Quantifying biodiversity impacts of bioenergy system

Latest developments in the scope of LCA

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Forword

Growing awareness of the biodiversity crisis has resulted in an urge from businesses and decision makers to address biodiversity loss to contribute to a transition to sustainable use of natural resources (see e.g. TNFD Reporting, Task Force for Nature-related Financial Disclosures). For this, there is a need for governance systems that assist and accelerate the integration of biodiversity considerations into decision-making. This Task 45 report leverages the capacity of an ongoing project, *Pathways towards an efficient alignment of the financial system with the needs of biodiversity (BIOPATH)*, funded by MISTRA - the Swedish Foundation for Strategic Environmental Research. BIOPATH involves systematically mapping and evaluating existing approaches (methods, metrics, and tools) currently employed to link decisions in industry and financial systems to biodiversity impacts, as mediated by land-use change. These mapped approaches will be evaluated from the perspective of how transparently, efficiently and credibly they capture biodiversity impacts. The evaluation outcomes form the basis for guidance on the use, refinement of existing, and development of new approaches, to inform decision-making affecting land-use about consequences for biodiversity and ecosystem services.

This report specifically focuses on how impacts on biodiversity of land-use consequences of biomass production associated with bioenergy systems can be quantified using life-cycle assessment (LCA). In addition to the input from the BIOPATH project, the report builds on dedicated knowledge gained at the workshop "Quantifying biodiversity impacts in bioenergy systems", 13th of June 2024, at the Royal Swedish Academy **of Agriculture and Forestry (KSLA), Stockholm, and on-line (hybrid event). The workshop included participants from the BIOPATH project, members from IEA Bioenergy Task 45, and international experts within the field. The workshop included presentations from: Laura Garcia Herrero (on behalf of Serenella Sala), Joint Research Centre, European Commission; Jan Paul Lindner, University of Augsburg, Germany; Eva Lindberg, Swedish University of Agricultural Sciences; Cecilia Akselsson, Lund University, Sweden; Anton Kvarnbäck, Lund University, Sweden, and Sara Kralmark, Kraftringen AB, Sweden.**

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Lund, Sweden

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The authors

1 Introduction

Life-cycle assessment (LCA) is a standardised method to assess the environmental impacts of products or services from a full life-cycle perspective. The LCA methodology has been under constant development since the mid-70’s. In the 90’s, the LCA-governing ISO 14040 was established, and several commercial software solutions were launched (Bjørn et al., 2018). Initially, LCA was primarily used as an analytical tool to aid decisions by companies relating to sustainability issues but is increasingly applied in policy tools, particularly for assessing greenhouse gas emissions. One notable example is the EU Renewable Energy Directive, RED, which uses threshold values calculated through LCA approaches to define sustainability criteria for the carbon footprint of bioenergy and biofuels (European Commission, 2018).

Currently sustainability criteria involving biodiversity are rarely based on LCA approaches. This paucity is due to the more complex nature of assessing biodiversity loss compared to greenhouse gas emissions. For example, sustainability criteria for biodiversity in the EU RED are based on guidelines regarding exempting land with assumed high biodiversity values (e.g., protected land, old-growth forests etc.) and not on comprehensive LCA approaches. However, the urge to consider biodiversity has resulted in a growing interest among companies and policy makers to measure biodiversity impacts adhering to the well-established methodology of LCA.

The overall aim of this report is to provide an overview of current progress in assessing biodiversity impacts and using assessment outcomes to inform decisions affecting land-use and land-use change, with a focus on bioenergy systems. We examine research relevant to enhancing and adapting LCA to incorporate biodiversity impact assessments in bioenergy systems and enable analyses of biodiversity - climate trade-offs. We also touch on what requirements need to be fulfilled regarding the availability of specific quantitative biodiversity data to do this.

2 Biodiversity in LCA

There has been extensive research and discussion about how to include biodiversity in LCA (see reviews by Lammerant et al., 2021; Crenna et al., 2020; UNEP-WCMC, 2020; Winter et al., 2017; Curran et al., 2016; 2011; Penman et al., 2010). For many reasons, it is more difficult to account for effects on biodiversity than for other environmental impact categories in LCA, such as global warming potential. Regarding impacts on biodiversity, important issues are, for example, the scale-dependency and non-additivity of impact assessments making it difficult to aggregate impact. Also, the multivariate character of biodiversity constraining the possibility to summarize impact in a simple metric, the lack of data to parameterize and validate models, and the difficulty of defining counterfactuals when ecosystems are dynamic (e.g. Crenna et al. 2020). Furthermore, many drivers of biodiversity loss are not represented in impact categories, and impact may not only reflect local but also landscape scale changes making consequences highly contextual. A specific difficulty compared with greenhouse gas emissions is that biodiversity loss is non-linearly related to the scale at which this is measured (King et al., 2021). However, there is already sufficient scientific understanding and models to allow current LCA-approaches for biodiversity to be improved to allow credible assessments (Curran et al. 2016).

In a survey, Crenna et al. (2020), distinguished between “within LCA” methods that could already be available in LCA software or developed in research and “beyond LCA” methods that used similar principles but without the formalism of LCA_methods (Fig. 1). For example, multiregional input-output (MRIO) analysis and biophysical accounting allowing impacts across sectors, regions, and entire economies to be captured. Crenna et al. (2020) also included methods focussing on the outcome of biodiversity/ecosystem function (ecosystem services) in this category. Many of the business application currently in use, falls in between the standard LCA and the beyond LCA methods, being tailored to the specific needs of business.

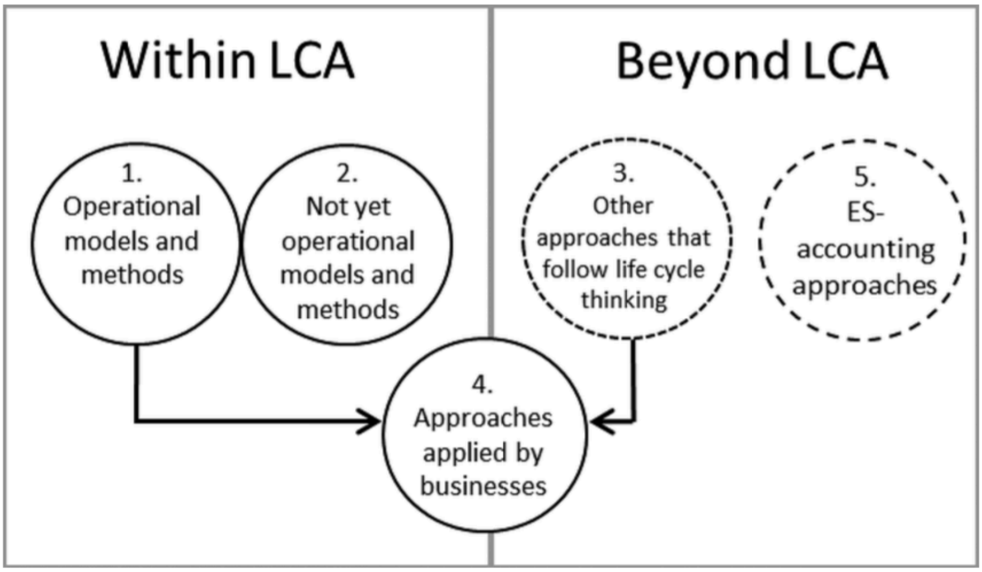

Figure 1. *Classification of the approaches selected by Crenna et al. (2020), concerning biodiversity impact assessment (see text for detailed explanation). The figure is published in Crenna et al. (2020).*

For biomass extraction for bioenergy production, the impact assessment may seem relatively straightforward, since it often mostly concerns land-use and management. However, while land-area requirement is straightforward to capture, management such as extracting a higher proportion of forest residues for bioenergy production may be harder to capture. Furthermore, the consequences of biomass extraction for bioenergy may depend on the larger landscape context, because biodiversity may not persist in highly fragmented landscapes. Even if these questions would be solved, how should impact of for example bioenergy plantations be assessed; as loss of local biodiversity, regional extinctions, global extinctions, or loss of biodiversity-based ecosystem services?

Several efforts have been made to develop a comprehensive and operational framework and indicators capable of supporting decision-making, particularly related to the need for sustainability disclosures in business. This is exemplified by work initiated by the the EU Business and Biodiversity Platform initiated by the European Commission, to facilitate the implementation of the EU Biodiversity Strategy (European Commissions, 2020). This initiative performed a critical review of methods and models for biodiversity impact assessment and their applicability in the LCA context (Damiani et al., 2023). Their conclusion is that there is a need to develop biodiversity impact assessment to expand coverage of drivers of biodiversity loss (i.e. not only cover land-use change, but also climate change, pollution, overexploitation of resources and the spread of invasive species), cover a broader range of ecosystems and taxonomic groups, include ecosystem services, and develop robust biodiversity indicators (Damiani et al., 2023).

The ongoing work by the European Commission also includes the development of the Product Environmental Footprint (PEF) with the aims in helping companies to calculate their environmental performance based on reliable, verifiable and comparable information (Herrero and Sala, 2024). Besides the results from the critical review of biodiversity impacts methods discussed above, this work also builds on stakeholder consultations, as well as testing and mapping coherence with Environmental Footprint (EF). Different options have been considered, such as selecting specific existing biodiversity models, calculation of a normalisation factor for the land use biodiversity impact category, and re-calibration of EF weights and to assign a weight to the land use biodiversity impact category. The goal is to select the most suitable method to assess the biodiversity impact in the EF and add biodiversity impact as the 17th impact category to the existing 16th impact categories in the existing EF (Herrero and Sala, 2024).

Another initiative is the Global Guidance on Environmental Life Cycle Impact Assessment Indicators (GLAM), part of the United Nations Environmental Programme umbrella Life Cycle Initiative is to enhance global consensus on environmental life cycle impact assessment indicators (LCI, 2024). GLAM method categorizes environmental impacts into main Areas of Protection, including ecosystem quality, which addresses potential damage to biodiversity, focusing on species vulnerability.

In terms of LCA for bioenergy use, the direct drivers of landcover change and land use on biodiversity are of special interest, since biomass feedstock production in agriculture and forestry often affect these. When land use is primarily for biomass production for bioenergy, the focus will be on land-cover change where natural ecosystems are contrasted with for the purpose suitable crops or tree plantations. Such productions generally have a strong effect on biodiversity. However, in reality the management practices used may vary in intensity and my modify existing land-uses to varying extents and the impact on biodiversity may be highly context dependent. In response to such complexities, the models used to assess land-use impacts on biodiversity in LCA have evolved to intensity of land uses and to consider where biomass is produced. The first issue is targeted by categorizing land use into different intensity levels or even using graded characterization of intensity in land use (Lindner et al. 2021). The second issue is targeted by using relative measures, which are dependent of broad geographical areas such as ecoregions.

LCA considers the relationship between quantity produced and the impact, in this case the bioenergy produced in relation to the biodiversity impact. Since product LCA is about environmental consequences in relation to the volumes produced, i.e. in this case the amount of biomass produced for bioenergy production, information on the productivity of different production methods and locations are also required. These are generally identified through inventory databases. For completeness, also the efficiency of converting different sources of biomass to bioenergy needs to be accounted for. Similarly, there is a need to identify the specific sites where bioenergy is produced, because the impact on biodiversity may vary. Such information is not available in easily accessible databases. All this information is important, for example since bioenergy production in high biodiversity areas may prove to be the best solution if also the production of bioenergy overcompensates for the impact. Furthermore, low-intensity bioenergy production systems with less per area impact on biodiversity may prove to have a higher per product biodiversity impact if productivity per area is low. Note that complications of this generally simplistic reasoning has been extensively covered in the land-sharing vs land-sparing debate (Fischer et al. 2014).

2.1 Conceptual framework of land use change models for impacts on biodiversity

One assumption in the land use framework in LCA is that one land occupation prevents another land occupation, this can be fundamentally attributed to land use classes being derived from land cover classifications (Koellner et al., 2013). This simply means that production of biomass raw materials for bioenergy will occur at the expense of the alternative land use consisting of natural ecosystems with, in most cases, more biodiversity. It considers land use change as a sequential process (see e.g. Lindeijer, 2000*)*. First, the framework distinguish the *transformation* of the original habitat to land used for bioenergy production from *occupation* by the current land use, or the land use of interest of the study. If the land use for bioenergy would cease, this land would hypothetically undergo *regeneration* to approach a more natural state, that may however not be perfectly similar to the original one. To allow quantification of biodiversity impacts, a *quality* is attributed to the different stages of the ecosystem, the original, the transformed and the regenerated. They way quality is measured in terms of indicators is not specified by the land use framework itself (Koellner et al., 2013). The land use framework combines the impact of the quality of the different land uses that accounts for the severity of the transformation, the duration of the occupation in relation to a reference state (Figure 2).

Arguably, a critical issue in the framework is the difference in quality between the original state and the hypothetical regenerated state, or the hypothetical biotic potential. Such differences may occur because of historical legacies that may take very long time to overcome. Quantification of the hypothetical quality of regenerated land may be a larger challenge than the quantification of the quality of original land used as the reference state.

The importance of the coherency between the choice of reference land use and carbon and biodiversity footprints has also been explored in, e.g. Soimakallio et at (2024) and Koponen et al., (2018). Soimakallio et al. (2024) conclude that if the aim of a footprint study is to assess the effects of land use, reference land use should describe dynamically non-use of land which could be natural regeneration or continuation of natural state. However, if the aim is to assess the effects of a decision to change land use or its management, such as producing bioenergy crops instead of food crops or start to harvest logging residues in forest operations, reference land use should describe dynamically alternative use of land expected without the studied decision. A resulting coherent reference land use could then be business as usual or projected future. Historic baseline or target reference land use may be useful in comparing the ecosystem value of the studied land use with a pre-set historical baseline or target (Soimakallio et al., 2024).

The *occupation impact* is a value attributed to the land use for occupying a piece of land temporarily that can be interpreted as a figure reflecting the relative biodiversity value of the land use compared with natural regeneration. The framework also suggests the quantification of permanent impacts, which is also dependent on this principle of relaxation. Permanent impacts are quantified by the difference between the quality of the *reference state*, which is the reference situation in the absence of the studied product system, or a human-free situation, and the quality of the steady state after relaxation. This permanent change of quality is referred to as *transformation impacts* (Koellner et al., 2013).

When it comes to the information flow of critical factors in land-use activities within the land-use framework (e.g. the Biodiversity Value Increment method presented by Lindner, 2024, see also Fig. 2), this can be exemplified by crop cultivation in agriculture. Here, information about structural elements, ground cover, crop rotation, weeds, Red Listed species, field size, tillage, fertilizer use and pesticide use, are of importance (Lindner, 2024). When it comes to the information flow in land-use activities in forestry, somewhat other critical parameters are of relevance which are exemplified later in Section 4. An important distinction regarding applied approaches to assess biodiversity impacts is if they focus on species richness per se, or if they focus on using indicators of biodiversity or ecosystem state (Myllyviita et al., 2019). Species richness approaches, which rely on species diversity data, are based on the assumption that land use change directly influence biodiversity. Ecosystem indicator approaches are, on the other hand, based on the assumption that ecosystem indicators, such as deadwood volume and tree species diversity, tree canopy cover and age of canopy trees, etc., indirectly indicate the biodiversity. According to Myllyviita et al. (2019), the two approaches based on different indicators are not fully coherent, but a limitation of the species richness approach is today the availability of reliable datasets. Thus, in the case study presented in Section 4, an ecosystem indicator approach has been selected since acquiring data on ecosystem indicators related to forest management is more direct and easier (see further discussion in Section 4.2).

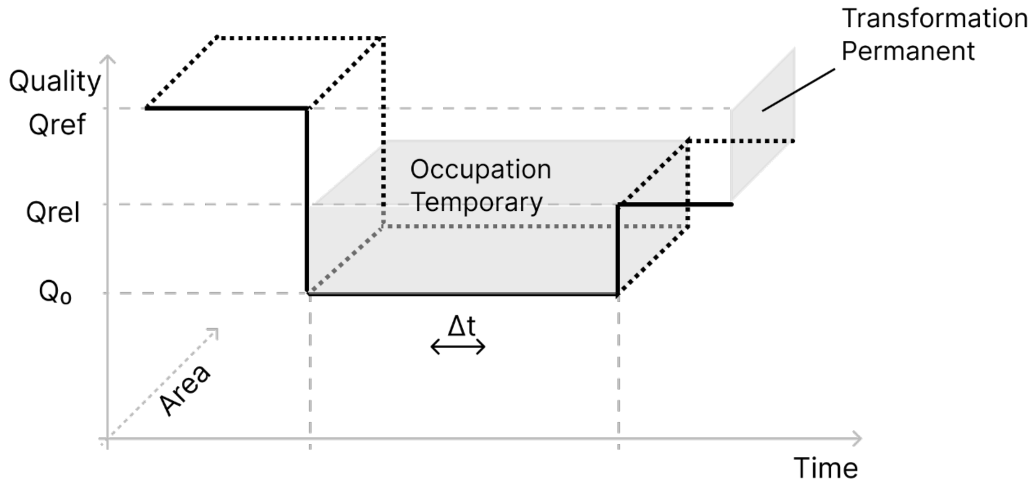

Figure 2. *The land use framework in LCA. Where Q represents quality of the land in the y axis.* Q_{ref} *is the reference state,* Q_{rel}*, which is the quality potentially achieved after re-naturalisation, and* Q_0 *the quality of the intervention. The impact of occupying a piece of land is modelled as the difference of the quality of that intervention* Q_0 *after relaxation* Q_{rel}*, which is the quality potentially achieved after re-naturalisation, this is referred to as occupation impacts. The difference of the natural state Qref and Qrel is a permanent quality loss, or transformation impact. Adapted from Lindner et al., (2014) after simplifications from Milà i Canals et al. (2007), Koellner et al. (2013)*

2.1 Important assumptions in LCA for biodiversity impact from land use and land use change

As concluded in the previous section, the impact of land use and land-use change in LCA are calculated in relative terms between the ideal state of the land and that of the intervention. This means that the impact is calculated in relation to a reference state. Although reference states are not something that is modified by users of LCA, they have a crucial impact on the results. Thus, selecting reference states in relation to the evaluation at hand is a contentious issue. Also, long-term climate changes may indirectly affect the reference state and the human land use history may vary between regions, which have relevance when defining timeframe and level of degradation (Nielsen et al., 2007) There are several references exploring this topic in a biodiversity context (see e.g. Curran et al., 2016; Vrasdonk et al., 2019, and references in these).

The reference states are not set by the framework as such, but defined by the method developer. The reference state often relates to land use not affected by human interventions (pre or post anthropogenic land use, cf. Figure 2). For natural states, a map of ecoregions published by the World Wildlife Fund (WWF) is often used, with ecoregions being relatively large units of land or water representing the original distribution of distinct assemblages of species and communities (Olsson et al., 2001). These ecoregions characterize biodiversity with a very coarse resolution. For example, in Sweden the dominating biome boreal forests (also present in Alaska, Canada, Iceland, Norway, Finland, and Russia) is subdivided into ecoregions such as the Scandinavian and Russian taiga. An important avenue to improve LCA is to define ecoregions at a much higher resolution based on transparent methodology and use of data. Furthermore, in a boreal forest, management is an important factor, however, in a tropical forest, these methodologies may not be the most appropriate for assessing impacts on biodiversity.

Another particularly troublesome assumption is that relaxation is assumed to be independent of the social and spatial context. The use of reference states such as potential vegetation also implicitly puts a priority on biodiversity per se, and fail to account for other values of biodiversity such as the value for humans in terms of ecosystem services, nature's contribution to people (Vrasdonk et al., 2019). While GLAM 1 suggested that the reference should be "hypothetical biodiversity potential", the method that is currently recommended by GLAM (Scherer et al., 2023), uses it as a natural area in the ecoregion. The problem with this assumption is that the reference state may not be the desirable state that produce goods and services we use. Another aspect is that biodiversity is determined by historical legacies and paths to recovery may be strongly contingent on the land use surrounding the site at which relaxation occurs. For example, for sugar cane production in Brazil it may be highly unlikely that biodiversity after relaxation would return to the ideal state of the ecoregion, since the landscapes where sugar cane dominates are highly fragmented and in general very intensively farmed.

When assessing biodiversity impacts of land use in LCA, many simplifying assumptions are made due to lack of more sophisticated data and approaches to take various factors into account. Koellner et al. (2013) in a guideline on how to build methods for land use impact assessment in LCA, described such simplification as: "discrete land use types are sufficient for an assessment of land use impacts; ecosystem quality remains constant over time of occupation; time and area of occupation are substitutable; transformation time is negligible; regeneration is linear and independent from land use history and landscape configuration; biodiversity and multiple ecosystem services are independent; the ecological impact is linearly increasing with the intervention; and there is no interaction between land use and other drivers such as climate change".

Not all aspects can be defined objectively. One relevant assumption is the attribution of land use or more often land cover change. How to assign a deforestation area to a product, such as biomass, or to the energy generated from it? If assigning impacts to the product that cause it may unfairly attribute the demand for land to displaced crops. The framework distinguishes land occupation and transformation making the allocation of converted land important to the final results. In life cycle assessment, the method developers define impact factors for land occupation and transformation, but how much of transformed land is attributed to the and use occupation is part of the inventory. However, whether or not the assumptions are required for technical reasons, they are extremely relevant to LCA of biomass-based product such as biofuels.

3 Challenges to implementing biodiversity in LCA of bioenergy systems

A critical element of LCA for biodiversity impacts is the spatial resolution of the information used for the assessment (Hellweg et al., 2023; Winter et al., 2017). Whenever impacts depend on location, spatial information of where impact occurs is critical. In terms of biomass for bioenergy, this may be very critical since different localities may have different background levels of biodiversity, even at relatively limited distances. Furthermore, localities may differ in the proportion of threatened or endemic species they harbour, affecting any impact assessment that considers global extinction risks. In addition, different localities may have very different productivity, such that the production of a unit of bioenergy occupies different areas of land. However, for biomass production for bioenergy such as the production of sugar cane for ethanol or recovery of forestry residues for heat and power production, specific information about localities where this is produced may be lost along the value chain. This results much coarser resolution data being used than what would be ideal for capturing spatial variation in biodiversity impacts per unit bioenergy, in terms of estimating the area occupied (determined by productivity) and the characterization factors (difference in ecological quality to references) for biodiversity.

In LCA performed by, for example, consultants, economic considerations may also come into play, limiting analyses to on-the-shelf applications that lack the capacity to consider the effect of spatial context on biodiversity impact factors even when such exist, instead capture these at global or regional scales. A visualization of these gaps is summarised in Figure 3.

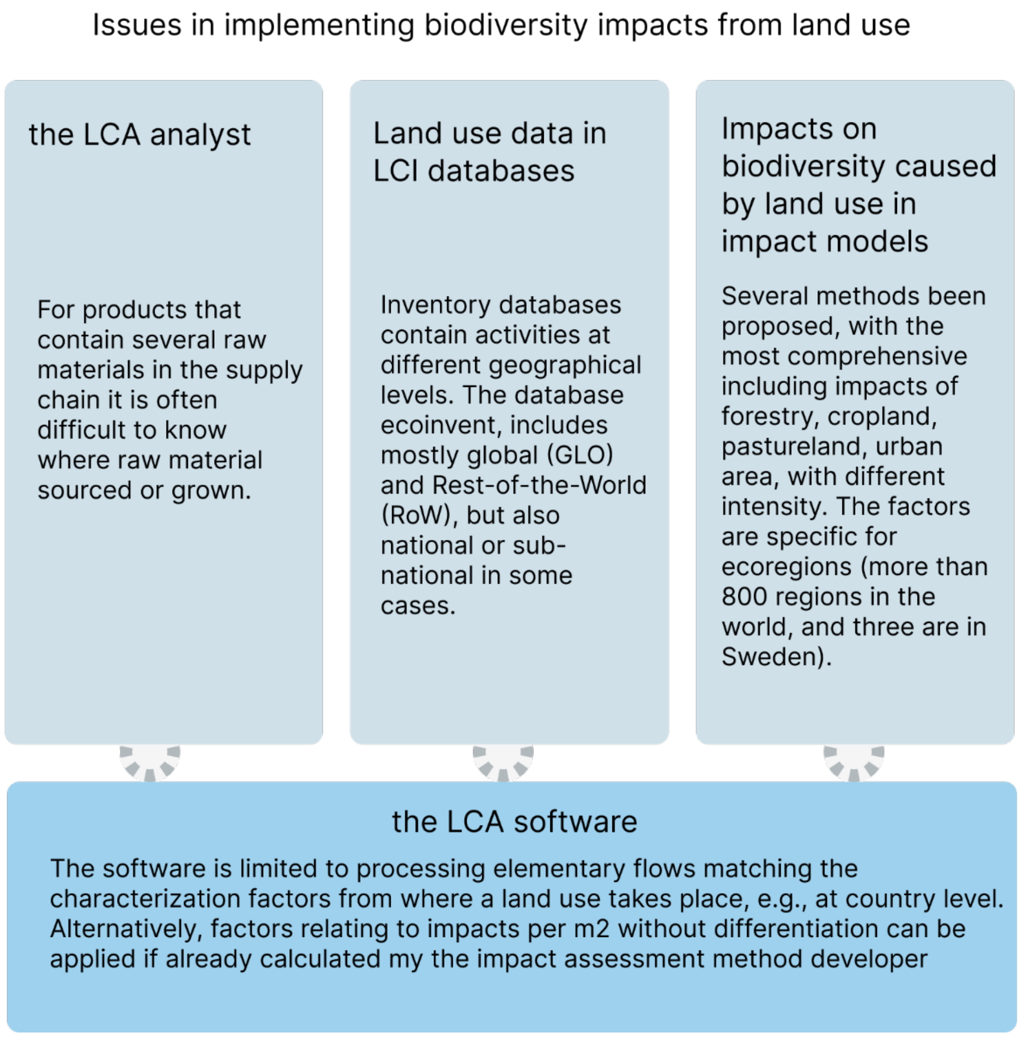

Figure 3. *Diagram summarising the current main barriers to the implementation of the assessment of land use impacts on biodiversity within life cycle assessment.*

In contrast to LCA studies focussing on climate impacts, where it is possible to use a common indicator that translates the different degrees to which greenhouse gases affect global warming, and GWP-100 is normally used, biodiversity is undoubtedly more complex. There is broad but not unanimous, consensus that a similar approach is not possible for biodiversity. Most methods used focus explicitly or implicitly (by the use of indicators) on the loss of species at some spatial scale, but such indicators fail to represent the full extent of values that biodiversity represents, including intrinsic, relational and instrumental values. Although a plethora of valuation method exist (IPBES, 2022), it remains unclear what are the relevant indicators that can be used in LCA, and some values may even be impossible to capture in generalized metrics (Michelsen and Lindner, 2015).

A further issue related to creating biodiversity metrics in LCA, is the sparsity of biodiversity data, including large spatial heterogeneities in the availability of data for different geographic regions and land uses. A consequence of this is that general estimates based on data from the level of ecoregions, may misrepresent local biodiversity consequences. Furthermore, it is unclear to what extent data from certain well-represented taxa can be generalized to other taxa. Although assessments are often based on models, these models are in turn informed by data and thus affected by the data issues mentioned here (Damiani et al., 2023).

In an extensive overview, Damiani et al. (2023) captured the latest methodological development in the vast plethora of biodiversity impact assessment methods that are in use or are proposed in the LCA. They found that several methods were dedicated to capturing impacts from land use as a sole pressure, but a combination with other pressures was common e.g. climate, and toxicity, but also although less commonly invasive species, or fragmentation of natural ecosystems were also included in at least one method. They found that the reference stage was conveyed in different ways across methods, but it was referred to a state of natural undisturbed or to a state before additional perturbation. The different methods were developed from variety of data, including literature data, databases, maps, and ecological surveys. The number of land use classes varied extensively between methods, ranging from methods that covered a broad range of land use classes, including eight classes and three land use intensities, to more specific methods developed specifically for one land use type, with eight intensity classes. Some of the evaluated methods used biodiversity impact metrics that took the species populations class in Essential Biodiversity Variables (EBV) into account. EBV includes genetic composition, species population, species traits, community composition, ecosystem functioning, and ecosystem structure (for more information, see Pereira et al, 2013). There was a variable accounting for which biodiversity was represented in terms of taxonomic or functional coverage. Some methods used biodiversity impact not directly related to an EBV class, e.g. by using indicators for human influence the environment (see Section 2.1).

As evidently shown by Damiani et al. (2023), no single method is able to simultaneously capture the variety of pressures on biodiversity, ecosystems, and taxonomic groups. Due to the large number of methods and tools that has been developed, there is a need of improved guidance regarding their relative merits (Hawkins et al., 2024; Zhu et al., 2024). There is in particular a need to increase the coverage of methods in term of pressures, taxonomic groups, and levels of biological organization as well as to consider consequences for ecosystem functioning (Damini et al. 2023). A potential beneficial use of impact assessment is as a way to increase the understanding of what impact a biomass-based product, such as a biofuel, has on biodiversity. Here, the focus should be on aspects of biodiversity that are particularly eroded by that bioenergy production, e.g. loss of endemic species or loss of local biodiversity for ecosystem services.

4 Case study of forestry residues using the Biodiversity Potential method

4.1 Background to the case study

LCA is an instrument that can be used to characterise and change the impact an energy company has on biodiversity, by considering which biomass is used, where and how it is produced. It is a way to systematically consider impact along the value chain. However, the value of the instrument will depend on adequate methods being used and relevant information to inform the assessment to be available. For bioenergy companies, a particular contentious issue may be to trace the sourcing of bioenergy feedstocks and the methods by which they were produced. It may be particularly difficult when the distance to the production is long and value chains complex. Both the ease to obtain this information and the possibility to let any LCA influence the sourcing may differ between energy companies, with for example small energy producers having smaller possibilities to demand information from suppliers or require that they modify their sourcing of production methods, as relevant.

LCA can be regarded both as a learning process and as a way to inform business decisions. By conducting a meaningful LCA that assesses the potential biodiversity impacts of bioenergy requires information about the bioenergy supply chain. To obtain this information, the energy company conducting the LCA needs to gather this information. The very process of gathering the relevant information will increase the company's *supply chain awareness*, which in turn will enabling it to target actions to reduce impact on biodiversity. For example, to adequately conduct an LCA, the company may need to require (i.e. demand from their suppliers) an increased level of spatialized information on where the biomass feedstock is sourced from. Hence, the LCA process may incite the company to increase the transparency in the supply chain, allowing actions and practices to support biodiversity to percolate through a bioenergy supply chain. For new adopters, this may be the most important part of the LCA process, but meaningful decision making will undoubtedly require increased learning about the supply chain.

Most importantly, the LCA may be used for decision-making, including targeting sourcing to less biodiversity sensitive areas or reducing impact by using biomass produced by alternative production methods. Energy companies may also want to consider production methods, such as the use of certified biomass feedstock (see e.g. Forestry Stewardship Council, FSC). However, the LCA can only adequately inform about the impact of using such alternative methods, if data relating to this systems is capture in the study, such as the consequences in terms of area demand. LCA results, as any model, are just are representative as the data that feed in the model.

4.2 Forestry residues in combined heat and power (CHP) production

This section is based on a recent MSc thesis (Kvarnbäck, 2024) analysing the biomass supply chain to the energy company *Kraftringen Energi*, located in the very south of Sweden close to the city of Lund. The company runs a large, combined heat and power plant (CHP) delivering district heat and electricity to surrounding municipalities. The plant is mostly fuelled with forestry residues, or logging residues such as tops and branches after final felling's and thinning's, from the region. The main supplier of forestry residues is the forest company *Södra Skogsägarna*, which is the largest forest owners' association in Sweden located in the south part of Sweden. Within the BIOPATH project, in which both *Kraftringen* and *Södra* participate, a case study was performed assessing the potential biodiversity impact from the harvest of forest residues by *Södra* delivered to *Kraftringen's* CHP plant, from a life cycle perspective (Kvarnbäck, 2024).

The approach selected for the study was the one suggested by Lindner et al. (2021), which focuses on landscape attributes indicative of ideal conditions for biodiversity and relates biodiversity impact to a number of land management parameters. Examples of parameters when applied to forestry are the number of native tree species, where a high number is an indicator of high quality of biodiversity, whereas the occurrence of exotic vegetation is an indicator of low quality. The age structure of the tree population is also of importance, as well as the presence of deadwood, where a high number of old trees and a high volume of deadwood are indicators of high quality of biodiversity. The presence of microbiotopes in the forest, such as water holes and riparian zones, is important for the biodiversity, and another indicator is the share of protected areas designed to protect biodiversity. A final example of a parameter relevant to biodiversity is the frequency of disturbances, such as low-intensity fires (Lindner et al., 2021).

Thus, this approach, as explained above, does not focus on species richness as, for example, approaches considering the potentially disappeared fraction (PDF) of species (see e.g. Scherer et al., 2023). Instead, the selected method represents a conditions-based approach (see e.g. Myllyviita et al., 2019). The rational for choosing this method is that, it can capture management practices, does not require categorization in intensity levels of land management, it is not restricted to a particular type of biodiversity (genetic, species, population, community and ecosystems) and that it is flexible regarding the availability of data, e.g.

collecting data on finer spatial levels from individual forest owners. This approach was chosen because acquiring primary data on species richness for a particular forest stand or property is typically challenging and resource demanding, while obtaining data on structural attributes related to forest management is more straightforward (Mattsson et al., 2022). However, it should be noted that the method is only as good as the ability of the structural variables to reflect biodiversity, where a plethora of structural variables have been suggested to be relevant for forests (Wagenaar et al., 2024).

The method, named the Biodiversity Potential (BP) method, differs from most other approaches regarding its use of a continuous scale for biodiversity impact and not predefined land-use intensity classes (Lindner et al., 2021). This enables high resolution assessment and differentiation between forest properties that would otherwise be assigned to the same predefined land-use intensity class (see e.g. Scherer et al., 2023; Chaudhary and Brooks, 2018). For a more in-depth description of the BP method, see Lindner et al. (2021).

In the work by Kvarnbäck (2024) the BP method was further developed and adapted to the regional conditions of forest management and practices for harvest of logging residues. The approach starts with defining model parameters based on biodiversity expert interviews, followed by guidance of the construction of mathematical relations between parameters and biodiversity, as well as the relation between parameters (Lindner, 2021). The interviews included the judgement of five forest experts in Sweden, covering following areas of expertise (besides general knowledge of biodiversity): ecology of temperate broadleaf forests, forest conservation, saproxylic insects, fungi, and epiphytic lichens. The development and adaption of the method resulted in eight management parameters and their contribution to the BP according to Figure 4. It was identified that the forestry residues consist of logging residues in the form of tops and branches from both deciduous trees, such as beech (*Fagus sylvatica*) and oak (*Quercus robur*), and coniferous trees, such as spruce (*Picea abies*) and pine (*Pinus sylvestris*). The model was created based on this and input from the expert interview (Kvarnbäck, 2024).

In the modified model, illustrated in Figure 4, logging residues were divided between fine woody debris (FWD), including most tops and branches, and coarse woody debris (CWD), including snags, standing dead trees and high stumps (diameter >10 cm) after felling's. Existing dead wood (mainly spared stem wood) was divided into three classes based on diameter, where class 1 is most important for biodiversity (diameter >50 cm), followed by class 2 (diameter 10-50 cm), and finally class 3 (diameter <10 cm). The eight management parameters in Figure 4 include specific units (e.g. m^3/ha, % logged area etc.) and a range, which together leads to a quantification of each management parameter for a specific forest area. Then, the biodiversity contribution from each of the eight management parameters was described by a contribution curve which was drawn based on the expert judgements. The x-axis in the contribution curve is the specific metric to quantify the management parameter (m^3/ha, % logged area, etc.), while the y-axis is the biodiversity contribution in the interval between 0 to 1. For example, the contribution curve for old trees specifies that approximately 25-30 old trees per hectare are required to reach the full biodiversity contribution, equivalent to a value of 1, from this specific management parameter (biodiversity contribution = 1), whereas approximately 5 old trees lead to half of the full biodiversity contribution, equivalent to a value of 0.5. Each contribution curve is presented in detail in Kvarnbäck (2024).

Relations of dependence are present where more than one management parameter is related to the same biodiversity attribute, which is illustrated in Figure 4 by the different arrows. For example, experts agreed that a diversity of native tree species needed to be combined with a limited area cover of exotic tree species to reach the full biodiversity potential. Regarding dead wood, native logging residues were judged to be able to replace some naturally occurring dead wood of similar coarseness. Each of the five biodiversity contributions was then assigned a contribution weight, illustrated in the right column in Figure 4. For example, the presence and quantity of trees older than 150 years was identified as the single most important contribution, leading to a weight of 40% of the total biodiversity potential. The two remaining attributes, native tree species diversity and dead wood, was assigned a contribution weight of 30% each, and where the weight of dead wood was further divided between the subcategories shown in Figure 4 (Kvarnbäck, 2024). The total Biodiversity Potential value of logging residues recovery from a specific logging site can then be calculated by multiplying the biodiversity contribution of each parameter, ranging from 0 to 1, with the weight of the parameter, ranging from 5% to 40%, followed by adding all parameters together.

The case study by Kvarnbäck (2024) was also based on a literature review focusing on forest residue harvest for energy purposes and potential environmental impacts, including some key references and conclusions. For example, a synthesis by Camia et al. (2021) regarding the sustainability of using woody biomass for energy production in the EU, concludes that the removal of fine woody debris is associated with low or no biodiversity risk as long as the extraction levels are below local or regional landscape thresholds, whereas the extraction of coarse woody debris should be avoided due to potential negative impacts. Furthermore, long-term research in Sweden regarding logging residue harvest show that, on average, a 50% harvest of slash is possible on clear cut areas without harming biodiversity, but with local restrictions (Akselsson, 2024; de Jong et al., 2017). Similar thresholds are also included in the regulations from the Swedish Forest Agency (2019). In parallel to these findings summarised in Kvarnbäck (2024), thresholds for forest residue removal are also implemented in other parts of the world, such as in New Zealand by the National Environmental Standards for Commercial Forestry (New Zealand Government, 2024).

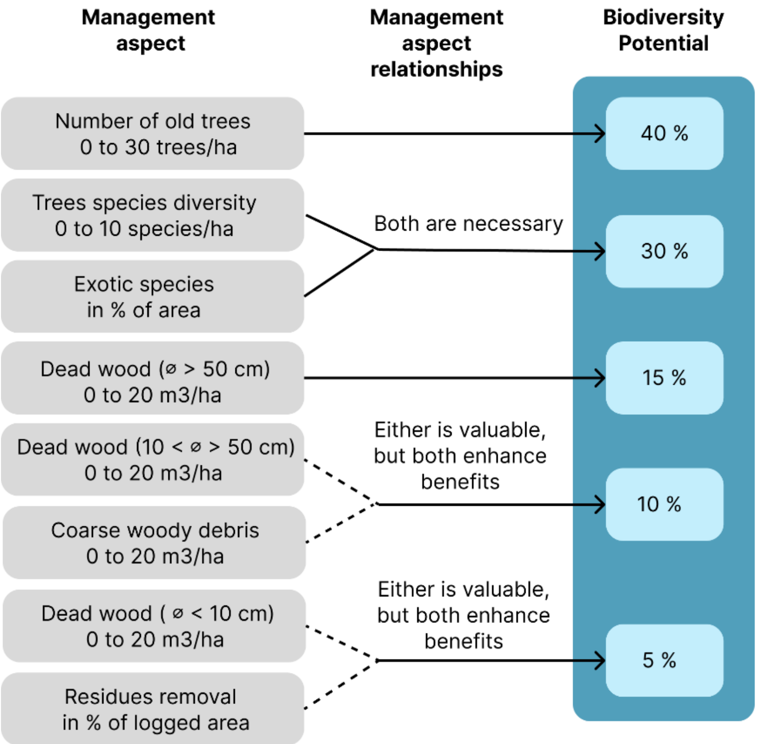

Figure 4. *Biodiversity parameters and their contribution to biodiversity as a whole, regarding forest residue recovery in southern Sweden according to Kvarnbäck (2024).*

The assessment of the logging residue fuels used by *Kraftringen's* CHP plant was based on data from seven selected forest owners (forest properties) in *Södra* delivering forestry residues (in form of wood chips) to the CHP plant. The primary data was collected by structured interviews with respective forest owner covering 12 separate forest stands from where logging residues were harvested. Respondents were asked to answer questions based on the current situation on their forest property, followed by questions regarding forest management measures, including both logging and harvest of logging residues, and considering the management parameters in the BP method (presented in Figure 4) (Kvarnbäck, 2024). Typical piles of logging residues from these suppliers, before chipping and transport to the CHP plant, are shown in Figure 5.

Figure 5. *Logging residues from deciduous trees (left) and spruce (right) (Photo: A. Kvarnbäck).*

The process of applying the BP method can be summarised in three steps. In the first step, the maximum BP value of a forest stand corresponding to ideal conditions for biodiversity is set. This maximum biodiversity value is then compared with the actual BP of the forest stand based on the quality of the current land use from a biodiversity perspective. The BP value ranges between 0 and 1, where BP = 1 represent the reference state where the biodiversity potential is fully achieved, and where the individual BP for each stand reflects the actual forest management in this stand. Thus, this step is coherent with the model approach discussed in Section 2 regarding land use. In the second step, the additional biodiversity quality change related to logging residue removal is compared with the alternative scenario leaving all logging residues in the stand. This additional biodiversity quality change is thus not directly compared with the reference state where the biodiversity potential is fully achieved (as in the first step), which could be perceived as not coherent with the BP model, and where this additional land use change is assessed towards a more quasi-natural state. However, by adding this additional change in land use in form of logging residue harvest to the primary land use in form of timber production, both the differences between various stands in relation to the reference state can be measured, and the magnitude of the additional impact caused by logging residue recovery. In the third step, the biodiversity quality change is related to the functional unit in the LCA (described below), to connect the biodiversity impact to the studied function of the biomass fuel.

Figure 6 shows the BP value with and without logging residue harvest for the 12 individual forest stands in the case study by forest property (A-G) and the dominant tree species. One conclusion from Figure 6 is that there is a rather large variability between properties, ranging from BP = 0.15 to BP = 0.5, and that the actual BPs are far from the reference state (BP=1). Another conclusion is that stands dominated by spruce have lower BP than stands dominated by deciduous trees and pine, and that the removal of logging residues often results in minor negative impacts on biodiversity, with some exceptions. For example, logging residue removal from properties with initially low BP, such as property A in Figure 6, may cause a more significant negative impact on biodiversity, especially in deciduous tree dominated forest stands (in this case beech). A reason for this is that the amount of available dead beech wood is in general low at property A, and consequently the removal of beech logging residues has a relatively high impact on BP in the model output. The opposite conditions can also appear, such as in property F, where logging residue removal has an almost insignificant impact due to a much higher initial availability of dead wood at the property (and thereby a higher BP). One important conclusion from these results is that an efficient strategy to reduce the biodiversity impact of logging residue recovery is to make sure that the overall availability of coarse dead wood is high at the forest property, since this can compensate for the logging residue removal. This can be achieved in practice by saving more old trees in final felling's at the felling sites, creating "high stumps" in thinning's (cutting the top of the tree), sparing groups of old trees in microecosystems, leave fallen trees in the forest, etc., which all together will build the pool of coarse dead wood.

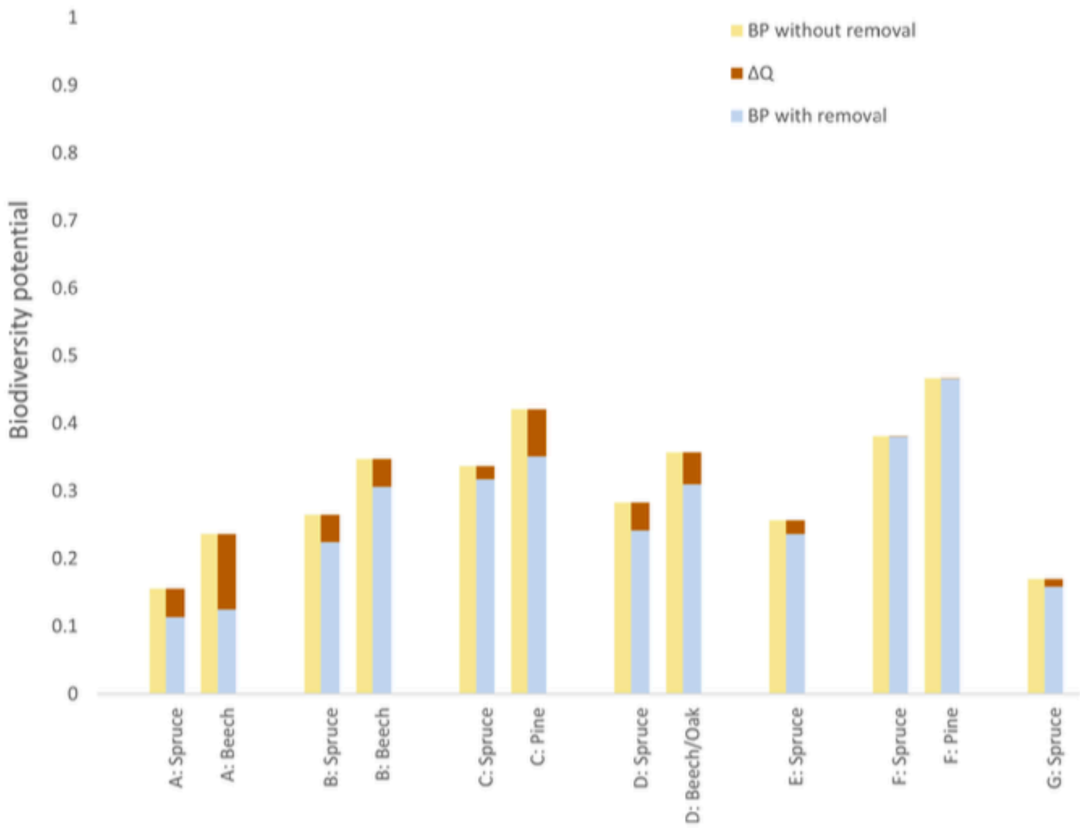

Figure 6. *Biodiversity potential of seven different forest properties with and without logging residue removal. The y-axis represents the BP value of each stand, with and without logging residue removal, and in relation to the reference state where the fully biodiversity potential is achieved (BP = 1). The red column illustrates the additional contribution to BP from logging residues (corresponding to ΔQ) (Kvarnbäck, 2024).*

The BP method was developed to be applied in an LCA context. While the system boundaries, i.e. the included process that affect biodiversity, are limited to the primary production stage, impacts are calculated in terms of a common denominator to allow for comparisons. This common denominator is the functional unit in the LCA, it connects the impact (in this case, impact on biodiversity) to the studied function delivered (here energy output from the biomass fuel). The functional unit is set to *1 kWh of hot water produced and distributed to a customer*. To allow the comparison to the same output of 1 kWh, the logging residue harvest yield per hectare and year, as well as the energy content in the various logging residues and the conversion efficiency in the production of district heat, need to be taken into account.

The case study show that the results expressed per functional unit, see Figure 7, will be almost synonymous to the biodiversity impact indicated in Figure 6, but with one exception. The yield of logging residues in the pine stand at property C was estimated to be rather low, expressed as kWh biomass per hectare and year, compared with the other stands included, thus this led to a rather significant increase in the biodiversity impact per functional unit. Finally, when interpreting the results of this case study is important to keep in mind that information regarding the different stands was gathered from interviews and is therefore subject to owners' perceptions. Thus, there is room for further developments of the applied method, including a more efficient and objective collection of data about actual forest management parameters, which will be discussed in Section 5. For example, parameter values could be based on structured processes involving a multitude of stakeholders including scoring of uncertainty.

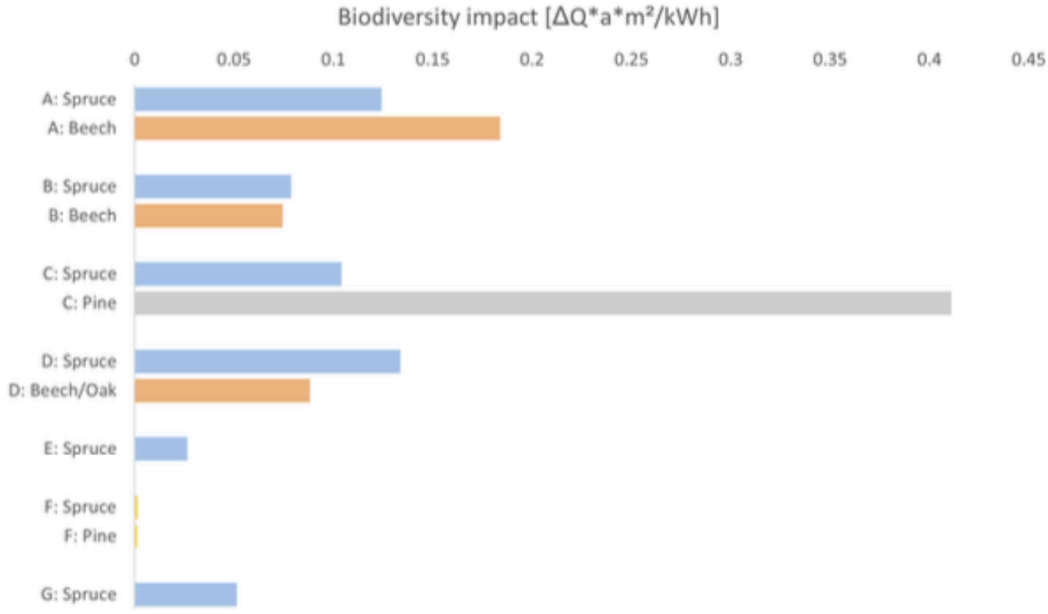

Figure 7. *Biodiversity impact per functional unit (1 kWh of hot water produced and distributed to a customer). Blue bars represents spruce stands, orange stands represents beech stands and grey bars represents pine stands (Kvarnbäck, 2024).*

5 opportunities for future development

Life cycle assessment tools for bioenergy systems and biofuels carbon footprint are more established than those for assessing biodiversity. Future development of tools and modelling approaches regarding biodiversity is thus needed, e.g. to secure coherent choices of reference land use to avoid misleading results (Agostini et al., 2020; Soimakallio et al., 2015). Applying the LCA methodology to assess biodiversity requires also more detailed information about the bioenergy systems and biofuels life cycle and especially regarding the biomass production phase and the local biodiversity conditions, which has been in focus in this report. Today, the availability of data on land-use management parameters and biodiversity attributes on a high-resolution scale is a limiting factor to make coherent and efficient assessments of biodiversity impact from specific bioenergy systems. Collecting inventory data from, e.g., field inventories, interviews with landowners, etc., is often very resource consuming which was demonstrated in the case study presented above. One promising opportunity is, however, the development of remote sensing methodologies based on satellite images, airborne laser scanning, digital photogrammetry and 3D radar data using interferometry (Lindberg, 2024). Remote sensing is suitable to assess biodiversity indicators on a regional landscape level (including identity, distribution, richness and proportion of habitat type), but also on a somewhat lower level, such as community-ecosystem (including identity, relative abundance, frequency, richness, evenness and diversity of species and guilds). When it comes to even lower levels, such as population-species and genetic, other methods are however needed (Lindberg, 2024). Monitoring of species are, for example, limited by the wavelength's restrictions from satellite instrumentation, and here is a need for a methodology that includes proxies and correlations to assess biodiversity (e.g. soil biodiversity).

Remote sensing technologies and concepts are developed quite rapidly today. For example, remote sensing of Swedish boreal forests is estimated to be applicable for quantifying important biodiversity indicators, such as stand age, tree species composition, dead wood (standing and lying), vertical layering, horizontal structure (gaps and patchiness), topography and soil wetness (Lindberg, 2024). These indicators correspond, to a large extent, to the management parameters in LCA methodologies using landscape attributes, such as the Biodiversity Potential method exemplified above. However, one potential limitation regarding the public useability of property-level data is the ownership of the data. Using data of forest management parameters and biodiversity indicators on a property level may require permission from the landowner which can restrict the availability. Thus, legal aspects should also be taken into account in the development of biodiversity databases for life cycle assessment of bioenergy systems and biofuels.

Another area of importance is to enhance the work to verify the accuracy of biodiversity indicators by studying the relation between the indicators with more direct measures of biodiversity and observations of species. The development of the BP method so far has been based on expert judgment but it will be essential to strengthen the biodiversity data validation in future developments to verify the parameters contribution, its relationship with other parameters and the weighting. More information about species habitat needs will result in improved indicators and this could preferably be done on a more local scale and over small but representative areas to reduce the expense of the research since it would be too expensive to cover the whole landscape (Lindberg, 2024). Improving biodiversity indicators taking into account varying local conditions are paramount for the assessment of land use and biomass production, and thereby in LCA of bioenergy systems and biofuels.

6 Conclusions

Quantifying biodiversity impacts of bioenergy systems using LCA approaches requires adequate biodiversity indicators, or generic biodiversity measures designed to reflect and track important features of the ecological system. The rationale for using concrete and quantified biodiversity indicators is that they can assess something that either cannot or would be too expensive to measure. It is also important that they should be management and policy relevant and interpretable in terms of environmental trends or progress towards policy goals.

The lack of a single biodiversity indicator or the complexity of measurement shall not justify general exclusion of biodiversity assessments and actions regarding biomass production systems. While these issues can be perceived as barriers, using quantitative tools such LCA brings undoubtedly an opportunity to companies and policy makers to better understand various biomass supply chains, when applied coherently. Greater biomass production systems awareness and control enables for a realm of actions to reduce environmental impacts, which are of high urgency for solving the climate and biodiversity crisis. Furthermore, the ongoing development of improved biodiversity impact models and collection of life cycle inventory data, e.g. based on appropriate biodiversity indicators and remote sensing methodologies, will open for increasingly applicable assessments. Most importantly, life cycle assessment is a quantitative tool that can be used to complement bioenergy companies' efforts in improving their environmental performance.

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