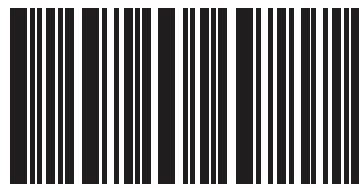Plant
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Nutritional and environmental comparison
of chicken and plant protein

by S. G. Wiedemann, J. Dunn, N. Senior, and L. Biggs
October 2020

AgriFutures®
Chicken Meat

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October 2020

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In submitting this report, the researcher has agreed to AgriFutures Australia publishing this material in its edited form.

Foreword

This project was conducted to assist the chicken meat industry to understand the environmental impacts and nutritional comparability of plant-based alternatives (PBAs) to Australian chicken meat. The findings of the literature review and scoping study suggest that many of the claims made suggesting significantly lower environmental impacts of PBAs compared to meat products are not accurate when compared to Australian chicken meat, and when nutritional equivalence is taken into account, though further research is needed to confirm this.

This research is the first step towards enabling comparison of chicken meat and PBAs on 'like terms' by considering their nutritional and environmental impacts, which aims to build a strong knowledge base for the industry.

This report was funded by AgriFutures Australia as part of Objective 2 of the Chicken Meat Program RD&E Plan 2019-2021 *Develop and implement measures to improve industry's impact on the environment.*

This confidential report has been developed for the AgriFutures Australia, AgriFutures Chicken Meat Advisory Panel, the Managers, Research for AgriFutures Chicken Meat Program and Australian Chicken Meat Federation staff. Provided there is consensus of the benefit to the industry this information may be extended beyond the industry. Based on the findings presented here and depending on further research to substantiate the scoping results.

John Smith
General Manager, Research
AgriFutures Australia

Abbreviations

GHG	Greenhouse gas
LUC	Land-use change
PBA	Plant-based alternative
CM	Chicken meat
QP	Quorn Pieces
PB	Pea-based product
LM	Laboratory manufactured meat
LCA	Life Cycle Assessment
NQCFU	Nutritional Quality Corrected Functional Unit
NQS	Nutritional Quality Score

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Executive summary

Increasing ethical and environmental concerns of consumers surrounding traditional meat products has resulted in the emergence and expansion of meat alternatives in the market. These products include plant-based alternatives (soy, wheat, pea, oat), animal-based alternatives (milk and insects) microbial products (mycoprotein), and cultured meats (Filho et al., 2019). Plant-based alternatives (PBAs) and meat analogues are manufactured food products which are designed to mimic the taste and texture of meat products. Meat substitutes aim to reduce environmental impacts caused by livestock production and are often marketed as a more environmentally friendly option compared to meats, commonly beef (Smetana et al., 2015). However, research shows that Australian chicken meat has relatively low environmental impacts compared to other meat products (Wiedemann, 2018) and it is therefore less clear whether these general comparisons are reasonable for comparing Australian chicken and PBAs or meat analogues. For comparisons to be informative, they also need to consider the nutritional factors associated with different products, and such knowledge is lacking for Australian chicken meat and alternatives. This project aimed to address these knowledge gaps by (1) determining the nutritional comparability of chicken meat to common PBAs and cultured meat via a literature review, and (2) by conducting a scoping life cycle assessment (LCA) to investigate the environmental impacts of the alternative foodstuffs and Australian chicken meat.

Background

The marketing and consumption of vegetarian, vegan and reduced meat diets are rapidly increasing. According to the Good Food Institute, sales of plant-based alternatives to meat (PBAs) grew 38 % in the US in the past two years, compared to total growth in retail food of 4 % over the same period (The Good Food Institute, 2020). In Australia, surveys show a substantial increase in the number of consumers identifying as vegetarian and vegan in the past five years (Roy Morgan, 2019). In the food market, PBAs and cultured meat companies are actively competing with traditional meat, using environmental credentials as a major point of difference. However, most comparisons have been done with beef, not chicken, and most studies were conducted overseas, where the environmental impact of PBA manufacturing may be lower than in Australia because of Australia's reliance on fossil fuels. Moreover, product comparisons must take into account nutritional factors, but this is often not done, as shown by the high-profile study in Lancet, which compared chicken with dry soybean despite the substantial differences in these two products, and recommended an 80 % lower chicken intake than the Australian average (Willett et al., 2019). These knowledge gaps raise the possibility the domestic market is misinformed about the environmental and nutritional comparisons of PBAs and Australian chicken.

Aims/objectives

The project outcome is a report for the AgriFutures Chicken Meat Program that provides a robust comparison of Australian chicken meat and PBAs, based on a critical review of the literature and a scoping life cycle assessment (LCA) comparing Australian chicken meat with PBAs available on the supermarket shelf.

Specific objectives of this project are:

1. Conduct a critical literature review of environmental impacts and nutritional value of PBAs and cultured meats, assessing impacts from plant production and key additives, manufacturing (including the impact of the country of manufacture and transport on energy use and emissions), nutrition profile, and the comprehensiveness and representativeness of the analysis.

2. Determine requirements for a 'like-for-like' comparison of environmental impacts between chicken meat and up to four common PBAs and a cultured meat product, based on Australian chicken meat LCA results, and the literature. The like-for-like comparison will use portions

that balance nutrition and culinary requirements and will investigate assessment methods required to ensure equivalent supply chain attributes for products sold in the Australian market.

3. Conduct a preliminary LCA scoping study comparing PBAs and cultured meat to Australian chicken meat using readily available datasets, augmenting the literature review by enabling sensitivity analysis and uncertainty analysis.

Results/key findings

The net result of our analysis of the literature and the scoping analysis results suggested that the comparison was much closer to equivalent than is commonly claimed. For example, the claimed 70 % lower carbon footprint of Quorn than chicken is much more than our scoping study results, which showed 15% higher impacts for chicken than Quorn, which considering the degree of uncertainty in the results is marginal and likely to be not statistically significant. For other impact categories of interest, the scoping results showed impacts such as fossil fuel energy may be lower for chicken, though this was very sensitive to the region of the world where the PBA was manufactured. This finding was similar with other products available on the Australian market.

Implications for relevant stakeholders

In the present study, we found it was possible to determine comparable products considering nutritional quality. Interestingly, this resulted in less mass of the plant-based product being required for the comparison than the chicken product. In other words, comparisons made without accounting for nutritional aspects would slightly disadvantage the PBA, not the chicken product.

Our review of the literature showed that soy-based and laboratory meat products could have higher impacts than chicken meat, and that these findings were sensitive to assumptions. Particularly with respect to soy, the impact of LUC emissions in some regions of the world is highly significant.

Multiple PBA products marketed to compete with chicken in the Australian market utilise environmental claims to support their product with several claiming very large differences between the PBA product and either 'meat' or more specifically in the case of Quorn, with chicken.

We observed several problems when evaluating these findings. Firstly, the comparisons were all done in overseas markets, where impacts from chicken are typically higher, and impacts from PBAs are expected to be lower because the transport requirement for these products from country of manufacture to Australia were not taken into account. Thus, there is doubt about the claims when they are made in the Australian market.

Further to this, the actual magnitude of the differences between chicken and PBAs should be taken into account; for a given portion, the difference between chicken and the pea-based products is less than the emissions from driving a car about 1km. In the context of choices available to consumers, other factors are more significant than this issue, but current marketing mistakenly emphasises the importance of this dietary choice.

Methods used

A search of the literature was conducted to find relevant PBA and cultured meat LCA studies. Studies were screened to ensure only those that assessed actual products that could be used to replace chicken meat in a meal. The overall quality of each study was assessed, and the findings were summarised. These studies were also utilised to provide data for the scoping analysis.

Three PBAs were compared to chicken meat (CM); a pea-based product (PB), Quorn pieces (QP; a mycoprotein-based alternative), and laboratory meat cultured from cyanobacteria (LM). A cradle-to-consumption scoping LCA was then conducted for products consumed in Australia. A sensitivity

analysis was performed on the production and processing of the PBAs either being produced in Australia or produced and imported from abroad. The chicken meat was assumed to be produced in Australia, based on previous research (Wiedemann et al., 2017)

Conclusions and recommendations

The literature review highlighted the variability in methods applied, particularly regarding system boundaries, functional units, allocation and location of production. More detailed research into these evolving meat alternative technologies is needed. Production conditions and product inputs for each product vary between different countries. Many PBA and cultured meat LCAs relied heavily upon secondary data, calculations and assumptions. The use of up to date primary data would allow for a more accurate comparison between the products and traditional meats.

The nutritional qualities of meats, plant-based alternatives and cultured meats vary and therefore, a comparison of environmental impacts should be based on nutritional equivalence. In order to make fair and meaningful comparisons between chicken and plant-based food alternatives, Nutrition Quality Corrected Functional Units (NQCFU) were developed based on nutritional equivalence (based on a nutrition quality score) and culinary equivalence (incorporating serving size) that incorporate a cooking method to represent foods as consumed. Future research could utilise a system such as this to ensure the different nutritional profiles of each product are considered. Interestingly, considering the comparisons made here, we found that chicken meat is not disadvantaged by being compared on a 'product' basis without accounting for nutritional aspects meaning this aspect may be less critical.

The results of our scoping study and literature review indicated that while there are noticeable differences in some impact categories between products, the impacts of chicken meat and PBAs are overall relatively similar. Specifically, the study found that:

1. The differences are relatively modest and not consistent: GHG impacts were slightly higher for Australian chicken meat compared to PBA, though the differences are modest when compared with other common activities. In contrast, the impacts were often higher for PBAs when energy and water is considered.

2. In comparison with laboratory-based meat, Australian chicken meat typically had lower environmental impacts across most categories, though it is noted the laboratory meat studies are heavily reliant on assumptions as few operational production processes exist at present.

A comparison between selected PBA products promoted in the Australian market for their apparent superior environmental credentials over traditional meats found that the literature and the scoping analysis did not strongly support many of the claims and in some cases, the claims were contradicted. This was partly because the supporting results were not based on the Australian market context. This implies that consumers may be misinformed by these claims in the Australian market.

Based on these findings, we recommend that the study is progressed according to the project plan, to complete a comparative analysis focused on chicken meat compared to PBAs in the Australian market.

Introduction

Increasing ethical and environmental concerns of consumers surrounding the production meat products has resulted in the emergence and expansion of meat alternatives in the market. These products include plant-based alternatives (soy, wheat, peat, oat, etc), animal-based alternatives (milk and insects) microbial products (mycoprotein), and cultured meats (Filho et al., 2019). Plant-based alternatives and meat analogues are manufactured food products which are designed to mimic the taste and texture of meat products. These products sometimes claim to reduce the environmental impacts caused by livestock production.

The marketing and consumption of vegetarian, vegan and reduced meat diets are rapidly increasing. According to the Good Food Institute, sales of plant-based alternatives to meat (PBAs) grew 38 % in the US in the past two years, compared to total growth in retail food of 4 % over the same period (The Good Food Institute, 2020). In Australia, surveys show a substantial increase in the number of consumers identifying as vegetarian and vegan in the past five years. In the food market, PBAs and cultured meat companies are actively competing with traditional meat, using environmental credentials as a major point of difference. However, while multiple comparisons have been done between PBAs and beef (Goldstein et al., 2017; Heller & Keoleian, 2018; Lynch & Pierrehumbert, 2019) few have been done comparing PBAs and chicken. These studies were conducted overseas, raising the possibility that the findings were influenced by factors that are dissimilar to Australian conditions. Previous research showed that Australian chicken meat has relatively low environmental impacts compared to other meat products (Wiedemann, 2018) and it is not clear if comparisons between PBAs and chicken meat in the Australian market would reach similar conclusions to the limited research from overseas. Moreover, product comparisons should consider nutritional factors, but this is often not done. For example, Willett et al. (2019) compared chicken with dry soybean, which are nutritionally very different products. Thus, robust knowledge regarding environmental impacts and nutritional comparability of PBAs and chicken meat in Australia requires further research.

Objectives

Specific objectives of this project were:

1. Conduct a critical literature review of environmental impacts and nutritional value of PBAs and cultured meats, assessing impacts from plant production and key additives, manufacturing (including the impact of the country of manufacture and transport on energy use and emissions), nutrition profile, and the comprehensiveness and representativeness of the analysis.

2. Determine requirements for a 'like-for-like' comparison of environmental impacts between chicken meat and up to four common PBAs and a cultured meat product, based on Australian chicken meat LCA results, and the literature. The like-for-like comparison will use portions that balance nutrition and culinary requirements and will investigate assessment methods required to ensure equivalent supply chain attributes for products sold in the Australian market.

3. Conduct a preliminary LCA scoping study comparing PBAs and cultured meat to Australian chicken meat using readily available datasets, augmenting the literature review by enabling sensitivity analysis and uncertainty analysis.

Methodology

Review

Literature search strategy

To find articles relevant to the specific research question Google Scholar was used. The literature search used keywords including "environmental", "impact", "LCA", "chicken meat", "plant-based alternative", "meat alternative", "meat analogue", "cultured meat", "lab meat", "haem", "pea", "soy", "tofu", "wheat", "mycoprotein", and "Quorn". The language was restricted to English, and no time restrictions were set. Articles were also found through a search of citations in relevant articles. This was done to ensure all relevant articles were included.

Inclusion and exclusion criteria

The focus for this literature review was on LCA studies that assessed a product comparable to chicken meat which could be used to replace chicken meat in a meal. With the exclusion of haem, other major ingredients in plant-based alternative products were screened out as they are not a comparable product.

The critical review assessed a total of 21 articles which covered 41 products (including different meats and PBAs) and focused on i) the coverage of impact categories (GHG, energy use, land occupation, water use, eutrophication and acidification), ii) system boundaries, iii) functional unit, iv) country of manufacture, v) key ingredients, and vi) the overall quality of the article.

A review of nutritional properties to enable comparison of PBAs and chicken meat was also conducted based on human nutrition requirements. In order to make fair and meaningful comparisons between chicken and plant-based food alternatives, Nutrition Quality Corrected Functional Units (NQCFU) will be developed based on nutritional equivalence (based on a nutrition quality score) and culinary equivalence (incorporating serving size) that incorporate a cooking method to represent foods as consumed.

Scoping study

Three PBAs were compared to chicken meat (CM); a pea-based product (PB), Quorn pieces (QP; a mycoprotein-based alternative), and laboratory meat cultured from cyanobacteria (LM). A cradle-to-consumption scoping LCA was then conducted for products consumed in Australia. A sensitivity analysis was performed on the production and processing of the PBAs either being produced in Australia or produced and imported from abroad. The chicken meat was assumed to be produced in Australia, based on previous research (Wiedemann, et al., 2017)

Functional unit

A Nutritional Quality Corrected Functional Unit was applied to establish functional comparability between chicken, PBAs and laboratory meat. Impacts were reported for a 0.1 kg portion of chicken meat nutritional equivalent and equivalencies are presented in the scoping study results section.

Inventory data and impact assessment

Impact assessment included global warming and water stress, and aggregated inventory results for fossil fuel energy use, freshwater consumption and land occupation calculated using methods described previously (Wiedemann, et al., 2017). The available datasets and impact assessment methods were insufficient to enable assessment of a broader range of impact categories. However, it is noted that other impacts, including chemical toxicity and eutrophication, may be relevant for the product, and these require further investigation in the future. Inventory data are shown in the Appendix.

Sensitivity analysis

Imported product vs domestic production

A sensitivity analysis was run looking at the impacts of producing and importing the different PBA products from outside of Australia. The impacts of producing and importing each product from the UK, EU and US were investigated. In the case of the pea-based product, we understand some such products are made and imported from New Zealand, and the impacts of this were modelled alongside the other points of origin. To calculate the differences between different regions of production, the electricity supply, water supply and transport distances were specified. All transport of the products was assumed to be frozen past the point of processing and packaging.

General sensitivity analysis

A general sensitivity analysis of each product was run to establish the most sensitive inputs. This information was used to refine parameters and increase the accuracy of the LCA. Further refinement of all parameters, especially sensitive parameters should be completed should a full LCA be completed in the future.

Review of environmental impacts

Environmental impacts of chicken and other meats

Chicken

Global meat consumption is rising and, consequently, the production of meat continues to increase. The main drivers in this increase in consumption are growing global populations, increased consumption per capita and a shift in diet towards meat products (Alexandratos & Bruinsma, 2012). Specifically, in Australia, there has been a shift from red meat to white meat consumption, with white meat consumption levels now higher than red meat (Wong et al., 2015). In addition to this growing demand, industries are also facing increasing pressure to quantify and reduce the environmental impacts associated with the production of chicken meat (Leinonen et al., 2012; Wiedemann, et al., 2017). The major environmental impacts of chicken meat production reported in the literature include GHG emissions, non-renewable energy consumption, land occupation, water consumption, eutrophication and acidification. In the Australian context, more focus has been placed on GHG emissions, energy and water. Eutrophication is a relevant environmental impact but has received less attention in the literature. Acidification, which refers to atmospheric acidification from aerial emissions of acidifying gases, is less relevant in Australia because the large landmass, topography and relatively low levels of industrialisation and intensification of livestock production mean impacts are less apparent than in some regions such as Europe.

Chicken is produced in three main systems: 'conventional' systems where birds live on the floor of the shed on bedding material referred to as litter, often with cooling and heating as required; 'free-range' where birds have access to an outdoor range area, and 'organic' where birds have access to an outdoor range area and are also fed an organic diet with only approved medications and feed additives. These different systems can have different levels of environmental impact, with some being more environmentally efficient than others. Regardless of the production system, chicken meat has a low environmental impact compared to the production of other meat such as beef, lamb and pork (Roy et al., 2011; Wiedemann, 2018; Williams et al., 2006) The improved environmental efficiency of meat chicken systems in comparison to other species can be attributed to the bird's fast growth rates and low feed conversion ratio.

There is a general consensus among the literature that feed production for meat chickens is the largest contributor to environmental impacts along the supply chain (Leinonen et al., 2012; Prudêncio da Silva et al., 2014; Wiedemann et al., 2017; Williams et al., 2006).

Most chicken meat studies have investigated impacts from production, including primary production to the farm gate (Leinonen et al., 2012; Pelletier, 2008; Williams et al., 2006) or primary production and meat processing (Prudêncio da Silva et al., 2014; Wiedemann et al., 2017). Some studies, such as Smetana et al. (2015), included all stages up to the point of consumption (cradle-to-plate). Thus, the comparison of impacts is difficult due to the different system boundaries used, functional units and different data sets. Nevertheless, the studies individually provide an overview of environmental impacts from different meat chicken production systems in different countries. The results from these six studies are reviewed in the following sections based on the impact category.

GHG emissions

All seven studies that were reviewed reported GHG emissions for chicken meat production. When compared to other meat products typically produced in Australia, such as beef, lamb and pork, the production of chicken meat emits significantly lower GHG emissions (Nijdam et al., 2012; Roy et al., 2011; Wiedemann, 2018).

A study conducted by Wiedemann et al. (2017) which investigated resource use and environmental impacts from chicken meat in Australia found GHG emissions ranged from 2.2 to 3.4 kg CO_2-e/kg carcass weight (CW). Emissions were lower in free-range systems (2.2 CO_2-e/kg CW), while in conventional systems results were higher, ranging from 2.8 – 3.4 CO_2-e/kg CW, though this was because of differences that led to lower impacts from the free range diet, rather than the production system. These results considered emissions from feed production (including land-use (LU) and land-use change (LUC)), meat processing, and the growth phases. Inclusion of the meat processing stage in the GHG emission results increased emissions by 8 %. Another Australian study conducted by Bengtsson and Seddon (2013) reported Australian meat chickens to emit an average of 2.6 kg of CO_2-e per kg of live weight (LW), which is higher than the values reported by Wiedemann et al. (2017) when converted to CW basis. The study also excluded assessment of (LU) and (LUC) so is likely to be an under-estimate if any imported soymeal was used in the ration.

In contrast to the findings of Wiedemann et al. (2017), which showed GHG emissions to be lower in Australian free-range systems, Leinonen et al. (2012) found free-range systems in the United Kingdom (UK) to emit more GHG emissions than conventional systems. Leinonen et al. (2012) studied the environmental impacts of chicken meat production in the UK. The study found conventional systems to emit 4.4 kg of CO_2-e/kg edible CW and the free-range system emitted 5.1 kg of CO_2-e/kg of edible CW. These results were similar to values reported in another UK meat chicken study (Williams et al., 2006). However, Williams (2006) did not take into account land-use change while Leinonen et al. (2012) did. Neither of these studies considered meat processing, and therefore emissions were underestimated. Pelletier (2008) found chicken meat systems from the United States of America (US) to emit 1.4 kg of CO_2-e/kg LW. These values excluded LU and LUC and are comparable to results from Wiedemann et al. (2017) study which showed GHG emissions excluding LU and LUC to range from 1.1 to 1.3 kg CO_2-e per kg of LW. Pelletier (2008) did not consider the meat processing stage.

A study conducted by Prudêncio da Silva et al. (2014) on meat chicken systems in France and Brazil showed higher GHG emission in French systems in comparison to the Brazilian systems. GHG emissions in the French system ranged from 2.2 kg CO_2-e /kg LW to 2.7 kg CO_2-e/kg LW. In the Brazilian system, GHG emissions were reported to be between 1.5 kg CO_2-e/kg LW and 2.1 kg CO_2-e/kg LW. A Finnish study conducted by Katajajuuri (2008) reported GHG emissions from meat chicken production to be 2.07 kg CO_2-e/kg LW. These results show a trend towards higher impacts from chicken meat produced overseas compared to Australian production, when the system boundaries and method choices are harmonised.

Numerous studies have identified the production of feed for meat chickens as being the largest contributor to GHG emissions in the supply chain (Leinonen et al., 2012; Prudêncio da Silva et al., 2014; Wiedemann et al., 2017). In the US meat chicken supply chain, the production of feed was responsible for 82 % of GHG emissions (Pelletier, 2008). Similarly, Prudêncio da Silva et al. (2014) reported that 83 % of all GHG emissions from the meat chicken supply chain in France and Brazil could be attributed to the production of feed. Leinonen et al. (2012) reported feed production to contribute 71 – 72 % of total GHG emissions. In Australia, the production of feed represented the largest contribution to GHG in the supply chain, accounting for 64 – 75 % (including LUC and LU) of total GHG emissions (Wiedemann et al. 2017). Although being the largest contributor to GHG emissions in the supply chain, feed production in Australia contributes less GHG emission in comparison to other countries. This can be attributed to the lower nitrous oxide emissions during cropping and lower inputs relative to crop yield for Australian crop production (Wiedemann et al., 2017).

Smetana et al. (2015) reported emissions from the production of chicken meat to range between 5.2 and 5.8 kg CO_2-e per kg of ready-to-eat chicken. These results are not directly comparable to other studies due to the differences in system boundaries and functional units used. This study used a cradle-to-plate system, therefore incorporating impacts from cooking the product, which could result in higher results in comparison to other studies.

Energy use

All studies reviewed reported energy use for chicken meat production. Wiedemann et al. (2017) reported that fossil fuel energy demand to range from 18.1 – 21.4 MJ/kg CW for conventional systems, while in free-range systems energy demand was 18.3 MJ/kg CW. Fossil fuel energy use included meat processing, grow out and breeding phases and the production of feed. These values are lower than those reported by Leinonen et al. (2012) for meat chickens in the UK because their conventional system's energy use was 25.4 MJ/ kg CW, 25.65 MJ/ kg CW for free-range systems and 40.3 MJ/ kg CW for organic systems. Energy use results for UK meat chicken systems reported by Williams et al. (2006) are significantly lower than Leinonen's results and slightly lower than Wiedemann et al. (2017) results. Values were reported on a per kg CW basis. Conventional systems used 12 MJ, free-range required 14.5 MJ and organic energy demand was 15.8 MJ. Both studies, conducted by Leinonen et al. (2012) and Williams et al. (2006), did not include energy use in the meat processing stage; therefore, results were underestimated. Other studies have found meat processing to contribute between 9 – 22 % of total energy use (Prudêncio da Silva et al., 2014; Wiedemann et al., 2017).

In US meat chicken systems Pelletier (2008) reported energy use to be 15.0 MJ/ kg LW. These results were similar to those published in a Finnish study conducted by Katajajuuri (2008), which found energy use values to be 16.0 MJ per kg LW. These energy use values are lower than those found by Prudêncio da Silva et al. (2014) who reported 18.0 – 19.1 MJ/ kg LW of energy use in Brazilian systems and 19.1 – 29. MJ/kg LW of energy use in French systems. When results from the study conducted by Wiedemann et al. (2017) are converted into LW basis results were 11.5 – 13.1 MJ/kg for conventional systems, which is lower than the US, French and Brazilian results. Although using a different functional unit to other studies, the energy use values reported by Smetana et al. (2015) appear significantly higher than the other studies, with values ranging from 51.6 – 63.4 MJ/kg of ready to eat product. The assessment conducted by Smetana et al. (2015), included impacts generated after the farm gate, such as processing, transport and cooking, which may explain these higher results.

According to Wiedemann et al. (2017), over half (53 – 59 %) of the total fossil fuel energy demand was used in the feed production process, particularly in field operations and the manufacture of fertiliser in Australian systems, while Pelletier (2008) reported that 80 % of energy use in the US meat chicken supply chain was related to the production of feed. The production of feed was also the major contributor to energy use in French and Brazilian systems, demanding 71 % and 57 % respectively. The meat processing stage also required a large portion of energy (22 % for Brazilian systems, 9 % for French systems and between 13 – 16 % in Australian systems (Prudêncio da Silva et al., 2014; Wiedemann et al., 2017).

Similarly to Leinonen et al. (2012), who conducted a study on meat chicken systems in the UK, Williams et al. (2006) also found organic systems to have the greatest energy demand, followed by free-range and then conventional systems with the lowest contribution. High energy use values in UK organic systems is due to the importation of feed ingredients resulting in increased energy demand for transportation (Leinonen et al., 2012). Williams et al. (2006) reported that organic systems have a higher feed conversion ratio. Growing periods are typically longer in organic systems, which requires increased energy demand.

Land occupation

Six out of the seven studies reported land occupation requirements for chicken meat. Wiedemann et al. (2017) found that arable land occupation ranges from 14.0 to 22.5 m^2/kg CW, while in free-range systems arable land occupation was 18.2 m^2/kg CW. Leinonen et al. (2012) reported land occupation values of 5.6 m^2/kg CW for conventional systems, 7.2 m^2/kg CW for free-range and 25 m^2/kg CW in organic systems. The values comparable to those reported by Williams et al. (2006) who found conventional systems to use 6.4 m^2/kg CW, 7.3 m^2/kg CW for free-range systems and 14.0 m^2/kg CW for organic systems. Organic systems in the UK have relatively high land occupation values which are

likely a result of lower yields in organic crops and consequently higher land area requirement. Land occupation in Brazilian systems were reported to be 2.5 m^2/kg LW and 2.6 to 3.9 m^2/kg LW in French systems (Prudêncio da Silva et al., 2014). Land occupation values from Finnish systems were reported to be higher at 5.5 m^2/kg LW (Katajajuuri, 2008). The higher values in Australia were associated with lower crop yields for Australian crops than most other regions of the world.

Eutrophication

Eutrophication impacts were reported in four of the studies reviewed. Eutrophication principally considers nitrogen and phosphorus losses to water. In meat chicken production systems, these losses can arise from crop production, or manure management, particularly in free-range systems.

Leinonen et al. (2012) found eutrophication potential to be relatively similar for conventional and free-range systems. In conventional systems, the eutrophication potential was 0.023 kg of PO_4 equivalent and 0.024 kg of PO_4 equivalent in free-range systems. Williams et al. (2006) reported higher eutrophication potential in UK systems. The eutrophication potential of conventional systems was 0.049 kg PO_4 [3]equivalent, in free-range systems, it was 0.063 kg PO_4 (Williams et al. 2006). Prudêncio da Silva et al. (2014) reported significantly lower eutrophication emissions in both French and Brazilian systems. French systems ranged from 0.013 to 0.019 kg PO_4^3 equivalent and in Brazilian systems it was reported to be 0.014 kg of PO_4^3 equivalent.

In US systems, Pelletier (2008) reported a significantly lower eutrophication potential result 0.0039 kg of PO_4 equivalent. Pelletier (2008) reported that feed production accounted for 97 % of the eutrophying emissions associated with the production of meat chickens in the US (Pelletier, 2008). Juuri (2008) also reported similar results for Finnish systems (0.002 kg of PO_4^3 equivalent). In contrast, Leinonen et al. (2012) reported about half of the eutrophication potential was due to feed production, with manure also having a high eutrophication potential (Leinonen et al. 2012). Prudêncio da Silva et al. (2014) reported contributions to eutrophication potential in the French system were largely from the production of feed (57 %) and the chicken production stage (36 %) of eutrophication potential. The meat processing stage accounted for an additional 7 %. Contributions to eutrophication differed slightly in the Brazilian system, with 69 % of the eutrophication potential related to feed production, 23 % related to the production of birds and 8 % due to the meat processing stage. Emissions from the chicken sheds and the use of fertilisers were major contributors to eutrophication potential. The Australian study by Wiedemann et al. (2017) did not assess eutrophication.

Acidification

Acidification potential varied throughout the literature and was reported in four of the six studies reviewed. A major source of acidification potential is NH_3 emissions from manure management, and SO_2 emissions from fossil fuel combustion (Leinonen et al., 2012). Leinonen et al. (2012) reported acidification potential results to be 0.05 kg of SO_2 equivalent in conventional systems and 0.06 kg of SO_2 equivalent in free-range systems. These results were significantly lower than those reported by Williams et al. (2006). They found acidification potential to be 0.17 kg SO_2 equivalent in conventional systems and 0.230 kg SO_2 equivalent in free-range systems.

Pelletier (2008) reported acidification potential in US meat chicken systems to be 0.02 kg SO_2 equivalent. The acidification potential of French Systems was reported by Prudêncio da Silva et al. (2014). They found it to range between 0.03 and 0.05 kg SO_2 equivalent, while for Brazilian systems acidification potential ranged between 0.03 and 0.04 kg SO_2 equivalent. Similarly, Katajajuuri (2008) reported acidification in Finnish systems to be 0.04 kg SO_2 equivalent.

Prudêncio da Silva et al. (2014) found that the largest contribution to acidification in French systems was a result of the chicken production stage, specifically the emissions from the chicken sheds. These emissions ranged from 69 – 77 % of total acidification potential. In Brazilian systems, the main contributors to acidification were the chicken sheds (44 – 48 %) and the use of urea N fertiliser in the production of maize for chicken feed (39 – 44 %) (Prudêncio da Silva et al., 2014). Pelletier (2008)

reported that feed contributed to 96 % of acidifying emissions. Leinonen et al. (2012) found that ammonia emissions from manure were a primary source of acidification potential in UK meat chicken systems. Acidification potential from manure was particularly high in organic systems because of the long production cycle.

Freshwater consumption

Only two studies assessed the water consumption of chicken meat production and of these, the latter only included direct water use. Wiedemann et al. (2017) found freshwater consumption to range from 38 – 111 L/kg CW in conventional Australian meat chicken systems, while in free-range systems, freshwater consumption was 70 L/kg CW. Freshwater consumption is expected to be higher in Australia as a result of increased cooling requirements in meat chicken sheds and potentially increased crop irrigation. 69 – 86 % of water consumption in the supply chain was related to feed production, in particular irrigation of grain crops. The provision of drinking and cooling water during the grow-out phase required 5 – 21 % of total freshwater consumption. Feed inputs, irrigation of feed, housing and meat processing were included in freshwater consumption results.

Leinonen et al. (2012) reported that water use values to range be 4.41 L/kg CW in conventional systems and 6.86 L/kg in free-range. These values incorporated drinking and cleaning water; however, water consumption in crop production was not included, which explained the lower values in comparison to the previous study. Other studies did not report water use in detail.

Table 1. Summary of environmental impacts of chicken meat production across the literature

Study	FU	Country	Exclusions	Climate change (kg CO₂-e)	Energy (MJ)	Land occupation (m²)	Acidification (kg SO₂ equiv.)	Eutrophication (kg PO₄ equiv.)	Water consumption (L)
Wiedemann et al. (2017)	1 kg CW	Australia		2.5 – 3.1	18.1 – 21.4	19.5 – 31.3	-	-	25.1 – 29.8
Leinonen et al. (2012)	1 kg edible CW	UK	Meat processing	4.4 – 5.7	25.4 – 40.3	5.60 – 25.0	0.05 – 0.06	0.023 – 0.024	4.4 – 6.9
Williams et al. (2006)	1 kg CW	UK	Land use change Meat processing	4.57 – 5.48	12.0 – 15.0	6.4 – 7.3	0.17 – 0.23	0.049 – 0.063	-
Pelletier (2008)	1 kg LW	US	Land use change	1.4	15.0	-	0.02	0.004	-
Prudêncio da Silva et al. (2014)	1 kg LW	Brazil and France	Land use change	1.5 – 2.7	19.1 – 29.5	2.5 – 3.90	0.03 – 0.05	0.014 – 0.019	-
Katajajuuri (2008)	1 kg LW	Finland		2.1	16.0	5.50	0.04	0.002	-
Smetana et al. (2015)	1 kg ready to eat			5.2 – 5.82	51.6 – 63.4	3.9 – 3.9	-	-	-

Notes: The studies are reported here as recorded in the original studies and utilise different functional units and apply different methodological choices. Results are not directly comparable between studies and are provided for indicative purposes only.

While these results were not directly comparable, when the studies were harmonised to the extent possible, we found a trend showing lower GHG impacts and higher land occupation from Australian chicken meat compared to international chicken meat studies. This result indicates that to be valid, comparisons for the Australian market would need to focus on Australian chicken meat. Considering this, only results from the Australian study were deemed relevant for drawing comparisons with alternatives.

Other meats (beef, lamb and pork)

When compared to other meats such as beef, lamb and pork, chicken has lower environmental impacts. Wiedemann (2018) reported environmental impacts of Australian beef, lamb, pork and chicken on a per kg of boneless fat corrected meat basis, reporting impacts separate of land use and direct land use change. The study reported GHG emissions for Australian pork ranged between 4.4 – 9.3 kg CO_2-e per kg of boneless fat corrected meat. Australian lamb emissions ranged from 17.2 – 22.3 kg CO_2-e per kg of boneless fat corrected meat, and Australian beef emissions ranged from 21.5 – 29.5 kg CO_2-e per kg of boneless fat corrected meat. The production of Australian chicken meat generated lower GHG emissions in comparison, with values ranging from 2.5 – 3.1 kg CO_2-e per kg of boneless fat corrected meat.

Fossil energy demand range from 30 – 38.2 MJ per kg of boneless fat corrected meat for pork, 14.1 – 28.3 MJ per kg of boneless fat corrected meat for lamb and 19.8 – 47.8 MJ per kg of boneless fat corrected meat for beef (Wiedemann, 2018). Fossil energy demand for Australian chicken ranged from 25.5 – 29.8 MJ per kg of boneless fat corrected meat (Wiedemann, 2018).

Land occupation ranged from 120.4 – 946.6 m^2 per kg of beef 8.8 – 8005.6 m^2 per kg of lamb, and 20.6 – 30.5 m^2 per kg of boneless fat corrected pork (Wiedemann, 2018). The land occupation required for chicken meat production is comparable to that required for pork, ranging from 19.5 – 31.3 m^2 per kg of boneless fat corrected meat (Wiedemann, 2018). It was noted that lamb and beef utilised a large proportion of rangeland areas for grazing while chicken and pork was reliant on arable land for crop production. As rangeland is not generally able to be converted to cropland and is less intensive, comparison with cropland is of limited value.

Water required to produce Australian beef was reported to be 1162.8 L per kg as a national average, while for lamb, water use ranged from 162.6 – 644.8 L per kg, and for pork, water use ranged from 44.6 – 315.1 L per kg (Wiedemann, 2018). Water required for Australian chicken meat production ranged from 52.7 – 154.7 L per kg of boneless fat corrected meat (Wiedemann, 2018). This comparison of environmental impacts of Australian meats which utilised common system boundaries indicates that chicken has lower impacts than other meats in most impact categories. For this reason, comparative studies that aim to demonstrate the environmental impacts of plant-based products compared to meat must specify the type of meat.

Table 2. Environmental impacts of Australian chicken, pork, lamb and beef.

Meat	Study	FU	Country	Climate change (kg CO_2-e)[1]	Energy (MJ)	Land occupation (m²)	Water consumption (L)
Chicken	Wiedemann (2018)	kg boneless fat corrected meat	Australia	2.5 – 3.1	25.1 – 29.8	19.5 – 31.3	29 – 50.7
Pork	Wiedemann (2018)	kg boneless fat corrected meat	Australia	4.4 – 9.3	30 – 38.2	20.6 – 30.5	44.6 – 315.1
Lamb	Wiedemann (2018)	kg boneless fat corrected meat	Australia	17.2 – 22.3	14.1 – 28.3	8.8 – 8005.6	162.6 – 644.8
Beef	Wiedemann (2018)	kg boneless fat corrected meat	Australia	21.5 – 29.5	19.8 – 47.8	120.4 – 946.6	274.4 – 1162.8

[1] Does not include impacts from land use and direct land use change

Environmental impacts of plant-based alternatives and cultured meats

Thirteen studies that assessed the environmental impacts of 31 plant-based alternative products and cultured meats through life cycle assessment (LCA) were included in this literature review. This literature review focused on products that could be used as a substitute for chicken meat in the diet. Six categories of meat alternatives and one major additive were included; Mycoprotein, wheat-based, soy-based, wheat-soy based, pea-based, cultured meats and haem. The impact categories reported varied between studies. All the studies reviewed reported GHG emissions, seven assessed energy consumption, eight assessed land occupation, four included water use, two assessed eutrophication potential and two assessed acidification that was generated by the production of plant-based alternatives. The system boundaries and functional units of each study varied. The impacts of each product are summarised in Table 3.

Mycoprotein

Mycoprotein is an edible protein source that is made up of filamentous fungal biomass. It is obtained from a naturally occurring fungus, *Fusarium venenatum* and is produced via the continuous fermentation of the fungus on a defined medium. Typically, a wheat derived glucose source is used as the carbon source and ammonium as the nitrogen source (Wiebe, 2004). Mycoprotein can also be produced using various other crop-based carbon sources such as sugar beet (Harrison, 2020). Mycoprotein products have been commercialised by Quorn, a UK company, and sold in 19 different countries (Souza Filho et al., 2019). This review includes six studies which assessed the environmental impacts of mycoprotein by conducting a life cycle assessment. Five out of six of the studies focused specifically on Quorn products. All studies reported GHG emissions, three reported energy consumption and four reported land occupation.

GHG emissions

Finnigan (2010) reported GHG emissions for mycoprotein to be 3.1 kg CO_2-e per 1 kg of product and Hsu et al. (2018) reported GHG emissions from mycoprotein to be 1.1 kg CO_2-e per 1 kg of product. For Quorn pieces, Tuomisto et al. (2014) reported GHG emissions to be 5.6 kg CO_2-e per 1 kg. Head et al. (2011) conducted a life cycle assessment on a Quorn mince product and reported GHG emissions to be 2.4 kg CO_2-e per 1 kg of ready to eat product. Similarly, Tuomisto and Roy (2012)

reported GHG emission of 2.3 kg CO_2-e per kg of product and Blonk et al. (2008) found GHG emissions of a mycoprotein based product to be 2.6kg CO_2-e per 1 kg of ready to eat product. Hsu et al. (2018) reported the lowest GHG emissions for a Quorn product – 1.7 kg CO_2-e per 1 kg of product and 1.1 kg CO_2-e per 1 kg of product for mycoprotein. Smetana et al. (2015) found GHG emissions to range from 5.6 – 6.2 kg CO_2-e per 1 kg of ready to eat mycoprotein based product.

The system boundaries of the peer-reviewed study conducted by Smetana et al. (2015) were cradle-to-cooking. In contrast, studies conducted by Head et al. (2011) and Blonk et al. (2008) used cradle-to-the point of retail as system boundaries, which are not inclusive of any impacts which occur after the point of retail such as transport and emissions generated by the consumer at the cooking stage. The lack of incorporation of the cooking stage may result in results being underestimated and makes a comparison between other studies difficult. The mycoprotein product assessed by Smetana et al. (2015) had the highest GHG emissions, which, to some extent, was explained due to the additional stages included in the assessment. However, being a mycoprotein product rather than a Quorn product, which has undergone additional processing, it would be expected to be lower in comparison to a Quorn product (when using the same system boundaries). Tuomisto and Roy (2012) and Finnigan (2010) reported impacts from the cradle-to-factory gate. Hsu et al. (2018) did not state what system boundary was used for the Quorn product. A cradle-to-gate boundary was used for the meats and soy products in the study, but the boundary used for the Quorn products were not reported. For a fair comparison between the Quorn and other products in the study, it would be expected that a cradle-to-gate boundary was also used for the Quorn product. However, this is not confirmed. The methodology for assessing impacts generated by meat and soy are reported in detail. However, the methodology used to calculate the impacts of Quorn products are not reported in the study. Although the study has acknowledged the differences in production methods and resource requirements for each source of protein, without clearly defined system boundaries and methodology for the assessment of Quorn products it is difficult to make a fair comparison between the meats.

The estimates of Finnigan (2010) were based on secondary data and were only inclusive of CO_2 impacts associated with energy and water consumption prior to distribution and consumption, highlighting potential inaccuracy in the results. The study conducted by Smetana et al. (2015) used data from databases, published data including Finnigan (2010) and calculations based on assumptions. Blonk et al. (2008) used data from databases, literature and data obtained from food companies. Tuomisto and Roy (2012) used data from Blonk et al. (2008) as well as data from databases. The only study to use entirely primary data was the study conducted by Hsu et al. (2018). This study has been published by Quorn and uses data from their certified carbon footprints for their products. Country of manufacture was not stated in most studies; however, Quorn products are manufactured in the UK.

The main inputs in the mycoprotein product assessed by Smetana et al. (2015) were molasses from sugar beet, nitrogen fertiliser and egg white. The main ingredients used in the other studies were not stated. All studies used a weight-based functional unit, making results comparable. Tuomisto and Roy (2012) presented results as per tonne of product. These results were divided by 1,000 to give results in kg to compare to the other studies. The mycoprotein product assessed in the study conducted by Head et al. (2011) was Quorn mince, and the Quorn product assessed by Blonk et al. (2008) is unknown. Finnigan (2010) also reported LCA results for a variety of Quorn products. This review has included results for the Quorn Pieces which is marketed as a chicken meat alternative.

Similarly, Hsu et al. (2018) also assessed a variety of Quorn products and mycoprotein. This review includes the results from only mycoprotein and Quorn Pieces. The differences between Quorn products and the difference between mycoprotein and Quorn products, which have undergone additional processing, would account for some of the variations in results. Although the study was the only one to use primary data, the lack of transparency regarding the system boundaries and methodology makes the reliability of the study questionable. The peer-reviewed study conducted by Smetana et al. (2015) was the most comprehensive and transparent, clearly outlining the methods used, the data used and limitations of the study. The study also provided a contribution analysis of each stage to the overall impact and conducted a sensitivity analysis using alternative functional units

and methodology. One limitation of the study is that the country of manufacture of the products was not stated. However, much of the data used was based on European products.

Energy use

Smetana et al. (2015) reported fossil fuel demand for mycoprotein products to range from 60.07 – 76.8 MJ per kg of ready to eat mycoprotein product, while Blonk et al. (2008) and Tuomisto and Roy (2012) both reported fossil fuel demand to be 38 MJ per kg of ready to eat Quorn product. Energy consumption at the processing stage had the largest impact, contributing 45 % of the overall impact. Frying for consumption also had a high energy demand, contributing 25 % of the overall impact (Smetana et al., 2015). The assessment of mycoprotein conducted by Head et al. (2011), Hsu et al. (2018) and Finnigan (2010) did not include fossil fuel demand as an impact category.

Land occupation

Land occupation from mycoprotein and Quorn products vary among the literature. Head et al. (2011) reported land occupation for Quorn to be $0.41m^2$, Smetana et al. (2015) reported $0.79 – 0.84$ m^2 per kg of ready to eat mycoprotein product, Blonk et al. (2008) reported a Quorn product required 1.2 m^2 of land and Tuomisto and Roy (2012) reported Quorn required 1.7 m^2 of land.

Soy-based PBAs

Six studies conducted LCAs on soy-based meat alternative products, covering a total of seven products. Out of the seven products, five of these were tofu and the other two were described as soy-based. All seven studies reported GHG emissions, two reported energy use and five reported land occupation. No other impact categories were reported.

Soy-based products include impacts from crop production, transport, manufacture, packaging, and potentially cooking, depending on the system boundary of the study. One key variable is the inclusion of GHG emissions from land-use change (LUC) in the results, which is a highly significant impact for soybean sourced from some major production regions such as South America (Castanheira & Freire, 2013; Henders et al., 2015). Of the studies reviewed, only one included LUC while the five other studies either did not include impacts from land-use change or did not state whether this was included or not. This suggests impacts from these studies were potentially strongly under-estimated, resulting also in comparisons which are not equivalent or comparable to chicken meat studies that did include land use change.

GHG emissions

Mejia et al. (2017) conducted a partial life cycle assessment to quantify the GHG emissions generated by tofu production in the US. The functional unit used in the study was 1 kg of packaged tofu ready for sale. GHG impacts were found to be 1.0 kg CO_2-e per 1 kg of packaged tofu, though this rose to 1.2 kg CO_2-e per kg when field nitrous oxide emissions were included. Without including field emissions, manufacturing of the product was identified as the largest contributor to GHG emissions, contributing 52 % of total emissions. This was followed by packaging (23 %), soybean production (16 %) and transport (9 %). Soybeans used in the production of the tofu assessed were grown in the US. Soybeans produced in the US can be expected to have lower impacts in comparison to soybeans grown in Brazil due to lower emissions from land-use change (Mejia et al., 2017).

Head et al. (2011) found GHG emissions generated from the production of tofu to range from 2.5 kg CO_2-e per 1 kg of tofu for certified organic tofu to 3.7 kg CO_2-e per 1 kg of tofu for uncertified organic tofu. The certification relates to soy which has been produced on land that has not been transformed or soy from North America or Europe. At the same time, uncertified soybeans could potentially be grown in places such as Brazil on land that has previously undergone deforestation,

resulting in increased emissions due to this land transformation. Smetana et al. (2015) found GHG emissions from a soy-based product to range from 2.7 – 2.8 kg CO_2-e per 1 kg of ready to eat product excluding LUC, increasing to 3.8 - 4.9 kg CO_2-e per 1 kg of ready to eat product when LUC was included. This estimation is based on global LUC data from FAOSTAT and therefore may not be region-specific to the soybeans used. Lower emissions for soy products were reported by Blonk et al. (2008) who found GHG emissions from tofu manufactured in the Netherlands to be approximately 2.2 kg CO_2-e per 1 kg of tofu. Although Blonk et al. (2008) did not state where soybean was grown, the paper stated that soybean producers have claimed that the soy was not produced on deforested land. However, these claims cannot be verified. The peer-reviewed study conducted by Fresán et al. (2019) found GHG emissions from a 65 % soy product were 2.1 kg CO_2-e per 1 kg of product and Tuomisto and Roy (2012) who reported GHG emissions were 2.0 kg CO_2-e per 1 kg of tofu. The lack of transparency regarding the location of soybean production and LUC impacts means results may be underestimated as LUC impacts from soy grown on land which has undergone deforestation are often significant. Reported land-use change values from imported soy from areas such as Argentina (which accounts for 96 % of all soy imports) to be 5.2 kg CO_2-e (Wiedemann & Watson, 2018).

Results from the partial LCA conducted by Mejia et al. (2017), Fresán et al. (2019) and Tuomisto and Roy (2012) would be expected to be lower compared to the other studies as a result of the LCA ending at the factory gate. The system boundary excluded impacts associated with retailing the product, preparation for consumption and disposal. Studies conducted by Head et al. (2011) and Blonk et al. (2008) state that the system boundaries are cradle-to-retail which includes additional impacts involved in the retailing process such as transport and electricity, however, does not include consumption and disposal. In contrast to these studies, Smetana et al. (2015) assessed impacts from the production of soy through to consumption. Therefore results from this latter study are expected to be higher due to the additional impacts at the cooking stage.

LCA data was obtained from published literature and databases for studies conducted by Tuomisto and Roy (2012) and Head et al. (2011). Fresán et al. (2019) utilised primary data from food manufacturers for the LCA. Mejia et al. (2017) also used data from a food manufacturer and obtained additional data related to soybean production from databases and published literature. Smetana et al. (2015) used a combination of database data, data from published literature and primary data for their LCA. Blonk et al. (2008) obtained data from a variety of sources including databases, published literature and from food companies.

Smetana et al. (2015), Mejia et al. (2017) and Fresán et al. (2019) used alternative functional units as part of the sensitivity analysis. These three studies used a primary functional unit which was based on product weight, another based on protein content and the other on the energy content of the product. Smetana et al. (2015) compared multiple meat alternative products and demonstrated the impact that using different functional units can have on the results. This highlights the importance of using a relevant and comparable functional unit as meat substitutes have different nutritional values and attributes.

The findings of Mejia et al. (2017) were the lowest of all studies. However, considering these results did not include LUC or N_2O emissions and used the shortest system boundary, this would explain to some extent the lower results. Head et al. (2011) and Blonk et al. (2008) also investigated the GHG emissions of 1 kg of tofu in the Netherlands at the point of retail. Head et al. (2011) included LUC in the results and reported the highest value out of all the studies. GHG emissions reported by Smetana et al. (2015) did not include LUC but gave an estimation of LUC impacts, increasing the result to be similar to those reported by Head et al. (2011). Fresán et al. (2019) and Tuomisto and Roy (2012) reported similar emissions for a soy-based product up to the factory gate exit. The peer-reviewed studies conducted by Mejia et al. (2017) and Fresán et al. (2019) are comprehensive and transparent. Both used primary data from food processing factories, clearly outlined the methods in detail, location of the production of soybeans, and the system boundaries of the study. Other studies were reliant on secondary data and assumptions, had unclear methodology or were not transparent about the location of soybean production.

Energy use

Fossil fuel demand for soy-based products was only reported in two of the six studies that assessed soy-based products. Smetana et al. (2015) reported energy consumption for a soy-based product to range from 27.8 – 37.0 MJ per kg of tofu at consumption. Energy use at frying was identified as a major impact, contributing 58 % to the overall impact of the soy-based product. Tuomisto and Roy (2012) reported lower energy consumption for the production of tofu at the factory gate (15.6 MJ per kg of tofu).

Land occupation

Tuomisto and Roy (2012) reported land occupation to be 3.0 m^2 per 1 kg of tofu, Blonk et al. (2008) reported 2.6 m^2 per 1kg of tofu at point of sale, Head et al. (2011) reported land occupation for tofu to be 2.1 m^2 per 1 kg of tofu at point of retail and Smetana et al. (2015) reported land occupation to be 1.1 – 1.4 m^2 per kg of ready to eat soy-based product.

Wheat-based PBAs

Two LCA studies investigated the environmental impacts of wheat-based meat alternatives. Both studies included GHG emissions; one included energy usage and land occupation. Other impact categories were not reported.

GHG emissions

The main ingredients used in the gluten-based product in the study conducted by Smetana et al. (2015) was wheat grain and oat hull fibre. Smetana et al. (2015) found GHG emissions generated by gluten-based product to be 3.6 – 4.0 kg CO_2-e per 1 kg of ready to eat product, while Fresán et al. (2019) reported GHG emissions for wheat-based meat alternative products containing at least 65 % wheat to average 2.1 kg CO_2-e per 1 kg of product at the factory gate.

Fresán et al. (2019) included impacts generated in production of raw materials through to the factory gate. Primary data were obtained from food factories over a 12-day period. These data were then audited, ensuring the accuracy of results. The wheat-based product Smetana et al. (2015) assessed was ready for consumption and included impacts from cradle-to-grave, therefore would be expected to have greater impacts due to additional stages such as cooking and transport. Both of these peer-reviewed LCAs were comprehensive and provided transparency concerning the methods used and the data used, however Smetana et al. (2015) obtained data from databases, and published literature results may not be as accurate as the primary data used by Fresán et al. (2019).

Energy use

Smetana et al. (2015) reported that energy use was 39.7 – 49.2 MJ per kg of ready to eat product. Fresán et al. (2019) did not report energy use. The greatest impact of the wheat-based product assessed by Smetana et al. (2015) was energy requirements for frying at the cooking stage.

Land occupation

Smetana et al. (2015) reported land occupation to be 5.5 – 5.8 m^2 per kg of ready to eat product. Fresán et al. (2019) did not report land occupation.

Wheat and soy-based PBAs

Two studies reported the impacts of wheat and soy-based meat alternative products. Both studies only reported GHG emissions. Mejia et al. (2017) assessed 57 different plant-based products and found the average GHG emissions to be 2.2 kg CO_2-e per kg of product. The specific ingredients of each product were not stated as products were grouped by product type (mince, patty, sausage, etc.). However, it was stated that soy and wheat were the main ingredients in most products, but the proportions of each ingredient were not reported. Fresán et al. (2019) used data from Mejia et al. (2017) for their LCA and reported GHG emissions for wheat-soy based product to be 2.3 kg CO_2-e. Mejia et al. (2019) obtained data from three different factories over a 12-day period. These data were reportedly audited, which should increase the reliability of the data and results. Both studies used the same system boundaries: cradle-to-factory gate. Fresán et al. (2019) reported results per 100 g of product, so in the present study we have converted these to be per 1 kg of product to allow for comparison against other products. The country of production of the crops and country of manufacture of the products was not stated in either study. Neither study was clear on whether LUC impacts were included in results, which may lead to a potential underestimation of results. The quantities of wheat, soy and other ingredients for each product were not reported. Therefore, some products may be almost entirely wheat-based, others may be almost entirely soy-based, while others may have more equal proportions of wheat and soy. This lack of transparency surrounding the proportions of ingredients makes grouping all products as wheat-soy based potentially inaccurate.

Pea-based PBAs

Two studies used pea protein as a major ingredient in a plant-based alternative product, and both reported GHG emissions and energy use. However, only one reported land occupation and the other included acidification and eutrophication, and neither of the studies reported water consumption.

GHG emissions

The system boundaries used by Davis et al. (2010) were cradle-to-consumption, whereas Heller and Keoleian (2018) assessed impacts from cradle-to-retail, so additional stages such as transport and cooking which were included by Davis et al. (2010) were not included in this study. Davis et al. (2010) presented results as one meal which was a 275 g pea burger with a protein content of 33.7 g and Heller and Keoleian (2018) presented results as a 0.113 g Beyond Burger patty. The functional units of both products were modified to be show results per kg of product to allow for comparison. The pea burger was not only the pea patty but also included bread, tomato, and water. In contrast, the Beyond burger LCA just included the patty only which was produced from peas, canola oil and coconut oil. It is evident that there are differences between each product, the functional units used, and the system boundaries included in each study, and this makes comparison difficult. The results from Davis et al. (2010) when reported per kilogram and including all ingredients for the burger product, produced in Sweden, were 1.7 kg CO_2-e per 1 kg of product and the same product produced in Spain emitted 4.2 kg CO_2-e per 1 kg of product. Heller and Keoleian (2018) assessed impacts from the beyond burger, which uses pea protein as a major ingredient and found GHG emissions to be 3.5 kg CO_2-e per 1 kg of product.

Energy use

Results from Davis et al. (2010) reported per kilogram of the burger product produced in Sweden were 99.3 MJ per kg product and the same product produced in Spain to consume 137.8 MJ per kg of product. Energy requirements to produce the Beyond burger patty were lower, using 53.1 MJ per kg of product.

Impacts were higher in the Spanish scenario compared to the Swedish scenario. Ninety per cent of Swedish electricity production is based on nuclear and hydropower, while the Spanish electricity is

generated from coal, nuclear and hydropower which results in greater emissions. The improved environmental performance of the Swedish pea burger in comparison to the Spanish pea burger may also be attributed to the lower yield of peas grown in Spain compared with the peas grown in Germany used for the Swedish burger (Davis et al., 2010). These results highlighted the sensitivity of results obtained to the region of production.

Land occupation

Land occupation for the pea-based patty was 2.7 m^2 per kg of product (Heller & Keoleian, 2018).

Eutrophication and acidification

Eutrophication for the Swedish pea burger was reported to be 0.02 kg PO_4 equivalent and 0.05 for the Spanish pea burger. Acidification was reported to be 0.01 for the Swedish burger and 0.04 for the Spanish burger (Davis et al., 2010).

Cultured meat

Cultured meat, also known as lab-grown meat or in vitro meat, is a meat substitute product that is derived from animal stem cells. Tissue engineering techniques are utilised to grow animal muscle tissue in vitro (Tuomisto & Teixeira De Mattos, 2011). Media can be cyanobacteria based or plant-based (Tuomisto et al., 2014). Five studies were reviewed that included an assessment of the environmental impacts of cultured meat. All the studies reported GHG emissions and energy use, but only four studies reported land occupation, only three reported water use, and only one study reported both acidification and eutrophication.

GHG emissions

Smetana et al. (2015) reported GHG emissions of 23.9 – 24.6 kg CO_2-e per kg of cultured meat produced, where the major input was urea used for cyanobacteria cultivation. These results were higher than other published results for cultured meat (Mattick et al., 2015; Tuomisto et al., 2014; Tuomisto & Teixeira De Mattos, 2011; Tuomisto & Roy, 2012). Tuomisto and Teixeira De Mattos (2011) reported GHG emissions for cultured meat of 1.9 kg CO_2-e per kg of cultured meat produced in Thailand, 2.2 kg CO_2-e per kg of cultured meat produced in California and 1.9 kg CO_2-e per kg of cultured meat produced in Spain. Of all the GHG emission produced, 71 % were produced as a result of muscle cell cultivation. The production of cyanobacteria contributed 28 % of GHGs. Tuomisto and Roy (2012) reported GHG emissions to be 1.9 kg CO_2-e per kg of cultured meat, Tuomisto et al. (2014) reported GHG emissions to range from 2.3 – 4.4 kg CO_2-e per kg of cultured meat. Mattick et al. (2015) reported GHG emissions of cultured meat to be 7.5 kg CO_2-e per kg of edible cultured meat.

The LCA of a cultured meat product conducted by Smetana et al. (2015) used data obtained from databases and data from Tuomisto and Teixeira De Mattos (2011). The system boundaries of the study were cradle-to-plate, which considered raw material extraction through to consumer use of the product. Cyanobacteria were used as the feedstock in this study. Mattick et al. (2015) conducted an anticipatory life cycle, analysis using the system boundaries cradle-to-factory gate. Data used in this LCA was obtained from published literature, and the functional unit used was 1 kg of CHO cell biomass. There is a high level of uncertainty associated with Mattick et al. (2015) anticipatory LCA. Although the study was based on published data, it relied heavily on assumptions. Results of this study were higher than those of Tuomisto and Teixeira De Mattos (2011). These results may be higher due to the inclusion of basal media production and the cleaning phase. Higher land occupation requirements can be attributed to the different feedstocks used and additional inputs such as basal media and soy hydrolysate.

The studies conducted by Tuomisto and Roy (2012) and Tuomisto et al. (2014) used system boundaries of production inputs through to the factory gate. Data used in these studies was based off Tuomisto and Teixeira De Mattos (2011). Tuomisto et al. (2014) considered alternative production scenarios including wheat and corn as well as cyanobacteria. These studies were largely based on assumptions, creating uncertainty within the results. In addition, this study focused on impacts from production inputs until the factory gate, therefore cannot be accurately compared against Smetana et al. (2015), which assessed the entire life cycle of the product.

The functional unit used by Tuomisto and Teixeira De Mattos (2011) in their peer-reviewed study was 1,000 kg of a mince type cultured meat product, which had 30 % dry matter content and 19 % of the mass was protein. Results were divided by 1,000 here to allow comparison with other results presented on a 1 kg basis. The study selected three countries, Spain, California, and Thailand due to their differing climatic conditions and availability of cyanobacteria production data in these regions. The system boundaries of this study were cradle-to-factory gate. Impacts related to growth factors, vitamins and cell culture were not included. This was justified by the authors because these amounts were less than 0.1 % of the DM weight, but this could be misleading, considering impacts from highly processed products such as amino acids have been found in previous work by the authors to be a hotspot in animal feed supply chains. At the time of this study, large scale cultured meat production did not exist. Therefore, calculations were based on a hypothetical scenario. Energy inputs were based on many assumptions, creating uncertainty within the results. Additional energy impacts may be generated as a result of the increased energy required to enhance the texture of the product. The study did not include impacts from indirect LUC and the production of energy inputs. The data used to calculate impacts in this study was heavily based on assumptions, increasing the uncertainty of the results.

Results were presented in alternative functional units by Smetana et al. (2015) and Tuomisto and Roy (2012). Both studies presented results primarily on a weight basis and used alternative functional units that took into account the protein content and energy content.

The environmental impacts of cultured meat products are largely based on assumptions rather than actual primary data as the product and production process are under development. The variation between results throughout the literature emphasises this uncertainty. Once the production of this emerging technology begins, and actual data from working commercial-scale cultured meat factories can be used to conduct a full LCA, the true environmental impacts of cultured meat can be fully understood.

Energy use

Smetana et al. (2015) reported energy use to be 290.7 – 373.0 MJ per kg of cultured meat at consumption, while Tuomisto and Teixeira De Mattos (2011) reported significantly lower values of 25.2 – 31.8 MJ per kg of cultured meat at the factory gate. Tuomisto and Teixeira De Mattos (2011) attributed the majority of energy use to muscle cell cultivation. Cultured meat impacts vary depending on the feedstock used (cyanobacteria, wheat, corn, etc.); however, both studies used cyanobacteria (Tuomisto & Teixeira De Mattos, 2011).

According to Smetana et al. (2015), the largest impact of cultured meat production is energy use which accounts for 75 % of all impacts. The majority of this energy is consumed in the cultivation of the medium and growing the meat. Urea required for the cultivation of cyanobacteria accounted for 16 % of the total impact while cooking accounted for 6 % of the total impact. Similarly, to the study conducted by Tuomisto and Teixeira De Mattos (2011), found the cultivation process of muscle cells had the largest impact, accounting for 72 % of total energy use and the production of Cyanobacteria accounted for 23 % of total energy use.

Land occupation

Smetana et al. (2015) found land occupation requirements of cultured meat to be 0.39 – 0.77 m^2 per kg of cultured meat, while Tuomisto and Roy (2012) found land occupation to be 0.2 m^2 per kg of cultured meat, and Tuomisto et al. (2014) reported land occupation for cultured meats to be 0.5 – 2.8 m^2 per kg of cultured meat.

Water use

Water use was reported in three studies for seven products. Most studies that reported water 'use' did not clarify whether this referred to freshwater consumption or not. The lowest water consumption for cultured meat production was reported by Tuomisto and Teixeira De Mattos (2011) for cultured meat produced in California consuming 368 L of water for 1 kg of cultured meat. Cultured meat production in Thailand was reported to consume 376 L of water per kg of cultured meat and production in Spain uses 521 L of water per kg of cultured meat produced. Tuomisto and Roy (2012) reported 500 L of water use per kg of cultured meat produced and Tuomisto et al. (2014) reported 332.5 L per kg of cultured meat for cultured meat produced using cyanobacteria as the feedstock. The study conducted by Tuomisto et al. (2014) demonstrated that the choice of feedstock has a significant impact on water use. Water use impacts were lower when cultured meat was produced using wheat as a feedstock (332.5 L) and significantly higher when using corn (843.8 L) (Tuomisto et al., 2014). Tuomisto and Teixeira De Mattos (2011) reported that the cultivation process of muscle cells had the largest impact on water use, accounting for 82 % of water use. Cyanobacteria production accounted for 17 % of the water use.

Haem

Haemoglobin is a protein found in living plants and animals. It can be extracted from the root nodules of legumes such as soybean plants or produced via fermentation of genetically engineered yeast. Haem is used as an additive in plant-based alternative products as a substitute for myoglobin which gives meat its distinct flavour, appearance and cooking characteristics (Goldstein et al., 2017). No studies were found that assessed the environmental impacts of haem individually.

Haem produced through fermentation, where a genetically modified yeast strain expresses the natural occurring leghemoglobin protein, is a major ingredient in the Impossible Burger and is used to give the burger its meat-like characteristics such as flavour, colour and texture. Other major ingredients in the Impossible Burger are potato and coconut oil. Khan et al. (2019) conducted an LCA for impossible foods to assess the environmental impacts of the Impossible Burger. The study reported GHG emissions, land occupation, eutrophication and water use. Energy use was not reported. GHG emissions generated from the production of 1 kg of the product were reported to be 3.5 kg CO_2-e, land occupation was reported to be 2.5 m^2 per unit. The study did not state whether impacts from LUC were included in GHG results.

Table 3. Summary of environmental impacts of PBAs and cultured meats across the literature

Study	Product	FU	Country	Climate change (kg CO$_2$-e)	Energy (MJ)	Land occupation (m^2)	Acidification (kg SO$_2$ equiv.)	Eutrophication (kg PO$_4$ equiv.)	Water consumption (L)
Smetana et al. (2015)	Gluten-based meat alternative	1 kg ready to eat product	n.r.	3.59 – 4.03	39.7 – 49.2	5.5 – 5.82			
	Mycoprotein-based meat alternative	1 kg ready to eat product	n.r.	5.55 – 6.15	60.07 – 76.8	0.79 – 0.84			
	Dairy-based meat alternative	1 kg ready to eat product	n.r.	4.38 – 4.95	48.79 – 59.1	3.32 – 3.41			
	Lab grown meat alternative	1 kg ready to eat product	n.r.	23.9 – 24.64	290.7 – 37	0.39 – 0.77			
	Soy-based meat alternative	1 kg ready to eat product	n.r.	2.65 – 2.78	27.78 – 36.9	1.06 – 1.44			
Mejia et al. (2018)	Tofu	1 kg packaged tofu	US	0.98					
Head et al. (2011)	Mycoprotein (Quorn)	1 kg of mycoprotein	n.r.	2.4		0.41			
	Tofu	1 kg tofu at point of retail	n.r.	2.54 – 3.72		2.1			
Blonk et al. (2008)	Mycoprotein (Quorn)	1 kg mycoprotein	Netherlands	2.6	38	1.2			
	Tofu	1 kg tofu	Netherlands	2.2	2.6				
Tuomisto and Teixeira De Mattos (2011)	Cultured meat	1 kg cultured meat (DM 30 % and protein 19 % of mass)	Thailand	1.9	25.2	0.2			376
			California	2.2	31.8	0.2			368
			Spain	1.9	31.7	0.2			521
Tuomisto et al. (2014)	Cultured meat	1 kg cultured meat (DM 30 % and protein 19 % of mass)	Spain	2.8	49.8	0.5			516

Reference	Product	Functional unit	Region						
			Spain	3.8	46.6	2.6			332.5
			Spain	3.5	45.6	2.8			844
Mattick et al. (2015)	Cultured meat	1 kg CHO cell biomass	US	7.5	106	5.5	70.2	7.9	
Tuomisto and Roy (2012)	Cultured meat	1 kg edible product	EU	1.9	31.7	0.2			449
	Quorn	1 kg edible product	EU	2.3	38	0.17			
	Tofu	1 tonne edible product	EU	2.0	15.6	0.30			
Fresán et al. (2019)	Wheat-based	1 kg product	n.r.	2.1					
	Soy-based	1 kg product	n.r.	2.1					
	Wheat/soy-based	1 kg product	n.r.	2.3					
	Nut-based	1 kg product	n.r.	2.1					
Davis et al. (2010)	Pea burger	1 kg pea-based meal	Primary production: Germany, Sweden	1.96	99.3		0.009	0.021	
			Primary production: Spain	4.22	137.8		0.036	0.051	
Heller and Keoleian (2018)	Beyond burger (pea)	1 kg uncooked burger patty delivered to retail outlets	US	3.54	53.1	2.65			9.7
Khan et al. (2019)	Impossible burger	1 kg impossible burger	US	3.5		2.5		1.3	106.8
Hsu et al. (2018)	Quorn pieces (comparable to chicken)	1 kg product	UK	1.72		2.5			1339
	Mycoprotein	1 kg product	UK	1.14	n.r.	1.8			776
Mejia et al. (2019)	Wheat-soy based	1 kg product	US	2.19					

Comparing meat and plant-based alternatives

In order to compare literature results for PBAs and chicken meat, we used a standard product mass which represented edible product. As system boundaries varied between studies results, they have been standardised so that all results presented include impacts from cradle-to-factory gate. The contribution analyses presented by Smetana et al. (2015) and Davis et al. (2010) were used to subtract additional impacts generated at the consumption stage from these studies. All chicken meat results were standardised to report impacts relative to 1 kg of edible chicken meat as per Wiedemann and Yan (2014) and impacts for meat processing were added to studies that previously excluded these.

GHG emissions for selected PBAs and chicken meat can be seen in Figure 1. Soy-based products had the highest average GHG emissions across the literature, followed by cultured meat, while soy-wheat products had the lowest average GHG emissions. There was a large variation in the cultured meat GHG results across the literature, with one product (Smetana et al., 2015) significantly higher than all others. This could be explained by the assumptions and high uncertainty in data used across the five cultured meat LCA studies as the production process is still under development. There was also some variation with the mycoprotein results. Five of the seven studies looked at Quorn products, while other remaining two looked at mycoprotein. It would be expected that mycoprotein products would have lower impacts in comparison to Quorn products as a result of the additional processing stage required to turn mycoprotein into a Quorn product. It would also be expected that there would be variation amongst Quorn products as different products will have differing additional ingredients and different levels of processing. (i.e. crumbed vs no crumb, pieces vs mince). Two of the Quorn products were pieces; one was mince while the others were unknown.

Only one of the seven soy products stated that LUC was included in the results. Most other PBA products and cultured meats did not report LUC impacts. Not incorporating emissions from LUC may result in results being underestimated. LUC value of 5.2 kg CO_2-e was added onto the other soy products to account for this impact (Watson et al., 2018). There is also variation amongst the impacts of chicken meat. The country of production largely influences this. Australian chicken meat produced lower GHG emissions than most of the chicken meat studies included and is represented on the graph by the red dot.
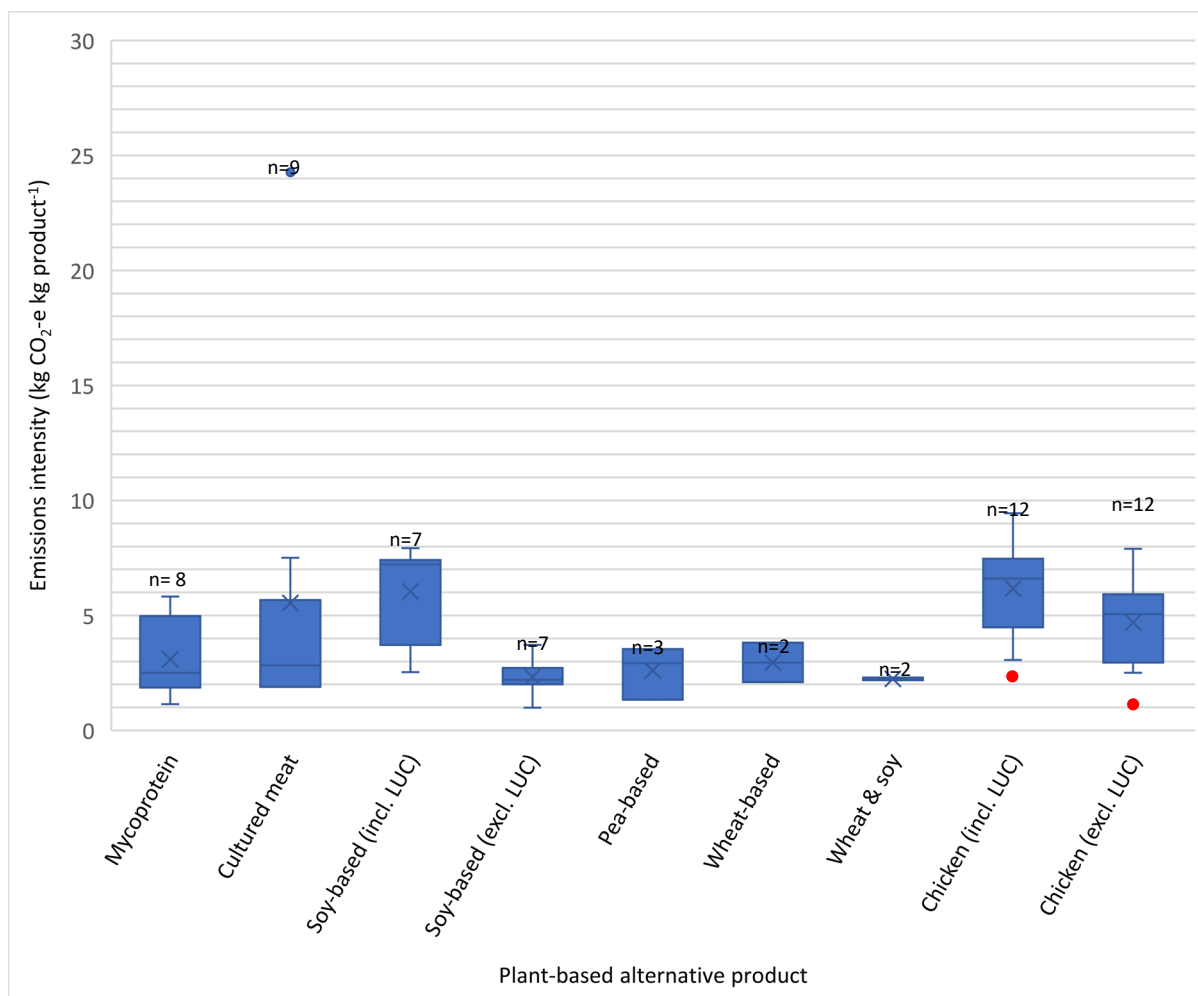
Figure 1. GHG emissions of PBAs and chicken meat (kg CO_2-e per kg of product^{-1})

The highest average energy consumption was for cultured meat products, followed by pea-based products, mycoprotein, chicken, wheat, and lastly soy (Figure 2). The highest individual energy consumption was for a cultured meat product, and this value was significantly higher than all other energy consumption values. The greatest variation can be seen between cultured meat results. As previously mentioned, the variation in the cultured meat products is likely a result of the uncertainty in the production process between the nine cultured meat products. The variation within the three pea products is likely a result of one product being just a patty. In comparison, the other two products included the whole meal, so additional impacts from the production of additional products were included. The range of energy results for the 12 chicken meat products is likely a result of different production countries. The energy impacts associated with chicken production in Australia were among the lowest of all the chicken studies included.
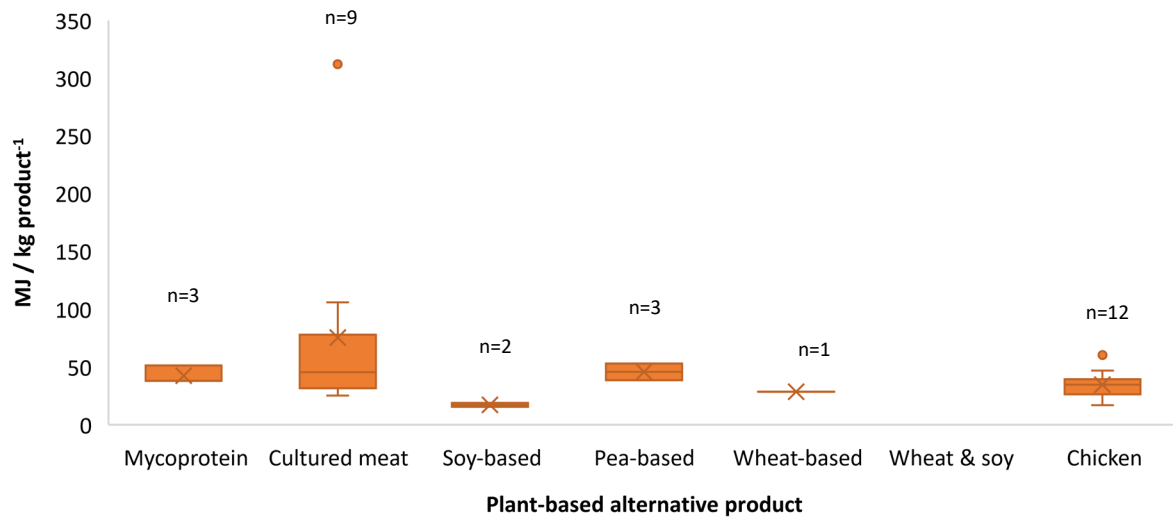
Figure 2. Energy use of PBAs and chicken meat (MJ per kg of product^{-1})

In contrast, land occupation for chicken meat production was the highest while mycoprotein was the lowest. Land occupation was not reported for wheat-soy products. There was a large variation in the land occupation for chicken meat. This again is likely due to different countries having different climates and therefore variation in yields of crops used for chicken feed. Cultured meat is often claimed to be a more environmentally sustainable meat replacement option due to its reduced land occupation requirements. The results presented in Figure 3 do not support these claims as cultured meat products have the second-highest average land occupation requirements of all PBAs.
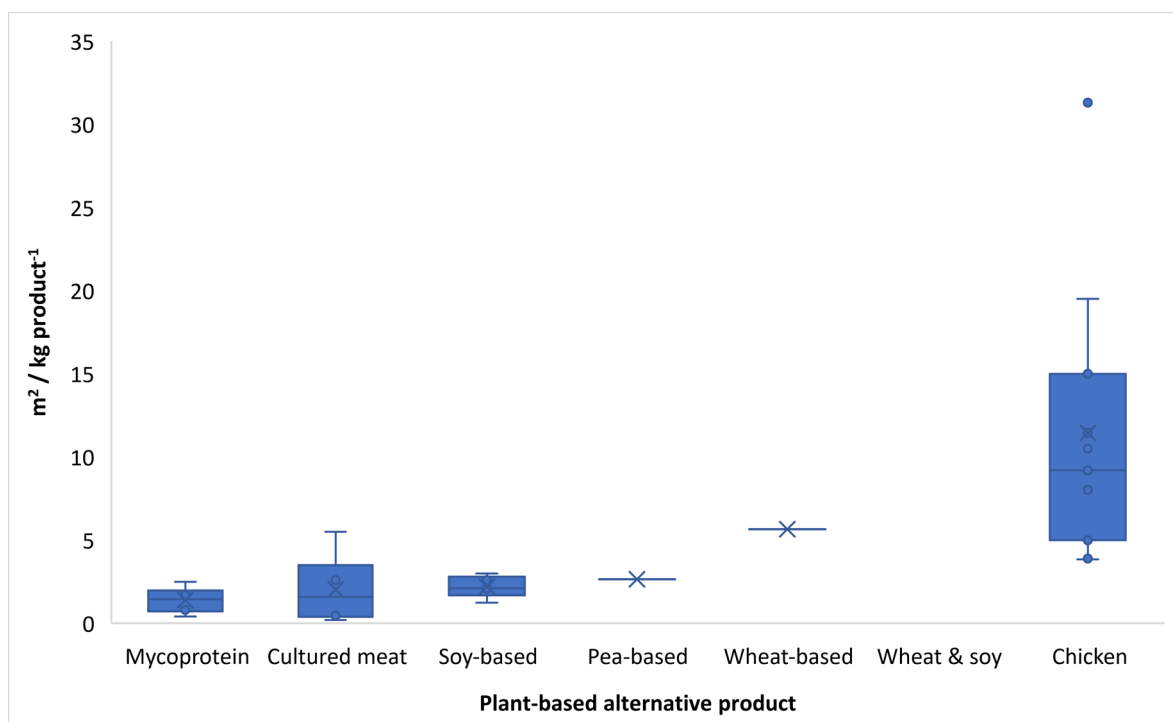
Figure 3. Land occupation of PBAs and chicken meat (m^2 per kg of product^{-1})

Water consumption was not reported in the majority of the PBA and cultured meat studies. The figures above use a weight-based functional unit to compare different products (1 kg of product). The questionable accuracy of this weight-based functional unit makes this a problematic comparison method for different products since there are varying nutritional contents and energy values in 1 kg of each product. Results from the Smetana et al. (2015) study demonstrated the large impact that the functional unit can have on the results, highlighting the importance of choosing a relevant and comparable functional unit as meat substitutes have different nutritional values and attributes. To reflect the varying quantities of each nutritional parameter, such as protein, fat and vitamin content, a more complex nutritional score would be a more accurate representation of the quality of products and would allow for an accurate comparison.

Review of nutritional qualities

Introduction

Life Cycle Analysis (LCA) has been described by Weideman (2004) as requiring a functional unit that captures obligatory properties of a product for use as a base for comparison across products on environmental impact. Obligatory properties are those that a product must have to be considered as an appropriate alternative when compared with other products.

In a review examining the application of nutritional, functional units in a commodity-level life cycle assessment (McAuliffe et al., 2020), the authors said the nutritional quality of final products is attracting an increased level of attention within the LCA literature, and work should continue to focus on human nutrition across the value chain. They also say that methods for assessing nutritional quality have not reached consensus. LCA comparisons have either been on a whole diet or individual products, with most examining diet. Additionally, a large body of research has focused on commodities and their comparison, which is removed by an additional step from the nutritional qualities of the product consumed. A systematic search of the literature found Functional Units are either based on single nutrients, composites of multiple nutrients, or commodity level analysis in a dietary context. The intent of this comparative assessment of chicken versus four PBA food products will be to compare individual foods that could be served as alternatives in a meal, which is not a typical approach when compared with existing literature. It will utilise a selection of several nutrients to assess nutritional composition, which we have called a Nutrition Quality Score (NQS).

In order to make fair and meaningful comparisons between chicken and plant-based food alternatives, Nutrition Quality Corrected Functional Units (NQCFU) will be developed based on nutritional equivalence (based on a nutrition quality score) and culinary equivalence (incorporating serving size) that incorporate a cooking method to represent foods as consumed.

A criticism of the use of nutrient profiling in environmental assessments raised in the review by McAuliffe et al. (2020) is the lack of rationale of the nutrients chosen and comparing unlike foods. A solution proposed by McAuliffe is to limit comparisons to foods within the same food group. This analysis will address this criticism by only comparing foods within a single food group: the meat and alternatives food group described below. Rather than compile as many nutrients as possible into a nutrition quality score, regardless of importance, nutrients will be prioritised according to the expectation of delivery via this food group.

Nutrition and dietary recommendations on chicken and PBAs in Australia

Chicken meat is included in the 'Lean meats and poultry, fish, eggs, tofu, nuts and seeds, legumes/beans' core food group within the Australian Dietary Guidelines (ADGs) (NH&MRC, 2013a). This group will henceforth be referred to as 'The meat and alternatives' group. There is a high level of biological and nutrient diversity within this food group, and plant-based foods in this group are not nutritionally equivalent, but rather placed in this group as an alternate source of some nutrients for people who choose not to eat meat/poultry/fish.

The quantities of plant-based foods suggested by the ADGs are given to match an energy limit (see Figure 4) rather than a nutrient target and their nutrient profile varies significantly. In terms of energy (kilojoule) content, a serve of lean meat and poultry, fish, eggs, tofu, nuts and seeds, legumes/beans provides 500 – 600 kJ.
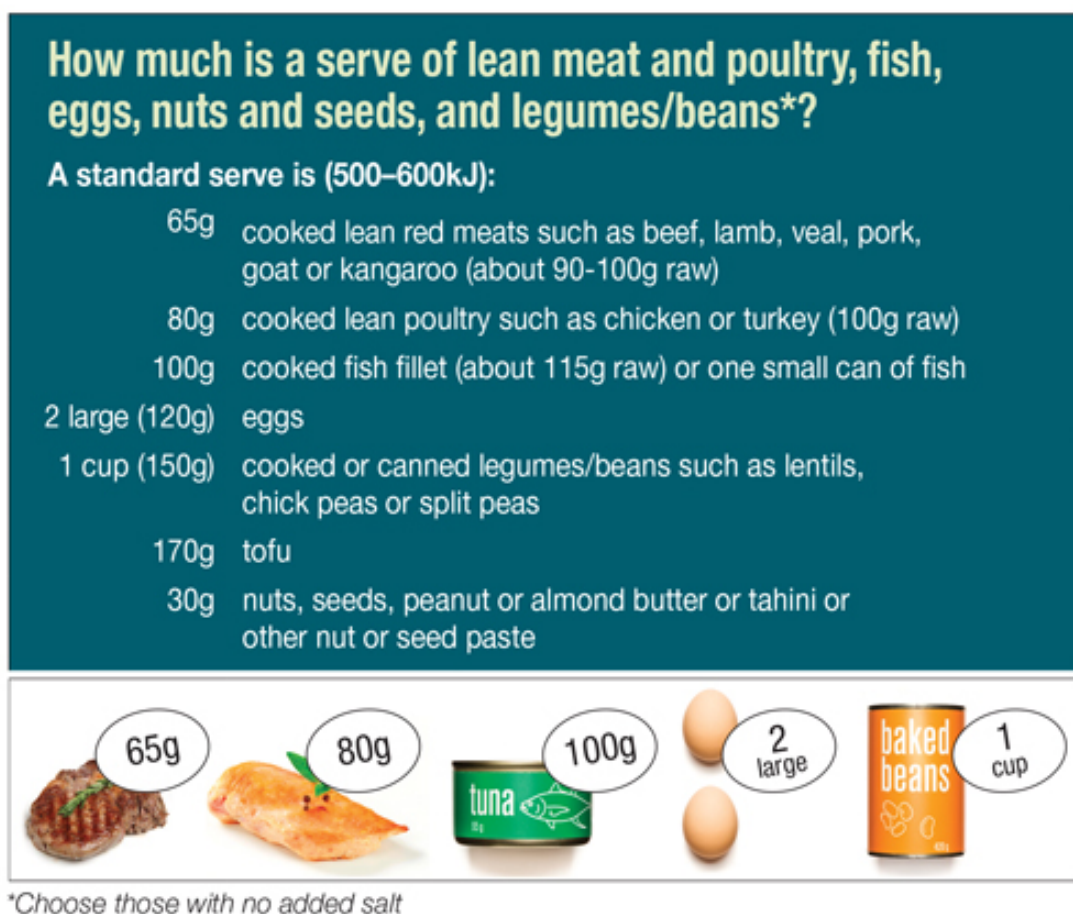
Figure 4. Government nutrition advice about serve size from the meat and alternatives food group (reproduced from NH&MRC, 2013b)

Notes: A serve of chicken meat used in this analysis is 80 g cooked (100 g raw), as suggested by the Australian Dietary Guidelines (ADGs).

What nutrients are expected to be delivered from this food group?

The ADGs nominate the key or distinguishing nutrients for this food group as iron, zinc, vitamin B12 and long-chain omega-3 fats – protein is not considered a distinguishing nutrient. However, this group is acknowledged as a good source: "In general, the foods in this group are a good source of many nutrients including protein, iron, zinc and other minerals and vitamins of the B-group" (NH&MRC, 2013a). The reason protein is not considered a distinguishing nutrient by the authors is that other food groups also contribute protein (e.g. dairy, cereals/grains), and there is no evidence of inadequate protein intakes in the Australian population. This is a departure from previous dietary guidelines in Australia (1998 and prior). It is also inconsistent with other dietary guidelines around the world, such as Canada's Food Guide (Health Canada, 2016) and the US Dietary Guidelines (USDHHS & USDoA, 2015) that refer to the meat and alternatives food group as 'protein foods'.

Distancing protein from the meat and alternatives food group in Australia is also out of step with consumer understanding. Traditional and long-standing nutrition education links the meat and alternatives food group with protein. There is also a strong protein marketing trend within the food industry that reinforces the importance of protein, especially in the growth market of plant-based meat alternatives. A supermarket audit conducted in 2019 found that 60 % of the 137 PBAs carried a protein claim on the pack (Curtain & Grafenauer, 2019). Furthermore, de-linking protein with meat

and alternatives is also out of step with the evidence base examining LCA of animal versus plant-based foods that typically incorporate a protein quality metric (Berardy et al., 2019).

For these reasons, we use a protein quality metric within the Nutrition Quality Score in this assessment.

Plant-based alternative (PBA) products for comparison rationale

The plant-based alternatives (PBAs) selected to compare with chicken in this analysis are:

1. Quorn pieces (substitute roast chicken pieces)
2. Tofurky plant-based Chick'n (lightly seasoned)
3. Sunfed Chicken Free Chicken
4. Unreal Co. vegan Chick'n sliders

The four products were chosen because they are marketed as plant-based chicken alternatives and are widely available in Australia:

- They are all sold in chilled format and require cooking, the same as raw chicken.

- They are also culinarily equivalent to chicken meat cuts; that is, they are used as a replacement for chicken in familiar meal formats.

- The preparation instructions could be easily interchanged with chicken cuts.

- They are available in major Australian supermarkets and have wide distribution and availability.

Table 4. Plant-based alternatives selected to compare with chicken

PBA	Format	Primary ingredients	Country of origin	Company	Australian retailer	Website	Pack image
Quorn pieces	Chilled 200 g	Mycoprotein	UK	Quorn Marlow Foods, UK	Coles, Woolworths, Foodworks, IGA	https://www.quorn.com.au/	
Tofurky Chick'n	Chilled 227 g	Wheat, soybean,	US	Tofurky, US	Woolworths	https://tofurky.com/what-we-make/chickn/lightly-seasoned/	
Sunfed Chicken Free Chicken	Chilled 300 g	Pea protein	New Zealand	Sunfed Limited Level 1, Quad 7, 6 Leonard Issitt Drive, Auckland	Coles	https://sunfedfoods.com/	
Unreal Co. Chick'n sliders	Chilled 310 g	Soy, rice	Australia (7%)	Unreal Co. 17 Research Drive, Croydon South, Victoria 3136	Woolworths	https://unrealco.com.au/	

Culinary equivalence of PBAs

Dietitians often educate about healthy balanced meals using the 'half-quarter-quarter' plate principle: aim for half the plate as vegetables; one quarter as meat/alternative; and one-quarter grains (or starchy vegetable). Chicken or PBAs represent the one-quarter of the plate for meat/alternatives, or the protein meal component. The protein component of a meal includes chicken and what other foods you could neatly swap for chicken in a meal context. From a culinary perspective, a meal component approach underlies ingredient decisions for shopping and meal planning. For example, for a roast dinner, will the purchase be chicken or Tofurky®? Or, for a stir-fry; will the ingredient be chicken strips or Quorn® pieces?

Nutritional equivalence of PBAs

The chicken meat nutrition data used for comparison are based on a weighted composite cut of raw chicken. The composite is derived from a weighted sample of raw chicken cuts (see Appendix) and has a 10 % fat content to align with the chicken meat commodity data used for the LCA. This ensures the nutrition information used for comparison is representative of the marketplace and typical consumption in Australia.

Table 5. Nutrient data for composite chicken meat per 100 g serve with 10 % total fat content

Energy (kJ)	Protein (g)	Fat, total (g)	Saturated fat (g)	Sodium (mg)	Vitamin B12 (ug)	Iron (mg)	Zinc (mg)	Omega-3 fats (g)
688	18.8	10g	3	49	0.465	0.538	1	0.021

We compared the ADGs serving size for chicken (100 g raw) with the manufacturer's recommended serving size for the PBAs. The PBA products chosen for comparison do not match for energy, with all falling outside the Dietary Guidelines food group energy target of 500 – 600 kJ. Some contained more energy than the food group standard, and some contained less (see Table 6). The Unreal Co. Chick'n Sliders contain the least number of kilojoules at 350 and the smallest serving size at 75 g. In contrast, Tofurkey Chick'n pieces contain the highest energy content at 971 kilojoules, and the highest fat content at 12 g per serve.

Table 6. Energy content of chicken versus PBAs

PBA Product	Serving size	Energy per serve (kJ)
Quorn pieces	100 g	432
Tofurky Chick'n	90 g	971
Sunfed Chicken Free Chicken	100g	939
Unreal Co. Chick'n sliders	75 g	350
Chicken meat (composite)	100 g	688

Despite differing in energy content, the standard serve for chicken (100 g) raw and the manufacturer's serving size for the PBAs (75 – 100 g) are culinary equivalents and therefore can be directly compared for the purposes of this LCA.

Proposed nutrition quality score for meat, poultry and PBAs

Nutrition quality score (NQS)

In order to develop the required Nutritional Quality Corrected Functional Unit for PBAs to compare with chicken meat, several target nutrients will be considered based on the expected nutrients to be delivered by the meat and alternatives food group. The NQS will incorporate the following key components:

- Protein quality (Weighted Protein Score; WPS),
- Presence of vitamin B12,
- Bioavailable iron content, and
- Bioavailable zinc content.

The NQS for each nutrient will be developed, which will then be entered into an algorithm to produce a Nutritional Quality Corrected Functional Unit (NQCFU). Chicken meat will be used as the sentinel food and the PBAs compared with it.

Long-chain omega-3 fats

Fish and seafood are the main sources (71 %) of long-chain omega-3 fats in the Australian diet (Fayet-Moore et al., 2015), meat and poultry are minor sources, and plant foods do not contain long-chain omega-3 fats (unless they are fortified). Poultry provided 10 % of long-chain omega-3 fats in the 1995 National Nutrition Survey, and 28 % came from red meat (DAA, 2011). From a regulatory, marketing and communication perspective, chicken meat contains less than the 30 mg per 100 g required to make a nutrient claim for long-chain omega-3 fats under the Food Standards Code (Schedule 4 - ANZFS, 2020). It is therefore not valid to compare long-chain omega-3 content in PBAs with chicken.

While the Australian Dietary Guidelines have determined long-chain omega-3 fats are a distinguishing nutrient for the meat and alternatives food group, it will not be included in the calculation of a Nutrition Quality Score (NQS) in this assessment.

Nutrients of public health concern

According to an audit of the four major supermarkets in Australia, the PBA market grew 429 % between 2015 and 2019 (Curtain & Grafenauer, 2019), however many products might be considered "ultra-processed" and contain added salt and other additives of consumer concern. Vegan products have attracted media scrutiny, including the Guardian newspaper's article titled "Vegan products sold in supermarkets loaded with unhealthy amounts of salt" (The Guardian, 2019, 11th of September).

From a public health nutrition perspective, the main concern is sodium (salt) as Australians currently consume more than the World Health Organisation (WHO) maximum of 5 g/day and this does not appear to be declining (Land et al., 2018). The supermarket audit by Curtain (2019) found that only 4 % of the 137 products audited were low in sodium and sodium content ranged from 58 – 1200 mg per 100 g. While chicken meat is very low in sodium (49 mg/100 g, raw weight), it may be prepared with sodium-containing ingredients and served with sodium-containing condiments. For this reason, it was decided not to account for differences in sodium, despite the generally higher levels in the PBAs. In addition, the largest source of sodium in the Australian diet comes from the bread and cereal food group and not the meat and alternatives group (FSANZ, 2015).

Sodium content will not be considered as part of nutritional equivalence in this analysis.

Other nutritional aspects not covered

Plant-foods are well known for their nutritional benefits. They are good sources of dietary fibre, and phytonutrients and plant-based diets are associated with health benefits (Kim et al., 2018). In this analysis, however, we have taken a food group approach as described by the Australian Dietary Guidelines (ADGs: NH&MRC, 2013a). The meat and alternatives group and the nutrients expected to be delivered from this group do not include fibre and phytonutrients. These nutrients and phytonutrients are expected to come from other food groups, i.e.

- Vegetables and legumes
- Fruit
- Grain (cereal) foods

Positive nutritional attributes of PBAs, such as fibre content, vitamins or phytonutrients, will not be considered in the nutritional quality score in this analysis.

Protein quality scores

High-quality protein is particularly important for the growth of infants and young children, and possibly in older people losing muscle mass in later life (Willett et al., 2019).

Protein quality (defined by the effect on growth rate) reflects the amino acid composition of the food source, and animal sources of protein are of higher quality than most plant sources. The scores for protein quality of animal products (milk, beef) have been found to be between 1.4 and 1.87 times higher than plant proteins (Ertl et al., 2016). Specifically, l-leucine appears to play a particularly strong stimulatory role for triggering muscle protein synthesis, and animal protein is typically higher in leucine than plant protein (Ciuris et al., 2019). Cereal grains such as wheat, rice and maize are limiting in indispensable amino acids (also known as essential amino acids) such as lysine. Although legumes have higher amino acid scores (i.e. DIAAS, see definition below) than cereal grains, they are limiting in methionine. They may contain anti-nutritional factors that can often reduce the absorption of amino acids (Bailey & Stein, 2019).

Until 2011 the Protein Digestibility Corrected Amino Acid Score (PDCAAS) was the United Nations reference standard method for scoring protein quality. It compares the amino acids with a test protein and then adjusts for digestibility via nitrogen ingestion versus excretion. It scores foods up to 100 (or 1.0) (Katz et al., 2019). From the list of foods in Table 7, chicken meat achieves a perfect PDCAAS score of 1.0, as does Quorn, while soy protein scores 0.94, pea flour 0.69 and wheat gluten 0.25.

Table 7. Protein digestibility corrected Amino Acid score (PDCAAS) for selected foods (reproduced from S. Miller & Dwyer, 2001)

Protein source	Score
Quorn pieces	1.0[d]
Casein	1.0[a]
Egg white	1.0[a]
Chicken (light meat, roasted)	1.0[c]
Turkey (ground, cooked)	0.97[c]
Fish (Cod, dry cooked)	0.96[c]
Soybean protein	0.94[b]
Beef	0.92[a]
Mycoprotein	0.91[d]
Pea flour	0.69[a]
Kidney beans (canned)	0.68[a]
Rolled oats	0.57a
Lentils (canned)	0.52[a]
Peanut meal	0.52[a]
Whole wheat	0.40[a]
Wheat gluten	0.25[a]

[a] From FAO/WHO Joint Report (1989); [b] From Sarwar and McDonough (1990); [c] Calculated from amino acid data in USDA Nutrient Database for Standard Reference. March 12, 1998 (assumed a digestibility equivalent to beef = 90 %); [d] Calculated from Marlow Foods data.

Forty per cent of the total amino acids in chicken are essential (or non-dispensable) amino acids. Protein and amino acid content increase after cooking due to moisture losses, despite some losses occurring during cooking. The most predominant essential amino acids are arginine, leucine and lysine (Honggyun et al., 2017).

Criticism of protein quality scoring

Some public health nutritionists are critical of protein quality scores because they do not reliably improve the quality of diet or health, do not take environmental impact into account and unfairly penalise plant-based proteins (Katz et al., 2019). This view is, in part, based on the way proteins are communicated, regulated, and marketed to the public in the US, which differs from Australia. In the US, food manufacturers can only make a protein claim on food based on the amount present as well as the protein digestibility-corrected amino acid score (PCDAAS). Food Standards Australia and New Zealand (FSANZ) only requires the amount of protein and not the protein quality score to make a protein claim.

Another factor raised in criticism of scoring protein quality is that 'protein combining' in the same meal is not necessary. Shortfalls of some essential amino acids in plant foods (such as lysine in cereals) can be complemented by high levels in other plant foods consumed throughout the day and do not need to be consumed in the same meal (Marsh et al., 2013).

Criticism of protein quality scoring has included the argument that animal foods with high protein quality scores also contain nutrients of concern associated with adverse public health impact. This

criticism does not apply to chicken. While red meat and processed meat are associated with a greater risk of chronic disease such as coronary artery disease and higher mortality overall, poultry is not associated with health risk in epidemiological studies (Battaglia et al., 2015). In high-income countries like Australia, the greatest dietary risk to health is associated with consumption of sodium and inadequate consumption of plant foods such as whole grains, vegetables, fruits, nuts and seeds (Afshin et al., 2019).

In response to the criticism of traditional protein quality scores, a new metric for protein quality that incorporates health outcomes and environmental impacts was proposed by Katz et al. (2019), albeit in the context of the US. Using this proposed new metric, chicken meat scores better than beef because it has high protein quality, is recommended for health (in the US) and has a low environmental impact (as rated by GHGEs per gram of protein). Beef scores poorly at 0.31, soy scores highly at 0.97, and chicken scores highly (0.98 for chicken breast without skin) or moderately (0.65 for dark meat with skin). In Australia, the Dietary Guidelines do not distinguish between breast meat and leg meat.

The Digestible Indispensable Amino Acid Score (DIAAS)

The Digestible Indispensable Amino Acid Score (DIAAS) is the preferred method for scoring protein quality of foods recommended by FAO (2013). The DIAAS is an advanced evaluation of how amino acids are digested and assimilated by the body. The DIAAS calculation utilises the ileal digestibility coefficients of each amino acid in a particular food to determine the true ileal digestibility of the indispensable amino acids present. While the previously used PDCAAS scores were 100 (or 1.0) or less, the DIAAS scores can exceed 100 for higher quality proteins, for example, whole milk powder has a DIAAS of 115 (Ertl et al., 2016).

The DIAAS has been shown to better describe protein quality for some dairy and plant proteins than the PDCAAS (Mathai et al., 2017). The DIAAS can be used in the analysis of meal plans to determine the adequacy of protein sources to meet recommended intakes (0.8 g/kg of body weight/day), which is particularly relevant for individuals following vegetarian and vegan diets (Berardy et al., 2019).

The DIAAS calculation method is set out below.

$$DIASS\ (\%) = \frac{x}{y} \times 100$$

Where:
x is mg of digestible indispensable amino acid in 1 g of the dietary protein
y is mg of the same indispensable amino acid in 1 g of the reference protein.

Using DIAAS cut-off values, FAO (2013) says that protein quality can be described as 'Excellent' if DIAAS is greater than 100 and 'Good' if DIAAS is between 75 and 99. The FAO (2013) suggests that protein quality claims should not be permitted for DIAAS scores of < 75. Using these cut-offs, chicken, beef and milk can be described as 'excellent'; soy and pea protein can be described as 'good', and no protein quality claim can be made for rice or wheat protein. This ranking of the protein quality of PBA ingredients is the same as that obtained using the PDCAAS method.

Because digestibility values of amino acids in individual food proteins are additive in mixed meals, DIAAS values for mixed meals may be calculated (Bailey & Stein, 2019).

Table 8. DIAAS (%) calculations of various protein foods

Protein source	Marinangeli & House (2017)	Ertl et al. (2016)	Cervantes-Pahm (2014)	Rutherfurd et al. (2015)	Mathai et al. (2017)	Han et al. (2019)	FAO (2013)
Rice protein				37		37	
Wheat		40.2	44		45		40
Peas		64.7					64
Pea protein				82			
Soybean		99.6					
Soy protein isolate				89, 91	84		
Soya flour					89		
Soybean expeller		100.3					
Soybean cake		97					
Tofu	52						
Chicken meat	108 (chicken breast)	108.2					

Note: Highlighted rows are ingredients in the PBAs in this analysis.

The DIAAS can be used to compare animal and plant proteins. For example, because wheat protein is around 42 and chicken meat is 108, an individual would need to eat more than twice as much wheat protein compared with chicken to meet human amino acid requirements. This example illustrates how both the quantity and quality of protein are important when assessing a food's amino acid adequacy. The Weighted Protein Score below addresses both aspects.

Weighted Protein Score (WPS)

The preferred method for this analysis was a slightly modified version of the Weighted Protein Score (WPS) developed by Berardy et al. (2019). This is shown in the formula below based on a food's protein content and DIAAS.

WPS equals the product of DIAAS (D) as a percentage, serving size in grams (SS), and grams of protein per 100 grams of a given food (P), divided by 100.

$$WPS = (D \times SS \times P) \div 100$$

Where:
D = DIAAS (%),
SS = Serving size in grams (g)
P = Protein per 100 g of food (%)

The Berardy et al.(2019) method used Reference Amounts Customarily Consumed (RACCs) developed by the US Food and Drug Administration (FDA, 1990) to provide a reference for identifying actual typical consumption amounts for the purposes of nutrition labelling. There are no equivalent specified reference amounts for Australia, and manufacturers are free to determine their own serving sizes of food products. We have used the manufacturer's serving size (SS) when calculating the WPS.

Table 9. Weighted Protein Scores for PBAs and chicken

PBA product (serving size)	Serving size	% protein in product	Primary protein ingredient (%)	DIAAS (%)	Weighted Protein Score
Quorn pieces	100g	14	Mycoprotein (13.2 %)	100* (Mathai et al., 2017)	14
Tofurkey Chick'n	90g	25	Wheat gluten	40 (Ertl et al., 2016)	9
Sunfed Chicken Free Chicken	100g	36	Pea protein	82 (Rutherfurd et al., 2015)	29.52
Unreal Co. Chick'n sliders	75g	13	Soy protein	99 (Ertl et al., 2016) Soy Protein Isolate 84 (Mathai et al., 2017)	9.65
Chicken meat (raw, composite)	100g	18.8	Chicken protein	108.2 (Ertl et al., 2016)	20.3

* DIAAS not available for mycoprotein.

Discussion of Weighted Protein Score (WPS) results

The highest WPS was obtained by the Sunfed Chicken Free Chicken, which is a pea protein-based PBA. Whilst pea protein has a low DIAAS, the protein content (36 %) of this product is the highest of all the PBA foods, and this contributed to its high WPS score.

Methodological issues

Where there are two sources of protein in a PBA food, we used the primary protein source as indicated in the ingredients list, which is listed in descending order by weight. We were unable to determine the protein split between the two sources in the manufactured products as the manufacturers declined to supply the data on request. This will affect the accuracy of the WPS and is likely to have adversely affected the score of Tofurkey Chick'n, albeit to a small degree, because the secondary protein ingredient (tofu) has a higher DIAAS (52) than the primary ingredient (wheat gluten) (DIAAS 40). The underestimate is expected to be small as tofu is 9.8 % by weight in the ingredients list and tofu has a relatively low protein content of 8 %.

We were unable to determine the DIAAS for Quorn, so the PDCAAS was used instead. This may be an underestimate.

It is likely that greater moisture content in the raw chicken meat compared with the PBAs has produced a nutrient dilution effect. That is, chicken as consumed (cooked) has higher protein and other nutrient levels than raw chicken used in this analysis because the moisture content is reduced by cooking, and this concentrates nutrient content by weight. We hypothesise the PBAs are processed then chilled and require reheating rather than cooking. They have a dry appearance and minimal shrinkage during cooking, suggesting lower moisture content. Therefore, the effect of cooking on their nutrient content is expected to be less.

To confirm the reheating versus cooking hypothesis, we asked the manufacturers whether their product could be eaten raw. Their responses did not confirm their products could be eaten raw, but rather that cooking made them taste better. There may also be microbiological safety risks. Below are their published Q&A responses on their website, or reply to our email inquiry:

Quorn pieces: "We do not recommend you eat Quorn (pieces, *sic*) without cooking them first" https://www.quorn.co.uk/faqs/product

Tofurky: "All chilled Tofurky products are pasteurized to ensure product safety. However, we do not recommend eating any chilled or frozen product without cooking according to packaging instructions." https://tofurky.com/faqs/#aa65d3520c108c62e701c064c10615ca

Sunfed Chicken Free Chicken: "You should cook Sunfed Chicken Free Chicken" no reason given. (The Sunfed Team, personal communication 21 April 2020).

Unreal Co. Chick'n sliders: "Although our Chick'n Sliders are plant-based, we still recommend cooking them first. We do not think anyone will experience any issues from uncooked Sliders, but we haven't performed enough testing for us to recommend eating them raw." (J. Guillaumier, personal communication, 9 April 2020).

Vitamin B12

Vitamin B12 is almost exclusively found in animal foods including red meat, poultry, seafood, milk, cheese and eggs. The Estimated Average Requirement (EAR) of vitamin B12 is 2.0 ug per day for adult men and women (NH&MRC, 2006). Vitamin B12 is not present in plant foods and is a nutrient of concern in a vegetarian or vegan diet. Vegans or vegetarians who significantly limit animal-based foods must regularly consume foods fortified with vitamin B12 or take supplements (Zeuschner et al., 2013). The Dietary Guidelines for Australian (DGAs) say, "Vitamin B12 is only found in animal-based foods unless it has been added to a plant-based product." The DGAs also say supplementation of vitamin B12 may be required for people with strict vegan dietary patterns (NH&MRC, 2013a).

In Australia, plant-based meat analogues, including the PBAs used in this assessment, are permitted by the Food Standards Code to voluntarily add vitamin B12 to their products at levels between 10 – 100 % RDI (Recommended Dietary Intake) per serve, depending on their composition and food format (ANZFS, 2020).

However, fortification of vitamin B12 is voluntary, and a supermarket audit completed in 2019 showed less that than a quarter (24 %) of PBAs in the Australian marketplace are fortified with vitamin B12 (Curtain & Grafenauer, 2019). None of the PBAs chosen for this assessment have added vitamin B12.

Because the meat and alternatives food group are expected to deliver vitamin B12, and PBAs can add it, vitamin B12 will be used in the Nutrient Quality Score calculation in this assessment.

Iron

Animal foods contain haem iron and plant foods contain non-haem iron, but the haem form is more bioavailable (NH&MRC, 2006). However, non-haem iron absorption depends on physiological need and is regulated in part by iron stores. Adaptation to low intakes can occur. Non-haem iron absorption can be as much as ten times greater in iron-deficient individuals compared with iron-replete individuals. Absorption also varies depending on meal composition (Melina et al., 2016).

The individual adaptation to low iron intakes makes nutritional equivalence comparisons between chicken and PBAs difficult. The 80 % increased intakes of iron from plant sources recommended for vegetarians as suggested in the NRVs (NH&MRC, 2006) is for whole diets rather than individual foods. In this analysis, we have taken a population approach and calculated theoretical iron bioavailability that excludes individual variability of iron absorption.

Absorption of iron from chicken

In order to adjust for differences in bioavailable iron in chicken meat compared to PBAs, it is necessary to consider that iron is not completely absorbed from animal foods either. Normal iron absorption varies from 50 % in breast milk to 10 % in infant cereals (NH&MRC, 2006). Approximately half of the iron in meat, fish, and poultry is haem iron. Depending on an individual's

iron stores, 15 – 35 % of haem iron is absorbed. Non-haem iron is absorbed at 2 – 20 %. It is markedly increased by enhancing factors such as ascorbic acid (vitamin C) and meat, fish and poultry that can increase non-haem bioavailability by four times (Monsen, 1988). We can assume that non-haem iron absorption from chicken would be on the high end of the 2 – 20 % range since chicken meat itself is an iron absorption enhancer.

Calculation of haem iron in chicken

Chicken meat contains 0.54 mg total iron per serve (100 g, raw), and half of this is haem iron which is 0.27 mg. If 15 – 35 % of this haem iron is absorbed, this equates to between 0.04 – 0.09 mg available iron per serve, depending on an individual's iron stores (iron absorption is enhanced where there are fewer stores). We can take the median absorption of 25 % to represent a typical consumer and calculate 25 % of 0.27 mg yields 0.07 mg available haem iron per serve.

Calculation of non-haem iron absorption in chicken

Half the iron in a serve of chicken (0.27 mg) is non-haem iron, which is absorbed at an estimated 20 %, which gives a total non-haem available iron content of 0.05 mg.

Calculation of total iron absorption from chicken

The total iron absorption from chicken is the haem iron (0.07 mg) added to the non-haem iron (0.05 mg) absorption, which gives a total of 0.12 mg per serve (see Table 12).

Non-haem iron absorption from the chicken will be estimated at 20 %

Zinc

The bioavailability of zinc from vegetarian diets is lower than from non-vegetarian diets. In addition, vegetarians typically eat high levels of legumes and whole grains, which contain phytates that bind zinc and inhibit its absorption (Hunt, 2003). Vegetarians have lower dietary zinc intakes and serum zinc concentrations, although evidence is lacking in the elderly, children and women during pregnancy and lactation (Foster & Samman, 2015). Unlike iron, adults have no adaptive response to increase zinc absorption from high phytate diets, and poor absorption from high-phytate diets is a major factor in the development of zinc deficiency (Gibson et al., 2018).

Despite this, overt zinc deficiency is not evident in Western vegetarians, and there is insufficient evidence to determine if zinc status is lower in vegetarian at-risk groups such as older people, children, pregnant and lactating women (2015).

Iron and zinc bioavailability

Iron and zinc are nutrients of concern when considering nutritional adequacy of vegetarian diets due to the absence of meats which contain bioavailable iron and zinc. In addition, vegetarian diets contain higher levels of phytate in legumes and whole grains that reduces the absorption of iron and zinc compared to omnivorous diets. While iron and zinc deficiencies are associated with plant-based diets in developing countries, the health consequences of lower iron and zinc availability are not clear in developed countries with abundant and varied food supplies. They warrant research attention as plant-based diets become more popular (Hunt, 2003).

The Estimated Average Requirement (EAR) for zinc is 12 mg per day for adult men and 6.5 mg for adult women. The EAR for iron is 6 mg per day for men and 8 mg for women (NH&MRC, 2006).

The Dietary Guidelines for Australia (DGAs) describe the meat and alternatives, food group, as follows (NH&MRC, 2013a):

"…Within this group, lean meats are a particularly good source of iron, zinc and vitamin B12. The iron and zinc in lean meat, poultry and fish is more easily absorbed by the body than the iron and zinc from eggs, and plant foods…Those following a vegan diet should choose foods to ensure adequate intake of iron, zinc and calcium and to optimise the absorption and bioavailability of iron, zinc and calcium..."

The dietary modelling that underpins the DGAs (DAA, 2011) also address the issue of nutrient adequacy of iron and zinc.

"…deficiency of some nutrients including iron, calcium, iodine, folate and vitamin D amongst some groups is also of concern…iron deficiency remains an area of concern, particularly for younger women and adolescent girls… Iron is also a key or distinguishing nutrient from the cereals food group, however, it has relatively low bioavailability compared to haem sources".

The reduced bioavailability of iron and zinc from plant foods is addressed in the Nutrient Reference Values (NRVs) for Australia and New Zealand (NH&MRC, 2006). They suggest that because absorption is lower from vegetarian diets, intakes need to be 80 % higher for iron and 50 % higher for zinc. These higher values are set on best estimates of relative bioavailability, but there is only limited clinical evidence available to confirm estimates.

Iron and zinc are frequently assessed together because they share common dietary sources; absorption is enhanced and inhibited by similar compounds, and deficiency is thought to occur simultaneously (Lim et al., 2013).

In general, the best sources of bioavailable iron and zinc are pulses (legumes); however the presence of phytic acid and other antinutritional factors reduces bioavailability to 5 – 15 % (Gupta et al., 2015). In a clinical study measuring the absorption of iron and zinc from red kidney beans (Phaseolus vulgaris) in women in a single food, test meal found absorption was 1.5 % for iron and 13 % for zinc (King et al., 2000).

Phytate

Phytic acid and its salts are collectively termed phytate and are present in unrefined cereals, legumes and oilseeds. There is no phytate in animal products. In unrefined cereals, phytate is typically concentrated in the outer aleurone (bran) layer. In legumes and oilseeds, phytate is distributed throughout the endosperm or kernel.

Phytic acid comprises 1 – 5 % by weight in cereals, legumes, oilseeds and nuts. It is the major storage form of phosphorous in plants representing 50 – 85 % of the total and is stored in leguminous seeds (such as soybeans and peas) within the protein bodies (found in the endosperm). While the phytate in most cereal grains is found in the bran fraction, in corn it is seen in the endosperm.

Population phytate intakes in adult omnivorous diets have been estimated at between 395 mg to 1,293 mg per day in the US and the UK (Lim et al., 2013). The PBAs in this assessment contain between 198 mg and 2,100 mg of phytate per serve (see Table 10), although please note the phytate content data used to derive these figures are not matched.
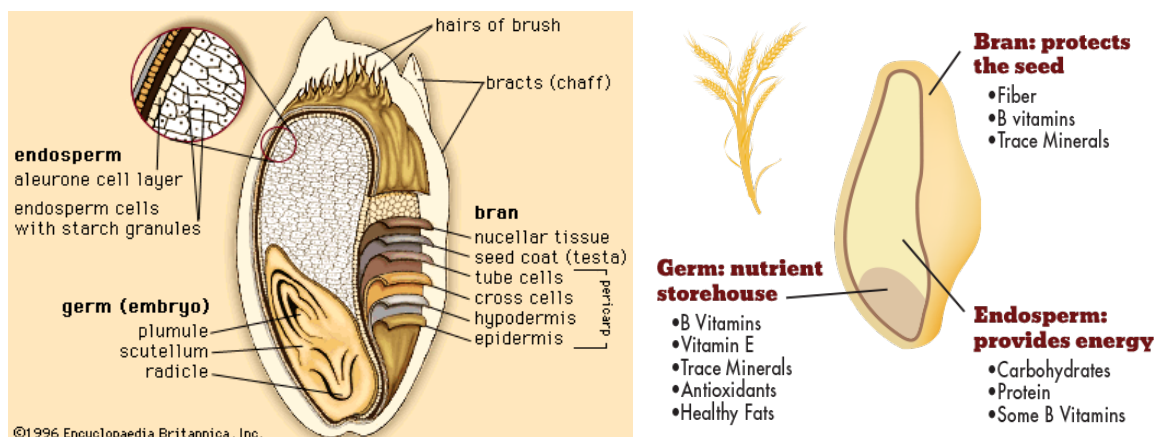
Figure 5. Anatomy of a cereal grain and nutrient content of components (reproduced from Encyclopedia Britannica, 2020)

Dietary phytate is a potent inhibitor of iron and zinc absorption. However, some processing practices and preparation techniques can reduce the inhibitory effect of phytate and increase its bioavailability. Milling of cereals can reduce phytate, whereas dehulling legumes does not, because phytate is evenly distributed through the endosperm. Phytate is relatively heat stable during normal household cooking, but there is some loss in higher temperature industrial processing such as extrusion cooking. Techniques to increase zinc bioavailability include soaking beans (legumes), grains, and seeds in water for several hours before cooking them (Gibson et al., 2018). Soaking and sprouting seeds activates phytase enzymes that hydrolyse phytic acid. Milling also reduces phytic acid but also removes the majority of minerals (Perera et al., 2018).

As the PBAs in this analysis are processed, it may be expected that the bioavailability may be greater than if the component ingredients were consumed minimally processed (e.g. soybeans vs tofu). However, the ingredients used are also added in a more concentrated form (e.g. soy protein, wheat gluten vs soybeans, wheat) which may also concentrate phytate. Measured bioavailability data is not available; therefore, approximations will be used. See Table 10 for the estimated phytate content per serve of the PBAs under analysis in this report followed by an explanation of how the estimates were determined.

Table 10. Phytate content of plant-based alternatives

PBA product (serve size)	Ingredients list	Main phytate containing ingredients	Estimated phytate content (g/100 g dry weight) (DW)	Estimated phytate content per serve (g)
Quorn pieces (100 g)	Mycoprotein (94 %), rehydrated free-range egg white, seasoning (natural flavour, yeast extract, onion), forming agent (calcium chloride), acidity regulator (calcium acetate).	**Mycoprotein**	0	0
Tofurkey Chick'n (90 g)	Water, wheat gluten, canola oil, organic tofu (filtered water, organic soybeans, firming agents (magnesium chloride, calcium chloride) (9.8 %), natural vegan flavour, oat fibre, vegan sugar, corn-starch, flavour enhancer (potassium chloride), soy sauce (water, soybeans, wheat, salt), granulated garlic, sea salt, acidity regulator.	**Wheat gluten**	0.27	0.243
		Tofu	0.22	0.198
		Total	0.49	0.44
Sunfed Chicken Free Chicken (100 g)	Water, pea protein (43 %), rice bran oil, pea fibre, NZ pumpkin, natural yeast extract, NZ maize starch.	**Pea protein**	2.1	2.1
Unreal Co. Chick'n sliders (75 g)	Water, protein [soy, rice], starches [potato, rice], fats [canola], vegan chicken flavouring, sugar, vegan egg replacer, nutritional yeast, herbs and spices, citric acid, potassium sorbate.	**Soy protein**	1.69	1.27
		Rice protein	0.3	0.23
		Rice starch	0.036	0.027
		Total	2.026	1.53
Chicken meat (raw) composite	Chicken meat composite, raw, 10 % fat.	**n.a.**	0	0

Key: Grey shading indicates a significant source of phytate.

Phytate in mycoprotein

One of the PBAs assessed in this report is mycoprotein (Quorn®). The conclusions of a US expert panel on mycoprotein include that rat studies have shown no significant differences between mycoprotein and a casein control in availability and balance of iron and zinc. British studies showed that mycoprotein does not contain any phytic acid or phytic salts (phytate) and, as a result, had no significant effect on the absorption of calcium, magnesium, phosphorus, zinc, or iron (S. Miller & Dwyer, 2001).

For the purposes of this analysis, the estimation of phytate content in mycoprotein is zero.

Phytate in wheat gluten

Wheat gluten is also known as gluten flour or vital wheat gluten or seitan. It is essentially wheat flour with the starch removed and is about 80% protein. As the brand of wheat gluten used in Tofurky Chick'n is unknown, the WHETPRO-80 (with 80% protein) brand will be used as a proxy. The phytate content of WHETPRO-80 wheat gluten has been analysed at 0.27g/100g (dry weight) (Naczk et al., 1986).

For the purposes of this analysis, the estimation of phytate content in wheat gluten is 0.27.

Phytate in soy protein

Soy protein is likely to have lower iron and zinc bioavailability than other ingredients.

The soy protein used in the Unreal Co. Chick'n sliders is unknown, so SUPRO 620 brand soy protein isolate will be used as a proxy. The phytate content of soybeans varies with environmental conditions and hereditary factors (Ishiguro et al., 2006). Soybeans contain 1 – 2.22 g phytic acid per 100 g (dry weight) (Gupta et al., 2015)., whereas phytic acid in SUPRO 620 soy protein isolate is 1.69 g/100 g (dry weight) (Naczk et al., 1986).

Tofu is a less concentrated product and is made from coagulated soy milk. Phytate content of soy milk is around 0.22 g/100 g (Ishiguro et al., 2006).

Up to 30 g soy protein has been demonstrated to be an inhibitor of iron and zinc absorption in single meal studies. While phytate contributes to this effect, some effect persists when phytate is removed, suggesting other compounds are implicated (Lim et al., 2013). In a test meal study of iron absorption in 32 subjects the absorption of iron was shown to increase four- to five-fold when the phytic acid concentration was reduced from 4.9 – 8.4 mg/g to less than 0.1 mg/g in soy protein isolate. Even small amounts were strongly inhibitory and had to be reduced to less than 10 mg phytic acid per meal before a meaningful increase in iron absorption was observed (Hurrell et al., 1992).

For the purposes of this analysis, the phytate content of soy protein has been allocated a value of 1.69 as per the SUPRO 620 proxy.

Phytate in pea protein

The phytic acid content of peas is 0.22 – 1.22 g per 100 g (dry weight) whereas phytate content is higher in pea protein (2.1 g/100 g, av.) (Gupta et al., 2015).

For the purposes of this analysis, the phytate content of pea protein will be allocated a value of 2.1.

Phytate in rice protein

In rice, 90 % of phytate is concentrated in the bran and 4 – 5 % in the endosperm (Goufo & Trindade, 2014). Rice bran consisting of pericarp, aleurone and germ is high in phytic acid (around 6 g/100 g) (Canan et al., 2011).

The phytic acid content of rice varies according to cultivar, genetics and plant nutrition. Phytate content ranges from 0.4 – 0.7 g/100 g in three *indica* (long-grain) brown rice cultivars (Perera et al., 2018) to 0.87 g/100 g in an average of 29 *japonica* (short-medium grain) varieties tested in China (Wei et al., 2007). Other analyses have shown 0.6 g/100 g for Egyptian brown rice (Shallan et al., 2010). A review paper that compiled a database of phytochemicals in several different rice types found an average phytate content of around 0.3 g/100 g in non-pigmented rice varieties (Goufo & Trindade, 2014).

For the purposes of this analysis, the phytate content of rice will be allocated a value of 0.58, which is an average of the figures cited above.

The rice protein ingredient of the Unreal Co. Chick'n sliders PBA is unknown. Generally, rice protein isolate is made by an enzymatic separation of the carbohydrates from the proteins in brown rice (bran, germ and endosperm). According to the US FDA definition, the first hydrolysis and separation produce an 80 % protein product is called concentrate, and the second step produces a 90 % protein product, called an isolate. In the absence of a known branded ingredient, the nutrient composition of a US commercial branded rice protein product called Oryzatein™ is used as a proxy in the table below (FDA, 2015). This product is used in the manufacture of plant protein products.

Table 11. Nutrient content of rice versus rice protein products

Nutrient	Content in grams per 100 g		
	Oryza sativa	Oryzatein™ 80 (concentrate)	Oryzatein™ 90 (isolate)
Fat	2.92	0.3	0.38
Carbohydrate (total)	77.24	11.2	7.54
Dietary fibre	3.5	6.33	6.4
Protein	7.94	85.9	91
Iron	1.47	5.1	7.7
Phosphorous	0.33	0.19	0.20

The phytate content of rice protein could not be found in any publicly available database or commercial product information. In order to estimate the phytate content of rice protein, phosphorous content of Oryzatein™ was used as a proxy to estimate the reduction in phytate from the original rice grain for which the phytate data is available. This is because 50 – 85 % of a plant's phosphorous is present as phytic acid. There is a correlation between phytate and phosphorous content in seeds (Perera et al., 2018).

Using this estimation method, the phytate content of rice protein can be estimated as 60 % of that of rice (0.58 g/100 g) or 0.3 g/100 g.

Phytate in rice starch

Rice starch is listed as the fourth ingredient in the ingredients list of the Unreal Co. Chick'n sliders PBA and is not present in large amounts as ingredients are listed in order by weight. Rice starch is a finely textured flour made from the endosperm of rice and is added for functional rather than nutritional purposes. For example, the soft gel it forms conveys a 'melt-in-mouth' quality (Wani et al., 2012).

There is no data available on the phytate content of rice starch, so an estimate was developed. Polished rice was used as a proxy as it is high in carbohydrate (80 % according to FoodData Central (USDA, 2019)). This is likely to underestimate phytate content as cereal starches contain only 1 – 2 % protein and lipid (Wani et al., 2012) and approximately 98 % carbohydrate. The phytate content of rice depends on the method of milling. In polished rice, phytate content varies from 0.012 – 0.06 g/100 g (median 0.036 g/100 g), based on data from 45 Thai rice mills (Tuntawiroon et al., 1990).

The median figure of 0.036 g/100 g has been allocated to rice starch in the calculations of phytate content in Table 10.

Estimated bioavailability of iron and zinc in Plant-Based Alternatives (PBAs)

All the PBAs selected are not fortified with iron or zinc (they do not have these nutrients added). This means the nutrients present are naturally occurring, and their bioavailability depends on the plant food ingredient present, the type of food matrix and the level of processing. In all the cereal and legume-based PBAs in this analysis, the primary protein has been isolated or extracted, and this effects phytate content.

Estimating iron bioavailability in PBAs

Single meal studies of phytate in bran or as sodium phytate can reduce iron absorption by 18 %. These studies show that the inhibitory effect increases with phytate content such that 250 mg phytate can reduce absorption by 82 %. However, interactions throughout the whole food matrix in a meal may temper the effect of phytate on iron absorption (Lim et al., 2013).

Completely removing phytate from cereal- and legume-based complementary foods has been shown to increase the percentage of iron absorption by as much as 12-fold (increasing from 0.99 % to 11.54 %) in a single-meal study when the foods were reconstituted with water (Hurrell et al., 1992).

The absorption of iron from legumes such as soybeans, black beans, lentils, mung beans, and split peas has been shown to be very low (0.84 – 1.9 %) and similar to each other (S. Lynch et al., 1984). A clinical study measuring the absorption of iron from red kidney beans (*Phaseolus vulgaris*) in women in a single food test meal found iron absorption was 1.5 % (King et al., 2000). In a mixed vegetarian meal, non-haem iron absorption was measured at 2.5 % (Hallberg & Hulthén, 2000). The relative absorption of non-haem iron of the PBAs with inhibitors present in this study will be estimated at 1.5 %.

From a whole diet perspective, vegetarians are advised to consume 80 % more iron than those consuming an omnivorous diet. However, this is not directly applicable to comparing iron bioavailability in plant-based foods in a single meal. Cook and colleagues (1991) found that non-haem iron absorption was 6.1 % on average for a single meal. However, when they compared non-haem iron availability from single meals with that of a diet consumed over a two-week period they found while there was a 4.5-fold difference in absorption in single meals, there was only a two-fold difference in absorption measured over a two week period. This suggests there are factors that ameliorate inhibition of iron absorption across the day or week. Single meal studies may exaggerate the impact of factors affecting iron bioavailability.

The Food and Agriculture Organisation of the World Health Organisation (FAO/WHO, 1989) identified three levels of bioavailability and associated compositional characteristics of different diets. The iron in a typical western mixed diet was judged to have 15 % bioavailability, mostly vegetarian diets with small amounts of meat and fish were judged to have 10 % bioavailability. In comparison, strict vegetarian diets were judged to be 5 % bioavailable. Other studies have found slightly higher estimates, but the US used a benchmark of 18 % bioavailability for adults consuming a mixed diet in the development of their Dietary Reference Intakes (FNB & IM, 2001). An absorption rate of 18 % is also assumed in the Nutrient Reference Values for Australia and New Zealand (NH&MRC, 2006). They estimated that the bioavailability of iron from a vegetarian diet is approximately 10%, compared with 18 % from a mixed Western diet. Hence, they recommend that the requirement for iron is 1.8 times higher for vegetarians. They say even lower bioavailability diets (approaching 5 % overall absorption) may be encountered with strict vegetarianism (FNB & IM, 2001).

The relative absorption of non-haem iron of the PBAs without inhibitors present (i.e. phytate, soy protein) in this study will be estimated at 5 %. This 5 % figure is the overall FAO/WHO (1989) estimate of iron bioavailability of a vegetarian diet. However, it also aligns with the mixed vegetarian meal absorption rate of 2.5 % if it were doubled to account for the other ameliorating factors in the whole diet as found by Cook (1991).

The relative absorption of non-haem iron of the PBAs with inhibitors present in this study will be estimated at 1.5 %.

The relative absorption of non-haem iron of the PBAs without inhibitors present (i.e. phytate, soy protein) in this study will be estimated at 5 %.

Table 12. Estimated bioavailable iron per serve of chicken versus PBAs

PBA product (serve size)	Iron (mg)	Main ingredient	Phytate per serve (g)	Estimated iron absorption	Bioavailable iron per serve (mg)
Quorn pieces (100 g)	2 non-haem	mycoprotein	0	5 %	0.1
Tofurkey Chick'n (90 g)	2 non-haem	Wheat gluten tofu	0.44	5 %	0.1
Sunfed Chicken Free Chicken (100 g)	9.1 non-haem	Pea protein	2.1	1.5 % (very high phytate)	0.14
Unreal Co. Chick'n sliders (75 g)	1.2 non-haem	Soy protein Rice protein Rice starch	1.53	1.5 % (very high phytate & soy inhibition)	0.06
Chicken meat (raw) composite	0.54 haem & non-haem	Chicken meat	0	25 % haem iron 20 % non-haem iron	0.12

Estimating zinc bioavailability in PBAs

Predicting zinc absorption is less complex than predicting iron absorption because only two factors account for a considerable degree of variance, *viz* the amount of zinc consumed and phytate content, and absorption is less influenced by factors in the individual. Several models have been proposed to predict zinc absorption with varying phytate intake.

Hunt (2010) has externally validated one model by Miller (2007) (see Figure below). Choosing an example point on this graph and converting mmol to grams (0.1 mmol = 6.54 mg zinc) of around 6 mg daily zinc intake; at a (high) phytate to Zn ratio of 30 (0.1 mmol TDZ = 6.54 mg; TAZ 0.23 mmol = 1.5 mg) when there is a high phytate to zinc ratio, zinc absorption is approximately 23 % (1.5 mg/6.53 mg).

With a zero phytate content, absorption reaches 0.45 mmol zinc (2.9 mg) TAZ against a daily intake of 6.54 or an absorption rate of 44 % (2.9 mg/6.54 mg).
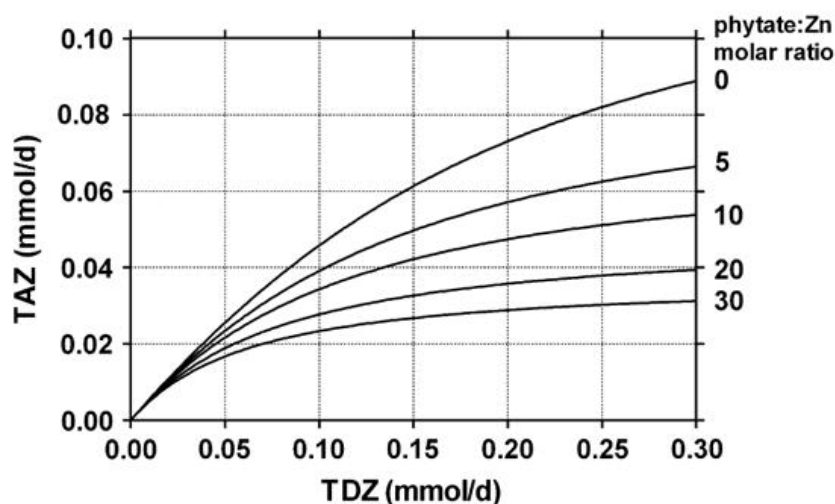
Figure 6. Predicted Total Absorbed Zinc (TAZ) from Total Daily Zinc (TDZ) from diets with different phytate contents (reproduced from L. Miller et al., 2007)

The European Food Safety Authority (ESFA, 2014) has set adult dietary zinc requirements for four levels of phytate intake (300, 600, 900, 1200 mg/day), acknowledging phytate food composition data is incomplete (Gibson et al., 2018).

Table 13. Average dietary zinc requirements depending on phytate and body weight (ESFA, 2014)

Bodyweight (kg)	Physiological requirement (mg Zinc/day)	300 mg/day dietary phytate	600 mg/day dietary phytate	900 mg/day dietary phytate	1200 mg/day dietary phytate
72.7	3.4	8.2	10.2	12.1	14
59.1	2.9	6.3	7.7	9.1	10.4
Average of two body weights	**3.15**	**7.25**	**8.95**	**10.6**	**12.2**

The EFSA average daily zinc requirements for different phytate intake levels are based on saturation response model predictions of total absorbed zinc, as shown in Figure 5. Again, this is data used for recommending population intakes rather than zinc bioavailability of individual foods; however, it is the best available approximate for this analysis. Examination of this curve suggests that absorption is around 20 % with high levels of phytate and 60 % at zero phytate.

The zinc absorption from chicken is estimated at 60 % to reflect zero phytate content.
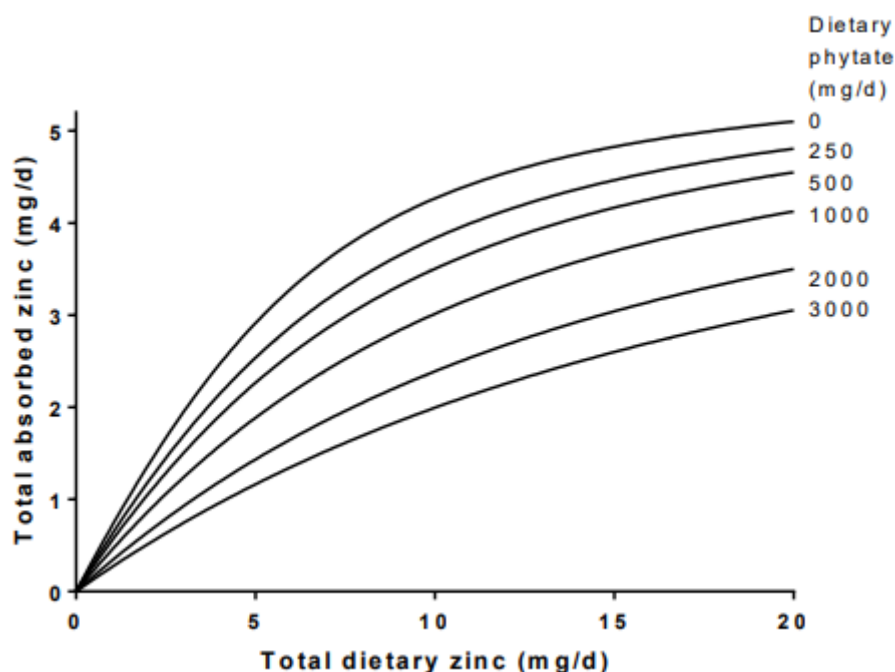
Figure 7. Saturation response model prediction of total absorbed zinc for selected levels of phytate intake (ESFA, 2014)

Nordic nutrition recommendations for zinc assume 40% absorption in an omnivorous diet (Nordic Council of Ministers, 2014). The Nutrient Reference Values for Australia and New Zealand assume a 24 % zinc absorption rate for men and 31 % for women (NH&MRC, 2006).

A zinc absorption study in young women consuming vegetarian meals over five days compared with meat containing meals (two kinds of pork), found that total zinc absorption was increased by 45 – 50 % in the meat diets compared with the vegetarian diet when both diets contained the same high levels of phytate. The subjects absorbed an average 1.8 mg of the 7.5 mg (24 %) of zinc contained in the vegetarian meal (Kristensen et al., 2006). Another absorption study of micronutrients from vegetarian meals found that 20.2 % of zinc was absorbed (Agte et al., 2005). A clinical study measuring the absorption of zinc from red kidney beans (*Phaseolus vulgaris*) in women in a single food test meal found zinc absorption was 13 % (King et al., 2000). A rate of zinc absorption for the plant-based PBAs with absorption inhibitors (phytate, soy protein) will be estimated at 20 %.

Quorn pieces have a zinc content of 9 mg per serve and zero phytate, therefore according to the Miller model (2007) this is 0.14 mmol of total zinc, and 0.56 mmol (3.7 mg) of this will be absorbed (41 %). According to the EFSA model, 4 mg is expected to be absorbed (44 %), which is good agreement between the two models. Quorn pieces will be allocated a zinc absorption of 40 %.

A rate of zinc absorption for the plant-based PBAs with absorption inhibitors (phytate, soy protein) will be estimated at 20 %.

Quorn pieces will be allocated a zinc absorption of 40 %.

Table 14. Estimated zinc bioavailability of chicken versus plant-based alternatives (PBAs)

PBA product (serve size)	Main ingredient	Zinc (mg)	Estimated phytate per serve (g)	Estimated zinc absorption	Bioavailable zinc content per serve (mg)
Quorn pieces (100 g)	mycoprotein	9	0	40 %	3.6
Tofurkey Chick'n (90 g)	Wheat gluten tofu	1^	0.44 (440 mg)	20 %	0.2
Sunfed Chicken Free Chicken (100 g)	Pea protein	3.2	2.1 (2,100 mg)	20 %	0.64
Unreal Co. Chick'n sliders (75 g)	Soy protein Rice protein Rice starch	0.36*	1.53 (1,530 mg)	20 %	0.072
Chicken meat (raw) composite	Chicken meat	1	0	60 %	0.6

*Data unavailable from the company and approximated using soy protein isolate ingredient. ^Data unavailable from the company and calculated using nutritional data of wheat gluten and tofu's primary ingredients.

Calculation of the Nutritional Quality Corrected Functional Unit (NQCFU)

Nutritional Quality Score (NQS) is a tool to account for nutritional differences between chicken meat and the PBAs for the four target nutrients. When conducting the LCA, the NQS for each nutrient will be used to calculate a Nutritional Quality Corrected Functional Unit (NQCFU) for each PBA. The nutrients used are target nutrients expected to be delivered by the meat and alternatives food group in the Dietary Guidelines for Australians (DGAs).

- Protein quality (Weighted Protein Score; WPS) - comparative

- Presence of vitamin B12 (beneficial component) – presence/absence

- Bioavailable iron content - comparative

- Bioavailable zinc content – comparative

A simple method of calculating the NQCFU is to average the comparative nutrient scores of the PBAs against chicken. Nutrients in PBAs that score less than chicken would require a corrective increase in the amount of PBA entered into the LCA to account for the shortfall. In the cases where the PBA provides more of a nutrient than chicken, the amount entered in the LCA for the PBA would be less. This comparative method does not allocate increased importance of any nutrient, but rather provides an overview of nutrient equivalence. However, it does not account for vitamin B12 that is absent from the PBAs and thus precludes a comparative calculation. This is a significant methodological limitation because it fails to account for the nutritional contribution of vitamin B12 from chicken.

The NQCFU is calculated as follows:

$$NQCFU = [WPS\,(chicken/PBA) + Iron\,(chicken/PBA) + Zinc\,(chicken/PBA)] \div 3$$

Where:
WPS: Weighted Protein Score
Iron: bioavailable iron per serve
Zinc: bioavailable zinc per serve

NQCFU calculation for Quorn pieces

$$Quorn\,NQCFU = [WPS\,(20.3/14) + Iron\,(0.12/0.1) + Zinc\,(0.6/3.6)] \div 3 = 0.92$$

NQCFU calculation for Tofurky Chick'n pieces

$$Tofurky\,NQCFU = [WPS\,(20.3/9) + Iron\,(0.12/0.1) + Zinc\,(0.6/0.2)] \div 3 = 2.15$$

NQCFU calculation for Sunfed Chicken Free Chicken

$$Sunfed\,NQCFU = [WPS\,(20.3/29.52) + Iron\,(0.12/0.14) + Zinc\,(0.6/0.64] \div 3 = 0.83$$

NQCFU calculation for Unreal Co. Chick'n sliders

$$Unreal\,NQCFU = [WPS\,(20.3/9.63) + Iron\,(0.12/0.06) + Zinc\,(0.6/0.072)] \div 3 = 4.15$$

Table 15. Nutritional Quality Corrected Functional Unit (NQCFU) calculation from protein, iron and zinc nutritional quality score (NQS) data for chicken meat versus plant-based alternatives (PBAs)

PBA product (serve size)	Main ingredient	Weighted Protein Score (WPS)	Vitamin B12 per serve (ug)	Bioavailable iron per serve (mg)	Bioavailable zinc content per serve (mg)	Quality corrected functional unit ratio (chicken/PBA)	Quality corrected functional unit (g)
Quorn pieces (100 g)	Mycoprotein	14	0	0.1	3.6	0.92	92
Tofurkey Chick'n (90 g)	Wheat gluten	9	0	0.1	0.2	2.15	193.5
Sunfed Chicken Free Chicken (100 g)	Pea protein	29.52	0	0.14	0.64	0.83	83
Unreal Co. Chick'n sliders (75 g)	Soy protein	9.65	0	0.06	0.072	4	300
Chicken meat (raw) composite	Chicken meat	20.3	0.465	0.12	0.6	1	100

Scoping study results

Greenhouse gas emissions

For ease of comparison, the product names were abbreviated to CM (chicken meat), PB (pea-based), QP (Quorn pieces) and LM (laboratory meat). Results are provided per 0.1 kg portion of Australian CM or equivalent portion consumed. To be nutritionally equivalent, the following product mass was required, 0.083 kg (PB), 0.092 kg (QP) and 0.1 kg (LM). Scenarios were run for the PBAs and laboratory meat where production was in the regions where the products were typically made prior to import to Australia. The regions used for the scenarios were the European Union (EU), New Zealand (NZ), the United Kingdom (UK) and the United States (US).

Impacts from GHG were highest for LM, followed by CM and QP, with PB having the lowest reported impacts (see Figure 9). The differences were not substantial between Australian chicken and Quorn (15 % higher for CM compared to QP) but were larger between chicken and pea (56 % higher for chicken). This difference was less apparent if Australian chicken was compared with Australian or US produced pea products, which were around 24 - 30% lower impacts for the pea-based product. We note that these results assumed no LUC for the pea product, which may be an under-estimate depending on the location of the pea production.
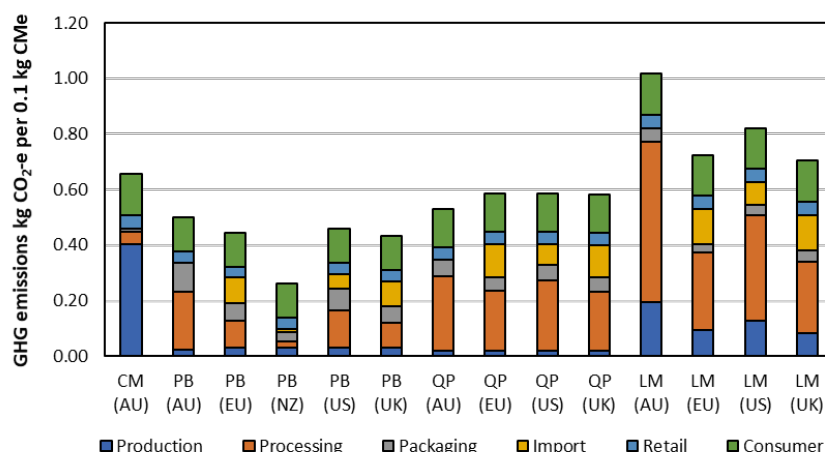
Figure 8. Greenhouse gas emissions (kg CO_2-e per 0.1 kg equivalent chicken meat portion) for chicken meat (CM), pea-based (PB), Quorn pieces (QP) and laboratory meat (LM) produced and processed domestically or produced, processed and imported from abroad

With respect to the PBAs, the greatest difference between products produced domestically or imported from abroad were seen in the products which used a large quantity of electricity during processing. The PB and LM products both fell into this category, whereas QP uses steam during processing instead, which in the present study was produced with lower impacts than electricity, though this will be strongly dependant on the fuel source for the steam generation and that would be a sensitive and uncertain factor in the present model. This difference is driven by the impacts of producing electricity in each of these regions, with New Zealand and the EU, who both utilise renewable energy technology, having significantly lower impacts than Australia which is still largely reliant on coal.

Transporting the PB and LM products from abroad resulted in lower impacts than producing these products in Australia, because of the high impact energy network in Australia. The reverse was true for QP, which used significantly less electricity during processing.

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Fossil energy use

Fossil energy use for CM was 5.4 MJ which was similar to PB (with the exception of PB produced in New Zealand) and was substantially lower than QP or LM. (Figure 9).

Figure 9. Fossil energy use (MJ per 0.1 kg of equivalent chicken meat) portion for chicken meat (CM), pea-based (PB), Quorn pieces (QP) and laboratory meat (LM) produced and processed domestically or produced, processed and imported from abroad

Fossil energy use impacts followed the same general trends as GHG emissions, with the benefits of producing PB and LM products abroad being greater than the impacts of transporting these products to Australia. Again, the reverse was true for QP.

Water stress

Water stress impacts for CM were 7.1 WSI-e L which was similar, and in some cases less than PBAs, and was significantly less than the 42.2 to 57.1 WSI-e L for LM. Water stress impacts for the value chain are provided as Figure 10, with hotspots for each section of the value chain shown.

Figure 10. Water stress (WSI-e litres per 0.1 kg equivalent chicken meat portion) for chicken meat (CM), pea-based (PB), Quorn pieces (QP) and laboratory meat (LM) produced and processed domestically or produced, processed and imported from abroad

The LM product had significantly greater water consumption and subsequent water stress than all the other products, and this was a result of the high freshwater consumption in the production of cyanobacteria used as the basis of cultured meat. The differences in regional water stress for the PB and LM products were largely a result of differences in how water-stressed each region was and was not reflective of changes in absolute water consumption, which remained fairly constant between products.

Land occupation

Land occupation impacts for CM were 1.66 m^2 yr^{-1} and this compared to between 0.67 and 1.13 m^2 yr^{-1} for PB, 0.09 and 0.10 m^2 yr^{-1} for QP and 0.11 and 0.28 m^2 yr^{-1} for LM. Land occupation impacts for the value chain are provided as 1, with hotspots for each section of the value chain shown.

Land occupation impacts for CM were greater than the PBAs, which was expected, because chickens need to convert grain to meat, adding an additional biological inefficiency in the process. That noted, the PB product also had significant land requirements to grow the peas and other vegetable ingredients used in the product. In contrast, QP and LM products had minimal land requirements.
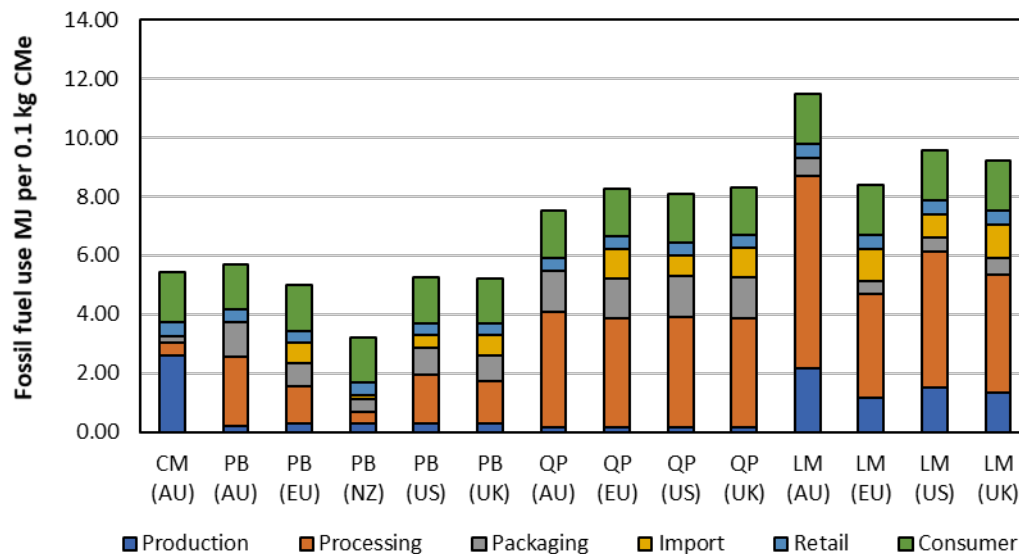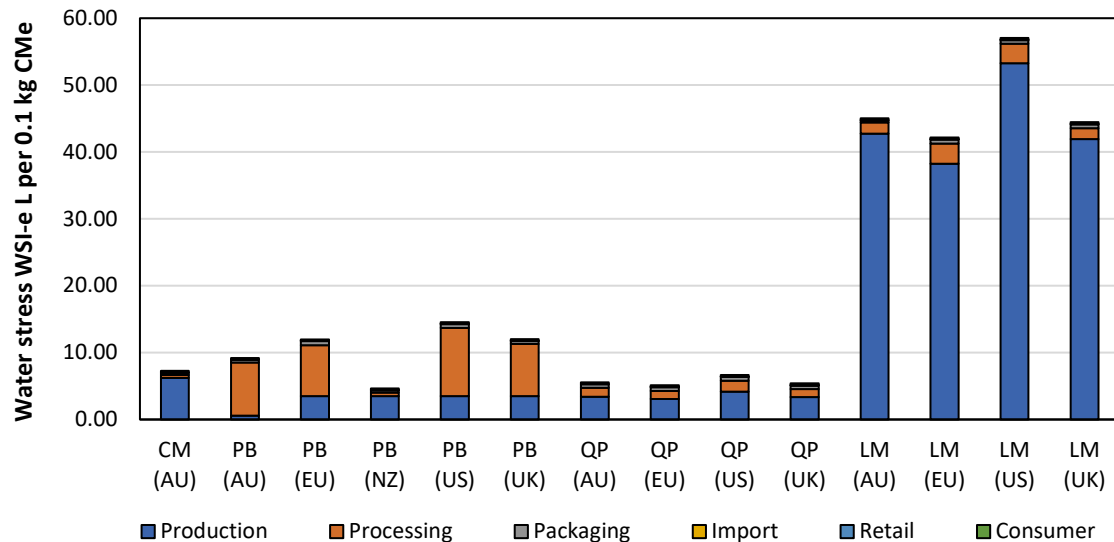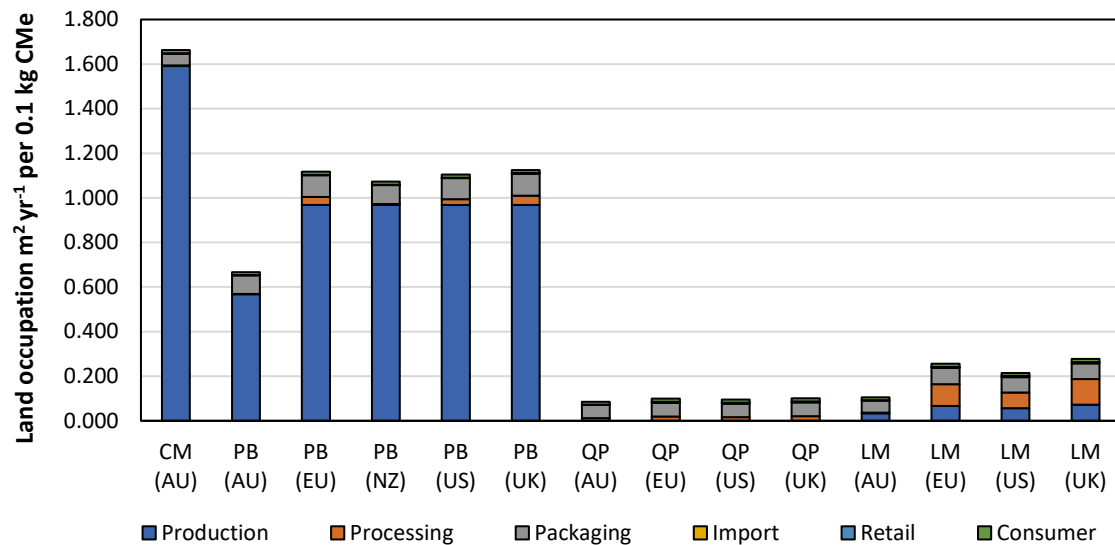

Figure 11. Land occupation (m^2 yr^{-1} per 0.1 kg equivalent chicken meat portion) for chicken meat (CM), pea-based (PB), Quorn pieces (QP) and laboratory meat (LM) produced and processed domestically or produced, processed and imported from abroad

Sensitivity analysis

A sensitivity analysis was performed by increasing sensitive model parameters, which were identified as those that resulted in a 2.5 % change in PBA value chain impacts when altered by 50 % or less. For simplicity, this was scaled to a percentage change per increase of the parameter by 100 % (i.e. doubled) for presenting values in this section of the report. The sensitivity analysis results are presented for products produced domestically in Australia, except for the section detailing the sensitivity during product import. This impact change could have been in GHG emissions, fossil energy use, water stress or land occupation. These parameters are detailed in the following sections.

PBA production

Water consumption for mycoprotein cultivation was the sole sensitive parameter in the production phase of the Quorn product, and doubling this value increased the value chain water stress impacts by 18.2 %. The cyanobacteria cultivation stage of laboratory meat production had four sensitive parameters: electricity consumption in fertiliser production, and electricity consumption, water consumption and land occupation in cultivation. Doubling fertiliser production electricity consumption resulted in an increase in both GHG and fossil energy use impacts of 7.0 % across the value chain. Whereas, doubling electricity consumption resulted in an increase in both GHG and fossil energy use impacts of 8.0 %. Doubling water consumption during cyanobacteria cultivation resulted in a 94 % increase in the value chain water stress impacts and represented an extremely sensitive parameter.

PBA processing and packaging

Electricity and water consumption during the processing of the PB product were found to be sensitive. Doubling general electricity consumption for processing increased both GHG and the fossil energy use value chain impacts by 13.8 %. Doubling the electricity consumption attributed to liquid nitrogen used in processing increased both GHG value chain impacts by 21.2 % and fossil energy use by 21.0 %. Doubling water consumption during processing increased water stress value chain impacts by 10.2 %.

The QP product used significant quantities of steam during processing, and doubling this quantity resulted in increases in value chain impacts of between 3.2 % and 37.6 %. This was a very sensitive parameter in the calculation of GHG and fossil energy impacts across the value chain. Water consumption during processing was also significant, and doubling this value resulted in a 19.2 % increase in water stress impacts.

Electricity consumption in the processing of the LM product was very sensitive for GHG and fossil energy use impacts. Combined, doubling electricity during processing increased both GHG and fossil energy impacts by 50.6 %.

There were several parameters in the packaging processes that were sensitive across all PBA products. These included the quantity of packaging materials (cardboard, plastic and wood pallets) and the electricity consumed during packaging. The number of wood pallets used to transport the product was very sensitive for land occupation: doubling this value increased the land occupation value chain impacts by 5.4 % for PB, 46.8 % for QP and 41.4 % for LM products. Doubling the quantity of cardboard and plastic packaging increased value chain impacts by up to 16.4 % and was reasonably sensitive. Doubling the electricity consumption during packaging increased GHG and the fossil use value chain impacts by between 27.4 % and 27.8 % for PB, and 9.0 % for LM products. The electricity consumption in QP product packaging was reported in the processing stage and could not be separated. Packaging impacts were unexpectedly significant and merited further investigation.

Product import

The distance travelled when shipping any of the PBA products to Australia was sensitive for all PBA products. For example, doubling this value for the QP product imported to Australia increased GHG value chain impacts by 16.4 % and fossil energy use by 8.2 %. In this case, the imported product was already shipped a distance representing nearly the maximum possible distance (UK to Australia) suggesting distance is unlikely to substantially increase impacts despite this sensitivity.

Retail and consumption

Electricity consumption was sensitive during the retail phase of all PBA products. Doubling retail electricity consumption increased both GHG and fossil energy use value chain impacts by between 3.2 % and 5.6 % for the PBA products.

The distance consumers travelled to their local supermarket, and the fraction of their shop attributed to the product were sensitive parameters. Doubling this distance from 5 to 10 kilometres (10 to 20 for a round trip) was sensitive for all impacts investigated, increasing value chain impacts for the PBAs by up to 16.2 %. Doubling the fraction of the shopping trip attributed to the product had the same effect.

As expected, losses during both retail and consumption were very sensitive. These included retail, fridge, plate and cooking losses. Any losses require the production and processing of more product to maintain the same functional unit, and thus impacts are increased throughout the value chain. Doubling the losses of each of these losses increased value chain impacts between 7.8 % and 20.6 % for PB, 7.6 % and 20.0 % for QP, and between 7.8 % and 21.0 % for LM products.

Doubling the electricity consumption for cooking the product increased both GHG and fossil energy use value chain impacts by between 6.6 % and 12.8 %.

The equivalence of each of the PBAs to chicken meat, also effectively the quantity of product consumed, was the most sensitive parameter in the study. As expected, doubling the quantity of product required, doubled the environmental impacts of each PBA.

Implications

Comparison of chicken and PBAs

In the present study, we found it was possible to determine comparable products taking into account nutritional quality. Interestingly, this resulted in less mass of the plant-based product being required for the comparison than the chicken product. In other words, comparisons made without accounting for nutritional aspects would slightly disadvantage the PBA, not the chicken product.

Our review of the literature showed that soy-based and laboratory meat products could have higher impacts than chicken meat, and that these findings were sensitive to assumptions. Particularly with respect to soy, the impact of LUC emissions in some regions of the world is highly significant.

Multiple PBA products marketed to compete with chicken in the Australian market utilise environmental claims to support their product (see Table 16) with several claiming very large differences between the PBA product and either 'meat' or more specifically in the case of Quorn, with chicken.

We observed several problems when evaluating these claims and findings. Firstly, the comparisons were all done in overseas markets, where impacts from chicken are typically higher, and impacts from PBAs are expected to be lower because the transport requirement for these products from country of manufacture to Australia were not taken into account. Thus, there is doubt about the claims when they are made in the Australian market.

Secondly, in the present scoping study we included LUC impacts for Australian chicken, which was associated with limited amounts of soymeal used in the feed supply chain. It is quite possible that LUC impacts also exist in at least some PBA supply chains, but without sufficient data, this couldn't be assessed.

The net result of our analysis of the literature and the scoping analysis results suggested that the comparison was much closer to equivalent than is commonly claimed. For example, the claimed 70 % lower carbon footprint of Quorn than chicken is much more than our scoping study results, which showed 15 % higher impacts for chicken than Quorn, which considering the degree of uncertainty in the results is marginal and likely to be not statistically significant. For other impact categories of interest, the scoping results showed impacts such as fossil fuel energy may be lower for chicken, though this was very sensitive to the region of the world where the PBA was manufactured.

New Zealand company Sunfed claims its pea-based met alternative product is "healthier for the planet" (see Table 16). These claims are not supported by any specific research that could be reviewed. Our review of pea-based products indicated that mean impacts were lower than chicken meat for GHG, land occupation and water in the literature and in our scoping study, though the comparison to Australian chicken meat showed less of a contrast. The scoping results showed GHGs were 36 % lower than Australian chicken meat on average. In contrast, energy was found to be higher for the pea-based product as it required high levels of processing.

Impossible Foods have claimed that their meat alternative product, the Impossible Burger, uses 96 % less land, 87 % less water, and generates 89 % less GHG emissions when compared to beef (Khan et al., 2019). However, compared to Australian chicken meat, the Impossible Burger required 74 % more water and GHG emissions were 12 % lower. This result for GHG is marginal. The land occupation requirements for this product were 90 % lower than that of Australian chicken and energy impacts were not reported in by Khan et al. (2019) so could not be compared, though energy associated with other pea-products is high and potentially higher than Australian chicken meat.

Further to this, the actual magnitude of the differences between chicken and PBAs should be taken into account; for a given portion, the difference between chicken and the pea-based products is less

than the emissions from driving a car about 1km. In the context of choices available to consumers, other factors are more significant than this issue, but current marketing emphasises the importance of this dietary choice.

Table 16. Public claims regarding chicken alternatives

Product	Claim	Where claims were made	Evidence cited to support the claim	Source
Quorn pieces	"70 % lower carbon footprint than chicken."	Quorn Sustainable Development Report, 2019, pg. 7.	Hsu et al. (2018) 'Quorn Footprint Comparison report.'	https://www.quorn.us/files/content/Sustainable-Development-Report2019.pdf
Sunfed Chicken Free Chicken	"Sunfed meats are … healthier, both for you and the planet."	Sunfed Foods website homepage	No evidence provided	https://sunfedfoods.com/
Beyond Meat	"By shifting from animal to plant-based meat, we can address four growing global issues: human health, climate change, constraints on natural resources, and animal welfare."	Beyond meat website	LCA by University of Michigan – Found BB uses less water, land, energy, and GHG compared to a beef burger, (Heller and Keoleian, 2018)	ht54tps://www.beyondmeat.com/about/
Impossible Burger	"Just one Impossible Burger (instead of a burger made from cows) will save the equivalent of: - 96 % less land - 87 % less water - 89 % fewer GHG emissions."	Impossible Foods website	Khan et al. (2019) – Comparative environmental LCA of the Impossible Burger with a conventional ground beef burger	https://impossiblefoods.com/mission/

Limitations

This study was a literature review and scoping analysis. As such, it is a preliminary analysis and further work is required to generate more robust comparisons. We note the following limitations:

- The nutritional comparison was limited in some respects. In particular, NQCFU does not account for vitamin B12 that is absent from PBAs. This means one benefit from chicken meat was not accounted for in the present review and further work would be required to incorporate this.
- The laboratory meat studies were heavily based on assumptions because the process and products are under development. These results may change substantially in the future.
- Data for the scoping analysis of the PBAs were drawn from multiple, disparate sources depending on what could be found. These data sources would need improvement to substantiate a robust comparison.
- Information around the impact of LUC on PBAs was lacking. This could increase impacts from some PBAs.

Conclusions and recommendations

As more PBA products emerge on Australian supermarket shelves and continue to be marketed as more environmentally sustainable than traditional meats, the need to accurately quantify these impacts increases. There is growing consumer interest in vegan, vegetarian and reduced meat diets and products; however, there are knowledge gaps, and the lack of transparency from some PBA companies and global application of specific comparisons is concerning.

The literature review highlighted the high variability in the literature data, particularly regarding system boundaries, functional units, allocation and location of production. More detailed research into these evolving meat alternative technologies is needed. Production conditions and product inputs for each product vary between different countries. For a fair comparison between the different PBA products and cultured meats, uniform production conditions would be required. Many PBA and cultured meat LCAs relied heavily upon secondary data, calculations and assumptions. The use of up-to-date primary data would allow for a more accurate comparison between the products and traditional meats.

The nutritional qualities of meats, plant-based alternatives and cultured meats vary and, therefore, a comparison of environmental impacts should be based on nutritional equivalence. In order to make fair and meaningful comparisons between chicken and plant-based food alternatives, the concept of Nutrition Quality Corrected Functional Units (NQCFU) was developed based on nutritional equivalence (based on a nutrition quality score) and culinary equivalence (incorporating serving size) that incorporate a cooking method to represent foods as consumed. Future research needs to utilise a system such as this to ensure the different nutritional profiles of each product are taken into account. Interestingly, considering the comparisons made here, we found that chicken meat is not disadvantaged by being compared on a 'product' basis without accounting for nutritional aspects. In contrast, PBAs may in fact be disadvantaged by such a comparison.

The results of our scoping study and literature review indicated that while there are noticeable differences in some impact categories between products, the impacts of chicken meat and PBAs are overall relatively similar. Specifically, the study found that:

1. The differences are relatively modest and not consistent. GHG impacts were slightly higher for Australian chicken meat compared to PBAs, though the differences are modest when compared with other common activities. In contrast, the impacts were often higher for PBAs when energy and water is considered.

2. In comparison with laboratory-based meat, Australian chicken meat typically had lower environmental impacts across most categories, though it is noted the laboratory meat studies are heavily reliant on assumptions as few operational production processes exist at present.

A comparison between selected PBA products promoted in the Australian market for their apparent superior environmental credentials over traditional meats found that the literature and the scoping analysis did not strongly support many of the claims and, in some cases, the claims were contradicted. This was partly because the supporting results were not based on the Australian market context. This implies that consumers may be misinformed by these claims in the Australian market.

Based on these findings, we recommend that a comparative analysis be undertaken, focused on comparing chicken meat with PBAs in the Australian market. Considering laboratory meat is not currently available and the comparison must be based heavily on assumptions, we recommend not including this in a follow up study if the purpose is to counter public claims, because the results would be more exposed to criticism for being assumption laden.

While the outcomes of this analysis can't be guaranteed until the data for the PBAs and laboratory meat are updated, we consider it likely that the study will find the differences between Australian chicken and PBAs will be modest and, in some cases, PBAs impacts may be higher than chicken meat.

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Appendix

How the generic (composite) raw chicken meat nutrient composition data was derived

In order to establish nutrient data for raw chicken, the nutrient composition for a composite sample of raw chicken cuts was calculated. The weighted composite was based on the four largest (by volume) selling chicken cuts produced for the Australian retail market – breast fillet, thigh fillet, drumstick and whole bird. This industry data was supplied by ACMF (Yates & Senior, 2020).

- Whole birds: 40.5 %

- Breast fillet: 33.1 %

- Thigh fillet: 16.3 %

- Drumstick: 10.1 %

These percentages were converted to grams and entered into FoodWorks 10 Professional nutritional analysis software (Xyris Software Pty Ltd, 2009) to produce nutrient data per 100 g for a composite 'chicken meat' product. The database underpinning the software is AusFoods that draws on the new Australian Food Composition Database (AFCD) and AUSNUT 2011 – 13 from FSANZ (2015).

The nutrient data is based on the edible portion only (without bones). In order to match the fat content of 10 % used in the LCA, the breast allocated a 'with skin' option was tweaked slightly to bring the total fat content up to 10 % (see Figure 12). This fat content of 10 % is slightly higher than the composite chicken nutritional data figure of 7 % fat used for consumer and industry communications. However, this data is for cooked chicken (as consumed), and some fat is rendered off during cooking. Therefore a 10 % fat content for raw chicken is a nutritionally reasonable allocation:

Chicken, whole, with skin, homemade, uncoated, raw **40.5 g**

Chicken breast, without skin, uncoated, raw **29 g**

Chicken breast, with skin, raw **4.1 g**

Chicken, thigh, with skin, raw **16.3 g**

Chicken drumstick with skin, uncoated, raw **10.1 g**
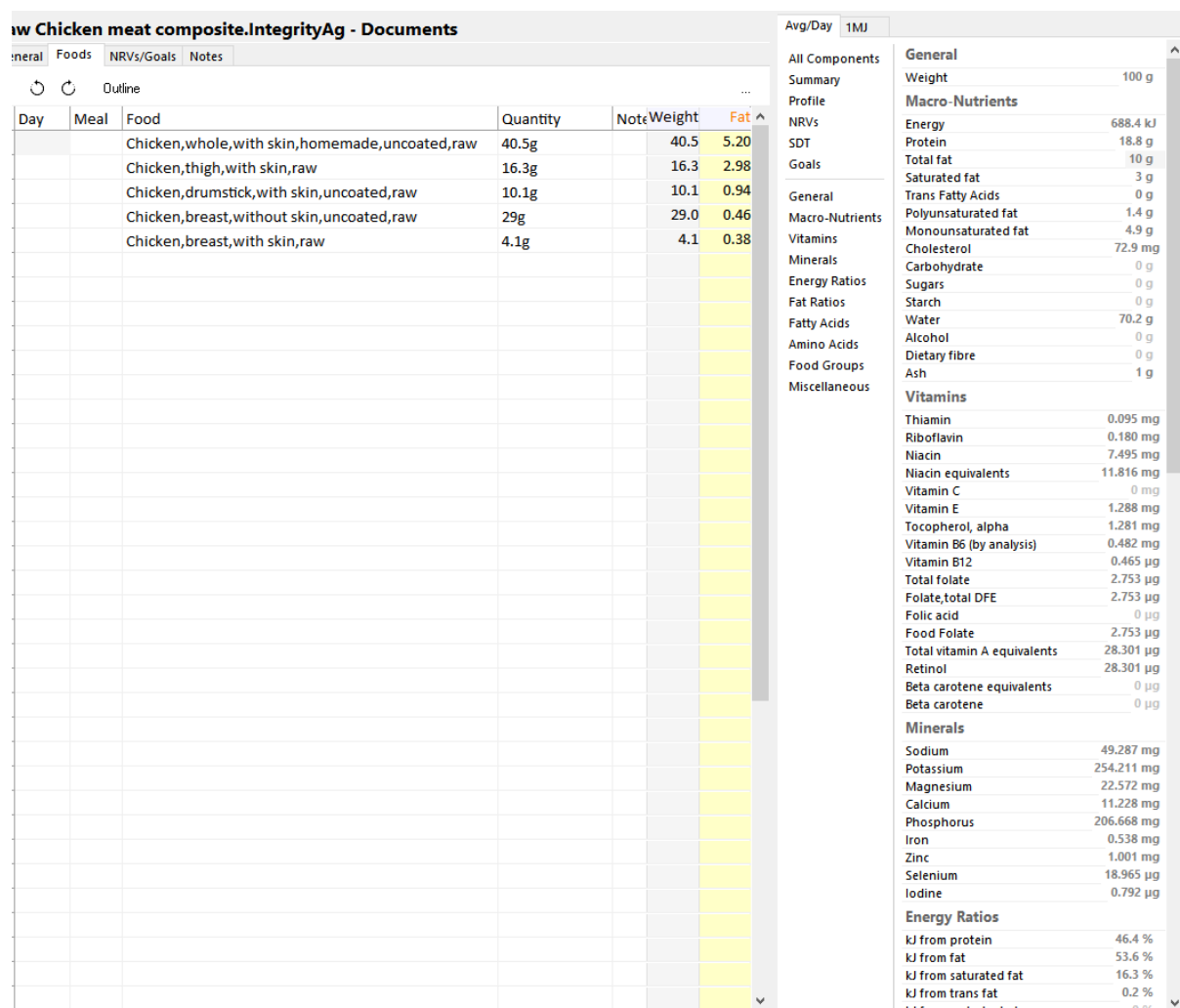
Figure 12. Screenshot of FoodWorks Nutritional Analysis showing the breakdown of chicken cut allocation to create chicken meat composite data and nutrient content data results (Xyris Software Pty Ltd, 2009)

Table 17. Nutrient data for composite chicken meat with 10 % total fat content

Energy (kJ)	Protein (g)	Fat, total (g)	Saturated fat (g)	Sodium (mg)	Vitamin B12 (ug)	Iron (mg)	Zinc (mg)	Omega-3 fats (g)
688	18.8	10g	3	49	0.465	0.538	1	0.021

ACMF industry data derivation

The weighting data was provided by the ACMF in 2019 and was based on data provided by companies supplying approximately 50 % of all chicken cuts to the Australian market. The ACMF considers this is to be representative of the industry more broadly.

Approximately 51 % of all raw chicken cuts (by volume) leaving Australian primary processing plants, for in-home consumption, fall into the categories of whole birds (35 %), raw breast fillet (8.75 %), raw thigh fillet (4 %) and raw drumsticks (3.25 %) available in those formats at retail. In the case of whole birds, the volume of both whole raw birds and rotisserie/BBQ chickens available at retail outlets was used. These volumes were used in the determination of the product weightings.

The remaining (49 %) of chicken cuts leaving processing plants comprise cuts which are individually sold in smaller volumes (e.g. wings; mini-drums, bone-in product lines etc.), or are enhanced or partially value added (e.g. diced, marinated, minced, crumbed, coated or formed into kebabs or schnitzels etc.), or which are supplied to cooking plants for use in value-added (cooked) product ranges.

The weightings were then adjusted to account for that proportion of production volume that leaves the primary processing plant as whole chicken but is broken down before reaching the consumer into specific cuts by secondary processors. The industry estimated that approximately 40 % of raw whole chickens that leave the primary processing plant go to secondary boners/processors, who break the whole birds down to other cuts that are then sold at retail, mostly as raw products. This 40 % of whole birds was therefore reallocated to the other cuts, based on the percentage of a whole carcass that they represent (breast fillet 33 %, thigh fillet 16 %, and drumstick 16 % of the whole bird by weight).

Pea-based product inventory

The pea-based product used either Australian or a mix of global processes, according to whether the product was manufactured in Australia, or manufactured and imported from abroad. Electricity and water consumption were country-specific.

Pea-based product processing

Processes from the AusLCI and ecoinvent databases (ALCAS, 2017; ecoinvent, 2020) were used to capture the impacts from the production of the pea, potato starch and canola oil ingredients. The quantity of each ingredient and electricity consumption was taken from Davis et al. (2010), a study of pea-based PBAs. Proxy values from the mycoprotein processing within Carbon Trust Quorn LCA (Hsu et al., 2018) were used for water consumption and land occupation. The number of packaging trays per kg of PBA product was calculated based on the assumption of 300 grams of product per tray. Table 1 summarises the inventory.

Table 18. Inventory per kilogram of pea-based product

	Material/process	Unit	Amount	Reference
Input	Peas, dried	kg	0.440	Davis et al. (2010)
	Potato, starch	kg	0.016	Davis et al. (2010)
	Canola oil	kg	0.090	Davis et al. (2010)
	Water, ingredient	kg	0.454	Davis et al. (2010)
	Water, processing	litres	96.000	Hsu et al. (2018)
	Electricity, processing	MJ	2.950	Davis et al. (2010)
	Electricity, liquid nitrogen production	MJ	4.500	Davis et al. (2010)
	Land occupation, industrial	m²a	0.035	Hsu et al. (2018)
	Packaging	trays	3.333	
Output	Pea-based product	kg	1.000	
	Water, evaporation	litres	96.000	

Pea-based product packaging

Inventory for packaging the pea-based product was taken from Heller and Keoleian (2018) with electricity values back-calculated using the fossil energy impacts from the paper (Table 19). Packaging also includes a period of cold storage before transportation to retail as per Heller and Keoleian (2018). The packaging is given per tray of packaged PBA product weighing 300 grams. The original packaging inventory was for two pea-based burgers (230 grams total). However, this was modified assuming the same quantity of packaging would be used for 300 grams of general pea-based product. It was assumed the thermoformed tray would no longer be required as the product would be contained within LDPE plastic wrapping.

Table 19. Inventory for the packaging of 300 grams of the pea-based product

	Material/process	Unit	Amount	Reference
Input	PBA product	g	300.0	
	Thermoformed tray	g	0.0	
	PE lid film	g	1.7	Heller and Keoleian (2018)
	Cardboard sleeve (incl. 12 % losses)	g	30.7	Heller and Keoleian (2018)
	Printing ink	g	0.3	Heller and Keoleian (2018)
	Patty paper	g	1.3	Heller and Keoleian (2018)
	Corrugated carton	g	29.9	Heller and Keoleian (2018)
	Wood pallet	no.	0.0008	Heller and Keoleian (2018)
	Pallet wrap	g	0.4	Heller and Keoleian (2018)
	Electricity, packaging	MJ	1.77	Calculated using Heller and Keoleian (2018)
	Electricity, cold storage	MJ	0.10	Calculated using Heller and Keoleian (2018)
Output	Packaged PBA product	g	360.6	
	Disposal, municipal waste	g	3.7	Heller and Keoleian (2018)

Quorn pieces inventory

To produce the QP product, mycoprotein is cultivated from molasses before being processed into the Quorn product. Inventory from LCAs of Quorn products by The Carbon Trust (Hsu et al., 2018) and Broekema et al. (2009) was used.

Mycoprotein production

Table 20 details the inventory for the production of a tonne of mycoprotein.

Table 20. Inventory for the production of a tonne of mycoprotein

	Material/process	Unit	Amount	Reference
Input	Electricity	kWh	5.0	Broekema et al. (2009)
	Fertiliser, nitrogen	kg	8.0	Broekema et al. (2009)
	Fertiliser, phosphorous	kg	8.0	Broekema et al. (2009)
	Fertiliser, potassium	kg	6.0	Broekema et al. (2009)
	Fertiliser, N_2O volatilisation	kg	0.3	Broekema et al. (2009)
	Diesel	kg	4.0	Broekema et al. (2009)
	Water, consumptive	litres	35,000	Hsu et al. (2018)
	Land occupation, industrial	m²a	90	Hsu et al. (2018)
Output	Mycoprotein	kg	1,000	

Mycoprotein processing

Table 21 details the inventory for the processing of a tonne of mycoprotein into Quorn pieces.

Table 21. Inventory for the processing of one tonne of Quorn pieces

	Material/process	Unit	Amount	Reference
Input	Mycoprotein	kg	1,000.0	
	Product splitting, electricity	kWh	172.0	Broekema et al. (2009)
	Product splitting, steam	kg	528.0	Broekema et al. (2009)
	Product splitting, diesel	kg	68.0	Broekema et al. (2009)
	Product processing, electricity	kWh	217.0	Broekema et al. (2009)
	Product processing, steam	kg	4,490.0	Broekema et al. (2009)
	Product packaging	kg	154.0	Broekema et al. (2009)
	Water, consumptive	litres	12,000.0	Hsu et al. (2018)
	Land occupation, industrial	m²a	35.0	Hsu et al. (2018)
Output	Quorn pieces	kg	1,000.0	

Mycoprotein packaging

Packaging for the mycoprotein used the inventory in Table 22. The processing inventory in Table 21 above included the electricity consumed in packaging. The quantity of polyethylene, polypropylene and cardboard used was taken from Broekema et al. (2009), and for additional completeness, inventory was used from Heller and Keoleian (2018) including cold storage.

Table 22. Inventory for the packaging of one tonne of mycoprotein pieces

	Material/process	Unit	Amount	Reference
Input	PBA product	kg	1,000.0	
	Polypropylene	kg	97.0	Broekema et al. (2009)
	Polyethylene	kg	6.0	Broekema et al. (2009)
	Cardboard	kg	51.0	Broekema et al. (2009)
	Printing ink	kg	1.1	Calculated using Heller and Keoleian (2018)
	Wood pallet	no.	2.7	Calculated using Heller and Keoleian (2018)
	Pallet wrap	kg	1.2	Calculated using Heller and Keoleian (2018)
	Electricity, cold storage	MJ	341.8	Calculated using Heller and Keoleian (2018)
Output	Packaged PBA product	kg	1,156.3	

Laboratory meat inventory

Laboratory cultured meat can be grown from a variety of inputs. However, inventory from the most frequently referenced LCA study (Tuomisto & Roy, 2012) was used, which was based on the cultivation of cyanobacteria; a photosynthetic family of bacteria.

Cyanobacteria cultivation

Inventory for cyanobacteria cultivation was taken directly from Tuomisto and Roy (2012) with electricity back-calculated using the fossil energy impacts from the paper (Table 23). To calculate electricity, it was assumed that all fossil energy impacts were a result of electricity consumption. It was assumed that all water used was consumptive.

Table 23. Inventory for the cultivation of one kilogram of cyanobacteria

	Material/process	Unit	Amount	Reference
Input	Electricity, high voltage	MJ	5.7441	Calculated using Tuomisto and Roy (2012)
	Water	litres	449	Tuomisto and Roy (2012)
	Land occupation, industrial	m²a	0.232	Tuomisto and Roy (2012)
Output	Cyanobacteria	kg	1.0	
	Water, consumption	litres	449	

Cyanobacteria processing

The cyanobacteria are transported to a processing plant where it is sterilised in a processing step (Table 24) as per Tuomisto and Roy (2012), with electricity back-calculated using fossil energy impacts from the paper (Table 23). To calculate electricity, it was assumed that all fossil energy impacts were a result of electricity consumption and diesel used during transportation. Energy values for transportation were converted to litres of diesel consumed by using the energy density of diesel. It was assumed that all water used was consumptive.

Table 24. Inventory for the processing (sterilisation) of one kilogram of cyanobacteria

	Material/process	Unit	Amount	Reference
Input	Cyanobacteria	kg	1.0	Tuomisto and Roy (2012)
	Electricity, high voltage	MJ	2.1	Calculated using Tuomisto and Roy (2012)
	Diesel, transport	litres	0.01	Calculated using Tuomisto and Roy (2012)
	Water, indirect	litres	8.0	Tuomisto and Roy (2012)
	Water, direct	litres	7.2	Tuomisto and Roy (2012)
Output	Sterilised cyanobacteria	kg	1.0	
	Water, consumption	litres	15.2	

Muscle production

Once sterilised, the cyanobacteria are cultured to create muscle. As with the other steps, inventory for was taken directly from Tuomisto and Roy (2012), with electricity back-calculated using the fossil energy impacts from the paper (Table 25). To calculate electricity, it was assumed that all fossil energy impacts were a result of electricity consumption. It was assumed that all water used was consumptive.

Table 25. Inventory for the production of one kilogram of cultured muscle

	Material/process	Unit	Amount	Reference
Input	Sterilised cyanobacteria	kg	1.0	Tuomisto and Roy (2012)
	Electricity, high voltage	MJ	15.0	Calculated using Tuomisto and Roy (2012)
	Water, indirect	litres	56.1	Tuomisto and Roy (2012)
Output	Cultured muscle	kg	1.0	
	Water, consumption	litres	56.1	

Laboratory meat packaging

The packaging inventory used (Table 26) was a combination of Wiedemann et al. (2015) and Heller and Keoleian (2018). For consistency with the other products, packaging also included impacts from cold storage. As less packaging is used for this product and the chicken meat, it was assumed that packaging electricity was half that used for the pea-based product.

Table 26. Inventory for the packaging of one tonne of product

	Material/process	Unit	Amount	Reference
Input	Product	kg	1000.0	
	LDPE	kg	3.9	Wiedemann et al. (2015)
	Corrugated board	kg	28.2	Wiedemann et al. (2015)
	Printing ink	kg	1.1	Heller and Keoleian (2018)
	Wood pallet	no.	2.67	Heller and Keoleian (2018)
	Pallet wrap	kg	1.2	Heller and Keoleian (2018)
	Electricity, packaging	kWh	300	Calculated using Heller and Keoleian (2018)
	Electricity, cold storage	MJ	341.8	Calculated using Heller and Keoleian (2018)
Output	Packaged product	kg	1034.4	

Chicken meat inventory

Chicken production and processing

Australian average chicken meat production inventory was taken from Wiedemann et al. (2017) for comparison with the PBAs. Table 27 shows the inventory for the grow-out phase of chicken production per tonne of liveweight produced. Australian average production was based on a 50:50 split between Queensland and South Australia production.

Table 27. Inventory for the grow-out phase of chicken production per tonne of liveweight

Materials	Queensland	South Australia
Feed ration, kg as-fed	1,886.0	1,853.0
Day-old chicks	423.4	402.5
Electricity, kWh	99.8	96.2
LPG, L	13.2	26.5
Natural gas, m³	25.7	n.a.
Diesel, L	0.4	1.6
Petrol, L	0.6	0.5
Staff transport, km	9.6	3.7
Freshwater consumption – animal houses, L	3,206.0	5,890.0
Freshwater consumption – water supply system losses, L	83.2	n.a.
Bedding – shavings/straw, kg	127.0	162.0
Pesticides, L	0.1	0.1
Disinfectant, L	0.8	0.5

Chicken processing

Chicken meat processing inventory was taken from Wiedemann et al. (2017). The meat processing inventory per tonne of carcase weight processed is shown in Table 28.

Table 28. Inventory for meat processing per tonne of carcase weight processed

Inputs	Queensland	South Australia
Electricity, kWh	239.3	155.0
LPG, L	3.1	7.7
Natural gas, m³	6.6	6.6
Diesel, L	2.5	0.2
Petrol, L	0.3	0.0
Total water supply, L	5,804.5	6,846.7
Freshwater consumption, L	2,472.1	1,505.8
Cleaning chemicals, L	7.3	3.4

Chicken meat packaging

Inventory from Table 26 was used.

Retail inventory

The retail inventory is shown in Table 29. The quantity of product retailed was calculated using a serve of chicken equivalent product consumed and the losses experienced through the retail and consumption inventories. Retail electricity was calculated by using an annual value of 1,500 kWh per m^2 to retail chilled products in a supermarket (Tassou et al., 2011). It was assumed that 100 serves of the product are displayed per m², and each product is kept on the shelf for one week on average. Using the same logic, the annual retail land occupation was calculated. Retail losses for meat products as per (WRAP, 2020) were used for all products and assumed to go to municipal waste. In reality, it is expected that losses for PBAs could be substantially greater as they are less popular in Australia. Transport distances from processor to the point of retail were calculated for both domestically produced and imported products. Sydney was assumed to be port of transit in Australia, Auckland in New Zealand, Felixstowe in the UK, Amsterdam (Netherlands) in the EU and Los Angeles in the US. Transport of all products was assumed to be frozen.

Table 29. Inventory for the retail and transportation of the PBAs and chicken meat per 100 gram chicken meat equivalent serving size (including losses)

	Material/process	Unit	Pea	Quorn	Lab	Chicken	Reference
Input	Product retailed	kg	0.125	0.127	0.138	0.138	Nutritional study
	Retail electricity	kWh/kg	0.288	0.288	0.288	0.288	Calculated value
	Retail losses	%	10.0 %	10.0 %	10.0 %	10.0 %	WRAP 2020
	Retail land occupation	m²/kg/yr	0.00019	0.00019	0.00019	0.00019	Calculated value
	Transport foreign warehouse to port	km	40	572	572	n.a.	Calculated value
	Transport port to port (NZ to Aus)	km	2361	n.a.	n.a.	n.a.	Calculated value
	Transport port to port (UK to NZ)	km	20,901	n.a.	n.a.	n.a.	Calculated value
	Transport port to port (UK to Aus)	km	n.a.	21,299	21,299	n.a.	Calculated value
	Transport port to port (EU to Aus)	km					Calculated value
	Transport port to port (US to Aus)	km					Calculated value
	Transport port to warehouse	km	110	110	110	n.a.	Calculated value
	Transport warehouse to retailer	km	200	200	200	200	Estimate
Output	Waste, municipal	kg	0.013	0.013	0.014	0.014	

Consumption inventory

The inventory for detailing consumer inputs up until the point of consumption is detailed in Table 30. The chicken equivalent serve size was taken from the nutritional study to determine the mass of the product consumed. The fraction of each product that is packaging was calculated using the packaging inventory in Table 19 and Table 22 and was used alongside losses to determine the quantity of product disposed to municipal waste. Fridge and plate losses of meat products from WRAP (2020) were used for all the products. The change in the moisture content of chicken before and after cooking was used to estimate cooking losses for all products (Meat Science, 2020).

The fridge electricity consumption was calculated by using a value of 2 kWh per day (Reduction Revolution, 2020) for a typical fridge/freezer with a kilogram of product assumed to be 1 % capacity of the appliance and was estimated to be stored for a week on average before consumption. In reality, a kilogram of any of the products could account for a higher % of capacity and could be stored for considerably longer, especially if frozen. The electricity used to cook the product was based on frying on a hob that uses 1.5 kWh per hour (Energy Use Calculator, 2020). It was assumed that each product was cooked for 15 minutes and 0.5 kg of the product was cooked at a time (approximately five serves).

The distance of the consumer to the closest supermarket was estimated at 5 km one way or 10 km for a return trip. Based on a single-serve of chicken or PBA costing approximately $1.50 and an average weekly supermarket shop of $100, the serve accounted for 1.5 % of items purchased. Consumer transport impacts were assigned accordingly. It was assumed that consumers accessed supermarkets by car.

Table 30. Inventory for the consumption of the PBAs and chicken meat by the consumer at home per 100 gram chicken meat equivalent serving size

	Material/process	Unit	Pea	Quorn	Lab	Chicken	Reference
Input	Product consumed	kg	0.083	0.092	0.100	0.100	Nutritional study
	Packaging fraction	%	16.8 %	3.7 %	3.7 %	3.7 %	Various
	Fridge & plate losses	%	16.0 %	16.0 %	16.0 %	16.0 %	WRAP (2020)
	Cooking losses	%	8.0 %	8.0 %	8.0 %	8.0 %	Meat Science (2017)
	Fridge electricity	kWh/kg/wk	0.14	0.14	0.14	0.14	Calculated value
	Cooking electricity	kWh/kg	0.75	0.75	0.75	0.75	Calculated value
	Transport consumer to retailer	km	10	10	10	10	Estimated value
	Purchase fraction	%	1.5 %	1.5 %	1.5 %	1.5 %	Calculated value
Output	Waste, municipal	kg	0.027	0.018	0.020	0.020	
	Water, evaporation	kg	0.007	0.007	0.008	0.008	

Limitations

The inventories used were limited, and the majority did not include mass losses of product, nor a transport component between production and processing. In the case of laboratory meat, it is anticipated that production and processing would occur on the same site, but this would not be true of the pea-based or mycoprotein PBAs. Some of the electricity consumption was back-calculated from impact results of the products, where the inventory was not reported. As such, this may over-estimate electricity consumption as fossil energy use can come from other products in the supply chain. Inventory detail, in general, was lacking for all the PBAs and care should be taken if completing a full LCA to ensure completeness and a fair representation of the impacts of each product. This has been achieved to as great an extent as possible with this scoping LCA. Still, the chicken meat product is the only one that could be considered as 'complete', and the impacts of the others are likely to be underestimated in comparison.

Mass loss through water content was considered. However, not enough information was known about each product, and a value for chicken breast was used for all products. It is also expected that there will be greater retail losses of PBAs than chicken as a result of the lower popularity and higher price of these products.

Land-use change impacts were calculated for chicken feed production and were included in the GHG impacts. However, land-use change was not included for the PBAs and could be particularly relevant for pea production depending on the region of production.

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