A Tale of To Go: A Comparative Life Cycle Assessment of Compostable Single Use Foodware

Jenny Y. Chiu

ABSTRACT

With increasing pressure to find alternatives to petrochemical single use foodware, compostable foodware has risen to the forefront. In order to compare the environmental impacts of polylactic acid (PLA) and sugarcane fiber-based bagasse compostable takeout containers, I created a comparative cradle-to-grave life cycle assessment of bagasse and PLA that evaluated greenhouse gas emissions and energy consumption per kilogram of clamshell takeout containers. I approximated a calculation of the greenhouse gas emissions associated with the transportation processes and established four potential end-of-life scenarios. I concluded that PLA clamshell takeout containers resulted in marginally lower greenhouse gas emissions than bagasse (8.121 – 9.30 kg CO_2 equivalents per kg bagasse and 6.383 – 9.109 kg CO_2 equivalents per kg PLA). PLA clamshell takeout containers were 75 times more energetically costly to manufacture and process than bagasse (826.95 MJ per kg PLA and 10.97 MJ per kg bagasse). I also prepared a brief literature review of the environmental toxicants, namely per- and poly-flouroalkyl substances (PFAS), associated with the two types of compostable foodware materials.

KEYWORDS

polylactic acid (PLA), sugarcane fiber (bagasse), cradle-to-grave, greenhouse gas emissions, energy consumption

INTRODUCTION

Single use packaging and foodware is a complex issue that intersects economic, political, and environmental spheres. As of 2018, global production of plastics was at 359 million tons (Plastics Europe 2019). Disposable plastic packaging and foodware constituted approximately 30-40% of this total (Vendries et al. 2020; Luijsterburg and Goossens 2014). Waste management systems have been struggling to keep up with the heterogeneous nature of materials found in packaging (Marsh and Bugusu 2007). In recent years, there has been increased pressure to shift away from petrochemical plastics. From a technical perspective, recycling plastics can be energetically costly and can degrade the quality of the polymer, resulting in a flawed system that fails to capture the majority of plastics produced (Arena et al. 2003). China, the major importer of plastic waste, began to reduce the quantity of plastics it would accept for recycling in 2013 with the Green Fence campaign and in 2017 tightened regulations on contamination tolerance with the National Sword policy (Brooks et al. 2018; CalRecycle 2020). Countries such as Taiwan, Costa Rica, Belize, and India have also made major strides toward limiting specific types of single use plastics (Schnurr et al. 2018). As of 2019, the City of Berkeley and other municipalities have passed single use foodware reduction ordinances, creating economic incentives to decrease demand for petrochemical plastic foodware (City of Berkeley 2020). These policies have had rippling impacts on the makeup of food packaging. Foodware tends to have unacceptable levels of food contamination, making it desirable to design foodware so that it can degrade in a composting facility (Ingrao et al. 2017). There is therefore unprecedented demand for sustainable and compostable foodware.

Multiple alternatives to petrochemical plastic have been developed to address this demand, including fiber-based and biodegradable plastic foodware. Here it is important to make a distinction between biodegradable plastics and bioplastics, which are often mistakenly conflated. Although bioplastics are sourced from naturally occurring materials, they are not inherently biodegradable. Biodegradable materials must be able to break down in the environment within a reasonable time frame (Johansson, et al. 2012). The ISO 14855 standard sets this time frame to 90 days, although some materials will take longer to degrade (ISO 2018). The rate of biodegradation also depends on the biotic and abiotic factors of composting conditions, including salinity, moisture content, the presence or absence of oxygen, and microbial makeup (Kale et al. 2007).

There are four major types of biodegradable materials: polymers derived from biotechnology processes (such as polylactic acid), agricultural resources (such as bagasse), microbial extraction, and chemical synthesis (Trinetta 2016). These biodegradable materials make composting, a natural process that converts organic materials into the soil-like humus, a viable end-of-life option (Kale et al. 2007). However, a meta-analysis of 71 studies found that biodegradability alone is not a strong predictor of environmental impact (Vendries et al. 2020). It is too focused on the end-of-life of a product and does not take the holistic life cycle into account. As it is difficult to determine whether or not a material is sustainable without evaluating greenhouse gas emissions and other byproducts associated with its raw material acquisition, production, transportation, and disposal, life cycle assessments (LCAs) have become a valuable tool when it comes to evaluating environmental impacts (Al-Salem et al. 2009). Existing life cycle assessments traditionally analyze the environmental impacts of each material separately (Zabaniotou and Kassidi 2003). As more and more alternatives to single use plastics appear on the market, there needs to be a comparative analysis of the environmental impacts of these materials.

Toxicant analyses are another aspect of product life cycles that can offer insight on a type of biodegradable foodware's environmental impacts. In order to maintain their desirable qualities and shelf life, foodware made of biodegradable plastics and fiber-based materials can rely on additives that are toxic to environmental and human health (Dilkes-Hoffman et al. 2018). This is also a public health concern, as any leaching from foodware carries higher risk of accidental ingestion due to the nature of this type of packaging. Packaging also has known environmental impacts. Plastics have been found to adsorb other toxicants, such as bisphenol A, in sea water and have the potential to move up through the food chain through biomagnification (Teuten et al. 2009). Current studies typically focus on the toxicants that occur as a direct result of the manufacture of materials, but environmentally-based analyses should have a more holistic perspective (Huijbregts et al. 2000). Both upstream and downstream effects of toxicants should be examined, as toxicants may be leached into the environment during manufacturing, use, and disposal. As it is difficult to completely quantify and compare the interactions between different toxicants and their impacts on public and environmental health, I will review the literature on toxicants related to the life cycle of compostable foodware to supplement my research.

This study will examine differences in environmental impacts throughout the life cycle of two biodegradable alternatives to single use plastic foodware: polylactic acid (PLA) and bagasse (sugar cane fiber). I will track greenhouse gas emissions (kg CO_2 eq.) and energy consumption (MJ) from cradle-to-grave of these two materials. As PLA is a polymer made of agricultural products, high levels of processing are expected in order to synthesize this polymer (Karamanlioglu et al. 2017). I thus hypothesized that the life cycle of PLA will consume more energy than that of bagasse, as this initial difference is quite pronounced. Due to the relatively more intense processing needed to create PLA as compared to fiber-based foodware, I hypothesized that the life cycle of PLA will also result in higher greenhouse gas emissions (to be measured as carbon dioxide equivalents). The environmental impact of PLA and bagasse will thus be compared quantitatively across greenhouse gas emissions and energy consumption, and more qualitatively on environmental toxicants throughout the life cycle.

METHODS

Study system

I split the study into two segments: (1) a quantitative comparative life cycle assessment (LCA) of bagasse and PLA in compostable foodware (specifically clamshell takeout containers) and (2) a supplementary qualitative literature review that would explore toxicological impacts. The cradle-to-grave LCA spanned four phases: (1) raw material acquisition, (2) production, (3) consumption, and (4) disposal. As more robust LCA databases and software were cost-prohibitive to this specific study, and this study relied heavily on literature references for data, I tracked my calculations and sources in Microsoft Excel. These are attached as Appendix B, C, and D.

Comparative life cycle assessment

Goal and scope

The two products I compared were single use bagasse (sugar cane fiber-based) and PLA clamshell takeout containers. When bagasse and PLA clamshell containers were standardized to the dimensions 20.32 cm ×20.32 cm ×7.62 cm, the weights of each container were nearly identical—one PLA container was estimated to weigh 43.9 g, and one bagasse container was estimated to weigh 43.8 g (WorldCentric 2018). As the PLA container was only 1.001 times heavier than the bagasse container, I determined that it was reasonable to use the weight of clamshell takeout containers rather than the raw number of containers as my functional unit. One functional unit was considered to be 1 kg of clamshell takeout containers produced from each respective material.

The inventory data was extracted from the literature (Vink et al. 2010; Marsolek 2003; Madival, et al. 2009). For bagasse, I relied on the processes used by StalkMarket Products under Asean Corporation. This report relied on data obtained by consistently interviewing manufacturers, and adhered to guidelines set by the 2004 Greenhouse Gas Protocol's Corporate Standard (Marsolek 2003). For PLA, I relied on the cradle-to-gate Ingeo PLA Eco-profile for production (Vink et al. 2010). The Ingeo PLA Eco-profile is heavily referenced in literature, as Natureworks is a major producer of PLA. I supplemented the rest of the life cycle (gate-to-grave) with data from a comparative approach that complied with ISO 14040, 14044, and 14044 standards (Madival et al. 2009). I also referenced emission factors from the EPA in my calculations and relied on their end-of-life data from the 2016 and 2019 Waste Reduction Models (EPA 2016, EPA 2019, EPA 2020).

In order to ground this study, I chose the University of California, Berkeley (UC Berkeley) as the location of consumption. This provided a controlled location for any transport calculations linked to greenhouse gas emissions and energy consumption. As disposal options and waste sorting behaviors vary region by region, end-of-life scenarios were also dependent on UC Berkeley's waste management contracts.

System boundaries

The system boundaries presented in Fig. 1 are comprehensive of the LCA and literature review aspect of the project and track greenhouse gas emissions (kg CO_2 eq. per kg material) and energy consumption (MJ per kg material) across the raw material acquisition, production, consumption, and disposal phases. These phases are further broken down into stages, with land and maritime transport accounted for separately within each phase. Raw material acquisition encompasses the agricultural aspects of the life cycle. Production spans from transport from mill to clamshell manufacture. It also includes fiberboard manufacture or PLA synthesis. The consumption phase includes travel from the factory gate to a distributor and finally to the place of consumption. Disposal includes transport to the end-of-life option.

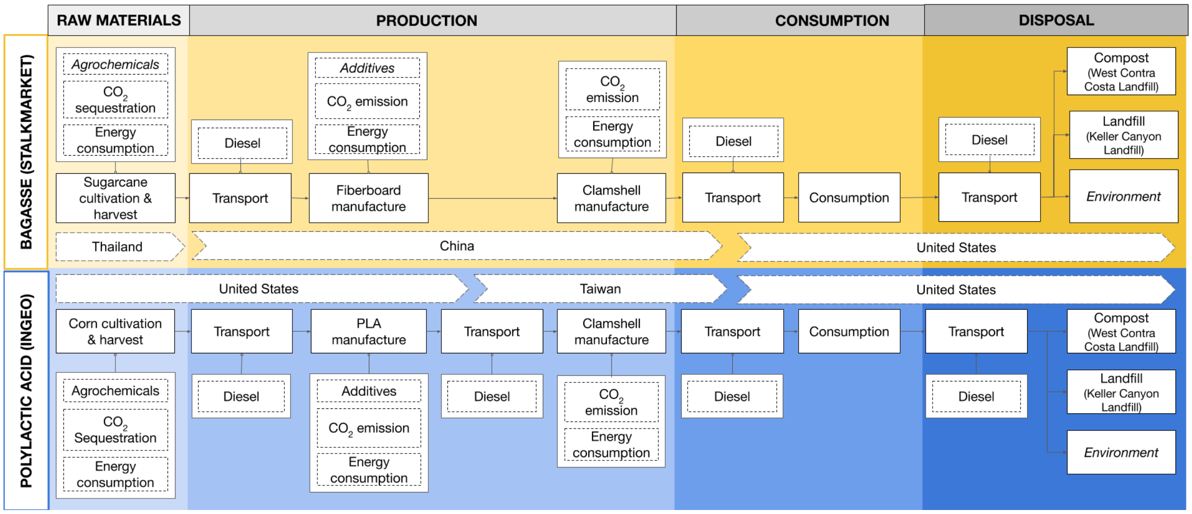

Figure 1. Comparative life cycle definition. This flowchart details the life cycle of both bagasse and PLA from raw materials to production, consumption, and finally disposal. The life cycle of bagasse is on the top (yellow gradation), whereas the life cycle of PLA is on the bottom (blue gradation). Inputs are denoted in rectangles bordered with a dashed line. The dashed arrows running through the center of the chart denote changes in location. An enlarged version of this life cycle definition is included as Appendix A.

Raw material acquisition. Raw material acquisition for bagasse occurred in Thailand (Marsolek 2003), whereas the feedstock for PLA was produced in the Midwest (Vink et al. 2010). Carbon sequestration from photosynthesis was factored into the LCA as a negative quantity to represent CO_2 removed from the atmosphere. Biogenic carbon is typically unaccounted for in cradle-to-grave systems, as the material itself is assumed to be carbon neutral if 100% degradation occurs (Morao and de Bie 2019). However, this has not been demonstrated to be the case with PLA especially, so carbon sequestration is accounted for separately in this phase (Bohlmann 2004).

Transport. Transport is not listed as a separate phase in this LCA, but can be extracted from the life cycle and isolated into its own category when considering the impact of localized versus international systems. After raw materials were acquired in Thailand, sugar cane fibers were then transferred to China for fiberboard and clamshell manufacture and finally shipped to the United States for distribution and consumption (Marsolek 2003; Harnoto 2013). PLA raw material acquisition began in the United States, and PLA pellets were then shipped to Taiwan for manufacture (Vink et al. 2010; Kuo 2017). As the location for the manufacturer in Taiwan was redacted, I assumed that the factory was located in Changhua County, which housed multiple packaging manufacturers, including some that create PLA clamshells (Kuo 2017; G.T. Internet Information n.d.). With UC Berkeley as my set location for consumption, composted waste was sent to the West Contra Costa Landfill in Richmond, CA and landfilled waste was sent to the Keller Canyon Landfill in Pittsburg, CA (Cal Zero Waste n.d.).

To calculate greenhouse gas emissions occurring as a result of transport, I used the following formula:

$$= container\ weight\ (kg_{product}) \times \#of\ containers \times distance\ (km) \times emission\ factor \left(\frac{kg\ CO_2}{kg_{product} \times km}\right)$$

Emission factors were derived from the EPA Emission Factors for Greenhouse Gas Inventories, with an emission factor of 0.001418 $\frac{kg\ CO_2}{kg_{product} \times km}$ for transport by truck and an emission factor of 0.00002740 $\frac{kg\ CO_2}{kg_{product} \times km}$ for maritime transport (EPA 2020). I assumed that diesel trucks were the only type of land transportation utilized. Distances were estimated using Google Maps (Google n.d.). The inputs for these calculations are listed in Appendix B.

Production. In bagasse production, moisture is removed from bagasse and it is then pressed into fiberboard sheets, which are then molded and coated with a hydrophobic surfactant. There is immaterial loss of bagasse during this process (Marsolek 2003). Transport from Thailand to China and within China was factored into this stage for bagasse.

Melt extrusion is currently the most common technique for manufacturing foodware and other consumer goods from PLA (Castro-Aguirre et al. 2016). Afterwards, PLA is thermoformed into sheets and shaped into containers (Madival et al. 2009). Transport within the United States, from the United States to Taiwan, and within Taiwan was factored into this stage for PLA.

Consumption. The consumption phase encompassed transport from the manufacturer to the distributor and finally to the place of consumption, which was assumed to have occurred at UC

Berkeley. Beyond transport to and from the place of consumption, no significant carbon emissions or energy consumption were expected to have occurred in this stage. Consumption controlled what disposal options were available.

Disposal. Due to the limitations around recycling bioplastics and fiberboard containers, recycling was not considered to be a viable end-of-life scenario at this point in time in the United States (Hottle et al. 2017; EPA 2016). There is also high organic contamination of foodware, again making recycling infeasible for bagasse and PLA clamshell containers (Ingrao et al. 2017).

I examined four end-of-life scenarios involving landfill and compost, which are summarized in Table 1. Scenario I and IV served as controls and assumed that 100% of each material would wind up in a landfill (Keller Canyon Landfill) or an industrial composting facility (West Contra Costa Landfill) respectively. Scenario II represented 80% landfill and 20% compost, and Scenario III represented 20% landfill and 80% compost.

	% Landfill	**% Compost**	**Most applicable for**
Scenario I	100	0	Control
Scenario II	80	20	PLA
Scenario III	20	80	Bagasse
Scenario IV	0	100	Control

Table 1. End-of-life scenarios. I explored four disposal scenarios for each of the material life cycles.

I chose the ratio for compost and landfill in Scenario II based on the case study of Lawrence Berkeley National Laboratory (LBNL). I selected LBNL as it had nearly identical waste signage to UC Berkeley's, but did not allow for PLA foodware in their compost. Using the waste audit data from their cafeteria (Building 54), I extrapolated that approximately 20% of PLA foodware by weight was wrongly composted, and that 20% of fiber-based foodware was incorrectly sorted into landfill (LBNL 2020). Scenario II was thus considered to be the most likely scenario for PLA foodware, as confusion around bioplastics has led to high rates of incorrect disposal (Ingrao et al.

2017). Scenario III was assumed to be the most likely scenario for bagasse foodware. Having contributed to the LBNL waste audits between 2018-2019, I assumed that these ratios were reasonable and extrapolated these values to the UC Berkeley campus as the last full-scale waste audit reported at UC Berkeley did not have a breakdown of materials within waste streams (King 2013).

It is commonly assumed that PLA is a carbon sink in an industrial composting facility, but more recent studies have observed that this is a faulty assumption (Krause and Townsend 2016). Degradation rates noted in literature tend to be under stringent laboratory conditions, so I instead relied on measurements of fugitive emissions from industrial composting facilities and landfills from the EPA Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (EPA 2016). I used values for mixed organic waste to encapsulate bagasse-related greenhouse gas emissions and energy consumption.

Literature review

For the literature review, I searched combinations of the following keywords: “life cycle assessment,” “polylactic acid (PLA),” “fiber-based,” “foodware,” “greenhouse gas emissions,” “energy consumption,” and “environmental toxicant” across material science, toxicology, waste, and environmental publications in Google Scholar. For the disposal phase of the life cycles, I compiled the literature across two separate categories (“landfill” and “composting facility”) in order to fill the gap in my data references.

RESULTS

The largest share of greenhouse gas emissions for both bagasse and PLA were associated with the production phase. The greenhouse gas emissions associated with the disposal phase were almost negligible. The most energy-intensive phase for both materials was also production, and the energy use associated with consumption was almost negligible.

Comparative life cycle assessment (LCA)

Greenhouse gas emissions

Total cradle-to-grave greenhouse gas emissions ranged from 8.121 – 9.30 $\frac{kg\ CO_2}{kg\ bagasse}$ for bagasse and from 6.383 – 9.109 $\frac{kg\ CO_2}{kg\ PLA}$ for PLA. The maximum greenhouse gas emissions occurred during production for both materials. For bagasse, production was responsible for 91.86 – 112.32 % of greenhouse gas emissions. For PLA, production was responsible for 61.47 – 87.72% of greenhouse gas emissions. In comparison, relatively minimal greenhouse gas emissions were associated with consumption and disposal. Each accounts for <1.25% of greenhouse gas emissions across scenarios and materials. Refer to Appendix C for the raw values.

		Scenario I		**Scenario II**		**Scenario III**		**Scenario IV**	
Phase	**Stage**	Bagasse	PLA	Bagasse	PLA	Bagasse	PLA	Bagasse	PLA
Raw Material Acquisition	Feedstock cultivation, harvest	-22.08%	-27.11%	-18.07%	-19.54%	-18.11%	-30.40%	-22.09%	-27.12%
Production	Manufacture	112.26%	78.23%	91.86%	61.47%	92.07%	87.72%	112.32%	78.27%
	Transport (land)	3.44%	4.11%	2.81%	3.23%	2.82%	4.61%	3.44%	4.11%
	Transport (maritime)	1.18%	3.73%	0.97%	2.93%	0.97%	4.19%	1.18%	3.73%
Consumption	Transport (land)	0.93%	0.09%	0.76%	0.07%	0.76%	0.10%	0.93%	0.09%
	Transport (maritime)	0.08%	0.09%	0.07%	0.07%	0.07%	0.10%	0.08%	0.09%
Disposal	Transport	0.08%	0.09%	0.11%	0.12%	0.25%	0.38%	0.04%	0.04%
	Facility	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%

Table 2. Greenhouse gas emissions normalized by scenario. This table describes the % of greenhouse gas emissions each stage is responsible for within each scenario. Note that due to carbon sequestration being a negative value, some stages such as manufacture will account for over 100% of greenhouse gas emissions in a scenario.

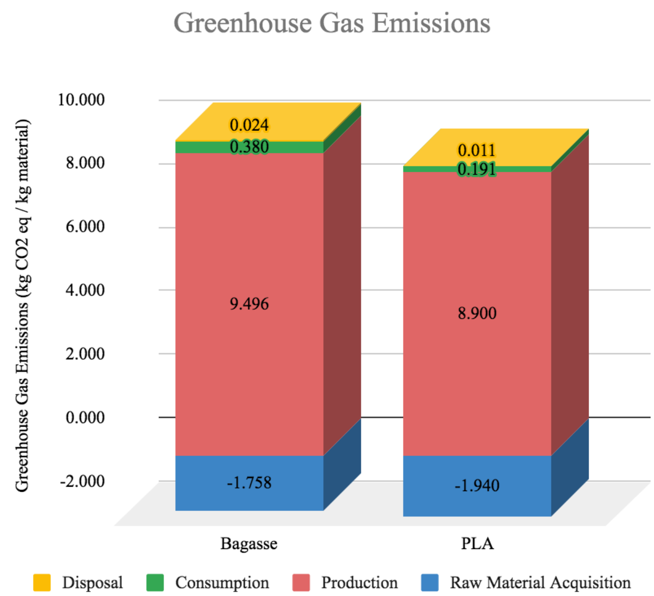

Figure 2. Greenhouse gas emission comparison. The greenhouse gas emissions associated with the bagasse life cycle (Scenario III) and the PLA life cycle (Scenario II) are visualized in Fig. 2.

Energy consumption

Cradle-to-grave energy consumption totaled 10.97$\frac{MJ}{kg\ bagasse}$ for bagasse and 826.95 $\frac{MJ}{kg\ PLA}$ for PLA. The largest portion of energy consumption occurred during production for both materials. The production phase for creating a kilogram of bagasse clamshell containers was responsible for 9.859 MJ of energy consumption. For a kilogram of PLA, production was responsible for 759.624 MJ of energy consumption . Relatively minimal greenhouse gas emissions were associated with consumption, and energy usage during disposal was negligible.

Table 3. Energy consumption normalized by scenario. This table describes the % of energy consumption each stage is responsible in each end-of-life scenario. Refer to Appendix D for the raw values.

		Scenario I		Scenario II		Scenario III		Scenario IV	
Phase	**Stage**	*Bagasse*	*PLA*	*Bagasse*	*PLA*	*Bagasse*	*PLA*	*Bagasse*	*PLA*
Raw Material Acquisition	Feedstock cultivation, harvest	9.87%	8.14%	9.87%	8.14%	9.87%	8.14%	9.87%	8.14%
Production	Material manufacture	2.17%	0.07%	2.17%	0.07%	2.17%	0.07%	2.17%	0.07%
	Clamshell manufacture	87.48%	91.78%	87.47%	91.78%	87.47%	91.78%	87.48%	91.78%
	Transport (truck)	0.18%	0.00%	0.18%	0.00%	0.18%	0.00%	0.18%	0.00%
	Transport (ocean freight)	0.06%	0.00%	0.06%	0.00%	0.06%	0.00%	0.06%	0.00%
Consumption	Transport (truck)	0.05%	0.00%	0.05%	0.00%	0.05%	0.00%	0.05%	0.00%
	Transport (ocean freight)	0.19%	0.00%	0.19%	0.00%	0.19%	0.00%	0.19%	0.00%
Disposal	Transport (truck)	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
	Facility	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%

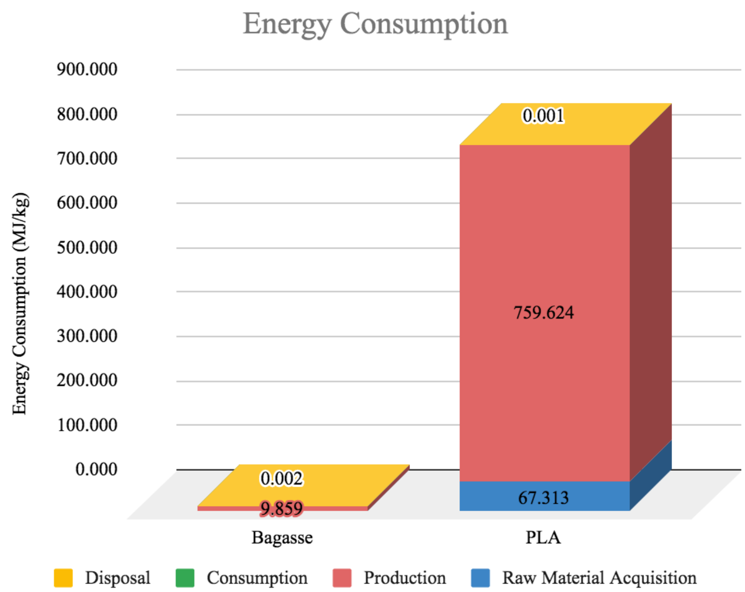

Figure 3. Energy consumption comparison. The energy consumption associated with the bagasse life cycle (Scenario III) and the PLA life cycle (Scenario II) are visualized in Fig. 3.

End-of-life scenarios

Scenario IV yielded the lowest greenhouse gas emissions for bagasse (8.121$\frac{kg\ CO_2\ eq}{kg\ bagasse}$), and Scenario II yielded the highest (9.930$\frac{kg\ CO_2\ eq}{kg\ bagasse}$). Scenario III yielded the lowest greenhouse gas emissions for PLA (6.383$\frac{kg\ CO_2\ eq}{kg\ bagasse}$), and Scenario II yielded the highest (9.109$\frac{kg\ CO_2\ eq}{kg\ PLA}$). Along with Scenario IV, Scenario I had the lowest energy consumption for the bagasse (10.969 $\frac{MJ}{kg\ bagasse}$) and PLA life cycles (826.952 $\frac{MJ}{kg\ PLA}$). However, Scenario II and III only had 0.001 – 0.002 MJ more energy consumption per kilogram of each respective material. In terms of energy consumption, there was no significant difference between end-of-life scenarios for each material at the scale of the functional unit. Refer to Appendix C and D for the detailed record of greenhouse gas emissions and energy consumption that occurs in each stage of bagasse and PLA life cycles.

Literature review

I explored environmental toxicants associated with bagasse and PLA in a supplementary literature review. Sugar cane is itself used to produce edible sugar and is considered to be safe. Industrially, the main concern for human health is the fine bagasse dust that is a side product of processing sugar cane fibers (Bhattacharjee et al. 1980). PLA is also considered to be non-toxic for humans and is Generally Recognized as Safe (GRAS) by the US Food and Drug Administration (FDA), and only trace amounts of PLA are expected to migrate from foodware onto food (Castro-Aguirre et al. 2016; Jamshidian et al. 2010). The main source of exposure to toxicants in food packaging therefore exists in its additives and any other chemicals used in its manufacture (Vink et al. 2010). Additives are added at various phases of its life cycle in order to promote desired properties. For example, one type of PLA food packaging is coated in silicon oxide to decrease permeability for O_2, moisture, and aroma compounds (Castro-Aguirre et al. 2016). Another LCA notes the comparative impact value of respiratory inorganics that occur as a result of processing PLA food ware (Madival et al. 2009).

The toxicants of greatest concern in my literature review were per- and polyfluoroalkyl substances (PFAS). PFAS are an additive that increases grease-resistance in foodware (Schaider et al. 2017). PFAS can have negative health impacts in humans, and most people have already been exposed to PFAS, as it has many industrial applications and its rates of degradation in the environment are exceedingly low. Health impacts include adverse effects on reproduction and development, liver and kidneys, and the immune system, with animal studies that also suggest potential for tumors (EPA n.d.). It is mostly a concern with fiber-based foodware, as PFAS is more commonly part of its production, and can migrate from packaging into food to be ingested by humans (Rosenmai et al. 2016). Fluorinated compounds are also a major concern in other types of fiber- or paper- based foodware, with 20–46% of 400 fast food packaging samples containing PFAS (Schaider et al. 2017).

DISCUSSION

Comparative life cycle assessment (LCA)

Greenhouse gas emissions

The significance of the calculated greenhouse gas emissions mostly lies in its global warming potential. I discovered that two materials were associated with approximately the same amount of greenhouse gas emissions, with PLA being marginally less (8.121 – 9.30 $\frac{kg\ CO_2}{kg\ bagasse}$ for bagasse and from 6.383 – 9.109 $\frac{kg\ CO_2}{kg\ PLA}$ for PLA). This could be due to the larger carbon sequestration capacity of PLA feedstock. PLA and bagasse are both alternatives to petrochemical plastic packaging and were derived from renewable feedstock. As this feedstock can sequester carbon before their degradation releases the carbon back into the atmosphere, it is important to acknowledge this key component of the carbon cycle in a cradle-to-grave LCA (Marsolek 2003).

A high percentage of each material's greenhouse gas emissions was associated with production. This may be an area for future improvement. Although PLA does require more processing than bagasse, as it must undergo fermentation and polymerization in addition to extrusion and thermoforming, the NatureWorks PLA factory is unique in its use of renewable energy (Vink et al. 2010). This greatly offset the greenhouse gas emissions associated with PLA material manufacture.

The current system boundary spans both Asia and the United States. Transport accounted for approximately 5% of the greenhouse gas emissions for both materials, including the majority of emissions in consumption and disposal phases, suggesting that should localized manufacturing systems be created, greenhouse gas emissions may drop correspondingly. As I assumed transport by diesel-fueled truck (over rail) when transport by land was needed, I may also have overestimated the impact of transport for greenhouse gas emissions. However, transport by rail occurs on more fixed paths, so additional modeling would be required to verify this. Although in this study I have attempted to be explicit in the distance travelled by each product, there appears to be wide variability depending on the region of production and consumption, as the EPA estimates that PLA travels on average 799.8 km per shipment, which is at least 400 km less than what this study estimated (EPA 2019).

Energy Consumption

Since the United States economy is currently fueled by petroleum, it is important to quantify the energy consumption of each product's life cycle.As different fuels are becoming more available, quantifying energy consumption could be useful when theorizing processes with a smaller carbon footprint (Vink et al. 2010). As I hypothesized, PLA does consume more energy in its life cycle than bagasse (75 times more). Around 80-90% of each material's energy consumption occurs during the clamshell manufacturing stage, suggesting that this could be a target for improved production design. In contrast, very little energy is consumed during the consumption and disposal phases. In the case of bagasse and PLA clamshell takeout containers, increased energy efficiency upstream in the production phase would have more of an impact than improved waste management design. Again the parameters set in this LCA have a great influence on the total energy consumption I estimated in each scenario, as the EPA's cradle-to-gate estimate of 34.2 MJ/kg PLA in their streamlined LCA is far below what this study predicted (EPA 2019). However, this can be attributed to the differences in system boundaries.

End-of-life scenarios

Depending on the temperature and moisture conditions, there has been no consensus on if PLA undergoes degradation over radical and concerted non-radical reactions, a hydroxyl end-initiated ester interchange, or random scission (Castro-Aguirre et al. 2016). Without a specific pathway, it was difficult to estimate the percentage of carbon in PLA that would be released during its degradation in a landfill or industrial composting facility. My results suggested a near-zero emission and energy consumption from end-of-life, which could be an underestimate (Krause and Townsend 2016).

Literature review

Although environmental toxicity is harder to directly compare than greenhouse gas emissions and energy consumption, it is important to understand the upstream and downstream chemicals released that could harm the environment and human health. PFAS is currently a family of chemicals that has caused concern among environmental scientists and public health officials (Clara et al. 2008). It is often used as an additive to induce favorable physical properties in materials and also increases stable shelf life (Schaider et al. 2017). Currently, PFAS is more predominant in fiber-based foodware, so it may be interesting to research safer chemicals that can achieve similar effects, or innovate on food packaging design such that PFAS is not needed to begin with.

Limitations

I examined a specific subset of fiber-based packaging in very specific scenarios, but environmental impacts of fiber-based packaging can vary based on feedstock and % waterproof coating added (Rodriguez et al. 2018). My study was also limited by the data available. For Ingeo PLA, the literature aggregated production into simplified steps to protect proprietary information (Vink et al. 2010). There is also variability among the literature. I relied on the EPA Waste Reduction Model to estimate greenhouse gas emissions and energy consumption at the end-of-life, calculating that greenhouse gas emissions from landfills were negligible (EPA 2016). However, in another study, 1 kg of PLA could result in 1.24 kg CO_2 eq. of methane emissions in a worst case scenario (Bohlmann 2004). These scenarios were difficult to translate from one landfill system to another, especially as many LCAs examine end-of-life in Europe facilities that utilizes different processes than what is standard in the United States (Krueger et al. 2009).

My study was also highly specific to its location, as waste management is highly dependent on municipality and can even vary within a city. For example, UC Berkeley utilizes a different waste removal service than the City of Berkeley, and its waste follows slightly different paths (Cal Zero Waste 2020). Another challenge of a comparative LCA is that certain assumptions about supply chain and distribution have to be made, but this limits the applicability of these results to other scenarios. As my environmental toxicants literature review was qualitative, it was difficult to use as a point of comparison as well.

Future directions

Due to the complexity of supply chains, transport is often an excluded category in comparative LCAs. Depending on the nature of and the distance a product travels, however, the carbon footprint that results from transport is not negligible (Ingrao et al. 2015). This could also be a consideration when localizing economies. There is flexibility in feedstock and raw material acquisition, as PLA can be manufactured from other plants. Currently PLA is predominantly manufactured from corn, but it has also been produced with sugarcane and other feedstocks (Morao and de Bie 2019).

In addition, future research could incorporate more impact categories and more detailed end-of-life analyses with empirical evidence. There is a lack of literature and data on PLA and bagasse degradation in industrial composting facilities and landfills, as most focus on a laboratory setting. There is also potential to explore the impacts of the materials should they wind up in the environment instead. For environmental dumping, it has been noted that many bio-based materials require specific temperature, humidity, and microbial conditions to efficiently degrade, therefore degrading slowly or not at all outside of industrial-scale composting facilities (Hottle et al. 2017). However, more specific studies could be done to provide more specific end-of-life data.

Future studies should focus on a combination of LCA and actionable items. There has been a global movement toward limiting petrochemical plastic use (Brooks et al. 2018; CalRecycle 2020; City of Berkeley 2020). The culture of convenience that currently exists in the United States has created a great need for alternatives to petroleum-based plastics. By creating LCA-driven policies, the environmental impact of the policies will be strengthened.

Conclusions and broader implications

Currently, bagasse seems to be the more attractive option based on energy consumption, whereas PLA has a slightly lower carbon footprint. Whereas both materials may introduce additives, PFAS tends to be more commonly associated with fiber-based foodware such as that made of bagasse. Despite the specific nature of this study, it provides a unique perspective on LCAs as a comparative tool especially as policies move us toward biodegradable packaging. With more robust databases, comparative LCAs can be a powerful tool for purchasing departments to decide which products have the least environmental impact, or for policymakers to decide how to word the language of an ordinance.

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REFERENCES

Al-Salem, S. M., Lettieri, P., and J. Baeyens. 2009. Recycling and recovery routes of plastic solid waste (PSW): A review. Waste Management 29(10): 2625-2643.

Arena, U., Mastellone, M. L., and F. Perugini. 2003. Life cycle assessment of a plastic packaging recycling system. The International Journal of Life Cycle Assessment 8(2): 92.

Bhattacharjee, J. W., Saxena, R. P., and S.H. Zaidi. 1980. Experimental studies on the toxicity of bagasse. Environmental Research 23(1), 68-76.

Bohlmann, G. M. 2004. Biodegradable packaging life-cycle assessment. Environmental Progress 23(4), 342-346.

Brooks, A. L., Wang, S., and J.R. Jambeck. 2018. The Chinese import ban and its impact on global plastic waste trade. Science Advances, 4(6).

Cal Recycle. 2020. Text of Timeline of China's Import Policies and Response from Affected Parties. <https://www.calrecycle.ca.gov/markets/nationalsword/timelinetext>. Retrieved on 30 April 2020.

Cal Zero Waste. N.d. Cal Zero Waste Operational Questions. <https://facilities.berkeley.edu/operating-units/campus-operations/cal-zero-waste/cal-zero-waste-faqs>. Retrieved on 30 April 2020.

Castro-Aguirre, E., Iniguez-Franco, F., Samsudin, H., Fang, X., and R. Auras. 2016. Poly (lactic acid)—Mass production, processing, industrial applications, and end of life. Advanced Drug Delivery Reviews 107, 333-366.

City of Berkeley. 2020. Single Use Foodware and Litter Reduction. <https://www.cityofberkeley.info/uploadedFiles/Public_Works/Level_3_-_Solid_Waste/2019-02-19%20Item%2001%20Ordinance%207639.pdf>. Retrieved on 29 March 2020.

Clara, M., Scharf, S., Weiss, S., Gans, O., and C. Scheffknecht. 2008. Emissions of perfluorinated alkylated substances (PFAS) from point sources—identification of relevant branches. Water Science and Technology 58(1), 59-66.

Dilkes-Hoffman, L. S., Lane, J. L., Grant, T., Pratt, S., Lant, P. A., and B. Laycock. 2018. Environmental impact of biodegradable food packaging when considering food waste. Journal of Cleaner Production 180: 325-334.

EPA [Environmental Protection Agency]. 2016. Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (WARM). Office of Resource Conservation and Recovery. EPA, Washington, D.C., USA.

EPA [Environmental Protection Agency]. 2019. Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model (WARM). Office of Resource Conservation and Recovery. EPA, Washington, D.C., USA.

EPA [Environmental Protection Agency]. 2020. Emission Factors for Greenhouse Gas Inventories. Greenhouse Gas Reporting Program. EPA, Washington, D.C., USA.

EPA [Environmental Protection Agency]. N.d. PFOA, PFOS and Other PFASs. Toxics Release Inventory Program. EPA, Washington, D.C., USA.

Google. N.d. Google Maps. <http://maps.google.com/>. Retrieved on 1 May 2020.

G.T. Internet Information. N.d. Clamshell Manufacturers in Taiwan. <http://www.manufacturers.com.tw/showroom-3700-1-1-0-0.php>. Retrieved on 6 May 2020.

Harnoto, M.F. 2013. A Comparative Life Cycle Assessment of Compostable and Reusable Takeout Clamshells at the University of California, Berkeley. <https://nature.berkeley.edu/classes/es196/projects/2013final/HarnotoM_2013.pdf. Retrieved on 17 April 2020.

Hekster, F. M., Laane, R. W., and P. de Voogt. 2003. Environmental and toxicity effects of perfluoroalkylated substances. In Reviews of Environmental Contamination and Toxicology 99-121.

Hottle, T. A., Bilec, M. M., and A.E. Landis. 2017. Biopolymer production and end of life comparisons using life cycle assessment. Resources, Conservation and Recycling 122, 295-306.

Huijbregts, M. A. J., Thissen, U., Jager, T., Van de Meent, D., and A.M.J. Ragas. 2000. Priority assessment of toxic substances in life cycle assessment. Part II: assessing parameter uncertainty and human variability in the calculation of toxicity potentials. Chemosphere 41(4) 575-588.

Ingrao, C., Gigli, M., and V. Siracusa. 2017. An attributional Life Cycle Assessment application experience to highlight environmental hotspots in the production of foamy polylactic acid trays for fresh-food packaging usage. Journal of Cleaner Production 150, 93-103.

Ingrao, C., Tricase, C., Cholewa-Wójcik, A., Kawecka, A., Rana, R., and V. Siracusa. 2015. Polylactic acid trays for fresh-food packaging: A Carbon Footprint assessment. Science of the Total Environment 537, 385-398.

ISO [International Organization for Standardization]. 2018. Determination of the ultimate aerobic biodegradability of plastic materials under controlled composting conditions. <https://www.iso.org/obp/ui/#iso:std:iso:14855:-2:ed-2:v1:en>. Retrieved on 29 March 2020.

Jamshidian, M., Tehrany, E. A., Imran, M., Jacquot, M., and S. Desobry. 2010. Poly-Lactic Acid: production, applications, nanocomposites, and release studies. Comprehensive reviews in food science and food safety 9(5), 552-571.

Johansson, C., Bras, J., Mondragon, I., Nechita, P., Plackett, D., Simon, P., and D. G. Svetec. 2012. Renewable fibers and bio-based materials for packaging applications–a review of recent developments. BioResources 7: 2506-2552.

Kale, G., Kijchavengkul, T., Auras, R., Rubino, M., Selke, S. E., and S.P. Singh. 2007. Compostability of Bioplastic Packaging Materials: An Overview. Macromolecular Bioscience 7: 255-277.

Karamanlioglu, M., Preziosi, R., and G.D. Robson. 2017. Abiotic and biotic environmental degradation of the bioplastic polymer poly (lactic acid): a review. Polymer Degradation and Stability 137, 122-130.

King, L., Brown, J., Miller, H., Kondo, C., Stapley, B., Ezaki, J., and L. McDowell. 2013. Getting to Zero Waste: Waste Audits. <http://tgif.berkeley.edu/overview/grant-cycles/2012-grant-awards/zero-waste/>. Retrieved on 6 May 2020.

Krause, M. J. and T.G. Townsend. 2016. Life-cycle assumptions of landfilled polylactic acid underpredict methane generation. Environmental Science & Technology Letters 3(4), 166-169.

Krueger, M., Kauertz, B., and A. Detzel. 2009. Life cycle assessment of food packaging made of Ingeo biopolymer and (r) PET. IFEU Report, Heidelberg.

Kuo, T. 2017. Workplace Conditions Assessment Report: PLA Deli Centers Factory Fair Labor Audit. <http://www.worldcentric.org/files/PLA%20Deli%20Cntrs%20Factory%20Fair%20Labor%20Audit%20Dec%202017.pdf>. Retrieved on 27 April 2020.

LBNL [Lawrence Berkeley National Laboratory]. 2020. Mini Waste Audit Waste Diversion. Sustainable Berkeley Lab. LBNL, Berkeley, CA, USA.

Luijsterburg, B. and H. Goossens. 2014. Assessment of plastic packaging waste: Material origin, methods, properties. Resources, Conservation and Recycling 85, 88-97.

Lundie, S., and G. M. Peters. 2005. Life cycle assessment of food waste management options. Journal of Cleaner Production 13(3), 275-286.

Madival, S., Auras, R., Singh, S. P., and R. Narayan. 2009. Assessment of the environmental profile of PLA, PET and PS clamshell containers using LCA methodology. Journal of Cleaner Production 17(13), 1183-1194.

Marsh, K., and B. Bugusu. 2007. Food packaging—roles, materials, and environmental issues. Journal of Food Science 72 (3): 39-55.

Marsolek, M. 2003. Carbon Footprint Analysis of StalkMarket Bagasse Tableware. Asean Corporation.

Morão, A. and F. de Bie. 2019. Life Cycle Impact Assessment of Polylactic Acid (PLA) Produced from Sugarcane in Thailand. Journal of Polymers and the Environment 27(11), 2523-2539.

Plastics Europe. 2019. Plastics – the Facts 2019. <https://www.plasticseurope.org/application/files/9715/7129/9584/FINAL_web_version_Plastics_the_facts2019_14102019.pdf>. Retrieved on 2 May 2020.

Rodríguez, L. J., Orrego, C. E., Ribeiro, I., and P. Peças. 2018. Life-cycle assessment and life-cycle cost study of banana (Musa sapientum) fiber biocomposite materials. Procedia CIRP 69, 585-590.

Rosenmai, A. K., Taxvig, C., Svingen, T., Trier, X., van Vugt-Lussenburg, B. M. A., Pedersen, M., and A.M. Vinggaard. 2016. Fluorinated alkyl substances and technical mixtures used in food paper-packaging exhibit endocrine-related activity in vitro. Andrology 4(4), 662-672.

Schaider, L.A., Balan, S.A., Blum, A., Andrews, D. Q., Strynar, M. J., Dickinson, M. E., Lunderberg, D. M., Lang, J. R., and G.F. Peaslee. 2017. Fluorinated Compounds in U.S. Fast Food Packaging. Environmental Science & Technology Letters 4(3), 105–111.

Schnurr, R. E., Alboiu, V., Chaudhary, M., Corbett, R. A., Quanz, M. E., Sankar, K., and T.R. Walker. 2018. Reducing marine pollution from single-use plastics (SUPs): A review. Marine Pollution Bulletin 137, 157-171.

Teuten, E. L., Saquing, J. M., Knappe, D. R., Barlaz, M. A., Jonsson, S., Björn, A., Rowland, S.J., and D. Ochi. 2009. Transport and release of chemicals from plastics to the environment and to wildlife. Philosophical Transactions of the Royal Society B: Biological Sciences 364(1526): 2027-2045.

Trinetta, V. 2016. Biodegradable Packaging. Food Science 1: 2-4.

U.S. DOT [United States Department of Transportation]. 2011. Freight Facts and Figures 2013. Freight Management and Operations. U.S. DOT, Washington, D.C., USA.

U.S. EIA [United States Energy Information Administration]. 2018. Frequently Asked Questions: Tools. Energy Information Administration. U.S. EIA, Washington, D.C., USA.

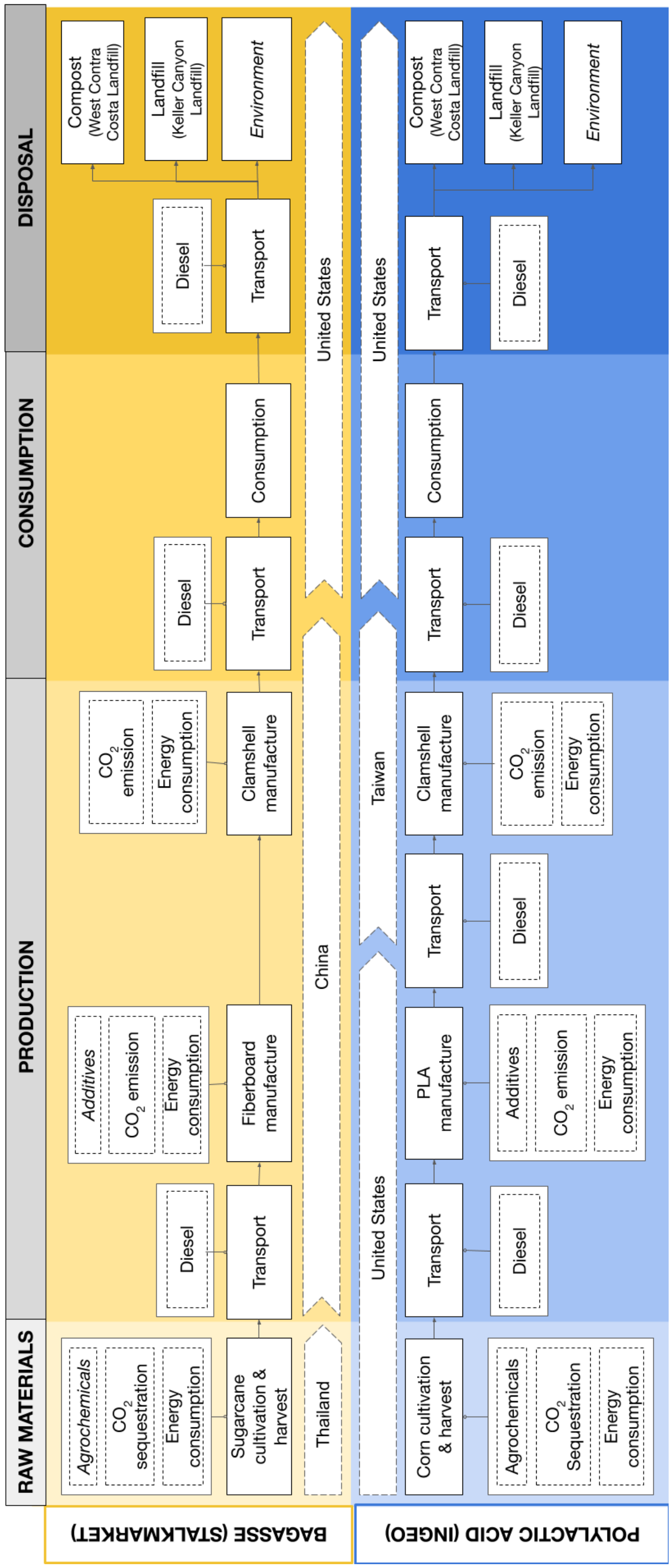
Vendries, J., Sauer, B., Hawkins, T. R., Allaway, D., Canepa, P., Rivin, J., and M. Mistry. 2020. The significance of environmental attributes as indicators of the life cycle environmental impacts of packaging and food service ware. Environmental Science & Technology.

Vink, E. T., Davies, S., and J.J. Kolstad. 2010. The eco-profile for current Ingeo® polylactide production. Industrial Biotechnology 6(4), 212-224.

World Centric. N.d. Contact. <http://www.worldcentric.org/about-us/contact>. Retrieved 6 May 2020.

Zabaniotou, A. and E. Kassidi. 2003. Life cycle assessment applied to egg packaging made from polystyrene and recycled paper. Journal of Cleaner Production 11(5): 549-559.

APPENDIX A: Comparative Life Cycle Definition



APPENDIX B: Transport Assumptions

Material	Life Cycle Phase	Description	Distance (km)*	Source
Bagasse	Production	Truck: sugarcane mill (Thailand) → fiberboard manufacture (Thailand)	30	Marsolek 2003
		Truck: fiberboard manufacture (Thailand) →Port of Thailand	440	Marsolek 2003
		Ocean freight: Port of Thailand → Yantian Port (Shenzhen, China)	3500	Harnoto 2013
		Truck: Yantian Port (Shenzhen, China) →clamshell manufacturer	499	Harnoto 2013
	Consumption	Truck: Clamshell manufacturer → Yantian Port (Shenzhen, China)	499	Harnoto 2013
		Ocean freight: Yantian Port (Shenzhen, China) →Port of Oakland (Oakland, CA)	11100	Harnoto 2013
		Truck: Port of Oakland (Oakland, CA) →Distributor (San Leandro, CA)	11.3	Harnoto 2013
		Truck: Distributor (San Leandro, CA)) →UC Berkeley (Berkeley, CA)	24	Harnoto 2013
	Disposal	Truck: UC Berkeley (Berkeley, CA) →West Contra Costa Landfill (Richmond, CA)	20.3	Cal Zero Waste n.d.
		Truck: UC Berkeley (Berkeley, CA) →Keller Canyon Landfill (Pittsburg, CA)	46.4	Cal Zero Waste n.d.
Polylactic Acid (PLA)	Production	Truck: corn farm (Midwest) →Ingeo factory (Blair, NE)	N/A	Vink et al. 2010
		Truck: Ingeo factory (Blair, NE) → Port of New Orleans (New Orleans, LA)	1704	U.S. DOT 2011
		Ocean freight: Port of New Orleans (New Orleans, LA) →Port of Taipei (Taipei, Taiwan)	9750	U.S. DOT 2011
		Truck: Port of Taipei (Taipei, Taiwan) →clamshell manufacturing factory	186	Kuo 2017; G.T. Internet

	(Changhua County, Taiwan)		Information n.d.
Consumption	Truck: clamshell manufacturing factory (Changhua County, Taiwan) →Port of Taipei (Taipei, Taiwan)	186	Kuo 2017; G.T. Internet Information n.d.
	Ocean freight: Port of Taipei (Taipei, Taiwan) →Port of Oakland (Oakland, CA)	6750	U.S. DOT 2011
	Port of Oakland (Oakland, CA) → Distributor (San Leandro, CA)	16.9	U.S. DOT 2011; World Centric n.d.
	Distributor (San Leandro, CA) →UC Berkeley (Berkeley, CA)	24.8	World Centric n.d.
Disposal	Truck: UC Berkeley (Berkeley, CA) →West Contra Costa Landfill (Richmond, CA)	20.3	Cal Zero Waste n.d.
	Truck: UC Berkeley (Berkeley, CA) →Keller Canyon Landfill (Pittsburg, CA)	46.4	Cal Zero Waste n.d.

* Distances estimated using Google Maps when magnitudes not offered in literature (Google n.d.).

APPENDIX C: Greenhouse Gas Emissions Data from Literature

		Bagasse		PLA	
Phase	**Stage**	**Emissions** $\left(\frac{kg\ CO_2\ eq}{kg\ bagasse}\right)$	**Source**	**Emissions** $\left(\frac{kg\ CO_2\ eq}{kg\ PLA}\right)$	**Source**
Raw Material Acquisition	Carbon sequestration	-1.794	Marsolek 2003		
	Feedstock cultivation, harvest	0.036	Marsolek 2003	-1.940	Vink et al. 2010
Production	Material manufacture	5.419	Marsolek 2003	2.739	Vink et al. 2010
	Clamshell manufacture	3.702	Marsolek 2003	5.599	Madival et al. 2009
	Transport (truck)	0.279	Appendix B	0.294	Appendix B
	Transport (ocean freight)	0.095	Appendix B	0.267	Appendix B
Consumption	Transport (truck)	0.076	Appendix B	0.006	Appendix B
	Transport (ocean freight)	0.304	Appendix B	0.185	Appendix B
Disposal	Scenario I: Transport	0.007	Appendix B	0.007	Appendix B
	Scenario I: Emissions	-0.0000003	EPA 2016	-0.00000183	EPA 2016
	Scenario II: Transport	0.011	Appendix B	0.011	Appendix B
	Scenario II: Emissions	-0.0000002	EPA 2016	-0.00000145	EPA 2016
	Scenario III: Transport	0.024	Appendix B	0.024	Appendix B
	Scenario III: Emissions	0.0000000	EPA 2016	-0.000001448	EPA 2016

	Scenario IV: Transport	0.003	Appendix B	0.003	Appendix B
	Scenario IV: Emissions	0.0000001	EPA 2016	0.00000008	EPA 2016
TOTALS	Scenario I	8.125	kg CO2 / kg bagasse	7.157	kg CO2 / kg PLA
	Scenario II	9.930	kg CO2 / kg bagasse	9.109	kg CO2 / kg PLA
	Scenario III	9.907	kg CO2 / kg bagasse	6.383	kg CO2 / kg PLA
	Scenario IV	8.121	kg CO2 / kg bagasse	7.154	kg CO2 / kg PLA

APPENDIX D: Energy Consumption Data from Literature

		Bagasse		**PLA**	
Phase	**Step**	**Energy consumption** $\left(\frac{MJ}{kg\ bagasse}\right)$	Source	**Energy consumption** $\left(\frac{MJ}{kg\ PLA}\right)$	Source
Raw Material Acquisition	Feedstock cultivation, harvest	1.083	Harnoto 2013	67.3131	Vink et al. 2010
Production	Material manufacture	0.238	Marsolek 2003	0.5847	Vink et al. 2010
	Clamshell manufacture	9.595	Marsolek 2003	759	Madival et al. 2009
	Transport (truck)	0.019	Appendix B	0.0204	Appendix B
	Transport (ocean freight)	0.007	Appendix B	0.0185	Appendix B
Consumption	Transport (truck)	0.005	Appendix B	0.000432	Appendix B
	Transport (ocean freight)	0.021	Appendix B	0.0128	Appendix B
Disposal	Scenario I: Transport	0.000455	Appendix B	0.000456	Appendix B
	Scenario I: Facility	-0.000000136	EPA 2016	0	EPA 2016
	Scenario II: Transport	0.000764	Appendix B	0.000764	Appendix B
	Scenario II: Facility	0.0000001834	EPA 2016	0.0000000459	EPA 2016
	Scenario III: Transport	0.00169	Appendix B	0.00169	Appendix B
	Scenario III: Facility	0.000000230	EPA 2016	0.000000184	EPA 2016
	Scenario IV: Transport	0.000199	Appendix B	0.000199	Appendix B

	Scenario IV: Facility	0.000000230	EPA 2016	0.000000230	EPA 2016
TOTALS	Scenario I	10.969	MJ / kg bagasse	826.950	MJ / kg PLA
	Scenario II	10.970	MJ / kg bagasse	826.952	MJ / kg PLA
	Scenario III	10.970	MJ / kg bagasse	826.952	MJ / kg PLA
	Scenario IV	10.969	MJ / kg bagasse	826.950	MJ / kg PLA