in Portugal) results in the production of 27.73 (=27.1+0.63+0.01) g palm oil from Colombia and -2.62 (=2.56+0.06) g soybean meal produced in Portugal based on soybean imported from Brazil (chain C3). The iteration stopped at the point where the consequences are so small, that any further expansion of the boundaries would yield no significant information for decision support (Ekvall and Weidema, 2004).

For each 1.00 g of palm kernel meal produced there is an avoided production of 0.34 g soybean meal. The amount of soybean meal that substitutes palm kernel meal was calculated on the basis of the protein content (Reinhard and Zah, 2009): 440 g kg^{-1} soybean meal and 149 g kg^{-1} palm kernel meal. Although other factors, such as fatty acid compositions and the energy content, also determine the use of a specific meal, in this research only protein content was taking into account. The substitution relationship prevailing between soybean oil and palm oil is assumed to exist on a one to one ratio (the oils are treated as equivalent, since they are assumed to be substitutable in the most important applications) (Schmidt and Weidema, 2008).

Two substitution scenarios were defined considering the avoided impacts associated to the soybean meal production in Brazil or in Portugal. The soybean meal impacts were adopted from biodiesel chain C, where in soybean cultivation in four states in Brazil (Mato Grosso, Goiás, Paraná and Rio Grande do Sul) and three alternative pathways (C1, C2 and C3) were considered. Based on these results, the lowest and the highest impact of soybean meal obtained in each category were considered, respectively, as the substitution scenario A and B.

BIODIESEL CHAIN C

Soybean meal determines the cultivation of soybean, i.e. soybeans are primarily planted because of the revenues related to soybean meal. Therefore, it is not likely that an increased demand for soybean oil will be compensated for by an increased production of soybean oil (Reinhard and Zah, 2009). Instead, the increased production of soybean biodiesel occurs at the expense of the available soybean oil and the increase in soybean biodiesel production in Portugal avoids additional palm oil production and import. **Figure 3.9** shows the resulting soybean oil-palm oil loop that results when the increased demand for soybean biodiesel in Portugal is met at the expense of the available soybean oil.

To produce 110.17 g soybean meal and 27.10 g soybean oil (amount required to produce 1 MJ of soybean biodiesel in Portugal and Brazil), 137.27 g soybean is needed. Increased soybean meal production involves increased soybean oil (co-)production, causing a decrease in palm oil production (which is a mix of palm oil and palm kernel oil) and consequently in co-produced palm kernel meal. For each 110.17 g of soybean meal produced, there is an avoided production of 7.56 g palm kernel meal, which is substituted by soybean meal based on the protein content (7.56 g of palm kernel meal contains

the same amount of protein as 2.56 g of soybean meal). Consequently, after the first turn in the loop, the result showed that an increased demand for 110.17 g of soybean meal caused a production of 112.79 (=110.17+2.56+0.06) g of soybean meal and -27.74 (=27.10+0.63) g of palm oil. By making this iteration for each turn, the extra amount of soybean meal produced is getting smaller.

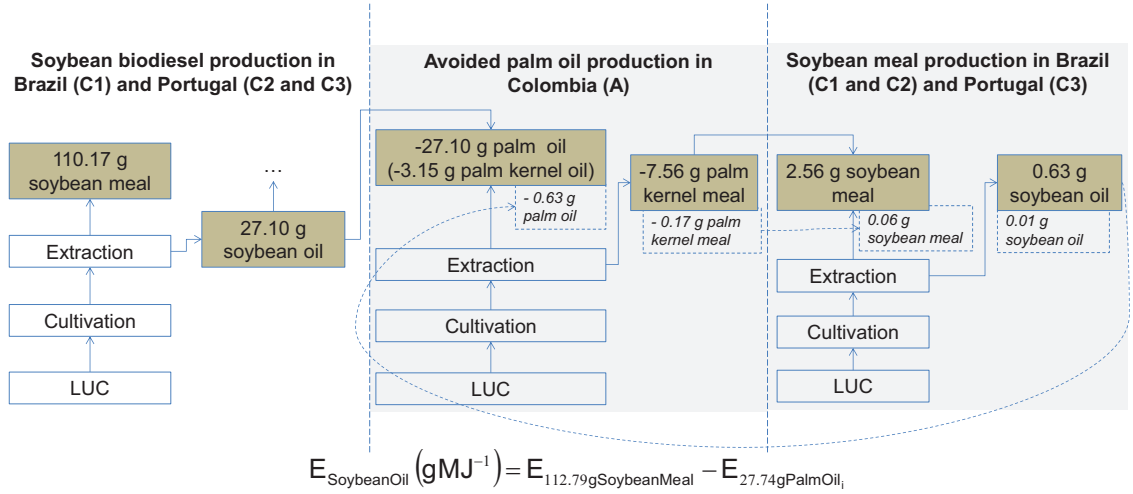


Figure 3.9. Soybean oil-palm oil loop associated to the additional production of 1 MJ of soybean biodiesel (shadowed boxes show the beginning of second loop).

Two substitution scenarios were defined considering the avoided impacts associated to the palm oil production in Colombia (assuming the expansion of the Colombian palm area from 1990 to 2010). The palm oil impacts were adopted from biodiesel chain A, wherein four fertilization schemes (#AS, #CAN, #U and #Poultry) and two POME treatment options (biogas is captured and flared or is released into the atmosphere) were considered. From these results, the lowest and the highest impact of palm oil obtained in each category were considered as the substitution scenario A and B.

3.11 CONCLUDING REMARKS

In this chapter the three biodiesel chains considered in this research was presented: A) Biodiesel produced in Portugal based on palm oil imported from Colombia, B) Biodiesel produced in Portugal based on soybean imported from Brazil and Argentina and C) Biodiesel produced in Brazil and Portugal based on soybean produced in four Brazilian states. The LC modeling and inventories of these chains were presented, including a detailed description of the calculation of carbon stock changes due to the various scenarios for LUC and methodologies adopted for field emissions (nitrogen, phosphorus, heavy metals and pesticides) calculations. The different scenarios for LUC, land-use practices, production

schemes and pathways are also well described. The alternative multifunctionality approaches (mass, energy and price allocation and substitution) adopted to deal with co-products were also presented and explained in this chapter. The LC environmental impacts of soybean and palm biodiesel were calculated based on the LC inventory presented in this chapter.

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4 LIFE-CYCLE IMPACT ASSESSMENT OF BIODIESEL

4.1 PURPOSE AND SCOPE

In this chapter the life-cycle environmental impacts of soybean and palm biodiesel systems are presented and discussed. The results are compared and the parts of such chains that impact the environment the most ("environmental hotspots") are identified. The overall objective is to increase the knowledge about the environmental sustainability of such systems, as well as to identified and discussed the major sources of uncertainty of the assessment. ReCiPe method (midpoint 1.07, hierarchist) (Goedkoop et al., 2012) was adopted to calculate the following impact categories: GHG intensity (climate change/global warming[a]), freshwater and marine eutrophication/eutrophication, ozone depletion/ozone layer depletion, photochemical oxidant formation/photochemical oxidation, terrestrial acidification/acidification, human toxicity and aquatic toxicity.

Sub-chapter 4.2 presents the GHG intensity results, focusing on the contribution of LUC emissions for the three biodiesel chains. For palm oil biodiesel chain (A), the results are calculated based on 12 LUC scenarios (it was assumed that 100% of palm area was converted from a previous land-use) and on the expansion of the Colombian palm area (from 1990 to 2010). In the soybean biodiesel chain B, the results for the 45 LUC scenarios for the expansion of the soybean area in Brazil and Argentina are compared. The specific area variation (from 1985 to 2006) of the different land-uses and soybean area in four states in Brazil is analyzed in biodiesel chain C. The differences on LUC emissions related to modeling assumptions and the major sources of uncertainty of LUC emissions are discussed. The research question 1 *"How can we account for the effects associated with direct LUC in the LCA of biodiesel?"* is answered.

Sub-chapter 4.3 focuses on the effects of various land-use practices, production schemes and pathways on the life-cycle impact assessment (LCIA) results of the three biodiesel chains (A, B and C). Palm biodiesel chain (A) is used to investigate the influence of adopting alternative fertilization schemes

[a] *A global warming potential (GWP) factor of zero was adopted for biogenic CO_2 emissions since the carbon emitted is equal to the carbon fixed by the palm fresh fruit bunches and soybean grains. The net CO_2 from clearing and crop sequestration associated with replanting was considered zero, assuming that there are no difference in growth between successive oil palm and soybean crops (Chase et al., 2012).*

for palm cultivation and biogas management options at oil extraction mill on the LCIA results. The influence of time horizon considered for GHG intensity is also assessed for this chain. The effect of adopting different soybean cultivation systems (no-tillage, reduced-tillage and tillage) in Brazil and Argentina was assessed in soybean biodiesel chain B. The influence of locations of soybean plantations and ports on the GHG intensity of soybean transportation is also assessed in this chain. Moreover, sensitivity analyses to nitrous oxide (N_2O) field emission calculations is performed in chains A and B. Regarding the soybean biodiesel chain C, the environmental consequences of adopting different LC inventories of soybean cultivation in four Brazilian states and different pathways to produce biodiesel in Portugal (import soybean, oil or biodiesel from Brazil?) are evaluated. The influence of the approach adopted for field nitrogen emission calculations (IPCC or site-specific models) is assessed in biodiesel chains A and B. Research question 2 *"What are the land-use practices, production schemes and pathways that lead to lower impacts?"* and research question 3 *"Are the environmental impacts of biodiesel influenced by the emission calculation approach and LCIA method adopted?"* are partially answered in this sub-chapter.

The research question 3 is also answered in **sub-chapter 4.4**, in which the influence of LCIA method on the results is analyzed. Two LCIA methods are adopted in biodiesel chains A and B to determine the extent to which the non-toxicity environmental impacts are influenced by the method apply: ReCiPe 1.07 and CML 2001, v2.05 (Guinée et al., 2002). To compare the environmental impacts, the CML and ReCiPe recommended methods for normalization were adopted in biodiesel chains A and B. The goal of normalization of impact is a better understanding of the magnitude of category indicator results relative to reference information (ISO, 2006b). It allows checking how the impact category contributes to significantly to global environmental problem (Requena et al., 2011). ReCiPe 1.07 and USEtox 1.01 (Rosenbaum et al., 2008) methods are adopted in soybean biodiesel chain C to compare the toxicity impacts.

Research question 4 *"How does the selected multifunctionality approach influence biodiesel environmental impacts?"* is answered in **sub-chapter 4.5**. A sensitivity analysis of alternative multifunctionality approaches is performed to biodiesel chains A and C: allocation based on mass balance, energy content and market prices of products, as well as two substitution scenarios were implemented for each chain. **Sub-chapter 4.6** answer research question 5 *"What are the GHG emission savings when palm and soybean biodiesel replace diesel?"*. The results are compared with the GHG saving criteria stipulated by the Renewable Energy Directive (RED) and the effect of multifunctionality approach on the GHG savings is also analyzed. In **sub-chapter 4.7** the results presented in previous sub-chapters are discussed to better understand what are the best practices for production of soybean and palm biodiesel, the most persistent shortcomings, site-dependency in agricultural LCA and opportunities for improvement.

4.2 THE INFLUENCE OF LUC ON GHG INTENSITY OF BIODIESEL

GHG intensity of soybean and palm biodiesel are presented and discussed in this sub-chapter, focusing on different LUC scenarios which aims to respond to the following objectives:

> *"1.1 Model and calculate the carbon stock changes from direct LUC resulting from the expansion of soybean and oil palm areas in South America."*
>
> *"1.2 Calculate N_2O emissions due to the nitrogen released by the mineralization of soil organic matter, as a result of land-use change."*
>
> *"1.3 Determine the influence of LUC in the GHG intensity of various scenarios for palm and soybean biodiesel production."*

The GHG intensity was calculated using the IPCC approach (IPCC, 2007) adopted in ReCiPe 1.07 method and energy allocation approach. The contribution of LUC and each process for the GHG intensity are presented individually for the biodiesel chains (A, B and C). The remaining environmental impacts are not presented in this sub-chapter since they are not influenced by LUC.

BIODIESEL CHAIN A

GHG intensity of palm biodiesel chain is presented in **Figure 4.1**, considering four fertilization schemes (#AS, #CAN, #U and #Poultry), twelve LUC scenarios (L0 to L12) and the expansion of the Colombian palm area in the 1990 to 2010 period (Expansion 1990-2010). Despite the GHG intensity of palm oil extraction, refining, biodiesel production and transport remain constant (since it was not influenced by the fertilization scheme or LUC), the contribution of each LC phase is also presented in **Figure 4.1**. It should be noted that these results were obtained based on site-specific approach adopted for N_2O emission calculations and considering that biogas from POME treatment was captured and flared (if biogas was released into the atmosphere the total results will increase about 17 g CO_2eq MJ^{-1}).

LUC makes a strong contribution to the results in almost all scenarios, except in scenario L10 (shrubland) in which cultivation is the phase that contributes most to the GHG intensity. The process contribution is as follows: 7 to 80% LUC (-3 to 118 g CO_2eq MJ^{-1}), 9 to 50% cultivation (13 to 17 g CO_2eq MJ^{-1}), 6 to 31% transport (9 g CO_2eq MJ^{-1}), 2 to 8% extraction (2 g CO_2eq MJ^{-1}), 4 to 18% oil refining and biodiesel production (6 g CO_2eq MJ^{-1}). A huge variation can be observed for the various LUC scenarios: between -59 and 152 g CO_2eq MJ^{-1}. The highest results were calculated for the scenarios where tropical forest is converted into palm plantation (L1 to L4) and the lowest values for the conversion of cropland into palm plantation (L12). Negative results were obtained for all scenarios of savanna (except for nominally managed savanna in Orinoquía region) and cropland conversion.

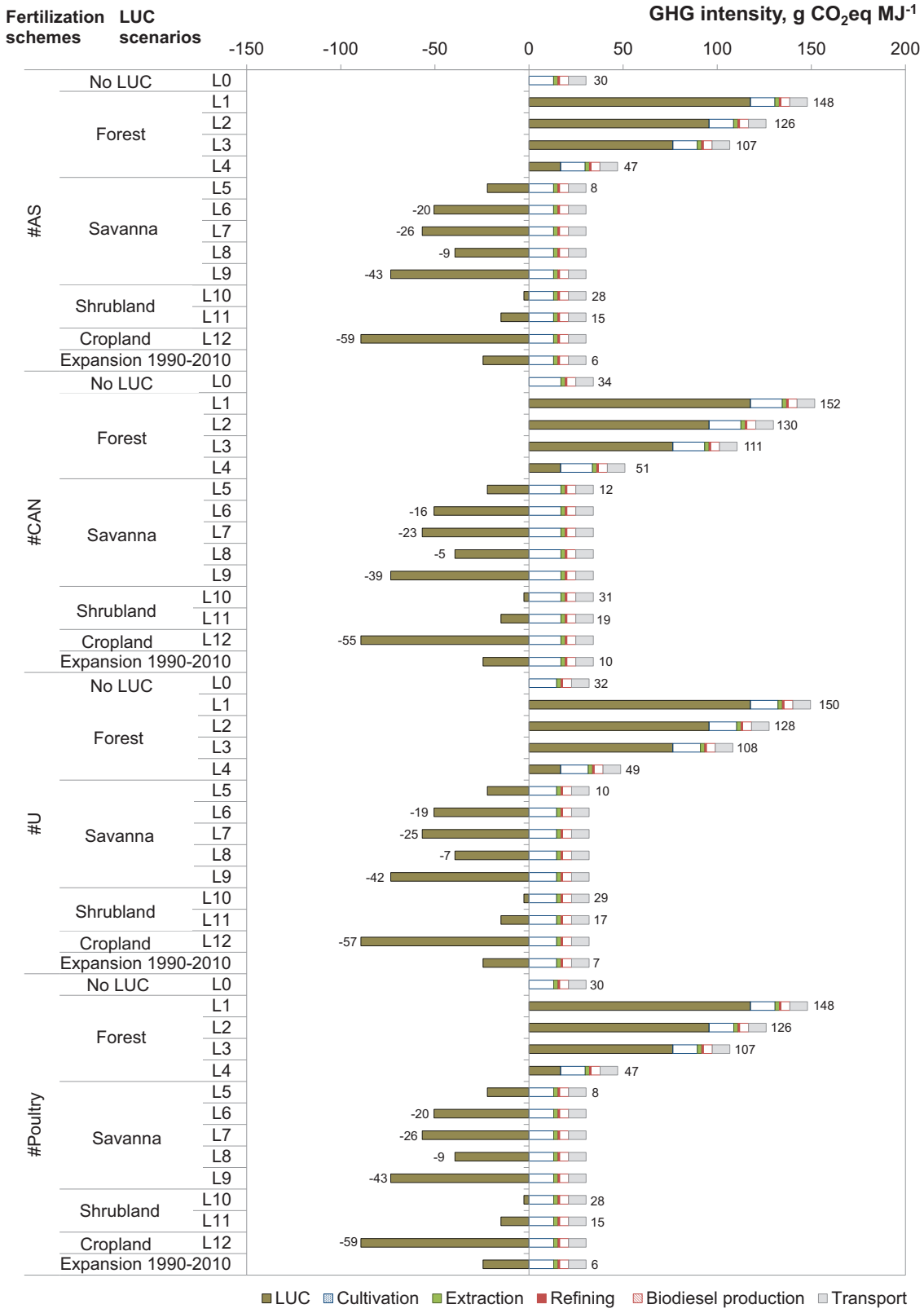


Figure 4.1. GHG intensity of palm biodiesel: contribution of LC phases for alternative LUC and fertilization scenarios.

The GHG intensity of palm biodiesel, considering the expansion of the Colombian oil palm area from 1990 to 2010, varies from 6 to 10 g CO_2eq MJ^{-1} due to an increase on the organic carbon in the vegetation and a high rate of soil organic carbon storage of palm plantation. Regarding the LUC, negative carbon stock changes were obtained for the expansion of the Colombian oil palm area and all scenarios of savanna, shrubland and cropland conversion (-3 to -89 g CO_2eq MJ^{-1}) due to higher carbon stock (mainly in ΔC_{veg}) in the palm plantation than in previous land-uses. LUC carbon stock changes are positive (17 to 118 g CO_2eq MJ^{-1}) for the scenarios in which forest is converted into palm plantation.

Using default (IPCC, 2006 or EC, 2010c) or specific values for Colombia (Anaya et al., 2009) for C_{vegR} results in important differences in the calculation of GHG intensity, as can be seen when primary forest (L1 to L3), nominally managed savannas (L5 to L7) and shrublands (L10 to L11) are converted into palm plantations. For example, carbon stock change of scenario L1 (C_{vegR} calculated using default values from IPCC, 2006) is 35% higher than scenario L3 (C_{vegR} calculated using the specific values of above ground living biomass for Colombian primary forest given by Anaya et al., 2009). Also, the carbon stock change of scenario L7 (default C_{vegR} from EC, 2010c) is almost 3 times higher than scenario L5 (nominally managed savannas adopting the specific value of aboveground biomass for Orinoquía region from Anaya et al., 2009). These results show the importance of using site-specific carbon stock values in the calculations.

Figure 4.2 shows the carbon stock changes for the 12 LUC scenarios, disaggregated in ΔSOC, ΔC_{veg} and N_2O, to enable a better understanding of their contribution to GHG intensity. More than 60% of the LUC GHG intensity occurs due to a high carbon stock change in the vegetation (ΔC_{veg}) for forest, savanna and cropland conversion scenarios. Changes in SOC (ΔSOC) contribute 55-60% to LUC GHG intensity in the scenarios of shrubland conversion. N_2O emissions due to the N mineralization of soil organic matter as a result of improved savanna conversion represent less than 1% for LUC GHG intensity.

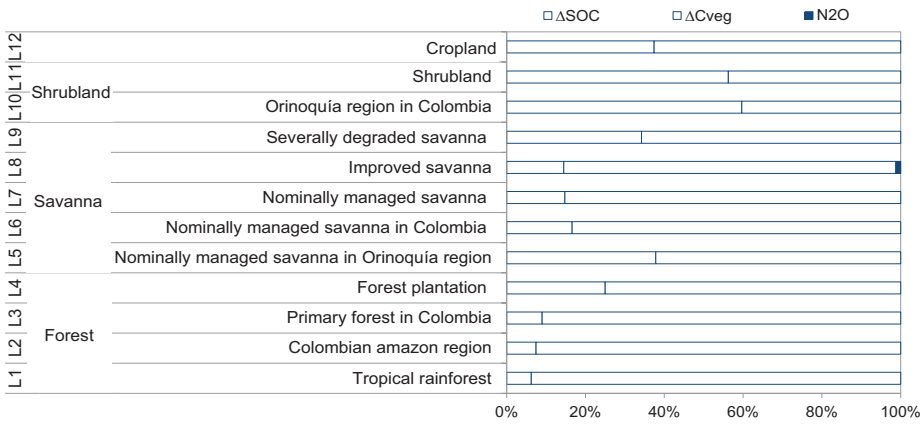


Figure 4.2. Contribution of ΔSOC, ΔC_{veg} and N_2O emissions for LUC GHG intensity in the alternative LUC scenarios for palm plantation in Colombia.

BIODIESEL CHAIN B

GHG intensity of biodiesel produced with soybean imported from Brazil and Argentina are presented in **Figure 4.3** for the alternative LUC scenarios and cultivation systems, in three different climate regions of both countries. The results show a huge variation, ranging from 14 g CO_2eq MJ^{-1} (conversion of severely degraded grassland in no-tillage soybean cultivation in warm temperate dry region) to 852 g CO_2eq MJ^{-1} (conversion of tropical rainforest in tillage soybean cultivation in tropical region). In Brazil the results vary from 37 to 852 g CO_2eq MJ^{-1} and in Argentina from 14 to 201 g CO_2eq MJ^{-1}.

The contribution of each LC phase to the GHG intensity is also shown in **Figure 4.3**: LUC represents 7 to 95% (-12 to 809 g CO_2eq MJ^{-1}), cultivation 1 to 39% (7 to 17 g CO_2eq MJ^{-1}), transport 2 to 49% (10 and 18 g CO_2eq MJ^{-1}) and oil extraction, refining and biodiesel production 1 to 34% (9 g CO_2eq MJ^{-1}). LUC dominates the results, contributing more than 80% to GHG intensity in all tropical region scenarios, in 4 scenarios for warm temperate moist region and 6 in warm temperate dry region. LUC amounts to less than 40% of GHG intensity in the scenarios in which severely degraded grassland was converted in warm temperate region. In warm temperate dry region, negative GHG intensity due to LUC was obtained (-2 to -12 g CO_2eq MJ^{-1}), due to the fact that the soil organic carbon of soybean plantations (SOC_A) is higher than the SOC_R in the severely degraded grassland in this region.

It can be observed that higher results were obtained when previous land was converted in soybean cultivation under tillage systems than the corresponding reduced or no-tillage systems in each region. Batlle-Bayer et al. (2010) also showed that no-till practices reduce soil organic carbon losses (0-30 centimeter topsoil layer) after land-use conversion from conventional tillage (primary and secondary tillage). According to the Product Board MVO (2011), the main reason is that no-till farming protects the soil from erosion and structural breakdown. No-tillage offers the possibility not only of reducing carbon loss from the soil as a result of cultivation, but also of increasing soil carbon in the form of organic matter, with positive impacts on both soil productivity and climate change reductions (Cavalett and Ortega, 2009, 2010).

Figure 4.4 shows the carbon stock changes for the 45 LUC scenarios, disaggregated in ΔSOC, ΔC_{veg} and N_2O. More than 50% of the LUC GHG intensity occur due to a high carbon stock change in the vegetation (ΔC_{veg}) in the following scenarios: i) all LUC scenarios in the tropical region, ii) forest and perennial crop conversions in warm temperate regions, iii) severely degraded grassland conversion in warm temperate moist region. Changes in SOC (ΔSOC) contribute more than 50% to LUC GHG intensity in the scenarios of grassland conversion in warm temperate dry region and improved management and moderately degraded grassland in warm temperate moist region. N_2O emissions due to the N mineralization vary from 0% (conversion of severely degraded grassland) to 6% (conversion of improved management grassland).

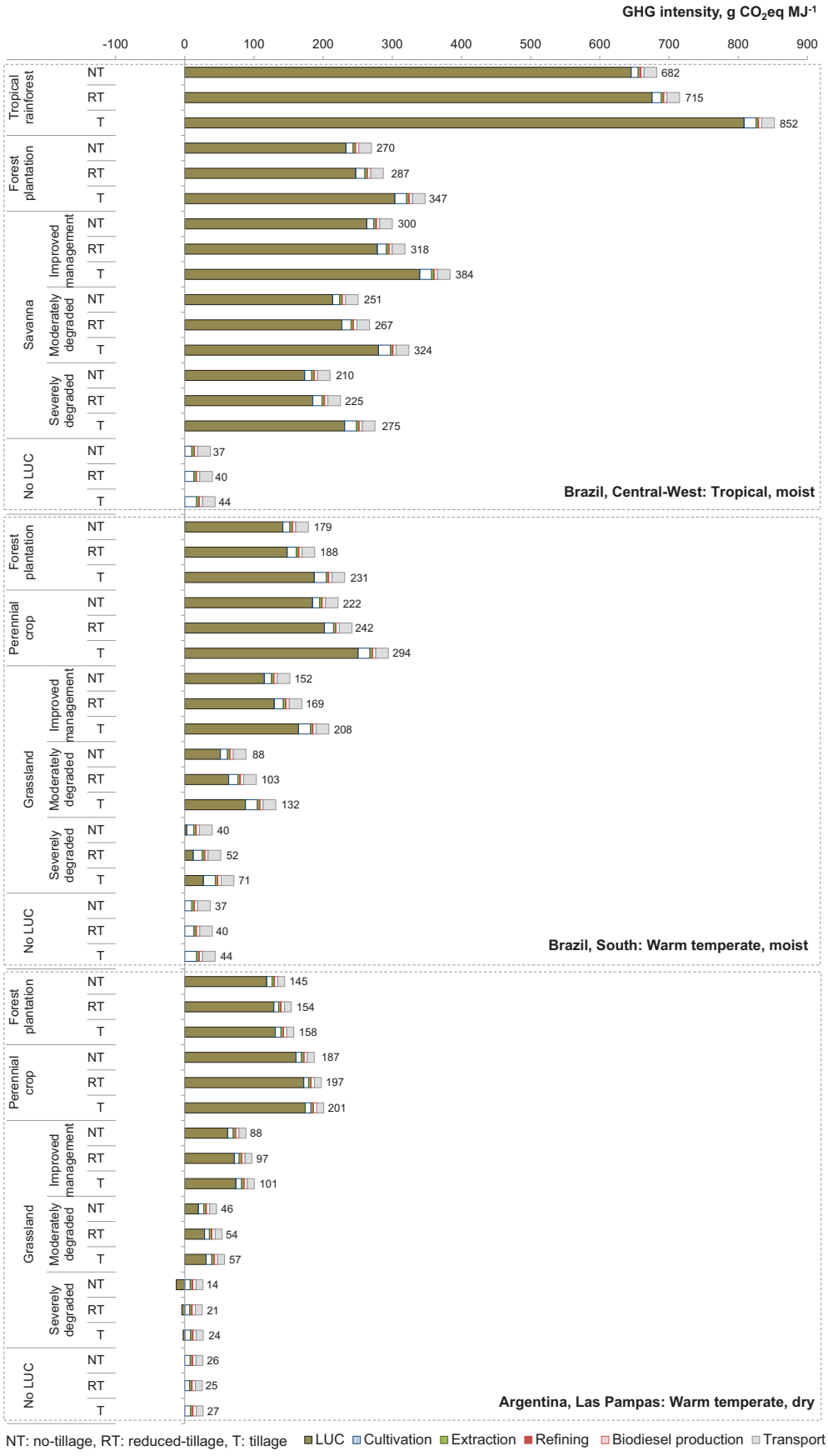


Figure 4.3. GHG intensity of soybean biodiesel: alternative LUC scenarios and cultivation systems, in 3 climate regions in Brazil and Argentina.

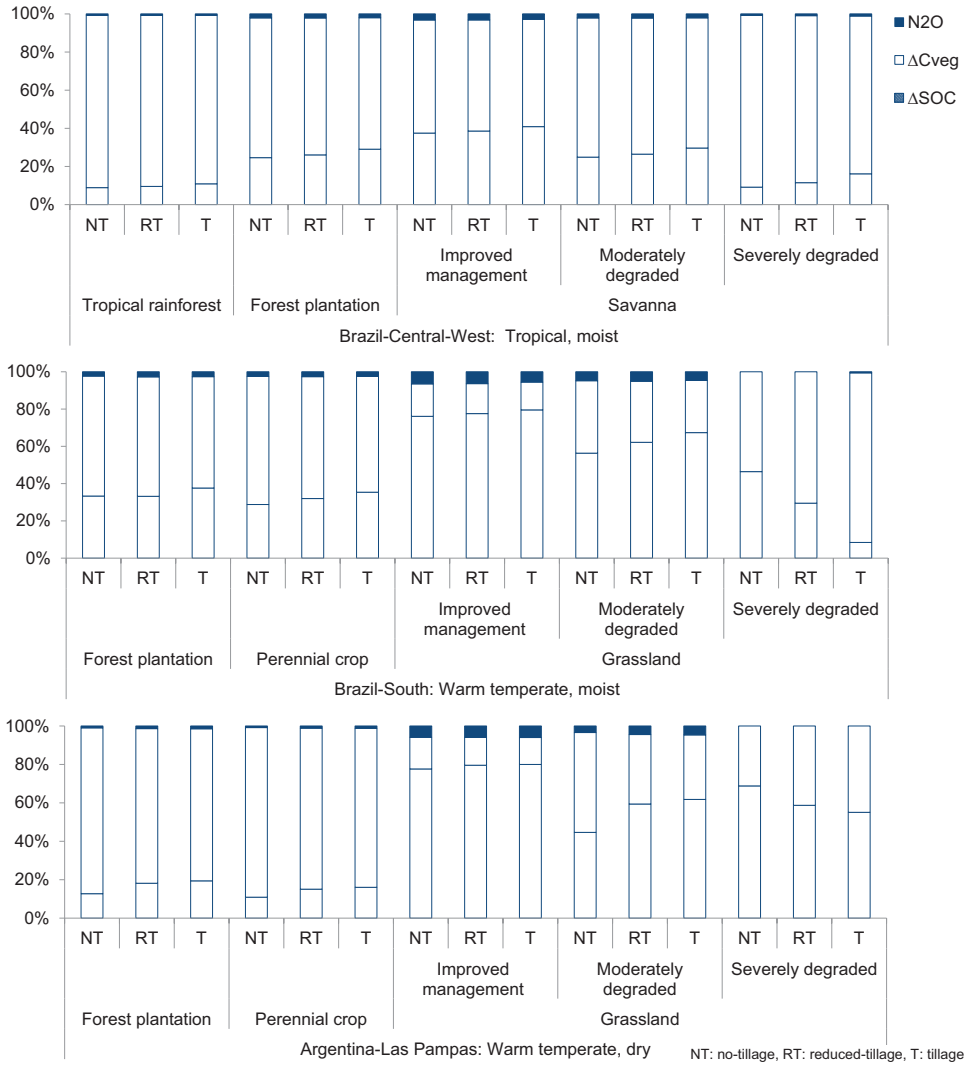
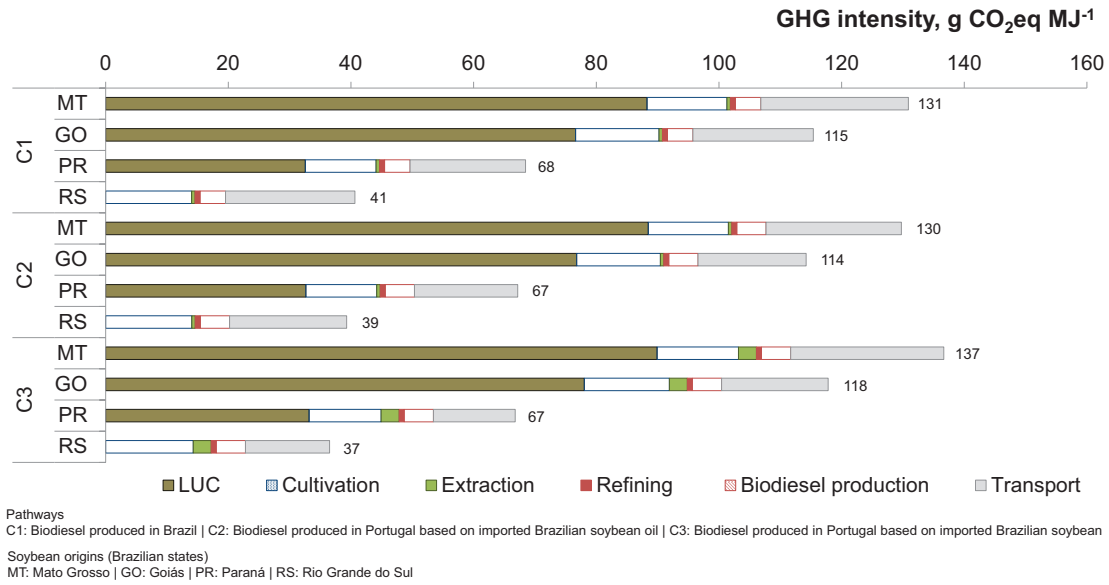


Figure 4.4. Contribution of ΔSOC, ΔC_{veg} and N_2O emissions for LUC GHG intensity in the alternative LUC scenarios and soybean cultivation systems in 3 climate regions in Brazil and Argentina.

BIODIESEL CHAIN C

GHG intensity of soybean biodiesel totally produced in Brazil and exported to Portugal (C1), produced in Portugal using soybean oil imported from Brazil (C2) and produced in Portugal using soybean imported from Brazil (C3) are presented in **Figure 4.5**. For the three pathways, it was considered that soybean was produced in four states in Brazil: Mato Grosso (MT), Goiás (GO), Paraná (PR) and Rio Grande do Sul (RS). The contribution of LC phases is also presented, including the contribution of LUC due to the expansion of soybean area in each state, from 1985 to 2006. The results vary widely according to the different states where soybean was cultivated but no significant differences among the pathways were found, since the results depend mostly on the LUC than transport.



Pathways
C1: Biodiesel produced in Brazil | C2: Biodiesel produced in Portugal based on imported Brazilian soybean oil | C3: Biodiesel produced in Portugal based on imported Brazilian soybean

Soybean origins (Brazilian states)
MT: Mato Grosso | GO: Goiás | PR: Paraná | RS: Rio Grande do Sul

Figure 4.5. GHG intensity of soybean biodiesel: contribution of each LC phase for three pathways and four Brazilian soybean origins.

The lowest result (37 g CO_2eq MJ^{-1}) were obtained when soybean was produced in Rio Grande do Sul (since soybean area reduced, no LUC occurred) and the highest when soybean was cultivated in Mato Grosso (137 g CO_2eq MJ^{-1}). When soybean was cultivated in Mato Grosso and Goiás the lowest results were calculated for pathways C1 and C2, while pathway C3 has the lowest results when soybean was cultivated in Paraná and Rio Grande do Sul.

The process contribution to GHG intensity of soybean biodiesel chain C is as follows: 0 to 68% LUC, 15 to 52% transport, 10 to 39% cultivation, 3 to 13% biodiesel production and less than 10% oil extraction and refining. The results depend greatly on the LUC, which contributes more than 48% for the GHG intensity of biodiesel produced from soybean from Mato Grosso, Goiás and Paraná. Transport and cultivation are the processes that contribute most to GHG intensity of biodiesel produced with soybean from Rio Grande do Sul. The influence of LUC on GHG intensity is related to the type and area of land that is converted into soybean and in which state this conversion occurred: LUC in Mato Grosso and Goiás were due to a savanna/shrubland conversion, whereas in Paraná were in pastures and forest plantations. CO_2 emissions (79-96%) play a major role in the results.

4.3 THE EFFECT OF LAND-USE PRACTICES, PRODUCTION SCHEMES AND PATHWAYS

This sub-chapter presents the LCIA of the three biodiesel chains, considering the ReCiPe 1.07 method and using energy allocation. It regards the following objectives:

"2.1 Extend the standard LCA methodology to address local aspects associated with land-use, including oil crop production in different countries, states and climate regions."

"2.2 Determine the environmental impacts of alternative cultivation systems, fertilization schemes and production options."

"2.3 Assess the environmental impacts of different pathways for biodiesel consumed in Portugal."

"2.4 Provide a better knowledge and understanding of agricultural systems and their environmental hotspots."

"3.1 Perform a sensitivity analysis for nitrogen field emission calculations."

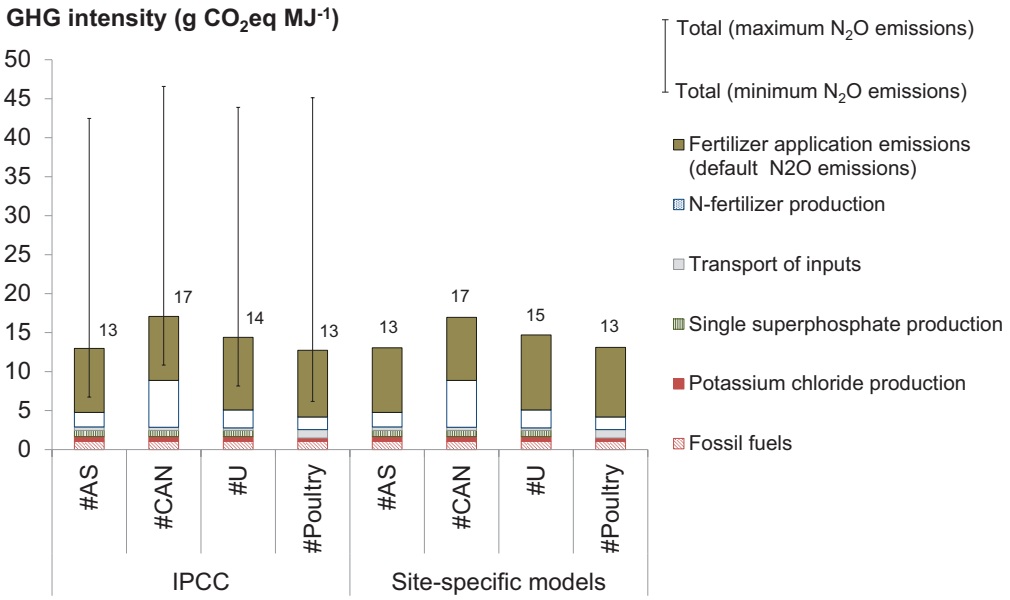
"3.2 Assess the influence of the time horizon considered for the GHG intensity calculation."

"3.3 Determine which LC stages and processes contribute most to the environmental impacts of biodiesel."

BIODIESEL CHAIN A

GHG intensity of palm cultivation in Colombia are shown in **Figure 4.6**, focusing on the contribution of the main inputs and nitrogen field emissions in the 4 fertilization schemes (#AS, #CAN, #U and #Poultry). Nitrogen field emissions were calculated based on the IPCC approach (default parameters and emission factors) and site-specific models. The results ranges obtained from the sensitivity analysis performed for field N_2O emissions (maximum and minimum parameters and emission factors from IPCC, 2006) are presented in the chart as error (range) bars.

GHG intensity of palm cultivation varies between 13 and 17 g CO_2eq MJ^{-1}. The lowest results were obtained when poultry manure was applied as fertilizer (#Poultry) and the highest when calcium ammonium nitrate was used (#CAN). This variation can be explained by the difference in fertilization type, mainly due to the fertilizer production. However, emissions from fertilizer application represent more than 48% of the overall cultivation emissions. The contribution of N-fertilizer production to the cultivation emissions vary between 12% (#Poultry) and 36% (#CAN). Phosphorus and potassium fertilizers production are responsible for less than 11% of the cultivation GHG emissions. Fossil fuels consumed in agricultural operations represent less than 8%, similarly to transport of inputs.



#AS: ammonium nitrate | #CAN: calcium ammonium nitrate | #U: urea | #Poultry: poultry manure

Figure 4.6. GHG intensity of palm cultivation: alternative fertilization schemes and N_2O emission calculations.

Comparing the two approaches adopted for nitrogen field emission calculations, it can be observed that no significant differences were obtained. Regarding the sensitivity analysis performed for field N_2O emissions, the results vary between 6 (#Poultry) and 11 g CO_2eq MJ^{-1} (#CAN) when minimum field N_2O emissions were calculated, and between 42 (#AS) and 47 g CO_2eq MJ^{-1} (#CAN) for maximum field N_2O emissions. The results show that field N_2O emissions play a major role in the GHG intensity of palm cultivation, suggesting more research on the sources of N_2O.

Figure 4.7 shows the direct and indirect field N_2O emissions and the effect of adopting default, minimum and maximum parameters and emission factors in the IPCC approach calculations. A significant variation in N_2O emissions can be observed. When default parameters were adopted about 22-26% of N_2O emissions from fertilizer application are indirect emissions (from leaching and volatilization), which are frequently ignored in most LCA studies (Smeets et al., 2009). However, adopting maximum parameters results in 49-53% of indirect N_2O emissions (relatively to total field N_2O emissions). On the other hand, indirect N_2O emissions represent less than 3% when minimum parameters and emission factors were adopted. It can also be observed that the contribution of indirect emissions was higher for the organic fertilizer (#Poultry), as a result of a high fraction of NH_3 volatilization when this fertilizer was applied. This fact emphasizes the importance of using fertilizer-specific NH_3 volatilization fractions (site-specific models).

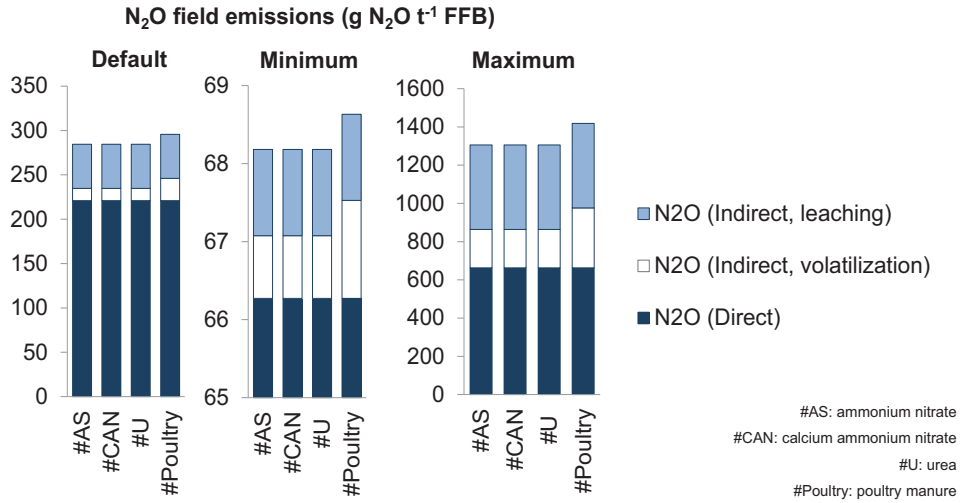


Figure 4.7. Sensitivity analysis of field N_2O emissions of the alternative fertilization schemes of palm cultivation.

The contribution of the main inputs and fertilizer application emissions for the remaining environmental impacts (acidification, freshwater and marine eutrophication, ozone depletion and photochemical oxidation) of palm cultivation under 4 fertilization schemes is presented in **Figure 4.8**. For the impacts that are influenced by the nitrogen emissions (acidification and marine eutrophication) the results are presented for the two approaches adopted (IPCC and site-specific models). It can be seen that fertilizer application emissions were that contribute most to terrestrial acidification and eutrophication impacts (more than 49% and 79%), fertilizers production contributed most to ozone depletion (more than 37%) and fossil fuels consumption to photochemical oxidation (more than 29%). Ammonia and nitrate emissions from fertilizer application were that contribute most to terrestrial acidification and marine eutrophication impacts, respectively. Freshwater eutrophication was mainly due to phosphorus emissions.

Contrasting to GHG intensity, terrestrial acidification and marine eutrophication impacts are greatly influenced by the nitrogen emission calculation approaches. The highest acidification impact was obtained for #Poultry fertilization scheme, independently of the nitrogen emission calculation approach. However, the results vary significantly for the remaining fertilization schemes when site-specific models were adopted and are similar with IPCC approach. When IPPC approach was adopted the lowest acidification impact was obtained for #AS fertilization scheme, whereas the lowest result were obtained for #CAN when site-specific model was adopted. Marine eutrophication impact is similar for the four fertilization schemes but when site-specific models were adopted, the results are 30-35% lower than those obtained with IPCC approach.

Comparing the remaining environmental impacts of the four fertilization schemes, the lowest freshwater eutrophication impact was obtained when poultry manure was applied as fertilizer (#Poultry) and the

highest when calcium ammonium nitrate was used (#CAN). The highest photochemical oxidant formation impact was also obtained when calcium ammonium nitrate was used. For #U fertilization scheme it was calculated the highest ozone depletion result and the lowest photochemical oxidant formation impact. The lowest ozone depletion impact was obtained for #AS fertilization scheme.

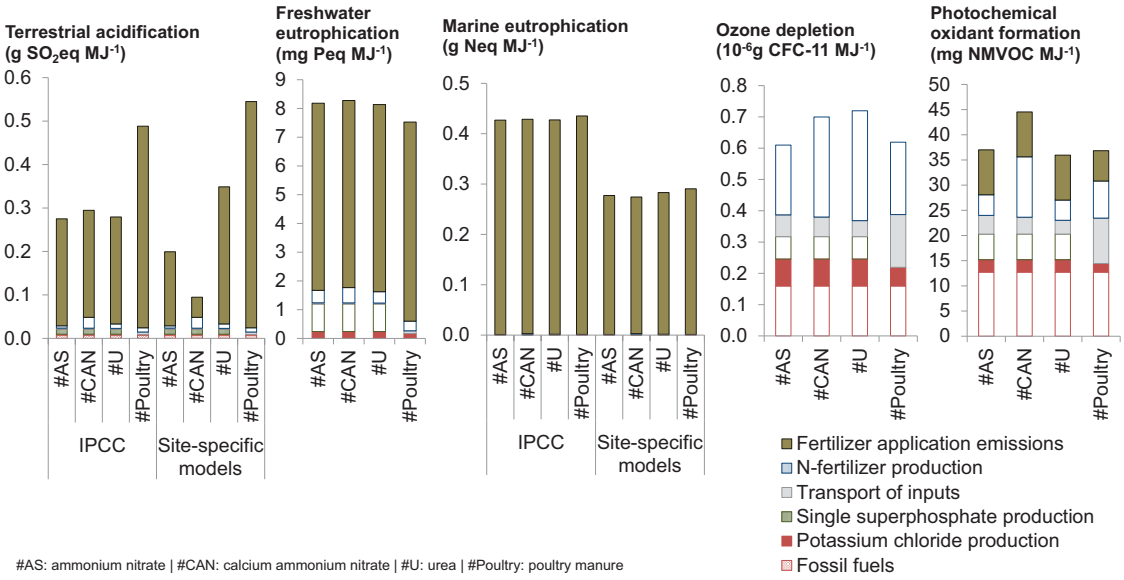


Figure 4.8. LCIA of palm cultivation: alternative fertilization schemes and nitrogen emission calculation approaches.

The LC environmental impacts of palm oil extraction, considering two scenarios for POME treatment (biogas captured and flared and biogas released into the atmosphere), are presented in **Figure 4.9**. A huge variation on the impacts of the two scenarios can be observed, except for ozone depletion and freshwater eutrophication, which are not influence by the emissions from POME treatment. In both scenarios POME treatment is the process that contributes most to GHG intensity (82-98%). GHG intensity of palm oil extraction considering that biogas is flared (2 g CO_2eq MJ^{-1}) was just about ten times lower than considering that biogas is released into the atmosphere (19 g CO_2eq MJ^{-1}).

For both POME treatment scenarios, electricity was the process that contributes most to ozone depletion impact (89%), whereas the combustion of fibers and shells at the cogeneration plant (to produce steam and electricity) was the process that contribute most to photochemical oxidant formation (68-90%), freshwater and marine eutrophication (84% and 61-94%) impacts. When biogas was flared, marine eutrophication, photochemical oxidant formation and acidification impacts were about 35%, 24% and 50% lower than when biogas was released. The process contribution to terrestrial acidification varies depending on the POME treatment scenario: POME treatment was the process that contributes

most (51%) if the biogas was released, but if biogas was flared the process that contributes most was co-generation (95%).

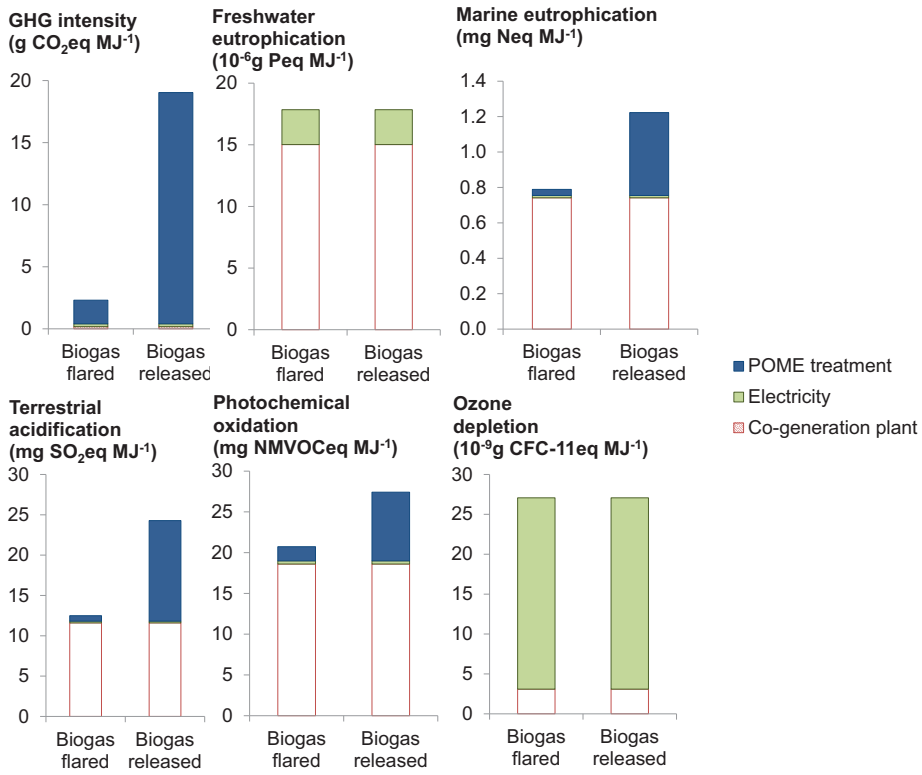


Figure 4.9. Environmental impacts of palm oil extraction considering different POME treatment options (biogas captured and flared or released).

A sensitivity analysis of the contribution of each GHG to the GHG intensity of palm cultivation and oil extraction considering three time horizons (100 years: GWP100, 20 years: GWP20 and500 years: GWP500) is presented in **Figure 4.10**. GWP for biogenic methane was calculated based on the fossil methane GWP for each time horizon and taking into account that 2.75 kg (=(44/12)/(16/12)) of biogenic carbon dioxide was not released per 1 kg of biogenic methane emitted (GWP100 = 25, GWP20 = 72 and GWP500 = 7.6). N_2O emissions were calculated based on IPCC approach and the results ranges obtained from the sensitivity analysis performed for field N_2O emissions (maximum and minimum parameters and emission factors) are presented in the chart as error (range) bars.

N_2O emissions (mainly due to N-fertilizer application) dominate the results for all fertilization schemes and GWP time horizons. The contribution of N_2O emissions for the GHG intensity of palm cultivation ranges between 42% (#U, GWP500) and 71% (#CAN, GWP100). The contribution of CO_2 emissions vary between 27% (#CAN, GWP20 and GWP100) and 58% (#U, GWP500). Methane emissions represent less than 5% of GHG emissions for all GWP time horizons. The highest results were obtained for GWP20 but no significant differences (lower than 2%) were found between GHG intensity of

cultivation calculated using GWP20 and GWP100. However, a reduction of 30-36% was obtained for a time horizon of 500 years (GWP500) comparing to 20 years.

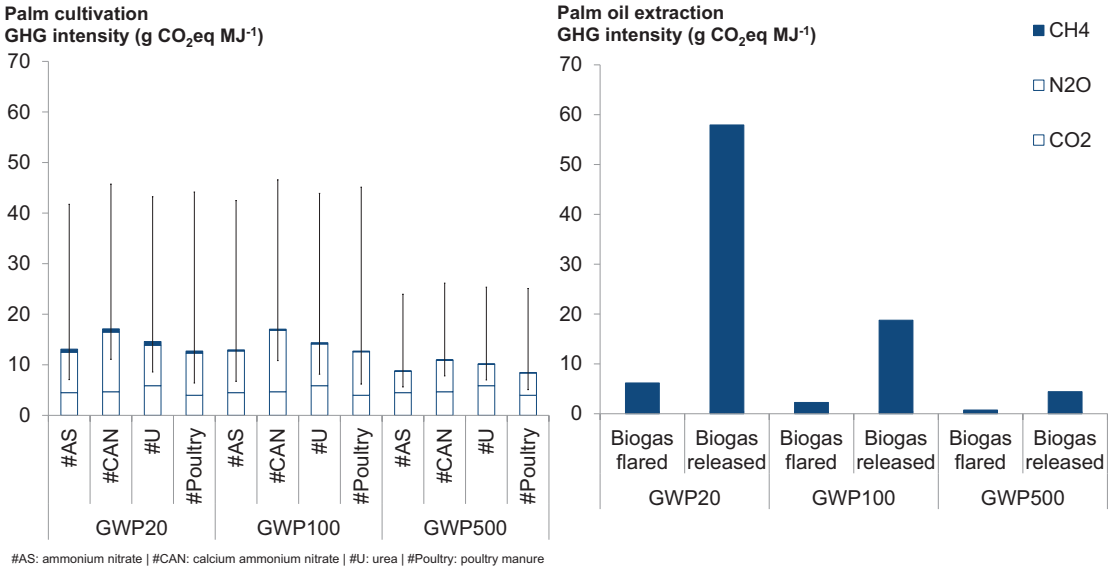


Figure 4.10. Contribution of GHG emissions to GHG intensity of palm cultivation and oil extraction: fertilization schemes, biogas management options and GWP time horizons.

GHG intensity of palm oil extraction varies between 1 g CO_2eq MJ^{-1} (GWP500) and 6 g CO_2eq MJ^{-1} (GWP20) when biogas was captured and flared. On the other hand, when biogas was released into the atmosphere, the results vary between 4 g CO_2eq MJ^{-1} (GWP500) and 58 g CO_2eq MJ^{-1} (GWP20). The results show that there are important GHG emissions when biogas is released into the atmosphere, mainly CH_4, with important differences between the three GWP time horizons. The results showed that biogas capture and flaring compared with biogas release to the atmosphere contributes to a reduction of 80-90% of the GHG intensity of palm oil extraction.

BIODIESEL CHAIN B

GHG intensity of alternative soybean cultivation systems (NT: no-tillage, RT: reduced tillage and T: tillage) in Brazil and Argentina, including the contribution of main inputs, are shown in **Figure 4.11**. GHG intensity ranges obtained from the sensitivity analysis performed for field N_2O emissions (maximum and minimum parameters and emission factors) are presented in the chart as error (range) bars. Adopting default values in the calculation of field N_2O emissions, the results vary between 7 (reduced-tillage, Argentina) and 17 g CO_2eq MJ^{-1} (tillage, Brazil). These results can be justified by the higher soybean yields and lower diesel requirements (for machinery) in no- and reduced tillage, since direct seeding was performed without primary tillage.

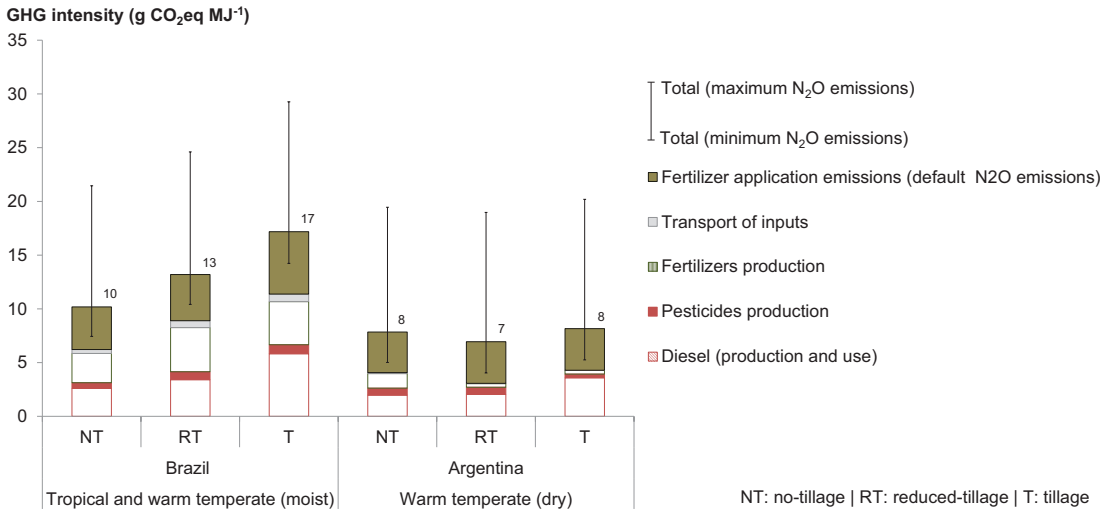


Figure 4.11. GHG intensity of alternative soybean cultivation systems in Brazil and Argentina.

Soybean cultivation had higher GHG intensities in Brazil (10-17 g CO_2eq MJ^{-1}) than in Argentina (7-8 g CO_2eq MJ^{-1}). This difference is due to the use of limestone and greater quantities of fertilizer in Brazil. Field N_2O emissions (default) are the most important contribution to the GHG intensity of soybean cultivation (between 33% and 56%) except under the tillage system in Brazil, where the emissions from the use of machinery contribute 34%. Diesel for agricultural machinery represents 25% to 44% of GHG intensity of cultivation, with a higher contribution under tillage system than the corresponding no- or reduced tillage systems. The main reason for the variation on GHG intensity of the different cultivation systems is diesel consumption, although the reason for the different results from Brazil and Argentina is the amount of fertilizer and lime applied to the soil.

Regarding the sensitivity analysis of the field N_2O emissions, it can be observed that the uncertainty in N_2O emission calculations is very high. When minimum parameters and emission factors were adopted, the results are reduced by 17% to 42%. If the maximum parameters and emission factors are adopted, the results increase by 70% to 173% and the field N_2O emissions dominate the GHG intensity for all cultivation systems. These results show that GHG intensity of soybean cultivation is very sensitive to the parameters and emission factors adopted for field N_2O emission calculations.

An analysis of the contribution of each GHG (CO_2, N_2O and CH_4) to the GHG intensity of soybean produced by the various cultivation systems (expressed in CO_2 equivalents) is presented in **Figure 4.12**. When default N_2O emissions were considered, CO_2 emissions from diesel combustion and the production of fertilizers are the main factors contributing to the GHG intensity of soybean produced in Brazil. N_2O contributes less than 37% in Brazil, but more than 48% in Argentina (due to field N_2O emissions). However, when minimum values were adopted for the field N_2O emission calculations, the

results are significantly different and CO_2 represents a higher contribution to GHG intensity in all regions (73-89%). It can also be observed that if maximum values were adopted 55% to 84% of GHG intensity is due to N_2O emissions. Methane emissions represent less than 3% of GHG intensity in all the scenarios considered.

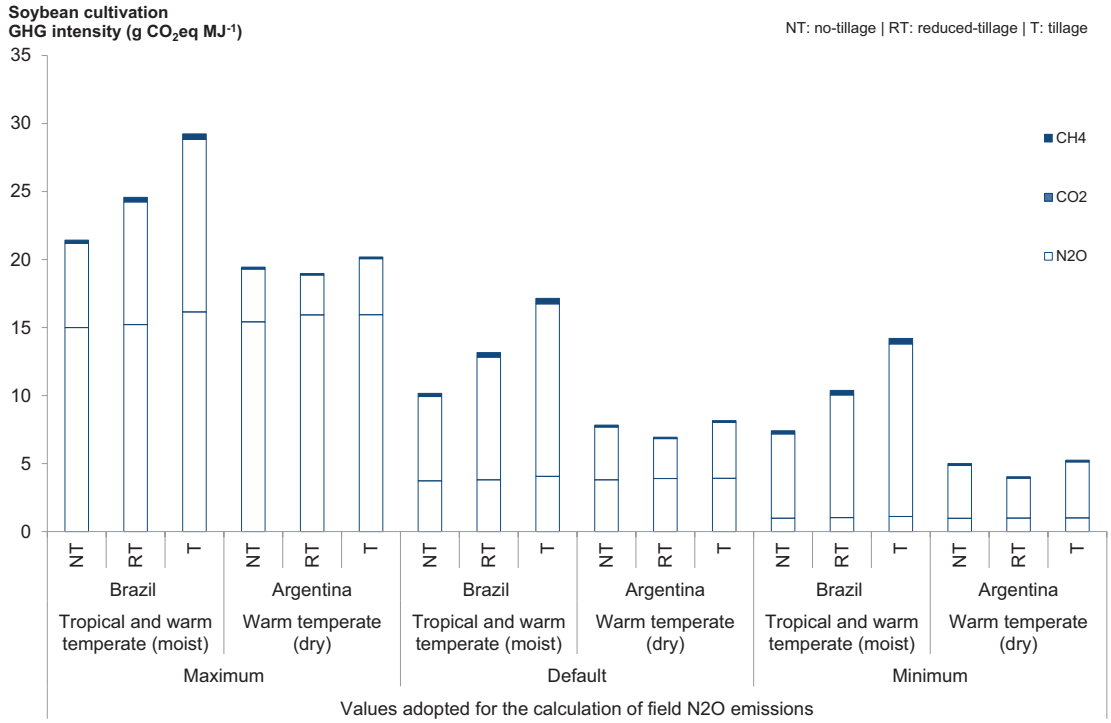


Figure 4.12. Contribution of each GHG emission to the GHG intensity of alternative soybean cultivation systems and N_2O emission calculations.

The contribution of the main inputs and fertilizer application emissions for the remaining environmental impacts of alternative soybean cultivation systems in Brazil and Argentina is presented in **Figure 4.13**. It can be seen that fertilizer application was the process that contribute most to eutrophication impact (more than 76%), diesel contributed most to ozone depletion (34-77%) and photochemical oxidant formation (56-93%). Fertilizers production was the process that contribute most to terrestrial acidification impact of all alternative soybean cultivation systems in Brazil and in no-tillage system in Argentina, whereas diesel production and consumption made the highest contribution for tillage and reduced-tillage systems in Argentina. In Brazil, the highest impacts were obtained for soybean cultivated under tillage system and the lowest values for no-tillage, in all environmental impact categories. The highest impacts were also obtained for tillage system in Argentina, while reduced-tillage had the lowest impacts, except for eutrophication (in which no-tillage has the lowest results). Marine eutrophication impact is similar for the alternative cultivation systems in both countries.

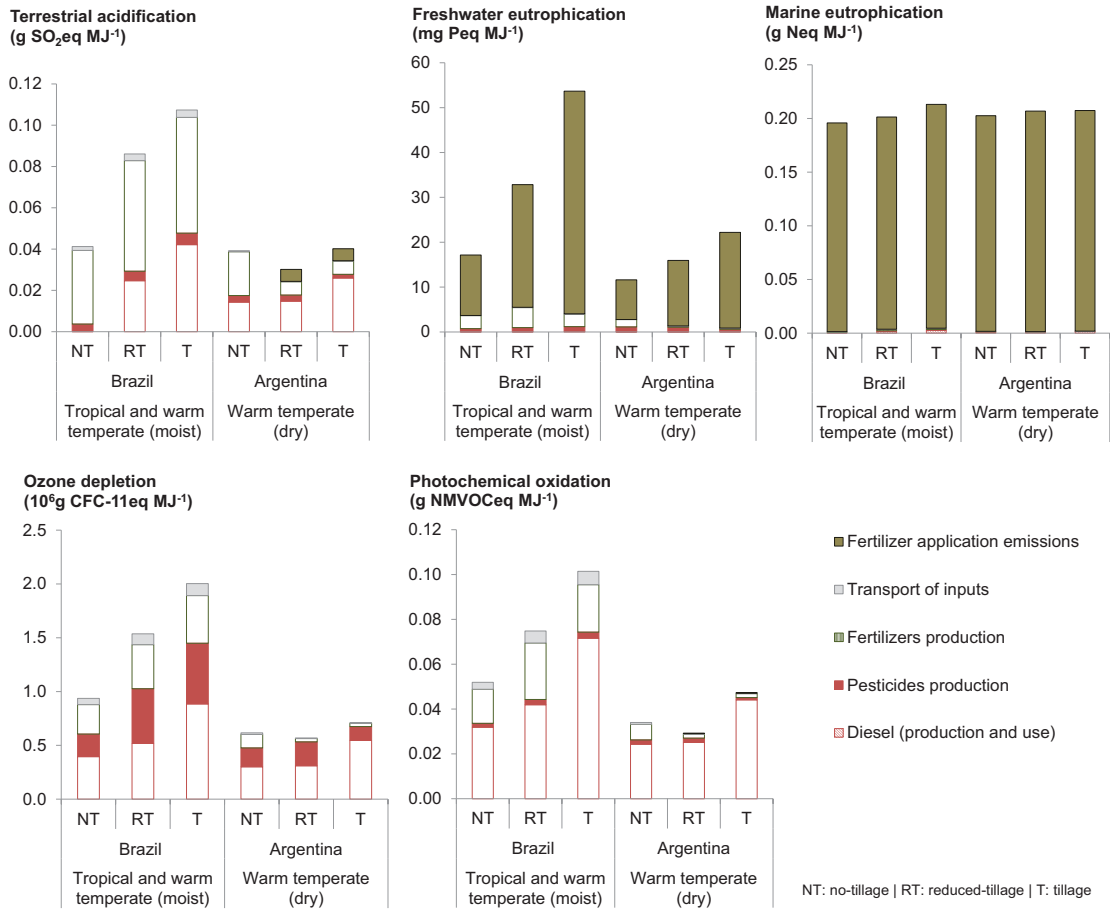


Figure 4.13. LCIA of alternative soybean cultivation systems in Brazil and Argentina.

Figure 4.14 shows the GHG intensity of alternative soybean transportation scenarios, calculated on the basis of the weighted average distances for the transoceanic and road transportation of soybean from plantations to the ports. The error range bars represent the variation associated with eleven types of lorry and the maximum and minimum distances for each route. The highest impact was calculated for the "3.5-16 t" lorry (fleet average) and the lowest impact for the ">32t" lorry (EURO4). Transportation of soybean from Brazil to Portugal involves higher emissions (9-28 g CO_2eq MJ^{-1}) than Argentina (7-12 g CO_2eq MJ^{-1}) due to the greater road transport distances in Brazil. Approximately 69% of the emissions in Brazil are from road transportation, whereas in Argentina this only represents 34% of the total transportation emissions. In Brazil, soybean imported from Mato Grosso has higher GHG emissions than other states. Regarding the ports, it can be observed that the emissions are in general lower for soybean imported from Santos and Paranaguá. In Argentina, no significant differences in the results were observed.

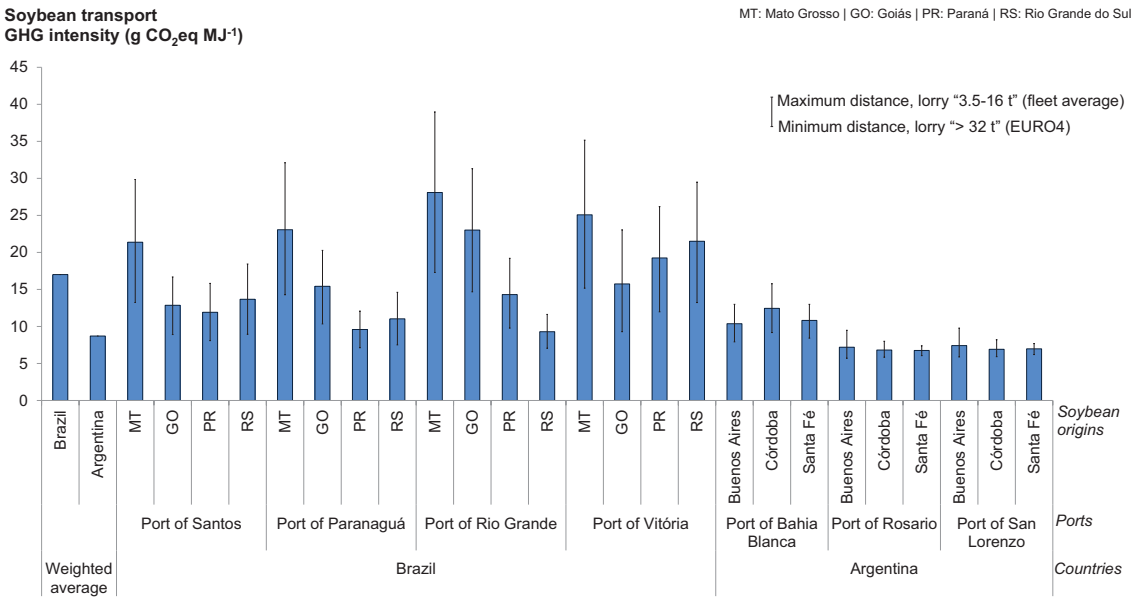


Figure 4.14. GHG intensity of alternative soybean transportation scenarios.

BIODIESEL CHAIN C

The environmental impacts of soybean cultivated in four states in Brazil (Mato Grosso, Goiás, Paraná and Rio Grande do Sul), including the contribution of main inputs and fertilizer application emissions, are presented in **Figure 4.15**. For the impacts that are influenced by the nitrogen emissions (climate change, acidification and marine eutrophication) the results are presented for the two approaches adopted for the calculations (IPCC and site-specific models). It can be seen that impacts vary widely according to the different states where soybean was cultivated and the approaches adopted for nitrogen emission calculations. The process contribution varies with the environmental impact category. GHG intensity, eutrophication and ecotoxicity (terrestrial, freshwater and marine) impacts are mainly caused by fertilizers and pesticides application. Fertilizers production and diesel (production and use) are the processes that contribute most to ozone depletion, photochemical oxidant formation, acidification and human toxicity impacts.

GHG intensity varies according to the different states where soybean was cultivated (11 to 14 g CO_2eq MJ^{-1}). Although the approach adopted for N_2O emission calculation do not have a significant influence on the GHG intensity (since the main difference of both approaches is the calculation of indirect N_2O emissions), when site-specific models were adopted the highest result were obtained for soybean produced in Rio Grande do Sul (RS), while the highest result were calculated for soybean from Goiás (GO) when IPCC approach were used. For both nitrogen emission calculation approaches, the lowest GHG intensity was calculated for soybean from Paraná (PR).

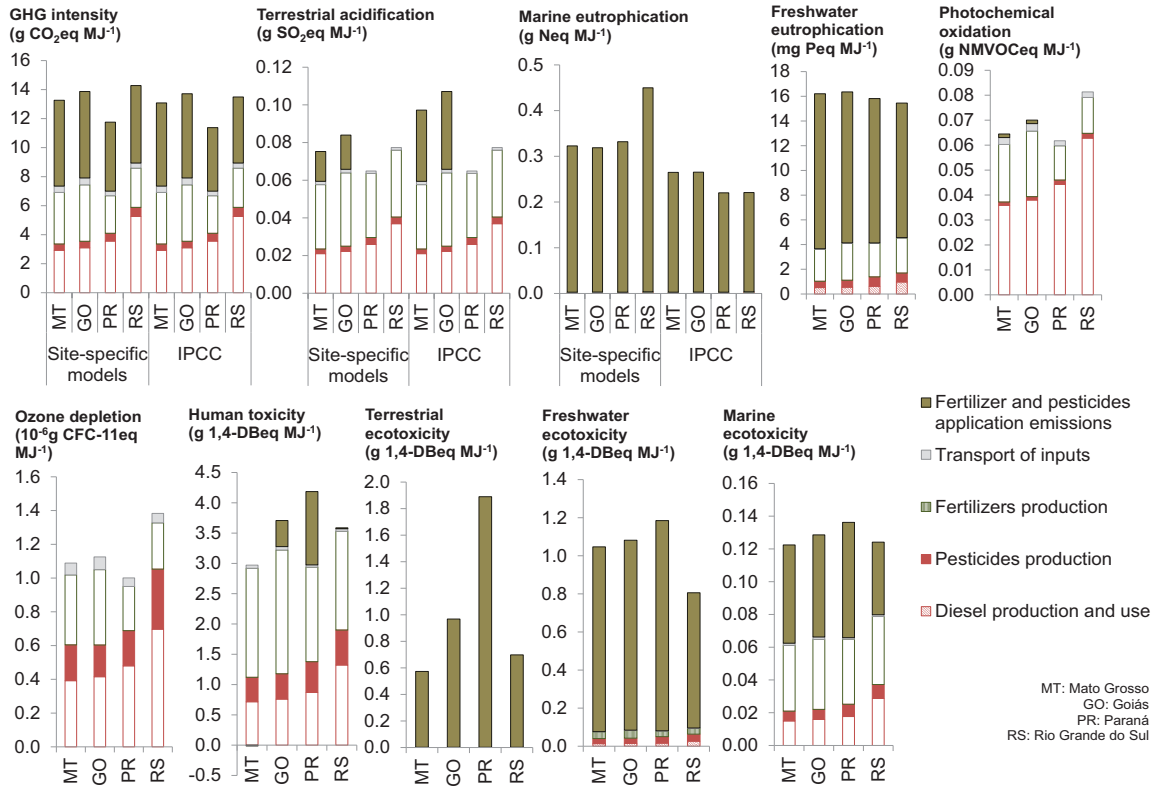


Figure 4.15. LCIA of soybean cultivation in four Brazilian states: Mato Grosso, Goiás, Paraná and Rio Grande do Sul.

The lowest terrestrial acidification impact was calculated for soybean from PR (0.06 g SO_2eq MJ^{-1}) and the highest for soybean from GO (0.08-0.11 g SO_2eq MJ^{-1}). The results for PR and RS were the same for both nitrogen emission calculation approaches, since no nitrogen fertilizer was applied to the soils in these states and consequently no ammonia emissions were calculated in both approaches. However, for soybean from Mato Grosso (MT) and GO, acidification impact obtained with IPPC approach is approximately 29% higher than those calculated using site-specific models. Marine eutrophication impact also varies according to the nitrogen emission calculation approaches and contradictory results were obtained: the site-specific approach results are about 20% (for MT and GO) and 50-100% (PR and RS) higher than those obtained with IPCC approach. Marine eutrophication impacts of soybean from MT and GO (0.32 g Neq MJ^{-1}) are the lowest when site-specific models were adopted, while soybean from PR and RS has the lowest results (0.22 g Neq MJ^{-1}) when IPCC approach were adopted.

With regard to the other environmental impact categories, the highest ozone depletion and photochemical oxidant formation results were obtained for soybean cultivated in RS and the lowest values for soybean from PR. Soybean from GO had the highest freshwater eutrophication impact, while the lowest was calculated to soybean cultivated in PR and RS (less P-fertilizer was applied in this states). Soybean from Paraná presents the highest toxicity impacts due to the emissions related to pesticides application, in particular pyretroid- and organophosphorus-compounds.

Figure 4.16 shows the LC environmental impacts of soybean oil extraction and biodiesel production in Portugal and Brazil. The environmental impacts of biodiesel production vary in both countries. GHG intensity and ozone depletion impacts of biodiesel production in Portugal are about 15% and 7% higher than those impacts in Brazil. For the remaining environmental categories, biodiesel production in Portugal presents lower results comparing with Brazil, particularly for acidification (-41%), photochemical oxidant formation (-31%) and marine eutrophication (-26%) impacts.

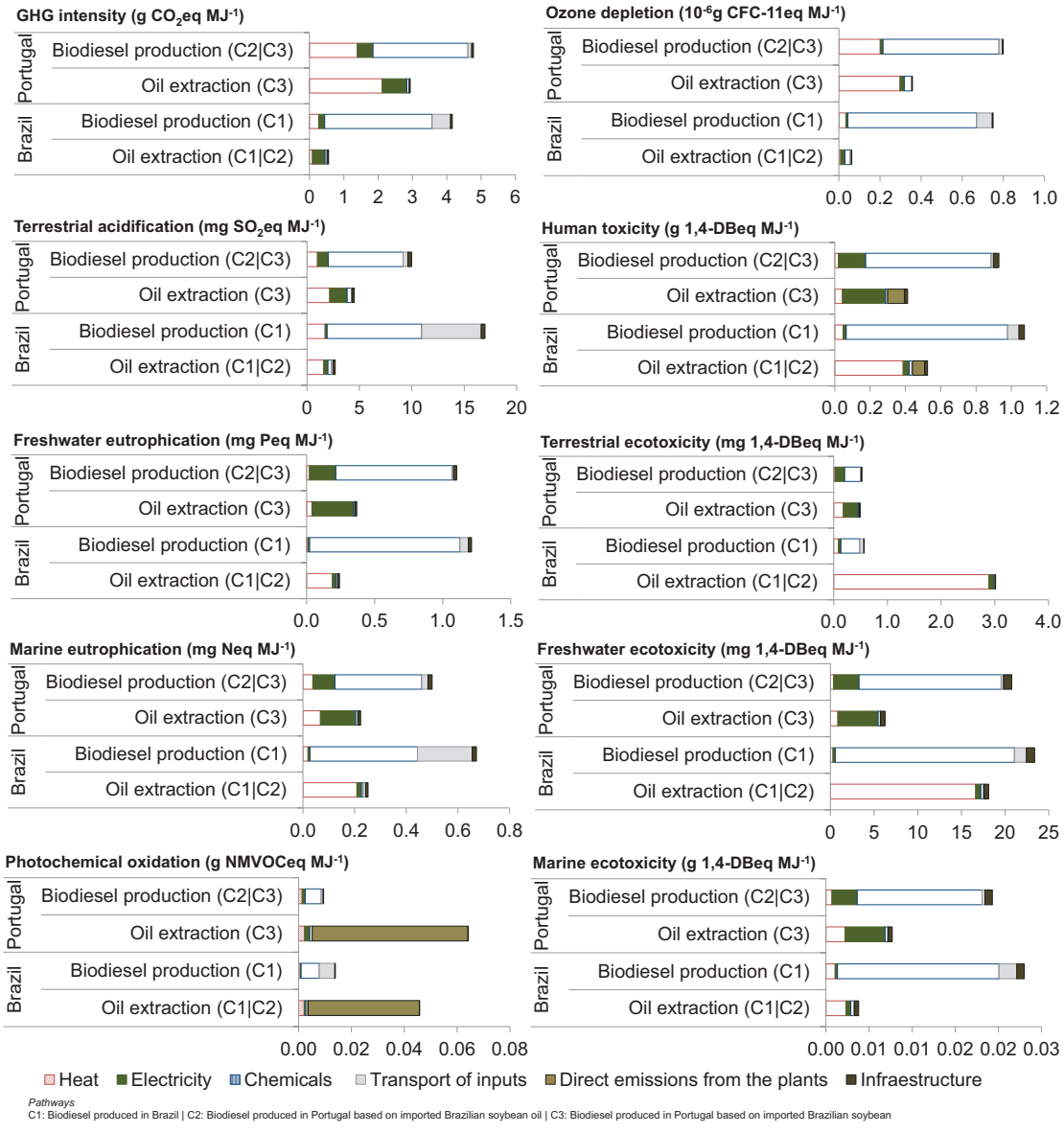


Figure 4.16. LCIA of soybean oil extraction and biodiesel production in Portugal and Brazil.

GHG intensity and ozone depletion impacts related to oil extraction in Portugal are approximately 4-5 times higher than in Brazil. This difference is related to the heat production and consumption in the oil extraction plants: heat consumption in Portugal is about 3 times higher than in Brazil and fuel oil and

natural are used in Portuguese plants (in Brazil biomass is used as a fuel). Acidification, freshwater eutrophication, photochemical oxidant formation and marine ecotoxicity impacts of oil extraction in Portugal are also higher than in Brazil, mainly due to the emissions related to electricity production in Portugal (mostly from hard coal burned in the power plants, which represented 29% of the Portuguese electricity mix in 2012) and the higher hexane emissions. On the opposite, marine eutrophication, human toxicity, terrestrial and freshwater ecotoxicity impacts of oil extraction in Brazil are higher than in Portugal mainly due to the emissions due to heat production (wood production and combustion).

The contribution of each process to the environmental impacts is also presented in **Figure 4.16**. The production of chemicals used in the biodiesel production in both countries (mainly methanol and sodium methoxide) is the process that contributes most (50-91%) to all environmental impact categories. Regarding the oil extraction in Portugal, electricity is the process that contributes most to eutrophication (61-83%) and toxicity (57-73%) impacts, while heat production is that contributes most to the GHG intensity (72%), ozone depletion (83%) and acidification (48%). For oil extraction in Brazil, heat production is the process that contributes most to acidification (59%), eutrophication (79-83%) and toxicity (61-96%) impacts, while electricity contribute most to the GHG intensity (65%) and hexane production to ozone depletion (42%). Approximately 92% of photochemical oxidant formation of oil extraction is due to the hexane emissions in both countries.

Figure 4.17 shows the LC environmental impacts of biodiesel produced from soybean cultivated in four states in Brazil (MT, GO, PR and RS) and taking into account three alternative pathways: biodiesel totally produced in Brazil and exported to Portugal (C1), biodiesel produced in Portugal based on soybean oil imported from Brazil (C2) and biodiesel produced in Portugal based on soybean imported from Brazil (C3). The contribution of each LC phase to the environmental impacts is also presented. The impacts were calculated based on the site-specific models adopted for field nitrogen emission calculations. GHG intensity and toxicity impacts were also calculated for soybean biodiesel chain C but the results are presented and analyzed in detail in **sub-chapter 4.2** and **4.4**.

Terrestrial acidification impact varies between 0.20 and 0.28 g SO_2eq MJ^{-1} for the alternative pathways and states where soybean was cultivated. The lowest terrestrial acidification was obtained for pathway C2 and for soybean produced in PR, while the highest were obtained for pathway C3 when soybean was produced in MT and GO and for pathway C1 when soybean was from PR or RS. The contribution of each LC phase to the acidification impact of soybean biodiesel is as follow: 56-67% from transportation, 27-37% from cultivation, 4-7% from biodiesel production and less than 3% from oil extraction and refining. Nitrogen oxides (NO_x) and sulfur dioxide (SO_2) emissions related to fuel combustion and fertilization are that contribute most to acidification impact of soybean biodiesel.

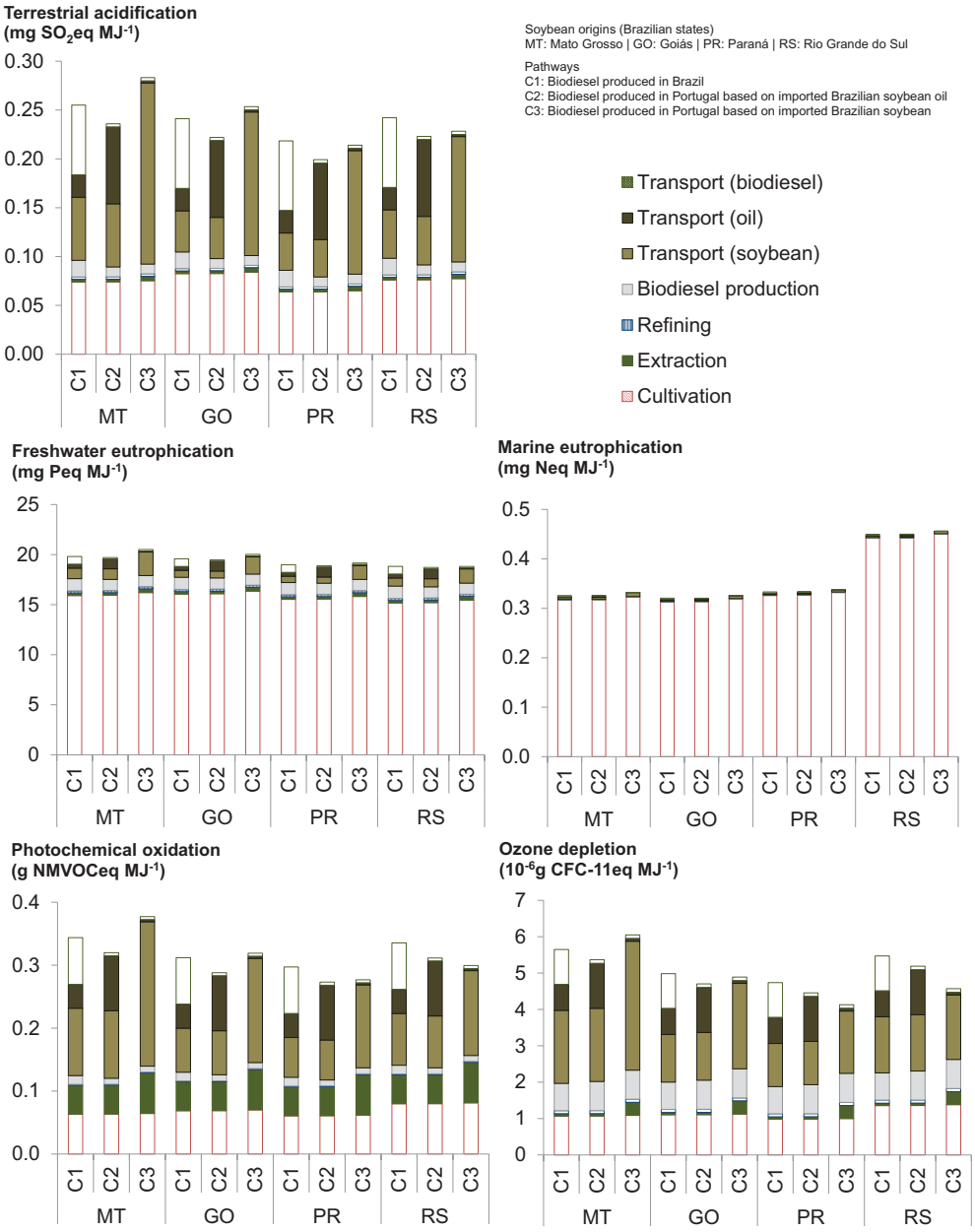


Figure 4.17. LCIA of soybean biodiesel: contribution of each LC phase for 3 pathways and 4 Brazilian soybean origins.

Freshwater eutrophication impact of soybean biodiesel varies between 18.7 and 20.5 mg Peq MJ^{-1}, whereas marine eutrophication between 0.32 and 0.46 g Neq MJ^{-1}. Because cultivation emissions contribute more than 79% for both categories, the results vary more significantly for the different soybean origins than the pathways adopted. However, for both impact categories, the lowest results were obtained for pathway C2, when soybean is from MT and GO and for pathway C3, when soybean is from PR and RS. Nitrate (NO_3^-) and phosphorus (field) emissions from soybean cultivation dominate marine and freshwater eutrophication results, respectively. Transesterification and oil extraction

together represent less than 1% to total marine eutrophication impact and less than 10% to total freshwater eutrophication impact of soybean biodiesel.

Photochemical oxidant formation impact of soybean biodiesel varies from 0.27 to 0.34 gNMVOCeq MJ^{-1}. Biodiesel produced with soybean from PR and RS had the lowest results for pathway C2 and C3, while for soybean cultivated in MT and GO the highest results were obtained for pathway C3 and the lowest results for pathway C2. Transportation is the phase that contributes most to photochemical oxidant formation impact (48-64%). Cultivation represents 17-27%, extraction and refining 13-24% and transesterification less than 5% to total photochemical oxidant formation impact of soybean biodiesel. NO_x due to transportation and agricultural operations, as well as hexane emitted from the extraction mill, are the main emissions that contribute to photochemical oxidant formation impact.

Ozone depletion impact of soybean biodiesel varies from $4.1x10^{-6}$ to $6.0x10^{-6}$ g CFC-11eq MJ^{-1}. The lowest result was calculated for biodiesel produced with soybean from PR, while the highest for soybean from MT. Comparing pathways, the lowest results were calculated for C2 when soybean was cultivated in MT and GO but, when soybean was cultivated in PR or RS, pathway C3 had the lowest results. About 43-65% of total ozone depletion impact was due to transportation, 18%-30% to cultivation, 13-19% to transesterification and less than 11% to oil extraction and refining. The emissions of halon 1301 (bromotrifluoromethane), halon 1211 (bromochlorodifluoromethane) and CFC-10 (tetrachloromethane), mostly related to fossil fuels combustion (transport, agricultural operations and energy production), are that contribute most to ozone depletion impact.

4.4 THE INFLUENCE OF LCIA METHODS

The influence of the LCIA method on the results (energy allocation) is analyzed in this sub-chapter, which aims to respond to the following objectives:

> *"3.3 Determine which LC stages and processes contribute most to the environmental impacts of biodiesel."*
>
> *"3.4 Compare the LCIA results calculated using different LCIA methods and determine the extent to which the results are influenced by the method applied."*

Two LCIA methods (CML 2001 and ReCiPe 1.07) were adopted for biodiesel chains A and B. LCIA results obtained from two methods are presented, focusing on the contribution of each LC phase for the similar non-toxicity environmental impact categories in both methods (ReCiPe/CML). Relative comparison of ReCiPe and CML, i.e. the impacts of the various scenarios relatively (as a percentage) to the scenario with the highest impact (100%), is also presented for each set of similar impact categories. To compare the environmental impacts, normalized results are also presented for chains A and B.

Normalized results were obtained by dividing the characterization results by a reference value: the average yearly environmental load of the world (1995) in the CML method (Huijbregts et al., 2003) and the average yearly environmental load of each citizen (considering the world population of 6 billion, year 2000) in the ReCiPe method (Sleeswijk et al., 2008).

The toxicity impacts of soybean biodiesel chain C, calculated using ReCiPe 1.07 and USEtox 1.01 methods, are also presented and compared in this sub-chapter. It should be noted that USEtox method provides “recommended” and “interim” characterization factors, reflecting the level of reliability of the calculations in a qualitative way (Huijbregts et al., 2010). Although “interim” factors should be used in LCA studies with great caution and under awareness of their large inherent uncertainty, both “recommended” and “interim” factors were adopted and compared in this research. Please note that characterization factors for metals are all considered "interim". The non-toxicity impacts calculated using ReCiPe method for biodiesel chain C were presented in **sub-chapters 4.2 and 4.3**.

BIODIESEL CHAIN A

The environmental impacts of biodiesel produced in Portugal from palm oil imported from Colombia are presented, considering ReCiPe and CML methods. The contribution of each LC phase for each impact category is presented for the four fertilization schemes (use of ammonium sulphate #AS, calcium ammonium nitrate #CAN, urea #U and poultry manure #Poultry as fertilizers) and the two palm oil mill effluent (POME) treatment options (biogas is captured and flared or is released into the atmosphere). Note that the results for the remaining processes are constant, since they were not influenced by the fertilization scheme or biogas management option. The results presented were obtained adopting the site-specific models for field nitrogen emission calculations and the LUC emissions related to the expansion of the Colombian palm area in the 1990 to 2010 period.

Figure 4.18 shows the climate change/global warming (CC/GW) of palm biodiesel obtained from ReCiPe (a) and CML (b) methods. The results vary from 6 to 27 g CO_2eq MJ^{-1} and the process contribution is as follows: 32 to 45% LUC (-24 g CO_2eq MJ^{-1}), 18 to 29% cultivation (13 to 17 g CO_2eq MJ^{-1}), 4 to 27% extraction (2 to 19 g CO_2eq MJ^{-1}), 12 to 17% transport (9 g CO_2eq MJ^{-1}) and 7 to 10% refining and biodiesel production (6 g CO_2eq MJ^{-1}). The relative comparison of LCIA methods is also presented in **Figure 4.18 (c)**, showing that similar results were obtained since emissions due to LUC dominate the results. However, ReCiPe results are slightly higher (3-10%) than CML results due to the higher CH_4 and N_2O characterization factors considered.

Comparing fertilization schemes and biogas management options, the lowest GHG intensity was obtained for ammonium sulphate (#AS) and biogas flared and the highest for calcium ammonium nitrate (#CAN) and biogas released, in both methods. The difference among fertilization schemes is more accentuate when biogas was flared and oil extraction results represent less than 4% for the total results.

When biogas was released into the atmosphere oil extraction results represent 23-27% for the total CC/GW, thus the difference among fertilization schemes is less evident.

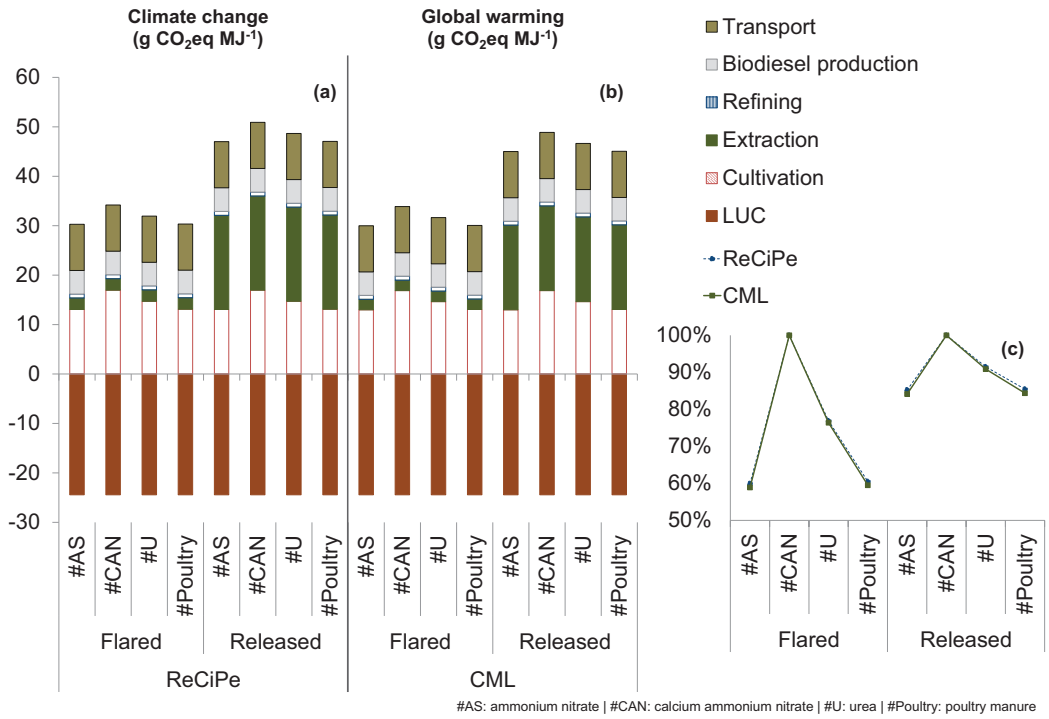


Figure 4.18. Climate change (a) and global warming (b) impact of palm biodiesel: alternative fertilization scenarios, biogas management options and LCIA methods (c).

Terrestrial acidification/acidification (TA/A) of palm biodiesel is presented in **Figure 4.19**. This impact vary from 0.2 to 0.7 g SO_2eq MJ^{-1} when ReCiPe method was adopted and 0.2 to 0.5 g SO_2eq MJ^{-1} with CML. Ammonia (NH_3), NO_x and SO_2 emissions are that contribute most to TA/A. The ReCiPe results are 9-39% higher than the CML results since the characterization factor of NH_3 emissions in CML method is 35% lower than in ReCiPe. This effect is more evident for #Poultry scheme and less for #CAN, since the NH_3 emissions were calculated on the basis of a rate of N-volatilization of 2% for #CAN and 25% for #Poultry (Asman,1992; Erisman et al., 2010).

The process contribution to TA/A is as follows: 41 to 84% cultivation (0.08 to 0.55 g SO_2eq MJ^{-1}), 12 to 44% transportation (0.08 g SO_2eq MJ^{-1}) and 4 to 18% extraction, refining and biodiesel production (0.03-0.04 g SO_2eq MJ^{-1}). A huge variation can be observed for the different fertilization schemes (0.2 to 0.5-0.7 g SO_2eq MJ^{-1}). With regard to biogas management options, it can be seen that TA/A of oil extraction (biogas released) is the double than the result obtained for oil extraction when biogas was flared. However, these results do not affect the total TA/A of palm biodiesel since extraction represents less than 12% of the total results.

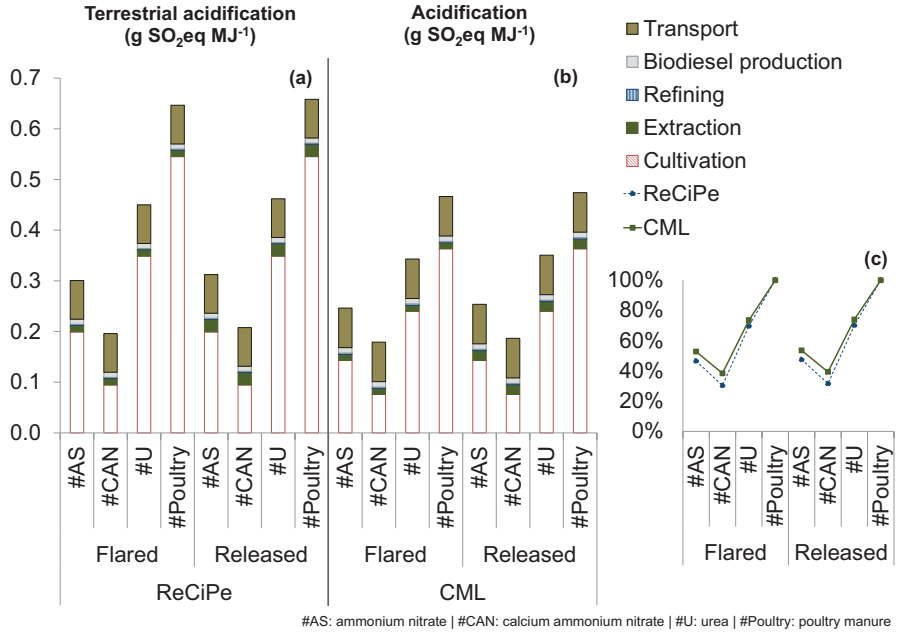


Figure 4.19. Terrestrial acidification (a) and acidification (b) impact of palm biodiesel: alternative fertilization scenarios, biogas management options and LCIA methods (c).

Figure 4.20 presents freshwater eutrophication (a), marine eutrophication (a') and eutrophication (b) impacts of palm biodiesel (the ReCiPe method distinguishes between freshwater and marine eutrophication). FE varies from 9.8 mg Peq MJ^{-1} to 10.5 mg Peq MJ^{-1} and ME from 0.28 to 0.30 g Neq MJ^{-1}, whereas eutrophication varies from 0.18 to 0.24 g PO_4^{3-}-eq MJ^{-1}. The results show that cultivation is the LC phase that contributes most to these impact categories (more than 77%), in all fertilization schemes (no variation occurs among biogas management options). Nitrate (NO_3^-) emissions in the cultivation are the most important emissions for marine eutrophication (ME) and eutrophication impacts whereas phosphorus and phosphate emissions are that contribute most to freshwater eutrophication (FE) impact. The fertilization scheme #CAN has the highest FE impact and the lowest ME and eutrophication impacts (lower nitrate emissions). On the opposite, #Poultry is the scheme with the highest ME and eutrophication impacts and the lowest FE. The variation among fertilization scenarios is more evident for eutrophication (b) than FE (a) and ME (a').

Ozone depletion/ozone layer depletion (OD) impacts are presented in **Figure 4.21**. Similar OD results were obtained with the different LCIA methods, fertilization schemes and biogas management options; however, the lowest OD impact was calculated using CML method for #AS fertilization scheme (2.67×10^{-6} g CFC-11eq MJ^{-1}), whereas the highest were calculated using ReCiPe for #U fertilization scheme (2.90×10^{-6} g CFC-11eq MJ^{-1}). OD was caused essentially by halon 1301 (bromotrifluoromethane) and halon 1211 (bromochlorodifluoromethane) emissions. CML results are slightly lower than ReCiPe results (3-5%), due to the lowest CML characterization factor for halon 1211. Emissions due to transportation are that contribute most to OD impact of palm biodiesel (44-47%),

whereas emissions from cultivation (fertilizers production and agricultural operations) and biodiesel production (mainly from methanol production) also make an important contribution to total OD impact (22-25% and 27-29%, respectively). Emissions from oil extraction and refining represent less than 4% for OD impact.

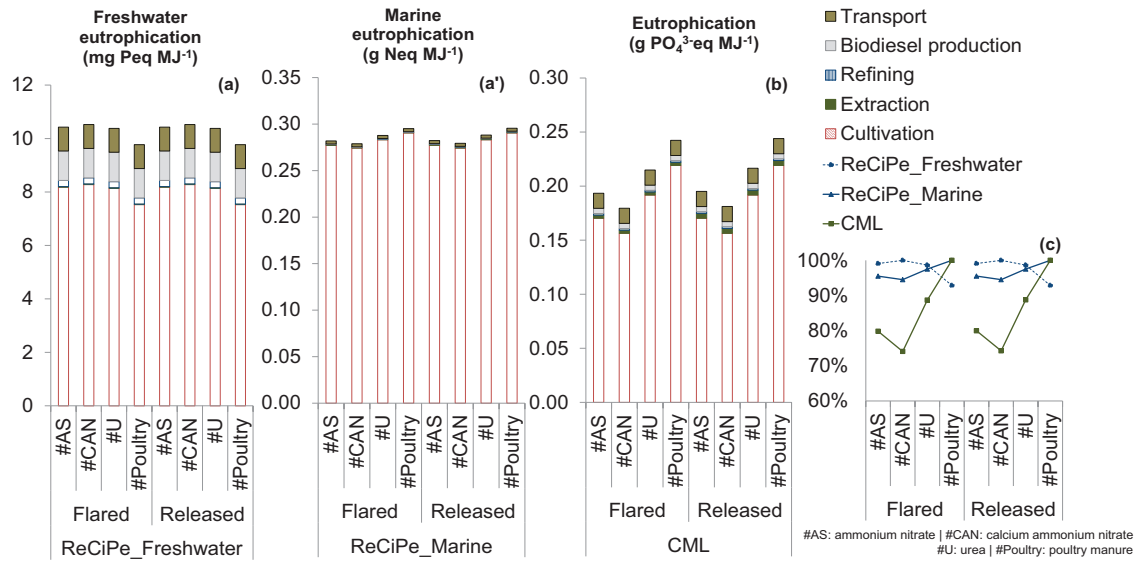


Figure 4.20. Freshwater (a), marine eutrophication (a') and eutrophication (b) impact of palm biodiesel: alternative fertilization scenarios, biogas management options and LCIA methods (c).

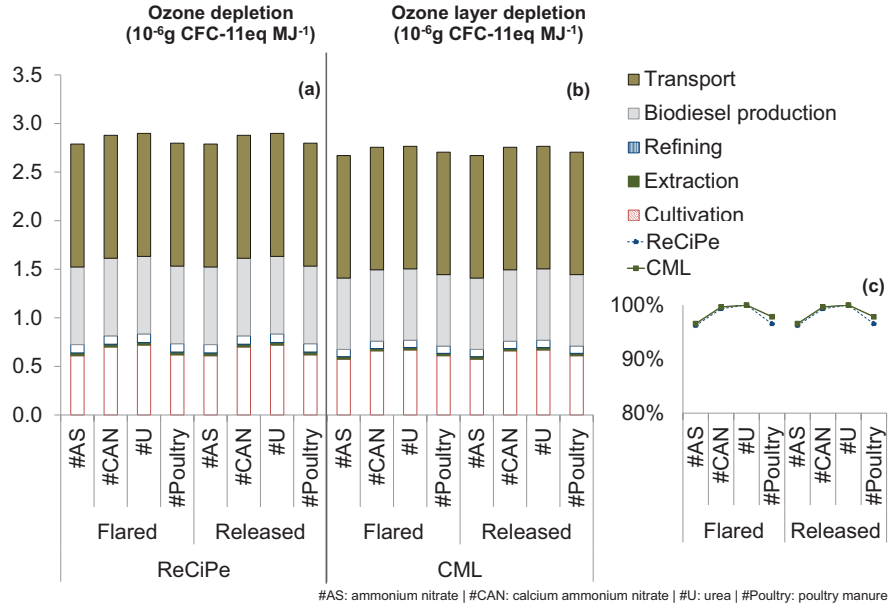


Figure 4.21. Ozone depletion (a) and ozone layer depletion (b) impact of palm biodiesel: alternative fertilization scenarios, biogas management options and LCIA methods (c).

Photochemical oxidant formation/photochemical oxidation (POF) impact of palm biodiesel are presented in **Figure 4.22**. Regarding the ReCiPe method, a little variation among fertilization schemes and biogas management options was obtained (less than 5%): from 0.16 g NMVOCeq MJ^{-1} (#U and biogas flared) to 0.17 g NMVOCeq MJ^{-1} (#CAN and biogas released). In contrast, POF impact varies greatly when CML method was adopted: from 4.75 g C_2H_4eq MJ^{-1} (#Poultry and biogas flared) to 9.78 g C_2H_4eq MJ^{-1} (#CAN and biogas released). This variation is particularly marked among biogas management options: the results calculated based on biogas releasing are 85-94% higher than those obtained when biogas was flared.

The contribution of each LC phase for POF impact depends on the LCIA method applied. Transportation emissions are that contribute most to POF impact (52-57%) when ReCiPe method was adopted, followed by palm cultivation emissions (22-27%). Regarding the CML method, transportation is also the LC phase that contribute most to POF impact (45-50%), but only when biogas was flared. When biogas was released, palm oil extraction emissions are that contribute most to POF impact (54-57%). Nitrogen oxides emissions are that contribute most to POF ReCiPe results, whereas POF impact calculated using CML method are mostly due to SO_2 emissions (when biogas was flared) and to biogenic CH_4 emissions (when biogas was released).

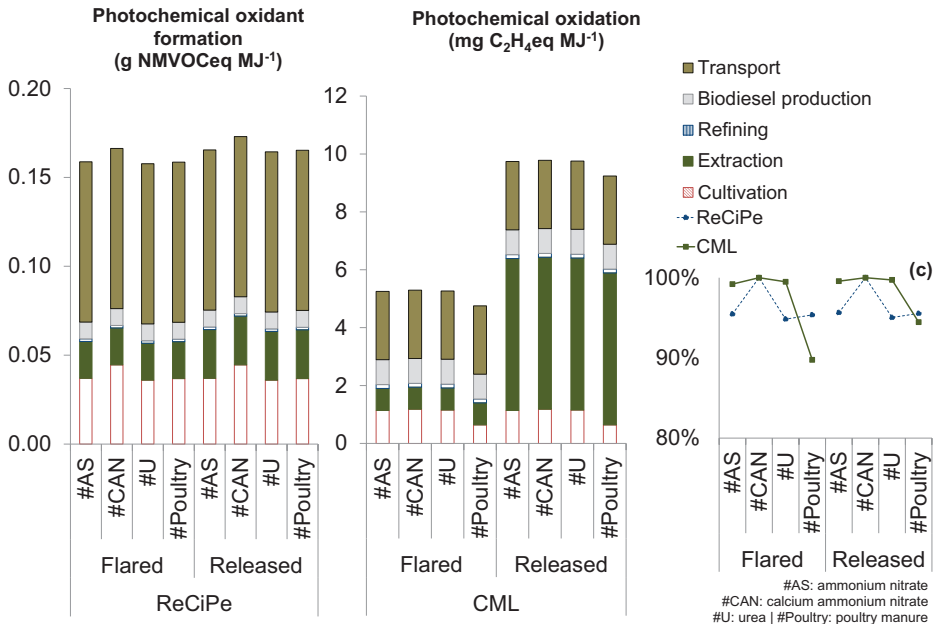


Figure 4.22. Photochemical oxidant formation (a) and photochemical oxidation (b) impact of palm biodiesel: alternative fertilization scenarios, biogas management options and LCIA methods (c).

Normalized LCIA results are presented in **Figure 4.23**. Marine and freshwater eutrophication (ReCiPe) and eutrophication (CML) are the most dominating impact categories but the differences between fertilization schemes are more significant in CML. Terrestrial acidification (ReCiPe), acidification and

global warming (CML) are also important categories. Ozone depletion/ozone layer depletion and photochemical oxidant formation/photochemical oxidation results are substantially lower than the other categories in both methods.

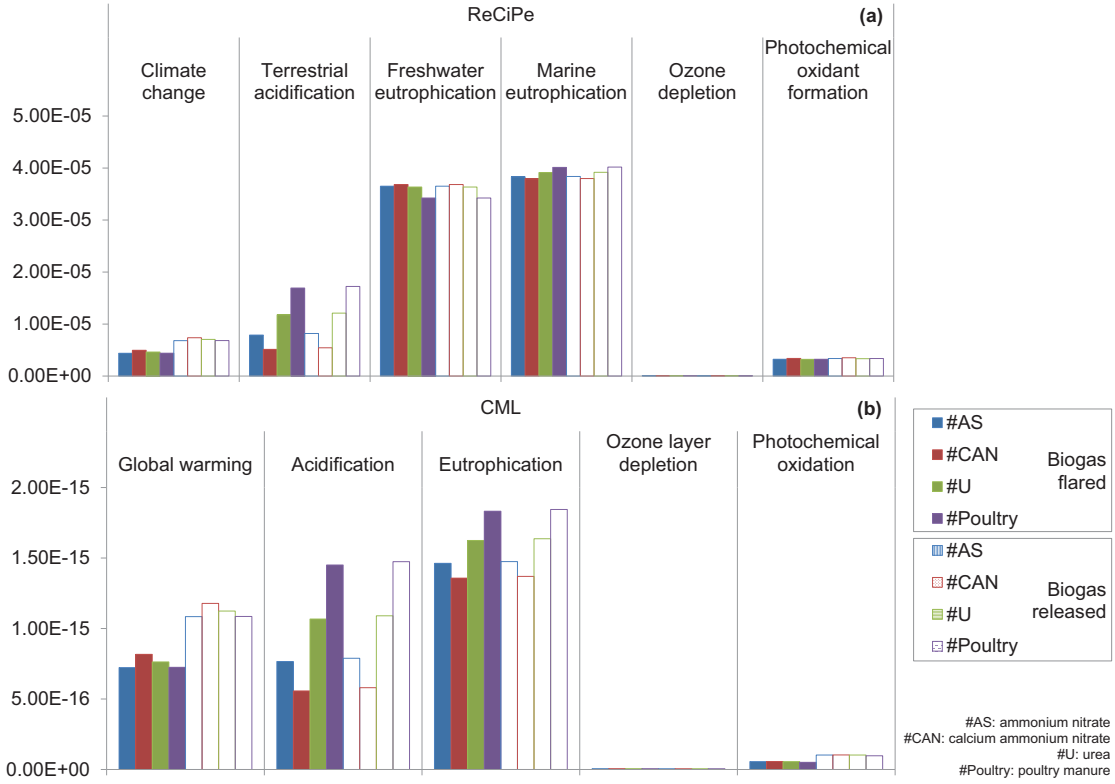


Figure 4.23. Normalized LCIA results of palm oil biodiesel: (a) ReCiPe versus (b) CML.

Comparing the results for the four fertilization schemes, it can be seen that they vary more significantly for TA/A impact categories (in both methods) and for eutrophication in CML method: the lowest results were obtained for #CAN and the highest for #Poultry. Regarding the biogas management no significant differences (less than 6%) were obtained for both LCIA methods, except for GHG intensity and POF impact categories. GHG intensity calculated using both LCIA methods are 40-55% higher when biogas was released than when biogas was flared. For CML method, POF impact is also higher (83-92%) for biogas released than biogas flared.

BIODIESEL CHAIN B

The LCIA of biodiesel produced in Portugal from soybean imported from Brazil and Argentina is presented, considering ReCiPe and CML methods. The contribution of LC phase for each impact category is presented for the alternative cultivation systems (tillage, reduced-tillage and no-tillage) and

adopting the weighted average distances of soybean transportation. The results were obtained considering the IPCC approach for field nitrogen emission calculations and the emissions related to the LUC scenarios in which the lowest results were obtained for Brazil (conversion of severely degraded grassland in warm temperate moist region) and Argentina (conversion of severely degraded grassland in warm temperate dry region) (**sub-chapter 4.2**).

Figure 4.24 shows the climate change/global warming (CC/GW) of soybean biodiesel obtained from ReCiPe (a) and CML (b) methods. The results vary from 14 to 71 g CO_2eq MJ^{-1} and the process contribution is as follows: 7 to 38% LUC (-12 to 27 g CO_2eq MJ^{-1}), 20 to 28% cultivation (7 to 17 g CO_2eq MJ^{-1}), 4 to 10% extraction (3 g CO_2eq MJ^{-1}), 26 to 46% transport (10 to 18 g CO_2eq MJ^{-1}) and 8 to 19% refining and biodiesel production (6 g CO_2eq MJ^{-1}). The relative comparison of LCIA methods (**Figure 4.24 (c)**) shows that similar results were obtained with both methods: in Brazil and Argentina the lowest GHG intensity was calculated for no-tillage (NT) systems and the highest for tillage (T) systems. GHG intensity of T systems in Brazil are 79% higher than NT and 36% than RT, while GHG intensity of T systems in Argentina are 74% and 15% higher than NT and RT, respectively.

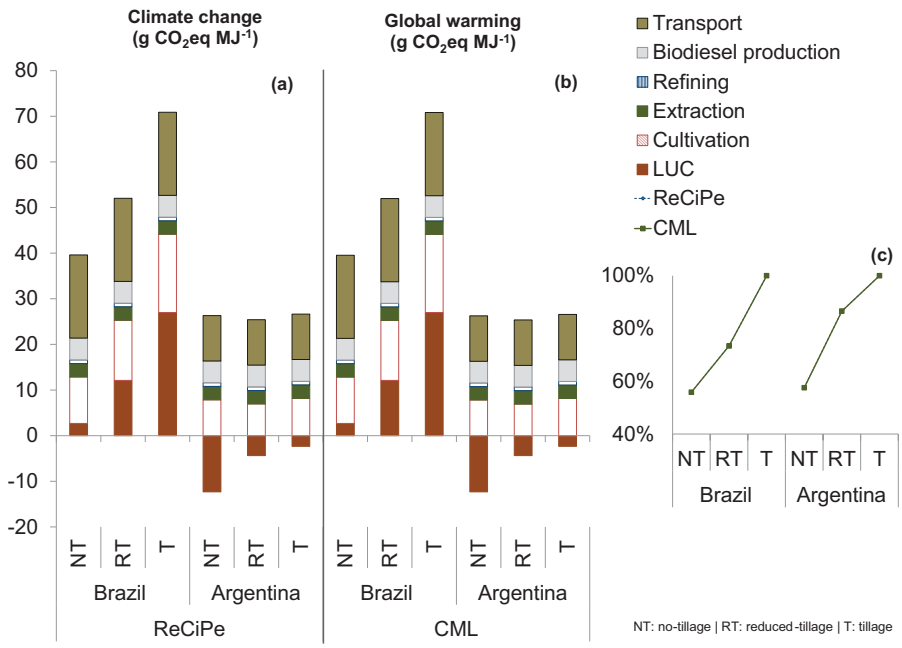


Figure 4.24. Climate change (a) and global warming (b) impact of biodiesel produced in Portugal based on soybean from Brazil and Argentina: alternative cultivation systems and LCIA methods (c).

Terrestrial acidification/acidification (TA/A) impact of soybean biodiesel is presented in **Figure 4.25**. The results vary from 0.2 to 0.3 g SO_2eq MJ^{-1} when ReCiPe and CML methods were adopted. The LC phase contribution to TA/A is as follows: 16 to 39% cultivation (0.03 to 0.11 g SO_2eq MJ^{-1}), 55 to 74% transports (0.13 to 0.16 g SO_2eq MJ^{-1}) and 6 to 10% extraction, refining and biodiesel production (0.02 g

SO_2eq MJ^{-1}). Sulfur dioxide (SO_2) emissions due to fossil fuels combustion are that contribute most to TA/A impact. In both methods, the lowest TA/A impact were obtained for NT systems in Brazil and RT in Argentina. The highest ReCiPe and CML results were calculated for tillage system in Brazil, while the highest result in Argentina depends on the method: tillage system when ReCiPe was used and no-tillage when CML was adopted. Nevertheless, a higher variation was obtained for the cultivation systems in Brazil than in Argentina: tillage system in Brazil are 20% and 8% higher than NT and RT systems, while in Argentina tillage (ReCiPe) and no-tillage (CML) systems are 1-6% and 2-7% higher than remaining systems.

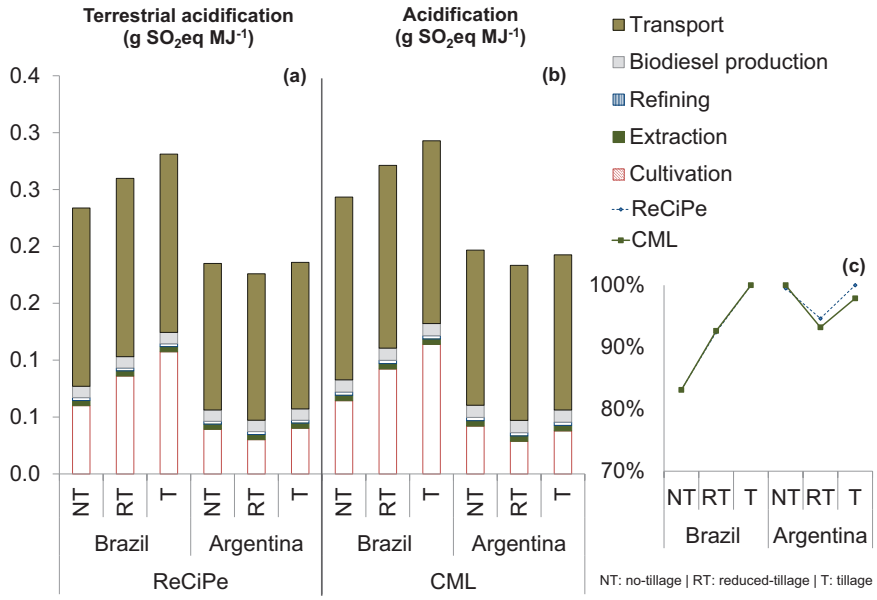


Figure 4.25. Terrestrial acidification (a) and acidification (b) impact of biodiesel produced in Portugal based on soybean from Brazil and Argentina: alternative cultivation systems and LCIA methods (c).

Figure 4.26 presents freshwater eutrophication (a), marine eutrophication (a') and eutrophication (b) impacts of soybean biodiesel. Nitrate field emissions are those contribute most to marine eutrophication (ME) and eutrophication impacts, whereas phosphorus and phosphate field emissions are that contribute most to freshwater eutrophication (FE) impact. For this reason, cultivation is the LC phase that contributes most to these impact categories (more than 79%), for all LCIA methods and cultivation systems in Brazil and Argentina. Biodiesel produced from soybean cultivated under tillage system in Brazil and Argentina have the highest FE, ME and eutrophication impacts.

A huge variation can be observed for the FE impact. In both countries, the lowest FE impact was calculated for NT systems (15-21 mg Peq MJ^{-1}) and the highest for T systems (25-57 mg Peq MJ^{-1}). FE results for Brazil are significantly higher than the results obtained for Argentina (42-127%). In contrast, ME impact varies from 0.20 to 0.22 g Neq MJ^{-1} and no significant variation (less than 8%) occurs

among all cultivation systems and countries. Eutrophication impact varies from 0.18 g PO_4^{3-}eq MJ^{-1} (NT and RT, Argentina) to 0.30 g PO_4^{3-}eq MJ^{-1} (T, Brazil). Eutrophication results of tillage system in Brazil are 31-54% higher than NT and RT, whereas in Argentina are 11-12% higher than NT and RT systems.

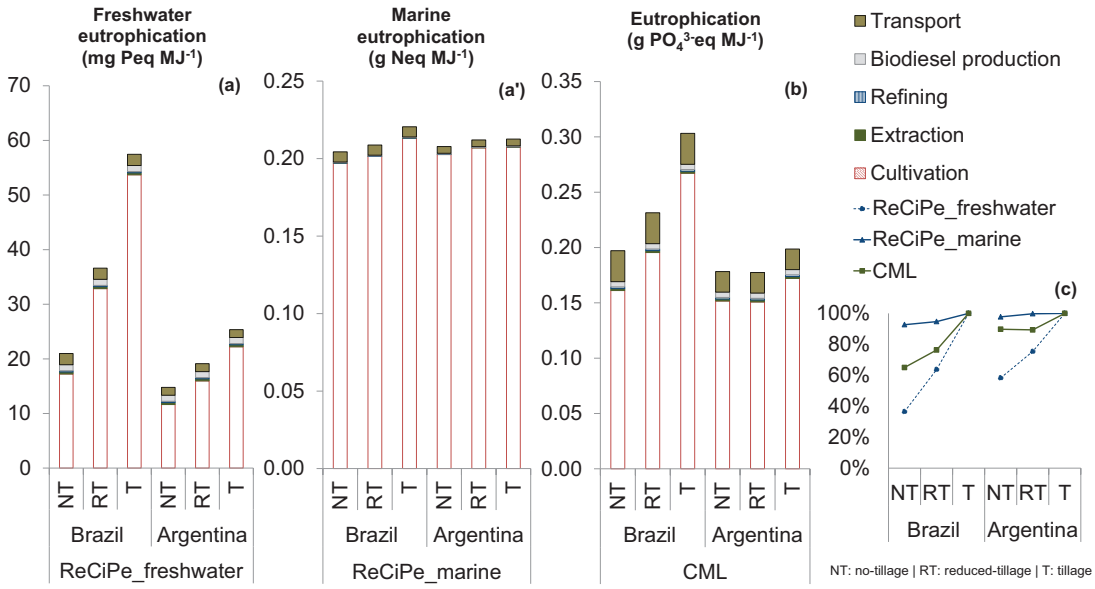


Figure 4.26. Freshwater (a), marine eutrophication (a') and eutrophication (b) impact of biodiesel produced in Portugal based on soybean from Brazil and Argentina: alternative cultivation systems and LCIA methods (c).

Ozone depletion/ozone layer depletion (OD) impact varies from $3.1x10^{-6}$ g CFC-11eq MJ^{-1} (RT, Argentina) to $6.0x10^{-6}$ g CFC-11eq MJ^{-1} (T, Brazil) and the results are presented in **Figure 4.27**. The LC phase contribution to OD is as follows: 40 to 55% transport, 18 to 38% cultivation, 5 to 11% extraction and 13 to 28% oil refining and biodiesel production. Similar OD impact was obtained for the different LCIA methods. The variation among cultivation systems occur but are more evident for Brazil than Argentina. The lowest OD impact was calculated for NT systems in Brazil and RT in Argentina, while the highest was obtained for tillage system in both countries. OD impact is caused essentially by halon 1301 (bromotrifluoromethane) emissions.

Photochemical oxidant formation/photochemical oxidation (POF) impact of soybean biodiesel are presented in **Figure 4.28**. POF impact calculated using ReCiPe method varies from 0.22 to 0.36 g NMVOCeq MJ^{-1} and from 41.6 to 45.4 mg C_2H_4eq MJ^{-1} with CML. In both methods the lowest POF impact was calculated for RT system in Argentina and the highest for tillage system in Brazil. The ReCiPe results are higher for tillage system than NT and RT systems in Brazil (8-16%) and Argentina (6-8%). A slight variation is observed for the CML results: tillage results in Brazil are 1-4% higher than NT and RT, while NT results in Argentina are 2% higher than RT and T. The contribution of LC phases

for POF impact varies for both methods. Transportation emissions were that contribute most to POF impact (49-59%) when ReCiPe method was adopted, followed by soybean cultivation emissions (13-28%). Regarding the CML method, soybean oil extraction emissions were that contribute most to POF impact (78-85%), while transportation emissions represented less than 12%. Nitrogen oxides emissions were that contribute most to ReCiPe results, whereas POF results obtained with CML method were mostly due to hexane emissions.

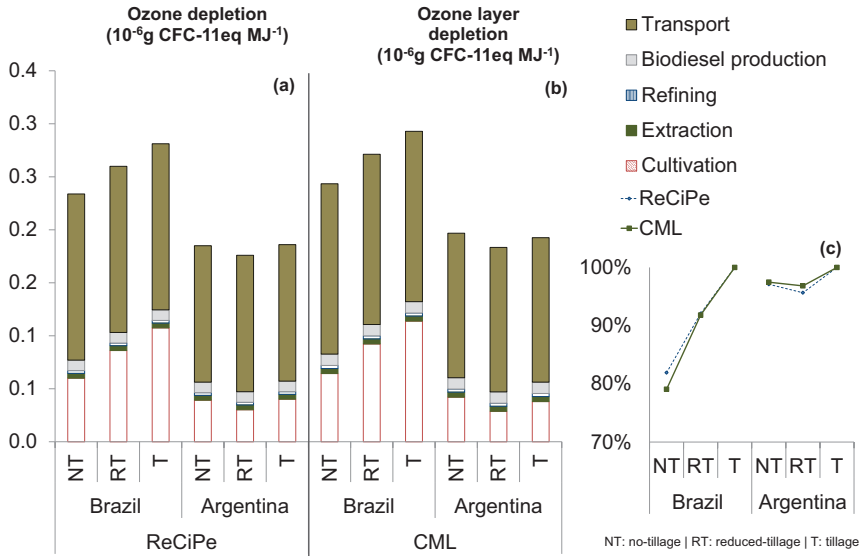


Figure 4.27. Ozone depletion (a) and ozone layer depletion (b) impact of biodiesel produced in Portugal based on soybean from Brazil and Argentina: alternative cultivation systems and LCIA methods (c).

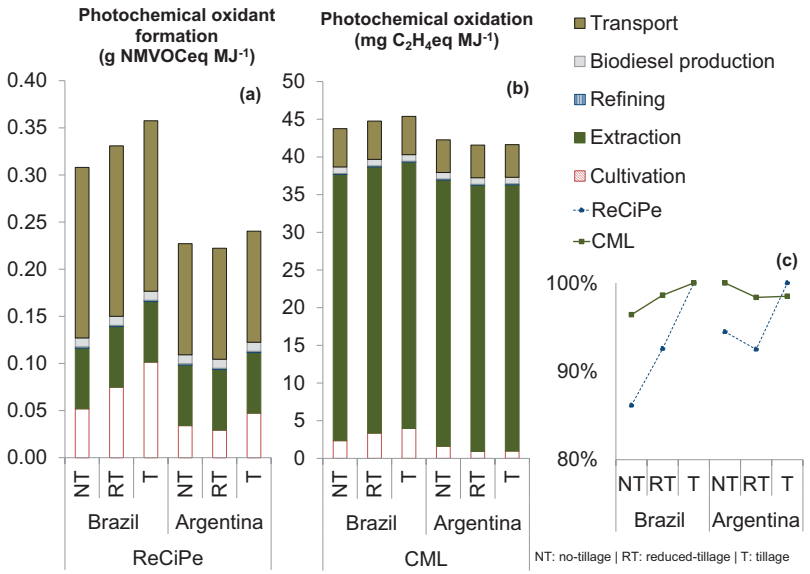


Figure 4.28. Photochemical oxidant formation (a) and photochemical oxidation (b) impact of biodiesel produced in Portugal based on soybean from Brazil and Argentina: alternative cultivation systems and LCIA methods (c).

Normalized LCIA results of biodiesel based on soybean produced in Brazil and Argentina are presented in **Figure 4.29**. Marine and freshwater eutrophication (ReCiPe) and eutrophication (CML) are the most important environmental impacts. GHG intensity, acidification and photochemical oxidation are also important categories, in particular when CML method was adopted. The importance of ozone depletion/ozone layer depletion is substantially lower than the other categories in both methods. The results in both countries vary according to the three cultivation systems and LCIA method. However, in Argentina, the variation in the CML results among cultivation systems is more significant than the variation in the ReCiPe results. The CML results for tillage system in Argentina are 16-134% higher than remaining systems, while tillage system results are 1-72% higher than remaining systems with ReCiPe method. In contrast, the variation in the ReCiPe results for Brazilian tillage system (6-174%) is higher than the variation in the CML results (1-54%).

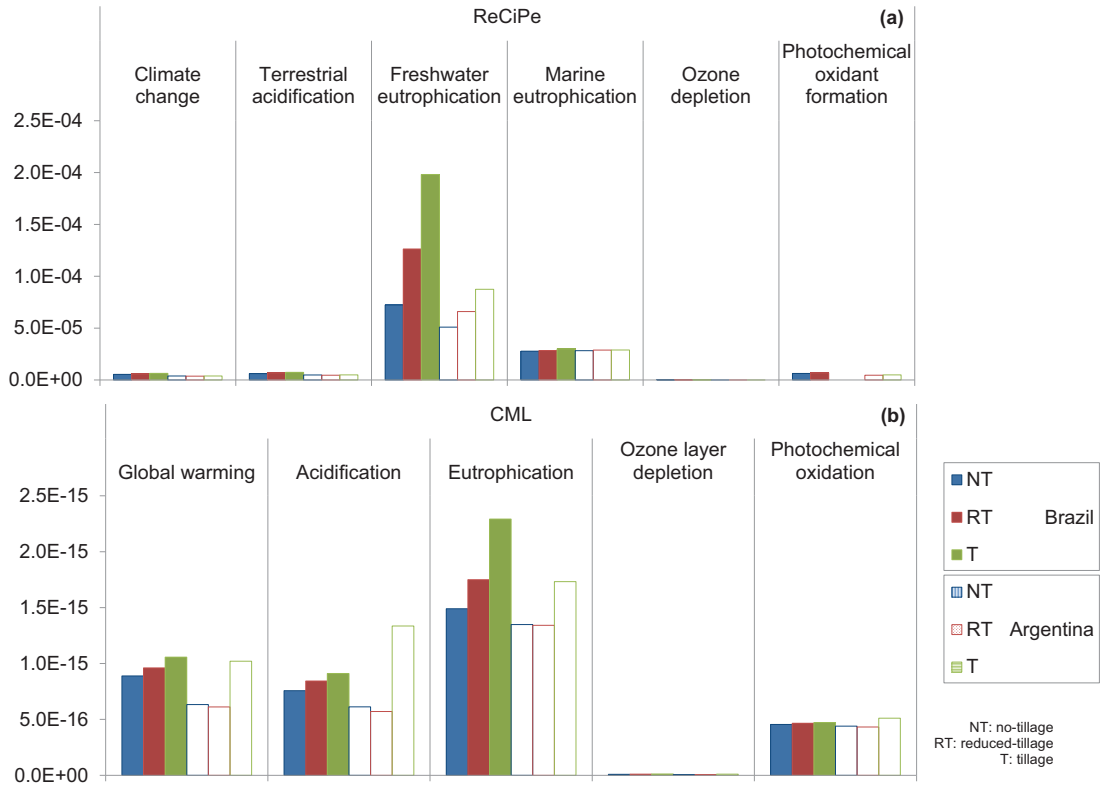


Figure 4.29. Normalized LCIA results of biodiesel produced in Portugal based on soybean from Brazil and Argentina: (a) ReCiPe versus (b) CML.

ReCiPe results shows that tillage system in Brazil has the highest values for all impact categories. The lowest CC, TA, OD and POF normalized results were calculated for RT systems in Argentina, while NT system in Argentina and Brazil has the lowest FE and ME results, respectively. The highest normalized CML results were calculated for tillage system in both countries: Brazil has the highest GW, eutrophication and OD results, while Argentina presents the highest results for acidification and POF.

The lowest results were obtained for NT and RT systems in Argentina for all CML environmental impact categories.

BIODIESEL CHAIN C

The human toxicity and ecotoxicity impacts of soybean biodiesel chain C were calculated and the results obtained from the application of ReCiPe 1.07 and USEtox 1.01 ("recommended" and "recommended+interim") methods were compared to determine the extent to which the results are influenced by the method applied. It should be noted that ReCiPe non-toxicity impact categories were presented and detailed analyzed previously (**sub-chapter 4.2** and **4.3**).

Figure 4.30 shows the human toxicity (HT) impact of soybean biodiesel, focusing on the contribution of each LC phase for the alternative pathways (C1, C2, C3) and soybean cultivation locations (MT, GO, PR, RS). ReCiPe results vary from 6.8 to 8.0 g 1,4-DBeq MJ^{-1} and the contribution of LC phases is as follow: 39-57% cultivation, 23-40% transport, 12-15% biodiesel production and 8-10% oil extraction and refining. HT impact obtained from USEtox varies from $1.8x10^{-9}$ to $6.2x10^{-9}$ CTUh MJ^{-1} ("recommended") and from $-6.3x10^{-8}$ to $-6.0x10^{-8}$ CTUh MJ^{-1} ("recommended+interim"). Cultivation emissions contribute more than 90% to HT impact when USEtox method was adopted.

The emissions contribution to human toxicity impact varies depending on the method applied. Manganese emissions due to energy consumption and acephate emissions from pesticides application (organophosphorus-compounds) are that contribute most to ReCiPe human toxicity results. Also USEtox ("recommended") HT results are mostly related to acephate emissions. The differences on the emission contribution are related to the use of the "recommended" version of USEtox, which does not include characterization factors for manganese emissions. Because soybean grains have higher zinc content than the inputs (the fertilizers applied) there is a zinc uptake, which contributes to the negative USEtox ("recommended+interim") results. Additionally, zinc uptake also makes an important contribution to ReCiPe HT results.

For ReCiPe and USEtox ("recommended+interim") a slight variation on HT impact was obtained among the four states where soybean was cultivated. In contrast, USEtox ("recommended") HT impact varies significantly (99-242%): the highest HT impact was obtained for biodiesel produced with soybean from PR and the lowest HT impact for biodiesel produced with soybean from MT. Comparing pathways, the highest ReCiPe results were obtained for pathway C1, when soybean was cultivated in GO, PR and RS, while the lowest were calculated for pathway C3 (except when soybean is from MT, in which C2 has the lowest result). Regarding the USEtox results, no significant variation among pathways was observed.

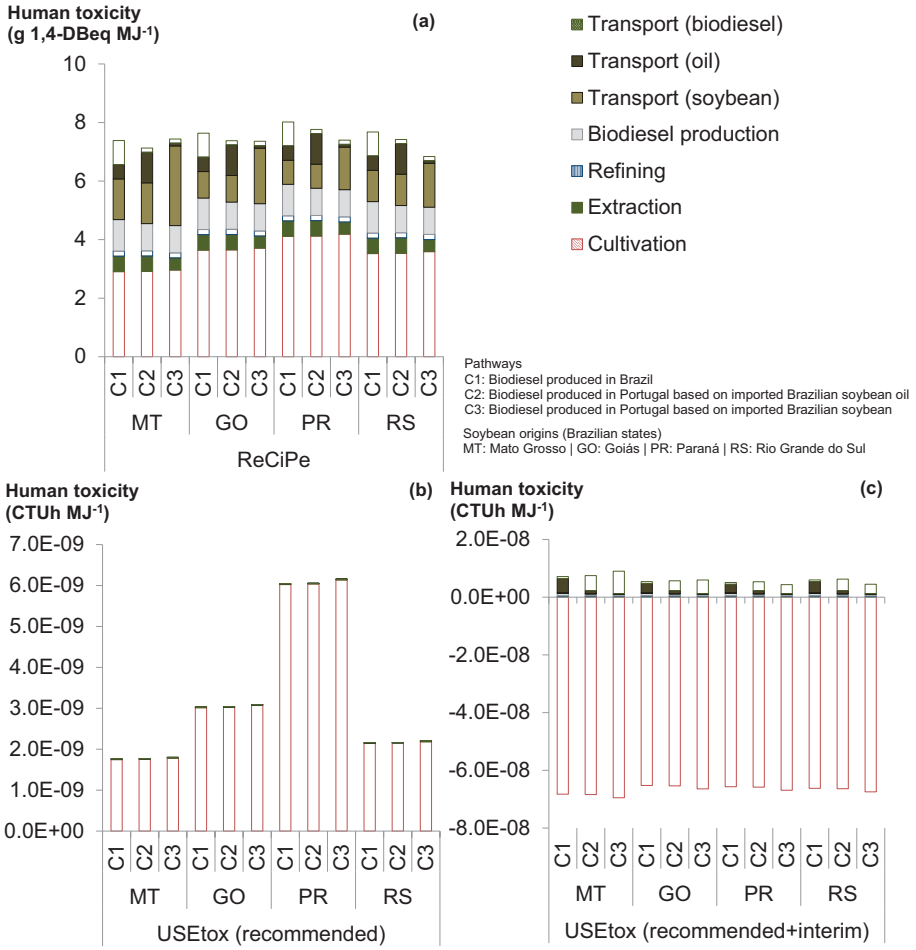


Figure 4.30. Human toxicity impact of soybean biodiesel: contribution of each LC phase and comparison of ReCiPe (a), USEtox "recommended" (b) and "recommended+interim" (c) results.

ReCiPe ecotoxicity impacts are presented in **Figure 4.31**. The ecotoxicity impacts calculated using ReCiPe method are separated in freshwater, marine and terrestrial ecotoxicity. It can be seen that soybean cultivation is the LC phase that contributes most to these impact categories (50-100%). Freshwater ecotoxicity (FE) impact varies from 0.9 to 1.3 g 1,4-DBeq MJ^{-1} and marine ecotoxicity (ME) from 0.21 to 0.24 g 1,4-DBeq MJ^{-1}, mainly due to phosphorus emissions from fertilizers application. Biodiesel produced with soybean from PR has the highest FE impact and the lowest with soybean from RS. No significant variation on ME impact occurs among soybean origins. Concerning terrestrial ecotoxicity (TE), the results vary from 0.6 g 1,4-DBeq MJ^{-1} (soybean from MT) to 1.9 g 1,4-DBeq MJ^{-1} (soybean from PR), due to the cypermethrin emissions from pesticides application (pyretroid-compounds).

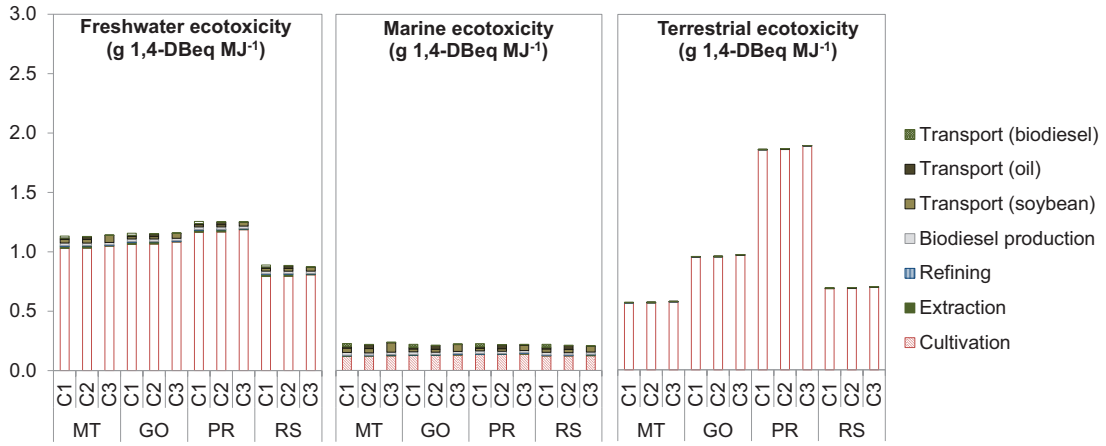


Figure 4.31. Aquatic ecotoxicity impact of soybean biodiesel using ReCiPe method: contribution of each LC phase.

Figure 4.32 presents the freshwater ecotoxicity (ET) impacts of soybean biodiesel calculated using ReCiPe, USEtox "recommended" and "recommended+interim" methods. As shown in **Figure 4.31**, ReCiPe results vary from 0.9 to 1.3 g 1,4-DBeq MJ^{-1} and phosphorus emissions from soybean cultivation are that contribute most to these results. USEtox ecotoxicity impact using "recommended" characterization factors varies little ($5.6x10^{-2}$ to $8.1x10^{-2}$ CTUe MJ^{-1}) compared to the results obtained with USEtox using "recommended and interim" characterization factors ($6.2x10^{-2}$ to $7.6x10^{-2}$ CTUeMJ^{-1}). However, the contribution of each LC phase to the ET impact varies greatly. USEtox "recommended" results are mostly (about 99%) due to the carbendazim soil emissions from fungicide application (benzimidazole-compound) in soybean cultivation phase. The contribution of LC phases when USEtox "recommended+interim" was adopted is as follow: 33-66% cultivation, 28-61% transport and 6-9% biodiesel production, oil extraction and refining.

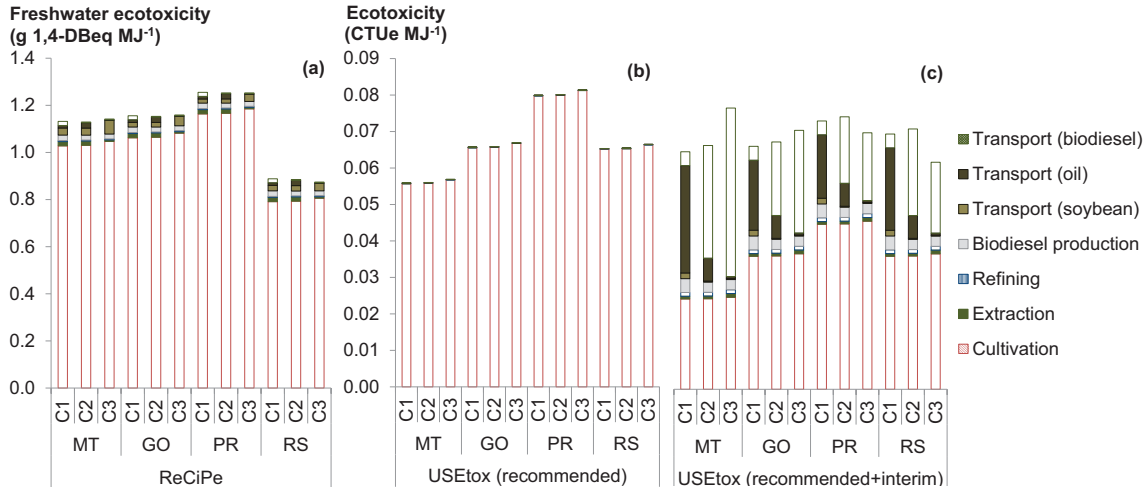


Figure 4.32. Aquatic ecotoxicity of soybean biodiesel: contribution of each LC and comparison of ReCiPe (a), USEtox "recommended" (b) and "recommended+interim" (c) results.

ReCiPe and USEtox ("recommended") ET impacts were the highest when biodiesel was produced with soybean from PR and the lowest when biodiesel was produced with soybean from RS and MT, respectively. Regarding the USEtox ("recommended+interim") ET impact, pathway C1 had the lowest results and C3 the highest, when biodiesel was produced with soybean from MT and GO biodiesel. When biodiesel was produced with soybean from PR and RS, the lowest ET impact was obtained for pathway C3 and the highest for pathway C2.

4.5 THE INFLUENCE OF MULTIFUNCTIONALITY APPROACH

This sub-chapter concerns the following objectives:

"4.1 Perform a sensitivity analysis for alternative multifunctionality approaches."

"4.2 Evaluate the influence of various multifunctionality approaches on LCA results."

Two substitution scenarios and alternative allocation procedures (energy, price and mass) were adopted for palm and soybean biodiesel chain (A and C). The effect of multifunctionality approach is analyzed based on the LCIA performed with ReCiPe 1.07 method, adopting the site-specific models for field nitrogen emission calculations and the LUC emissions related to the expansion of the Colombian palm area (from 1990 to 2010) and the soybean area in each Brazilian state (from 1985 to 2006). The following impact categories are analyzed: GHG intensity, terrestrial acidification, freshwater eutrophication and photochemical oxidant formation.

BIODIESEL CHAIN A

The LCIA results were calculated using allocation and two substitution scenarios (A and B). In the substitution scenarios it was assumed that palm kernel meal substitute soybean meal on the market. The lowest (scenario A) and highest (scenario B) impact of soybean meal in each category was obtained from soybean cultivation in different states and alternative pathways (biodiesel chain C), as presented in **Table 4.1**. The effect of multifunctionality on the environmental impacts of palm biodiesel is presented in **Figure 4.33**. It can be seen that multifunctionality influences the results, but the extent of this influence depends on the impact categories, the fertilization schemes and biogas management options assessed. The reason for this is the variation on the contribution of each LC phase among the various environmental impact categories, considering all co-products (no multifunctionality approach).

Table 4.1. Biodiesel chain A: substitution scenarios.

Impact categories	Substitution scenarios	
	A: Lowest soybean meal impact	B: Highest soybean meal impact
GHG intensity	RS, C1 and C2	MT, C3
Terrestrial acidification	PR, C2	PR, C1
Freshwater eutrophication	PR, C3	PR, C1
Photochemical oxidant formation	PR, C2	PR, C1

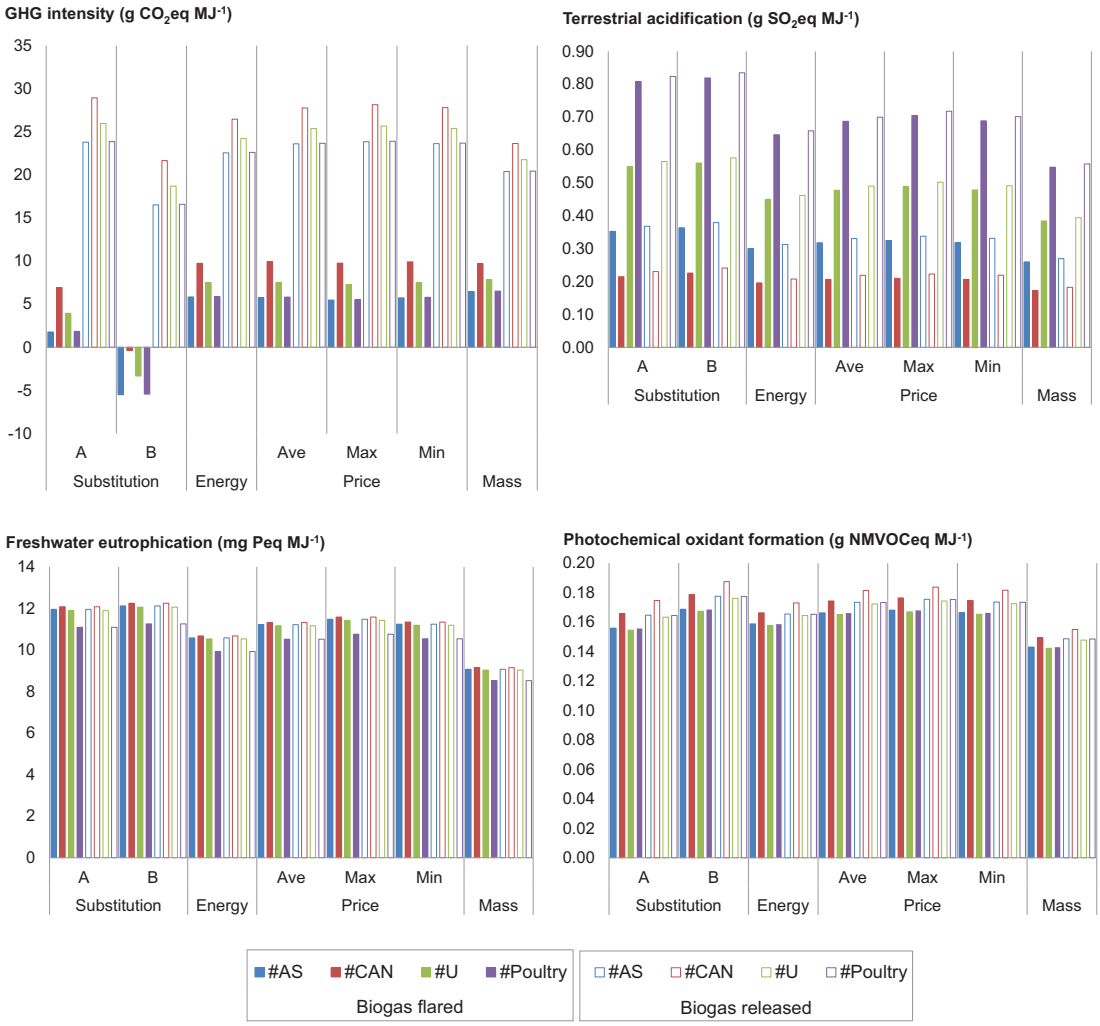


Figure 4.33. Effect of multifunctionality on LCIA results of palm biodiesel chain A: alternative fertilization schemes (#AS, #CAN, #U and #Poultry) and biogas management options (flared and released).

There is no significant variation (less than 8%) on acidification, eutrophication and photochemical oxidant formation impacts among substitution scenarios A and B. However, GHG intensity varies significantly according to the substitution scenarios. When scenario A was adopted (lowest GHG intensity of soybean meal produced in Brazil with soybean from RS was avoided), the GHG intensity results of palm biodiesel are higher (more than 34%) than when scenario B was adopted (highest GHG intensity of soybean meal produced in Portugal with soybean from MT). Comparing the three

approaches adopted for allocation based on price of products (average prices, prices for maximum and minimum ratio palm oil/palm kernel meal), it can be seen that there is a minor variation on the impacts. Although annual prices of palm oil and palm kernel oil vary widely, this finding can be explained by the relatively high mass share of the main product (palm oil: 72%) compared with the co-products.

GHG intensity varies widely according to the different fertilization schemes, biogas management options and multifunctionality approaches: -5.5 g CO_2eq MJ^{-1} (#AS, biogas captured and flared, substitution_B) and 28.9 g CO_2eq MJ^{-1} (#CAN, biogas released, substitution_A). The lowest GHG intensity was obtained in the substitution scenario B (highest GHG intensity of soybean meal produced in Portugal with soybean imported from Mato Grosso) for all fertilization schemes and biogas management options. However, the highest result among the multifunctionality approaches depends on the fertilization schemes and biogas management options. When biogas was released, the highest GHG intensity was obtained with price (max) allocation (#AS and #Poultry) and substitution A (#CAN and #U). On the opposite, when biogas is released captured and flared, mass (#AS, #U and #Poultry) and price (#CAN) allocation results are the highest. This variation is related to the contribution of LC phases to the GHG intensity of palm biodiesel. Emissions associated with carbon stock changes due to LUC (about -32 g CO_2eq MJ^{-1}, no allocation) are that contribute most to this impact and the second main contribution depends on the biogas management option: oil extraction contributes about 26-28% when biogas was released and less than 4% when biogas was captured and flared.

The variation on acidification, eutrophication and photochemical oxidant formation impacts among multifunctionality approaches is 31-50%, 32-34% and 18-21%, respectively. The lowest acidification, eutrophication and photochemical oxidant formation impacts were obtained when mass allocation was adopted, while the highest were obtained in the substitution scenario A (lowest impacts of soybean meal). Energy and price allocation (average, min and max ratio) results are similar in all this categories. Terrestrial acidification impact of palm biodiesel varies from 0.17 g SO_2eq MJ^{-1} (#CAN, biogas captured and flared) to 0.83 g SO_2eq MJ^{-1} (#Poultry, biogas released) and freshwater eutrophication from 8.5 mg Peq MJ^{-1} (#Poultry) to 12.2 mg Peq MJ^{-1} (#CAN). Photochemical oxidant formation impact varies from 0.14 g NMVOCeq MJ^{-1} (#U, biogas captured and flared) to 0.19 g NMVOCeq MJ^{-1} (#CAN, biogas released).

BIODIESEL CHAIN C

Figure 4.34 shows the LC impacts of soybean biodiesel, based on soybean from Mato Grosso (MT), Goiás (GO), Paraná (PR) and Rio Grande do Sul (RS) and alternative pathways: totally produced in Brazil (C1), export soybean oil (C2) or soybean (C3) to Portugal where oil extraction and biodiesel production occur. The results presented were calculated based on two substitution scenarios (A and B), energy, price and mass allocation. The substitution scenarios are presented in **Table 4.2** and were

defined based on the lowest and highest impact in each category of substituted product: palm oil (biodiesel chain A) produced under different fertilization schemes (#AS, #CAN, #U and #Poultry) and biogas management options (biogas released or captured and flared).

Table 4.2. Biodiesel chain C: substitution scenarios.

Impact categories	Substitution scenario			
	A: Lowest palm oil impact		B: Highest palm oil impact	
Climate change	#AS	Biogas captured and flared	#CAN	Biogas released
Terrestrial acidification	#CAN		#Poultry	
Freshwater eutrophication	#Poultry		#CAN	
Photochemical oxidant formation	#U		#CAN	

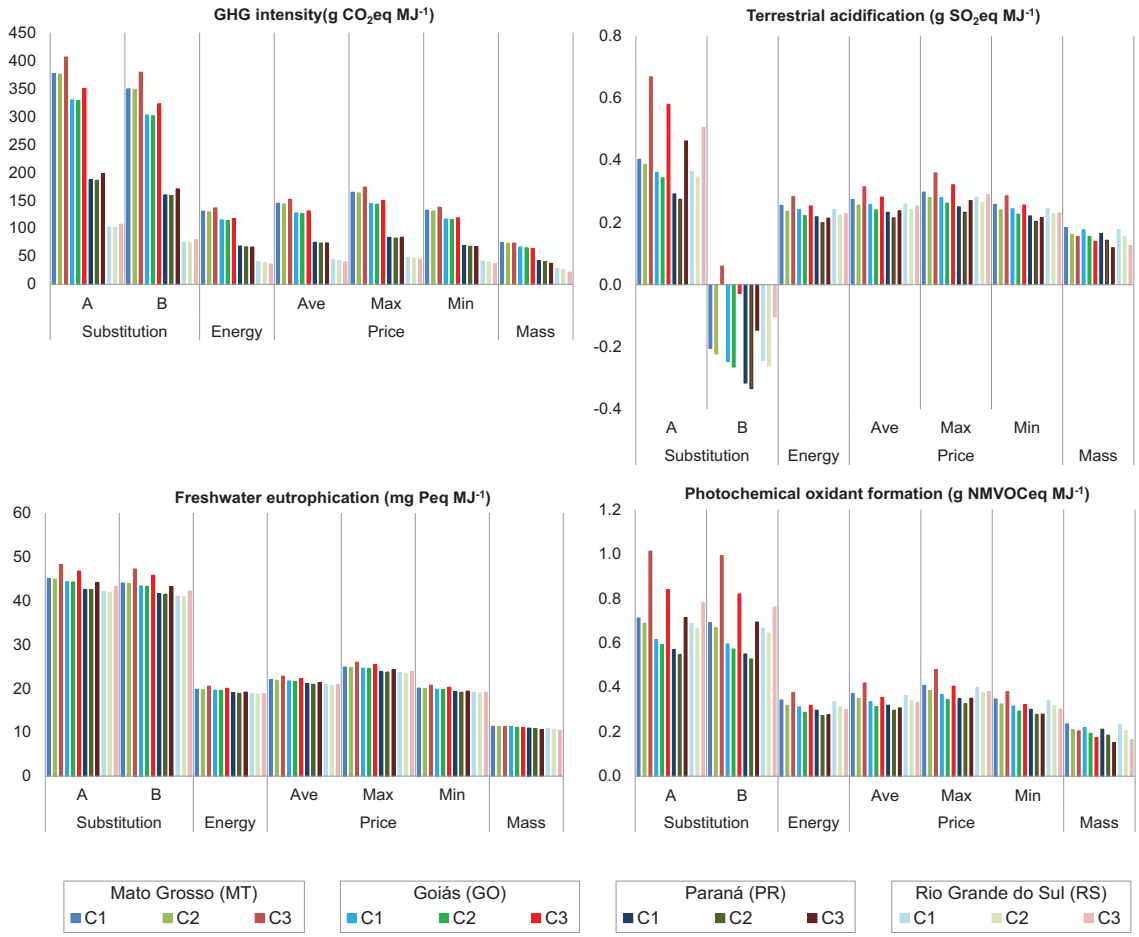


Figure 4.34. Effect of multifunctionality on LCIA results of soybean biodiesel chain C: alternative soybean origins (Brazilian states) and pathways (C1, C2 and C3).

The sensitivity analysis of multifunctionality approach shows that the environmental impacts of soybean biodiesel depend greatly on the approach adopted. The highest impacts were obtained in the substitution scenario A (lowest impacts of palm oil produced in Colombia) in the four environmental

impact categories. The lowest GHG intensity, freshwater eutrophication and photochemical oxidant formation impacts were calculated using mass allocation, while the lowest terrestrial acidification impact was obtained in the substitution scenario B (highest impacts of palm oil produced in Colombia).

The results obtained in the four environmental impact categories are similar when energy and price allocation were adopted. Energy and price allocation results are about the 32-135% higher than mass allocation results for all impact categories, soybean origins and pathways. Comparing the price allocation approaches (price_ave, price_max and price_min), there is some variation between the results: the highest were obtained when soybean oil and meal ratio was the maximum and they are 13-26% higher than the lowest results. Regarding the two systems expansion scenarios, it can be seen that there is no significant variation (less than 4%) on eutrophication and photochemical oxidant formation impacts among scenarios A and B. However, GHG intensity varies from 7% to 37% among substitution scenarios and terrestrial acidification impacts obtained in scenario A are 2 to 20 times higher than the results obtained in scenario B.

GHG intensity ranged from 22 g CO_2eq MJ^{-1} (RS, C3) to 406 g CO_2eq MJ^{-1} (MT, C3), terrestrial acidification from -0.33 g SO_2eq MJ^{-1} (PR, C2) to 0.67 g SO_2eq MJ^{-1} (MT, C3), freshwater eutrophication from 10 mg Peq MJ^{-1} (RS, C3) to 48 mg Peq MJ^{-1} (MT, C3) and photochemical oxidant formation ranged from 0.15 g NMVOCeq MJ^{-1} (PR, C3) to 1.01 g NMVOCeq MJ^{-1} (MT, C3). It can be observed that the highest result in all impact categories was obtained for biodiesel produced in Portugal based on soybean imported from Mato Grosso, Brazil (MT, C3). However, the lowest and the highest environmental impacts calculated for the alternative pathways depend on the multifunctionality approach adopted. For instance, mass allocation results for all impact categories shows that they are the highest for pathway C1, whatever the soybean origin. In contrast, the impacts calculated using substitution (A and B) are the highest in pathway C3, whatever the soybean origin. Regarding the soybean origin, the highest and lowest results obtained in each category are the same for all multifunctionality approaches, whatever the pathway.

4.6 GREENHOUSE GAS SAVINGS

This sub-chapter concerns the following objective:

"5.1 Assess the GHG emission savings when palm and soybean biodiesel replace fossil diesel."

The GHG emission savings calculated for soybean and palm biodiesel are compared with the GHG saving criteria for biodiesel stipulated by the Renewable Energy Directive (RED, Directive 2009/28/EC, EC, 2009): GHG savings should be at least 35% until the end of 2016, 50% until the end of 2017 and 60% after 2017, taking into account emissions from whole biofuel production and distribution chain,

including the emissions from carbon stock changes caused by direct LUC. The GHG intensity was calculated based on ReCiPe 1.07 method and on energy allocation approach (as defined in the RED); however, the effect of multifunctionality approach on the GHG savings is also analyzed in this sub-chapter. Site-specific models for field nitrogen emission calculations in biodiesel chain A and C were adopted (IPCC model was adopted in biodiesel chain B).

Figure 4.35 shows the GHG savings from replacing diesel with palm and soybean biodiesel (chain A and C). The results for palm and soybean biodiesel were obtained considering the LUC emissions related to the expansion of the Colombian palm area (from 1990 to 2010) and the soybean area in each Brazilian state (from 1985 to 2006). Typical and default[a] GHG savings for palm and soybean biodiesel, as well as the GHG saving criteria for biodiesel defined in the RED (EC, 2009), are also presented.

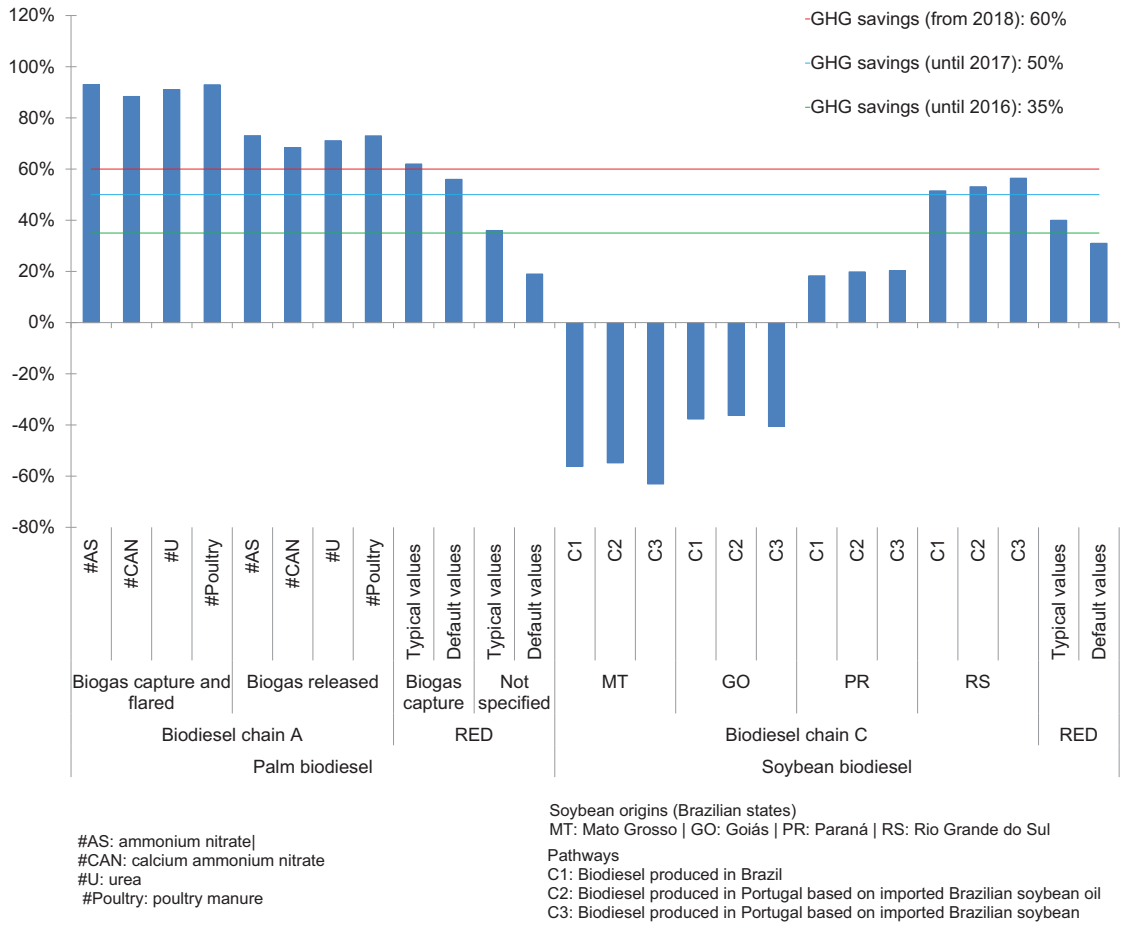


Figure 4.35. GHG savings from replacing diesel with palm and soybean biodiesel (chain A and C): alternative palm fertilization schemes, biogas management options, soybean origins and pathways.

[a]The Directive includes 'default values' which economic operators can use to show compliance with the sustainability criterion on greenhouse gas savings. The default values are set at a conservative level to make it unlikely for economic operators — by using default values — to be claiming values that are better than their actual value (EC, 2010a).

It can be seen that palm biodiesel (chain A) achieved the GHG saving criteria stipulated by RED (35%, 50% and 60%), regardless of the fertilization scheme or biogas management option, whereas soybean biodiesel (chain C) only achieves 35% and 50% of savings when soybean is produced in Rio Grande do Sul. Since LUC emissions are that contribute most to GHG intensity in both chains, these results can be explained by the fact that negative LUC GHG intensity (-24 g CO_2eq MJ^{-1}) due to the expansion of the Colombian oil palm area, while the LUC emissions as a result of the expansion of the soybean area vary from 33 g CO_2eq MJ^{-1} in Paraná to 88 g CO_2eq MJ^{-1} in Mato Grosso. Although the type (savanna and shrubland) and area (70-80%) of land that is converted into palm and soybean is similar in Colombia and in the Brazilian states of Mato Grosso and Goiás, the LUC emissions vary widely because palm is a perennial crop (vegetation carbon stock, C_{veg} = 60 t ha^{-1}) while soybean is an annual crop (C_{veg} = 0 t ha^{-1}).

It was found that GHG savings (typical and default) defined in RED differ considerably from the GHG savings calculated in this thesis. The GHG savings calculated for palm biodiesel (biogas capture) are 43-50% higher than typical savings and 58-66% higher than default values from RED. Also, even when it was considered that biogas at palm oil mill was released, the GHG savings calculated are 90-103% and 260-285% higher than typical and default savings of palm biodiesel (process not specified) from RED. On the other hand, there is no GHG savings when biodiesel was produced based on soybean cultivated in Mato Grosso and Goiás, whereas the GHG savings from biodiesel produced with soybean from Paraná are significantly lower than typical and default values from RED. However, when soybean was produced in Rio Grande do Sul (no LUC occur), the GHG savings are higher than the typical and default values given by the RED. It should be noted that default GHG savings from palm biodiesel (process not specified) and soybean biodiesel given by the RED do not meet the GHG saving criteria for biodiesel defined by the RED.

The effect of multifunctionality approach on the GHG savings is presented **Figure 4.36**. Two scenarios of substitution, energy, price and mass allocation were adopted as described in **sub-chapter 3.10**. There is a significant difference between the effects of multifunctionality on the GHG savings. Comparing the GHG savings from replacing diesel with palm oil biodiesel, considering the various multifunctionality approaches, it can be seen that they are similar and both are lower than the GHG saving criteria of 35%, 50% and 60% sets out by the RED. Regarding the soybean biodiesel chain C, the GHG savings are greatly influenced by the multifunctionality approach adopted. Adopting the substitution scenarios A and B, GHG saving criteria established by RED were not achieved, whereas the GHG savings calculated using energy and price allocation only achieved 35% when soybean was cultivated in Rio Grande do Sul. The minimum of 35% of GHG savings was also reached when mass allocation was adopted, however, only if soybean was from Paraná or Rio Grande do Sul.

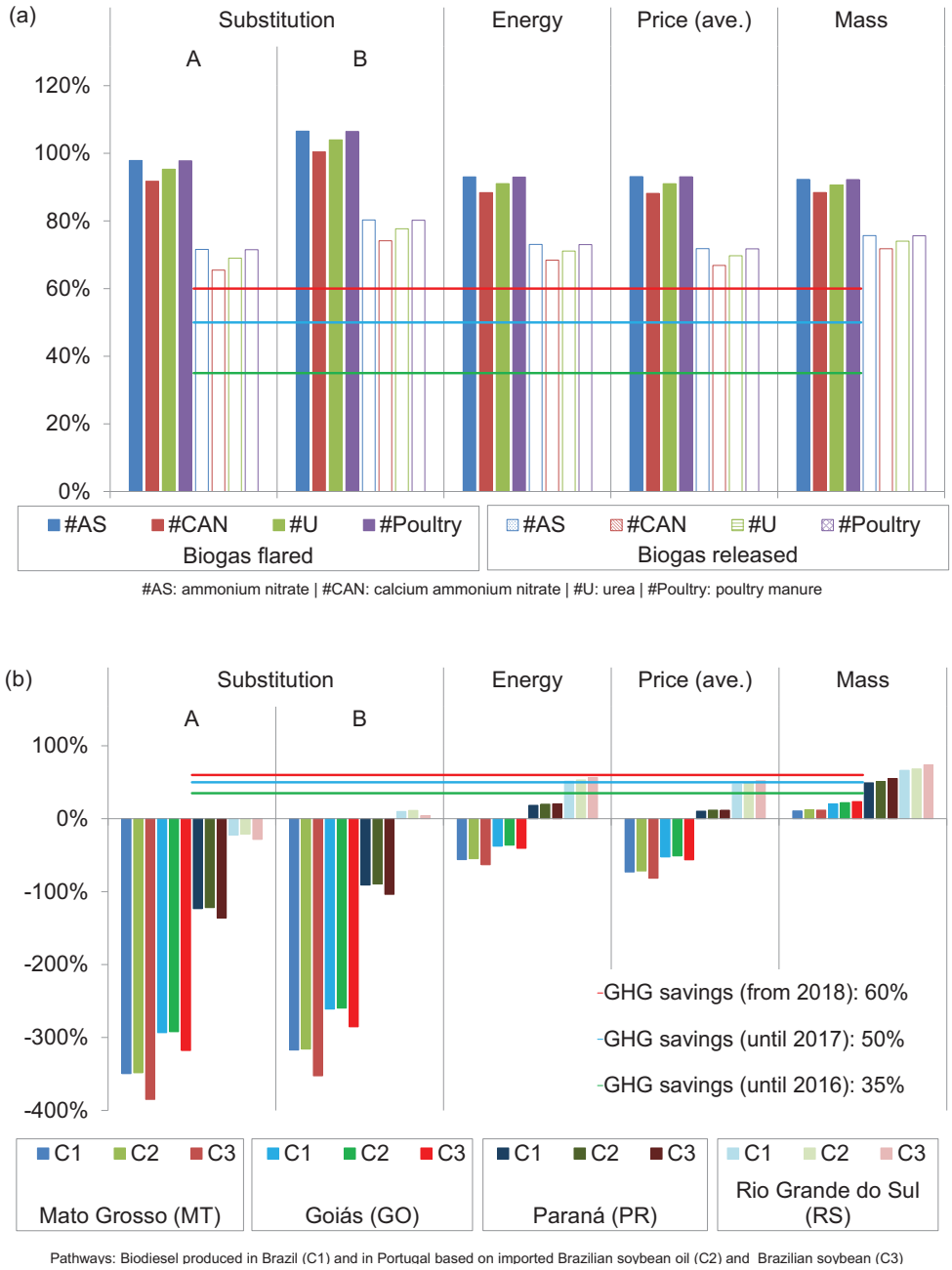


Figure 4.36. GHG savings from replacing diesel with palm (a) and soybean (b) biodiesel using various multifunctionality approaches.

As noted already, carbon stock changes caused by LUC largely influence the GHG savings from palm and soybean biodiesel. For this reason it is important to know how the GHG savings vary among alternative LUC scenarios due to the expansion of palm and soybean areas. **Figure 4.37** presents the GHG savings from replacing diesel with palm biodiesel considering twelve LUC scenarios (L1 to L12) and alternative fertilization schemes and biogas management options. **Figure 4.38** shows the GHG savings from replacing diesel with soybean biodiesel (chain B) considering alternative LUC scenarios established on the basis of a combination of alternative previous land-uses (conversion of tropical forest

land, forest plantations, perennial crop plantations, savanna and grasslands), cultivation systems (tillage, reduced tillage and no-tillage) and climate regions (tropical moist, and warm temperate, moist and dry) in Brazil and Argentina.

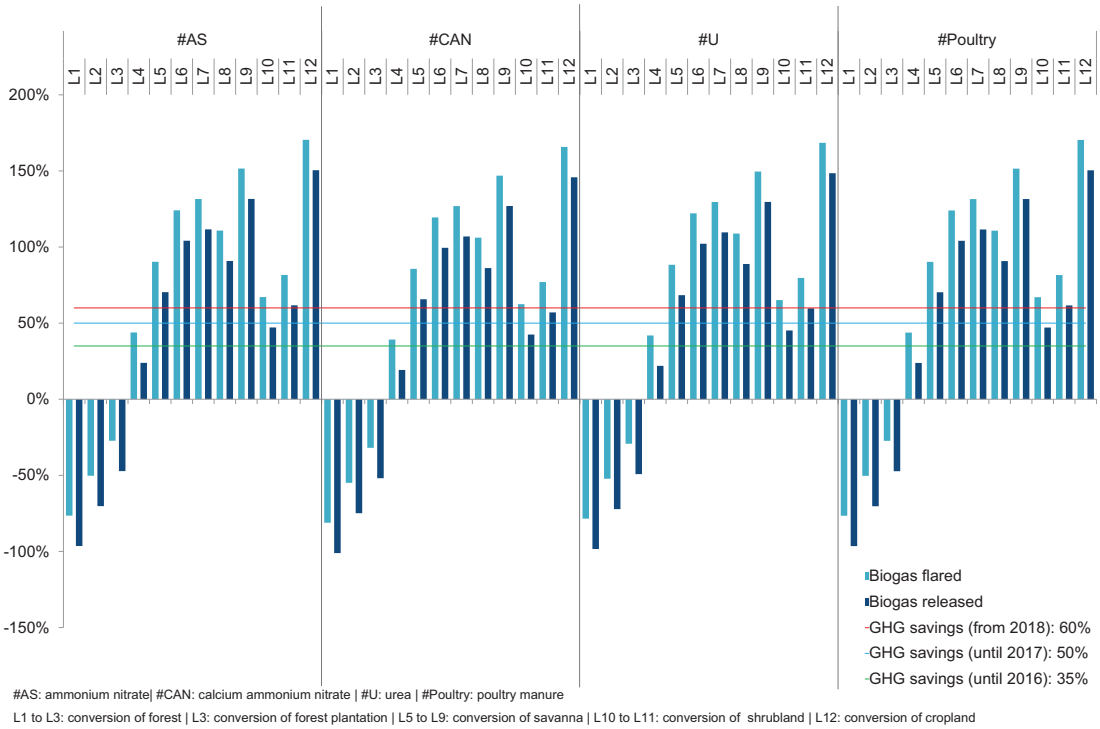


Figure 4.37. GHG savings from replacing diesel with palm biodiesel: alternative LUC, fertilization schemes and biogas management options.

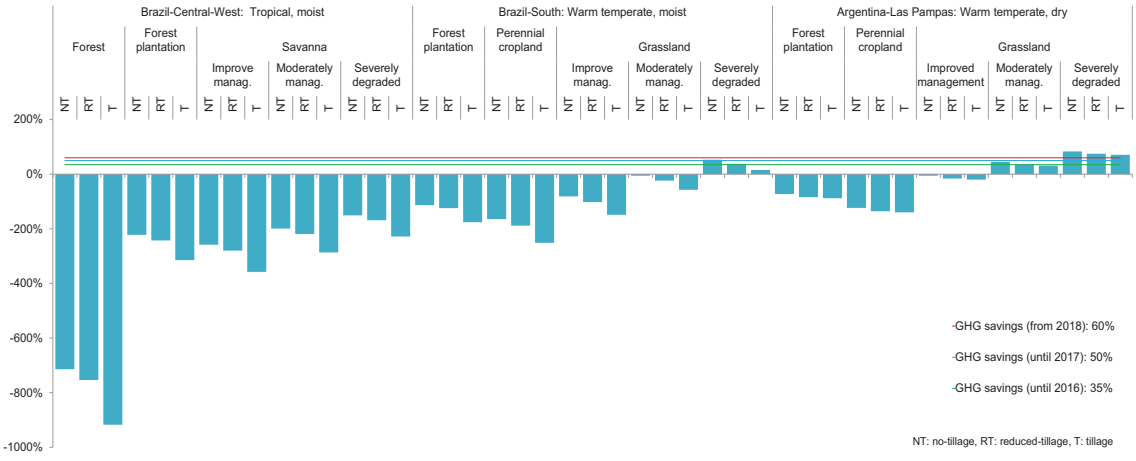


Figure 4.38. GHG savings from replacing diesel with soybean biodiesel: alternative LUC and agricultural systems in three climate regions in Brazil and Argentina.

It can be observed that the GHG savings from replacing diesel with palm biodiesel are higher than 60% in all the scenarios in which savanna and cropland are converted in palm plantations. In the scenarios of shrubland conversion a GHG saving of 60% can be also achieved if biogas was captured and flared. Negative GHG savings (i.e. increased emissions) occur when palm is planted in previous natural forestland (L1 to L3 scenarios). When forest plantations are converted in palm areas the RED GHG saving criteria of 35% is achieved if biogas is captured and flared.

Regarding the soybean biodiesel, GHG emissions are higher than diesel (negative savings) for all LUC scenarios in the tropical region (Brazil, Central-West). Also, when forest plantation, perennial cropland and improved management grassland are converted into soybean plantations in warm temperate regions of Brazil and Argentina, GHG emissions of soybean biodiesel are higher than diesel. A GHG saving of 35% is achieved when severely degraded grassland is converted in a no- and reduced-tillage soybean plantation in warm temperate moist region (Brazil, South), whereas more than 60% is achieved when severely degraded grassland is converted in soybean plantation in warm temperate dry region (Argentina).

In order to determine the main reasons for the differences between the GHG savings from palm and soybean biodiesel chains calculated in this research with those presented in the RED, the GHG emissions of these chains should be compared with the typical and default GHG emissions presented in the RED. This comparison should be made without LUC, since the RED only includes the disaggregated GHG emissions of cultivation, processing (extraction, refining and transesterification) and transport. The process contribution to GHG emissions of palm and soybean biodiesel chains (A, B and C), without LUC, is presented in **Figure 4.39**. The results (energy allocation) are compared with the total GHG emissions of fossil diesel and with the typical and default GHG emissions of each LC phase presented in the RED.

The chart in **Figure 4.39** indicates that GHG emissions calculated in all palm and soybean biodiesel scenarios ensure the GHG saving criteria of 35%, if LUC is not considered. On the opposite, if default GHG emissions from RED were adopted for palm (process not specified) and soybean biodiesel the GHG saving was lower than 35%. Comparing the process contribution to the GHG emissions of biodiesel chains (no LUC), it can be seen that cultivation was the process that contribute most to palm biodiesel emissions if biogas was capture (43-50%), while processing was that contribute most to the total emissions (48-52%) if biogas was released. On the other hand, transportation was the process that contributes most to the soybean biodiesel emissions in all regions and pathways (37-56%).

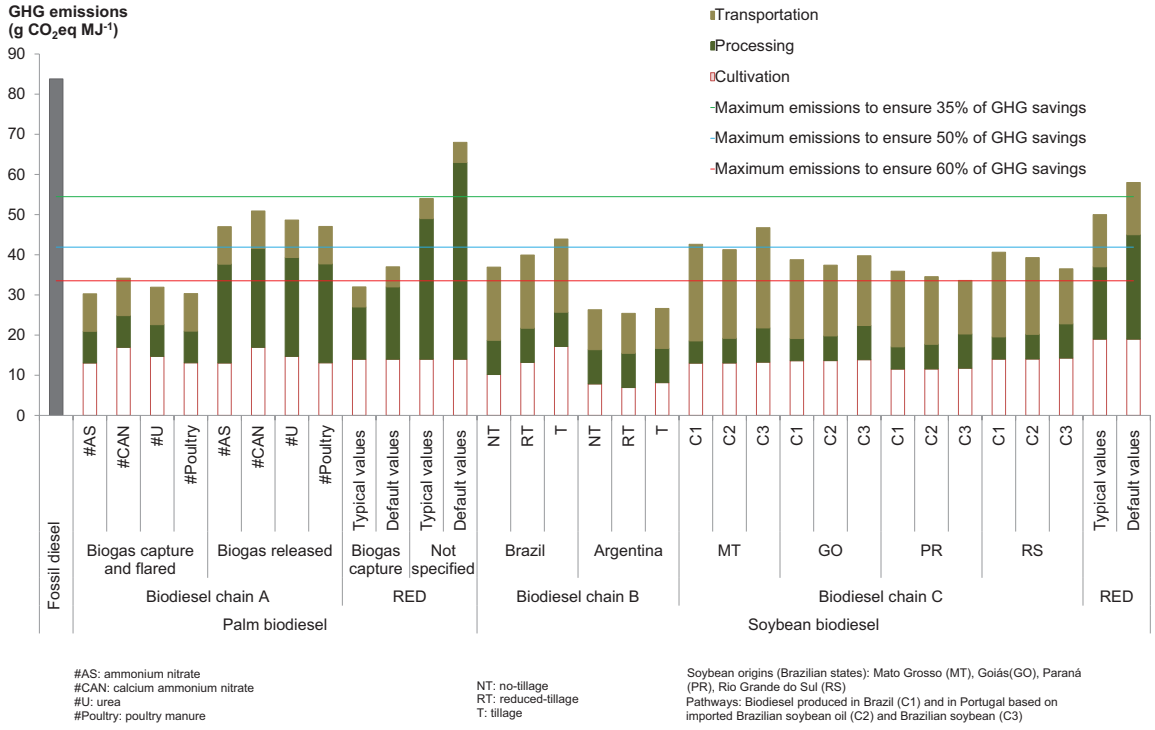


Figure 4.39. GHG emissions of fossil diesel, palm and soybean biodiesel: contribution of LC phases and comparison with RED values.

GHG emissions from cultivation calculated for all palm fertilization schemes (13-17 g CO_2eq MJ^{-1}) are similar to the RED palm cultivation values (14 g CO_2eq MJ^{-1}). However, the calculated emissions from processing (8 and 25 g CO_2eq MJ^{-1}, when biogas was captured and released) are significantly lower than the typical (13 and 35 g CO_2eq MJ^{-1}) and default RED values (18 and 49 g CO_2eq MJ^{-1}). The emissions from transportation (9 g CO_2eq MJ^{-1}) are almost the double than RED transportation emissions (5 g CO_2eq MJ^{-1}).

Regarding the soybean biodiesel, the contribution to GHG emissions also differ from the typical and default values from RED. The soybean cultivation and processing emissions calculated vary, respectively, from 7 to 14 g CO_2eq MJ^{-1} and from 6 to 9 g CO_2eq MJ^{-1}. These values are significantly lower than the cultivation (19 g CO_2eq MJ^{-1}) and processing (18-26 g CO_2eq MJ^{-1}) emissions given by RED. RED transport emissions (13 g CO_2eq MJ^{-1}) are lower than the transport emissions calculated if soybean was cultivated in Brazil (13-25 g CO_2eq MJ^{-1}) and higher if is from Argentina (10 gCO_2eq MJ^{-1}). Nevertheless, transportation emissions calculated vary widely depending on the soybean origin and pathway (10-25 g CO_2eq MJ^{-1}), which greatly influence the total GHG emissions of soybean biodiesel.

4.7 DISCUSSION: CRITICAL ISSUES AND HOTSPOTS

In this sub-chapter the most important methodological aspects and the major sources of uncertainty regarding the environmental sustainability assessment of soybean and palm biodiesel are discussed. Also, the LCIA results and the environmental hotspots of these biodiesel chains are discussed.

LAND-USE CHANGE

The influence of LUC in the GHG intensity of palm and soybean biodiesel was assessed. In palm biodiesel chain A, a combination of alternative fertilization schemes and LUC scenarios (as well as the expansion of the palm area in the 1990-2010 period) was considered for palm production in Colombia. In biodiesel chain B, a comprehensive evaluation was carried out of LUC scenarios, resulting from a combination of LUC and soybean cultivation systems for Brazil (Central-West and South regions) and Argentina (*Las Pampas* region). In biodiesel chain C the LUC emissions were calculated based on the expansion of the soybean area (from 1985 to 2006 period) in four Brazilian states (Mato Grosso, Goiás, Paraná and Rio Grande do Sul). **Table 4.3** summarizes the results obtained for each biodiesel chain.

Table 4.3. GHG intensity of palm and soybean biodiesel chains: LUC emissions contribution.

Biodiesel chain	Country	Region/state	LUC type	GHG intensity	
				Total results (g MJ^{-1})	LUC contribution
A [a]	Colombia	Orinoquía (*Los Llanos*)	LUC scenarios [b]	-59–152	7–80%
			Expansion of palm area (from 1990 to 2010)	6–10	42–45%
B	Brazil	Central-West	LUC scenarios [b]	210–852	82–95%
	Brazil	South	LUC scenarios [b]	40–294	7–85%
	Argentina	*Las Pampas*	LUC scenarios [b]	14–201	8–87%
C [c]	Brazil	Mato Grosso	Expansion of soybean area (from 1985 to 2006)	130–137	66–68%
		Goiás		114–118	66–67%
		Paraná		67–68	48–50%

[a] The results presented were obtained considering that biogas was captured and flared.

[b] Scenarios in which LUC not occur are not included.

[c] No LUC occur in the 1985 to 2006 period in Rio Grande do Sul.

Comparing the GHG intensity of palm and soybean biodiesel chains (A and C), considering the area expansion in Colombia and Brazil, respectively, it can be seen that the GHG intensity of palm biodiesel is significantly lower than of soybean biodiesel. Although the GHG intensity for both chains are mainly due to the LUC emissions, palm area expansion resulted in negative GHG emissions and soybean area expansion in Brazil in positive and with very high LUC emissions (33 to 88 g CO_2eq MJ^{-1}). Even though the type (savanna and shrubland) and area (70-80%) of land that is converted into palm and soybean

areas are similar in Colombia and in Brazilian states of Mato Grosso and Goiás, the LUC emissions is different because palm is a perennial crop (vegetation carbon stock, C_{veg}=60 t ha^{-1}) while soybean is an annual crop (C_{veg}=0 t ha^{-1}).

The findings of this research are consistent with other LCA studies that accounted for carbon emissions from direct LUC in biodiesel chains or biodiesel feedstock. In general, a wide range of results was also reported in previous studies that addressed LUC (see **sub-chapter 2.3.2**). For instance, Reinhard and Zah (2009) showed that the GHG intensity of biodiesel produced with soybean from Brazil may vary from -40 to 210 g CO_2eq MJ^{-1} (consequential LCA), whereas Reijnders and Huijbregts (2008b) presented a ranged values of 146–951 g CO_2eq MJ^{-1} (assuming a LHV of biodiesel of 37 MJ kg^{-1}). van Dam et al. (2009) showed that biodiesel produced from soybean from Argentina had lowest GHG intensity (-39 to 152 and 8 to 95 g CO_2eq MJ^{-1}, respectively). Regarding the palm biodiesel, the GHG intensity vary widely according to the studies and countries where palm was produced: -85 to 3300 g CO_2eq MJ^{-1} in Malaysia (Hassan et al., 2011), 1 to 248 g CO_2eq MJ^{-1} in Thailand (Silalertruksa and Gheewala, 2012a), 53 and 150 g CO_2eq MJ^{-1} in Indonesia (Harsono et al., 2012) or -125 to 182 g CO_2eq MJ^{-1} in Southeast Asia (Lange, 2011).

The contribution of carbon stock changes in vegetation (ΔC_{veg}) and soil (ΔSOC), as well as of N_2O emissions due to a change on land-use, was calculated in palm and soybean biodiesel chains A and B. The results show that LUC emissions occur due to a high ΔC_{veg} in both chains, except when shrubland was converted in palm plantation, as well as when grassland in warm temperate dry region (Argentina) and improved management and moderately degraded grassland in warm temperate moist region (Brazil, South) are converted into soybean plantation. N_2O emissions contribute less than 6% for the LUC GHG intensity calculated in each LUC scenario of these biodiesel chains.

The results also show the importance of adopting site-specific values for the parameters used in the calculation of annualized CO_2 emissions from carbon stock change. For instance, in palm oil biodiesel chain different LUC emissions were obtained when specific aboveground living biomass (B_{AGB}) values for Colombian natural regions and vegetation indexes were adopted, compared with the default values from IPCC (2006) and from RED (EC, 2009). Also, when LUC emission calculations were made at a state level based on the Brazilian land-use statistics (biodiesel chain C) the results vary widely among the states. However, information on the carbon stocks at site-specific level is scarce and difficult to obtain.

LAND-USE PRACTICES AND NITROGEN FIELD EMISSIONS

The environmental impacts of palm cultivation in Colombia vary for the four fertilization schemes (ammonium sulphate, calcium ammonium nitrate, urea and poultry manure). The lowest GHG intensity was calculated when poultry manure was used as a fertilizer and the highest for calcium ammonium

nitrate. However, the opposite results were calculated for terrestrial acidification: the lowest result was obtained for calcium ammonium nitrate (adopting nitrogen site-specific models) and if poultry manure was used as a fertilizer the results was at least 50% higher than in the remaining fertilization schemes.

Regarding the soybean cultivation in Brazil and Argentina (chain B), it was found that the impacts also varied among the alternative systems (no-tillage, reduced-tillage and tillage). However, the results were similar for the various impact categories: soybean had the highest environmental impacts when tillage system was adopted in both countries. Comparing the LCIA results calculated on the basis of the different inventories of soybean cultivation in four states in Brazil (Mato Grosso, Goiás, Paraná and Rio Grande do Sul), it can be seen that the state in which soybean had the lowest impacts is not straightforward. For instance, soybean cultivated in Paraná had the lowest GHG intensity, acidification, eutrophication, photochemical oxidant formation and ozone depletion impacts but the highest toxicity impacts.

GHG intensity and marine eutrophication impacts of palm (13-17 g CO_2eq MJ^{-1} and 0.3-0.4 g Neq MJ^{-1}) and soybean cultivation (7-17 g CO_2eq MJ^{-1} and 0.2-0.4 g Neq MJ^{-1}) are similar. Terrestrial acidification impact related to palm cultivation varies from 0.09 to 0.54 g SO_2eq MJ^{-1}, while for soybean the results are lower (0.03-0.11 g SO_2eq MJ^{-1}). In contrast, freshwater eutrophication impact of soybean cultivation (12-54 mg Peq MJ^{-1}) is higher than those calculated for palm cultivation (less than 9 mg Peq MJ^{-1}). Regarding the ozone depletion and photochemical oxidant formation, the impacts are also different for palm (less than $0.7x10^{-6}$ g CFC-11eq MJ^{-1} and 0.05 g NMVOCeq MJ^{-1}) and soybean cultivation ($0.6\text{-}2.0x10^{-6}$ g CFC-11eq MJ^{-1} and 0.03-1.00 g NMVOCeq MJ^{-1}).

It was shown that fertilizers application emissions (nitrogen and phosphorus field emissions) make an important contribution (more than 48%) to the GHG intensity, terrestrial acidification and (marine and freshwater) eutrophication impacts of palm cultivation. GHG intensity and eutrophication impacts of soybean cultivation are also mainly caused by fertilizers application emissions. This concurs with other studies, showing that field N_2O emissions play a major role in the GHG intensity of palm and soybean cultivation (Achten et al., 2010; Choo et al., 2011; Harsono et al., 2012; de Souza et al., 2010; Castanheira and Freire, 2013; Knudsen et al., 2010). Terrestrial acidification impact of soybean cultivation is mostly caused by the emissions from fertilizers production and fossil fuel consumed in agricultural operations, which are also the processes that contribute most to ozone depletion and photochemical oxidant formation impacts of palm and soybean cultivation.

Nitrogen field emissions (NH_3, NO_3^- and N_2O) from palm and soybean cultivation were calculated based on site-specific models and IPCC tier 1 methodology (IPCC, 2006). Comparing the two approaches adopted for ammonia (NH_3) emission calculations contradictory results were obtained for terrestrial acidification impact. Regarding the palm cultivation, when site-specific model approach was adopted, higher acidification impact was calculated for the #U and #Poultry fertilization schemes, while #AS and

#CAN schemes had the highest acidification impact with IPCC approach. It was also demonstrated that the palm fertilization scheme with the lowest acidification impact depends on the approach adopted: when IPPC approach was adopted the lowest impact was obtained for #AS fertilization scheme, whereas when site-specific model was adopted #CAN had the lowest impact. Likewise, when IPCC approach was adopted the acidification impact of soybean cultivated in Mato Grosso and Goiás are higher than the results obtained with the site-specific-models.

Marine eutrophication impact also varies according to the nitrate (NO_3^-) emission calculation approaches. When IPPC approach was adopted, marine eutrophication impact of palm cultivation is approximately 50% higher than when site-specific model approach was applied. By contrast, when a site-specific model approach was adopted, marine eutrophication impact of soybean cultivation is about 20-100% higher than the results obtained with IPCC approach. Also, the state in which soybean cultivation had the lowest marine eutrophication impact depends on the NO_3^- emission calculation approach: when site-specific model was adopted the lowest impact was calculated for soybean from Mato Grosso and Goiás, while soybean from Paraná and Rio Grande do Sul had the lowest impact with IPCC approach.

The approach adopted for nitrous oxide (N_2O) emission calculations does not have a significant influence on the GHG intensity of palm and soybean cultivation (chain A and C). This result may be explained by the fact that only indirect N_2O emissions were calculated differently in the two approaches and because indirect N_2O emissions contribute less than 35% for the total N_2O emissions from palm and soybean cultivation calculated in both approaches. However, a sensitivity analysis performed for field N_2O emissions was also presented and showed a significant variation in the GHG intensity results. When minimum parameters and emission factors from IPCC (2006) were adopted, GHG intensity of cultivation was reduced by 37-51% in the case of palm and 17-42% in soybean (chain B). If the maximum parameters and emission factors were adopted, the GHG intensity increased by 173-255% for palm and 70-173% for soybean.

Regarding the influence of time horizon on the GHG intensity of palm cultivation, the results are the highest when time horizons of 20 and 100 years (GWP20 and GWP100) were considered and a reduction of 30-36% on the GHG intensity of palm cultivation was obtained when a time horizon of 500 years (GWP500) was adopted, comparing to GWP20 results.

PRODUCTION SCHEMES

The environmental impacts of palm oil extraction were analyzed, considering two scenarios for palm oil mill effluent (POME) treatment: biogas captured and flared and biogas released into the atmosphere. A huge variation on the environmental impacts was found, except for ozone depletion and freshwater eutrophication impacts, which are not influence by the emissions from POME treatment. Biogas capture

(and flaring) system reduces GHG intensity, terrestrial acidification, marine eutrophication and photochemical oxidant formation impacts of palm oil extraction by 88%, 49%, 35% and 24%, respectively. POME treatment is the process that contributes most to GHG intensity of palm oil extraction (and to acidification if the biogas was released), whereas energy processes (electricity from grid and steam and electricity from cogeneration plant) are that contributes most to ozone depletion, photochemical oxidant formation, freshwater and marine eutrophication (and to acidification if the biogas was flared). The GHG intensity of palm oil extraction vary from 1 g CO_2eq MJ^{-1} (biogas captured and flared, GWP500) to 58 g CO_2eq MJ^{-1} (biogas released, GWP20) for the different time horizons and POME treatment scenarios. Similar GHG intensity of palm oil was obtained by Kaewmai et al. (2012).

Turning now to the environmental impacts of soybean oil extraction, it was found that energy processes (electricity and heat) are that contribute most to all impact categories, except for photochemical oxidant formation impact (which is mainly caused by hexane emissions). Comparing soybean oil extraction in Brazil and Portugal, it was found that GHG intensity, ozone depletion, acidification, freshwater eutrophication, photochemical oxidant formation and marine ecotoxicity impacts are higher in Portugal than in Brazil. On the opposite, marine eutrophication, human toxicity, terrestrial and freshwater ecotoxicity impacts are higher for oil extraction in Brazil. **Table 4.4** presents the LC environmental impacts of mechanical palm oil extraction and chemical soybean oil extraction (ReCiPe method, energy allocation).

Table 4.4. LC environmental impacts of mechanical palm oil extraction and chemical soybean oil extraction (ReCiPe method, energy allocation).

	Mechanical palm oil extraction		Chemical soybean oil extraction	
	Biogas flared	Biogas released	Brazil	Portugal
GHG intensity (g CO_2eq kg^{-1}oil)	89	737	22	114
Terrestrial acidification (g SO_2eq kg^{-1} oil)	0.5	0.9	0.1	0.2
Freshwater eutrophication (mg Peq kg^{-1} oil)	0.7	0.7	9	14
Marine eutrophication (mg Neq kg^{-1} oil)	31	47	10	9
Ozone depletion (10^{-6} g CFC-11eq kg^{-1} oil)	1.0	1.0	2	14
Photochemical oxidant formation (g NMVOCeq kg^{-1} oil)	0.8	1.1	1.8	2.5

In this thesis the environmental impacts of biodiesel production (transesterification process) were calculated on the basis of specific data collected in two mills in Brazil and five in Portugal. It was also considered that the process is similar palm and soybean oil. GHG intensity and ozone depletion impacts of biodiesel production in Portugal are higher than in Brazil; however, for the remaining environmental categories, biodiesel production in Portugal presents lower impacts comparing with Brazil. These results

were obtained due to the more efficient use of chemicals and the use of natural gas to produce heat in Portuguese mills.

Transportation emissions are that contribute most to GHG intensity (no LUC), ozone depletion, acidification and photochemical oxidation impacts of soybean biodiesel chains. For instance, the sensitivity analysis of the influence of locations of soybean plantations in Brazil and Argentina showed that the GHG intensity of soybean transportation can vary from 6 to 39 g CO_2eq MJ^{-1}. These findings reinforce the importance of transportation on the environmental impacts of soybean biodiesel chains.

LCIA METHOD

Both ReCiPe and CML method were adopted to compare the environmental impacts of soybean and palm biodiesel, to determine the LC phases that contribute most to the impacts and the extent to which the impacts are influenced by the method apply. The normalized environmental impacts calculated for palm and soybean biodiesel chains with both methods were also presented. In both ReCiPe and CML methods the highest normalized result was calculated for eutrophication impact categories, while ozone depletion and photochemical oxidation impacts are substantially lower than the remaining categories. The characterized environmental impacts are discussed in the following paragraphs.

The GHG intensity and terrestrial acidification of palm biodiesel (calculated using both LCIA methods) vary significantly amongst the different biogas management options and palm fertilization schemes, respectively. Nonetheless, the extent of impacts variation also depends on the LCIA method adopted: CML eutrophication and photochemical oxidation impacts vary widely among palm fertilization schemes and biogas management options, respectively, whereas no significant variation occurs when ReCiPe was adopted. The environmental impacts of soybean biodiesel vary more markedly among the alternative cultivation systems and soybean origins (Brazil and Argentina or different Brazilian states) comparing with palm biodiesel scenarios. The extent of variation of soybean biodiesel impacts also depends on the LCIA method adopted: when ReCiPe was adopted, photochemical oxidation impact varies widely among Brazil and Argentina, whereas no significant variation occurs when CML was adopted.

Regardless of the LCIA method, the lowest GHG intensity and ozone depletion impacts of palm biodiesel were calculated for the use of ammonium sulphate and poultry manure as fertilizers, whereas the lowest acidification impact was calculated for the use of calcium ammonium nitrate. Likewise, the use of calcium ammonium nitrate as fertilizer allows lower marine eutrophication (ReCiPe) and eutrophication (CML) impacts of palm biodiesel, while the lowest freshwater eutrophication impact of palm biodiesel was obtained when poultry manure was applied as fertilizer. The lowest photochemical oxidation impact of palm biodiesel was obtained if urea or poultry manure were used as fertilizers. Biodiesel produced with palm oil extracted in a mill in which biogas is captured and flared, instead of

released into the atmosphere, had lowest GHG intensity and photochemical oxidation impact. The biogas management option does not have a significant influence on the remaining environmental impacts of palm biodiesel since oil extraction represents less than 12% of the total impact in each category.

Regarding the biodiesel produced with soybean imported from Brazil and Argentina, the same conclusions can be drawn from the results obtained with the two LCIA methods: the lowest GHG intensity, freshwater and marine eutrophication impacts were obtained when no-tillage system was adopted in both countries, while the lowest impacts in the remaining environmental categories were obtained for no-tillage system in Brazil and reduced-tillage in Argentina.

The contribution of each LC phase of palm and soybean biodiesel for the different environmental impacts is not significantly affected by the LCIA method adopted, except for photochemical oxidation category. Photochemical oxidation impact of soybean biodiesel was mainly caused by the emissions from transportation when ReCiPe method was adopted whereas extraction emissions were that contribute most to this impact when CML method was applied. The process contribution to photochemical oxidation impact of palm biodiesel also depends on the LCIA method, as well as on the biogas management option: when CML method was adopted, the transportation emissions were the main contribution if biogas was flared and the extraction emissions were the main contribution if biogas was released. When ReCiPe method was adopted, transportation made the main contribution to photochemical oxidation impact of palm biodiesel, regardless of the biogas management option.

The human toxicity and aquatic ecotoxicity impacts of soybean biodiesel chain vary depending on the application of ReCiPe and USEtox (“recommended” and “recommended+interim”) methods. However, due to the highest uncertainty associated with the calculation of emissions that contribute most to these impacts (e.g., heavy metals, pesticides), the generalizability of the results obtained is limited. Cultivation is the LC phase that contributes most to the human toxicity impact when both methods were adopted. Using ReCiPe and USEtox (“recommended+interim”) methods, a slight variation on human toxicity impact of soybean biodiesel was obtained among the four Brazilian states where soybean was cultivated and the alternative pathways. In contrast, with USEtox (“recommended”) method the human toxicity impact vary significantly. Cultivation is also the process that contribute most to the aquatic ecotoxicity impact of soybean biodiesel, except for the case in which soybean was produced in Mato Grosso and USEtox (“recommended+interim”) method were adopted. When ReCiPe and USEtox (“recommended”) methods were adopted the ecotoxicity impact of soybean biodiesel was the lowest when soybean was produced in Rio Grande do Sul and Mato Grosso, respectively.

MULTIFUNCTIONALITY

The results show that the effect of multifunctionality on the results is more significant for soybean than palm biodiesel. This can be explained by the relatively high mass share of the palm oil compared with the palm kernel meal. For the environmental impacts analyzed in this research, the results obtained with energy and price allocation are similar (also demonstrated by Huo et al., 2009) and higher than those obtained with mass allocation. A slight variation on results of both chains was observed amongst the three approaches adopted for allocation based on price of products (average prices, prices for maximum and minimum ratio oil/meal).

The variation on acidification, eutrophication and photochemical oxidant formation impacts of palm biodiesel among multifunctionality approaches is 31-50%, 32-34% and 18-21%, respectively. Those environmental impacts are the lowest when mass allocation was adopted, while the highest results were obtained in the substitution scenario (A), in which the lowest impacts of soybean meal (the avoided product) were used. The lowest GHG intensity of palm biodiesel was obtained in the substitution scenario (B), in which the highest GHG intensity of soybean meal was adopted. However, the highest GHG intensity depends on the fertilization schemes and biogas management options. When biogas was released the highest result was calculated using substitution A (#CAN, #U and #Poultry) and price (max) allocation (#AS), whereas mass (#AS, #U and #Poultry) and price (#CAN) allocation results are the highest when biogas was captured and flared.

The environmental impacts of biodiesel produced from Brazilian soybean are greatly influenced by the state in which soybean was cultivated, the pathway adopted and the multifunctionality approach. The highest impacts were obtained in the substitution scenario (A) in which palm oil from Colombia (the avoided product) had the lowest impacts. The lowest GHG intensity, freshwater eutrophication and photochemical oxidant formation impacts were calculated using mass allocation, while the lowest terrestrial acidification impact was obtained with the substitution scenario (B) in which acidification of palm oil from Colombia (the avoided product) was the highest. With regard to soybean origin, the highest and the lowest impacts are the same for all multifunctionality approaches. The lowest and highest environmental impacts of alternative pathways depend on the multifunctionality approach: when mass allocation was adopted soybean biodiesel produced in Brazil and exported to Portugal had the highest impacts (pathway C1), whereas biodiesel produced in Portugal based on soybean imported from Brazil had the highest impacts when substitution was adopted (pathway C3). Also, different impacts were obtained when energy allocation approach was adopted: the lowest impacts were obtained for biodiesel produced in Portugal based on soybean oil imported from Brazil (C2), whereas the highest impacts were obtained for biodiesel produced in Portugal based on soybean imported from Brazil (C3).

The energy allocation method is problematic when the co-products have distinctly different uses, such as when one co-product is used for energy purposes (palm or soybean oil) and another co-product (palm or soybean meal) is used for human food or animal feed. In fact, even though human food and animal feeds have energy content, their function/use is related to nutritional properties and not to the product energy content. The energy allocation method seems to be appropriate where all the co-products or, at least, the main co-product is used for energy purposes, as is the case of palm oil. When there is production of co-products in which the main product will not be used for energy purposes, as is the case of soybean meal, the substitution method and/or an allocation approach that reflects the function of the main product should be employed. In the case of soybean, the market is mostly driven by the production of soybean meal for the livestock feed industry and the adoption of the energy allocation is, thus, not seem appropriate.

GREENHOUSE GAS SAVINGS

Considering the LUC emissions related to the expansion of the Colombian palm area (from 1990 to 2010), the GHG savings from replacing diesel with palm biodiesel vary from 68 to 93%, complying the GHG saving criteria stipulated by RED (35%, 50% and 60%). Soybean biodiesel (considering the expansion of the soybean area in each Brazilian state from 1985 to 2006) achieves the RED GHG saving criteria of 35% and 50% when soybean is produced in Rio Grande do Sul (no LUC occur). These results contradict the default GHG savings defined in the RED: GHG savings from replacing diesel with palm biodiesel (process not specified) and soybean biodiesel given by the RED do not meet the minimum GHG saving criteria of 35% for biodiesel.

The GHG savings results also indicate that they depend greatly on the multifunctionality approach. The RED adopted the energy allocation approach because of concerns with uncertainties associated with the substitution method. Furthermore, the RED probably assumed that the various biofuel co-products are used mainly for energy purposes. However, when the co-products are not used for energy purposes (e.g. soybean meal) energy-based allocation should not be the preferred method. This is particularly important if significantly different GHG emissions are obtained with the substitution method and/or with other allocation approaches, since the RED has assumed that the energy allocation approach has "*results that are generally comparable with those produced by the substitution method".* However, the findings of this research do not support this statement, in particular for soybean biodiesel.

5 CONCLUSIONS AND RECOMMENDATIONS

5.1 MAJOR CONTRIBUTIONS AND KEY RESEARCH FINDINGS

This thesis presents an environmental sustainability assessment of selected biodiesel produced from soybean and palm cultivated in South America. A framework was developed and implemented for various biodiesel chains, with the aim of contributing to the life-cycle (LC) modeling of multifunctional bioenergy systems. Critical modeling issues were addressed and assessed using alternative scenarios, such as: land-use change (LUC), different land-use practices, production options and pathways, as well as the nitrogen field emission calculations and the life-cycle impact assessment (LCIA) method adopted. To this end, six research questions were formulated in **Chapter 1**. The main responses and related findings are discussed below.

1. How can we account for the effects associated with direct LUC in the LCA of biodiesel?

The carbon stock changes and N_2O emissions (as a result of nitrogen released by the mineralization of soil organic matter) due to direct LUC were calculated in **Chapter 3**. Firstly, LUC due to the expansion of the palm area in Colombia and the soybean area in Argentina and Brazil were modeled using different scenarios established on the basis of a combination of alternative previous land-uses, different fertilization schemes and cultivation systems. Secondly, LUC was modeled on the basis of actual palm expansion in Colombia from 1990 to 2010 and soybean expansion in four Brazilian states (Mato Grosso, Goiás, Paraná and Rio Grande do Sul) from 1985 to 2006.

The influence of LUC in the GHG intensity of soybean and palm biodiesel was presented and discussed in **sub-chapter 4.2**. The results demonstrated the importance of LUC in the GHG intensity of biodiesel based on palm or soybean, but a significant intensity range was calculated for the alternative LUC scenarios assessed: the highest results were calculated for the scenarios in which tropical forest was converted into palm or soybean plantations, whereas the lowest levels were for the conversion of annual cropland and degraded savanna (in palm plantations) and severely and moderately degraded grassland/savanna (in soybean plantations). In addition, different LUC emissions were obtained when site-specific data was used in the calculation instead of generalized data from the literature. These

results showed the importance of promoting bioenergy crop cultivation in previous degraded land in order to minimize GHG emissions. The relevance of use site-specific data in the calculations was also demonstrated.

The GHG intensity of palm biodiesel was significantly lower (11-107%) than soybean biodiesel when the actual expansion of the palm and soybean area was adopted (in Colombia from 1985 to 2006 and in Brazil from 1985 to 2006). This difference is mainly due to the fact that palm is a perennial crop (with a high carbon stock in the vegetation) while soybean is an annual crop (with a low carbon stock). N_2O emissions make a slight contribution to the total LUC emissions (less than 1% for palm and less than 10% for soybean).

2. ***What are the land-use practices, production schemes and pathways that lead to lower impacts?***

The LC environmental impacts (using energy allocation and the ReCiPe method) are presented in **sub-chapter 4.3**) for:

i) four oil palm fertilization schemes (use of ammonium sulphate, calcium ammonium nitrate, urea and poultry manure) and two biogas management options (captured and flared or released into the atmosphere) in a specific plantation and oil extraction mill in Colombia;

ii) three soybean cultivation systems (tillage, reduced-tillage and no-tillage) in Brazil and Argentina;

iii) alternative soybean transportation systems (from plantations to ports in Brazil and Argentina and from ports to Portugal);

iv) soybean production in four states in Brazil (Mato Grosso, Goiás, Paraná and Rio Grande do Sul) and three alternative pathways (soybean biodiesel totally produced in Brazil and exported to Portugal, produced in Portugal using soybean oil imported from Brazil and produced in Portugal using soybean imported from Brazil).

The environmental impacts of biodiesel chains vary according to the alternatives analyzed. The selection of the alternative with the lowest impacts is complex and depends on the environmental impact category considered. However, some findings can be highlighted:

- The environmental impacts of palm and soybean biodiesel are greatly influenced by land-use practices and the location (country or state) where cultivation takes place.
- The lowest GHG intensity and freshwater eutrophication impacts of palm cultivation were obtained when poultry manure was used as a fertilizer. The lowest acidification, photochemical oxidant formation and ozone depletion impacts were obtained when calcium ammonium

nitrate, urea and ammonium sulphate were used as fertilizers, respectively. The lowest environmental impacts of soybean cultivation in Brazil and Argentina were obtained when no- and reduced-tillage systems were adopted.

- The GHG intensity and marine eutrophication impacts are similar for palm and soybean cultivation. In contrast, freshwater eutrophication, ozone depletion and photochemical oxidant formation impacts are very different for palm and soybean cultivation. Fertilizer application emissions, fertilizer production and the fossil fuel consumed in agricultural operations have an important contribution to the environmental impacts of palm and soybean cultivation.
- Palm oil mill effluent treatment (including biogas management) is the process that contributes most to GHG intensity of palm oil extraction, whereas the energy (electricity and heat) consumed contributes most to ozone depletion, photochemical oxidant formation, freshwater and marine eutrophication impacts. The emissions from the energy consumed in soybean oil extraction are also the emissions that contribute most to environmental impacts, except for photochemical oxidant formation, which is mainly caused by hexane emissions.
- Soybean oil extraction in Brazil has lower environmental impacts than in Portugal in almost all impact categories, due to the use of wood as a fuel and lower energy consumption. Biodiesel production in Portugal has the lowest impacts in almost all the categories (except GHG intensity and ozone depletion) due to having the lowest chemical consumption and shortest distances for inputs transportation.
- The lowest GHG intensity was obtained for biodiesel produced in Portugal based on soybean oil imported from Brazil, whereas the highest intensity was calculated for biodiesel produced with imported soybean from Brazil. Regardless of the pathway, the lowest GHG intensity was calculated for biodiesel produced with soybean from Paraná and Rio Grande do Sul and the highest for soybean cultivated in Mato Grosso and Goiás.
- The transportation emissions are that contribute most to GHG intensity (no LUC), ozone depletion, acidification and photochemical oxidation impacts of soybean biodiesel chains. For instance, the sensitivity analysis for the influence of the location of soybean plantations in Brazil and Argentina showed that the GHG intensity of soybean transportation can vary from 6 to 39 g CO_2eq MJ^{-1}. These findings reinforce the significance of transportation in relation to the environmental impacts of soybean biodiesel chains.

3. Are the environmental impacts of biodiesel influenced by the emission calculation approach and LCIA method adopted?

The third research question was addressed by comparing the environmental impacts of palm and soybean biodiesel calculated using two LCIA methods: ReCiPe and CML. The toxicity impacts of soybean biodiesel were also calculated based on the LCIA ReCiPe and USEtox methods. The LC phases that contribute most to the results and the extent to which the results are influenced by the method applied were determined. The results were presented in **sub-chapter 4.4**. The main findings are:

- The variation in the environmental impacts of palm and soybean biodiesel according to the different production schemes depends on the LCIA method adopted. For instance, regarding the photochemical oxidation impact of palm biodiesel, a small variation among biogas management options was obtained using ReCiPe method (less than 5%), whereas with the CML method the results varied significantly (more than 85%).
- The contribution made by the various palm and soybean biodiesel LC phases to the different environmental impacts is not significantly affected by the LCIA method adopted, except for the photochemical oxidation impact category.
- The normalized LCIA results calculated for palm and soybean biodiesel chains using the ReCiPe and CML methods show that the highest value was calculated for eutrophication, whereas the ozone depletion and photochemical oxidation impacts are substantially lower than the remaining categories.
- The ReCiPe and USEtox ("recommended" and "recommended+interim") methods considered different substances in the impact calculations and the characterization models greatly depend on the estimation of emissions for the different environmental compartments (air, water and soil), which is inherently problematic and was not assessed in this thesis. Due to the high level of uncertainty associated with the calculation of emissions that contribute to toxicity impacts (e.g., heavy metals, pesticides), the generalizability of the results obtained is limited.
- The results show that field N_2O emissions play a major role in the GHG intensity of palm and soybean cultivation, although the GHG intensity is very sensitive to the parameters and emission factors adopted for field N_2O emission calculations. The high level of uncertainty in field N_2O emission calculations is more evident for palm than soybean cultivation because more N-fertilizer is applied.
- The time horizon (20, 100 and 500 years) has a great influence on the GHG intensity of palm cultivation and palm oil extraction. The highest results were obtained when time horizons of 20 and 100 years were considered in the calculations, due to the highest GWP of CH_4 and N_2O.

- Different nitrogen (nitrous oxide, ammonia and nitrate) field emission calculation approaches (IPCC approach and site-specific models) were employed, demonstrating the importance of calculating nitrogen field emissions based on site-specific models and cultivation data. Terrestrial acidification and marine eutrophication impacts were greatly influenced by ammonia and nitrate field emission calculation approaches and contradictory results were obtained for the various cultivation scenarios.

4. How does the selected multifunctionality approach influence biodiesel environmental impacts?

The fourth research question was addressed in **sub-chapters 3.10** and **4.4**. A sensitivity analysis for alternative multifunctionality approaches was performed to illustrate the consequences, in terms of results, of using different multifunctionality approaches. A number of general conclusions can be drawn as a result of this research:

- The effect of multifunctionality on the results is considerably more significant for soybean biodiesel than palm biodiesel and varies among the various impact categories, land-use practices, pathways or production options.
- The impacts calculated with energy and price-based allocation are similar and higher than those obtained with mass allocation, both for palm and soybean biodiesel.
- The lowest acidification, eutrophication and photochemical oxidant formation impacts of palm biodiesel were calculated using mass allocation, while the highest were obtained using a substitution scenario involving the lowest impacts of the avoided product (soybean meal). With regard to GHG intensity, the lowest result was obtained for the substitution scenario in which the GHG intensity of soybean meal was the highest. However, the highest GHG intensity depends on the fertilization schemes and biogas management options.
- The highest impacts of soybean biodiesel were obtained for the substitution scenario in which palm oil from Colombia (the avoided product) had the lowest impacts. The lowest GHG intensity, freshwater eutrophication and photochemical oxidant formation impacts were calculated using mass allocation, while the lowest terrestrial acidification impact was obtained with the substitution scenario in which acidification of palm oil from Colombia (the avoided product) was the highest. With regard to soybean origin, the categories for which the highest and lowest results were calculated were the same for the various multifunctionality approaches, whereas the lowest and highest environmental impacts calculated for the alternative pathways depended on the multifunctionality approach.

- Soybean biodiesel produced in Brazil and exported to Portugal has the highest impacts mass-based allocation, whereas biodiesel produced in Portugal based on soybean imported from Brazil has the highest impacts when substitution is adopted.

5. What are the GHG emission savings when palm and soybean biodiesel replace diesel?

The GHG emission savings when palm and soybean biodiesel replace fossil diesel were assessed and compared with the GHG saving criteria stipulated by RED (35% until the end of 2006, 50% from 2017 onwards and 60% from 2018 onwards). The GHG savings from palm biodiesel vary from 68% to 93% among the alternative fertilization schemes and biogas management options. Soybean biodiesel results in savings of 35% and 50% for soybean produced in Rio Grande do Sul (no LUC). These results contradict the default GHG savings defined in RED: the GHG savings from palm biodiesel (process not specified) and soybean biodiesel given by RED do not meet the minimum GHG saving of 35%.

6. How can the environmental sustainability of biodiesel be improved by applying the LCA methodology?

The sixth research question is partially addressed in the next two **sub-chapters (5.2** and **5.3)**.

5.2 SUGGESTIONS TO IMPROVE THE ENVIRONMENTAL SUSTAINABILITY OF SOYBEAN AND PALM BIODIESEL

The main findings regarding the environmental impacts of palm and soybean cultivation show that more effort should be made to promote the use of more efficient fertilizers and the adoption of best soil management practices, which could maximize fertilizer uptake by crops and reduce the quantity of fertilizer used and/or nutrient losses. In order to reduce the amount of nitrogen and phosphorus fertilizers lost due to leaching and volatilization, developing seeds that require significantly less nitrogen fertilizer could be a key factor in reducing the environmental impacts of bioenergy crops. In addition, more research is needed to develop site-specific models (at national level, at least) that can support nitrogen field emission calculations. Furthermore, significant reductions in the environmental impacts of biodiesel can be achieved if transportation routes are optimized, in particular for soybean transportation in Brazil.

The biogas capture system will reduce the environmental impacts of palm biodiesel. However, more efficient biogas management, namely recovery for energy generation instead of flaring, should be

implemented in order to reduce the impacts further. Significant reductions can also be achieved by optimizing the use of residues and the co-generation system in the palm oil extraction mill.

With regard to biodiesel production, it is important to identify opportunities for using bioethanol in the transesterification reaction. The technical and economic success of replacing methanol (fossil in origin) with bioethanol to produce biodiesel (resulting in fatty acid ethyl ester instead of fatty acid methyl ester) is expected to present a challenge for the environmental sustainability of biodiesel, particularly for Brazil (Castanheira et al., 2014a). Chemically, the methyl and ethyl routes are very similar but in practice there are differences between these two routes in terms of reaction time, catalyst amount and reaction temperature. According to Hamelinck et al. (2007), difficulties in the separation phase are a major obstacle in ethyl ester production for any feedstock used. Beyond the technical aspects, the main limitations on the implementation of ethyl transesterification are the price and availability of bioethanol. Brazil has the cheapest bioethanol in the world but this depends greatly on geographical location and fluctuations over time (Castanheira et al., 2014a).

5.3 POLICY RECOMMENDATIONS

The impacts of the biodiesel production chain can be mitigated by appropriate policies aimed at the integrated optimization of food and bioenergy production. Conflicts between food and biodiesel can be avoided through agro-economic-ecological zoning for soybean and oil palm, allowing for the appropriate use of land for each purpose. The implementation of policies and regulations to avoid the cultivation of palm and soybean in high carbon stock land (e.g. primary forest) are also crucial to guaranteeing palm and soybean biodiesel production with low GHG emissions. Despite the limitations and weaknesses of some sustainability tools and initiatives, policies, legislation, standards and certification schemes, they can play an important role in the sustainability assessment and development of the emerging biodiesel production. However, it is a fact that in many cases the legislation is not properly implemented, indicating that stronger enforcement is required (Castanheira et al., 2014a).

In addition, the sustainability criteria for biofuels should include other environmental impact categories, as demonstrated by the impacts calculated in this thesis. It was found that in some cases a biodiesel chain has a lower GHG intensity than others with significantly higher impacts in other environmental categories. For example, biodiesel produced in Portugal based on soybean imported from Rio Grande do Sul has a substantially higher GHG intensity (37 g CO_2eq MJ^{-1}) than palm biodiesel (7 g CO_2eq MJ^{-1}, biogas flared and using urea as fertilizer). However, in the same scenarios, palm biodiesel has a higher terrestrial acidification impact (0.45 g SO_2eq MJ^{-1}) than soybean biodiesel (0.23 g SO_2eq MJ^{-1}).

Last but not least, the harmonization of multifunctionality approaches in the various policies, legislation, standards and certification schemes is crucial to producing consistent LCIA results. The energy allocation method may be appropriate if all the co-products or, at least, the main co-product are used for energy purposes, as is the case with palm oil. However, when important co-products are not used for energy purposes, as in the case of soybean meal, the substitution method or an allocation approach reflecting the function of the co-product should be employed. Although avoiding allocation by system expansion is an attractive way of dealing with the multifunctionality problem, its main drawbacks are that the system becomes more complicated because of the need for additional data on other subsystems to be included in the system boundary.

5.4 LIMITATIONS AND FUTURE RESEARCH

In this thesis the indirect land-use change (iLUC) effects on the environmental impacts of soybean and palm biodiesel were not considered. This is one important limitation of the thesis. However, many authors have been arguing that iLUC modeling approaches have limitations and result in an unacceptable range of uncertainty (Broch et al., 2013; Fritsche et al., 2010; Mathews and Tan, 2009). In addition, the attributional LCA approach has limited or no capacity to model iLUC, which requires a consequential approach in which system boundaries are expanded to determine how changes in supply and demand affect the markets in question. This requires linking economic models that simulate market behavior to predict LUC with carbon stock and emission factor databases to determine the net GHG emissions (Broch et al., 2013). Regardless of the debate, regulators are developing biofuel policies that include the effects of iLUC together with traditional LCA approaches. The European Commission is working to understand the issues in order to include iLUC in the Renewable Energy Directive (EC, 2012).

The assessment of land-use impacts should also include the impacts on land quality, as described by Milà i Canals et al. (2006): biodiversity, biotic production potential (e.g., soil fertility), ecological soil quality (filter and buffer capacity, water carbon and nutrients cycling). Land provides ecological functions and these functions should therefore be included in the LCA, although detailed bio-geographical parameters based on spatial differentiation will be required.

The implications of the uncertainty regarding the LCA of soybean and palm biodiesel were addressed, based on the scenario modeling approach. However, parameter uncertainty should also be analyzed and the results compared with those obtained in this thesis. Incorporating parameter uncertainty into LCA is crucial: Malça and Freire (2010), for example, concluded that in certain cases parameter uncertainty is more significant in the calculation of GHG emissions than scenario uncertainty.

There are a number of related topics that could benefit from further study on the basis of the results obtained from this research:

- *Land-use change*: the quantification of CO_2 emissions related to LUC is a great challenge. LUC may lead to slash and burn practices, whose impacts may be still not fully addressed and quantified. Moreover, perennial crops such as oil palm are commonly associated with other crops in agroforestry systems and assessing the land-use impacts in these cases may require specific research and methodological developments. Since information on carbon stocks at site-specific level is scarce and difficult to obtain, high-resolution mapping using geographic information systems (GIS) is required to provide consistent data sets for land-use and carbon stocks (in particular aboveground carbon) and to obtain the most robust results.
- *Incorporating parameter uncertainty*: parameter uncertainty must be addressed using various approaches (e.g. Monte-Carlo simulation) and the results should be compared with the scenario uncertainty results obtained in this thesis. Several statistical measures have to be calculated to quantify the uncertainty in the results, including the coefficient of variation (which measures the dispersion of data around the mean) and the 5th, 25th, 75th, and 95th percentiles. The probability distributions also have to be selected to fit the input parameters of the study.
- *The application of Multi-Criteria Decision Analysis*: LCA provides a basis for assessing and identifying the options for potential improvements to the environmental performance of palm and soybean biodiesel. In order to support decision-making on sustainability, there is a need for research to combine the LCA framework and Multi-Criteria Decision Analysis (MCDA). In general, published work combining MCDA and LCA yields a ranking of several alternatives by summarizing the impacts of each alternative into a single numerical value. A much less developed idea is the use of standards-based MCDA sorting methods such as ELECTRE TRI (Dias et al., 2002). The original ELECTRE TRI has been used with LCA for screening technologies (Basson and Petrie, 2007) and can be used for screening biodiesel chains.
- *Field emissions and related impacts*: LCA is a valuable tool for the environmental assessment of biodiesel, especially when used in combination with site-specific indicators. However, there still is a need to develop the methodology to obtain a more robust quantification of nitrogen, phosphorus, heavy metals and pesticides emissions.
- *Social and economic impacts*: This thesis provides a partial contribution towards improving insights into the sustainability potential of palm and soybean biodiesel systems, since the social and economic impacts were not considered. Hence, it is important that the environmental LCA results presented here are supplemented with results obtained by using other tools, such as Social Life-Cycle Assessment and Life-Cycle Costing.

- *Methanol versus Ethanol*: despite the difference between using a renewable (ethanol) and fossil (methanol) source, it is important to compare the environmental impacts of the two routes. Biodiesel production using the methyl and ethyl routes should be compared, based on the life-cycle assessment approach.
- *Biodiesel consumption:* the environmental impacts of biodiesel consumption on transport or thermal/electric energy production should also be addressed.
- *Water footprint (WF):* the production of energy crops for biofuel production can have substantial impacts on water demand and quality. The WF of soybean and palm biodiesel should be calculated, considering the volume of freshwater used, measured in terms of water volume consumed (evaporated) or polluted in the various stages of the production chain (Hoekstra, 2012).

6 REFERENCES

Achten, W.M.J., Van den Bempt, P., Almeida, J., Mathis, E., Muys, B., 2010. Life cycle assessment of a palm oil system with simultaneous production of biodiesel and cooking oil in Cameroon. Environmental Science and Technology 44 (12), 4809-4815.

Achten, W.M.J., Verchot, L.V., 2011. Implications of Biodiesel-Induced Land-Use Changes for CO_2 Emissions: Case Studies in Tropical America, Africa, and Southeast Asia. Ecology and Society 16 (4), 14.

Althaus, H.J., Chudacoff, M., Hischier, R., Jungbluth, N., Osses, M., Primas, A., 2007. Life Cycle Inventories of Chemicals. Final report ecoinvent data v2.0. Volume: 8. Swiss Centre for LCI, Empa - TSL. Dübendorf, CH.

Anaya, J.A., Chuvieco, E., Palacios-Orueta, A., 2009. Aboveground biomass assessment in Colombia: A remote sensing approach. Forest Ecology and Management 257, 1237-1246.

ANDA, 2011. Anuário Estatístico do Setor de Fertilizantes 2010. Associação Nacional para Difusão de Adubos-ANDA. São Paulo.

Andrei, E., 2005. Compêndio de defensivos agrícolas. 7ª ed. São Paulo, 2005.

Angarita, E.Y., Lora, E.S., 2009. The energy balance in the palm oil-derived methyl ester (PME) life cycle for the cases in Brazil and Colombia. Renewable Energy 34, 2905–2913.

Asman, W.A.H., 1992. Ammonia emission in Europe: updated emission and emission variations. Rep. 228471008. Bilthoven, the Netherlands: National Inst. of Public Health and Environmental Protection.

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

Azapagic, A., 1996. Environmental System Analysis: The Application of Linear Programming to Life Cycle Assessment, Vol 1. PhD thesis, University of Surrey.

Baker, C.J., Saxton, K.E., Ritchie, W.R., Chamen, W.C.T., Reicosky, D.C., Ribeiro, M.F.S., Justice, S.E., Hobbs, P.R., 2007. No-tillage seeding in conservation agriculture. Second Edition. Edited by C.J. Baker and K.E. Saxton. Food and Agriculture Organization of the United Nations.

Barros, S., 2013. Brazil Biofuels Annual - Annual Report 2013. USDA Foreign Agricultural Service: GAIN-Global Agricultural Information Network. GAIN Report Number: BR13005.

Basset-Mens, C., van der Werf, H.M.G., Robin, P., Morvan, Th., Hassouna, M., Paillat, J.-M., Vertes, F., 2007. Methods and data for the environmental inventory of contrasting pig production systems. Journal of Cleaner Production 15, 1395-1405.

Basson, L., Petrie, J.G., 2007. A Critical Systems Approach to Decision Support for Process Engineering. Computers and Chemical Engineering 31, 876-888.

Batlle-Bayer, L., Batjes, N.H., Bindraban, P.S., 2010. Changes in organic carbon stocks upon land use conversion in the Brazilian Cerrado: A review. Agriculture, Ecosystems and Environment 137, 47–58.

Bauer, C., 2007. Holzenergie. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent data v2.0. Editors: Dones R. Volume: 6. Swiss Centre for LCI, PSI. Dübendorf and Villigen, CH.

Berndes, G., Bird, N.D., Cowie, A., 2010. Bioenergy, Land Use Change and Climate Change Mitigation. IEA Bioenergy. Available at www.ieabioenergy.com (acessed December 26, 2012).

Bessou, C., Basset-Mens, C., Tran, T., Benoist, A., 2013. LCA applied to perennial cropping systems: a review focused on the farm stage. International Journal of Life Cycle Assessment 18, 340–361.

Boone, F.R., 1988. Weather and other environmental factors influencing crop responses to tillage and traffic. Soil and Tillage Research 11, 283-324.

Brandão, M., Milà i Canals, L., Clift, R., 2011. Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. Biomass and Bioenergy 35, 2323–2336.

Brandão, M., Clift, R., Milà i Canals, L., Basson, L., 2010. A Life-Cycle Approach to Characterising Environmental and Economic Impacts of Multifunctional Land-Use Systems: An Integrated Assessment in the UK. Sustainability 2, 3747-3776.

Broch, A., Hoekman, S.K., Unnasch, S., 2013. A review of variability in indirect land use change assessment and modeling in biofuel policy. Environmental Science & Policy 29, 147–157.

Buchgeister, J., 2012. Comparison of sophisticated Life Cycle Impact Assessment Methods for Assessing Environmental Impacts in a LCA Study of Electricity Production. 25th International Conference on Efficiency, Cost, Optimization, Simulation and Environmental Impact of Energy Systems ECOS 2012. Perugia, Italy, 26-30 June 2012.

Buratti, C., Barbanera, M., Fantozzi, F., 2012. A comparison of the European renewable energy directive default emission values with actual values from operating biodiesel facilities for sunflower, rape and soya oil seeds in Italy. Biomass and Bioenergy 47, 26-36.

Castanheira, É.G., Freire, F., 2011. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja em Portugal. Relatório Final elaborado para a APPB, no âmbito do Protocolo de colaboração entre a APPB e a ADAI. Coimbra, Dezembro de 2011.

Castanheira, É.G., Acevedo, H., Freire, F., 2014b. Greenhouse gas intensity of palm oil produced in Colombia addressing alternative land use change and fertilization scenarios. Applied Energy 114, 958-967.

Castanheira, É.G., Dias, A.C., Arroja, L., Amaro, R., 2010. The environmental performance of milk production on a typical Portuguese dairy farm. Agricultural Systems 103, 498–507.

Castanheira, É.G., Freire, F., 2013. Greenhouse gas assessment of soybean: implications of land use change and different cultivation systems. Journal of Cleaner Production 54, 49-60.

Castanheira, É.G., Grisoli, R., Freire, F., Garcilasso, V., Coelho, S., 2014a. Environmental sustainability of biodiesel in Brazil. Energy Policy 65, 680–691.

Cavalett, O., Chagas, M.F., Seabra, J.E., Bonomi, A., 2012. Comparative LCA of ethanol versus gasoline in Brazil using different LCIA methods. The International Journal of Life Cycle Assessment 18(3), 647–658.

Cavalett, O., Ortega, E., 2009. Emergy, nutrients balance and economic assessment of soybean production and industrialization in Brazil. Journal of Cleaner Production 17, 762-771.

Cavalett, O., Ortega, E., 2010. Integrated environmental assessment of biodiesel production from soybean in Brazil. Journal of Cleaner Production 18, 55-70.

CENBIO, 2013. Comparação da ACV de Biodiesel produzido a partir de Óleo de Soja e Gordura Bovina via rota Metílica e Etílica. CENBIO – Centro Nacional de Referência em Biomassa. Final Report – Project BIOACV nº 558733/2010-7, financial support the CNPq. São Paulo, 2013.

Chamberlain, J.F., Miller, S.A., Frederick, J.R., 2011. Using DAYCENT to quantify on-farm GHG emissions and N dynamics of land use conversion to N-managed switchgrass in the Southern U. S. Agriculture, Ecosystems and Environment 141, 332-341.

Chase, L.D.C., Henson, I.E., Abdul-Manan, A.F.N., Agus, F., Bessou, C., Milà i Canals, L., Sharma, M., 2012. The PalmGHG Calculator: The RSPO greenhouse gas calculator for oil palm products, Beta-version. The Roundtable for Sustainable Palm Oil – RSPO. Kuala Lumpur, Malaysia, 2012.

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

Cherubini, F., Birda, N., Cowie, A., Jungmeier, G., Schlamadinger, B., Woess-Gallasch, S., 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: key issues, ranges and recommendations. Resources, Conservation and Recycling 53(8), 434–47.

Choo, Y.M., Muhamad, H., Hashim, Z., Subramaniam, V., Puah, C.W., Tan, Y.A., 2011. Determination of GHG contributions by subsystems in the oil palm supply chain using the LCA approach. The International Journal of Life Cycle Assessment 16, 669-681.

Corley, R.H.V., Tinker, P.B., 2003. The Oil Palm. World Agriculture Series. Fourth Edition. Blackwell Publishing: Oxford, UK.

Crutzen, P.J., Mosier, A.R., Smith, K.A., Winiwarter, W., 2008. N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmospheric Chemistry and Physics 8, 389–395.

Dalgaard, R., Schmidt, J., Halberg, N., Christensen, P., Thrane, M., Pengue, W. A., 2008. LCA of Soybean Meal. The International Journal of Life Cycle Assessment 3, 240-254.

De Souza, S.P., Pacca, S., de Avila, M.T., Borges, J.L.B., 2010. Greenhouse gas emissions and energy balance of palm oil biodiesel. Renewable Energy 35, 2552-2561.

Deike, S., Pallutt, B., Melander, B., Strassemeyer, J., Christen, O., 2008. Long-term productivity and environmental effects of arable farming as affected by crop rotation, soil tillage intensity and strategy of pesticide use: A case-article of two long-term field experiments in Germany and Denmark. The European Journal of Agronomy 29, 191-199.

Del Grosso, S.J., Ojima, D.S., Parton, W.J., Stehfest, E., Heistemann, M., DeAngelo, B., Rose, S., 2009. Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. Global Planetary Change 67, 44-50.

Del Grosso, S.J., Parton, W.J., Mosier, A.R., Walsh, M.K., Ojima, D.S., Thornton P.E., 2006. DAYCENT National-Scale Simulations of Nitrous Oxide Emissions from Cropped Soils in the United States. Journal of Environmental Quality 35, 1451-1460.

Derpsch, R., Friedrich, T., Kassam, A., Hongwen, L., 2010. Current status of adoption of no-till farming in the world and some of its main benefits. International Journal of Agricultural and Biological Engineering 3 (1), 1-26.

DGEG, 2012. Matéria-Prima na Produção de Biocombustíveis em Portugal: 2011 e 2012. Direcção-Geral de Energia e Geologia (DGEG) - Ministério do Ambiente, Ordenamento do Território e Energia.

DGEG, 2013. Preço máximo dos biocombustíveis. Direcção-Geral de Energia e Geologia (DGEG) - Ministério do Ambiente, Ordenamento do Território e Energia. Available at: http://www.dgge.pt/ (accessed 11 December 2013).

Dias, A.C., Arroja, L., 2012. Comparison of methodologies for estimating the carbon footprint - case study of office paper. Journal of Cleaner Production 24, 30-35.

Dias, L., Mousseau, V., Figueira, J., Clímaco, J., 2002. An Aggregation/Disaggregation Approach to Obtain Robust Conclusions with ELECTRE TRI. European Journal of Operational Research 138 (2), 332-348.

Diaz-Chavez, R.A., 2011. Assessing biofuels: Aiming for sustainable development or complying with the market? Energy Policy 39, 5763–5769.

Dreyer, L.C., Niemann, A.L., Hauschild, M.Z., 2003. Comparison of Three LCIA Methods Comparison of Three Different LCIA Methods: EDIP97, CML2001 and Eco-indicator 99. The International Journal of Life Cycle Assessment 8(4), 191-200.

Dros, J.M., 2004. Managing the soy boom: Two scenarios of soy production expansion in South America. Report commissioned by the World Wild Life Fund, Available at: http://assets.panda.org/downloads/managingthesoyboomenglish_nbvt.pdf (accessed 5 March 2012).

Earles, J.M., Halog, A., 2011. Consequential life cycle assessment: a review. International Journal of Life Cycle Assessment 16, 445–453.

EC, 2009. Directive 2009/28/EC of the European Parliament and of the council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC, Official Journal of the European Union, L140/16 of 5.6.2009.

EC, 2010a. Communication from the Commission on voluntary schemes and default values in the EU biofuels and bioliquids sustainability scheme (2010/C 160/01), Official Journal of the European Union of 19.6.2010.

EC, 2010b. Report from the Commission on indirect land-use change related to biofuels and bioliquids, COM (2010) 811 final, Brussels, 22.12.2010.

EC, 2010c. Commission Decision of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC, Official Journal of the European Union, L151/19 of 17.6.2010.

EC, 2012. Commission Proposal for a Directive of the European Parliament and of the Council amending Directive 98/70/EC relating to the quality of petrol and diesel fuels and amending Directive 2009/28/EC on the promotion of the use of energy from renewable sources. COM(2012) 595 final. Brussels, 17.10.2012

EIA, 2013. International Energy Statistics. Prepared by the U.S. Energy Information Administration (EIA). Available at: http://www.eia.gov (accessed January 9, 2014).

Eisentraut, A., 2010. Sustainable Production of Second-General Biofuels: Potential and Perspectives in Major Economics and Developing Countries. Information Paper. International Energy Agency (IEA), February 2010.

Ekvall, T., Finnveden, G., 2001. Allocation in ISO 14041—a critical review. Journal of Cleaner Production 9, 197–208.

Ekvall, T., Tillman, A., 1997. Open-loop recycling: criteria for allocation procedures. The International Journal of Life Cycle Assessment 2(3), 155–162.

Ekvall, T., Weidema, B., 2004. System boundaries and input data in consequential life cycle inventory analysis. The International Journal of Life Cycle Assessment 9, 161–71.

EMBRAPA, 2010. Tecnologias de Produção de Soja: Região Central do Brasil 2011. EMBRAPA – Empresa Brasileira de Pesquisa Agropecuária. Londrina: Embrapa Soja, 2010.

Environment Canada, 2009. Biogas Flare and Sour Gas calculator. NPRI Toolbox, Env. Canada based on AP-42 US EPA Clean Air Criteria emission factors are from the US EPA's WebFIRE database. Retrieved from http://www.ec.gc.ca/pdb/npri/documents/2004ToolBox/toolBox_e.cfm

EPE, 2013. Brazilian Energy Balance 2013: Year 2012. Rio de Janeiro: EPE-Empresa de Pesquisa Energética.

Erisman, J.W., van Grinsven, H., Leip, A., Mosier, A., Bleeker ,A., 2010. Nitrogen and biofuels; an overview of the current state of knowledge. Nutrient Cycling in Agroecosystems 86, 211-223.

ERSE, 2013. Informação sobre Produção em Regime Especial (PRE) - Portugal Continental 2013. ERSE - Entidade Reguladora dos Serviços Energéticos.

EurObserv'er, 2013. Biofuels barometer. Systèmes Solaires, Le journal des énergies renouvelables 216, July 2013.

Faist Emmenegger, M., Heck, T., Jungbluth, N., 2007. Erdgas. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent data v2.0. Editors: Dones R.. Volume: 6. Swiss Centre for LCI, PSI. Dübendorf and Villigen, CH.

Faist Emmenegger, M., Reinhard, J., Zah, R., 2009. Sustainability Quick Check for Biofuels – intermediate background report. With contributions from T. Ziep, R. Weichbrodt, Prof. Dr. V. Wohlgemuth, FHTW Berlin and A. Roches, R. Freiermuth Knuchel, Dr. G. Gaillard, Agroscope Reckenholz-Tänikon. Dübendorf.

FAO and IFA, 2001. Global estimates of gaseous emissions of NH3, NO and N_2O from agricultural land. First version, published by Food and Agriculture Organization of the United Nations (FAO) and International Fertilizer Industry Association (IFA). Rome, 2001.

FAO, 2006. World agriculture: towards 2030/2050 - Interim report. Prospects for food, nutrition, agriculture and major commodity groups. Global Perspective Studies Unit Food and Agriculture Organization of the United Nations, Rome, June 2006.

FAO, 2013a. FAOSTAT. FAO - Food and Agriculture Organisation of the United Nations. Retrieved from http://faostat.fao.org

FAO, 2013b. Commodity Prices. Food and Agriculture Organization of the United Nations. Retrieved from http://www.fao.org/economic/est/prices

Fargione, J., Hill, J., Tilman, D., Polasky, S., Hawthorne, P., 2008. Land clearing and the biofuel carbon debt. Science 319, 1235–1238.

Fedepalma, 2009. Anuario estadistico 2009-la agroindustria de la palma de aceite en Colombia. Bogota, Colombia.

Fehrenbach, H., Giegrich, J., Gärtner, S., Reinhardt, G., Rettenmaier, N., 2007. Greenhouse Gas Balances for the German Biofuels Quota Legislation. Methodological Guidance and Default Values. Prepared for the Federal Environmental Agency Germany, Draft, Heidelberg, December 2007.

Finnveden, G., Lindfors, L., 1998. Data quality of life cycle inventory data—rules of thumb. International Journal of Life Cycle Assessment 3(2), 65–66.

Flach, B., Wilson, J., Wideback, A., Bettini, O., Guerrero, M., Bendz, K., 2012. EU-27 Oilseeds and Products Annual. USDA Foreign Agricultural Service: GAIN-Global Agricultural Information Network. GAIN Report Number: E70016.

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

FNP, 2010. Agrianual 2010: Anuário da Agricultura Brasileira. *FNP* Consultoria e Comércio - São Paulo.

FNP, 2011. Agrianual 2011: Anuário da Agricultura Brasileira. *FNP* Consultoria e Comércio - São Paulo.

Friedrich, T., 2005. Does No-till Farming require more herbicides? Outlooks on Pest Management, 188-191.

Frischknecht, R., Jungbluth, N., Althaus, H.J., Doka, G., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., Wernet, G., 2007a. Overview and Methodology. Ecoinvent report No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf.

Frischknecht, R., Tuchschmid, M., Faist Emmenegger, M.,Bauer, C., Dones, R., 2007b. Strommix und Stromnetz. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent data v2.0. Editors: Dones R.. Volume: 6. Swiss Centre for LCI, PSI. Dübendorf and Villigen, CH.

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

Gabe, U., Rodella, A.A., 1999. Trace elements in Brazilian agricultural limestones and mineral fertilizers. Communications in Soil Science and Plant Analysis 30 (5-6), 605-620.

Gawel, E., Ludwig, G., 2011. The iLUC dilemma: How to deal with indirect land use changes when governing energy crops? Land Use Policy 28, 846–856.

GEA, 2012. Global Energy Assessment - Toward a Sustainable Future. Cambridge University Press, Cambridge, UK and New York, USA and the International Institute for Applied Systems Analysis, Laxenburg, Austria.

Gerendas. J., Heng, A., 2010. Oil Palm Fertilization – Sharing Some Perspectives. ISP Selangor Seminar (8-Jun, 2010) "Plantation industry competitive strategies in achieving a sustainable future", 2-7.

Germer, J., Sauerborn, J., 2008. Estimation of the impact of oil palm plantation establishment on greenhouse gas balance. Environment, Development and Sustainability 10(6), 697–716.

Gibbs, H.K., Johnston, M., Foley, J.A., Holloway, T., Monfreda, C., Ramankutty, N., Zaks, D., 2008. Carbon payback times for crop-based biofuel expansion in the tropics: The effects of changing yield and technology. Environmental Research Letters 3, 34001–34011.

Gnansounou, E., Dauriat, A., Villegas, J., Panichelli, L., 2009. Life cycle assessment of biofuels: energy and greenhouse gas balances. Bioresource Technology 100(21), 4919–30.

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A.M., Struijs, J., van Zelm, R., 2012. ReCiPe 2008: a life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level; First edition (revised); Report I: Characterisation, July 2012. Available at: http://www.lcia-recipe.net (accessed January 2013).

González-García, S., Bacenetti, J., Negri, M., Fiala, M., Arroja, L., 2013. Comparative environmental performance of three different annual energy crops for biogas production in Northern Italy. Journal of Cleaner Production 43, 71-83.

Guinée, J., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., Huijbregts, M., 2002. Handbook on Life Cycle Assessment – Operational Guide to the ISO Standards. I. LCA in Perspective; IIa: Guide; IIb: Operational Annex III: Scientific Background. Dordrecht: Kluwer Academic Publishers.

Gutser, R., Ebertseder, T., Weber, A., Schraml, M., Schmidhalter, U., 2005. Short-term and residual availability of nitrogen after long-term application of organic fertilizers on arable land. Journal of Plant Nutrition and Soil Science 168, 439-446.

Hamelinck, C., Schober, S., Mittelbach, M., Verolet, J., Dehue, B., 2007. Fatty acid ethyl esters. Final report for Lot 3a of the Bioscopes project, June 2007.

Hansen, S.B., Olsen, S.I., Ujang, Z., 2012. Greenhouse gas reductions through enhanced use of residues in the life cycle of Malaysian palm oil derived biodiesel. Bioresource technology 104, 358–66.

Harsono, S.S., Prochnow, A., Grundmann, P., Hansen, A., Hallmann, C., 2012. Energy balances and greenhouse gas emissions of palm oil biodiesel in Indonesia. GCB Bioenergy 4(2), 213–228.

Hassan, M.N.A., Jaramillo, P., Griffin, W.M., 2011. Life cycle GHG emissions from Malaysian oil palm bioenergy development: The impact on transportation sector's energy security. Energy Policy 39(5), 2615–2625.

Heijungs, R., 2013. Ten easy lessons for good communication of LCA. International Journal of Life Cycle Assessment (October 2013).

Henson, I.E., Rodrigo Ruiz, R., Romero, H.M., 2012. The greenhouse gas balance of the oil palm industry in Colombia : a preliminary analysis II. Greenhouse gas emissions and the carbon budget. Agronomía Colombiana 30(3), 370-378.

Heriansyah, 2008. Optimizing the use of oil palm by-product (EFB) as fertiliser supplement for oil palm. Jakarta, Indonesia.

Hoekstra, A.Y., 2012. The hidden water resource use behind meat and dairy. Animal Frontiers 2, 3-8.

Hokazono, S., Hayashi, K., 2012. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. Journal of Cleaner Production 28, 101-112.

Hou, J., Zhang, P., Yuan, X., Zheng, Y., 2011. Life cycle assessment of biodiesel from soybean, jatropha and microalgae in China conditions. Renewable and Sustainable Energy Reviews 15, 5081– 5091.

Huijbregts, M. A. J., Breedveld, L., Huppes, G., de Koning, A., van Oers, L., Suh, S., 2003. Normalisation figures for environmental life-cycle assessment. Journal of Cleaner Production 11(7), 737–748.

Huijbregts, M., Hauschild, M., Jolliet, O., Margni, M., McKone, T., Rosenbaum, R. K., van de Meent, D., 2010. USEtox User manual. Report version: 1.01, February 2010. Available at: http://www.usetox.org/model/publications

Huo, H., Wang, M., Bloyd, C., Putsche, V., 2009. Life-cycle assessment of energy use and greenhouse gas emissions of soybean-derived biodiesel and renewable fuels. Environmental Science and Technology 43, 750-756.

IBAMA, 2010. Produtos agrotóxicos e afins comercializados em 2009 no Brasil: uma abordagem ambiental. Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis – Ibama, Brasília.

IBGE, 2012. Produção Agrícola Municipal. Área plantada, área colhida, quantidade produzida e valor da produção da lavoura temporária. Instituto Brasileiro de Geografia e Estatística - IBGE. Available at: http://www.sidra.ibge.gov.br (accessed August 10, 2012).

IEA, 2008. Electricity/Heat in Colombia in 2008. Paris, France: International Energy Agency-IEA, 2008.

IFA, 2013. Statistics-IFADATA. IFA-International Fertilizer Industry Association. Retrieved from http://www.fertilizer.org/ifa/ifadata/results (accessed October 12, 2013).

IPCC, 1995. IPCC Second Assessment Report: Climate Change 1995. Cambridge, United Kingdom and New York, USA: Cambridge University Press.

IPCC, 2001. IPCC Third Assessment Report: Climate Change 2001: The Scientific Basis. (X. D. Houghton J T, Ding Y, Griggs D J, Noguer M, van der Linden P J, Ed.). Cambridge, United Kingdom and New York, USA: Cambridge University Press.

IPCC, 2006. IPCC guidelines for national greenhouse gas inventories. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.). Hayama, Japan: Institute for Global Environmental Strategies.

IPCC, 2007. IPCC Fourth Assessment Report: Climate Change 2007. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007. Cambridge, United Kingdom and New York, USA: Cambridge University Press.

Isitekhale, H., Osemwota, I., Amhakhian, S., 2013. Poultry manure and NPK fertilizer application and their residual effects on the yield and yield components of tomato (*Lycopersiconesculentum. mill*) in two distinct ecological zones of Central Southern Nigeria. IOSR Journal of Agriculture and Veterinary Science 3 (2), 40-47.

ISO, 2006a. Environmental management - Life cycle assessment - Principles and framework, ISO 14040. International Organization for Standardization, Geneve, Switzerland.

ISO, 2006b. Environmental management- Life cycle assessment - Requirements and guidelines, ISO 14044. International Organization for Standardization, Geneve, Switzerland.

James, C., 2003-2009. Global status of commercialized transgenic crops. International Service for the Acquisition of Agri-biotech Applications, ISAAA. Available at: http://www.isaaa.org

Janssen, R., Rutz, D.D., 2011. Sustainability of biofuels in Latin America: Risks and opportunities. Energy Policy 39, 5717–5725.

Jungbluth, N., 2007. Erdöl. Sachbilanzen von Energiesystemen. Final report No. 6 ecoinvent data v2.0. Editors: Dones R.. Volume: 6. Swiss Centre for LCI, PSI. Dübendorf and Villigen, CH.

Jungbluth, N., Chudacoff, M., Dauriat, A., Dinkel, F., Doka, G., Faist Emmenegger, M., Gnansounou, E., Kljun, N., Spielmann, M., Stettler, C., Sutter, J., 2007. Life Cycle Inventories of Bioenergy. Final report ecoinvent data v2.0. Volume: 17. Swiss Centre for LCI, ESU. Duebendorf and Uster, CH.

Kaewmai, R., H-Kittikun, A., Musikavong, C., 2012. Greenhouse gas emissions of palm oil mills in Thailand. International Journal of Greenhouse Gas Control 11, 141–151.

Kellenberger, D., Althaus, H.-J., Jungbluth, N., Künniger, T., 2007. Life Cycle Inventories of Building Products. Final report ecoinvent data v2.0, Vol. 7, Swiss Centre for LCI, Empa – TSL, Dübendorf.

Kendall, A., Chang, B., 2009. Estimating life cycle greenhouse gas emissions from corn-ethanol: a critical review of current U.S. practices. Journal of Cleaner Production 17, 1175-1182.

Kendall, A., Yuan, J., 2013. Comparing life cycle assessments of different biofuel options. Current Opinion in Chemical Biology 17, 439–443.

Khalid, H., Zin, Z.Z., Anderson, J.M., 2000. Decomposition processes and nutrient release patterns of oil palm residues. Journal of Oil Palm Research 12, 46-63.

Kim, S., Dale, B.E., 2009. Regional variations in greenhouse gas emissions of biobased products in the United States—corn-based ethanol and soybean oil. The International Journal of Life Cycle Assessment 14, 540–546.

Kløverpris, J., Wenzel, H., Nielsen, P.H., 2008. Life Cycle Inventory Modelling of Land Use Induced by Crop Consumption. Part 1: Conceptual Analysis and Methodological Proposal. International Journal of Life cycle Assessment 13 (1), 13–21.

Knudsen, M.T., Yu-Hui, Q., Yan, L., Halberg, N., 2010. Environmental assessment of organic soybean (Glycine max.) imported from China to Denmark: a case study. Journal of Cleaner Production 18, 1431-1439.

Krautgartner, R., Henard, M.C., Rehder, L. E., Boshnakova, M., Dobrescu, M., Flach, B., Wilson, J., Wideback, A., Bettini, O., Guerrero, M., Bendz, K., 2012. EU-27 Oilseeds and Products Annual. USDA Foreign Agricultural Service: GAIN-Global Agricultural Information Network. GAIN Report Number: E70016.

Krautgartner, R., Henard, M.C., Rehder, L. E., Boshnakova, M., Dobrescu, M., Flach, B., Wilson, J., Bettini, O., Guerrero, M., Bendz, K., 2013. EU-27 Oilseeds and Products Annual 2013. Ample Soybean World Supplies to Boost EU-27 Soybean Meal Consumption. USDA Foreign Agricultural Service: GAIN-Global Agricultural Information Network. GAIN Report Number: AU13002.

Lal, R., 1983. No-till farming: Soil and water conservation and management in the humid and sub-humid tropics. IITA Monograph Nº 2, Ibadan, Nigeria.

Lam, M.K., Lee, K.T., 2011. Renewable and sustainable bioenergies production from palm oil mill effluent (POME): win-win strategies toward better environmental protection. Biotechnology advances 29, 124-41.

Lam, M.K., Lee, K.T., Mohamed, A.R., Pinang, P., 2009. Life cycle assessment for the production of biodiesel : A case study in Malaysia for palm oil versus jatropha oil. Biofuels, Bioproducts and Biorefining (3), 601–612.

Landis, A.E., Theis, T.L., 2008. Comparison of life cycle impact assessment tools in the case of biofuels. 2008 IEEE International Symposium on Electronics and the Environment, 1–7.

Lange, M., 2011. The GHG balance of biofuels taking into account land use change. Energy Policy 39, 2373-2385.

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

Lee, K.T., Ofori-Boateng, C., 2013. Sustainability of Biofuel Production from Oil Palm Biomass. Green Energy and Technology. Springer Science+Business Media Singapore 2013. ISBN 978-981-4451-69-7.

Lehuger, S., Gabrielle, B., Gagnaire, N., 2009. Environmental impact of the substitution of imported soybean meal with locally-produced rapeseed meal in dairy cow feed. Journal of Cleaner Production 17, 616–624.

Leip, A., Weiss, F., Wassenaar, T., Perez, I., Fellmann, T., Loudjani, P., Tubiello, F., Grandgirard, D., Monni, S., Biala, K., 2010. Evaluation of the livestock sector's contribution to the EU Greenhouse Gas Emissions (GGELS) e final report. European Commission, Joint Research Center, Ispra.

Lynd, L.R., Aziz, R.A., de Brito Cruz, C.H., Chimphango, A.F.A., Cortez, L.A.B., Faaij, A., Greene, N., Keller, M., Osseweijer, P., Richard, T.L., Sheehan, J., Chugh, A., van der Wielen, L., Woods, J., van Zyl, W.H., 2011. A

global conversation about energy from biomass: the continental conventions of the global sustainable bioenergy project. Interface Focus 1, 271–279.

Macedo, M. N., DeFries, R. S., Morton, D. C., Stickler, C. M., Galford, G. L., Shimabukuro, Y. E., 2012. Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. Proceedings of the National Academy of Sciences 109(4), 1341-1346.

MAGyP, 2012.Exportaciones de Granos, Aceites y Subproductos - Volúmenes exportados de granos. Ministerio de Agricultura, Ganadería y Pesca-MAGyP. Available at: http://www.minagri.gob.ar(accessed August 10, 2012).

Malça, J., Freire, F. ,2006. Renewability and life-cycle energy efficiency of bioethanol and bio-ethyl tertiary butyl ether (bioETBE): assessing the implications of allocation. Energy – The International Journal 31, 3362-3380.

Malça, J., Freire, F., 2009. Energy and Environmental Benefits of Rapeseed Oil Replacing Diesel. International Journal of Green Energy 6, 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, 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 and Sustainable Energy Reviews 15, 338–351.

Manik, Y., Halog, A., 2012. A Meta-Analytic Review of Life Cycle Assessment and Flow Analyses Studies of Palm Oil Biodiesel. Integrated Environmental Assessment and Management 9 (1), 134-141.

MAPA, 2012. Sistema de Agrotóxicos Fitossanitários - AGROFIT. Ministério Brasileiro da Agricultura, Pecuária e Abastecimento - Coordenação-Geral de Agrotóxicos e Afins. Available at: http://extranet.agricultura.gov.br

Mathews, J., Tan, H., 2009. Biofuels and indirect land use change effects: the debate continues. Biofuels Bioproducts & Biorefining 3, 305–317.

Matheys, J., van Autenboer, W., Timmermans, J-M, van Mierlo, J., van den Bossche, P., Maggetto, G., 2007. Influence of functional unit on the life cycle assessment of traction batteries. International Journal of Life Cycle Assessement 12(3), 191–196.

Menichetti, E., Otto, M., 2009. Energy balance and greenhouse gas emissions of biofuels from a life-cycle perspective. In: R.W. Howarth and S. Bringezu (eds) Biofuels: Environmental Consequences and Interactions with Changing Land Use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE). International Biofuels Project Rapid Assessment, 22-25 September 2008, Gummersbach Germany. Cornell University, Ithaca NY, USA.

Meyer, D. E., Cederberg, C., 2010. Pesticide use and glyphosateresistant weeds – a case study of Brazilian soybean production. SIK-Rapport Nr 809.

Middelaar, C.E., Cederberg, C., Vellinga, T.V., van der Werf, H.M.G., de Boer, I.J.M., 2013. Exploring variability in methods and data sensitivity in carbon footprints of feed ingredients. The International Journal of Life Cycle Assessment 18 (4), 768-782.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R.F., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2006. Key Elements in a Framework for Land Use Impact Assessment Within LCA. International Journal of Life Cycle Assessment 12 (1), 5–15.

Milazzo, M.F., Spina, F., Primerano, P., Bart, J.C.J., 2013. Soy biodiesel pathways: Global prospects. Renewable and Sustainable Energy Reviews 26, 579–624.

Miller, S., Landis, A., Theis, T., 2006. Use of Monte Carlo Analysis to Characterize Nitrogen Fluxes in Agroecosystems. Environmental Science and Technology 40, 2324-2332.

Miller, S., Landis, A., Theis, T., Reich, R., 2007. A Comparative Life Cycle Assessment of Petroleum and Soybean-Based Lubricants. Environmental Science and Technology 41, 4143-4149.

Miller, S.A., 2010. Minimizing Land Use and Nitrogen Intensity of Bioenergy. Environmental Science and Technology 44, 3932–3939.

Morais, S., Mata, T.M., Ferreira, E., 2010. Life Cycle Assessment of Soybean Biodiesel and LPG as Automotive Fuels in Portugal. Chemical Engineering Transactions 19, 267-272.

Mourad, A.L., Walter, A., 2011. The energy balance of soybean biodiesel in Brazil: a case article. Biofuels, Bioproducts and Biorefinery 5, 185-197.

Nassar, A.M., Antoniazzi, L. B., Moreira, M. R., Chiodi, L., Harfuch, L., 2010. An allocation methodology to assess GHG emissions associated with land use change – Final Report. ICONE (Institute for International Trade Negotiations).

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

Nemecek, T., Kägi, T., 2007. Life Cycle Inventories of Swiss and European Agricultural Production Systems. Final report ecoinvent V2.0 No. 15a. , Agroscope Reckenholz-Taenikon Research Station ART, Swiss Centre for Life Cycle Inventories, Zurich and Dübendorf, CH.

Oberholzer, H.R., Weisskopf, P., Gaillard, G., Weiss, F., Freiermuth, R., 2006. Methode zur Beurteilung der Wirkungen landwirtschaftlicher Bewirtschaftung auf die Bodenqualität in Ökobilanzen – SALCA-SQ. Agroscope FAL Reckenholz.

OECD/FAO, 2013. OECD – FAO Agricultural Outlook 2013-2022 Highlights. Retrieved from http://www.oecd.org/

Opara-Nadi, O.A., 1993. Chapter 8 - Conservation tillage for increased crop production. In: FAO (Ed), Soil tillage in Africa: needs and challenges. FAO Soils Bulletin 69.

Ortega, E., Cavalett, O., Bonifácio, R., Watanabe, M., 2005. Brazilian soybean production: emergy analysis with an expanded scope. Bulletin of Science, Technology & Society 25, 323-334.

Padula, A.D., Santos, M.S., Ferreira, L., Borenstein, D., 2012. The emergence of the biodiesel industry in Brazil: Current figures and future prospects. Energy Policy, 44, 395-405.

Panichelli, L., Dauriat, A., Gnansounou, E., 2009. Life cycle assessment of soybean-based biodiesel in Argentina for export. The International Journal of Life Cycle Assessment 14, 144-159.

Papong, S., Chom-In, T., Noksa-nga, S., Malakul, P., 2010. Life cycle energy efficiency and potentials of biodiesel production from palm oil in Thailand. Energy Policy 38(1), 226–233.

Pardo, R., Acevedo, H., Freire, F., 2006. Life Cycle Inventory of Palm oil for Colombia. Second International Seminar on Biofuels. National University of Colombia, Bogotá, August 9-11, 2006.

Patthanaissaranukool, W., Polprasert, C., Englande, A.J., 2013. Potential reduction of carbon emissions from Crude Palm Oil production based on energy and carbon balances. Applied Energy 102, 710–717.

Patyk, A., Reinhardt, G., 1997. Düngemittel - Energie- und Stoffstromsbilanzen. Vieweg. Umweltvissenschaften. Friedr. Vieweg & Sohn Verlagsgesellschaft mbH, Braunschweig/Wiesbaden, Germany.

Pawelzik, P., Carus, M., Hotchkiss, J., Narayan, R., Selke, S., Wellisch, M., Weiss, M., Wicke, B., Patel, M.K., 2013. Critical aspects in the life cycle assessment (LCA) of bio-based materials – Reviewing methodologies and deriving recommendations. Resources, Conservation and Recycling 73, 211– 228.

Payraudeau, S., van der Werf, H.M.G., Vertès, F., 2007. Analysis of the uncertainty associated with the estimation of nitrogen losses from farming systems. Agricultural Systems 94 (2), 416–430.

Pennington, D.W., Potting, J., Finnveden, G., Lindeijer, E., Jolliet, O., Rydberg, T., Rebitzer, G., 2004. Life cycle assessment Part 2: Current impact assessment practice. Environment International 30, 721–739.

Pinzon, L., 2012. Biofuels Annual: Colombian Biofuels Use Close to reaching E10 and B10 Levels. USDA Foreign Agricultural Service: GAIN-Global Agricultural Information Network.

Pleanjai, S., Gheewala, S.H., 2009. Full chain energy analysis of biodiesel production from palm oil in Thailand. Applied Energy 86, S209–S214.

Pleanjai, S., Gheewala, S.H., Garivait. S., 2009. Greenhouse gas emissions from production and use of palm methyl ester in Thailand. International Journal of Global Warming 1(4), 418–431.

Ponsioen, T.C., Blonk, T.J., 2012. Calculating land use change in carbon footprints of agricultural products as an impact of current land use. Journal of Cleaner Production 28, 120-126.

Pradhan, A., Shrestha, D. S., Van Gerpen, J., Duffield, J., 2008. The energy balance of soybean oil biodiesel production: a review of past studies. American Society of Agricultural and Biological Engineers 51, 185-194.

Prasuhn, V., 2006. Erfassung der PO4-Austräge für die Ökobilanzierung SALCA Phosphor. Agroscope Reckenholz - Tänikon ART, 20p.

Product Board MVO, 2011. Fact sheet Soy. Rijswijk, The Netherlands: Product Board Margarine, Fats and Oils.

Prudêncio da Silva, V., van der Werf, H.M.G., Spies, A., Soares, S.R., 2010. Variability in environmental impacts of Brazilian soybean according to crop production and transport scenarios. Journal of Environmental Management 91, 1831-1839.

Reap, J., Roman, F., Duncan, S., Bras, B., 2008. A survey of unresolved problems in life cycle assessment Part 2: impact assessment and interpretation. International Journal of Life Cycle Assessment 13, 374–388.

Reijnders, L., Huijbregts, M., 2008a. Palm oil and the emission of carbon-based greenhouse gases. Journal of Cleaner Production 16, 477–482.

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

Reijnders, L., Huijbregts, M., 2011. Nitrous oxide emissions from liquid biofuel production in life cycle assessment. Current Opinion in Environmental Sustainability 3(5), 432–437.

Reinhard, J., Zah, R., 2009. Global environmental consequences of increased biodiesel consumption in Switzerland: consequential life cycle assessment. Journal of Cleaner Production 17, S46-S56.

Reinhard, J., Zah, R., 2011. Consequential life cycle assessment of the environmental impacts of an increased rapemethylester (RME) production in Switzerland. Biomass and Bioenergy 35, 2361-2373.

REN, 2013. Dados Técnicos/Technical Data 2012. REN - Redes Energéticas Nacionais. Available at: http://www.centrodeinformacao.ren.pt (acessed January 2, 2013).

REN21, 2013. Renewables 2013 - Global Status Report. Renewable Energy Policy Network for the 21st century. Available at: http://www.map.ren21.net (accessed October 10, 2013).

Renard, K.G., Freimund, J.R., 1994. Using monthly precipitation data to estimate the R factor in the revised USLE. Journal of Hydrology 157, 287-306.

Requena, J.F.S., Guimaraes, A.C., Quirós Alpera, S., Relea Gangas, E., Hernandez-Navarro, S., Navas Gracia, L.M., 2011. Life Cycle Assessment (LCA) of the biofuel production process from sunflower oil, rapeseed oil and soybean oil. Fuel Processing Technology 92(2), 190-199.

Rincón, V., 2009. Dinámica de la expansión del área cultivada com palma de aceite y su impacto en la cobertura del suelo: Zona Oriental palmera de Colombia (1972-2009). MSc. Universitat de Girona, Girona, Spain.

Rochette, P., Janzen, H.H., 2005. Towards a revised coefficient for estimating N_2O emissions from legumes. Nutrient Cycling in Agroecosystems 73, 171-179.

Rodríguez, M., Van Hoof., B., 2004. Desempeño ambiental del sector palmero en Colombia: evaluación y perspectivas. Fedepalma, Bogota.

Romanelli, T. L.; Milan, M., 2010. Material flow determination through agricultural machinery management. Scientia Agricola 67 (4), 375-495.

Rosenbaum, R.K., Bachmann, T.M., Gold, L.S., Huijbregts, M.A.J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H.F., MacLeod, M., Margni, M., McKone, T.E., Payet, J., Schuhmacher, M., van de Meent, D., Hauschild, M.Z., 2008. USEtox—the UNEP/SETAC-consensus model: recommended characterisation factors for human toxicity

and freshwater ecotoxicity in life cycle impact assessment. The International Journal of Life Cycle Assessment 13, 532-546.

Rosillo-Calle, F., Pelkmans, L., Walter, A., 2009. A Global Overview of Vegetable Oils, with Reference to Biodiesel. A Report for the IEA Bioenergy Task 40.

Santos, J., 2006. Environmental Life Cycle Assessment of Palm oil in Colombia (in Portuguese). Mechanical Engineering Integrated Master Thesis, 2006. University of Coimbra, Portugal.

Santos, O.I.B., Rathmann, R., 2009. Identification and analysis of local and regional impacts from the introduction of biodiesel production in the state of Piauí. Energy Policy 37, 4011–4020.

Schaffel, S.B., La Rovere, E.L., 2010. The quest for eco-social efficiency in biofuels production in Brazil. Journal of Cleaner Production 18, 1663-1670.

Scharlemann, J.P., Laurance, W.F., 2008. Environmental Science: How Green Are Biofuels? Science 319 (5859), 43-44.

Schmidt, J.H., 2007. Life cycle assessment of rapeseed oil and palm oil. Ph.D. thesis, Part 3: Life cycle inventory of rapeseed oil and palm oil. Department of Development and Planning, Aalborg Universit.

Schmidt, J.H., 2010. Comparative life cycle assessment of rapeseed oil and palm oil. The International Journal of Life Cycle Assessment 15(2), 183–197.

Schmidt, J.H., Weidema, B.P., 2008. Shift in the marginal supply of vegetable oil. International Journal of Life Cycle Assessment 13(3), 235–239.

Searchinger, T., Heimlich, R., 2009. Estimating greenhouse gas emissions from Soy-Based US biodiesel when factoring in emissions from land use change. Biofuels, Food and Feed Tradeoffs, 45-55.

Searchinger, T., Heimlich, R., Houghton, R. A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.H., 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change. Science 319, 1238-1240.

Siangjaeo, S., Gheewala, S. H., Unnanon, K., Chidthaisong, A., 2011. Implications of land use change on the life cycle greenhouse gas emissions from palm biodiesel production in Thailand. Energy for Sustainable Development 15 (1), 1-7.

SIIA, 2012. Producción Agrícola Por Provincias. Sistema Integrado de Información Agropecuaria - SIIA, Ministerio de Agricultura, Ganadería y Pesca. Available at: http://www.siia.gov.ar (accessed August 10, 2012).

Silalertruksa, T., Gheewala, S. H., 2012a. Food, Fuel, and Climate Change. Is Palm-Based Biodiesel a Sustainable Option for Thailand? Journal of Industrial Ecology 16(4), 541–551.

Silalertruksa, T., Gheewala, S. H., 2012b. Environmental sustainability assessment of palm biodiesel production in Thailand. Energy 43, 306-314.

Silva, D., Lahr, F., Garcia, R., Freire, F., Ometto, A., 2013. Life cycle assessment of medium density particleboard (MDP) produced in Brazil". International Journal of Life Cycle Assessment 18 (7), 1404-1411.

Silva, G.S., 2010. Os Desafios da Soja no Brasil. Aprosoja- Associação dos Produtores de Soja e Milho do Estado de Mato Grosso. FASUL – Faculdade Sul Brasil, Maio 2010.

Sleeswijk, A.W., van Oers, L.F.C.M., Guinée, J.B., Struijs, J., Huijbregts, M.A.J., 2008. Normalisation in product life cycle assessment: an LCA of the global and European economic systems in the year 2000. The Science of the Total Environment 390 (1), 227–40.

Smaling, E.M.A., Roscoe, R., Lesschen, J.P., Bouwman, A.F., Comunello E., 2008. From forest to waste: Assessment of the Brazilian soybean chain, using nitrogen as a marker. Agriculture, Ecosystems and Environment 128, 185–197.

Smeets, E.M.W., Bouwman, L.F., Stehfest, E., van Vuuren, D.P., Posthuma, A., 2009. Contribution of N_2O to the greenhouse gas balance of first-generation biofuels. Global Change Biology 15, 1-23.

Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agriculture, Ecosystems and Environment 133, 247–266.

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

Spielmann, M., Dones, R., Bauer, C., 2007. Life Cycle Inventories of Transport Services. Final report ecoinvent Data v2.0, Vol. 14, Dübendorf and Villigen, Switzerland, Swiss Centre for LCI, PSI.

Stichnothe, H, Schuchardt, F., 2010. Comparison of different treatment options for palm oil production waste on a life cycle basis. International Journal of Life Cycle Assessment 15(9), 907–915.

Stichnothe, H., Schuchardt, F., 2011. Life cycle assessment of two palm oil production systems. Biomass and Bioenergy 35(9), 3976–3984.

Suppalakpanya, K., Ratanawilai, S.B., Tongurai, C., 2010. Production of ethyl ester from esterified crude palm oil by microwave with dry washing by bleaching earth. Applied Energy 87, 2356-2359.

Sutter, J., 2007. Life Cycle Inventories of Highly Pure Chemicals. Final report ecoinvent Data v2.0. Editors: 0. Volume: 19. Swiss Centre for LCI, ETHZ. Duebendorf and St. Gallen, CH.

Tan, K.T., Lee, K.T., Mohamed, A.R., Bhatia, S., 2009. Palm oil: addressing issues and towards sustainable development. Renewable and Sustainable Energy Reviews 13, 420-427.

Thamsiriroj, T., Murphy, J.D., 2009. Is it better to import palm oil from Thailand to produce biodiesel in Ireland than to produce biodiesel from indigenous Irish rapeseed? Applied Energy 86(5), 595–604.

The Royal Society, 2008. Sustainable biofuels: prospects and challenges. Report prepared by The Royal Society working group on biofuels. Policy document 01/08. Available at: http://www.royalsociety.org

Tillman, A.M., Ekvall, T., Baumann, H., Rydberg, T., 1994. Choice of system boundaries in life cycle assessment. Journal of Cleaner Production 2, 21–29.

Tosi, J., Mosciaro, M., Borda, M., Forján, H., Marinissen, A., Pereyra, E. S., 2005. Haciendo numeros para la campaña 05/06 de soja. Instituto Nacional de Tecnología Agropecuaria - INTA. Available at: http://riap.inta.gov.ar/ (accessed August 14, 2012).

Udo de Haes, H.A., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E.G., Hofstetter, P., Jolliet, O., Klopffer, W., Krewitt, W., Lindeijer, E., Mueller-Wenk, R., Olsen, S.I., Pennington, D.W., Potting, J., Steen, B. (eds), 2002. Life-cycle impact assessment: striving towards best practice. Society of Environmental Toxicology and Chemistry (SETAC), Pensacola.

USDA, 1999. Soil Taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys. Agriculture Handbook. Number 436, United States Department of Agriculture Natural Resources Conservation Service.

USDA, 2013a. Oilseeds: World Markets and Trade. USDA - United States Department of Agriculture, Foreign Agricultural Service. Circular Series FOP 11-13, November 2013. Available at: http://apps.fas.usda.gov/psdonline/psdHome.aspx (accessed December 4, 2013).

USDA, 2013b. Agricultural Production, Supply, and Distribution. USDA - United States Department of Agriculture. Available at: http://www.indexmundi.com (accessed August 14, 2013).

van Dam, J., Faaij, A.P.C., Hilbert, J., Petruzzi, H., Turkenburg, W.C., 2009. Large-scale bioenergy production from soybeans and switchgrass in Argentina Part B. Environmental and socio-economic impacts on a regional level. Renewable and Sustainable Energy Reviews 13, 1679–1709.

van der Voet, E., Lifset, R.J., Luo, L., 2010. Life-cycle assessment of biofuels, convergence and divergence. Biofuels 1(3), 435–449.

Von Uexkull, H., n.d. Oil Palm (*Elaeis guineensis Jacq*.). Asia Progr Potash Phosphate Institute/International Potash Institute, Singapore.

Walter, A., Dolzan, P., Quilodrán, O., de Oliveira, J. G., da Silva, C., Piacente, F., Segerstedt, A., 2011. Sustainability assessment of bio-ethanol production in Brazil considering land use change, GHG emissions and socio-economic aspects. Energy Policy 39, 5703-5716.

Wang, G., Chen, S., 2012. A review on parameterization and uncertainty in modeling greenhouse gas emissions from soil. Geoderma 170, 206-216.

Wardenaar, T., van Ruijven, T., Beltran, A. M., Vad, K., Guinée, J., Heijungs, R., 2012. Differences between LCA for analysis and LCA for policy: a case study on the consequences of allocation choices in bio-energy policies. International Journal of Life Cycle Assessment 17, 1059–1067.

Weidema, B., Wenzel, H., Petersen, C., Hansen, K., 2004. The product, functional unit and reference flows in LCA. Environmental news no. 70. Danish Ministry of the Environment.

Weidema, B.P., 2003. Market information in life cycle assessment. Copenhagen: Danish Environmental Protection Agency (Environmental Project no. 863). Available at: http://www2.mst.dk

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

Wilke, B., Schaub, D., 1996. Phosphatanreicherung bei Bodenerosion. Mitt. Deutsche Bodenkundl. Gesellsch. 79, 435-438.

Wischmeier, W. H., Smith, D. D., 1978. Predicting rainfall erosion losses – a guide to erosion planning. Agriculture Handbook No. 537, U.S. Department of Agriculture.

Witcover, J., Yeh, S., Sperling, D., 2013. Policy options to address global land use change from biofuels. Energy Policy 56, 63–74.

World Bank, 2013. World Bank Commodity Price Data. World Bank. Available at: http://www.indexmundi.com/commodities/?commodity=food-price-index

Xue, X., Collinge, W.O., Shrake, S.O., Bilec, M.M., Landis, A.E., 2012. Regional life cycle assessment of soybean derived biodiesel for transportation fleets. Energy Policy 48, 295–303.

Yang, H., Zhou, Y., Liu J., 2009. Land and water requirements of biofuel and implications for food supply and the environment in China. Energy Policy 37, 1876-1885.

Yee, K.F., Tan, K.T., Abdullah, A.Z., Lee, K.T., 2009. Life cycle assessment of palm biodiesel: Revealing facts and benefits for sustainability. Applied Energy 86, S189-S196.

Yusoff, S., Hansen, S.B., 2007. Feasibility study of performing an LCA on crude palm oil production in Malaysia. International Journal of Life Cycle Assessment 12, 50-56.

Zamagni, A., Buttol, P., Porta, P.L., Buonamici, R., Masoni, P., Guinée, J., Heijungs, R., Ekvall, T., Bersani, R., Biekowska, A., Pretato, U., 2008. Critical review of the current research needs and limitations related to ISO-LCA practice. Deliverable D7 of Work Package 5 of the CALCAS project: http://fr1.estis.net/sites/calcas

Zimmer, Y., 2010. Competitiveness of rapeseed, soybeans and palm oil. Journal of Oilseed Brassica 1(2), 84-90.

APPENDIX I: CORE ARTICLES FOR PHD THESIS (ABSTRACTS)

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ENVIRONMENTAL SUSTAINABILITY OF BIODIESEL IN BRAZIL[a]

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HIGHLIGHTS

- Biodiesel production (based on soybean and beef tallow) increased sharply in Brazil
- Land use change, biodiversity, water impacts, GHG and energy balance are critical.
- Diversifying feedstock and adopting ethyl transesterification can minimize impacts.
- Zoning and certification can play an important role for biodiesel sustainability.

ABSTRACT

Biodiesel production in Brazil has grown from 736 m^3 in 2007 to 2.7 Mm^3 in 2012. It is an emergent bioenergy for which it is important to guarantee environmental sustainability. The objective of this article is to characterize the biodiesel production chain in Brazil, to identify potential environmental impacts and to analyze key drivers and barriers for biodiesel environmental sustainability. This article explores these aspects and focuses on the increasing demand for the main feed stocks for biodiesel production in Brazil: soybean oil and beef tallow. The impacts of land use and land use change on greenhouse gas emissions, biodiversity and water, as well as the energy balance were found to be critical for the environmental sustainability assessment and development of biodiesel chains. Increasing agriculture yields, diversifying feed stocks and adopting ethyl transesterification can contribute to minimize environmental impacts. It was also found that environmental impacts can be mitigated by appropriate policies aiming at an integrated optimization of food and bioenergy production and through agro-economic-ecological zoning, allowing adequate use of land for each purpose. Despite the limitation and weakness of some sustainability tools and initiatives, certification and zoning can play an important role for the sustainability of the emerging biodiesel production in Brazil.

Keywords: Soybean biodiesel; Sustainability assessment; Tallow biodiesel.

[a]Castanheira, É.G., Grisoli, R., Freire, F., Garcilasso, V., Coelho, S., 2014. Environmental sustainability of biodiesel in Brazil. *Energy Policy* 65, 680-691. http://dx.doi.org/10.1016/j.enpol.2013.09.062

GREENHOUSE GAS INTENSITY OF PALM OIL PRODUCED IN COLOMBIA ADDRESSING ALTERNATIVE LAND USE CHANGE AND FERTILIZATION SCENARIOS[a]

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HIGHLIGHTS

- A comprehensive evaluation of alternative LUC and fertilization schemes
- The GHG intensity of palm oil greatly depends on the LUC scenario
- Colombian palm area expansion resulted in negative or low Palm Oil GHG intensity
- GHG emissions from plantation vary significantly with N_2O emission parameters

ABSTRACT

The main goal of this article is to assess the life-cycle greenhouse gas (GHG) intensity of palm oil produced in a specific plantation and mill in Colombia. A comprehensive evaluation of the implications of alternative land use change (LUC) scenarios (forest, shrubland, savanna and cropland conversion) and fertilization schemes (four synthetic and one organic nitrogen-fertilizer) was performed. A sensitivity analysis to field nitrous oxide emission calculation, biogas management options at mill, time horizon considered for global warming and multifunctionality approach were also performed. The results showed that the GHG intensity of palm oil greatly depends on the LUC scenario. Significant differences were observed between the LUC scenarios (-3.0 to 5.3 kg CO_2eq kg^{-1} palm oil). The highest result is obtained if tropical rainforest is converted and the lowest if palm is planted on previous cropland, savanna and shrubland, in which almost all LUC from Colombian oil palm area expansion occurred between 1990 and 2009. Concerning plantation and oil extraction, it was shown that field nitrous oxide emissions and biogas management options have a high influence on GHG emissions.

Keywords: carbon footprint; carbon stock change; fertilization; global warming; palm oil biodiesel; vegetable oils

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[a]Castanheira, É.G., Acevedo, H., Freire, F., 2014. Greenhouse gas intensity of palm oil produced in Colombia addressing alternative land use change and fertilization scenarios. *Applied Energy* 114, 958-967. http://dx.doi.org/10.1016/j.apenergy.2013.09.010

ENVIRONMENTAL ASSESSMENT OF PALM OIL PRODUCED IN COLOMBIA[a]

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HIGHLIGHTS

- A LCA of palm oil produced in Colombia is presented.
- Environmental impacts of palm plantation are higher than impacts of oil extraction.
- Fertilization scheme strongly influence the environmental impacts of palm oil.
- Negative palm oil GHG intensities were obtained due to LUC.
- The life-cycle impact assessment method (ReCiPe and CML) influence the results.

ABSTRACT

The majority of palm oil life-cycle assessment (LCA) studies were performed for Asia and most of the studies focused on climate change. The purpose of this article is to develop a LCA palm oil produced in a specific plantation and extraction mill in Colombia, accounting for direct effects of land-use change (LUC) and assessing the implications of different fertilization schemes. Two life-cycle impact assessment (LCIA) methods (ReCiPe and CML) were adopted to determine the extent to which the results are influenced by the method applied. The results showed that the environmental impacts of plantation are higher than impacts of oil extraction. LUC has a strong contribution to the climate change and negative results were obtained. The type of fertilizer does not have much influence on climate change when compared with LUC. However, significant variation on results for the remaining categories was observed for the alternative fertilization schemes. The lowest climate change, photochemical oxidation and ozone layer depletion results were calculated when ammonium sulphate and poultry manure were used as fertilizers. Acidification and marine eutrophication lowest impacts occur when calcium ammonium nitrate was applied. Different results were obtained from the LCIA methods adopted, mainly for eutrophication and photochemical oxidation categories. Normalized results showed that eutrophication is the most important impact category for both LCIA methods.

Keywords: Fertilization; Land-use change (LUC); Life-cycle assessment (LCA); Palm oil; ReCiPe; CML.

[a] Castanheira, É.G., Freire, F., 2014. Environmental assessment of palm oil produced in Colombia *(submitted)*.

GREENHOUSE GAS ASSESSMENT OF SOYBEAN PRODUCTION: IMPLICATIONS OF LAND-USE CHANGE AND DIFFERENT CULTIVATION SYSTEMS[a]

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HIGHLIGHTS

- LC GHG balance of soybean is dominated by LUC emissions.
- Significant GHG variation was calculated for LUC scenarios and cultivation systems.
- Tillage systems have higher GHG emissions than reduced-(no-)tillage systems.
- Uncertainty in N_2O is high and dominates cultivation GHG emissions

ABSTRACT

The increase in soybean production as a source of protein and oil is being stimulated by the growing demand for livestock feed, food and numerous other applications. Significant greenhouse gas (GHG) emissions can result from land use change due to the expansion and cultivation of soybean. However, this is complex to assess and the results can vary widely. The main goal of this article is to investigate the life-cycle GHG balance for soybean produced in Latin America, assessing the implications of direct land use change emissions and different cultivation systems. A life-cycle model, including inventories for soybean produced in three different climate regions, was developed, addressing land use change, cultivation and transport to Europe. A comprehensive evaluation of alternative land use change scenarios (conversion of tropical forest, forest plantations, perennial crop plantations, savanna and grasslands), cultivation (tillage, reduced tillage and no-tillage) and soybean transportation systems was undertaken. The main results show the importance of land use change in soybean GHG emissions, but significant differences were observed for the alternative scenarios, namely 0.1-17.8 kg CO_2eq kg^{-1} soybean. The original land choice is a critical issue in ensuring the lowest soybean GHG balance and degraded grassland should preferably be used for soybean cultivation. The highest GHG emissions were calculated for tropical moist regions when rainforest is converted into soybean plantations (tillage system). When land use change is not considered, the GHG intensity varies from 0.3 to 0.6 kg CO_2eq kg^{-1} soybean. It was calculated that all tillage systems have higher GHG emissions than the corresponding no-tillage and reduced tillage systems. The results also show that N_2O emissions play a major role in the GHG emissions from cultivation, although N_2O emission calculations are very sensitive to the parameters and emission factors adopted.

Keywords: Carbon footprint; Carbon stocks; Land conversion; Life Cycle Assessment (LCA); Soil management; Tillage.

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[a]Castanheira, É.G., Freire, F., 2013. Greenhouse gas assessment of soybean: implications of land use change and different cultivation systems. *Journal of Cleaner Production* 54, 49-60.
http://dx.doi.org/10.1016/j.jclepro.2013.05.026

LIFE-CYCLE ASSESSMENT OF SOYBEAN-BASED BIODIESEL IN EUROPE: COMPARING GRAIN, OIL AND BIODIESEL IMPORT FROM BRAZIL[a]

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HIGHLIGHTS

- A LCA of soybean-based biodiesel (SME) is presented.
- The effects of importing grain, oil or biodiesel from Brazil are assessed.
- LUC strongly influence the GHG intensity of SME.
- Cultivation and transport are the phases that most contribute for SME impacts.
- The origin of soybean greatly influences the LCA results.
- Toxicity impacts results are greatly influenced by the LCA method.

ABSTRACT

The majority of life-cycle assessment (LCA) studies for soybean-based biodiesel (soybean methyl ester - SME) have been focused on climate change. The purpose of this article is to present a LCA of SME delivery to a fuel blending facility in Portugal, accounting for the implications of produce soybean in four states in Brazil. The influence of the location of the oil extraction and transesterification mills (Brazil or Portugal) was assessed based on different pathways: import biodiesel, soybean oil or grain. A life-cycle inventory and model of soybean biodiesel was implemented, including land-use change (LUC), soybean cultivation, soybean oil extraction, transesterification and transport. The ReCiPe 1.07 method was adopted for the environmental impact assessment. USEtox 1.01 method was also adopted to determine the extent to which the toxicity impacts are influenced by the method applied. A sensitivity analysis of alternative allocation procedures was performed to evaluate the influence on the results. The LC environmental impacts of SME are greatly influenced by the state where soybean is produced. The choice of pathway in which environmental impacts of SME is the lowest is miscellaneous and depends greatly on the allocation approach adopted. The toxicity impacts of SME are greatly influenced by the LCA method applied.

Keywords: land-use change (LUC), life-cycle impact assessment (LCIA), ReCiPe, Roundup Ready® (RR), soybean methyl-ester (SME), USETox.

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[a]Castanheira, É.G., Grisoli, R., Coelho, S., da Silva, G.A., Freire, F., 2014. Life-cycle assessment of soybean-based biodiesel in Europe: comparing grain, oil and biodiesel import from Brazil *(submitted)*.

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APPENDIX II: FULL LIST OF PUBLICATIONS

1. Articles in internationally reviewed scientific journals

1.1. Published

1. Castanheira, É.G., Grisoli, R., Freire, F., Garcilasso, V., Coelho, S., 2014. Environmental sustainability of biodiesel in Brazil. *Energy Policy* 65, 680–691.
http://dx.doi.org/10.1016/j.enpol.2013.09.062 **JCR® impact factor (2012): 2.743**

2. Gülşen, E., Olivetti, E., Malça, J., Castanheira, É.G., Freire, F., Dias, L., Kirchain, R., 2014. Impact of Policy on Greenhouse Gas Emissions and Economics of Biodiesel Production. *Environmental Science & Technology* 48 (13), 7642–7650.
http://dx.doi.org/10.1021/es405410u **JCR® impact factor (2012): 5.257**

3. Castanheira, É.G., Acevedo, H., Freire, F., 2014. Greenhouse gas intensity of palm oil produced in Colombia addressing alternative land use change and fertilization scenarios. *Applied Energy* 114, 958–967.
http://dx.doi.org/10.1016/j.apenergy.2013.09.010 **Citations (Scopus): 1** **JCR® impact factor (2012): 4.781**

4. Castanheira, É.G., Freire, F., 2013. Greenhouse gas assessment of soybean: implications of land use change and different cultivation systems. *Journal of Cleaner Production* 54, 49-60.
http://dx.doi.org/10.1016/j.jclepro.2013.05.026 **Citations (Scopus): 2** **JCR® impact factor (2012): 3.398**

5. Castanheira É.G., Silva, P., 2010. Governance of the emerging biofuel markets in European Union: the Portuguese context. *Global Business and Economics Review* 12, 3:230-251(22).
http://dx.doi.org/10.1504/GBER.2010.034895 **Citations (Scopus): 2**

1.2. Submitted to ISI Journals, in review

6. Castanheira, É.G., Grisoli, R., Coelho, S., da Silva, G.A., Freire, F., 2014. Life-cycle assessment of soybean-based biodiesel in Europe: comparing grain, oil and biodiesel import from Brazil.

7. Castanheira, É.G., Freire, F., 2014. Environmental assessment of palm oil produced in Colombia.

8. Figueiredo, F., Castanheira, É.G., Freire, F., 2014. Life-cycle assessment of irrigated and rainfed sunflower: implications of alternative land use change scenarios.

1.3. LCA research published in articles not directly related to the thesis (written during the time period of PhD)

9. González-García, S., Castanheira, É. G., Dias, A. C., Arroja, L., 2013. Environmental life cycle assessment of a dairy product: the yoghurt. *The International Journal of LCA* 18:4, 796-811.
http://dx.doi.org/10.1007/s11367-012-0522-8 **Citations (Scopus): 2** **JCR® impact factor (2012): 2.773**

10. González-García, S., Castanheira, É. G., Dias, A.C., Arroja, L., 2013. Using Life Cycle Assessment methodology to assess UHT milk production in Portugal. *Science of the Total Environment* 442, 225-234.
http://dx.doi.org/10.1016/j.jclepro.2012.10.010 **Citations (Scopus): 2** **JCR® impact factor (2012): 3.258**

11. González-García, S., Castanheira, É. G., Dias, A.C., Arroja, L., 2013. Environmental performance of a Portuguese mature-cheese-making dairy mill. *Journal of Cleaner Production* 41, 65-73.
http://dx.doi.org/10.1016/j.jclepro.2012.10.010 **Citations (Scopus): 3** **JCR® impact factor (2012): 3.398**

12. Castanheira, É.G., Dias, A.C., Arroja, L., Amaro, R., 2010. The environmental performance of milk production on a typical Portuguese dairy farm. *Agricultural Systems* 103:7, 498-507.
http://dx.doi.org/10.1016/j.agsy.2010.05.004 **Citations (Scopus): 16** **JCR® impact factor (2012): 2.504**

2. Papers at international conferences and symposia

2.1. Published in full in proceedings

1. Castanheira, É.G., Acevedo, H., Freire, F., 2013. Avaliação da intensidade de gases com efeito de estufa do ciclo de vida de óleo de palma produzido na Colômbia. 10ª CNA-XII CNEA, 10ª Conferência Nacional do Ambiente-XII Congresso Nacional de Engenharia do Ambiente, 6-8 Novembro 2013, Aveiro, Portugal *(Oral communication)*.

2. Figueiredo, F., Castanheira, É., Freire, F., 2013. Avaliação de ciclo de vida do óleo de girassol produzido em Portugal. 10ª CNA-XII CNEA, 10ª Conferência Nacional do Ambiente-XII Congresso Nacional de Engenharia do Ambiente, 6-8 Novembro 2013, Aveiro, Portugal *(Oral communication).*

3. Castanheira, É., Freire, F., 2013. Greenhouse Gas Footprint of Palm Oil from Colombia. Energy for Sustainability 2013, Sustainable Cities: Designing for People and the Planet, 8-10 September 2013, Coimbra, Portugal *(Oral communication).*

4. Coelho, S.T., Silva, G. A., Freire, F., Grisoli, R., Castanheira, É., Garcilasso, V.P., Nogueira, A., 2013. A comparative life cycle assessment of biodiesel from soybean oil and beef tallow in Brazil. Energy for Sustainability 2013, Sustainable Cities: Designing for People and the Planet, 8-10 September 2013, Coimbra, Portugal *(Oral communication).*

5. Figueiredo, F., Castanheira, É., Freire, F., 2013. Life cycle assessment of sunflower addressing land use change. Energy for Sustainability 2013, Sustainable Cities: Designing for People and the Planet, 8-10 September 2013, Coimbra, Portugal *(Oral communication).*

6. Malça, J., Castanheira, E., Figueiredo, F., Carvalho, R., Freire, F., 2013. Comparative GHG assessment of biodiesel produced from rapeseed, soybean and sunflower. Energy for Sustainability 2013, Sustainable Cities: Designing for People and the Planet, 8-10 September 2013, Coimbra, Portugal *(Oral communication).*

7. Castanheira, É. G., Freire, F., 2013. Greenhouse gas intensity of palm oil from Colombia: implications of land use change and nitrogen fertilization. International Society for Industrial Ecology's 2013 Conference: Strategy for Green Growth. June 25-28th, Ulsan, South Korea *(Oral communication).*

8. Grisoli, R., Nogueira, A., Castanheira, É. G., Freire, F., Silva, G.A., Coelho, S., 2013. A comparative Life Cycle Assessment of biodiesel from soybean oil and beef tallow in Brazil. International Society for Industrial Ecology's 2013 Conference: Strategy for Green Growth. June 25-28th, Ulsan, South Korea *(Poster).*

9. Queiróz, J., Caldeira, C., Castanheira, É., Freire, F. (2013). Life-cycle energy and GHG assessment of biodiesel from waste cooking oil, addressing alternative collection scenarios. CILCA2013 – V International Conference on Life Cycle Assessment. 24-27th March, Mendonza, Argentina (*Poster*).

10. Castanheira, É.; Freire, F., 2012. System expansion and allocation in the life-cycle GHG assessment of soybean oil. 8th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCAFood 2012). 2-4 October, Saint Malo, France (O*ral communication*).

11. Figueiredo, F., Castanheira, É.; Freire, F., 2012. LCA of sunflower oil addressing alternative land use change scenarios and practices. 8th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCAFood 2012). 2-4 October, Saint Malo, France (*Oral communication*).

12. Grisoli, R., Pecora, V., Lora, B., Guardabassi, P., Nogueira, A., Silva, S.P.R., Freire, A., Castanheira, É. G., Freire, F., Silva, G.A., Coelho, S., 2012. Emissões de gases de efeito de estufa no ciclo de vida do biodiesel de soja produzido no Brasil. III Congresso Brasileiro em Gestão do Ciclo de Vida de Produtos e Serviços "Novos desafios para um planeta sustentável", 03 a 06 de setembro de 2012, Maringá – PR – Brasil *(Poster).*

13. Castanheira, É.G.; Freire, F., 2012. Biodiesel de Soja: Gases com Efeito de Estufa e a Relevância da Alteração de Uso dos Solos. III Congresso Brasileiro em Gestão do Ciclo de Vida de Produtos e Serviços "Novos desafios para um planeta sustentável", 03 a 06 de setembro de 2012, Maringá – PR – Brasil *(Oral communication*).

14. Caldeira, C.; Gülsen, E.; Olivetti, E.; Castanheira, É.; Figueiredo, F.; Malça, J.; Dias, L.; Kirchain, R.; Freire, F., 2012. Capturing Uncertainty in Biofuels for Transportation. Resolving Environmental Performance and Enabling Improved Use. Gordon Research Conference on Industrial Ecology, Les Diablerets, Switerzland, June 18 - 22, 2012 (*Poster*).

15. Castanheira, É.; Freire, F., 2012. Life-cycle GHG assessment of soybean biodiesel. 2012 IEEE ISSST International Symposium on Sustainable Systems and Technology. Boston, USA, May 16-18 2012 *(Oral communication*). doi:10.1109/ISSST.2012.6227977

16. Gulsen, E., Olivetti, E., Kirchain, R., Freire, F., Malça, J., Castanheira, É., 2012. Impact of Blending Practices on Estimated Uncertainty in Biodiesel Lifecycle Carbon Footprint. 2012 IEEE ISSST International Symposium on Sustainable Systems and Technology. Boston, USA, May 16-18 2012 *(Oral communication*).

17. Castanheira, É.; Freire, F., 2011. GHG life-cycle assessment of soybean-based biodiesel: assessing the implications of alternative land use change scenarios. Workshop on land use impact of bioenergy - Quantifying and managing land use impacts of bioenergy. Organized by IEA Bioenergy (Tasks 38, 40 and 43). Campinas, Brazil, 19-21 September 2011 *(Oral communication*).

18. Castanheira, É.; Freire, F., 2011. Environmental life cycle assessment palm oil biodiesel in Colombia addressing alternative land use change and practices. Workshop on land use impact of bioenergy - Quantifying and managing land use impacts of bioenergy. Organized by IEA Bioenergy (Tasks 38, 40 and 43). Campinas, Brazil, 19-21 September 2011 *(Poster).*

19. Castanheira, É.; Freire, F.; 2011. LC GHG Assessment of Soybeans. LCM 2011 - Towards Life Cycle Sustainability Management. Berlin, August 28–31, 2011 *(Oral communication).*

20. Castanheira, É.; Freire, F., 2011. Life cycle assessment of palm oil biodiesel addressing land use and land use change. ECOS2011, the 24th International Conference on Efficiency, Cost, Optimization, Simulation and Environmental Impact of Energy Systems. Novi Sad, Serbia, July 4–7, 2011 *(Oral communication).*

21. Castanheira, É.; Freire, F., 2011. Environmental performance of palm oil biodiesel – a life-cycle perspective. 2011 IEEE ISSST International Symposium on Sustainable Systems and Technology. Chicago, Illinois, USA, May 16-18 2011 *(Oral communication).* doi:10.1109/ISSST.2011.5936843

22. Castanheira É.; Freire, F., 2010. CO_2 and N_2O emissions from palm oil plantations: impacts of land use and land use change. 7th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCAFood 2010). 22-24 September, Bari, Italy *(Oral communication).*

23. Castanheira, É.; Freire, F., 2010. GHG Emissions Assessment of Palm Oil: uncertainty and scenario analysis for increased production. International Conference "Uncertainty and Robustness for Planning and Decision Making" (URPDM2010), 15-17 April Coimbra, Portugal *(Oral communication).*

24. Castanheira, É.G.; Freire, F. 2009. Life cycle modeling addressing direct land use changes: the palm oil biodiesel chain. International Society for Industrial Ecology's 2009 Conference, June 21-24th, Lisbon, Portugal *(Poster).*

3. Technical reports

1. Castanheira, É.G.; Freire, F., 2013. Cálculo das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja na Prio. Relatório elaborado para a Prio Biocombustíveis S.A. - Grupo Martifer no âmbito do sistema de certificação ISCC - International Sustainability & Carbon Certification. Coimbra, Fevereiro de 2013.

2. Castanheira, É.G.; Freire, F., 2012. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja na Iberol. Relatório Específico (Confidencial) elaborado para a Iberol – Sociedade Ibérica de Biocombustíveis e Oleaginosas, S.A., no âmbito do protocolo de colaboração entre a APPB e a ADAI. Coimbra, Março de 2012.

3. Castanheira, É.G.; Freire, F., 2012. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja na Biovegetal. Relatório Específico (Confidencial) elaborado para a Biovegetal – Combustíveis Biológicos e Vegetais, S.A. - Grupo SGC, no âmbito do protocolo de colaboração entre a APPB e a ADAI. Coimbra, Janeiro de 2012.

4. Castanheira, É.G.; Freire, F., 2012. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja na Prio. Relatório Específico (Confidencial) elaborado para a Prio Biocombustíveis S.A. - Grupo Martifer, no âmbito do protocolo de colaboração entre a APPB e a ADAI. Coimbra, Janeiro de 2012.

5. Castanheira, É.G.; Freire, F., 2012. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja na Sovena. Relatório Específico (Confidencial) elaborado para a Sovena Oilseeds Portugal, S.A. -Grupo Nutrinveste, no âmbito do protocolo de colaboração entre a APPB e a ADAI. Coimbra, Janeiro de 2012.

6. Castanheira, É.G.; Freire, F., 2012. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja na Torrejana. Relatório Específico (Confidencial) elaborado para a Fábrica Torrejana de Biocombustíveis, S.A., no âmbito do protocolo de colaboração entre a APPB e a ADAI. Coimbra, Janeiro de 2012.

7. Castanheira, É.G.; Freire, F., 2011. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja em Portugal. Relatório Final elaborado para a APPB, no âmbito do Protocolo de colaboração entre a APPB e a ADAI. Coimbra, Dezembro de 2011.

8. Castanheira, É.G.; Freire, F., 2011. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja em Portugal. Relatório Intermédio – Parte II elaborado para a APPB, no âmbito do Protocolo de colaboração entre a APPB e a ADAI. Coimbra, Dezembro de 2011.

9. Castanheira, É.G.; Freire, F., 2011. Avaliação de Ciclo de Vida das emissões de Gases com Efeito de Estufa da Produção de Biodiesel de Soja em Portugal. Relatório Intermédio – Parte I elaborado para a APPB, no âmbito do Protocolo de colaboração entre a APPB e a ADAI. Coimbra, Dezembro de 2011.