Incorporating local species data into global biodiversity assessment framework

A methodological analysis of the GLAM framework for Swedish grasslands

Tilda Lagerhäll

Degree project/Independent project • 15 credits
Swedish University of Agricultural Sciences, SLU
Faculty of Natural Resources and Agricultural Sciences
Department of Energy and Technology
Biology and Environmental Science – Bachelor's Programme
Examensarbete (Institutionen för energi och teknik, SLU) 2026:14 • ISSN 1654-9392
Uppsala 2026

Incorporating local species data into global biodiversity assessment framework

Inkorporering av lokala artdata i globalt ramverk för bedömning av biologisk mångfald

Tilda Lagerhäll

Supervisor:	**Lindsay Holsen, Swedish university of Agricultural Sciences, Department of Energy and Technology**
Assistant supervisor:	Johan O Karlsson, Swedish University of Agricultural Sciences, Department of Energy and Technology
Examiner:	Elin Röös, Swedish University of Agricultural Sciences, Department of Energy and Technology

Credits:	15 credits
Level:	First cycle, G2E
Course title:	Independent project, environmental science
Course code:	EX0896
Programme/education:	Biology and Environmental Science – Bachelor’s Programme
Course coordinating dept:	Department of Energy and Technology
Place of publication:	Uppsala
Year of publication:	2026
Cover picture:	Photo by author
Title of series:	Examensarbete (Institutionen för energi och teknik, SLU)
Part number:	2026:14
ISSN:	1654-9392
Keywords:	species richness, semi-natural grasslands, biodiversity loss, life cycle assessment, land occupation

Swedish University of Agricultural Sciences
Faculty of Natural Resources and Agricultural Science
Department of Energy and Technology

Abstract

Biodiversity is declining at an alarming rate. Human land use causes land fragmentation and habitat loss, both of which contribute to the decline in species richness. Therefore, it is important to improve the way biodiversity is assessed in life cycle assessments (LCAs) to more accurately reflect the impact of land use. Currently, there is a limitation in global biodiversity assessments used in LCA as they are too general regarding land types and management practices. One consequence of this is that land use with positive biodiversity impacts risks being overlooked. This is particularly relevant for semi-natural grasslands, which are biodiversity rich ecosystems in need of maintained management to preserve many species. This study investigated whether local Swedish grassland data could be integrated into the GLAM biodiversity assessment framework recommended by the UNEP Life Cycle Initiative to better reflect its biodiversity impact. An exploratory methodology was employed, involving the deconstruction of the framework to trace the flow of equations and map them against the available local data, to identify data gaps. Despite good access to empirical species data for grasslands in Sweden, the findings suggest that integrating these into GLAM involves several data and resource demanding steps. For examples, parameters such as area of habitat patches and inter-patch distance were missing in order to capture the consequences of fragmentation intended to be reflected in the characterisation factors (CFs) generated. Ultimately, the study raises doubt about the relevance of incorporating country-based site-specific empirical biodiversity data into a global model like GLAM that relies heavily on modelled biodiversity.

Keywords: species richness, semi-natural grasslands, biodiversity loss, life cycle assessment, land occupation

Sammanfattning

Den biologiska mångfalden minskar i en alarmerande takt. Mänsklig markanvändning orsakar fragmentering av mark och förlust av livsmiljöer, vilket bidrar till minskad artrikedom. Därför är det viktigt att förbättra sättet att bedöma den biologiska mångfalden i livscykelanalyser (LCA) för att bättre återspegla effekterna av vår markanvändning. För närvarande finns det en begränsning i bedömningarna av den biologiska mångfalden, då förvaltningsmetoder generaliseras. Detta innebär att markanvändning med positiva effekter på den biologiska mångfalden riskerar att förbises. Detta är särskilt relevant för naturbetesmarker, som har hög biologisk mångfald och som behöver upprätthållna förvaltningsmetoder. I denna studie undersöktes om lokala svenska gräsmarksdata kunde integreras i GLAM-ramverket för bedömning av biologisk mångfald, som rekommenderas av UNEP Life Cycle Initiative, för att bättre återspegla dess påverkan på den biologiska mångfalden. En explorativ metod användes, som innebar att ramverket dekonstruerades för att följa ekvationernas flöde och kartlägga dem mot tillgängliga lokala data, i syfte att identifiera dataluckor. Trots god tillgång till empirisk artdata för svenska gräsmarker visar resultaten att en integrering av dessa data i GLAM kräver flera resurskrävande steg. Till exempel krävs, parametrar såsom area av habitatfläckar, avstånd mellan fläckarna och motståndskraften hos det omgivande landskapet saknades för att kunna fånga konsekvenserna av fragmenteringen som skulle återspeglas i de genererade karakteriseringsfaktorerna (CF). Sammantaget ifrågasätter studien relevansen av att integrera landsbaserade, platsspecifika empiriska biodiversitetsdata i en global modell som GLAM, vilken i hög grad bygger på modellerad biodiversitet.

Nyckelord: artrikedom, naturbetesmark, biodiversitetsförlust, livscykelanalys, markanvändning

Table of contents

List of tables

List of figures

Abbreviations

Abbreviation	Description
CF	Characterisation Factor
GLAM	Global Guidance for Life Cycle Impact Assessment Indicators and Methods
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
SLU	Swedish University of Agricultural Sciences
PDF	Potentially Disappeared Fraction

Introduction

Biodiversity is fundamentally important for functioning ecosystems (Zhen et al. 2025). During the last decades biodiversity has decreased at an alarming rate while the human activities driving biodiversity loss continue to intensify (Winter et al. 2017; Järviö et al. 2025). This development is critical both due to the intrinsic value of biodiversity and since the loss risks to degrade the contribution of ecosystem services such as fresh water and air, climate regulation and food provision (Winter et al. 2017). One of the biggest contributors for terrestrial species loss is land use and land use change (De Baan et al. 2013; Scherer et al. 2023). Due to the land fragmentation and degradation of habitats caused by land use, species richness has decreased by 13.6% (Newbold et al. 2015). The importance of assessing the biodiversity loss is therefore essential to reduce the impacts caused by anthropogenic land use (Scherer et al. 2023).

Life Cycle Assessment (LCA) is a widely used tool for quantifying the environmental impact of products and services throughout their entire life cycle (Winter et al. 2017; Scherer et al. 2023). Originally developed to assess the impact of material and energy flows in industrial products, it is now employed for a wide range of products and services, as well as for various environmental impacts including carbon footprint, land use, water use, and biodiversity loss (Zhen et al. 2025). Foods in meals and diets typically originate from multiple countries which calls for global methods. However, there are limitations to the methods of biodiversity assessment within the frameworks of global LCAs since the scale is general (Zhen et al. 2025). Even though it is important to assess the global state of biodiversity, resilient delivery of ecosystem functions is highly dependent on diversity on a local level (Newbold et al. 2015). The complexity of biodiversity, with its differentiated landscapes and spatial variations, is therefore often obscured in global models. This is due to the lack of nuance regarding the effect of specific land management on local environments. It is central to incorporate more empirical data and local level ecological details to bridge the gap between biodiversity impacts and its modelling within the Life Cycle Impact Assessment (LCIA) phase of LCA, in which inventory data is translated to an impact score (Holsen et al. in prep). Otherwise, these methods risk misrepresenting actual biodiversity impacts due to overly generalised land use classifications, thereby missing potential solutions to reduce biodiversity loss. (Holsen et al. in prep).

The necessity of differentiated assessments to accurately represent biodiversity impacts becomes evident for land use types that have the potential of having positive impacts on biodiversity. Semi-natural grasslands are biodiversity rich ecosystems in European ecoregions and are critical habitats for about a third of Sweden's red listed species (Eriksson 2022; Prangel et al. 2024). Vascular plants are one taxon commonly used to represent terrestrial biodiversity and can indicate key differences of biodiversity across grassland types. An extensive monitoring program, Remiil (Lundin et al. 2016) is one example of extensive, field data from a grassland survey that can be applied toward LCA, as in Holsen et al. (in prep). These datasets offer an opportunity to be integrated in existing LCIA methods, such as the biodiversity assessment framework GLAM, recommended by the United Nations Environment Programme (UNEP) Life Cycle Initiative. This could have the potential of increasing the resolution of biodiversity impacts in LCA.

This study investigates whether local Swedish grassland data could be integrated into the GLAM biodiversity assessment framework recommended by the UNEP Life Cycle Initiative to better reflect its biodiversity impact.

1.1 Objective

The thesis explores the possibility of integrating local species richness data in the global biodiversity assessment framework GLAM, to more accurately reflect local ecological conditions for vascular plants in Swedish grasslands. The aim is to bridge the gap between theoretical global LCIA frameworks with empirical data for local conditions and site-specific managements by addressing the following research questions:

How can available local species data be used in the LCIA method GLAM to develop characterisation factors that assess the potential regional and global biodiversity loss from land use?

What parameters and additional inputs are required in the GLAM framework to calculate such regional and global characterisation factors for land-use-related biodiversity impacts?

1.2 Background

1.2.1 LCA and biodiversity assessments

Biodiversity is one of the most complex impacts to quantify in LCAs. The methods developed for water use or carbon footprint are well established, easier to quantify and relatively straight forward (Crenna et al. 2020). While as for biodiversity there is not one accepted method but approximately 17 methods (Damiani et al. 2023), all with limitations and shortcomings. They are commonly inadequate at capturing biodiversity's complexities (Zhen et al. 2025), emphasizing the importance of developing the biodiversity assessment methods further and with greater consensus among methods. Biodiversity loss is evaluated as both a midpoint (e.g. land occupation) and an endpoint indicator (damage to ecosystems), representing the final consequences effected by other indicators such as land use.

The land use impact on biodiversity in LCA is calculated using a characterisation factor (CF) indicating the potential species loss from using or transforming a unit of land e.g. one m^2 (Chaudhary & Brooks 2018). When assessing the biodiversity impact of a product in LCA, the CF is multiplied by the area of land use or land use change from the life cycle inventory, e.g. the amount of land used of a certain type to produce one unit of product. In this thesis only land use occupation, and not land use change or transformation, will be examined.

During the last decade biodiversity assessment methods have developed rapidly. To gain consensus the United Nations Environment Programme (UNEP) Life Cycle Initiative's 'Global Guidance for Life Cycle Impact Assessment Indicators and Methods' (GLAM) initiated a task force (Coelho et al. 2025). They attempt to capture the complexity of biodiversity by incorporating several factors that affect the species interaction with and between different land uses. The first recommended method was based on the well-known ecological theory, species-area relationship (SAR), that states the dependence on the area for species richness in a sample region (Conor & McCoy 2013; De Baan et al. 2013; Chaudhary et al. 2015). To better account for mosaic landscapes this concept was developed into countryside species-area models (cSAR), which includes species groups' affinity in different habitats within a landscape (Coelho et al. 2025). The method was further developed by including five types of land uses and three intensities and how they affect different taxonomic groups, and thus also include whether the region contains endemic or endangered species (Chaudhary & Brooks 2018). The most recently published method by Scherer et al. (2023) is based on the previous research and further include effects created by fragmentation in the landscape proposed by (Kuipers et al. 2021b) and global scale perspectives by including global extinction probability proposed by Verones et al. (2022) (Coelho et al. 2025). As Scherer et al. (2023) is the most recent and comprehensive recommended method, it will be the focus of this thesis.

1.3 The GLAM framework

The GLAM framework is based on multiple equations and parameters, aiming to capture species survival ability for different land use types and intensity levels in the ecoregions of the world by comparing them to a natural reference state.

(Scherer et al. 2023). Compared to previous methods, the scope of this framework is widened since it considers not only the direct effect of land use on biodiversity, but also the potential of species to migrate between different habitat patches, taking into account the resistance of the surrounding land use type, sizes of habitat patches and the distance required for migration (Figure 1.). In this way, the effects of land fragmentation on biodiversity are considered. The method has enabled an impact assessment where the biodiversity impact for 766 ecoregions (e.g. Sarmatic Mixed Forests), five species groups (plants, birds, amphibians, mammals and reptiles), five land use types (pasture, urban land, cropland, plantations and managed forests) and three land intensities (minimal, light and intense) are accounted for. The impact is expressed as a characterisation factor (CF) with an average or marginal value for land occupation or land transformation on a regional or global level.

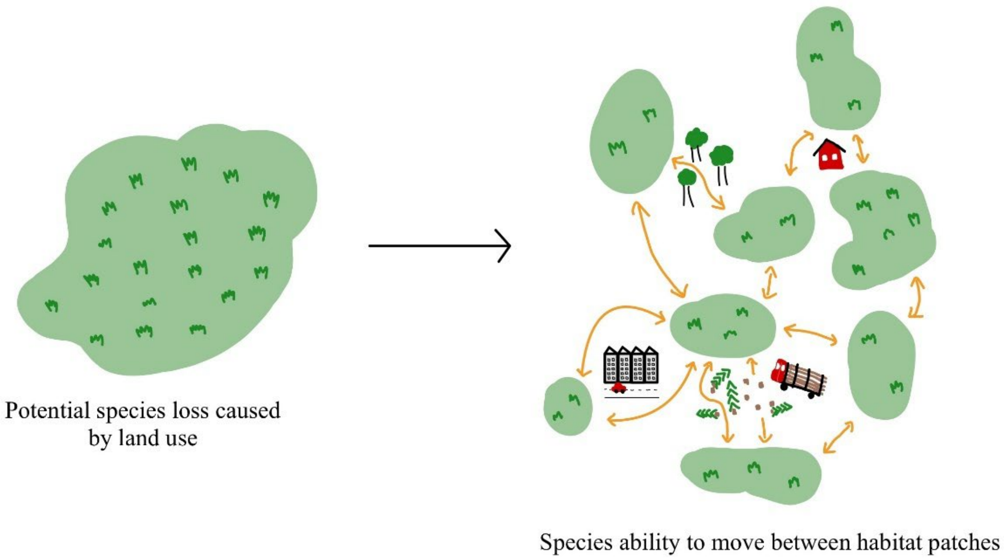

Figure 1. Visualisation of the factors included in the GLAM framework. The scope is widened in comparison to previous methods since it not only includes the potential species loss caused by land use but also by how the fragmentation affects the species ability to move between habitat patches. This includes the sizes of the patches, the resistance of mobility in surrounding landscape and the interpatch distances.

The CF of biodiversity impact is generally defined as the 'potentially disappeared fraction' per square meter of land used (PDF/m^2) indicating the *potential* loss of unique species caused by anthropogenic pressures per unit of land (Verones et al. 2022). The PDF is defined as the difference in species richness between natural and human-managed habitats (Järviö et al. 2025). It is an indicator of the biodiversity impact caused by land use at the endpoint level (Scherer et al. 2023).

They quantify potential relative species loss and indicate the effect of land use on ecosystem quality.

The GLAM method provides average or marginal CFs (Scherer et al. 2023). These two approaches both estimate ‘relative species loss’ (RSL) for the ecoregion *j* and species group *g*. For the average value the total regional land use (A_{lu}) and the allocation factor (a) are estimated directly to reflect the impact of the land use type *i* and intensity level *m*. For the marginal CFs a mathematical derivative is used to incorporate the marginal impact from using more of a certain land type. Both average and marginal CFs are suitable for attributional and consequential LCAs. However, average CFs are more suitable for national footprints while marginal CFs works well for standard product LCAs. Nevertheless, the CFs should not be used for scenario assessments involving significant additional land use (Scherer et al. 2023).

The CFs can be assessed in terms of land occupation (i.e. land use) and land transformation (i.e. land use change) (Kuipers et al. 2021b). Occupational CFs quantify the PDF/m^2 of the land used, while transformational CFs also consider time, specifically the regeneration time after land use has ceased. This is estimated by multiplying the occupational CF by half the regeneration time, aggregating the unit PDF/m^2*year (Scherer et al. 2023). This thesis will only consider occupational (i.e. land use) CFs. To derive the biodiversity damage in a LCA the CF will be multiplied with the inventory (Chaudhary & Brooks 2018). For land use the inventory will be m^2year, which by using the GLAM framework will result in the damage of PDF*year (Chaudhary & Brooks 2018; Scherer et al. 2023).

The CFs can also reflect different scopes by assessing the regional or global impact. Where the regional impact is considered reversible since species can migrate back if the habitat recovers whereas the global species loss is representing the risk for irreversible global extinction (Scherer et al. 2023). The global CF is estimated by multiplying the regional CF by the 'global extinction probability' (GEP). GEP is a scaling approach using species richness, species range sizes and global conservation status to indicate how the regional loss in each area would contribute to species loss globally (Kuipers et al. 2019; Verones et al. 2022).

1.3.1 Semi-natural grasslands

The inability of capturing positive effects of land management becomes relevant when it comes to semi-natural grassland as these grasslands can have higher biodiversity under management than if they would be abandoned and able to revert to ‘natural conditions’. These habitats are maintained through low-intensity management such as grazing or mowing, rather than through conventional high-input agriculture (Eriksson 2022). They are called ‘semi-natural’ since they are not fully natural as they need grazing or mowing to prevent them from succeeding into forests. Even though grasslands are among the most biodiverse ecosystems on earth with significant importance for ecosystem services, the importance is often neglected leading to deteriorated and disappeared habitats (Petermann & Buzhdygan 2021; Glimskär et al. 2023).

The semi-natural grasslands high diversity of plants is partly because grazing hinders competitive species from becoming dominant, which allows many species to coexist (Eriksson 2022). Continuing management is therefore essential to maintain the high biodiversity. Ignoring these positive effects can alter the results of biodiversity impacts within LCA, misrepresent reality and lead to decision-making that further reduces the areaof biodiverse grasslands.

The biodiversity of Sweden's semi-natural grasslands is monitored in multiple ways, providing opportunities to integrate the results into existing LCIA methods. Holsen et al. (in prep) classified the sites into grassland categories based on land use, management, soil type, soil moisture index, and tree cover. This resulted in 16 classes of grassland, divided into ten mainland classes and six classes for Öland and Gotland. The median species richness for each class was calculated using the vascular plant richness data from Remiil (Lundin et al. 2016). This paper will investigate the potential of integrating these sixteen grassland classes of land use types and intensities, along with their corresponding species richness data in the GLAM framework.

To derive the CFs of the grassland class, a reference state is needed. The most frequently used reference is ‘potential natural vegetation landscape’ which defines as the landscape in absence of human intervention (Scherer et al. 2023). Each grassland class was matched to a reference site from the National Meadow and Pasture Inventory (TUVA) at the Swedish Board of Agriculture based on soil type and soil moisture to match underlying characteristics (Holsen et al. 2026). The reference state for grassland, in the absence of grazing or mowing, is a forested landscape.

Holsen et al. (in prep) present an approach to derive local, detailed CFs for different grasslands in Sweden based on data from existing plant monitoring programmes. Their results showed that the CF of most semi-natural grasslands was below zero, indicating a positive impact on ecosystem health. This emphasises the potential of semi-natural grasslands for biodiversity. Gaining a fuller understanding of how these local impacts relate to regional or global biodiversity loss is an essential perspective in LCA, since feed and other inputs often originate from different countries or regions other than where the product is produced. This calls for a more comprehensive methodology in impact assessments. This work explores how this can be achieved within GLAM.

Methods

This study uses an exploratory methodology to investigate the possibility of integrating existing species richness data into the GLAM framework. A literature review of the framework and its underlying studies has been conducted, primarily focusing on Scherer et al. (2023) and the supporting research by Verones et al. (2022), De Baan et al. (2013), and Kuipers et al. (2021a). Review of existing applications of the methods were also employed to gain a better understanding of the potential challenges and opportunities of incorporating local parameters into the global method. As this field of study is rapidly evolving with a lot of recent research on the application of the method, some of the reviewed research has not yet been peer-reviewed, such as Järviö et al. (2025). This was taken into account, resulting in a more critical view of the conclusions.

2.1 Grassland survey data

The data source for this study is vascular plant richness data from Remiil which is a national grassland monitoring program that started in 2009 (Lundin et al. 2016). While Holsen et al. (in prep) previously employed this dataset to estimate local CFs, this study instead handles their underlying species richness data derived from Lundin et al. (2016) (specifically, median species richness per class). This is needed to enable incorporation into the GLAM framework, which requires the raw ecological data rather than already calculated local CFs.

2.2 Framework analysis

The species richness of vascular plants in Swedish grasslands was mapped against the requirements of the GLAM framework to identify data gaps. The mathematical structure of the GLAM framework was systematically deconstructed to trace the flow of the equations, the necessary input data to aggregate CFs was determined and the functional gaps for the Swedish grasslands were identified. These input data were then evaluated to determine the required level of precision. Furthermore, the parameters available through the grassland monitoring programme were connected to the correct equations, while parameters requiring further investigation outside the scope of this study were identified.

2.3 Delimitations

This paper only considers occupational CFs. This is due to the lack of reliable regeneration time data for vascular plants, the species group on which this study will focus (Kuipers et al. 2021). The limitation of exploring the implementation of only vascular plants is due to the high level of available data. There are large-scale monitoring programmes for plants that do not exist for other taxa. While plants do not universally represent biodiversity as a whole, they can serve as an estimated indicator of species richness. (Holsen et al. in prep).

Results & discussion

The GLAM framework is a complex model based on a system of equations (Scherer et al. 2023). This part of the paper will explain the equation flow and consider how local data can be incorporated. The equations require many parameters. The indexes indicate the specifics of the parameters and are clarified in Table 1. The parameters are identified and clarified in Table 2 to help in the understanding the flow of the equations examined later in this paper.

Tabell 1. Clarification of indexes used

INDEX	DESCRIPTION
g	Species group
j	Ecoregion
i	Land use type
m	Intensity level
lu	Total land use within ecoregion
x	Habitat patch
y	Habitat patch
k	Land type separating patches
occ	Occupational
reg	Regional
glo	Global
avg	Average
mar	Marginal

Table 2. The parameters of the GLAM framework with accompanying explanation. Based on the work of Scherer et al. (2023)

PARAMETER	EXPLANATION
$\mathbf{RSL}_{g,\,j,\,reg}$	Relative species loss for a species group in an ecoregion.
$\mathbf{H}_{g,\,j,\,1}$	Suitable connected area for the species group in the ecoregion. The area of functional habitat excluding anthropogenic land use such as cropland or forestry.
$\mathbf{H}_{g,\,j,\,0}$	Suitable connected area for the species group in the ecoregion. Hypothetical state where all land is used as natural reference state. Equivalent to total area of ecoregion.

$\mathbf{z}_{g,\,j}$	Slope of the species-area relationship for biome and species group.
$\mathbf{h}_{g,\,j,\,i,\,m}$	Habitat affinity for the species group with assessed ecoregion, land use type and intensity level, i.e. a species group ability to survive.
$\mathbf{S}_{g,\,i,\,m,\,1}$	Species richness for the species group for assessed land use type and intensity.
$\mathbf{S}_{g,\,i,\,m,\,0}$	Species richness for the species group at reference state for assessed land use type and intensity.
$\mathbf{rr}_{g,\,i,\,m}$	Relative species richness.
$\mathbf{rr}_{g,\,i,\,m,\,k}$	Relative species richness for surrounding land type.
$\mathbf{ECA}_{g,\,j,\,i,\,m,\,1}$	Equivalent Connected Area. Higher value equals more fragmented area.
$\mathbf{A}_{j,\,i,\,m,\,x/y}$	Habitat patch area (x or y) for the land use type and intensity in ecoregion.
$\mathbf{p}_{g,\,j,\,i,\,m,\,x,\,y}$	Probability of dispersal for species group in assessed ecoregion, land use type and intensity between patch x and y.
$\boldsymbol{\alpha}_{g,\,j,\,i}$	Median dispersal distance. Distance that the species group can migrate.
$\mathbf{w}_{g,\,j,\,i,\,m,\,x,\,y}$	Least-cost distance, i.e. the most efficient distance for the species group to migrate between two habitat patches.
$\mathbf{d}_{j,\,i,\,m,\,x}$	Distance between habitat patches.
$\mathbf{r}_{g,\,j,\,i,\,m,\,k}$	Resistance of the surrounding land type separating the habitat patches.
$\mathbf{A}_{j,\,lu}$	Area of total human land use in ecoregion.
$\mathbf{a}_{g,\,j,\,i,\,m}$	Allocation. Equals the proportion of the total share of the biodiversity that should be allocated to the assessed land use type and intensity.
$\mathbf{A}_{j,\,i,\,m}$	Area of the land use type and intensity level in the ecoregion.
GEP	Global extinction probability.

3.1 Regional average CF

The regional occupational average characterisation factor ($CF_{occ, avg, reg}$) is estimated to reflect the direct impact of the land use type *i* and intensity level *m*. The equation flow is visualised in Figure 2. It is based on Equation 1:

Equation 1

$$CF_{g,\,j,\,i,\,m,\,occ,\,avg,\,reg} = RSL_{g,\,j,\,reg} \cdot A_{j,\,lu}^{-1} \cdot a_{g,\,j,\,i,\,m} \tag{1}$$

Where $RSL_{g,j,reg}$ is relative species loss (3.1.1), $A_{j,lu}$ accounts for the total area of land use in the ecoregion (3.1.2), and $a_{g,j,i,m}$ is the allocation factor (3.1.3). The CF is expressed as PDF/m^2 since RSL is expressed as PDF, A as m^2 and *a* is dimensionless. This CF indicates the land use type's impact on biodiversity in the region.

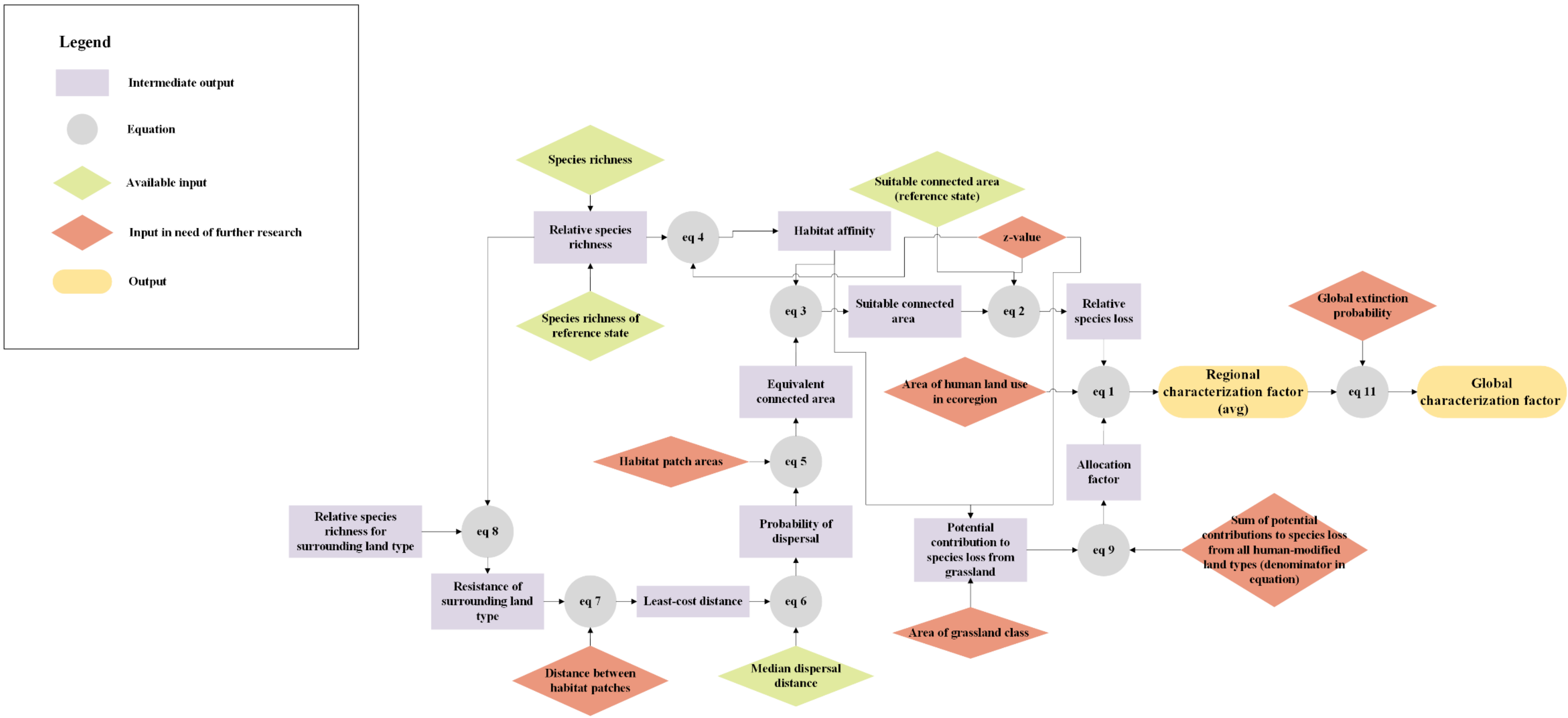

Figure 2. Flowchart of the calculation of average characterization factor. The different shapes and colours are described in the legend in the upper left corner.

3.1.1 Relative species loss

Relative species loss (RSL) measures the total species loss caused by anthropogenic land use in the ecoregion assessed, compared to the natural reference state. This is calculated using Equation 2, which considers the suitable connected area of functional habitat for a species group (g) in an ecoregion (j) ($H_{g,j,1}$, eq 3) and the suitable connected area of the reference state ($H_{g,j,0}$), as well as the slope of the species-area relationship (z) for the species group (g) and ecoregion (j). Z-values for vascular plants for different biomes can be derived from Gerstner et al. (2014). Two biomes are present in Sweden, 'Temperate broadleaf and mixed forests' and 'Boreal forest/taiga'. A spatial mismatch occurs as each grassland class is distributed across these biomes.
Equation 2

$$RSL_{g,j,reg} = 1 - \left(\frac{H_{g,j,1}}{H_{g,j,0}}\right)^{z_{g,j}} \qquad (2)$$

$H_{g,j,1}$ refers to the suitable connected area and the contribution from the total anthropogenic land use. For a given species group (g) and ecoregion (j), the equation (eq 3) summarizes the product of habitat affinity (h) and equivalent connected area (ECA) across all combinations of land use type (i) and intensity level (m). (eq 3). The parameter $H_{g,j,1}$ account for the fact that species viability depends not only on total area but also on the connectivity between patches.

Equation 3

$$H_{g,j,1} = \sum_{i,m} h_{g,j,i,m} \cdot ECA_{g,j,i,m} \qquad (3)$$

Put simply, $H_{g,j,1}$ represents the total suitable habitat available for a given species group (g) in a given ecoregion (j). It combines two things: how much a species group *likes* a particular land use type and intensity (habitat affinity, h), and how much of that land is actually connected and accessible (ECA). By summing this across all land use types and intensity levels, H gives a single number that captures both the quality and the connectivity of the available habitat.

The suitable connected area of the reference state $H_{g,j,0}$ refers to a hypothetical state where all land is used as the natural reference state. It is therefore equivalent to the total area of the ecoregion as the habitat affinity is assumed to be 1 and the landscape to be fully connected. Hence, the ECA for the reference state is equivalent to the area of the ecoregion. When assessing the Swedish grassland classes, the proportion of land in the ecoregions needs to be considered. This could be done by multiplying the areas of the ecoregions with the classes' proportions. However, this would introduce uncertainty, as it would entail estimating how the sampled grasslands in each grassland class are distributed in the landscape. This would require both ecological and statistical verification to ensure that the $H_{g,j,0}$ is reflected in the desired way.

The habitat affinity ($h_{g, j, i, m}$) is a value that describes a species group ability to survive in a certain ecoregion, land use type and intensity level in comparison to the natural reference state in the same ecoregion (Equation 4). It is calculated by dividing the species richness in the assessed land use type and intensity ($S_{g, i, m, 1}$) with the species richness for the natural reference state ($S_{g, i, m, 0}$) which aggregates a relative species richness value (rr). The equation further considers the invert the slope of species-area relationship (z) for the species group (g) and ecoregion (j).

Equation 4

$$h_{g,j,i,m} = \left(\frac{S_{g,i,m,1}}{S_{g,i,m,0}}\right)^{1/z_{g,j}} = rr_{g,i,m}{}^{1/z_{g,j}} \tag{4}$$

Scherer et al. (2023) uses values for rr from Gallego-Zamorano et al. (2022) who merged four databases with pair-wise measured comparisons of plant species richness between specific land-use types and nearby 'natural habitats' (original vegetation with no evidence of prior destruction and minor, if any, disturbance by humans) and performed a meta-analysis using linear mixed-effects models to determine the changes in plant species richness in relation to different land-use types.

For the Swedish grasslands, optimally sixteen different $h_{g, j, i, m}$ would be needed where the relative species richness ($rr_{g, i, m}$) is calculated by dividing the median species richness for the assessed grassland class by its reference state median species richness. Note, that the reference state used for the Swedish grasslands are slightly different, since it represents a 'regeneration state' rather than a 'natural or original state', which are difficult to establish for Sweden.

The equivalent connected area aims to quantify the functional habitat area in a fragmented landscape. There are general values of ECA possible to derive from databases of CFs by Scherer et al. (2023). They are general for the ecoregion. These values are based on Equation 5 by multiplying every habitat area patch ($A_{j, i, m}$) combination (symbolized with x and y) for the ecoregion (j), land use type (i) and intensity level (m) with the probability of dispersal ($p_{g,j,i,m,x,y}$) for the species group between the two habitat patches. This means that the model mathematically covers every single possible combination of patches in the ecoregion to derive the ECA.

Equation 5

$$ECA_{g,\,j,\,i,\,m} = \left(\sum_{x,\,y} A_{j,\,i,\,m,\,x} \cdot A_{j,\,i,\,m,\,y} \cdot p_{g,\,j,\,i,\,m,\,x,\,y} \right)^{0,5} \tag{5}$$

Incorporating local parameters for grassland into the ECA would require a large amount of data. The area of each habitat patch for the sixteen grassland classes is needed. Meaning that it is not enough to know the total area of each grassland class. The precise data of the distribution of the land is necessary to know how well the area is connected. Another challenge is that the ECAs in Scherer et al. (2023) were derived from a relatively coarse land use map (around 10x10 km resolution). To derive ECAs for semi-natural grasslands in Sweden a finer resolution would likely be necessary as the grasslands are often small and scattered, which may pose compatibility problems. Furthermore, more variables of information are required to be able to derive the probability of dispersal. This is explained further below in relation to the underlying equations.

The probability of dispersal (p) is a dimensionless value expressing a species group possibility of moving between the habitat patches of the same land use type and intensity level, in a certain ecoregion. The probability is based on Equation 6, which is dependent on the least-cost distance ($w_{g,\,j\,,i,\,m,\,x,\,y}$) and the median dispersal distance ($\alpha_{g,\,j,\,i}$). The least-cost distance is based on Equation 7, while the median dispersal distance is a value that needs to be determined. Median dispersal distance is defined as the median distance that a species group can move between sites. For the vascular plants in our examples, estimated values derived from existing research can be used (Duan et al. 2024). To obtain more detailed values on how far the plants in the different grasslands can move, it would be necessary to classify which plants are located where.

Equation 6

$$p_{g,\,j,\,i,\,m,\,x,\,y} = e^{-w_{g,\,j,\,i,\,m,\,x,\,y}/\alpha_{g,\,j,\,i}} \tag{6}$$

The least-cost distance ($w_{g,\,j,\,i,\,m,\,x,\,y}$) is determined by the resistance ($r_{g,\,j,\,i,\,m,\,k}$) of the surrounding landscape (k) and the interpatch distance ($d_{j,\,i,\,m,\,x}$) (Equation 7). It aims to find the way between the habitat patches that has the least amount of resistance (cost). The resistance (r) is, in turn, calculated in Equation 8. The interpatch distance for the land use classes and intensities included in Scherer et al. (2023) is already determined and estimated in the GLAM model of Scherer et al. (2023). This means that to derive the global general land use and land intensity CFs that GLAM currently refers to, the information already exists. However, to derive CFs in the grassland example of this paper, a substantial amount of data on the physical distances between each habitat patch and a mapped layer of these patches is needed.

Equation 7

$$w_{g,j,i,m,x,y} = min\left(\sum_k d_{j,i,m,x} \cdot r_{g,j,i,m,k}\right) \tag{7}$$

The landscape resistance ($r_{g,j,i,m,k}$) for plants is a dimensionless value and is estimated by considering the relative resistance of the habitat patches ($rr_{g,i,m}$) and the surrounding landscape ($rr_{g,i,m,k}$) (Equation 8). The model depends on the logic that the landscape resistance is dependent on the species overlap between the habitat patches and the surrounding landscape (k). If species richness is equally high, resistance will be zero and considered non-existent. Resistance measures how difficult it is for species to move between habitat patches. A high resistance value would increase the least-cost distance, lower the probability of dispersal and therefore reduce the ECA of the area.

Equation 8

$$r_{g,i,m,k} = 1 - \frac{min(rr_{g,i,m}, rr_{g,i,m,k})}{rr_{g,i,m}} \tag{8}$$

3.1.2 Area of total land use

The area of total land use ($A_{j,lu}$, m^2) should include the full area of anthropogenic land use in the ecoregion. Hence, the area of cropland, pasture, plantations, managed forests and urban land. The reference states such as untouched forests and areas with no vegetation are excluded. These are values unchanged for the Swedish semi-natural grasslands as they, in the scope of GLAM, are already included as pastures with a minimal intensity. Once again, the fact that the grassland data extends over multiple ecoregions needs to be considered to make it usable in the grassland classes. One option could be to calculate grasslands CFs for Sweden as the geographical boundary.

3.1.3 Allocation

The allocation factor determines how much of the biodiversity loss should be allocated to the different land use types and intensity levels in an ecoregion. For example, how much of the PDF in the ecoregion is caused by pasture on a minimal intensity level. This is calculated by using Equation 9. The numerator considers the habitat affinity (h) of the assessed land use type (i), managed with intensity (m) in ecoregion (j) and the z-value for species group (g) in ecoregion (j) and the total area of the corresponding land use ($A_{j, i, m}$). The denominator sums the corresponding products of (1-h) and total land of all human modified land use types in the ecoregion. The allocation is a critical factor for the average CF, since it determines the level of impact accounted for the land use type and intensity.

In Equation 9, the remaining parameter to cover for Swedish grasslands is the total area ($A_{j, i, m}$). If spatial data for all individual habitat patches were available, the total area for each grassland class could be calculated simply by summing the areas of the patches belonging to that specific class.

Equation 9

$$a_{g, j, i, m} = \frac{\left(1 - h_{g, j, i, m}{}^{z_{g, j}}\right) \cdot A_{j, i, m}}{\sum_{i, m}\left(1 - h_{g, j, i, m}{}^{z_{g, j}}\right) \cdot A_{j, i, m}} \tag{9}$$

3.2 Regional marginal CF

The regional occupational marginal characterisation factor ($CF_{occ, mar, reg}$) estimated by the mathematical derivate of Equation 1 (Equation 10). The marginal CF gives the impact caused by the use of one additional unit of land converted in contrast to the while the average CF calculates the total number of species lost after conversion. As such, the marginal value is more suitable for product LCAs and considers the species loss if more of the land use would occur.

The parameters used in the equation for marginal CF are also included in the average value. Therefore, if the parameters required for the average CF have been calculated for Swedish grasslands, it would be no more complicated than fitting the same parameters to Equation 10. The equation flow is visualised in Figure 3.

Equation 10

$$CF_{g,\,j,\,i,\,m,\,occ,\,mar,\,reg} = \left(H^{-1}_{g,\,j,\,0} \cdot z_{g,\,j} \cdot \left(\frac{H_{g,\,j,\,1}}{H_{g,\,j,\,0}}\right)\right)^{z_{g,\,j}-1} \cdot \frac{H_{g,\,j,\,0} - H_{g,\,j,\,1}}{A_{j,\,lu}} \cdot a_{g,\,j,\,i,\,m} \qquad (10)$$

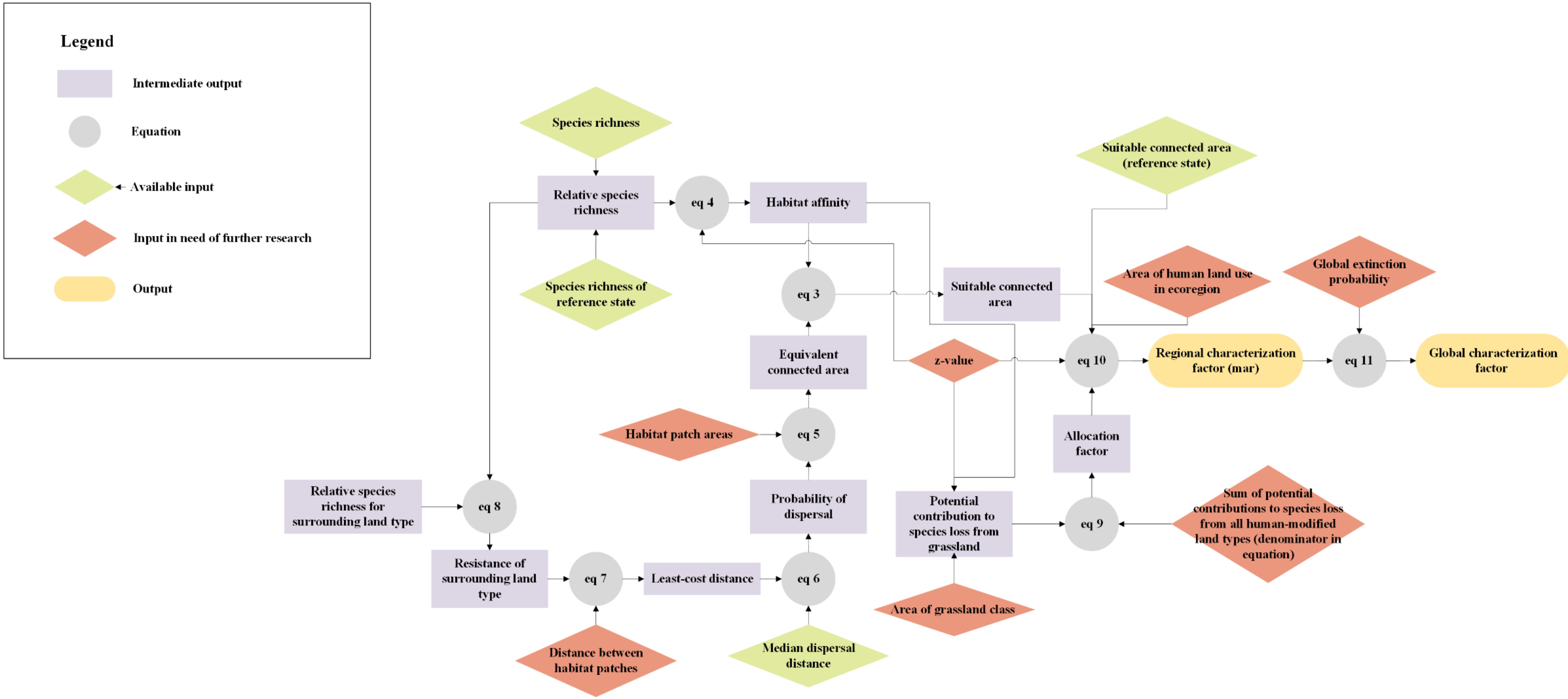

Figure 3. Flowchart of the calculation of marginal characterization factor. The different shapes and colours are described in the legend in the upper left corner.

3.3 Global CF

The global CF converts the regional CFs into a value that relates how the regional species richness loss will affect the risk of global species richness loss. In other words, it converts regional reversible loss to global irreversible permanent loss of species (extinction). The global CF is calculated by using Equation 11 regardless of if the intermediate aim involves the average or marginal CF. The regional CF is in both cases multiplied with the 'global extinction probability' (GEP). The GEP is based on an approach presented by Kuipers et al. (2019) that considers the range area of species, the occurrence weight value and the threat level weight value. For the vascular plants in Swedish grasslands this can be simplified by deriving the GEPs from Verones et al. (2022) as suggested by Scherer et al. (2023). The GEP values are representative for ecoregions and not country-based, highlighting the challenge of deriving country-based CFs in an ecoregional framework.

Equation 11

$$CF_{g,\,j,\,i,\,m,\,glo} = CF_{g,\,j,\,i,\,m,\,reg} \cdot GEP_{g,\,j} \tag{11}$$

3.4 Data gaps for Swedish grasslands

Even though there are reliable monitoring programmes for species richness in Swedish grasslands, it becomes clear that this data does not cover the full framework to derive CFs based on local ecological conditions through GLAM. The local species data relates only to the species richness while GLAM requires data that describes the land distribution of the grassland classes.

GLAM is based on a large number of equations, with parameters built upon one another. This is visualised in Figure 2 and Figure 3, while Table 3 reflects the explanations and equations needed for each parameter, as well as the required input data. For input data on parameters that can be kept general, it is possible to use values from existing research papers. This is possible for the z-value and the GEP, since these values are general to the ecoregion rather than specific to grassland classes. These values must be considered in such a way that the ecoregional approach can be adapted to be compatible with Swedish grasslands located in multiple ecoregions.

Table 3. The parameters of the GLAM framework with explanation connected to the example of Swedish grasslands. The table examines where each parameter is derived from. The blue colour in the left-hand column indicates that this is input data. The column on the far-right shows if it is built on an equation (grey) and whether the input

data is available within the scope of this study (green colour) or requires further investigation (red colour).

PARAMETER	EXPLANATION	DERIVED FROM	
$CF_{g, j, i, m, occ, avg, reg}$	Regional characterization factor for occupation with an average value.	Equation 1.	
$CF_{g, j, i, m, occ, avg, glo}$	Global characterization factor for occupation with an average value.	Equation 11	
$CF_{g, j, i, m, occ, mar, reg}$	Regional characterization factor for occupation with a marginal value.	Equation 10	
$CF_{g, j, i, m, occ, mar, glo}$	Global characterization factor for occupation with a marginal value.	Equation 11	
$RSL_{g, j, reg}$	Relative species loss for a species group in an ecoregion.	Equation 2.	
$H_{g, j, 1}$	Suitable connected area for vascular plants in the ecoregion.	Equation 3.	
$H_{g, j, 0}$	Suitable connected area vascular plants in the ecoregion, reference state.	Equivalent to total area of ecoregion. Proportion of land in each ecoregion needs to be considered for the grasslands, or CFs at national level calculated.	
$z_{g, j}$	Slope of the species-area relationship for vascular plants in ecoregion.	Ecoregional values need to be converted to be compatible with the grassland's distribution.	
$h_{g, j, i, m}$	Habitat affinity for vascular plants in each grassland class	Equation 4.	
$S_{g, i, m, 1}$	Species richness of vascular plants for each grassland class	Data needed for species group.	
$S_{g, i, m, 0}$	Species richness of vascular plants at reference state for each grassland class	Data needed for species group at reference state.	
$rr_{g, i, m}$	Relative species richness.	Equation 4.	
$rr_{g, i, m, k}$	Relative species richness for surrounding land type.	Equation 4 but for species richness data for the surrounding land type (k).	
$ECA_{g, j, i, m, 1}$	Equivalent Connected Area.	Calculated using Equation 5.	
$A_{j, i, m, x/y}$	Habitat patch area (x or y) for the grassland class in ecoregion.	Value needed. Can probably be derived from the Swedish Land Parcel Identification System in combination with	

		the Swedish grassland monitoring programme.	
$p_{g, j, i, m, x, y}$	Probability of dispersal for vascular plants between patch x and y of each grassland class.	Equation 6	
$\alpha_{g, j, i}$	Median dispersal distance. Distance that vascular plants can migrate.	Either value derived from existing research or if possible own data.	
$w_{g, j, i, m, x, y}$	Least-cost distance.	Equation 7	
$d_{j, i, m, x}$	Distance between habitat patches.	Value needed. Can probably be derived from the Swedish Land Parcel Identification System in combination with the Swedish grassland monitoring programme.	
$r_{g, j, i, m, k}$	Resistance of the surrounding land type separating the habitat patches.	Equation 8	
$A_{j, lu}$	Area of total human land use in ecoregion (or Sweden).	Value needed.	
$a_{g, j, i, m}$	Allocation. Equals the proportion of the total share of the biodiversity should be allocated to the grassland class.	Equation 9.	
$A_{j, i, m}$	Area of the grassland class in the ecoregion (or Sweden).	Value needed. Sum of habitat patch areas.	
GEP	Global extinction probability.	Ecoregional values need to be converted to be compatible with the grassland's distribution.	

The bridge between local monitoring and GLAM is currently limited by the lack of readily available data connected to spatial factors. This applies the total area of each grassland class, along with precise information on each habitat patch and the distances between them. These data could likely be derived from the Swedish Land Parcel Information System, which contains the spatial extent of each grassland managed with agri-environmental support in combination with information from the Swedish grassland monitoring programme. However, questions remain regarding compatibility issues with rest of the GLAM framework, as map layers of higher resolution than the one used by Scherer et al. (2023) would likely be needed to calculate ECA for the different grassland classes. The main challenge of the integration is that, although the level of detail must be increased, compatibility with GLAM must be maintained. This requires a comprehensive mapping of land use that accurately depicts the distribution of the classes. Simply estimating how much land each class occupies in Sweden is not enough.

The allocation factor determines the proportion of total species loss that can be attributed to the assessed loss. A positive biodiversity impact, like for several of the grasslands, would yield a negative allocation factor since the grasslands demonstrate an increase in species richness compared to the reference state. Although, negative allocation factors are not used in GLAM due to high uncertainty, it would be possible in theory according to the framework. There is also uncertainty surrounding the consequences of using reference states that differ from those in the GLAM framework. This could cause inconsistencies in the baseline, which would make the results unreliable. Specific reference states would be needed to maintain local detail, but there is a risk that this detail would be lost in the broader general factors also included in the framework.

Overarching discussion

The species richness data available for Swedish grasslands is not enough to derive characterisation factors through the GLAM framework as spatial data also is required. Even though it would be possible to incorporate local species data into GLAM and thereby calculate site-specific CFs it is questionable if incorporating detailed local data on species richness is meaningful in a global framework build on modelling of impacts. Since GLAM is a model with the aim to capture biodiversity damage at the global level with complete coverage, and therefore requiring substantial simplifications, it is important to question whether it becomes mathematically and conceptually meaningful to characterize local (and positive) biodiversity impacts. This emphasises the need for further research into the assessment of local and positive impacts through LCIA frameworks.

Another potential barrier in implementing local species richness data for Swedish grasslands within the GLAM framework is the scope of the monitoring. GLAMs regional impacts are fundamentally based on ecoregions, while the data considered in this study is focused on Sweden. Sweden contains multiple ecoregions, none of which it fully covers. This means that even though there are comprehensive data within Sweden, this is put in a framework assessing ecoregions which makes the data insufficient in covering grasslands in the ecoregion. However, to exclusively assess Sweden's grasslands multiple ecoregions must be included. This highlights the difference between monitoring and modelling, where monitoring programmes usually takes place by country due to fundings while modelling is more easily adapted to ecoregion. One option would be to calculate CFs for the grasslands using Sweden as the geographical boundary.

The GLAM framework is not designed to incorporate local values straight into the modelling framework as the rr values build on results from a meta-analysis of empirical species richness data. It is a general, global and complex framework that has developed the assessments of biodiversity, with many positive functions by including additional important parameters. The GLAM framework favours the implementation of modelled species richness over empirical data incorporation, as evidenced by the incompatibilities experienced during this process. The resolution of local data also risks being lost in general parameters, such as GEP. The result will not be more exact than the weakest link, meaning that even though detailed values are incorporated, the CFs risks to be as uncertain as the GEP is general. Future research will require these to be connected in a way that simplifies usage. Where local species richness data is available, like for the Swedish grasslands, it might be favourable to use local CFs, i.e. the rr directly, in LCA like in for example Trydeman – Knutsen et al. (2017).

The current version of the GLAM framework incorporates land use intensity but fails to capture the specific ecological effects of different management practices (Coelho et al., 2025). Although the framework distinguishes between minimal, light, and intense land use intensity, it neglects the differences in ecological outcomes caused by variations in management practices. As a result, the framework overlooks land use practices with positive potential, such as those relating to semi-natural grasslands.

Conclusion

This study provides further insights into the possibilities and challenges of incorporating local species richness data for Swedish semi-natural grasslands in the recommended global framework GLAM for biodiversity impact assessment. It was found that while species richness data are available, it is insufficient to derive regional or global CFs, as the specific spatial requirements are not met. This paper proposes exploratory methods on how to bridge the current data gaps. Although, it is also a question whether the detailed ecological data will be accurately reflected in a global model as it is built to assess ecoregional biodiversity damage rather than country-based site-specific biodiversity gain.

References

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

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

Coelho, C.R.V., Börjesson, P. & Smith, H.G. (2025). Understanding land use impacts of croplands on biodiversity through UNEP's Global Guidance for Life Cycle Impact Assessment. *Resources, Conservation and Recycling*, 222, 108420. https://doi.org/10.1016/j.resconrec.2025.108420

Conor, E.F. & McCoy, E.D. (2013). Species–Area Relationships. I: *Encyclopedia of Biodiversity*. Elsevier. 640–650. https://doi.org/10.1016/B978-0-12-384719-5.00132-5

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

Damiani, M., Sinkko, T., Caldeira, C., Tosches, D., Robuchon, M. & Sala, S. (2023). Critical review of methods and models for biodiversity impact assessment and their applicability in the LCA context. *Environmental Impact Assessment Review*, 101, 107134. https://doi.org/10.1016/j.eiar.2023.107134

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

Duan, J., Yang, L., Tang, T., Rao, J., Liu, W., Chen, X., Li, R. & Shen, Z. (2024). Environment and management jointly shape the spatial patterns of plant species diversity of moist grasslands in the mountains of northeastern Yunnan. *Plant Diversity*, 46 (6), 744–754. https://doi.org/10.1016/j.pld.2024.04.005

Eriksson, O. (2022). Coproduction of Food, Cultural Heritage and Biodiversity by Livestock Grazing in Swedish Semi-natural Grasslands. *Frontiers in Sustainable Food Systems*, 6, 801327. https://doi.org/10.3389/fsufs.2022.801327

Gallego-Zamorano, J., Huijbregts, M. A. J., & Schipper, A. M. (2022). Changes in plant species richness due to land use and nitrogen deposition across the globe. Diversity and Distributions, 28, 745–755. https://doi.org/10.1111/ddi.13476

Glimskär, A., Hultgren, J., Hiron, M., Westin, R., Bokkers, E.A.M. & Keeling, L.J. (2023). Sustainable Grazing by Cattle and Sheep for Semi-Natural Grasslands in Sweden. *Agronomy*, 13 (10), 2469. https://doi.org/10.3390/agronomy13102469

Holsen, L., Glimskär, A., Olsson, E., von Brömssen, C., Röös, E. & Karlsson, J.O. (in prep). Development of characterization factors of Swedish grassland vascular-plant biodiversity for application in life cycle assessment. Department of Energy and Technology, SLU.

Järviö, N., Uusitalo, V. & Ekroos, J. (2025). Enhancing Life Cycle Biodiversity Impact Assessment Methods across Farming Systems through

Environmental DNA Analysis. In Review. https://doi.org/10.21203/rs.3.rs-6032493/v1
Knudsen, M. T., Hermansen, J. E., Kainz, M., Fjellstad, W., Wolfrum, S., Sarthou, J.-P., Cederberg, C., Herzog, F., Dennis, P., Balázs, K., Vale, J., & Jeanneret, P. (2017). Characterization factors for land use impacts on biodiversity in life cycle assessment based on direct measures of plant species richness in European farmland in the "Temperate Broadleaf and Mixed Forest" biome. Science of the Total Environment, 580, 358–366. https://doi.org/10.1016/j.scitotenv.2016.11.172
Kuipers, K.J.J., Hellweg, S. & Verones, F. (2019). Potential Consequences of Regional Species Loss for Global Species Richness: A Quantitative Approach for Estimating Global Extinction Probabilities. *Environmental Science & Technology*, 53 (9), 4728–4738. https://doi.org/10.1021/acs.est.8b06173
Kuipers, K.J.J., Hilbers, J.P., Garcia-Ulloa, J., Graae, B.J., May, R., Verones, F., Huijbregts, M.A.J. & Schipper, A.M. (2021a). Habitat fragmentation amplifies threats from habitat loss to mammal diversity across the world's terrestrial ecoregions. *One Earth*, 4 (10), 1505–1513. https://doi.org/10.1016/j.oneear.2021.09.005
Kuipers, K.J.J., May, R. & Verones, F. (2021b). Considering habitat conversion and fragmentation in characterisation factors for land-use impacts on vertebrate species richness. *Science of The Total Environment*, 801, 149737. https://doi.org/10.1016/j.scitotenv.2021.149737
Lundin, A., Kindström, M., Glimskär, A., Gunnarsson, U., Hedenbo, P. & Rygne, H. (2016). *Metodik för regional miljöövervakning av gräsmarker och våtmarker 2015-2020*. Department of Ecology, SLU. https://res.slu.se/id/publ/80655
Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Börger, L., Bennett, D.J., Choimes, A., Collen, B., Day, J., De Palma, A., Díaz, S., Echeverria-Londoño, S., Edgar, M.J., Feldman, A., Garon, M., Harrison, M.L.K., Alhusseini, T., Ingram, D.J., Itescu, Y., Kattge, J., Kemp, V., Kirkpatrick, L., Kleyer, M., Correia, D.L.P., Martin, C.D., Meiri, S., Novosolov, M., Pan, Y., Phillips, H.R.P., Purves, D.W., Robinson, A., Simpson, J., Tuck, S.L., Weiher, E., White, H.J., Ewers, R.M., Mace, G.M., Scharlemann, J.P.W. & Purvis, A. (2015). Global effects of land use on local terrestrial biodiversity. *Nature*, 520 (7545), 45–50. https://doi.org/10.1038/nature14324
Petermann, J.S. & Buzhdygan, O.Y. (2021). Grassland biodiversity. *Current Biology*, 31 (19), R1195–R1201. https://doi.org/10.1016/j.cub.2021.06.060
Prangel, E., Reitalu, T., Neuenkamp, L., Kasari-Toussaint, L., Karise, R., Tiitsaar, A., Soon, V., Kupper, T., Meriste, M., Ingerpuu, N. & Helm, A. (2024). Restoration of semi-natural grasslands boosts biodiversity and re-creates hotspots for ecosystem services. *Agriculture, Ecosystems & Environment*, 374, 109139. https://doi.org/10.1016/j.agee.2024.109139
Scherer, L., Rosa, F., Sun, Z., Michelsen, O., De Laurentiis, V., Marques, A., Pfister, S., Verones, F. & Kuipers, K.J.J. (2023). Biodiversity Impact Assessment Considering Land Use Intensities and Fragmentation. *Environmental Science & Technology*, 57 (48), 19612–19623. https://doi.org/10.1021/acs.est.3c04191
Verones, F., Kuipers, K., Núñez, M., Rosa, F., Scherer, L., Marques, A., Michelsen, O., Barbarossa, V., Jaffe, B., Pfister, S. & Dorber, M. (2022). Global extinction probabilities of terrestrial, freshwater, and marine

species groups for use in Life Cycle Assessment. *Ecological Indicators*, 142, 109204. https://doi.org/10.1016/j.ecolind.2022.109204

Winter, L., Lehmann, A., Finogenova, N. & Finkbeiner, M. (2017). Including biodiversity in life cycle assessment – State of the art, gaps and research needs. *Environmental Impact Assessment Review*, 67, 88–100. https://doi.org/10.1016/j.eiar.2017.08.006

Zhen, H., Goglio, P., Hashemi, F., Cederberg, C., Fossey, M. & Trydeman Knudsen, M. (2025). Toward Better Biodiversity Impact Assessment of Agricultural Land Management through Life Cycle Assessment: A Systematic Review. *Environmental Science & Technology*, 59 (15), 7440–7451. https://doi.org/10.1021/acs.est.5c02000

Publishing and archiving

Approved students' theses at SLU can be published online. As a student you own the copyright to your work and in such cases, you need to approve the publication. In connection with your approval of publication, SLU will process your personal data (name) to make the work searchable on the internet. You can revoke your consent at any time by contacting the library.

Even if you choose not to publish the work or if you revoke your approval, the thesis will be archived digitally according to archive legislation.

You will find links to SLU's publication agreement and SLU's processing of personal data and your rights on this page:

- https://libanswers.slu.se/en/faq/228318

☒ YES, I, Tilda Lagerhäll, have read and agree to the agreement for publication and the personal data processing that takes place in connection with this.

☐ NO, I/we do not give my/our permission to publish the full text of this work. However, the work will be uploaded for archiving and the metadata and summary will be visible and searchable.