Environmental, Social and Economic Sustainability of Biobased Plastics. Bio-polyethylene from Brazil and polylactic acid from the U.S.

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PREFACE

This research was performed at the Centre for Energy and Environmental studies (IVEM) at the University of Groningen. The research period at the IVEM was enriching for both my personal and academic development. During five hectic months of research, I gained hands on experience with conducting scientific research and performing under great time pressure. The semi-independent character of the learning thesis allowed me to follow my own interests and methods while still being under expert guidance. On a social level, I enjoyed the interaction with staff members and students of IVEM. I believe this combination of science and social life at the IVEM is a sound basis for my scientific education.

The research was performed under supervision of prof.dr. Ton Schoot Uiterkamp and dr. Cindy Visser. I want to thank Cindy Visser for her constructive criticism and her contributions on the structure and quality of the report. I want to thank Ton Schoot Uiterkamp for his guidance in the research, for sharing his knowledge and for the always entertaining and educational anecdotes. A special thanks to Ton, who enabled me to continue my scientific career in the United States. I also want to thank the people who helped me with specific problems; dr.ir. Sanderine Nonhebel on the greenhouse gas balance, dr. René Benders on Simapro and prof.ir. Michel Boesten, dr. ir. Gerald Jonker and dr. ir. Erik Heeres on the economic aspects. Finally, I want to thank Tiemen Folkers, who was of tremendous help when the used reference program decided to break down.

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Abbreviations and definitions

Several names and concepts are used on a frequent basis throughout the report. A definition of the used abbreviations is given below.

Bio-supply chain:	The supply chain of biobased products
Fossil-supply chain:	The supply chain of fossil-based products
Fossil-PE:	Polyethylene based on crude oil as feedstock
Fossil-PE supply chain:	Supply chain of fossil-based polyethylene
Bio-PE:	Polyethylene based on sugarcane as feedstock
Bio-PE supply chain:	Supply chain of biobased polyethylene
PLA:	Polylactic acid
PLA supply chain:	Supply chain of polylactic acid
GHG emissions:	Greenhouse gas emissions

SUMMARY

Ever depleting fossil resources, growing fossil feedstock prices and global environmental impact associated with continuously rising greenhouse gas emissions have led to increased attention for biobased products as alternatives for the present fossil-based ones. It is therefore important for scientists and academics to provide knowledge and explore practical routes for the sustainable transition towards biobased alternatives. This research focuses on the replacement of fossil-based polyethylene (fossil-PE) by biobased PE (bio-PE) from Brazilian sugarcane and polylactic acid (PLA) from US corn. The report gives an outlook on such a shift towards a sustainable biobased economy. The aim of this research is to assess the characteristics of the biobased product supply chains which could provide in the demand for biobased polyethylene and polylactic acid for the Netherlands on environmental, social and economic aspects in order to contribute to the existing fundamental research on biobased products. This led to the main research question:

How sustainable are the supply chains for biobased polyethylene and polylactic acid, which could meet the current demand for fossil-based polyethylene in the Netherlands, and how does this compare to the supply chain of fossil-based polyethylene?

An analytical framework is developed along which the three dimensions of sustainability can be evaluated; environmental (greenhouse gas emissions, biodiversity and the local environment), social (competition for food, welfare and wellbeing) and economic sustainability (market price). The report focuses on the demand for bio-PE and PLA which could replace the current Dutch demand for PE of approximately 500 kiloton per year.

The report shows that on environmental sustainability, bio-PE outperforms fossil-PE and PLA. Life cycle greenhouse gas emissions are particularly low for bio-PE due to the extensive use of bagasse as energy supply. PLA associated greenhouse gas emissions are slightly less than the greenhouse gas emissions for fossil-PE. Depending on the type of land that is converted to biomass feedstock, greenhouse gas emissions can increase due to the release of carbon from decaying biomass and the loss of soil organic carbon. This effect can be significant if rainforest is converted either by direct or indirect land use change. Even so, considering the relatively small demand, enough land is available in Brazil for the production of the biobased products without endangering bio-diverse regions. The main impact of biobased products on the local environment is the imbalance of NPK nutrients and for fossil-based products on- and offshore oil spills.

The main concern with regard to the social sustainability was found in the exploitation of sugarcane and corn field workers. Case reports were found on slavery and exploitation, although no structural proof was found. For fossil-PE, decrease of the local welfare and wellbeing was found for several countries producing naphtha. Competition for food was considered as one of the main indicators. It is found that there is no competition for food if only the Dutch demand is considered, but a worldwide demand for multiple biobased products would inevitably lead to competition for food. This stresses the importance of alternative biomass sources that do not impact food supply, such as lignocelluloses.

Even though bio-PE shows more favourable results than fossil-PE, biobased products are still unsustainable due to the high market price with respect to the biobased product. With current feedstock prices, market prices for bio-PE and PLA are respectively 40% to 60% more expensive than fossil-PE. The price imbalance can be partially explained by the fact that the costs of environmental degradation are externalized for fossil-based products. Internalizing these externalities by for instance green VAT or carbon tax would level the playing field for biobased products. A reduced green vat of 6% for "green" products (19% for normal products) would reduce the difference to 25% and 40% for bio-PE and PLA respectively. Additionally implementing a carbon tax of 50 USD/t CO_2 would reduce the difference even further to 15% and 35% for bio-PE and PLA respectively.

1. INTRODUCTION

Depleting fossil reserves worldwide give rise to resource scarcity and increasing costs for all economic sectors using fossil materials. Driven by environmental concerns and rising costs, more and more attention is paid worldwide to alternatives for the use of fossil materials. This search for alternatives is part of the concept of a biobased economy, *i.e. an economy in which green materials instead of fossil-based materials are used for the production of energy, transport fuels, chemicals and other materials* (Commissie Duurzame Ontwikkeling., 2010). Biofuels receive most attention in this concept (Shen *et al.*, 2009), but current views state that biobased materials which replace fossil materials might be more interesting for both the environment and the economy (Platform Groene Grondstoffen., 2007; Wetenschappelijke en Technologische Commissie voor de Biobased Economy., 2011). This view is based on the concept of cascading, where biomass feedstock would first be used for the highest-value product, after which the remaining feedstock is used for the lower value products. If the value pyramid for biobased products is taken into account (figure [1.1]), biofuels are the bottom layer, meaning high volume and low added value. Chemicals from biomass feedstock are higher up the value pyramid and still have considerable volume. The project group "platform groene grondstoffen" (platform green resources) calculated that the most interesting sector for replacing fossil materials is the (bulk) chemical sector. First of all because pharmaceuticals and fine chemicals are more difficult to replace and therefore have higher initial investment costs. Second, because of its economic potential; (Bulk) Chemical production can lead to profits at ca. € 30/GJ, while only low profits can be gained from electricity production (ca. € 6/GJ) and transport fuels production (€ 8/GJ). (platform groene grondstoffen, 2007).

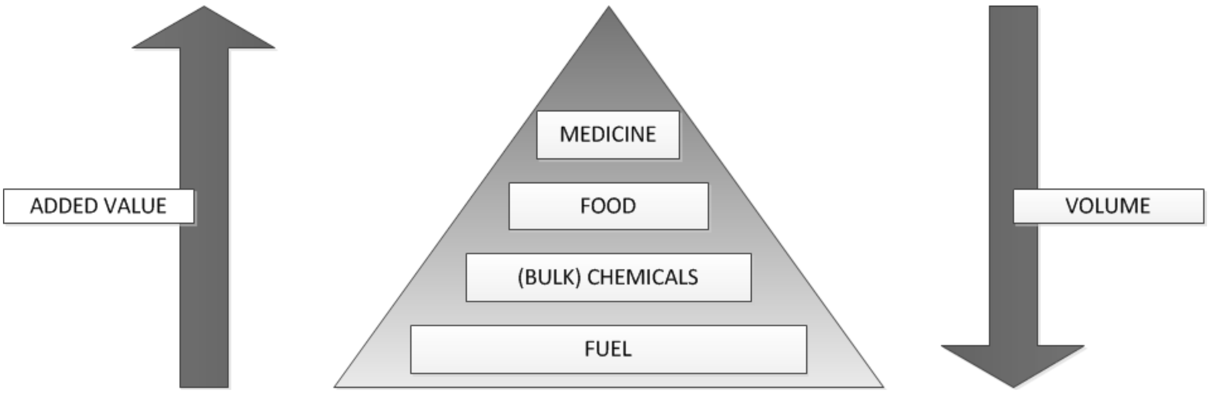


Figure [1.1] Value pyramid for biobased products

Developments

In light of current views on biobased products, the Dutch government is putting more and more emphasis on the replacement of fossil materials by biobased materials (Asveld *et al.*, 2011). The Dutch petrochemical sector itself aims to replace as much as 25% of its fossil feedstock by biomass feedstock by the year 2030 (Regiegroep Chemie., 2006). The sector states that this aim is driven by the increasing market demand for greener products. The demand for biobased plastics is expected to show a >20% annual growth rate (Shen *et al.*, 2009). Adding that fossil-based materials are being depleted makes it clear that a shift towards biobased products is imminent, if not now than certainly in the near future. The significance of chemicals in this transition is highlighted by the fact that 20% of all fossil materials used in the Netherlands are used by the chemical sector; 8% is used in direct energy use, and as much as 12% is used in end products (Platform groene grondstoffen, 2007).

Sustainability

The increase in demand spurs the increase in capacity. This in turn raises implications for the supply chain. Shifting from a fossil-based material supply chain to a biobased material supply chain means that production location, refining, storage, and transport must be redefined. This restructuring process leads to concerns on sustainability. Choices have to be made on logistic issues not only from an economical and technical point of view, but also from a social and environmental point of view. Destruction of habitat, depletion of minerals, local welfare and social welfare concerns are just some of the potential problems which might arise (Asveld *et al.*, 2011; Wetenschappelijke en Technologische Commissie voor de Biobased Economy., 2011; Projectgroep Duurzame Productie van Biomassa., 2006). Economic growth often prevails over environmental sustainability in developing countries, increasing the importance of sustainability frameworks (Dyllick & Hockerts, 2002). Certain sustainability criteria must be met for imported products to protect and preserve the local environment and welfare (Projectgroep Duurzame Productie van Biomassa., 2006), but also to reduce dependency on depleting resources (Wetenschappelijke en Technologische Commissie voor de Biobased Economy., 2011). These criteria must envelop all aspects of sustainability to prevent environmental and social damage.

1.1 Research aim and research questions

Biobased product supply chains play a key role in the transition towards a biobased economy. Current models mainly evaluate the economic optimization of biomass supply chains (Bowling *et al.*, 2011; Carolan *et al.*, 2007; Dornburg *et al.*, 2006; Krishnakumar & Ileleji, 2010). This is just one of the relevant factors associated with a sustainable supply chain. This research proposes an overall evaluation of a biomass supply chain with a special focus on sustainability. To limit the scope of the research, the Netherlands are chosen for market size, and fossil-based polyethylene and its' biobased substitutes are chosen for products. 29% of the volume of all the plastic produced is either low density (LDPE), linear low density polyethylene (LLDPE) or high density polyethylene (HDPE), making it the plastic with the highest demand worldwide (PlasticEurope., 2010). Replacing it by biobased materials requires a restructured biomass-to-product chain which could handle the large demand volume for polyethylene (Wetenschappelijke en Technologische Commissie voor de Biobased Economy., 2011). Because of this, replacing fossil-based polyethylene can be seen as a representation of a shift towards a biobased economy. Research into possible replacements reveals biobased polyethylene (non-degradable) and polylactic acid (degradable) as having a high potential for replacing fossil-based polyethylene (Shen *et al.*, 2009). Both products are already being marketed as a potential replacement, and demand for both is expected to grow considerably in the coming years (Shen *et al.*, 2009). This leads to the following research aim:

> *The research aim is to assess the characteristics of the biobased product supply chains which could meet in the demand for biobased polyethylene and polylactic acid for the Netherlands on economic, social and environmental aspects. This will contribute to the existing fundamental research on biobased products and especially on its supply chain aspects.*

In order to perform this research, a main research question and four sub-questions are defined.

Main research question:

> *How sustainable are the supply chains for biobased polyethylene and polylactic acid, which could meet the current demand for fossil-based polyethylene in the Netherlands, and how does this compare to the supply chain of fossil-based polyethylene?*

Sub questions:

1) Which general configuration can be identified for a biobased supply chain?

2) How can the sustainability of biobased supply chains be represented in an analytical framework with environmental, social and economic indicators?

3) What are the specific configurations and what are the specific characteristics of the supply chains of fossil-based polyethylene, biobased polyethylene and polylactic acid? How do they compare?

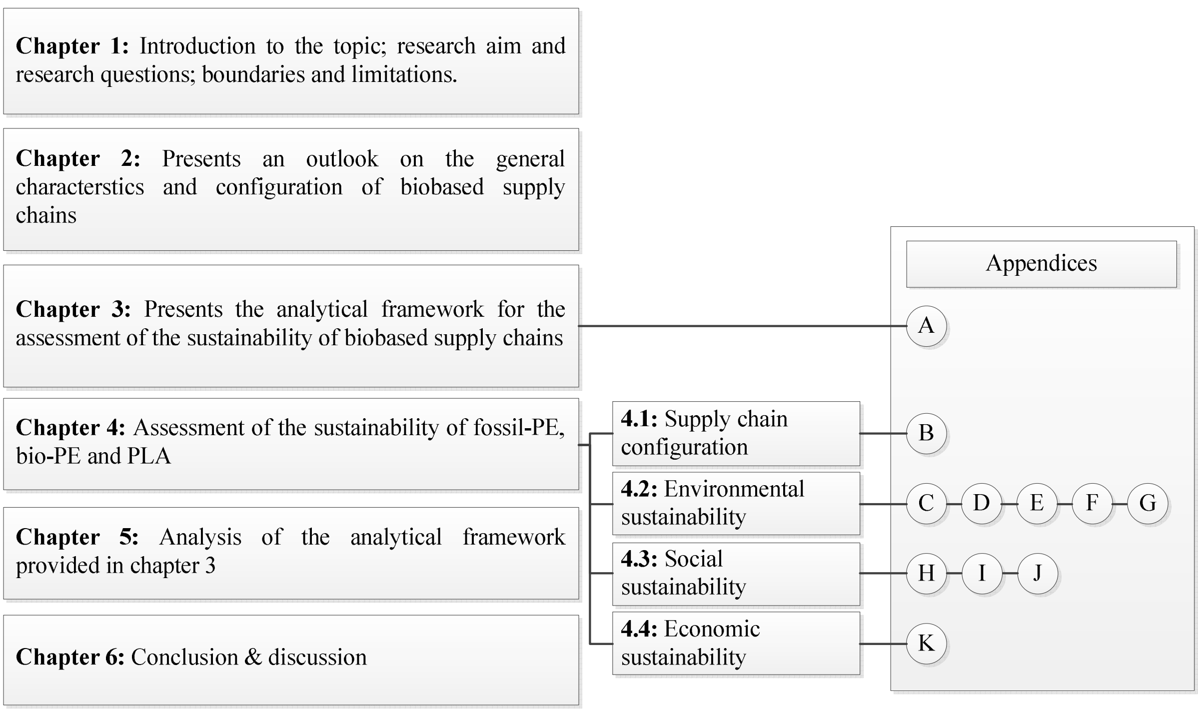
4) How can the framework proposed in sub-question 2 be improved using the case studies on biobased polyethylene and polylactic acid?

1.2 System boundaries and limitations

This research is characterized by two important boundaries. First, the research provides a theoretical framework aimed to evaluate the supply chain characteristics for different materials. By doing so, this research contributes to an overall assessment of replacing fossil-based materials by biobased materials. Second, the research focuses on the actual demand for the fossil product, as well as on a biobased product which has a demand-driven potential of replacing the fossil product. This means that the biobased product should not only have the technical potential to replace the fossil-based product, but should also be in demand. Such as is the case with biobased polyethylene and polylactic acid. This approach ensures that the research is not merely hypothetical. As stated in the research aim, the Netherlands are chosen to determine market size. Fossil-based polyethylene, biobased polyethylene and polylactic acid are chosen as products.

1.3 Reader's guide

The reader's guide provides the reader with a view on the subsequent chapters. Each chapter is briefly described.



2. GENERAL CONFIGURATION AND CHARACTERISTICS OF A BIOBASED SUPPLY CHAIN

Chapter 2 gives an overview of the general characteristics of a biobased product supply chain (referred to as “bio-supply chain” from here on). Paragraph 1 describes some general characteristics of a bio-supply chain. Paragraph 2 gives a methodology aimed at identifying the optimal configuration for a specific bio-supply chain.

2.1 Supply chain processes

Figure [2.1] shows a typical configuration of a bio-supply chain. The supply chain starts with local feedstock production (e.g. Sugarcane). The feedstock is then transported from the production site to the collecting facility. From these it is transported to the production facility where it is pre-treated and processed to the final product (e.g. ethanol). Depending on the final product, more production facilities can be involved. These facilities process the product to higher value products (e.g. ethylene). Note that this is a general configuration. Depending on the feedstock, the configuration might be different.

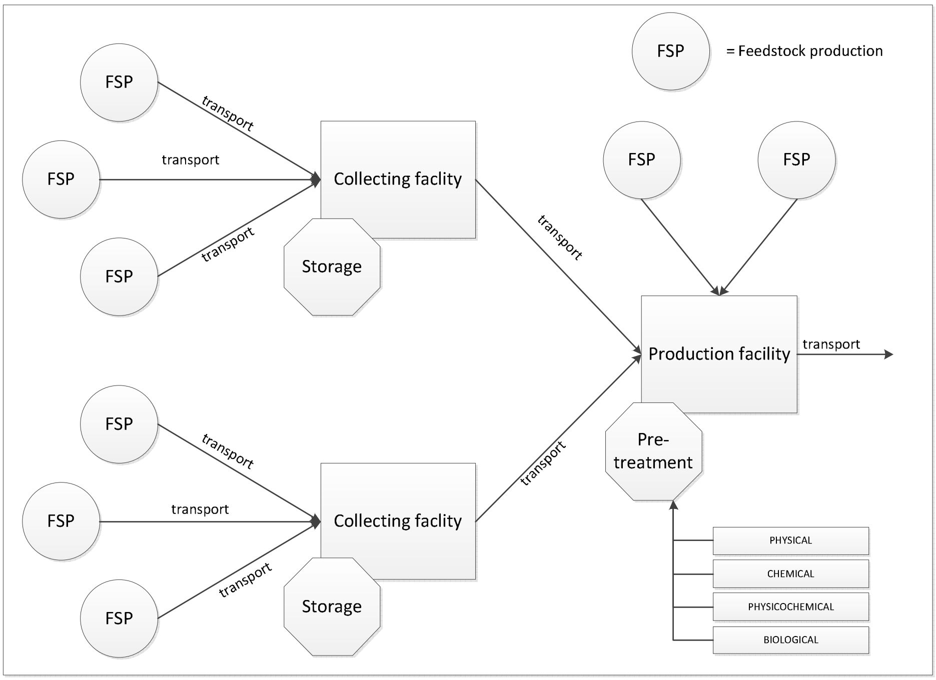


Figure [2.1] General configuration of a bio-supply chain

Current production of biomass is generally characterized by small quantities scattered over many sources and location (Annevelink & de Mol, 2005). This means that biomass production requires a large area for collection, variation in crop maturity with time and weather, a short collection window, and competition from concurrent harvest operations (Kumar *et al.*, 2006). Therefore, storage of biomass is needed to ensure a constant supply of biomass on an annual basis. The site where biomass is stored varies per biomass type and is subject to several variables like costs and pre-treatment options. There are three options for locating storage; on-field storage, intermediate storage between field and refinery, and storage at refining plant (Rentizelas *et al.*, 2009). Biomass is harvested from the production sites and transported to a regional collection facility. Because of the scattered nature of biomass production, adequate collection systems need to be in place to ensure an efficient supply chain for biomass. From these collection facilities, biomass is transported to the production facilities. Bio-supply chains are typically constructed with pre-processing and pre-treatment on-site at the production facility (Koukios *et al.*, 2010; Carolan *et al.*, 2007). During the pre-treatment phase the biomass is prepared for further production. Pre-treatment methods can be divided into four different categories; physical methods (e.g. chipping, milling, grinding and irradiation), chemical methods (e.g. using alkali, dilute acid, oxidizing agents and organic solvents), physicochemical methods (e.g. steam pre-treatment/autohydrolysis, hydrothermolysis, ammonia fibre explosion and wet oxidation) and biological methods (e.g. lignin degrading micro-organism) (Galbe & Zacchi, 2007; Mosier *et al.*, 2005). In the production facility the pre-treated biomass is converted into end-product or semi-end-product. Production facilities typically include the following processes: bio-feedstock handling and storage; biomass pre-treatment; biomass fractionation to main and co-products; downstream processing of primary and intermediate outputs; product and co-product upgrading; product and co-product marketing and integrated material/energy/economic flows (Koukios *et al.*, 2010). As stated, specific supply chains might differ from the general configuration.

2.2 Supply chain configuration

To assess the configuration of a specific Bio-supply chain, the following four steps are proposed which determine size, type and location of the biomass production.

1) Determine the demand for the fossil-based product in the Netherlands
2) Determine the type of biomass suitable for production of the biobased substitute
3) Determine the region and location of biomass production
4) Determine the bio-supply chains' configuration and geographical lay-out

The first step to be performed is to assess the demand for the fossil-based product that the biobased product aims to replace. In this research, fossil-PE will be investigated. With the determination of the demand, a target is set for the size of the bio-supply chain. As previously mentioned, this approach ensures that the research is not merely hypothetical. The next step is to determine which biomass is most suitable for the production of the biobased substitute. Not only technical, but also economic and environmental factors must be taken into account when assessing if a certain type of biomass is suitable. Again, the potential biobased product substitute should be subject to a certain demand. In this research, both bio-PE and PLA have proven to be in demand to some extent. When the biomass type is determined, the location or region for production is assessed. Technical, environmental, economic and social factors will be different for each region of production. To assess an optimal region, a quick scan will be performed to determine the most suitable region. The fourth step is to assess the configuration and geographical lay-out of the bio-supply chain using the following approaches:

- Literature survey on the best available techniques and optimum configuration
- Evaluate local characteristics of the region chosen
- Calculate total land use

By doing so, the theoretical best practice is tested against the actual situation of the location of biomass origin. The total land use will be calculated by determining a) biomass yield per hectare and b) needed amount of biomass per kg of biobased product.

Summarizing, this chapter provided a general view on biobased supply chains and a four-step template on the configuration of biobased product supply chains. The template is used in chapter four for the evaluation of the supply chains.

3. FRAMEWORK REPRESENTATIVE FOR A SUSTAINABLE BIOBASED SUPPLY CHAIN

Chapter 3 provides an analytical framework along which specific bio-supply chains can be analysed. Furthermore, a set of indicators and boundaries is provided which are used to evaluate the three dimensions of sustainability; environmental, social and economic sustainability. Finally, some limitations of the framework are given.

3.1 The analytical framework

Figure [3.1] shows the analytical framework along which the specific bio-supply chains will be analysed. The iterative process of constructing the framework is shown in appendix A. Figure [3.1] shows the three dimensions of sustainability as intrinsic aspects of the bio-supply chain. The dimensions are overlapping to indicate that they are interconnected. The focus of this research is on the bio-supply chain and its sustainability dimensions. Policies, market forces and other influences co-determine certain aspects of the bio-supply chain. This is represented graphically in figure [3.1].

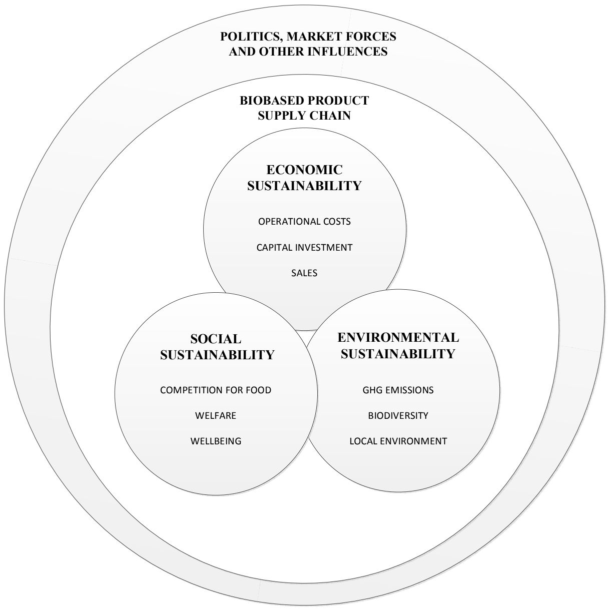


Figure [3.1] Analytical framework including economic, environmental and social sustainability aspects and policies, market forces and other influences

3.2 Indicators/boundaries

Table [3.1] gives the indicators and boundaries which will be used to evaluate the specific supply chains. The indicator/boundary set was constructed by comparing the NTA8080, ISCC and RSB certification programs for sustainable production of biomass (ISCC Association, 2010; NEN, 2009; RSB, 2010). Appendix A elaborates on how the indicator/boundary set was determined. The aspect "policies, market forces and other influences" is very broad. Focus will be put on the institutional character and regulatory framework to limit the extent. Where needed, policies and regulations are discussed along with the environmental, social and economic sustainability analysis. A separate paragraph (4.5) is devoted to possible policies for the Netherlands

Table [3.1] Indicators/boundaries for the assessment of the environmental, social and economic sustainability of supply chains

Environmental aspects	Indicator/boundary
Greenhouse gas emissions	• Net reduction of 50% life cycle GHG emissions compared to the fossil reference • Biomass is not produced on land with high carbon stock: a) Wetlands b) Continuously forested land c) Peat bog
Biodiversity	• No biomass production in designated protected areas or within 5 km of these areas • No biomass production in areas with high conservation value (HCV) or within 5 km of those areas. • 10% of the functional land use area should be used for original vegetation representative for the area.
Local environment	• Compliance with the Stockholm convention on persistent organic pollutants • Soil organic matter is maintained/preserved • Maintenance of soil nutrients balance concerning nitrogen (N), phosphorus (P) and potassium (K) • Burning as part of cultivation is restricted. • No run-off of applied fertilizer to surface water • Measurements to prevent soil erosion should be adapted
Social aspects	Indicator/boundary
Competition for food (and local use of biomass)	• No reduction of available biomass for food, local energy supply, medicine and building materials. Analysed by: a) Nature of biomass feedstock b) Production location c) Land use surface d) Development of land use, food availability and prices of land and food.
Welfare	• Stimulating local welfare defined by reporting on Global Reporting Initiative (GRI) EC1,EC6, EC7(GRI, 2000) EC1: Generated and distributed economic value like operational costs, capital investment, voluntary payments to community EC6: Supporting local businesses by attracting local providers of materials, products, and services that are based in the same geographic market as the reporting organization. EC7: Hiring local residents for management to support local community and the organization's ability to understand local needs measured by local resident manager percentage.
Wellbeing	• Compliance with: a) Tripartite declaration of principles concerning multinational enterprises and social policy b) Universal declaration of human rights c) Legal ownership or law of custom • Contribution must be made to wellbeing local inhabitants • Global reporting initiative (GRI) SO2, SO3, SO4 SO2: reduce corruption risk, either by analysing business units for corruption, or including a formal percentage for corruption in the risk assessment SO3: Anti-corruption training for management and non-management, expressed in percentages. SO4: Report on corruption incidents and disciplining
Economic aspects	Indicator/boundary
Sales	• No large increase in market price compared to the fossil reference
Total capital investment	• Facility investment • Machinery investment • Land value

Operational costs	• Man-hours • Fuel use • Power requirement • Input materials • Operation & maintenance • Governmental payments • Insurance

[a] For a detailed description of carbon stock land see ISCC criteria (ISCC Association, 2010)

3.3 Framework limitations

The proposed framework is intended to evaluate the main sustainability aspects of bio-supply chains. This includes economic sustainability, which is a market-orientated evaluation. The framework is limited since it leaves out profit and revenues.

Also, data might be unavailable for several of the indicators. Short reports will be given as small proxies on those indicators for which data is unavailable. This will be especially true for social sustainability indicators. Since the supply chains that are evaluated might be partially non-existent at the time of writing, social sustainability status can only be determined on the basis of the situation in the country where the research is performed.

The last limitation is that the framework is not an optimization framework. Optimization models do exist, mainly on the basis of economic evaluations (Bowling *et al.*, 2011; Carolan *et al.*, 2007; Dornburg *et al.*, 2006; Krishnakumar & Ileleji, 2010). The goal of this framework is to integrate environmental, social, economic and political aspects. This will give a holistic view of the aspects, in contrast to the general optimization models.

3.4 Concluding remarks

Note that the full explanation of the analytical framework is presented in appendix A. The framework and indicator/boundary set the basis for the sustainability evaluation of the bio-supply chains in chapter 4. Chapter 5 gives an evaluation of the sufficiency of the used indicators/boundaries for the determination of sustainability.

4. SUSTAINABILITY ANALYSES OF THE FOSSIL-BASED POLYETHYLENE, BIOBASED POLYETHYLENE AND POLYLACTIC ACID SUPPLY CHAINS

Chapter 4 analyses the supply chains of fossil-PE, bio-PE and PLA. During the research, the main focus was put on the sustainability of the bio-PE supply chain. Information on the bio-PE supply chain will therefore be more detailed than information on the PLA supply chain. This chapter discusses the supply chain in general and discusses the results on environmental, social and economic sustainability. Furthermore, some policies surrounding biobased products are discussed. The chapter concludes with some key findings.

4.1 Supply chains

This paragraph gives an overview of the production and supply chains of fossil-PE, bio-PE and PLA. The discussion on the fossil-PE supply chain will conclude with an estimate of the Dutch demand for polyethylene. This demand will be used to evaluate the biobased supply chains on the sustainability dimensions.

Fossil-based polyethylene

Two main feedstock routes exist for the production of fossil-PE; natural gas (main feedstock United States) and naphtha (main feedstock Europe). Since this study focuses on the demand for polyethylene in the Netherlands, only naphtha is discussed. Information on the naphtha supply chain and refining is derived from "the chemistry and technology of petroleum" (Speight, 1999).

Naphtha is a complex mixture of hydrocarbons. In petroleum engineering *full range naphtha* refers to a hydrocarbon mixture with a boiling point between 30°C and 200°C. The naphtha production process begins with the extraction of crude oil from wells. Extraction is carried out in three phases; primary recovery, secondary recovery and enhanced recovery. During primary recovery, natural pressure in the well is used to extract the crude oil. During secondary extraction, pressure is added through injection wells to drive the crude oil to the production well. Enhanced recovery involves specialized techniques to extract the remaining crude oil. After recovery, the crude oil is pre-treated for transport. Pre-treatment involves primarily degasification, water extraction and the removal of impurities like sand. Crude oil is mainly transported through pipelines and ocean tankers, and occasionally by road transport. At the refinery site, crude oil is stored in special (cylindrical) tanks (tank farm) or in salt cavities, former coal mines or artificial caverns. During refinery, the crude oil for naphtha first enters a separation process, after which it is fed into a catalytic cracker which produces the naphtha. Naphtha in turn is then transported to an ethylene plant where it undergoes a process called pyrolysis or steam cracking. Typically 3.17 tonne of naphtha is used to produce 1 tonne of ethylene. The remaining fraction of the naphtha is processed to other petrochemical products.

At the refining site, the naphtha is refined into various products. Most of the worldwide annual commercial production of ethylene is based on thermal cracking, also called pyrolysis or steam cracking. Steam cracking is typically done at a temperature of 850°C. Typical conversion ranges of commercial furnaces are 60–70%. Much of the ethylene is consumed locally, and otherwise transported to polymerization facilities using mainly pipelines. The pipelines are usually pressurized between 4 and 100 MPa. Pipelines are often kept above 4 °C to prevent liquid ethylene from forming. Ethylene is transformed into fossil-PE by polymerisation, after which it is ready for the plastic industry. The actual demand for polyethylene in the Netherlands is difficult to assess using open literature. Exact data can be found in market reports, but these are costly and are therefore not used for this research. Estimation on demand in the Netherlands is made by comparing total plastic demand in the Netherlands (1653 kt plastic in 2009) with typical polyethylene market share; 29%, combined demand for HDPE, LDPE and LLDPE (PlasticEurope., 2010). This results in a total demand of ~500kt/yr (481.11kt in 2009) for (unprocessed) polyethylene in the Netherlands.

Biobased polyethylene

To assess the impact of meeting the demand for polyethylene in the Netherlands, a plant is considered with an annual capacity of 500 kiloton. For bio-PE, sugarcane was chosen as most suitable for this research because 1) bio-PE from sugarcane is commercially available and 2) a quick scan of the environmental impact of sugarcane showed that it might be the most environmentally friendliest feedstock. The Braskem plant is currently the only commercial scaled plant producing bio-PE from sugarcane. The supply chain configuration is therefore based on the Braskem bio-PE plant in Brazil. The bio-PE supply chain starts with the cultivation of sugarcane. Farms which cultivate sugarcane range between 10,000 and 45,000 ha per farm. The harvest season in the central south (mainly São Paulo), where 80-85% of all Brazilian sugarcane is produced (figure [4.1]), begins in May and ends in November or December (Laluce, 1991). Yield per hectare per year varies for different regions from 60 ton/ha/year till 100 ton/ha/year, with an average of 68.7 ton/ha/year (Macedo, 2004). Sugarcane harvesting in the State of São Paulo is carried out on average 63.8% manually and 36.2% with machines. Pre-harvest burning is applied on 75% of the total area (Ometto *et al.*, 2009).

Figure [4.1] Sugarcane production in Brazil as occupation percentage of the municipality. Source: Sparovek *et al.* (2007)

Partially burned tops and stalks are left on the field after sugarcane is harvested. Burning has substantial effects on human health and on emissions to air, and is therefore phased out under Brazilian legislation (Macedo *et al.*, 2008). After harvest, the sugarcane is transported by truck to a sugarcane mill, which is on average 20-23 km from the sugarcane fields (Brehmer & Sanders, 2009; Macedo, 2004). In the sugarcane mill, sugarcane is processed to ethanol. The sugarcane is first cleaned, after which the sugar is extracted. The sugar juice is processed and subsequently the juice is fermented with yeast to so-called wine. After fermentation, the wine goes through a distillation process, from which ethanol is produced (Dias *et al.*, 2011). A graphical overview of the process is given in appendix B. The ethanol is transported to the Braskem bio-ethylene facility in the state of Rio Grande del Sul. Ethylene is produced through a process of dehydration. The ethylene produced is used as feedstock for the polymerization plant which was already located there for polymerizing petrochemically produced ethylene. Since petrochemically produced ethylene and biobased ethylene are essentially the same molecules, the polymerization phase is the same. The bio-ethylene is therefore integrated into the existing fossil-PE supply chain (Braskem, 2012). From this point, bio-PE is ready to be shipped to the Netherlands.

Polylactic acid

As with bio-PE, a plant is considered with an annual capacity of 500 kiloton, which matches the demand for polyethylene in the Netherlands. For PLA feedstock, corn was chosen as most suitable for this research. This choice was based on available literature on corn for PLA. The biggest producer of PLA today is Naturework LLC in the United states. The supply chain configuration is therefore based on the PLA plant of Naturework LLC. Natureworks uses mainly corn as feedstock for the production of PLA (Vink *et al.*, 2003). The Naturework manufacturing facility is located in Nebraska, which is situated in the Corn belt. Figure [4.2] shows the production of corn in the United States. Corn is harvested and transported to wet mills. In the wet mills, the starch is separated from other components. The starch is converted into dextrose by hydrolysis. At the Natureworks facility, dextrose is fermented into lactic acid. The monomer can be polymerized in one of two ways; direct condensation polymerization of lactic acid or ring-opening polymerization through the lactide intermediate. Natureworks uses the second process (Vink *et al.*, 2003).

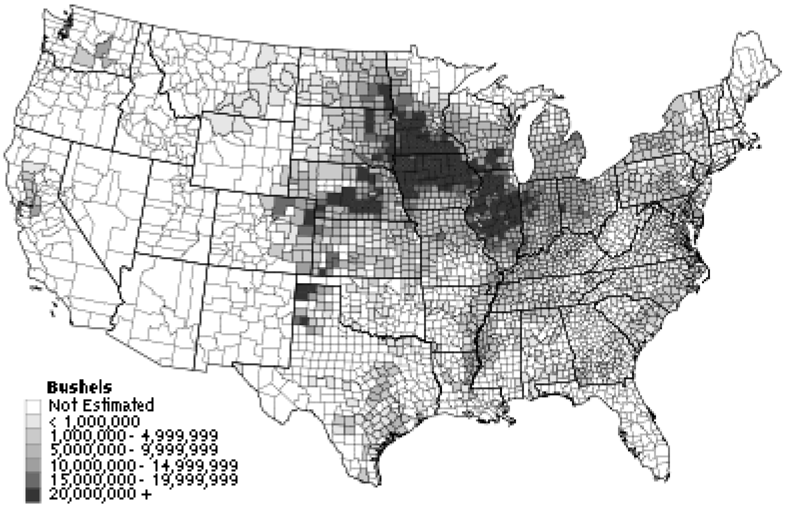


Figure [4.2] Corn production in the United States in bushels (~39 bushels corn/tonne corn). Source: U.S. departement of agriculture. National agricultural statistic service (http://www.nass.usda.gov/)

4.2 Environmental sustainability

This paragraph compares the environmental sustainability of the fossil-PE, bio-PE and PLA supply chains described above. The three different aspects, GHG emissions, biodiversity and the local environment are compared.

Greenhouse gas emissions

The first aspect of environmental sustainability is GHG emissions and the related matter of energy requirements. They will be discussed separately. GHG emissions and energy requirements are analysed by a cradle-to-grave approach, with full incineration as end-of-life scenario. The analysis is based on low-density polyethylene (LDPE) since crucial data was only found on low-density polyethylene and not on high-density polyethylene (HDPE) or linear low-density polyethylene (LLDPE).

Greenhouse gases – GHG emissions are evaluated on CO_2, CH_4 and N_2O and converted to CO_2-equivalent/kg product. The results for fossil-PE, bio-PE and PLA are given in figure [4.3]. This research focuses on short cyclic impacts on the CO_2-balance only. This excludes the uptake of CO_2 by plants which over time became fossil materials (e.g. long cyclic CO_2). This uptake of CO_2 occurred in a pre-industrialized era, with no (or very little) anthropomorphic influences. The uptake of this CO_2 has no longer effect on the current CO_2-levels. The short cyclic CO_2 uptake by plants for biobased products is taken into account, since this does have an impact on the current imbalance of CO_2-levels. An extensive overview of how the results were obtained, as well as the exact values, is given in appendix C.

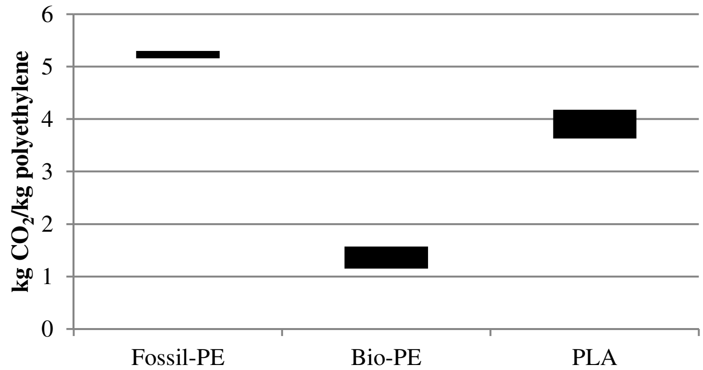


Figure [4.3] Life cycle GHG emissions for PLA and polyethylene, including incineration as end-of-life scenario

The results for bio-PE show a GHG emission reduction ranging from ~70% till ~80%, which is well above the 50% reduction given in table [3.1]. The results for PLA show a GHG emission reduction ranging from ~19% till ~34%, which is below the 50% reduction given in table [3.1]. The low emissions resulting from the production of bio-PE result from the extensive use of bagasse as an energy source. The results shown in figure [4.3] do not include land use change and the restriction of pre-harvest burning. Restricting pre-harvest burning has only minor influence, with a potential reduction of ~0.2 kg CO_2-eq/kg product (Lisboa *et al.*, 2011). The resulting reduction of GHG emissions would range from ~74% till ~83%. Land use change could significantly effect GHG emissions (Anderson-Teixeira *et al.*, 2009). Extensive research is performed on GHG emission due to land use change, but still large differences are reported due to different system boundaries, different time frames, difficulties in quantifying emissions and whether or not indirect land use is included (Searchinger *et al.*, 2008). Sugarcane cultivation was primarily on previously deforested areas and displaced mainly pastures and perennial crops (Pacca & Moreira, 2009). This would mean that the GHG emissions from land use change would be small, and repayment by carbon uptake in sugarcane would be within several years. However, indirect land use change of forested areas and Cerrado would result in a large initial carbon release (Fargione *et al.*, 2008; Lapola *et al.*, 2010; Searchinger *et al.*, 2008). An analysis was performed on land use change for the production of bio-PE to determine the effects of land use change on GHG emissions. Three scenarios were examined; the conversion of pasture to sugarcane fields (P-S), the conversion of Cerrado to sugarcane fields (C-S) and the conversion of rainforest to sugarcane fields (R-S). It is assumed that all carbon released from land use change is converted to CO_2. No indirect land use change scenario was examined, but the R-S scenario reflects the situation if displacing cattle would cause rainforest to be converted to pastures. The initial carbon loss during conversion was attributed to 5, 10 and 15 years of products for each scenario. A table with the used values is given in appendix D.

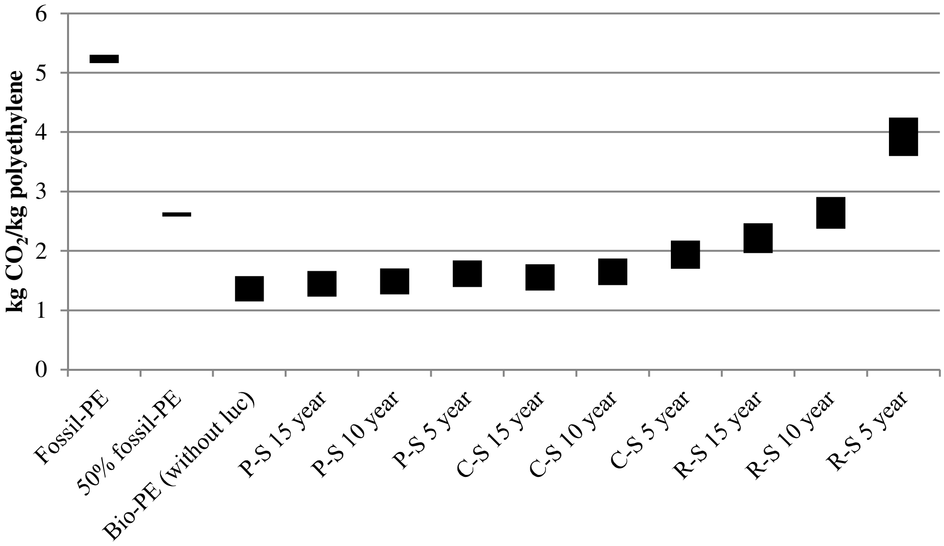


Figure [4.4] Life cycle GHG emissions for bio-PE including land use change (LUC) and end-of-life incineration. P-S: The conversion of pasture to sugarcane; C-S: the conversion of Cerrado to sugarcane; R-S: the conversion of rainforest to sugarcane.

Figure [4.4] shows the effect of land use change on GHG emissions. Although the results are only for bio-PE, similar effects are expected to result from land use change for the production of PLA. For bio-PE, direct land use change occurred mainly as conversion of pasture to sugarcane fields (Pacca & Moreira, 2009). However, indirect land use change could displace cattle to current rainforest, causing the conversion of rainforest to pastures, which in turn are converted to sugarcane fields. The effects of direct land use change (P-S) on GHG emissions are small, as is shown in figure [4.4]. The effects of indirect land use change could be significant depending on the type of land that is converted. It should be pointed out that soil organic carbon **increase** after conversion was not considered. It is reported that soil organic carbon can potentially return to pre-conversion levels after some period of time depending on how pastures or sugarcane fields are managed (Neill *et al.*, 1997).

As the paragraph “biodiversity” will show, there is sufficient land available to prevent direct or indirect land use change to cause great impact on the GHG emissions by the conversion of high carbon stock land. Note that this is only true for the relatively small production volume needed to meet in the demand of polyethylene in the Netherlands.

Energy requirements - Energy requirements were evaluated on MJ/kg product. The results for fossil-PE, bio-PE and PLA are given in figure [4.5]. A distinction is made between energy requirements derived from fossil resources and energy requirements derived from renewable resources. The main renewable resource used in the production of bio-PE and PLA is excess bagasse from sugarcane and corn (Ensinas *et al.*, 2007). The intrinsic energy stored in the product (oil-equivalent stored in product, ~43 MJ/kg product) and energy production at end-of-life incineration are excluded. Detailed information on how the values were obtained is given in appendix E.

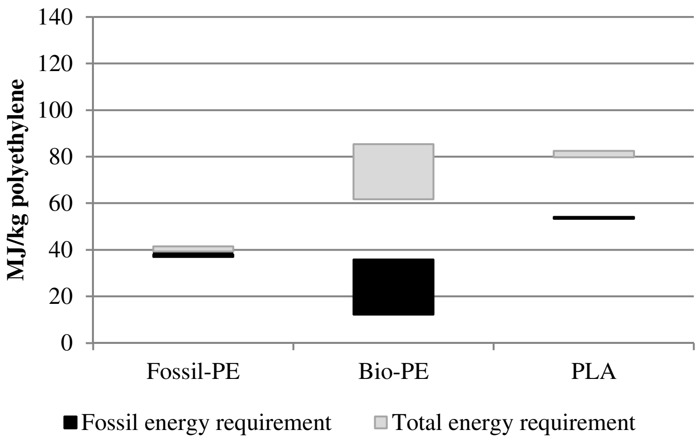


Figure [4.5] Life cycle energy requirements for polylactic acid and polyethylene. Total energy requirement = fossil energy requirement + renewable energy requirement.

The results for bio-PE show an increase in energy requirements ranging from ~49% till ~124% compared to fossil-PE. The results for PLA show an increase in energy requirements ranging from ~92% till ~117% compared to fossil-PE. Bio-PE shows a decrease in fossil energy requirements ranging from ~3% till ~70% compared to fossil-PE. PLA shows an increase in fossil energy requirements ranging from 32% till 47% compared to fossil-PE. Note that even though fossil-PE requires less fossil energy than PLA, GHG emission levels are higher than those of PLA. This is due to the fact that CO_2-sequestering is taken into account for PLA and bio-PE.

Biodiversity

Biodiversity is evaluated on 1) current violation of protected areas and 2) additional land use for the production of fossil-PE, bio-PE and PLA, compared to the available unprotected land.

Current violation of protected areas - Direct impacts on biodiversity by the production of fossil-PE encompasses deforestation, opening of forest canopy along seismic lines, drop zones and helipads, soil and ground water contamination and oil spills, as well as hydrocarbon, water and mud discharges. Indirect impacts mostly stem from road creation for oil and gas processes (Matea *et al.*, 2011). These roads are used by (illegal) logging industries and bush meat hunters, increasing deforestation and species extinction. Two major studies recently released show the interference of oil projects with protected land on GIS maps for the western Amazon (Finer *et al.*, 2008) and sub-Saharan Africa (Matea *et al.*, 2011). Both studies found that large shares of designated protected areas are being used for oil and gas extraction, which shows that protected areas are not always respected.

Bio-PE is produced in Brazil with sugarcane as feedstock. Direct impact on the biodiversity is small, since sugarcane expansion mainly takes place on former pastures (Pacca & Moreira, 2009). The impact of indirect land use change from sugarcane production is as of yet unclear. Displaced cattle might lead to the conversion of Cerrado and rainforest.

PLA is produced in the United States with corn as feedstock. Numerous small areas are protected in the Corn Belt (World database on protected areas[1]). However, no reports were found on the effect of corn production on legally protected land. As for Bio-PE, the impact of indirect land use change is as yet unclear.

Additional land use for the production of fossil-PE, bio-PE and PLA - The additional land use for the production of 500kt/yr fossil-PE is considered to be negligible. Additional land use for PLA was

[1] http://www.wdpa.org/

calculated assuming that 2.5 kg corn is needed per kg PLA (Vink *et al.*, 2003) and corn yield per hectare is 5000 kg/yr. Actual corn yield might be higher due to excellent weather conditions, but a relative small yield is assumed to assess a worst case scenario. Determining the additional land use for bio-PE was not as straightforward. A detailed calculation method is given in appendix F. The results for fossil-PE, bio-PE and PLA are shown in table [4.1].

Table [4.1] Additional land requirement and the impact on protected areas for the production of fossil-PE, bio-PE and PLA

	Fossil-PE	Bio-PE	PLA
Additional land use for 500kt/yr product	0 (1000 ha)	179 – 196 (1000 ha)	249 (1000 ha)
Direct impact on protected areas	Low	Low	Low
Indirect impact on protected areas	Uncertain	Uncertain	Uncertain

The results for bio-PE show an increase in land used for economic utilization of 0.08%[2] or 0.32%[3] compared to the total arable land. The results for PLA also show an increase in land used for economic utilization of 0.06%[4] or 0.15%[5] compared to the total arable land. Considering the relatively small increase in land needed, it is assumed that production of biomass for replacing fossil-PE in the Netherlands does not need to impact protected land. A few notions need to be remarked on this statement. First, if biomass production for biobased products co-exists with biomass production for biofuels, the impact of the two production processes is accumulative and they have to be considered separately. Sugarcane production has already increased with ~75% over the last 6 years in the south-central region of Brazil due to the demand for sugar and ethanol, and is expected to continue to increase (Silva *et al.*, 2010). The demand for sugarcane for bio-PE would add to this increase. Second, the area of land designated as protected might not be sufficient, taking into account the actual value and biodiversity of the land. This is illustrated with the Cerrado region in Brazil. Figure [4.6] shows the areas suitable (green) for sugarcane production in Brazil. Important biomes are excluded as potential land for sugarcane (white).

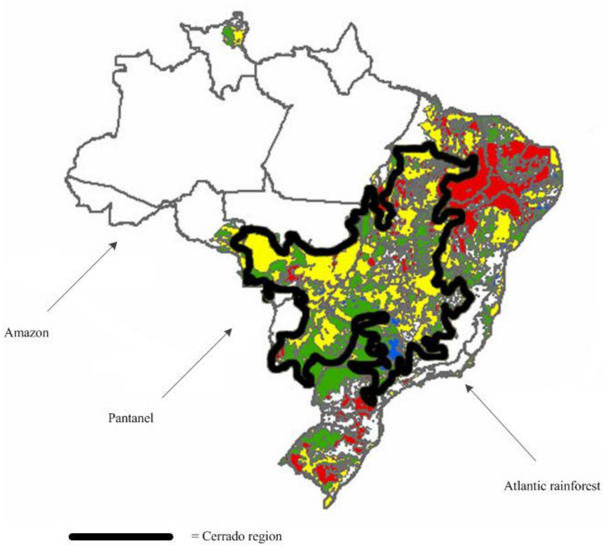


Figure [4.6] Land suitable for sugarcane production. Green areas are considered very suitable. Source: Smeets (2006)

[2] Total land used for economic utilization in Brazil is 236 million ha (FAO, 2004)
[3] Total arable land in Brazil is 61 million ha (FAO, 2012)
[4] Total land used for economic utilization in the United States is 403 million ha (FAO, 2012)
[5] Total arable land in the United States is 163 million ha (FAO, 2012)

The bio-diverse Cerrado region is for a large part not officially protected. The Cerrado region is a bush- and grassland biome. It was once seen as one of the last great land frontiers. Only 2.7% of the Cerrado region is national park. The Brazilian government designated 20% of the land as mandatory for natural preservation by private land owners (Gauder *et al.*, 2011). It is estimates that out of the 204 MHa Cerrado, 90 MHa are suitable for sugarcane production (Smeets *et al.*, 2006). This shows that even though the Cerrado region is an important biome, which might deserve protection, only a small part is designated as protected area. Third, the impact of indirect land use is still unclear. Displaced cattle might cause the conversion of protected land.

Local environment

The local environment was evaluated on multiple criteria. Each criterion is discussed below for bio-PE and PLA. Fossil-PE is discussed separately since the biomass indicators did not apply for the fossil-based product supply chain.

Fossil-PE and the local environment - The indicators provided in chapter 3 are applicable on bio-supply chains, but to a lesser extent on fossil-supply chains. Therefore, instead of following the biomass sustainability indicators, a short evaluation on the effects of oil production on local environments is given for soil, water and air quality. Soil quality is mainly affected during production, but more important during pipeline ruptures. Studies on soil quality are often based on Soil Organic Matter (SOM) and microbial biomass (MB). They show a significant effect of oil spills on soil and biological quality, which can persist over several decades (Timmerman *et al.*, 2003). Pipeline ruptures are inevitable due to the sheer length of the pipelines system, making it virtually impossible to control every risk like corrosion, external interference, construction & material defects and "acts of god" like natural disasters (Dey, 2004). The largest threats for water quality are offshore oil spill, with a special focus on oil tanker accidents. Figure [4.7] gives an overview of marine tanker oil spills larger than 700 tonnes from 1965 until 2002 (Vieites *et al.*, 2004). The graph shows that spills are most common in coastal regions. Studies performed on oil spills show that they cause extensive damage on marine biodiversity, bird populations and coastal regions (Duke *et al.*, 1997; Guzman *et al.*, 1991; Shriadah, 1998). Especially, the disaster with the Exxon Valdes triggered a global response against oil spill pollution, spurred by the United States (Paine *et al.*, 1996). Despite efforts to prevent further oil spills, they are considered inevitable and effects of oil spills on cellular, organismic and community levels can persist for decades (Van der Meulen, 1982). Air quality is impeded by the oil refining process through exploration, production and flaring (Villasenor *et al.*, 2003), as well as gasses emitted from the refining process. During combustion, hydrocarbons, nitrogen oxide and sunlight can lead to ozone formation, or smog and sulphur and nitrogen oxide emissions can result in (localized) acid rain formation (Speight, 1999).

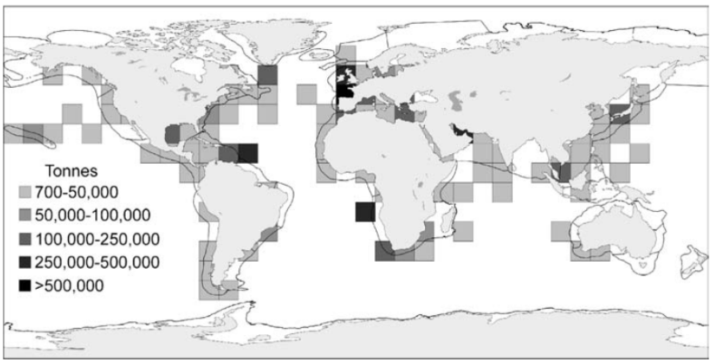


Figure [4.7] Worldwide distribution of oil spilled in the seas by maritime transport from 1965 to 2002

The chapter continues with the evaluation of the bio-PE and PLA supply chains along the local environment indicators given in table [3.1].

Bio-PE and PLA: Compliance with the Stockholm convention on persistent organic pollutants - Brazil ratified the Stockholm convention on the 16th of June 2004. On the 4th of November 2010 Brazil submitted the national report on persistent organic pollutants. Brazil reported a ban on production, import, export and use of all chemicals stated in annex A (elimination) of the Stockholm convention. Furthermore, Brazil reported that no chemicals stated in annex B (restriction) of the Stockholm convention were produced, imported or exported. The evaluation of pollutants given in annex C (unintentional) is reported to be in progress. Next to compliance with the Stockholm convention, the environmental crimes law foresees in a legal penal framework to those who endanger the environment (National congress, 1999). Chapter 5, section 3 of the "environmental crimes law" on pollution and other environmental crimes provides a framework for penalties on those actions that cause pollution of any nature at such level that it results or could result in damage to human health, or that it could cause death of animals or significant destruction of flora.

The United States signed the Stockholm convention on persistent organic pollutants on the 23rd of 2001. However, it has yet to ratify the Stockholm convention. The U.S. regulates persistent organic pollutants in the Toxic Substances Control Act (TSCA) (EPA, 1976). The original persistent organic pollutants in the Stockholm convention are also covered in the TSCA. In order to be compliant with newly added persistent organic pollutants to the Stockholm convention, U.S. legislation needed to be changed. In 2011, Senator Frank Lautenberg signed the Safe Chemicals Act to modernize the U.S. chemical policy, and to allow the U.S. to ratify the Stockholm convention (Ditz *et al.*, 2011).

Bio-PE and PLA: Soil organic matter is maintained/preserved and maintenance of soil nutrients balance concerning nitrogen (N), phosphorus (P) and potassium (K) – For Brazil, a comprehensive study was performed by Malavolta (1994) on soil nutrients and organic matter. Malavolta identified four ways in which nutrients are extracted from the soil: uptake by sugarcane, burning of sugarcane, leaching and erosion. Nutrient extraction can vary depending on variety of sugarcane, soil conditions, length of the crop cycle and type of crop (Malavolta, 1994). Table [4.2] shows the application rate of fertilizers which are considered needed for sustainable sugarcane production, i.e. without shifting the nutrient balance (Patzek & Pimentel, 2005). Table [4.2] also provides the reported/estimated application rates given by different authors. When the recommended application rates are compared with the reported/estimated application rate, it can be seen that there is a negative nutrient balance for N-P-K nutrients. This result is supported by the findings of the FAO (2004). The FAO states that the nutrient balance for the Sao Paulo region is on average negative for N-P-K nutrients. Even though the nutrient balance is negative, sugarcane cultivation in Brazil has been successful for decades. For N this might be explained by biological nitrogen fixation (Hartemink, 2008), which could be as high as 60-80% of plant N for certain species of sugarcane (Martinelli & Filoso, 2008). Furthermore, application of filter cake and vinasse to sugarcane fields increases nutrients levels in the soil. The process is called ferti-irrigation, since it both fertilizes and irrigates the sugarcane fields. Both filter cake and vinasse contain N-P-K nutrients (de Resende *et al.*, 2006). In Brazil, several laws constrain the use of vinasse because of possible harmful effects on the local environment (Smeets *et al.*, 2006). Note: The evaluation of the NPK balance was limited due to the fact that multiple studies are only available in Portuguese (Busato *et al.*, 2005; de Andrade *et al.*, 2011; Otto *et al.*, 2010).

Table [4.2] Application rates of N-P-K nutrients for sugarcane production in Brazil

	N (kg/ha/yr)	P_2O_5 (kg/ha/yr)	K_2O (kg/ha/yr)
Recommended application rates			
Patzek & Pimentel (2005) unburnt	100	80	170
Patzek & Pimentel (2005) burnt	145	93	314
Reported/estimated application rates			
FAO (2004)	61	57	118
Macedo (2008) [a]	81	42	115
Malavolta (1994)[a]	50-90	29-60	12-87
Seabra et al. (2011) [b]	53	17	67

[a] Average of 1 plant and 5 ratoon cycles
[b] Assuming 68.7 tonne sugarcane per hectare

Table [4.3] shows the applied fertilization rate for corn production in the United States (Christensen, 2002). The rates are averaged for the Corn Belt, lake states, plains states and the southeast regions. No information was found on sustainable application rates. It was found that nitrogen fertilization is reported to be too high (EWG, 2011) causing high run-off levels as referred to later. Potassium and phosphorus levels in the soil are reported to drop[6]. Note that the latter statement is reported on the website mentioned, and validity is highly uncertain because of lacking peer review or site reviews.

Table [4.3] Application rates of N-P-K nutrients for corn production in the U.S.

Reported/estimated application rates	N (kg/ha/yr)	P_2O_5 (kg/ha/yr)	K_2O (kg/ha/yr)
Christensen (2002)	62	25	34

Bio-PE and PLA: Burning as part of cultivation is restricted - Sugarcane burning is common practice in Brazil. Prior to harvest, the sugarcane is burnt to make the harvesting process easier and to reduce the need for manual labour. Burning prior to harvesting was applied to 75% of the total area of sugarcane cultivation in 2009 (Ometto *et al.*, 2009). Burning sugarcane has several negative effects on the environment like increasing soil temperature, decreasing soil water content and bulk density, PAH (polycyclic aromatic hydrocarbon) emission, aerosol pollution, elevation of CO-levels, O_3-levels and NO_x-levels concentrations, water runoff, soil compaction and higher rates of surface erosion (Martinelli & Filoso, 2008). Furthermore, burning sugarcane has negative effects on human health, like increased respiratory morbidity. A correlation was found between hospital admissions for inhalation treatment for respiratory diseases near pre-harvest burned sugarcane fields (Martinelli & Filoso, 2008). Brazil recognizes these threats to the environment and public health and therefore pre-harvest burning is being phased out under law 11,241/2002 (National congress, 2002). Figure [4.8] shows the timeframe of this phase out. From 2030 onwards, pre-harvest burning will no longer be applied according to national legislation. This is in accordance with the indicator on the sustainability of the local environment.

[6] http://www.back-to-basics.net/soil_test_summary.htm

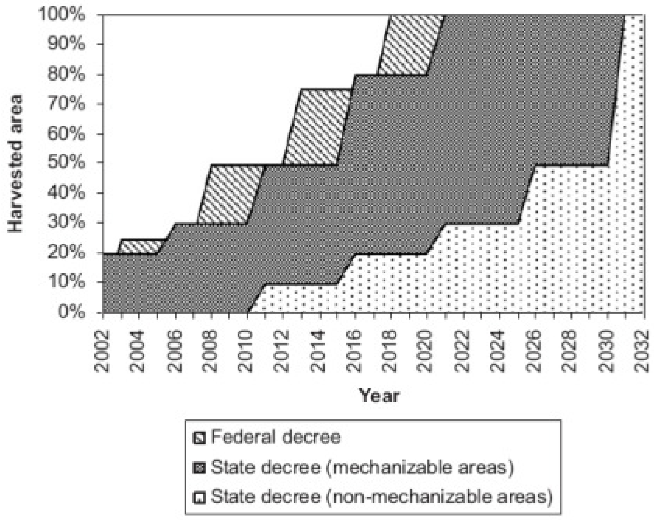


Figure [4.8] Time frame for the phase out of pre-harvest burning. Source: Macedo *et al.* (2008)

In the United States, pre- and post-harvest crop residue burning is common practice. Sugarcane, wheat and rice are crops that are most commonly burned pre- or post-harvest. Corn is less extensively burned (McCarty *et al.*, 2009). Many states in the United States are so-called “freedom to farm” or “right to farm” states, which prevents legislation against pre- and post-harvest burning. However, considering air quality and visibility levels through smoke, many states are reconsidering policies on crop burning (McCarty & Hartmann, 2010). Appendix G gives an overview of current policies in some states, showing that restrictive legislation is being implemented.

Bio-PE and PLA: No run-off of applied fertilizer to surface water - According to Goldemberg *et al.* (2008) no water pollution due to the run-off of applied fertilizer to surface water is reported in Brazil. Sugarcane is rated as level 1, meaning that there is no impact on water quality (Goldemberg *et al.*, 2008). Since sugarcane fields are under-fertilized (see the paragraph on the nutrient balance), sugarcane fields act more as N-P-K sinks than that they are sources of fertilizer run-off. Nonetheless, water pollution partly due to fertilizer run-off is reported (Ometo *et al.*, 2000). The reported impact is however significantly lower than water pollution due to urban waste.

High run-off of applied fertilizer during agricultural operations in the United States is reported by several authors (Sistani *et al.*, 2010; Smith *et al.*, 2007; Udawatta *et al.*, 2006). The Corn Belt contributes to a large extent to fertilizer run-off. High fertilizer input and drainage techniques divert water and fertilizers to drainage ditches (EWG, 2011). Three recommendations are made by the “Environmental working group” on reducing fertilizer run-off in the United States. First, the amount of applied fertilizer should be reduced, and improved techniques for fertilizer uptake should be applied. Second, nutrient management plans should be obligatory. Third, funding’s should be restored to government agencies that monitor water quality (EWG, 2011). Although not mentioned in the recommendations, it should be noted that enforcement of laws and regulations is of utmost importance.

Bio-PE and PLA: Measurements to prevent soil erosion should be adapted – Soil erosion is often high in sugarcane cultivation areas in Brazil compared to forests and pastures because of extensive areas of bare soil associated with sugarcane production. High winds, extensive rainfall on the bare soil, slope, soil properties, crop type and management systems are key determinants of erosion levels (Martinelli & Filoso, 2008; Mann *et al.*, 2002). Potential erosion losses can cause loss of surface soil particles that contain high concentrations of soil organic carbon and nutrients. Soil losses for sugarcane may vary dramatically, from 0.1 ton/ha/yr to 109 ton/ha/yr, depending on the slope, the annual rain fall, the management and harvesting system. Soil erosion during sugarcane production is therefore a site-specific problem, rather than an inherent problem to sugarcane production (Smeets *et al.*, 2006).

Common measures to reduce erosion include drains, bunds, ridges, strip cropping, and on heavy clays and strip tillage. Bench terracing is also applied to avoid run-off and erosion. Most attention in Brazil is paid to no-tillage systems (Hartemink, 2008). Tillage of the land can cause high levels of erosion, especially in sloped areas. Tillage techniques include ploughing, digging, stirring and overturning the land. The tillage systems in Brazil are being replaced by no-till systems (FEBRAPDP, 2003), as shown in figure [4.9] (data was found up to 2006). In 2006, this was ~40% of the total arable land.

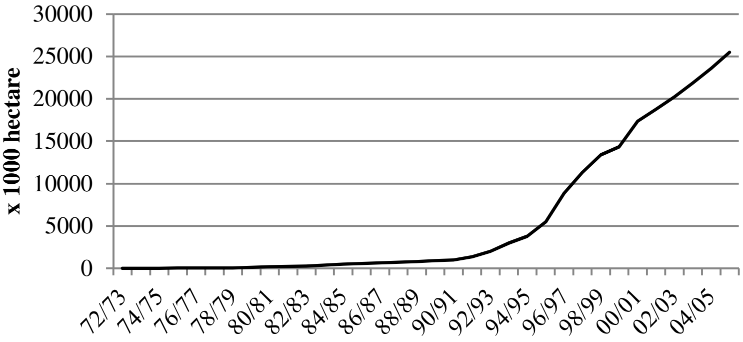


Figure [4.9] Area cultivated with no-till practices for Brazil. Source: FEBRAPDPD (2003)

Soil erosion is an on-going concern for agricultural operations in the United States. A recent report by the "environmental working group" reported that over six million acres of land suffer erosion above sustainability levels (EWG, 2010). Possible solution are adding riparian forest and bush strips, which filter soil from water before it enters drainage ditches, reducing tillage practices and leaving more crop residues on the field (EWG, 2010; Petrolia, 2008; Shipitalo & Edwards, 1998). Table [4.4] shows the current tillage systems applied to highly erodible land (HEL) and non-highly erodible land (NHEL). The table shows that systems that reduce erosion are being applied, but only to some extent. Based on the information given above, improvements in soil erosion control are recommended

Table [4.4] Applied tillage systems in corn production in the U.S. 1996 source: Christensen (2002)

	% of total acres	
Tillage system	HEL	NHEL
Conventional	15	35
Reduced	28	33
Mulch-till	28	15
No-till	29	15
Ridge till	0	2

Table [4.5] shows an overview of the results of this chapter. The production of bio-PE shows better results on the compliance with the Stockholm convention, pre- and post-harvest burning and run-off of applied fertilizer when compared to PLA. Both the production of bio-PE and PLA do not comply with the indicator on fertilizer use, but for other reasons. Erosion measures are being taken for both the production of bio-PE and PLA.

Table [4.5] Comparing bio-PE and PLA: Impact on local environment

	Fossil-PE	Bio-PE (Brazil)	PLA (United states)
Compliance with the Stockholm convention	-	Yes	No
SOC and NPK balance is preserved	-	No, under-fertilization	No, over-fertilization (N)
Pre- and post-harvest burning is restricted	-	Phased out in 2030	Only in certain states
No run-off of applied fertilizers	-	Low levels of run-off reported	High levels of run-off reported
Prevention of soil erosion	-	No-tillage systems are being implemented	Most highly erodible land has reduced tillage systems

4.3 Social sustainability

This paragraph compares the social sustainability of the fossil-PE, bio-PE and PLA supply chains. The three different aspects competition for food, welfare and wellbeing are compared.

Competition for food

Oil and gas exploration and production has its' effects on the surrounding land as described under "environmental sustainability". Land use change occurs mainly on-site and along transport routes. This has some impact on local land use, but it is very much restricted to a relative small production site and has no significant competition with food. The competition of crops planted for biobased products with food and other local use of biomass can be more drastic. The competition is determined through reporting on four different points (table [3.1]):

a) Nature of biomass feedstock
b) Production location
c) Land use surface
d) Development of land use, food availability and prices of land and food

The production location was discussed in paragraph 4.1. The land use surface was discussed in paragraph 4.2. The development of land use, food availability and prices of land and food are determined by using the FAO statistics database.

Figure [4.10] shows the production of crops and meat in megatons for Brazil for the last decade. It clearly shows the increase in sugarcane production. Even though an increase in sugarcane production took place, the production of other important food items did not decrease. The main reason for this is that sugarcane expansion took place on former pasture lands, not on crop land (Gauder *et al.*, 2011).

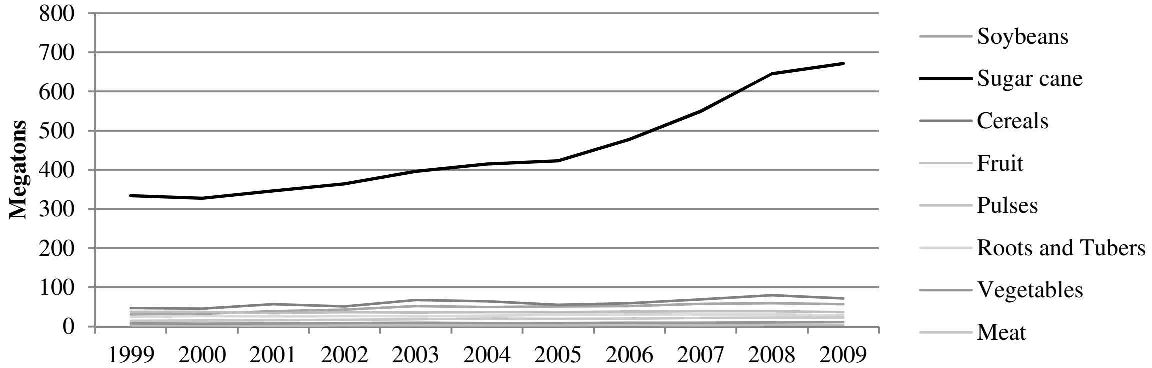


Figure [4.10] Crop production in Brazil. Source: FAO database

The impact of increased sugarcane production on food prices is difficult to determine. To give an analysis, world average producer prices for important food production items in Brazil are compared to producer prices in Brazil. Figure [4.11] shows the producer prices of sugarcane in Brazil, compared with the average producer price of the top ten producers' worldwide (FAO, 2012). From figure [4.11] it can be concluded that the average world producer price follows roughly the same trend as the producer price in Brazil. To determine the impact on the price of other crops, the world average producer prices are determined for these crops in appendix H using the FAO price statistics database. Considering the development of producer prices in the world and in Brazil, it is assumed that the rise in producer prices in Brazil is not correlated with the increase of sugarcane production but by the general rise in producer prices worldwide. This is consistent with the studies done by other authors (Gauder *et al.*, 2011; Sparovek *et al.*, 2009). Gauder *et al.* (2011) discusses three different scenarios on the impact of sugarcane expansion for ethanol on food production. They conclude that even without arable land expansion, food production could grow up to 2.8% per year by increasing production efficiency, which is more than the annual population growth of 2.1%. Land sales prices are increasing due to the expansion of the sugarcane cultivation area. Sales prices for the Sao Paulo state have increased more rapidly than the average sales prices for crop land in Brazil. This rapid increase is caused by the high level of investment in sugarcane and the massive wave of increased sugarcane plantations in the area (Sauer & Leite, 2011). Appendix I shows the development of sales prices of crop lands in different states in Brazil.

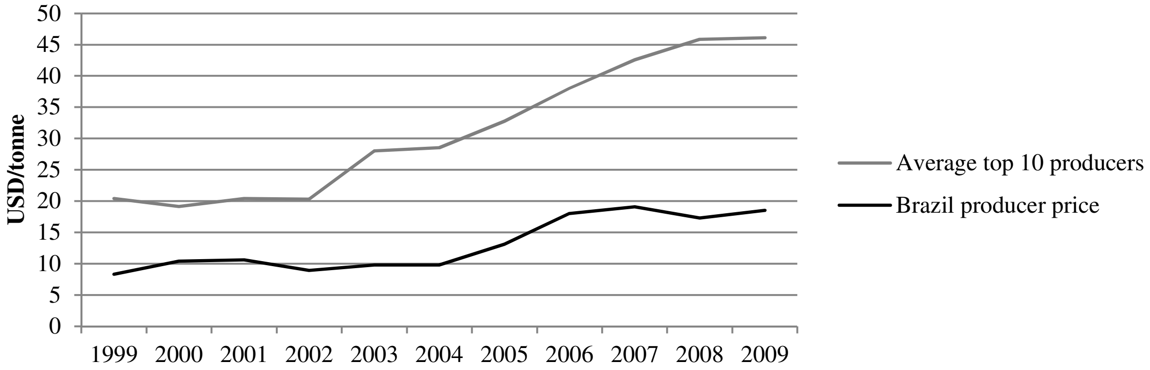


Figure [4.11] Sugarcane producer prices. Source: FAO

Figure [4.12] shows the production of important crops (EPA, 2012) in the United States. It is shown that corn production increased over the last decade, but relatively less when compared to the increase in sugarcane production in Brazil. From figure [4.12] it can also be seen that the production of other crops did not decrease over the last decade.

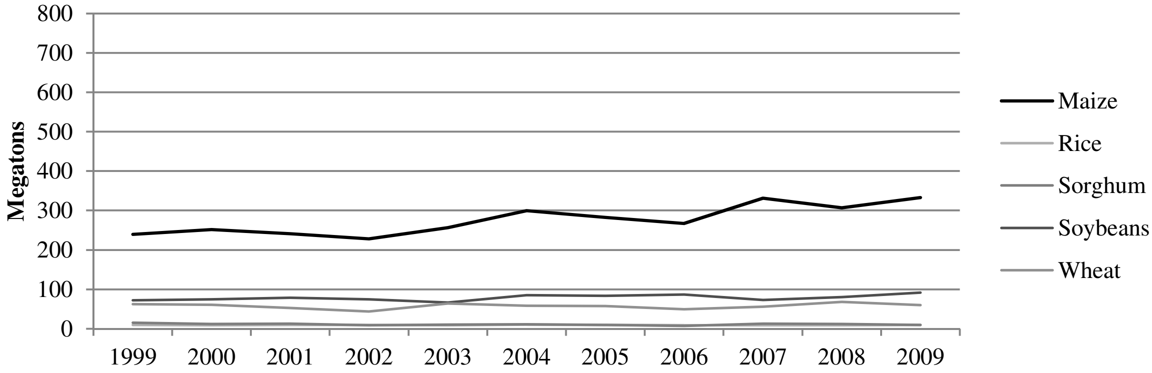


Figure [4.12] Crop production in the United States. Source: FAO database

Figure [4.13] shows that the average producer prices of the top ten corn producers seem to follow the price trend set by the United States. Recently, predictions for a drawdown of corn inventories led to a corn prize surge. By may May 2011, corn export prices were on average 80 percent above their May 2010 quoted values (FAO, 2011). If the United States decide to limit export to keep domestic supply at acceptable levels, other regions in the world might suffer (Banerjee, 2011).

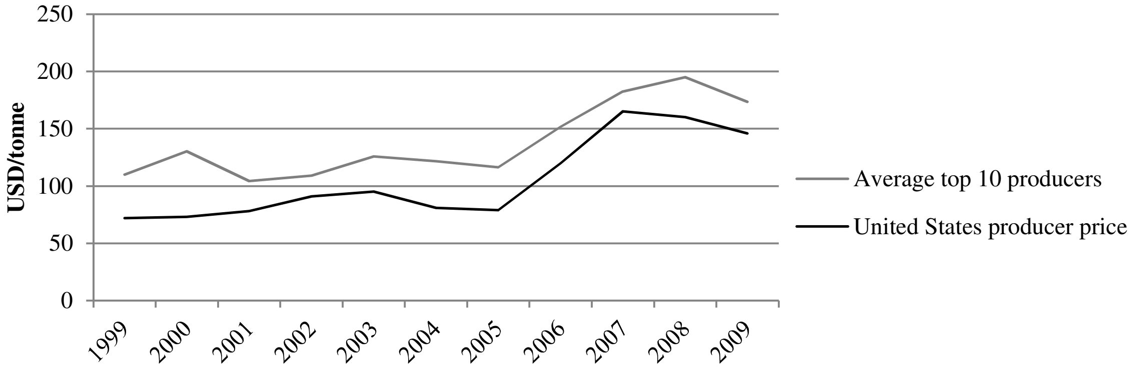


Figure [4.13] Corn producer prices. Source FAO

For both bio-PE and PLA no evidence was found that food prices of *other* crops rose due to the production of feedstock for purposes other than food. However, US corn prices and Brazils' sugarcane prices do affect the world market price for those specific crops. Especially price rises in corn, which is one of the major cereal products, could affect the food prices and therefore food availability worldwide. However, considering that the amount of feedstock needed for bioplastic production is small when compared to food and fuel demand, one might argue that the effects on food competition are negligible for both products.

Welfare

The aspect welfare of social sustainability is evaluated on the basis of the Global Reporting Initiative (GRI). The evaluation is based on indicators EC1 "Direct economic value generated and distributed", EC6 "Policies, practices and proportion of spending on local suppliers" and EC7 "Local hiring" as described in table [3.1].

Since the production of fossil-PE was not linked to a specific case, evaluating these indicators separately is more difficult. Therefore, a general view on the impact of oil production on welfare is given. The information on oil and welfare is derived from the report "Oil-led development: social, political and economic consequences." composed by the Stanford University Centre on Democracy, Development and the Rule of Law (Karl, 2004). During the first stages of exploration and production, petroleum revenues have positive influences on welfare. Employment rates go up, infrastructure is constructed and improved, and income per capita increases. This positive influence can potentially diminish when countries do not shift from oil-dependency to more sustainable industries, like agriculture and other labour-intensive industries. Especially the volatility of petroleum revenue causes negative impacts on investment, income distributions and poverty alleviation. Furthermore, the capital-intensive nature of the oil and gas industry provides only a limited number of jobs. Because of the high ratio of capital with respect to employment, and the skills required, local population does not benefit from the few jobs created. This results in high percentages of people living in oil and gas exporting countries to remain unemployed and in poverty. This effect has been evident in countries like Algeria, Angola, Congo, Ecuador, Gabon, Iran, Iraq, Kuwait, Libya, Qatar, Saudi Arabia, and Trinidad Tobago. Nigeria and Venezuela showed the most negative impact, reducing real per capita income back to 1960 levels. Furthermore, the large amounts of capital and technological resource needed results in foreign oil and gas companies to become dominant as an internal social force. Partnerships are formed with the elite of the country, but the economic presence, capital and technological advantages does not give benefits for local population, and domestic entrepreneurs are pushed out of the market. So even though the oil and gas industry is subjected to high demand and high revenues, these benefit only the local elite, while it can have great negative impact on the welfare of local populations. Note that not all countries fall in the same category as the countries described above. Australia, Botswana, Canada, Chile, Hong Kong, Norway, Singapore, South Korea, Taiwan, and the United States successfully exploited their natural resource without impeding welfare. Furthermore, some Southeast Asian countries manage to avoid the *resource curse*. The difference can be found in the pre-existing political, social and economic institutions which were in place to manage the natural resources as they came into production (Sovacool, 2010).

The Brazilian Sugarcane Industry Association (UNICA) published the Sustainability Report in 2010 (UNICA, 2010), which is based on the GRI indicators. Important to note is that the quality of reporting is approved by the GRI. The GRI explicitly notes that it does not give an opinion on the sustainability of the reporter, nor on the quality of information in the report. EC1: Investments to build new mills and expand existing ones should total $33 billion through 2012. The majority of new projects involve Brazilian investors, but the share of foreign investors in the capital of companies in the sector is projected to increase from the current 7% to 12% by 2012/2013. Job creation is higher than for other sectors in Brazil (40% mean formal jobs compared to 72.9% formal jobs in sugarcane regions), and the average income of families with jobs in the sugarcane industry is higher than that of 50% of the Brazilian population (Sauer & Leite, 2011). However, Issues with underpayment are reported, mainly due to the calculation method for the amount of sugarcane cut per fieldworker (Martinelli & Filoso, 2008). The unemployment rate in Sao Paulo, where sugarcane is the dominant industry was 6.3% in 2010, which was slightly below the national average of 6.7% (IBGE, 2012). EC6: UNICA reports that of 93 reporting members, 79% stated that policies were applied for purchasing and investment aimed at enhancing socio-economic development in the community where they operate. No scientific literature was found which could verify this statement. EC7: UNICA reports that of 93 reporting members, 89% stated that programs are established within the company to hire as many local people as possible. Furthermore, training aimed at increasing skill levels in the community is provided to hired personnel. In contrast to this report, it is found that sugarcane field workers are often migrants from poorer regions in Brazil, which are lured under the promise of better wages and good living conditions (Martinelli & Filoso, 2008).

For the United States, no report was found that explicitly detailed the GRI indicators as given in table [3.1]. Welfare was instead analysed on average household incomes for farms, and the percentage of local hiring. In 2008, the average farm household income was $79,796. The average U.S. household income was $68,424, which means that the average farm household income was 17% higher than the average U.S. household income (Heller & Keoleian, 2000). In contrast, farm workers are reported to be one of the most economically disadvantaged work groups in the United States (USDA, 2012). In 2001-2002, 30% of all farm workers had total family incomes that were below the poverty line

(Carroll *et al.*, 2005). Furthermore, on the topic of local hiring, 42% of hired farmworkers were migrants (defined as travelling more than 75 miles during a 12 month period to obtain a farm job) (Carroll *et al.*, 2005).

Wellbeing

The aspect wellbeing of social sustainability is evaluated on the basis of human rights, contributions made to local wellbeing and the GRI. The evaluation of the GRI is based on the indicators SO2, SO3 and SO4 as described in table [3.1]. These indicators assess the anticorruption measures taken.

Since the production of fossil-PE was not linked to a specific case, evaluating these indicators separately is more difficult. Therefore, a general view on the impact of oil production on wellbeing is given. The information on oil and wellbeing is derived from the report "Oil-led development: social, political and economic consequences." composed by the Stanford University Centre on Democracy, Development and the Rule of Law (Karl, 2004). The effects of wellbeing in countries which fall into the resource curse can be significant on social indicators. Developing countries dependent on oil and gas export typically have high poverty rates, poor health care, high rates of child mortality, poor educational performance, low life expectancies and high malnutrition rates. Indirect effects on wellbeing occur, like the increase of prostitution, AIDS and crime by the influx of immigrants looking for a job in the oil and gas industry. Furthermore, states with the greatest natural resources have very high levels of corruption. In the example of Africa, oil producing countries score lowest on the Government Effectiveness Index and they spiral towards the bottom of the Transparency International's Corruption Perceptions Index (Shaxson, 2007).

The main issue in Brazil surrounding wellbeing during bio-ethylene production is the exploitation of cane cutters, as reported by Martinelli & Filoso (2008) and NGO reporter Brazil. Poor working conditions and health threats resulted in law suits against sugarcane employers by the Ministry of Labour. NGO reporter Brazil reports that several cases of slavery occur each year. In Sao Paulo, no slavery was reported, but an excess of working hours and violations to workers' health and safety conditions were documented (NGO, 2009). Compliance with the universal declaration of human rights is violated by these incidents. Contributions to local wellbeing are reported in the sustainability report of UNICA (2010). Table [J-1] in appendix J shows the number of projects and the investment (in Real) as reported by the members of UNICA. The total investment is 32 million Real, or 18 million USD. A quarter of the total investment is reported to be spent on the area of environment. Table [J-2] in appendix J shows the initiatives per category, and the number of UNICA members which have such initiatives. Note that 89 of the 93 companies report that programs are in place to hire as much local people as possible in the community where the company operates. This is not in line with the extensive migrant hiring reported in the paragraph "welfare". On a national level, Brazil scored +0.06 in 2010 on the *control of corruption index* (Worldbank, 2012) (-2.5 = very poor performance, 2.5 = excellent performance). UNICA reports that given the characteristics of UNICA, no process is identified as having a significant risk on reputation or image. Table [4.6] shows the percentage of UNICA members, and their formal commitment for combating forms of corruption.

Table [4.6] Commitment to combating corruption by UNICA members. Source: UNICA (2010)

Commitment	Number of companies (%)
In the code of conduct	62%
As a specific corporation policy on the subject	24%
Via formal adhesion or public declaration relating to the commitments and voluntary initiatives	25%

The figures presented raise questions on the implementation of SO2 and SO3 of the Global Reporting initiative in the sugarcane industry, since over 75% of the companies have no specific corporation policy on corruption. As stated in the previous paragraph, no guarantee whatsoever can be given on the validity of the reports of the members, and the report of UNICA in general.

For the United States, no report was found that explicitly detailed the GRI indicators for corn production as given in table [3.1]. Wellbeing was instead analysed on compliance with human rights and the corruption control index. No academic articles were found which elaborated on human rights for U.S farm workers. The CIW (Coalition of Immokalee Workers) and anti-slavery organisations do report that at any time, 5% of farm workers in the U.S. are subject to forced labour(Anti-Slavery International, 2012; CIW, 2012). Although this could not be verified, it is an indication that farm workers rights are not up to standard with worker rights in other sectors. On a national level, the U.S. scored a +1.23 in 2010 on the *control of corruption index* (Worldbank, 2012) (-2.5 = very poor performance, 2.5 = excellent performance).

4.4 Economic sustainability

This paragraph compares the economic sustainability of the fossil-PE, bio-PE and PLA supply chains. Note that the economic sustainability is only evaluated on market prices. The analysis is based on LDPE as stated in the paragraph "greenhouse gas emissions".

Market price

The market price was evaluated on spot market prices. Appendix K gives detailed information on how the market prices for fossil-PE and bio-PE were obtained. Figure [4.14] shows the results. Two linear equations are shown which indicate the relation between the feedstock price and the LDPE market price for both fossil-PE and bio-PE. Furthermore, the actual (current) market prices of fossil-LDPE and bio-LDPE are given. The market price of PLA is also given for comparison, but it should be noted that the feedstock is not comparable to ethanol or crude oil in terms of market price dependency on the feedstock. The market price for PLA was obtained from the pro-bip report (Shen *et al.*, 2009). At the time of writing, the reported market price was 1.2 USD/lb PLA. When converted, this results in 2645 USD/tonne. Note that that the current fossil-LDPE price deviates from the given relation. This is explained by price volatility and a delayed response to feedstock prices.

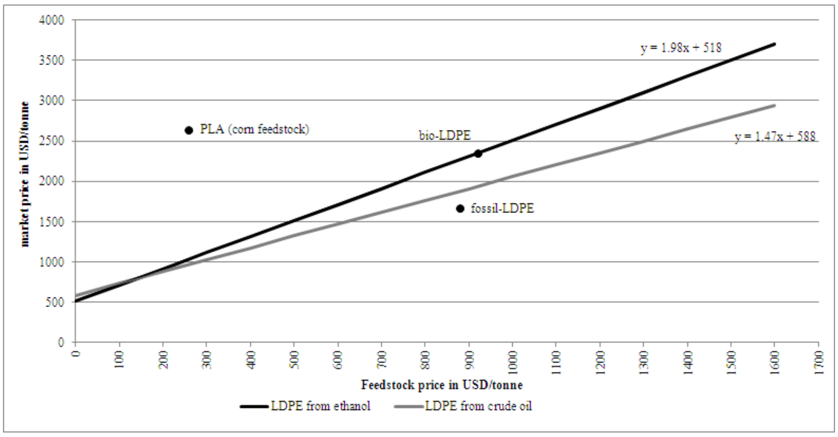


Figure [4.14] Market price analyses for ethylene at different feedstock prices. The figure also includes current market prices for PLA (with corn as feedstock), bio-LDPE and fossil-LDPE.

From figure [4.14] the conclusion can be drawn that the market price for bio-LDPE and PLA is significantly higher than the market price of fossil-LDPE. The prices are shown in detail in appendix K. Furthermore, the market prices are strongly dependent on the price of the feedstock. At current feedstock prices, bio-PE is ~40% more expensive than fossil-PE, and PLA is ~60% more expensive than fossil-PE. The indicator set for market price was "no large increase in market price compared to the fossil reference". Although no percentage was given, a 40-60% increase can be considered large, rendering bio-PE and PLA economically unsustainable. Note that products are being sold. This indicates that not only the market price determines the economic sustainability. Marketing and public green awareness could spur a demand for more expensive, but greener, products. One remark on future market prices can be made. While fossil-PE and bio-PE production technologies are mature, PLA production technology has the potential to improve (Shen *et al.*, 2009). This could potentially lower the market price of PLA, and close the market price gap with fossil-PE. This could be especially true with rising oil prices. Nevertheless, Shen *et al.* report an expected upward price shift for PLA.

4.5 Policies: Green VAT and carbon tax

As stated in paragraph 3.2, this paragraph discusses some sustainable policies for the Netherlands. Policies and regulations that affect the sustainability dimensions were, where necessary, discussed in previous sections of the report.

The Dutch government can stimulate the transition towards a biobased economy in numerous ways. Two policies are shortly discussed below; one "pull" measure and one "push" measure. VAT tax reduction can be defined as a "pull" measure. The Netherlands has three categories of VAT taxes; 19%, 6% and 0 %. Most products have 19% VAT taxes. Environmental friendly products could be taxed at 6% VAT taxes. The European parliament recently passed an initiative resolution which calls for such "greening VAT" strategies (European parlement, 2011). Reducing VAT taxes for bio-PE would mean that it becomes more economic competitive with fossil-PE. Carbon-tax implementation: Carbon tax implementation can be defined as a "push" measure. Companies and consumers would pay per tonne of CO_2 emitted during the life cycle of the product. Figure [4.14] shows the price difference between fossil-PE and bio-PE. Bio-PE is ~40% more expensive than fossil-PE and PLA is ~60% more expensive than fossil-PE. Figure [4.15] shows the price difference between fossil-PE and bio-PE if green VAT was applied (trend lines were removed for clarity). Bio-PE is now ~25% more expensive than fossil-PE and PLA is now ~40% more expensive than fossil-PE.

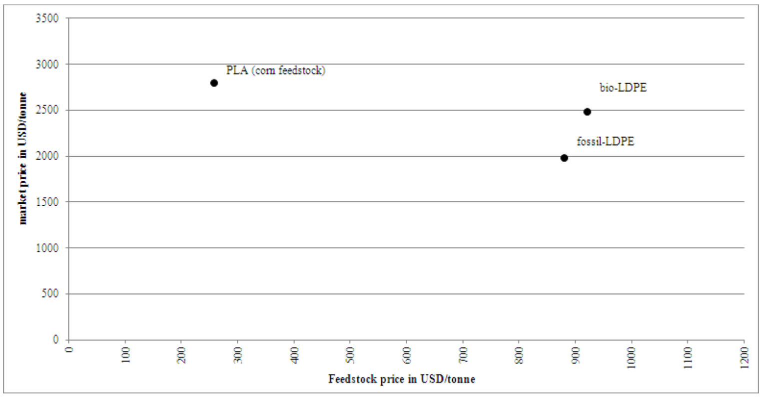


Figure [4.15] Current (estimated) market prices for fossil-PE, bio-PE and PLA including green VAT

Figure [4.16] shows the price difference between fossil-PE and bio-PE if green VAT was applied together with 50 USD/tonne CO_2 carbon tax. This is a very high carbon tax if compared to the current carbon emission rights trade price. However, it might be a more realistic carbon tax, and in time might become accepted by the public. The total carbon tax per tonne product was determined by multiplying the average CO_2 emissions as given in paragraph 4.2 with 50 USD. Bio-PE is now ~15% more expensive than fossil-PE and PLA is now ~35% more expensive than fossil-PE. One could argue that bio-PE and PLA are not too expensive, but that fossil-PE is too cheap. This is further discussed in the conclusion of this report.

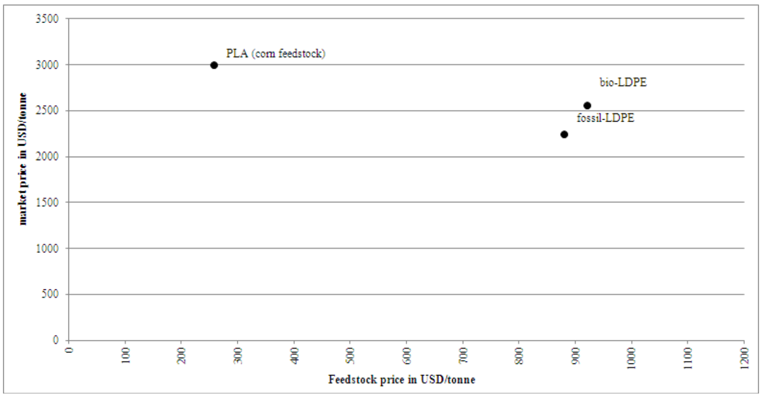


Figure [4.16] Current (estimated) market prices for fossil-PE, bio-PE and PLA including green VAT and carbon tax

4.6 Key findings

Chapter 4 compared the results on sustainability for fossil-PE, bio-PE and PLA. Bio-PE GHG emissions were found to be 70-80% lower than fossil-PE GHG emissions, and PLA GHG emissions were found to be 19-33% lower than fossil-PE GHG emissions. The effect of direct or indirect land use change could lower the reduction potential, depending on which type of land is converted and how the initial carbon loss is attributed to the products. Additional land use was relatively small for both bio-PE and PLA, even considering indirect land use. However, it can be argued that land use for sugarcane and corn for biobased products cannot be seen separately from the land use for sugarcane and corn for fuel. Furthermore, land important for biodiversity might not always fall under national or internationally protected areas. Considering the small production volume needed, the competition for food is small for both bio-PE and PLA. Note that, with a growing biobased economy, total land use (for all applications) and impact on competition for food should be considered cumulative. Production of fossil-PE has no significant impact on the competition for food. The local environment seems to potentially be at risk for fossil-PE, bio-PE and PLA. Special concern on this issue results from the imbalance in fertilization. NPK levels are below sustainable levels in Brazil, while in the United States over fertilization causes problems with run-off into water bodies. Issues on welfare and wellbeing were found for bio-PE, PLA and fossil-PE. Exploitation is reported for both Brazil and the United States, as well as for oil-producing countries. When viewed from an economic aspect, bio-PE is at the moment 40% more expensive than fossil-PE. PLA is 60% more expensive. This is a major hurdle in the economic sustainability of bio-PE and PLA. Paragraph 4.5 shows that the gap might be closed by implementing policies like green VAT and carbon tax.

5. EVALUATION OF THE ANALYTICAL FRAMEWORK AND INDICATORS / BOUNDARIES

Chapter 5 evaluates the analytical framework and indicators/boundaries proposed in chapter 3. The evaluation is based on the case studies presented in chapter 4. Paragraph 5.1 evaluates the representation of the analytical framework. Paragraph 5.2 evaluates the criteria and boundaries given in table [3.1]. An evaluation is given in paragraph 5.3 on the NTA 8080 certification criteria, which served as the basis for the indicator development in this research. The chapter concludes with some key findings.

5.1 The analytical framework

The analytical framework covered the three dimensions of sustainability. Each dimension was subdivided into three main aspects. Each dimension could be extended with an aspect called for example “minor influences”. This aspect could be integrated to cover for minor characteristics that are not mentioned under one of the other aspects. Furthermore, the economic sustainability can to a large extend be determined by the marketing of “green” products. Although this is reflected in sales, the importance might justify the construction of a separate aspect. The policies, market forces and other influences cover a wide range of aspects. This research focussed on sustainability, so grouping all these aspects for a short evaluation is considered adequate. However, to fully comprehend the bio-supply chain on all these aspects, it is recommended that the group “policies, market forces and other influences” is extended to give more attention to these aspects.

5.2 Indicators/boundaries

This paragraph discusses the indicators and boundaries that are given in table [3.1]. An evaluation is given on the sufficiency of these indicators and boundaries to analyse the different aspects of sustainability. The evaluation is done on the basis of the case studies provided in chapter 4.

Environmental indicators/boundaries

Environmental indicators/boundaries were constructed for three aspects of environmental sustainability; GHG emissions, biodiversity and the local environment.

GHG emissions – The indicators/boundaries set on GHG emissions seem to be adequate. The reduction minimum can be adjusted when global, European, or national goals for GHG emissions reduction are increased to more than 50%. The fact that current PLA production does not meet the target could indicate that the 50% reduction minimum is too strict for bioplastics. However, bio-PE does meet the target. Furthermore, the study by Vink *et al.* (2003) shows that further reduction in GHG emissions is possible for PLA in the near future. A 50% reduction target is therefore considered adequate. More attention needs to be paid to indirect land use change, since this can have a large effect on the life cycle GHG emissions.

Biodiversity - Although the indicators/boundaries set on biodiversity can be considered adequate, major drawback is the dependence on local, national and international law for the determination of protected land. Protecting land is more than often under debate, especially when the land is suitable for economic use.

Local environment – The local environment is a broad definition which encompasses the whole local system. Many more indicators and boundaries could be constructed which address the local environment, for example on toxic waste material. For this research, the indicators used are assumed to address the most important issues surrounding the production of biomass. For further research, it is recommended that all issues are examined and evaluated according to their importance.

Social indicators/boundaries

Social indicators/boundaries were constructed for three aspects of social sustainability; competition for food, welfare and wellbeing.

Competition for food – The main boundary is that there should be no reduction of available food. This also means that prices of food cannot rise to a great extent due to biomass use for other purposes. Considering the global poverty and malnourishment problems, this indicator is considered vital for the determination of sustainability.

Welfare – The aspect welfare is centred on stimulating the local economy. Welfare is mainly determined by reports on several qualitative indicators. Certain quantitative indicators like job creation, income compared to national average and percentage of local hiring could be added to ease the process of comparison, and to set strict indicators as opposed to soft reporting obligations.

Wellbeing – Wellbeing is centred on the general compliance with human rights. There is no discussion on the importance of meeting these demands. More important might be the compliance with, and implementation of, local and international law. Therefore it is recommended that an indicator is added to the aspect wellbeing which allows for the assessment of law implementation.

Economic indicators/boundaries

Economic indicators/boundaries were constructed for three aspects of economic sustainability; Sales, total capital investment and operational costs. Only the market price indicator for sales was evaluated in this research.

Market price – Because of limited available data, the market price was used to assess the potential of economic sustainability. This approach, however inevitable in this research, is considered to be unsatisfactory for determining the economic sustainability. For a full assessment on economic sustainability, a range of additional indicators should be analysed. Consider for instance the marketing of “green” products, company reputation etc. Incorporating such indicators would allow for a more in-depth approach. Even so, market prices do give an indication on the substitution potential. Especially for identical products like fossil-PE and bio-PE, consumers would not be willing to pay a price which is significantly higher than what they are used to. Considering this, the indicator market price could be extended with a limitation on premium paid for biobased products compared to fossil-based products.

Policies, market forces and other influences

In this research, policies, market forces and other influences were described separately from the sustainability dimensions. During the writing of the report, it proved to be difficult to separately describe sustainability and policies. Policies, market forces and other influence effect all aspects of the supply chain and its’ sustainability aspects. For further research, it is recommended that policies, market forces and other influences are not viewed separately, but are incorporated into the sustainability assessment.

NTA 8080

The indicators given in table [3.1] were constructed mainly on the basis of the Dutch NTA 8080 certification. Therefore, it is deemed appropriate to give a few comments on the certification scheme in light of this research. First, the NTA 8080 certification is constructed on biomass for energy. Therefore, adopting biomass for products in the certification could facilitate sustainability assessments of biomass products as well as energy. Second, compliance with local law and regulation is mentioned for several indicators as a boundary. However, local law and regulation might be ill-designed to comply with desirable sustainability levels. To assess sustainability in a desired fashion, two options might be applied; 1) integrating desired standards for law and regulation into the certification or 2) using Dutch laws and regulations as standard to which sustainability is tested. Third, several indicators only require business plans or reporting obligations from the involved companies. This approach leaves considerable room for freedom of interpretation. Assessment of the validity of such business plans and reports, as well as on-site inspection by a third independent party could cancel the risk of false reporting (either knowingly or unknowingly).

5.3 Key findings

Improving the indicators can be done by 1) adding indirect land use change effects as a separate indicator for GHG emissions and biodiversity, 2) adding the evaluation of conservation value of land, since designated protected areas might not suffice, 3) adding compliance to law, which might be of more importance than the written presence of law and regulations and 4) extending the economic indicators with consumer acceptance of price premiums, marketing, profits, corporate image etc.

6. CONCLUSION & DISCUSSION

In this research report, an analysis was performed on the sustainability of biobased product supply chains compared to a fossil reference. The contribution of this research to current scientific research is twofold. First, it provides a framework for the sustainability assessment of biobased products. Second, the research provides an assessment of the current state of two products with could become important pillars of a biobased economy. The framework was constructed to answer the main research question:

How sustainable are the supply chains for biobased polyethylene and polylactic acid, which could meet the current demand for fossil-based polyethylene in the Netherlands, and how does this compare to the supply chain of fossil-based polyethylene?

To answer this, the report discusses several indicators on the three dimensions of sustainability; environmental, social and economic. The results are visualized in figure [6.1].

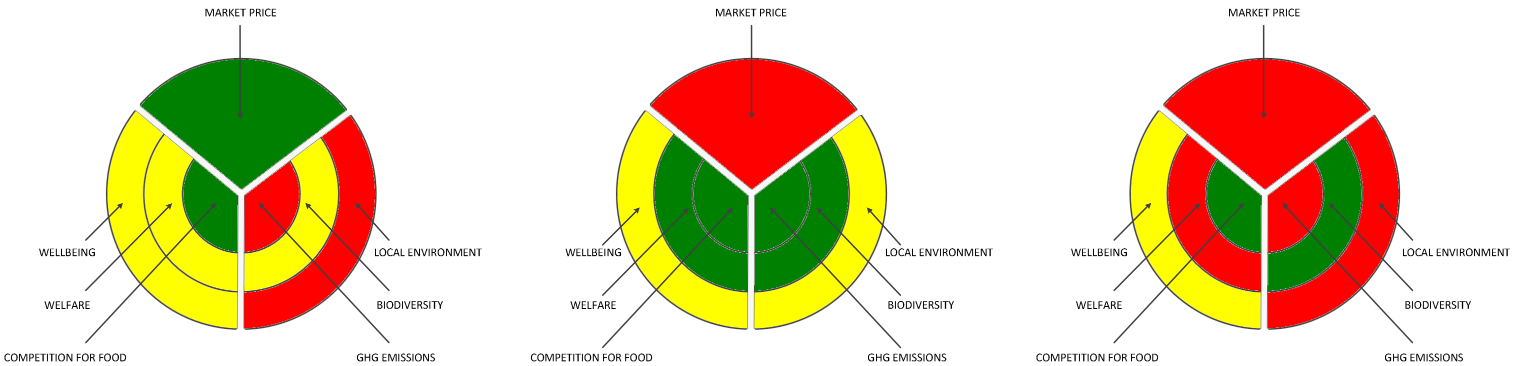


Figure [6.1] Visualization of the sustainability for fossil-PE, bio-PE and PLA. Green: all indicators/boundaries are met for the related aspect. Yellow: either the indicators/boundaries are only partially met, or the indicators/boundaries are only met in some situations. Red: Some or all of the indicators related to the aspect are not met.

The main answer to the research question is that if sustainability implies satisfying all indicators, bio-PE and PLA are currently not sustainable. Bio-PE is more sustainable than fossil-PE on social and environmental dimensions, but the high market price limits the economic sustainability. PLA is currently less sustainable than fossil-PE[7].

Both bio-PE and PLA showed negative results for important indicators. For bio-PE, the main obstacles are the negative nutrient balance (*local environment*) and concerns about exploitation of workers (*wellbeing*). For PLA, the main obstacles are the high level of GHG emissions, the negative score on most indicators on the local environment, and the disadvantaged position of field workers (*welfare & wellbeing*). Note that bio-PE scores better on overall environmental and social sustainability than fossil-PE. PLA scores worse on social sustainability, depending on which country for fossil-PE is taken into account. PLA scores slightly better on environmental sustainability, due to the positive results on biodiversity.

Both bio-PE and PLA showed negative results for the market price, compared to fossil-PE. For biobased product supply chains to be economically sustainable, the market should drive the demand. Market prices for biobased products are too high with respect to their fossil-based counterparts. However, stating that biobased products are too expensive is misleading. A more correct statement would be that fossil-based product prices are not based on the total true costs, as environmental damage is not taken into account in the market price. Therefore, fossil-based product prices are in fact artificially low. This report shows that with regard to the aspect environmental sustainability, fossil-PE scores worse than both bio-PE and PLA. GHG emissions are higher, production takes place in

[7] Note that this might depend on the reference country for fossil-PE (mainly influences welfare and wellbeing).

protected regions and the impact on the local environment can be considerable. These environmental externalities are for a large part not included in the market price of the product. Society as a whole pays for such externalized costs. This leads to a "tragedy of the commons", where resources held in common responsibility deteriorate because the rational choice of the individual (company) is to exploit the resource (Hardin, 1968). This process is being stimulated by the subsidies on fossil resources. For instance, the EU subsidized fossil energy production with 30 billion euro in 2005 (Jefferson, 2008). Instead of internalizing the externalized costs, the use of fossil resources is encouraged, creating an imbalance of market prices of fossil-based and biobased products. This imbalance gives a disproportional advantage to the demand-driven potential of fossil-based products. Public awareness of depleting resources, sink pollution and the environmental impacts in general are often not enough to overcome this imbalance in product price. For biobased products to become truly economically sustainable, the market price should to a large extent be levelled with the fossil-based product it aims to replace. Implementing subsidies would create artificially low prices for the biobased product. A more sustainable approach would be to include true cost (externalities) in the market price of fossil-based products. Including externalities and cutting subsidies levels the economic playing field for fossil-based versus biobased products. Green VAT strategies and carbon taxes are examples of internalizing the externalities. However, these strategies include only some of the costs for environmental damage. Many more costs are involved than those costs that can be directly expressed in monetary value (McKinney *et al.*, 2007). For example, the aesthetic value of biodiversity is difficult to express in monetary value. These costs are so-called intangible costs. Furthermore, there might be hidden costs that are not recognized yet by current scientific knowledge. The same holds for future costs, which represent long-term impacts that cannot be foreseen in the short run. Including such costs is difficult, but it might be the most effective way of constituting a shift towards a more sustainable, biobased economy.

A further remark can be given about the economic sustainability of biobased products. Currently, the demand for biofuel is larger than the demand for other biobased products. World use of fuels is ~100 EJ, while the use of plastics is only (converted to energy) ~8 EJ (Wetenschappelijke en Technologische Commissie voor de Biobased Economy., 2011). Sugarcane and corn are extensively used to meet the demand as a "sustainable" alternative for fossil fuel. As is discussed in the introduction, the value gained from using biomass for fuel is significantly less than the value gained if biomass is used for chemicals. If the above statements are combined with the fact that the demand, and therefore impact on the environment, is smaller for biobased products, it is hard to see why so much emphasis should be put on biofuels. It would make more sense to use biomass for biobased products like chemicals, and to invest into potentially clean and environmentally benign systems like solar-hydrogen for fuel.

Finally, an important comment must be given on the choice of biomass feedstock for a biobased economy. By current estimates, 500 million people are chronically hungry worldwide and 8 to 11 million people die each year because of hunger and malnutrition (McKinney *et al.*, 2007). With an ever growing world population, the limit of the carrying capacity of our planet is almost reached. One might even argue that we are already in an overshoot situation, and that hunger and malnutrition are therefore likely to increase. Using food crops as feedstock to such an extent that it competes with food is therefore morally undesirable. As this research shows, a small demand for food crop feedstock does not significantly impact food prices or availability. This research, however, only focuses on the demand for one plastic for the Netherlands. If multiple products are considered with worldwide demand, the use of food crop feedstock inevitably leads to competition with food. This notion coincides with the growing consumption of products and goods on a global scale. Especially using main cereals like corn as biobased feedstock would give rise to enormous pressures on the availability and affordability of food. The focus should therefore shift to other biomass feedstock sources, such as lignocellulose material like the roots and stems of food crop or woody biomass like trees. Roots and stems are co-products of food crop, and are thus produced on crop land without direct competition for food. Woody biomass can grow on marginal lands, preventing competition with crop land. Only if the competition for food is avoided will a biobased economy be truly sustainable.

RECOMMENDATIONS FOR FURTHER RESEARCH

Main recommendation is to evaluate the indirect land use change caused by increased production of biomass. This is not only true for biomass production for feedstock, but also for food and fuel. Indirect land use change is as important as direct land use change, but is still badly understood. Additional research would therefore greatly attribute to current scientific knowledge.

Since this research focused on sustainability and not on policies and other influences, it is recommended that the research is extended with an in-depth view of such influences. As an addition to policies, it is recommended that the compliance level is also taken into account.

Finally, to gain a better view of a biobased economy, it is recommended that this research is repeated for several products, for different levels of demand, for different regions of production and for different feedstocks for each product.

APPENDIX A: FRAMEWORK CONSTRUCTION AND INDICATOR DETERMINATION

This appendix provides a framework along which the sustainability of specific bio-supply chains can be analysed. First, a general view on sustainability is given. Second, the three dimensions of sustainability (economic, environmental and social sustainability) are discussed. Each dimension is divided into three aspects. For each aspect, indicators and boundaries are given, along which bio-supply chains can be analysed. Third, additional influences are discussed which might impact the characteristics of bio-supply chains, but do not fit into the sustainability dimensions.

Supply chain sustainability

Sustainability is a key concept in this research. Sustainability was first described as sustainable development and was defined by the Brundtland committee in 1987 as followed: "sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (Barnaby, 1987). In his book "Cannibals with Forks: the Triple Bottom Line of 21st Century Business" Elkington described corporate sustainability as a triple bottom line concept, also referred to as people, planet, profit or economic, environmental and social sustainability (Elkington, 1997). These three dimensions of sustainability are interconnected and cannot be viewed separately. The sustainability dimensions are shown in figure [A-1].

Figure [A-1] The three dimensions of sustainability

Important to note is that even though these definitions are on corporation and businesses, they are just as applicable to bio-supply chains since supply chains are effectively built up from company activities along the value chain. The following three paragraphs discuss each dimension in detail.

Environmental sustainability

"Ecologically sustainable companies use only natural resources that are consumed at a rate below the natural reproduction, or at a rate below the development of substitutes. They do not cause emissions that accumulate in the environment at a rate beyond the capacity of the natural system and absorb and assimilate these emissions. Finally, they do not engage in activities that degrade eco-system services." (Dyllick & Hockerts, 2002)

For the Netherlands, a system of criteria and indicators was proposed by the committee "sustainable production of biomass", which essentially provides assessable criteria for sustainable biomass production from field to consumer (Projectgroep Duurzame Productie van Biomassa., 2006; Projectgroep Duurzame Productie van Biomassa., 2006). The criteria were named Cramer criteria after the committee chair. The goal of the committee was to provide the Dutch government with assessable criteria for sustainable import of biomass. Sustainability was viewed not only on environmental levels, but on social levels as well. These criteria were formulated in such a way that they were applicable to all types of biomass and all countries of origin (Projectgroep Duurzame Productie van Biomassa., 2006; Projectgroep Duurzame Productie van Biomassa., 2006)). Six different areas of importance were defined. For the purpose of this research, the six different areas of importance are divided over environmental and social sustainability. The three areas on social sustainability are discussed later.

Cramer criteria on environmental sustainability:

- Greenhouse gas emissions
- Biodiversity
- Local environment

The sustainability dimensions given in figure [A-1] characterize the bio-supply chain. Figure [A-2] shows the sustainability dimensions in relation to the bio-supply chain. Furthermore, it shows the aspects of environmental sustainability mentioned above.

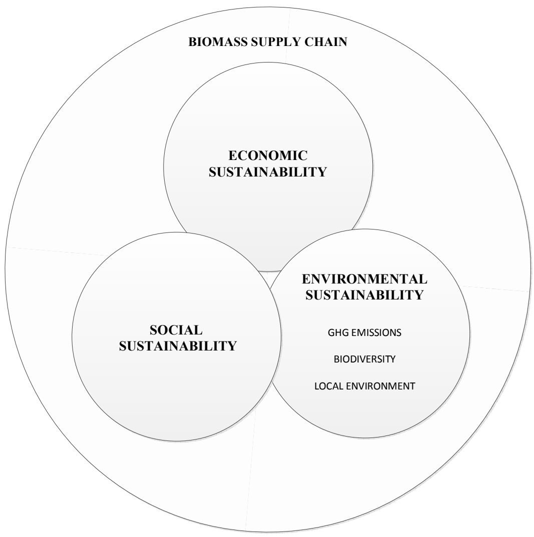


Figure [A-2] The three dimensions of sustainability including environmental sustainability aspects

Environmental sustainability indicators/boundaries

The indicators and boundaries for environmental sustainability are mainly based on the NTA 8080 norm system. The NTA 8080 norm system and the NTA 8081 certification system were developed based on the Cramer criteria. The NTA 8080 system provides an extensive list of demands which have to be met in order to be depicted as sustainable (NEN, 2009). The NTA 8081 is the certification process for companies involved in the bio-supply chain. On an international level other certification systems are developed to assess the sustainability of bio-supply chains, like the ISCC certification (ISCC Association, 2010) and the RSB certification (RSB, 2010). The three systems are all based on biomass production for energy supply. To a large extent, the same criteria can be used for bio-supply chains. The NTA 8080, ISCC and RSB are compared and analysed to provide a set of indicators which could evaluate the bio-supply chain. The indicators based on the norms provided in the NTA 8080, ISCC and RSB are given in table [3-1]. The criteria were adapted so that they were applicable for a supply chain for biobased products where needed.

Social sustainability

"Social sustainable companies add value to the communities within which they operate by increasing the human capital of individual partners as well as furthering the social capital of these communities. They manage social capital in such a way that stakeholders can understand its motivations and can broadly agree with the company's value system." (Dyllick & Hockerts, 2002)

The Cramer criteria given below on social sustainability hold the same values as given by Dyllick.

Cramer criteria on social sustainability:

- Competition for food (and local energy supply, medicine and building materials)
- Welfare
- Wellbeing

As with the criteria on environmental sustainability, the aspects are described in the NTA 8080. Norms given for each aspect are described below and form the basis for research on social sustainability of a bio-supply chain. Figure [A-3] shows the extended framework.

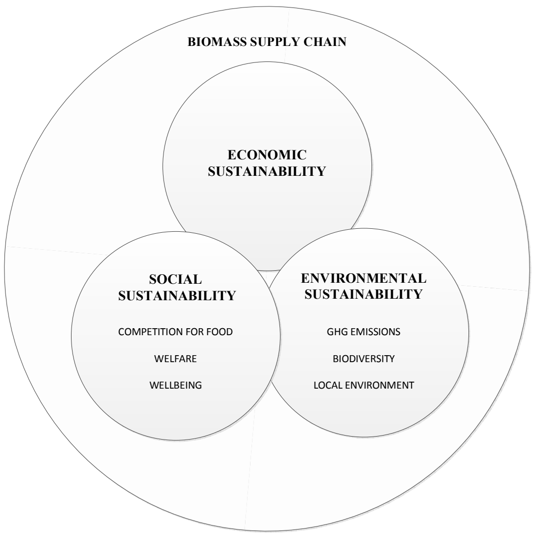


Figure [A-3] The three dimensions of sustainability including environmental and social sustainability aspects

Social sustainability indicators/boundaries

The social sustainability indicators are composed in the same matter as the environmental sustainability indicators. The NTA 8080, ISCC and RSB norms are compared and adapted for this research. The indicators and boundaries that will be used in the analytic framework are summarized in table [3-1]. It should be noted that the indicators on social sustainability are more on basis of reporting than on measurable indicators.

Economic sustainability

"Economically sustainable companies guarantee at any time cash-flow sufficient to ensure liquidity while producing a persistent above average return to their stakeholders" (Dyllick & Hockerts, 2002)

The economic sustainability of a bio-supply chain is of particular interest when viewing the potential of biobased products to replace fossil-based products. Since consumers are still driven to a large extent by cost, rather than environmental awareness, the additional costs incurred for more expensive biobased products might withhold consumers from purchases. This would lead to an economically unsustainable supply chain. Biobased products should therefore not exceed the price for a fossil-based product it aims to replace to a large extent. Traditionally, when assessing economics, one would consider fixed capital and current operational capital. However, when considering sustainability a distinction between three forms of capital can be made (Dyllick & Hockerts, 2002).

- Financial capital (i.e. equity, debt)
- Tangible capital (i.e. Machinery, land, stocks)
- Intangible capital (i.e. reputation, inventions, know-how)

However, when analysing the sustainability by methods and models, intangible capital is difficult to assess. Current studies on bio-supply chains (Bowling *et al.*, 2011; Dornburg *et al.*, 2006) therefore focus on financial capital and tangible capital. They use the following primary indicators to assess the economic sustainability of bio-supply chains:

- Total sales
- Total capital investment
- Operational costs

The sustainability framework given in figure [A-3] can be extended with the economic aspects given above. The extended sustainability framework is shown in figure [A-4].

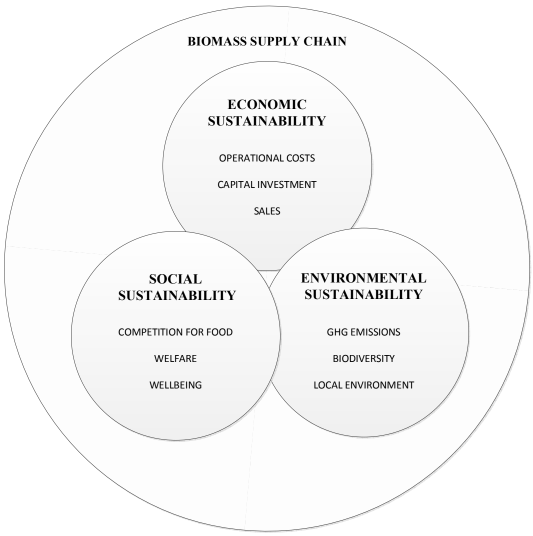


Figure [A-4] The three dimensions of sustainability including environmental, social and economic sustainability aspects

Economic sustainability indicators/boundaries

As shown, economic sustainability aspects are divided into sales, capital investment and operational costs. Sales are determined by products sold times the price of product. For bulk goods this is often done by weight or, in case of liquids, volume. However, sales depend on various aspects which are beyond the scope of this research. The aspects are for example marketing and reputation of the product/company. Market price is expected to be available, and will serve as the main indicator for this research.

A model is constructed by van Dam which divides the operational costs and capital investments into *levels of input*, *required man- hours*, *fuel use and machinery* and *land value* (van Dam *et al.*, 2007). Additional costs are incurred like facility investments, taxes, royalties and lab charges (Bowling *et al.*, 2011). To gain a more complete overview, the model given by van Dam is adapted to include such costs, see figure [A-6]. Information on costs and investment are market sensitive information, and are expected to be hard to find. The relevant indicators to determine sales, total capital investment and operational costs are summarized in table [3-1].

Policies, market forces and other influences

When determining the different characteristics of bio-supply chains, some forces and influences need to be taken into account. Although these forces are not part of the supply chain, the effect they have can be rather significant. Two of the major influences on production of goods are policies and market forces. Market forces are perhaps most influential, since production of goods finds its origin in consumer demand. With the current tendency towards greener products, consumers facilitate a shift towards biobased products. However, an obstacle to biobased product introduction could be the might of the petrochemical industry, which is well-established, entrenched and profitable, relying on low feedstock prices (Dale, 2003). Policies can influence this consumer behaviour by instigating additional taxes on fossil fuels (like carbon tax), instigating import & export tariffs or subsidizing biobased products. Furthermore, when production of biomass takes place in a developing country, policy emphasis can be put on economic growth rather than on social or environmental considerations. The institutional character and regulatory profile can therefore to a great extent determine many variables of the bio-supply chain, like configuration and costs, which in turn impact environmental and social issues. Even though institutional character and regulatory profile are not characteristics of the bio-supply chain itself, they need to be investigated to gain a complete view all aspects of biomass production. In the framework proposed in the next paragraph a report phase is proposed in which these aspects are analysed.

Policies, market forces and other influences indicators/boundaries

Policies, market forces and other influences are difficult concepts to evaluate in concrete numbers or indicators. To be able to integrate such influence into the analytical framework, a qualitative report will be given on the status of such influences. A focus will be put on the institutional character and regulatory framework. The final framework including these influences is shown in figure [A-5].

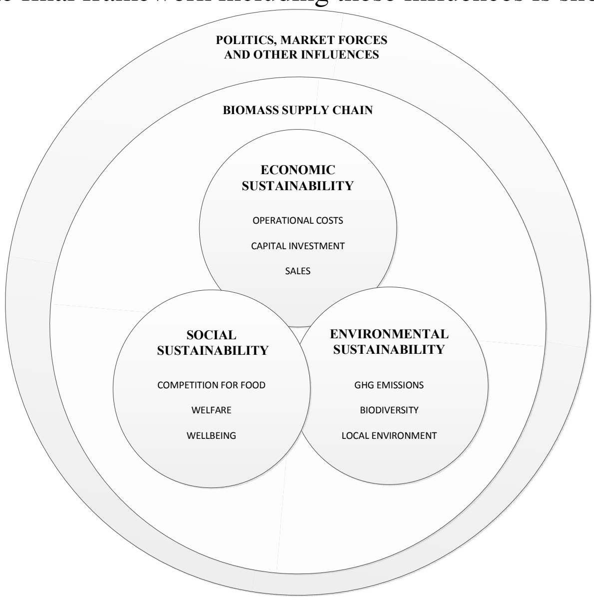


Figure [A-5] Analytical framework including Economic, environmental and social sustainability aspects and policies, market forces and other influences

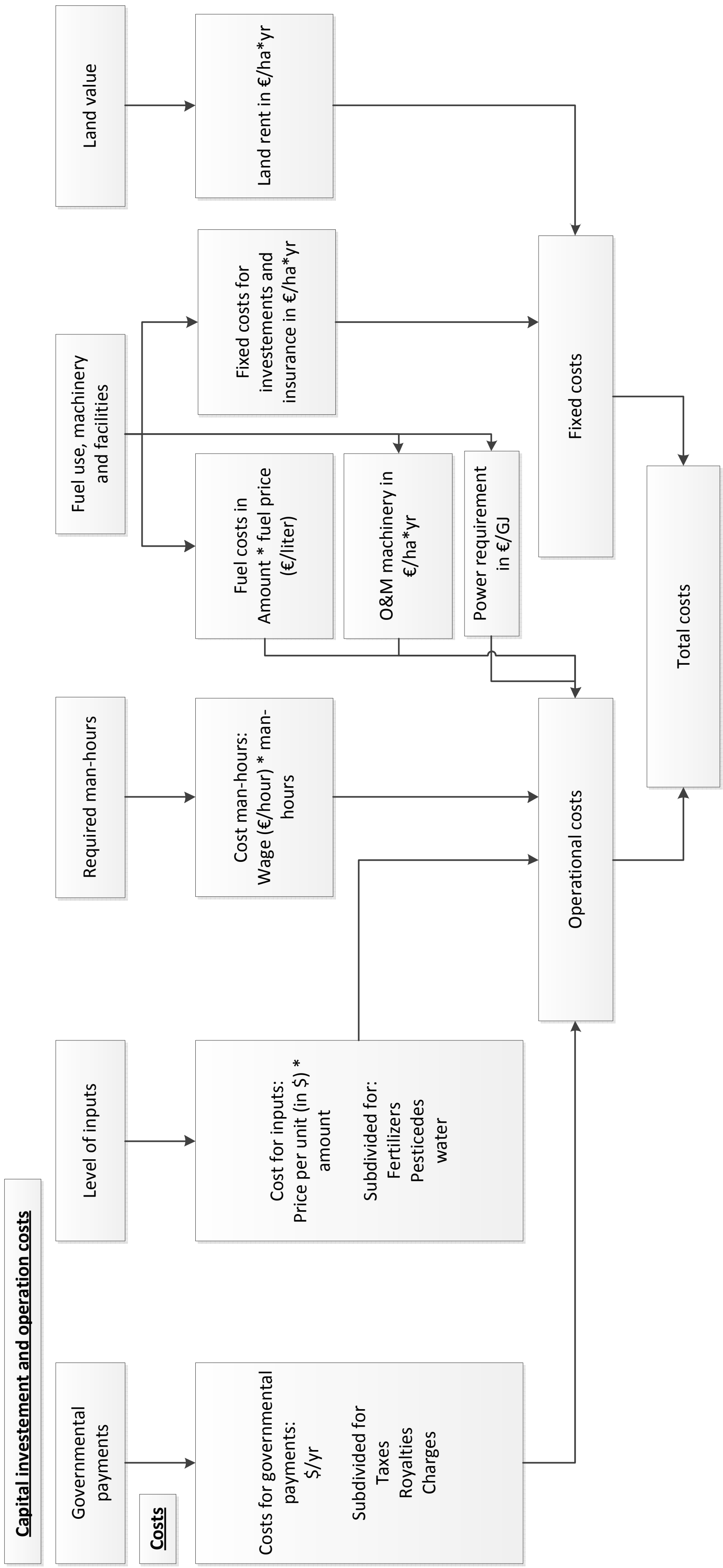


Figure [A-6] Extended van Dam model

APPENDIX B: SCHEMATIC OVERVIEW OF ETHANOL PRODUCTION

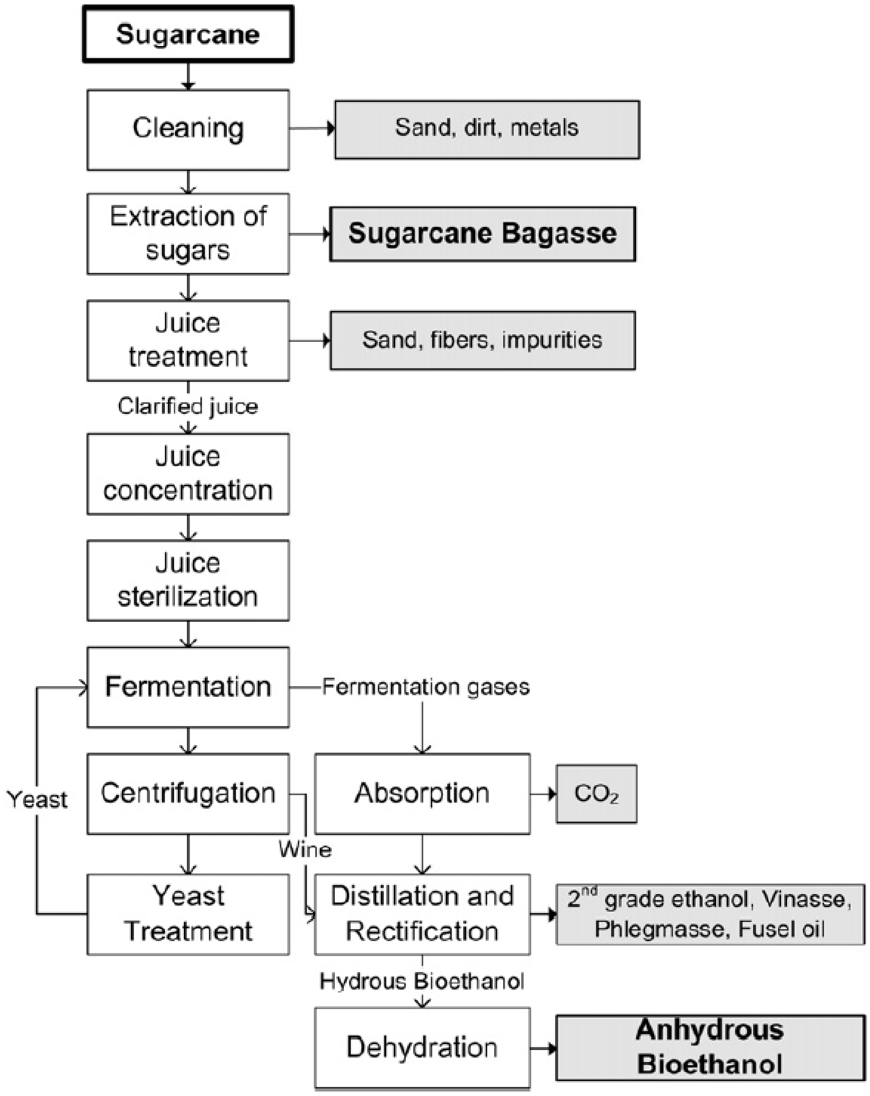


Figure [B-1] Schematic overview of ethanol production. Source: (Dias *et al.*, 2011)

APPENDIX C: GREENHOUSE GAS EMISSIONS

The values given for GHG emissions are scarcely the same for two different reports. Differences occur for example through the definition of system boundaries or differences in production methods. For this research, cradle-to-factory gate studies were used and emissions for incineration were added to the results. Only analyses were used that calculated emissions for CO_2, CH_4 and N_2O either separately or as CO_2 equivalent. The original values were converted to kg CO_2-eq/kg product for comparison using the conversion table [C-2]. Full incineration following stoichiometry was used, resulting in 3.14 kg CO_2-equivalent/kg polyethylene and 1.83 kg CO_2-equivalent/kg PLA. Since no information was found on HDPE or LLDPE, the evaluation was based on LDPE.

Greenhouse gas emissions for fossil-PE

The outcomes of the SimaPro database, the SPINE LCI dataset (CPM, 2012), the EAP database and a study by Liptow and Tillman (2009) were combined to give a range of emission output. This is shown in table [C-1]. The emissions are given in kg CO_2 equivalent. The range of emissions will serve as the fossil reference for the research on bio-supply chains. Note that the factory gate-to-grave cycle is the same for fossil-PE and bio-PE. However, excluding this cycle from the analysis would result in different reduction percentages if both types of polyethylene are cross referenced. Since the given indicators are based on life cycle GHG emissions and energy requirements, the complete life cycle is taken into account.

Table [C-1] Life cycle GHG emissions for bio-PE

Study/database	kg CO_2-equivalent / kg LPDE	kg CO_2-equivalent / kg HDPE
SimaPro database	5.20	5.03
SPINE LCI dataset	5.16	4.95
EAP	5.21	5.10
Liptow & Tillman (2009)[a]	5.30	-
Total range	5.16-5.30	4.95-5.10

[a] For this research, the attributional LCA given in the report is used.

Greenhouse gas emissions for bio-PE

Data on cradle-to-grave GHG emissions from the production of bio-PE is very limited. Several studies are however available on the cradle-to-gate GHG emissions for the production of ethanol. These studies are presented in table [C-5]. Furthermore, each study was analysed on which specific aspects of the life cycle GHG emissions were included. Missing values were extrapolated for all studies using values which were available. All values are converted to CO_2-equivalent/kg product using table [C-2].

Table [C-2] Conversion values

	Value	source
L/ha/yr	5900	Macedo (2004)
density ethanol (kg/L)	0.789	www.densityofethanol.com[a]
Conversion rate (eth/et) high	1.67	stoichiometry & Morschbacker (2006)
Conversion rate (eth/et) low	1.82	stoichiometry & Morschbacker (2006)
CO2 emission dehydration	0.368	Liptow & Tillman (2009)
CO2 emission polymerization	0.246	Liptow & Tillman (2009)
LHV ethanol (MJ/L)	24.81	Seabra *et al.* (2011)

[a] at 20 degrees centigrade

Original and added values

Table [C-5] shows the different LCAs used. The light grey markings in table [C-5] show those values which were presented in the original LCA. The dark grey marking shows those values which were extrapolated to the original LCA's. The extrapolated values are discussed below.

Value for CO_2 sequestering: By using stoichiometry, the CO_2 emissions from burning bio-polyethylene are calculated as being 3.14 kg CO_2/kg bio-polyethylene. Since all carbon in bio-polyethylene comes from sugarcane, and since all carbon in sugarcane is taken up by photosynthesis, the same amount of CO_2 sequestering can be attributed to bio-polyethylene. This value is extrapolated to all LCA's.

Values for dehydration and polymerization: One study was found which gave values for dehydration and polymerization. The values giving by Liptow & Tillman (2009) for dehydration (0.37 kg CO_2/kg bio-polyethylene) and polymerization (0.25 kg CO_2/kg bio-polyethylene) are extrapolated to all LCA's. Note that Liptow & Tillman only give values for LDPE.

Value for transport of ethanol: Two studies gave values for the transport of ethanol. Seabra et al. (2011) gave a higher value than De Oliviera et al. (2005). The value given by Seabra et al. (0.04 kg CO_2/kg bio-polyethylene) is based on 340 km of transport. The value given was extrapolated for all LCA's.

Value for transport Porto Alegre – Rotterdam: The value for GHG emitted during sea transport was calculated by using the eco transit tool[8]. Entered parameters were: 500 kton of goods, average goods in bulk, only sea transport. Start at Porto Alegre, finish at Rotterdam. The resulting 0.08 kg CO_2/kg bio-polyethylene was extrapolated to all LCA's

Value for incineration: As stated above, the value for incineration was obtained by using stoichiometry. 3.14 kg CO_2/kg bio-polyethylene was extrapolated to all LCA's.

The resulting values are given in the last column of table [C-6]. The BREW study was excluded from further analysis because of strong deviating values compared with the other eight LCA's, which could not be explained by analysing the study.

[8] http://www.ecotransit.org/index.en.phtml

Differences in including LCA aspects

Certain aspects were included in some studies, while excluded in others. No changes were made to the different outcomes, but the different views are given below.

Emissions from methane: Only Seabra et al. excluded emissions from methane because it contributed only in a minor way to the total greenhouse gas emissions.

Emissions from trash burning: Pereira & Ortega and Oliviera *et al.* argue that emissions from thrash burning were taken up by plants making the GHG balance zero, and are therefore not taken into account. Other studies however also include N_20 emissions from trash burning, which is not taken up by plants.

Embodied emissions in buildings, machinery and equipment: Due to the relatively small contribution to the overall greenhouse gas emissions, some authors chose to exclude those numbers. This has no significant effect on the outcome of the LCA's.

Reduction of emission by using surplus biomass used for process energy - Emissions can be reduced by replacing certain boilers in the process by more efficient ones. Some authors chose to include these avoided emissions, while others did not.

Reduction of emissions by using surplus biomass for electricity generation – Emissions can be reduced by using surplus biomass to generate electricity. This "green" electricity would replace standard electricity. Oliviera et al. argues that since the standard electricity mix for Brazil is mostly renewable (i.e. hydrogen dams), no replacement of greenhouse gas emissions takes place. However, greenhouse gas emission from Brazilian electricity supply has an emission factor of 80 gr CO2eq/kWh as reported by the IEA (IEA, 2009). The most influential other, Macedo et al., did include the replacement of fossil-based energy supply. However, he used the world average of 560 t CO2eq/GWh, which is seven times higher than the Brazilian emission factor. Note, Macedo et al. did use the Brazilian emission factor to argue that electricity supply to the sugarcane mill did not significantly contribute towards GHG emissions.

The resulting values for kg CO_2-eq/kg bio-LDPE given in Table[C-6] are used to determine a bandwidth for GHG emissions. For the studies the mean and standard deviation were determined. The bandwidth is taken as the result of the *mean ± standard deviation,* which is shown in table [C-3].

Table [C-3] Review results for life cycle GHG emissions for bio-PE compared to fossil-PE

	kg CO_2-eq/kg bio-LDPE
Mean	1.36
Standard deviation	0.21
Bandwidth of GHG emissions (Mean ± Standard deviation)	1.15 - 1.57
Fossil reference[a]	5.16 – 5.47

[a] See paragraph 4.2

Figure [C-1] gives a graphical representation of the values in table [C-3]. The resulting GHG emissions reduction ranges from ~70% (equation 1) till ~80% (equation 2).

(1) $$\left(1-\frac{High\ range\ GHG\ emissions\ from\ bioLDPE}{Low\ range\ GHG\ emissions\ from\ fossil\ LDPE}\right)\times 100\%$$

(2) $$\left(1-\frac{Low\ range\ GHG\ emissions\ from\ bioLDPE}{High\ range\ GHG\ emissions\ from\ fossil\ LDPE}\right)\times 100\%$$

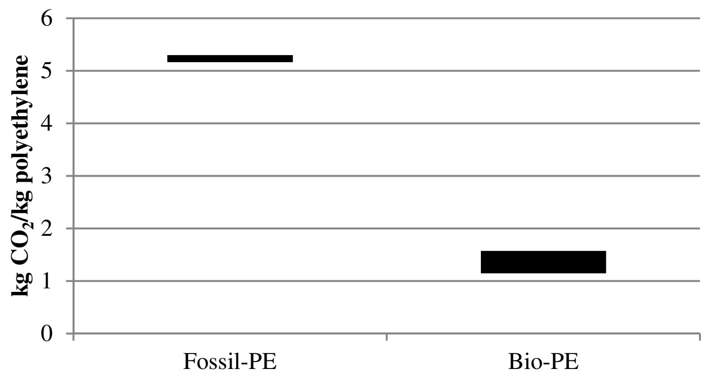


Figure [C-1] Life cycle GHG emissions for bio-PE compared with fossil-PE

This comparison excludes unburnt sugarcane practices for comparison reasons. Burning of sugarcane is however being legally phased out (Macedo *et al.*, 2008). Subtracting unburnt GHG emissions from burnt emissions shows a reduction of GHG emissions, of (Lisboa *et al.*, 2011); Liptow & Tillman, 2009)~0.2 kg CO_2-eq/kg bio-LDPE (Lisboa *et al.*, 2011). The resulting reduction of GHG emissions would range from ~74% (equation 1) till ~83% (equation 2).

Greenhouse gas emissions for PLA

An life cycle analysis study on natureworks' PLA was used to show the life cycle GHG emissions and energy requirements (Vink *et al.*, 2003). These results were verified by data from the Simapro database. Note that since the Simapro database also used a significant amount of information from the study by Naturework result may be biased. CO_2 sequestering and incineration was added to both the study of Vink *et al*. (2003) and the SimaPro results. Full incineration following stoichiometry was considered, resulting in 1.83 kg CO_2-equivalent/kg PLA, which is equal to the CO_2 sequestering. The results are shown in table [C-4].

Table [C-4] Review results for life cycle GHG emissions for PLA compared to fossil-PE

Study/database	kg CO_2-equivalent / kg PLA
SimaPro database	4.18
Vink *et al.* (2003)	3.63
Total range	3.63 - 4.18
Fossil reference[a]	5.16 – 5.47

[a] See paragraph 4.2

Figure [C-2] shows the resulting values of table [C-4] in comparison to the values found for fossil-PE. The resulting GHG emissions reduction ranges from ~19% (equation 1) till ~34% (equation 2).

(1) $$\left(1 - \frac{High\ range\ GHG\ emissions\ from\ bioLDPE}{Low\ range\ GHG\ emissions\ from\ fossil\ LDPE}\right) \times 100\%$$

(2) $$\left(1 - \frac{Low\ range\ GHG\ emissions\ from\ bioLDPE}{High\ range\ GHG\ emissions\ from\ fossil\ LDPE}\right) \times 100\%$$

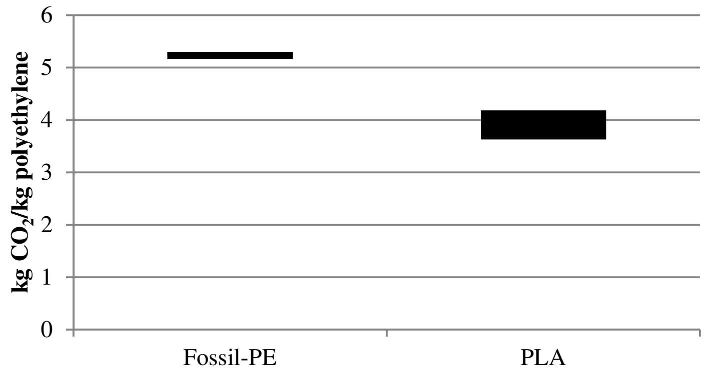


Figure [C-2] Life cycle GHG emissions for PLA compared with fossil-PE

Table [C-5] Conversion of LCA's to bio-PE LCA's by extrapolation of missing values

		Macedo (2008)	Seabra (2011)	Crago (2010)	Garcia (2011)	Lisboa (2011)	Liptow & Tillman (2009)	Oliviera (2005)	Pereira (2010)	BREW, Patel (2006)
Agricultural	CO_2 sequestering	v	v	v	v	v	v	v	v	v
	fossil fuel use	x	x	x	x	x	x	x	x	x
	fertilizer, pesticide production	x	x	x	x	x	x	x	x	x
	Fertilizer, pesticide application	x	x	x	x	x	x	x	x	x
	emissions from methane	x	n.i.	?	x	x	x	x	?	x
	trash burning	x	x	x	x	x	?	n.i.	n.i.	x
Industrial	embodied GHG emission in buildings, machinery and equipment	x	n.i.	x	n.i.	x	n.i.	n.i.	x	x
	chemical production	x	x	x	?	x	x	x	x	x
	surplus Biomass used for process energy	x	x	?	x	x	n.i.	?	n.i.	x
	surplus Biomass used for electricity	x	x	x	x	x	n.i.	n.i.	n.i.	x
	dehydration	v	v	v	v	v	x	v	v	v
	polymerization	v	v	v	v	v	x	v	v	v
Transport	transport of sugarcane (20 km)	x	x	x	x	x	x	x	x	x
	transport of ethanol (340 km)	v	x	v	v	v	v	x	v	v
	transport Porto Alegre - Rotterdam (11323,23 km)	v	v	v	v	v	v	v	v	v
End-of-Life	incineration	v	v	v	v	v	v	v	v	v

x	Original in LCA
n.i.	Not included in LCA
v	Added to LCA
?	not mentioned or unclear

Table [D-3] Life cycle greenhouse gas emissions for bio-PE

Study	Original value	kg CO2-eq/kg bio-LDPE (excl. extrapolated values as indicated in table [D-2])[a]	kg CO2-eq/kg bio-LDPE (incl. extrapolated values as indicated in table [D-2])
Patel *et al.* (2006)	-0.9 t CO2-eq/t ethylene	-0.9	-0.15b
Macedo *et al.* (2008)[b,c]	215 kg CO2-eq/m3 ethanol	0.30	1.05
Seabra *et al.* (2011)[b,d]	21.3 g CO2-eq/MJ ethanol	0.73	1.49
Crago *et al.* (2010)[b]	362 kg CO2-eq/m3 ethanol	0.50	1.26
Garcia *et al.* (2011)[d]	26.6 kg cCO2-eq/GJ ethanol	0.91	1.67
Lisboa *et al.* (2011)N_2O 3.87[b,e]	4976 kg CO2-eq/ha/yr	0.79	1.55
Liptow & Tillman (2009)[f]	1.04 kg CO2-eq/kg bio-LDPE	-	1.18
De Oliveira *et al.* (2005)[g]	572 kg CO2-eq/m3 ethanol	0.79	1.55
Pereira & Ortega (2010)	0.28 kg CO2-eq/L ethanol	0.39	1.14

[a] conversion of original values by conversion factors given in table [D-1]
[b] Includes avoided emissions due to use of biomass surplus for process energy generation and energy surplus in form of electricity due to the use of biomass surplus. Excludes the replacement potential of fossil fuels of ethanol in cars.
[c] Based on hydrous ethanol. Avoided emissions based on HDE as given by Macedo et al. Includes trash burning as given by Macedo et al.
[d] Converted by using a Lower Heating Value (LHV) of 24.81 MJ/Litre (Lloyd, 2005)
[e] N_2O emission factor of 3.87%
[f] Attributional LCA. Without indirect land use change, without transport to Europe
[g] worst case situations was assumed

APPENDIX D: LAND USE CHANGE EFFECTS ON GREENHOUSE GAS EMISSIONS

Table [D-1] shows the values that were used to construct figure [4.2] in paragraph 4.2. The initial value for the "low" scenario is 1.15 kg CO_2 / kg bio-PE. The initial value for the "high" scenario is 1.57 kg CO_2 / kg bio-PE.

Table [D-1] Effect of land use change on life cycle GHG emissions

	Additional kg CO_2-eq/kg bio-polyethylene		cumulative kg CO_2-eq [a]/kg bio-polyethylene	
	Low	High	Low	High
Pasture-to-sugarcane conversion (20 Mg C/ha)				
5 year attribution	0.24	0.27	1.39	1.84
10 year attribution	0.12	0.13	1.27	1.71
15 year attribution	0.08	0.09	1.23	1.66
Cerrado-to-sugarcane conversion (45 Mg C/ha)				
5 year attribution	0.55	0.60	1.70	2.17
10 year attribution	0.27	0.30	1.42	1.87
15 year attribution	0.18	0.20	1.33	1.77
Rainforest-to-sugarcane conversion (201 Mg C/ha)				
5 year attribution	2.45	2.67	3.60	4.24
10 year attribution	1.23	1.34	2.37	2.91
15 year attribution	0.82	0.89	1.96	2.46

[a] original bandwidth (1.15-1.57) + additional kg CO_2-eq/kg bio-polyethylene

APPENDIX E: ENERGY REQUIREMENTS

The values given for energy requirements are scarcely the same for two different reports. Differences occur for example through the definition of system boundaries or differences in production methods. Therefore, multiple studies were analysed to provide a bandwidth of energy requirements. Full incineration without energy recovery is considered as end-of-life scenario. The analysis focusses on LDPE, since crucial data on dehydration and polymerization was only available for LDPE.

Energy requirements for fossil-PE

Multiple databases were cross referenced to gain a range of energy requirements. The SPINE LCI dataset did not provide information on energy requirements. Instead the study of Harding *et al.* was used to validate the values given by SimaPro and EAP. Simapro and EAP initially included the fossil feedstock from which the product was manufactured as inherent energy requirement (43MJ/kg polyethylene, Liptow & Tillman). In this research, the energy which is incorporated into the product is excluded for fossil-PE, bio-PE and PLA. The results are shown in table [E-1].

Table [E-1] Life cycle energy requirement for fossil-PE

Study/database	MJ/kg LDPE	MJ/kg HDPE
SimaPro database	38.1	35.7
Harding *et al.* (2007)	38.8	30.7
EAP	40.4	32.7
Liptow & Tillman (2009)[a]	41.5	-
Total range	38.1-41.5	30.7-35.7

[a] attributional LCA

Energy requirements for bio-PE

Data on cradle-to-grave energy requirements of bio-PE is very limited. Several studies are however available on the cradle-to-gate energy requirements for the production of ethanol. These studies are presented in table [E-5]. The original values were converted to MJ/kg bio-PE for comparison using the conversion table given in table [E-4]. Furthermore, each study was analysed on which specific aspects of the life cycle GHG emissions were included. Missing values were extrapolated for all studies using values which were available. This process is shown in below.

Original and added values

The same procedure as for GHG emissions is followed to evaluate energy requirements. The light-grey marking in table [E-5] shows those values which were presented in the original LCA. The dark grey marking shows those values which were extrapolated to the original LCA's. The extrapolated values are discussed below.

Value for ethanol production: The energy requirement for ethanol production was excluded from most studies because there is no fossil-based energy requirement. Energy for ethanol production is produced by burning excess bagasse. In this study, the values are extrapolated from the study done by Liptow & Tillman (2009). The value given by Liptow & Tillman (49.5 MJ/kg bio-polyethylene) is extrapolated for all LCA's that did not include the energy requirement for ethanol production.

Values for dehydration and polymerization: One study was found which gave values for dehydration and polymerization. The values giving by Liptow & Tillman for dehydration (7.4 MJ/kg bio-polyethylene) and polymerization (13.7 MJ/kg bio-polyethylene) are extrapolated to all LCA's. Note that Liptow & Tillman only give values for LDPE.

Value for transport of ethanol: Two studies gave values for the transport of ethanol. Seabra et al. (2011) gave a higher value than De Oliviera et al. (2005). The value (converted to MJ/kg bio-polyethylene) given by Seabra et al. (0.75 MJ/kg bio-polyethylene) is based on 340 km of transport. The value given was extrapolated for all LCA's.

Value for transport Porto Alegre – Rotterdam: The value for the energy requirements during sea transport was calculated by using the eco transit tool[9]. Entered parameters were: 500 kton of goods, average goods in bulk, only sea transport. Start at Porto Alegre, finish at Rotterdam. The resulting 1.21 MJ/kg bio-polyethylene was extrapolated to all LCA's

Value for incineration: The value for incineration was based on the assumption that all inherent energy is converted to heat or electricity. The value for inherent energy was given by Liptow & Tillman (43 MJ/kg bio-polyethylene) and extrapolated to all LCA's.

The resulting values are given in the last column of table [E-6]. The BREW study was excluded from further analysis because of strong deviating values compared with the other eight LCA's, which could not be explained by analysing the study.

Differences in including LCA aspects

Certain aspects were included in some studies, while excluded in others. The different views are given below.

Embodied emissions in buildings, machinery and equipment: Due to the relatively small contribution to the overall energy requirements, some authors chose to exclude those numbers. This has no significant effect on the outcome of the LCA's

Reduction of emission by using surplus biomass used for process energy – Overall system energy requirements can be reduced by replacing certain boilers in the process by more efficient ones. Some authors chose to include these avoided emissions, while others did not. In contrary to the Greenhouse gas LCA's, the values were easy to obtain in the LCA's. For reason of comparison, the values were excluded.

Reduction of emissions by using surplus biomass for electricity generation – Overall system energy requirements can be reduced by using surplus biomass to generate electricity. Some authors chose to include these avoided emissions, while others did not. In contrast to the Greenhouse gas LCA's, the values were easy to obtain in the LCA's. For reason of comparison, the values were excluded.

The resulting values for MJ/kg bio-LDPE given in table [E-6] are used to determine a bandwidth for Energy requirements. For the studies the mean and standard deviation were determined. The bandwidth is taken as the result of the *mean ± standard deviation,* which is shown in table [E-2].

[9] http://www.ecotransit.org/index.en.phtml

Table [E-2] Review results for life cycle energy requirements for bio-PE compared to fossil-PE

	MJ/kg bio-LDPE
Mean	73.5
Standard deviation	11.9
Bandwidth of GHG emissions (Mean ± Standard deviation)	61.6 – 85.4 (of which is fossil: 12.2-35.9)
Fossil reference[i]	38.1 – 41.5

The resulting energy requirement increase ranges from ~49% (equation 1) till ~124% (equation 2).

(1) $\left(\frac{Low\ range\ energy\ requirement\ from\ bioLDPE}{High\ range\ energy\ requirement\ from\ fossil\ LDPE} - 1\right) \times 100\%$

(2) $\left(\frac{High\ range\ energy\ requirement\ from\ bioLDPE}{Low\ range\ energy\ requirement\ from\ fossil\ LDPE} - 1\right) \times 100\%$

Figure [E-1] gives a graphical representation of the values in table [E-2]. The total energy requirements are calculated as fossil-based energy requirements + renewable energy requirements.

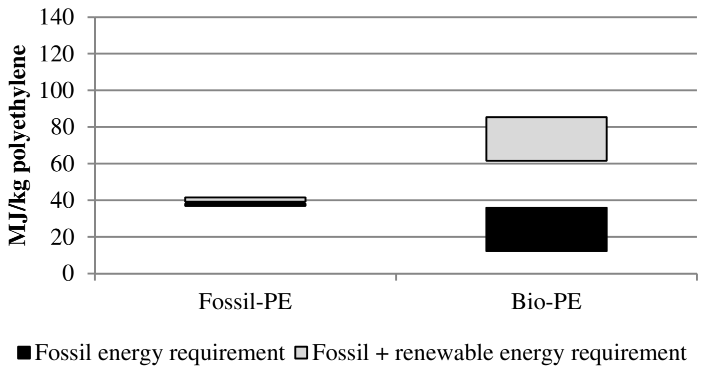


Figure [E-1] Life cycle energy requirement for bio-PE compared with fossil-PE

Energy requirement for PLA

The same approach as for fossil-PE was used to determine the energy requirements of PLA. The results are shown in table [E-3]. Energy production from incineration was deducted from the results of the Simapro database and the study by Vink *et al.* (2003).

Table [E-3] Life cycle energy requirement for PLA compared to fossil-PE

Study/database	MJ/kg PLA
SimaPro database	79.7
Vink *et al.*	82.5
Total range	79.7 – 82.5 (of which is fossil: 53.4-54.1)
Fossil reference[i]	38.1 – 41.5

The resulting energy requirement increase ranges from ~92% (equation 1) till ~117% (equation 2).

$$(1)\quad \left(\frac{Low\ range\ energy\ requirement\ from\ bioLDPE}{High\ range\ energy\ requirement\ from\ fossil\ LDPE} - 1\right) \times 100\%$$

$$(2)\quad \left(\frac{High\ range\ energy\ requirement\ from\ bioLDPE}{Low\ range\ energy\ requirement\ from\ fossil\ LDPE} - 1\right) \times 100\%$$

Figure [E-2] shows the resulting values of table [E-3] in comparison with the values found for fossil-PE.

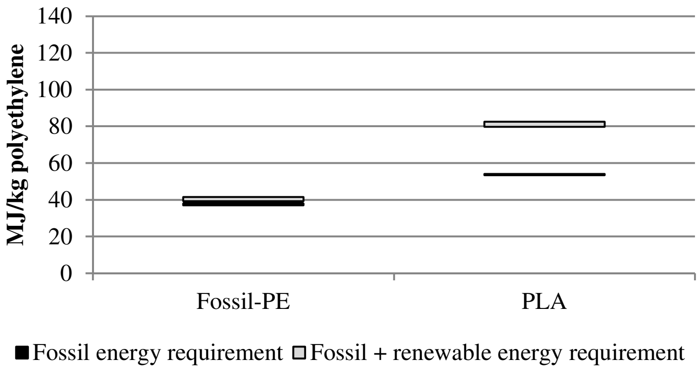


Figure [E-2] Life cycle energy requirements for PLA compared to fossil-PE

Table [E-4] Conversion values

	Value	source
L/ha/yr	5900	Macedo (2004)
density ethanol (kg/L)	0.789	www.densityofethanol.com[a]
Conversion rate (eth/et) high	1.67	stoichiometry & Morschbacker (2006)
Conversion rate (eth/et) low	1.82	stoichiometry & Morschbacker (2006)
CO_2 emission dehydration	0.368	Liptow & Tillman (2009)
CO_2 emission polymerization	0.246	Liptow & Tillman (2009)
LHV ethanol (MJ/L)	24.81	Seabra *et al.* (2011)

[a] at 20 degrees centigrade

Table [E-5] Conversion of LCA's to bio-PE LCA's by extrapolation of missing values

		Macedo (2008)	Seabra (2011)	Garcia (2011)	Liptow & Tillman (2009)	Oliveira (2005)	Boddey (2008)	Brehmer & Sanders (2009)	Pimentel and Patzek (2008)	BREW, Patel (2006)
Agricultural	fossil fuel use	x	x	x	x	x	x	x	x	x
	fertilizer, pesticide production	x	x	x	x	x	x	x	x	x
Industrial	Embodied energy emission in buildings, machinery and equipment	x	n.i.	n.i.	n.i.	n.i.	x	x	x	x
	chemical production	x	x	x	x	x	x	x	x	x
	Ethanol production	v	v	v	x	v	v	x	V	x
	Surplus Biomass used for process energy	n.i.	n.i.	?	n.i.	n.i.	n.i.	n.i.	n.i.	n.i.
	surplus Biomass used for electricity	n.i.	n.i.	n.i.	n.i.	?	n.i.	n.i.	n.i.	n.i.
	Dehydration	v	v	v	x	v	v	v	v	x
	Polymerization	v	v	v	x	v	v	v	v	v
Transport	Transport of sugarcane (20 km)	x	x	x	x	x	x	x	x	x
	Transport of ethanol (340 km)	v	x	v	v	x	v	v	v	v
	Transport Porto Alegre - Rotterdam (11323,23 km)	v	v	v	v	v	v	v	v	v
end-of-life	Incineration	v	v	v	x	v	v	v	v	v

x	Original in LCA
n.i.	Not included in LCA
v	Added to LCA
?	Not mentioned or unclear

Table [E-6] Life cycle greenhouse gas emissions for bio-PE

Study	Original value	MJ/kg polyethylene (excl. extrapolated values as indicated in table [E-2])[a]	MJ/kg polyethylene (incl. extrapolated values as indicated in table [E-3])
Patel *et al.* (2006)	109.4 GJ/tonne ethylene	109.5[b]	123.6[c]
Seabra *et al.* (2011)[d]	173 $KJ_{fossil}/MJ_{ethanol}$	5.9	78.5[e]
Pimentel and Patzek (2008)[f.]	51.59 GJ/ha	12.0	84.6[e]
Macedo *et al.* (2008)[f]	18.1 GJ/ha	4.2	76.8[e]
Boddey *et al.* (2008)[f]	15.32 GJ/ha	3.6	75.4[e]
De Oliveira *et al.* (2005)[f]	42.43 GJ/ha	9.9	82.5[e]
Garcia *et al.* (2011)	8.4 $GJ_{ethanol}/GJ_{fossil}$	4.1	76.6[e]
Brehmer & Sanders (2009)	9.1GJ/ha + 11.42 GJ/tonne ethanol	22.1	44.2
Liptow & Tillman (2009)[f]	68.8 MJ/kg LDPE	-	69.8

[a] conversion of original values by conversion factors given in table [E-4]
[b] Values for the BREW study include energy requirement for dehydration and renewable energy use for ethanol production.
[c] Polymerization and dehydration energy were determined by comparing the values of ethylene (petrchPELD) and PELD. A value of 12.2 GJ/tonne was found. Value was based on sugarcane in Brazil
[d] Credits for electricity and bagasse burning not taken into account
[f] Derived from the comparative analysis by Ramirez Triana (2011)
[e] 49.50 MJ/kg polyethylene from bagasse for the production of ethanol was added based on Liptow & Tillman
[f] Attributional LCA. Without transport to Europe

APPENDIX F: LAND USE CALCULATIONS

In order to assess the impact on biodiversity, additional land use was calculated for the supply of 500kt of polyethylene to the Netherlands. This land use is compared to the total land with economic utilization and the total arable land.

Land use for Fossil-PE production

The additional land use required for the production of 500kt of fossil-PE for the Netherlands can be considered negligible. Capacity from current production facilities can be increased, without needed additional land. The land needed for factories is very small compared to the land needed for the production of crops.

Land use for Bio-PE production

Table [F-1] shows the conversion values used for determining the additional land use.

Table [F-1] Conversion values

Source	Description	Value
Paragraph 4.1	Kt/yr demand for the Netherlands	500 kt/yr
Macedo (2004)	Tonne sugarcane/hectare*year	68.7 tonne/ha/yr
Macedo (2004)	Litre ethanol/tonne sugarcane	86 L/ tonne
Morschbacker (2006)	Conversion rate ethanol-ethylene	90.3-98.5%
	Molecular weight ethylene	46 g/mol
	Molecular weight ethanol	28 g/mol
	Density ethanol	0.789 kg/L

The conversion of ethylene to polyethylene is close to 100%. Equation (1) shows the calculation of litre per hectare using the values in table [F-1]. Note that Braskem reports 6400 litre per hectare. To assess a worst case scenario, ~5900 litre per hectare was used for calculating land use.

(1) $68.7\, t/(ha \times yr) \times 86\ L/t = 5908\, L/ha$

The equations (2)-(6) show the calculation of the additional number of hectares needed. The calculation of the "high" scenario is shown, which considers a low conversion rate for ethanol to ethylene. The calculation of the "low" scenario is not shown, but follows the same approach.

(2) $\frac{500.000.000\, kg\ ethylene}{0.028\, kg/mol} = 18 \times 10^9 mol\ ethylene$

(3) $\frac{18\times10^9 mol\ ethylene}{90.5\ \%} = 20 \times 10^9 mol\ ethanol$

(4) $20 \times 10^9 mol\ ethanol \times 0.046\, g/mol = 91 \times 10^7 kg\ ethanol$

(5) $\frac{91\times10^7 kg\ ethanol}{0.789\, kg/L} = 12 \times 10^8 L\ ethanol$

(6) $\frac{12\times10^8 L\ ethanol}{5900\, L} = 196.000\ ha$

Table [F-2] shows the results of two different runs through the equation above. The first is the additional land use needed for meeting the polyethylene demand in the Netherlands (low and high). The second is the additional land use needed for meeting the polyethylene demand in Europe (low and high). Furthermore, the land used for sugarcane production is shown. The values are compared to the total land with economic utilization and the total arable land.

Table [F-2] Estimated additional land use for sugarcane in Brazil

	Ha*1000	% compared to total land with economic utilization[a]	% compared to total arable land[b]
Total land used for sugarcane production 2010	8210[c]	3.5 %	13.4 %
Additional land use to meet PE demand in the Netherlands	179 – 196	0.08 %	0.32 %
Additional land use to meet PE demand in Europe	4675 - 5103	2.0 - 2.2 %	7.6 – 8.3 %

[a] Total land with economic utilization (ha*1000) = 236100 (FAO, 2004)
[b] Total arable land (ha*1000) = 61200 (FAO, 2004)
[c] (Schlesinger, 2010)

Land use for PLA production

It is assumed that 2.5 kg corn is needed per kg PLA (Vink *et al.*, 2003). Furthermore, corn yield per hectare is assumed to be 5000 kg. Actual corn yield might be higher due to weather conditions, but this value is assumed to assess a worst case scenario. Table [F-3] shows the results for the Dutch demand and the European demand. The values are compared to the total land with economic utilization and the total arable land.

Table [F-33] Estimated additional land use for the production of PLA

	Land use in 1000 ha	% compared to total land with economic utilization[a]	% compared to total arable land[b]
Current production Natureworks (150 kt/yr)[c]	75	0.02 %	0.05 %
Production for the Netherlands (500 kt/yr)	249	0.06 %	0.15 %
Production for Europe (13 Megaton/yr)[a]	6481	1.61 %	3.98 %

[a] Total land with economic utilization: 403 * 10^6 ha. Source; FAO database
[b] Total arable land: 163 * 10^6 ha. Source; FAO database
[c] http://www.natureworksllc.com/

APPENDIX G: CROP RESIDUE BURNING REGULATIONS IN THE UNITED STATES

Table [G-1] Crop residue burning regulations. Source: McCarthy (2011)

State	Crop Residue Burning Regulations
California	• Requires a burning permit; • Burning only on burn days determined by local Air Districts in consultation with the California Air Resource Board; • Residues required to be shredded and piled when possible.
Florida	• Sugar cane farmers initiated burning oversight with Florida Department of Forestry (FLDOF) in 2004; • FLDOF issues burn permits between November and March.
Louisiana	• Farmers can burn during the daytime and are required to have certified Burn Managers at the burn.
Oregon	• In 1991, House Bill 3343 established an open field burning acreage phase-down, propane flaming limitation, and residue burn permitting issued by the Oregon Department of Agriculture (ODA) for the Willamette Valley; • 102,500 acres of grass seed and cereal residues can be burnt per year, which is enforced through aerial and ground surveys; • ODA has the right to fine growers that burn on no-burn days.
Washington	• Washington Department of Ecology (DOE) under the 1991 Clean Air Act of Washington issues all burning permits and determines burn days based on atmospheric conditions and U.S. Forest Service fire danger ratings; • Cost of permits are $2.00 per acre to be paid by the farmers; • DOE can fine farmers $10,000 for any illegal crop residue burning; • DOE uses aerial photography, tip hotline, and remote sensing for enforcement.

APPENDIX H. WORLD AVERAGE PRODUCER PRICES VERSUS BRAZILIAN PRODUCER PRICES.

This appendix shows the graphs on world average producer prices versus Brazilian producer prices. Not that no distinction is made between the size of production for each country. This might cause the actual world price to be misrepresented. However, this analysis focusses on trends, not on actual prices. Therefore, the approach is considered valid. Data on producer prices was derived from the FAO database. A time frame of 10 years is chosen. Only countries which have data from 1999-2009 are taken into account. The results show that the development of Brazilian producer prices is consistent with the development of average world producer prices (except for oranges, which decreased for Brazil). Circumstances for development of producer prices in producer countries can differ. All countries were included that had data from 1999 until 2009. Figure [H-1] untill [H-7] do not seem to indicate a diviating price pattern for Brazil, which could have indicated that the increased sugarcane production was leading to increases in prices of other crops.

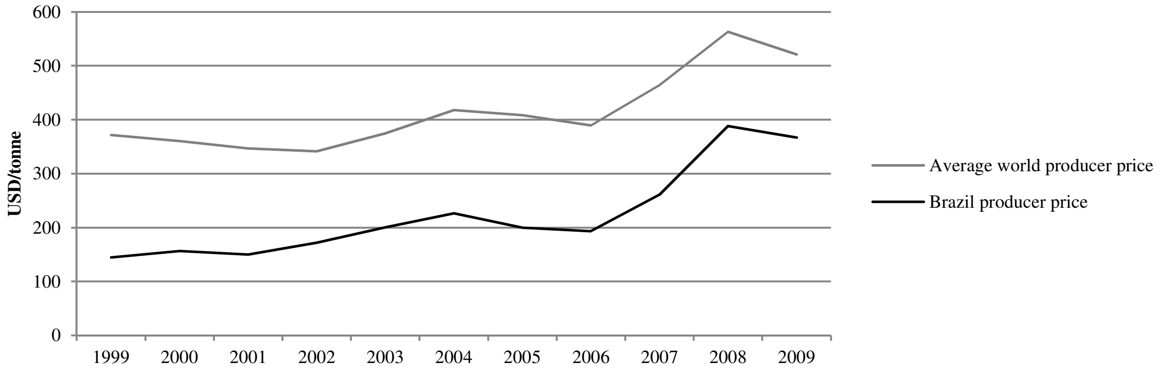


Figure [H-1] Soybean producer prices

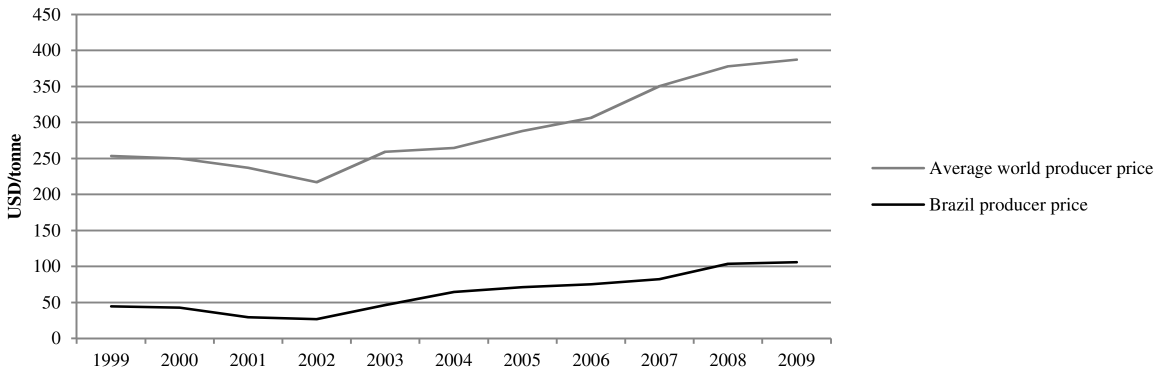


Figure [H-2] Cassave producer prices

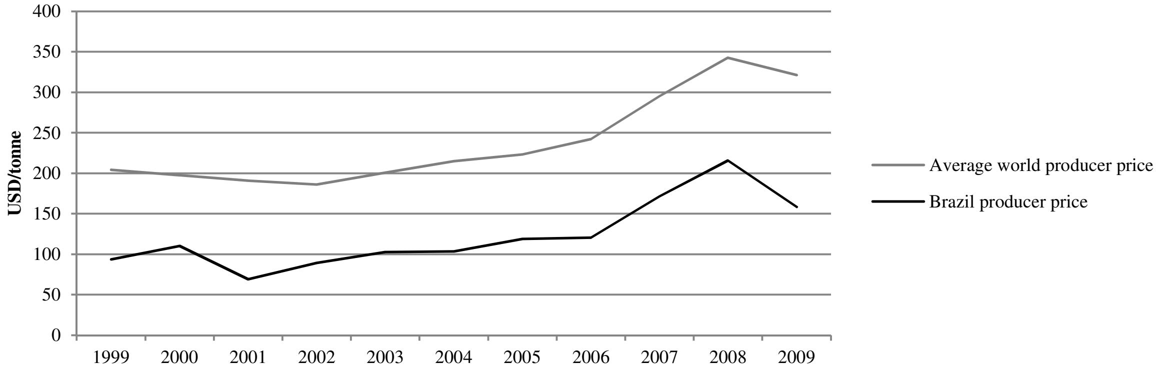


Figure [H-3] Maize producer prices

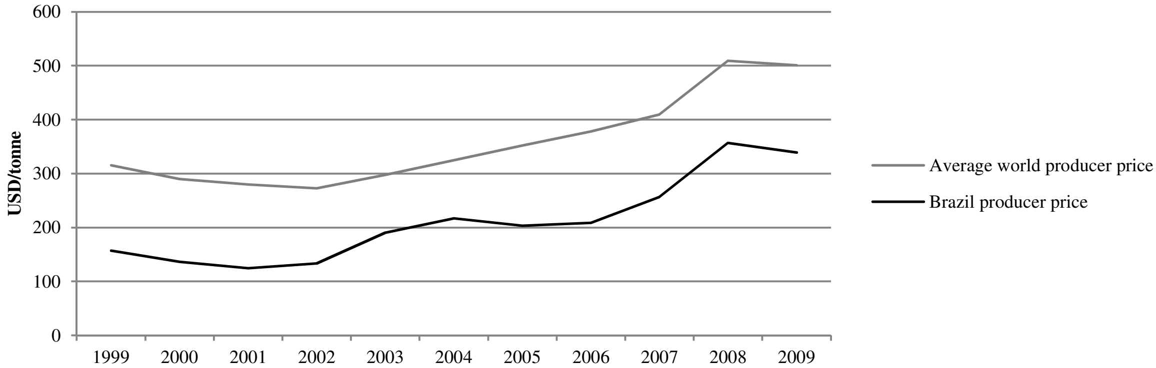


Figure [H-4] Rice (paddy) producer prices

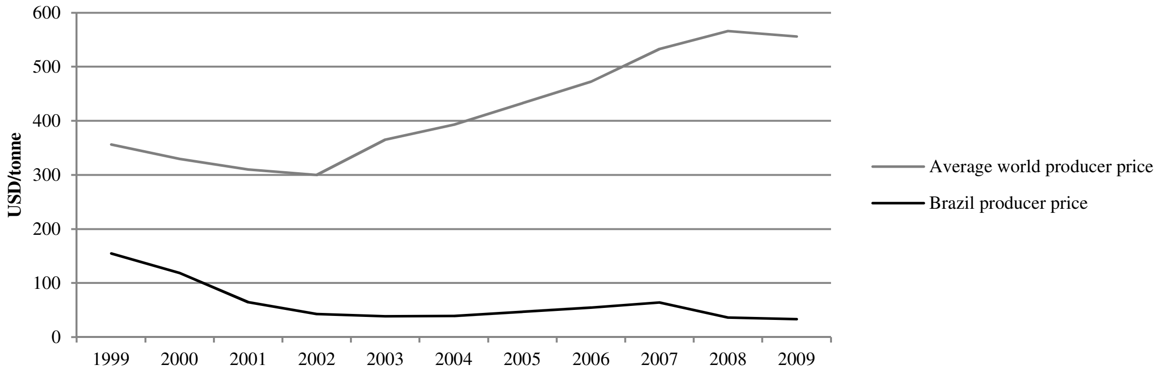


Figure [H-5] Oranges producer prices

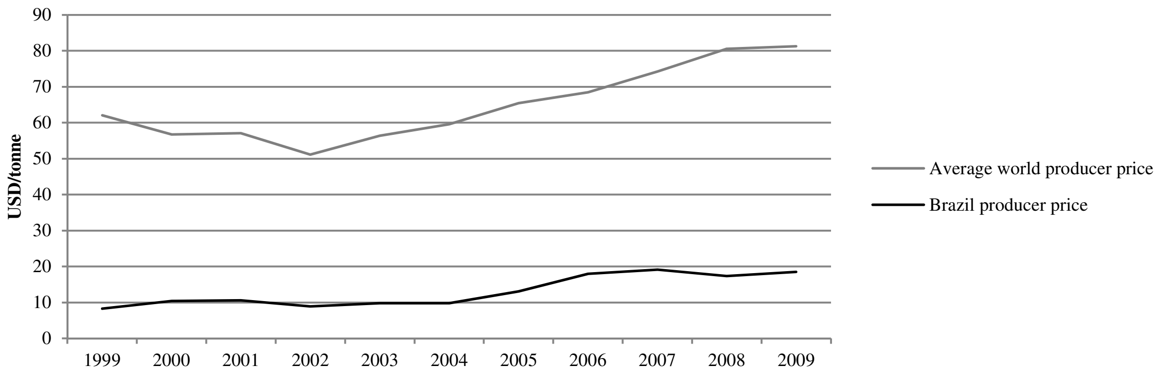


Figure [H-6] Sugarcane producer prices

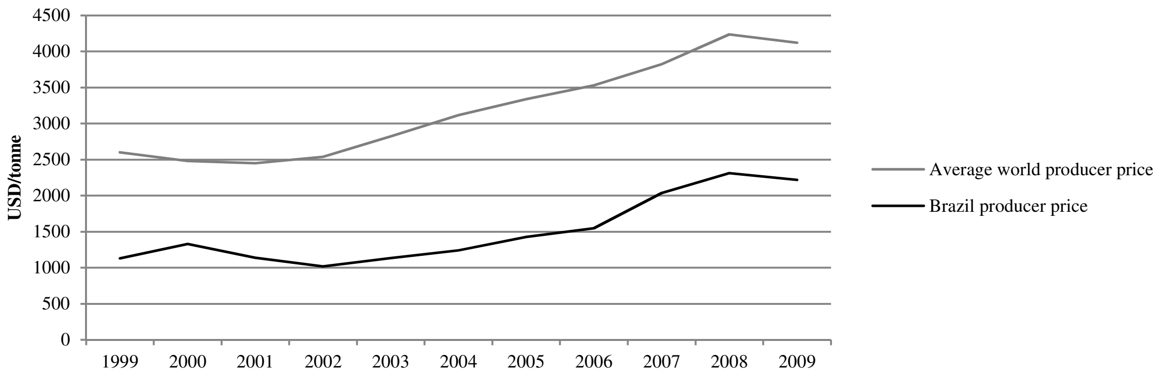


Figure [H-7] Cattle meat producer prices

APPENDIX I: SALES PRICES OF CROP LAND IN BRAZIL

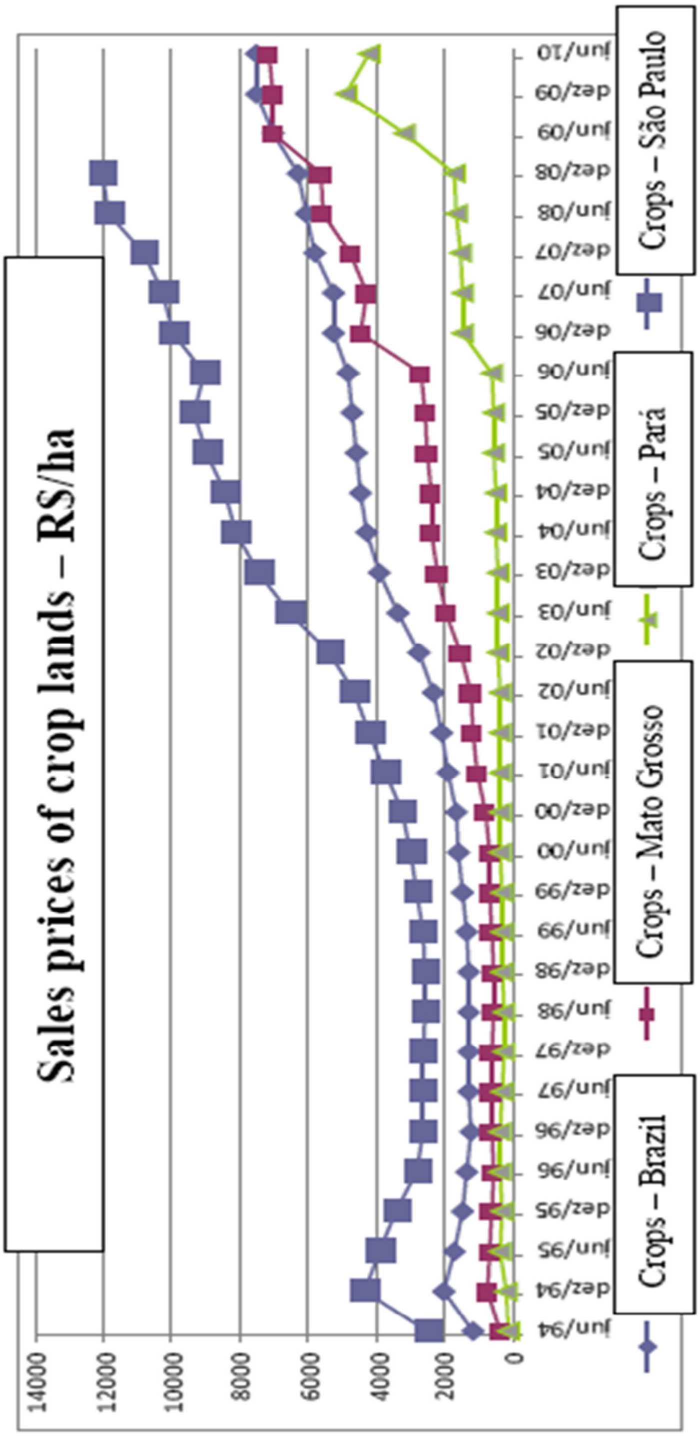


Figure [I-1] Sales prices of cropland in Brazil. The vertical axis displays R$/ha. Source: Sauer & Leite (2011)

APPENDIX J: PROJECTS AND COMPANIES DEDICATED TO LOCAL WELLBEING IN BRAZIL

Table [J-1] Projects contributing to local wellbeing as reported by UNICA members in 2010. Source: Sustainability report UNICA

AREA	PROJECTS	PEOPLE AFFECTED	INVESTMENT
Culture	22	91,333	R$1,564,432.66
Environment	43	69,243	R$8,596,047.34
Sport and Leisure	12	23,645	R$1,743,830.28
Quality of life	53	80,982	R$7,005,617.57
Health	36	28,698	R$2,975,886.53
Education	46	26,988	R$6,618,190.76
Training	61	88,718	R$3,851,518.39
Total	273	409,607	R$32,355,523.53

Table [J-2] Number of companies involved in local wellbeing initiatives as reported by UNICA members in 2010. Source sustainability report UNICA

INITIATIVES	# COMPANIES
Participate in local forums.	61
Internalize this relationship within the company.	51
Work in partnership with the community to build networks to solve local problems, offering technical support and/or physical space, or other types of support.	48
Participate in the formulation of public policies, engaging in solving problems where the company is located.	41
Recognize the community where it is present as an important stakeholder in the company's decision-making processes.	67
Contribute to improvements in infrastructure or in the local environment that can be enjoyed by the community (housing, roads, bridges, schools, hospitals, etc.).	83
Have a program to hire as many local people as possible in the community where it operates, giving them training, aiming to increase skill levels in that community in cooperation with unions, NGOs, community representatives or public authorities.	89
Have procurement and investment practices to improve socioeconomic development of the community where it operates.	71

APPENDIX K: FOSSIL-PE AND BIO-PE MARKET PRICES

This appendix shows the relation of feedstock prices and the market prices of fossil-LDPE and bio-LDPE. LDPE was evaluated to be consistent with the evaluation of GHG emissions and energy requirements.

Market price fossil-PE

Since oil prices are volatile, a linear regression analysis was performed on relation between the ethylene prices and crude oil prices in the period October 2008 until May 2011 (not corrected for inflation) (Plastemart, 2012). The ethylene and crude oil prices for this period are given in table [I-1]. The result is shown in figure [I-1].

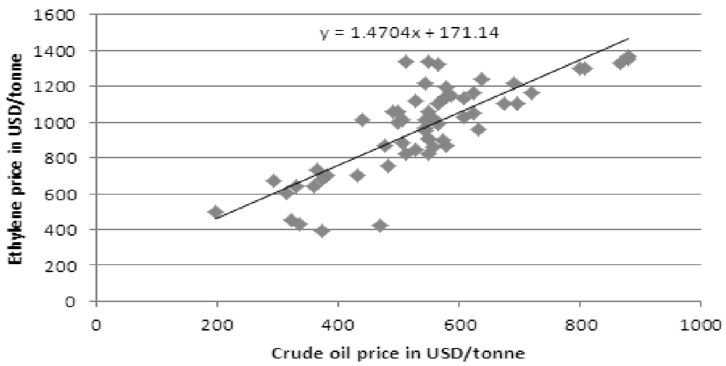


Figure [I-1] Market price of ethylene from crude oil

The average mark-up (May 2010-May2010) from ethylene to LDPE (film) is 417 as is shown in the bottom row of table [I-2]. The average was taken to limit the volatile price respond of PE and ethylene to feedstock price fluctuations. This value was added to the equation of figure [I-1], resulting in figure [I-2].

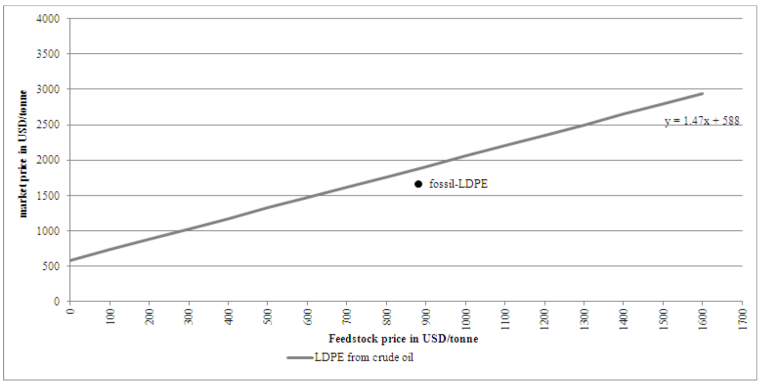


Figure [I-2] Current market price of Fossil LDPE

The exact market price of 1670 USD/tonne for May 2011 (Table [I-2]) for fossil-LDPE is also shown in figure [I-2] (Plastemart, 2012). The crude oil price at the time was 880 USD/tonne.

Market price bio-PE

Seddon et al. gave a relation between the ethanol price and the ethylene production costs (Seddon, 2010). This relation is shown in figure [I-1]. The current prize for Brazilian ethanol is 0.73 USD/gallon, or 920 USD/tonne[10].

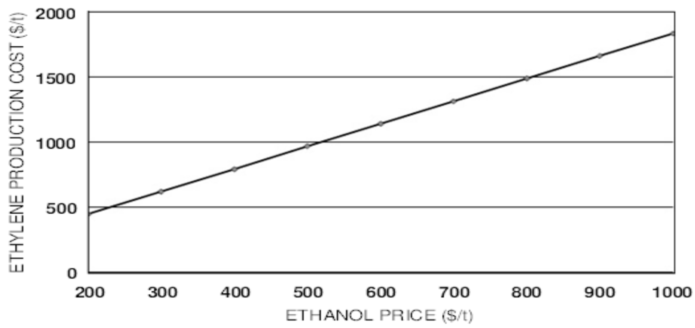


Figure [I-3] Production costs of ethylene from ethanol

To determine the price of polymerization, it is assumed that the price is the same for fossil- and bio-ethylene polymerization. As with fossil-PE, the average 417 USD/tonne is used to limit the effect of the price respond of PE and ethylene. Furthermore, since the fossil-ethylene prices are market prices, and the bio-ethylene prices production prices, a profit margin of 15% was added to the bio-PE. When the polymerization and profit margin are added, the linear relation shown in figure [I-3] is obtained. The figure [xx] was merged into the graph to show the relation between bio-LDPE and fossil-LDPE.

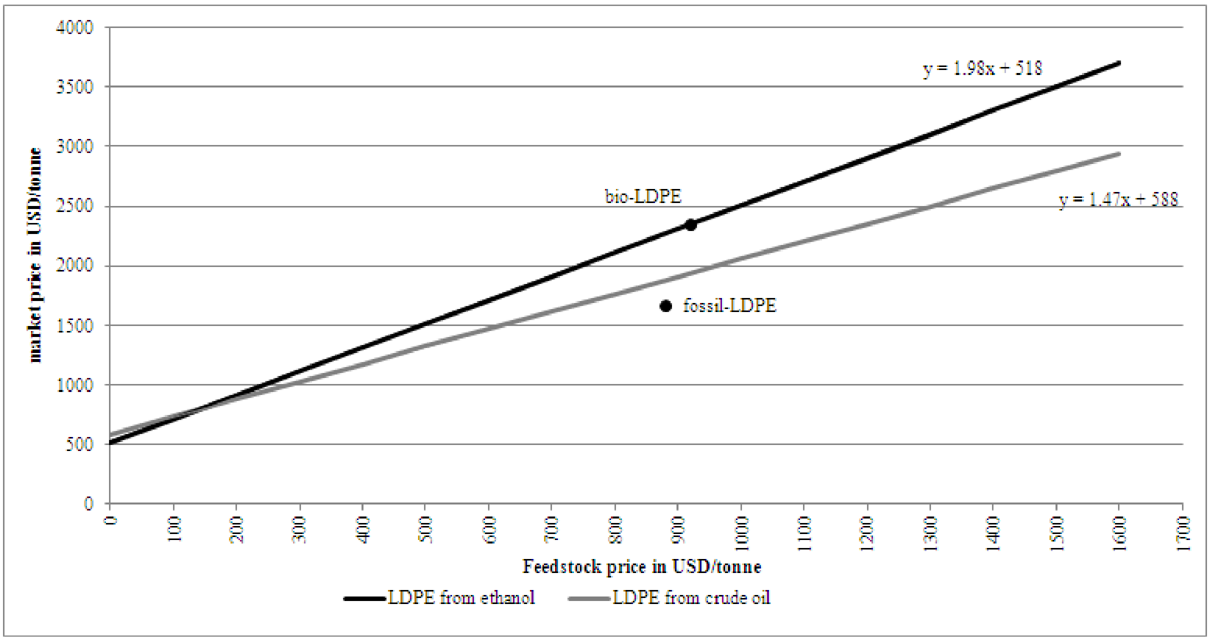


Figure [I-4] Market prices of ethylene from ethanol and crude oil

[10] http://www.cepea.esalq.usp.br/english/ethanol/

Table [I-1] ethylene prices and crude oil prices for the period Oct '08 until May '11. Source: (Plastemart, 2012)

Date	Ethylene price in USD/t	Crude oil price in USD/bbl	Crude oil price in USD/tonne	Date	Ethylene price in USD/t	Crude oil price in USD/bbl	Crude oil price in USD/tonne
May 01`11	1365	120	879.60	Jan 01`10	1320	77	564.41
Apr 15`11	1350	120	879.60	Dec 15`09	1125	78	571.74
Apr 01`11	1330	118	864.94	Dec 01`09	1120	72	527.76
Mar 15`11	1300	109	798.97	Nov 15`09	1060	75	549.75
Mar 01`11	1300	110	806.30	Nov 10`09	1020	76	557.08
Feb 15`11	1215	94	689.02	Oct 15`09	905	75	549.75
Jan 15`11	1160	98	718.34	Oct 01`09	820	75	549.75
Jan 01`11	1100	95	696.35	Sep 15`09	870	65	476.45
Dec 15`11	1100	92	674.36	Sep 01`09	1000	68	498.44
Dec 01`10	1050	85	623.05	Aug 15`09	1015	69	505.77
Nov 15`10	960	86	630.38	Aug 01 '09	1060	68	498.44
Nov 10`10	1030	83	608.39	Jul 15 '09	1060	67	491.11
Oct 15`10	1135	83	608.39	Jul 01 '09	1015	60	439.80
Oct 01`10	1100	77	564.41	Jun 15 '09	885	69	505.77
Sep 15`10	990	77	564.41	Jun 01 '09	845	72	527.76
Sep 01`10	970	74	542.42	May 15 '09	755	66	483.78
Aug 15`10	910	75	549.75	May 01 '09	700	59	432.47
Aug 01 '10	865	79	579.07	Apr 15`09	700	52	381.16
Jul 15 '10	860	76	557.08	Apr 01`09	730	50	366.50
Jul 01 '10	900	78	571.74	Mar 15`09	680	51	373.83
Jun 15 '10	950	74	542.42	Mar 01`09	645	49	359.17
Jun 01 '10	1015	74	542.42	Feb 15`09	640	45	329.85
May 15 '10	1220	74	542.42	Jan 15`09	675	40	293.20
May 01 '10	1240	87	637.71	Jan 01`09	605	43	315.19
Apr 15`10	1160	85	623.05	Dec 15`08	500	27	197.91
Apr 01`10	1140	79	579.07	Dec 01`08	455	44	322.52
Mar 15`10	1150	80	586.40	Nov 15`08	430	46	337.18
Mar 01`10	1195	79	579.07	Nov 10`08	395	51	373.83
Feb 15`10	1335	75	549.75	Oct 15`08	425	64	469.12
Jan 15`10	1335	70	513.10	Oct 01`08	820	70	513.10

Table [I-2] Market prices for polyethylene and ethylene for the period may 2010-May 2011. Source: (Plastemart, 2012)

	HDPE		LDPE		LLDPE		feedstock
Price lists	film	injection	film	injection	film	injection	ethylene
May 01`11	1400	1380	1670	1680	1375	1385	1365
Apr 15`11	1380	1370	1680	1690	1380	1390	1350
Apr 01`11	1355	1345	1710	1720	1400	1410	1330
Mar 15`11	1340	1330	1710	1720	1410	1420	1300
Mar 01`11	1310	1300	1700	1710	1400	1410	1300
Feb 15`11	1320	1310	1700	1710	1425	1435	1215
Jan 15`11	1310	1300	1690	1700	1395	1405	1160
Jan 01`11	1260	1250	1650	1660	1360	1375	1100
Dec 15`11	1240	1230	1630	1640	1350	1365	1100
Dec 01`10	1265	1255	1610	1620	1365	1380	1050
Nov 15`10	1290	1280	1615	1625	1410	1425	960
Nov 10`10	1270	1260	1580	1590	1320	1330	1030
Oct 15`10	1245	1235	1500	1410	1280	1510	1135
Oct 01`10	1195	1185	1380	1390	1190	1200	1100
Sep 15`10	1195	1185	1380	1390	1190	1200	990
Sep 01`10	1170	1160	1335	1340	1170	1180	970
Aug 15`10	1175	1165	1335	1340	1165	1175	910
Aug 01 '10	1055	1045	1265	1270	1090	1100	865
Jul 15 '10	1070	1060	1280	1285	1110	1120	860
Jul 01 '10	1120	1110	1325	1330	1165	1175	900
Jun 15 '10	1125	1115	1340	1345	1180	1175	950
Jun 01 '10	1150	1140	1385	1390	1225	1220	1015
May 15 '10	1270	1260	1470	1480	1330	1325	1220
May 01 '10	1300	1290	1485	1495	1380	1375	1240
average (24 points)	1242	1232	1518	1522	1294	1312	1101
Polyethylene-ethylene[a]	141	131	417	421	193	211	

[a] Average polyethylene market price/average ethylene market price

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