Environmental impact of biodegradable food packaging when considering food waste

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ABSTRACT

From a waste management perspective, high-barrier, multi-layer, biodegradable food packaging could be a useful replacement for current multi-layered packaging that is non-recyclable and non-degradable. Whilst there is still technical research required, it is envisioned that a biodegradable thermoplastic starch (TPS) and polyhydroxyalkanoate (PHA) layered material could be a promising target. However, there is currently limited research identifying what environmental trade-offs are associated with using such a material – meaning there is no guidance regarding what design characteristics are important to consider during development of such packaging. The aim of this study was to quantify the greenhouse gas (GHG) trade-offs associated with using the proposed biodegradable packaging and identify the important design considerations. To our knowledge this is the first study to discuss the implications of including food wastage when assessing biodegradable food packaging materials. It also considers the impacts of landfill methane capture efficiency, which is an important aspect as biodegradable packaging may release methane when disposed of in a landfill whereas non-biodegradable packaging is inert. However, a key result is that when food waste is included in the system boundaries, it contributes over 50% of the GHG emissions associated with the system, regardless of whether the package is biodegradable or not. This shows that even for biodegradable packaging, reducing food waste is a key design consideration. In fact, the negative environmental impacts associated with disposal of a PHA-TPS packaging in landfill with low gas capture rates can actually be offset if the package reduces food wastage (beef) by approximately 6%. The overarching result is that a PHA-TPS food packaging only delivers positive GHG outcomes if it reduces food wastage or increases the viability of biological food waste processing.

1. Introduction

Whilst recycling is viewed as the primary mechanism to reduce the environmental and waste management issues associated with the use of plastics, 30% of plastic packaging materials may never be eligible for recycling or reuse without fundamental redesign of the materials used (Ellen MacArthur Foundation, 2017). This is due to a variety of reasons. Some niche materials are used in too small a quantity to justify the recycling infrastructure, some plastic products are considered too small for practical sorting, and some packaging products are prone to being contaminated with organics (particularly food) or chemicals (Ellen MacArthur Foundation, 2017).

Another category of materials that pose a particular challenge for recycling, are those that comprise a combination of different plastic types – whether that be blended plastics or multi-layered materials (with a different plastic type in each layer). In either case, the cost and technology constraints of separating and recycling the different plastic polymers can be prohibitive and alternative strategies are required to address the waste management challenges associated with plastic disposal (Barlow and Morgan, 2013; Ellen MacArthur Foundation, 2016). On land, the focus of materials redesign and waste management is to reduce the reliance...
on landfilling for waste disposal, as well as reduce the amount of plastic that is disposed of in open-dumps and on the streets in developing countries (Hoornweg and Bhada-Tata, 2012; Hopewell et al., 2009). This will then help to reduce the amount of plastic waste that enters the oceans. It has been estimated that ~30% of all packaging is not disposed of appropriately, and thus has the potential to accumulate in the world’s oceans (Ellen MacArthur Foundation, 2016). Given this, it is also important that materials design focuses on reducing the marine life impact of the plastics that inevitably make their way to the oceans.

Food packaging is a particularly problematic part of the global challenge with plastics waste management as it represents the largest demand for plastic packaging (PlasticsEurope, 2016) and comprises a large portion of the objects identified in coastal surveys (Andrady, 2015). This seems likely to increase, as urbanisation and dietary change in developing countries is leading to an increased global reliance on processed foods (World Packaging Organisation, 2008). Improved recycling systems can only go so far to solving this problem, in part because of food contamination, and in part because the food industry is increasing its use of multi-layered packaging materials. This is because multi-layered materials can provide a much higher resistance to water and gas transfer than single-layered materials, and thus reduce food spoilage (Barlow and Morgan, 2013).

Given that materials redesign has a useful role to play for food packaging, a multi-layered combination of thermoplastic starch (TPS) and polyhydroxyalkanoate (PHA) could warrant inclusion in the list of materials being considered (Fabra et al., 2016; Shogren, 1997). TPS is one of the best oxygen barriers of all polymeric materials (Krotha and De Mulder-Johnston, 1997), while it has been shown that PHA exhibits the best water barrier properties of biodegradable polymers (Shogren, 1997). The combination might therefore offer impressive barrier properties, with the potential to lower food spoilage rates compared to more conventional packaging materials.

The biodegradation characteristics of a PHA-TPS combination also seem promising. The appeal of biodegradable food packaging is that it might broaden the waste management options for materials that can’t easily be recycled (by including biological processing). However, while there are a number of plastic polymers that exhibit high biodegradation rates in landfill or composting, TPS and PHA are two of the very few materials that also show potential to biodegrade in sea-water (O’Brine and Thompson, 2010; Volova et al., 2010).

A mixed-layer PHA-TPS material for food packaging might therefore be rather unique in its potential to both reduce food spoilage rates and alleviate the marine pollution impacts caused by recalculating plastics making their way to the ocean. The PHA-TPS combination has been successfully formulated at lab-scale using a number of different techniques (Fabra et al., 2016; Martin et al., 2001), although further research is required to better understand its food preservation and marine-degradation performance.

The potential PHA-TPS combination also offers an intriguing mix of greenhouse gas (GHG) implications in a carbon-constrained world. Focusing just on the production processes, some studies suggest the energy demand to produce biopolymers can be higher than for conventional products, but the overall carbon footprint can be lower depending on assumptions about the biological feedstock used (Hottle et al., 2013; Yates and Barlow, 2013). Furthermore, biodegradability is not always viewed in the literature as a GHG benefit, depending on the assumptions used for degradation rates and waste handling systems (Yates and Barlow, 2013). Importantly, while most carbon-footprint studies of food packaging pay no attention to the food itself, the few that do consider the role of packaging choice on food preservation show that this could be the single biggest influence on overall GHG results (Conte et al., 2015; Williams and Wikström, 2011).

Here, the aim was to identify those design characteristics most useful for minimising environmental tradeoffs associated with a multi-layered PHA-TPS food packaging material. The primary focus for this paper is on GHG implications, although life-cycle water demand is also considered, given the potential importance of food production to the conclusions. The analysis covers the full life-cycle of the biodegradable packaging and the food it contains. To our knowledge, this is the first study to discuss the implications of including food wastage when assessing biodegradable food packaging materials. The study also considers the impacts of landfill methane capture efficiency.

Two very different food products (beef and cheese) that would likely benefit from a packaging material with improved water and oxygen barrier properties were considered (Barlow and Morgan, 2013). Sensitivity analysis was used to consider potential variation in those parameters found to be influential in previous literature (Clune et al., 2017; Yates and Barlow, 2013). A key outcome is that food waste is shown to contribute at least 50% of the GHG emissions in a food packaging life-cycle analysis regardless of whether the package is biodegradable or not. Also, that the difference in the relative impact of the two types of packaging are minimal compared to the overall, particularly when including food waste in the calculation. As a consequence, reducing food waste is a key design consideration for PHA-TPS food packaging. Such packaging was only found to deliver positive GHG outcomes if it reduced food wastage or contributed to building the viability of biological food waste processing (e.g. anaerobic digestion). Reducing food wastage also has the potential to offset the emission released from biodegradable packaging in inefficient landfills.

2. Methods

2.1. Life-cycle assessment

Life-cycle assessment (LCA) is the most popular tool for estimating the environmental impact of a product or process throughout its entire lifecycle (Reap et al., 2008) and guidelines for its implementation are defined by the International Organisation for Standardisation 14040 series (ISO, 2006). An LCA involves collating information on the inputs and outputs of a system and subsequently converting these to environmental consequences (such as global warming potential). It was thus selected as the tool to explore the research question. Although not a perfect tool (e.g. it is unable to capture the marine impacts of plastic litter), nor able to provide definitive answers, it is a transparent way in which to assess specific environmental tradeoffs (Schnoor, 2009).

2.2. Literature review

In food packaging LCAs, the default has been to analyse the packaging in isolation, overlooking the implications of the food system (Madival et al., 2009; Toniolo et al., 2013). However, there is a movement towards including the function of the package (i.e. delivering food to the consumer) and associated indirect effects (e.g. food waste) within the system boundaries (Wikström et al., 2016). Results of such studies have demonstrated the importance of using this expanded system boundary to reduce total environmental impact when considering development of alternate packaging materials (Williams and Wikström, 2011). In particular, Conte et al. (2015) assessed single layer and multi-layer conventional packaging and showed that multi-layer materials emerge as environmentally superior only when food waste is included in the
system boundaries. To further develop the idea, Grant et al. (2015) have developed a packaging design tool that includes product loss in the assessment of packaging materials. Whilst the impact of food waste in a packaging LCA has been documented, a biodegradable packaging option has not been considered. Thus, the expanded LCA methodology was selected for this study.

2.3. Methodology of the study

2.3.1. Impact factors

Climate change and water use were calculated as the impact factors. The study was limited to these impact factors as there was not sufficient data available over the entire scope of the analysis for the consideration of other impact factors.

2.3.1.1. Global warming potential (GWP). Global warming potential using a 100-year time horizon (GWP100) was used as the default impact factor across the entire study. However, the need to check the sensitivity of an LCA’s conclusions to different modeling choices is discussed by Levasseur et al. (2016). It was for this reason that a global warming potential using a twenty-year time horizon (GWP20) was also calculated for selected scenarios to consider impacts in the nearer term. Methane is a short-lived GHG, and because of this using a twenty year time horizon places greater emphasis on its emissions and its associated global warming potential than using a 100-year time horizon (Levasseur et al., 2016). The GWP is reported as CO₂-equivalent (CO₂e) emissions.

2.3.1.2. Water use. There is a considerable amount of water use associated with food production (Mekonnen and Hoekstra, 2010). Thus, a water use analysis was conducted for the food and packaging system as it was predicted that it would be an important factor when considering the potential sustainability impact of a package in the context of the food system. Water use was added on a volumetric basis across the lifecycle from available inventory flows for water inputs from water storages, rivers and lakes. Rainwater inputs to agriculture were not included and no account was included for water stress in the regions where water was taken from. Water stress was not included in an attempt to keep the results independent of location.

2.3.2. Scope and functional unit

The entire system of a hypothetical supply chain from production of the raw materials to end-of-life (EOL) was considered for the functional unit ‘1 kg of packaged product at the house’ as presented in Fig. 1. Mass flows for the figure are recorded in Table 1. Production of the polymers occurred in China, with subsequent transport to Victoria (Australia) where the package is produced and combined with a food product, followed by transport to Queensland (Australia) where the product is sold in a supermarket, consumed at the house and the waste disposed of.

The influence on the environmental outcomes of changing packaging types, food types, waste disposal scenarios and food wastage levels was explored.

2.3.2.1. Packaging types. Two types of food packages were compared, the proposed biodegradable PHA-TPS layered material and the most commonly used meat packaging material, polypropylene (PP) (based on a visual survey in local shopping centres).

PHAs are produced through bacterial fermentation of a carbon source under limited nutrient conditions with intracellular accumulation of the polymer. This can then be purified, usually through solvent extraction (Jiang and Zhang, 2013). The carbon sources are most commonly sugars derived from an agricultural feedstock (Braunegg et al., 1999) although the use of other feedstocks has also been explored (Khardenavis et al., 2005; Strong et al., 2016). Production of PHA from sucrose (from sugar-cane) was used as the default method in this study following the inventory of Harding et al. (2007). Thermostatic starch is produced through treatment of native starch with either heat or shear in the presence of plasticisers (Jiang and Zhang, 2013). The starch can come from a variety of sources including maize, wheat, potato and cassava. Production of thermostatic starch from maize starch and glycerol was used as the default method in this study. PP is produced by from crude oil via naphtha (PlasticsEurope, 2014).

2.3.2.2. Food types. Beef and cheese were selected as model food types as they are typical food types that require high-barrier packaging to prevent food spoilage. Cheese represents a mid-range impact product for emissions associated with food production and beef represents a high-range impact product (Clune et al., 2017). Australian production was modeled for both food types, with the CO₂ emissions associated with production close to the world mean in both cases (Clune et al., 2017). Further details are provided in Table 2. Only the environmental impact due to the production, transportation and storage of the food WASTE associated with the
Table 1
Mass flows associated with Fig. 1.

<table>
<thead>
<tr>
<th>Material flows as per Figure 1</th>
<th>Beef</th>
<th>Cheese</th>
</tr>
</thead>
<tbody>
<tr>
<td>Packaging (g)</td>
<td>Food (g)</td>
<td>Total (g)</td>
</tr>
<tr>
<td>Wasted</td>
<td>Consumed</td>
<td>Wasted</td>
</tr>
</tbody>
</table>
| P1 | 11.5 | 27.6 | 11.5 | 27.6 | 59.8 | Note: The remainder of the total of the biopolymer packaging is water (15% of 89%)
| P2 | 57.7 | 878.8 | 57.7 | 878.8 | 902.5 |
| A | 57.7 | 58.0 | 0 | Packaging (beef or cheese) = B + 5% wastage |
| B | 54.8 | 55.3 | 0 | |
| C | 52.1 | 110.8 | 878.8 | 52.4 | 92.4 | 902.5 |
| D (functional unit) | 50 | 71.2 | 878.8 | 1000 | 50 | 47.5 | 902.5 | 1000 |

Mass balance In: 368.5, 150.3

Mass balance Out: 368.5, 150.3

Table 2
Inventory inputs and GWP100 results for the modeled system.

<table>
<thead>
<tr>
<th>Process</th>
<th>Description</th>
<th>kg CO2e/kg processed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Production of the packaging</td>
<td>PHA-TPS: The production of PHA from maize was based on data published by Harding et al. (2007). Thermoplastic starch production was from maize starch, glycerol and water. Maize and glycerol production processes were from Ecoinvent.</td>
<td>3.35</td>
</tr>
<tr>
<td>Production of the food</td>
<td>PP: Polypropylene granulate production and extrusion process were from Ecoinvent. Beef: The inventory for cattle production was from personal correspondence. Water use was taken from the report by Mekonnen and Hoekstra (2010). Cheese: The inventory for milk production was from a Dairy Australia report (Dairy Australia, 2012). Water use was taken from the report by Mekonnen and Hoekstra (2010). A model was created to represent the inputs for production of the packaged product.</td>
<td>3.41</td>
</tr>
<tr>
<td>Produced waste</td>
<td>Waste was taken as 5% packaging waste and 0% food waste. Storage at the supermarket: Food waste at the store was modeled as 4% meat waste or 4.5% dairy waste according to a report by the European Commission (European Commission, 2015). Storage at the house: Food waste at the house was modeled as 7.5% meat waste or 5% dairy waste according to a report by the European Commission (European Commission, 2015). Waste processing</td>
<td>0.133</td>
</tr>
<tr>
<td>PP: The Ecoinvent process for disposal of PP to a sanitary landfill was used. PHA-TPS: Landfill: The AusLCI process for food waste in landfill was used as the model with minor adjustments. Anaerobic Digestion: The AusLCI process for AD of food waste was used as a model with the production of compost and methane. Methane combustion was assumed to offset electricity whilst compost use offset fertilizer application. Composting: The AusLCI process for aerobic composting of food waste was used as the model and compost use offset fertilizer application.</td>
<td>Landfill: 2.13</td>
<td></td>
</tr>
<tr>
<td>AD: –1.86</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Composting: –0.0337</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Food: Landfill: The AusLCI process for food waste in landfill was used as the model. Anaerobic Digestion: The AusLCI process for AD of food waste was used as a model with the production of compost and methane. Methane combustion was assumed to offset electricity whilst compost use offset fertilizer application. Composting: The AusLCI process for aerobic composting of food waste was used as the model and compost use offset fertilizer application.</td>
<td>Landfill: 1.29</td>
<td></td>
</tr>
<tr>
<td>AD: –0.119</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Composting: –0.0591</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
functional unit was included (this waste can occur at the supermarket or the house) as this is the portion of food production which packaging can actually influence.

2.3.2.3 Waste disposal scenarios. Landfill, anaerobic digestion and composting were considered as waste disposal scenarios for the PHA-TPS package and the food so that the benefits of biological waste disposal options could be explored. It was assumed that landfill was the only waste disposal option for PP due to a range of barriers still slowing the uptake of recycled materials in food contact applications (Australian Packaging Covenant, 2014). Incineration was not considered, as it is not a common practice in Australia (Randell et al., 2013).

2.3.2.3 Food wastage levels. It was assumed that the packaging could play a role in increasing the shelf life of the food through providing improved barrier properties. This was assumed to lead to a decrease in food wastage levels at both the supermarket and the house. However, there is no specific numerical relationship that describes how increased shelf life translates into reduced food wastage.

2.4. Inventory inputs

Mass flows and inventory inputs were collected for each section of the supply chain from the databases Ecoinvent (version 3.2), AusLCI (version 1.26) and AustralasianLCI (version 2011.8) as well as a variety of publications, reports and personal correspondences. Energy use was adjusted to reflect the Australian grid for all back-ground processes taken from Ecoinvent. Simplified descriptions of the inputs for the modeled system as well as selected GWP100 results (for comparative purposes) are presented in Table 2 (A more detailed description of the inventory inputs can be found in the supplementary information). Transport was included for all processes and descriptions for the vehicle models are presented in Table 3 whilst distances are shown in Fig. 1. All transport distances were estimated from Google Maps.

2.5. Sensitivity inputs

Low and high range scenarios for the emissions associated with production of the packaging, production of the food and the waste processing were included for sensitivity analysis and are presented in Table 4. The results obtained when using these alternative scenarios are then presented as ranges in the main results.

2.5.1. Production of the packaging

Both PHA and starch can be produced from a variety of different sources as described in section 2.3.2.1. This can introduce a large variability in the GHG emissions associated with production. Furthermore, choices concerning farming methods, production technologies, energy use, and energy source can also influence the environmental outcome (Yates and Barlow, 2013).

PHA: Based on a review of biopolymer LCAs by Yates and Barlow (2013), the lowest and highest GWP100 values were selected to demonstrate the possible range. The PHA production modeled in the main system corresponded to mid-range when compared to this review. The most extreme range was associated with genetically modified plant production of the polymer, with environmental impacts varying based on the type of electricity input used (biomass vs. coal) (Kurdikar et al., 2000).

Starch: Fewer reports have been published on the LCAs of pure starch materials and the range recorded is much smaller than for other biopolymers (Hottle et al., 2013; Yates and Barlow, 2013). Due to the lack of literature, the range for TPS production was determined from the range of starch production processes present in Ecoinvent.

2.5.2. Production of the food

The environmental impact of food production can vary

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**Table 3**

<table>
<thead>
<tr>
<th>Transport Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Non-refrigerated ground transport</td>
<td>A model for articulated truck freight with 100% backhaul, 90% rural driving from AusLCI was used.</td>
</tr>
<tr>
<td>Ground transport</td>
<td>A model for articulated truck freight with 100% backhaul, 90% rural driving from AusLCI was modified by including additional fuel consumption for refrigeration of 2.875 L/h.</td>
</tr>
<tr>
<td>Ocean transport</td>
<td>A model for transoceanic freight ship from AusLCI was used.</td>
</tr>
<tr>
<td>Personal transport</td>
<td>A model from AusLCI for the average fuel use of a car per km travelled was used with 30% of the trip to the store allocated to alternative activities.</td>
</tr>
<tr>
<td>Waste transport</td>
<td>A model for municipal waste collection service from Ecoinvent was used.</td>
</tr>
</tbody>
</table>

**Table 4**

<table>
<thead>
<tr>
<th>Process</th>
<th>Parameter</th>
<th>Results from the LCA (kg CO₂e/kg processed)</th>
<th>High alternative value (kg CO₂e/kg processed)</th>
<th>Low alternative value (kg CO₂e/kg processed)</th>
<th>Reference for alternative values</th>
</tr>
</thead>
<tbody>
<tr>
<td>Production of the packaging</td>
<td>Emissions: PHA production process</td>
<td>2.87</td>
<td>5.7</td>
<td>-4.0</td>
<td>Yates and Barlow (2013), Harding et al. (2007) and AusLCI</td>
</tr>
<tr>
<td></td>
<td>Emissions: TPS production process</td>
<td>1.35</td>
<td>2.31</td>
<td>1.24</td>
<td>Rossi et al. (2015) and Ecoinvent.</td>
</tr>
<tr>
<td>Production of the wasted food</td>
<td>Emissions: beef production</td>
<td>26.0</td>
<td>31.6 (upper quartile value)</td>
<td>22.26 (Lower quartile value)</td>
<td>Clune et al. (2017)</td>
</tr>
<tr>
<td>Waste processing</td>
<td>AD: Compost offsets</td>
<td>340 (food), 150 (biopolymer)</td>
<td>N/A</td>
<td>0</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Compost: Compost offsets</td>
<td>450 (food), 420 (biopolymer)</td>
<td>N/A</td>
<td>0</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Landfill: gas capture efficiency</td>
<td>0</td>
<td>0.97</td>
<td>N/A</td>
<td>Schuz et al. (2009)</td>
</tr>
</tbody>
</table>
substantially due to differences in either physical factors such as geography and production methods or methodological factors such as system boundaries and allocation method (Clune et al., 2017). Based on a comprehensive meta-analysis by Clune et al. (2017) (which collated all the reported GHG footprints of production for different food types) a range for the impact of food production was tested. The upper quartile and lower quartile values were used as representation of the most probable range.

2.5.3. Waste processing

The influence on the results due to disposal of the packaged product in either a sub-optimally functioning composting or AD facility, or state-of-the-art landfill facility were explored. To achieve this, offsets (i.e. compost and electricity generation) were removed from composting and AD scenarios, whilst 97% methane capture (Schuetz et al., 2009) was modeled for the landfill. These extreme scenarios, although perhaps unrealistic, were selected so that ‘what-if’ situations could be understood. For example, it allows an understanding of whether landfilling is an inherently poor option for biodegradable plastics (from a GHG emissions perspective) or whether it would be on par with AD if gas capture infrastructure could be improved. These extreme scenarios also allow for a visualisation of the entire range of results.

3. Results and discussion

3.1. Comparison of packaging materials (not including food impacts)

Firstly, the CO$_2$e emissions of the different food packaging materials in isolation (no food included in the system boundary) were considered so that different contributors within packaging production and disposal could be analysed (Fig. 2). The results of this study fall within the range of life-cycle impacts for biodegradable polymers reported in the literature (Hottle et al., 2013). The production of the polymers emerges as a significant GHG contributor relative to polymer processing and distribution, although there is little difference between the production of a PP package and the alternative biodegradable package. Of note though, as indicated by the range bars which represent the emissions associated with alternative starch and PHA production, the Australian average is 30% methane capture efficiency (Australian Greenhouse Office Department of the Environment and Heritage, 2011). These results indicate the large variability in environmental impact biodegradable plastics will have depending on methane capture rates during waste processing. Transport (excluding personal car transport) was a minor part of the life-cycle emissions with fuel use allocated on a mass basis.

3.2. Including food waste in the system boundary

Whilst it is important to understand the specifics with regards to the production of the plastic packaging, it is also important to understand its impact in the context of the food packaged within it. As shown in Fig. 3, once the wasted food is included in the system boundary, the impact of the production of the packaging becomes a small part of the impact of the total system. This finding has been reported in previous studies (Conte et al., 2015; Williams and Wikström, 2011), and the considerable impact of food production and wastage has also been noted for other indicators such as acidification and eutrophication (Silvenius et al., 2014). Calculating either GWP100 or GWP20 delivers the same message, although it is more pronounced when using a twenty-year time horizon. Due to methane emissions associated with both food production and waste disposal in landfill, these sectors show an increase in the magnitude of the impact relative to the other aspects of the system when moving to a shorter time horizon. Fig. 3(A) shows a beef scenario with disposal in landfill whilst...
3.3. Considering waste management options

Biodegradable packaging has an inherent value compared to non-biodegradable packaging as it presents an opportunity to direct food away from landfill (Razza and Innocenti, 2012). This is important as it has been reported that the waste sector accounts for 3% of total GHG emissions with food waste in landfills (DEFRA, 2011). Anaerobic digestion and composting are considered to be the best options for food waste disposal (Eriksson et al., 2015). The greatest value for this would be in situations where there is a higher proportion of unopened, packaged food that needs to be disposed of, such as from supermarkets. Legislation has been introduced to encourage more effective waste management, with countries such as Sweden setting environmental targets for 50% of food waste from supermarkets and bulk food preparation to be biologically treated by 2018 (Eriksson et al., 2015).

Thus, the implications of different waste handling technologies were considered and are presented in Fig. 4 (with the assumption that the alternatives can only be accessed for the biodegradable packaging). Results are similar for the cheese scenarios and are presented in the supplementary information (Fig. S1). Each of the alternative options are benchmarked to the emissions from ‘1 kg of PP packaged product at the house, disposed of in landfill with 0% methane capture’ and the results are shown as percentage differences. All of the scenarios perform better than the benchmark but it is interesting to note that the two most commonly proposed alternatives for food waste handling (composting and AD) do not perform better than recovering full methane from landfill (although, as this is just a GHG focused study other factors such as nutrient recovery have not been considered, also, as noted previously, 97% methane capture in a landfill is very optimistic). However, this demonstrates that it is avoidance of methane emissions that is important, not necessarily the specific way in which this is achieved.

The breakdown within the AD and composting scenarios show that a majority of the benefit of these alternative processes is derived from diverting food waste from a 0% methane capture landfill. Brancoli et al. (2017) recently reported that if food is disposed of in its packaging from supermarkets to an AD facility, up to 40% by mass of this food will then be rejected along with the plastic fraction. This sort of scenario could be avoided if the disposed food was packed in biodegradable packaging, leading to a reduction in GHG emissions from the waste sector.

3.4. Considering food waste prevention

Redirecting inevitable food waste to a more appropriate waste management system is important but there is general agreement that prevention of food waste should be a top priority when designing sustainable food packages (Grönnman et al., 2013). This redesign also carries an economic incentive for food distributors. There are a variety of behavioral factors that would play the central role in reducing food waste, but as discussed, packaging can play a role through limiting the exposure of the food to oxygen and water.

Figs. 5 and 6 present the associated difference in GWP100 for a variety of beef waste scenarios and disposal scenarios. In Fig. 5 the scenario is benchmarked relative to ‘a PP package disposed of in landfill with 0% methane recovery’ whilst in Fig. 6 the scenarios are benchmarked to ‘a PP package disposed of in landfill with 97% methane recovery’. Fig. 5 indicates that when landfill methane is not captured, the GHG emissions of the biodegradable scenario are ~7% higher than a PP scenario when food wastage is considered to be equal. However, if the package can reduce food wastage by ~6% (through reducing food spoilage) the emissions associated with the biodegradable package in landfill can be negated. It is therefore important, not necessarily the specific way in which this is achieved.
possible that the GHG benefits in reducing food spoilage could more than outweigh the GHG burdens of making and disposing of a PHA-TPS food-packaging product for beef, even with disposal in a 0% methane capture landfill. When using a twenty-year time horizon (results presented in supplementary information, Fig. S2) a slightly greater reduction in food wastage is required to overcome the impact of packaging in landfill (~8%).

On the other hand, as presented in Fig. 6, even under conditions of maximum methane recovery, if a biodegradable packaging does not match the functional performance of a PP package, a small increase in food waste can negate any benefits obtained by disposal in optimised waste management infrastructure.

The combination of these results demonstrates that if all methane emissions associated with waste management can be captured, AND food wastage can be reduced then the reduction in GHG emissions of using a PHA-TPS material relative to a PP material can be maximised. A 15% increase in the shelf-life of beef has been reported for other starch-based packaging materials compared to traditional systems (Cooper, 2013), although it is not exactly known what percentage reduction in food waste this correlates to. The benefit of a biodegradable package is also that it enables more flexibility in regards to waste processing options.

Results for a cheese scenario, presented in Fig. S3 in the supplementary information, show similar trends, however, because cheese has a lower GWP associated with its production, a greater reduction in food waste would be required to overcome the emissions of biodegradable plastic in landfill. In this case alternative waste management systems are of greater importance.

### 3.5. Considering potential impact

Further to the results presented, the opportunities should be considered in light of relative consumption levels of the food products. In Australia, approximately 580,000 tonnes of beef (Meat and Livestock Australia, 2016) and 260,000 tonnes of cheese (Dairy Australia, 2017) are domestically produced and consumed per year. Given that more beef than cheese is consumed, and also that it has a higher wastage rate and carbon footprint of production, it appears that beef would be the more immediate focus for high-barrier, biodegradable packaging applications. However, to provide quantitative conclusions as to the total potential impact of the new packaging material, the following parameters would need to be known:

- The frequency of each food type being disposed of in packaging;
- What percentage of food is wasted due to reaching its ‘use-by-date’;
- What the quantitative relationship is between increased food lifetime and reduced food wastage.

Confirmation of the PHA-TPS packaging’s ability to reduce food spoilage is also required.

### 3.6. Water use

Although the initial scope was limited to GHG emissions, it is useful to know whether considering other resources common to both the food and packaging sectors would change the conclusions of the study. The results for a water use analysis are presented in Fig. 7. Approximately 60% more water is required to produce the PHA-TPS packaging when compared to a PP packaging, however, similar to the GHG analysis, this is dwarfed by the water use for the production of the wasted food. Thus, the change in the consumption of water in producing the packaging is minor, but any reductions in food wastage could substantially reduce the water use of the system. This supports the original conclusion that reducing food waste should be a key consideration in biodegradable packaging design.

![Fig. 6. Kg CO2e difference (GWP100) for a variety of beef wastage scenarios with disposal in a 97% methane capture landfill or AD. The results are calculated relative to a PP beef package disposed of in landfill with 97% methane capture.](image1)

![Fig. 7. Water use by sector for the full system boundary. *Only the impact of the production and disposal of the wasted food associated with 1 kg of the packaged food product at the house is included in the system boundary.](image2)
4. Conclusion

On a basic level, the results of this work show that the main differences in GHG emissions between a PP and PHA/TPS food packaging are due to landfill emissions. However, this only holds true until food waste is included in the system boundary, at which point differences become dominated by changes in the quantity of food waste. This then leads to three main insights. Firstly that food packaging design needs to focus on the reduction of food waste (e.g. focus on high barrier properties), even if a biodegradable material is used. Secondly, that the GHG emissions associated with disposal of a PHA-TPS packaging in landfill can be offset if the package reduces beef wastage by approximately 6% (demonstrating the viability of a high-performing biodegradable packaging providing GHG benefits even if disposed of in landfill). Thirdly, that a biodegradable packaging could provide GHG benefits through increasing the amount of food waste available for biological processing (e.g. anaerobic digestion with subsequent biogas processing). As a final note and word of caution, it should be acknowledged that whilst this LCA has provided some interesting comparisons between biodegradable and conventional plastic packaging, it can only tell a small part of the story. LCA is a useful tool but is currently not configured to quantify many of the important environmental impacts associated with plastics, particularly accumulation in the oceans.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jclepro.2018.01.169.

References


