Integrating Biodiversity Impacts into Life Cycle Assessment

Applied to a case study on onshore wind power production

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Abstract

In order to support the transitioin towards sustainable energy systems, the future expansion of renewable energy systems is imperative. Among them, is wind energy – method which is commonly associated with low GHG emissions during the operation and energy generation. While this is true, the operation of wind farms is associated with significant impacts on biodiveristy. Life Cycle Assessment (LCA) is widely applied for quantification of environmental impacts and has become the main tool for decision-makers and scientific community to assess biodiveristy. However, recent research suggests that the inclusion of biodiveristy impacts in LCA framework is limited and does not provide sufficient assessment of the phenomenon.

To address this gap, current thesis aims to (1) *examine* methodological robustness of the current LCA approaches for biodiversity assessment, and (2) *understand* how the effects of onshore wind power production on biodiversity can be assessed in a more comprehensive way within the LCA framework. A combination of literature review, comparative assessment of available biodiveristy footprinting methods and a comparative LCA study of a wind farm was conducted to inform the research objectives.

The findings reveal several aspects. *First*, the methodological choices regarding the life cycle impact analysis (LCIA) method significantly influence the outcomes of the LCA. *Second*, the majority of the available beyond-LCA methods – methods which are based on life cycle thinking approaches or use alternative frameworks – only partially capture key biodiversity loss drivers relevant to wind power, highlighting the important gap – lack of sector-specific methodological approaches. Lastly, methods which could be offer good coverage of biodiversity aspects are difficult to incorporate in existing LCA framework. To address this, a stepwise approach has been suggested explaining how practitioners can start closing the gap for sector-specific assessment. The steps include (1) quantificaiton of relevant biodiveristy drivers, (2) translation of pressures into relevant impact pathways for inventory and (3) selection of suitable methods; if there are not relevant LCIA methods available, combine the existing approch with alternative biodiversity footprinting methods which cover the relevant impacts.

The research highlights the importance of method selection in biodiveristy footprinting and provides actionable guidance for LCA practitioners and decision-makers who want to improve their practices.

Key words: onshore wind energy, life cycle assessment, biodiversity footprinting, energy production, environmental impact

Sammanfattning

Energisystem är absolut nödvändiga. Bland dem finns vindkraft – en metod som ofta förknippas med låga utsläpp av växthusgaser under drift och energiproduktion. Även om detta är sant, är driften av vindkraftsparker förknippad med betydande påverkan på den biologiska mångfalden. Livscykelanalys (LCA) används i stor utsträckning för kvantifiering av miljöpåverkan och har blivit det viktigaste verktyget för beslutsfattare och forskarsamhället för att bedöma biologisk mångfald. Ny forskning tyder dock på att inkluderandet av effekter på biologisk mångfald i LCA-ramverket är begränsat och inte ger en tillräcklig bedömning av fenomenet.

För att åtgärda denna brist syftar den aktuella avhandlingen till att (1) undersöka metodologisk robusthet hos de nuvarande LCA-metoderna för bedömning av biologisk mångfald, och (2) förstå hur effekterna av landbaserad vindkraftsproduktion på biologisk mångfald kan bedömas på ett mer omfattande sätt inom LCA-ramverket. En kombination av litteraturgenomgång, jämförande bedömning av tillgängliga metoder för biodiversitetsavtryck och en jämförande LCA-studie av en vindkraftspark genomfördes för att informera forskningsmålen.

Resultaten avslöjar flera aspekter. För det första påverkar de metodologiska valen avseende livscykelkonsekvensanalys (LCIA) resultaten avsevärt. För det andra fångar majoriteten av de tillgängliga metoderna bortom livscykelanalys – metoder som är baserade på livscykeltänkande eller använder alternativa ramverk – endast delvis viktiga drivkrafter för förlust av biologisk mångfald som är relevanta för vindkraft, vilket belyser den viktiga bristen – bristen på sektorspecifika metodologiska tillvägagångssätt. Slutligen är metoder som skulle kunna erbjuda god täckning av aspekter av biologisk mångfald svåra att införliva i befintliga LCA-ramverk. För att hantera detta har en stegvis metod föreslagits som förklarar hur yrkesverksamma kan börja minska bristen för sektorspecifik bedömning. Stegen inkluderar (1) kvantifiering av relevanta drivkrafter för biologisk mångfald, (2) översättning av påfrestningar till relevanta påverkansvägar för inventering och (3) val av lämpliga metoder. Om det inte finns relevanta LCA-metoder tillgängliga, kombinera den befintliga metoden med alternativa metoder för biologisk mångfaldsavtryck som täcker de relevanta effekterna.

Forskningen belyser vikten av metodval vid fotavtryck för biologisk mångfald och ger handlingskraftig vägledning för LCA-yrkesverksamma och beslutsfattare som vill förbättra sina metoder.

Nyckelord: landbaserad vindkraft, livscykelanalys, biologisk mångfaldsavtryck, energiproduktion, miljöpåverka

List of abbreviations

EM	Ecosystem Multifunctionality
ES	Ecosystem Services
GHG	Greenhouse gas
GIS	Geographic Information Systems
IEA,	International Energy Agency
ILCD	International Reference Life Cycle Data System
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MCA	Multi-criteria Analysis
MRIO	Multi-Regional Input-Output
MSA	Mean Species Abundance
MW	Megawatts
NCP	Nature’s contribution to people
NREL	National Renewable Energy Laboratory
PAF	Potentially affected fraction
PBF	Product Biodiversity Footprint
PDF	Potentially Dissapeared Fraction of species
REMPD	Renewable Energy Materials Properties Database
SBTN	Science Based Targets Network
WDPA	World Database on Protected Areas
WWF	World Wide Fund for Nature

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1 Introduction

Throughout human history, society has systematically exploited natural resources and has paid little attention to the consequences of its actions (Rockström et al., 2009). There is no doubt that human activity has had a detrimental effect on the natural processes governing life on Earth, as evidenced by the loss of biodiversity, the increased frequency of natural disturbances, and ultimately, climate change. Consequently, many planetary boundaries have either already been breached or are on the verge of being breached (ibid.).

Modern society has reached a significant level of awareness regarding its actions and is taking steps to mitigate negative effects and adapt to the new reality using contemporary technologies and industrial innovations. One of the chosen pathways is the transition from fossil fuel-based energy generation to one based on renewable sources (IPCC, 2023). Wind energy systems are among the most prevalent options, and many countries have committed to their expansion (Wind Europe, 2021; Bošnjaković et al., 2022). Wind energy production is associated with minimal greenhouse gas (GHG) emissions, making it one of the most viable options for tackling modern issues, and it can potentially reduce dependence on fossil fuels (Oebels & Pacca, 2013; Das & Nandi, 2022). Besides low GHG emissions, wind power production does not contribute to air or water pollution, can be integrated into other forms of land use (e.g., integration into agriculture and grazing), and generates minimal toxic waste.

However, the manufacture of the components comprising the wind farm, their installation, and the use phase of the farm itself are still associated with numerous environmental impacts, particularly on biodiversity and ecosystems (Oebels & Pacca, 2013; Gasparatos et al., 2017; Das & Nandi, 2022). Studies on wind power's effects on marine, terrestrial and aquatic species have identified several issues pertaining to biodiversity and ecosystem services (Sander, Jung & Schindler, 2024). During the operational phase of the turbines, these impacts include habitat fragmentation or destruction, alteration of landscapes, creation of barriers to the seasonal migration of birds and other species, noise pollution, and vibrations from their operation, as well as significant demand for materials and resources for manufacturing and installation (Drewitt & Langston, 2006; Gasparatos et al., 2017; Tolvanen et al., 2023). Driven by the rapid expansion of the wind power systems, there is a strong need for comprehensive assessment of biodiversity loss pertaining to their operations (Schöll & Nopp-Mayr, 2021).

The Life Cycle Assessment (LCA), the standard and widely accepted approach to assessing the environmental impacts of different systems, allows practitioners to estimate the environmental impacts of different systems throughout their whole life cycle – from raw material acquisition until the end of life (Matthews et al., 2014; Jolliet et al., 2016). The LCA methodology has become a “go-to” tool for many organisations during research and design phases as well as for business management and policymaking to get a representation of the potential environmental impacts that different systems might have (ibid.). As such, LCA has been quoted as one of the approaches capable of calculating the life cycle impacts on biodiversity from products, services or organisations and providing an estimate of their biodiversity footprints (Bromwich et al., 2025). On the other hand, practitioners specialised in the topics of biodiversity assessment and LCA state that the assessment methodologies for biodiversity impacts require further refinement and development (Crenna et al., 2019; Asselin et al., 2020).

Despite the many benefits, traditional LCA approaches represent environmental impacts in aggregated categories (mid-point or end-point indicators), which do not comprehensively cover all nuances of different systems, like in the case of biodiversity (Crenna et al., 2019). The LCA methodology focuses on such categories as land use, GHG emissions, and resource depletion, and several life cycle impact assessment (LCIA) methods include modules targeted towards assessing biodiversity impacts, through proxy metrics (Asselin et al., 2020; Verones et al., 2020), which represent a proportion of all biodiversity impacts of onshore wind power, however, still fail to capture their full complexity (Damiani et al., 2023). So, despite their many advantages, the LCA methodologies focus on aggregated outputs which are summarised into one metric, and several practitioners raise concerns about their maturity and their ability to comprehensively account for the complexity of biodiversity-related parameters (Crenna et al., 2019, Asselin et al., 2020; Verones et al., 2020; Damiani et al., 2023). Even within the LCA framework, different LCIA methods utilise different assumptions for the relative importance of environmental pathways, leading to differing outcomes between LCA studies (Bromwich et al., 2025).

Another pitfall of existing biodiversity assessment methods within LCA is their reliance on simplified indicators (Marques et al., 2021). While such indicators provide a standardised metric which can be used in future comparisons, they often fail to capture the specifics of the systems and local pressures on biodiversity. Furthermore, the lack of a standardised methodology for integrating ecological data in LCA hinders the comparability of different studies (Damiani et al., 2023). Thus, there is an apparent need to combine LCA with ecological field data to improve the robustness of the biodiversity impact assessment methods (Asselin et al., 2020).

This master's thesis project is part of a broader work on the sustainability of wind energy in the frame of the VindEl research program funded by the Swedish Energy Agency. It aims to assess the environmental, economic, and social impacts associated with wind power production in Sweden in the past, present, and future, using LCA methodologies. While this is the overall goal of the research project, the current study will delve into the questions of biodiversity assessment methodologies and their integration into the bigger framework of the life cycle sustainability assessment (LCSA) and LCA, specifically.

1.1 Aim and objectives

Given the modern challenges and the urgent need to better understand the ecological trade-offs of onshore wind energy production due to its rapid expansion, the current report aims to (1) *examine* methodological robustness of current LCA approaches for biodiversity assessment, and (2) *understand* how the effects of onshore wind power production on biodiversity can be assessed in a more comprehensive way within the LCA framework.

To achieve these two aims, the project will fulfil the following objectives:

1. Explore the influence of methodological differences on evaluation of biodiversity within the LCA framework using a case study
2. Identify and examine relevant biodiversity footprinting methods
3. Evaluate their suitability for integration into the LCA framework using a set of criteria
4. Propose an approach to integrate the biodiversity impact assessment methodologies into the LCA framework.

The methodological robustness of current LCA approaches will be addressed by Objective 1, while the question of a more comprehensive integration of biodiversity assessment within LCA framework will be addressed by Objectives 2-4. The case study approach will be adopted to inform results to satisfy Objective 1, focusing on the LCA study of a wind farm with two alternative configurations of tower. Unlike the rest of the work, where the focus is on the use phase impacts, the analytical focus of the case study will be on the raw materials and manufacturing stages. The remaining objectives will be addressed by means of literature review and multi-criteria analysis of methods.

To achieve said purpose of the study, the study will answer the following research question:

"How can practitioner improve the current practice for assessment of biodiversity impacts within LCA framework for onshore wind power production?"

To support it, the following questions will be answered:

RQ1: What is the effect of the differences in LCIA methods on results of biodiversity assessment?

RQ2: Which biodiversity footprinting methods beyond LCA framework are currently available?

RQ3: How can these methods be integrated into conventional LCA to improve the estimation of the impacts of onshore wind power production?"

1.2 Delimitations

To better frame the scope of the study, certain delimitations have been applied. Given that biodiversity is a multidimensional entity and there exist many methods to assess biodiversity on different levels and with different applications (IPBES, n.d.; Damiani et al., 2023; Barth et al., 2024; De Ryck et al., 2024), the current study focuses only on methods which are intended for a decision-making context. Such a choice is motivated by the fact that the original intended purpose of LCA is to support the decisions, which places significant restrictions on the scope of its methodology (Jolliet et al., 2016, p.1). As such, the methods which are assessed for their potential compatibility for LCA integration have to be from the same domain and focus on supporting the decision-making efforts.

2 Theoretical background

This section will provide background information necessary to understand the topic of biodiversity assessment within the framework of Life Cycle Assessment (LCA). *First*, the section will cover aspects related to biodiversity, biodiversity loss. *Then,* the section will cover the LCA methodology in general and its inclusion of biodiversity considerations to give a sufficient understanding of the components which comprise the analysis. Lastly, the section will outline the impacts of wind power operation on biodiversity, since they are much broader than captured by conventional LCA methodology.

2.1 Biodiversity loss

Before addressing the issues of biodiversity assessment, it is important to clarify what is understood under the term "biodiversity". According to the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, n.d.), biodiversity is "*the variability among living organisms from all sources, including terrestrial, marine and other aquatic ecosystems and ecological complexes of which they are part of. This includes variation in genetic, phenotypic, phylogenetic, and functional attributes, as well as changes in abundance and distribution over time and space within and among species, biological communities and ecosystems*". This means that biodiversity is a multidimensional entity and considers changes to not only species but also ecosystems and genetic variability, which are specific to a geographical location (ibid.).

The main drivers for biodiversity loss have been identified to be *land use change*, *resource overexploitation*, *invasive species*, *climate change* and *pollution* (Millennium Ecosystem Assessment, 2005; Damiani et al., 2023). The environmental drivers in most cases are caused by the changes in the socioeconomic sphere, such as the growth of population, changing consumption patterns, rapid urbanisation, growth of trade and industrialisation (ibid). Because of this, biodiversity loss, along with climate change, has been recognised to be one of the most pressing issues challenging the well-being of human society and nature (IPBES, 2019). In response to the growing loss of biodiversity, the European Commission has expressed the need for better integration of biodiversity considerations on every level of decision-making and for better methods, criteria and standards to inform the decisions (European Commission, 2020).

2.2 Life cycle assessment as a tool to assess supply chain impacts

One of the most common approaches to address the environmental impacts, including those on biodiversity, of a product system in the supply chain is LCA. According to Jolliet et al. (2016, p. 1), "*Life cycle assessment (LCA) is a decision-making tool which specifically addresses the need of selecting and optimising available technological solutions*". The authors further state that the relevancy of the LCA as a decision-making tool comes from the fact that it covers the entire life cycle of a product or service from its inception to final disposal and links the environmental performance of said product or service to its functionality, thus providing a quantification of the environmental impact due to the functional performance. Another advantage of LCA is linked to its ability to represent the product as a system, which allows for better quantification and analysis (Matthews et al., 2014, p. 29).

According to ISO (14040:2006, p. 7), the LCA study consists of four stages: (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation phase. An LCA study is an iterative process, where the results of the previous stages inform the next stage (ibid.). As the study progresses, more information often becomes available, so the inputs to each stage will inevitably be revised until the optimal setup is reached (Joint Research Centre, Institute for Environment and Sustainability, 2010, p. 25). The overall LCA process is illustrated in Figure 1. The LCA study can also be conducted in a shortened format as a Life Cycle Inventory (LCI) study, which excludes the impact assessment stage (ISO 14040:2006, p. 7).

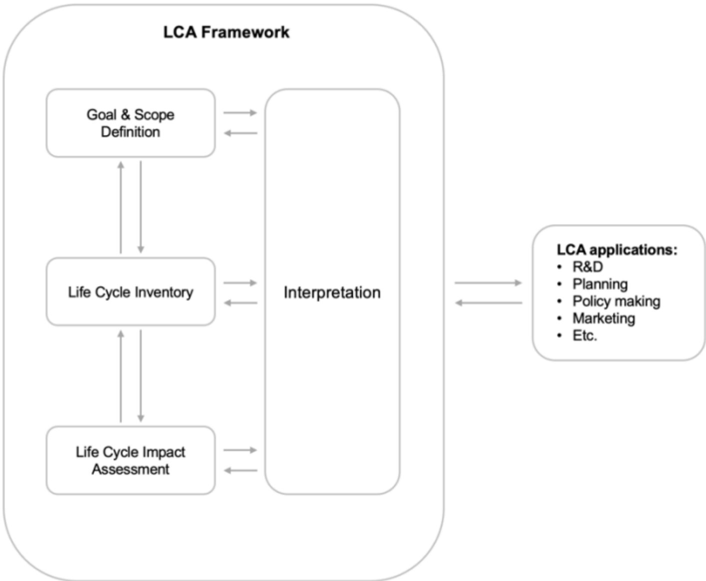

Figure 1. LCA structure (Adapted from ISO 14040:2006).

The first stage in the LCA process is the definition of the goal and scope. During this stage, practitioners outline the purpose of the study, describe the system being studied, the extent of the study, and the methods used (Matthews et al., 2014, pp. 83-85; Jolliet et al., 2016, p. 23). According to ISO (14040:2006, p. 11), the goal statement (usually in the form of a few sentences (Matthews et al., 2014, p. 86)) of the LCA study should include four key considerations: the *intended application* of the study, the *reasons* for conducting it, the *audience* for the study, and whether *the results are intended for comparisons*. In some cases, the goal definition may impose specific limitations on the applicability of the LCA results due to the declared methodology, assumptions, and impact coverage (Joint Research Centre, Institute for Environment and Sustainability, 2010a, p. 32). In such cases, the limitations should be declared and accounted for throughout the study (ibid.). Practitioners need to be precise when defining the goal of the study, as this has direct implications for decisions made during future stages and the study overall (Joint Research Centre, Institute for Environment and Sustainability, 2010a, pp. 33-34).

The scope of the study includes a comprehensive description of the qualitative and quantitative information encompassed within the study and the key parameters that characterise the process (Matthews et al., 2014, p. 86). The breadth, depth, and level of detail in this scope description should align with the objective of the study as defined by the goal (Jolliet et al., 2016, p. 25). To achieve this, the scope definition must describe the product system, its functions and functional units, the system boundaries, allocation procedures, impact assessment methodologies and categories, data quality requirements, assumptions, and limitations of the study, as well as the reporting format (ISO 2006:14040, p. 11; Joint Research Centre, Institute for Environment and Sustainability, 2010a, p. 51). The scope of the study is also influenced by the nature of the LCA study – attributional or consequential (Jolliet et al., 2016, p. 25). The *attributional* LCA describes the environmental impacts of the significant flows in and out of the system, while the *consequential* LCA, on the other hand, illustrates the environmental consequences of decisions concerning the system under analysis (ibid.).

The most important processes in the scope definition are the description of the system functions, their quantification and the setting of the system boundary. *The product system* is a collection of unit processes connected by intermediate flows which satisfy a certain function (ISO 14040:2006, p. 4; Matthews et al., 2014, pp. 92-94). The system is also characterised by elementary flows going into the system from outside the boundary as well as other inputs and outputs (ibid.). As such, *the functional unit* is a quantitative measure of the system function and acts as a reference for all processes happening within and outside the system (Matthews et al., 2014, p. 87; Jolliet et al., 2016, p. 27). Lastly, *the system boundary* is the description of the processes and flows which are included in the study according to the goal (Matthews et al., 2014, p. 90). One common approach to quantifying the system boundary is the cradle approach, where the delimitation is set along the life cycle stages of the product system (Verghese & Carre, 2012).

The next stage of the LCA study is the inventory analysis. During this stage, the data about the flows within and outside the product system is collected, and the inputs and outputs are quantified (ISO 14040:2006, p. 13). The inventory analysis is an iterative process, so as more data is collected, new limitations and requirements are identified, which require adaptation of processes as well as the goal and scope of the study (ibid.). The inventory analysis process is done in several stages: data collection, data calculation, and allocation of flows and releases. During data collection, practitioners collect and classify all information about unit processes within the system boundary. Then, the data is validated and related to unit processes and reference flows of the functional unit (ibid.). Lastly, in case of multiple outputs from one process, the allocation procedures are considered and performed. As a result, a life cycle inventory table is formed, which aggregates the emissions across substances and resources involved in the processes of the analysed system (Jolliet et al., 2016, p. 105).

After completing the inventory, the potential environmental impacts of the system are estimated against a set of impact categories and indicators. This process is called Life Cycle Impact Assessment (LCIA) (ISO 14040:2006, p. 14). There are many LCIA methods available which allow the quantification of the environmental impacts across different categories and indicators. However, they all follow a similar general process (Jolliet et al., 2016, p. 107). First, the results of the inventory analysis, which have similar effects, are aggregated under the *midpoint categories,* emphasising the fact that they are between the inventory and the final damage quantification (ibid.). Here, the impacts are represented in terms of the midpoint indicators, and the contribution of each flow is calculated through a *characterisation factor*. The midpoint categories are then aggregated into *damage categories*, which are quantified by *damage indicators* and correspond to one of the different *areas of protection* (ibid.). It is important to keep in mind that the aggregation of impacts introduces uncertainty, and it grows the further practitioners move from the inventory to the damage categories. In some cases, to aid the interpretability of the results, practitioners opt for one of the following approaches: *normalisation* (the individual impacts are represented relative to the total impact of the category), *grouping* (prioritisation of results by sorting or ranking) or *weighing* (assigning "value" to the impact within an aggregated indicator) (Jolliet et al., 2016, pp. 112-114).

The last phase of LCA is interpretation. As emphasised before, interpretation is the only phase performed during each of the three phases of the LCA. The main purpose of the interpretation phase is thus to identify the hotspots in the product system's life cycle and assess the quality and robustness of the analysis results (Jolliet et al., 2016, p. 149).

2.3 Biodiversity assessment in LCA

Within the scope of LCA, biodiversity is mostly assessed as part of the 'Natural Environment' protection area through quantification of the negative impacts on the natural ecosystems, functional and structural, as a result of exposure to human activity (Joint Research Centre, Institute for Environment and Sustainability, 2010b, p. 19). Despite biodiversity having many dimensions – ecological diversity, population diversity and genetic diversity – the current LCA and LCIA frameworks do not cover them entirely, largely focusing on the species abundance and using the Potentially Disappeared Fraction (PDF) of species concept for impact quantification (Joint Research Centre, Institute for Environment and Sustainability, 2010b, p. 19; Crenna et al., 2020). The review of biodiversity assessment methods by Damiani et al. (2023) concluded that the majority of the assessed methods rely on PDF to represent the impacts of biodiversity. The study also revealed another indicator that some of the LCIA methods use for quantification of biodiversity impacts – Mean Species Abundance (MSA).

The PDF can be understood as the fraction of species which have a high probability of no occurrence within a specified region due to varying stressors (Joint Research Centre, Institute for Environment and Sustainability, 2010b, p. 21). The PDF quantifies the species richness decline in the range between 0 and 1, where 0 represents the complete intactness of the original species and 1 represents the complete disappearance of species (Kuipers et al., 2025). The MSA, on the other hand, quantifies the abundance of original species in a disturbed environment relative to their abundance before the disturbance. Despite two indicators being used by LCA methods to quantify the biodiversity impacts, they measure different aspects. The PDF measures loss of species while the MSA quantifies abundance of species (ibid.). As such, the indicators tell different stories – the decline in abundance of species usually precedes the complete extinction of species, so it is expected that the "footprint" from using MSA as an indicator will be larger than from using PDF. It also means that MSA can capture the loss of biodiversity earlier than PDF, leading to better quantification of biodiversity impacts, especially the ecosystem services and multifunctionality (Marques et al., 2021).

Despite the LCA methodology being widely recognised as a keystone approach to quantifying various impacts of different products and systems, it cannot capture the complexity of biodiversity loss (Damiani et al., 2023). Among the aspects which are not sufficiently covered in any of the operational LCIA methods are the *overexploitation* and *invasive species*. Another aspect, not covered by the LCIA methods, is *Ecosystem Multifunctionality* (EM) – the ability of the ecosystems to deliver more than one service (Marques et al., 2021; Damiani et al., 2023). According to Marques et al. (2021), EM is an important and complementary factor to consider when assessing the ability of biodiversity to perform its functions. For the LCA methodology to incorporate biodiversity considerations at a higher level, the models should include more dimensions and drivers of biodiversity loss, which would require the development of additional quantification factors, as well as spatial details (Crenna et al., 2020).

Out of all assessed methods by Damiani et al. (2023), only Product Biodiversity Footprint (PBF) addresses overexploitation and invasive species, thus having the largest coverage over the drivers of biodiversity loss. Other methods covered in the investigation, including the most used ones like ReCiPe, LC-IMPACT, and Impact World+, mostly cover categories of land and water use, GHG emissions and nutrient emissions, but not the overexploitation, invasive species or EM (ibid.). Even PBF, which was considered to be the best performing, has issues due to the complexity of the method and requires further refinement (Asselin et al., 2020).

Another issue with biodiversity quantification is reliance on proxy metrics and aggregates such as PDF or MSA (Marques et al., 2021). While MSA can be considered a better indicator than PDF for biodiversity quantification (Kuiper et al., 2025), both of them quantify biodiversity from the position of species diversity and fail to describe other aspects of biodiversity, like genetic diversity or ecosystem changes (Damiani et al., 2023). To improve the current way of quantifying impacts, Marques et al. (2021) suggest that the indicators should be refined further to preserve the species granularity (while currently the "species" is an aggregated unit), include mechanistic models of biodiversity to account for species behaviour and dynamics, and expand the overall coverage of the indicators to include more realms and species groups.

2.4 Use phase biodiversity impacts of onshore wind power

While the environmental benefits associated with the use of wind power compared to fossil-based energy production systems are undisputable, the production of wind power requires large areas for the installation and operation of the wind farms, which in turn have negative effects on the surrounding areas. The impacts on biodiversity from wind power production can be roughly separated into two groups: (1) *direct effects from land use change* and (2) *indirect effects from fauna avoidance behaviour* (Gasparatos et al., 2017). Alternatively, Sander, Jung & Schindler (2024) have conducted a review of 152 scientific articles and have outlined several areas of impact of onshore wind power utilisation, which cover a wide range of environmental effects. Several of the identified impacts are linked to loss of biodiversity (see Table 1): *impacts on fauna* (birds, bats, insects and mammals), *impacts on land use* (habitat loss and fragmentation, impacts on forests and vegetation), and *other impacts* (microclimatic changes, noise pollution and indirect effects).

The observed pressures can be combined on a more general level (though not precisely) into the categories which have been identified by MEA (2005). As such, *land use change* (which includes habitat transformation and fragmentation) along with *climate change* play a significant role in driving the loss of biodiversity as a result of wind farm operation. Additionally, *invasive species* and *pollution* (specifically noise and vibration) drivers need to be included to account for indirect effects caused by land use change and turbine operation.

Table 1. Main drivers of biodiversity loss and pressures on biodiversity due to onshore wind farm operation (Sander, Jung & Schindler, 2024).

Category	Drivers	Pressures / Impacts on Biodiversity
Fauna	Wind turbine operation Sensory limitations of species	Collision mortality (especially birds and bats) Barotrauma from air pressure changes Avoidance behaviour leading to habitat displacement and interspecies competition Insect attraction and mortality, affecting pollination and food webs Mammal displacement and migration disruption
Land Use	Land clearing for infrastructure Expansion into natural habitats	Habitat loss and fragmentation (especially in forests, wetlands, grasslands) Disruption of ecological corridors and genetic exchange Edge effects altering species composition Vegetation modification impacting herbivores and pollinators
Microclimate Alteration	Wind-induced changes in local climate patterns	Increased nighttime temperatures Altered humidity and soil moisture Changes in plant growth and species suitability, leading to ecosystem imbalances
Noise and Vibration	Turbine operation	Disruption of communication and navigation (especially for echolocating species like bats) Reduced foraging efficiency Population dynamics changes due to avoidance or adaptation
Socioeconomic Drivers	Population growth Urbanisation Consumption and trade patterns	Indirect impact through increased land use demand and infrastructure development Policy and planning challenges from public opposition, potentially leading to rushed or poorly sited projects

2.4.1 Impacts on fauna

According to Sander, Jung & Schindler (2024), the most studied biodiversity impacts of onshore wind power utilisation are the effects on fauna, particularly on birds and bats, with some studies focusing on insects, pollinators and mammals. Wind turbines have significant impacts on the population of aerial species, like bats and different kinds of birds, with collision mortality being the most common cause for their decline. Some species rely on sensory input for orientation and do not have enough time to avoid the rotating blades of the wind turbine or are susceptible to "barotrauma" – a phenomenon caused by the pressure changes near the turbines, leading to internal haemorrhaging (ibid.). Certain species, like raptors and bats, are at high risk because of their life-history traits (low reproductive activity and long lifespans), making their populations vulnerable when mortality rates increase (Laranjeiro, May & Verones, 2018).

Additionally, some species employ avoidance behaviour – they shift their habitats and migratory routes to other areas away from turbines, which in turn causes them to migrate to lower-quality habitats and increases interspecies competition for resources, making them more open to threats from predators. The avoidance and displacement behaviours have been shown to reduce the activity of bats in the presence of turbines by 20 times as compared to sites devoid of wind farms (Millon et al., 2018). The displacement process constitutes functional habitat loss with long-term implications for populations of species, thus further intensifying indirect biodiversity impacts.

Insects are another taxon affected by wind energy production (Sander, Jung & Schindler, 2024). Research shows that insects are attracted to wind turbines due to heat emission and reflective surfaces, making them easy targets for predators or leading to their or their predator's demise through collisions with the blades (Laranjeiro, May & Verones, 2018). The decline in insect populations, in turn, has cascading effects on surrounding ecosystems, as insects play a vital role in pollination and food web stability.

Similar to aerial species, terrestrial mammals may also experience habitat displacement due to land-use changes in areas surrounding wind farms to accommodate necessary infrastructure. This leads to habitat fragmentation and alteration or even disruption of established migratory pathways for terrestrial species, further intensifying population stress (Sander, Jung & Schindler, 2024).

2.4.2 Impacts on land use

Another area of impact of wind power on biodiversity is land use. The construction and operation of wind farms demand reshaping and adjusting the landscape, leading to fragmentation and loss of habitats for many species (Sander, Jung & Schindler, 2024). The most affected areas are forests, which are cleared to accommodate wind farms and their infrastructure, altering the habitats through fragmentation and increasing the edge effects. This leads to a shift in forest-dependent species composition and exposes them to new external threats (ibid.). In non-forest areas, like wetlands, grasslands and coastal areas, similar effects are observed where wind farm construction and operation lead to the modification of local vegetation structures, subsequently affecting herbivore and pollinator communities. Such shifts in vegetation coverage impact population dynamics of herbivores, disrupt pollinator networks, and alter soil conditions, leading to broader ecological shifts across multiple trophic levels (Kati et al., 2021).

The "red thread" which connects all effects of the land use change is habitat fragmentation (Sander, Jung & Schindler, 2024). It disrupts the ecosystems on local and regional levels by disrupting the ecological corridors, preventing species from migrating, acquiring food and maintaining genetic diversity through interbreeding. Some species have managed to adapt to such changes, but others, specifically those that display specific habitat requirements, are under significant pressure (ibid.).

2.4.3 Other impacts

Beyond its direct effects on fauna and habitats, wind energy also brings about wider environmental changes that indirectly impact biodiversity (Sander, Jung & Schindler, 2024). One of them is *microclimatic alternation*.

The operation of wind turbines is associated with the modifications of wind patterns, atmospheric turbulence and surface temperature disruptions (Sander, Jung & Schindler, 2024). These effects are especially noticeable in areas which are characterised by stable atmospheric conditions, so wind farms can cause an increase in nighttime temperatures, a change in humidity levels and the ability of soil to retain moisture at a local level. In turn, this influences the plant growth rate and seasonal phenology and can create microhabitats which favour specific species while not being suitable for others (ibid.). The long-term impacts of microclimatic alterations are not fully understood yet, but the current understanding indicates cascading effects on the entire ecosystem.

Another aspect is the *noise and vibration pollution* caused by wind farms, which affects not only wildlife but also human communities (Sander, Jung & Schindler, 2024). Wind turbines generate low-frequency noise during operation, which impacts species that rely on echolocation and vocalisation for communication, navigation, and foraging. As mentioned earlier, bats and birds are among the most affected species, as the noise from turbines masks important auditory cues, leading to disorientation and reduced foraging efficiency (ibid.). Noise pollution influences reproductive behaviour, disrupts territory establishment and can cause stress in wildlife, having lasting effects on entire populations (Laranjeiro, May & Verones, 2018). Evidence suggests that some species have adapted by altering their vocalisation patterns to different frequencies or by avoiding wind farms completely, contributing to changes in population dynamics and alterations in food webs.

Lastly, the *social aspects* associated with the development of wind farms also contribute indirectly to the negative effects on biodiversity (Sander, Jung & Schindler, 2024). Public opposition to wind farm construction often arises from concerns about aesthetics, noise, and wildlife protection, which in turn influences the policy development process regarding the location and implementation methods of the project (ibid.). If biodiversity concerns are not adequately addressed during the planning stage, the rate of renewable energy adoption slows down. Conversely, when ecological concerns are sufficiently considered, project implementation can proceed without delays or disruptions. Therefore, developing renewable energy infrastructure should always strike a balance between biodiversity conservation and development (ibid.).

3 Methodology

This section describes the overall methodological approach used to achieve the objectives of the study. The schematic description of the steps undertaken and their interrelations can be seen in Figure 2 below. The following discussion will explain the process for the literature review and development of the assessment criteria.

Figure 2. Schematic representation of the methodological approach employed to achieve the objectives of the study.

The following discussion explains the process for the literature review and development of the assessment criteria. The processes of method selection & screening are explained in Section 4.

3.1 Literature review

The process begins with a literature review, which serves to inform the current research through systematic review and analysis of previous scientific works (Machi & McEvoy, 2016, p. 3). In this study, a literature review was used for two purposes: to identify the biodiversity impacts of onshore wind power during the use phase to inform the assessment criteria developed for the purpose of biodiversity impact method assessment and to identify relevant biodiversity assessment methods.

The literature review was performed as a two-stage process. First, the relevant scientific articles were identified and assessed. The articles were searched for in the scientific databases like Scopus, Web of Science, and KTH Library using different combinations of the following keywords: “biodiversity”, “ecosystem”, “assessment”, “methodology”, “approach”, “impact”, “onshore wind power”. After the initial search, it became apparent that certain limitations must be imposed to narrow down the scope of the analysis. As such, the exclusion criteria involved the removal of the literature published earlier than 2020 and articles concerning marine ecosystems, as they have no direct relevance to onshore wind power production. The final compilation of articles was then thematically separated into two categories – “environmental impacts of onshore wind power production on biodiversity” and “biodiversity assessment methods” to inform the two research objectives outlined at the beginning of the section.

3.2 Method selection

After identifying a list of biodiversity footprinting methods using a literature review, a list of methods covering a broad perspective was compiled. Since the focus of the current study is on the biodiversity impacts of onshore wind power production, certain requirements had to be satisfied by the selected methods. They need to be suitable for performing biodiversity assessment of impacts from onshore wind farm operation, thus allowing for quantification of energy system impacts, and include all relevant species.

As such, from the initial list of 41 tools, 12 were excluded from the final analysis. These tools were excluded because they either (1) *focused on aquatic species only*, (2) *did not support renewable energy system assessment,* (3) *did not support footprint quantification,* (4) *could not be classified as a “method”* or (5) *are add-ons to others.* This resulted in a list of 29 tools and methods (see Table 15 in Appendix 9.2), which were assessed against the criteria presented in the next section.

3.3 Assessment criteria development

To identify the relevant methods to be integrated into LCA, the assessment criteria were devised. As previously stated, the information used to inform the criteria dimensions was obtained through a literature review. The advantage of using the multi-criteria analysis (MCA) approach is that it allows to streamline the decision-making process when multiple aspects are involved by structuring them in a matrix where the performance of the subject under study can be benchmarked against the identified criteria (Mateo, 2012, pp. 7-8).

Following the framework of similar works, which assessed the different methods for their suitability to be integrated into LCA, specifically Crenna et al. (2020), Damiani et al. (2023) and Bergman et al. (2024), the criteria was constructed to include three perspectives: *inclusion of relevant biodiversity aspects, methodological aspects,* and *LCA compatibility.* The evaluation criteria are summarised in Table 2 below.

Table 2. Criteria for method assessment.

Criteria	Sub-criteria	Evaluation aspects
1. Inclusion of biodiversity aspects	*1.1 Drivers of biodiversity loss*	1.1.1 Land use change 1.1.2 Habitat fragmentation (incl. change) 1.1.3 Direct exploitation 1.1.4 Invasive species 1.1.5 Pollution 1.1.6 Climate change 1.1.7 Ecosystem multifunctionality
	1.2 Species	1.2.1 Inclusion of species in assessment framework
	1.3 Habitats	1.3.1 Inclusion of habitats in assessment framework
2. Methodological aspects	*2.1 Scalability*	2.1.1 Spatial and temporal scales
	2.2 Data availability & Documentation	2.2.1 Availability of required input data 2.2.2 Use of public databases 2.2.3 Ease of use 2.2.4 Clarity and availability of documentation
3. LCA compatibility	*3.1 Quantification of results*	3.1.1 Output expressed in numerical form 3.1.2 Unit of quantification
	3.2 Link to LCI	3.2.1 Supports development or use of characterization factors 3.2.2 Supports midpoint or endpoint quantification

The first dimension is concerned with *drivers for biodiversity loss* as defined by MEA (2005), aspects defined by Marques et al. (2021) as crucial for inclusion in LCA-based assessment of biodiversity footprint (like ecosystem multifunctionality), *coverage of relevant species* and *habitats,* which are specific to the case of onshore wind power production. The choice to rely on the drivers defined by MEA instead of drivers specific to the wind power production is grounded in the results of the literature review, which confirms that all MEA drivers are relevant for the onshore wind power case. The driver for *habitat fragmentation* (which has been separated in the current assessment) is included under the *land use* driver, and the *pollution* driver includes all aspects ranging from aspects pertaining to air quality, ecotoxicity, eutrophication to noise, light and other disturbances (Millenium Ecosystem Assessment, 2005). So current set up allows to capture make conclusions regarding the state of biodiversity loss assessment both on a wind power specific and on a more general levels allowing for broader contribution potential.

For methodological aspects, the focus will be on *scalability*, i.e. usability at different spatial and temporal resolutions, *data availability* and *extent of documentation*. The last two aspects of methodology are especially important as they constitute the extent to which the methodology is easy to use and adapt to the needs of practitioners. Lastly, for the third dimension of LCA compatibility, the focus will be on the *quantifiability* of the results of the assessment and the *possibility to link the data to LCI approaches*.

Similarly to the first dimension, a more generalist approach has been adopted with the evaluation aspects for the last two dimensions. The reason behind such choice is motivated by the desire to leave the assessment on a broad and more general level. The main focus of the thesis is to provide an assessment of existing biodiversity footprinting methods and use the findings from this to suggest improvement pathways for existing LCA frameworks. So, if the methodological compatibility criteria was more focused on LCA, many of the methods which might provide additional perspective would have been "screened out" as non-compatible, which is not the aim of the thesis.

During the evaluation of methods against the criteria, the assessment was conducted qualitatively for each of the criteria.

3.4 Case study

During the literature review, it became apparent that the methodological differences span not only between biodiversity footprinting approaches from different categories (e.g. LCA-based and remote sensing approaches) but also between the methods following similar approaches, as is the case with LCA (Bromwich et al., 2025). To analyse and illustrate these differences between LCIA methods, a case study has been conducted.

The case study approach allows practitioners to design the setup in a way that reflects the conceptual understanding formulated in the literature and highlights the specific aspects that the study aims to explore (Pan & Tan, 2011). This supports the overall explorative nature of the research aims, which, among other things, aspire to explore the influence of methodological assumptions on the outcomes of an LCA study. By contextualising the theoretical understanding with the real-life application, the case study provides actionable insights for both researchers and practitioners (ibid.).

As such, the case study focused on a comparative evaluation of two alternative constructions of wind turbine tower – one made of steel, and one made of wood – assessed using two LCIA methods – ReCiPe 2016 (H) and IMPACT 2002+. The setup of the LCA study follows the ISO 14040 (2006) standard. The more in-depth explanation of the case study is provided further in Section 5.

4 Results – biodiversity assessment

The literature on biodiversity assessment methods classifies them into two categories – *within LCA* and *beyond LCA* (Crenna et al., 2020; Damiani et al., 2023). The first category includes the operational LCIA methods and approaches, which include biodiversity assessment in their endpoint quantification. The second group is comprised of methods which quantify the biodiversity impacts from a life cycle perspective but use alternative approaches to quantifying the life cycle inventory. These methods include multiregional input-output (MRIO) or biophysical accounting tools and approaches (Crenna et al., 2020; Barth et al., 2024). Additionally, the beyond-LCA category includes ecosystem service (ES) accounting approaches, which are developed for specific context applications.

The biodiversity quantification within LCA has been discussed in Section 2.3. The following discussion will cover how biodiversity is addressed within beyond-LCA category.

4.1 Biodiversity assessment beyond LCA

As stated before, apart from the LCA-based approaches, biodiversity can be assessed through other methods which either partially rely on the LCA methodology or use completely different quantification and measurement approaches (Crenna et al., 2020). These tools can be roughly separated into MRIO modelling tools or biophysical accounting methods (Barth et al., 2024). Some methods provide the quantification of biodiversity impacts using financial flows between regions, allowing for a proxy assessment of biodiversity through economic activity, species extinction or abundance. Alternatively, biodiversity is assessed through geodata, where the geographic information systems (GIS) approaches are common (ibid.). They allow practitioners to capture spatial parameters, relationships and patterns related to species and ecosystems.

To aid the decision-making process in organisations on different levels, the World Wide Fund for Nature (WWF), Science Based Targets Network (SBTN) and IBAT Alliance have developed their guidelines and suggestions on how to approach biodiversity assessment. WWF has developed the biodiversity impact assessment framework, where they lay down principles and approaches for informing stakeholders regarding the potential biodiversity benefits from investments in enterprises and projects (WWF Switzerland & The Biodiversity Consultancy, 2024). The SBTN has developed a toolbox which contains resources which are supposed to help organisations assess their environmental impacts, including biodiversity (SBTN Step 1 Toolbox, 2024). This toolbox contains a list of 45 tools which allow for assessing biodiversity impacts from three categories – species, ecosystems and nature's contribution to people (NCP). The IBAT Alliance has developed the Integrated Biodiversity Assessment Tool, which allows organisations to estimate early-stage biodiversity-related risks through spatial data for terrestrial and marine ecosystems (De Ryck et al., 2024).

4.2 Biodiversity footprinting method assessment

As a result of the literature review, a list of 29 biodiversity footprinting methods has been compiled. These methods correspond to the general beyond-LCA category and can be separated into five groups: MRIO tools utilising life cycle perspective, portfolio and risk dashboards, spatial tools (incl. remote-sensing approaches), and ecosystem-service assessment tools. The magnitude of each group, as well as the methods which correspond to them, can be found in Table 3 below.

Table 3. Distribution of methodological types for final analysis.

Group	Methods	Number of methods
MRIO tools	Biodiversity Footprint Method (BFM); BIGER Footprint; Biodiversity Impact Assessment tool (BIAT); BioScope; Corporate Biodiversity Footprint (CBF); Environmental Profit & Loss (EP&L); Global Biodiversity Score (GBS); Global Impact Database (GID); GLOBIO 4; LUCI-LCA; Product Biodiversity Footprint (PBF)	11
Spatial tools (incl. remote-sensing)	IBAT; SLAM; Biodiversity Net Gain Calculator (BNGC); Statutory Biodiversity Metric (UK); High-Conservation-Value (HCV) Approach; Site Biodiversity Footprint (SBF), Leeana; Xylo Systems; Trend.Earth	9
Portfolio & risk dashboards	Biodiversity Risk Filter (BRF); ENCORE; Nature Risk Profile (NRP); Link; Nala.Earth	5
Ecosystem-service assessment tools	InVEST (habitat & collision modules); ECOPLAN-Scenario Evaluator; Ecosystem Intelligence Platform (EIP); TESSA	4

The following discussion will cover the assessment based on each of the three criteria dimensions – inclusion of biodiversity aspects, methodological aspects and LCA compatibility.

4.2.1 Criteria 1 – Inclusion of biodiversity aspects

The first criteria for biodiversity footprinting methods assessment are concerned with coverage of biodiversity loss driver, species and habitats by the methods which are being assessed. First, the result presentation will cover the drivers.

Among the biodiversity drivers, the most addressed one is *Land-use change* (see Table 4). It has been included in every assessed method to a varying degree. It is important to mention that *Link* provides an assessment of land use change only through the Land occupation aspect (De Ryck et al., 2024). All methods except two (*GLOBIO 4* and *HCV Approach*), which include *habitat fragmentation* (a total of 15 methods currently include this driver), assess it as part of the *land-use change* driver. Two more approaches (*ECOPLAN-SE* and *EIP*) do not currently include quantification of impacts associated with habitat fragmentation, but the implementation is planned during the next methodological update cycle. Approximately half of the assessed methods, which include *direct exploitation* driver (a total of 21 include this driver) for impact quantification, do so using *water use* metric, while the other half also includes other resources and species. *Invasive species* are assessed by 12 methods, with *CBF* providing quantification of impacts only from the transport sector. *Pollution* is included by 24 methods to various degrees of completeness. The majority of methods focus on acidification, eutrophication, ecotoxicity on different ecosystem levels, and non-GHG air pollution, while some (six methods) also include pollution from noise, light and other disturbances. Two methods, *BNGC* and *SBMT,* quantify pollution exclusively by noise, light and disturbance pollution, while *GBS* quantifies this type of pollution through terrestrial encroachment pressures. *Climate change* is included in 20 methods, the majority of which quantify it on the terrestrial ecosystem level. *Ecosystem multifunctionality* is assessed by 19 methods, for all of which it is included through the consideration and quantification of impacts on ecosystem services. It is important to mention that two methods, *ENCORE* and *SBF,* included additional drivers and pressures which were not included in the assessment criteria, but which are very important for the case on onshore wind power production – *collisions and electrocutions of fauna* (De Ryck et al., 2024).

Table 4. Criteria 1: Biodiversity aspects included in the biodiversity footprinting methods. The symbol "X" denotes that the aspect is being covered within the assessed method, and "-" means the aspect is not being covered. In some cases, additional information is added instead of the symbol, meaning additional clarifications are required. For *Land-use change*, "Land occupation" is the only aspect covered by *Link* method within this category; all other methods provide sufficient coverage of this driver. *Habitat fragmentation* is largely included as part of the *Land-use change* driver as a sub-category; for two methods its implementation is planned for future updates. "Water" is included for all cases when it is the only aspect which assesses *Direct exploitation* driver. The "transp. sector" in *Invasive species* means that only impacts of transport are included in the assessment for this driver. In case of *Pollution* when the text in brakets says "incl. noise" that means that among others, noise and light pollution as well as other disturbances are included; in all other cases, the text in brakets indicates the only categories which are included in this driver.

	Drivers of biodiversity loss							**Species**	**Habitats**
	Land-use change	*Habitat fragmentation*	*Direct exploitation*	*Invasive species*	*Pollution*	*Climate change*	*EM*		
BFM	X	(part of LUC)	(Water; NL)	-	X	X	X	X	X
BIGER	X	-	(Water)	-	X	X	-	X	-
BIAT	X	-	(Water)	X	X	X	X	X	-
BNGC	X	(part of LUC)	-	X	(Noise)	-	-	X	X
BRF	X	(part of LUC)	X	X	X	X	X	X	X
BioScope	X	-	(Water)	-	X	X	-	X	-
CBF	X	(part of LUC)	(Water)	(transp. sector)	(incl. noise)	X	X	X	X
ECOPLAN-SE	X	(planned)	(Water)	-	(incl. noise)	X	X	-	X
EIP	X	(planned)	-	X	(incl. noise)	X	X	-	X
ENCORE	X	-	X	X	X	-	X	X	-
EP&L	X	-	-	-	X	X	-	X	-
GBS	X	(part of LUC)	X	-	(Enroachment)	X	X	X	X
GID	X	(part of LUC)	(Water)	-	X	X	X	X	X
GLOBIO 4	X	X	-	-	X	X	-	X	-
HCV Approach	X	X	-	-	-	-	X	X	X
IBAT	X	-	X	X	(nitrogen atm; nutrients to water)	X	-	X	X
InVEST	X	(part of LUC)	X	-	(incl. noise)	X	X	X	X
LUCI-LCA	X	-	-	-	-	-	X	X	-

Leeana	X	(part of LUC)	X	-	X	-	X	X	X
LIFE	X	-	(Water)	-	(Waste)	X	X	X	X
Link	(Land occupation)	-	X	X	-	-	X	X	X
Nala.Earth	X	-	(Water)	-	(incl. noise)	-	X	X	-
NRP	X	(part of LUC)	(Water)	-	-	X	X	X	X
PBF	X	-	X	X	X	X	-	X	
SBF	X	(part of LUC)	X	X	(incl. noise)	X	-	X	X
SLAM	X	-	(Water)	-	X	X	-	X	-
SBMT	X	(part of LUC)	-	X	(Noise)	-	-	X	X
TESSA	X	(part of LUC)	X	-	(Non-GHG; nutrients to water)	X	X	X	X
Trend.Earth	X	-	-	-	-	X	-	-	-
Xylo Systems	X	(part of LUC)	-	X	-	-	X	X	X

All but three methods (*ECOPLAN-SE*, *EIP*, and *Trend.Earth*) include the quantification of *species* in their impact assessment (see Table 4). On the other hand, the quantification is largely based on aggregated parameters, so during the assessment, it will be difficult to understand which groups of species are affected and to what degree. Similar pattern is observed for the inclusion of impacts on *habitats* – all of the 19 methods which include them in the quantification assess the impacts through proxy metrics which aggregate the impacts under broader categories. It is important to mention that the majority of methods included the overall quantification of biodiversity impacts in terms of abstract metrics, like risk, coverage or points, or established metrics for quantifying the species abundance. Only two methods (*LIFE* and *Xylo Systems*) addressed another dimension of biodiversity – genetic diversity, thus covering two out of three dimensions.

4.2.2 Criteria 2 – Methodological aspects

When it comes to methodological aspects (see Table 5), the majority of methods cover wide spatial scale (from local to global), four methods (BNGC, Statutory Biodiversity Metric Tool, HCV Approach, and Site Biodiversity Footprint) are site-specific, and one method (Trend.Earth) provide only landscape-level overview. When it comes to time resolution, the majority of methods provide a static overview, while eight methods allow the creation of annual dashboards and monitor the evolution of impacts over time. Among them are GLOBIO 4, Trend.Earth, Global Impact Database, Nala.Earth, Ecosystem Intelligence Platform, Leeana, Xylo Systems and Link.

Table 5. Criteria 2: Methodological aspects related to assessed biodiversity footprinting methods. For "Availability of data": Y = data is available; P = requires additional user input; N = not available. For "Ease of use": Y = data is available; P = summary only; N = not available.

	Scalability	**Data availability & doccumentation**			
	Spatial/temporal scales	*Availability of data*	*Use of public databases*	*Ease of use*	*Documentation*
BFM	Site → corporate; snapshot & user scenarios	Y	GLOBIO, FAOSTAT	Y	Open
BIGER	Company → global portfolio; annual reporting	P	LC-IMPACT CF set	P	Fee
BIAT	Company → portfolio; annual	P	EXIOBASE v3, IMPACT World+	P	Fee
BNGC	Project site; construction & 30-yr management	P	UK habitat & condition tables	Y	Open
BRF	Asset → global supply chain; current year	Y	HydroSHEDS, KBA, WDPA	P	Open (web)
BioScope	Supply chain, investor fund; single year	Y	EXIOBASE 3.4, ReCiPe 2016	Y	Open
CBF	Company → portfolio; annual & 2030 target	P	GLOBIO curves, MRIO trade	P	Fee
ECOPLAN-SE	Municipality → regional plan; 5- to 30-yr scenarios	P	InVEST biophysical coeffs	Y	Open
EIP	Site; real-time to annual	P	Sentinel-2, WDPA	N	Fee
ENCORE	Sector, corporate; present-day	Y	WDPA, WRI Aqueduct, EXIOBASE	Y	Open
EP&L	Product → corporate group; fiscal-year	Y	ecoinvent, ReCiPe	Y	Open
GBS	Site → corporate; annual & scenario	Y	GLOBIO, ecoinvent	Y	Open
GID	Country → sector; 2014–2022 time-series	Y	EXIOBASE, LC-IMPACT	P	Fee (API)

GLOBIO 4	Pixel (300 m) → global; 2015–2100	P	HYDE 3.2, RoadNet, CLM GHG	Y	Open
HCV Approach	Site estate; 5-yr management cycle	P	WDPA, KBA (support layers)	Y	Open
IBAT	Single asset; real-time download	Y	WDPA, KBA, Red List	Y	Fee (low)
InVEST	Parcel → landscape; annual & scenario	P	GlobCover, FAO soils (optional)	Y	Open
LUCI-LCA	Country / scenario; 2000–2050	P	GLOBIO, InVEST land-change	Y	Open
Leeana	Parcel (10 m) → regional; quarterly	N	Sentinel-2, GBIF	N	Fee
LIFE	Facility → corporate; annual	P	internal KPIs + optional public stats	Y	Open
Link	Facility → corporate; present	Y	WDPA, GBIF, Copernicus	P	Fee
Nala.Earth	Site → supply chain; quarterly & target years	Y	ESA CCI, WDPA, TNFD ref layers	P	Fee
NRP	Listed company; annual	Y	EXIOBASE, Aqueduct	P	Fee
PBF	Product system; cradle-to-gate year	P	LC-IMPACT CFs	Y	Open (report)
SBF	Facility; baseline & 10-yr action plan	P	GLOBIO, on-site surveys	P	Fee
SLAM	Asset; snapshot	Y	WDPA, KBA, GLAD forest loss	N	Fee
SBMT	Project site; baseline + post-dev 30 yr	P	UK habitat maps	Y	Open
TESSA	Local habitat; present + one alternate scenario	N	Field plots, local stats	Y	Open
Trend.Earth	Landscape → national; 2001–present yearly	Y	ESA CCI land-cover	Y	Open
Xylo Systems	Asset buffer; monthly updates	Y	GBIF, IUCN species, Sentinel-2	N	Fee

Most of the assessed methods (26 out of 29) rely on publicly available databases for their analysis (see Table 5). The most common databases are the World Database on Protected Areas (WDPA), used by nine methods, the GLOBIO evidence base, used by six methods, and EXIOBASE MRIO tables are used by five methods. In addition to data from publicly available sources, 16 methods require primary data to perform calculations and assessments.

When it comes to the availability of methodology, 16 methods have their documentation published, including full equations and characterisation factor tables and are available for free. The remaining 14 methods require a fee for their use, out of which nine have published the summary of the approaches, and five methods do not have data publicly available. For these methods, the assessment was based on information provided by De Ryck et al. (2024) in their report "Assessment of Biodiversity Measurement Approaches for Businesses and Financial Institutions".

4.2.3 Criteria 3 – LCA compatibility

Despite assessing some of the drivers of biodiversity loss qualitatively, 23 out of 29 methods provide numerical output to quantify the results (see Table 6). Out of 23 methods, 13 methods use PDF- or MSA-based quantifications, five methods assess impacts as a risk % or dimensionless score, three methods use ecosystem service indices, and two methods quantify impacts based on biodiversity units. Methods like Biodiversity Risk Filter, ENCORE, HCV Approaches, SLAM and IBAT provide qualitative quantification with no numerical output.

The characterisation factors are supplied by 13 methods. These are the methods which rely on LCA logic to perform the calculations and use LCI databases. There are also five methods which support the development of characterisation factors by the user. The remaining 12 methods do not support the development of characterisation factors. Most of the methods (17) support the quantification on the endpoint level, while only 5 support the quantification on the midpoint level.

Table 6. Criterion 3: LCA compatibility of biodiversity footprinting methods. For "Numerical output": Y = output is quantified, N = not available. For "Characterisation factors": Y = available, P = supports development, N = does not support development.

	Quantification of results		Link to LCI	
	Numeric output	*Unit*	*Characterisation factors*	*Mid- / End-point quantification*
BFM	Y	MSA·ha	Y	End
BIGER	Y	PDF * km²	Y (LC-IMPACT)	Mid
BIAT	Y	PDF/MSA per company portfolio	Y (IMPACT World+)	Both
BNGC	Y	Biodiversity units	N	End (site-units only)
BRF	N	1-5 risk score	N	–
BioScope	Y	PDF years	Y (ReCiPe)	Mid
CBF	Y	MSA·km²	Y (GLOBIO CF set)	End
ECOPLAN-SE	Y	ES scores & maps	P	End
EIP	Y	ES indices	P	End
ENCORE	N	Qualitative risk levels	N	–
EP&L	Y	€ cost / pressure; MSA	Y (internal CF tables)	Both
GBS	Y	MSA·km²	Y	End
GID	Y	PDF yrs & €	Y (LC-IMPACT)	Mid
GLOBIO 4	Y	MSA (%)	Y (per pressure)	End
HCV Approach	N	Qualitative HCV map & mgmt plan	N	–
IBAT	N	Presence / STAR units	N	–
InVEST	Y	Service-specific (e.g., tons C, habitat ha)	P	End
LUCI-LCA	Y	ΔMSA (%) / ES indices	Y (GLOBIO-linked)	End
Leeana	Y	Biodiversity condition %	N	End
LIFE	Y	LIFE score	N	End
Link	Y	Multiple KPIs incl. MSA & ES value	Y (GLOBIO / LC-Impact)	End
Nala.Earth	N	Risk/impact dashboard	Y (bespoke CF library)	Both
NRP	Y	Scores per driver & revenue-at-risk	N	–
PBF	Y	PDF·year single score	Y (LC-Impact)	Mid

SBF	Y	km²·MSA·year	P	End
SLAM	N	Exposure score	N	–
SBMT	Y	Units (ha × condition × distinctiveness)	P	End (habitat units)
TESSA	Y	ES physical & $ value	N	End
Trend.Earth	Y	% degraded / improved land	N	Mid (land-condition)
Xylo Systems	Y	Condition & risk scores	N	End

4.2.4 Summary – methods suitable for LCA integration

As evident from the above analysis, there are 14 methods which provide readily available characterisation tables and five more methods which support the development of characterisation factors by the user. Out of these 19 methods, only Nala.Earth does not provide numerical output, meaning the remaining 18 methods have the potential to be incorporated into the LCA framework. In this selection, eight methods require a fee for their implementation and provide only a top-level description of their methodology. This results in 10 methods which can be deemed suitable for future integration into the LCA framework (see Table 7).

Table 7. Final selection of methods which are suitable for integration in the LCA framework.

Method	Output units	Addressed biodiversity aspects
	MRIO tools (LC-based)	
BFM	km²/ha MSA	Land-use change, habitat fragmentation, direct exploitation, pollution, climate change, ecosystem multifunctionality; species; habitats
BioScope	PDF year	Land-use change, direct exploitation, pollution, climate change; species
EP&L	PDF/MSA €	Land-use change, pollution, climate change; species
GBS	MSA km²	Land-use change, habitat fragmentation, direct exploitation, pollution, climate change, ecosystem multifunctionality; species; habitats
GLOBIO 4	Δ MSA (% or grid values)	Land-use change, habitat fragmentation, pollution, climate change; species; habitats
LUCI-LCA	Δ MSA per ha	Land-use change, ecosystem multifunctionality; species
PBF	PDF year	Land-use change, direct exploitation, invasive species, pollution, climate change; species; habitats
	Ecosystem service assessment tools	
InVEST	Habitat-quality index (0–1) or Δ-species abundance	Land-use change, habitat fragmentation, direct exploitation, pollution, climate

		change, ecosystem multifunctionality; species; species; habitats
ECOPLAN-SE	Biodiversity intactness multifunctionality index (%)	Land-use change, direct exploitation, pollution, climate change, ecosystem multifunctionality; habitats
	Spatial tools	
SBMT	Biodiversity units (area × condition × distinctiveness)	Land-use change, habitat fragmentation, invasive species, pollution; species; habitats

The majority of the methods which are ready for integration into LCA framework come from the MRIO tools category and are based on the life cycle thinking approach, which makes their future integration much more simple. The remaining three methods are not based on life cycle thinking, but are standalone methods which are either ecosystem service assessment specific or are from the spatial domain.

When it comes to the indicators used for quantification of impacts, there is a clear dependance on the category which the method belongs to. All of the suitable MRIO methods rely on impact quantification through PDF- or MSA-based metrics. The ecosystem service assessment tools quantfy the impacts through metrics which are better suited for their purpose – either quantifying the level of habitat quality, species or biodiveristy intactness index which have been shown to have good compatibility with this domain (Marques et al., 2021). The spatial tool has employed the quantification of self-derived unit based on area, condition and distinctiveness.

Among the drivers for biodiversity loss, invasive species remains the most underaddressed with just two methods covering the impacts – *PBF* and *SBMT.* On the other hand, the drivers which are addressed the most are land use change, pollution and climate change. The rest of the drivers are addressed by half of the methods each. For all methods which cover species (all except *ECOPLAN-SE*), the quantification of the impacts happens on the aggregated level through proxy metrics (usually they coincide with the impact quantification unit). The habitats are explicitly covered in half of the methods and quantification is also done on the aggregated level.

As discussed before, biodiversity is a concept which has multiple dimentions which cover species, genetic variations and ecossytems (IPBES, n.d.). The purpose of assessing the beyond-LCA methods for their suitability for LCA was to improve the current framework and address the impacts and biodiversity dimentions which are currently underaddressed. The literature and the results of this study has shown that both LCA and beyond-LCA methods provide sufficient coverage of the species diversity dimension of biodiversity. The ecosystem health, despite general concensus in the examined literature, is covered by two methods exlusevely and is included in the assessment of other methods through proxy metrics like MSA. Unfortunately, there was no evidence of inclusion of genetic diversity by any of the "suitable" biodiversity footprinting methods.

4.3 Integrating biodiversity into LCA

On a general level the results show that there is a great potential for integration of biodiversity impacts into the LCA framework. On the other hand, the full integration of the biodiversity impacts , especially those which are relevant to the wind power production, remain a challenge. The majority of methods which have been selected as compatible with LCA utilise the life cycle thinking approaches and rely on LCI for impact quantification, so they suffer from the same drawbacks as the general LCA framewrok albeit can still provide additional insights on top of generic LCIA methods.

So, LCA methodology requires further development to capture its dynamic and complex nature and refine the existing cause-and-effect relationships through the development of new characterisation factors (Crenna et al., 2020). The even with support for characterisation factor development on the methodological level, it is a long and difficult process, so to attempt and bridge the existing gap, the practitioners can employ the following framework to better account for biodiversity impacts during their LCA studies.

The first step in the process would be to carefully identify and map all relevant biodiversity pressures along the whole life cycle of the system. For onshore wind power, land use change, habitat fragmentation and noise pollution would be the primary concerns. The practitioners need to consider the ecological context of these pressures in order to be able to correctly establish the damage pathways and cause-and-effect relationships.

In *the second step*, the practitioners need to connect the LCI flows to the identified pressures and understand which impact categories are necessary. Since the LCA methodology aggregates the impacts into single scores, practitioners need to consider integrating additional metrics and data to capture aspects not available for the conventional LCA methodology. For example, they can use spatially explicit data from GLOBIO to enrich the existing inventory.

As identified during the method review, existing methods do not cover all relevant biodiversity loss drivers. To combat this, when selecting the methods in *the third step*, practitioners should consider combinations of different methods to support the complexity of the analysis. The ultimate goal is not to replace the LCIA methods, but to enhance the findings from them with information and data from other methods, which would support the local spatial dynamics and risk exposures. It is important to remember, however, that the methods should be relevant to the system under study and have solid scientific grounding. Otherwise, the credibility of the study might be called into question.

5 Results – case study of a wind farm

To better understand the performance of different LCIA methods to assess the impact of biodiversity loss, a case study has been conducted. The following sections explain the set-up of the LCA study as prescribed by the ISO (14040:2006) standard. The analysis has been performed using the Activity Browser – a graphical interface to Brightway2, a Python-based framework to perform advanced LCA calculations (Steubing et al., 2020).

5.1 Goal and scope

The goal of the LCA study to compare the impacts on biodiversity associated with a wind power plant using either steel or wood as main material in the tower. Additionally, two different LCIA methods were used to evaluate the influence of methodological choice on the final result. The reason behind this is to verify whether practitioners can draw similar or different conclusions when using different LCIA methods. The analysis will compare the magnitude of the difference in outputs for two different configurations of the tower using two different LCIA methods to assess the degree of similarity between them.

The attributional LCA approach has been chosen for this study. The attributional LCA approach assessed the causes of the impacts thus looking in the "past", while consequential approach examines the effects of the changes thus looking in the future (Arvidsson, Sanden & Savnström, 2023). Additionally, the attributional approach allows to assess the share of total impacts associated with different product systems, instead of investigating the consequences of technological development in the consequential LCA approach (Schaubroeck et al., 2021). Since the case study aims to investigate the causes of the impacts rather than the effects of the two alternative configurations, this makes the attributional approach to LCA more suitable for the study objectives.

The *functional unit* of the study is 1 MW installed on a generic wind power plant in Sweden. The system boundary of the study is set to cradle-to-gate, excluding all stages after construction of the wind farm due to limitations in data availability and to reduce the complexity of the system (see Figure 3). Additionally, all processes related to transportation of the components between and during the life cycle stages and the production of energy required for the completion of the stages has been excluded from the boundary. Such choice is motivated by a few factors. First of all, previous LCA studies on the topic (Oebels & Pacca, 2013) have shown that the majority of the impacts are arrtibuted to the raw materials and manufacturing of components stages. Moreover, the effects of the material substitution for the tower construction are noticeable only in these stages as the remaining stages are unaffected by the material choice.

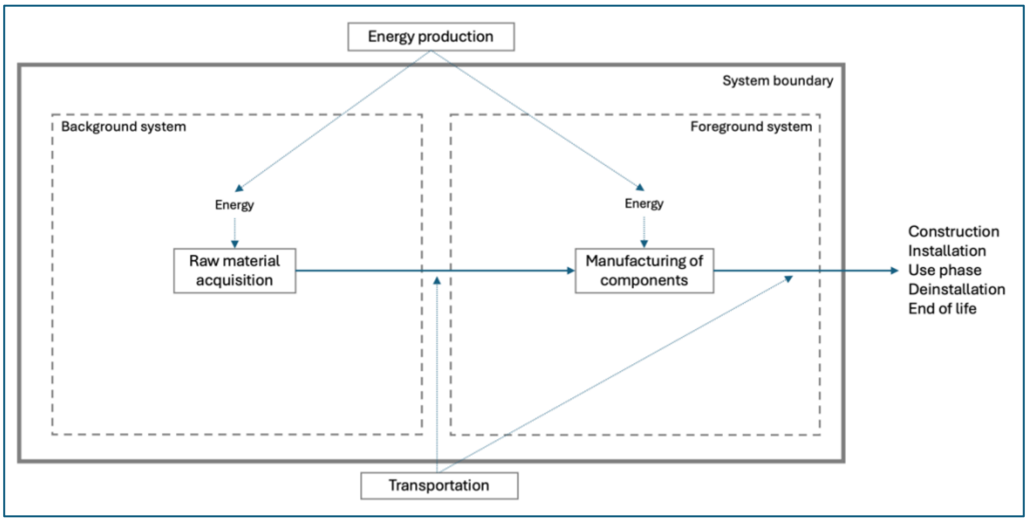

Figure 3. System diagram for the LCA study of a generic Swedish wind power farm.

The foreground information used in the study is based on the Renewable Energy Materials Properties Database (REMPD), which was transformed into an LCI format, connecting it to the ecoinvent 3.9.1 (cut-off) database. The REMPD was developed by the US National Renewable Energy Laboratory (NREL) and is publicly available (Cooperman et al., 2023; Eberle et al., 2023). This database includes information and bill of materials of a generic onshore wind power plant that consists of the following components: array and export cables, roads, towers, blades, hub, nacelle, foundation, and substation.

Two LCIA methods have been chosen for this study: ReCiPe 2016 v1.03 Endpoint (H), IMPACT 2002+ (Endpoint). These particular methods have been chosen for the study because they are the only LCIA methods implemented in Activity Browser which offer adequate coverage of biodiversity-related impact categories to conduct a meaningful study. The endpoint quantification has been chosen because the quantification of pressures on the Ecosystem Quality damage category happens at the endpoint level with accurate representation of their magnitude. The brief description of both methods is given below.

5.2 Life cycle inventory

This section will describe the construction and assumptions employed when developing foreground and background systems of the LCA study. On a general level, the study consists of two scenarios reflecting the changes to the tower materials – either steel or wood. As part of the sensitivity analysis, one additional variation will be added to test for the influence of the LCIA method choice on the outcome by using IMPACT 2002+ instead of ReCiPe 2016 (H).

Apart from the differences in materials for the tower, the construction of the two systems is identical and employs the same approaches to foreground and background systems' construction and assumptions. The following section will describe the process of building the foreground database and the origin of the underlying assumptions. The background system relies on assumptions for the production processes of individual components available in the ecoinvent 3.9.1 (cut-off) database and the work of Eberle et al. (2023) and is applied to the European and Swedish case.

5.2.1 Foreground system

For the current study, the foreground data were modelled using information available in the REMPD database. The data on the onshore wind farm is based on the reference turbine construction by the International Energy Agency (IEA) in the Wind Task 37. The resulting information is expressed in terms of annual capacity in megawatts (MW), with a capacity of 3.4 MW used as the basis (Bortolotti et al., 2019).

The construction of a generic wind farm consists of the following components: blades, hub, nacelle, tower, foundation, array and export cables, substation, roads and some other additional materials.

According to Eberle et al. (2023), the materials used in the construction of all of the above listed components can be expressed in terms of several categories – concrete, road aggregate, i.e. crushed stones, rock and gravel, steel, composites and polymers (including carbon-fibre reinforced polymers which are primarily used in blade production), cast iron and other metals and alloys (see Table 8 for distribution of materials by farm component).

Table 8. Summary of the components included in the onshore wind power plant model. Adapted from Eberle et al. (2023).

Component	Description
Blades	Composed of composite materials, including polymer resin, glass or carbon fibers, and a wood or foam core. Three blades per turbine are standard.
Hub	Primarily made of cast iron, with a steel-based pitch system for blade orientation.
Nacelle	Built with fiberglass, steel, and cast iron. Configurations vary by turbine type, sometimes using rare-earth elements in direct-drive generators and additional materials like transformers.
Tower	For steel construction: Predominantly made of tubular steel; can also include concrete or a mix of steel and concrete. Additional materials are used for cabling and personnel access equipment. For wood construction: Predominantly same construction; 90% of steel substituted with wood.
Foundation	Primarily concrete with steel reinforcement; represents a major portion of the turbine's mass in land-based installations.
Array and export cables	Use aluminum or copper with polymer insulation. Submarine cables have added protective layers of lead or steel.
Substation	Requires steel and copper for transformers and wiring, with structural differences between land-based (concrete foundations) and offshore (steel support) substations.
Roads	Made from aggregate materials like crushed stone, gravel, or recycled concrete, used to provide site access within the wind plant.
Miscellaneous	Includes additional materials such as protective coatings (zinc for corrosion resistance), electronic controls, sensors, lighting, and safety equipment containing semiconductors and other critical materials.

The current design of wind power plants requires approximately 1,200 metric tonnes (t) of material per megawatt (MW), with the following breakdown of components by mass: 53% road aggregate, 34% concrete, 9% steel, 2% composites and polymers, 1% cast iron, 1% other metals and alloys, and less than 1% other materials.

To account for differences in the wind farm's configuration parameters (e.g. plant size, turbine number, rotor diameter, etc.), the REMPD model applies scaling relationships to each material type's fractional contribution (e.g., % concrete, % steel). For this case study, the standard configuration of the wind farm is assumed (see Table 9).

Table 9. Reference data used to build the foreground database for the LCA study. Adapted from Eberle et al. (2023).

Parameter	Value	Unit
Lifespan	30	year
Number of turbines	72	-
Turbine capacity	2.8	MW

Plant rated capacity	202	MW
Hub height	90	m
Rotor diameter	125	m

Appendix 9.1 provides a detailed bill of materials for the components included in the foreground database as described by Eberle et al. (2023), with links to the proxy materials in the ecoinvent 3.9.1 (cut-off) database.

5.3 Sensitivity analysis

The sensitivity analysis for this LCA study will aim to check for the methodological differences which might occur from using two alternative LCIA methods. As described before, the main analysis will be conducted using the ReCiPe 2016 (H) Endpoint method and for sensitivity analysis the LCIA method will be switched to IMPACT 2002+. The discussion below covers how two methods estimate and calculate the impacts on biodiversity through the ecosystem quality damage category to provide background for future methodological differences.

5.3.1 ReCiPe 2016

The ReCiPe 2016 is an LCIA method which provides a harmonized way to calculate characterization factors on midpoint and endpoint levels (Huijbregts et al., 2017). It allows to charaterise impacts on three areas of protection: human health, ecosystem quanlity, and resource scarcity. Each of these three endpoint categories is connected to the 17 midpoint categories through 8 damage pathways (ibid.).

For the scope of this thesis, the further focus will be directed to only one endpoint damage category – ecosystem quality – as it characterises impacts on biodiversity. Within the ReCiPe 2016 framework, the ecosystem quality is measured in terms of potentially dissapeared fraction (PDF) of species over space and time (PDF*m^2*year or PDF*m^3*year) for three ecosystems: marine, terrestrial, and freshwater (Huijbregts et al., 2017). The aggregation of the impacts on three ecosystems is performed through species density in each of the three ecosystems. The midpoint-endpoint connection for ecosystem quality is done through three damage pathways on terrestrial, freshwatrer and marine ecosystems (ibid.). Figure 4 depicts the relationships between endpoint and midpoints for ecosystem quality as well as their units of measure within the ReCiPe 2016 framework.

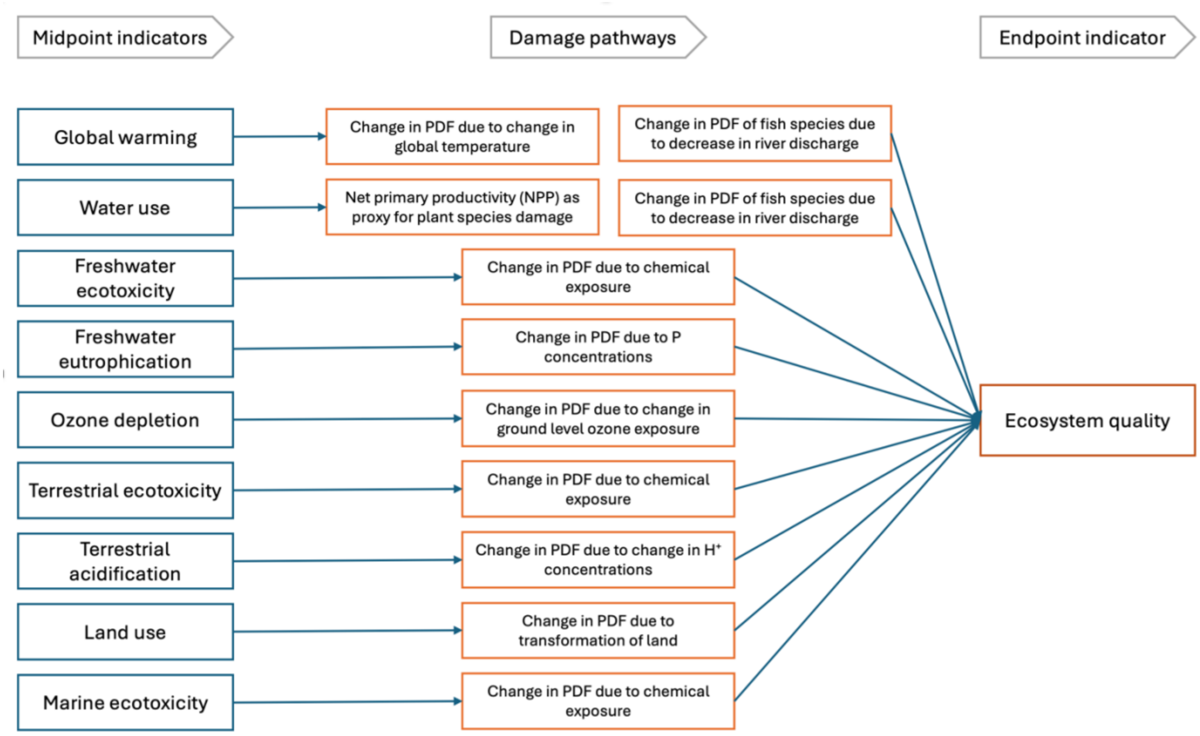

Figure 4. Midpoint-endpoint relationships for ecosystem quality under ReCiPe 2016 framework. Adapted from Huijbregts et al. (2017).

The authors of the ReCiPe framework have outlined a few aspects where their model can be improved. Firstly, the half of the midpoint indicators connected to ecosystem quality (land use and ecotoxicity) lack regionalised or spatial characterisation factors which would greatly improve the quantification of impacts (Huijbregts et al., 2017). Secondly, the ecosystem quality could greatly benefit from quantification through *global risk of extinction* in addition to the existing quantification through PDF. Lastly, the framework would greatly benefit from introduciton of additional impact categories at the midpoint level to more accurately model the impacts to the ecosystems (ibid.). The additions could include marine eutrophication, invasive species, plastic debris, and noise pollution.

5.3.2 IMPACT 2002+

IMPACT 2002+ is a successor to Eco-Indicator 99 and CML 2002 methods (with successive improvement through IMPACT World + framework) and is an LCIA method which links the LCI results to 14 midpoint and four endpoint categories (Jolliet et al., 2003; Jolliet et al., 2016, p. 138). The four endpoint (damage) categories are (1) human health, (2) ecosystem quality, (3) climate change, and (4) resources (ibid.). Similarly to the discussion on ReCiPe 2016 framework, the following discussion will primarily focus on the methodology and midpoint-to-endpoint relationships pertaining to ecosystem quality.

Table 10. Midpoint categories contributing to ecosystem quality damage category and their characterisation units. Adapted from Jolliet et al. (2003).

Midpoint impact category	Midpoint unit	Endpoint unit
Aquatic ecotoxicity	kg_{eq} triethylene glycol into water	
Terrestrial ecotoxicity	kg_{eq} triethylene glycol into water	
Terrestrial acidification/nutrification	kg_{eq} SO_2 into air	PDF * m^2 * yr
Land occupation	m^2_{eq} organic arable land·year	

Within the IMPACT 2002+ framework, 6 midpoint impact categories (Table 10) contribute towards the endpoint category of ecosystem quality (Jolliet et al., 2003). The midpoint damage categories of *ecotoxicity*, both aquatic and terrestrial, *terrestrial acidification/nutrification*, and *land occupation* are characterised in terms of PDF over area and time at the endpoint damage category level (ibid.). The *aquatic acidification* and *aquatic eutrophication* are characterised in terms of potentially affected fraction (PAF) of species over area and time per kg of emitted substance on the midpoint level and is then converted to the PDF representation using an extrapolation factor (ibid.). For the remaining categories of *aquatic acidification* and *aquatic eutrophication* the characterisation factors are being developed (by the time of the original publication) while the contribution of *photochemical oxidation* and *ozone depletion* to ecosystem quality is unclear due to lack of scientific data (ibid.).

5.4 Results of LCA study

This section will present the results of the case study to compare the two LCIA methods – ReCiPe 2016 (H) Endpoint and IMPACT 2002+ Endpoint. The section will first look at the results from the two methods separately to provide a general overview and build an understanding of how the subassemblies were ranked depending on the tower material selection. Then, the section will compare and contrast the results from the two methods. The overall results for the two damage categories are presented in Table 11 below.

Table 11. Total scores for the two LCIA methods and two tower configurations for the Ecosystem quality damage category.

Method	Score	Unit
ReCiPe 2016 (H) Steel	0,004762011	species*year
ReCiPe 2016 (H) Wood	0,004944691	species*year
IMPACT 2002+ Steel	38,156722077	PDF*year*m2
IMPACT 2002+ Wood	36,969780825	PDF*year*m2

The comparison of the two methods has been carried out not in “raw” numbers but rather in contribution percentages because both methods have different ways to estimate the total impacts for the ecosystem quality damage category and can not be compared one-to-one on the midpoint level for a deeper analysis.

5.4.1 Damage to Ecosystem quality as assessed by ReCiPe 2016

The assessment using ReCiPe 2016 (H) for two alternative tower configurations has shown that the wood configuration has a higher total impact on the ecosystem quality than the steel tower, as seen by the results in Table 11. On the other hand, the distribution of the contribution of the individual components towards the final score has remained the same, with tower, nacelle and blades having the largest impacts (see Table 12).

Table 12. Contribution of wind farm components to the final score for the Ecosystem quality damage category for ReCiPe 2016 (H) LCIA method.

Component	Contribution (Steel)	Contribution (Wood)
Tower	28,03%	30,69%
Nacelle	20,46%	19,70%
Blade	19,68%	18,95%
Foundation	14,17%	13,64%
Array and export cables	11,77%	11,34%
Hub	3,98%	3,83%
Substation	1,55%	1,49%
Roads	0,36%	0,35%

A similar trend can be observed when looking at the midpoint categories associated with ecosystem quality (see Figure 5). The magnitude of impacts is consistent across the two configurations for all midpoint indicators except for land use – here, there is a large "spike" in the impact for the wood tower configuration, making it the third largest impact for this construction. Otherwise, the impacts from steel configuration scored higher for the remainder of the impact categories with *Climate change (terrestrial ecosystems) and Acidification (terrestrial)* constituting the majority of impacts, while the categories of *Ecotoxicity (marine), Climate change (freshwater ecosystems), Eutrophication (marine)* and *Water use (aquatic ecosystems)* having almost negligible impacts (below 1% from the total score).

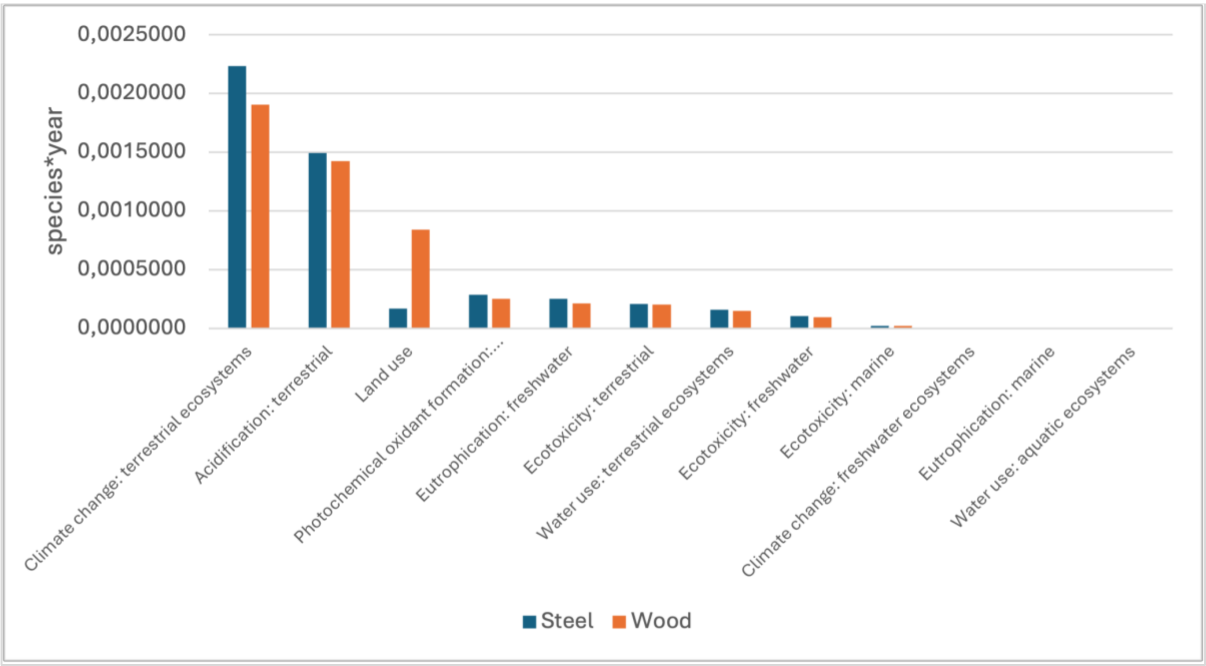

Figure 5. Impact assessment results per midpoint damage category for steel and wood towers using ReCiPe 2016 (H) Endpoint LCIA method.

On a general level, the overall contribution of the components of the wind farm to the impact category is consistent across the two compositions except for the tower (see Figure 6). This is expected, given that only this component has a difference in the bill of materials. For all but one impact category, the steel tower has a larger contribution than the wood one however, the overall magnitude of the impact is approximately the same. The story is different for the land use category, where the wood tower has a significantly larger contribution and impact than the steel one, which ultimately contributes to a larger total score for the whole damage category.

Figure 6. Contribution of the components of wind farm to the midpoint indicators for (a) steel and (b) wood tower configurations using ReCiPe 2016 (H) LCIA method.

5.4.2 Damage to Ecosystem quality as assessed by IMPACT 2002+

The results from IMPACT 2002+ show that the wood tower construction has a slightly better overall performance than the steel tower option (see Table 11). Conversely, the overall magnitude of contribution of wind farm components to the total score has stayed the same, regardless of the construction, with nacelle, tower, and cables accounting for the majority of impacts (see Table 13).

Table 13. Contribution of wind farm components to the total score for the Ecosystem quality damage category for IMPACT 2002+ LCIA method.

Component	Steel	Wood

Nacelle	29,33%	30,27%
Tower	26,46%	24,10%
Array and export cables	16,54%	17,07%
Foundation	12,14%	12,53%
Blade	8,41%	8,68%
Hub	4,00%	4,13%
Substation	2,78%	2,87%
Roads	0,34%	0,35%

When looking at the distribution of impacts between the midpoint categories contributing towards the ecosystem quality, *terrestrial ecotoxicity* has the largest share for both the steel and wood scenarios (see Figure 7). On a general level, the wood option scores lower than the steel one for all the midpoint categories except *land occupation*. This is expected due to the higher pressure on land from using wood compared to steel.

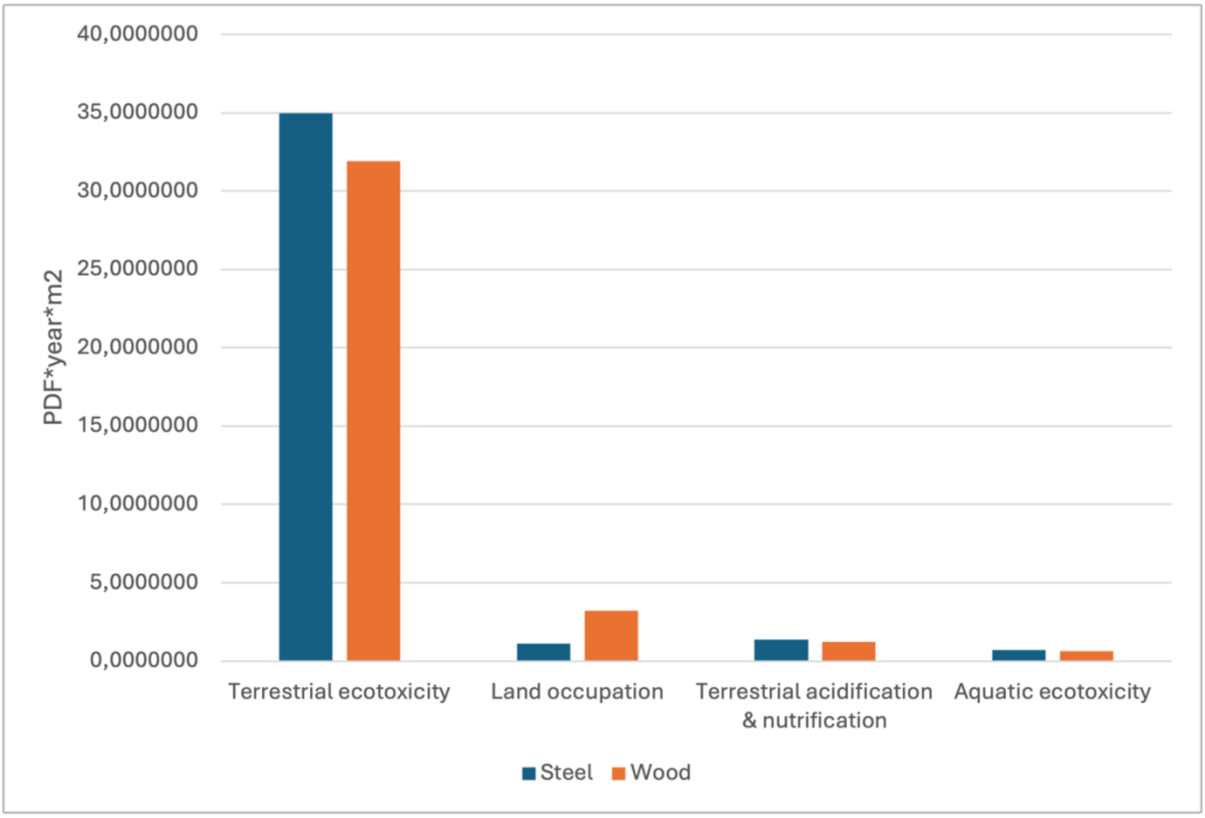

Figure 7. Impact assessment results for per midpoint damage category for steel and wood towers using the IMPACT 2002+ LCIA method.

When looking at the contributions of wind farm components per impact category, the general trend for magnitude and distribution of impacts between the components is consistent across the two scenarios (see Figure 8). The changes can be seen with diminished contribution from the wood tower for all categories except *land occupation,* where the contribution has increased by 20%.

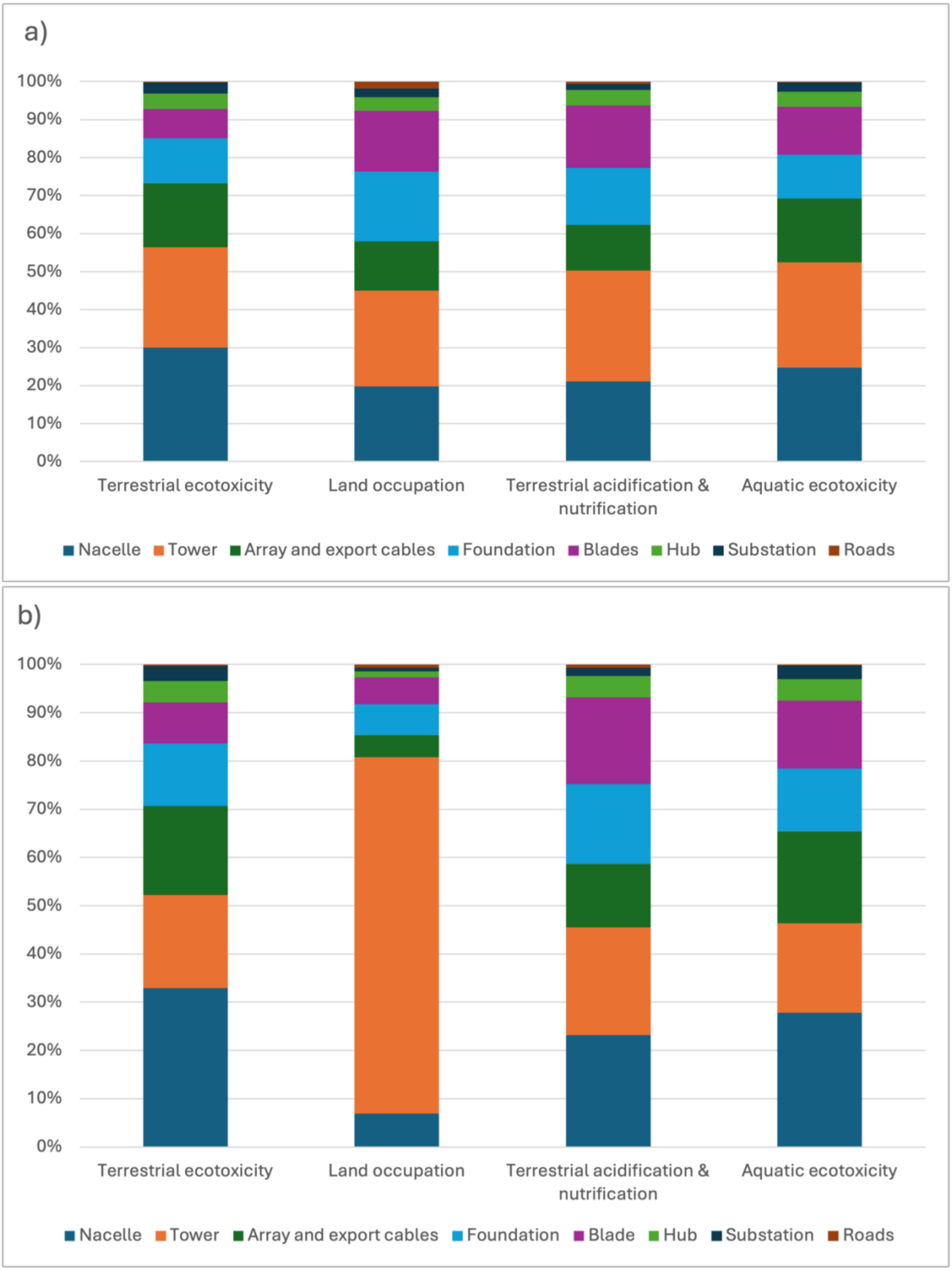

Figure 8. Contribution of wind farm components to the midpoint impact indicators for (a) steel and (b) wood tower configurations using IMPACT 2002+ LCIA method.

5.4.3 Comparison of two LCIA methods

As can be seen from the results, the results produced by two LCIA methods tell a different story. The magnitude of the total scores for the ecosystem quality damage category provides conflicting results –

the ReCiPe shows that the wooden alternative has a larger impact than the steel one, while the IMPACT 2002+ shows that the steel one is worse from the environmental point of view. Despite showing similar trends in the distribution of impacts between the components of the wind farm within the same methods, depending on the composition of the tower, across the methods, the contribution of components is different (see Figure 9).

The three lowest contributors for the two methods are the same – hub, substation and roads. The other components are "ranked" differently for the two methods. ReCiPe ranks tower's contribution higher than IMPACT, while for IMPACT nacelle has the highest contribution. A similar pattern is seen for the rest of the components, where ReCiPe ranks the contributions of blades, foundation and array cables in this order, while IMPACT ranks them as array cables, foundation and blades.

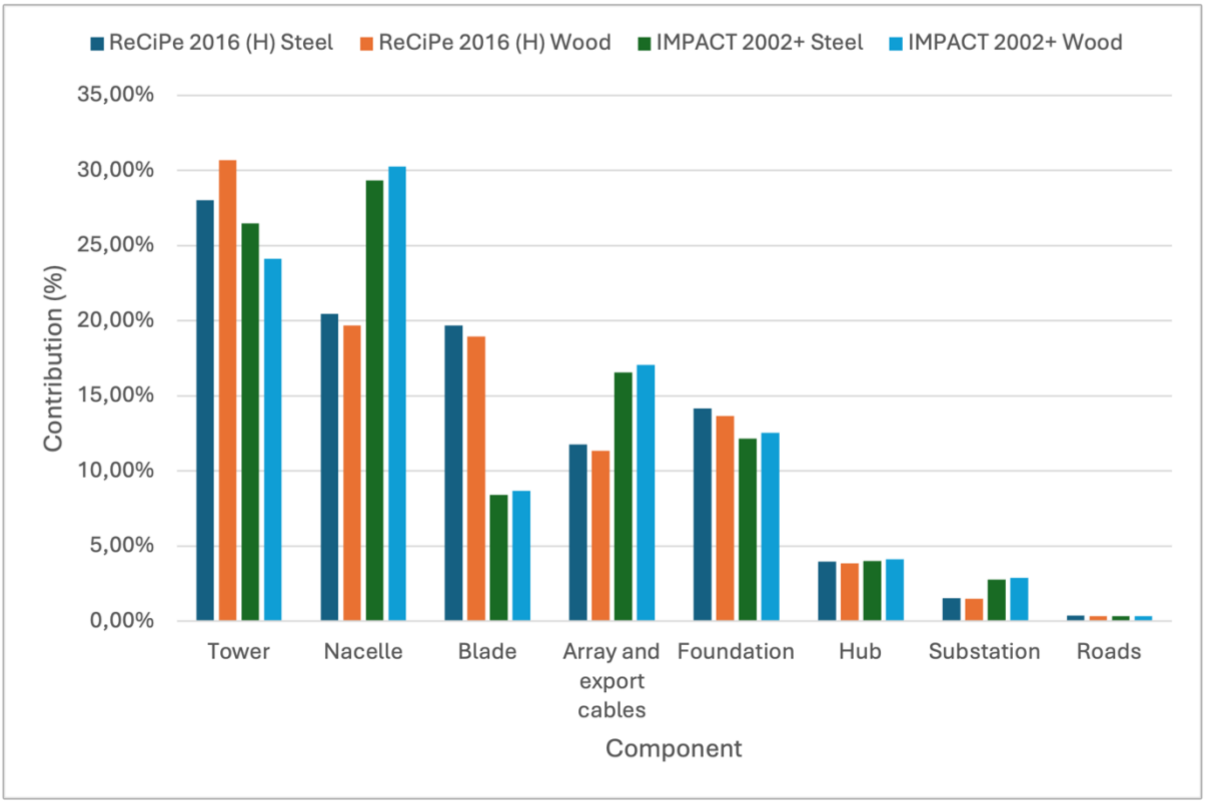

Figure 9. Comparison of the components' contributions to the total score for the Ecosystem quality damage category between the two LCIA methods and tower constructions.

From Figure 5 and Figure 7 it is obvious that the two methods place different emphasis on different midpoint indicators when calculating the final scores for the endpoint damage categories. This way, ReCiPe 2016 assigns the majority of the impacts to *climate change (terrestrial) and acidification (terrestrial)* (these two categories account for more than 50% of the total impact cumulatively in both scenarios). For IMPACT 2002 +, the majority of the impacts are attributed to *terrestrial ecotoxicity* (around 90% for both scenarios). This highlights the methodological differences which have to be accounted for when selecting the method for the LCA study.

The full one-to-one comparison of the two methods is not possible due to highlighted methodological differences. On the other hand, despite the major disagreement of the two methods in regard to the environmentally "best" choice between the steel or wood tower options, there is a general agreement between the methods on the midpoint level. This way, both LCIA methods have ranked the steel tower as the worst option in regard to terrestrial ecotoxicity, terrestrial acidification and aquatic ecotoxicity. There was also an agreement in the land use category where both methods assigned a larger impact to the wood construction. In this light, despite the major methodological differences between the two methods on a more granular, midpoint level, the two methods generally agree, and score impacts similarly.

6 Discussion

6.1 Influence of methodological choices and uncertainty in LCA

The results of the case study have verified the fact that the choice of LCIA methodology has a substantial influence on the outcomes of the biodiversity assessment from the product system. When conducting the assessment of identical configurations of wind farms with the exception of tower, where the materials differed between two scenarios, ReCiPe showed a higher ecosystem quality damage score for the scenario with wooden tower construction as compared to the steel one. The other method, IMPACT 2002+, showed the opposite result, where larger damage was attributed to the steel tower. Given both methods conducted assessment on the same LCI data, such differences raise concerns in regard to robustness of conclusions about biodiversity footprints derived from two LCA studies.

The divergence most likely stems from differences in characterization models and midpoint-to-endpoint pathways of the two LCIA methods. In case of ReCiPe, the method emphasizes land use change effects more heavily, while IMPACT put emphasis on ecotoxicology pathways assigning more weight to fossil-based emissions and resource extraction. Since wind tower uses more wood which involves greater land use impacts from forest clearing, ReCiPe interprets them as more damaging. Conversely, the steel production is a more toxic process which involves the use of chemicals which is evident from the results produced by IMPACT. Due to analytical setup of the LCA study, where only raw material acquisition (background system) and component manufacturing (foreground system) were included in the system boundaries, the variation in final results can only be explained due to the differences in assumptions and damage pathways of the two LCIA methods.

What is concerning is not the credibility of standalone results – both methods are well established in the community and are widely applied for impact assessments – but the revealed limitation of LCIA studies to deliver consistent and comparable guidance for biodiversity-related decisions. This ambiguity reflects negatively on the efforts of practitioners to integrate biodiversity footprinting into LCA framework as it involves large uncertainty in result interpretation and assumes adoption of certain level of risk.

This issue is highlighted in various literature (Crenna et al., 2020; Damiani et al., 2023; Bromwich et al., 2025) which explores the biodiversity assessment within LCA framework. Some practitioners (Asselin et al., 2020; Crenna et al., 2020) also highlight that even the units used to quantify the biodiversity impacts in LCA, like PDF of species, are ambiguous and cause confusion among experienced practitioners which struggle to interpret the results.

As such, this result has important implications for the aim of the thesis which attempts to conceptualise the link between biodiversity and LCA. It highlights the fact that practitioners need to interpret the results of the LCA and their scores with caution and consider applying multiple methods to comprehensively capture all relevant impacts. The potential mitigation strategy could include use of hybrid methods which would cover multiple perspectives and include extensive account for uncertainty and assumptions like it is currently done in climate modeling. Ultimately, the challenge of integrating biodiversity perspective into LCA is not only technical but also requires strong contextual judgement and interpretation.

6.2 Biodiversity loss in footprinting methods

The results also show that while there is a subset of methods which provide sufficient coverage of biodiversity loss drivers such as climate change, pollution, habitat fragmentation, direct exploitation and invasive species, there is also a large subset of tools which do not do that. This conclusion reflects the inherent trade-offs of the methodological design. While some methods cover many indicators, they might not do that comprehensively and aggregate the impacts over the whole driver not being able to capture the specifics. Other methods, while they cover less drivers overall, are able to provide a much deeper quantification of impacts for drivers which are covered, like in the example of spatial tools, which are predominantly targeted towards land-use quantification but are able to provide deeper insights because of their reliance on spatial data.

The final methodology selection in this thesis involved screening their applicability for LCA, so many methods which do not follow the life cycle thinking logic have been eliminated despite offering good biodiversity coverage, for example ENCORE, SBF, which offer collision mortality quantification or Life and Xylo Systems, which provide quantification impacts on genetic diversity. As such, the reliance solely on the criteria of "compatible with LCA" inhibits the ability of practitioners to capture relevant biodiversity pressures. To mitigate this issue, it is important to consider the aim of the study and use methods appropriate for the aim regardless of their compatibility.

However, as it currently stands, even with sufficient coverage of general biodiversity loss drivers, almost all of the evaluated methods (both selected for LCA implementation and not) provide insufficient coverage of drivers which are wind energy systems specific – like vibration, collusion mortality, microclimatic alternations, and behavioral changes. This issue is expected, since the vast majority of the currently available methods is designed for broad application or agricultural practices, which results in loss of sector-specific aspects.

While this result is expected, this further highlights the major limitations of the existing biodiversity footprinting methods within and outside of LCA which are unable to provide comprehensive assessment of systems with specific requirements. This also suggests that the results obtained from using the methods will underestimate the overall impacts of wind power production on biodiversity, since a big proportion of relevant drivers are not included on methodological level.

The assessed literature agrees with the results on the issues surrounding sector-specific quantification of biodiversity impacts. Bergman et al. (2025) outlined similar problem for the assessment of biodiversity impacts for marine food systems where the relevant drivers are excluded from most of the LCIA methods. However, there has been some improvement in the field: Damiani et al. (2023) mentioned that there are currently a few LCIA methods under development which aim to specifically address issues related to bird diversity due to wind energy by May et al. (2021).

Despite that, the majority of available methods still systematically suffer from generalization of impacts over multiple sectors. To address this issue, the practitioners should focus on the development of sector specific models for impacts and characterization factors. A possible approach for wind farms case would be to assign specific weights for land use impacts based on their type (forest, grassland, cropland, etc.), species sensitivity and relative importance for wind power production. Thus, practitioners can incorporate hybrid approaches to link spatial data with on ground measurements of species abundance to feed into the LCI.

6.3 Integrating biodiversity perspective in LCA

The key result of this thesis is the proposed approach to integrate biodiversity impacts into the LCA of wind power systems. The proposed approach involves several steps: (1) identification of relevant pressures, (2) linking LCI flows to the pressures, and (3) using appropriate methodology to quantify the pressures – be it LCIA alone, or a combination with one of the methods from beyond-LCA category to improve the coverage of analysis. This approach allows for a certain level of flexibility for practitioners. Instead of prescribing the use of single methods to try and capture all relevant impacts, the proposed approach acknowledges the complexity of the task and directs the practitioners towards informed choices which will minimise the potential risks and uncertainty of biodiversity footprinting within LCA.

The literature on the topic (Winter et al., 2017; Sanyé-Mengal et al., 2023; Bergman et al., 2024) supports the proposed stepwise approach for integration pathways of biodiversity footprinting into LCA.

Despite the advantages for practical use, this approach has some limitations which might affect the final application. The first one is related to data availability and quality. Despite the availability of methods to quantify the desired impacts, the data quality and availability remains the issue. In connection to this, practitioners need to realise that the same issues that were described previously still exist here – methodological differences. Different methods have varying scope of application, both spatial and temporal, so practitioners need to select methods which allow quantification on the same scale spatial and temporal scales. No matter how good the existing footprinting methods are, they still have sector-specific gaps which can not be covered due to limitations of technology evolution. Lastly, unlike LCA methods, which are highly standardised, the alternative methods might not follow any standard as they use unique methodological approaches. These issues could be partially solved by future technological and standard development which will homogenise the requirements on all methods used for footprinting and decision-making purposes making methods more comparable and uniform. As this recommendation concerns future developments, the actions which practitioners can currently apply will be to use primary data as much as possible to remove discrepancy in data quality and depth of assumptions and include extensive accounting for methods and uncertainty during assessment to remove doubts from result validity.

7 Conclusion

The aim of the current study was to explore the influence of methodological differences on biodiversity quantification in LCA and understand how biodiversity impacts can be included in the LCA, using onshore wind power production as a case. To address the two aims the following objectives were defined:

1. Explore the influence of methodological differences on evaluation of biodiversity within the LCA framework using a case study
2. Identify and examine relevant biodiversity footprinting methods
3. Evaluate their suitability for integration into the LCA framework using a set of criteria
4. Propose an approach to integrate the biodiversity impact assessment methodologies into the LCA framework.

The results and discussion of this study provide several key insights which fulfill the objectives.

Firstly, the case study has revealed that the results of the LCA have a clear dependence on the choice of a LCIA method. Given the same system configurations, the two methods – ReCiPe 2016 (H) and IMPACT 2002 – have provided opposite results when examining which tower configuration has the largest impact on biodiversity. The results were also different for the hotspot analysis as the two methods emphasized the contribution of the midpoint indicators differently. On the midpoint level, for the categories which could be directly compared, the contributions of components to the total impact for the given category were consistent between the methods. Due to these differences, when conducting the LCA studies practitioners need to be aware of the potential uncertainty and select the impact methodology which will aid the goal of the study. The practitioners need to acknowledge the risks of the methodological uncertainties and ensure the results obtained from the studies are robust to them and could be safely used to inform the adaptation strategies.

Secondly, the literature review has shown that the current approaches for biodiversity footprinting can be separated into two large categories – within-LCA and beyond-LCA. The first group includes LCIA methods which assess biodiversity on the ecosystem quality damage category level, while the second group consists of multiple sub-groups of methods which either build on top of the life cycle thinking framework or use alternative approaches to biodiversity quantification.

The subsequent assessment of the beyond-LCA method category has been conducted to evaluate the potential of methods to be incorporated into the LCA framework. The evaluation showed that the methodological limitations of the LCA framework support the integration of methods which already utilise the life cycle thinking in their framework or those which support the development of characterization factors for inventory by the user. Other methods, despite their additionality in terms of covered biodiversity aspects could not be recommended for integration due to methodological bottleneck.

In connection to this, the study has revealed an important gap – the biodiversity footprinting methods which are currently available largely focus on general biodiversity loss without offering specifics for the particular sector, like it is for wind power production. Even if the general coverage of biodiversity loss drivers by assessed methods is sufficient, sector specific drivers are not included.

Lastly, the proposed stepwise approach on how to include biodiversity loss impacts in LCA study offers practical guidance on how practitioners can start integrating more perspectives into their studies. The proposal emphasizes use of hybrid methods where the selection is grounded in the specific drivers for biodiversity loss and not only compatibility with LCA framework.

On a general level, the study contributes to the growing scientific research field on inclusion of biodiversity into LCA. It highlights important methodological blind spots and proposes feasible pathways to bridge the gaps between methods. The findings of the paper can be of value not only to the narrow scientific community but also for policymakers and any LCA practitioner who want to make informed decisions regarding ecological trade-offs and biodiversity loss mitigation strategies.

The paper highlights the gaps in sector-specific biodiversity footprinting methodology and importance of its inclusion in the assessment frameworks. This could give rise to new methodological developments not only for wind energy sector but for other sectors as well which will improve reliability and comprehensiveness of available methods.

As such, the future research could focus on several fronts:

- Development of modular, open-source biodiversity plugins for LCA software which would lower the entry barrier to the field
- Integrate dynamic spatial data and information on species vulnerability into LCI datasets to support regionally relevant assessment
- Development of hybrid methods which combine the characteristics of all methodological groups could help accelerate the closure of the identified gaps in biodiversity footprinting.

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9 Appendix

9.1 Life cycle inventory table

Table 14. Life cycle inventory table of the main material inputs for the generic wind farm in Sweden.

Component	Reference material	Quantity	Unit	Location
Wind farm	array and export cables construction	1	unit	SE
	blade	1	unit	SE
	foundation	1	unit	SE
	hub	1	unit	SE
	nacelle	1	unit	SE
	roads	1	unit	SE
	substation	1	unit	SE
	tower	1	unit	SE
Array and export cables	market for aluminium, primary, ingot	2405,696	kilogram	RoW
	market for chromium	2,927	kilogram	GLO
	market for copper, cathode	1178,791	kilogram	GLO
	market for manganese	0,637	kilogram	GLO
	market for nickel, class 1	-0,049	kilogram	GLO
	market for polyethylene, low density, granulate	4426,481	kilogram	GLO
	market for titanium	0,013	kilogram	GLO
	market for zinc	0,089	kilogram	GLO
Blades	carbon fibre reinforced plastic, injection moulded	801,103	kilogram	GLO
	market for chromium	6,809	kilogram	GLO
	market for cobalt	0,010	kilogram	GLO
	market for copper, cathode	22,737	kilogram	GLO
	market for epoxy resin, liquid	4735,934	kilogram	RER
	market for glass fibre reinforced plastic, polyamide, injection moulded	13500,946	kilogram	GLO
	market for manganese	7,069	kilogram	GLO
	market for neodymium oxide	2,010	kilogram	GLO
	market for nickel, class 1	5,985	kilogram	GLO
	market for polyethylene terephthalate, granulate, amorphous	432,596	kilogram	GLO
	market for polyethylene, low density, granulate	266,249	kilogram	GLO
	market for polyvinylchloride, bulk polymerised	334,578	kilogram	GLO
	market for praseodymium oxide	0,002	kilogram	GLO

	market for sawnwood, hardwood, raw, dried (u=20%)	1,617	cubic meter	RER
	market for steel, low-alloyed	374,634	kilogram	GLO
	market for titanium	0,225	kilogram	GLO
	market for zinc	0,100	kilogram	GLO
	niobium mine operation and beneficiation, from pyrochlore ore	0,001	kilogram	RoW
	polyester resin production, unsaturated	1578,645	kilogram	RER
Foundation	market for aluminium, primary, ingot	1,423	kilogram	RoW
	market for boron carbide	13,593	kilogram	GLO
	market for chromium	189,931	kilogram	GLO
	market for cobalt	0,462	kilogram	GLO
	market for copper, cathode	7,643	kilogram	GLO
	market for epoxy resin, liquid	12,494	kilogram	RER
	market for gallium, semiconductor-grade	0,009	kilogram	GLO
	market for lithium	0,003	kilogram	GLO
	market for manganese	412,357	kilogram	GLO
	market for nickel, class 1	431,122	kilogram	GLO
	market for praseodymium oxide	0,112	kilogram	GLO
	market for reinforcing steel	9336,392	kilogram	GLO
	market for steel, chromium steel 18/8	878,719	kilogram	GLO
	market for steel, low-alloyed	12494,289	kilogram	GLO
	market for tin	0,032	kilogram	GLO
	market for titanium	10,618	kilogram	GLO
	market for zinc	3,854	kilogram	GLO
	market group for concrete, normal strength	180,778	cubic meter	GLO
	niobium mine operation and beneficiation, from pyrochlore ore	0,064	kilogram	RoW
Hub	market for cast iron	4320,735	kilogram	GLO
	market for chromium	42,022	kilogram	GLO
	market for cobalt	0,154	kilogram	GLO
	market for copper, cathode	13,436	kilogram	GLO
	market for dysprosium oxide	0,000	kilogram	GLO
	market for gallium, semiconductor-grade	0,003	kilogram	GLO
	market for glass fibre reinforced plastic, polyamide, injection moulded	101,432	kilogram	GLO
	market for manganese	152,806	kilogram	GLO
	market for nickel, class 1	140,951	kilogram	GLO
	market for polyethylene, low density, granulate	48,740	kilogram	GLO
	market for praseodymium oxide	0,041	kilogram	GLO

	market for reinforcing steel	13,832	kilogram	GLO
	market for silicone product	5,480	kilogram	RER
	market for steel, chromium steel 18/8	189,691	kilogram	GLO
	market for steel, low-alloyed	8088,205	kilogram	GLO
	market for tin	0,012	kilogram	GLO
	market for titanium	3,926	kilogram	GLO
	market for zinc	1,357	kilogram	GLO
	niobium mine operation and beneficiation, from pyrochlore ore	0,024	kilogram	RoW
Nacelle	iron pellet production	562,655	kilogram	RoW
	market for aluminium, primary, ingot	274,597	kilogram	RoW
	market for cast iron	5949,601	kilogram	GLO
	market for casting, steel, lost-wax	168,258	kilogram	GLO
	market for chromium	901,863	kilogram	GLO
	market for cobalt	0,872	kilogram	GLO
	market for copper, cathode	880,326	kilogram	GLO
	market for epoxy resin, liquid	260,059	kilogram	RER
	market for gallium, semiconductor-grade	0,025	kilogram	GLO
	market for glass fibre reinforced plastic, polyamide, injection moulded	559,962	kilogram	GLO
	market for manganese	214,293	kilogram	GLO
	market for nickel, class 1	748,411	kilogram	GLO
	market for polyethylene terephthalate, granulate, amorphous	7,856	kilogram	GLO
	market for polyethylene, low density, granulate	70,803	kilogram	GLO
	market for praseodymium oxide	0,055	kilogram	GLO
	market for reinforcing steel	6,434	kilogram	GLO
	market for silicone product	30,152	kilogram	RER
	market for steel, chromium steel 18/8	4280,482	kilogram	GLO
	market for steel, low-alloyed	11679,794	kilogram	GLO
	market for tin	0,034	kilogram	GLO
	market for titanium	5,600	kilogram	GLO
	market for zinc	8,292	kilogram	GLO
	niobium mine operation and beneficiation, from pyrochlore ore	0,033	kilogram	RoW
	polyester resin production, unsaturated	0,649	kilogram	RER
Roads	market for gravel, crushed	465807,321	kilogram	CH
Substation	market for aluminium, primary, ingot	5,239	kilogram	RoW
	market for boron carbide	0,022	kilogram	GLO
	market for cast iron	3,689	kilogram	GLO
	market for chromium	9,222	kilogram	GLO

	market for cobalt	0,103	kilogram	GLO
	market for copper, cathode	236,560	kilogram	GLO
	market for epoxy resin, liquid	0,156	kilogram	RER
	market for gallium, semiconductor-grade	0,001	kilogram	GLO
	market for glass fibre reinforced plastic, polyamide, injection moulded	16,904	kilogram	GLO
	market for lead	0,068	kilogram	GLO
	market for manganese	17,856	kilogram	GLO
	market for nickel, class 1	11,681	kilogram	GLO
	market for polyethylene terephthalate, granulate, amorphous	36,353	kilogram	GLO
	market for polyethylene, low density, granulate	1,959	kilogram	GLO
	market for praseodymium oxide	0,003	kilogram	GLO
	market for silicone product	64,687	kilogram	RER
	market for steel, low-alloyed	1280,365	kilogram	GLO
	market for tin	0,009	kilogram	GLO
	market for titanium	0,361	kilogram	GLO
	market for zinc	0,218	kilogram	GLO
	market group for concrete, normal strength	0,298	cubic meter	GLO
	niobium mine operation and beneficiation, from pyrochlore ore	0,002	kilogram	RoW
Tower (steel)	market for aluminium, primary, ingot	462,242	kilogram	RoW
	market for chromium	41,770	kilogram	GLO
	market for steel, chromium steel 18/8	56,982	kilogram	GLO
	market for cobalt	1,311	kilogram	GLO
	market for copper, cathode	46,476	kilogram	GLO
	market for epoxy resin, liquid	62,024	kilogram	RER
	market for gallium, semiconductor-grade	0,016	kilogram	GLO
	market for steel, low-alloyed	80285,548	kilogram	GLO
	market for graphite	4,177	kilogram	GLO
	iron pellet production	20,843	kilogram	RoW
	market for lithium	0,821	kilogram	GLO
	market for manganese	1512,792	kilogram	GLO
	market for nickel, class 1	1142,998	kilogram	GLO
	niobium mine operation and beneficiation, from pyrochlore ore	0,234	kilogram	RoW
	market for polyethylene terephthalate, granulate, amorphous	118,645	kilogram	GLO
	market for praseodymium oxide	0,410	kilogram	GLO
	market for tin	0,105	kilogram	GLO

	market for titanium	38,744	kilogram	GLO
	market for zinc	21,011	kilogram	GLO
Tower (wood)	iron pellet production	20,843	kilogram	RoW
	market for aluminium, primary, ingot	462,242	kilogram	RoW
	market for chromium	41,770	kilogram	GLO
	market for cobalt	1,311	kilogram	GLO
	market for copper, cathode	46,476	kilogram	GLO
	market for epoxy resin, liquid	62,024	kilogram	RER
	market for gallium, semiconductor-grade	0,016	kilogram	GLO
	market for glued laminated timber, average glue mix	81,820	cubic meter	Europe without Switzerland
	market for graphite	4,177	kilogram	GLO
	market for lithium	0,821	kilogram	GLO
	market for manganese	1512,792	kilogram	GLO
	market for nickel, class 1	1142,998	kilogram	GLO
	market for polyethylene terephthalate, granulate, amorphous	0,983	kilogram	GLO
	market for polyethylene, low density, granulate	117,662	kilogram	GLO
	market for praseodymium oxide	0,410	kilogram	GLO
	market for steel, chromium steel 18/8	11,396	kilogram	GLO
	market for steel, low-alloyed	16057,110	kilogram	GLO
	market for tin	0,105	kilogram	GLO
	market for titanium	38,744	kilogram	GLO
	market for zinc	21,011	kilogram	GLO
	niobium mine operation and beneficiation, from pyrochlore ore	0,234	kilogram	RoW

9.2 Biodiversity assessment methods screening

Table 15. Overview of the final selection of methods for analysis.

Method (name)	Reference	Type	Brief description	Documentation
Biodiversity Footprint Method	Damiani et al. (2023), De Ryck et al. (2024)	Calculation method, tool (web application)	Plansup & WEnR. Scenario tool for companies, products or sectors; pressure-based calculator derived from GLOBIO. Users enter land-use areas, GHG quantities and (in full method) N&P to water; naturalness factors convert pressures to impact.	http://www.plansup.nl https://biodiversity-footprint-calculator
Biodiversity Impact and Global Extinction Risk Footprinting tool (BIGER Footprint)	De Ryck et al. (2024)	Database, tool	GIST Impact. Corporate/portfolio engine that traces operational & value-chain emissions/pressures; applies LC-IMPACT CFs to terrestrial, freshwater & marine realms.	https://gistimpact.com/biodiversity-solutions/
Biodiversity Impact Assessment Tool (BIAT)	De Ryck et al. (2024)	Calculation method, tool	ISS ESG + Quantis. Investor-oriented LCIA platform; bottom-up company model + EXIOBASE trade flows; reports both PDF and MSA plus ES dependencies (ENCORE/CICES).	https://www.issgovernance.com/esg/biodiversity-impact-assessment-tool/
Biodiversity Net Gain Calculator (BNGC)	De Ryck et al. (2024)	Calculation method, tool (excel)	Arcadis. Site-level "no-net-loss / net-gain" accounting; habitat area×condition×significance	https://www.arcadis.com/
Biodiversity Risk Filter (BRF)	De Ryck et al. (2024)	Calculation method, tool (web application), database	WWF. Quick global risk heat-map for assets / supply chains	https://riskfilter.org
BioScope	Damiani et al. (2023), De Ryck et al. (2024), SBTN STEP 1 TOOLBOX	Tool (web application)	Dutch Govt, Arcadis, PRé. Portfolio hot-spot screener (EXIOBASE + ReCiPe)	https://bioscope.info

	(2024), Barth et al. (2024)			
Corporate Biodiversity Footprint (CBF)	De Ryck et al. (2024), SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024), Damiani et al. (2023)	Calculation method, tool	Iceberg Data Lab. Bottom-up footprint for investors	https://icebergdatalab.com
ECOPLAN-Scenario Evaluator	De Ryck et al. (2024)	Calculation method, tool	Belgian academic consortium. ES trade-off model for plans	https://www.uantwerpen.be/en/projects/ecoplan/
Ecosystem Intelligence Platform	De Ryck et al. (2024)	Tool	EcoMetrix. AI-assisted site ES quantification	https://ecosystemintelligence.com
ENCORE	SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024), De Ryck et al. (2024)	Database, tool (web application)	UNEP-FI/Global Canopy. Dependency/impact materiality by sector & geography	https://encore.naturalcapital.finance
Environmental Profit & Loss (EP&L)	Damiani et al. (2023), SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024)	Calculation method	Kering. Corporate accounting of nature costs along value-chain; hybrid LCA + economic valuation	https://kering-group.opendatasoft.com/pages/environmental-profit-loss/
Global Biodiversity Score (GBS)	Damiani et al. (2023), De Ryck et al. (2024)	Calculation method, tool (software)	CDC Biodiversité. Company/portfolio footprint. Pressures are quantified through MSA via GLOBIO	https://www.cdc-biodiversite.fr/wp-content/uploads/2024/01/DOSSIER-MEB-49-GBS-MD-WEB.pdf
Global Impact Database (GID)	De Ryck et al. (2024)	Database, tool	Impact Institute. EE-IO model with biodiversity extensions	https://impactinstitute.com

GLOBIO 4	SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024)	Calculation method, tool	PBL Netherlands. Pressure–response biodiversity model	https://www.pbl.nl
High-Conservation-Value (HCV) Approach	SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024)	Calculation method	HCV Network. Identification & management of critical biodiversity & ES values	https://hcvnetwork.org
IBAT	Damiani et al. (2023), SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024), De Ryck et al. (2024)	Tool, database, metric (STAR)	BirdLife/IUCN/WCMC. Subscription GIS of WDPA, KBA, Red List & STAR	https://www.ibat-alliance.org
InVEST	Damiani et al. (2023), SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024), De Ryck et al. (2024)	Calculation method, tool	Natural Capital Project. Open-source ES & habitat models	https://naturalcapitalproject.stanford.edu/software/invest
Land Use Change Improved LCA method (LUCI-LCA)	Damiani et al. (2023)	Calculation method	Chaplin-Kramer et al., 2017. Hybrid extended LCA that couples spatial land-change modelling (InVEST) with GLOBIO MSA to project future biodiversity loss and ES shifts across 8 land-use classes & intensities.	DOI:10.1016/j.ecolind.2017.10.040
Leeana	De Ryck et al. (2024)	Tool (web application)	Leeana.ai. Satellite + AI site scanners for corporates	https://leeana.ai
LIFE Methodology	Damiani et al. (2023), De Ryck et al. (2024)	Calculation method, tool (web application)	Instituto LIFE (BR). Corporate certification – pressure & performance indices	https://www.lifebrazil.org

Link	De Ryck et al. (2024)	Tool	Metabolic SaaS – nature risk & footprint hub	https://link.metabolic.io
Nala.Earth	De Ryck et al. (2024)	Tool	Cloud platform converting corporate data to TNFD/SBTN STEP 1 TOOLBOX (2024)-ready metrics	https://nala.earth
Nature Risk Profile	De Ryck et al. (2024)	Database, tool	UNEP-WCMC & S&P dataset of company nature dependencies & impacts	https://www.spglobal.com/esg/
Product Biodiversity Footprint	De Ryck et al. (2024), Damiani et al. (2023)	Calculation method, tool	I Care & Sayari. Eco-design metric for products; three-module framework merges LC-IMPACT with ecological refinements to cover all five IPBES drivers (semi-quantitative for invasive species & over-exploitation).	https://www.productbiodiversityfootprint.com
Site Biodiversity Footprint	De Ryck et al. (2024)	Calculation method, tool	I Care. Site-level assessment for corporate facilities; merges local surveys with pressure modelling to yield normalised km^2*MSA*year and guide action plans; aligned with 5 IPBES drivers.	https://www.i-care-consult.com
SLAM	De Ryck et al. (2024)	Database, tool	GIST Impact. Geofence assets vs. sensitive areas (TNFD)	https://gistimpact.com
Statutory Biodiversity Metric Tool (UK)	De Ryck et al. (2024)	Calculation method, tool (excel)	DEFRA/NE. Excel for habitat-based biodiversity units	https://naturalengland.org.uk
TESSA	Damiani et al. (2023), De Ryck et al. (2024)	Tool	Multi-NGO toolkit for site ES assessment with low-cost methods	https://tessa-tool.org
Trend.Earth	SBTN STEP 1 TOOLBOX (2024), Barth et al. (2024)	Tool (web application)	Conservation International. Land-degradation monitoring portal	https://trend.earth
Xylo Systems	De Ryck et al. (2024)	Calculation method, tool	Australian AI platform for asset biodiversity dashboards	https://xylosystems.org