JNCC Report 764

Methods to compare domestic to overseas commodity production impacts
(*Guidance Report*)

C. Brice, O. Birks, I. Hartley and M. Harris

April 2024

ISSN 0963 8091

JNCC's report series serves as a record of the work undertaken or commissioned by JNCC. The series also helps us to share, and promote the use of, our work and to develop future collaborations.

For further information on JNCC's report series please contact:
Joint Nature Conservation Committee, Quay House, 2 East Station Road, Fletton Quays, Peterborough, PE2 8YY.

https://jncc.gov.uk/

Communications@jncc.gov.uk

This report should be cited as:
Brice, C., Birks, O., Hartley, I. & Harris, M. 2024. Methods to compare domestic to overseas commodity production impacts. *JNCC Report 764 (Guidance report)*, JNCC, Peterborough, ISSN 0963-8091.
https://hub.jncc.gov.uk/assets/126ddb88-5db2-4ddd-9735-c2ed10b4ed5f.

This report is compliant with JNCC's Evidence Quality Assurance Policy https://jncc.gov.uk/about-jncc/corporate-information/evidence-quality-assurance/.

Summary

There is an increasing policy focus upon sustainable consumption due to the recognised environmental impacts from commodity production. To guide policy related to land use, food security and importation, it is useful for policy makers to be able to compare the production impacts of a given commodity produced domestically to a production system in another country. Various methods could be applied to enable these comparisons, though they vary in terms of data availability, ease of application, the impacts they assess and the production countries and commodities which can be compared. This literature review identified methods that could be used to compare production impacts and assessed how useful they would be for the proposed purpose. Life Cycle Assessments (LCAs) were found to be the most suitable method due to data availability, ease of application using existing studies or software, the range of impacts which can be assessed and the ability to apply the method to a wide range of commodities and countries to enable comparisons to be made. The review is followed by a case study demonstrating the use of LCAs to understand the differences in the environmental impacts of rapeseed oil and palm oil produced, as substitutable commodities that are largely produced domestically and overseas (Appendix 1). The case study does not undertake a primary LCA study but synthesises existing knowledge from the literature.

Contents

1 Introduction

A robust method is required that can quantify impacts and enable a comparison between production systems to enable a shift towards more sustainable food systems and to reduce the negative environmental impacts associated with the production of commodities consumed in the UK. There is often a lack of available data and information on the environmental impacts of food systems, which can be seen as a barrier to a move to more sustainable systems (Clark *et al.* 2022). This literature review investigates methods which could be used to compare production systems for a given commodity. To enable the development of policy to support more sustainable food systems, it is essential to be able to assess the impacts of commodity production and make comparisons between different production systems, including between production systems typically found domestically and overseas. Findings from these comparisons could be used to guide domestic land use policy, to ensure that commodities which can be produced with fewer and less severe impacts domestically than overseas are prioritised domestically, and reliance on imports of these are reduced. Additionally, these investigations could guide trade policy and decisions, and enable the prioritisation of the importation of commodities which result in greater impacts when produced in domestic systems compared to specific production systems in other producer countries. Therefore, importation from specific production systems overseas that have lower associated impacts could be prioritised where possible.

1.1 Policy context

Consumption is associated with environmental impacts including biodiversity loss and greenhouse gas emissions and is also relevant to concerns around food and resource security and supply chain resilience. Therefore, there has been increased interest and policy development around sustainable consumption. The topic has recently been covered in high profile reports including the National Food Strategy, the Dasgupta review and the first draft of the Convention on Biological Diversity's post-2020 framework. The following policy documents from the four UK countries also recognise the issue:

- England's 25 Year Environment Plan aims to "avoid improving our domestic environment at the expense of the environment globally".
- Wales' 2015 Wellbeing of Future Generations Act has a goal for "a globally responsible Wales," which includes "ensuring that our supply chains are fair, ethical and sustainable," "supporting sustainable behaviour," and "efficient use of resources".
- Scotland's 2020 Environment Strategy includes an outcome that "we are responsible global citizens with a sustainable international footprint".
- Northern Ireland's Sustainable Development Strategy has a guiding principle of "living within environmental limits".

Previous JNCC work has developed the Global Environmental Impacts of Consumption (GEIC) indicator which estimates the tropical deforestation, biodiversity loss and water use associated with consumption. It breaks down each impact type by the commodities and producing countries associated with the impact. It is used as an indicator both nationally as a UK experimental statistic, and internationally as a component indicator in the Convention on Biological Diversity's *Kunming-Montreal Global Biodiversity Framework*. It is based on combining modelled trade flows with environmental data (Croft *et al.* 2022) and was designed to be used as a hot-spotting tool to prioritise the sectors and geographies where interventions by a consumer country are most likely to make a difference.

It would be possible to use data from the indicator as a starting point for comparing the average production systems of two countries: users could filter the dashboard to show information about production rather than consumption and perform some simple analyses to get comparable statistics for two countries of interest (dividing the impact by the tonnes produced). However, as the indicator is global in scope, it relies on global and publicly accessible data sources. It is therefore likely that, except when comparing to the most data poor regions, more detailed data on production impacts could be found if narrowing the scope to directly compare two specific and defined production systems. This report therefore aims to build on previous work by exploring alternative methods that could be used more effectively when wishing to compare production systems at a more granular level.

1.2 Aims and scope

The aim of this review is to give an overview of potential methods that could be employed to compare the impacts of commodities produced in defined production systems domestically and overseas. The goal was to identify which methods could be applied with relative ease by governments to investigate the potential impacts of policies and trade agreements influencing where commodities consumed in the UK are produced. Multiple factors interact when determining where it is best to produce a given commodity, such as the efficiency of producing a commodity in each climate or land type, the complexity of the supply chain, the negative and positive impacts production may have locally and globally, food security considerations, existing trade agreements and transportation implications. Therefore, to enable informed decisions to be made around which commodities should be produced domestically and which should be imported from specific overseas production systems, a wide range of considerations need to be considered. This is a challenging task due to the amount of data required to enable such an assessment and the lack of data often available, particularly for some overseas systems.

Assessing the impacts of domestic versus overseas production is a complex issue and the situation will differ greatly between commodities. For example, for commodities which can be produced with low impacts domestically, such as crops suited to the UK climate, it may be better to produce these within the UK rather than import them, offshoring the impacts and requiring additional transportation. However, some commodities which are produced overseas with lower impacts than domestically, such as crops which thrive in a tropical climate, would be more sensible to import even with the additional transportation emissions. Additionally, it is important to consider comparisons between alternative commodities. For example, with oil crops, palm oil may be more commonly grown in tropical regions, whereas rapeseed, which can be used as an alternative, would be a more suitable oil seed crop to grow in the UK. Therefore, a suitable impact comparison method to guide policy decisions on domestic land use and trade should enable an impact comparison between alternative, substitutable commodities.

This review will focus on the methods available for agricultural commodity production comparisons. However, the review will also consider whether the methods are also applicable for additional commodity types including mined resources and fisheries. The types of impacts this review will consider include biodiversity loss, greenhouse gas emissions, land use, water stress, eutrophication potential and acidification potential. This was a time limited review and is therefore not exhaustive but can be used to indicate which methods have been used previously and guide method selection for a given interest.

2 Intercomparison methods

2.1 Background and considerations

Considerations for a suitable comparison method include:

1. Data availability and quality.
2. Level of expertise required for calculation.
3. The impacts that can be compared using the method.
4. The commodities that can be compared using the method.
5. The production countries that can be compared using the method.

For a judgement to be made about which methods may be useful to enable a comparison of production impacts for a given commodity between countries, suitable criteria need to be considered and established. Once the components of a suitable method have been established it is possible to assess methods against this ideal solution. With the variety of potential methods available, it is likely that of the methods that could be applied for the case in question, some will not be suitable. However, these methods are likely to have another use case to which they are more suited. The following section will introduce considerations when identifying a method for the current use case, which is to enable individuals involved in policy decision making and research to make a comparison of the environmental impacts associated with producing a given commodity in each domestic production system versus in a specific overseas country production system as a case study analysis.

There are various factors to consider when determining what a useful method would look like, the first of which being the data required and its availability and quality. Some methods may require primary data collection, which may enable more accurate measurements and calculations, and guarantee a level of data quality and suitable data for the analysis, however, this would also be more costly and less practical for a quick comparison. Therefore, methods which can use existing data would be more suitable for the current purpose, noting that the quality and source of the data needs to be considered. Another key point is the ease with which the method can be applied by the practitioners in question. In this case it would be useful for individuals involved in policy decision making and research to be able to conduct a case study comparison without needing a technical expert to apply the method or calculation.

The impacts that can be compared vary between methods and it is essential to consider which impact types need to be investigated and ensure the selected method is suitable for comparing these. For example, of two studies investigating the impacts associated with cattle production in South America, one considered deforestation and land cover (Seidl *et al*. 2001), while the other looked at land management, agricultural inputs, and soil health (Ferguson *et al.* 2013). This raises the importance of using the same or comparable methods to compare the impacts of two different production systems for the same commodity. Additionally, it is useful to be able to look at a variety of impact types within the same analysis to increase the breadth of its relevance. Therefore, a suitable method would allow the comparison of as many key impacts as possible, including biodiversity loss, greenhouse gas emissions, land use, water stress, eutrophication potential and acidification potential (Pelletier *et al.* 2010; Alvarenga *et al*. 2012; Clark *et al.* 2022).

For the current research, a method is required to enable a case study comparison between production systems in different countries for a given commodity (or set of substitutable commodities). Some methods may be more suited to certain types of commodities, and some may be limited in terms of the commodities they can be applied to. For a single method to be recommended, it would be most practical for this method to be suitable for application to a wide range of commodities, enabling a comparison of whichever commodity is of relevance with the same method. However, it may also be possible to recommend different methods for comparisons of different types of commodities if they meet the other specified requirements. The same applies to the countries that the method could compare production systems between.

The most widely used methods identified from the research include footprint methods, which can be applied to specific commodities, and Life Cycle Assessment (LCA) methods.

2.2 Recommended comparison method: Life Cycle Assessments (LCAs)

Various comparison methods were reviewed considering the assessment criteria described in the previous section. The result of this was the recommendation of LCAs. This section will outline and discuss LCA usage and how these methods meet the assessment criteria. Other methods assessed will be discussed in following sections.

LCAs are widely used to calculate the environmental impacts resulting from the production of a good or the provision of a service (Basset-Mens *et al.* 2022).

2.2.1 ISO standards and LCA stages

The LCA method is specified in the International Organisation for Standardisation (ISO) 14040 and 14044 standards (ISO 2006a, 2006b). These are the basis for all additional standards and guidelines. The four LCA stages are defined in these standards:

1. **Goal and scope definition**
    a. The level of detail and system boundaries of an LCA vary based on the subject and goal of the LCA.
    b. Cradle to (farm) gate approach considers all activities from extraction of input materials to the point of the product leaving the factory/farm gate. Cradle to grave approach additionally covers impacts linked to product use and end of life (Circular Ecology 2023).

2. **Life cycle inventory analysis (LCI)**
    a. An inventory of input/output data for the studied system. Data are collected based on the LCA goals.

3. **Life cycle impact assessment (LCIA)**
    a. Additional information is included to supplement the LCI to aid understanding of the environmental significance of the impacts. This section can be further broken down into the following steps (Mu *et al*. 2020):
        i. Selection of impact categories – these come under three main areas, ecosystem impacts (e.g. climate change, eutrophication), human impacts (e.g. ozone depletion, carcinogens) and resource depletion (e.g. fossil fuels, water).
        ii. Classification – assigning the LCI results to appropriate impact categories.
        iii. Characterisation – calculation of the impacts in each category.

iv. Normalisation – optional step to relate the impacts to reference conditions, including weighting impact categories based on judgement of importance if a single score is being calculated.

4. **Interpretation**
   a. LCI/LCIA results are summarised and discussed in the context of the goals and scope to inform conclusions, recommendations, and decisions.

2.2.2 LCA usage

LCAs can be used to compare the impacts from different production systems for a given commodity and identify the key drivers from each system. For example, Pelletier *et al.* (2010) compared conventional to deep-bedded niche swine production systems in the upper Midwestern US (deep-bedded niche is a smaller scale rearing method with guidelines described by Honeyman, 2005). They used ISO compliant LCA to quantify cradle-to-farm gate flows of resource and energy inputs, outputs and resulting waste. Conventional commodity systems were found to be more efficient in general, and the two production systems had different drivers of impacts. Feed production was key in both, but more important in niche systems. Management of liquid manure in conventional systems strongly influenced greenhouse gas emissions, whereas in deep bedded niche, eutrophication potential was increased due to the outdoor storage of solid manure. This example demonstrates how LCAs can be applied to examine how the environmental impacts vary and overlap between comparable production systems.

One of the potential limitations with using LCAs to compare the impacts from producing a commodity in different regions is the methodological differences between different LCA types. Poore & Nemecek (2018) attempted to overcome this by developing a global database consolidating data from 570 studies which were assessed to ensure standardised methods. The dataset included inputs and outputs from each study and covered 38,700 farms across 119 countries and included 40 products which represent 90% of protein and calories consumed globally. Five impact indicators were covered: land use, freshwater withdrawals weighted by local water scarcity, greenhouse gas, acidifying and eutrophying emissions. These were recorded at each supply chain stage, though greenhouse gas emissions at the farm stage were disaggregated into 20 sources. The authors found that impacts from the production of a given commodity varied greatly (up to 50-fold) between producers. As this method enabled impacts to be quantified at each stage in the supply chain, it would allow producers to identify potential ways to reduce impacts. Despite the great variation, the lowest impact animal products were associated with greater impacts than vegetable substitutes, demonstrating how LCAs can be used to identify differences in impacts across different commodities as well as across production systems for a given commodity.

Another consideration when using LCAs for this type of comparison is that current LCA resources are generally tailored to temperate regions in developed economies (Basset-Mens *et al.* 2022). This is partly because within the LCIA step, the characterisation methods for different impact categories may not be relevant for tropical production systems as they have been specified with western environmental conditions and problems as their basis (Eshun, Potting & Leemans 2011). This may then pose a problem with applying these methods to developing, tropical countries to enable a comparison with a developed, temperate producer country such as those within the UK. However, the demand for LCAs of agricultural commodities in developing regions is increasing, fuelled by export and local markets (Basset-Mens *et al.* 2022). 'Life Cycle Assessment of agri-food systems: An operational guide dedicated to developing and emerging economies' identifies key considerations when applying LCA methods in these contexts and provides best practice guidance with a focus on a fieldwork approach (Basset-Mens *et al.* 2022).

Although LCA analyses can be seen as a thorough and in-depth assessment of production impacts, it is also possible that some key impacts and considerations are not necessarily considered. WWF highlighted the importance of acknowledging the limitations of LCA when using the method to guide decision making (Stevenson 2016; WWF 2016). This was demonstrated with the example of a forestry site which is poorly managed, dominated by tree stumps, deep ruts in the ground, streams brown with soil runoff, the removal of bird nests and fences to keep out the local community. The alternative may be a well-managed site, with wildlife habitats and connectivity, logging limited to an appropriate quantity and location and the engagement of the local community with the forest management. WWF (2016) argue that LCA methods may not capture much of the damage described in the poorly managed scenario, nor some of the positive, beneficial aspects of the well managed scenario, due to limits in LCA methods and their flexible boundaries. For the forestry example, they recommend combining LCA with the Forest Stewardship Council (FSC) certification for responsible forest management. This was developed by WWF and others to determine the sustainability of forest products using performance-based indicators to assess environmental and social impacts at the extraction location (Stevenson 2016). They argue that these two assessment types should also be applied in tandem to other supply chains such as agriculture, aquaculture, and fisheries. An alternative recommendation is to supplement LCAs with analysis using other models to generate more robust assessments. Fan *et al.* (2022) describe factors that are not considered within LCAs such as economic and societal concerns which shape the agricultural system, proposing that these factors should be modelled (for example using agent-based modelling (ABM) or emergy analysis (EmA)) alongside LCAs to overcome these current LCA shortfalls.

The boundaries of LCA studies need to be carefully considered to ensure that reliable conclusions can be drawn from the study. For example, when considering crops that are grown on land that was historically deforested to allow for their cultivation, it is important that the LCA incorporates these deforestation impacts to provide a full and realistic assessment of the impacts resulting from this production system. If the LCA only looks at the current snapshot of time when assessing such a production system, ongoing impacts from historic land use changes will not be considered, and the study will under report the impacts. If historic land change isn't considered, countries which have previously deforested to grow crops will unfairly benefit by LCAs not reflecting this historic damaging activity. This will make it challenging to use LCAs to assess the real impact of products from these countries. Cradle to gate or grave assessments should incorporate this historic land use information, but it is possible to omit these data when completing the analysis, so when referring to existing LCAs this would need to be confirmed by the LCA user. This is one example of potential data misuse in LCAs that users should be aware of. This highlights the need for guidance when interpreting LCAs and involving an expert could reduce the use of unreliable LCAs in comparison studies.

When comparing commodities consumed in the UK and produced overseas or domestically it is important to consider the impacts of transportation. The distance that commodities are transported from production site to consumer country is often marketed as an area where the public can reduce associated impacts by reducing 'food miles' and buying local. However, a review of LCA studies for US food commodities concluded that transportation accounts for a relatively small proportion of the impacts associated with most food commodities when compared to production (Heller 2017). A USDA study found that transportation (not including consumer transport to shops) accounted for 4% of energy use in the food system in 2002 (Canning *et al.* 2010). Another study found food distribution accounted for 4% of GHG emissions, with an additional 7% linked to indirect transport such as delivery of inputs to farms whereas food production was found to account for 83% of the total GHG emissions (Weber & Matthews 2008). The review also concluded that the mode of transport used can have a greater influence on the impacts than the distance travelled, and that consumer shopping journeys can account for a surprising proportion of impacts in food commodity cradle to grave LCAs (Heller 2017).

This demonstrates that although impacts can be reduced by reducing the transportation associated with a commodity, the impacts from the distribution of food from the production site to the consumer location are often relatively small compared to impacts from production. There are however exceptions to this, demonstrated by Meisterling *et al.* (2009) when they compared the global warming potential of conventional and organic wheat in the US. They found that the global warming potential of a loaf of organic bread was 30 g CO_2 less than the conventional, but if the organic loaf was shipped 420 km further, this would negate the difference in impacts. Therefore, though a greater proportion of the impacts associated with commodities may come from the production stage than the transportation stage, if goods are transported further, it has the potential to negate the reduction in impacts from choosing a more sustainable production system.

2.2.2.1 Hestia

The Hestia database was developed as an expansion of the multi-indicator global database developed by Poore & Nemecek (2018; Hestia 2023). The database includes data from 286 peer reviewed studies and reports and is open access, available to download through the API on the website. The agri-environmental data are provided in a standardised format and contain data from farms, farm surveys, field experiments and LCAs. The platform also contains models using LCA methods to enable users to calculate the impacts of farms or foods.

The readily accessible, standardised data from Hestia increases the potential for studies investigating the interaction of commodity production impacts with other factors. Clark *et al.* (2022) used Hestia to estimate impacts for manufactured food products, inferring their composition from ingredients lists, for four indicators: greenhouse gas emissions, land use, water stress and eutrophication potential. These impact data were paired with a nutrition measure (Nutriscore) to reveal that generally more nutritious foods tend to be more sustainable. Clark *et al.* (2022) noted that consumers want to have impact information available to make informed decisions on sustainable food choices, and therefore a method such as this is required which can be applied across food products to enable comparisons.

However, it is important to apply some additional context when using these aggregated data and ensure the time when and location where it was reported are representative of the current situation. For example, following high deforestation rates in Australia, legislation was introduced in Queensland and New South Wales in the 2000s that effectively tackled this (Evans 2016). However, policy changes in the mid-2010s enabled the resumption of deforestation activities, resulting in a significant increase in deforestation (33% more in 2015-16 in Queensland than the previous year, Slezak 2018). Following this, if the database was still using LCA data from before the mid-2010s policy change, it would not be representative of the increased deforestation impacts. Clearly this is an issue that can also arise when comparing existing LCAs, so the date of LCAs and the context at the time need to be considered in all studies and comparisons.

2.2.3 Biodiversity in LCAs

One key impact that needs to be assessed when comparing agricultural commodities is biodiversity. Whilst biodiversity remains more challenging than many other impact types due to its complexity, over the last 15 years, LCAs have been used extensively to measure the impacts of agriculture upon biodiversity (Fan *et al.* 2022). One example of a study that used two evaluation methods to compare biodiversity loss for wood production in north and south Finland is Myllyviita *et al.* (2019). They found that the species richness index showed minimal difference between the production systems, whereas the ecosystem indicators approach revealed a higher impact in the north. The authors concluded the lack of coherence was in part due to the species richness index considering all species, whereas the ecosystem indicators only consider endangered species. They highlighted that biodiversity LCAs are highly sensitive to reference state, model and data, and the variance of these factors between methods can impact their coherence.

2.2.4 Considerations for a suitable comparison method

2.2.4.1 Data availability and quality

Due to the increasing use of LCA over the last decade, data from producers across the world on inputs, outputs and production practices have been collected and have become more widely available (Poore & Nemecek 2018). LCA studies must follow ISO standards (ISO 2006a) which ensure that data of a sufficient quality are required to make an assessment.

2.2.4.2 Level of expertise required for calculation

Although calculating an LCA from scratch can be a complex task, there are alternative solutions which do not require a high level of expertise. Many LCA studies have been published, making it possible to compare existing results. These can be accessed individually or through a centralised database such as Hestia (Hestia 2023). Additionally, user friendly LCA software, such as openLCA (GreenDelta 2022) and SimaPro (SimaPro 2023), can be used with access to a vast number of datasets to enable non-experts to conduct LCAs with existing data.

2.2.4.3 The impacts that can be compared using the method

A wide variety of impacts can be assessed using LCAs, and different methods can be selected based on the impacts of interest. There has been ongoing work to enable biodiversity impact assessment and various metrics have been developed to enable this (Fan *et al.* 2022).

2.2.4.4 The commodities that can be compared using the method

Any commodities can be assessed using an LCA. However, if using existing LCA analyses to compare production systems, it is important to make sure that the same or comparable method has been used, as many variations on LCA methods exist. A potential solution to this could be to use a standardised database of existing LCA analyses (Poore & Nemecek 2018; Hestia 2023).

2.2.4.5 The production countries that can be compared using the method

The method can be applied to any countries where data are available. It is important to note that concerns have been raised about applying Eurocentric LCA methods to production systems in developing countries, so where possible guidance should be followed to ensure comparable results (Basset-Mens *et al.* 2022).

2.3 Other comparison methods

During the literature review, a range of other methods were identified that could be used to compare commodity production systems. These include the Ecological Footprint, carbon footprinting, Cumulative Energy Demand and the Farm Energy Analysis Tool (FEAT). Whilst useful for the purposes for which they were designed, these did not as fully meet the criteria set out above.

The Ecological Footprint method was originally developed by Wackernagel and Rees in 1990 to assess the sustainability and impacts of regions and countries (Wackernagel & Rees 1996; Ecological Footprint 2023). The Ecological Claim was developed at the Netherlands Environment Assessment Agency as an alternative to the original method (Rood *et al.* 2004). The Ecological Footprint method can also be applied to specific commodities to enable a comparison (Alvarenga *et al*. 2012). However, previous research has concluded that the Ecological Footprint method may not be suitable for measuring and comparing the impacts of agricultural commodities, as some important impacts are neglected from this analysis, such as eutrophication and acidification (Alvarenga *et al*. 2012). Additionally for the purpose of this review, the method is not suitable for comparing impacts of the production of a specific commodity in different countries as it does not allow for comparative advantages and specialisation and can be seen as biased against trade (van den Bergh & Verbruggen 1999).

The term 'carbon footprint' has been widely used to quantify the greenhouse gas emissions, and therefore impact on climate change, associated with the production of a commodity or other human activity or organisation (Yan *et al.* 2015; Climate Change & the Carbon Footprint 2019). The direct and indirect greenhouse gas emissions associated with each stage of a process can be assessed using a LCA to calculate the carbon footprint (Wiedmann & Minx 2008; Yan *et al.* 2015). However, as Wiedmann and Minx (2008) point out, the term carbon footprint has been used to describe various approaches including more basic online calculators. Generally, the carbon footprint method results in a figure of CO_2 equivalent and can be applied to specific commodity production systems to compare the impacts between producer countries. For example, Yan *et al.* (2015) used the carbon footprint method to compare the impacts from two rice production systems in China. However, as this method only considers greenhouse gas emissions and does not allow for an assessment of other key impacts such as biodiversity loss, it is not recommended for the proposed purpose.

Another potential method that could be used to compare the sustainability of production systems is environmental certification. The production systems meeting these voluntary certification standards (examples include Organic, Fairtrade, Roundtable on Sustainable Palm Oil (RSPO), and Rainforest Alliance) should in theory result in reduced greenhouse gas emissions, improved soil health, and socio-economic benefits (Jena *et al*. 2022). When looking at two comparable production systems, in theory it would be possible to conclude that the one with the environmental certification is more sustainable with less impacts. However, it is also possible that the other system meets the required standards for certification but has not been assessed or applied for certification. Therefore, making comparisons using certifications is binary and does not allow for a thorough comparison of impacts. Additionally, certification options are market based and involve costs for producers to become certified, and there is mixed evidence around improved socio-economic conditions for producers who do buy in (Jena *et al*. 2022). Therefore, the reduced impacts from certification may not be guaranteed for all impact areas, making comparisons less viable. For further discussion of this, see Section 2 of the previous JNCC report, Harris *et al.* (2019).

Additional methods identified which were not deemed suitable for the current purpose include greenhouse gas specific LCA methods such as Cumulative Energy Demand (Huijbregts *et al.* 2010), and the Farm Energy Analysis Tool (FEAT) which is also greenhouse gas and energy specific (FEAT 2011; Camargo *et al*. 2013).

It appears that greenhouse gas emissions have been more extensively studied in the context of commodity footprinting than other types of environmental impact. This suggests a need for further research to gain a greater understanding of impacts such as biodiversity loss, especially given the growing recognition that we are facing a joint biodiversity and climate crisis.

3 Conclusions

A variety of methods exist which could be used to compare the production impacts of a commodity across production systems and countries. Most of these can be broadly split into footprint methods, such as the ecological and carbon footprint, which can be applied at an individual commodity scale, and LCAs. There are limitations with the footprint methods in terms of the impacts they assess and their application to comparable production systems in different countries. LCAs can be completed for many commodity production systems with existing data, and with data collection the method can be applied to potentially any commodity. By using existing LCA studies or LCA software, production systems in international and domestic locations can be compared with relative ease. There are some concerns around using LCA methods for tropical production systems, though emerging methods and guidelines are addressing this. When comparing LCAs it is important to consider the study boundaries to ensure the assessments are representative and account for the key impacts associated with the production systems. Data misuse is a potential issue when using existing LCA studies, highlighting the need to interrogate the data and if possible, involve an expert. An applied case study example of using a LCA to compare substitutable commodities can be found in Appendix 1.

References

Alcock, T.E., Salt, D.E., Wilson, P. & Ramsden, S.J. 2022. More Sustainable Vegetable Oil: Balancing Productivity with Carbon Storage Opportunities. *Science of The Total Environment*, **829** (154539). Available at: https://doi.org/10.1016/j.scitotenv.2022.154539.

de Alvarenga, R.A.F., da Silva Júnior, V.P. & Soares, S.R. 2012. Comparison of the ecological footprint and a life cycle impact assessment method for a case study on Brazilian broiler feed production. *Journal of Cleaner Production*, **28**, 25–32. Available at: https://doi.org/10.1016/j.jclepro.2011.06.023.

Basset-Mens, C., Avadi, A., Bessou, C., Acosta-Alba, I., Biard, Y. & Payen, S. 2022. *Life Cycle Assessment of agri-food systems*. éditions Quae. Available at: https://doi.org/10.35690/978-2-7592-3467-7.

van den Bergh, J.C.J.M. & Verbruggen, H. 1999. Spatial sustainability, trade and indicators: an evaluation of the "ecological footprint". *Ecological Economics*, **29** (1), 61–72. Available at: https://doi.org/10.1016/S0921-8009(99)00032-4.

Camargo, G.G.T., Ryan, M.R. & Richard, T.L. 2013. Energy Use and Greenhouse Gas Emissions from Crop Production Using the Farm Energy Analysis Tool. *BioScience*, **63** (4), 263–273. Available at: https://doi.org/10.1525/bio.2013.63.4.6.

Canning, P., Ainsley, C., Huang, S., Polenske, K.R. & Waters, A. 2010. Energy Use in the U.S. Food System. Available at: https://www.ers.usda.gov/webdocs/publications/46375/8144_err94_1_.pdf?v=0

Circular Ecology. 2023. Environmental Glossary of Terms – LCA & Footprinting, Circular Ecology. Available at: https://circularecology.com/glossary-of-terms-and-definitions.html (Accessed: 8 June 2023).

Clark, M., Springmann, M., Rayner, M. & Harrington, R.A. 2022. Estimating the environmental impacts of 57,000 food products. *Proceedings of the National Academy of Sciences*, **119** (33), e2120584119. Available at: https://doi.org/10.1073/pnas.2120584119.

Climate Change & the Carbon Footprint. 2019. *Global Footprint Network*. Available at: https://www.footprintnetwork.org/our-work/climate-change/ (Accessed: 22 February 2023).

Collado-Ruiz, D. & Ostad-Ahmad-Ghorabi, H. 2010. Comparing LCA Results out of Competing Products: Developing Reference Ranges from a Product Family Approach. *Journal of Cleaner Production*, **18** (4), 355–64. Available at: https://doi.org/10.1016/j.jclepro.2009.11.003.

Croft, S., West, C., Harris, M., Green, J., Molotoks, A., Harris, V. & Way, L. 2022. Technical documentation for an experimental statistic estimating the global environmental impacts of consumption: 2022 version. *JNCC Report 715*, JNCC, Peterborough, ISSN 0963-8091. https://hub.jncc.gov.uk/assets/91efc19d-f675-426f-9333-ed0195cc729d

Damiani, M., Sinkko, T., Caldeira, C., Tosches, D., Robuchon, M. & Sala, S. 2023. Critical Review of Methods and Models for Biodiversity Impact Assessment and Their Applicability in the LCA Context. *Environmental Impact Assessment Review*, **101**, 107134. Available at: https://doi.org/10.1016/j.eiar.2023.107134.

De Rosa, M. & Schmidt, J. 2022. Life Cycle Assessment of Palm Oil at United Plantations Berhad 2022, Results for 2004-2021. Summary report. United Plantations Berhad, Teluk Intan, Malaysia. Available at: https://lca-net.com/files/Life-Cycle-Assessment-of-Palm-Oil-at-United-Plantations-Berhad-2022.pdf.

Ecological Footprint. 2023. *Global Footprint Network*. Available at: https://www.footprintnetwork.org/our-work/ecological-footprint/ (Accessed: 16 February 2023).

Eshun, J.F., Potting, J. & Leemans, R. 2011. LCA of the timber sector in Ghana: preliminary life cycle impact assessment (LCIA). *The International Journal of Life Cycle Assessment*, **16** (7), 625–638. Available at: https://doi.org/10.1007/s11367-011-0307-5.

Evans, M. 2016. Deforestation in Australia: Drivers, trends and policy responses. *Pacific Conservation Biology*, 22. Available at: https://doi.org/10.1071/PC15052.

Fan, J., Liu, C., Xie, J., Han, L., Zhang, C., Guo, D., Niu, J., Jin, H. & McConkey B.G. 2022. Life Cycle Assessment on Agricultural Production: A Mini Review on Methodology, Application, and Challenges. *International Journal of Environmental Research and Public Health*, **19** (16), 9817. Available at: https://doi.org/10.3390/ijerph19169817.

FEAT. 2011. Available at: https://www.ecologicalmodels.psu.edu/agroecology/feat/ (Accessed: 5 April 2023).

Ferguson, B.G., Diemont, S.A.W., Alfaro-Arguello, R., Martin, J.F., Nahed-Toral, J., Alvarez-Solis, D. & Pinto-Ruiz, R. 2013. Sustainability of holistic and conventional cattle ranching in the seasonally dry tropics of Chiapas, Mexico. *Agricultural Systems*, **120**, 38–48. Available at: https://doi.org/10.1016/j.agsy.2013.05.005.

Giam, X. 2017. Global Biodiversity Loss from Tropical Deforestation. *Proceedings of the National Academy of Sciences*, **114** (23), 5775–77. Available at: https://doi.org/10.1073/pnas.1706264114.

GreenDelta. 2022. openLCA is a free, professional Life Cycle Assessment (LCA) and footprint software with a broad range of features and many available databases, created by GreenDelta since 2006. Available at: https://www.openlca.org/ (Accessed: 6 April 2023).

Harris, M., Hawker, J., Croft, S., Smith, M., Way, L., Williams, J., Wilkinson, S., Hobbs, E., Green, J., West, C. & Mortimer, D. 2019. Is the Proportion of Imports Certified as Being from Sustainable Sources an Effective Indicator of UK Environmental Impact Overseas? Contracted Report to Defra (Project No. BE0166). Available at: https://randd.defra.gov.uk/ProjectDetails?ProjectID=20329&FromSearch=Y&Publisher=1&SearchText=BE0166&SortString=ProjectCode&SortOrder=Asc&Paging=10#Description

Heller, M. 2017. Food Product Environmental Footprint Literature Summary: Food Transportation. Center for Sustainable Systems, University of Michigan, State of Oregon Department of Environmental Technology. Available at: https://www.oregon.gov/deq/FilterDocs/PEF-FoodTransportation-FullReport.pdf

Hestia. 2023. Available at: https://www.hestia.earth/ (Accessed: 21 February 2023).

Honeyman, M.S. 2005. Extensive bedded indoor and outdoor pig production systems in USA: current trends and effects on animal care and product quality. *Livestock Production Science*, **94** (1), 15–24. Available at: https://doi.org/10.1016/j.livprodsci.2004.11.029.

Huijbregts, M.A.J., Hellweg, S., Frischknecht, R., Hendricks, H.W.M., Hungerbuhler, K. & Hendricks, A.J. 2010 Cumulative Energy Demand As Predictor for the Environmental Burden of Commodity Production. *Environmental Science & Technology*, **44** (6), 2189–2196. Available at: https://doi.org/10.1021/es902870s.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A. & van Zelm, R. 2017. ReCiPe2016: A Harmonised Life Cycle Impact Assessment Method at Midpoint and Endpoint Level. *The International Journal of Life Cycle Assessment*, **22** (2), 138–47. Available at: https://doi.org/10.1007/s11367-016-1246-y.

ISO. 2006a. *ISO 14040 Environmental management – Life cycle assessment – Principles and framework*. The International Standards Organisation.

ISO. 2006b. *ISO 14044 Environmental management – Life cycle assessment – Requirements and guidelines*. The International Standards Organisation.

Jena, P.R., Lippe, R.S. & Stellmacher, T. 2022. Editorial: Sustainable certification standards: Environmental and social impacts. *Frontiers in Sustainable Food Systems*, **6**. Available at: https://www.frontiersin.org/articles/10.3389/fsufs.2022.922672 (Accessed: 6 July 2023).

Koh, L.P. & Wilcove, D.S. 2008. Is Oil Palm Agriculture Really Destroying Tropical Biodiversity?. *Conservation Letters*, **1** (2), 60–64. Available at: https://doi.org/10.1111/j.1755-263X.2008.00011.x.

Ludwig, F., Biemans, H., Jacobs, C., Supit, I., van Diepen, K. & Fawell, J. 2011. Water Use of Oil Crops: Current Water Use and Future Outlooks. *ILSI Europe Environment and Health Task Force*. Available at: https://library.wur.nl/WebQuery/wurpubs/fulltext/193177.

Manning, F.C., Kho, L.K., Hill, T.C., Cornulier, T. & Teh, Y.A. 2019. Carbon Emissions From Oil Palm Plantations on Peat Soil. *Frontiers in Forests and Global Change*, **2**. Available at: https://www.frontiersin.org/articles/10.3389/ffgc.2019.00037.

Mattila, T., Helin, T., Antikainen, R., Soimakallio, S., Pingoud, K. & Wassman, H. 2011. 'Land Use in Life Cycle Assessment. *The Finnish Environment*, **24**. Available at: https://core.ac.uk/download/pdf/14925706.pdf.

Meisterling, K., Samaras, C. & Schweizer, V. 2009. Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat. *Journal of Cleaner Production*, **17** (2), 222–230. Available at: https://doi.org/10.1016/j.jclepro.2008.04.009.

Michelsen, O. 2008. Assessment of Land Use Impact on Biodiversity. *The International Journal of Life Cycle Assessment*, **13** (1), 22–31. Available at: https://doi.org/10.1065/lca2007.04.316.

Mohammad, S., Baidurah, S., Kobayashi, T., Ismail, N. & Leh, C.P. 2021. Palm Oil Mill Effluent Treatment Processes – A Review'. *Processes*, **9** (5), 739. Available at: https://doi.org/10.3390/pr9050739.

Mu, D., Xin, C. & Zhou, W. 2020. Chapter 18. Life Cycle Assessment and Techno-Economic Analysis of Algal Biofuel Production', in A. Yousuf (ed.), *Microalgae Cultivation for Biofuels Production*. Academic Press, pp. 281–292. Available at: https://doi.org/10.1016/B978-0-12-817536-1.00018-7.

Myllyviita, T., Sironen, S., Saikku, L., Holma, A., Leskinen, P. & Palme, U. 2019. Assessing biodiversity impacts in life cycle assessment framework - Comparing approaches based on species richness and ecosystem indicators in the case of Finnish boreal forests. *Journal of Cleaner Production*, **236**, 117641. Available at: https://doi.org/10.1016/j.jclepro.2019.117641.

Pelletier, N., Lammers, P., Stender, D. & Pirog, R. 2010. Life cycle assessment of high- and low-profitability commodity and deep-bedded niche swine production systems in the Upper Midwestern United States. *Agricultural Systems*, **103** (9), 599–608. Available at: https://doi.org/10.1016/j.agsy.2010.07.001.

Pendrill, F., Persson, U.M., Godar, J. & Kastner, T. 2019. Deforestation Displaced: Trade in Forest-Risk Commodities and the Prospects for a Global Forest Transition. *Environmental Research Letters*, **14** (5), 55003. Available at: https://doi.org/10.1088/1748-9326/ab0d41.

Poore, J. & Nemecek, T. 2018. Reducing food's environmental impacts through producers and consumers. *Science*, **360** (6392), 987–992. Available at: https://doi.org/10.1126/science.aaq0216.

Raum, S. 2020. Land-Use Legacies of Twentieth-Century Forestry in the UK: A Perspective. *Landscape Ecology*, **35** (12), 2713–22. Available at: https://doi.org/10.1007/s10980-020-01126-1.

Reijnders, L. & Huijbregts, M.A.J. 2008. Biogenic Greenhouse Gas Emissions Linked to the Life Cycles of Biodiesel Derived from European Rapeseed and Brazilian Soybeans. *Journal of Cleaner Production*, **16** (18), 1943–48. Available at: https://doi.org/10.1016/j.jclepro.2008.01.012.

Rood, G.A., Willing, H.C., Nagelhout, D., ten Brink, B.J.E., Leewis, R.J. & Nijdam, D.S. 2004. *Tracking the influence of the Dutch on global biodiversity*. 500013005/2004. Bilthoven, The Netherlands: National Institute of Public Health and the Environment (in Dutch): National Institute of Public Health and the Environment (RIVM).

Saavedra-Rubio, K., Thonemann, N., Crenna, E., Lemoine, B., Caliandro, P. & Laurent, A. 2022. Stepwise Guidance for Data Collection in the Life Cycle Inventory (LCI) Phase: Building Technology-Related LCI Blocks. *Journal of Cleaner Production*, **366**, 132903. Available at: https://doi.org/10.1016/j.jclepro.2022.132903.

Saifuddin, N.M., Salman, B., Hussein, R. & Ong, M. 2017. Microwave Pyrolysis of Lignocellulosic Biomass – a Contribution to Power Africa. *Energy, Sustainability and Society*, **7**. Available at: https://doi.org/10.1186/s13705-017-0126-z.

Schmidt, J. 2007. Life Cycle Assessment of Rapeseed Oil and Palm Oil: Ph.D. Thesis, Part 3: Life Cycle Inventory of Rapeseed Oil and Palm Oil. Aalborg: Department of Planning and Development, Aalborg University. Available at: https://citeseerx.ist.psu.edu/document?repid=rep1&type=pdf&doi=872cd9a9228d0f24c44ebd5c5c0ae679b1b8b2d1.

Schmidt, J.H. 2008. Development of LCIA Characterisation Factors for Land Use Impacts on Biodiversity. *Journal of Cleaner Production*, **16** (18), 1929–42. Available at: https://doi.org/10.1016/j.jclepro.2008.01.004.

Schmidt, J.H. 2010. Comparative Life Cycle Assessment of Rapeseed Oil and Palm Oil. *The International Journal of Life Cycle Assessment*, **15** (2), 183–97. Available at: https://doi.org/10.1007/s11367-009-0142-0.

Schmidt, J.H. 2015. Life Cycle Assessment of Five Vegetable Oils. *Journal of Cleaner Production*, **87**, 130–38. Available at: https://doi.org/10.1016/j.jclepro.2014.10.011.

Schmidt, J.H., Weidema, B.P. & Brandão, M. 2015. A Framework for Modelling Indirect Land Use Changes in Life Cycle Assessment. *Journal of Cleaner Production*, **99**, 230–38. Available at: https://doi.org/10.1016/j.jclepro.2015.03.013.

Searchinger, T.D., Wirsenius, S., Beringer, T. & Dumas, P. 2018. Assessing the Efficiency of Changes in Land Use for Mitigating Climate Change. *Nature*, **564** (7735), 249–53. Available at: https://doi.org/10.1038/s41586-018-0757-z.

Seidl, A.F., Silva, J. dos S.V. de & Moraes, A.S. 2001. Cattle ranching and deforestation in the Brazilian Pantanal. *Ecological Economics*, **36** (3), 413–425. Available at: https://doi.org/10.1016/S0921-8009(00)00238-X.

SimaPro. 2023. *SimaPro LCA software for informed change-makers*, *SimaPro*. Available at: https://simapro.com/ (Accessed: 6 April 2023).

Slezak, M. 2018. “Global deforestation hotspot”: 3m hectares of Australian forest to be lost in 15 years. *The Guardian*, 4 March. Available at: https://www.theguardian.com/environment/2018/mar/05/global-deforestation-hotspot-3m-hectares-of-australian-forest-to-be-lost-in-15-years (Accessed: 24 October 2023).

Stevenson, M. 2016. *What’s Your Footprint? Use LCA and FSC to Find Out | Blog Posts | WWF*, *World Wildlife Fund*. Available at: https://www.worldwildlife.org/blogs/sustainability-works/posts/what-s-your-footprint-use-lca-and-fsc-to-find-out (Accessed: 22 February 2023).

Verones, F. *et al.* (2020) ‘LC-IMPACT: A Regionalized Life Cycle Damage Assessment Method’, *Journal of Industrial Ecology*, **24** (6), 1201–19. Available at: https://doi.org/10.1111/jiec.13018.

Wackernagel, M. & Rees, W.E. 1996. *Our ecological footprint: reducing human impact on the earth*. Gabriola Island, BC: New Society Publishers (New catalyst bioregional series). Available at: http://bvbr.bib-bvb.de:8991/F?func=service&doc_library=BVB01&doc_number=007624923&line_number=0001&func_code=DB_RECORDS&service_type=MEDIA (Accessed: 15 February 2023).

Weber, C.L. & Matthews, H.S. 2008. Food-miles and the relative climate impacts of food choices in the United States. *Environmental Science & Technology*, **42**(10), 3508–3513. Available at: https://doi.org/10.1021/es702969f.

Wiedmann, T. & Minx, J. 2008. A Definition of Carbon Footprint. *CC Pertsova, Ecological Economics Research Trends*, **2**, 55–65.

WWF. 2016. *Detailed Comparison of FSC and LCA as Sustainability Assessment Tools for Forest Products*. WWF. Available at: https://www.worldwildlife.org/publications/detailed-comparison-of-fsc-and-lca-as-sustainability-assessment-tools-for-forest-products (Accessed: 22 February 2023).

Yan, M., Luo, T., Bian, R., Cheng, K., Pan, G. & Rees, R. 2015. A comparative study on carbon footprint of rice production between household and aggregated farms from Jiangxi, China. *Environmental monitoring and assessment*, **187**, 4572. Available at: https://doi.org/10.1007/s10661-015-4572-9.

Zulkifli, H., Halimah, M., Chan, K.W., Choo, Y.M. & Mohd Basri, W. 2010. Life cycle assessment for oil palm fresh fruit bunch production from continued land use for oil palm planted on mineral soil (part 2). *Journal of Oil Palm Research, Malaysia*, **22**, 887–894.

Appendix 1. Literature review case study: Overseas Palm Oil versus Domestic Rapeseed Oil in the UK

Introduction

Given that LCAs have been recommended as the most appropriate methods to use when comparing two production systems, a case study was undertaken to demonstrate this for a selected commodity group. Rapeseed oil and palm oil are two prominent oils in the UK market that, despite having vastly dissimilar sources of production – largely domestic (rapeseed) and 100% imported (palm oil) – are considered substitutable commodities in use (Schmidt & Weidema 2008). Global trends have shown that the demand for vegetable oils has been on a steady rise and that it will continue to grow (Schmidt 2010). This has prompted initial evaluations of the alternative commodity supply solutions for vegetable oils to explore fresh solutions to reducing the environment impact of consumption. For example, Jannick Schmidt has consistently used LCAs (2015, 2010, 2007) to highlight the varied environmental impacts associated with the production systems of different commodities, namely, comparisons between vegetable oils.

When using LCAs to evaluate alternative commodity supply systems under different time and resource constraints, it is critical to assess the considerations for a suitable comparison method that are presented in Section 2.2.4. Generally, if you were to complete a full LCA from scratch it would firstly involve collecting primary data from each individual section of the production system and inputting it into LCA software. This is a necessary step when comparing commodities that have not been studied before. However, collecting primary data for each production system stage can be time-consuming and resource-intensive (Saavedra-Rubio *et al.* 2022). Utilising secondary databases to power LCAs such as Hestia (see Section 2.2.2) or using approaches that incorporate the analysis of published LCAs can be used in the instances of time and resource limitations, providing there is data availability. Hence, with these restraints considered, this review aims to provide a comparative analysis of secondary LCA literature to assess the differences in environmental impacts associated with overseas palm oil production and domestic rapeseed oil production, making conclusions and applications to the UK system.

In conducting a case study analysis of rapeseed and palm oil LCA literature, it was crucial to consider that there may be variations in methods used between assessments. One general practice was to ensure that selected studies followed the standards set out by the ISO14040/44 standards (ISO 2006a, 2006b) and followed a similar Goal and Scope, that were also aligned with the aims of the review. Additionally, before making any direct comparisons of impact statistics, it was essential to ensure that the functional unit remained consistent between compared studies (Collado-Ruiz & Ostad-Ahmad-Ghorabi, 2010; Alcock *et al.* 2022). This refers to the constant reference unit for the product that enables its performance to be quantified between each impact area (ISO 2006b). Schmidt's "Life cycle assessment of five vegetable oils" (2015) incorporated a comparison of European rapeseed oil and Indonesian/Malaysian palm oil production systems using a functional unit of "1 tonne of refined oil", thus, this was selected as the baseline for making comparisons to data within other assessments in this study.

Flow Chart: Cradle-to-Gate approach

To ensure that LCA results were comparable between studies, it was necessary to ensure that system boundaries remained constant. A side-by-side analysis was completed following a cradle to gate approach (see Section 2.2.1), separating the production systems into three main stages that are displayed in the flow charts in Figure 1 and 2: Agricultural (cultivation), Oil Mill and Oil Refinery. Additionally, due to time limitations, the results of LCAs that incorporated attributional modelling were used primarily, which exclude the impacts from system expansions. The following impact areas were used to quantify the differences in overall environmental performance: global warming, water stress, eutrophication and acidification potential, and biodiversity.

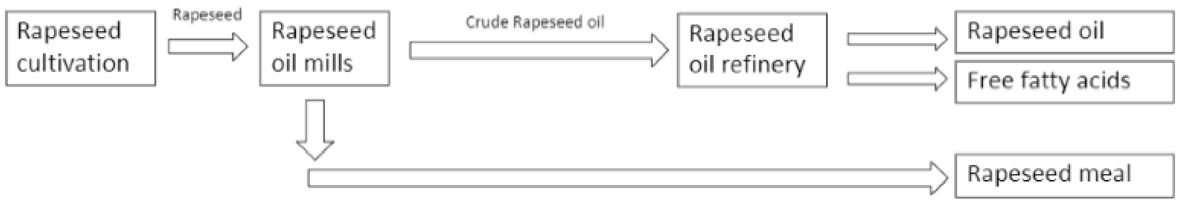

Figure 1. Rapeseed Oil production.

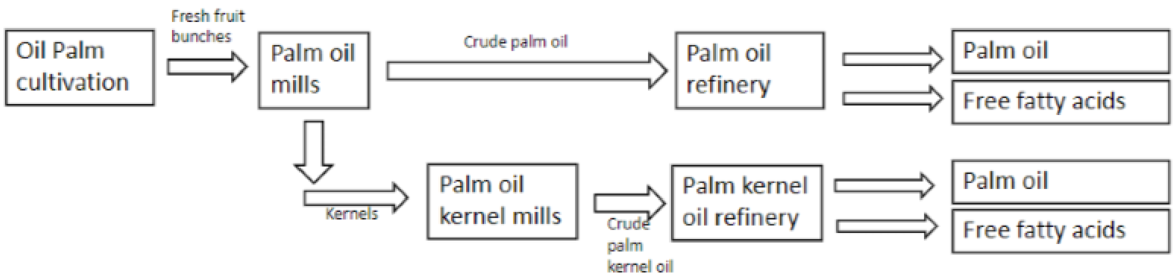

Figure 2. Palm Oil production. Produced incorporating the information from Schmidt (2015).

Land use: land occupation and land transformation

Traditionally, within LCAs, land use impacts have been separated into two areas: the associated impacts from cultivating land over a specified period (land occupation) and the impacts from converting land for agricultural use (land transformation). More recently, land transformation has been referred to as land use changes (Mattila *et al.* 2011). The impact areas associated with land occupation are clearly defined and commonly feature throughout LCAs. For example, greenhouse gas emissions (GHG), water use, biodiversity, and measures of toxicity. However, research has uncovered variations in the methods that are used to categorise and quantify the impacts from land transformation and land use changes. Typically, LCA studies that include impacts relating to land transformation focus on global warming (GHG equivalent) and biodiversity.

In Schmidt's (2010) comparative LCA of rapeseed and palm oil, impact areas were modelled using an "assumption allocation" method that estimated the impacts from transforming specific regional land types to agricultural land - which were adjusted to the functional unit of oil produced. The results in Table 1 assume that "set-aside land" was transformed to rapeseed crops in Denmark, whilst "secondary forests" were transformed to palm oil crops in Malaysia and Indonesia (Schmidt 2010). However, conclusions warn that these interpretations are heavily based on uncertainties and do not consider indirect impacts of land transformation, only direct impacts. Hence, a different method was used to model the impacts of land transformation in more recent LCAs. In Schmidt's LCA of five vegetable oils (2015) it is modelled using a measure for indirect Land Use Change (iLUC). The model aims to provide more of a comprehensive link between the "use of land and the effects on land use changes and intensification" (Schmidt 2012). Using multiple data frameworks, the measure incorporates the modelling of direct impacts of deforestation and intensification processes as well as indirect impacts from land use changes to accommodate new crops (Schmidt *et al.* 2015). The uncertainties relating to modelling iLUC are also less detrimental to interpretations, with results falling within acceptable ranges of uncertainty (Schmidt 2012). Despite these improvements, literature is consistent in stating the need for a consensus around a standardised method for quantifying both indirect and direct land use changes in LCAs. Nevertheless, current LCAs (Dalgaard *et al.* 2014; Schmidt 2015) are leaning towards the use of this framework for iLUC modelling, presented in Schmidt *et al.* (2015). The overall results for land use impacts within LCA studies have been summarised in Table 1 and although they follow different methods to quantify land use impacts, indicate similar findings.

Global warming: land transformation

The results in Table 1 show that there is a clear distinction between the impacts of land use in palm oil and rapeseed oil production systems. On average, to produce 1 tonne of refined rapeseed oil, the land mass needed for crop cultivation is much higher than to produce the same functional unit of refined palm oil. Therefore, palm oil production is more efficient in terms of relative productivity – producing a higher yield of refined oil over a much smaller land mass. This is reinforced in Zulkifli *et al.* (2010), stating that palm oil crops have the capacity to produce up to eight times more yield than any other oil crop over 1 hectare of land. Despite this, when compared with rapeseed oil production, palm oil production releases considerably higher volumes of GHG emissions during both land occupation and land transformation processes. The discrepancy in land transformation impacts have been linked to the different practices that occur prior to and during cultivation. Palm oil cultivation is commonly associated with large quantities of deforestation, resulting in high GHG emission and biodiversity loss (Koh & Wilcove 2008). Typically cultivated in tropical regions such as South-east Asia (Indonesia and Malaysia) and South America (Brazil), palm oil exports are partly responsible for drastic losses to carbon stocks through deforestation as native forests are cleared to meet growing demands (Pendrill *et al.* 2019). The removal of carbon rich biomass (often through burning or logging) causes huge volumes of GHGs to be released, as well as further disrupting processes that support the biodiversity of the ecosystem. Thus, the benefits of higher productivity are achieved at the expense of increased negative impacts from the transformation practices that are associated with palm oil production specifically.

Table 1. Land use impacts results from Schmidt (2015) and (2010) per 1 tonne of refined oil.

Study	**Land Mass (m^2/yr.)**		**Global warming from land occupation (kg CO_2)**		**Global warming from iLUC/ land transformation (kg CO_2)**		**Total (kg CO_2)**	
	Rapeseed oil	**Palm oil**	**Rapeseed oil**	**Palm oil**	**Rapeseed oil**	**Palm oil**	**Rapeseed oil**	**Palm oil**
Schmidt (2015)	2,352	1,712	-95	1,418	357	606	262	2,024
Schmidt (2010)	5,480	2,420	2,220	2,470	521	978	2,741	3,448

On the other hand, the production of rapeseed oil tends to be less associated with deforestation and other land transformation practices than palm oil. For example, the data in Table 1 indicates that for both studies, the equivalent GHG emissions associated with land transformation (Schmidt 2010) and iLUC (Schmidt 2015) are nearly half for rapeseed production compared to palm oil production. However, due to the uncertainties that are still present in quantifying land use in LCAs, differences in data can also be accounted for by additional factors. One factor that needs to be considered when comparing the impacts of land use changes from different production systems is the spatial and temporal differences in land types (Schmidt *et al.* 2015). Schmidt (2010) presents evidence that indicates that the transformation of primary and secondary forests which support palm oil cultivation will incur more severe land use impacts than the conversion of set-aside land for rapeseed crops. These characterised results prove that the variations in regional land types used for crop cultivation have a vital role in the global warming impacts from transforming the land. It is also worth noting that rapeseed crops in the UK may be more likely to be grown on land transformed from another crop rather than set-aside land.

Biodiversity

Although studies have demonstrated a comprehensive link between land transformation processes and negative impacts on biodiversity (Xingli 2017), many current LCA studies still lack the specific data required to accurately quantify this relationship (Damiani *et al.* 2023). Thus, within some LCA studies biodiversity impacts are not included, even though they are regarded as key to the comparison of different production systems (see Section 2.2.3). It is broadly recognised in the literature that habitat changes incurred by land use (both occupation and transformation processes) are now one of the main drivers of biodiversity loss (Souza *et al.* 2015). Despite this, there is no clear consensus for the most appropriate method to estimate this due to the complexities of the different aspects of biodiversity (Damiani *et al.* 2023). In recent years, a wide range of models have been developed to assist LCAs in evaluating biodiversity loss which mostly rely on ‘species richness’ as an indicator. For example, the Species-Area Relationship (SAR) calculates the loss in species according to land use and the Species Distribution Model (SDM) assesses impacts based on predicted changes in distribution due to observed land use patterns (Souza *et al.* 2015). However, these models are criticised for being too one-dimensional in measuring the diverse impacts on biodiversity from land use (Souza *et al.* 2015). To appropriately quantify the biodiversity impacts within LCAs, it has been suggested that the focus should be on expanding the depth of analysis used within multiple indicators (Souza *et al.* 2015), for example, enhancing the current species richness index by including population and landscape analyses. ReCiPe 2016 (Huijbregts *et al.* 2017) and LC-Impact (Verones *et al.* 2020) represent some of the most recent developments in biodiversity reporting within LCAs. However, even with these improvements, the diverse methods that are used throughout LCA literature hinder the ability to compare results effectively.

In Schmidt’s 2015 study, biodiversity impacts were not considered. However, in Schmidt (2010), it was quantified using wS100 (weighted species 100), a method developed to provide insights into biodiversity with application to variable land types (Schmidt 2008). This involves “multiplication of occupied area, the number of species affected per standard area (100 m^2), the duration of occupation and re-naturalisation from transformation, and a factor for ecosystem vulnerability” (Schmidt 2008). The factor for ecosystem vulnerability is based on data from a surrogate species of vascular plant, which some authors believe to be applicable to other species (Souza *et al.* 2015). However, this is also criticised by others, with warnings around the assumptions and unsuitability to some biodiversity measures (Michelsen 2008).

The overall results of the study indicate that the differences in biodiversity impacts between rapeseed and palm oil are not entirely conclusive (Schmidt 2010). This is accounted for by the varied uncertainties that are present in quantifying biodiversity impacts from land transformation, with the use of available data based heavily on assumptions. Consequently, these results are dealt with separately to the impacts from land occupation.

The results show that the impacts on biodiversity from land occupation are more significant from rapeseed oil than palm oil production. Schmidt (2010) accounts for this through differences in biodiversity capacity and vulnerability of the regional land that is used for oilseed crop cultivation in Denmark. Schmidt (2010) states that the occupation of the same amount of land in Denmark has a higher impact than in Brazil, as a smaller proportion of Denmark is naturally vegetated, therefore there is less available land to support natural species richness levels.

Looking solely at the required land for producing 1 tonne of refined oil set-aside land in Denmark and a combination of alang-alang grassland and degraded forest in Malaysia and Indonesia, the carbon and biodiversity impacts are as follows: 1 tonne of rapeseed oil would require 0.00751 hectares, resulting in 0.71 tonnes of CO_2 equivalent and 0.71 wS100, while 1 tonne of palm oil would require 0.00247 hectares, causing 0.50 tons of CO_2 equivalent and 0.30 wS100 (Schmidt 2010). These calculations suggest that rapeseed oil has a more significant impact compared to palm oil if converted from the same starting land use type. However, the results for palm oil are highly sensitive to the distribution between transformed alang-alang grassland and forest into oil palm. If degraded secondary forest (with no alang-alang land) were transformed into oil palm, 1 tonne of palm oil would cause 1.08 tons of CO_2 equivalent and 0.95 wS100. If the transformed forest were primary forest, the impact on biodiversity could be as high as 17 wS100.

However, it is not typically the case that land used for rapeseed oil production and land used for palm oil production are converted from the same starting land use type. Palm oil is more typically associated with land conversion than rapeseed oil, which is often grown on land that has been occupied for production systems for many years. As well as this, palm oil is typically grown in tropical areas which are often biodiversity hotspots, whereas rapeseed oil is often grown in areas with a lower starting biodiversity and therefore less to lose when conversion does take place. A more nuanced analysis that considers data based on both occupation and transformation in the cases of the specific production systems being compared and taking into account specific previous land use types (rather than a theoretical direct comparison) would therefore be needed to gain a full insight of effects on biodiversity.

Climate change

Land occupation

The contributions to climate change from the occupation of land in production systems of both oils have been quantified within several LCA studies and highlight significant differences between the two. Results indicate that of the production systems studied, overall, palm oil consistently releases higher volumes of GHG than rapeseed oil and therefore affects climate change more intensely (Schmidt 2010, 2015). This can be linked to some of the activities associated with land occupation, such as the agricultural and cultivation practices and processing methods in the oil mill and refinery stage, that are specific to palm oil production (Schmidt 2015).

Cultivation

Within both production systems, the agricultural stage (cultivation) produces the highest climate change impact. In the palm oil cultivation taking place in Malaysia and Indonesia that was studied in Schmidt (2015), it was recorded that per tonne of refined oil produced, emissions equivalent to 3,259 kg CO_2 were released, with 3,016 kg CO_2 from field emissions alone. The respective GHG emissions for 1 tonne of refined rapeseed oil produced during cultivation was 2,247 kg CO_2 with 976 kg CO_2 from field emissions (Schmidt, 2015: 135). These GHG emission data are based on only the agricultural stage of the LCA, without any adjustments made for emissions from other stages or biogenic CO_2 uptake from the cultivated oilseed plants, hence the data are different to the figures in Table 1.

The process of peatland decay in tropical forests has been recognised as a high source of CO_2 field emissions that are released in the cultivation of palm oil and that is often quantified within LCAs (Schmidt & Rosa 2019; Schmidt 2010, 2015). Undisturbed, the peatland in Malaysia and Indonesia provides a long-term carbon sink. However, the drainage processes that are used to prepare land for palm oil cultivation disrupts this and releases high volumes of CO_2 into the atmosphere (Manning *et al.* 2019). This process is also exacerbated by the removal of forest cover during deforestation and land transformation practices. Of the LCA studies found as part of this review that have quantified these data, all have referenced complexities and assumptions that have been made measuring these emissions. However, studies remain consistent in the accounting of high GHG figures to peat emissions in the agricultural stage (Schmidt 2010, 2015; Schmidt & De Rosa 2019).

Oil mill/refinery stage

Studies have also highlighted sizeable differences in the contribution to climate change from the oil mill stages of both production systems. For example, in Schmidt (2015) it was recorded that producing 1 tonne of refined oil releases emissions equivalent to 423 kg CO_2 in the oil mill stage of rapeseed oil production compared with 917 kg CO_2 for palm oil. This has mostly been accounted as the treatment of palm oil mill effluent (POME) which, if not captured, releases methane (CH_4) into the atmosphere (Schmidt 2010, 2015). POME is a by-product that is released in the form of solid oil and grease during the oil mill stage of palm oil production. Without treatment, it holds a toxic potential that poses a huge risk to the environmental health of living ecosystems (Mohammad *et al.* 2021). Schmidt's 2015 study estimated that in 2012 only 5% of oil mills in Malaysia and Indonesia treated POME using methane capture technology. Data in Alcock *et al.* (2022) estimates figures for 2022 in Malaysia to be 6% and in Indonesia to be 27%, hence this demonstrates that currently, it remains a prominent source of GHG emissions in the production of palm oil.

Beyond the gate

Although this case study focuses primarily on data from cradle to gate LCA studies, it is important to consider the environmental impacts that occur beyond the gate. An extension of this approach, cradle to grave LCAs (see Section 2.2.1), further explore the impacts that are linked to transportation, product use and disposal (Circular Ecology 2023). However, there is a lack of quantified evidence within current literature surrounding the LCA of this chosen commodity group. Despite this, these additional impact areas remain crucial to providing a comprehensive answer to which commodity production system is more environmentally preferred, particularly in the context of comparing domestic and international production systems. Schmidt (2007) touches on the huge discrepancies between the impacts from the transportation of rapeseed oil produced in Denmark and palm oil that was produced in Malaysia and Indonesia. For example, this specific study uses Amsterdam as a representative for Central Europe so that the transportation distances could be compared from production to consumption. Rapeseed oil was transported approximately 800 km by lorry compared with palm oil that was transported over 15,200 km by ocean tanker and lorry (Schmidt 2007). Within the time constraints of this study, no quantitative data were found on the environmental impacts of rapeseed and palm oil transportation, but it is important to recognise the affected impact areas, such as the contribution to climate change from fossil fuel combustion engines. Within cradle to cradle or cradle to grave LCAs, it is necessary to quantify these impacts.

Water consumption and water stress

Literature suggests that water consumption is low in the production systems of both oilseed crops assessed here, so may be a less relevant impact to include compared to its importance for other commodities. The results in Schmidt (2015) do not include a standalone value for blue water use during the cultivation of rapeseed oil. However, the results do indicate that there is an overall blue water saving, when considering the displacement of marginal crops (system expansion). Although this approach goes beyond the system boundaries that are defined for this case study, this is explained in a further section (Modelling system expansions). Within the scope of this review, no specific study data were found to quantify the impacts on water use from rapeseed oil production. However, there is a clear consensus within studies that water consumption is higher in rapeseed cultivation as irrigation is rarely used in palm oil cultivation (Ludwig *et al.* 2011). Despite this, Ludwig *et al.* (2011) states that as rapeseed oil crops are commonly grown in regions of Northern Europe such as Denmark, Ukraine and the UK, little irrigation is needed for cultivation due to the areas' low water scarcity (Ludwig *et al.* 2011).

On the other hand, the water consumption requirements for producing palm oil are less clear (Ludwig *et al.* 2011). Palm oil is grown predominantly in tropical climates with high humidity and sustained rainfall throughout the year, such as Indonesia and Malaysia (Ludwig *et al.* 2011). Although some studies are unclear over the use of irrigation in palm oil production, Schmidt's LCA (2015) results show crops are not irrigated, but with the results indicating a net consumption of water during cultivation. Despite this, further results have indicated that contributions from the oil mill and refinery stage are more responsible for blue water use within the production system of palm oil (Schmidt, 2015). Specifically, this is accredited to machinery production and water that is used in POME treatment. However, the overall blue water consumption value remains minimal, at 7.13 m^3. Equivalent estimates for water use in rapeseed oil mill and refinery stages were not found within the scope of this review but this is likely to be lower as no POME treatment is required.

Given the similar landscapes and climates of Denmark and the UK, it is expected that the results would be similar. Ludwig *et al.* (2011) also identifies that some UK regions (areas in the north of the UK) offer some of the most optimal growing conditions in Europe for producing rapeseed crops efficiently, and so may be useful to target for increases in rapeseed oil production.

Acidification and Eutrophication potential

Within the time constraints of this study, the assessment of acidification and eutrophication potential found existing LCAs methods in the literature to be limited in relation to the selected commodity group. Despite this, of the studies that have explored them, the comparative results are conclusive. For example, the results in Table 2 conclude that across both acidification and eutrophication impact areas, palm oil production is more environmentally favourable than rapeseed oil. The was calculated using the EDIP97 method in SimaPro (see Section 2.2.4), which equates to the impacts of producing 1 tonne of refined oil (Schmidt 2010). Further analysis in Schmidt (2010) also revealed that the major contributors to acidification and eutrophication potential were pesticides and heavy metals from fertilisers. Schmidt also indicates that more fertilisers are used in the overall cultivation of rapeseed, therefore meaning that the eutrophication and acidification potential is higher.

Table 2. Eutrophication, acidification results from Schmidt (2010) per 1 tonne of refined oil.

Impact area: Oilseed	Eutrophication (kg NO_3)	Acidification (kg SO_2)
Rapeseed oil	140,000	20.2
Palm oil	124,000	14.8

Modelling system expansions

System expansion is an approach that is used with LCAs to account for the impacts that are associated with displacement or implications on marginal production systems from the production of another commodity (Schmidt 2010). Although the concept is not explored within the system boundaries of this review, it is important to consider the potential implications of expansions in the overall decision-making process when comparing domestic and international production systems. For example, Schmidt's (2010) study explores the hypothetical impacts of different expansion scenarios for rapeseed oil (Denmark) and palm oil production (Malaysia and Indonesia). However, it is reliant on many assumptions. Regarding water use, the consideration of displaced marginal crops in the production of rapeseed oil resulted in a water saving, as opposed to net water consumption. An example of a similar consideration is that in the flow chart (Figure 1), it is highlighted that the production system of rapeseed oil results in multiple outputs, one of which is rapeseed meal. This is produced as a by-product during the oil mill stage and can be used for animal feed (Schmidt 2015). Hence, by producing rapeseed meal as a by-product of rapeseed oil, the need for cultivating and irrigating other crops (such as soybean or barley) to produce animal feed, is reduced, or eliminated. For example, the results of Schmidt's LCA of Danish rapeseed oil (2015) state that producing 1 tonne of refined oil is associated with a net blue water saving of 362 m^3, despite still being irrigated, if comparing against the alternative of producing both an oilseed and a separate animal feed crop. With less demand to produce meals from marginal crops, growing rapeseed can lead to potential water savings and reduce impacts on agricultural water consumption.

This study provides scope for further analysis, with progressions towards the exploration of different system expansion scenarios for the domestic production of rapeseed oil and overseas production of palm oil. The use of a consequential modelling approach for a future LCA, like the one used in Schmidt (2010), is beneficial in the incorporation of expansion scenarios. The aim of this study though, was to test the application of LCAs to a real-life commodity group and evaluate the findings that can be drawn from previous literature.

Limitations and uncertainties

Reliability

The analysis faces several limitations and uncertainties that impact its reliability. One significant limitation is the scarcity of available data found within the scope of this study that concern the rapeseed and palm oil production systems. Consequently, the analysis heavily relies on a limited pool of information, primarily derived from two studies that examine Danish rapeseed oil and Malaysian/Indonesian palm oil production (Schmidt 2010, 2015). Despite this, the landscape and climate similarities allow for the applicability of Danish rapeseed production results to UK rapeseed production systems and provide some comprehensive findings. To address these limitations, providing that time and resource constraints enable this, future studies would benefit from the collection of primary data.

Incorporating primary data would contribute to a more accurate and conclusive analysis, enhancing the overall reliability of the findings.

Another limiting factor that may impact the analysis of life cycle impact areas is the reliance on assumptions regarding specific indicators and models. For example, the current methods for quantifying biodiversity and iLUC within LCA are heavily based on assumptions.

Contextual considerations

Within Europe and the UK, deforestation already occurred many years ago (Susanne 2020), but this historical context is not considered within the current indicators used within LCAs. This limitation may result in an incomplete assessment of the environmental impacts in these regions in a more holistic sense. Some of the indicators that are used to quantify impact areas within LCAs could be considered more sensitive to commodity groups that are produced in developing countries. For example, palm oil production in Indonesia and Malaysia is situated on arable land that has been newly established from the deforestation and clearing of tropical forests, whereas rapeseed oil production tends to be cultivated in Europe on land that has already undergone transformation process or previous cultivation. Hence, the indicators used within LCAs fail to consider that although the current impacts of land transformation are low for specific commodity groups, there should still be contextual considerations for the historical data.

Conclusions

Overall, this case study concludes that for GHG emissions, rapeseed production tends to have lower overall impacts than palm oil production using a cradle-to-gate attributional LCA approach. Amongst both production systems, the cultivation stage provides the main hotspot for GHG emissions. Within LCA literature, this is consistently accounted for by the decay of the peatlands soil that palm oil crops occupy. Additionally for palm oil, high volumes of CH_4 are released during the current practices used for the treatment of POME in the oil mill stage, which contributes to the overall GHG impact area. With respect to the impacts associated with land use changes, studies have indicated that they are a high contributor to GHGs, especially during the deforestation of primary and secondary forests. However, the varied methods for quantifying these impacts on GHGs still remains full of assumptions and uncertainties. Despite not being included within a cradle-to-gate LCA assessment, the contribution to GHG emissions from transportation will need to be considered in the overall context of comparing domestic rapeseed oil against overseas palm oil production, as the huge disparities in distance will lead to a higher contribution from palm oil importation.

For the impact areas of acidification and eutrophication, literature is conclusive that palm oil production systems are more environmentally preferred to rapeseed oil. This is due to the high requirement for pesticide and fertiliser input in rapeseed oil cultivation to ensure that the ideal growing conditions are met. The use of pesticides and fertilisers have been linked to acidification and eutrophication potential due to the release of the toxic chemicals and heavy metals that they are composed of. However, many studies opt to not include these measures within impact assessments.

The conclusions for biodiversity and water use are less clear. There is ongoing debate around the most suitable methods for assessing impacts upon biodiversity and most studies use a species richness measure to quantify this impact. The findings for the comparison of biodiversity impacts between the two oil-seed crops were inconclusive, though land occupation from rapeseed oil in Denmark was found to have greater impacts on biodiversity than palm oil production in Brazil. The land transformation impacts rely heavily upon the habitat being converted for palm oil cultivation, though the conversion of primary or secondary forest for palm oil results in higher biodiversity impacts than the conversion of set-aside land for rapeseed oil. Studies have indicated that water use during the cultivation of rapeseed is higher than palm oil because of the contributions from irrigation methods. Despite not being irrigated, palm oil production has a higher water use during the oil mill stage due to machinery use and treatment of POME. Additionally, when considering the displacement of marginal crops in a system expansion scenario, rapeseed oil is associated with a net water saving.

The literature also puts huge emphasis on the complex nature of a concrete answer for which oil is more environmentally preferable, with some studies suggesting that solutions should focus more on improving the applied technology (Schmidt 2010) and the methods used to communicate LCA data. For example, developments around land use change modelling and muti-dimensional biodiversity indicators would enhance the accuracy of LCA reporting and minimise the reliance on assumptions.