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Comparative life cycle analysis of a biodegradable multilayer film and a conventional multilayer film for fresh meat modified atmosphere packaging – and effectively accounting for shelf-life

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ABSTRACT

Life cycle analyses of novel food packaging materials do not often account for the environmental impact of a change in shelf-life, which can result in misleading comparisons. This paper established a methodology for comparative life cycle analyses, whereby the direct effects of the lidding films were compared whilst ensuring the indirect effects of the wasted food portion remained stable. Global warming potential and non-renewable energy use were analysed for a conventional (low-density polyethylene/ethylene vinyl alcohol) versus a biodegradable (polyhydroxyalkanoate/butenediol vinyl alcohol) multilayer lidding film for modified atmosphere packaging of minced beef. Two methodologies were investigated. The first (metric one) changed the barrier layer thickness in the biodegradable film to match the carbon dioxide transmission rate with that of a conventional film. The second (metric two) changed the barrier layer thickness to match a carbon dioxide transmission rate predicted by a mathematical model to ensure the same shelf-life as the conventional film. Using metric two over metric one resulted in 1) a thinner film 2) 2.3 times lower global warming potential. When using sugar beet as the biopolymer feedstock and the current UK disposal system, the biodegradable film had 135% higher global warming potential than the conventional film. By incorporating waste products and better farming practices, the global warming potential of the biodegradable film could be up to 92% lower than that of the conventional film. This work demonstrates how shelf-life can be incorporated into life cycle analyses and the importance of accounting for it, in particular when evaluating biodegradables which often have higher permeabilities.

1. Introduction

The change in public perceptions of plastic packaging has encouraged the European Union (EU) to commit to making all plastic packaging recyclable or reusable in the EU market by 2030 (European Commission, 2018). Meanwhile, in the United Kingdom (UK) the majority of the largest supermarkets and suppliers have signed up to the Plastics Pact (Wrap, 2020), a collaborative initiative between governmental and non-governmental organisations that has set targets for companies to help to create a plastic circular economy and ultimately transition waste into an added-value resource. The Plastics Pact regards a 'problem plastic' as one that is not recyclable or hampers the recycling process and has set the aim for contributing companies to replace all packaging with recyclable, reusable, or compostable materials by 2025. Conventional multilayer films currently used for meat packaging fall under the definition of 'problem plastic' and alternatives are therefore required.

Packaging necessity is debated, but it is generally agreed that some form is required for high-value products, such as red meat, to prevent contamination, extend shelf-life and minimise waste. Modified atmosphere packaging (MAP) typically comprises a tray and lidding film, with the headspace atmosphere designed to extend shelf-life and product quality. There are currently no commercial solutions for dealing with MAP films at end-of-life. Meat packaging films are often highly contaminated (e.g. with plasticizers) which negatively affects recycled product quality (Horodytska et al., 2018). Mechanical recycling of

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multilayer films is inefficient due to low volume and requirement for polymer separation (Horodytska et al., 2018). Reusable MAP films are also impractical due to food-contamination and distribution logistics. There is a need for more sustainable solutions.

Compostability may present a solution, as high barrier biodegradable multilayer lidding films could be degraded through biological activity into their constituent elements. If this occurs within a set timeframe in the industrial or home composting environment, the multilayer film would comply with the compostable requirement of the Plastics Pact. However, there are currently limited managed routes for the disposal of compostable plastics through either industrial composting or anaerobic digestion (AD), therefore, typically disposal is with municipal waste. Because of this the intended environmental benefits may not be realised. Studies often assume AD and composting as the sole waste management strategy despite, currently, only a small percentage of biodegradable products being disposed of in this manner (Kakadellis and Harris, 2020). It is important that new products and a region's waste management system are developed concurrently.

Also important is the functionality of alternative materials as this influences shelf-life and therefore food waste. Often packages that give a product a longer shelf-life have higher environmental burdens through the manufacture of more complex materials (Conte et al., 2015) or packaging systems (Gutierrez et al., 2017). Recent studies highlighted the small proportion of environmental impacts associated with packaging in comparison to the whole product-packaging system – sometimes as low as 1% (Silvenius et al., 2014). By increasing the shelf-life and reducing food waste, the increased burden from packaging manufacture is negated and the burden from the product life cycle is decreased (Conte et al., 2015). Sustainability assessment of packaging should account for the impacts of the packaging, food waste and the product's circularity (Pauer et al., 2019). A method for comparing packaging materials, proposed by Grant et al. (2015), suggests accounting for wasted food production and disposal, enabling the impacts from a reduction in waste to be quantified and better understood.

From a review of the literature, the indirect effects of food waste have come to the forefront of food packaging life cycle analyses (LCAs), however the impact of variations in packaging functionality remains unclear (Kakadellis and Harris, 2020). Some previous LCAs have assumed that packages have similar functionality, yet do not support this statement with data (Vidal et al., 2007). Others have modelled how a change in food waste affects the outcome of the LCA but did not determine the relationship between shelf-life and food waste (Dilkes-Hoffman et al., 2018). Attempts have been made to quantify this relationship using various methods such as surveying consumer habits to find the impact of packaging design (Silvenius et al., 2014) and shelf-life extension due to incorporation of nanomaterials (Zhang et al., 2019). Another method uses statistical predictions from models using empirical market data (Spada et al., 2018). Although these methods are successful at quantifying the impact of a large change in shelf-life on food waste, small changes in shelf-life on food waste are difficult to quantify due to the uncertainty of these methods. There is a clear need to better account for shelf-life in food packaging LCAs.

Based on the literature reviewed, gaps in knowledge exist around (1) the impacts from the end-of-life of biodegradable plastics based on current disposal systems and (2) the relationship between shelf-life and food waste. The aim of this study is to assess the environmental impacts of a conventional and biodegradable film throughout their life cycles whilst maintaining the functionality of the films. Therefore, the objectives of this paper are to (1) determine barrier layer thickness for a biodegradable and conventional film that provide the same shelf-life (2) carry out an LCA on the two films (3) investigate current and future disposal and production options for biodegradable films in a sensitivity analysis. This paper will focus on an ethylene vinyl alcohol (EVOH)/low-density polyethylene (LDPE) film (a conventional film whereby EVOH forms an inner barrier layer) and a butenediol vinyl alcohol (BVOH)/polyhydroxyalkanoate (PHA) film (a biodegradable film, with a BVOH barrier layer).

2. Methodology

LCA is a technical approach for evaluating the environmental impacts of products throughout the different stages of their lives, from the extraction of the raw materials to product manufacture, and disposal at end-of-life (Scientific Applications International Corporation, 2006). The International Organization for Standardization (ISO) 14040 and 14044 series (The International Standards Organisation, 2006a; 2006b) and publicly available specification, PAS 2050 (BSI, 2011), were followed in this analysis.

2.1. Goal, scope & boundary

The goal of this LCA was to evaluate the environmental burdens associated with the life cycle of a biodegradable multilayer lidding film (PHA/tie/BVOH/tie/PHA) in comparison with a conventional multilayer lidding film (LDPE/tie/EVOH/tie/LDPE). Both films are high barrier films intended for use for fresh meat MAP (Polymers Database, 2020). Minced beef was assumed for this analysis. The films were assumed to be manufactured and used in the UK. The PHA and LDPE layers act as barriers to water vapor while the BVOH (known commercially as Nichigo G-Polymer™) and EVOH (known commercially as EVAL™ F type (32 mol% ethylene)) layers act as barriers to oxygen and carbon dioxide. The direct and indirect effects of these two films were analysed from cradle-to-grave using secondary data.

Elements of the supply chain (packing, retail and consumer stages) that were assumed to be the same for both films were excluded from the system boundary, although waste associated with these stages was included in the end-of-life (Fig. 1). The impacts from the production of tie layers are often neglected in LCAs and were not included in this study due to assumed negligible impacts and lack of information on production and environmental impacts (Kliaugaitė and Staniškis, 2013). The end-of-life stage was calculated with the simplification that the conventional and biodegradable films were made entirely of LDPE and PHA, respectively. This was justified because properties of the inner (EVOH and BVOH) and outer (LDPE and PHA) layers mean they would act similarly in disposal systems since BVOH, PHA and the intended tie layer are fully biodegradable (Mitsubishi Chemical, n.d.) and EVOH and LDPE are non-biodegradable; therefore, this simplification was not expected to have a significant impact on the end-of-life assessment.

Transport was included to account for the movement of the feedstock and polymer granules, which allowed for a sensitivity analysis of crops used to produce PHA that are native to different countries. The refrigerated transport of meat wasted due to expired shelf-life was accounted for in the 'Wasted Meat Production' stage, as this transport stage is attributed to the food product in line with BSI (2011). Transport was not included in the end-of-life stages other than for transporting the digestate and compost to their destination.

Sugar beet, sugarcane, and corn (FS 1–3) are pre-processed into monosaccharides before being converted into PHA through cellular conversion. The 'Farming & Pre-processing' stage (Fig. 1) for scenarios FS 1–4 includes field emissions, agricultural production, processing into monosaccharides and energy and emissions credits for displaced products. Production routes that make use of waste products and do not require farming were identified (FS 6 & 7) and were compared with the impacts arising from using crops farmed for food as a feedstock (FS 1–4). FS5, 'Corn & Stover', uses sustainably farmed corn to limit emissions, as well as electricity generated from combustion of the lignin rich stover (Kim and Dale, 2008). Corn is not commonly grown in the UK and is more common in mainland Europe, therefore the crop was assumed to be shipped from France through the Calais to Dover route.

FS6, 'Biogas', assumes that methane generated from an AD plant treating PHA and other organic waste is the feedstock for PHA production (Rostkowski et al., 2012). Carbon emissions from the disposal of

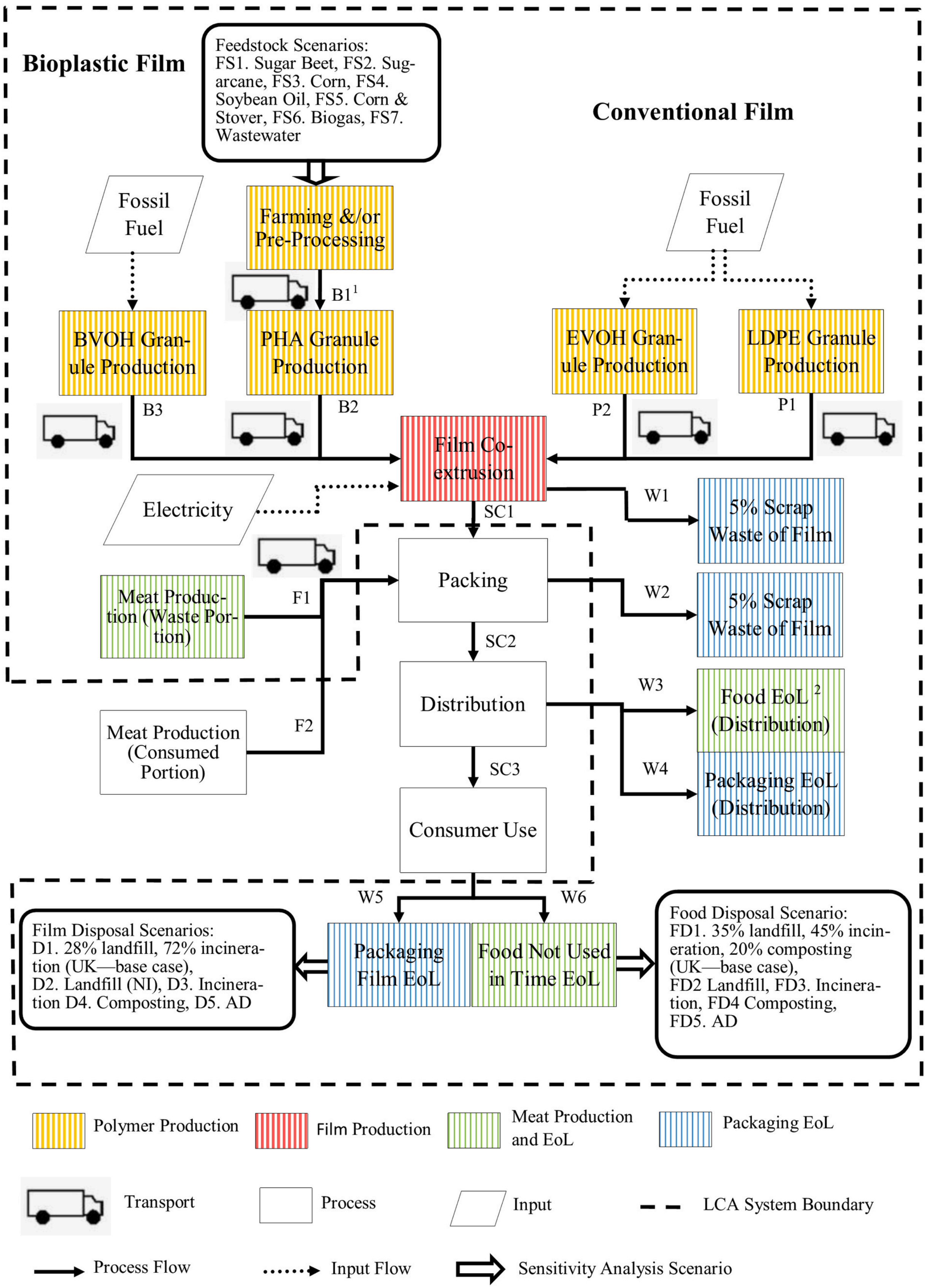

Fig. 1. System boundary and material flow between processes. [1]Letters B, P, F, SC and W refer to the production stage of the biodegradable film, conventional plastic film and food, the flow through the supply chain, and the material going to waste, respectively. The numbers refer to different steps within that stage. [2]EoL is end-of-life.

PHA could be recycled into new PHA; however, this relies on the disposal infrastructure to be in place to ensure circularity. FS7, 'Wastewater', refers to a process that ferments wastewater from a paper mill or food waste industry into volatile fatty acids, which are then used to selectively grow PHA producing bacteria and to provide feedstock for the bacteria's growth (Fernández-Dacosta et al., 2015). Further information on the LCA of each feedstock can be found in Chapter A1, Table A1.1.

2.2. Functional unit

The functional unit was taken as the amount (g) of film required for 1 kg of produce; 1 kg of produce is commonly made up of two 500 g packs of minced beef and the area of film required to cover two packs is 0.079 m^2 (based on packs on display in a major UK supermarket chain that were produced by a major manufacturer of plastic trays). For the purposes of this LCA, the atmosphere inside the package was assumed to be 20% carbon dioxide and 80% oxygen, as is common for red meat in MAP (McMillin, 2008). Conventional multilayer films are commonly 50 μm thick (Dixon, 2011), containing 5 μm of EVOH (Mullan and McDowell, 2003).

Since the function of meat packaging is to maintain the integrity of the product for a set amount of time, the shelf-life was used to define the functional unit of the film. The high concentration of carbon dioxide in the package headspace is designed to inhibit bacterial growth and prolong shelf-life, and a drop in the concentration due to a more permeable film can cause a decrease in shelf-life (Marcinkowska-Lesiak et al., 2016). If a barrier layer has a higher permeability to carbon dioxide, then the thickness of the layer can be increased to decrease the gas transmission across it (transmission rate = permeability/thickness). Two metrics were trialled to determine the BVOH thickness: one) equalising the carbon dioxide transmission rate (CTR) for both packages, and two) ensuring the minimum required CTR to maintain the shelf-life was reached in both packages.

To calculate the film's CTR, the permeability of each layer was required before combining them with the layer thicknesses to calculate the overall CTR (Eq. (1)). The permeabilities, and therefore the CTR, of the inner layers, BVOH and EVOH, are both dependent on the relative humidity (RH). Since the RH can vary between 35% (in consumer refrigerators) (Howell et al., 1997) and 85% (in refrigerated vans) (Baston and Barna, 2012), the CTR was modelled across a range of RHs to ensure the package functions in all environments it will be subjected to along the supply chain. To find the permeability of the barrier inner layer, the RH experienced by this layer was calculated using Eq. (2). These values are influenced by the ability of the outer layers to keep water vapor out and the position in the film. To reduce the RH experienced, the barrier layer is positioned nearer the outer layer, 35 μm from the inner layer (Kuraray, 2012). Using published permeability values of the inner and outer layers, the CTR was calculated. From the calculations (Chapter A2, Table A2.3), the BVOH layer has a higher permeability at high RH, therefore, to decrease the CTR, the thickness of the BVOH layer was increased.

$$CTR_T = 1 \Big/ \left(\frac{x_1}{P_{CO_2 1}} + \frac{x_2}{P_{CO_2 2}} + \cdots + \frac{x_N}{P_{CO_2 N}} \right) \qquad 1$$

where CTR_T: film's carbon dioxide transmission rate (m^3 m^{-2} hr^{-1} atm^{-1}), x_1, x_2, …, x_N: thicknesses of the film's layers (m), and $P_{CO_{2_1}}$, $P_{CO_{2_2}}$, …,$P_{CO_{2_N}}$: carbon dioxide permeability of the film's layers (m^3 m^{-2} hr^{-1} atm^{-1}).

$$RH_B = \frac{RH_O\left(\frac{x_B}{P_{H_2O_B}} + \frac{2x_I}{P_{H_2O_I}}\right) + RH_I\left(\frac{x_B}{P_{H_2O_B}} + \frac{2x_O}{P_{H_2O_O}}\right)}{2\left(\frac{x_I}{P_{H_2O_I}} + \frac{x_B}{P_{H_2O_B}} + \frac{x_O}{P_{H_2O_O}}\right)} \qquad 2$$

where RH_B: relative humidity of the barrier layer (%), RH_O: relative humidity of the outside environment (%), RH_I: relative humidity of the inside environment (%), x_B: the thickness of the barrier layer (m), x_I: the thickness of the inside layer (m), x_O: the thickness of the outside layer (m), $P_{H_2O_B}$: water vapor permeability of the barrier layer (g m m^{-2} hr^{-1} atm^{-1}), $P_{H_2O_I}$: water vapor permeability of the inside layer (g m m^{-2} hr^{-1} atm^{-1}), and $P_{H_2O_O}$: water vapor permeability of the outside layer (g m m^{-2} hr^{-1} atm^{-1})).

As per metric one, if the BVOH thickness is increased to meet the CTR exhibited by a 5 μm EVOH layer (Fig. 2), then it can be assumed that, at high RH, the BVOH layer becomes too thick to be practically incorporated into a lidding film. This is because lidding films are commonly around 50 μm thick; films above 200 μm thickness are difficult to produce on a blown film line and once the thickness is above 1 mm, the material becomes a sheet (Dixon, 2011). For metric two, the shelf-life was compared using a predictive model on MATLAB (R2020a) (Hutchings et al., 2021). In brief, the code calculates the minimum required carbon dioxide permeability, above which the permeability begins to shorten shelf-life. The code is based on the predicted growth of *Pseudomonas* spp. since this spoilage bacterium is particularly susceptible to changes in carbon dioxide concentration in the headspace (Gill and Tan, 1979). Results from the code were validated against 13 datasets of *Pseudomonas* spp. concentration over time using different red meat products, package sizes, temperatures and lidding permeabilities. Inputs were entered into the model to replicate a pack of 500 g of minced beef and the resulting CTR requirement to maintain the shelf-life (to within 1 h) was predicted to be 9.70×10^{-6} m^3 m^{-2} hr^{-1}.

To carry out the calculation for metric two, first the CTR of both barrier layers was calculated at a thickness of 5 μm at each RH, to assess whether the required CTR was met (Fig. 3). While 5 μm is sufficient for the EVOH barrier layer, the BVOH barrier layer exceeded the maximum allowable CTR at high RH (75%) (Fig. 3). Therefore, the thickness of the BVOH was increased to 6 μm to ensure similar performance at high RH. By meeting the minimum required CTR (metric two), the food waste in both products can be assumed equal and the impacts from the films can be fairly compared by incorporating this change in the inner layer thickness in the functional unit. This metric was used as it gives a more practical result than metric one. The remaining film was assumed to be composed of PHA and LDPE for the biodegradable and conventional film, respectively, and the overall film thickness was the same as for the conventional film. The mass flows through the hypothetical system are in Chapter A3, Table A3.1.

A third option of altering the PHA thickness (rather than the BVOH thickness) in order to reduce the RH experienced by the BVOH inner layer was considered. An increase of 10 μm in the thickness of the PHA layer resulted in an overall decrease of 1% RH experienced by the inner layer (when the outside RH was 75%) which also decreased the CTR of

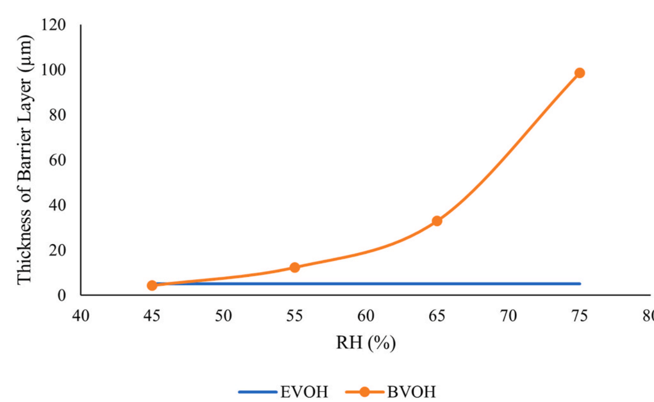

Fig. 2. Thickness of BVOH film required to match the CTR of EVOH with a thickness of 5 μm at different relative humidity values. Calculations are detailed in Table A1.3.

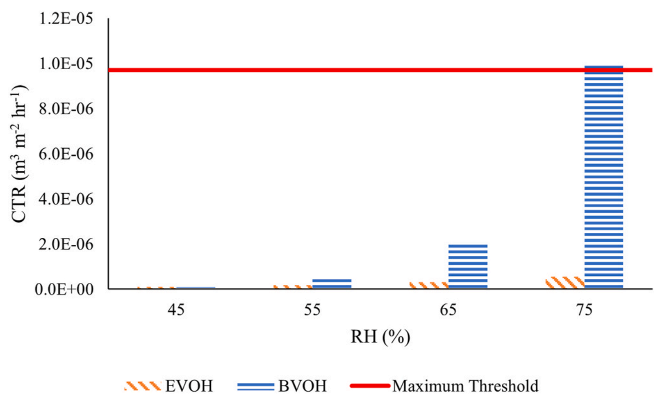

Fig. 3. Carbon dioxide permeability exhibited by EVOH and BVOH, whilst keeping the thickness of both films at 5 μm, compared to the maximum threshold of 9.70×10^{-6} m^3 m^{-2} hr^{-1} to maintain shelf-life.

the lidding film to below the desired level (9.70×10^{-6} m^3 m^{-2} hr^{-1}) to maintain shelf-life. However, this method increased the weight of the biodegradable film by 19%, which would increase both the GWP and NREU by the same proportion (19%). Therefore, changing the thickness of the BVOH layer was considered a more sustainable option, since it did not involve increasing the thickness or significantly changing the weight of the film.

2.3. Impact factors and inventory

2.3.1. Overview and data sources

Two impact category indicators were assessed:

a) Global warming potential (GWP) for a 100-year time horizon (GWP 100) reported as equivalent carbon dioxide greenhouse emissions (kg CO_2 eq.).
b) Non-renewable energy use (NREU) (also known as fossil fuel depletion) (MJ).

This LCA included only these two categories due to insufficient data for the inclusion of more categories. However, both factors are important in helping to reach the Sustainable Development Goals set out by the United Nations and the UK's target of becoming carbon neutral by 2050 ("Climate Change Act," 2008), and the inclusion of fossil fuels specifically helps to give an idea of the circularity of the product and the long-term circularity of production (Pauer et al., 2019). The data for the life cycle was sourced from Ecoinvent (version 2.2) (Frischknecht et al., 2005), Plastics Europe (Plastics Europe, 2018), Environmental Footprint Secondary Database (EFSD) (version 1.0 in OpenLCA) (European Commission Single Market for Green Products, 2019), European reference Life Cycle Database (ELCD) (version 3.2 in OpenLCA) (Joint Reference Centre European Commission, 2015), UK Government data and literature values (Chapter A4, Table A4.1.). Where data was not directly available in the literature, specific calculations were carried out and these are detailed in the following paragraphs.

2.3.2. Electricity generation

Impact factors from external references were updated to be in line with current emissions and fossil fuel depletion from UK electricity generation. The details of fossil fuel depletion per unit of electricity consumed can be found in Chapter A5, Table A5.1. (Gov.UK Department for Business Energy and Industrial Strategy, 2019). Emissions from electricity were 0.077 kg CO_2 eq./MJ electricity (Hill et al., 2019).

2.3.3. BVOH production

Butenediol vinyl alcohol is a relatively new polymer, and the authors are not aware of existing life cycle studies in the literature, however, information on its production is available in the form of patents and these were used to estimate life cycle impacts (GWP and NREU). Butenediol vinyl alcohol is made through copolymerisation of vinyl acetate with a butene diol monomer, the most favourable of which is 3,4-diacetoxy-1-butene (3,4-DAB) due to its excellent reactivity ratio with ethanol vinyl acetate (Shibutani and Sakai, 2011). The resulting copolymer is then saponified to achieve a hydrolysis degree of 80–97.9%.

The reaction between 3,4-DAB and vinyl acetate requires 1 mol of 3,4-DAB to between 5 and 9 mol of vinyl acetate (Shibutani and Sakai, 2011). The production pathway and environmental impacts of vinyl acetate are well documented (Frischknecht et al., 2005). 3,4-DAB can be produced as a co-product in the production of 1,4-diacetoxy-2-butene (1,4-DAB) through acetoxylating 1,3-butadiene with acetic acid. The selectivity for 1,4-DAB is 90% with approximately the remaining 10% becoming 3,4-DAB (Weissermel and Arpe, 1997), which is then distilled off (Kuni et al., 2004). 1,4-DAB is then further processed to become 1,4-butanediol or benzylalcohol, an important chemical in the manufacture of plastics, whereas 3,4-DAB has few well-known other uses (Komatsu, 2003). Because of the large number of unknowns, the impacts were compared using two allocation approaches. Impacts were calculated (Chapter A6, Table A6.1 and Table A6.3) from both upper and lower molar ratio values; due to the uncertainties surrounding molar ratios and the specifics of the processes, the average of the four scenarios was used (Table A6.5).

2.3.4. Meat production (wasted portion)

Of the total amount of beef purchased in the UK (1,187,000 tonnes in 2017), 112,000 tonnes are wasted (Jeswani et al., 2021) and 29% of that is due to 'not being used in time' (Quested et al., 2013). There fore 2.7% of the purchased beef is wasted due to expired shelf-life ($0.29 \times 112{,}000\ \textit{tonnes}/1{,}187{,}000\ \textit{tonnes} \times 100 = 2.7\%$). The waste from the distribution (including distribution centres, retail, wholesalers and transport between them) of beef products is 3.8%, with the primary reason for this being expired shelf-life (Jeswani et al., 2021). A fraction of wasted food portion was allocated to the lidding film based on the ratio of the film's active area to the total active area of the whole packaging (tray plus film) which represents the film's preservation functionality. This ratio was found to be 0.28:1 which means that for each 1 kg of wasted beef, 0.28 kg is allocated to the film.

2.3.5. End-of-life

It was assumed for the base case scenario that plastic waste was managed by landfill and incineration in line with the proportions of black bin municipal waste managed through these routes in the UK. The UK was until recently subject to the EU Waste Framework Directive, however, the ratios of municipal waste going to incineration (with energy recovery) and landfill (72:28) (scenario D1) are significantly different from the EU average (55:45) (Eurostat, 2020). Within the UK practices differ between the regions; the film disposal scenario, D2, considers the Northern Ireland (NI) context where all film waste goes to landfill (Fisher, 2020). Due to variation in waste management strategies across the EU, both within and between countries, the impacts associated with biodegradable waste disposal can vary widely. Therefore, by considering the UK strategy (D1) and NI purely landfill strategies (D2), along with 100% incineration (D3), 100% composting (D4) and 100% AD (D5), the results present the wide range of strategies currently employed. The calculations and impact values from the end-of-life scenarios can be found in Chapter A7.

3. Life cycle impact assessment – results and discussion

3.1. Base case scenario

The GWP and NREU of the biodegradable film are 135% and 9% higher respectively than for the conventional film (Fig. 4a). The reason there is a smaller difference between the NREU values than the GWP

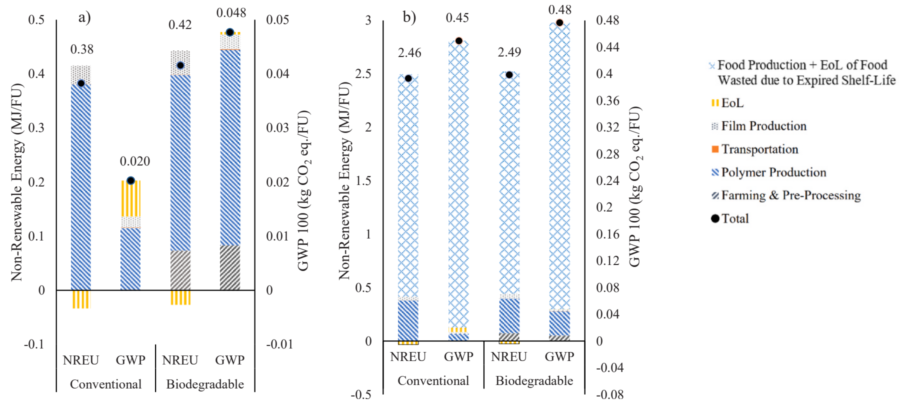

Fig. 4. a) NREU and GWP per functional unit (1000 g of beef at the consumer home) of the conventional and biodegradable film (base case: sugar beet (FS1) and UK disposal scenario (D1)) from cradle to grave, b) including the portion of food not used in time (FD1) allocated to the film from cradle to grave.

values is due to the use of crude oil to make plastics; 66% of the NREU of LDPE resin production is attributed to the energy potential in the feedstock (Hammond and Jones, 2011), whereas the PHA feedstock is treated as a carbon credit. The biodegradable film is 30% heavier than the conventional film, due to PHA having a higher density than LDPE (Chapter A3, Table A3.1). The polymer production stage is the highest contributing stage to both GWP and NREU. When excluding the energy credits from the end-of-life stage, 92% of the total NREU is attributed to polymer production for the conventional film. When end-of-life is included, only 57% of the GWP is from the production stage of the conventional film. GWP is likely to fall in the future due to the decarbonisation of the electricity grid, however fossil fuel use will remain high due to fossil fuels being the primary feedstock for most conventional plastics. In the case of the biodegradable film, 90% of NREU and 93% of GWP result from the farming and polymer production stages, while the next most impacting stage is end-of-life. The farming and end-of-life hotspots are investigated in the sensitivity analysis.

When food waste is considered (Fig. 4b), the GWP is over 9 and 20 times higher than the base case for the biodegradable and conventional films respectively. The higher GWP arises because of the production and disposal of waste beef, and primarily from the methane emissions over the lifetime of cattle due to their digestive processes (enteric fermentation). In comparison, the NREU was only around six times higher in both cases, primarily from the production of feed, either grass or concentrate (Williams et al., 2006). These results highlight the importance of including food waste, which is in agreement with recent literature on the subject (Kakadellis and Harris, 2020). As little as a 0.3% change in the amount of product being wasted would cause an increase in the GWP equal to the impacts of the conventional film itself.

3.2. Sensitivity analysis

3.2.1. Farming stage

The base case considered utilises sugar beet as the feedstock for biodegradable polymer production as it is commonly grown in the UK. Although it is a high-yielding crop, it requires arable land, competes with food, and must be grown in rotation (Trimpler et al., 2017). Another potential temperate crop is corn. Corn PHA (FS3) has the highest impacts of the options examined, which is due to a low crop yield (9.1 tonne crop/hectare) (Kim and Dale, 2004), but by incorporating the corn stover for electricity generation, as well as getting rid of the tilling process to increase the sustainability of the farming practices, it becomes the least impacting option for NREU and has a negative GWP. However, although incorporating stover (the parts of the crop that are not consumed), a portion of the crop that would otherwise be eaten is used raising concerns for the food-fuel-fibre debate.

Options for imported crops include soybean and sugarcane. Soybean oil (FS4) is converted directly into a PHA (poly-3-hydroxybutyrate (P(3HB)) through cellular conversion; the yield of this conversion (0.76–0.78 g P(3HB)/g-soybean oil (Kahar et al., 2004)) is higher than for glucose (0.3–0.4 g P(3HB)/g-glucose (Ryu et al., 1997)) due to the high concentration of linoleic acid, resulting in low GWP and NREU impacts compared to the feedstocks (sugar beet, sugarcane and corn) that require conversion to glucose (Fig. 5). However, soybean farming is linked to deforestation in the Amazon, causing depletion of carbon stocks (Bonini et al., 2018) and displacement of local communities, as well as aggravating socio-economic issues such as inequality and territorial disputes (Sauer, 2018). Sugarcane (FS2) performs the best (Fig. 5), due to a high crop yield (85 tonne crop/hectare) (Renouf et al., 2008) but it has also been linked to similar social and environmental impacts (Machado et al., 2017).

Waste feedstocks were also considered. Using wastewater (FS7) results in a low GWP, but the energy consumption during downstream processing of the mixed culture is large (76.59 MJ/kg polyhydroxybutyrate) due to the electricity required for evaporation of the solvent used (Fernández-Dacosta et al., 2015). Biogas (FS6) presents a viable alternative with a modest GWP and NREU but would require investment in AD infrastructure and waste management processes. It could, however, offer a new income stream to AD plant operators which could potentially improve financial viability particularly of farm-scale plants which struggle in the absence of subsidies (Cucchiella et al., 2019).

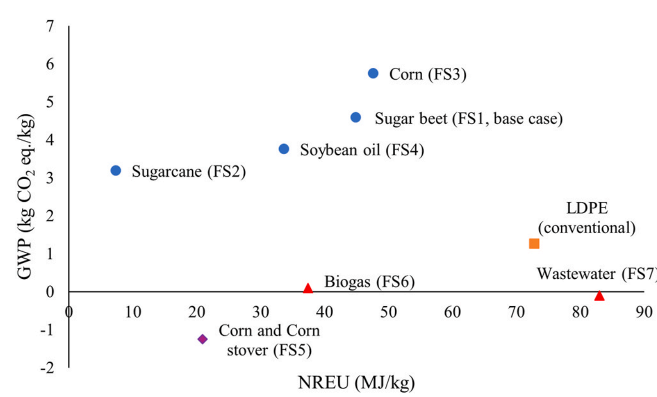

Fig. 5. Scatter graph of GWP versus NREU per kg of PHA made from various feedstocks (FS 1–7) (food waste excluded), blue circles represent feedstocks that are otherwise used for food, red triangles represent waste streams being used as feedstocks, purple diamonds represent a feedstock made from a combination of food and waste stream (Table A5.1.) and the orange square represents the GWP and NREU per kg of LDPE (the conventional film outer layer).

3.2.2. End-of-life scenario

The end-of-life scenario is an important factor for overall life cycle impacts. With all biodegradable films going to landfill in NI (D2), the GWP from the biodegradable film in NI contributes 14% of the LCA and is 16% higher than in the UK (Fig. 6a). The use of energy recovery in the UK results in a negative value for NREU due to the electricity credits. The conventional film has a higher energy content than PHA, therefore energy recovery is more beneficial for conventional films (Fig. 6b). Food, when not used in time, is often thrown away in its entirety to avoid contamination, potentially ending up in landfill. Similarly, remnants of produce on the plastic packaging can also end up in landfill. AD or composting of biodegradable plastic packaging and its contents could help divert food waste from landfill and incineration and would help with the UK's target to eliminate food waste to landfill by 2030 (HM Government, 2018). The GWP reduction from food waste disposal in AD instead of landfill is 0.48 CO_2 eq./kg (Chapter A8, Table A8.1 and Table A8.2). Composting has a higher GWP and NREU than AD (Fig. 7a and b), but AD has higher impacts in other categories not investigated in this paper, such as eutrophication, acidification, and toxicology, largely due to resources required for treating wastewater and exhaust gas (Salemdeeb et al., 2018). AD and incineration perform well primarily due to the displacement of fossil fuels. As energy systems decarbonise in the future, the impacts will need to be reassessed.

3.3. Interpretation and recommendations for life cycle improvement

3.3.1. Impact of functional unit

By looking specifically at the food wasted due to expired shelf-life, a relationship between the food waste and shelf-life was assumed: food waste will remain the same if the shelf-life remains the same (assuming no change in packaging design). Rather than quantifying how food waste is affected based on shelf-life, the quantity of food waste was maintained by altering the functional unit to maintain shelf-life. If equal carbon dioxide transmission rates (CTRs) were used as the basis for defining the functional unit (metric one), the GWP would be 2.3 times larger than with the shelf-life functional unit (metric two) (Fig. 8). The result emphasises the importance of selecting the most appropriate functional unit so as not to skew the results, as well as highlighting the importance of accounting for the transient nature of the permeability of these barrier layers at different RHs.

The choice of functional unit can change the results of an LCA dramatically, but it is not well studied in the literature, with the majority of LCAs using a functional unit of m^2 for films and kg for trays (Kakadellis and Harris, 2020). However, with decreasing food security, the importance of including food in the LCA of food packaging is more relevant than ever. One study (Lorite et al., 2017), altered the functional unit of kg of produce consumed based on different shelf-life durations (which were based on the probability of purchase depending on whether the product was within two days of the 'use-by' or 'best before' date) to find the critical shelf-life extension to ensure environmental benefit. However, only one study (Siracusa et al., 2014) has investigated the thickness of the film, by reducing a polyamide/polyethylene film from 85 μm to 65 μm and running tests to ensure no reduction in shelf-life, 25.3% less environmental damage was recorded.

This current paper developed a novel method and therefore provides a new basis for conducting LCAs of food packaging where justification of maintaining shelf-life through experimental or theoretical calculations should be performed to increase the accuracy of comparative LCA results concerning the wasted food production stage. Despite PAS 2050 (BSI, 2011) and ISO 14040 and 14044 (The International Standards Organisation, 2006a; 2006b) having no mention of the inclusion of food waste in the life cycle of packaging, one cannot be fully assessed without the other if a truly sustainable product is to be achieved.

3.3.2. Impact of feedstock and end-of-life

The type of feedstock used can go a long way to lowering the GWP. Different crops give different yields of PHA (Somleva et al., 2013), however the concerns around food security remain (Karan et al., 2019). Making use of waste streams offers a circular route for production whilst helping with the disposal of agricultural waste. In the UK, numerous waste streams could be used to produce PHA or alternative biobased plastics. Identified waste streams in the UK include wheat straw, barley straw, oat hull, carrot tops and sugar beet tops (Bolaji et al., 2021). Using lignin-rich by-products, such as corn stover, could also give lower impacts (Kim and Dale, 2005). Other possible feedstocks for PHA include waste vegetable and animal oils (Surendran et al., 2020), lignocellulosic waste materials (de Souza et al., 2020), and brewery wastewater (Tamang et al., 2019). Findings from a consumer and industry focus group showed high enthusiasm for the incorporation of waste products into biobased materials for food packaging purposes, although there were some concerns when the waste product originated from animal matter (Mehta et al., 2021).

By using wastes, biodegradable and biobased plastics can contribute to the circular economy, but circular does not necessarily mean sustainable (Haupt and Hellweg, 2019). To ensure sustainability of bioplastics (biobased and/or biodegradable plastics) sustainability criteria

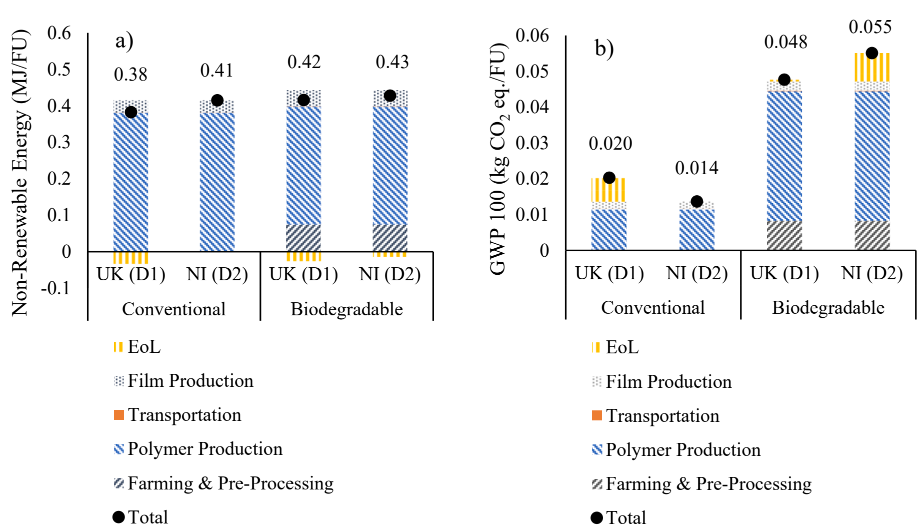

Fig. 6. Impacts of the conventional and biodegradable (FS1) films per functional unit considering disposal routes 72:28 incineration: landfill (UK scenario D1) and 100% (NI scenario D2) landfill, a) GWP and b) NREU.

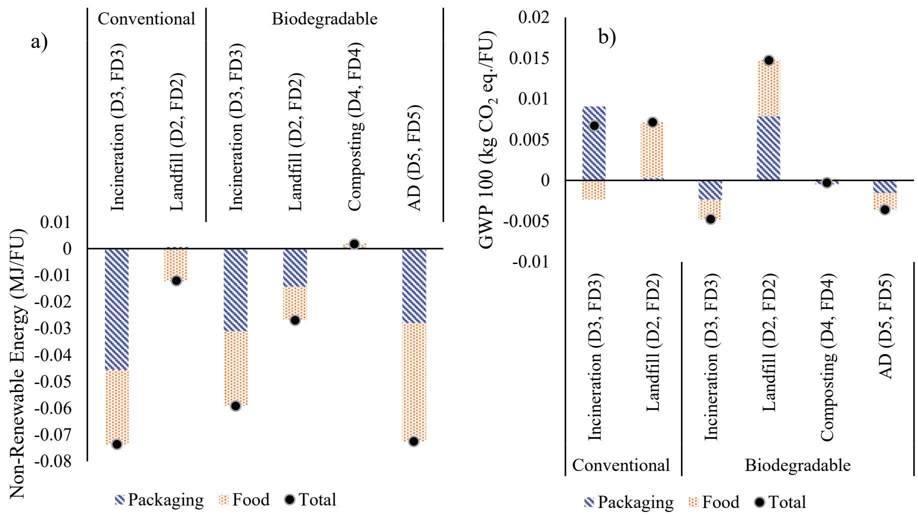

Fig. 7. Impacts from the end-of-life of packaging and wasted portion of beef going to different end-of-life destinations (D2-5 and FD1-5), a) GWP and b) NREU.

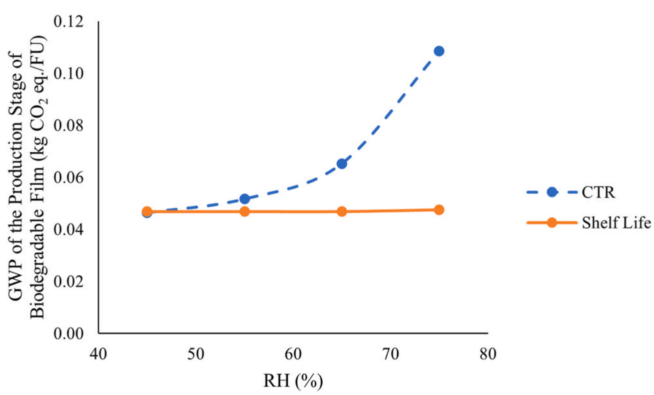

Fig. 8. The GWP of the production of the biodegradable film per functional unit (mass of film per kg of product) based on both CTR and shelf-life.

could be enforced in a similar way to the criteria in the bioenergy industry that necessitate a 60–80% reduction in greenhouse gas emissions (European Parliament, 2018). By changing the end-of-life disposal from the current UK scenario (landfill and incineration) to AD and changing the feedstock from sugar beet to biogas, the NREU and GWP would decrease by 31% and 89% respectively; this is 25% and 75% lower than the values for the conventional film (Fig. 9). Provided that the feedstock and disposal method are carefully selected, switching from a conventional film to a biodegradable film achieves these targets and also aligns with the EU goal to reduce greenhouse gases by at least 55% by 2030

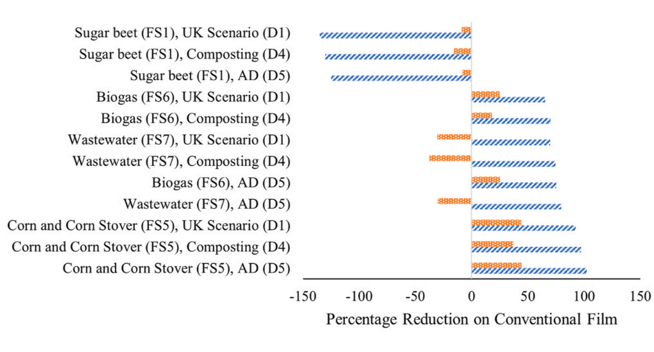

Fig. 9. Percentage life-cycle savings of the biodegradable film compared to the conventional film according to crop used and end-of-life.

(Fig. 9) (European Commission, 2020). The EU has laid out objectives to increase circularity in production processes in the future, including 'supporting the sustainable and circular bio-based sector through the implementation of the Bioeconomy Action Plan' (European Commission, 2020). The gap between the NREU of biodegradable and conventional films will also increase as the UK decarbonises its electricity grid and PHA production becomes more efficient. A study comparing the use of a PHA versus a polypropylene carrier bag showed that, by switching from the US energy mix to geothermal, the GWP of the PHA bags is 80% below that of the polypropylene bags (Khoo et al., 2010).

3.3.3. *Limitations and recommendations*

This LCA focused on GWP and NREU. Further work is recommended to investigate a suite of environmental and socio-economic impacts. Following standards such as the EU Product Environmental Footprint to consider a wider range of impacts has been recommended and this would also help to make LCAs from different studies more comparable (Walker and Rothman, 2020). The use of inconsistent methodologies in existing studies in the literature makes it difficult to fairly evaluate bio-based and fossil-based plastics as the results from LCAs vary widely even for the same polymer type. This is especially true for bio-based polymers due to the different processing methods, feedstock source and end-of-life calculations (Walker and Rothman, 2020).

A limitation of the current investigation was the lack of available data on BVOH production and end-of-life scenarios. The work could be expanded by including a tray in the LCA to compare the entire package, as other research suggests that the GWP of the lidding film of meat packaging only accounts for 22% of the overall GWP of the package (Firoozi Nejad et al., 2021). However, the importance of the lidding film in maintaining shelf-life needs to be recognised, and as such, accurate assessment of its impacts is required. Current work provides the basis for future research on novel packaging.

4. Conclusions

This paper successfully established a methodology for comparative LCAs of lidding films, whereby the direct effects of the lidding films were compared whilst ensuring that the wasted food portion remained stable. The work highlights that the difference in barrier properties between biodegradable and conventional plastics should be acknowledged and accounted for in the functional unit to allow accurate comparisons to be made. The use of shelf-life to determine the functionality is advised when carrying out LCAs of bioplastics to ensure a fair comparison is made with conventional plastics. Results showed that a small increase

(0.3%) in the proportion of food going to waste would be equivalent to the GWP of the conventional lidding film, highlighting the importance of ensuring the functionality of the packaging.

The investigation into PHA feedstocks has led to the conclusion that waste streams should be incorporated into the production of bioplastics to avoid the food-fuel-fibre debate and increase the circularity and sustainability of the product by reducing the GWP to 70% below conventional films. Although disposal systems in the UK are currently ill-equipped to deal with lidding films, they are also not prepared for biodegradable plastics, causing emissions from landfill and incineration and this is exacerbated in NI where the reliance is solely on landfill. Composting and AD could be incorporated to decrease the GWP from the life cycle of the biodegradable film by 15 or 18% respectively in NI. These disposal systems in conjunction with biodegradable packages could reduce the number of unopened packages and plastics containing food remnants going to landfill and incineration.

Developing a waste management stream for biodegradable films and incorporating waste products into their production could lead to a circular product. If bioplastics were incorporated meaningfully into society, they could help to reduce production of 'problem plastics' and contribute to targets for zero food waste to landfill and carbon neutrality by 2050.

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CRediT authorship contribution statement

Natasha Hutchings: Conceptualization, Methodology, Investigation, Writing – original draft. **Beatrice Smyth:** Conceptualization, Methodology, Writing – review & editing, Supervision. **Eoin Cunningham:** Writing – review & editing, Supervision. **Mahamad Yousif:** Investigation, Methodology. **Chirangano Mangwandi:** Writing – review & editing, Supervision, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2021.129423.

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