INSTITUT FÜR ENERGIE- UND UMWELTFORSCHUNG HEIDELBERG

Product Life Cycle Assessment

Footprints and LCA concepts

Report on WP4 of the SYMOBIO 2.0 project:
Consolidation of systemic monitoring and modeling of the bioeconomy

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Abbreviations

AP	Acidification Potential
CCU	Carbon Capture and Utilization
DNP	Distance to Nature Potential
EoL	End of Life
EP	Eutrophication Potential
eq	equivalents
FU	Functional Unit
GHG	Greenhouse Gas
GWP	Global Warming Potential
IE	Inhabitant Equivalent
IPCC	Intergovernmental Panel on Climate Change
JRC	Joint Research Center
LCA	Life Cycle Assessment
LUC	Land Use Change
PE	Polyethylene
PET	Polyethylene terephtalate
PMF	Particulate Matter Formation
UBA	Umweltbundesamt

1 Background and goal

The aim and purpose of SYMOBIO is to provide a system for monitoring the sustainability of the bioeconomy in Germany. This system comprises a number of so-called footprints as indicators:

- Material footprint
- Forest footprint
- Agricultural footprint
- Water footprint
- Climate footprint
- Biodiversity footprint (new in SYMOBIO 2.0)
- Socio-economic footprint (new in SYMOBIO 2.0)

These footprints represent the resources used-or the emissions generated-as a result of domestic consumption of biomass-based products. For doing so, the production and supply chains are tracked, the exports are subtracted from the sum of domestic production and domestic production is added to imports. Imports are traced back to the countries of origin of raw materials using global multi-regional input-output datasets.

The SYMOBIO footprints therefore fundamentally follow the principle of the life cycle assessment (LCA), as established and standardized for product LCAs. The difference ultimately lies in the fact that the SYMOBIO footprints consider total material flows, whereas a life cycle assessment generally focuses on a product in terms of a specific functional unit. On the other hand, the results of the SYMOBIO footprints could also be understood as an aggregation of numerous product life cycle assessments.

The main difference lies in the granularity: for reasons of practicability alone, the SYMOBIO footprints cannot map ALL biomass products in ALL details. Generalized data sets are required here. An individual product LCA, on the other hand, can be designed in any level of detail, depending on the objective and framework. The approach of a macroeconomic material flow balance is therefore also referred to as "top-down" as opposed to the "bottom-up" approach of a product LCA.

A special difference between the SYMOBIO footprints and a LCA is that the system examined by SYMOBIO depicts a concrete actual situation of material flows in the bioeconomy. As a monitoring system, the footprints describe the development of resource expenditure or rather emissions derived from the bioeconomy over the years. However, they do not include a counterfactual scenario: what would it be like without the production of all these products? It would indeed be difficult to integrate such a scenario, as there are simply no real alternative products available for some products (e.g. staple foods). For other products, on the other hand, it is by no means clearly defined what

exactly the alternative products would be. In most cases, this requires assumptions and case distinctions, which then leads to ranges of results. A monitoring system would be overwhelmed by this, or at least overloaded.

Product LCAs, on the other hand, are specifically designed for such specialized cases and variant studies. For this reason, case studies on individual bioeconomy products are carried out as part of WP4 of SYMOBIO 2.0 using product LCAs. These case studies are the contents of the second deliverable of AP4 of the SYMOBIO project.

The focus is on how to handle biogenetic carbon and which methodologies are useful for footprinting biodiversity in the context of LCA.

First and foremost, it should be emphasised that both approaches follow the same methodology in principle, i.e. both are based on ISO 14040/44 for life cycle assessments and, as far as specific footprints are concerned, also on ISO 14067:2018 (Product Carbon footprint), ISO 14046:2014 (Water footprint) standards. The differences mainly arise in the area of specific applications and scopes. While the footprints of SYMOBIO 2.0 are very specifically designed to ensure clarity in the results – as required for monitoring – the LCA framework offers a great deal of variation in principle.

2 Overview on footprints and LCA impact categories

At the start of the SYMOBIO 2.0 project, one of the leading questions was what the differences and discrepancies between ‘standard LCA’ and SYMOBIO FPs are:

- What differences are caused by different approaches to system boundaries?
- Are there relevant environmental impacts not covered by SYMOBIO FPs?
- Where are the differences in the data bases?

2.1 System boundary questions

The major difference between

- Top down (SYMOBIO):
- Bottom up (LCA)

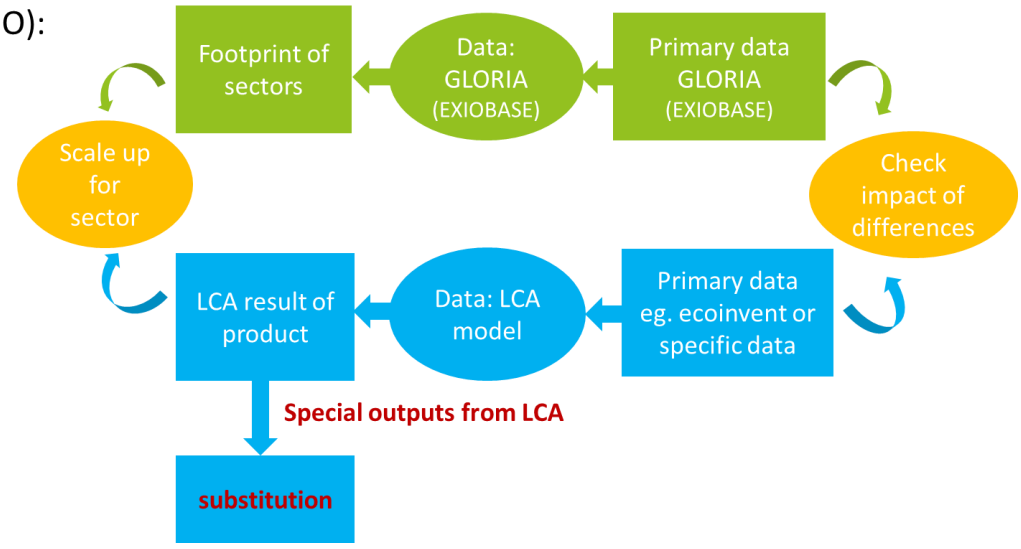


Figure 1:Life cycle assessment framework following ISO 14040 and ISO 14044

The results should provide a reliable overview of all environmental issues identified as relevant in the case studies under consideration. To this end, scenarios of complete life cycles are modeled for the individual biomass and product types. As the entire life cycle of a product is examined, the term "product systems" is used rather than "products"

2.2 LC Impact categories and SYMOBIO 2.0 footprints

Some key terms that are important for interpreting and assessing the relevance of impact assessment results in the LCA are explained below on the basis of the four phases "Goal definition and scope of investigation", "Life cycle inventory", "Impact assessment" and "Interpretation" and illustrated using examples of land use and land quality. Table 1 gives an overview of characterization approaches in impact assessment from LCAs reflected by the SYMOBIO 2.0 footprints.

Table 1: Assignment of different characterization approaches from LCAs to the SYMOBIO 2.0 footprints

SYYMBIO 2.0 footprint	LC impact category or indicator	Compatibility / deviations
WP2.3 Material footprint	Abiotic depletion (ADP) Kumulierter Rohstoffaufwand (KRA) (cumulated raw material demand CRD) Raw Material Index (RMI) Total Material Requirement (TMR)	In principle good match Depends on the selection of the indicator
WP2.3 agricultural land footprint	Land occupation (inventory date) structured align with the Corine land-use classes Flächenrucksack, land rucksack (UBA)	Good match
WP2.5 Water footprint	Water consumption (inventory date) z.T. differenciated in „blue“, „green“, „grey“ water Considering water scarcity according to the AWARE approach	Good match however, data base may differ
WP2.6 Climate footprint	GWP100 (GWP20) incl. Land-use change (LUC) Which approach? How to handle biogenic carbon: storage, emission, time dimension (so far only in LUC considered))	Good match (LUC is meanwhile also considered by the SYMOBIO 2.0 footprint)
WP2.7 Biodiversity footprint	Chaudhary & Brook; Köllner; de Baan; Biodiversity Value Increment (BVI) Lindner/Fehrenbach Naturfernepotenzial (NFP) distance to nature potential (DNP)	Degree of land use intensity in LCA is key in LCA approaches, SYMOBIO footprint is strongly concentrated of geographical background situation

3 Carbon balancing in product LCA

Product life cycle assessments (LCAs) are used to investigate the environmental impact of using products in comparison to alternative materials. This involves recording and evaluating the inputs, outputs and potential environmental impact of a product system throughout its entire life cycle. The greenhouse gas (GHG) balance as part of an LCA is suitable for determining GHG emissions and GHG savings along the product process chain and comparing them with alternatives, e.g. fossil and mineral products. When considering biomass products – either in a material application or as an energy source – the question arises as to how biogenic carbon is handled in GHG balances. Biomass is usually described as 'greenhouse gas neutral', i.e. the carbon released during combustion (either for energy use or disposal) is not included in the balance. However, this raises the following questions:

- How can we take into account the fact that carbon is stored for longer when used in durable wood products?
- How do we deal with the effects associated with removing wood from the forest in terms of its carbon storage or sink capacity?
- How can emissions from land use changes be taken into account?
- How can alternatives to land use be taken into account?

3.1 The methodological framework for accounting for carbon emissions in bio-based products

Biomass products represent a biogenic carbon store during the product's lifetime. While some bio-based products, such as biofuels, food or packaging, tend to have a short lifespan, other product applications may potentially store biogenic carbon over a period of decades to centuries. However, there is no consensus on how the temporary carbon storage of a product should be taken into account in product life cycle assessments. The following sections provide an overview of how such stores are handled and how they change at different levels. An increase in a biogenic store is referred to as a sink, while a decrease is referred to as a source.

3.1.1 International standard for the carbon footprint of products (PCF)

The PCF-ISO standard (ISO 14067 2018) makes a distinction between short-lived and long-lived products, whereby a product is considered long-lived if the emissions from its final disposal occur more than ten years after the start of its use. The effects of delayed emissions for long-lived products are reported for information purposes in the documentation of a PCF, i.e. they are not included in

the actual footprint. The standard specifies calculation methods, time-dependent characterisation factors and discounting approaches.

European standard for life cycle assessments of bio-based products (EN 16760:2016): Temporal effects of GHG emissions should be taken into account when relevant. However, no distinction is made between short-lived and long-lived products. An exception is storage in products that exceed a time horizon of 100 years: these are considered to be indefinite storage. For periods shorter than this, the principle of net zero carbon balance is applied, assuming that carbon dioxide sequestration during plant growth is equal to the release at the end of the life cycle (combustion or decomposition).

3.1.2 EU – Product Environmental Footprint Category Rules (PEFCR):

With version 6.3 (May 2018), the PEF (EU Commission 2019) is extended to include the issue of (temporary) biogenic carbon storage. The service life of the product is taken into account, and a distinction is also made between end products and intermediate products. For the latter, there are no credits under any circumstances, as further processing and final service life cannot be known. In the case of an end product and a cradle-to-grave approach, the service life of the product and – if the product originates from a forest – the storage period in the forest determine the storage potential. A credit is applicable for products with a service life of more than 100 years. This credit must be distributed across all (secondary) products of the system.

3.1.3 International methods (IPCC):

The IPCC guidelines were developed for national emission inventories (not for products) and contain a specific method for so-called 'harvested wood products (HWP)' (IPCC 2019) . The carbon bound in HWP forms a pool of carbon that was originally bound by the above-ground biomass carbon pool of forests and other wood-producing land categories. The carbon originally sequestered by vegetation is transferred to the HWP pool, similar to the transfer from the above-ground biomass carbon pool to the dead wood, litter and soil carbon pools in the LULUCF sector.

3.1.4 Further approaches in the literature:

Based on a comprehensive literature review, Soimakallio et al. (2022b) propose a solution for how the temporal C storage can be attributed to bio-based products with different lifespans. The rationale refers to the following fact: It is common practice to convert non-CO_2 greenhouse gases into carbon dioxide equivalents, typically using their cumulative radiative forcing over 100 years. The same basis can also be used for temporary carbon storage and delayed emissions in life cycle assessment. The authors refer to (Helin et al. 2016), who calculate that temporary carbon storage of 5, 10, 20, 40, 60 or 80 years results in a cumulative radiative forcing over 100 years that is approximately -5%, -9%, -17%, -34%, -52% and -72% lower cumulative radiative forcing over 100 years compared to the immediate release of the same amount of carbon dioxide.

(Fehrenbach, Bischoff, et al. 2022) take a similar approach, linking the carbon storage balance in forests with the carbon storage of wood products. The LCA model used covers the period from 2020 to 2050, which is in line with the general political goal of reducing greenhouse gas emissions by 2050.

3.2 The storage balance and sink capacity in forests

The CO_2 storage balance describes the influence of timber harvesting on the CO_2 storage capacity of a forest. Trees store CO_2 through their growth, which increases the CO_2 storage capacity of the forest area. When trees die, they become part of the deadwood pool. At the end of the decomposition process, a small proportion of the CO_2 previously bound in living biomass is stored in the soil as humus. The majority, however, is released again, reducing the CO_2 storage capacity of the forest area. If wood is harvested in the course of forest management, this also reduces the CO_2 storage capacity. The balance between CO2 sequestration and release through growth, decomposition after natural mortality and timber extraction leads to a net increase or decrease in the CO2 storage capacity of the forest area. This change in the CO2 storage capacity is referred to as sink capacity; it can be positive (net CO2 sequestration = sink) or negative (net CO2 release = source). If a forest is used less intensively, the sink capacity improves. If use is intensified, the sink capacity decreases. The storage balance represents how much the CO_2 storage capacity changes when one cubic metre of wood is harvested and is expressed in tonnes of CO_2 per cubic metre of wood harvested.

The storage balance can be quantified using forest models and a comparison of different intensity scenarios for timber use. In the modelling, the difference in CO_2 storage capacity in the forest is compared with the difference in timber extraction for alternative forest management scenarios. The extent of the impact of timber harvesting on the CO_2 storage balance depends in particular on the forest structure, e.g. high storage balance in young and fast-growing forests, low in old, unstable forests.

A literature study determined an average CO_2 storage balance of 1.2 t CO_2/m³ (standard deviation ±0.7 t CO_2/m³) for boreal and temperate forests (Soimakallio et al. 2022). This includes all changes in all carbon pools in the forest area. As described in the introduction, wood is considered 'greenhouse gas neutral' in product-related GHG balances. This means that when wood is used for energy or recycling, the carbon bound in the wood and released during combustion is not taken into account in the GHG balance. This is based on the argument that the CO2 released during combustion of wood was previously bound by the growth of the wood. However, this neutrality is contradicted by the above-described relationships regarding the storage balance. The situation is similar with the rules of the GHG balance according to the IPCC (see above). Accordingly, product GHG balances should at least take into account the amount of CO2 stored in wood as carbon. This parameter can be determined as a characteristic value for tree species without complex forest modelling (e.g. spruce 0.7 t CO2 / m³ and beech 1.0 t CO2 / m³). This factor could be used to represent a large part of the effects on the forest area (comparison with the values determined in (Soimakallio et al. 2022).

The amount of CO2 bound in wood must also be taken into account when using woody residues and waste materials, as many of these residues and waste materials can alternatively be used in durable products, i.e. they could remain in the product storage.

3.3 The assessment of land-use change in LCA

Land-use change is typical case in practice for carbon balancing. While it is common practice in LCA to excluded CO_2 emissions from the use of biomass (be combustions or fermentation or just decay) from adding to Global Warming Potential (GWP), emissions from land-use change are to considered. This interpretation somehow relates to the IPCC GHG methodology for national inventory reports. As for CO_2 emissions from wood this is somehow misinterpretation of the methodology which accounts the removal of wood from forests by a complete release of the contained carbon as CO_2. However for land-use change the methodology is different, it just accounts the change of carbon stock before and after an change in use.

For the application of LUC in LCA several questions can be considered when assessing GHG emissions from land use and land use change (LULUCF), such as:

- Counterfactual approaches: 'What could be instead?'
- Retrospective approaches: 'What was there before?'

The following two sections describe both possibilities.

3.3.1 Carbon opportunity costs (COC)

The storage balance described in the previous section describes the consideration of effects on forest area. In contrast, the cultivation of annual crops in agriculture does not have such effects, as the carbon pool 'living biomass' is built up (plant growth) and broken down again (harvest, dead plant residues) within a year. However, in the case of such areas, the question could be asked as to how much CO2 could be stored if arable land were converted to forest. This is based on the idea that arable use (soil cultivation, harvesting, weed control, etc.) prevents succession to forest or afforestation (cf.(Searchinger et al. 2018)).

These so-called carbon opportunity costs (COC) describe the lost storage effect of natural vegetation that could develop on the occupied area instead of biomass production.

If these CO_2 opportunity costs are taken into account in the GHG balance of biofuels, for example, it becomes apparent that renaturation or reforestation of arable land leads to a higher GHG reduction of approximately 40% compared to the use of biofuels (Fehrenbach and Bürck 2022).

These opportunity costs are not taken into account for agricultural residues and waste, as it is assumed that these are not associated with land use. The application of COC requires an alternative to the product. The approach is therefore hardly applicable to essential staple foods.

3.3.2 Attributive land use change (aLUC)

While the COC approach described in the previous section can be used to evaluate existing land use, the following section deals with changes in land use. Emissions from land use change (LUC) are regularly reported and disclosed in national reporting. They must also be calculated as part of the sustainability requirements of the Renewable Energy Directive (RED). A distinction must be made here between 'direct' and 'indirect' land use changes. Direct land use changes refer specifically to the actions of a producer. Under the RED, affected farmers must assign their product to a specific cultivation area within the supply chain. If there is a change in land use, this effect can be calculated on a product-specific basis. However, application to generic products/markets within the framework of product life cycle assessments is problematic.

Indirect land use effects, on the other hand, are triggered by the manufacture of a product elsewhere. The calculation is complex and is carried out using combinations of land use models with economic equilibrium or partial models. Taking such effects into

account in product life cycle assessments is problematic for reasons of consistency of system boundaries.

However, real land use changes are taking place, which can also be verified using various data sources. Using the same system as for attributing real emissions to the processes of a life cycle, this is also possible with real land use changes. The calculation considers the actual situation of land use change due to agricultural activity on an empirical basis and provides a general emission factor that attributes the actual and measurable LUC in a defined agricultural area to all producers in that agricultural area: The land use changes actually caused by agriculture (in a defined region) are attributed to all agricultural products (including bioenergy) in proportion to the amount of land required. This is therefore an attributive attribution, which is why this variable is referred to as attributive LUC (aLUC). The attributive land use change (aLUC) approach was developed as part of the UBA BioEm project (Fehrenbach et al. 2016) on the basis of such relationships and has already been applied to relevant bioenergy sources (Keller et al. 2021), (Fehrenbach et al. 2020).

This approach disregards ‘consequential’ relationships. Even if a specific use, such as corn for biogas, is suspected to be a ‘driver’ for LUC, the aLUC does not differentiate here. Agricultural products are treated equally.

Implications

The concepts described above show that **the narrative of ‘CO_2-neutral' biomass needs to be questioned.** The use of agricultural biomass for additional energetic and/or material use can be a driver of (indirect) land use changes, which annually leads to immense greenhouse gas emissions worldwide (according to (Friedlingstein et al. 2022) over the period of 2012 and 2021, 17 % of global GHG emissions are associated with land use changes / deforestation. The use of forest wood has significant impacts on the forest as a CO_2 sink, while long-lived “harvested wood products” can also be a strategy to “store” carbon. Against this backdrop, the consumption of 'land' and 'forest-based wood' should be weighed against the goals for a sustainable and balanced bioeconomy (see Section 2.1). This means not only focusing on the sustainable production of biomass, but also on the sustainable consumption of biomass-based products. For this, monitoring tools like LCA need to be adapted, augmented or complemented by approaches that reflect the system-wide impacts of that consumption, especially as regards GHG emissions.

4 Biodiversity in LCA

4.1 Fundamentals of life cycle assessment/footprinting and biodiversity:

The difficulty lies in reducing the complexity of biodiversity to a single impact indicator. Unlike climate impact (--> GWP100), for example, biodiversity is difficult to express in a single measure. The complexity can be reduced by limiting the analysis to individual elements of biodiversity, such as

- alpha biodiversity
- certain taxonomic groups.
- Use of proxies, such as
  - (physico-chemical) soil properties
  - intensity of human intervention (hemeroby)

In this context, the question arises, particularly with regard to biodiversity in LCA, whether the **midpoint** level is appropriate or whether the **endpoint** is required. The endpoint level means that everything that affects biodiversity is aggregated in an integral indicator: impact on land use, through climate change, through substance inputs (acidification, eutrophication, ecotoxic substances), for which other LCA categories already exist. impact on land use, climate change, substance inputs (acidification, eutrophication, ecotoxic substances), for which other LCA categories already exist and are evaluated independently.

Since the complexity increases enormously with the goal of an endpoint indicator and the transparency of the final results is hardly possible, midpoint has been preferred for biodiversity so far.

For biodiversity, this is usually a midpoint indicator characterised by area (occupation). The area is the initial parameter, which, depending on the model approach, is assigned an ecological value rating using a characterisation factor.

With regard to land use and biodiversity, the work carried out within the framework of UNEP/SETAC still forms a central guideline for land use impacts on biodiversity in LCA: a global approach (Koellner et al. 2013), (Verones et al. 2017), (De Baan et al. 2013). These early works refer to the so-called the bathtub diagram Figure 2, which begins with the initial state of land use, describes the ecological loss of the area through use over the period of use and includes a partial restoration of the initial state at the end of use. However, since both the initial state and the period of use are not available as information in most cases, this model is rarely used in its entirety.

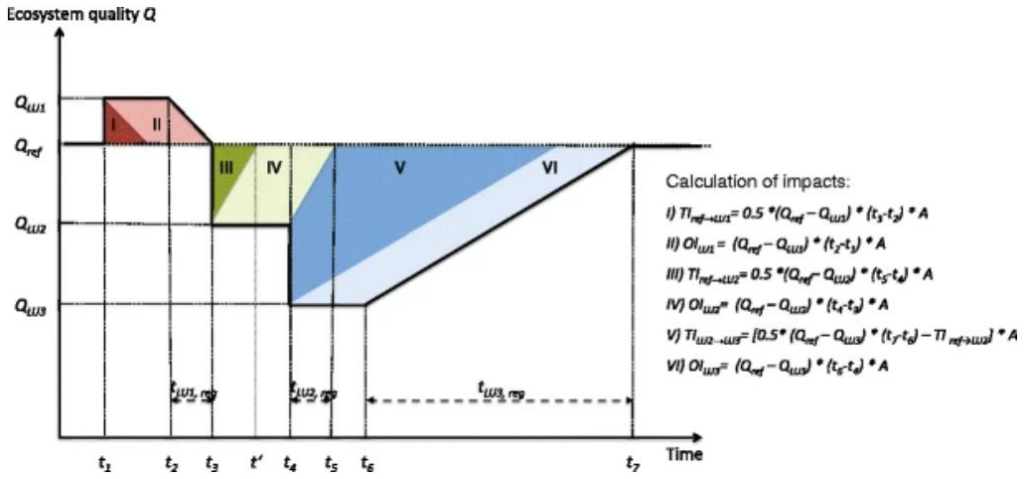


Figure 2: Simplified illustration of transformation impact (TI) and occupation impact (OI) for three land use types with different regeneration rates (t LU1, reg, t LU2, reg and t LU3, reg). For simplicity, the area A of occupation or transformation, which would embrace the third dimension, is not shown in the graph, but in the equation (Koellner et al. 2013)

4.2 Methods for evaluating biodiversity in LCA

There are now a large number of methods for evaluating biodiversity in life cycle assessments. Within the scope of this work package, three methods were examined in particular, including the BVI according to (Lindner et al. 2020), the “Land Rucksack” according to (Fehrenbach, Mascha Bischoff, et al. 2022) which is based on hemeroby and the method by (Chaudhary and Brooks 2018), which were compared with each other.

4.2.1 The land rucksack – hemeroby approach

At the life cycle inventory level, the land rucksack applies both quantitative data in the physical unit square metre and qualitative data, i.e. information on the hemeroby of the area in use. For the quantitative data, a distinction is made between two basic variables:

1. Land occupation during the time of use (temporary land occupation)
2. Land transformation associated with land use.

The principle of **hemeroby classes** has a in tradition in landscape ecology (Sukopp 1976). Seven-part class systems are widespread and first applications in LCA were introduced by (Schmitz et al. 1999) for the land—use category in the UBA-*Bewertungsmethode in Ökobilanzen*. The first stages for forest systems were further developed by (Fehrenbach et al. 2015) for agricultural systems With the Land Rucksack (Fehrenbach, Mascha Bischoff, et al. 2022) the method was further mainstreamed.

The sorting of land into discrete classes instead of application of a continuous scale does justice to the complexity and multi-layered nature of hemeroby. This cannot be properly

expressed with a single parameter (e. g. nitrogen input per unit area or proportion of deadwood). The seven-part classification system has proven its worth, e. g. to allow allocation of common land use types. The first step is to classify the basic land use types of forest, grassland, arable land and various types of settlements with their respective ranges according to the classification system (see Figure 3).

The hemeroby class system with indicative allocations of the distribution of different land use types

Hemeroby class	Forest/Forestry	Agriculture		Settlement areas		
I natural	Primary forest, no use					
II	More					
III	to	Grassland, extensive, species-rich to intensive management				
IV	less natural forest		Arable land, extensive to intensive management	Derelict land		
V	management				Mining areas	
VI	Timber plantations					
VII						Sealed surface

Figure 3: The hemeroby class system with indicative allocations of the distribution of different land use types (Fehrenbach, Mascha Bischoff, et al. 2022)

The **allocation of land in use to a specific hemeroby** class is carried out by means of criteria or, more precisely, criteria sets with metrics for determining the intensity of intervention associated with respective human use. Since the type of intervention differs greatly between forest, grassland, arable land or settlement areas, a uniform catalogue for all types of use is not appropriate. What may be a criterion for increased hemeroby in a forest (e.g. fragmentation) may result in lower hemeroby in an agricultural setting (e.g. diversity, structural diversity). For this reason, separate sets of criteria with their own metrics were developed for the different types of land use.

The **impact assessment** summarises life cycle inventory results in one impact category. Consequently, individual land use results, differentiated according to hemeroby classes, are summarised into one indicator value on the level of the LC inventory. But how are x m^2 of class II, y m^2 of class III and z m^2 of class VII added into a total? For this, (Fehrenbach et al. 2015) proposed a characterisation model defined on the fining that the minimum range between the class with the lowest impact of an land in use (hemeroby class II, since class I means no use at all) and the highest hemeroby (class VII) must be at least a factor 33. Another finding was, that the steps between the classes do not follow a linear function: a near-nature forest (e.g. class II) and an extensively managed forest (e.g. class III) are much closer in terms of ecological value and biodiversity than a sealed areas (class (VII) from an intensive cropland (class VI). The most logical mathematic behind the characterisation of hemeroby classes has been detected as the factor 2. This is shown in

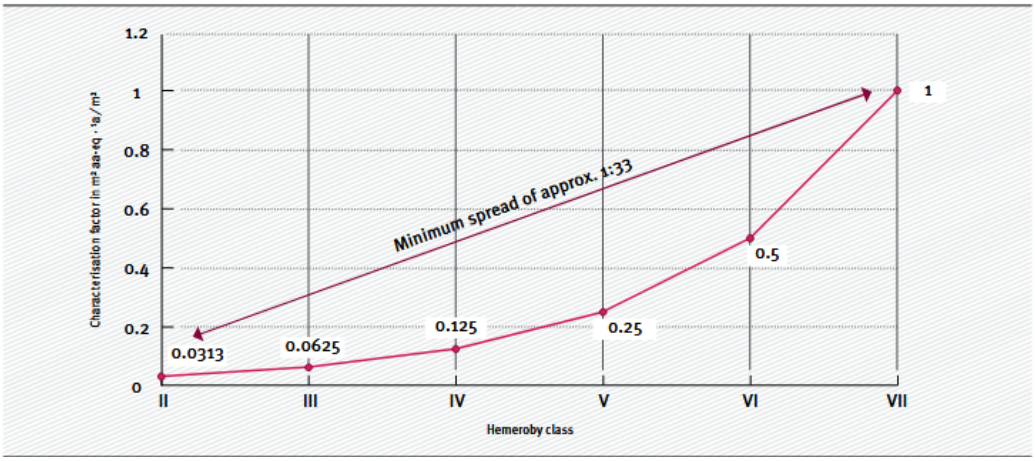


Figure 4: Allocation of individual hemeroby classes to characterisation factors (Fehrenbach, Mascha Bischoff, et al. 2022)

4.2.2 The BVI method

The BVI method combines the fuzzy framework of (Lindner et al. 2020)with the hemeroby approach of (Fehrenbach et al. 2015) in a framework previously proposed by Maier et al. [18]. According to hemeroby concept, the degree of naturalness (reciprocal to decreasing human influence) is set as a proxy for biodiversity value. However, it only provides a rough calculation structure in seven hemeroby classes. The BVI converts the hemeroby approach into a relatively fine-grained calculation structure for biodiversity value, but refrains from clearly defining this value. Maier et al. [18] use a regional weighting factor, fixed intervals per land use type and a set of management parameters per land use type. The BVI was developed to meet the above-mentioned requirements of various interest groups and to close research gaps in this area. It produces a quality indicator for a given part of land, described by a Q value for use in the Q, A, t diagram of the land use framework referenced in Figure 2. The input data from which the Q value is calculated are properties of the land; mostly management parameters. The calculation follows a sequence of steps that is structurally the same regardless of which kind of land use is assessed.

Out of these steps, two are relevant for the development beyond the mere hemeroby approach, covered by the land rucksack:

1. The dissolution of the seven ordinal classes into fluid values on a scale from zero (minimal biodiversity or full hemeroby) to one (maximum biodiversity, no hemeroby).
2. Multiplying these values by an **ecoregion factor** as a weighting factor that makes the biome-specific and land use-specific biodiversity values globally comparable.

4.2.3 The method according to Chaudhary and Brooks

Unlike the two previous methods, this approach is not based on hemeroby but on actual or potential **species loss**. However, here too, the basis is land use (or land use change). This is multiplied by the characterisation factor (potential species loss).

(Chaudhary and Brooks 2018) distinguish regional and global CFs. Regional species loss per unit of land use and land use change takes into account that some of these species are endemic and their loss leads to global species loss. Global CF (the regional CFs for each taxon per ecoregion are multiplied by a vulnerability value for that taxon in that ecoregion). Figure 5 gives an overview of the major components of the method.

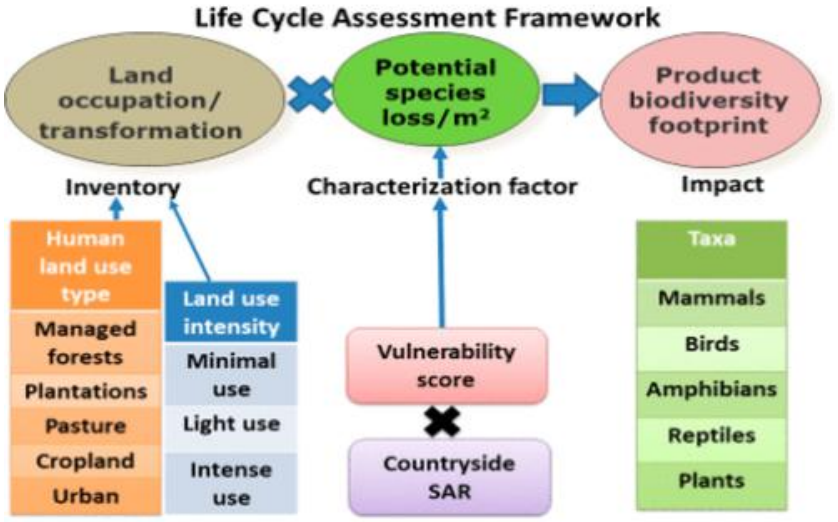


Figure 5: Main components of the method by (Chaudhary and Brooks 2018)

4.3 The biodiversity footprint of SYMOBIO 2.0

The biodiversity footprint developed by iDiv as part of SYMOBIO 2.0 – as described in the The Bioeconomy Monitoring Report 2024 (Beck-O´Brien et al. 2024) – quantifies the impact of biomass-based raw material production on biodiversity. The focus is on the impact of agricultural and forestry production on species and ecosystems. It thus represents a 'production footprint' or 'consumption footprint'. The latter measures the impact of production on biodiversity contained in the raw materials consumed by a specific country and sector, such as the German bioeconomy.

Methodologically, this biodiversity footprint measures changes in the local persistence of individual species and their habitats within larger ecosystems that are affected either by habitat displacement due to land use changes or by habitat degradation due to more intensive production practices.

The footprint expresses how both the expansion of production areas and the intensification of land use affect suitable habitats, thereby threatening the survival of species and the integrity of ecosystems. The framework for calculating the biodiversity footprint provides a detailed understanding of the relationship between consumption patterns in one region and biodiversity loss in production areas elsewhere.

The maximum level of geographical resolution, by which the impacts can be separately presented is a 10 × 10-km pixel and year, referring to each species and ecosystem type, for each commodity, and for land-use expansion and intensification. Alternatively, comprehensive biodiversity footprints can be calculated that reflect, for example, the cumulative impacts of production or consumption of all commodities on all vertebrates and all ecosystems worldwide over multiple decades.

For the biodiversity footprint, two primary types of **land-use data** are considered:

1. land-use expansion data, provided by satellite observations and/or land-use models, which detail the conversion of natural habitats into agricultural or forestry production areas, and
2. land-use intensification data, capturing changes in the share of the maximally achievable yields in a given region that are currently achieved within already cultivated areas, and which serve as a proxy for capturing intensification-associated management practices, such as increased pesticide or fertilizer use or higher livestock densities.

In this sense the SYMOBIO biodiversity footprint is perfectly in line with the approaches for LCA explained before, which also refer to land-use typology. The valuation of biodiversity here refers to models of species distribution dynamics which map the geographic areas where species occur and indicate the areas covered by the habitats providing the specific environmental conditions that each species needs to survive. These datasets provide the foundation for modelling how production activities affect local habitats and species.

Where biomass production leads to an **expansion of land use**, the impacts are quantified in terms of the area of natural habitat lost to raw material production activities (e.g. conversion of forests to arable land). The loss of habitats means a reduction in local population persistence.

Where biomass production leads to **land use intensification**, the impacts are estimated by considering these affect habitat quality (e.g. through pollution) and reduce the suitability of remaining habitats for species survival. By combining these two types of impacts (expansion and intensification), the method calculates the changes in the probability of local survival for each affected species.

Starting at the **species level**, the impacts are extrapolated to larger taxonomic groups (e.g. all primates, all mammals or all vertebrates) or larger geographical regions. The global rarity of species is taken into account by relating local changes in persistence to the habitat areas and persistence values of the species. If the aggregate persistence score of all species in a larger region falls by 1, this indicates that one species has become extinct worldwide or that 50 species have each lost 2% of their total habitat. The result is a biodiversity footprint for the taxonomic group or region that reflects the cumulative impact of raw material production.

The link between species or habitat loss and **specific products** is established by tracking land use changes associated with certain crops (e.g. soy, oil palms). By identifying the areas where these raw materials are produced and assessing the impact on biodiversity in these regions, a biodiversity footprint can be assigned to specific raw materials. At the same time, the impacts can also be calculated at the level of broader land use sectors (e.g. all agricultural land) by aggregating the impacts of several raw materials

produced in the same region on biodiversity. This enables a sector-wide assessment of the impacts of land use changes on biodiversity. Subject to further methodological adjustments, the workflow can be expanded to comprehensively cover agricultural or forestry products (e.g. including livestock, wood, etc.).

Case study Biodiversity impacts of German consumption of soy and sugar imported from Brazil

The case study focusses on **soy** and **sugar** from the world leader in soybean and sugarcane production, Brazil. Soybeans mainly grown in the species-rich Cerrado and sugarcane in the even more biodiverse Mata Atlântica biome. Between 1997–2007 and 2008–2018, Germany's soy imports from Brazil declined significantly (−55%), while sugarcane imports rose sharply (+162%). Contrary to the expectation that these consumption trends would have a proportional impact on the biodiversity footprint, the data paint a paradoxical picture: biodiversity loss due to sugarcane imports fell by 87%, while that due to soy imports rose by 134%.

These opposing developments can be explained by differences in production systems and their spatial extent. The expansion of sugarcane production took place mainly within already intensively used agricultural areas, where biodiversity had previously been greatly reduced. In contrast, soybean production expanded into previously extensively used, species-rich peripheral areas of the Cerrado. These regions were home to largely intact species communities, including many rare and sensitive species that were particularly affected by intensification.

The case study emphasises that high biodiversity footprints cannot necessarily be avoided by extensive production systems, as these require larger areas of land and thus cause other ecological impacts. In addition, the biodiversity losses recorded are exclusively attributable to new land use changes during the respective study periods. Historical losses and indirect land use shifts – such as those caused by the displacement of other production systems – are not taken into account.

A precise assessment of biodiversity footprints requires fine-scale, time-differentiated data that takes into account the spatial distribution of species, their endangerment and specific land use dynamics. Coarse-grained maps or blanket species counts lead to significant misinterpretations. Further development of monitoring methods is therefore necessary in order to adequately capture indirect effects such as spatial spillover and historical land use legends and to realistically assess the ecological sustainability of global supply chains.

4.4 Expert workshop on this topic in the course of SYMOBIO 2.0

A workshop was held in February 2024 as part of the project to address the question of how biodiversity is taken into account in life cycle assessments and what rate is used for the biodiversity footprint in SYMBIO 2.0. After a brief introduction to the core aspects of biodiversity, the workshop discussed

1. What levels of biodiversity exist and how are they quantified?

2. Which drivers of biodiversity loss play which roles?

The focus was on the question of which methods for quantifying biodiversity impacts are used in LCA and footprinting, and which metrics are commonly used. It was reflected on the immensely complex dimension of real biodiversity to the need for simplifications and ‘proxy approaches’. The discussion was deepened using two example methods:

- Biodiversity in life cycle assessments: The BVI method
- The biodiversity footprint in the context of SYMOBIO 2.0

Discussion on the possibilities and limitations of the approaches presented and their consideration in product life cycle assessments (with a focus on existing standards)

5 Conclusions

This report addresses a number of aspects from an LCA perspective that are relevant in the context of developing footprints for monitoring the bioeconomy according to SYMOBIO. First, the SYMOBIO footprints are discussed in light of the usual impact categories of LCA. It becomes clear that, in principle, the selection of footprints provides good coverage. It should be noted that in LCA, a selection is often made and a comprehensive list of categories is not calculated in all cases. Experience shows that a reduced selection of categories is usually sufficient to achieve the LCA result. The case studies in the second report on this WP4 are also limited to a selection.

Finally, the decisive factor is the data used for the assessment. LCA differs from SYMOBIO FP primarily in that LCA can go much deeper and into more detail in individual cases, allowing for a broader data basis or specific data to be used. This question of (possibly higher) granularity can be considered the essential difference between LCA and SYMOBIO FP. It is directly connected with the two different basic approaches of both: the bottom-up and the top-down perspective. In addition to that, the question of substitution calculation makes another relevant difference, as can be seen from the case studies in the second report.

Carbon balancing in LCA leaves still a lot of development or refinement. This area is still marginally standardised. There are frameworks but no real consensus. The most controversial issue is indirect land use change. In principle, the consideration of LUC is widely accepted. However, the question of which method to use has not been clarified. It should be noted that the addition of LUC to the carbon footprint of SYMOBIO is methodologically very similar to the attributional LUC (aLUC) approach presented here.

There is also little standardisation for biodiversity to date, even though numerous methods exist. The SYMOBIO FP were supplemented by a biodiversity FP as part of SYMOBIO 2. Like almost all LCA biodiversity approaches, this has an area reference. However, the impact on biodiversity is directly related to the loss of species or habitats through land conversion or intensification, whereby a direct link to biomass products can be established. In this sense, this approach is also closely related to the methods used in LCA and may even help to refine it further.

Ultimately, it can be said that the FP and what the LCA offers are basically in line with each other. The main difference remains: the LCA is the appropriate instrument for individual cases and differentiation, and it offers the possibility of analysis through direct comparison with reference products and thus also alternative products or substitution effects. Monitoring at the macroeconomic level naturally has a completely different scope.

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