Assessment and Solutions for Waste Handling of Compostable Biopolymers

by

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ABSTRACT

Fossil resources have enabled the development of the plastic industry in the last century. More recently biopolymers have been making gains in the global plastics market. Biopolymers are plastics derived from plants, primarily corn, which can function very similarly to fossil based plastics. One difference between some of the dominant biopolymers, namely polylactic acid and thermoplastic starch, and the most common fossil-based plastics is the feature of compostability. This means that biopolymers represent not only a shift from petroleum and natural gas to agricultural resources but also that these plastics have potentially different impacts resulting from alternative disposal routes. The current end of life material flows are not well understood since waste streams vary widely based on regional availability of end of life treatments and the role that decision making has on waste identification and disposal.

This dissertation is focused on highlighting the importance of end of life on the life-cycle of biopolymers, identifying how compostable biopolymer products are entering waste streams, improving collection and waste processing, and quantifying the impacts that result from the disposal of biopolymers. Biopolymers, while somewhat available to residential consumers, are primarily being used by various food service organizations trying to achieve a variety of goals such as zero waste, green advertising, and providing more consumer options. While compostable biopolymers may be able to help reduce wastes to landfill they do result in environmental tradeoffs associated with agriculture during the production phase. Biopolymers may improve the management for compostable waste streams by enabling streamlined services and reducing non-compostable fossil-based plastic contamination. The concerns about incomplete degradation of biopolymers in composting facilities may be ameliorated using alkaline amendments sourced from waste streams of other industries.

While recycling still yields major benefits for traditional resins, bio-based equivalents may provide addition benefits and compostable biopolymers offer benefits with regards to global warming and fossil fuel depletion. The research presented here represents two published studies, two studies which have been accepted for publication, and a life-cycle assessment that will be submitted for publication.

DEDICATION

This dissertation, including all the time and effort that has gone into it, is dedicated to my wife, Andrea, and our newborn son, Logan. Andrea has worked and sacrificed to help make this possible while Logan has given me the motivation, if not the sleep, to finish strong.

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INTRODUCTION

Motivation and Vision

Over the last century the plastics industry has grown to become the third largest sector of U.S. manufacturing with plastic goods exceeding a $373 billion value annually (Carteaux 2013). This market, which is comprised in large part of containers, packaging, and non-durable goods, contributes nearly 13% of the waste generated in the United States (USEPA 2014a). The feedstocks for traditional plastics such as polyethylene (PE), polyethylene terephthalate (PET), polystyrene (PS), and polyvinyl chloride (PVC) are fossil based resources. These feedstocks have provided a consistent source of raw materials for the development and improvement of plastics but increasing cost and scarcity of fossil resources could have effects on the plastics industry (Anastas and Kirchhoff 2002). Even without concern about availability of finite raw materials, the sheer size of the plastics industry represents a significant potential for gains in environmental performance if plastics, production methods, or waste handling could be improved.

Biopolymers have emerged as plant-based alternatives to fossil-based plastics. The concept of industrial scale biopolymer production began in the 1990s with the introduction of green chemistry and increasing fossil fuel prices (Iles and Martin 2013). Biopolymers constitute one of the fastest growing segments in the global plastics market with a growth rate around 25% annually and nearly one million tons annual production (Rapra 2012). Biopolymers are plastics that can be produced from renewable feedstocks such as sugar, corn, soy, hemp, algae, and methane captured from various wastes (Du and Yu 2002, Flieger, Kantorova et al. 2003). These feedstocks have varying amounts of both positive and negative environmental tradeoffs associated with agriculture-based production (Landis, Miller, and Theis 2007, Álvarez-Chávez, Edwards et al. 2012). Many biopolymer products are not completely bio-based but are comprised of blends of both conventional and renewable resources which is characteristic of plastic production (Hartmann 1998, Shen, Haufe, and Patel 2009, Shen, Worrell, and Patel 2010). For example, the predominant globally produced biopolymer, Bio-PET, is a blend of conventional feedstocks and bio-based ethylene derived from sugar (Morschbacker 2009).

While being made from plants provides an interesting selling point, some biopolymers have material characteristics, such as PLA which has good thermal properties and stiffness to density ratio, which may make them more attractive for some applications than the commodity plastics that currently dominate global markets (Patel, Crank et al. 2006, Braasch 2015). However, some biopolymers, such as Bio-PET which is used in Coca-Cola's PlantBottle™, are the chemical equivalent to their conventional counterpart with identical properties and the ability to be a 'drop-in' substitution when extruding and forming products and recycling them in the same waste streams (Philp, Ritchie, and Guy 2013). While there is the option to recycle those biopolymers which fall into the traditional 1 through 6 plastic resin recycling codes, such as Bio-PET which is a number 1 along with fossil PET, other biopolymers such as polylactic acid (PLA) and thermoplastic starch (TPS) may not be recyclable given the current infrastructure and are given the resin code 7 (a number 7 designates "other" plastics which are not typically recycled). However, these number 7 biopolymers often come with the additional feature of compostability (Lopez, Vilaseca et al. 2012, Roland-Holst, Triolo et al. 2013, Landis 2010).

In the United States, about 91% of plastics are not recovered and end up in a landfill or are incinerated. Generally, 13% of municipal solid waste (MSW) that is not recovered is incinerated with energy recovery as a waste management strategy (USEPA 2015a). While the overall recycling rate for plastics is only about 9%, certain plastic products and specific resins have higher rates of recovery. PET soft drink bottles were recovered at a rate of 31% in 2013, while high-density polyethylene (HDPE) milk and water bottles were estimated at about 28%. Packaging and nondurable plastics in MSW totaled 20.5 million tons, of which 10% were recovered (USEPA 2015a). The ability to compost biopolymers changes the traditional pathways for plastics at end of life (EOL) and creates the opportunity for material recovery from organic waste streams comingled with plastics via composting. Non-compostable plastics and organic wastes would need to be sorted prior to being sent to separate waste streams or all of the waste would be destined for a landfill.

Compostable plastics must conform to American Society of Testing and Materials (ASTM) standards including ASTM D6400-04 Standard Specification for Compostable Plastics, ASTM D6868-03 Standard Specification for Biodegradable Plastics Used as Coatings on Paper and Other Compostable Substrates, and ASTM D5338-98(2003) Standard Test Method for Determining Aerobic Biodegradation of Plastic Materials Under Controlled Composting Conditions (Song, Murphy et al. 2009, ASTM 2004, 2003b, a). Of compostable biopolymers, PLA is the most common in the United States, but TPS and polyhydroxyalkanoates (PHAs) are also common (EuBP 2014b, Tabone, Cregg et al. 2010, Rapra 2012). One significant problem for biopolymers is that some, primarily thicker PLA products, conform to the ASTM standards but in fact are reported to not degrade in compost facilities.

Further complicating the problems associated with composting is the fact that some bioplastics are labeled 'biodegradable' rather than compostable, a distinction which is not always well understood by stakeholders. The plastics with the 'biodegradable' label are defined in ASTM D6400-04 as "a plastic that degrades because of the action of naturally occurring microorganisms such as bacteria, fungi, and algae" but compostable plastics have more stringent requirements regarding the process and time frame of degradation (ASTM 2004). Biodegradable plastics take more time to break down which does not conform to commercial composting processes (Kale, Auras et al. 2007). Biodegradable plastics are used in products like grocery bags, trash bags, packaging, diapers, and agricultural mulch films (Ammala, Bateman et al. 2011). While ASTM standards are an important classification for this burgeoning industry, many composting facilities are having trouble with biopolymers not degrading at rates consistent with their composting processes.

With increasing interest in zero waste efforts (i.e. the practice of avoiding sending waste to landfills), there are more and more compostable products being sold with the promise of compostability and sustainability. The US Environmental Protection Agency (EPA) is encouraging a general shift towards composting as a means of avoiding emissions from organic materials, primarily food and yard waste, in the waste management industry (USEPA 2013b). Part of this transition is encouraged by the availability of biobased compostable plastics. The ability to market these products with phrases like "plant-based," "made from corn," and "we turn into soil" is a compelling argument for individuals, institutions, and cities that are seeking to reduce their environmental footprints or try to achieve zero waste goals (World Centric 2014, Coca-Cola 2012). Figure 1 shows a PLA cold cup labeled as "made from corn" (Appendix A shows additional labeling used for common biopolymer products). Biopolymers have the added benefit as disposable food service ware because they can be composted with food waste and do not need to be cleaned. Plastics fouled by food wastes cannot be recycled without significant pretreatment and compost operations will not accept organic wastes with excessive non-compostable plastic contamination(Wood 2014, Pfiefer 2014, Hill 2014).

Figure 1. An image of a PLA cold cup advertised as being "made from corn." Image source http://www.staples.com.

This research helps identify optimal disposal options and infrastructure for compostable biopolymers based on sustainability metrics. This dissertation reports the life-cycle environmental impacts of different disposal options, organizational and structural barriers to the adoption of compostable products, consumer motivation and disposal habits, and methods to improve disposal habits and waste handling.

Research Objectives

This dissertation reports the EOL impacts of biopolymers and provides solutions to improve the management of compostable plastics. The scope of this research extends from tracking waste scenarios at public events to evaluating how people identify different wastes to solving the challenges faced in industrial composting operations. The following chapters will also explore solutions to real barriers experienced by the industry and focus on developing solutions that reduce contamination and increase the value of industrial compost. This work was guided by and seeks to address the following research objectives:

1. Quantify end of life of waste flows of biopolymers via waste audits at public venues and waste handling facilities
2. Identify best practices for facilities management and waste handling of compostables via surveys and focus groups
3. Evaluate sustainable solutions for composting infrastructure options and factors influencing feasibility of scaling
4. Quantify the end of life environmental impacts of compostable biopolymer scenarios via life-cycle assessment incorporating findings from objectives 1-3

Intellectual Merit

The environmental impacts associated with a shift toward using more biopolymers in lieu of fossil-based polymers in the US are unknown, particularly with respect to EOL. In addition the impacts that purchasing (i.e. supply chain decisions) and use of biopolymers have not been studied extensively in the life-cycle literature despite the fact that biopolymers' ultimate environmental performance relies heavily on these factors. Similarly, while the U.S. EPA recommends that municipalities and facilities managers consider composting as a more sustainable method for landfill diversion, the sustainability of landfill diversion strategies has not been critically evaluated for the changing landscape of organic wastes, which now include bioplastics. The environmental profiles of compostable materials will aid in science driven decision making for new approaches of municipal waste handling.

The findings herein can be used to quantify the environmental EOL profile for biopolymers and contribute to the understanding of compostable biopolymer use in real-world scenarios. Waste management defines a key interface between humans and the environment, determining atmospheric emissions and the fate of materials entering the waste stream. Decision making at EOL alters the impacts of products and can also change the environmental profile for new biopolymers and energy that can be recovered from waste. The results of the research presented in this dissertation can help guide future research and policy aimed at improving environmental performance of biopolymers and the larger organic and recyclable waste streams.

Broader Impacts

This research is relevant to a broad audience of businesses and policymakers who are looking for ways to improve the environmental performance of consumer goods. Determining a management approach for waste is a real decision that has far reaching impacts on the life-cycle of nearly every product. Only with a comprehensive knowledge of the cradle to grave impacts of biopolymers and an understanding of the factors contributing to biopolymer use and decision making associated with EOL will it be possible to make informed decisions related to appropriate selection of products for specific applications and the development of infrastructure for optimal disposal. This and subsequent efforts are needed to assess the environmental sustainability of compostable products and inform the decision making of researchers, manufacturers, and consumers.

This research connects to real-world dynamics through collaborations with industrial, commercial and governmental partners. Through engagement with organizations like Waste Management Inc. (WM), NatureWorks LLC., Garick, City of Phoenix and Arizona State University, this research provides a unique perspective with significant insights. The results of this research can help cities and facilities managers evaluate a wider range of options when selecting products with different properties and policies to reduce their environmental footprint.

With funding through the Diane and Gary Tooker's K-12 STEM education grants a module was created based on personal waste audits and the US Environmental Protection Agency's Waste Reduction Model (WARM). This module was designed to scale to the appropriate educational level whether it is engaging high school students or engineering seniors at college. The activities and modeling encourage students to consider not just how a product is made but also what happens to it when it is thrown away and the module has students explore the substitution of different materials. Hundreds of students have taken part of this hands-on, experiential module that encourages them to use tools and metrics to understand their own connection to the environment.

Organization of Thesis

Table 1 shows the chapters within this document and the peer-reviewed journal in which each study was published. The findings of the life-cycle assessment (LCA) in Chapter 6 will also be submitted to a peer-reviewed journal which has yet to be determined.

Table 1. Contributing publications and the corresponding objectives addressed by each chapter

Thesis Chapter	Publication	Journal	Objective(s)
Ch. 2	Sustainability Assessment of Bio-Based Polymers	Polymer Degradation and Stability (Published 2013)	review and background 4
Ch. 3	Toward Zero Waste: Composting and Recycling for Sustainable Venue Based Events	Waste Management (Published 2015)	1 & 2
Ch. 4	Alkaline Amendment for the Enhancement of Compost Degradation for Polylactic Acid (PLA)Biopolymer Products	Compost Science & Utilization (Accepted 2015)	3
Ch. 5	Compostable Biopolymer Use in the Real World: Stakeholder Interviews to Better Understand the Motivations and Realities of Use and Disposal in the US	Resources Conservation and Recycling (Accepted 2015)	1 & 2
Ch. 6	Biopolymer Production and End of Life Comparisons Using Life-Cycle Assessment	TBD	4

Literature Review and Background

Drivers of Biopolymer Use

The demand for biopolymers has been driven in large part by increasing interest in environmental performance of plastics. This is due in part to a concern over the impacts associated with fossil-based plastics which have led to calls to reduce plastic wastes as well as outright bans on various plastic products (Rochman, Browne et al. 2013, Steinmetz 2014, Goldstein 2013). Simultaneously demand for biopolymers has increased because of bio-preferred policies and increases in efforts to divert organics from landfills as part of zero waste programs or specifically to avoid landfill emissions (Yepsen 2009, BioCycle 2014, EEA

2014, ASU 2014, San Francisco 2014). These various drivers have led major retailers with immense buying power to begin demanding greater use of biopolymers in the products they sell. It is just this sort of demand that promises to drive NatureWorks' PLA production from one billion pounds of production in 2013 to double that in the near future (Miel 2015).

Food waste reduction efforts have the potential to act as an indirect driver for the transition to compostable plastics. The EPA has developed a food recovery hierarchy which seeks to minimize the amount of organic wastes going to landfills. Composting is one of the treatment methods suggested for this effort (USEPA 2015b). It is the least prioritized method of reducing food wastes, but also is the easiest to implement on a significant scale. Biopolymers can be implemented in settings with high amounts of organic wastes without risking contamination that would prevent reduction efforts via composting.

Biopolymers: Production and Feedstocks

Biopolymers can be produced using a wide variety of feedstocks by implementing different technologies to refine the raw materials into formable plastics. The production process for biopolymers can be as simple as isolating starches from corn and forming products using a few additives but production can also require complex fermentation and polymerization processes to create high value products (de Jong, Higson et al. 2012). While 'drop-in' replacements can be utilized in existing infrastructure, unique biopolymers require a significant capital investment for production facilities such as the NatureWorks PLA production facility in Blair, Nebraska and the company's planned facility in Thailand (Vink, Davies, and Kolstad 2010, Esposito 2012). The following chapters detail the production and applications of the most common biopolymers in the U.S. including PLA, TPS, PHA, PDO, and Bio-PET. PLA, a major focus of this research, is found in two main types, crystalline and amorphous (Figure 2). Amorphous PLA is used for clear thermoformed products versus crystalline PLA, which undergoes additional processing with a number of different nucleating agents depending on the application and is used in high heat applications (NatureWorksLLC 2004, 2002b, a).

Figure 2. Amorphous PLA cup and crystalline PLA spoons, products by World Centric. The crystal clear cold cup is produced using NatureWorks' Ingeo PLA while the spoons are produced using a PLA/talc blend.

Feedstock development and availability remains a top priority for bio-based materials which can use a wide variety of agricultural feedstocks (Dijkstra and Langstein 2012, Morschbacker 2009, Álvarez-Chávez et al. 2012). There has been a lot of research dedicated to feedstock improvement in order to develop crops that minimize processing while maximizing the amount of usable biomass. The goal of feedstock improvement is to enable more efficient production which is supported by more dependable and consistent agricultural yields (Babu, O'Connor, and Seeram 2013, Gerngross and Slater 2000, van Beilen and Poirier 2007, de Jong et al. 2012). As investments in biopolymers increase, the industry will continue find new uses and markets just as the fossil plastic industry did in the last century, for example, PLA has become one of the primary plastics used in the 3-D printing industry because of superior thermal properties. The simultaneous investments in biofuels technologies and biorefineries may also create positive synergies for bio-based chemicals, for example the same technology that is used to create ethanol from the gasification of waste could simply be used to create ethanol as a plastic feedstock or the syngas produced during gasification could be fermented to produce 1,3 butadiene, a chemical used in the production of nylon (Aylott and Higson 2013).

The development of new feedstocks and polymers as well as the refinement of existing resins will broaden the applications for biopolymers and has the potential to improve the environmental footprints associated with products manufactured using biopolymer technologies (Snell, Singh, and Brumbley 2015, Mohan 2014, Reddy, Vivekanandhan et al. 2013, Brigham and Sinskey 2012). Near-term developments include the expansion of PLA and starch markets as well as the use of bio-based ethylene in the production of PE (including HDPE and LDPE) and PET (Iwata 2015, Laycock and Halley 2014, Mohammadi Nafchi, Moradpour et al. 2013). Although the initial development of biopolymers for food and drink packaging has been expensive, the price stabilization of biopolymers will lead to "technological and environmental competitiveness" (Byrne 2015).

End of Life Options

Landfilling is the primary form of waste management for MSW generally as well as for plastics. Landfills can have dramatically different internal conditions and a wide variety of technologies employed to reduce harmful environmental emissions. Landfills in the US are typically lined to reduce leaching into the surrounding soil and capped to control the flow of landfill gasses (Weitz, Thorneloe et al. 2002, Denison 1996). The degradation of organic materials is particularly important for compostable and biodegradable plastics because under landfill conditions they can create methane which is a powerful greenhouse gas. While there are technologies to capture landfill gas and either flare it or burn it for energy recovery, these is a large amount of uncertainty around capture efficiency which may vary anywhere between 50 to 95% with the EPA recommending the use of 75% capture efficiency for modeling purposes (Spokas, Bogner et al. 2006, USEPA 2008). Due to the emissions associated with landfills there have been efforts to increase the availability of options for material recovery and landfill avoidance.

Recycling is another EOL option for plastics and the primary alternative to landfilling. Recycling can be accomplished through different pathways depending upon the resin characteristics, some methods are as simple as chipping, heating, and reforming while others require more intensive chemical synthesis to yield a useful feedstock (Al-Salem, Lettieri, and Baeyens 2009). In the US, recycling has been increasingly utilized as an EOL option as a percent of total waste but rates remain low, with an average plastic recycling rate of 9% (USEPA 2015a). The process of recycling can have significant environmental benefits because it offsets the production of virgin materials and upstream energy use, however the quality of the recycled material can diminish the more it is recycled. The reduced quality of recycled plastics leads to 'down-cycling' through different types of applications. The balance between demand for recycled feedstocks and cost to process recyclables results in narrow profit margins for the industry (Hopewell, Dvorak, and Kosior 2009).

Composting is method of processing organic wastes that utilizes the natural biological processes of aerobic bacterial communities to convert organics into stable soil carbon which serves to buffer nutrients in the soil, control moisture content, and sequester biogenic carbon. The benefits of composting allow for the elimination of waste and the creation of a useful soil amendment which can be sold for landscaping, gardening, and farming. Despite the benefits, composting has largely been the purview of a few farmers and gardeners but with increasing levels of municipal-scale waste management. The composting process reduces the volume of wastes by up to 50% (CalRecycle 2006) which has made it useful in niche municipal services like neighborhood leaf disposal. Composting has also been used as a pretreatment for landfill wastes to reduce the overall volume and convert organics to CO_2 rather than risk CH_4 generation once the wastes are in the landfill (Komilis, Ham, and Stegmann 1999), avoiding the severe climate change implications of methane emissions (Bogner, Pipatti et al. 2008, Adhikari, Barrington, and Martinez 2006, Wang, Odle et al. 1997).

Incineration of MSW, with or without energy recovery, has been in decline since the 1980s with 80 waste-to-energy facilities nationwide in 2013 down from 102 in 2000 with the majority of capacity in the Northeast and the least in the West (USEPA 2015a). Some combustion facilities, like composting of MSW, are used simply to reduce the volume of waste sent to landfill. The decline of combustion facilities was due in part to pressure from environmental groups and social movements who expressed concerns about the effects MSW incineration can have on air quality (MacBride 2013). There are some new efforts to expand the use of combustion facilities with energy recovery as an alternative to landfill, including pyrolysis and plasma technologies but because incineration facilities are expensive to site and there is significant opposition from neighborhoods and environmental groups, few have been built (Arena 2012, Zhang, Dor et al. 2012, MacBride 2013).

Increasing Interest in End of Life

While there have been many LCAs of biopolymers, a majority of them focus on the production related impacts with far fewer evaluating EOL. Chapter 2 details the use of LCA and the findings of previous biopolymer studies which largely focus on global warming and fossil fuel depletion impacts through July 2012. Despite the lack of scenarios evaluating composting in LCA, compostability is a critical feature of some biopolymers and a dramatic shift from recycling or landfilling.

The technology for municipal scale composting is well developed with published best practices and equipment available for the chipping, mixing, and hauling organic wastes (Chen, Inbar et al. 1997, Campbell, Glasser et al. 2009). Composting is not widely adopted as a result of the ease and historically low costs of landfilling, which is the status quo, and the complications of diverting wastes for feedstock acquisition. This is due in large part to contamination in consumer separated wastes or the complexity in sorting unseparated wastes (Platt, Goldstein et al. 2014, Yepsen 2013, Bernstad 2014, Graham-Rowe, Jessop, and Sparks 2014). However, the potential of food wastes to drive methane generation in landfills and the challenges of climate change have driven interest in composting as a more environmentally friendly approach to waste management.

The ASTM specifications and ISO test methods used to determine whether a plastic is compostable under controlled composting conditions do not necessarily reflect performance under real composting conditions (ISO). The standard test methods differ from real composting processes due to climatic differences, types of organic feedstocks, and other variables such as internal temperature, moisture, and pH (Kale, Auras, et al. 2007). There are many published and anecdotal reports of biopolymers not composting sufficiently in commercial composting facilities (Ghorpade, Gennadios, and Hanna 2001, Mohee and

Unmar 2007, Gómez and Michel Jr 2013). Additionally, crystalline PLA has been shown to be stable within a landfill environment, which is an important feature in order to avoid unwanted CH_4 emissions or premature degradation, while amorphous PLA does result in emissions under landfill conditions (Kolstad, Vink et al. 2012), leading to further questions about the life-cycle impacts of biopolymers.

Decision Making Implications on EOL

Consumer decision making, facilities management, and available waste services dictate how biopolymers, or any other products, are entering the EOL phase. The behavioral aspects of waste collection are complex and involve a wide variety of factors which contribute to different patterns of consumer behavior across demographics and locales.

For individual consumers decision making is often based on awareness of environmental impacts and the ease with which they can alter their behaviors to conform to their preferences. By providing convenient services and communicating environmental goals, facility managers and waste collection companies can empower their customers to act in a way that conforms to the environmental preferences of their customers while achieving the goals of the businesses (Graham-Rowe, Jessop, and Sparks 2014, Bernstad 2014). Behavioral research which investigated waste collection strategies in public venues highlights the benefits of an approach which incorporates techniques to encourage positive behavior change through social dynamics and traditional outreach. Early adopters of new waste collection schemes can provide a model of behavior which in turn encourages others to follow suit (Zhang, Williams et al. 2011, Sussman and Gifford 2013, Sussman, Greeno et al. 2013, Ohtomo and Ohnuma 2014, Schwab, Harton, and Cullum 2014, Mickaël 2014).

Waste management companies have focused largely on systems models and collection optimization to improve their performance, primarily to reduce costs associated with waste handling. Most of the work that has been done on modeling waste scenarios has focused on optimizing the current waste handling paradigm with recycling only becoming a piece of most models in the 1990s. Since then, there has been a slow progression towards the development of tools to identify more sustainable methods to waste management (Morrissey and Browne 2004). Waste management, which is characterized by a relationship between sources, sinks and offsets, has resulted in decreasing environmental impacts even while municipal solid waste (MSW) increased by 60% from 1974 to 1997. Significant improvements were largely due to methane emissions reduction from landfills and increased recycling rates (Weitz, Thorneloe et al. 2002).

Approaches to waste management have been evolving as the cost of landfilling and the environmental concerns about siting new landfills increases (Lund 1990). Dynamic models of the waste management system have been designed to evaluate costs and efficacy policy interventions designed to measure flows to landfills (Dyson and Chang 2005, Kollikkathara, Feng, and Yu 2010). An increased desire to move wastes out of landfill and demand from large customers, including ASU, the City of Phoenix, and Intel, has driven more companies, like WM, to expand their services to include composting.

Waste handling is a dynamic activity with constantly changing variables including the introduction of new types of waste and new approaches to managing wastes which have the potential to change impacts on a broad scale. While compostable biopolymers may help reduce landfill wastes, for plastics and other organics, this effort must be accomplished at scale. If biopolymers can be produced with similar or preferable impacts compared to fossil-based plastics, EOL processes may govern which plastics perform more favorably under different use paradigms.

Furthermore, decision making must be informed by accurate assessments of current and near-term performance of biopolymers. In addition to LCAs, many environmental assessments of biopolymers have been done which can contribute to the development of biopolymer LCAs including investigation of compost emissions (Hermann, Debeer et al. 2011), investigation of degradation rates in landfill (Kolstad et al. 2012), updating life-cycle inventories (Vink and Davies 2015a), and evaluating useful service life of biopolymers (Miller, Srubar Iii et al. 2015).

CHAPTER 2

SUSTAINABILITY ASSESSMENTS OF BIO-BASED POLYMERS

This Chapter was published in the peer reviewed journal *Polymer Degradation and Stability* and appears as published with the exception of text and figure formatting. The citation for this article is:

Hottle, T.A.., M.M. Bilec, and A.E. Landis. 2013. "Sustainability assessments of bio-based polymers." *Polymer Degradation and Stability* 98 (9):1898-1907.

This chapter addresses research objective 3) Evaluate sustainable solutions for composting infrastructure options and factors influencing feasibility of scaling

Introduction

As biopolymers capture a larger market share, the measurement of their life-cycle environmental impacts will be important to enable consumers and producers to identify more sustainable methods of use, production, and disposal for such products. This paper summarizes the range of reported findings from peer-reviewed life-cycle assessments (LCAs) and commonly used LCA databases. LCA is a tool that quantifies the environmental sustainability of bio-based polymers from their 'cradle to grave'. A review of LCAs and LCA databases provides the research and polymer community with guidance toward the use of LCA in furthering the sustainability of the use, design, and disposal of bio-based polymers.

Plastics are used in all aspects of life including textiles, electronics, healthcare products, toys, packaging for foods, and many other goods. Approximately 31 million tons of plastic were used in the United States in 2010 with 14 million tons used in packaging, 11 million tons used in durable goods, and 6 million tons used in non-durable goods such as disposable diapers, cups, and plates (USEPA 2010). Globally, plastic production exceeded 260 billion kilograms of plastic in 2009 (Thompson, Swan et al. 2009). According to the US Census Bureau the population of the US in 2010 was nearly 309 million people (Mackun 2011), which means an average of about 200 pounds of plastic per person was consumed that year.

Currently the dominant feedstocks for plastic production are derived from the fossil fuel industry. The chemistry of plastics lends itself to the readily accessible constituents of petroleum and natural gas. These sources have been able to provide reliable, consistent feedstocks for plastics development over the last 60 years. Over time, plastics have become more and more prevalent in daily life and new technologies are improving the performance of plastics, but just as gasoline and diesel will decrease in availability due to the increasing cost or scarcity of petroleum and other fossil-based fuels, so too will plastics made from fossil resources (Anastas and Kirchhoff 2002). This increasing scarcity of resources emphasizes the need for alternative methods of creating plastics. Further, if resource availability were not a concern, it would be desirable to find methods of production that decrease the environmental impacts of ubiquitous materials because of the sheer scale of the industry. Petroleum-based plastics are crafted from carbon that has been locked up in the earth for millions of years. If this carbon were released through the incineration of the plastics, or some other form of degradation, it would result in a net increase of greenhouse gases in the atmosphere.

Plastics have different useful lifespans and are disposed of in a number of ways with varied recycling rates. According to the US Environmental Protection Agency (EPA), in 2009, plastics contributed to 12%, by weight, of the municipal solid waste (MSW) in the US, and 7% of plastics that were disposed of in MSW were recovered for recycling, though recovery rate is not necessarily indicative of a final recycling rate. Of total plastics, about 93% end up in a landfill or are incinerated. Generally, 12% of MSW that is not recovered is incinerated as a waste management strategy. When burned, 1 kg of plastic produces an average of 2.8 kg of carbon dioxide (EIA 2011). While overall recovery of plastics for recycling was only 7%, recovery of certain plastic containers is more significant. Polyethylene terephthalate (PET) soft drink bottles were recovered at a rate of 28% in 2009, while high-density polyethylene (HDPE) milk and water bottles were estimated at about 29%. Packaging and nondurable plastics in MSW totaled 19.2 million tons, of which 9% were recovered (USEPA 2009).

Biopolymers come in many different forms; they can be derived from renewable resources and may not be defined within the traditional plastics classification numbering system 1–6, like polylactic acid (PLA) (Landis 2010) or they can be partially made from renewables and synthesized like traditional plastics as in the case of bio-based PET (Tabone et al. 2010, Morschbacker 2009). Biopolymers offer a renewable alternative to traditional petroleum-based plastics and can be derived from a wide variety of feedstocks including agricultural products such as corn or soybeans and from alternative sources like algae or food waste (Flieger et al. 2003, Du and Yu 2002, Landis, Miller, and Theis 2007). Biopolymers can replace petroleum-based polymers in nearly every function from packaging and single use to durable products.

Biopolymers are being designed with features such as biodegradability and compostability, which are standardized in the US according to ASTM D6400-04 Standard Specification for Compostable Plastics, ASTM D6868-03 Standard Specification for Biodegradable Plastics Used as Coatings on Paper and Other Compostable Substrates, and

ASTM D5338-98(2003) Standard Test Method for Determining Aerobic Biodegradation of Plastic Materials Under Controlled Composting Conditions (ASTM 2004, 2003b, a). Biopolymers offer the opportunity to reduce fossil resources required to produce the 21 million tons of plastic annually consumed for packaging and non-durable goods, as well as divert the 16.7 million tons of plastic waste entering landfills. However, being derived from renewable resources does not guarantee that biopolymers will perform favorably when compared to petroleum-based polymers (Miller, Landis, and Theis 2007), and as such, sustainability assessments like LCA are conducted to compare and improve the environmental impacts of biopolymers.

This review presents a broad summary of the current status of environmental impact assessments for biopolymers. We begin with an overview of biopolymers and an introduction to LCA. Then we review the output data from the commonly used life-cycle inventory (LCI) database, ecoinvent, and impact assessment tool. Finally, we review and analyze the findings of LCA studies on biopolymers that have been published within the peer reviewed literature.

Common Biopolymers

The studies reviewed in this paper focused on the LCA results of PLA, PHA, and thermoplastic starch (TPS). These are the most prevalent biopolymers currently represented in life-cycle literature. While there are other biopolymers on the market and in development, such as bio-based 1,3-propanediol (PDO) and bio-based polyethylene terephthalate (Bio-PET), publicly available data and life-cycle assessment results were not available at the time of this review.

The applications of PLA include clear and opaque rigid plastics for packaging, disposable goods, durable goods, and bottles, as well as films and fibers for a variety of purposes (Vink, Rábago et al. 2004, Vink, Davies, and Kolstad 2010). PLA is made from lactic acid, which is produced through the fermentation of dextrose typically sourced from corn, however any starch-rich feedstock could be used. Lactic acid can be polymerized in a number of different ways to create granules that are used to make commercial products (Giorno, Chojnacka et al. 2002, Tay and Yang 2002, Gruber, Hall et al. 2001). PLA can be blended with petroleum-based polymers or fibers, either synthetic or natural, to improve the heat resistance or durability of the plastic (Landis 2010). PLA-based plastics can be biodegradable and compostable, features that offer a wider variety of options for disposal (Gerngross and Slater 2000).

PHA had a short history of use in packaging and bottles but is not widely used in these applications today (Gerngross and Slater 2000). PHA is increasingly being used in more niche applications in a variety of industries from medicine to agriculture. PHA is produced through the bacterial fermentation of renewable feedstocks containing monomers such as glucose, sucrose, and vegetable oil, resulting in the formation of the polymer (Akiyama, Tsuge, and Doi 2003, Philip, Keshavarz, and Roy 2007). Similar to PLA, PHA can also be combined with other materials to form composites with improved properties. PHA is also biodegradable and can be used to create compostable plastics (Philip, Keshavarz, and Roy 2007).

Another biopolymer included in the studies reviewed herein is TPS. It is created using the starch polymers from renewable sources, primarily corn, which is then processed and combined with additives and formed into shape (Bastioli 1998). TPS is generally incorporated into composites with synthetic polymers to create materials appropriate for the market. These materials can be used in making films, rigid materials, such as plates and cutlery, packaging, and foams, and, depending upon the constituents may be biodegradable and compostable. Current research efforts are focused on creating new TPS based composites by incorporating fibers or nano-materials to improve or completely change the characteristics of starch products (Bastioli 1998, Lopez et al. 2012, Mohammadi Nafchi et al. 2013, Cyras, Manfredi et al. 2008).

Two other important plant-based materials in the polymer industry are bio-based 1,3-propanediol (PDO) and bio-based polyethylene terephthalate (BIO-PET); however these polymers are not well represented in the LCA literature and thus were not included in the subsequent review. PDO is made through biological fermentation processes in conjunction with petroleum products to create materials comparable with nylon. The primary biological feedstock used in the fermentation process is corn grain, which makes up 37% of the polymer by weight. The remaining content is derived from fossil-based products (Álvarez-Chávez et al. 2012). Current applications of polymers made with PDO include carpeting, apparel, and films, which are reported to outperform traditional petroleum-based materials (Kurian 2005). BIO-PET, which is made from combining bio-based ethylene and other petroleum-based feedstocks, is most notably used in clear plastic bottles. The ethylene portion is made from corn fermentation similar to the corn ethanol process, and is then synthesized in the same manufacturing process as traditional PET. This results in a product identical to traditional PET that is recyclable but not biodegradable (Shen, Haufe, and Patel 2009). Efforts exist to create a completely bio-based PET product (Shiramizu and Toste 2011).

PDO and BIO-PET products should be evaluated in an ongoing basis, similar to PLA and PHA, to determine the environmental impacts of these products. Additionally, as manufacturing methods are refined and additional biopolymers are created, future research will be needed to identify the impacts of these products and identify where improvements can be made.

LCA as a Method for Quantifying Environmental Impacts

To determine the environmental impacts of a product or process, a LCA is often conducted. LCA provides a comprehensive and quantitative analysis of the environmental impacts of a product or process throughout its entire life-cycle. LCA is a powerful and widely used tool for measuring the sustainability of an enterprise or concept and informing decisions with respect to sustainability and environmental considerations. Guidelines for conducting an LCA are defined by the International Organization for Standardization (ISO) 14040 series (ISO 2006). There are four main steps to an LCA according to ISO:

1) Goal and Scope Definition – defines the extent of the analysis including the goals and the system boundaries. The functional unit for the LCA is defined within this step. The functional unit describes a reference for what is being studied and how much or over what time frame (i.e. 1 kg plastic resin).

2) Inventory Analysis – documents material and energy flows that occur within the system boundaries, often referred to as Life-cycle Inventory (LCI).

3) Impact Analysis – characterizes and assesses the environmental effects using the data obtained from the inventory, often Life-cycle Impact Assessment (LCIA). LCIA expresses the LCI data in common terms, usually with respect to an equivalency

factor, such as CO_2-equivients for greenhouse gas emissions. Common LCIA categories include global warming potential, non-renewable resource depletion, eutrophication, ecotoxicity, acidification, ozone depletion, smog formation, and human health (e.g. carcinogens, respiratory impacts, and non-carcinogens).

4) Interpretation – reviews the results of the LCA, identifies opportunities to reduce the environmental burden throughout the product's life, and provides conclusions and recommendations.

Many organizations have established guidelines for performing detailed LCAs, including the Environmental Protection Agency (EPA), and the Society for Environmental Toxicology and Chemistry (SETAC) (ISO 2006, Vigon, Tolle et al. 1992, Fava, Denison et al. 1991, UNEP/SETAC 2005). All of these organizations describe LCA steps similar to those listed above. Assessments can range from conceptual models to detailed quantitative analyses. There are a host of LCA software packages and tools available to aid in the construction of an LCA. Additional information is available in the further reading section of this chapter.

LCAs can be comparative or stand-alone (i.e. an analysis of a single product). Through use of a stand-alone LCA, it is possible to observe which stage (creation, use, or disposal) causes the most impact and may offer suggestions to minimize impacts throughout the product life. Comparative LCAs among possible products help to determine the environmentally preferable alternative.

The system boundaries of an LCA may extend from cradle to gate, cradle to grave, or cradle to cradle. As depicted in Figure 3, cradle to gate (dashed lines) implies that the life-cycle assessment covers activities prior to the use phase while cradle to grave (dashed and dotted lines) includes the product's use and end-of-life. Cradle to cradle (whole chart) indicates a product that can be disposed of and returned back to the natural environment; the life-cycle of PLA can be considered as such because when PLA degrades its carbon is recycled back into the environment for uptake by biomass. In LCAs of polymers, practitioners may also describe boundaries from the cradle to pellet; where the analysis stops at the creation of resin pellets, and excludes processes downstream including product manufacture, use, and disposal. Furthermore, LCA practitioners also refer the system boundaries in terms of ‘scope’. The system boundaries for a scope 1 LCA would include only the material and energy flows associated with the manufacturer; for example, Scope 1 system boundaries for the polymer manufacturer would extend only to the direct inputs and outputs associated with their factory. Scope 2 system boundaries include Scope 1 in addition to all supply chain and raw materials extraction data. Scope 3 system boundaries are used to define the entire life-cycle, including any further production or processing downstream of Scope 1 as well as use and disposal.

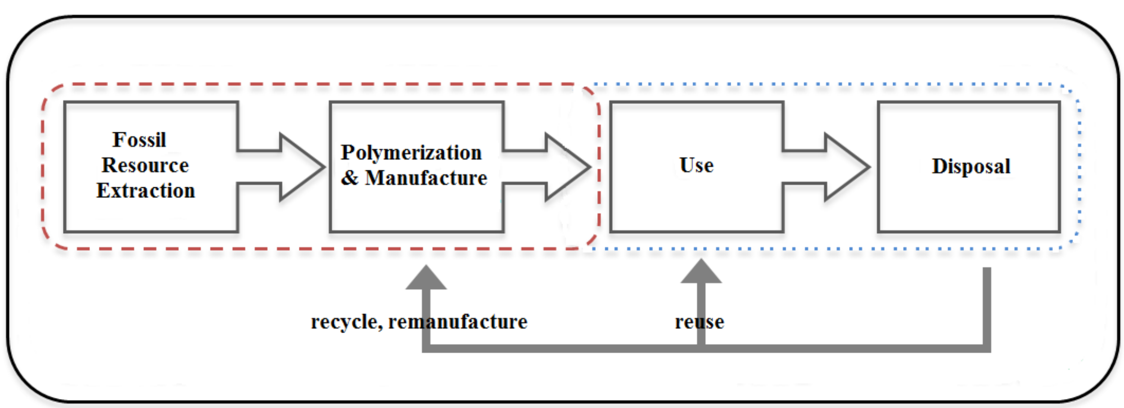

Figure 3. Generic life-cycle stages for polymers. The dashed line indicates a cradle to gate system boundaries, the dotted line is an extension of that system boundary to cradle to grave, and the entire figure is indicative of a cradle to cradle assessment.

Review of Environmental Impacts of Polymers Reported in Existing LCA Databases

LCA data is readily accessible in existing LCA tools for some biopolymers and most petroleum-based polymers. To review the environmental impacts reported in these existing LCA databases, life-cycle data for biopolymers and petroleum-based polymers were obtained from the ecoinvent v2.2 database (Frischknecht, Jungbluth et al. 2005). LCA data is available within ecoinvent for high density polyethylene (HDPE), low density polyethylene (LDPE), polyethylene terephthalate (PET), polypropylene (PP), polystyrene (PS), and two biopolymers: PLA and TPS. This data is reported in ecoinvent by the "European plastics industry" for petroleum polymers, while biopolymer data is reported by NatureWorks for PLA compiled in 2007, and from Novamont from 2004 for TPS. In order to review and succinctly present the ecoinvent data (which otherwise would be tables of hundreds of material and energy flows), LCA software such as SimaPro enables LCI data from ecoinvent to be used in conjunction with the TRACI v2.00 (Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts) (Bare 2011), developed by the US EPA, to compare life-cycle impacts from the database. TRACI calculates impacts for global warming potential (GWP), eutrophication, ecotoxicity, acidification, ozone depletion, smog formation, human health – carcinogens, human health – non-carcinogens, and human health – respiratory. The limitations to reviewing existing biopolymer LCI data in this manner are discussed in subsequent sections.

The comparative life-cycle environmental impacts from existing databases for petro- and biopolymers are shown in Figure 4 and Figure 5, reported directly from ecoinvent and TRACI with no modifications. These figures present a simplified analysis of the ecoinvent data using TRACI to demonstrate life-cycle methodology, and can provide a baseline for the environmental impacts of PLA and TPS with commonly used data and tools. The results reported from ecoinvent represent a cradle to granule (i.e. gate) system boundary for the production of 1 kg of granules for the five common petroleum-based plastics and PLA. Since TPS is not formed into granules, the functional unit for TPS was 1 kg of processed starch. While the polymers are compared in Figure 4 and Figure 5 by weight, the functional units and thus the comparative impacts of the subsequent products may vary.

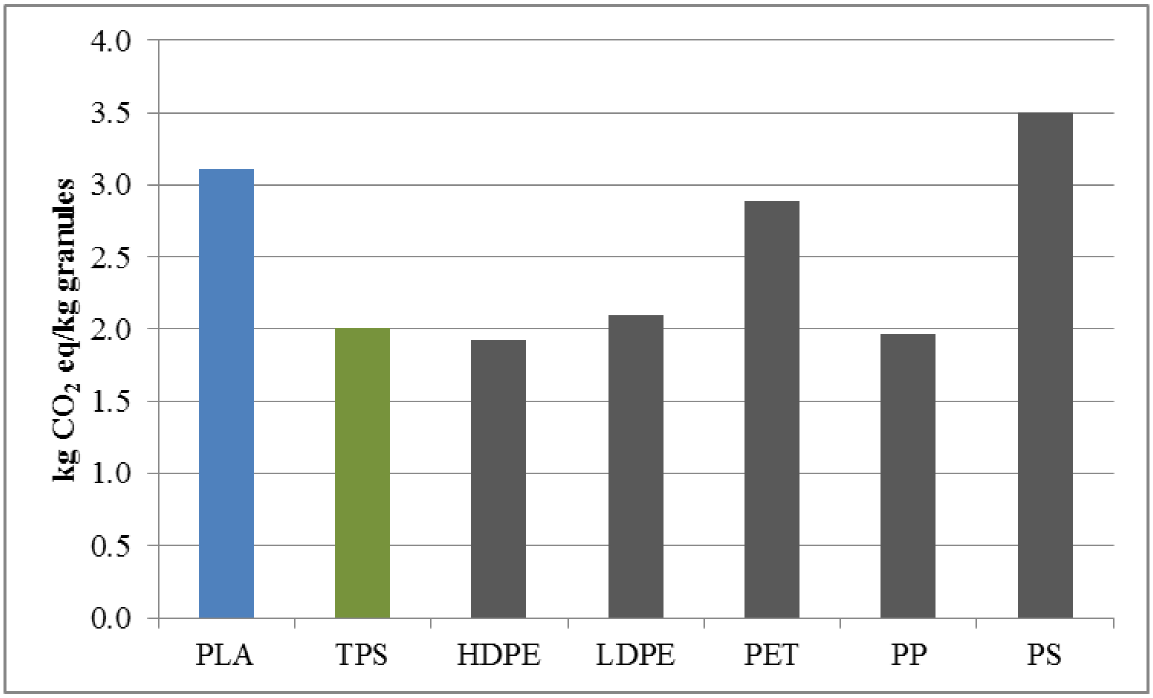

Figure 4. Global warming potential for cradle to granule (gate) of PLA and TPS compared to five common petroleum-based plastics. Data taken from ecoinvent v2.2 and TRACI v2.00. PLA = polylactic acid, TPS = thermoplastic starch, HDPE = high density polyethylene, LDPE = low density polyethylene, PET = polyethylene terephthalate, PP = polypropylene, PS = polystyrene.

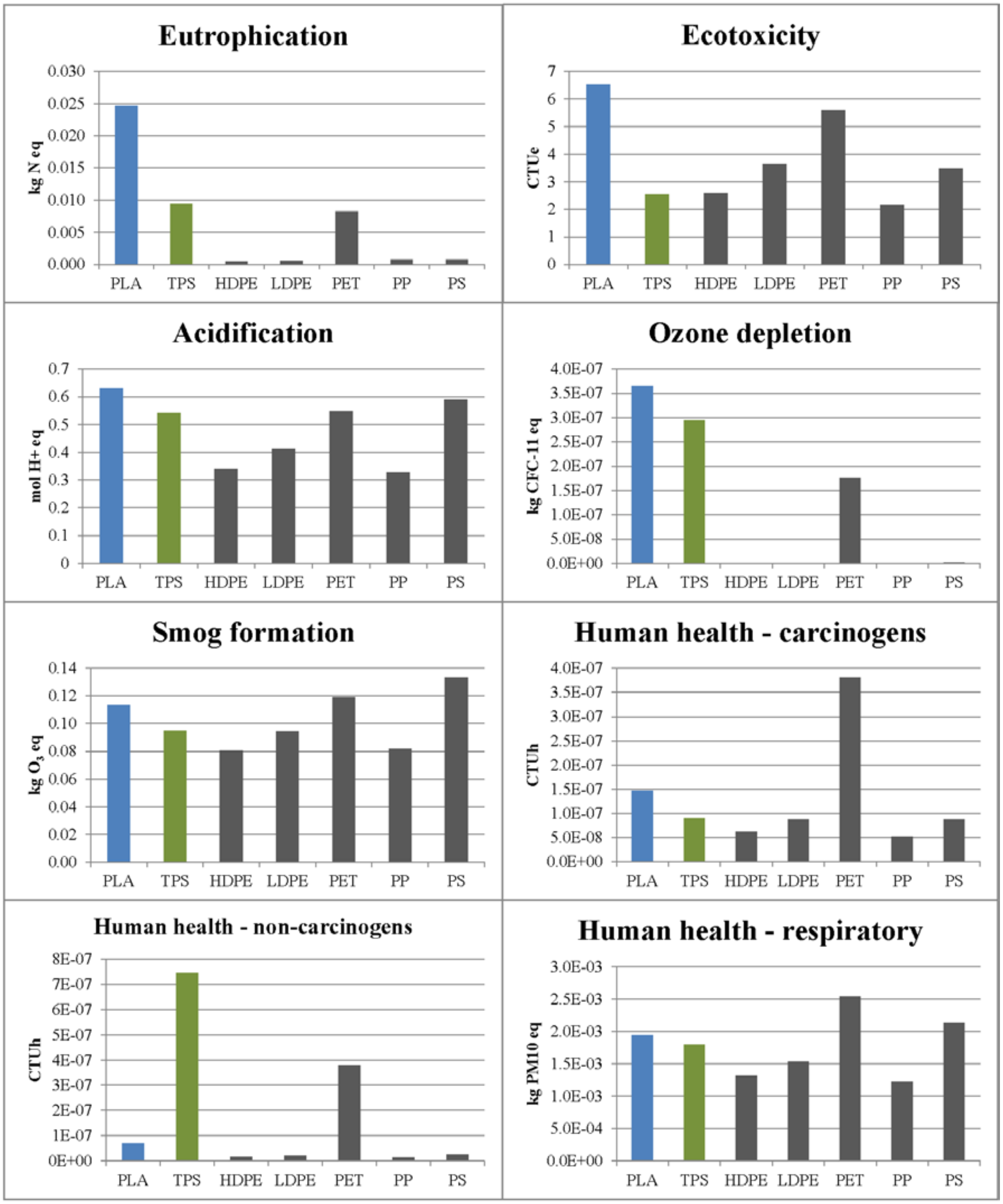

Figure 5. Life-cycle environmental impacts of PLA and TPS compared to petroleum-based polymers per kg of granule (starch). Data taken from ecoinvent v2.2 and TRACI v2.00. PLA = polylactic acid, TPS = thermoplastic starch, HDPE = high density polyethylene, LDPE = low density polyethylene, PET = polyethylene terephthalate, PP = polypropylene, PS = polystyrene. CTUe = Comparative Toxic Unit ecosystem, CTUh = Comparative Toxic Unit human health.

While Figure 4 shows little difference between the biopolymers and petroleum-based plastics with regards to GWP, it is clear from the results of other impact categories included in Figure 5 that there are environmental tradeoffs between biopolymer and traditional polymer production. There are environmental impact categories, notably eutrophication, ozone depletion, and non-carcinogenic human health, in which the biopolymers exhibit higher environmental impacts when compared to the petroleum-based plastics. However, the relative significance of these impacts should be evaluated to determine whether or not some impacts occur at such small rates that there is no real impact as a result of the production process. The significance of the impacts can be determined through normalization and weighting, which assigns weighted values to the different impact categories and helps compare them to known impact values. This review paper does not apply weighting or normalization to determine significance; the reader is referred to Development of normalization factors for Canada and the United States and comparison with European factors (Lautier, Rosenbaum et al. 2010) for an example of normalization factors for TRACI.

Using LCI data combined with an LCIA tool in a black box approach as depicted in Figure 4 and Figure 5 may result in overestimation of impacts or even an inaccurate model of the system. Many environmental impact categories have certain conditions that must be modeled accurately. For example, in the case of the eutrophication category, water bodies tend to be either nitrogen or phosphorus limited, which means that only N compounds would contribute to eutrophication in N-limited water bodies. As such, P emissions would not contribute to eutrophication in those areas. When LCI data is analyzed using LCIA tools without customization, these details related to regional implications and limitations of water bodies are not taken into account in existing LCA tools. Thus, eutrophication impacts can be overestimated. Similarly, impacts related to smog tend to only manifest in urban areas. However, black box LCA tools do not account for regional variations in emissions, and again may overestimate the impacts of smog-related emissions. In the case of PLA, smog related emissions occur primarily at power production facilities (40%) and on farms (33%); farms in particular are generally located where they are less likely to contribute to smog. When conducting a comprehensive LCA of specific products with known production locations and known supply chains, it is important to understand and model the particulars of the system, while simultaneously utilizing sensitivity analysis and scenario analysis to determine any changes to LCA impacts resulting from changes to supply chains or production facilities.

Based on the outputs from TRACI and the data supplied by ecoinvent, the eutrophication attributed to the biopolymers is due in large part to agricultural N and P emissions from intensive farming, which contributes 52% of the eutrophication impacts for PLA and 56% for TPS. Energy use during the production process was the secondary driver of eutrophication for biopolymers. Eutrophication from PET production results primarily from the use of ethylene and ethylene glycol, which is one of the primary compounds used to make PET (with either dimethyl terephthalate (DMT) or terephthalic acid (TPA)). Biopolymers' ozone depletion impacts are largely attributable to emissions that result from the transport of fossil fuels used in the process of creating the biopolymers. The non-carcinogenic human health impacts from TPS can be associated with the ecoinvent data for intensive corn farming in Switzerland which results in zinc emissions to the soil. TPS exhibited more than 7 times greater noncarcinogenic human health impacts compared to PLA, which are both made from corn. A sensitivity analysis was performed to determine if the methods of farming defined by ecoinvent were impacting the results. Ecoinvent data for

PLA is derived from average corn production occurring in the US while ecoinvent data for TPS corn production is from Switzerland (Frischknecht et al. 2005).

In order to evaluate the difference that the ecoinvent regional agricultural data has on resultant LCA environmental agricultural impacts, the original ecoinvent TPS results were compared to a modified TPS process that was built using the PLA corn agricultural data. Only the agricultural data was changed in the modified TPS process; and all data was obtained without modification from ecoinvent. Similarly, an additional sensitivity analysis was conducted to determine the effects of different electricity mixes between the TPS (ecoinvent utilizes Italy's mix) and PLA (ecoinvent uses a generic European mix) ecoinvent data. The results of the sensitivity analysis can be seen in Figure 6. This sensitivity analysis demonstrates minimal change due to the electricity mix. However, the effect of the data from ecoinvent's regions for corn production created dramatic effects on the results for non-carcinogens that would bring the non-carcinogenic human health impacts for TPS down to 1.99E-7 CTUh, falling between PLA and PET. When we reviewed the inventory data in ecoinvent we found that corn produced in the US resulted in negative zinc emissions, while corn grown in Switzerland results in zinc emissions to the soil. This is a discrepancy in the data that is characteristic of the types of problems that can manifest in LCA results if LCI data is not evaluated and properly validated.

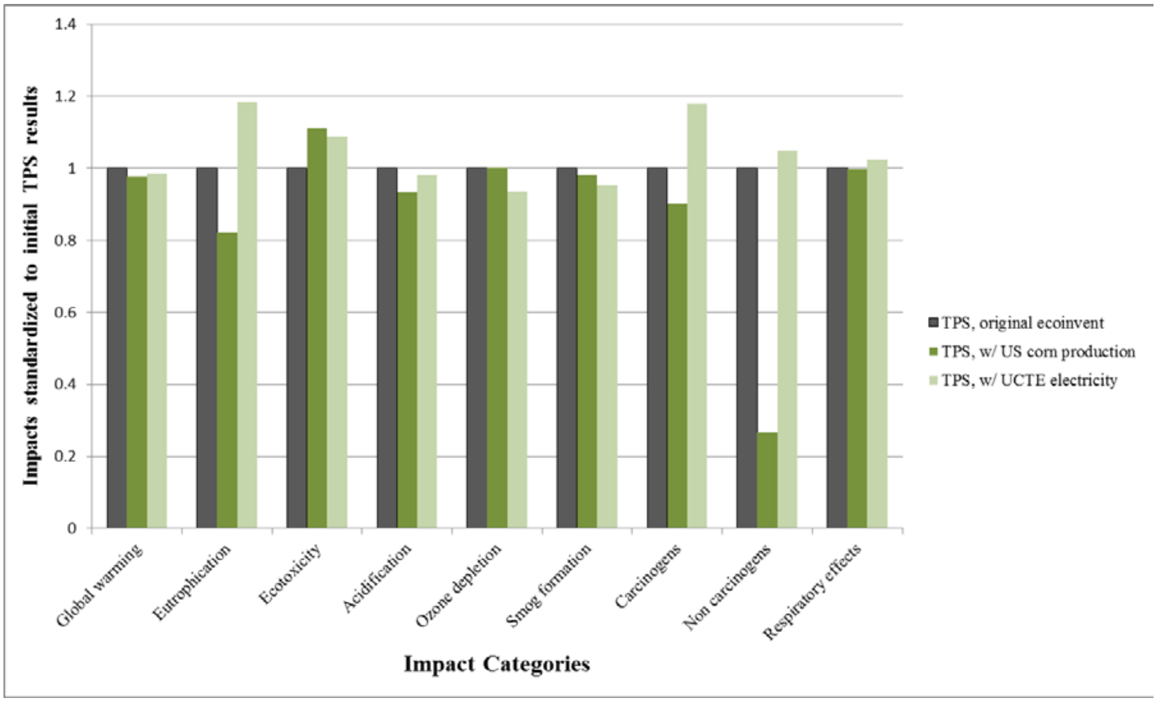

Figure 6. Sensitivity analysis for the corn production and energy mix of TPS using the TRACI v2.00 results and data from ecoinvent 2.2 database. TPS original ecoinvent uses German corn production and an Italian electricity mix. TPS w/ US corn production swaps TPS original ecoinvent agricultural data with US corn production data used in ecoinvent PLA data. TPS w/ UCTE electricity swaps TPS original ecoinvent electricity data with ecoinvent electricity data for PLA processes, which were derived from Union for the Coordination of the Transmission of Electricity averages.

When taken together, Figure 4 and Figure 5 make it difficult to determine if there is a significant difference between the cradle to gate production of biopolymers and petropolymers. Some LCAs attempt to answer this question through a normalization process to show whether or not different impact categories are significant. With ecoinvent system boundaries only from cradle to granulate (or kg of starch in the case of TPS), biopolymers do not exhibit a clear win or lose across any of the environmental indicators when compared to petroleum polymers. It is not clear that PLA and TPS are 'better' or 'worse' within the acidification, smog formation, ecotoxicity, carcinogen or respiratory categories. PLA's ecotoxicity impacts range from 3 times greater than PP and only 1.2 times greater than PET.

However, PLA and TPS result in higher eutrophication and ozone depletion impacts than their petroleum counterparts. Finally, TPS non-carcinogenic human health impact results from ecoinvent and TRACI may be overestimated by a factor of four as described in the sensitivity analysis.

Review of Published LCAs

Table 2 summarizes LCAs of biopolymers published in the peer-reviewed literature through 2012. The environmental performance of PHA, PLA, and TPS has primarily been evaluated from the cradle to the production of resin or pellet with fewer than half of the studies including EOL in the system boundaries (Table 2). The studies compare biopolymers to petroleum-based polymers on the basis of GWP and nonrenewable energy use, while other EPA criteria air pollutants and nonpoint aqueous emissions are included in only a few of the studies. Figure 7 summarizes the distribution of environmental impact areas that were evaluated in recent biopolymer LCAs. Of the twenty-one studies reviewed, nearly all of them evaluated GWP and energy use, while other areas of impact were largely ignored with fewer than 25% of the studies evaluating ecosystem quality, eutrophication, human health, land use, or water use. The emphasis is clearly on greenhouse gas emissions and energy use. LCAs that assess only GHGs and nonrenewable energy consumption may miss potential unintended consequences resulting from switching from petro to biopolymers. For example, bio-based products have been shown to exhibit tradeoffs in the form of decreased greenhouse gas emissions while experiencing increased water quality degradation (Miller, Landis, and Theis 2007). Biopolymers may outperform petroleum-based polymers, but that cannot be confirmed unless LCA practitioners are creating clear assessments of the environmental impacts of both petroleum-based polymers and their bio-based counterparts.

Table 2. Impact categories and system boundaries of previous biopolymer LCAs. End of life (EOL) scenarios: I = incineration without energy recovery, IR = incineration with energy recovery, L = landfill without methane capture (* indicates the study assumed no methane production), LC = landfill with methane capture, R = recycle, T = litter, C = compost.

Study Details			Life Cycle Impact Assessment Categories Reported							System Boundaries, Cradle to . . .			
Author	Date Accepted	Product	Global Warming	Fossil Fuels/Energy	Ecosystem Quality	Eutrophication	Human Health	Land Use	Water Use	Resin	Pellet	Product	EOL
Gerngross	August 2000	General		■						■			
Dornburg	July 2003	General	■	■				■		■	■	■	I
Kijchavengkul	February 2008	General	■	■						■			
Tabone	September 2010	General	■	■	■	■	■			■	■		
Heyde	July 1997	PHB	■	■						■	■	■	I, L, R
Kurdikar	July 2000	PHA	■	■						■			
Akiyama	November 2002	PHA	■	■						■	■		
Kim	June 2004	PHA	■	■	■	■	■			■			
Pietrini	April 2007	PHB	■	■						■	■	■	IR
Yu	July 2008	PHA	■	■						■			
Zhong	November 2008	PHA	■	■	■		■			■			
Kendall	January 2012	PHB	■	■						■			
Vink	November 2002	PLA	■	■					■	■	■		
Bohlmann	November 2004	PLA	■	■						■	■	■	L*, LC
James	June 2005	PLA	■			■				■	■	■	C, LC, R, T
Vink	April 2007	PLA	■	■						■	■		
Madival	March 2009	PLA	■	■	■	■	■	■		■	■	■	IR, L*, R
Vink	June 2010	PLA	■	■						■	■		
Groot	July 2012	PLA	■	■	■	■	■	■		■	■		
Shen	May 2008	TPS	■	■						■	■	■	IR
Piemonte	December 2010	TPS	■					■		■	■	■	

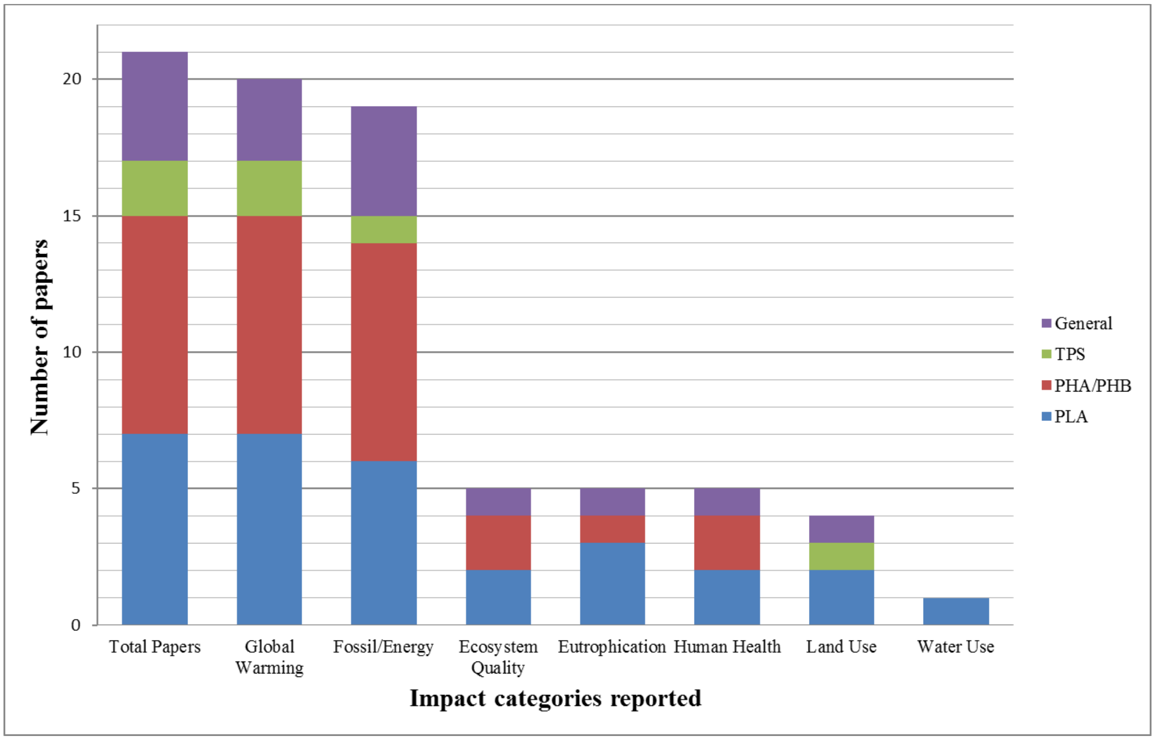

Figure 7. Impact categories included in peer reviewed biopolymer LCAs. 'General' refers to studies that looked at biopolymers in general without focusing on TPS, PHA, or PLA.

The LCAs summarized in Table 2 run the gamut in regards to system boundaries, research goals, product types, assumptions, and EOL scenarios. These are important differences that help to explain the variability of the results and determine which studies should inform the larger debate. A majority of the LCAs were conducted to evaluate specific products, such as PLA (Vink, Davies, and Kolstad 2010, Dornburg, Lewandowski, and Patel 2003, Heyde 1998, Pietrini et al. 2007, Zhong, Song, and Huang 2009, Vink et al. 2003, Bohlmann 2004, James 2005, Madival et al. 2009, Shen and Patel 2008). Other studies focused primarily on the manufacturing process (Akiyama, Tsuge, and Doi 2003, Yu and Chen 2008, Kendall 2012, Vink et al. 2007), the types of feedstocks being used (Kurdikar et al. 2000, Kim and Dale 2004, Groot and Borén 2010), the overall sustainability of the polymers (Tabone et al. 2010, Gerngross and Slater 2000), the compostability of products made from biopolymers (Kijchavengkul and Auras 2008), or the land use change associated with the production of biopolymers (Piemonte and Gironi 2011).

Similarly, when the system boundaries extended to the product or end-of-life, there were significant differences in the types of products assessed; Heyde, James, and Piemonte looked at plastic bags (Heyde 1998, James 2005, Piemonte and Gironi 2011), Bohlmann and Madival looked at containers for food (Bohlmann 2004, Madival et al. 2009), and Shen assessed generic packaging (Shen and Patel 2008), all of which would be considered non-durable goods or packaging. The study by Pietrini looked at monitor casings and car panels and was the only LCA reviewed that evaluated durable goods (Pietrini et al. 2007).

The remaining figures are related to GWP reported for the biopolymers in these studies. Aside from greenhouse gases being significant factors in the environmental impacts of a product, GWP was the most consistent measure used in the studies and for which most of the data could be converted to a single unit of measure for comparative purposes. Of the twenty-one studies, fifteen reported data that could be used in the subsequent comparisons.

Figure 8 illustrates the range of reported CO_2 equivalents among fifteen LCAs published over a twelve-year period. Of the twenty-one studies, fifteen reported data that could be used in these comparisons, while the remaining six did not report in values that could be converted to CO_2 equivalents. Study (f), (l), and (n) provided single data points while the others provided ranges due to scenario changes and uncertainty. Figure 8 also illustrates the overall range of CO_2 equivalents reported among the reviewed LCAs of biopolymers that included greenhouse gases in their life-cycle inventory. The TPS data was sourced from a limited number of studies and does not indicate a very broad range.

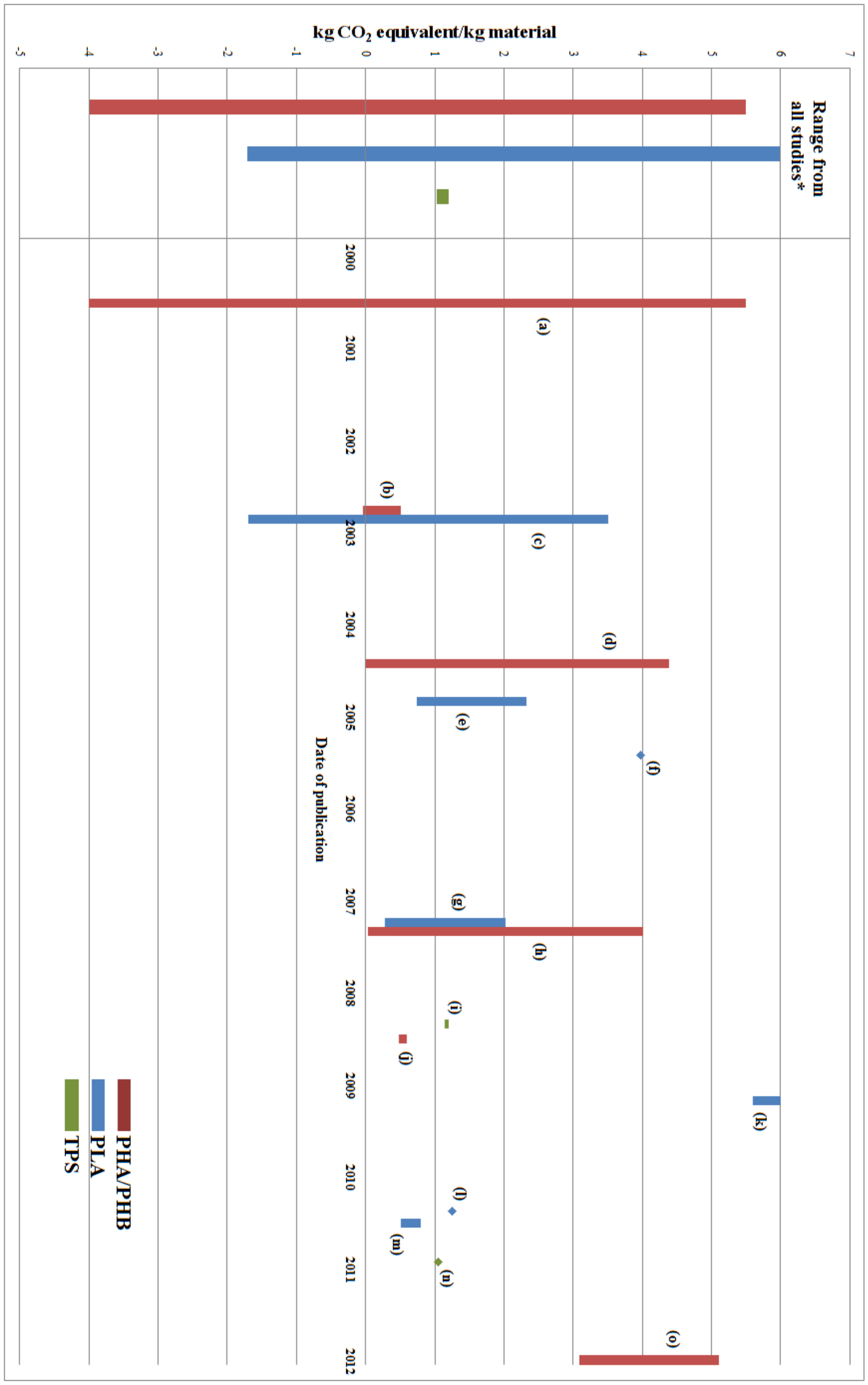

Figure 8. Reported LCA ranges of CO_2 emissions equivalents for PHA, PLA, and TPS from the reviewed studies. *Each study had different system boundaries and scenarios that contributed to this range. (a) Kurdikar, (b) Akiyama, (c) Vink, (d) Kim, (e) Bohlmann, (f) James, (g) Vink, (h) Pietrini, (i) Shen, (j) Yu, (k) Madival, (l) Vink, (m) Groot, (n) Piemonte, and (o) Kendall.

The system boundaries and assumptions within the studies significantly influence the GWP results reported in Figure 8. For example, transportation emissions and EOL scenarios in Madival's study (k) resulted in significantly higher GWP numbers than the other LCAs of PLA that omitted these emissions.

Likewise some of the lower values reported can be attributed to the use of alternative energy scenarios in the LCAs; this is particularly visible in two studies by Vink (c, g) and the one by Groot (m). Vink's use of wind power and renewable energy certificates (RECs) in some scenarios results in the lower range of GWP reported for the biopolymers (Vink et al. 2003, Vink et al. 2007). Vink's most recent paper does not incorporate wind energy offsets in this way (Vink, Davies, and Kolstad 2010). Similarly, the study by Groot (m) includes carbon offsets from burning left over biomass for energy, reducing the GWP of PLA by offsetting the need for fossil fuels in the manufacturing process (Groot and Borén 2010).

Kurdikar's wide range of findings for PHA (a) are a result of divergent scenarios that assume single source heat and power generation with biomass power resulting in net negative carbon emissions. Kurdikar's coal scenario topped out the charts with around 5.5 kg CO_2 equivalent per kilogram of PHA (Heyde 1998). Kurdikar's early study reports both the upper and lower limits in Figure 8 for the LCAs of PHA/PHB. The later studies, able to refine their scenarios and system boundaries, provided greater certainty with their results.

The system boundaries of the studies and which EOL scenarios, if any, were included in the LCA can greatly influence the GWP results, which is explored in more detail in Figure 9 which depicts the average of the fifteen studies included in Figure 8, breaking them up based on the system boundaries used in the LCAs. Though the EOL scenarios are different in each LCA, Figure 9 demonstrates that when EOL is included in the system boundaries of an LCA for PHA and PLA, the GWP of the products increases up to 12-fold and 6-fold from the pellet, respectively. Table 2 shows that three of the LCAs included only one method of disposal while the other four studies evaluated two or more methods of disposal. Furthermore, methane capture or lack thereof during landfill scenarios (Spokas et al. 2006) and uncertainty regarding composting methods, which may include too much moisture or not enough oxygen leading to methane emissions (Vink et al. 2003), may also drastically affect the greenhouse gas emissions from the biopolymers.

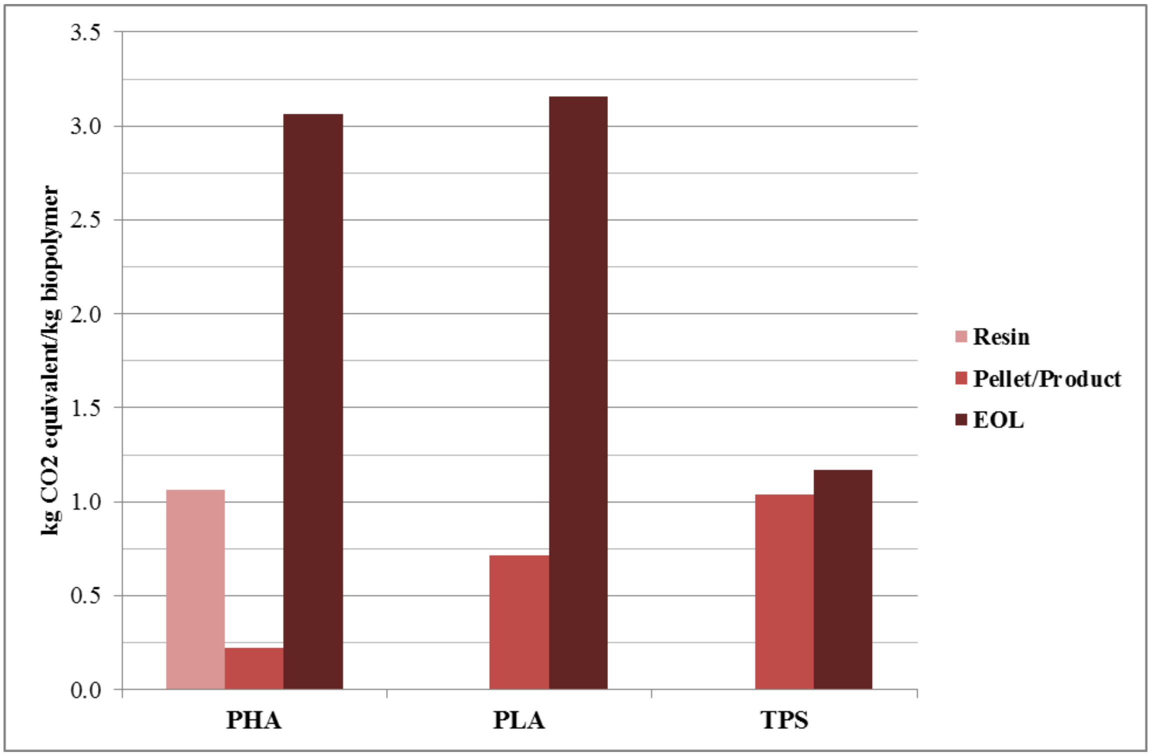

Figure 9. Average global warming potential of biopolymers based on the extent of the system boundaries of the fifteen different LCAs assessed in this study.

Including EOL provides a more comprehensive estimate of the life-cycle environmental impacts of biopolymers. However, the specific method of disposal may also provide varied results. In fact, despite the compostability of many biopolymers, the predominant method of plastics disposal in the US is recycling or landfilling. Few US cities have compost infrastructure. A sensitivity analysis of EOL studies is important to ensure the results do not hinge on a single assumption, especially when the assumption is based on consumer behavior. Additionally, the scenarios must be representative of how the biopolymers are being disposed of or viable alternatives. Madival provided a nice example of scenarios for comparison: Scenario 1) 40% recycling, 30% incineration, 30% landfill, Scenario 2) 100% landfill, Scenario 3) 100% recycling, Scenario 4) 50% incineration, 50% landfill, and the Current Scenario for the US – 23.5% incineration, 76.5% landfill (Madival et al. 2009). These scenarios provide a wide variety of EOL combinations in order to explore the upper and lower limits of the impacts that result from the different methods of disposal.

Summary

Life-cycle Assessment is a tool that can quantify the environmental impacts of biopolymers. However, the environmental impacts associated with the creation, use, and disposal of biopolymers remains unclear, since biopolymers can be made into a variety of products for a variety of uses, and ultimately are disposed of in many different ways. One role of LCA practitioners is to identify current production benchmarks and to analyze future scenarios to help guide the development of manufacturing, use, and disposal for sustainable products.

There are other factors and tradeoffs that are important to the life-cycle environmental impacts of biopolymers beyond a simple cradle to gate LCA study. The end-of-life options for biopolymers may prove to alter the product's environmental profile. The general assumption is that biopolymers quickly degrade and recycle their carbon to other plants via biological recycling. However, recommendations for waste management of PLA state that the material should be composted at industrial facilities since the appropriate degradation conditions cannot be met in the typical ‘backyard’ compost pile (Vink et al.

2004, Gruber and O'Brien 2002, Kale, Kijchavengkul et al. 2007). Industrial composting may add additional cost and transportation dimensions to the disposal of compostable products that make composting more comparable to a landfilling scenario which may further decrease any GWP benefits that were achieved. Despite the ability to compost most biopolymers, landfilling is currently the dominant waste management method in the US, and a major effort would be required to change personal disposal habits, including education of consumers and infrastructure for composting (Denison 1996, Gross and Kalra 2002, Sartorius 2002). While PLA has been shown to be stable in landfills (Kolstad et al. 2012), landfilling may alter the environmental profile of other biopolymers with the potential for methane emissions and uncertainty surrounding the capture of landfill gas (Spokas et al. 2006).

Biopolymers can be made to be biodegradable and compostable, but most US cities do not have composting infrastructure. The LCA by James was the sole study that included composting as an EOL option (James 2005). This highlights a major discrepancy between the feature of compostability and LCAs that simply analyze current polymer disposal practices rather than exploring the potential of creating new waste management pathways that may provide future environmental benefits or unintended negative consequences.

Little work has been done to assess the recycling options for biopolymers using LCA methodologies. It has been shown that of the different disposal options for petroleum-based products, recycling offers substantial environmental advantages over other alternatives (Denison 1996, Patel, von Thienen et al. 2000). However, proper labeling and handling procedures for recycling facilities will be needed to ensure that biopolymers do not foul other recycling streams. Recycling may provide other life-cycle benefits by reducing feedstock requirements and energy input (Piemonte 2011). However, similar to composting, the current infrastructure and logistics required may be a barrier to recycling of certain biopolymers like PHA and PLA. As noted by Song and colleagues, "although it is feasible to mechanically recycle some bioplastic polymers such as PLA a few times without significant reduction in properties, the lack of continuous and reliable supply of bioplastics polymer waste in large quantity presently makes recycling less economically attractive" (Song, Murphy et al. 2009). While this is a problem with a number of recyclable materials, as the biopolymer market continues to grow it may eventually became economically feasible to recycle biopolymers, which in turn could have a beneficial impact on the environmental performance of these products. Organic contamination caused by the use of biopolymers in food packaging further complicates the recycling of biopolymers (Razza, Fieschi et al. 2009), but the ability to compost them provides alternative waste management strategies when compared to traditional plastics. The optimal disposal scenario could involve incineration, composting, recycling, landfilling, or a combination of the aforementioned alternatives and may be different depending on the type of biopolymer.

Many of the EOL scenarios reported in the literature are based on typical MSW ratios for plastics, but with most disposable biopolymers being advertised as compostable, it is unclear how consumers are disposing of these biopolymers. In addition, it is not clear what mix of EOL scenarios would provide the greatest benefits to biopolymers' life-cycle environmental impacts. While there have been studies clearly stating that biopolymers will not break down under landfill conditions, particularly PLA (Kijchavengkul and Auras 2008, Madival et al. 2009, Kale, Kijchavengkul, et al. 2007, Kolstad et al. 2012), others still report methane emissions from landfilling as an impact in the life-cycle for these products (Heyde 1998, Bohlmann 2004, James 2005, Groot and Borén 2010, Shen and Patel 2008), thus increasing the GWP associated with biopolymers. Scenario development and sensitivity analyses can help identify where assumptions such as methane emissions from landfilling of biopolymers have significant impacts and in what areas additional research is needed to provide more accurate data for the LCA.

It is important for LCA practitioners to present transparent data and assumptions and to provide the necessary context of their findings in order to accurately quantify the impacts from biopolymers and traditional plastics as well as to aid the industry in making effective improvements. Databases, while convenient, must be scrutinized for accuracy and completeness. As production methods change, updates to inventories must be made. For example, Vink has continued to provide updated profiles for the creation of NatureWorks' PLA which should be incorporated into any future LCAs of that product (Vink, Davies, and Kolstad 2010). The quality of LCA findings is not only a function of determining appropriate system boundaries but is also determined by the quality of the inventory data. The ecoinvent database with TRACI produced results that were not commensurate with the results from NatureWorks (Vink, Davies, and Kolstad 2010), as described in the review of environmental impacts of polymers. These types of discrepancies result from a wide range of variables, namely differences in system boundary selection, in methods of calculation for environmental impacts, and the quality of the inventory data. All of these factors and assumptions can influence resultant impacts like GWP which, depending on system boundaries, may exclude carbon uptake by plants or ignore the GHG emissions that result from end of life treatments. Further, the environmental impacts of polymer production estimated in peer-reviewed LCAs are not consistent throughout the literature. Exploring the impacts of the polymer industry will help determine which impacts are relevant to the discussion and those that prove to be insignificant.

The potential benefits of biopolymers in regards to GWP will not be realized until the material and energy demands from the farming and production processes are reduced. The use of REC's in Vink's 2003 and 2007 studies of PLA demonstrate the power of a low fossil fuel energy paradigm combined with carbon negative feedstocks (Vink et al. 2003, Vink 2007). There is a great potential to sequester atmospheric carbon into everyday material. As noted by Gerngross and Slater, "any manufacturing process, not just those for plastics, would benefit from the use of renewable energy" (Gerngross and Slater 2000). The mitigation of the other environmental impacts, such as water quality degradation from agricultural practices, will further enhance the environmental profile for biopolymer products.

Biopolymers are relatively new to the market when compared to their petroleum counterparts. The industry has made significant gains over a short period of time (Vink et al. 2004, Vink, Davies, and Kolstad 2010, Vink et al. 2003, Vink et al. 2007), and any comparison between biopolymers and fossil-based polymers must take these technology improvements and productivity into account. The fact that biopolymers are currently on par with traditional plastics means that further technology improvements and economies of scale have the potential to tip the scales in favor of biopolymers. Environmental impacts resulting from agricultural production will need to be managed in order to maintain and improve any benefits gained by transitioning to bio-based production. Better agricultural nutrient management practices or the development of new feedstocks that require minimal energy and nutrient inputs are two ways to improve the impacts from agriculture (Landis, Miller, and Theis 2007, Mohanty, Misra, and Drzal 2002). Taking a whole-systems approach to designing sustainable biopolymers will lead to a path of biopolymers with clear life-cycle environmental benefits compared to their petroleum counterparts. However, without a clear understanding of the distribution for biopolymers in waste streams, the resultant cradle to grave life-cycle environmental impacts of biopolymers remain uncertain. Life-cycle analysis will continue to be a useful tool to identify more sustainable methods of production, use, and disposal of biobased products.

CHAPTER 3

TOWARD ZERO WASTE: COMPOSTING AND RECYCLING FOR SUSTAINABLE VENUE BASED EVENTS

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This chapter addresses research objectives 1) Quantify end of life of waste flows of biopolymers via waste audits at public venues and waste handling facilities, and 2) Identify best practices for facilities management and waste handling of compostables via surveys and focus groups

Introduction

Venue-based events are a major part of Americans' lifestyles and often include food and drink services which generate a significant amount of waste. There were over 125 million people who attended NBA, NHL, MLB, and NFL games in the 2013 seasons (ESPN 2014a, c, b, d) while domestic movie theater attendance exceeded 1.3 billion in 2013 (Nash Information Services LLC 2014). These figures do not include non-professional sporting events, festivals, fairs, concerts, and other theaters all of which are venue-based events. The food and drink services provided by all of these venues represent a large amount of disposable products which must enter the waste stream. As institutions shift toward more environmentally friendly operations and aim for zero waste facilities, compostable products and alternative material pathways like composting and recycling become more attractive. These options can complicate decision making as facility managers must consider factors including source material selection, employee training, public awareness, simplicity of collection, and environmental tradeoffs for different approaches to waste management.

Sporting events and other venue based activities have increasingly become a focus for environmental improvement through sustainability initiatives. The environmental assessment of major sporting events, such as the Olympics and international championships, has become an important factor in determining where and how these events are hosted (Collins and Flynn 2008, Collins, Jones, and Munday 2009). Sustainability efforts at sporting events in North America have been spearheaded by the Natural Resources Defense Council (NRDC) and the Green Sports Alliance who view waste management as one of the key areas of focus for improvement of environmental performance (Hershkowitz, Henly, and Hoover 2012). Waste management has also been highlighted as a major operational focus for event managers looking to incorporate more sustainable practices. Not only can managers ensure the impacts of events are lessened but that "the event itself can be used to promote a green message" (Laing and Frost 2010). This suggests that events may serve as a model of public waste disposal in the same way that behavioral models function, altering the way people behave collectively. Similar efforts aimed at waste reduction have been evaluated for airports and air travel, which is functionally similar to venue based events because of the captive audience and the control over what types of materials enter the waste stream (Coggins 1994, Li, Poon et al. 2003, Pitt and Smith 2003).

This research seeks to address public waste generation at venue based events through a two-pronged approach, (1) evaluate total waste generation from venues to model alterative handling scenarios and (2) evaluate different methods to reduce contamination in recycling and compost. Modeling was conducted using USEPA's Waste Reduction Model (WARM), which is designed to assess greenhouse gas emissions and energy impacts related to material types and waste management practices (USEPA 2014b). WARM was developed to help waste planners model impacts by selecting material inputs and providing model parameters which include transportation distances and landfill conditions (e.g. methane capture efficiency and landfill moisture content). WARM enables scenario development to create comparisons between differing approaches to waste treatment and the subsequent impacts.

The two-pronged approach focuses on the evaluation of waste scenarios that incorporate more compostable products and the public's ability to adjust to new methods of collection through a case study at Arizona State University's (ASU) Packard Stadium at the Tempe, Arizona campus where collegiate baseball games are played. ASU's goal is to become a "zero waste institution" producing no landfill wastes, which corresponds to a larger regional effort by the City of Phoenix to achieve a 40% reduction in landfill wastes by the year 2020 (ASU 2014, Phoenix 2013, Reid 2013).

Waste handling is an actively changing system with the introduction of new types of waste and new approaches to waste management that can alter impacts on a broad scale. Biodegradable and compostable plastics that meet the ASTM and ISO standards (ASTM 2003a, b, 2004, ISO), such as PLA, may prove to be useful in reducing landfill wastes but the composting for these materials needs to happen at an industrial facility. The requirements for the degradation of these products requires sustained temperatures and biological activity that are difficult to reproduce in home composting systems (Song et al. 2009). Additionally, PLA, the most common compostable biopolymer in the US, can be recycled but the current quantity and flow of the material is not significant enough to be financially competitive with landfill disposal (Song et al., 2009). Cradle-to-gate life-cycle environmental impacts of biopolymers are similar to traditional petroleum based plastics; however, end of life has not been well studied. The few life-cycle assessments of biopolymers which include waste management, use generic scenarios based on traditional plastics, with even fewer assessments including composting as an option despite the fact that recycling is not viable and composting as a waste management option is one of the main benefits of these products (Hottle, Bilec, and Landis 2013). Disposal is a key area of inquiry as it is a large source of uncertainty and is largely unexplored for these products.

Consumer involvement in waste systems dictates how materials enter the different waste streams. This is true for residential and commercial waste handling and public collection. The social components of waste management are complex, including factors like convenience, perceived efficacy, consumer awareness, outreach, and participation; ultimately resulting in patterns of consumer behavior. Behavior change is a central factor, necessary for shifting to more sustainable waste management but there is a lack of research with regards to behavior change interventions (Zhang et al. 2011). Sussman and Gifford, 2013 and Sussman et al., 2013 found that prompts, such as signage, and people modeling how to sort compost in public settings have a significant influence on the behavior of the people around them. Additionally, they found that the positive behavior changes persisted even after the behavior models were removed from the research setting (Sussman and Gifford 2013, Sussman et al. 2013). These studies, which have investigated behavior, inform the development of new waste collection strategies in public venues and highlight the benefits of an approach that incorporates techniques to encourage positive behavior change through social dynamics and traditional outreach.

Methods

The methods are divided into three specific areas that were integral to this research. The first section, raw data collection, describes the process of conducting the waste audits including weighing, sorting, and determining the mix of materials in the waste stream. The second section, modeling and scenario analysis, describes the parameters used for the WARM tool and the seven scenarios that were modeled for the assessment. The final section, waste disposal behavior at public events, describes the use of signage and volunteers to aid in material identification that were used throughout the four games so that behavioral factors could be assessed.

Through a partnership with ASU Athletics and University Sustainability Practices (USP) waste audits were conducted at four university baseball games. Events like the ASU baseball games provide an opportunity to run large experiments with reasonable control since most of the waste materials being handled are generated within the game and are limited to products that vendors offer. This research tracked the efficacy for different methods of waste collection that have been implemented at the games as well as determining the quantity and composition of the waste. ASU's Packard Stadium has over 3000 grandstand seats and can accommodate over 7800 attendees, with an average attendance of 2809 per game in 2013 (Cutler 2013).

As a part of ASU's zero waste effort, Packard Stadium has moved to a two bin collection system offering receptacles for composting and recycling only. The bins are colored to help distinguish the two material streams, blue for recycling and green for composting (Figure 10). In addition to the bin color, ASU has attached signs to the front of each bin which labeled and described which materials belong in each bin. The composting signs listed food, liquid, napkins, plates & cups, and compostable spoons & forks. The recycling signs listed paper, plastic, aluminum, glass, and cardboard. Using EPA's WARM (USEPA 2014b) this research explored the dynamics between different management scenarios, carbon emissions, and energy use.

Figure 10. Compost and Recycling containers being staffed by a volunteer bin guard at ASU's Packard Stadium.

Data collection occurred over a span of four baseball games at Packard Stadium, three of which were in a consecutive weekend series. Figure 11 charts raw data collection and the subsequent analyses of this research which addresses two separate areas of inquiry: (1) Characterization of game-day wastes (quantification and categorization) and the assessment of different management scenarios using WARM and, (2) The public's ability to adjust to new methods of material collection at venue-based events.

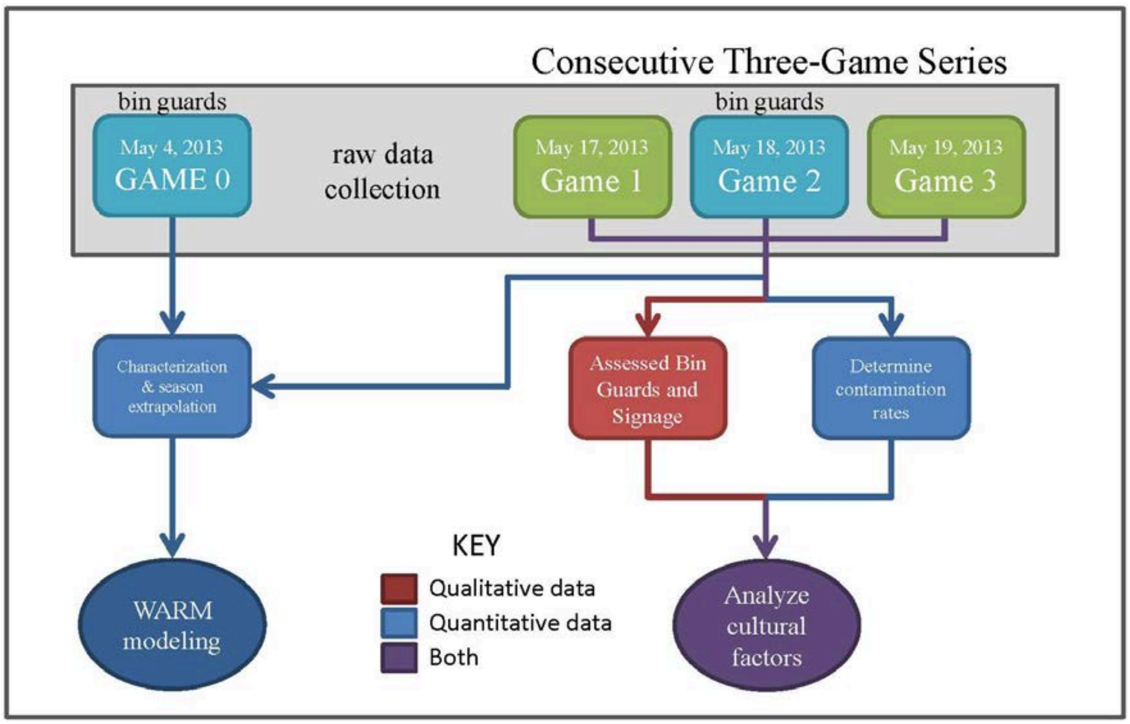

Figure 11. Data collection from waste audits at four baseball games. The flow diagram depicts quantitative data collection and modeling as well as the qualitative analysis from the consecutive series.

Raw Data Collection

Materials from the compost and recycling bins were weighed for each of the four games. At the games, attendees were required to discard all waste from outside of the stadium before entering the venue, so nearly all of the bin materials in the stadium originated from vendors in the stadium. The bin materials were collected and weighed when full bags were removed during the game and at the end of the game when the bins were no longer in use. The material was weighed using a CPWplus L Floor Scale with a capacity of 75 kg and accurate to 20 g. The goal for the audit was to collect at least 50% of the waste generated at the games. The audits were successful, achieving an average of 66% of the bin waste from each game was collected, sorted, and weighed. The materials were sorted by hand, classified as either compostable or recyclable, and compared with the bin type in which it was found to determine the efficacy of the collection strategy. Data collection then followed a step by step process to record weights, contamination rates, and material types.

The initial weight of each sampled bag was recorded followed by a sorting process to remove any contamination. An average bag weight was used to calculate an estimated mass for the unsorted bags from each game. After sorting, both the non-contaminated material and the contaminating materials were reweighed. Some products like polystyrene, plastic films, and foils (i.e. chip bags), which foul both the material streams, were categorized as contamination regardless of where they were placed. The composition, or the percentages of which types of materials make up the items in the bins, was estimated and recorded one bag at a time with the assistance of an ASU Recycling Technician. Estimation was used to determine composition in order to reduce sort time (three measurements of mass per bag as opposed to eleven) and because direct measurements of mass would have been confounded by food waste contamination which tended to stick to some materials (paper and cardboard) more than it would to others (plastic and polystyrene). The waste compositions included eight categories of managed waste which correspond to the WARM tool; aluminum, high density polyethylene (HDPE), polyethylene terephthalate (PET), polypropylene (PP), corrugated containers, newspaper, mixed paper, and mixed organics.

Modeling and Scenario Analysis

The data for the mass of material generated at each game was used to estimate total material generation numbers for the games and the full season. The average waste generation of a baseball game at Packard Stadium was calculated by extrapolating the total material generation for each game then averaging the totals. This average was then multiplied by the games played in the stadium over the 2013 season, giving an estimate of total generation for a season. A ninety-five percent confidence interval was determined for game data and the season total to determine the uncertainty involved in the calculation of the data. Total seasonal generation was used along with the composition data (percent of each waste type) for the assessment of different handling scenarios to annualize the impacts for comparison purposes.

USEPA's WARM was used to calculate CO_2 equivalent emissions and energy use that result from different waste management strategies. The energy calculations in WARM are limited to transportation, direct use, and generation. The model is designed to compare baseline strategies to alternative approaches and calculates emissions accordingly. The results from WARM, which was designed to help inform decision making on the municipal scale, are the total impacts calculated and not the difference between a baseline scenario and an alternative. The model assumes the end of life treatment inputs represent the end of life treatment impacts excluding the possibility of some materials being rerouted to other methods of treatment.

All of the inputs for WARM were selected based on correspondence with ASU's waste service provider (McLaren 2013). The preset Arizona electricity mix and current mix for source reduction were used. The following inputs were selected to represent the Butterfield Station Landfill, located in Mobile, Arizona, southwest of Phoenix; the landfill is considered dry (which means organics do not degrade as quickly and generate less methane overtime) and has gas recovery systems that fall under the "typical operation" category for WARM. The methane gas is flared at Butterfield to meet regulatory compliance and all landfill scenarios included flared gas in the modeling. Transportation distances were estimated using Google Maps to represent the transport of waste to the tipping floor (where collection trucks are dumped and a decision is made determining the final destination for that waste) and the subsequent management sites. The distances are: landfill – 45 miles, recycling – 41 miles, and composting – 35 miles. Incineration, the combustion of wastes, is not part of the local waste management regime and was not used in the model.

A scenario analysis was conducted to assess the variability associated with the model parameters. Each scenario described in the next paragraph and Table 3 was also analyzed using WARM inputs for national averages of gas recovery landfills which flare their gas. Additionally, the default WARM distance for management sites of 20 miles was used. This comparison highlights the categories that are most influenced by a change in the model parameters emphasizing the need for attention to accuracy within models and the importance of decision making in the real-world.

Table 3. Waste management scenarios by percent mass. The first three scenarios represent the current waste mix, which has no PLA. The subsequent scenarios investigate methods of altering the materials in the waste stream for improved performance.

Seven Scenarios for Waste Management

	1	2	3	4	5	6	7
Material Type	Landfill Only	Landfill/ Recycle	Compost/ Recycle	Aluminum, PET, PLA Landfill	Aluminum/ PET Recycle, PLA Compost	PLA/ Organics Landfill	PLA/ Organics Compost
Aluminum Cans	2.7%	2.7%	2.7%	2.7%	2.7%		
HDPE	8.1%	8.1%	8.1%				
PET	21.6%	21.6%	21.6%	21.6%	21.6%		
PP	10.8%	10.8%	10.8%				
Corrugated Containers	2.7%	2.7%	2.7%				
Newspaper	2.7%	2.7%	2.7%				
Mixed Paper	5.4%	5.4%	5.4%				
Mixed Organics	46.0%	46.0%	46.0%	56.8%	56.8%	56.8%	56.8%
PLA				18.9%	18.9%	43.2%	43.2%

landfill | recycle | compost

The seven scenarios were developed with USP and the ASU Recycling Technicians to represent realistic options for waste handling. The treatment methods are associated with tradeoffs beyond the impacts measured within the model and are the topic of debate within USP and among the Recycling Technicians. While a single stream system (e.g. an all compostable waste stream) simplifies collection and achieves high rates of landfill diversion, mixed stream systems including recycling provides economic return for the university but increases the amount of waste sorting required. The scenarios described below represent three options for waste mixes and possible treatment options across the three different mixes, resulting in seven scenarios.

The seven scenarios for managing waste were explored using the WARM model (Table 3). They included management approaches for the current waste mix and two alternative mixes that substitute a varying amount of PLA for plastics and metals. The first three scenarios, (1) landfill only, (2) landfill/recycle, and (3) compost/recycle, were options for the management of the current waste composition at Packard. Scenario 1 and 2 are typical approaches to waste handling while Scenario 3 represented the actual approach to waste management at Packard Stadium.

The next two scenarios, (4) aluminum, PET, PLA landfill and (5) aluminum/PET recycle, PLA compost, explored the possible substitution of all non-compostable materials with PLA except for PET and aluminum, which are commonly recycled products (USEPA 2010). The substitution of PLA assumed 100% substitution by mass. The mixed organics waste was not altered. Scenario 4 served as a landfill scenario and Scenario 5 described a compost and recycling approach.

The final two scenarios, (6) PLA/organics landfill and (7) PLA/organics compost, explored the possibility of shifting to a single stream waste handling approach in which all materials are compostable. Scenario 6 modeled a landfill approach to this organics only waste stream, while 7 looked at the possibility of all the materials being composted. In these scenarios PLA replaced plastics and aluminum while paper and cardboard were shifted into the mixed organics category. These scenarios were created to test a single stream approach while still achieving a 100% diversion rate. WARM does not account for composting as a waste management approach for paper or cardboard, but the EPA recommended using the mixed organics category for this modeling (USEPA 2013a). All the other compostable materials have specific categories for composting in WARM. The inclusion of a landfill scenario for the primary mix and the two alterative mixes provides a comparison for materials that may not end up at the expected end of life treatment.

Waste Disposal Behavior at Public Events

In addition to modeling impacts from waste handling, this research sought to investigate the behavior of composting in public venues by exploring people's understanding of and willingness to segregate material types for some perceived environmental benefits. Behavior herein is defined as the collective attitude and actions of people attending the ballgames toward waste disposal. The ability and willingness of attendees to change disposal habits was gauged through both quantitative and qualitative assessments of behavior.

Two approaches were developed by ASU's USP office for helping consumers identify in which bin to put their wastes were used to determine efficacy of the different approaches. The first was signage on the recycling and composting bins that describe what material types are appropriate to put in each bin. The second approach uses volunteer "bin guards" who received ten minutes of training prior to each game. Bin guards stood next to each waste station and aided ballpark visitors with their decision making when they had waste to throw away (see a bin guard at Packard Stadium in Figure 10). Each volunteer had a lanyard with a card to help them identify what types of materials go into each bin. Bin guards were instructed to put all inherently contaminating materials (e.g. polystyrene, plastic films, and foils) into the recycling bins because the recycling system is less sensitive to contamination than the composting system.

The three game series between ASU and the University of Arizona was used to evaluate changing consumer behavior with education and outreach efforts (Figure 11). The first game in the series, May 17, 2013, served as a control with signs on the bins only. The second game, May 18, 2013, tested whether there was increased efficacy of collection (i.e. decreased contamination) when volunteer bin guards served by assisting patrons with sorting. And, finally, the third game, May 19, 2013, returned to a sign only approach to determine if game attendees' behavior had altered in response to their previous experiences that weekend. A learning period could only have occurred if the same audience (or some of it) returned to each game so attendance figures were tracked for each game. These metrics also included the number of season ticket holders and ticket sales for people attending more than one game, however, there was no way to confirm the tickets were used by the same individual for each game.

Results

Raw Data Collection

The baseball game waste audits collected more than 180 kg of material produced over four games. This was an average collection rate of 66% for each game; the remaining bags were counted so total game wastes could be calculated. Table 4 presents the waste audit data that was collected at the games. First the total mass of materials collected during each game is reported under the collected bin material category. This is followed by a breakdown of that data into two categories, wastes correctly placed and contamination. The final section in Table 4 presents the estimated total game wastes as well as recorded attendance data, which are both used to calculate a per capita waste generation figure.

Table 4. Waste audit data for four games. Waste audit collection data from four ASU baseball games at Packard Stadium. *Collected Bin Material* only represents the bin materials that were sampled and weighed at each game, which accounts for 66% of total waste. *Extrapolated total bin material* was calculated by applying the average bag mass to the uncollected bags for each game and totaling that figure with *collected bin material*.

Waste Generation at Four Games

Collected Bin Material					
	4-May-13	17-May-13	18-May-13	19-May-13	All Games
Total (kg)	54.00	35.94	51.52	40.32	**181.78**
Recycle %	44%	55%	59%	60%	**54%**
Compost %	56%	45%	41%	40%	**46%**

Recycle					
	4-May-13	17-May-13	18-May-13	19-May-13	All Games
Total (kg)	23.78	19.66	30.46	24.10	**98.00**
Correct (kg)	20.16	14.06	26.14	18.70	**79.06**
Incorrect (kg)	3.62	5.60	4.32	5.40	**18.94**
Contamination	15%	28%	14%	22%	**19%**

Compost					
	4-May-13	17-May-13	18-May-13	19-May-13	All Games
Total (kg)	30.22	16.28	21.06	16.22	**83.78**
Correct (kg)	28.72	9.84	19.56	12.56	**70.68**
Incorrect (kg)	1.50	6.44	1.50	3.66	**13.10**
Contamination	5%	40%	7%	23%	**16%**

Extrapolated Total Bin Material and Attendance					
	4-May-13	17-May-13	18-May-13	19-May-13	Averages
Material (kg)	77.14	59.90	90.16	51.80	**69.75**
Attendance	3052	3198	3260	1500	**2753**
Per capita (g)	25.28	18.73	27.66	34.53	**25.34**

Shading indicates bin guards were used

The uncertainty was evaluated by calculating confidence intervals for the extrapolated game wastes for each game and the calculation of seasonal waste generation using the average total of the four games. The confidence intervals for the total mass of each game were ±4.24 kg (5.5%), ±4.99 kg (8.3%), ±10.49 kg (11.6%), and ±2.72 kg (5.2%) respectively. The confidence interval for the game average that was used to calculate the season total was ±16.88 kg (24.2% of the calculated average). The uncertainty regarding the total seasonal waste generation is quite high, however, the composition of material types was assessed across all games by a Recycling Technician for each disposable product category resulting in material ratios that are consistent across each scenario. Since the impacts increase in direct proportionality to the wastes produced, any increase or decrease in the amount of seasonal waste does not result in different relative impacts between the scenarios.

To determine material composition, the types of contamination were also evaluated. Some contamination (7.6 kg, 24% of contamination) was a result of products such as films, foils, and polystyrene which do not belong in either compost or recycle but are still part of the vendors' offered products. However, the majority of contamination (24.4 kg, 76% of contamination) was the result of recyclables in the compost or vice versa. This indicates that improved collection methods or a more developed understanding of composting and recycling could lead to decreased contamination improving the overall diversion rate of material generated at events.

Scenario Analysis Using WARM

Figure 12 presents the WARM findings for CO_2 equivalent emissions of the seven scenarios for the materials that are generated at Packard Stadium over one season. WARM does not include a complete life-cycle assessment of products or waste handling; it accounts for CO_2, CH_4, N_2O, and perfluorocarbons emissions resulting from end of life, waste transportation, and offsets to future production of recycled materials which are reported as CO_2 equivalent emissions (USEPA 2014a, b). The benefits of recycling are seen in Scenarios 2, 3, and 5 where refined products are retained for use in future products, offsetting the emissions associated with processing virgin materials. The PLA landfill Scenarios (4 & 6) result in greater emission reductions when compared to composting (5 & 7) because the WARM model assumes PLA does not breakdown under normal landfill conditions (Kolstad et al. 2012, USEPA 2014a, b), essentially sequestering its carbon; however PLA releases CO_2 when it is composted. Despite the emissions from composted PLA, Scenario 5 outperforms Scenario 4 due to the benefits of recycling aluminum and PET. The organics only Scenarios (6 & 7) showed better performance in the landfill than composting. This difference in performance is due exclusively to the degradation of PLA in the compost while it is assumed to sequester its carbon when in the landfill. This assessment is limited to CO_2 equivalent emissions and does not account for benefits of creating a saleable soil amendment, which should be the focus of additional research.

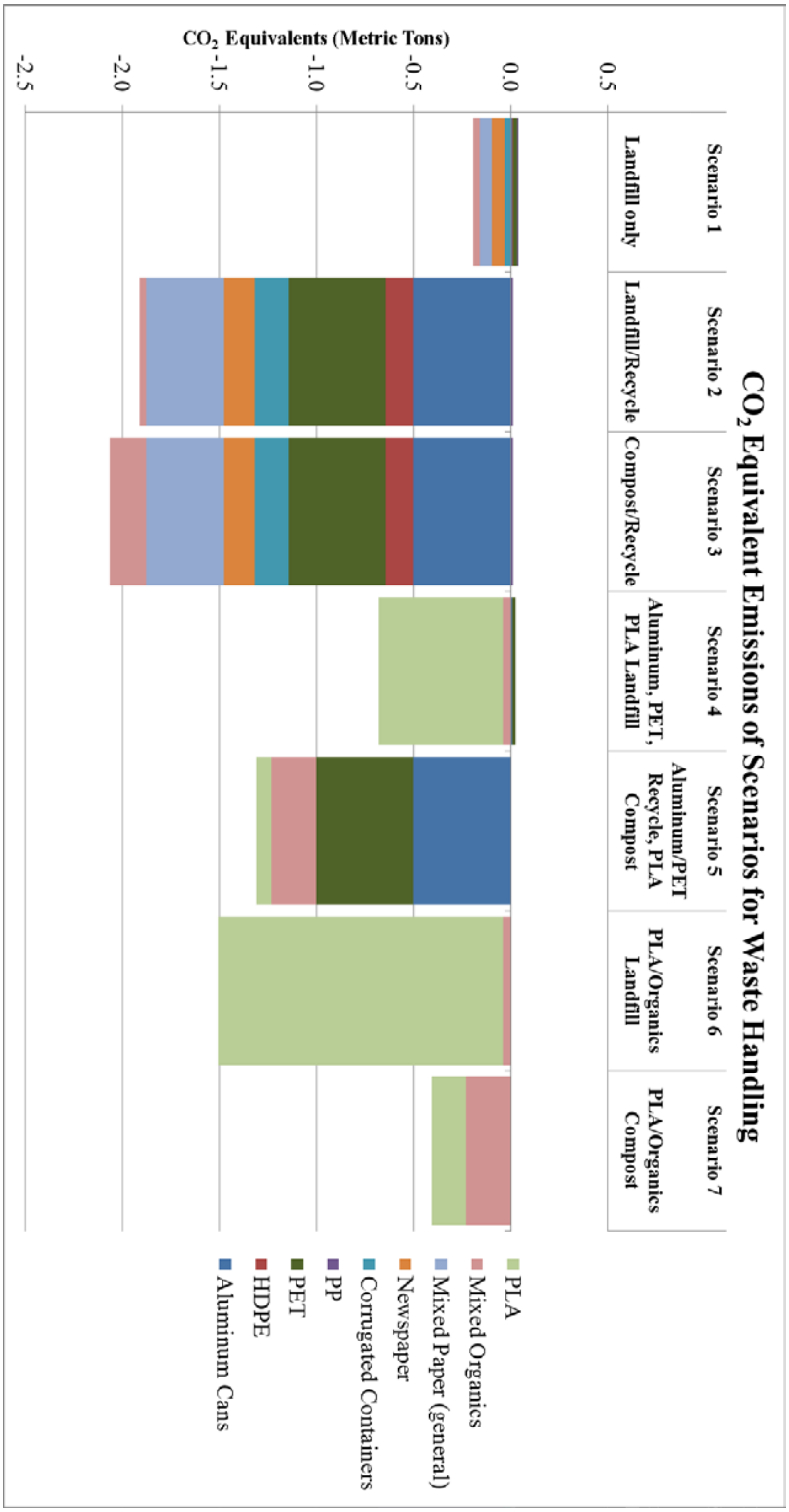

Figure 12. Scenario results of CO_2 equivalent emissions in metric tons by waste type using WARM for one season of waste generated at ASU's Packard Stadium.

Figure 13 presents the WARM findings for energy consumption of the seven scenarios for the waste that is generated at Packard Stadium over one season. Again there is a stark difference between the recycling Scenarios (2, 3, & 5) and the others. The recycling scenarios save energy and the non-recycling scenarios are net energy consumers. While recycling, landfilling, and composting use energy to process the waste material, recycling is the only approach that takes advantage of the embedded energy in the products, offsetting production energy of virgin products. This model does not balance the economics of different strategies or the energetic benefits of compost when applied to land (i.e. reduction of fertilizer use or reduced land use change impacts).

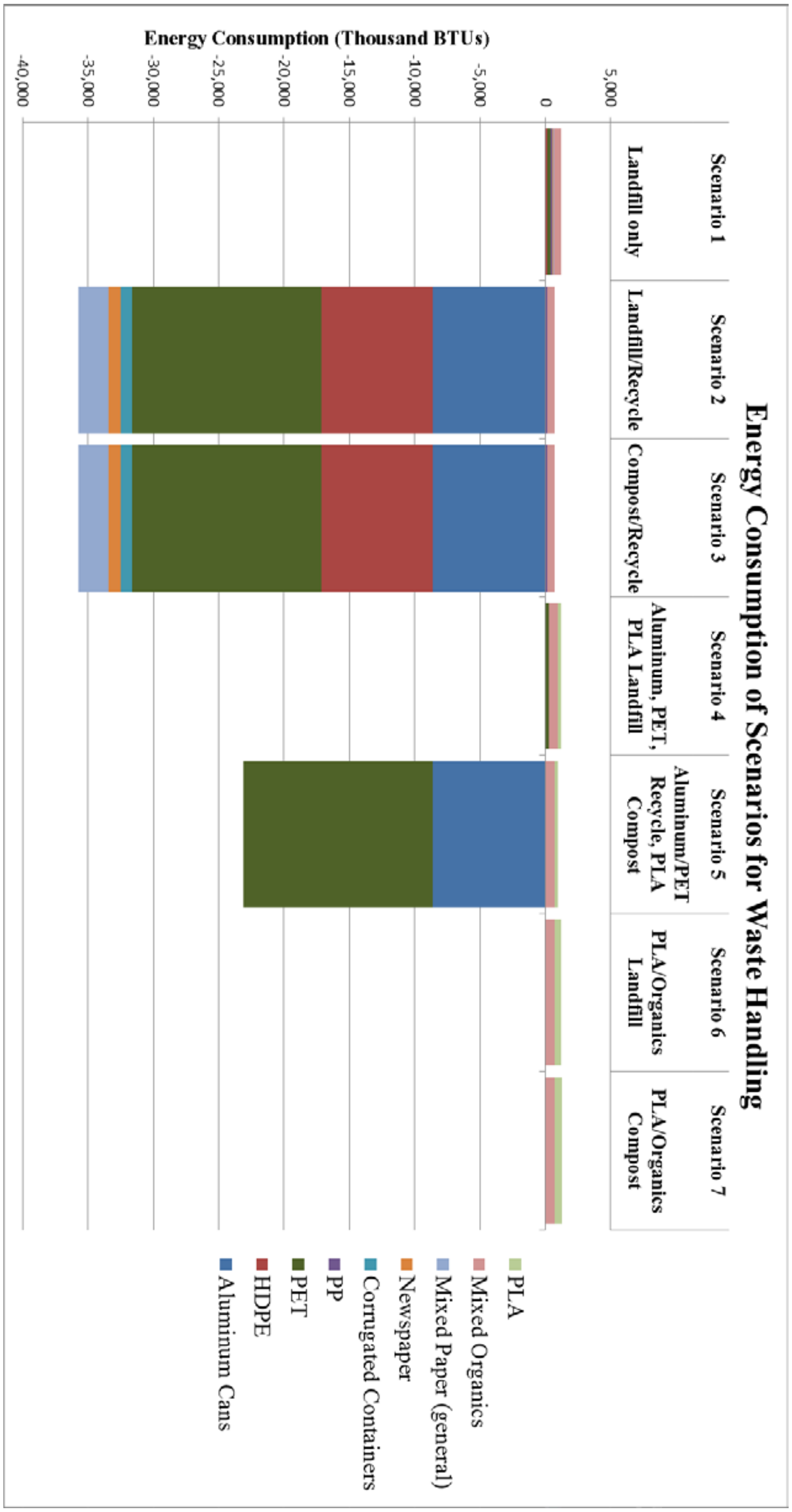

Figure 13. Scenario results of energy consumption in thousand BTUs by waste type using WARM for one season of waste generated at ASU's Packard Stadium.

The current approach for composting and recycling materials (Scenario 3) coming out of Packard Stadium at ASU performs comparatively well in terms of CO_2 equivalent emissions and energy consumption, but these impacts are not all that needs to be considered; The contamination rates, especially for composting (an average of 15.6%) are too high for the waste handler to accept and will be rejected at the tipping floor; the maximum contamination rate is less than 1% (Wood 2014). When contamination levels exceed the waste handler defined limit, the compost is diverted to landfill. The recycling stream can tolerate more variable materials and contamination (up to 15%) because of the efficacy of sorting that is accomplished by the material recovery facility, but food waste fouling can be problematic for recyclables. While food waste fouling may not result in diversion to landfill, it complicates the processing at the material recovery facility requiring additional cleaning and may reduce the value of the final products (McLaren 2013). Significant contamination in the compost and recycling streams may result in diversion to landfill and results in waste management Scenarios 2, 3, and 5 having an actual environmental impact that looks like Scenario 1, landfill only. Thus, reducing contamination in recycling and composting waste streams is an important factor for successful waste management diversion strategies.

The modeling using the national average data and default distances within WARM did not produce results very different from the ASU scenarios with one notable exception. With methane being flared in both the experimental and national average modeling, the mixed organics in the landfill under the national average scenarios results in 0.27 metric tons (Scenarios 1 & 2) and 0.33 metric tons of CO_2 equivalent emissions (Scenarios 4 & 6), an increase of 0.30 and 0.37 metric tons respectively when compared to the modeling specific to the Butterfield Station Landfill. This is due to increased degradation of organic materials in landfills with higher moisture levels. The consequences of methane generation could result in two very different outcomes. Methane can be captured and burned for energy, providing additional value from the waste and reduced energy use. On the other hand, both methane and the CO_2 that results from burning the landfill gas (for energy or flaring to meet compliance) are greenhouse gases. Other regions that receive more rain or are characterized by distributed, rural sources of waste have different impacts resulting from methane and transportation that will alter the results and may suggest a different approach for sustainable management of materials disposal.

Waste Disposal Behavior at Public Events

Despite the descriptive signage on the bins at the three game series, the data shows an improvement in identification and proper placement of materials when the bin guards were used during Game 2. The Saturday night game had twelve volunteer bin guards, each one assigned to a pair of bins, who covered all of the publicly available bins. The bin guards helped attendees sort their compost and recycling. Prior to the game the bin guards were trained by two staff members. The training was completed in ten minutes and at the end of the training the bin guards were given lanyards with cards which explain which materials go in each bin, similar to the signage on the bins.

Attendance data (Table 4) supplied by the athletics department shows there are 1,392 season ticket holders and 121 tickets purchased prior to the series for two or three games out of the series with University of Arizona. With an average attendance of 2,653 per game, more than half of the baseball fans had tickets for at least two of the three games. The attendance data and the different contamination rates between Game 1 (33.5%) and Game 2 (11.3%) suggests that the bin guards may have served as a successful intervention that enabled visitors to learn about proper disposal of venue materials. Contamination was reduced in unsupervised waste bins (i.e. no bin guards helping visitors) from Game 1 to Game 3 (22.5%) by 32.8%. This is especially encouraging for repeating events and venues that are continually providing reinforcement for correct disposal behaviors. In addition, for one-time events expected to generate large amounts of waste, bin guards may be employed to help reduce contamination rates.

Discussion

The benefits of recycling stand out in both the CO_2 equivalent emissions and energy impacts (Figure 12 and Figure 13). However, recycling cannot exist alone in a zero waste strategy since food waste makes up a portion of the waste. While composting and recycling of the current mix results in the best modeled performance, the rates of contamination remained prohibitively high without manned bins, and a zero waste policy would not be possible under these conditions. Without significant efforts to increase outreach and public awareness, through strategies in addition to the bin guards and signage, it will be difficult for ASU to attain acceptable levels of contamination for multi-bin systems at Packard Stadium. Even with extensive levels of outreach, recyclables caked in food wastes will foul either waste stream without postconsumer treatment.

Alternatively, simplifying the supply chain (and subsequently the waste stream) by limiting recyclables and having all other products compostable, like in Scenario 5 (aluminum/PET recycle, PLA compost), may be more effective in reducing contamination. This simplification approach reduces recyclables to PET and aluminum, the two materials providing the greatest benefits in both energy and CO_2 equivalent emissions (Figure 12 and Figure 13), while finding compostable alternatives for all the other products. Thus the outreach effort (e.g. bin signage, bin guards, public announcements) can focus on clear disposal instructions based on product function, "all beverage containers get recycled, everything else gets composted." Simplification can achieve relatively low CO_2 equivalent emissions and energy impacts as well as divert all the materials generated from the landfill, thus achieving zero landfill waste.

Despite higher greenhouse gas and energy impacts relative to traditional plastics that can be recycled, the use of biopolymers alters the composition of the material streams which can help to simplify management approaches. Compostable products provide an alternative waste management strategy for organically fouled materials, which are caked in food wastes. This makes it easier to eliminate landfill wastes; biopolymers, which perform as well as traditional plastics, can simply be composted along with food wastes without concern about organic matter contamination. In comparison, recyclable plastics that are contaminated with organic matter are typically sorted out of recycling and enter the landfill. This study of venue-based events suggests that the real benefits of biopolymers may be a result of these indirect benefits to management and contamination that are not necessarily captured within traditional modeling. Future research could evaluate the consequential impacts resulting from the use of biopolymers and the subsequent rates of organics that are diverted from landfills. If the methane avoided from the landfill is significant enough it may outweigh the additional impacts incurred from the use of biopolymers. If not, biopolymers may only be useful in landfill avoidance through simplifying waste streams and provide few environmental benefits. The modeling conducted for this research represented the regional dynamics for the case study of ASU. Waste services are diverse and regionally dependent with many variables, some of which are accounted for in the WARM model. The Phoenix metropolitan area is an urban environment with an arid climate and nearby waste facilities offering diverse services. The Arizona-centered modeling represents a dry landfill which is not conducive for the breakdown of organics and could serve as a lower limit for climate emissions from landfills. The scenario analysis using the national averages highlights the potential for methane emissions from landfills that are more conducive to the degradation of organics. Increased degradation rates combined with uncertainty in capture efficiency and different methane handling approaches (e.g. flaring and energy recovery) may result in dramatically different impacts (Spokas et al. 2006).

From a facilities management perspective a single stream system, in which there is only one bin, is much simpler with regards to contamination, collection, and handling. Scenario 7, compostables only, represents the only single stream system that also diverts all materials from landfill. Zero landfill wastes achieved via use of compostables requires a shift in management focus from the back-end of waste management to supply chain procurement. All disposable products must be compostable, a challenge in events which requires that service providers and vendors conform to purchasing standards. Such an approach also begs the question of who pays for potential additional costs of using compostable only materials.

While achieving zero waste, energetic and emissions benefits of recycling are lost in this single stream composting approach, which is clear in Figures 12 and 13. By treating all materials the same, handlers are unable to take advantage of the strategies that maximize environmental goals for each type of material. Not analyzed in this research is the price of the different management approaches for the customer and the waste handler, which can dictate whether or not any given approach is economically viable. Additionally, the economics become even more important when valued products created from these materials like gas, energy, raw materials, or compost are taken into account.

An additional factor when considering the ease of single stream management versus a two stream system is consumer involvement. If the facility or institution is most interested in reducing its own landfill wastes a single stream may be best, but if the goal is to engage people in sustainability initiatives and introduce new norms there is an advantage to having consumers sort their wastes even if it reduces efficiency at the facility. The benefits of strategies like the use of bin guards may create new norms that will influence people in their lives beyond the stadium walls which is supported by previous literature on behavior modeling (Sussman and Gifford 2013, Sussman et al. 2013, Zhang et al. 2011). Taking this into account it may be better for one-time events to use single stream management while repeat events, like baseball games, may be more beneficial as a platform for education.

The modeling assumed that once materials enter their respective end of life stream (compost, landfill, or recycle) they are actually processed in that system. However, some compostable materials like thermoplastic starch or PLA can be hard to distinguish from more common recyclable plastics. Biopolymer products could end up at a recycling facility and be diverted to the landfill or, worse yet, actually make it to the composting facility only to be screened out with other plastics. Screening at compost facilities may happen prior to the composting process to remove unwanted materials or once the compost is finished, and in either of these situations if the biopolymers remain intact they will be removed from the compost and discarded. These dynamics must be assessed to gain a complete understanding of how recyclables and compostables are moving through the compost and recycling streams.

The increasing focus on venue-based events can help improve the environmental performance of sporting events, theaters, festivals and other gatherings (Collins and Flynn 2008, Collins, Jones, and Munday 2009, Hershkowitz, Henly, and Hoover 2012, Laing and

Frost 2010), which may also provide insights into larger waste management trends. Venues such as Packard Stadium allow for the exploration of different aspects of sustainability and enable researchers and facility personnel to experiment with new management approaches. In addition to management scenarios, venues can control for the types of materials that enter the system and work with service providers to develop contracts which specify new standards. Venues can be a proving ground for improved environmental performance for shared spaces.

Conclusions

The findings of this research demonstrate how waste materials and end of life treatment have real tradeoffs for energy and climate change emissions. In addition to the impacts calculated using WARM, there are several management practices that achieve different objectives such as climate emissions reduction, landfill avoidance, maximum financial benefits, or ease of management. The assessment of collection methods at venues suggests that there are effective ways to teach consumers, over time, how to sort materials and that simple signage may aid in reinforcement of behavior but is not necessarily effective as a standalone strategy. These findings will help venue operators and facility managers better understand their options when deciding to change management approaches or improve the environmental impacts of their operations. This study helped inform the efforts of ASU to meet both its "Carbon Neutrality Action Plan" (ASU 2010) as well as the zero waste commitment and can provide insights for other organizations to meet their environmental goals.

CHAPTER 4

ALKALINE AMENDMENT FOR THE ENHANCEMENT OF COMPOST DEGRADATION FOR POLYLACTIC ACID (PLA) BIOPOLYMER PRODUCTS

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This chapter addresses research objective 3) Evaluate sustainable solutions for composting infrastructure options and factors influencing feasibility of scaling

Introduction

Compostable biopolymers have become increasingly popular in the single-use disposable products market due to their compostability (Lunt 1998); however, in actuality they may not degrade in the time frame expected during the composting process (Gómez and Michel Jr 2013). While these plastics may be instrumental in altering the traditional approach to waste management because of their biodegradable and compostable properties (Razza et al. 2009), the role of compostable plastics in municipal solid waste is still somewhat uncertain because, despite ASTM certification as compostable materials (ASTM 2004, 2003a), these products are slow to decompose compared to other organic wastes (Ghorpade, Gennadios, and Hanna 2001, Mohee and Unmar 2007, Gómez and Michel Jr 2013).

In order for polylactic acid (PLA, the most common compostable biopolymer in the US) products to degrade the wastes need to be processed at an industrial composting facility to meet the minimum thermal requirements for degradation. Composters can generally accept food waste and turn it into a product that can be utilized in landscaping, erosion control, and farming applications in anywhere from a couple weeks to a couple months (Goyal, Dhull, and Kapoor 2005); however, PLA products may take up to six months to breakdown under ideal conditions (World Centric 2014). While most PLA products meet the ASTM technical requirements for degradation, this may not translate to the reality of compost facilities where compostable food and yard waste is processed much more quickly. One possible solution to the degradation problems in commercial compost operations is creating limits as to what kind of products will be accepted at their facilities based on degradation rates. A good example of this is Cedar Grove's program which tests compostable products to determine if the material will break down in their composting system in order to ensure a high standard of compost quality (Cedar Grove 2012).

When biopolymers do not compost at the same rate as food and other organic waste, the biopolymers remain in the compost piles even when the food and yard waste compost is finished and ready to sell. This results in the accumulation of biopolymers in compost facilities. If biopolymers are returned to active compost piles to allow more time for degradation, the additional handling and processing results in size reduction of traditional plastics, increasing the difficulty of identifying and removing non-biodegradable plastics contamination from those piles (Figure 14). Ultimately biopolymers end up being screened out of compost and when too there is too much accumulation, are sent to landfills (Pfiefer 2014). Landfilling incurs additional costs, both financial and environmental due to the additional handling and change in end of life treatment. Many compost facilities in the U.S.

are removing bioplastics from compost and sending them to landfills and others have begun to outright refuse bioplastics due to these challenges (Hill 2014, Elliot and Leif 2014).

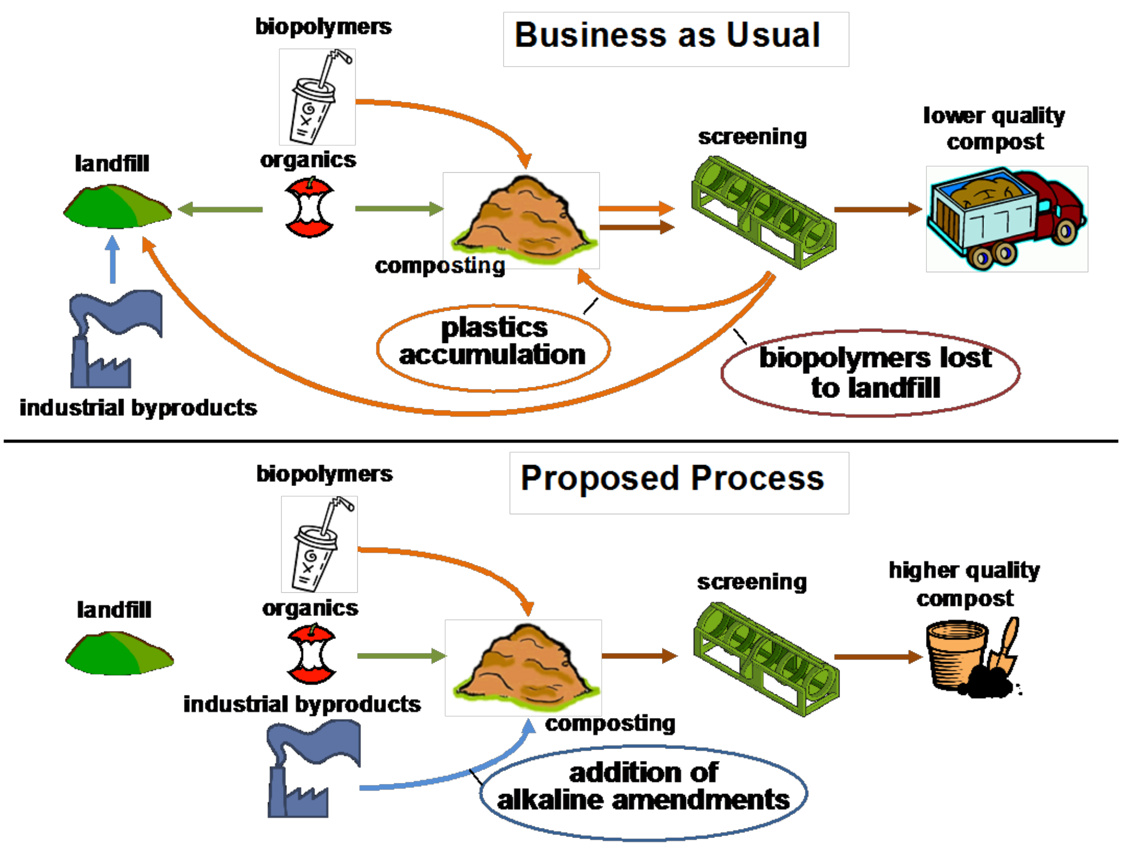

Figure 14. Proposed material flow for enhanced commercial composting process compared with the current process. The green arrow represents the organics, which includes food waste, grass clippings, etc. The blue arrow represents the industrial byproducts utilized as alkaline amendment. The orange arrows represent the biopolymers. The proposed process results in a product of higher quality that has neither uncomposted PLA.

Research testing the compostability of PLA has relied on ASTM and ISO standards (Cadar, Paul et al. 2012, Kale, Kijchavengkul, et al. 2007, Pradhan, Misra et al. 2010, Kale, Auras, et al. 2007). ASTM D5338, Standard Test Method for Determining Aerobic Biodegradation of Plastic Materials Under Controlled Composting Conditions (ASTM 2003a), and ISO 14855-1, Determination of the Ultimate Aerobic Biodegradability and

Disintegration of Plastic Materials under Controlled Composting Conditions—Method by Analysis of Evolved Carbon dioxide (ISO). The ASTM and ISO test methods are used to determine whether a material can be considered compostable under controlled composting conditions, but do not evaluate performance in realistic composting conditions. The standard test methods differ from real composting processes due to climatic differences, types of compost materials, and other parameters such as internal temperature, moisture, and pH (Kale, Auras, et al. 2007). The degradation of PLA follows two steps: hydrolysis to low molecular weight oligomers followed by microbial degradation to CO_2, water, and humus (Drumright, Gruber, and Henton 2000). Hydrolysis is primarily influenced by temperature, moisture, and pH (Drumright, Gruber, and Henton 2000, Kale, Kijchavengkul, et al. 2007).

Additives can be used to enhance the composting process of traditional organic wastes. There are several types of commercially available additives that include mixtures of different amounts of microorganisms, mineral nutrients, or readily available forms of carbon (Himanen and Hänninen 2009). Additives can also include pH-balancing compounds, like alkaline minerals, lime, and ash, which increase the pH levels of the compost (Gabhane, William et al. 2012). While alkaline amendments have been shown to enhance composting and certain soil environments, no work has been done to assess how they may impact biopolymers (Yu and Huang 2009, Cox, Bezdicek, and Fauci 2001, Himanen and Hänninen 2009).

Industrial alkaline wastes provide a resource which could be used as compost alkaline additives to improve PLA degradation and improve the quality of the finished compost. ALCOA Inc., a leading aluminum manufacturer, produces bauxite residue as a byproduct of aluminum manufacturing that has been used in remediation of soil in abandoned mine lands due to its nutritional properties for the soil (McConchie, Clark et al. 2000, Doye and Duchesne 2003, ALCOA 2007). Another source, wood fly ash is composed of potassium carbonate (K_2CO_3) and magnesium oxide (MgO). Previous research evaluated the use of wood fly ash as a soil amendment, and both K_2CO_3 and MgO contribute to a pH of over 13 as well as other nutritional components to soil (Kurola, Arnold et al. 2011, Weyerhaeuser 2014). Mineral CSA produced by Harsco Minerals is another option. The product is composed largely of calcium silicate and is obtained from slag generated during stainless steel production (Harsco 2009).

This research investigates the use of an alkaline amendment to enhance the degradation of biopolymers under thermophilic composting conditions. By utilizing waste products as amendments to improve composting, landfill wastes can be avoided in both systems (Figure 14). By enhancing the rate of PLA degradation, compostable biopolymers may be composted along with food and yard waste without sacrificing the quality of the final product.

Materials and Methods

This study was conducted in a laboratory setting using flasks as bioreactors to simulate a composting environment over a twenty-two day incubation period. In order to maintain a uniform environment, the bioreactors were monitored daily and moisture and temperature were kept constant.

The compost experiment conducted herein differs from the ASTM standard lab methods to better represent actual composting processes. As opposed to the ASTM testing period of up to 180 days at 58°C, this experiment was truncated to 22 days to represent the lower limit of composting while taking in consideration the actual length of the key stage of aerobic composting when the microbial community is dominated by thermophilic organisms. The thermophilic process of composting occurs within a period of 21 days which is also the most intensive stage of degradation (Barrington, Choinière et al. 2002, Goyal, Dhull, and Kapoor 2005). We were also looking for differentiation in degradation based upon the additive rather than complete degradation in all cases and this shorter time frame enabled this comparison.

For this study raw food was included in the bioreactors in addition to the compost inoculant so the bioreactors would better simulate the actual composting process that occurs at commercial composting facilities in which food and yard wastes are processed along with biopolymers. This is in contrast to the ASTM method of using only mature compost which aids in tracking CO_2 evolution for compostable materials (ASTM 2003a), but may simultaneously be withholding nutrients necessary for a robust microbial community capable of degrading biopolymers that may also react to changes in pH (Drumright, Gruber, and Henton 2000, Himanen and Hänninen 2009).

Materials

The compost environment was simulated in a bench-scale setting; the various experimental conditions contained different amounts of alkaline amendment but a constant amount of PLA, inoculant, and food scraps. Two different types of PLA were tested: 16oz clear PLA drinking cups designed for cold liquids, manufactured by World Centric using NatureWorks' Ingeo™ amorphous PLA resin 2003D (NatureWorksLLC 2002b), and opaque PLA flatware, manufactured by World Centric using a 70% PLA (using Chinese grown corn), 30% talc blend (World Centric 2015). They are referred to as clear and opaque PLA throughout the remainder of this paper. Both sets of samples were cut into 2 cm^2

pieces for use in the experimental bioreactors. These products were chosen because both are readily available, marketed as compostable, and the two products have different material characteristics including surface area to mass and volume ratios.

The compost included food scraps, compost inoculant, and the alkaline additive. The food scraps used in the compost sample included one head of broccoli, two large apples, 2 large carrots, 1 bag of lettuce (283 g), and one large cucumber. These food scraps were chosen based on an audit of back of the house food waste conducted at the Hasayampa dining hall at Arizona State University and characterize a mixture of wastes destined for composting. The food scraps were grated using a food processor for a consistent texture prior to being combined with the PLA and compost inoculant. The food scraps were added to the compost mixture in order to introduce new carbon, nitrogen, and moisture to the already decomposed inoculant so that the composting process is a similar environment to the commercial composters' piles. The alkaline amendment was added at rates of 0%, 2%, and 10% as similar proportions have been used in previous experiments testing commercially available compost additives (Himanen and Hänninen 2009).

A compost inoculant was used to introduce a microbial community necessary for composting to boost the initial stages of the composting process. The compost inoculant was collected on January 8, 2014 from a food waste compost pile at Garick's Maricopa Organics Recycling Facility located in Maricopa, Arizona. The experiment was conducted within fifteen days of collecting the inoculant; the inoculant was stored in the lab in a plastic bin to achieve a consistent moisture content. The compost inoculant was collected from one to two feet within the food waste compost pile, which was four months old with a C:N ratio of 14.1:1 and 40.68% organic matter when dry (see Appendix B for Garick's analysis of the compost sampled). The compost was filtered by hand to remove plastics and any large debris such as twigs and rocks before being screened at 1/8", Garick's standard for finished compost.

The alkaline additive used is a soil amendment distributed by Harsco Minerals. It is a processed alkaline material from steel operations, which would otherwise be waste. The alkaline material, called Mineral CSA, is produced and sold for remediating acid mine lands. Mineral CSA undergoes additional processing to remove heavy metals and results in a uniform size and texture, which provides the consistency needed for this study.

Product Density and Surface Area

The density and surface area of the two PLA products were measured in order to determine which basic properties may be contributing factors in the degradation during the composting process. The densities of each type of PLA were calculated using a simple water displacement method. A 100 mL graduated cylinder (± 0.8 mL) was used to determine the volume for samples of a known mass for each PLA material type. The density was calculated using the volume and mass for each sample. The surface area of each type of PLA was calculated by measuring the dimensions of the cup and the spoon and dividing the shapes into two dimensional calculable geometrical shapes (Appendix C). The area of each shape was calculated, summed, and doubled in order to produce a total surface area of each of the PLA products. The final surface area of each type of PLA was divided by their respective volumes and masses in order to find the surface area ratios of both products.

Bioreactor Setup

Six flasks served as bioreactors containing the materials described in section 2.1. Each bioreactor contained 400g of compost inoculant, 200g of food scraps, and 40g of PLA.

Mineral CSA was added in 2% and 10% concentrations (based on the combined weight of the inoculant and food scraps) to the bioreactors for both types of PLA. The experiment consisted of these six bioreactors; the conditions and contents for each bioreactor are described in Table 5. Three of these bioreactors contained the clear PLA while the other three contained the opaque PLA. For both types of PLA there was a control bioreactor and two test bioreactors with different concentrations of alkaline additive. The bioreactors were maintained at controlled temperatures using a water bath, and supplied with heated and humidified air. The bioreactors were exposed to an incubation temperature profile of 35°C for two days and following at 58±2 °C for twenty days. The controlled temperature increase over time was used to better simulate an actual compost process with a transition from a mesophilic biota to an environment favoring thermophilic organisms (Goyal, Dhull, and Kapoor 2005, Barrington et al. 2002, Kijchavengkul, Auras et al. 2006). Water bath heat was supplied by a thermostatically controlled heating mat placed under aluminum trays (Appendix D). A thermistor was used for the controllers and a thermometer was used for calibration and continuous monitoring. The supplied air passed through a bubble humidifier, which was also heated by the water bath, and was then distributed via pipettes to the bottom of each individual bioreactor. At the beginning of the incubation period 100 mL of deionized water was added to each bioreactor.

Table 5. Contents of each bioreactor prior to incubation. The abbreviations for the bioreactor names include the percentage of the amendment added (0, 2, 10), followed by the type of plastic (clear, opaque).

Bioreactor	Compost inoculant (g)	Food Scraps (g)	PLA (g)	Mineral CSA (g)	Initial pH
0 - clear	400	200	40	0	7.0
2 - clear	400	200	40	12	7.3
10 - clear	400	200	40	60	7.6
0 - opaque	400	200	40	0	7.0
2 - opaque	400	200	40	12	7.3
10 - opaque	400	200	40	60	7.6

Monitoring and Maintenance

Moisture and pH were recorded every four days using an analog electric resistance moisture meter and a digital electrode probe for pH measurements, both Rapitest® products are specifically designed for use with soils. The moisture meter was calibrated at the beginning of the trials using soil that was saturated but not sitting in water. This baseline was used to determine whether water needed to be added to the flasks throughout the trial. Water was added in 10mL increments until the desired moisture level was confirmed with the meter. Occasionally, because of the composting process, moisture levels increased and additional water was not needed. The pH was measured after moisture adjustments were made. Testing was conducted outside of the water bath on a lab bench. The probes were inserted into the flasks to maintain the integrity of the samples over the test period. In order to preserve a stable incubation environment the bioreactors were checked on a daily basis with deionized water being added as needed to maintain consistent moisture levels, an average of 72.7 mL per day.

Upon completion of 22 days of composting, the entire contents of each bioreactor were removed. The PLA samples were hand sorted from the compost and washed in a rinsing tray using an irrigation bottle. This process of cleaning was repeated three times followed by a final rinse. PLA pieces that were too small to be separated by hand using this irrigation bottle approach were considered to be composted and were not included in final mass measurements. The samples were then spread out and allowed to air dry under a vented hood. The final masses were recorded using a scale with a sensitivity of ± 0.5 g.

Visual Inspection of PLA Degradation

After the PLA was hand sorted from the final compost, visual evidence of degradation was recorded using photographs for both types of PLA and microscopic images for the clear PLA. An Olympus Stylus digital camera was used to photograph the opaque PLA under the Super Macro setting. The microscope images of the clear PLA were taken using an Olympus IX70 microscope at 100X power and at maximum lightning with a Canon EOS 10D digital SLR camera. The photographed samples were selected as representative samples from each bioreactor to allow for comparisons across treatment scenarios.

Results

The degradation and compost conditions were evaluated based on pH, visual inspection, and PLA mass loss. These evaluations showed marked differences in the clear PLA with regards to mass loss and visual signs of degradation. There was no change with regard to mass loss for opaque PLA degradation; however, the opaque PLA bioreactors exhibited changes in pH as well as visual evidence of PLA degradation.

Results of this study support the findings of previous studies which demonstrate that PLA products with low density and high surface area to volume ratios take less time to degrade (Kale, Auras, et al. 2007, Kale, Kijchavengkul, et al. 2007). The surface area to volume ratio, surface area to mass ratio, and the density measurements are shown in Table 6. While the densities of the two materials are similar, clear PLA has nearly 4 times the surface area to volume ratio.

Table 6. Material characteristics for two PLA products. This table reports material properties for the PLA prior to blending and the properties of the specific products evaluated within this study. The clear PLA is NatureWorks' Ingeo™ biopolymer 2003D (NatureWorksLLC 2002b, Karamanlioglu and Robson 2013). *The data presented here for opaque PLA is for NatureWorks' Ingeo™ biopolymer 4032D, which is used in blends with talc for hot applications such as flatware and is assumed to have similar properties to the Chinese PLA used in the product (NatureWorksLLC 2002a, 2004, Bondeson and Oksman 2007).

	PLA Density (g/cm^3)	PLA Molecular Weight (kg/mol)	Product Surface Area (cm^2)	Product Volume (cm^3)	Product Density (g/cm^3)	Surface Area: Volume (cm^2/cm^3)	Surface Area: Mass (cm^2/g)
Clear	1.24	159	654.4	10.0	1.2	65.5	53.5
Opaque	1.24*	220-240*	52.5	3.8	1.4	13.8	9.6

Variations in pH and Effect of Alkaline Amendment on pH

Over the twenty-two day period the pH value of the compost decreased slightly over the course of the incubation period, as seen in Figure 15. In the case of the control bioreactors this pH change was not as great. The alkaline amendment introduced high levels of pH at the early stages of the composting process. In the composting process as CO_2 and NH_3 begin to form, the pH levels increase and reach a peak. After that, microbial activity increases and begins to breakdown the food scraps and PLA, a process corresponding to degradation during compost that has been previously described (Nakasaki, Yaguchi et al. 1993).

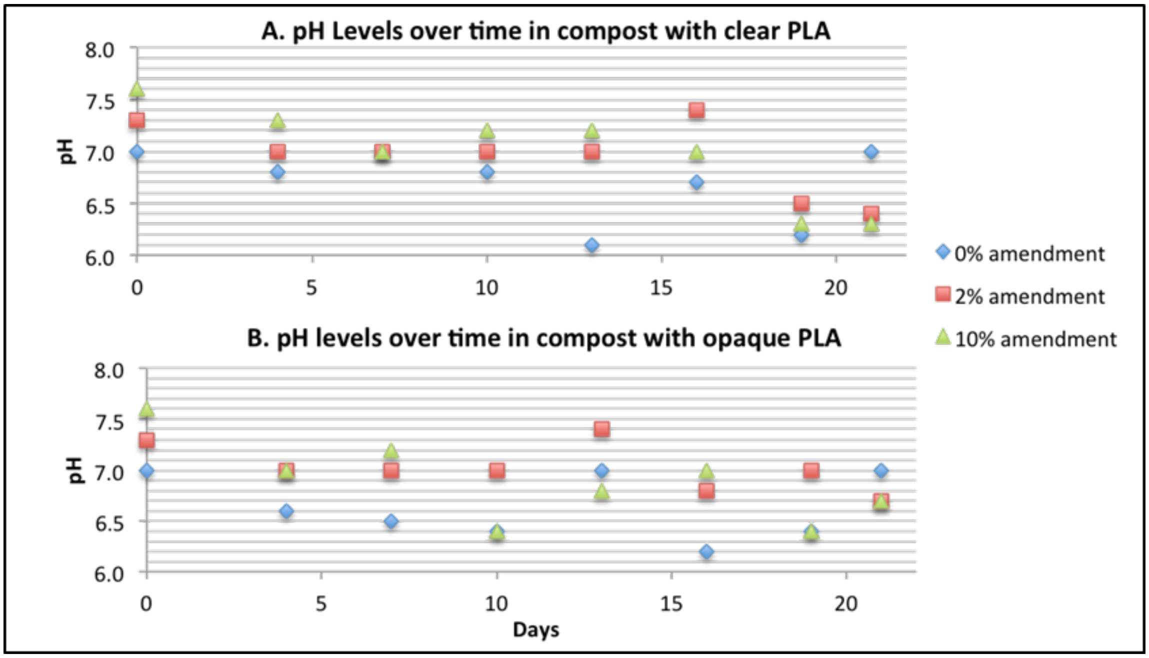

Figure 15. pH levels over time in compost bioreactors with clear PLA drinking cups designed for cold liquids and PLA flatware, both cut into 2 cm^2 squares.

The pH levels within each bioreactor that received an alkaline treatment decreased over time as a result of the composting process. The pH for the bioreactors with no Mineral CSA (i.e. 0-clear and 0-opaque) varied throughout the process, however, in the initial and final values were 7.0 for both bioreactors (Figure 15). The acidification of the compost is characteristic of decomposition of organic materials (Yu and Huang 2009). While there is a large amount of variation in the pH readings our data suggests a general pattern of acidification within all of the bioreactors.

Visual Inspection of PLA Degradation

Based on a visual inspection and microscopic evaluation, the clear PLA with the alkaline amendment showed more signs of physical decomposition. The clear PLA plastic squares were more distorted and discolored when subjected to higher rates of alkaline amendments. Figure 16 shows the clear plastic before the composting process and the PLA samples after they were removed from the bioreactors.

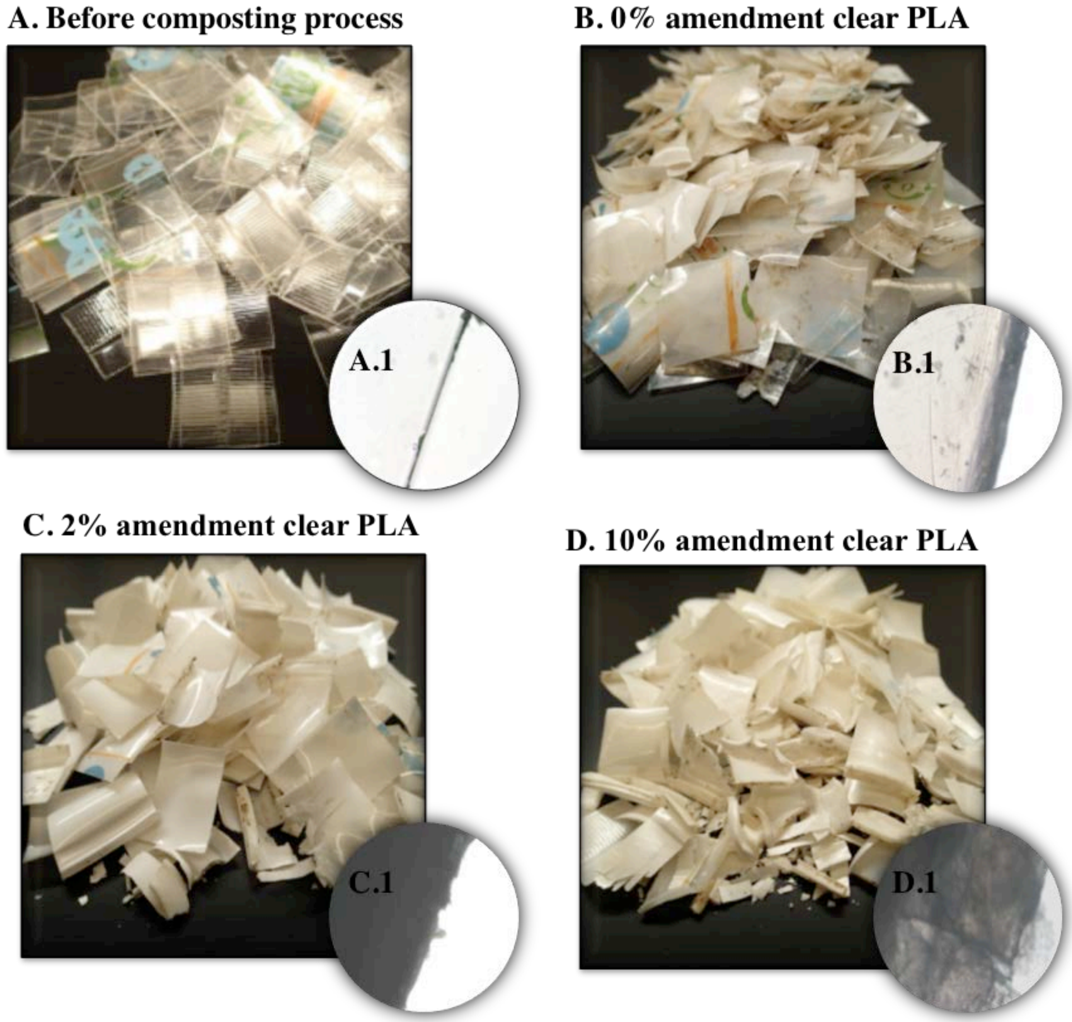

Figure 16. PLA products before (A) and after composting process for 0% (B), 2% (C), and 10% (D) amendment samples. The initial plastic (A) was cut into 2cm2 pieces. Images B.1-D.1 are microscopic images of representative samples from each bioreactor under 100X power. Image A.1 is the uncomposted plastic under the same power.

As seen in Figure 16, the microscope images of the clear PLA demonstrate that the samples in the control bioreactor became slightly cloudy when compared with the uncomposted PLA. The clear PLA that was composted with 2% alkaline amendment appeared darker due to the solid white color resulting from the composting process over the 22 day incubation period. The light of the bottom lit microscope could not penetrate the PLA as readily resulting in a darkened sample in the image (Figure 16, C.1). The most visual degradation was apparent in the sample that was composted with 10% alkaline amendment. The microscope images of the PLA with 10% alkaline amendment show that the PLA became translucent again because the physical degradation fragmented the material into many thinner layers (Figure 16, D.1). These visual signs of degradation may be associated with an increase crystallinity of the PLA, which has been previously described (Kale, Auras, and Singh 2006, Tsuji and Ikada 1998)

Similar to the clear PLA samples, the opaque samples showed greater discoloration in proportion to the amount of alkaline amendment present (Appendix E). The opaque PLA under 10% alkaline treatment was clearly darker and more distorted than the other samples. There was almost no visual change in the opaque PLA in the control bioreactor.

Given the visual inspection and the changes in pH, the clear PLA and the opaque PLA differed in degradation resulting from the composting process. The greater density and reduced surface area/volume ratio may mean the opaque PLA, blended with talc and is manufactured to have higher heat resistance, requires more time to degrade than the clear PLA with or without an alkaline amendment.

Mass Loss

The degradation of the PLA was also measured by comparing the final mass with the initial mass (40g) for the PLA contained within each bioreactor. The clear PLA degraded more than the opaque PLA, which showed no loss of mass in any of the bioreactors. At the end of the twenty-two days the clear PLA in the control bioreactor (representing normal composting conditions) had lost 2.5g, the 2% treatment bioreactor had lost 7.5g, and the 10% alkaline treatment bioreactor had lost 6.6g (Table 7). That equates to 3 and 2.6 times greater total degradation for the PLA in the alkaline treatments respectively when compared to the PLA in the control bioreactor.

Table 7. Initial and final mass calculations for the PLA in each bioreactor. The abbreviations for the bioreactor names include the percentage of the amendment added (0, 2, 10), followed by the type of plastic (clear, opaque).

Sample	Initial mass (g)	Final mass (g)	Percent Loss
0-clear	40.0	37.5	6.25%
2-clear	40.0	32.5	18.75%
10-clear	40.0	33.4	16.50%
0-opaque	40.0	40.0	0%
2-opaque	40.0	40.0	0%
10-opaque	40.0	40.0	0%

Discussion

The results of the amendment in the compost samples, including visual changes of the PLA and reductions in mass, suggest that the alkaline amendment may have aided in enhancing the degradation rate of the PLA during the incubation period. The mass loss data suggests that incorporating even small quantities of alkaline amendment may help to degrade PLA as shown by the percent mass loss in clear sample with 2% alkaline amendment (Table 7). Even if it is difficult to obtain alkaline byproducts from industry, it remains promising that only a small portion of alkaline amendments are needed in order to greatly increase the level of compostability of the PLA.

While the time frame that the compost samples spent in the bioreactors was relatively short for this study, we saw complete degradation of the raw food and the PLA samples showed differentiated degradation results. Assuming a typical time frame for a compost pile in a commercial composting operation, which can be a couple months or more, the clear PLA with the amendment may degrade completely and the opaque PLA may show improved degradation. Additional research would need to be conducted to determine the reduction in time for complete degradation of the PLA or the efficacy of amendments at a commercial scale. The complete degradation of PLA would enable operators of commercial composting facilities in optimizing the screening process for the compost. If the PLA were to degrade by the end of the commercial composting process, then screening could remove any non-degradable plastic residues and reduce the amount of waste going to landfill.

The ASTM test methods for compostable products could be improved to better represent real world composting conditions. Currently, ASTM tests for compostability by isolating test materials in finished compost inoculant, allowing for 180 days of decomposition at 58°C. By improving on methods described herein that rely on real composting conditions, biobased polymers such as PLA can be properly labeled as "compostable" and thus can be processed within the existing composting infrastructure. Other updates to ASTM methods may include the shortened time frame with an alternative temperature profile or raw food, like this study, or other variables that can be incorporated in lab scale trials.

There are several intriguing options of alkaline waste products from industry including alkaline clay (ALCOA 2007), coal ash (Belyaeva and Haynes 2009), wood ash (Kurola et al. 2011, Weyerhaeuser 2014), slag (Harsco 2009), or other industrial alkaline products. According to the Material Safety Data Sheet, Mineral CSA complies with the 40CFR 503.13 EPA Standards for the use or disposal of sewage sludge (Harsco 2009, USEPA 1993). This standard must be considered with the application of other alkaline amendments to compost.

Industrial symbiosis, a concept developed out of the field of Industrial Ecology, focuses on identifying waste in one industry which can be used as a raw material or energy source in another industry (Chertow 2000). Industrial symbiosis can be applied in this case to benefit both the industries that create alkaline wastes as well as composting facilities that need new ways to enhance their processes and products. Industrial symbiosis encourages material flows among local industries, benefitting all parties in the system through collaboration and the synergistic possibilities offered by geographic proximity.

One example of an industrial symbiosis system would be to connect The Frito-Lay facility in Casa Grande, AZ, which produces an alkaline ash waste product with the Maricopa Organics Recycling Facility operated by Garick LLC who has received a lot of biopolymers in the organic wastes delivered to their compost facility. Frito Lay has implemented a net zero waste plan where they installed high pressure biomass boiler that uses wood and agricultural waste as its fuel source. The biomass boiler provides steam to help power the plant and the byproduct of this process is wood ash (Drevensek 2011). The Maricopa Organics Recycling Facility operated by Garick LLC provides biomass for combustion to Frito-Lay while simultaneously providing composting services for other organic material streams. Through a mutually beneficial relationship with Frito-Lay, Garick could reclaim the ash waste to reintegrate into their composting operations in order to enhance biopolymer degradation at their facility while simultaneously reducing the waste generated through biomass combustion at Frito-Lay.

A new system benefitting from enhanced biopolymer degradation may also include organic wastes that would otherwise be contaminated by plastics. Improving the composting of biopolymers may also significantly alter the life-cycle impacts of these products (Hottle, Bilec, and Landis 2013). The enhanced degradation of biopolymers in compost may enable greater diversion rates for institutions and cities by enabling the acceptance of biopolymers and any mixed organics stream which includes biopolymers wastes. The development of an effective pathway for compostable products is crucial in realizing waste reduction goals (Hottle, Bilec et al. 2015). Further, the proposed system could create industrial symbiosis between the composting industry and manufacturing industries, both of which can reduce their waste outputs as well as increase their profits by cutting their costs in waste treatment.

The research presented herein lays the groundwork for further inquiry into the use of alkaline amendments to enhance biopolymer degradation. This study demonstrated improvements in PLA degradation when composted with the alkaline material called Mineral CSA. The use of other alkaline materials should be explored and amendment ratios can be optimized to enhance degradation of PLA under different compost conditions. Additional trials and more replicates will be needed to determine more precise rates of decomposition associated with the addition of an amendment and the associated statistical significance. Other methods of investigation could be employed to determine the exact nature of the interactions involved with introducing an alkaline amendment, both chemical and biological. This topic of research has the potential to help improve biopolymer degradation in the composting industry and ultimately improve the life-cycle impacts of bio-based products.

CHAPTER 5

COMPOSTABLE BIOPOLYMER USE IN THE REAL WORLD: STAKEHOLDER INTERVIEWS TO BETTER UNDERSTAND THE MOTIVATIONS AND REALITIES OF USE AND DISPOSAL IN THE US

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This chapter addresses research objectives 1) Quantify end of life of waste flows of biopolymers via waste audits at public venues and waste handling facilities, and 2) Identify best practices for facilities management and waste handling of compostables via surveys and focus groups

Introduction

Over the past five decades the use of plastic has become ubiquitous. Plastics are regularly used in the manufacturing of many products, from grocery bags to synthetic lumber, and from toothbrushes to sutures. Over 15,000 plastics manufacturers operate in the U.S. with facilities located in every state. The value of shipped plastic goods in the U.S. was over $373 billion, and the plastics industry is ranked as the third largest sector of U.S. manufacturing (Carteaux 2013). In addition, plastics make up approximately 13% of the country's municipal solid waste stream, which is roughly equivalent to 32 million tons of plastic waste generated annually (USEPA 2012).

Biopolymers are one of the fastest growing segments within the global plastics market. Biopolymers (or bioplastics) are plastics that can be produced from renewable materials, including sugar, corn, soy, hemp and captured methane from waste. Biopolymers do not have to be made entirely out of renewable materials, as many produced today are blends of conventional and renewable feedstocks (Hartmann 1998, Shen, Worrell, and Patel 2010, Shen, Haufe, and Patel 2009). Furthermore, some biopolymers such as Bio-PET have an identical polymeric structure as their conventional counterpart and can be recycled along with regular PET. With such a variety of feedstocks and manufacturing processes not all biopolymers are biodegradable or compostable (Roland-Holst et al. 2013, Hottle, Bilec, and Landis 2013, Lopez, Vilaseca et al. 2012b). Worldwide consumption of all biopolymers including compostable and non-compostable plastics in 2012 reached 981,056 tons (less than 1% of total polymer consumption), and the market is expected to continue to grow in the United States (USDA 2008) and globally (Rapra 2012, Shen, Haufe, and Patel 2009). The growth of the biodegradable and compostable subset of biopolymers is predicted at a rate of around 13% annually (Platt 2006). Of total global biopolymer production, 43% are biodegradable plastics including compostable polymers (EuBP 2014b).

Compostable plastics must be able to degrade in a commercial composting setting according to set American Society of Testing and Materials (ASTM) standards including ASTM D6400-04 Standard Specification for Compostable Plastics, ASTM D6868-03 Standard Specification for Biodegradable Plastics Used as Coatings on Paper and Other Compostable Substrates, and ASTM D5338-98(2003) Standard Test Method for Determining Aerobic Biodegradation of Plastic Materials Under Controlled Composting

Conditions (Song et al. 2009, ASTM 2004, 2003b, a). Of compostable plastics, polylactic acid (PLA) is the most abundant, but thermoplastic starch (TPS) and polyhydroxyalkanoates (PHA) are also common (EuBP 2014b, Tabone et al. 2010). Biodegradable plastics still degrade but do not conform to the timeframe in which commercial composting occurs and have a different set of ASTM standards (Kale, Auras, et al. 2007). This technology is used in products like grocery bags, trash bags, packaging, diapers, and agricultural mulch films (Ammala et al. 2011). It is important to note that while ASTM standards are an important industry codification, many commercial compost facilities are struggling to process them; this issue is discussed in more detail below.

The drivers behind the growth of compostable biopolymers vary across regions, often relating to bans on conventional plastics, bio-preferred purchasing, and zero waste initiatives. According to the literature these drivers are associated with concern over increased fossil fuel use, greenhouse gas emissions, plastics pollution, decrease in landfill space, and human health (Gironi and Piemonte 2011, Ren 2003, Gómez and Michel Jr 2013, Kijchavengkul and Auras 2008, Álvarez-Chávez et al. 2012). For example, there are many conventional plastic bans being implemented and compostable product mandates being established. Recently the State of California has banned single use plastic bags (Steinmetz 2014), and it is estimated that over 100 U.S. cities have banned poly(styrene) (PS) food and beverage containers (Goldstein 2013). The U.S. federal government's BioPreferred Program mandates federal bio-based product purchasing, and it is likely that it has inspired cities across the U.S. to implement similar programs. After speaking with a city representative, it is clear that the City of Phoenix is one example of this (Carsberg 2014). Organizations in every state are either voluntarily adopting or being mandated to create waste to landfill reduction plans. Additionally, growth in the composting industry and new organics waste diversion policies, such as the newly passed legislation in both California and Massachusetts, which requires all commercial organic wastes be diverted from landfill, will continue to encourage waste to landfill reduction goals (EEA 2014, BioCycle 2014, Yepsen 2009).

Though compostable biopolymer use is growing in response to the aforementioned trends, there have also been well documented challenges and concerns related to their use. The U.S. Composting Council has identified five key challenges which include: labeling & identification, enforcement & legislation, ASTM standards, consumer education, and impacts to the National Organics Program (California Organics Recycling Council 2011). Clear labeling or demarcation of compostable bioplastics is crucial for helping consumers (here consumers are defined as individuals who are using compostable biopolymer products, in either a residential or commercial setting) accurately identify and separate their waste in the right disposal bins. Enforcement and legislative challenges refer to the lack of federal regulations for labeling products compostable, biodegradable, or biobased. Without enforcement concerning the use of these labels, some companies may mistakenly market products as compostable when they are not. In addition, some products that have been designed to meet ASTM compostability standards still are not degrading adequately compared to other organic wastes (Ghorpade, Gennadios, and Hanna 2001, Mohee and Unmar 2007, Gómez and Michel Jr 2013). The reasons for this are varied, but one may be that some ASTM standards include decomposition times that are longer than actual commercial composting timeframes. For example, a variety of ASTM certified compostable biopolymers take over three months to decompose in a commercial compost facility and one of the largest composters in the Pacific Northwest States they have a ninety day turn around time for creating finished compost (Worldcentric 2014, CedarGrove 2015). The challenges associated with consumer education are many as there is profound misunderstanding between the terms biodegradable, compostable, bio-based, as so forth. Moreover, many consumers and compostable biopolymer users do not have a general knowledge of the differences in composting and landfilling compostable plastics. Lastly, compost that has been made with compostable bio-plastic feedstock has caused problems for organic growers as there has been debate over whether compost made with these products violates USDA Organics label rules and regulations (California Organics Recycling Council 2011).

In addition to these challenges, there has been concern over which disposal method is ideal for compostables (Yates and Barlow 2013, Weiss, Haufe et al. 2012, Rossi, Cleeve-Edwards et al. 2014), the use of GMO feedstocks for bioplastics (Gerngross and Slater 2000, Snell, Singh, and Brumbley 2015, van Beilen and Poirier 2007), and possible impacts to human health (Roes and Patel 2007, Thompson, Moore et al. 2009, Álvarez-Chávez et al. 2012). Research around compostable bioplastics is ongoing, and many stakeholders who currently handle these products are also trying to determine best practices. For example, cities now working to divert more waste from landfill are grappling with many of the aforementioned challenges. Trying to weigh the potential costs and benefits to determine the overall sustainability of these products has become an important task for many managers, purchasers, and policy makers.

To help inform decision makers various tools have been developed to accurately assess what the impacts of different plastics products may be. Over the past decade there has been a proliferation of life-cycle assessments for biopolymers but the assumptions that underpin assessment can drastically affect overall findings (Hottle, Bilec, and Landis 2013). To date many environmental assessments of biopolymers have been done, including inventory improvements for life-cycle assessments (Vink, Davies, and Kolstad 2010, Hermann et al. 2011) but few life-cycle assessments adequately address end of life and findings vary widely (Hottle, Bilec, and Landis 2013, Yates and Barlow 2013, Koller, Sandholzer et al. 2013, Shen and Patel 2008, Weiss et al. 2012). Moreover, gaps exist in the available literature which document how compostable biopolymers are being used and their exact method of disposal. This US-based study provides information on where compostable biopolymers are most commonly found, who is using them, and how organizations using these products are actually disposing of them. In addition, the study evaluates the motivations behind purchase and disposal decisions. Our overall intent is to provide understanding for how these products are being used so that assessments are not limited by wide ranging assumptions and can produce more meaningful results.

Through stakeholder and user interviews, this paper identifies where compostable plastics are being used and disposed, and the motivation behind purchase and disposal decisions. Stakeholders include producers and distributors in the compostable biopolymer industry, compostable biopolymer experts, and decision makers who currently manage these products like municipal solid waste professionals or commercial composters. Users include organizations that use compostable biopolymers, such as cafes, cafeterias, and recreational concessions. The findings from these interviews provide insight into how these products are now being managed and in doing so we hope to contribute key information for important environmental assessment tools, decision makers, and compostable biopolymer users, both food service businesses and customers.

Methods

To determine where compostable biopolymers are being used and by whom, we began with audits of bioplastics in eight local grocery stores and three preliminary interviews with stakeholders, including producers and distributors in the industry, in order to identify where consumers were using compostable biopolymers. Following the preliminary interviews, we conducted twelve interviews with a variety of regional compostable biopolymer users, such as public and private cafeterias, restaurants, and sporting venues, to understand the motivations behind their purchasing and disposal practices. A limited number of participants were interviewed through non-representative qualitative expert elicitation, an established social science interviewing methodology (Trost 1986, Sandelowski 1995).

Grocery Store Audits

In order to help define the scope of the research and gauge the availability of compostable biopolymers for use and disposal in a residential setting, an audit of eight local grocery stores was conducted. The audits were conducted over three days in the Phoenix metropolitan area. Costco, Wal-Mart, and Fry's are food stores who also sell many other types of retail items such as clothes, toys, and electronics. Safeway, Albertsons, Trader Joes, Whole Foods, and Sprouts are food stores who carry mainly food items but could also have a small selection of other assorted retail items. The stores were selected as they cater to a wide range of consumers, affluence levels, and consumer preferences. Three stores were visited on June 16th, 2014: Fry's, Trader Joes and Whole Foods. Two more were audited on June 17th: Costco and Wal-Mart, and the remaining three were visited on June 18th, 2014: Albertsons, Safeway, and Sprouts. For all grocers, the store manager was contacted and approval for the audit was received.

The data (i.e. number and type of polymer) was visually collected and documented while walking through each aisle or section of the grocery store. In order to maintain a consistent review of product categories, any areas that fell outside of the baby, beverage, bread and bakery, breakfast and cereal, canned goods, condiments, cookies and snacks, dairy and eggs, the deli, frozen foods, fruits and vegetables, grains and pasta, international foods, meats and seafood, and cleaning and home products were not audited as some larger grocers sell many non-food items, including personal care or clothing. All rigid plastic packaging in each aisle was inspected. In addition to packaging, we also looked for plastic products that were made out of biopolymers (of any type, compostable, biodegradable, or non-biodegradable), such as PLA flatware. The item's name, brand, size, and type of plastic were documented for all plastic packaging or products that were labeled with number seven recycling symbol, PLA, plant-based, or PlantBottle™. Plastics are often labeled with the number 1 through 7. Plastics labeled with a number 2-6, or that had no recycling symbol or any information about the plastic material were not documented. Number one plastics, which are PET, were inspected further to determine if they were bio-PET products. After compiling the data from the grocery stores, all products with a number seven were logged and a search was conducted through company websites to determine plastic type.

It is possible that some biopolymers were not accounted for. We sought to capture all of the Bio-PET, but it is visually indistinguishable from PET, shares the same resin recycling code (number one), and is not always labeled as plant based or have a PlantBottle™ trademark so it is possible not all Bio-PET products were identified. Film, or flexible plastic packaging, was not inspected because it is difficult to determine what thin films are made from as they are not often labeled. In addition, global production of rigid bioplastics packaging greatly exceeds that of flexible packaging (EuBP 2014a).

Interviews

To scope and refine the interviews, which aimed to understand where compostable biopolymers are being used, we first conducted preliminary, unstructured interviews. We reached out to six producers and/or distributors in the supply chain who either make or sell compostable biopolymer products: Sodexo, Arizona Restaurant Supply, Western Paper, a Sprout's Farmers Market, NatureWorks LLC, and EcoProducts. Out of the six contacted three were available for interviews: a Sprouts manager, a representative from Natureworks, and a representative from EcoProducts. Both NatureWorks and EcoProducts produce and distribute compostable biopolymers, with NatureWorks being one of the largest producers of compostable PLA resin in the United States (Nampoothiri, Nair, and John 2010).The preliminary interviews were semi-structured and broad themes were set out beforehand with follow up questions that varied based on interviewees' responses. Themes included: where individual consumers are most likely coming into contact with compostable bioplastics, the distribution of compostable biopolymers, and where the majority of product sales occur. Preliminary interviews lasted between 15-45 minutes and were all conducted over the phone. During the preliminary interviews, responses were documented on a laptop by the interviewer. After each preliminary interview, the interviewer immediately reviewed the questions to ensure each one was answered adequately, check for errors, and follow up with clarifying questions.

In addition to this, a variety of other stakeholders connected to compostable biopolymer use were also interviewed. These stakeholder interviews included three governmental employees who help manage municipal solid waste, two from the City of Portland and one from the City of Phoenix, three commercial-scale composters (Recycled City LLC, Roots Composting LLC, and the University of New Hampshire), and a biopolymers industry expert to further develop our knowledge of current practices, challenges, and implications of compostable biopolymer use. These stakeholder interviews followed the same protocol as before with the exception that contact with the City of Phoenix was in the form of an email exchange.

The grocery store audit and the first three preliminary interviews with producers and distributors suggested that residential consumers were not coming in contact with biopolymers (of any type, compostable, biodegradable, or non-biodegradable), as the overall number of biopolymer products in the store was low and products that were there were not selling quickly. As such, the interview process was modified to gain an understanding of where compostable biopolymers were being used and disposed so that we could identify organizations (compostable biopolymer users) that would be appropriate for this research (Sandelowski 1995). Since compostable biopolymers are largely found in the food service industry, we utilized the food service market segmentation strategy developed by the USDA to create categories where compostable biopolymers are being used (USDA 2010). This statistically non-representative stratified sampling allowed for a wider elicitation in overall participant experiences (Trost 1986). We delineated five main categories which included: limited service eating places (organizations where customers pay prior to receiving food or drink, such as a café), cafeterias (both public and private), recreational food concessions (such as at sporting events), caterers, and hospitals. A list of establishments, within the Phoenix Metropolitan area, which had the possibility of carrying compostable biopolymers was made for each category, upon which each establishment was contacted to confim the use of compostable plastic. A total of twelve establishments confirmed using compostable biopolymers; and were interviewed about their use and disposal practices. The second set of twelve interviewees are summarized in Table 8. Stanford was the one exception, being located outside of the Phoenix area, and was chosen as an organization to interview because no other large cafeterias were available and they are well known for their waste reduction goals and as users of compostable biopolymers.

Table 8. Interviews conducted with various stakeholders across the compostable biopolymer supply chain. ~ indicates the second set of interviews, * indicates preliminary interviews.

Completed Interviews		
Compostable Biopolymer Users ~	**Location**	**Date**
Cafeterias		
Arizona State University	Tempe, AZ	8/13/2014
Intel	Chandler, AZ	8/1/2014
Stanford University	Stanford, CA	7/31/2014
Catering Companies		
Atlasta Catering and Event Concepts	Phoenix, AZ	7/25/2014
Bruce Brown Catering Company	Phoenix, AZ	7/21/2014
Santa Barbara Catering Company	Tempe, AZ	7/9/2014
Limited Food Service Establishments		
Anonymous Café	Tempe, AZ	7/8/2014
Pomegranate Café	Phoenix, AZ	7/30/2014
The Cutting Board Bakery and Café	Mesa, AZ	8/11/2014
Recreational Concessions		
Arizona Diamondbacks	Phoenix, AZ	8/27/2014
Desert Botanical Gardens and Arizona Science Center	Tempe & Phoenix, AZ	8/29/2014
Phoenix Convention Center	Phoenix, AZ	7/28/2014
Other Compostable Biopolymer Stakeholders *		
Composters		
Recycled City LLC	Phoenix, AZ	7/23/2014
Roots Composting LLC	Flagstaff, AZ	7/31/2014
Universtiy of New Hampshire	Durham, NH	7/10/2014
Industry Expert		
Brenda Platt, Institute For Local Self Reliance	Washington, D.C.	8/29/2014
Government		
City of Phoenix	Phoenix, AZ	7/30/2014
City of Portland, Solid Waste and Recycling: Residential Composting	Portland, OR	6/23/2014
City of Portland, Solid Waste and Recycling: Commercial Composting	Portland, OR	6/24/2014
Producers and Distributors		
EcoProducts	Boulder, CO	7/25/2014
NatureWorks LLC	Minnetonka, MN	6/20/2014
Sprouts Farmers Market	Tempe, AZ	6/29/2014

For the second set of twelve interviews, initial contact was made through email or phone, where upon the interviewer explained the research and scheduled an interview in order to speak with a representative about the organization's use and disposal of compostable biopolymers. All interviews were over the phone or in person except one exchange with a catering company (Bruce Brown Catering) that was conducted over email. The interviews were semi-structured and each category of food service had a list of questions and general themes to address. In all cases, respondents answered questions about the types of compostable biopolymers they used, why they chose to purchase them, and the method of disposal. The interviewers also asked follow up questions to gain further insight and elucidate their compostable biopolymer use and disposal stories. Again, while interviewing, answers to questions and notes were typed in real time. After each interview the interviewer immediately reviewed the questions to ensure each one was answered adequately, check for errors, and follow up with clarifying questions. The original interview questions can be found in Appendix F.

The interviews were analyzed using qualitative content analysis (Hsieh and Shannon 2005). The results and interview analysis only represent organizations from the second set of interviews with compostable biopolymer users (Table 7). Responses were classified according to three critical questions identified based on gaps in the literature including: motivations behind compostable biopolymer purchase, disposal practice, and motivation behind disposal choice. Next we searched for the challenges each organization associated with using and disposing of biopolymers. In addition, special attention was paid to how much influence individual consumers had on the purchase decision and disposal of these products.

Results and Discussion

Grocery Store Audit

Eight out of nine grocers carried items that were made from or packaged with biopolymers. Figure 17 presents the findings of the audit for all grocers audited. There were a variety of different types of products found with some of the most common being bio-PET bottles, PLA utensils, and compostable trash bags. Figure 18 shows the types of products found at all of the grocers. This represents the total number of biopolymer products available in each store and does not account for the total number of plastic products in each store. The percentage of biopolymer products, compared to all conventional plastic, was very small, and the biopolymer products are not always clearly identifiable. For example, the Stonyfield yogurt cup label does not mention anywhere on it that the packaging is plant based, instead the bottom of the yogurt cup states "this cup is made from plants." There were an abundance of number seven products, over forty items across the eight stores, including items such as 4 oz. Motts Applesauce packs, Nescafe Tasters Choice packaged coffee, and some of the one gallon bottles of Arizona Tea. According to the ASTM a number seven resin code on plastics incorporates all other possible types of polymers and materials which are made out of multiple resins or are multi-layered (Wilhelm 2008).

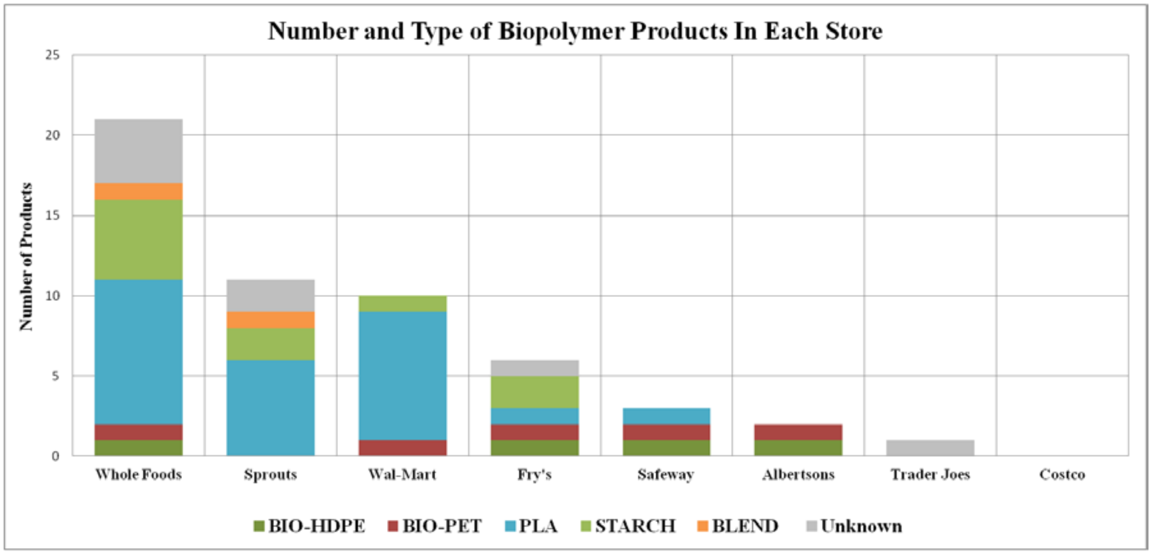

Figure 17. The number of biopolymer products found at each grocery store categorized by the type of biopolymer material. Products where no information on the type of biopolymer material used are labeled as unknown.

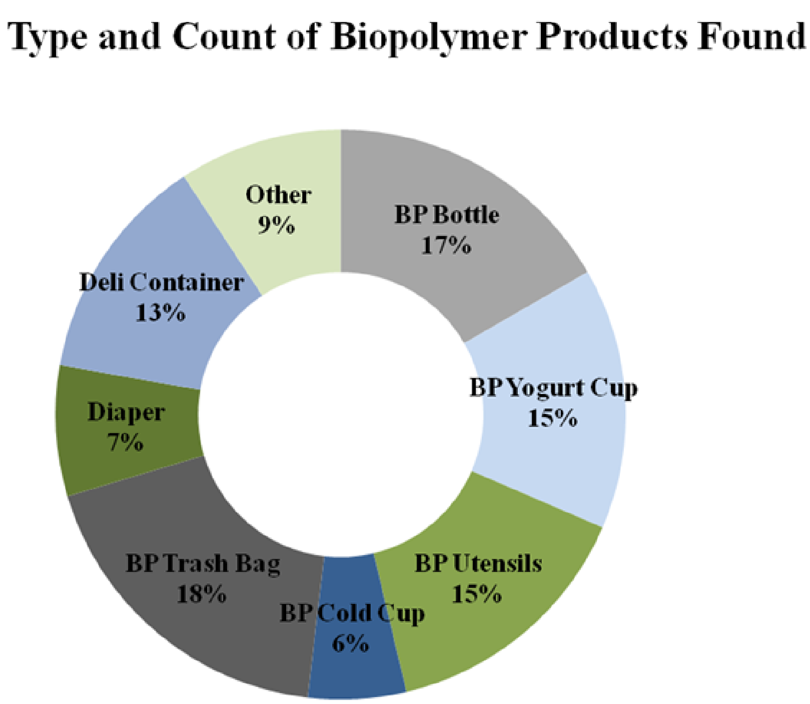

Figure 18. The total count of biopolymer products found categorized by product type. The "other" type consists of sponges, a soap bottle, straws, and a party pack with assorted biopolymer products such as compostable utensils and cups. BP = biopolymer

There are a limited number of biopolymer products (of any type, compostable, biodegradable, or non-biodegradable) available for residential consumers to buy and the Sprouts store manager described the sales volume for compostable utensils as low. Furthermore, even with the number seven plastics, the total number of products identified represents a very small portion of all the plastic products and packaging in the grocery section of the stores, which the Spouts manager estimated ranged from hundreds for smaller grocers, to thousands for larger grocery stores. The results from this audit show that individuals are not coming into contact or purchasing many biopolymer products, of any kind, via their local grocers, and as such, use and disposal of any type of biopolymers in a residential setting is still quite low.

Preliminary interviews with NatureWorks LLC and EcoProducts supported these findings, and suggested that if and when consumers do come into contact with compostable biopolymers, it is most likely occurring in a commercial foodservice setting (e.g., restaurant) rather than at home. A representative from NatureWorks, stated that though they have some sales in grocery retail and food packaging they have more contact with the commercial food service sector. EcoProducts, a large manufacturer and distributor of compostable plastic products, reported that the vast majority of their sales are to commercial food service businesses. The main types of businesses EcoProducts sells to fall into six main categories: colleges and universities, corporate campuses, health care, large venues (e.g. professional sports arenas), restaurants, and the hospitality industry. The EcoProducts respondent also noted that as these products are not as competitive in a retail setting, such as a grocery store, compostable biopolymers do not see as much use in homes. In addition to this she stated that because of new mandates, such as the ones banning conventional plastics, larger organizations are increasingly turning towards compostable biopolymers. Though consumers are using compostable plastics in a limited way in a residential setting, the majority of contact is within institutional settings, specifically commercial food service.

Compostable Biopolymer User Interviews

More than thirty organizations with a commercial food service component and possible compostable biopolymer use were contacted. Out of those thirty, twelve interviews were conducted between June 1st, 2014 and September 1st, 2014. Each food service category had three interviews attributed to it, except hospitals as we were not able to find any in the Phoenix area that used compostable biopolymer products. The interview process proved helpful because it revealed information typically missed in quantitative data collection related to compostable biopolymer disposal, particularly related to the importance of communication in the overall waste management system. Generally, most organizations using compostable biopolymers sent their waste to landfill. Out of the twelve organizations, three composted their compostable biopolymers – all from the cafeteria category. None of the organizations interviewed disposed of their compostable biopolymers by recycling, which is logical as these products are not accepted in municipal recycling facilities (Song et al. 2009).The motivation behind these disposal decisions, and reasons given for purchase, will be discussed in the subsequent sections.

Understanding Motivation

For each food service category there were a variety of reasons cited for the decision to purchase and use these products. Figure 19 is a graphical representation of the motivations that food service organizations shared for purchasing compostable plastics. Many of the reasons given from recreational concessions, limited food service establishments, and caterers were related or overlapped, and out of the four food service types all but cafeterias sent their compostable biopolymers to landfill. All companies who landfilled their compostable biopolymers (recreational concessions, limited food service establishments, and caterers) stated that using compostable biopolymer products aligned with the organizations' intention and desire to be a sustainable company. In addition, they wanted to use biopolymers for their perceived environmental benefits, to have the "greenest" footprint possible, and to align with their environmental branding. Another common reason given among the landfillers of biopolymers was that integrating sustainability into business practices is considered the norm and that using compostable products helped them fulfill that expectation. Many recreational concessions noted the need to stay competitive in contract renegotiations and used compostable biopolymers as a way to signal a move towards sustainable business operations and to align with their contractors' goals. Other reasons given across the organizations who landfill compostable biopolymers included wanting to use products that broke down (they believed the PLA products would degrade in landfills), wanting to avoid the use of conventional plastics, and that they wanted to support products that used bio-based feedstocks. There was only one case where the main reason for purchase was driven by individual consumer preference. In this instance, a caterer bought biopolymers to have on hand in case a client specifically asked for them.

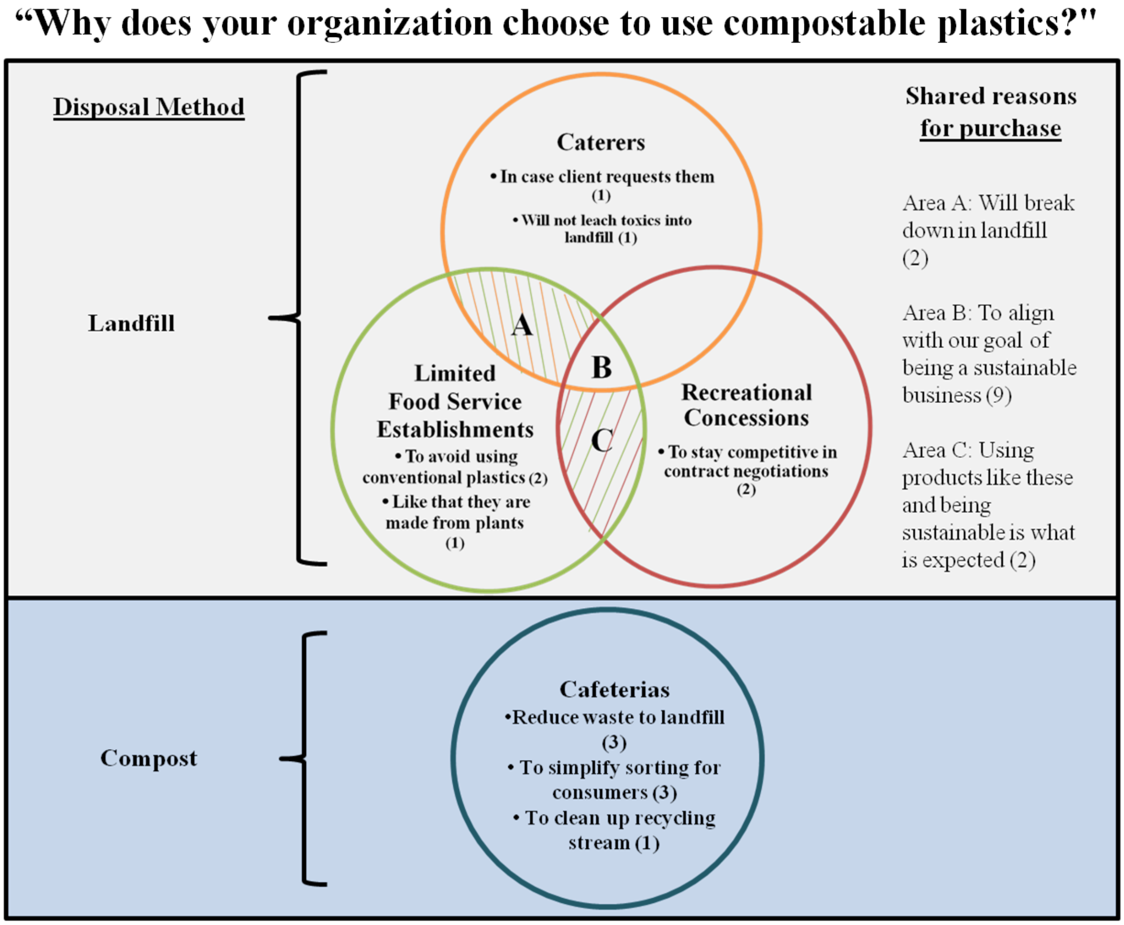

Figure 19. Responses given to the question "why does your organization choose to use compostable plastics?" For each food service category there were three organizations interviewed; the four categories are Caterers, Limited Food Service Establishments, Recreational Concessions, and Cafeterias. The numbers in parentheses next to the stated reasons indicate how many organizations gave that particular response.

Cafeterias, the only food service category where compostable biopolymers were being sent to compost facilities, had noticeably different reasons for purchasing biopolymers. It is important to note that this is not likely the case for all cafeterias across the nation, as an elementary school or correctional facility cafeteria may operate in a much different manner. Like the other food service types, all three cafeterias valued integrating sustainability into their business practices or are motivated by broader sustainability goals to use compostable bioplastics, but unlike the rest of the organizations they all cited specific and measurable waste reduction goals that they were trying to achieve. All three organizations also said that they used biopolymers in order to simplify sorting so as to achieve greater waste diversion. Using compostable biopolymers can reduce the time individuals spend sorting trash and help simplify the process, which reduces contamination and thus helps drive diversion rates higher, as previous research has shown (Hottle, Bilec et al. 2015). Other reasons given were that that switching to completely reusable products (e.g. ceramic plates and cups) was cost prohibitive and that compostable biopolymers were able to replace a wide variety of products typically destined to landfill which would further reduce the overall waste of the organization.

Out of all the reasons given between cafeterias and the other food service categories, interviewers documented very few instances of greenwashing, which is defined as "a superficial or insincere display of concern for the environment" (Collins English Dictionary, 2014). In most cases the organizations felt strongly about working to make decisions that produced positive impacts for both the environment and the organization. For most all food service categories, these products were more expensive than conventional disposables, but purchasers were willing to pay more because they believed they were making the right choice. One limited food service establishment was so committed to buying compostable biopolymers that after a period of financial hardship where they were not able to afford compostable bioplastics they promptly resumed buying them even before they had completely recovered financially.

Though all organizations cared about the environment and the responsibility of the choices they were making, not all organizations had the resources to allocate to detailed analysis and management of these products. This can been seen in the two instances where respondents' purchase decision was motivated in part, because they believed the compostable biopolymers would degrade in landfills; compared to cafeteria managers who thoroughly understood where these products would and would not degrade. All of the cafeterias interviewed (ASU, Stanford, and Intel) were part of larger organizations that employ hundreds or thousands of people and have substantial annual operating budgets. Similarly, all cafeterias also had strategic sustainability plans and measurable sustainability goals. Even for recreational food courts, which are relatively large, their concessions were contracted (in two out of the three cases) by smaller local companies. In addition, each cafeteria had a dedicated project manager who specifically focused on issues related to sustainability and waste management.

For the most part, companies from the other three categories were much smaller, and the individual deciding what to purchase had many other duties and responsibilities. For example, for all limited food service establishments the owner was the purchaser, as well as the marketing director, human resources, the kitchen manager, and they also often worked in the café during the day cooking or serving. All organizations from the different food service categories were trying to make good choices but the disparity in overall organizational resources impacted decision making. In the case of organizations with limited resources, some switched over traditionally recyclable products (such as cold cups) to a compostable biopolymer product which resulted in an increase in waste being sent to landfill as they could not compost the cups, which could have previously been sent to a recycling facility.

Larger drivers, i.e. conventional plastic bans, organics recycling mandates, and a growing trend to reduce waste to landfill could also be seen in organizations' decision to purchase compostable plastics. It is most clearly reflected in the cafeteria food service segment, especially in Stanford's case where they are working to meet state and city waste diversion goals and abide by laws that ban PS and conventional plastic bags. Over the past few years both Intel and ASU decided, independent of regional laws, to establish waste to landfill reduction goals, with ASU originally having set a goal to reach zero waste by 2015. In every case organizations were using compostable biopolymer products in response to the growing social trend to integrate sustainability into business practices and, for a variety of reasons, they believed using compostable biopolymers represented a more sustainable option. Aside from cafeterias, the decision to purchase compostable plastics did not seem linked to organizations' ability to compost them. In addition, purchasing decisions had very little to do with individual consumer preference. Out of the twelve organizations only one stated that they bought compostable biopolymers because of customer demand. Many organizations stated that few customers had ever explicitly commented on the use of compostable bioplastics or seemed to have any awareness of them. Overall, our findings suggest that neither residential consumers nor food service patrons are driving the purchase and use of compostable biopolymers; the primary drivers are linked to organizational waste diversion and sustainability goals. Legislative bans were not found to be the exclusive drivers among interviews; though they did accompany organizational drivers in states with bans.

Understanding Disposal

As noted before, all organizations in the catering category, the limited food service category, and the recreational concessions category sent compostable biopolymers to landfill. In contrast, all three organizations in the cafeteria category did their best to send the compostable biopolymers to composting facilities. Out of the nine facilities that sent their compostable bioplastics to landfill, all stated lack of access to commercial composting infrastructure which were also able to accept these products, as the main reason for landfilling. Two of the three limited food service establishments have a commercial composter, but explained that their composter did not accept compostable biopolymers. For recreational concessions and caterers a handful of organizations had some kind of pre-consumer organics disposal stream, so that organics could be composted or anaerobically digested. Pre-consumer organic wastes are the kitchen food scraps, and other organic waste such as cardboard, that is generated behind the counter by the kitchen or the organizations staff. The Phoenix Convention Center used an ORCA on-site organic waste aerobic digester which allows food waste to be sent to wastewater treatment for disposal, Desert Botanical Gardens had staff that came and picked up food scraps to use for composting, and Atlasta Catering composted all pre-consumer organics with a local commercial facility. All nine of the organizations that did not compost their compostable biopolymers explicitly stated a desire to find a commercial facility that would accept them, even if it meant paying more for the service. One of the catering companies stated that they have been looking for two years to find a composter in the Phoenix valley that will accept their post-consumer organics waste. The Arizona Diamondbacks, which have occasionally held zero waste events, noted that they were only able to do so because the events were special occurrences and as such they had organics trucked approximately 140 miles away to a facility that accepted compostable plastics. It was decided that long-term transport to this facility was neither economically or environmentally sustainability for the organization. For the Phoenix area, and many other parts of the country, there is no easily accessible composting infrastructure. Even if there are commercial composters, it can be a challenge to find one that will accept organics mixed with compostable biopolymers.

Despite the infrastructure challenges, cafeterias that were located in the Phoenix metropolitan area were able to find a composter for their pre and post-consumer organics waste. Even though the cafeterias have been able to compost their compostable plastics, all three also stated sufficient composting infrastructure as one of the biggest challenges to using compostable biopolymers. In each of the three organizations project managers worked hard to find, collaborate with, or create the necessary composting infrastructure. For example, ASU worked with their hauler, Waste Management Inc., to find a location to which they could send their organic waste. In contrast, Stanford has had access to more commercial composting facilities, but finding a good fit was still a challenge. Stanford's respondent explained that development of the composting market has been crucial. In Stanford's vicinity three composting facilities are now operating: one that only accepts and sells high quality organics and soil amendment, one that creates a low quality compost for fill in construction projects and accepts most anything, and a composter that sits in between – they will accept compostable biopolymers and work to create a medium quality soil amendment that can be sold to residential and commercial customers. For Stanford it was the development of the regional compost market that dictated their ability to find a facility that would accept their post-consumer organics and compostable biopolymers. In sum, it seems that three components created the necessary conditions that enabled cafeterias to send their compostable biopolymers to a compost facility: they each had measurable waste to landfill goals, waste diversion and organics programs were actively managed and monitored, and each cafeteria had the resources to dedicate to the above tasks and to secure a commercial composter or connect to robust regional infrastructure that was already intact.

It is important to note that even with organizations who have commercial composter services, some compostable biopolymers still ended up being sent to landfill. For all cafeterias interviewed, this was the case, though the percentage lost to landfill could not be determined. For post-consumer separated waste streams this is a common occurrence, and can be seen with recycling as well as with separated organic streams. For a number of reasons, it is very difficult to get 100% of waste sorted correctly and to the desired waste treatment facility. For compostable biopolymers, organizations noted that diversion loss can occur in two ways, onsite and then at the commercial composters facility itself. Within the organization, individuals not sorting their waste into the correct bin (i.e. throwing their compostable cup into the landfill bin), custodial staff not correctly sorting bags at dumpster, and staff being directed to toss composting waste because it looks as if it has too much contamination, are all ways compostable biopolymers could end up being sent to landfill. At the compost facility, composters could reject entire organic waste loads because of too much contamination, and composters may screen biopolymers out of compost because it cannot be sorted from the other conventional plastic contaminates. Both of these decisions result in biopolymers being sent to landfill.

Contrary to what was observed for purchasing decisions, it is clear that individual consumers have more of an impact on compostable biopolymer end of life. Though individuals have more impact via their disposal decision, every type of organization was working to alter the overall system design to reduce this impact, both purposefully and otherwise. For example, caterers have many events where trained servers clear and sort trash (regardless of the type of disposables used), some of the limited food service establishments do a post sort of all their organic waste, recreational concessions may utilize bin-guards (staff that stand by the waste bins and help consumers sort all waste correctly). Cafeterias also identified a number of additional ways they mitigate losses in order to get organics diversion rates as high as possible. Intel closely monitors all landfill and compost dumpsters and follows up immediately with staff if there are unexpected tonnage increases or decreases. Strong relationships, supported by regular meetings and trainings with property management and their contract cleaning company is used to drive better diversion rates as well. ASU and

Stanford use a variety of different management strategies to correct individual sorting error including effective signage and bin placement, bin guards, and post event sorting. In addition, they also work closely with their contractors be they food service, custodial staff, or waste handlers.

For all organizations the most important factor related to compostable biopolymer disposal decisions was access to compost infrastructure and the overall compost market development. For key decision makers, such as municipal solid waste managers or directors, especially in cities where bio-preferred purchasing is encouraged, it may be beneficial to devote equal attention and resources to support the composting infrastructure for products that demonstrate improved environmental impacts for composting rather than landfilling (Yates and Barlow 2013). The choice to compost biopolymers may also result in consequential diversion of food waste for composting, improving the environmental impacts of food waste and those associated with biopolymer disposal (Levis and Barlaz 2011). This would include the opportunity for all organizations with pre and post-consumer organics access to commercial composting, and for composters to be supported by a robust market that supplies compost to a variety of different sectors.

Conclusion

After the grocery store audits which identified relatively few biopolymers were available in retail settings, this research focused on compostable biopolymers in commercial food service settings. The decision to purchase these products is impacted by larger social trends, such as zero waste initiatives and plastics bans, but individual user motivation was based on a variety of different factors. For all food service categories disposal decisions relied heavily on the regional waste infrastructure that was available. In Phoenix where municipal commercial composting is not readily available, for the organizations we spoke with, most compostable biopolymers were being landfilled. Consequently, in regions where there is no commercial composting infrastructure, this research suggests that most food service providers are sending biopolymers to landfill, however quantifying the mass of composting and landfill waste streams will require waste audits and material flow analyses. This research also found that motivation to purchase was not explicitly linked to the ability to compost the compostable biopolymers nor driven by individual consumer preference.

Sustainability of biopolymers with a potential use in food service industries must consider the available waste infrastructure and disposal methods of commercial food service providers. In addition, the most appropriate method of disposal for compostable biopolymers may depend on individual business factors and with which impacts the organization is most concerned. For example, with a commercial food service business which uses large quantities of disposable cold cups, and is most concerned with decreasing waste to landfill, it may be more sustainable to stay with conventional plastic products that can be readily recycled (Hottle, Bilec et al. 2015). Alternatively, a business that creates large quantities of disposable, food-soiled products, and is concerned with decreasing waste to landfill, may find that the most appropriate option for their business is compostable biopolymers as most material recovery facilities do not accept small plastics like utensils and have trouble with organic contamination in the recycling process.

However, it is beyond the scope of this paper to decide if compostable bio-polymers can ultimately be considered a sustainable product or which end of life treatment is the most environmentally beneficial. It is important to note that the peer reviewed literature lacks evidence and consensus one way or the other related to the sustainability of compostable biopolymers. Most compostable biopolymer assessments to date focus on plastic production and ignore the complicated realities of waste handling (Hottle, Bilec, and Landis 2013, Tabone et al. 2010, Vink and Davies 2015b, Gerngross and Slater 2000). Many studies on municipal solid waste treatment methods vary widely. For example, composting has been found to be one of the best ways to treat food and food soiled waste because of the reduced methane generation compared to landfill while on the other hand it has been demonstrated to be one of the worst options because there is no opportunity for energy recovery via anaerobic digestion or landfill gas capture (Finnveden, Björklund et al. 2007, Marchettini, Ridolfi, and Rustici 2007, Kim and Kim 2010, Saer, Lansing et al. 2013, Favoino and Hogg 2008). For compostable bioplastics which are not food soiled some studies show it may be preferable to landfill them (Lundie and Peters 2005).

This research demonstrates a demand for compostable biopolymer plastics among various food service providers but the ambiguity regarding end of life is pervasive. The uncertainty concerning end of life could undermine the investments and efforts of stakeholders throughout the supply chain who are creating and using products they hope will have improved environmental performance. Though there is clearly a need for further research around what end of life treatments are the most beneficial, the compostable biopolymers continue to expand into the plastics market. In order to improve the overall environmental performance of compostable biopolymers it is important to understand the motivations behind purchasing, and for compostable biopolymers that perform better in composting situations, create robust waste systems that can accommodate increasing volumes of compostable waste. Increased communication along the life-cycle for compostable biopolymers can help stakeholders create a dialogue, clarifying their goals and expectations as they assume greater responsibility for the impacts of the products they use.

CHAPTER 6

BIOPOLYMER PRODUCTION AND END OF LIFE COMPARISIONS USING LIFE-CYCLE ASSESSMENT

This chapter addresses research objective 4) Quantify the end of life environmental impacts of compostable biopolymer scenarios via life cycle assessment incorporating findings from objectives 1-3

Introduction

Biopolymers are a growing segment of the plastics market primarily in packaging and disposable products (Shen, Worrell, and Patel 2010). These bio-based plastics have gained interest as an alternative to fossil based plastics because of the potential to shift to more environmentally friendly production and waste handling. Some commercially available biopolymers are compostable, though many require conditions only attainable in commercial-scale composting facilities (Hottle, Bilec, and Landis 2013, Yates and Barlow 2013). This life cycle assessment (LCA) seeks to explore the impacts associated with the production and disposal of biopolymers compared to fossil-based plastics; most literature on biopolymers has neglected EOL, which studies say could significantly impact the overall life-cycle impacts of biopolymers.

Traditional plastic production is a mature industry with materials produced using fossil resources such as petroleum and natural gas. The US Environmental Protection Agency (EPA) reported that in 2013, plastics contributed to 13%, by weight, of the municipal solid waste (MSW) in the US totaling about 32.5 million tons of plastic waste generated annually (USEPA 2015a). Nine percent of plastics entering the waste stream were recovered for recycling, though recovery rate is not necessarily indicative of a final recycling rate. Of total plastics, about 91% are discarded to a landfill or are incinerated. While overall recovery of plastics for recycling was only 9%, recovery of certain plastic containers is more significant. In 2013 polyethylene terephthalate (PET) soft drink bottles were recovered at a rate of 31% while high-density polyethylene (HDPE) milk and water bottles were estimated at about 28%. Packaging and nondurable plastics in MSW totaled 20.5 million tons, of which 10% were recovered (USEPA 2015a).

Despite the ability to increase recovery rates of fossil based plastic materials, concerns over increased fossil resource use, greenhouse gas emissions, pollution, costs of landfilling, and human health impacts associated with plastics have driven an increasing interest in the use of biopolymers (Gironi and Piemonte 2011, Ren 2003, Gómez and Michel Jr 2013, Kijchavengkul and Auras 2008, Álvarez-Chávez et al. 2012). Biopolymers are plastics that can be produced, at least in part, from renewable materials such as corn and sugar cane. Some biopolymers, like Bio-PET, are blends of conventional and renewable feedstocks (Hartmann 1998, Shen, Worrell, and Patel 2010, Shen, Haufe, and Patel 2009).

Bio-PET is the same polymer as PET aside from the fact that the ethylene component is bio-based rather than fossil-based and they can be recycled together since they are chemically identical (Tabone et al. 2010, Morschbacker 2009). Bio-PET is used in Coca-Cola's Plantbottle™ and is the most prevalent biopolymer in the market (European Bioplastics 2014, Coca-Cola 2012). Other biopolymers such as PLA and thermoplastic starch (TPS) are compostable but not recyclable given the current infrastructure and markets (Song et al. 2009). PLA is the most prevalent compostable biopolymer in the market (European Bioplastics 2014). Compostable biopolymers must conform to American Society of Testing and Materials (ASTM) standards; ASTM D6400-04 Standard Specification for Compostable

Plastics, ASTM D6868-03 Standard Specification for Biodegradable Plastics Used as Coatings on Paper and Other Compostable Substrates, and ASTM D5338-98(2003) Standard Test Method for Determining Aerobic Biodegradation of Plastic Materials Under Controlled Composting Conditions (Song et al. 2009, ASTM 2004, 2003b, a). Some biopolymer products are coming under increasing scrutiny because they are not fully degrading in commercial composting facilities, complicating the waste management for users and waste handlers of these products (Cedar Grove 2012, Ghorpade, Gennadios, and Hanna 2001, Mohee and Unmar 2007, Gómez and Michel Jr 2013).

LCA is a well-established method used to quantify the environmental impacts of products and processes. Previous life cycle assessments of biopolymers are largely limited to assessments of global warming potential and fossil fuel depletion which may favor biopolymers because of the inherent properties of plastics made from biogenic carbon compared to fossil based plastics and may miss the potential environmental tradeoffs that can occur when shifting to agriculturally produced feedstocks (Miller, Landis, and Theis 2007, Landis, Miller, and Theis 2007, Hottle, Bilec, and Landis 2013).

End of life (EOL), the processes involved in handling and treating a material after it enters the waste stream, has been shown to be significant for traditional plastics (Björklund and Finnveden 2005, Hopewell, Dvorak, and Kosior 2009, USEPA 2015a). Few of the past LCAs for biopolymers adequately address EOL. When waste scenarios are included in LCAs, findings vary widely based on the chosen EOL scenarios (e.g. landfilling, recycling, incinerating, composting) which are not always based on realistically available disposal methods (Hottle, Bilec, and Landis 2013, Yates and Barlow 2013, Koller et al. 2013, Shen and Patel 2008, Weiss et al. 2012). Of the twenty-one LCAs reviewed in Hottle et al. (2013) only seven included EOL within the system boundaries. Of the seven studies which evaluated EOL, only one study included composting as an EOL scenario for compostable biopolymers while five included scenarios for incineration which is a less common disposal method for plastics compared to landfilling and recycling (USEPA 2015a). Increasingly there have been calls to further investigate the EOL for biopolymers (Yates and Barlow 2013, Weiss et al. 2012, Rossi et al. 2014, Hottle, Bilec, and Landis 2013, Hermann et al. 2011). At the same time, there have been inventory improvements for the production of biopolymers and waste streams in the United States that can be used for refining LCAs (Vink, Davies, and Kolstad 2010, Hermann et al. 2011, Pressley, Levis et al. 2015, Kolstad et al. 2012, Vink and Davies 2015a).

The LCA presented herein models the production of biopolymers and traditional fossil based plastics as well as EOL scenarios including landfill, compost, and recycle. For each recyclable polymer (PET, Bio-PET, HDPE, Bio-HDPE, LDPE, and Bio-LDPE) landfill and recycling scenarios are used, while the remaining compostable polymers (PLA and TPS) are modeled using compost and landfill scenarios. PLA has two landfill scenarios to model two different forms of PLA, amorphous and crystalline, both of which are used in products on the market. Amorphous PLA is used in applications such as clear cups, films, and coatings. Crystalline PLA is used in applications such as cutlery and hot coffee cup lids. The amorphous PLA emits some methane as it breaks down in the landfill, while the crystalline PLA does not degrade under landfill conditions (Kolstad et al. 2012).

Methods

Scope and System Boundary

Figure 20 depicts the system boundaries for the LCA. The functional unit for the study is 1 kg of polymer. Eight polymers were evaluated in the study: PLA, TPS, PET, Bio-

PET, HDPE, Bio-HDPE, LDPE, and Bio-LDPE. These polymers were selected because PLA, TPS, Bio-PET, and Bio-PE represent the most common biopolymers globally, with 11.4%, 11.3%, 12.3% and 37% market shares of biopolymers respectively (European Bioplastics 2014). PLA is made from corn grown in the US and produced at the NatureWorks facility in Blair, Nebraska. The Bio-PET is synthesized in part from ethylene glycol which comes from sugar cane grown and distilled in Brazil. Bio-HDPE and Bio-LDPE modeled the use of Brazilian sugar cane based ethylene. Both the ethylene glycol and ethylene processes included freighter shipment to the U.S.

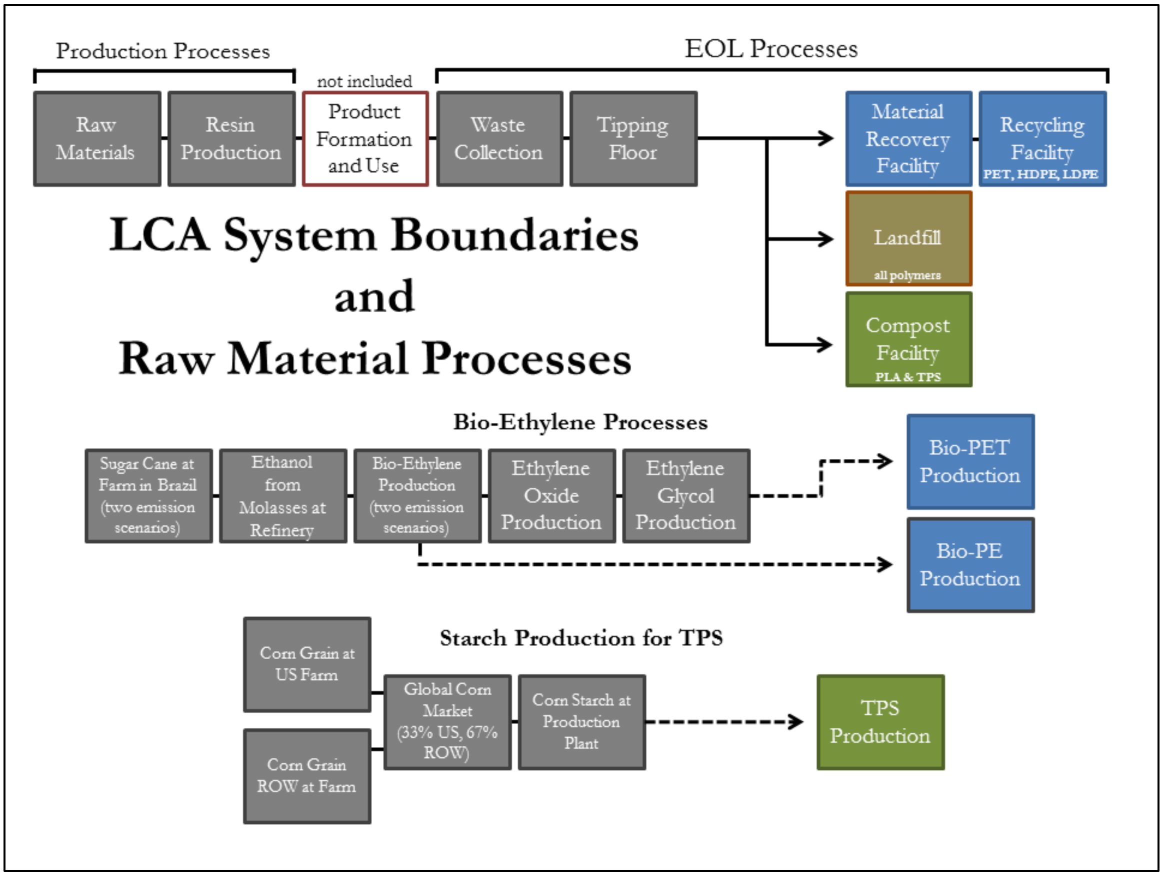

Figure 20. System boundaries for the LCA of eight biopolymers including production and EOL processes. Included in the assessments is transportation between EOL processes and energy inputs for all of the processes. All of the polymers undergo the same initial EOL processes of 'Waste Collection' and 'Tipping Floor' handling. The colors represent different EOL pathways with recyclable polymers and recycling processes in blue, compostable polymers and compost processes in green, and landfill in brown which is a scenario for all polymers. The bio-ethylene and starch processes contribute to the raw materials used in the respective inventories. PLA production was modeled using gross emissions data and not a process flow within the modeling software (Vink, Davies, and Kolstad 2010).

In addition to the production of the polymer resins, this assessment included EOL processes starting with waste services common to all polymers which are waste collection and tipping floor activities (Figure 20) and ended with three possible options for final EOL; recycle, compost, or landfill. Each polymer was evaluated using two EOL scenarios; landfill and one alternative EOL treatment. All polymers were evaluated using a landfill EOL scenario because landfilling is the predominant destination for plastics in the United States. The second EOL scenario was based upon whether the material is recyclable or compostable. The different methods of transportation used to move waste materials between the EOL processes were included in the assessment. The EOL scenarios, travel distances, and handling processes were based on services in the Phoenix metropolitan region, where this study took place. The recycling process for plastics included baling at a material recovery facility (MRF), shipment via semi-truck to Long Beach, California, and transportation on an ocean freighter to Hong Kong (Campbell 2014). The benefits of offsetting virgin material production via recycling are allocated to the plastics during EOL in this LCA.

The product formation (e.g. blown film extrusion, sheet extrusion, thermoforming, and injection molding) and use of plastics were excluded from this LCA because these processes are similar for both biopolymers and their fossil-based counterparts.

Inventory

Table 9 presents the various processes used to compile the life cycle inventory (LCI) data for the eight polymers; detailed LCI is presented in the SI. The process used to model the tipping floor is based on data collected during site visits to the Waste Management Inc. (WM) Sky Harbor Tipping Floor located in Tempe, Arizona from 11 July 2014 to 17 July 2014 and follow-up communications with WM employees. The remaining processes were created or customized using data available within existing databases and literature so the inventories are applicable to the scope and boundaries of the study. These processes are described in further detail.

Table 9. Sources of inventory data for the processes used in the LCA and the name of the processes within the database or literature. More details are presented in the SI..

Process	Database or Source	Process Name
PLA production	Vink 2010	2009 Ingeo PLA
TPS production	ecoinvent v2/Hottle 2013	revised TPS production including: Modified starch, at plant/RER unit process
PET production	Franklin Associates 2011	cradle-to-resin life-cycle inventory results for PET resin
HDPE production	Franklin Associates 2011	cradle-to-resin life-cycle inventory results for HDPE resin
LDPE production	Franklin Associates 2011	cradle-to-resin life-cycle inventory results for LDPE resin
Bio-ethylene production	ecoinvent v2/Tabone 2010	multiple scenarios
PLA landfilling	ecoinvent v2/Kolstad 2012	custom PLA landfilling including: Disposal, polyethylene, 0.4% water, to sanitary landfill/CH unit process
PLA composting	average – ecoinvent v3/site visit, personal communication, and Kruger 2009 (Kruger, Kauertz, and Detzel 2009)	custom PLA composting and Biowaste [CH] treatment of, composting | Alloc Def unit process
TPS landfilling	ecoinvent v2	Disposal, packaging cardboard, 19.6% water, to sanitary landfill/CH unit process
TPS composting	average – ecoinvent v3/site visit, personal communication, Piemonte 2011 and Hermann 2010	custom TPS composting and Biowaste [CH] treatment of, composting and custom TPS composting | Alloc Def unit process
PET landfilling	ecoinvent v2	Disposal, polyethylene terephtalate, 0.2% water, to sanitary landfill/CH unit process
PET recycling	Franklin Associates 2011	custom process including: PET (waste treatment) [GLO] recycling of PET
PE landfilling	ecoinvent v2	Disposal, polyethylene, 0.4% water, to sanitary landfill/CH unit process
PE recycling	Franklin Associates 2011	custom process including: PE (waste treatment) [GLO] recycling of PE
MRF operations	Pressley 2015	multiple scenarios based on polymer type
tipping floor operations	site visit, personal communication	custom process
waste collection	ecoinvent v3	Municipal waste collection service by 21 metric ton lorry [GLO] market for | Alloc Def, unit process
local transportation & recyclables ground shipping	USLCI v1.6	Transport, single unit truck, diesel powered/US
recyclables international shipping	USLCI v1.6	Transport, ocean freighter, average fuel mix/US

Production Inventories

The production of TPS was modeled using the 'Modified starch, at plant/RER U' process in the ecoinvent 2 (Frischknecht et al. 2005) database and adjustments to base the process on global corn production and US natural gas (Hottle, Bilec, and Landis 2013). This was done by modifying the sub-process 'Maize starch, at plant/DE U' which contributes 0.33561 kg/kg to TPS. Within the 'Maize starch' process 1.261 kg/kg 'Grain maize IP, at farm/CH U' was replaced with an equal quantity of 'Maize grain GLO market for Alloc Def, U.' Within the 'Modified starch, at plant/RER U,' 12.547 MJ of 'Natural gas, high pressure, at consumer/IT U' was replaced by an equivalent energy content of 'Natural gas, at consumer/RNA U.'

The ethylene glycol derived from Brazilian grown sugar cane was used to synthesize Bio-PET. Ethylene production, which is a sub-process of ethylene glycol production, was used to synthesize Bio-HDPE and Bio-LDPE (Figure 20). The production processes of bio-ethylene and bio-ethylene glycol were modeled using data from Tabone et al. (2010) which details the structure of these scenarios in SimaPro. The scenarios included both high and low rates of emissions for sugar cane production and two methodologies for modeling the production of ethylene. Ocean transport for Bio-PET was adjusted from Tabone's (Tabone et al. 2010) 15,000 miles to 12,482 km (6,740 nautical miles) representing transport from Santos, Brazil to Port of Houston, which equates to 0.01248 tkm per kilogram of material shipped. The Bio-PET inventories were created by subtracting 0.4101 kg of ethylene glycol within the ecoinvent database from the Franklin Associates inventory for PET resin (Franklin Associates 2011) and replacing it with an equivalent amount of bio-based ethylene glycol for each of the four emissions scenario. The Bio-LDPE inventories were created by subtracting 1.008 kg/kg 'ethylene, average GLO market for Alloc Def, U' from the Franklin

Associates inventory for LDPE resin(Franklin Associates 2011) and replacing it with 1.008 kg/kg of bio-ethylene production for each of the four emissions scenario. The Bio-HDPE inventories were created by subtracting 0.990 kg/kg 'ethylene, average GLO market for Alloc Def, U' from the Franklin Associates inventory for HDPE resin(Franklin Associates 2011) and replacing it with 0.990 kg/kg of bio-ethylene production for each of the four emissions scenarios.

EOL Inventories

The tipping floor process was developed using data collected at the WM Sky Harbor Tipping floor. The facility handles 1,995,806 kg (2,200 short tons) of waste material per day. The facility uses around 27,255 L (7,200 gal) of water and 530 L (140 gal) of diesel fuel per day. The process was created by adding the following sub-process together: 0.0225788526 kg 'Tap water, at user ROW market for Alloc Def, U,' 0. 000266 L 'Diesel, at refinery/l/US,' and 0.0095494 MJ 'Equipment use - diesel, burned in building machine GLO' (based on the energy content of the diesel fuel) per kilogram of waste handled(Frischknecht et al. 2005, Weidema, Bauer et al. 2013). Diesel fuel was removed as a sub-process for the 'Equipment use' process since there was data available for the amount of diesel consumed and the inventory for US diesel was used. For PET, Bio-PET, HDPE, Bio-HDPE, LDPE, and Bio-LDPE MRF inventories were developed based on literature which includes wire for baling, diesel, and electricity at varying rates depending upon the waste stream entering the MRF (e.g. single stream, pre-sorted, mixed waste, dual stream) and the type of polymer being sorted (Appendix G). LDPE was assumed to be film waste and was modeled as such in the MRF scenarios.

For the transportation processes, the average distance to processing centers for the Phoenix, Arizona service area were mapped using Google Earth and used as a baseline for the model (Appendix H). Recycling incurred additional transport of the baled materials; 610 km ground transport (Phoenix, AZ to Long Beach, CA) and 11,686 km ocean freighter transport (Long Beach, CA to Hong Kong, China).

TPS and PLA composting were modeled using two separate inventories for each material. One compost scenario used the 'Biowaste [CH] treatment of, composting' unit process with customized carbon dioxide and methane emissions based on stoichiometric calculations for the degradation of PLA and TPS. The calculations for both PLA and TPS were based on 60% degradation with 95% of the degraded material converting to CO_2 and the remaining 5% converting to CH_4 (Piemonte 2011, Hermann et al. 2011). For PLA this equated to 0.010 kg of biogenic methane and 1.05 kg of biogenic carbon dioxide per kilogram of PLA. For TPS this equated to 0.0089 kg of biogenic methane and 0.93 kg of biogenic carbon dioxide per kilogram of TPS, which has 88.8% of the carbon content of PLA (Hermann et al. 2011). The second scenario used to model composting for PLA and TPS used the same CO_2 and CH_4 emissions described above but the underlying processes were based on data collected during site visits to the Maricopa Organics Recovery Facility operated by Garick in Maricopa, Arizona between January 2013 and June 2014. The facility handled 6,000 kg (reported as lbs 400,000 lbs/month) of organic waste per day. The facility used around 68,000 L (reported as 18,000 gal/day) of water and 150 L (reported as 40 gal/day) of diesel fuel per day. The composting inventory developed to model Garick's composting operations was created by adding the following sub-process together: 5.1 kg 'Tap water, at user ROW market for Alloc Def, U,' 0.0050197608 L 'Diesel, at refinery/l/US,' and 0.180063801 MJ 'Equipment use - diesel, burned in building machine

GLO' (based on the energy content of the diesel fuel) for one kilogram of organics processed (Frischknecht et al. 2005, Weidema et al. 2013). Diesel fuel was replaced as a sub-process for the 'Equipment use' process since there was data available for the actual amount of diesel consumed and the upstream inventory for US diesel was used.

The two PLA landfill scenarios distinguished between different forms of PLA, amorphous and crystalline. The amorphous PLA emits some methane as it breaks down in the landfill, while crystalline PLA does not degrade under landfill conditions (Kolstad et al. 2012). Both scenarios use the 'Disposal, polyethylene terephtalate, 0.2% water, to sanitary landfill/CH' process from the ecoinvent v2 database (Frischknecht et al. 2005) as a template. The "Disposal, polyethylene terephtalate' process was modified by eliminating outliers when compared to PE and cardboard landfill processes as summarized in Table 10.

Table 10. Modifications for PLA landfill scenarios based on the ecoinvent process for landfilling of PET

Original Emission	Unit	Value	New emission for crystalline PLA landfill scenario	New emission for amorphous PLA landfill scenario	Value	Rationale
Carbon dioxide, fossil burden from direct release or incineration of landfill biogas	kg/kg	1.39e-2	None	None	-	Fossil emissions based on degradation do not apply to PLA
Carbon monoxide, fossil burden from direct release or incineration of landfill biogas	kg/kg	7.86e-7	None	None	-	Fossil emissions based on degradation do not apply to PLA
Methane, fossil burden from direct release or incineration of landfill biogas	kg/kg	2.13e-3	None	Methane, biogenic	3.38e-2	Fossil emissions based on degradation do not apply to PLA. Biogenic methane added based on 40% degradation yielding 189 L/kg (Kolstad et al. 2012). All emissions were modeled as methane with no conversion to CO_2 via combustion.
Antimony, groundwater, long-term, emissions from long-term leachate	kg/kg	1.60e-5	None	None	-	Antimony to groundwater was removed because it was an outlier more than 1 order of magnitude higher than the landfill process for polyethylene and 5 orders higher than the landfill process for cardboard.

TPS landfilling was modeled using a proxy process, 'Disposal, packaging cardboard, 19.6% water, to sanitary landfill/CH.' There were no EOL inventories available for TPS; TPS and cardboard have similar material properties and carbon content (Hermann et al. 2011, Shen, Haufe, and Patel 2009).

The recycling processes for PET and PE, developed by Pré Consultants, were modified to represent the Franklin Associates inventories for US produced plastics. This was accomplished simply by replacing the sub-process for the offset of virgin plastic production from the ecoinvent process for PET and HDPE production to the corresponding Franklin Associates inventories.

Using the LCI data described above, the midpoint life cycle impact assessment factors (Appendices I, J, and K) were calculated using TRACI v2.1 (Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts) (Bare 2012). TRACI calculates impacts based on the inventories for global warming, eutrophication, ecotoxicity, acidification, ozone depletion, smog formation, carcinogens, non-carcinogens, respiratory effects, and fossil fuel depletion.

A sensitivity analysis was conducted to determine the impact that variables and assumptions within the study affect the overall results of the LCA and which life cycle phases may yield the greatest return when looking for system improvement. The sensitivity analysis was conducted using the minimum and maximum values for the variables and assumptions within the model based on processes with multiple inventory scenarios (e.g. ethylene and ethylene glycol production scenarios) as well as regionally specific transportation and infrastructure (e.g. compost facility and MRF type). The results are reported in terms of percent change from the baseline scenario which was used for the LCA.

Results and Discussion

Figure 21 illustrates the differences for the life-cycle impacts of production and the two EOL scenarios for each polymer type. The production impacts for the bio-ethylene plastics (i.e. Bio-PET, Bio-HDPE, and Bio-LDPE) manufactured using sugar cane grown in

Brazil are greater for all the impact categories except for global warming and fossil fuel depletion. Bio-ethylene production introduces different production processes with the most increases across impact categories resulting from distillation and farming. The process for distillation of ethanol from sugar contributes significantly to ozone depletion, smog, acidification, carcinogens, and respiratory effects. The process for sugar cane farming contributes significantly to ozone depletion, acidification, eutrophication, carcinogens, non-carcinogens, and ecotoxicity. The processes of ethylene dehydration from ethanol, phosphorus based fertilizers, and ocean freighter transportation also contributed to the differences between fossil plastics and the bio-ethylene based alternatives. The task of using low density feedstocks such as sugar cane to produce chemical feedstocks (i.e. ethylene) for plastic production requires processes like farming, distillation, and dehydration.

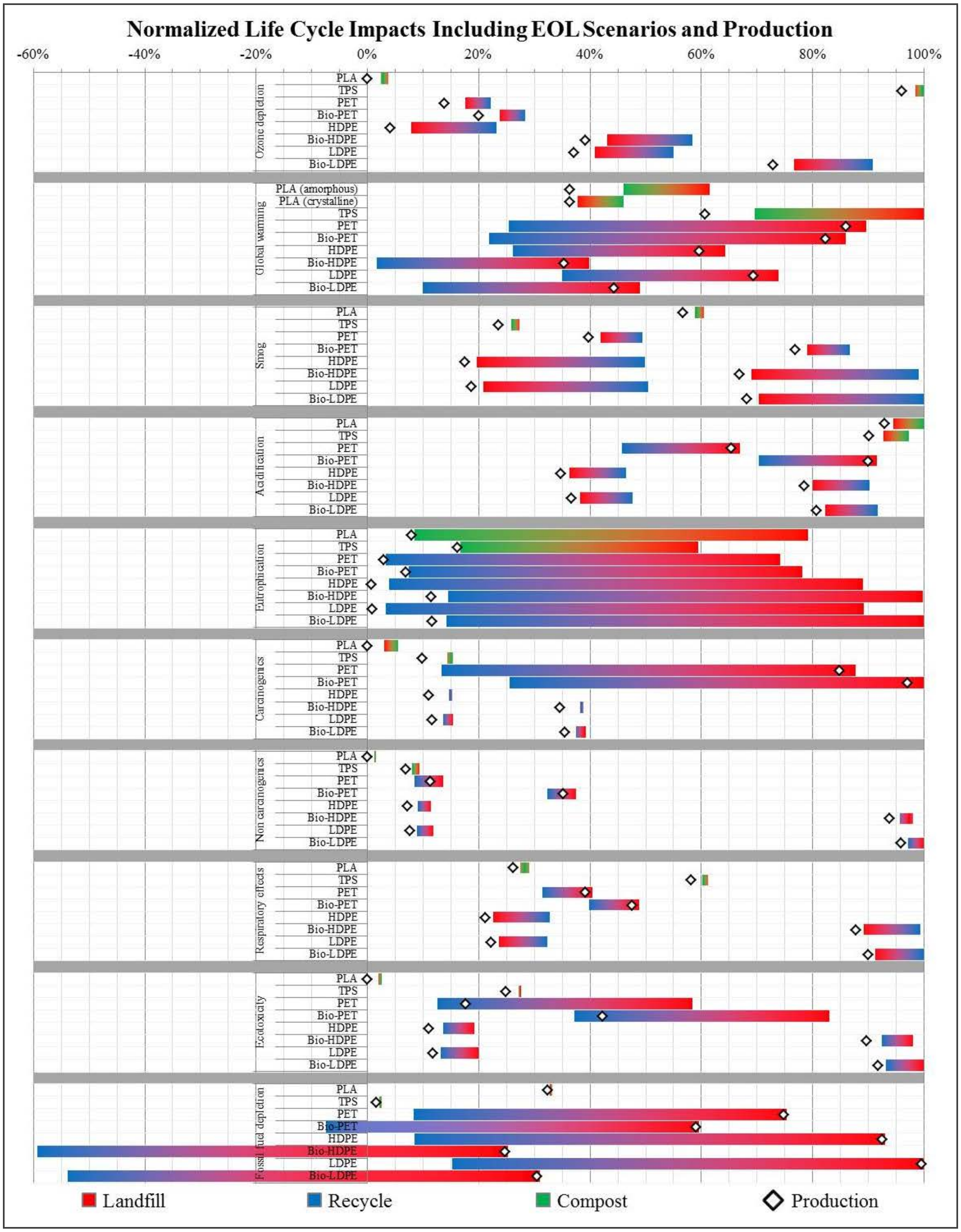

Figure 21. LCA results using TRACI 2.1(Bare 2012) for impact assessment. The graph presents findings for the production and EOL of eight different plastic materials. The production impacts are indicated by a black diamond. The net impacts from EOL are presented as a shaded bar ranging between two EOL options for each polymer; EOL impacts were added to production, but the range can be less than (due to offsets from

avoiding production of virgin plastics) and more than production. PLA has an amorphous and crystalline scenario in the 'Global warming' category. For all other categories there was no more than 1% difference between the two scenarios so only the impacts for crystalline PLA were graphed. All values for net life-cycle impacts are included in Appendix L.
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PLA and TPS, which are both manufactured using corn, have greater impacts to acidification and eutrophication than any of the fossil-based plastics. These impacts are the result of phosphorous, nitrogen, and sulfur which are used in fertilizers for the agricultural processes (e.g. diammonium phosphate, urea, and ammonium nitrate) as well as effluent wastes generated in starch production. TPS has additional nitrogen impacts from natural gas based plasticizers which contributed to acidification.

The EOL scenarios alter the life-cycle impacts in several different ways. Eutrophication impacts from landfilling for all polymers and fossil fuel depletion impact reductions from recycling stand out as the largest changes relative to production impacts. On the other hand, none of the EOL scenarios in the non-carcinogenics category change very dramatically one way or the other. Global warming impacts vary significantly based on EOL scenario and polymer degradation under landfill conditions. Amorphous PLA and TPS, which degrade under landfill conditions (Kolstad et al. 2012, Hermann et al. 2011), emit methane that causes higher global warming impacts than composting. Methane can be captured from landfills at various rates and flared or burned for energy recovery (Spokas et al. 2006). The emissions from TPS and PLA degradation in landfill cause higher global warming impacts than the alternative composting processes. Composting results in 57% CO_2 and 3% CH_4 based on the carbon content of the biopolymer and the remaining carbon persists in the soil (Piemonte 2011).

The TPS landfill scenario includes landfill gas capture and combustion resulting in 1.18e-2 kg of CO_2, 2.30e-5 kg of CH_4, and 7.57e-6 kg of CO per kilogram of TPS (Frischknecht et al. 2005). The amorphous PLA landfill scenario evaluated the worst case scenario for methane generation and did not include any conversion to CO_2 via combustion, providing an upper bound for impacts for PLA in landfills. Crystalline PLA does not degrade under landfill conditions (Kolstad et al. 2012) and therefore landfilling does not incur any emissions from the breakdown of the crystalline PLA.

Recycling has major implications for the life-cycle impacts of the traditional plastics and their bio-based counterparts. In some categories, recycling decreases the impacts associated with production because the benefits from offsetting virgin plastic production are allocated to the materials. This is the case for global warming and fossil fuel depletion for all of the recyclable plastics, as well as carcinogenics for PET and Bio-PET. For ozone depletion and smog, recycling incurs additional impacts beyond those of landfilling due to the transportation of the recyclable plastics from the material recovery facility to the recycling facility in China.

The offsets gained by recycling in the global warming and fossil fuel depletion categories are further accentuated by the use of bio-based ethylene for the production of Bio-PET, Bio-HDPE, and Bio-LDPE. Because recycling these polymers offsets the fossil-based resins, the recycled bio-polymers can achieve negative life-cycle impacts in the fossil fuel depletion category, shown in Figure 21. While the initial production processes are important, including EOL demonstrates that waste scenarios are also a critical factor in the environmental performance of plastics.

Figure 22 shows the process contribution to each impact category for EOL. For the recyclable materials, the most notable impacts result from the energy used during recycling, the equipment and fuel used at landfills, and the freighter shipping for bailed recyclables. The waste collection, ground transportation, and intermediate handling (i.e. tipping floor and MRF) are not major contributors for most of the impact categories; these processes combined for recyclables contribute, at most, to eleven percent (HDPE) of the maximum category impact in the ozone depletion and ten percent (HDPE) of the maximum impact for respiratory effects. International shipping does not need to be inherent to the process of recycling and the shipping incurs significant impacts, particularly for smog (47%), acidification (31%), non-carcinogenics (24%), respiratory effects (20%), and ecotoxicity (17%). Greater environmental benefits from recycling could be achieved if recyclable materials did not need to be shipped long distances.

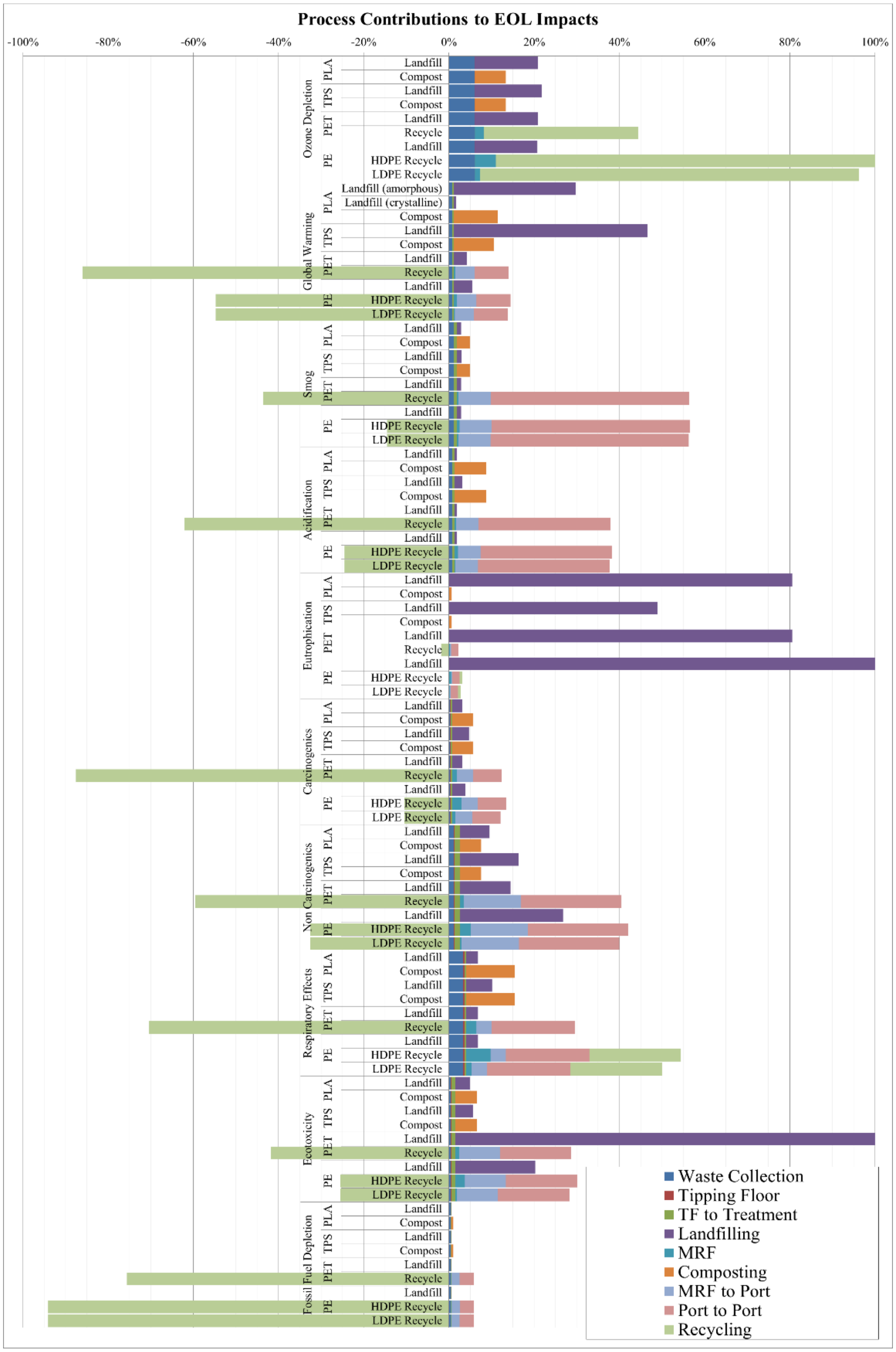

Figure 22. Process contributions to EOL impacts. TF = tipping floor, MRF = material recovery facility, port to port = shipping from Long Beach, CA to Hong Kong, China.

For PLA and TPS the most notable impacts result from the equipment and diesel fuel used at landfills and compost facilities are the dominant contributors to the EOL impacts for PLA and TPS. Composting PLA and TPS results in higher impacts than landfilling in six categories, smog, acidification, carcinogenics, respiratory effects, ecotoxicity, and fossil fuel depletion. The impacts that are higher for composting result from the active management and additional water use associated with the composting process. Similar to the recyclable materials, PLA and TPS have contributions to ozone depletion and respiratory effects resulting from waste collection, ground transportation, and intermediate handling. In the case of PLA and TPS these processes are an even larger proportion of the total EOL impacts including 46% of the ozone depletion impacts for composting of TPS and PLA, 71% of the global warming impacts for landfilling crystalline PLA, 68% of smog for landfilling of PLA, 68% of smog impacts for landfilling of PLA, and 100% of the fossil fuel depletion impacts for landfilling of PLA and TPS. The large difference between the global warming impacts of amorphous and crystalline PLA in the landfill scenarios is due entirely to the $CH_{4,}$ which contributes 94% of the global warming emissions for landfilled amorphous PLA. For composting PLA, 82% of the global warming impacts are due to the biogenic emissions from the PLA degrading. Likewise, biogenic emissions from composting and landfilling TPS equate to 80% and 98% respectively for global warming impacts of TPS EOL scenarios. The landfill emissions for amorphous PLA and TPS are worse than for composting. Because degradation of these materials is inherent to the composting process, any improvements aimed at minimizing global warming impacts must focus on reducing the contributions from transportation and handling. Transportation and handling contribute to the global warming emissions for the composting scenario for PLA with 10.5% resulting from the waste collection, tipping floor processes, and transportation while 5.5% are due to operations of the compost facility. For TPS those values are 11.5% and 8.5% respectively.

Figure 23 presents the binary EOL options on a sliding scale from 100% landfill and 0% recycling/compost to 0% landfill and 100% recycling/compost. Figure 23 shows the difference in impacts from landfilling amorphous (A) and crystalline (B) PLA. This figure highlights the additional impacts resulting from the CH_4 emissions associated with the degradation of amorphous PLA in the landfill scenario. Additionally, Figure 23 also shows a traditionally recyclable plastic, PET (C), and the results for TPS (D), which incurs greater impacts when landfilled due to methane generation from anaerobic degradation like amorphous PLA. Figure 23 also shows the current recycling rate of PET in the United States and the potential impact reductions that could be achieved if the recycling rate increased.

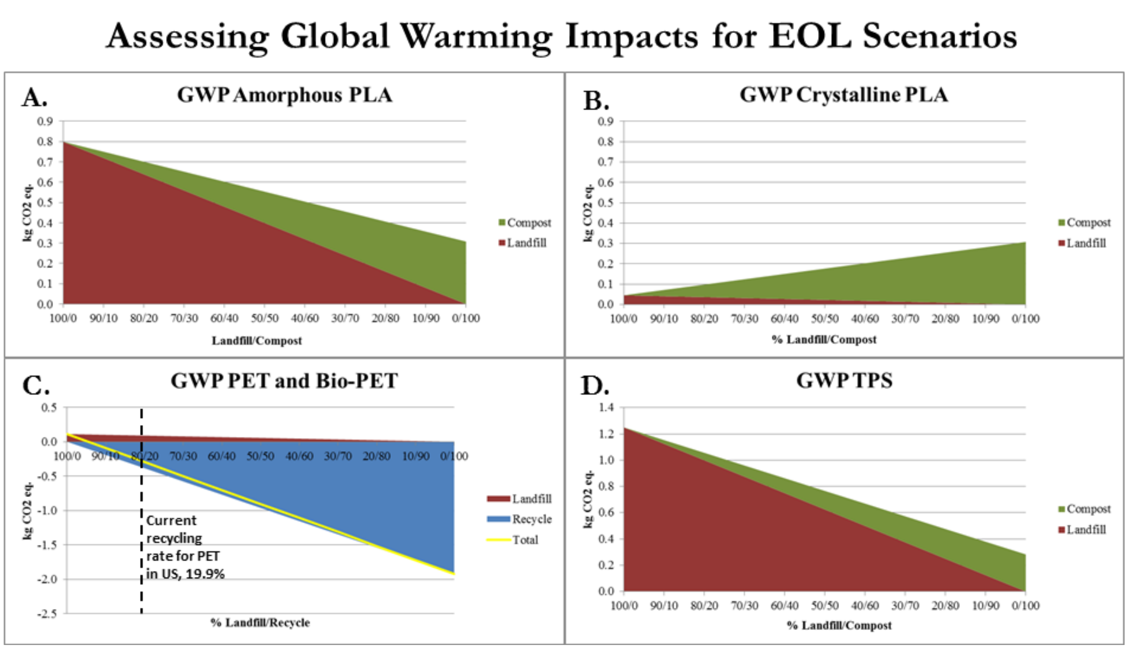

Figure 23. Global warming impacts for EOL scenarios scaled from 100% landfilling and 0% recycling/composting to 0% landfilling and 100% recycling/composting for amorphous (A) and crystalline (B) PLA, PET and Bio-PET (C), and TPS (D). The current recycling rate for PET is indicated (C). A zero waste strategy would be represented by reading the values at the far right of the graphs. For crystalline PLA (B) a zero waste approach would result in greater impacts. Results for all polymers and impact categories can be found in Appendix M.

The graphs in Figure 23 enable EOL decision makers to visualize the impacts associated with disposal and quantify their environmental goals. An organization with specific environmental goals could use these results to optimize their material selection and waste treatment strategy to achieve their objectives. The results suggest that a zero waste (i.e. landfill avoidance) strategy, as has been the approach for some institutions and municipalities, may not achieve the greatest environmental performance in all cases. Specifically; zero waste strategies should not be used for venues or organizations that have crystalline PLA in their waste streams; Figure 21 and Figure 22 show that landfilling crystalline PLA has lower impacts than composting in seven categories. However, the systemic impacts of the entire waste stream can also be important. If, for example, food waste to landfill could be avoided using crystalline PLA cutlery and a single stream composting collection at event venues and food service establishments, the benefits of creating an easily compostable waste stream to avoid methane generation from food waste may outweigh the crystalline PLA compost emissions.

Figure 24 displays the results for sensitivity analyses for global warming impacts (Appendix N presents the full results for all of the polymers and impact categories). For each set of variables (e.g. MRF technology, ethylene inventory, and local transportation) the minimum and maximum scenarios (ethylene scenarios are described in Methods section and Appendices G and H address inventories for MRFs and transportation) were tested against the scenario which was used to model the life-cycle impacts displayed in Figure 21. Bio-LDPE (A) and LDPE (B) are representative of the ratios for all of the bio-ethylene and fossil based plastics but the exact percent difference changes due to differing baseline impact values.

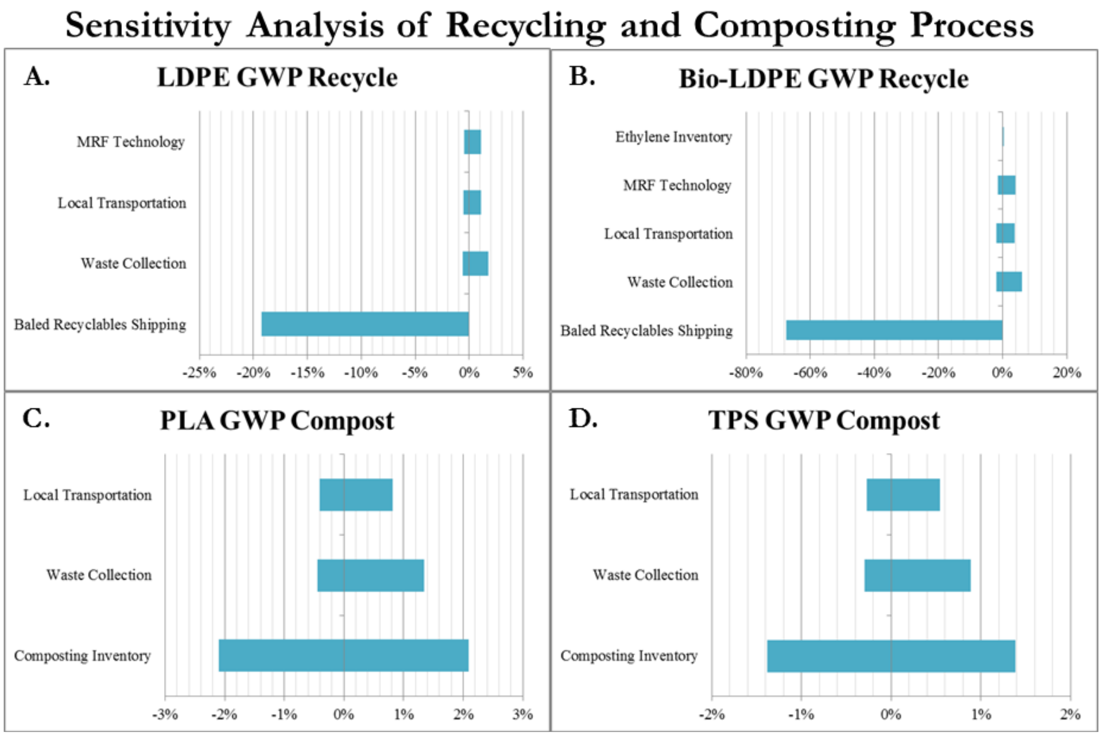

Figure 24. Sensitivity analysis of global warming impacts based on high and low ranges for variables used within the LCA. The 0% baselines are the findings for the recycled LDPE and Bio-LDPE as well as the composted PLA and TPS.

The Bio-LDPE the ethylene inventories, which were developed by Tabone et al. (2010) and revised for this study, were very similar and do not create significant sensitivities in the model. 'Baled Recyclables Shipping' was represented in the LCA by the high scenario, which is 610 km ground transport (Phoenix, AZ to Long Beach, CA) and 11,686 km ocean freighter transport (Long Beach to Hong Kong). The lower boundary represents not sending recyclable plastics to China (no international freight) but includes the same 610 km of ground transport. Since the LCA modeled shipping to China based on current real-world operations the sensitivity represented here demonstrates only negative values. The process 'Baled Recyclables Shipping' stood out as the most significant in terms of sensitivity for global warming impacts with shifts in the results as much as -63.5% for Bio-LDPE and -

18.9% for LDPE, suggesting a large share of the global warming emissions in the recycling scenarios are driven by international shipping (also shown in Figure 22).

This study demonstrates that production and EOL of biopolymers and fossil-based plastics must be accounted for when evaluating the life cycle environmental impacts of plastics. Environmental performance is not limited to purchasing decisions (i.e. production of plastics), and availability of disposal options can significantly influence the overall life-cycle impacts. Despite shipping or recyclables in the current market, recycling proves to have major benefits due to avoidance of creating new plastics from virgin resources. The benefits of recycling could extend to biopolymers if they can be recycled in existing waste streams. TPS and PLA may find additional life-cycle benefits if they can scale adequately to warrant recycling. The benefits of fossil based plastics are most notable in the global warming and fossil fuel depletion impact categories. Decision makers along the supply chain looking to improve environmental performance for packaging and disposable goods can use these results to better understand purchasing and waste handling in the context of real-world waste handling options. For example; zero waste, or landfill avoidance, waste management strategies do not always result in lower environmental impacts, as shown in Figure 24.

CHAPTER 7

CONCLUSION

For Objective 1) Quantify end of life of waste flows of biopolymers via waste audits at public venues and waste handling facilities, this research found that people are not coming into contact with or purchasing many biopolymer products on an individual consumer level. Furthermore, there are several management practices that can optimize different objectives such as climate emissions reduction, landfill avoidance, maximum financial benefit, or ease of management. The assessment of collection methods at venues suggests that there are effective ways to teach consumers, over time, how to sort materials and that simple signage may aid in reinforcement of behavior but is not necessarily effective as a standalone strategy.

For Objective 2) Identify best practices for facilities management and waste handling of compostables via surveys and focus groups, this research found that best practices for disposal of biopolymers may be regionally dependent and producers as well as consumers in this growing market need to be adaptable as systems adjust to achieve greater environmental outcomes for plastic materials. Increased communication along the life-cycle for compostable biopolymers can help stakeholders create a dialogue, clarifying their goals and expectations as they assume greater responsibility for the impacts of the products they use.

For Objective 3) Evaluate sustainable solutions for composting infrastructure options and factors influencing feasibility of scaling, this research suggests collection schemes should be considered based on the available treatment options and ease of management to reduce sorting time and contamination while minimizing environmental impacts. Further, compost facilities and biopolymer manufacturers need to work to find solutions, such as alkaline enhanced degradation to aid in the composting process for biopolymers.

For Objective 4) Quantify the end of life environmental impacts of compostable biopolymer scenarios via life-cycle assessment incorporating findings from objectives 1-3, this research demonstrated that recyclability remains as a highly effective method of reducing life-cycle impacts for plastics regardless the feedstock and is more widely understood in public collection schemes. For compostable biopolymers to be effective as a strategy to minimize environmental impacts during EOL the materials must avoid landfill if they are prone to anaerobic degradation. Zero-waste policies may not be resulting in optimal environmental performance and a better metric may be overall material performance including production and EOL.

Compostable biopolymers can find a niche in helping to optimize waste streams if there is a concerted effort between consumers, facilities, and waste handlers. By helping other organics avoid landfills biopolymers may have consequential impacts that can reduce the overall impacts of waste, however they do not provide any real advantages in systems where recycling is already efficient or where waste is sent to landfill by default. Recommendations for improving the overall performance of compostable biopolymers include revising standards for compostability to reflect real-world conditions, creating effective messaging in public collection areas, empowering consumers to make environmentally friendly decisions easily, optimizing biopolymer degradation in compost facilities by incorporating alkaline wastes from other industries, rather than using a zero waste policy organizations should focus on the life-cycle performances for products to optimize purchasing and disposal patterns, and, finally, use biopolymers which do not break down in landfills in durable goods or non-organic waste streams so the plastic will remain intact in landfills, reducing the potential for harmful emissions.

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APPENDIX A

EXAMPLES OF BIOPOLYMER LABELS ON CONSUMER PRODUCTS

Coca-Cola label image source: http://www.coca-colacompany.com/press-center/image-library/plantbottle-labels/

Nature House image source: http://www.savannahnaturehouse.com/

APPENDIX B

ANALYSIS OF THE COMPOST FROM WHICH THE INOCULANT WAS COLLECTED

REPORT NO.
F14066-6000
ACCOUNT NUMBER
30548

A & L GREAT LAKES LABORATORIES, INC.

3505 Conestoga Drive • Fort Wayne, IN 46808 • Phone 260-483-4759 • Fax 260-483-5274
www.algreatlakes.com • lab@algreatlakes.com

QUALITY ANALYSES FOR INFORMED DECISIONS

TO: GARICK LLC - TOPEKA, KS
2200 NW WATERWORKS WAY
TOPEKA, KS 66606-0000

FOR: GARICK MARICOPA FACILITY

ATTN: JIM BAMBRICK

COMPOST ANALYSIS REPORT

LAB NUMBER: 70633
SAMPLE ID: MARICOPA

DATE RECEIVED: 03/07/2014
DATE REPORTED: 03/31/2014 PAGE: 1

PARAMETER	UNIT	ANALYSIS RESULT	DRY BASIS RESULT	ANALYSIS METHOD
Moisture @ 70 C	%	32.14		TMECC 03.09-A
Dry Matter	%	67.86		TMECC 03.09-A
Total Nitrogen (N)	%	0.98	1.44	TMECC 04.02-D
Phosphorus (P)	%	0.13	0.19	TMECC 04.03-A
Phosphate (P205)	%	0.30	0.44	TMECC 04.03-A
Potassium (K)	%	0.39	0.57	TMECC 04.04-A
Potash (K2O)	%	0.46	0.68	TMECC 04.04-A
Magnesium (Mg)	%	0.01	0.02	TMECC 04.05-MG
Calcium (Ca)	%	1.70	2.50	TMECC 04.05-B
Arsenic	mg/kg	2.613	3.850	SW846-6020 04.06-As
Cadmium	mg/kg	<0.122	< 0.180	SW846-6020 04.06-Cd
Chromium	mg/kg	8.28	12.20	SW846-6020 04.06-Cr
Copper	mg/kg	42.21	62.20	SW846-6020 04.06-Cu

TMECC - Test Methods for the Examination of Composting and Compost, The U.S. Composting Council.

Report Approved By: Gregg T. Neyman
Greg Neyman - Vice President / COO

Approval Date: 3/31/2014

COMPOST

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DATE RECEIVED: 03/07/2014
DATE REPORTED: 03/31/2014 **PAGE:** 2

PARAMETER	UNIT	ANALYSIS RESULT	DRY BASIS RESULT	ANALYSIS METHOD
Mercury	mg/kg	<0.40	< 0.59	SW846-6020 04.06-Hg
Molybdenum	mg/kg	1.02	1.51	SW846-6020 04.06-Mo
Nickel	mg/kg	5.63	8.29	SW846-6020 04.06-Ni
Lead	mg/kg	11.88	17.50	SW846-6020 04.06-Pb
Selenium	mg/kg	<0.38	< 0.56	SW846-6020 04.06-Se
Zinc	mg/kg	49.1	72.3	SW846-6020 04.06-Zn
503 Metals PASS/FAIL	pass/fail		PASS	EPA 503 Metal Limits
pH	-	7.8		TMECC 04.11-A
Soluble Salts	dS/m	4.27		TMECC 04.10-A
Fecal Coliform/MPN	MPN/g dry		0	SM(20th)-9221E TMECC
Salmonella	MPN/4g dry		< 3	SM(20th)-9260D
Pathogen Reduction - PASS/FAIL	pass/fail		PASS	40 CFR 503 Class A
Ash @ 550 C	%	40.25	59.32	TMECC 03.02-B

TMECC - Test Methods for the Examination of Composting and Compost, The U.S. Composting Council.

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SAMPLE ID: MARICOPA

DATE RECEIVED: 03/07/2014
DATE REPORTED: 03/31/2014 **PAGE:** 3

PARAMETER	UNIT	ANALYSIS RESULT	DRY BASIS RESULT	ANALYSIS METHOD
Organic Matter by LOI @ 550C	%	27.61	40.68	TMECC 05.07-A
Organic Carbon by LOI @ 550C	%	13.80	20.34	Estimated
Carbon:Nitrogen Ratio (C:N)	-	14.1:1	14.1:1	TMECC 05.02-A
Foreign Material	%		0.28	TMECC 03.08-A
Germination - Emergence	%	97		TMECC 05.05-A
Germination - Vigor	%	72		TMECC 05.05-A
Ave Ht of Seedlings in Control	inches	3.4		TMECC 05.05-A
Ave Ht of Seedlings in Compost	inches	3.5		TMECC 05.05-A
Respiration - CO2-C/g TS	mg CO2-C / g TS/Day		1	TMECC 05.08-B
Respiration - CO2-C/g OM	mg CO2-C / g OM/Day		1	TMECC 05.08-B
Compost Stability Index	-		Very Stable	TMECC 05.08
Retained on U.S. 2-inch Sieve	%		0.00	TMECC 02.02-B
Retained on U.S. 1-inch Sieve	%		0.00	TMECC 02.02-B

TMECC - Test Methods for the Examination of Composting and Compost, The U.S. Composting Council.

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LAB NUMBER: 70633
SAMPLE ID: MARICOPA

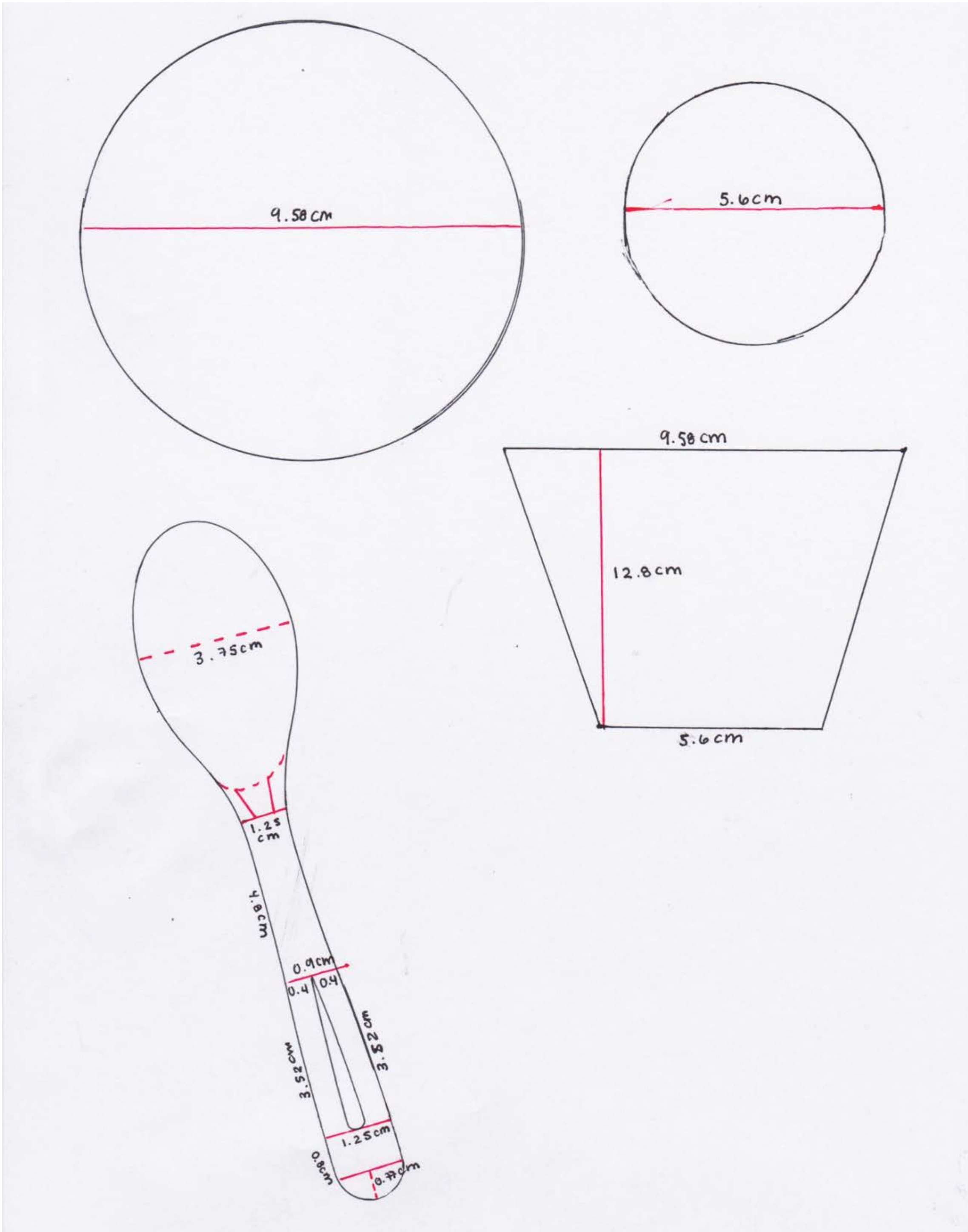
DATE RECEIVED: 03/07/2014
DATE REPORTED: 03/31/2014 **PAGE:** 4

PARAMETER	UNIT	ANALYSIS RESULT	DRY BASIS RESULT	ANALYSIS METHOD
Retained on U.S. 5/8-inch Sieve	%		0.00	TMECC 02.02-B
Retained on U.S. 3/8-inch Sieve	%		1.14	TMECC 02.02-B
Retained on U.S. 1/4-inch Sieve	%		5.76	TMECC 02.02-B
Retained on U.S. 5/32-inch	%		8.60	TMECC 02.02-B

TMECC - Test Methods for the Examination of Composting and Compost, The U.S. Composting Council.

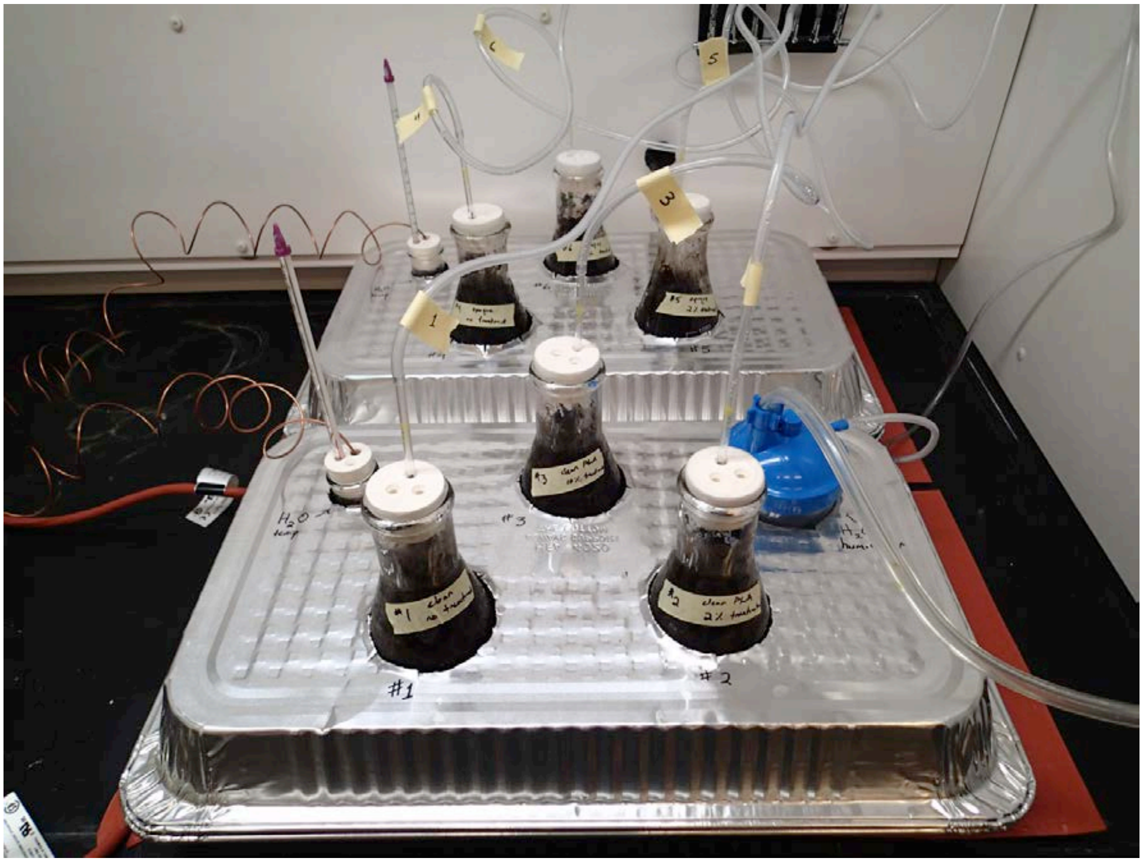
APPENDIX C

GEOMETRIC SHAPES USED FOR SURFACE AREA CALCULATIONS OF WORLD CENTRIC PLA SPOONS


9.58 cm
5.6cm
9.58 cm
12.8cm
5.6cm
3.75cm
1.25
cm
4.8cm
0.9cm
0.4
0.4
3.52cm
3.52 cm
1.25cm
0.8cm
0.77cm

APPENDIX D

PHOTOGRAPH OF BIOREACTOR SETUP



The large tube on the bottom right of the image is the incoming air going into the bubble humidifier (blue lid). The humidified air passes through a tub splitter to distribute the air to the six flasks evenly. The water baths are warmed by the thermostatically controlled pads (orange). The thermistors are immersed in DI water in the small flasks on the left of the water baths.

APPENDIX E

IMAGES OF DECOMPOSITION OF OPAQUE PLA/TALC BLEND

A. Opaque PLA before composting process

B. 0% opaque PLA after composting

C. 2% opaque PLA after composting

D. 10% opaque PLA after composting

Images taken with Olympus TG-2 on Super Macro mode.

APPENDIX F

INITIAL INTERVIEW QUESTIONS FOR BIOPOLYMER STAKEHOLDERS

Interview Questions for Recreational Concessions

Organization Name: Date:

Person Spoken To: Title:

Does your venue use compostable plastic service ware?

For how long have you had them available?

What types of compostables do you use (type of service ware and product name)?

Why did your company choose to offer these types of compostable products?

Does your company have sustainability goals they are trying to meet, and if so what are they?

What are the challenges, if any, associated with using these products?

How have your employees or customers reacted to these products?

Do you feel there is a general level of understanding in your staff about these products?

Have you had any specific feedback about them, from customers or staff?

Compared to conventional food service ware how much more expensive are these products?

How are they disposed of at your organization?

What kind of disposal options does your company have?

Does your company compost?

If so, is post-consumer compost included in your composting stream?

If you don't have compost and it became available to your company through as a city service would you participate?

What if they required post-sorting?

What if it cost more than land filling?

Do you train service staff on these products?

Do you train your janitorial staff on how to handle these products?

APPENDIX G

INVENTORY DATA FOR MATERIAL RECOVERY FACILITY SCENARIOS

		Single-stream MRF Inventory Values		
Process	**Unit**	**PET/Bio-PET**	**HDPE/Bio-HDPE**	**Film Wastes (LDPE/Bio-LDPE)**
Bailing Wire	kg/Mg	0.6	0.6	0.6
Diesel Fuel	MJ/Mg	25	25	25
Energy Use	kWh/Mg	9.9	32.7	3.0

		Mixed-waste MRF Inventory Value		
Process	**Unit**	**PET/Bio-PET**	**HDPE/Bio-HDPE**	**Film Wastes (LDPE/Bio-LDPE)**
Bailing Wire	kg/Mg	0.3	0.3	0.3
Diesel Fuel	MJ/Mg	25	25	25
Energy Use	kWh/Mg	36.9	116.1	4.7

		Dual-stream MRF Inventory Value		
Process	**Unit**	**PET/Bio-PET**	**HDPE/Bio-HDPE**	**Film Wastes (LDPE/Bio-LDPE)**
Bailing Wire	kg/Mg	0.6	0.6	0.6
Diesel Fuel	MJ/Mg	25	25	25
Energy Use	kWh/Mg	6.9	22.0	2.1

		Pre-sorted MRF Inventory Value		
Process	**Unit**	**PET/Bio-PET**	**HDPE/Bio-HDPE**	**Film Wastes (LDPE/Bio-LDPE)**
Bailing Wire	kg/Mg	0.7	0.7	0.7
Diesel Fuel	MJ/Mg	25	25	25
Energy Use	kWh/Mg	3.1	3.1	3.1

Adapted from Pressley et al. (2015)

APPENDIX H

BASELINE TRANSPORATION VALUES AND SCENARIOS FOR SENSITIVITY ANALYSIS

Transportation scenarios (km)	baseline	min	max
Waste Collection to Tipping Floor (TF) =	15	10	30
TF to Landfill/MRF/Composting =	60	30	120
From MRF to Port =	610	610	610
Port to Port (Recycling Facility) =	11686.12	0	11686.12

APPENDIX I

PLASTIC PRODUCTION – PER KILOGRAM OF PLASTIC MIDPOINT IMPACT FACTORS CALCULATED USING TRACI

Plastic Production - Per Kilogram of Plastic Midpoint Impact Factors Calculated Using TRACI		HDPE (Franklin Associates 2011) with Bioethylene (Tabone 2010)					LDPE (Franklin Associates 2011) with Bioethylene (Tabone 2010)				
Impact category	**Unit**	**Bio-HDPE - Ethylene Scenario 1**	**Bio-HDPE - Ethylene Scenario 2**	**Bio-HDPE - Ethylene Scenario 3**	**Bio-HDPE - Ethylene Scenario 4**	**Bio-HDPE Average**	**Bio-LDPE - Ethylene Scenario 1**	**Bio-LDPE - Ethylene Scenario 2**	**Bio-LDPE - Ethylene Scenario 3**	**Bio-LDPE - Ethylene Scenario 4**	**Bio-LDPE Average**
Ozone depletion	kg CFC-11 eq	5.94E-08	5.94E-08	5.94E-08	5.94E-08	**5.94E-08**	1.10E-07	1.10E-07	1.10E-07	1.10E-07	**1.10E-07**
Global warming	kg CO2 eq	1.12E+00	1.12E+00	1.12E+00	1.12E+00	**1.12E+00**	1.41E+00	1.41E+00	1.41E+00	1.41E+00	**1.41E+00**
Smog	kg O3 eq	2.45E-01	2.45E-01	2.45E-01	2.45E-01	**2.45E-01**	2.50E-01	2.50E-01	2.50E-01	2.50E-01	**2.50E-01**
Acidification	kg SO2 eq	1.38E-02	1.38E-02	1.38E-02	1.38E-02	**1.38E-02**	1.42E-02	1.42E-02	1.42E-02	1.42E-02	**1.42E-02**
Eutrophication	kg N eq	1.89E-03	1.89E-03	1.89E-03	1.89E-03	**1.89E-03**	1.92E-03	1.92E-03	1.92E-03	1.92E-03	**1.92E-03**
Carcinogenics	CTUh	1.53E-08	1.60E-08	1.53E-08	1.60E-08	**1.57E-08**	1.57E-08	1.64E-08	1.57E-08	1.64E-08	**1.61E-08**
Non carcinogenics	CTUh	7.09E-07	7.11E-07	7.09E-07	7.12E-07	**7.10E-07**	7.24E-07	7.26E-07	7.24E-07	7.26E-07	**7.25E-07**
Respiratory effects	kg PM2.5 eq	1.69E-03	1.69E-03	1.69E-03	1.69E-03	**1.69E-03**	1.73E-03	1.73E-03	1.73E-03	1.73E-03	**1.73E-03**
Ecotoxicity	CTUe	6.69E+00	7.52E+00	6.69E+00	7.52E+00	**7.11E+00**	6.85E+00	7.69E+00	6.85E+00	7.69E+00	**7.27E+00**
Fossil fuel depletion	MJ surplus	3.00E+00	3.00E+00	3.00E+00	3.00E+00	**3.00E+00**	3.68E+00	3.68E+00	3.68E+00	3.68E+00	**3.68E+00**

Plastic Production - Per Kilogram of Plastic Midpoint Impact Factors Calculated Using TRACI		PET (Franklin Associates 2011) with Bioethylene (Tabone 2010)									
Impact category	**Unit**	**Bio-PET - Ethylene Scenario 1**	**Bio-PET - Ethylene Scenario 2**	**Bio-PET - Ethylene Scenario 3**	**Bio-PET - Ethylene Scenario 4**	**Bio-PET Average**	HDPE (Franklin Associates 2011)	LDPE (Franklin Associates 2011)	PET (Franklin Associates 2011)	PLA (Vink 2010)	TPS (Franklin Associates 2011)
Ozone depletion	kg CFC-11 eq	3.03E-08	3.03E-08	3.03E-08	3.03E-08	**3.03E-08**	6.20E-09	5.62E-08	2.09E-08	0.00E+00	1.46E-07
Global warming	kg CO2 eq	2.62E+00	2.62E+00	2.62E+00	2.62E+00	**2.62E+00**	1.90E+00	2.20E+00	2.73E+00	1.15E+00	1.93E+00
Smog	kg O3 eq	2.82E-01	2.82E-01	2.82E-01	2.82E-01	**2.82E-01**	6.44E-02	6.83E-02	1.45E-01	2.08E-01	8.61E-02
Acidification	kg SO2 eq	1.58E-02	1.58E-02	1.58E-02	1.58E-02	**1.58E-02**	6.12E-03	6.46E-03	1.15E-02	1.64E-02	1.59E-02
Eutrophication	kg N eq	1.14E-03	1.14E-03	1.14E-03	1.14E-03	**1.14E-03**	1.22E-04	1.34E-04	4.85E-04	1.30E-03	2.67E-03
Carcinogenics	CTUh	4.39E-08	4.40E-08	4.39E-08	4.40E-08	**4.39E-08**	5.01E-09	5.29E-09	3.84E-08	1.75E-11	4.47E-09
Non carcinogenics	CTUh	2.67E-07	2.67E-07	2.67E-07	2.67E-07	**2.67E-07**	5.42E-08	5.76E-08	8.62E-08	1.93E-10	5.18E-08
Respiratory effects	kg PM2.5 eq	9.15E-04	9.15E-04	9.15E-04	9.15E-04	**9.15E-04**	4.09E-04	4.29E-04	7.54E-04	5.03E-04	1.12E-03
Ecotoxicity	CTUe	3.25E+00	3.45E+00	3.25E+00	3.45E+00	**3.35E+00**	8.74E-01	9.31E-01	1.40E+00	3.40E-04	1.98E+00
Fossil fuel depletion	MJ surplus	7.14E+00	7.14E+00	7.14E+00	7.14E+00	**7.14E+00**	1.12E+01	1.20E+01	9.05E+00	3.91E+00	1.94E-01

APPENDIX J

WASTE TREATMENT – PER KILOGRAM OF PLASTIC MIDPOINT IMPACT FACTORS CALCULATED USING TRACI

Waste Treatment - Per Kilogram of Plastic Midpoint Impact Factors Calculated Using TRACI		HDPE, LDPE, Bio-HDPE, Bio-LDPE	HDPE, LDPE, Bio-HDPE, Bio-LDPE	PET, Bio-PET	PET, Bio-PET	PLA				TPS			All Polymer Types
Impact category	**Unit**	**Recycling**	**Landfill**	**Recycling**	**Landfill**	**Composting (ecoinvent v3)**	**Composting (custom)**	**Amorphous, Landfilled (Kolstad 2012)**	**Crystalline, Landfilled (Kolstad 2012)**	**Composting (ecoinvent v3)**	**Composting (custom)**	**Landfill, Cardboard Proxy (ecoinvent v2)**	**Tipping Floor (custom)**
Ozone depletion	kg CFC-11 eq	2.48E-08	4.11E-09	1.01E-08	4.12E-09	3.74E-09	3.08E-10	4.12E-09	4.12E-09	3.08E-10	3.74E-09	4.39E-09	5.05E-12
Global warming	kg CO2 eq	-1.47E+00	1.13E-01	-2.31E+00	8.01E-02	3.06E-01	2.45E-01	7.67E-01	1.30E-02	2.20E-01	2.81E-01	1.22E+00	9.01E-04
Smog	kg O3 eq	-4.05E-02	2.50E-03	-1.22E-01	2.53E-03	1.15E-02	5.35E-03	4.54E-03	2.53E-03	5.35E-03	1.15E-02	3.06E-03	2.66E-04
Acidification	kg SO2 eq	-3.52E-03	8.92E-05	-8.90E-03	9.00E-05	1.93E-03	2.04E-04	1.48E-04	9.00E-05	2.04E-04	1.93E-03	2.63E-04	9.04E-06
Eutrophication	kg N eq	9.95E-05	1.46E-02	-2.63E-04	1.17E-02	1.36E-04	2.77E-05	1.17E-02	1.17E-02	2.77E-05	1.36E-04	7.13E-03	7.17E-07
Carcinogenics	CTUh	-4.55E-09	1.32E-09	-3.79E-08	9.87E-10	3.17E-09	1.08E-09	9.87E-10	9.87E-10	1.08E-09	3.17E-09	1.69E-09	3.26E-11
Non carcinogenics	CTUh	-3.86E-08	2.87E-08	-7.06E-08	1.41E-08	7.68E-09	4.04E-09	8.20E-09	8.20E-09	4.04E-09	7.68E-09	1.64E-08	1.49E-10
Respiratory effects	kg PM2.5 eq	8.07E-05	1.04E-05	-2.64E-04	1.03E-05	5.95E-05	2.62E-05	1.22E-05	1.03E-05	2.62E-05	5.95E-05	2.27E-05	1.10E-06
Ecotoxicity	CTUe	-8.22E-01	6.04E-01	-1.35E+00	3.17E+00	2.07E-01	1.15E-01	1.12E-01	1.12E-01	1.15E-01	2.07E-01	1.34E-01	3.07E-03
Fossil fuel depletion	MJ surplus	-1.09E+01	2.38E-05	-8.72E+00	2.46E-05	7.13E-02	3.60E-02	2.46E-05	2.46E-05	3.60E-02	7.13E-02	4.08E-05	1.73E-03

Waste Treatment - Per Kilogram of Plastic Midpoint Impact Factors Calculated Using TRACI		LDPE, Bio-LDPE				HDPE/Bio-HDPE				PET/Bio-PET			
Impact category	**Unit**	**Recycle Dual Stream MRF**	**Recycle Mixed Waste MRF**	**Recycle Pre-Sorted MRF**	**Recycle Single Stream MRF**	**Recycle Dual Stream MRF**	**Recycle Mixed Waste MRF**	**Recycle Pre-Sorted MRF**	**Recycle Single Stream MRF**	**Recycle Dual Stream MRF**	**Recycle Mixed Waste MRF**	**Recycle Pre-Sorted MRF**	**Recycle Single Stream MRF**
Ozone depletion	kg CFC-11 eq	3.30E-10	4.59E-10	3.86E-10	3.78E-10	1.38E-09	6.31E-09	3.86E-10	1.94E-09	5.82E-10	2.15E-09	3.86E-10	7.40E-10
Global warming	kg CO2 eq	4.18E-03	5.92E-03	4.95E-03	4.83E-03	1.86E-02	8.67E-02	4.95E-03	2.64E-02	7.66E-03	2.93E-02	4.95E-03	9.83E-03
Smog	kg O3 eq	7.70E-04	8.72E-04	8.13E-04	8.07E-04	1.59E-03	5.45E-03	8.13E-04	2.03E-03	9.67E-04	2.20E-03	8.13E-04	1.09E-03
Acidification	kg SO2 eq	3.18E-05	4.29E-05	3.64E-05	3.58E-05	1.20E-04	5.37E-04	3.64E-05	1.68E-04	5.31E-05	1.86E-04	3.64E-05	6.64E-05
Eutrophication	kg N eq	8.42E-06	1.50E-05	1.12E-05	1.08E-05	6.18E-05	3.14E-04	1.12E-05	9.05E-05	2.13E-05	1.01E-04	1.12E-05	2.93E-05
Carcinogenics	CTUh	3.43E-10	3.06E-10	4.09E-10	3.69E-10	9.22E-10	3.55E-09	4.09E-10	1.23E-09	4.82E-10	1.24E-09	4.09E-10	5.70E-10
Non carcinogenics	CTUh	4.22E-10	7.28E-10	5.59E-10	5.37E-10	2.97E-09	1.50E-08	5.59E-10	4.34E-09	1.04E-09	4.85E-09	5.59E-10	1.42E-09
Respiratory effects	kg PM2.5 eq	4.90E-06	6.98E-06	5.76E-06	5.65E-06	2.15E-05	9.99E-05	5.76E-06	3.04E-05	8.90E-06	3.38E-05	5.76E-06	1.14E-05
Ecotoxicity	CTUe	1.22E-02	1.80E-02	1.59E-02	1.49E-02	7.14E-02	3.49E-01	1.59E-02	1.03E-01	2.65E-02	1.14E-01	1.59E-02	3.54E-02
Fossil fuel depletion	MJ surplus	5.99E-03	7.37E-03	6.57E-03	6.49E-03	1.71E-02	6.96E-02	6.57E-03	2.31E-02	8.67E-03	2.54E-02	6.57E-03	1.03E-02

APPENDIX K

TRANSPORTATION – PER TONNE-KILOMETER MIDPOINT IMPACT FACTORS CALCULATED USING TRACI

Transportation - Per Tonne-Kilometer Midpoint Impact Factors Calculated Using TRACI				
Impact category	**Unit**	**Municipal Waste Collection**	**Transport, Ocean Freighter**	**Transport, Single Unit Truck**
Ozone depletion	kg CFC-11 eq	1.13E-07	6.91E-13	7.58E-12
Global warming	kg CO2 eq	1.30E+00	1.83E-02	1.99E-01
Smog	kg O3 eq	2.05E-01	1.11E-02	3.45E-02
Acidification	kg SO2 eq	6.95E-03	3.79E-04	1.24E-03
Eutrophication	kg N eq	6.63E-04	2.05E-05	6.96E-05
Carcinogenics	CTUh	9.53E-09	2.48E-10	2.72E-09
Non carcinogenics	CTUh	9.19E-08	2.39E-09	2.62E-08
Respiratory effects	kg PM2.5 eq	8.50E-04	6.28E-06	2.21E-05
Ecotoxicity	CTUe	1.01E+00	4.62E-02	5.07E-01
Fossil fuel depletion	MJ surplus	2.59E+00	3.26E-02	3.58E-01

APPENDIX L

SUMMARIZED LIFE-CYCLE IMPACT RESULTS FOR FULL SYSTEM BOUNDARY AND EOL ONLY

Impact category	Unit	Bio-HDPE Landfilled	Bio-HDPE Recycled (average)	Bio-HDPE EOL ONLY Landfilled	Bio-HDPE EOL ONLY Recycled (average)	HDPE Landfilled	HDPE Recycled (average)	HDPE EOL ONLY Landfilled	HDPE EOL ONLY Recycled (average)	Bio-LDPE Landfilled	Bio-LDPE Recycled (average)	Bio-LDPE EOL ONLY Landfilled	Bio-LDPE EOL ONLY Recycled (average)	LDPE Landfilled	LDPE Recycled (average)	LDPE EOL ONLY Landfilled	LDPE EOL ONLY Recycled (average)
Ozone depletion	kg CFC-11 eq	6.53E-08	8.85E-08	5.80E-09	2.90E-08	1.20E-08	3.52E-08	5.80E-09	2.90E-08	1.16E-07	1.37E-07	5.80E-09	2.72E-08	6.20E-08	8.31E-08	5.80E-09	2.72E-08
Global warming	kg CO2 eq	1.27E+00	5.46E-02	1.45E-01	-1.07E+00	2.04E+00	8.30E-01	1.45E-01	-1.07E+00	1.55E+00	3.14E-01	1.45E-01	-1.09E+00	2.35E+00	1.11E+00	1.45E-01	-1.09E+00
Smog	kg O3 eq	2.53E-01	3.63E-01	7.91E-03	1.18E-01	7.23E-02	1.83E-01	7.91E-03	1.18E-01	2.58E-01	3.66E-01	7.91E-03	1.17E-01	7.62E-02	1.85E-01	7.91E-03	1.17E-01
Acidification	kg SO2 eq	1.41E-02	1.59E-02	2.77E-04	2.07E-03	6.40E-03	8.19E-03	2.77E-04	2.07E-03	1.45E-02	1.61E-02	2.77E-04	1.92E-03	6.74E-03	8.36E-03	2.77E-04	1.92E-03
Eutrophication	kg N eq	1.65E-02	2.40E-03	1.46E-02	5.16E-04	1.47E-02	6.38E-04	1.46E-02	5.16E-04	1.65E-02	2.33E-03	1.46E-02	4.23E-04	1.47E-02	5.42E-04	1.46E-02	4.23E-04
Carcinogenics	CTUh	1.73E-08	1.75E-08	1.66E-09	1.88E-09	6.67E-09	6.89E-09	1.66E-09	1.88E-09	1.77E-08	1.68E-08	1.66E-09	8.74E-10	6.95E-09	6.00E-09	1.66E-09	8.74E-10
Non carcinogenics	CTUh	7.42E-07	7.25E-07	3.18E-08	1.41E-08	8.60E-08	6.83E-08	3.18E-08	1.41E-08	7.57E-07	7.34E-07	3.18E-08	9.70E-09	8.94E-08	6.66E-08	3.18E-08	9.70E-09
Respiratory effects	kg PM2.5 eq	1.72E-03	1.91E-03	2.55E-05	2.22E-04	4.35E-04	6.31E-04	2.55E-05	2.22E-04	1.76E-03	1.92E-03	2.55E-05	1.93E-04	4.54E-04	6.17E-04	2.55E-05	1.93E-04
Ecotoxicity	CTUe	7.76E+00	7.32E+00	6.53E-01	2.10E-01	1.53E+00	1.08E+00	6.53E-01	2.10E-01	7.92E+00	7.36E+00	6.53E-01	1.08E-01	1.58E+00	1.02E+00	6.53E-01	1.08E-01
Fossil fuel depletion	MJ surplus	3.06E+00	-7.17E+00	6.21E-02	-1.02E+01	1.12E+01	1.02E+00	6.21E-02	-1.02E+01	3.74E+00	-6.51E+00	6.21E-02	-1.02E+01	1.21E+01	1.84E+00	6.21E-02	-1.02E+01

Impact category	Unit	Bio-PET Landfilled	Bio-PET Recycled (average)	Bio-PET EOL ONLY Landfilled	Bio-PET EOL ONLY Recycled (average)	PET Landfilled	PET Recycled (average)	PET EOL ONLY Landfilled	PET EOL ONLY Recycled (average)	PLA Landfilled (amorph-ous)	PLA Landfilled (crysta-lline)	PLA Compost (average)	PLA EOL ONLY Landfilled (average)	PLA EOL ONLY Compost (average)	TPS Landfilled	TPS Compost (average)	TPS EOL ONLY Landfilled	TPS EOL ONLY Compost (average)
Ozone depletion	kg CFC-11 eq	3.61E-08	4.31E-08	5.82E-09	1.28E-08	2.67E-08	3.37E-08	5.82E-09	1.28E-08	5.82E-09	5.82E-09	3.72E-09	5.82E-09	3.72E-09	1.52E-07	1.49E-07	6.08E-09	3.72E-09
Global warming	kg CO2 eq	2.73E+00	6.93E-01	1.13E-01	-1.92E+00	2.85E+00	8.09E-01	1.13E-01	-1.92E+00	1.95E+00	1.20E+00	1.46E+00	4.22E-01	3.08E-01	3.18E+00	2.21E+00	1.25E+00	2.83E-01
Smog	kg O3 eq	2.90E-01	3.18E-01	7.94E-03	3.59E-02	1.53E-01	1.81E-01	7.94E-03	3.59E-02	2.18E-01	2.16E-01	2.22E-01	8.95E-03	1.38E-02	9.46E-02	1.00E-01	8.47E-03	1.38E-02
Acidification	kg SO2 eq	1.61E-02	1.24E-02	2.78E-04	-3.44E-03	1.18E-02	8.06E-03	2.78E-04	-3.44E-03	1.67E-02	1.66E-02	1.76E-02	3.07E-04	1.25E-03	1.63E-02	1.71E-02	4.51E-04	1.25E-03
Eutrophication	kg N eq	1.29E-02	1.22E-03	1.18E-02	7.46E-05	1.22E-02	5.60E-04	1.18E-02	7.46E-05	1.31E-02	1.31E-02	1.40E-03	1.18E-02	9.68E-05	9.82E-03	2.77E-03	7.14E-03	9.68E-05
Carcinogenics	CTUh	4.53E-08	1.16E-08	1.33E-09	-3.24E-08	3.97E-08	6.03E-09	1.33E-09	-3.24E-08	1.34E-09	1.34E-09	2.48E-09	1.33E-09	2.46E-09	6.50E-09	6.94E-09	2.03E-09	2.46E-09
Non carcinogenics	CTUh	2.84E-07	2.45E-07	1.72E-08	-2.16E-08	1.03E-07	6.46E-08	1.72E-08	-2.16E-08	1.15E-08	1.15E-08	9.15E-09	1.13E-08	8.96E-09	7.13E-08	6.08E-08	1.95E-08	8.96E-09
Respiratory effects	kg PM2.5 eq	9.41E-04	7.69E-04	2.55E-05	-1.47E-04	7.79E-04	6.07E-04	2.55E-05	-1.47E-04	5.31E-04	5.29E-04	5.61E-04	2.64E-05	5.80E-05	1.16E-03	1.18E-03	3.78E-05	5.80E-05
Ecotoxicity	CTUe	6.57E+00	2.95E+00	3.22E+00	-4.03E-01	4.62E+00	9.98E-01	3.22E+00	-4.03E-01	1.61E-01	1.61E-01	2.10E-01	1.61E-01	2.10E-01	2.16E+00	2.18E+00	1.82E-01	2.10E-01
Fossil fuel depletion	MJ surplus	7.20E+00	-9.03E-01	6.21E-02	-8.04E+00	9.11E+00	1.00E+00	6.21E-02	-8.04E+00	3.97E+00	3.97E+00	4.03E+00	6.21E-02	1.16E-01	2.57E-01	3.10E-01	6.21E-02	1.16E-01

APPENDIX M

EOL IMPACTS FOR WASTE TREATMENT OPTIONS AT DIFFERENT RATIOS FOR SIX POLYMERS INCLUDING AMORPHOUS AND CRYSTALLINE PLA

PET	Units	Landfill/ Recycle	100% landfill 0% recycle	90% landfill 10% recycle	80% landfill 20% recycle	70% landfill 30% recycle	60% landfill 40% recycle	50% landfill 50% recycle	40% landfill 60% recycle	30% landfill 70% recycle	20% landfill 80% recycle	10% landfill 90% recycle	0% landfill 100% recycle
Ozone depletion	kg CFC-11 eq	Landfill	5.8E-09	5.2E-09	4.7E-09	4.1E-09	3.5E-09	2.9E-09	2.3E-09	1.7E-09	1.2E-09	5.8E-10	0.0E+00
	kg CFC-11 eq	Recycle	0.0E+00	1.3E-09	2.6E-09	3.8E-09	5.1E-09	6.4E-09	7.7E-09	9.0E-09	1.0E-08	1.2E-08	1.3E-08
	kg CFC-11 eq	Total	5.8E-09	6.5E-09	7.2E-09	7.9E-09	8.6E-09	9.3E-09	1.0E-08	1.1E-08	1.1E-08	1.2E-08	1.3E-08
Global warming	kg CO2 eq	Landfill	1.1E-01	1.0E-01	9.0E-02	7.9E-02	6.8E-02	5.6E-02	4.5E-02	3.4E-02	2.3E-02	1.1E-02	0.0E+00
	kg CO2 eq	Recycle	0.0E+00	-1.9E-01	-3.8E-01	-5.8E-01	-7.7E-01	-9.6E-01	-1.2E+00	-1.3E+00	-1.5E+00	-1.7E+00	-1.9E+00
	kg CO2 eq	Total	1.1E-01	-9.1E-02	-2.9E-01	-5.0E-01	-7.0E-01	-9.1E-01	-1.1E+00	-1.3E+00	-1.5E+00	-1.7E+00	-1.9E+00
Smog	kg O3 eq	Landfill	7.9E-03	7.1E-03	6.4E-03	5.6E-03	4.8E-03	4.0E-03	3.2E-03	2.4E-03	1.6E-03	7.9E-04	0.0E+00
	kg O3 eq	Recycle	0.0E+00	3.6E-03	7.2E-03	1.1E-02	1.4E-02	1.8E-02	2.2E-02	2.5E-02	2.9E-02	3.2E-02	3.6E-02
	kg O3 eq	Total	7.9E-03	1.1E-02	1.4E-02	1.6E-02	1.9E-02	2.2E-02	2.5E-02	2.8E-02	3.0E-02	3.3E-02	3.6E-02
Acidification	kg SO2 eq	Landfill	2.8E-04	2.5E-04	2.2E-04	1.9E-04	1.7E-04	1.4E-04	1.1E-04	8.3E-05	5.6E-05	2.8E-05	0.0E+00
	kg SO2 eq	Recycle	0.0E+00	-3.4E-04	-6.9E-04	-1.0E-03	-1.4E-03	-1.7E-03	-2.1E-03	-2.4E-03	-2.8E-03	-3.1E-03	-3.4E-03
	kg SO2 eq	Total	2.8E-04	-9.4E-05	-4.7E-04	-8.4E-04	-1.2E-03	-1.6E-03	-2.0E-03	-2.3E-03	-2.7E-03	-3.1E-03	-3.4E-03
Eutrophication	kg N eq	Landfill	1.2E-02	1.1E-02	9.4E-03	8.2E-03	7.1E-03	5.9E-03	4.7E-03	3.5E-03	2.4E-03	1.2E-03	0.0E+00
	kg N eq	Recycle	0.0E+00	7.5E-06	1.5E-05	2.2E-05	3.0E-05	3.7E-05	4.5E-05	5.2E-05	6.0E-05	6.7E-05	7.5E-05
	kg N eq	Total	1.2E-02	1.1E-02	9.4E-03	8.3E-03	7.1E-03	5.9E-03	4.7E-03	3.6E-03	2.4E-03	1.2E-03	7.5E-05
Carcinogenics	CTUh	Landfill	1.3E-09	1.2E-09	1.1E-09	9.3E-10	8.0E-10	6.6E-10	5.3E-10	4.0E-10	2.7E-10	1.3E-10	0.0E+00
	CTUh	Recycle	0.0E+00	-3.2E-09	-6.5E-09	-9.7E-09	-1.3E-08	-1.6E-08	-1.9E-08	-2.3E-08	-2.6E-08	-2.9E-08	-3.2E-08
	CTUh	Total	1.3E-09	-2.0E-09	-5.4E-09	-8.8E-09	-1.2E-08	-1.6E-08	-1.9E-08	-2.2E-08	-2.6E-08	-2.9E-08	-3.2E-08
Non carcinogenics	CTUh	Landfill	1.7E-08	1.5E-08	1.4E-08	1.2E-08	1.0E-08	8.6E-09	6.9E-09	5.2E-09	3.4E-09	1.7E-09	0.0E+00
	CTUh	Recycle	0.0E+00	-2.2E-09	-4.3E-09	-6.5E-09	-8.6E-09	-1.1E-08	-1.3E-08	-1.5E-08	-1.7E-08	-1.9E-08	-2.2E-08
	CTUh	Total	1.7E-08	1.3E-08	9.4E-09	5.5E-09	1.7E-09	-2.2E-09	-6.1E-09	-1.0E-08	-1.4E-08	-1.8E-08	-2.2E-08
Respiratory effects	kg PM2.5 eq	Landfill	2.5E-05	2.3E-05	2.0E-05	1.8E-05	1.5E-05	1.3E-05	1.0E-05	7.6E-06	5.1E-06	2.5E-06	0.0E+00
	kg PM2.5 eq	Recycle	0.0E+00	-1.5E-05	-2.9E-05	-4.4E-05	-5.9E-05	-7.3E-05	-8.8E-05	-1.0E-04	-1.2E-04	-1.3E-04	-1.5E-04
	kg PM2.5 eq	Total	2.5E-05	8.2E-06	-9.0E-06	-2.6E-05	-4.3E-05	-6.1E-05	-7.8E-05	-9.5E-05	-1.1E-04	-1.3E-04	-1.5E-04
Ecotoxicity	CTUe	Landfill	3.2E+00	2.9E+00	2.6E+00	2.3E+00	1.9E+00	1.6E+00	1.3E+00	9.7E-01	6.4E-01	3.2E-01	0.0E+00
	CTUe	Recycle	0.0E+00	-4.0E-02	-8.1E-02	-1.2E-01	-1.6E-01	-2.0E-01	-2.4E-01	-2.8E-01	-3.2E-01	-3.6E-01	-4.0E-01
	CTUe	Total	3.2E+00	2.9E+00	2.5E+00	2.1E+00	1.8E+00	1.4E+00	1.0E+00	6.8E-01	3.2E-01	-4.0E-02	-4.0E-01
Fossil fuel depletion	MJ surplus	Landfill	6.2E-02	5.6E-02	5.0E-02	4.3E-02	3.7E-02	3.1E-02	2.5E-02	1.9E-02	1.2E-02	6.2E-03	0.0E+00
	MJ surplus	Recycle	0.0E+00	-8.0E-01	-1.6E+00	-2.4E+00	-3.2E+00	-4.0E+00	-4.8E+00	-5.6E+00	-6.4E+00	-7.2E+00	-8.0E+00
	MJ surplus	Total	6.2E-02	-7.5E-01	-1.6E+00	-2.4E+00	-3.2E+00	-4.0E+00	-4.8E+00	-5.6E+00	-6.4E+00	-7.2E+00	-8.0E+00

HDPE	Units	Landfill/ Recycle	100% landfill 0% recycle	90% landfill 10% recycle	80% landfill 20% recycle	70% landfill 30% recycle	60% landfill 40% recycle	50% landfill 50% recycle	40% landfill 60% recycle	30% landfill 70% recycle	20% landfill 80% recycle	10% landfill 90% recycle	0% landfill 100% recycle
Ozone depletion	kg CFC-11 eq	Landfill	5.8E-09	5.2E-09	4.6E-09	4.1E-09	3.5E-09	2.9E-09	2.3E-09	1.7E-09	1.2E-09	5.8E-10	0.0E+00
	kg CFC-11 eq	Recycle	0.0E+00	2.9E-09	5.8E-09	8.7E-09	1.2E-08	1.5E-08	1.7E-08	2.0E-08	2.3E-08	2.6E-08	2.9E-08
	kg CFC-11 eq	Total	5.8E-09	8.1E-09	1.0E-08	1.3E-08	1.5E-08	1.7E-08	2.0E-08	2.2E-08	2.4E-08	2.7E-08	2.9E-08
Global warming	kg CO2 eq	Landfill	1.5E-01	1.3E-01	1.2E-01	1.0E-01	8.7E-02	7.3E-02	5.8E-02	4.4E-02	2.9E-02	1.5E-02	0.0E+00
	kg CO2 eq	Recycle	0.0E+00	-1.1E-01	-2.1E-01	-3.2E-01	-4.3E-01	-5.3E-01	-6.4E-01	-7.5E-01	-8.5E-01	-9.6E-01	-1.1E+00
	kg CO2 eq	Total	1.5E-01	2.4E-02	-9.7E-02	-2.2E-01	-3.4E-01	-4.6E-01	-5.8E-01	-7.0E-01	-8.2E-01	-9.4E-01	-1.1E+00
Smog	kg O3 eq	Landfill	7.9E-03	7.1E-03	6.3E-03	5.5E-03	4.7E-03	4.0E-03	3.2E-03	2.4E-03	1.6E-03	7.9E-04	0.0E+00
	kg O3 eq	Recycle	0.0E+00	1.2E-02	2.4E-02	3.5E-02	4.7E-02	5.9E-02	7.1E-02	8.3E-02	9.5E-02	1.1E-01	1.2E-01
	kg O3 eq	Total	7.9E-03	1.9E-02	3.0E-02	4.1E-02	5.2E-02	6.3E-02	7.4E-02	8.5E-02	9.6E-02	1.1E-01	1.2E-01
Acidification	kg SO2 eq	Landfill	2.8E-04	2.5E-04	2.2E-04	1.9E-04	1.7E-04	1.4E-04	1.1E-04	8.3E-05	5.5E-05	2.8E-05	0.0E+00
	kg SO2 eq	Recycle	0.0E+00	2.1E-04	4.1E-04	6.2E-04	8.3E-04	1.0E-03	1.2E-03	1.5E-03	1.7E-03	1.9E-03	2.1E-03
	kg SO2 eq	Total	2.8E-04	4.6E-04	6.4E-04	8.2E-04	1.0E-03	1.2E-03	1.4E-03	1.5E-03	1.7E-03	1.9E-03	2.1E-03
Eutrophication	kg N eq	Landfill	1.5E-02	1.3E-02	1.2E-02	1.0E-02	8.8E-03	7.3E-03	5.8E-03	4.4E-03	2.9E-03	1.5E-03	0.0E+00
	kg N eq	Recycle	0.0E+00	5.2E-05	1.0E-04	1.5E-04	2.1E-04	2.6E-04	3.1E-04	3.6E-04	4.1E-04	4.6E-04	5.2E-04
	kg N eq	Total	1.5E-02	1.3E-02	1.2E-02	1.0E-02	9.0E-03	7.5E-03	6.1E-03	4.7E-03	3.3E-03	1.9E-03	5.2E-04
Carcinogenics	CTUh	Landfill	1.7E-09	1.5E-09	1.3E-09	1.2E-09	9.9E-10	8.3E-10	6.6E-10	5.0E-10	3.3E-10	1.7E-10	0.0E+00
	CTUh	Recycle	0.0E+00	1.9E-10	3.8E-10	5.6E-10	7.5E-10	9.4E-10	1.1E-09	1.3E-09	1.5E-09	1.7E-09	1.9E-09
	CTUh	Total	1.7E-09	1.7E-09	1.7E-09	1.7E-09	1.7E-09	1.8E-09	1.8E-09	1.8E-09	1.8E-09	1.9E-09	1.9E-09
Non carcinogenics	CTUh	Landfill	3.2E-08	2.9E-08	2.5E-08	2.2E-08	1.9E-08	1.6E-08	1.3E-08	9.5E-09	6.4E-09	3.2E-09	0.0E+00
	CTUh	Recycle	0.0E+00	1.4E-09	2.8E-09	4.2E-09	5.6E-09	7.1E-09	8.5E-09	9.9E-09	1.1E-08	1.3E-08	1.4E-08
	CTUh	Total	3.2E-08	3.0E-08	2.8E-08	2.6E-08	2.5E-08	2.3E-08	2.1E-08	1.9E-08	1.8E-08	1.6E-08	1.4E-08
Respiratory effects	kg PM2.5 eq	Landfill	2.6E-05	2.3E-05	2.0E-05	1.8E-05	1.5E-05	1.3E-05	1.0E-05	7.7E-06	5.1E-06	2.6E-06	0.0E+00
	kg PM2.5 eq	Recycle	0.0E+00	2.2E-05	4.4E-05	6.7E-05	8.9E-05	1.1E-04	1.3E-04	1.6E-04	1.8E-04	2.0E-04	2.2E-04
	kg PM2.5 eq	Total	2.6E-05	4.5E-05	6.5E-05	8.4E-05	1.0E-04	1.2E-04	1.4E-04	1.6E-04	1.8E-04	2.0E-04	2.2E-04
Ecotoxicity	CTUe	Landfill	6.5E-01	5.9E-01	5.2E-01	4.6E-01	3.9E-01	3.3E-01	2.6E-01	2.0E-01	1.3E-01	6.5E-02	0.0E+00
	CTUe	Recycle	0.0E+00	2.1E-02	4.2E-02	6.3E-02	8.4E-02	1.1E-01	1.3E-01	1.5E-01	1.7E-01	1.9E-01	2.1E-01
	CTUe	Total	6.5E-01	6.1E-01	5.6E-01	5.2E-01	4.8E-01	4.3E-01	3.9E-01	3.4E-01	3.0E-01	2.5E-01	2.1E-01
Fossil fuel depletion	MJ surplus	Landfill	6.2E-02	5.6E-02	5.0E-02	4.3E-02	3.7E-02	3.1E-02	2.5E-02	1.9E-02	1.2E-02	6.2E-03	0.0E+00
	MJ surplus	Recycle	0.0E+00	-1.0E+00	-2.0E+00	-3.1E+00	-4.1E+00	-5.1E+00	-6.1E+00	-7.1E+00	-8.1E+00	-9.2E+00	-1.0E+01
	MJ surplus	Total	6.2E-02	-9.6E-01	-2.0E+00	-3.0E+00	-4.0E+00	-5.1E+00	-6.1E+00	-7.1E+00	-8.1E+00	-9.1E+00	-1.0E+01

LDPE	Units	Landfill/ Recycle	100% landfill 0% recycle	90% landfill 10% recycle	80% landfill 20% recycle	70% landfill 30% recycle	60% landfill 40% recycle	50% landfill 50% recycle	40% landfill 60% recycle	30% landfill 70% recycle	20% landfill 80% recycle	10% landfill 90% recycle	0% landfill 100% recycle
Ozone depletion	kg CFC-11 eq	Landfill	5.80E-09	5.22E-09	4.64E-09	4.06E-09	3.48E-09	2.90E-09	2.32E-09	1.74E-09	1.16E-09	5.80E-10	0.00E+00
	kg CFC-11 eq	Recycle	0.00E+00	2.72E-09	5.44E-09	8.16E-09	1.09E-08	1.36E-08	1.63E-08	1.90E-08	2.18E-08	2.45E-08	2.72E-08
	kg CFC-11 eq	Total	5.80E-09	7.94E-09	1.01E-08	1.22E-08	1.44E-08	1.65E-08	1.86E-08	2.08E-08	2.29E-08	2.51E-08	2.72E-08
Global warming	kg CO2 eq	Landfill	1.45E-01	1.31E-01	1.16E-01	1.02E-01	8.70E-02	7.25E-02	5.80E-02	4.35E-02	2.90E-02	1.45E-02	0.00E+00
	kg CO2 eq	Recycle	0.00E+00	-1.09E-01	-2.18E-01	-3.27E-01	-4.36E-01	-5.45E-01	-6.55E-01	-7.64E-01	-8.73E-01	-9.82E-01	-1.09E+00
	kg CO2 eq	Total	1.45E-01	2.14E-02	-1.02E-01	-2.26E-01	-3.49E-01	-4.73E-01	-5.97E-01	-7.20E-01	-8.44E-01	-9.67E-01	-1.09E+00
Smog	kg O3 eq	Landfill	7.91E-03	7.12E-03	6.32E-03	5.53E-03	4.74E-03	3.95E-03	3.16E-03	2.37E-03	1.58E-03	7.91E-04	0.00E+00
	kg O3 eq	Recycle	0.00E+00	1.17E-02	2.34E-02	3.50E-02	4.67E-02	5.84E-02	7.01E-02	8.17E-02	9.34E-02	1.05E-01	1.17E-01
	kg O3 eq	Total	7.91E-03	1.88E-02	2.97E-02	4.06E-02	5.14E-02	6.23E-02	7.32E-02	8.41E-02	9.50E-02	1.06E-01	1.17E-01
Acidification	kg SO2 eq	Landfill	2.77E-04	2.49E-04	2.22E-04	1.94E-04	1.66E-04	1.38E-04	1.11E-04	8.31E-05	5.54E-05	2.77E-05	0.00E+00
	kg SO2 eq	Recycle	0.00E+00	1.92E-04	3.84E-04	5.76E-04	7.69E-04	9.61E-04	1.15E-03	1.35E-03	1.54E-03	1.73E-03	1.92E-03
	kg SO2 eq	Total	2.77E-04	4.41E-04	6.06E-04	7.70E-04	9.35E-04	1.10E-03	1.26E-03	1.43E-03	1.59E-03	1.76E-03	1.92E-03
Eutrophication	kg N eq	Landfill	1.46E-02	1.31E-02	1.17E-02	1.02E-02	8.75E-03	7.29E-03	5.83E-03	4.38E-03	2.92E-03	1.46E-03	0.00E+00
	kg N eq	Recycle	0.00E+00	4.23E-05	8.46E-05	1.27E-04	1.69E-04	2.12E-04	2.54E-04	2.96E-04	3.39E-04	3.81E-04	4.23E-04
	kg N eq	Total	1.46E-02	1.32E-02	1.18E-02	1.03E-02	8.92E-03	7.50E-03	6.09E-03	4.67E-03	3.26E-03	1.84E-03	4.23E-04
Carcinogenics	CTUh	Landfill	1.66E-09	1.49E-09	1.33E-09	1.16E-09	9.94E-10	8.29E-10	6.63E-10	4.97E-10	3.31E-10	1.66E-10	0.00E+00
	CTUh	Recycle	0.00E+00	8.74E-11	1.75E-10	2.62E-10	3.49E-10	4.37E-10	5.24E-10	6.12E-10	6.99E-10	7.86E-10	8.74E-10
	CTUh	Total	1.66E-09	1.58E-09	1.50E-09	1.42E-09	1.34E-09	1.27E-09	1.19E-09	1.11E-09	1.03E-09	9.52E-10	8.74E-10
Non carcinogenics	CTUh	Landfill	3.18E-08	2.86E-08	2.54E-08	2.22E-08	1.91E-08	1.59E-08	1.27E-08	9.53E-09	6.35E-09	3.18E-09	0.00E+00
	CTUh	Recycle	0.00E+00	9.70E-10	1.94E-09	2.91E-09	3.88E-09	4.85E-09	5.82E-09	6.79E-09	7.76E-09	8.73E-09	9.70E-09
	CTUh	Total	3.18E-08	2.96E-08	2.74E-08	2.52E-08	2.29E-08	2.07E-08	1.85E-08	1.63E-08	1.41E-08	1.19E-08	9.70E-09
Respiratory effects	kg PM2.5 eq	Landfill	2.55E-05	2.30E-05	2.04E-05	1.79E-05	1.53E-05	1.28E-05	1.02E-05	7.66E-06	5.11E-06	2.55E-06	0.00E+00
	kg PM2.5 eq	Recycle	0.00E+00	1.93E-05	3.87E-05	5.80E-05	7.73E-05	9.66E-05	1.16E-04	1.35E-04	1.55E-04	1.74E-04	1.93E-04
	kg PM2.5 eq	Total	2.55E-05	4.23E-05	5.91E-05	7.59E-05	9.26E-05	1.09E-04	1.26E-04	1.43E-04	1.60E-04	1.76E-04	1.93E-04
Ecotoxicity	CTUe	Landfill	6.53E-01	5.88E-01	5.22E-01	4.57E-01	3.92E-01	3.26E-01	2.61E-01	1.96E-01	1.31E-01	6.53E-02	0.00E+00
	CTUe	Recycle	0.00E+00	1.08E-02	2.16E-02	3.24E-02	4.32E-02	5.39E-02	6.47E-02	7.55E-02	8.63E-02	9.71E-02	1.08E-01
	CTUe	Total	6.53E-01	5.98E-01	5.44E-01	4.89E-01	4.35E-01	3.80E-01	3.26E-01	2.71E-01	2.17E-01	1.62E-01	1.08E-01
Fossil fuel depletion	MJ surplus	Landfill	6.21E-02	5.59E-02	4.97E-02	4.35E-02	3.73E-02	3.11E-02	2.49E-02	1.86E-02	1.24E-02	6.21E-03	0.00E+00
	MJ surplus	Recycle	0.00E+00	-1.02E+00	-2.04E+00	-3.06E+00	-4.07E+00	-5.09E+00	-6.11E+00	-7.13E+00	-8.15E+00	-9.17E+00	-1.02E+01
	MJ surplus	Total	6.21E-02	-9.63E-01	-1.99E+00	-3.01E+00	-4.04E+00	-5.06E+00	-6.09E+00	-7.11E+00	-8.14E+00	-9.16E+00	-1.02E+01

PLA Amorphous	Units	Landfill/ Recycle	100% landfill 0% compost	90% landfill 10% compost	80% landfill 20% compost	70% landfill 30% compost	60% landfill 40% compost	50% landfill 50% compost	40% landfill 60% compost	30% landfill 70% compost	20% landfill 80% compost	10% landfill 90% compost	0% landfill 100% compost
Ozone depletion	kg CFC-11 eq	Landfill	5.816E-09	5.234E-09	4.653E-09	4.071E-09	3.490E-09	2.908E-09	2.326E-09	1.745E-09	1.163E-09	5.816E-10	0.000E+00
	kg CFC-11 eq	Compost	0.000E+00	3.719E-10	7.439E-10	1.116E-09	1.488E-09	1.860E-09	2.232E-09	2.604E-09	2.975E-09	3.347E-09	3.719E-09
	kg CFC-11 eq	Total	5.816E-09	5.606E-09	5.397E-09	5.187E-09	4.977E-09	4.768E-09	4.558E-09	4.348E-09	4.139E-09	3.929E-09	3.719E-09
Global warming	kg CO2 eq	Landfill	7.992E-01	7.193E-01	6.393E-01	5.594E-01	4.795E-01	3.996E-01	3.197E-01	2.398E-01	1.598E-01	7.992E-02	0.000E+00
	kg CO2 eq	Compost	0.000E+00	3.077E-02	6.153E-02	9.230E-02	1.231E-01	1.538E-01	1.846E-01	2.154E-01	2.461E-01	2.769E-01	3.077E-01
	kg CO2 eq	Total	7.992E-01	7.500E-01	7.009E-01	6.517E-01	6.026E-01	5.534E-01	5.043E-01	4.551E-01	4.060E-01	3.568E-01	3.077E-01
Smog	kg O3 eq	Landfill	9.952E-03	8.957E-03	7.962E-03	6.967E-03	5.971E-03	4.976E-03	3.981E-03	2.986E-03	1.990E-03	9.952E-04	0.000E+00
	kg O3 eq	Compost	0.000E+00	1.383E-03	2.765E-03	4.148E-03	5.531E-03	6.914E-03	8.296E-03	9.679E-03	1.106E-02	1.244E-02	1.383E-02
	kg O3 eq	Total	9.952E-03	1.034E-02	1.073E-02	1.111E-02	1.150E-02	1.189E-02	1.228E-02	1.266E-02	1.305E-02	1.344E-02	1.383E-02
Acidification	kg SO2 eq	Landfill	3.360E-04	3.024E-04	2.688E-04	2.352E-04	2.016E-04	1.680E-04	1.344E-04	1.008E-04	6.721E-05	3.360E-05	0.000E+00
	kg SO2 eq	Compost	0.000E+00	1.253E-04	2.507E-04	3.760E-04	5.014E-04	6.267E-04	7.521E-04	8.774E-04	1.003E-03	1.128E-03	1.253E-03
	kg SO2 eq	Total	3.360E-04	4.278E-04	5.195E-04	6.113E-04	7.030E-04	7.948E-04	8.865E-04	9.783E-04	1.070E-03	1.162E-03	1.253E-03
Eutrophication	kg N eq	Landfill	1.176E-02	1.058E-02	9.409E-03	8.232E-03	7.056E-03	5.880E-03	4.704E-03	3.528E-03	2.352E-03	1.176E-03	0.000E+00
	kg N eq	Compost	0.000E+00	9.683E-06	1.937E-05	2.905E-05	3.873E-05	4.841E-05	5.810E-05	6.778E-05	7.746E-05	8.714E-05	9.683E-05
	kg N eq	Total	1.176E-02	1.059E-02	9.428E-03	8.262E-03	7.095E-03	5.929E-03	4.762E-03	3.596E-03	2.430E-03	1.263E-03	9.683E-05
Carcinogenics	CTUh	Landfill	1.326E-09	1.193E-09	1.061E-09	9.281E-10	7.956E-10	6.630E-10	5.304E-10	3.978E-10	2.652E-10	1.326E-10	0.000E+00
	CTUh	Compost	0.000E+00	2.464E-10	4.928E-10	7.392E-10	9.855E-10	1.232E-09	1.478E-09	1.725E-09	1.971E-09	2.217E-09	2.464E-09
	CTUh	Total	1.326E-09	1.440E-09	1.554E-09	1.667E-09	1.781E-09	1.895E-09	2.009E-09	2.122E-09	2.236E-09	2.350E-09	2.464E-09
Non carcinogenics	CTUh	Landfill	1.130E-08	1.017E-08	9.044E-09	7.913E-09	6.783E-09	5.652E-09	4.522E-09	3.391E-09	2.261E-09	1.130E-09	0.000E+00
	CTUh	Compost	0.000E+00	8.961E-10	1.792E-09	2.688E-09	3.584E-09	4.481E-09	5.377E-09	6.273E-09	7.169E-09	8.065E-09	8.961E-09
	CTUh	Total	1.130E-08	1.107E-08	1.084E-08	1.060E-08	1.037E-08	1.013E-08	9.899E-09	9.664E-09	9.430E-09	9.196E-09	8.961E-09
Respiratory effects	kg PM2.5 eq	Landfill	2.742E-05	2.468E-05	2.194E-05	1.919E-05	1.645E-05	1.371E-05	1.097E-05	8.226E-06	5.484E-06	2.742E-06	0.000E+00
	kg PM2.5 eq	Compost	0.000E+00	5.803E-06	1.161E-05	1.741E-05	2.321E-05	2.902E-05	3.482E-05	4.062E-05	4.643E-05	5.223E-05	5.803E-05
	kg PM2.5 eq	Total	2.742E-05	3.048E-05	3.354E-05	3.660E-05	3.966E-05	4.273E-05	4.579E-05	4.885E-05	5.191E-05	5.497E-05	5.803E-05
Ecotoxicity	CTUe	Landfill	1.609E-01	1.448E-01	1.287E-01	1.126E-01	9.654E-02	8.045E-02	6.436E-02	4.827E-02	3.218E-02	1.609E-02	0.000E+00
	CTUe	Compost	0.000E+00	2.097E-02	4.195E-02	6.292E-02	8.390E-02	1.049E-01	1.258E-01	1.468E-01	1.678E-01	1.888E-01	2.097E-01
	CTUe	Total	1.609E-01	1.658E-01	1.707E-01	1.755E-01	1.804E-01	1.853E-01	1.902E-01	1.951E-01	2.000E-01	2.049E-01	2.097E-01
Fossil fuel depletion	MJ surplus	Landfill	6.213E-02	5.592E-02	4.970E-02	4.349E-02	3.728E-02	3.106E-02	2.485E-02	1.864E-02	1.243E-02	6.213E-03	0.000E+00
	MJ surplus	Compost	0.000E+00	1.157E-02	2.315E-02	3.472E-02	4.630E-02	5.787E-02	6.945E-02	8.102E-02	9.260E-02	1.042E-01	1.157E-01
	MJ surplus	Total	6.213E-02	6.749E-02	7.285E-02	7.821E-02	8.358E-02	8.894E-02	9.430E-02	9.966E-02	1.050E-01	1.104E-01	1.157E-01

PLA Crystalline	Units	Landfill/ Recycle	100% landfill 0% compost	90% landfill 10% compost	80% landfill 20% compost	70% landfill 30% compost	60% landfill 40% compost	50% landfill 50% compost	40% landfill 60% compost	30% landfill 70% compost	20% landfill 80% compost	10% landfill 90% compost	0% landfill 100% compost
Ozone depletion	kg CFC-11 eq	Landfill	5.816E-09	5.234E-09	4.653E-09	4.071E-09	3.490E-09	2.908E-09	2.326E-09	1.745E-09	1.163E-09	5.816E-10	0.000E+00
	kg CFC-11 eq	Compost	0.000E+00	3.719E-10	7.439E-10	1.116E-09	1.488E-09	1.860E-09	2.232E-09	2.604E-09	2.975E-09	3.347E-09	3.719E-09
	kg CFC-11 eq	Total	5.816E-09	5.606E-09	5.397E-09	5.187E-09	4.977E-09	4.768E-09	4.558E-09	4.348E-09	4.139E-09	3.929E-09	3.719E-09
Global warming	kg CO2 eq	Landfill	4.538E-02	4.084E-02	3.631E-02	3.177E-02	2.723E-02	2.269E-02	1.815E-02	1.361E-02	9.077E-03	4.538E-03	0.000E+00
	kg CO2 eq	Compost	0.000E+00	3.077E-02	6.153E-02	9.230E-02	1.231E-01	1.538E-01	1.846E-01	2.154E-01	2.461E-01	2.769E-01	3.077E-01
	kg CO2 eq	Total	4.538E-02	7.161E-02	9.784E-02	1.241E-01	1.503E-01	1.765E-01	2.027E-01	2.290E-01	2.552E-01	2.814E-01	3.077E-01
Smog	kg O3 eq	Landfill	7.938E-03	7.144E-03	6.350E-03	5.556E-03	4.763E-03	3.969E-03	3.175E-03	2.381E-03	1.588E-03	7.938E-04	0.000E+00
	kg O3 eq	Compost	0.000E+00	1.383E-03	2.765E-03	4.148E-03	5.531E-03	6.914E-03	8.296E-03	9.679E-03	1.106E-02	1.244E-02	1.383E-02
	kg O3 eq	Total	7.938E-03	8.527E-03	9.116E-03	9.705E-03	1.029E-02	1.088E-02	1.147E-02	1.206E-02	1.265E-02	1.324E-02	1.383E-02
Acidification	kg SO2 eq	Landfill	2.778E-04	2.500E-04	2.222E-04	1.944E-04	1.667E-04	1.389E-04	1.111E-04	8.333E-05	5.555E-05	2.778E-05	0.000E+00
	kg SO2 eq	Compost	0.000E+00	1.253E-04	2.507E-04	3.760E-04	5.014E-04	6.267E-04	7.521E-04	8.774E-04	1.003E-03	1.128E-03	1.253E-03
	kg SO2 eq	Total	2.778E-04	3.753E-04	4.729E-04	5.705E-04	6.680E-04	7.656E-04	8.632E-04	9.608E-04	1.058E-03	1.156E-03	1.253E-03
Eutrophication	kg N eq	Landfill	1.176E-02	1.058E-02	9.406E-03	8.230E-03	7.054E-03	5.879E-03	4.703E-03	3.527E-03	2.351E-03	1.176E-03	0.000E+00
	kg N eq	Compost	0.000E+00	9.683E-06	1.937E-05	2.905E-05	3.873E-05	4.841E-05	5.810E-05	6.778E-05	7.746E-05	8.714E-05	9.683E-05
	kg N eq	Total	1.176E-02	1.059E-02	9.425E-03	8.259E-03	7.093E-03	5.927E-03	4.761E-03	3.595E-03	2.429E-03	1.263E-03	9.683E-05
Carcinogenics	CTUh	Landfill	1.326E-09	1.193E-09	1.061E-09	9.281E-10	7.956E-10	6.630E-10	5.304E-10	3.978E-10	2.652E-10	1.326E-10	0.000E+00
	CTUh	Compost	0.000E+00	2.464E-10	4.928E-10	7.392E-10	9.855E-10	1.232E-09	1.478E-09	1.725E-09	1.971E-09	2.217E-09	2.464E-09
	CTUh	Total	1.326E-09	1.440E-09	1.554E-09	1.667E-09	1.781E-09	1.895E-09	2.009E-09	2.122E-09	2.236E-09	2.350E-09	2.464E-09
Non carcinogenics	CTUh	Landfill	1.130E-08	1.017E-08	9.044E-09	7.913E-09	6.783E-09	5.652E-09	4.522E-09	3.391E-09	2.261E-09	1.130E-09	0.000E+00
	CTUh	Compost	0.000E+00	8.961E-10	1.792E-09	2.688E-09	3.584E-09	4.481E-09	5.377E-09	6.273E-09	7.169E-09	8.065E-09	8.961E-09
	CTUh	Total	1.130E-08	1.107E-08	1.084E-08	1.060E-08	1.037E-08	1.013E-08	9.899E-09	9.664E-09	9.430E-09	9.196E-09	8.961E-09
Respiratory effects	kg PM2.5 eq	Landfill	2.545E-05	2.291E-05	2.036E-05	1.782E-05	1.527E-05	1.273E-05	1.018E-05	7.636E-06	5.090E-06	2.545E-06	0.000E+00
	kg PM2.5 eq	Compost	0.000E+00	5.803E-06	1.161E-05	1.741E-05	2.321E-05	2.902E-05	3.482E-05	4.062E-05	4.643E-05	5.223E-05	5.803E-05
	kg PM2.5 eq	Total	2.545E-05	2.871E-05	3.197E-05	3.523E-05	3.848E-05	4.174E-05	4.500E-05	4.826E-05	5.152E-05	5.478E-05	5.803E-05
Ecotoxicity	CTUe	Landfill	1.609E-01	1.448E-01	1.287E-01	1.126E-01	9.654E-02	8.045E-02	6.436E-02	4.827E-02	3.218E-02	1.609E-02	0.000E+00
	CTUe	Compost	0.000E+00	2.097E-02	4.195E-02	6.292E-02	8.390E-02	1.049E-01	1.258E-01	1.468E-01	1.678E-01	1.888E-01	2.097E-01
	CTUe	Total	1.609E-01	1.658E-01	1.707E-01	1.755E-01	1.804E-01	1.853E-01	1.902E-01	1.951E-01	2.000E-01	2.049E-01	2.097E-01
Fossil fuel depletion	MJ surplus	Landfill	6.213E-02	5.592E-02	4.970E-02	4.349E-02	3.728E-02	3.106E-02	2.485E-02	1.864E-02	1.243E-02	6.213E-03	0.000E+00
	MJ surplus	Compost	0.000E+00	1.157E-02	2.315E-02	3.472E-02	4.630E-02	5.787E-02	6.945E-02	8.102E-02	9.260E-02	1.042E-01	1.157E-01
	MJ surplus	Total	6.213E-02	6.749E-02	7.285E-02	7.821E-02	8.358E-02	8.894E-02	9.430E-02	9.966E-02	1.050E-01	1.104E-01	1.157E-01

TPS	Units	Landfill/ Recycle	100% landfill 0% compost	90% landfill 10% compost	80% landfill 20% compost	70% landfill 30% compost	60% landfill 40% compost	50% landfill 50% compost	40% landfill 60% compost	30% landfill 70% compost	20% landfill 80% compost	10% landfill 90% compost	0% landfill 100% compost
Ozone depletion	kg CFC-11 eq	Landfill	6.084E-09	5.475E-09	4.867E-09	4.259E-09	3.650E-09	3.042E-09	2.433E-09	1.825E-09	1.217E-09	6.084E-10	0.000E+00
	kg CFC-11 eq	Compost	0.000E+00	3.719E-10	7.439E-10	1.116E-09	1.488E-09	1.860E-09	2.232E-09	2.604E-09	2.975E-09	3.347E-09	3.719E-09
	kg CFC-11 eq	Total	6.084E-09	5.847E-09	5.611E-09	5.374E-09	5.138E-09	4.901E-09	4.665E-09	4.429E-09	4.192E-09	3.956E-09	3.719E-09
Global warming	kg CO2 eq	Landfill	1.250E+00	1.125E+00	1.000E+00	8.751E-01	7.501E-01	6.250E-01	5.000E-01	3.750E-01	2.500E-01	1.250E-01	0.000E+00
	kg CO2 eq	Compost	0.000E+00	2.827E-02	5.655E-02	8.482E-02	1.131E-01	1.414E-01	1.696E-01	1.979E-01	2.262E-01	2.545E-01	2.827E-01
	kg CO2 eq	Total	1.250E+00	1.153E+00	1.057E+00	9.599E-01	8.632E-01	7.664E-01	6.697E-01	5.729E-01	4.762E-01	3.795E-01	2.827E-01
Smog	kg O3 eq	Landfill	8.468E-03	7.621E-03	6.774E-03	5.928E-03	5.081E-03	4.234E-03	3.387E-03	2.540E-03	1.694E-03	8.468E-04	0.000E+00
	kg O3 eq	Compost	0.000E+00	1.383E-03	2.765E-03	4.148E-03	5.531E-03	6.914E-03	8.296E-03	9.679E-03	1.106E-02	1.244E-02	1.383E-02
	kg O3 eq	Total	8.468E-03	9.004E-03	9.540E-03	1.008E-02	1.061E-02	1.115E-02	1.168E-02	1.222E-02	1.276E-02	1.329E-02	1.383E-02
Acidification	kg SO2 eq	Landfill	4.511E-04	4.060E-04	3.609E-04	3.158E-04	2.707E-04	2.256E-04	1.805E-04	1.353E-04	9.023E-05	4.511E-05	0.000E+00
	kg SO2 eq	Compost	0.000E+00	1.253E-04	2.507E-04	3.760E-04	5.014E-04	6.267E-04	7.521E-04	8.774E-04	1.003E-03	1.128E-03	1.253E-03
	kg SO2 eq	Total	4.511E-04	5.314E-04	6.116E-04	6.918E-04	7.721E-04	8.523E-04	9.326E-04	1.013E-03	1.093E-03	1.173E-03	1.253E-03
Eutrophication	kg N eq	Landfill	7.144E-03	6.429E-03	5.715E-03	5.001E-03	4.286E-03	3.572E-03	2.857E-03	2.143E-03	1.429E-03	7.144E-04	0.000E+00
	kg N eq	Compost	0.000E+00	9.683E-06	1.937E-05	2.905E-05	3.873E-05	4.841E-05	5.810E-05	6.778E-05	7.746E-05	8.714E-05	9.683E-05
	kg N eq	Total	7.144E-03	6.439E-03	5.734E-03	5.030E-03	4.325E-03	3.620E-03	2.916E-03	2.211E-03	1.506E-03	8.015E-04	9.683E-05
Carcinogenics	CTUh	Landfill	2.027E-09	1.824E-09	1.621E-09	1.419E-09	1.216E-09	1.013E-09	8.106E-10	6.080E-10	4.053E-10	2.027E-10	0.000E+00
	CTUh	Compost	0.000E+00	2.464E-10	4.928E-10	7.392E-10	9.855E-10	1.232E-09	1.478E-09	1.725E-09	1.971E-09	2.217E-09	2.464E-09
	CTUh	Total	2.027E-09	2.070E-09	2.114E-09	2.158E-09	2.202E-09	2.245E-09	2.289E-09	2.333E-09	2.376E-09	2.420E-09	2.464E-09
Non carcinogenics	CTUh	Landfill	1.945E-08	1.751E-08	1.556E-08	1.362E-08	1.167E-08	9.726E-09	7.781E-09	5.836E-09	3.890E-09	1.945E-09	0.000E+00
	CTUh	Compost	0.000E+00	8.961E-10	1.792E-09	2.688E-09	3.584E-09	4.481E-09	5.377E-09	6.273E-09	7.169E-09	8.065E-09	8.961E-09
	CTUh	Total	1.945E-08	1.840E-08	1.735E-08	1.630E-08	1.526E-08	1.421E-08	1.316E-08	1.211E-08	1.106E-08	1.001E-08	8.961E-09
Respiratory effects	kg PM2.5 eq	Landfill	3.784E-05	3.405E-05	3.027E-05	2.648E-05	2.270E-05	1.892E-05	1.513E-05	1.135E-05	7.567E-06	3.784E-06	0.000E+00
	kg PM2.5 eq	Compost	0.000E+00	5.803E-06	1.161E-05	1.741E-05	2.321E-05	2.902E-05	3.482E-05	4.062E-05	4.643E-05	5.223E-05	5.803E-05
	kg PM2.5 eq	Total	3.784E-05	3.986E-05	4.187E-05	4.389E-05	4.591E-05	4.793E-05	4.995E-05	5.197E-05	5.399E-05	5.601E-05	5.803E-05
Ecotoxicity	CTUe	Landfill	1.823E-01	1.641E-01	1.459E-01	1.276E-01	1.094E-01	9.117E-02	7.293E-02	5.470E-02	3.647E-02	1.823E-02	0.000E+00
	CTUe	Compost	0.000E+00	2.097E-02	4.195E-02	6.292E-02	8.390E-02	1.049E-01	1.258E-01	1.468E-01	1.678E-01	1.888E-01	2.097E-01
	CTUe	Total	1.823E-01	1.851E-01	1.878E-01	1.906E-01	1.933E-01	1.960E-01	1.988E-01	2.015E-01	2.043E-01	2.070E-01	2.097E-01
Fossil fuel depletion	MJ surplus	Landfill	6.214E-02	5.593E-02	4.972E-02	4.350E-02	3.729E-02	3.107E-02	2.486E-02	1.864E-02	1.243E-02	6.214E-03	0.000E+00
	MJ surplus	Compost	0.000E+00	1.157E-02	2.315E-02	3.472E-02	4.630E-02	5.787E-02	6.945E-02	8.102E-02	9.260E-02	1.042E-01	1.157E-01
	MJ surplus	Total	6.214E-02	6.750E-02	7.287E-02	7.823E-02	8.359E-02	8.895E-02	9.431E-02	9.967E-02	1.050E-01	1.104E-01	1.157E-01

APPENDIX N

SUMMARIZED RESULTS OF SENSITIVITY ANALYSIS FOR EIGHT POLYMERS

Bio-PET Recycling			Ethylene Inventory		MRF Technology		Local Transportation		Waste Collection		Baled Recyclables Shipping	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	4.31E-08	0%	0%	-1%	3%	0%	0%	-1%	4%	0%	0%
Global warming	kg CO2 eq	6.93E-01	0%	0%	-1%	2%	-1%	2%	-1%	3%	-31%	0%
Smog	kg O3 eq	3.18E-01	0%	0%	0%	0%	0%	1%	0%	1%	-41%	0%
Acidification	kg SO2 eq	1.24E-02	0%	0%	0%	1%	0%	1%	0%	1%	-36%	0%
Eutrophication	kg N eq	1.22E-03	0%	0%	-2%	5%	0%	0%	0%	1%	-20%	0%
Carcinogenics	CTUh	1.16E-08	-1%	1%	-2%	5%	-1%	1%	0%	1%	-25%	0%
Non carcinogenics	CTUh	2.45E-07	0%	0%	-1%	1%	0%	1%	0%	1%	-11%	0%
Respiratory effects	kg PM2.5 eq	7.69E-04	0%	0%	-1%	2%	0%	0%	-1%	2%	-10%	0%
Ecotoxicity	CTUe	2.95E+00	-3%	3%	-1%	2%	-1%	1%	0%	1%	-18%	0%
Fossil fuel depletion	MJ surplus	-9.03E-01	0%	0%	1%	-1%	1%	-2%	1%	-4%	-42%	0%

PET Recycling			MRF Technology		Local Transportation		Waste Collection		Baled Recyclables Shipping	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	3.37E-08	-2%	4%	0%	0%	-2%	5%	0%	0%
Global warming	kg CO2 eq	8.09E-01	-1%	2%	-1%	1%	-1%	2%	-26%	0%
Smog	kg O3 eq	1.81E-01	0%	1%	-1%	1%	-1%	2%	-72%	0%
Acidification	kg SO2 eq	8.06E-03	-1%	1%	0%	1%	0%	1%	-55%	0%
Eutrophication	kg N eq	5.60E-04	-5%	11%	0%	1%	-1%	2%	-43%	0%
Carcinogenics	CTUh	6.03E-09	-4%	9%	-1%	3%	-1%	2%	-48%	0%
Non carcinogenics	CTUh	6.46E-08	-2%	4%	-1%	2%	-1%	2%	-43%	0%
Respiratory effects	kg PM2.5 eq	6.07E-04	-2%	3%	0%	0%	-1%	2%	-12%	0%
Ecotoxicity	CTUe	9.98E-01	-3%	7%	-2%	3%	-1%	2%	-54%	0%
Fossil fuel depletion	MJ surplus	1.00E+00	-1%	1%	-1%	2%	-1%	4%	-38%	0%

Bio-HDPE Recycling			Ethylene Inventory		MRF Technology		Local Transportation		Waste Collection		Baled Recyclables Shipping	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	8.85E-08	0%	0%	-2%	4%	0%	0%	-1%	2%	0%	0%
Global warming	kg CO2 eq	5.46E-02	0%	0%	-53%	96%	-11%	22%	-12%	36%	-392%	0%
Smog	kg O3 eq	3.63E-01	0%	0%	0%	1%	0%	1%	0%	1%	-36%	0%
Acidification	kg SO2 eq	1.59E-02	0%	0%	-1%	2%	0%	0%	0%	1%	-28%	0%
Eutrophication	kg N eq	2.40E-03	0%	0%	-5%	8%	0%	0%	0%	0%	-10%	0%
Carcinogenics	CTUh	1.75E-08	-2%	2%	-6%	12%	0%	1%	0%	1%	-17%	0%
Non carcinogenics	CTUh	7.25E-07	0%	0%	-1%	1%	0%	0%	0%	0%	-4%	0%
Respiratory effects	kg PM2.5 eq	1.91E-03	0%	0%	-2%	3%	0%	0%	0%	1%	-4%	0%
Ecotoxicity	CTUe	7.32E+00	-6%	6%	-2%	3%	0%	0%	0%	0%	-7%	0%
Fossil fuel depletion	MJ surplus	-7.17E+00	0%	0%	0%	-1%	0%	0%	0%	-1%	-5%	0%

HDPE Recycling			MRF Technology		Local Transportation		Waste Collection		Baled Recyclables Shipping	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	3.52E-08	-6%	11%	0%	0%	-2%	5%	0%	0%
Global warming	kg CO2 eq	8.30E-01	-4%	6%	-1%	1%	-1%	2%	-26%	0%
Smog	kg O3 eq	1.83E-01	-1%	2%	-1%	1%	-1%	2%	-71%	0%
Acidification	kg SO2 eq	8.19E-03	-2%	4%	0%	1%	0%	1%	-54%	0%
Eutrophication	kg N eq	6.38E-04	-17%	30%	0%	1%	-1%	2%	-38%	0%
Carcinogenics	CTUh	6.89E-09	-16%	29%	-1%	2%	-1%	2%	-42%	0%
Non carcinogenics	CTUh	6.83E-08	-8%	14%	-1%	2%	-1%	2%	-41%	0%
Respiratory effects	kg PM2.5 eq	6.31E-04	-5%	10%	0%	0%	-1%	2%	-12%	0%
Ecotoxicity	CTUe	1.08E+00	-11%	20%	-1%	3%	0%	1%	-50%	0%
Fossil fuel depletion	MJ surplus	1.02E+00	-2%	4%	-1%	2%	-1%	4%	-37%	0%

Bio-LDPE Recycling			Ethylene Inventory		MRF Technology		Local Transportation		Waste Collection		Baled Recyclables Shipping	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	1.38E-07	0%	0%	0%	1%	0%	0%	0%	1%	0%	0%
Global warming	kg CO2 eq	3.18E-01	0%	0%	-2%	4%	-2%	4%	-2%	6%	-67%	0%
Smog	kg O3 eq	3.66E-01	0%	0%	0%	0%	0%	1%	0%	1%	-35%	0%
Acidification	kg SO2 eq	1.61E-02	0%	0%	0%	0%	0%	0%	0%	1%	-27%	0%
Eutrophication	kg N eq	2.35E-03	0%	0%	-1%	2%	0%	0%	0%	0%	-10%	0%
Carcinogenics	CTUh	1.70E-08	-2%	2%	-1%	3%	0%	1%	0%	1%	-17%	0%
Non carcinogenics	CTUh	7.35E-07	0%	0%	0%	0%	0%	0%	0%	0%	-4%	0%
Respiratory effects	kg PM2.5 eq	1.93E-03	0%	0%	0%	1%	0%	0%	0%	1%	-4%	0%
Ecotoxicity	CTUe	7.37E+00	-6%	6%	0%	1%	0%	0%	0%	0%	-7%	0%
Fossil fuel depletion	MJ surplus	-6.50E+00	0%	0%	0%	0%	0%	0%	0%	-1%	-6%	0%

LDPE Recycling			MRF Technology		Local Transportation		Waste Collection		Baled Recyclables Shipping	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	8.34E-08	0%	1%	0%	0%	-1%	2%	0%	0%
Global warming	kg CO2 eq	1.11E+00	0%	1%	-1%	1%	-1%	2%	-19%	0%
Smog	kg O3 eq	1.85E-01	0%	0%	-1%	1%	-1%	2%	-70%	0%
Acidification	kg SO2 eq	8.38E-03	0%	1%	0%	1%	0%	1%	-53%	0%
Eutrophication	kg N eq	5.58E-04	-3%	8%	0%	1%	-1%	2%	-43%	0%
Carcinogenics	CTUh	6.16E-09	-4%	8%	-1%	3%	-1%	2%	-47%	0%
Non carcinogenics	CTUh	6.73E-08	-1%	3%	-1%	2%	-1%	2%	-41%	0%
Respiratory effects	kg PM2.5 eq	6.22E-04	-1%	2%	0%	0%	-1%	2%	-12%	0%
Ecotoxicity	CTUe	1.04E+00	-2%	5%	-1%	3%	0%	1%	-52%	0%
Fossil fuel depletion	MJ surplus	1.84E+00	0%	1%	-1%	1%	-1%	2%	-21%	0%

PLA Composting			Composting Inventory		Local Transportation		Waste Collection	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	3.72E-09	-46%	46%	0%	0%	-15%	45%
Global warming	kg CO2 eq	1.46E+00	-2%	2%	0%	1%	0%	1%
Smog	kg O3 eq	2.22E-01	-1%	1%	0%	1%	0%	1%
Acidification	kg SO2 eq	1.76E-02	-5%	5%	0%	0%	0%	1%
Eutrophication	kg N eq	1.40E-03	-4%	4%	0%	0%	0%	1%
Carcinogenics	CTUh	2.48E-09	-42%	42%	-3%	7%	-2%	6%
Non carcinogenics	CTUh	9.15E-09	-20%	20%	-9%	17%	-5%	15%
Respiratory effects	kg PM2.5 eq	5.61E-04	-3%	3%	0%	0%	-1%	2%
Ecotoxicity	CTUe	2.10E-01	-22%	22%	-7%	14%	-2%	7%
Fossil fuel depletion	MJ surplus	4.03E+00	0%	0%	0%	1%	0%	1%

TPS Composting			Composting Inventory		Local Transportation		Waste Collection	
Impact category	Unit	Baseline	Min	Max	Min	Max	Min	Max
Ozone depletion	kg CFC-11 eq	1.49E-07	-1%	1%	0%	0%	0%	1%
Global warming	kg CO2 eq	2.21E+00	-1%	1%	0%	1%	0%	1%
Smog	kg O3 eq	1.00E-01	-3%	3%	-1%	2%	-1%	3%
Acidification	kg SO2 eq	1.71E-02	-5%	5%	0%	0%	0%	1%
Eutrophication	kg N eq	2.77E-03	-2%	2%	0%	0%	0%	0%
Carcinogenics	CTUh	6.94E-09	-15%	15%	-1%	2%	-1%	2%
Non carcinogenics	CTUh	6.08E-08	-3%	3%	-1%	3%	-1%	2%
Respiratory effects	kg PM2.5 eq	1.18E-03	-1%	1%	0%	0%	0%	1%
Ecotoxicity	CTUe	2.18E+00	-2%	2%	-1%	1%	0%	1%
Fossil fuel depletion	MJ surplus	3.10E-01	-6%	6%	-3%	7%	-4%	13%