Assessment of an LCIA method for evaluating biodiversity impact from food production

An LCA case study of pork

Master's thesis in Industrial Ecology

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Gothenburg, Sweden 2022
www.chalmers.se
Report No. E2022:048

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Abstract

The global loss of biodiversity is mainly driven by land use and land use change due to e.g., agriculture and food production. This study presents a case study of pork that assesses the applicability of the biodiversity life cycle impact assessment method (LCIA) developed by Chaudhary & Brooks (2018). The purpose of the study was to increase the understanding of the method in three aspects: (1) aspects of biodiversity captured in relation to food production, (2) the potential inclusion of land transformational impact, and (3) the spatial resolution. The outcome of the study was intended to provide valuable insights to the development of a database for biodiversity impact from food carried out by RISE (Research Institutes of Sweden), a database to be used for consumer communication purposes.

The case study consists of a life cycle assessment (LCA) of three pork production systems with different feed compositions. The functional unit was 1 kg of edible pork meat and covered production phases from cradle to farm-gate. Considerable weight was put on the inventory data collection for land occupation and land transformation flows, where modelling the feed composition and the feed crop cultivation was an essential part. Several methods to assess biodiversity impact in LCA are being developed, in this study one of the most promising methods was selected. What aspects of biodiversity the method capture in relation to food production, was assessed through mapping drivers of biodiversity loss and characteristics of biodiversity to illustrate what parts of these were covered. The evaluation of which spatial resolution of the method to be preferred and how to allocate the impacts from land transformation was assessed through applying different characterization factors to demonstrate the variation in result depending on methodological choices. In some respects, the studied method fails to cover the complexity of biodiversity loss, as it only assesses species loss and how it is impacted by land use interventions, but no other indicator or driver. Yet, the method was found relatively easy to apply and could be used when comparing products of similar character for indicating land use impacts on species. For cropland, three intensity levels are included. However, the method is not developed enough to show differences in a comparative LCA on crops cultivated under different agricultural systems (e.g., differences between organic and conventional).

When applying the higher spatial resolution provided by the method, ecoregional approach, the results differ from when applying a higher resolution using the country approach, although not by much. Since applying high spatial resolution was found to be more complicated and time consuming, one should consider focusing on other parameters that might have a larger influence, such as assessing a detailed feed composition. The method is not yet developed enough to include several agricultural land use types nor taxa groups (e.g., insects and microorganisms), two important factors potentially providing a larger difference in result between products of different character. Altogether, the lower spatial resolution might in many cases be good enough.

A harmonized way of applying the characterization factors for land transformation was not found. However, when included using the approach of this study, the impact from land transformation had a large contribution to the total biodiversity damage. Whether to include land transformation is suggested to be decided by if the food products under study contributes or have contributed to land use change, and not through system delimitations.

Altogether, despite highlighted drawbacks of the method, it serves a purpose by capturing potential differences in biodiversity damage between food products. One should however keep in mind that the method accounts for one only driver of biodiversity decline and one biodiversity indicator.

Keywords: *biodiversity*, *life cycle assessment*, *life cycle impact assessment*, *land use*, *land use change*

Acknowledgements

We would like to thank our supervisor Ulrika Palme who has helped us throughout our project. We would also like to thank her for including us in the land use research team, where we got invaluable insight and guidance. Thanks to all team members, and a special thanks also to Allison Perrigo and Martin Persson for being great sounding boards when in doubt.

Moreover, a huge thanks to the whole biodiversity team at RISE, and especially our supervisor Serina Ahlgren for holding our hand throughout this rollercoaster. We would also like to thank Britta Florén and Karin Morell at RISE for being the best cheerleading supporters.

Lastly, we would like to thank our family and friends for being there for us through thick and thin.

Viktor Lundmark Harrison and Maja Häggström
Gothenburg, June 2022

Abbreviations

BD	Biodiversity damage
CBD	Convention on biological diversity
DM	Dry matter
EBV	Essential biodiversity variables
EU	European union
FAOSTAT	Food and agriculture organization corporate statistical Database
FU	Functional unit
IBGE	Brazilian institute of geography and statistics
ILUC	Indirect land use change
IPBES	The intergovernmental science-Policy platform on biodiversity and ecosystem services
ISO	International Organization for Standardization
IUCN	International Union for Conservation of Nature and Natural Resources
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LU	Land use (also referred to as land occupation)
LUC	Land use change (also referred to as land transformation)
MAGyP	Ministerio de agricultura, ganadería y pesca
NUTS	Nomenclature of territorial units for statistics
PDF	Potentially disappeared fraction of species
PSL	Potential species-equivalents lost
RISE	Research institutes of Sweden
SAR	Species-area relationship
SCB	Statistiska centralbyrån/Statistics Sweden
UNEP-SETAC	United nations environment programme and society of Environmental toxicology and chemistry
VS	Vulnerability score
WWF	World wildlife fund

Contents

1 Introduction

Consumer communication is one of many reasons to study the environmental impact of food and has the potential of transforming consumption patterns to more sustainable choices (Chaudhary & Brooks, 2018). Since 2015, Research Institutes of Sweden (RISE) has provided a climate database with data on climate footprint of food products. The overall purpose of this database is to inform consumers about the carbon footprint of food, and it has mostly been used by restaurants, grocery retailers and for public meals served in for example schools and hospitals (RISE, n.d.-a). However, the environmental impact from food production is much more multifaceted than what could be represented through climate footprint. RISE has therefore started developing a database for biodiversity impact, a database aiming to be being used in a similar way as the climate database, and to broaden the communication of environmental footprint of food products (RISE, n.d.-b).

Life cycle assessment (LCA) is a common method to assess potential environmental impacts of products or processes. Within LCA, the method where inventory data (e.g., land use or emissions) is translated to an indicator of environmental impact is called life cycle impact assessment (LCIA). An environmental impact can be expressed in midpoint indicators, such as climate change, eutrophication, acidification or land use, or aggregated into an endpoint category of damage to the environment. LCA-studies and methodologies have been used for developing the climate database and will also be used for developing the biodiversity database.

Until present, climate change has been the most used metric to evaluate environmental impact of food, and the impact on biodiversity loss has been given much less attention (Willett et al., 2019). The main reason is that, compared to climate impact, it is more complex and there is yet no harmonized way of assessing the various pressures on biodiversity loss, such as land use/land use change, climate change and pollution. Moreover, many LCIA methods have been developed that quantifies different aspects of biodiversity, which also confirms the complexity of measuring the impact on biodiversity (see e.g., Gabel et al., (2016) for a review). The LCIA method that will be used for the RISE biodiversity database is developed by Chaudhary & Brooks (2018), and estimates global species-equivalents potentially lost following land occupation (also called land use, LU) and land transformation (also called land use change, LUC). This method will henceforth be referred to as the *CB method.*

1.1 Aim

The aim of the study is to assess the applicability of the CB method when being used for evaluating biodiversity impact from food production, by focusing on three essential areas for biodiversity indicators within LCA. The *first* focus area includes analysing how well the indicator acts as a proxy for biodiversity loss, including analysing what reference situation is applied. The *second* focus area covers an assessment of how to apply and include the impact from land transformation, and the *third* covers the choice of spatial resolution and how it affects the result. The intended purpose is to highlight important factors that demonstrate the usefulness of the method in the context of food production. The outcome is intended to provide valuable insight to the development of the RISE biodiversity database. To assess the CB method, an LCA case study of pork will be conducted using the research questions presented below.

1. **Aspects of biodiversity captured.** *How well does the indicator of the CB method act as a proxy for biodiversity loss in terms of biodiversity attributes covered, drivers of biodiversity decline and by the reference situation used? What are the strengths and weaknesses of the method when used for assessing biodiversity impact from food production?*

2. **Land occupation and land transformation.** *How is the impact from land transformation measured and allocated? If both land occupation and land transformation are included in the impact assessment, can they be summarized?*

3. **Spatial resolution.** *At what level of spatial resolution should biodiversity impacts be assessed, and are there any limitations caused by data availability?*

1.2 Delimitations

The LCA case study is limited to the investigation of pork. Assessment of the CB method usability for other types of food products will be analyzed qualitatively. The study does not include practicalities of the RISE database, apart from aspects of LCA character. The study only includes aspects of direct land occupation and land transformation, meaning indirect land use interventions are excluded. The study is also limited to the investigation of the spatial resolution within the CB method. Delimitations related to the modelled system of the LCA are presented in section 3.1.3.

1.3 Report structure

The report is structured as follows. First, a *background* on biodiversity and the causes of decline are presented. The background is complemented with a methodological background on the CB method and how it can be applied. The *method* for the LCA case study of pork is presented and includes the definition of goal and scope, system boundary, functional unit, and the inventory analysis, impact assessment and interpretation. To be noted is that the LCA case study was complemented with a review of previous studies, some of them being LCA studies applying the CB method. Following the method section, results are presented for each focus area. The results from the LCA were analysed together with the complementary literature for the respective focus areas. As the format of the study is an LCA case study assessing the LCIA method used, these complementing literature findings are first presented in the result section as part of the LCA analysis. A *discussion* on the CB method applicability is further done which highlights the strengths and weaknesses of the CB method and our study. Finally, a *conclusion* of this study is drawn. The LCA calculations are available in Appendix, provided as a Microsoft Excel file.

2 Background

This section provides a background on what biodiversity is, what causes biodiversity loss and presents ways of categorizing and measuring biodiversity. It also includes a methodological background of the CB method and how the method can be applied.

2.1 Biodiversity

Biodiversity is defined as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (CBD, 1992). Biodiversity is fundamental to human life on Earth (WWF, 2020) and supports the ecosystem services, in regulating, providing provision, support and cultural services (Millennium Ecosystem Assessment, 2005). It regulates the Earth temperature (Willett et al., 2019), the quality of air, water, and soil, contributes to pest control as well as provides pollination. In other words, biodiversity is a fundamental part of stabilizing the conditions of Earth that are necessary for the existence of all life on Earth. It is also crucial for the supply of fuel, fibres, medicines and food to the (human) society (Benton et al., 2021).

2.2 Drivers of biodiversity decline

Since the industrial revolution, the world's ecosystems have been increasingly pressured. The WWF Living planet Index indicates a 68% global average decrease in population of mammals, birds, amphibians, reptiles, and fish between the years 1970 and 2016 (WWF, 2020). To understand the causes of this decline, one commonly uses the distinction of *indirect* and *direct* drivers, see Figure 1 (IPBES, 2019). Indirect drivers are those that contribute to the direct drivers through human activities or pressures. For example, an increase in population leads to an increased demand for food, and thus increased need for arable land that potentially drives deforestation. The indirect driver is the change of population or demography, the human pressures are the agricultural activities, and the direct drivers of biodiversity loss are represented by deforestation. The five largest direct drivers constitute of land use/land use change, direct exploitation, climate change, pollution and invasive alien species (IPBES, 2019).

Converting natural habitats for agricultural purposes is the major direct driver of biodiversity loss in terrestrial systems (WWF, 2020). The agricultural sector is of today accountable for around 80% of deforestation worldwide mainly through clearing of forests and burning of biomass and is responsible for around 70% of the terrestrial biodiversity loss (WWF, 2020). Agricultural land occupies around 49% of habitable land, of which the vast majority (78%) is used for animal food production. Important human pressures of the agricultural sector are the intensive agricultural system, the use of fertilizers and pesticides, high water usage, high meat consumption and monocultural farming (Benton et al., 2021).

The pressure from the food system on biodiversity, is the sum of the current agricultural systems and the food products produced and demanded for. The pressure and size of the impact from food products varies depending on the food characteristics, the agricultural systems applied, as well as the geographical location. For example, bananas are highly connected to terrestrial LUC in Latin America, whereas rice is responsible for high land occupation. Tomatoes and almonds on the other hand are more connected to high pesticide use and water consumption, respectively. These four food product examples all have different environmental impact pathways, for example, pesticide use leads to habitat degradation and water contamination, whereas LUC leads to direct loss of species' habitat. (Crenna et al., 2019)

The type of agricultural system plays a role for the loss of biodiversity, as different systems have different impact pathways. A distinction is often done between an intensive and an extensive agricultural system. An intensive system has high mechanisation, uses fertilizers and pesticides with the purpose of generating high output per area occupied, whereas the opposite generally goes for an extensive system. The two systems are driving biodiversity loss in different ways, either through the intensive use of chemicals aiming for land-sparing, or an extensive use but where large amount of land is required to generate the same output (land-sharing) (Benton et al., 2021). Different results of the impact from these two systems can be generated depending on the product and drivers included in the assessment. For example, a food product from an intensive system can have a lower climate impact as it generates high output (food produced) per input (energy and resources needed) compared to an extensive system. On the other hand, an extensive system could have a lower biodiversity impact if the impact of pesticides is considered. (van der Werf et al., 2020) Nevertheless, both systems are part of the human pressures of the agricultural system. The food characteristics determine what type of land is suitable or used, where some crops are cultivated annually (e.g., wheat) and some being permanent crops (e.g., oil palm), and thus requiring different types of land. Others could be picked in the wild (e.g., blueberries) or produced in greenhouses (e.g., some tomatoes). Therefore, the drivers of decline and the agricultural pressure on biodiversity is not a distinct concept.

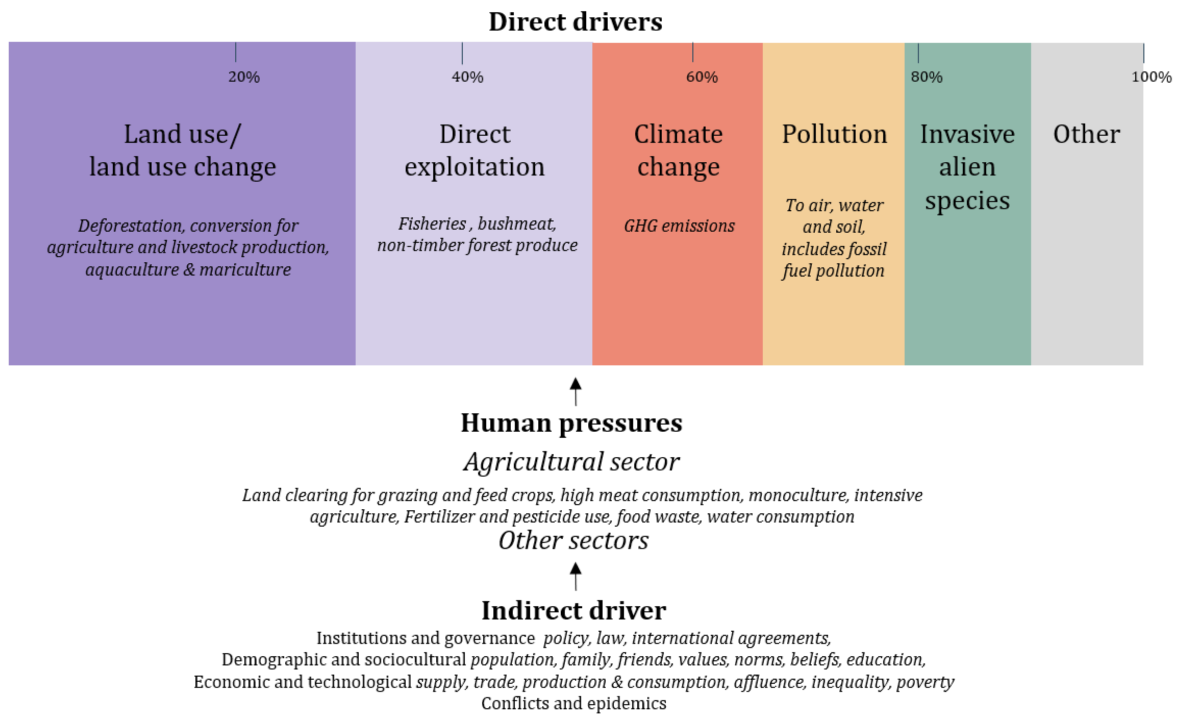

Figure 1. Indirect and direct drivers of biodiversity decline, and agricultural activities that pressures biodiversity. Adapted from IPBES (2019), IPBES (2021) and Benton et al. (2021). The percentages show the contribution of the direct drivers to biodiversity decline for all sectors

2.3 Categorizing and measuring biodiversity

As the definition by the CBD (1992) of biodiversity indicates, it covers many areas and similarly, there are many ways to quantify biodiversity. On way is to apply The Essential Biodiversity Variables (EBV), developed by the Group on Earth Observations' Biodiversity Observation Network. The EBV's are mapped areas of biodiversity that aims to guide what biodiversity parameters to observe and how it should be done in a harmonized way. The EBVs consists of six classes of biodiversity variables: genetic composition, species populations, species traits, community composition, ecosystem structure and ecosystem function (Pereira et al., 2013) and the expectation is that they can demonstrate minimum measures for monitoring change in biodiversity.

Another way is to apply the Noss' Hierarchy of Biodiversity, which is another framework that aims at illustrating the various aspects of biodiversity (Noss, 1990). It consists of twelve elements divided into the three biodiversity attributes, *composition, structure* and *function,* at four different scales, *landscape, community/ecosystem, species and genetic,* see Figure 2. Each element holds several measures of biodiversity, which indicates that there is not a single measurement that can cover all aspects of biodiversity, but different measurements are needed for different purposes. For example, measuring landscape structure could be done through satellite images to generate an indication of the spatial heterogeneity or fragmentation of landscapes. Another example is the community composition, which is connected to measures of species richness, (the number of species within a given sample) and to the relative species abundance (i.e., the relative coverage/amount of each species). Species interact with each other, for example predators eat other species, parasites live in or on and take advantage of other organisms and many organisms is part of a symbiotic relationship where each part is benefitted. These interactions shape the structure of the community but are generally not considered when measuring community structure (Starr et al., 2010). As an example, a zoo could hold many species (high in species richness) but lack the interactions in-between.

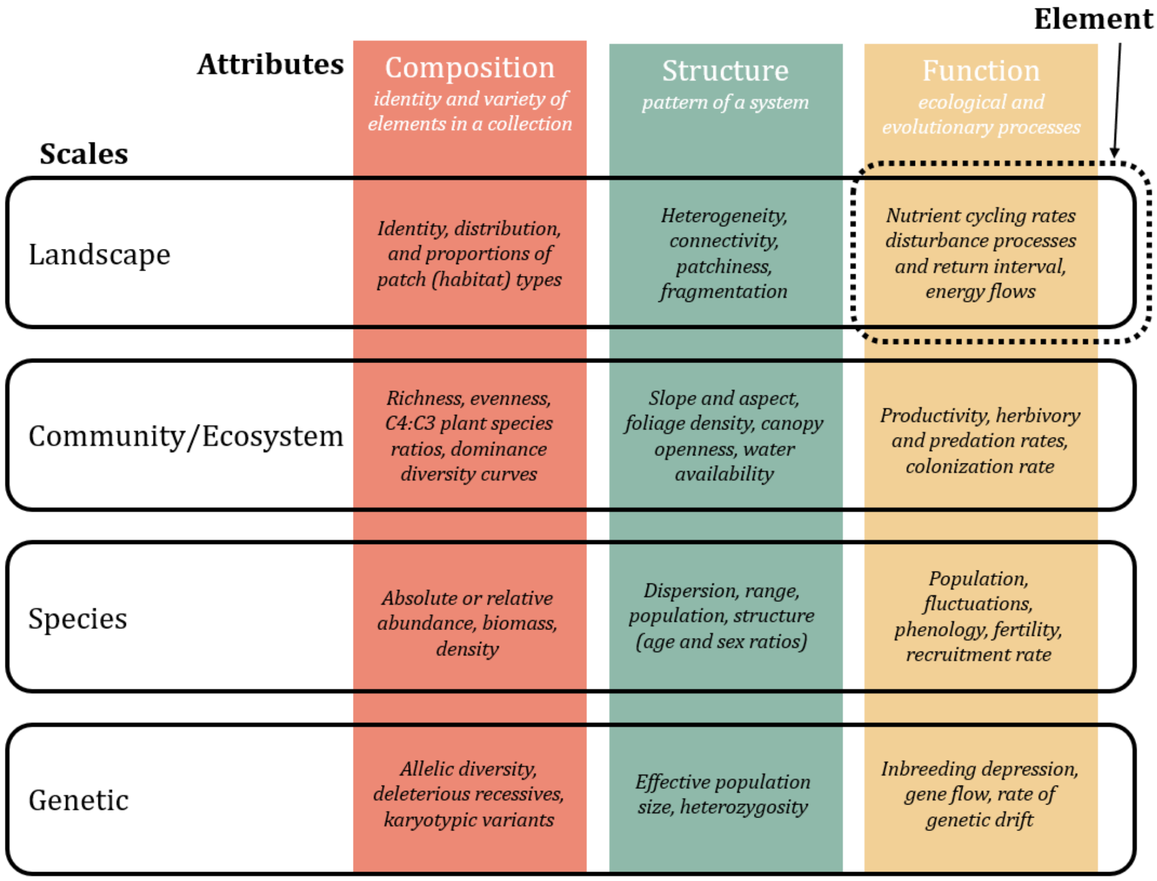

Figure 2. Noss' Hierarchy of Biodiversity where each element holds indicators for measuring biodiversity. Adapted from Bracy Knight et al. (2020)

2.3.1 Biodiversity in LCA

Biodiversity as an impact category in LCA is found both on midpoint and endpoint level, which means that different drivers are included depending on the level of aggregation. In one common

LCIA method, ReCiPe (Huijbregts et al., 2017), the impact categories contributing to the endpoint level of ecosystem damage (or biodiversity damage), include several midpoint categories, such as ecotoxicity, acidification, land use interventions, water use and climate change. The endpoint covers some of the direct drivers and human pressures demonstrated in Figure 1. The aggregated result in this method is expressed in the unit of potential disappeared fraction of species (PDF) and represents a projected fraction of species going extinct due to all of the midpoint categories. Biodiversity as a midpoint indicator is often based on land use interventions as the sole driver of biodiversity loss. However, regardless of aggregation level, most LCIA methods for biodiversity are based on the species level in the Noss' framework demonstrated in Figure 2, and very few on the genetic level (Vrasdonk et al., 2019).

Crenna et al. (2020) highlights three drawbacks with current LCIA methods on biodiversity. First, most methods quantify the impact through species richness, which only covers the community composition in the EBVs and on the species scale of the Noss' framework. Thus, important aspects of biodiversity are left out, such as those on the genetic level, the abundance of species as well as those on the scale of ecosystem or landscape diversity. Secondly, many drivers of biodiversity loss are excluded, for example the impacts from LUC, direct exploitation, and invasive species. Thirdly, the spatial resolution is often poor, and the impact is often assessed on the country level. The spatial aspect is an important parameter for biodiversity assessment as it, in contrast to global issues such as climate change, is a local concern not bound to national borders. As for both measuring biodiversity in general, and within LCA, one needs to pose the question "What are we monitoring or assessing, and why?", a question highlighted by Noss (1990) that helps to recall that there is a difference between the indicators, what they measure and for what they are useful. For literature reviews of biodiversity LCIA indicators, see Gabel et al. (2016) and Crenna et al. (2020).

2.3.1.1 Biodiversity impact from land use interventions

One frequently discussed concept when assessing biodiversity in LCA from land use interventions is the *reference situation* (or *reference state*). By comparing the reference situation, i.e., the reference quality of biodiversity with the studied situation, it is possible to measure the change of the quality of biodiversity (Vrasdonk et al., 2019). The reference situation applied, therefore determines the total impact of the change of land quality from the studied situation. Temporally, a reference situation can be a point in the future, present or past (de Baan et al., 2013a). A future reference state could represent the land area after an occupation period, meaning it could regenerate and after a given time period would reach an equilibrium where the quality of biodiversity no longer can be improved. One such future target reference state could be the level of quality stated in national or international agreements. A present reference situation is the quality of biodiversity that is believed to have had no, or little, impact from human intervention, sometimes referred to as *natural counterfactual* (Vrasdonk et al., 2019). A past reference state can be a situation that existed prior to human intervention of land. In LCA studies it is most common to use a situation that has, supposedly, not been affected by humans (Milà i Canals et al., 2007).

The evolution of land quality from land use interventions is often described as a process of a transformation process (LUC) with instant quality decline, followed by an occupational process (LU), also affecting the quality of the land. The land area and period of occupation is defined in the modelled system in the LCA. After the occupational period, the land quality is assumed to undergo regeneration. These processes are illustrated in Figure 3. The time it takes for an area to naturally regenerate [1] to a reference state is referred to as the regeneration time. The transformational impact is often referred to in terms of deforestation (IPBES, 2019), but can also

[1] Passive or active regeneration

include for example excavation of a wetland and is quantified by the change in diversity from natural habitat conversion. Occupation on the other hand, indicates a land being used for human activities that hinders natural regeneration, for example due to human activities in forest plantations, agriculture, and urban areas. Conceptually, the occupied land is gradually degraded over time, see for example the illustrated decline in land quality in Figure 3 during the occupational period. Permanent impact or irreversible damage occurs when the quality after the regeneration process is lower than the initial quality ($Quality_{initial} - Quality_{reg}$). The size of the area occupied or transformed further defines the total impact. The change of land quality and the regeneration time are geographically dependent and are determined by the specific climate and biotic conditions of a region. Further, the size of the land under study together with the temporal allocation, methodological choices to be done within the LCA, determines the total impact from the studied system. (Milà i Canals et al., 2007)

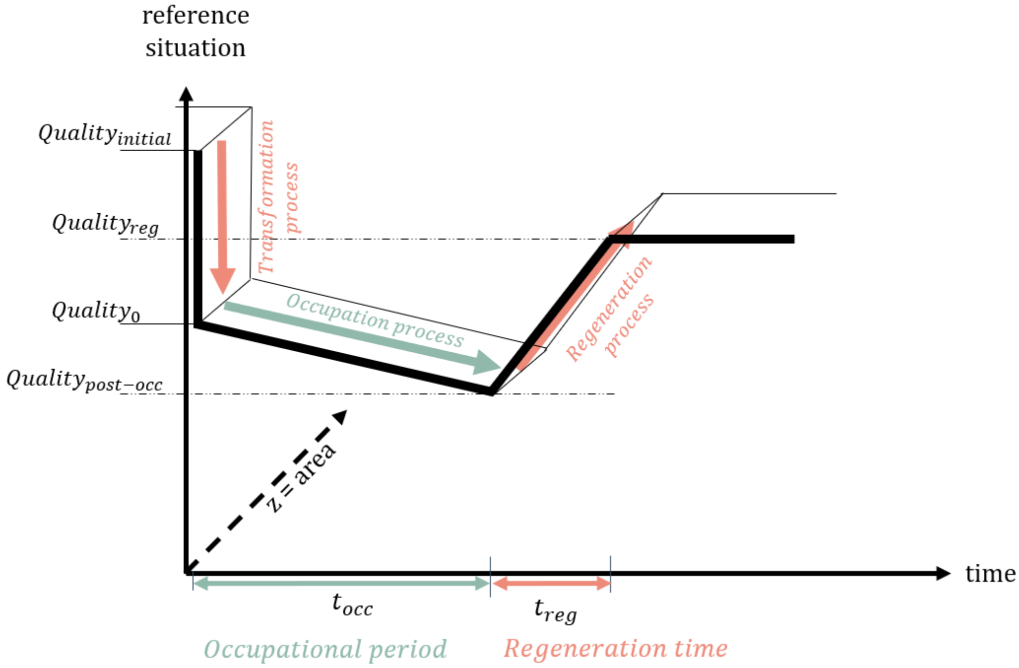

Figure 3. Schematical overview of the evolution of land quality (y-axis) from land use interventions, represented by the orange and green arrows, the x-axis shows the time duration of the intervention, and the z-axis shows the land area that is occupied or transformed. Adapted from Milà i Canals et al. (2007)

2.4 Chaudhary & Brooks LCIA method

The CB method is an LCIA method that projects global species-equivalents potentially lost from land use interventions. The method is an updated version of the method developed by Chaudhary et al. (2015) and is recommended by the UNEP-SETAC (Koellner et al., 2013) for assessing biodiversity impact. As the CB method expresses species richness decline following land use interventions, it does not include other direct drivers seen in Figure 1, such as invasive species, climate change, pollution and overexploitation. The method provides global characterization factors (CF) for five different taxa (mammals, birds, amphibians, reptiles, and plants), differentiated by 804 ecoregions (terrestrial areas with certain ecosystem characteristics) and 5 different land use types (managed forests, plantation forests, pasture, cropland and urban, each with three intensity levels), see Figure 4.

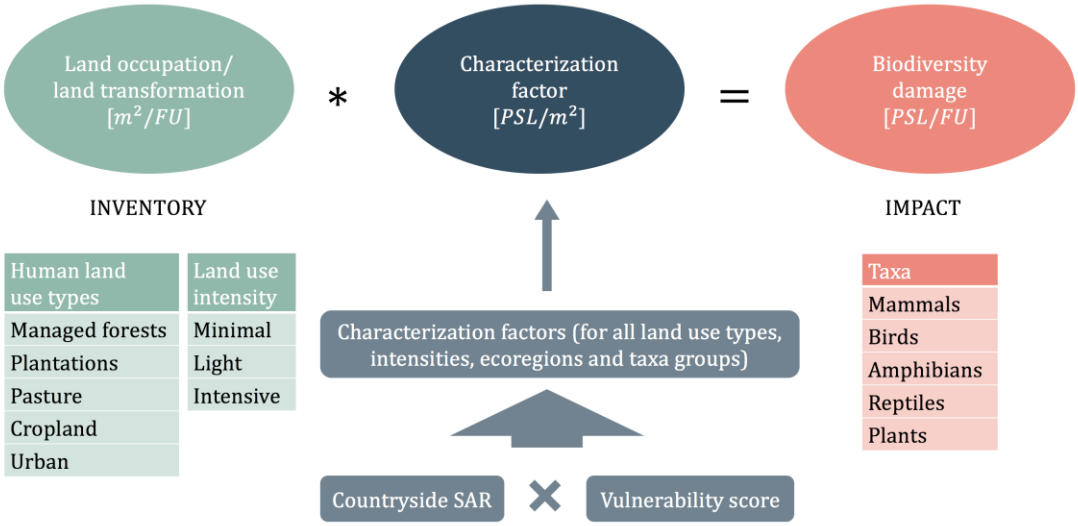

Figure 4. Overview of the CB method, adapted from Chaudhary & Brooks (2018)

The CB method provides CF for both occupational and transformational impact, as well as a taxa-aggregated and a country-aggregated CF. The impact, expressed as biodiversity damage (BD) is a measure of global extinctions in the unit of potential global-species-equivalence lost (PSL), or potential disappeared fraction of species (PDF) for the taxa-aggregated impact. The list below summarizes the main methodological choices within the CB method, where the *boundaries* reflect important LCA specific parameters to consider.

- Spatial resolution – ecoregion or country approach → *geographical boundaries*
- Land use intervention – occupational and/or transformational → *temporal boundaries*
- Taxa resolution – a specific taxa, several taxa or taxa aggregated
- Land use types, including intensities – which land use type to be used

2.4.1 Methodological background

This section presents the background calculations of the CB method and what parameters are included to calculate the CFs.

2.4.1.1 Spatial resolution

Three types of ecologically and geographically defined areas are used to develop the CFs: ecoregions, biomes and biogeographic realms, which are each in turn divided depending on their characteristics. Biogeographic realms are the largest geographical entities and are divided into areas depending on the flora and fauna found within. In each realm there are several biomes, the second largest type of areas, which are characterized by similar climate and vegetation structure. The smallest type of ecologically and geographically defined areas are ecoregions, which are "relatively large units of land that contain a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of the natural communities prior to major land use change" (Olsson, 2001). The ecoregions used in the CB method are based on the WWF Wildfinder database. CFs are developed for an ecoregional resolution, i.e., the highest spatial resolution and country aggregated resolution, i.e., the lowest spatial resolution. The country-aggregated CFs are based on the ecoregional CFs and the ecoregional area-share within each country. An ecoregion with large area shares in a country will therefore contribute more to the country-CF. Naturally, large countries or countries with high ecosystem variety can hold many different ecoregions. For example, China, Canada, Brazil, and Indonesia have many ecoregions within their national territories, see Figure 5.

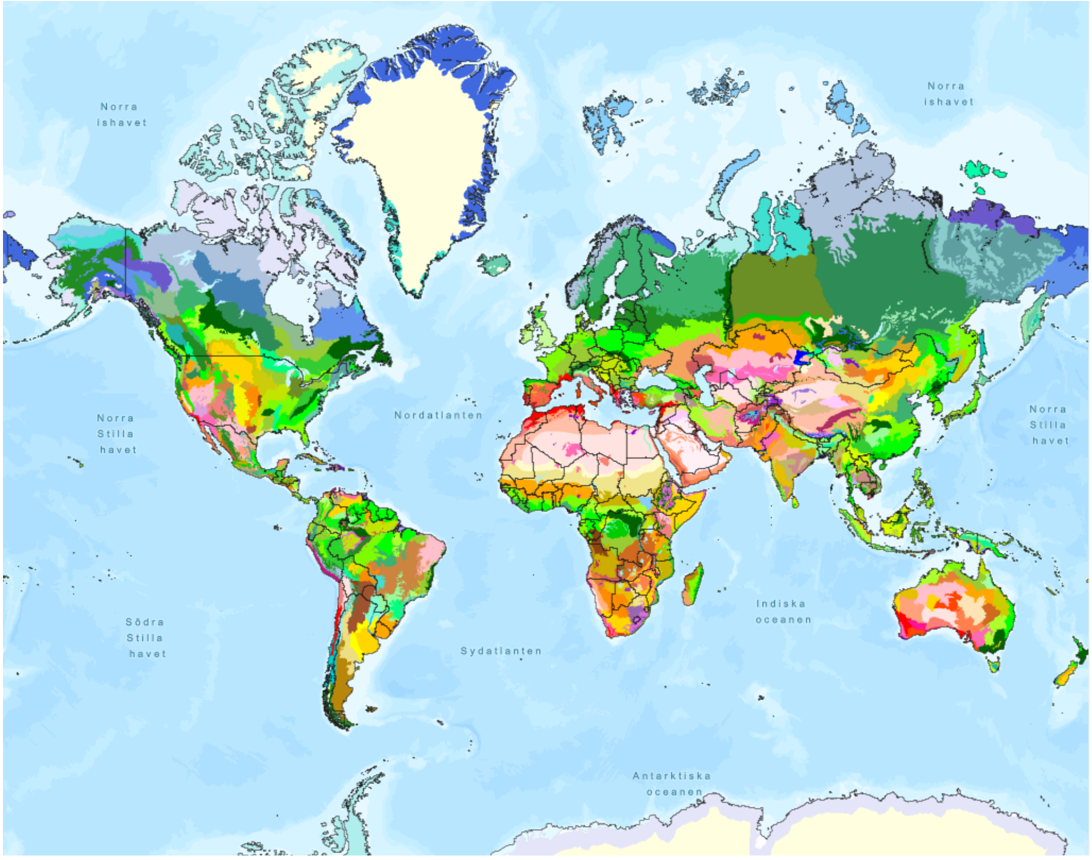

Figure 5. The WWF Terrestrial ecoregions displayed in color. National territories are marked with black lines.

2.4.1.2 Calculating the characterization factors

A species-area-relationship (SAR) model is the biodiversity model most used within LCIA (Vrasdonk et al., 2019) and is used for estimating a reduction of species, S, in relation to a change of area in natural vegetation, A. The CB method builds on the countryside SAR (cSAR) model, which, compared to the classic SAR, considers that species can survive in a human shaped landscape. See Equation 1 for the cSAR model, where *g = taxa, i = land use types, j=ecoregion.* The exponent z could be termed the "slope" as it describes the rate of species reduction per habitable area reduced (Drakare et al., 2006) and is in the CB method differentiated per ecoregion.

The species loss S_{loss} is a proxy of biodiversity loss, illustrated by the comparison of the state of species diversity before any human intervention S_{org} and the species diversity after human intervention. Without any human intervention, the area of the natural habitat A_{new} would be equal to the original size of natural habitat A_{org} (the area of the ecoregion), and thus there would be no species loss, S_{loss}. In other words, it is the size of the numerator that determines the relative species loss, by the sum of the current size of natural habitat A_{new} and current land use types A in an ecoregion.

The original species abundance S_{org}, within each taxa group, could be defined as a *species abundance reference situation.* The data on original species abundance in the method is derived from the WWF Wildfinder database (WWF, n.d.) and Kier et al. (2005) and is compiled from both current and historically recorded data. The species reference situation is a form of natural counterfactual such, where the counterfactual reference situation is the current sum of species abundance in an ecoregion.

The S_{loss} is calculated for all ecoregions and taxa groups, and further the different land use types are given a relative share of loss depending on a taxon affinity, a factor that projects potential species abundance in a human land use type.

$$S_{loss,g,j} = S_{org,g,j}\left(1-\left(\frac{A_{new,j}+\sum_{i=1}^{16} h_{g,i,j}*A_{i,j}}{A_{org,j}}\right)^{z_j}\right) \qquad \textit{Equation 1}$$

The CFs are further calculated using the cSAR and the taxon affinity and provide projected species loss from an isolated ecoregion. Species might however also occur elsewhere and thus to express the species loss in a global context, the CFs are multiplied with a vulnerability score ($0 \leq VS_{g,j} \leq 1$) according to Equation 2.

$$CF_{global,g,i,j} = CF_{g,i,i} * VS_{g,j} \qquad \textit{Equation 2}$$

The VS represents the endemic richness of an ecoregion according to the WWF Wildfinder database (WWF, n.d.) and the species threat level according to the International Union for Conservation of Nature (IUCN) Red List, with a score range of 0 ≤ VS ≤ 1. A value of 1 indicates that all species in an ecoregion are *critically endangered* and only present in that ecoregion. If a species is present in an ecoregion this is considered, however, whether the species is present in the whole ecoregion or only in a specific place is not considered. Translating the initial calculated CFs into global CFs enables a global assessment on biodiversity loss as well as it demonstrates a global reference value for biodiversity conservation.

The transformational CFs are calculated using regeneration times (t_{reg} in Equation 3) based on Curran et al. (2014) and represents the time it takes for secondary growth habitat to reach average old growth diversity values.

$$CF_{trans,g,i,j} = CF_{occ,g,i,j} * 0{,}5\ t_{reg,g,i,j} \qquad \textit{Equation 3}$$

The regeneration times have been created for different realms, biomes[2], land use intensities and taxonomic groups. They highly depend on the climatic conditions in an ecoregion, with boreal ecoregions having the longest regeneration times and tropical ecoregions the shortest (Chaudhary & Brooks, 2018). Parameters found to have high influence were latitude, altitude, and the biome type[2] (de Baan et al., 2015). The Sørensen index, an indicator used to measure biodiversity, was used to develop the regeneration times used in the CB method. It is a presence-based indicator, meaning it considers that a species occurs in a certain area. There are drawbacks with this, such as that there is a risk that when a species is detected, that individual of the species might only have been passing through the area or that it had just recently come to the area and hence the species might not yet be established there (M. Curran. personal communication, May 10, 2022) and does not consider a potential risk of disturbance that might hinder the regeneration process. A successful regeneration is in that way assumed and is not accounted for. As the species abundance reference state is based on a *natural counterfactual,* a risk for, or the size of permanent impact is neither considered.

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[2] Simplified into forest and non-forest (de Baan et al., 2015)

The CB method assumes that both old growth and secondary growth habitats are close enough to each other for species from the old growth area to recolonize the second growth area. An increased spatial resolution or the inclusion of landscape structure parameter would allow for estimating the potential of recovery. The method does not account for a distinction between passive and active recovery, i.e., the land is left undisturbed and will recover naturally, or active recovery, meaning that human actions are taken to facilitate the recovery (Curran et al., 2014). Global CF will henceforth only be referred to as CF.

2.4.2 Applying the method

One of the choices to be made when applying the CB method is how to use occupational and transformational impact, a decision to be made within the goal and scope of the LCA. None of the previous LCA studies that were reviewed using the CB method (see sections 4.1 and 4.3) have chosen to include the impact from transformation, and the approaches for LUC presented in this section are therefore not found used with the CB method.

The impact from occupation, BD_{occ} is calculated by Equation 4 and is a function of the occupation time t_{occ} and the size of the occupied land area, $area_{occ}$. See Figure 6 for an illustration of the BD from occupation generated by the CB method.

$$BD_{occ}\ [PDF * y] = CF_{occ} \left[\frac{PDF}{m^2}\right] * t_{occ}[y] * area_{occ}\ [m^2] \qquad \textit{Equation 4}$$

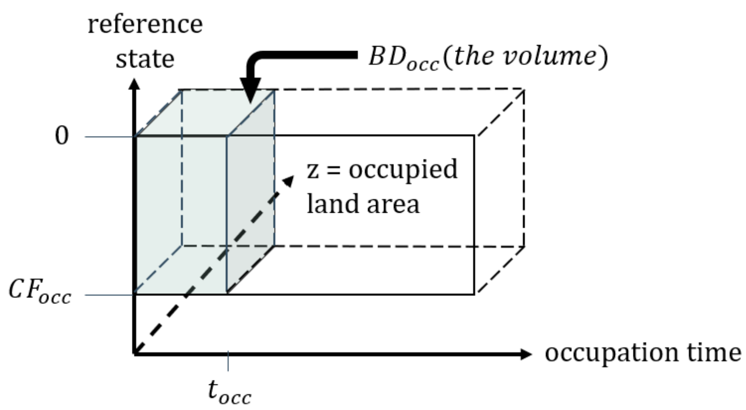


Figure 6. Schematic illustration of occupational impact, which is illustrated by the colored volume in the rectangle. The volume depends on the size of the CF_{occ} (y-axis), the occupied land area (z-axis) and the occupation time (x-axis)

The impact from occupation, BD_{trans} is a function of the transformed land area, $area_{trans}$. The size of the BD_{trans} provides information on the reversibility of the transformation. See Equation 5 and Figure 7 for how the BD from transformation is calculated and visually expressed.

$$BD_{trans}[PDF] = CF_{trans} \left[\frac{PDF}{m^2}\right] * area_{trans}\ [m^2] \qquad \textit{Equation 5}$$

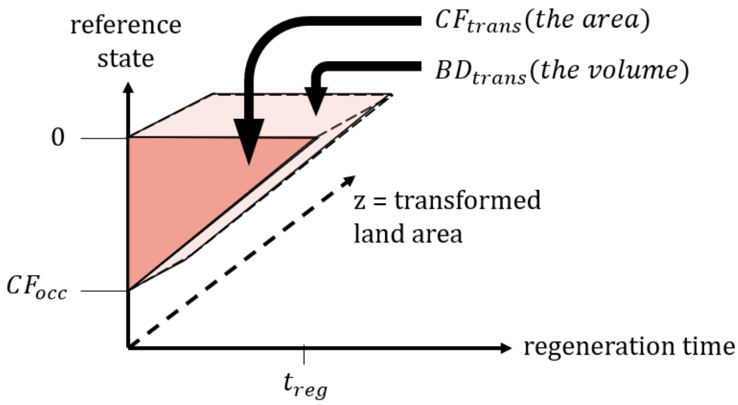


Figure 7. Schematic illustration of transformational impact, which is illustrated by the colored volume in the rectangle. The volume depends on the size of the CF_{occ} (y-axis), the transformed land area (z-axis) and the regeneration time (x-axis)

As the CFs in the CB method are developed for attributional LCA (marginal CFs are available in Chaudhary et al. (2015)), they are not aimed to be used for consequential LCA, as in "what would happen if we transformed this land?". However, the impact from LU and LUC are estimations based on potential consequences if transformed or occupied compared to a reference habitat. In attributional LCA studies, one usually has data on the size of the occupied area per functional unit (often yield) but not always how much LUC a commodity has contributed to. An overview of LUC approaches for commodities linked to deforestation analyzed by Persson et al. (2014) shows large differences in carbon footprint from LUC depending on assumptions, amortizing period and allocation approach. According to the UNEP-SETAC (Koellner et al., 2013) one should amortize the impact over the production on the land during the succeeding 20 years after transformation. The choice of 20 years is a compromise compared to a shorter allocation time, such as 1 year, which would mean that the transformational impact would only seem relevant for a short period. If one used a longer time frame, such as 100 years, there would instead be a risk of not including the impact in calculations (Koellner et al., 2013). This way of amortizing is illustrated in Figure 8 and Equation 6.

$$BD_{trans} = \frac{BD_{trans,tot}}{20} \qquad \textit{Equation 6}$$

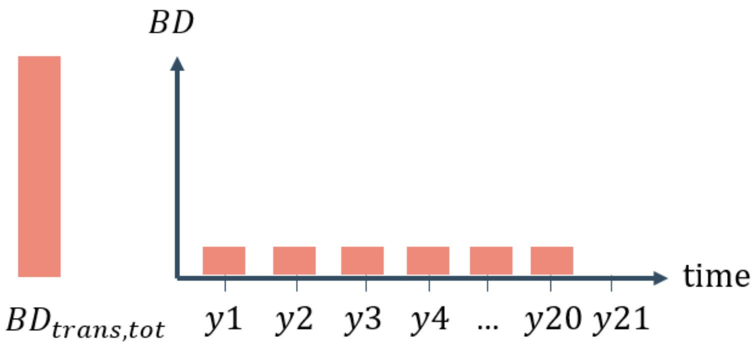


Figure 8. Illustration of amortizing the LUC over the production output the succeeding 20 years

In attributional LCA, one needs to rely on historical data of the LUC of the commodity under study. Allocating LUC to the current production output could therefore instead be based on how much land transformation a commodity has been accountable for in the past. Both Milà i Canals et al. (2013) and the LUC Impact Tool developed by Blonk Consultants (2021) use this type of approach for allocating transformational impact. The LUC Impact Tool linearly estimates the current LUC for a commodity based on the past 20 years of LUC connected to that commodity, expressed as a LUC_{factor}, see Equation 7 and Figure 9. LU_{exp} is the expansion of a certain crop in a certain country during the past 20 years and the $LU_{current}$ is the current crop production output. The LUC_{factor} is in the equation divided to match the timeframe of the LCA, here 1 year of occupation. Factors are provided for different crops and countries, also distinguished by what land use type was there before, and to what land use type it was transformed to. Both Milà i Canals et al. (2013) and the LUC Impact Tool have similar approaches but use different assumptions and allocation modes. While the LUC Impact Tool is based on how much LUC a certain crop in a certain country has contributed to, Milà i Canals et al. (2013) allocates 20 years of impact to the respective land use type from that commodity. The transformational impact can further be calculated using the Equation 8 and thus, the transformational CF and the occupational elementary flow. Equation 7, 8 and Figure 9 are based on the LUC Impact Tool.

$$LUC_{factor} = LUC_{exp} / LU_{current} / 20 years \; [m^2/ha/y] \qquad \textit{Equation 7}$$

$$BD_{trans} = CF_{trans}\left[\frac{PDF}{m^2}\right] * area_{occ}\,[ha] * t_{occ}[y] * LUC_{factor}\left[\frac{m^2}{ha*y}\right] \qquad \textit{Equation 8}$$

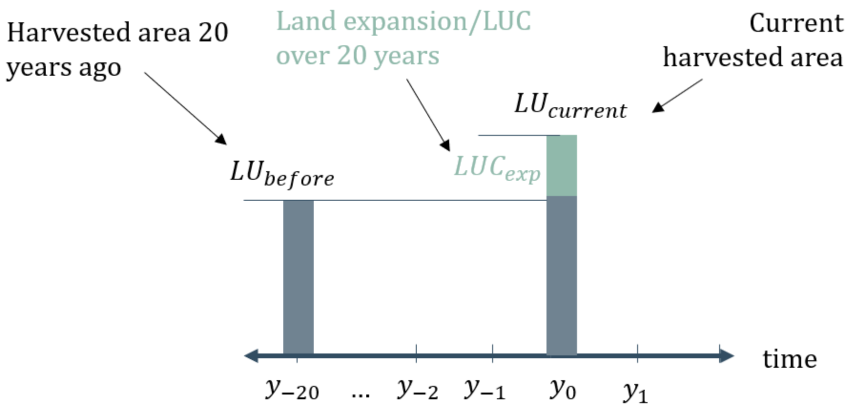


Figure 9. Illustration of how the LUC_{factor} is calculated by Equation 7. The y-axis represents the production output in hectare (ha)

3 LCA case study method

An LCA case study was made to evaluate the applicability of the CB method, and included goal and scope definition, inventory analysis, impact assessment and interpretation.

3.1 Goal and scope

The goal of the LCA was to assess the potential biodiversity impact from the production of pork supplied to the Swedish market by applying the CB method, and to provide information on how methodological choices affect the result.

3.1.1 Products under study

The products under study were Swedish and Danish pork meat. Two Swedish pork production systems and one Danish pork production system was chosen to reflect what is sold on the Swedish market. Swedish pork makes up around 70% of the market and Danish pork around 10% (Swedish Board of Agriculture, 2020). The main differences between the three production systems were the feed composition, i.e., the pig diet and the country of origin of the feed crops. Further details about the differences between the three pork products is described in section 3.2.

- Swedish pork, feed alternative A
- Swedish pork, feed alternative B
- Danish pork

3.1.1.1 Functional unit

The functional unit (FU) describes the function of a product and serves as the reference basis for the calculations in the impact assessment (Baumann & Tillman, 2004). A mass-based FU of 1 kg of bone free meat was used for the LCA case study.

3.1.2 Scope and system boundary

The pork production system included impact from land use interventions from pork production from a cradle-to-farm-gate perspective. The system included four main steps illustrated in Figure 10, the cultivation and harvesting of animal feed, animal feed processing, as well as the animal production phases pork production and slaughter phase. Downstream processes such as post-production processing, packaging, retailing, the use phase and end-of-life phases were excluded. The cradle corresponded to land transformation, step 0, when LUC was considered, and land occupation, step 1 for cases when only LU was considered.

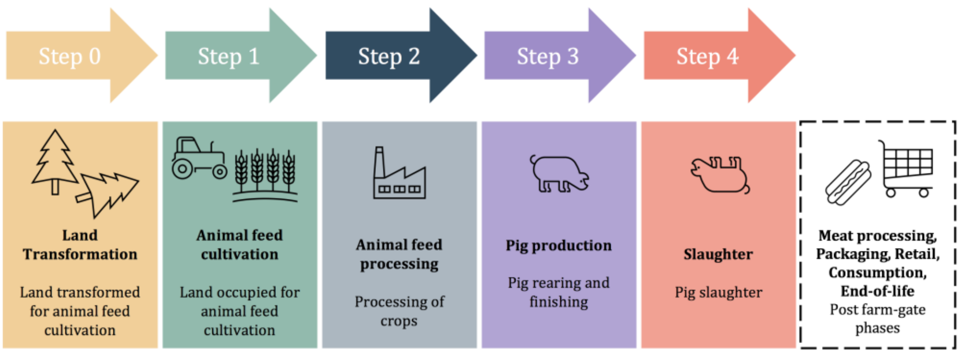


Figure 10. The five pork production phases included in the LCA. Post farm-gate processes (dashed box) were excluded

The pork production system was defined as an integrated pig farm that comprises sows, piglets, and slaughter pigs, and thus captures the total feed needed for the production of pork. The Swedish and Danish production systems differ in the animal feed intake, country of origin of the feed as well as the feed conversion efficiency. For most European countries, the largest share of feed intake (in mass) are grains (e.g., wheat, oats, barley), followed by protein feed (e.g., rapeseed meal/cake, soymeal/cake, peas) and by-products from the cereal and sugar industry (Cederberg, 2009). Figure 11 shows the system boundary, with LU and LUC as both input and output parameters, in terms of pre-LU/LUC and post-LU/LUC. It includes all feed ingredients used in this study whether used in all pork production systems or not, to generate the output of 1kg bone-free meat.

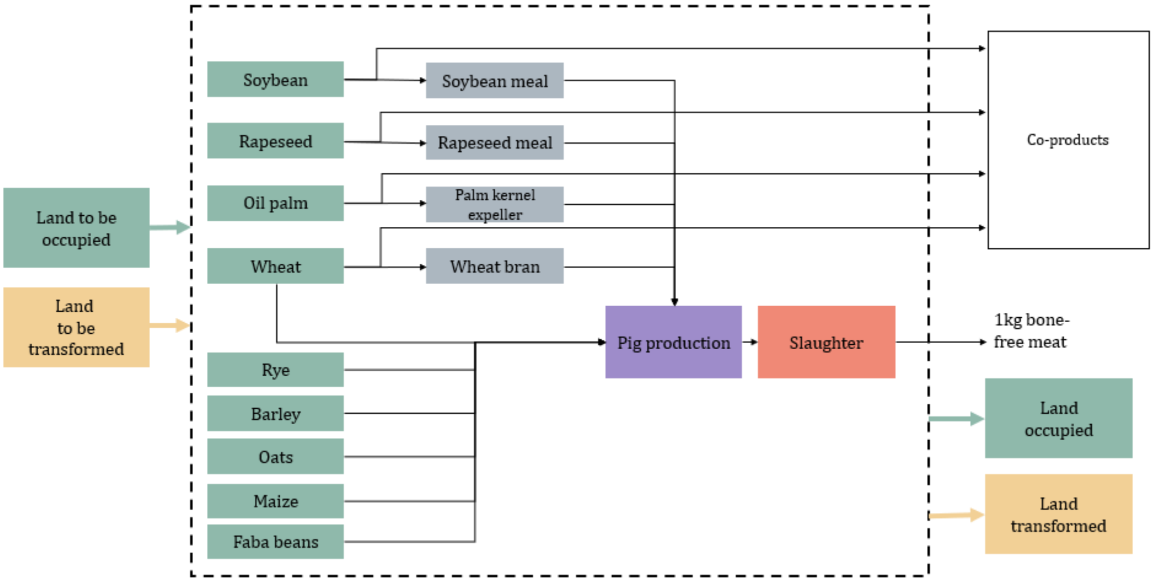


Figure 11. The system boundary of the pork production process. Transformational and occupational flows are shown both as inflow (before land use intervention) and outflow (after land use intervention)

The geographical boundaries are set to the countries of origin of the animal feed as well where the pig production and slaughter takes place. However, as the CB method assesses global biodiversity impact, the actual loss in the countries or ecoregion was not derived, but rather the global relative loss of biodiversity through the land use interventions in the countries and ecoregions considered. The temporal boundary for occupation were set to one year of animal feed cultivation necessary to provide the functional unit. The allocation of the transformational impact was determined by how much LUC the current harvested area is accountable for per year. Therefore, the temporal boundary for transformation followed the one of occupational.

3.1.3 Method delimitations

Land use interventions for farm buildings were excluded based on the assumption that the impact is negligible compared to the BD from the cultivation of animal feed (see e.g., Basset-Mens & van der Werf (2005)). Impact from the background system was excluded, e.g., biofuel for transport or wind farm for electricity production. Indirect land use intervention was also not included as any elementary flow of resources and energy, other than land area, was excluded.

If an area is harvested several times during the same year, it will be reported several times, and the yield data will therefore not be representative for how much area of land is under occupation. In that sense, national accounts and statistics does not account for multi-cropping, which according to the numbers by (Röös et al., 2017) differs depending on crop type and geographical location (i.e., land and climate conditions). Accounting for multi-cropping was not applied in this study, which therefore excludes the fact that and a lower land use for crops suitable for multi-cropping could lead to lower BD compared to what is presented in this study.

The yield statistics were most often found reporting harvested production (tons) per harvested area (ha) when reporting crop productivity. Cultivated areas that are not harvested, for example in the case of drought or damage, are thus not included. This means more land might be occupied for cultivation of crops than reported. The yield used in the study is based on the area harvested and does therefore not account for damaged cropland area.

3.2 Life Cycle Inventory

In this section, the inventory of the LCA is presented and includes the data used for the animal feed, ecoregion productivity and allocation and conversion factors. Inventory data was derived from previous LCA studies, as well as from national accounts and statistics. All calculations were computed using Microsoft Excel. For full details and references for the LCI, see Appendix.

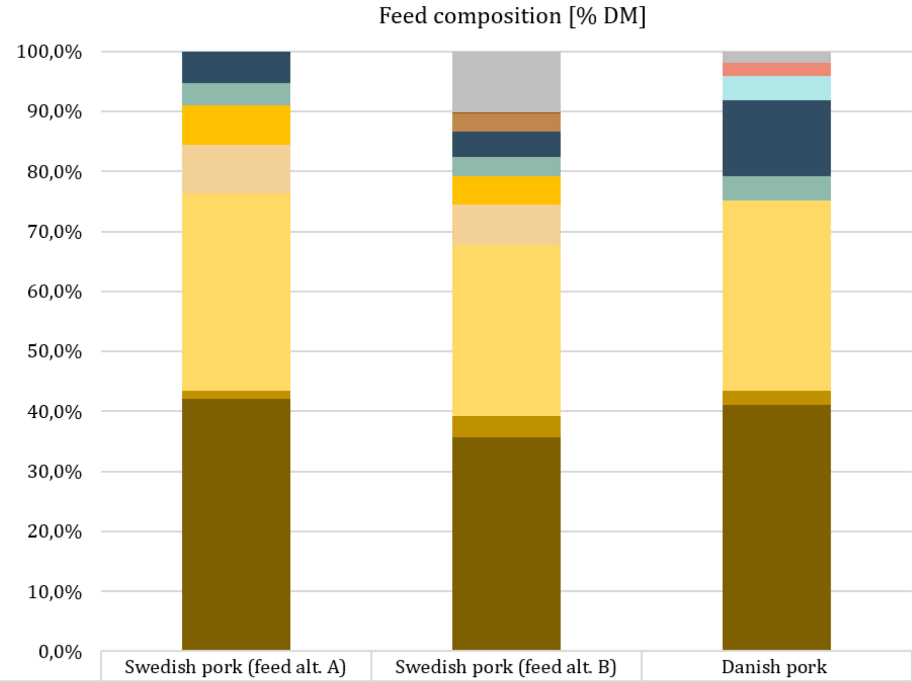
3.2.1 Animal feed cultivation and processing

Inventory within the animal feed cultivation included information on the animal feed composition, country of origin, allocation factors as well as harvested production and losses, both at country and ecoregion level.

3.2.1.1 Animal feed composition

The types and shares of the feed ingredients in a pig diet depend on the pork production system. Variations are also found between countries, within countries, between farms, but also during the year. The latter could be because of the seasonal variability of crop cultivation, but also due the variation of flows of by-products (see e.g., Zira et al. (2021), Landquist et al. (2020), Cederberg (2009) and Wirsenius et al. (2020)). Therefore, two Swedish feed compositions were included, with main differences in the share of by-products. Country of origin for the feed crops has been determined by an import/export approach based on national trade statistics and previous LCAs on the topic. The feed composition and country of origin[3] are presented in Figure 12. For full detail on how the feed composition and country of origin was withdrawn from the original sources, see Appendix, Table S3.

[3] Argentina = ARG, Brazil = BRA, Denmark = DK, Indonesia = IND, Poland = PL, Russia = RUS Sweden = SWE



	Swedish pork (feed alt. A)		Swedish pork (feed alt. B)		Danish pork	
by-products	0,0%	-	10,1%	SE	2,0%	DK
palm kernel expeller	0,0%	-	0,0%	-	2,2%	IND
Sunflower cake	0,0%	-	0,0%	-	4,1%	RUS
maize	0,0%	-	0,3%	PL	0,0%	-
rye	0,0%	-	3,0%	SE	0,0%	-
soybean meal	5,2%	BRA	4,2%	BRA	12,6%	ARG
rapeseed meal	3,7%	DK	3,2%	DK	4,1%	DK
oats	6,7%	SE	4,8%	SE	0,0%	-
faba beans/peas	8,0%	SE	6,7%	SE	0,0%	-
barley	33,0%	SE	28,5%	SE	31,6%	DK
wheat bran	1,3%	SE	3,6%	SE	2,4%	DK
wheat	42,0%	SE	35,6%	SE	41,1%	DK

Figure 12. Animal feed composition in percentage dry matter weight, and country of origin[3] Feed composition for Swedish pork (feed alt. A) adapted from Zira et al., (2021), Swedish pork (feed alt. B) adapted from Landquist et al., (2020) and Danish pork adapted by (Nguyen et al., 2010)

3.2.1.2 Harvested production

The data for harvested production was derived from EUROSTAT[4] in NUTS2[5] resolution for feed crops originating from the European Union (EU). For feed crops outside the EU, data from national accounts and statistics were used in the resolution of provinces[6]. The provinces with the highest production that together made up 60% of the total domestic production were included for countries outside EU.

The ecoregion area-share per NUTS2-region or province was measured using layers in ArcGIS [7]online to determine the ecoregion share of harvested production. This was done to be able to distribute the ecoregion of origin of the animal feed. The yield in m^2/ha was also derived using the area-share and productivity of the state. This means that all ecoregions in a province were

[4] European Union statistical database
[5] Nomenclature of Territorial Units for Statistics, a geocode standard for referencing subdivisions of countries in the European Union. NUTS2 is the 2nd level resolution.
[6] Federative units of Brazil, Oblasts of Russia, Provinces of Indonesia, Provinces of Argentina
[7] Software and online Geographic Information System

included even if some do not have crop production, as this is a result of an ecoregion existing in a region that has high productivity. The method applied for calculating harvested production per ecoregion is explained with Brazil as an example. Figure 13 shows ecoregions and provincial borders of Brazil, where the three provinces with check marks were included in the study as they covered around 60% of total domestic soy production. The ecoregions in these provinces were used in the study and their productivity based on their area-share in each province. See Appendix, Table S2 for harvested production and yield per country and ecoregion.

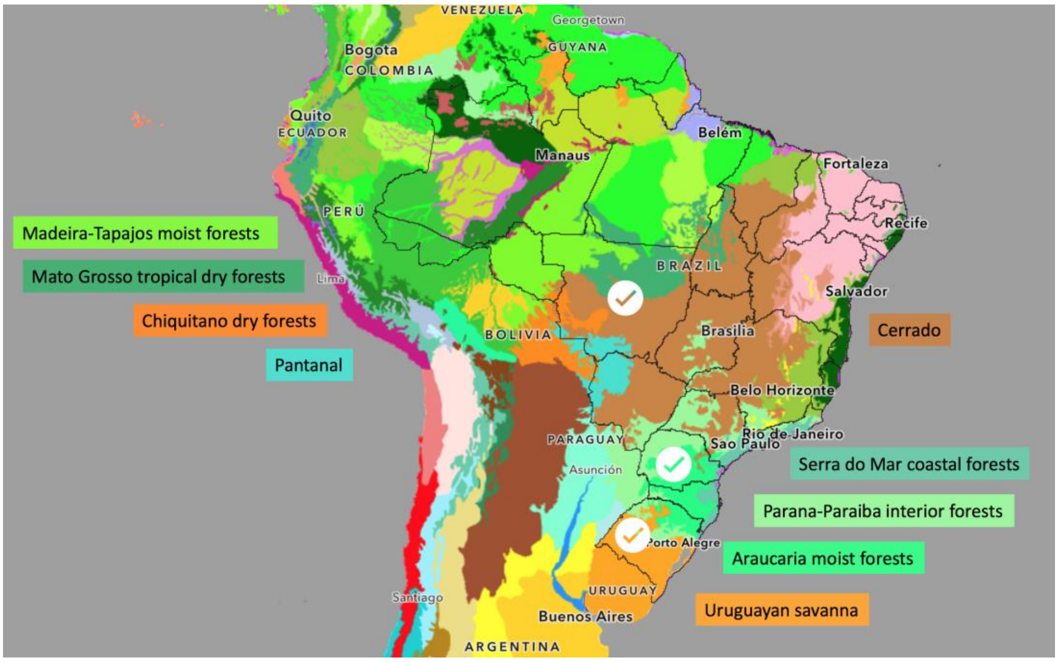


Figure 13. The WWF Terrestrial ecoregions in Brazil in color. The three check-marked provinces and their respective ecoregions were included in the study

3.2.1.3 Waste and losses

Waste fractions based on Gustavsson et al. (2011) were included for losses during post-harvesting handling and storage, see Table 1. Harvesting losses were already included in the EUROSTAT database. In the non-EU national accounts, harvesting losses were assumed to be included as data was reported in tons harvested.

Table 1. 'Post-harvest handling and storage' waste fractions (Gustavsson et al. 2011)

	Cereals	**Oilseeds and pulses**
Europe	0,04	0,01
Latin America	-	0,03
South and Southeast Asia	-	0,12

3.2.1.4 Animal feed processing

The animal feed processing phase is not accountable for land use interventions, and thus does not hold an inventory of elementary flow. For this step, economic allocation factors were used for the processed feed ingredients according to Table 2. The feed ingredients that were not processed, were allocated 1 to 1 (e.g., wheat, barley, oats). The by-products included in two of the animal feed diets were given the economic allocation factor of zero due to time limitations. For more details, see Appendix, Table S4. No waste/losses were included for animal feed processing.

Table 2. Economic allocation factors (Moberg et al. 2019)

Animal feed type	Economic allocation factor
wheat bran	0,04
rapeseed meal	0,28
soy meal	0,65
palm kernel expeller	0,03
sunflower cake	0,28

3.2.2 Pig production and slaughter

In the pig production and slaughter phase, land use interventions from farm buildings and slaughterhouses were excluded. The impact from these two phases stems therefore only from conversion factors through feed conversion and losses. The amount of feed needed to produce an amount of meat is referred to as feed conversion efficiency and expressed in the unit of kilo dry matter (DM) feed per kilo carcass weight (CW). The feed conversion efficiency depends highly on the pork production system. According to Wirsenius et al. (2020), the Danish pork production system has a 13% higher feed conversion efficiency than the Swedish production system. See Table 3 for feed conversion efficiencies and conversion factors, and see Appendix, table S4 for more details. Waste in animal production was assumed to be included in the feed conversion efficiency. No economic allocation was made for the pig production and slaughter phases, instead, all impact was allocated to the edible meat.

Table 3. Feed conversion efficiency and carcass weight conversion factor (Moberg et al., 2019, and Wirsenius et al., 2020)

	Swedish pork (Alt. A and Alt. B)	**Danish pork**
Feed conversion efficiency [kg DM/kg CW]	4,20	3,65
Carcass weight to bone free meat conversion factor [kg CW/kg bone free meat]	1,66	1,66

3.3 Life Cycle Impact Assessment

The Chaudhary & Brooks (2018) LCIA method was used for estimating the biodiversity impact. As previously described, there is one CF for each combination of taxa, land use type (incl. intensity level), land use intervention type and ecoregion. All calculations used the CF for the taxa aggregated impact, and cropland under the intensity level "intense use". This land use type was used for both occupational and transformational impact. Occupational CFs were used in all calculations, transformational CFs on the other hand were only used in those cases where the crop contributed to land transformation. Country aggregated CFs and ecoregion CFs were both used. See Appendix Table S6A for BD of the feed types and Table S6B for BD of the pork products.

3.3.1 Calculating the occupational biodiversity damage

The occupational biodiversity damage, $BD_{occ,f}$ was first calculated per kilo feed type f, by multiplying the land use per kg feed type with the corresponding CF according to Equation 9.

$$BD_{occ,f}[\ PDF_{occ}/kg_f] = area_{f,j} * t_{occ} * CF_{occ,i,j} \qquad \textit{Equation 9}$$

The total biodiversity damage $BD_{occ,p}$ for each pork product p, was further calculated by adding the BD of the (up to 12) feed types with the respective feed ratio FR_f, and further multiplied with the pork production and slaughter conversion factors, se Equation 10.

$$BD_{occ,p}[\,PDF_{occ}/kg_p] = (\textstyle\sum_{f=1}^{12} BD_{occ,f} * FR_f) * conversion\ factors \qquad \textit{Equation 10}$$

3.3.2 Calculating the transformational biodiversity damage

To determine if land transformation should be allocated to the feed crop, and if so, how large share of the occupied area should be considered to contribute to land use change, the LUC Impact Tool (Blonk Consultants, 2021) was used. The transformation from the land use types; grassland, forests and unattributed was included to derive the LUC_{factor}, which is the average transformed area per current production output in $m^2\ /ha/y$. Equation 11 was further used to calculate the transformational BD of the four feed crops shown in Table 4, using the LUC_{factor}. The four feed crops accountable for LUC was soybean from Brazil and Argentina, palm oil from Indonesia and sunflower from Russia. Equation 12 was finally used to calculate the transformational BD of the pork products.

$$BD_{trans,f}[\,PDF_{trans}/kg_f] = area_{occ,f,j} * t_{occ} * CF_{trans,i,j} * LUC_{factor} \qquad \textit{Equation 11}$$

$$BD_{trans,p}[\,PDF_{trans}/kg_p] = (\textstyle\sum_{f=1}^{12} BD_{trans,f} * FR_f) * conversion\ factors \qquad \textit{Equation 12}$$

Table 4. Factor for allocating LUC in m^2 to the harvesting production output in ha/y

Country	**Feed crop**	**LUC_{factor} $[m^2/ha/y]$ LUC attributed to the crop and country**
Argentina	Soybean	251
Brazil	Soybean	291
Indonesia	Palm	392
Russia	Sunflower	92

3.4 Interpretation

As the study aimed at assessing the CB method rather than the biodiversity impact of pork, the interpretation was done by setting the LCA results into context of the focus areas. This means that the LCA results were interpreted and analyzed for each focus area and compared to relevant literature findings of the respective focus area.

4 Results and analysis

The aim of the study was to assess the applicability of the CB method when used for evaluating biodiversity impact from food production. This was done by examining three essential areas for biodiversity indicators within LCA. This section presents the findings for these three focus areas, presented in separate subsection. Results from the LCA case study of pork and reflections on how other studies have mastered the method, are presented connected to each focus area in the respective subsections. For full details on the calculations, see Appendix.

4.1 Aspects of biodiversity captured

This section presents the findings of the first focus area on how well the CB method acts as a proxy for biodiversity loss and its ability to capture different aspects of biodiversity impact from food production. First, the general LCA results are presented using occupational CFs and country CFs. These are followed by findings from literature on how the land use parameter stands in relation to other drivers as well as the methods ability to capture different agricultural systems. Lastly, a reflection is made on the cSAR theory of the use of species richness as a proxy for biodiversity.

4.1.1 LCA results

The LCA results show that the Danish pork product accounts for higher BD compared to the Swedish pork products, with 32% and 44% lower BD for the feed alternatives A and B respectively. This difference is illustrated in Figure 14, which also shows the process contribution to the total BD. To be noted is that the figure includes both direct land use, i.e., cultivation of animal feed, and the contribution of the upstream processes to more land use needed due to losses. NB LCA results in this section are based on the occupational and country approach.

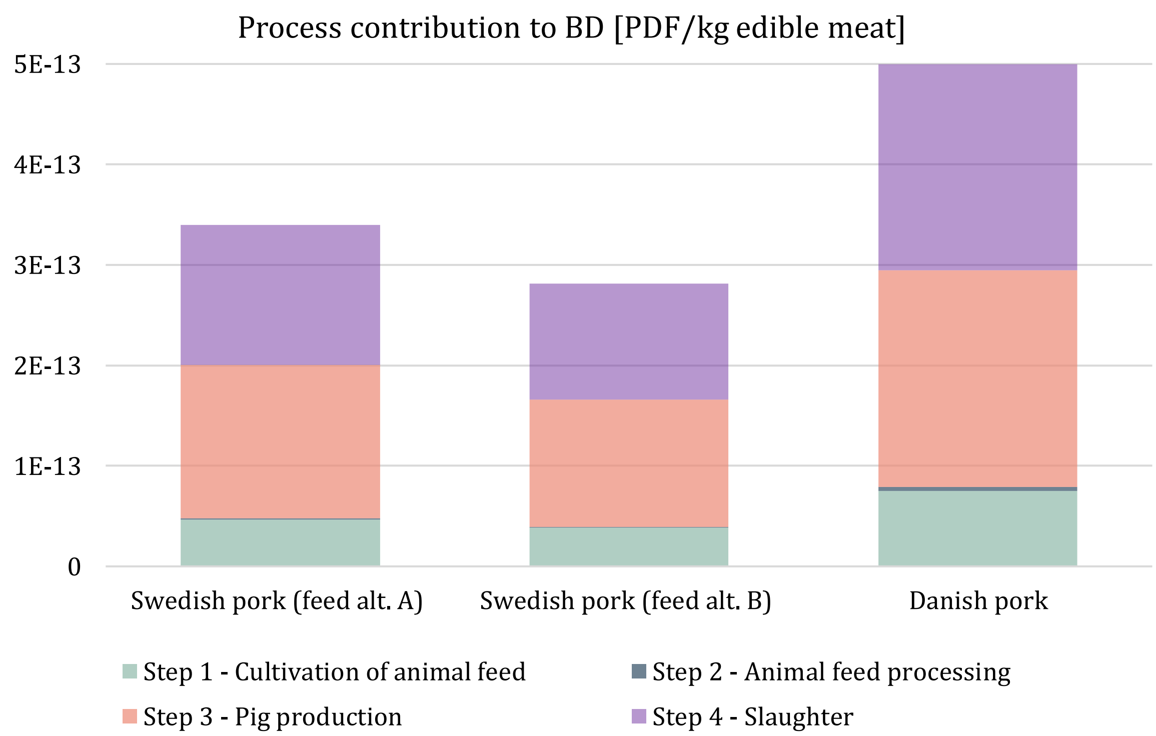


Figure 14. Process contribution. The impact from the succeeding steps after animal feed cultivation is based on the indirect need for more land to produce the functional unit of 1 kg bone free meat.

The contribution of the feed ingredients to the total BD compared to the feed ratio in DM is illustrated in Figure 15. The feed ratio in DM is withdrawn from Figure 12 to illustrate the differences in BD contribution of the feed types. The result shows that despite the low inclusion of soybean meal, 5,2% for the Swedish alt. A, its contribution to the total impact is around 65%. The same pattern is found for all three products. The differences in BD between the Danish and Swedish pork products are mainly due to the higher share of soybean meal in the Danish feed composition and the inclusion of palm kernel expeller from Indonesia. The difference in BD between the two Swedish alternatives is mainly due to the difference in share of by-products, where the feed alternative A requires more alternative feed. This is a direct result of allocating the impact to the co-products rather than the by-products in the feed and the differences in BD between the Swedish pork products should be interpreted with care.

The lower land use of soybean meal from Brazil (Swedish pork products) compared to that of Argentina (Danish pork product), displayed in Figure 16, does not make up for a higher CF in Brazil compared to Argentina. Producing a kilo of soybean meal with the origin of Argentina will have a lower BD than producing it from Brazil. The higher feed conversion efficiency of the Danish pork production system does not make up for the larger share of soybean meal and palm kernel expeller. Therefore, in these cases, the feed composition and the country of origin of the feed seems to have a higher influence on the total BD over the feed conversion efficiency and yield parameters.

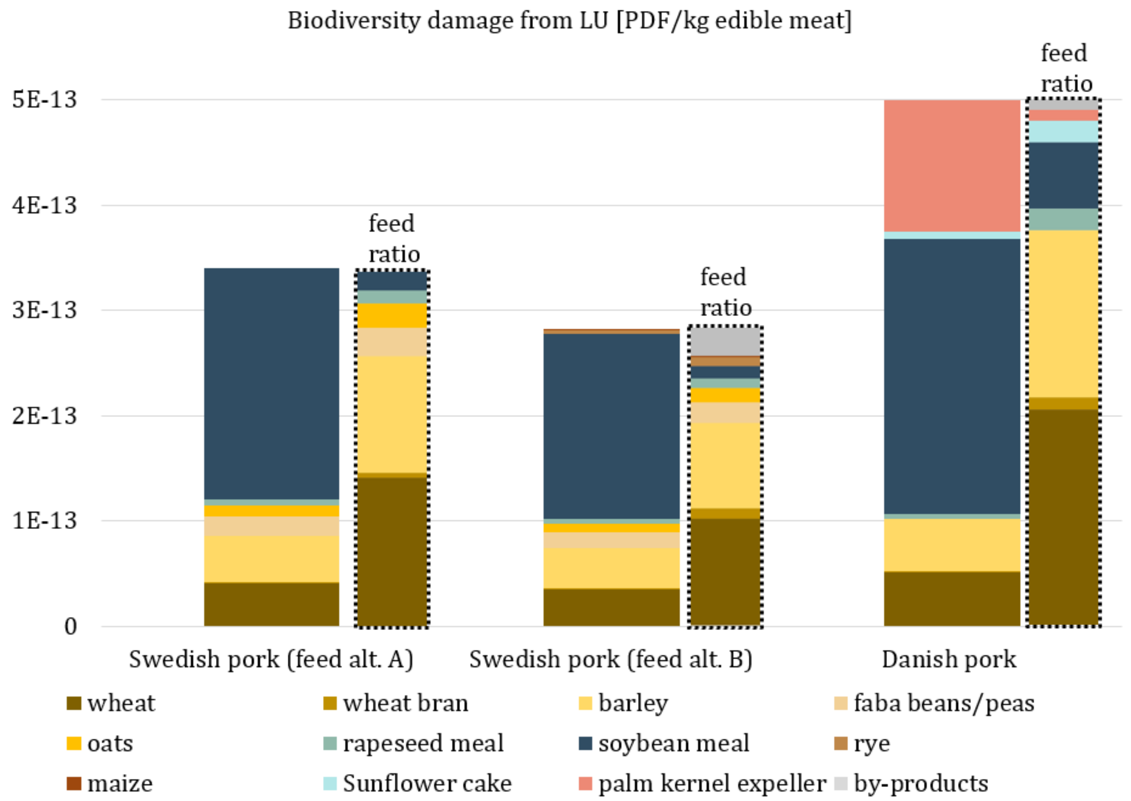


Figure 15. Biodiversity damage contribution of the feed ingredients (left bars) and visual comparison of each product's respective feed ratio in percentage DM (right bars, dashed borders). NB, the scale represents the BD contribution of the feed ingredients (left bars).

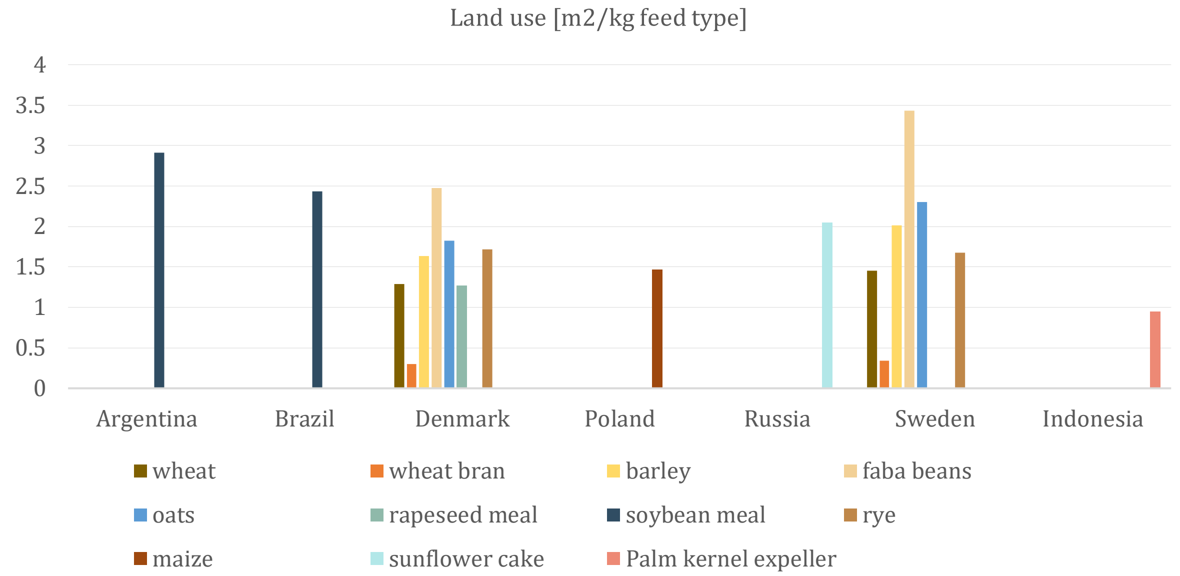


Figure 16. Land use per kg feed type (includes economic allocation, waste and losses along the supply chain)

The results of the LCA case study were compared to two other LCA studies presenting BD results on pork products using the CB method, Karlsson Potter & Röös (2021) and Møller et al. (2022) (though only for occupational impact). In the former, Swedish pork was assessed and got a BD just over $3 * 10^{-13}\ PDF/kg$, which is similar to our results of the occupational impact of $3{,}4 * 10^{-13} PDF/kg$ (feed alt. A) and $2{,}82 * 10^{-13} PDF/kg$ (feed alt. B). Møller et al., (2022) who assesses Norwegian pork got a BD of $1{,}1 - 1{,}7 * 10^{-13} PDF/kg$ depending on the scenario.

4.1.2 The land use parameter

The CB method covers the direct drivers of land use and land use change, the most important driver of terrestrial biodiversity loss. To illustrate how the land use parameter stands in relation to other drivers and human pressures, the results from a study by Crenna et al. (2019) on food consumption in Europe was used. This study showed that the relative contribution of midpoint categories to the endpoint category of ecosystem damage differs between food products, see Figure 17. For the food products included in the study, the midpoint category of land was the overall largest single contributor, but with large differences between the products. NB, land transformation is included in the land use category in ReCiPe 2016 (Huijbregts et al., 2017). For example, the contribution of the land use parameter to the total impact for cod and tomatoes was around 1%, to between 80-90% for olive oil and sunflower, with pork in the middle of and around 60%.

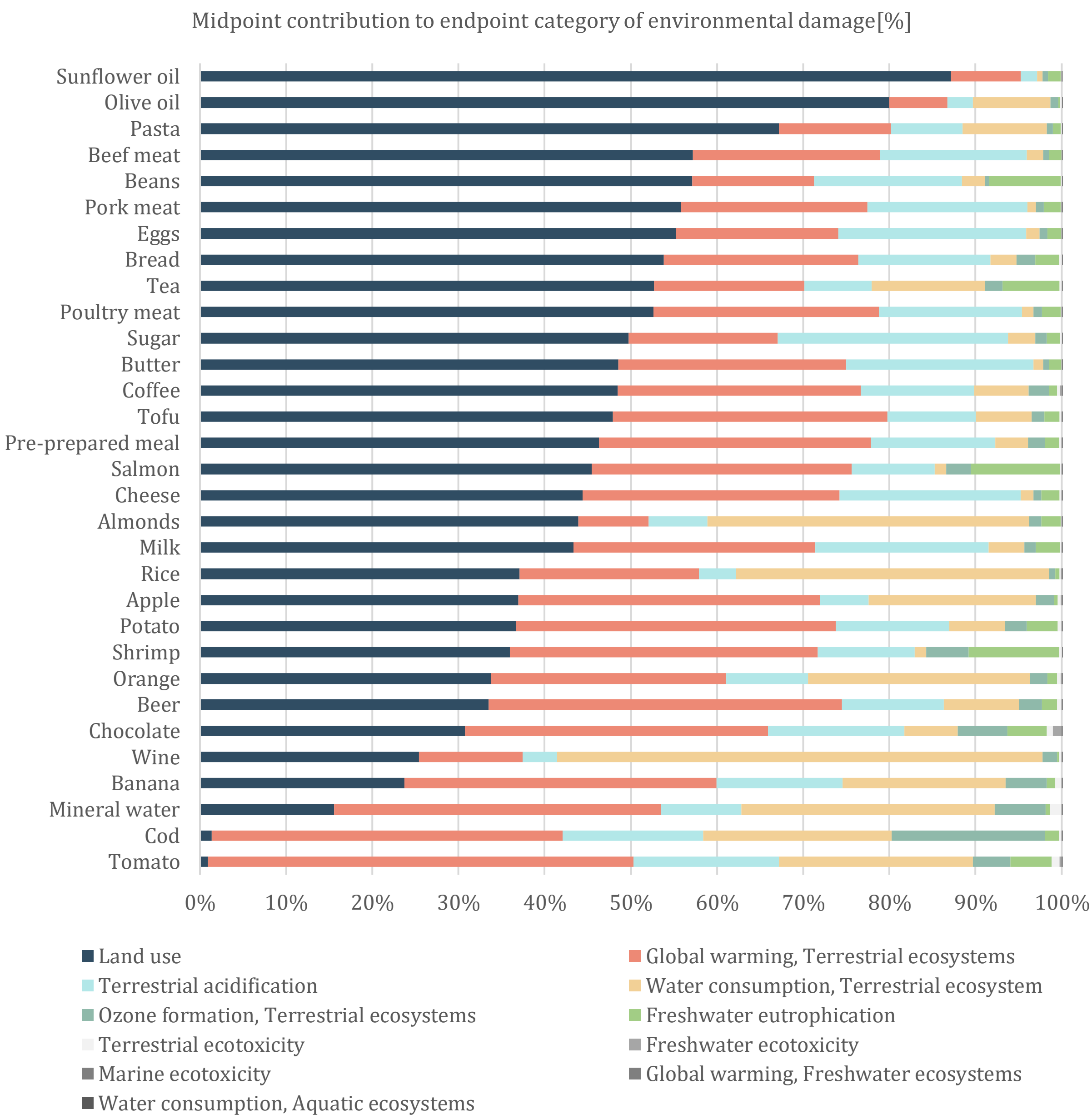


Figure 17. Midpoint contribution to endpoint for different food products, ReCiPe 2016. Adapted from Crenna et al. (2019)

The study by Crenna et al. (2019) also estimated the BD by the CFs developed by Chaudhary et al. (2015). Normalizing these results with the ReCiPe 2016 show an indication of how an aggregated endpoint category (that includes more drivers) correlates to a midpoint category of BD (that uses land use as a driver). In Figure 18, one can see that the two approaches follow similar patterns yet display some differences. The largest differences were found for cod, shrimp, and salmon, followed by poultry, wine, tofu and mineral water. The food products have less than 40% contribution from land use (in Figure 17) also demonstrates lower BD using the ReCiPe endpoint method.

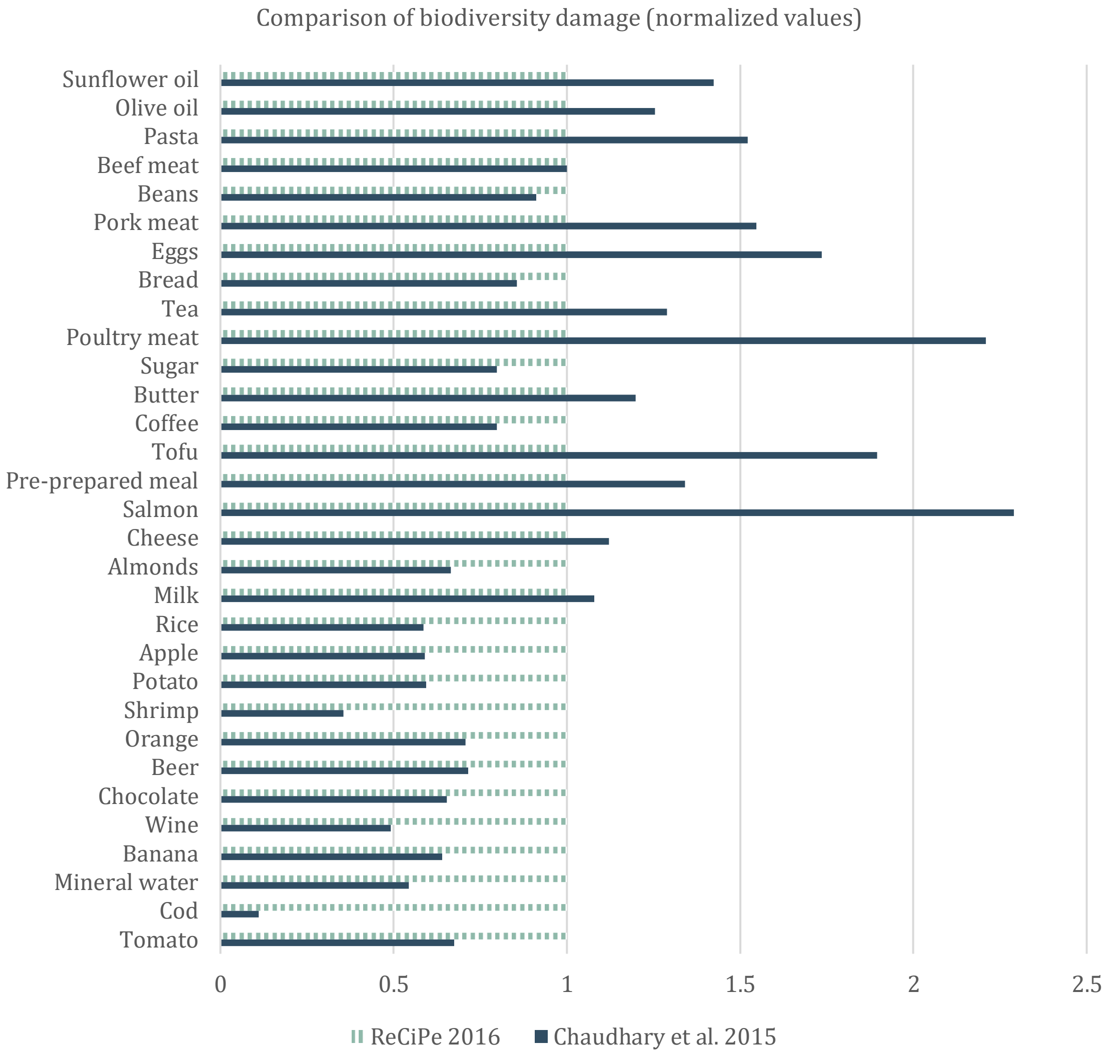


Figure 18. Comparison of Chaudhary et al. (2015) and ReCiPe 2016, normalized to beef meat and ReCiPe-results. Data from Crenna et al. (2019).

The midpoint contribution analysis and the comparison of BD between the two methods presented show that the land use parameter is important, even though it does not manage to capture other midpoint categories (drivers to some extent). It illustrates that the choice of which driver(s) to include, could present different results depending on which product is assessed. Important to note is the fact that the study by Crenna et al. (2019) also builds on an LCIA method and LCA methodology, and therefore includes assumptions and uncertainties. Another important note is that the overexploitation and invasive species are not captured in either ReCiPe 2016 or CB.

4.1.2.1 Agricultural systems captured

Apart from understanding what drivers the CB method covers, its possibility to capture a spectrum of land use related parameters should be improved for a more nuanced assessment of food products. Van der Werf et al., (2020) highlights the narrow perspective on agricultural systems as an area of concern within current practices of LCA in the agricultural sector. The CB method is no exception, which is largely linked to the land use types developed for agricultural land. As the agricultural system is much accountable for land use interventions and biodiversity decline, one would assume including more land use types would be favourable. For example, de Baan et al. (2013a) found that differentiating between annual and permanent cropland[8] could have an influence on the result. This was also the finding by Milà i Canals et al. (2013) who, in an LCA case study on margarine, showed that the relative contribution from the permanent crops was lower than from annual crops. A differentiation of permanent and annual cropland in the CB could therefore have generated lower contribution of BD from the palm kernel expeller compared to the current results.

Organic agriculture could have a potential of 30% higher species richness compared to conventional farming (van der Werf et al., 2020). This is due to a lower amount of fertilizers and pesticides used, but also due to the types of agricultural system that are often applied in organic production. The three intensity levels for cropland in the CB method aim to capture the differences in agricultural systems. However, they are not large enough to account for a lower yield generated by organic farming. Therefore, comparing a conventional crop with an organic grown crop in the same ecoregion or country would generate lower impact for the conventionally grown crop. As the CB method is designed today, it therefore tends to favor intensive farming and the land-sparing theory. A potential reason is that any land use intervention influences species richness, regardless of the intensity of the agricultural system.

Large-scale intensive agriculture and pesticide use are the largest contributors to insect decline (Sánchez-Bayo & Wyckhuys, 2019) and organic farming demonstrates positive effects on microbial abundance in soils (van der Werf et al., 2020). The exclusion of certain taxa groups, for example insects and microorganisms (e.g., bacteria and fungi) could be one reason of the small differences between the CF of the intensity levels, which instead of including more land use types, could compensate for the differences in intensity levels. However, this is partly explained by insects (and other arthropods) and microorganisms are, in comparison to the taxa groups included in the CB method, not covered by monitoring programs globally and they are thus to a large extent unknown. If they were included this could also be highly biased to the countries that have more data.

The CB method aims at differentiating between agricultural intensity levels (i.e., organic and conventional). Yet, as the method is designed today it does not manage to capture potential differences in species richness between different agricultural systems. Applying techniques such as agroforestry and crop rotation as well as using little or no fertilizers or pesticides to decrease the pressure from agricultural activities on biodiversity will therefore not be captured.

4.1.3 The species richness parameter

The cSAR model estimates the reduction of species resulting from the conversion of natural habitat. The structure of the cSAR model, on which the CFs are based, could lead to higher CFs for ecoregions with little natural habitat left, and the opposite for undisturbed habitat. This was found to be the case for the CF developed by the de Baan et al. (2013b). However, the results from our LCA case study show that the overall impact does not follow this pattern. Sweden had the lowest CF with much converted natural habitat in the past, whereas Indonesia had the highest. The BD contribution of the animal feed was highest for feed crops from tropical regions, which is also in line with the rate of change of the Living Planet Index (WWF, 2020).

Species richness could be a good proxy for measuring the quality of biodiversity, but one should be aware of what it captures (Nilsson, 2019). In Noss' hierarchical framework, the CB method could fit in the elements of species composition (see section 2.3). No source has been found

[8] Included in Chaudhary et al. (2015)

showing that one indicator is more important than another, but one can imagine that they all have their place and purpose. It is therefore not possible to say that using one indicator is more accurate than using another. When applying the CB method, it is highly recommended to understand what elements of biodiversity it captures when interpreting and making use of the results for a transparent communication.

4.2 Land occupation and land transformation

For the LCA results presented in the previous section, 4.1, occupational CFs were used. The coming section investigates how the methodological choice of including transformational CFs can affect the result.

The results from the LCA shows that the BD is higher for land transformation compared to that of land occupation for all three products, see Figure 19. The BD from transformation for the Danish pork is almost three times higher than the occupational BD, whereas around 1.5 times for the Swedish pork products. The differences between the Swedish and Danish pork products are found in the feed composition, and the country of origin of the feed crops of the Danish pork. As stated in previous sections, the Danish pork has higher inclusion of soybean meal, which is one explanation of the differences. Another is that the Danish pork is including palm kernel expeller from Indonesia that also has the highest LUC-factor. See Figure 20 for contribution of BD from both LU and LUC, and how it is divided among feed ingredients.

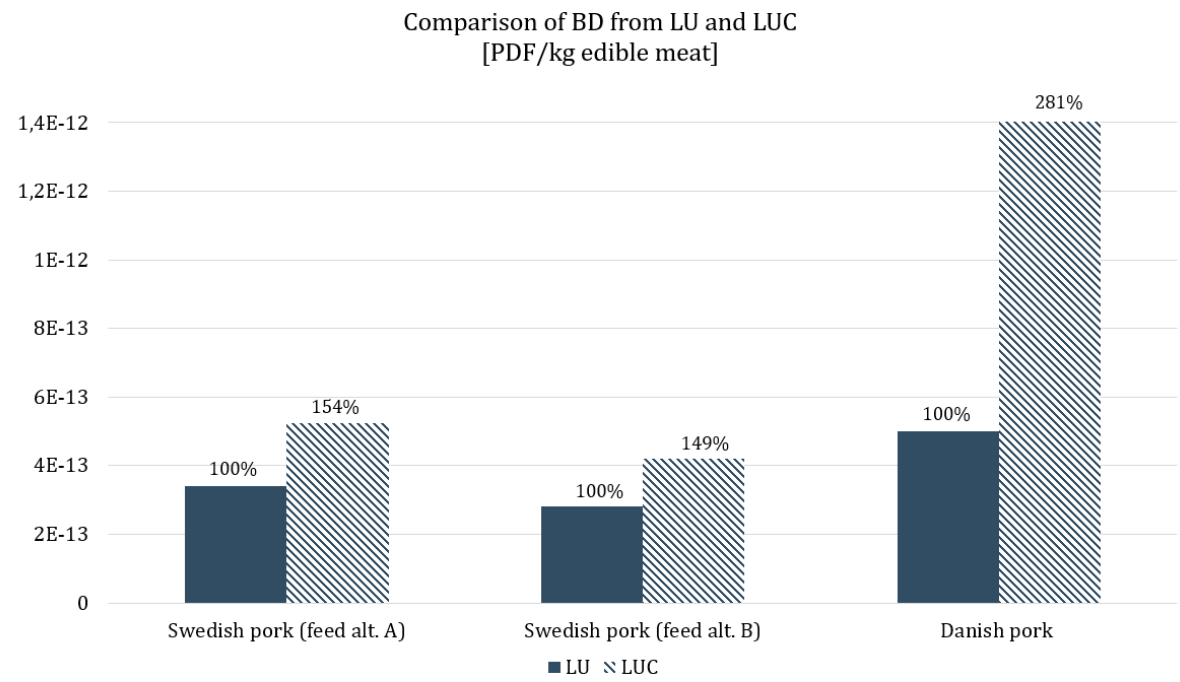


Figure 19. Biodiversity damage from land occupation, LU and land transformation, LUC. The percentages show normalized values for each product

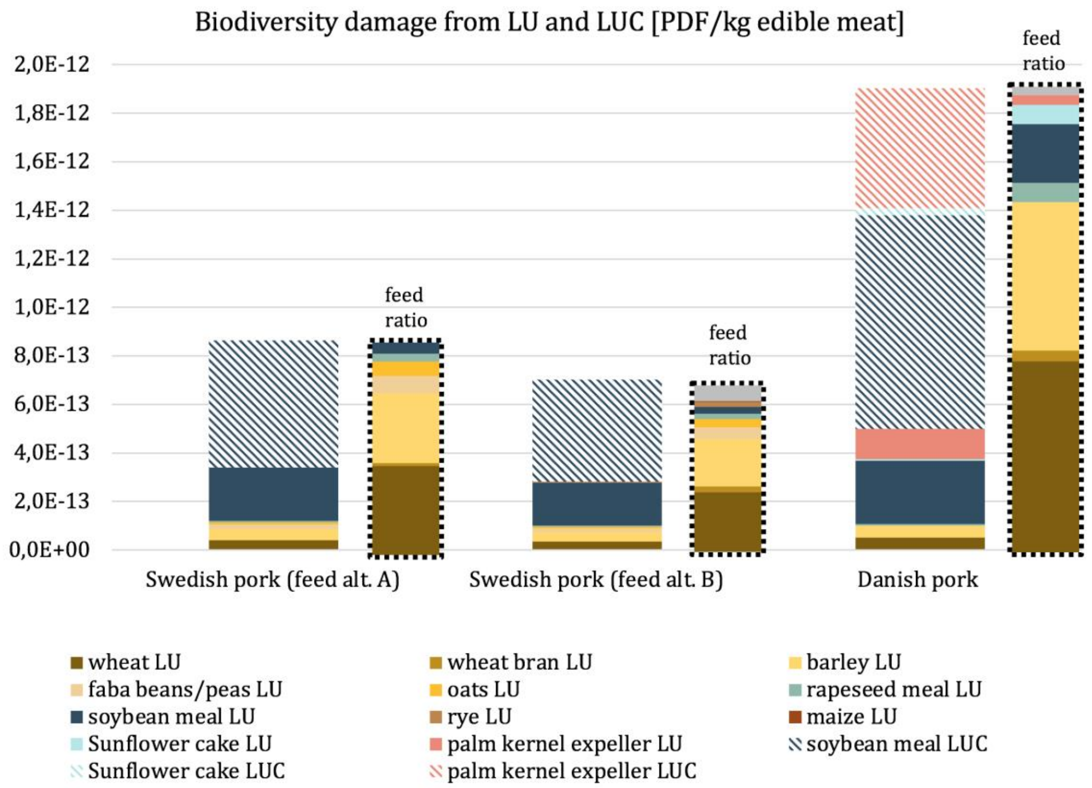


Figure 20. Biodiversity damage contribution of the feed ingredients (left bars) and visual comparison of each product's respective feed ratio in percentage DM (right bars, dashed borders). NB, the scale represents the BD contribution of the feed ingredients (left bars)

These results indicate that the question of how and whether to include LUC in LCA on food products deserves to be highlighted. It also indicates the importance of accuracy in terms of country of origin of the feed ingredients, as well as the feed ratio. Altogether, three methodological choices were identified giving three different results as illustrated in Figure 21. Either one only includes the occupational impact, (the bottom bars, LU), only the transformational impact, LUC (the bars in the middle, LUC), or including both (the bars at the top (LU + LUC). By comparing the results of these three methodological choices, one can see that the summed impact, i.e., LU+LUC, from the Danish pork is almost 7 times higher compared to the Swedish pork (feed alt. B) when only including LU. The percentage shows how the impact varies compared to the Swedish pork (feed alt. B) for LU.

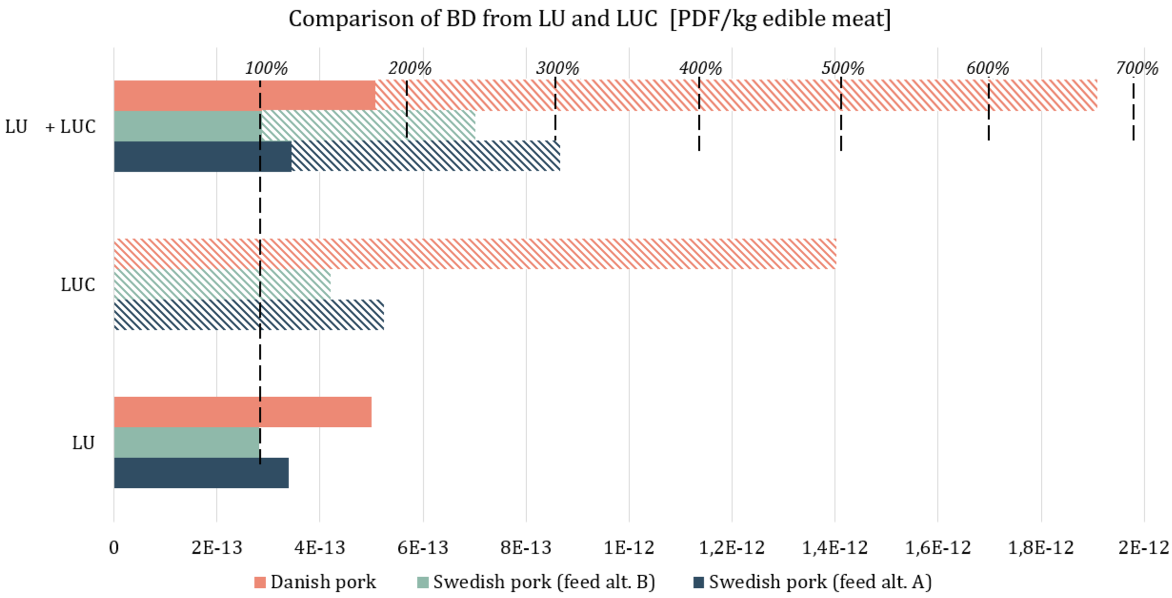


Figure 21. Biodiversity damage comparison of three land use intervention approaches, (1) an occupational impact approach, LU, (2) a transformational impact approach, LUC, and (3) an approach which adds both the occupational, LU and transformational impact, LUC.

No previous studies were found using the transformational CF of the CB method. Thus, it was not possible to verify the results or the applied allocation method for transformational impact.

4.3 Spatial resolution

For the LCA results presented in the previous sections, 4.1 and 4.2, country level CFs were used. The coming section presents how results can vary depending on the methodological choice of applying the higher geographical resolution CFs provided by the method, the ecoregion CFs. The section also presents other studies' approaches to the ecoregional CFs.

Varying results were found when comparing the two spatial resolutions in the CB method, i.e., comparison at country or ecoregional level. Figure 22 shows an overview of the differences between the two approaches, and for the three pork products. For both the Swedish pork products, the ecoregion approach gives a higher BD and was found to be around 109-110% of the country approach. The opposite result was found for the Danish pork, where the BD using the ecoregional approach was around 83% of the country approach. The order of the impact between the products is however the same, with highest BD for the Danish pork, followed by the Swedish pork with feed alternatives A and B.

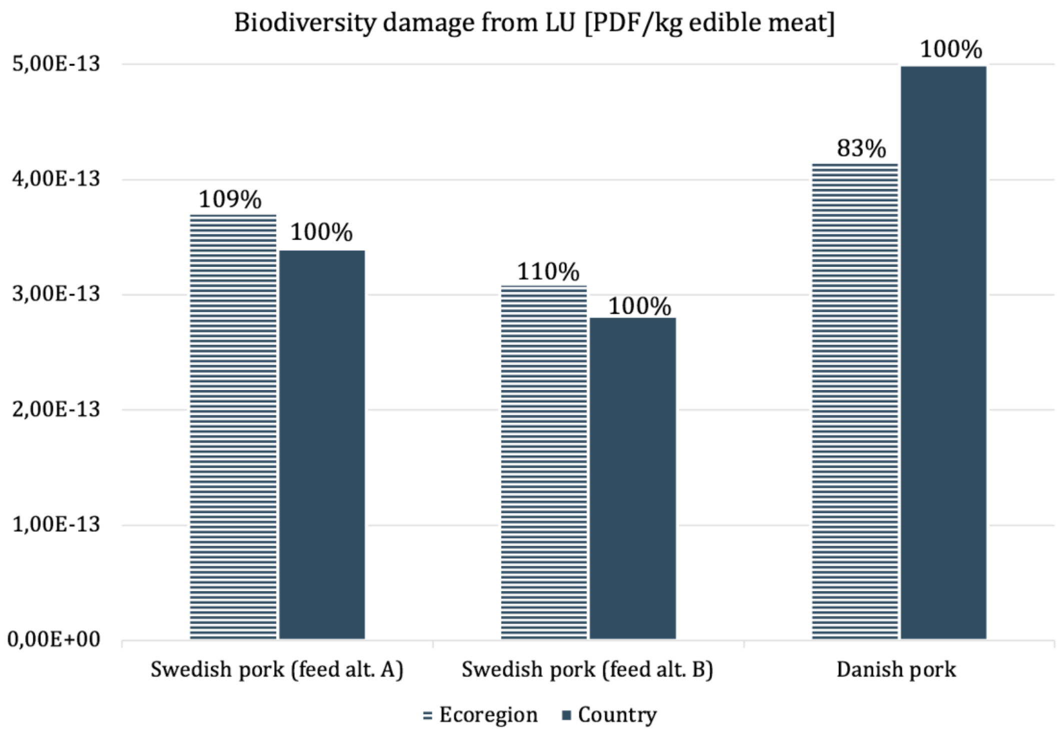


Figure 22. Biodiversity damage comparison between the ecoregional and country approach. The percentages show normalized values for each product.

The difference between the impact using ecoregional approach and country approach is a result of the feed crops included and their country of origin. Figure 23 shows the BD for the different feed types where all ingredients, except soybean meal and palm kernel expeller, receive a higher impact using the ecoregion approach. For soybean meal and palm oil, the opposite applies. The differences between the Swedish and Danish pork are therefore mainly found in the country of origin of soybean meal (Brazil and Argentina respectively) and the inclusion of palm kernel expeller in the Danish pig diet. The ecoregions included in the study for the Brazilian soybean meal (Swedish pork) show similar impact as the country aggregated, whereas the Argentinian, included in the Danish pig diet, account for higher impact using country approach. The country approach is also higher for the palm kernel expeller from Indonesia.

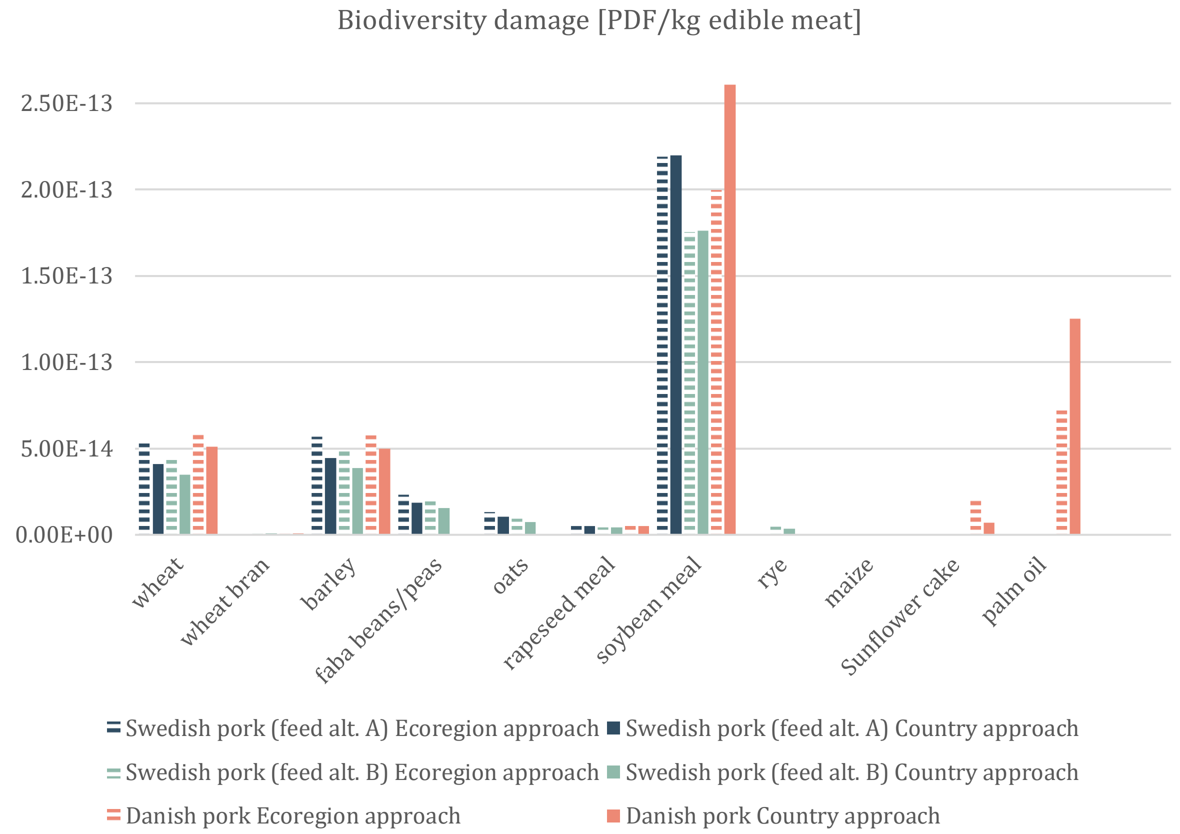


Figure 23. Biodiversity damage comparison between the ecoregional and country approach for the feed types included in the study. NB, the scale represents the impact per kg edible meat, and thus the biodiversity damage per kilo feed type is not displayed in this figure

The differences in impact between ecoregion and country approach is simply a result of that the impact is allocated where the harvested production takes place. Figure 24 illustrates the ecoregion share of harvested production for the Swedish feed crops and where the ecoregions are located. A large share of the crops was harvested in the southern two ecoregions, Baltic mixed forests and Sarmatic mixed forests, which also have a higher CF compared to the northern Scandinavian and Russian taiga. The northern ecoregion is however overrepresented in the country CF due to the area-share aggregation method used to develop the country CF in the CB method. With the same yield, the impact would therefore be higher by the ecoregion approach, but the yield is lower in the northern ecoregion compared to the southern ecoregions. Therefore, despite the lower CF of the northern ecoregion, barley, faba beans and oats, generated higher BD as a result of the lower yield. As such, the differences between ecoregion and country approach, for the Swedish crops, was relatively low because of the differences in yield and CFs were balancing each other out.

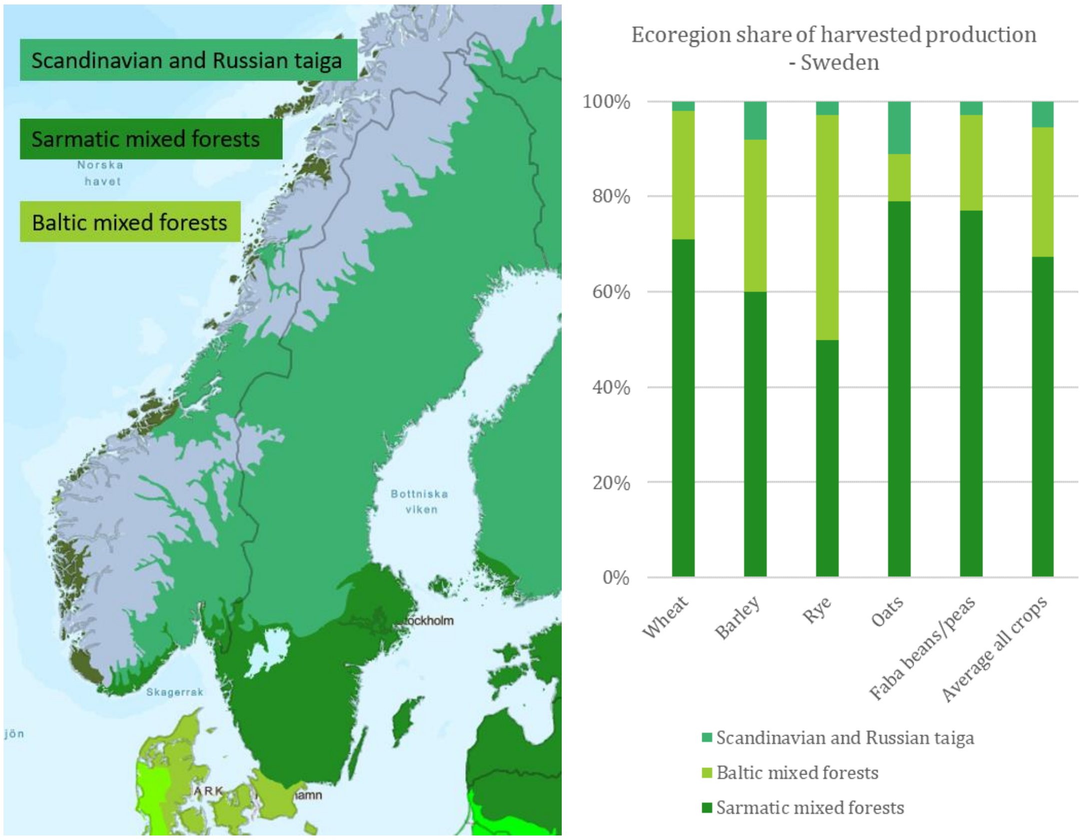


Figure 24. Left: The three Swedish ecoregions included in the study. Right: Bar chart showing the Swedish ecoregions' share of harvested production for wheat, barley, rye, oats, faba beans/peas and average all crops.

The variation within a country between the ecoregion CF and country CF included in the study shows large differences between countries. These differences are displayed in Table 5, which shows the lowest and highest ecoregion CF (of the ecoregions included in the study) normalized to the country CF. Russia and Brazil stands for the highest variation, and Denmark, Argentina, and Sweden for the lowest. However, only 2% of the harvested production of sunflower oil in Russia were allocated to the ecoregion with the highest CF. The harvested production allocated to the Brazilian ecoregion with the highest CF was 1%. Despite the minor production allocated, the differences in impact from Russian sunflower cake using ecoregion CF were 3 times higher than using the country approach (see Figure 23). For the Brazilian soybean meal, however, the differences in biodiversity impact by country and ecoregional approach were found to be negligible despite a large variation among the CFs of ecoregions within the country.

Table 5. Variation of ecoregional CFs relative to country CF. Only ecoregions included in the study are included

Country	Lowest ecoregion CF	Highest ecoregion CF
Russia	83%	1812%
Brazil	44%	982%
Indonesia	39%	182%
Poland	73%	224%
Denmark	88%	135%
Argentine	56%	89%
Sweden	90%	142%

The area-aggregated country CF fails to reflect the different land types as it does not consider whether a cropland (or e.g., urban area) exists in an ecoregion, the size of the land type or the ecoregion productivity. Cultivation and harvesting might only take place in a few ecoregions (composing only a small share of the country's total area) within a country, something also pointed out by Chaudhary et al. (2015). Therefore, applying higher geographical resolution (i.e., using the ecoregion approach) might be more favourable for larger countries with large differences in biodiversity potential between ecoregions, compared to smaller countries where these differences are not as large. Two countries were in this case study standing out in terms of ecoregion variation, Russia and Brazil, which could either be a consequence of that some ecoregions in e.g., Indonesia and Argentina being left out or that Russia and Brazil hold highly fragile ecoregions compared to the other countries included. Another reason could be the translation of provincial production to ecoregion production where ecoregions that do not necessarily have production were included. Either way, it clearly marks the importance of knowing where the harvested production takes place as the impact on biodiversity differs depending on the approach chosen.

Both country and ecoregion approaches have been found in previous studies applying the CB method. Some studies found applying the country approach, e.g., Karlsson Potter & Röös (2021) and Moberg et al. (2020) was of broad character whereas the studies applying the ecoregion approach, e.g., Tidåker et al. (2021), Lucas et al. (2021), Gaudreault et al. (2020), Hayashi (2020), Chaudhary & Tremorin (2020) and Møller et al. (2022) were of another character, for example a comparative LCA of a few food products. This could be an indication of that the country-CFs are more feasible for broader types of LCA studies. Chaudhary & Tremorin (2020) re-calculated the ecoregional CFs to provincial CFs by area-weighting, which therefore also includes ecoregions not necessarily having cultivation. Møller et al. (2022) estimated the share of cropland in each Norwegian ecoregion from national accounts on counties[9] and distributed the crop production and impact accordingly. The latter method therefore assumes all crops had similar distribution and yield, which potentially is a result of data trade-offs to make the study more feasible.

[9] NIBIO Area barometer

5 Discussion

The assessment of the CB method's usability for estimating the impact on biodiversity from food production was done through an LCA case study of pork. This section contains discussions on the chosen method for evaluating the CB method and identifies strengths and weaknesses of the assessment.

5.1 Aspects of biodiversity captured

Despite various ways of measuring biodiversity exists, there is yet no standard of which drivers of biodiversity loss to account for and how to measure their impact, as pointed out by Bracy Knight et al. (2020). The CB method assesses the direct drivers of LU and LUC, and the indicator of species richness, and it is therefore important to remember that it represents those specific drivers for that indicator. Claiming that the CB method represents the *impact on biodiversity* could depend on the recipient, and for some cases, be misleading. However, as the CB method is a relatively feasible method, the question is if the measure it provides of biodiversity damage is good enough. For example, there might be a correlation between species richness and other biodiversity elements, which would mean that the results do capture a broader spectrum of biodiversity. Answers to this were not provided by this study but will potentially be proven by future research. However, the chances of applying a more complex method compared to the CB method could be lower as an LCIA method needs to be feasible enough to be commonly used, meanwhile, the results need to be detailed enough to be relevant.

The relative importance of the land use parameter between different food products was highlighted through the study by Crenna et al. (2019), presented under section 4.1.2. The contribution from the land use parameter to the endpoint damage varied from 1% to around 90% of the total impact, but between 40-60% for most of the products included in that study. The CB method could for that reason possibly give a fairer representation of biodiversity impact for products where land use interventions are the main driver, or for comparative assessments of products having similar share of land use contributions. To note is also that, other drivers than what was included in the ReCiPe 2016 method were disregarded, such as invasive species and direct exploitation. Including more LCIA methods for biodiversity damage that includes other drivers, could have given a more diverse indication of what aspects of biodiversity the CB method captures in relation to food production and how well it performs as a proxy for biodiversity loss. Other methods for estimating the impact on biodiversity should have been assessed and compared to the CB method, such as those on soil quality or Environmental DNA.

5.2 Land occupation and land transformation

The results obtained by using the chosen LUC allocation method (see section 4.2), indicate that most of the BD from LUC stems from regions shown to also have the highest biodiversity decline according to the WWF Living Planet Index (WWF, 2020). When including both LU and LUC, the contribution to BD is higher from crops originating from these areas, compared to if one only assesses LU. Since biodiversity is largely driven by conversion of natural habitats, it would be misleading to not include any aspects of LUC.

The transformational CFs for cropland were chosen to calculate the BD from LUC. The CB method do not distinct between what land use type is transformed *from,* and CFs are therefore the same even if the transformed land was forest or grassland. This is the case for all CFs, regardless of what land type the area is being transformed into. To not be able to make distinctions between an impact from e.g., deforestation or grassland conversion for cultivation of crops, should be noted for future users of the CB method.

The CFs for transformational impact are based on both the CFs for occupation and the regeneration times (see Equation 3). However, the indicators used to develop the occupational CFs and regeneration times are different, being based on species richness and the Sørensen index respectively. The Sørensen index is not currently being used in LCA as an indicator of impact, i.e., to develop characterization factors. It is interesting that these two parameters of the CB method are based on different indicators, even though it is not known what consequences this has. It surely has some effect, and even though it is not covered in this study, it could be interesting to explore how this affects the results in future studies.

Since the regeneration times is the only factor that distinguishes occupational and transformational CFs, the differences in BD from LUC and BD from LU for a crop grown in a boreal ecoregion (where the regeneration times are longer compared to a tropical one) will therefore be larger compared to a crop in a tropical region. If the regeneration times would have been based on an indicator that captures more complex change in community structure[10], they would in general be up to ten times longer (M. Curran, personal communication, May 10, 2022). They would be faster in the palearctic realm than in Australasia[11] or the tropics, which is the opposite for the regeneration times that are currently used. As for the results of the LCA, the difference between occupational and transformational BD would be even greater if the regeneration times would be based on an indicator such as the Morisita-Horn Index, since they would be longer in general but also since they would be longer in tropical regions than in boreal ones, this indicates that the transformational impact is even more important to include when measuring BD.

A harmonized way of including and allocating direct land transformation using the CB method was not found. Additionally, no previous studies, that applies the CB method, were found including the impact from LUC. Nor did the reviewed studies include discussions on how the calculation of biodiversity impact could be applied using the recommendations of the amortization period of 20 years. The choice of including land transformation was, despite the uncertainties, found superior to excluding it, as examining the inclusion of transformational impacts was one of the three focus areas of this study. Therefore, the result of the LUC reflects the applied allocation method of the transformational CFs provided by the CB method.

5.3 Data limitations and spatial resolution

A few challenges were found when collecting and compiling the life cycle inventory for the LCA. One example was that the animal feed composition varied between sources, which was especially crucial for the share of ingredients with large contribution to the total BD, such as soy and palm. In this study the ratios of soybean varied between 4,2% for the Swedish pork (alt B) and 12,6% for the Danish pork, of the total feed ingredients, whereas the soy share in Moberg et al. (2019) was set to 1,6% (feed composition not included in the study). This generated a contribution of the BD from the soy being from 52% to 62% of the impact from LU, and from 60% to 85% if LUC also is included. If other feed composition alternatives had been included in the study, one would probably have seen other results. However, the method succeeds in highlighting the differences between different pork products, much due to the different CF between the countries of origin.

[10] The Morisita-Horn index is used to measure the level of similarity or difference between two data sets and is more sensitive to complex change in community structure than for example the Sørensen index (M. Curran, personal communication, May 10, 2022).
[11] A region comprising of Australia, New Zealand, and neighboring islands

The sources used for animal feed are relatively new for the Swedish pork (alt. A from 2020 and alt. B from 2021), however for the Danish pork the source is from 2010.

According to O. Karlsson (personal communication, February 28, 2022), the feed can change significantly over a relatively short time period and the feed used today is not the same as twenty years ago. One example is that the share of protein has decreased due to an increase in price. This might be an explanation of why the percentage of soybean is much larger for the Danish pork, (12,6%) compared to the Swedish alternatives (4,2% and 5,2%). One could for example imagine that the results for the Danish pork hence would be different if based on more newly assessed feed ratios. The results in this study should therefore be interpreted with caution, and it should be stressed that the main purpose is to highlight the applicability of the CB method. O. Karlsson (personal communication, February 28, 2022) also highlights the fact that the feed composition may not be the same in the future, with protein from grass or the inclusion of insects in the diet. The BD results presented are thus only a reflection of the feed composition and their country of origin and will most certainly need to be updated with changing agricultural production practices and technological developments.

A second challenge regarding data handling was the waste and losses along the supply chain. In this study, harvesting losses were included through national statistics, with additional losses during post-processes according to the compiled data in Gustavsson et al. (2011). Additionally, the pork production losses were assumed to be included in the conversion factors. As this waste data is presented per sub-continents and process step, a lot of information is needed on the downstream supply chain of the commodity being assessed. The downstream processes after harvesting are not necessarily carried out in the same region as the country of origin of the crop. Surely, other results would have been generated if other waste factors had been used. This is especially true if the system boundary had included the use phase, as the use phase waste of meat in Gustavsson et al. (2011) is 11%. Read et al. (2022) found that halving food waste in the US would have a larger effect on biodiversity decline compared to shifting diets (in the scenario and diets modelled). However, as all products are assumed to be consumed in Sweden, the relative difference had been the same if the use phase would have been included.

A third aspect that needs to be emphasized in regard to the data handling is that the inventory compiled from national accounts and statistics on harvested production and yield data does not always consider the cultivated area nor if multi-cropping is included. As multi-cropping was excluded in this study and differences between continents are large, the relative contribution of BD from feed crops from Latin America and Asia compared to those from Europe needs to be taken with caution. This is due to a higher potential yield increase in the former areas according to the multi-cropping conversion factors by Röös et al. (2017), and further that a yield increase has a direct effect on the result. If these multi-cropping conversion factors would have been applied in this study, they would have generated a potential yield increase factor of around 1.7 for the soybean meal leading to a decrease of its BD contribution, whereas being negligible for the European crops.

5.3.1 Applying the ecoregional approach

The fact that the availability of data on ecoregion level was found to be a major limitation of the ecoregional approach, favors the use of the country CF approach. For example, FAOSTAT was found to be reporting on national levels, and TRASE, an initiative mapping global supply chains with the purpose of highlighting deforestation, was found reporting on national, regional and in some cases biome level. As these, or the national statistics and accounts used in this study, were not reporting on ecoregion level, a recalculation of the collected data was done from provincial harvesting production to ecoregional production using area-weighting. This brings many uncertainties, one being the inclusion of ecoregions not necessarily having crop production. The fact that an ecoregion was located in a province with high production, led to those ecoregions not necessarily having cropland included in the calculations as actually having harvested production. This was the case for example for the ecoregions Serra do Mar coastal forests in Brazil and Carpathian Mountain Conifer forests in Poland.

In this study, ecoregion specific yields were used, which has the potential of lowering uncertainty compared to distributing national yield statistics equally. The fact that other studies (Chaudhary & Tremorin (2020) and Møller et al. (2022)) apply similar approaches as in this study, highlights the need and limitations of data availability of crop production on ecoregion level. One should also carefully interpret the results as the harvesting production data is based on different nations' statistical databases that might have different ways of collecting and reporting data to governmental institutions. In the study, the choice of estimating ecoregion productivity was also based on national accounts and for the nations outside EU it included 60% of the total harvested production. Doing so made the study more feasible but might have led to the exclusion of relevant ecoregions for biodiversity loss and the comparison between ecoregions might have looked different if all ecoregions had been included.

Instead of using national statistics and basing the ecoregion productivity through provincial productivity, one could have used GIS software to map ecoregion productivity of the feed ingredients. In that only the ecoregions having crop production would have been included. This would have generated a more accurate comparison of ecoregional and country approach and most likely led to other results and conclusions.

6 Conclusions

This study explored the applicability of the CB method for assessing biodiversity impact from food, based on the three focus areas: (1) how well the indicator acts as a proxy for biodiversity loss, (2) applying land transformation, and (3) the spatial resolution. The purpose was to highlight important factors demonstrating the usefulness of the method in the context of food production, as the intended outcome was to provide valuable insight to the development of a biodiversity database for food.

As for the first area and the question of how the method acts as a proxy for biodiversity loss, no uniform answer was found as there are many ways to define and measure biodiversity. The CB method presents biodiversity impact based on the indicator of species richness, projected by the driver of land use interventions, which is the largest driver of biodiversity loss. In general, it was not clearly presented in the CB method what levels of biodiversity were covered and not. It was found that the type of indicator for developing the CFs and regeneration times are different, being based on species richness and Sørensen index respectively. Naturally, the consequences are that there are aspects of biodiversity that are not covered by the method.

The CB method was also found to have a narrow perspective on biodiversity impact from agricultural systems due to only including one land use type, in three intensity levels, for the cultivation of crops. A sparse differentiation between agricultural systems could lead to misleading comparisons of commodities cultivated on different land use types and under different agricultural practices. The method does therefore not manage to generate results that could potentially incentivize improvements in agricultural practices. The type of product chosen for this case study was pork, and the result showed that firstly, using the correct feed composition, and secondly using the right origin of the feed ingredients is crucial, as these are to parameters that have a large impact of the total BD.

The conclusions of the first focus area are also relevant for the second focus area, the choice of applying an ecoregional or a country approach. As the method does not yet cover land use types and taxa groups detailed enough to provide a more nuanced comparison between food products under different agricultural systems, the country approach might in some cases be enough. Ecoregional CF can in some cases however provide a more detailed result compared to using country CF but for that, more detailed inventory data is required. As production data on crops were presented on national or provincial level, the collected data had to preprocessed, which might lower the feasibility of the ecoregion CF.

To our knowledge, no previous studies had used the CB method to calculate a transformational impact. This study has highlighted the need for a harmonized way of applying the transformational CF. The applied allocation method for the transformational impact have demonstrated that its impact on biodiversity is substantial and should be taken into consideration. The study has also shown that the impacts from occupation and transformation can be added, yet that they could be presented separately for transparency. To be noted, is that the results presented from the LCA case study reflect our interpretation of how LUC can be applied using the CB method. For future research it is recommended that studies measure the transformational impact, but also compare the CB method to other methods.

To summarize, despite drawbacks of the CB method highlighted in this study, the method serves a purpose by capturing potential differences between food products, yet one need to be aware of the background of the indicator and the method, but also to remember that the method should continuously be developed.

References

Basset-Mens, C., & van der Werf, H. M. G. (2005). Scenario-based environmental assessment of farming systems: The case of pig production in France. *Agriculture, Ecosystems and Environment*, *105*(1–2), 127–144. https://doi.org/10.1016/j.agee.2004.05.007

Benton, T. G., Bieg, C., Harwatt, H., Pudasaini, R., & Wellesley, L. (2021). *Food system impacts on biodiversity loss Three levers for food system transformation in support of nature*.

Blonk Consultants. (2021). *LUC Impact Tool*.

Cederberg, Christel. (2009). *Greenhouse gas emissions from Swedish production of meat, milk and eggs 1990 and 2005*. SIK - Institutet för livsmedel och bioteknik.

Chaudhary, A., & Brooks, T. M. (2018). Land Use Intensity-Specific Global Characterization Factors to Assess Product Biodiversity Footprints. *Environmental Science and Technology*, *52*(9), 5094–5104. https://doi.org/10.1021/acs.est.7b05570

Chaudhary, A., & Tremorin, D. (2020). Nutritional and environmental sustainability of lentil reformulated beef burger. *Sustainability (Switzerland)*, *12*(17). https://doi.org/10.3390/SU12176712

Chaudhary, A., Verones, F., de Baan, L., & Hellweg, S. (2015). Quantifying Land Use Impacts on Biodiversity: Combining Species-Area Models and Vulnerability Indicators. *Environmental Science and Technology*, *49*(16), 9987–9995. https://doi.org/10.1021/acs.est.5b02507

Crenna, E., Marques, A., la Notte, A., & Sala, S. (2020). Biodiversity Assessment of Value Chains: State of the Art and Emerging Challenges. In *Environmental Science and Technology* (Vol. 54, Issue 16, pp. 9715–9728). American Chemical Society. https://doi.org/10.1021/acs.est.9b05153

Crenna, E., Sinkko, T., & Sala, S. (2019). Biodiversity impacts due to food consumption in Europe. *Journal of Cleaner Production*, *227*, 378–391. https://doi.org/10.1016/j.jclepro.2019.04.054

Curran, M., Hellweg, S., & Beck, J. (2014). Is there any empirical support for biodiversity offset policy? In *Ecological Applications* (Vol. 24, Issue 4).

de Baan, L., Alkemade, R., & Koellner, T. (2013a). Land use impacts on biodiversity in LCA: A global approach. *International Journal of Life Cycle Assessment*, *18*(6), 1216–1230. https://doi.org/10.1007/s11367-012-0412-0

de Baan, L., Curran, M., Rondinini, C., Visconti, P., Hellweg, S., & Koellner, T. (2015). High-resolution assessment of land use impacts on biodiversity in life cycle assessment using species habitat suitability models. *Environmental Science and Technology*, *49*(4), 2237–2244. https://doi.org/10.1021/es504380t

de Baan, L., Mutel, C. L., Curran, M., Hellweg, S., & Koellner, T. (2013b). Land use in life cycle assessment: Global characterization factors based on regional and global potential species extinction. *Environmental Science and Technology*, *47*(16), 9281–9290. https://doi.org/10.1021/es400592q

Gabel, V. M., Meier, M. S., Köpke, U., & Stolze, M. (2016). The challenges of including impacts on biodiversity in agricultural life cycle assessments. In *Journal of Environmental Management* (Vol. 181, pp. 249–260). Academic Press. https://doi.org/10.1016/j.jenvman.2016.06.030

Gustavsson, Jenny., Food and Agriculture Organization of the United Nations., & ASME/Pacific Rim Technical Conference and Exhibition on Integration and Packaging of MEMS, N. (2011). *Global food losses and food waste : extent, causes and prevention*. 29.

Huijbregts, M. A. J., Steinmann, Z. J. N., Elshout, P. M. F., Stam, G., Verones, F., Vieira, M. D. M., Hollander, A., Zijp, M., & Zelm, R. (2017). *ReCiPe 2016 v1.1 A harmonized life cycle* *impact assessment method at midpoint and endpoint level Report I: Characterization*. www.rivm.nl/en

IPBES. (2019). *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (S. M. Díaz, J. Settele, E. Brondízio, H. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. Brauman, S. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. Subramanian, G. Midgley, P. Miloslavich, Z. Molnár, D. Obura, … C. Zayas, Eds.). https://ri.conicet.gov.ar/handle/11336/116171

Jolliet, O., Antón, A., Boulay, A. M., Cherubini, F., Fantke, P., Levasseur, A., McKone, T. E., Michelsen, O., Milà i Canals, L., Motoshita, M., Pfister, S., Verones, F., Vigon, B., & Frischknecht, R. (2018). Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use. *International Journal of Life Cycle Assessment*, *23*(11), 2189–2207. https://doi.org/10.1007/s11367-018-1443-y

Karlsson Potter, H., & Röös, E. (2021). Multi-criteria evaluation of plant-based foods –use of environmental footprint and LCA data for consumer guidance. *Journal of Cleaner Production*, *280*. https://doi.org/10.1016/j.jclepro.2020.124721

Kier, G., Mutke, J., Dinerstein, E., Ricketts, T. H., Küper, W., Kreft, H., & Barthlott, W. (2005). Global patterns of plant diversity and floristic knowledge. *Journal of Biogeography*, *32*(7), 1107–1116. https://doi.org/10.1111/j.1365-2699.2005.01272.x

Koellner, T., Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., Canals, L. M., Saad, R., Souza, D. M., & Müller-Wenk, R. (2013). UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *International Journal of Life Cycle Assessment*, *18*(6), 1188–1202. https://doi.org/10.1007/s11367-013-0579-z

Landquist, B., Woodhouse, A., Axel-Nilsson, M., Sonesson, U., Elmquist, H., Velander, K., Karlsson, O., Eriksson, I., Åberg, M., & Elander, J. (2020). *Uppdaterad och utökad livscykelanalys av svensk grisproduktion*. RISE Bioekonomi och hälsa jordbruk och livsmedel. .

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R. F., Gaillard, G., Michelsen, O., Müller-Wenk, R., & Rydgren, B. (2007). Key elements in a framework for land use impact assessment within LCA. In *International Journal of Life Cycle Assessment* (Vol. 12, Issue 1, pp. 5–15). https://doi.org/10.1065/lca2006.05.250

Milà i Canals, L., Rigarlsford, G., & Sim, S. (2013). Land use impact assessment of margarine. *International Journal of Life Cycle Assessment*, *18*(6), 1265–1277. https://doi.org/10.1007/s11367-012-0380-4

Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being : synthesis*. Island Press, Washington, DC.

Moberg, E., Walker Andersson, M., Säll, S., Hansson, P. A., & Röös, E. (2019). Determining the climate impact of food for use in a climate tax—design of a consistent and transparent model. *International Journal of Life Cycle Assessment*, *24*(9), 1715–1728. https://doi.org/10.1007/s11367-019-01597-8

Møller, H., Samsonstuen, S., Øverland, M., Modahl, I. S., & Olsen, H. F. (2022). Local non-food yeast protein in pig production–environmental impacts and land use efficiency. *Livestock Science*, *260*, 104925. https://doi.org/10.1016/j.livsci.2022.104925

Nguyen, T. L. T., Hermansen, J. E., & Mogensen, L. (2010). *Monitoring of stored and processed agricultural biomass by remote wireless technology*. Faculty of Agricultural Sciences, Aarhus University.

Nilsson, E. (2019). Biologisk mångfad. In *Det här är biologi*. Svenska Nationalcommitén för biologisk mångfald. https://soundcloud.com/user-620168986/biologisk-mangfald?fbclid=IwAR1m5ZETWc2tE5Zf7ufkqRrucaqQ8fHvOZdItdksRxd-xrtbnrC9Xt91g7A

Noss, R. F. (1990). *Indicators for Monitoring Biodiversity: A Hierarchical Approach* (Vol. 4, Issue 4).
Olsson, D. (2001). *Terrestrial Ecoregions of the World. A new map of life on earth*.
Pereira, H. M., Ferrier, S., Walters, M., Geller, G. N., Jongman, R. H. G., Scholes, R. J., Bruford, M. W., Brummitt, N., Butchart, S. H. M., Cardoso, A. C., Coops, N. C., Dulloo, E., Faith, D. P., Freyhof, J., Gregory, R. D., Heip, C., Höft, R., Hurtt, G., Jetz, W., … Wegmann, M. (2013). Essential biodiversity variables. In *Science* (Vol. 339, Issue 6117, pp. 277–278). American Association for the Advancement of Science. https://doi.org/10.1126/science.1229931
Persson, U. M., Henders, S., & Cederberg, C. (2014). A method for calculating a land-use change carbon footprint (LUC-CFP) for agricultural commodities - applications to Brazilian beef and soy, Indonesian palm oil. *Global Change Biology*, *20*(11), 3482–3491. https://doi.org/10.1111/gcb.12635
Read, Q. D., Hondula, K. L., & Muth, M. K. (2022). *Biodiversity effects of food system sustainability actions from farm to fork*. https://doi.org/10.1073/pnas
RISE. (n.d.-a). *RISE klimatdatabas för livsmedel*. A. Retrieved February 4, 2022, from https://www.ri.se/sv/vad-vi-gor/expertiser/rise-klimatdatabas-for-livsmedel
RISE. (n.d.-b). *Utveckling av en unik biodiversitetsdatabas för livsmedel*. B. Retrieved February 4, 2022, from https://www.ri.se/sv/vad-vi-gor/projekt/utveckling-av-en-unik-biodiversitetsdatabas-for-livsmedel
Röös, E., Bajželj, B., Smith, P., Patel, M., Little, D., & Garnett, T. (2017). Greedy or needy? Land use and climate impacts of food in 2050 under different livestock futures. *Global Environmental Change*, *47*, 1–12. https://doi.org/10.1016/j.gloenvcha.2017.09.001
Starr, C., Taggart, R., Evers, C., & Starr, L. (2010). *Which factors shape community structure?" in Biology: The Unity and Diversity of Life* (12th ed.). Yolanda Cossio.
Swedish Board of Agriculture. (2020). *Marknadsrapport grisköttt 2020*. https://jordbruksverket.se/download/18.430a570b1743edf421c5ddbe/1598881202835/Marknadsrapport-griskott-2020.pdf
United Nations, & Convention On Biological Diversity. (1992). *Article 2. Use of Terms*. 3.
van der Werf, H. M. G., Knudsen, M. T., & Cederberg, C. (2020). Towards better representation of organic agriculture in life cycle assessment. *Nature Sustainability*, *3*(6), 419–425. https://doi.org/10.1038/s41893-020-0489-6
Vrasdonk, E., Palme, U., & Lennartsson, T. (2019). Reference situations for biodiversity in life cycle assessments: conceptual bridging between LCA and conservation biology. *International Journal of Life Cycle Assessment*, *24*(9), 1631–1642. https://doi.org/10.1007/s11367-019-01594-x
Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., Jonell, M., Clark, M., Gordon, L. J., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J. A., de Vries, W., Majele Sibanda, L., … Murray, C. J. L. (2019). Food in the Anthropocene: the EAT-Lancet Commission on healthy diets from sustainable food systems. *Lancet (London, England)*, *393*(10170), 447–492. https://doi.org/10.1016/S0140-6736(18)31788-4
Wirsenius, S., Searchinger, T., Zionts, J., Peng, L., Beringer, T., & Dumas, P. (2020). *Comparing the life cycle greenhouse gas emissions of dairy and pork systems across countries using land-use carbon opportunity costs*.
WWF. (2020). *Living Planet Report 2020 : Bending the Curve of Biodiversity Loss* (R. E. Almond, M. Grooten, & T. Peterson, Eds.). WWF, Gland, Switzerland. https://livingplanet.panda.org
WWF Wildfinder database. (n.d.). Retrieved May 30, 2022, from https://www.worldwildlife.org/pages/wildfinder-database
Zira, S., Rydhmer, L., Ivarsson, E., Hoffmann, R., & Röös, E. (2021). A life cycle sustainability assessment of organic and conventional pork supply chains in Sweden. *Sustainable Production and Consumption*, *28*, 21–38. https://doi.org/10.1016/j.spc.2021.03.028

Appendix

The appendix is named “Appendix master's thesis report no. E2022.048“, and is available as a Microsoft Excel file, accessible on the Chalmers ODR website.

The appendix includes data and case-study calculations of the LCA process; crop production; feed composition; miscellaneous data; characterization factors; biodiversity damage from feed ingredients and the pork products; and pivot charts and tables.

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