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PHD THESIS

Environmental and nutritional sustainability potential of
legume-based food production pathways

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DECLARATION

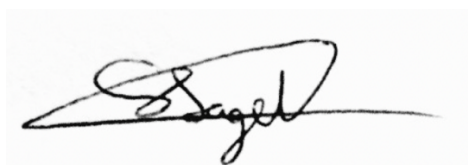
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Peas and love





The TRUE family – Peas up!

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TERMS AND DEFINITIONS

Allocation: Apportioning the environmental flows of a product system between that product system and other product systems.

Biophysical allocation: Apportioning the environmental flows of a product system between that product system and other product systems based on metabolic energy requirements to produce and maintain body tissues, (excluding tissues treated as waste).

Economic allocation: Apportioning the environmental flows of a product system between that product system and other product systems based on the relative economic value of the co-products.

Elementary flow: Material/energy that enters directly (if inputted) the studied system without having been adjusted by human action, or leaves directly (if outputted) the studied system with no further human modification. Therefore, these directly come from or are released to the environment.

Functional unit: Quantified performance of the system(s) being studied in the life cycle assessment, providing a basis for ensuring comparability of systems.

Life cycle assessment (LCA): Quantitative decision support tool for determining the environmental burdens associated with a product or service throughout its life cycle.

Life cycle impact assessment (LCIA): Phase in the Life Cycle Assessment in which the environmental burdens of the system studied are calculated and evaluated.

Life cycle interpretation: Phase in the Life Cycle Assessment in which the results of the life cycle impact assessment are interpreted and analysed in line with the goal and scope to reach conclusions.

Life cycle inventory (LCI): Phase in the Life Cycle Assessment in which the inputs and outputs of the system studied are collected, quantified, and recorded.

Mass allocation: Apportioning the environmental flows of a product system between that product system and other product systems based on the relative mass of the co-products.

Multifunctional process: Process that yields multiple output products that are not waste.

Normalisation: Optional step in a Life Cycle Assessment in which results are characterised against a common reference to obtain results against a single unit.

Process: Element in the system in which input and output flows are quantified.

Product flow: Material/energy that was transformed by human action and will be used further, and are thus exchanged between processes within the system.

Waste flow: Material/energy leaving the product system as waste.

NOMENCLATURE

aLCA	Attributional Life Cycle Assessment	-
BB (BR)	Beef burger made with Brazilian beef	-
BB (FR)	Beef burger made with French beef	-
BB (IE)	Beef burger made with Irish beef	-
BB (US)	Beef burger made with American beef	-
BB (ZA)	Beef burger made with South African beef	-
BG	Bulgaria	-
CA	Aquafaba-based mayonnaise from Canada	-
COC	Carbon Opportunity Cost	-
cLCA	Consequential Life Cycle Assessment	-
DV_{EFA}	Recommended daily value intake of essential fatty acids	grams
DV_{prot}	Recommended daily value intake of protein	grams
DV_{fib}	Recommended daily value intake of fibre	grams
EFA	Amount of essential fatty acids in 100 g of product	grams
EGG	Egg-based mayonnaise	-
EoL	End of Life	-
EP	Eutrophication potential	-
Fib	Amount of fibre in 100 g of product	grams
FU	Functional unit	-
GHG	Greenhouse gas	-
GWP	Global warming potential	-
HH	Human health	-
LCA	Life Cycle Assessment	-
LCIA	Life Cycle Impact Assessment	-
LU	Land Use	-
MB (IE)	Meatball made with Irish beef	-
MB (BR)	Meatball made with Brazilian beef	-
NDU	Nutrient Density Unit	-
PCR	Product category rules	-
PEF	Product environmental footprint	-
PPB	Pea protein ball	-
Prot	Amount of protein in 100 g of product	grams
RU	Resource use	-
S_i	Amount of kilocalories in 100 g of product	kcal
SP	Spain	-
VB	Vegetarian burger	-
VEG	Vegan mayonnaise	-

Delivering sustainable nutrition is a key challenge for the global food sector, including in Europe, where 50% of the population is overweight, 20% obese, and where food production and consumption accounts for 30% of the continent's greenhouse gas (GHG) emissions (Crippa et al., 2021), and is a main driver of biodiversity loss, nutrient pollution, and resource depletion (Willett et al., 2019). A substitution of locally-grown legumes for popular unhealthy products in European diets holds great potential in assisting sustainable food transformation. This PhD thesis assesses the potential delivery of nutrition at a comparatively low environmental cost of these innovative foods in a thesis-by-publication format. Comparative attributional Life Cycle Assessments (LCA) from cradle to fork or factory gate were performed between traditional foods and their legume-based alternatives. Products were compared across the sixteen impact categories recommended by the Product Environmental Footprint (PEF) guidelines (European Commission, 2018a), a breadth of environmental issues that are seldom assessed, and statistical significance assessed through Monte Carlo analyses and modified Null Hypothesis Significance Tests, a step that is rarely performed in LCA. Moreover, in addition to a mass-based functional unit (FU), a nutrient density unit (NDU) as proposed by Van Dooren (2016) was used as an additional FU for foods to capture positive nutrition in the comparison, an aspect that is often omitted in food LCAs. Primary data was collected from chickpea pasta, pea protein balls, pea-soy burger patties, and aquafaba-based (chickpea cooking water) vegan mayonnaise producers. We find that the environmental burdens of producing and consuming the legume-based foods were overall statistically significantly lower ($p < 0.05$) than those of the conventional foods across most categories, including a 41% average carbon footprint reduction. This advantage is further extended when using the NDU as a FU, due to the legume-based foods being on average 66% more nutrient-dense. Moreover, if consumption habits undergo minor traditional-legume substitutions, the carbon storage potential from resulting spared land represents a significant climate change mitigation opportunity. This thesis therefore highlights the potential of innovative legume-based products to deliver sustainable nutrition. Further research should focus on the environmental and nutritional effects of integrating these products at a diet level in Europe through a consequential LCA showing the effects of substituting these for their traditional alternatives, integrating full crop rotation cycles.

LAY SUMMARY

Our relationship with food – the way we produce, prepare, and consume it – has large negative consequences on the environment and our health. It accounts for more than a quarter of global greenhouse gas emissions and around 80% of nutrient pollution in water, utilises more than half of habitable land and nearly 75% of available freshwater, and reduces biodiversity critically. Our current diet, which is high in foods with unhealthy nutrients and low in foods with positive nutrients, is more hazardous than unsafe sex, alcohol, drug, and tobacco altogether. Legumes, which are vegetables from the *Fabaceae* family – such as peas, chickpeas, lentils, and faba beans – are highly nutritious foods that are at the same time more eco-friendly than most other foods. Their key characteristic is their ability to fix atmospheric nitrogen thanks to a partnership with soil bacteria, thereby reducing the need for synthetic fertiliser, which causes high environmental harm. Substituting legumes for popular unhealthy products in European diets holds great potential in assisting sustainable food transformation. Minimally processed legumes are however not palatable to many, hence consumers increasingly turn towards highly processed legume-based products that mimic a popular food. Limited research assesses simultaneously the environmental and nutritional effects of substituting these innovative products for their popular alternatives. Life Cycle Assessment (LCA) is a method used to calculate the environmental impact of a product's life cycle, from the extraction of raw materials to its consumption or disposal. To perform a LCA, all inputs and outputs (also called flows) of a product life cycle are connected to environmental databases, which associate factors of environmental harm to each flow. This thesis investigates the environmental performances of legume-based foods and their respective popular alternatives, namely pasta made from chickpea or wheat, protein balls from peas or beef, burger patties from soy and peas or beef, and mayonnaise from aquafaba (chickpea cooking water) or egg yolk. I compare the environmental performance of the legume and non-legume products across 16 environmental issues, including carbon footprint, water scarcity, and nutrient pollution. I integrate nutrition in the comparison where possible, to identify foods that perform better across both nutrition and environment. I find that the environmental harm of producing and consuming the studied legume-based foods are overall significantly lower than those of the conventional foods across most environmental issues, including a 41% average carbon footprint reduction. This advantage further extends when incorporating nutrition to the equation, due to the legume-based foods being on average 66% more nutritious. Moreover, if consumption habits undergo minor substitutions of animal-based foods with the legume products, large amounts of carbon can be

stored in the resulting freed land. This thesis therefore highlights the potential of innovative legume-based products to deliver sustainable nutrition. Further research should focus on the environmental and nutritional effects of integrating these products in whole diets in Europe, looking at the consequences of a diet change in the market over several years, so that the entire effects of legumes introduction in cropping sequences are taken into account, as their nitrogen fixation also benefits the following crops.

1. INTRODUCTION

1.1. TRUE PROJECT

This PhD is part of the Horizon 2020 “TRAnsition paths to sUustainable legume based systems in Europe” (TRUE) research and innovation programme (GFE, 2020). The objectives of the TRUE project are to recognise the optimal transition pathways to increase sustainable legume cultivation and consumption across Europe. The PhD is part of the Work Package 5 “Environmental LCA and Nutrient Quality Assessment” that aims to answer the following questions:

- What is the environmental footprint of animal feed & food produced from legumes, considering nutrient cycling and break-crop effects in legume-rotations across major EU agro-climatic zones?
- What are the optimum legume-enriched diets/food choices for improving health, that decrease the environmental footprint – including indirect effects incurred during supply chain transitions - and reduce direct costs to the consumer?

1.2. SUMMARY

Delivering sustainable nutrition is a key challenge for the global food sector. Large anthropogenic pressures on Earth ecosystems result from the intensive agricultural practices implemented to meet the rising global food demand (Steffen et al., 2015). These damage the environment, causing land degradation, freshwater depletion, and biodiversity loss, among others. The food sector is estimated to be responsible for up to 30% of global greenhouse gas (GHG) emissions (Vermeulen et al., 2012), and 70% of freshwater use (Steffen et al., 2015), and is predicted to increase by 50-90% between 2010 and 2050 if no further measures are taken (Springmann et al., 2018). At the same time, health deterioration due to poor diet quality is driven by increased consumption of animal protein and energy-dense, refined foods that are high in added fats and sugars (Kendall et al., 2010; Willett et al., 2019). In Europe, this is linked to more than 50% of the population being overweight and more than 20% obese (WHO, n.d.), and to numerous non-communicable diseases, such as type 2 diabetes and cardiovascular diseases (Aune et al., 2009; Hu, 2011; Huang et al., 2012). Unhealthy diets pose a great threat to human health, hence global food production should aim to produce healthier foods from sustainable agricultural systems (Willett et al., 2019).

Legumes, from the *Fabaceae* family, are key crops for achieving environmental and health improvements. The FAO termed them as “nutritious seeds for a sustainable future” (FAO, 2016a)

due to their numerous environmental and nutritional benefits. Despite these benefits, however, legumes are only grown on 2.1% of Europe's arable land, versus 54% for cereals (European Commission, 2020a). Their consumption represents only 1% of the daily energy intake (FAO, 2019a), and is far from meeting the 17% recommended energy intake of the sustainable diet devised by the EAT Lancet report (Willett et al., 2019).

Innovative legume-based foods that aim to substitute popular food products are increasingly present on the market. At first sight, they appear to require less resources and therefore to be more environmentally-friendly, and healthier, due to the absence of meat. However, robust scientific studies showing their relative impact on the environment and on health are scarce, as well as their relative role in Europe to achieve sustainable food systems. Moreover, existing studies do not often compare these foods from a nutritional standpoint, identifying solely environmental sustainable products. Considering the economic and health impacts of unhealthy diets in Europe, it is an imperative to identify foods that are both environmentally sustainable and nutritious.

This thesis aims to address the following:

- **Benchmarking of the environmental sustainability performance of nutrient delivery of novel legume-based products against their conventional counterparts.** Using an attributional life cycle assessment (LCA) approach, chickpea pasta, pea protein balls, and a pea-soy meatless burger are compared with wheat pasta, beef meatballs, and a beef burger, so that foods that provide the highest nutrient density at the lowest environmental cost are identified. An additional legume-based product is analysed –aquafaba mayonnaise – and is compared with traditional egg-based mayonnaise. However, nutrition is not included in the analysis of these two last products, due to their nutritional values being negligible.

Six chapters follow this introductory chapter, and the objectives of each are described below.

- Chapter 2: to present the full methodology of Life Cycle Assessment used across this thesis: its steps, the environmental issues covered, and its limitations.
- Chapter 3 “Substituting wheat with chickpea flour in pasta production delivers more nutrition at a lower environmental cost”: to compare the environmental performance of chickpea pasta versus durum wheat pasta using a serving and nutrient density functional units.
- Chapter 4 “Substitution of beef with pea protein reduces the environmental footprint of meat balls whilst supporting health and climate stabilisation goals”: to compare the

environmental performance of pea protein balls versus beef meatballs using a serving and nutrient density functional units.

- Chapter 5 “Comparative Life Cycle Assessment of plant and beef-based patties, including carbon opportunity costs”: to compare the environmental performance of a different meat analogue than the previous chapter, this time with many more ingredients in the vegetarian option, involving extensive processing.
- Chapter 6 “Does circular reuse of chickpea cooking water to produce vegan mayonnaise reduce environmental impact compared with egg mayonnaise?”: to compare the environmental impact of producing chickpea cooking water-based mayonnaise with conventional egg-based mayonnaise.
- Chapter 7 presents an overall discussion where all the case studies of this thesis are brought together. The aim of this chapter is to provide an overview of the potential of legumes to offer a more sustainable pathway to achieve high nutrient density in Europe from an environmental perspective.

1.3. FOOD CHALLENGES IN EUROPE: ENVIRONMENTAL AND NUTRITIONAL SUSTAINABILITY

The European food sector currently faces a key challenge of providing nutritious foods that have a minimal impact on the environment, as part of a sustainable diet. The concept of “sustainable diet” was first termed in 1986 (Gussow and Clancy, 1986), and represents a diet that has a low impact on the environment, respecting ecosystems biodiversity, while being healthy, culturally adequate, accessible, and affordable (FAO, 2010). This diet comprises of a variety of plant-based foods (nuts, fruits, vegetables, legumes), few animal-based foods, unsaturated instead of saturated fats, and little or no refined grains, highly processed foods, and added sugar (Willett et al., 2019). As shown in Figure 1.1, the mean European diet, however, does not comply with at least the requirements of health and environment, as it contains significantly more red meat, sweeteners, starchy vegetables, dairy, and less legumes and nuts (Williams et al., 2020b) than the EAT reference diet from Willett et al. (2019). Substituting part of animal protein with plant protein in diets is recommended for health and environmental purposes (Birt et al., 2017; Harwatt, 2019).

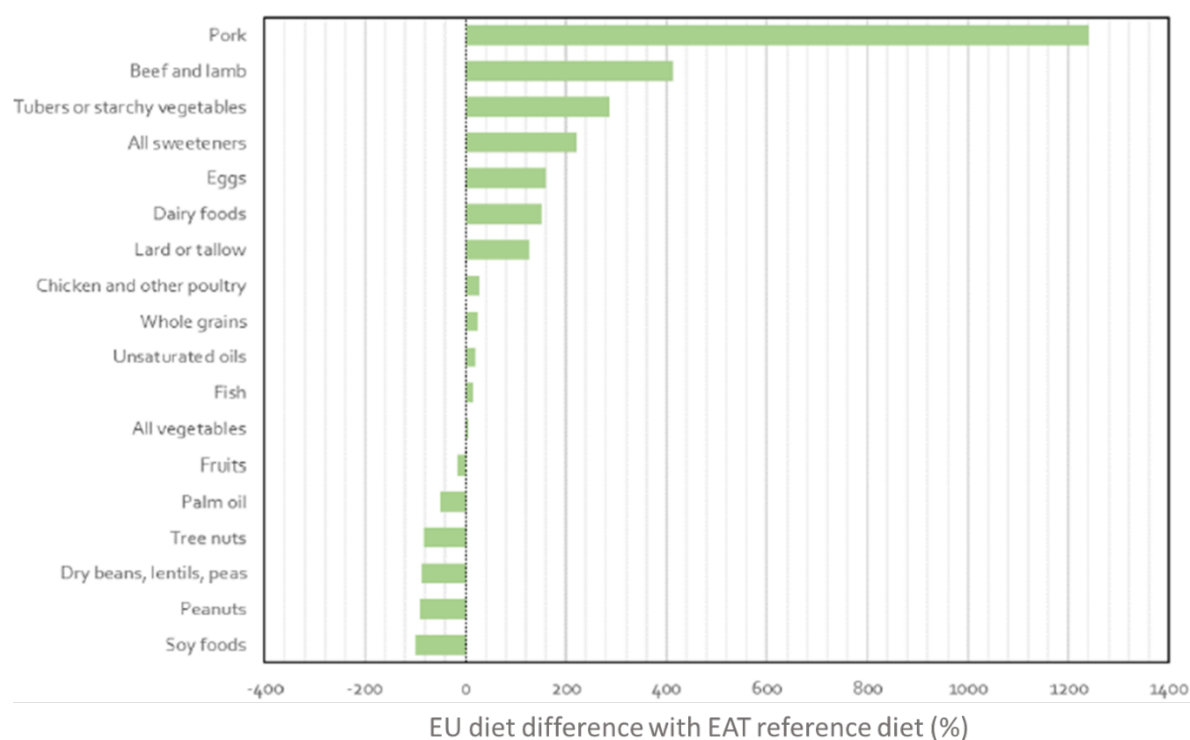


Figure 1.1: Dietary intake of mean European diet from 2009-2013 versus the EAT Lancet reference diet. Reproduced from Williams et al. (2020).

1.3.1. Nutritional and health challenges

The European population is facing major health issues, due to a poor diet quality, with increasingly high consumption of animal protein and of energy-dense, nutrient-poor refined foods (Global Panel on Agriculture and Food Systems for Nutrition, 2016; Kendall et al., 2010). As a result, more than 50% of the European population is overweight and more than 20% is obese (WHO, n.d.). These issues are associated with non-communicable diseases (NCDs), such as cardiovascular diseases, diabetes, cancer, and chronic respiratory diseases (Aune et al., 2009; Hu, 2011; Huang et al., 2012). Unhealthy diets are more risky in terms of morbidity and mortality than is the combination of unsafe sex, alcohol, drug and tobacco consumption (Global Panel on Agriculture and Food Systems for Nutrition, 2016).

This increased demand for animal protein and decreased demand for plant protein is associated with growing wealth within a population and intensification of food systems (Tilman and Clark, 2014). It is widely accepted that overconsumption of animal protein, especially red and processed meat, impacts negatively human health (Godfray et al., 2018; Rust et al., 2020). Numerous chronic diseases are associated with excessive consumption of these. Typical diseases include type 2 and gestational diabetes, coronary heart disease, heart failure, stroke, colorectal

cancer, and ultimately a higher death rate, as statistically significant results were shown by numerous studies (Larsson and Wolk, 2006; Norat et al., 2005; Pan et al., 2012; van Dooren et al., 2014; Wolk, 2017). How meat is processed and cooked can also be detrimental to health, with the addition of nitrate and nitrite compounds and the formation of carcinogenic chemicals (Bouvard et al., 2015; Etemadi et al., 2017). Intake of processed meat is 90% higher than advised (Afshin et al., 2019). Recently, numerous news articles were citing a recent review that claimed that red meat consumption had very small detrimental health effects, and that these effects were anyways marked by a very low quality of evidence (Johnston et al., 2019). However, the authors received numerous criticisms about their methodological choices, which included the use of grading criteria for examining clinical drug trials, which have very different behaviours than effects of dietary choices, and used studies that were not directly evaluating the effects of reducing meat consumption (Harvard T.H. Chan School of Public Health, 2019). Aside from this, the global antimicrobial resistance crisis is exacerbated by the overuse of antibiotics in the meat industry (Spellberg et al., 2016), and meat adulteration is common (Ali et al., 2015; Rahman et al., 2014).

Besides the large issue of overconsumption of red meat, the high popularity of energy-dense but nutrient-poor foods, with refined fats and sugars, alcohols and oils, increase the occurrence of obesity and NCDs (Tilman and Clark, 2014). These foods are very easily accessible, affordable, easily prepared, have a large portion size, and are marketed to children and adults, who succumb to convenience in today's fast pace environment (Rolls, 2003; Wright and Aronne, 2012). As shown in Figure 1.2, the availability of ultra-processed foods in households is correlated to the prevalence of obesity at the national level. For example, 50% of the purchased dietary energy available was from ultra-processed foods in British households, and 24.5% of the British population is obese (Monteiro et al., 2018).

Consumers of nutrient-poor foods ultimately need to eat other nutritious foods in order to maintain their health, otherwise causing economically-expensive medical costs, lost productivity, and premature death burdens to their community (Frazão, 1999; Popkin et al., 2006; Scarborough et al., 2011).

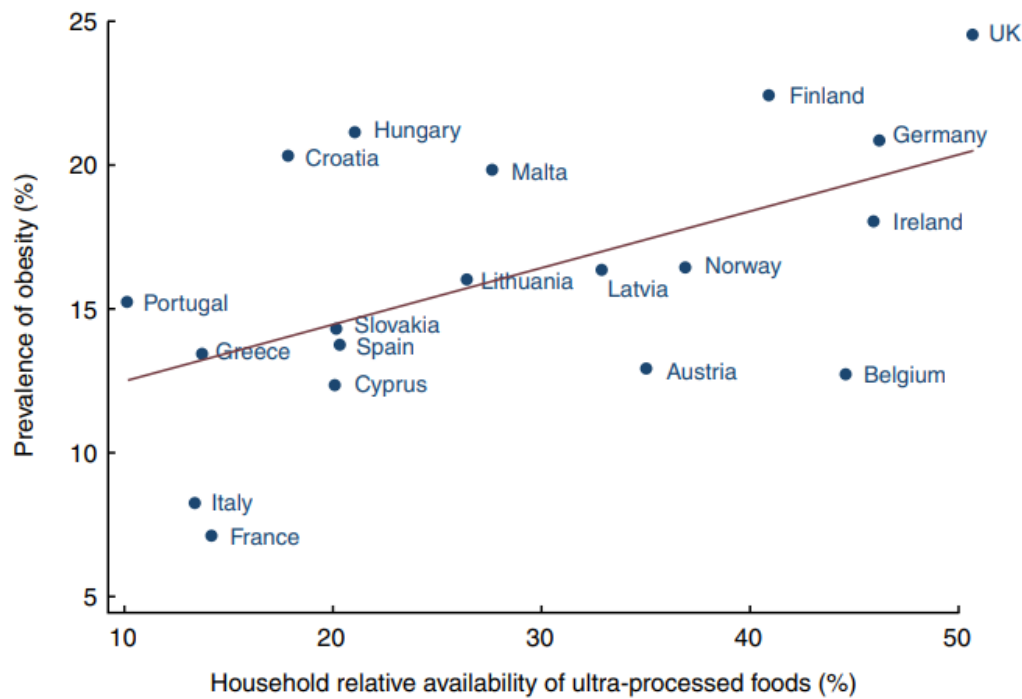


Figure 1.2: Correlation of the purchased dietary energy in households with the national obesity prevalence. Figure from Monteiro et al. (2018).

Besides the direct effects of consumption of these foods, indirect effects on health occur from agricultural production, due to environmental degradation from food systems. Low-quality air from agricultural and land clearing biomass burning results in premature deaths (Kopplitz et al., 2016). Additionally, yield and nutritional content of major crops are forecasted to decrease significantly as an effect of climate change, contributing further to food insecurity (Challinor et al., 2014).

1.3.2. Environmental challenges

In addition to the high burdens on health and their economic consequences, the increasingly large demand for animal products is associated with amplified environmental impacts of the food system, due to the comparatively high environmental burdens of animals (Poore and Nemecek, 2018). The livestock sector is omnipresent in Europe. It occupies around two thirds of the agricultural area, and around 75% of protein-rich feed is imported, mostly from Brazil and Argentina, with associated negative consequences on deforestation and biodiversity loss (Westhoek et al., 2011). Calories necessary for livestock production represent 40% of grains produced worldwide, for which the production is associated with non-negligible environmental impacts (Foley et al., 2011). Beef production involves substantial greenhouse gas (GHG) emissions

to the atmosphere – enteric fermentation releases methane during feed digestion and manure management, nitrous oxide is released from manure management and fertilisation, and carbon dioxide is released from manufacture of fertilisers, combustion of fuels, and feed production. High use of limited resources is also an issue, with cattle rearing consuming large amounts of land, water, and energy (Steinfeld et al., 2006). The calories obtained from cattle meat obtained are, moreover, considerably lower than the calories fed to the cattle (Shepon et al., 2016).

Aside from animal rearing-related environmental issues, the intensification of agricultural crop cultivation drives many negative effects on soil and air quality, biodiversity, climate, and energy consumption. Since the invention of the Haber-Bosch process in 1908, which produces ammonia from atmospheric nitrogen and hydrogen, the use of relatively cheap synthetic nitrogen fertilisers (SNFs) amplified greatly, and is widely used worldwide (Erisman et al., 2008). SNF production, however, is a highly resource-intensive process, and its over-application contribute to the imbalance of biogeochemical flows, due to N leaching and ammonia emissions to air, degrading the quality of air, water, and soil (Sutton et al., 2011), changing the climate and promoting biodiversity loss (Mozumder and Berrens, 2007).

In addition to this, land use for cattle rearing and its feed production represent a substantial Carbon Opportunity Cost (COC) (Hayek et al., 2020). COC represents the quantified difference between the carbon dioxide amount sequestered by a land, and the potential carbon sequestration that would be achieved by restoring the native ecosystem on that specific portion of land. The relative COC of animal products is significantly higher than that of plant-based foods, due to the lower macronutrient conversion rate from feed to food, and it is estimated that going from the current, unsustainable diets to plant-based diets would sequester globally between 332 and 547 Gt CO₂ globally (Hayek et al., 2020).

In light of these data and the breadth of research in this topic, the experts call for a “Great Food Transformation” (Willett et al., 2019).

1.4. THE ROLE OF LEGUMES IN ENVIRONMENTAL SUSTAINABILITY AND HEALTH

1.4.1. Legumes and the environment

Legumes (*Fabaceae*) are the third biggest family of flowering plants, with 800 genera and 20 000 species (Lock and MN, 2005). By virtue of their numerous agro-ecological benefits, they are key in achieving sustainable intensification of agricultural systems. Legumes release up to 7 times less GHGs than other crops (Stagnari et al., 2017). The need for SNF is greatly reduced when

legumes are introduced in a cropping system, thanks to their symbiotic relationship with N₂-fixing bacteria in their roots. SNF is costly and contributes to the imbalance of biogeochemical flows, due to N leaching from its application. Using legumes instead of SNF saves around 277 kg ha⁻¹ CO₂ per year (Stagnari et al., 2017). Improved biodiversity and weed control are achieved through legumes in rotations, intercropping, green manures, enriched pastures (Sturludóttir et al., 2014). The use of legumes also reduces incidence of diseases (Foyer et al., 2016), lessens GHG emissions, increases yields (Peoples et al., 2009), improves Nitrogen Use Efficiency (NUE) when compared with the use of SNFs and increases soil carbon content (Peoples et al., 2019, 2017). Some studies indicated that legume cultivation may increase leaching (Nemecek et al., 2008), however this was shown not to be a significant trade-off when considering the superior nutritional output of rotations in which legumes were introduced (Costa et al., accepted).

1.4.2. Legumes, nutrition and health

In addition to their environmental benefits, legumes are highly nutritious foods. Numerous macro- and micronutrients are found in legumes, including fibre, protein, low glycaemic carbohydrates, phosphorus, magnesium, B vitamins, iron, copper, manganese, and zinc (Polak et al., 2015). In addition of legumes being cholesterol-free, fat and saturated fat are scarcely present (Polak et al., 2015). Therefore, if legumes were partially replacing livestock products that are high in fats, the overconsumption of saturated fatty acids could be drastically reduced, being 30% higher than the maximum recommended value (Westhoek et al., 2011). The occurrence of NCDs is reduced with the consumption of legumes (Flight and Clifton, 2006; Jayathilake et al., 2018; Kouris-Blazos and Belski, 2016; Vigiouliou et al., 2017). Myocardial infarction incidence is also reduced with legume consumption (Miller et al., 2017). Nutrients like fibre, potassium, magnesium, and phytochemicals in legumes lower cholesterol levels and assist in blood pressure control (Bazzano et al., 2011). Gut health improvement is also a key effect of legume intake (Clemente and Olias, 2017). Fibre and other nutrients, such as protein and slow-digesting carbohydrates in legumes, help manage weight (McCrory et al., 2010). The presence of anti-cancer peptides (Luna-Vital and González de Mejía, 2018) possibly assists in the prevention of prostate (Park et al., 2008) and colorectal cancers (Zhu et al., 2015).

The term “poor people’s meat” is frequently used for legumes, due to their low price when compared to foods from animal origin and their high protein content. Their comparatively lower protein quality when aside of meat was often pinpointed by researchers in the past, and is still by the wider community (Katz et al., 2010). Although lower methionine contents in legumes doesn’t make their protein of highest quality, the consumption of cereals alongside compensates for this

imbalance, thanks to complementary amino acids (Elango et al., 2009). Furthermore, highest protein quality in foods provides cells with the capacity to grow and replicate excessively fast in adults, and this phenomenon is linked with increased risk of cancer (Tomasetti et al., 2017; Willett et al., 2019). Besides, European diets are characterised by a 70% higher protein intake per capita than the recommended intake, and therefore lower protein quality intake should not be of concern for adults in this geographical area (European Commission, 2020b; Westhoek et al., 2011).

1.4.3. Role of legumes in Europe

Despite the potential of legumes as key elements in a diet that contributes to environmental and nutritional sustainability of food systems, their production and consumption in Europe is limited. While 54% of Europe's arable land contains cereals, legumes cover only 2.1% (European Commission, 2020a). Low production of legumes in Europe is partly due to a technological lock-in resulting from unstable grain legume prices, variable yields, a government price support policy for cereals (Byerlee and White, 2000). Another major reason for low production is low consumer demand for these crops. Only 1% of an average European's daily energy intake is from legumes (FAO, 2019a), in contrast with the 17% recommended energy intake of the environmentally and nutritionally sustainable diet devised by the EAT Lancet report (Willett et al., 2019). Reasons for this low consumption include comparatively longer cooking times required, flatulence, and perceptions that legumes are old-fashioned (Magrini et al., 2016; Niva et al., 2017). A fundamental challenge for the food industry is therefore to produce legume-based foods that appeal to consumers' senses, while being healthy and environmentally sustainable.

Innovative legume-based food products that present similar flavour and texture than traditional meat and other foods are potentially the key to achieve a sustainable diet transition with minimal consumer effort. Global popularity of these innovative foods is growing. For example, the meat substitute market is forecasted to be multiplied by 10 fold between 2019 and 2029, reaching USD 140 billion (Statista, 2019a). Vegan and vegetarian foods have increased by 140% and 21% globally, respectively (Statista, 2017). These foods are increasingly popular with the rise of flexitarian diets (Derbyshire, 2017; European Commission, 2018b; NatCen, 2016). Flexitarians are individuals who eat mainly a vegetarian diet, with an occasional intake of meat and fish (Derbyshire, 2017).

1.5. LIFE CYCLE ASSESSMENT AS A SUSTAINABILITY TOOL FOR LEGUME-BASED FOODS

1.5.1. Life Cycle Assessment principles

Life Cycle Assessment (LCA) is a popular quantitative decision support tool for determining the environmental burdens associated with a product or service life cycle, from the extraction of resources to production, use, disposal, and end of life (ISO, 2006a). It relies on the connection of environmental flows of a defined system, and their characterisation with environmental impacts. The LCA methodology is explained fully in Chapter 2 of this thesis, and presented briefly below.

The methodology consists of four stages: (i) goal and scope definition, (ii) collecting the Life Cycle Inventory (LCI), (iii) performing the Life Cycle Impact Assessment (LCIA), and (iv) interpretation (ISO, 2006b). These stages are iterative, and the LCA practitioner can return to any stage and modify it to suit best the wished aims along the process. In (i) goal and scope definition, it is stated what the purpose of conducting the LCA is, whether it is to compare specific products/services, quantify the impacts of and improve a single product, or eco-labelling, amongst others. The functional unit (FU), which is the quantified performance of the system(s) being studied in the life cycle assessment, is defined in this step. From there, system boundaries are demarcated, and other modelling choices set up. In (ii), the inventory data is collected as environmental flows entering and leaving the system. In (iii), the environmental flows collected in (ii) are characterised for the impact assessment methods selected, providing as an output the environmental impact results in a single unit for each environmental issue assessed. Finally, in (iv), results are analysed, determining, in the case of a comparative LCA, which option is the most environmentally sustainable, whether there are any trade-offs, contribution of individual stages to the total burdens, and conclusions drawn.

1.5.2. Evaluating environmental and nutritional sustainability of foods with life cycle assessment

The use of LCA to assess the sustainability of food systems is booming (Figure 1.3). The concept of nutrition integration in the assessment is increasingly present. The choice of the FU depends on the question to be answered (Figure 1.4). Several approaches can be taken to assess both environmental and nutritional sustainability of foods in LCAs:

- 1) Comparing foods that are nutritionally equivalent

2) Using nutritional content as an additional impact category in the Life Cycle Impact Assessment (LCIA) stage

3) Using a FU that is based on the nutritional content of food.

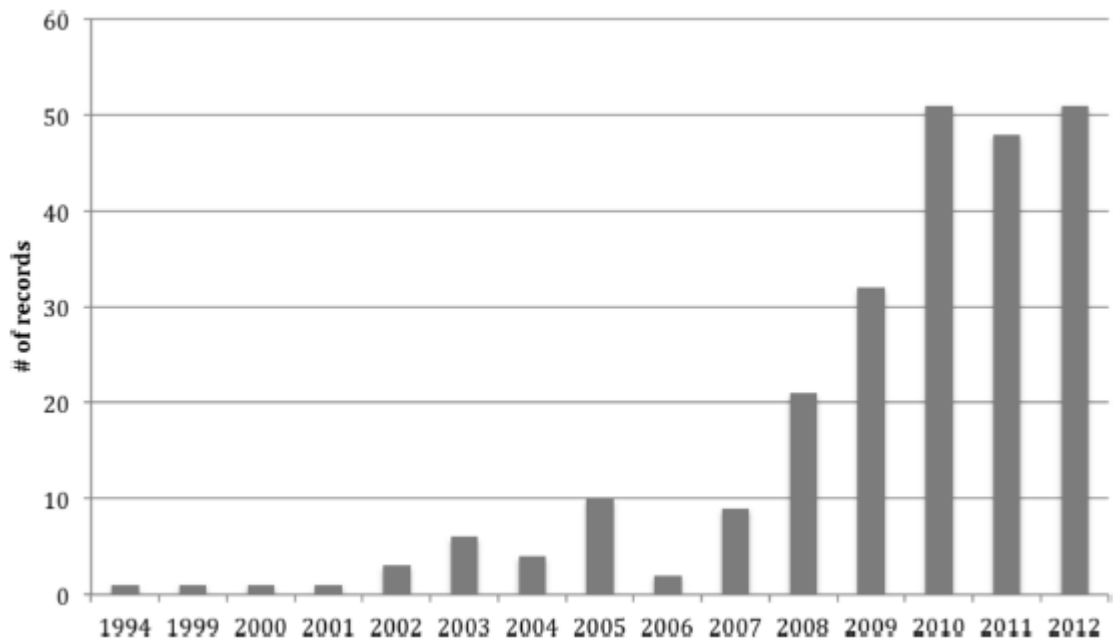


Figure 1.3: Histogram of the number of articles on the topic of “food” and “life cycle assessment” from a search on Web of Science from 1994 to 2012 (Figure from Heller, Keoleian, Willett, 2013).

The first option, comparing foods that are nutritionally equivalent is often not possible when comparing alternatives that are not made with the same main ingredient, such as a plant-based burger and a beef burger. The second option, using nutritional content as an additional impact category in LCIA, unless weighed heavily, does not place nutrition as a key aspect of food. The third option, using a FU that is based on the nutritional content of food is a key approach to award environmental credits to foods that are more nutritious than their less nutritious alternatives. The following paragraphs describe the various FUs used in food LCAs, including the FU used in the LCIA presented in this thesis.

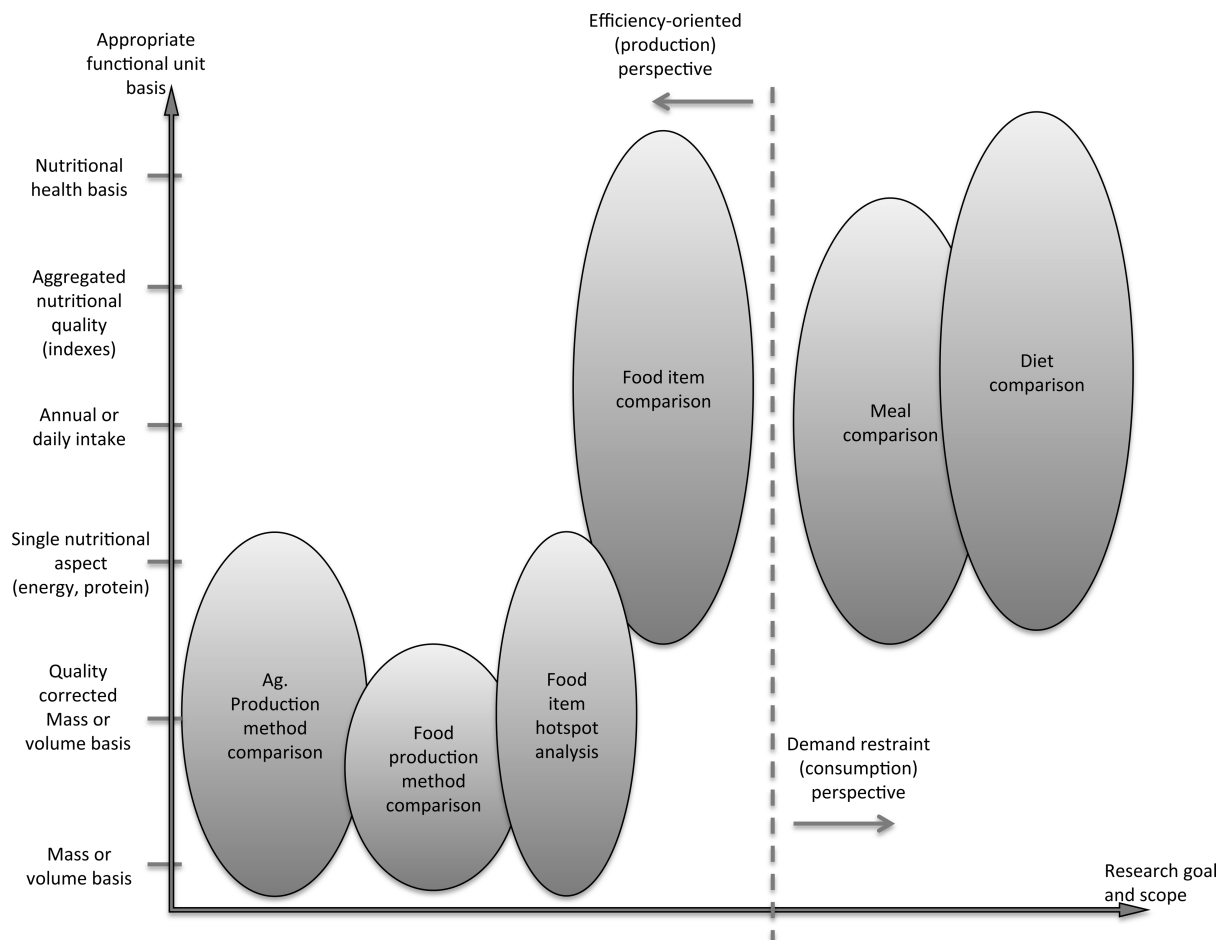


Figure 1.4: Conceptual map displaying the use of different functional units depending on the research question in food LCAs (Figure from Heller, Keoleian, Willett, 2013).

1.5.2.1. Single-aspect functional units

Traditionally, most food LCAs used mass or volume as a FU (Schau and Fet, 2008; Smetana et al., 2015; Sonesson et al., 2017). These FUs are sufficient to compare two identical end products. Heller & Keoleian (2018) performed a comparative LCA of the plant-based Beyond Meat Burger with a mass-based FU, arguing that nutritional profiles of the two burgers are similar. However, they only stated that the content of protein, iron, saturated fat, cholesterol, total fat, and calories in both products was similar, without justifying why they chose these elements over others, therefore possibly integrating biases. From a nutritional perspective, there are many more nutrients to look at to assess whether a food is nutritious or not. When the aim is to identify which food provides nutrition at a lower environmental cost, a mass, volume, or serving-based FU is inappropriate when comparing a legumes-based food with a non-legumes-based food, as nutritional contents might differ greatly.

An energy-based FU is more comprehensive than a mass-based FU, however it can favour unhealthy foods such as fast-food products, which are energy-dense but nutrient-poor, over healthier products. De Laurentiis, Hunt, Lee, & Rogers (2019) found that using energy as a FU lead to some vegetarian meals having a higher environmental impact when compared to meals with meat, due to their low energy content. However, the authors suggest that the developed tool should be used to evaluate existing recipes that are nutritionally equal, as the meals are otherwise incomparable. Therefore, they recommend considering nutrition in the analysis, which can be captured in the FU instead. Tyszler, Kramer, & Blonk (2014) performed product substitution with system expansion and linear optimisation as a solution to compare different food products. System expansion was performed to account for changes in consumption to achieve equal nutritional intake, and linear optimisation was used to define likely alternative diets that change as little as possible from the original diet. This approach eliminates the need for a nutritional FU, however it requires the modelling of a weekly diet, which has to be representative for each country. Moreover, 40 nutritional properties were used to define the lower and upper boundaries, baseline diet, and before and after scenarios, which make the method difficult to implement.

Another approach is to use protein content as a comparison unit (Sonesson et al., 2017). However, although protein is a crucial nutrient in food guidelines (WHO, 2007), there are numerous foods that are high in protein but contain differing amounts of unfavourable nutrients and should not be credited equally. The consumption of protein in developed countries per capita being currently 70% greater than the recommended values (Westhoek et al., 2011), a FU focusing on protein solely is unnecessary, as the nutrient is easily available. Sonesson et al. (2017) go even further by suggesting a digestible protein FU, as they argue that plant protein is of lower quality than animal protein due to a different amino acids balance. However, although plant protein has been shown to be less digestible than animal protein in children, net protein utilisation from animal or plant sources is comparable among adults (WHO, 2007), so distinguishing between simple grams of protein and digestible protein is an unnecessary complication in the FU. Other key nutrients should be considered for developed countries that are lacking nutrient-dense foods, such as fibre.

Therefore, FUs that consider only one aspect of foods are limited by their narrowness, and in a context of developed countries, it is not relevant to solely focus on protein, an abundant nutrient.

1.5.2.2. *Nutrient profiling methods as functional units*

Existing indexes that combine several nutrients could be used as FUs. These are extensively explained by Saarinen, Fogelholm, Tahvonen, & Kurppa (2017). Many consider nutrient density, based on the reasoning that today's diets are energy rich but nutrient poor (WHO, 2017). Nutrient density is the proportion of the amount of nutrients to the energy in a food product. Kägi et al. (2012) performed an LCA of a meal adjusted by the nutrient density score (NDS) and the nutrient rich foods index 9.3 (NRF 9.3). Both scores are very extensive, with the NDS considering 21 nutrients and NRF 9.3 considering 12, including 9 nutrients to favour and 3 to limit. Smedman et al. (2010) proposed a Nutrient Density to Climate Impact (NDCI) index based on 21 important nutrients in the Nordic Nutrition Recommendations (NNR). However, analysing the food products for all these nutrients in order to perform an LCA can be tedious and expensive, and the LCA practitioner might refrain from using these. The Food Quality Score (FQS) algorithms (Kennedy et al., 2008) were developed to be used as indicators of healthy foods. These calculate the percentage daily values of several nutrients (11 for the universal algorithm) and can be food group-specific (looking at different nutrients depending on whether the product is dairy, grains, or fruits and vegetables). Although looking at different numbers of nutrients, all FQS algorithms show similar food rankings. Drewnowski & Fulgoni (2008) also indicated that health rankings did not differ significantly between models considering 9 or 23 positive nutrients. Moreover, it was shown that models that looked at more than 12 micronutrients were less correlated to energy density than models that simply looked at protein, fibre, and three vitamins and minerals, suggesting that more complex indexes are not necessarily superior if we consider that nutrient density is key (Drewnowski, Maillot, & Darmon, 2009).

Validation against an approved health index is necessary when wanting to provide a FU that is indicative of health. Fulgoni, Keast, & Drewnowski (2009) criticised the fact that many indexes are not. They validated the NRF indexes with the Healthy Eating Index (HEI), which is a recognised American index linking food types and nutrients with recommended intakes. They compared the NRF indexes and observed the best correlation between the NRF9.3 and the HEI. NRFs with higher numbers of nutrients had a weaker correlation to the HEI.

However, many indexes such as the NRF index consider nutrients to limit, and although they seem key in identifying unhealthy foods, using them as a FU represents a major issue, as foods that are high in limiting nutrients may score negatively, which is not applicable in LCAs. This can be solved with numeric applications, however these would make the FU less transparent (Saarinen et

al., 2017). Another issue with some nutrient indexes is that algorithms used are questionable, such as the NDCI, due to the use of non-transparent threshold of recommended intake.

The Nutrient Density Unit (NDU) was developed by Van Dooren (2016) and solves the issues of opacity, data-hungry, and numbers negation of most FUs, by requiring nutritional values per 100 grams of fibre, protein, and essential fatty acids, and including nutrient density. Nutrients are weighed equally. Fibre is an essential macronutrient for maintaining a healthy diet and controlling weight (Kendall et al., 2010), and it was shown that diets in developed countries are too poor in fibre (Deehan and Walter, 2016). Validation against a recognised health index has not been performed yet for the NDU, however Williams et al. (2020) showed the correlation between the NDU and NRF 12.3 (Figure 1.5). The formulae of the NDU and NRF 12.3 were recorded in Equations 1 and 2, respectively. The NRF 9.3 was validated against the HEI. Using NDU as a FU becomes relevant for assessing foods in diets of developed countries, as it does not make a distinction between plant-based and animal protein. As mentioned previously, net protein utilisation is similar for plant and animal-based proteins in adults, and protein consumption far exceeding daily requirements, the different balance of amino acids in plants is no longer limiting. However, a more precise assessment of amino acid content may become necessary when assessing foods in developing countries, in which protein is a limiting nutrient.

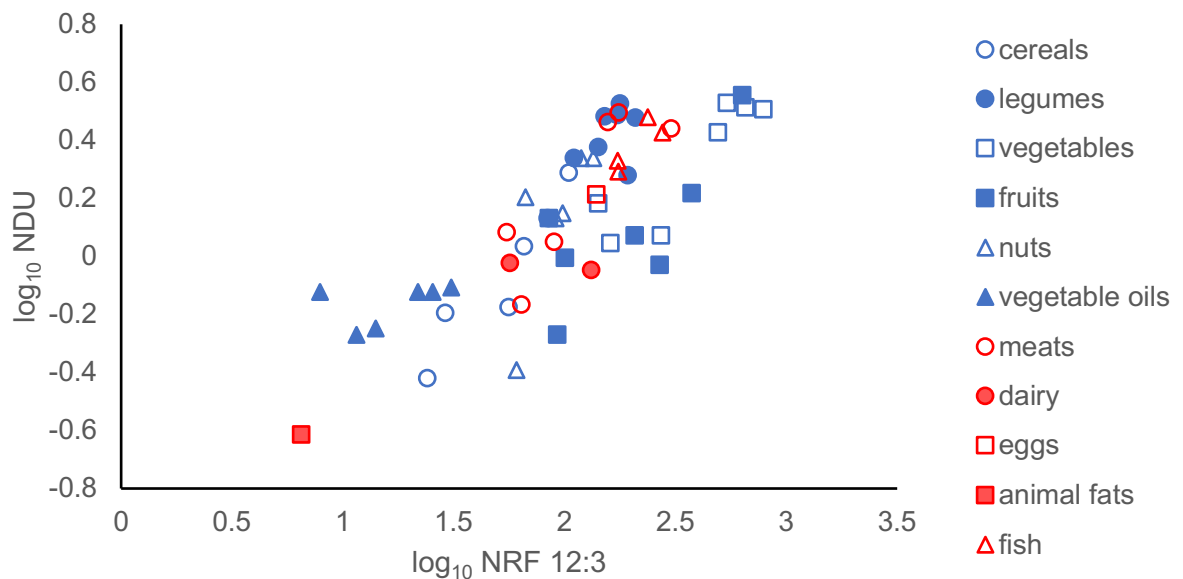


Figure 1.5: Correlation of NDU with NRF 12.3 for 55 separate foods across 11 food groups. $r^2 > 0.64$ (Williams et al., 2020b)

$$NRF_{12.3} = \frac{\sum_{1-12} \frac{nutrient_i}{DV_i} - \sum_{1-3} \frac{nutrient_i}{MRV_i}}{\frac{S_i}{2000}} \quad \text{Eq. (1)}$$

$$NDU = \frac{\left(\frac{EFA}{DV_{EFA}}\right) + \left(\frac{protein}{DV_{prot}}\right) + \left(\frac{fibre}{DV_{fibre}}\right)}{3 \times \left(\frac{S_i}{2000 \text{ kcal}}\right)} \quad \text{Eq. (2)}$$

Drewnowski (2005) criticises the fact that many nutrient scores attribute the same weight to all nutrients considered. Assigning different weights to the nutrients would allow distinction between highly available nutrients, such as protein, which is consumed in larger quantities in developed countries, and less available ones, such as omega-6 fatty acids. Bioavailability was also recommended as a weighting factor (Scarborough and Rayner, 2010). Katz, Njike, Rhee, Reingold, & Ayoob (2010) proposed the Overall Nutritional Quality Index (ONQI) algorithm for the NuVal™ system, which incorporates proprietary weights to take into account relative impacts of nutrients on health conditions. However, weighing nutrients decreases transparency of the FU, and in the case of the ONQI these weights are not accessible.

Criticism applicable to the NDU are as follow. The aggregation of all nutrients into one number is limitative, as two foods with different nutrients could score equally (Drewnowski, 2005; Rööß et al., 2015). This aggregation is however an advantage when wanting to communicate results easily to the public. Recently, it was argued that nutritional content should not be present in the FU, but rather as an impact in the LCA (Weidema and Stylianou, 2019). This reasoning is based on the claim that consumers do not consider alternative foods based on their nutritional content, but rather on the satisfaction they provide. However, nutrition remains the key role of food consumption, without which our bodies would be malfunctioning. Masset et al. (2015) used 100g and 100 kcal FUs, linking them to environmental impacts (using GHG emissions as a proxy), nutritional quality (through SAIN, LIM and UK Ofcom nutrient profiling models), and price. By doing so, nutrients were not captured in the FU, only juxtaposed with the FU, therefore excluding nutrition as the function of foods. The fact that consumers don't take into account nutritional content when evaluating two alternatives is highly debatable, as studies on food purchasing behaviours showed that nutrition is one of the aspects consumers consider, along with degree of liking and price (Aschemann-Witzel et al., 2013; Bower et al., 2003).

In the context of food sustainability, and considering the prominent health issues linked to overconsumption of unhealthy foods, delivery of nutrients was considered throughout this thesis as the key function of food to capture in the FU of LCAs. Table 1.1 summarises all indexes mentioned in this review. Several criteria were defined after analysing the literature: 1) it has to remain positive to be used in LCAs 2) data required should be easy to collect for simplicity 3) it should correlate with more extensive nutrient indexes 4) nutrient density should be included to maximise the amount of nutrient one receives from consuming calories, due to today's double paradox of obesity and undernutrition (Zelman and Kennedy, 2005) 5) the algorithm has to be transparent (Drewnowski and Fulgoni, 2008) 6) daily recommended values used should be extracted from a recognised authority (Drewnowski and Fulgoni, 2008) 7) the FU should be validated against a recognised health index. The NDU seemed to offer the best combination of features, by removing the nutrients to limit therefore remaining positive, requiring relatively simple data, correlating with the more complete NRF 9.3 index, which itself was validated against the HEI, including nutrient density in the denominator, being transparent due to no additional factors, and using the daily recommended values from the USDA and the Institute of Medicine.

Table 1.1: Potential functional units discussed in this review, with nutrients considered and their limitations.

FU	Nutrients considered	Limitations
Mass/Volume/Serving	NA	Does not consider nutrition, cannot compare foods with differing nutritional values
Food contribution to the nutrient composition of a weekly diet	>40	High data required
Energy content	NA	Can favour high energy but poor nutrient-dense foods, does not consider nutrition
Protein	Protein	Protein is not a limiting nutrient in Europe
Digestible protein	Digestible protein	Protein is not a limiting nutrient in Europe, net protein utilisation does not differ in adults
Nutrient Density Score NDS	Protein, fibre, vitamins A, B6, B12, C, E, D, thiamine, riboflavin, niacin, pantothenic acid, folate, calcium, iron, magnesium	High data required
Nutrient Rich Foods NRF9.3	Protein, fibre, vitamins A, C, E, calcium, iron, potassium, magnesium, saturated fat, added sugar, sodium	High data required, negative score possible
Nutrient Density to Climate Impact	Protein, carbohydrates, fat, retinol, vitamins D, E, C, B6, B12, thiamine, riboflavin, niacin, folate, phosphorus, iron, potassium, calcium, magnesium, selenium, zinc, iodine	High data required, non-transparent threshold of recommended intake
Food Quality Score (universal)	fibre, vitamins A, C, E, D, and B12, folate, calcium, magnesium, iron, potassium	High data required
NuVal™ (Overall Nutritional Quality Index)	Fibre, folate, vitamins A, C, D, E, B12, B6, potassium, calcium, zing omega-3 fatty acids, bioflavonoids, carotenoids, magnesium, iron, saturated fat, trans fat, sodium, sugar, cholesterol, protein quality, fat quality, glycaemic load, energy density	High data required, non-transparent
Nutrient Density Unit	Protein, fibre, essential fatty acids	Should be validated against an approved health index

1.5.3. Existing literature on life cycle assessment of legume-based processed foods

Studies that looked at the environmental effects associated with the production of legume-based food alternatives were recorded in Table 1.2. Some used functional units based on mass, such as the meatless burgers from Beyond Meat (Heller and Keoleian, 2018) and Impossible Foods (Khan et al., 2019). These showed that the plant-based burgers have a significantly lower environmental burden than beef burgers, with the Beyond Meat Burger causing 90% less GHG emissions, 46% less energy, >99% less water scarcity, and 93% less land use burdens than a conventional beef burger. Per kg of uncooked patty at factory, the Impossible Burger causes 92% less GHG emissions, 89% less aquatic eutrophication, and uses 96% less land 87% less water than a typical kg of beef burger. The two LCAs justified the use of a mass FU with the claim that nutritional profiles of the plant-based burgers were similar to the beef burgers. However, they only looked at protein, iron, saturated fat, total fat, carbohydrates, cholesterol, and calories, without justifying why these were selected. Several other nutrients are key when looking at whether a food is nutritious or not. Fibre, for example, is a key nutrient lacking in Western diets, and diabetes, obesity, cancer, and heart diseases are associated with poor fibre consumption (Brennan, 2005; Kendall et al., 2010; van Dooren et al., 2014). Moreover, the fact that there is no cholesterol and comparatively more iron in the Beyond Meat burger (Heller and Keoleian, 2018) should provide an advantage when comparing at the product's sustainability, which is not directly taken into account in the current mass-based FU LCA.

Table 1.2: Life cycle assessment studies of plant-based processed foods

Food	Functional unit	Impact categories	Reference
Meat substitutes			
Pea and mung bean protein	Mass	GHGE, energy use, water use, land use	(Heller and Keoleian, 2018)
Soy protein	Mass	GHGE, water use, land use	(Broekema and van Paassen, 2017)
Wheat	Mass	Eco-indicator 99	(Berardy, 2012)
Soy, lupine	Mass	GHGE	(Blonk et al., 2008)
Soy protein	Mass	GHGE, land use, water use	(Khan et al., 2019)
Soy meal	Mass, energy, digested dry matter protein	ReCiPe (18 indicators)	(Smetana et al., 2015)
Pea	Meal	GHGE, eutrophication, acidification, land use, energy use	(Davis et al., 2010)
Soy	Mass	GHGE	(Mejia et al., 2020)
Pea protein	Protein	GHGE, acidification, eutrophication, toxicity, water use, land use.	(Zhu & Van Ierland, 2003)
Pasta			
Pea protein	Mass	GHGE	(Nette et al., 2016)
Chickpea	Mass	GHGE	(Güngörmüşler et al., 2020)
Yellow pea	Nutrient balance score (27 macro- and micronutrients)	GHGE	(Chaudhary et al., 2018)

Legume-based pasta LCAs also exist, with Chaudhary et al. (2018) looked at pasta made from 30% pea and 70% wheat flours, and determined its associated GHG emissions and nutritional content, comparing it to traditional wheat pasta. Another comparative LCA of pasta made with egg or pea protein was from Nette et al. (2016). The latter study, like several other food LCAs, disregarded nutrition by using a mass-based functional unit (Heller et al., 2013), which could have possibly misinformed researchers on the broader sustainability of these foods. Chaudhary et al.

(2018), on the other extreme, used the Nutrient Balance Score for comparing the two alternatives, looking at the content of 32 macro and micronutrients. Such extensive scores involve an expensive process of nutrient analysis, as well as being lengthy in time, discouraging LCA practitioners to use these as a FU. Moreover, only the nutrient content was calculated for the raw products, and cooking ultimately changes these values (Fabbri and Crosby, 2016). Lastly, the two pasta studies analysed solely the carbon footprint of the products, and it is now widely accepted that it is not necessarily representative of other impact categories.

The attributional LCA approach is suitable to calculate the environmental burdens associated with the current production of these foodstuffs. Another main question is, nevertheless, the effects on the environment of integrating these foods in diets in case consumers substitute traditional alternatives with these. The following section provides a brief overview of existing LCAs of diets, and their limitations.

1.5.4. Existing literature on life cycle assessment of diets

Besides comparing the sustainability of individual food products, these will ultimately be part of a wider diet. It is therefore important to evaluate these environmental impacts from a wider perspective. In light of identifying which diet type is the most sustainable, numerous LCA studies compared the current diet of a country with vegetarian, vegan, and diets from recommended guidelines. Although many of them looked at GHG emissions only (Berners-Lee et al., 2012; Heller and Keoleian, 2015; Macdiarmid et al., 2012; Scarborough et al., 2014), several had more extensive combinations of impact categories, such as land, water, energy use, eutrophication, and ecotoxicity (Baroni et al., 2007; Meier and Christen, 2013; Saxe, 2014). Compared diets were sometimes adjusted to have the same energy and protein contents (Saxe et al., 2013; Tukker et al., 2011). All studies found that diets with animal products have a comparatively higher environmental impact than the vegetarian and vegan ones, yet, few studies have properly assessed the environmental impact of switching the current diet to another one using a consequential approach.

Some studies highlight the fact that a healthy diet does not necessarily have a low impact on the environment (Macdiarmid et al., 2012), and vice-versa (Berners-Lee et al., 2012), depending on the foods chosen. It is therefore crucial to consider both environmental sustainability and nutrition when devising diets. Heller & Keoleian (2015) also found that if the switch to a healthy diet was done while still keeping the same energy intake in the US, the switch would actually be associated with higher environmental burdens, and that it was necessary to reduce the energy intake to recommended levels in order to achieve a reduction of environmental burdens. However, to come to such conclusions, a cLCA approach is necessary.

Goldstein et al. (2016) compared the average Danish diet with vegan and vegetarian diets, using the provision of 2000 kcal a day (excluding non-dairy beverages) as a FU in a cLCA. They justified their scope ending at farm gate with the fact that most burdens are generated at the agricultural stage. However, as has been shown in some studies (Davis et al., 2010), including our Meatless burger LCA in Chapter 4, the high energy processing costs of some legume-based products can be major hotspots, and should therefore not be neglected.

Very few studies have calculated the effects of direct land use change (dLUC), and none have accounted for indirect land use change (iLUC) on the environment from diet change (Hallström et al., 2015). These effects, however, contribute significantly to climate change (Hallström et al., 2015), and it is therefore key to include them. In addition, very few studies have specifically included the use of processed plant-based alternatives in diet change scenarios, as the majority substituted animal products with unprocessed foods like pulses, cereals, vegetables, fruits, and nuts. To determine whether these innovative legume-based processed foods can contribute to the sustainable diet transition, it is therefore crucial to integrate them in a cLCA of diet change.

1.6. AIMS AND OUTLINE

Based on the current needs to determine the relative environmental and nutritional effects of innovative legume-based foods when compared to their conventional alternatives, and to assess their role in diet change in Europe to support a sustainable transition, this PhD project aims to answer the following question:

- What is the environmental footprint of legume-based processed foods in Europe, considering their provision of nutrition?

This question is answered using an attributional LCA approach. Chapter 2 follows this introduction and contains the methodology used throughout this thesis, presenting aLCAs and the identified limitations of this approach. Following the methodology chapter are the aLCAs of chickpea pasta (Chapter 3), pea protein balls (Chapter 4), pea and soy burger patty (Chapter 5), aquafaba mayonnaise (Chapter 6). Chapters 3 to 5 make use of the Nutrient Density Unit proposed by Van Dooren (2016), so that foods are compared from a combined environmental and nutritional perspective. The discussion bringing all the food items into perspective is reported in Chapter 7.

2. MATERIALS AND METHODS

In the previous chapter, I introduced key challenges of the food sector in Europe: to produce and consume nutritious foods at a low environmental cost, and the role that legumes play in this context. I also presented the LCA approach as a quantitative decision support tool that can be used to benchmark foods to identify more environmentally and nutritionally sustainable foods. In this chapter, I present the full methodology of LCA used across this thesis: its steps, the calculation of the nutrient density functional unit, and the environmental issues covered.

2.1. LIFE CYCLE ASSESSMENT PRINCIPLES

Life Cycle Assessment (LCA) is a popular tool for quantifying the environmental burdens associated with a product or service life cycle, from the extraction of resources to production, use, disposal, and end of life (ISO, 2006a). As shown in Figure 2.1, the methodology consists of four iterative stages: goal and scope definition, inventory analysis, impact assessment, and interpretation (ISO, 2006b). LCA results can then be used for various purposes, such as eco-design, policy making, and marketing.

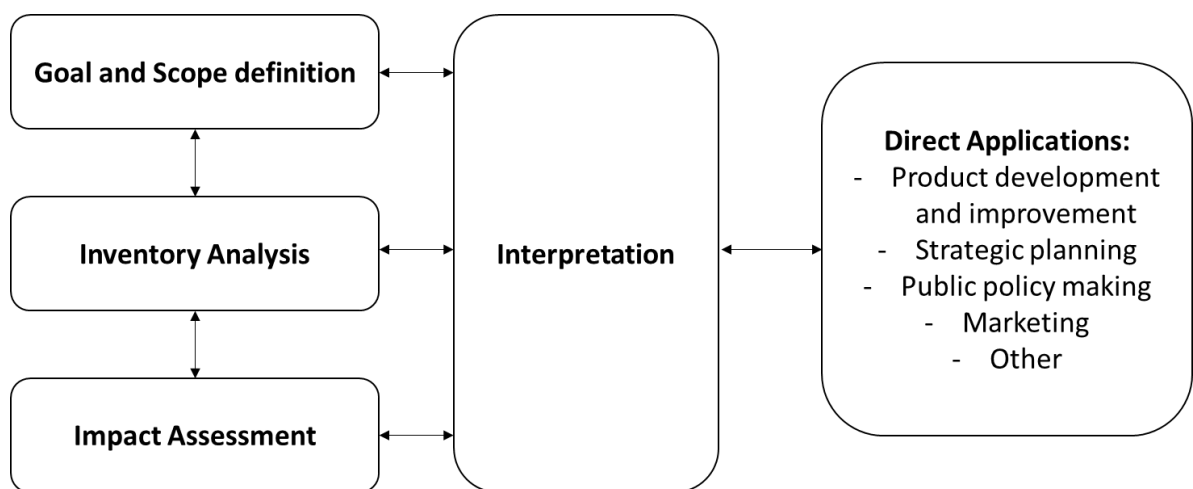


Figure 2.1: Stages of a Life Cycle Assessment. Adapted from ISO 14040 (2006)

2.1. ATTRIBUTIONAL AND CONSEQUENTIAL LIFE CYCLE ASSESSMENTS

There are currently two main types of LCAs, depending on the aim of the study: attributional and consequential LCAs. These differ in the system boundaries selected. Which activities are included, and how individual processes are linked determine the modelling type (Figure 2.2). This

thesis uses an attributional approach, which is best suited to answer the question recorded in Section 1.1.6. The aim of an aLCA is to determine the environmental burdens associated with the production of a certain product. This is done by allocating processes of the system following a normative rule (Sonnemann et al., 2011). Therefore, the inventory of an aLCA is based on data from a market average or from a specific supplier.

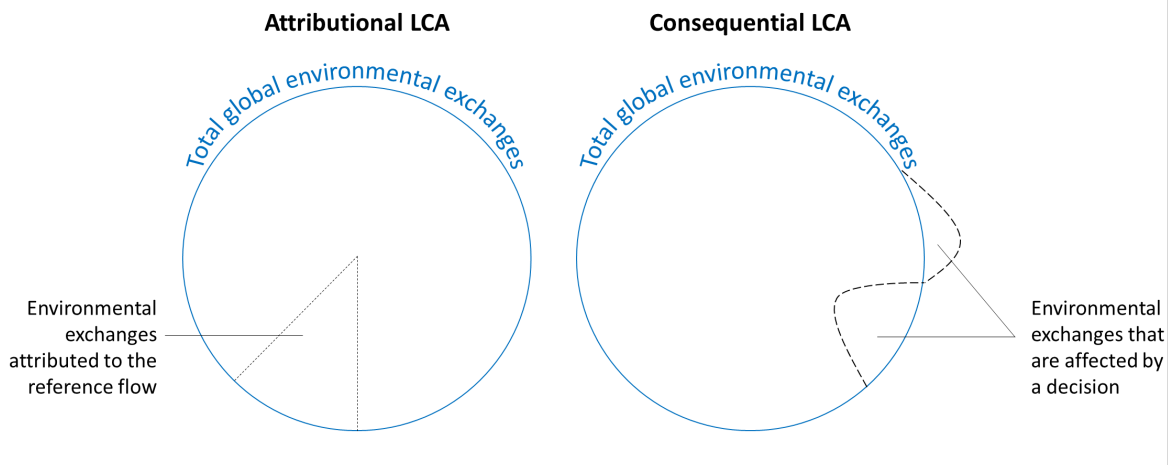


Figure 2.2: Schematic representation of attributional and consequential life cycle assessment (Adapted from Weidema, (2003)): The circle represents the total environmental impact of the world. An aLCA model defines a piece of this world to be attributed to the functional unit. The dotted lines aim to highlight the use of a normative rule as to which specific part of the world environmental exchanges can be attributed to the functional unit, due to the fact that every process is theoretically linked. On the other hand, cLCA only assesses what part of the world environmental exchanges will change when the demand for the functional unit is altered, with some exchanges increasing and others decreasing.

2.2. LIFE CYCLE ASSESSMENT SOFTWARE: OPENLCA

All the LCAs in this thesis were modelled in OpenLCA v1.10, an open-source LCA software (GreenDelta, 2020). Two databases were used: Ecoinvent v3.6 (Wernet et al., 2016) and Agri-footprint v3.0 (Durlinger et al., 2017). These databases are made up of elementary flows, product flows, waste flows, and processes. An elementary flow represents a material/energy that enters directly (if inputted) the studied system without having been adjusted by human action, or leaves directly (if outputted) the studied system with no further human modification. Therefore, these directly come from or are released to the environment. Examples of these include groundwater, and emissions to the air. Product flows represent the material/energy that was transformed by human action and will be used further, and are thus exchanged between processes within the system. These include, for example, low voltage electricity or refined palm oil. Waste flows represent material/energy leaving the product system as waste, such as biowaste or wastewater.

Processes represent the elements in the system in which input and output flows are quantified. Ecoinvent v3.6 (Wernet et al., 2016) is an international database that is most commonly used for all background processes, such as electricity generation and transport, while Agri-footprint (Durlinger et al., 2017) is specialised in the agri-food domain and has specific food processes that are absent in Ecoinvent v3.6. Agri-footprint v3.0 was used only when Ecoinvent v3.6 did not have the desired data. The next sections, **2.3-2.6** explain the four steps in a LCA.

2.3. GOAL AND SCOPE DEFINITION

This step defines particularly why, what, for who, and how the LCA study will be carried out. The goal of the LCAs in this thesis was to assess the potential of legume-based products as healthy and environmentally-sustainable alternatives to traditional food and beverage products in such a way as to inform researchers and policy makers. To achieve this, for each of the product categories selected, a comparison of one legume-based product and its traditional alternative was performed.

To perform an LCA, a Functional Unit (FU) has to be decided upon. Defining the FU refers to quantifying the task to be fulfilled by the studied system(s) (ISO, 2006a). For one FU then, several quantifiable reference flows can be selected. The LCA studies presented in this thesis were carried out using two FUs, one based on mass/serving, and the second FU based on the NDU. As discussed in Chapter 1, the NDU, although is a simple calculation, correlates with significant analyses of a number of nutrients. For example, in the pasta LCA study (Chapter 3), the mass functional unit was a serving of pasta, cooked. The corresponding reference flows were 80 g of dry pasta, cooked. In addition to this, when the products examined had a nutritional purpose (as opposed to alcohol and mayonnaise), a second FU was used: the Nutrient Density Unit (Van Dooren, 2016). With a NDU of 2.3 for chickpea pasta and 0.9 for durum wheat pasta, the reference flows corresponding to one NDU of pasta were thus 35 g of dry chickpea pasta, cooked, and 89 g of dry durum wheat pasta, cooked. Hence, the NDU is an example of FU that quantifies the nutrient density as the performance achieved by food. Instead of comparing foods on a serving size basis, the NDU integrates the nutritional aspect of foods, which is key to identify the optimal foods to integrate in European diets, which need to integrate more nutrient-dense foods and less energy-rich, nutrient-poor ones.

To ensure comparability between the products assessed, a proper system boundary definition is key to make sure any biases are limited. The system boundary defines the elements of the product system that are assessed, and those that are left out. These boundaries can include

production inputs and outputs of raw materials, manufacturing, transport, storage, consumer cooking, and waste disposal. The LCAs carried out in this thesis were from field to fork, for products that require cooking (pasta and meat analogues), and from field to factory gate for those that do not (mayonnaise). This is because distribution, storage, and use are the same for the compared products that do not require cooking in this study. Once the contact with the food company was established, constant communication was required to understand the steps involved all along the production chain, to build a flow diagram like the one presented in Figure 2.3.

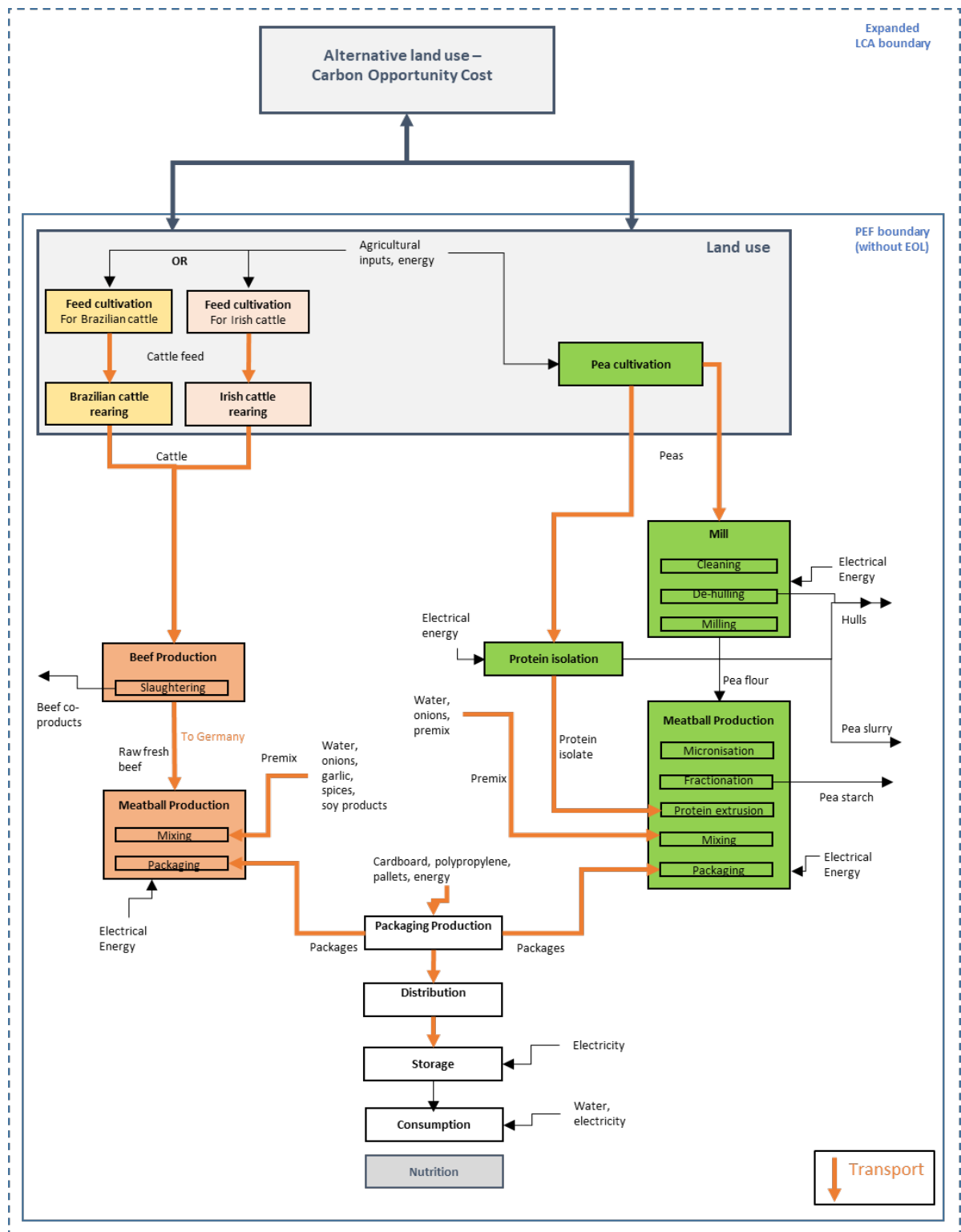


Figure 2.3 from Saget et al. (2021): Example of a system boundary flow diagram.

Here, three products are assessed: protein balls made from peas, Irish beef, or Brazilian beef. The reference flow is a 100 g serving of protein balls, cooked at consumer house. Two system boundaries are presented: the baseline system boundary (PEF boundary without End of life), and the expanded LCA boundary.

2.3.1. Multifunctionality and allocation

Multifunctional processes are processes that yield multiple output products that are not waste. For example, dairy cow rearing provides milk, cattle for slaughter, and calves. LCA modelling requires decision-making on how the environmental burdens can be attributed to a single product from multifunctional processes. The ISO 14044 guidelines (ISO, 2006b) provide a stepwise procedure to deal with multifunctionality:

1. Avoid allocation by:

- 1.A. Dividing the process into several sub-processes containing individually the corresponding input and output data.

- 1.B. Expanding the system boundaries to include all of the functions provided by all of the co-products.

Following 1.A. is not possible when co-products cannot be obtained independently. Where it is possible, it would require extensive additional data, as most processes are multifunctional, and would decrease transparency, using numerous assumptions (Mackenzie et al., 2017). Following 1.B. would imply accepting that the environmental burdens calculated will no longer correspond to those of the unique product.

If option 1 cannot be performed, ISO 14044 guidelines (ISO, 2006b) recommend to:

2. Allocate burdens between co-products based on their “physical relationships”, i.e. their causal relationships.
 3. If no physical relationship can be determined, allocate burdens between co-products based on another relationship, such as economically.

Opting for option 2 could for example be to allocate burdens based on mass, such as in a dehulling of faba beans process, 90% of the environmental burdens would be allocated to the bean kernels, and 10% to the hulls, for the case in which 90% of the bean mass were kernels and 10% were hulls. However, in the case of meat products, this approach is not representative of the researched cause-effect relationship, as a same mass of different body tissues requires different energy inputs. Therefore, option 2 for meat systems could also be allocating burdens based on a biophysical approach as shown in Figure 2.4, which partitions burdens based on metabolic energy requirements to produce and maintain body tissues, excluding tissues treated as waste (Chen et al., 2017).

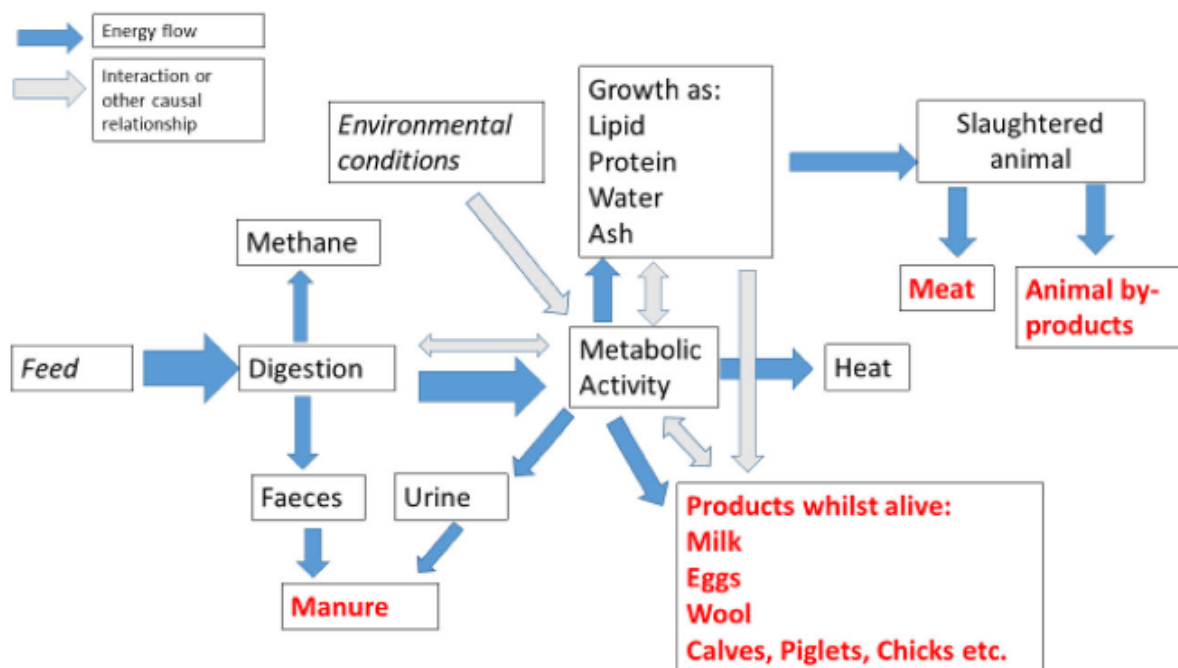


Figure 2.4 from Mackenzie et al. (2017): Example of livestock flowchart showcasing co-products and with energy flows and other causal relationships.

Finally, option 3 designates most commonly economic allocation, for which environmental burdens are allocated to specific co-products based on the relative economic value of these co-products. Taking the example from Figure 2.4, all of the co-products in red would be allocated the corresponding proportion of the environmental burdens of the whole system based on their relative economic value and amounts produced. The limitations of this approach stem from price changes due to market fluctuations.

Following the PEF guidelines, economic or mass allocation of co-products along the processes in the life cycles was performed in the LCA studies of this thesis (European Commission, 2018a). For LCAs involving beef, a sensitivity analysis with biophysical allocation for co-products of beef production was performed, and compared with the legume-based product for which a simple mass allocation was performed for legume processing co-products. Relevant allocation factors were extracted from the PEF guidelines (European Commission, 2018a) where possible. If no data was available in the PEF guidelines, personal communication with industry experts was performed, as well as literature search, to obtain robust economic allocation factors.

2.4. INVENTORY ANALYSIS AND MODELLING

Table 2.1 lists the food and drinks manufacturers contacted via email or on their webpage about carrying out possible LCAs. Of this list of 92 companies, no plant-based milk companies responded, and 7 meat analogue and pasta companies replied requesting more information. After the first call, only three pasta companies and one meat analogue company were willing to participate. However, after contacting the suppliers of raw materials, only one pasta and one meat analogue companies were disposed to take part in the study. Besides this, we decided to perform LCAs for one mayonnaise company that established contact independently, one TRUE partner that makes meat analogues, and one legume brewer. For the selected products, a comparative LCA was performed, using on top of the selected legume products a traditional alternative that was either produced directly from the same food company if available, or from an already published LCA.

Table 2.1: List of companies that were contacted for this PhD, with the legume type used in their products.

Pasta	
Company name	Legume used
Banza	chickpeas
Tolerant	lentil, chickpea, red lentil, black bean
Modern Table	lentil
Ancient Harvest	black beans
Explore Cuisine	black beans, mung beans, edamame
Sainsburys	red lentil
Napolina	green pea, red lentil
Felicia	red lentil
Dalla Costa	red lentil, green pea, chickpea
Pedon (more than pasta)	chickpea, green pea, red lentil
Pasta Lensi (Tree House)	red lentil
Bentilia	lentil, bean
Choose Chickpea	chickpea
Really Healthy Pasta	red lentil, mung bean, black bean, chickpea
Barilla	chickpea, lentil
Tutto Bene	green peas, chickpeas, red lentils
Sam Mills	black bean, green pea, chickpea, red lentil

San Remo	peas, lentils, chickpeas and borlotti beans
Pasta Gustosa	red lentil, green pea, yellow lentil
Prairie Harvest	chickpeas, green split peas, lentils, red kidney beans
Green Banana	green banana and legumes
Variva	chickpeas
Pasta Armando	chickpea, lentil, bean, pea
Profusion Organic	lentil
Seggiano	lentil
Doves Farm (Freee)	lentil
Little Pasta Organics	lentil
La Bio Idea	lentil
Milk substitute	
Eden Foods	soy
Pure Harvest	soy
Silk (Danone)	soy
American Soy Products	soy
WestSoy (Hain Celestial)	soy
Pacific Natural Foods	soy
Better Than Milk (Panos Brands)	soy
Sanitarium	soy
ZenSoy	soy
SunOpta	soy
Vitasoy International Holdings	soy
Alpro (Danone)	soy
So Good	soy
Provamel	soy
Valsoia	soy
Granarolo	soy
Shoyce	soy
Bjorg	soy
WhiteWave Foods (Danone)	soy
So Delicious (Danone)	soy
Soy	soy
Sojade	soy

Lima Food	soy
Lea Nature	soy
Isola Bio	soy
Monsoy = Liquats	soy
Bio-regain	soy
Naturlii	soy
Sojasun	soy
Meat substitute	
Impossible Foods	soy
Sunfed Meats	pea
New Wave Foods	soy
Tofurky	soy
Ripple Foods	yellow peas
Lightlife Foods	soy, pea, beans, chickpeas
NotCo	lupine and others
Good Catch Foods	pea, soy, chickpea, lentil, fava, navy
JUST	mung bean (egg), yellow pea (mayo)
Ojah	soy protein, yellow pea protein
ChickP	chickpea
Moving Mountains	soya and pea protein
Gold and Green Foods	fabia beans, yellow peas
The Vegetarian Butcher	soy protein
Beyond Meat	soy, pea
Boca (Kraft Heinz)	soy
Gardein	soy, pea
Morning Star Farms (Kellogg's)	soy
Schouten	soy, wheat, pea
Vivera	lupine
Oumph!	soy
Vbites	soy
Sgaia	soy
Mock	soy
ADM	soy, black bean, navy bean,
No Evil Foods	chickpea, white beans for the pardon

So meat	soy
Naturlii	peas
Meatless	fava bean
The Meatless Farm	pea, soy
Linda McCartney	soy
Goodlife	lentils, soy, pea
Garden Gourmet	soy
Gosh!	broad bean
Heck	pea

Guided by the system boundary diagram, the inventory analysis stage records and quantifies all the input and output flows belonging to air, water, and land that circulate in and out of the defined product system that provides the FU. For each stage across the product life cycle and following the reference flow, inputs and outputs were recorded and quantified as shown in Table 2.2. Where data was missing, research was carried out using existing literature and by personal communication with the relevant industry body.

Table 2.2: Example of a life cycle inventory recording, here for the pea cleaning and dehulling stage part of the pea protein ball production presented in Chapter 4.

Pea cleaning and dehulling				
Inputs	Flow	Amount	Unit	Description
	Green/yellow peas ready for cleaning	0.051	kg	
	Electricity, medium voltage	5.09	Wh	electricity for dehulling
	Electricity, medium voltage	2.14	Wh	electricity for cleaning/screening
	Transport, freight, lorry with refrigeration machine, 3.5-7.5 ton, EURO4, R134a refrigerant, cooling	15.1	kg*km	Pea transport to cleaning/dehulling facility
Outputs	Flow	Amount	Unit	Description
	Cleaned pea kernels	0.043	kg	
	Pea hulls	0.008	kg	

Once the inventory data was collected, the whole product system was modelled in OpenLCA v1.10 (GreenDelta, 2020), downloaded from OpenLCA Nexus (GreenDelta GmbH, 2020). The aim of the LCA software is to make the process of performing a LCA easier. Databases containing environmental flows, processes, and impact assessment methods can be uploaded to the software. The systems studied can be modelled within it, so that the environmental burdens of these can be calculated directly, instead of having to connect manually all of the environmental flows, via Excel, for example. The databases “Ecoinvent 3.6 Cut-Off Unit Processes Regionalised” and “Agri-footprint 3.0” were downloaded from OpenLCA Nexus (GreenDelta GmbH, 2020) and imported in OpenLCA v1.10 (GreenDelta, 2020). In the software, the two databases Ecoinvent v3.6 (Wernet et al., 2016) and Agri-footprint v3.0 (Durlinger et al., 2017) were merged into a single database. Processes were created for each of the life cycle stages of the product systems, as well as process flows. In these processes, existing processes present in the Ecoinvent v3.6 (Wernet et al., 2016) and Agri-footprint v3.0 (Durlinger et al., 2017) databases were inputted as secondary data, for electricity for example. When primary data was available, additional processes were built. If, for a certain stage, a process or flow was inexistent and no primary data were available, proxy processes were used, such as cultivation of chives substituted for onion cultivation. Allocation factors were also inputted in the “Allocation” tab in OpenLCA v1.10.

All stages were linked, with the corresponding output of a process stage inserted as an input of the following process stage, so that by the end the whole product system to be evaluated was connected, as shown in Figure 2.5.

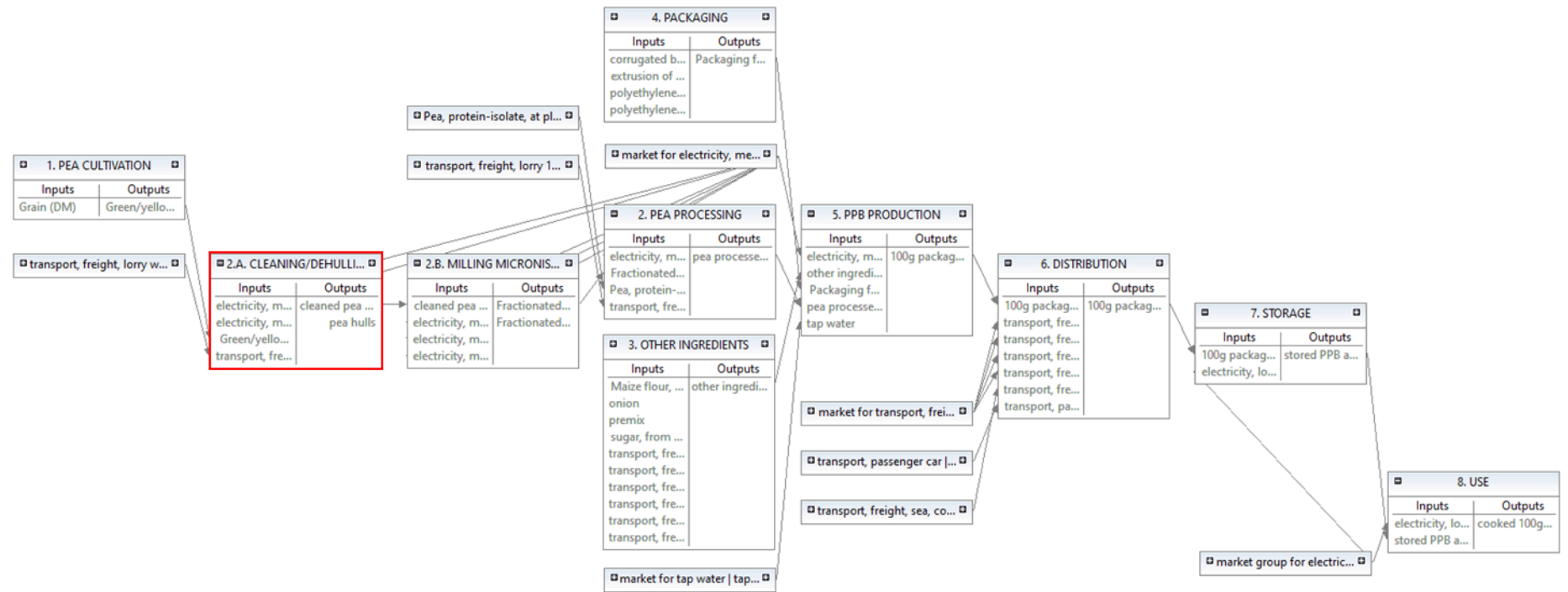


Figure 2.5: Example of a product system modelled in OpenLCA v1.10, here of pea protein balls, from cultivation to use. The rectangle outlined in red represents the process presented in Table 2.2.

Direct and indirect emissions from crop residues were quantified using IPCC equations 11.2, 11.9, and 11.10 (IPCC, 2019). Carbon dioxide emissions from lime application were calculated following IPCC (2006). Phosphorus leaching to water from fertiliser application was calculated using the cropland loss coefficients indicated by Styles et al. (2015). Country-specific fertiliser type breakouts were extracted from the International Fertilizer Association (2020). Crop protection was outside of the scope of the LCAs in this thesis, due to highly variable types and amounts applied. An annual application of 400 kg.ha⁻¹ was consistently modelled for all crops including legumes, as a conservative approach. Transportation and use phases were modelled following the general PEF guidelines (European Commission, 2018a), or product-specific PEF guidelines, such as for pasta (European Commission, 2018c).

2.4.1. Nutritional analysis

For those foods that brought some positive nutrition and for which the NDU FU was also used (pasta, burger patty, and protein balls), samples of these products and their selected alternatives were sent to the partner laboratory at the Centre for Biotechnology and Fine Chemistry of the Catholic University of Portugal (Portuguese Catholic University, 2020). Cooking was performed following the cooking guidelines of each product. Methods for determining nutritional composition and energy content of the foods were recorded in Table 2.3. Random replicates were performed to 7% of the analysis by the laboratory, which follows standard and certificated protocols.

Table 2.3: Methods for nutritional and energy analyses carried out for the food products in this thesis.

Nutrient/Energy	Method	Reference
Protein	Kiedahl (ISO 1871:2009)	(ISO, 2009)
Energy	EU regulation 1169/2011	(European Union, 2011)
Fibre	Enzymatic-Gravimetric from AOAC 991.43 and AOAC 985.29	(Lee et al., 1992; Prosky et al., 1985)
Essential fatty acids	Gas chromatography (FID) from ISO 12966-1:2014; 12966-2:2011; 12966-3:2016	(ISO, 2016, 2014, 2011)

2.5. IMPACT ASSESSMENT

The LCAs undertaken for this thesis assessed products across the sixteen environmental impact categories recommended by the European Commission, as presented in Table 2.4 (European Commission, 2018a) and developed further in Subsections A-G. The methods package “EF method, adapted” from OpenLCA LCIA methods v.2.0.6 was used (GreenDelta GmbH, 2020). This package aims to offer the methods recommended by the European Commission, in its initiative to standardise LCAs of products, to facilitate reproducibility, comparability, and verifiability (European Commission, 2019a). This package was adapted from the European PEF methods package to be compatible with the Ecoinvent and Agri-footprint data libraries. PEF normalisation factors per global person equivalents were applied to the results to assess results under a single unit (Crenna et al., 2019). These normalisation factors were obtained by characterising the environmental impacts across the assessed categories at the global scale, and scaling them down to a single person.

For all the chapters of the thesis, LCIA results were presented for each reference flow raw and normalised. Process contributions were shown as individual bar charts for each step of the life cycles across every category. This part of the results is key to identify hotspots in the products life cycles, to understand which steps contribute the most to specific impact categories and represent the greatest improvement opportunities.

Table 2.4: Life cycle impact assessment methods employed in the studies. Adapted from European Commission (2018c)

Impact category	Indicator	Unit	Recommended default LCIA method	Normalisation Factor
Climate change	Radiative forcing as Global Warming Potential (GWP100)	kg CO ₂ eq.	Baseline model of 100 years of the IPCC (based on (IPCC, 2013))	7758
Ozone depletion	Ozone Depletion Potential (ODP)	kg CFC-11 eq.	Steady-state ODPs as in (WMO, 1999)	2.34E-02
Human toxicity, cancer*	Comparative Toxic Unit for humans (CTUh)	CTUh	USEtox model (Rosenbaum et al., 2008)	3.85E-05
Human toxicity, non-cancer*	Comparative Toxic Unit for humans (CTUh)	CTUh	USEtox model (Rosenbaum et al., 2008)	4.75E-04
Ionising radiation, human health	Human exposure efficiency relative to U235	kBq U ²³⁵ eq.	Human health effect model as developed by Dreicer et al. 1995 (Frischknecht, et al., 2000)	4219
Photochemical ozone formation, human health	Tropospheric ozone concentration increase	kg NMVOC eq.	LOTOS-EUROS model (van Zelm et al., 2008) as implemented in ReCiPe 2008	40.6
Acidification	Accumulated Exceedance (AE)	mol H ⁺ eq.	Accumulated Exceedance (Posch et al., 2008; Seppälä et al., 2006)	55.6
Eutrophication, terrestrial	Accumulated Exceedance (AE)	mol N eq.	Accumulated Exceedance (Posch et al., 2008; Seppälä et al., 2006)	177
Eutrophication, freshwater	Fraction of nutrients reaching freshwater end compartment (P)	kg P eq.	EUTREND model (Struijs et al., 2009) as implemented in ReCiPe	2.55
Eutrophication, marine	Fraction of nutrients reaching marine end compartment (N)	kg N eq.	EUTREND model (Struijs et al., 2009) as implemented in ReCiPe	28.3

Ecotoxicity, freshwater*	Comparative Toxic Unit for ecosystems (CTUe)	CTUe	USEtox model, (Rosenbaum et al., 2008)	11815
Resource use, minerals & metals	Abiotic resource depletion (ADP ultimate reserves)	kg Sb eq.	CML 2002 (Guinée et al., 2002; van Oers et al., 2002)	5.79E-02
Resource use, fossils	Abiotic resource depletion – fossil fuels (ADP-fossil)	MJ	CML 2002 (Guinée et al., 2002; van Oers et al., 2002)	65274
Land use	Soil quality index	Dimensionless (point)	Soil quality index based on LANCA (Beck et al., 2010; Bos et al., 2016)	1334401
Water scarcity	User deprivation potential (deprivation-weighted water consumption)	m ³ world eq.	Available Water Remaining (AWARE) as recommended by (UNEP, 2016)	11469
Particulate Matter	Impact on human health	Disease incidence	PM method recommended by (UNEP, 2016)	6.37E-04

*Based on total human land appropriation

2.5.1. Climate change

Climate change represents the change in global temperatures from anthropogenic GHG emissions, due to, amongst others, burning of fossil fuels, agriculture, cement production, and deforestation. The consequences of climate change range from modifying weather patterns and rising sea levels, increasing the risk of droughts, fires, and floods, to increasing risks of extinction of numerous species. To represent climate change in LCA, the Global Warming Potential (GWP) is quantified through radiative forcing impacts of GHGs, i.e. the ability of GHGs to trap heat into the atmosphere over a defined timespan, here of 100 years. In this category, carbon dioxide is the reference gas, with a GWP of 1.

In this thesis, Carbon Opportunity Cost (COC) was also taken into account when comparing the climate change burdens of a legume product versus a meat one (Chapters 4 and 5). COC represents the difference between how much atmospheric CO₂ is stored under a specific land use,

and how much carbon it could potentially store if that same land was restored to its native ecosystem (Hayek et al., 2020). (Searchinger et al., 2018). It is estimated that the total global COC of animal agricultural rearing amounts to 152.5 gigatons of carbon (Hayek et al., 2020). In Chapters 4 and 5, the COCs were calculated per serving and per NDU for each products, and were also used in the extrapolation scenarios at a country level. Factors for COCs specific to the foods produced were extracted from Searchinger et al. (2018) and are represented in Table 2.5.

Table 2.5: Carbon opportunity costs of agricultural products used in this thesis. Adapted from Searchinger et al. (2018)

Product	COC (kg CO ₂ per kg fresh weight)
Soybean	5.9
Vegetable	0.7
Wheat	1.9
Potato	0.6
Vegetable oil	9.7
Rice	2.6
Pulses	10.5
Beef	144
Maize	2.1

2.5.2. Ozone depletion

Ozone depletion represents the damage of the ozone layer caused by anthropogenic sources. Ozone depletion increases the amount of carcinogenic rays reaching the Earth, due to ultraviolet (UV) light readily entering the atmosphere. Ozone depletion is due to emission of chlorofluorocarbons (CFCs), halons, and hydrochlorofluorocarbons (HCFCs). These stable compounds are found in cooling, cleaning, and aerosol systems, and are broken down in the stratosphere when in contact with UV light from the sun. Free chlorine, one of the compounds released, interferes with the ozone layer and depletes it. In LCA, ozone depletion is represented by the Ozone Depletion Potential (ODP) in kg CFC-11 equivalents.

2.5.3. Human toxicity, cancer effects and non-cancer effects

Human toxicity, cancer effects and non-cancer effects are two of the three toxicity-related categories in this thesis. They represent the discharge of toxic chemicals into the environment, which are carcinogenic in one category, such as chromium and nickel, and non-carcinogenic in the second, such as mercury and cadmium. Use of fossil fuels to generate energy releases these compounds into the environment. Human toxicity categories are expressed as Comparative Toxic Units for human toxicity (CTUh), representing the projected increase of illness cases in the whole population per amount of compound emitted.

2.5.4. Ionising radiation, human health

Ionising radiation human health, represents the impacts on human health from the release of high energy particles that can ionise atoms and molecules, i.e. which remove electrons from them. This can alter DNA, causing cancer and other illnesses. The ionised atoms and molecules (radionuclides), such as Carbon-14 and Cesium-134, result from energy production from nuclear or other sources, phosphate rock, oil, and, gas extraction (Frischknecht et al., 2000). The indicator representing this impact is the Human exposure efficiency relative to U^{235} .

2.5.5. Photochemical ozone formation, human health

Photochemical ozone formation, human health is a category capturing the impact of the reaction of methane, carbon monoxide, and non-methane volatile organic compounds (NMVOCs) with nitrogen oxide and sun, forming ozone in the troposphere. This creates smog, and causes respiratory illnesses, eye, nose, and throat irritations. Smog also affects agricultural yields and resistance to diseases. This impact therefore captures the secondary pollution resulting from vehicle, factory, and agricultural emissions. The LCA indicator is the Tropospheric ozone concentration increase, measured in NMVOC equivalents.

2.5.6. Acidification, terrestrial and freshwater

Acidification refers to the pH reduction from the reaction between acidic gases, such as ammonia, nitrogen oxide, and sulphur oxide, with atmospheric water, which causes acid rain. This

acid rain releases aluminium from the soil, and challenges the survival of aluminium and acid-sensitive organisms, including fish and plants. Acidification, terrestrial and freshwater is measured in LCA with the Accumulated Exceedance (AE), in mol H⁺ equivalents.

2.5.7. Eutrophication: terrestrial, freshwater, and marine

Eutrophication refers to a system being overly enriched with nutrients – mostly P and N - and minerals. Leakage of P and N in water systems causes extreme algae propagation, hypoxia due to bacterial consumption of the dead algae, and results in the death of aquatic flora and fauna. In a terrestrial ecosystem habitually characterised by low nutrient availability, more competitive plant species can overtake other species that habitually thrive in the low-nutrient systems, resulting in a decrease of biodiversity. Terrestrial eutrophication is measured in LCA with Accumulated Exceedance in N equivalent, freshwater eutrophication with the fraction of nutrients reaching the freshwater end compartment in P equivalent, and marine eutrophication with the fraction of nutrients reaching the marine end compartment in N equivalent. Exceedance of N and P result mainly from agricultural systems with fertiliser usage.

2.5.8. Freshwater ecotoxicity

Freshwater ecotoxicity quantifies the effect of toxic substances emission into freshwater. These substances include heavy metals like mercury, arsenic, and cadmium. This environmental impact is measured through the Comparative Toxic Unit for ecosystems (CTUe). The CTUe refers to the proportion of species potentially affected per cubic meter per day per kilogramme of substance emitted.

2.5.9. Resource use, minerals and metals and fossil fuels (energy carriers)

These two categories represent the use of an abiotic resource - lifeless chemical and physical matter - such as coal, relative to its scarcity. This concerns all activities that consume abiotic resources, including energy extraction, transport and agriculture, which consume copper, natural gas, and crude oil amongst others. Resource use, minerals and metals is quantified in the LCAs of this thesis through abiotic resource depletion, in antimony equivalents, and Resource use, fossil fuels in Mega Joules.

2.5.10. Water scarcity

The water scarcity category aims to capture the potential impacts of water use in a specific location after all types of water uses were performed (anthropogenic and naturogenic), therefore determining the deprivation potential if more water was to be used in that area. Hydroelectricity production and irrigation are examples of processes that affect the water scarcity category, captured with the user deprivation potential in m³ world equivalent.

2.5.11. Land occupation

Land occupation is measured with the LANCA[®] tool (Bos et al., 2016). It combines the following land capacities:

- 1) *Erosion resistance*. This refers to the extent to which land can resist to the erosion created in addition to naturogenic erosion, and is measured in amount of erosion taking place as a result of the FU over the corresponding time period.
- 2) *Mechanical filtration*. This represents the ability of the soil to filter contaminating particles. Permeability is evaluated, depending on soil texture, and mechanical filtration capacity is determined as the quantity of water that was prevented from being filtered as a result of the FU over the corresponding time period.
- 3) *Physicochemical filtration*. This refers to the cation exchange capacity of the soil, which varies with texture and pH of the soil. Physicochemical filtration capacity is determined as the quantity of cations that was prevented from being fixed as a result of the FU over the corresponding time period.
- 4) *Groundwater replenishment*. This aspect refers to land ability to restock groundwater resources, depending on the structure of the vegetation, the climatic conditions, and layer permeability. Groundwater replenishment is evaluated for each land type, based on annual precipitation, available field capacity, and evaporation. Groundwater replenishment capacity is determined by the groundwater quantity that was prevented from being replenished as a result of the FU over the corresponding time period.
- 5) *Biotic production*. This aspect refers to the biomass prevented from being produced due to the FU. This depends on the type of land use, which can for example be coniferous forest, railway system, mining area, or permanent crops.

2.5.12. Particulate matter (respiratory inorganics)

This impact category quantifies fine particulate matter (PM_{2.5}) pollution, which is associated with respiratory illnesses. PM precursors are nitrogen oxides, sulphur oxides, and ammonia. This impact is measured in disease incidence per amount of PM.

2.6. INTERPRETATION

The interpretation phase is the fourth stage of the LCA process. A key aim of the interpretation phase was to identify issues linked to modelling, and reiterate the other stages to fix them. The stages responsible for the majority of the environmental impact of the system studied were identified, and the sensitivity of the results was investigated with a further analysis (ISO, 2006b). The latter is termed sensitivity analysis, and an additional LCA was run to test the sensitivity of modifying parameters in some stages of the inventory, or changing the allocation approach.

Finally, conclusions and recommendations based on the interpretation of results were built, such as the environmental hotspots of the system studied, and how to possibly modify current practices to mitigate these effects. Identifying the limitations of the model was key, such as data quality and potential effect of assumptions in the model.

2.6.1. Uncertainty in LCA

Uncertainty is present across all phases of a LCA, and can be classified as parameter, model, or scenario uncertainty (Bamber et al., 2020). Parameter uncertainty, or stochastic/data uncertainty, represents uncertainty arising from the fact that the measured data points are part of a population pool with variable data points. It can also be uncertainty due to suboptimal data quality. Scenario uncertainty encompasses the uncertainty due to modelling choices, for example with the selection of a FU or the allocation approach. Thirdly, model uncertainty represents the uncertainty arising from the LCIA methods chosen, which have their own defined characterisation factors.

In this thesis, scenario uncertainty was dealt with through sensitivity analyses that calculated the results with a different modelling approach, such as going from economic to physical and biophysical allocations, using the NDU as a FU instead of the mass-based FU, etc. Parameter and model uncertainties can be dealt with by performing an uncertainty analysis. Monte Carlo (MC) analysis is the most common type of stochastic model used to propagate uncertainty to LCIA results, so that the total uncertainty of parameter and model uncertainties is captured through a confidence interval (Bamber et al., 2020). The MC approach in LCA consists of running several times

the LCIA to obtain the probability of the results distribution. The OpenLCA software has a function to run MC analyses to assess variable data, data quality, and uncertainty stemming from characterisation factors (GreenDelta, 2020). In this thesis, MC analyses were run in Chapters 4-6. Once the MC analysis was run, a modified Null Hypothesis Significance Testing (NHST) was performed, as in Mendoza Beltran et al. (2018), to identify whether results were statistically significant. A hypothesis was statistically tested: “the different production systems delivering product X are associated with different environmental impacts”, with a null hypothesis being that the environmental impacts of the different production systems were equal. A Bonferroni correction of the significance level to exclude false positives was also applied, as $\alpha_b = 0.05/Y$, with Y representing the number of environmental impact categories assessed and the number of alternative products. Threshold values were tested at $\delta_0 = 0.2$.

2.7. ISSUES WITH LCA TOOLS USED: AWARE METHOD MODELLING

A significant modelling issue was found when considering water scarcity within the link of the AWARE method present in the “EF method, adapted” package and Ecoinvent v3.6 (Wernet et al., 2016). In the “electricity production, hydro, pumped storage” processes, a “water GLO” elementary flow was placed in inputs, with a global AWARE characterisation factor, while the water output flow was country-specific, with a regional AWARE characterisation factor. In Ecoinvent 3.5, water characterisation factors evened out (global water flow in both input and output). After personal communication with GreenDelta and Ecoinvent, it was agreed that the characterisation factors were supposed to even out, as in Ecoinvent v3.5. Therefore, all water flows in the “electricity production, hydro, pumped storage” processes were modified manually to have the same input and output characterisation factors (X and -X).

3. SUBSTITUTING WHEAT WITH CHICKPEA FLOUR IN PASTA PRODUCTION DELIVERS MORE NUTRITION AT A LOWER ENVIRONMENTAL COST

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Keywords: Chickpea; Legumes; Life Cycle Assessment; Pasta; Food sustainability; Nutrient Density Unit

3.1. ABSTRACT

The modern food system is characterised by 1) unsustainable agricultural practices, heavily dependent on agrochemical inputs and leaking large amounts of reactive nitrogen (N) whilst degrading soils, and 2) the consumption of energy-rich but nutrient-poor foods, contributing to non-communicable diseases related to malnutrition. Substituting cereals with low-input, protein- and fibre-rich legumes in the production of mainstream foods offers a promising solution to both issues. Chickpea (*Cicer arietinum*) is a leguminous crop that can be grown with little or no synthetic N fertiliser. We performed life cycle assessment (LCA) to compare the environmental footprint of pasta made from chickpeas with conventional pasta made from durum wheat (*Triticum durum*) from cradle to fork. Two functional units were used, an 80g serving of pasta, and a Nutrient Density Unit (NDU). Environmental burdens per serving were smaller for chickpea pasta across at least 10 of the 16 impact categories evaluated. Global warming, resource use minerals and metals, freshwater eutrophication, marine eutrophication, and terrestrial eutrophication burdens were smaller than those of durum wheat pasta by up to 45%, 55%, 50%, 86%, and 76%, respectively. Cooked chickpea pasta contains 1.5 more protein, 3.2 times more fibre and 8 times more essential fatty acids than cooked durum wheat pasta per kcal energy content. Thus, the environmental advantage of chickpea pasta extended to 15 of the 16 impact categories when footprints were compared per unit of nutrition. Global warming, resource use and eutrophication burdens per NDU

were 79–95% smaller for chickpea pasta than for durum wheat pasta. The one major trade-off was land use, where chickpea pasta had a burden 200% higher per serving, or 17% higher per NDU, than wheat pasta. We conclude that there is high potential to simultaneously improve the environmental sustainability and nutritional quality of food chains through simple substitution of cereals with legumes in staple foods such as pasta. Breeding and agronomic management improvements for legumes could reduce the yield gap with cereals, mitigating the land use penalty. Meanwhile, the higher protein content of chickpea pasta could contribute towards wider environmental benefits via animal protein substitution in diets, and merits further investigation. Consumers who look for the traditional taste and texture of wheat pasta can achieve these aspects by cooking the chickpea pasta al dente and combining it with a typical pasta sauce, which will hide its subtle nutty taste.

3.2. INTRODUCTION

In the first chapter, I introduced key challenges of the food sector in Europe: to produce and consume nutritious foods at a low environmental cost, and the role that legumes play to achieve this goal. I also presented the LCA approach as a quantitative decision support tool that can be used to benchmark foods to identify more environmentally and nutritionally sustainable foods. In the second chapter, I presented the full methodology of LCA used across this thesis: its steps, the environmental issues covered, and its limitations. This chapter is the first of the comparative LCAs presented in this thesis. It compares the environmental performance of chickpea pasta versus durum wheat pasta from a serving and nutrient density functional units. It aims to answer the following questions: is chickpea pasta more sustainable from an environmental and nutritional perspective than durum wheat pasta? How does the use of a nutritional functional unit affect the environmental impact of this food product compared to its conventional alternative?

The global food sector faces a major challenge to deliver sustainable nutrition. Intensive agricultural practices adopted to meet growing food demand have driven massive anthropogenic pressures on the Earth's ecosystems, notably via land occupation, fertiliser use and animal-related greenhouse gas (GHG) and ammonia emissions (Steffen et al., 2015). Synthetic Nitrogen Fertiliser (SNF) use causes significant environmental and economic damage, as its production is resource-intensive, and over-application of SNF causes N leaching and ammonia emissions to air, degrading the quality of air, water, and soil (Sutton et al., 2011), changing the climate and promoting biodiversity loss (Mozumder and Berrens, 2007). In terms of cereal production, the use of SNF represents a major source of greenhouse gas emissions to the atmosphere, with typical farm gate values per kg of grain of the order of 0.50 kg CO₂e for oats, wheat and barley (Williams et al., 2020b). To put this in perspective, the cultivation of grain legumes (faba bean, chickpea and pea), where

nitrogen is provided by biological fixation of atmospheric nitrogen by bacteria present in root nodules, produces typical farm gate emission values of the order of 0.18 kg CO₂e (Williams et al., 2020b). Accumulation of this biologically fixed N in plants boosts yields (Peoples et al., 2009), while the use of legumes in agriculture increases biodiversity, reduces weed invasion (Sturludóttir et al., 2014), and can enhance carbon sequestration in the soil (Peoples et al., 2019).

Grain legumes are also more beneficial than cereal grains in term of human health, providing a rich source of nutrients. The risk of type-2 diabetes and cardiovascular diseases decreases with consumption of legumes (Flight and Clifton, 2006; Jayathilake et al., 2018; Kouris-Blazos and Belski, 2016; Viguioliouk et al., 2017), while other positive effects include a reduction in the relative risk of developing myocardial infarction (Miller et al., 2017). The presence of fibre and phytochemicals in legumes lowers cholesterol and helps regulate blood pressure (Bazzano et al., 2011). Moreover, the consumption of legumes improves gut health (Clemente and Olias, 2017) and assists in managing weight (McCrary et al., 2010). Legumes are also a source of anti-cancer peptides (Luna-Vital and González de Mejía, 2018) that can potentially assist in the prevention against prostate (Park et al., 2008) and colorectal cancers (Zhu et al., 2015).

A growing demand for grain legumes for feed and food coupled with supportive policies and yield enhancements in the EU has led to a record production of legume crops in Europe in 2017-2018 (Barel et al., 2018). This demand for pulses, especially chickpeas and lentils is forecast to increase further (Barel et al., 2018). Recently, a growing demand for vegetarian products has been observed across Europe with the rise of flexitarian diets (Derbyshire, 2017; European Commission, 2018b; NatCen, 2016). Launches of vegan and vegetarian food products have increased worldwide by 140% and 21%, respectively (Statista, 2017), and highlight the opportunity to increase legume-based substitutes for staple foods. One such possibility is legume pasta, where substituting durum wheat with pulses in pasta production could dramatically expand legume cultivation and consumption in Europe, with associated environmental and nutritional benefits. Such a substitution could contribute towards crop diversification in the EU and biodiversity restoration (Zander et al., 2016).

Pasta is a popular staple food with high versatility. It is typically made from semolina flour obtained by milling durum wheat, and mixing with water, and sometimes eggs. In Europe alone, 5.4 million tonnes of pasta were produced in 2017 (Eurostat, 2018) and consumption of durum wheat pasta amounted to around 3.4 thousand tonnes per year (Statista, 2019b). Despite the high variability in consumption amongst EU countries, with Italians consuming 23.5 kg of pasta per capita annually, Greeks 11.2 kg, British 3.5 kg, and Irish 1 kg (Statista, 2018) for example, pasta production and consumption are growing worldwide, with a compound annual growth rate of 4.4% projected between 2019 and 2023 (Statista, 2019b).

Numerous pasta products made from ingredients other than durum wheat are appearing commercially. Major categories include other types of wheat pasta (spelt), other cereals pasta (quinoa, rice), 0-calories pasta (konjac), and legume-based pasta (edamame, adzuki, black bean, chickpea, red and or green lentil, and mung bean vermicelli). Legume-based pasta also represents a food opportunity for the 1.4% individuals with celiac disease in the world, due to the legumes pasta being gluten-free (Celiac Disease Foundation, 2018). A sensory analysis of legumes pasta showed consumer acceptability of legumes pasta (Turco et al., 2019). Chickpea pasta can be cooked *al dente*, conserving a similar texture to that of traditional durum wheat pasta, and the difference in taste can be hidden with the sauces typically added to pasta dishes.

In this paper we report results from a ‘farm to fork’ analysis of the environmental burden of both durum wheat and chickpea dry pasta using Life Cycle Assessment (LCA) methodology. LCA is a defined protocol used in assessing the environmental impacts of a product system, by looking at the inputs and outputs of its life cycle (ISO, 2006a). It has been widely used in the food sector, and is a powerful tool to support decision making when considering the sustainability of food systems (Sala et al., 2017). Despite the wealth of LCA data on food systems (Heller et al., 2013), our literature search on Google Scholar (2000-2010) revealed only two LCA studies that have considered the use of legumes in pasta: Chaudhary et al. (2018) estimated the carbon footprint and nutritional content of a partial substitution of refined wheat flour with Canadian yellow pea flour in pasta (30% pea-70% wheat flours), while Nette et al. (2016) performed a comparative LCA of pasta made with egg or pea protein. Nette et al. (2016), like in numerous other food LCAs, disregarded nutrition by using a mass-based functional unit for comparing different foodstuffs, omitting the key nutritional role of the products involved (Heller et al., 2013), and thus potentially supporting misleading conclusions on the wider sustainability of these foodstuffs. On the other extreme, Chaudhary et al. (2018) compared the Nutrient Balance Score of the food products, requiring the knowledge of 32 macro and micronutrients contents, an expensive process that can dissuade LCA practitioners to use a nutritional functional unit. The study also used nutritional content of raw ingredients, which will ultimately change when cooked (Fabbri and Crosby, 2016). Moreover, both studies found that the legume alternative had a lower global warming potential, but did not assess the foods across other impact categories. Product Environmental Footprint (PEF) guidelines (European Commission, 2018a) recommend a more comprehensive evaluation of 16 environmental impact categories when assessing the environmental sustainability of products. Furthermore, most food LCA studies have used mass-based functional units for comparing

The comparative LCA of chickpea and durum wheat pasta reported here has been assessed over the sixteen impact categories recommended by PEF Guidance (European Commission, 2018a), and has two objectives, 1) to compare the environmental burdens of a serving of cooked pasta

using a conventional functional unit of 80g dry weight of pasta, cooked), and 2) to compare the same but use a functional unit based on the nutritional density of the pasta serving, as proposed by Van Dooren (2016).

3.3. MATERIALS AND METHODS

3.3.1. Goal, scope, and boundary definition

This LCA study is a comparative assertion of the overall environmental burden from cradle to fork arising from the consumption of chickpea pasta and conventional durum wheat pasta. The open source software OpenLCA 1.10.2 (GreenDelta, 2020) was used to calculate the environmental footprint of the two pasta products, using Agrifootprint 3.0 (Durlinger et al., 2017) and Ecoinvent 3.6 (Wernet et al., 2016) international databases. Inventory data on chickpea pasta were collected specifically for this study from Variva Ltd. (www.variva.bg, 2019), the Bulgarian manufacturer of chickpea pasta Variva®. Data on durum wheat pasta manufacture were adapted from Bevilacqua et al. (2007) and modelled as though the durum pasta was manufactured in Bulgaria to make the geographical origin of the two products identical.

In order to represent the differing nutritional profiles of the two types of pasta in the LCA analysis, two functional units were used; 80 grams dry weight pasta, cooked (a mass-based FU) and one NDU. The mass-based functional unit is referred to as a serving in this paper. The NDU represents the integration of nutrient density as a comparative basis between the chickpea and wheat pasta in the LCA, such that the environmental burdens are quantified per integrated content of protein, essential fatty acids, fibre, and calories. The reference flow was defined as a cooked portion of pasta. The mass balance flow for both pastas is represented in Figure 3.1.

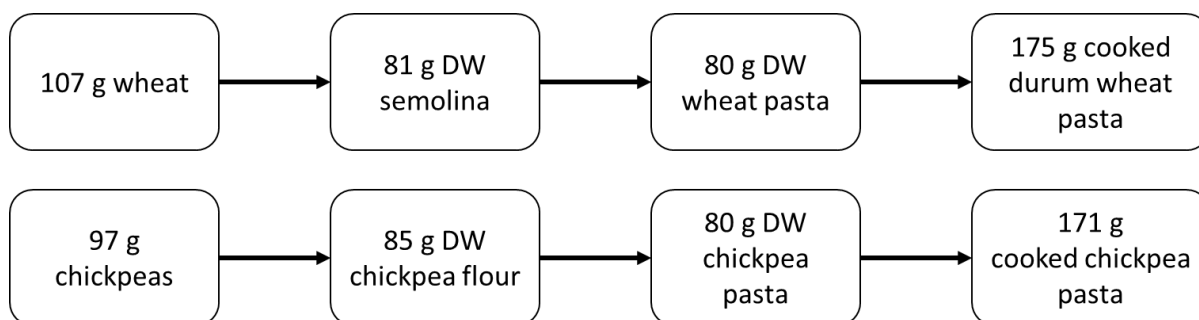


Figure 3.1: Mass balance flow of chickpea and durum wheat pasta.

An attributional LCA was performed with economic allocation for harvested wheat straw and wheat feed co-products from wheat semolina production. Allocation factors were of 92.5% for wheat grains and 7.5% for wheat straw; 84% for semolina and 16% for wheat feed (EPD International, 2016; European Commission, 2018c). Small amounts of biowaste obtained from

chickpea sorting and cleaning were assumed to be discarded in the field, although they could potentially be used as a soil conditioner (which could reduce the environmental burden of chickpea pasta, thus the applied assumption of allocating all burdens to chickpea pasta is conservative). To ensure compatibility between the LCA of durum wheat and chickpea pasta, a second-order approach was used, where the system boundaries included all stages of the life cycle from cradle to fork. Figure 3.2 illustrates both the system boundaries used and manufacturing steps for the cradle to fork assessment of chickpea and durum wheat pasta value chains. In accordance with the ILCD handbook (JRC, 2010), identical product use was considered, the same life cycle stages were included, and the inventory data had roughly matching degrees of accuracy. The LCA methodology followed PEF guidelines (European Commission, 2018a) as far as possible, excluding end of life, in line with the need to harmonise approaches for improved transparency and comparability. The recycling stage was not modelled, as the focus of the study was to compare the differences between pasta made with different raw materials, wheat or chickpea. Therefore, including recycling for the same packaging would not have contributed any useful differentiation in the study.

Results obtained from running the impact assessment of the LCA procedure were normalised by annual person equivalents, using the factors recommended in the PEF guide (European Commission, 2018a). This was done to facilitate interpretation of impact scores by providing a joint reference impact (Benini et al., 2014).

Four scenarios were assessed in this study to compare different assumptions about cultivation of the wheat and chickpeas used to make the pastas:

1) *Wheat (0% straw)* - pasta made from durum wheat with none of the wheat straw harvested during field operations

2) *Wheat (80% straw)* - pasta made from durum wheat with 80% of the wheat straw harvested during field operations, and cultivation burdens economically allocated between grain (for pasta) and straw, with a 7.5% allocation factor for wheat straw and 92.5% for wheat grain (EPD International, 2016)

3) *Chickpea (Bulgaria)* - pasta made from chickpeas using a Bulgarian case study, with 100% chickpea cultivation residues remaining in the field

4) *Chickpea (Spain)* - pasta made from chickpeas using the Bulgarian case study for all steps but cultivation, which was modelled from a Spanish case study, representing chickpea cultivation with no added fertilisers, chemicals/pesticides based on best practice. As for the other chickpea scenario, 100% chickpea cultivation residues remained in the field

Scenarios 2) *Wheat (80% straw)* and 3) *Chickpea (Bulgaria)* are the baseline scenarios, and scenarios 1) *Wheat (0% straw)* and 4) *Chickpea (Spain)* are alternative scenarios. In scenario 2) *Wheat (80% straw)*, 80% was an estimate of the amount of above ground straw residue that is removed during straw harvesting operations, as in Lienhardt et al. (2019).

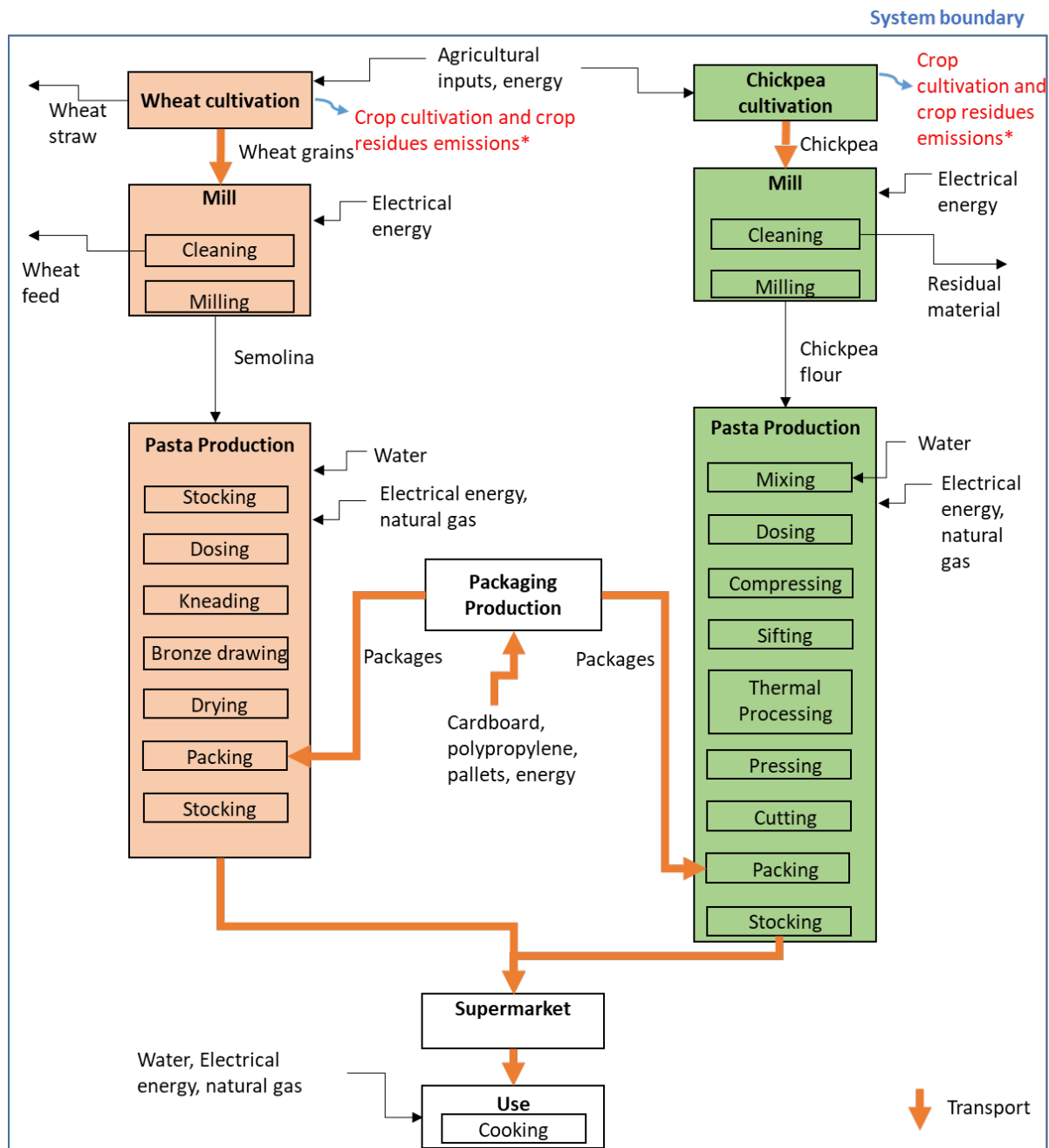


Figure 3.2: System boundary of chickpea pasta, from cradle to fork.

The manufacturing steps of chickpea pasta are indicated in green (right side), those of durum wheat pasta in red (left side), and those shared by the two products are white (middle). For more information on these emissions, see the following section.

3.3.2. Chickpea pasta inventory

Post-farm gate data were provided by Variva Ltd., and cultivation data provided by the main grower supplying Variva Ltd. with chickpea. Chickpea cultivation was modelled with emission factors of the IPCC 2019 guidelines; N content of ground residues were of 0.008 N for below-ground residues, and the same amount for above-ground residues per hectare (IPCC, 2019). Amount of NPK applied per hectare and yield ($1\,820\text{ kg}\cdot\text{ha}^{-1}$ dry matter) were obtained from the farmer working with Variva Ltd. Distribution of fertiliser types specific to Bulgaria was extracted from the International Fertilizer Association (2020). Field activities included from Ecoinvent 3.6 processes were sowing, tillage and ploughing, fertilising, and harvesting. Two applications of fertiliser were assumed. Due to the high variability in crop protection application, the difficulty of finding trustworthy sources that describe which type and what quantities are needed, and the fact that amounts are usually much smaller than those of fertilisers, crop protection was excluded from the study. Lime application was fixed for all scenarios at 400 kg/ha , as a corrector of acidity. This was a conservative approach, as lime is typically applied to prevent soil acidification as a result of the application of ammonium-based fertilisers (Defra, 2010), which are applied in greater amounts in cereal crops than to legume crops.

Direct field emission of nitrous oxide - from crop residues remaining on field and synthetic N fertilisers (SNFs) – were calculated following equation 11.2 of IPCC (2019). Direct emissions of nitrogen oxides resulting from the application of SNF were modelled according to Nemecek and Kägi (2007), while ammonia emissions from the volatilising N fraction of SNF were modelled according to IPCC (2019). Indirect emissions of nitrous oxide resulting from a) the volatilisation of SNF and b) from SNF and on field crop residues leaching were calculated with equations 11.9 and 11.10 of IPCC, respectively (IPCC, 2019). Indirect emissions of nitrate losses to water were determined following Reckling *et al.* (2016), and those of CO_2 emissions from lime and urea applications were calculated according to IPCC (2006). Finally, indirect emissions of phosphorus losses to water from the use of synthetic P fertilisers were determined with the cropping system loss coefficients, as in Styles *et al.* (2015).

Chickpeas were harvested and transported at most 50 kilometres to the cleaning facility. The sorting machine consumed 40 Wh per kg , and 5-10% of the material was separated as biowaste. The remaining 90-95% of clean chickpeas were then thermally processed in a 1000 kg capacity oven, transported 150 kilometres to the milling plant where the obtained flour was mixed with water, pressed, cooked in a pasta oven, and finally packaged.

The fertiliser applications in the Bulgarian scenario, *Chickpea (Bulgaria)*, were likely used for boosting yields, though chickpea cultivation guides indicate that high use of nitrogen fertilisers is needless, and may be detrimental to chickpea growth by inhibiting nodulation and atmospheric

nitrogen fixation (Corp et al., 2004). Moreover, in some areas, use of phosphorus fertiliser is not required (Corp et al., 2004; GRDC, 2012). Consequently, an additional scenario, *Chickpea (Spain)*, was modelled, using yields from a Spanish field experiment for which no fertiliser, insecticides or pesticides were added. High variability of yields was observed in the Spanish case study, ranging from 0 to 3520 kg.ha⁻¹ chickpeas dry matter across the sixteen plots studied. This high variance was due to fungal pathogen ascochyta blight. The average yield, 2014.5 kg.ha⁻¹ dry matter was selected for the *Chickpea (Spain)* scenario.

The amount of energy and volume of water used for cooking were determined by adapting data from PEF guidelines for dry pasta (European Commission, 2018c) and information provided by Variva® pasta (variva.bg, 2018). The energy required for cooking for chickpea pasta is less than wheat pasta, due to the shorter cooking time required (6 minutes versus 11). Adding salt was not mentioned in the chickpea pasta cooking recommendations. However to maintain a consistent comparison, it was assumed that the same amount of salt was added in cooking both products. Electric cooking was modelled with a flow specific to Bulgarian electricity. All of the cooking water was assumed to go to residential wastewater treatment. Information on packaging was provided directly by Variva Ltd.

3.3.3. Durum wheat pasta inventory

Data for durum wheat pasta cultivation was obtained from the same farm in Bulgaria that supplies the chickpeas, using a combination of Agrifootprint 3.0 and Ecoinvent 3.6 processes. Data for durum wheat pasta manufacturing was adapted from Bevilacqua et al. (2007) using Ecoinvent 3.6 processes. Inputs and outputs for all processes involved in the delivery of an 80 g dry weight serving of *al dente* – a cooking technique in which pasta is still firm - cooked pasta, from cradle to fork, were recorded and are included in full in Table 3.1. Relevant background processes for chickpea and durum wheat pasta production were extracted from LCA databases Ecoinvent 3.6 and Agrifootprint 3.0.

Agricultural data for both wheat and chickpea production came from the same farm in Bulgaria. Amount of NPK applied per hectare and yield (5 340 kg ha⁻¹ dry matter) for wheat were obtained from the farmer working with Variva Ltd. As for chickpea cultivation, wheat cultivation was modelled with emission factors of the IPCC 2019 guidelines based on N contents of above- and below-ground residues, with 0.006 and 0.009 kg N per kg residue, respectively (Liang and Noble, 2019). The breakout of fertiliser types specific to Bulgaria was extracted from the International Fertilizer Association (2020) since the specific fertilisers used in the farm assessed was unknown; one application of fertiliser was assumed along with 400 kg of lime per hectare. Data were collated

for the two wheat scenarios, 1) *Wheat (0% straw)* and 2) *Wheat (80% straw)*. Following Product Category Rules (PCR) for arable crops, economic allocation was performed for allocating the burdens between wheat grain and wheat straw during the cultivation phase (EPD International, 2016). Wheat straw and wheat grain co-products were assigned 7.5% and 92.5% of cultivation environmental burdens respectively.

Durum wheat is transported after cultivation to a milling facility. Milling uses 70 Wh of electricity, 10 J of natural gas, and 0.02 L of water for 110 g of durum wheat, producing 80 g of semolina, and 30 g of co-product used as cattle feed. The semolina is transported to another facility where it is dosed, kneaded (using 0.02 L of water per 80 g of semolina), bronze drawn (shaped), and then dried. These manufacturing steps account for 3.02 Wh of heat gas and 13 Wh of electricity, 80 J of natural gas and 60 J of crude oil for the FU. The dried pasta is packaged in cardboard boxes of 250 g and transported to a retail centre and subsequently to the consumer. Transport modes, distances, and weights of products transported within Bulgaria, within Europe, and outside Europe, were assumed to be the same for durum wheat pasta and chickpea pasta so as to avoid any false differentiation between pasta types based on factors independent of the main flour type. Transport modes and distances were modelled following the PEF guidelines (European Commission, 2018a). Based on 2018 data provided by Variva Ltd, 20% of Variva® chickpea pasta produced is sold within Bulgaria, 20% is sold in Europe (excluding Bulgaria), and the remaining 60% is sold in Turkey. The distance between the centroids of Bulgaria and Turkey was determined to be 1162 km with Google Maps. Based on the PEF guidelines, consumer transport to the supermarket included product volume in the equation to determine the share of the product in the shopping environmental burdens. 80 grams of raw pasta are then cooked in 0.8 L of boiling water for 11 minutes with 5 grams of salt (Barilla, 2018; European Commission, 2018c).

Table 3.1: Inventory of inputs and outputs for an 80 g dry pasta serving of al dente cooked chickpea or wheat pasta.

*Depending on whether wheat straw was harvested or not, with 148 g belonging to scenario 2) and 0 to scenario 1)

Stage	Input/output/process	Units	Wheat		Chickpea (Bulgaria)		Chickpea (Spain)	
			Input	Output	Input	Output	Input	Output
Cultivation	Fertiliser –N	kg	0.0040		0.0016			
	Fertiliser – P ₂ O ₅	kg	0.0025		0.0052			
	Fertiliser – K ₂ O	kg	0.0024		0.0077			
	Urea	kg	0.0021		0.0009			
	Lime	kg	0.008		0.008		0.008	
	SO ₃	kg	0.001					
	Energy, diesel burned in machinery	MJ	0.13		0.04		0.04	
	Seed	kg	0.004		0.004		0.004	
	Land	m ²	0.20		0.53		0.48	
	Chickpeas (dry matter)	g				97		97
	Wheat straw	g		148 / 0*				
	Wheat grain (dry matter)	g		110				
Flour production	Transport 16-32 t lorry (200km)	kg.km	21.27		18		18	
	Electricity	Wh	70		17.4		17.4	
	Organic residual material	kg				0.005		0.005
	Chickpea flour	kg				0.08		0.08
	Water	L	0.0200		0.0200		0.0200	
	Natural gas	J	10					
	Semolina	kg		0.08				
	Wheat grain, feed	kg		0.03				
Pasta production	Machines electricity	Wh	13		16		16	
	Water	L	0.0200		0.0242		0.0242	
	Heat, other than nat. gas	Wh	3.02		19.73		19.73	
	Nat gas- thermal energy	MJ	0.08					
	Heat and power co-generation, oil	MJ	0.06					

Table 3.1 (continued): Inventory of inputs and outputs for an 80 g dry pasta serving of al dente cooked chickpea or wheat pasta

Stage	Input/output/process	Units	Wheat		Chickpea (Bulgaria)		Chickpea (Spain)	
			Input	Output	Input	Output	Input	Output
Packaging	Film, low density PET	g	1.6		1.6		1.6	
	Kraft paper, unbleached	g	4.5		4.5		4.5	
	Folding boxboard	g	4.5		4.5		4.5	
	Polypropylene, granulates	g	0.0029		0.0029		0.0029	
	Extrusion, plastic film	g	0.0029		0.0029		0.0029	
	Trans PP film >32 t lorry	kg.km	7.40E-4		7.40E-4		7.40E-4	
	Corrugated board box production	g	3.7		3.7		3.7	
	Flat pallet	Unit	8.53E-6		8.53E-6		8.53E-6	
	Transport of boxes >32t lorry	kg.km	2.11		2.11		2.11	
	Packaging electricity	Wh	1.2		1.2		1.2	
	Printing ink, offset	g	0.0149		0.0149		0.0149	
Transport factory- retail-	Transport factory-DC (BG) lorry 3.5-7.5	kg,km	19.2		19.2		19.2	
	Transport factory-DC (EU) lorry 16-32 t	kg,km	56.0		56.0		56.0	
	Transport factory-DC (outside EU)	kg,km	55.8		55.8		55.8	
	Consumer transport by car	km	1.56E-7		1.56E-7		1.56E-7	
Cooking	Tap water	L	0.80		0.80		0.80	
	Boiling and cooking electricity	kWh	0.694		0.394		0.394	
	Salt	g	5		5		5	

Durum wheat and semolina are very similar in terms of energy and protein content (USDA, 2020). Therefore, we assumed that co-products of semolina production were used as wheat-based feed substitutes. Following the product category rules of uncooked pasta, economic allocation was performed for the milling stage, and the co-products semolina and wheat for feed were assigned 84% and 16% of upstream burdens respectively (EPD International, 2010). Because co-products were in small quantities, and because the wheat feed co-products from semolina production are likely to largely replace similar wheat-derived feeds, system expansion was deemed not relevant to perform in this case. For simplification, apart from stated co-products and waste flows, no loss was assumed during any of the life cycle stages of either product. 80 g of dried pasta was used as the reference flow, assuming the same packaging for both pasta products based on packaging information collected from Variva Ltd.

3.3.4. Nutrient Density Unit (NDU)

A key function of food is to deliver nutrition to the body. LCAs of food products which are based solely on mass FUs do not take this into account, and make inter-food comparisons difficult. LCAs which use a protein FU are still problematic in that nutrition is far more complex than just protein delivery, and that in Europe, daily protein intake is above recommended levels (Westhoek H. et al., 2016). Therefore, in this study, the Nutrient Density Unit (NDU) developed by Van Dooren (2016) was selected a good proxy (see Discussion).

Nutritional composition of both cooked pastas was obtained by nutritional analysis of Variva® chickpea pasta and of durum wheat pasta from the supermarket. Analyses of protein using the Kiedahl method (ISO 1871:2009) (ISO, 2009), energy following the EU regulation 1169/2011 (European Union, 2011), fibre by Enzymatic-Gravimetric Method from the AOAC 991.43 and AOAC 985.29 (Lee et al., 1992; Prosky et al., 1985), and essential fatty acids through gas chromatography (FID) from ISO 12966-1:2014; 12966-2:2011; 12966-3:2016 (ISO, 2016, 2014, 2011) were performed. Random replicates were performed to 7% of the analysis by a credited laboratory that follows standard, verified, and certificated protocols, and that work mostly with industry. Nutritional characteristics of the two pastas are summarised in Table 3.2.

The Nutrient Density Unit (NDU) was applied following Van Dooren's (2016) formula (1).

$$NDU = \frac{\left(\frac{EFA}{DV_{EFA}}\right) + \left(\frac{Protein}{DV_{prot}}\right) + \left(\frac{Fibre}{DV_{fibre}}\right)}{3 \times \left(\frac{S_i}{2000 \text{ kcal}}\right)} \quad (\text{Eq. 1})$$

Where:

EFA is the amount of essential fatty acids in 100g of product, expressed in grams.

Protein is the amount of protein in 100g of product, expressed in grams.

Fibre is the amount of fibre in 100g of product, expressed in grams.

DV_{EFA} is the recommended daily value intake of essential fatty acids, expressed in grams.

DV_{prot} is the recommended daily value intake of protein, expressed in grams.

DV_{fibre} is the recommended daily value intake of fibre, expressed in grams.

S_i is the amount of kilocalories in 100g of product, expressed in kilocalories.

Table 3.2: Energy, protein, dietary fibre, and essential fatty acids content of 80 g dry weight durum wheat and chickpea pasta, cooked.

	Durum wheat pasta	Variva® pasta chickpea
Energy (kcal)	144	150
Protein (g)	5.3	8.1
Dietary fibre (g)	1.8	5.7
EFA (g)	0.2	1.6

$$NDU_{variva\ pasta(cooked)} = \frac{\frac{1.6}{12.4} + \frac{8.1}{50} + \frac{5.7}{25}}{3 \times \frac{150}{2000}} = \mathbf{2.3} \quad (\text{Eq. 2})$$

$$NDU_{wheat\ pasta(cooked)} = \frac{\frac{0.2}{12.4} + \frac{5.3}{50} + \frac{1.8}{25}}{3 \times \frac{144}{2000}} = \mathbf{0.90} \quad (\text{Eq. 3})$$

The energy content of the two products does not differ greatly, with 144 kilocalories per 80 g dry weight durum wheat pasta, cooked versus 150 kilocalories per 80 g dry weight chickpea pasta, cooked. The protein content of chickpea pasta is 1.5 times higher than that of durum wheat pasta (8.1 g and 5.3 g for chickpea and durum wheat pasta, respectively). The EFA content of chickpea pasta is 8 times higher than that of durum wheat pasta (1.6 g and 0.2 g for chickpea and durum wheat pasta, respectively). The fibre content is 3.2 times higher in chickpea pasta (5.7 g and 1.8 g for chickpea and durum wheat pasta, respectively). Applying these data to NDU equations (2) and

(3) shows that chickpea pasta is 2.6 times more nutrient dense than durum wheat pasta overall – delivering 2.3 vs 0.90 NDU.

3.3.5. Impact Assessment

The environmental burden of the four scenarios was assessed using sixteen impact categories from the PEF recommended methodology (European Commission, 2018a). Impact indicator data were normalised according to PEF guidelines as person equivalents.

3.4. LIFE CYCLE IMPACT ASSESSMENT RESULTS

Table 3.3 lists the derived environmental burdens across the sixteen impact categories. Results for each of the two functional units are shown. For both functional units, scenario 4) chickpea pasta with the Spanish agronomic data has a smaller environmental burden across most categories when compared to the other 3 scenarios, and scenario 1) *Wheat (0% straw harvest)* has the highest burdens overall. Per serving, chickpea pasta from the Bulgarian case study (scenario 3) has a smaller environmental burden across 10 of the 16 environmental impact categories. However, chickpea pasta from the Bulgarian case study (scenario 3) requires more than twice the arable land than durum wheat pasta, with 0.53 m².yr versus 0.20 m².yr, thus has a land use burden that is between 1.8 and 2.0 times the one of durum wheat pasta (Table 3.3). This is due to significantly lower yields for chickpea than for wheat.

Per serving, chickpea pasta from the Spanish case study (scenario 4) has a smaller environmental burden across 13 of the 16 environmental impact categories when compared to both wheat pasta scenarios (scenarios 1 and 2). For the same weight of pasta, the acidification, climate change, marine eutrophication, terrestrial eutrophication and water scarcity of durum wheat pasta (scenarios 1 and 2) is between 0.7 and 6 times higher than chickpea pasta from the Spanish case study (scenario 4), respectively. However, chickpea pasta from the Spanish case study (scenario 4) still requires more than twice the arable land use than durum wheat pasta (scenarios 1 and 2), with 0.48 m².yr versus 0.20 m².yr, respectively (Table 3.1), and thus has a land use burden that is up to 1.7 higher than the one of durum wheat pasta (Table 3.3).

When one NDU is used as the FU, the comparative environmental efficiency of chickpea pasta is improved further (Table 3.3). Chickpea pasta from the Spanish case study (scenario 4) generates smaller environmental burdens than durum wheat pasta per unit of nutrient density across all impact categories, except for the land use one, for which scenario 2) *Wheat (80% straw)* has the lowest burden of all scenarios. For scenario 3) *Chickpea (Bulgaria)*, all environmental burdens are

smaller than those of durum wheat pasta per unit of nutrient density except for land use, which is 8-15% smaller for durum wheat pasta. The differences amongst the other impact categories are magnified, with one NDU from wheat pasta generating between 2 and 3 times more climate change, terrestrial eutrophication, and water scarcity burdens than one NDU from chickpea pasta derived from Bulgarian chickpeas, and between 3 and 10 times more in the same categories than one NDU from chickpea pasta derived from Spanish chickpeas. Therefore, to achieve the same nutrition, chickpea pasta from both scenarios (3 and 4) has a considerably lower overall environmental burden than durum wheat pasta (scenarios 1 and 2), with a small land use trade-off.

Table 3.3: Summary of environmental burdens for durum wheat pasta and chickpea pasta, expressed per serving and per Nutrient Density Unit.

Impact category	Unit	Impact per serving				Impact per Nutrient Density Unit			
		Wheat (0% straw)	Wheat (80% straw)	Chickpea (Bulgaria)	Chickpea (Spain)	Wheat (0% straw)	Wheat (80% straw)	Chickpea (Bulgaria)	Chickpea (Spain)
Acidification ter. & freshwater	mol H ⁺ eq	0.00208	0.00198	0.00132	0.00067	0.00289	0.00275	0.000717	0.000419
Cancer human health	CTUh	1.89E-09	1.86E-09	2.23E-09	1.85E-09	2.62E-09	2.59E-09	1.21E-09	1.00E-09
Climate change	kg CO ₂ eq	0.207	0.196	0.163	0.114	0.287	0.272	0.088	0.062
Ecotoxicity freshwater	CTUe	0.217	0.214	0.293	0.254	0.301	0.298	0.159	0.138
Eutrophication freshwater	kg P eq	0.00019	0.00019	0.00012	9.55E-05	0.00026	0.00026	0.000065	0.000052
Eutrophication marine	kg N eq	0.00133	0.00111	0.00104	0.00019	0.00185	0.00154	0.00057	0.0001
Eutrophication terrestrial	mol N eq	0.00627	0.00587	0.00388	0.00153	0.00871	0.00815	0.00211	0.00083
Ionising radiation, HH	kBq U ⁻²³⁵ eq	0.0424	0.0423	0.0243	0.023	0.0589	0.0588	0.0132	0.0125
Land use	Point	24.3	22.5	67.4	60.6	33.7	31.3	36.7	32.9
Non-cancer human health	CTUh	3.50E-08	3.43E-08	7.54E-08	6.43E-08	4.86E-08	4.77E-08	4.10E-08	3.50E-08
Ozone depletion	kg CFC11 eq	1.53E-08	1.51E-08	1.37E-08	1.14E-08	2.12E-08	2.09E-08	7.43E-09	6.17E-09
Photochem. ozone form.	kg NMVOC eq	0.00048	0.00047	0.00052	0.00042	0.00067	0.00065	0.00028	0.00023
Resource use, energy carriers	MJ	2.55	2.53	1.9	1.62	3.54	3.51	1.03	0.883
Resource use mins. & metals	kg Sb eq	1.36E-09	1.26E-09	2.12E-09	6.16E-10	1.89E-09	1.75E-09	1.15E-09	3.35E-10
Respiratory inorganics	disease inc.	1.70E-08	1.63E-08	1.12E-08	6.30E-09	2.36E-08	2.26E-08	6.06E-09	3.43E-09
Water scarcity	m ³ depriv.	0.37	0.368	0.259	0.221	0.514	0.511	0.141	0.12

From smallest (1) to largest (4) environmental impact:

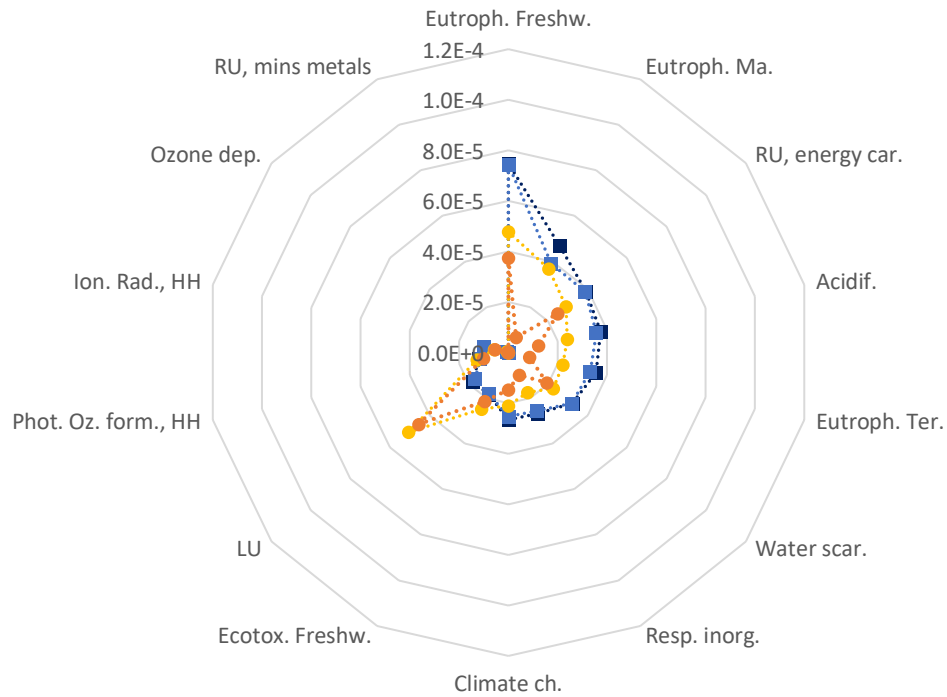


When comparing the differences between the chickpea pasta scenarios and scenario 1) *Wheat (0% straw)* with the differences between the chickpea pasta scenarios and scenario 2) *Wheat (80% straw)*, no significant change in the pattern of results was observed. Straw harvest not only “dilutes” the environmental burden of grain production, but reduces the quantity of straw residues in the field that give rise to nitrate leaching to water. In the long term, straw incorporation may increase soil organic carbon, though this effect is dependent on factors such as average C:N ratio of soil organic matter, climate, soil type, etc. and is beyond the scope of this study.

Figures 3.3.A. and 3.3.B. illustrate the environmental burdens normalised per person equivalents as described in Section 2. Figure 3.3.A. refers to the environmental burdens per serving. Here, we can see the same picture of environmental impact differences between the two products than in the first two result columns of Table 3.3. Both pasta types contribute relatively more to the global burdens of freshwater eutrophication, and chickpea pasta to land use. The contribution of one serving of wheat pasta to an average person’s annual freshwater eutrophication footprint is nearly times higher than the contribution to an average person’s climate change footprint. For a serving of chickpea pasta, ~~from~~ it is around 2.5 times higher. Concerning land use, the contribution of one serving of chickpea pasta from Bulgaria to an average person’s annual land use footprint is 2 times higher than the contribution to an average person’s climate change footprint. For a serving of chickpea pasta from Spain, it is 3 times higher.

Figure 3.3.B. refers to the environmental burdens per NDU. This graph represents the same broad picture as the last four result columns of Table 3.3, but in person equivalents. Per NDU, chickpea pasta (scenarios 3 and 4) has a lower environmental burden than durum wheat pasta (scenarios 1 and 2) across all fourteen impact categories displayed in Figure 3.3.B, excepted for land use, for which the impact is similar. The gap in every other category is greatly widened when comparing with Figure 3.3.A.

3.A.



3.B.

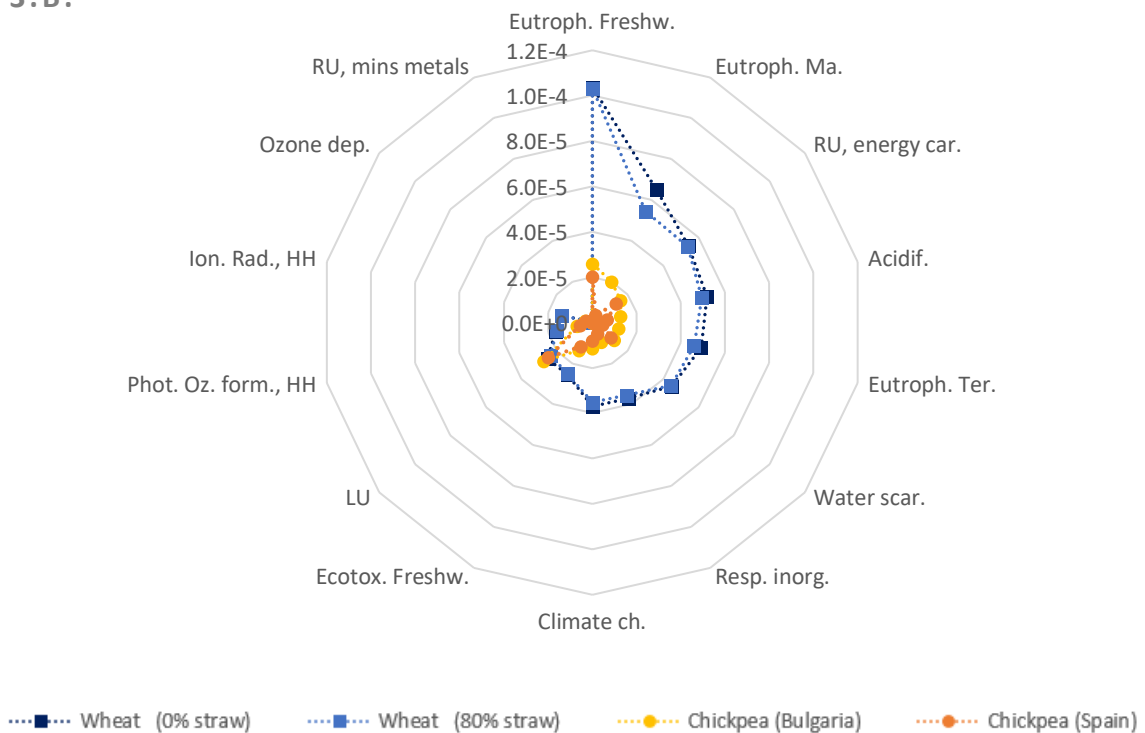


Figure 3.3: Environmental burden of cooked chickpea and durum wheat pasta across 13 impact categories, using as a FU: **A.** weight: serving **B.** Nutrient Density Unit.

Human toxicity-related impact categories were excluded from the graphs, following PEF recommendations (JRC, 2018). Results are normalised per person equivalents.

A second aim of attributional LCA is to identify improvement opportunities within a product life cycle. Hotspots in the production chain of chickpea pasta from scenario 3) *Chickpea (Bulgaria)*

were identified by examining the burden of each life cycle stage for all impact categories, and were recorded in process contribution graphs (Figure 3.4). Scenario 2) *Wheat (80% straw)* process contributions are presented as a comparison.

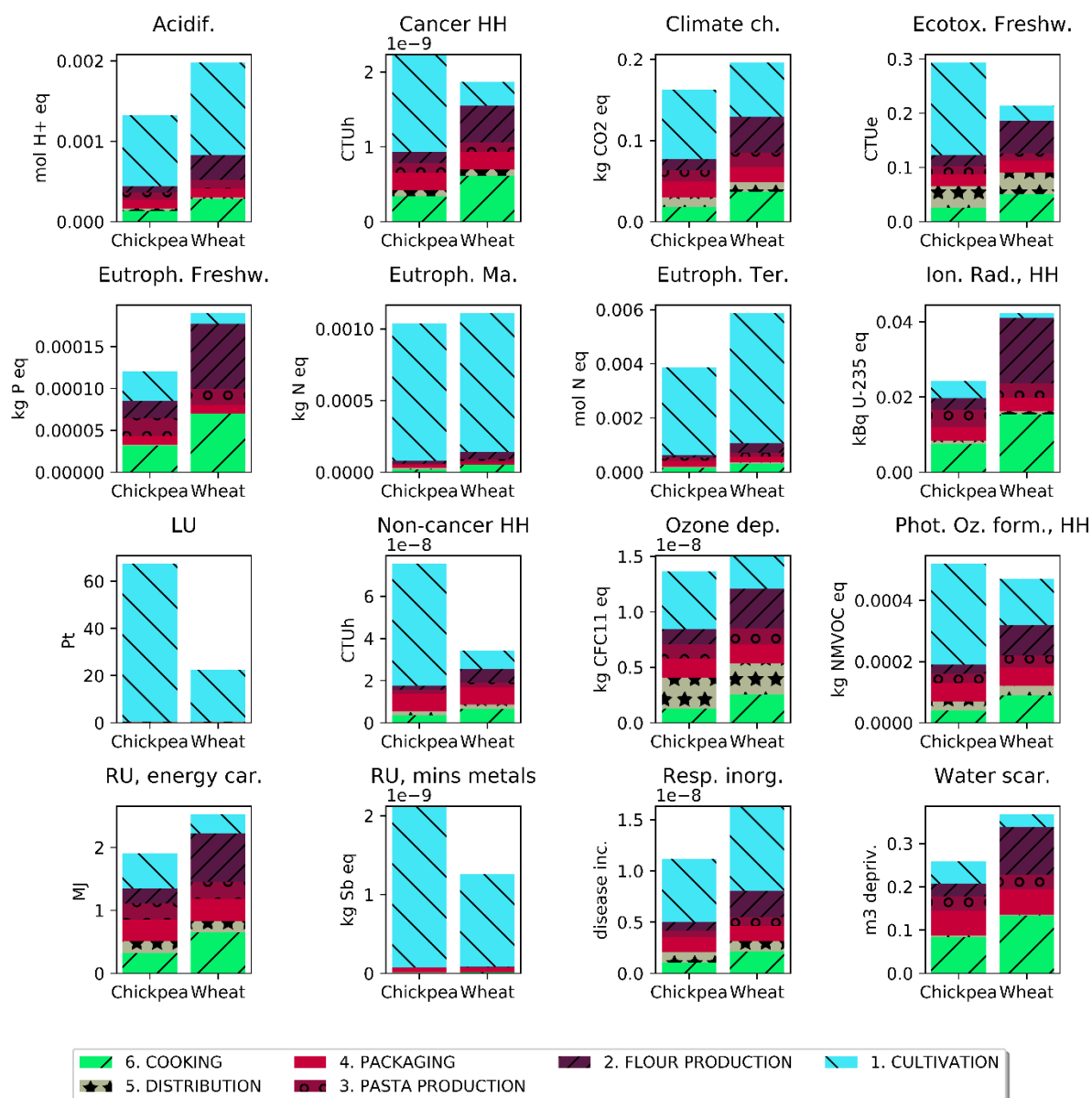


Figure 3.4: Process contributions of scenario 3) Chickpea (Bulgaria) and scenario 2) Wheat (80% straw) across 16 impact categories.

Agricultural stage

The agriculture stage is the major hotspot of the life cycles of both pastas, contributing to at least 20% of total burdens in 15 out of 16 impact categories for scenario 3) *Chickpea (Bulgaria)*, and 10 out of 16 categories for scenario 2) *Wheat (80% straw)*. The agricultural stage in scenario 3)

Chickpea (Bulgaria) is responsible for more than half of the total GHG, acidification, toxicity, eutrophication marine and terrestrial, land use, photochemical ozone formation, minerals and metals use, and respiratory inorganics emissions of the whole life cycle. A significant proportion of these burdens is due to fertiliser production and application. Acidification was mainly caused by ammonia emission from N fertiliser application. GHG emissions from agriculture are caused mainly by emission of dinitrogen monoxide, mainly from direct N₂O emissions arising from crop residue and fertiliser N inputs. Marine and terrestrial eutrophication is mostly caused by nitrate emission to water and air, from applied N fertilisers and residues.

Chickpea cultivation causes scenario 3) *Chickpea (Bulgaria)* to have a higher environmental burden than durum wheat pasta across the resource use minerals and metals, land use, photochemical ozone formation, and toxicity-related categories. Around a third of the freshwater ecotoxicity burden of scenario 3) *Chickpea (Bulgaria)* comes from the production of chickpea seeds. In the land use category, chickpea cultivation has a higher burden because of its much lower yields than durum wheat, with chickpea yields of 1 820 kg DM/ha and 2 015 kg DM/ha in the Bulgarian and Spanish case studies, respectively, compared to wheat yields of 5 340 kg DM/ha. On the other hand, wheat cultivation is responsible for the higher environmental burden of durum wheat pasta compared with chickpea pasta in the terrestrial eutrophication and respiratory inorganics categories. This is mainly due to wheat cultivation releasing ammonia and nitrogen oxides to the air from use of N fertilisers.

Cooking

Cooking was identified as the second major hotspot behind cultivation, contributing to more than 20% of total burden across 3 impact categories in scenario 3) *Chickpea (Bulgaria)* and across 6 in scenario 2) *Wheat (80% straw)*. In the Climate Change category, cooking is responsible for nearly 20% of the total burden of 2) *Wheat (80% straw)* mainly due to electricity usage, which is higher than that of cooking chickpea pasta, due to the comparatively longer cooking time. Electricity production and the treatment and distribution of tap water in Europe are responsible for the water scarcity burdens from cooking.

Milling

The high burden of freshwater eutrophication of scenario 2) *Wheat (80% straw)* is due to the high use of electricity in milling. The process “market for electricity, medium voltage- Bulgaria” from Ecoinvent 3.6 was used. This mix relies mainly on lignite, which also remains important in countries

such as Germany and Poland. The freshwater eutrophication burden of the Bulgarian electricity mix is around 4.2 times higher than the UK's, and around 1.5 times lower than Germany's. Therefore, this hotspot is geographic-dependent, and the burden should significantly decrease once EU countries move away from polluting coal and lignite fuel sources in response to climate change policy obligations.

3.5. DISCUSSION

3.5.1. Crop type

Environmental sustainability is affected across multiple dimensions (Steffen et al., 2015), therefore assessing products across multiple impact categories is critical to provide a more precise and holistic indication as to which mitigation options should be adopted to improve the overall sustainability of food systems.

Owing to the importance of the cultivation stage, uncertainty in chickpea yield has a major influence on environmental footprints of chickpea pasta. Comparatively low yields for chickpeas mean that modest per hectare inputs translate into relatively high burdens on a mass basis for the final pasta. Relatively little research has been undertaken on yield improvement in legumes when compared to cereals such as wheat and barley (van Loon et al., 2018). The reason for little improvement in chickpea productivity can be attributed to their subordinate position in consumer likings after cereals, volatile prices due to irregular yields and lack of government incentives as opposed to cereals (Merga and Haji, 2019). There is considerable scope for yield improvement in chickpea cultivation, and therefore potential for the environmental footprint of chickpea pasta to reduce further relative to durum wheat pasta.

Limited available evidence suggests that fertiliser usage is still common in chickpea cultivation, despite research and agronomy guides stating that the practice of N application (and P application in some cases) is not necessary (Corp et al., 2004; Gan et al., 2009; GRDC, 2012). On the European scale, reducing or eliminating fertiliser application will reduce environmental degradation across all impact categories, as shown with 4) *Chickpea (Spain)*. For example, Climate Change will decrease from 0.16 kg CO₂ eq. to 0.11 kg CO₂ eq. per serving. An additional benefit of not applying fertiliser is that costs of production would decrease – the cost of SNF is increasing (Abi-Ghanem et al., 2012; Saghir Khan et al., 2007). The alternative scenario, 4) *Chickpea (Spain)*, used data from existing plots in Spain, showing the feasibility of such cultivation methods. However, even under plot trials, yield variability was very high. Therefore, it is crucial to educate farmers on proper chickpea cultivation practices, and on cultivation of legumes in general, to achieve sustainability

from environmental, economical, and social perspectives. More breeding is also important to find varieties that produce more reliable yields with improved resilience.

3.5.2. Impact study (Nutrient density functional unit)

The advantage of the NDU as a FU as opposed to more extensive nutrient indexes lies in its simplicity whilst maintaining much of the nutritional differentiation achieved by the latter indexes (Van Dooren, 2016). It requires only three 'nutrient' inputs - energy, essential fatty acid, protein and fibre content - and is an excellent proxy for more nutrient data demanding density indices, such as the NRF12:3 nutrient rich food index of (Drewnowski and Fulgoni, 2008), which requires 15 'nutrient' data inputs per food item. To illustrate the convenience of NDU above more conventional NRF indices, Figure 3.5 adapted with permission from Williams et al. (2020), shows the correlation of NDU with NRF12:3 for a total of 55 separate food items spanning 11 food groups ($r^2 > 0.64$).

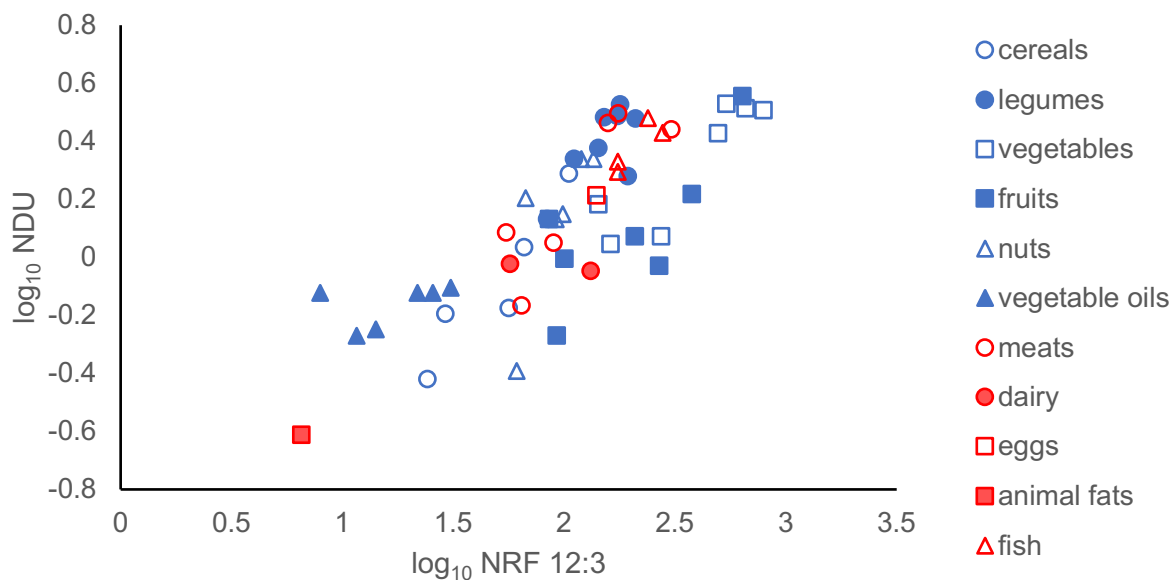


Figure 3.5: Correlation of the Nutrient Density Unit with the Nutrient Rich Foods 12:3 for different food categories.

The three macronutrient groups present in the NDU, essential fatty acids, protein, and fibre are essential for the human diet. The two groups of dietary essential fatty acids, linoleic (omega-6) and alpha-linolenic (omega-3) cannot be synthesised by the human body (Di Pasquale, 2009). The main source of omega-3 is in fish and flaxseed oil, while vegetable oils are the main source of omega-6 (Di Pasquale, 2009). These nutrients are crucial for proper growth as constituents of cell membranes (Simopoulos, 1999). Dietary fibre denotes the dietary constituents that mammalian enzymes cannot degrade (Bach Knudsen, 2001). It is a key nutrient, decreasing the risk of obesity and type 2 diabetes (Brennan, 2005; van Dooren et al., 2014), cancer and heart disease (Kendall et

al., 2010). A diet high in fibre also decreases bowel ailments, including constipation through increased faecal substance (Wood and Grusak, 2007). Dietary protein is key to achieve body growth and protein maintenance, and net protein utilisation from animal or plant sources is comparable among adults (WHO, 2007). Consuming chickpea pasta instead of durum wheat pasta has a clear nutritional benefit in terms of fibre, essential fatty acids, and protein, resulting in considerably higher NDU values. Diets in developed countries are too high in omega-6 fatty acids and too low in omega-3 fatty acids (Simopoulos, 2016). Therefore, an additional analysis should be made to see the content of different essential fatty acids in the two products to determine the ratio of omega-3 to omega-6. One could argue that a higher amount of protein in pasta is unnecessary, as the average European consumer already consumes considerably more than the daily recommended protein intake (Westhoek et al., 2011), and that consumers will not reduce the amount of protein consumed from other (higher burden) sources, leading to wasteful protein intake. However, increasing plant protein intake to substitute animal-based protein is a shift that is crucial for environmental and health objectives (Willett et al., 2019). In addition, the presence of viscous fibre in foods is linked to the achievement of satiety (Slavin and Green, 2007), preventing over-eating, suggesting that chickpea pasta could contribute towards reducing excess calorie consumption in industrialised countries. Ideally, chickpea pasta would substitute other sources of protein, such as meat, dairy and eggs, further reducing the environmental burdens. In their meta-analysis of food products from cradle to retail, Poore & Nemecek (2018) showed that the carbon footprint of pulses versus meat products per kg protein is significantly lower, with the 10th percentile of 100g protein ruminant meat, pork, and poultry being 5, 2.5, and 1.3 times higher than the 90th percentile of pulses other than peas. Therefore, our conclusions here are likely to be conservative, and the aforementioned wider effects of substituting wheat with chickpeas in pasta production require further investigation.

Because the *al dente* texture of legume pasta is possible, and pasta is commonly eaten with sauces, the subtle taste of chickpea pasta can easily be masked if desired. An additional incentive for consumers to opt for legume pasta is the high nutritional value, promoting a healthy diet with weight management due to high fibre content. Finally, a third incentive is the comparatively shorter cooking time of chickpea pasta, which suits a market that takes little time to prepare food.

The PEF guidelines for pasta do not consider any nutritional aspects, recommending a simple mass-based FU as the only FU to be used (European Commission, 2018c), potentially leading to incomplete footprint labelling from a nutritional perspective. We recommend that PEF guidelines incorporate a nutritional FU as an optional alternative to the mass-based FU, in order to support more accurate comparison of different foods.

3.5.3. Limitations

Processing legumes into “staple” foods such as chickpea pasta increase legume cultivation in EU rotations while encouraging diet change, at least from cereals to legumes and, ideally, from meat to legume protein. This study did not include the benefits of N carryover legumes provide to the following crops in rotations. This is a conservative choice that favours wheat pasta, and enables attributional footprints to be compared. LCAs with legume cropping systems should account for these benefits.

In Bevilacqua et al.'s (2007) study, it was only mentioned that compost and feed were co-products of milling. We adopted a conservative approach in this study, assuming that all co-products of milling (24%) went to cattle feed, as this has a higher value than compost. An economic allocation was performed for the co-products of semolina production and sorting of chickpeas as recommended by the PEF guidelines for dry pasta (European Commission, 2018c). This allocation method is limited by price fluctuation over time due to changes in demand, leading to changes in allocation factors (Nijdam et al., 2012).

The application of crop protection agents was excluded in all scenarios. Pesticides are often a hotspot in agricultural LCAs (Zortea et al., 2018), therefore the real impact of both types of pastas should be higher, excepted for scenario 4) *Chickpea (Spain)*, in which no pesticides or chemicals were applied in reality. However, chickpea seed production, which contributed significantly to several environmental impact categories including ecotoxicity, is a generic background process for chickpea seed production in the United States taken from Agrifootprint 3.0. These high burdens are in part due to fungicide application, releasing compounds like chlorothalonil to the environment, a highly toxic substance for aquatic animals (IPCS, 1996). A European seed production plant could vary in terms of management, as for example, the use of chlorothalonil has been banned in the European Union (European Commission, 2019b).

Limitations of current LCA methodologies relevant to this study include the fact that land use is modelled as part of the technosphere, leading to potentially misleading conclusions around higher yields per unit of land always improving eco-efficiency (Richi et al., 2015). Important impacts and ecosystem services associated with land use are not represented in life cycle impact assessment methods, including, *inter alia*, impacts on pollinators, wider biodiversity (habitats) and soil quality (Ingrao et al., 2019).

3.6. CONCLUSIONS

This study highlights the potential of chickpea pasta to play an important role in the shift from animal protein to plant protein and higher dietary fibre consumption that is critical to achieve more sustainable, healthy diets in industrialised countries. Managed appropriately, chickpea

cultivation in Europe could help to diversify crop rotations and decrease the use of synthetic fertilisers through biological nitrogen fixation. However, variable and often low chickpea yields reflect lack of crop breeding and poor agronomic practises, and represent significant potential barriers to more widespread chickpea cultivation. Breeding programmes and targeted advice via extension services could improve chickpea yields, putting this promising sustainable food crop on a level playing field with more-intensively developed wheat.

The use of the Nutrient Density Unit proposed by Van Dooren (2016) as a functional unit to compare different foods, or evaluate modification of foods, is an important advance on food LCA studies that commonly use mass, calories, or protein content as functional units. Foods made with different ingredients have different nutritional qualities that cannot be reliably represented by just one nutritional component. In this case study, the environmental burden per nutrient density unit delivered by chickpea pasta is significantly smaller than for durum wheat pasta across all categories, except for land use, which remains a significant challenge due to the scarcity of the resource. The use of the Nutrient Density Unit as a functional unit is an elegant approach to identify the nutritional (health) and environmental sustainability of different foods. Using the NDU extended the overall environmental efficiency advantage of chickpea pasta.

Further research needs to be carried out to evaluate the indirect burdens associated with a change of production of pasta types. These indirect burdens will be captured in a consequential LCA, taking into account the effects of European diet changes through partial substitution of durum wheat pasta with chickpea pasta, such as indirect land use change, the effect of higher intake of fibre on satiety leading to less food intake, and changing market prices of inputs, substitute products and co-products.

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4. SUBSTITUTION OF BEEF WITH PEA PROTEIN REDUCES THE ENVIRONMENTAL FOOTPRINT OF MEAT BALLS WHILST SUPPORTING HEALTH AND CLIMATE STABILISATION GOALS

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4.1. ABSTRACT

Recent environmental footprint comparisons between meat and plant-based meat analogues do not consider nutritional density holistically, nor the high carbon opportunity costs (COC) of land requirements, which are critical in terms of climate stabilisation targets. We performed an attributional life cycle assessment (LCA) of a 100 g serving of cooked protein balls (PPBs) made from peas (*Pisum sativum*), and Swedish-style beef meatballs (MBs) made from Irish or Brazilian beef. Per serving, PPB production and consumption was associated with lower environmental burdens across all 16 categories assessed. Global warming, acidification, and land use burdens of PPBs were at least 85%, 81%, and 89% smaller, respectively, than MBs. The scale of environmental advantage was sensitive to the allocation method, with biophysical allocation across cattle co-products decreasing MB burdens by at least 35%, 38%, and 46% in the acidification, climate change, and land use categories, respectively. Furthermore, PPBs have a higher nutritional density than MBs, and hence their environmental footprint per unit of nutrition was considerably lower across all 16 impact categories. Per Nutrient Density Unit, global warming, acidification, and land use burdens of PPBs were at least 89%, 87%, and 93% smaller, respectively, than MBs. Results were tested with Monte Carlo simulations and a modified null hypothesis significance test, which supported the main findings. Finally, when COC of land was factored in, the climate advantage of PPBs extended greatly. Assuming MBs equivalent to just 5% of German beef consumption are replaced by PPBs, total carbon savings including COC could amount to 8 million tonnes CO₂e

annually, an amount equal to 1% of Germany's annual GHG emissions. Therefore, this study highlights the potential of PPBs to meet health and climate neutrality objectives.

Keywords: Life-Cycle Assessment, dietary behaviour, health impacts, carbon footprint, carbon opportunity cost, sustainable products

4.2. INTRODUCTION

The previous chapter presents the first case study of this thesis: a comparative LCA of chickpea pasta versus durum wheat pasta. Results show that chickpea pasta had lower environmental burdens than wheat pasta across 10 categories out of 16 when using a serving FU, against 15 out of 16 when using a NDU FU. This was due to the chickpea pasta having a NDU that was 1.6 times higher than the NDU of the durum wheat pasta. The major trade-off, however, was a higher land use for the chickpea pasta, which yields need improving. This fourth chapter presents the comparative LCA of pea protein balls (PPBs) with beef meatballs (MBs). Two questions were answered: are the PPBs more sustainable from an environmental and nutritional perspective than their beef equivalent? How does the use of a nutritional functional unit affect the environmental impact of this food product compared to its conventional alternative? In addition, we included the carbon sequestration potential from land use savings if just 5% of German beef consumption was replaced by PPBs.

The European Union set a target in September 2020 to become the first climate-neutral continent by 2050, and achieve a 55% reduction of net GHG emissions by 2030 from 1990 levels (European Commission, 2020c). The food sector is currently responsible for 26% of total GHG emissions worldwide (Our World in Data, 2019), and is therefore a key area in which finding sustainable alternatives is crucial to mitigate the associated environmental footprint. Furthermore, food production dominates global land use, in competition with alternative land uses such as forestry that are increasingly being promoted to address biodiversity and climate crises (Arneth et al., 2019). At the same time, European diets differ from national dietary recommendations, resulting in more than 50% of individuals being overweight and more than 20% obese (WHO, n.d.). The Farm to Fork strategy is at the centre of the European Green deal, and aims to build “fair, healthy and environmentally-friendly food systems” (European Union, 2020). Delivering nutrition with a minimal environmental impact on the environment is therefore key.

The excessive production and consumption of meat, especially beef, is of significant concern globally, driving major environment damage (Eshel et al., 2014; Poore and Nemecek, 2018) and, in

industrialised countries, negative health outcomes through overconsumption (Godfray et al., 2018; Rust et al., 2020). The EAT-Lancet Commission 'planetary healthy' diet recommends an intake of red meat not exceeding 28 grams a day, representing at most around 10 kg of red meat per person per year (Willett et al., 2019). Current red meat consumption is nearly four times higher than this in developed countries, at more than 39 kg retail weight per capita annually (FAO, 2019b). Furthermore, intake of processed meat is 90% higher than recommended globally (Afshin et al., 2019). High red meat consumption has been linked with increased risk of several chronic diseases (type 2 diabetes, gestational diabetes, coronary heart disease, heart failure, stroke, and cancer) and an increased death rate (Larsson and Wolk, 2006; Norat et al., 2005; Pan et al., 2012; van Dooren et al., 2014; Wolk, 2017). Some types of meat processing and cooking result in the formation of carcinogenic chemicals, and red meat consumption has been categorised as "probably carcinogenic to humans" (Bouvard et al., 2015). Other issues related to meat include the global antimicrobial resistance crisis catalysed by the improper use of antibiotics in the meat industry (Spellberg et al., 2016) and meat adulteration (Ali et al., 2015; Rahman et al., 2014). Limiting the consumption of processed and unprocessed red meat is now advised by several authoritative bodies (Richi et al., 2015; Willett et al., 2019).

From an environmental perspective, pasture-based beef production can enhance carbon sequestration in soils and improve nitrogen availability for plants (Henderson et al., 2015). However, these benefits are at best counterbalanced by the substantial greenhouse gas (GHG) emissions and losses of reactive nitrogen (N) to air and water from beef production. Beef production is an important source of methane emission globally, through the processes of enteric fermentation during feed digestion and manure management (Godfray et al., 2018). It is also an important source of ammonia and nitrous oxide from manure management and fertilisation, and carbon dioxide from manufacture of fertilisers, combustion of fuels, and feed production (Beauchemin et al., 2009; Chadwick, 2005; Nguyen et al., 2010). In addition to GHG emissions, extensive grazing and feed demand in particular drive deforestation, biodiversity loss, water pollution, nutrient leakage and land degradation (Godfray et al., 2018; Steinfeld et al., 2006). Cattle rearing is an inefficient way to provide calories and protein nutrition to humans (Shepon et al., 2016).

Substituting meat protein with plant protein in the human diet, in particular legumes, has the potential to significantly improve the sustainability of food systems (Jensen et al., 2012; McCrory et al., 2010; Peoples et al., 2019; Wagner, 2011). Legumes, including peas, fix atmospheric N through their symbiotic relationship with rhizobia in their roots, therefore do not necessitate the addition of synthetic N fertiliser (Stagnari et al., 2017; Wagner, 2011; Westhoek H. et al., 2016). A study by Harwatt et al. (2017) showed that replacing beef calories and protein intake by grain

legume consumption in the United States would accomplish between half to three quarters of the 2020 GHG emissions reduction target and free nearly half of US cropland. However, consumers are reluctant to reduce meat consumption (Macdiarmid et al., 2016). Indeed, global diets have changed considerably since the 1960s. More calories are being consumed per person, and the proportion of fat and animal protein consumed has increased significantly with wealth. In contrast, the consumption of plant protein has remained static with increasing GDP. This has led to a marked decline in the healthiness of diets as personal wealth increases (Williams et al., 2020b).

Current legume consumption in Europe represents just 1% of daily energy intake (FAO, 2019a; Williams et al., 2020b). However, the growth of innovative legume-based food products that emulate meat products in terms of flavour and texture offers a potential sustainable diet transition that requires little consumer effort while meeting protein demand. The popularity of these products is increasing worldwide, and the meat substitute market is expected to reach USD 140 billion by 2029, ten times higher than in 2019 (Statista, 2019a). One such substitute product is the plant-based 'meatball.' Meatballs are a popular dish globally, and countries have their own variations in terms of key ingredients that typically include beef, veal, pork, spices, and breadcrumbs. In 2015, IKEA sold 2.9 million Swedish-style meatballs every day (IKEA, 2015). Our study is the first LCA of vegetarian protein balls, a meat alternative that has fewer processed ingredients than meatless burger patties, and are hence potentially healthier. Further, this study evaluates environmental impacts in terms of nutrition provided, and carbon opportunity costs.

Existing literature has evaluated the comparative environmental footprints of some legume and beef products. Smetana et al. (2015) compared the environmental footprints of common meat substitutes, using mass, energy, and protein functional units (FU), while Zhu & Van der Land (2003) used a protein FU only. Davis et al. (2010) compared the environmental impacts of different protein sources using a meal as a FU. Mejia et al. (2020) compared the environmental impact of producing 57 different meat analogues, looking at GHG emissions only and using mass as a FU. However, the use of FUs that take into account more complete aspects of nutrition has not been used in LCA studies of meat analogues. Considering the prevalence of current Western diets that are nutrient-poor and energy-rich, with a high consumption of meat products, the surge in popularity of meat analogues, and the lack of correlation between nutrient content and environmental impact of foods (Saarinen et al., 2017), using a FU that integrates several nutrients is key to assess the sustainability of meat analogues versus meat. Sustainable nutrition is evaluated in this study through the use of the Nutrient Density Unit as a FU, as first proposed by Van Dooren (2016) and used in Saget et al. (2020). This FU allows the comparison of different types of protein balls on a basis of what modern diet formulations focus on: environmental sustainability and nutrition (Donati et al., 2016; Lawrence et al., 2015; Willett et al., 2019).

Beef from contrasting global export regions is considered to represent trade in these food commodities. However, the recent free trade agreement with the Mercosur is likely to compromise the achievement of carbon reduction targets, due to illegal deforestation that may contaminate the increased EU imports of Brazilian beef (Rajão et al., 2020). At present, Brazilian beef already represents up to 40% of European beef imports (Askew, 2020), with 200,000 tonnes of beef imported from the Mercosur every year (European Commission, 2020d). Germany is a major importer of Brazilian beef. In 2019, the country imported 12.9 thousand tonnes of boneless meat of Brazilian bovine animals, fresh, chilled, or frozen, for a value of 89.1 million dollars (United Nations, 2020). The production of animal-based food also represents a substantial carbon opportunity cost (COC), due to the potential of carbon sequestration by land restoration (Hayek et al., 2020).

This study goes beyond the scope of previous food footprint comparisons to explore the consequences of different land requirements in the context of targets for Net Zero GHG emissions in the second half of this century (Masson-Delmotte et al., 2019), using the carbon opportunity cost method recently proposed by Searchinger et al. (2018).

4.3. MATERIALS AND METHODS

4.3.1. Goal, scope, and boundary definition

This study is a comparative assertion of the overall environmental impacts arising from consumption of Pea Protein Balls (PPBs) produced in Germany with conventional beef meatballs (MBs) produced in Germany using Brazilian (BR) or Irish (IE) beef, accounting for their respective nutritional values. The aim was to assess the relative advantages and disadvantages of the three products by performing an attributional LCA. The open source software OpenLCA v1.10.2 (GreenDelta, 2020) was used to calculate the environmental footprint of the three products from cradle to fork, using Agri-footprint v3.0 (Durlinger et al., 2017) and Ecoinvent v3.6 (Wernet et al., 2016) international databases. Agri-footprint (Durlinger et al., 2017) is a life cycle inventory database specialised in the food and agricultural sector, and Ecoinvent v3.6 (Wernet et al., 2016) is a life cycle inventory database covering extensively numerous fields (electricity generation, chemicals production, recycling, agricultural processes, etc.). In accordance with the ILCD handbook (JRC, 2010), identical product use and patterns were analysed, and the same life cycle stages were included. Packaging recycling was excluded from the system boundaries, as it was assumed to be similar in both cases. Primary data on the German PPBs were collected from IGV GmbH, a company specialising in food and feed processing located in the Potsdam-Mittelmark district (IGV, accessed 12/2019). Data on processing of beef and other ingredients to produce the MBs were adapted from

Biswas & Naude (2016) and modelled as though the beef meatballs were produced in Germany, to align with the PPB geographical production.

As nutritional delivery is a key function of food, two FUs were used: a serving-based FU, 100 g cooked protein balls, and a nutritional FU, the Nutrient Density Unit (NDU) (Van Dooren, 2016) of cooked protein balls. Both economic and physical allocation was performed for beef co-products and pea co-products. Biophysical allocation, which allocates burdens to co-products based on their metabolic energy requirements, was also performed for MBs (BR and IE) as a sensitivity analysis. The aim of biophysical allocation is to allocate based on physical and causal relationships between co-products, to offer an alternative to economic allocation, since the latter can be subject to market fluctuations (Sandin et al., 2015). Physical allocation based on mass was performed for the pea dehulling and fractionation co-products in PPBs as a comparison.

Figure 4.1 illustrates the system boundaries and manufacturing steps for the cradle to fork assessment of PPBs and MBs (BR and IE). The LCA methodology followed the Product Environmental Footprint (PEF) guidelines (European Commission, 2018a) for the agricultural, distribution (40% distributed in Germany, 40% in Europe, and 20% outside Europe), storage and cooking stages. End of Life (EoL) was excluded from the boundaries, as it was considered to be the same across all products. Environmental impacts of the three alternatives were compared across sixteen environmental impact categories recommended in PEF guidelines (European Commission, 2018a). Results were then normalised using global per person environmental loadings recommended in the PEF guide. This was done to facilitate interpretation of impact scores.

A Monte Carlo analysis with 1000 runs was performed in OpenLCA v1.10.2 (GreenDelta, 2020) for each of the three products across all sixteen categories for all scenarios: baseline, NDU FU, no beef transport, and biophysical allocation. The majority of parameters used an underlying lognormal distribution but in some places the default normal distribution was used. The negative Monte Carlo results were set to zero, as advised by Muller et al. (2016). A modified Null Hypothesis Significance Test (NHST), as presented in Mendoza Beltran et al. (2018) was implemented, after being identified as the most appropriate uncertainty statistics method for interpreting our propagated uncertainty LCA outcomes. The hypothesis “the different production systems delivering protein balls are associated with different environmental impacts” was tested statistically. The null hypothesis tested assumed that the environmental impacts of the inventory of the different production systems were equal. We applied the Bonferroni correction of the significance level to exclude false positives, as $\alpha_b = 0.05/48 = 0.001042$. The 48 factor stands for the sixteen environmental impact categories and for the three pairs of alternatives. All threshold values

were tested with an effect size (δ_0) of 0.2. The results that were consistent across all scenarios were recorded.

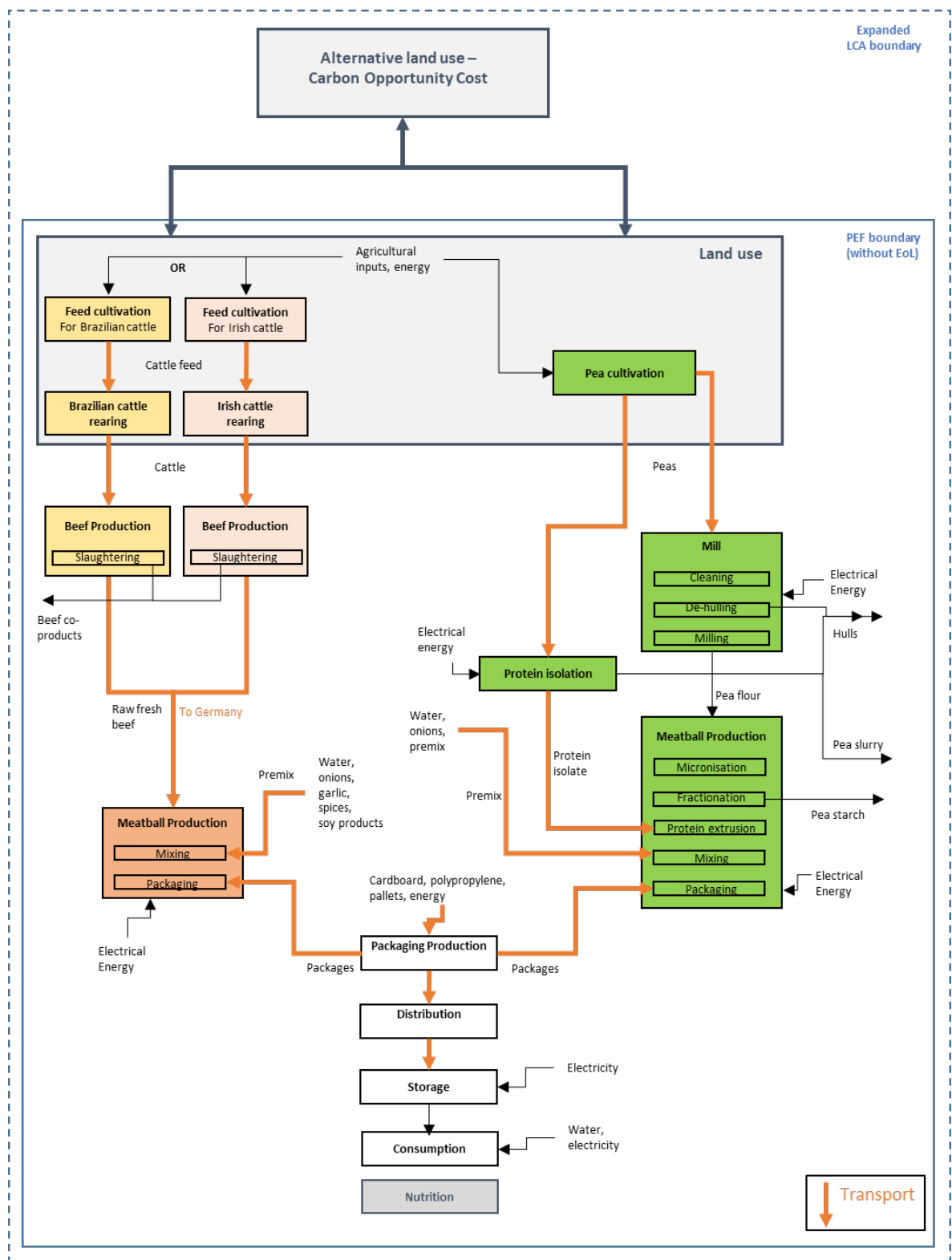


Figure 4.1: System boundary of pea protein balls and Swedish-style beef meatballs (from Irish or Brazilian beef) production, from cradle to fork.

Expanded boundary includes carbon opportunity cost of land that could be used for alternative purposes such as forestry to meet Net Zero GHG targets.

4.3.2. Nutrient Density Unit Functional Unit

The Nutrient Density Unit (NDU) FU was applied following Van Dooren's (2017) formula (1).

$$\text{NDU} = \frac{\left(\frac{EFA}{DV_{EFA}}\right) + \left(\frac{Prot}{DV_{prot}}\right) + \left(\frac{Fib}{DV_{fib}}\right)}{3 \times \left(\frac{S_i}{2000 \text{ kcal}}\right)} \quad (1)$$

Where:

EFA is the amount of essential fatty acids in 100 g of product, expressed in grams.

Protein is the amount of protein in 100 g of product, expressed in grams.

Fibre is the amount of fibre in 100 g of product, expressed in grams.

DV_{EFA} is the recommended daily value intake of essential fatty acids, expressed in grams.

DV_{prot} is the recommended daily value intake of protein, expressed in grams.

DV_{fibre} is the recommended daily value intake of fibre, expressed in grams.

S_i is the amount of kilocalories in 100 g of product, expressed in kilocalories.

Nutritional data for PPBs and MBs (BR and IE) were obtained from analyses undertaken by the Catholic University of Porto (Portugal). Details of the nutritional analysis method are recorded in Section 9.1.1 of the Appendix. The nutritional composition for MBs and PPBs and calculated NDUs of cooked PPBs and MBs (IE and BR) are recorded in Table 4.1. The simplicity of the NDU is a major advantage, requiring few nutritional analyses when compared with more extensive FUs, such as the Nutrient Rich Foods index 11.3 (Fulgoni et al., 2009), while still correlating with them (Saget et al., 2020).

Table 4.1: Summary of nutritional composition and Nutrient Density Units of pea protein balls and Swedish-style beef meatballs (from Irish or Brazilian beef) cooked, per 100 g.

Content per 100 g cooked	PPBs	MBs (BR and IE)
Energy (kcal)	209	240
Protein (g)	22.33	17.5
Dietary fibre (g)	1	2
EFA (g)	1.6	0.6
NDU	1.96	1.33
NDU (adjusted)	1.85	NA

Even though net protein assimilation is similar amongst adults regardless of whether the protein comes from plants or animals (WHO, 2007), and PPBs and MBs are present in countries in which protein is not a limiting nutrient, we calculated the NDU of PPB with adjusted protein digestibility, assuming all protein in the PPBs come from peas for simplicity. The protein digestibility of pea protein concentrate is 92% (Gilani and Lee, 2003), which makes the amount of protein available in the PPB 20.5 g per 100 g cooked PPBs, versus 22.33 g when not adjusted. The resulting NDU of adjusted protein digestibility for PPBs is 1.85, versus 1.96 for PPBs when protein digestibility was not accounted for, as shown in Table 4.1.

4.3.3. Pea protein balls and Meatballs inventory

The three products assessed in this study are shown in Figure 4.2: Pea Protein Balls (PPB) made in Germany with German peas, Swedish-style meatballs made in Germany with Brazilian beef (MB (BR)), and Swedish-style meatballs made in Germany with Irish beef (MB (IE)). These products reflect globally important and contrasting beef systems that supply export markets, to Germany and beyond. Brazil is the largest global exporter of beef, and Brazilian beef systems are diverse but dominated by low-input, low-output production across vast areas of degraded pastures. Meanwhile, Irish beef systems based on productive pastures with grazing through most of the year are considered to be amongst the most sustainable beef systems in Europe, with comparatively low carbon footprints (Leip et al., 2010). Irish beef cattle can be completed on grass with minimum concentrate feed inputs (Teagasc, 2020), and may be awarded a protected geographical indication

status in Europe as “Irish grass-fed beef” (The Irish Times, 2020). Over 90% of Irish beef produced is exported, largely into the rest of Europe.

To assess water scarcity, Ecoinvent v3.5 electricity mixes (Wernet et al., 2016) were used instead of those of v3.6, due to an identified issue with the latter. Unrealistic water scarcity burdens were otherwise associated to the electricity use, as AWARE from EF method (adapted) was associating global water scarcity factors to the input water flows, and regional water scarcity factors to the output water flows. Instead, X and -X factors would normally to be attributed to input and output water flows for electricity (hydropower), respectively, as in Ecoinvent v3.5 (Wernet et al., 2016). Full recipes of the PPBs and MBs were recorded in Section 9.1.2 of the Appendix. Inputs and outputs for all processes involved in the life cycle of 100 g of cooked MBs and PPBs, from cradle to fork, were recorded and are included in full in Section 9.1.3 of the Appendix. MBs were made of beef, onions, garlic, vegetable oil, salt, soy protein, and water. Further details on the MBs inventory were recorded in Section 9.1.4 of the Appendix. PPBs were made of peas, potato starch, sugar, chives, onions, paprika, cornflakes, salt, and water. Production of PPBs included dehulling, milling, micronisation, fractionation, and extrusion. All details on the PPBs inventory were recorded in Section 9.1.5 of the Appendix. Geographical locations for all steps from cattle rearing to sale of final products are recorded in Figure 4.2. Transport distances were modelled following PEF guidelines, include sea-transport distance calculations (Searates, accessed 01/2020). As mentioned above, the scenario MB (BR) is likely to take place in reality, due to Brazil being a major beef exporter to Germany, and the probability of this scenario happening will increase with the recent trade agreement between the Mercosur and the EU. Nevertheless, a sensitivity analysis was performed to assess the importance of transport of beef in the life cycle of MB (IE and BR).

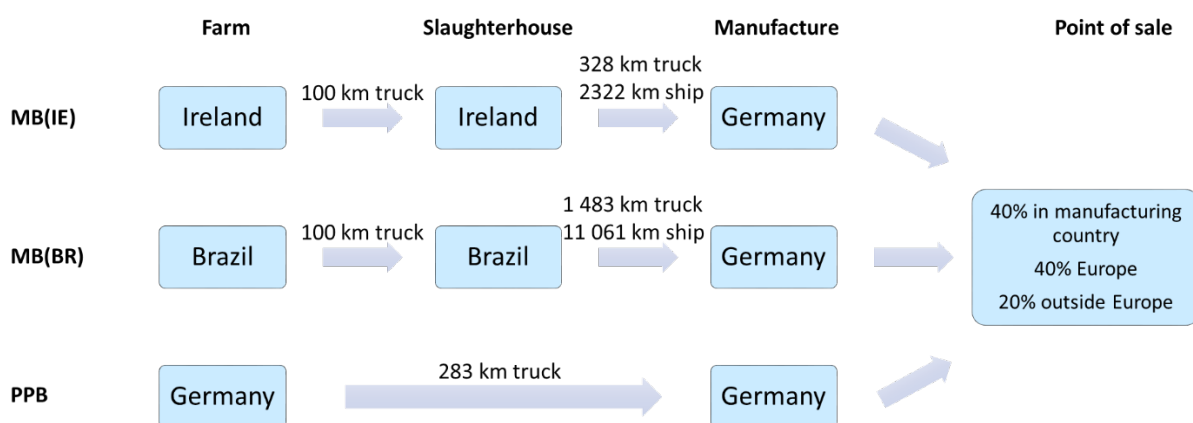


Figure 4.2: Geographical locations where steps across the life cycle of pea protein balls and Swedish-style meatballs (with Brazilian or Irish beef) take place. The percentages in the point of sale step represent the portion of total products sold around the world.

4.3.4. Carbon opportunity cost calculations

The COC factors of crops and livestock used in this study were extracted from the carbon loss method presented in Searchinger et al. (2018), and were applied to the recipe ingredients for the PPBs and MBs recorded in Tables 9.1 and 9.2 of the Appendix. The carbon loss method calculates the COC with the global carbon loss from native vegetation and soils caused by producing a certain crop and divides it by the global annual production of this crop. The ingredients' COC were then summed proportionally to match the recipes. The COC for a 100 g serving and for one NDU of MBs and PPBs was multiplied by the corresponding factors to match the economic or physical allocation approaches. These were added to the production GHG emissions, and reported as a separated COC-inclusive GWP footprint.

Following the extrapolation scenario in which it is assumed that 5% of the beef consumed in Germany is processed into MBs, the COC involved with the consumption of MBs was determined using the COC calculation described above. This COC was multiplied by the average beef consumption per capita in Germany and the German population. To this was added the COC of the corresponding amounts of other ingredients to make the MBs. The COC was also calculated for an equal weight of PPBs using the same approach in order to calculate the potential COC saving associated with a simple 5% substitution of beef, in the form of MBs, with PPBs.

4.4. RESULTS

4.4.1. General results

Across all scenarios, the burdens of PPBs were statistically significantly lower than those of MBs (IE and BR) across the acidification, climate change, marine and terrestrial eutrophication, land use, photochemical ozone formation, resource use energy carriers and minerals and metals, and respiratory inorganics categories. The modified NHST results with the Bonferroni correction common to all scenarios were recorded in Table 9.8 of the Appendix. All Monte Carlo simulations and modified NHST results were recorded in the Microsoft Excel workbooks in the Supplementary Data.

The economically-allocated environmental impact results for the sixteen categories are recorded in Table 4.2. The results per 100 g cooked MBs (BR and IE) and PPBs, and per NDU are shown. In terms of 100 g cooked balls, the PPBs have the lowest environmental burdens across all impact categories, irrespective of whether the beef comes from Ireland or from Brazil. The environmental burdens of PPBs were statistically significantly lower than those of MBs across 11 categories out of 16 when comparing PPBs to MBs (IE), and across 12 categories out of 16 when comparing PPBs to MBs (BR) (Supplementary Data). In terms of climate change, land use, and

eutrophication, the impact of MBs (IE) is between 4 and 34 times that of PPBs, and for MBs (BR), it is between 3 and 31 times that of PPBs (Table 4.2). Brazilian beef meatballs were shown to perform better than those from Irish beef across 11 out of 15 categories, including resource use energy carriers, and the three eutrophication categories. However, MB (BR) was associated with 40% more GHG emissions than MB (IE).

Relative results normalised per person equivalents per NDU are recorded in Figure 4.3. Besides the commonly highlighted higher climate change and land use burdens involved with beef production, this study provided a larger overview of the other relative impacts on the environment. Notably, the comparatively higher marine (53% higher than a serving of MB with Brazilian beef) and terrestrial (3.4 times higher than a serving of MB with Brazilian beef) eutrophication involved with beef production in Ireland, as well as acidification (2.9 times higher than a serving of MB with Brazilian beef) and respiratory inorganics (3.1 times higher than a serving of MB with Brazilian beef). These four categories were the ones with the higher person equivalent burdens, with 100 g of MB (IE) corresponding to 0.1%, 0.2%, 0.1%, and 0.1% of an average person's annual marine eutrophication, terrestrial eutrophication, acidification, and respiratory inorganics impacts, respectively. The main reasons for this marine eutrophication burden lie in the nitrate emissions from grass cultivation and grazing (75% of total impact). The terrestrial eutrophication burden is mainly due to ammonia emission from cattle (43% of total impact) and from grass cultivation and grazing (45% of total impact). The acidification burden is due to ammonia emission from cattle (41% of total impact) and from grass cultivation and grazing (43% of total impact). Finally, the respiratory inorganics burdens are due to ammonia emissions from cattle (40% of total impact) and from grass cultivation and grazing (42% of total impact).

The results of the sensitivity scenario in which transport of beef was excluded was recorded in Table 9.9 of the Appendix. The environmental burdens associated with beef transport were not large enough to substantially reduce their difference from PPBs, excepted for in the water scarcity category (Supplementary Data). Ignoring the transport burdens reduced the climate change burden of MB (IE) by 1% and of MB (BR) by 2%. There was also a reduction between 1 and 8% across other impact categories, and an ozone depletion burden reduction of 18% for the MB (BR).

Per NDU, the differences between the PPBs and the MBs increased, as the NDU of PPBs was 19% higher than the NDU of MBs (IE and BR). Per NDU, in terms climate change, land use, and eutrophication, the impact of MBs (IE) was between 4.5 and 55 times that of PPBs. The Monte Carlo simulation and modified NHST supported these findings of the differences between the results per serving and per NDU (Supplementary Data). The environmental impact of PPBs per NDU with

adjusted protein digestibility remained lower across all impact categories than per NDU of either MBs.

With economically allocated burdens, adding the COC to the GHG emissions of producing and consuming the three products resulted in the MBs having a climate change burden 11 and 12 times greater than that of PPBs, for MBs (IE) and MBs (BR), respectively (Table 4.2). When using one NDU as the FU, these differences increased to 19 and 21 times greater, for MBs (IE) and MBs (BR), respectively (Table 4.2). With (bio)physically allocated burdens, per serving, the differences were 5 and 7 times greater, for MBs (IE) and MBs (BR), respectively (Table 9.10).

Table 4.2: Summary of economically-allocated environmental burdens for pea protein balls, meatballs (Brazil), and meatballs (Ireland), expressed per 100 g cooked product and per Nutrient Density Unit.

Impact category	Unit	Impact per serving			Impact per NDU			
		PPB	MB(IE)	MB(BR)	PPB	MB(IE)	MB(BR)	PPB (adj)
Acidification ter. & freshwater	mol H ⁺ eq	0.0034	0.07	0.018	0.0017	0.052	0.013	0.0018
Cancer human health	CTUh	1.10E-08	5.00E-08	7.90E-07	5.40E-09	3.70E-08	5.90E-07	5.75E-09
Climate change (w/o COC)	kg CO ₂ eq	0.5	3.3	4.6	0.27	2.5	3.4	0.29
Climate change + COC	kg CO ₂ eq	1.1	13.4	14.7	0.5	10.1	11.1	0.6
Ecotoxicity freshwater	CTUe	1.1	5.7	271	0.55	4.3	203.4	0.58
Eutrophication freshwater	kg P eq	0.0003	0.0012	0.0008	0.0002	0.0009	0.0006	0.0002
Eutrophication marine	kg N eq	0.0013	0.029	0.019	0.0007	0.021	0.014	0.0007
Eutrophication terrestrial	mol N eq	0.0088	0.3	0.067	0.004	0.22	0.05	0.005
Ionising radiation, HH	kBq U ²³⁵ eq	0.17	0.3	0.3	0.08	0.23	0.23	0.09
Land use	Points	38	354	1186	19.6	266.3	891.4	20.8
Non-cancer human health	CTUh	6.60E-07	2.30E-06	4.20E-04	3.40E-07	1.80E-06	3.20E-04	3.58E-07
Ozone depletion	kg CFC-11 eq	6.50E-08	9.10E-08	1.10E-07	3.30E-08	6.80E-08	8.50E-08	3.50E-08
Photochem. ozone form.	kg NMVOC eq	0.0014	0.0076	0.0055	0.00072	0.0057	0.0041	0.0008
Resource use, energy carriers	MJ	8.6	19.8	15.4	4.4	14.9	11.6	4.6
Resource use mins. & metals	kg Sb eq	3.40E-09	2.90E-07	4.40E-09	1.70E-09	2.20E-07	3.30E-09	1.82E-09
Respiratory inorganics	disease inc.	2.20E-08	5.00E-07	1.20E-07	1.10E-08	3.70E-07	9.10E-08	1.18E-08
Water scarcity	m ³ depriv.	0.27	1.03	0.43	0.14	0.78	0.32	0.14

From smallest (1) to largest (3) environmental impact:

1  2  3 

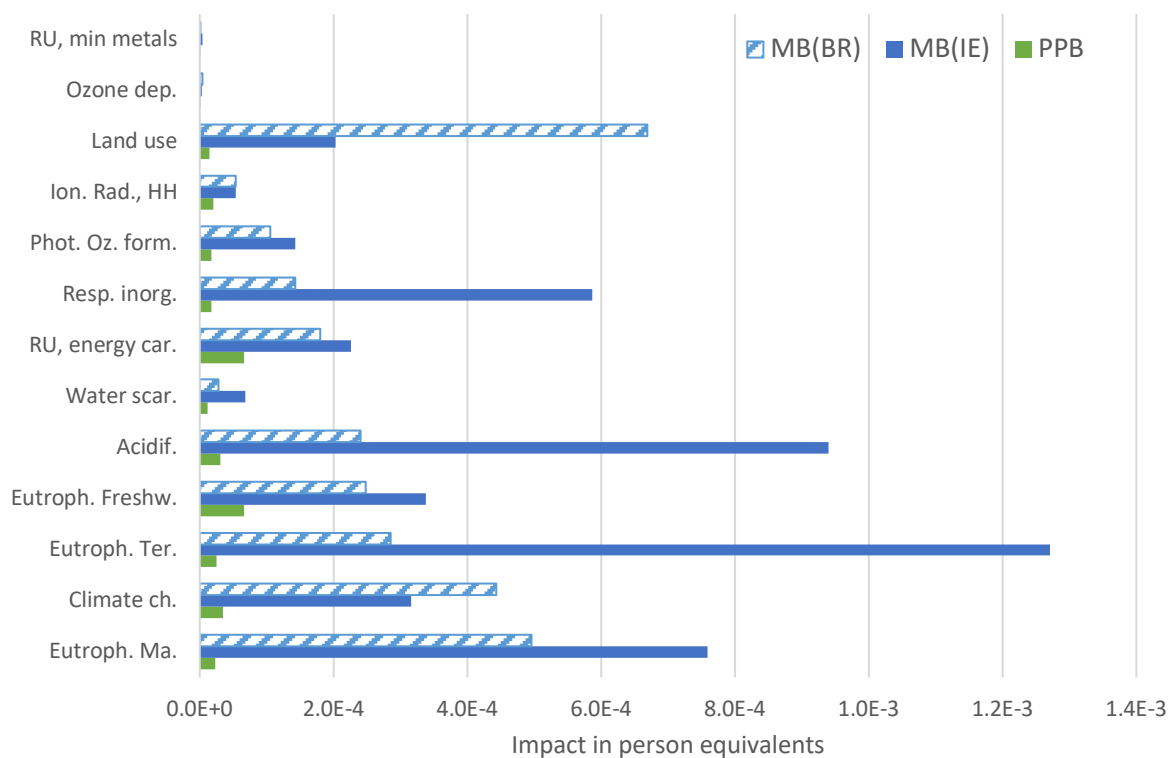


Figure 4.3: Environmental burden of cooked pea protein balls and meatballs from Irish and Brazilian beef across 13 impact categories, using as a FU the Nutrient Density Unit. Results are normalised per person equivalents.

To evaluate the sensitivity of results to the allocation choice for partitioning of flows between co-products, a physical and biophysical approach was tested and compared with the aforementioned impact assessment results based on economic allocation (Table 4.2). The environmental burdens of PPBs with physical allocation between kernels and starch co-products, starch-rich and protein-rich co-products, and MBs with biophysical allocation between beef and other cattle co-products are available in Table 9.10. With a biophysical allocation factor for beef of 26.8% of total cattle production burdens, the environmental burdens associated with production of MBs (BR and IE) were significantly reduced across most categories when compared to the same system economically-allocated. For example, the burdens of MBs (IE) when biophysical allocation was performed were between 55 and 71% smaller in the climate change, acidification, and land use categories, than when economic allocation was performed. Biophysical allocation reduced these burdens for MBs (BR) by between 36 and 46%. The environmental burdens of PPBs only decreased between 2 and 37% across all categories when economic allocation was replaced with biophysical allocation. Nonetheless, the environmental burdens of PPBs remained smaller than those of MBs (BR and IE), regardless of the allocation method.

4.4.2. Process contributions

Burden contributions across each stage of the value chain for PPBs and MBs (BR and IE) are recorded in Figure 4.4 for all sixteen environmental impact categories. For MBs (IE and BR), production of the main ingredient (beef) is responsible for most of the impacts across all categories, except for freshwater eutrophication and ionising radiation (cooking contributed 41-57% and 85% of these burdens, respectively, along with 50-60% of total energy use). Beef production was responsible for 77-95% of total acidification, 76-82% of total climate change, 90-98% of total marine and terrestrial eutrophication, and 96-99% of total land use burdens. Acidification and terrestrial eutrophication were in part due to ammonia emission to air from cattle raising and grass production. Non-fossil methane emitted to the air from cattle was responsible for more than one third of the climate change impact. Nitrate emission to water was responsible for most of the marine eutrophication burden. Land use was mostly due to grassland occupation for pasture. The high differences in land use burdens were due to Brazilian beef cattle grazing at much lower stocking density than Irish beef cattle, often on degraded pastures, hence requiring more area. Animals are also slaughtered later, therefore generating less beef per animal (and grazed area) per year. Additionally, land use change impacts associated with clearing of large areas of forest for Brazilian beef production was not included in the climate change footprint.

Large differences were revealed between cattle rearing in Brazil and Ireland across the cancer and non-cancer human health categories. The high non-cancer burden associated with Brazilian beef was due to zinc and lead emitted to the soil, while those of the cancer burden were due to emissions of chromium, nickel, lead, and cadmium to the soil. These emissions of heavy metals stem from mineral fertilisation and liming, and were regionalised to Brazil (Canals, 2003; Wernet et al., 2016).

The majority (75%) of water scarcity burdens of MBs (IE) were due to compound feed production, with oat production responsible for more than half of the burdens, and maize production for 16%. For MBs (BR), water scarcity burdens were much more spread across beef production and other ingredients' production, with no specific process responsible for a majority of the impact. Water scarcity burdens of PPBs were mainly due to irrigation required for onion and bell pepper production. Aside from the agricultural stage, cooking of each product was responsible for 10 to 24% of the total water scarcity burdens.

Cooking was the stage contributing the most towards the environmental impacts of PPBs, with a percentage contribution of at least 20% across 10 out of 16 impact categories. It contributed to 47% of the total climate change burden and 56% of total resource use energy carriers. The second stage that contributed the most to the environmental burdens was the cultivation and processing

of the other ingredients. This stage was responsible for 53% of the total marine eutrophication, and 29% of total land use. Pea cultivation had a low share of burdens across all categories, except marine eutrophication for which it was responsible for 22% of the total, and land use for 57%. Across the other categories, its contribution was less than 7%. The burdens associated with cooking were higher for MBs due to a longer cooking time and different cooking instructions.

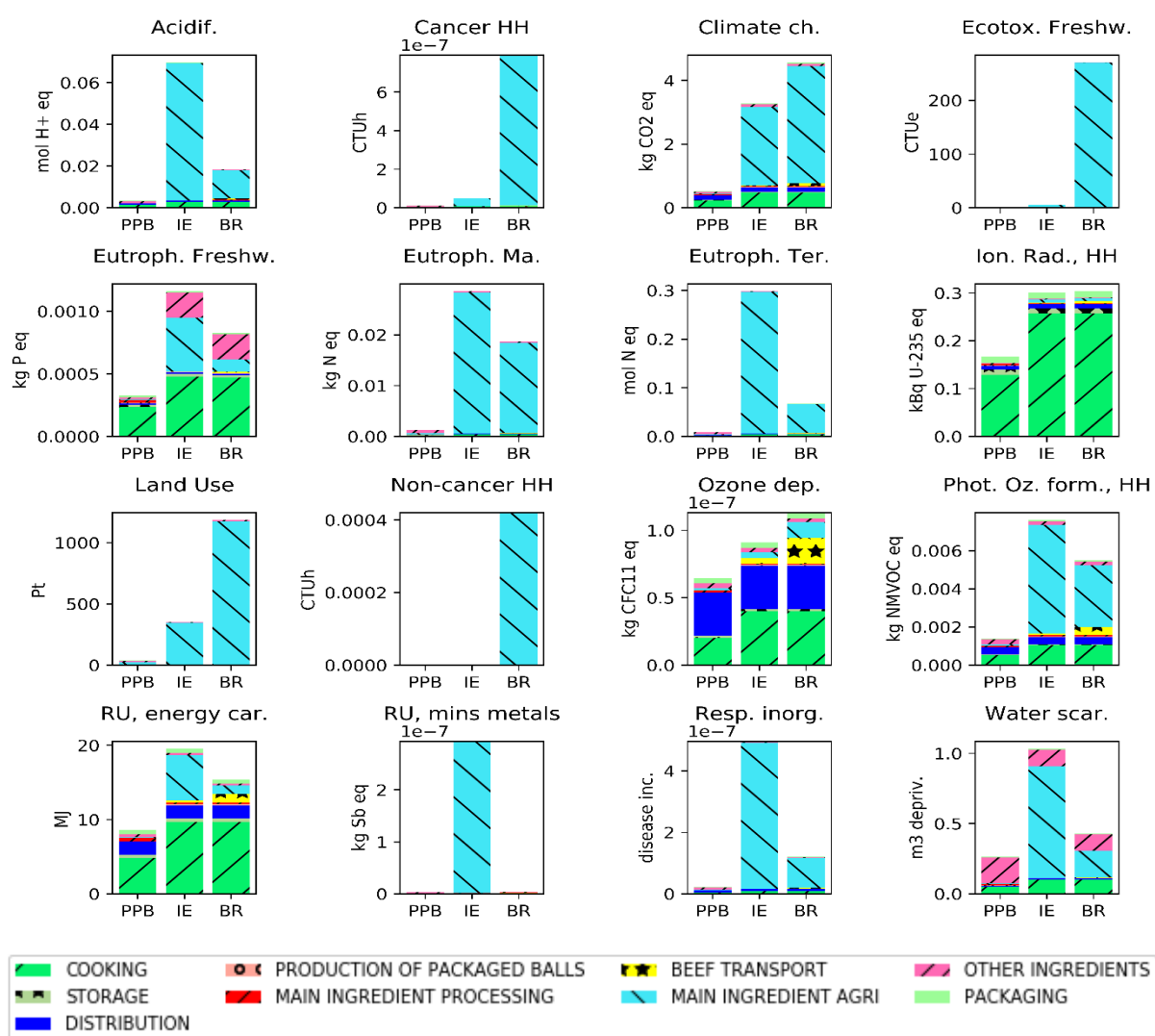


Figure 4.4: Process contributions of pea protein balls and meatballs (Irish and Brazilian) across the 16 impact categories.

4.4.3. Extrapolated scenario impacts

With a consumption of 10 kg of beef on average per capita (Statista, 2020a) and a population of more than 83 million inhabitants in 2018 (FAOSTAT, 2020), 5% of beef consumed in Germany represents an amount of around 42 thousand tons. Extrapolating this as the amount of MBs (IE) produced and consumed amounts to a climate change burden of 2.0 million tons CO₂ equivalent, and for MBs (BR) 2.7 million tons CO₂ equivalent. For a same weight of protein balls, the

corresponding climate change impact of PPBs was of 216 thousand tons CO₂ equivalent, 90% smaller than the climate change burden of the corresponding amount of MBs (IE) and 93% smaller than that of MBs (BR).

With a COC of 144 kg CO₂ equivalent per kg of beef (Searchinger et al., 2018), the COC of just 5% of German beef consumption (in meatballs) would be 5.6 million tonnes CO₂ equivalent a year. A COC of 10.5 kg CO₂ equivalent per kg fresh weight of pulses (Searchinger et al., 2018) would translate into an aggregate COC of 0.3 million tonnes CO₂ equivalent a year for a weight of PPBs equivalent to 5% of German beef consumption in the form of MBs. Therefore, the COC saving of substituting 5% of beef consumption in Germany (as MB) with an equivalent weight of PPBs is approximately 5.3 million tonnes CO₂ equivalent a year. Including the lower emissions from PPB production, the total emission saving of such a substitution would amount to around 7 million tonnes of CO₂ equivalent a year (MB IE), or 8 million tonnes CO₂ equivalent a year (MBs BR). Besides this, considerable savings in this scenario include a positive difference of 195 to 688 billion land use points, 370 to 540 quadrillion global person equivalents in resource use, energy carriers, and 6.4 to 31 quadrillion global person equivalent in terrestrial eutrophication, among others.

4.5. DISCUSSION

4.5.1. A meatball alternative at a lower environmental cost

The working hypothesis that “a serving of PPB has a significantly overall lower environmental impact than a serving of MB (IE or BR) per unit of nutrition” was validated. The results showing that the environmental burdens of a serving of a vegetarian alternative are lower than those of a meat product are coherent with those from other studies. For example, soybean versus chicken (Smetana et al., 2015), peas versus pork (Zhu and Van Ierland, 2003), soy burger versus beef (Khan et al., 2019), pea burger versus beef (Heller and Keoleian, 2018) and versus pork (Davis et al., 2010). However, Davis et al. (2010) found that the pea burger required around the same amount of energy than other meat products, due to high energy requirements for processing. We present an example of how a vegetarian meat analogue can also require less energy using few ingredients and little processing. Moreover, PPBs are a pilot product from IGV GmbH and are not commercialised yet, thus energy efficiency may improve when produced at a larger scale.

Growing peas in Germany appears to be associated with a low environmental impact when compared with other processes in the value chains assessed here, as shown by the contribution analysis. Cooking at the consumer’s house was the main contributor of the overall environmental impact of PPBs, while cattle rearing was the main contributor to the overall environmental impact of MBs. For all three products, cooking was found to be a major contributor across the freshwater

eutrophication (between 41% and 72% of total impact), ionising radiation human health (between 77% and 85% of total impact), ozone depletion (between 31% and 44% of total impact), and resource use energy carriers (between 50% and 63% of total impact). Water scarcity burdens were high for onion irrigation, which represent 12% of the PPBs in weight. However secondary data was used, and irrigation may not be required in northern countries like Germany as it would in Mediterranean countries (Abdelkhalik et al., 2019).

In their study comparing the climate change impact of beef with cultured meat, Lynch & Pierrehumbert (2019) questioned the relevance of standard GWP₁₀₀ values used in most environmental foot-printing studies. They claimed that because beef production emits more methane, a short-lived climate forcing gas, whilst cultured meat that emits more CO₂, a long-lived climate forcing gas, cultured meat could cause a greater warming effect over the very long term (1000 year scenarios) - if meat consumption and associated methane emissions decrease over time. However, the CO₂ emitted from processing the veggie options could potentially be compensated hundreds of times over by the alternative use of the land spared from meat production for carbon sequestration activities – as indicated by the COC method (Searchinger et al., 2018). Climate change burdens of processing 100 g of PPBs are 99.7% smaller than the COC of 100 g of MBs. The large land footprint of beef production reduces opportunity for land-based GHG offset necessary to achieve climate stabilisation (Arneth et al., 2019). Moreover, besides GHG emissions, beef production is associated with multiple environmental burdens pertinent to exceedance of planetary boundaries (Steffen et al., 2015), such as water use.

Beef production was by far the most impactful process across most impact categories, reflecting the environmental challenges posed by ruminant livestock rearing in relation to climate change, land use and reactive N losses in particular, as well documented in the literature (Eshel et al., 2014). The large environmental variation of different beef production systems was captured in this study with the use of the Irish beef on one hand, and the Brazilian beef on the other hand. Beef production burdens taken from Ecoinvent v3.6 represent a market average for widely differing Brazilian cattle systems. This is representative of beef originating from Brazil, a major global exporter of beef, but masks large differences in environmental efficiency of beef production across Brazil. When compared to Irish beef, Brazilian beef has a higher land and GHG footprint, while Irish beef has higher acidification, eutrophication, and energy use. Irish beef requires more inputs, such as fertilisers, but less land, and emits less enteric methane, as shown by the FAO (2018b). These results highlight that the environmental impact of beef is very sensitive where and how it is produced, as previously indicated by Poore & Nemecek (2018), but these differences are not enough to achieve better environmental sustainability than a vegetarian alternative like PPBs.

Biophysical allocation for beef co-products greatly decreased the calculated environmental footprints of beef meatballs, though they remained considerably larger than for PPBs. For example, the burdens of MBs (IE) when biophysical allocation was performed were between 55 and 71% smaller in the climate change, acidification, and land use categories, than when economic allocation was performed. Biophysical allocation reduced these burdens for MBs (BR) by between 36 and 46%. The aim of biophysical allocation is to allocate based on physical and causal relationships between co-products, to offer an alternative to economic allocation, since the latter is subject to market fluctuations and uncertainties. However, despite biophysical allocation of beef co-products being recommended by the FAO (2016), Mackenzie, Leinonen, & Kyriazakis (2017) showed that the approach still relies on economic values of co-products. Nguyen, Van der Werf, & Doreau (2012) compared economic, mass, and protein allocation in beef systems and found that results from economic allocation yielded the smallest differences with the other allocation methods, suggesting that economic allocation was a suitable allocation reference. In any case, results demonstrate the particular sensitivity of beef product footprints to choice of allocation method, which should be explored making comparisons with plant-based alternatives. The environmental burdens of PPBs only decreased between 2 and 37% across all categories when economic allocation was replaced with mass allocation. Therefore in this study, a physical allocation approach instead of an economic one decreased the environmental burdens of all three types of protein balls and had a greater reduction impact on the beef products than the pea product.

Besides the commonly highlighted higher climate change and land use burdens involved with beef production, this study provides a comprehensive overview of other important impacts on the environment, underlining the significantly higher marine, terrestrial eutrophication, acidification, and respiratory inorganics burdens involved with production of Irish beef due to ammonia emissions from cattle and grass cultivation and grazing. The high respiratory inorganics burden for the MBs (IE) adds on to the negative impacts on human health from overconsumption of red meat. Respiratory inorganics are one of the major environmental factors damaging human health, and are linked to respiratory and cardiovascular morbidity, mortality, lung cancer, and diabetes, amongst others (Fantke et al., 2015).

Restoring the land currently occupied by cattle holds great potential in storing atmospheric CO₂. Our model shows that there is a 5.3 million tons CO₂ equivalent a year difference between 5% of beef consumption in Germany and the same weight in PPBs for only COC. This amount is equal to 1% of Germany's annual GHG emissions (when including international aviation and excluding land use change and forestry) in 2018 (Eurostat, 2020a).

4.5.2. Delivering nutrition at a lower environmental cost

Besides the potential for mitigating the environmental burdens of food production and consumption, identifying healthier food options in developed countries is an urgent matter, with unhealthy diets representing a greater risk to morbidity and mortality than the sum of unsafe sex, alcohol, drug and tobacco use (Willett et al., 2019). Using a nutrient density unit as a functional unit provides a simple approach to compare the environmental efficiency of nutrition delivered by different foods, in this case protein balls made from primarily beef or peas. This is the first time in the literature that a more complete nutritional FU (as opposed to protein or calorie content individually) was used to compare meat alternatives. Existing LCA studies of meat alternatives have been so far limited by the use of a serving size or single aspect FU only, constraining their conclusiveness regarding the environmental efficiency of wider nutrition. The use of a nutrient density FU such as the NDU when evaluating different food alternatives, and for formulating diets in developed areas such as Europe, is a major advance when compared to using a simple serving size FU. Further research should be undertaken on consumer behaviour, specifically on whether an environmental sustainability label on unhealthy foods incites certain individuals to purchase these.

The choice of the NDU followed Saget et al. (2020), and is a practical approach based on analysis of a few crucial nutrients that correlate with the more extensive index of the Nutrient Rich Foods Index (NRF) 12.3, which includes all the nutrient of NRF 11.3, plus essential fatty acids: protein, fibre, vitamins A, C, E, B₁₂, calcium, iron, magnesium, potassium zinc, added sugars, saturated fats, and sodium (Fulgoni et al., 2009). Sonesson et al. (2017) claim that comparing plant and animal-based foods should be done on a protein digestibility basis. Adjusting for protein digestibility in the PPBs did not affect the NDU significantly, nor the comparative environmental performance between pea- and beef-derived protein balls.

Although the NDU correlates with the NRF 12.3 (Saget et al., 2020), which takes into account other nutrients such as added sugar, sodium, and saturated fats, the NDU does not itself consider nutrients to limit. These may be present in high amounts in processed meat alternatives. There is a general concern that these types of foods may be detrimental to health (Hu, 2011), and further study is needed to determine the comparative health disadvantages of “negative” nutrients in protein balls made from both beef and peas. The use of the NDU, however, was sufficient to show that PPBs are more nutrient-dense than MBs, thus extending their advantage over beef in terms of environmental efficiency per unit of nutrition delivered.

Despite the fact that being a healthier and more environmentally-friendly alternative to meat, consumption of PPBs and other innovative plant-based meat analogues is hindered by several factors, such as taste and texture (consumers are looking for satisfying mouthfeels), availability,

and price, as they are not as easily accessible and often more expensive than industrially-produced meat (Gapper, 2018). However, plant protein products may have a further public health advantage over meat products insofar as they avoid the use of growth hormones and antibiotics used in some livestock systems.

4.5.3. Future context

Due to the energy decarbonisation strategy of the EU to achieve climate neutrality by 2050, the climate change burden of PPBs has a large potential to decrease considerably with the use of cleaner energy sources, whereas the substantial decrease in climate change burden of cattle remains limited in comparison. Between 2004 and 2018, the average share of electricity from renewable energy sources went from 14% to 32% (Eurostat, 2020b). Table 9.11 shows the environmental burdens of replacing the electricity mixes from Europe and Germany to the Swedish electricity mix. The Swedish electricity mix was chosen due to it having the highest share of “clean” energy, with 55% of its electricity coming from renewable sources (Eurostat, 2020c). Therefore, the mix represents a good example of how other European countries like Germany will evolve towards a cleaner electricity supply. The climate change footprint of a serving of PPBs went from 0.5 kg CO₂ equivalents with the current mix to 0.3 kg CO₂ equivalents with the Swedish mix, incurring a 45% reduction. The environmental burdens were also reduced across all other categories, excepted for the ionisation human health and land use ones, across which the Swedish mix had a 69% and 3% increase, respectively.

Current carbon footprint comparisons of foods may underestimate the critical importance of land requirements in the context of future climate objectives, as they do not include COCs. Land is increasingly required to offset residual emissions and achieve Net Zero Targets (Committee on Climate Change, 2019). Land use is a resource that can be potentially spared if beef production and consumption is substituted with legume products like the PPBs, with 5% of annual beef consumption in Germany used in protein balls representing a positive difference land use impact of 195 and 688 quadrillion global person equivalents. Water scarcity savings were also substantial, ranging between 1.7 and 5.8 trillion person equivalents in m³ deprivation.

4.5.4. Limitations

Notwithstanding the fact that life cycle inventories for Irish and Brazilian beef could only be sourced from separate databases, the respective environmental profiles correspond with an increasing body of literature on the environmental footprints of European and Latin American beef systems (Nguyen, Hermansen, Mogensen, 2010; Willers et al., 2017). Several studies have identified large differences in the environmental impacts of beef systems (Capper, 2012; Costa et al., 2018;

Pelletier et al., 2010; Picasso et al., 2014). Picasso et al. (2014) compared five different types of systems in Uruguay and found that cattle fed with a combination of improved pasture and feedlot had the largest environmental impacts across six of seven categories assessed, and the smallest impact in the seventh category, GWP. The other systems, native grasslands, improved pasture, or a combination of grassland and improved pasture or grassland and feedlot had lower environmental impacts across the same six impact categories and had a higher GWP. Capper (2012), on the other hand, found that in the United States grass-fed systems (cow-calf, pre-grass and grass finishing) required more inputs and produced more waste than typical American systems (cow-calf, stocker, feedlot, and dairy) with or without growth enhancing technologies. Costa et al. (2018) showed that an integrated livestock-crop system production could decrease land requirement six-fold and halve GHG emissions compared to traditional systems. Most of these results were linked with a more efficient cattle system. In this study, we deliberately looked at beef from major beef-exporting countries previously shown to span the spectrum of carbon footprints (FAO, 2018; A. Leip et al., 2010), including Brazil, which is the largest exporter of beef (Willers et al., 2017), thus evaluating a representative range of beef meatball footprints.

Other limitations of this study include the lack of background LCA data available for seasoning ingredients present in the PPBs and MBs, which could have a significant impact due to energy requirements for processing. Biswas & Naude's (2016) LCA on MB and the PPB inventory did not include trace elements in the study, such as food additives. However, because trace elements were excluded in both products, the relative burdens of products were not affected.

4.6. CONCLUSIONS

Our approach shows the potential of combining environmental and nutritional attributes into quantitative footprints that can be used to assess sustainable food options in the food sector. Our attributional life cycle assessment of 100 g of cooked pea protein balls versus beef meatballs made from Irish or Brazilian beef from cradle to fork showed that pea protein balls have a smaller environmental impact across most impact categories assessed, regardless of the origin of the beef or the allocation method employed. Using a biophysical instead of an economic allocation approach across cattle co-products lowered significantly the environmental footprint of the meatballs. Due to their higher nutrient density, the comparative environmental advantage of pea protein balls over beef meat balls was extended when footprints were expressed per unit of nutrition.

The carbon opportunity cost associated with land occupation is usually not represented within food footprints, yet has an enormous influence on the climate impact of foods. We show

that substituting 5% of Germany's beef consumption with pea protein balls could spare enough land to offset 1% of total annual GHG emissions. Spared land could also be used to produce bio-based feedstocks supporting the transition to a bio-based circular economy. In addition to reducing GHG emissions, substitution of beef meatballs with pea protein balls could lead to large reductions in other forms of resource use and pollution, including energy demand, and reduced emission of health-impacting respiratory inorganics. Through holistic environmental foot-printing and accounting for carbon opportunity costs of land we demonstrate that substitution of beef meat balls with pea protein balls could play an important role on the path to climate neutrality and a more circular economy.

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5. COMPARATIVE LIFE CYCLE ASSESSMENT OF PLANT AND BEEF-BASED PATTIES, INCLUDING CARBON OPPORTUNITY COSTS.

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Keywords: plant protein, meat analogues, nutrient density unit, diet change, climate change, sustainable food

5.1. ABSTRACT

Legume-derived foods have been shown to have comparatively low greenhouse gas (GHG) intensities whilst providing high amounts of nutrients. However, processing legumes into meat analogues can incur significant energy costs. Here, we undertake a comprehensive life cycle assessment of plant-based and (Brazilian and Irish) beef burger patties. Sixteen impact categories are supplemented with the carbon opportunity cost of land occupation, and benchmarked against nutrient density units (NDU) to provide holistic evidence on the potential contribution of plant-based patties to environmentally-sustainable nutritional density. Plant-based patties have a smaller environmental footprint across most categories, including a 77% smaller climate change burden, but incur 8% more energy use compared with Brazilian beef patties. Normalised scores (person equivalents) were significantly larger ($p < 0.05$) for the beef products across key categories including land use, acidification, and marine and terrestrial eutrophication. Sensitivity analyses indicated significant variance across impact categories if beef cattle are reared in South Africa, France or the United States, including a 16-fold difference in land occupation. Biophysical allocation of co-products reduced environmental burdens of beef burgers. However, owing to a 68% higher NDU per serving, reflecting higher fibre and essential fatty acid content, plant-based patties are associated with 81-87% less climate change and 92-95% less marine eutrophication per NDU compared with beef burger patties. Accounting for carbon opportunity cost of land further

increased the climate change advantage of plant-based patties by 25-44%. A simple extrapolation indicates that switching from beef to vegetable patties in the UK could save between 9.5 and 11 million tonnes CO₂e annually, representing up to 2.4% of territorial GHG emissions.

5.2. INTRODUCTION

The previous chapter presented a comparative LCA of pea protein balls versus beef meatballs. Results show that per serving or NDU, PPB production and consumption was associated with lower environmental burdens across all 16 categories assessed, regardless of the allocation approach adopted, and results were significant ($p < 0.05$) across most categories. If 5% of German beef consumption was replaced by pea protein balls, we showed that total carbon savings including carbon opportunity costs of land use could amount to 8 million tonnes CO₂e annually, an amount equal to 1% of Germany's annual GHG emissions. This fifth chapter presents another case study of meat analogue, this time with many more ingredients in the vegetarian option, involving extensive processing. It is a comparative LCA of a pea and soy burger patty with a beef burger patty. Two questions were answered: is the vegetarian patty more sustainable from an environmental and nutritional perspective than the beef equivalent? How does the use of a nutritional functional unit affect the environmental impact of this food product compared to its conventional alternative? The carbon sequestration potential from land use savings if burger patties annually consumed in the United Kingdom were replaced by the vegetarian patties were determined. In addition to the Brazilian and Irish beef systems, three additional cattle rearing inventories were investigated: from France, America, and South Africa.

Beef burger patties are a major component of Western diets, the average adult consuming approximately 1 kg burger patties per year when eating out in the United Kingdom (Department for Environment Food & Rural Affairs, 2020). Due to growing concerns about the effect of excessive red meat consumption on human health and the environment (Gerber et al., 2013; Richi et al., 2015; Steinfeld et al., 2006; Willett et al., 2019), purchases of animal-free alternatives, such as legume-based patties are on the increase (Forbes, 2019).

There are many environmental and health benefits to considering legume-derived foods. Primarily, they have a much reduced greenhouse gas (GHG) intensity (aka "carbon footprints") per unit of nutritional density when compared with other foods (Williams et al., 2020b, 2020a). Legume cropping doesn't require the application of nitrogen fertiliser, by virtue of their capacity to biologically fix atmospheric nitrogen (McCrary et al., 2010; Wagner, 2011). Legume cropping bypasses the inefficient livestock production stage involved in producing animal-derived protein

foods (Nadathur et al., 2017), fertilises soils naturally, and improves soil structure and health (Meena and Lal, 2018), and grain legumes provide a rich source of protein and fibre to the human diet (Foyer et al., 2016).

However, without substantial processing, legumes are not as palatable as meat for many consumers, and current legume consumption in Europe represents only 1% of daily calorie intake (FAO, 2019). Processing legumes into meat analogues that have the same appearance, texture, and taste as meat could support a sustainable and healthy transition away from meat consumption with little consumer effort while meeting growing global protein demand and complying with dietary guidelines (Harwatt et al., 2017; Heller and Keoleian, 2018; Khan et al., 2019; Kumar et al., 2017; Rööß et al., 2018; Stehfest et al., 2009). In the United Kingdom, more than half of consumers over 16 years in age reported eating meat substitutes over the last six months to September 2020, and 55% reported eating alternatives to meat processed products, such as legume-based burgers (Mintel Group Ltd., 2020). The marketing of legume-based products could benefit by environmental and nutritional labelling. There remains a need therefore, to validate the simultaneous environmental and health credentials of these vegetarian alternatives, many of which contain ingredients of varying geographical provenance, and involve a high degree of processing associated with significant environmental burdens.

Life Cycle Assessment (LCA) is a method used to quantify the environmental impacts of a product, taking into account inputs and outputs at all stages of the product life cycle, from the extraction of raw materials to manufacturing, transport, use, and disposal (ISO, 2006). Existing LCA studies comparing vegetarian burger patties with beef patties though, fail to integrate nutritional content as part of the functional unit (Heller and Keoleian, 2018; Khan et al., 2019), despite the fact that improving the nutrition of global diets is one of the key challenges of the food sector today (Willett et al., 2019). Fibre, for example, is a key nutrient lacking in Western diets, and a lack of it is associated with diabetes and obesity (Brennan, 2005; van Dooren et al., 2014), cancer, and heart disease (Kendall et al., 2010). Here, we use the Nutrient Density Unit (NDU) as proposed by van Dooren (2016) as a functional unit to compare the environmental burdens of the meat and vegetarian patties.

Another significant limitation of previous LCA studies on plant versus beef-based patties is their limited approach to calculating the carbon impact of the patties. Existing studies do not include the Carbon Opportunity Costs (COCs) associated with the comparatively large land requirements of cattle rearing as opposed to plant protein production (Heller and Keoleian, 2018; Khan et al., 2019; Lynch and Pierrehumbert, 2019). Land is a critically constrained resource with respect to multiple competing uses in relation to net zero GHG emission targets. As large areas will be needed for GHG

offset activities (Masson-Delmotte et al., 2019), including COCs of land use in the comparison is key in determining the environmental benefits of plant and beef-based patties.

In addition to the carbon footprint limitations of existing comparative LCA studies of meat substitutes, the range of environmental impacts investigated is scarce, looking solely at carbon footprint, aquatic eutrophication, non-renewable energy use, land occupation, and water consumption (Heller and Keoleian, 2018; Khan et al., 2019). The Product Environmental Footprint (PEF) guidelines used in our study follow a standardisation initiative and offer a much more complete impact assessment package (European Commission, 2018). Additionally, although different food production methods have a wide range of environmental impacts, especially for beef systems (Poore and Nemecek, 2018), the existing comparative studies of meat and meat substitutes only include one beef system. To address this limitation, this study assesses different cattle rearing systems, from Brazil, Ireland, South Africa, France, and America.

The purpose of our study was to determine whether the consumption of a vegetarian burger patty is associated with a lower environmental impact than that of a burger patty made with beef from different cattle rearing systems, based on their nutrient density. Furthermore, we expand the boundaries to include carbon opportunity costs associated to the production of the different burger alternatives, and contextualise their impacts to achieve the net zero GHG emission and wider climate neutrality targets of the United Kingdom (Shepherd, 2020).

5.3. MATERIALS AND METHODS

5.3.1. Goal, scope, and boundary definition

This study is a comparative assertion of the overall environmental impacts arising from the consumption of a vegetarian burger patty (VB) with a conventional beef burger (BB) with Irish (IE) or Brazilian (BR) beef, accounting for their nutritional functionality. Inputs and outputs for all processes involved in the life cycle of the VB were recorded, but due to intellectual property protection, ingredients were grouped into categories of product in Table A.1 of the Appendix. The aim was to assess the relative environmental performance of the three products by performing an attributional LCA. Three central products were compared:

- 1) VB – 4 oz burger patty made in the UK with plant-based ingredients sourced globally.
- 2) BB (IE) – 4 oz burger patty made with Irish beef in the UK.
- 3) BB (BR) – 4 oz burger patty made with Brazilian beef in the UK.

The open-source software OpenLCA v1.10.2 (GreenDelta, 2020) was used to calculate the environmental footprint of the two products from cradle to fork, using Agrifootprint v3.0 (Durlinger et al., 2017), Ecoinvent v3.6 (Wernet et al., 2016) international databases. End of life was excluded from the analysis, as it was assumed to be similar for all three products, and not a major contributor to life cycle burdens of meat products. Inventory data for the VB were collected specifically for this study from a British company manufacturing the VB (Pers. Comm., 2019). Data on the BB were modelled as though the burgers were produced in the UK with beef received from either Brazil or Ireland using secondary data. Economic allocation was used to allocate burdens between the beef co-products and the processing co-products of the VB. Economic allocation percentages for beef co-products were extracted from the PEF guidelines (European Commission, 2018). As a sensitivity analysis, the environmental performance of the three products were also compared when using a simple mass allocation used for VB co-products, and biophysical allocation for the BB (IE) and BB (BR) beef co-products (Chen et al., 2017). Allocation factors are reported in Table A.2 of the Appendix. Biophysical allocation represents allocation of co-products based on energy flows and other causal relationships (Chen et al., 2017). To determine whether results were statistically significant, a modified Null Hypothesis at the additional beef systems, the Bonferroni correction was $\alpha_b = 0.05/96 = 0.0005208$ due to the 6 pairs of alternatives. The effect size was set at $\delta_0=0.2$.

Figure 1 illustrates the system boundaries for the cradle to fork assessment of the VB and BBs (IE and BR). The environmental footprints of the burgers were assessed across the sixteen environmental impact categories recommended in the PEF Category Rules Guidance (European Commission, 2018). To assist in the interpretation of environmental burdens across several categories, results were normalised by global person equivalents with the PEF recommended factors to generate comparable normalised scores across impact categories (European Commission, 2018).

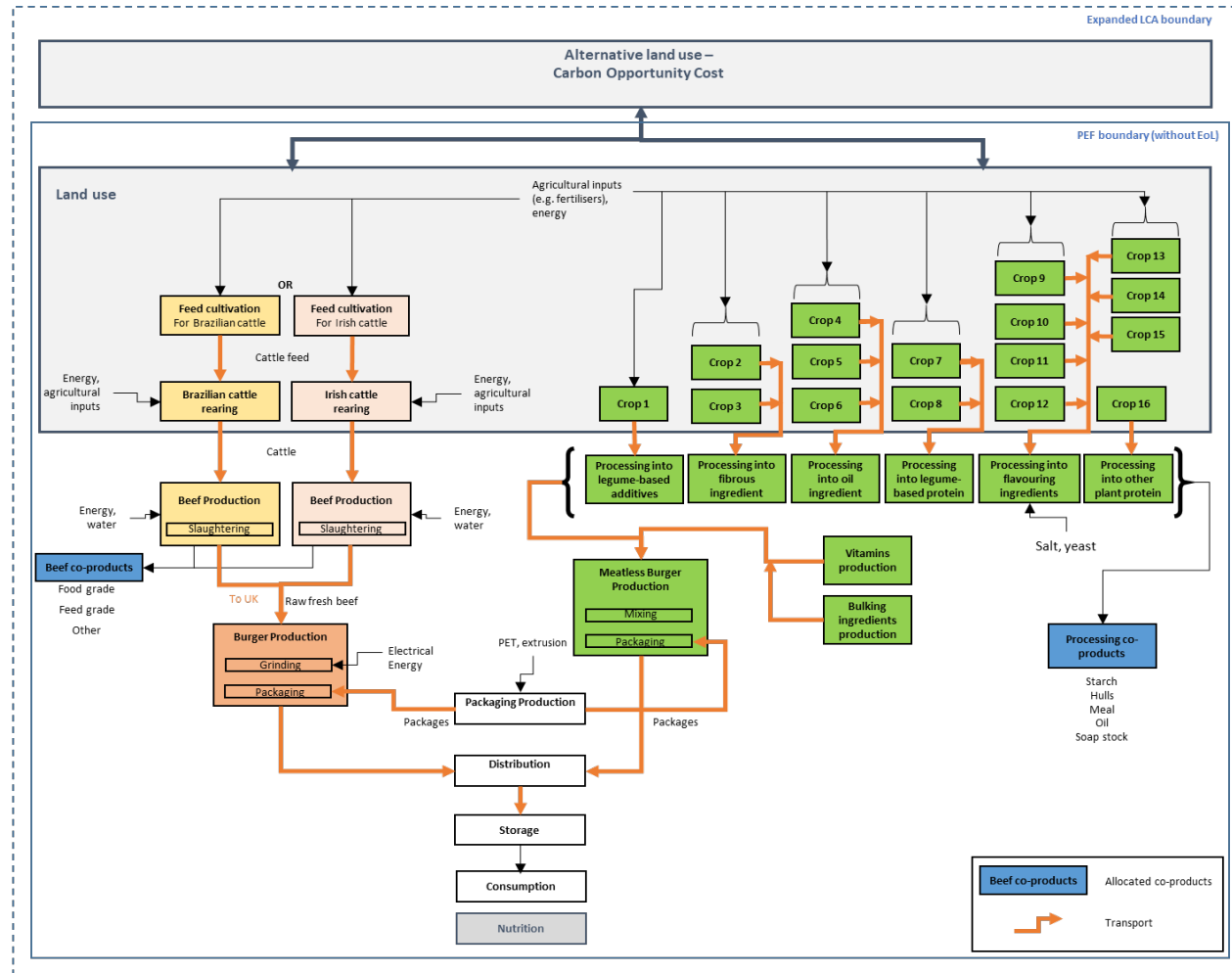


Figure 1: System boundary of vegetarian (VB) and beef burger production BB (BR and IE), from cradle to fork. Beef burger production is represented in red (left), vegetarian burger in green (right), and the bottom middle represents the common stages to both products in white.

5.3.2. Functional Units

Two functional units (FUs) were used. First, a simple weight FU expressed as a single 4 oz burger patty was applied, as per recent vegetarian burger LCAs (Heller and Keoleian, 2018; Khan et al., 2019). Second, the Nutrient Density Unit (NDU) was used to account for differences in nutrient density between products (Saget et al., 2020; Van Dooren, 2016; Williams et al., 2020b, 2020a). The use of the latter was to ensure that products were compared per unit of nutrient density, which favours foods that have a high ratio of nutrients to energy. The NDU correlates with the more complete Nutrient Rich Foods index 12.3 (Saget et al., 2020). A nutritional analysis of the VB and BBs was performed with the pan-broiled method with one sample in triplicates, so that no additional fat was added and nothing else but cooking itself would alter the nutritional values. Protein content was analysed using the Kieldahl method (ISO 1871:2009) (ISO, 2009), energy content following the EU regulation 1169/2011 (European Union, 2011), fibre content using the Enzymatic-Gravimetric Method from the AOAC 991.43 and AOAC 985.29 (Lee et al., 1992; Prosky et al., 1985), and essential fatty acids content using gas chromatography (FID) from ISO 12966-1:2014; 12966-2:2011; 12966-3:2016 (ISO, 2016, 2014, 2011). It was assumed that the nutritional content of BBs was the same, regardless of the geographical origin of beef (BR or IE).

The NDU was applied following Van Dooren's (2017) formula (1).

$$NDU = \frac{\left(\frac{EFA}{DV_{EFA}}\right) + \left(\frac{Protein}{DV_{prot}}\right) + \left(\frac{Fibre}{DV_{fibre}}\right)}{3 \times \left(\frac{S_i}{2000 \text{ kcal}}\right)} \quad (1)$$

Nutritional results for the pan-broiled products, NDU results for the cooked VB and for the cooked BB (IE and BR) are presented in Table 1.

Table 1: Summary of the nutritional composition of 100 g of the vegetarian and beef burger patties, pan-broiled.

Content per 100 g cooked	VB	BB (IE and BR)
Energy (kcal)	210	248
Protein (g)	15.9	23.4
Dietary fibre (g)	5.8	0
EFA (g)	2.1	0.40
NDU	2.28	1.35
NDU (adjusted)	2.20	Non applicable

The energy content of the two products was similar, with 210 kcal per 100 g VB, cooked, versus 248 kcal per 100 g BB (IE and BR), cooked. Nutritional content varied greatly between the

VB and BB (IE and BR), with the BBs (IE and BR) having 32% more protein, but no dietary fibre versus 5.8 g of fibre per 100 g cooked VB, and the VB having 81% more essential fatty acids.

The WHO (2007) states that adults have a net protein utilization that is comparable across all protein types. Moreover, the BB and VB assessed are sold in countries in which protein requirements are largely exceeded. Therefore, protein digestibility should not be of concern when comparing the two products. Nevertheless, we calculated the NDU of the VB with adjusted protein digestibility, assuming all protein in the VB originates from peas for simplicity. Pea concentrate protein digestibility amounts to 92% (Gilani and Lee, 2003), versus 100% digestibility assumed for animal protein. Therefore, the NDU for the VB was adjusted to account for 14.6 grams digestible protein per serving (Equation 4), versus 15.9 grams when not adjusted. The adjusted NDU delivered by a VB portion is 2.20 (versus 2.28 unadjusted).

5.3.3. Vegetarian burger inventory

The VB is made of 16 different crops, with legume-based additives, fibrous and oil ingredients, legume- and cereal-based proteins, flavouring and seasoning ingredients, bulking ingredients, and vitamins. The VB ingredients come from various countries worldwide, and often are shipped to a first factory where they are processed, then shipped to the UK. In the UK, the ingredients are assembled and the obtained burger is packaged. The amount of energy required to mix the ingredients and form the patty was extracted from Davis & Sonesson (2008). The VB is then sent to the point of sale, with 73% of the products sold within the United Kingdom, 5% in other European countries, and 22% outside of Europe (Pers. Comm., 2019). Transportation modes and distances were defined using the PEF guidelines, using Searates (2020) to estimate transoceanic distances. Land transport distances were calculated using centroids of countries (Google, n.d.). Following the PEF guidelines (European Commission, 2018), 0.38 km of consumer transport from the home to the retail centre was attributed to the patty component of an average shopping trip. The VB is then stored for two days in the fridge, requiring 0.099 Wh of energy, calculated with the formula from EPD International AB (2019). The consumption phase was modelled following EPD International AB (2019), with a cooking time of 3 minutes each side for the VB, using 0.55 kWh. Trace elements that represent less than 0.5% of the VB were excluded from the LCA, due to lack of data on the environmental impact of these elements.

Table 2: Inventories for a 4 oz (113 g) vegetarian (VB) and beef burgers (Irish and Brazilian beef), from “field to fork”

Stage	Input / output / process	Units	VB		BB (IE)		BB (BR)		Source of information
			Input	Output	Input	Output	Input	Output	
Beef production and transport	Fertiliser (N) w/o grain and meal	kg			0.020		0.007		Database
	Urea, as N w/o grain and meal	kg			0.005		0.005		Database
	Fertiliser (P ₂ O ₅) w/o grain and meal	kg			0.003		0.012		Database
	Fertiliser (K ₂ O) w/o grain and meal	kg			0.003		0.010		Database
	Feed for cattle (grain+meal)	kg			0.64		0.067		Database
	Manure, from cow	kg			1.6				Database
	Lime	kg			0.1		0.2		Database
	Land, total of crops and grazing	m ²			7.7		17.3		Database
	Water	L			0.3		0.03		Database
	Beef cattle for slaughter	kg				0.23		0.23	Database
	Beef transport, lorry with refrigeration	kg.km			14.7		113.4		(European Commission, 2018)
	Beef transport, train with refrigeration	kg.km			27.2				(European Commission, 2018)
	Beef transport, barge/ship with refrigeration	kg.km			30.6		1136		(European Commission, 2018)
Legume-based additive	Legume-based additive	kg	0.0011						Company
	Transport, transoceanic ship with	kg*km	1829						Company
	Transport, lorry with refrigeration machine	kg*km	295						Company
Fibrous ingredients	Fibrous ingredients	kg	0.2						Company
	Transport, lorry >32t	kg*km	7.7						Company
	Transport, train	kg*km	1.6						Company
	Transport, barge	kg*km	1.8						Company
Oil ingredients	Oil ingredients	kg	0.009						Company
	Transport, lorry >32t	kg*km	3.8						Company
	Transport, train	kg*km	2.1						Company
	Transport, transoceanic ship	kg*km	47.8						Company
	Transport, barge	kg*km	2.4						Company
Legume-based protein	Protein extract	kg	0.030						Company
	Protein extract transport, lorry, with	kg*km	23.4						Company
	Protein transport, transoceanic ship, with	kg*km	171.5						Company
	Protein extract transport, train, with	kg*km	2.7						Company
	Protein extract transport, barge, with	kg*km	3.1						Company
Cereal-based protein	Protein extract	kg	0.006						Company
	Protein extract, >32t lorry	kg*km	0.7						Company
	Protein extract transport, barge	kg*km	1.5						Company
	Protein extract transport, train	kg*km	1.4						Company
Flavouring and seasoning ingredients	Flavouring & seasoning ingredients	kg	0.037						Company
	Heat, natural gas	kWh	0.028						Company
	Flavouring & seasoning transport, >32t lorry	kg*km	23						Company
	Flavouring & seasoning transport, barge	kg*km	43.5						Company
	Flavouring & seasoning transport, train	kg*km	38.7						Company
Bulking ingredients	Bulking ingredient	kg	0.012						Company
	Bulking ingredient transport, >32t lorry	kg*km	11.4						Company
	Bulking ingredients transport, transoceanic	kg*km	68.5						Company
	Vitamins	kg	0.0015						Company

Vitamins	Vitamin transport, >32t lorry	kg*km	1.4			Company
	Vitamin transport, transoceanic ship, with	kg*km	27.2			Company
	Vitamin transport, train, with refrigeration	kg*km	0.11			Company
	Vitamin transport, barge, with refrigeration	kg*km	0.13			Company
Packaging	Extrusion, plastic film	g	6.55	6.55	6.55	Company
	Folding boxboard production	g	4.05	4.05	4.05	Company
	Packaging transport, lorry, >32t	kg*km	1.56	1.56	1.56	Company
	Packaging transport, train	kg*km	2.97	2.97	2.97	Company
	Packaging transport, ship	kg*km	3.82	3.82	3.82	Company
Burger production	Energy for grinding/mixing electricity	MJ	4.20	4.20	4.20	(Kamdem and Hardy, 1995)
Distribution	Burger transport, lorry with refrigeration	kg*km	130.3	130.3	130.3	(European Commission, 2018)
	Burger transport, transoceanic ship with	kg*km	491.0	491.0	491.0	(European Commission, 2018)
	Burger transport, passenger car	km	0.159	0.159	0.159	(European Commission, 2018)
Storage	Energy for cooling	kWh	0.099	0.099	0.099	(European Commission, 2018)
Cooking	Energy for cooking gas	kWh	0.28	0.69	0.69	(European Commission, 2018)
	Energy for cooking electricity	kWh	0.09	0.23	0.23	(European Commission, 2018)
	Cooked burger patty	Item	1	1	1	

5.3.4. Beef burger inventory

Beef of Irish origin was assessed since it represents 38% of beef imports to the UK, with a total of 384 thousand tonnes of Irish beef import reported by the UK (United Nations, 2020). Brazilian beef was also assessed, as Brazil is the largest international exporter of beef and is targeting greater exports into European countries, for example with the Mercosur agreement (European Commission, 2019; Zu Ermgassen et al., 2020). These two origins of beef are representative of different grass-fed systems. Sensitivity analyses were also run using beef from cattle systems in South Africa, France, and the United States in order to show the full range of possible outcomes for BBs. Data on the VB was collected from a manufacturing company in the UK. The BBs modelled were made of 100% beef using secondary life cycle inventory data from Agri-footprint v3.0 and Ecoinvent v3.6 (Durlinger et al., 2017; Wernet et al., 2016). Farm level processes and transport to and from the slaughterhouse are specific to BB (IE) and BB (BR). For the BB (IE), the beef system used in the inventory was of an average beef farm in Ireland from Agrifootprint v3.0 (Durlinger et al., 2017). For the BB (BR), the beef system is a representative combination of several cattle farming systems in Brazil. Intensive beef cattle and fat steers rearing dominated the market, representing half of the market process. An amount of 4 oz (0.113 kg) raw fresh beef requires 0.247 kg of beef cattle live weight. This involves 0.69 kg of compound feed made with wheat, barley, corn, oats, rapeseed, soybean, and sugar cane products. In addition to the compound feed, 2.58 kg of grass silage and 13.1 kg of grazed grass over two years complete the cattle diet to obtain the 4 oz (0.113 kg) patty. A distance of 100 km is covered by truck to transport the cows between the farm and the slaughterhouse (Durlinger et al., 2017). The slaughtering process of 0.247 kg of cattle produces 0.113 kg of fresh meat, the rest being category 1, 2, and 3 co-products, food grade bones and fat, and hides and skins. Beef transport from the slaughterhouse to the factory was modelled according to the PEF guidelines (European Commission, 2018). To grind the meat into a burger, 4.2 kJ of electricity were used, based on Kamdem & Hardy (1995). The obtained BB is then packaged, transported, and stored as described in Section 2.3. Energy use during the cooking stage was calculated based on a cooking time of 5 minutes on each side for the BB, using 0.92 kWh of electricity (EPD International AB, 2019a).

Because of the diversity of cattle rearing systems globally, a sensitivity analysis was run with additional cattle systems from South Africa (ZA), France (FR), and the United States (US) across the categories in which at least one burger patty had a burden of at least $5E-4$ global person equivalents, as determined by our assessment in the first part of the publication (Section 3.1). These beef cattle systems were selected due to the limited availability of cattle data in existing databases. The South African system was taken from Ecoinvent v3.6 (Wernet et al., 2016), and is representative of large commercial feedlots, with a combination of pastures and pens. Pastures are mainly natural,

with few inputs. The French system was taken from Agribalyse v3.0 (ADEME, 2020) and is a mix of conventional production in extensive and semi-intensive grassland areas. The US system was taken from the EF database (Green Delta, 2019) and is characterised by intensive systems with no grazing in open-front barns. These cattle systems were then connected to the burger manufacturing and consumption steps, with transport from the beef country of origin to the UK calculated following the same methodology as for the BB (IE) and BB (BR). As a result, three additional BB scenarios were created for sensitivity analyses:

- 1) BB (FR) – 4 oz burger patty made with French beef in the UK.
- 2) BB (ZA) – 4 oz burger patty made with South African beef in the UK.
- 3) BB (US) – 4 oz burger patty made with US beef in the UK.

5.3.5. Carbon opportunity costs for UK scenario

The average person in the UK consumes around 1.06 kg of burger patty yearly (Department for Environment Food & Rural Affairs, 2020). Assuming all beef burger patties consumed by the UK population of 67 million (Office for National Statistics, 2020) could be substituted with vegetable patties, we calculated the potential carbon sequestration achieved by the resulting land sparing (Searchinger et al., 2018). For this, we utilised the COC data per ingredient type defined by Searchinger et al. (2018): 144 kg CO₂ eq per kg of beef, and at most 10.5 kg CO₂ eq per kg ingredient used in the VB.

5.4. RESULTS

5.4.1. General results

The environmental impact results of the three products across the sixteen categories are recorded in Table 3 for the two functional units described previously. Per patty, The VB had a significantly lower ($p < 0.05$) environmental burden across 9 categories out of 16 per when compared to both BBs (IE and BR). The VB patty was associated with an environmental burden that was between 67% and 95% smaller than the BB (IE) and between 77% and 96% smaller than the BB (BR) in the climate change, marine and terrestrial eutrophication categories, land use, and acidification categories, respectively. Including COC increased the climate change burdens of the BBs by 15.2 kg CO₂e, and the VB by 1.5 kg CO₂e, resulting in burdens 3.3, 2.3, and 1 times higher than without COC for the BB (IE), BB (BR), and VB, respectively. The VB was not associated with significantly higher ($p < 0.05$) environmental burdens across any categories when compared to BBs (IE and BR), although its energy use resource carriers burdens were insignificantly different to those of the BB (BR).

The wide discrepancy in nutritional composition between the BBs (IE and BR) and VB resulted in significantly different NDU values. Due to a comparatively higher NDU even after adjustment for protein digestibility, the VB had a significantly lower ($p < 0.05$) environmental burden across 11 categories out of 16 per NDU (adjusted) when compared to both NDUs of BBs (IE and BR). One VB NDU was associated with an environmental burden that was between 80% and 97% smaller than the BB (IE) and 79% and 98% smaller than the BB (BR) in the climate change, marine and terrestrial eutrophication categories, land use, and acidification categories, respectively.

Table 3: Summary of environmental burdens for the 4 oz (0.113 kg) vegetarian burger (VB) and beef burgers (BB) made with beef from Ireland or Brazil, expressed per cooked product and per NDU with economic allocation performed for co-products. Cells in green indicate that the product has a significantly lower environmental burden ($p < 0.05$) than the other two alternatives in the respective impact category, while cells in red indicate that the product has a significantly higher environmental burden ($p < 0.05$) than the other two alternatives in that category.

Impact Category	Unit	Impact per 4 oz burger, cooked			Impact per NDU		
		BB (IE)	BB (BR)	VB	BB (IE)	BB (BR)	VB (adj.)
Acidification ter. & freshwater	mol H ⁺ eq	0.109	0.025	0.009	0.080	0.018	0.004
Cancer human health	CTUh	7.1×10^{-8}	1.3×10^{-6}	1.4×10^{-8}	5.3×10^{-8}	9.4×10^{-7}	6.5×10^{-9}
Climate change (w/o COC)	kg CO ₂ eq	4.5	6.6	1.5	3.4	4.9	0.7
Climate change (with COC)	kg CO ₂ eq	19.7	21.8	3.0	14.6	16.1	1.4
Ecotoxicity freshwater	CTUe	8.4	436.8	23.9	6.2	323.6	10.9
Eutrophication freshwater	kg P eq	0.0008	0.0003	0.0003	0.0006	0.0002	0.0001
Eutrophication marine	kg N eq	0.045	0.029	0.004	0.034	0.022	0.002
Eutrophication terrestrial	mol N eq	0.48	0.10	0.02	0.35	0.08	0.01
Ionising radiation, HH	kBq U-235 eq	0.34	0.35	0.24	0.25	0.26	0.11
Land use	Point	561	1906	75	415	1411	34
Non-cancer human health	CTUh	3.7×10^{-6}	6.8×10^{-4}	3.3×10^{-7}	2.8×10^{-6}	5.0×10^{-4}	1.5×10^{-7}
Ozone depletion	kg CFC11 eq	6.7×10^{-8}	1.0×10^{-7}	9.1×10^{-8}	5.0×10^{-8}	7.4×10^{-8}	4.1×10^{-8}
Photochem. ozone form.	kg NMVOC eq	0.011	0.007	0.005	0.008	0.005	0.002
Resource use, energy carriers	MJ	21.7	14.4	15.5	16.1	10.7	7.1
Resource use mins. & metals	kg Sb eq	4.7×10^{-7}	4.4×10^{-9}	3.7×10^{-8}	3.5×10^{-7}	3.3×10^{-9}	1.7×10^{-8}
Respiratory inorganics	disease inc.	7.8×10^{-7}	1.7×10^{-7}	7.1E-08	5.8×10^{-7}	1.3×10^{-7}	3.2×10^{-8}
Water scarcity	m ³ depriv.	1.39	0.41	0.40	1.03	0.31	0.18

To determine which categories were the most relevant in the comparison, environmental burdens normalised per person equivalents across all categories (except for the toxicity-related categories owing to uncertain normalisation data) are shown per burger patty in Figure 2. Categories with scores that exceeded $5\text{E-}4$ person equivalents in at least one product were the land use, marine and terrestrial eutrophication, acidification, respiratory inorganics, and climate change. Normalised scores were comparatively lower for the few impact categories where VB had higher burdens than the BBs (resource use minerals and metals and ozone depletion).

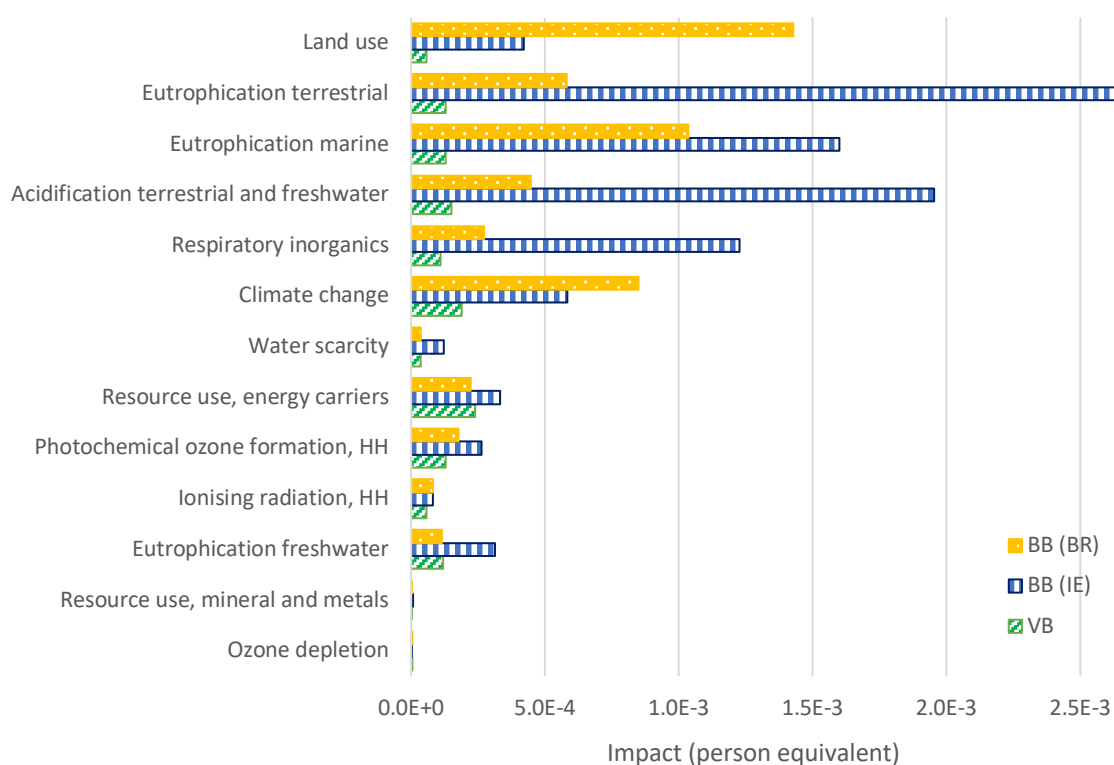


Figure 2: Normalised environmental burdens of a vegetarian (VG) or beef patty from Irish BB (IE) or Brazilian beef BB (BR) across 13 impact categories per 4 oz burger Toxicity-related impact categories were excluded from the graphs.

5.4.2. Process contributions

Process contributions across the sixteen environmental impact categories for the burger patties are recorded in Figure 3. Beef production is responsible for most of the burdens across all categories in the BBs. It was responsible for at least 75% of the total climate change, land use, marine and terrestrial eutrophication, cancer and non-cancer human health, freshwater ecotoxicity, resource use minerals and metals, respiratory inorganics, water scarcity, and acidification burdens. The climate change burden of the BB was due mostly to enteric methane

emissions from cattle, a significant proportion of acidification and terrestrial eutrophication due to ammonia emission to air from cattle raising and grass cultivation, and the high land use for raising the cattle (pastures) and growing all the crops for feed.

For the VB burger, flavouring and seasoning contributed to at least 20% of the total burdens across 8 categories out of 16. The climate change, photochemical ozone formation human health, resource use energy carriers and freshwater eutrophication burdens from flavouring were partly due to heat for production of one natural raw material. This aspect of the life cycle of the VB was the main reason for an insignificant difference between the energy use resource carriers burdens of VB and those of BB (BR), as mentioned in Section 3.1. The cereal-based protein production also contributed to at least 20% of the total burdens across 8 categories out of 16. It represented 25%, 21%, 18%, 43%, 31%, and 36% of total acidification, cancer human health, climate change, resource use minerals and metals, and terrestrial and marine eutrophication, respectively. The high freshwater ecotoxicity burden in the VB was due to cereal-based protein cultivation, which emitted cypermethrin, a neurotoxin insecticide, to the water.

It should be noted that the burger production happened to take place in the UK, and that producing it elsewhere would have little impact on the overall environmental performance of the burger patties assessed in this study, as the only process using British data was the electricity required for assembling the ingredients/grinding the beef, for which the electricity amount was assumed to be the same across all products. Transport was responsible to a relatively low share of the overall environmental burdens across all categories.

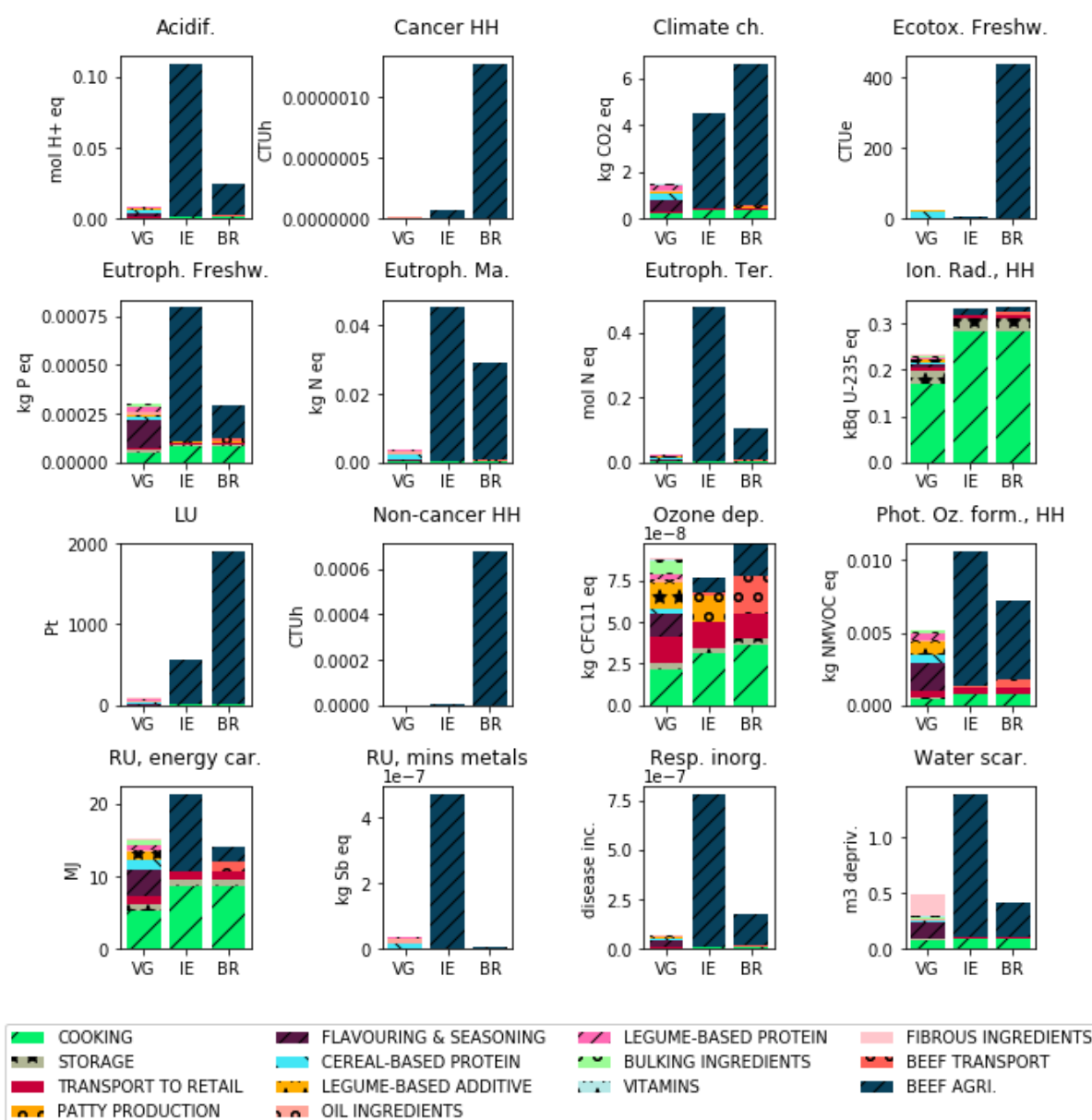


Figure 3: Process contributions of one 4 oz (0.113 kg) patty cooked, from Irish beef (IE), Brazilian beef (BR), and vegetarian ingredients (VB) across 16 impact categories with economically-allocated burdens for beef production and plant ingredients processing co-products.

5.4.3. Extrapolation scenario results

Substituting the consumption of 19 grams of BB per week over the period of one year across the entire UK population with the consumption of VB could derive considerable environmental savings. Acidification saving could amount to 4 to 56 million mol H⁺, and marine eutrophication between 13 and 23 thousand tonnes N equivalents.

The climate change burden including COC associated with the scenario in which all beef burgers were from Irish or Brazilian beef was of 12 or 14 million tonnes CO₂e, respectively. This represents a positive difference of 9.5 to 11 million tonnes CO₂e when compared to the equivalent amount of

VB burgers consumed (eating the same quantity of VB burgers would represent 2.8 million tonnes CO₂e). This amount is equal to 2.1 to 2.4% of the annual territorial GHG emissions in the UK in 2019 (Department for Business, 2020).

5.4.4. Sensitivity analyses

Table 4 represents the environmental impact of one VB with physically-allocated burdens for the protein concentrates, oils, starches, and emulsifiers co-products, and one BB (IE and BR) biophysically-allocated between beef and beef processing co-products. Per patty, The VB had a significantly lower ($p < 0.05$) environmental burden across 6 categories out of 16, and significantly higher photochemical ozone formation and resource use energy carriers burdens when compared to both BBs (IE and BR). Biophysical allocation between beef and beef co-products greatly reduced the cattle rearing burdens attributable to beef (Table 5). For example, beef carbon footprints reduced by 63 and 65% when shifting from economic to biophysical allocation for the BB (IE) and BB (BR), respectively. In comparison, going from economic allocation to physical allocation of VB co-products decreased environmental burdens by a smaller magnitude (Table 5). For example, the climate change burden of the VB decreased by 10% when going from economically allocated burdens to physically allocated ones. On the other hand, the energy use resource carriers burden of the VB was higher than the one of both the BBs (IE and BR) when physical allocation was applied, due to a 35% or 10% reduction of the BBs footprints. Per NDU, however, the VB had no significantly higher burdens per NDU when compared to both NDUs of BBs (IE and BR).

The results of the modified NHST that were significantly ($p < 0.05$) higher or lower than the other alternatives and that were common to all four modelling choices (economic allocation with weight or NDU functional units, and (bio)physical allocation with weight or NDU functional units) were recorded in Table A.3 of the Appendix. The VB alternative had a significantly lower environmental burden than both the BBs (IE and BR) across 6 categories out of 16. The VB had significantly higher cancer human health and ecotoxicity freshwater burdens than the BB (IE) for all scenarios.

Table 4: Summary of environmental burdens of the burger patties with biophysically allocated beef co-products for the beef patties (Irish and Brazilian) physically allocated co-products of the vegetarian patty, expressed per burger patty and per Nutrient Density Unit adjusted, cooked. Cells in green indicate that the product has a significantly lower environmental burden ($p < 0.05$) than the other two alternatives in the respective impact category, while cells in red indicate that the product has a significantly higher environmental burden ($p < 0.05$) than the other two alternatives in that category.

Impact Category	Unit	Impact per 4 oz burger, cooked			Impact per NDU		
		BB (IE)	BB (BR)	VB	BB (IE)	BB (BR)	VB (adj.)
Acidification terrestrial and freshwater	mol H ⁺ eq	0.033	0.009	0.008	0.024	0.007	0.004
Cancer human health effects	CTUh	2.5×10^{-8}	3.7×10^{-7}	1.3×10^{-8}	1.9×10^{-8}	2.8×10^{-7}	6.0×10^{-9}
Climate change (w/o COC)	kg CO ₂ eq	1.7	2.3	1.3	1.2	1.7	0.6
Climate change (with COC)	kg CO ₂ eq	6.0	6.7	2.3	4.5	5.0	1.0
Ecotoxicity freshwater	CTUe	3	126	4	2	94	2
Eutrophication freshwater	kg P eq	0.0003	0.0002	0.0003	0.0002	0.0001	0.0001
Eutrophication marine	kg N eq	0.013	0.009	0.003	0.010	0.007	0.001
Eutrophication terrestrial	mol N eq	0.14	0.03	0.02	0.10	0.03	0.01
Ionising radiation, HH	kBq U-235 eq	0.33	0.34	0.25	0.25	0.25	0.11
Land use	Point	171	559	84	126	414	38
Non-cancer human health effects	CTUh	1.1×10^{-6}	0.0002	2.3×10^{-7}	8.3×10^{-7}	1.5×10^{-4}	1.0×10^{-7}
Ozone depletion	kg CFC11 eq	6.1×10^{-8}	8.6×10^{-8}	1.2×10^{-7}	4.5×10^{-8}	6.3×10^{-8}	5.4×10^{-8}
Photochemical ozone formation, HH	kg NMVOC eq	0.004	0.003	0.006	0.0030	0.0025	2.7×10^{-3}
Resource use, energy carriers	MJ	14.1	12.9	16.0	10.5	9.6	7.3
Resource use, mineral and metals	kg Sb eq	1.4×10^{-7}	1.7×10^{-9}	1.9×10^{-8}	1.0×10^{-7}	1.3×10^{-9}	8.4×10^{-9}
Respiratory inorganics	disease inc.	2.3×10^{-7}	6.0×10^{-8}	6.6×10^{-8}	1.7×10^{-7}	4.4×10^{-8}	3.0×10^{-8}
Water scarcity	m ³ depriv.	0.47	0.20	0.83	0.35	0.14	0.38

Table 5: Summary of the climate change and resource use energy carriers footprints of the burger patties obtained with different allocation methods (economic and physical), different functional units (serving and nutrient density unit), and different beef origins (Irish and Brazilian), as well as the vegetarian product.

	Functional Unit	Allocation method	Impact per 4 oz burger, cooked		
			BB (IE)	BB (BR)	VB
Climate change (kg CO ₂ eq)	Serving	Economic	4.5	6.6	1.5
		(Bio)physical	1.7	2.3	1.3
	Nutrient Density Unit	Economic	3.4	4.9	0.7
		(Bio)physical	1.2	1.7	0.6
Resource use energy carriers (MJ)	Serving	Economic	21.7	14.4	15.5
		(Bio)physical	14.1	12.9	16.0
	Nutrient Density Unit	Economic	16.1	10.7	7.1
		(Bio)physical	10.5	9.6	7.3

Figure 4 displays the relative environmental burdens arising from the production and consumption of a VB or a BB made of French, American, South African, Irish, and Brazilian beef across the categories in which at least one burger patty had a burden of at least $5E-4$ global person equivalents. The water scarcity burden of the French cattle system was excluded due to the water data being not calculated across the full cattle life cycle (ADEME, 2020). Results of the modified NHST for these burgers were recorded in Table A.4 of the Appendix. Categories that were excluded because burdens from all systems were below the defined threshold were ozone depletion and ionising radiation human health. In addition to these excluded categories, the toxicity-related categories were also excluded, following the PEF guidelines, owing to uncertain normalisation data (European Commission, 2018).

The two products that had a higher footprint than 0.0015 person equivalents across some categories were the BB (ZA) and BB (IE). The BB (ZA) appeared to have the highest land use footprint, with a burden that was between 0.7 and 5 times higher than the other BBs. The land use footprint of BB (ZA) was 43 times higher than that of the VB. Freshwater eutrophication was also the highest for the BB (ZA), with a burden between 3 and 16 times higher than that of all patties. The BB (IE) had a terrestrial eutrophication burden between 33 and 363% higher than that of all beef patties. The BB (US) appeared to have the highest burdens across water scarcity and energy use, resource carriers. The BB (FR) did not have the highest burden across any of the high impact categories displayed in Figure 4, although data was missing across the water scarcity category.

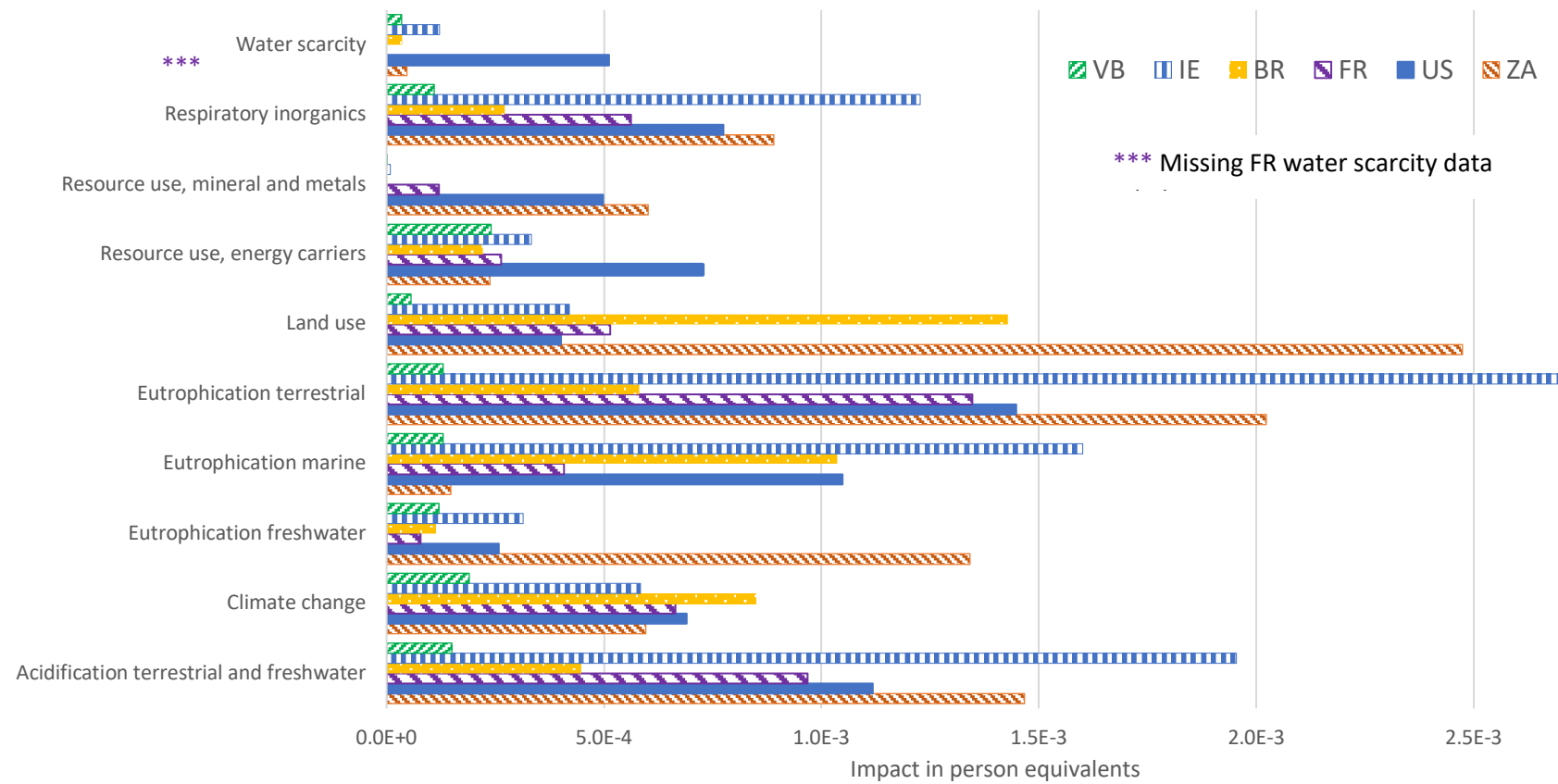


Figure 4: Environmental impact per 4 oz burger of a vegetarian (VB) or beef patty from Irish BB (IE), Brazilian BB (BR), French BB (FR), American BB (US), or South African beef BB (ZA) across the nine impact categories with the highest environmental burdens, excluding the toxicity-related categories.

5.5. DISCUSSION

5.5.1. Superior nutrient density with lower environmental impacts

Overall, the VB is more nutrient dense whilst incurring a dramatically smaller environmental footprint across most impact categories than the BB. In addition to the higher environmental burdens in most categories incurred by the BB comes a comparatively lower NDU, due to a relatively higher protein content not compensating enough for the absence of dietary fibre and less essential fatty acids in the BB. To these nutritional contents, one can add the negative health effects stemming from red meat consumption mentioned in the introduction, namely cardiovascular diseases, cancer, and type 2 diabetes. Therefore, from a nutritional stance alone VB may carry a significant advantage over BB. Using an energy content FU (e.g. kcal) as a proxy for nutritional delivery can be misleading, as it results in the favouring of energy-dense nutrient-poor foods. Moreover, the vegetarian and beef burgers in this study have comparable amounts of kilocalories, hence the resulting environmental burdens per energy content would not yield significant differences when comparing with the serving-based FU. Using a simple protein FU as a proxy for nutritional delivery (Nijdam et al., 2012; Sonesson et al., 2017) would have been misleading, as the BB contains more protein than the VB, and as discussed previously, protein is not a limiting nutrient in developed countries, whilst nutrients such as fibre are lacking in modern diets. Vegetarian burger patties are alternatives to beef patties that consumers can opt for at the supermarket. With the double challenge of malnutrition and large negative environmental consequences of the food system in Europe, utilising the NDU as a FU allowed the integration of nutritional aspects for food alternatives in a more complete way than a limited energy content or single nutrient FU, yet in a more simple way than more extensive indices like the NRF 9.3 (Fulgoni et al., 2009). It would nevertheless be relevant to explore other nutritional FUs that are specifically adapted to the nutritional needs of the European population.

Besides the inclusion of nutritional contents in the FU, other approaches to integrate nutritional performance in LCA - and ultimately health effects - exist. One is to link epidemiological data to nutrients (Stylianou et al., 2015; Weidema and Stylianou, 2019), attributing health burdens scores to foods based on dietary risk factors. However, robustness of epidemiological data needs to be improved, and may differ between different populations.

The VB belongs to Group 4 in the NOVA classification, and is thus an ultra-processed food. However, in contradiction with the description of ultra-processed foods in (Monteiro et al., 2018), this study showed that an ultra-processed food can be nutrient-dense, and a higher source of dietary fibre and essential fatty acids when compared to its direct meat alternative. Nevertheless, it would be relevant to analyse and compare the burgers' contents of saturated fats, trans-fats, free

sugars, and key micronutrients, as these are typically decreased in ultra-processed foods (Monteiro et al., 2018). It is important to keep in mind that such a vegetarian patty could contribute towards a transition to eating less meat and more whole, plant-based foods (Hu, 2003).

For those categories in which the VB was associated with higher burdens, significant improvements could be achieved by modifying the flavouring/seasoning mix and the cereal-based protein. Substituting the cereal-based protein with a legume-based one may significantly reduce its related burdens, as the assessment showed that the legume-based proteins assessed had a lower environmental burden across all categories, with 99%, 88%, and 64% less freshwater ecotoxicity, water scarcity, and climate change, for an equal quantity, respectively. Another mitigation option could be to source the cereal and legume ingredients from a cereal-legume crop rotation (Costa et al., 2020).

However, in this study as shown in Table 5, per burger patty in only the scenario using (bio)physical allocation of co-products, the energy use resource carriers burden of the VB was significantly higher than the one of both the BBs (IE and BR). The fact that results were highly different when biophysical allocation was selected compared to economic allocation for the beef co-products shows that results are very sensitive to modelling choices. Nevertheless, the Product Category Rules (PCR) guidance of meat of mammals imposes an economic allocation for slaughterhouse activities, as mass of inputs is not linearly correlated to mass of outputs (EPD International AB, 2019b). Moreover, some studies put forward the limits of the biophysical approach, which is nonetheless sensitive to the economic values of products (Mackenzie et al., 2017).

5.5.2. Contribution to climate neutrality targets

Recent research highlights that the Paris Agreement target of maintaining a temperature elevation of 1.5° or 2°C since the pre-industrial age will not be possible without reducing the food system's GHG emissions (Clark et al., 2020). In June 2019, the British parliament passed an act to commit the UK to a "Net Zero" GHG emission target by 2050, meaning that all carbon emissions will have to be offset (Committee on Climate Change, 2019; Shepherd, 2020). Our proposed scenario in which all burger patties in the UK would be vegetarian instead of beef-based showed a potential climate change saving of 9.5 to 11 million tonnes CO₂e. This is an example of a small change in consumer habit that can make a significant difference towards achieving climate stabilisation.

The high energy use burdens for producing the VB may be criticised with similar arguments presented in Lynch and Pierrehumbert (2019), saying that beef burgers, which emit comparatively

more methane and less carbon dioxide than cultured meat over their life cycle, could cause less warming over the next 1000 next years compared with culture meat – if meat consumption decreases over time. In response to the limitation of the GWP₁₀₀ approach that does not make an adequate distinction between long- and short-lived climate pollutants, GWP* was created to take into account the temporal evolution of warming-equivalent emissions instead of simple GWP₁₀₀ CO₂ equivalents (Lynch et al., 2020). However, the findings of Lynch and Pierrehumbert (2019) are significantly constrained by the exclusion of COCs, thus neglecting the important implications of the comparatively very large land requirements of cattle rearing. Quite apart from having a much smaller production footprint, VB patties had a COC of just 1.5 kg CO₂e compared with 15.2 kg CO₂e for BBs. Moreover, the UK energy mix is decarbonising at a rapid pace (UK Government, 2020), and this will reduce the carbon footprint of energy used in VB patty production at a faster rate than cattle rearing is projected to decarbonise (Lanigan et al., 2018). In fact, the minor trade-offs identified for VB across a small number of impact categories were partly the result of heat requirements for production of one natural raw material, and could be mitigated by a shift towards a more sustainable heat source. Huge improvements in the efficiency of cattle rearing would be needed to match the low burdens of the vegetarian burger patty, and the potential for GHG emissions reduction in beef systems is ultimately constrained by the inherent characteristics of ruminant livestock feed conversion (Audsley and Wilkinson, 2014; Beauchemin et al., 2011; Clark et al., 2001). Environmental burdens of the beef burger could still decrease by adopting a more sustainable cattle raising system, as shown with the great contrast of environmental profiles stemming from the five cattle systems presented in this study, which are coherent with results of previous studies (FAO, 2018; Leip et al., 2010; Poore and Nemecek, 2018). Nonetheless, irrespective of how methane is dealt with in climate neutrality targets (and substantial reductions in methane emissions are required (Rogelj et al., 2018)), it is clear that a shift from beef towards vegetarian burgers could make a substantial contribution towards climate neutrality goals.

5.5.3. Limitations

A conservative approach was undertaken when selecting proxies due to lack of primary data. While energy required to make the vegetable extracts was ignored due to the lack of data, the total burdens were attributed to the ingredients used, even though the co-products of these ingredients have an economic value, for which the allocation was not modelled. Allocating burdens between co-products used in the VB would have decreased the overall burdens of the ingredients. Regarding the beef burger inventory, obtaining data for British beef cattle would have been highly relevant, however this data was not available in any of the databases. Additionally, the use of different databases was a key limitation to compare different products. However, the primary aim

of the study was to present a detailed footprint for a plant based burger and compare it with what was readily available in existing databases. The wide range of beef cattle processes available provided an overview of the relative comparisons between all burger patties. COC values used in this study were generic, as they were not distinguishing the differences between land spared in the different countries assessed. However, this was not necessary to convey the comparatively larger magnitude of the COC effect of beef versus that of plants used in the VB.

5.5.4. CONCLUSIONS

Considering Nutrient Density Unit as a functional unit highlighted the more positive nutritional profile of the vegetarian burger compared with the beef burger considered here, especially the higher content of fibre, a key nutrient for health lacking in Western diets. The results of our attributional LCA are aligned with other studies showing that vegetarian burgers have smaller environmental burdens than beef burgers. Overall, with economically-allocated burdens for beef and plant ingredients processing co-products, per burger patty, the vegetarian burger was associated with significantly lower ($p < 0.05$) environmental burdens than beef burgers made from Irish or Brazilian beef, across 9 out of 16 impact categories analysed. The relative differences between the Irish, Brazilian, French, South African, and American cattle systems highlighted the potentially large variance within a particular food product depending on origin and production processes, leaving scope for improvements. This range, however, was generally not sufficient to surpass the comparatively lower environmental impact of the VB.

Our study also demonstrated the significance of carbon opportunity costs in relation to the much smaller land requirement for the vegetarian burger versus the beef burgers. This could be critical as land is subject to more intense competition among alternative uses in the context of climate change, especially for carbon dioxide removal – and negates any doubt about the comparative climate advantage of vegetarian burgers in light of debate about how to quantify the warming effect of methane emissions from beef production. Replacing the beef burger patties eaten in the UK with vegetarian ones was shown to potentially extend the climate change savings to 9.5-11 million tonnes CO₂e, which is equivalent to 2.1 to 2.4% of the annual territorial GHG emissions in the UK in 2019. However, the ultra-processing required to produce the vegetarian patties is responsible for their NOVA group 4 classification, suggesting their potential as a “transition” food only. This “transition” product can support individuals to switch to a more nutrient-dense and environmentally sustainable diet that comprises more legumes and less red meat, in harmony with the UK’s net zero targets.

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6. DOES CIRCULAR REUSE OF CHICKPEA COOKING WATER TO PRODUCE VEGAN MAYONNAISE REDUCE ENVIRONMENTAL IMPACT COMPARED WITH EGG MAYONNAISE?

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Keywords: comparative life cycle assessment, vegan, legumes, food

6.1. ABSTRACT

We performed a comparative attributional life cycle assessment (LCA) of mayonnaise made with chickpea cooking water (aquafaba) as the emulsifying agent, and traditional mayonnaise made with egg yolk, from cradle to factory gate. The vegan mayonnaise was found not to be as environmentally sustainable as the egg-based product. The vegan mayonnaise had a significantly ($p < 0.05$) lower impact across 4 categories, cancer human health, freshwater ecotoxicity, land use, and minerals and metals use, but a significantly higher impacts across 8 categories out of 16, acidification, climate change, terrestrial eutrophication, ozone depletion, photochemical ozone formation, and resource-use-energy-carriers. The majority of categories under which vegan mayonnaise underperformed are wholly related to the electricity needed for aquafaba processing. We found that the environmental impacts arising from the comparatively high electricity use for aquafaba processing can be mitigated with a “cleaner” electricity grid, or onsite renewable electricity generation. Substituting the Mexican grid, where the aquafaba is currently processed, for the Canadian grid, where the mayonnaise is produced, reduced the carbon footprint of the vegan mayonnaise by 37%, making it similar to the egg-based product, but the share of electricity and therefore the environmental burdens associated with it are still higher than in the traditional egg-based product across most categories. Sunflower oil production was also linked to extensive environmental burdens, contributing to 37% and 74% of total climate change, and 99% and 98% of total land use burdens for the vegan and egg mayonnaises, respectively. The environmental footprint of the vegetable oil used could be lowered by adopting a different oil production method.

In contrast, a different agricultural method/geographical origin or substituting sunflower with soybean, or rice bran were associated with important trade-offs across the climate change and eutrophication categories, amongst others. Our study shows that substituting egg yolk with aquafaba could cause an increase in the environmental footprint of mayonnaise due to high processing costs, illustrating that vegan options do not always have a smaller environmental footprint.

6.2. INTRODUCTION

The previous chapter presented a comparative LCAs of a vegetarian pea and soy burger patty versus a beef patty. Results show that overall, the vegetarian burger had a lower environmental footprint across most categories, although the vegetarian patty had a higher energy use burden than the Brazilian beef patty, due to high requirements for ingredient processing. In addition, with a higher nutrient density for the vegetarian patty, its comparative environmental performance was further improved. This chapter looks at vegan mayonnaise production in Canada from water that was used to cook chickpeas (aquafaba) in Mexico. Because mayonnaise is very poor in nutrition, this chapter aims to answer a single question: is chickpea mayonnaise more sustainable than the conventional egg-based mayonnaise? It was hypothesised that the chickpea mayonnaise would have a smaller environmental impact when compared to traditional mayonnaise, due to the use of a waste product (cooking water).

Mayonnaise is a popular condiment traditionally composed of egg yolk, oil, and vinegar or lemon juice. More than 6.5 million people reported consuming mayonnaise more than once a week in the UK in 2018, alongside 3.4 million in Spain, and 4.5 million in France (Statista, 2020b, 2020c, 2020a). The global mayonnaise market was estimated at US\$ 10.8 billion in 2019 and is expected to reach US\$ 13.27 billion by 2026 (IMARC, 2019). In Spain, it was reported that 52.8 thousand tons of mayonnaise were consumed in 2019 (CMR Group, 2020). This consumption habit does not come burden-free. Although eggs have a comparatively lower environmental footprint than other animal sources, they are often associated with a higher carbon footprint than autotrophic protein sources (Poore & Nemecek, 2018).

Life Cycle Assessment (LCA) is a technique used to quantify the environmental impact of a product or service, by looking at all inputs and outputs in a defined system, from farm to factory gate, or down through to consumption, disposal, or recycling (ISO, 2006). These inputs and outputs

are linked to elementary flows, which are associated with specific impact factors corresponding to environmental issues, such as climate change, water scarcity, energy use, or nutrient pollution. To our knowledge, only one published LCA study looked at the environmental impact of traditional, egg-based mayonnaise (Hetherington et al., 2012), and none of other mayonnaise types. In the existing mayonnaise study, rapeseed oil was found to contribute to slightly more than 50% of the mayonnaise total carbon footprint, followed by packaging glass (21%) and energy consumption at the manufacturing plant (13%). The authors found that one 600 g jar of mayonnaise was responsible for 1.17 kg CO₂ equivalent.

Besides these environmental concerns, consumers are increasingly aware of the animal welfare issues related to the egg industry (Cleveland et al., 2020). The number of vegans in the world varies by sources, and a recent survey by ComRes (2018) estimated the share of vegans in selected European countries to be of around 1% of the total population. What is certain is that veganism is gaining popularity globally, with the number of individuals identifying themselves as vegan rising by 600% in the US between 2014 and 2017 (Global Data, 2017), and quadrupling in the UK between 2014 and 2018 (Statista, 2020d). Egg is commonly substituted in foods such as in pasta, meringue, and mayonnaise with legume-based ingredients. Numerous vegan egg substitutes are commercially available and made, for example, from mung bean protein isolate (JUST Egg, 2020), soy or pea protein isolates (Eunite Foods, 2020; Vegg, 2020), chickpea flour (Eggcitables, 2020), or potato starch (Bob's Red Mill, 2020). Some studies compared the carbon footprint of the egg product and its plant-based alternative (Nette et al., 2016), finding that the legume pasta emits 35.5% less GHG emissions than the egg alternative.

Numerous LCAs compared the environmental performance of different diets. Studies found a better performance of vegan diets over vegetarian diets (Chai et al., 2019; van Dooren et al., 2014), Mediterranean diets (Castañé & Antón, 2017), omnivorous diets (Harwatt et al., 2017; Scarborough et al., 2014; van Dooren et al., 2014). Some found that vegan and vegetarian diets had a very similar environmental impact (Goldstein et al., 2016), though these diets could also have a higher impact than omnivorous diets (Rosi et al., 2017). Numerous studies highlighted that omnivorous and vegetarian diets could have a similar environmental impact than vegan diets, if managed properly and with substantial reductions in animal-based foods (Chai et al., 2019; Tyszler et al., 2016). At a product level, vegan choices were often more environmentally sustainable than meat options (Heller & Keoleian, 2018; Khan et al., 2019; Saget et al., 2021).

One step further to increase food sustainability may be the replacement of animal products with waste products. Aquafaba – “bean water” in Italian – is a term used to describe the cooking water of chickpeas. Originally a waste stream, it can be used as an egg replacer thanks to its foaming

and emulsifying properties, with the perspective to increase sustainability of food production, being cruelty-free (Buhl et al., 2019; Mustafa et al., 2018), and more healthy, through the reduction of cholesterol intake (Mirzanajafi-Zanjani et al., 2019; Raikos et al., 2020). Aquafaba can be found in vegan sweet and savoury products, such as in meringues, dairy substitutes and mayonnaise (Mustafa & Reaney, 2020). To date, no study has compared the environmental impacts of producing a vegan, aquafaba-based mayonnaise with those of a traditional egg-based mayonnaise. There remains the need, therefore, to validate whether vegan, aquafaba-based mayonnaise has a lower environmental impact than traditional egg-based mayonnaise, as the former involves processing and transportation steps that have never been investigated. In this study, we hypothesised that the aquafaba-based vegan mayonnaise (VEG) has a lower environmental footprint than the conventional egg-based mayonnaise (EGG).

6.3. MATERIALS AND METHODS

6.3.1. Goal, scope, and boundary definition

This study is a comparative attributional LCA. The open source software OpenLCA v1.10 (GreenDelta, 2020) was used to calculate the environmental footprint of the two products from cradle to factory gate, using Agri-footprint v3.0 (Blonk Consultants, 2019) and Ecoinvent v3.6 (Wernet et al., 2016) international databases. Inventory data on the VEG and EGG were collected specifically for this study from a company manufacturing both products.

One functional unit (FU) was used: a 473 mL glass pot filled with mayonnaise at factory gate from the anonymous company. Two reference flows were compared: a 473 mL glass pot filled with VEG at factory gate, and a 473 mL glass pot filled with EGG at factory gate. Figure 1 illustrates the system boundaries used and manufacturing steps for the cradle to gate assessment of VEG and EGG. The environmental impact of the two products were compared across sixteen environmental impact categories recommended in the Product Environmental Footprint (PEF) Category Rules Guidance (European Commission, 2018). Results were then normalised by global person equivalents, using the factors recommended in the PEF guide (European Commission, 2018). This was done to facilitate interpretation of impact scores.

To assess the sensitivity of aquafaba environmental footprints to different regional electricity mixes used for processing, we investigated the environmental burdens stemming from a hypothetical vegan mayonnaise product (CA). The only difference between VEG and the hypothetical scenario is that the Mexican electricity grid used for aquafaba processing was substituted with the electricity grid from Canada, Ontario. This electricity mix was chosen, as it is the same as the one for egg yolk separation and mayonnaise ingredient assembly. We performed a modified Null Hypothesis Significance Test (NHST) following the same approach as in Mendoza

Beltran et al. (2018), with 1000 Monte Carlo simulation results for each VEG, EGG, and CA products. The modified NHST investigated whether the production of mayonnaise with egg yolk or aquafaba had a statistically ($p < 0.05$) significant different environmental impacts for each of the 16 impact categories. To exclude false positives, a Bonferroni correction of $\alpha_b = 0.05/48 = 0.001042$ was applied. The denominator value was determined with the sixteen environmental impact categories assessed and the three alternative pairs. The effect size was set as $\delta_0=0.2$.

With sunflower oil being a key ingredient amounting to around 70% of the total recipe ingredients by weight, three additional aspects of the LCA were investigated: A) how sensitive are the environmental burdens of sunflower oil production to a different production method? B) How sensitive are the environmental burdens of sunflower oil production to different geographical origins? C) How does sunflower oil production compare with oils from different crops? To answer A), two methods of sunflower oil production available in the Agri-footprint v3.0 database – pressing or solvent extraction – were compared using oil yield as a comparison unit (Blonk Consultants, 2019). To answer B), processes for sunflower oil production from Argentina, China, and Ukraine from the Agri-footprint v3.0 database were compared (Blonk Consultants, 2019). Finally, the environmental impacts of using of a different type of oil to produce mayonnaise was investigated to answer C). Rapeseed, soybean, rice bran oils were selected after asking the mayonnaise company which oils could be used in its mayonnaise.

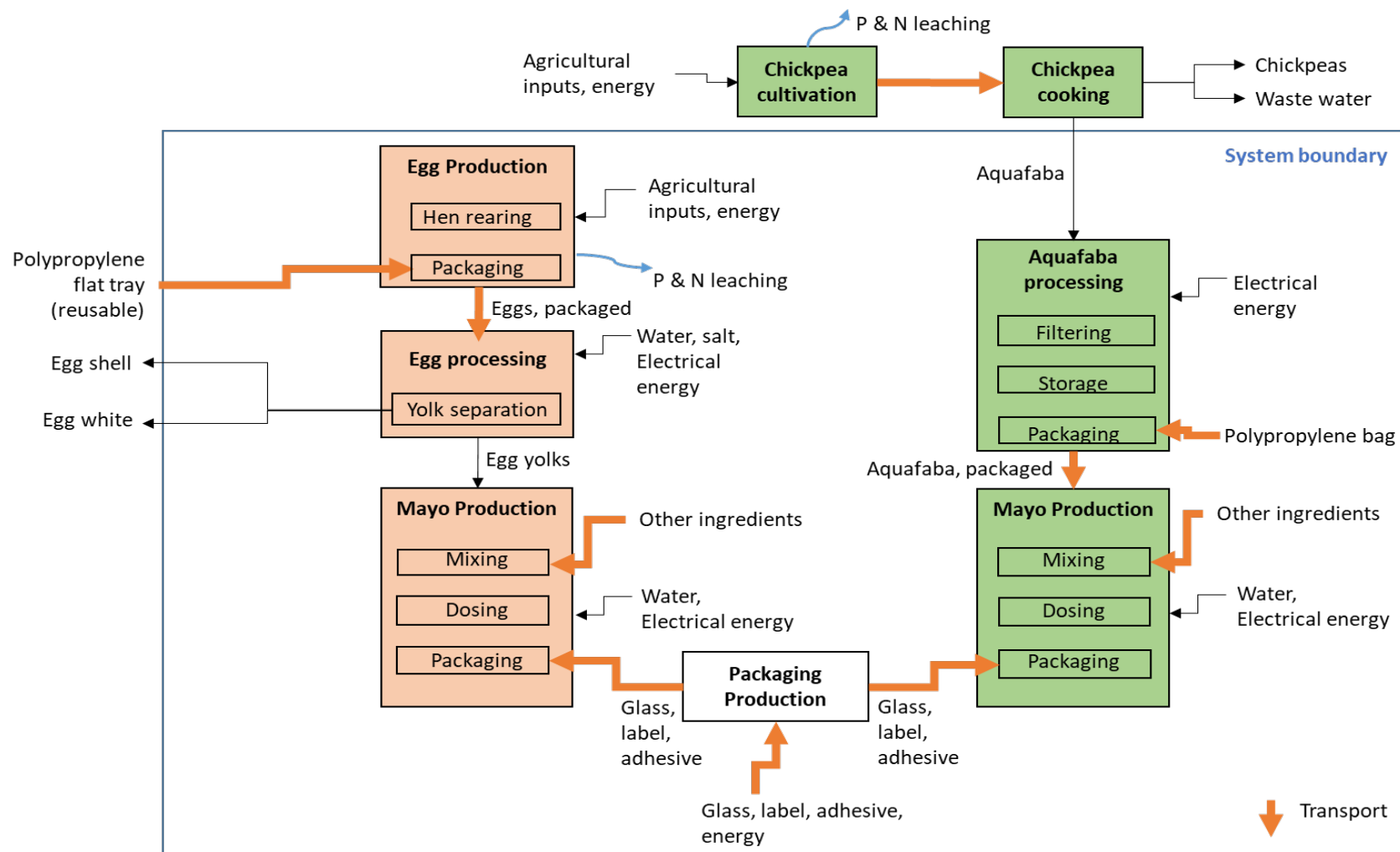


Figure 1: System boundary of vegan (in green) and egg (in red) mayonnaise production, from cradle to factory gate.

6.3.2. Mayonnaise inventory

Data on the recipe, origin of ingredients, and chickpea agricultural phase of the VEG were collected from the mayonnaise manufacturing company. Inputs and outputs to all processes involved in the life cycle of the VEG and EGG, from cradle to factory gate, were recorded but due to intellectual property information, ingredients were grouped into a black box and presented in Table 2. Relevant background processes for production of both mayonnaise types were extracted from LCA databases Ecoinvent 3.6 (Wernet et al., 2016) and Agri-footprint 3.0 (Blonk Consultants, 2019). Sunflower oil was imported from Argentina. As a conservative approach, the pressing method for sunflower oil production was selected.

All ingredients are transported to the mayonnaise factory in Canada. The energy required to mix and package the mayonnaise ingredients into the final pot was not available, and was therefore assumed to be the same as in Morgan, Stichnothe, & Azapagic (2010) for both the VEG and EGG. The obtained mayonnaise undergo the same packaging process, into a 473 mL glass pot.

6.3.2.1. *Vegan mayonnaise*

The main ingredient of differentiation in the VEG was the use of the aquafaba instead of egg yolk. The aquafaba is produced in Mexico. The International Reference Life Cycle Data System (ILCD) handbook stipulates that the economic value of the co-products is determined at the point, condition, and amount provided by that particular multifunctional process, and not at a later stage (European Commission, 2010). Therefore, because at the point of chickpea cooking, the cooking water is treated as waste, the environmental burdens of chickpea cultivation and processing were entirely attributed to the cooked chickpeas, leaving the unprocessed aquafaba as a burden-free product. The aquafaba is then further processed and packaged in polypropylene bags, consuming 3.59 kWh, and sent to a factory in Canada, where it is mixed with water and other ingredients, and the obtained mayonnaise is packaged in a 473 mL glass pot, consuming 0.4 Wh.

6.3.2.2. *Egg mayonnaise inventory*

The agricultural data for eggs was obtained directly from the mayonnaise manufacturer and used to modify the existing the Agri-footprint v3.0 egg production process (Blonk Consultants, 2019). After collection in Canada, eggs are packaged into polypropylene flat trays and transported to the mayonnaise factory in Canada, where the egg white, shell, and yolk are separated. The egg packaging was assumed to be reused 50 times, as a conservative approach. The energy and water required for this separation step was unknown, and was therefore assumed to be 1.29 kWh for 2.18 kg pasteurized egg yolk, as in Tsai, Huang, & Wilson (2021). All energy sources (electricity,

natural gas, diesel, and gasoline) from Tsai, Huang, & Wilson (2021) were modelled as electricity in this study. A study from Pelletier, Ibarburu, & Xin (2013) found that the emissions stemming from the egg processing and breaking stages were relatively minor, representing 1 to 2% of the total emissions from the whole egg supply chain, hence approximating these was not considered to be a major limitation. After being checked, the egg is washed and broken down. Egg yolk and white are then separated. The egg yolk is homogenised, refrigerated and subsequently pasteurised. The environmental burdens of the egg chain were economically attributed between egg whites and yolks, as egg shells were considered as waste. Dried egg white is worth between 4.5 to 8 times more than dried egg yolk (Belyavin, 2015). In this study, we assumed it was worth the median of this range, 6.25 times more than dried egg yolk. With a water content of 55% in egg yolk and 87.7% in egg whites (Réhault-Godbert et al., 2019), a fresh weight content of 30% of egg yolk and 60% of egg white in one egg, the allocation factors were economically calculated and amounted to 0.39 for the egg yolks and 0.61 for the egg whites. The obtained EGG mayonnaise is then packaged in the same fashion as for the VEG, as described in Section 2.2.

Table 1: Inventory for the 473 mL pot of vegan (VEG) and egg (EGG) mayonnaise modelled in this study. “Confid.” relates to the confidential recipe of the mayonnaise company.

Stage	Input / output / process	Units	VEG		EGG	
			In	Out	In	Out
Egg production	Feed and supplements	kg			0.28	
	Land	kg			0.66	
	Energy	kWh			0.01	
	Laying hen	Item			0.01	
	Water	L			0.45	
	Transport, truck >32 t	kg.km			0.28	
	Laying hen	kg				0.008
	Egg	kg				Confid.
Aquafaba processing	Energy	kWh	3.59			
	Aquafaba	kg		Confid.		
Aquafaba/egg packaging and transport	Polypropylene	kg	0.11		0.027	
	Forming of packaging	kg	0.11		0.027	
	Packaging transport, truck >32 t	kg.km	25.0		0.12	
	Packaging transport, train	kg.km	30.4		0.15	
	Packaging transport, ship	kg.km	39.1		0.19	
	Transport to mayo factory, truck >32 t	kg.km	805.5		653.5	
Egg processing	Water	L			0.42	
	Processing energy	kWh			0.022	
	Salt	kg			0.004	
	Egg yolk	kg				Confid.
	Egg shell	kg				Confid.
	Egg white	kg				Confid.
Other	Sunflower oil from crushing (pressing)-	kg	Confid.		Confid.	
	Oil transport, truck >32 t	kg.km	893		995	
	Oil transport, ship	kg.km	4360		4860	
	Water	L			0.037	
	Other ingredients	kg	0.04		0.38	
	Ingredients transport truck >32 t	kg.km	39		31	
	Ingredients transport ship	kg.km	476		232	
Packaging	Glass	kg	0.203		0.203	
	Graphic paper, 100% recycled	kg	0.0009		0.0009	
	Dichloromethane	kg	0.0001		0.0001	

	Steel cap	kg	0.011		0.011
	Packaging trans, lorry, >32t	kg*km	484.3		484.3
Mayonnaise production	Electricity for assembling Mayonnaise pot, at factory	kWh item	0.0004	1	0.0004 1

6.-1. RESULTS

6.-1.1. General environmental results

The environmental burdens of the two mayonnaise types are recorded in Table 2. The two products had large environmental footprint differences across various categories, with VEG having an 80% higher carbon footprint and a 1.3 times higher resource-use-energy-carriers burdens than those of EGG. The EGG mayonnaise had a 10% and 12% higher terrestrial eutrophication and land use burdens than VEG. The Monte Carlo analyses and the modified NHST showed that the VEG had a significantly lower environmental impact from EGG across the cancer human health, ecotoxicity freshwater, land use, and resource use minerals and metals categories, while EGG had a significantly lower environmental impact across 8 categories out of 16, including acidification, climate change, and resource-use-energy-carriers (Table 2).

To understand which categories may represent relatively greater environmental loadings, we performed a normalisation of the results in person equivalents (Figure 2). Apart from the toxicity-related categories, for which the normalisation factors are highly uncertain (European Commission, 2018), between the VEG mayonnaise categories, the highest burdens were resource-use-energy-carriers, land use, climate change, acidification, and the three eutrophication types, with a burden of at least 3E-4 person equivalents per pot of mayonnaise. Between the EEG mayonnaise categories, these were the same excepted for climate change and freshwater eutrophication, for which the burdens were smaller than 3E-4 person equivalents. VEG had a significantly higher ($p < 0.05$) environmental impact across 4 categories with the highest top 7 normalised scores – acidification, climate change, resource-use-energy-carriers, freshwater eutrophication – while EGG had a significantly higher ($p < 0.05$) environmental impact across 1 of them – land use.

Table 2: Summary of environmental burdens for the vegan (VEG) and egg-based (EGG) mayonnaises, expressed per 473 mL pot of mayonnaise. Shaded cells in grey indicate the impact categories in which the product has a significantly larger environmental impact than the other mayonnaise.

Name	Unit	VEG	EGG
Acidification terrestrial and freshwater	mol H ⁺ eq	0.022	0.019
Cancer human health effects	CTUh	6.3E-08	5.5E-08
Climate change	kg CO ₂ eq	3.6	2.0
Ecotoxicity freshwater	CTUe	25.7	27.6
Eutrophication freshwater	kg P eq	0.00089	0.00038
Eutrophication marine	kg N eq	0.011	0.011
Eutrophication terrestrial	mol N eq	0.057	0.063
Ionising radiation, human health	kBq U- ²³⁵ eq	0.243	0.096
Land use	Point	799	897
Non-cancer human health effects	CTUh	1.1E-06	1.1E-06
Ozone depletion	kg CFC-11 eq	2.7E-07	1.2E-07
Photochemical ozone formation, human health	kg NMVOC eq	0.012	0.008
Resource use, energy carriers	MJ	46.9	20.4
Resource use, mineral and metals	kg Sb eq	3.3E-07	3.7E-07
Respiratory inorganics	disease inc.	1.7E-07	1.6E-07
Water scarcity	m ³ depriv.	1.13	1.18

A process contribution graph was recorded in Figure 3, so as to identify hotspots across the life cycle. Sunflower oil production was responsible for most of the impact across the majority of categories. Sunflower oil production was responsible for 59% of the total cancer human health burdens of VEG and 76% for EGG, mainly due to the emission of chromium to water from sunflower seed production. It was also responsible for 82% and 88% of the total non-cancer human health effects of VEG and EGG, respectively, due to zinc emission to soil. In the ecotoxicity freshwater category, sunflower oil production was responsible for 90% of the total burdens VEG and 93% of EGG due to the emission of the insecticide alpha-cypermethrin to water. Sunflower oil production was also responsible for 99% and 98% of land use burdens for VEG and EGG, respectively. For the EGG mayonnaise, resource-use-energy-carriers burdens were due to sunflower seed production with 67% of the total burdens, and an additional 21% from the glass pot for the mayonnaise. For the VEG mayonnaise, 26% of total burdens in this category were due to sunflower seed production, and 14% from the aquafaba polypropylene packaging. In the climate change category, sunflower oil was responsible for 37% of total VEG burdens.

Besides the high environmental share of sunflower oil, the electricity usage for aquafaba processing was responsible for a major portion of the environmental burdens, including 44% of the total carbon footprint, 47% of total resource-use-energy-carriers, and 90% of the total freshwater eutrophication. Sources of the Mexican electricity mix are a combination of natural gas, oil, and

lignite, amongst others, which are an important source of carbon dioxide emissions to the air. Egg production for the EGG mayonnaise was responsible for 23% of its total acidification burden and 29% of its total terrestrial eutrophication, due to the release of ammonia to the air. Egg production was also the cause of 35% of its total water scarcity footprint due to feed irrigation, and 21% of the total respiratory inorganics burden due to ammonia and particulate matters that are less than 10 μm release to the air.

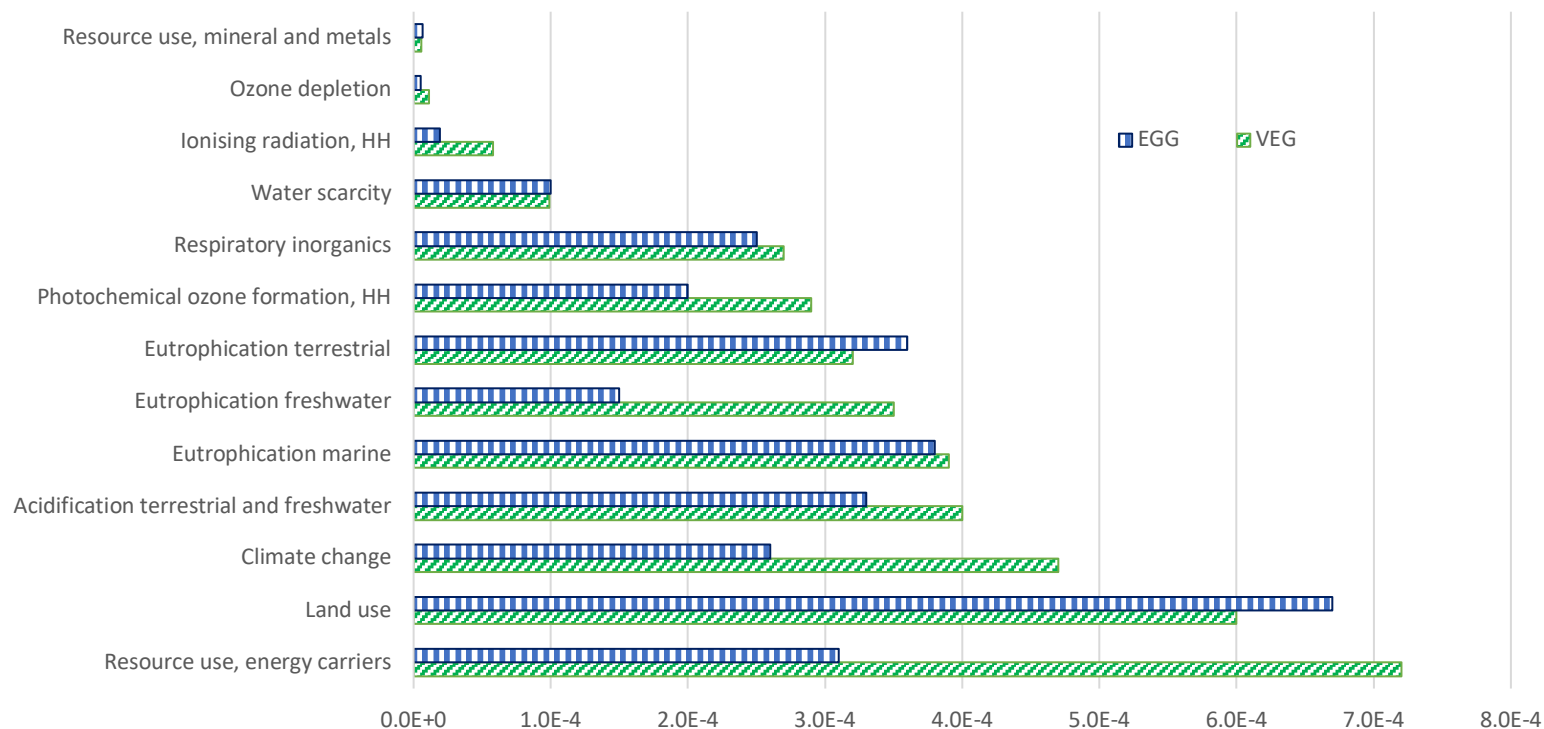


Figure 2: Normalised environmental results of a pot of vegan (VEG) and egg (EGG) mayonnaise across all impact categories assessed, excepted for the toxicity-related ones, due to highly uncertain normalisation factors, following Product Environmental Footprint guidelines (European Commission, 2018).

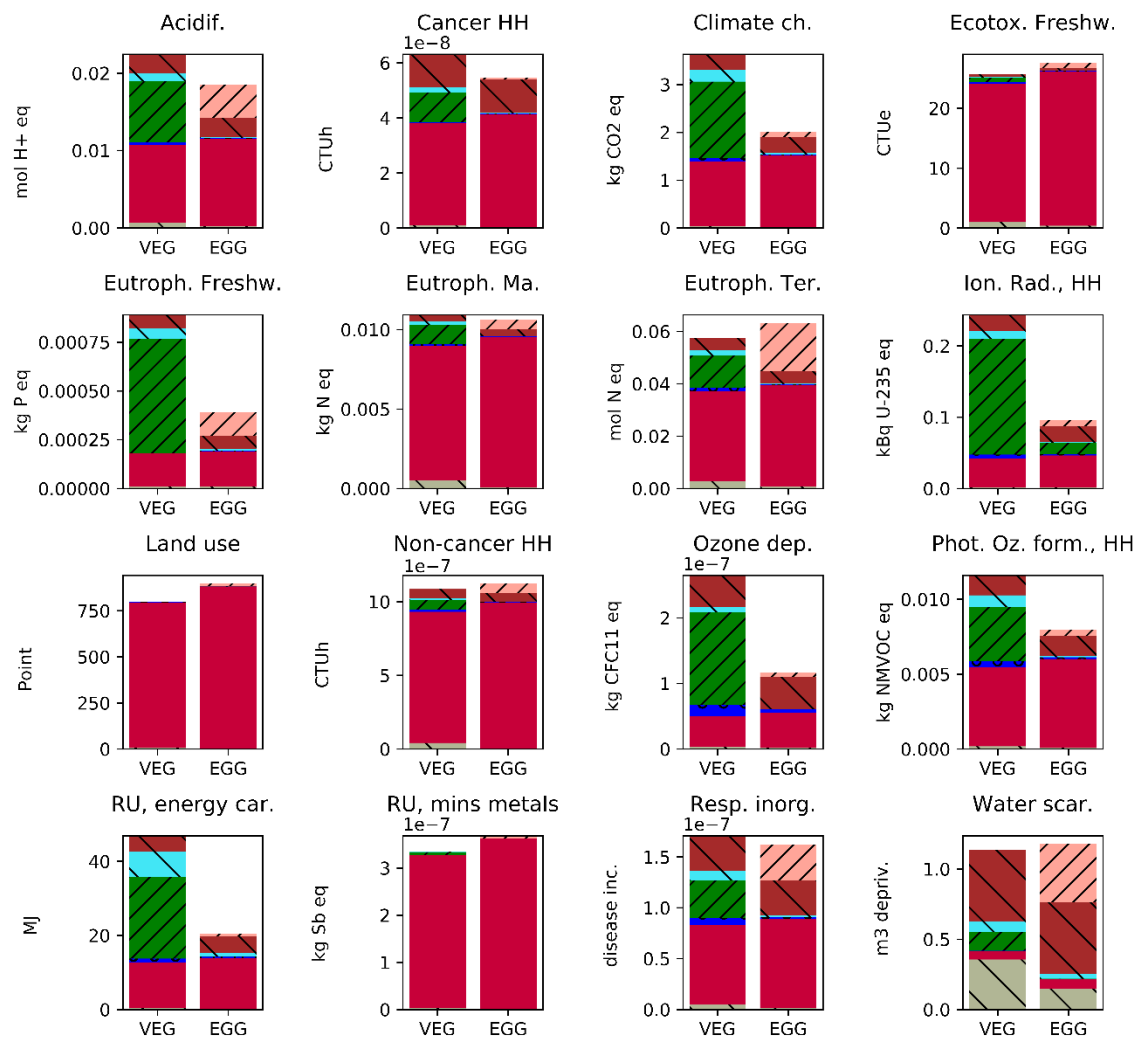


Figure 3: Process contributions of vegan (VEG) and egg (EGG) mayonnaise across the sixteen environmental impact categories recommended by the Product Environmental Footprint guidelines.

6.-1.2. Sensitivity analyses

6.-1.2.1. Electricity for aquafaba processing

The electricity use for aquafaba processing has a major impact on the comparative environmental performance of the vegan mayonnaise. If this step were to be removed, the VEG mayonnaise would have lower environmental burdens than the EGG version across 13 categories out of 16, with remaining higher resource-use-energy-carriers, ozone depletion, and ionising radiation burdens. The process contribution results that comprise the additional scenario, CA are shown in Figure 4. The environmental performance of CA was considerably smaller than that of VEG

across most categories, with a 37% lower climate change footprint, a 32% lower acidification burden, and a 62% lower freshwater eutrophication burden. The carbon footprint of CA is 13% higher than that of EGG, while the carbon footprint of VEG is 80% higher than that of EGG. However, CA was associated with a 15% higher resource-use-energy-carriers burden, and a 4 times higher water scarcity burden than VEG.

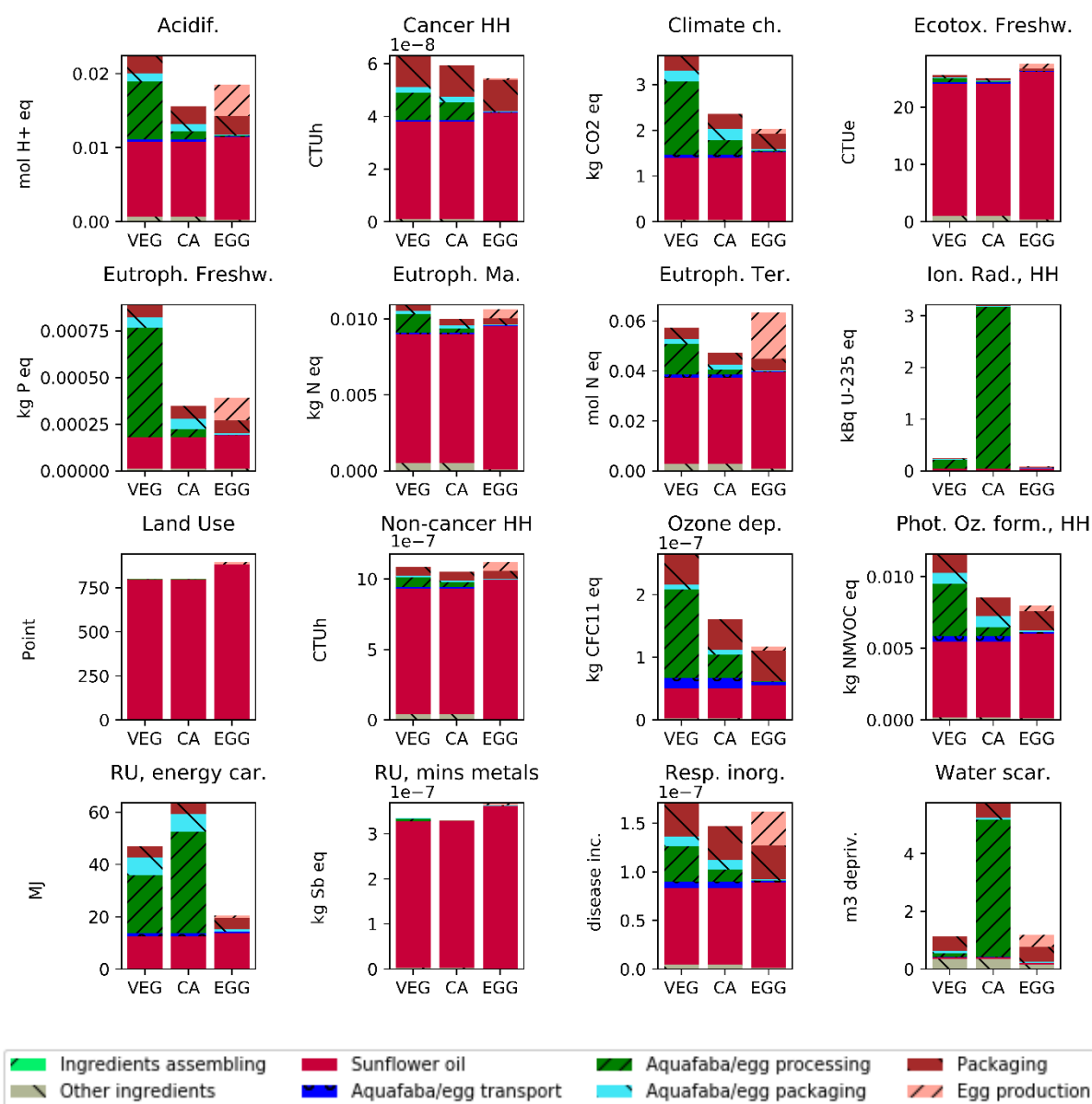


Figure 4: Process contributions of egg (EGG), vegan mayonnaise (VEG), and vegan mayonnaise with the Canadian electricity mix for the aquafaba processing (CA) across the sixteen environmental impact categories recommended by the PEF guidelines.

6.-1.2.2. Type of oil used

A. Environmental differences of alternative production method of sunflower oil

The comparison of Argentinian sunflower oil from pressing and solvent extraction was recorded in Figure 5. Apart from photochemical ozone formation, the solvent extraction method appeared to have an impact between 7% and 16% lower than the pressing method. The yield of sunflower oil from the solvent method appeared to be 30% higher than from the pressing method, hence the comparatively lower overall environmental footprint. However, when applied to the whole vegan mayonnaise production system, this different processing method did not change significantly the overall environmental impact of the product (data not shown).

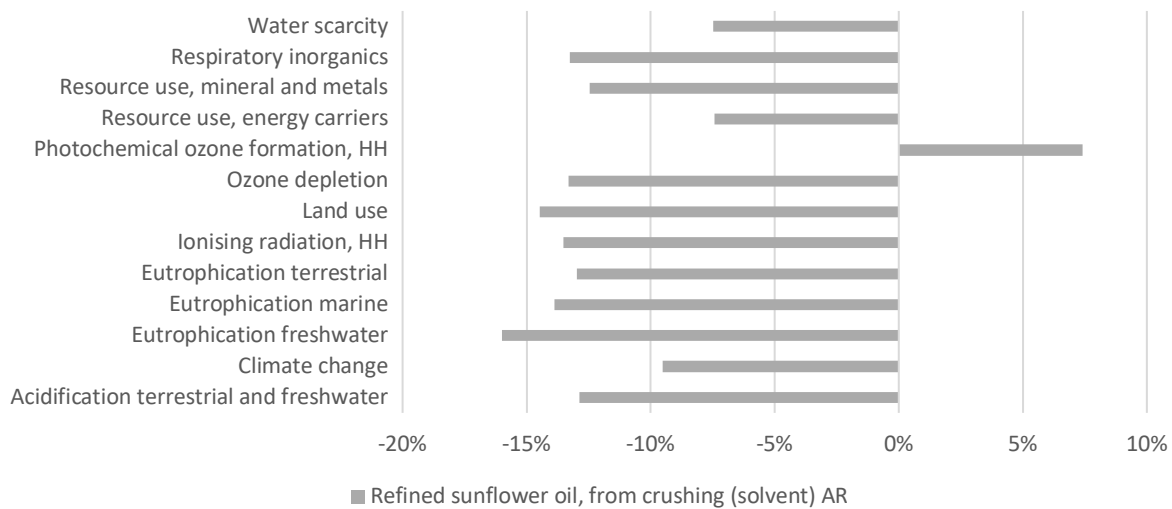


Figure 5: Comparison of Argentinian sunflower oil production from pressing (reference, on Y axis at 0%) and from solvent extraction across the non-toxicity categories recommended by the PEF guidelines

B. Environmental differences of sunflower oil production from varying geographical origins

Figure 6 presents the environmental differences between refined sunflower oil made by pressing from the different countries from cradle to farm gate. The systems compared differed in the sunflower agronomy stage as well as the electricity grid for oil processing, which was specific to the country in which the sunflower oil was produced. The sunflower oil from Ukraine was associated with higher environmental burdens across all categories apart from land use, which was 1% lower. Burdens from other categories excluding the toxicity ones were between 5% (photochemical ozone formation) and 266% (freshwater eutrophication) higher than those of sunflower oil from Argentina. Sunflower oil from China appeared to have a lower environmental impact across 6 categories out of 16, including a 28% lower land use burden than the Argentinian sunflower oil. Yields were higher for the Chinese sunflower seed production, with 2519 kg seeds

per hectare against 1797 kg Argentinian seeds per hectare, and 1820 kg Ukrainian seeds per hectare. Although both the Ukrainian and Chinese systems had a higher yield per hectare than the Argentinian one, they had a higher environmental impact due to more fertiliser inputs per final output (Table 3).

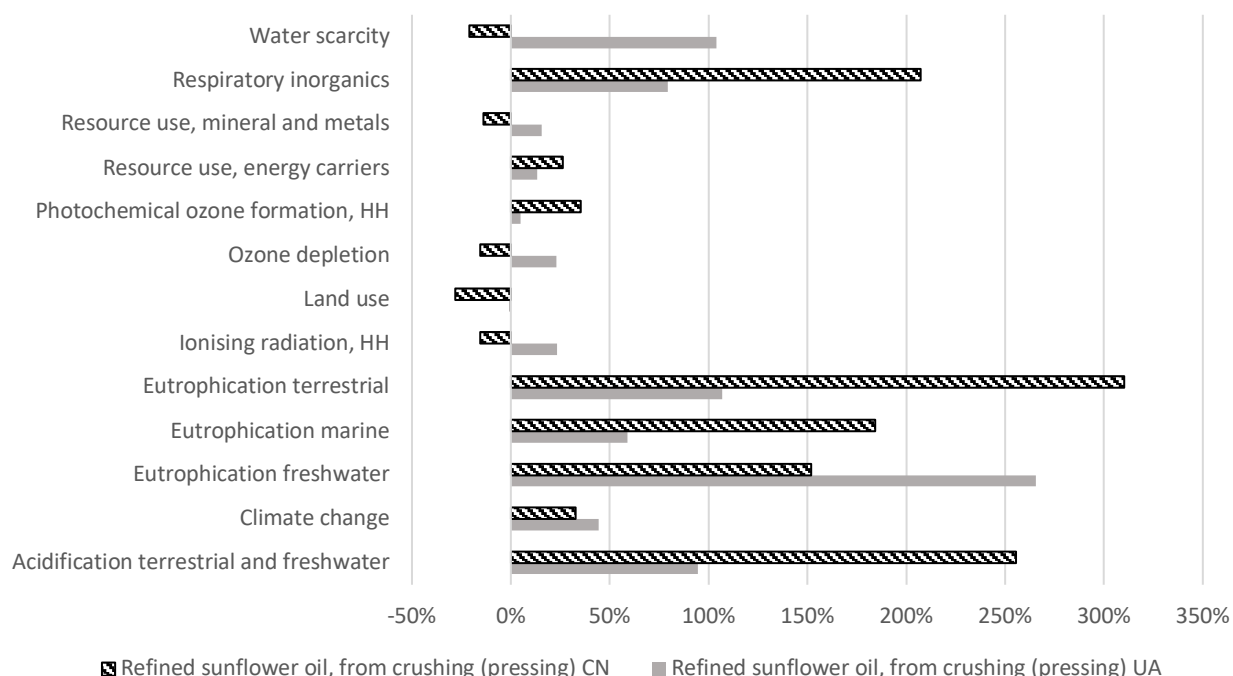


Figure 6: Comparison of Argentinian sunflower oil production from pressing (reference, on Y axis at 0%) and from Ukraine (UA) and China (CN) across the non-toxicity categories recommended by the PEF guidelines

Table 3: Comparison of fertiliser inputs per kg of sunflower seed obtained in the Ukrainian, Chinese, and Argentinian agronomic processes from Agri-footprint v3

	N	Manure	P	K
Ukraine	0.0158	0.2693	0.0203	0.0066
China	0.0423	0.6135	0.0106	0.0052
Argentina	0.0056	0.1493	0.0039	0

C. Environmental differences of other vegetable oils

The environmental impact results of rapeseed (from pressing in Germany), soybean (from pressing in Argentina and Brazil), and rice bran (from China) oil types per kg of product are presented in Figure 7. All other options had a lower land use and resource use minerals and metals burdens than the sunflower oil. The only option that was associated with a lower carbon footprint than the current sunflower oil was the rapeseed oil from Germany, with a 20% lower climate change

burden. However, the German rapeseed oil had eutrophication burdens that were between 60% and 180% higher, and a 125% higher acidification burden than those of the Argentinian sunflower oil.

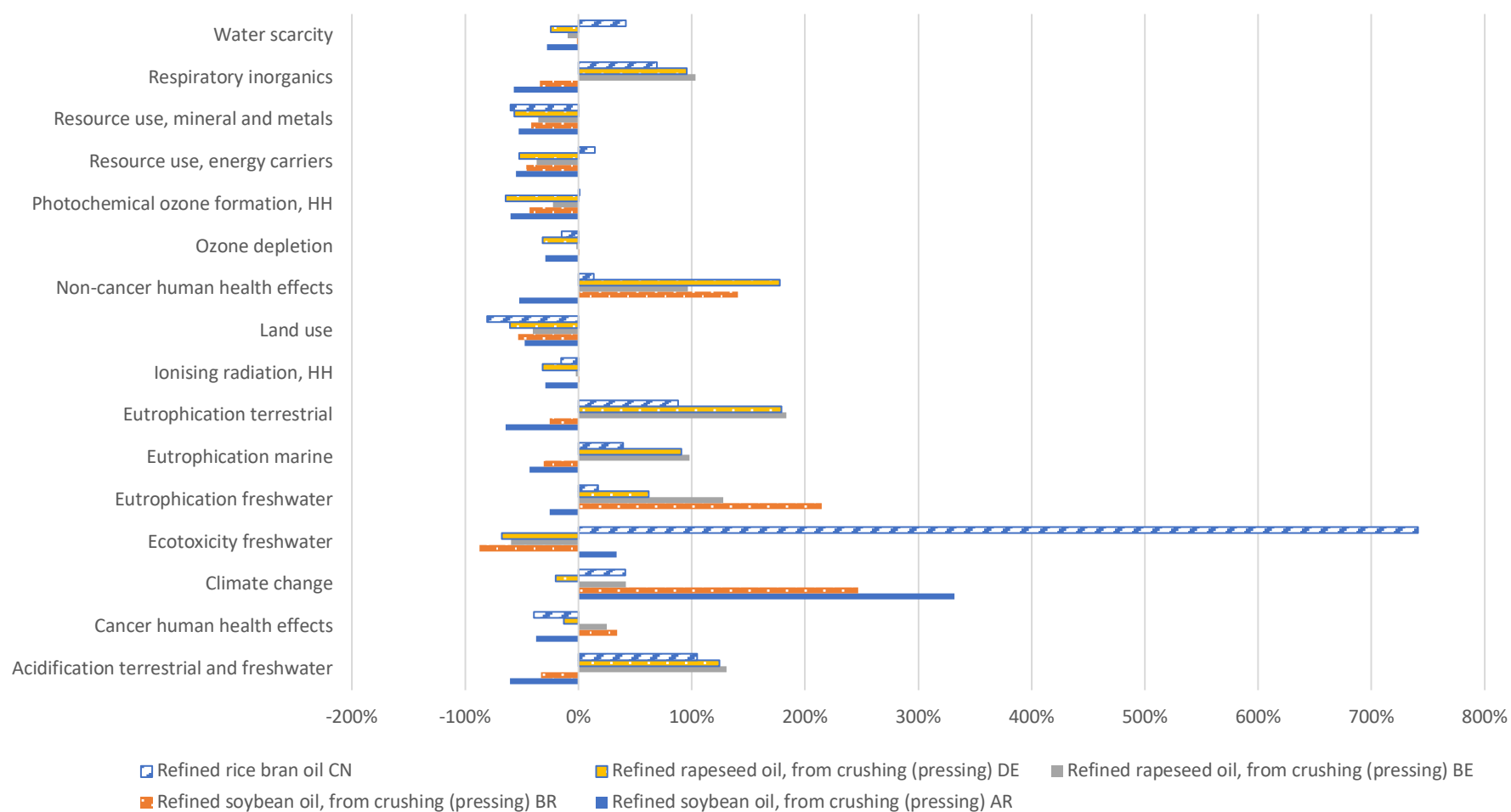


Figure 7: Comparison of Argentinian sunflower oil production from pressing (reference, on Y axis at 0%) and from soybean oil (Brazil –BR and Argentina –AR), rice bran oil (China –CN), and rapeseed oil (Belgium –BE and Germany –DE) across the non-toxicity categories recommended by the PEF guidelines

6.0. DISCUSSION

The comparative LCA showed that using aquafaba instead of egg yolk can significantly affect the overall environmental performance of mayonnaise. Our hypothesis that the vegan mayonnaise was associated with lower overall environmental burdens than the egg alternative was shown to be false in our case study, due to the high electricity use for aquafaba processing. We showed that if the electricity of the aquafaba processing step was removed, the environmental impacts stemming from the vegan mayonnaise would be smaller across all but 3 categories from the studied egg-based mayonnaise. Other studies have pointed out that additional processing steps and a comparatively high energy use can critically worsen the environmental advantage of an innovative product (Alexander et al., 2017; van der Weele et al., 2019). Because aquafaba is a recent ingredient, more efficient and less resource-intensive processing methods may be developed in the future, resulting in a significantly lower overall environmental burden for aquafaba-based mayonnaise. According to Smetana et al. (2018), modelling of waste streams at low Technology Readiness Levels (TRLs) is not efficiently performed and the end result may not be representative of the future potential environmental performance of the products. Moreover, the environmental performance of other aquafaba-based products should be investigated, as processing techniques may vary from this case study.

With electricity mixes differing greatly between countries, the contribution of the current aquafaba processing electricity use to the overall mayonnaise environmental impact can vary greatly, depending on where this step takes place. The sensitivity analysis performed with the CA scenario reduced the environmental burdens of the vegan mayonnaise across most categories, including a 37% reduction of the total carbon footprint, for example. In total, CA had significantly ($p < 0.05$) lower burdens than EGG across 8 categories out of 16, and 12 out of 16 when compared to the VEG mayonnaise (Table A.1). The EGG mayonnaise still had significantly ($p < 0.05$) lower environmental burdens across the same categories when compared with VEG or CA, apart for the freshwater eutrophication and respiratory inorganics categories, across which CA mayonnaise had a significantly lower burden. The CA mayonnaise also had a significantly lower burden than VEG and EGG across the terrestrial eutrophication categories. However, EGG had a significantly lower water scarcity burden than that of the CA mayonnaise. It is important to note that the Mexican electricity grid mix process used in this case study is from Ecoinvent v3.6 (Wernet et al., 2016), which is based on 2014 electricity data. As the Mexican government is working to increase its share of clean energy, aiming for it to reach 25% of total electricity production by 2024, 43% by 2030, and 50% by 2050 (Mexican Government, 2020), the current and future environmental impact of VEG is likely to be lower than the one presented in this case study, even if the amount of electricity use remains the same.

The contribution analysis showed that sunflower oil was responsible for a major part of the environmental impact in both the VEG and EGG mayonnaises. This finding is in line with the existing published LCA of traditional egg mayonnaise, which found that the oil (rapeseed in their case) was responsible for more than half of the carbon footprint of the product (Hetherington et al., 2012). This is not very surprising, as the oil is the main ingredient of the mayonnaise quantity-wise. Finding a more environmentally-sustainable oil or oil substitute would therefore have the greatest impact common to both mayonnaises. Our assessment showed that the agronomic approach in various countries had a large influence on the environmental impact of sunflower seed production, with total carbon footprints of oil varying between 33% and 45% more, land use up to 28% less, and terrestrial eutrophication up to 310% more than the current sunflower oil from Argentina, amongst others. This high variance between agricultural approaches was also revealed in other studies (Poore & Nemecek, 2018). In addition to environmental issues related to oil production, alternatives that are more healthy are being researched extensively (Mirzanajafi-Zanjani et al., 2019), due to high intakes of dietary fats being linked with poor gut microbiota and consequently decreased metabolic health, causing high obesity rates with increased incidence of type 2 diabetes, cardiovascular diseases, and hypertension (Cândido et al., 2018; Wolters et al., 2019). With more than one in two adults and nearly one in six children being overweight or obese in the Organisation for Economic Co-operation and Development (OECD) countries (OECD, 2017), it is therefore crucial to find healthier alternatives without compromising environmental sustainability, as was shown with the aquafaba substitution with egg yolk in this study.

As displayed with the sixteen impact categories considered, it is key to evaluate the impact beyond a simple carbon footprint in order to identify trade-offs. Our additional analyses showed that the environmental burdens of oil in the mayonnaise across most categories may be reduced by up to 16% by adopting a different oil processing method (solvent), and that substituting sunflower with soybean or rice bran were associated with important trade-offs, including across the climate change category, across which the total carbon footprint of oil was 330% higher. A consequential LCA that aimed to capture the effects of substituting the average mix of oils with different oils was carried out by Schmidt (2015). The author concluded that an increase in rapeseed and sunflower oils would have the lowest carbon footprint, palm and soybean oils had the lowest land use, and sunflower oil had the smallest water use. Due to the key role of restoration of native ecosystems on anthropogenic land use to store atmospheric carbon and maintain temperature levels below 1.5 °C (Hayek et al., 2020; Searchinger et al., 2018), the fact that land use burdens of all other oils were between 40% and 80% lower than the land use burden of the current Argentinian sunflower oil selected as the study baseline is important.

The environmental impact of the aquafaba packaging was higher than that of the egg packaging, as the eggs were packaged and transported as a whole, with a subsequent separation of yolk, white and shell. This implies that the egg packaging burdens were distributed between the various components, and due to the egg yolks having a lower economic value than egg whites, the egg packaging burdens were therefore mostly allocated to the egg whites, which are not part of the EGG mayonnaise boundaries. If no allocation had been performed, or if the egg yolks had been separated prior to transport to the mayonnaise production factory, the respective environmental impact may have varied significantly.

The study is limited by the fact that secondary processes were used for sunflower seed production and processing into oil. In order to accurately calculate the impact of the use of sunflower oil, primary data on sunflower seed cultivation and oil processing need to be collected. However, this study provides a good overview of the potential environmental consequences of modifying the supply chain of the oil ingredient, and the potentially wide ranges of environmental impact that can be observed for the same end product depending on ingredient sourcing and processing location. This is the first study that compares an aquafaba-based mayonnaise with an egg-based one. Therefore, results are strictly specific to this case and only provide an overview of the potential impacts and hotspots for the specific products described. Additional environmental assessments need to be performed with data from other mayonnaise manufacturers, as we showed that impacts were very sensitive to the location of aquafaba processing and the agronomic data of sunflower oil, amongst other points.

6.1. CONCLUSIONS

Our comparative LCA shows that substituting egg yolk with aquafaba in mayonnaise can result in a significantly higher environmental footprint across major categories including climate change, due to high electricity use for aquafaba processing. Therefore, we highlighted the absolute necessity to perform environmental assessments of novel products, as, even with a burden-free waste vegan product substituting for an animal product, associated environmental burdens of the former can be comparatively higher. We showed that increasing the share of cleaner electricity generation on national grids is likely to reduce the overall impact of the vegan mayonnaise, although important trade-offs can occur across some categories. The environmental hotspot common to both mayonnaises is sunflower oil production, and we showed that there is potential to reduce this impact by adopting a different oil production method. In contrast, changing agricultural practises or source regions, and substituting sunflower with soybean or rice bran, were

associated with important trade-offs. Thus, for mayonnaise, vegan substitutes and circular reuse of a waste (aquafaba) do not necessarily derive lower environmental impact.

7. DISCUSSION

The first two chapters introduced key challenges of the food sector in Europe, the role of legumes in this context, and the LCA approach and methodology used throughout this thesis to evaluate the potential of legumes in innovative food products to provide more nutrition at a lower environmental cost than their traditional equivalents. Chapters 3 to 6 present the comparative LCAs as case studies of this thesis: chickpea pasta versus durum wheat pasta (Chapter 3), pea protein balls versus beef meatballs (Chapter 4), pea and soy burger patty versus beef patty (Chapter 5), and aquafaba mayonnaise versus egg mayonnaise (Chapter 6). For each chapter, the products were compared from a mass standpoint (functional unit). For Chapters 3, 4, and 5, nutrient density was integrated in the comparison as the Nutrient Density Unit (NDU) to compare foods based on how much nutrition they provide. This chapter brings all the case studies together to provide an overview of the potential of legumes to offer a more sustainable pathway to achieve high nutrient density in Europe from an environmental perspective, meeting the aim introduced at the start of the thesis:

- Benchmarking of the environmental sustainability performance of nutrient delivery of novel legume-based products against their conventional counterparts

I show that the extra processing legumes undergo and the addition of other ingredients to the recipe required to make legumes palatable to unused consumers do not add excessive additional environmental burdens that would make the products less environmentally sustainable than the traditional ones across most categories, excepted for the vegan mayonnaise, due to a high electricity usage increasing significantly the burdens across the acidification, freshwater eutrophication, ionising radiation, and climate change categories, amongst others. I also show that when including nutrient density as a functional unit, the environmental advantage of the legume-based products further extends, due to a comparatively higher nutrient density. Therefore, the environmental sustainability performance of nutrient delivery of novel legume-based products is superior across most impact categories assessed when compared to their conventional counterparts.

Key trends across products and impact categories are identified in this chapter by looking first at the results of all the LCA case studies performed in this thesis, and second at those using a nutrient density functional unit. The discussion then extends around carbon sequestration potential resulting from land sparing and restoration. Finally, the limitations of legume LCAs and agricultural LCAs in general are briefly deliberated.

7.1. STANDARD MASS FUNCTIONAL UNIT SUMMARY

Legume-based products that mimic a traditional popular food option hold potential in providing healthier and more environmentally-conscious alternatives to some foods present in lose-lose diets — diets that are damaging both human and planetary health (Garnett, 2016). Table 7.1 presents the statistically significant ($p < 0.05$) differences between the environmental burdens associated with the various legume-based and non-legume-based products compared in this thesis. To include the pasta products in the comparison table, a Monte Carlo analysis and a modified NHST were performed with data from Chapter 3, using the same methodology as for the statistical significance tests performed in Chapters 4 to 7.

Through the case studies results presented in Table 7.1, we can conclude that the extra processing legumes undergo and the addition of other ingredients to the recipe required to make legumes palatable to unused consumers do not add excessive additional environmental burdens that would make the products less environmentally sustainable than the traditional ones across most categories, excepted for the vegan mayonnaise, for which the additional electricity used to process the aquafaba attributed a comparatively higher environmental footprint across several impact categories, including climate change. Besides the mayonnaise product, the additional legume processing steps had in some cases few significant drawbacks, such as a higher resource and photochemical ozone formation burdens (Chapter 5) when compared to the traditional alternative. These drawbacks were however minor when looking at the magnitude of the normalised scores across impact categories for which the legume alternative had a comparatively lower environmental footprint.

For all the case studies presented in this thesis, a same mass of legume-based product had significantly lower acidification, climate change, marine and terrestrial eutrophication, and respiratory inorganics burdens than the non-legume product, excepted for the mayonnaise comparison, for which the corresponding environmental burdens of the legume-based product were either insignificant or higher. For these other non-mayonnaise products, their associated lower burdens across the mentioned categories are in line with existing literature (Lienhardt et al., 2019; Poore and Nemecek, 2018; Williams et al., 2020a, 2020b). Therefore, these case studies support the potential of innovative legume-based products to facilitate the transition towards a more environmentally and nutritionally sustainable diet in Europe, so as to remain within planetary boundaries (Steffen et al., 2015).

The legume-based foods that were compared to animal-based products had a significantly ($p < 0.05$) lower land use burden. This highlights the land use efficiency of autotrophs versus heterotrophs and the yield gap of legumes when compared to cereals, especially in Europe, as recorded in other studies (Alandia et al., 2020; Nemecek et al., 2008; Reckling et al., 2018; Zander et al., 2016). In Chapter 3, the comparison did not include an animal-based product, and it was clear that yield was a limiting factor for legumes to be undoubtedly more environmentally sustainable than cereal-based products. It is worth noting however that the land use burden differences between the pasta products and those comparisons with a beef component were at a different scale, with the beef products having a land use burden difference that is between 1 and 2 orders of magnitude higher. This suggests that chickpea pasta could be integrated in food production and consumption systems and provide the numerous agro-ecological and health benefits without greatly impeding on land use, if this is combined with a substitution of the legume-based mock meats with the beef products presented in this thesis. Furthermore, the land spared from a decreased demand for and production of beef could store large quantities of carbon, if restored to its native state. This is further discussed in Section 7.3.

Most cancer and non-cancer human health, ecotoxicity freshwater, ozone depletion, and water scarcity burdens differences between the legume and non-legume products were insignificant. Existing literature shows that water demand for pulses is 1.5 times smaller than water demand for milk, eggs, and chicken, and 6 times smaller than that of beef (Mekonnen and Hoekstra, 2010). This highlights the fact that there are trade-offs associated with incorporating legumes into ultra-processed foods, and that these innovative products are potential transition foods to help consumers shift their diet to a more sustainable one. However, the end sustainable diet goal would be to consume legumes that have undergone little processing.

Table 7.1: Difference between the environmental score of the legume product and the traditional product with economic allocation of co-products.

Significance of the results ($p < 0.05$) that is common to all scenarios assessed in the respective case studies (such as allocation methods) is shown in colour, with green meaning that the legume product has a significantly lower burden than the traditional product, red significantly higher, and white non-significant.

	Pasta				Protein balls		Burger patties					Mayonnaise	
Legume Product	Chickpea (BG)	Chickpea (BG)	Chickpea (SP)	Chickpea (SP)	PPB	PPB	VB	VB	VB	VB	VB	VEG	CA
Traditional Product	Wheat (0% straw)	Wheat (80% straw)	Wheat (0% straw)	Wheat (80% straw)	MB (IE)	MB (BR)	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)	EGG	EGG
Is the mean impact of the legume product at least 0.2 standard deviation units significantly lower or higher than that of the traditional product?													
					lower	higher	NS						
Acidification ter. & freshwater	-37%	-33%	-68%	-66%	-95%	-81%	-91%	-50%	-88%	-83%	-80%	21%	-17%
Cancer human health	18%	20%	-2%	-1%	-78%	-99%	-80%	-99%	-71%	-89%	-82%	16%	6%
Climate change	-21%	-17%	-45%	-42%	-85%	-89%	-67%	-77%	-68%	-72%	-71%	80%	13%
Ecotoxicity freshwater	35%	37%	17%	19%	-81%	-100%	185%	-95%	419%	-49%	382%	-7%	-9%
Eutrophication freshwater	-37%	-37%	-50%	-50%	-75%	-63%	-63%	3%	-91%	-55%	50%	134%	-11%
Eutrophication marine	-22%	-6%	-86%	-83%	-96%	-93%	-92%	-87%	0%	-87%	-60%	3%	-7%
Eutrophication terrestrial	-38%	-34%	-76%	-74%	-97%	-87%	-96%	-80%	-94%	-92%	-92%	-9%	-26%
Ionising radiation, HH	-43%	-43%	-46%	-46%	-43%	-43%	-29%	-31%	500%	-4%	100%	153%	2429%
Land use	177%	200%	149%	169%	-89%	-97%	-87%	-96%	-98%	-86%	-89%	-11%	-11%
Non-cancer human health	115%	120%	84%	87%	-71%	-100%	-91%	-100%	-78%	-89%	-91%	-3%	-7%
Ozone depletion	-10%	-9%	-25%	-25%	-29%	-41%	36%	-9%	-9%	-17%	-24%	128%	30%
Photochem. ozone form.	8%	11%	-13%	-11%	-82%	-75%	-9%	43%	150%	-23%	67%	46%	6%
Resource use, energy carriers	-25%	-25%	-36%	-36%	-57%	-44%	-29%	8%	0%	-67%	-10%	129%	163%
Resource use mins. & metals	56%	68%	-55%	-51%	-99%	-23%	-92%	741%	-100%	-100%	-99%	-9%	-11%
Respiratory inorganics	-34%	-31%	-63%	-61%	-96%	-82%	-91%	-58%	-88%	-86%	-80%	6%	-11%
Water scarcity	-30%	-30%	-40%	-40%	-74%	-37%	-71%	-2%	-26%	-93%	NA	-4%	389%

7.2. NUTRIENT DENSITY FUNCTIONAL UNIT SUMMARY

Besides the environmental benefits and trade-offs of legume-based innovative foods at a mass level, this thesis integrated an additional key aspect of sustainability in the LCA equation – nutrition. With Europe facing a diet crisis characterised by an exceeding intake of energy-rich but nutrient-poor foods, this thesis highlighted the potential of legume-based products to provide better nutrition at a lower environmental cost, facilitating the transition to win-win diets – diets that provide healthy nutrition at an acceptable environmental cost. Integrating nutrient density to the functional unit for those foods that provide nutrition resulted in all legume-based products having a significantly lower environmental footprint across the acidification, climate change, marine and terrestrial eutrophication, resource-use-energy-carriers, and respiratory inorganics (Table 7.2). These categories are the same as the ones across which a serving of the legumes foods have a significantly lower environmental burden when excluding the mayonnaise comparison, excepted for the resource-use-energy-carriers category, for which the difference was not statistically significant in two of the burger patties compared when using a mass-based functional unit (Table 7.1). This suggests that the additional energy required for processing the legumes into palatable food products is worth the effort, as better nutrition is achieved.

With the NDU as the functional unit, the lower land use advantages of the legume-based products were further extended (Table 7.2). However, the land use limitation identified in the mass functional unit comparisons was still present in the NDU comparisons, with three pasta comparisons having a significantly higher land use burden for the chickpea pasta option. Therefore, the challenge of increasing legume yield remains. Other studies have highlighted several points to focus on to achieve this, including improving yield stability and tolerance to (a)biotic stress of legumes (Alandia et al., 2020), for example through intercropping (Viguier et al., 2018). It is nevertheless worth noting that legume yields modelled in existing agricultural production processes from Ecoinvent v3.6 (Wernet et al., 2016) and Agri-footprint v3.0 (Durlinger et al., 2017) were highly conservative, and we found that they were much lower than official numbers shown in FAOSTAT (FAO, 2019a). For example, pea production in Germany had a yield of around 2443 kg dry matter per hectare in Ecoinvent v3.6 (Wernet et al., 2016) and 3204 kg dry matter per hectare in Agri-footprint v3.0 (Durlinger et al., 2017), against an average of 4899 kg dry matter per hectare in FAOSTAT between 2017 and 2019 (FAO, 2019a), representing up to twice the amount recorded in the databases.

Table 7.2: Difference between the environmental score of one nutrient density unit of the legume product and one nutrient density unit of the traditional product with economic allocation of co-products.

Significance of the results ($p < 0.05$) that is common to all scenarios assessed in the respective case studies (such as allocation methods) is shown in colour, with green meaning that the legume product has a significantly lower burden than the traditional product, red significantly higher, and white non-significant.

Legume Product	Pasta				Protein balls		Burger patties	
	Chickpea (BG)	Chickpea (BG)	Chickpea (SP)	Chickpea (SP)	PPB	PPB	VB	VB
Traditional Product	Wheat (0% straw)	Wheat (80% straw)	Wheat (0% straw)	Wheat (80% straw)	MB (IE)	MB (BR)	BB (IE)	BB (BR)
Is the mean impact of 1 NDU of the legume product at least 0.2 standard deviation units significantly lower or higher than that of the traditional one?								
					lower	higher	NS	
Acidification ter. & freshwater	-75%	-74%	-86%	-85%	-97%	-86%	-95%	-78%
Cancer human health	-54%	-53%	-62%	-61%	-84%	-99%	-88%	-99%
Climate change	-69%	-68%	-78%	-77%	-88%	-91%	-79%	-86%
Ecotoxicity freshwater	-47%	-47%	-54%	-54%	-87%	-100%	76%	-97%
Eutrophication freshwater	-75%	-75%	-80%	-80%	-78%	-67%	-83%	-50%
Eutrophication marine	-69%	-63%	-95%	-94%	-97%	-95%	-94%	-91%
Eutrophication terrestrial	-76%	-74%	-90%	-90%	-98%	-90%	-97%	-88%
Ionising radiation, HH	-78%	-78%	-79%	-79%	-61%	-61%	-56%	-58%
Land use	9%	17%	-2%	5%	-92%	-98%	-92%	-98%
Non-cancer human health	-16%	-14%	-28%	-27%	-80%	-100%	-95%	-100%
Ozone depletion	-65%	-64%	-71%	-70%	-49%	-59%	-18%	-45%
Photochem. ozone formation	-58%	-57%	-66%	-65%	-86%	-80%	-75%	-60%
Resource use, energy carriers	-71%	-71%	-75%	-75%	-69%	-60%	-56%	-34%
Resource use mins. & metals	-39%	-34%	-82%	-81%	-99%	-45%	-95%	415%
Respiratory inorganics	-74%	-73%	-85%	-85%	-97%	-87%	-94%	-75%
Water scarcity	-73%	-72%	-77%	-77%	-82%	-56%	-83%	-42%

7.3. CARBON OPPORTUNITY COST AND THE NEED TO SPARE LAND

As mentioned in Section 7.1, land use is a key resource with great carbon sequestration potential. Reestablishment of native ecosystems from anthropogenic transformation is key to prevent global warming levels going above 1.5 degrees Celsius, especially land used for animal agricultural production (Hayek et al., 2020). The land area savings or costs achieved by substituting the traditional products with their legume-based alternatives were recorded in Table 7.3. The product that provided the greatest environmental savings in terms of constrained land area was the vegetarian burger patty when compared to the burger made from Brazilian or South African beef, with around 15.5 m² and 13.8 m² spared per serving, respectively. The product comparison with the highest percentage land use save was the pea protein ball when compared to the meatball made from Brazilian beef, with a 3281% saving. This thesis consequently highlights the large environmental benefits of substituting meat with legumes, specifically beef, to preserve human health and the environment, including land.

Hayek et al. (2020) estimated the effect of a shift to plant-based diet worldwide to sequester 332-547 GtCO₂, representing 99-163% of CO₂ emissions budget. In Chapters 4 and 5, we performed extrapolation scenarios and estimated the carbon saving effects of small changes in the consumption habits of Germany and the UK. We showed that a simple substitution of the vegetarian patty for all beef patties eaten in the UK could save to 9.5-11 million tonnes CO₂e once COC of land use are accounted for, which is equivalent to 2.1 to 2.4% of the annual territorial GHG emissions in the country in 2019 (Chapter 5). Substituting just 5% of German beef consumption for pea protein balls was estimated to save 8 million tonnes CO₂e annually, representing 1% of Germany's annual GHG emissions. These innovative, processed legume-based foods therefore hold potential in assisting in the sparing of land globally if substituted for beef products similar to those presented in the thesis.

7.4. FINAL CONSIDERATIONS

It is worth noting that at the product level, across these case studies as well as across the majority of the literature, environmental benefits brought by legumes when considering entire cropping-cycles (rotations) – instead of individual years of cultivation – are not included (Costa et al., 2020). Indeed, improved nutrient cycling and yield increase are benefits brought by integrating legumes into a farm's agricultural strategy which are not taken into account in product-level LCAs, and can make a key difference if break-crop effects mitigate the comparatively low yields of numerous legumes. This exclusion is also reflected in the current economical ecosystem of Europe, which favours specialisation of crops instead of diversification (Zander et al., 2016). The LCA approach also tends to promote intensive over organic agriculture, for which the latter is partly

characterised by crop diversification. Indeed, LCAs looking at agricultural products are often limited by their constricted product-based approach, weakness around assessing land degradation and the effects of pesticides, including loss of biodiversity (van der Werf et al., 2020). To understand the effects of introducing these products at a diet level, it is necessary to perform a consequential LCA, which would take into account market effects of such a substitution. The product-level data collected in this thesis will feed in the consequential LCA performed as part of the TRUE project.

This thesis provided product-based case studies highlighting the potential of legumes to assist in the transition towards more sustainable diets in Europe. The innovative legume-based foods that have the same aspect, texture, and ease of preparation facilitate the transition to more sustainable and healthy diets for consumers who are not ready to embrace dishes with minimally-processed legumes (Michel et al., 2021). However, existing literature shows that few consumers are willing to substitute meat for meat mocks (Hartmann and Siegrist, 2017). One of the reasons for this is that the majority of consumers is unaware of the large negative impact of meat on the environment, and the misconception that soy-based foods have a similar environmental impact than meat (Camilleri et al., 2019; Siegrist and Hartmann, 2019). Efforts should thus concentrate on raising awareness amongst the general public of the catastrophic consequences of current diets, and how they can be tackled substituting unhealthy foods with legumes, including processed legumes. However, scientific papers and policy documents are not targeted to the general public and there is a notable gap between the relevant parties. The aim of the TRUE project, within which this thesis is part of, is to facilitate knowledge exchange between stakeholders of the food value chain and therefore improve dissemination of results. In addition to TRUE, other European projects focusing on legumes are aiming to raise legume awareness amongst consumers, such as the INCREASE project (INCREASE, 2020) and the LEGVALUE project (LEGVALUE, 2017). In addition to European projects, the Legume Innovation Network (LEGVALUE, 2020) was established end of 2020 to bond legume stakeholders in Europe, including any legume-focused business and Non-Governmental Organisation (NGO), with the goal to facilitate knowledge exchange and European sustainable commercial development of legumes. This unity strengthens legume value chains and facilitates a viable transition to legume cultivation and consumption across Europe to achieve the desired goal: a more sustainable food system that remains within planetary boundaries.

Table 7.3: Differences in land area when substituting the traditional products with their legume-based alternatives presented in this thesis.

Results are per serving (pasta, protein balls, and burger patties) or per pot (mayonnaise). Green cells indicate a land saving while red cells indicate a land cost.

Legume Product	Pasta				Protein balls		Burger patties					Mayonnaise	
	Chickpea (BG)	Chickpea (BG)	Chickpea (SP)	Chickpea (SP)	PPB MB (IE)	PPB MB (BR)	VB BB (IE)	VB BB (BR)	VB BB (ZA)	VB BB (US)	VB BB (FR)	VEG EGG	CA EGG
Traditional Product	Wheat (0% straw)	Wheat (80% straw)	Wheat (0% straw)	Wheat (80% straw)									
Land area difference (m ²)	-0.33	-0.35	-0.28	-0.29	3.03	9.82	4.51	15.48	13.73	4.47	5.63	0.72	0.72
Land area percentage difference	-58%	-60%	-53%	-56%	1014%	3281%	536%	1840%	1631%	531%	669%	11%	11%

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9. APPENDIX

9.1. CHAPTER 4 APPENDIX

9.1.1. Nutritional analyses

Nutritional content of MBs was assumed to be the same regardless of the geographical provenance of the beef. The Kiedahl method (ISO 1871:2009) was used to analyse protein content (ISO, 2009). Energy content was determined following the EU regulation 1169/2011 (European Union, 2011). The enzymatic-Gravimetric Method from the AOAC 991.43 and AOAC 985.29 was used for determining the fibre content (Lee et al., 1992; Prosky et al., 1985), and gas chromatography (FID) from ISO 12966-1:2014; 12966-2:2011; 12966-3:2016 (ISO, 2016, 2014, 2011) for essential fatty acids. Random replicates were performed to 7% of the analysis by a credited laboratory that follows standard, verified, and certificated protocols, and that regularly works with industry.

9.1.2. Recipes

Table 9.1: Recipe per 100 g of pea protein balls.

Quantity	Ingredient
6.05 g	Fractionated peas rich in protein
6.63 g	Pea protein isolate
10.2 g	Potato starch
5.5 g	Cornflakes (100% corn)
2.0 g	Sugar
2.0 g	Spice mix (paprika, onion, and chives powder)
11.5 g	Onion
57.4 mL	Water

Table 9.2: Recipe per 100 g meatballs (Irish and Brazilian beef)

Quantity	Ingredient
70 g	Raw fresh beef
8.5 g	Vegetable oil
0.6 g	Salt
4.8 g	Soy protein
2.8 g	Onion
2.8 g	Garlic
8.5 mL	Water

9.1.3. Inventory of the protein and meatballs

Table 9.3: Inventory for a reference flow of 100 g of packaged pea protein balls and Swedish-style meatballs (BR and IE).

Stage	Input / output / process	Units	Pea		Beef (IE)		Beef (BR)	
			Input	Output	Input	Output	Input	Output
Main ingredient agricultural production	Fertiliser (N) w/o grain and meal	kg	0		0.012		0.004	
	Urea, as N w/o grain and meal	kg			0.003		0.003	
	Fertiliser (P ₂ O ₅) w/o grain and meal	kg	0.0012		0.002		0.007	
	Fertiliser (K ₂ O) w/o grain and meal	kg	0.0015		0.002		0.006	
	Feed for cattle (grain+meal)	kg			0.39		0.041	
	Manure, from cow	kg			0.96			
	Harvesting	m ²	0.34					
	Sowing	m ²	0.34					
	Lime	kg	0.014		0.082		0.10	
	Tillage, ploughing	m ²	0.34					
	Seed	kg	0.012					
	Herbicide	kg						
	Land, total of crops and grazing	m ²	0.34		4.7		10.5	
	Pea (dry matter)	kg		0.0927				
	Transport, truck >20t	kg*km			15		1.4	
	Water	L			0.18		0.019	
	Beef cattle for slaughter	kg				0.14	0.14	
Cleaning/	3.5-7.5 t truck with refrigeration	kg.km	5.5					
Screening/ Slaughtering	Electricity	Wh	2.0					
	Cleaned peas	kg		0.0509				
	5% loss from cleaning	kg		0.0025				
	Drinking water, water purification treatment	kg			0.31		0.31	
	Electricity mix, <1kV	MJ			0.060		0.060	
	Heat, from residual heating systems	MJ			0.023		0.023	
	Transport, truck	kg.km			20		20	
	Waste water, untreated	kg			0.31		0.31	
	Beef meat, fresh	kg				0.075		0.075
	Beef co-product, Cat.1/2 and waste	kg				0.034		0.034
	Beef co-product, Cat.3 by-products	kg				0.011		0.011
	Beef co-product, food grade bones	kg				0.012		0.012
	Beef co-product, food grade fat	kg				0.011		0.011
	Beef co-product, hides and skins	kg				0.011		0.011
De-hulling	Screening energy	Wh	2.1					
	De-hulling energy	Wh	5.1					
	De-hulled peas	kg		0.0432				
	Hulls	kg		7.63E-3				
	hulls transport to pellet mill, 16-32 t truck	kg.km	0.76					

Hulls as feed	pelleting energy	Wh	0.74		
	pellet transport to cattle farm, 16-32 t truck	kg.km	0.38		
Milling	Milling electricity	Wh	7.9		
	Milled peas	kg	0.0432		
Micronisation	Micronisation electricity	Wh	3.9		
	Micronised peas	kg	0.0432		
Fractionation	Fractionation electricity	Wh	27		
	Fractionated peas rich in starch	kg	0.0372		
	Fractionated peas rich in protein	kg	6.05E-3		
Protein extrusion	Protein isolates (>85% protein)	kg	6.1E-3		
	Protein isolate transport, 16-32 t truck	kg.km	6.6E-3		
	Electricity	Wh	0.21		
	Pea extrudates	kg	0.0115		
Premix production	Potato starch	kg	0.010		
	Potato starch transport, >32t	kg.km	1.3		
	Potato starch transport, train	kg*km	2.5		
	Potato starch transport, barge	kg*km	2.8		
	Sugar from sugar beet	kg	2.0E-3		
	Sugar truck transport, >32t	kg.km	0.26		
	Sugar transport, train	kg*km	0.47		
	Sugar transport, barge	kg*km	0.53		
	Bell pepper	kg	8.0E-3		
	Onion	kg	8.0E-3		
	Chives	kg	8.0E-3		
	Electricity, medium voltage (drying)	kWh	0.74		
	Mix Truck transport, >32t	kg.km	3.1		
	Mix transport, train	kg*km	5.8		
	Mix transport, barge	kg*km	6.5		
	Premix electricity	Wh	0.30		
	Premix	kg	0.020		
Onions	Onion	kg		2.9E-3	2.9E-3
	lorry with refrigeration machine, 3.5-7.5 t	kg*km		5.7E-3	5.7E-3
Garlic	Garlic (onion as proxy)	kg		2.9E-4	2.9E-4
	Garlic Truck transport, >32t	kg*km		0.024	0.024
	Garlic transport, train	kg*km		0.068	0.068
	Garlic transport, barge	kg*km		0.077	0.077
Vegetable oil	Vegetable oil	kg		8.5E-3	8.5E-3
	Oil Truck transport, >32t	kg*km		0.71	0.71
	Oil transport, train	kg*km		2.0	2.0
	Oil transport, barge	kg*km		2.3	2.3
Spice and soy products	Soy protein	kg		4.8E-3	4.8E-3
	Soy protein transport, lorry, >32t	kg*km		4.8	4.8
	Soy protein transport, transoceanic ship	kg*km		56	56
	Maize flour	kg	5.5E-3		
	Flour Truck transport, >32t	kg.km	0.72		
	Flour transport, train	kg*km	1.3		

Meatball/ protein ball production	Flour transport, barge	kg*km	1.5		
	Onions	kg	0.10		
	Salt	kg		5.7E-4	5.7E-4
	Onions transport, 3.5-7.5 t truck	kg.km	0.13		
	Beef transport, truck	kg.km		24.6	111
	Beef transport, ship	kg.km		174	830
	Water	L	0.057	0.011	0.011
Packaging	Electricity	Wh	3.8	0.028	0.028
	Corrugated board box	kg	0.02	0.02	0.02
	Packaging transport, lorry, >32t	kg*km	2.3	2.3	2.3
	Packaging transport, train	kg*km	4.3	4.3	4.3
	Packaging transport, ship	kg*km	5.5	5.5	5.5
	Packaging at filling plant	kg			
			0.015	0.015	0.015
Distribution	Meatball transport, lorry w ref machine, >32t	kg*km	5.5	5.5	5.5
	Meatball transport, lorry w ref machine, >32t	kg*km	1.6	1.6	1.6
	Meatball transport, lorry w ref machine, >32t	kg*km	2.3	2.3	2.3
	Meatball transport, transoceanic ship w ref	kg*km	415	415	415
	Meatball transport, lorry w ref machine, >32t	kg*km	35	35	35
	Meatball trans, passenger car	km	7.6E-7	7.6E-7	7.6E-7
	Energy for cooling	kWh	0.047	0.047	0.047
Use	Energy for cooking	kWh	0.55	1.1	1.1
	Cooked meatballs	g		100	100

9.1.4. Meatballs inventory

The study from Biswas & Naude (2016) was a streamlined LCA in which trace elements were excluded from the assessment of MBs, representing 0.33% of the total product. In the MB (BR) scenario, the process “market for cattle for slaughtering, live weight, BR” from Ecoinvent v3.6 (Wernet et al., 2016) was selected. It includes a mix of the different cattle systems in Brazil, from native or planted pastures, proteic supplementation, weaned systems, intensive production, feedlot, and from the Pantanal region or outside. The most common systems were the ones of intensive beef cattle and fat steers production on pasture, representing 50% of total beef cattle production. Included in these processes were farm management practices, such as sowing, fertilising, and tillage, as well as lime application, feeds, and additives. Live cattle was modelled to be transported by truck for 100 km to the slaughterhouse, and the obtained beef transported for 1483 km by truck and 11061 km by ship to Germany (Searates, 2020).

In the MB (IE) product, the process “Beef cattle for slaughter, at beef farm, PEF compliant Economic- IE” was used instead of the “market for cattle for slaughtering, live weight, BR”. This process was derived from average Irish beef farm characteristics, and supplied by Agri-footprint v3.0 (Durlinger et al., 2017). The Irish system is characterised by calves feeding on grass pastures for 60% of the year, and the remaining time on grass silage and compound feed in sheds. Calves are slaughtered from 2 years of age. To obtain 0.07 kg of raw fresh beef, 0.153 kg of beef cattle is fed with 0.429 kg of compound feed beef (29% of barley grain, 21% of maize, 9% of oat grain, 15% of oat rapeseed meal, 12% of soybean, 5% of sugar cane molasses, and 9% of wheat grain), 1.6 kg of grass silage, and grazes 8.09 kg of grass during its two years of existence. Live cattle was modelled to be transported by truck for 100 km to the slaughterhouse, and the obtained beef transported for 327 km by truck and 2322 km by ship to Germany (Searates, 2020).

The slaughtering process for the reference flow of both MB (IE) and MB (BR) was recorded in Table 9.3. Since the United States, Brazil, and Argentina are the top producers of soybean (FAO, accessed 03/2019), the production of soybean protein concentrate destined as an ingredient in MBs (IE) and (BR) was modelled as 50% produced in Brazil and 50% in Argentina. In Germany, the ingredients are then pre-cooked in 8.47 g of vegetable oil and cooked in a heating tunnel. The obtained MBs are pre-cooled with a pre-cooler, frozen, and packaged, consuming 0.027 kWh of electricity. A distance of 0.38 km of the consumer transport from the house to the retail centre was allocated to the purchasing of meatballs, following the PEF guidelines (European Commission, 2018a). The MBs are then stored for one day in the fridge, consuming 47 Wh per 100 g serving. The consumption phase was modelled using the “Preserves and preparation of meat PCR” (EPD International AB, 2019a) and the cooking instructions of IKEA meatballs (IKEA, 2019). The MBs are cooked in the oven for 15 minutes at 225 degrees Celsius, with 15 minutes of pre-heating. Following

the Product Category Rules of preserves and preparation of meat, cooking in the oven was modelled with an energy requirement of 1.1 kWh (EPD International AB, 2019a).

No data were available for the environmental burdens of producing the seasoning ingredients pepper, paprika, onion powder and chive powder in either databases. We substituted bell pepper production for paprika while onion production substituted onion and chive powder with processes from Agri-footprint v3.0 (Durlinger et al., 2017). Pepper production is a data gap that no proxy was found for, and due to its presence in small quantities, it was excluded from the analysis. The electricity amount needed to dry the ingredients for the spice mix was excluded from the analysis, due to the lack of information available on spice production. Packaging was composed of a simple cardboard carton (Biswas and Naude, 2016).

Beef meat for MB (IE) and MB (BR) was allocated 26.8% of the environmental burdens of cattle rearing when biophysical allocation was applied, as recommended by the European Commission Cattle Model Working Group (Chen et al., 2017), and 92.9% when economic allocation was applied (European Commission, 2018a) – see Table 9.4.

Table 9.4: Economic and biophysical allocation from beef production (Chen et al., 2017; European Commission, 2018a).

Products	Biophysical allocation (%)	Economic allocation (%)
Cat. 1/2 and waste	16.3	0
Cat.3 by-products	36.0	0.8
Other food grades	15.1	2.8
Hides and skins	5.8	3.5
Beef meat, fresh	26.8	92.9

9.1.5. Pea protein balls inventory

Agricultural production data for peas were extracted from the German State Institute for Agriculture, Food and Rural Areas (LEL Schwäbisch Gmünd, 2020). Per hectare, 44 kg of K₂O and 35 kg of P₂O₅ fertilisers were applied. Yields (91% dry matter grain) were 2700 kg (dry yield) per hectare. In terms of machinery operations, one fertiliser and one lime application, tillage-ploughing, sowing, and harvesting were modelled using Ecoinvent v3.6 processes (Wernet et al., 2016). Post-farm gate data were provided by IGV GmbH (IGV, 2019). Germany is the second largest European

producer of field peas behind France (Eurostat, 2019). It was assumed that the region for pea cultivation for PPBs was Lower Saxony, as the larger farms in the west of Germany are mainly in Schleswig-Holstein and Eastern Lower Saxony (Britannica, 2019), and Lower Saxony is closer to IGV GmbH. IPCC equations 11.2, 11.9, and 11.10 were used to determine direct and indirect emissions from crop residues, based on a residue N content of 0.008 kg N per kg dry matter (IPCC, 2019). Carbon dioxide emissions stemming from lime application were determined following IPCC (2006), and P losses to water following fertiliser application were based on cropland loss coefficients used by Styles et al. (2015). Distribution of fertiliser types specific to Germany was extracted from the International Fertilizer Association (2020). Crop protection was excluded from the study, due to the extensive variability in specifications about quantity and types. Although lime is commonly applied to counter soil acidification caused largely by ammonium-based fertilisers (Defra, 2010), which were not applied in our case for peas, we modelled average lime input to cropland (400 kg/ha/yr) as a conservative approach.

For the reference flow relating to 100 g of protein balls made from peas and other vegetables, 92.7 g of peas are harvested, from which 39.3 g are inputted in an existing Ecoinvent v3.6 pea protein isolation process (Wernet et al., 2016) mentioned in the following paragraph. The other 53.4 g are transported by lorry for 283 km to the factory, where they are screened and cleaned, consuming 2.1 Wh of electricity. The screening stage separates around 5% of the total weight for peas, composed of any unfit peas due to insect damage, under sized seeds, discolouration, stain, varietal admixture, stones or pieces of chalk, and other defects (Askew & Barrett pulses ltd, 2020). The clean peas are then dehulled with a vertical peeler with corundum insert. The amount of electricity required for this process was extracted from Draganovic et al. (2013), and amounts to 5.1 Wh. Two products are obtained from this process, 43.2 g of dehulled peas, and 7.6 g of hulls. The dehulled peas are then dry-milled, using 7.9 Wh of electricity (Colantoni et al., 2017). The obtained flour is then micronized with a classifier mill and subsequently fractionated with a classifier, consuming 3.89 and 27 Wh of electricity, respectively (IGV, pers. com., 2019). A 37.2 g starch-rich fraction and a 6.05 g protein-rich fraction with 40-50% protein content are obtained.

The protein-rich fraction is then added to 6.6 g of pea protein isolate purchased from another company, for which the upstream burdens were modelled in the same way as for the pea protein-rich fraction previously mentioned, and extruded with a twin-screw extruder, consuming 0.21 Wh of electricity. 5% of the isolate and fraction is lost during this process, yielding 12.1 g of extrudates. The pea protein isolation process produces three by-products, 4.04 g of hulls, 235 g of slurry, and 15.1 g of starch. Other ingredients in the PPBs include a premix made from potato starch, sugar, bell pepper, onion, chives, and cornflakes, onions, and water, which are mixed together with the

pea extrudates to form the balls. Onions were used as a proxy for chives, due to the lack of specific data on the ingredient. The processes of onion and soybean production were the same as the ones used for MBs. Onions were grown in the village 2 km away from the IGV GmbH factory, and the process for onion production in the Netherlands from Ecoinvent v3.6 (Wernet et al., 2016) was used due to the absence of such a process for Germany. The global process of bell pepper production from Ecoinvent v3.6 (Wernet et al., 2016) was used as a proxy for paprika, and energy required for drying was excluded. Maize flour (Durlinger et al., 2017) was used as a proxy for cornflakes. The packaging, distribution, and storage phases were the same as for the MBs. The Ecoinvent v3.6 global process “market for sugar, from sugar beet” was used for the sugar in the premix. Cooking instructions were obtained from IGV GmbH. The PPBs are cooked for six minutes with medium heat in the pan, requiring 0.55 kWh of electricity (EPD International AB, 2019a).

Mass balance diagrams of pea milling and of pea protein isolation were recorded in Figure 9.1. Economic and physical allocation factors for pea dehulling and fractionation were recorded in Tables 9.5-9.7.

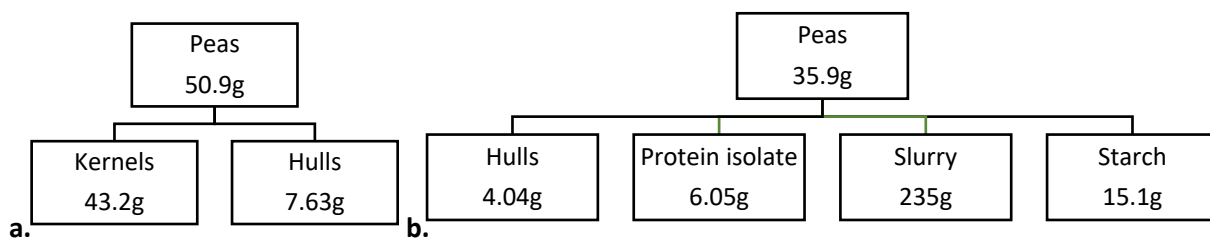


Figure 9.1: Mass balance diagrams of pea co-products.

Mass balance of the harvested peas used to produce pea flour. Information obtained from IGV GmbH (IGV, 2019) 9.1.b. Mass balance of the harvested peas used to produce pea protein isolate. Data obtained from Agri-footprint (Durlinger et al., 2017). Note: different proportions of hulls are obtained in the two cases: in case a., hulls represent nearly 15% of the peas, while in case b. hulls represent less than 11.5% of the peas.

Table 9.5: Allocation percentages of pea co-products. Adapted from (Davis and Sonesson, 2008)

Components of whole peas	Mass (%)	Economic allocation (%)
Dehulled peas (protein+starch)	85	88
Pea residuals (hulls, fibre)	15	12

Table 9.6: Economic allocation from fractionation step. Extracted from Agri-footprint 5.0 (van Paassen et al., 2019).

Products	Mass fraction (%) from case study	Price (euro/ton) from Agri-footprint 5.0	Economic allocation (%), calculated
Pea, protein-concentrate	14	1600	34

Pea, starch-concentrate	86	495	66
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Table 9.7: Economic allocation from fractionation step. Extracted from Agri-footprint 5.0 (van Paassen et al., 2019).

Products	Mass fraction (%) from Agri-footprint 3.0	Economic allocation (%), from Agri-footprint 3.0
Pea, hulls	12	3
Pea, protein-isolate	19	63
Pea, slurry	23	0
Pea, starch	46	34

At the time of the study, the hulls obtained from the PPB process were discarded, as the product was manufactured in small quantities. However, in the case where production increases to an industrial level, pea hulls could become a valuable co-product as an additive in foods to increase gastrointestinal health (Dahl et al., 2003; Flogan and Dahl, 2010) and in the production of crisps and puffs (Pulse Canada, 2016), or for animal feed (Lienhardt et al., 2019). Pea starch obtained from pea fractionation can be used as a binding and thickening agent in the food industry (Emsland Group, 2019), thus substituting various starches when expanding the system boundaries. However, due to the difficulty of performing system boundary expansion with product substitution for beef co-products due to the numerous substitution possibilities, no boundaries expansion was performed in this study.

9.1.6. Modified Null Hypothesis Significance Testing

Table 9.8: Summary of the results of the modified null hypothesis significance tests common to all scenarios of the research comparing the pea protein balls, meatballs (Brazil), and meatballs (Ireland).

Consistent across all scenarios				
Meaning of result >		Is the mean impact of j at least 0.2 standard deviation units significantly lower than that of k ?		
		no	yes	
Impact				
Acidification terrestrial and freshwater	$j \downarrow \quad k \rightarrow$	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		no
	MB (BR)	no	yes	

Cancer Human Health	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		no	yes
	MB (IE)	no		yes
	MB (BR)	no	no	
Climate Change	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		yes
	MB (BR)	no	no	
Ecotoxicity freshwater	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		no	no
	MB (IE)	no		no
	MB (BR)	no	no	
Eutrophication freshwater	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		no	no
	MB (IE)	no		no
	MB (BR)	no	no	
Eutrophication marine	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		no
	MB (BR)	no	no	
Eutrophication terrestrial	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		no
	MB (BR)	no	yes	
Ionising radiation, HH	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		no	no
	MB (IE)	no		no
	MB (BR)	no	no	
Land use	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		yes
	MB (BR)	no	no	
Non-cancer human health effects	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		no	yes
	MB (IE)	no		no
	MB (BR)	no	no	
Ozone depletion	j↓ k→	PPB	MB (IE)	MB (BR)

	PPB		no	yes
	MB (IE)	no		no
	MB (BR)	no	no	
Photochemical ozone formation, HH	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		no
	MB (BR)	no	no	
Resource use, energy carriers	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		no
	MB (BR)	no	no	
Resource use, mineral and metals	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		no
	MB (BR)	no	yes	
Respiratory inorganics	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		yes	yes
	MB (IE)	no		no
	MB (BR)	no	yes	
Water scarcity	j↓ k→	PPB	MB (IE)	MB (BR)
	PPB		no	no
	MB (IE)	no		no
	MB (BR)	no	no	

9.1.7. Additional analyses

Table 9.9: Summary of economically-allocated environmental burdens for pea protein balls, meatballs (Brazil), and meatballs (Ireland), expressed per 100 g cooked product without beef transport.

From smallest (1) to largest (3) environmental impact:

1  2  3 

Impact category	Unit	Impact per serving		
		PPB	MB(IE)	MB(BR)
Acidification ter. & freshwater	mol H ⁺ eq	0.0034	0.07	0.02
Cancer human health	CTUh	1.10E-08	5.0E-08	7.9E-07
Climate change (w/o COC)	kg CO ₂ eq	0.5	3.3	4.5
Climate change + COC	kg CO ₂ eq	1.1	13.4	14.6
Ecotoxicity freshwater	CTUe	1.1	5.7	270.9
Eutrophication freshwater	kg P eq	0.0003	1.2E-03	7.9E-04
Eutrophication marine	kg N eq	0.0013	0.03	0.02
Eutrophication terrestrial	mol N eq	0.0088	0.3	0.1
Ionising radiation, HH	kBq U ²³⁵ eq	0.17	0.3	0.3
Land use	Points	38	354	1185
Non-cancer human health	CTUh	6.6E-07	2.3E-06	4.2E-04
Ozone depletion	kg CFC-11 eq	6.5E-08	8.7E-08	9.1E-08
Photochem. ozone form.	kg NMVOC eq	0.0014	0.008	0.005
Resource use, energy carriers	MJ	8.6	19.5	14.3
Resource use mins. & metals	kg Sb eq	3.40E-09	2.9E-07	4.3E-09
Respiratory inorganics	disease inc.	2.20E-08	5.0E-07	1.2E-07
Water scarcity	m ³ depriv.	0.27	1.03	0.43

Table 9.10: Summary of physically-allocated environmental burdens for pea protein balls and biophysically-allocated burdens for both meatballs (Irish and Brazilian), expressed per 100 g cooked product.

From smallest (1) to largest (3) environmental impact:		1	2	3
Impact category	Unit	Impact per serving (100 g, cooked)		
		PPB	MB(IE)	MB(BR)
Acidification terrestrial and freshwater	mol H+ eq	0.003	0.02	0.01
Cancer human health	CTUh	1.0E-08	2.1E-08	4.3E-07
Climate change w/o COC	kg CO ₂ eq	0.5	1.5	2.9
Climate change + COC	kg CO ₂ eq	0.7	4.2	5.6
Ecotoxicity freshwater	CTUe	1.0	2.2	146
Eutrophication freshwater	kg P eq	0.0003	0.001	0.001
Eutrophication marine	kg N eq	0.001	0.009	0.01
Eutrophication terrestrial	mol N eq	0.008	0.1	0.04
Ionising radiation, HH	kBq U-235 eq	0.2	0.3	0.3
Land use	Point	24	113	645
Non-cancer human health	CTUh	6.6E-07	7.4E-07	2.3E-04
Ozone depletion	kg CFC11 eq	6.3E-08	8.7E-08	1.1E-07
Photochemical ozone formation	kg NMVOC eq	0.001	0.004	0.004
Resource use, energy carriers	MJ	8.2	15.1	14.82
Resource use minerals and metals	kg Sb eq	2.6E-09	8.6E-08	3.4E-09
Respiratory inorganics	disease inc.	2.1E-08	1.6E-07	7.6E-08
Water scarcity	m ³ depriv.	0.26	0.47	0.34

Table 9.11: Summary of economically-allocated environmental burdens for pea protein balls with German and European electricity mixes, and pea protein balls with Swedish electricity mix, expressed per 100 g cooked product.

From smallest (1) to largest (2) environmental impact: 1  2 

		German and European mix	Swedish mix
Impact category	Unit	PPB	PPB
Acidification ter. & freshwater	mol H ⁺ eq	0.003	0.002
Cancer human health	CTUh	1.1E-08	9.3E-09
Climate change	kg CO ₂ eq	0.5	0.3
Ecotoxicity freshwater	CTUe	1.1	1.0
Eutrophication freshwater	kg P eq	0.0003	0.0001
Eutrophication marine	kg N eq	0.001	0.001
Eutrophication terrestrial	mol N eq	0.009	0.007
Ionising radiation, HH	kBq U ²³⁵ eq	0.2	0.3
Land use	Points	38	40
Non-cancer human health	CTUh	6.6E-07	6.5E-07
Ozone depletion	kg CFC11 eq	6.5E-08	5.8E-08
Photochem. ozone form.	kg NMVOC eq	0.001	0.001
Resource use, energy carriers	MJ	8.6	6.8
Resource use mins. & metals	kg Sb eq	3.4E-09	3.3E-09
Respiratory inorganics	disease inc.	2.2E-08	1.9E-08
Water scarcity	m ³ depriv.	0.27	0.26

9.2. CHAPTER 5 APPENDIX

Table 9.12: Ingredients in one vegetarian burger patty.

1.1 g	Legume-based additive
45.5 g	Fibrous ingredients
11.4 g	Oil ingredients
29.9 g	Legume-based protein
5.7 g	Cereal-based protein
36.3 g	Flavouring and seasoning ingredients
11.4 g	Bulking ingredients
1.5 g	Vitamins

Table 9.13: Physical, biophysical, and economic allocation factors for all of the co-products used as ingredients in the study.

	(Bio)physical allocation	Economic allocation
Other legume-based ingredients	0%	1%
Lecithin	0%	1%
Oil 1	64%	92%
Legume protein concentrate 1	56%	63%
Starch 1	74%	86%
Oil 2	96%	99%
Starch 2	10%	3%
Legume protein concentrate 2	47%	93%
Oil 2	98%	99%
Cereal protein	7%	77%
Oil 3	53%	72%
Beef meat	27%	93%

Table 9.14: Positive results of the modified null hypothesis significance tests that were common to the economic allocation with mass or Nutrient Density functional units, and the (bio)physical allocation with mass or Nutrient Density functional units.

Negative values from the Monte Carlo analyses were adjusted to zero (Muller et al., 2016).

Common to all scenarios investigated				
Is the mean impact of j at least 0.2 standard deviation units significantly lower than that of k ?				
	no	yes		
$j \downarrow k \rightarrow$	VB	BB (IE)	BB (BR)	
VB		yes	yes	
BB (IE)	no		no	
BB (BR)	no	yes		
$j \downarrow k \rightarrow$	VB	BB (IE)	BB (BR)	
VB		no	yes	
BB (IE)	yes		yes	
BB (BR)	no	no		

		VB	MB (IE)	MB (BR)
	j↓ k→			
VB			yes	yes
MB (IE)		no		yes
MB (BR)		no	no	
		VB	BB (IE)	BB (BR)
	j↓ k→			
VB			no	yes
BB (IE)		yes		yes
BB (BR)		no	no	
		VB	BB (IE)	BB (BR)
	j↓ k→			
VB			yes	no
BB (IE)		no		no
BB (BR)		no	no	
		VB	MB (IE)	MB (BR)
	j↓ k→			
VB			yes	yes
MB (IE)		no		no
MB (BR)		no	yes	
		VB	MB (IE)	MB (BR)
	j↓ k→			
VB			yes	yes
MB (IE)		no		no
MB (BR)		no	yes	
		VB	MB (IE)	MB (BR)
	j↓ k→			
VB			no	no
MB (IE)		no		no
MB (BR)		no	no	
		VB	MB (IE)	MB (BR)
	j↓ k→			
VB			yes	yes
MB (IE)		no		yes
MB (BR)		no	no	
		VB	BB (IE)	BB (BR)
	j↓ k→			
VB			no	yes
BB (IE)		no		yes
BB (BR)		no	no	
		VB	BB (IE)	BB (BR)
	j↓ k→			
VB			no	no

BB (IE)	no		yes
BB (BR)	no	no	
j↓ k→	VB	BB (IE)	BB (BR)
VB		no	no
BB (IE)	no		no
BB (BR)	no	yes	
j↓ k→	VB	BB (IE)	BB (BR)
VB		no	no
BB (IE)	no		no
BB (BR)	no	yes	
j↓ k→	VB	BB (IE)	BB (BR)
VB		yes	no
BB (IE)	no		no
BB (BR)	yes	yes	
j↓ k→	VB	BB (IE)	BB (BR)
VB		yes	yes
BB (IE)	no		no
BB (BR)	no	yes	
j↓ k→	VB	BB (IE)	BB (BR)
VB		yes	no
BB (IE)	no		no
BB (BR)	no	no	

Table 9.15: Results of the modified null hypothesis significance tests comparing all burger options modelled in this study with economically-allocated burdens for co-products, with the mass functional units.

Negative values from the Monte Carlo analyses were adjusted to zero (Muller et al., 2016).

Meaning of result >		Is the mean impact of j at least 0.2 standard deviation units significantly lower than that of k ?					
		no		yes			
Impact							
Acidification terrestrial and freshwater	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	yes	yes	yes
	BB (IE)	no		no	no	no	no
	BB (BR)	no	yes		yes	yes	yes
	BB (ZA)	no	yes	no		yes	no
	BB (US)	no	yes	no	no		no
	BB (FR)	no	yes	no	no	yes	
Cancer human health effects	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		no	yes	yes	no	no
	BB (IE)	yes		yes	yes	yes	yes
	BB (BR)	no	no		no	no	no
	BB (ZA)	no	no	no		no	no
	BB (US)	yes	no	yes	yes		no
	BB (FR)	yes	no	yes	yes	yes	
Climate change	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	yes	yes	yes
	BB (IE)	no		yes	no	yes	yes
	BB (BR)	no	no		no	no	no
	BB (ZA)	no	no	yes		yes	yes
	BB (US)	no	no	yes	no		no
	BB (FR)	no	no	yes	no	no	
Ecotoxicity freshwater	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		no	yes	yes	yes	no
	BB (IE)	yes		yes	yes	yes	no
	BB (BR)	no	no		no	no	no
	BB (ZA)	no	no	yes		no	no
	BB (US)	no	no	yes	yes		no
	BB (FR)	yes	yes	yes	yes	yes	
Eutrophication freshwater	i↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)

	VB		yes	no	yes	yes	yes
	BB (IE)	no		no	yes	no	no
	BB (BR)	no	yes		yes	yes	yes
	BB (ZA)	no	no	no		no	no
	BB (US)	no	yes	no	yes		no
	BB (FR)	no	yes	no	yes	yes	
Eutrophication marine	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	yes	yes	yes
	BB (IE)	no		no	no	no	no
	BB (BR)	no	yes		no	yes	no
	BB (ZA)	no	yes	yes		yes	yes
	BB (US)	no	yes	no	no		no
	BB (FR)	no	yes	yes	no	yes	
Eutrophication terrestrial	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	yes	yes	yes
	BB (IE)	no		no	no	no	no
	BB (BR)	no	yes		yes	yes	yes
	BB (ZA)	no	yes	no		no	no
	BB (US)	no	yes	no	yes		no
	BB (FR)	no	yes	no	yes	yes	
Ionising radiation, HH	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	no	yes	yes	yes
	BB (IE)	no		no	no	yes	yes
	BB (BR)	no	no		yes	yes	yes
	BB (ZA)	no	no	no		yes	yes
	BB (US)	no	no	no	no		no
	BB (FR)	no	no	no	no	yes	
Land use	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	yes	yes	yes
	BB (IE)	no		yes	yes	no	yes
	BB (BR)	no	no		yes	no	no
	BB (ZA)	no	no	no		no	no
	BB (US)	no	yes	yes	yes		yes
	BB (FR)	no	no	yes	yes	no	
Non-cancer human health effects	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	yes	yes	yes
	BB (IE)	no		yes	yes	no	no
	BB (BR)	no	no		no	no	no
	BB (ZA)	no	no	yes		no	no

	BB (US)	no	yes	yes	yes		yes
	BB (FR)	no	no	yes	yes	no	
Ozone depletion	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		no	no	yes	yes	yes
	BB (IE)	yes		yes	yes	yes	yes
	BB (BR)	no	no		yes	yes	yes
	BB (ZA)	no	no	no		no	no
	BB (US)	no	no	no	no		no
	BB (FR)	no	no	no	no	no	
Photochemical ozone formation, HH	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	no	yes	yes
	BB (IE)	no		no	no	yes	no
	BB (BR)	no	yes		no	yes	no
	BB (ZA)	yes	yes	yes		yes	yes
	BB (US)	no	no	no	no		no
	BB (FR)	no	yes	no	no	yes	
Resource use, energy carriers	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	no	yes	yes	yes
	BB (IE)	no		no	no	yes	no
	BB (BR)	no	yes		no	yes	yes
	BB (ZA)	no	yes	no		yes	yes
	BB (US)	no	no	no	no		no
	BB (FR)	no	yes	no	no	yes	
Resource use, mineral and metals	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	no	yes	yes	yes
	BB (IE)	no		no	yes	yes	yes
	BB (BR)	yes	yes		yes	yes	yes
	BB (ZA)	no	no	no		no	no
	BB (US)	no	no	no	yes		no
	BB (FR)	no	no	no	yes	yes	
Respiratory inorganics	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)
	VB		yes	yes	yes	yes	yes
	BB (IE)	no		no	no	no	no
	BB (BR)	no	yes		yes	yes	yes
	BB (ZA)	no	yes	no		no	no
	BB (US)	no	yes	no	yes		no
	BB (FR)	no	yes	no	yes	yes	
Water scarcity	j↓ k→	VB	BB (IE)	BB (BR)	BB (ZA)	BB (US)	BB (FR)

	VB		yes	no	no	yes	NA
	BB (IE)	no		no	no	yes	NA
	BB (BR)	no	no		no	yes	NA
	BB (ZA)	no	yes	no		yes	NA
	BB (US)	no	no	no	no		NA
	BB (FR)	NA	NA	NA	NA	NA	

9.3. CHAPTER 6 APPENDIX

Table 9.16: Results of the modified null hypothesis significance test for the vegan mayonnaise with aquafaba processed in Mexico (VEG), aquafaba processed in Can-ada (CA), and egg mayonnaise (EGG).

Negative values from the Monte Carlo analyses were adjusted to zero (Muller et al., 2016).

Meaning of result >		Economic allocation, mass FU		
		Is the mean impact of <i>j</i> at least 0.2 standard deviation units significantly lower than that of <i>k</i> ?		
		no	yes	
Impact				
Acidification terrestrial and freshwater	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	yes		no
	CA	yes	yes	
Cancer Human Health	j↓ k→	VEG	EGG	CA
	VEG		yes	no
	EGG	no		no
	CA	yes	yes	
Climate Change	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	yes		yes
	CA	yes	no	
Ecotoxicity freshwater	j↓ k→	VEG	EGG	CA
	VEG		yes	no
	EGG	no		no
	CA	yes	yes	
Eutrophication freshwater	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	yes		no
	CA	yes	yes	
Eutrophication marine	j↓ k→	VEG	EGG	CA
	VEG		no	no

	EGG	no		no
	CA	yes	no	
Eutrophication terrestrial	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	no		no
	CA	yes	yes	
Ionising radiation, HH	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	yes		no
	CA	yes	no	
Land use	j↓ k→	VEG	EGG	CA
	VEG		yes	no
	EGG	no		no
	CA	no	yes	
Non-cancer human health effects	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	no		no
	CA	no	no	
Ozone depletion	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	yes		yes
	CA	yes	no	
Photochemical ozone formation, HH	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	yes		yes
	CA	yes	no	
Resource use, energy carriers	j↓ k→	VEG	EGG	CA
	VEG		no	yes
	EGG	yes		yes
	CA	no	no	

Resource use, mineral and metals	j↓ k→	VEG	EGG	CA
	VEG		yes	no
	EGG	no		no
	CA	yes	yes	
Respiratory inorganics	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	yes		no
	CA	yes	yes	
Water scarcity	j↓ k→	VEG	EGG	CA
	VEG		no	no
	EGG	no		yes
	CA	no	no	

9.4. OTHER PUBLICATIONS

The links to publications for which I was a co-author and that are part of the TRUE project are recorded in this Section. A meta-analysis of climate change and eutrophication values at farm gate of raw food products was combined with the nutrient density scores of these foods in 9.4.1. The report showed that fruits, vegetables, and legumes had overall lower environmental burdens than red meat and dairy. When applying these values to the average European diet and the reference diet from the EAT-Lancet commission (Willett et al., 2019), the former scored comparatively low, highlighting the potential for food sustainability improvement in Europe. The report also showed that legumes had the lowest carbon footprint per nutrient density unit. In 9.4.2, the analysis is extended to the average Irish diet and combined environmental, nutritional, and economic assessments. The report showed that it is possible to increase nutritional quality and reduce environmental burdens while decreasing costs per capita if the Irish diet transits towards the EAT-Lancet reference diet (Willett et al., 2019).

Section 9.4.3 is a comparative LCA of the alcohol gin made from peas or from wheat grains. The study, published in Environment International, showed that the pea-gin was associated with overall lower environmental burdens than the wheat-gin, although the pea-gin required 112% more land. A simple system boundary expansion suggested that 2.2 kg CO₂ equivalents are avoided for each litre of pea gin produced instead of wheat-gin. Finally, Section 9.3.4 is a literature review of the approaches undertaken when performing LCA studies of legumes, more specifically the modelling around crop rotations, N carryover and leaching, and the choice of functional unit. The review showed that legume interactions in rotations are often ignored in LCAs, limiting general awareness of their sustainable benefits.

9.4.1. TRUE report: The environmental assessment of diets

(Williams et al., 2020b) <https://zenodo.org/record/3732026#.YFZGbp1KjIU>

9.4.2. TRUE report: A combined environmental and nutri-economic assessment of diets

(Williams et al., 2020a) <https://zenodo.org/record/3747542#.YFZGzZ1KjIU>

9.4.3. Just the tonic! Legume biorefining for alcohol has the potential to reduce Europe's protein deficit and mitigate climate change

(Lienhardt et al., 2019) <https://www.sciencedirect.com/science/article/pii/S0160412019308773>

9.4.4. Representing crop rotations in life cycle assessment: a review of legume LCA studies

(Costa et al., 2020) <https://link.springer.com/article/10.1007/s11367-020-01812-x>