Life Cycle Assessment of a Hybrid Poly Butylene Succinate Composite

by

Hassan Moussa

A thesis
presented to the University of Waterloo
in fulfillment of the
thesis requirement for the degree of
Master of Environmental Studies
in
Environment and Resource Studies

Waterloo, Ontario, Canada, 2014

AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Poly butylene succinate (PBS) is a biodegradable plastic polymer that has physical and mechanical properties similar to common petroleum plastics like polypropylene (PP) and polyethylene (PE). PBS may be produced from petroleum or bio-based feedstocks, or by a hybrid combination of petroleum and bio-based resources. Producers are reducing content of petroleum components used for the production of PBS, and by doing so are seeking potential environmental performance improvements. In this study, “hybrid” PBS refers to the production of PBS polymer from bio-based succinic acid (SAC) sourced from sorghum and petroleum-based 1, 4-butanediol (BDO).

Given its biodegradability, PBS is commercially used for compostable bags and agricultural mulching film applications. A recent study in Ontario identified composite materials made with PBS blended with natural fibres like switchgrass (SG) as promising for applications in automotive products. Such novel composite materials are touted as potential bio-based alternatives to conventional petroleum-based plastics.

Of the few studies that have considered the environmental performance of PBS materials, none have assessed the potential environmental impacts of a hybrid PBS composite. Therefore, this study undertook a life cycle assessment (LCA) of SG reinforced hybrid PBS composite (hybrid composite). LCA is an environmental management technique that is used to assess environmental aspects (inputs and outputs) and potential environmental impacts of a product or service throughout its life cycle. The analysis considered a cradle-to-gate system boundary and evaluated eleven environmental performance indicators. The environmental performance of the hybrid composite was compared to a conventional glass fibre (GF) reinforced polypropylene (PP) composite (baseline composite), a material that is widely used in automotive components.

Results showed that the production of the hybrid composite in comparison to the baseline composite decreased potential impact for most of the assessed indicators: cumulative energy demand by 40%, waste heat by 23%, global warming potential by 35%, smog by 2%, carcinogens by 54%, non-carcinogens by 172%, respiratory effects by 22% and ecotoxicity by 45%. Increases in the values of impact indicators were apparent for ozone depletion, acidification, and eutrophication by 43%, 16%, and 322%, respectively.

Analysis revealed that dominant influences on results were not related directly to the bio-based make-up. Rather, the biggest influence on the environmental performance of composite production were the sources of heat used in petroleum-based materials, the energy mix in electricity for bio-based materials, the type of reinforcing fibre and the co-product treatment methodology used.

The study helps fill a gap in knowledge regarding bio-based chemicals and hybrid biodegradable plastic composites, and points to opportunities for future research on feedstocks for industrial composite materials.

The importance of this study is that it helps to identify the environmental strengths and weaknesses associated with the production of the hybrid composite specifically, and bio-based materials more generally. It points to alternative material substitution options for use in the automotive industry. In this study, life cycle assessment exemplifies multidisciplinary methodologies, which seek to traverse the boundaries between the social and natural sciences and disciplines to support more sustainable policy decisions for a bio-economy. The systematic nature and the widely applicable consequences of this LCA study have the potential to contribute to industrial and business management, and reach the public policy arena in an effort to drive environmental and social change.

Acknowledgements

I would like to thank each and every one who provided me with help and guidance, and made this journey an experience of a life-time. First, my advisor, Dr. Steven B. Young for his unlimited support in spite of the difficulties that we have overcome, Dr. Goretty Dias, the technical committee member for her valuable contribution to this work, Prof. Ian Rowlands for being the examiner, and Dr. Derek Armitage, the chair of my exam.

I also acknowledge the importance of the collaboration between the University of Waterloo and Myriant Corporation, U.S.A. and their valuable contribution to this work. In particular, I would like to thank Dr. Michael Mang, Product Technology Manager and Mrs. Amy Miranda Manager, Communications and Government Affairs.

Finally, this research was supported by the Province of Ontario, Ontario Ministry of Agri-foods and Rural Affairs (OMAFRA), Project Number: 200004-200005-200006, Renewable, Recyclable and Lightweight Hybrid Green Composites from Lignin, Switch grass, Miscanthus and Bioplastics for Sustainable Manufacturing.

Dedication

This thesis is dedicated to:

My wife and my two daughters for their unconditional love and support

To the memory of my beloved dad

My mom and my siblings

And to all my true friends

Table of Contents

List of Figures

List of Tables

List of Abbreviations

ACC	American Chemistry Council
AcC	Acetyl cellulose
AMS	Ammonium sulfate
ASTM	American Society for Testing and Materials
BDO	1, 4-butanediol
Bio-SAC	Bio-succinic Acid
BPF	British Plastics Federation
CED	Cumulative Energy Demand
CH_4	Methane
CO_2	Carbon Dioxide
DIN	German Institute for Standardization
ECCP	European Climate Change Program
EN	European Standards
E.O.L.	End of Life
EPA	United States Environmental Protection Agency
EP	Epoxy
EPS	Expandable Polystyrene
GHG	Greenhouse Gas
GWP	Global Warming Potential
HDPE	High Density Polyethylene
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization of Standardization
LCA	Life Cycle Assessment (analysis)
LCI	Life Cycle Inventory
LDPE	Low Density Polyethylene
LLDPE	Linear Low Density Polyethylene
MF	Melamine Formaldehyde
N_2O	Nitrous Oxide or Dinitrogen Oxide

NAFTA	North American Free Trade Agreement
OECD	Economic Co-operation and Development
OMAFRA	Ontario Ministry of Agriculture, Food and Rural Affairs
PA	Polyamides
PBS	Poly Butylene Succinate
PCL	Polycaprolactone
PE	Polyethylene
PES	Polyethylene Suberate
PET	Polyethylene Terephthalate
PF	Phenolic
PHA	Poly Hydroxyalkanoates
PHB	Poly Hydroxybutyrate
PLA	Poly Lactic Acid
PlasticsEurope	Plastics Industry Association in Western Europe
PP	Polypropylene
PS	Poly Styrene
PVC	Polyvinyl Chloride
SAC	Succinic Acid
SG	Switchgrass
SimaPro	System for Integrated Environmental Assessment of Products
SPI	Society of the Plastics Industry
TRACI	Tool for the Reduction and Assessment of Chemical and other environmental Impacts
UF	Urea Formaldehyde
UNFCCC	United Nations Framework Convention on Climate Change
USDA	United States Department of Agriculture
USDE	United States Department of Energy

1. Introduction

1.1 Bio-based Materials

Pollution Probe and BIOCORP Canada (2004) defined bio-products as “commercial or industrial products that rely on energy, chemicals or processes available from living organisms” (p. 67). Similarly, the U.S. Farm Security and Rural Investment Act of 2002 (Farm Bill, 2002) defines a bio-based product as “a commercial or industrial product (other than food or feed) that is composed, in whole or in significant part, of biological products or renewable domestic agricultural materials (including plant, animal, and marine materials) or forestry materials” (U.S. Department of Agriculture [USDA], 2002, SEC. 9001). This definition was revised in the U.S. Farm Bill of 2008 to incorporate bio-based intermediate ingredients or feedstocks (USDA, 2008a). The bio-based market program (BioPreferred Program) was created and included in the 2002 Farm Bill section 9002 and reauthorized in the 2008 Farm Bill (USDA, 2002; USDA, 2008). The BioPreferred Program categorizes bio-based products in groups with standard minimum bio-based content. Products in these groups must meet or exceed the minimum bio-based content (USDA, 2013). However, the BioPreferred and other U.S. programs based on bio-based products are largely silent on the sustainability benefits and costs of these products.

While there are no conclusive results when comparing sustainability of bio-based with petroleum-based materials, interest in biomass and bio-based materials is partially due to the increased attention on climate change and greenhouse gas (GHG) emissions mitigation, the future of energy supply, air quality, and industrial technological advancements (Wood & Layzell, 2003; Bradley & Bradburn, 2010). Nevertheless, the expectation is that the potential environmental impacts of bio-based materials, at least from a GHG emissions perspective, will be lower than those of petroleum-based materials. In addition to GHG emissions, there are other impact categories that play key roles in evaluating the environmental performance of a product (Goedkoop et al., 2010). Studies by Zah et al. (2007), Khoo et al. (2010), and

Gironi and Piemonte (2010) suggest that the use of bio-based materials may potentially reduce GHG emissions and dependency on non-renewable resources, but may also contribute to problems of human health and eco-system quality due to impacts from intensive agricultural practices.

Generally speaking, modern agriculture crop production systems are heavily dependent on fossil fuels for the production of synthetic fertilizers, pesticides and herbicides, farm machinery, and irrigation (Pimentel, Hepperly, Hanson, Douds, & Seidel, 2005). This energy-intensive industry has been associated with a broad range of ecological impacts, including water pollution (Carpenter et al., 1998), soil erosion (Gerhardt, 1997), pesticide toxicity, and pest resistance (Carvalho, 2006). According to the Intergovernmental Panel on Climate Change (IPCC)'s 4th assessment report in 2007, agriculture is responsible for emitting 5.1-6.1 Gt of CO_2-eq in 2005, which is equivalent to 10-12% of synthetic greenhouse gas (GHG) emissions globally (Smith et al., 2007). Moreover, and as discussed by Wood & Layzell (2003) and Bradley & Bradburn (2010), the finite nature of fossil fuel reserves and the environmental impacts associated with the extraction and consumption of these reserves highlights the increased concerns about the severe agricultural practices used for the cultivation of different bio-logical feedstocks that are used in the production of bio-materials (Pimentel et al., 2005).

Parameters on bio-based materials vary as a result of type, quantity, quality, and location of the biomass used, as well as its intended use in the production process. For that reason it is essential to have a data system that provides precise and reliable information on the availability and environmental profiles and impacts of biomasses and materials (Agriculture and Agri-Food Canada, 2008). This body of information would be broadly useful, for example, as biomass is used for the production of bio-fuels for ethanol and biodiesel, biomaterials for food packaging, bio-chemicals like bio-succinic acid, bio-polymers (e.g., hybrid poly butylene succinate), and reinforcement materials (e.g., switchgrass fibres).

Though the environmental advantages are largely unclear, bio-based materials are gaining prominence due to the strong support by governmental policies, environmental regulations, environmental awareness, and the potential to create jobs. These are also motivated by economic (e.g., escalating prices of petroleum counterparts), international security issues and political disputes in oil-exporting countries, and natural disasters related issues. Additionally, industrial leadership, technological advancements, and positive feedback from end-of-use industries have increased the demand and acceptance of end users to bio-based materials.

1.2 Poly Butylene Succinate (PBS)

PBS is an aliphatic polyester polymer that is versatile and stable in normal conditions, originally produced from petroleum resources. Its processability is similar to conventional polymers like polyethylene. However, PBS is presented as being totally biodegradable (Showa Denko K.K., 2010; Showa Denko, 2012; Ichikawa & Mizukoshi, 2012; Tserki, Matzinos, Pavalidou, Vachliotis, & Panayiotou, 2006; Kim, Kim, Jin, Park, & Yoon, 2001). PBS has limitations due to its low elasticity and low gas-barrier properties (Terasawa et al., 2008; Twarowska-Schmidt & Tomaszewski, 2008; Ray, Okamoto, Maiti, & Okamoto, 2002). One approach to overcome low elasticity issues is to reinforce PBS with fibres that add structural integrity to the final material. PBS is used in agricultural mulch film, packaging, civil engineering materials and construction products, and consumer goods (Ichikawa & Mizukoshi, 2012; Theunissen, 2012). The material was recently tested for manufacturing automotive parts (Terasawa et al., 2008; Theunissen, 2012).

Since the late 1990s, a body of literature has grown on the mechanical and physical properties and biodegradability of petroleum-based PBS polymers (Fujimaki, 1998; Kim, 2001; Zhao et al., 2005; Tokiwa, Calabia, Ugwn, & Aiba, 2009). Yet few studies have assessed the environmental impacts of either the polymer or its composites throughout their life cycle (Ichikawa & Mizukoshi, 2012; Terasawa, Tsuneoka, Tamura, & Tanase, 2008). Ichikawa and Mizukoshi (2011) and Terasawa et al. (2008) carried out research on “cradle-to-grave” [1] system boundary. To our knowledge, however, no study has assessed the environmental impacts associated with the production of a hybrid or bio-based PBS polymer nor any of its composites.

1.3 Switchgrass (SG)

Studies on bio-based materials have identified switchgrass as a promising biomass crop suitable for bio-energy and bio-based applications (Hewson & Oo, 2009; Karp & Shield, 2008; Keoleian & Volk, 2005; Sanderson et al., 1996), and recently as a potential reinforcing material for plastic composites (Mohanty, Misra, & Sahoo, 2010).

Switchgrass (SG) is a perennial non-food grass crop native to North America that grows with moderate to high productivity, and is (1) adaptable to marginal farmlands, (2) drought-resistant, (3) requires less intensive agricultural practices (fertilizer, herbicide, and pesticide use) than other crops, (4) provides good resistance to pests and diseases, and (5) offers flexibility in different value-added uses (McLaughlin et al., 1999; Samson et al., 1993). Switchgrass has a high photosynthetic efficiency (60% greater than corn), which provides a high energy gain per hectare (Rinehart, 2006). Moreover, its high cellulosic content makes switchgrass an ideal candidate for production of bio-products (Rinehart, 2006). For all of these reasons, switchgrass fibres are a viable potential reinforcing material for composites, including PBS-based plastics.

1.4 Problem Statement and Objectives

This study asks “What is the comparison in environmental profiles between bio-based and conventional composite materials?” This question is addressed by looking at a specific SG / PBS “hybrid” composite material produced for a hypothetical automotive part application. The study considers the viability of this

[1] Cradle-to-grave life-cycle assessment is the full LCA from resource extraction (cradle) to the use phase and disposal phase (grave).

hybrid composite in comparison to a conventional reinforced plastic that would typically be used in the auto part: glass fibre (GF) reinforcement in polypropylene (PP) polymer matrix. This conventional material is the "baseline" product used in this study.

A life-cycle assessment (LCA) study was conducted across a cradle-to-gate [2] system boundary, drawing on new primary data collected on the production of bio-based succinic acid, which is the major precursor chemical in the production of PBS. This approach allowed the study to consider multiple environmental emissions and resource categories, across multiple production steps from raw materials to a finished composite material product.

The specific objectives of this study are to:

1. Quantify the potential life cycle environmental impacts associated with the production of the hybrid composite and the baseline composite materials.
2. Compare the environmental performance of the hybrid composite vs. the baseline composite, examining the role of the bio-based composition in the material.
3. Identify issues and areas of significance ("hotspots") in the environmental life-cycle profile of the hybrid composite, and to discuss potential opportunities for improvement of the product system.

It is anticipated that the results of this study will help inform at least two audiences to drive environmental and social change:

1. Policy-makers should be interested in this study as the results would provide information relevant to the potential environmental costs and benefits of a more bio-based economy. For example, Ontario is developing strategies that support a more sustainable, effective, and diverse economy with high employment that has a societal and geographical cohesion (OMAFRA, 2011).

[2] Cradle-to-gate life-cycle assessment is a partial product life cycle from raw material extraction (cradle) through materials processing, and manufacturing to the factory gate.

Additionally, this study may help policy-makers to initiate dialogues with economic sectors through supporting the research and development of new bio-based technologies and materials with incentives that are specifically developed for that (OMAFRA, 2011). This may lead to the implementation of new Ontario's policies and regulations for agricultural, rural, and industrial developments that plan for a balanced and secure supply and demand of affordable industrial feedstocks (Confederation of European Paper Industries, 2012).

2. Companies should be interested as they seek reliable supply chains that link biomass growers, producers of bio-based materials that use biomass, and the manufacturers of bio-based products that may potentially replace conventional fossil-based products. The Biobased Industries Consortium (2013) and the Confederation of European Paper Industries (2012) state that such an approach will help create an integrated economy that is more competitive technologically, environmentally, and socially.

1.5 Thesis Structure

In Chapter 2, the study presents information on plastics and composite materials in general and on bio-plastics in particular, including their industrial applications. Additionally, criteria that are used in assessing a product environmental performance are discussed. The LCA methodological framework is also introduced in Chapter 2. Chapter 3 communicates the specific methods and data used in the modelling of the hybrid composite and the baseline composite. Chapter 4 presents life cycle impact assessment (LCIA) results looking at the comparison between the hybrid and the baseline composite and the different environmental emissions and impact categories. Chapter 5 discusses the study results, provides conclusions and indicates potential future research opportunities.

2. Literature Review

2.1 Plastics and Composites

2.1.1 Background

The history of synthetic plastics goes back to the year 1855, when Alexander Parkes [3] synthetized and patented the first man-made thermoset (The Robinson Library, 2012). "Parkesine", as the newly developed plastic was called, was derived from organic cellulose. The intention was to replace natural rubber with a cheaper mass-produced synthetic plastic (Society of the Plastics Industry, 2013). In 1909, Erinoid developed Casein plastics, derived from milk. The first biodegradable plastic was commercialized in 1990 by ICI under the name "Biopol" (British Plastics Federation [BPF], 2012).

Today, plastics are widely used in nearly every aspect of our lives, steadily replacing materials such as metals and glass in many applications including packaging, construction, transportation, health, electricity and electronics, and agriculture. This is driven by factors related to cost, performance, processability, weight, and ability to resist corrosion (Industry Canada, 2012). However, in spite of the technological advancements and enhancements in the environmental, mechanical and physical performance of plastic products and their composites, there is still an ongoing debate as to whether plastics are "good" or "bad" (PlasticsEurope, 2012a; PlasticsEurope, 2012b; Andrady & Neal, 2009; Thompson, Moore, vom Saal, & Swan, 2009; Ryan, Moore, van Franeker, & Moloney, 2009).

On average, the global production of plastics has increased annually since the mid-20th century. In 1950 the global production was 1.3 million tonnes, and in 2010 reached 265 million tonnes (PlasticsEurope, 2007; PlasticsEurope, 2011), as illustrated in Figure 1.

[3] Alexander Parkes (1813-1890) was an English scholar who held over 80 patents in chemistry.

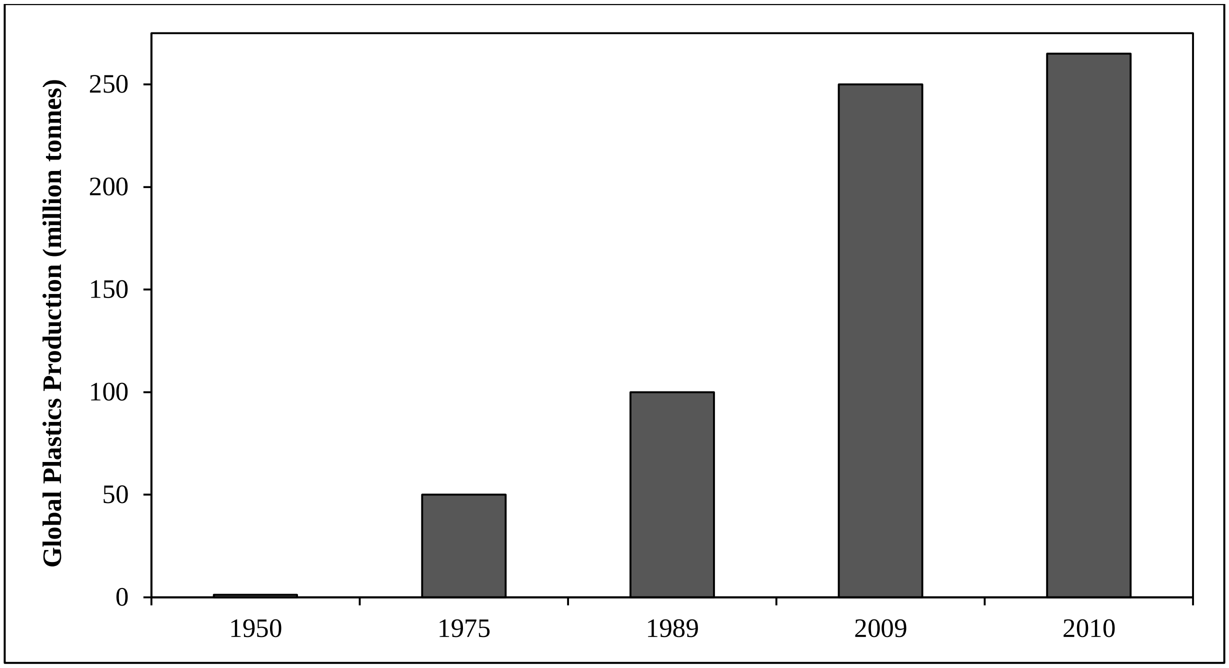

Figure 1*: The global production of plastics (1950-2010) measured by millions of tonnes (PlasticsEurope, 2007; PlasticsEurope, 2011)*

By 2010, three geopolitical zones accounted for 65.5% of the world's plastics production (PlasticsEurope, 2011). These zones are the North American Free Trade Agreement (NAFTA) countries, the European Union states (EU-27), and China (PlasticsEurope, 2011). In 2009, NAFTA countries, the EU-27 and China contributed 23.5%, 24.0% and 18%, respectively, to global plastic production (Haensel, 2011). However, in 2010, China led the global plastics production, followed by the EU-27 and NAFTA countries by at 23.5%, 21.5% and 20.5% respectively (PlasticsEurope, 2011), as in Figure 2.

Another indicator that can help understand the increasing trend in the use of plastics is the global annual plastic consumption per capita. Statistics reveal an increase in the average plastic consumption per capita from 43 kg to 121.5 kg between 1980 and 2010 in North America and the EU-27 (PlasticsEurope, 2007), Figure 3.

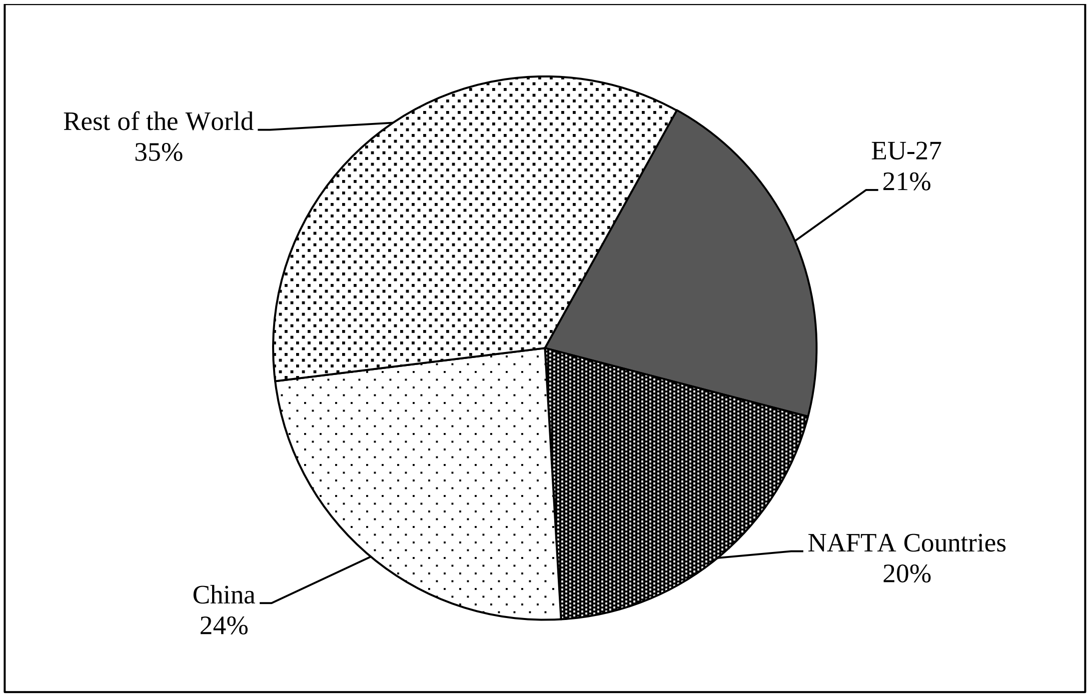

Figure 2: *Global plastic production (2010) by country and region (PlasticsEurope, 2011)*

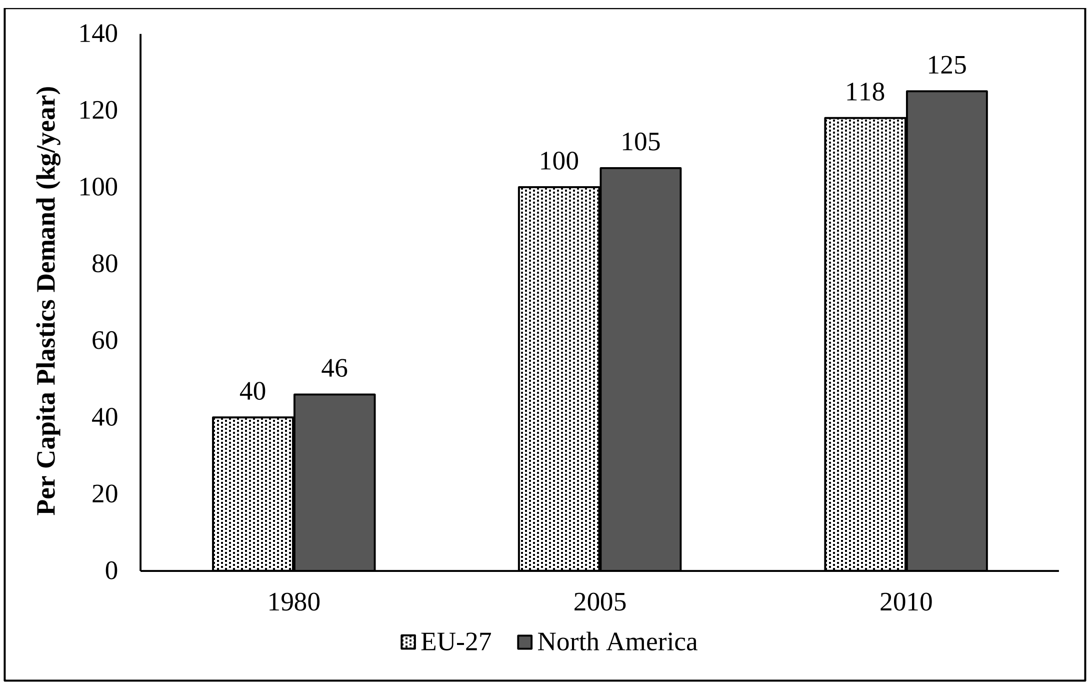

Figure 3*: Per Capita Demand Plastic Consumption kg per year (1980-2010) (PlasticsEurope, 2007)*

A report by PlasticsEurope (2011) indicated that, in 2010, the packaging industry dominated the European industrial consumption of plastics by 39%, followed by the building and construction industry by 20.6%, the automotive industry by 7.5%, and electrical and electronics by 5.6% (PlasticsEurope, 2011). In Canada, as in EU, the industrial sectors that dominated the use of plastics were similar but with approximately double the consumption in the automotive sector (Industry Canada, 2012). The shipping and packaging industry led Canadian plastics consumption by 39%, followed by construction (33%), automotive (14%), and others (14%), which represents the End-Use-Market (Industry Canada, 2012; Industry Canada, 2011a), Figure 4.

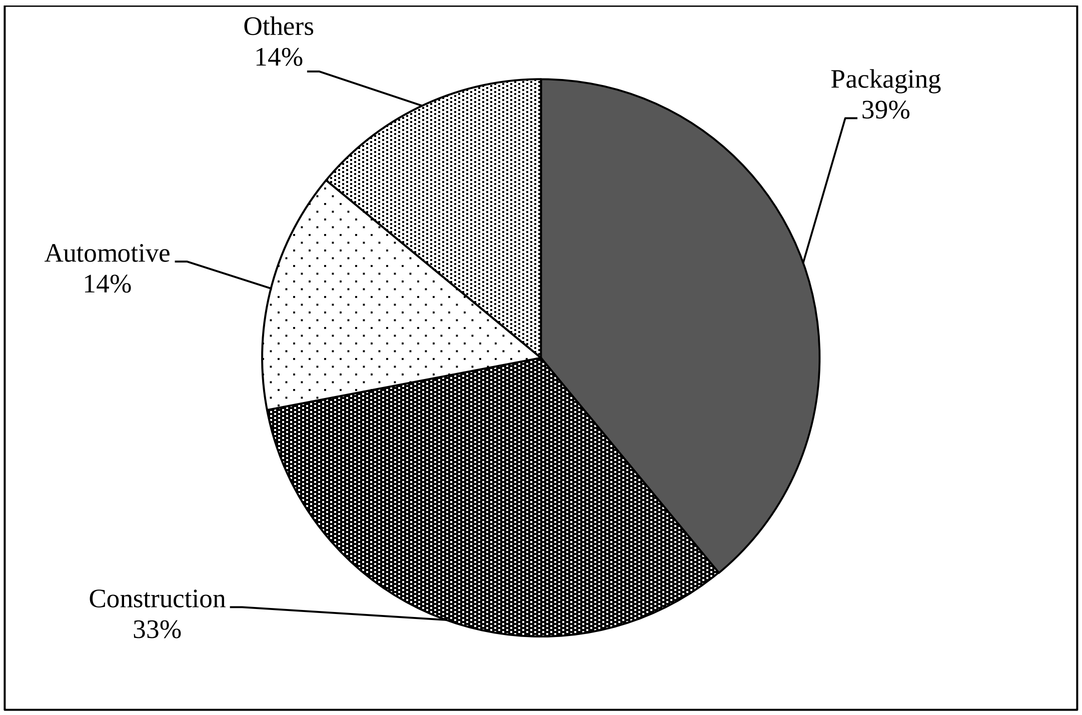

Figure 4*: Canadian Plastics Consumption per Industrial Sectors (percentage of the total) (Industry Canada, 2012; Industry Canada, 2011a)*

A study by the American Chemistry Council (2012a) indicates that the use of plastics and plastic composites in North American will continue to rise. The average weight of plastics and plastic composites increased approximately from 135 kg to 171 kg per vehicle by 27% from 2001 and 2011. This is equal to 9.2% of a car’s total weight.

In 2010, the European demand for plastics was dominated by five main resins: Polypropylene (PP) at 19%; polyethylene (PE), including low-density poly ethylene (LDPE) and linear low-density polyethylene (LLDPE) at 17%; high-density polyethylene (HDPE) at 12%; polyvinyl chloride (PVC) at 12%; and polystyrene (PS) at 8% (PlasticsEurope, 2011).

In 2010 and 2011, American plastics demand was led by PE (HDPE, LLDPE and LDPE), followed by PP, PVC, PS and expandable polystyrene (EPS) (ACC, 2012b). During approximately the same time period, industry output in Canada was dominated by polyethylene (PE) resins (64%), followed by melamine formaldehyde (MF), urea formaldehyde (UF) and phenol formaldehyde (PF) resins (24.5%), and other resins (e.g., polystyrene, nylon, polyvinyl chloride, and rubber) (11.5%) (Industry Canada, 2011b) as illustrated in Figure 5.

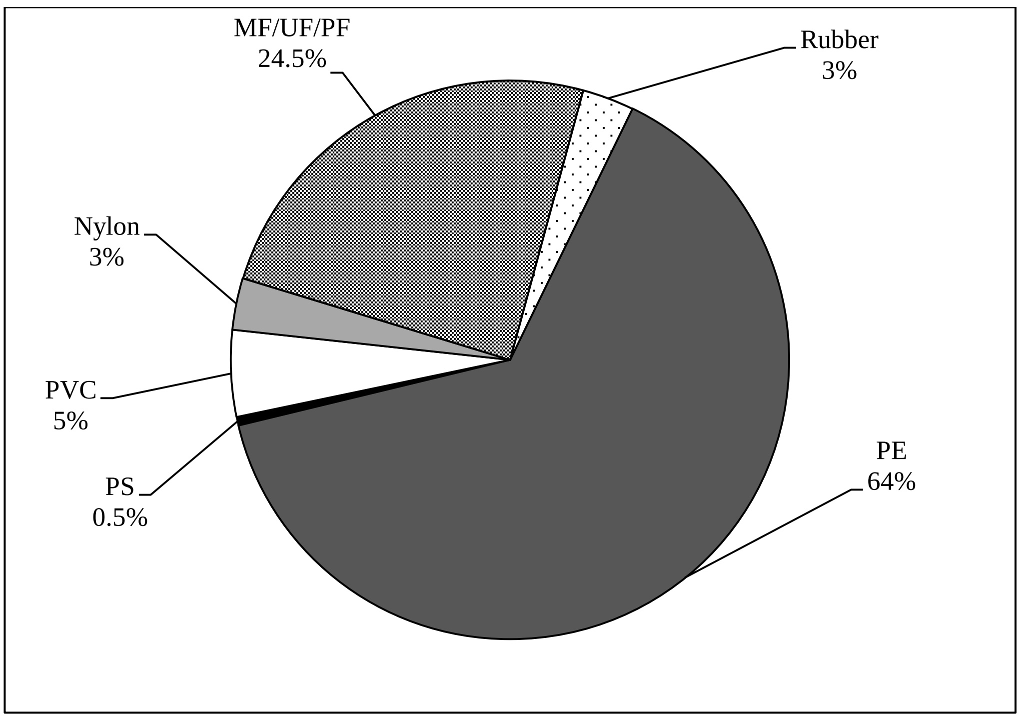

Figure 5*: Canadian industry output by Resin Type (percentage of total)*

By 2011, the use of plastics and composites in the North American automotive industry was dominated by polypropylene resins at 24%, followed by polyurethane at 15%, other engineering resins at 13% and nylon at 12% (ACC, 2012a), Figure 6.

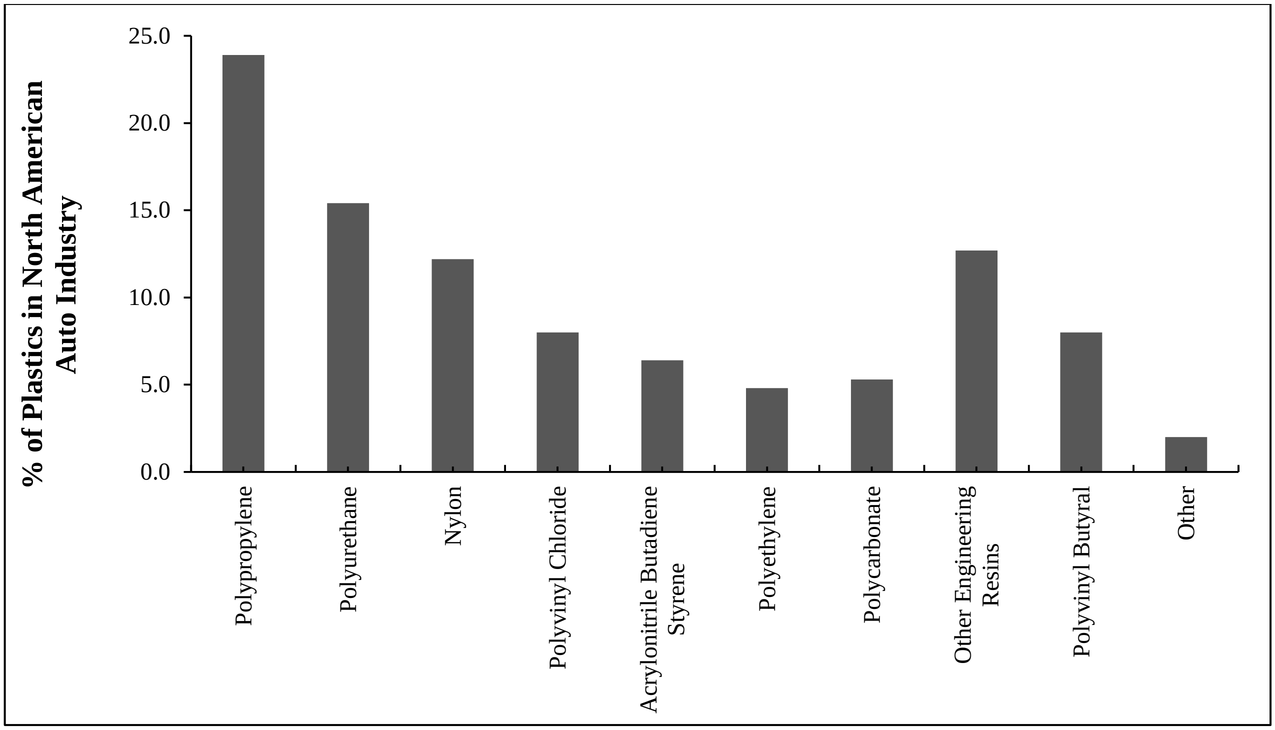

Figure 6*: The percentage of average plastics/composites used in light vehicles in 2011 (ACC, 2012a).*

The evolution of the Canadian plastics industry in the past half-century has been impressive, robustly supporting the Canadian economy and even surpassing the growth of the total economy and other sectors (Empey & Clark, 2005). There are similarities between Canada's and the rest of the world's plastic industries, especially with regards to the problems they face, such as globalization, the cost and depletion of non-renewable resources, global warming, waste generation, and investment in new technologies (Empey & Clark, 2005).

PlasticsEurope (2007 and 2011) and Pilz, Brandt and Fehringer (2010) claim that, in comparison to steel, the main advantages of plastics (relevant to this study) are the reduction in energy and non-renewable resource use and carbon dioxide (CO_2) emissions, especially during the use phase (PlasticsEurope, 2007; PlasticsEurope, 2011; Pilz, Brandt, & Fehringer, 2010). Specifically, non-renewable resource use may be reduced by 22.4 million tonnes (PlasticsEurope, 2007; PlasticsEurope, 2011), GHG emissions by 124 million tonnes, and energy consumption in the total life-cycle by 2.42 million GJ; additionally, a 107 million-tonne reduction of total mass for the same functional units is also anticipated (Pilz et al., 2010; UNFCCC, 2009).

According to their response when exposed to heat, plastic polymers are categorized into two basic groups: thermosets and thermoplastics. Thermoset plastics are polymers that can be supplied in liquid or powder form, are produced and formed upon heating, and cannot be re-processed or re-melted. Thermoset materials cannot be recycled to produce new thermosets, but they can be used as cheap fillers in other products. Examples of thermosets are epoxies (EP), furan, polyurethane cast elastomers (EP), unsaturated polyesters (UP), and phenolic (PF) (Society of the Plastics Industry [SPI], 2012; BPF, 2012).

Thermoplastic polymers are mass-produced in pellet form and become soft, more fluid and formable when heated, but do not cure or set. They can be melted and hardened repetitively by heating and cooling, allowing them to be reused to produce new plastic shapes (Hudson, 2011; Society of Plastics Engineers, 2001; SPI, 2012). Examples of thermoplastics include acrylonitrile butadiene styrene (ABS), nylons (polyamides) (PA), polypropylene (PP), polyethylene (high- and low-density) (HDPE & LDPE), polyvinyl chloride (PVC), and polyethylene terephthalate (PET) (BPF, 2012). Polybutylene succinate (PBS), the subject of this study, is a thermoplastic (Han, Wang, & Wu, 2012; Tsai & Wertheim, 2001).

2.1.2 Bio-plastics

When the grassroots-spawned environmental movement of the late 1900s sparked an interest in the general public to reduce environmental burdens associated with concerns over the landfill volume those plastics occupied, biodegradable plastics were introduced, leading in recent years to the production of PBS (Gironi & Piemonte, 2011).

Although not all bio-based polymers are biodegradable and not all biodegradable polymers are bio-based, both are called bio-plastics (Chin & Uematsu, 2011). There are three types of bio-plastics, as identified by Tokiwa et al. (2009) (see Figure 7): (1) bio-based non-biodegradable polymers such as polyethylene synthetized from biomass or renewable sources, nylon 11 and acetyl cellulose (AcC); (2) bio-based biodegradable polymers such as poly lactic acid (PLA) and poly hydroxyalkanoates (PHAs) from starch (Corn Refiners Association, 2012), poly hydroxybutyrate (PHB) and poly butylene succinate (PBS); and (3) non-bio-based biodegradable polymers such as poly butylene succinate (PBS), polycaprolactone (PCL) and polyethylene suberate (PES).

Corn and potato are currently the most popular sources for starches used in bio-material manufacturing (Agriculture and Agri-Food Canada, 2009).

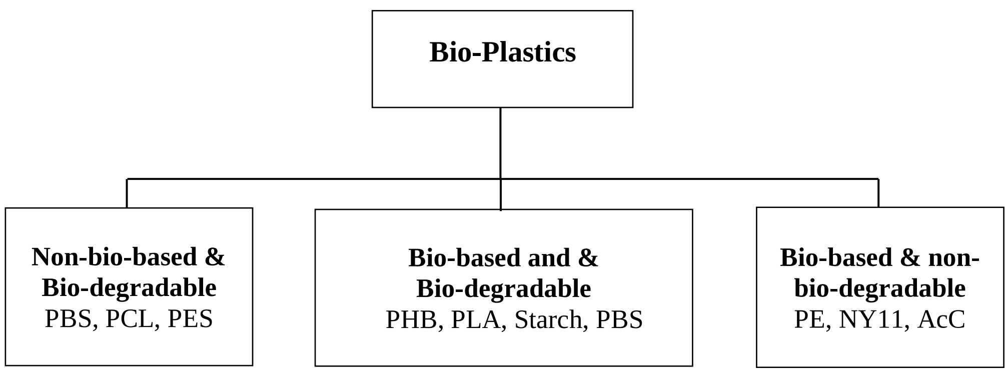

Figure 7*: The three types of bio-plastic (Tokiwa et al., 2009).*

The social and financial impacts associated with the use of food crops as industrial feedstock are outside this study's scope, but the problem of global food supply shortages and the increase in food prices stemming from such agricultural pursuits are significant considerations (Bailey, 2013; Haines & Van Gerpen, 2012; Hao, Colson, Karali, & Wetzstein, 2013; Chen & Khanna, 2013), although some researchers disagree (Zilberman, Hochman, Rajagopal, Sexton, & Timilsina, 2012). The study by Zilberman et al. (2012) argues that the relationship between bio-fuels and food prices is not defined and that there are other factors needed to be taken into account such as the location and the tupe of crops and fuels considered. Nonetheless, studies acknowledge the potential negative impacts associated with the extensive use of water, fertilizers, herbicides, deforestation and habitat loss on farms that provide feedstock to the bio-plastics industry (Stockholm Environment Institute, 2007; Reijnders, 2006).

2.1.3 Bio-based Polymers

Bio-based polymers are identified as polymers that are entirely or partly produced from biological sources of materials and feedstock, such as food crops, wood, grass and agricultural by-products (Flieger, Kantorova, Prell, Rezanka, & Votruba, 2003; Mohanty, Misra, & Drzal, 2002). These polymers are supposedly non-toxic and can be designed to be biodegradable or recyclable (Mohanty et al., 2002). According to Bray (2008) as cited by Tullo (2008), in 2007 natural rubber represents 68% of all bio-based polymers on the market, while cellulosic and other polymers account for 29% and 3%, respectively. Other polymers refer to PLA with 38%, urethanes with 26%, and glycerin-based materials, nylon resins and polyhydroxy-alkanoates (PHA) with 12% each, Figure 8.

According to Flieger et al. (2003) and Babu, O'Connor, & Seeram (2013), bio-based polymers could be produced either naturally or chemically. Naturally means by biological (living) systems (e.g., micro-organisms, plants and animals), while chemically means synthesized from biological materials (e.g., sugar, starch, natural fats or oils).

Pilz et al. (2010) identified two technologies to produce bio-based polymers. The first starts with the production of a monomer (e.g., lactic acid) to produce a polymer in a subsequent stage. The second technology starts with the production of a high volume monomer (e.g., polyethylene) or its derivatives. Both technologies have been proven feasible. However, what is important and relevant to this study is the non-renewable energy used in production chain (e.g., the energy mix), which has an influence on the environmental profile of polymers in general (Pilz et al., 2010).

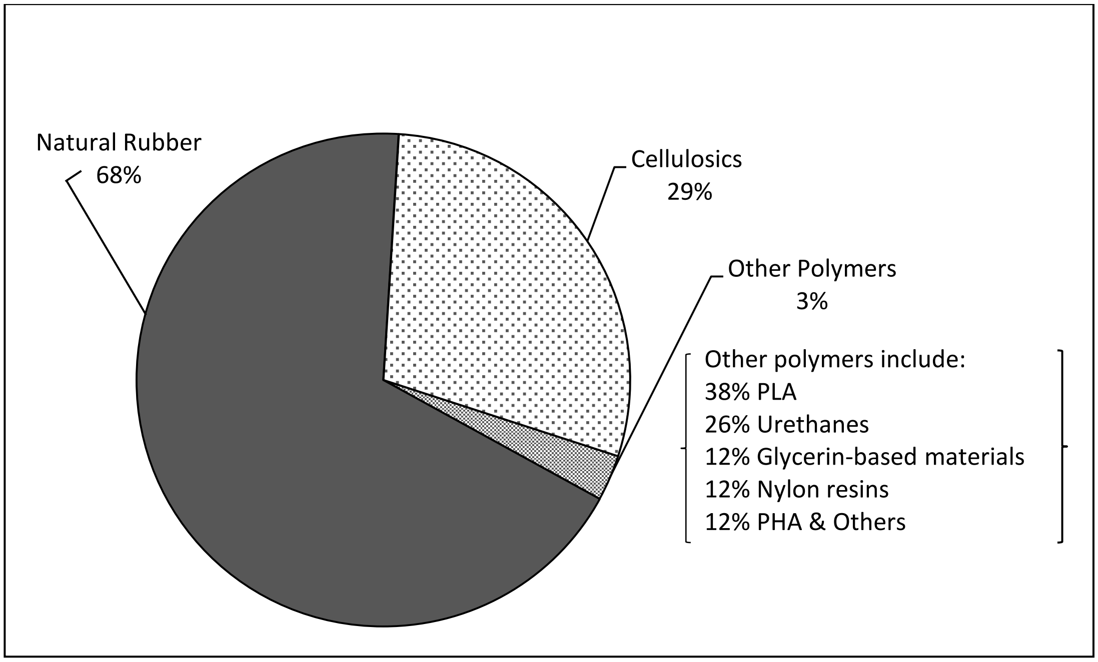

Figure 8*: Global bio-based polymers based on 2007 production data (Source: SRI Consulting)*

Despite bio-based polymers still being in their preliminary development phase, interest in them and their composites is growing (Flieger et al., 2003; Mittal, 2011). Compared to conventional counterparts, the market share of bio-based polymers is relatively low, and thus the scale of environmental benefits associated with their production is minimal. Bachmann (2003) as cited by (USDA, 2008b) predicted the world bio-based polymers market to grow from 5-10% to 10-20% between 2010 and 2025. Here, it is important to understand that growth in bio-based polymers development and production is a potential step for the chemical industry towards a better environmental profile compared to conventional counterparts (Gross & Kalra, 2002; Pilz et al., 2010), as measured by the potential for lower energy consumption and GHG emissions (OECD, 2011; Drachman, 2009; Gielen, Newman, & Patel, 2008). A study by de Jong, Higson, Walsh, & Wellisch (2012) confirmed the latter and indicated that the potential positive impact will affect the petrochemical industries, and the downstream users. That is claimed to be achievable if only “substantial progress is made in White Biotechnology and if the use of lignocellulosic feedstocks is successfully developed" (Patel et al., 2006, p. V), and confirmed by The European Association for Bioindustries, EuropaBio (2008).

Studies suggest that reductions in GHG emissions are attributable to the assumption that bio-based materials are carbon neutral and balance CO_2 emissions emitted during the production and disposal phases (Hobson & Carus, 2012; Dahlke, Larbig, Scherzer, & Poltrock, 1998; Toyota Motor Corporation, 2012; Gross & Kalra, 2002).

Mittal (2011) and Hermann, Blok, & Patel (2007) suggest that the increased interest in the production of bulk bio-based chemicals in general and polymers in particular are influenced by two environmental urgencies. The first is related to the uncertainty of future petroleum supplies and associated prices, given that conventional chemicals and polymers are petrochemically derived. The second urgency is related to the need to produce chemicals and polymers with lower environmental impacts, in particular global warming. In some regions, legislation has encouraged industry leaders to focus more on bio-based polymers rather than conventional ones (Wallach, 2012; Bianco, Litz, Gottlieb, & Damassa, 2010; Hsu & Elliot, 2009; Bennett Jones 2008; Mittal, 2011). These necessities have led to a new perspective based on sustainable development and eco-design values, leading to interest in the production of more environmentally safe bio-based materials (Averous & Boquillon, 2004).

In 2010, the European Climate Change Program (ECCP) estimated the savings in GHG emissions from the bio-based plastics produced in the European Union to be 2.0-4.0 million tonnes (Crank et al., 2005). A more optimistic expectation estimated the reduction at 500-1,000 million tonnes (Hermann et al., 2007). Meanwhile, Drachman (2009) claimed that the use of bio-based materials will result in a 20% reduction in GHG emissions and energy consumption, and Gross and Karla (2002) stated that the potential reduction in CO_2 emissions to be 10 million tonnes as a result of the anticipated production of 3.6 million tonnes of PLA alone by the year 2020. In Canada, bio-based materials have the potential to reduce GHG emissions, improve environmental sustainability, and stimulate the economy (Sparling, Laughland, & Mitura, 2009; Pollution Probe and BIOCAP Canada, 2004).

A study by BCC Research (2012) indicated that the global market trend for the use of bio-based plastics reflects a continuous average annual growth of more than 33% between 2010 and 2016 (estimated). The European market achieved nearly 34% growth in 2010 and 35.3% in 2011 and is expected to increase another 32.1% by 2016. The NAFTA market grew at 37.5% and 36.5% in 2010 and 2011, respectively, and is estimated a 33% by 2016 (BCC Research, 2012) (Figure 9).

2.1.4 Biodegradable Polymers

Conventional plastics end-of-life (E.O.L) management is a significant concern of environmentalists (Bohlmann, 2005; Rochman et al., 2013; European Commission, 2011a)**,** sociologists (Eriksson & Finnveden, 2009), and planners. Currently, most plastics end up as solid wastes in already overfilled and continuously declining landfill locations (Bohlmann, 2005). While plastics could be recycled and re-used, there are restrictions when it comes to, for example, food packaging (Health Canada, 2011). In "Division 23 of the Food and Drugs Regulation", Health Canada published guidelines addressing concerns for using recycled plastics in food packaging due to the permeablity of plastics (Health Canada, 2011; Stewardship Ontario, 2005). Incineration is another solution that is less environmentally preferable in comparison to recycling as there are concerns regarding emissions from incinerators (Eriksson & Finnveden, 2009; Bohlmann, 2005).

Most important, however, is that conventional plastics do not degrade easily in natural landfills and composting environments. Some researchers consider the disposal of conventional plastic wastes in landfills in comparison to recycling and incineration end-of-life treatment options an environmental burden (Tokiwa et al., 2009; Thompson, Moore, vom Saal, & Swan, 2009; Barnes, Galgani, Thompson, & Barlaz, 2009; Huda, Drzal, Mohanty, & Misra, 2008).

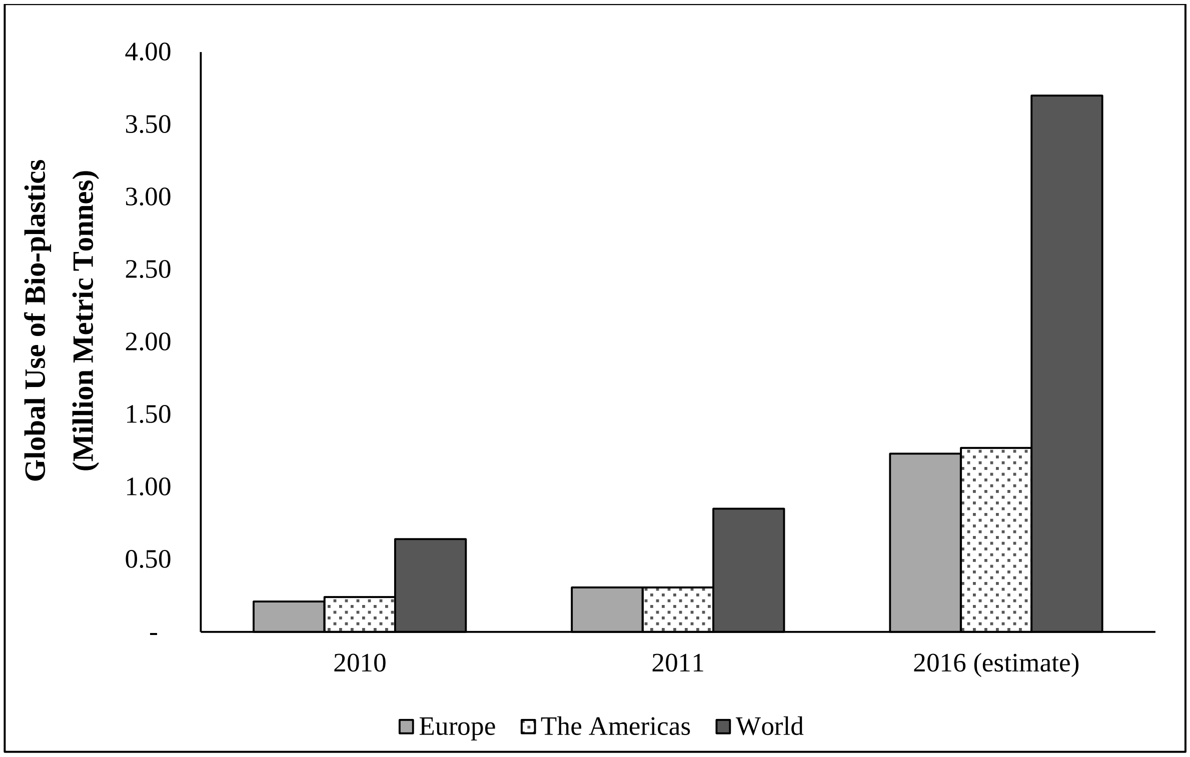

Figure 9*: Global Use of Bio-based Plastics (2010-2016) (BCC Research, 2012)*

The development of biodegradable polymers such as polybutylene succinate (PBS) and polylactic acid (PLA) resulted from an exploration of potential solutions to the environmental problems of plastic waste's end-of-life (Kim, Yang, & Kim, 2005; Tokiwa et al.,2009; Zheng, Yanful, & Bassi, 2005) especially for products like agricultural films and chainsaw lubricants that are in direct contact with the environment (Drachman, 2009).

Three types of degradable plastics are identified: biodegradable plastics that naturally degrade under the influence of microorganisms; biologically degradable plastics known as compostable plastics; and photodegradable/Oxo-degradable plastics that degrade in sunlight (Gautam, 2009; European Comission, 2011b). The natural degradation of biodegradable plastics occurs at the EOL in landfills under the effect of micro-organism enzymes such as bacteria, fungi and algae. The degradation can happen within a few weeks, as claimed by Flieger et al. (2003) and Gautam (2009), although others disagree (Nolan-Itu, 2002; Narayan, 2009; Song, Murphy, Narayan, & Davies, 2009; Cho, Moon, Kim, Nam, & Kim, 2011).

During degradation, plastics undergo a process called depolymerisation, where polymers become low-weight molecular monomers, dimers and oligomers. Subsequently, these oligomers become biomass, minerals, salt, water, and gases such as CO_2 (under aerobic conditions) and methane (CH_4) (under anaerobic conditions). Humic substances are the end-product of any degrading organic material (Tokiwa et al., 2009; Chin & Uematsu, 2011; Bohlmann, 2005). Standards define biodegradable plastics as those plastics that undergo substantial and irreversible changes in their chemical structure under the influence of naturally existing microorganisms (Narayan & Pettigrew, 1999; Hemjinda, Krzan, Chiellini, & Miertus, 2007; Czichos, Saito, & Smith, 2011). These changes are accompanied by the loss of standard physical and mechanical properties (Kolybaba et al., 2003). Standards were created to test the degradability of biodegradable plastics, such as ASTM D6400-99, ISO 14855-2, DIN V49000 and EN 13432 (Chin & Uematsu, 2011; Ichikawa & Mizukoshi, 2012; Hemjinda et al., 2007; Scott, Lemaire, Jakubowics, & Ojeda, 2011).

In recent years, global chemical producers have started commercializing a wide range of biodegradable polymers including well known synthetized polyesters, like PBS and PLA, (Shih & Jeng, 2011) and aliphatic-aromatic copolyesters (Nakajima-Kambe, Ichihashi, Matsuzoe, Kato, & Shintani, 2009). This was initiated to match the demand for biodegradable polymers that is expected to grow by 30% on an annual basis, as claimed by Leaversuch (2002). A recent study by Malveda & Yokose (2012) on biodegradable polymers challenged those growth expectations, and indicated that the annual growth in the North America, Europe and Asia consumption of biodegradable polymers is expected to reach only 13% in 2014. This growth is influenced mainly by increasing demand from the packaging industry (Malveda & Yokose, 2012). In North America, the growth in the consumption of biodegradable polymers is essentially driven by competitive pricing, continuous improvement in processability, overall competitive properties, and concerns on local, state and federal levels regarding the end-of-life of non-biodegradable polymers (Malveda & Yokose, 2012).

Based on their physical, chemical and structural properties, synthesized biodegradable polymers were categorized in two main groups. The first is highly crystalized and flexible, similar to the low density polyethylene (LDPE). The second is semi-amorphous and more rigid, similar to polypropylene (PP), polystyrene (PS) and polyethylene terephthalate (PET) (Leaversuch, 2002). PBS falls under the second group of biodegradable polymers. Averous & Boquillon (2004), Gautam (2009) and Mittal (2011) suggested a different classification to categorize biodegradable polymers based on their production routes. Three routes employ biomass, and the fourth is petrochemically synthesized, such as PBS.

2.1.5 Natural Fibres

Mohanty et al. (2002) and Lucintel (2011) classified natural fibres into wood fibres and non-wood natural fibres (Figure 10).

Non-wood natural fibres such as switchgrass are increasingly gaining attention as reinforcing materials as an alternative to synthetic fibres (i.e., glass fibres) in industrial applications (Williams & Wool, 2000; Corbiere-Nicollier et al., 2001; Joshi, Drzal, Mohanty & Arora, 2003).

In comparison with synthetic fibres, natural fibres are abundant, biodegradable, lightweight, cheaper, have a lower density, require less energy to be manufactured, have excellent chemical resistance and good molding ability, and are processable in conventional processing equipment (Tantatherdtam et al., 2009; Joshi et al., 2003; Corbiere-Nicollier et al., 2001; Xu, Jayaraman, Morin, & Pecqueux, 2008; Westman, Fifield, Simmons, Laddha, & Kafentzis, 2010). Among the issues that require research and development are the dimensional stability of natural fibres (Kolybaba et al., 2003), resin compatibility and water absorption (Westman et al., 2010). Importantly, natural fibres are not considered to be a health hazard in comparison with glass fibres (Suddell, Evans, Isaac, & Crosky, 2002).

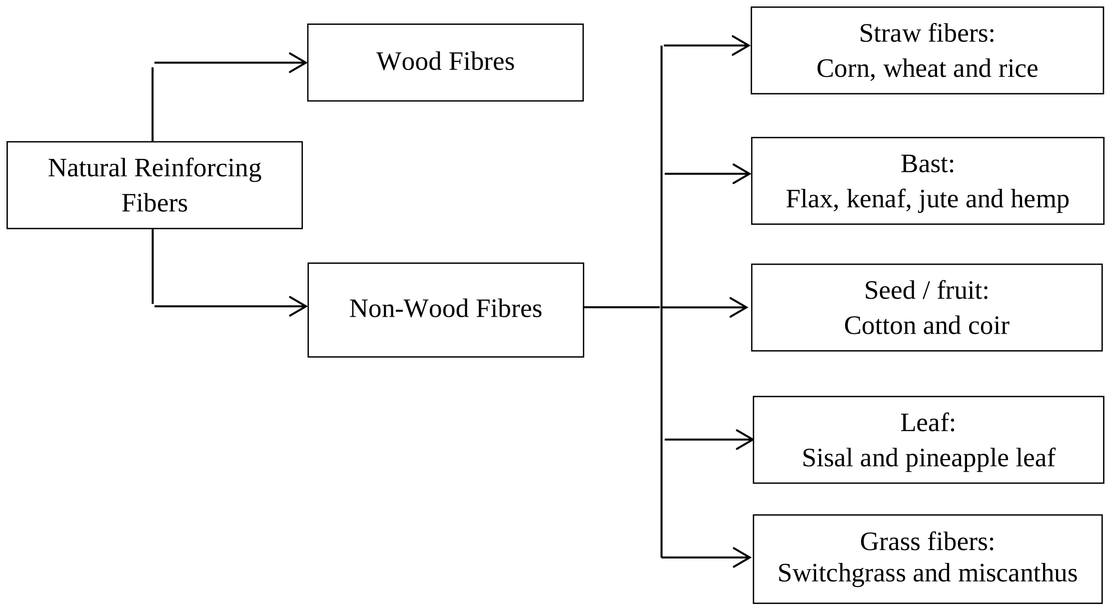

Figure 10*: Natural fibres categories and sub-categories that are used as reinforcing materials in plastic composites (Mohanty et al., 2002)*

Claims have been made that the production of natural fibre composites will result in a better environmental profile compared to glass fibre composites (Wotzel, Wirth, & Flake, 1999; Joshi et al., 2003; Nabi Saheb & Jog, 1999; Santulli, Janssen, & Jeronimidis, 2005). Joshi et al. (2003) claim that natural fibre composites are less dependent on non-renewable energy and materials, and result in lower pollutant and GHG emissions, while enhancing energy recovery at the end of life. However, the question is, whether this superiority can be generalized to all plastic composites made with natural fibres (Corbiere et al., 2001; Xu et al., 2008). A comprehensive cradle-to-grave life-cycle assessment is needed to determine the potential environmental impacts of natural fibre composites in comparison with glass fibre composites (Joshi et al., 2003; Corbiere-Nicollier et al., 2001; Xu et al., 2008). As there are some factors that negatively affect the use perception of natural fibre composites, such as lower mechanical properties (i.e., tensile strength and flexural strength), lower thermal stability, less colour stability, increased odour, and volatile organic chemicals releases throughout the life-cycle of these composites (Tran et al., 2011).

The extraction and translocation of materials and agricultural practices have the biggest effect on the environmental impacts of biomass, including factors such as human toxicity, eco-toxicity, and water eutrophication (Corbiere-Nicollier et al., 2001; Xu et al., 2008). Poor agricultural practices can cause contamination of soil and water by heavy metals and phosphate due to the use of fertilizers, while the use of herbicides has little influence (Corbiere-Nicollier et al., 2001). Moreover, if the same lands were used for food crop cultivation during the rotation period, there is a possibility that the crops will be contaminated making it inedible or a threat to human health (Corbiere-Nicollier et al., 2001).

Plastic composites that contain natural fibres and biodegradable polymers such as PBS and PLA are the focus of the research and development of many companies. Their aim is to produce plastics with a better environmental profile to replace conventional counterparts (Tantatherdtam et al., 2009; Santulli et al., 2005; van Dam, 2008).

The increase in demand for natural fibre composites (Figure 11), is strongly influenced by government support, environmental regulations, economic, political and natural disasters, technological advancements, customer acceptance, positive feed-back from end-of-use industries, and escalating prices of petroleum-based counterparts (Lucintel, 2011; Suddell, 2008; BioProducts Canada, 2004).

Between 2005 and 2010, the global natural fibre composites market grew annually by 15%, from $1.09 billion to $2.17 billion. It is expected to reach $3.81 billion by 2016, with an annual growth of 10%, as in Figure 11(Lucintel, 2011). Automotive and construction industries are leading the growth as the major consumers of natural fibre composites (Lucintel, 2011; Xu et al., 2008; Beckwith, 2008). Led by the construction industries, North America is the biggest global market for wood fibre composites (Lucintel, 2011). The future demand of this market is estimated by Kline & Company INC. (2000) as cited by (Mohanty, Misra, & Drzal (2005) to be equal to 0.2 million tonnes. It was estimated that the North American natural fibre composites market value would grow tenfold between 2001 and 2006, from $150 million to $1.5 billion (Kline & Company INC., 2002).

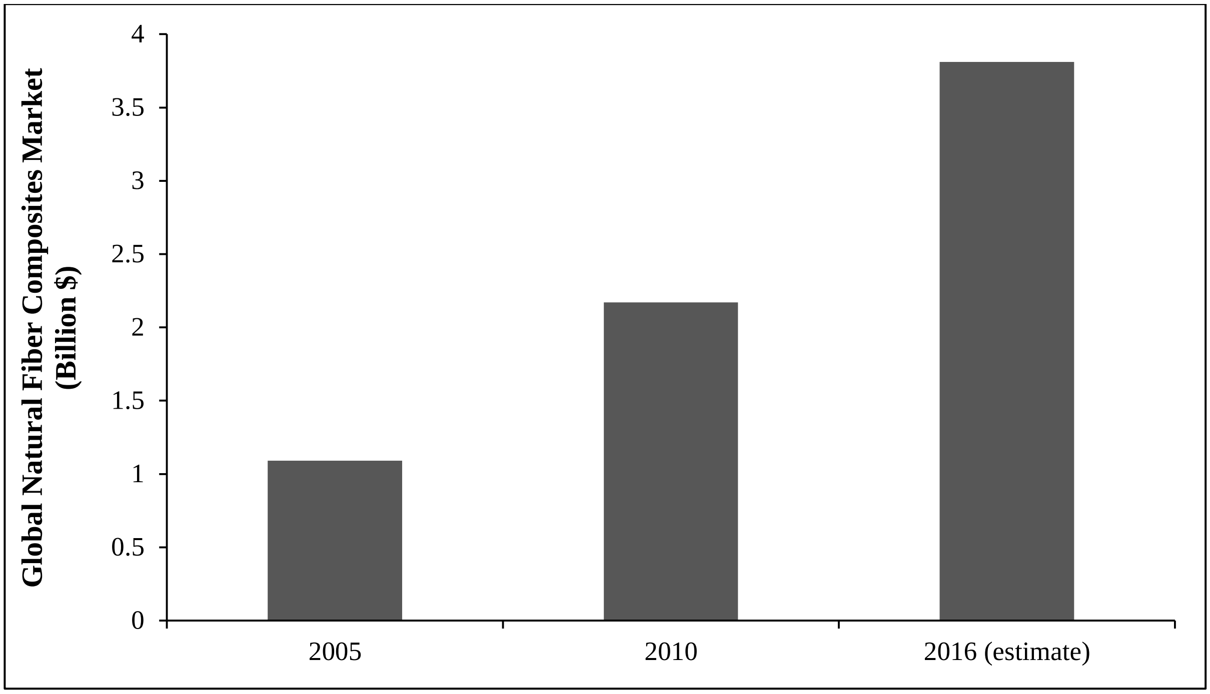

Figure 11*: Global natural fibre composites market value 2005-2016 (Lucintel, 2011)*

Europe, led by the automotive industry, is the biggest global market for non-wood fibre composites, mostly bast fibres (i.e., flax, kenaf and hemp) composites (Carus & Gahle, 2008; Lucintel, 2011).

In 2001, the European market was estimated to be equal to 30,000 to 40,000 tonnes, with a market value of $50 million to $65 million. This market was projected to grow at the same rate as the North American market (Kline & Company INC., 2002). The main advantage of using natural fibre composites (e.g., flax fibre and China reed fibre reinforced polypropylene composites) in the automotive industry is that these composites are lighter in weight and have good thermal and acoustic insulation properties (Lucintel, 2011; Joshi et al., 2003).

2.1.6 Bio-based Plastic Composites in the Automotive Industry

Henry Ford's (1863-1947) ideas included the use of bio-based plastics in the 1930s (Crawford, 2009). Ford used hemp, wood pulp, cotton, flax and ramie as sources for natural fibres to reinforce plastic composites for auto interiors (Crawford, 2009). In 1940, the world witnessed the production of the first plastic car using 70% cellulose-based plastic (PlasticsEurope, 2010). The car was 30% lighter than its metal counterparts, but the cheap prices of steel and gasoline in the late 1940s made the plastic car financially unviable (PlasticsEurope, 2010). In 2009, the Ford Motor Company continued the bio-based trend in the auto industry, becoming the first company to use wheat straw as reinforcing material in back seats plastic bins on the 2010 Ford Flex model (de Guzman, 2010). As minor as these bins may seem in the larger scheme of automotive construction, they will nonetheless help reduce petrol consumption by 9 tonnes per year and CO_2 emissions by 14 tonnes per year, as claimed by the Ford Motor Company (de Guzman, 2010). A report by Hill, Swiecki, and Cergger (2012) proposes that by the year 2020, a strict 54.5 mile per gallon (mpg) fuel economy target could only be met if lighter materials are used. This mpg is the new 2020 fuel economy standard target that the Obama administration and the automotive companies have agreed upon (Hill et al., 2012). Several North American regulations related to vehicle GHG emissions are already in use and are continously under revision in efforts to improve them (EPA, 2011a; EPA, 2011b; Ministry of Justice, 2010; Government of Canada, 2012; EPA & National Highway Traffic Safety Administration, 2012). To understand what drives all of the previously mentioned regulations, we need to take into consideration the magnitude of passenger vehicles' contribution to GHG emissions. In 2011, passenger vehicles accounted for 17% of the United States total GHG emissions. It is expected that these regulations will help reduce car-related GHG emissions by up to 40% in 2025 (Center for Climate and Energy Solutions, 2012). In a Canadian context, it is estimated that the implementation of new regulations the new models from 2017 to 2025 will result in 5% lower GHG emissions annually in comparison to 2008 models (Environment Canada, 2012).

However, the Organisation Internationale des Constructeurs d'Automobiles (OICA) (2010) disagrees with the current legislators' approaches to reduce automotive GHG emissions and considers them incomplete and narrow (McCurdy, 2010). OICA (2010) states that the current worldwide automotive legislative approaches are focused on shifting the burden of GHG emissions reduction totally on to the automotive industry (International Organization of Motor Vehicle Manufacturers [OICA], 2010). Instead, an integrated approach that includes all relevant stakeholders should be adopted (OICA, 2007). The proposed OICA integrated approach focuses on five pillars: 1) Automaker and vehicle technology, e.g., the production of high-performance light-weight cars with low emissions; 2) energy providers, such as the production of alternative highly efficient bio-fuels and ensuring that the required infrastructure is available; 3) driver behavior by improving drivers' driving techniques to reduce fuel consumption and GHG emissions that eventually improve road safety*;* 4) traffic and infrastructure planning and management, by improving traffic flow and eradicating traffic congestions*;* and 5) CO_2 related taxation, by influencing consumers' decisions when buying a new vehicles (OICA, 2010). Our study falls concerns the first 'pillar' of automaker and vehicle technology.

Recent initiatives by the automotive sector worldwide to become a more environmentally responsible industry arose as a response to societal concerns and pressure from governments (Hill et al., 2012; Lammers & Kromer, 2002; Suddell et al., 2002). For instance, the use of bio-based plastic materials and natural fibre composite materials was strongly motivated by European Union (EU) regulations (Suddell et al., 2002). One representative regulation is the European Directive 2000/53/EC, which stipulates that 95% of vehicles must be partly degradable or recyclable by 2015 (Suddell et al., 2002). In Asia, strict regulations that are similar to those implemented in the EU have been instituted by several governments, while in the U.S. such regulations are not yet in place or are not as strict as in other parts of the world (Hill et al., 2012). Yet, there are already roughly 1.5 million vehicles in the U.S. already using natural fibre (e.g., kenaf, jute, flax, hemp and sisal) composite materials (e.g., polypropylene and polyester) (Bledzki, Farouk, & Sperber, 2006). Through the use of bio-plastics and bio-fibre composites, the trend is to design and manufacture interior automotive parts that are lighter in weight in comparison to current counterparts (PlasticsEurope, 2012a; Chiaberge, 2011), safer for passengers, more fuel and energy efficient, shock absorbent, durable, and economically sound (Lammers & Kromer, 2002; PlasticsEurope, 2012a; de Guzman, 2010). PlasticsEurope (2010) claim that 12-15% of a modern car's total weight (average 120 kg) is made of plastic and plastic composites, leading to a 20-35% improvement in the car's fuel economy (PlasticsEurope, 2010; PlasticsEurope, 2012a). If these parts, materials and composites are biodegradable (e.g., PBS, PLA, PHA), that would be considered an additional advantage, especially at the E.O.L. phase.

Richard Bell, DuPont's Global Development Manager for Renewable Materials, stated that while the automotive industry "is one of several market segments driving toward greater use of high-performance bio-based materials to meet their sustainability challenges, the vast majority of available bio-based polymers fail to achieve the balance between the high performance required by the automotive industry and the requisite cost-effectiveness" (de Guzman, 2010, para. 20). Today, bio-plastic and bio-plastic composite auto parts are used in door panels and inserts, rear parcel shelves, seatbacks, headliners, package trays, dashboards, trunk liners, glove box doors, gear shift knobs, horn buttons, accelerator pedals, distributor heads, interior trims, steering wheels, dashboards, body panels, spare tire covers, and spare-wheel pans (Terasawa et al., 2008; Xu et al., 2008; Crawford, 2009; Lucintel, 2011). In addition, they can also be found as glass replacement in headlight lenses, bezels and windscreens, steel for fenders, spoilers, and dashboards (PlasticsEurope, 2010). PlasticsEurope claims that the stability of the replacement plastic parts is equal to or greater than the replaced parts regarding safety requirements (PlasticsEurope, 2010).

By 2015, the global automotive sector is expected to consume 66,000 tonnes of natural fibre composites for door panels, inserts, and rare shelves, 106,000 tonnes for spare wheels pan covers and headliners, and 147,000 tonnes for bumpers and protection trims (Lucintel, 2011). A study by Kline & Company INC. (2000) (as cited by Mohanty et al., 2005) reports that North American demand for natural fibres (wood and non-wood) as reinforcing materials for plastic composites is projected to grow annually by 30% for automotive applications and 60% for designated building products. A study by Bledzki et al. (2006) confirms the latter, and predicts that the North American and European automotive industries will lead the global annual growth of natural fibres use by nearly 54% (Bledzki et al., 2006). The reason for this growth, according to Suddell et al. (2002), is that almost all major car manufacturers use natural fibre composites for automotive interior parts. Germany, Europe’s biggest automotive producer, continues to lead the European demand for natural fibres. Germany’s demand for natural fibres grew from 4,000 tonnes in 1996 to 18,000 tonnes in 2003, and from 70,000 tonnes in 2005 to 100,000 tonnes in 2010 (Suddell et al., 2002; Bledzki et al., 2006). Although Karus & Gahle (2006) agree that the trend is emerging, they dispute its alleged magnitude.

2.1.7 Poly (butylene succinate) (PBS)

There are three production routes for the production of PBS polymer. These routes are influenced by the source of carbon (renewable and non-renewable) and the type of feedstock used in the production process. They then adhere to the same technological sequence that starts with the esterification of succinic acid and 1, 4 Butanediol (BDO) to produce PBS oligomers, followed by the polycondensation to produce PBS the polymer (Jacquel et al., 2011; Xu & Guo, 2010; Fujimaki, Hatano, Hokari, Seki, & Takiyama, 1994), Figure 12.

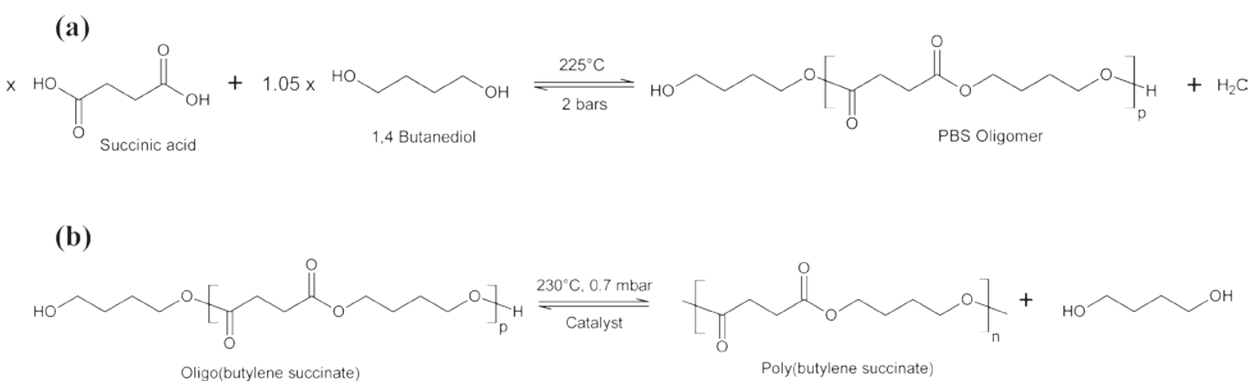

***Figure 12**: Poly (butylene succinate) production sequence. (a) Esterification, (b) Polycondensation (Jacquel et al., 2011)*

Different feedstocks and production routes will result in different potential environmental impacts. The three production routes are petroleum-based, bio-based and hybrid. Currently, almost all PBS produced globally is petroleum-based and mainly manufactured by Showa Denko K.K. of Japan (Showa Denko, 2012; Xu & Guo, 2010). A bio-based route is theoretically a feasible route, because bio-succinic acid is commercially available and produced by more than one producer (Myriant, 2012; BioAmber, 2013; Smidt, 2011). The production of bio-based BDO is still in its early stages and is not yet produced commercially on a large scale (Balboa, 2011; Nexant, 2012; Burk, Van Dien, Burgard, & Niu, 2011). A hybrid production route is the third option using bio-succinic acid and petroleum-based BDO. This route is the scenario to be used for this study. In addition, the envrionmental performace data from the first production route (petroleum-based), as generated by Moussa et al. (2012), will be used for comparison with the hybrid production route.

While a sizeable body of literature has grown recently that discusses the mechanical/physical properties and biodegradability of PBS and its composites (Frollini, Bartolucci, Sisti, & Celli, 2013; Wan & Chen, 2013; Wang et al., 2012; Zeng et al., 2012; Qu, Tan, Ferg, & Hu, 2011; Zhu, Wang, Chen, & Xu, 2009; Tokiwa et al., 2009; Zhao et al., 2005; Kim et al., 2001; Fujimaki, 1998), few studies have addressed the environmental impacts throughout the life cycle of the material (Ichikawa & Mizukoshi, 2012; Moussa et al., 2012; Terasawa, Tsuneoka, Tamura, & Tanase, 2008). This research is intended to help bridge this gap.

2.1.8 Switchgrass

Switchgrass is a non-wood short fibre source (Lips & Elbersen, 2001; Mohanty et al. 2002) and a non-food perennial warm-season C4 prairie grass (WSG) native to North America (Samson, 2007; Elbersen, Christian, Yates, EL Bassam, & Sauerbeck, 2001). Is well-suited to marginal and fallow lands and can grow up to 180 centimeters in height (McLaughlin et al., 1999; McLaughlin & Walsh, 1998). During the North American summer seasons, the productivity yield of switchgrass is 8-12 tonnes per hectare, which is moderate to high compared to, for instance, corn productivity, which is 6.5 tonnes per hectare (Samson, 2007). In the winter, switchgrass productivity decreases but nevertheless remains steady throughout its life-cycle (Samson, 2007), which makes the production cost of switchgrass lower in comparison with other annual crops, as claimed by Vogel (2004) as cited by Torrez, Johnson, & Boe, (2013). In addition to having a relatively high and steady yield, switchgrass is resistant to draught, diseases and pests (Carvalho, 2006) and requires minimal agricultural management practices in terms of fertilizers, herbicides, pesticides, water usage (McLaughlin et al., 1999; Samson et al., 1993; Carvalho, 2006) and the use of heavy machinery, which means less use of non-renewable energy for the cultivation process (Pimentel et al., 2005). Moreover, switchgrass roots can reach 80 centimeters deep in soil, which aids in the protection of soil erosion (Prairie Lands Biomass LLC, 2012; Ma, Wood, & Bransby, 2000; Lips & Elbersen, 2001) and improves natural and wildlife habitat (Duffy & Nanhou, 2001). Switchgrass also helps decrease the amount of residues by 55% and phosphorous and nitrogen run-offs of into surface water by 36% and 39%, respectively (Prairie Lands Biomass LLC, 2012). The Intergovernmental Panel on Climate Change (IPCC), 4th assessment (2007) acknowledged in 2005 that, as a result of extensive agricultural practices, farming was responsible for emitting 5.1-6.1 Gt. Of CO_2-eq (54.1% CH_4 and 45.9% N_2O), which is equivalent to 10-12% of synthetic GHG emissions globally (Smith et al., 2007). The benefits of switchgrass are illustrated in Figure 13. Switchgrass is usually used as forage (Duffy & Nanhou, 2001; Elbersen et al., 2001). However, because of its high lingo-cellulosic content (Samson, 2007; Rinehart, 2006), studies found it suitable to be used in paper production (Law, Kokta, & Mao, 2001; Girouard & Samson, 2000). It has also been tested as an energy crop for the production of electricity and ethanol as a biomass source instead of the conventional non-renewable resources (Duffy & Nanhou, 2001; Elbersen et al., 2001).

In a Canadian context, Samson (2007) claimed that using warm-season grasses like switchgrass as energy crops for the production of electricity and ethanol, instead of conventional feedstock, may have a major impact on the reduction of GHG emissions. It is estimated that using switchgrass pellets to generate electricity will result in 90% lower GHG emissions in comparison with conventional feedstocks (Samson, 2007).

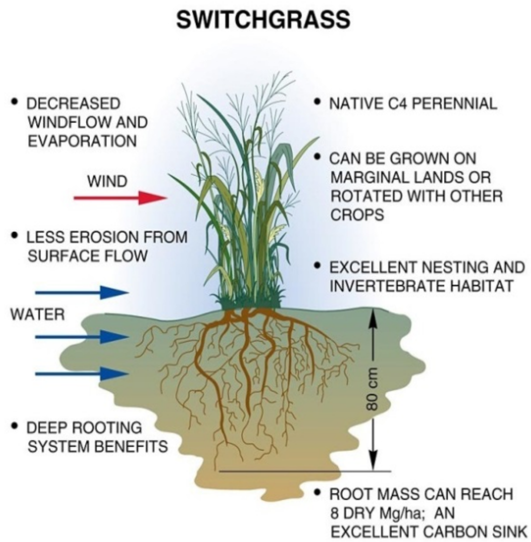

Figure 13*: The ecological benefits of switchgrass (Randolph & Masters, 2008)*

However, switchgrass's biggest advantage is its high photosynthesis yield (Samson, Ho Lem, & Bailey, 2012; Samson, 2007; Rinehart, 2006) compared with other crops that are currently used as biomass. Corn, for example, has a 60% lower photosynthesis yield than switchgrass (Rinehart, 2006). Switchgrass could also help in reducing GHG emissions via soil carbon sequestration process (Duffy & Nanhou, 2001; Samson, 2007). In the U.S., the result of using switchgrass on Conservation Reserve Program (CRP) [4] lands was the storage of 40 tonnes of CO_2 per hectare in comparison to traditional crops (Samson, 2007). In North America, the switchgrass carbon sequestration rate is 6.2-37.0 tonnes CO_2 per hectare per year (Frank, Berdahl, Hanson, Liebig, & Johnson, 2004; Lee, Owens, & Doolittle, 2007), 20-30% higher than corn's sequestration rate (McLaughlin & Walsh, 1998). Hence, switchgrass can be considered a possible

[4] CRP is a voluntary program intended to take highly erodible cropland out of production and stabilize soil loss through planting permanent cover crops. http://www.forestry.state.al.us

solution to mitigate the effects of increased GHG emissions on global warming and climate change (Samson, 2007).

Although the advantages of switchgrass cultivation have been identified by many studies, few have researched the disadvantages. Miller, Chamberlain, Alfaro and Sharp (2011) identified the potential negative impacts for growing switchgrass on fallow lands. Similarly, Hartman, Nippert, Brozco, & Springer (2011) highlighted the ecological disadvantages of cultivating switchgrass on marginal lands and the impacts on wildlife (e.g., possible spread of diseases) and soil quality. Harper and Keyser (2008) emphasize the importance of switchgrass in properly planned harvesting practices in order not to negatively affect the natural habitat. Tulbure, Wimberly and Owens (2012) suggested incorporating climate change impacts on switchgrass yield and economical viability when planning for future development using the plant, while Shastri, Hansen, Rodriguez & Ting (2012) focused on economical and technological related issues that negatively stimulate its industrial use.

Despite these negative aspects, studies identified the potential benefits of using switchgrass as a source for natural fibres in plastic composites (Pfister & Larock, 2012; Van Den Oever, Elbersen, Keijsers, Gosselink, & De Klerk-Engels, 2003; Reddy & Yang, 2007). A recent study on the LCA of switchgrass cultivation in Ontario, Canada, highlighted the potential positive impacts associated with replacing glass fibres with switchgrass fibres as reinforcing material in plastic composites (Kalita, 2012). For this and the previously mentioned reasons, the choice of using switchgrass as the reinforcing material for the proposed hybrid plastic composite was based on its potential positive environmental impacts.

2.1.9 Bio-based vs. Petroleum-based Products

It is essential to understand that bio-based materials are not as-a-rule "environmentally superior" to their conventional counterparts, despite the general assumption that they are. For contributions to climate change this is the case if "GHG emissions from indirect land use change are neglected" (Weiss et al., 2012, p. S176). However, in addition to GHG emissions, there are other impact categories linked to land use, water intake, eutrophication, acidification and ozone depleting materials that play key roles in evaluating a product's environmental impact (Weiss et al., 2012; Tabone, Cregg, Beckman, & Landis, 2010; Goedkoop et al., 2010; Zah et al., 2007; Miller, Landis, & Theis, 2007; Broekema, 2013; Zah et al. 2007; Miller 2007, Milà I Canals et al., 2011). In comparing bio-based materials with petroleum counterparts, Detzel and Kruger (2006) highlighted the importance of "trade-offs" of various potential environmental categories in the process of decision-making. Based on the priority and importance of these trade-offs to the different environmental categories, preference may be given to either bio-based plastics or to conventional resources (Gironi & Piemonte, 2011).

The unavailability of sufficient and reliable data was identified as the main reason for inconclusive results on the environmental impacts of bio-based materials (Milà I Canals et al., 2011). The inconclusive results will increase the uncertainty in LCA results, according to Von Blottnitz & Curran (2007). In addition to the previously mentioned reasons, the continuously changing environment and agricultural conditions add extra complexity to the data collection (Milà I Canals et al., 2011; Malça & Freire, 2010).

A number of LCA studies and reports on bio-based materials were reviewed and discussed below to determine what criteria scholars use to identify the environmental perfromance of a product and in which areas bio-based materials are compared favourly to conventional counterparts. Generally, there is no unified criteria to make such a determination.

Zah et al. (2007) assessed the environmental impact (cradle-to-grave) associated with the production and use of 26 bio-fuels from Switzerland and other countries in comparison with three fossil fuels. They concluded that "not all bio-fuels can reduce environmental impacts as compared to fossil fuels" (Zah et al., 2007, Abstract, para. 4). Although some bio-fuels offered a 30% reduction in cumulative energy demand (CED) and GHG emissions, they increased other environmental impacts, such as smog formation, eutrophication, and ecotoxicity. Hence, the study concluded that with the majority of bio-fuels, a trade-off decision needs to be made between the benefits of GHG reduction and the negative influences of other environmental impacts (Zah et al., 2007).

Studies by Haufe and Carus (2011) and La Rosa et al. (2013) showed that hemp reinforced PP composites in comparison to GF reinforced PP composites used in auto parts required less energy demand and generated fewer GHG emissions. Additionally, the study noted a proportional relationship between the content of hemp fibres in a composite, and the environmental performance of an auto part.

Khoo et al. (2010) compared global warming potential (GWP), acidification, and photochemical ozone formation impacts for the production of carrier bags made from bio-based PHA and petroleum-based PP polymer. The study concluded that bio-based bags (based on U.S. electricity grid mix) caused a 69% higher production of environmental impacts in comparison with petroleum-based bags. The coal-fired energy source used for the production of the bio-based bags had the largest impact on GWP, acidification and photochemical ozone formation. The use of coal-fired electricity in the production of PHA bags resulted in a five-fold greater impact compared to PP bags. Moreover, while using natural gas-fired electricity resulted in the same impacts for both bio- and petroleum-based bags, using geothermal-generated electricity resulted in an 80% less impact than petroleum-based bags (Khoo et al., 2010).

James and Grant (2005) compared degradable and bio-based plastic bags with single-use and reusable petroleum-based plastic bags, using three envrionmental indicators: GHG emissions, resource depletion, and eutrophication. Results indicate that bio-based PLA bags - if not properlly disposed - have higher GHG emissions and higher eutrophication value in comparison to petroleum-based single-use HDPE bags. While HDPE bags had a higher resource depletion value in comparison with PLA bags.

Gironi and Piemonte (2011) conducted an LCA (cradle-to-gate) comparison study between drinking water bottles made of bio-based polylactic acid (PLA) and petroleum-based polyethylene terephthalate (PET). Results indicated that using PLA could help conserve non-renewable resources but will cause stresses to human health and the eco-system quality due to intensive agricultural practice (Gironi & Piemonte, 2011).

Kimura and Horikoshi (2005), in a cradle-to-grave LCA study of bio-based PLA in comparison with petroleum-based polymer (ABS), concluded that the production of PLA caused higher CO_2 emissions than the production of ABS, but only when the sequestration of CO_2 by the crops due to photosynthesis was considered, the production of PLA produced 14% lower CO_2 emissions (Kimura & Horikoshi, 2005). However, this study did not take into consideration any of the other impact categories, such as eutrophication, acidification, land use and water intake, so results are not adequate to give a conclusive result regarding the environmental performance of the bio-based polymer.

Tabone et al. (2010) compared the environmental impacts of 12 bio- and petroleum-based polymers using global warming, fossil fuel depletion, human health and ecotoxicity impacts category. Study results indicated that bio-based polymers caused lower global warming and fossil fuel depletion but had a higher impacts with regards to human health and ecotoxicity. The latter is attributable to extensive agricultural practices.

A study by Terasawa et al. (2008) using CO_2 emissions only to compare automotive interior parts made from PP with the same parts made of a bamboo-fibre-reinforced PBS composite indicated that as a result of the production of PP resins, fewer CO_2 emissions will be emitted in comparison with the composite. On the other hand, when taking into consideration the full life cycle, from harvesting to end-of-life, the composite has almost 50% fewer CO_2 emisisons. The study did not provide a proper justification for using CO_2 emissions as an adequate indicator to describe the environmental performance and whether the use of other impact indicators will provide the same results.

A study by Ichikawa and Mizukoshi (2012) compared GHG emissions associated with the production of PBS, a petroleum-based and biodegradable resin and starch-PBS compound with petroleum-based LDPE and polystyrene (PS) resins. The study concluded that both PBS and the starch-PBS compound have lower GHG emissions than LDPE and PS resins. Still, this impact indicator is an inadequate representation for a product's environmental performance.

With exclusion of PLA, the number of LCA studies on bio-based polymers in general and PBS and its composites in particular is limited. Nevertheless, there are no similar studies on PBS composites in general and hybrid PBS composites in particluar that quantify other potential environmental impacts than global warming potentials were found for comparison. Though most bio-based polymers LCA studies identifed the need for assessments that take into consideration broader environmental concerns, with exception of a few LCA studies that evaluate GWP and energy demand plus other impacts such as eutrophication, acidification and ozone depletion. The present study considers nine potential environmental impacts in addition to GWP and energy demand in its assessment of the production of PBS and PBS composites.

Thus, and based on all of the above, this study looks to bridge the gap in knowledge on bio-based chemicals and hybrid biodegradable plastic composites in general and PBS in particular. As well, it hopes to provide an in-depth cradle-to-gate life-cycle assessment that considers several environmental emission and resource-use categories, with the overall intention of framing and communicating a better understanding of the environmental performance of PBS, the hybrid composite under investigation.

2.2 Life Cycle Assessment

According to PRé Consultants (2010) and Helgeson and Lippiatt (2009), the multidisciplinary character of LCA tool is one of its most important characteristics and complexities. In this study, some of the multiple disciplines sciences involved are: industrial and chemical engineering, environmental management, data management, chemistry, economics, environmental science, and health science.

2.2.1 Background

The best way to understand the concept of life cycle assessment (LCA) and its importance is consider the evolution of the concept and the conditions that helped develop it. LCA was previously known by many names such as environmental profile analysis, integral environmental analysis (IEA), eco-balance, and resource and environmental profile analysis (REPA). In 1997, ISO 14040 standard “Environmental

Management -- Life Cycle Assessment -- Principles and Framework" was the first LCA standard to be published by the International Organization for Standardization (ISO), which was revised in 2009. In 1998, another LCA standard was published entitled ISO 14041 Life Cycle Assessment – Goal and Scope Definition and Inventory Analysis, which was revised in 2003. In 2000 ISO published two more LCA standards: ISO 14042 Life Cycle Assessment **--** Life Cycle Impact Assessment and ISO 14043 Life Cycle Assessment – Life Cycle Interpretation. In 2006, ISO 14044 Life Cycle Assessment – Requirements and Guidelines was introduced to replace ISO 14041, 14042 and 14043. Together the ISO 14040 series forms the basis of good LCA.

ISO defines LCA as the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO, 2006a, p. 2). The U.S. Environmental Protection Agency defines life cycle assessment as "a technique to assess the environmental aspects and potential impacts associated with a product, or service, by:

1. compiling an inventory of relevant energy and material inputs and environmental releases;
2. evaluating the potential environmental impacts associated with identified inputs and releases; and
3. Interpreting the results to help you make a more informed decision" (Scientific Applications International Corporation [SAIC], 2006, p. 2).

The Society of Environmental Toxicology and Chemistry (SETAC) define LCA in greater detail as:

> "A process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment; assess the impact of those energy and materials used and releases to the environment; and to identify and evaluate opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing, extracting and processing raw materials;

> manufacturing, transportation and distribution; use, re-use, maintenance; recycling, and final disposal." (Fava et al., 1994, p. 150)

Contrary to traditional environmental impact assessments like IEA and REAP, life cycle assessment was neither meant to measure or predict impacts nor it was intended to be absolute or precise (ISO, 1998). LCA is recognized by practitioners as the best integrated tool available to assess the potential environmental impacts of a product or a service (Reap, Roman, Duncan, & Bras, 2008). Other qualitative and analytical environmental management and assessment tools such as Material Input per Service Unit, Environmental Performance Evaluation, Environmental Accounting, and Material, Energy and Toxic-analysis are available, but not as useful as LCA (Fet, 1998). However, LCA is an imperfect method with limitations that continuously undergoes refinement (Reap, Roman, Duncan, & Bras, 2008; Nygren & Antikainen, 2010). Though, the socio-economic aspects of a product are, typically, not addressed in an LCA study, these factors can addressed qualitatively as recommended by ISO 14040. LCA has been criticized for its results could be customized as desired by practitioners (Baumann & Tillman, 2004). In LCA, it is recommended to include qualitative and quantitative data if both data are available (Hochschorner & Finnvenden, 2003).

According to ISO 14040 and 14044 standards and Consoli et al. (1993) a life cycle assessment study consists of four methodological components. The first is the goal and scope definition which includes the purpose, system boundaries and functional unit identification. The second is the life cycle inventory (LCI) analysis, which includes input and output data collection and setting the co-product procedures. This is followed by the life cycle impact assessment (LCIA), which includes the potential environmental impacts evaluation and the LCA results calculation. The fourth and final component is the life cycle interpretation that focuses on the interpretation of LCI or LCIA phases' results (ISO, 2006a; ISO, 2006b), Figure 14. Baumann and Tillman (2004) and ISO (2006a) identified two system boundaries to perform life cycle assessment studies: the cradle-to-grave and the cradle-to-gate. The difference between the two models is related to the system boundary identification (Baumann & Tillman, 2004). A cradle-to-grave LCA is defined as a full life cycle assessment of a product starting from the extraction, acquisition, cultivation and harvesting of raw materials, through to the production of a product, then to the use phase and finally to the end of life stage (ISO, 2006a).

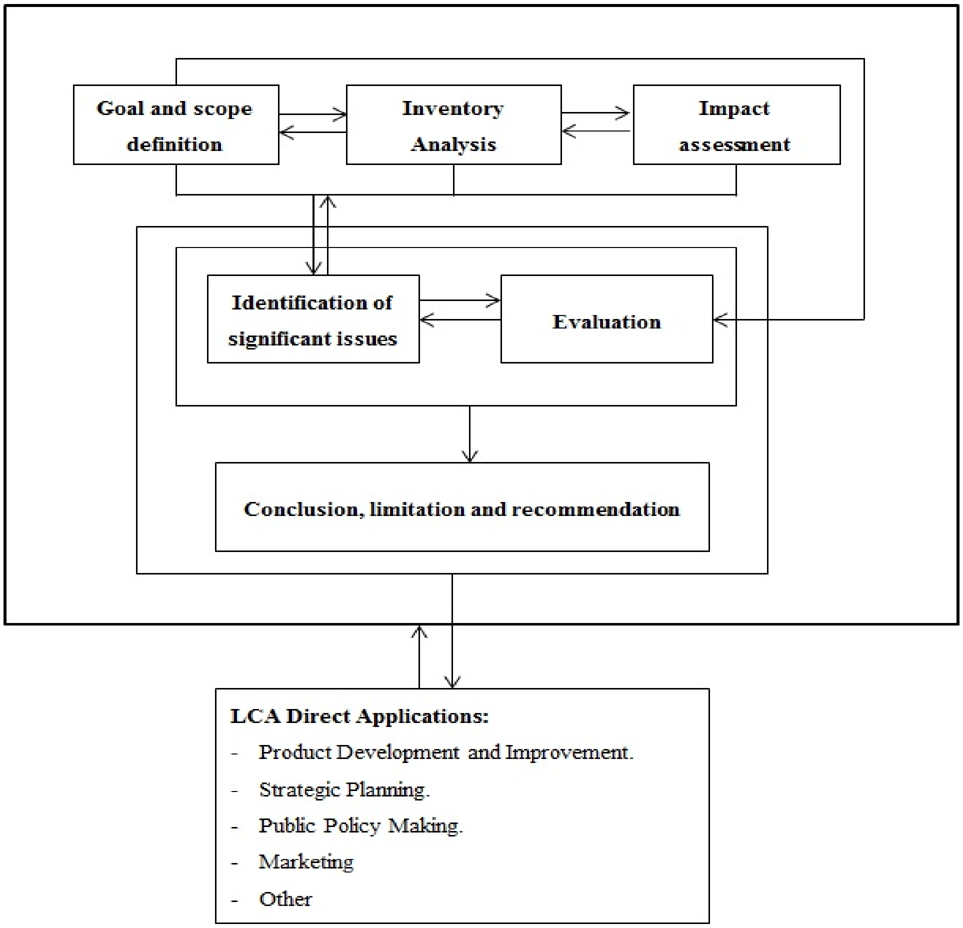

***Figure 14**: The relationship between the different stages of LCA based on ISO (2006a) and ISO (2006b).*

A cradle-to-gate LCA is a partial life cycle assessment that begins exactly like the cradle-to-grave process but ends when the final product is produced and leaves the factory gate (Hammond & Jones, 2008), Figure 15.

Cradle-to-gate LCAs are commonly used for assessing the environmental profiles of polymers and composites for their ability to provide a preliminary impression of the environmental impacts of the materials (Yu & Chen, 2008). However, the results from cradle-to-gate LCAs will not be adequate for use in comprehensive decision making processes concerning a product's environmental performance (ISO, 1998).

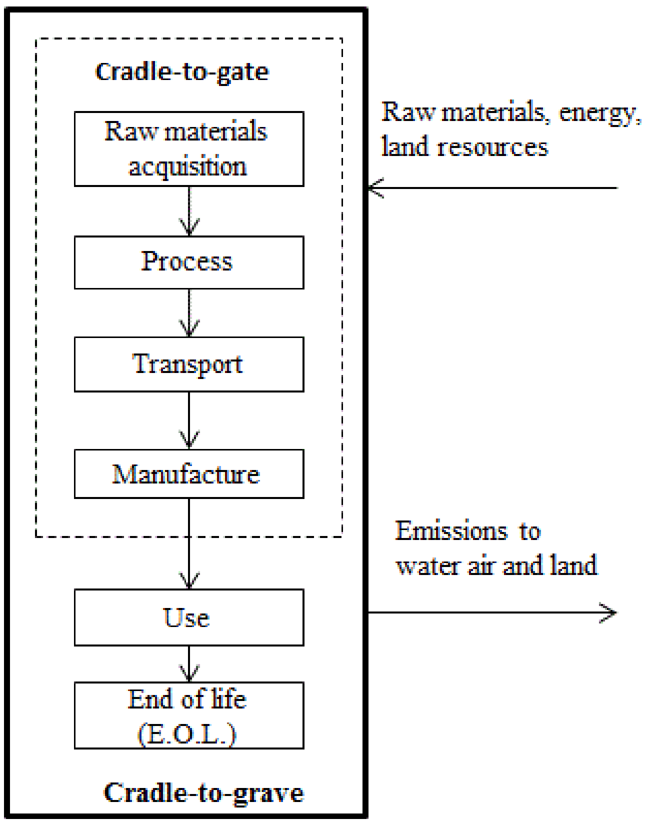

***Figure 15**: Life cycle models and system boundaries (Baumann & Tillman, 2004).*

The choice of which unit processes are included in the process system boundary is an essential step in the LCA study goal and scope. It is preferable to set a system boundary in such way that all the system inputs and outputs are elementary flow [5] and product flow [6] (ISO, 2006a; ISO, 2006b).

[5] Elementary flow: a material or energy entering or leaving the system without previous human transformation
[6] Product flow: product entering or leaving the system to another product system

ISO 14040 standard and Baumann & Tillman (2004) identified four main areas in which LCA can help with: improving products' environmental performance; policies, regulations and decision making (including design, development and purchasing of products and processes); green marketing and public relations (including eco-labeling, environmental product declaration and benchmarking); and the selection of environmental performance indicators. Wenzel (1998) and Baumann & Tillman (2004) describe LCA as a supportive tool for decision making which can benefit public policy makers, industry, environmental NGOs and consumers.

2.2.2 Co-product Problem and Allocation Procedures

One scholarly debate in LCA concerns situations when multiple products from a single process are co-produced. These process outputs are not perceived as waste but that have economic or social function. The challenge in LCA is to distribute appropriately the environmental burdens of the single process among its multiple co-products. This is important as the allocation of system impacts among different products can have a major influence on the environmental performance of a product system.

The ISO 14044 (2006) standard defines allocation as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" (p. 4). Two issues are necessary to be taken into account when allocation decisions are made in an LCA study: 1) All allocation procedures must be clearly stated and 2) for a unit process the total number of the allocated inputs and outputs must be equal before and after allocation (ISO, 2006b). The ISO 14044 standard recommends that allocation should be avoided, whenever possible, and this recommendation is followed by many LCA studies (Moussa et al., 2012; Kim and Dale 2002; Wang, Saricks, & Santini, 1999).

Alternatively, avoiding allocation can happen by dividing a unit process into multiple sub-processes and using the output data from these sub-processes. Nevertheless, this will not completely solve the allocation problem state Ekvall & Finnveden (2001); or by expanding the system which is preferable to traditional allocations (Weidema, 2000). System expansion means that the boundaries of the product system are expanded to include the use of the co-product(s) through the use of alternatives of other products that are already used in the market (ISO, 2006b; Weidema, 2000). The ISO 14044 standard suggests that if the process cannot be subdivided, co-products are required to be assigned burdens based on either physical properties (mass, energy or heating values) (Weidema, 2000; Mortimer, Elsayed, & Evans, 2007; Nuss & Gardner, 2013; Iriarte & Villalobos, 2013), or based on economic value (Jungbluth, Bauer, Dones, & Frischknecht, 2005; ISO, 2006b; Luo, van der Voet, Huppes, & Udo de Haes, 2009).

Economic-based allocation is comparable to the methodology suggested and used by Guinée, Heijungs, and Huppes (2004). Pawelzik et al. (2013) argue that when product and co-products economic values are comparable, the use of mass allocation is preferable. Energy-based allocation is a preferable option when the energy content of products and co-products is essential for the LCA goal and scope (Pawelzik et al., 2013). The choice of allocation procedures has a considerable impact on the results of an LCA study (Jungmeier, Werner, Jarnehammar, Hohenthal, & Richter, 2002; Luo, 2010; Edwards & Anex, 2009; Hetherington, McManus, & Gary, 2011; Morais, Martins, & Mata, 2010; Halleux, Lassaux, Renzoni, & Germain, 2008; Curran, 2007). LCA results can vary significantly based on the allocation method employed (Luo et al., 2009; Ekvall & Finnveden, 2001). For this reason, Luo et al. (2009) and Hetherington et al. (2011) have suggested and it is a good practice to use multiple allocation procedures in a study. It is essential to acknowledge that different studies require different allocation approaches and procedures as suggested by Tsiropoulos, Cok, & Patel (2013).

2.2.3 Life Cycle Inventory Data

The biggest challenge that faces LCA practitioners when performing an LCA study is data collection. Life cycle inventory (LCI) data are defined as “the primary inputs for conducting LCA studies” (U.S. DOE, 2009, p.1). The results of LCA studies are considered robust, defensible, and meaningful only when input data are consistent, accurate, and relevant (U.S. DOE, 2009). It is critical to differentiate between two types of data, the foreground and background, (PRé Consultants, 2010; Nebel, Alcorn, & Wittstock, 2011). PRé Consultants (2010) defines foreground data as specific data, generally empirical that are required to model a product system. In contrast, background data are generic for materials, energy, etc. Background data are generally secondary and tertiary sources and usually found in literature and databases (PRé Consultants, 2010). de Bruijn, van Duin, & Huijbregts (2002) and Nebel, Alcorn, & Wittstock (2011) define the latter similarly.

2.2.4 Assumptions and Limitations

International standards demand that assumptions in an LCA study be defined and clearly stated. In addition, the assumptions must be in accordance with the goal and scope defined in the study (ISO, 2006a). According to Wenzel et al. (2006) different assumptions will lead to different conclusions and "in general, differences in these assumptions proved to be more significant than differences in environmental impacts associated with different elements of each study" (p. 2).

The ISO 14040 standard requires that the limitations of a study be communicated in a transparent and sufficient way to demonstrate the inherent uncertainties, difficulties and trade-offs in the LCA. It is essential to understand how the accuracy, transparency, quality and credibility of an LCA study will be affected by these uncertainties and limitations (Weidema, 2000; Heijungs & Huijbregts, 2004; Guldbrandsson et al., 2011; Baker & Lepech, 2009). LCA practitioners understand these limitations and uncertainties while emphasizing the importance of reducing uncertainty where possible (Baker & Lepech, 2009). However, a systematic framework to resolve uncertainties is still unavailable (Heijungs & Huijbregts, 2004). ISO 14040 recommends that "in some cases, the goal and scope of the study may be revised due to unforeseen limitations, constraints or as a result of additional information. Such modifications should be documented" (p. 7). Limitations associated with choices that are made during the LCI phase, must be clear and communicated (ISO, 2006a). These value-choices are related to the system boundary, the inclusion and exclusion of unit processes in addition to their inputs and outputs, cut-offs criteria and data gaps. In addition, it is essential to communicate the level of data quality used in the study, in relation to uncertainties and allocation procedures (ISO, 2006a; UT Center for Clean Products, 2009). Assessing the data quality level is a limitation which needs thorough attention when performing an LCA study as it is unviable to collect site-specific or primary data for all the unit processes (ISO, 2006a; ISO, 2006b). Often, LCA studies use publicly available data, with secondary and tertiary data used to fill the data gaps, knowing the uncertainties associated with the use of unrelated temporal, geographical, and technological specific system set of data (Strongwell Corporation, 2009). To understand the potential impacts of the limitations of an LCA study, statistical uncertainty, sensitivity and scenario analyses were performed in Chapter 4, sections 4.5 as possible solutions (ISO, 2006b).

2.2.5 Life Cycle Impact Assessment (LCIA)

LCIA is “aimed at evaluating the significance of potential environmental impacts using the LCI results” (ISO, 14040, 2006a, p. 14). The United States *National Risk Management Research Laboratory* indicates that “a life cycle impact assessment attempts to establish a linkage between the product or process and its potential environmental impacts” (NRMRL, 2012, p. 46). LCIA classifies and characterizes these potential impacts in a thorough process to provide a stronger basis for products’ comparison.

LCIA involves three obligatory steps: the selection of relevant impact categories (e.g. energy demand, global warming, and acidification), establishing the connection between LCI results and each of the impact categories, and modeling the results using characterization factors (NRMRL, 2012; ISO, 2006a; ISO, 2006b). These steps may include reviewing the LCA study goal and scope to assure the objectives have been met (ISO, 2006a; ISO, 2006b). The ISO 14040 states that “the LCIA addresses only the environmental issues that are specified in the goal and scope” (ISO, 2006a, p. 15). Therefore, LCIA does not include the assessment of all environmental issues in a product system. The interrelation between the different phases of LCA as illustrated in Figure 14 needs to be taken into account when choosing LCIA impact categories (ISO, 2006b). In this study, two indicators and nine impact categories were considered.

Cumulative Energy Demand (CED) Indicator

The CED is a pre-existing midpoint method in SimaPro (System for Integrated Environmental Assessment of Products), expanded by PRé Consultants based on the method published by EcoInvent (Goedkoop, Oele, de Schryver, & Vieira, 2008), and developed initially as a response to the oil crises in the early 1970s as indicated by Boustead & Hancock (1979). Energy demand is a well-developed indicator that is widely used in LCA studies (Gironi and Piemonte, 2011; Zah et al., 2007). This indicator is not only used to identify the possibilities and significances of energy reduction (VDI, 2012) or systems' energy efficiency (Klöpffer, 1997), but could also be used as a tool to evaluate products based on energy performance (VDI, 2012; Frischknecht et al., 2007b). Modahl, Lyng, Raadal, & Askham (2012) indicate that "the CED results are heavily dependent on the LCA practitioner's knowledge of how to manipulate the pre-defined CED methods" (Abstract). For those reasons CED is criticized (Huijbregts et al., 2010). CED method uses the characterization, weighting, and single score factors, however, normalization is excluded from this method (Goedkoop et al., 2008).

Waste Heat Indicator

Goedkoop et al. (2008) described waste heat as an inventory (elementary flow) indicator that is used to provide potential insights on the energy and environmental efficiency and performance of a product system. SimaPro LCA tool records all the waste heat discharged from a product system without differentiating between the sources of waste heat (Althaus et al., 2007).

A study by the United States Department of Energy [U.S. DOE] (2004) indicated that in chemical processes, substantial amounts of energy inputs are released as waste heat emissions to air, water and soil. The U.S. DOE (2004) as cited by Bai (2010) and BCS (2010) estimated these amounts to be between 20-50% energy inputs. Barker, Turner, Napier-Moore, Clark, & Davison (2009) as cited by Bundela & Chawla (2010) identified two general potential environmental impacts of waste heat: the impacts of energy use and the several pollutants (chemicals and particles) contained in the released waste heat. The reduction of waste heat releases to the environment represents an opportunity for the reduction of energy demand and the environmental impacts associated with its use (Huijbregts et al., 2006; BCS Incorporated, 2008). Similarly, a study by Zhang & Akiyama (2009) indicated that in industry, waste heat reduction by improving systems' efficiency or heat recovery will probably reduce energy that is used for electrical, and heat generation. In addition to the proportional relationship with energy demand, waste heat (as a heat source) contributes to global warming (Zhang, Cai, & Hu, 2013). Bundela & Chawla (2010) pointed out that the greatest potential environmental impact of waste heat discharges is on the aquatic ecosystem. Huijbregts et al. (2006) and BCS Incorporated (2008) emphasized on the potential environmental benefits and energy savings associated with waste heat reduction (improve efficiency), mainly because it is an emission-free source of energy.

Other Impact Categories calculated using TRACI LCIA method

The Tool for the Reduction and Assessment of Chemical and other environmental Impacts (TRACI) was developed by the U.S. Environmental Protection Agency for the US as a LCIA tool, pollution prevention tool and set of sustainability metrics (Frischknecht et al., 2007b). A stand-alone, mid-point LCIA method, TRACI uses characterization, normalization, and weighting factors (Goedkoop et al., 2008; European Commission, 2010; Bare, Norris, Pennington, & McKone, 2003; Bare, 2011; Goedkoop et al., 2008; Frischknecht et al., 2007b) to quantify potential environmental impacts across several impact categories: ozone depletion, global warming, photochemical oxidation (smog) formation, acidification, eco-toxicity, eutrophication, human health criteria air pollutants, human health carcinogen, human health and non-carcinogen (Bare, 2011; Rossi & Margni, 2010; Goedkoop et al., 2008). TRACI does not include resource consumption in any impact categories (Frischknecht et al., 2007b), which is considered as a potential weakness in the approach compared to other LCIA methods; however, TRACI is still a valuable method since it is widely used in LCA studies in North America, the geographical location of this study intended audience.

2.2.6 Sensitivity and Scenario Analyses

Sensitivity analysis is defined as "systematic procedures for estimating the effects of the choices made regarding methods and data on the outcome of a study" (ISO, 2006b, p. 5). In a similar way, PRé Consultants (2010) defines sensitivity analysis as "a way to understand the influence of the most important assumptions that were made, in order to get a better understanding of the magnitude of the effect of the assumptions that was made" (p. 37). ISO 14044 (2006) indicates that a sensitivity analysis considers whether variation in methods and parameters causes significant changes in results. These changes are considered significant if larger than a certain percentage value that is set by LCA practitioners. The values of these changes are calculated as a percentage of the initial results of methods and data used (ISO, 2006b). Guo & Murphy (2012) indicate that a sensitivity analysis is often performed on boundary definition, allocation procedures, LCIA methods, and inventory data.

According to Björklund (2002) a "scenario analysis involves calculating different scenarios, to analyse the influence of discrete input parameters on either output parameter values or priorities" (p. 67). Raynolds, Checkel, & Fraser (1998) as cited by Björklund (2002), describe a scenario analysis as a change in LCA results due to potential future conditions based on assumptions.

One of the sensitivity and scenario analyses drawbacks is that it takes into account only one variable at a time, whereas the other variables are kept constant. The accuracy and effectiveness of each of the variables used in the analyses depend to a great extent on the reliability and availability of data, which is deemed to be insufficient in bio-based materials (Grabowski, 2013).

2.2.7 LCA and Sustainability Assessment

It is relevant to consider how LCA relates or contributes to broader sustainability assessment, which arises from concept of sustainable development defined by the Brundtland Commission's report in 1987 as "a development that meets the needs of the present without compromising the ability of future generation to meet their own needs” (p. 8). According to Gibson (2006) and Pope, Annandale, & Morrison-Saunders (2004), sustainability assessment is an integrative conceptual tool to help move toward sustainability. However, since the three dimensions of sustainability are summed-up separately, issues of incomplete perspective could result (German & Schoneveld, 2012). To complete the picture of sustainability, it is suggested that the social, economic and environmental dimensions, which are interdependent, be taken into consideration and assessed separately, yet, re-integrated under one assessment, which as noted by Gibson (2006) is a struggle. Pope, Annandale, & Morrison-Saunders (2004) and Sadler (1996) stated that such an “integrated assessment” is typically performed to minimize uncertainties, or to fulfil the “triple bottom line” objectives, while equally weighting the impacts of the three dimensions in the decision-making process. However, in comparison to environmental and economic sustainability, social sustainability perceives less attention (Bell, 2000 as cited by Molnar & Morgan, 2001). Recently, the interest in social sustainability started to gain more recognition and legitimacy (German & Schoneveld, 2012; Benoît & Vickery-Niederman, 2010), yet, and according to Foot & Ross (2004) there is no definition of social sustainability that is precise, complete, functional and agreed upon. Hence, the principles of social sustainability were made less sophisticated in order to be comfortably assess such a complex issue (German & Schoneveld, 2012). Gibson (2001), Devuyst (1999) and Sadler (1999) suggested that environmental assessment could be used as a building block for sustainability by including other dimensions (e.g., social) along with the environmental dimension.

A study by Pope, Annandale, & Morrison-Saunders (2004) suggested that the theory of sustainability assessment is an evolution of environmental impact assessment, therefore, it is considered to be the “next generation” of environmental assessment (Sadler, 1999). As the latter, relates to environmental LCA, there is connection to other types of LCA methods with different scopes, for example, social LCA (SLCA) and economic LCA or life cycle cost (LCC) (Zamagni et al., 2009). Klöpffer (2008) suggested that a “life cycle sustainability assessment” (LCSA) could be done by conducting an LCA, an LCC and an SLCA consecutivelly. According to the UNEP/SETAC Life Cycle Initiative (2011), LCSA refers “to the evaluation of all environmental, social and economic negative impacts and benefits in decision-making processes towards more sustainable products throughout their life cycle” (p. 3). Practically, and according to Hunkeler, Lichtenvort, & Rebitzer (2008), combining the LCA and LCC tools would raise issues of conflict in methodological choices (e.g., co-product treatment and system boundary definition).

3. Methodology and Data

The current study follows the methodological and structural framework provided by the ISO 14040 and 14044 international standards on life cycle assessment. These standards are intended to provide consistent, credible, transparent, and comparable results and are broadly accepted by stakeholders (Finkbeiner, Inaba, Tan, Christiansen, & Klüppel, 2006). As introduced in Section 2.2.1, the required methodological steps are addressed, and discussed in order here.

3.1 Goal and Scope

In this section, the goal and scope of this study are identified and stated in relation to the intended use of this LCA study.

3.1.1 Goal

The goal and objectives of this thesis were identified and described in Chapter 1. Results will be shared with research, industrial and government partners, to help facilitate the decision-making as it relates to the aspects and potential environmental impacts of the production of novel industrial bio-based products.

3.1.2 Scope

The LCA study of SG / PBS composite pellets was conducted for a cradle-to-gate production system described by seven major unit processes (Figure 16). These seven unit processes are described in details in Section 3.2.2.

The cradle-to-gate system boundary ends when the hybrid composite is produced into a pellet form, which is then suitable for subsequent processing, for example into auto parts. The product system does not include the shipping of the hybrid composite pellets or further downstream activities like manufacturing of auto parts, car assembly, car use, and end-of-life.

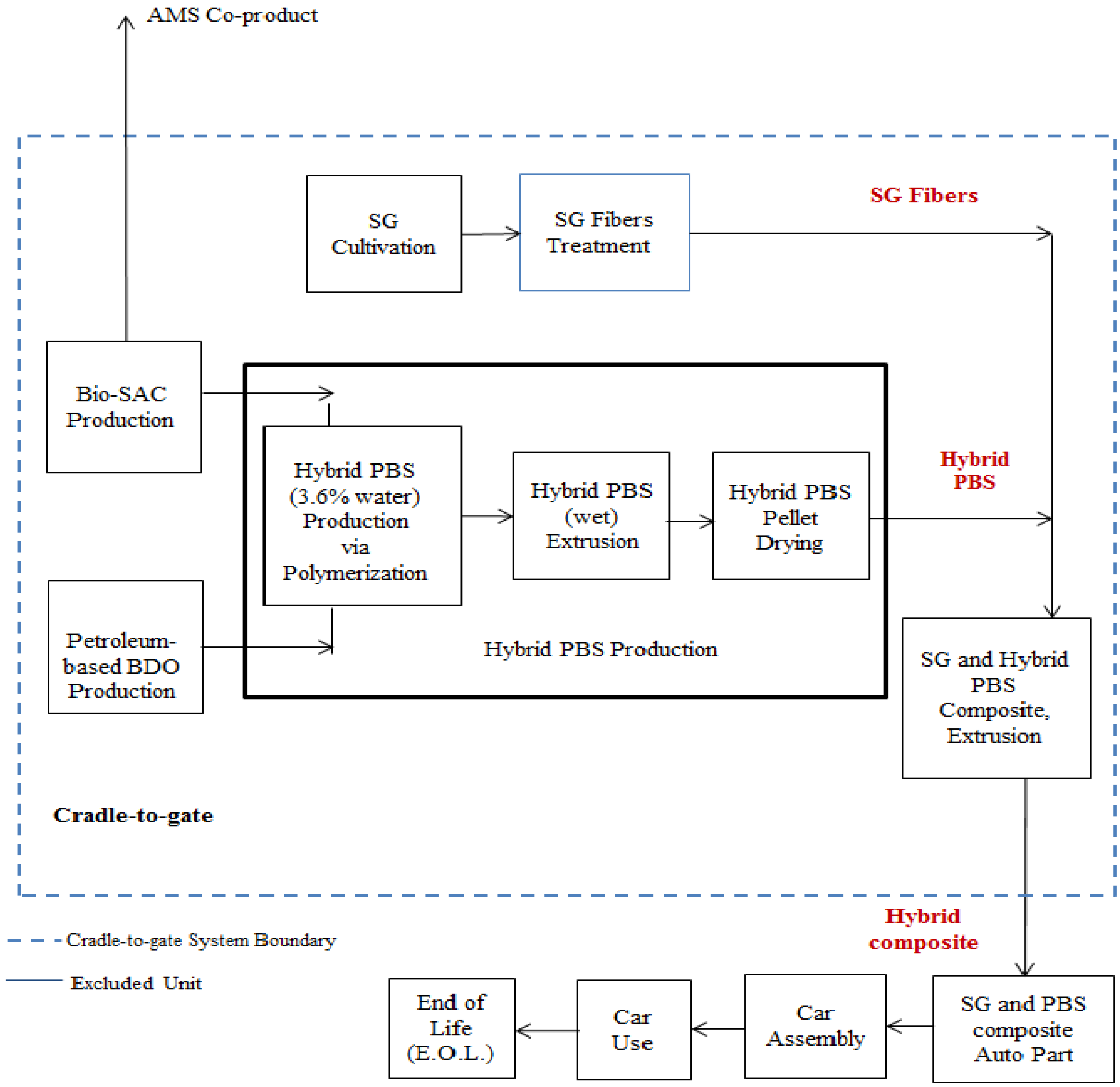

Figure 16*: System boundary for the production of the hybrid composite used in the auto part.*

Concerning foreground data, the boundary of this study includes the operation of capital equipment, but not its production, following established practice (PRé Consultants, 2010). This assumption is justified given that the environmental impacts of capital equipment are negligible when used for bulk production systems (Allen, 2002), and given the limited reliable information on the environmental impacts associated with the production of different capital equipment in the product system (Gorrée, Guinée, Huppes, & van

Oers, 2002). The present study acknowledges the fact that exclusion of the production of some of the capital equipment may result in an error in the results (PRé Consultants, 2010). For EcoInvent background data the profile of capital equipment is included (PRé Consultants, 2010).

In order to accomplish objective #2 of this study, which is to compare the environmental performance of the hybrid composite to a baseline composite, glass fibre (GF) reinforced polypropylene (PP) composite in pellet form was chosen as the baseline. PP is a petroleum-based polymer that is widely used in the automotive industry for its suitable physical and mechanical properties (Borealis AG, 2010a). GF is an inorganic and nonmetallic reinforcement fibre that is widely used in the automotive industry given its good mechanical and thermal properties. Many industrial sectors including the automotive industry are attracted to the light-weight, strength, and non-corrosive properties of PP-GF composites (ASD Reports, 2013). The baseline composite was modeled similarly to the hybrid composite using a cradle-to-gate system boundary with three unit processes: glass fibre production, polypropylene production, and baseline composite production via extrusion (Figure 17).

3.1.3 The Function and the Functional Unit

This study assumes that both the hybrid and baseline composites can be used to manufacture an automotive part, represented here as a hypothetical automotive battery tray. The hypothetical part is assumed to be designed and produced in compliance with technical specifications and performance requirements that would be typical of an original equipment manufacturer (OEM). For example, this would include the volume (dimensions and thicknesses), parameters of scratch resistant, low gloss, low scrap, good processability, good impact resistance, low thermal expansion rate, and no emissions, fogging or odour when processed (Borealis AG, 2010b). The functional unit (FU) is one automotive battery tray made from composite materials.

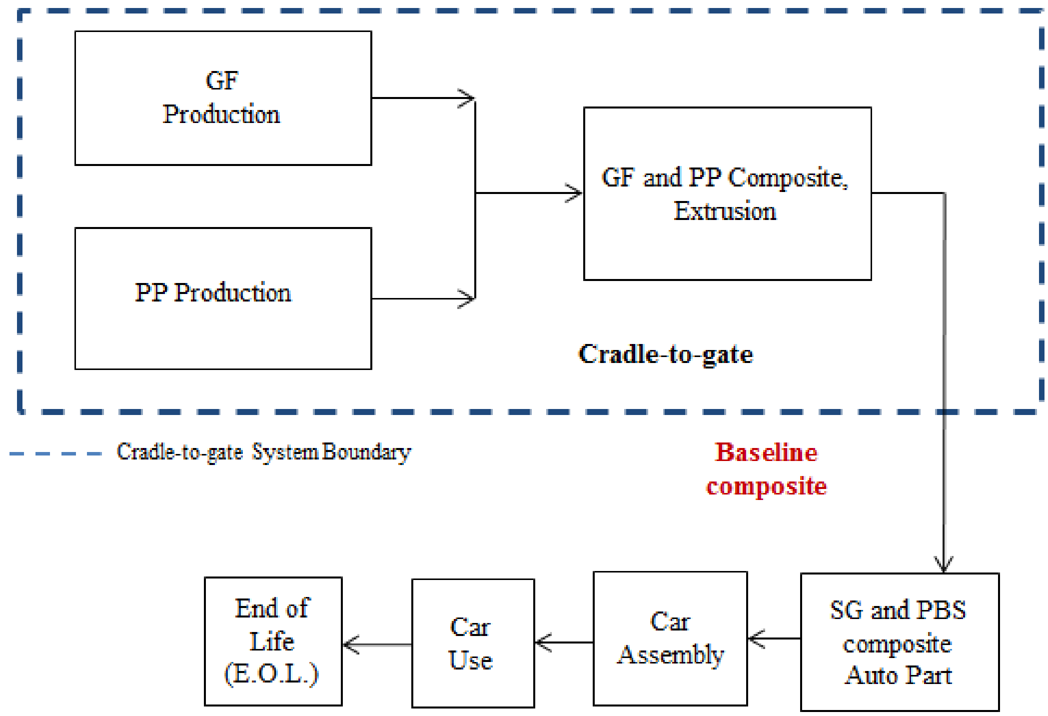

Figure 17*: System boundary for the production of the baseline composite material for use in an auto part*

3.1.4 Reference Flows

A reference flow measures "the outputs from processes in a given product system required to fulfill the function expressed by the functional unit" (ISO, 2006b, p.5). In this study, the automotive battery tray can be produced from either the hybrid composite or the baseline composite. The two produced battery trays are equivalent in function and have the same volume and dimensions. A patent by Mohanty et al. (2010), a research partner, used in this study describes and confirms the substitutability of conventional composite with the hybrid material. The patent states that the mechanical and physical performance of materials made from either composite is essentially equal; this was not questioned within the current study. On this basis it is reasonable and fair to compare the composite materials in this manner. Table 1 describes the average physical and mechanical properties of the hybrid and baseline composite.

Table 1: *Physical and mechanical properties of the two composite materials*

Material Property		Hybrid Composite (Mohanty, Misra, & Sahoo, 2010)	Baseline Composite (MatWeb, 2012; Anderson, 2011)
Material Composition		SG-PBS (50%:50%)	GF-PP (30%:70%)
Density	kg/m^3	1,063	1,150
Tensile Strength	MPa	34±0.2	77.1
Tensile Modulus	GPa	3.1±0.04	6.25
Flexural Strength	MPa	46±0.6	110
Flexural Modulus	GPa	36±0.11	5.17
Izod Impact Strength	J/m	25±2.1	109
Heat Deflection Temperature	^oC	106.4	123

The baseline composite is slightly denser than the hybrid composite (Table 1); hence there is a slight difference in mass. The reference flow for the hybrid composite battery tray (50% SG reinforced hybrid PBS composite) is 1 kg hybrid composite. The reference flow for the conventional composite (30% GF reinforced PP composite) is 1.1 kg baseline composite.

3.1.5 Co-product Treatment

Several co-products are encountered in the product systems analyzed. In the production of the hybrid composite, ammonium sulfate (AMS) is produced during the production of bio-SAC. The study applied a system expansion approach to this process (as described in Section 2.2.2). This was based on information received from Myriant Corporation on its operation in Louisiana, USA that all the AMS produced is sold in the region as fertilizer for agricultural use. The AMS market was further examined and it was determined that the co-product AMS fertilizer is functionally equivalent to petrochemically-produced AMS and directly displaces it in the regional market (Myriant Corporation, 2011). The USA AMS market is more than 3.3 million tonnes per year (Maxwell, 2002), and the AMS co-product is a small fraction of it, therefore the operation of Myriant does not affect market prices. Under these circumstances the use of the system expansion approach is justified.

Other co-products are apparent in background processes used in this study (Frischknecht et al., 2007b). On examination of the EcoInvent data, it is clear that system expansion approach and other allocation procedures were used to address the co-products problem (Frischknecht et al., 2007b). These other co-products, approaches and procedures were not addressed or taken into consideration in this study, and are assumed to have minor influence on the LCA results.

3.1.6 Limitations

1. Like most LCA studies there are weaknesses in specific model assumptions and data quality. The study is limited by the use of published datasets primarily based on European production technologies, practices, conditions and assumptions. These data have technological and resource limitations when used in a different geographical context.
2. The majority of data used in this study are from secondary or tertiary sources, and may be associated with high uncertainty, except for bio-SAC data that are empirical. To mitigate the high uncertainty concern, an assessment of uncertainty was calculated and uncertainty values are assigned by SimaPro pedigree uncertainty calculations.
3. Transportation between the different production facilities (switchgrass, bio-succinic acid, 1, 4-butanediol, poly butylene succinate, and the composite) was not considered.
4. Switchgrass and soil carbon sequestration in Southern Ontario were not considered as a conservative assumption due to the unavailability of data specific to Ontario (as explained in Section 3.2.2.1). Also, after farming, SG is subjected to thermal, mechanical and chemical

treatments of fibres in preparation for use as a reinforcing material in the hybrid composite. These activities were not included (see Figure 16), as it was assumed that the potential environmental impacts of these processes are negligible in the whole LCA.

5. The study uses a limited set of LCIA categories to assess the environmental performance of the product system. Eleven (11) indicators were used; however, several other category areas, such as land use, water use and resource depletion are not considered.

3.2 Life Cycle Inventory (LCI)

This section discusses the collection of input and output data for the product system under study. A combination of pre-existing and newly developed unit process data were utilized in the examination of the hybrid composite system. Secondary and tertiary data used for the modelling of all unit processes, and were obtained from either the EcoInvent database or the U.S. Life Cycle Inventory (U.S. LCI) database. For the production of bio-SAC empirical data were used, as provided by Myriant Corporation. Modifications were made to electricity and heat profiles, as described below (section 3.2.3). Data sources for the baseline composite system were entirely based on tertiary data sources.

- EcoInvent is a life cycle inventory (LCI) database from the Swiss Centre for Life Cycle Inventories (the EcoInvent Centre). The EcoInvent database covers environmental activities at system and unit process levels, and is intended to be used as background data. The quality of EcoInvent data is supported and assured by the EcoInvent Centre validation and review system (Weidema et al., 2013).
- The US LCI database provides reliable and transparent information on energy and material flows and is compatible with other LCI databases. Flows are associated with specific unit processes used for material production and services in the USA (Deru, 2009).

The geographical locations of the different production facilities in the product system are discussed in Section 3.2.2. The transportation activities were assumed negligible given experience by others, suggesting that transportation aspects are minor to the overall LCA study for cases similar to the current study (Gironi and Piemonte, 2011; Franklin Associates, 2010; Khoo et al., 2010; and Zah et al., 2007). The potential environmental impacts of packaging materials used for packaging of the bio-SAC, hybrid PBS and the hybrid composite were not included and were assumed to be negligible.

3.2.1 LCA Tool

All unit processes were modelled using SimaPro software version 7.3.0. First released in 1990, SimaPro was created by PRé Consultants in the Netherlands and is widely used in more than 80 countries (Earth Shift, 2012; Boureima, Sergeant, & Wynen, 2007; Glišović, 2000). SimaPro is used to assess and analyze the environmental performance of a product or a service, and designed based on the framework of the ISO 14040 and 14044 standards (Earth Shift, 2012).

SimaPro is widely used by industry and academia, has an intuitive and user-friendly interface, is flexible in implementing LCA, contains a number of good databases (e.g., system and unit processes, and impact assessment methods) and datasets (e.g., EcoInvent and US LCI). SimaPro enables users to adjust and modify pre-existing LCIA methods, create new methods, adjust analyzing options, and compare two or more different products and processes (Boureima, Sergeant, & Wynen, 2007; Glišović, 2000; Menke, Davis, & Vigon, 1996). Additionally, the results of Kalita (2012), which this current study relied on, contained information on switchgrass in SimaPro files.

A study by Zamagni, et al. (2009) discussed the limitations of LCA softwares noting that "main LCA software are not sufficiently capable of facilitating broadening and deepening LCA" (p. 11). In SimaPro, practitioners have little control over the availability and quality of background data (e.g., catalysts used in chemical processes) and the methodological choices related to co-product treatment in background data. Another limitation is how LCI data are assigned to the different indicators and impact categories, especially if one aspect has the potential to affect more than one impact category or indicator. For example, nitrous oxide has the potential to affect both ozone depletion and GWP. Additionally, in SimaPro the uncertainties of LCIA methods are not calculated as part of the total uncertainty analyses, as only LCI uncertainties are calculated by the Monte Carlo Simulation (Andrae, 2010).

3.2.2 Unit Processes

3.2.2.1 Switchgrass Cultivation

The modeling of switchgrass cultivation was based on a field data and research by Kalita (2012) from the University of Guelph. The work by Kalita was supported by the same partners also supporting this current study. This allowed for direct access to foreground data. The SG data time-line is from 2006 to 2011 and the technology used in switchgrass cultivation represents farm practices in current use in Ontario. The study by Kalita (2012) indicated areas of uncertainty in regards to switchgrass yield data.

Kalita (2012) also employed SimaPro and used a combination of primary and secondary data (Table 2). Empirical data were obtained from one farm in Southern Ontario that provided well documented information on switchgrass cultivation practices. These data represent all input materials used on-farm in the cultivation and harvesting of switchgrass, including diesel, seeds, herbicides, and nitrogen-based fertilizer used to produce one tonne of switchgrass. Secondary data were obtained from literature and reports.

The use of synthetic nitrogen-based fertilizer in SG cultivation results in emissions of nitrous oxide (N_2O) and ammonia (NH_3). In agricultural LCA studies, N_2O is often a significant source of GHGs. In the current study, direct and indirect N_2O emissions from ammonium nitrate fertilizer were calculated based on the *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (IPCC, 2006) (Table 2). Kalita (2012) used an emission factor for N_2O direct emissions calculated for Ontario as 0.017 kg N_2O-N per kg of nitrogen based on a study by Rochette et al. (2008). In the current study, NH_3 emissions were calculated the same as by Kalita (2012) based on a study by Sheppard et al. (2010) as 6% of the nitrogen fertilizer used as in Table 2.

***Table 2**: Process input and outputs for the cultivation of 1 kg of Ontario switchgrass*

Unit Process and Location	Flow Type	Input/output	Unit	Value	Data Source
Switchgrass - Cultivation (1.0 kg). Ontario, Canada	Input Materials	Diesel Fuel	L	3.34E-03	Kalita (2012)
		Switchgrass Seed	kg	1.10E-04	
		Glyphosate - pesticide		1.30E-05	
		2,4 -dichlorophenoxyacetic Acid - pesticide / herbicide		2.20E-05	
		Atrazine - herbicide		2.40E-05	
		Ammonium Nitrate (35% Nitrogen) - fertilizer		15.55E-03	
	Airborne Emissions	Nitrous Oxide (N_2O) Direct and Indirect Emissions	kg	1.94E-04	IPCC (2006)
		Ammonia (NH_3)	kg	3.26E-1	Kalita (2012) based on Sheppard et al. (2010)

Background data used by Kalita are from both secondary and tertiary sources, and represent the production of input materials used on-farm. These background data were based on previous studies, reports, and the US-EI database adjusted for use in a Canadian context (Kalita, 2012). 0.5 kg of switchgrass was used for the production of 1 kg of the hybrid composite.

3.2.2.2 Bio-Succinic Acid Production

Primary data on the production of bio-SAC were provided by Myriant Corporation. This firm is an industrial biotechnology company based in Quincy, Massachusetts with a stated mission to become "a low-cost producer of high-value biochemicals” from renewable and non-food crop feedstocks (Myriant Corporation, 2011, p.1). Much of their business plans target bio-based products that are functionally similar to petroleum counterparts. Myriant manufactures a variety of chemical building blocks using patented technologies that are based on a single-step anaerobic fermentation process using biocatalysts that are capable of consuming sugar from different sources. Their aim is higher productivity and yield in comparison with other known biotechnologies (Myriant Corporation, 2011). In 2013, Myriant started the commercial production of bio-SAC at a new plant in Louisiana, USA. The plant is the first of its kind in North America, with an annual nominal production capacity of 13 million kg. As a part of the collaboration with the University of Waterloo, Myriant provided data (Table 3) that were collected by Myriant from their operations at a pilot-scale plant in 2012 and later validated by the new plant operations at Louisiana. These data were use in the LCA model in this study. Ammonium sulfate (AMS) is a significant co-product produced during the production of bio-SAC at the Myriant Corporation’s Louisiana plant.

Background processes on the bio-SAC are based on the EcoInvent and US LCI databases. Electricity mix used was adjusted to fit the geographical location in Louisiana (see below Table 9, Section 3.2.3). Sorghum is used by Myriant Corporation as a feedstock for the production of bio-SAC because it is a non-food crop in the USA (Myriant Corporation, 2011). Moreover, it is locally available and has low handling costs. However, the current study used Chinese data for the unit process for sorghum cultivation as there is a lack of data on USA production. The sorghum cultivation unit process includes the use of diesel, machines, fertilizers, and pesticides. In addition to sorghum, dextrose is used in a small amount for the production of bio-SAC. The dextrose GHG emissions value was assumed to be 0.62 kg CO2-eq per kg of dextrose based on a study by Tsiropoulos et al. (2013). 0.33 kg of bio-SAC is required for the production of 1 kg of the hybrid composite.

__Table 3:__ Process input and outputs for the production of 1 kg bio-SAC

Unit Process and Location	Flow Type	Input/output	Unit	Value	Data Source
This Table Intentionally Left Blank. Data Protected by Non-disclosure Agreement					

3.2.2.3 1, 4-Butanediol Production

Data on the production of 1, 4-butanediol (BDO) are from tertiary sources, based on an EcoInvent pre-existing process in SimaPro for Europe from 2001. This pre-existing process is the only available BDO production process in SimaPro. For this reason it was used in this study and altered to fit the Japanese production context in regards to electricity and heat (Table 4). Additionally, it was assumed that the technology used for the production of BDO in Japan is similar to the original BDO production process from EcoInvent available in SimaPro. The technology used for the production of BDO is based on a one-step continuous hydrogenation of 2-butyne-1, 4-diol from acetylene over modified nickel catalysts (Sutter, 2007). 0.27 kg of petroleum-based BDO is required for the production of 1 kg of the hybrid composite.

3.2.2.4 Hybrid PBS

The hybrid PBS production was modeled in stages as three unit processes: hybrid PBS (3.6% water) production via polymerization, extrusion of hybrid PBS (wet) and drying of hybrid PBS pellet. Data for the modeling of the hybrid PBS were obtained and calculated (Table 5) from patents that represent a real production facility by Showa Denko of Japan, one of the biggest producers of PBS (Taka, Yasukawa, & Takiyama, 1994; Takiyama, Fujimaki, Seki, Hokari, & Hatano, 1994). The technologies described in these patents are the only publicly available technologies. The three unit processes were altered to fit the Japanese production context in regards to electricity and heat.

Polymerization

Data for the modeling of the hybrid PBS (3.6% water) production via polymerization were obtained and collected from patents (see Table 5). The production yield of the hybrid PBS polymerization process was assumed to be 100%, based on patents by Jentsch, Martin, & Zirngiebl (1997), Fujimaki et al. (1994), Imaizumi et al. (1998), and Takiyama et al. (1995).

__Table 4:__ Process input and outputs for the production of 1 kg petroleum-based BDO

Unit Process and Location	Flow Type	Input/output	Unit	Value	Data Source
Petroleum-based BDO Production (1.0 kg). Japan.	Input Materials and Energy	Acetylene	kg	3.20E-01	EcoInvent data. Electricity based on Vivoda (2011) and heat based on Althaus et al. (2007). Both adjusted for Japanese context.
		Formaldehyde		7.33E-01	
		Water, Cooling	m^3	4.90E-01	
		Water, Deionized	kg	1.00E-00	
		Liquid Nitrogen		3.12E-02	
		Liquid Hydrogen	kg	5.40E-02	
		Sodium Hydroxide 50% in Water		3.00E-03	
		Copper - Catalyst		2.04E-04	
		Nickel 99.5% - Catalyst		2.14E-04	
		Electricity Mix, Japan	kWh	0.18	
		Heat, Japan	MJ	24.75	
	Airborne Emissions	Ethane	kg	4.61E-05	
		Waste, Heat	MJ	6.48E-01	
		Formaldehyde	kg	5.70E-04	
		1-butanol		3.10E-05	
		NMVOC		1.29E-03	
	Waste Materials	Copper Catalyst Waste		2.26E-6	
		Nickel Catalyst Waste		1.11E-08	
	Waterborne Emissions	Formaldehyde	kg	2.98E-05	
		Copper ion		1.34E-06	
		Nickel Ion		7.70E-7	
		1-butanol	kg	1.10E-05	
		BOD		5.78E-05	
		COD		5.78E-05	
		DOC		1.91E-05	
		TOC		1.91E-05	

The unit processes were altered to fit the Japanese production context in regards to electricity and heat consumed. 0.55 kg of hybrid PBS (3.6% water) is required for the production of 1 kg of the hybrid composite.

Extrusion

The extrusion of hybrid PBS (wet) into pellets was modeled based on a pre-existing process from EcoInvent in SimaPro, verified by Ecoinvent in 2007. The extrusion unit process was modified to fit the Japanese production context in regards to electricity use. The fractions of the extruded plastic that are lost during the extrusion process were modeled as waste disposal to municipal incineration (see Table 5). 0.50 kg of hybrid PBS (wet) is required for the production of 1 kg of the hybrid composite.

Drying

The drying unit process was modelled simply as electricity consumption. This accounts for the fact that the process is essentially an electric oven. The consumption value was calculated for 1 kg of the hybrid PBS dry pellet based on a 0.35kW power source for 0.02 hour contact time and assumed an emission for drying of 0.041 kg of water vapour to air. Vapor emissions were calculated from Showa Denko of Japan’s process mass balance (Taka et al., 1994) (see Table 5). 0.50 kg of hybrid PBS pellets (dry) is required for the production of 1 kg of the hybrid composite.

3.2.2.5 Composites Extrusion

The study assumes that the extrusion technology was used for the production of both the baseline and hybrid composites (Gardner & Murdock, 2002; Zou, 2010; Van Den Oever, Elbersen, Keijsers, Gosselink, & De Klerk-Engels, 2003). The extrusion process was based on a pre-existing process from EcoInvent in SimaPro, verified in 2007 by EcoInvent. The extrusion unit process represents a small number of European and Swiss production conditions (Hischier, 2007) that were altered in this study to fit the Japanese production context in regards to electricity for the hybrid composite and the baseline composite.

The fraction of the extruded plastic that is lost during the extrusion process (0.4%) was modeled as waste disposal to municipal incineration.

Table 5: *Process input and outputs for the production of 1 kg hybrid PBS*

<table>
<tr><th>Unit Process and Location</th><th>Flow Type</th><th>Input/output</th><th>Unit</th><th>Value</th><th>Data Source</th></tr>
<tr><td rowspan="13">Hybrid PBS (3.6% Water) Production - Polymerization (1.0 kg) Japan.</td><td>Avoided Product</td><td>BDO (unreacted)</td><td rowspan="5">kg</td><td>3.50E-02</td><td rowspan="5">Taka et al. (1994)</td></tr>
<tr><td rowspan="6">Input Materials and Energy</td><td>Titanium in Ground</td><td>1.60E-05</td></tr>
<tr><td>Petroleum-based BDO</td><td>5.31E-01</td></tr>
<tr><td>Bio-SAC (99.5%)</td><td>6.50E-01</td></tr>
<tr><td>Liquid Nitrogen</td><td>1.16E-01</td></tr>
<tr><td>Electricity, Japan Mix</td><td>kWh</td><td>1.37E-02</td><td>Vivoda (2011).</td></tr>
<tr><td>Heat, Japan Mix</td><td>MJ</td><td>1.82E-01</td><td>Althaus et al. (2007).</td></tr>
<tr><td rowspan="4">Airborne Emissions</td><td>Heat, Waste</td><td>MJ</td><td>9.31E-02</td><td rowspan="4">Taka et al. (1994)</td></tr>
<tr><td>Tetrahydrofuran</td><td rowspan="4">kg</td><td>1.16E-01</td></tr>
<tr><td>Nitrogen</td><td>3.97E-03</td></tr>
<tr><td>Water Vapour</td><td>1.99E-01</td></tr>
<tr><td>Final Waste</td><td>Catalyst</td><td>9.90E-05</td><td>Titanium Tetra Isopropoxide</td></tr>
<tr><td colspan="5"></td></tr>
<tr><td rowspan="2">Hybrid PBS (Wet) – Extrusion (1.0 kg). Japan.</td><td>Input Materials and Energy</td><td>Hybrid PBS (3.6% water)</td><td>kg</td><td>1.10E-00</td><td>Taka et al. (1994)</td></tr>
<tr><td>Waste Treatment</td><td>Disposal Plastics Mixture, to Municipal Incineration</td><td>kg</td><td>6.20E-02</td><td>Hischier (2007)</td></tr>
<tr><td rowspan="3">Hybrid PBS Pellets - Drying (1.0 kg). Japan</td><td rowspan="2">Input Materials and Energy</td><td>Hybrid PBS (wet)</td><td>kg</td><td>1.04E-00</td><td rowspan="3">Taka et al. (1994)</td></tr>
<tr><td>Electricity, Japan Mix</td><td>kWh</td><td>0.70E-02</td></tr>
<tr><td>Airborne Emissions</td><td>Water Vapour</td><td>kg</td><td>4.00E-02</td></tr>
</table>

Data on the production of the two composites were based on secondary data obtained from the patent by Mohanty et al. (2010) (Table 6 and Table 7). The production processes for the two composites via extrusion were created as unit processes. Background processes for both composites are from EcoInvent.

Table 6: *Process input and outputs for the production of 1 kg of hybrid composite via extrusion*

Unit Process and Location	Flow Type	Input/output	Unit	Value	Data Source
Hybrid Composite Production - Extrusion (1.0 kg). Japan	Input Materials	Hybrid PBS Pellets	kg	5.02E-01	Mohanty et al. (2010)
		Switchgrass Fibres		5.02E-01	Kalita (2012)
	Waste Treatment	Disposal Plastics Mixture, to Municipal incineration		0.40E-02	Hischier (2007)

Table 7: *Process input and outputs for the production of 1 kg of baseline composite via extrusion*

Unit Process and Location	Flow Type	Input/output	Unit	Value	Data Source
Baseline Composite Production – Extrusion (1.0 kg). Japan.	Input Materials	Polypropylene granulate	kg	7.03E-01	Mohanty et al. (2010)
		Glass Fibres		3.01E-01	
	Waste Treatment	Disposal Plastics Mixture, to Municipal incineration		0.40E-02	

3.2.2.6 Polypropylene Production

Plastics used in packaging, consumer goods, and construction have been the subject of LCA studies for decades. These studies were supported and influenced by industry associations (e.g., *Association of Plastics Manufacturers in Europe* [APME]), and governmental departments (e.g., US EPA). Two major LCI databases for plastics have been developed by industry associations, one in Europe and one in the USA. This study used EU data from the Eco-profile of the European plastics industry (PlasticsEurope) because the US LCI data are incomplete and missing information on uncertainty and data quality (Suh, Leighton, Tomer, & Chen, 2013).

Polypropylene (PP) is a bulk commodity plastic that is produced by the polymerization of propylene. Propylene is produced from naphtha through steam cracking (Boustead, 2005). In SimaPro, EcoInvent provides data on the production of PP based on the average of 28 European production facilities from PlasticsEurope. These datasets represent cradle-to-gate system boundary that include all unit processes from the extraction of raw materials until the production of the PP (Hischier, 2007). This unit process was altered in this study to fit the Japanese production context in regards to electricity and heat.

3.2.2.7 Glass Fibre Production

Glass fibre (GF) is a non-metallic mineral-based reinforcing material that is widely used for automotive applications. GF is produced via the handling, melting and refining of raw materials (silica, soda ash, limestone, and cullet); the formation of glass fibres; and the finishing process. Sand the primary material for the production of GF is melted along with other chemicals in high-temperature furnaces resulting in a consistent melt (Kellenberger et al., 2007). The melted glass is formed into fibres and then treated chemically with water-soluble sizing and / or coupling agents (U.S. EPA, 1995).

This study used inventory data for GF based on reports for the European Glass Manufacturing Industry.

3.2.3 Auxiliary Unit Processes

3.2.3.1 Electricity Generation

Electrical energy underlies activities across almost all industrial processes. The choice of electricity generation sources, its emissions, and environmental profiles, are often of great impact on the results of an LCA study.

Data on electricity in this study were specifically modelled to fit the geographical locations of the major unit processes for the production of the hybrid composite. Two electricity production profiles were developed, one for Japan and the other for the south eastern USA, specifically the region defined by the

SERC (South Eastern Reliability Council). SERC grid coverage includes Louisiana where the Myriant bio-SAC production facility is located. The SERC electricity mix was based on data from the Form EIA-923, "Power Plant Operations Report" (U.S. Energy Information Administration 2012). These data are observed and collected from a model-based sample of plants by the U.S Energy Information Administration (U.S EIA) on a monthly basis (U.S. EIA, 2012).

Electricity mixes were altered for all unit processes in the system boundaries (see Figure 16, Page 51) except for the cultivation of switchgrass and sorghum, and for the production of bio-SAC. The electricity used in the majority of unit processes were based on European electricity mixes, while in this study these were altered to be geographically located in Japan. A Japanese electricity mix based on 2009 generation by fuel type was used (Vivoda, 2011) (Table 8). Nevertheless, the emission factors associated with the Japanese electricity mix were still based on European averages. Electricity generated from coal and oil power plants were based on Belgian and French averages respectively, while electricity generated from nuclear power plants was based on Swiss averages. Based on economic trade statistics[7], the electricity distribution and transmission losses in 2009 were calculated to be 4.85% of electricity generation output. This percentage represents the losses that occur during distribution and transmission between the source and the consumer.

Table 8*: Japanese 2009 electricity generation mix by fuel type (Vivoda, 2011)*

Natural Gas	Oil	Hard Coal	Hydro	Nuclear	Wind Power	Solar Power
29.3%	29.3%	24.9%	8.3%	7.1%	0.84%	0.26%

The electricity mix used for the modeling of bio-SAC unit process was based on 2011 fuel mixes (Table 9) and emission factors that are representative of SERC electricity generation.

[7] www.tradingeconomics.com

Table 9*: SERC 2011 electricity generation mix by fuel type (U.S. EIA, 2012)*

Hard Coal	Bituminious Coal	Lignite Coal	Natural Gas	Nuclear	Hydro	Wood	Others
32.7%	16.7%	0.2%	20.6%	23.8%	2.9%	1.4%	1.7%

3.2.3.2 Heat Generation for Chemical Processes

Similar to electricity, heat is utilized in almost all industrial processes, and has a great influence on LCA study results. In this study, all unit processes use heat to help carry out the processes, except for the cultivation of sorghum and switchgrass. For unit processes that are geographically located in Japan there are no available data on the average energy mixes used to generate heat for the chemical and plastic industry in SimaPro. The study created a new unit process for heat based on the energy mix provided by EcoInvent (Althaus et al., 2007). It represents the average technology that is used to generate heat energy in the European chemical and plastics industry (Table 10). To generate 1 MJ of heat 0.3636 kg of cold water is required. However, the amount of input water was not taken into account because steam is normally used in a closed loop system. It is assumed that the technologies used to generate heat in Japan are similar to the European ones, with one exception, which is the electricity mix used in this case is the Japanese electricity mix (see Table 8).

Table 10*: Japan energy mix used for generation 1 MJ of heat (Althaus et al., 2007)*

Electricity	Coal	Fuel Oil	Natural Gas
0.008 kWh	0.193 MJ	0.284 MJ	0.800 MJ

3.3 LCIA Impact Categories

Pre-existing mid-point LCIA methods were used in this study to convert the input and output data into environmental performance indicators and impact categories for the hybrid and baseline composites. The methods used were: cumulative energy demand (CED), waste heat calculated as an inventory indicator method, and other impact categories calculated using the TRACI LCIA method.

3.3.1 Cumulative Energy Demand (CED)

CED calculates the accumulated direct and indirect energy requirements associated with a product system (see Section 2.2.5) (Frischknecht et al., 2007b; Kim & Dale, 2008; Kraatz, Reinemann, & Berg, 2008; Huijbregts et al., 2010). In SimaPro the CED value is automatically calculated using preset conversion factors. When calculating CED, five energy resources were taken into account: non-renewable fossil, non-renewable nuclear, renewable biomass, renewable water, and renewable wind, solar, and geothermal.

3.3.2. Waste Heat

In this study, waste heat is employed as an indicator to measure the energy efficiency performance of the product system. In SimaPro waste heat is calculated by accounting for all waste heat released in form of emissions to air, water, and soil (Althaus et al., 2010). The ISO 14044 (2006) standard indicates that after characterization, “a set of inventory results that are elementary flows” (clause 4.4.2.5) could be presented as results. Under this logic, this study uses waste heat as a LCIA indicator method.

In SimaPro, waste heat inventory indicator is a part of EcoInvent *Selected LCI Results* method (Goedkoop, Oele, de Schryver, & Vieira, 2008). Goedkoop et al. (2008) indicate that the use of any inventory indicator “does not replace the use of the complete set of LCI results and the application of LCIA methods” (p. 42). The selection of inventory indicators (e.g., waste heat and carbon monoxide) was meant to help LCA practitioners get access to a group of LCI results of products without any indication of their environmental significance.

The growing concerns of non-renewable energy use and future economic and environmental costs (see Chapter 1, Section 1.1) in addition to the potential influence of waste heat reduction and recovery on products’ environmental performances as discussed in Chapter 2, Section 2.2.5.3, highlighted the importance of incorporating waste heat indicator in this study. Yet, none of the assessed studies in the literature reviewed in Chapter 2 have quantified this indicator or used it as a metric for environmental performance of products.

3.3.3 Other Impact Categories

To provide a broader set of LCIA categories, the current study uses TRACI, an impact assessment method that is used to calculate the other impact categories that is widely applied in LCA studies in North America. It was developed by the U.S. EPA and is therefore highly credible (Bare, 2011). TRACI uses emissions factors for USA locations assessing multiple impact categories. Since this study is intended for North American audiences including research, industrial, and governmental partners, TRACI method is a relevant and useful method. Other approaches considered included alternative methods (e.g., Eco-Indicator 95 and CML 2001) that are widely employed in LCA but that are based on European conditions.
TRACI includes nine impact categories: ozone depletion, global warming potential, acidification, eutrophication, photochemical oxidation (smog) formation, carcinogens, non-carcinogens, respiratory effects and ecotoxicity. In this study all nine categories were used. Values were calculated in SimaPro software.

4. Results

This chapter presents the LCA cradle-to-gate results for the reference flow of 1kg of the hybrid composite and 1.1 kg of the baseline composite, and provides results of the comparison between the hybrid and the baseline composites.

The term "hybrid PBS" is used to describe the product that results from three unit processes: hybrid PBS (3.6% water) production via polymerization, hybrid PBS (wet) via extrusion, and hybrid PBS pellet via drying. The term "hybrid composite" refers to the 50% SG reinforced hybrid PBS composite and "baseline composite" refers to the 30% GF reinforced PP composite (Figures 16 and 17).

4.1 Hybrid and Baseline Composite Comparison

In this section the environmental impacts associated with the production of the 50% SG-reinforced hybrid PBS composite and the 30% GF-reinforced PP composite are presented, consistent with objective # 1 of this study. The results of cumulative energy demand, waste heat indicator, and other impact categories (calculated using TRACI method) for the production of the 1 kg hybrid composite and 1.1 kg baseline composite are summarized in Table 11.

The hybrid composite had lower values for eight of the eleven assessed indicators and impact categories compared to the baseline composite (Table 11). Whereas, the values of ozone depletion, acidification, and eutrophication impact categories of the hybrid composite were higher in comparison to the baseline composite.

Because these are composites, it is also important to understand the influence of the fibre component on the composite environmental impacts. Table 12 shows that switchgrass fibres have a small contribution to all of the indicators for the hybrid composite, whereas, glass fibres have a higher contribution to almost all the indicators of the baseline composite.

Table 11*: Hybrid composite vs. baseline composite environmental profile results for all eleven indicators and impact categories for a reference flow of 1.0 kg hybrid composite and 1.1 kg baseline composite. Bold numbers represent the higher indicators' values.*

Description		Unit	Baseline Composite	Hybrid Composite	% of Difference
Indicator	Cumulative Energy Demand	MJ	**80.9**	48.7	-40%
	Waste Heat	MJ	**38.3**	29.6	-23%
Impact Category	Ozone Depletion	kg CFC-11 eq	1.29E-07	**1.85E-07**	+43%
	Global Warming Potential	kg CO_2 eq	**2.88E+00**	1.86E+00	-35%
	Smog	kg O_3 eq	**1.60E-01**	1.57E-01	-2%
	Acidification	mol H+ eq	6.56E-01	**7.64E-01**	+16%
	Eutrophication	kg N eq	4.48E-03	**1.89E-02**	+322%
	Carcinogens	CTUh	**1.15E-07**	5.24E-08	-54%
	Non-Carcinogens	CTUh	**3.73E-07**	-2.67E-07	-172%
	Respiratory Effects	kg PM10 eq	**2.50E-03**	1.95E-03	-22%
	Ecotoxicity	CTUe	**3.89E+00**	2.14E+00	-45%

In the context of this study, what is important is whether the hybrid composite's higher values of ozone depletion, acidification and eutrophication impact categories in comparison to the baseline composite are worth taking into consideration in the decision-making process comparing the two composites. These category indicators were found important to consider given experience by Schindler (2012), Keller (2009), Selman & Greenhalgh (2009), Environment Canada (2001), Keller et al. (2004), Great Lakes Water Quality Agreement (2012) and Ontario Regulation 463/10 (2010).

In general, a comparison between GF and SG fibres is not considered fair unless it takes into account the mass difference between the two fibres. Yet, what is important in this study is that the two composites are functionally equivalent regardless of the fact that fibre content is different. Having said that, in comparison with SG cultivation, the production of GF is a highly intensive heat, electricity, and waste heat process. This is due to the glass melting process, which mainly uses energy generated from natural gas and electricity (Elvebakken, 2002 as cited by Kellenberger et al. 2007). The energy requirement and waste heat associated with the melting process are 12 MJ/kg and 5.8 MJ/kg, respectively as calculated by

Table 12: *Proportional influence of polymer and fibre on environmental indicators for the two composites for a reference flow of 1.0 kg hybrid composite and 1.1 kg baseline composite.*

Description		Hybrid Composite		Baseline Composite	
		50% SG Fibre	**50% Hybrid PBS Polymer**	**30% GF Fibre**	**70% PP Polymer**
Indicator	Cumulative Energy Demand	1.2%	98.8%	20.6%	79.4%
	Waste Heat	2.4%	97.6%	47.5%	52.5%
Impact Category	Ozone Depletion	3.0%	97.0%	99.8%	0.2%
	Global Warming Potential	5.1%	94.9%	37.5%	63.5%
	Smog	2.0%	98.0%	70.0%	30.0%
	Acidification	4%	96.0%	72.0%	28.0%
	Eutrophication	0.5%	99.5%	92.0%	8.0%
	Carcinogenics	4.0%	96.0%	75.0%	25.0%
	Non Carcinogenics	2.0%	98.0%	99.0%	1.0%
	Respiratory Effects	2.0%	98.0%	73.0%	27.0%
	Ecotoxicity	2.0%	98.0%	72.0%	28.0%

Windsperger (2002), and cited by Kellenberger et al. (2007). Additionally, the production process of GF causes a higher level of combustion emissions particularly CO_2, nitrogen oxide (NO_x), and hydrogen fluoride (HF). These emissions also contain metallic dust and chlorides which are accounted in the TRACI impact assessment method under acidification, respiratory, and ozone depletion. The SG cultivation, on the other hand, is a low intensive, non-industrial production process. These grasses are grown and harvested with minimal energy and chemical inputs compared even to other commercial agriculture (McLaughlin et al., 1999; Samson et al., 1993; Carvalho, 2006).

Similarly, in the comparison between the fraction of polymer contribution to the environmental profile of the composite, PP has a lower contribution (as a percentage) to CED, GWP, waste heat, and other category indicators (see above Table 12) in comparison to PBS. This is due to the economy of scale and efficiency of the one-step production process of PP via polymerization. PBS, on the other hand, is produced by means of two-stage polymerization process.

In this comparison, it is noticeable that what influence the environmental performance of the composite is not whether the polymer is bio or petroleum based, but the type of reinforcing materials used in the composite.

4.2 Hybrid Composite - Cumulative Energy Demand Indicator

The CED required for the production of 1 kg hybrid composite is 48.7 MJ (Figure 18). The contribution of the major unit processes to CED for the production of 1 kg of the hybrid composite shows that the production of petroleum-BDO uses 28.2 MJ of total CED, and contributes 58% of CED.

By looking at the production of BDO unit process in details, results indicate that heat generation and the production of formaldehyde are the two biggest contributors to BDO energy demand with 41%, and 34%, respectively. Natural gas for heat generation is responsible for 59% of heat CED value, while methanol used for the production of formaldehyde contributes to 93% of formaldehyde CED value and that is a result to the natural gas used for methanol production.

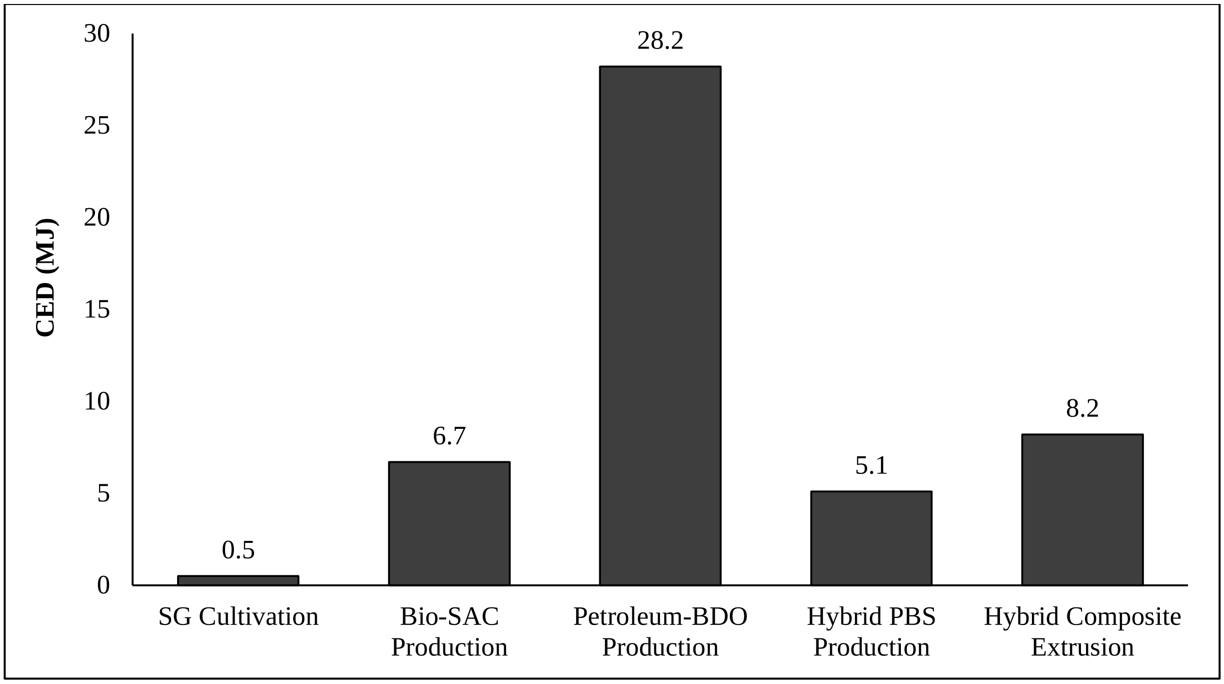

Figure 18*: Contribution of major unit processes to CED required for the production of 1 kg of hybrid composite.*

4.3 Hybrid Composite - Waste Heat Indicator

The amount of waste heat identified as "emissions to air, water and soil" is 29.6 MJ per 1 kg of the hybrid composite (Figure 19), and represents 61% of the cumulative energy demand required for the production of the hybrid composite.

The production of petroleum-based BDO generates 22.3 MJ of waste heat, and contributes to 75% of waste heat (Figure 19). This is mainly resulting from the waste heat from natural gas burned in Japan to generate heat for the production of petroleum-BDO, which accounts for 61% of heat generation waste heat and 32% of BDO production total waste heat.

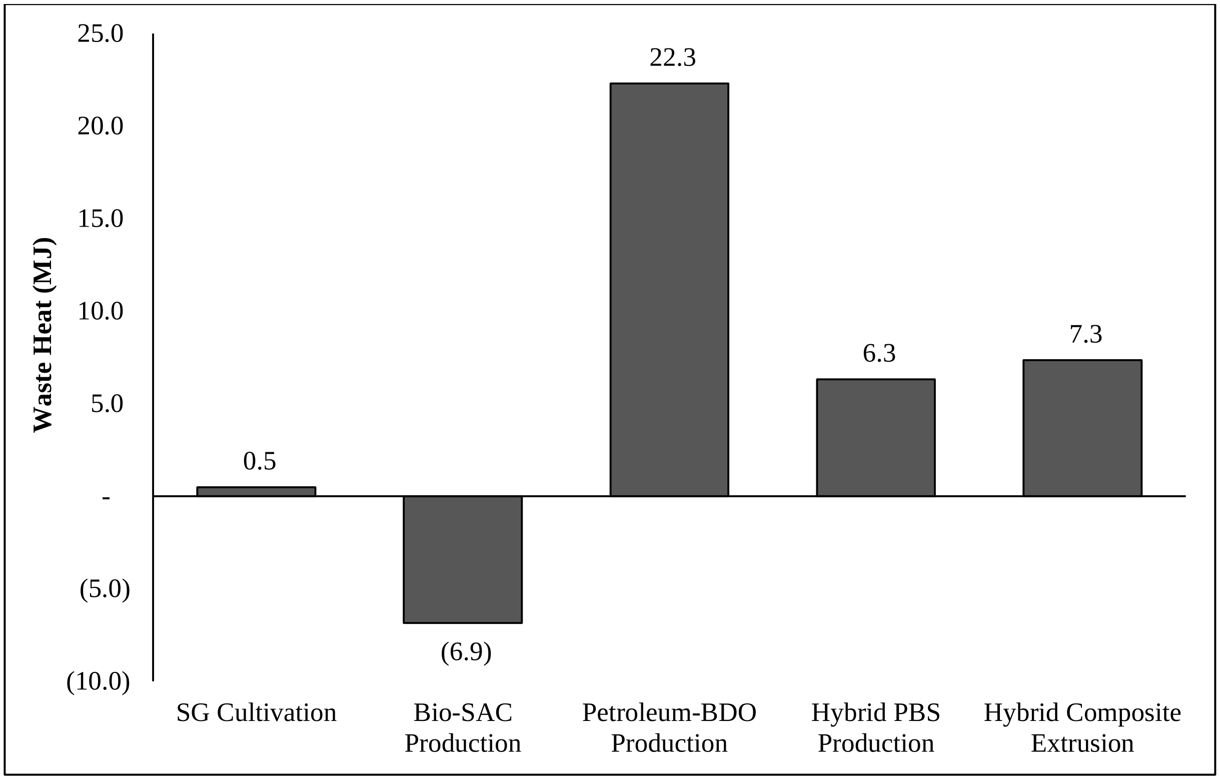

Figure 19*: Contribution analysis of each unit process to waste heat indicator per 1 kg of hybrid composite*

The BDO production model in SimaPro was built by EcoInvent based on the assumption that 100% of the electricity used is converted to waste heat that is released to air (Sutter, 2007). A more in depth analysis on the waste heat associated with the production of petroleum-BDO indicates that waste heat from electricity use contributes 1.4% of BDO production waste heat. The technology used for the production of petroleum-based BDO goes back to the year 2001, yet, still represents the production technology in current use (Sutter, 2007). It is assumed that neither reactors heat insulation nor the heat recovery systems are therefore as effective as in more modern production facilities. Another possible explanation for the higher contribution of BDO production to waste heat indicator is that the generated waste heat has a temperature that makes the recovery process technologically and financially unviable (BCS Incorporated, 2008).

Figure 19 shows a negative value of waste heat generated as a result of the production of bio-SAC. This is due to the emission and energy credit assigned as a result of the system expansion approach that was employed to address the AMS co-product problem. Therefore, there is a subtraction of generated waste heat from the production of petroleum-based AMS was considered as an avoided product and subtracted from the waste heat generated during the production of bio-SAC. Since the AMS waste heat value is greater than the combined contribution of all other unit processes, the value of bio-SAC waste heat is negative (Figure 19).

Additionally, we can assume that most of the heat generated in the production of bio-SAC is recovered, since the factory in Louisiana is newly built using state of art technologies in regards to the insulation of reactors and heat recovery systems (Myriant Corporation, 2011).

4.4 Hybrid Composite - Other Impact Categories

TRACI characterization factors are used to calculate the results of other impact categories as discussed in Section 3.3.3.The results for the production of 1 kg of the hybrid composite are presented in Figure 20. Similar to the CED and waste heat indicators, the production of BDO contributes the most to ozone depletion, global warming potential, smog, carcinogens, non-carcinogens, respiratory effects, and ecotoxicity impacts (Figure 20), while bio-SAC production contributes the most to acidification. Figure 20 shows values of ozone depletion, global warming, carcinogens, and non-carcinogens category indicators for the production of bio-SAC are negative. The reason is that some of the LCI values that are known to contribute to these category indicators result from the production of petroleum AMS (e.g., HCFC-22, arsenic, chromium, lead, and mercury). Here, as a result of the system expansion approach used to address the AMS co-product problem, these emissions are considered as avoided and thus subtracted from category indicators (see Figure 20).

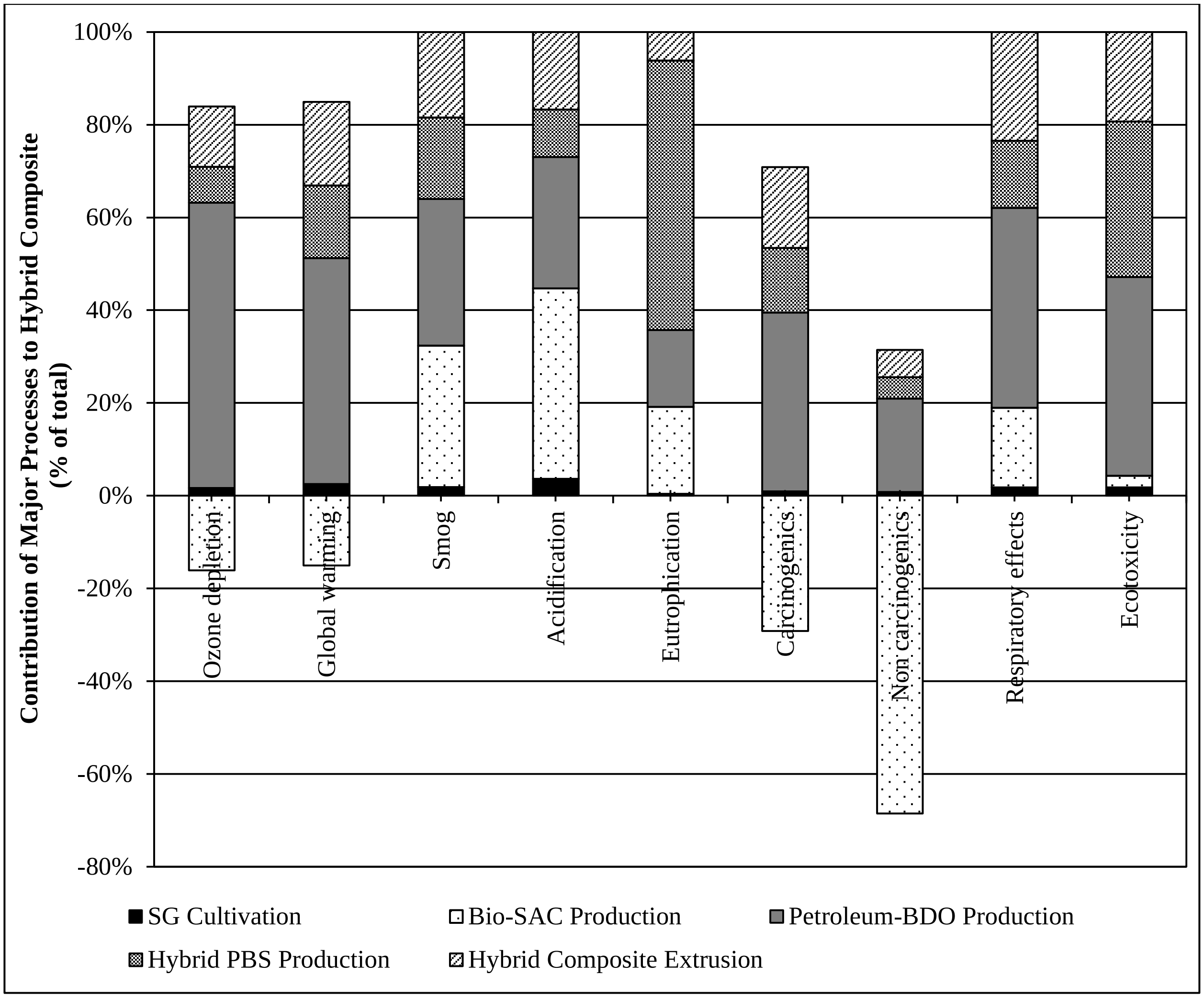

Figure 20*: Contribution of major processes to other impact categories for the hybrid composite production.*

The production of hybrid PBS contributes most to eutrophication. The electricity required for the production of oxygen used for the production of acetylene and required for the production of petroleum BDO contributes 99.8% of eutrophication potential as an impact category. This was found true given experience by Odeh and Cockerill (2008). Similar to CED and waste heat, it is the use of natural gas for heat in the production of BDO that contributes to ozone depletion, global warming, smog, acidification, non-carcinogens and respiratory effects (Figure 21). Additionally, the production of acetylene required for the production of BDO contributes to eutrophication, carcinogens, and ecotoxicity indicators. This is mainly due to the influence of oxygen production used for the production of acetylene.

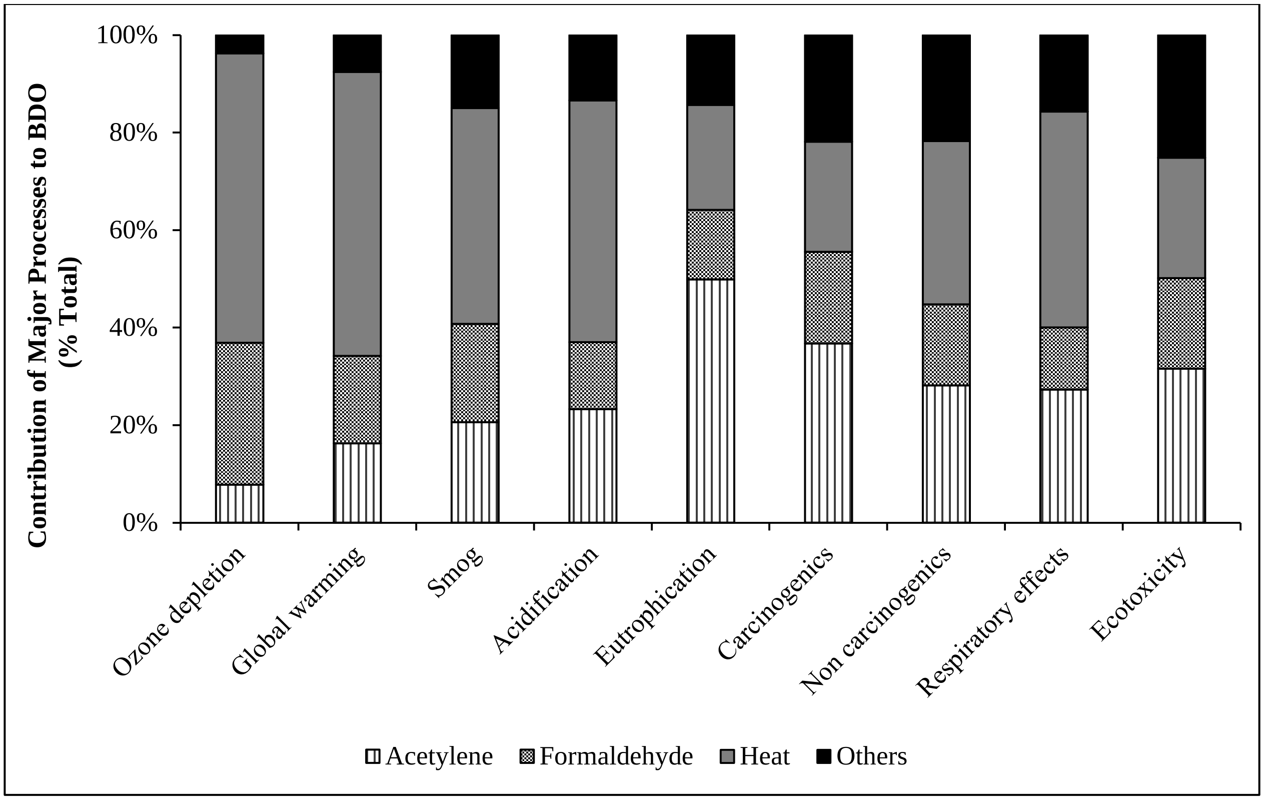

Figure 21: *Major processes contribution to other impact categories for BDO production.*

The contribution analysis of all the different unit processes and materials to other impact categories associated with the production of 1 kg bio-succinic acid are as in Figure 22. Similarly, the contribution of unit processes to the production of bio-SAC indicates that electricity use contributes the most to global warming, smog, acidification, carcinogens, non-carcinogens, and respiratory effects. This is mainly due to the impact of hard coal used as a part of SERC electricity grid mix as discussed in Section 3.2.3.1. The cultivation of sorghum contributes most significantly to eutrophication, and ecotoxicity. Natural gas for heat generation contributes the most to ozone depletion.

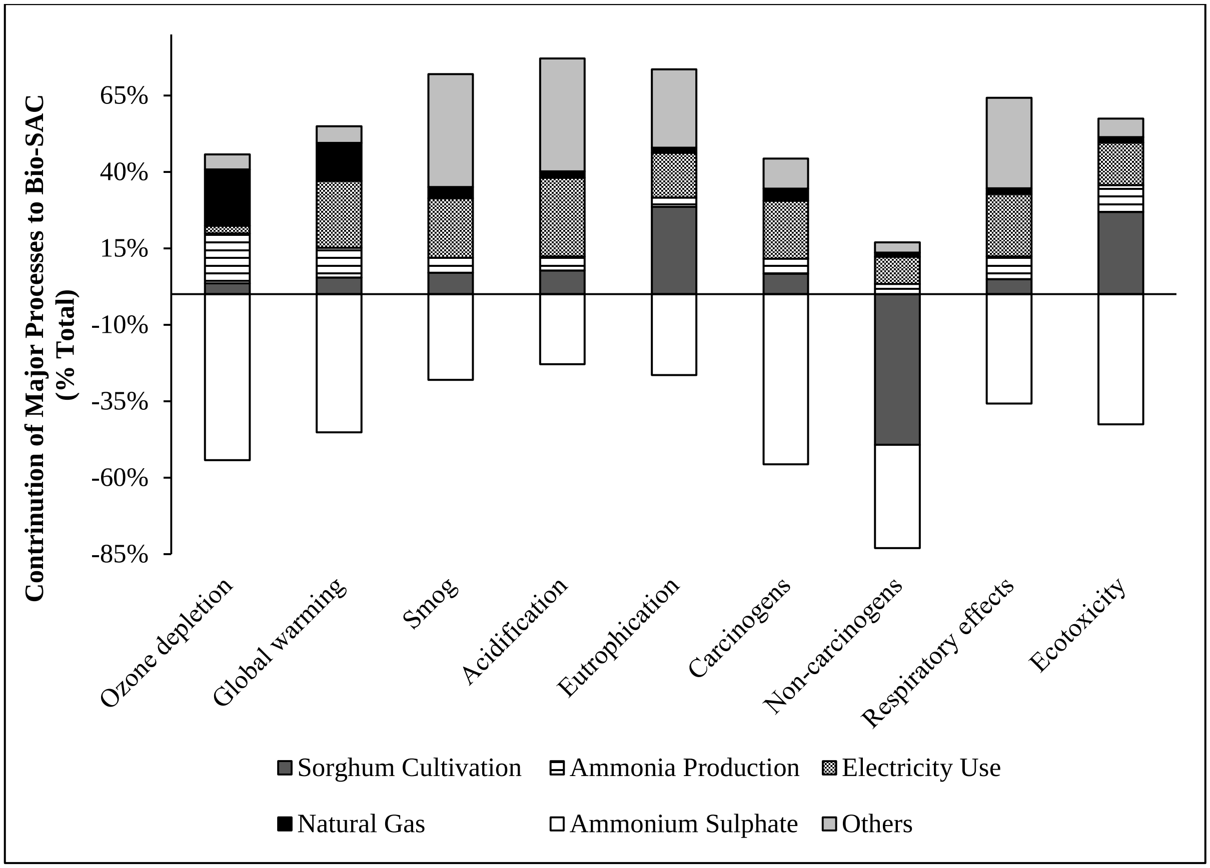

***Figure 22**: Major processes contribution to other impact categories for bio-SAC production.*

Figure 22 indicates that total values of ozone depletion and carcinogens impact categories are negative. This is a result of the co-product treatment where some emissions in the production of petroleum AMS (e.g., R-40, chromium, and lead) are avoided, hence, are subtracted.

The contribution analyses results (Figure 20, Figure 21, and Figure 22) identify electricity use, heat generation, and the cultivation of sorghum as the biggest contributors to other impact categories calculated using the TRACI impact assessment method. The influence of electricity on the other impact indicators is important for businesses to consider electricity from renewable sources with better environmental performance, and policy-makers to create more incentives for using renewable energy.

4.5 Sensitivity and Scenario Analyses

In this study, changes are considered significant if variation values shift by more than 10%. This is an arbitrary value but is consistent with suggested international practice (ISO, 2006b).

Uncertainty analyses were performed assessing the level of confidence in data used in the product system. These analyses were done using the Monte Carlo simulation in SimaPro. The range of calculated uncertainty values are in the acceptable range of LCA studies (PRé Consultants, 2010).

4.5.1 Sensitivity Analysis

The influence of co-product methodology choice on results was tested. One sensitivity analysis is carried out to test co-product treatment method for the bio-SAC and the hybrid composite. The parameters considered were CED and GWP. These two indicators are widely used in literature (see Section 2.1.9) to assess the environmental performance of products, and are therefore easily understood across a variety of systems.

By default, for the production of bio-SAC this study used the system expansion approach to address the co-product problem. For variation the study uses mass allocation instead. This decision was made in compliance with the ISO 14044 standard recommendations, which suggest that if allocation cannot be avoided, co-products are required to be divided based on physical association (mass, energy or heating values) (Weidema, 2000; Mortimer, Elsayed, & Evans, 2007; Nuss & Gardner, 2013; Iriarte & Villalobos, 2013). Mass allocation was used in this study as the approach for the co-product. Table 3, Section (3.2.2.2) indicates that the bio-SAC: AMS mass ratio is 1:1.49. Results for the sensitivity analyses are as in Table 13.

There is a significant change in CED and GWP of both bio-SAC and hybrid composite when comparing system expansion with mass allocation (Table 13). For bio-SAC, values increased by 19% for CED and 111% for GWP. Similarly for hybrid composite, indicator's values are 15% and 58% higher for CED and

GWP, respectively. In the allocation method a co-product is not considered as avoided product, but is considered as a burden and its potential environmental impacts are allocated based on the amount of the co-product produced, in this case the AMS (as discussed in Section 3.2.2).

Table 13: *Sensitivity and scenario analyses results and percentage of change on CED and GWP category and impact indicators for bio-SAC and hybrid composite.*

Sensitivity & Scenario Analyses	**Bio-SAC**							
	Default				**Variation**			
Category and Impact Indicator	**Co-product Treatment**	**Feed-stock**	**Production Route**	**Geographical Location**	**Co-product Treatment**	**Feed-stock**	**Productio n Route**	**Geographical Location**
CED (MJ)	31.4				37.2	69.9	87.7	-
Change (%)	-				+19%	+123%	+179%	-
GWP (kg CO_2 –eq)	0.87				1.84	1.50	4.22	-
Change (%)	-				+111%	+72%	+385%	-
Sensitivity / Scenario Analyses	**Hybrid Composite**							
	Default				**Variation**			
Category and Impact Indicator	**Co-product Treatment**	**Feed-stock**	**Production Route**	**Geographical Location**	**Co-product Treatment**	**Feed-stock**	**Production Route**	**Geographical Location**
CED (MJ)	48.7				56.0	68.0	74.8	44.6
Change (%)	-				+15%	+40%	+54%	-8%
GWP (kg CO_2 –eq)	1.9				3.0	2.8	3.84	1.58
Change (%)	-				+58%	+47%	+102%	-17%

4.5.2 Scenario Analyses

In this study, three scenario analyses were completed to test different initial assumptions around bio-based feedstocks, SAC production routes, and geographical location of production sites. Again, as in the sensitivity analysis, the LCIA category indicators used were CED and GWP. The scenario analyses criteria are as shown in Table 14.

Table 14: *Scenario analyses criteria for assessing the production of bio-SAC and hybrid composite*

Scenario Analysis Criteria	Default	Variation
Bio-based Feedstock to Bio-SAC Production	Sorghum and Dextrose	Dextrose
SAC Production Route	Bio-SAC	Petroleum-SAC
Hybrid Composite Production Geographical Location	Japan	Ontario

4.5.2.1 Feedstock

Initially, this study used a combination of sorghum grains and dextrose as feedstock for the production of bio-SAC. For variation, the study uses only dextrose. Myriant Corporation considers dextrose (from any source) as a potential replacement for the currently used feedstock. For this scenario analysis, the study used 1.82 kg of dextrose instead of 1.66 kg of sorghum grains and 1.33E-02 kg of dextrose. Table 13 illustrates the scenario analyses results for CED and GWP impact indicators associated with the production of bio-SAC and the hybrid composite.

There is a significant change in CED and GWP of both bio-SAC and hybrid composite when comparing sorghum and dextrose (default feedstock) with dextrose (variation feedstock). For bio-SAC, values increased by 123% for CED and 72% GWP. For the hybrid composite, values are 40% and 47% higher for CED and GWP, respectively.

This is expected since the production of dextrose is an energy intensive process (Tsiropoulos et al., 2013) in comparison to the cultivation of sorghum. Additionally, the amount of dextrose used for variation is greater than the amount used initially. The magnitude of change of bio-SAC values is greater than the hybrid composite indicator values. This is due to the fact that the contribution of bio-SAC production on the hybrid composite CED and GWP values is small at 14% and 15%, respectively.

4.5.2.2 SAC Production Route

By default, bio-SAC was used in the production of the hybrid composite; for variation, the study used petroleum-based SAC instead. The results of the scenario analyses are as illustrated in Table 13.

Results indicate greater changes in CED and GWP values for bio-SAC and less significance change in regards to the hybrid composite when using petroleum-SAC instead of bio-SAC. Bio-SAC, The values of indicators are increased by 179% for CED and 385% for GWP. This is due to the high environmental burden associated with the production of petroleum-SAC via the catalytical hydrogenation of maleic anhydride followed by the hydrolysis of succinic anhydride (Patel et al. 2006; Jentsch et al. 1997). These processes are highly energy intensive in comparison to production of bio-SAC. The values of hybrid composite indicators are increased by 54% for CED and 102% for GWP. The magnitude of change for the hybrid composite indicators' values is less in comparison to bio-SAC is a result for the low contribution of SAC to the environmental performance of the hybrid composite.

4.5.2.3 Geographical Location

Initially, the hybrid composite was assumed to be produced in Japan; as an alternative, the study uses Ontario, Canada since it is the location of the intended audience of this study. Scenario analyses results assessing the hybrid composite CED and GWP values are as illustrated in Table 13.

As a result of the change in geographical location of the hybrid composite production, the study notes a slight improvement in CED and GWP values. For Ontario, values decreased by 8% and 17% for CED and GWP, respectively. This is mainly due to influence of the electricity grid mix used for the production of the hybrid composite in the different geographical locations. The Japanese electricity grid mix per fuel type is based on Vivoda (2011) as discussed in Chapter 3, Section 3.2.3.1. Information on Ontario's electricity grid mix by fuel type is based on database from the Independent Electricity System Operator (IESO) (2012) as described in Table 15.

Table 15: *Ontario 2012 electricity supply mix by fuel type (IESO, 2013)*

Natural Gas	**Bituminous Coal**	**Lignite Coal**	**Hydro**	**Nuclear**	**Wind**	**Other**
14.6%	2.7%	0.1%	22.3%	56.4%	3.0%	0.8%

* Other includes: wood waste, biogas, etc.

Non-renewable fossil energy represents 83.5% of Japan electricity grid mix, with 25% being coal-fired electricity. In Ontario, 17.4% of electricity is generated using non-renewable fossil energy, with less than 3% being coal-fired electricity. Results were expected since the coal-fired electricity has a drastic influence on the overall environmental performance of a product as found by others (e.g., Khoo & Tan, 2010 and Puri, Compston, & Pantano, 2008). Nonetheless, the magnitude of change was less than expected.

5. Discussion

The original objectives of this study (see Section 1.4) are re-visited in this chapter. This section discusses whether the application of bio-based materials in the automotive industry is an environmentally desirable alternative option, when replacing the baseline composite that is in common use. Three areas are explored specifically: the area of significance in the hybrid system, potential opportunities for enhancing the environmental performance of the hybrid composite, and the geographical influence.

5.1 Comparison to Baseline Composite

This study aimed to explore whether bio-based materials in general and the SG / hybrid PBS composite in particular are environmentally alternative options for use in different industries, mostly the automotive industry.

The results of the current study align with the analysis of Weiss et al. (2012). However, there are some deviations from the general pattern of Weiss et al. (2012). The study by Weiss et al. (2012) assessed forty-four LCA studies comparing bio and petroleum-based materials and concluded that bio-based materials in general have lower energy demand, global warming potential, carcinogens, non-carcinogens and ecotoxicity indicator values. However, Weiss et al. (2012) suggest that indicators of ozone depletion, smog, eutrophication and respiratory effects may be greater. In regards to acidification values, Weiss et al. (2012) found no conclusive result due to the level of high uncertainty; yet, the study stated that, in general bio-based plastics and composites seem to reduce acidification.

Bio-fuels studies are a relevant point of reference for the evaluation of bio-based materials. The upstream production of bio-based materials and bio-fuels are similar, as both start from the cultivation of the agricultural feedstocks and continue to the production process via fermentation. Zah et al. (2007) compared 26 bio-fuels with three fossil fuels and stated that bio-fuels have lower CED and GWP, but at the same time results show higher value for smog, eutrophication and ecotoxicity indicators.

The results of this study support the choice of SG-reinforced hybrid PBS composite as an alternative option for use in the automotive industry. The potential change in study results from the initial ones is acknowledged if different sets of criteria, assumptions, and other impact categories were taken into account. Based on the latter, the decision making process in regards to the potential use of the hybrid composite as a replacement to the baseline composite is influenced to a great extent by the impact indicators used in the assessment. Studies by Weiss et al. (2012), Kimura & Horikoshi (2005), and Huijbregts et al. (2005) supported this inference, and indicated that the inclusion of land use change, for example, in the LCIA will result in higher GHG emissions from the production of bio-based materials in comparison with conventional counterparts.

The content of fibres in a composite represents an essential part of its structure: 50% by mass in the case of the hybrid composite and 30% by mass in the case of the baseline composite. For this reason, understanding the contribution of reinforcing fibre to the environmental performance of the composite is important. In the comparison between the two composites (see Section 4.1) it is apparent that the SG fibres contribute minimally to the environmental performance of the hybrid composite (in the range of 1%-5%); whereas, GF has significant influence on the environmental performance of baseline composite (varying from 21% to 99.8%). Therefore, the type of reinforcing material is a significant influence on the environmental profiles of plastic composite, regardless of whether the polymer matrix is biobased or petroleum based.

Policy makers and industry experts should be aware that other combinations and compositions of composites can be produced, and would have different environmental profiles. Based on the results of this study, it is interesting to consider a hypothetical SG/PP composite. This could potentially have lower LCIA indicator values than either the baseline or the hybrid composite. The lower LCIA indicator values are explained since PP in comparison to PBS has lower values across all assessed environmental impact categories and indicators (see Table 12, Section 4.1). Moreover, the SG environmental impact categories and indicators are lower in comparison to GF. These findings align with those from Terasawa et al. (2008) in regards to the polymer GHG emissions, and by Haufe and Carus (2011) and La Rosa et al. (2013) in regards to the fibres.

A broader analysis of product performance could be helpful to understand the sustainability performance of a product system. This is achievable by addressing different types of profiles or dimensions of the product system, like economic and social performance in addition to environmental performance, as in the spirit of LCSA (UNEP/SETAC Life Cycle Initiative, 2011).

For industries, the economic performance of of plastic composites in general and the hypothetical SG/PP composite in particular is also important.In general, the cost of the bio-based materials remains higher than that of their conventional counterparts (Jong et al., 2012; Bell, 2013; Koronis, Silva, & Fontul, 2013). Whether such a new composite is suitable for applications in the automotive industry would, of course, need to be explored.

A consideration that has not entered the analysis above is a potential weakness in the technical functionality of bio-based polymers. This was not considered in the functional unit in the analysis (see section 3.1.3) since the scope of study is cradle-to-gate and did not include downstream activities (e.g., use or end of life). Specifically, biodegradable composites (e.g., PBS composites) are more suitable for use in interior applications, as there is a risk that biodegradable components will hydrolyze, weaken, and degrade if exposed to sunlight, heat and moisture (T. Mizukoshi, personal communication, February 15, 2012; Koronis et al., 2013; Bledzki, Faruk, Jaszkiewicz, 2010). Adding hydrolysis inhibitor to biodegradable composites was found helpful to improving the hydrothermal aging, but not to stop it (Terasawa et al., 2008). Currently, bio-based and biodegradable polymers and composites are widely used for automotive interior parts (Bledzki et al., 2010; Terasawa et al., 2008), for example, battery trays. The potential benefits of using these materials for automotive applications were discussed in details in Section (2.1.6).

An additional consideration that differentiates the hybrid / bio-based systems from conventional materials is end-of-life (E.O.L.) performance. Conventional plastics, where recycling is an option, retain some economic value at E.O.L; bio-based composites do not. The inclusion of end-of-life scenarios is important since the baseline composite retains an economic value at its end of life if properly disposed not as waste to landfill. Conventional composite materials can be recycled, and re-used or incinerated to recover energy. In contrast, the hybrid composite is supposedly biodegradable, which means it retains no financial value. This is because biodegradable plastics are classified as neither recyclable (have no material value) nor compostable (have no nutritional value), but disposed as waste to landfill (Regional Public Works Commissioners of Ontario [RPWCO], 2009). Furthermore, there are no proper infrastructure and landfill conditions (temperature, humidity, and microorganisms) that can naturally activate the biodegradation process, which means more anticipated problems in the landfills (RPWCO, 2009).

When comparing the environmental performance (cradle-to-gate) of bio and petroleum based composites, this study suggests that it is important to focus on three main issues: the type of reinforcing materials in use, the energy mix used for heat production and the co-product treatment (allocation procedures).

5.2 Areas of Significance and Enhancement Opportunities

The generation of heat in the production of petroleum-based BDO and the use of electricity for the production of bio-SAC are the two areas of great influence on the environmental profile of the hybrid composite. These findings align with results from previous reports on conventional materials (U.S. EIA, 2013; Wettestad, 2008) and with the literature on bio-based materials (Haufe & Carus, 2011; La Rosa et al., 2013; Khoo et al., 2010; Puri, Compston, & Pantano, 2008).

There is a potential to ameliorate the environmental performance of the hybrid composite through improvements on the energy use (electricity and heat) and efficiency of the product system. A study by PE International (2011) indicates that the most widely used approaches to improve the environmental performance of a chemical product are to employ alternative energy sources, use different feedstocks, or apply varying production routes and technologies.

Two potential paths of change in process technology could help improve the environmental performance of the hybrid composite. The first relates to heat sources and electricity consumption in the production of BDO (Figure 21). If less fossil energy is used by means of substitution with other forms of energy there may be improvements in environmental performance. The second potential path is replacing the petroleum-based BDO with bio-based BDO. Similar to SAC, such a material would be produced using less energy and emitting fewer GHGs. Opportunities would also include bio-based BDO produced via the hydrogenation of bio-SAC using the Davy Process (Mang, 2012) or by means of the direct fermentation of sugar and biomass (Nexant, 2012). Bio-based BDO produced using the Davy Process offers a 33% reduction in non-renewable fossil energy demand and global warming potentials in comparison to petroleum-based BDO. This direction is supported by results from studies by Ichikawa and Mizukoshi (2012), Gironi and Piemonte (2011), Khoo et al. (2010) and Tabone et al. (2010), which all indicated that bio-based materials in general may potentially cause lower GWP and CED in comparison with petroleum-based counterparts.

This current study notes that waste heat and CED (energy use) are proportionally interrelated. Results indicate that similar to energy demand and global warming, most of the waste heat is related to the production of BDO. Energy efficiency and environmental performance of chemical processes in general are a function of modern and innovative production technologies that use renewable feedstocks, and may help in saving energy demand, in addition to the use of a better energy and electricity sources (Mata, Smith, Young, Costa, 2003; Patt & Banhilzer, 2009).

5.3 Geographical Influence

In this study, electricity drives some of the hybrid composite environmental performance impact categories. The mix of electricity on a power grid differs based on location. Given the novelty of bio-based technologies, it is relevant to consider whether the choice of geographical location, and therefore, electricity mix, might be a potential factor for industries to consider toward improving the environmental performance of bio-based materials produced (Collet et al., 2011).

In the product system considered, for example, processes assumed to be in Japan, could instead be geographically located in Ontario. Such a scenario assumes that all the production facilities of materials are technologically identical except for their choice of electricity mix. The hybrid composite would then have lower values for CED, GWP, and waste heat indicators. This can be explained by the fact that in Onatrio, electricity from fossil sources accounts for 17.4%, while in Japan it accounts for 83.5%. The magnitude of change in results were less than expected (between 8-15% for CED, GWP, and waste heat), yet, such a shift in location is environmentally significant taking into consideration the volume of production through-out the operational life-time of the production facility.

Similar, although the impact of electricity on the environmental profile of BDO is minimal, petroleum-based BDO produced in Ontario instead of Japan would likely have lower CED and GWP indicators. This shifts the assumption that bio-SAC production is geographically located in Ontario instead of Louisiana, USA. As a result of such change it would be likely to have a lower energy demand and GWP indicators' values due to the lower content of non-renewable fossil energy (mainly coal) in electricity grid mix in Ontario (2.8% as of 2012) in comparison with SERC electricity grid mix (49.6%). The magnitude of change in the environmental performance for the production of petroleum-based BDO and bio-SAC as a result of geographical change is likely to be greater in comparison with the hybrid composite. This observation is drawn based on results from Section 4.5.2 that show that the level of change in environmental performance of the intermediate products are greater than those of final products. Based on the above, a broad observation can be drawn that the electricity mix in general has an influence on a system environmental performance, whether these materials are bio or petroleum-based.

A shift in location would affect other decisions made in the LCA study. There would be a logical concern whether the co-product treatment methodology would remain valid or would need to be reconsidered. AMS produced as a co-product during the production of bio-SAC is sold locally in Lousiana as fertilizer. However, if an alternative location had no demand for the co-product, or if the co-product had significant impact on market prices, a different co-product treatment methodology would be necessary. Section 4.5.1 noted significant influences in the environmental performance indicators for the hybrid composite as a result of changes in co-product treatment method.

According to Weidema and Wesnaes (1996) addressing the potential influence of location on a product's environmental performance is important. The findings of the current study in particular and any other LCA studies in general are not generalizable, unless, the significance of location is considered (McKone et al., 2011; Khoo et al., 2010; Quantis, 2009; Ciroth et al., 2002).

5.4 Policy and Business Implications

For the intended audience of this study as introduced in Chapter 1, it is hoped that the results will be of value.

1. The results of this study should be useful to policy-makers for future developments of Ontario energy infrastructure since, that energy-in-process of the production of bio-based materials are just as critical, if not more so, than the materials used in the manufacturing processes. This study showed that energy is the major contributor to the environmental impacts of bio-based materials, using clean energy will potentially result in a better environmental performance. Ontario has relatively clean energy at grid, in addition to the abundance of natural resources, and reliable infra-structure, which support the agricultural and economic development opportunities in the province,

to ensure a competitive, innovative, and continuously growing economy that may potentially, help to position Ontario as a destination for global investors in the bio-industries in general and auto industry in particular. Furthermore, this study should inform policy-makers that supporting the development of future agricultural activities to mass produce bio-fibres for reinforcing plastic materials is not the ideal choice from an environmental prospective. This study results, showed that the contribution of SG fibres to the environmental performance of the hybrid composite is very minimal. For this reason, whether the natural fibres are locally cultivated and produced or imported has no significant environmental influence.

Additionally, policy-makers could use the results of this LCA study to explore the potentials of future implementation of regulations and industry incentives, to provide transparent information on the environmental performance of bio-based materials. Moreover, acknowledging the fact that bio-based industries are one of the bases of a bio-based economy, that is not only based on food, feed, and bio-fuels, but on advance industrial chemicals, and materials that may potentially replace the conventional counterparts in several industrial sectors, mainly automotive. However, the environmental performance of bio-based materials in general and the newly developed hybrid composite alone is not sufficient to build a solid decision to support moving forward toward a bio-economy. Decisions should be based on the big picture that includes the development of agricultural, business, economic, social, food safety, and rural policies, which ensure balance between economic and industrial prosperity, social and environmental justice.

2. Businesses could use this study to understand the potential environmental significance and benefits of bio-based materials, through identifying the environmental impacts of the different design and production stages, processes, and flows used on the environmental performance of the bio-based materials and the efficiency of its production system. Moreover, to recognize the potential to improve the overall environmental profile of these materials and the different

production stages and processes. Businesses are continuously looking for replacement materials that are mass produced and technically equivalent for the currently used materials, yet, with a better environmental and economic performance. Additionally, this study informs businesses of alternative potential options, which could offer better environmental and economic performances in comparison to both the hybrid and baseline composites.

Ontario automotive parts manufacturers are trying to regain their position among the top 10 global exporters[8] through exploring future opportunities in the global bio-based auto parts market. However, recapturing this position is only possible if new generations of bio-based materials with a better financial, technological, and environmental performance are developed.

This study holds valuable environmental insight for businesses, since; their procurement decisions have more power to influence the environmental impacts of products more than any other life cycle stage in their supply chain.

5.5 Future Research

A number of future research opportunities were identified:

1. There is value in a future study using bio-based BDO instead of conventional BDO as a potential path for improving the environmental performance of the hybrid composite. Technically, such a polymer would no longer be “hybrid” but would then be almost all bio-based.

[8] http://www.newswire.ca/en/story/1160775/canada-s-auto-parts-industry-gaining-ground-still-room-for-growth-scotiabank

2. It is recommended that future study should use indicators in addition to the LCIA metrics assessed here. Other considerations should include land use change (direct and indirect), loss of biodiversity, water and non-renewable resource depletion.

3. It is recommended that future researches consider the environmental performance of the hybrid composite through its whole life cycle (cradle-to-grave). Specifically, it is desirable to include biodegradation activities and impacts in a more complete analysis of potential environmental performance.

4. It is recommended that future researches pursue the interesting opportunity to explore the environmental performance of a SG-PP composite as a potential environmental and economic replacement to the hybrid and baseline composite.

5. It is recommended that, in an ambitious and wider-scoped research, a system-thinking approach to LCA that looks at the environmental performance of bio-based materials as a part of a socio-economic whole system. Conventional LCA is an imperfect tool as it does not yet address social and economic aspects and impacts of a product.

5.6 Conclusion

This study presents the first cradle-to-gate LCA of switchgrass reinforced hybrid poly (butylene succinate) composite. This was done to investigate the environmental performance of this newly developed composite in comparison to the baseline composite, identify the hotspots (heat and electricity), and recognize potential improvements that could help to improve the environmental impacts associated with the production of the hybrid composite.

When comparing the environmental performance (cradle-to-gate) of bio and petroleum based composites, it is important to focus on three main issues: the sources of energy used in the production process, the reinforcing materials in use and the co-product treatment methodology. Results revealed that the reinforcing material, not the polymer is what drives the potential environmental impacts of the produced composites, especially if the sources of energy in use are identical.

The intention of the current multidisciplinary study is not to provide an absolute answer or a conclusive result, yet, it is expected to contribute to the general knowledge and understanding of the potential environmental impacts associated with the production of bio-based materials.

References

Agriculture and Agri-Food Canada. (2008, December 11). *Biomass Inventory Mapping and Analysis Tool (BIMAT)*. Retrieved from Agriculture and Agri-Food Canada: http://www4.agr.gc.ca/

Agriculture and Agri-Food Canada. (2012, September 05). *New Agriculture and Agri-Food Canada Research Creating Environmentally Friendly Bioplastics from Potatoes*. Retrieved from Agriculture and Agri-Food Canada: http://www4.agr.gc.ca/AAFC-AAC/display-afficher.do?id=1261422415821&lang=eng

Allen, D. T. (2002). *Center for Energy and Environmental Resources, Environmental Science and Management 282, Life-Cycle Inventories* . Retrieved from The University of Texas: http://www.utexas.edu/

Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Frischknecht, R., Hellweg, S., . . . Nemecek, T. (2010). *Implementation of Life Cycle Impact Assessment Methods. ecoinvent report No. 3.* (R. Hischier, & B. Weidema, Eds.) St. Gallen, Switzerland: Swiss Centre for Life Cycle Inventories. EcoInvent Centre.

Althaus, H.-J., Hischier, R., Osses, M., Primas, A., Hellweg, S., Jungbluth, N., & Chudacoff, M. (2007). *Life Cycle Inventories of Chemicals. Data v2.0 (2007).* Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories. EcoInvent Centre.

American Chemistry Council [ACC]. (2012a). *Chemistry and Light Vehicles.* Washington, DC: Economics & Statistics Department - American Chemistry Council. Retrieved from http://www.plastics-car.com/lightvehiclereport

American Chemistry Council [ACC]. (2012b, March). *Plastics Statistics - 2011 Year In Review - Resins.* Retrieved from American Chemistry Council: http://www.americanchemistry.com/Jobs/EconomicStatistics/Plastics-Statistics/Year-in-Review.pdf

Anderson, B. (2011, February 22). *EXTRON 300 GF HS.* Retrieved from PolyPacific: http://www.polypacific.com.au/products/data-sheet/appliances

Andrady, A. L., & Neal, M. A. (2009). Applications and societal benefits of plastics. *Philosophical Transactions of The Royal Society B, 364*(1526), 1977-1984. doi:10.1098/rstb.2008.0304

Andrae, A. S. G. (2010). *Global life cycle impact assessments of material shifts*. London: Springer.

ASD Reports. (2013, March 04). *The World Glass Fiber Reinforced Plastic (GFRP) composites market to be worth $ 29.4bn in 2013*. Retrieved from ASDReports: https://www.asdreports.com

Averous, L., & Boquillon, N. (2004). Biocomposites based on Plasticized Starch: Thermal and Mechanical Behaviours. *Carbohydrate Polymers, 56*(2), 111-122. doi:10.1016/j.carbpol.2003.11.015

Babu, R. P., O'Connor, K., & Seeram, R. (2013). Review Paper: Current progress on bio-based polymers and their future trend. *Progress in Biomaterials, 2*(8). doi:10.1186/2194-0517-2-8

Bachmann, R. (2003). Industrial Biotech-New Value Creation Opportunities. *BIO Third Wave: Analyst Briefing on Industrial Biotechnology*. New York: Biotechnology Industry Organization .

Bai, L. (2010). *Life Cycle Assessment of Electricity Generation from Low Temperature Waste Heat. The Influence of Working Fluid (Master's Thesis).* Trondheim, Norway: Norwegian University of Science and Technology. Department of Energy and Process Engineering.

Bailey, R. (2013). The “Food Versus Fuel” Nexus. In A. Goldthau (Ed.), *The Handbook of Global Energy Policy*. Oxford, UK: John Wiley & Sons Ltd. doi:10.1002/9781118326282.ch16

Baitz, M., Partl, J., Braube, A., & Brauner, E. (2011). *White Paper: Improving the environmental performance of chemical products.* Leinfelden-Echterdingen, Germany: PE International AG. Retrieved from http://www.pe-international.com/

Baker, J. W., & Lepech, M. D. (2009). Treatment of Uncertainties in Life Cycle Assessment. *10th International Conference on Structural Safety and Reliability. :*. Osaka, Japan: International Association for Structural Safety and Reliability (IASSAR). Retrieved from http://www.stanford.edu/~mlepech/pubs/icossar.lca.uncertainty.09.pdf

Balboa, B. (2011, May 09). *INSIGHT: Is bio-based BDO for real?* Retrieved from Chemical Industry News & Chemical Market Intelligence (ICSI): http://www.icis.com/Articles/2011/05/09/9457930/insight-is-bio-based-bdo-for-real.html

Bare, J. (2011). TRACI 2.0: the tool for the reduction and assessment of chemical and othe renvironmental impacts 2.0. *Clean Technology and Environmental Policy, 13*(5), 687-696. doi:10.1007/s10098-010-0338-9

Bare, J. C., Norris, G. A., Pennington, D. W., & McKone, T. (2003). TRACI The Tool for the Reduction and Assessment of Chemcial and Otehr Environmental Impacts. *Journal of Industrial Ecology, 6*(3-4), 49-78. doi:10.1162/108819802766269539

Barker, D., Turner, S., Napier-Moore, P., Clark, M., & Davison, J. (2009). CO2 Capture in the Cement Industry. *Energy Procedia, 1*(1), 87-94. doi:10.1016/j.egypro.2009.01.014

Barnes, D. K., Galgani, F., Thompson, R. C., & Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B, 364*(1526), 1985-1998. doi:10.1098/rstb.2008.0205

Baumann, H., & Tillman, A.-M. (2004). *The Hitch Hiker's Guide to LCA.* Lund, Sweden: Studentlitteratur AB.

Bayart, J., Barteau, S., Worbe, S., von Eichhorn, R., & Pauly, D. (2012, June 22). Life Cycle Assessment and Water Footprint: tools for measuring the sustainability of water use. Paris, France: Veolia Environnement Recherche & Innovation. Retrieved from http://www.veolia.com

BCC Research. (2012, February). *Plastics - Global Markets and Technologies for Bioplastics*. Retrieved from BCC Research: http://www.bccresearch.com/report/bioplastics-markets-technologies-pls050b.html

BCS Incorporated. (2008). *Waste Heat Recovery: Technology and Opportunities in the U.S. Industry.* Laurel, Maryland : Department of Energy. Industrial Technologies Program.

BCS Incorporated. (2010). *Summary of Characteristics and Energy Efficiency Demand-side Management Programs in the Southeastern United States.* Laurel, USA: The U.S. Department of Energy - Industrial Technologies Program.

Beckwith, S. W. (2008). Natural Fibers: Nature Providing Technology for Composites. *Society for the Advancement of Material and Process Engineering (SAMPE) Journal, 44*(3), 64-65. Retrieved from http://www.sampe.org/news/TechTidbitsMayJune08.pdf

Bell, S. L. (2013). *Process Economics Program. Report 265A - Bio-Based Polymers.* Englewood, U.S.: IHS Chemicals. Retrieved from http://www.ihs.com/chemical

Bennett Jones LLP. (2008). *Canada: Key Federal & Provincial Actions to Address Greenhouse Gas Emissions.* Calgary, Canada: Bennett Jones LLP. Retrieved from http://www.bennettjones.com/Images/Guides/guide4062.pdf

Benoît, C., & Vickery-Niederman, G. (2010). *Social Sustainability Assessment Literature Review.* Tempe, USA: The Sustainability Consortium.

Bianco, N. M., Litz, F. T., Gottlieb, M., & Damassa, T. (2010). *Reducing Greenhouse Gas Emissions in the United States Using Existing Federal Authorities and State Actions.* Washington: World Resources Institute.

BioAmber. (2013, April 25). *BIO-SUCCINIC ACID (BIO-SA™)*. Retrieved from BioAmber: http://www.bio-amber.com/products/en/products/succinic_acid

Confederation of European Paper Industries [CEPI]. (2012). *Biobased for Growth. A Public-Private Partnership on Biobased Industries.* Brussels, Belgium: The European Commission. Retrieved from http://www.cepi.org

Biobased Industries Consortium [BIC]. (2013). *Biobased and Renewable Industries for Development and Growth in Europe. Strategic Innovation and Research Agenda (SIRA).* European Union: Biobased Industries Consortium (BIC).

BioProducts Canada. (2004). *Innovation Roadmap on Bio-based Feedstocks, Fuels and Industrial Products: Capturing Canada's Natural Advantage.* Ottawa: Industry Canada. Retrieved from http://www.titan-projects.com/assets/Files/Bio-based-Feedstocks-Fuels-and-Industrial-Products.pdf

Björklund, A. E. (2002). Survey of Approaches to Improve Reliability in LCA. *International Journal of Life Cycle Assessment, 7*(2), 64-72. doi:10.1065/lca2001.12.071

Bledzki, A. K., Faruk, O., & Jaszkiewicz, A. (2010). Cars from Renewable Materials. *Composites, 10*(3), 282-288.

Bledzki, A. K., Faruk, O., & Sperber, V. E. (2006). Cars from Bio-Fibers. *Macromolecular Materials and Engineering, 291*(5), 449-457. doi:10.1002/mame.200600113

Bohlmann, G. M. (2005). General Characteristics, Processability, Industrial Applications and Market Evolution of Biodegradable Polymers. In C. Bastioli (Ed.), *Handbook of Biodegradable Polymers* (pp. 183-218). Shawbury: Smithers Rapra Technology. Retrieved from http://www.knovel.com/web/portal/browse/display?_EXT_KNOVEL_DISPLAY_bookid=1206&VerticalID=0

Borealis AG. (2010a). *Polypropylene for the global automotive industry Higher demands, innovation solutions.* Vienna, Austria: Borealis Group. Retrieved from http://www.borealisgroup.com/

Borealis AG. (2010b). *Industry Solutions - Mobility - Lightweight Vehicles - Interior*. Retrieved from Borealis Group: http://www.borealisgroup.com/industry-solutions/mobility/lightweight-vehicles/interior

Bos, H. L. (2004). *The Potential of Flax Fibres as Reinforcement for Composites Materials.* Eindhoven, The Netherlands: Eindhoven University of Technology. Retrieved from http://alexandria.tue.nl/extra2/200411364.pdf

Boustead, I. (2005). *Eco-profiles of the European Plastics Industry. Polypropylene (PP).* Brussels, Belgium: Association of Plastics Manufacturers - PlasticsEurope.

Boustead, I., & Hancock, G. F. (1979). *Handbook of Industrial Energy Analysis. First Edition edition.* New York: Ellis Horwood Ltd.

Bray, R. (2008). *Bio-Based Polymers.* Menlo Park, California: SRI Consulting.

British Plastics Federation [BPF]. (2012, April 03). *A History of Plastics*. Retrieved from British Plastics Federation - Plastipedia: http://www.bpf.co.uk/plastipedia/plastics_history/default.aspx

Broekema, R. (2013). *Bio-based versus petroleum-based plastic: A choice between climate change and land use?* Retrieved from Blonk Consultants: http://blonkconsultants.nl/en/publications/news-letter/bio-vs-petro-based-plastic.html

Bundela, P., & Chawla, V. (2010). Sustainable Development through Waste Heat Recovery. *American Journal of Environmental Sciences*, *6*(1), 83-89. doi:10.3844/ajessp.2010.83.89

Burk, M. J., Van Dien, S. J., Burgard, A. P., & Niu, W. (2011). *Patent No. 8067214.* United States.

Canada Mortgage and Housing Corporation [CMHC]. (2004). *Sensitivity and Uncertainty: Annex 31 Energy-Related Environmental Impact of Buildings.* Ottawa: Canada Mortgage and Housing Corporation. Retrieved from http://www.iisbe.org/annex31/pdf/K_sensitivity.pdf

Canals, L., Azapagic, A., Doka, G., Jefferies, D., King, H., Mutel, C., . . . Williams, A. (2011). Approaches for Addressing Life Cycle Assessment Data Gaps for Bio-based Products. *Journal of Industrial Ecology, 15*(5), 707-725. doi:10.1111/j.1530-9290.2011.00369.x

Carpenter, S., Caraco, N., Correll, D., Howarth, R., Sharpley, A., & Smith, V. (1998). Non-point Pollution of Surface Waters with Phosphorus and Nitrogen. *Ecological Applications*, *8*(3), 559-568. doi:10.1890/1051-0761(1998)008[0559:NPOSWW]2.0.CO;2

Carus, M., & Gahle, C. (2008, September 22-23). Natural fibre reinforced plastics - Materials with Future. *11th International AVK-convention and the 6th Natural Fiber Conference*. Essen, Germany: Federation of Reinforced Plastics (AVK).

Carvalho, F. P. (2006). Agriculture, pecticides, food security and food safety. *Environmental Science & Policy, 9*(7-8), 685-692. doi:10.1016/j.envsci.2006.08.002

Center for Climate and Energy Solutions. (2012). *Federal Vehicle Standars - Light-Duty Vehicle Standards (Model Years 2012 to 2025)*. Retrieved from Center for Climate and Energy Solutions: http://www.c2es.org/federal/executive/vehicle-standards#ldv_2012_to_2025

Chen, X., & Khanna, M. (2013). Food vs. Fuel: The Effect of Biofuel Policies. *American Journal of Agricultural Economics, 95*(2), 289-295. doi:10.1093/ajae/aas039

Chiaberge, M. (Ed.). (2011). *New Trends and Developments in Automotive Industry.* Rijeka, Coratia: InTech Open Access Publisher. Retrieved from http://www.intechopen.com/books/new-trends-and-developments-in-automotive-industry

Chin, I.-J., & Uematsu, S. (2011). Biodegradation of Polymeric Systems. In V. Mittal (Ed.), *Nanocomposites with Biodegradable Polymers: Synthesis, Properties, and Future Perspectives* (pp. 28-57). New Youk, U.S.: Oxford University Press.

Cho, H., Moon, H., Kim, M., Nam, K., & Kim, J. (2011). Biodegradability and biodegradation rate of poly(caprolactone)- starch blend and poly(butylene succinate) biodegradable polymer under aerobic and anaerobic environment. *Waste Management, 31*(3), 475-480. doi:10.1016/j.wasman.2010.10.029

Ciroth, A., Hagelüken, M., Sonnemann, G. W., Castells, F., & Fleischer, G. (2002). Geographical and Technological Differences in Life Cycle Assessment Inventories. Part II. Technological and Geographical Differences. *The International Journal of Life Cycle Assessment, 7*(6), 363-368. doi:10.1065/lca2002.10.095.2

Collet, P., Hélias, A., Lardon, L., Ras, M., Goy, R.-A., & Steyer, J.-P. (2011). Life-cycle assessment of microalgae culture coupled to biogas production. *Bioresource Technology, 102*(1), 207-214. doi:10.1016/j.biortech.2010.06.154

Consoli, F., Allen, D., Boustead, I., Fava, J., Franklin, W., & Jensen, A. (1993). *Guidelines for Life-cycle Assessment: A Code of Practice.* Pensacola, USA: Society of Environmental Toxicology and Chemistry (SETAC).

Cooney, C. (2009). LCA finally takes water into account. *Environmental Science and Technology, 43*(11), 3986-3886. doi:10.1021/es901078v

Corbière-Nicolliera, T., Gfeller Labana, B., Lubdquist, L., Leterrier, Y., Manson, J.-A., & Jolliet, O. (2001). Life cycle assessment of biofibres replacing glass fibre reinforcement in plastics. *Resources, Conservation and Recycling, 33*(4), 267-287. doi:10.1016/S0921-3449(01)00089-1

Corn Refiners Association. (2012, April 12). *Bioproducts*. Retrieved from Corn Refiners Association: http://www.corn.org/products/bioproducts/

Crank, M., Pattel, M., Marscheider-Weidemann, F., Schleich, J., Hüsing, B., & Angerer, G. (2005). *Techno-economic Feasibility of Large-scale Production of Bio-based Polymers in Europe.* Sevilla, Spain: European Comission - Director General Joint Research Centre. Retrieved from http://ftp.jrc.es/EURdoc/eur22103en.pdf

Crawford, C. (2009). Automotive Bioplastics; Back to the Future. *Canadian Chemical News*, 28-30. Retrieved from http://www.bioautocouncil.com/news/File.aspx?4e9f5201-2357-4847-b57a-cce4de3f4095

Curran, M. (2007). Studying the Effect on System Preference by Varying Coproduct Allocation in Creating Life-Cycle Inventory. *Environmental Science & Technology, 41*(20), 7145–7151. doi:10.1021/es070033f

Czachorski, M., & Leslie, N. (2009). *Source Energy and Emission Factors for Building Energy Consumption.* Washington, DC: Natural Gas Codes and Standards. Retrieved from http://ad.aga.susqtech.com/SiteCollectionDocuments/KnowledgeCenter/OpsEng/CodesStandards/0008ENERGYEMISSIONFACTORSRESCONSUMPTION.pdf

Czichos, H., Saito, T., & Smith, L. (Eds.). (2011). *Springer Handbook of Metrology and Testing. 2nd edition.* Berlin Heidelberg: Springer-Verlag. doi:10.1007/978-3-642-16641-9

Dahlke, B., Larbig, H., Scherzer, H., & Poltrock, R. (1998). Natural Fiber Reinforced Foams Based on Renewable Resources for Automotive Interior Applications. *Journal of Cellular Plastics, 4*, 361-379. doi:10.1177/0021955X9803400406

de Bruijn, H., van Duin, R., & Huijbregts, M. A. (2002). *Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards.* (J. B. Guinée, Ed.) London: Kluwer Academic Publishers. doi:10.1007/0-306-48055-7

de Guzman, D. (2010, June 02). *Bio-Car Zooms In - The Use and Development of Renewable Chemicals in Automotive Parts is Rising*. Retrieved from Chemical Industry News & Chemical Market Intelligence (ICSI): http://www.icis.com/Articles/2010/06/07/9364336/use-of-bio-based-auto-parts-is-increasing.html

de Jong, E., Higson, A., Walsh, P., & Wellisch, M. (2012). *Bio-based Chemicals - Value Added Products from Biorefineries.* Wageningen, The Netherlands: IEA Bioenergy.

Deru, M. (2009). *U.S. Life Cycle Inventory Database Roadmap.* Washington, D.C.: National Renewable Energy Laboratory (NREL), U.S. Department of Energy (DOE). Retrieved from http://www.nrel.gov/lci/pdfs/45153.pdf

Detzel, A., & Kruger, M. (2006). *Life Cycle Assessment of PolyLactide (PLA): A compariosn of food packaging made from NatureWorks PLA and alternative materials - Final Report.* Heidelberg: IFEU GmbH.

Devuyst, D. (1999). Sustainability assessment: the application of a methodological framework. *Journal of Environmental Assessment Policy and Management, 01*(04), 459-487. doi:10.1142/S1464333299000351

Dlamini, N. G., Fujimura, K., Yamasue, E., Okumura, H., & Ishihara, K. N. (2011). The environmental LCA of steel vs HDPE car fuel tanks with varied pollution control. *International Journal of Life Cycle Assessment, 16*(5), 410-419. doi:10.1007/s11367-011-0277-7

Drachman, P. (2009). *Industry Insight - Carbon Neutrality and Life Cycle: Analysis for Biodegradable Plastics.* Surrey, UK: Pira International Ltd.

Duffy, M., & Nanhou, V. Y. (2001). *Costs of Producing Switchgrass for Biomass in Southern Iowa, PM 1866.* Ames, Iowa: Iowa State University (ISU) Extension. Retrieved from http://iowaswitchgrass.com/__docs/pdf/Costs%20of%20Switchgrass.pdf

EarthShift. (2012). *SimaPro LCA Software. LCA Software for Measuring Sustainability Impacts*. Retrieved from EarthShift: http://www.earthshift.com/software/simapro

Edwards, K., & Anex, R. P. (2009). *Co-product Allocation in Life Cycle Assessment: A Case Study. Extended Abstract 2009-645-AWMA.* Stillwater, MN: Trinity Consultants. Retrieved from http://www.trinityconsultants.com/

Ekvall, T., & Finnveden, G. (2001). Allocation in ISO 14041—a critical review. *Journal of Cleaner Production, 9*(3), 197-208. doi:10.1016/S0959-6526(00)00052-4

Elbersen, H., Christian, D., Yates, N., EL Bassam, N., & Sauerbeck, G. (2001). Switchgrass in NW Europe. In *Switchgrass (Panicum virgatum L.) as an alternative energy crop in Europe. Initiation of a productivity network. Final Report* (pp. 5-12). The Netherlands: Agrotechnological Research Institute (ATO-DLO). Retrieved from http://www.switchgrass.nl/pdf/Sw_FinalRep_full2.pdf

Empey, B., & Clark, P. (2005). *Plastics Technology Roadmap: Looking Forward - Executive Summary.* Mississauga: Canadian Plastics Industry Association. Retrieved from http://www.bioautocouncil.com/pdf/CPIAroadmapExecutiveSummary.pdf

Environment Canada. (2012, November 27). *Backgrounder: Regulating Greenhouse Gas Emissions from Light-Duty Vehicles (2017-2025)*. Retrieved from Environment Canada: http://www.ec.gc.ca/default.asp?lang=En&n=56D4043B-1&news=1F13DA8A-EB01-4202-AA6B-9E1E49BBD11E

Environment Canada. (2001). *Threats to sources of drinking water and aquatic ecosystem health in Canada.* Burlington, Ontario. NWRI Scientific Assessment Report Series No. 1 72 p.: National Water Research Institute.

Environmental Protection Act: Ontario Regulation 452/09. Greenhouse Gas Emissions Reporting. (2011). Retrieved from http://www.e-laws.gov.on.ca/

Eriksson, O., & Finnveden, G. (2009). Plastic waste as a fuel - CO2- neutral or not? *Energy & Environmental Science, 2*(9), 907-914. doi:10.1039/B908135F

EuropaBio. (2008). *How Industrial Biotechnology can Tackle Climate Change.* Brussels, Belgium: The European Association for Bioindustries. Retrieved from http://www.bio-economy.net/reports/files/ib_and_climate_change.pdf

European Comission. (2011b, June). Future Brief - Plastic waste: redesign and biodegradability. *Science for Environmental Policy*(1). Retrieved from http://ec.europa.eu/environment/integration/research/newsalert/pdf/FB1.pdf

European Commission. (2010). *Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment (First Edition).* Ispra, Italy: European Commission, Joint Research Centre, Institute for Environment and Sustainability.

European Commission. (2011a, June). *In-Depth Report - Plastic Waste: Ecological and Human Health Impacts.* Brussels, Belgium: European Commission. Retrieved from http://ec.europa.eu/environment/integration/research/newsalert/pdf/IR1.pdf

Fava, J. A., Denison, R., Jones, B., Curran, M., Vigon, B., Selke, S., & Barnum, J. (1994). *A Technical Framework for Life-cycle Assessments* (Vol. Second printing). Pensacola, USA: SETAC Foundation for Environmental Education.

Fet, A. M. (1998). Environmental management tools and their application - A review with reference to case studies. *e 2nd International Conference on Technology Policy and Innovation.* Lisboa, Portugal: Center for Innovation, Technology and Policy Research.

Finkbeiner, M., Inaba, A., Tan, R., Christiansen, K., & Klüppel, H.-J. (2006). The New International Standards for Life Cycle Assessment: IS 14040 and ISO 14044. *International Journal of Life Cycle Assessment, 11*(2), 80-85. doi:10.1065/lca2006.02.002

Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., . . . Suh, S. (2009). Recent developments in Life Cycle Assessment. *Journal of Environmental Management, 91*(1), 1-21. doi:10.1016/j.jenvman.2009.06.018

Flieger, M., Kantorová, M., Prell, A., Řezanka, T., & Votruba, J. (2003). Biodegradable Plastcis from Renewable Sources. *Folia Microbiol*, *48*(1), 27-44. doi:10.1007/BF02931273

Foot, D., & Ross, S. (2004). Social Sustainability. In C. Galea (Ed.), *Teaching Business Sustainability: From Theory to Practice* (pp. 107-125). Sheffield, UK: Greenleaf Publishing.

Frank, A., Berdahl, J., Hanson, J., Liebig, M., & Johnson, H. (2004). Biomass and carbon partitioning in switchgrass. *Crop Science, 44*(4), 1391-1396. doi:10.2135/cropsci2004.1391

Franklin Associates. (2010). *Cradle-to-gate Life Cycle Inventory of Nine Plastics Resins and Four Polyurethane.* Kansas: The Plastics Division of The American Chemistry Council.

Frischknecht, R., Althaus, H.-J., Bauer, C., Doka, G., Heck, T., Jungbluth, N., . . . Nemecek, T. (2007a). The Environmental Relevance of Capital Goods in Life Cycle Assessments of Products and Services. *International Journal of Life Cycle Assessment*, 806-816. doi:10.1065.lca2007.02.308

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., & et al. (2007b). *Implementation of Life Cycle Impact Assessment Methods. ecoinvent report No. 3, v2.0.* Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories.

Frollini, E., Bartolucci, N., Sisti, L., & Celli, A. (2013). Poly(butylene succinate) reinforced with different lignocellulosic fibers. *Industrial Crops and Products, 45*, 160-169. doi:10.1016/j.bbr.2011.03.031

Fujimaki, T. (1998). Processability and properties of aliphatic polyesters, 'BIONOLLE', synthesized by polycondensation reaction. *Polymer Degradation and Stability, 59*(1-3), 209-214. doi:10.1016/S0141-3910(97)00220-6

Fujimaki, T., Hatano, Y., Hokari, T., Seki, S., & Takiyama, E. (1994, May 10). *Patent No. 5,310,782A.* United States.

Gardner, D., & Murdock, D. (2002). Extrusion of Wood Plastic Composites. Brewer, Maine, United States. Retrieved from http://entwoodllc.com/PDF/Extrusion%20Paper%2010-11-02.pdf

Gautam, S.-P. (2009). *Bio-degradable Plastics- Impact on Environment.* Delhi, India: Central Pollution Control Board - Ministry of Environment and Foresets. Retrieved from http://www.cpcb.nic.in/upload/NewItems/NewItem_150_PlasticsWaste.pdf

Gerhardt, R.-A. (1997). A comparative Analysis of the Effects of Organic and Conventional Farming Systems on Soil Structure. *Biological Agriculture and Horticulture, 14*(2), 139-157. doi:10.1080/01448765.1997.9754803

German, L., & Schoneveld, G. (2012). A review of social sustainability considerations among EU-approved voluntary schemes for bio fuels, with implications for rural livelihoods. *Energy Policy, 51*, 765-778. doi:10.1016/j.enpol.2012.09.02

Gibson, R. B. (2001). *Specification of sustainability-based environmental assessment decision criteria and implications for determining ''significance'' in environmental assessment.* Ottawa, Canada: Canadian Environmental Assessment Agency: Research and Development Programme.

Gibson, R. B. (2006). Beyond the Pillars: Sustainability Assessment as a Framework for Effective Integration of Social, Economic and Ecological Considerations in Significant Decision-Making. *Journal of Environmental Assessment Policy and Management, 8*(3), 259-280.

Gielen, D., Newman, J., & Patel, M. K. (2008). Reducing Industrial Energy Use and CO2 Emissions: The Role of Materials Science. *MRS Bulletin*, *33*(04), 471-477. doi:10.1557/mrs2008.92

Gironi, F., & Piemonte, V. (2011). Life Cycle Assessment of Polylactic Acid and Polyethylene Terephthalate Bottles for Drinking Water. *Environmental Progress & Sustainable Energy, 30*(3), 459-468. doi:10.1002/ep.10490

Girouard, P., & Samson, R. (2000). *The potential role of perennial grasses in the pulp and paper industry.* Retrieved from REAP-Canada: http://www.reap-canada.com/online_library/agri_fibres_forestry/3%20The%20Potential.pdf

Girouard, P., Mehdi, B., Samson, R., & Blais, P.-A. (1999). Commercial Production of Switchgrass in Eastern Ontario: A Management Guide. Research Report prepared by REAP Canada and Research Technology Transfer Section College d'Alfred of Guelf University.

Goedkoop, M., De Schryver, A., Oele, M., Durksz, S., & de Roest, D. (2010). *Introduction to LCA with SimaPro 7.* Amersfoort, The Netherlands: PRé Consultants.

Goedkoop, M., Oele, M., de Schryver, A., & Vieira, M. (2008). *SimaPro 7 Database Manual. Methods Library.* the Netherlands: PRé Consultants.

Gorrée, M., Guinée, J. B., Huppes, G., & van Oers, L. (2002). Environmental Life Cycle Assessment of Linoleum. *The International Journal of Life Cycle Assessment, 7*(3), 158-166. doi:10.1007/BF02994050

Government of Canada. (2012). Canadian Environmental Protection Act, 1999. *Canada Gazette: Part I, 146*(15). Retrieved from http://www.gazette.gc.ca/rp-pr/p1/2012/2012-04-14/pdf/g1-14615.pdf#page=20

Grabowski, A. A. (2013). *LCA data gaps in feedstocks of biobased plastics (Master's Thesis).* Michigan: Michigan State University.Gross, R. A., & Kalra, B. (2002). Biodegradable Polymers for the Environment. *Science, 297*(5582), 803-807. doi:10.1126/science.297.5582.803

Great Lakes Water Quality Agreement. Protocol Amending the Agreement Between the United States of America and Canada. (2012). Retrieved from http://www.ec.gc.ca/grandslacs-greatlakes

Guinée, J. B., Heijungs, R., & Huppes, G. (2004). Economic Allocation: Examples and Derived Decision Tree. *International Journal of Life Cycle Assessment, 9*, 23-33. doi:110.1065/lca2003.10.136

Guinée, J. B., Udo de Haes, H., & Huppes, G. (1993). Quantitative life cycle assessment of products: 1:Goal definition and inventory. *Journal of Cleaner Production, 1*(1), 3-13. doi:10.1016/0959-6526(93)90027-9

Guldbrandsson, F., Malmodin, J., & Bondesson, A. (2011). Quantifying the life cycle assessment uncertainty in the information and communication technology sector. *Life Cycle Management Conference LCM 2011.* Berlin, Germany: LCM 2011 Scientific Committee.

Guo, M., & Murphy, R. (2012). LCA data quality: Sensitivity and uncertainty analysis. *Science of the Total Environment, 435-436*, 230-243. doi:10.1016/j.scitotenv.2012.07.006

Haensel, W. (2011). Are We on Target? Is Europe on Track to Achieve its Ambitious Climate Change Targets? *A panel discussion at the European Parliament* . Brussels, Belgium: The Parliament Magazine and PlasticsEurope.

Haines, D., & Van Gerpen, J. (2012, June 28). Biodiesel and the Food vs. Fuel Debate. *Farm Energy Home*. University of Minnesota Extension. Retrieved from http://www.extension.org/pages/27136/biodiesel-and-the-food-vs-fuel-debate

Halleux, H., Lassaux, S., Renzoni, R., & Germain, A. (2008). Comparative Life Cycle Assessment of Two Biofuels Ethanol from Sugar Beet and Rapeseed Methyl Ester. *International Journal of Life Cycle Assessment, 13*(3), 184-190. doi:10.1065/lca2008.03.382

Hammond, G., & Jones, C. (2008). *Inventory of Carbon & Energy (ICE): Version 1.6a.* Bath, UK: University of Bath. Retrieved from http://www.bath.ac.uk/mech-eng/research/sert/

Han, H., Wang, X., & Wu, D. (2012). Mechanical properties, morphology and crystallization kinetic studies of bio-based thermoplastic composites of poly(butylene succinate) with recycled carbon fiber. *Journal of Chemical Technology & Biotechnology*. doi:10.1002/jctb.3956

Hao, N., Colson, G., Karali, B., & Wetzstein, M. (2013). Food before Biodiesel Fuel? *the Southern Agricultural Economics Association (SAEA) Annual Meeting.* Orlando, Florida. Retrieved from http://ageconsearch.umn.edu/bitstream/143078/2/Food%20before%20Biodiesel%20Fuel.pdf

Harper, C. A., & Keyser, P. D. (2008). SP704-A-Potential Impacts on Wildlife of Switchgrass Grown for Biofuels. *The University of Tennessee Agricultural Extension Services*. Retrieved from http://nativegrasses.utk.edu/publications/SP704-A.pdf

Hartman, J. C., Nippert, J. B., Brozco, R. A., & Springer, C. J. (2011). Potential ecological impacts of switchgrass (Panicum virgatum L.) biofuel cultivation in the Central Great Plains, USA. *Biomass and Bioenergy, 35*(8), 3415-3421. doi:10.1016/j.biombioe.2011.04.055

Haufe, J., & Carus, M. (2011). *Hemp Fibres for Green Products - An Assessment of Life Cycle Stucdies on Hemp Fibre Aapplications.* European Industrial Hemp Association (EIHA). Hürth, Germany: nova-Institute GmbH. Retrieved from www.nova-institute.eu

Health Canada. (2011, July 10). *Guidelines for Determining the Accepatability and Use of Recycled Plastics in Food Packaging Applications* . Retrieved from Health Canada: http://www.hc-sc.gc.ca/fn-an/legislation/guide-ld/recycled_guidelines-directives_recycle01-eng.php

Heijungs, R., & Huijbregts, M. A. (2004). A Review of Approaches to Treat UIncertainty in LCA. *2nd Biennial Meeting of the International Environmental Modelling and Software Society* (pp. 332-339). Osnabruck, Germany: International Environmental Modelling and Software Society. Retrieved from http://www.iemss.org/iemss2004/pdf/lca/heijarev.pdf

Helgeson, J. F., & Lippiatt, B. C. (2009). Multidisciplinary Life Cycle Metrics and Tools for Green Buildings. *Integrated Environmental Assessment and Management*, *5*(3), 390-398. doi:10.1897/IEAM_2008-069.1

Hemjinda, S., Krzan, A., Chiellini, E., & Miertus, S. (2007). *Environmentally Degradable Polymeric Materials and Plastics: Guidelines to Standards and Testing Practices.* Trieste, Italy: United Nations Industrial Development Organization and the International Centre for Science and High Technology -. Retrieved from http://institute.unido.org/

Hermann, B., Blok, K., & Patel, M. (2007). Producing Bio-Based Bulk Chemicals Using Industrial Biotechnology Saves Energy and Combats Climate Change. *Environmental Science and Technology, 41*(22), 7915-7921. doi:10.1021/es062559q

Hetherington, A. C., McManus, M. C., & Gary, D. A. (2011). Comparison of allocation and impact assessment methodologies on the life cycle assessment of rape and sunflower seed oils. *LCM 2011 - Towards Life Cycle Sustainability Management.* Berlin: Technische Universität Berlin. Retrieved from http://www.lcm2011.org/papers.html

Hewson, D., & Oo, A. (2009). *Report on Energy Crop Options for Ontario Power Generation.* London, Canada: The University of Western Ontario Research Park.

Hobson, J., & Carus, M. (2012, February 10). Document for The Commission & The Parliament: EIHA is supporting the "Greening" in Brussels CAP reform: It is a step in the right direction. Concerns about transition period. European Industrial Hemp Association. Retrieved from http://eiha.org/attach/690/12-02-10_Document_for_The_Commission__The_Parliament_December_2012_on_CAP.pdf

Hochschorner, E., & Finnveden, G. (2003). Evaluation of Two Simplified Life Cycle Assessment Methods. *The International Journal of Life Cycle Assessment, 8*(3), 119-129.

Hoekstra, A. Y. (2003). *Virtual water trade: Proceedings of the International Expert Meeting on Virtual Water Trade, Value of Water Research Report Series No. 12.* Delft, Netherlands: UNESCO-IHE. Retrieved from www.waterfootprint.org

Hoekstra, A. Y., Chapagain, A. K., Aldaya, M. M., & Mekonnen, M. M. (2011). *The Water Footprint Assessment Manual. Setting the Global Standard.* London, UK: Earthscan.

Hsu, S.-L., & Elliot, R. (2009). Regulating Greenhouse Gases in Canada: Constitutional and Policy Dimensions. *McGill Law Journal, 54*(3), 463-516.

Huda, M., Drzal, L., Mohanty, A., & Misra, M. (2008). Effect of fiber surface-treatment on the properties of laminated biocomposites form poly(lactic acid) (PLA) and kenaf fibers. *Composites Science and Technology, 68*(2), 424-432.

Huijbregts, M. A., Hellweg, S., Froschkecht, R., Hendriks, H. W., Hungerbuhler, K., & Jan Hendriks, A. (2010). Cumulative Energy Demand As Predictor for the Environmental Burden of Commodity Production. *Environmental Science and Technology, 44*(6), 2189–2196. doi:10.1021/es902870s

Huijbregts, M. A., Rombouts, L. J., Hellweg, S., Frischkecht, R., Hendriks, A. J., van de Meent, D., . . . Struijs, J. (2006). Is Cumulative Fossil Energy Demand a Useful Indicator for the Environmental Performance of Products? *Environmental Science and Technology, 40*(3), 641-648. doi:10.1021/es051689g

Hunkeler, D., Lichtenvort, K., & Rebitzer, G. (Eds.). (2008). *Environmental Life Cycle Costing.* Brussels, Belgium: Society of Environmental Toxicology and Chemistry (SETAC).

Ichikawa, Y., & Mizukoshi, T. (2012). Bionolle (Polybutylenesuccinate). *Advances in Polymer Science, 245*, 285-313. doi:10.1007/12_2011_145

Imaizumi, M., Kimura, H., Fujihira, R., Ichikawa, Y., Suzuki, J., Moteki, Y., . . . Takiyama, E. (1998). *Patent No. 5714569A.* United States.

Independent Electricity System Operator [IESO]. (2012, January 06). *Composition of Ontario's Electricity Supply Mix Continues to Change: Consumer Response Supports Reliability.* Retrieved from Independent Electricity System Operator: http://www.ieso.ca

Industry Canada. (2011a, November 16). *Canadian Synthetic Resins Industry*. Retrieved from Industry Canada: http://www.ic.gc.ca/eic/site/plastics-plastiques.nsf/eng/pl01384.html

Industry Canada. (2011b, November 16). *Description for the Canadian Synthetic Industry*. Retrieved from Industry Canada: http://www.ic.gc.ca/eic/site/plastics-plastiques.nsf/eng/pl01385.html

Industry Canada. (2012, January 09). *Industry Profile for the Canadian Plastic Products Industry*. Retrieved from Industry Canada: http://www.ic.gc.ca/eic/site/plastics-plastiques.nsf/eng/pl01383.html

Intergovernmental Panel on Climate Change [IPCC]. (2006). *2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4, Chapter 11.* (S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanane, Eds.) Kanagawa, Japan: Institute for Global Environmental Strategies (IGES). Retrieved from http://www.ipcc-nggip.iges.or.jp

International Organization of Motor Vehicle Manufacturers [OICA]. (2007). *Climate Change & CO2*. Retrieved from International Organization of Motor Vehicle Manufacturers: http://oica.net/category/climate-change-and-co2/

International Organization of Motor Vehicle Manufacturers [OICA]. (2010). *Sustainable Mobility: CO2 in the Road Transport Sector - The Integrated Approach - A study by OICA.* Paris:

International Organization of Motor Vehicle Manufacturers. Retrieved from http://oica.net/wp-content/uploads/oica-co2_report_final_11june2010.pdf

IPCC National Greenhouse Gas Inventories Programme . (2006). *2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4, Chapter 11.* Kanagawa, Japan: Institute for Global Environmental Strategies (IGES). Retrieved from http://www.ipcc-nggip.iges.or.jp

Iriarte, A., & Villalobos, P. (2013). Greenhouse gas emissions and energy balance of sunflower biodiesel: Identification of its key factors in the supply chain. *Resources, Conservation and Recycling, 73*, 46-52. doi:10.1016/j.resconrec.2013.01.014

ISO. (1998). *ISO 14041: Environmental Management - Life Cycle Assessment - Goal and Scope Definition and Inventory Analysis.* Geneva, Switzerland: International Organization for Standardization.

ISO. (2006a). *ISO 14040: Environmental Management - Life Cycle Assessment - Principles and Framework.* Geneva, Switzerland: International Organization for Standardization.

ISO. (2006b). *ISO 14044: Environmental management - Life cycle assessment - Requirements and guidelines.* Geneva, Switzerland: the International Organization for Standardization.

Jacquel, N., Freyermouth, F., Fenouillot, F., Rousseau, A., Pascault, J., Fuertes, P., & Saint-Loup, R. (2011). Synthesis and Properties of Poly(butylene succinate): Efficiency of Different Transesterification Catalysts. *Polymer Chemistry, 49*(24), 5301-5312. doi:10.1002/pola.25009

James, K., & Grant, T. (2005). LCA of Degradable Bags. *4th Australian LCA Conference.* Melbourne, Australia: Centre for Design - Royal Melbourne Institute of Technology (RMIT) Univeristy. Retrieved from http://conference.alcas.asn.au/2005/Papers/James_and_grant.pdf

Jefferies, D., Muñoz, I., Hodges, J., King, V. J., Aldaya, M., Ercin, A., . . . Hoekstra, A. Y. (2012). Water Footprint and Life Cycle Assessment as approaches to assess potential impacts of products on water consumption. Key learning points from pilot studies on tea and margarine. *Journal of Cleaner Production, 33*, 155-166. doi:10.1016/j.jclepro.2012.04.015

Jentsch, J.-D., Martin, G., & Zirngiebl, E. (1997). *Patent No. 5616730.* Germany.

Joshi, S.-V., Drzal, L.-T., Mohanty, A.-K., & Arora, S. (2003). Are Natural Fiber Composites Environmentlly Firendly Superior to Glass Fiber Reinforced Composites? *Composites Part A: Applied Science and Manufacturing, 35*(3), 371-376. doi:10.1016/j.compositesa.2003.09.016

Jungbluth, N., Bauer, C., Dones, R., & Frischknecht, R. (2005). Life Cycle Assessment for Emerging Technologies: Case Studies for Photovoltaic and Wind Power. *International Journal of Life Cycle Assessment, 10*(1), 24-34. doi:10.1065/lca2004.11.181.3

Jungmeier, G., Werner, F., Jarnehammar, A., Hohenthal, C., & Richter, K. (2002). Allocation in LCA of Wood-based Products. Experience of Cost Action E9 - Part I Methodology. *International Journal of Life Cycle Assessment, 7*(5), 290-294. doi:10.1ossnca2002.08.091.1

Kalita, B. (2012). *Life Cycle Assessment of Switchgrass (Panicum virgatum L.) Biomass Production in Ontario (Master's thesis).* Guelph: University of Gurlph.

Karp, A., & Shield, I. (2008). Bioenergy from plants and the sustainable yield challenge. *New Phytologist, 179*(1), 15-35. doi:10.1111/j.1469-8137.2008.02432.x

Kellenberger , D., Althus, H.-J., Kunniger, T., Lehmann, M., Jungbluth, N., & Thalmann, P. (2007). *Life Cycle Inventories of Building Products. ecoinvent report No. 7.* Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories. EcoInvent Centre.

Keller, W. (2009). Limnology in Northeastern Ontario: from Acidification to multiple Stressors. *Canadian Journal of Fishiries and Aquatic Sciences, 66*(7), 1189-1198. doi:10.1139/F09-080

Keller, W., Heneberry, J., Gunn, J. M., Snucins, E., Morgan, G., & Leduc, J. (2004). *Recovery of Acid and Metal - Damaged Lakes Near Recovery of Acid and Metal - Damaged Lakes Near*. Sudbury, Ontario: Laurentian University. Retrieved from http://www3.laurentian.ca/

Keoleian, G. A., & Volk, T. A. (2005). Renewable Energy from Willow Biomass Crops: Life Cycle Energy, Environmental and Economic Performance. *Critical Reviews in Plant Science, 24*(5-6), 385-406. doi:10.1080/07352680500316334

Khoo, H. H., Tan, R. B., & Chng, K. W. (2010). Environmental impacts of conventional plastic and bio-based carrier bags. *The International Journal of Life Cycle Assessment, 15*(3), 284-293. doi:10.1007/s11367-010-0162-9

Kim, H.-S., Yang, H.-S., & Kim, H.-J. (2005). Biodegradability and mechanical properties of agro-flour–filled polybutylene succinate biocomposites. *Journal of Applied Polymer Science, 97*(4), 1513-1521. doi:10.1002/app.21905

Kim, M.-N., Kim, K.-H., Jin, H.-J., Park, J.-K., & Yoon, J.-S. (2001). Biodegradability of ethyl and n-octyl branched poly(ethylene adipate) and poly(butylene succinate). *European Polymer Journal, 37*(9), 1843-1847.

Kim, S., & Dale, B. E. (2002). Allocation Procedure in Ethanol Production System for Corn Grain. I. System Expansion. *The International Journal of Life Cycle Assesmsnet, 7*(4), 237-243. doi:10.1065/lca2002.05.081

Kim, S., & Dale, B. E. (2008). Cumulative Energy and Global Warming Impact from the Production of Biomass for Biobased Products. *Journal of Industrial Ecology, Volume 7*(Issue 3-4), 147-162. doi:10.1162/108819803323059442

Kimura, K., & Horikoshi, Y. (2005). Bio-Based Polymers. *FUJITSU Scientific & Technical Journal, 41*(2), 173-180. Retrieved from http://www.fujitsu.com/downloads/MAG/vol41-2/paper07.pdf

Kline & Company INC. (2000). *Opportunities for Natural Fibers in Plastic Composites.* Little Falls, NJ: Kline & Company INC.

Kline & Company INC. (2002). *Opportunities for Natural Fibers in Plastic Composites, Western Europe 2002.* Little Falls, NJ: Kline & Company INC.

Klöpffer , W. (1997). In Defence of the Cumulative Energy Demand [Editorial]. *The International Journal of Life Cycle Assessment, 2*(2), 61. doi:10.1007/BF02978754

Klöpffer, W. (2008). Life Cycle Sustainability Assessment of Products. *The International Journal of Life Cycle Assessment, 13*(2), 89-95. doi:10.1065/lca2008.02.376

Koehler, A. (2008). Water use in LCA: managing the planet's freshwater resources. *The International Journal of Life Cycle Assessment, 13*(6), 451-455. doi:10.1007/s11367-008-0028-6

Köhler, A., & Aoustin, E. (2008). Assessment of Use and Depletion of Water Resources within LCA. *SETAC Europe Annual Meeting.* Warsaw, Poland: UNEP/SETAC Life Cycle Initiative.

Kolybaba, M., Tabil, L., Panigrahi, S., Crerar, W., Powell, T., & Wang, B. (2003). Biodegradable Polymers: Past, Present, and Future. *2003 CSAE/ASAE Annual Intersectional Meeting.* Fargo, North Dakota: The Society for Engineering in Agricultural, Food, and Biological Systems.

Koronis , G., Silva, A., & Fontul, M. (2013). Green composites: A review of adequate materials for automotive applications. *Composites Part B: Engineering, 44*(1), 120-127. doi:10.1016/j.compositesb.2012.07.004

Kraatz, S., Reinemann, D. J., & Berg, W. E. (2008). Energy Inputs for Corn Production in Wisconsin and Germany. *2008 ASABE Annual International Meeting.* Rhode Island: American Society of Agricultural and Biological Engineers. Retrieved from http://fyi.uwex.edu/uwmril/files/2011/05/08-ASABE-Kraatz-Corn-Energy-Inputs.pdf

La Rosa, A. D., Cozza, G., Latteri, A., Mancini, G., Recca, A., & Cicala, G. (2013). A Comparative Life Cycle Assessment of a Composite Component for Automotive. In S. Pierucci, & J. J. Klemeš (Ed.), *11th International Conference on Chemical and Process Engineering (ICheaP-11). 32*, pp. 1723-1728. Milan, Italy: AIDIC Servizi S.r.l. doi:10.3303/CET1332288

Lammers, P., & Kromer, K. (2002). Competitive Natural Fiber Used in Composite Materials for Automotive Parts. Paper No. 026167. *American Society for Agricultural Engineers Annual Meeting.* Chicago, Illinois: American Society for Agricultural Engineers (ASAE).

Law, K., Kokta, B., & Mao, C. (2001). Fibre morphology and soda-sulphite pulping of switchgrass. *Biosource Technology, 77*(1), 1-7. doi:10.1016/S0960-8524(00)00140-1

Leaversuch, R. D. (2002, September). Biodegradable Polymers: Packaging Goes Green. *Plastics Technology*. (R. D. Leaversuch, Ed.) Retrieved from http://www.ptonline.com/articles/biodegradable-polyesters-packaging-goes-green

Lee, D., Owens, V., & Doolittle, J. (2007). Switchgrass and Soil Carbon Sequestration Response to Ammonium Nitrate, Manure, and Harvest Frequency on Conservation Reserve Program Land. *Agronomy Journal, 99*(2), 462-468. doi:10.2134/agronj2006.0152

Lips, S., & Elbersen, H. (2001). Non-energy uses of switchgrass. In *Switchgrass (Panicum virgatum L.) as an alternative energy crop in Europe. Initiation of a productivity network. Final Report* (pp. 51-53). The Netherlands: Agrotechnological Research Institute (ATO-DLO). Retrieved from http://www.switchgrass.nl/pdf/Sw_FinalRep_full2.pdf

Lucintel. (2011, March). Lucintel Brief - Opportunities in Natural Fiber Composites. Las Colinas, TX, U.S.: Lucintel. Retrieved from http://www.lucintel.com/LucintelBrief/PotentialofNaturalfibercomposites-Final.pdf

Luo, L. (2010). *Biomass Refining for Sustainable Development: Analysis and Directions (Doctoral dissertation).* Leiden: Leiden University. Retrieved from https://openaccess.leidenuniv.nl/bitstream/handle/1887/15394/Chapter%207.pdf?sequence=9

Luo, L., van der Voet, E., Huppes, G., & Udo de Haes, H. A. (2009). Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol. *International Journal of Life Cycle Assessment, 14*(6), 529-539. doi:10.1007/s11367-009-0112-6

Ma, Z., Wood, C., & Bransby, D. (2000). Soil management impacts on soil carbon sequestration by switchgrass. *Biomass and Bioenergy, 18*(6), 469-477. doi:10.1016/S0961-9534(00)00013-1

Malça, J., & Freire, F. (2010). Unceratinty Analysis in Biofuel Systems: An Application to Life Cycle of Rapeseed Oil. *Journal of Industrial Ecology, 14*(2), 322-334. doi:10.1111/j.1530-9290.2010.00227.x

Mallia, E., & Lewis, G. (2013). Life cycle greenhouse gas emissions of electricity generation in the province of Ontario, Canada. *International Journal of Life Cycle Assessment, 18*(2), 377-391. doi:10.1007/s11367-012-0501-0

Malveda, M., & Yokose, K. (2012). *Biodegradable Polymers - Abstract.* Englewood, U.S.: IHS Chemicals. Retrieved from http://chemical.ihs.com/CEH/Public/Reports/580.0280

Mang, M. (2012, May). Bio-based succinic acid: Transforming the chemicals industry. *Speciality Chemicals Magazine, 32*(05).

Mata, T. M., Smith, R. L., Young, D. M., & Costa, C. A. (2003). Evaluating the Environmental Friendliness, Economics and Energy Efficiency of Chemical Processes; Heat Integration. *Clean Technologies and Environmental Policy, 5*(3), 302-309. doi:10.1007/s10098-003-0203-1

Mattila, T., Kujanpää, M., Dahlbo, H., Soukko, R., & Myllymaa, T. (2011). Uncertainty and Sensitivity in the Carbon Footprinting of Shopping Bags. *Journal of Industrial Ecology, 15*(2), 217-227.

MatWeb. (2012, June 12). *Overview of materials for Polypropylene with 30% Glass Fiber Filler.* Retrieved from MatWeb: http://matweb.com/search/datasheet.aspx?matguid=e2a28c79390342e4904502d582e69c62&ckck=1

McCurdy, D. (2010). “Our Global Road Transport Priority: Reducing CO2 Emissions through an Integrated Approach”. *Ministerial Conference on Global Environment and Energy in Transport.* Rome, Italy.

McKone, T., Nazaroff, W., Berck, P., Auffhammer, M., Lipman, T., Torn, M., . . . Horvath, A. (2011). Frand Challenges for Life-Cycle Assessment of Biofuels. *Environmental Science and Technology, 45*(5), 1751-1756. doi:10.1021/es103579c

McLaughlin, S., & Walsh, M. (1998). Evaluating environmental consequences of producing herbaceous crops for bioenergy. *Biomass and Bioenergy, 14*(4), 317-324. doi:10.1016/S0961-9534(97)10066-6

McLaughlin, S., Bouton, J., Bransby, D., Conger, B., Ocumpaugh, W., Parrish, D., . . . Wullschleger, S. (1999). Developing Switchgrass as a Bioenergy Crop. In J. Janick (Ed.), *Proceedings of the Fourth National Symposium New Crops and New Uses: Biodiversity and Agricultural Sustainability* (pp. 282-299). Alexandria, VA.: ASHS Press. Retrieved from http://www.hort.purdue.edu/newcrop/proceedings1999/pdf/v4-282.pdf

Miller, S. A., Landis, A. E., & Theis, T. L. (2007). Feature: Environmental Trade-offs of Biobased Products. *Environmental Science and Technology, 41*(15), 5176-5182. doi:10.1021/es072581z

Miller, S., Chamberlain, J., Alfaro, J., & Sharp, B. (2011). Assessing The Non-Carbon Impacts Of An Emerging Bioenergy Industry. *6th International Conference of the International Society for Industrial Ecology (ISIE) Proceedings* (p. Abstract # 822). Berkeley, CA: International Society for Industrial Ecology (ISIE).

Ministry of Justice. (2010). *Passanger Automobile and Light Truck Greenhouse Gas Emission Regulations.* Ottawa: Ministry of Justice. Retrieved from http://laws-lois.justice.gc.ca/PDF/SOR-2010-201.pdf

Mittal, V. (2011). Bio-nanocomposites: Future High-value Materials. In V. Mittal (Ed.), *Bio-nanocomposites: Future High-value Materials* (pp. 1-27). New York, U.S.: Oxford University Press.

Modahl, I. S., Lyng, K.-A., Raadal, H. L., & Askham, C. (2012). Cumulative Energy Demand - How valid and Comprehensive are our CED Practices Today? *Poster presented at the SETAC Europe 22nd annual meeting / 6th SETAC World Congress.* Berlin, Tyskland: SETAC Europe.

Mohanty, A. K., Misra, M., & Drzal, L. T. (Eds.). (2005). *Natural Fibers, Biopolymers, and Biocomposites.* Boca Raton, FL: Taylor & Francis Group, LLC.

Mohanty, A., Misra, M., & Sahoo, S. (2010, June 01). *Patent No. WO/2010/139056.* USA.

Mohanty, A.-K., Misra, M., & Drzal, L.-T. (2002). Sustainable Bio-Composites from Renewable Resources: Opportunities and Challenges in the Green Materials World. *Journal of Polymers and the Environment, 10*(1/2), 19-26.

Molnar, D., & Morgan, A. J. (2001). *Defining Sustainability, Sustainable Development and Sustainable Communities: A Working Paper for the Sustainable Toronto Project.* Toronto, Canada: Sustainable Toronto Project.

Morais, S., Martins, A., & Mata, T. (2010). Comparison of allocation approaches in soybean biodiesel life cycle assessment. *Journal of the Energy Institute, 38*(1), 48-55. doi:10.1179/014426010X12592427712073

Morrison-Saunders, A., & Pope, J. (2013). Conceptualising and Managing Trade-Offs in Sustainability Assessment. *Environmental Impact Assessment Review, 38*, 54-63. doi:10.1016/j.eiar.2012.06.003

Mortimer, N., Elsayed, M., & Evans, A. (2007). *Environmental Assessment Tool for Biomaterials: Environmental Assessment Primer. Final Version.* Sheffield: North Energy Association, Department of Environment, Food and Rural Affairs. Retrieved from http://www.nnfcc.co.uk/tools/environmental-assessment-primer

Moussa, H., Young, S. B., & Gerand, Y. (2012). Polybutylene Succinate Life Cycle Assessment Variations and Variables: Environmental and Sustainability Impact Measurement. *American Institute of Chemical Engineers (AIChE), 2012 International Congress on Energy*. Pittsburgh: American Institute of Chemical Engineers (AIChE).

Myriant. (2012). *Bio-Succinic Acid: A high Performing Green Chemical without a Green Premiun Price* . Retrieved from Myriant: http://www.myriant.com/media/press-kit-files/Myriant-BioSFactSheet.pdf

Myriant Corporation. (2011). *Form S-1 Registeration Statement under The Securities Act of 1933: Myriant Corporation.* Washington, D.C.: United States Securities and Exchange Commission. Retrieved from http://www.sec.gov/Archives/edgar/data/1485026/000095012311054952/b86680sv1.htm

Nabi Saheb, D., & Jog, J. (1999). Natural Fiber Polymer Composites: A Review. *Advances in Polymer Technology*, *18*(4), 351-363.

Nakajima-Kambe, T., Ichihashi, F., Matsuzoe, R., Kato, S., & Shintani, N. (2009). Degradation of aliphatic-aromatic copolyesters by bacteria that can degrade aliphatic polyesters. *Polymer Degradation and Stability*, *94*(11), 1901-1905.

Narayan, R. (2009). Biodegradability Sorting through Facts and Claims. *Bioplastics Magazine, 4*(1), 28-31. Retrieved from http://www.bpiworld.org/resources/Documents/Narayan%20Bioplastics%20Article%202009.pdf

Narayan, R., & Pettigrew, C. A. (1999, December). ASTM Standards Help Define and Grow a New Biodegradable Plastics Industry. *ASTM Standardization News*, pp. 36-42. Retrieved from http://www.ftc.gov/os/comments/greenguiderevisions/00181-56737.pdf

Nebel, B., Alcorn, A., & Wittstock, B. (2011). *Life Cycle Assessment: Adopting and adapting overseas LCA data and methodologies for building materials in New Zealand.* Rotorua, New Zealand: The Ministry of Agriculture and Forestry.

Nexant. (2012). *Prospectus Is Bio-Butanediol Here to Stay?* New York: Nexant, Inc.

Nolan-Itu. (2002). *Biodegradable Plastics - Development and Environmental Impacts.* Melbourne: Environment Australia. Retrieved from http://www.europeanplasticfilms.eu/docs/AustralianReportonBiodegradablePlastics.pdf

Nova Institute. (2012). *Bio-based Polymers in the World. Capacities, Production and Applications: Status Quo and Trends towards 2020.* Hürth, Germany: Nova Institute for Ecology and Innovation.

Nuss, P., & Gardner, K. H. (2013). Attributional life cycle assessment (ALCA) of polyitaconic acid production from northeast US softwood biomass. *International Journal of Life Cycle Assessment*, *18*(3), 606-612. doi:10.1007/s11367-012-0511-y

Nygren, J., & Antikainen, R. (2010). *Use of life cycle assessment (LCA) in global companies.* Helsinki: Finnish Environmental Institute - Centre for Sustainable Consumption and Production. Retrieved from http://www.environment.fi/publications

Oak Ridge National Laboratory. (2007). *Alternative, Renewable and Novel Feedstocks for Producing Chemicals.* Washington, DC: U.S. Department of Energy.

Odeh, N. A., & Cockerill, T. T. (2008). Life cycle GHG assessment of fossil fuel power plants with carbon capture and storage. *Energy Policy, 36*(1), 367-380. doi:10.1016/j.enpol.2007.09.026

OECD. (2011). *Industrial Biotechnology and Climate Change: Opportunities and Challenges.* Paris, France: OECD Directorate for Science, Technology and Industry (DSTI). Retrieved from http://www.oecd.org/sti/biotech/49024032.pdf

OMAFRA. (2011). *Ontario's Bio Advantage. Programs for Ontario's Bioproducts Sectors - Bioenergy, Biomaterials and Biochemicals.* Retrieved from OMAFRA: http://www.omafra.gov.on.ca/

Ontario Regulation 463/10 made under the Environmental Protection Act: Ozone Depletion Substances and Other Halocarbons. (2010). *e-Laws.* Retrieved from http://www.e-laws.gov.on.ca/html/source/regs/english/2010/elaws_src_regs_r10463_e.htm

Patel, M., Crank, M., Domburg, V., Hermann, B., Roes, L., Husing, B., . . . Recchia, E. (2006). *Medium and Long-term Opportunities and Risks of the Biotechnological Production of Bulk Chemicals from Renewable Resources - The Potential of White Biotechnology - The BREW Project.* Utrecht: Utrecht University under the European Commission's GROWTH Programme (DG Research). Retrieved from http://brew.geo.uu.nl/BREW_Final_Report_September_2006.pdf

Patt, J. J., & Banholzer, W. F. (2009). Improving Energy Efficiency in the Chemical Industry. *The Bridge, 39*(2), pp. 16-21. Retrieved from http://www.nae.edu/

Pawelzik, P., Carus, M., Hotchkiss, J., Narayan, R., Selke, S., Wellisch, M., . . . Patel, M. (2013). Critical aspects in the life cycle assessment (LCA) of bio-based materials – Reviewing methodologies and deriving recommendations. *Resources, Conservation and Recycling, 73*, 211-228. doi:10.1016/j.resconrec.2013.02.006

Pelletier, N., Audsley, E., Brodt, S., Garnett, T., Henriksson, P., Kendall, A., . . . Troell, M. (2011). Energy Intensity of Agriculture and Food Systems. *The Annual Review of Environmental and Resources, 36*, 223-246.

Pembina Institute. (2012, June 14). *New report examines impacts of coal-fired power in Canada*. Retrieved from Pembina Institute: http://www.pembina.org

Pfister, D. P., & Larock, R. C. (2012). Green Composites Using Switchgrass as a Reinforcement for a Conjugated Linseed Oil-Based Resin. *Journal of Applied Polymer Science, 127*(3), 1921-1928. doi:10.1002/APP.37536

Pfister, S., Koehler, A., & Hellweg, S. (2009). Assessing the Enviornmental Impacts of Freshwater Consumption in LCA. *Environmental Science and Technology, 43*(11), 4098-4014. doi:10.1021/es802423e

Pilz, H., Brandt, B., & Fehringer, R. (2010). *The Impact of Plastics on Life Cycle Energy Consumption and Greenhouse Gas Emissions in Europe. Summary report.* Vienna, Austria: Association of Plastics Manufacturers.

Pimentel, D., Hepperly, P., Hanson, J., Douds, D., & Seidel, R. (2005). Environmental, Energietic, and Economic Comparisons of Organic and Conventional Farming Systems. *BioScience, 55*(7), 573-582. doi:10.1641/0006-3568(2005)055[0573:EEAECO]2.0.CO;2

PlasticsEurope. (2007). *The Compelling Facts about Plastics - an analysis of plastics production, demand and recovery for 2005 in Europe.* Brusseks, Belgium: PlasticsEurope.

PlasticsEurope. (2008). *Environmental Product Declaration of the European Plastics Manufacturers. Polypropylene (PP).* Brussels, Belgium: Association of Plastics Manufacturers (PlasticsEurope).

PlasticsEurope. (2010). *Plastics - The Facts 2010: An analysis of European Plastics Production, demand and recovery for 2009.* Brussels, Belgium: PlasticsEurope.

PlasticsEurope. (2011). *Plastics - the Facts 2011: An analysis of European Plastics Production, Demand and Recovery 2010.* Brussels, Belgium: PlasticsEurope.

PlasticsEurope. (2012a, February 28). *Use of Plastics?: Transportation*. Retrieved from PlasticsEurope.org: http://www.plasticseurope.org/use-of-plastics/transportation.aspx

PlasticsEurope. (2012b, March 22). *What is Plastic?: Innovation*. Retrieved from PlasticsEurope.org: http://www.plasticseurope.org/what-is-plastic/innovation.aspx

Pollution Probe and BIOCAP Canada. (2004). *Primer on Bioproducts.* Toronto and Kingston: Pollution Probe and BIOCAP Canada. Retrieved from http://www.biocap.ca/images/pdfs/BioproductsPrimerE.pdf

Pope, J., Annandale, D., & Morrison-Saunders, A. (2004). Conceptualising sustainability assessment. *Environmental Impact Assessment Review, 24*(6), 595-616. doi:10.1016/j.eiar.2004.03.001

Prairie Lands Biomass LLC. (2012, April 09). *Farm-Related Local Environmental Benefits*. Retrieved from Iowa Switchgrass: http://www.iowaswitchgrass.com/benefits~onfarmbenefits.html

PRé Consultants. (2010). *Introduction to LCA with SimaPro 7.* The Netherlands: PRé Consultants.

PRé Consultants. (2012). *Life Cycle-Based Sustainability Standards & Guidelines.* Retrieved from PRé Sustainability: http://www.pre-sustainability.com/download/Pre_Whitepaper.pdf

Puri, P., Compston, P., & Pantano, V. (2008). Life Cycle Assessment of Australian Automotive Door Skin. *Int'l Journal of Life Cycle Assessment, 14*(5), 420-428. doi:10.1007/s11367-009-0103-7

Qin, X., Mohan, T., El-Halwagi, M., Cornforth, G., & McCarl, B. (2006). Switchgrass as an alternate feedstock for power generation: an integrated environmental, energy and economic life-cycle assessment. *Clean Technologies and Environmental Policy, 8*(4), 233-249.

Qu, J.-p., Tan, B., Ferg, Y.-h., & Hu, S.-x. (2011). Mechanical Properties of Poly(Butylene Succinate) Reinforced with Continuously Steam-Exploded Cotton Stalk Bast. *Polymer-Plastics Technology and Engineering, 50*(14), 1405-1411. doi:10.1080/03602559.2011.593081

Quantis. (2009). *What is Life Cycle Assessment?* Retrieved from Quantis International: http://www.quantis-intl.com/

Randolph, J., & Masters, G. M. (2008). *Energy for Sustainability:Technology, Planning, and Policy.* Washington, D.C.: Island Press.

Rattner, A. S., & Garimella, S. (2011). Energy harvesting, reuse and upgrade to reduce primary energy usage in the USA. *Energy, 36*(10), 6172-6183. doi:10.1016/j.energy.2011.07.047

Ray, S. S., Okamoto, K., Maiti, P., & Okamoto, M. (2002). New Poly(butylene succinate)/Layered Silicate Nanocomposite. 1: Preparation and Mechanical Properties. *Journal of Nanoscience and Nanotechnology, 2*(2), 171-176. doi:10.1166/jnn.2002.086

Raynolds, M. A., Checkel, M., & Fraser, R. A. (1998). Uncertainty, Sensitivity and Data Quality Assessment for Life Cycle Value Assessment (LCVA). *International Congress & Exposition, Design and Manufacture for the Environment* (pp. 207-215). Detroit, Michigan, USA: SAE Special Publications, SP-1342. doi:10.4271/980479

Reap, J., Roman, F., Duncan, S., & Bras, B. (2008). A survey of unresolved problems in life cycle assessment. Part 1: goal and scope and inventory analysis. *The International Journal of Life Cycle Assessment, 13*(4), 290-300. doi:10.1007/s11367-008-0008-x

Reddy, N., & Yang, Y. (2007). Natural Cellulose Fibers From Switchgrass With Tensile Properties Similar to Cotton and Linen. *Biotechnology and Bioengineering, 97*(5), 1021-1027. doi:10.1002/bit.21330

Regional Public Works Commissioners of Ontario (RPWCO). (2009, June 19). *RPWCO Position Paper: Biodegradable Plastic Products and Packaging.* Retrieved from City of Toronto: http://www.toronto.ca/garbage/pdf/rpwco_bio_plastic_position.pdf

Reijnders, L. (2006). Conditions for the sustainability of biomass based fuel use. *Energy Policy, 34*(7), 863-876. doi:10.1016/j.enpol.2004.09.001

Rinehart, L. (2006). Switchgrass as a Bioenergy Crop. (P. Discoll, Ed.) *ATTRA - National Sustainable Agriculture Information Service*. Retrieved from www.attra.ncat.org/attra-pub/PDF/switchgrass.pdf

RLHudson. (2011). *Thermoset vs. Thermoplastic Materials*. Retrieved from RLHudson: http://www.rlhudson.com/publications/techfiles/thermo.htm

Rochette, P., Worth, D. E., Lemke, R. L., McConkey, B. G., Pennock, D. J., Wagber-Riddle, C., & Desjardins, R. L. (2008). Estimation of N2O emissions from agricultural soils in Canada . I . Development of a country-specific. *Canadian Journal of Soil Science, 88*(5), 641-654. doi:10.4141/CJSS07025

Rochman, C. M., Browne, M.-A., Halpern, B. S., Hentschel, B. T., Hoh, E., Karapanagioti, H. K., . . . Thompson, R. C. (2013). Policy: Classify plastic waste as hazardous. *Nature, 494*(7436), 169-171. Retrieved from http://www.nature.com/nature/journal/v494/n7436/pdf/494169a.pdf

Rossi, V., & Margni, M. (2010). TRACI. In R. Hischier, & B. Weidema (Eds.), *Implementation of Life Cycle Impact Assessment Methods - ecoinvent report No. 3* (pp. 149-155). St. Gallen: Swiss Centre for Life Cycle Inventories - ecoinvent centre.

Ryan, P. G., Moore, C. J., van Franeker, J. A., & Moloney, C. L. (2009). Monitoring the abundance of plastic debris in the marine environment. *Philosophical Transactions of The Royal Society B, 364*(1526), 1999-2012. doi:10.1098/rstb.2008.0207

Sadler, B. (1999). framework for environmental sustainability assessment and assurance. In J. Petts (Ed.), *Handbook of Environmental Impact Assessment, Vol. 1* (pp. 12-32). Oxford: Blackwell Science.

Samson, R. (2007). *Switchgrass Production in Ontario: A Management Guide.* Retrieved from Resource Efficient Agricultural Production (REAP): http://www.reap-canada.com/online_library/feedstock_biomass/2007%20SG%20production%20guide-FINAL.pdf

Samson, R., Girouard, P., Omielan, J., & Henning, J. (1993). Integrated Production of Warm Season Grasses and Agroforestry for Low Cost Biomass Production. *First Biomass Conference of the American's: Energy, Environment, Agriculture, and Industry* (pp. 235-247). Burlington, Vermont: National Renewable Energy Laboratory. Retrieved from . http://www.reap-canada.com/online_library/feedstock_biomass/30%20Integrated%20Production.pdf

Samson, R., Ho Lem, C., & Bailey, S. (2012). The Potential pf Switchgrass as a Commercial Pellet Fuel in Ontario. *6th Annual Growing the Margins: Rural Green Energy Conference and Exhibition.* London, Canada: Resource Efficient Agriculture Production (REAP).

Sanderson, M., Reed, R., McLaughlin, S., Wullschleger, S., Conger, B., Parrish, D., . . . Tischler, C. (1996). Switchgrass as a sustainable bioenergy crop. *Bioresource Technology, 56*(1), 83-93. doi:10.1016/0960-8524(95)00176-X

Santulli, C., Janssen, M., & Jeronimidis, G. (2005). Partial replacement of E-glass fibers with flax fibers in composites and effect on falling weight impact performance. *Journal of Materials Science, 40*(13), 3581-3585. Retrieved from http://www.carlosantulli.net/santulli_janssen_jeronimidis.pdf

Schindler, D. W. (2012). The Dilemma of Controlling Cultural Eutrophication of Lakes. *Proceedings of The Royal Society B*, 1-12. doi:10.1098/rspb.2012.1032

Scientific Applications International Corporation [SAIC]. (2006). *Life Cycle Assessment: Principles and Practices.* Cincinnati, U.S.: National Risk Management Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency. Retrieved from http://www.epa.gov/nrmrl/std/lca/pdfs/chapter1_frontmatter_lca101.pdf

Scott, G., Lemaire, J., Jakubowics, I., & Ojeda, T. (2011). *Briefing Note on Biodegradable Plastics.* London, England: Oxo-Biodegradable Plastics Association. Retrieved from http://www.biodeg.org/files/uploaded/biodeg/briefing_notes/OPA_Briefing_Note_15.6.12(3)(1).pdf

Selman , M., & Greenhalgh, S. (2009, September). *Eutrophication: Policies, Actions, and Strategies to Address Nutrient Pollution.* Washington, DC: World Resources Institute.

Shastri, Y. N., Hansen, A. C., Rodriguez, L. F., & Ting, K. (2012). Switchgrass - practical issues in developing a fuel crop. *CAB Reviews, 7*(037). doi:10.1 079/PAVSNNR20127037

Sheate, W. R., Dagg, S., Richardson, J., Aschemann, R., Palerm, J., & Steen, U. (2003). Integrating the environment into strategic decision-making: conceptualizing policy SEA. *European Environment, 13*(1), 1-18. doi:10.1002/eet.305

Shih, Y.-F., & Jeng, R.-J. (2011). Biodegradable Poly(butylene succinate) / multi-walled Carbon Nanotube Nanocomposites. In V. Mittal (Ed.), *Nanocomposites with Biodegradable Polymers: Synthesis, Properties, and Future Perspectives* (pp. 101-122). New York, U.S.: Oxford University Press.

Showa Denko. (2010, June 25). *Bionolle - The pioneer in biodegradable*. Retrieved from Showa Denko Europe GmbH: http://www.showa-denko.com/index.php?id=6&L=1&tx_ttnews%5Btt_news%5D=126&cHash=7f1b840168

Showa Denko. (2012). *Bionolle™*. Retrieved from Showa Denko Singapore: http://showadenko.us/product/bionolle.php

Siler-Evans, K., Azevedo, I. L., & Morgan, M. F. (2012). Marginal Emissions Factors for the U.S. Electricity System. *Environmental Science and Technology, 46*(9), 4742-4748. doi:10.1021/es300145v

Smidt, M. (2011). A sustainable supply of succinic acid. *Euro Biotech News, 10*(11-12), 70-71.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., . . . Sirotenko, O. (2007). *Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer (eds)]*. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Society of Plastics Engineers [SPE]. (2001). *Thermoplastic and Thermosetting Polymers.* Retrieved from Mid-Michigan Society of Plastics Engineers: http://www.midmichiganspe.org/pdfs/documents/thermoplasticandthermoset.pdf

Society of the Plastics Industry [SPI]. (2012, April 02). *Definitions of Resins*. Retrieved from PlasticsIndustry.org: http://www.plasticsindustry.org/AboutPlastics/content.cfm?ItemNumber=656&navItemNumber=1128

Society of the Plastics Industry [SPI]. (2013, November 06). *History of Plastics*. Retrieved from PlasticsIndustry.org: http://www.plasticsindustry.org/AboutPlastics/content.cfm?ItemNumber=670

Song, J., Murphy, R., Narayan, R., & Davies, G. (2009). Biodegradable and compostable alternatives to conventional plastics. *Philosophical Transactions of The Royal Society B, 364*(1526), 2127-2139. doi:10.1098/rstb.2008.0289

Sparling, D., Laughland, P., & Mitura, V. (2009). *Vista on the Agri-Food Industry and the Farm Community - The Canadian Bioproducts Industry, 2003 and 2006.* Ottawa, Canada: Statistics Canada - Agriculture Division. Retrieved from http://www.statcan.gc.ca/pub/21-004-x/21-004-x2009001-eng.pdf

Stewardship Ontario. (2005). *Ontario Blue Box Market Overview: 2002/2003 Blue Box Materials Generated and Municipally Marketed - Mass Balance Report.* Toronto, Canada: Stewardship Ontario.

Stockholm Environment Institute. (2007). *Industrial Biotechnology and Biomass Utilisation - Prospects and Challenges for the Developing World.* Vienna: United Nations Industrial Development Organization (UNIDO).

Suddell, B. C. (2008). Industrial Fibers: Recent and Current Developments. *The Symposs on Natural Fibers* (p. Technical Paper No. 56). Rome: FAO.

Suddell, B., Evans, W., Isaac, D., & Crosky, A. (2002). A Survey into the Application of Natural Fiber Composites in the Automotive Industry. *Fourth International Symposium in Natural Polymers and Composites (ISNaPol)*, (pp. 455-461). Sao Pedro, Brazil.

Suh, S., Leighton, M., Tomer, S., & Chen, C. (2013). Interoperability between ecoinvent ver. 3 and US LCI database: a case study. *The International Journal of Life Cycle Assessment*, *18*(6). doi:10.1007/s11367-013-0592-2

Sutter, J. (2007). *Life Cycle Inventories of Petrochemical Solvents Data v2.0. ecoinvent report No. 22.* Dübendorf, Uster - Switzerland : ecoinvent centre - Swiss Centre for Life Cycle Inventories.

Tabone, M. D., Cregg, J. J., Beckman, E. J., & Landis, A. E. (2010). Sustainability Metrics: Life Cycle Assessment and Green Design in Polymers. *Environmental Science & Technology, 44*(21), 8264-8269. doi:10.1021/es101640n

Taka, T., Takiyama, E., & Yasukawa, Y. (1994). *Patent No. 5324794A.* United States.

Takiyama, E., Fujimaki, T., Seki, S., Hokari, T., & Hatano, Y. (1994). *Patent No. 5,310,782.* United States.

Takiyama, E., Hatano, Y., Fujimaki, T., Seki, S., Hokari, T., Hosogane, T., & Harigai, N. (1995). *Patent No. 5436056A.* United States.

Tantatherdtam, R., Tran, T., Chotineeranat, S., Lee, B.-H., Lee, S.-n., Sriroth, K., & Kim, H.-J. (2009). Preparation and Characterization of Cassava Fiber-Based Polypropylene and Polybutylebe Succinate Composites. *Kasetsart Journal: Natural Science*, *43*(5), 245-251.

Terasawa, I., Tsuneoka, K., Tamura, A., & Tanase, M. (2008). Development of Plant-Based Plastics Technology, 'Green Plastic'. *Mitsubishi Motors - Technical Review*(20), pp. 91-96.

The Association of German Engineers [VDI]. (2012). *VDI Guideline: VDI 4600. Cumulative energy demand (KEA) - Terms, definitions, methods of calculation.* Düsseldorf: Verein Deutscher Ingenieure (VDI). Retrieved from http://www.vdi.eu/engineering/vdi-guidelines/vdi-guidelines-details/rili/93594/

The Henry Ford. (2003). *Life of Henry Ford*. Retrieved from thehenryford.org: http://www.hfmgv.org/exhibits/hf/

The Robinsion Library. (2012, January 17). *Alexander Parks*. Retrieved from The Robinsion Library: http://www.robinsonlibrary.com/technology/technology/biography/parkes.htm

Theunissen, L. (2012). Biosuccinium™, enabling sustainable polymer solutions. *Biopolymer Symposium* . San Antonio, Texas. Retrieved from http://www.assobioplastiche.org/wp-content/uploads/2012/10/Theunissen-Lawrence.pdf

Thompson, R. C., Moore, C. J., vom Saal, F. S., & Swan, S. H. (2009). Plastics, the environment and human health: current cosensus and future trends. *Philosophical Transactions of The Royal Society B, 364*(1526), 2153-2166. doi:10.1098/rstb.2009.0053

Tokiwa, Y., Calabia, B. P., Ugwa, C. U., & Aiba, S. (2009). Biodegradability of Plastics. *International Journal of Molecular Sciences, 10*(9), 3722-3742. doi:10.3390/ijms10093722

Torrez, V. C., Johnson, P. J., & Boe, A. (2013, March). Infestation Rates and Tiller Morphology Effects by the Switchgrass Moth on Six Cultivars of Switchgrass. *BioEnergy Research*. doi:10.1007/s12155-013-9306-7

Toyota Motor Corporation. (Retrieved 2012, March 13). *Design for Recycling (Based on Eco-VAS) - Use of Renewable Resources - Eco Plastics*. Retrieved from Toyota Motor Corporation Global: http://www.toyota-global.com/sustainability/environmental_responsibility/automobile_recycling/design_for_recycling/use_of_renewable_resources.html

Tran, T., Lee, B.-H., Yang, H.-S., Chotineeranat, S., Sriroth, K., & Kim, H.-J. (2011). Use of starch granules melting to control the properties of bio-flour filled polypropylene and poly(butylene succinate) composites: Mechanical properties. *Starchh Journal, 63*(10), 637-648. doi:10.1002/star.201100024

Tsai, F.-J., & Wertheim, B. C. (2001). *Patent No. US6225388 B1.* U.S.

Tsiropoulos, I., Cok, B., & Patel, M. K. (2013). Energy and greenhouse gas assessment of European glucose production from corn e a multiple allocation approach for a key ingredient of the bio-based economy. *Journal of Cleaner Production, 43*, 182-190. doi:10.1016/j.jclepro.2012.12.035

Tukker, A., & Janseb, B. (2006). Environmental Impacts of Products: A Detailed Review of Studies. *Journal of Industrial Ecology, 10*(3), 159-182.

Tulbure, M. G., Wimberly, M. C., & Owens, V. N. (2012). Response of switchgrass yield to future climate change. *Environmental Research Letters, 7*(4). doi:10.1088/1748-9326/7/4/045903

Tullo, A. H. (2008). Growing Plastics. *Chemical & Engineering News, 86*(39), 21-25. doi:10.1021/cen-v086n039.p021

Twarowska-Schmidt, K., & Tomaszewski , W. (2008). Evaluation of the Suitability of Selected Aliphatic Polyester Blends for Biodegradable Fibrous Materials with Improved Elasticity. *Fibers & Textiles in Eastern Europe, 3*(68), 9-13.

U.S. Enery Information Administration [U.S. EIA]. (2013). *Annual Energy Outlook 2013 with Projection to 2040.* Washington, DC: U.S. Department of Energy (U.S. DOE). Retrieved from www.eia.gov

UNEP/SETAC Life Cycle Initiative. (2011). *Toward a Life Cycle Sustainability Assessment.* Paris, France: UNEP and SETAC Life Cycle Initiative.

UNFCCC. (2009). *National greenhouse gas inventory data for the period 1990-2007.* Copenhagen: UNFCCC - United Nation Frame Work Convention on Climate Change. Retrieved from http://unfccc.int/resource/docs/2009/sbi/eng/12.pdf

United States Department of Agriculture [U.S. DOA]. (2008). *U.S. Biobased Products Market Potential and Projections Through 2025.* Washington, DC: United States Department of Agriculture.

United States Department of Energy [U.S. DOE]. (2004). *Technology Roadmap. Energy Loss Reduction and Recovery in Industrial Energy Systems.* Washington, DC: United States Department of Energy. Retrieved from http://www1.eere.energy.gov/

United States Department of Energy [U.S. DOE]. (2009). *U.S. Life Cycle Inventory Database Roadmap.* Washington, D.C: National Renewable Energy Laboratory (NREL). U.S. Department of Energy. Retrieved from www.nerl.gov

United States Energy Information Administration [U.S. EIA]. (2012, October 02). *Electricity. 2011: EIA-923*. Retrieved from United States Energy Information Administration: http://www.eia.gov/electricity/data/eia923/

United States Environmental Protection Agency [U.S. EPA]. (1995). Chapter 11: Mineral Products Industry. Glass Fibre Manufacturing. In *AP 42, Fifth Edition, Volume I.* Washington, DC: United States Environmental Protection Agency. Retrieved from http://www.epa.gov

United States Environmental Protection Agency [U.S. EPA]. (2010, December 15). *Ozone Layer Protection Glossary*. Retrieved from U.S. Environmental Protection Agency: http://www.epa.gov/ozone/defns.html

United States Environmental Protection Agency [U.S. EPA]. (2011a). *Code of Federal Regulations - Title 40 CFR 86.1818-12.* Retrieved from Legal Information Institute - Cornell University Law School: http://www.law.cornell.edu/cfr/text/40/86.1818-12

United States Environmental Protection Agency [U.S. EPA]. (2011b). *Control of Air Pollution from Motor Vehicles: Tier 3 Motor Vehicle Emission and Fuel Standards.* Washington, DC: Environmental Protection Agency. Retrieved from http://www.epa.gov/otaq/documents/tier3/tier3-nprm-20130329.pdf

United States Environmental Protection Agency & National Highway Traffic Safety Administration . (2012). *Federal Register - Rules and Regulations: Vol. 77, No. 199.* Washington, D.C.: Environmental Protection Agency and Department of Transportation .

UT Center for Clean Products . (2009). *Life-Cycle Assessment of Cladding Products. A comparison of aluminum, brick, granite, limestoes, and precast concrete.* Knoxville, TN: Univerity of Tennessee - Center for Clean Products.

van Dam, J. E. (2008). Environmental benefits of natural fiber production and use. *The Symposium on Natural Fibres* (p. Technical Paper No. 56). Rome: FAO. Retrieved from ftp://ftp.fao.org/docrep/fao/011/i0709e/i0709e.pdf

Van Den Oever, M., Elbersen, H., Keijsers, E., Gosselink, R., & De Klerk-Engels, B. (2003). Switchgrass (Panicum virgatum L.) as a reinforcing fibre in polypropylene composites. *Journal of Materials Science, 38*(18), 3697-3707. doi:10.1023/A:1025959627950

Vivoda, V. (2011, December 14). Japan's energy security predicament in the aftermath of Fukushima Disaster. *Journal of Energy Security, 4*(4), 1-8. Retrieved from Singapore International Energy Week: http://www.ensec.org/

Vogel , K. P. (2004). Switchgrass. In L. E. Moser, B. L. Burson, & L. E. Sollenberger (Eds.), *Warm-season (C4) grasses. Agronomy 45* (pp. 561–588). Madison: the American Society of Agronomy.

von Blottnitz, H., & Curran, M. (2007). A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective. *Journal of Cleaner Production, 15*(7), 607–619. doi:10.1016/j.jclepro.2006.03.002

Wallach, P. A. (2012, October). U.S. Regulations of Greenhouse Gas Emissions. *Governance Studies at Brookings*. Retrieved from http://www.brookings.edu/~/media/research/files/papers/2012/10/26%20climate%20change%20wallach/26%20climate%20change%20wallach.pdf

Wan, C., & Chen, B. (2013). Reinforcement of biodegradable poly(butylene succinate) with low loadings of graphene oxide. *Journal of Applied Polymer Science, 127*(6), 5094-5099. doi:10.1002/app.38136

Wang, J., Zheng, L., Li, C., Zhu, W., Zhang, D., Xiao, Y., & Guan, G. (2012). Fully biodegradable blends of poly(butylene succinate) and poly(butylene carbonate): Miscibility, thermal properties, crystallization behavior and mechanical properties. *Polymer Testing, 31*(1), 39-45. doi:10.1016j.polymertesting.2011.09.005

Wang, M., Saricks, C., & Santini, D. (1999). *Effects of Fuel Ethanol Use on Fuel-Cycle Energy and Greenhouse Gas Emissions.* Chicago: Center for Transporation Research - Argonne National Laboratory, United States Department of Energy. Retrieved from http://www.transportation.anl.gov/pdfs/TA/58.pdf

Weidema, B. P. (2000). Avoiding Co-Product Allocation in Life Cycle Assessment. *Journal of Industrial Ecology*, *43*, 11-33. doi:10.1162/108819800300106366

Weidema, B. P., & Wesnæs, M. S. (1996). Data quality management for life cycle inventories—an example of using data quality indicators. *Journal of Cleaner Production*, *4*(3-4), 167-174. doi:10.1016/S0959-6526(96)00043-1

Weidema, B., Bauer, C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., . . . Wernet, G. (2013). *Overview and Methodology. Data Quality Guideline for the Ecoinvent Database version 3. Ecoinvent Report 1(v3).* St. Gallen: The Ecoinvent Centre.

Weiss, M., Haufe, J., Carus, M., Brandão, M., Bringezu, S., Hermann, B., & Patel, M. K. (2012). A Review of the Environmental Impacts of Biobased Materials. *Industrial Ecology, 16*(Supplement s1), S169-S181. doi:10.1111/j.1530-9290.2012.00468.x

Wenzel, H. (1998). Application dependency of LCA methodology: Key variables and their mode of influencing the method. *The International Journal of Life Cycle Assessment, 3*(5), 281-288.

Wenzel, H., Villanueva, A., Willum, O., Bey, N., Vrgoc, M., & Frees, N. (2006). The significance of boundary conditions and assumptions in the environmental life cycle assessment of waste management strategies. An analytical review of existing studies. *First NorLCA Symposium.* Lund, Sweden: Nordic Life Cycle Association (NorLCA).

Westman, M., Fifield, L., Simmons, K., Laddha, S., & Kafentzis, T. (2010). *Natural Fiber Composites: A Review.* Washington: Pacific Northwest National Laboratory. Retrieved from http://www.pnl.gov/main/publications/external/technical_reports/PNNL-19220.pdf

Wettestad , J. (2008). *EU Energy-intensive Industries and Emissions Trading. Losers becoming Winners? a CANES Working Paper.* Lysaker, Norway: Fridtjof Nansen Institute. Retrieved from www.fni.no

Williams, G. I., & Wool, R. P. (2000). Composites from Natural Fibers and Soy Oil Resins. *Applied Composite Materials, 7*(5-6), 421-432. doi:10.1023/A:1026583404899

Wotzel, K., Wirth, R., & Flake, M. (1999). Life cycle studies on hemp fibre reinforced components and ABS for automotive parts. *Die Angewandte Makromolekulare Chemie, 272*(1), 121-127.

Xu, J., & Guo, B.-H. (2010). Poly(butylene succinate) and its copolymers: Research, development and industrialization. *Biotechnology Journal, 5*(11), 1149-1163. doi:10.1002/biot.201000136

Xu, X., Jayaraman, K., Morin, C., & Pecqueux, N. (2008). Life Cycle Assessment of Wood-Fibre-Reinforced Polypropylene Composites. *Journal of Materials Processing Technology, 198*(1-3), 168-177. doi:10.1016/j.jmatprotec.2007.06.087

Yu, J., & Chen, L. X. (2008). The Greenhouse Gas Emissions and Fossil Energy Requirment of Bioplastics from Cradle to Gate of Biomass Refinery. *Environmental, 42*(18), 6961-6966. doi:10.1021/es7032235

Zah, R., Böni, H., Gauch, M., Hischier, R., & Wäger, L. (2007). *Life Cycle Assessment of Energy Products: Environmental Impact Assessment of Biofuels - Executive Summary.* Bern, Switzerland: Empa Technology and Society Lab.

Zamagni, A., Buttol , P., Buonamici, R., Masoni, P., Guinée, J., Huppes, G., . . . Rydberg, T. (2009). *D20 Blue Paper on Life Cycle Sustainability Analysis.* Leiden, Netherlands: Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability (CALCAS).

Zeng, J.-B., Huang, C.-L., Jiao, L., Lu, X., Wang, Y.-Z., & Wang, X.-L. (2012). Synthesis and Properties of Biodegradable Poly(butylene succinate-co-diethylene glycol succinate) Copolymers. *Industrial & Engineering Chemistry Research, 51*(38), 12258-12265. doi:10.1021/ie300133a

Zhang, G. J., Cai, M., & Hu, A. (2013). Energy consumption and the unexplained winter warming over northern Asia and North America. *Nature Climate Change, 3*(5), 466-470. doi:10.1038/nclimate1803

Zhang, L., & Akiyama, T. (2009). How to recuperate industrial waste heat beyond time and space. *International Journal of Exergy, 6*(2), 214-227. doi:10.1504/IJEX.2009.023999

Zhao, J.-H., Wanga, X.-Q., Zenga, J., Yanga, G., Shia, F.-H., & Yana, Q. (2005). Biodegradation of poly(butylene succinate--butylene adipate). Polymer Degradation and Stability. *Polymer Degradation and Stability, 90*(1), 173-179. doi:10.1016/j.polymdegradstab.2005.03.006

Zheng, Y., Yanful, E. K., & Bassi, A. S. (2005). A Review of Plastic Waste Biodegradation. *Critical Reviews in Biotechnology, 25*(4), 243-250. doi:10.1080/07388550500346359

Zhu, W., Wang, X., Chen, X., & Xu, K. (2009). Miscibility, Crystallization, and Mechanical Properties of Poly(3-hydroxybutyrate-co-4-hydroxybutyrate)/Poly(butylene succinate) Blends. *Journal of Applied Polymer Science, 114*(6), 3923-3931. doi:10.1002/app.30965

Zilberman, D., Hochman, G., Rajagopal, D., Sexton, S., & Timilsina, G. (2012). The Impact of Biofuels on Commodity Food Prices: Assessment of Findings. *American Journal of Agricultural Economics, 95*(2), 275-281. doi:10.1093/ajae/aas037

Zou, Y. (2010). *Lightweight Composites from Biological Materials and Polypropylene.* College of Education and Human Sciences . Lincoln: University of Nebraska. Retrieved from http://digitalcommons.unl.edu/cehsdiss/60