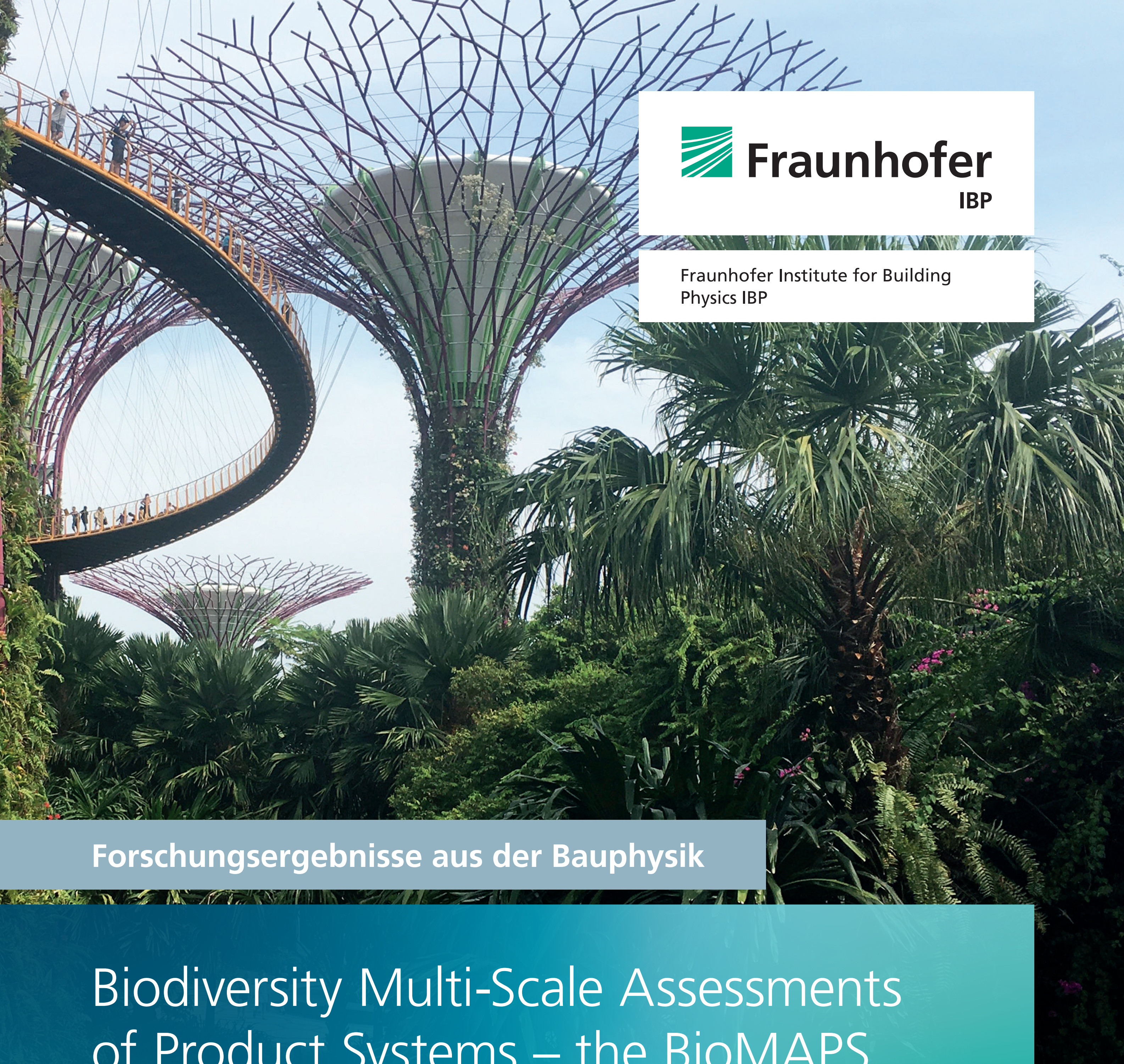

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Biodiversity Multi-Scale Assessments of Product Systems – the BioMAPS Method

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University of Stuttgart
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Biodiversity Multi-Scale Assessments of Product Systems – the BioMAPS Method

Von der Fakultät Bau- und Umweltingenieurwissenschaften der Universität Stuttgart
zur Erlangung der Würde einer Doktor-Ingenieurin (Dr.-Ing.)
genehmigte Abhandlung

Vorgelegt von

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Mitberichter: **Prof. Dr.-Ing. Jan Paul Lindner**

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Declaration

I hereby affirm that the present work was made by myself, using only the listed literature and without any help from others.

Stephanie Maier Stuttgart, den

"You, me and the rest of the human species are critically dependent on the health of the natural world. If the seas stop producing oxygen, we would be unable to breathe, and there is no food that we can digest that doesn't originate from the natural world. – If we damage the natural world, we damage ourselves."

Sir David Attenborough

Executive summary

Land use and land use changes belong to the major drivers of the continuing biodiversity loss. Life Cycle Assessment (LCA) is a well-established tool for measuring such impacts that arise during the life cycle of products and services. Although there are valuable methods for biodiversity impact assessment in LCA, as part of the Life Cycle Impact Assessment (LCIA), they are still rarely used by companies and municipalities. This is mainly due to the fact that existing methods are not globally applicable or do not provide sufficient decision support for LCA end users. Therefore, this dissertation deals with the development of a new, globally applicable method for the analysis of biodiversity impacts in LCA. The new **bio**diversity **m**ulti-scale **a**ssessment of **p**roduct **s**ystems (BioMAPS) method is based on key requirements from an ecological and nature conservation point of view as well as on technical requirements of LCA.

The methodological framework of BioMAPS is structured as a modular procedure in which the impacts can be assessed at global, regional and local scales for different organizational levels of biodiversity (namely, ecosystems, species and genes). All analytical steps can be evaluated independently, using different databases, models, concepts and maps. This flexible structure ensures customer-tailored and solution-driven suggestions for decision makers at three spatial scales.

With regard to the global scale, valuable research results from ecology and nature conservation are used to determine the global distribution of biodiversity and its risks. These sciences analyze and prioritize areas of global importance for the conservation of biodiversity. In general, three different prioritization concepts can be identified: reactive, proactive concepts and approaches that include areas with a high degree of irreplaceability. These three conservation concepts make use of different indicators and criteria, which leads to a spatial discrepancy between the risk areas. Therefore, a combination of the different schemes is used for the BioMAPS method to identify areas of global importance for biodiversity. Also, a unified biodiversity risk map is developed in the course of this thesis. The unified biodiversity risk map is used to assess impacts related to resources and products. Based on this map, the different locations of the land use types are analyzed. Within a country or region, the probability that land use falls within an area worthy of protection for biodiversity is calculated. The calculation is carried out for the different land use types such as arable farming, pastures, plantations, forestry or urban areas using global land use models.

Concerning the local scale, existing ecological models and databases are used and integrated into the method to quantify the impact of certain land use types on biodiversity in a given field or piece of land. These models are used to determine an average impact for each land use type. For example, forestry has, on average, a lower impact on biodiversity than arable farming. However, the impact on local biodiversity does not only depend on the type of land use, but is also significantly influenced by the intensity of land use. For example, intensive forest management with monocultures and even-aged trees can be more detrimental to biodiversity than extensively managed arable land with flowering strips. Thus, this method does not only account for an average value of each land use type, but also for an impact interval. This interval is determined by the intensity of land use and specific land management parameters. Depending on the type of land management and thus the intensity of land use, the effects on biodiversity are in the upper, middle or lower range of the interval. Therefore, for each type of land use a land use intensity index (LUI) is calculated, which is composed of management parameters that have a scientifically proven influence on biodiversity. These management parameters are determined from nature conservation databases. The LUIs are used to quantify the value within the interval in which the impact on biodiversity is found.

Finally, the results of local biodiversity risks (from the LUIs) are scaled up to a larger landscape context to assess regional impacts. This scale does not only include biodiversity risks on a field or individual land area but also all land use types that shape the landscape as a whole. This is particularly important to highlight landscapes with a higher proportion of native primary or secondary habitats or with a higher proportion of extensively farmed land. In nature conservation science a landscape matrix with extensively managed areas is known as "land sharing" while a matrix with a high proportion of primary and secondary habitats as "land sparing". Even though there has been an ongoing debate on which of these approaches is more suitable for conservation, there are many scientists that advocate for a combination of both approaches. Therefore, both approaches are integrated into the method by calculating biodiversity risks, which are derived from a landscape development index (LDI). The LDI contains the shares of the individual land use types in the overall landscape composition, as well as their land use intensities from the LUIs. In this step, a biodiversity risk value is determined for the entire landscape matrix.

In addition, background data for all scales are calculated using a GIS environment to operationalize the BioMAPS method for entire product systems across global value chains. For this purpose, global statistical data on biodiversity risk areas, land use and land management are combined with ecological models. Thus, information on the impact of land use on biodiversity at both coordinate and country level for all scales is obtained. In doing so, global land use intensities for different types of land use are calculated based on statistical data. The site-specific conditions of individual parameters, such as climate and soil properties, are taken into account via global agro-ecological zones. The global data sets can be used as background data for the supply chain. If a company has primary data on individual farming parameters, for example if it knows how much fertilizer is used for its own fields, this data is preferred and used in a foreground process. If the company can only provide the country of origin of a product, the background database will be used instead. Thus, the more precisely a company knows the production location, the more precisely the effects can be determined.

The BioMAPS method is then applied to a case study that assesses the biodiversity impact of different transport energy carriers for a passenger car. These include diesel and gasoline from fossil fuels, electricity from renewable (solar, wind) and non-renewable resources, methane from photovoltaic electricity, biodiesel from palm oil, bioethanol from sugar cane, bio-methane from corn, and methane from wood. This assessment uses both foreground and background data sets. In general, it can be shown that the area of land use is a decisive factor for the overall impact on biodiversity. Therefore, biofuel energy sources have a higher biodiversity impact compared to non-biofuel energy sources (including the electricity grid mix). Here, however, electrical energy from renewable sources such as wind or solar energy in particular have the least biodiversity impact. The highest impact is measured with biofuel systems, biogas from wood production, followed by bioethanol from sugar cane. Bioethanol from sugar cane and biodiesel from palm oil have the most severe impact on biodiversity at a global level, as the production facilities are located in areas particularly worthy of protection.

The BioMAPS method presented in this book is the first globally applicable impact assessment method that takes into account the multi-scale concept of biodiversity by building a bridge between the disciplines of ecology, nature conservation and life cycle assessment. It uses and harmonises the wide range of research results available, thus providing LCA end-users with a coherent framework for assessing the biodiversity impacts of product systems. As a result, this methodology aims to mitigate negative impacts by identifying concrete actions at the global, regional and local scale.

Kurzfassung

Landnutzung und Landnutzungsänderungen gehören zu den Hauptursachen für den anhaltenden Verlust an biologischer Vielfalt. Die Ökobilanz (Life Cycle Assessment, LCA) ist ein bewährtes Instrument zur Messung solcher Auswirkungen, die während des Lebenszyklus von Produkten und Dienstleistungen entstehen. Obwohl es im Rahmen der Folgenabschätzung (Life Cycle Impact Assessment, LCIA) bereits wertvolle Methoden zur Bewertung der Auswirkungen auf die biologische Vielfalt gibt, werden sie von Unternehmen und Kommunen noch immer selten eingesetzt. Dies liegt vor allem daran, dass die vorhandenen Methoden nicht global anwendbar sind oder keine ausreichende Entscheidungshilfe für die Endnutzer von Ökobilanzen bieten. Die vorliegende Dissertation befasst sich daher mit der Entwicklung einer neuen, global anwendbaren Methode zur Analyse von Biodiversitätsauswirkungen in Ökobilanzen. Die neue Methode für eine **Bio**diversitäts-**M**ultiskalen-**A**nalyse von **P**rodukt-**S**ystemen (BioMAPS) basiert auf zentralen Anforderungen aus ökologischer und naturschutzfachlicher Sicht sowie auf fachlichen Anforderungen der Ökobilanz.

Der methodische Rahmen von BioMAPS ist als modulares Verfahren aufgebaut, bei dem die Auswirkungen auf globaler, regionaler und lokaler Ebene für verschiedene Organisationseinheiten der Biodiversität (nämlich Ökosysteme, Arten und Gene) bewertet werden können. Alle Analyseschritte können unabhängig voneinander ausgewertet werden, wobei unterschiedliche Datenbanken, Modelle, Konzepte und Karten verwendet werden. Diese flexible Struktur gewährleistet kundenspezifische und lösungsorientierte Vorschläge für Entscheidungsträger auf drei räumlichen Ebenen.

Auf globaler Ebene werden wertvolle Forschungsergebnisse aus der Ökologie und dem Naturschutz genutzt, um die globale Verteilung der biologischen Vielfalt und deren Risiken zu bestimmen. Diese Disziplinen analysieren und priorisieren Gebiete von globaler Bedeutung für die Erhaltung der biologischen Vielfalt. Im Allgemeinen lassen sich drei verschiedene Priorisierungskonzepte unterscheiden: reaktive, proaktive und Ansätze, die Gebiete mit einem hohen Grad an Unersetzlichkeit einbeziehen. Bei diesen drei Schutzkonzepten werden unterschiedliche Indikatoren und Kriterien verwendet, was bisher zu einer räumlichen Diskrepanz zwischen den Risikogebieten führt. Daher wird bei der BioMAPS-Methode eine Kombination aus den verschiedenen Konzepten verwendet, um Gebiete von globaler Bedeutung für die biologische Vielfalt zu ermitteln. Außerdem wird im Rahmen dieser Arbeit eine einheitliche Risikokarte für die biologische Vielfalt entwickelt. Die einheitliche Biodiversitätsrisikokarte wird zur Bewertung der Herkunft von Ressourcen und Produkten verwendet. Auf der Grundlage dieser Karte werden die Standorte der verschiedenen

Landnutzungstypen analysiert. Innerhalb eines Landes oder einer Region wird die Wahrscheinlichkeit berechnet, dass eine Landnutzung in ein schützenswertes Gebiet für die biologische Vielfalt fällt. Die Berechnung erfolgt für die verschiedenen Landnutzungstypen wie Ackerbau, Weiden, Plantagen, Forstwirtschaft oder städtische Gebiete anhand globaler Landnutzungsmodelle.

Auf lokaler Ebene werden bestehende ökologische Modelle und Datenbanken verwendet und in die Methode integriert, um die Auswirkungen bestimmter Landnutzungsarten auf die biologische Vielfalt auf einem bestimmten Feld oder einer Landfläche zu quantifizieren. Diese Modelle werden verwendet, um eine durchschnittliche Auswirkung für jede Landnutzungsart zu bestimmen. So hat beispielsweise die Forstwirtschaft im Durchschnitt geringere Auswirkungen auf die biologische Vielfalt als der Ackerbau. Die Auswirkungen auf die lokale biologische Vielfalt hängen jedoch nicht nur von der Art der Landnutzung ab, sondern werden auch maßgeblich von der Intensität der Landnutzung beeinflusst. So kann sich beispielsweise eine intensive Waldbewirtschaftung mit Monokulturen und gleichmäßig alten Bäumen nachteiliger auf die biologische Vielfalt auswirken als extensiv bewirtschaftete Ackerflächen mit großen Flächen an Blühstreifen. Daher wird bei dieser Methode nicht nur ein Durchschnittswert für jede Landnutzungsart berücksichtigt, sondern auch ein Wirkungsintervall. Dieses Intervall wird durch die Intensität der Landnutzung und spezifische Parameter der Landbewirtschaftung bestimmt. Je nach Art der Landbewirtschaftung und damit der Intensität der Landnutzung liegen die Auswirkungen auf die biologische Vielfalt im oberen, mittleren oder unteren Bereich des Intervalls. Daher wird für jede Landnutzungsart ein Landnutzungsintensitätsindex (LUI) berechnet, der sich aus Bewirtschaftungsparametern zusammensetzt, die einen wissenschaftlich nachgewiesenen Einfluss auf die Biodiversität haben. Diese Bewirtschaftungsparameter werden aus Naturschutzdatenbanken ermittelt. Die LUIs werden verwendet, um den Wert innerhalb des Intervalls zu quantifizieren, in dem die Auswirkungen auf die biologische Vielfalt zu finden sind.

Abschließend werden die Ergebnisse der lokalen Biodiversitätsrisiken (aus den LUI) auf einen größeren Landschaftskontext hochgerechnet, um die regionalen Auswirkungen zu bewerten. Diese Ebene umfasst nicht nur die Biodiversitätsrisiken auf einem Feld oder einer einzelnen Fläche, sondern auch alle Landnutzungsarten, die die Landschaft als Ganzes prägen. Dies ist besonders wichtig, um Landschaften mit einem höheren Anteil an einheimischen primären oder sekundären Lebensräumen oder mit einem höheren Anteil an extensiv bewirtschafteten Flächen hervorzuheben. In der Naturschutzwissenschaft wird eine Landschaftsmatrix mit extensiv bewirtschafteten Flächen als "land sharing" bezeichnet, während eine Matrix mit einem hohen Anteil an primären und sekundären Lebensräumen als "land sparing" bezeichnet wird. Auch wenn es eine anhaltende Debatte darüber gibt, welcher dieser Ansätze für den Naturschutz besser geeignet ist, gibt es viele Wissenschaftler, die sich für eine Kombination beider Ansätze aussprechen. Daher werden beide Ansätze in die Methode integriert, indem Risiken für die biologische Vielfalt berechnet werden, die aus einem Landschaftsentwicklungsindex (LDI) abgeleitet werden. Der LDI enthält die Anteile der einzelnen Landnutzungstypen an der gesamten Landschaftszusammensetzung sowie deren Landnutzungsintensitäten aus den LUI. In diesem Schritt wird somit ein Biodiversitätsrisikowert für die gesamte Landschaftsmatrix ermittelt.

Um die BioMAPS-Methode für ganze Produktsysteme über globale Wertschöpfungsketten nutzbar zu machen, werden außerdem Hintergrunddaten für alle Ebenen in einer GIS-Umgebung berechnet. Dazu werden globale statistische Daten zu Biodiversitätsrisikogebieten, Landnutzung und Landmanagement mit ökologischen Modellen kombiniert. Auf diese Weise erhält man Informationen über die Auswirkungen der Landnutzung auf die biologische Vielfalt für bestimmte Koordinaten oder ganze Länder. Dabei werden auf der Grundlage statistischer Daten globale Landnutzungsintensitäten für verschiedene Landnutzungstypen berechnet. Die standortspezifischen Bedingungen einzelner Parameter, wie Klima und Bodeneigenschaften, werden über globale agro-ökologische Zonen berücksichtigt. Die globalen Datensätze können als Hintergrunddaten für die Lieferkette genutzt werden. Verfügt ein Unternehmen über Primärdaten zu einzelnen Bewirtschaftungsparametern, z.B. wenn es weiß, wie viel Dünger auf den eigenen Feldern eingesetzt wird, werden diese Daten bevorzugt und in einem Vordergrundprozess verwendet. Wenn das Unternehmen nur das Herkunftsland eines Produkts angeben kann, wird stattdessen die Hintergrunddatenbank verwendet. Je genauer ein Unternehmen den Produktionsstandort kennt, desto genauer lassen sich die Auswirkungen bestimmen.

Die BioMAPS-Methode wird schließlich auf eine Fallstudie angewandt, in der die Biodiversitätsfolgen verschiedener Verkehrsenergieträger für einen Pkw bewertet werden. Dazu gehören Diesel und Benzin aus fossilen Brennstoffen, Strom aus erneuerbaren (Sonne, Wind) und nicht-erneuerbaren Ressourcen, Methan aus Photovoltaik-Strom, Biodiesel aus Palmöl, Bioethanol aus Zuckerrohr, Biomethan aus Mais und Methan aus Holz. Bei dieser Bewertung werden sowohl Vordergrund- als auch Hintergrunddatensätze verwendet. Generell lässt sich zeigen, dass die Fläche der Landnutzung ein entscheidender Faktor für die Gesamtauswirkungen auf die biologische Vielfalt ist. Daher haben Biokraftstoff-Energiequellen eine höhere Auswirkung auf die biologische Vielfalt als Energiequellen ohne Biokraftstoff (einschließlich des Stromnetzmixes). Die geringsten Auswirkungen auf die biologische Vielfalt haben dabei jedoch insbesondere elektrische Energie aus erneuerbaren

Quellen wie Wind- oder Solarenergie. Die höchste Auswirkung wird bei Biogas aus der Holzproduktion gemessen, gefolgt von Bioethanol aus Zuckerrohr. Bioethanol aus Zuckerrohr und Biodiesel aus Palmöl haben auf globaler Ebene die stärksten Auswirkungen auf die biologische Vielfalt, da sich die Produktionsanlagen in besonders schützenswerten Gebieten befinden.

Die in dieser Abhandlung vorgestellte BioMAPS-Methode ist die erste weltweit anwendbare Folgenabschätzungsmethode in LCA, die das Multiskalenkonzept der Biodiversität berücksichtigt, indem sie eine Brücke zwischen den Disziplinen Ökologie, Naturschutz und Ökobilanz schlägt. Sie nutzt und harmonisiert das breite Spektrum an verfügbaren Forschungsergebnissen und bietet damit Ökobilanz-Endanwendern einen kohärenten Rahmen für die Bewertung der Auswirkungen der Biodiversität von Produktsystemen. Im Ergebnis zielt diese Methode darauf ab, negative Auswirkungen durch die Identifizierung konkreter Maßnahmen auf der globalen, regionalen und lokalen Ebene abzumildern.

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List of abbreviations

AZE	Alliance for Zero Extinction
BfN	Bundesamt für Naturschutz
BH	Biodiversity Hotspot
BioMAPS	Biodiversity Multi-scale Assessment of Product Systems
BR_globe	Biodiversity Risk at global scale
BR_locLU	Biodiversity Risk at local scale depending on land use
BR_locLUI	Biodiversity Risk at local scale depending on land use intensity
BR_regLDI	Biodiversity risk at regional scale depending on Landscape Development Index
CBD	Convention on Biodiversity
CE	Crisis Ecoregion
CIAT	Centro Internacional de Agricultura Tropical
CIESIN	Center for International Earth Science Information Network
CMB	Conditions for Maintained Biodiversity
CPD	Center of Plant Diversity
CR	Critical
cSAR	Countryside Species Area Relationship
EBA	Endemic Bird Area
ED	Evolutionary Distinct
EDGE	Evolutionary Distinct and Globally Endangered
EN	Endangered
EU	European Union
FAO	Food and Agriculture Organization
GAEZ	Global Agro-Ecological Zones
GIS	Geoinformation System
HBWA	High Biodiversity Wilderness Area
HII	Human Influence Index
HSM	Habitat Suitability Model
HVO	Hydrotreated Vegetable Oils
IBA	Important Bird and Biodiversity Area
IFL	Intact Forest Landscape
IIASA	International Institute for Applied Systems Analysis

ILCD	International Reference Life Cycle Data System
IPA	Important Plant Area
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
IUCN	International Union for Conservation of Nature
KBA	Key Biodiversity Area
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LDI	Landscape Development Index
LTW	Last of the Wild
LUI	Land Use Intensity Index
MTI	Maximum Tolerable Intensity
NABU	Naturschutzbund
NDVI	Normalized Difference Vegetation Index
NetCDF	Network Common Data Format
NGO	Non-governmental organization
PBR	Potential Biodiversity Risk
PREDICTS	Projecting Responses of Ecological Diversity In Changing Terrestrial Systems
SA	Secondary Area
SAR	Species Area Relationship
SSP	Shared Socioeconomic Pathways
TLU	Tropical Livestock Unit
UBR	Uniform Biodiversity Risk
UNECE	United Nations Economic Commission for Europe
UNEP/SETAC	United Nations Environment Programme and the Society of Environmental Toxicology and Chemistry
UNEP-WCMC	United Nations Environment Programme World Conservation Monitoring Centre
WCS	Wildlife Conservation Society
WGS	World Geodetic System
WWF	World Wide Fund For Nature

Glossary

Biome	Geographical area with similar climatic and ecological conditions.
Biodiversity	A multi-scale concept that covers the diversity of ecosystems, the diversity of species and the genetic diversity within a species.
Cause-effect chain	A cause-effect chain graphically shows the relationship between the environmental intervention and ist possible effects. Impact indicators can be selected at different stages of this chain, for example as mid-point or end point.
Characterization factor	Characterization factors are used to quantify the influence of a product or service in the various impact categories. In other terms, they express the contribution of a single mass unit of the intervention to an impact category.
Ecosystem services	Ecosystem services are assets that nature brings to mankind, such as pollination or carbon sequestration.
Endemism	Endemics are organisms that only occur in a specific, spatially delimited environment. These can be for example islands or mountains.
ILCD flow list	The ILCD flow list is a common reference for an elementary land use flow list for use in LCA land use impact assessment.
Irreplaceability	The degree to which conservation options are lost if a specific site is lost.
GIS	Geoinformation systems are information systems for the acquisition, processing, organization, analysis and presentation of spatial data.
Landscape	Area that consists of different patches of land use types.
Land use flows	All types of land use that are involved in a product’s life cycle.
Life Cycle Assessment	Standardized approach to assess environmental impacts throughout the whole life cycle of products or services.

Occupation	Occupation is used in LCA for the description of the current use of land.
Proactive conservation	Focuses on areas with low vulnerability (very low threat), that are usually quite pristine and undestroyed habitats.
Reactive conservation	Focuses on areas with a high vulnerability (very high threat). These are usually severely degraded habitats.
Scale	Level of hierarchy; in biodiversity science four different scales are distinguished (spatial, organizational, temporal, administrative).
Taxa	Group of living organisms within biological systematics.
Transformation	Transformation describes in LCA the change from one type of land use to another type.
Vulnerability	The likelihood that the biodiversity of a site is threatened (in the future).

1 Introduction

The climate crisis, the loss of biodiversity as well as the overuse of resources are symptoms of the Anthropogenic Age we currently live in – and humanity's greatest challenges of the 21st century. As highlighted in the planetary boundaries concept, the loss of biodiversity poses an even more serious risk than the other eight global ecological threats such as the climate crisis, ocean acidification, chemical spillovers or the human impact on the nitrogen cycle [1–5]. This loss of biodiversity is mainly driven by mankind while its rate far exceeds the extinction rate of the natural background; hence, the scientific community considers the current loss of biodiversity to be a major mass extinction, where the last happened around 65 million years ago [6–11]. Just recently, a warning on the status and trend of biodiversity loss has been issued by the Convention on Biodiversity (CBD) and by thousands of concerned scientists [12,13] which further illustrates the urgency of the biodiversity crisis.

With regard to this crisis, a multitude of studies show that the main cause of biodiversity loss around the world is attributable to land use such as agriculture or urban development [13,14]. As a result, we need to focus on how to assess the impact of land use and how to mitigate the effects. Even though there are other drivers of biodiversity loss besides land use, such as climate change and invasive species [13]. Global product chains can also amplify climate change, and long transport routes can introduce species from natural habitats to new regions. However, these impacts are not the scope of this investigation.

1.1 Relevance of biodiversity loss

The rapid loss of biodiversity is not only a challenge from an ecological point of view, but also for the functioning of our world economy and the vitality of societies around the world [5,15]. A highly topical example of social health impacts and related socio-economic effects is the emergence of zoonotic diseases such as Ebola or COVID-19, which is directly related to wildlife trafficking and the anthropogenic destruction of habitats and ecosystems [16–18]. Healthy and intact ecosystems are of fundamental importance for societies and economies in both developing and industrialized countries [19]. They are able to provide vital services for the economy and society, which are also referred to as ecosystem services [13]. Ecosystem services include nitrogen fixation in soils, the fixation of greenhouse gas emissions, soil protection, a cooling effect on urban microclimates or the pollination of agricultural crops. The services for society as a whole are e.g. the production of nutritious and varied food, materials for textiles or access to clean drinking water [13]. However, ecosystems require a variety of taxa that enable complex ecological interactions and thus ensure this variety of ecosystem services [20–22]. Thus, ecosystem services are directly related to biodiversity, as it has been proven that several ecosystem functions increase significantly with increasing biodiversity. This complex interaction is also called multifunctionality [20–22]. A prominent example of the socio-economic dependency on ecosystem services is the role of pollinator species within the production of many types of food. With the ongoing loss of such species, the ecosystem loses some of its functions, which in the worst case could lead to the disruption of the trophic cascade and the collapse of the entire ecosystem [5,12]. According to Webb et al. [23] in average about a quarter of all species are threatened with extinction, varying within each taxon. And as several ecosystem functions increase strongly with high biodiversity [21,22], the current loss of biodiversity has also a direct impact for industry and companies.

The Natural Capital Protocol stresses the importance of biodiversity for business and industry. It compares biodiversity to a stock market that generates flows in the form of ecosystem services that are profitable for companies [24]. The analogy of biodiversity as a stock market indicates the fundamental conflict of interest for industry and businesses because short-term profits are highest when biodiversity is destroyed without consideration, but then long-term profits will be at a low. For example: waste, land degradation, ecological disturbances or changes in habitats belong to the main drivers of biodiversity loss [15,25]. On the other hand, businesses are heavily dependent on ecosystem services such as the provision of raw materials or water regulation. Examples of such ecologic-economic dependencies are the extinction of pollinators, deforestation, soil degradation or a depletion of marine and terrestrial species. Examples of collapsed ecosystems that are no longer able to provide the vital services serve as a warning of the high consequential damage, as is already the case in China, where people have to pollinate agricultural crops by hand [26] or in Brazil, where deforestation has led to droughts and water scarcity in urban areas [27].

The primary economic sector in particular shows a high dependency on biodiversity and ecosystem services, as it concentrates on the cultivation and use of agricultural and forestry products [5,15]. Also for the secondary and tertiary sector, additional risks affect suppliers or customers as part of the supply chain, i.e. higher costs for resources, increasing scarcity of resources or tightening of regulations [15,24]. This shows the urgency of the biodiversity crisis also for the private sector and that the protection of biodiversity forms the basis for a sustainable development and the prosperity of future generations and would also enable business opportunities in the long run.

In the private sector, assessing the impact of land use processes on biodiversity (of products or services) is usually conducted by life cycle assessment (LCA). It is a prominent tool to assess the impact of materials, products and processes along the entire value chains, and which should provide decision support to producers and consumers on more biodiversity friendly alternatives. Even though there are valuable methods and approaches for the assessments of land use impacts on biodiversity in LCA, such as following authors [28–59], they only focus on certain aspects of biodiversity [5,60,61]. As ecologists highlight, biodiversity is a multi-scale concept [62,63], therefore there is a need for a method that is able to assess the impacts at different spatial and organizational scales (namely, ecosystems, species, genes). And as shown in the course of this thesis, so far there is no globally operational method that is able to account for this multi-scale concept and to assess biodiversity as holistically as it is defined by ecologists. Although the loss of biodiversity has a high relevance for companies, it is still rarely considered in their life cycle assessments [64]. This is often justified by the fact that existing methods for assessing biodiversity are still subject to restrictions in their global applicability and, moreover, cannot provide end users of LCA with sufficient decision support at all levels relevant to the product life cycle [5,64]. Thus, the awareness that in principle all socio-economic activities have disruptive impacts on biodiversity, on which they also depend, makes the development of a tool to quantify impacts on biodiversity at different scales and to derive recommendations to mitigate these negative effects an urgent priority [65].

1.2 Biodiversity methods in LCA: an interdisciplinary approach

The concept of biodiversity comes from a discipline that is naturally studied by ecologists and biologists. Yet, the technical assessment of impacts on biodiversity from products or processes is carried out by an LCA which is being developed and researched in the engineering sciences. Both areas, biodiversity research and LCA, have their own state of the art and requirements for the development of a biodiversity method. This results in research gaps since existing biodiversity methods in LCA are not yet able to assess biodiversity as it is defined by ecologists and biologists [5,61]. The aim of this work is to bridge the gap between the two disciplines by developing a method that takes into account the requirements of both sides while guiding LCA end users to make biodiversity-conscious decisions.

The purpose of this dissertation is therefore to develop a biodiversity multi-scale assessment method accounting for impacts at several scales. Since there is no globally operational method to model and compare the impacts of companies and products on biodiversity at different scales, it is difficult to make changes from a company management or consumer perspective. Simply put, you cannot manage what you cannot measure. Therefore, this method should combine underlying concepts, methods and databases from different disciplines. The importance for such an interdisciplinary multi-scale method and the underlying concepts and models are further described in chapter 2.

1.3 Chapter overview

Considering the need to assess the impact of products on biodiversity and the lack of a universally and globally applicable tool, this dissertation addresses this desideratum.

The dissertation is structured as follows (see also Figure 1):

After having highlighted the relevance for integrating biodiversity into LCA in this introduction of **chapter 1**,

chapter 2 illustrates the state of the art and requirements for the development of a new method from a conservational and ecological point of view as well as from the technical requirements of LCA, including an end user perspective.

In **chapter 3**, the research gaps are identified using the current state of the art of existing LCA methods. Based on this review, further specific requirements for the development of a biodiversity impact assessment method in LCA are derived.

Chapter 4 lays out the development of the method that addresses the research gaps and requirements identified in chapter 2 and chapter 3. Subsequently, the methodological framework is being described for a multi-scale assessment including the calculation rules and the adaptation of the land use flow list.

The method is then made globally operational in **chapter 5**. It illustrates the calculation steps for the biodiversity impact assessment at the global, regional and local scale in a GIS environment.

The applicability of the method is demonstrated in **chapter 6** by applying it to a case study of different transport energy systems. Afterwards, the results of the case study are presented and discussed.

Finally, the BioMAPS method is evaluated and discussed in **chapter 7**.

Chapter 8 concludes this investigation by describing further research needs and by providing an outlook and a short conclusion.

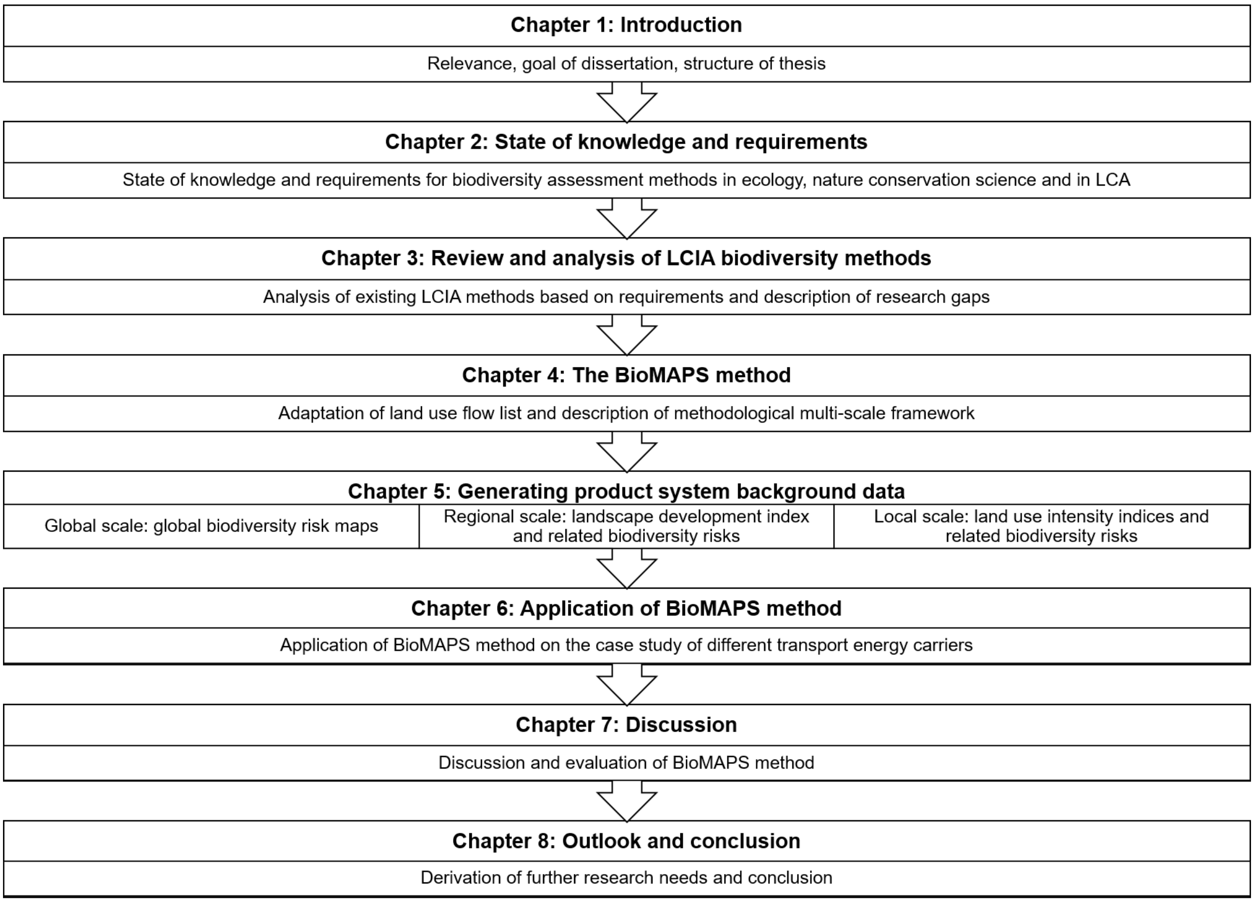


Figure 1: Structure of the thesis

2 State of knowledge and requirements

This section illustrates the requirements for a method which integrates biodiversity into LCA. Since the development of a method for a biodiversity impact assessment in LCA is a cross-cutting topic, the requirements take in two perspectives: first the field of biodiversity conservation science and ecology and second the field of LCA. Furthermore, general requirements for method development in LCA are described as formulated by the European Commission in their International Reference Life Cycle Data System (ILCD) handbook [66].

2.1 State of knowledge in ecology and nature conservation disciplines

In this sub-chapter the state of knowledge for biodiversity assessments is described from the perspective of biodiversity conservation science and ecology.

2.1.1 Biodiversity as multi-scale concept

Ecological and nature conservation considerations are key to understand and define biodiversity in order to find appropriate concepts, models and indicators that allow for a thorough assessment. Biodiversity is a complex and multi-scale concept [62,67,68]. Henle et al. describe three different scales that are important for biodiversity conservation and management [62]. Herein, the term scale is defined as a certain "level of hierarchy" which is based on the definition of Wu et al. [69]. The first scale deals with the organizational levels of biodiversity and its components which include genes, species and ecosystems [69]. The second scale relates to the spatial variability of the diversity of life that can be described and assessed on a global, regional as well as local scale. The third relates to the temporal scale, describing short term or long term impacts [62].

As Henle et al. stress all three different scales (the organizational, spatial and temporal scale) have to be considered in a holistic management of biodiversity [62]. Furthermore, they describe another scale which is not directly related to ecology but is essential in the conservation and management of biodiversity. This includes the administrative scale where decision makers can directly or indirectly influence biodiversity at different levels (e.g. governments by making new environmental regulations, municipalities by designing biodiversity friendly cities, and farmers or land owners by managing their fields environmentally friendly). Each of these scales follow their own hierarchy and can be further subdivided into several levels that directly influence each other (see Figure 2).

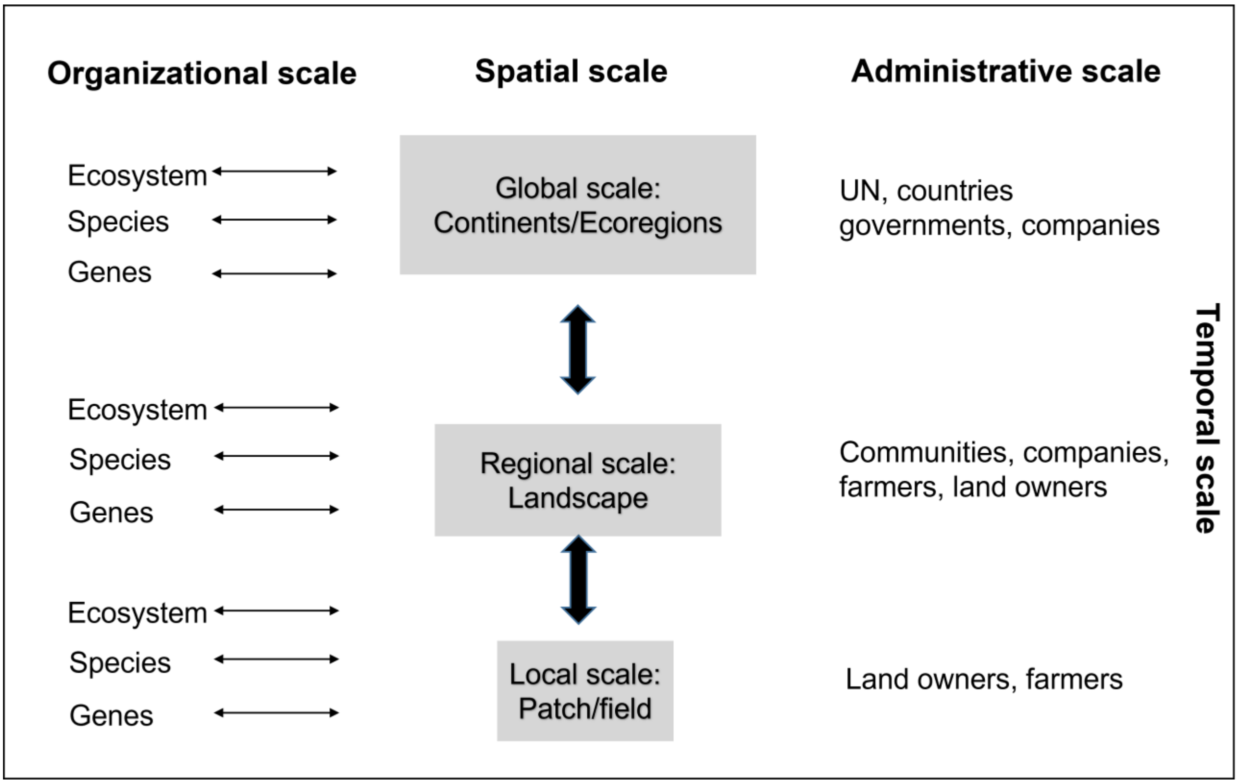


Figure 2: Scales in ecology and biodiversity conservation based on the concept of Henle et al. and Wu et al. [62,69]

This work takes the multiscale concept of Henle at al. and Wu et al. [62,69] as a basis together with other concepts that are presented here to develop a biodiversity impact assessment method in LCA. The administrative scale will be further explained in chapter 2.2 since LCA should provide support for decision makers at different scales with regard to products and processes. The temporal scale cannot be considered since the modelling of the temporal scale is subject to the LCA modeler. With that in mind it will only be integrated through the modelling of the LCA users (see chapter 4.1.1). The organizational and spatial scales are described from an ecological point of view in the subsequent subchapters. Due to the structural design of the method and the resulting logic of analysis, the local scale is in the following addressed before the regional scale, even though this may seem counterintuitive at first sight (see chapter 2.1.3).

2.1.2 Organizational scale: definition of biodiversity

The organizational scale is directly derived from the definition of biodiversity. With regard to the organizational scale of biodiversity, most natural scientists agree on the definition of the Convention on Biological Diversity [70], which states: *"'Biological diversity' means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems"* [70]. As emphasized in this definition of the CBD, the preservation of the wealth of biological processes is of fundamental importance, and this wealth can be described with three organizational levels: genes, species and ecosystems. These organizational levels of biodiversity are sometimes referred to as genetic, organismal and ecological diversity [71]. Herein using indicators at different organizational levels, such as the measurement of the number of species or the composition or intactness of an ecosystem can be used to assess impacts on biodiversity. Although this method takes up the three-level definition of biodiversity, it is important to note that, like all definitions and classifications, it is a simplified classification of reality. Furthermore, we have to realize that all levels are interrelated: Genes define species, and several species form a complex ecosystem (see Figure 3).

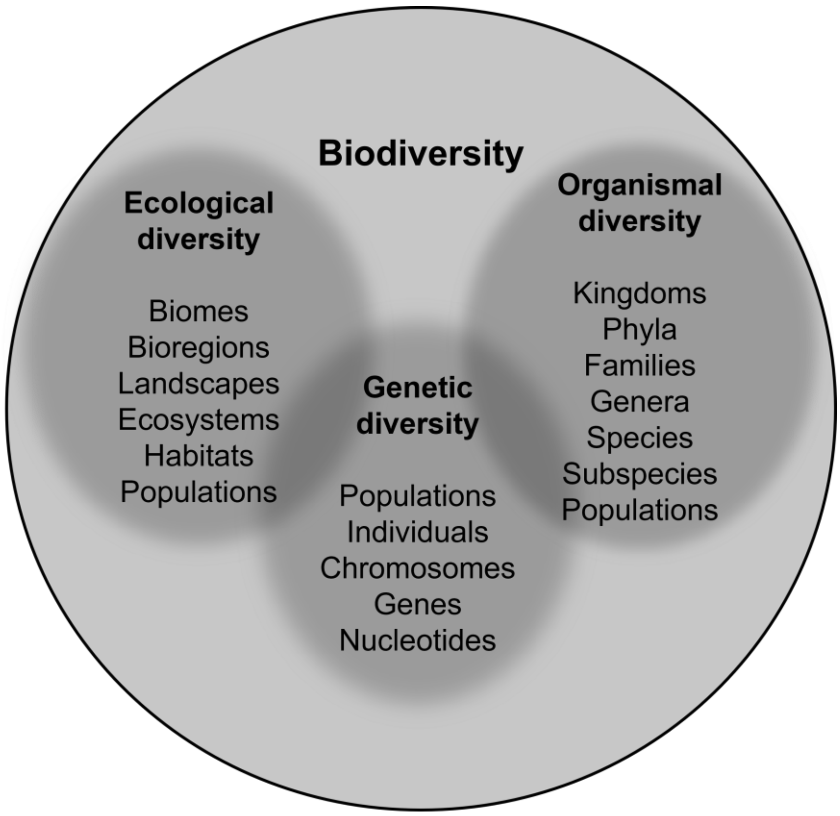


Figure 3: The organizational levels of biodiversity based on Heywood [71]

Since the three organizational levels of biodiversity are directly interconnected, double counting cannot be avoided. This is in contrast to the technical requirement of the LCA to avoid double counting [72]. But from an ecological point of view, the consideration of all organizational levels is crucial. This involves the genetic level, as it is critical to the evolutionary adaptability of species that define a community, a population and an entire ecosystem. Species adaptation, as defined by genetic selection in response to the changing environment, is of vital importance, especially in an age of global change and large-scale extinction [5].

For each of these organizational levels there is a set of indicators or biodiversity metrics for measuring biodiversity. These indicators either measure the conservation value of biodiversity (e.g. by analyzing the occurrence of rare ecosystems or the number of threatened species) or they target the ecological function and resilience of biodiversity, e.g. by measuring the composition, structure and function of each of the three levels [73]. Duelli et al. suggest using a “basket of indicators” for the holistic assessment of biodiversity [73]. Several main ecological indicators are depicted in table 1.

Table 1: Some ecological indicators at different levels of biodiversity (after Duelli, Obrist and Noss [73,74])

Organizational level	Indicator
Genes	Phenotypic diversity, genealogical, allelic diversity
Species	Species richness, abundance, species composition, functional diversity, evenness
Ecosystems	Ecosystem heterogeneity, habitat richness, landscape composition

Commonly used conservation indicators are depicted in Table 2. They can be placed in the concepts of vulnerability and irreplaceability:

Table 2: Conservation indicators at different levels of biodiversity (non exhaustive list) (after Gordon et al. and Brooks et al. [75,76])

Organizational level	Indicator
Genes	Phylogenetically unique species, rare gene pool
Species	Rare species, endemic species, threatened species
Ecosystems	Intact ecosystems, rare ecosystems, degraded ecosystems, habitat loss, fragmentation

The inclusion of all organizational levels of biodiversity which are accessed through suitable biodiversity metrics and indicators forms one requirement for the development of the new method.

2.1.3 Spatial scale: global, regional and local impacts

The spatial scale can be broadly differentiated into global, regional and local biodiversity impacts. For measuring global impacts one has to consider the distribution and threat patterns of biodiversity worldwide. Within a regional scale biodiversity impacts are measured for example at a landscape level, whereas local biodiversity impacts are directly assessed at a field or patch level.

2.1.3.1 Global distribution and risks of biodiversity

Since we want to develop a method that is able to assess biodiversity impacts worldwide, we have to bear in mind that biodiversity is not evenly distributed around the globe. With reference to the global distribution of biodiversity, there are general macro-ecological distribution patterns: terrestrial species richness is greatest at the equator, while it decreases towards the poles [77–79]. In addition to this unequal distribution, risks to biodiversity are also spread differently worldwide [77,80–82]. Because of this unequal distribution of species richness and threats, conservationists must prioritize between areas with higher and lower urgency for biodiversity conservation. Herein, the priorization criteria follow the organizational three level definition of biodiversity, including ecosystems, species and genes. At each level, one can also distinguish between ecological indicators or conservation indicators. The latter focus on the concept of vulnerability or irreplaceability. For the measurement of irreplaceability common taxa such as endemic plants, birds or vertebrates are often used. The higher the number of endemic species in a region, the higher the risk that critical biodiversity will be lost due to the transformation or use of the ecosystem. Some global protection systems also incorporate other indicators of irreplaceability, such as taxonomical distinctiveness, rare phenomena and the global scarcity of key ecosystems [5]. Conservation indicators to monitor vulnerability are the degree of threat to the species (based on the IUCN Red List) and the size of habitats lost in ecosystems or an endangered gene pool. The vulnerability level is then ranked as significant (therefore suitable for a reactive approach) or small (important for proactive conservation) [5], with both approaches being considered crucial for the preservation of biodiversity [75,76,83].

In order to analyze the difference in the distribution of biodiversity and its threats, nature conservation scientists and NGOs have identified so-called biodiversity conservation schemes. These schemes are typically presented as maps representing areas of higher and lower conservation value, based on a specific set of criteria. Scholars and non-governmental organizations have conducted extensive research to pinpoint global key areas where the preservation of biodiversity is most critical. As highlighted in [5], these are for example the High Biodiversity Wilderness Areas [84], Frontier Forests [85], Global 200

Ecoregions [86], the Last of the Wild [87], Endemic Bird Areas [88], Centers of Plant Diversity [89], Biodiversity Hotspots [90] or Crisis Ecoregions [91]. These global protection systems can all be placed in the context of irreplaceability or vulnerability, though some schemes are a combination of both [5,76]. As Brooks et al. have shown in their study, almost 80% of terrestrial areas worldwide are home to vulnerable or irreplaceable sites for biodiversity [76]. Based on these two different categories, vulnerable areas can be further divided into "reactive" (very high vulnerability) and "proactive" (very low vulnerability) systems [76]. However, researchers have also emphasized that there is as yet no "one-size-fits-all" solution for the representation and prioritization of biodiversity [83]. Each biodiversity conservation scheme focuses on a different aspect, e.g. certain taxa, the degree of threat to different species, or particularly intact versus severely degraded ecosystems [5]. As a consequence, there is a spatial mismatch between proactive and reactive conservation schemes since their focus is exactly opposite.

Here, some challenges for a holistic global biodiversity assessment become apparent: One reoccurring gap is a lack of including the genetic, the species and the ecosystem level at the same time. Herein, Brooks recommends that the phylogenetically different species, such as those identified by Isaac et al. [92], must be included in each biodiversity assessment [93]. Another problem is the under-representation of non-vertebrate species, such as insects or below-ground biodiversity [76,94–97]. This is mainly due to the fact that these taxa are currently less well studied than, for example, vertebrates. Taking below-ground biodiversity into account is, however, also crucial because the richness of biodiversity below and above the ground might vary significantly according to the findings of Cameron et al. [98]. Another approach is the hotspot concept [90,99,100] which was critically reviewed by Marchese et al. [101]. Hotspots include species from a wide variety of taxa using endemic plants as indicators for endemic mammals, birds and amphibians. Unfortunately, this approach is solely based on the indicator of species richness. Thus, each approach has merits by highlighting critical aspects of global biodiversity. At the same time, there is no single approach or study which would suffice to cover the manifold aspects of biodiversity. First, each approach lacks some aspects. Second, neither scientists nor representatives of NGOs have yet agreed on a common biodiversity conservation scheme. And third, it is crucial to include a biodiversity which is as geographically spread, as distinct from each other and as unique as possible to have the best ‘insurance’ for life on this planet. Consequently, taken together, the various global data sets of conservation schemes provide a good and well-founded database on the global distribution of biodiversity and its conservation status [83].

Bearing this in mind, a new LCA method must account for the global dimension. Therefore, a uniform global biodiversity risk map should be developed indicating areas of high biodiversity conservation value worldwide, including proactive and reactive areas, as well as areas that are irreplaceable for biodiversity. With such a biodiversity risk map the impact of land use can be assessed no matter where it is taking place. A global risk factor must therefore be included in the method to take account of the fact that biological diversity is neither homogeneously spread worldwide nor are species, genes or ecosystems equally threatened [5].

2.1.3.2 Local scale: field and patch perspective

On a local scale the impacts on biodiversity are assessed at a patch or a field level. At this level, ecological studies have shown a difference in the impacts of biodiversity due to different land use types such as agriculture, pasture, or forestry [102–104]. Studies by Martins et al. [102] and Newbold et al. [103] show that on average the impact of the land use type forestry is smaller than the impact of agriculture or urbanized areas in comparison to primary vegetation. Martins et al. [102] show that there is a better response of bird taxa to forestry and pasture than to agricultural land. Surprisingly, the results indicate no significant difference in the evaluated land use types throughout different broad climatic regions (e.g. tropical and temperate) except for pastures that had a higher negative impact in the tropics according to the findings of Martins et al. [102], whereas Newbold et al. [105] found no statistically significant difference also for pasture. This facilitates the development of a global and comparative assessment. The biodiversity indicators to measure impacts at the local scale are usually at the organizational biodiversity level of species. Herein, depending on the biodiversity indicators (=metrics) used, Newbold et al. [105] found different responses of the impact of land use on for example species richness, species abundance or composition or the functional diversity [105]. This requires again the use of a “basket of indicators” as highlighted by Duelli and Obrist [73].

Thus, the inclusion of a factor that accounts for land use type specific impacts on biodiversity at a local scale forms another requirement for the development of the method. Yet, these local impacts do not solely depend on the type of land use but also on the intensity. Several scientific studies have shown a statistical significance of the degree of land use intensity and the associated impacts on biodiversity [102,103,106–114]. While some studies found various effects of land use intensity for certain taxa [107,109], other studies found no statistically significant difference between different taxa [105]. Thus, no universal statement can be made on the impact of land use intensities on all taxa. Still, there is an agreement that reducing the land use intensity generally benefits biodiversity at all organizational levels [103,114]. Several scientists name the current increase in land use intensity one of the main reasons for the loss of biodiversity [114–118]. Land use intensity is described by the type and quantity of land management practices such as the application of fertilizers and pesticides, the management of crop rotation systems or the age structure in forests [111,119]. Depending on the management practices, each type of land use has a different spectrum of impacts on biodiversity. Therefore, the method should be able to quantify land use intensities to account for this broad variation in impacts.

2.1.3.3 Regional scale: landscape perspective – land sparing versus land sharing

Finally, the impacts on biodiversity should be evaluated in a landscape perspective, as well. Ecological studies show that biodiversity impacts of a certain land use type are not only restricted to local impacts at the patch or field level but also effect biodiversity of the surrounding areas [114,120]. This is especially the case for organisms with high dispersal abilities or migrating species. Also, the effects of management on a field influence biodiversity within the surrounding landscape. For example: the intensity of pesticide application also influences the adjacent patches and its fauna and flora, as shown for example by Hallmann et al. [121] for insects. The impacts at a regional scale are usually measured with organizational biodiversity indicators at the ecosystem level such as the landscape composition (the share of different landscape elements e.g. patches of land use types and native habitats) or the landscape configuration such as heterogeneity (e.g. the diversity of habitat types, clustered or widespread patches) or aspects of connectivity and fragmentation. These indicators also directly influence the species and genetic biodiversity levels as a fragmented landscape can negatively impact populations and their gene pool. The negative influences on species, from a changed landscape configuration increase significantly with the reduction of the total amount of habitat in the landscape [122]. Therefore, these authors are of the opinion that the size of residual habitat volume must first be considered before the effects of habitat fragmentation can be assessed independently. They believe that the fragmentation or perforation itself hardly affects the species and ecosystem dynamics of landscapes, but that habitat loss, regardless of its spatial configuration, is primarily responsible for the subsequent biodiversity impacts [122]. Based on their study we can conclude that especially the landscape composition (and the share of primary and secondary vegetation) is of highest importance for biodiversity conservation. Furthermore, Tscharntke et al. [114] illustrate how a structurally diverse landscape matrix improves even the local biodiversity of cropland land use systems, and how it might even compensate for high intensive land management at a local scale [114].

Therefore, the method should take into account not only the impacts of certain land use types at field or patch level, but also the impacts of biodiversity within the overall landscape in which the land use process is embedded. This requirement contributes to one of the most debated concepts in biodiversity science: the question of land sparing versus land sharing [123–125]. Following the land sparing approach, some areas are set-aside for biodiversity conservation while other areas are designated for intensive land use. In the land sharing approach, more area of land may be used economically, however, it must be used less intensively and "biodiversity friendly" through specific management practices [124]. Both strategies have proven to be beneficial for biodiversity, and many conservationists point out that a combination of both land sharing and land sparing strategies (depending on the local conditions) is required for effective biodiversity protection [126–128].

2.1.4 Summary of main requirements from an ecological perspective

To summarize, the main ecological and conservation requirements include the need for a multi-scale method. Herein, a "basket" of biodiversity indicators should be used at all spatial and organizational scales. This "basket" should be comprised of ecological and conservation indicators. At a global scale, a biodiversity risk map, based on vulnerability and irreplaceability concepts, shall be developed. Then, these risks have to be regionalized in the LCA method, accounting for the different types of land use. The local scale considers biodiversity impacts at the field and patch level, while the regional scale looks at the landscape perspective and considers land sharing and land sparing conservation strategies.

2.2 State of knowledge in LCA discipline

In this chapter the requirements for a biodiversity method in LCA are outlined from a LCA and end user perspective. This includes the general requirements for the development of impact assessment methods in LCA, as well as specific requirements for biodiversity assessment methods and decision support for LCA addressees. In regard to the general requirements the following section details the general standardized structure of a LCA in which the method will be embedded and it describes the different phases within a LCA while focusing on two of the phases that are most relevant for the development of a biodiversity impact assessment method – Life Cycle Inventory (LCI) and Life Cycle Impact Assessment (LCIA).

2.2.1 Requirements in LCA

In order to understand the technical requirements, we need to understand the standardized steps of a LCA first as they form the underlying basis for the development of a biodiversity method in LCA.

2.2.1.1 Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA)

LCA is a method for assessing and evaluating the environmental impact of a product or process. The underlying basis of a LCA is the concept of Life Cycle Thinking which divides the "life cycle" of e.g. a product into life cycle phases such as resource extraction, production, use and the end of a product or process. LCA is an established and well recognized method based on the ISO 14040/44 standards which encompasses four different steps that are always to be executed [129,130].

Goal and scope: In the first step, the goal of the LCA study should be described and the system boundaries defined. Herein, the functional unit is described. Furthermore, it is also defined whether an attributional or a consequential LCA is being performed. While the attributional LCA describes input and output flows assigned to the selected functional unit, the consequential LCA also describes changes in the system in the context of changes in the output of the functional unit [131].

Life Cycle Inventory (LCI): All input flows (e.g. energy and materials) and output flows (e.g. emissions, waste and products) are detailed and recorded for each life cycle phase.

Life Cycle Impact Assessment (LCIA): An impact assessment is carried out by linking input and output flows to characterization factors of specific impact categories such as climate change or biodiversity. In this phase the actual biodiversity impact is analyzed yielding the results. The herein developed method, therefore, will be embedded within the Life Cycle Impact (LCIA) assessment phase of the LCA framework.

Results and interpretation: The results of the LCA are described and interpreted. Since LCA is an iterative process the results and their interpretation can lead to modification in the previous phases.

The following standardized steps of a LCA are shown in Figure 4.

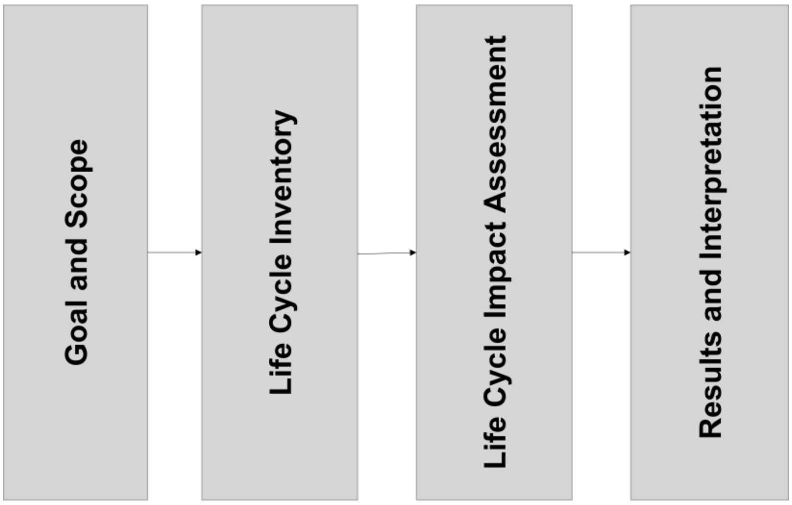


Figure 4: Standardized steps of a LCA after ISO [129,130]

2.2.1.2 Land Use Inventory and Land Use Flows

In order to conduct a Life Cycle Impact Assessment, the Life Cycle Inventory phase is crucial because it is during this phase that all input and output data of a product's life cycle is gathered. For a method that deals with the assessment of land use impacts, the data has to include all relevant information on past and current land use types that were/are involved throughout the life cycle of a product or process. This includes the specific area where land use occurred (e.g. country, region or specific location) and the timeframe of the land use activities [132,133]. All relevant input data for assessing the impact of land use processes on biodiversity are depicted in Figure 5.

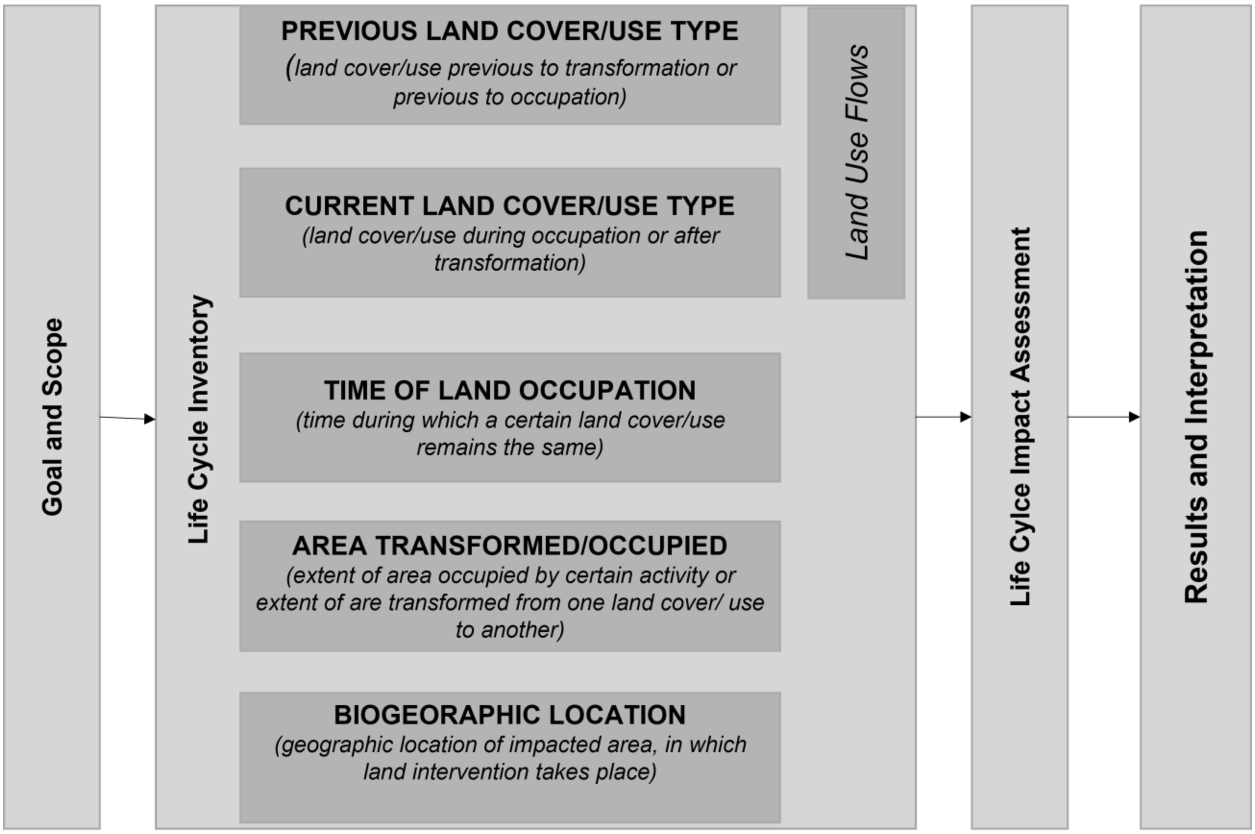


Figure 5: Input data gathered during the LCI phase, adapted after Taelman et al. and Mila i Canals et al. [132,133]

In the terminology of LCA, land use types that are involved in a product's life cycle are also referred to as land use flows. These land use flows are categorized and divided into classes and subclasses depending on the level of information in the elementary flow list.

2.2.1.2.1 Nomenclature of land use flows

The nomenclature of land use flows stems from the earth sciences and remote sensing that investigate the earth's surface with satellites and spatial maps and classify its land cover as well as the different types of land use accordingly. Land cover is defined as everything that covers the earth's surface and that can be determined by direct observation, whereas land use requires a sort of interpretation of an activity in a socio-economic context [134]. The land use flows in LCA are based on these land use and land cover classifications, such as the Global Land Cover 2000 [135], GlobCover [136] or the CORINE land cover classification system [137]. Depending on data availability and knowledge of the company and/or LCA modeler the information on the type of land use can be more generic or specific. Since there are several options of how to categorize land use types, this study builds upon the land use categorization of Koellner et al. [138] and Bos et al. [139,140] which has also been adopted and recommended by the EU as part of the International Reference Life Cycle Data System (ILCD) handbook [66] and is sometimes also referred to as ILCD flow list.

These authors differentiate between four different levels of land use flows (i.e. land use types). In the first level, general information on a land use class, though sometimes only on land cover, is described. Then, each subsequent level keeps refining the information on land use. For example: the first level might categorize land use as “agriculture”, whereas the second level might then provide more details such as "arable farming", the third level lists land use practices and the fourth level describes land use intensities [138]. Despite the advantages of this approach, there are also some inconsistencies: level 1 provides sometimes the land cover and sometimes the more detailed land use, some management parameters should be at level 3 listing land use practices such as “irrigation” and “fertilization” but they are not. In addition, the land use intensity of level 4 should be listed before the land use practices of level 3 because the information of “intensity” is a less detailed information than the question of specific “practices” and the practices combined determine the intensity. Furthermore, this nomenclature only provides a limited number of land use flows that can be assessed in the impact phase. In accordance with that, scientists call for the need to update and refine existing land use flows [61]. Therefore, as part of the method an updated land use flow list will be developed and provided which can also be matched to the ILCD flow list. Once all data and information on land use types, time and area of the land use as well as the region is gathered, the LCIA phase follows.

2.2.1.3 Life Cycle Impact Assessment – land use impacts in LCA

In the LCIA phase two steps of classification and characterization are executed. First, all input flows are assigned to an environmental impact category (e.g. biodiversity or soil quality), this step is called classification. Second, based on the classification the impact is quantified in the characterization step using a so-called characterization factor. The characterization factor in the LCA describes the impact on biological diversity. It is multiplied by the land used and thus characterizes an area with regard to its effects on biodiversity. From an ecological perspective the characterization factor describes the difference of biodiversity between the site where land use takes place and a reference site based on an ecological indicator. The resulting two values are called biodiversity quality in LCA. The difference between both values is the actual biodiversity impact, which can be analyzed at different spatial and organizational scales depending on the chosen indicator (see Figure 6).

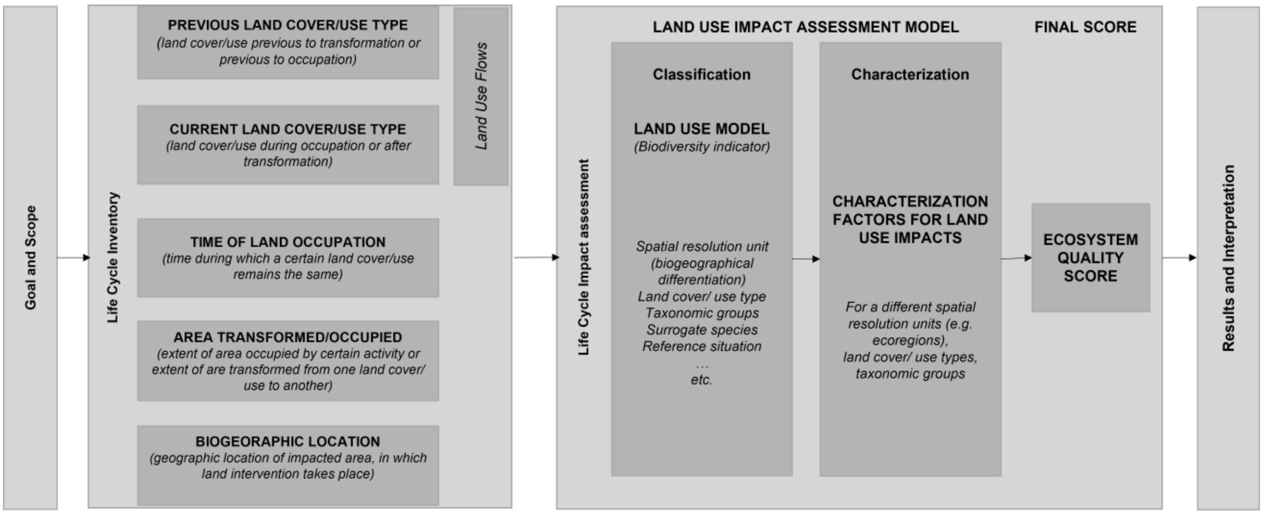


Figure 6: Connection of Life Cycle Inventory phase with LCIA adapted after Taelman et al. and Mila i Canals et al. [132,133]

In order to develop characterization factors, it is necessary to distinguish between the land use flows of land transformation and land occupation. Land occupation takes into account the biodiversity quality of the current land use type and area within a given time period and is therefore measured in area-time units (e.g. m² and year) [138,141]. Land transformation, on the other hand, is the change in the quality of biodiversity between one type of land use and another. There are two types of transformations that are considered in a LCA, namely permanent and reversible transformation [138]. The reversible transformation describes the quality of land after it is no longer used anthropogenically. It is assumed that an abandoned land can no longer achieve the biodiversity quality of the original natural land cover, but only a "quasi-natural state" [142]. The reversible transformation is therefore calculated by accounting for a regeneration factor for the time it takes for the land to reach this quasi-natural state. Permanent transformation measures the quality of land between different land use types. Permanent transformation does not consider a regeneration factor, so the calculation is time independent [142]. In the permanent transformation, both the pre-use and the post-use state of land are considered e.g. the transformation from past primary forest to current pasture land use. In the post-use state, the quality of biodiversity of possible future land use developments is being compared with the quality of current land use (e.g. the transformation from current pasture land use to future cropland land use) [5].

Both types of transformation are measured as area of transformed land. Although it is not necessary to model the effects of permanent transformation in relation to its time horizon, Koellner et al. [142] recommend nevertheless to consider the transformation time in the modelling in order to aggregate occupation and transformation flows with the same unit

[142]. The focus of this dissertation is on the assessment of occupation and permanent transformation as these are also considered in the LANCA model [139,143,144].

2.2.1.3.1 UNEP SETAC land use framework and biodiversity quality

For the calculation of characterization factors, most LCIA methods make use of the UNEP SETAC framework [138,141,142]. Therein, the biodiversity "quality" triggered by land occupation and land transformation, is compared with a reference state to receive the impact (ΔQ), depending on the characterization factor [5]. In order to assess the change in biodiversity quality, a great variety of ecological indicators are used in existing LCIA methods at different organizational levels of biodiversity, with most of them being at the species and ecosystem level [5]. The characterization factor is combined with the inventory of land use flows to determine the impacts on biodiversity. Figure 7 depicts the change in biodiversity quality transformation and for occupation. Herein, occupation takes place between t2 and t3 and is calculated as a change in biodiversity quality between the reference situation (which might be primary vegetation) LU0 (land use type) and LU3. The reversible transformation after the occupation takes place between time t3 and t4 and leads to a change in biodiversity quality between LU3 and LU2. The permanent transformation after occupation is calculated as the difference between the biodiversity quality of LU2 and LU1, it is time independent.

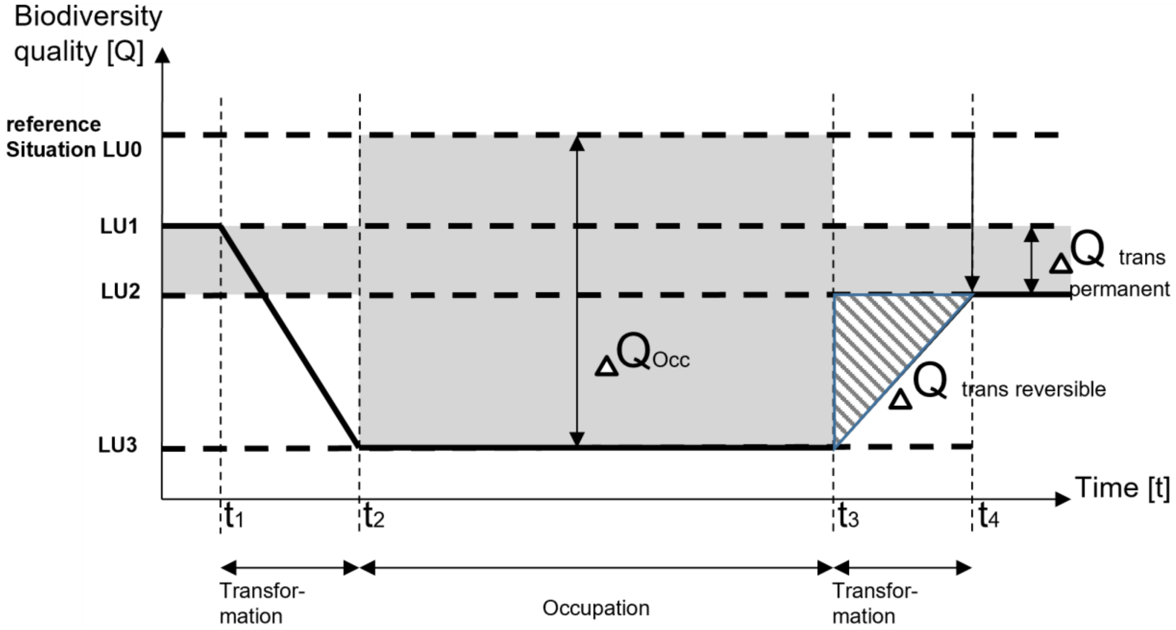


Figure 7: Change in biodiversity quality through land use based on Koellner et al, Mila i Canals et al. and Bos [141–143]

2.2.1.3.2 Reference situation

For the calculation of the characterization factor, the choice of the reference situation (LU0) has a decisive influence on the difference in the "quality" of biodiversity. According to the Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES), there are three general approaches for choosing a reference situation [145]. First, one can compare a "natural state" with an anthropogenically influenced state. The "natural state" reference situation can be defined either as a time-bound natural state or as a counterfactual natural state [145]. The time-bound natural state describes a historical state of ecosystems as it existed before human-induced "degradation". However, IPBES raises some concerns regarding this approach: First, it is difficult to obtain data on such a reference situation. And second, scientists and scholars do not agree on the time in history when human action became strong enough to cause "degradation" [145].The counterfactual, natural reference state can be described as a hypothetical ecosystem condition that would exist if there was no human impact. Again, this reference situation is problematic since this is only a hypothetical condition, collecting data and measuring the impact is therefore difficult. Furthermore, it is also possible that a reference situation changes naturally. In southern Finland, for example, the proportion of spruce is increasing and that of pine is decreasing. Hence, an area that was a pine forest 100 years ago would turn into a spruce forest under the conditions of today if it were left untouched.

The second approach for establishing a reference situation is the time-bound recent state approach. The chosen baseline is a state of the ecosystem that existed in recent historical times, e.g. 50 years ago. The ecosystem in question is then compared with the state of this most recent historical ecosystem. The advantage of this approach is that more data is available for actual measurements. The main disadvantage is that different regions and countries are treated unequally depending on their history: The ecosystems in the temperate zones of most developed countries as well as in tropical regions that were governed by European empires as plantation colonies were heavily degraded in Early Modern and Modern Times (ca.16th – 19th century). Therefore, the temporal reference of 50 years misses most loss of biodiversity. On the other hand, regions that have experienced a more recent economic "development" such as the temperate zones in the former Soviet Union and large parts of the tropics, come off badly accordingly. For the exception of the Greater Caribbean Basin and countries belonging to the former Soviet Union, this means that industrialized countries will have less impact on the quality of biodiversity than most developing countries [145]. An arbitrary temporal reference causes therefore postcolonial unequal treatment [146,147].

The third approach for selecting a reference situation is called space-for-time-substitution. Herein, ecosystems are identified and divided into analytical units, so-called plots, and afterwards the condition of those plots is assessed in respect to degradation. The different plots are then compared to each other in order to assess impacts. Plots are divided according to land use types and one area has therefore only one land use type. The degraded plots are compared with the relatively intact plots (e.g. typical primary vegetation in an 'area') of the same ecosystem. The differences between the two plots show the change in biodiversity quality [103]. Again, this approach is not flawless: for one, the identification of intact areas in one ecosystem and the subsequent selection of plots can vary depending on the criteria of the researcher. And second, as temporal dimension the current state is taken. Therefore, historical impacts are ignored. Still, there are several advantages to this approach as well: First, there is a large amount of monitoring data available on a global scale since the current state can be analyzed – this advantage cannot be stressed enough. Second, this is a standardized method in ecology for analyzing biodiversity at different organizational scales [105]. Third, the results can be validated empirically. Fourth, the data can be continuously updated and revised as new measurements and primary data become available. Revised data can be based on field data and monitoring data. And last, policy making is easier if the reference state is a current intact ecosystem, while decision makers might be harder to convince of an ecological state which existed in the last century/centuries.

2.2.1.3.3 Impact pathway and cause-effect chain

Many models which analyze the impacts in the LCIA phase use so-called cause-effect chains. Here, the Life Cycle Inventory (LCI) data is linked with the impact category, for example biodiversity, in order to estimate the environmental impacts. For example: The LCI provides the information on land occupation and land transformation that is classified and characterized to the midpoint category land use, which is linked in the damage assessment and normalization step to the endpoint category ecosystem quality [PDF m^2 a]. All endpoint categories can be grouped into one single index which is weighted to receive one common unit (see Figure 8).

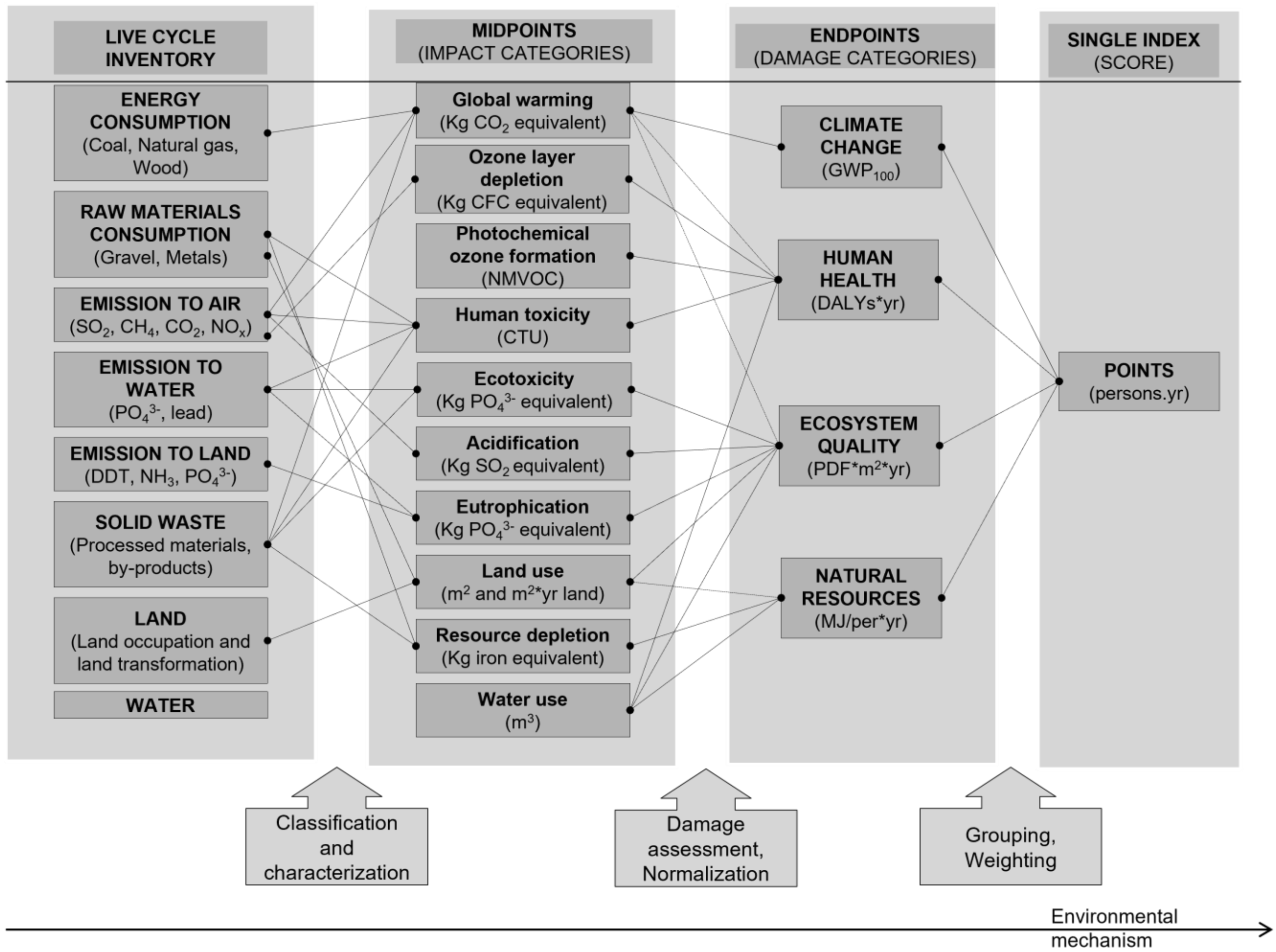


Figure 8: Impact pathway in LCIA, after Souza et al. [61]

As shown in Figure 8, cause-effect chains investigate how indicators influence the impact category at different stages. Often, these stages are either at midpoint or endpoint of the effect chain, for instance: an indicator at midpoint can be the “loss of species richness” while indicators at the endpoint often combine the results of various midpoint indicators. Such an assessment does not provide actual impacts of land use but rather an average approximation. With increasing size of the land use area within a plot and with increasing time of the occupation the factor is multiplied accordingly [61]. Despite the merits of the cause-effect chains, there are some pitfalls to this proceeding: First, the indicators at the endpoint are subject to great uncertainty due to the combination of various midpoint indicators and the high degree of complexity [61]. Second, usually only negative effects are depicted in the cause-effect chains for biodiversity impacts [5]. Yet, it is worth considering indicators that measure both a ‘pressure’ and a ‘relief’ for biodiversity in order to account and incentivize for positive land management.

2.2.1.4 General requirements for LCIA methods

The European Commission – Joint Research Centre [72] has formulated several general requirements and criteria for the development of impact assessment methodologies in LCA – although most requirements reflect general diligent scientific working. There are scientific requirements that must be met, such as the completeness of the scope. Herein, the Joint Research Centre stresses the importance of developing a method with a high level of spatial adaptability. This goes hand in hand with a specific validity of indicators based on geographical regions and different scales. In addition, a method should enable a comparison of products, materials and other processes in respect to their impacts. Herein, double counting should be avoided as far as possible. Also, all critical mechanisms influencing the impact categories, e.g. biodiversity, must be included in the method, as far as they are well understood. Another important aspect is the scientific robustness and reliability of the models, indicators and parameters used in the method. For example, the Joint Research Centre [72] stresses the need to use models and data that have been scientifically published and well-acknowledged by the scientific community. In addition, the method should reflect the latest scientific knowledge and it should be possible to update the method continuously as new research results become available. Where possible, the method should be validated and verified with monitoring and field data in order to make the impact assessment as accurate as possible. The scenario and model uncertainty should be considered and well described. Further requirements for a LCIA method development refer to a good model documentation, transparency of the results and uncertainties as well as the reproducibility of the results. Another aspect concerns the method's applicability and the acceptance by different stakeholders, including the communication of results to economic actors and policy makers. This goes hand in hand with a customer-tailored presentation of the results which are easily understandable for non-LCA experts [72]. Additionally, a method is only operational if it provides background data for the users. Herein, the ILCD differentiates between a foreground and a background system in LCA. For the foreground system primary data of a company is usually available and can be used, whereas the background system comprises processes that are not under the direct influence of a company (e.g. tier two or tier three supplier), therefore generic or average datasets have to be used, since no specific information on these processes is available [148]. Thus, the provision of a background dataset is of utmost importance for a method to be operational in LCA. All these requirements are to be considered in the herein presented development of a LCA method.

2.2.2 Administrative scale: decision support

As highlighted by the Joint Research Centre [72], the LCA is intended to evaluate different products, materials or production processes in comparison to alternatives in order to highlight those that have the least environmental impact over their entire life cycle [5]. This is usually done with the aim of optimizing existing products and processes. With regard to biodiversity, the herein proposed method should be globally applicable and able to quantify the impact of biodiversity at four different levels that revolve around the land use: the location in which the land use takes place, the concrete land use type, and the land use intensities applied as well as the landscape composition [5,138,142,149–151]. This four-level based information is not only important from an ecological and conservation perspective but also necessary at the administrative scale for LCA users, consumers and, in particular, decision makers in the economic and political sectors in order to optimize products with regard to biodiversity impacts.

These four levels of decision support are the pillars of the method in this thesis. Therefore, the following simplified example comparing cow's milk and soy milk is intended to illustrate the importance of the different levels of information for decision makers. The first information level refers to the location of the production or land use process. In this example this would be the grazing of cows in Brazil compared to soybean cultivation in Italy. Using this method, the location of the production site with a higher risk for biodiversity should be indicated by weighing up the species-rich Brazilian grazing site and the severely degraded ecosystem in which the Italian soy cultivation area is located. It is therefore indispensable that the method takes in a global perspective in order to provide such an analysis. This level of information is necessary for consumers who want to decide to buy a product from a country/region where the impact on biodiversity would be lower. For producers, the information is important because it enables them to shift the sourcing of their products to another country or region with a lower risk to biodiversity. If relocation is not possible, it is particularly important for landowners in such a risk area to make their land management as biodiversity-friendly as possible, which directly relates to the next relevant information level. This level of information relates to the question: What is produced and what kind of land use is part of the product? The method should be able to compare the biodiversity impacts of different land use types [138,142,150]. In this example the comparison of the land use type pasture in milk production and cropland for soy milk production. This information is essential for consumers when they decide whether it is more beneficial for biodiversity to buy products from one land use type (soy milk – land use type cropland) or another (cow's milk – land use type pasture). For producers, the information helps them to switch to resources from a land use type that is on average better for biodiversity. The third relevant information concerns the question of how a product was produced [150]. This includes, for example, the comparison of organic and conventional agriculture. This information relates to the intensity of land use and thus to specific land management parameters such as the amount of fertilizer applied. The method is intended to provide consumers with additional answers as to whether they would rather buy organic or conventional products. For landowners and companies, decision support is to be given on the following question: How would the impact on biodiversity change if they were to change their land management? The fourth relevant information relates to the question of whether it is better to farm some land intensively and conserve other land for biodiversity or to farm all land extensively by applying more biodiversity-friendly management practices.

In order to provide all these levels of information, the development of a hierarchical method is necessary. If the method were to account only the local field or patch scale, landowners who apply the land sharing strategy would no longer be encouraged to set-aside land for biodiversity conservation. The method should be able to consider all four levels and to quantify each level separately which can be aggregated into one final impact value. In addition, the method should be designed so that primary data can be used to make the results as accurate as possible while background data can be used when primary data are not available [5,150].

2.2.3 Summary of main requirements from a LCA perspective

In general, a method for assessing biodiversity in the life cycle assessment has to fit into the overall LCA framework and make use of the UNEP-SETAC framework in order to provide characterization factors. The land use flow list should be updated and cause-effect chains developed which are able to include positive biodiversity indicators. Additionally, it should be based on scientific evidence and data which can be constantly updated, and it has to be globally applicable both in a foreground and background system. Furthermore, a method should be able to assess where the loss of biodiversity is greatest and provide recommendations for LCA end users on how to mitigate the impact. The new method, therefore, has to provide decision support regarding the location of production, the comparison of different land use types and intensities as well as, the inclusion of management parameters. It furthermore should provide decision support to land owners on the landscape design including different land sharing and land sparing strategies. Therefore, the development of a hierarchical method is required that is able to provide separate results for each of the decision options. To make the method user-friendly, it should also be possible to add the individual results to one indicator so that complex system can be displayed in an understandable way.

2.2.3.1 Derivation of a catalogue of requirements for biodiversity impact assessment methods in LCA

Based on the herein described state of knowledge in the different disciplines of ecology and nature conservation as well as LCA, a catalogue of requirements is derived for the development of a biodiversity assessment method in LCA. This catalogue of requirements is used in order to assess the already existing LCIA methodologies and to highlight research gaps in the following chapter. Furthermore, based on the requirement catalogue, the new BioMAPS method is being developed and evaluated.

Requirements from the disciplines of ecology and nature conservation:

1. Biodiversity as multi-scale concept requires the development of a multi-scale assessment method.
2. The method should include different spatial scales (e.g. global, regional and local).
3. The method should include different organizational scales and indicators at the levels of ecosystems, species and genes (basket of indicators).
4. It should give recommendations at different administrative scales which represent in this thesis the various end users of a LCA (e.g. governments, companies, land owners, producers and consumers).
5. At the global scale ecological and conservation indicators should be used that account for the global distribution of biodiversity as well as the different levels of vulnerability and irreplaceability of biodiversity.
6. At the regional scale, the landscape matrix should be included enabling the assessment of land sharing and land sparing conservation strategies.
7. At the local scale, different biodiversity impacts due to the types of land use, land use intensities and land management parameters should be accounted for.

Requirements from the LCA discipline:

8. The method has to fit into the overall LCA framework and should follow the standardized structure of a LCA.
9. It should adapt the Life Cycle Inventory and refine the land use flow list to a multi-scale assessment.
10. It should be based on the UNEP-SETAC framework and provide characterization factors.
11. It should develop a cause-effect chain that includes “relief” indicators that are beneficial for biodiversity.

12. It should provide decision support to LCA end users regarding the location of production, the comparison of different land use types and intensities as well as, the inclusion of management parameters.
13. It should provide decision support to LCA end users regarding land sharing and land sparing strategies.
14. It should be based on scientific data which can be constantly updated.
15. It has to be globally operational both in a foreground and background system.

The state of fulfillment of these requirements will be evaluated in chapter 7.

3 Review of LCIA biodiversity methodologies and research gaps

A comprehensive literature review must precede the development of a LCA method in order to analyze the current state of the art and to identify research gaps. Therefore, in the following chapter 3.1 LCIA methods are being reviewed with regard to the catalogue of requirements outlined above. Based on the review of these LCIA biodiversity methods, further specific requirements are derived and added to the catalogue of requirements. Since Maier et al. [5] have published a review of the existing methodologies as part of this dissertation in 2019, this section will update the review including the most recent publications and will provide a summary of the most important results.

3.1 Organizational scale in LCIA methods

Since all LCA methods which assess biodiversity are based on concepts and tools from the fields of ecology and nature conservation, the LCIA methods herein identified have similar research gaps with regard to the inclusion of the organizational scales of biodiversity [5]. According to Winter et al. [60], there is still no LCA method that considers all three organizational levels of biodiversity [60]. Many methods concentrate either on the level of species or on the level of the ecosystem, and none include phylogenetic diversity [5]. Phylogenetic diversity refers to the evolutionary relationship between species – and thus the richness of a multi-species gene pool [5]. The preservation of phylogenetic diversity not only offers greater potential for adaptation to global change [152] but it is also most likely to enhance ecosystem functions, as close-related species are more likely to fill similar niches and thus to perform similar ecosystem functions [153,154]. While increasingly more ecological studies consider phylogenetic aspects, this process is still in its infancy in global land use assessments and LCA [5,152]. However, given the unpredictable effects of the ongoing global crisis, the preservation of the evolutionary potential of species is the most effective "insurance" to enhance adaptability [5,155]. The different ecological and conservation indicators used in the LCIA methods are further elaborated in the next subchapter, depending on the related spatial scale of assessment.

3.2 Spatial scale in LCIA methods

In this chapter the inclusion of different spatial scales in existing biodiversity LCIA methods is being reviewed.

3.2.1 Global biodiversity risks in LCIA methods

In order to assess global biodiversity risks, the LCIA methods take up the concepts of vulnerability and irreplaceability. However, LCA practitioners do not make use of the diverse range of already existing global conservation schemes, they assess different organizational levels of biodiversity and use different conservation indicators at a global scale [5]. Global species richness is the most commonly used metric for LCIA methods, followed by an assessment of rare habitats or ecosystems [5]. However, as Veach et al. [156] highlighted in their study, if only one single indicator such as species richness is used as a criterion for global biodiversity distribution, a lot of coverage of biodiversity is lost. Few LCIA methods include the presence of endemic species as an indicator of irreplaceability, while none of them accounts for key conservation areas, such as the Important Bird Areas, Alliance for Zero Extinction sites or Centers of Plant Diversity. Other factors such as areas with major migratory routes or high concentrations of species have also been neglected in the LCA to date [5], although they are emphasized as critical for the preservation of biodiversity [157] and are included in the Global 200 Ecoregion Conservation Scheme accordingly [86]. In addition, no method considers global areas that are important for phylogenetically different species and genetic traits [5], as highlighted in the EDGE areas [158]. The priority setting of LCIA methods and conservation science differs greatly. While conservation scientists mainly focus on areas with endemic species (irreplaceability), the majority of LCIA methods emphasizes vulnerability indicators and the minority of methods take endemism into account [5].

Although ecologists and conservationists have different conservation priorities than the developers of LCIA biodiversity methodologies, there is some similarity between the two fields. Both attempt to quantify biodiversity and highlight the most important areas for biodiversity conservation [5]. Biodiversity conservationists strive to prevent the loss of biodiversity, while life cycle managers try to quantify the impact of a product throughout its life cycle in different regions of the world in order to provide recommendations for biodiversity-friendly production and consumption. As we have seen, there is a discrepancy in the LCA between what is considered worthy of protection and what is not. Therefore, each LCIA method gives different recommendations on where resources should be extracted. Due to these differences, important risk areas of biodiversity are not yet sufficiently considered and mapped in LCIA methods. To date, there is no LCIA method that harmonizes the different research results from nature conservation in order to identify critical global biodiversity areas and integrate regions of irreplaceability as well as high and low vulnerability [5].

3.2.2 Regionalization of global biodiversity risks in LCIA methods

Most LCIA methods are capable of regionalizing global biodiversity risks. They use the results of their specific ecological or conservation indicators and aggregate the risks per spatial unit of analysis, such as grid cell, region, ecoregion, country level or biome. However, no method assesses the location where the actual land use takes place by using global land use models, instead they rely on data for the overall national or (bio-) regional average [5,140]. This generic use of data clearly carries the risk that some areas may be over- or underestimated with regard to biodiversity risks in a specific region [5]. Furthermore, only a limited number of methods is yet able to regionalize biodiversity risks at a global scale. Most of them focus only on individual and specific regions [5]. In addition, several risk areas of biodiversity are not yet considered in the regionalization of land use because, as emphasized in the previous subchapter, all methods concentrate only on specific ecological or conservation indicators of biodiversity (e.g. reactive vs. proactive approaches) [5]. Moreover, no method takes into account the occurrence of land use sites in areas of high value for the protection of biodiversity (e.g. in AZE sites), even though sufficient research results and data from nature conservation sciences are available. Furthermore, it has not yet been investigated how temporal changes in land use (transformation from and to) affect global biodiversity risk areas by using for example land use suitability maps or historical land use models [5], except for Chaudhary et al. [58] who analyze the loss of terrestrial vertebrates. Conservation scientists emphasize the role of companies in the conservation of such critical biodiversity areas, thus underlining the need to integrate such areas into a life cycle assessment [159].

3.2.3 Land use impacts at local scale in LCIA methods

As described in Maier et al. [5], most of the biodiversity methods are capable of differentiating between various types of land use and of evaluating their local impacts on biodiversity at a patch or field level. However, some of the methods only assess specific types of land use such as forestry [36,160], cropland [35,39,44,47], mining [34] or they do not take land use into account at all [46,49]. Since they are not able to compare the impacts of different resources from different types of land use, they do not sufficiently meet the requirements of the LCA.

Depending on the choice of indicator, LCIA methods use very different reference situations to assess the effects of land use types. This can be, for example, the changes in species richness under the specific land use type compared to primary vegetation. Other methods use the degree of naturalness of an ecosystem or the potential natural vegetation and hemeroby as a reference. Sometimes abstract values are also used as a reference, such as the maximum potential of biodiversity, which is defined by a set of conditions that must be met to achieve the highest quality of biodiversity [36,46,55]. Other methods suggest using policy objectives as a kind of reference to compare impacts with a target [161]. However, as IPBES clearly declares, policy objectives should not be used as a reference because they are too strongly influenced by social and economic factors and vary according to case and region [145,162].

Together with the reference situation, the ecological indicator used to quantify local impacts varies. Several LCA methods analyze the impact of land use on biodiversity using the species richness indicator [28,31,34,35,40,44,163]. Herein, a prominent ecological model that is based on the indicator species richness and which can predict the number of species lost due to land use is the Species Area Relationship (SAR) model as defined by Arrhenius [164]. This model has been taken up by a number of LCA methods [32,33,37,40,45,165]. The SAR model assumes that the number of species increases with the size of an area [164]. This is because larger areas allegedly inhabit more, and also more diverse habitats. As larger areas host more species, the probability of species formation over time is higher and the risk of extinction is lower due to the larger species pool [20]. Chaudhary et al. [45] for example measure the potential global extinction of five different taxa that depend on six different land use types and all ecoregions [166] of the world based on SARs. However, there is also criticism of the use of SARs to predict extinction rates. Henle et al. [62] claim that the species-area relationships vary greatly depending on the region on which they occur and the taxa that are affected. Furthermore, it was shown that the actual extinction rates were over- or underestimated on the basis of SARs [102,167,168]. This is mainly due to the fact that predictions based on SARs do not take into account specific habitat affinity of different species. They assume that the conversion from *Naturland* to anthropogenically used areas will ultimately lead to the extinction of all species in this area. But, as research has clearly shown, species could also live in anthropogenically altered habitats [169].

The countryside Species Area Relationships (cSARs) model is a modified version of the Species Area Relationship (SAR) model. cSARs account for the species-specific use of habitats, that have been modified by humans and predict that species which have adapted to such habitats survive even in the disappearance of their native habitat [168]. The cSAR model provides a measure to incorporate both the effect of species persistence on a plot and the proportion of species extinctions in an area. Hence, cSAR provides a more realistic picture than the original SAR which only measures the sensitivity of species richness to changes in the native habitat [102]. The cSAR model takes into consideration the different habitat use by various species groups. The richness of each species group is determined by a function of the area of each habitat in the environment. The proportion of remaining species depends on their affinity to habitats modified by humans [102]. This model is particularly suitable to describe diversity patterns in multi habitat landscapes even when the original habitat cover or species composition is not known. Proenca et al. [170] and Guilherme & Pereira [171], have shown that the performance of cSAR is better than the classic SAR in describing bird and plant diversity in such landscapes [169]. This model is also used in LCA for example by coupling cSARs with specific habitat affinities of endemic bird species and with economic input-output models [172]. The authors calculate global characterization factors for land use data sets of 13 agricultural commodities and the proportion of bird species that remains after a land use change.

However, the deployment of SAR-based methods to measure the loss of biodiversity is not without its drawbacks. As pointed out by Fattorini & Borges [167], SARs do not address the "indirect effects on biodiversity" resulting from the loss or degradation of habitats and may therefore not capture the full impact of land use. Furthermore, the models of SAR and cSAR are taxa-specific due to the different habitat affinities. This means that the habitat affinity of one taxon might not be directly transferred to another. This process is also known as cross-taxon surrogacy and its implementation in LCA methods has been criticized by Souza et al. [61].

Among the indicators at the ecosystem level used in LCA methods are i.a. hemeroby. Hemeroby is a measure of human influence on ecosystems and was first described by Kowarik [173]. Similar to the hemeroby concept is the concept of naturalness. While hemeroby describes how similar the ecosystem is to the natural state, naturalness focuses on the distance between the state of the ecosystem and a natural state. Both concepts are thus based on the measure of naturalness, but point in the other direction (distance versus similarity) [51]. The hemeroby levels of an area describe the state of an ecosystem in terms of land use and can therefore be used to characterize different types of land use. The degree of hemeroby depends on the degree of human influence that prevents the system from developing into a (quasi-)natural stage [173]. The hemeroby concept makes it possible to estimate the impact of a particular type of land use on the degradation of an area's naturalness and thus is a conservation indicator at the ecosystem level. Hemeroby is an integrated, descriptive measure of various human influences that prevent a system from developing into a situation without anthropogenic influence [174]. However, Baitz [175] criticizes the use of hemeroby in the LCA because of its focus on *Naturland*. Herein, *Naturland* receives the highest biodiversity value. Yet, it has been shown that some ecosystems with secondary vegetation can have an even higher number and abundance in species than primary vegetation [105]. The hemeroby concept classifies an area according to its proximity to its "natural state" using a scale. A “natural” area has a high value, while a degraded area has a low hemerobic value. The assessment of areas and their classification into hemerobial values is not standardized and clearly defined but depends on the subjectivity of the method developer. Among the LCA methods that include the hemeroby concept are Coelho & Michelsen [43], who formulate conditions for biodiversity conservation (CMB) while linking them to the hemeroby classes, as well as Lindner et al. [55]. Brentrup et al. [30] and Fehrenbach et al. [52] use the hemeroby concept in the LCA of land use in European biogeographical regions, while Rossi et al. [160] in respect to forestry practices in boreal forests. Cote et al. [51] developed a conceptual model for forestry practices in boreal forests in Canada using the naturalness concept.

There are two further concepts in ecological studies that provide a set of environmental variables and conditions to make predictions of a species’ fitness and survival or a species’ occurrence in a certain area [176]. The first one is called niche theory [177,178] and the second Habitat Suitability models (HSMs) [176]. According to Hirzel & Le Lay [176] Habitat Suitability models can be seen as the application of the niche theory since a set of conditions is used in order to make predictions on a species or taxa absence or presence in a given area. HSMs have been first applied in a biodiversity assessment in LCA by Baan et al. [47]. They use habitat suitability models based on the potential geographical distribution of species, the minimum and maximum range of the occurrence area, the threat levels and the rarity (e.g. endemism) of taxa. However, they also use cross-taxon surrogates by investigating only the habitat suitability of mammals that differ from other taxa. Amphibians, for example, are not included although they belong to the most threatened group of the IUCN Red List [179].

Other LCIA methods use an equivalent concept as the Habitat Suitability Models called Conditions for Maintained Biodiversity (CMB) [36]. Herein, a set of criteria or conditions for biodiversity are defined that must be met in order to receive the highest biodiversity value. Coelho & Michelsen [43] used the hemeroby concept as a proxy value for CMB. Another approach, which is based on Michelsen [36], was developed by Lindner [46] and further developed by Winter et al. [49] and Lindner et al. [55] using expert interviews in order to identify the best “criteria” for biodiversity. Yet this approach is not in line with the recommendation of the JRC to use scientifically published data that can be validated with monitoring data [72].

As has been shown, many models and indicators derived from ecology and conservation have been used in LCA methods for the impact assessment on biodiversity at the local scale. In summary, there is no consensus on the most promising approach for a biodiversity assessment in LCA and whether a common methodological framework is possible. Limitations of existing LCIA methods show a desideratum. In their literature review, Souza et al. [61] give a comprehensive overview of the ecological models and indicators used in LCA. They found that most methods focus on indicators that measure the impact on the number of species (species richness) and related models such as SARs and cSARs. They are generally specific to certain species or taxa or apply to particular land use types only [5]. Additionally, some methods at the ecosystem level are not empirically validated and sometimes subjective, which contradicts the recommendations of the ILCD [72].

Based on the review of the indicators and methods in this thesis and in the research report of Souza et al. [61] there are several aspects that are usually lacking in current methods and that should be considered in the future:

- the ‘quantification’ of biodiversity must be clearly defined, thorough, transparent and coherent while subjectivity of the results should be avoided,
- methods should be made globally operational,
- the classification of land use and land cover should be improved and spatial information through the coupling of GIS and LCA should be integrated,
- cross-taxon surrogacy should be avoided,
- the use of only one biodiversity metric might not be sufficient to assess biodiversity as a whole,
- the overestimation or underestimation of the biodiversity impacts of some land use types should be avoided.

3.2.4 Land use intensity impacts in LCIA methods

As shown in Maier et al. [5] Some LCIA methods make it possible to quantify the impact of land use intensity on biodiversity. However, most of them are only capable of assessing a few intensity classes such as low and high intensity or extensive and intensive farming. No globally operational method is yet capable of providing continuous intensity variables for the different types of land use [5]. This is a disadvantage, as Armengot et al. show in their study [180]. Herein, they illustrate that using an intensity gradient of different management parameters provides more reliable results.

There are only a few methods that are capable of assessing specific land use parameters and their impact on biodiversity [5]. These methods are usually very data intensive or only apply to a specific type of land use or to a particular region [181]. Therefore, there is no globally operational method that provides information on land use intensity derived from land management practices for all land use types. In addition, specific management parameters are rarely combined with continuous land use intensity variables [5]. Some methods evaluate management parameters and relate them to a kind of biodiversity metric by assuming any form of relationship. They are susceptible to uncertainty because the results depend on the assumed relationship between land management and biodiversity impacts. As Kleijn et al. point out, this relationship has not yet been fully understood in ecological research [182]. The inclusion of land management parameters in the LCA method is particularly important for the decision support of landowners [183]. Landowners usually cannot simply move the location of their plots or change the overall land use system to improve their negative impacts on biodiversity or even have positive impacts. But by adapting the farming methods they use to cultivate their land, they have a significant opportunity to directly influence local biodiversity [5].The inclusion and quantification of land management parameters in the LCIA methods is also valuable for companies, as they can make recommendations for their suppliers for management improvements and quantify the corresponding impact [5].

3.2.5 Landscape perspective at regional scale in LCIA methods

So far, the landscape perspective has been hardly addressed in any of the LCIA methods [54,184]. There are some valuable approaches that concentrate on certain aspects of the landscape’s spatial configuration such as fragmentation [53,54] or the perforation potential of Coelho et al. [57]. Yet, as highlighted by ecologists habitat loss, regardless of its spatial configuration, is primarily responsible for the subsequent biodiversity impacts [122]. Thus, especially the landscape composition and the availability of primary and secondary natural habitat is of highest importance for biodiversity conservation. Until now, however, there is no method which includes the overall landscape composition, including the last remnants of primary and secondary vegetation within a landscape and the landscape intensity. This goes hand in hand with a lack of consideration of the different land sharing and land sparing strategies in existing methods.

3.3 Summary of main research gaps and research objective

As has been highlighted in this chapter, there are still considerable research gaps regarding the fulfillment of the requirements from the ecology, nature conservation and LCA disciplines. These research gaps are further summarized in Table 3, where the requirements from the disciplines are listed for each method. This table also illustrates whether it meets them or not. As depicted, there is still no method which can conduct a globally applicable assessment of biodiversity which includes the different organizational scales of all three organizational levels of biodiversity (the genetic, the species and the ecosystem level) and the three spatial scales (global, regional and local). As it can be seen in Table 3 all methods either focus on one or two organizational scales, and on one or two of the spatial scales. Since they focus on individual parts for specific requirements they are only able to map individual aspects of biodiversity.

There is still no method that considers and researches the multiscality of biodiversity in its entirety and through which we can derive detailed information for decision makers. None of the methods fully meet the requirements while also being able to provide the required background data for the assessment of global value chains.

Table 3: Summary of LCIA methods and research gaps based on requirements from ecology, nature conservation and LCA further developed from Maier et al. [5]

Author	Organiza-tional scale			Global scale			Local scale			Regional scale		Opera-tional
	Genetic	Species	Eco-systems	Biodiversity pattern	Pro-, reac-tive, irre-placeability (p,r,i)	Regionaliza-tion	Land use types	Intensity gradient	Manage-ment param-eter	Landscape aspects	Landscape composition	Background data
[28]	-	x	-	-	i	-	x	-	-	-	-	-
[29]	-	x	-	x	i	x	x	-	-	-	-	x
[30]	-	-	x	-	p	-	x	x	-	-	-	Eu-rope
[31]	-	x	x	-	i	-	x	-	-	-	-	-
[32,33]	-	x	-	-	r,i	-	x	-	-	-	-	-
[34]	-	x	x	-	r,i	-	Only min-ing	-	-	-	-	-
[35]	-	x	-	-	r,i	-	Gras sland , cropl and	-	x	-	-	-
[36]	-	-	x	x	r,i	x	Only for-estry	x	x	-	-	-
[37]	-	x	x	x	r,i	x	x	-	-	-	-	x
[38]	-	x	x	-	r	-	x	-	-	-	-	-
[163]	-	x	-	-	i	-	x	-	-	-	-	-
[39]	-	-	x	-	r,i	-	Only cropl and	x	x	-	-	-
[40]	-	-	x	x	r,i	x	x	-	-	-	-	x
[41]	-	x	-	x	r	-	x	-	-	-	-	x
[42]	-	x	-	x	r	x	3 land use types	-	x	-	-	-
[43]	-	x	-	-	r,i	-	x	x	-	-	-	-
[44]	-	-	x	-	r	x	Only cropl and	Inten-sities for cropl and	-	-	-	x
[45]	-	x	-	x	r,i	-	x	-	-	-	-	x
[46]	-	x	-	-	i	x	x	x	x	-	-	-

Author	Organizational scale			Global scale			Local scale			Regional scale		Operational
	Genetic	Species	Eco-systems	Biodiversity pattern	Pro-, reactive, irreplaceability (p,r,i)	Regionalization	Land use types	Intensity gradient	Management parameter	Landscape aspects	Landscape composition	Background data
[47]	-	x	x	x	r,i	x	Only cropland	-	-	-	-	-
[48]	-	x	-	-	i	-	x	x	-	-	-	-
[58]	-	x	-	-	r	x	x	-	-	-	-	x
[49]	-	x	-	x	r,i	x	x	-	x	-	-	-
[50]	x	x	-	x	r	x	x	x	-	-	-	x
[160]	-	x	x	-	p	-	Only forestry	x	-	-	-	x
[51]	-	-	x	-	p	-	Only forestry	x	x	-	-	-
[52]	-	-	x	-	p	-	Forestry, cropland	x	x	-	-	-
[53]	-	-	x	-	-	-	Only mining	-	-	x	-	-
[54]	-	x	x	-	r,i	-	-	-	-	x	-	-
[55,185]	-	x	x	x	r,p	-	x	x	x	-	-	-
[56]	-	x	-	x	r	-	-	-	x	-	-	-
[57]	-	-	-	-	-	-	-	-	-	x	-	x
[59]	-	x	-	x	r,i	-	-	-	-	-	-	-

Therefore, the aim of this dissertation is to develop such a new method for a Biodiversity Multi-scale-Assessment of Product Systems (BioMAPS). It takes up geoscientific techniques that investigate land use and land use change using GIS and remote sensing. Note that it focuses exclusively on land use processes and their impacts on terrestrial biodiversity. Secondary effects, such as impacts due to climate change, are not the subject of this dissertation.

Furthermore, a new life cycle land use flow list will be provided that is required for the LCIA phase. The analysis of occupation and transformation flows will be based on global land use models, as well as global average values for management parameters required for the background system.

To this end, the BioMAPS method is being developed to meet the following requirements and to address the research gaps highlighted in Table 3, based on the catalogue of requirements, as summarized below.

Ecology and conservation science:

1. Development of a multi-scale assessment method for the holistic assessment of biodiversity.
2. Inclusion of different spatial scales (e.g. global, regional and local).
3. Inclusion of different organizational scales and indicators at the levels of ecosystems, species and genes (basket of indicators).
4. Be able to give recommendations at different administrative scales, the various end users of a LCA (e.g. governments, companies, land owners, producers and consumers).
5. Inclusion of ecological and conservation indicators that account for the uneven global distribution of biodiversity as well as the different levels of vulnerability and irreplaceability.
6. Inclusion of the landscape matrix enabling the assessment of land sharing and land sparing conservation strategies.
7. Inclusion of different biodiversity impacts due to the types of land use, land use intensities and land management parameters.

LCA discipline:

8. Be in line with the overall LCA framework.
9. Adaptation of the Life Cycle Inventory and the land use flow list to a multi-scale assessment.
10. Be in line with the UNEP-SETAC framework and provide characterization factors.
11. Development of a cause-effect chain that includes “relief” indicators that are beneficial for biodiversity.
12. Provision of decision support for LCA end users regarding the location of production, the comparison of different land use types and intensities as well as, the inclusion of management parameters.

13. Provision of decision support for LCA end users regarding land sharing and land sparing strategies.
14. Use of scientific data which can be constantly updated.
15. Be globally applicable both in a foreground and background system.

LCIA biodiversity methods:

16. Cross-taxon surrogacy should be avoided by including data on several taxa.
17. Use of land use models to assesses the location where the actual land use takes place for the regionalization.
18. Include effects of land transformation on biodiversity risk areas by using for example land use suitability maps or historical land use models.
19. Definition of the ‘quantification’ of biodiversity, shall be thorough, transparent and coherent while subjectivity of the results should be avoided.
20. Improvement of the classification of land use and land cover through the use of spatial information by coupling of GIS and LCA.
21. The use of more than one biodiversity metric is recommended.
22. The overestimation or underestimation of the biodiversity impacts of some land use types should be avoided.
23. An intensity gradient instead of intensity classes which are linked to different management parameters should be included.

4 The BioMAPS method

The following chapter describes the BioMAPS method which integrates biodiversity impacts in LCA while also accounting for the requirements described in the previous chapters. Herein, a detailed investigation of available methods, concepts as well as indicators is conducted.

To support LCA end users in their decisions on multiple spatial scales, a method must be developed that provides separate results for each of the scales. Therefore, the method will be modularly structured according to the three spatial scales: global, regional, local.

On a global scale, the LCA end user should receive information about the regions or countries from which resources or products should be preferably sourced. Since it is especially important to consider proactive and reactive areas as well as the factor of irreplaceability, the method analyzes and classifies the location and the number of different conservation sites based on the approach of Brooks et al. [76]. The BioMAPS method furthermore analyzes the unequal distribution of biodiversity and considers the different taxa as well as organizational levels. Due to the complex supply chains, we often do not know where exactly land use takes place. Hence, land use models are used to assess the estimated location of land use areas in global biodiversity conservation areas. In doing so, the method uses the global land use models [186], which indicate the locations of land occupation and transformation, this step is called regionalization of global biodiversity risks.

At the local scale, the main objective is to provide decision support for cultivation practices and land management. Herein, the relative decline in biodiversity can be compared with a local reference situation. In order to be able to draw conclusions about land use related local changes in biodiversity, the individual land management parameters are quantified with the help of a land use intensity index (LUI) and coupled with biodiversity metrics of the local level, such as the reduction of species richness due to the intensity of land use in a field. In addition, the BioMAPS method analyzes impacts on as many taxa as possible to avoid cross-taxon surrogacy by using the research results on different biodiversity metrics of Newbold et al. [104,105] based on the PREDICTS database [187]. The global land use intensity indices are calculated based on the concept of Herzog et al. [188] and Erb et al. [189] using global databases and land management statistics, such as from the FAO and UNECE [190–197]. The management parameters for calculating the LUIs are determined on the basis of the Conservation Evidence database [198], which collects scientific evidence for biodiversity-effective management activities. The LUIs are coupled with the biodiversity metrics of Newbold et al. [104,105] of PREDICTS database [187] in order to translate the intensity values into local biodiversity impact values.

On a regional scale, the design and management of an entire landscape is of particular importance. Here, the different decision makers are faced with the question whether it is better to use a field intensively and leave more patches of natural vegetation in the landscape, or whether its larger field should be cultivated with low intensity, which would be more biodiversity friendly but leave less area for natural vegetation. To support these decisions, the BioMAPS method examines the landscape composition, which includes the proportions of each type of land use, as well as the remaining proportion of natural areas in the landscape. Furthermore, it examines the land use intensity of each patch in the landscape and the influences on regional biodiversity of the different landscape management options. As with the local level, it is important to provide information on regional biodiversity impacts for the supply chain. Therefore, information on the composition of landscapes from global land use models are used and combined with the herein calculated global intensity information of the individual land use types. These global intensity maps are then used to calculate global landscape development indices (LDI) by adapting the approach of Brown & Vivas [199].

As a final step, the BioMAPS method offers the option to aggregate the individual results into a dimensionless single point result, if no further detailed information is required. For example, if only the question about the selection of either product A or B is to be answered.

The following subchapters describe first the adaptation of the Life Cycle Inventory to fit the modular approach of the method. In the subsequent subchapters the development of the method is being described for the global, regional and local scales in more detail. This includes the underlying models, databases, methods and the calculation rules. The calculation procedures in GIS, which make the method operational for global value chains, is described in chapter 5.

4.1 Adaptation of Life Cycle Inventory for multi-scale biodiversity method

Since a multi-scale method is being developed, adjustments of the LCI structure and the land use flow list must also be made in the Life Cycle Inventory phase, as further information is now required for the impact assessment on all three spatial scales.

4.1.1 Adaptation of overall LCI framework

The Life Cycle Inventory is now structured according to the global, local and regional level. At the global level, information on the location of land use must be provided in order to calculate a global risk factor for biodiversity. This risk factor accounts for the three different organizational scales of ecosystems, species and genes as well as for their vulnerability and irreplaceability. At the regional scale, the landscape composition is described as the proportion of patches of all land use types and their intensities. At the local scale, the land use type of the field or patch is described. Here, more detailed information can be used, if available, such as information on individual management parameters or intensity values. In addition, as it is already common practice in the LCI framework for land use, information about the area required for a specific process of the product per functional unit is used [132]. In respect to the temporal aspect, the time scale comprises the time of land use. This is usually modelled in the life cycle assessment over a period of one year during occupation and is measured in area-time. The information of the LCI is directly connected to the LCIA where, each of the three spatial scales provides decision support according to the impacts on biodiversity; this data can either be considered individually or combined into one final biodiversity score (see Figure 9).

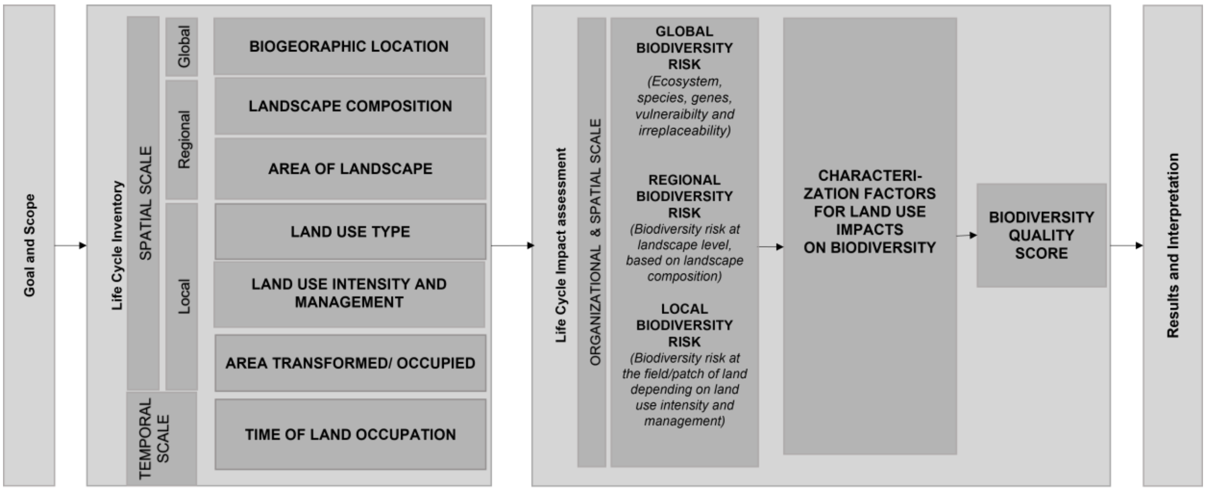


Figure 9: Life cycle Inventory of the BioMAPS method after Mila i Canals et al. [133] adapted by Taelman et al. [132]

4.1.2 Provision of new land use flow nomenclature

Part of the Life Cycle Inventory phase is the definition of all land use flows that are involved in the life cycle of the product. Since the overall LCI framework has been changed including the landscape composition and the area of land use patches at the regional scale, as well as the land use intensity and management parameters at the local scale, also a new land use flow list has to be provided that is able to reflect the characteristics of the multi-scale impact assessment method. For this method a land use nomenclature is proposed which is based on flow properties that define a land use type. Herein, also the land use flows are structured hierarchically as it is done in Koellner et al. [138]. Yet, instead of using separate flow names, the land use flows are characterized by so-called flow properties, such as a land use type specific set of management parameters, the land use intensity and its geolocation (see Table 4). The set of land use type specific management parameters, the intensities depending on the geolocation are further defined and calculated in a GIS environment in chapter 5 since they are directly connected to the LCIA method.

Table 4: Land use flow nomenclature based on flow properties

Level 1	Level 2	Flow properties			
Land use type	Land use subtype	Land use intensity	Land use intensity value	Management parameters	Geolocation
Cropland	C3 nitrogen-fixing	Minimum	0.2	E.g. not fertilized, low mechanization	E.g. Finland
Pasture	Rangeland	Light	0.5	E.g. medium livestock intensity, no pesticides, high mechanization rate	E.g. Iceland

The first two levels of the flow nomenclature (land use type and land use subtype) resemble those of Chaudhary et al. [50] and are based on the land use impact model of the PREDICTS database of several authors [105,187,200,201]. Furthermore, the new nomenclature is based on the land use model of Hurtt et al. [186]. The land use model of Hurtt et al. [186] distinguishes between 12 different land use classes and subclasses, whereas the PREDICTS database distinguishes only between six broad land use classes divided into the intensity classes of minimum, light and intense land management. In order to match the land use classes of Hurtt et al, [186] with the PREDICTS database and the ILCD land use flows, some of the subclasses are aggregated to a higher level of land use class: e.g. the land use model of Hurtt et al. [186] provides the land use subclasses *managed pasture* and *rangeland pasture*, that are aggregated to the overall land use class *pasture*. The land use subclasses *primary vegetation forested* and *secondary vegetation forested* are aggregated to the overall land use class *forest*. All five subclasses of crop types *C3 and C4 annual (C3 ann, C4 ann), C3 and C4 perennial (C3 per, C4 per) and C3 nitrogen-fixing (C3 nfx)* are aggregated to the overall land use class *cropland*. As the land use model of Hurtt et al. [186] is used for regionalization, the definition of their land use classes is mapped to the definition of the land use classes of Hudson et al. [187], whose data is used for the local impact assessment.

The following land use classes are used in the PREDICTS database [105,187]:

- primary vegetation = no evidence of prior destruction of vegetation, e.g. natural forest or natural grassland,
- secondary vegetation = vegetation that recovers after the destruction of the primary vegetation e.g. secondary forests or secondary grasslands,

- plantations = land that people have planted with tree plants and orchids such as oil palms, rubber, fruit and coffee,
- cropland = land that people have planted with herbaceous plants, even if these plants are fed to farm animals after harvesting,
- pasture = land on which livestock are known to graze regularly or permanently, or
- urban = areas with human dwellings, areas used for human residence and/or buildings including infrastructure such as streets.

These land use classes are already quite compatible with the land use classification of Hurtt et al. [186]. The land use class *plantation* is an additional class in the PREDICTS model that can be mapped to the land use class *cropland* with the sub classes *C3 and C4 perennials*. Planted native forests for wood and timber production, however, are mapped to the land use subclass *secondary vegetation forested*. The land use flows of the BioMAPS method expand the classification of Chaudhary et al., Hudson et al., Newbold et al. [50,105,187,200,201] and Hurtt et al. [186]. It follows a flow property based approach distinguishing between the broad land use classes (e.g. cropland, pasture), land use sub classes (e.g. rangeland pasture, managed pasture) and their flow properties land use intensity classes (e.g. minimum, light, intense) as well as land use intensity gradient values (ranging from 0 to 1). Where the intensity value is directly derived from the specific set of land management parameters and the geolocation. Thus, this new property based nomenclature allows for the use of a finite number of land use flows with infinite property combinations in the impact assessment.

The land use classes and sub classes used for this method are shown in Table 5 with a short definition as well as some examples. The land use flow nomenclature, including the intensity intervals, intensity values and the management parameters, is matched to the current ILCD land use flows in order to facilitate the integration of this method in existing LCA software and the LANCA model [139,143,144], that makes use of the ILCD flow list [138]. It is depicted in Annex I.

Table 5: Land use types and subtypes

Land use type adapted from several authors [50,103,186,187]	Land use subtype adapted from Hurtt et al. and Chaudhary et al. [50,186]	Definition after PREDICTS [103,187,200]	Examples
Primary vegetation		Native vegetation that has never been completely destroyed by human actions or by extreme natural events.	Amazon rainforest, tundra, taiga
	Forest	Primary forest areas	The tropical rainforests of the Amazon region belong to the largest existing primary forests. In the temperate zones there are still large areas of primary forests in North and Central America and Russia. Poland has (still) one of the last primary forests in Europe.
	Non-forest	Primary non-forest areas	Primary grasslands, savannahs, tundra, desserts, wetlands etc.
Secondary vegetation		Vegetation in former primary vegetation areas and recovering natural vegetation in other land use areas.	
	Forest	Forested areas with secondary vegetation, timber plantations also fall under this category, they are only classified as secondary vegetation, if the tree species are native to the area, often intensively managed for wood production, including monoculture.	Black forest, plantations of pine, eucalyptus or cork oaks
	Non-forest	Secondary vegetation that is not a forested area.	Secondary grasslands, meadows etc.
Pasture		Pasture is an area where farmers keep livestock for grazing.	
	Managed pasture	Areas used for the cultivation of modified, domesticated fodder crops for livestock. These have been planted with seeds, usually of alien species or, in occasional cases, with indigenous plants, often intensively cultivated with agronomic methods and under livestock control.	Managed pasture includes planted native or introduced vegetation such as tall fescue or switch grass.

Land use type adapted from several authors [50,103,186,187]	Land use sub-type adapted from Hurtt et al. and Chaudhary et al. [50,186]	Definition after PREDICTS [103,187,200]	Examples
	Rangeland pasture	Lands on which the native vegetation is predominantly made up of grass, grass-like plants, herbs or bushes that can be used for grazing or browsing, enhances native vegetation and includes areas planted with imported species, but which are extensively managed.	Grazing lands include natural grasslands, savannahs, wetlands, deserts, the tundra and certain forb and shrub communities.
Cropland		Land that people have planted with herbaceous crops, even if these crops will be fed to livestock, (if cropland is abandoned it becomes secondary vegetation).	
	C3 annual crops	Can only be harvested once a year and then need to be replanted, C3 photosynthesis mechanism means that there is no separation between the initial carbon dioxide fixation and calvin cycle.	Cassava, yellow yam, cocoyam, peanuts, (rice), wheat, about 85% percent of the plant species on the planet are C3 plants.
	C3 perennial crops (not plantations)	Do not need to be replanted each year, after harvest, they automatically grow back.	Red canary grass, (rice)
	C3 nitrogen-fixing crops	Nitrogen-fixing plants including Fabaceae plants, they usually require less fertilizer inputs.	Alfalfa, soy beans, clover, peanuts
	C4 perennial crops	C4 photosynthesis mechanism means that there is a spatial separation between the initial carbon dioxide and the calvin cycle, C4 plants are adapted to hot and sunny environments, perennial crops do not need to be replanted each year, after harvest, they automatically grow back.	Sugarcane, miscanthus, switch grass, sorghum.
	C4 annual crops	Annual crops can only be harvested once a year and then need to be replanted.	Corn
Plantation		Areas with vegetation for cash crop or timber production. Contrary to secondary vegetation forests, it does not grow back naturally but has to be planted.	

Land use type adapted from several authors [50,103,186,187]	Land use sub-type adapted from Hurtt et al. and Chaudhary et al. [50,186]	Definition after PREDICTS [103,187,200]	Examples
	C3 perennial crops plantation	Plantation with so called cash crops or orchards; no woody stem; planted for the fruits/crops not timber; the management is the same as practices of cropland which includes, pesticides, fertilizer or irrigation.	Banana, oil palm, olive, citrus, cocoa, tea, coffee, apple, grapes, cherries, peach, mango.
	Timber plantation	Plantations for wood production with exotic tree species.	Eucalyptus plantations in Brazil, short-rotation plantations.
Urban		Areas of human habitation and/or buildings in which the vegetation present is predominantly used for civilian or personal purposes, (remains of primary vegetation around which the suburbs might have developed are classified as primary vegetation, areas with commercial agricultural production, commercial wood plantations or pastures are classified as cropland, plantation forest or pasture).	Cities, villages, industrial sites, roads urban parks, village green areas and gardens,areas of abandoned or fallow land within built-up areas.

4.2 Description of the new BioMAPS LCIA framework

In this chapter the methodological framework of BioMAPS is further elaborated. Herein, all steps of the methodological framework for each of the scales are described on a theoretical basis. The operationalization of the framework, including the detailed calculation procedures in a GIS environment, is done in a second step in chapter 5.

4.3 Global scale

With regard to the global scale of the distribution of biodiversity and its risks, the method is based on research results from conservation scientists and NGOs. It takes into account the three different prioritization approaches, namely, reactive, proactive and irreplaceability. A harmonization of the different schemes is applied to identify important areas for biodiversity and to develop a unified biodiversity risk map. Based on this map, the different locations of land use types are analyzed. In doing so, the risk is calculated for a country or region that land use falls within a biodiversity risk area [5,15]. The calculation is carried out for the various types of land use of the land use flow list using global land use models in the regionalization step. An average risk per country or region is calculated for each type of land use for the background database. The foreground biodiversity risk map can be used to determine the risks based on primary information on land use such as coordinates or regions.

4.3.1 Development of global biodiversity risk map and gap analysis

In order to be able to assess global biodiversity risks existing conservation databases and conservation schemes are identified, analyzed and classified according to the concepts of proactive, reactive and irreplaceable. Furthermore, the conservation schemes are analyzed based on the coverage of the organizational levels of biodiversity. The similarity and coverage of congruence between the different schemes are assessed in order to avoid redundancy. Based on these databases a global unified biodiversity risk map is developed which highlights risk areas per grid cell worldwide. To this end, this method builds on the approach of Brooks et al. [76]. They take several maps representing global biodiversity risk areas and superimpose them to calculate the proportion of risk areas in a given grid cell. This is done separately for all proactive and reactive conservation schemes, resulting therefore in two world maps. However, the following changes are made for this method: First, risk areas of high (reactive) and low vulnerability (proactive) and high irreplaceability are superimposed to produce a single map. Second, according to the three-level organizational scale of biodiversity, all data that provide information on one of these levels of biodiversity will be included in the calculations. This encompasses priority areas not included in the approach of Brooks et al. [76], such as phylogenetically relevant areas for the genetic level. Finally, all currently relevant global protection schemes are combined into a uniform global risk map. In the end, we obtain three world maps: one with all proactive biodiversity risk areas, another with all reactive biodiversity risk areas and a uniform biodiversity risk map combining both proactive and reactive areas (UBR) (see Figure 10). As a next step a gap analysis is carried out to also include and assess areas that are not covered by the UBR map. Therefore, more suitable global biodiversity maps and indicators are included in the global assessment to cover the gaps identified within the step of the gap analysis.

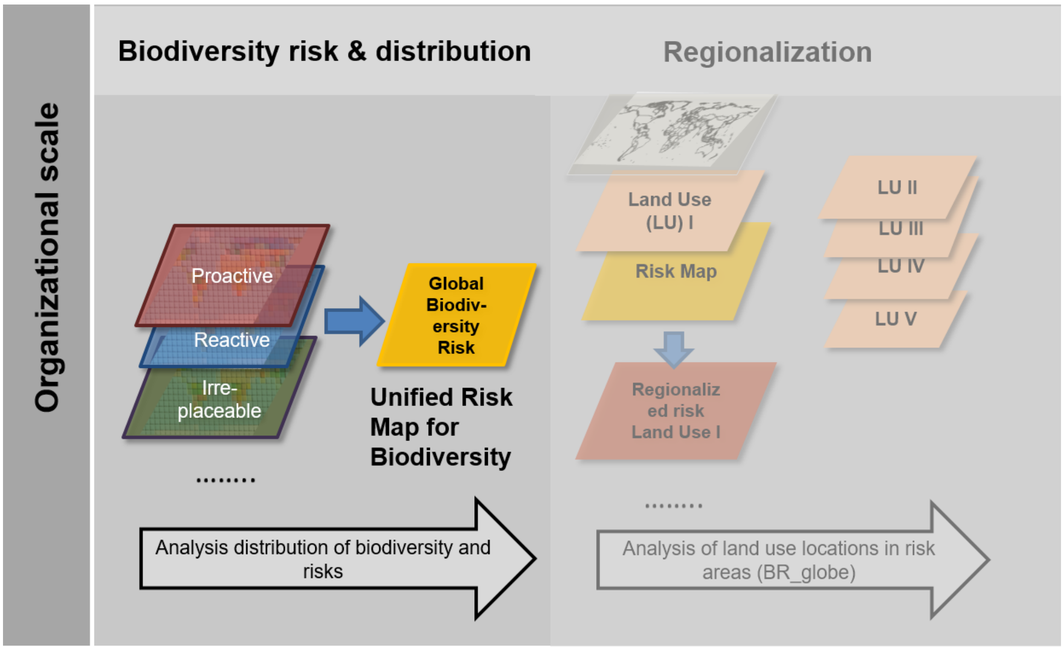


Figure 10: Global scale – development of global biodiversity risk maps (further developed from first version of Maier et al. [5])

4.3.2 Regionalization of global risks by land use type

For the regionalization, global land use models showing the current estimated location of different land use types are used for occupation and the scenarios analyzing past and future land use are used to calculate transformation risks. The calculation of the global risk factor is based on the approach of Dobrovolski et al. [202], who investigate the overlap between agricultural land use and biodiversity risk areas. However, this method is not limited by agriculture, but calculates the degree of overlaps between the different land use types for occupation and biodiversity risk areas of the UBR map. Furthermore, it analyzes land transformation by using historical records – a factor which was not integrated by Dobrovolski et al. [202], and future scenarios of land transformation [5]. However, as companies with global supply chains often do not know the exact product origin, country average values are often assumed in existing LCA methods. Going one step further, this method does not just calculate an average biodiversity risk value per country, but an average value per type of land use and country [5]. Herein, the land use map is overlaid with the unified biodiversity risk map (see Figure 11): In doing so, we can determine the degree of overlap as well as the proportion of proactive, reactive and proactive-reactive areas per land use type. This step yields the probability of land use types in risk areas of the UBR map (BR_overlay) and the proportion of risk areas per grid cell (PRA).

This way, some impacts will be significantly higher or lower than if we assumed only country average values regardless of the location of the land use types. Australia, for example, has a variety of ecosystems, ranging from rainforests in the north, vast deserts in its interior, to temperate zones in the south; the priority areas of biodiversity are unevenly distributed accordingly. If, for example, there is no pasture in the rainforest, a global risk factor for this land use excludes these areas and an impact assessment would be correspondingly smaller [5]. The assessment of the extent of land use in areas that are at risk for biodiversity has also been recommended by several nature conservation organizations [159,203] and is common practice in nature conservation research. However, this procedure has not yet been used in any other LCA method [5].

The final global risk factor per grid cell (BR_globe) for each land use flow is then calculated from the probability of land use in risk areas (BR_overlay), the proportion of risk areas per grid cell (PRA) and a so-called Jenkins Index values for all land use areas outside of biodiversity risk areas (Jenkins Index), which is derived from the gap analysis (see chapter 5.1.3). Since all maps have values from 0 to 1, we obtain a dimensionless global risk map per land use flow. Contrary to the risk factors of the regional and the local scale, the global risks factor is not normalized. This is based on the suggestion of conservation scientist, who state that a global impact would have higher implications for the overall biodiversity loss than a local or regional impact on biodiversity [204].

$$\text{BR\_globe}_{[i]:} = \text{BR\_overlay} + \text{PRA} + \text{Jenkins Index} \tag{1}$$

where

$\text{BR\_globe}_{[i]:}$: Global biodiversity risk at location i

BR_overlay: Probability of land use types in risk areas of UBR map (regionalized biodiversity risk) per grid cell

PRA: Proportion of risk areas per grid cell

Jenkins Index: Values of regionalized Jenkins Index outside of risk areas per grid cell

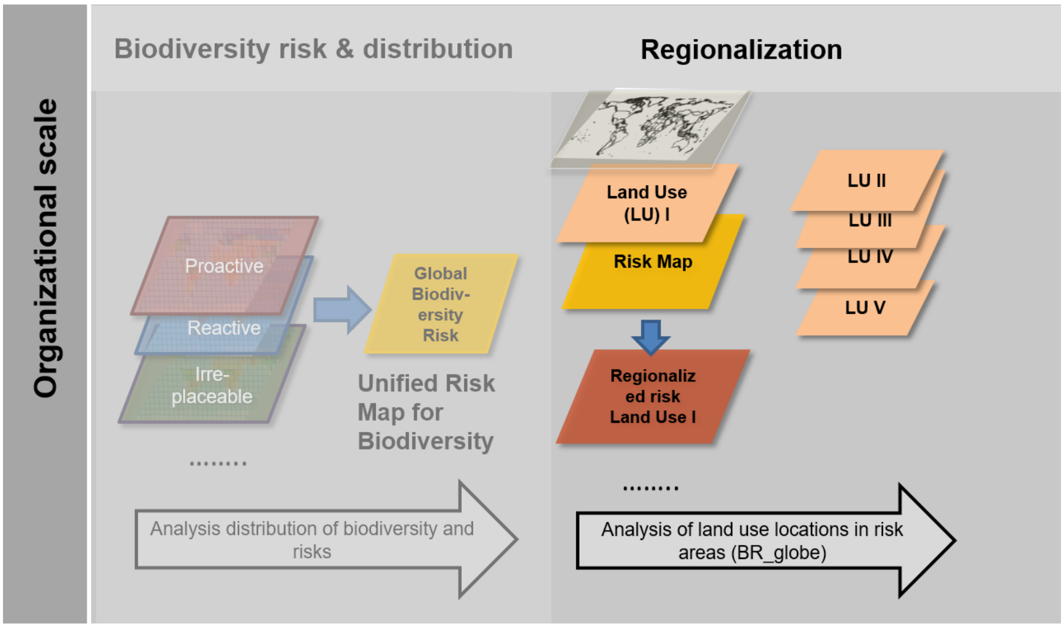


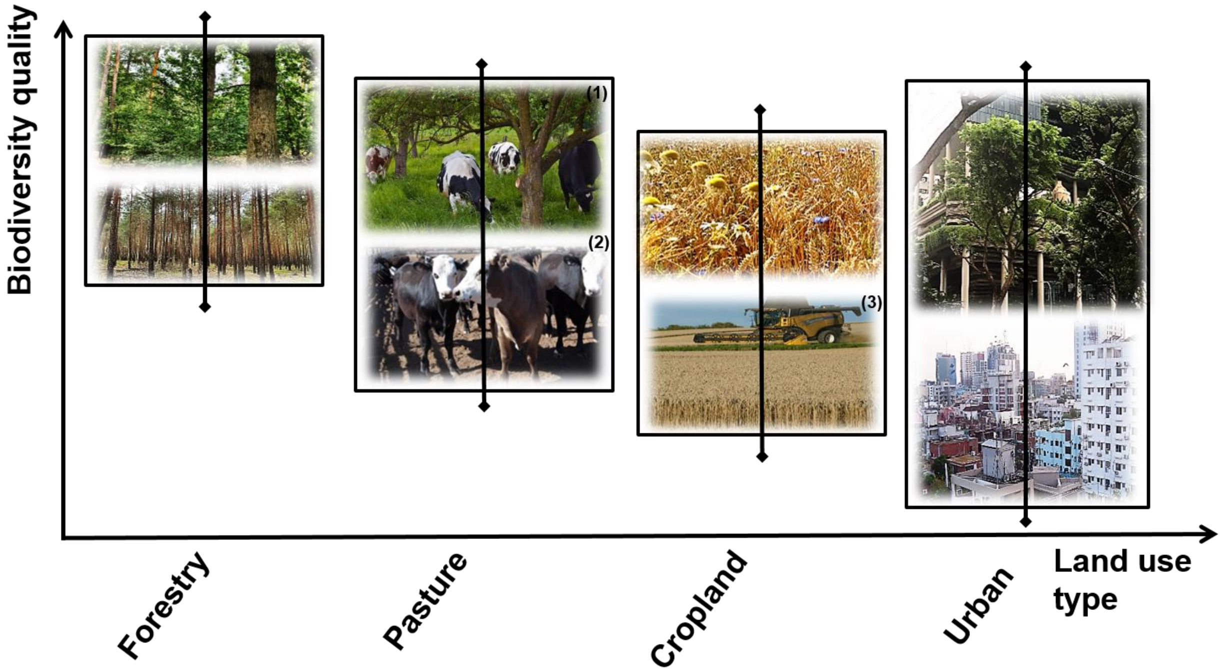
Figure 11: Regionalizing global biodiversity risks (further developed from first version of Maier et al. [5])

4.4 Local scale

In the second step, available research results and databases are used and integrated into the method to assess the impacts of different land use types at a local scale on biodiversity [5,15]. With these research results an average impact per land use type is determined as well as an impact interval. This interval is determined by the intensity of land use and its specific land management. Management parameters that have a proven impact on biodiversity are identified from conservation databases. These parameters are used to calculate the land use intensity indices (LUI). For this method, global land use intensities are calculated for the different types of land use, based on statistical data. The global data sets can be used as background data for the supply chain. If a company has primary data on individual management parameters, this method can be used to calculate in a foreground process how a shift in the interval impacts biodiversity.

4.4.1 Calculation of land use type specific impact intervals

For determining the impact on biodiversity at the local scale and for calculating the impact intervals, this method uses the values of Newbold et al. [105,106] based on the PREDICTS database [187,200,201]. The PREDICTS database is based on the field of macro ecology and offers the most comprehensive database of its kind to date. It contains data on different types of land use and how they affect biodiversity worldwide in relation to a reference situation. These impacts are measured by a variety of biodiversity metrics of which the information stems from published scientific studies, which in turn have measured such effects on site and thus provide empirical results. The PREDICTS database includes data on numerous taxa ranging from invertebrates to vertebrates, therefore avoiding cross-taxon surrogacy. The reference situation of PREDICTS is the primary vegetation under minimal human influence [105] and thus belongs to the concept of space for time substitution. Since the comparison of the impact of a type of land use is made against the reference state, the PREDICTS model is in line with the UNEP-Setac framework and the existing LCIA methods [5]. From the data, an average impact value of a land use type on biodiversity is calculated and an impact interval as a function of land use intensity is determined. This interval is defined by the management of the plot/field and its land use intensity [5] (see Figure 12).



(1) https://www.pxfuel.com/en/free-photo-xanqn
(2) https://pixnio.com/fauna-animals/cows-and-calves/cattle-ranch-farming-cows
(3) https://www.piqsels.com/de/public-domain-photo-zefdv

Figure 12: Change in biodiversity quality in comparison to a reference state for different types of land use and their intensities (own pictures, except 1-3 under creative commons)

This database allows the evaluation of land use types based on different biodiversity metrics. Herein, it is possible to provide impact values for each individual biodiversity metric or to summarize them within one index. For this method, an index is calculated to facilitate communication of the results. For the background data, the default biodiversity impact value is always the worst case scenario, which is equal to the average value for intensive land use [5]. Within the interval, changes in management parameters can lead to changes in impacts on biodiversity both positive and negative ones. For one type of land use, the size of the interval does not change based on the findings of Newbold et al. [105], but within the interval the value of a specific biodiversity impact can vary depending on the intensity of land use [5,15] (see Figure 13).

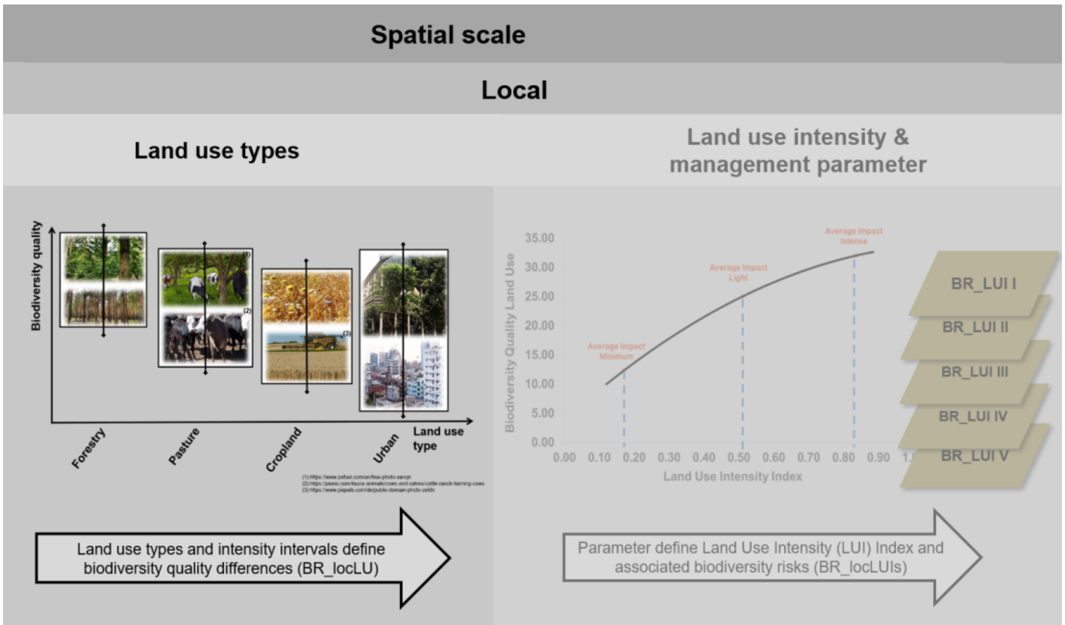


Figure 13: Land use intensity defines biodiversity quality within impact intervals (further developed from first version of Maier et al. [5])

As several studies show, there is no statistically significant difference in the relative impacts on biodiversity of the land use types within different biogeographical regions [105,205]. Therefore a value for the quality of land use for biodiversity can be used, which depends only on the type and intensity of land use, decoupled from its location [5]. The models by Newbold et al. [104,105] that analyze the PREDICTS database provide data for the biodiversity metrics “species richness”, “rarefied species richness”, “abundance” (alpha diversity), “functional diversity” and “community similarity” (beta diversity, measured as Jaccard similarity of species richness and abundance). These are used for the calculation of biodiversity differences within a given interval. The characterization factors for the local impact are calculated from these quality differences. The impact on species richness for example is calculated by comparing the number of species within a given type of land use with the number of species in the reference situation, thus creating the relative difference in the quality of biodiversity. Functional diversity is an indicator of differences in certain functional traits such as plant height or body size between the plot and the reference plot. PREDICTS provides information on the differences in functional properties of mean plant height for some of the land use types [105]. Further biodiversity indicators are the Jaccard traits, which provide information on the compositional similarity or dissimilarity of species between a type of land use and the reference situation. They are a biodiversity indicator for measuring ß-diversity. The Jaccard similarities are calculated as a Jaccard index based on richness and abundance [104]. Some examples for the calculation of biodiversity indicators and quality differences at the local scale are given in the following.

Species richness: $S_{LU} / S_{Ref} = SR_{LU}$ (2)

Quality difference species richness: $((S_{Ref} - SR_{LU})/S_{Ref}) * 100\% = \Delta R$ (3)

where
S_{LU}: Number of species under land use
S_{Ref}: Number of species under reference land use
SR_{LU}: Species richness under land use
ΔR: Quality difference of species richness for land use

Species abundance: $A_{LU} / A_{Ref} = SA_{LU}$ (4)

Quality difference abundance: $((A_{Ref} - SA_{LU})/A_{Ref}) * 100\% = \Delta A$ (5)

where
A_{LU}: Number of individuals under land use
A_{Ref}: Number of individuals under reference land use
SA_{LU}: Species abundance under land use
ΔA: Quality difference of species abundance for land use

For reasons of applicability, a simple assumption is used and an average index for the quality of biodiversity risks per land use as the average of all biodiversity metrics provided by Newbold et al. [104,105] based on PREDICTS, is calculated (referred to here as the local biodiversity risk quality index). As in their study they only provide data for species richness, abundance and rarefied species richness for different intensities, only these metrics will be used. However, in the future further metrics might be added from PREDICTS such as functional diversity or similarity richness and abundance. Since all values are listed as relative values in percent, they can be summarized as follows:

$$BRloc\_LU_i = \frac{1}{3} * (\Delta R_i + \Delta A_i + \Delta Rf_i) \quad (6)$$

where

$BRloc\_LU_i$: Biodiversity risk for land use i at local scale

ΔR_i: Quality difference of species richness for land use i

ΔA_i: Quality difference of species abundance for land use i

ΔRf_i: Quality difference of rarefied species richness for land use i

4.4.2 Land use intensity and management parameters

The intervals of the Biodiversity Risk Index derived from Newbold et al. [103,104] from the PREDICTS database are used to quantify the impact of land management on biodiversity. For this purpose, a land use intensity index is calculated for each type of land use. The concept of land use intensity indices (LUI) is based on [188,189,206]. It comes from the fields of agroecology, geography and geosciences, but has not yet been used in any LCIA biodiversity method [5]. The LUI index is an additive index that consists of various management parameters and provides a value for the intensity of land use. The LUI index is calculated for a specific land use type and then calibrated against the impact interval derived from the PREDICTS database. By linking the land use intensity indices with the information on local biodiversity impacts from the PREDICTS model, we can see how the values shift within an interval. Therefore, we can determine a final local risk value based on the land management (see Figure 14).

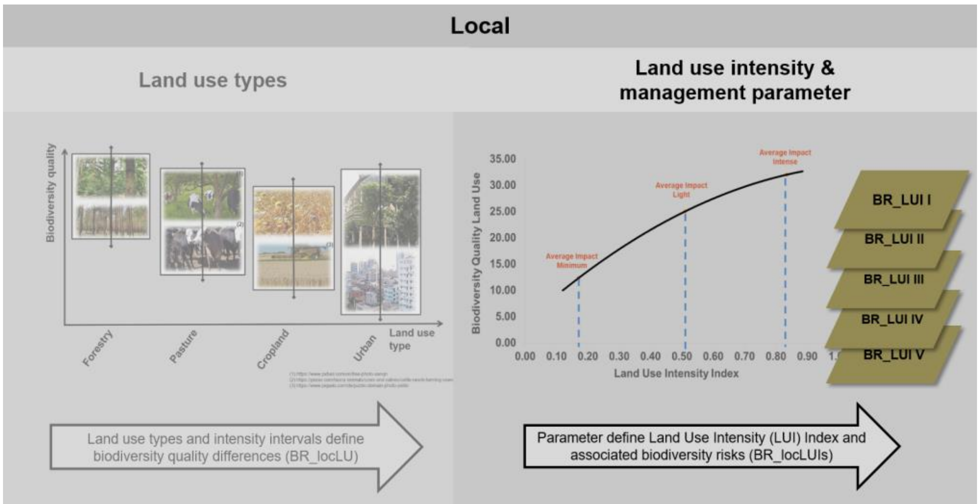


Figure 14: Calculation of local biodiversity impacts within the interval (further developed from first version of Maier et al. [5])

4.4.3 Assessment of management parameters

A review of different management parameters in relation to the land use intensity has been carried out in several publications, see [189,206,207]. Some indicators for the intensity of cropland, for example, include fertilizer application (mineral and organic), technologies or labor intensity, irrigation and crop yields [207]. Robinson & Sutherland [112] show possible causes for the change in populations of birds, plants and insects through different farming methods for arable land. For example, the intensity of mechanization or the use of pesticides have a direct influence on insects and the amount of fertilizer directly affects plant populations [112]. In various ecological studies, the pasture intensity in grassland was assessed using the indicators of mowing frequency, grazing intensity and fertilizer use [208–210].

For the BioMAPS method, management parameters are compiled for the calculation of land use intensities for all land use flows of the newly developed land use flow list. All management parameters are identified on the basis of the Conservation Evidence Database [198]. This database collects and summarizes scientific literature on land management activities. The effectiveness of the management activities is evaluated as well as the question whether that activity is beneficial or even harmful for biodiversity. The Conservation Evidence Database is based on over one million scientific publications reviewed by scientists on the impact of management activities on biodiversity [198].

The next step is to select appropriate indicators that can measure the intensity of the respective management parameters. A distinction can be made between pressure indicators to measure adverse impacts on biodiversity (e.g. use of pesticides) and relief indicators to measure positive impacts on biodiversity (e.g. provision of habitats in the form of set-aside area). The selection of indicators is also based on the availability of global data sets and GIS maps, since management parameters are identified to calculate global LUIs for the background database. Furthermore, management parameters and appropriate data sets are identified on the basis of the studies of [105,106,189,198,206,207].

4.4.4 Calculation of land use intensity indices

Various land use intensity indices have been developed, see [188,211–215]. The herein developed method builds on the land use intensity index developed by Herzog et al. [188] and applied and further developed by Blüthgen et al. [208], Erb et al. [189] and Kuemmerle et al. [206], since a correlation between their land use intensity index and effects on biodiversity was empirically measured and validated [208]. The LUI is calculated according to the following formula [188,189,206,208].

$$LUI_{\text{Land use type [i]}} = \frac{PI[i]}{PI[max]} + \frac{PII[i]}{PII[max]} + \frac{PIII[i]}{PIII[max]} + \frac{PIV[i]}{PIV[max]} + \frac{Pn[i]}{Pn[max]} \tag{7}$$

where

$LUI_{\text{Land use type [i]}}$: Land Use Intensity Index of a specific land use type at location [i]

P: Specific management parameter (I to n parameters)

max: Maximum value of specific management parameter P

For the management parameters related to the impacts on biodiversity within the same land use type, a land use intensity index is calculated as the sum of its management functions. According to Erb et al. [189], individual parameters are standardized either by a *maximum value, mean value* or *z standardization*. Depending on the type of standardization, an index with values from 0 to 1 is obtained for each type of land use flow, where 0 means a very low intensity of land use and 1 reflects a very high intensity of land use. The standardization of the values of management parameters is also referred to as benchmarking in this study, since the individual values are compared with a benchmark value.

Choosing a benchmark poses similar challenges to choosing the reference situation in the UNEP SETAC framework for assessing the impact of land use on the quality of biodiversity. Various approaches have been developed and used to calculate the land use intensity index. Kuemmerle et al. [206], Erb et al. [189] and Herzog et al. [188] propose to use either the mean value or the maximum value of all individual values for standardization. However, they also note that this is not the case for global calculations because the values vary too much depending on the region. And here outliers influence maximum and mean values too strongly [206]. Another possibility for benchmarking comes from the economic sector. Economic benchmark values are calculated as a function of maximum agronomic yields. For example, the response function of the amount of nitrogen supplied as a function of the crop yield shows a maximum at which the yield cannot be increased even though the nitrogen supply is increased. This maximum is referred to as the maximum agronomic yield [216]. Here, however, the benchmark value is mainly based on economic rather than ecological criteria.

Another approach could be to use policy regulations or scientific recommendations for benchmarks that should not be exceeded for each management parameter. However, policy recommendations vary from country to country and do not account for differences in geographic characteristics. For example, the German government uses 70 kg per hectare as the target value for the nitrogen surplus in arable land [217]. Environmental NGOs and scientists claim that this target is still far too high. The NGO 2030Watch [218], for example, proposes a target value of a maximum of 50 kg nitrogen surplus. Kim et al. [219] classify values with a nitrogen input of more than 250 kg per ha and year as high land use intensity at the suggestion of Temme & Verburg [220].

The FAO proposes different amounts of fertilization depending on the type of crop and the geographical region. For wheat in temperate regions, 200-250 kg N/ha is recommended for high yields. In subtropical regions, however, only about 120-150 kg N/ha is recommended. While pulse crops (nitrogen-binding plants) generally require less fertilizer [216]. The fertilizer application rates in grassland vary between 0 and up to 1000 kg nitrogen per hectare. On average, however, they are between 50 and 350 kg nitrogen per hectare [216]. The FAO stresses that the nitrogen input required varies depending on the stored soil moisture, plant and crop varieties and climatic and geographical conditions.

Based on these discussions, this method takes up the concept of Maximum Tolerable Intensity (MTI), developed by Sattler et al. [221] as a benchmark for all indicators that are influenced by geographic effects, for the other indicators the maximum value is taken. The maximum tolerable intensity level is calculated from the mean value and the standard deviation of all individual values in a specific geographical region. It should be noted that the MTI is not a conservation target. Rather, it is a threshold value that indicates a very high intensity of land use (LUI = 1) for current practices. Thus, any value above the maximum tolerable intensity level will automatically receive the highest intensity value of 1. Conservation targets of pressure indicators are usually oriented towards lower intensity levels. Taking into account the geographical variations of some management parameters, not one global benchmark value but several global benchmarks are calculated depending on the agro-ecological zone in which the land use takes place [5]. Agro-ecological zones are natural physical regions with similar climate, physiographic and soil characteristics. The division of the world into agro-ecological zones offers a standardized approach for the description of the relevant external factors influencing agricultural land use practices [222]. The FAO recommendations for land management, such as fertilization, also depend on the characteristics of the different agro-ecological zones [190,222]. Figure 15 shows 18 global agro-ecological zones as described by Ramankutty et al. [223]. For each of these zones, a benchmark value is calculated as the maximum tolerable intensity for each management parameter depending on site-specific conditions. Based on the individual benchmark value, the land use intensity index is calculated for each type of land use. The calculation of individual benchmark values per global agro-ecological zone has the advantage that the individual management parameters and intensity levels are only compared within the same agro-ecological zone that share the same geographic and climatic characteristics [5].

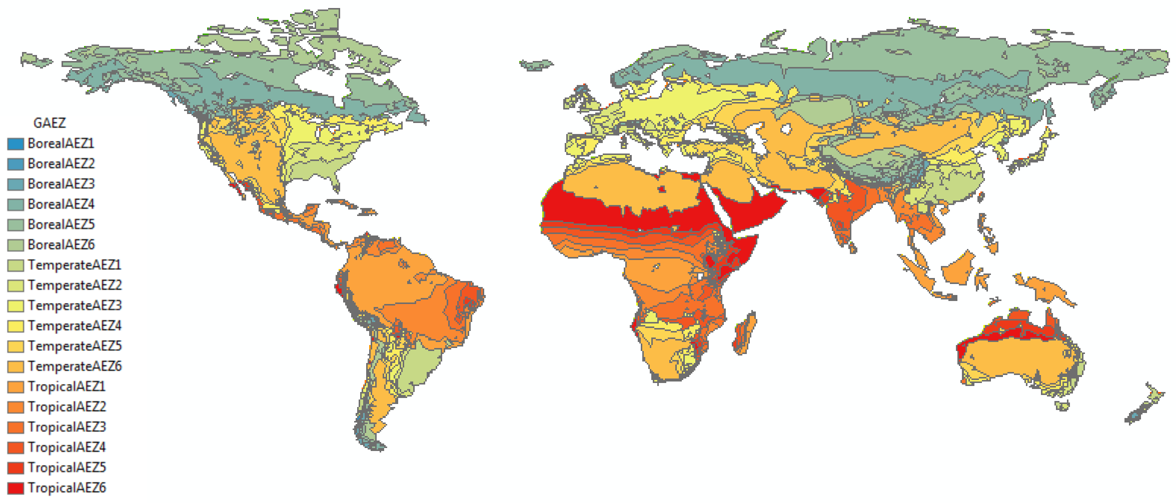


Figure 15: Global Agro-Ecological Zones (based on Ramankutty et al. [223])

The maximum tolerable intensity is calculated for each land use type that depends on geographic and climatic conditions (for example for each crop type within the same agro-ecological zone). The calculation uses the following equation:

$$\text{MTI} = \text{av(per land use type and major GAEZ)} + \text{av SD(per land use type and major GAEZ)} \quad (8)$$

where

MTI: Maximum tolerable intensity per crop type and GAEZ

av: Average value per land use type and GAEZ

av SD: Average standard deviation per land use type and GAEZ

The LUI is then calculated as follows:

$$LUI_{LandUseTypeI[i]} = \frac{PI[i]}{PI[MTI]} + \frac{PII[i]}{PII[MTI]} + \frac{PIII[i]}{PIII[MTI]} + \frac{PIV[i]}{PIV[MTI]} + \frac{Pn[i]}{Pn[MTI]} \quad (9)$$

where

$LUI_{LandUseTypeI[i]}$: Land use intensity of the specific land use type

Pn[i]: Management parameter

Pn[MTI]: Maximum tolerable intensity of management parameter based on [221]

In this method, land use intensity indices are calculated for all land use flows of the new flow list, namely primary and secondary vegetation (including forestry), pasture, cropland, plantations, and urban areas. This facilitates the translation of these land use intensity indices into the biodiversity risk values from the PREDICTS model. The general method for calculating global LUIs is illustrated in Figure 16, exemplarily for the land use type cropland. From existing global land management databases, intensity maps are developed to represent areas of high and low intensity of land management. These are then combined into a single LUI for each land use flow. If a company knows where the land use of its product takes place, it can be determined on the map what the average land use intensity in this area is. This land use intensity value can then be used to determine the likely biodiversity impacts from the PREDICTS model.

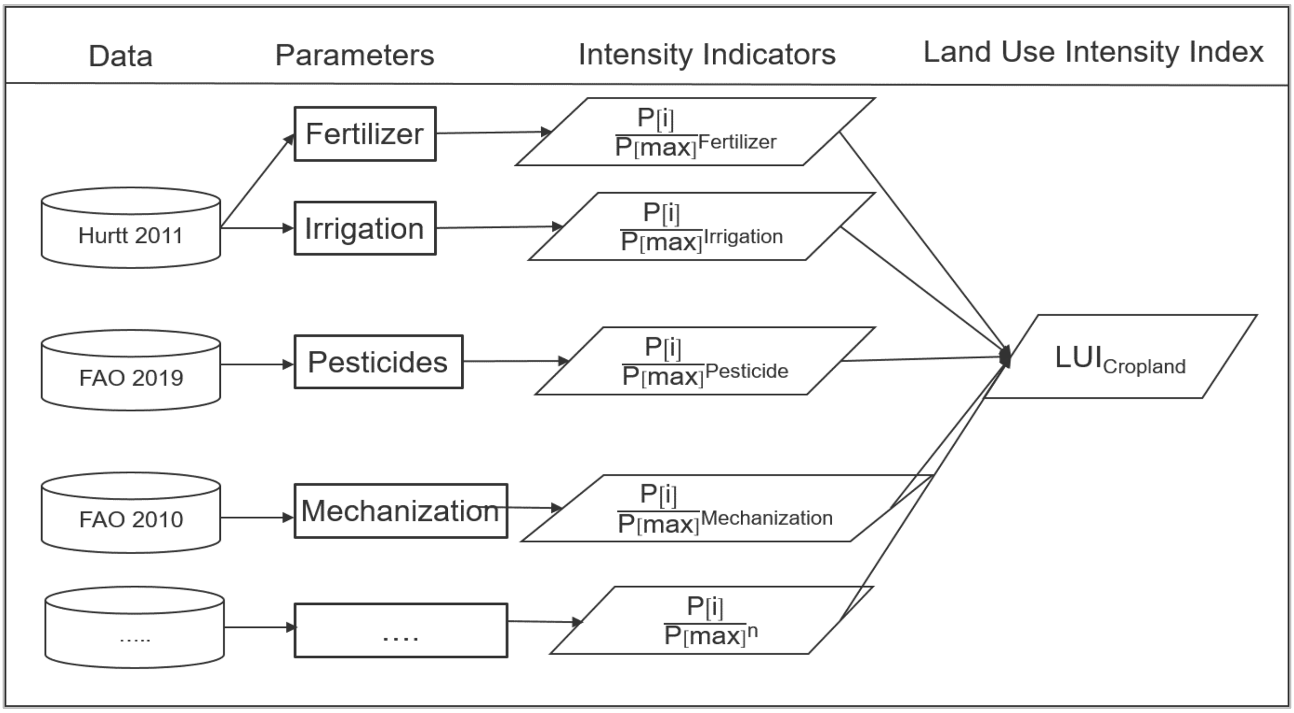


Figure 16: Calculation of LUI index

4.4.5 Translation of LUI into biodiversity risks

In this step, the land use intensity indices for each type of land use are translated into biodiversity risks using the biodiversity risk index impact intervals from [104,105] of PREDICTS. The biodiversity risk index provides an average value for each type of land use and an impact interval (sub chapter 4.4.1). In addition, average values are given for the land use intensities of minimum, light and intensive land use. Based on Maier et al. [5], the calculated LUI indices are translated into biodiversity risks by using an equation that fits a simple curve to three fixed points of average minimum, light and intensive land use intensity. The interval for minimum land use intensity lies between 0 and 0.33, with an average minimum intensity value of 0.17. The interval for light land use intensity is between 0.33 and 0.66, with an average light land use intensity value of 0.51. The interval for intensive land use intensity is between 0.66 and 1.0 with an average intensive value of 0.84 [5]. This average intensity value is used for the background database for management parameters for which no global data set is available (e.g. for set-aside areas). Translating the LUI values into biodiversity risks makes it possible to calculate changes in biodiversity risks within the impact interval caused by changes in individual management parameters and thus in land use intensity. Note, that a land use intensity value of 0 is only a theoretical value. If an area is not farmed, it is either abandoned and has turned into secondary vegetation, or it is pristine and has primary vegetation. Therefore, a land use intensity of 0 cannot be achieved on managed land and is therefore not shown in the following graph. An example for such a relationship is given in Figure 17 for the land use type pasture (see chapter 5.2.7.3).

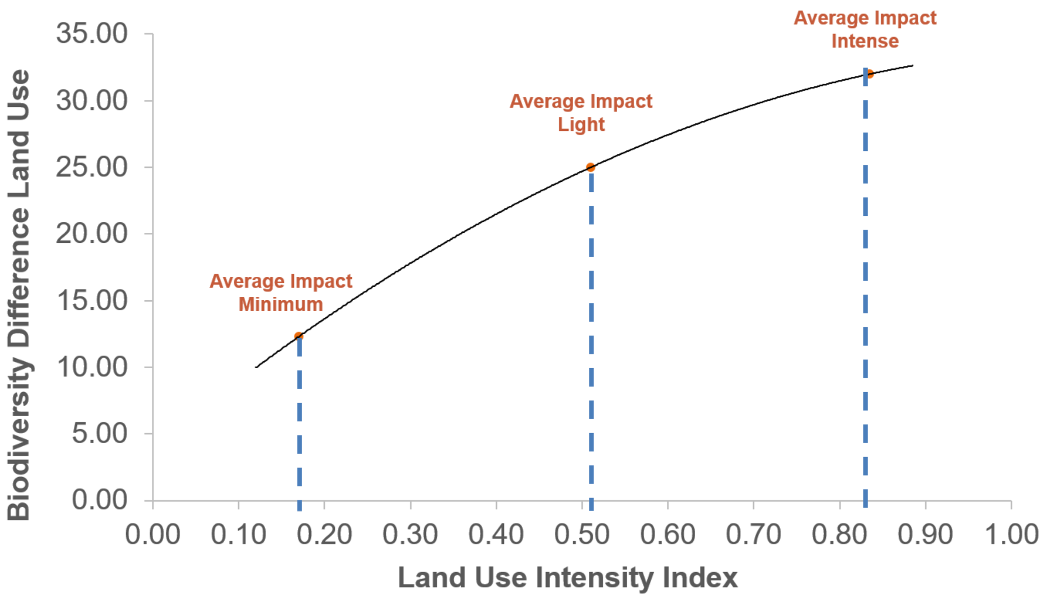


Figure 17: Translation of land use intensity values into biodiversity risks as proposed by Maier et al. [5]

As ecological studies show there are mainly two different shapes of the relationship between the intensity of land use and biodiversity which are most frequently described [182]. These relationships are considered in a sensitivity analysis. Thus, it is assumed that the biodiversity risks increase proportionately to the rate of increase in anthropogenically influenced land use similar to the approaches of Arunyawat & Shrestha [224] and McKinney [225] for habitat quality [226]. By adapting their approaches, polynomial degree 1 and 2 relationships of biodiversity quality will be tested in chapter 5.2.7:

$$BR\_LUI_i=(a*LUI_i^2+b*LUI_i+c) \quad \text{if degree 2 polynomial} \tag{10}$$

$$BR\_LUI_i=(d*LUI_i+e) \quad \text{if degree 1 polynomial} \tag{11}$$

where

BR_LUI_i: Specific biodiversity risk based on the LUI

LUI_i: LUI of the land use i

a, b, c, d, e: Set of land use type specific coefficients as defined in chapter 5.2.7

4.5 Regional scale

As a last step, the results of the local biodiversity risks (from the LUIs) are then scaled up to a larger landscape context in order to assess the regional impacts. Within this step we receive a biodiversity risk value for the entire landscape matrix.

4.5.1 Integration of the LUI index into a landscape matrix

In order to analyze the effects of land use intensity not only at the plot or field level, but also at the landscape level, the LUI indices (and their resulting biodiversity risks) obtained for each type of land use are used to calculate a landscape development index (LDI), which is adapted for this biodiversity method based on Brown & Vivas [199]. This is done to include biodiversity impacts not only at the local level, which refers to one specific land use type, but also to include the landscape level, which might include several land use types. Additionally, some of the management parameters do not directly relate to the field or patch level but to the landscape matrix (= area of influence) within the analytical unit. This method defines a landscape, therefore, as an area that consists of different patches of land use types, where the scope of influence in LCA depends on the management of a company, farm, city, or municipality. A landscape might be quite complex, for example: A farmer owns land on which he/she grows C4 annual and C3 nitrogen-fixing crops. In addition, the owner also uses some of the land for grazing livestock and sheep as part of a managed and rangeland pasture system. In an adjacent area he/she owns a small forest for wood production and a larger primary forest which he/she decides to leave unmanaged and therefore to set it aside for biodiversity. In addition, he/she sets aside a stretch of primary forest to connect both forest patches via a corridor. The area where the farmhouse is located is part of the urban land use type. The farmhouse might have a green façade and a garden with a wild flower mix and might therefore be under light land use intensity (see Figure 18).

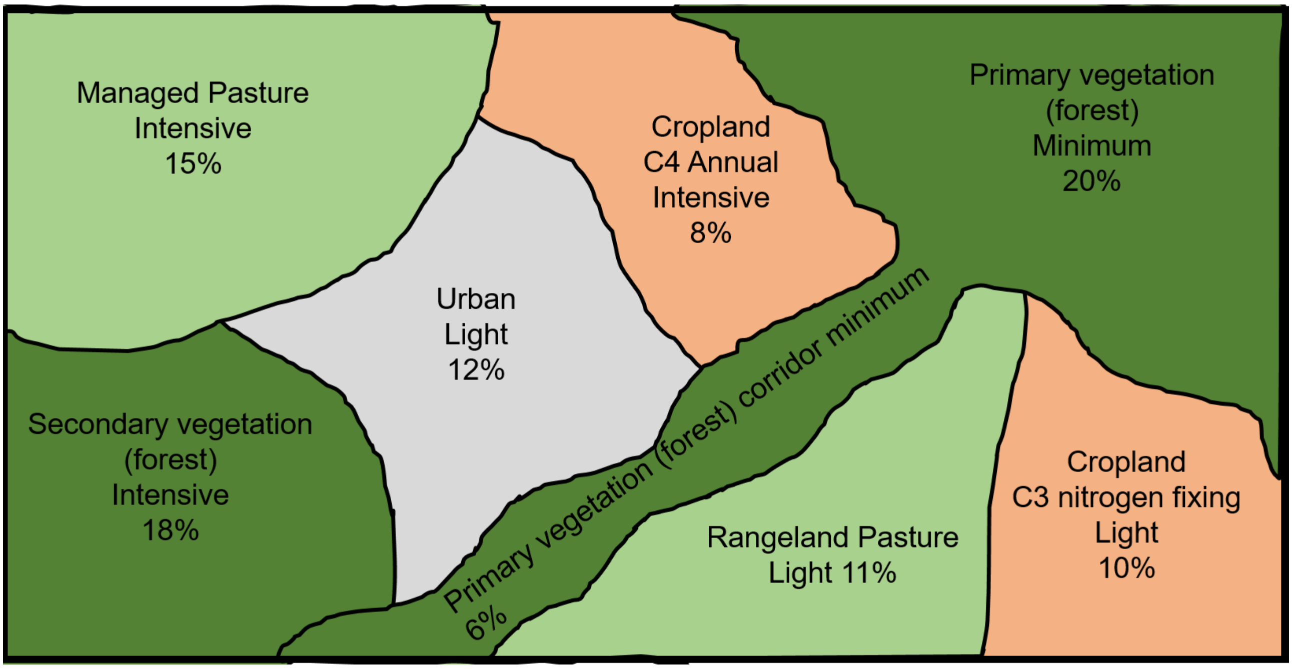


Figure 18: A landscape matrix composed of several patches of land use types and their intensities

Herein, each of the land use patches has its own land use intensity – depending on the farmer's management. This leads to separate biodiversity risks for each patch and a common biodiversity risk at the landscape level. Even if some areas are used intensively, there are other areas that might be spared for the conservation of biodiversity (in this example a patch of primary forests is set-aside as well as a corridor of primary forest connecting both forest patches). The advantage of calculating the biodiversity risk at a landscape level is, that those landscapes are highlighted with a higher share of primary and secondary vegetation. Furthermore, landscapes with a higher share of low intensity land use patches will also result in a lower biodiversity risk at the landscape level. Thus, landscapes with a higher share of set-aside areas of primary or secondary vegetation (e.g. natural forests) and/or with a higher share of land use types under low intensity management ultimately have a better LDI. Thus this step of the method includes both land sharing and land sparing strategies.

Therefore, the analysis of the impacts of land use on biodiversity is extended to the landscape level in order to capture the overall picture and to be able to include landscape scale management approaches. The equation for calculating biodiversity risks at landscape level is based on Brown & Vivas [199]. A landscape development index is calculated by determining the total area of the landscape and the percentage of all land use types within the landscape [144]. In addition, the biodiversity risk resulting from the land use intensity indices is considered for each patch resulting into an overall biodiversity risk value at the landscape level, calculated as arithmetic mean of BR_LUI weighted according to the land use area proportions (see Figure 19).

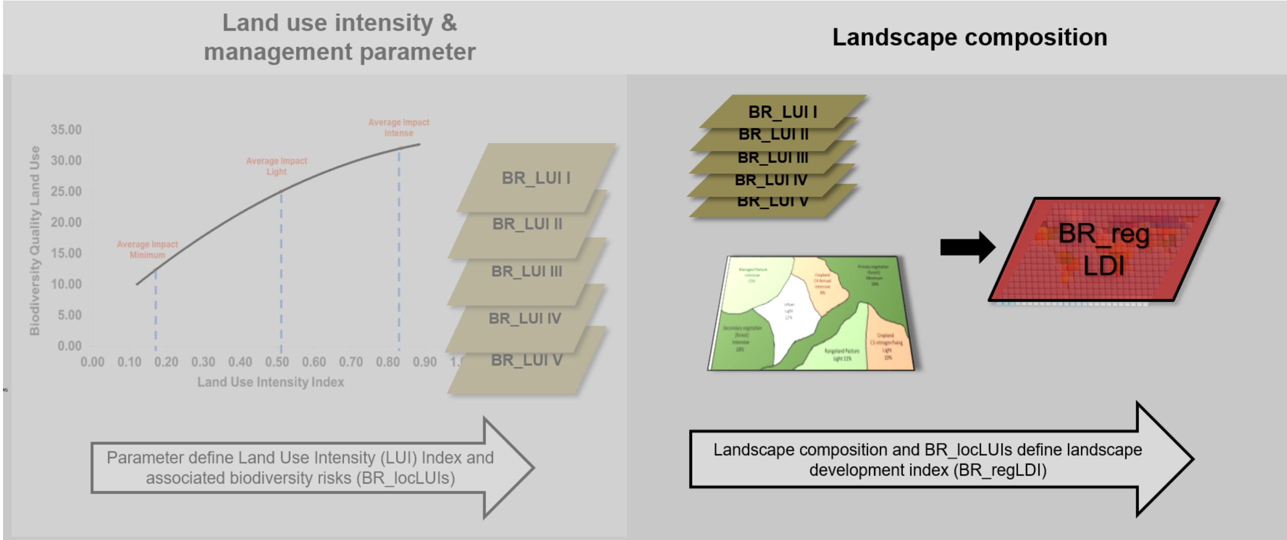


Figure 19: Regional assessment of biodiversity impacts (further developed from first version of Maier et al. [5])

The landscape development index (LDI) is calculated based on Maier et al. [144]:

$$BR\_regLDI_{total} = \sum \% \, LU_i * BR\_locLUI_i \qquad (12)$$

where

BR_regLDI_{total}: Biodiversity risk at landscape level

$\% \, LU_i$: Percent of land use i in the total area of influence

BR_locLUI_i: Biodiversity risk for land use i depending on the land use intensity

Note, that the land use type that is assessed for the local scale is also part of the landscape and therefore also included in the landscape development index. This is done, since a monotonous landscape can also consist of only a few land use types. For example, a landscape that consists mainly of extensive, large cropland areas and an urban area of the farmhouse would still be taken into account in the regional assessment. Furthermore, there is no specific de minimis threshold for a land use type area to be considered within the landscape. This is due to the fact that even very small land use patches can provide a habitat to different species [227].

4.6 Summary of BioMAPS methodological framework

In summary, the overall methodological framework of BioMAPS comprises three different parts for the three spatial scales. The calculation procedure of each of the three parts is elaborated in chapter 5, following this summarized structure:

1. Global scale:
   a. Assessment of global biodiversity risk areas and creation of a unified biodiversity risk (UBR) map with different organizational biodiversity levels and conservation indicators.
   b. Gap analysis of areas that are not covered by global conservation schemes.
   c. Regionalization of global risks with land use models for occupation and transformation. Calculation of the share of land use in risk areas (BR_overlay), the proportion of risk areas per grid cell (PRA) and the share of threatened, endemic and total species in non-risk areas (Jenkins Index). This step leads to a global biodiversity risk factor of different land use types (BR_globe).
2. Local scale:
   a. Provision of land use type specific impact intervals for different ecological indicators by calculating a local biodiversity risk index. Here, we obtain biodiversity risk values per land use type on the local scale (BR_locLU).
   b. Calculation of land use intensity indices (LUI) using management parameters and translation into the land use type specific values of the impact interval of the local biodiversity risk index (BR_locLUI). We obtain biodiversity risk values at the local level depending on the actual land use intensity.
3. Regional scale:
   a. Assessment of biodiversity risks of the individual land use types within a landscape matrix by calculating a landscape development index (LDI).
   b. Assessment of the share of land use types within a landscape (landscape composition) as well as the intensities of the individual land use types and its associated biodiversity risks from step 2.
      This enables a landscape-specific biodiversity risk value to be determined at a regional level (BR_regLDI).

The characterization factor is then calculated from the global (BR_globe), regional (BR_regLDI) and local biodiversity (BR_locLUI) risk values. Herein, a simple additive index is used.

This characterization factor is multiplied with the area required per functional unit as well as the time for occupation impacts.

$$CF = BR\_globe + BR\_locLUI/100 + BR\_regLDI \tag{13}$$

where
CF: characterization factor
BR_globe: Global biodiversity risk factor
BR_locLUI: Local biodiversity risk factor depending on land use type and intensity
BR_regLDI: Regional biodiversity risk factor depending on landscape composition and intensities of patches

The resulting value is expressed as Potential Biodiversity Risk multiplied by area and time.

$$PBR = CF*a*t \tag{14}$$

where
PBR: Potential Biodiversity Risk
CF: Characterization factor
a: Area
t: Time

In the following chapter, the individual steps are explained in more detail, including the necessary calculations in GIS, the algorithms for automation and parallelization as well as the underlying databases. Furthermore, the data calculated for the background system are presented, which operationalize the method for global value chains. The overall structure of the BioMAPS method is displayed in Figure 20.

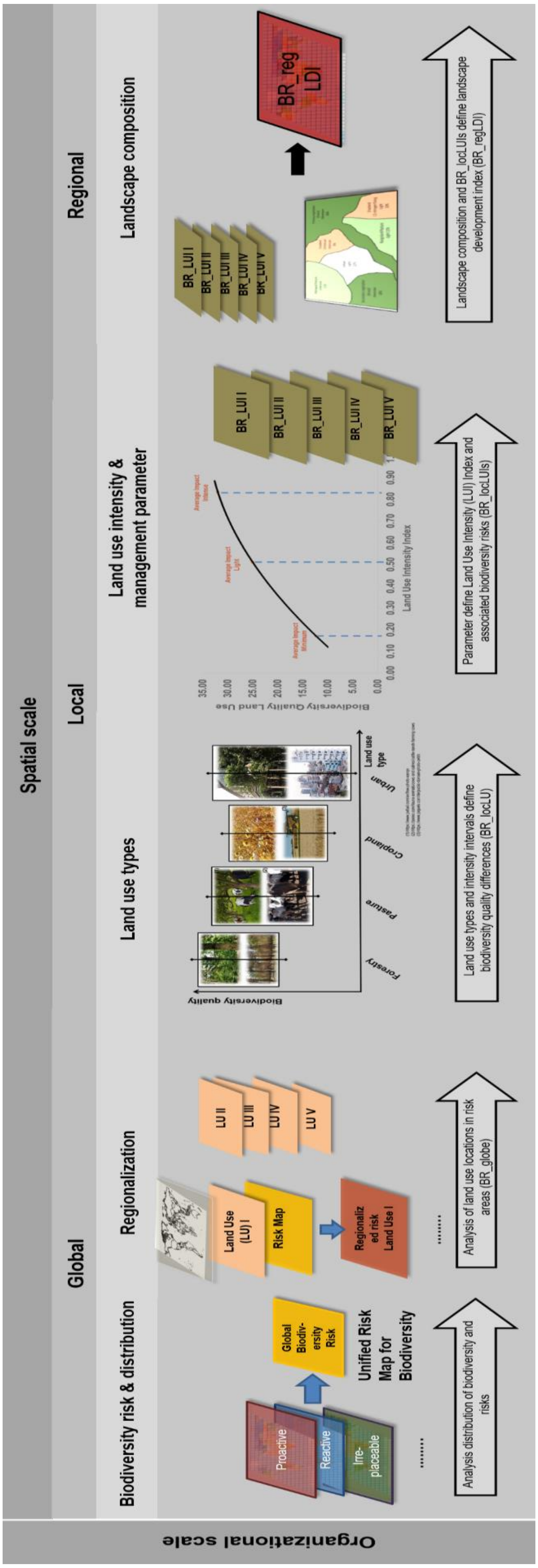


Figure 20: Biodiversity impact assessment framework of the BioMAPS method (further developed from first version of Maier et al. [5])

5 Generating product system background data in a GIS environment

This chapter describes the calculation of the background data for all spatial scales using a GIS environment to make the method applicable worldwide. If detailed information is known, such as the exact coordinates where land use takes place or information on specific land use parameters, this primary data can be used as part of the foreground system to enable a more accurate assessment. Since, the methodological framework of BioMAPS is structured as a modular procedure according to the global, regional and local scale of the assessment, all analytical steps are evaluated independently of each other, using different databases, models, concepts and maps for each step.

5.1 Global scale

This chapter describes the calculation steps and underlying databases for the biodiversity assessment at a global scale. It describes the design of the global unified biodiversity risk map, the analysis of suitable databases, the calculation procedures with GIS as well as the gap analysis. Furthermore, the regionalization of global biodiversity risks is executed using global land use models.

5.1.1 Development of global risk map

As a first analytical step a world map is developed showing the global risk areas of biodiversity. We obtain three world maps: one with all proactive biodiversity risk areas, another with all reactive biodiversity risk areas and a uniform biodiversity risk map combining both proactive and reactive areas (UBR). Furthermore, a gap analysis is conducted to assess the areas that are not covered by the biodiversity risk map.

5.1.1.1 Biodiversity conservation databases and analysis

The first step in producing a biodiversity risk map is to identify current global biodiversity protection schemes and databases and to assess their suitability for the inclusion in the risk map. To this end, each scheme is classified according to the concept of irreplaceability and vulnerability (further distinguished between reactive and proactive approaches) [76], its coverage of the level of biodiversity and the different taxa which are included at the species-level. Based on the proposals of Gordon et al. [75] the following inclusion criteria are applied:

- The map must represent the whole world where land use takes place (except for Antarctica), and

- it must prioritize areas for biodiversity, thus indicating areas of high and low biodiversity risks;
- the criteria must be scientifically sound (maps containing only policy objectives will not be considered);
- the data must be freely available;
- the map must include, even better focus on, terrestrial biodiversity, since this is the scope of the assessment method.

In the following, this chapter illustrates how the priority maps are developed. Based on the literature review of global conservation schemes of Brooks [93] and Maier et al. [5] taking into account the herein described criteria, 18 global conservation schemes have been identified. These are analyzed with regard to the coverage of biodiversity and its classification as proactive or reactive risk areas, resulting in two separate maps. These maps are combined in a second step to one unified biodiversity risk map. This is done to prevent, that either proactive or reactive areas are given more consideration. Out of the 18 global conservation databases identified, 15 schemes prioritize areas for biodiversity conservation. There are three additional databases developed by Jenkins et al. [228] that do not prioritize individual areas but instead reveal the global distribution patterns of species richness, endemism and threatened species. These three databases are not used as part of the risk map but in the regionalization step to evaluate and distinguish between the non-prioritized areas (see chapter 5.1.4). Out of the 15 remaining schemes, the Megadiversity Countries [229] scheme is based on national and political criteria since it shows the countries with the highest richness of endemic species. It is therefore not included in the analysis. Three global conservation schemes, the Crisis Ecoregions [91], the Endemic Bird Areas (EBA) [230,231] and the Important Plant Areas [230], met all inclusion criteria, yet their data were not freely available. Therefore, they could not be considered in the development of the risk map. However, should this change in the future, it would be advisable to include them as well. The World Database on Protected Areas, can be classified as proactive map since it shows biodiversity areas with low vulnerability [197]. Yet this map is based on management decisions and criteria and is therefore used as a data basis for the local scale for the management parameter set-aside areas (in chapter 5.2.4). Having sorted of, 8 global conservation schemes for the calculation of the biodiversity risk maps, there are still 10 remaining conservation schemes of which 5 can be classified as reactive and 3 schemes as proactive. The remaining 2 schemes only consider irreplaceability criteria (see Table 6 at the end of this chapter).

5.1.1.1.1 Reactive conservation schemes

In the following the schemes that can be classified as reactive are described. These include the Biodiversity Hotspot Scheme, which was identified and revised by several authors [90,99,232]. Biodiversity hotspots focus on endemic species and large habitat losses. The criterion for classification as a reactive site is that more than 70% of the habitat (defined as area with primary vegetation) has already been lost [99]. With regard to habitat loss, this scheme includes highly vulnerable biodiversity areas. In addition, it shows the sites where endemic species occur and thus irreplaceable biodiversity risk areas. Most biodiversity hotspots are still unprotected, so that land use activities can take place in these areas [90,99,232]. This underlines the importance of considering biodiversity hotspots as part of the global impact assessment for biodiversity in the life cycle assessment. With regard to taxa, the Biodiversity Hotspot scheme concentrates on the occurrence of endemic vascular plants, but also on vertebrate species (excluding fish). The organizational levels of biodiversity that are covered by biodiversity hotspots are species and ecosystems. The areas with biodiversity hotspots are shown in Figure 21.

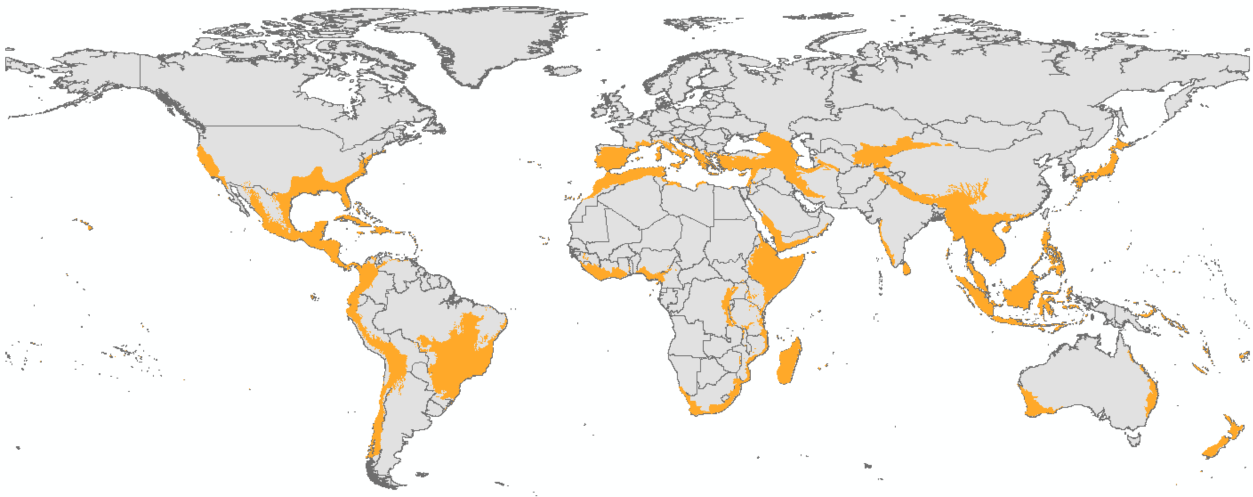


Figure 21: Biodiversity Hotspots (revisited 2016) [90,99,232]

The Key Biodiversity Areas (KBAs) are area-based programs for the protection of biodiversity that cover aspects of irreplaceability and high vulnerability. Vulnerability indicators include the IUCN Red List criteria for threatened species. Irreplaceability indicators include species in restricted areas, community species and bio-medically restricted species. Within a KBA all essential sites are included that are necessary for the preservation of vital populations of specific species [233]. The size of the individual KBAs is rather small compared to other conservation schemes. It builds on the scheme of Important Bird Areas by including more taxa such as mammals, reptiles, amphibians, vascular plants, conifers, algae, fungi, lichens, liver moss or moss. With regard to the organizational scale, the KBA focuses on the species and the population level. However, KBAs do not consider further ecological or conservation indicators such as endangered or unique ecosystems, large intact ecosystems [234], or the genetic or phylogenetic level. Since the KBA approach integrates several protection concepts such as the Important Bird Areas, Alliance for Zero Extinction Areas or Important Plant Areas (all with different foci), these data are included separately. All remaining KBAs that are not included in the IPA and AZE areas are shown in Figure 22.

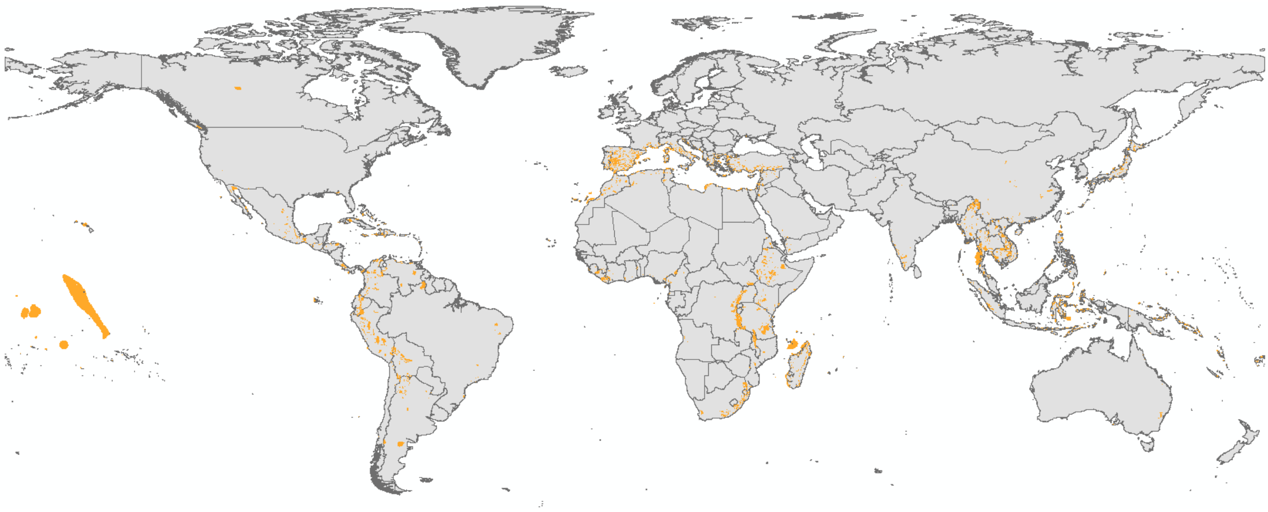


Figure 22: Key Biodiversity Areas not included in IPA and AZE sites [230]

The Alliance for Zero Extinction data set represents areas worldwide that contain 95% or more of the remaining population of a threatened species and is continuously developed by the Alliance for Zero Extinction (AZE, Alliance for Zero Extinction 2003). An area must meet all three of the following criteria to qualify as an AZE site: It must comprise a minimum of one endangered (EN) or highly endangered (CR) species that is on the IUCN Red List, the predominant known residual population (>95%) of EN or CR species or their biographical segment and a definable boundary within which the character of the site differs from that of the wider environment [235]. As a result of these strict criteria, AZE areas are the most accurate and smallest of all identified global schemes. So far, AZE sites for mammals, birds, some reptiles, amphibians, conifers and reef corals have been identified. Like IPAs and IBAs, AZE sites are intended to be an instrument for implementing nature conservation at the national level and are grouped under the umbrella concept of the KBAs. To date, 587 AZE areas for 920 different species have been classified worldwide [236,237]. AZE sites concentrate on the species level by prioritizing highly endangered species. Therefore, their focus is on high vulnerability (reactive) and irreplaceability. Since they concentrate exclusively on the species level, neither genetic diversity nor intact terrestrial ecosystems are taken into account. They are shown in Figure 23.

Figure 23: AZE sites (sites status confirmed) [237]

The data set Important Bird and Biodiversity Areas (IBAs) represents essential habitats for birds and their habitats worldwide and is continuously further developed by the nature conservation partnership BirdLife International.

The internationally agreed upon criteria for qualification as an IBA are that the area contains a significant number of a globally endangered species or important breeding sites for endemic species (also called Endemic Bird Area (EBA) or Secondary Area (SA)) or a species whose distribution is largely or completely limited to one biome and whose communities support 1% of the world's population. As with KBAs, the IBA site scale is open to practical conservation measures and is used worldwide as an instrument to protect biodiversity. However, only around 40% of all IBAs have received some official protection. To date more than 12,000 IBAs have been identified, all of which are also classified as KBAs.

Although birds are the only indicator of IBAs, they are often representative of greater biodiversity and are likely to be used as proxy for other animal and plant species. In 2013 the IBAs were retitled from *Important Bird Areas* to *Important Bird and Biodiversity Areas* to reflect this link [238]. Like the KBAs, the IBAs are part of the reactive systems with indicators of irreplaceability. Neither the ecosystem nor the genetic level are taken into account. They are shown in Figure 24.

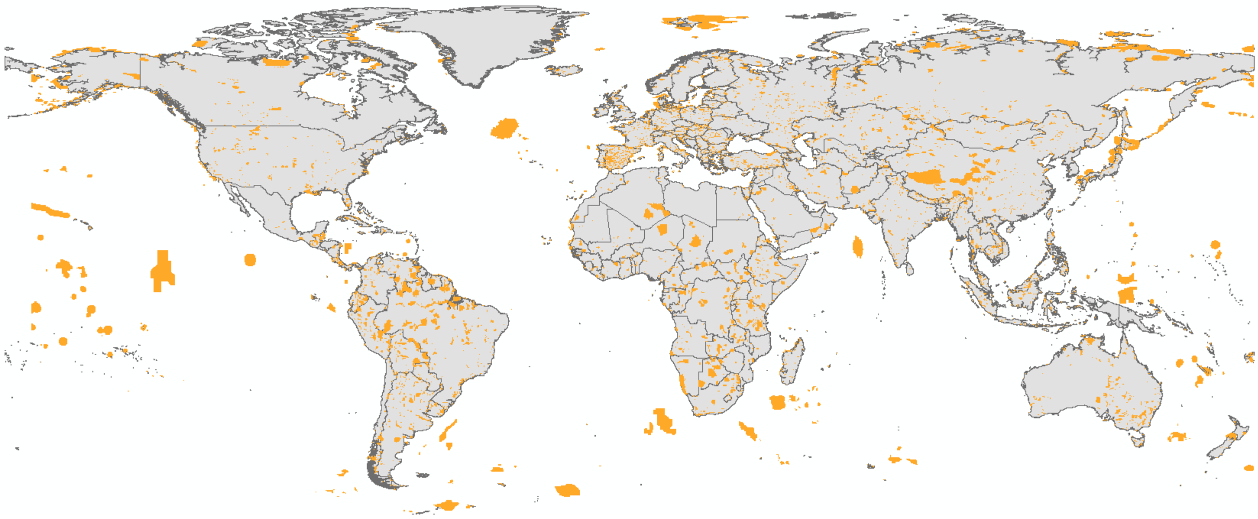


Figure 24: IBA sites (status confirmed) [230]

The global data set Evolutionarily Distinct and Globally Endangered (EDGE) zones developed by Safi et al. [239] show the distribution and threat of mammalian and amphibian species with a "unique" evolutionary history [239]. EDGE were launched as part of the Existence programme by the Zoological Society of London in 2007 [92]. Evolutionarily distinct species have few close relatives on the tree of life and thus an unusual genetic diversity, measured as phylogenetic difference between species. Such species have been developing independently for millions of years, and their extinction would lead to the disappearance of species with very unique genetic characteristics. EDGE species are evolutionarily distinct species with one of the three "threatened" IUCN statuses. The spatial prioritization EDGE mammalian and amphibian species was mapped as "zones“ [239]. Based on the EDGE score system, a global range pattern with a raster resolution of 25 x 25 km to 200 x 200 km was projected. EDGE priority zones of mammals and amphibians are distributed very differently worldwide. This underlines the importance of considering as many taxa as possible, as "representative" species cannot reflect this difference otherwise. In the future. EDGE of Existence also plans to develop and update further EDGE zones for terrestrial vertebrates, marine mammals, reptiles, corals and sharks. Those EDGE zones are thus biodiversity risk areas at species and genetic level, with an emphasis on high irreplaceability and high vulnerability (reactive) (see Figure 25).

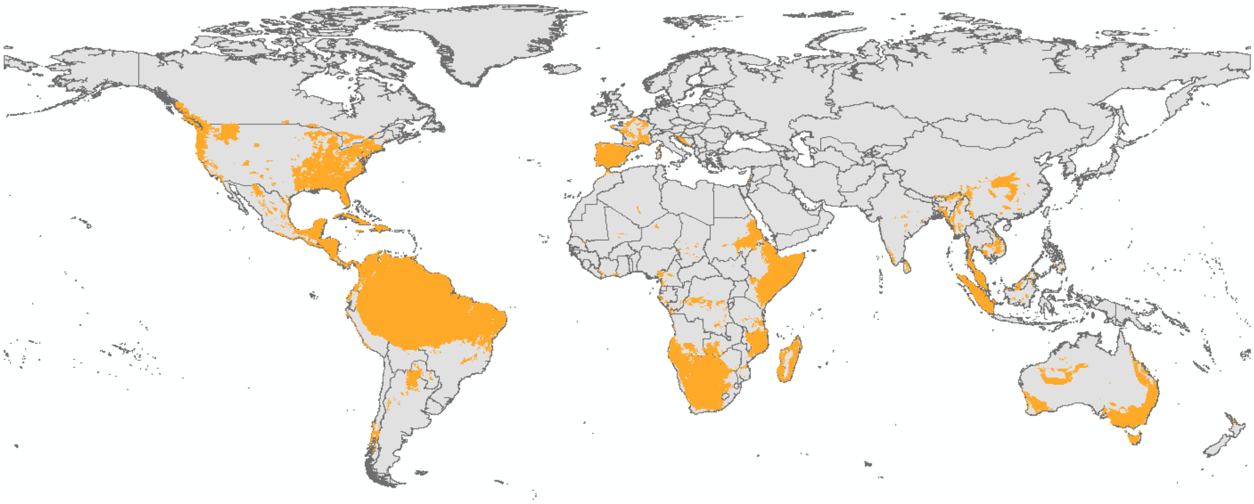


Figure 25: EDGE Areas (resolution 25 km) [239]

5.1.1.1.2 Irreplaceability conservation schemes

The Global 200 ecoregions belong to the risk areas of biodiversity which cannot be divided into reactive or proactive approaches since they take into account indicators of the irreplaceability of biodiversity. They include large areas that focus on ecological diversity and include outstanding and unique habitats that are representative of the ecosystem [86]. The criteria for classification as a Global 200 Ecoregion include ecosystems with a high species richness or endemic characteristics, ecological or evolutionary phenomena that are exceptional on a global scale (e.g. large migratory routes of vertebrate species) or the rarity of specific habitats and ecosystems [86]. They are not as detailed and precise as, for example, the KBAs (see Figure 26). Instead they draw more attention to important biodiversity sites and underline their significance for decision making [86].

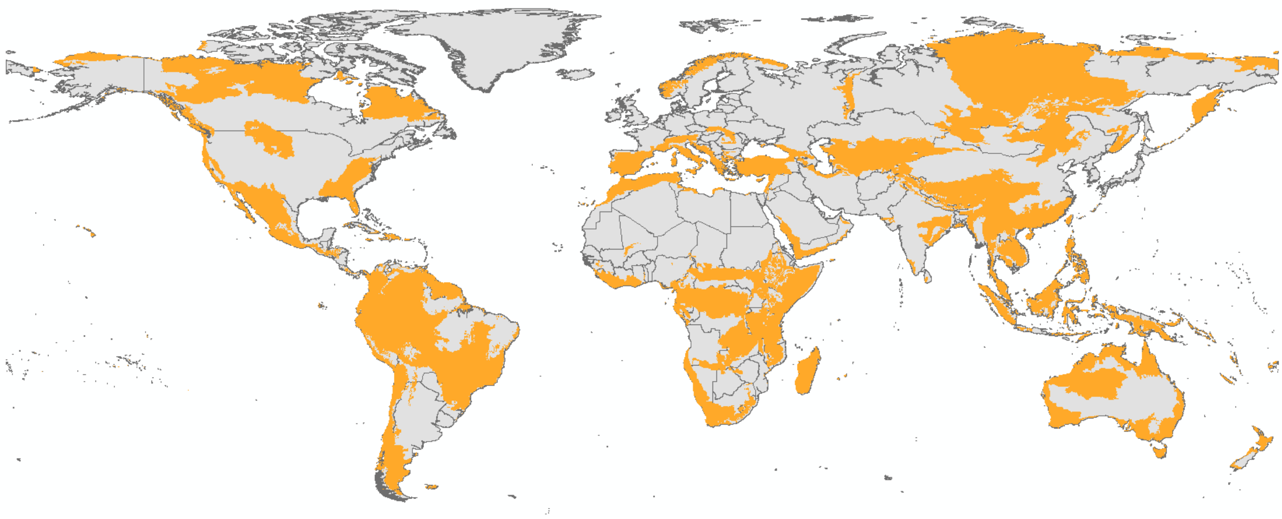


Figure 26: Global 200 Ecoregions [86]

The Centers of Plant Diversity (CPD) is another conservation scheme that does not consider vulnerability measures but focuses on a high degree of irreplaceability. CPD has 234 sites worldwide which are of great importance for the global diversity of vascular plant species. The UNEP-WCMC [240] points out that if these areas were protected, the largest number of plant species worldwide would be preserved. The criteria for classification as a CPD are a high species richness in the area and a high degree of endemicity of the plant species. In addition, sites must host an important gene pool of plant species that might be also of value to humans [240]. This makes CPD one of the few conservation schemes that takes into account the genetic level of biodiversity. As the CPD also includes unique habitat types, all three levels of biodiversity are included with respect to the taxa of vascular plants. Given their importance, NGOs explicitly advised against extractive land use activities in such CPD areas [159,240]. The CPD map shows that it is of utmost importance to include vulnerable and irreplaceable biodiversity areas in the risk map. If, for example, only endemic vascular plants (as part of the CPD map) are considered within the framework of a LCIA method, land use in the Congo Basin or the Amazon forest would be undervalued, although they are one of the areas with the highest biodiversity in terms of species richness and still offer largely intact ecosystems (see Figure 27).

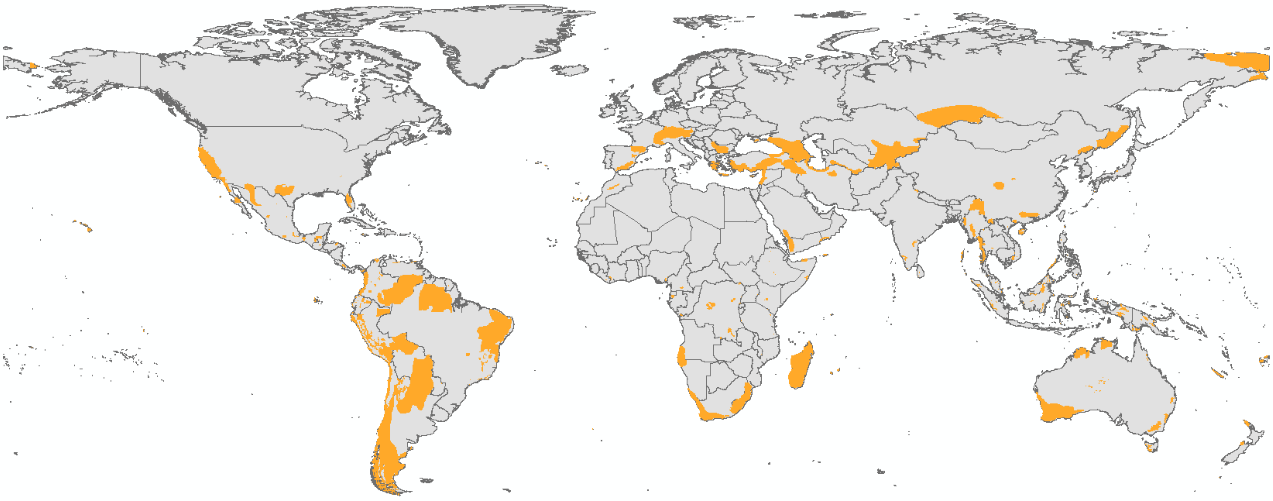


Figure 27: Centers of Plant Diversity [240]

5.1.1.1.3 Proactive conservation schemes

High Biodiversity Wilderness Areas (HBWAs) are areas with low anthropogenic pressure and therefore belong to the proactive conservation schemes. The criteria for classification as a HBWA are a minimum size of 10,000 km², a low human population density and at least 70% of an area must accommodate a natural soil and historical habitat as it was 500 years ago. HBWAs consist of a total of 24 species-rich wilderness areas with 18% endemic plant species and 10% of all global vertebrate species. For inclusion in the LCIA method, it is important that only 7% of all HBWAs are legally protected [84]. This means that future land use in these areas is likely to happen. The delineations of HBWAs are characterized only by population density and natural habitat (see Figure 28). The fragmentation and perforation of ecosystems caused by roads and infrastructure is not taken into account, as is the case with intact forest landscapes.

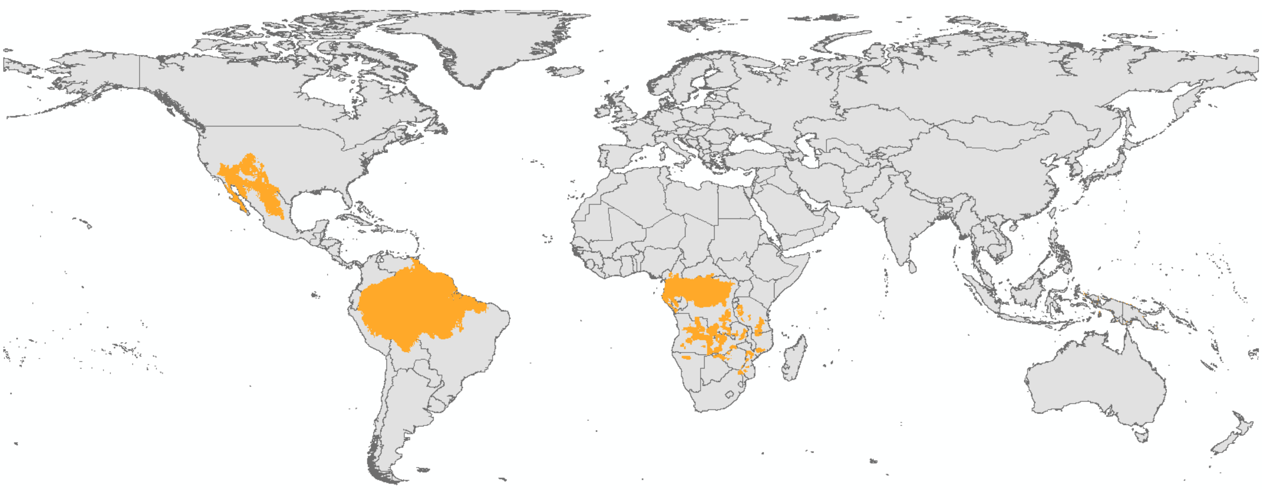


Figure 28: High Biodiversity Wilderness Areas (HBWAs) [84]

Intact forest landscapes (IFLs) cover areas with natural forests or naturally treeless areas (e.g. grassland or shrub land) that are under very low human pressure and activity. They are therefore proactive schemes with a low vulnerability. An IFL must be large enough to accommodate viable populations of native biodiversity with criteria at the ecosystem level without focusing on individual taxa. The area of an IFL must be at least 500 km², while its width must be at least 10 km [241–243]. The map for the classification of an IFL was developed by analyzing satellite images and the buffering of all disturbed areas (e.g. roads or settlements). Thus, an IFL is characterized by a high connectivity of an undisturbed ecosystem (see Figure 29). IUCN calls on governments and the private sector to preserve these undisturbed areas and primary habitats [244]. Land use activities in these areas should therefore be monitored and strictly prevented, which is why they are particularly important for the assessment of the land use flows of transformation in the LCIA method.

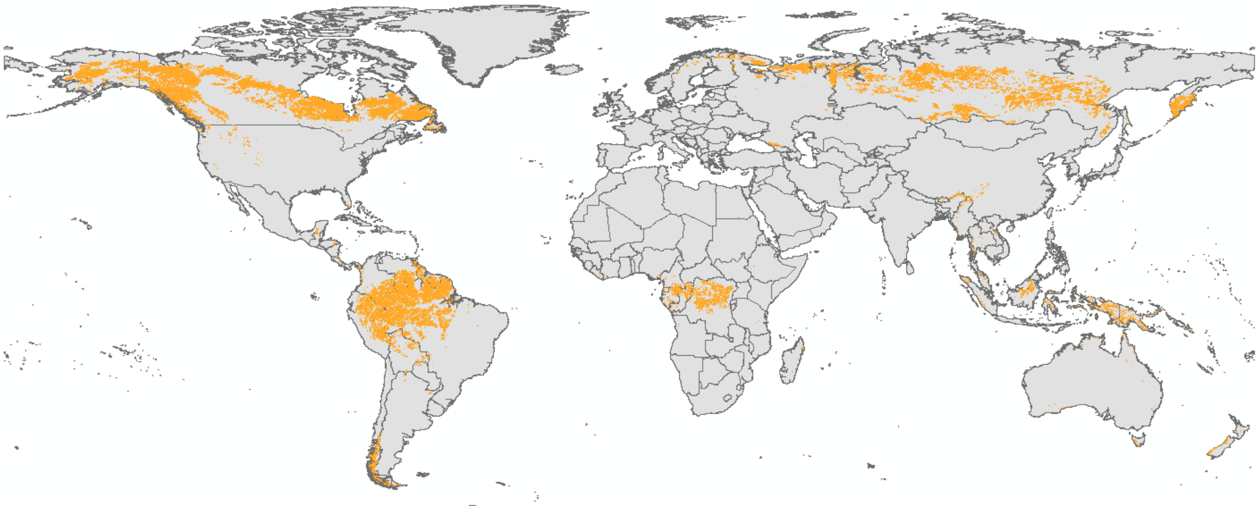


Figure 29: Intact forest landscapes [241,242]

The global data set Last of the Wild (LTW) was developed by the Wildlife Conservation Society (WCS) and the Center for International Earth Science Information Network (CIESIN) of Columbia University to identify the biomes with the least human impact worldwide. To identify the LTW areas, the Human Influence Index (HII) was developed to assess human influence at the global level, based on indicators such as population density, roads, navigable rivers and various agricultural land uses. Within each biome, the 10% least affected areas were identified by the HII. From these areas, the 10 largest contiguous areas were selected and classified as Last of the Wild areas, ranging from 5 km² to over 100,000 km² [241,242]. This scheme has been updated on several occasions, such as 2002 and 2005.

LTW areas are intact habitats with very low vulnerability which do not focus on the threat or occurrence of a particular species. Rather, they are relatively undisturbed ecosystems, typically large regions. Some areas are legally protected, others are under anthropogenic influence. Yet overall, less than 19% of all LTW areas have some legal protection, making most of them vulnerable to land transformation. The map with the last of the wild areas illustrates why it is important to specifically further look at the location of land use and the way of land use. Without overlaying the land use areas with the risk areas, the biodiversity risks in countries with a high proportion of anthropogenically unused land, such as desert areas in North Africa, or large parts of primary forests in Canada, would be overestimated because they have a high proportion of intact ecosystems. This means that if only the currently used areas (occupation) or future and historical land use areas (transformation) are assessed with regard to their biodiversity risks by regionalization, the other unsuitable areas are automatically excluded (see Figure 30).

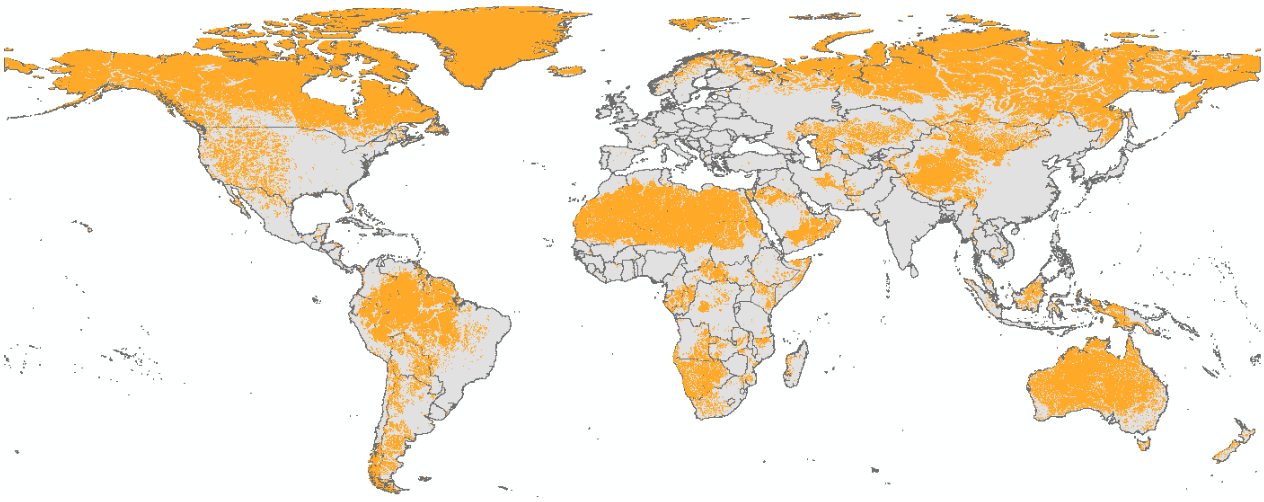


Figure 30: Last of the Wild (2005) [87]

5.1.1.2 Summary of global conservation schemes

In summary, three of the ten global conservation schemes concentrate exclusively on proactive areas at the ecosystem level. Their focus is on pristine and intact ecosystems, with HBWA placing an additional emphasis on intact ecosystems that are also rich in species diversity. HBWAs are also important for vascular plant species and vertebrates. The other proactive protection schemes do not focus on specific taxa but try to preserve intact ecosystems as a whole. Mittermeier et al. [84] argue that conservation efforts should be proactive wherever possible. Ultimately, it depends on the question and the location of land occupation or land transformation in a life cycle assessment. When dealing with land occupation, land use in reactive schemes are essential to be evaluated in order to mitigate the impact on biodiversity in these areas under high threat. Land transformation, however, has a greater impact on intact ecosystems, which is why proactive strategies are important. As shown, most conservation schemes focus on the reactive approach of biodiversity conservation, which explains the somewhat peculiar distribution of risk areas on the world map. For example, the Congo Basin is often not prioritized in a conservation scheme due to its relatively intact ecosystems. The focus of reactive areas is above all on the species level. Therefore, the inclusion of these reactive schemes covers critical areas for a wide range of taxa such as birds, mammals, reptiles, amphibians, vascular plants, conifers, algae, fungi, lichens, liver blossoms, and mosses in terms of endemism, threat level and number of species. Two conservation schemes also consider genetic aspects such as the CPD and EDGE sites, with a focus on vascular plants and mammals or amphibians. The inclusion of crisis ecoregions and important plant areas developed by Hoekstra et al. [91] would greatly benefit the priority biodiversity maps, as in the case of crisis ecoregions only a few schemes indicate reactive areas at the ecosystems level. However, the data are not freely accessible and therefore cannot be included in the biodiversity risk map.

A summary of the schemes can be found in Table 6.

Table 6: Biodiversity conservation schemes at a global scale based on Maier et al. [5]

Reference	Name	Creation date (update)	Organizational level	Taxa	Vulnerability*	Irreplaceability	Included in risk map
[90,99,232]	Biodiversity Hotspot (BH)	2016 (1999)	Ecosystem, species	Vascular plants	High	High	yes
[245,246]	Key Biodiversity Areas (KBA)	2014	Species, ecosystems	Birds, mammals, reptiles, amphibians, vascular plants, conifer, algae, fungi, lichens, liverworts, mosses, etc.	High	High	yes
[91]	Crisis Ecoregions (CE)	2005	Ecosystem	No focus	High	High	no
[84]	High Biodiversity Wilderness Areas (HBWA)	2002	Ecosystem	Vascular plants, vertebrates	Low	High	yes
[87]	Last of the Wild (LtW)	2002	Ecosystem	No focus	Low	-	yes
[85,241,242]	Intact Forest Landscapes (IFL)	2013 (1997)	Ecosystem	No focus	Low	-	yes
[230]	Important Bird Areas (IBA)	2014 (1980)	Ecosystem, species	Birds	High	High	yes
[230]	Important Plant Areas (IPA)	1995	Species, ecosystem	Vascular plants, algae, fungi, lichens, liverworts, mosses	High	High	no
[86]	Global 200 Ecoregions	1998	Ecosystem, species	No focus	-	High	yes
[230,231]	Endemic Bird Areas (EBA)	1998	Ecosystem, species	Birds	-	High	no
[89]	Centers of Plant Diversity (CPD)	2013 (data 1994–1997)	Species, genes	Vascular plants	-	High	yes
[236]	Alliance for Zero Extinction (AZE)	2005	Species	Birds, mammals, reptiles, amphibians, conifers	High	High	yes
[239]	Evolutionarily Distinct and Globally Endangered (EDGE)	2013	Species, genes	Mammals, amphibians	High	High	yes
[229]	Megadiversity countries	1997	Species	Non-fish vertebrates and higher plants	-	High	no
[197]	Protected areas	2018	Ecosystem	No focus	Low	-	no
[228]	Threatened Species	2013	Species	Mammals, amphibians, birds	High	-	no

Refer-ence	Name	Creation date (update)	Organi-zational level	Taxa	Vulner-ability*	Irre-placea-bility	Included in risk map
[228]	Species richness	2013	Species	Mammals, amphibians, birds	-	High	no
[228]	Endemic species	2013	Species	Mammals, amphibians, birds	-	High	no

* vulnerability high =reactive, low= proactive

5.1.1.3 Global risk map: data preparation and cleaning

In order to produce a uniform global risk map, some prior data preparation and cleaning of the maps must take place. All spatial calculations are performed with RStudio using the raster, sp and rgdal packages [247]. The maps are visualized with QGIS and ArcGIS. The geodatabase of biodiversity hotspots (version of 2016) is queried in QGIS via an SQL filter only for terrestrial areas, "TYPE" = 'hotspot_area', as the focus of this work is on terrestrial biodiversity. The EDGE map for the phylogenetic diversity of mammals and amphibians contains several resolutions within one layer, with the lowest resolution of 25 km being queried. The KBA protection map is queried after "IbaStatus" = "confirmed" or "candidate", "KbaStatus" = "confirmed" and "AZE-Status" = "confirmed" or "candidate". In order to display and integrate all three different schemes individually, the three queries are saved as individual vector layers. All priority areas are converted into a binary format with 1 = priority area and 0 = no priority area and then into a raster format with the same resolution of 0.25° x 0.25° of the land use map of Hurtt et al. [186] and the land use map of Kehoe et al. [248] with a resolution of 1 km, as these maps are used in the regionalization step. Therefore, the land use map of Kehoe et al. [248] is divided into the broad land use types, based on Newbold et al. [103]. The map with the lower resolution is used to assess global biodiversity risks in the background system to calculate average values per country and the map with the higher resolution is used to assess risks in the foreground system. All "no data" are set to zero to represent the grid cells without biodiversity risk sites. All maps are projected with the function *projectRaster()* in R into the same geographical coordination system WGS 84. The "nearest neighbor" method is used as a resampling method, since it retains the original values of all raster cells. The workflows for proactive and reactive areas (including irreplaceable areas) are shown in Figure 31 and Figure 32.

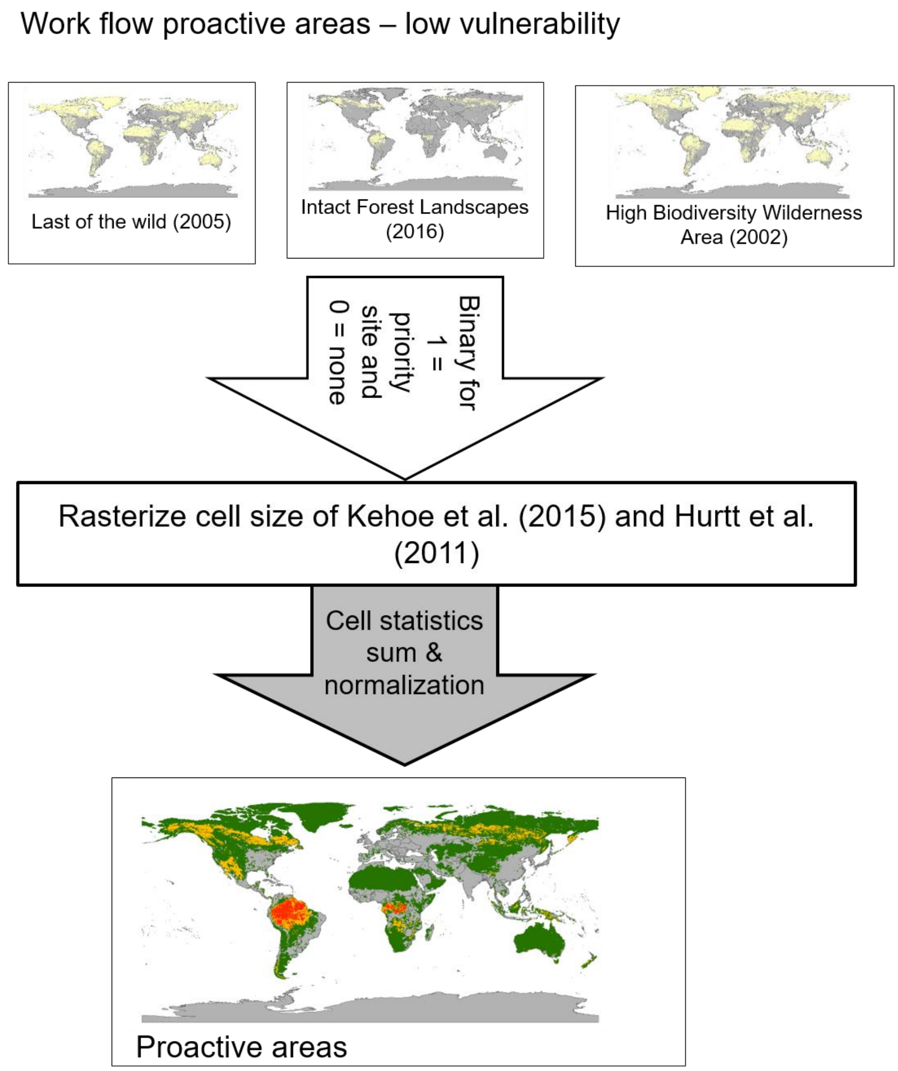


Figure 31: Work flow of proactive areas

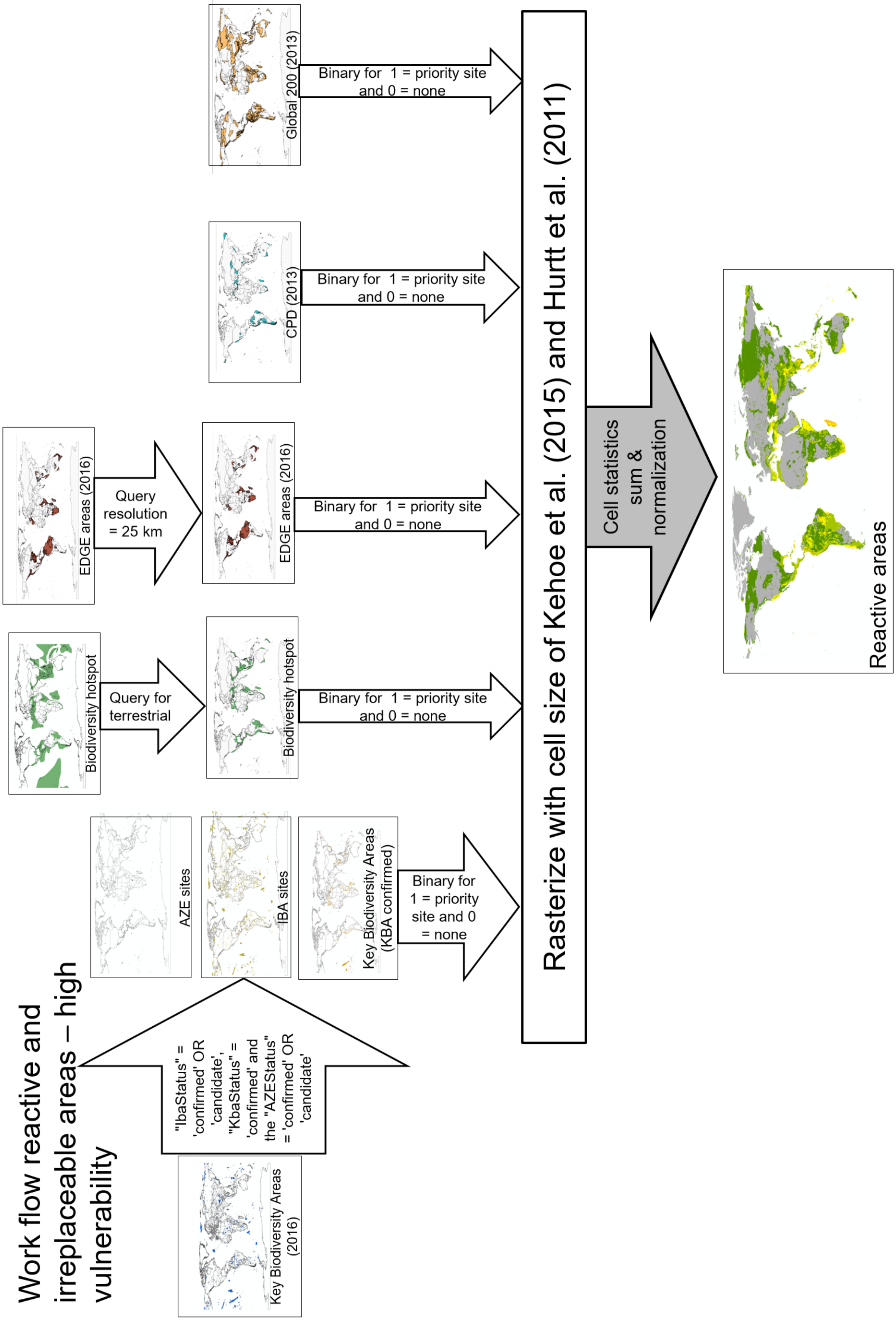


Figure 32: Work flow of reactive and irreplaceable areas

5.1.1.4 Biodiversity risk map: GIS calculations

Three maps are developed for assessing the risk to biodiversity, the reactive biodiversity risk map, the proactive biodiversity risk map and the unified biodiversity risk (UBR) maps. The schemes containing only irreplaceability (here: Global 200 Ecoregions and CPD) are integrated into the reactive map. This is due to the fact that most reactive schemes include some aspects of irreplaceability, while proactive schemes do not include them at all. For the reactive biodiversity risk map, an overlay analysis is performed for the seven conservation schemes of the Biodiversity Hotspot Scheme, the Global 200 Ecoregions, CPD and EDGE zones, IBAs, KBAs and AZE. For the proactive biodiversity risk map, an overlay analysis is performed for the conservation schemes of HBWA, IFL and LTW areas (see Figure 33).

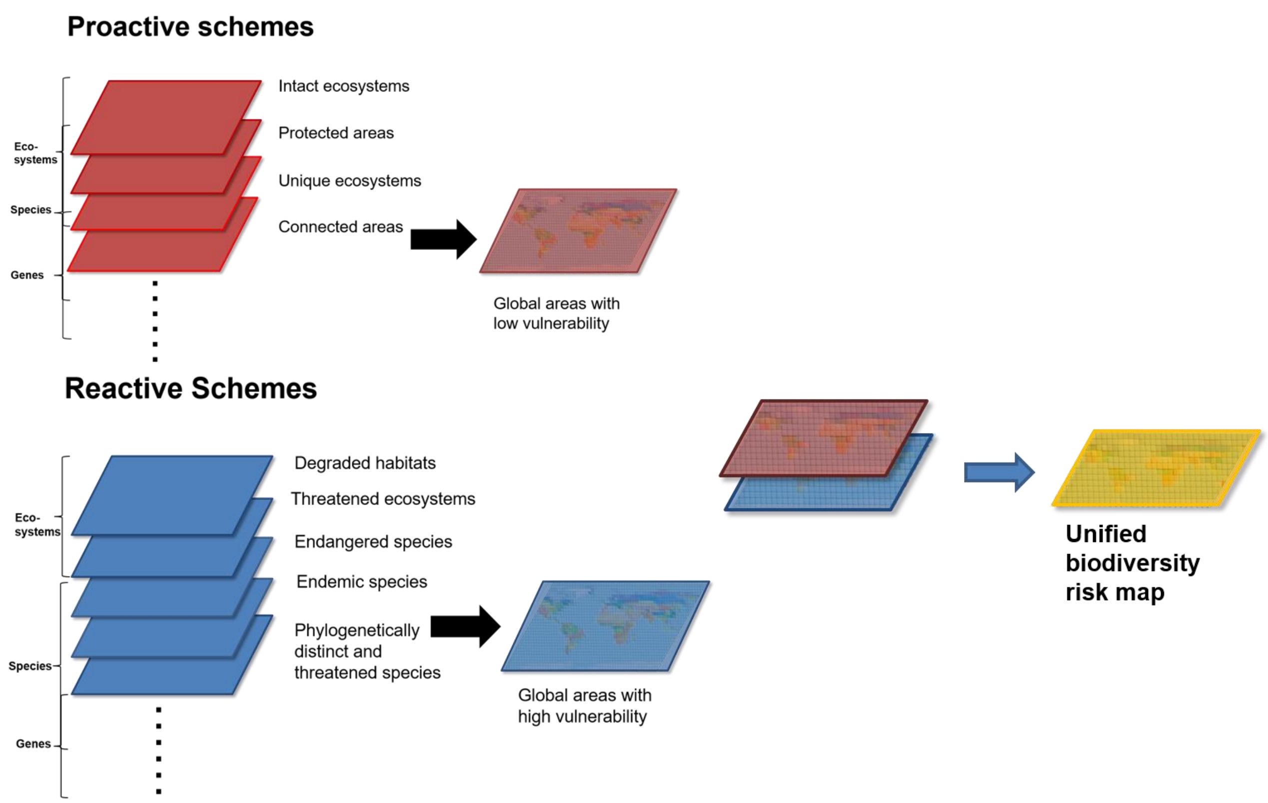


Figure 33: Overlay analysis of proactive, reactive and irreplaceable conservation areas

Afterwards, the proactive and reactive schemes are added up as the sum of cell values (normalized by the number of schemes) to take into account the overlapping proactive and reactive sites and to obtain a single UBR map. The resulting values can be calculated both for the average per country (background system) and for individual raster cell values (foreground system). The areas with the highest degree of overlap, i.e. the largest number of both proactive and reactive areas, have the highest protection priority and thus a high biodiversity risk, after Brooks et al. [76]. Grid cells in which we have both reactive and proactive and irreplaceable protection schemes receive the highest value and thus the highest priority. The lowest priority is given to cells if there is no reactive or proactive conservation scheme.

5.1.2 Results of global biodiversity risk maps

Figure 34 depicts the reactive biodiversity risk map, which shows important biodiversity areas from a high vulnerability perspective. Areas with a high number of reactive conservation schemes are located in large parts of Latin America, sub-Saharan Africa, Southeast Asia, Australia in the Pacific region and Europe's Mediterranean region. All these areas are home to either endemic species (as in Spain), phylogenetically unique species (as in Australia) or harbor a large number of endangered species (as in Indonesia). As all these areas belong to the reactive schemes, land use in these areas is very likely.

In Southeast Australia, Cuba or New Guinea, for example, there are species that have extremely rare phylogenetic lineages and at the same time are highly endangered. The Western long beaked echidna, for example, only occurs in a certain part of the island of New Guinea. It is directly threatened by the loss of its habitat through anthropogenic activities such as mining, logging and agriculture and is the last of its kind [158,249]. A life cycle assessment should therefore recommend avoiding anthropogenic activities in such areas or at least carrying them out with special care.

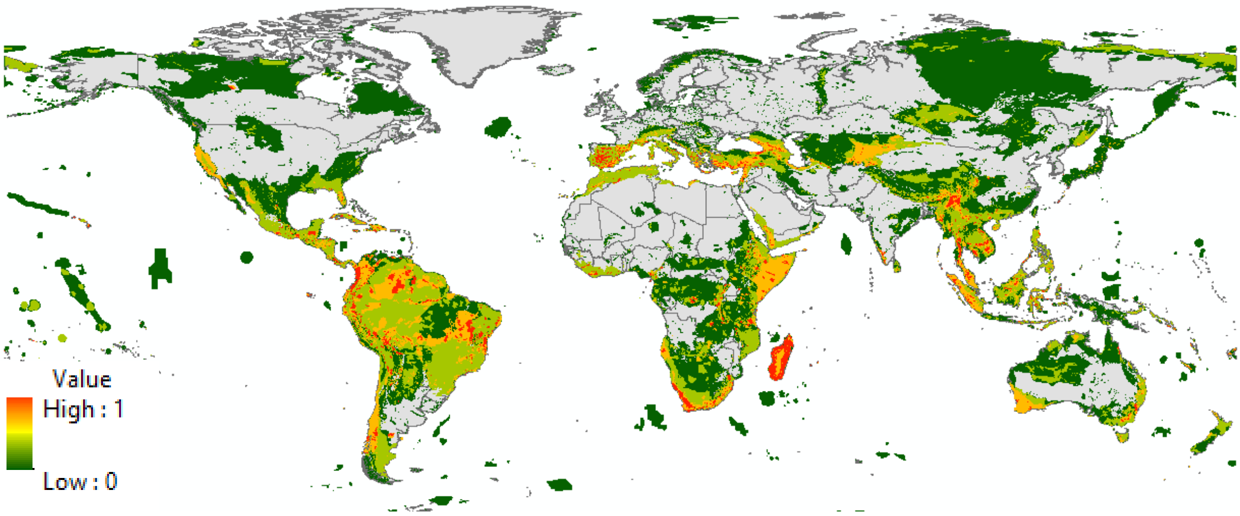


Figure 34: Global biodiversity risk map depicting the extent of reactive risk sites

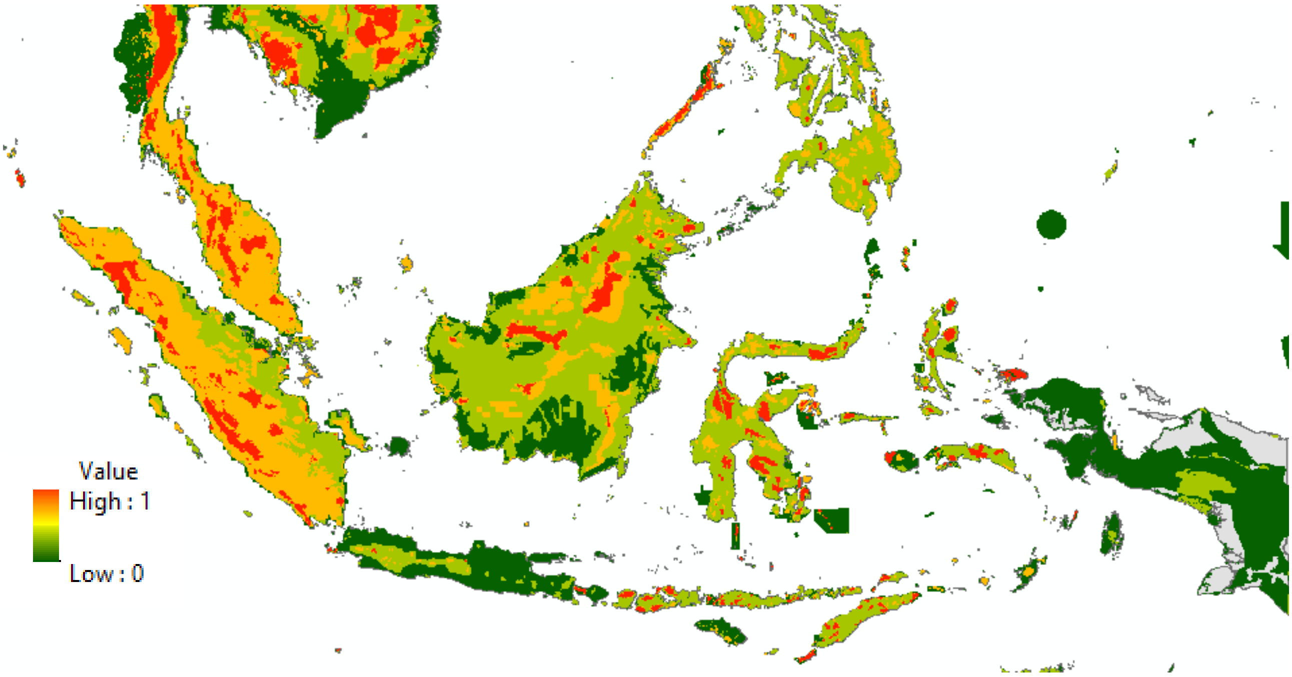


Figure 35: Reactive risk sites in Southeast-Asia

The reactive biodiversity risk map and proactive biodiversity risk map can be aggregated by region, illuminating regions around the world from different angles. By aggregating the RBR areas per ecoregion, transnational areas are highlighted. These include the ecoregion in Madagascar, Southeast Asia and the Iberian Peninsula. On the other hand, there are low-risk regions for biodiversity such as Scandinavia, the Sahara and most of North America (north of the Rio Grande).

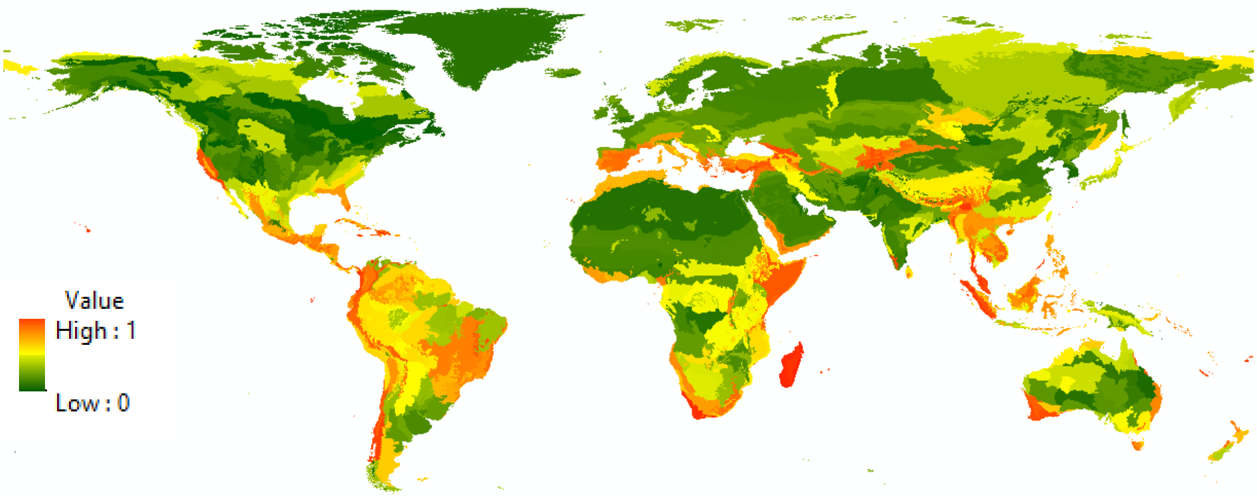


Figure 36: Reactive biodiversity risk map aggregated per ecoregion

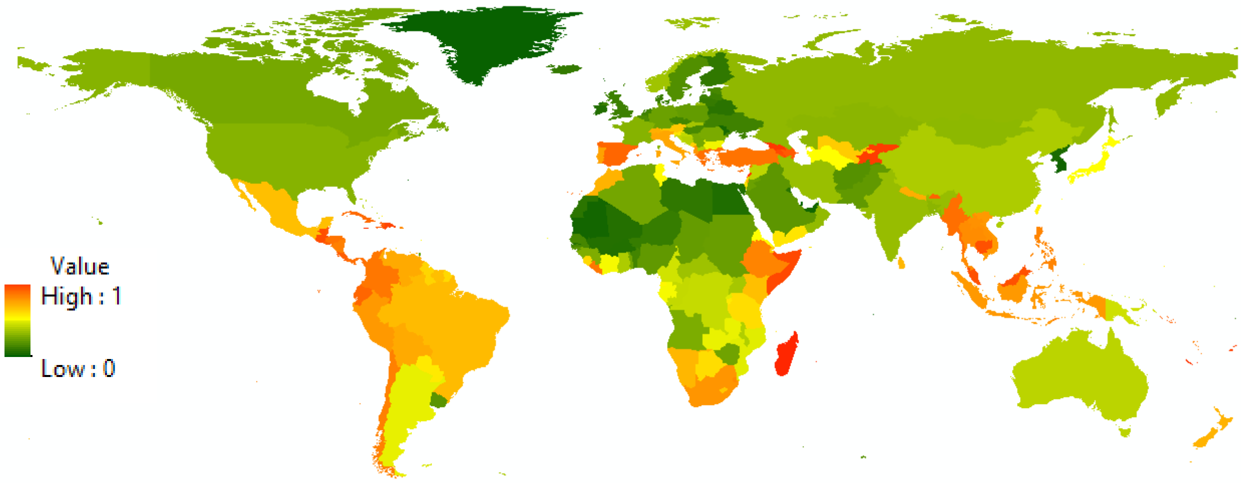


Figure 37: Reactive biodiversity risk map aggregated per country

The most important risk areas of the proactive biodiversity risk map are shown in Figure 38. This map shows areas with large, intact and undisturbed regions such as the tropical rainforests of the Congo, the island of New Guinea and the Amazon Basin as well as forests that include wilderness areas with valuable biodiversity and intact forest landscapes.

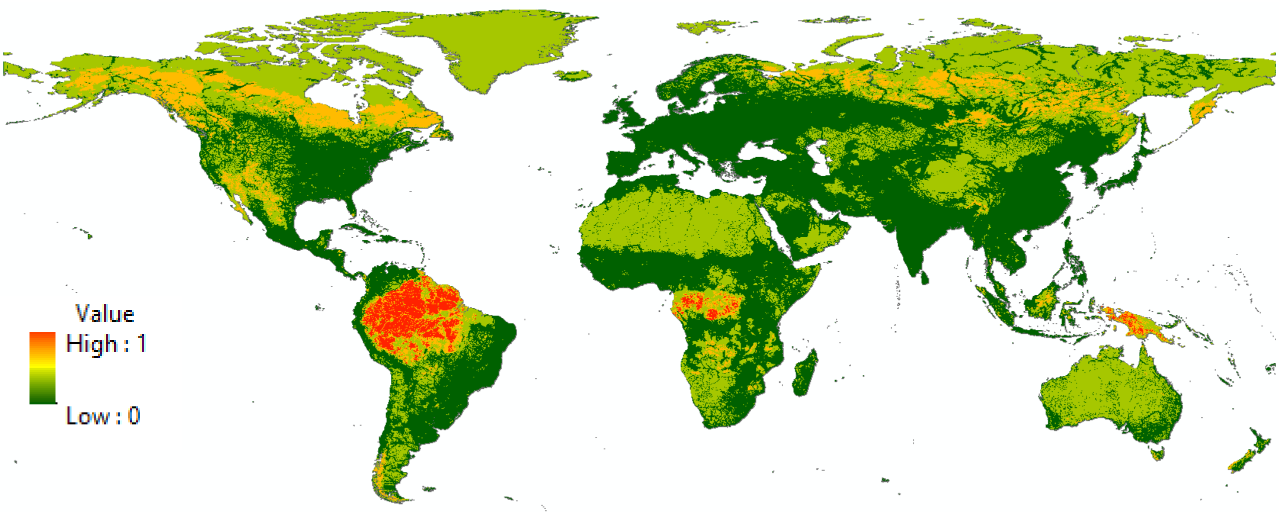


Figure 38: Spatial extent of proactive biodiversity risk areas

Looking at the proactive and reactive biodiversity maps, their complementary nature becomes evident. The region of Southeast Asia illustrates this fact: Southeast Asia is home to many reactive risk areas such as the Biodiversity Hotspot, AZE sites, KBAs or EDGE sites. For example, the Malay Peninsula and Sumatra are shown as important areas on the reactive biodiversity risk map, while the island of New Guinea is of minor importance. New Guinea still has intact ecosystems and only a few endangered species. However, this does not mean that biodiversity in New Guinea is not at risk – on the contrary, it still contains fairly intact ecosystems and is therefore not included in the reactive biodiversity risk map but instead in the proactive map. Only a unified biodiversity risk (UBR) map can therefore reveal the whole picture of biodiversity around the world (Figure 39). The UBR map contains all organizational levels of biodiversity with a combination of ecological and conservation indicators. At species level, it contains representative biodiversity risk areas for various taxa such as birds, mammals, reptiles, amphibians, vascular plants, conifers, algae, fungi, lichens, liver worms and mosses. In addition, areas with an important gene pool and phylogenetically diverse and threatened species, such as regions in South Africa or South West Australia, are highlighted. Intact and rare ecosystems are also part of the UBR map, as they still occur in the Congo Basin or the Amazon forest.

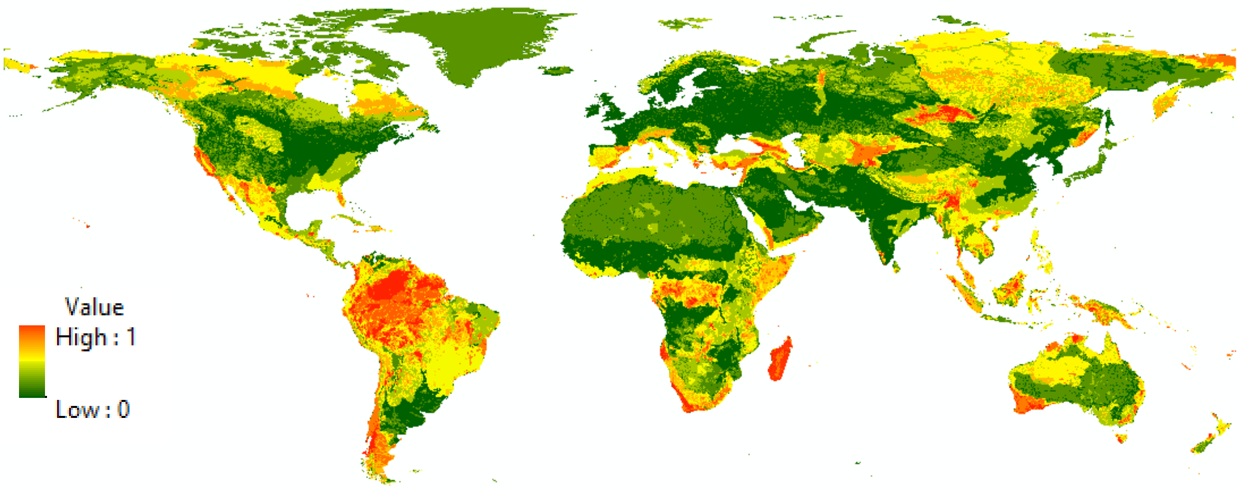


Figure 39: Spatial extent and overlap of both reactive and proactive biodiversity risk areas

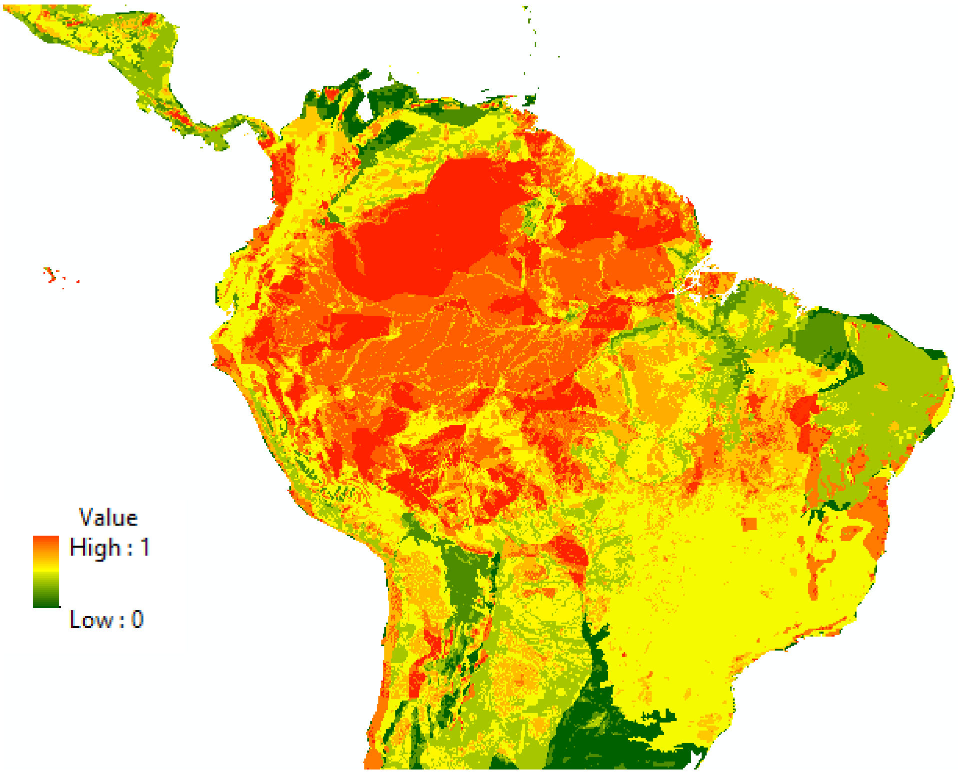


Figure 40: Reactive and proactive biodiversity risk areas in South America

Figure 41 shows countries that on average have a particularly high number of proactive and reactive risk areas and thus a higher global risk for biodiversity. These aggregated risk values can be used for the value chain if the precise location of land use is unclear, but only the country of origin is known [5]. With regard to the reactive biodiversity risk map, land use should be shifted to land use types with higher biodiversity quality (transformation) and land management should be improved to reach the upper range of the impact interval (occupation). The proactive biodiversity risk map shows where land use of all kinds should be avoided at all costs in order to protect our intact ecosystems. Finally, the UBR map shows all areas where land use poses a risk to biodiversity. Based on the UBR map, a global risk factor can be calculated so that each spatial unit of analysis receives a value for the impact of biodiversity.

However, in order to determine the actual risks depending on the location of land use, it is first necessary to analyze where what types of land use takes place in a country. For this purpose, land use models are used that have identified the locations of different types of land use at the global level. These are used to analyze whether a type of land use (e.g. cropland or pasture) falls within a biodiversity risk area or not and how many reactive and proactive areas are affected. This is conducted in the next step of regionalization.

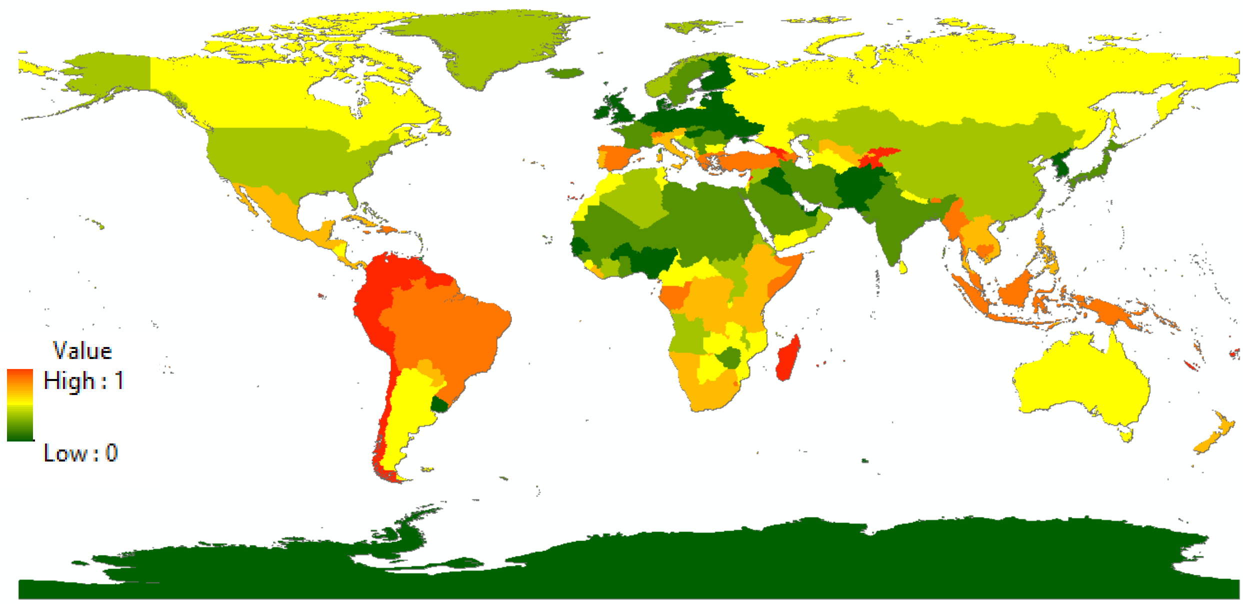


Figure 41: Reactive and proactive risk areas per country

5.1.3 Gap analysis

As can be seen in Figure 42 in orange and as it has been highlighted by Brooks et al. [76], about 20% of the total terrestrial area is not covered by any risk area (in orange). Therefore, these areas would be given a lower risk rating in the LCIA method. However, also these areas do not have the same distribution and "value" of biodiversity, but differ in the degree of risk to biodiversity.

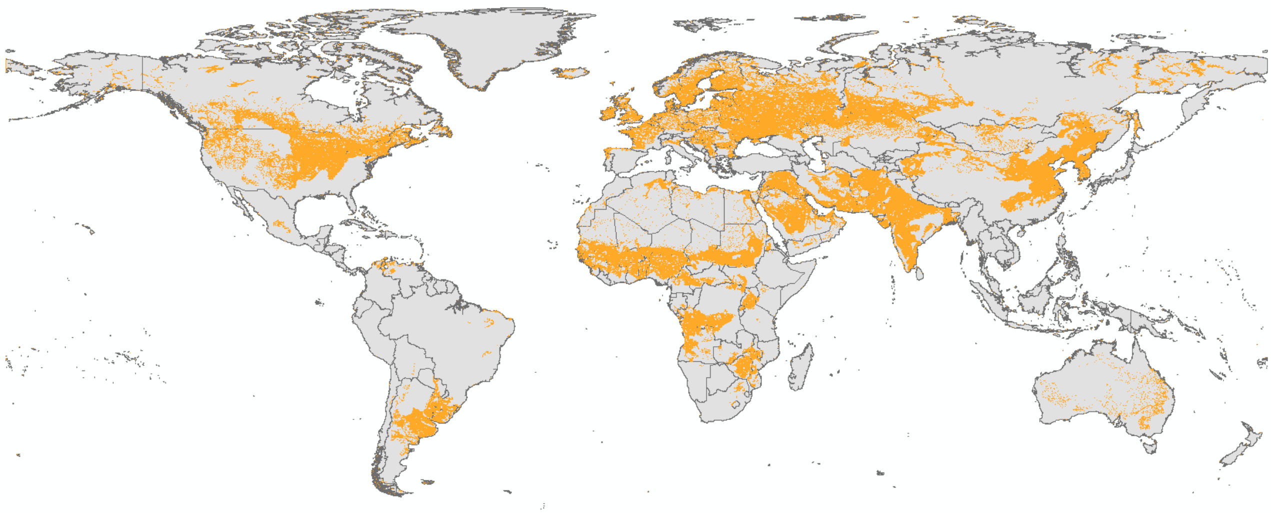


Figure 42: Areas without any conservation focus (orange)

Therefore, to be able to compare two areas that are not prioritized in the UBR map, a so-called Jenkins biodiversity value is calculated to fill the gaps analyzed. Jenkins et al. [228]

provide three global data sets, one for species richness, another for the number of threatened species and a third for the number of endemic species for the taxa mammalia, aves and amphibia. To obtain a Jenkins biodiversity value, the total number of threatened species of mammals, birds and amphibians is calculated. The same procedure is done for the total number of endemic species and the total number of species. For each of the three resulting data sets, the proportion of species per grid cell is then calculated, resulting in a value between 0 and 1, where 1 is the highest proportion of species listed. The average proportion is then calculated from the three individual data sets to obtain a Jenkins biodiversity value for each grid cell. The entire Jenkins index map is shown in Figure 43.

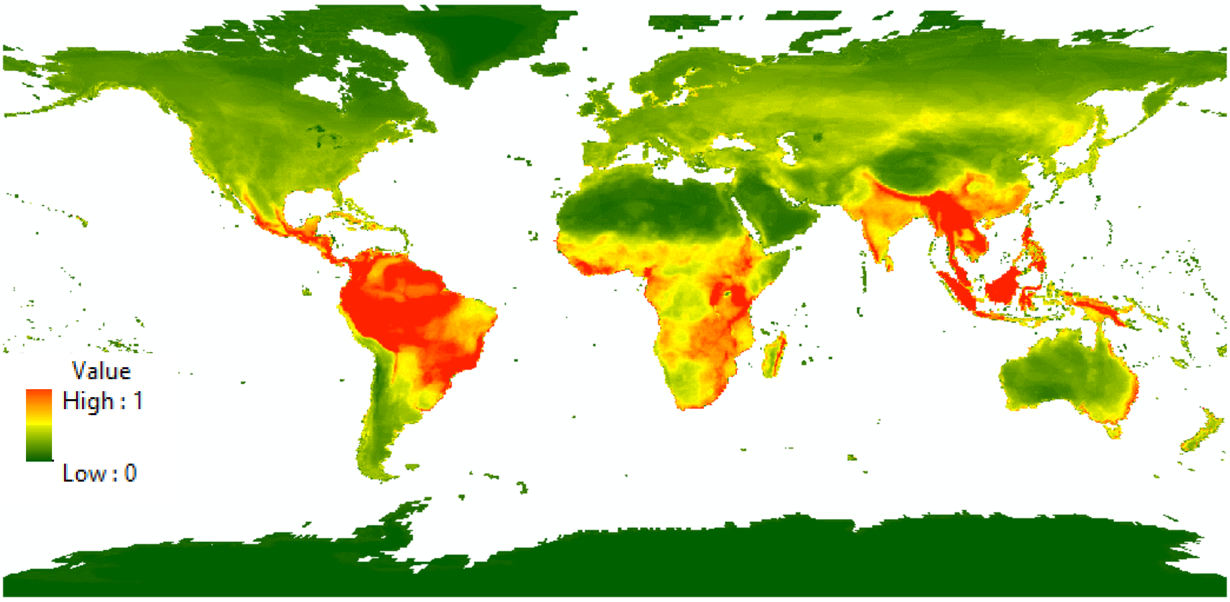


Figure 43: Jenkins Index

The resulting advantage of the Jenkins Index is that it rounds out the gaps left by the conservation schemes and therefore completes the map. For example: A company wants to relocate its timber production site from Indonesia/Borneo to a region that is not declared as a reactive or proactive site (in other words, it is not part of the UBR map). It is therefore reviewing production sites in Finland, Latvia and India. The Jenkins Index helps determine which production site would have the least negative impact on biodiversity. Figure 44 shows the Jenkins Index for all areas not covered by the UBR map. As can be seen, India has higher values than Finland or Latvia. Therefore, the LCIA analysis would indicate that the global risks would be lower in the latter two countries and therefore a production site would be preferable there in terms of biodiversity.

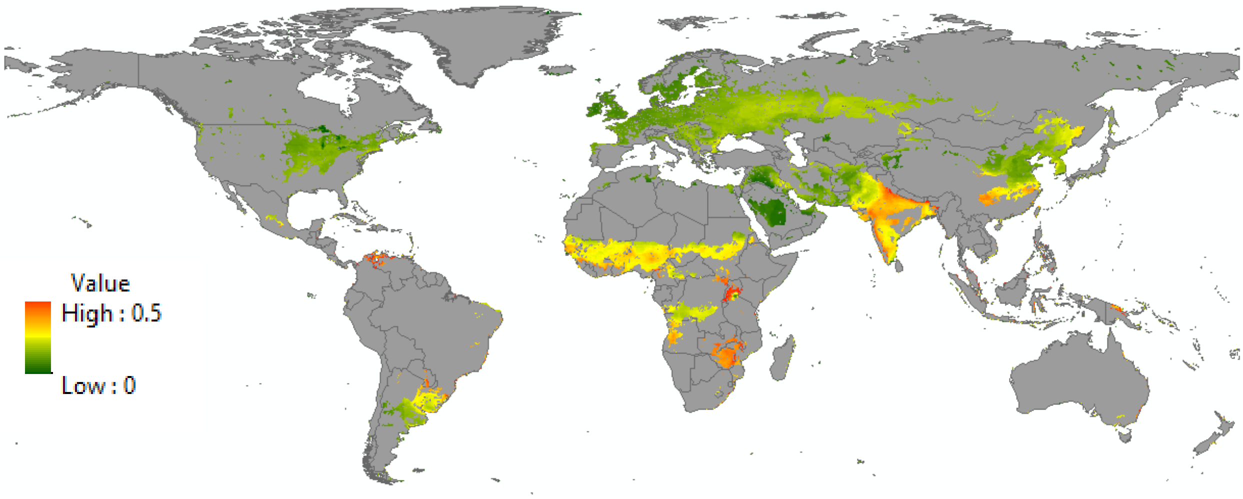


Figure 44: Jenkins Index in areas that are not covered by any conservation scheme

5.1.4 Regionalization of global risks by land use type

In this chapter, the regionalization of the global biodiversity risks with the land use models of Hurtt et al. [186] is executed. Herein, the land use map is compared with the unified biodiversity risk maps: In doing so, we can determine the degree of overlap as well as the proportion of proactive, reactive and proactive-reactive areas per land use type.

5.1.4.1 Data preparation and cleaning for regionalizing global risks

The land use model of Hurtt et al. [186] is used for the regionalization of the various land use types, as recommended by Kim et al. [219] for biodiversity assessments. The same land use model is also used by the IPCC to assess and predict current and future impacts of climate change [186]. The land use types and subtypes of Hurtt et al. [186] represent the current status of land use in 2015 for occupation. The land use model is stored in the Network Common Data Format (NetCDF). The NetCDF is a data cube with the variables coordinates, time and share of land use type per grid cell. The time change from one land use type to another can be selected for each land use type for transformation. Using ArcGIS, the NetCDF data cube is divided into hundreds of individual raster layers to obtain individual layers for all occupation and transformation flows for each land use type.

Each land use map is converted and stored as a single raster layer for all possible land transformation options and all occupation flows. The baseline year of the land use model (2015) represents the location of the occupation land use flows, therefore the current land use activities. The transformation flows are based on the chosen modelling period of 10 years, yet as Koellner et al. [142] highlight in the UNEP Setac Framework, the number of years of modelling time is always arbitrary. In order to analyze the transformation impacts over 10 years, the average share of each grid cell in the historical land use change from 2005 to 2015 is calculated. The same applies to the future land use classes from 2015 to 2025 with the virtual machine OSGeoLive Version 10 within the Lubuntu Virtual Box Version 5.2.12 r122591 (Qt5.6.2). The *$cdo timselsum* command is used to calculate the total land use change area for each transformation flow. *$cdo timselsum* processes values from 121 variables for historical land use flows. For the future area of land use change, 1,345,770,742 values from 121 variables are processed. The grid cells in which no land use occurs are automatically removed from each land use flow grid by traversing all grid levels with the raster package in RStudio with the *values()* function.

5.1.4.2 Regionalization global risks for land use flows for occupation

For the regionalization of land use for occupation, the overlapping area of the individual land use types in proactive and reactive areas is determined and divided by the total area of the land use type per analytical unit (grid cell and country). The location of the following 13 land use flows and sub flows was investigated:

- cropland (divided into the 5 subtypes: C3 perennials, C3 annuals, C4 perennials and C4 annuals, and C3 nitrogen-fixing plants),
- forestry (subdivided into primary vegetation forest and secondary vegetation forest),
- pasture (subdivided into the subtypes managed pasture and pasture),
- urban areas.

For the land use flow cropland a uniform map is produced as the sum of all subtypes of cropland and for the land use flow pasture a map is developed as the sum of rangeland pasture and managed pasture. For the land use flow forestry a map is calculated with the sum of the areas from the primary forest vegetation maps and the secondary forest vegetation maps. With RStudio version 1.1.338 the calculation is performed and the extent to which the land use areas fall into the risk areas of biodiversity is calculated. The packages *raster, sp, rgdal, doParallel* and *foreach* are used [247,250–254]. *doParallel* and *foreach* packages are used for parallelization and automation of the calculations. The *raster* package is used for processing raster data and *sp* and *rgdal* for vector data operations. Using the *doParallel* and *foreach* packages, an algorithm is written, which automatically scans all raster layers in a folder and stacks them in a virtual raster stack. The biodiversity risk map overlay is processed and looped through the raster stack to calculate the extent of overlap for each grid cell in a raster layer of the raster stack. The result of the land use types overlapping the unified biodiversity risk map is saved and written to individual raster files.

The same procedure applies to the biodiversity risk maps, which only considers proactive or reactive risk areas. All these calculation methods are performed for all land use flows for occupation as well as for the average change of historical and future land use flows. The resulting maps show the proportion of a land use type corresponding to risk sites for biodiversity. The maps can be used to assess impacts on biodiversity if the coordinates of the region or land use are known (foreground data). For the background data, the percentage of land use with the overlapping risk areas of biodiversity is calculated for all countries (the analytical unit). The availability of such data is crucial for assessing the impact of biodiversity within the supply chain, where only the country of origin is known. This excludes Antarctica and some oceanic islands from the analysis as there is no information on land use activities. For all other countries, the total area of each land use type per country is converted into km² for all land use flows (transformation and occupation). First, the share of each type of land use per grid cell is converted into area km² and summarized as total land use area per country. Given the large number of land use maps and the large amount of data, this is also done by using an algorithm that can automate and parallelize the processing operation. The shape file with the countries of the world is overlaid with the virtual grid stack containing all grid layers representing the area of land use per grid cell. The *extract()* function adds up the total land use area per country. The resulting information is written to a data frame and saved as a CSV file. The same calculation procedure applies to the land use area that falls into the biodiversity risk areas per land use type and country. Finally, the land use area within a biodiversity risk area is divided by the total land use area per country to calculate the share of land use of a country in areas with high biodiversity risks. In a final step, the CSV files are joined to a vector map representing the countries of the world. This is done to visualize the results calculated by country. By using the algorithms, it is also possible to obtain results for each desired unit of analysis (states/regions, etc.).

5.1.4.3 Regionalization of global risks for land use flows transformation

For the regionalization of the transformation flows, all possible variations from one type of land use to another were taken into account, resulting in 144 land use type specific global biodiversity risk maps for historical land use and the same number of risk maps for predicted future land use. The land use model of Hurtt et al. [186] provides different scenarios for land transformation depending on future socio-economic development paths. For this method, the future SSP5 scenarios (Shared Socio-Economic Pathways) of the IPCC are used as an example for the regionalization of the transformation flows. This global development scenario describes a development path that is strongly based on fossil fuels and represents a demanding development path for the environment. As the SSP5 scenario describes a conservative development path, this scenario was chosen because in a life cycle assessment a conservative scenario is usually assumed if no more precise data are available [255].

5.1.4.4 Regionalization of global risks, the Jenkins Index

For the regionalization of the Jenkins Index, the land use maps are overlaid with the Jenkins map to calculate the impact on the proportion of species and the proportion of endangered and endemic species affected by each land use flow. Yet, herein only the areas that are not covered by the UBR map, that were assessed during the gap analysis, are taken into account. Therefore, at first, all other areas are excluded by converting the values to no data points.

5.1.4.5 Calculation of final regionalized biodiversity risks

The final global risk factor for each land use flow is then calculated from the probability of land use in risk areas, the proportion of risk areas per grid cell and the Jenkins Index values for all land use areas outside of biodiversity risk areas. Since all maps have values from 0 to 1, we obtain a dimensionless global risk map per land use flow. Contrary to the risk factors of the regional and the local scale, the global risks factor is not normalized. This is based on the suggestion of conservation scientist, who state that a global impact would have higher implications for the overall biodiversity loss than a local or regional impact on biodiversity [204].

5.1.5 Results of the regionalization of global biodiversity risks

One exemplary result of regionalized global biodiversity risks is shown for the foreground database for crop production in Figure 45. Further results for the other land use flows for both the foreground and background database are presented in Annex II. These results show that by regionalizing risks with land use models, the risk values vary according to the type of land use. For example: The Democratic Republic of the Congo has a fairly high average risk that C3 perennial production takes place in biodiversity risk areas, while the risks for pasture and urban areas are lower because these land use activities mainly take place outside the risk areas. More detailed knowledge of the location of land use areas will also improve the global risk assessment using the foreground maps with a higher resolution. Depending on the region within a country, the land use process in a country may be within or outside risk areas.

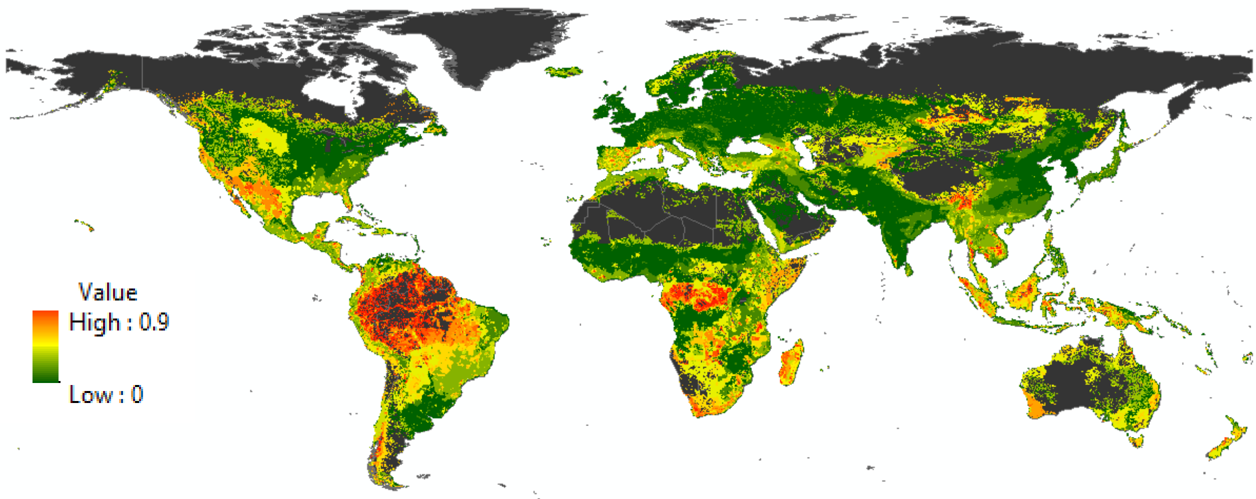


Figure 45: Biodiversity risk cropland foreground system

An exemplary result of regionalized land transformation flows is shown in Figure 46 for the foreground databases (further results are presented in Annex II). The future transformation maps show the risks involved that land transformation will take place within biodiversity risk areas within the next 10 years. The historical transformation maps show areas where land transformation has taken place in the last 10 years and where they overlap with biodiversity risk areas. Figure 46 shows for example the areas of transformation from primary forest to C3 annual crop production in biodiversity risk areas. As expected during the last 10 years there was a high risk that the transformation from primary forest to C3 annual crop took place in biodiversity risk areas in Brazil or in Indonesia and Malaysia.

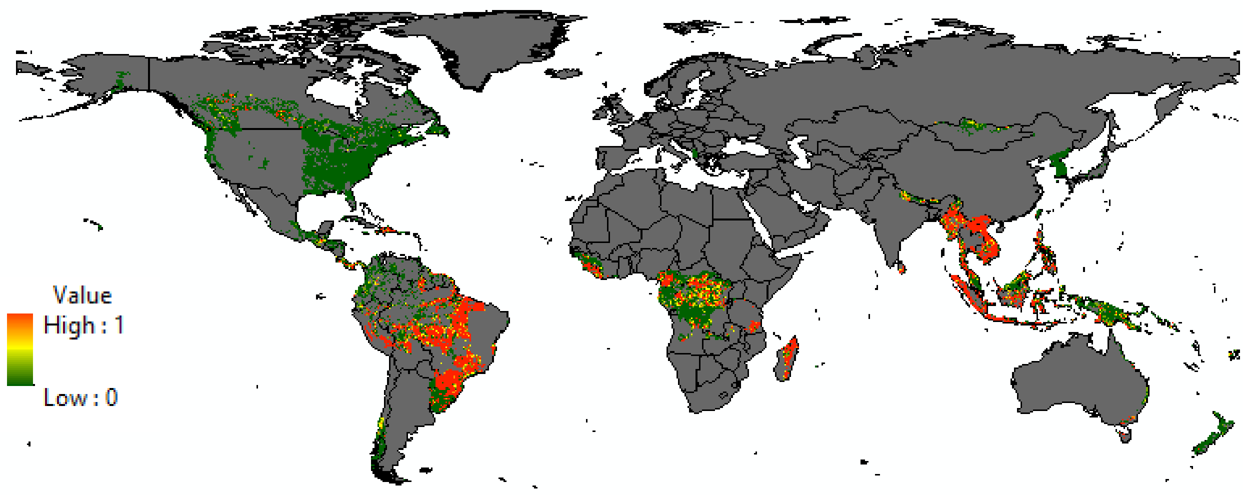


Figure 46: Biodiversity risk for transformation primary forest to C3 annual crops (historic)

An example for the Jenkins Index values of urban areas is depicted in Figure 47. In this map all areas that have been classified as urban by Hurtt et al. [186] and that do not coincide with global biodiversity risk areas of the UBR map are shown. They are evaluated with regard to the indicators of species richness, endemic species as well as threatened species. Red areas indicate that a large number of threatened and endemic species as well a high species richness occur in these urban areas.

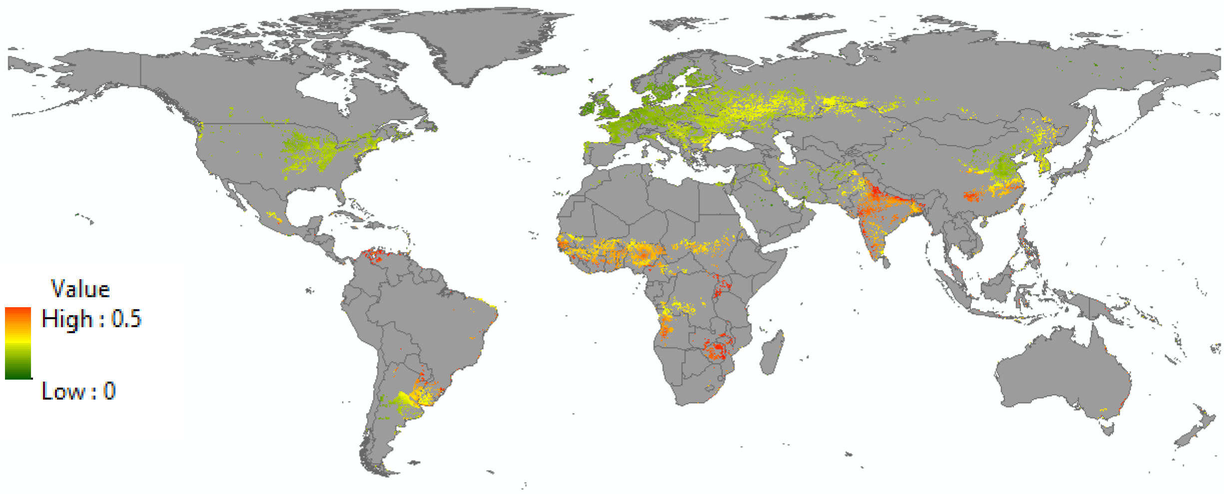


Figure 47: Jenkins Index for urban areas

An exemplary map of the overall global risk factor is shown in Figure 48 for the land use flow plantation (C3 perennial crops). Herein, we can see quite a high risk for biodiversity due to plantations in some areas in Southeast Asia, in the Amazon region as well as in the Congo Basin. Furthermore, relatively high risks exist in the Mediterranean regions of Europe.

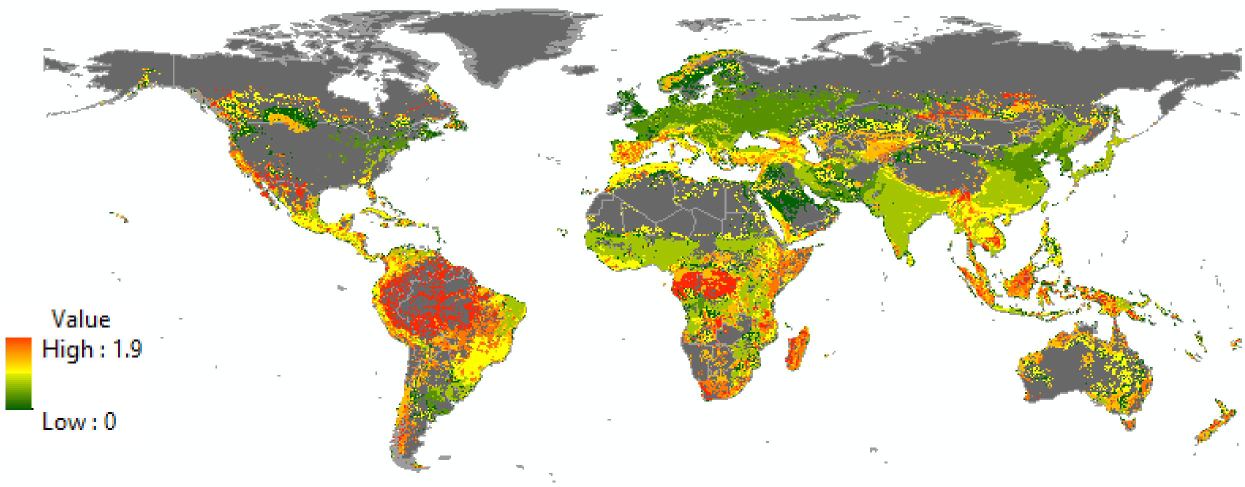


Figure 48: Global risk factor plantation

5.2 Local scale

In this chapter the land use type specific impact intervals for different ecological indicators are calculated. This step yields biodiversity risk values per land use type on the local scale. Furthermore, the calculation of land use intensity indices (LUI) is conducted by identifying and using biodiversity specific management parameters, as described in [5] similar as to [256] which was published during the publication process of the present work. These LUIs are then translated into the land use type specific impact interval. We obtain global datasets for biodiversity risk values at the local level depending on the land use intensity of a grid cell.

5.2.1 Calculation of land use type specific impact intervals

For determining the impact on biodiversity at the local scale, this method uses the values of Newbold et al. [104,105] of the PREDICTS database [187,200,201]. The PREDICTS database facilitates the evaluation of different types of land use based on biodiversity indicators, so that for each individual biodiversity indicator impact values are calculated and summarized in a local biodiversity risk index.

Based on the UNEP Setac Framework [142], the characterization factors for occupation effects are calculated by comparing the biodiversity risk quality index of the current land use type with the biodiversity risk quality index of the reference situation, which is primary vegetation as stated in Newbold et al. [104,105] and the PREDICTS database [187,200,201]. The data for the biodiversity quality of the secondary vegetation forest are averaged for the different successive phases (secondary vegetation young, medium and old) in order to obtain a single value for the secondary vegetation. To evaluate the effects of permanent transformation, the average biodiversity quality is compared between two different land use types [5]. If the current land use type is compared to a historical land use type it's a transformation from (pre-use conditions), if it is compared to a hypothetic future land use type it is referred to as transformation to (post-use conditions). The characterization factors for permanent transformation impacts are calculated by comparing the differences in the risk quality of biodiversity between two types of land use (permanent transformation) according to the following formulas of the UNEP-SETAC Framework. These characterization factors are then multiplied with the area of land use and, for occupation, the time of land use to assess the impacts. The rationale is depicted in Figure 49.

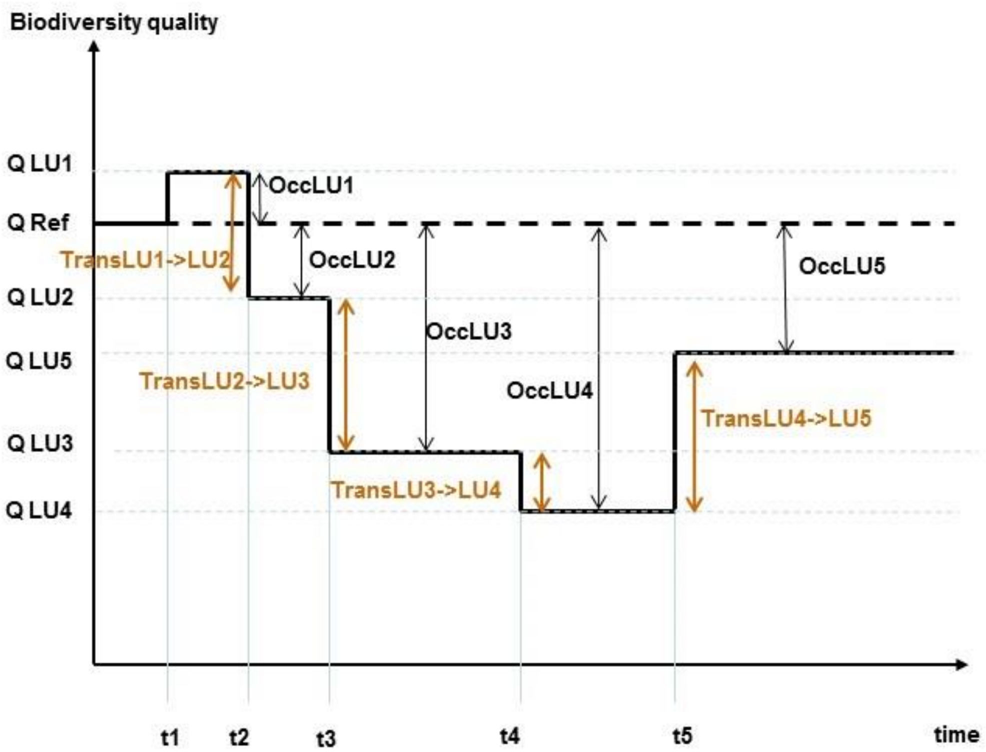


Figure 49: Calculation of occupation and permanent transformation biodiversity impacts adapted after Mila i Canals et al. [133]

The characterization factors are calculated according to these equations:

$$CFOcc_{LU1}=(Q_{ref}-Q_{LU1}) \quad (15)$$

$$CFOcc_{LU2}=(Q_{ref}-Q_{LU2}) \quad (16)$$

$$CFOcc_{LU3}=(Q_{ref}-Q_{LU3}) \quad (17)$$

$$CFOcc_{LU4}=(Q_{ref}-Q_{LU4}) \quad (18)$$

$$CFOcc_{LU5}=(Q_{ref}-Q_{LU5}) \quad (19)$$

$$CFTrans_{LU1\rightarrow LU2}=(Q_{ref}-Q_{LU1})-(Q_{ref}-Q_{LU2}) \quad (20)$$

$$CFTrans_{LU2\rightarrow LU3}=(Q_{ref}-Q_{LU2})-(Q_{ref}-Q_{LU3}) \quad (21)$$

$$CFTrans_{LU3\rightarrow LU4}=(Q_{ref}-Q_{LU3})-(Q_{ref}-Q_{LU4}) \quad (22)$$

$$CFTrans_{LU4 \rightarrow LU5}=(Q_{ref}-Q_{LU4})-(Q_{ref}-Q_{LU5}) \quad (23)$$

$$CFTrans_{LU5 \rightarrow LUn}=(Q_{ref}-Q_{LU5})-(Q_{ref}-Q_{LUn}) \quad (24)$$

where
$CFOcc_{LU}$: Characterization factor for occupation for the specific land use type
$CFTrans_{LU->LU}$: Characterization factor for transformation between land use types
Q_{ref}: Quality value of biodiversity of reference situation (primary vegetation)
Q_{LU}: Quality value of biodiversity value under land use type

5.2.2 Results of biodiversity risks due to land use types

Since the data of Newbold et al. [103,104] only provides values for the indicators species richness, abundance and rarefaction, based richness for all land use types as well as all intensity classes, only these three indicators could be taken into account in the overall biodiversity risk index. The values for the average biodiversity risk index for the land use flows of occupation are shown in Table 7. If no primary information is available (e.g. if the enterprise only has knowledge that its raw materials originate from Brazilian pasture land), the average intensive effect is used for the background database. Improvements in the impact on biodiversity are only possible if the resources are taken from another, on average less harmful land use type or if the origin of the resources is changed [5]. For example: If a company uses resources from cropland, there is a mean decrease of about 30%, whereas if it could obtain its resources from forestry the average intense biodiversity impact would be a reduction of biodiversity quality of about 20%, therefore mitigating the impact. The same logic applies to transformation, which can have positive or negative impacts depending on the new land use type. Furthermore, comparing the intervals, we can conclude that urban areas have the highest room for improvement within the same land use type, intensive urban land use has an average risk index of 47.2% but the urban land use under minimum intensity has an average risk index of only 4.2%. That is to say that urban areas have the greatest potential to mitigate their negative impact per area unit and instead to become biodiversity-friendly areas. The values of the Biodiversity Risk Index for transformation are listed in Annex III.

Table 7: Biodiversity Risk Index for occupation land use flows with reference to primary vegetation minimal use

Land Use Type	Biodiversity quality difference calculated with data from Newbold et al. [103,104] derived from PREDICTS database [187,201]						
	Average richness	Average abundance	Average rarefaction based richness	Functional diversity (mean plant height)	Average similarity (Jaccard richness)	Average similarity (Jaccard abundance)	Average BR index
Primary vegetation minimal	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Primary vegetation light	-1.40	-3.80	-1.50				-2.23
Primary vegetation intense	-5.40	-30.70	2.20				-11.30
Secondary vegetation avg.	17.35	26.55	16.10		1.29	2.99	20.00
Secondary vegetation minimal	17.40	22.50	14.20	4.00			18.35
Secondary vegetation light/intense	17.30	30.60	18.00				21.97
Cropland avg.	33.77	29.00	23.97		3.20	6.58	28.91
Cropland minimal	26.90	10.60	22.50	14.20			20.00
Cropland light	38.10	45.10	20.90				34.70
Cropland intense	36.30	31.30	28.50				32.03
Pasture avg.	29.43	22.50	17.37		5.13	10.11	23.10
Pasture minimal	21.80	4.80	10.30	28.00			12.30
Pasture light	29.40	27.80	17.80				25.00
Pasture intense	37.10	34.90	24.00				32.00
Plantation avg.	28.50	4.37	23.33				18.73
Plantation minimal	19.20	-13.40	11.80	9.80			5.87
Plantation light	26.90	22.20	14.80				21.30
Plantation intense	39.40	4.30	43.40				29.03
Urban avg.	29.63	41.83	12.00		-3.07	-6.73	27.82
Urban minimal	4.00	18.20	-9.70				4.17
Urban light	34.70	44.90	16.80				32.13
Urban intense	50.20	62.40	28.90				47.17

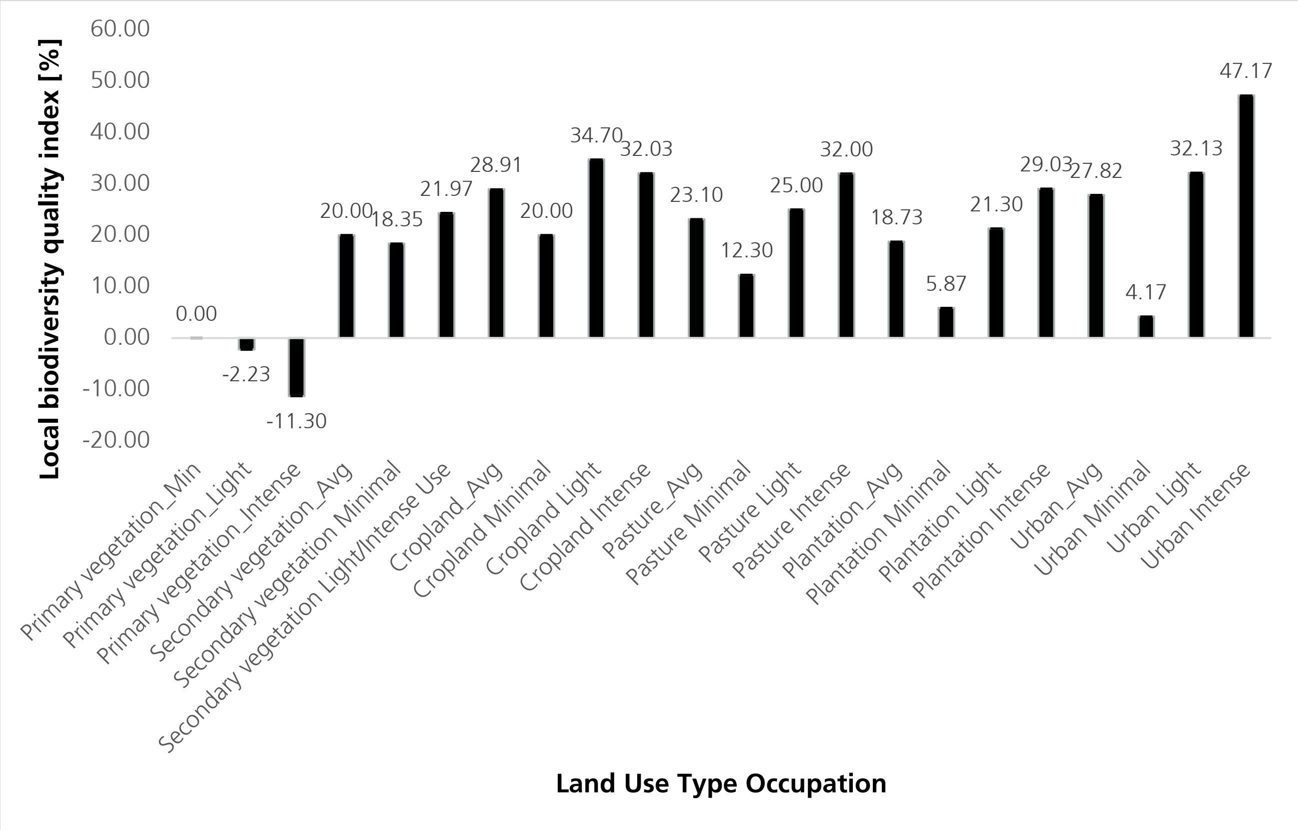


Figure 50: Biodiversity quality difference due to different land use types for occupation calculated with data from Newbold et al. [103,104] of the PREDICTS database

5.2.3 Land use intensity and management parameters

The intervals of the local Biodiversity Risk Index are used to quantify the impact of land management on biodiversity. For this purpose, a land use intensity index is calculated for each type of land use in this subchapter. The LUI index is calculated for all land use types and then translated into local biodiversity risks depending on the impact interval derived from Newbold et al. [105,106] of the PREDICTS database. By linking the land use intensity indices with the information on local biodiversity impacts from the PREDICTS model, we can see how the values shift within an interval. Therefore, we can determine a final local risk value based on the land management.

5.2.4 Identification of management parameters

In this chapter management parameters for the calculation of land use intensities for the same land use flows used in the PREDICTS database are compiled. Herein, a distinction is made between land management parameters that are attributable to the patch/field itself (within the same land use type, e.g. cropland) and landscape management parameters that involve a different land use type, such as the provision of natural corridors or buffer zones around the patch/field (which would be classified for example as land use type primary vegetation or secondary vegetation). The latter have to be measured at a landscape scale and are therefore part of the LDI calculations (Chapter 5.3 Regional scale). The management parameters within the same patch or field (the same type of land use) are measured at plot level (local scale) and are part of the LUI calculations. All management parameters are identified on the basis of the Conservation Evidence Database of Sutherland et al. [198], similar to the approach of Ulrich et al. [257] who analyzed the parameters only for the land use type cropland. However, for this method management parameters for all broad land use types will be taken into account.

As a first step, all management activities where *the effectiveness has been proven by clear evidence* or where there is *evidence of mean effectiveness* in the Conservation Evidence Database are firstly filtered. An effectiveness (defined as the size of benefit or harm) score of 0 means no effect, whereas an effectiveness of 100% implies that the management activity is always effective [198]. The management activities with at least an effectiveness score of 50% are categorized as *beneficial* or *likely to be beneficial* in the database [198]. Other activities in the Conservation Evidence Database are assessed to show a *trade-off between benefit and harm, unknown efficacy, are unlikely or likely to be ineffective* [198] and therefore not considered, as suggested by Ulrich et al. [257]. Should their effectiveness be proven in the future, an incorporation will be necessary. In a second step, the management activities are matched to the corresponding land use flows of this method (cropland, pasture, forestry, plantation, urban). Furthermore, in order to keep the long list of management activities shorter and to avoid double counting, all activities relating to the same management parameters are grouped together. Management activities related to policies or regulations are not included in the list (e.g. *paying farmers to cover the costs of conservation measures [198]*). The same applies to management activities related to awareness and education for biodiversity conservation or threats related to climate change, as they are not at the center of this method. Thus, only those management activities are included that are directly responsible for the intensity of land use. A complete list of management parameters matched to the land use types and a description of each management activity is attached in Annex IV.

The next step is to select appropriate indicators that can measure the intensity of the respective management parameters. A distinction can be made between pressure indicators to measure adverse impacts on biodiversity (e.g. use of pesticides) and relief indicators to measure positive impacts on biodiversity (e.g. provision of habitats in the form of set-aside area). In order to add all indicators within a single LUI, the relief indicators are scaled so that the highest value of a relief indicator has the intensity value of 0. Similar to Asselin et al. [56], all indicators are also classified with respect to the five main drivers of biodiversity loss as identified in the Millennium Ecosystem Assessment and the Global Biodiversity Report of IPBES [6,13]. These main drivers are habitat change, overexploitation (e.g. hunting and poaching), pollution, climate change and invasion of alien species; although, as mentioned, climate change assessment is not part of this method and pollution and alien species are included only through land use. The selection of indicators is also based on the availability of global data sets and GIS maps, since management parameters are identified to calculate global LUIs for the background database. For the indicators for which no global data set is available, a conservative value is taken. Further management parameters and appropriate data sets are identified on the basis of the studies of several authors [105,106,189,198,206,207].

5.2.4.1 Patch scale management parameters (within field/patch)

This chapter lists all management parameters applicable within the same type of land use. They are relevant to a local scale and are only applied within the same type of land use. They are used to calculate the land use intensity indices. The relation between the management activities described in the Conservation Evidence database, the classified management parameters, as well as the indicators will be depicted in a land use type specific cause-effect chain.

In order to describe all land management parameters the following structure is used:

1. **Land management of land use type**
   Description of the management activities identified in the Conservation Evidence Database for the respective land use type and depiction in a cause-effect chain by relating them to suitable management parameters.
   a. **Management parameter**
      Description of each land use type specific management parameter and its relevance for biodiversity.
      i. **Driver of biodiversity loss:** Assignment of parameter to related drivers of biodiversity loss in the Millennium Ecosystem Assessment and the Global Biodiversity Report of IPBES.
      ii. **Related management activities in the Conservation Evidence Database:** Listing of management activities from the Conservation Evidence Database that are related to the specific management parameter.
      iii. **Indicator:** Listing of suitable indicator to measure the impact of a management parameter on biodiversity. Description whether the indicator is a pressure or relief indicator.
      iv. **Global data set:** Description of the availability of a global dataset to be used in the LUI calculations for the background database.

By using this structure all management parameters of all land use types are being described in the subsequent subchapters.

1. Land management cropland and plantation crops

For the land use type cropland, a total of 45 management activities were identified in the Conservation Evidence Database that show a high effectiveness and are therefore categorized as either beneficial or likely to be beneficial for the conservation of biodiversity (see Annex IV). 35% of these activities relate to the management and provision of terrestrial habitats for different species by allocating part of the land use area to biodiversity. Thus, management activities targeting the same objective are assigned to the same management parameters and indicators, as some of the activities can be measured with similar parameters. This is done to make the method user-friendly and to avoid double counting. For example: The management activities *provide set-aside areas in farmland or create skylark plots* can be assigned to the same management parameter *set-aside area* within the land use type cropland since they refer to some area within the field or patch that is saved for biodiversity. This management parameter for example can then be measured with the indicator *share of set-aside area to total land use area*.

For all parameters well-established indicators could be found to measure the impact. The only management activities where no indicator could be established are those relating to the provision of artificial nesting/resting/foraging sites. The other management activities can be linked to management parameters that can be measured by appropriate indicators to calculate the LUI. The cause-effect chain for the local impacts of cropland management activities on biodiversity is presented in Figure 51. The parameters and indicators are described in more detail below.

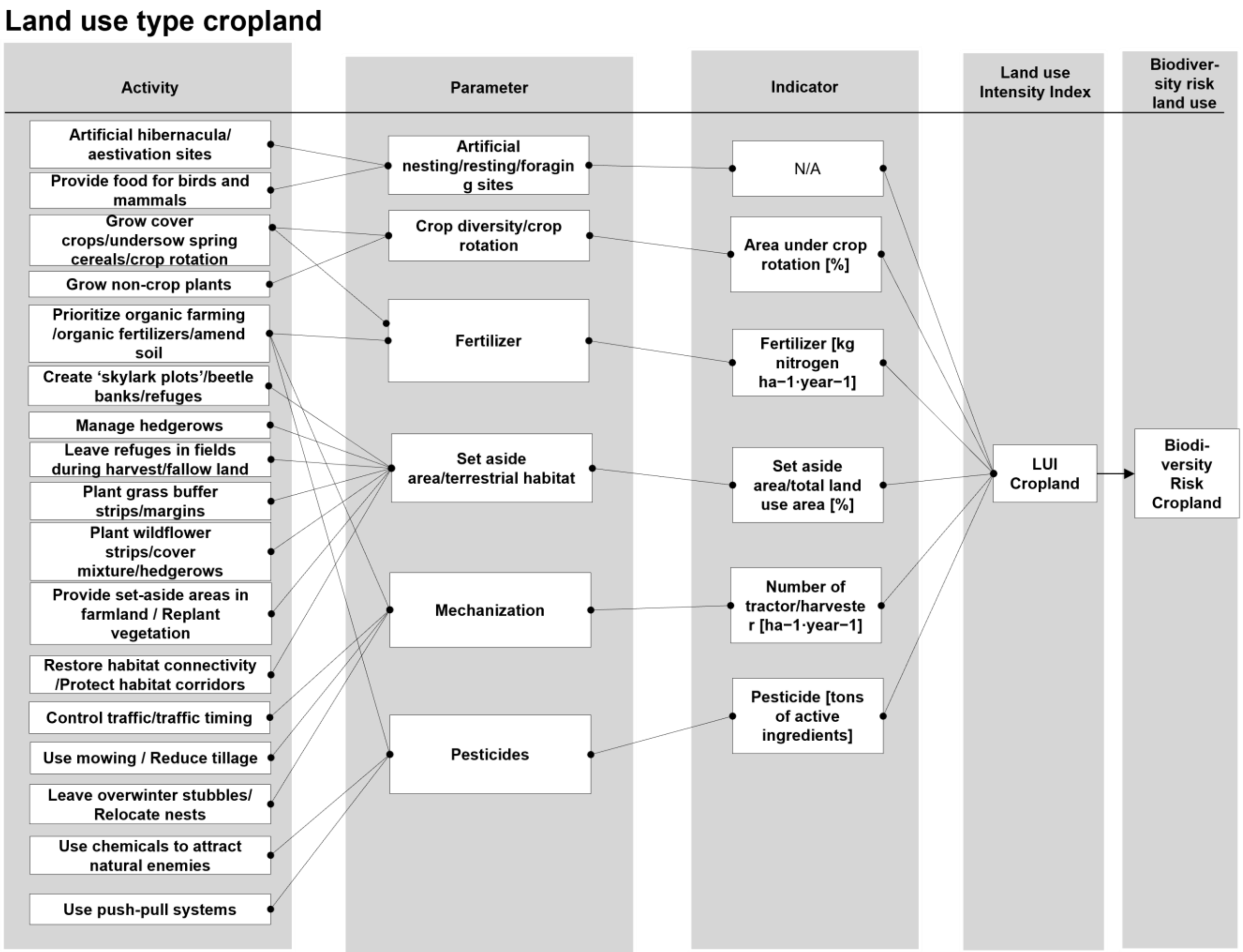


Figure 51: Management activities, parameters and indicators for the land use type cropland

a. Management parameter fertilizer (synthetic):

Synthetic fertilizer in the form of e.g. nitrogen is an indicator with a strong influence on biodiversity. Eutrophication through nutrient input and deposition is regarded as one of the most important global drivers of biodiversity loss [1]. High nitrogen inputs have a particularly negative impact on the composition of flora and below-ground biodiversity. Increasing fertilizer inputs leads to water quality problems [258] and have both direct and indirect (e.g. positive correlations between increased nitrogen consumption and plant diseases) impacts on biodiversity [259–262]. Many studies have demonstrated the positive effects on biodiversity by reducing fertilizer use. This management parameter was evaluated in the Conservation Evidence Database as extremely effective for the conservation of biodiversity with a high effectiveness score of 100% [198].

i. **Driver of biodiversity loss:** Pollution

ii. **Related management activities in the Conservation Evidence Database [198]:**
- *Use organic farming instead of conventional farming,*
- *Amend the soil using a mix of organic and inorganic amendments,*
- *use organic rather than mineral fertilizers, or*
- *reduce fertilizer, pesticide or herbicide use generally.*

iii. **Indicator (pressure):** Synthetic fertilizer [kg_N/(ha*a)]

iv. **Global data set:** Global data sets on the specific fertilizer application rates are available for various crops types of C3 annual, C3 perennial, C4 annual C4 perennial and C3 nitrogen-fixing crops provided by [186].

b. Management parameter pesticide

Several studies have shown that pesticide applications have negative effects on wildlife in agricultural areas [263–265]. Therefore, activities that relate to this management parameter were assessed as 100% effective for the conservation of biodiversity [198]. The study by Ewald et al. [266] found that the use of pesticides has a greater impact on the decline of insect populations than extreme weather events due to climate change. The conclusion of the study was that reducing the use of pesticides could even mitigate some of the negative impacts of climate change [266]. Another large-scale long-term study investigated a decline in insect biomass over several decades. Their results show that the biomass of insects in Germany has decreased by more than 70%. Although the study was conducted in nature reserves, the authors assume that the strong decline is very likely due to the intensification of agriculture and especially to the use of pesticides in adjacent fields [121]. Furthermore, a decline in insects leads to a decline of other dependent species in the food chain. The Conservation Evidence Database suggests several management measures that could contribute to reducing the amount of pesticide use, such as growing crops that attract natural enemies and other measures listed below. When such activities are carried out, their effect can be measured with the proposed indicator as a reduction in pesticide use.

i. **Driver of biodiversity loss:** Pollution

ii. **Related management activities in the Conservation Evidence Database [198]:**
   - *use organic farming instead of conventional farming,*
   - *grow non-crop plants that produce chemicals that attract natural enemies,*
   - *grow plants that compete with damaging weeds,*
   - *use chemicals to attract natural enemies,*
   - *combine trap and repellent crops in a push-pull system, or*
   - *reduce fertilizer, pesticide or herbicide use generally.*

iii. **Indicator (pressure):** Pesticide application [$t_{active\ ingredients}$/(ha*a)]

iv. **Global data set:** Global data sets on pesticide application are available on a national basis provided by the FAO [193].

c. Management parameter crop diversity/crop rotation

Several studies from European countries reported a beneficial effect of rotational crop management on ground beetles or vegetation [198]. Three studies found the richness and/or abundance of ground beetle species increased and one study found a higher richness of plant species on crop plants or on farms with more crop rotations compared to monoculture fields [198]. Further studies from Canada, Portugal and Zambia analyzed the impact of leguminous crop inclusion in crop rotations and observed an increase in the number of microbes and the diversity of soil fauna [198]. The heterogeneity of cultivated areas is therefore regarded as a key factor for increasing the biodiversity of cropland [267], so that the cultivation of several crops on a single field can contribute to increase the biological value. Management activities to increase crop diversity refer to crop rotation (temporal variation of different crop species) or mixed and intercropping (spatial variation of different crop varieties).

i. **Driver of biodiversity loss:** Habitat change, overexploitation

ii. **Related management activities in the Conservation Evidence Database [198]:**
   - *Grow cover crops beneath the main crop (living mulches) or between crop rows,*
   - *undersaw spring cereals, with clover for example,*

- *grow cover crops when the field is empty,*
- *leave overwinter stubbles, or*
- *use crop rotation.*

iii. **Indicator (pressure):** Field area under crop rotation per total cropland area [%]

iv. **Global data set:** Global data sets on crop rotations is available per grid cell provided by Hurtt et al. [186].

d. Management parameter mechanization

According to a literature review on the effects of mechanization on biodiversity, a total of 42 individual studies from Europe investigated the effects of the reduction of tillage on wildlife in agricultural areas [198]. Thirty-four studies found positive effects of reduced tillage compared to conventional tillage on a number of taxa, including earthworms and other invertebrates, weeds or birds. Positive effects were an increased biomass, species richness or abundance of earthworms and other invertebrates, an increased number of some weed species, a higher density of Eurasian skylark and other birds [198]. A review showed that mechanical tillage has a negative impact on the number of invertebrates and that no-till systems provide more feed resources for invertebrates. Further studies showed that a higher number of bacteria was found in controlled agricultural traffic and that soil compaction was reduced by relying on less heavy and less frequent machines. Although other studies did not show any effects or even negative effects on certain taxa, this management parameter was assessed as likely to be beneficial for the conservation of biodiversity with an overall certainty of 60% [198].

i. **Driver of biodiversity loss:** Habitat change, overexploitation

ii. **Related management activities in the Conservation Evidence Database**

- *Use organic farming instead of conventional farming,*
- *control traffic and traffic timing,*
- *reduce tillage,*
- *use mowing techniques to reduce mortality, or*
- *relocate nests at harvest time to reduce nestling mortality.*

iii. **Indicator (pressure):** Number of tractors per ha and year [$No_{tractor/harvester}$ /(ha*a)]

iv. **Global data set:** Global data sets on the number of tractors and harvesters per hectare and year is available on a national basis provided by the FAO [194].

e. **Management parameter set-aside area**

A meta-analysis of 127 studies investigating the effects of set-aside land on biodiversity showed that the number of species and population density of several taxa such as birds, insects, spiders and plants on set-aside land is significantly higher than in conventional agricultural areas [268]. In Europe, declining bird species in particular were supported by set-aside land. A positive effect was observed for all investigated species [269]. The study by Pywell et al. [270] showed that the provision of set-aside land can even lead to higher crop yields due to the increased ecosystem services in these areas. Set-aside areas with high structural heterogeneity such as rows of hedges or flower strips offer an improvement in biodiversity, and a wealth and diversity of beneficial organisms. They are of great importance for the conservation of biodiversity, e.g. of insect communities or birds. Flowering plants, for example, which can be sown in strips or blocks, provide a habitat and an important food source for pollinators and other insects. A higher number of insects in turn is a food source for birds and other insectivores [198]. According to Sutherland et al. [198], the wildflower mix may also include agricultural varieties such as clover. They found that out of a total of 80 individual studies investigating the impacts of flower margins on biodiversity, 64 of the studies showed positive impacts on at least one taxa [198]. Therefore, this management parameter was considered to be very effective for the conservation of biodiversity [198]. Conservation headlands allow the vegetation at the edge of the field to recover naturally without planting, although mowing can take place later. The edges of the field are not fertilized and are treated with herbicides only in case of the appearance of harmful weeds [198]. Other management activities that make a positive contribution to the indicator are the creation of skylark plots, lapwing plots or beetle banks in agricultural areas or the retention of near-natural shrubs and trees on arable land, as is the case in agroforestry plantations.

If several areas are reserved for biodiversity and managed differently, they can be added to the total set-aside area. It is important to note that for the LUI calculations only management within the same patch/field (i.e. within the same land use type) is considered. For the management parameter set-aside area, this means that an area is only considered if it is set-aside within the same patch/field. If the set-aside area lies outside the field/patch and belongs to a different type of land use, it is part of the landscape matrix and is therefore taken into account in the LDI calculations.

i. **Driver of biodiversity loss:** Habitat change
ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Create ‘skylark plots’ (undrilled patches in cereal fields),*
- *create beetle banks,*
- *create refuges,*
- *leave refuges in fields during harvest,*
- *leave uncropped cultivated margins or fallow land (includes lapwing and stone curlew plots),*
- *plant nectar flower mixture/wildflower strips,*
- *plant wild bird seed or cover mixture,*
- *leave headlands in fields unsprayed (conservation headlands), or*
- *retain or plant native trees and shrubs amongst crops (agroforestry).*

iii. **Indicator (relief):** Agricultural set-aside area (with reference to the listed management activities) per total agricultural land use area [%]

iv. **Global data set:** No global data set is available for set-aside areas. Since the BfN recommends between 10 and 20% of the cultivation area to be conserved for biodiversity, 20% of set-aside area is equated to the lowest intensity value (0.0). The value of 4% is assumed as the average highest intensity (0.8) for this method. The overall highest intensity would be achieved if no area were set-aside with an intensity value of 1 (see Table 8). Thus, the average highest intensity value of 0.8 per country is assumed for the background data if no further information is available. Within the foreground system, the indicator can be evaluated more accurately either by aerial or satellite imagery where the area of set-aside land is compared to the overall field area. Furthermore, primary data from farmers or landowners can be used.

Table 8: Set-aside area and its intensity values

Set-aside area [%]	Intensity value
0	1.0
4	0.8
8	0.6
12	0.4
16	0.2
20	0.0

2. Land management plantation

The cause-effect chain for the land use type plantation is shown in the next figure. The management parameters and indicators are the same as for cropland as plantations are part of the perennial crops classified by Hurtt et al. [186] and require similar land management. However, the management activities for biodiversity conservation differ slightly for cropland and plantations, as highlighted by Sutherland et al. [198]. And the risk intervals for biodiversity are different for both land use types as well, as Newbold et al. [105] indicate. The management of plantations is therefore described in a separate cause-effect chain and assigned its own land use class (see Figure 52).

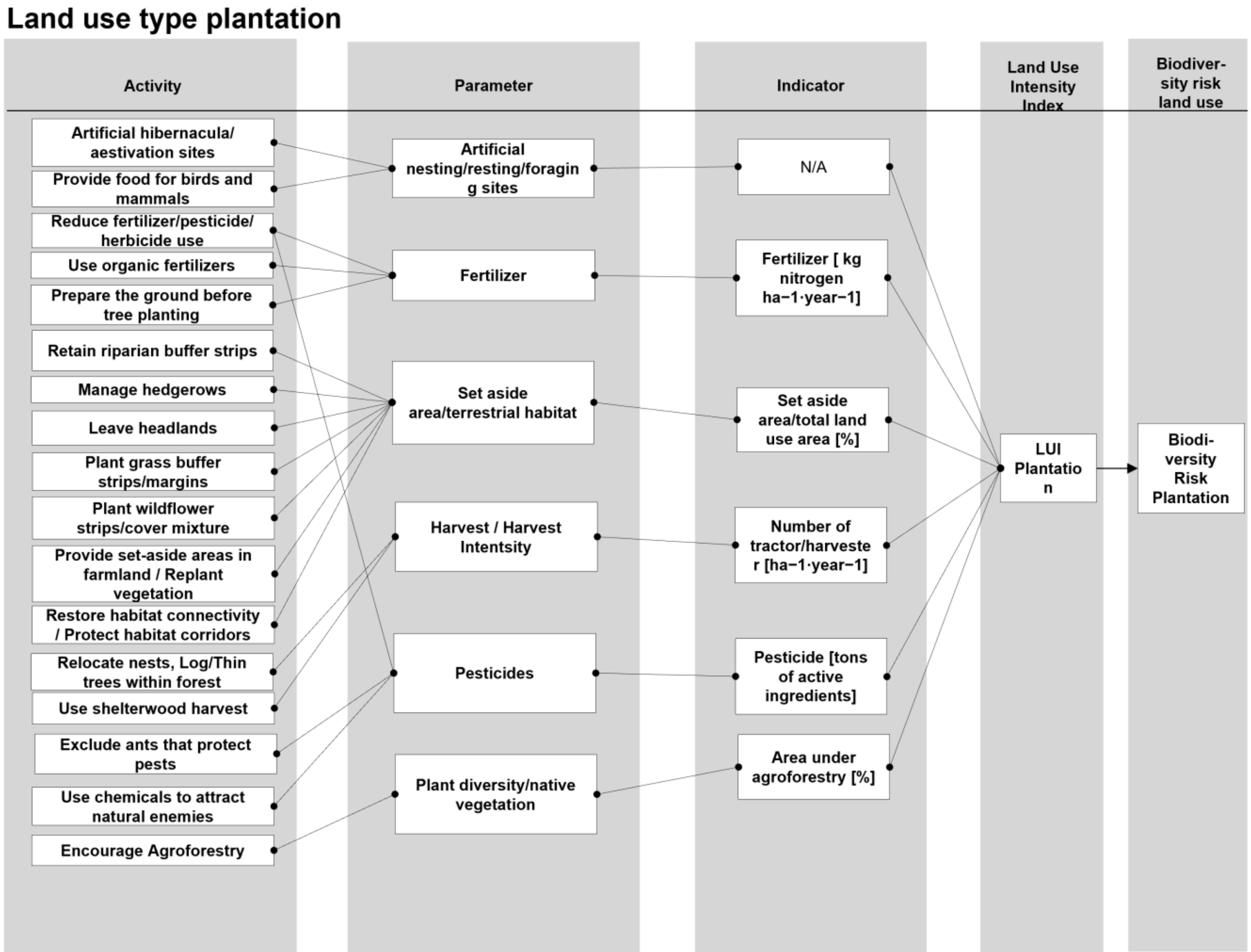


Figure 52: Management activities, parameters and indicators for the land use type plantation

3. Land management pasture

A total of 43 biodiversity-effective management activities of the Conservation Evidence Database were identified for the land use type pasture (see Annex IV). These management activities, associated parameters and indicators are mapped in the cause-effect chain in Figure 53 and are described in more detail below.

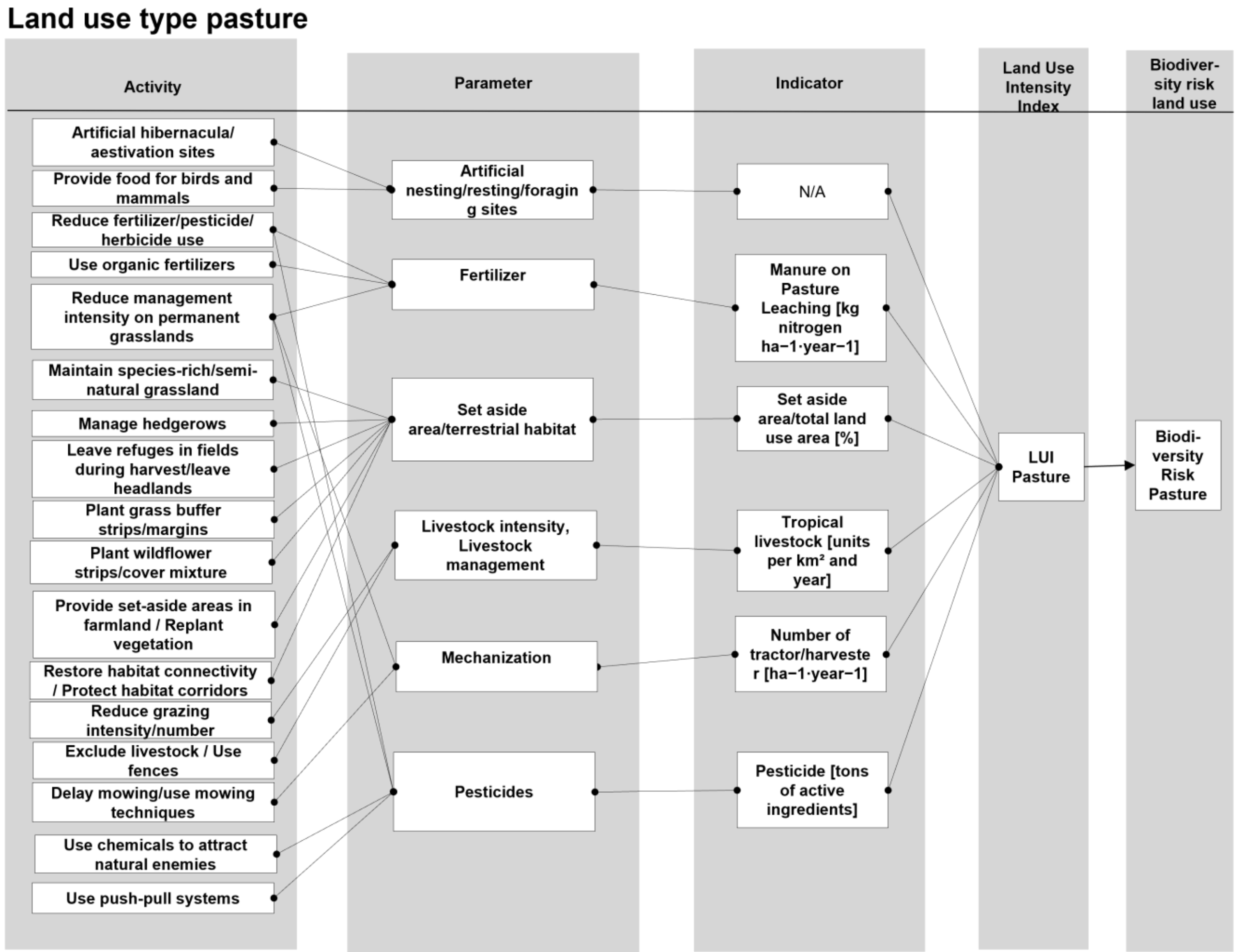


Figure 53: Management activities, parameters and indicators for the land use type pasture

a. Management parameter mowing

Mowing and harvesting can have a negative impact on wild fauna and flora in agricultural and pastoral grassland areas. Various harvesting machines or mowing patterns are used to reduce the impact of mowing on, for example, wildlife in the field. Adaptation of mowing techniques, frequency and timing of mowing can have a positive impact on ground breeders, who often stay in the long grass as long as possible [198].

i. **Driver of biodiversity loss:** Habitat change, overexploitation

ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Reduce management intensity on permanent grasslands (several interventions at once),*

- *delay mowing date on grasslands,*
- *relocate nests at harvest time to reduce nestling mortality,*
- *use mowing techniques to reduce mortality,*
- *delay mowing or first grazing date on grasslands, or*
- *convert to organic farming.*

iii. **Indicator (pressure):** Numbers of tractors or harvesters per hectare and year [$No_{tractor/harvester}$ /(ha*a)]

iv. **Global data set:** Global data sets on mowing intensity are available as number of harvesters per hectare per year at national level [194].

b. Management parameter fertilizer on pasture

Animal waste products in the form of liquid manure or slurry have become an ever-greater burden with the growth of livestock farming and the increase in the amount of biological waste. Livestock farmers are faced with considerable management problems. This leads to the conflict that operators are trying to reduce the cost of animal waste disposal by spreading excessive amounts of slurry on easily accessible land, thereby increasing the environmental impact that ultimately affects biodiversity [271]. The indicator of excessive manure on pasture that seeps into the soil is thus one suitable indicator of the intensity of grazing systems. The Conservation Evidence Database suggests several management activities related to the use of manure and fertilizer.

i. **Driver of biodiversity loss:** Pollution

ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Use organic rather than mineral fertilizers,*
- *reduce fertilizer, pesticide or herbicide use generally, or*
- *reduce management intensity on permanent grasslands (several interventions at once).*

iii. **Indicator (pressure):** Manure on pasture leaching [kg_N/(ha * a)]

iv. **Global data set:** Global data sets on the amount of manure on pasture that is leaching is available on a national scale [272].

c. Management parameter pesticide in grasslands

The use of pesticides in the form of insecticides, fungicides and herbicides does not only have adverse effects on biodiversity in croplands but also on grasslands. They affect for example mammals by removing plant food sources and changing the microclimate. Herbicides can cause changes in vegetation and habitat that threaten vertebrates, while insecticides can affect the availability of important feed insects and other invertebrates. Several studies have shown that the population density of some species or species richness increased when pesticide use was reduced and other measures were taken instead. Another study found that partridges had higher survival in grassland areas with reduced pesticide use [198].

i. **Driver of biodiversity loss:** Pollution
ii. **Related management activities in the Conservation Evidence Database [198]:**
- *Reduce fertilizer, pesticide or herbicide use generally,*
- *reduce management intensity on permanent grasslands (several interventions at once),*
- *grow non-crop plants that produce chemicals that attract natural enemies,*
- *use chemicals to attract natural enemies,*
- *reduce chemical inputs in grassland management,*
- *convert to organic farming, or*
- *grow plants that compete with damaging weeds.*

iii. **Indicator (pressure):** Pesticide application [$t_{\text{active ingredients}}$/(ha*a)]
iv. **Global data set:** No specific data sets on the global use of pesticides on grassland is available. However the FAO [193] provides approximate data for the average use of pesticides in arable land per country.

d. Management parameter livestock intensity

Grazing intensity has a direct influence on the diversity of the arthropods through mortality, soil compaction and loss of resources. Ruminants have a direct impact on plant communities by trampling, eating and compacting plants. Vegetation cannot maintain its integrity if pasture pressure is too high. Increased livestock density also affects the composition of flora. Invertebrates die from trampling as a cause of high stocking rates. A study in moors in Great Britain showed greater coverage of all vegetation at lower grazing intensities. Whereas grazing under very low intensity can have positive effects on biodiversity. The same study found that overgrazing can destroy entire habitats. It is also particularly harmful in arid areas where excessive grazing can lead to soil erosion. By reducing grazing intensity, damage to vegetation can be avoided. It also contributes to reducing bird disturbances and nest losses [198].

i. **Driver of biodiversity loss:** Overexploitation, habitat loss

ii. **Related management activities in the Conservation Evidence Database [198]:**
   - *Reduce grazing intensity,*
   - *reduce number of livestock,*
   - *exclude or remove livestock from degraded peatlands,*
   - *use fences to exclude livestock from shrublands,*
   - *use wire fences within grazing areas to exclude livestock from specific forest sections, or*
   - *delay mowing or first grazing date on grasslands.*

iii. **Indicator (pressure):** Livestock intensity [tropical livestock units/ (km^2 * a)]

iv. **Global data set:** Global maps are available for the number of the main ruminant species of sheep, goat, cattle and buffalo distribution on a grid cell level by Gilbert et al. [273–276].

e. Management parameter set-aside area pasture

As is the case with cropland, the management parameter set-aside land on pastures is also an important control parameter for the conservation of biodiversity. These include the creation of uncultivated edges around pasture fields or the planting of grass buffer strips.

i. **Driver of biodiversity loss:** Habitat loss

ii. **Related management activities in the Conservation Evidence Database [198]:**
   - *Manage hedgerows to benefit wildlife (includes no spray, gap- filling and laying),*
   - *leave headlands in fields unsprayed (conservation headlands),*
   - *leave refuges in fields during harvest,*
   - *maintain species-rich, semi-natural grassland,*
   - *plant grass buffer strips/margins around arable or pasture fields,*
   - *plant nectar flower mixture/wildflower strips,*
   - *plant wild bird seed or cover mixture,*
   - *provide or retain set-aside areas in farmland,*
   - *replant vegetation, or*
   - *create uncultivated margins around intensive arable or pasture fields.*

iii. **Indicator (relief):** Pastoral set-aside area (with reference to the listed management activities) per total pastoral land use area [%].
The design of the set-aside areas is defined by the above-mentioned activities and has to be managed accordingly.

iv. **Global data set:** The same approach is used as for the land use type cropland. As there is no global data set available for set-aside land. For the background data set the value of 14% is taken as the average highest intensity (0.8). The highest intensity would be reached if no area is set-aside with an intensity value of 1.

4. Land management urban areas

A total of 25 management activities of the Conservation Evidence Database are beneficial or likely to be beneficial for urban biodiversity. They could be mapped to eight suitable management parameters, whereby no suitable, well-established indicator could be found for the parameter artificial nesting/resting/forest areas. The relationship between management activities, parameters and indicators for the calculation of the land use intensity index for urban areas is shown in Figure 54, whereby each individual management parameter is described in more detail in this chapter.

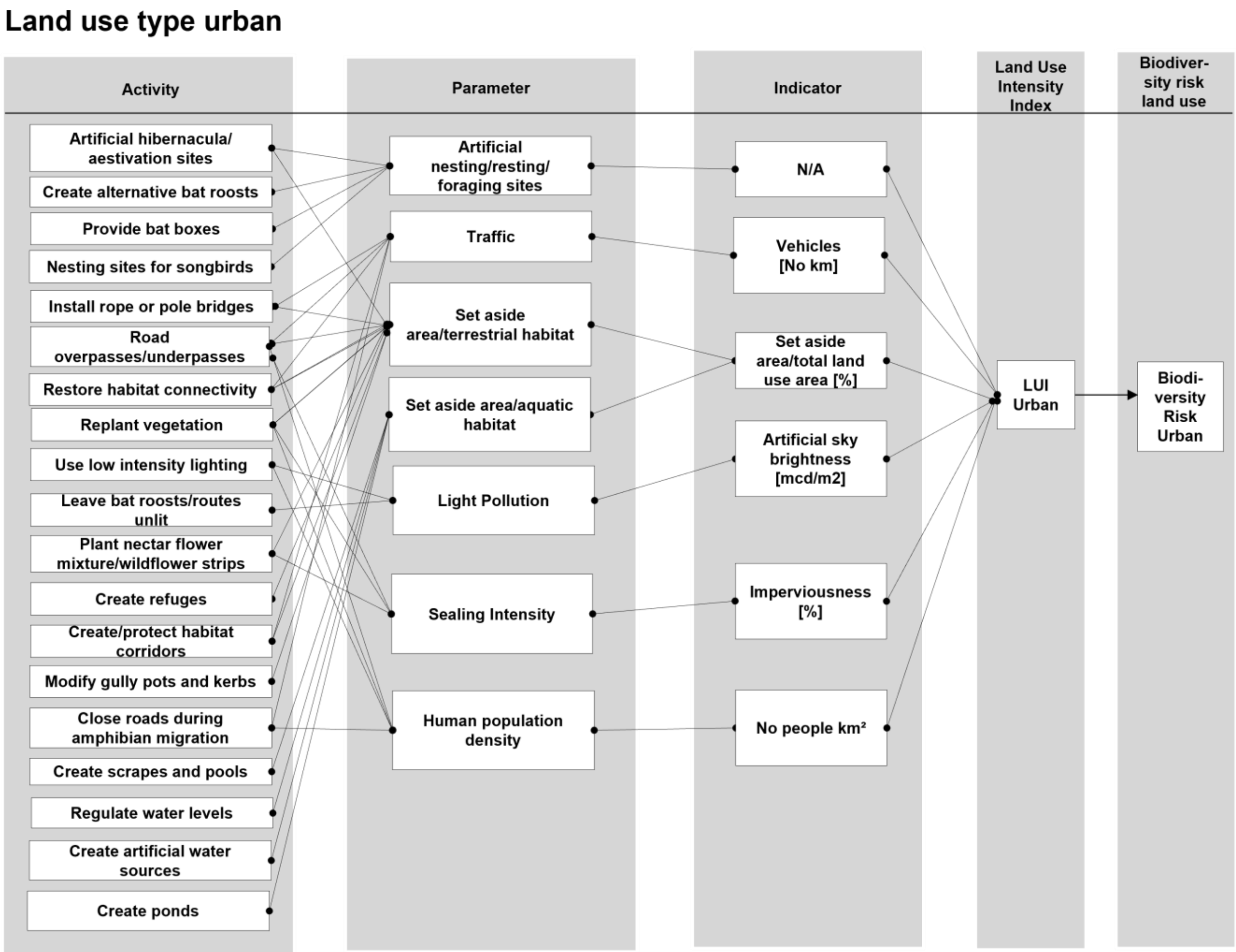


Figure 54: Management activities, parameters and indicators for the land use type urban

a. Management parameter artificial nesting/resting/foraging sites

The provision of artificial nesting, resting or feeding areas for wildlife is particularly important in urban areas where large parts of the natural habitat have been lost. Amphibians, for example, need moist, sheltered places to spend the winter or hot, dry summers whereas nesting boxes replace the original, destroyed natural nesting sites for songbirds [198]. Further shelters can be attached to house walls for bats.

i. **Driver of biodiversity loss:** Habitat change

ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Create artificial hibernacula or aestivation sites,*
- *create alternative bat roosts within developments,*
- *provide bat boxes for roosting bats,*
- *install overpasses as road crossing structures for bats, or*
- *install underpasses as road crossing structures for bats.*

iii. **Indicator (relief):** No well-established indicator is available.

iv. **Global data set:** No global datasets for this management parameter could be found. Therefore, it can be only included in the method as soon as data becomes available.

b. Management parameter traffic intensity

Road traffic can have a significant impact on several species, especially when their annual migration routes cross between wintering and breeding sites (e.g. amphibian populations). Suitable management strategies include (temporarily) closing roads to protect important migration routes or reducing the general traffic speed and overall traffic flow. In addition, gully pots on kerbstones form harmful traps, especially for amphibians. Animals crossing the roads are not only directly exposed to traffic, but if they manage to reach the curb, they often move along it, until they fall into a shaft [198]. There are several management activities to reduce the impact of traffic on migratory animals. Allowing for a safe passage of wild animals is also crucial, e.g. by installing road overpasses and underpasses or moving pits, change the structure of their grids, provide escape ladders or change the shape of kerbs [198]. Traffic intensity (measured as distance of vehicles movements per year) is used as a management parameter, as it is assumed that a reduction in overall traffic intensity in urban areas would reduce the need for safe passageways (such as road crossings, underpasses or bridges) for wildlife.

i. **Driver of biodiversity loss:** Habitat change

ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Install rope or pole (canopy) bridges,*
- *install overpasses as road crossing structures for bats,*
- *install underpasses as road crossing structures for bats,*
- *modify gully pots and kerbs, or*
- *close roads during seasonal amphibian migration*.

iii. **Indicator (pressure):** Distance of vehicle movements per time [vehicle-km/a)]

iv. **Global data set:** Data sets are only available for some countries of the European Union and refer to the total distance of (million) vehicle kilometer per year [277]. For other countries, the worst average intensity value is used for the background data.

c. Management parameter set-aside area/green space for biodiversity

Green network systems are a critical factor in urban ecology and are becoming an alternative approach to tackling environmental problems. Urban green spaces in the form of parks, gardens, green street landscapes, cemeteries or corridors are an integral part of the green network system [278,279]. In urban areas, the provision of green spaces for biodiversity can be vertical (e.g. greening of façades) or horizontal (see Figure 55 and Figure 56). They thus offer more area potential for green spaces than other types of land use. The use of native (also climatically adapted) vegetation and the composition of plant species (plant diversity) are particularly important for the design of green spaces for biodiversity. This management parameter refers only to set-aside areas designed with biodiversity in mind or specifically intended for biodiversity (e.g. through the use of native plant species, including a mixture of different plant species, including buffer strips along roads, or by connecting green areas for biodiversity via corridors). Urban green areas such as simple lawns or gardens with exotic species are considered in the management parameter sealing, as they reduce the sealing intensity in the urban environment.

Figure 55: Examples for biodiversity friendly urban management in Singapore (own photograph)

Figure 56: Green facade in Stuttgart (own photograph)

Just as green spaces can serve as terrestrial habitats in urban areas, the provision of water areas such as ponds and other artificial water sources is crucial for both terrestrial and aquatic species. According to Sutherland et al. [198] there are several studies showing that ponds in urban gardens are used by amphibians as habitats. Some studies have even found that urban rainwater ponds or natural urban drainage systems are used as aquatic habitats [280,281]. In addition, artificial water sources are beneficial for all species, as they are a source for food and drinking especially in hot and dry seasons [198].

i. **Driver of biodiversity loss:** Habitat change
ii. **Related management activities in the Conservation Evidence Database [198]:**
  - *Replant vegetation,*
  - *plant nectar flower mixture/wildflower strips,*
  - *create refuges,*
  - *restore habitat connectivity,*
  - *create/protect habitat corridors,*
  - *sowings of species-rich seed mixes for lawns and meadows [279],*
  - *leafy streetscapes designs include multiple types of green elements such as road-side trees, flower beds under trees, or green façades [279],*
  - *regulate water levels (maintain pond water levels),*
  - *create artificial water sources, or*
  - *create ponds.*
iii. **Indicator (relief):** Set-aside area (under listed management activities) per total urban area [%].

The design of the set-aside areas is defined by the above-mentioned activities and has to be managed accordingly.

iv. **Global data set:** There is no global data set available. Therefore, the average worst value (10% of urban set-aside land) is used for the background database until data becomes available. As urban areas offer more potential for the creation of green spaces for biodiversity (horizontally and vertically), 50% of set-aside areas are equated to the lowest intensity value. 0% of urban set-aside areas correspond to the highest intensity value of 1 (Table 9).

Table 9: Set-aside area in urban environments and its intensity values

Set-aside area [%]	Intensity Value
0	1
5	0.9
10	0.8
15	0.7
20	0.6
25	0.5
30	0.4
35	0.3
40	0.2
45	0.1
50	0

d. Management parameter degree of sealing

Soil sealing has a detrimental effect on biodiversity, as everything above ground is destroyed. However, it also has a negative impact on below-ground biodiversity as it interferes with natural cycles such as precipitation and solar radiation [282]. This management parameter can be seen as complementary to the proposed management parameters set-aside area/green space. At the same time, the increase in green areas in cities might lead to a reduction in sealed areas. Gravel gardens as depicted in Figure 57 and Figure 58 are also classified as sealed areas because they have the same effects on biodiversity as completely sealed areas [283].

Figure 57: Gravel garden I (own photograph)

Figure 58: Gravel garden II (own photograph)

i. **Driver of biodiversity loss:** Habitat change

ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Replant vegetation,*
- *plant nectar flower mixture/wildflower strips,*
- *create refuges, or*
- *leafy streetscapes designs include multiple types of green elements such as road-side trees, flower beds under trees, or green façades [279].*

iii. **Indicator (pressure):** imperviousness [%]

iv. **Global data set:** Elvidge et al. [284] provide global data sets on the intensity of sealing at a grid cell level. The unit of the data set is in percentage imperviousness.

e. **Management parameter light pollution**

Light pollution has a significant impact on the circadian and seasonal cycles of organisms and on their movements and spatial patterns. Overall, light pollution disrupts the spatial-temporal dynamics of biological communities and ecosystems [285]. A study in Great Britain, for example, showed that bats avoided flying on illuminated hedges and that their activity was lower than that of unlit hedges [198]. Therefore, the Conservation Evidence Database suggests to only use low intensity lighting and to avoid light pollution.

i. **Driver of biodiversity loss:** Pollution
ii. **Related management activities in the Conservation Evidence Database [198]:**
- *Use low intensity lighting,*
- *leave bat roosts, roost entrances and commuting routes unlit, or*
- *avoid illumination of bat commuting routes.*

iii. **Indicator (pressure):** Artificial sky brightness [mcd/m^2]
iv. **Global data set:** A global data set on the artificial sky brightness is developed by Falchi et al. [196] providing values at a grid cell level which can be used to assess the intensity of light pollution across the world.

f. **(Management) parameter population density**

Human population density is not a classical management parameter, but a proxy for the management and intensity of urban systems. For example, the study by Luck [286] showed that human population density is strongly related to the richness of exotic species. Since there are no global data sets on the number of exotic species in urban areas, human population density can serve as a proxy indicator. However, this proxy should be replaced as soon as specific data becomes available. In addition, human population density is closely linked to the increase in the number of threatened species, both geographically limited and invasive [286]. The human population competes with flora and fauna for space and habitat, especially in urban areas. Therefore, biodiversity-friendly planning and management of growing urban centers is of paramount importance.

i. **Driver of biodiversity loss:** Habitat change, pollution, overexploitation, invasive species
ii. **Related management activities in the Conservation Evidence Database** All
iii. **Indicator (pressure):** Number of persons per area [$No_{persons}/km^2$]
iv. **Global data set:** Global data-sets are available for the number of persons per km² on a grid cell level provided by the CIESIN [287].

5. **Land management forestry (primary and secondary vegetation forested)**

For the management of forests, the Conservation Evidence Database identified a total of 28 management activities that have been scientifically proven to be beneficial or are likely to be beneficial for biodiversity (see Annex IV). This method offers well-established indicators to measure the impact on a parameter for all parameters with the exception of the parameters anti-poaching and artificial nesting / rest / feeding sites as well as mixed-tree forests. In addition, no global data sets are available for these three parameters. The management activities, parameters and indicators for the calculation of the land use intensity index for forestry are shown in Figure 59. The management parameters are described in the following subchapter, highlighting the drivers of biodiversity loss to which they relate, the type of indicator (pressure or relief) and appropriate global data sets.

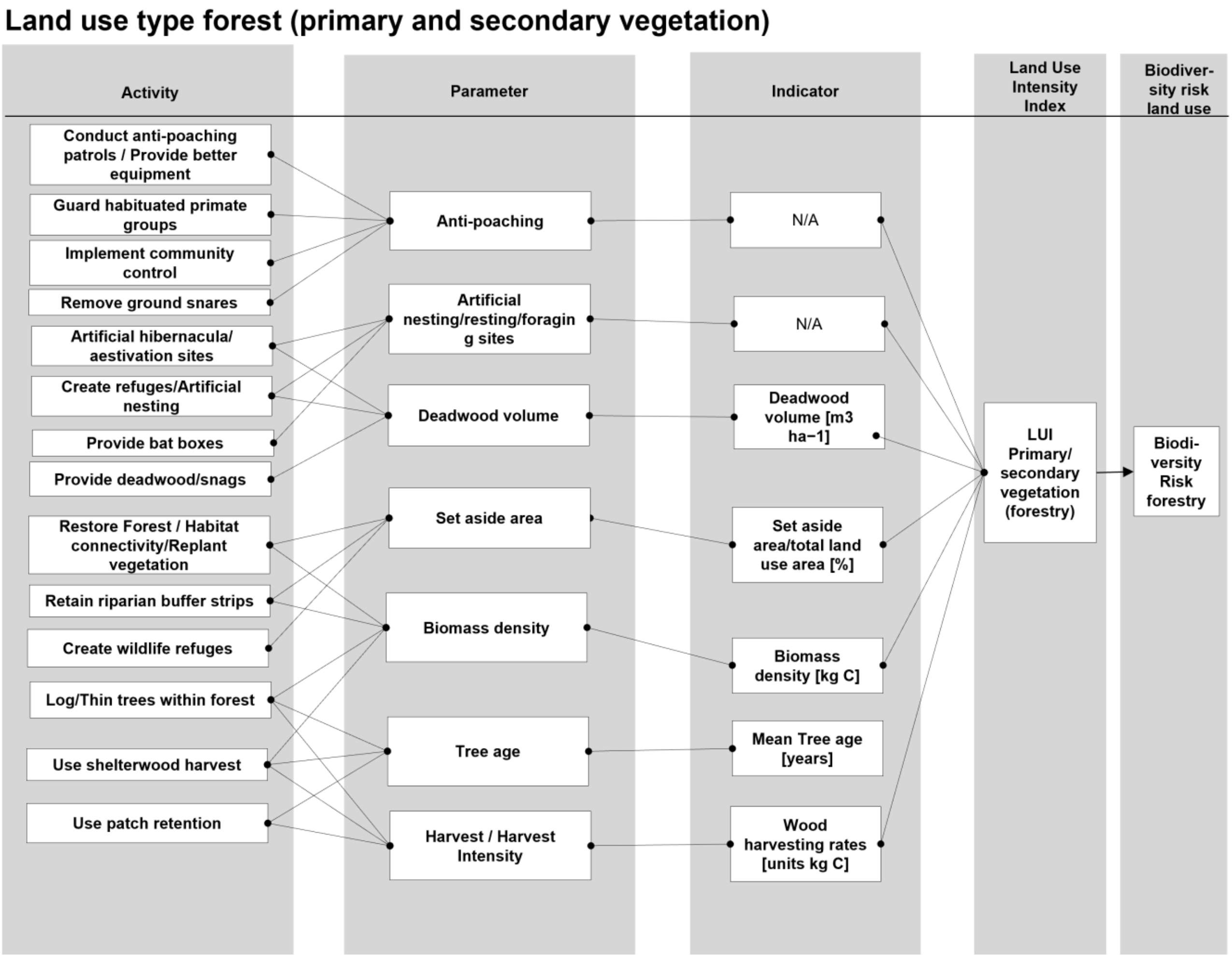


Figure 59: Management parameter for the land use type forestry

a. Management parameter anti-poaching

Poaching and illegal hunting or killing of wild animals pose a serious threat to biodiversity. The main type of land use affected is forestry, as poaching occurs predominantly in primary or secondary forests. Land managers could help reduce pressure on wildlife by providing equipment, patrolling and monitoring areas, or removing traps and snares. It should be noted that poaching and illegal killing of wildlife is not only a challenge for developing countries, but still exists in developed countries. For example, the study by Brochet et al. [288] shows that the illegal killing and taking of birds is a serious problem for biodiversity in Europe's Mediterranean regions.

i. **Driver of biodiversity loss:** Overexploitation

ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Guard habituated primate groups to ensure their safety/well- being,*
- *implement community control of patrolling, banning hunting and removing snares,*

- *provide better equipment (e.g. guns) to anti-poaching ranger patrols,*
- *regularly de-activate/remove ground snares,*
- *conduct regular anti-poaching patrols, or*
- *use wildlife refuges to reduce hunting disturbance.*

iii. **Indicator (pressure):** No suitable well-established indicator could be found. Thus, the indicator cannot yet be included in the LUI calculations.

iv. **Global data set:** No global data sets on poaching incidences or anti-poaching measures have been found.

b. Management parameter artificial nesting/resting/foraging sites

The provision of artificial nesting, resting or feeding places in forests is beneficial for a variety of species such as bats, birds, invertebrates or amphibians [198]. Here, the design and maintenance of the nesting, resting or feeding area is important and quite species-specific. In addition, the suitable location of the sites must be taken into account. With bat boxes, for example, height, orientation and solar radiation play a role. It is important for the feeding stations to use species-appropriate food.

i. **Driver of biodiversity loss:** Habitat change

ii. **Related management activities in the Conservation Evidence Database [198]:**
- *Create artificial hibernacula or aestivation sites,*
- *create refuges,*
- *provide bat boxes for roosting bats,*
- *provide artificial nesting sites for songbirds, or*
- *clean nest boxes to increase occupancy or reproductive success*.

iii. **Indicator (relief):** No well-established indicator to measure the impact of artificial nesting/resting/foraging sites could be identified.

iv. **Global data set:** No global data sets available.

c. Management parameter deadwood

In natural forests, deadwood is a typical feature and a key factor of biodiversity, as it is a microhabitat for a variety of species [289,290]. It was therefore chosen as an indicator for the sustainable management of European forests [291]. The decline of deadwood leads to the loss of habitat for a large number of species [292]. In Europe, this has been shown to be one of the main reasons for the loss of forest biodiversity. Therefore, increasing the proportion of deadwood is seen as an important management practice to promote biodiversity (see Figure 60 a-d). At least 40 m³ of deadwood per hectare has been identified as an important characteristic for communities of saproxylic organisms. A total forest area in Europe should contain at least 20 m³ of deadwood per hectare with a diameter of more than 40 cm [293].

Figure 60 a-d: Deadwood as biodiversity management parameter in the black forest (own photographs)

i. **Driver of biodiversity loss:** Habitat change
ii. **Related management activities in the Conservation Evidence database [198]:**
   - *Provide deadwood/snags in forests (use ring-barking, cutting or silvicides).*
iii. **Indicator (relief):** Deadwood volume, standing and lying [m^3/ha]
iv. **Global data set:** There are only data sets for some European countries available as average volumes of the total deadwood of Forest Europe [195]. The default value for the other countries for the background data is the worst average intensity value.

d. **Management parameter wood harvesting rates**

Studies have shown that the degradation of forests in the form of repeated felling, for example, has harmful effects on biodiversity [294]. Therefore, timber harvesting quotas can serve as an indicator of forest degradation. A decrease in timber harvest intensity is not only beneficial for biodiversity, but also has a positive effect on the global climate. Sutherland et al. [198] reviewed various studies on the intensity of timber harvesting and found that the majority of the studies are directly related to impacts on biodiversity.

i. **Driver of biodiversity loss:** Habitat change, overexploitation
ii. **Related management activities in the Conservation Evidence Database [198]:**
- *Log/remove trees within forests: effects on understory plants,*
- *thin trees within forests: effects on understory plants,*
- *thin trees within forests: effects on young trees,*
- *use patch retention harvesting instead of clearcutting, or*
- *use shelterwood harvest instead of clearcutting.*

iii. **Indicator (pressure):** Wood harvesting rates as carbon per area [units kg C/km^2]
iv. **Global data set:** Global data sets on wood harvesting rates are available on a grid cell basis for both primary and secondary forests provided by Hurtt et al. [186].

e. Management parameter tree species/tree diversity

Forest composition and the number of different native tree species have a direct positive influence on biodiversity. Typically, monoculture forests or forests with a low number of native tree species have lower biodiversity compared to natural forests [295].

i. **Driver of biodiversity loss:** Habitat change, overexploitation
ii. **Related management activities in the Conservation Evidence Database [198]:**
- *Restore or create forests,*
- *replant vegetation,*
- *use wildlife refuges to reduce hunting disturbance, or*
- *retain riparian buffer strips during timber harvest.*
- *use patch retention harvesting instead of clearcutting,*
- *use shelterwood harvest instead of clearcutting.*

iii. **Indicator (relief):** Number of native tree species
iv. **Global data set:** No global data sets on the number and composition of native tree species in forests are found.

f. **Management parameter tree age**

Tree age is a common indicator for measuring the intensity of forest systems. It is measured as the average age of all trees or the maximum age of the oldest trees [107]. Studies have shown that the age of tree species has a significant influence on the species richness of birds, insects, vascular plants or lichens and fungi and that the age of the canopy has a high influence on the Shannon Diversity Index [283,284].

i. **Driver of biodiversity loss:** Habitat change, overexploitation
ii. **Related management activities in the Conservation Evidence Database [193]:**
  - *Log/remove trees within forests: effects on understory plants,*
  - *thin trees within forests: effects on understory plants,*
  - *thin trees within forests: effects on young trees,*
  - *use patch retention harvesting instead of clearcutting,*
  - *use shelterwood harvest instead of clearcutting,*
  - *restore or create forests, or*
  - *replant vegetation.*

iii. **Indicator (relief):** Mean tree age [a]
iv. **Global data set:** Global data sets on the age of trees in forested areas are available on a grid cell basis for secondary forests provided by Hurtt et al. [181].

g. **Management parameter biomass density**

A study conducted by Lennox et al. [296] on the availability of biomass and its effects on biodiversity showed that biomass density is closely related to the distribution of species in secondary forests [296]. This makes it a suitable indicator for measuring the intensity of forest use and its impact on biodiversity. The FAO also recommends that the decline in biomass should be used as an indicator of forest degradation [297].

i. **Driver of biodiversity loss:** Habitat change, overexploitation
ii. **Related management activities in the Conservation Evidence Database [198]:**
  - *Log/remove trees within forests: effects on understory plants,*
  - *thin trees within forests: effects on understory plants,*
  - *thin trees within forests: effects on young trees,*
  - *use patch retention harvesting instead of clearcutting,*
  - *use shelterwood harvest instead of clearcutting,*
  - *restore or create forests, or*

- *replant vegetation.*

iii. **Indicator (relief):** biomass density [kg C/km²]

iv. **Global data set:** Global data sets on the density of plant biomass are available on a grid cell basis for both primary and secondary forests provided by Hurtt et al. [186].

h. Management parameter set-aside areas

The creation of set-aside land in forests has proved beneficial for biodiversity. It has been classified as beneficial and highly effective for the conservation of biodiversity [198]. In this way the vegetation can regrow naturally without being planted [198]. Ecotones with high structural heterogeneity, such as natural forest margins, offer an improvement in biodiversity and a wealth and diversity of beneficial organisms. They are of great importance for the conservation of biodiversity, e.g. of insect communities or birds. The destruction of marginal habitats (e.g. forest buffer zones along streams) thus has a negative impact on biodiversity.

i. **Driver of biodiversity loss:** Habitat change

ii. **Related management activities in the Conservation Evidence Database [198]:**
- *Restore or create forests,*
- *replant vegetation,*
- *use wildlife refuges to reduce hunting disturbance, or*
- *retain riparian buffer strips during timber harvest.*

iii. **Indicator (relief):** Set-aside area (uncultivated forest) per total forest area [%].

iv. **Global data set:** In order to calculate the set-aside areas in forests the world database on protected areas of UNEP-WCMC & IUCN [197] is used and the share of protected versus unprotected forests calculated.

5.2.5 Calculation of land use intensity indices

The management parameters, indicators and global data sets are used to calculate land use intensity indices. In general, it is preferable to use primary data of the real land use management and to calculate the intensity index in a foreground process. However, if this is not possible, the background data calculated in this thesis can be used. The background database uses maps available in a GIS environment and statistical data on global practices such as fertilizer use or pesticide use. This chapter presents the calculation of LUI indices for each management parameter and all land use flows for occupation at a global level in order to provide background data for the supply chain. In addition, benchmark values are provided for each management parameter to allow the use and comparison of primary data in the method.

5.2.5.1 Land use intensity index cropland and plantation cash crops

The following global data sets (see Table 10) are used to calculate the land use intensity indices for the background database in regard to the class and sub classes of cropland. Since no global databases are found, the default value for the background database is assumed for the management parameter set-aside area. Satellite images can be used to calculate the size of retention areas or ecotones, but information on the exact location of land use must be available.

Table 10: Data sets for the LUI cropland and plantations

Management parameter	Indicator [unit]	Data type	Global data source
Fertilizer	Fertilizer application rates [kg_N/(ha*a)]	Global maps, primary data	[186]
Pesticide	Pesticide application rates [$t_{active\ ingredients}$/(ha*a)]	Global maps, FAO statistics, primary data	[193]
Mechanization (tillage)	Number of machines [$No_{tractor/harvester}$ /(ha*a)]	FAO statistics, primary data	[194]
Set-aside areas/ecotones	Set-aside area per field area [%]	Primary data, satellite images	N/A
Crop rotation/crop diversity	Share crop rotation per field [%]	Global map, primary data	[186]
Global cropland data set	Area of cropland production [km²]	Global map, primary data	[186]

Fertilizer intensity

Based on the spatial distribution maps of different crops and the fertilization maps (in kg nitrogen ha-1-year-1) of Hurtt et al. [186], a weighted fertilizer map for all fertilizer rates and crops is generated:

$$Fertilizer_{WeightedSum[i]} = \sum \%Crop_{Type}[i]*Fertilizer_{CropType}[i] / \sum \%Crop_{Type[i]} \quad (25)$$

where

$Fertilizer_{WeightedSum[i]}$: Fertilizer application for all crop types and fertilizer rates at location i

$\%Crop_{Type[i]}$: Share of crop type at location i

$Fertilizer_{CropType[i]}$: Fertilizer application rate for crop type at location i

Based on the map of the weighted sum of all fertilizer quantities used for each crop type, the application quantities per global agro-ecological zone are calculated using the GAEZ map of Ramankutty et al. [223]. The mean and standard deviation of fertilizer application within each zone is calculated by using zone statistics. To calculate the benchmark value within each zone, the MTI is calculated by adding the mean to the corresponding standard deviation values. The maximum tolerable level of fertilizer intensity is calculated according to Sattler et al. [221].

The table with the benchmark value is linked to the GAEZ map and the new map showing the MTI values per GAEZ is exported as shape file. The resulting map is then converted into a raster map to standardize the individual fertilizer maps and to obtain fertilizer application intensity maps for the land use type cropland.

The benchmark value map used for standardization is shown in Figure 61. As can be seen, the benchmark values for fertilization in tropical agro-ecological zones are lower than the benchmark values in temperate regions. This also corresponds to the recommendations of the FAO, which recommends lower fertilizer application in tropical areas due to the specific climatic conditions and soil properties in these regions.

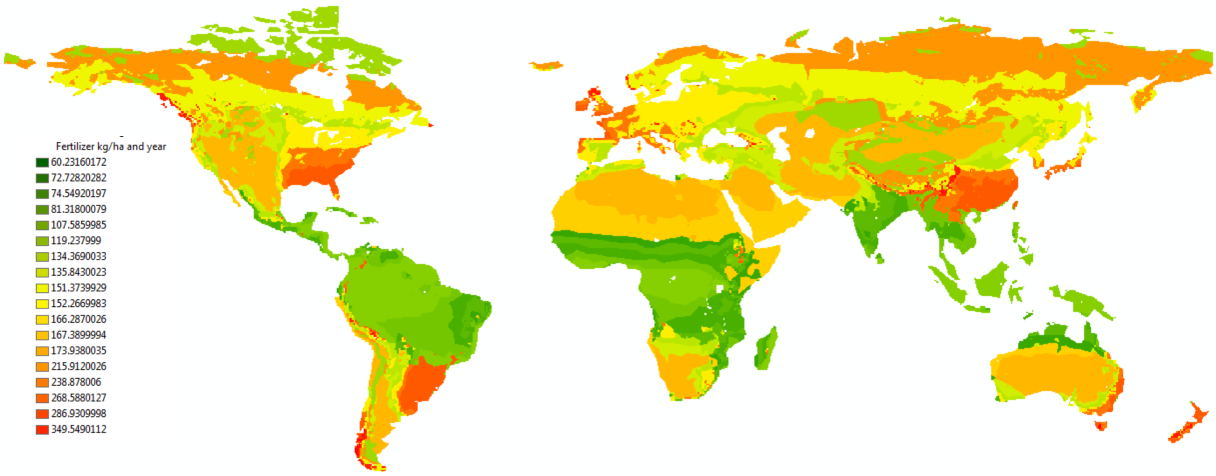


Figure 61: Maximum intensity levels of fertilizer application per agro-ecological zone

As a next step the map with the weighted sum of all fertilizers is standardized by the benchmark values per global agro-ecological zone. As a result, we obtain intensity values for the management parameter fertilizer use for the land use class cropland (see Figure 62).

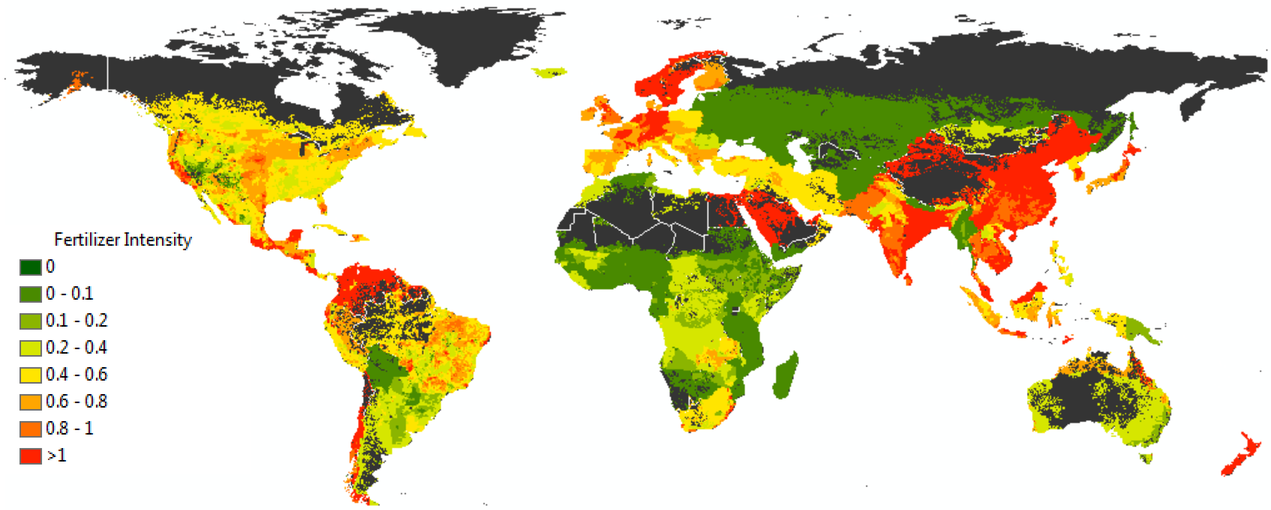


Figure 62: Global fertilizer intensity map for cropland

The same calculation method applies to all land use subclasses of cultivated areas namely C3 and C4 annual and C3 and C4 perennial crops and C3 nitrogen-fixing crops. However, only fertilization rates in areas where the specific crop variety is cultivated are taken into account. In Figure 63 and Figure 64 it can be shown that the fertilization intensity varies according to the type of crop and region. This underlines the need for regionalization of the specific amount of fertilizer in a given area, as the impact on biodiversity is directly related to the intensity of the individual management parameters. Thus, the effects on biodiversity will also vary depending on the type of crop and region.

For example: The land use subclass C3 nitrogen fixation has a generally lower intensity for the management parameters fertilizer use, as these plant varieties live symbiotically with nitrogen-fixing bacteria. They do not require as much fertilizer as other plant varieties. Therefore, at least for the management parameter fertilizer application, the cultivation of C3 nitrogen-fixing plants is more favorable for biodiversity compared to other crop types, for example C3 annual crops.

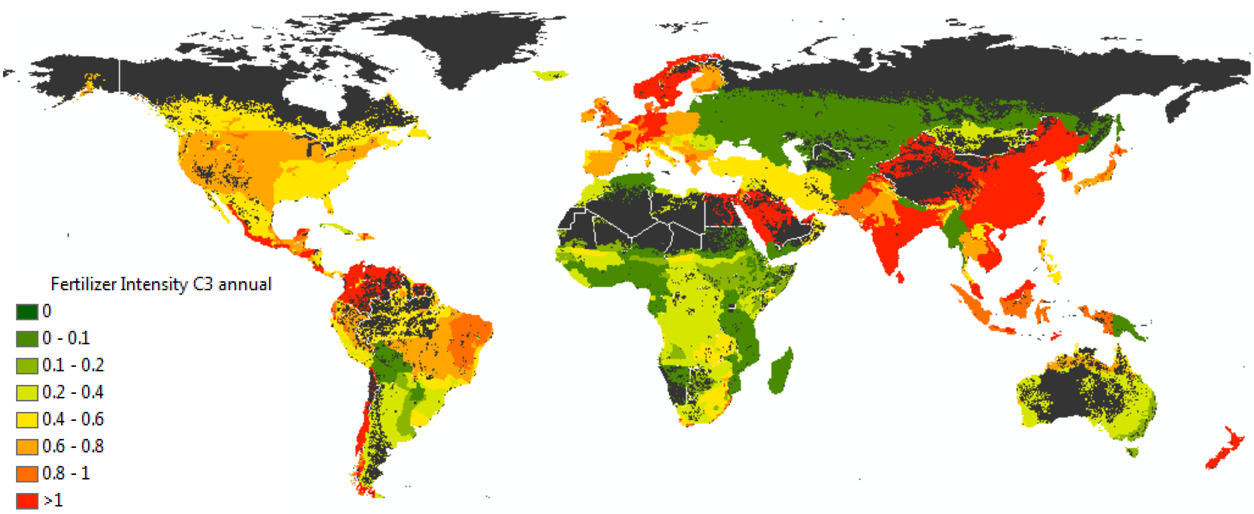


Figure 63: Fertilizer intensity of C3 annual crop production

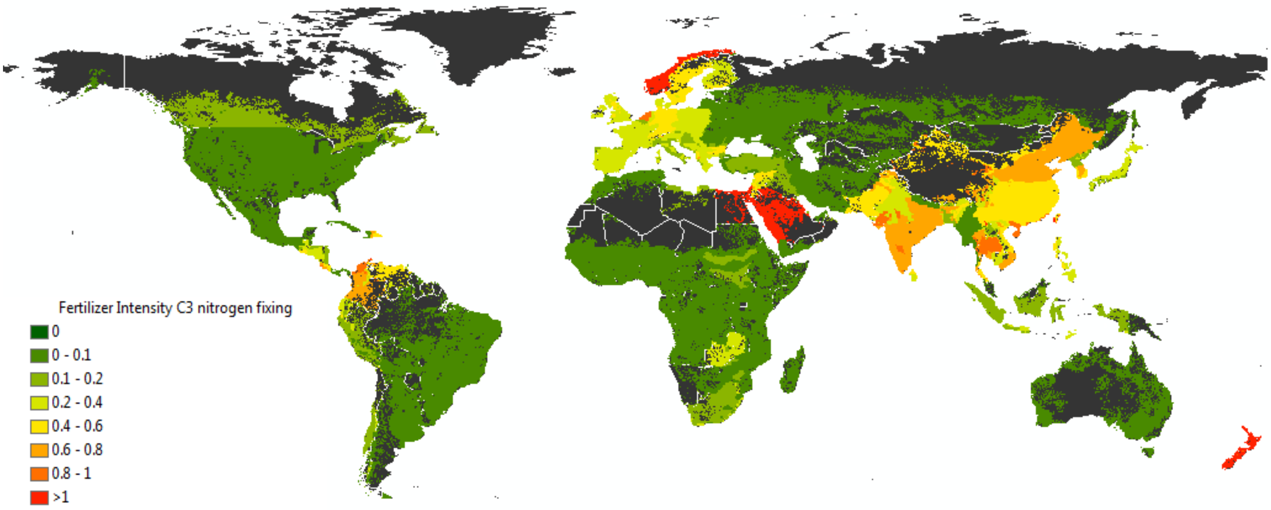


Figure 64: Fertilizer intensity of C3 nitrogen-fixing crop production

Mechanization intensity

The FAO's global data set on tractor and harvester use per hectare, year and country is used as an indicator of mechanization intensity. To produce a spatial intensity map, the tabular FAO data [194] are joined to a vector map based on the corrected country names. The most recent maximum number of tractors and harvesters per year and ha is used for countries where data was missing. Additional values are obtained from further statistics, as some countries such as Australia and Germany are missing in the FAO statistics. The total number of tractors and harvesting machines per year and land is then divided by the total agricultural area per land to obtain the number of agricultural machines per hectare cropland. The vector map is converted into a raster map with the same resolution as the land use model of Hurtt et al. [186], see Figure 65.

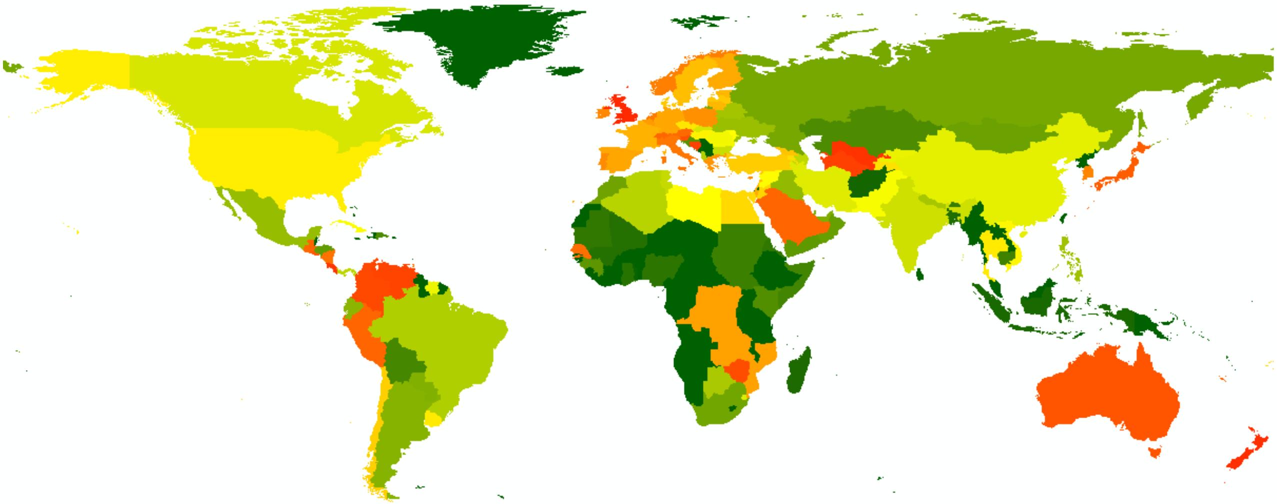


Figure 65: Mechanization number of tractors per ha and year (based on FAO statistics and additional data)

This raster map is then used to disaggregate the national data values, based on the location of different crops and areas using the land use model of Hurtt et al. 2011. For the spatial disaggregation of the national statistical data (number of tractors per ha and year per land) the proportional calculation approach of Milego & Ramos [298] is used. Herein, a grid cell receives its downscaled value depending on the fraction of a grid cell that shares the value within a cell, so that the cell value is calculated as follows:

$$\text{Grid cell Value}_i = \sum (V_i * \text{Share}_i) \tag{26}$$

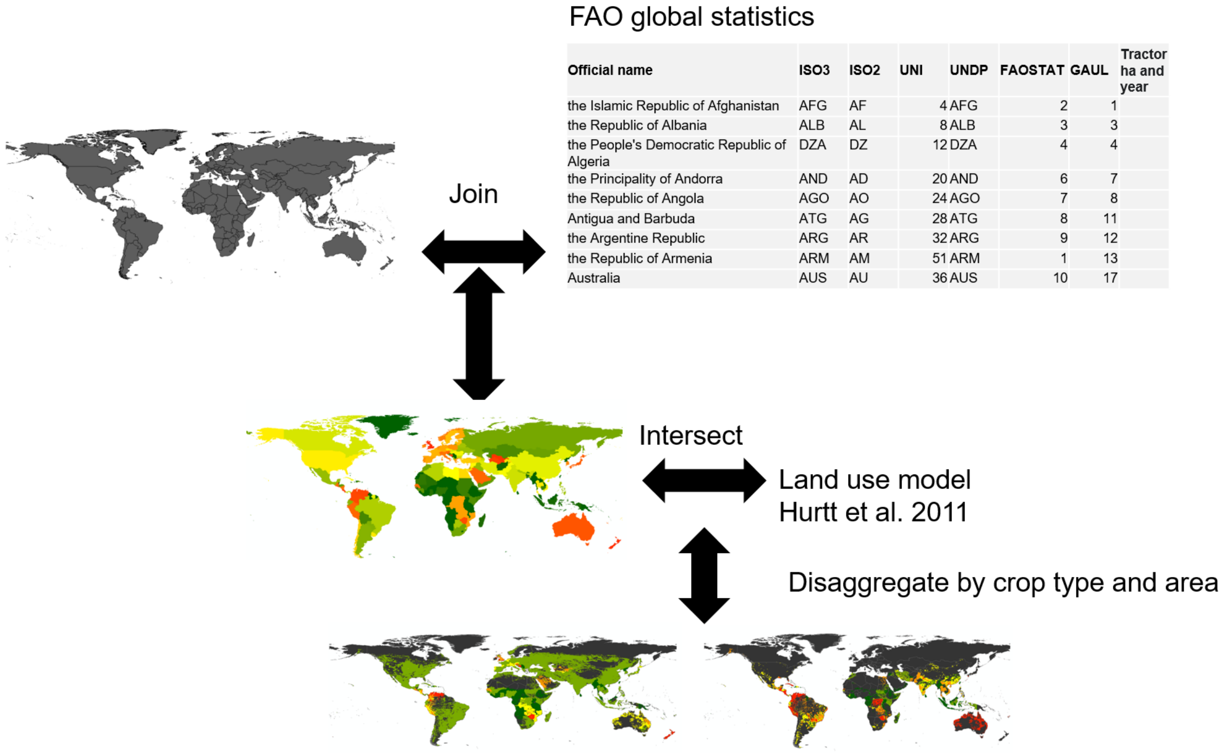
Where

V_i: Value of unit i

Share_i: Share of unit i within grid cell

The disaggregation of national statistics is carried out according to the procedure shown in Figure 66. Herein, a global map with the countries' boundaries is joined to the national statistics table based on the unique country code, containing the values of the management parameter per country. The land use model of Hurtt et al. [186] is intersected with the resulting map to receive only the information for areas where the land use type takes place. The values of the management parameter are then disaggregated by crop type and by crop area to receive the specific values for each land use flow, according to the equation (27).

The same calculation method applies to all data sources that are not available as spatial data but only as national statistics in a table.



Official name	ISO3	ISO2	UNI	UNDP	FAOSTAT	GAUL	Tractor ha and year
the Islamic Republic of Afghanistan	AFG	AF	4	AFG	2	1	
the Republic of Albania	ALB	AL	8	ALB	3	3	
the People's Democratic Republic of Algeria	DZA	DZ	12	DZA	4	4	
the Principality of Andorra	AND	AD	20	AND	6	7	
the Republic of Angola	AGO	AO	24	AGO	7	8	
Antigua and Barbuda	ATG	AG	28	ATG	8	11	
the Argentine Republic	ARG	AR	32	ARG	9	12	
the Republic of Armenia	ARM	AM	51	ARM	1	13	
Australia	AUS	AU	36	AUS	10	17	

Figure 66: Spatial disaggregation of national statistics by area and crop type after [298]

The disaggregation of the national statistics is done using R and RStudio. The number of tractors per ha and year for cropland production is calculated. In order to obtain the intensity value for the indicator mechanization, the benchmark value is calculated for each crop type and global agro-ecological zone according to the maximum tolerable intensity approach. The resulting benchmark map and the benchmark values for mechanization are shown in Figure 67. A high mechanization MTI value is depicted in red, with about four tractors per hectare and year, whereas areas in green show a low benchmark value for mechanization with less than 0.4 tractors per hectare.

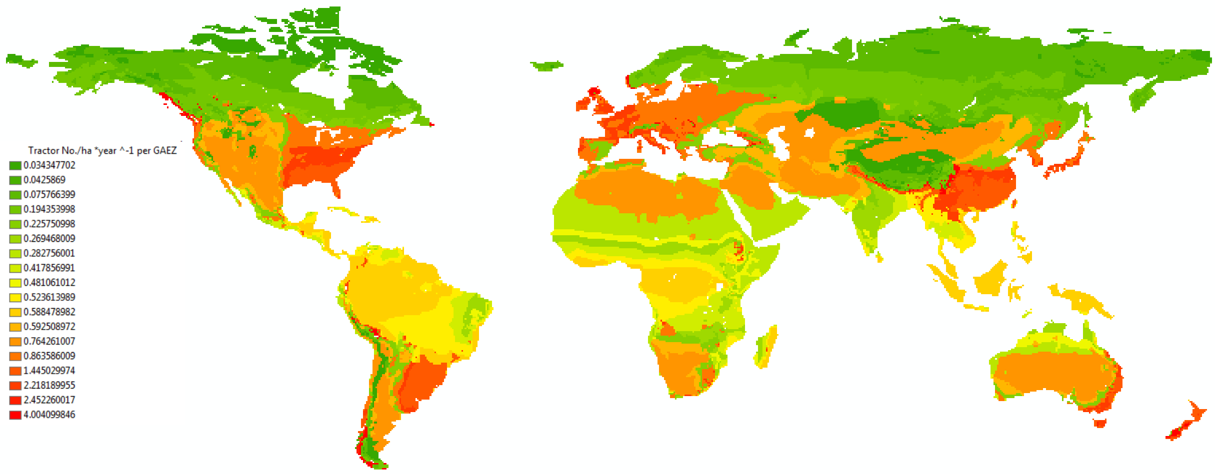


Figure 67: Mechanization No of tractors per ha and year, benchmark values per GAEZ

Management parameter pesticide

The FAO global database (FAO 2019) is used to calculate pesticide application intensities. Since the calculation procedure for the pesticide management parameters is based on the same approach as for mechanization, only the results are displayed and described for this parameter. Herein, the highest pesticide application values are found in some agro-ecological zones of China, Europe, Eastern US or Southern Brazil, Uruguay and parts of Argentina. Low values of pesticide application are found for example in some Scandinavian countries, northern Russia, Canada and Alaska and the Sahel countries (see Figure 68).

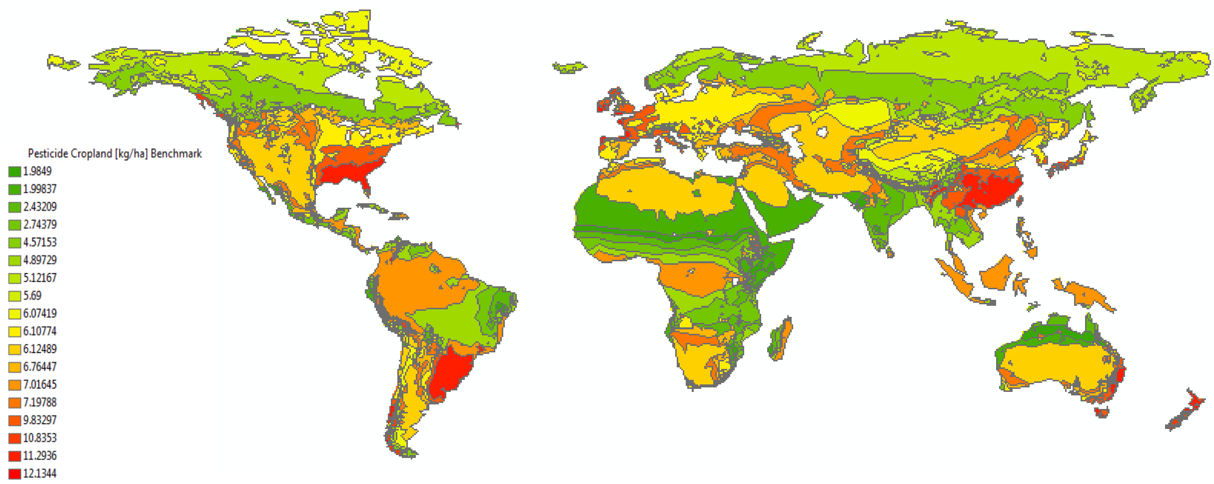


Figure 68: Pesticide application cropland benchmark value per GAEZ

Management parameter crop rotation

The transition data sets of Hurtt et al. [186] are used for the evaluation of crop rotation from one crop type to another per grid cell for the year 2015 (as base year of occupation). The values are aggregated per grid cell for the C3 annual, C4 annual and C3 nitrogen-fixing transitions (not for perennial crops). The total cultivated area is divided by the crop rotation area in order to obtain the percentage of crop rotation both per grid cell and as an average per country. Since this management parameter is one of the relief indicators (measurement of positive effects), its values are linearly transformed so that 0% of the crop area in crop rotation (all areas with monoculture) receives an intensity value of 1 and 100% a value of 0.

The proportion of the crop rotation area in relation to the total area under cultivation is shown in Figure 69 per grid cell. There is a higher proportion of crop rotation systems in countries in Africa, South America and the USA. The agricultural systems in Europe have a lower crop rotation share.

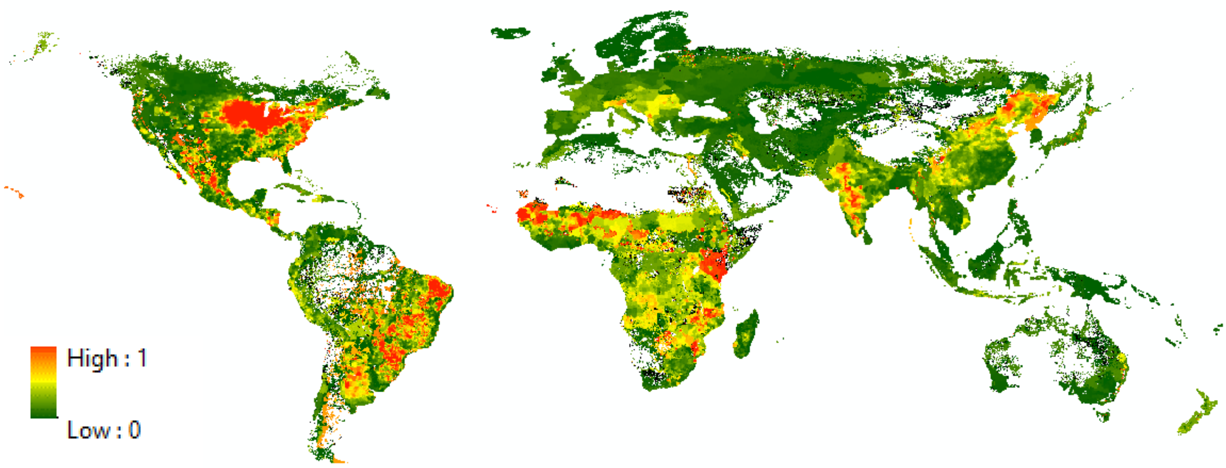


Figure 69: Share of area under crop rotation in comparison to total cropland area

Management parameter set-aside area

For the management parameter set-aside land, the average worst value (intensity value of 0.8) is assumed for the background data set, which corresponds to an average set-aside land of 4 percent per land and cultivated area (see Table 11).

Table 11: Set-aside area in cropland and its intensity values

Set-aside area [%]	Intensity value
0	1.0
4	0.8
10	0.6
14	0.4
16	0.2
20	0.0

Calculation of land use intensity index for land use type cropland

The total LUI for the background data in respect to cropland (flow and its sub flows) is calculated per grid cell as follows:

$$LUI_{Cropland[i]} = \frac{F[i]}{F[MTI]} + \frac{Me[i]}{Me[MTI]} + \frac{P[i]}{P[MTI]} + \frac{\text{Area Crop Rotation}}{\text{Total Crop Area}} + \frac{\text{Set-aside area}}{\text{Total crop area}} \qquad (27)$$

where

$LUI_{Cropland[i]}$: Land Use Intensity index for cropland at location i

F[i]/F[MTI]: Intensity of fertilizer application at location i

Me[i]/Me[MTI]: Intensity of mechanization at location i

P[i]/P[MTI]: Intensity of pesticide application at location i

Area Crop Rotation/Total Crop Area: Share of crop rotation in total crop area at location i

Set-aside area/Total crop area: Share of set-aside area in total crop area at location i

A mean LUI for the analytical unit “country” is then calculated for the land use type cropland and its sub land use types using zonal statistics and a vector map depicting state boundaries. The LUI maps are depicted in Figure 70 a & b exemplarily for the land use flow cropland and its sub flow cropland, C3 annual. The LUI maps for the other sub flows are depicted in Annex V.

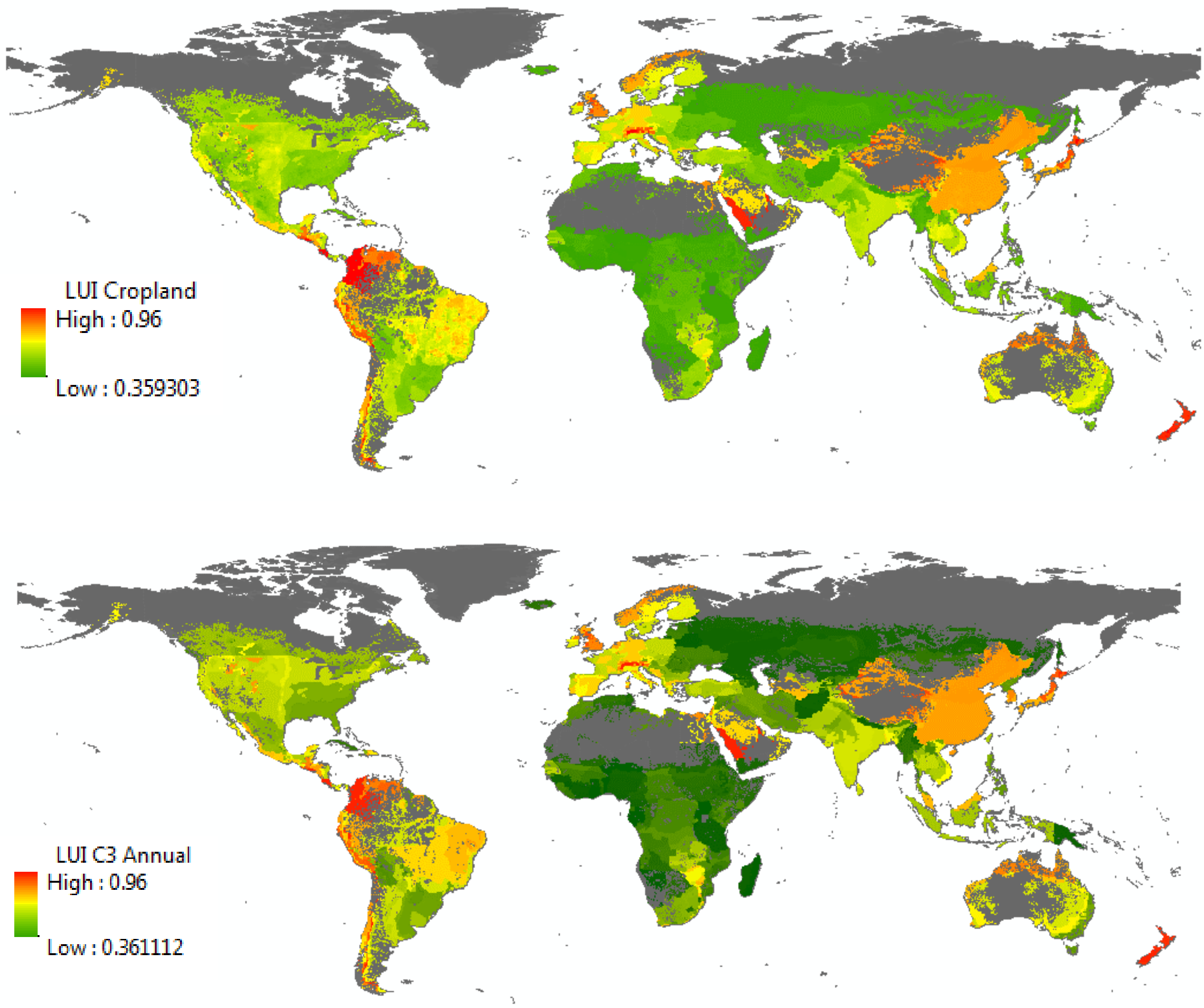


Figure 70 a & b: Land Use Intensity Index for cropland production

5.2.5.2 Land use intensity index pasture

The global data sets that are used to calculate the land use intensity index for the land use type pasture and its sub types managed pasture and rangeland pasture are depicted in Table 12.

Table 12: Data sets for LUI pasture

Management parameter	Indicator [unit]	Data Type	Global data source
Manure on Pasture Leaching	Amount of manure leaching [$kg_{N\text{-}leaching}/(ha*a)$]	Global maps, primary data	[186]
Livestock intensity	Tropical livestock per area [tropical livestock units/(km^2*a)]	Global maps sheep, goat, cattle, buffalo distribution, primary data	[273]
Pesticide	Pesticide application rate [$t_{active\ ingredients}/(ha*a)$]	Global maps, FAO statistics, primary data	[193]
Mechanization	Number of machines [$No_{tractor/harvester}$ /(ha*a)]	FAO statistics, primary data	[194]
Set-aside areas/ecotones	Ratio set-aside area per field area [%]	Primary data, satellite images	N/A

Livestock intensity

The data from Gilbert et al. [299] for the global distribution of ruminant species of goats, sheep, cattle and buffalos are used to calculate the intensity of the management parameter livestock intensity. Each species has a different body mass and thus a different need for plant uptake, which puts varying pressures on biodiversity in grazing systems. Therefore, the number of ruminant species per km^2 is converted into Tropical Livestock Units (TLUs). The TLUs take into account the difference in grazing intensity due to body mass and feed requirements and uses a “tropical cow” as a reference. The geographical differences in the availability of plant biomass are taken into account by comparing the different intensity levels only within the individual global agro-ecological zones. This means that the intensities are only compared in regions that naturally have similar amounts of biomass available. For example, the grazing intensities in dry areas with less biomass are compared only with each other. Thereby, also regional differences in climate and soil are considered, as recommended by Fetzel et al. [300].

The number of goats, sheep, cattle and buffaloes per area of land use type pasture and subtypes managed pasture and rangeland pasture is calculated from data from Hurtt et al. [186] and Gilbert et al. [299]. Therefore, the maps of Gilbert et al. [299] are converted to the same resolution of Hurtt et al. [186]. The livestock densities (head per km^2) of each of the four ruminant species are converted to TLU using the conversion factors of [219,300,301]. The conversion factor for the number of heads of ruminant species to TLU is 0.6 for cattle, 0.5 for buffalo and 0.1 for sheep and goats [219,300,301]. If it is known which livestock species belongs to the land use type pasture, the grazing intensity of this species can be calculated directly from these conversion factors and the proposed benchmark values. The specific TLUs are combined for the land use type pasture and the subtypes rangeland pasture and managed pasture. The maximum intensity level of TLU is calculated per agro-ecological zone. The intensity of the management parameter livestock is calculated by comparing the TLUs with the maximum intensity level of the TLU. The livestock intensity is calculated both for the individual grid cells and for the intensity of individual countries used as background data.

As can be seen in Figure 71, the maximum intensity levels for grazing are lower in dessert areas such as the Sahara or central Australia. The highest intensity levels are in temperate zones and in tropical regions where more biomass is available for grazing.

The TLU maps per km² in regard to sheep, cattle, goats and buffaloes as well as the resulting intensity maps are shown both at grid cell and country level in Annex V. For cattle, for example, there are up to 990 tropical livestock units per km² in some regions of South America, India, Europe or Eastern Africa (see Figure 72). For the land use class pasture there are quite high grazing intensities in e.g. Great Britain, New Zealand, India or Ethiopia. The grazing intensity of the subclass rangeland pasture is particularly high in regions of Brazil or Pakistan, whereas the intensity level of the land use class managed pasture is higher in India, China, Great Britain or Ethiopia (see Figure 73).

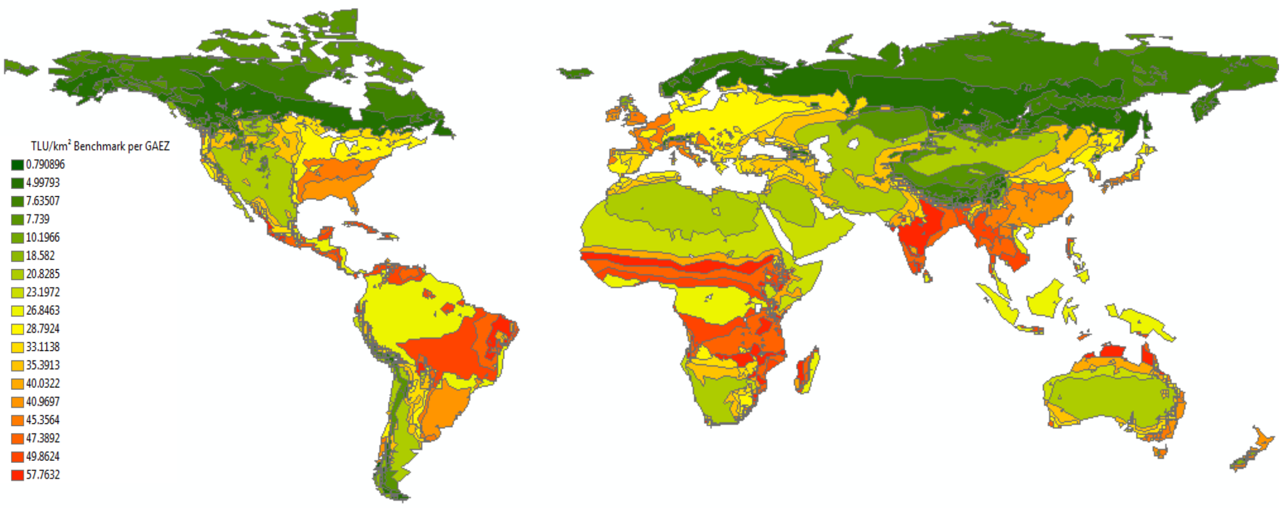


Figure 71: Maximum intensity levels for grazing intensity

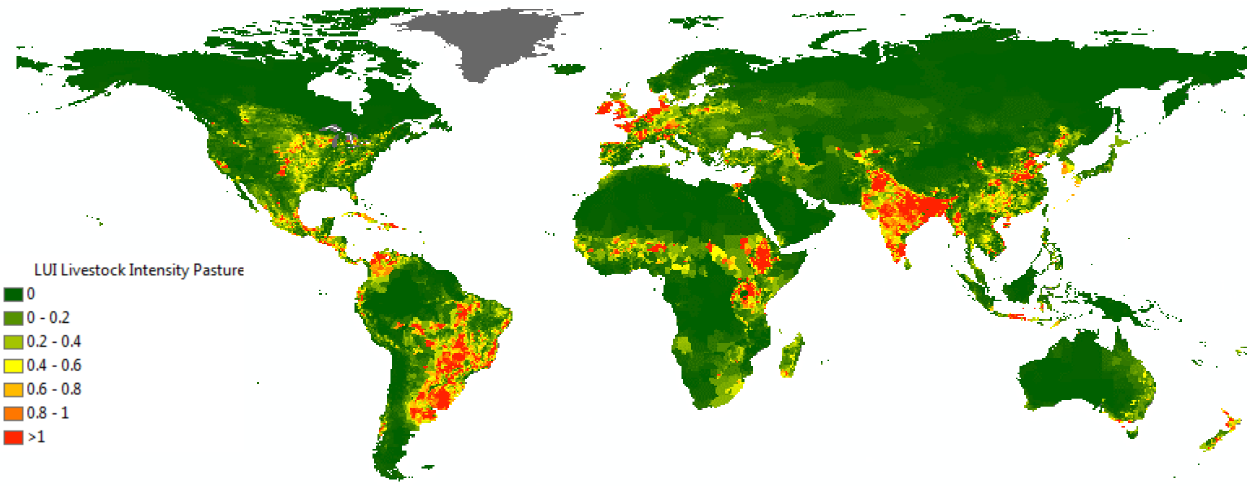


Figure 72: Number of cattle per km² expressed in tropical livestock units (TLU)

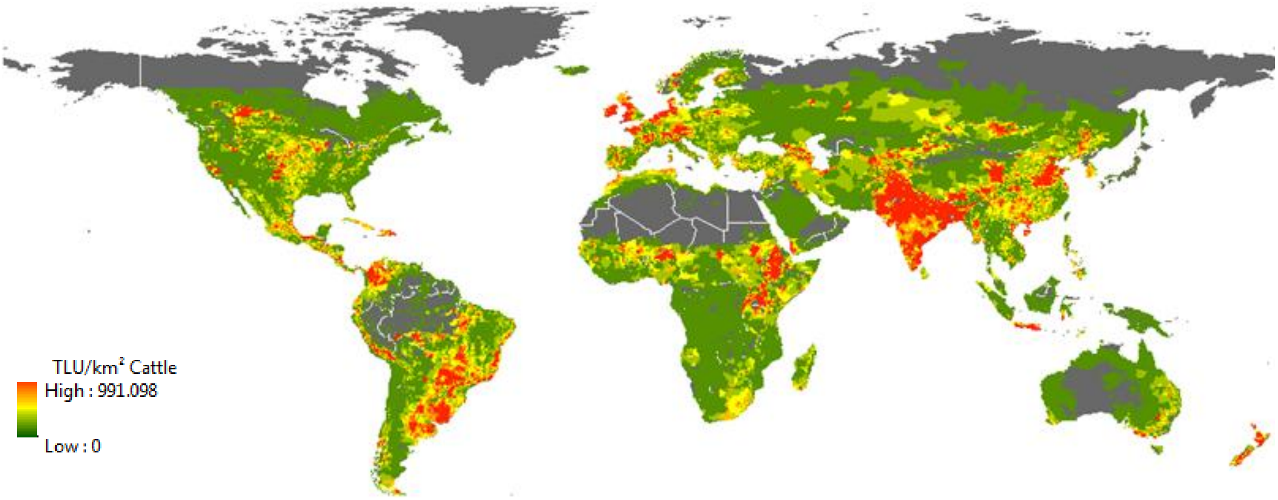


Figure 73: Livestock intensity for land use flow pasture

Management parameter pesticide

For the management parameter pesticide use, the same approach and data is used as for the calculations of the cropland pesticide intensity. This is done since no specific global data set for pasture is available [206]. Especially for this management parameter it is recommended to use primary data. Furthermore, the data has to be corrected in future as soon as global data sets for pesticide use on pastoral grassland become available. The maximum tolerable intensity level for pesticide application in pastoral systems is depicted in Figure 74.

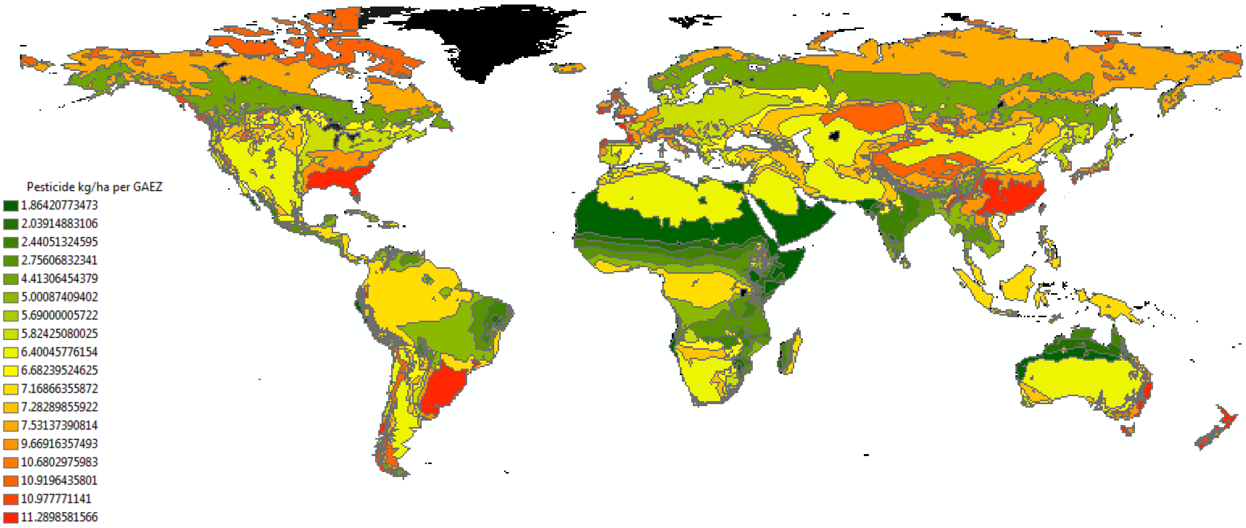


Figure 74: Maximum tolerable intensity level of pesticide application in pasture

In general, the pesticide maximum intensity levels are higher in boreal and temperate regions. Similar to the application of fertilizers, also pesticide applications depend on several external factors characteristic for the GAEZ, such as light, temperature, moisture, pH of the soils as well as the crop types cultivated [302]. High intensity levels of pesticide application can be found especially in China and some South American countries, such as Brazil. For the land use sub flow managed pasture, the pesticide intensity is quite high in Europe, such as Germany or Italy (see Figure 75).

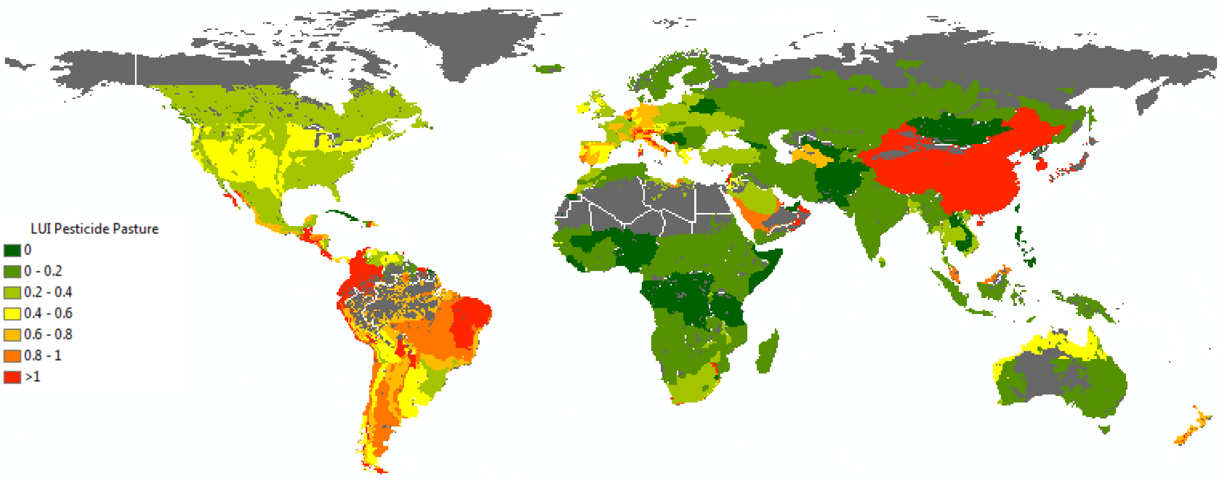


Figure 75: Pesticide use intensity for the land use flow pasture

Management parameter manure on pasture

The country statistics of the FAO [192,272] serve to calculate the intensity of fertilizer in the form of manure on pasture. Herein, not the total manure applied on pasture area is taken into account but only the data set on manure leaching, since the excess of nutrient is especially harmful for biodiversity. The data set is joined on the country shapefile and the kg of manure leaching is calculated by dividing the total value per country by the area of pasture land per country. The shape file is converted to a raster file using the same resolution as the model of Hurtt et al. [186]. The specific amount of manure on pasture leaching is then calculated for the land use flow pasture and the sub flows managed pasture and rangeland pasture:

$$Manure_{ManagedPasture}[kg/ha] = Manure_{PastureTotal}\,[kg/ha] * \frac{Area_{ManagedPasture}[ha]}{Area_{PastureTotal}\,[ha]} \tag{28}$$

where

$Manure_{ManagedPasture}$: Manure leaching of land use sub type managed pasture in kg per ha

$Manure_{PastureTotal}$: Amount (kg) of manure leaching at pastoral land per ha

$Area_{ManagedPasture}$: Area of land use sub type managed pasture in ha

$Area_{PastureTotal}$: Total Area of land use type pasture in ha

The maximum intensity level of manure leaching is calculated per global agro ecological zone and is depicted in Figure 76.

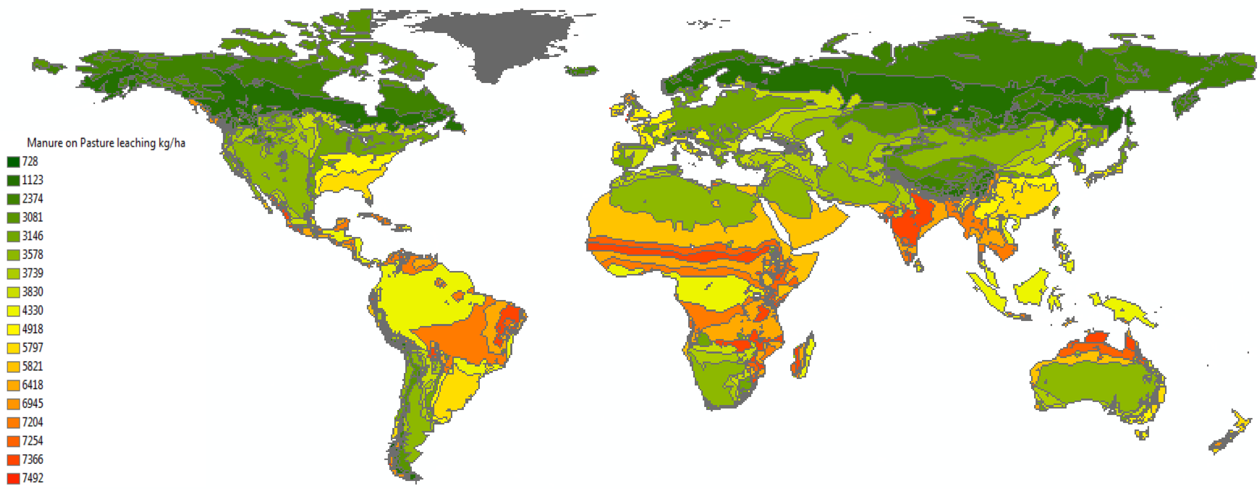


Figure 76: Maximum intensity level of manure leaching

The intensity of manure leaching on pasture, rangeland and managed pasture is derived by dividing the values per grid cells by the maximum intensity level of each agro ecological zone. The mean LUI for each land use flow is calculated for each country using the *extract()* function in RStudio. The intensity map is depicted in Figure 77 for rangeland pasture. Whereas the intensity of manure leaching of rangeland pasture is quite high in Pakistan, Ethiopia, New Zealand or some regions in China, the intensity of manure leaching in managed pasture systems is quite high in regions of Great Britain or India (see Annex V).

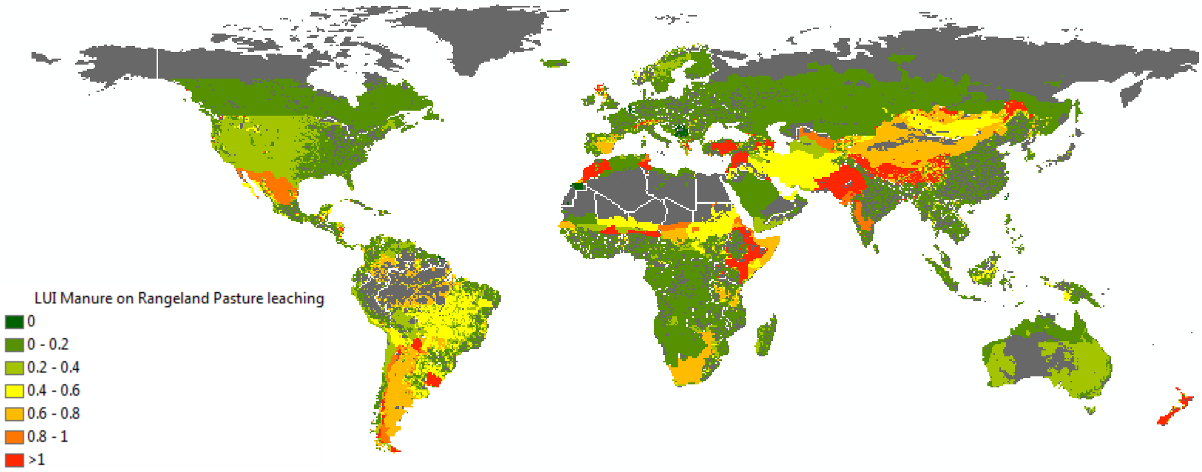


Figure 77: Intensity of manure leaching in rangeland pasture

Management parameter mechanization

For the calculation of the intensities of the management parameter mechanization in pasture, the same national statistics are used as for the mechanization of cropland, since no specific data are available for pasture. The national statistics provide values for the number of tractors used per hectare per year for each country. The specific number of tractors is disaggregated per hectare of the land use flow pasture and the sub flows rangeland pasture and managed pasture. The maximum intensity level is calculated per global ecological agricultural zone and is depicted in Figure 78. The intensity of mechanization is then standardized for each grid cell with the respective maximum intensity levels. The average mechanization intensity for the background database is calculated for each country and land use flow using the *extract()* function.

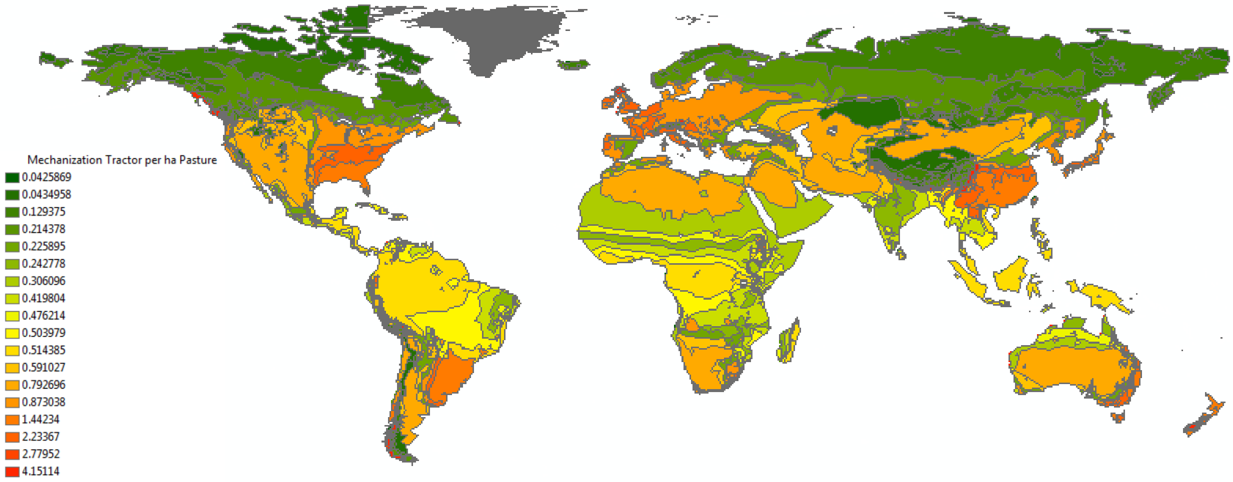


Figure 78: Maximum intensity level of mechanization per global agro ecological zone

Management parameter set-aside areas

Default values are used for the management parameter set-aside land because no global data sets are available on set-aside land in pastures. In contrast to managed pasture, rangeland pasture usually takes place as part of a nomadic and transhumance system in the native primary or secondary vegetation [303], livestock is constantly moved to different areas so that some other areas are spared at the same time. Since not all grazing land is used permanently, the lowest value for grazing land accounts for about 70% of set-aside land. Therefore, the lowest average intensity default value for rangeland pasture is 0.3 which means that about 50% of the area is not used. Nevertheless, degradation can occur on rangeland pasture if, for example, the livestock intensity or pesticide intensity is too high (see other management parameters). In contrast, managed pasture is defined as a form of pasture that transforms the previous land and uses human-planted vegetation. In addition, the livestock remains on the same land and is usually fenced in. As a standard value, it receives an average intensity of 0.8 for the background database. The average intensity of the set-aside areas of rangeland pasture and managed pastures is used for the overall land use flow pasture. The percentage of set-aside land for the land use type pasture and its related intensity values are depicted in Table 13.

Table 13: Set-aside area and its intensity values for pasture land

Set-aside area [%]	Intensity value
0	1
14	0.8
35	0.5
49	0.3
70	0.0

Land Use Intensity Index Pasture

The Land Use Intensity Index for pasture is calculated using mean overlay statistics for the three land use flows of pasture, rangeland and managed pasture.

$$LUI_{Pasture[i]} = \frac{GI[i]}{GI[MTI]} + \frac{Me[i]}{Me[MTI]} + \frac{Ma[i]}{Ma[MTI]} + \frac{Pe[i]}{Pe[MTI]} + \frac{SetAside[i]}{Totalarea[i]} \quad (29)$$

where

$LUI_{Pasture[i]}$: Land Use Intensity index for pasture at location i

GI[i]/GI[MTI]: Intensity of grazing at location i

Me[i]/Me[MTI]: Intensity of mechanization at location i

Pe[i]/Pe[MTI]: Intensity of pesticide application at location i

SetAside/Total area: Share of set-aside area in total pasture area at location i

The resulting maps can be seen in Figure 79 and Figure 80.

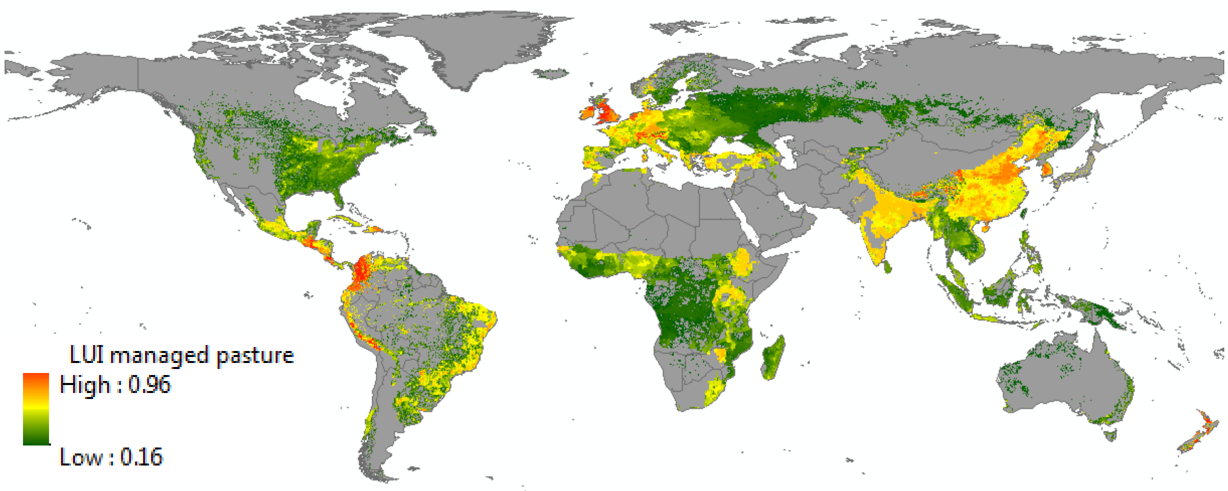


Figure 79: Land use intensity index managed pasture

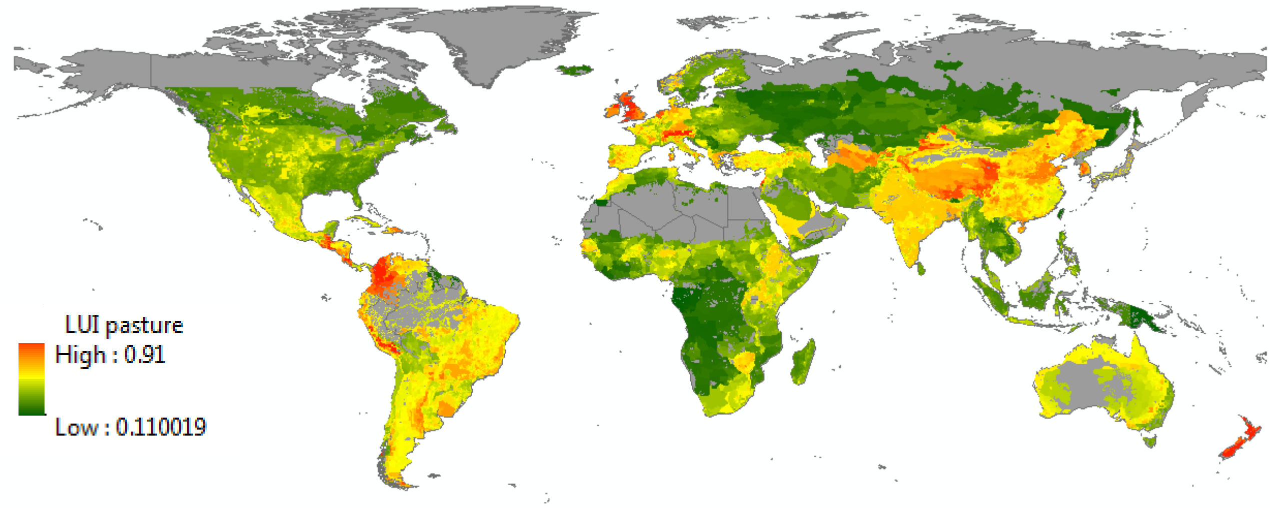


Figure 80: Land use intensity index pasture

5.2.5.3 Land use intensity index urban

The data sets to calculate the land use intensity index for urban areas is depicted in the following Table 14.

Table 14: Data sets for the calculation of the LUI urban

Land use class	Management parameters	Indicator [unit]	Data sources	References
Urban	Set-aside area	Set-aside area per urban area [%]	Satellite images, primary data	N/A
	Sealing	Imperviousness [%]	Global maps, primary data	[284]
	Light pollution	Artificial sky brightness [mcd/m²]	Global maps, statistics, primary data	[196]
	Population density	Number of persons per area [$No_{persons}/km^2$]	Global maps, primary data	[287]
	Traffic intensity	Number of vehicles per area and year [$No_{vehicles}/(km * a)$]	National statistics, primary data	[277]

Management parameter sealing

For the management parameter sealing, the data set of Elvidge et al. [284] is used. It shows the degree of sealing as the global density of the constructed impervious surface [284]. The unit of the data set is % sealed area per grid cell. The data set is only analyzed in areas classified as urban in the land use model of Hurtt et al. [186]. In contrast to the calculation of the land use intensity indices for other land use types, the management parameters of the urban land use type are not directly dependent on soil or climatic conditions. The degree of sealing in an urban area, for instance, does not depend on the soil or climate but only on the management decision of the land owner, whereas the required quantities of fertilizer are influenced by crop type and geographic characteristics. Therefore, the maximum intensity level is not calculated for each global agro ecological zone, but as one benchmark value for all urban areas. Note that due to the resolution of the data set of Elvidge et al. [284] the highest sealing intensity value for one grid cell can only reach 60%. For the foreground data, however, a highest sealing intensity of 100% has to be taken.

Management parameter light pollution

The management parameters light pollution use the data sets on artificial nightlights of Falchi et al. [196]. They produced maps showing the artificial sky brightness of the zenith. The simulated zenith radiation data are measured in mcd/m². The model contains only artificial light, without natural light. This makes it suitable as an indicator for light pollution. Areas classified as urban by Hurtt et al. [186] are included in the calculation. In order to obtain information about the data distribution, cell statistics are calculated to obtain the maximum, mean and standard deviation. Only a single benchmark value is calculated as also the intensity of artificial light pollution does not depend on geographic external variables. The data set is standardized by the maximum intensity level of the zenith radiation data (see Figure 81).

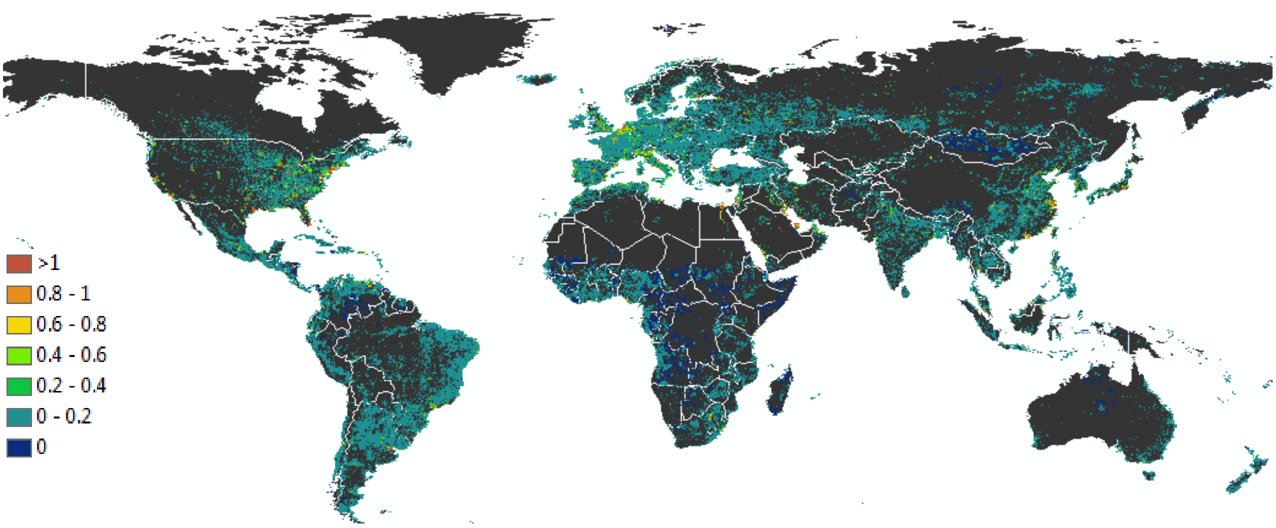


Figure 81: LUI light pollution standardized by global maximum intensity level

Management parameter population intensity

For the parameter population intensity, the data set of the CIESIN [287] is used. It is resampled to the resolution of the land use model of Hurtt et al. [186] using nearest neighbor interpolation. For the analysis, only those areas are considered that were classified as urban [186]. The benchmark value is calculated using the MTI approach. The population intensity map shows high intensity levels in areas in the Netherlands, India, Indonesia or China (see Figure 82).

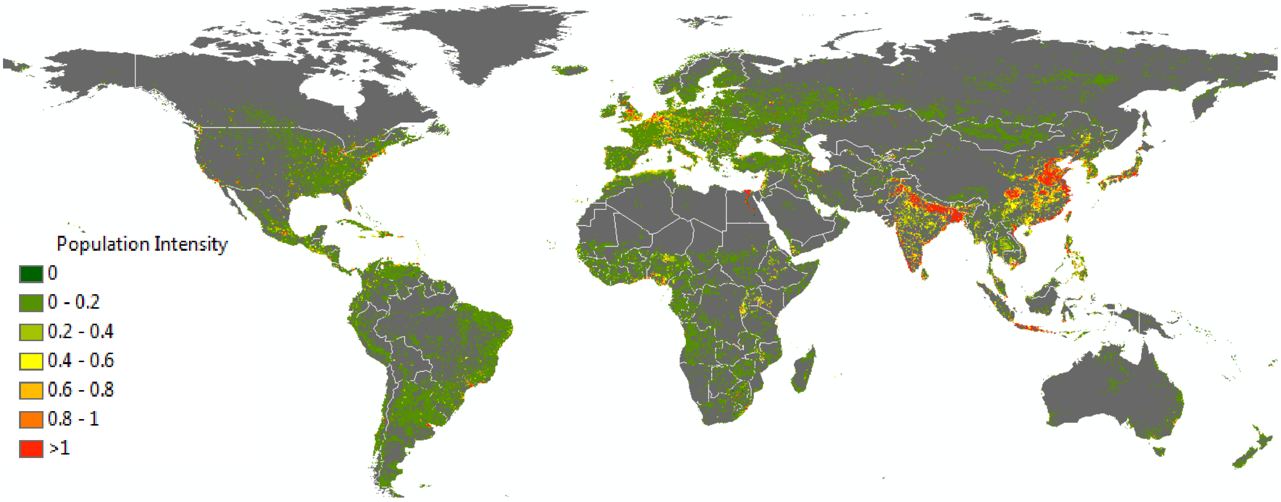


Figure 82: Population intensity

Management parameter traffic intensity

For the management parameters traffic intensity, the UNECE traffic intensity data are used as national statistics (motor vehicle movements on national territory by road) [277]. Values for 53 countries are available in this database. For these countries, data from 2017 are used, except for the Netherlands and Slovenia, where only the 2016 data are available. For the remaining countries, the average highest traffic intensity value is used as default for the background calculations. For these countries it is particularly important to use primary data when calculating the LUI in order to improve the results. The maximum intensity level of traffic is calculated from the data sets available. This value should also be updated in the future as new statistical values become available.

Management parameter set-aside areas/green space

No global data record is available for the management parameter set-aside land in urban areas. Nevertheless, green areas can be identified by satellite images. The Normalized Difference Vegetation Index (NDVI) can be calculated to provide information about the state of vegetation. The index provides values in the range of -1 to 1. The negative values can be used to identify water areas, while low values from 0 to 0.2 stand for vegetation-free areas. Values in the range of 0.9 to 1 indicate a high vegetation cover. Since no global data sets are available, this method calculates the NDVI as shown with an example of a town in Germany. The images are taken from the satellite Sentinel 2A of the European Union. The calculation is done with the band 8 and band 4 of the satellite images, which contain reflection values in the near infrared range and in the red visible range. The formula used to perform the calculations is listed below. The resulting NDVI map is shown in Figure 83.

$$NDVI = (NIR - Red)/(NIR + Red) \quad (30)$$

where

NDVI: Normalized Difference Vegetation Index

NIR: Spectral reflectance measurements in the near-infrared range

Red: Spectral reflectance measurements in the red (visible) range

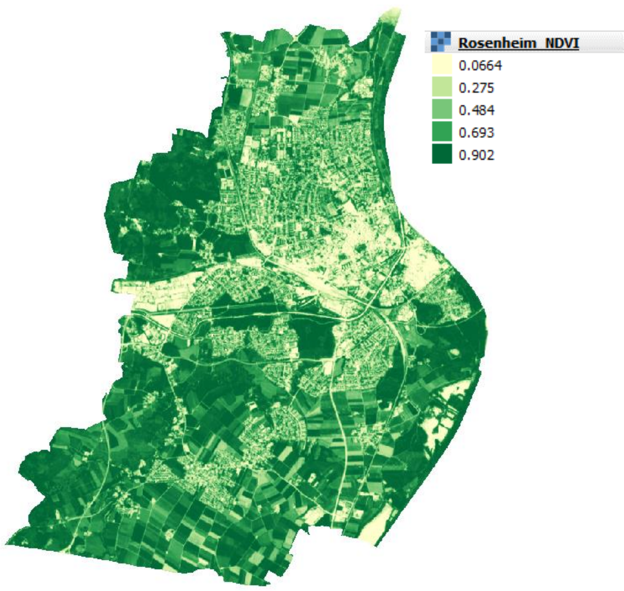


Figure 83: NDVI values indicating green areas

Areas with an NDVI greater than 0.6 can be classified as green areas. The sum of these green areas for the whole city is then divided by the total area of the city to obtain a value for the proportion of green area. It is important to exclude all areas that are classified as separate land use types, such as cropland or areas with primary vegetation, as they are by definition not part of the urban land use type (see Figure 84). These areas are taken into account in the LDI calculations.

Figure 84: NDVI values only larger than 0.6 indicate green areas, (areas of the land use type cropland are excluded)

If the specific area of land use is known, the set-aside area may be calculated according to the method described above. For the background data, however, the average intensity value is assumed.

Land use intensity index urban

The land use intensity index for urban areas is calculated from the intensity maps of the management parameters light pollution, degree of sealing as well as population and traffic intensity. No global data sets are available for the other management parameters, therefore the highest average intensity value per country is assumed. These management parameters should be considered in particular in the foreground calculations.

$$LUI_{Urban[i]} = \frac{LP[i]}{LP[MTI]} + \frac{Se[i]}{Se[MTI]} + \frac{Pop[i]}{Pop[MTI]} + \frac{T[i]}{T[MTI]} + \frac{SetAside[i]}{Totalarea[i]} \tag{31}$$

where

$LUI_{Urban[i]}$: Land Use Intensity index for urban at location i

LP[i]/LP[MTI]: Intensity of light pollution at location i

Se[i]/Se[MTI]: Intensity of sealing at location i

Pop[i]/Pop[MTI]: Intensity of human population application at location i

T[i]/T[MTI]: Intensity of traffic at location i

Set-aside/Total area: Share of set-aside area in total urban area at location i

The intensity map for the urban land use class shows a quite high land use intensity especially in the densely populated countries of India, Bangladesh, China, Indonesia and the Netherlands. Further urban areas in the United Kingdom, the USA or Egypt also show a high land use intensity (see Figure 85).

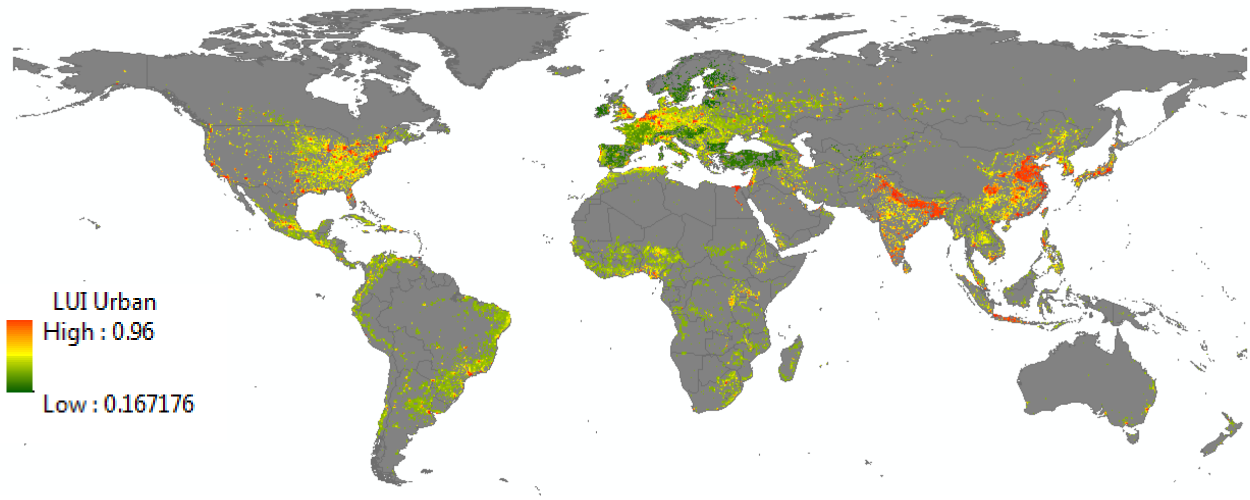


Figure 85: Land Use Intensity Index Urban

5.2.5.4 Land use intensity index forestry (primary and secondary vegetation)

The data sets for calculating the land use intensity index for the land use type forestry are presented in the following Table 15.

Table 15: Data sets for the land use intensity index forestry [5]

Land use type	Sub types	Management parameter	Data type	Indicator [Unit]	Data Source
Secondary vegetation	Forested	Mean age/tree age	Global maps, primary data	Years [a]	[186], primary data
		Wood harvesting rates	Global maps, primary data	Amount of carbon [units kg C]	[186], primary data
		Dead wood volume	National statistics (Europe), primary data	Average deadwood volume [m³/ha]	[195], primary data

Land use type	Sub types	Management parameter	Data type	Indicator [Unit]	Data Source
		Set-aside areas/buffer zones	World protected areas	Protected forest area per total forest area [%]	Satellite images, primary data
		biomass density	Global maps, primary data	Amount of carbon per area [kg C/m²]	[186], primary data

Management parameter tree age

For secondary forests, data are available for the mean tree age [186]. In the LUI's calculations, only areas classified as primary or secondary forests are taken into account. The global agro-ecological zones are not used to calculate the benchmark values for forestry, as they relate to agricultural characteristics. Instead the global ecological zones for forest assessment, as proposed by the FAO [304], are used since they take into account geographical, climatic and altitude characteristics of different forest types; all of which influence the parameters of the LUI. A total of 20 different forest zones are proposed (see Table 16).

Table 16: Global ecological zones for forest assessments based on the FAO [304]

Climate Group	Climatic criteria	Global ecological zone for forests	Criteria
Boreal	< 3 months over 10°C	Boreal coniferous forest	Vegetation physiognomy: coniferous forest dominant
		Boreal mountain system	Approximate > 600 m altitude
		Boreal tundra woodland	Vegetation physiognomy: woodland and sparse forest dominant
Polar	All months below 10°C	Polar	
Subtropical	8 months or more over 10°C	Subtropical dry forest	Seasonally dry, winter rains, dry summer
		Subtropical humid forest	Humid: No dry season
		Subtropical mountain system	Approximate>800 – 1000 m altitude
		Subtropical steppe	Semi-Arid: Evaporation >Precipitation
Temperate	4 to 8 months over 10°C	Temperate continental forest	Continental climate: coldest month under 0°C
		Temperate desert	Arid: All months dry
		Temperate mountain system	Approximate >800 m altitude

Climate Group	Climatic criteria	Global ecological zone for forests	Criteria
		Temperate oceanic forest	Oceanic climate: coldest month under 0°C
		Temperate steppe	Semi-Arid: Evaporation > Precipitation
Tropical	All months without frost: in marine areas over 18°C	Tropical dry forest	Dry/wet: 5 – 8 months dry, during winter
		Tropical moist forest	Wet/dry: 3 – 5 months dry, during winter
		Tropical mountain system	Approximate > 1000 m altitude (local variations)
		Tropical rainforest	Wet: 0 – 3 months dry. When dry period, during winter
		Tropical shrub land	Semi-Arid: Evaporation > Precipitation

With these characteristic forest zones, the benchmark value is also calculated as maximum intensity level of the individual parameter values.

The benchmarks for the mean age of trees are shown in Figure 86 for each characteristic forest zone. It can be seen that older secondary forests with a higher average tree age exist in boreal and subtropical mountain systems. Since the indicator tree age is classified as a relief indicator, the values are rescaled so that the lowest value has the highest intensity value.

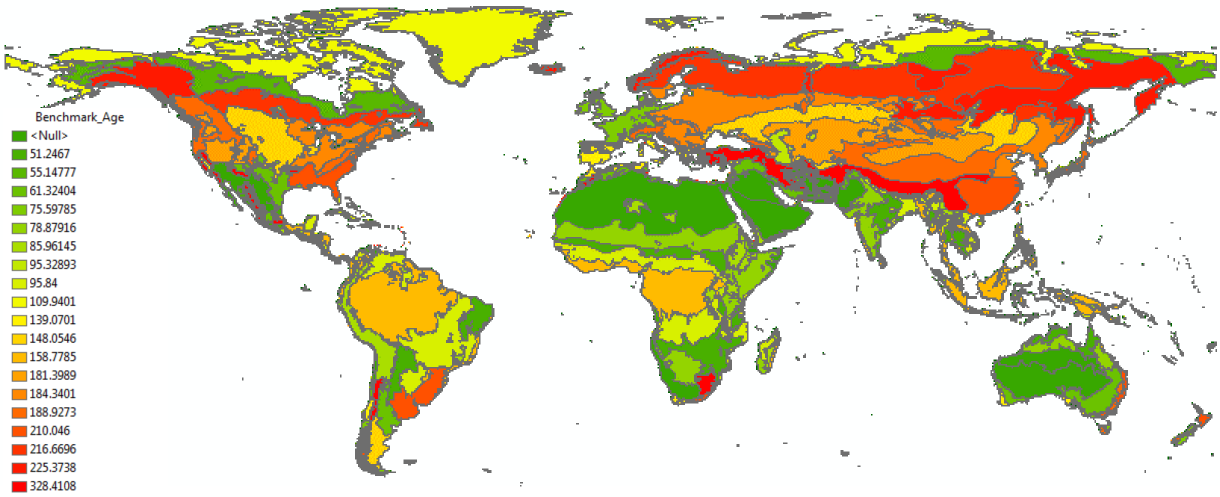


Figure 86: Benchmark values per forest zone for tree age in secondary forests

The intensity values for the management parameter tree age are depicted in Figure 87. Herein, only the areas are evaluated that have been classified as secondary forest by Hurtt et al. [186].

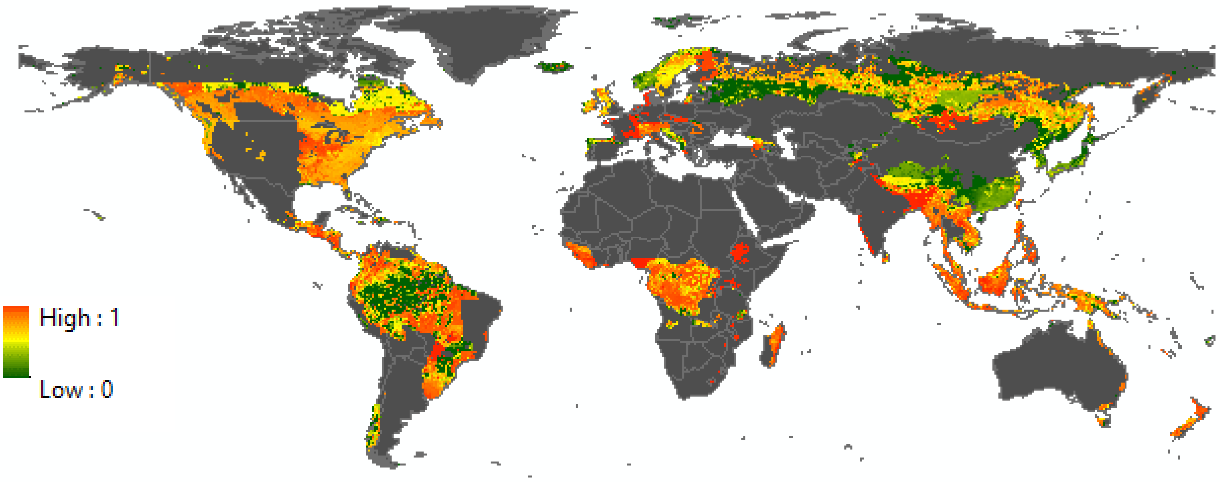


Figure 87: Intensity values of tree age for areas with secondary forest standardized by GEZ

Management parameter biomass density

Data for the management parameter biomass density are available for secondary forests supplied by Hurtt et al. 2011, measured in kg C per km². Only the grid cell values were analyzed in areas classified as forest by [186]. The benchmarks are calculated for each characteristic forest zone. As expected, the highest benchmark for biomass density is in tropical rainforests and the lowest in the Polar Regions. Since biomass density is also one of the relief indicators, the inverted values are used to calculate the LUI (see Figure 88).

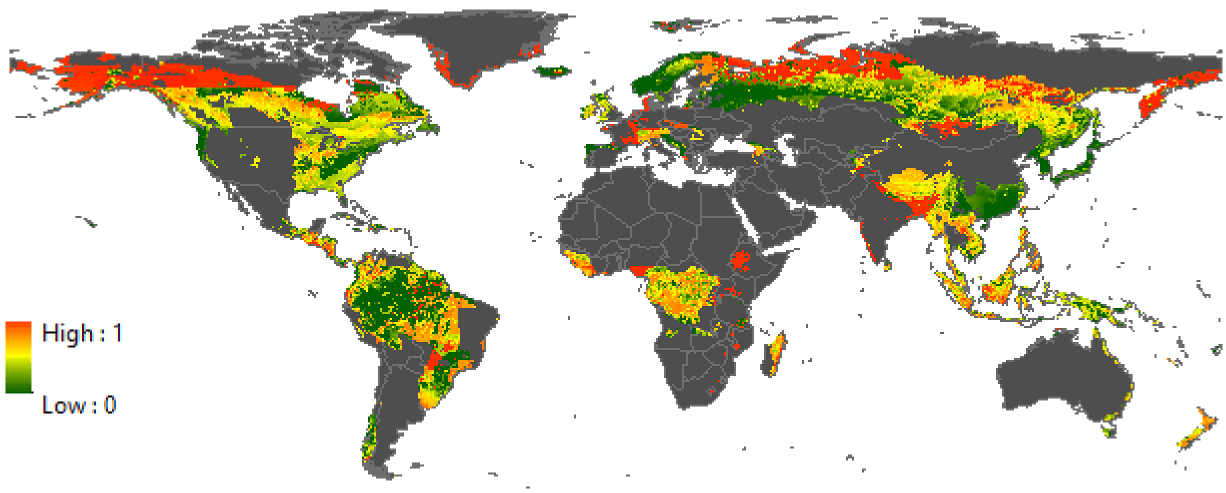


Figure 88: Intensity values for the management parameter biomass density in areas that are classified as secondary forests

Management parameters wood harvest rates

For the management parameter wood harvest rates (in kg C) the transitional data of Hurtt et al. [186] for the year 2015 and the SSP5 scenario are used. These include biomass from primary forests as well as primary non-forests and secondary forests for the succession stages of young and old forests. The total amount of timber harvest is calculated for the land use type forestry. In addition, the total harvest rates kg C per km² and year 2015 in secondary and primary forests are calculated on the basis of cell statistics. The benchmarks per global forest zone for primary and secondary forests are shown in Figure 89.

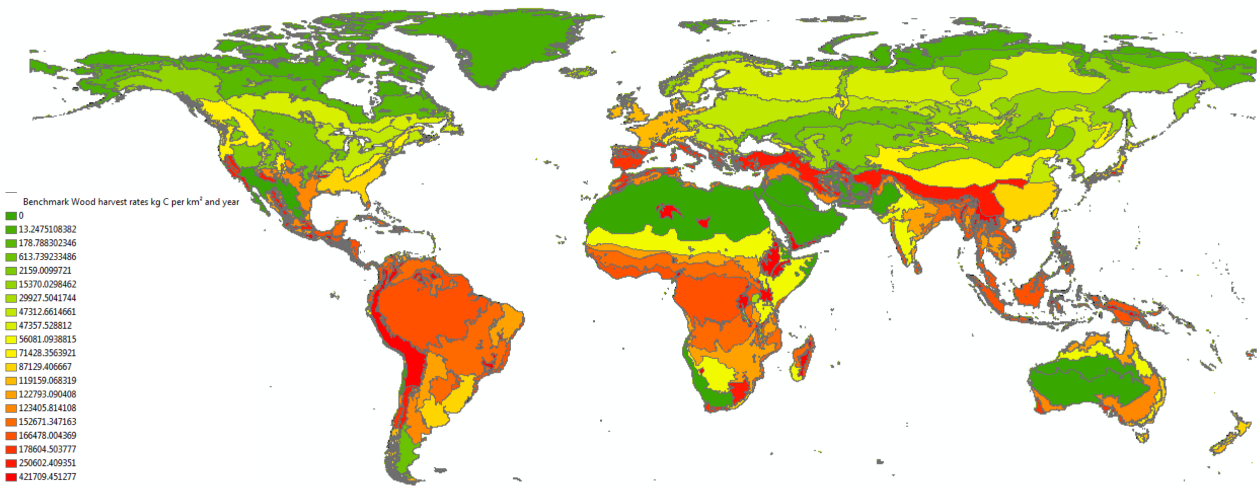


Figure 89: Benchmark wood harvest rates for primary and secondary forest per global ecological zone

The intensity values for the management parameter wood harvesting are depicted in Figure 90 for secondary forests and in Figure 91 for primary forests.

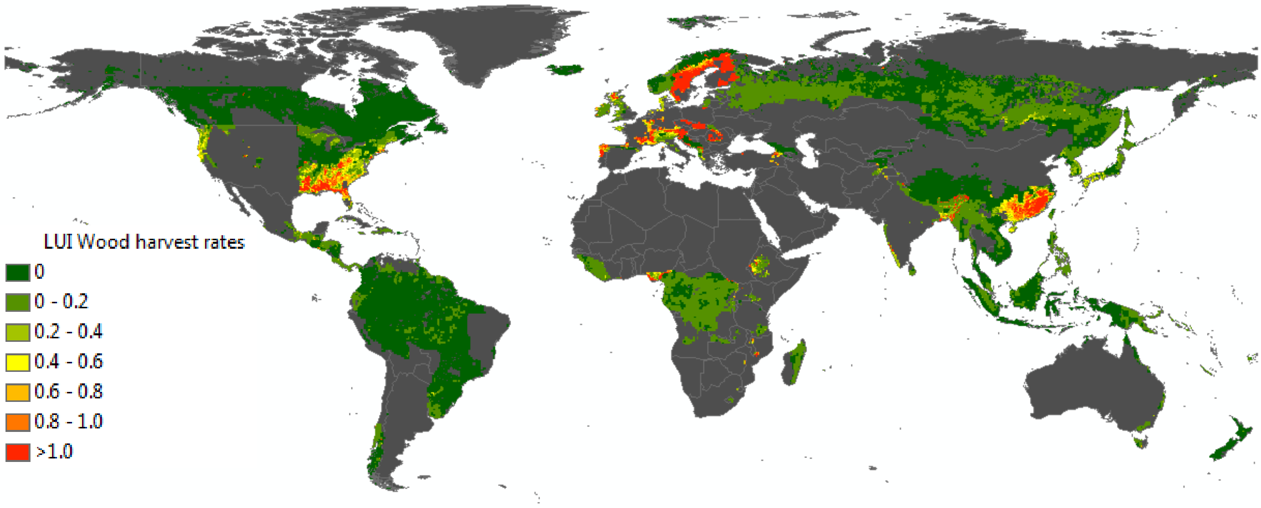


Figure 90: Intensity of wood harvest rates in secondary forests

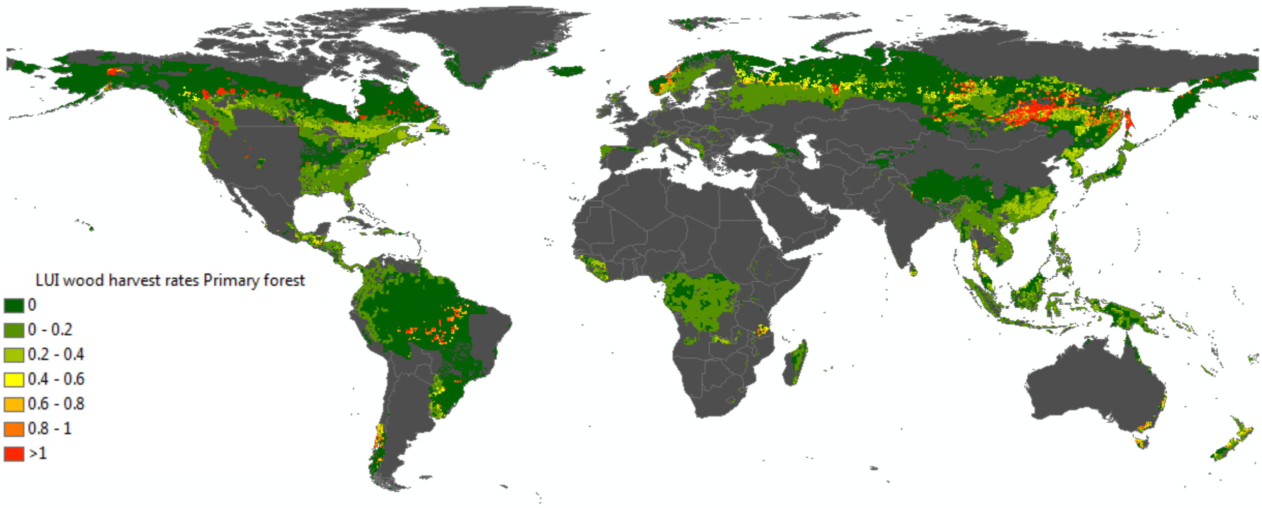


Figure 91: Intensity of wood harvest rates in primary forests

Management parameter deadwood volume

The data from Forest Europe [195] are used for the management parameter deadwood volume. It contains values for the deadwood volume of some European countries in m³ per hectare for the years 2000, 2005 and 2010. The average deadwood volume for 2010 is calculated for all countries. With the exception of Luxembourg, Norway and Portugal, where only older data were available (see

Table 39 in the Annex V). For the other countries, the average value of the worst intensity is assumed for the background data. As data are only available for European forests, no benchmarks could be calculated per global forest zone. However, this needs to be updated in the future as new data become available, as only forest areas with similar characteristics should be compared. The values are converted so that areas with a deadwood volume of 100 m^3 per ha have an intensity value of 0 and areas without deadwood have an intensity value of 1, such as Puletti et al. [305] suggest. For missing values, the average worst intensity is assumed, which corresponds to a deadwood volume of 20m^3 per ha and an intensity of 0.8.

Management parameter set-aside area

For the management parameters of set-aside land in forests, the proportion of protected areas per forest type and country is calculated. Therefore, the World Protected Area Database of UNEP-WCMC & IUCN [197] is used for the background database. Herein small set-aside areas are not covered. These should be analyzed if possible, in the foreground assessment. The database is filtered only for terrestrial protected areas and those falling under IUCN categories I to IV, as these categories mean that they are to some extent legally protected. The total area of primary and secondary forest that is protected is calculated. As a next step the total forest area per country and the proportion of protected primary and secondary forests per total forest area per country are calculated. Since the proportion of set-aside land in the forest does not depend on the characteristics of a global forest zone, no individual benchmarks are calculated. The resulting map with the proportions of set-aside land per forest area and land is scaled so that a proportion of 50% or more set-aside has the lowest intensity values (see Figure 92). 50% of set-aside land is assumed as the reference value, as Baillie & Zhang [306] suggest, that half of the world's natural forest area would need to be used to conserve all facets of biodiversity.

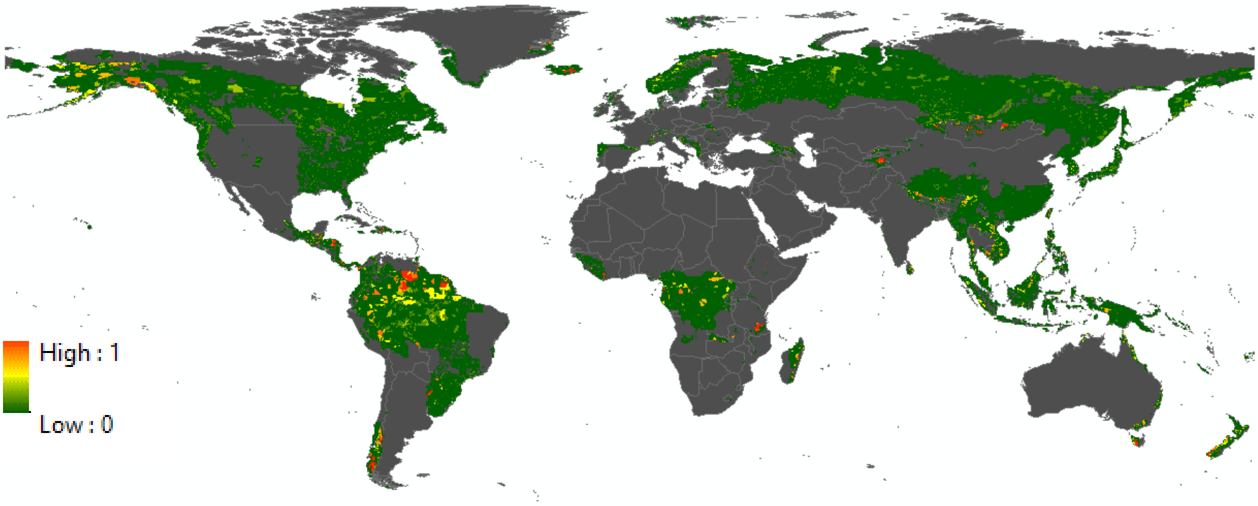


Figure 92: Share of protected set-aside land in forest areas

Land use intensity index forestry

The land use intensity index for the land use type forest and the sub types primary and secondary forest is calculated based on the following formula:

$$LUI_{Forest[i]} = \frac{WH[i]}{WH[MTI]} + \frac{BM[i]}{BM[MTI]} + \frac{TA[i]}{TA[MTI]} + \frac{DW[i]}{DW[MTI]} + \frac{SetAside[i]}{Totalarea[i]} \quad (32)$$

where

$LUI_{Forest[i]}$: Land Use Intensity index for forestry at location i

WH[i]/WH[MTI]: Intensity of wood harvest at location i

BM[i]/BM[MTI]: Density of wood biomass at location i

TA[i]/TA[MTI]: Tree age intensity at location i

DW[i]/DW[MTI]: Intensity of deadwood at location i

Set-aside/Total area: Share of set-aside area in total forest area at location i

The land use intensity index per grid cell for the background database is depicted in the next Figure 93 and Figure 94.

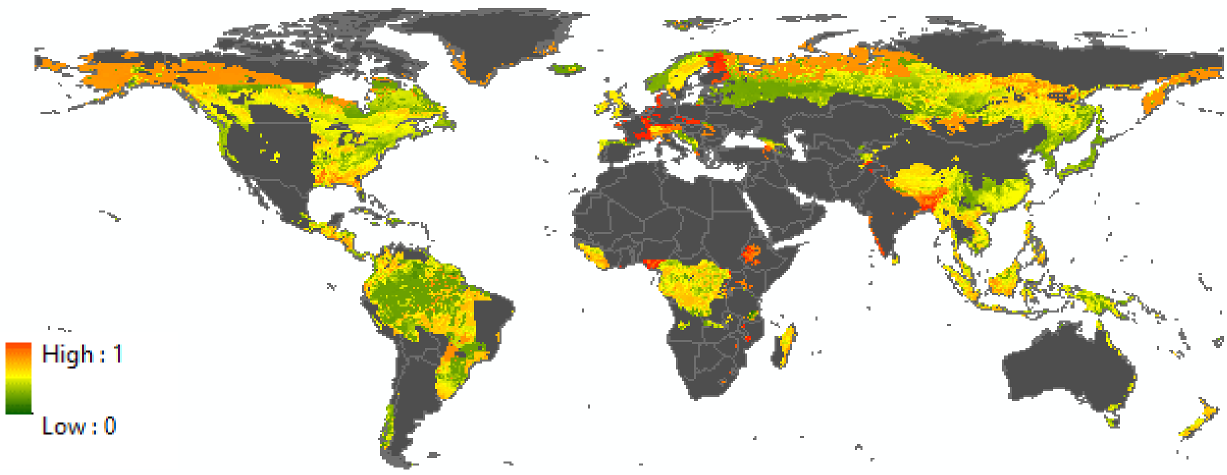


Figure 93: Land use intensity index for forests (primary and secondary)

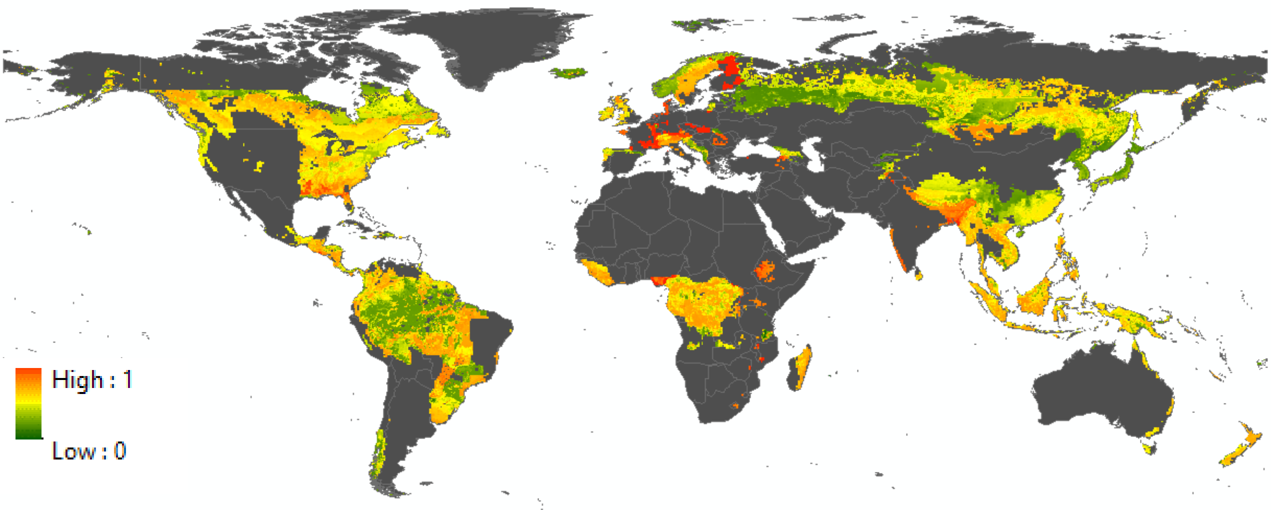


Figure 94: Land use intensity index for secondary forests

5.2.6 Overall results of land use intensity indices

The data distribution of the intensity values of land use per broad type of land use is shown in the next figure at a global scale. For the urban land use type, the highest proportion of values is in the average minimum intensity class, with another smaller peak in the average high intensity class. The majority of data points for the land use type pasture are in the lower intensity range. The land use class cropland and plantation also have two peaks, one in the lower intensity range and one in the higher intensity range. For plantations, on the other hand, the values of the higher range dominate in contrast to cropland where there are lower intensity values (see Figure 95). These results are comparable to the analysis of Chaudhary et al. [50] who investigated the proportion of the three intensity classes (minimal, light, intense) for several land use types. They found the highest proportion of forest area under intensive use (86%), for pasture land the highest proportion of area was under light use (90%), for cropland and plantations it was quite equally distributed within the three intensity intervals (cropland: minimal 22%, light 36%, intense 42%) and for urban areas the highest proportion of area was under minimal use (60%) [50].

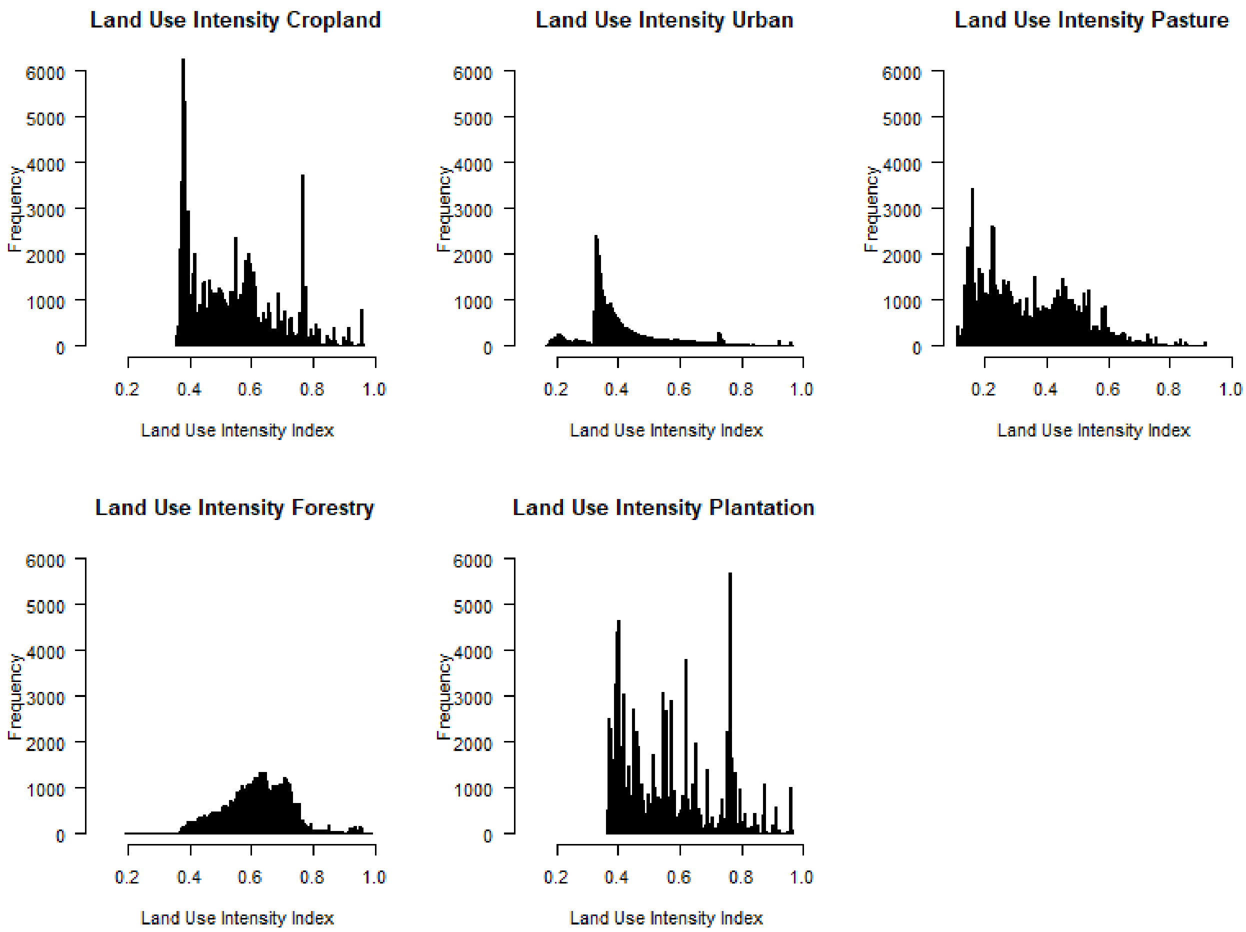


Figure 95: Data distribution of the land use intensity index values for the five land use types (frequency = pixel frequency of the raster data)

5.2.7 Translation of LUI into biodiversity risks

In this chapter, the land use intensity indices for each type of land use are translated into biodiversity risks using the biodiversity risk index intervals of the PREDICTS model [104,105]. The calculated LUI indices are translated into biodiversity risks by using an equation that fits a simple curve to three fixed points of average minimum, light and intensive land use intensity. Translating the LUI values into biodiversity risks makes it possible to calculate changes in biodiversity risks within the impact interval caused by changes in individual management parameters and thus in land use intensity.

5.2.7.1 Biodiversity risks cropland

The specific biodiversity risks for cropland are calculated on the basis of the land use-specific risk curve for biodiversity shown in Figure 96. By interpolating between the three fixed points of minimum (=0.17), light (=0.51) and intensive (=0.84) land use intensity and the associated risk values of biodiversity derived from Newbold et al. [104,105] from the PREDICTS model, we obtain an equation that enables the calculation of further land use intensity specific biodiversity risks. The formula for calculating biodiversity risks for each value of land use intensity of cropland is as follows:

$$BR\_locLUI_{Cropland} = -77.354*LUI^2 + 95.836*LUI + 5.9434 \tag{33}$$

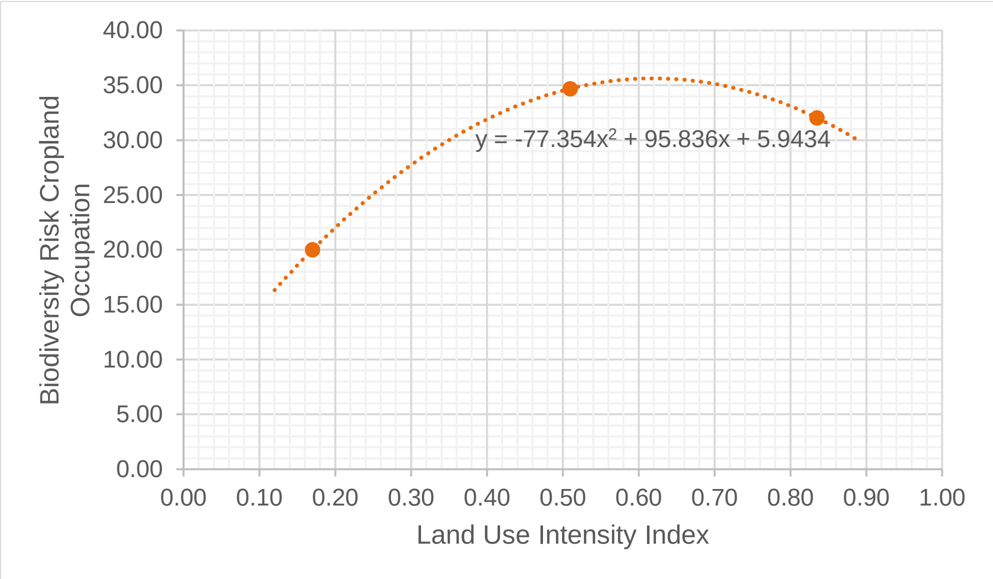


Figure 96: Interpolation based on a potential polynomial degree 2 relationship between LUI and biodiversity risks for cropland

By using a degree 2 polynomial fit, the higher land use intensity values get a lower biodiversity risk for cropland. This is based on the results of chapter 4.5, where the PREDICTS model shows in average lower biodiversity risk values for the highest land use intensity class of 0.8 than for the medium land use intensity class (0.5) for cropland. This could lead to an overestimation of biodiversity risks in areas with medium land use intensity and an underestimation of areas with a high land use intensity. Therefore, biodiversity risks for cropland are also calculated by assuming a degree 1 polynomial relationship between land use intensity and biodiversity risks (see Figure 97). Here the biodiversity risks for arable cropland are calculated using the following formula:

$$BR\_locLUI_{Cropland} = 18.288*LUI + 19.675 \quad (2)$$

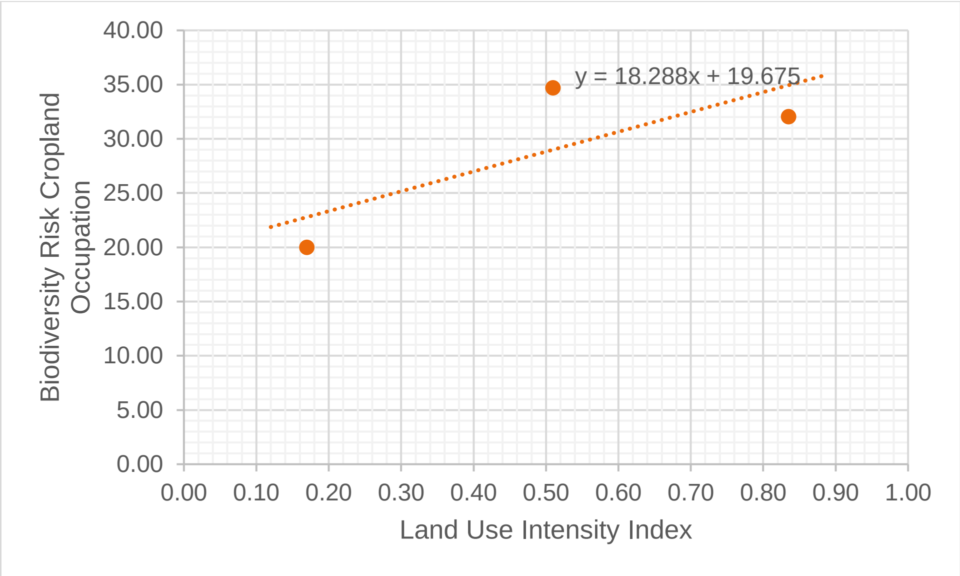


Figure 97: Interpolation based on a potential degree 1 polynomial relationship between LUI and biodiversity risks for cropland

The global biodiversity risks for the land use type cropland and the subtypes C3 annual and perennial, C4 annual and perennial as well as C3 nitrogen-fixing crops are presented in the following figure (assuming a degree 2 polynomial relationship). As can be seen in Figure 98, the risks in areas with an intermediate land use intensity seem to be overestimated. Therefore, a degree 1 polynomial relationship between LUI and biodiversity risks is assumed for the land use type cropland.

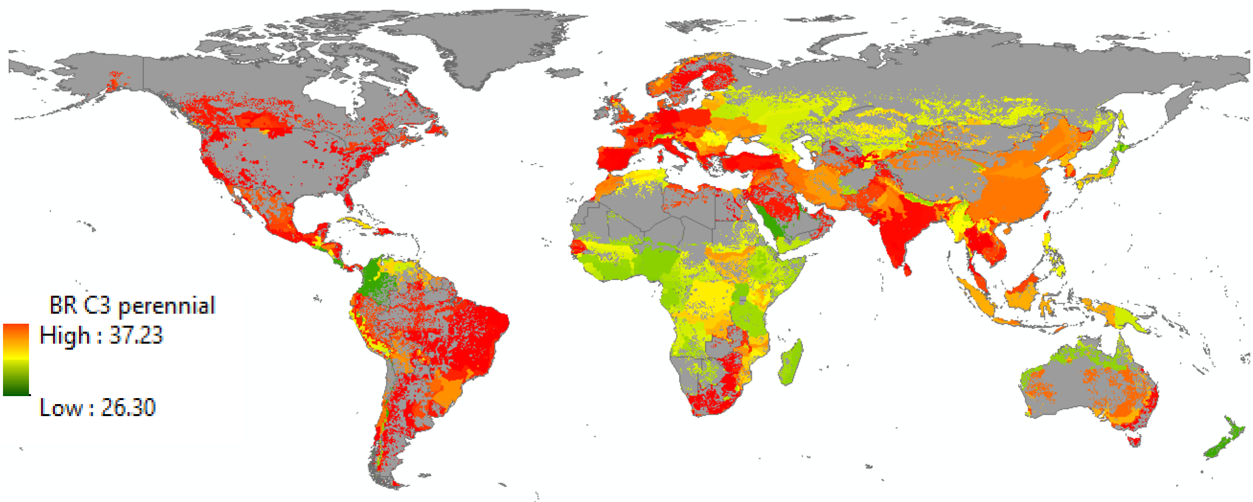


Figure 98: Biodiversity risks C3 perennial degree 2 polynomial fit

Figure 99 and Figure 100 illustrate the estimated biodiversity risks for the land use type cropland and its subtypes by assuming a polynomial of degree 1 relationship between land use intensity and biodiversity risks. As can be seen, this relationship shows a more realistic picture since the risks for the medium land use intensity seem less overestimated. Herein, the highest risks for biodiversity occur in the areas where we also have the highest land use intensity for C3 nitrogen-fixing crops such as in China, Great Britain or Colombia (see Figure 99).

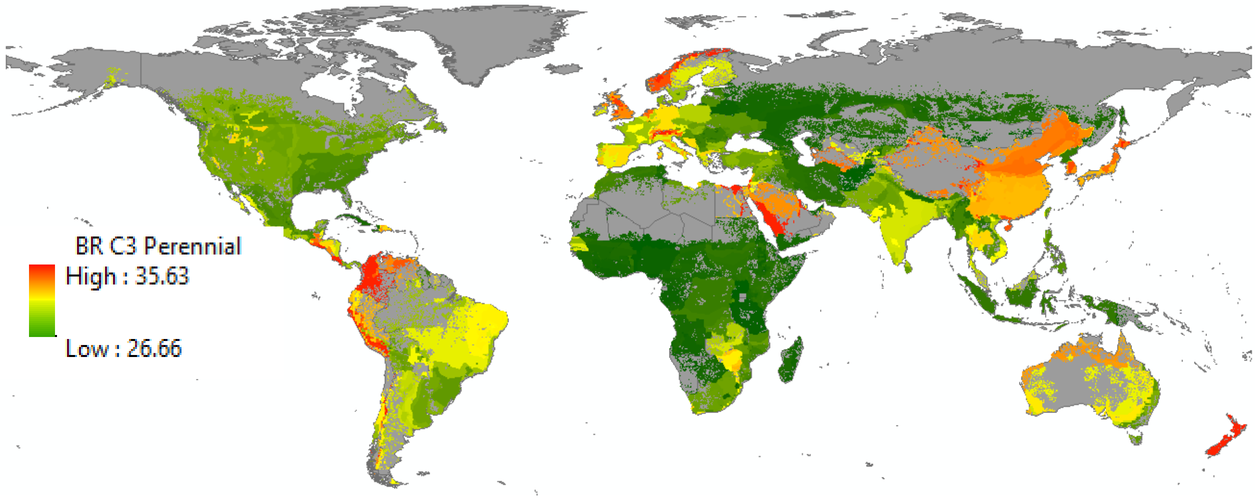


Figure 99: Biodiversity risks cropland – C3 nitrogen-fixing crops degree 1 polynomial

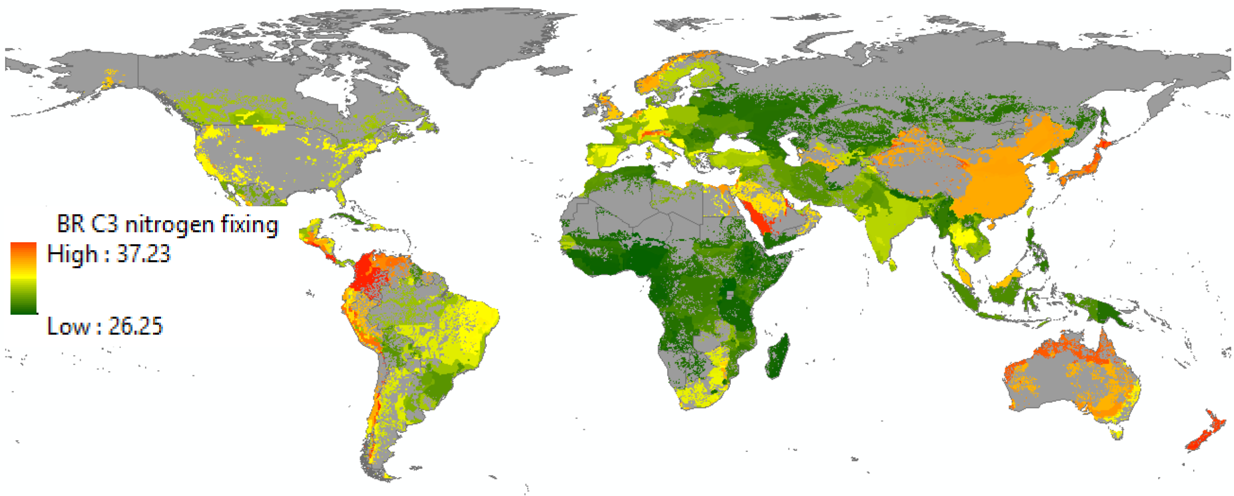


Figure 100: Biodiversity risks C3 perennial crop degree 1 polynomial

5.2.7.2 Biodiversity risks plantation

The biodiversity risks for plantations are calculated according to the equation:

$$BR\_locLUI_{Plantation} = -32.477*LUI^2 + 67.477*LUI - 4.6658 \quad (34)$$

For plantations, the land use intensity indices of C3 and C4 perennial plants are used, as they contain typical plantation plants such as coffee, cocoa or palm oil trees. The relationship between biodiversity risks for plantations and the land use intensity index is shown in Figure 101.

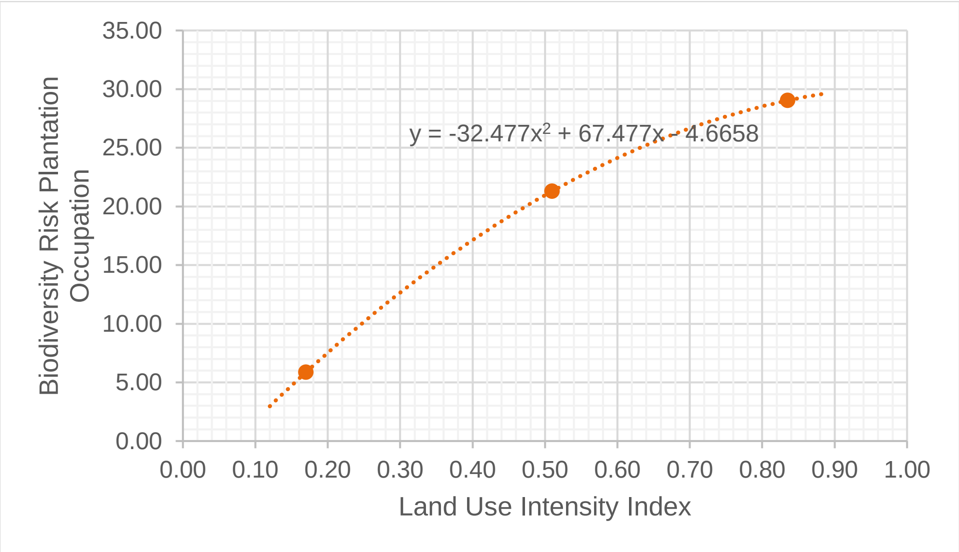


Figure 101: Interpolation between points based on a potential degree 2 polynomial relationship between LUI and biodiversity risks of plantations

The results indicate that the high LUI of plantations in Malaysia, Colombia, Venezuela, Australia or New Zealand, increases the risk for local biodiversity. A high intensity of land use in these countries could lead to a quality loss of biodiversity of about 30% compared to the biodiversity quality of the reference situation in these countries (see Figure 102).

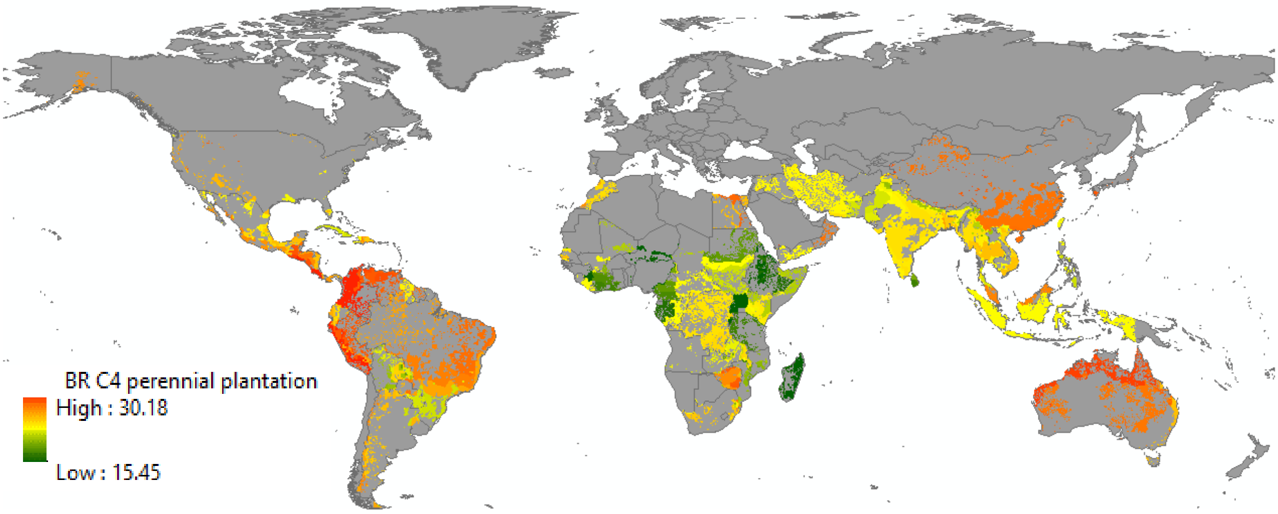


Figure 102: Biodiversity risk plantation C4 perennial

The biodiversity risks based on the LUI for plantations, assuming a polynomial of degree 1 relationship, are shown in Annex V. They are calculated according to this equation:

$$BR\_locLUI_{Plantation} = 34.918*LUI + 1.0996 \quad (35)$$

5.2.7.3 Biodiversity risks pasture

The relationship between land use intensity and biodiversity risks for the land use type pasture is shown in Figure 103. The biodiversity risks depending on the value of land use intensity can be calculated by assuming a polynomial of degree 2 relationship using the following equation. The degree 1 relationship is tested below:

$$BR\_locLUI_{Pasture} = -23.781*LUI2 + 53.524*LUI + 3.882 \quad (36)$$

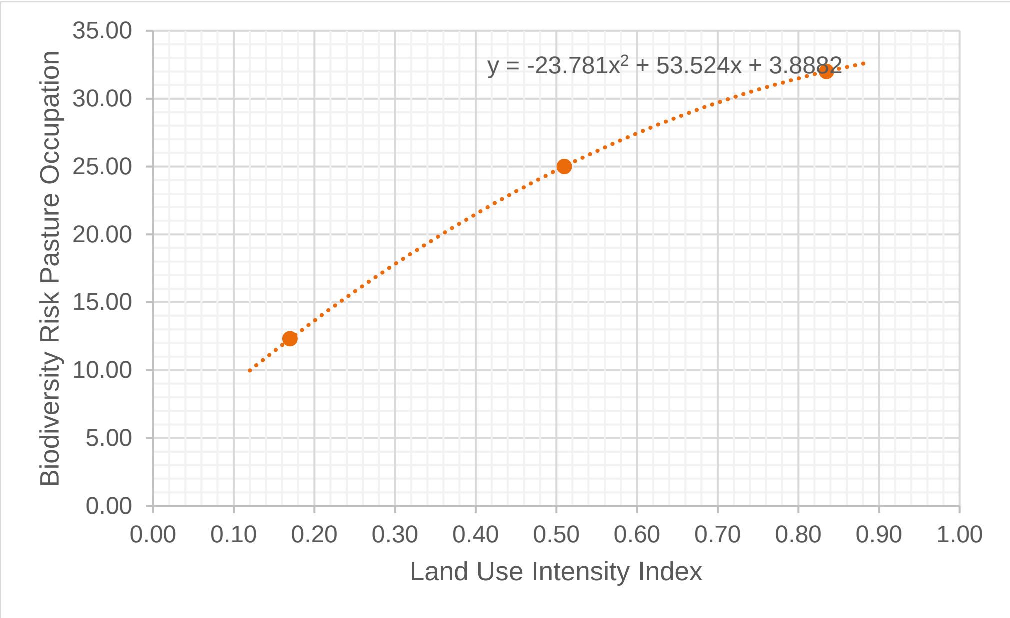


Figure 103: Interpolation between points based on a potential degree 2 polynomial relationship between LUI and biodiversity risks of pasture

$$BR\_locLUI_{Pasture} = 29.638*LUI + 8.1098 \quad (37)$$

To calculate the specific biodiversity risks for the land use type pasture for the background data, the map of the LUI index for pasture was used, and the index translated into biodiversity risk values for the land use type pasture and its subtypes, managed pasture and rangeland pasture, with R (Figure 104).

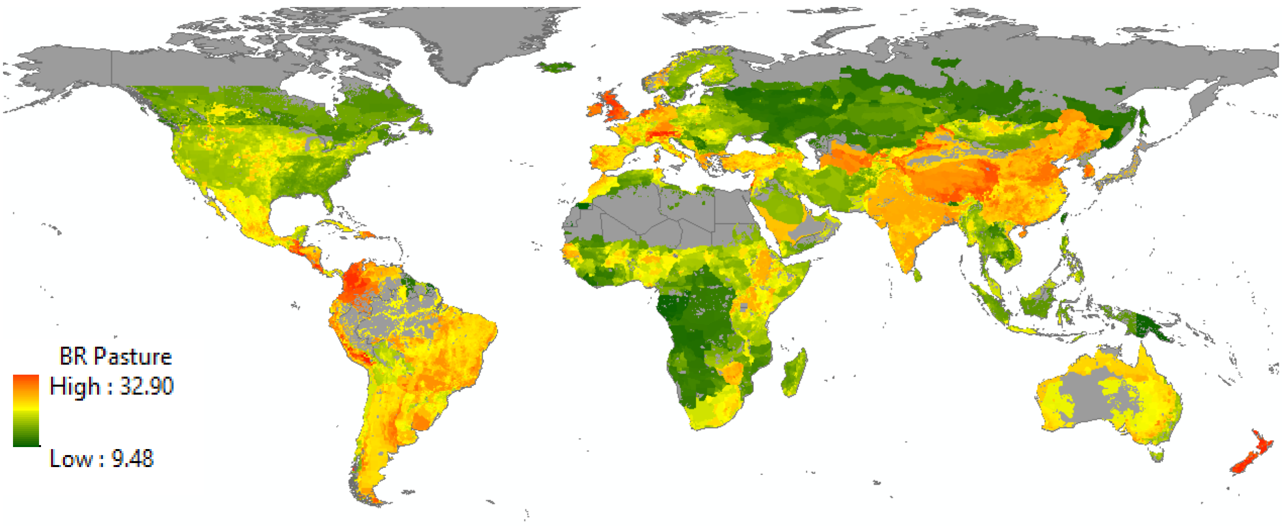


Figure 104: Biodiversity risk pasture

5.2.7.4 Biodiversity risks forestry

For secondary forests, the PREDICTS model provides only two intensity classes (first class: minimum intensity and second class: light and intense intensity). Thus, the average minimum intensity value is the same as for the other land use types = 0.17. The intensity value for the light/intense class, however, is calculated as the mean value between the light intensity value and the intense intensity value (Figure 105). Thus, we can only describe the relationship as a polynomial of degree 1 with the equation. In future, if more data is available it would be recommended to separate the light and intense intensity classes.

$$BR\_locLUI_{Forestry} = 12.396*LUI + 16.243 \quad (38)$$

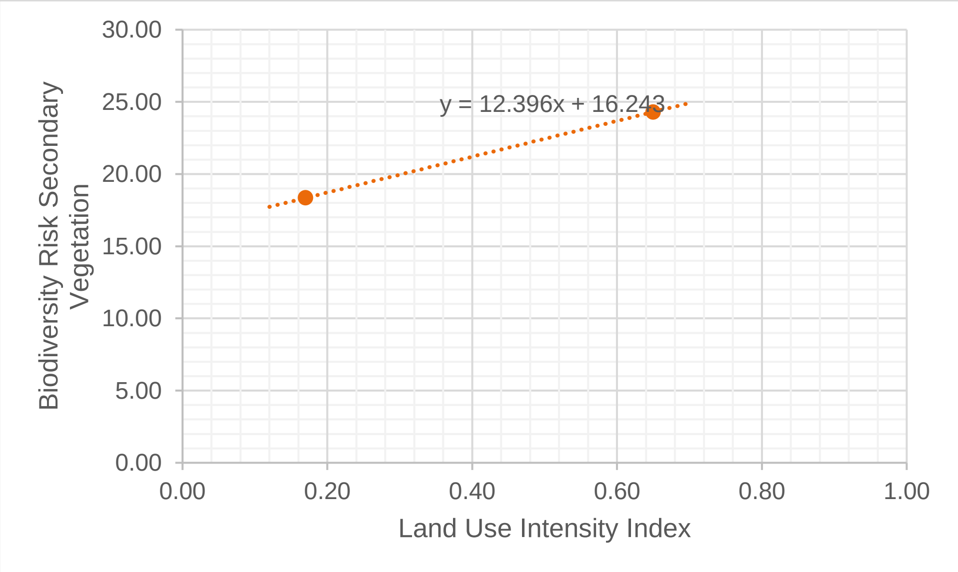


Figure 105: Interpolation between points based on a potential polynomial of degree 1 relationship between LUI and biodiversity risks of forest

The biodiversity risks for secondary forests are depicted in Figure 106. As can be seen, the biodiversity risks for forestry can reach in general lower impact values than the other land use classes, so that in some areas (highlighted in green) the difference in biodiversity quality of forest systems is only about 15% lower compared to the reference situation of primary forest. However, if we have intensive forest management the biodiversity quality difference can increase up to 30% (areas in red).

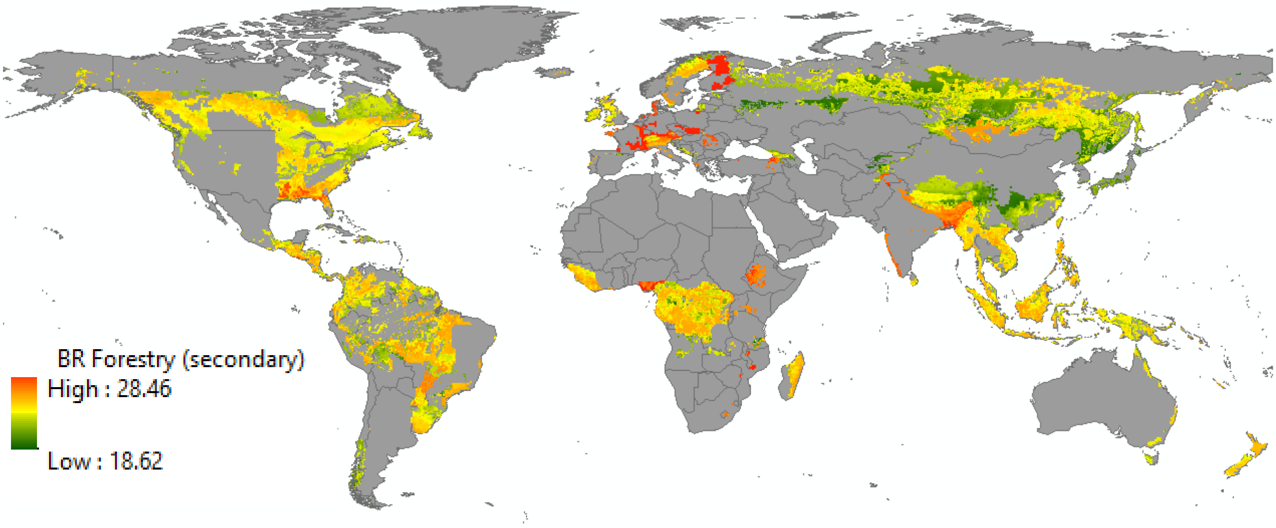


Figure 106: Biodiversity risks land use sub type secondary forests

5.2.7.5 Biodiversity risks urban

As shown, urban areas with intensive use can cause a drastic quality difference of more than 50%. However, urban areas also have the greatest potential for improvement, as the risk interval is the largest.

The following equation is used to convert the LUI index into urban biodiversity risks:

$$BR\_locLUI_{Urban} = -54.133*LUI^2 + 119.07*LUI - 14.51 \quad (39)$$

The degree 2 polynomial relationship between land use intensity and the associated biodiversity risk is shown in Figure 107.

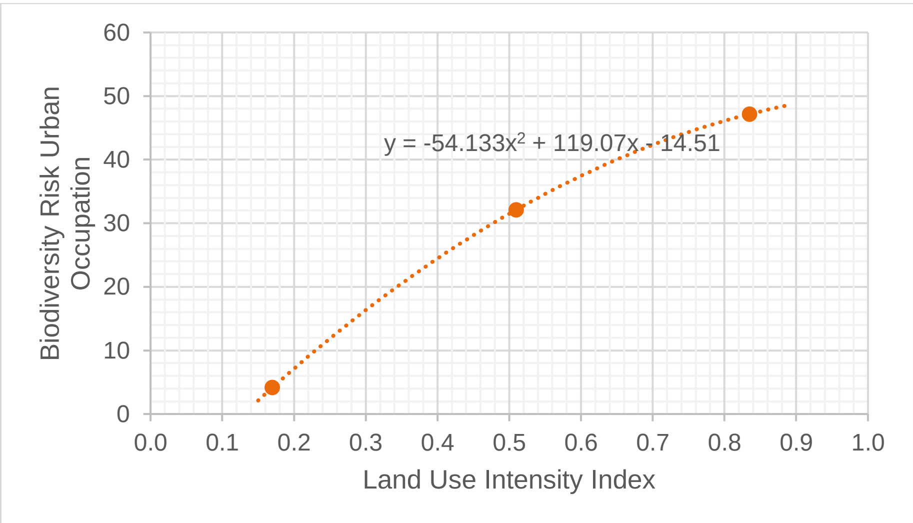


Figure 107: Interpolation between points based on a potential degree 2 polynomial relationship between LUI and biodiversity risks of cropland

The biodiversity risks for the polynomial of degree 1 relationship of urban intensity are calculated by using the following formula:

$$BR\_locLUI_{Urban} = 64.797 * LUI - 4.9002 \quad (2)$$

A global map for the background database is shown in Figure 108. The risks to urban biodiversity are quite high in dense urban centers in for example India, China, Japan, Egypt, some urban areas in the USA or urban areas in the Benelux. The intensity of land use is often caused by high sealing, a high population density or a high degree of light pollution. In the high intensity urban areas, a decline of biodiversity quality of about 50% is very likely.

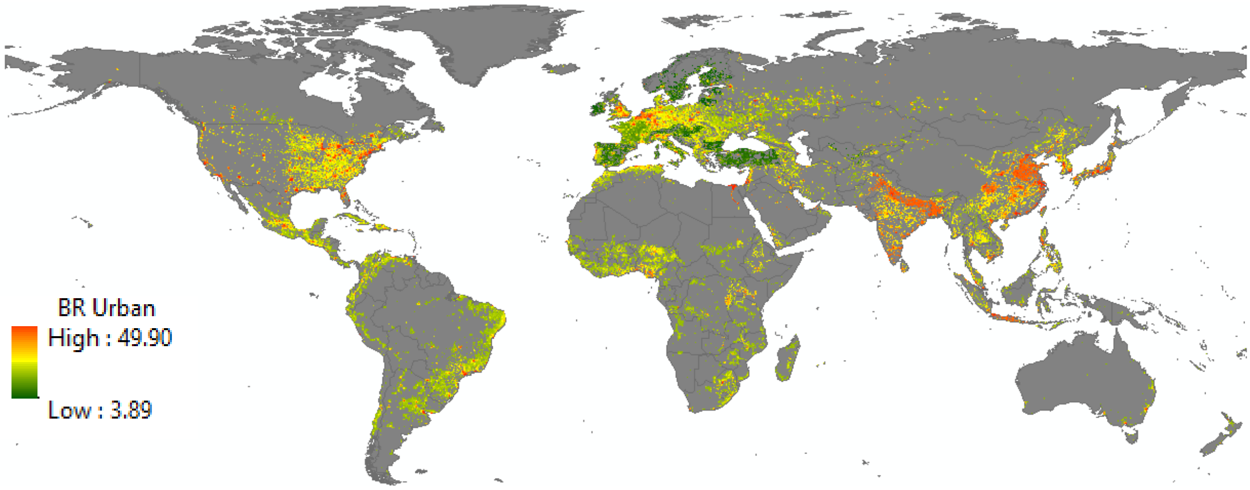


Figure 108: Biodiversity risks land use type urban

The overall range of biodiversity risks of the background database derived from the individual global land use intensity maps is depicted in Table 17 and Figure 109.

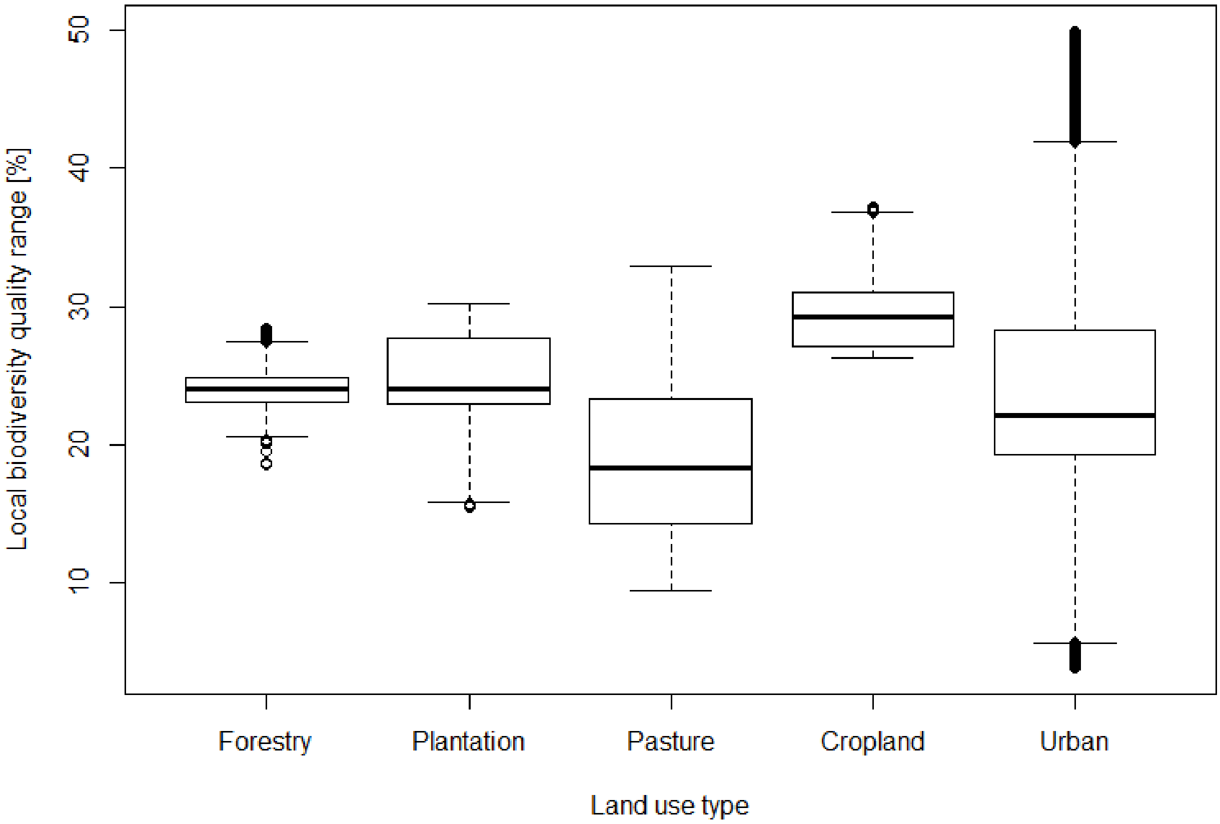


Figure 109: Biodiversity risks based on global LUIs (C4 perennial crops are shown for plantations)

Table 17: Biodiversity risks based on global LUIs

Data range	Urban	Cropland	Pasture	Forest	Plantation
Minimum	5.64	26.25	9.48	20.56	15.81
1. Quartile	19.25	27.09	14.36	23.08	23.00
Median	22.10	29.27	18.33	23.98	24.04
3. Quartile	28.33	31.01	23.38	24.84	27.73
Maximum	41.95	36.85	32.90	27.46	30.18

In line with the PREDICTS model we have the highest range of impacts in the urban land use class from about a quality loss of only 5% up to a loss of about 50% in high urbanized areas, whereas the median risks for urban biodiversity is relatively low (about 20% loss of quality). The highest global median risks are for the land use type cropland with a loss of biodiversity quality of almost 30% compared to the reference situation of primary vegetation under minimal use.

5.2.8 Sensitivity analysis of different relationships

Since the assumption of different relationships between biodiversity risks and LUI influences the results, a pixel wise standard deviation between the two approaches (polynomial of degree 1 and 2 relationship) of all biodiversity risk maps for each land use flow is calculated and shown in Table 18, except for the forest flows, where only a degree 1 relationship could be tested. The highest standard deviation applies to the land use type cropland, where the assumption of a polynomial of degree 1 relationship versus a degree 2 relationship could lead to a median standard deviation of about 3% of quality differences. The smallest standard deviation results for the land use flow pasture where the degree 2 polynomial relationship almost resembles a degree 1 relationship.

Table 18: Standard deviation biodiversity risks due to different relationships

Data range	Plantation	Pasture	Cropland	Urban
Minimum	0.03	0.00	1.95	0.55
1. Quartile	0.77	0.40	3.11	1.70
median	1.47	0.82	3.44	2.00
3. Quartile	1.61	1.16	3.88	2.46
Maximum	2.64	1.54	5.04	3.61

By comparing the standard deviation of the results of the two assumed relationships between the land use intensity and local biodiversity risks, a spatial distribution map of the standard deviations was generated for each broad land use type. As depicted in Figure 110 for all land use types most of the world is shown in dark green color. This implies that there is no great difference in local biodiversity risks between both relationships tested (standard deviation between 0 and 2%). There are some regions showing a yellow color and a limited number of areas showing an orange color, especially for cropland. In these regions there is a difference in the results between the degree 1 and 2 polynomial decay functions of about 6 to 8%. The highest standard deviation is shown for cropland in areas of Colombia.

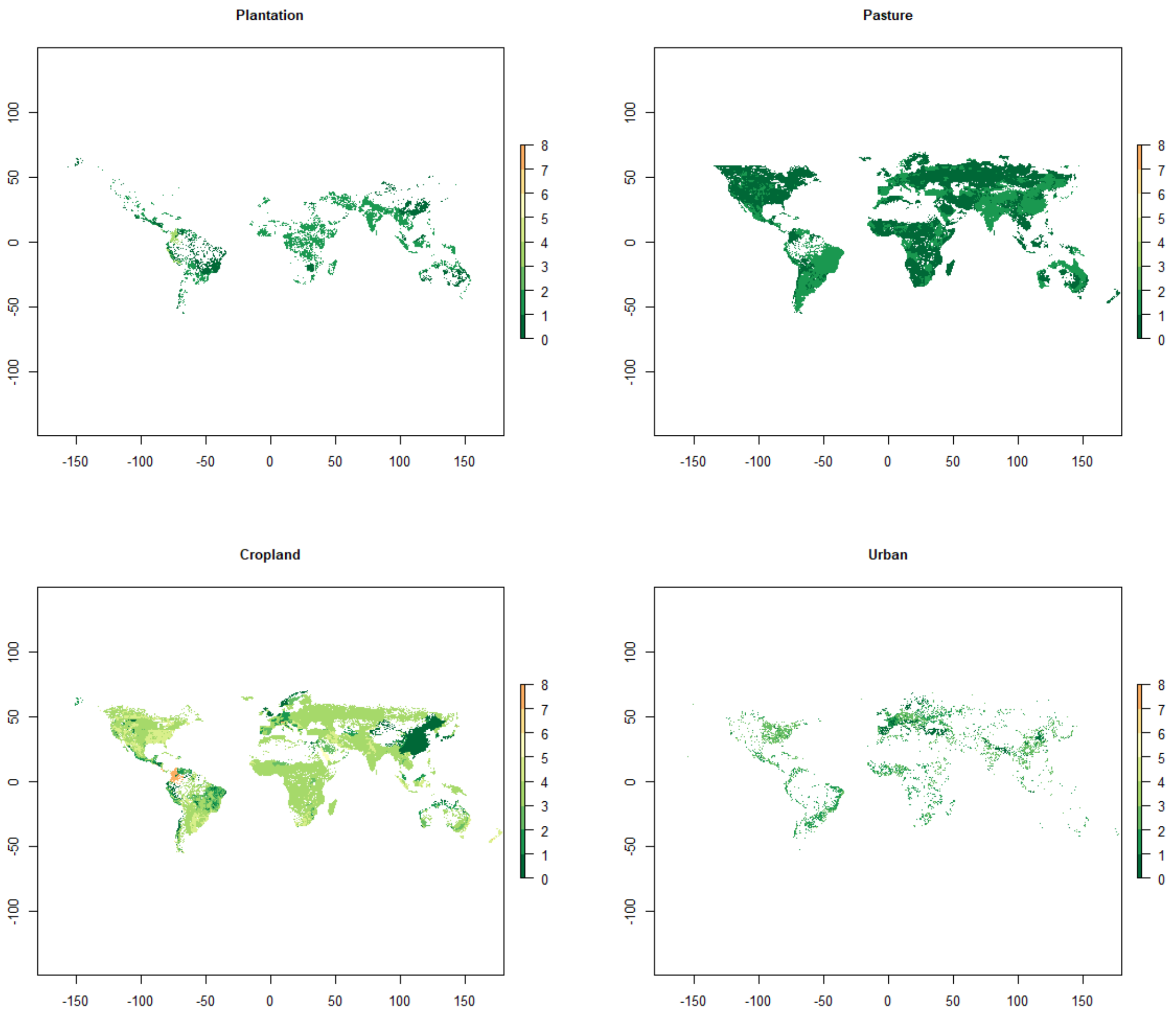


Figure 110: Spatial distribution of standard deviation of local biodiversity risks

5.3 Regional scale

In order to analyze the effects of land use at the landscape level, in this subchapter the LUI indices (and their resulting biodiversity risks) obtained for each type of land use are used to calculate the landscape development intensity index (LDI).

5.3.1 Landscape scale management parameters

As has been analyzed from the Conservation Evidence Database, there are some management parameters that are applicable in a landscape context. These management activities are taken into account in the calculation of the landscape development index. The management parameter set-aside land in a landscape context is always related to a different land use type since it takes place outside the patch or field. This can be the conservation and set-aside of primary habitat (land use type primary vegetation) or the creation, restoration and set-aside of secondary habitats (classified as land use type secondary vegetation) (see

Figure 111). In both cases terrestrial habitat as well as aquatic habitats can be set-aside. Since the management activities differ in the management of terrestrial or aquatic habitats both sub-parameters are described separately.

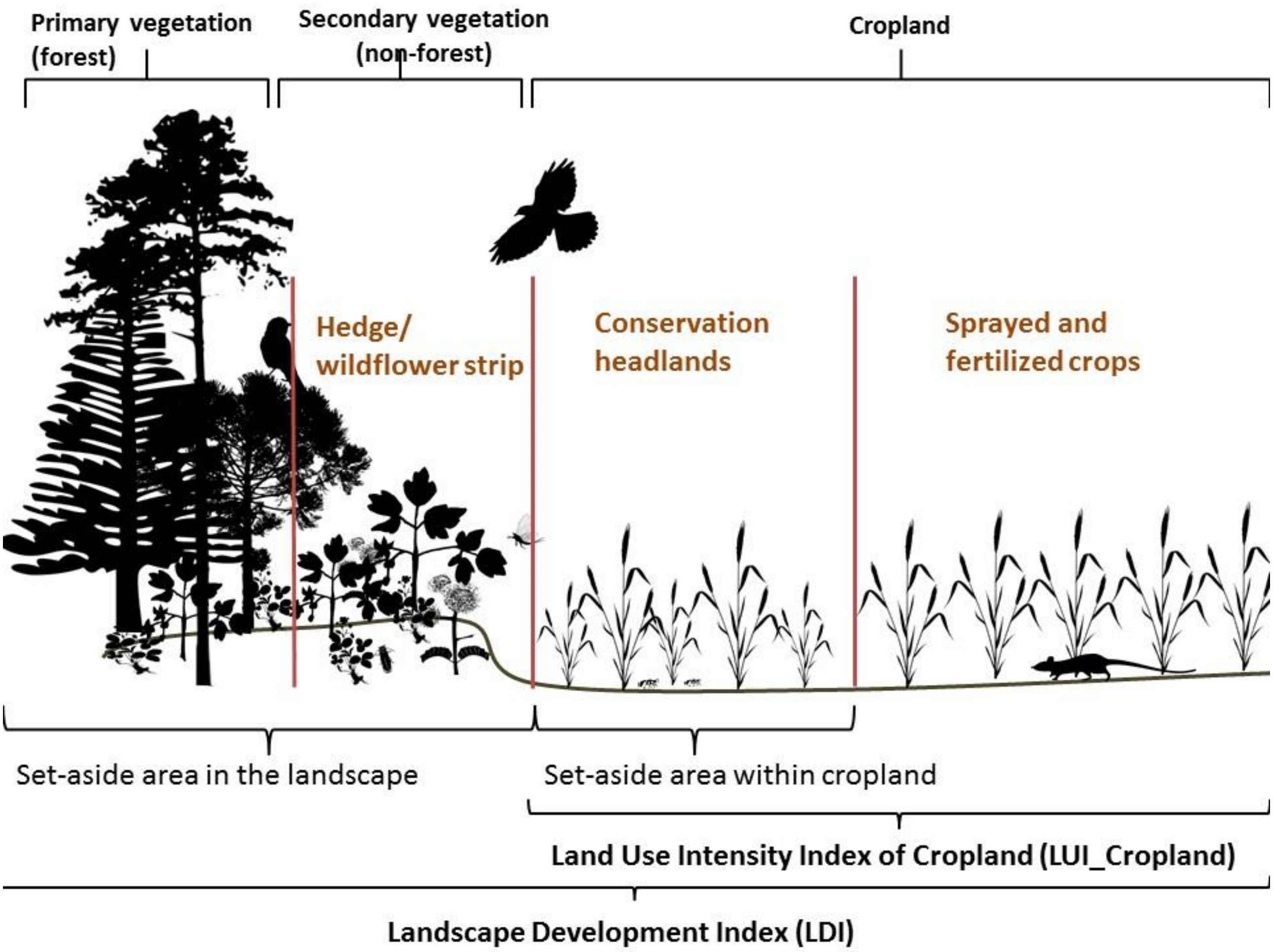


Figure 111: Set-aside area within the field and in a landscape context (adapted after Cooke [307])

a. **Management parameter set-aside area in the landscape (connectivity, corridor buffer zones)**

The creation of set-aside land in a landscape context has been found to be very beneficial to biodiversity. This management parameter has been classified as beneficial and highly effective for the conservation of biodiversity [198]. This intervention makes it possible to regenerate the field edge vegetation in a natural way (secondary vegetation) or to retain the natural primary vegetation. Set-aside areas preserve populations of endangered species if they are of high ecological quality and sufficiently distributed for example in an agricultural landscape [308]. Landscapes with high structural heterogeneity such as forest margins, hedge rows, ecotones or flower strips offer an improvement in biodiversity and an abundance and variety of beneficial organisms. The effectiveness of set-aside land was rated as very high (100%) [198]. The destruction of marginal habitats between different patches therefore has a negative impact on biodiversity. Set-aside land in a landscape is always classified either as the land use type primary vegetation, e.g. if parts of a natural forest or natural grassland are set-aside, or the land use type secondary vegetation, e.g. if a buffer strip of wildflower mixtures or hedges are planted/restored and set-aside.

b. Management parameter set-aside areas/aquatic habitats

Land which might be set-aside for biodiversity can be both a terrestrial as well an aquatic habitat, depending on the type of area that is set-aside. Many aquatic habitats, such as ponds, have been lost due to the conversion of land for agriculture and the intensification of agriculture. The creation of additional breeding sites for aquatic and semi-aquatic taxa, such as amphibians, can help to partially replace them and thus contribute to the protection of their populations [198]. Pond types that are beneficial for certain species can be created. Further aquatic habitats can be created by managing ditches. In addition, the drying of amphibious breeding sites prior to the development of terrestrial life stages can have significant negative impacts on populations. Sometimes it can be useful to sustain the water level until the terrestrial life stage is reached by using a local supply of water or by diverting water from outside. Temporary drying out of breeding sites can increase diversity, as it can help control predators, alien species or more dominant species [198]. As a natural resource, wetlands and water areas are therefore important for humans as well as for biodiversity. Since wetlands are rich in plant and animal species, the conservation, restoration and provision of wetlands and water surfaces is a high priority [309]. Wetlands represent a different type of land use. The quality value for wetlands corresponds to the impact interval of the primary vegetation non-forest (for natural wetlands) or the secondary vegetation non-forest (for restored or newly created natural aquatic habitats) of the PREDICTS model.

i. **Driver of biodiversity loss:** Habitat change

ii. **Related management activities in the Conservation Evidence Database [198]:**

- *Restore habitat connectivity,*
- *replant vegetation,*
- *create/protect habitat corridors,*
- *restore or create traditional water meadows,*
- *create scrapes and pools in wetlands and wet grasslands,*
- *regulate water levels (maintain pond water levels),*
- *restore or create wetlands and marine habitats (coastal and intertidal wetlands),*
- *maintain upland heath/moorland,*

- *maintain traditional water meadows,*
- *manage ditches to benefit wildlife, or*
- *create ponds.*

iii. **Indicator (relief):** Share of primary and secondary vegetation (minimum use) within the landscape [%] as part of the LDI.

iv. **Global data set:** A global landscape development index is calculated in this thesis per grid cell and as an average per country that can be used for the background database (chapter 5.3).

5.3.2 Calculation of biodiversity risks at regional scale

The land use model of Hurtt et al. [186] provides information on the share of different land use types within a grid cell. Herein, also information on the share of primary and secondary vegetation is available. Primary and secondary vegetation areas under very low human influence can be regarded as set-aside land in the landscape for the background system. The information on the intensity of each land use type is taken from the different land use intensity indices. Herein, the share of each land use type per grid cell is determined and multiplied with the associated risks due to the land use intensity index of each share of land use type. This results in landscape specific biodiversity risk values per grid cell. At a country level the total area of each country is calculated. As a next step the total area of each land use type is calculated in absolute numbers and as a percentage of the country area. Furthermore, the average land use intensity per country is calculated resulting in an average biodiversity risk value at the landscape level per country and land use type. For the land use type primary and secondary vegetation, non forest, the lowest land use intensity (0.17) is assumed, since scientists state there are no landscapes left really "untouched" by humans [310]. Set-aside areas or protected areas are classified as primary or secondary vegetation (forest or non-forest) under minimum land use intensity. The LDI is calculated using the information of the share of each land use type as well as the biodiversity risks of each land use type per grid cell of the following land use flows and subflows: C3 and C4 annual crops, C3 and C4 perennial crops, C3 nitrogen-fixing crops, managed pasture and rangeland pasture, secondary and primary forest and non-forest, as well as urban areas. Since we want to calculate the LDI in landscapes that are relevant in a product context, all bare areas and deserts where no land use activity is taking place at all are excluded. The regional biodiversity risk is calculated as follows:

$$BR\_regLDI_{total} = \sum \%\ LU_{LandUseType_i} * BR\_locLUI_{LandUseType_i} \qquad (40)$$

where

BR_regLDI_{total}: Biodiversity risk at landscape level

$LU_{LandUseType_i}$: Percent of specific land use type in the total area of influence at location i

$BR\_locLUI_{LandUseType_i}$: Biodiversity risk for land use type depending on the land use intensity at location i

5.3.3 Results of biodiversity risk due to landscape development index

The biodiversity risk at the landscape level for the background database is depicted in Figure 112 per grid cell. The figure shows a lower biodiversity risk where the landscape is either composed of a higher share of primary and secondary vegetation as is the case with Canada and the tropics or where the landscape has a higher share of land use under a lower land use intensity such as in regions in sub-Saharan Africa. The highest impact on biodiversity is for those landscapes that are composed of almost 100% of urban areas and that are under a very high land use intensity. For each land use type, a specific LDI is calculated by considering only the land areas in which the individual land use is embedded. One example is given in Figure 113 for the LDI of areas of the land use type managed pasture.

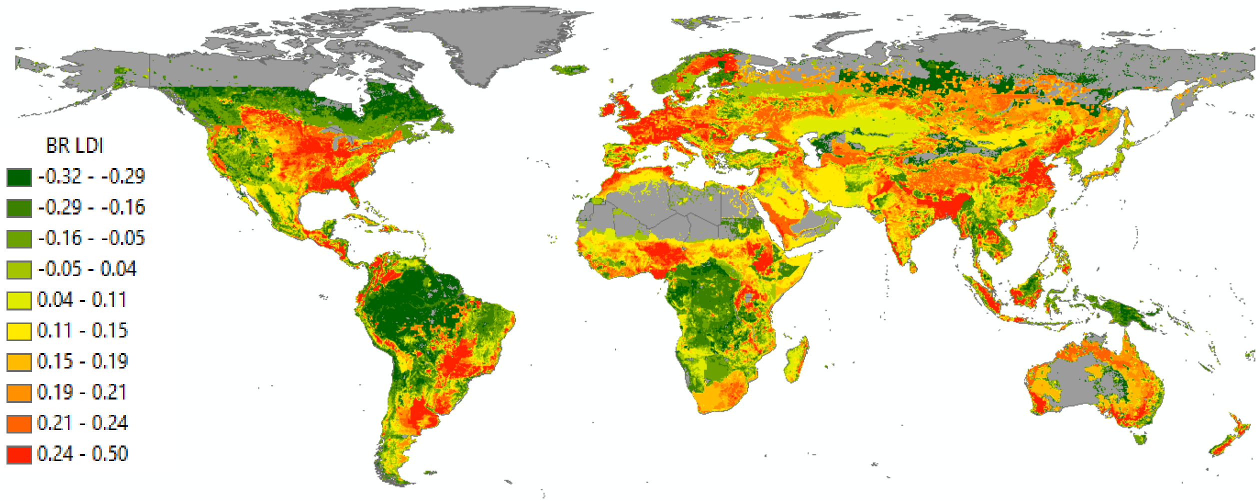


Figure 112: Biodiversity risks at the landscape level for all land use types per grid cell

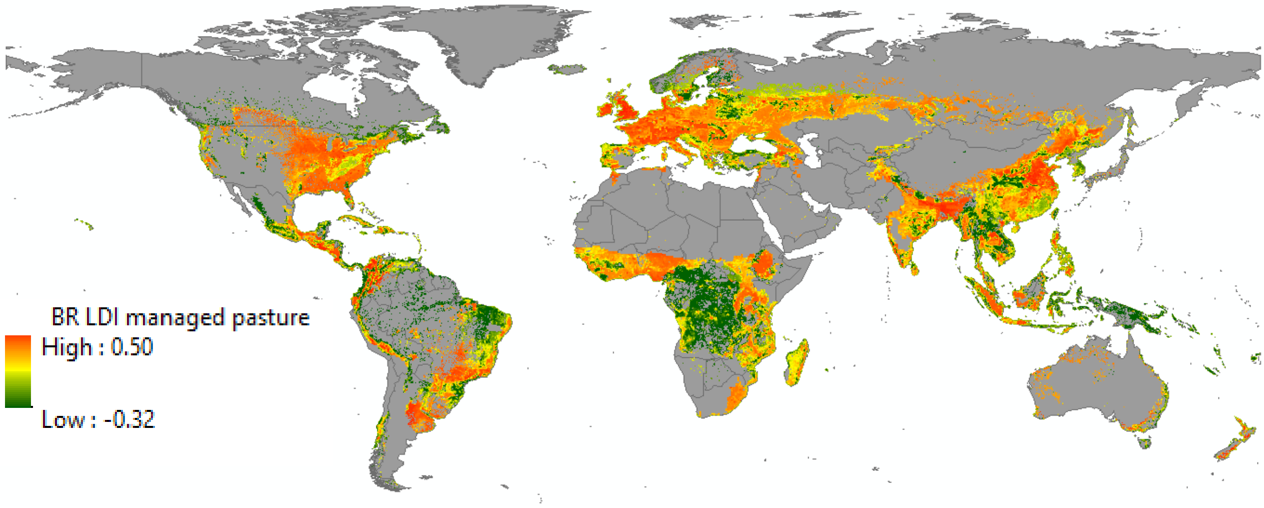


Figure 113: Biodiversity risks at the landscape level for areas with managed pasture and their landscapes

6 Application of the BioMAPS method

In order to apply the method, developed in the course of this thesis, there are five subsequent steps for the LCA modeler.

First, all land use processes that are involved in all life cycle phases of the product are analyzed and their geographic location is determined in the inventory phase. The identified land use processes are mapped according to the matching land use flows of this method. With respect to the regionalization of the land use types and the biodiversity risk areas, if the exact location or region of the land use process is available, the biodiversity risk map can be used and the ratio of the biodiversity risk areas to the land use area is calculated directly from the foreground map. If there is no specific information on the region available then the background data with average values for the probability that the land use process takes place in a biodiversity risk area is taken.

Second, the land use type specific impact on biodiversity is analyzed. If primary data on land management parameters or the land use intensity is available, one can go directly to the third step. If there is no further information other than the land use type (regardless of the intensity), we can use the calculated average impact on biodiversity for each land use type involved in the life cycle of the product.

Third, for the assessment of the impacts of the land use intensity or the management parameters on biodiversity, the average values of the respective land use intensity indices translated into biodiversity risks per country is taken. If there is further information on specific quantities for land management parameters, the land use intensity index is calculated in a foreground process depending on the benchmark value of the agro-ecological zone. The specific land use intensity index is then translated into biodiversity risks, using the formula provided by the biodiversity quality risk curve for the respective land use type.

Fourth, the landscape scale is assessed either by using country average data or the grid cell data (background data), or by calculating the landscape intensity index for the specific location in a foreground process as the share of each land use type and intensity in the surrounding landscape and its associated biodiversity risks.

As a final step, the value for the total biodiversity impact can be calculated using the following formula:

$$PBR = (BR\_globe + BR\_locLUI/100 + BR\_regLDI) * a * t \quad (41)$$

where

PBR: Potential biodiversity risk

BR_globe: Global biodiversity risk factor

BR_locLUI: Local biodiversity risk factor depending on land use type and intensity

BR_regLDI: Regional biodiversity risk factor depending on landscape composition and intensities of patches

t: Time of occupation of specific land use type

a: Area of occupation or transformation of specific land use type

The flow chart of the steps of the method is depicted in Figure 114:

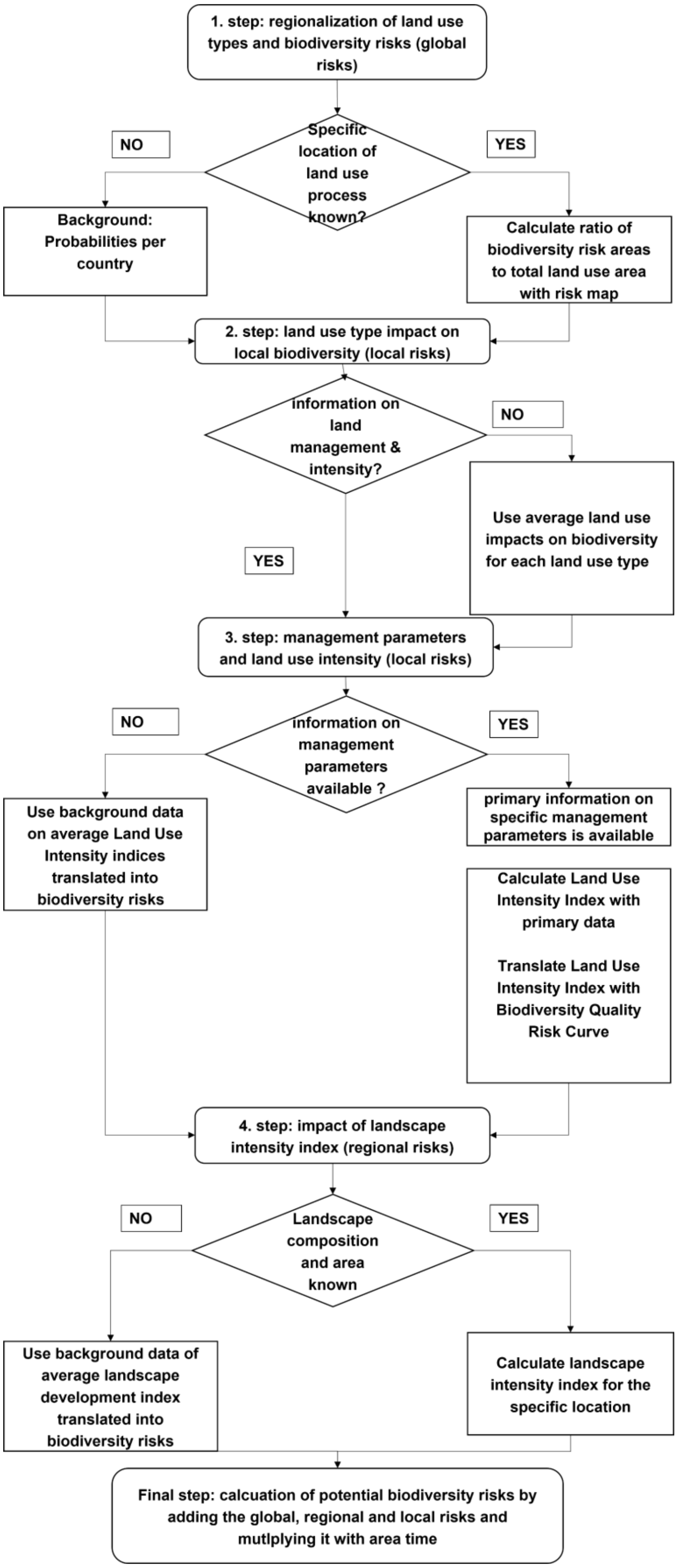


Figure 114: Analytical steps of the method

6.1 Preliminary work

The herein presented case study compares the biodiversity impacts of different renewable energy transport systems. The biodiversity impact analysis is based on the preliminary work of Uusitalo et al. [311] who analyze the land use requirements for different energy transport systems. These include also biofuels such as diesel made from palm oil from Malaysia, ethanol made from sugarcane from Brazil, methane made from pyrolysis (using wood) from Finland as well as methane made from anaerobic digestion from corn from Germany [311] (see Table 19).

Table 19: Biofuels energy carriers for transportation based on [311]

Product	Feedstock and production method	Land use type	Land use subtype	Land use intensity class	Location of production
Diesel	Palm oil	Plantation	C3 perennial crop	Intense	Malaysia
Ethanol	Sugarcane	Cropland	C4 perennial crop	Intense	Brazil
Methane	Pyrolysis from wood	Forestry	Secondary forest	Intense	Finland
Methane	Anaerobic digestion from corn	Cropland	C4 annual crop	Intense	Germany

6.2 Goal and scope

The goal of this case study is to analyze the biodiversity impact of different energy carriers for transportation with respect to their land use for occupation. For reasons of comparability, the same functional unit as in Uusitalo et al. [311] is used which is equal to an average of 18,600 km a^{-1} driven by a passenger car in the European Union. Herein, for each energy type (diesel, electricity, gas, ethanol) the average consumption values are taken as suggested in Uusitalo et al. [311]. An attributional LCA modelling approach is chosen, in order to be consistent with the study of Uusitalo et al. [311].

6.3 Inventory analysis

For the inventory analysis the land use flows identified by Uusitalo et al. [311] are matched to the land use flows of this method. The geographic location of land use as well as the area requirement for occupation is based on the calculations of Uusitalo et al. [311]. The geographic analytical unit is defined as the country and the region where the land use takes place. Herein, transformation impacts are not part of the study.

The feedstock and production methods of the biofuels can be classified according to the following land use types: Diesel from palm oil of Malaysia is classified as land use flow plantation, C3 perennial crop, intense. Sugarcane from Brazil is classified as cropland, C4 perennial crop, intense. Wood for the pyrolysis of methane is classified as secondary vegetation, forests, intense in Finland. And methane from corn in Germany is classified as cropland, C4 annual crop, intensively grown as biofuel (see Table 19). All production plants are classified as land use type urban, where the intensity and management parameters depend on the location on site. The same goes for the non-biofuels, if there is no primary data on the management available. Herein, all of the pathways are mature and commercially available except for the wood gasification which merely represents a theoretical pathway to show the direct use of wood raw materials.

The model for the production of the energy systems is shown in Figure 115. The processes in the model that require land are the four production processes for corn, wood, palm fruits as well as sugar cane. As well as the processing plants to further process the interim products to methane, diesel and ethanol. Further land use is required for the transportation processes of the primary and secondary products in the form of streets or refueling stations (urban land use). These land use areas, however, are estimated to be negligible per functional unit and are therefore only assessed in the background database [311].

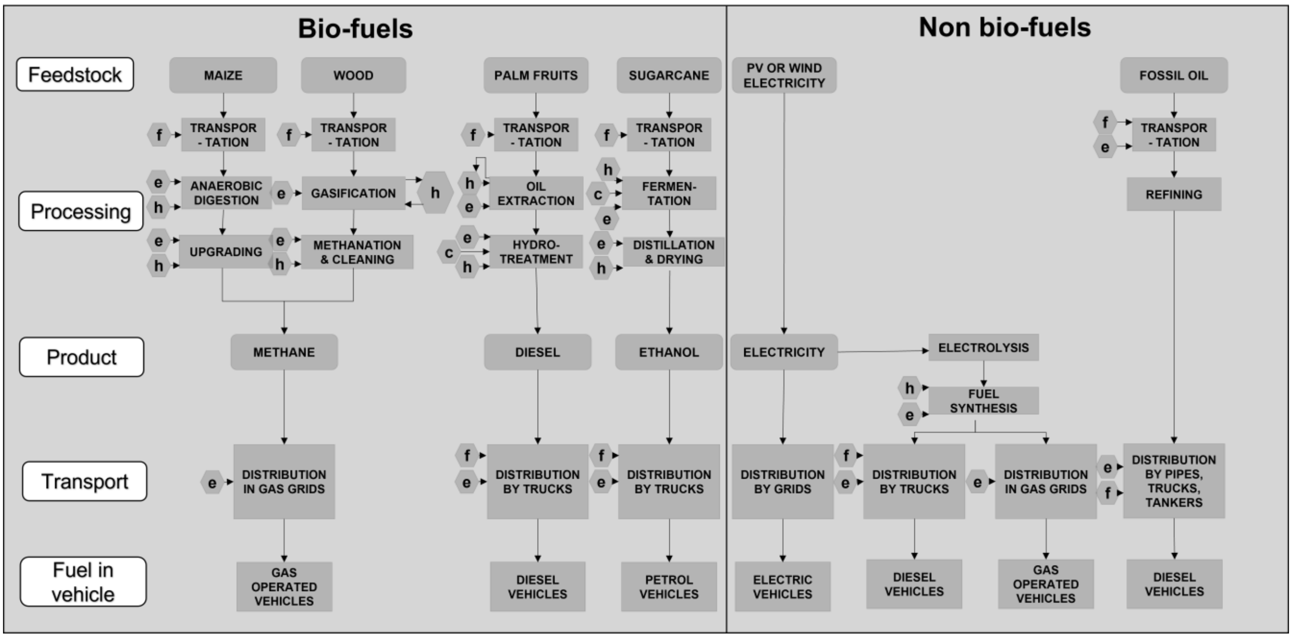


Figure 115: Model for energy carriers for transportation, adapted from Uusitalo et al. [311] (f = fuel, e = electricity, h = heat, c = chemical)

The area requirements per functional unit, the different land use flows for occupation, as well as their specific location (as country, region and coordinate) are depicted in Table 20 for the foreground calculations of the biofuels, based on Uusitalo et al. [311] and supplemented by own research for coordinates. For the non-biofuels only background information was available.

Table 20: Life Cycle Inventory for case study

Fuel type	Product	Feed-stock and produc-tion method	Land use type	Land use sub-type	Coun-try	Region	Longi-tude	Lati-tude	Area re-quire-ments [m²/FU]
Bio-gas wood	Methane from wood	Pyrolysis from wood	Forestry	Secondary forest	Finland	Southern Finland	26.845	61.403	27843.000
	Methane from wood	SNG plant	Urban		Finland	Southern Finland	25.645	60.917	0.018
Bio-diesel	Diesel from palm oil	Palm oil	Plantation	C3 peren-nial crop	Malaysia	Sabah	115.649	5.393	1719.060
	Diesel from palm oil	Extrac-tion	Urban		Malay-sia	Sabah	115.746	5.346	0.002
	Diesel from palm oil	Crude palm oil HVO process	Urban		Nether-lands	South Holland	4.034	51.968	0.071
Bio-me-thane corn	Methane from corn	Anaerobic digestion from corn	Cropland	C4 annual crop	Germany	Saxony	12.777	51.325	3554.000
	Methane from corn	Biogas plant	Urban		Ger-many	Saxony	12.381	51.328	0.280
Bio-ethanol	Ethanol from sugarcane	Sugarcane	Cropland	C4 perennial crop	Brazil	São Paulo	-48.006	-21.196	2442.000
	Ethanol from sugarcane	Ethanol pro-duction	Urban		Brazil	São Paulo	-47.800	-21.200	1.024

6.4 Biodiversity impact assessment

For the biodiversity impact assessment, information on the three different spatial scales is provided which can be aggregated into one single biodiversity impact value. Values are provided from both the background database using country average values and the foreground database using the values of the coordinates or regions and some primary data values, where available. For the assessment using the background database all fuel types that are included in the study of Uusitalo et al. [311] are taken into account. Herein, the GaBi inventory of Uusitalo et al. [311] is directly coupled to the impact model of this method by using the land use flow list provided in the Annex VI. The application of the method in a foreground system is shown by taking a closer look at the following biofuels: 1) methane from wood production, 2) methane from corn silage produced in Germany, 3) ethanol from sugar cane produced in Brazil, and 4) diesel from palm oil cultivated in Malaysia. Here, some foreground information (including more accurate area requirement calculations for occupation) is available and, as can be seen in Figure 116, theses bio-fuels show a higher biodiversity impact than all non-bio fuels or the electricity grid mix.

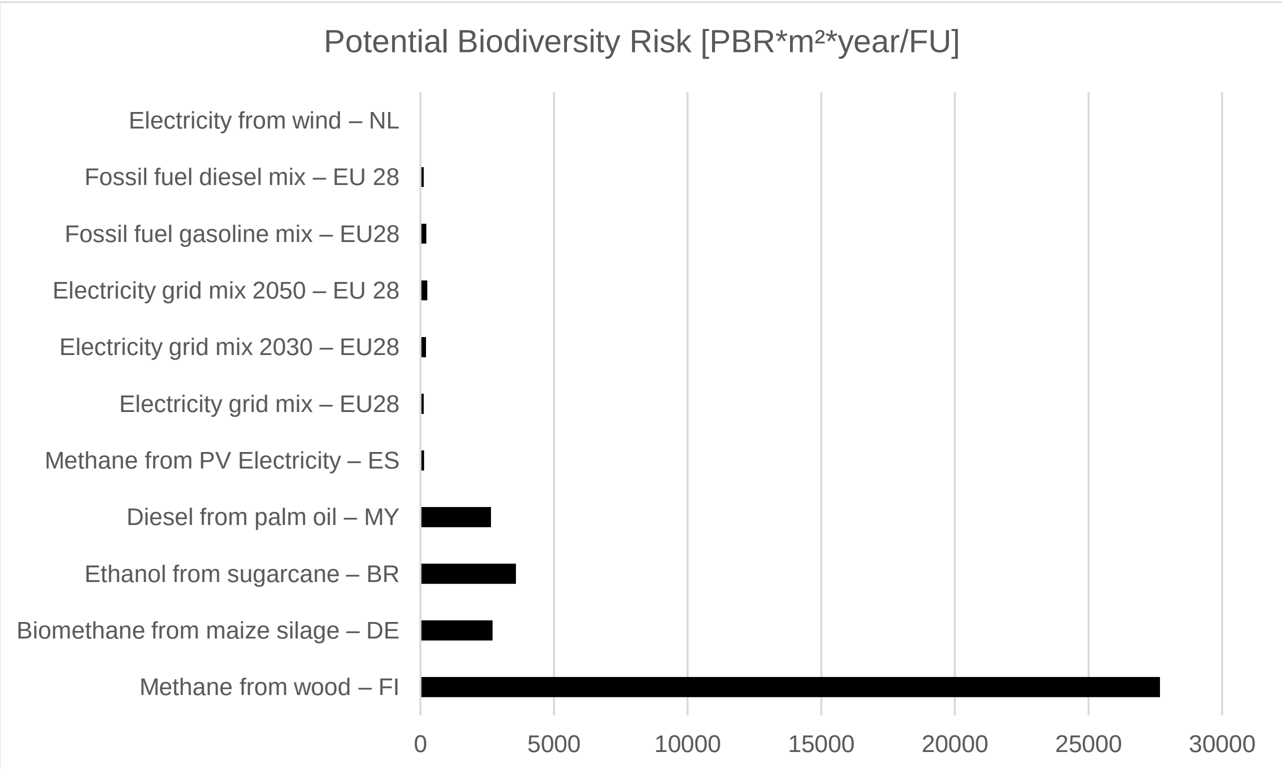


Figure 116: Potential biodiversity risk due to different transport energy carriers

The results only for the non-bio fuels (including the electricity grid mix) are depicted in Figure 117. Herein, it can be seen that especially the use of electricity as transport energy carrier derived from renewable sources such as wind energy in the Netherlands or solar energy in Germany has the least impact. The highest impact is due to the EU electricity grid mix as it is predicted to be in 2050 in the GaBi database. Herein, a mix of different energy carriers is assumed which also includes bio-fuels. For this energy scenario it is predicted that the share of renewable energies will increase which includes an increase in energy from biomass [312].

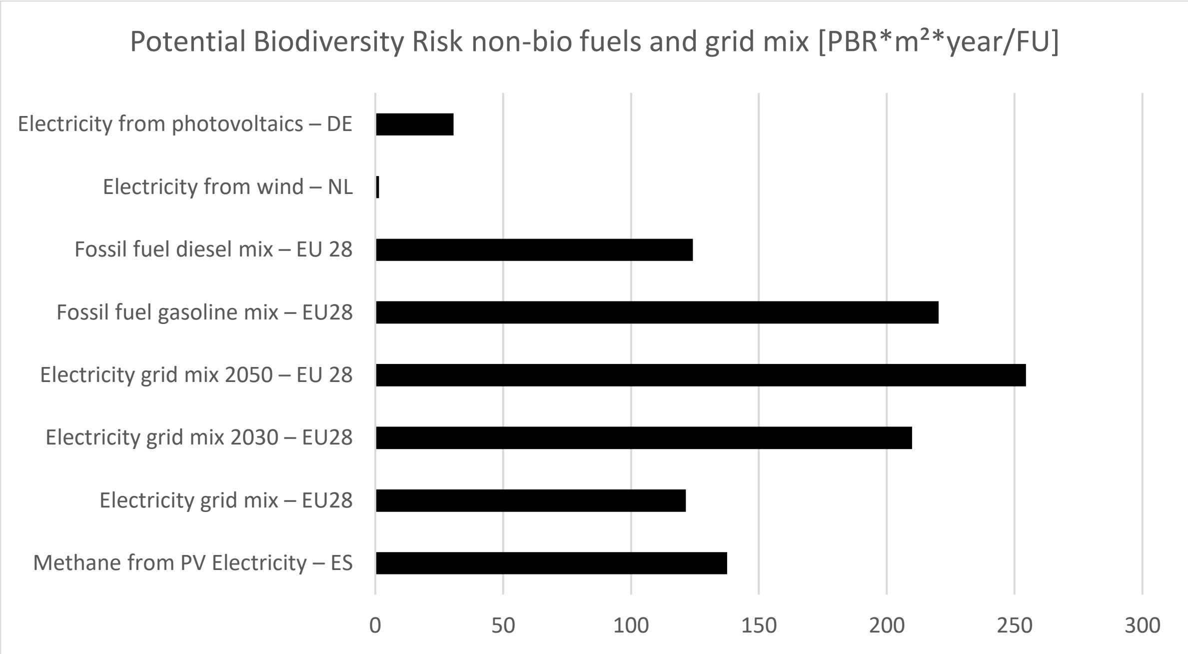


Figure 117: Potential biodiversity risk non-bio fuels and electricity grid mix

6.4.1 Global scale: regionalization of global biodiversity risks

By having a closer look at the bio-fuels using foreground information, with regard to the location where the land use process takes place, the highest probability that the land use process takes place in a biodiversity risk area is for the sugarcane production for bio-ethanol in Brazil as well as for bio-diesel made from palm oil in Malaysia. For these countries, there is a probability of more than 90% that the land use flows cropland, plantation or urban take place in risk areas. For the land use flow secondary forest in Finland there is a probability of about 45% that the land use takes place in biodiversity risk areas according to the background database. Yet, by looking at the specific location of forestry in Finland in the region of Southern Finland this probability is decreased to about only 5%. The same goes for the production of C4 annual crops (corn) in Germany. Whereas in all of Germany 24% of C4 annual crop production takes place in biodiversity risk areas, the risk decreases to 15% for the specific region of Saxony because the more detailed location of the crop production site is known. With regard to the proportion of risk areas affected of the UBR map, for the background data the highest share of proactive and reactive risk areas affected is in Brazil with an average of about 35% of proactive and reactive risk areas. In Malaysia there are about 28% of risk areas. The lowest number of risk areas affected is for the background database in the production site of Finland. Even though the probability that forestry takes place within a biodiversity risk area in Finland is quite high, only a low number of sites would be actually affected. This is indicated by the UBR value of 0.12 and is due to the fact that there are only few biodiversity risk areas in Finland that overlap with secondary forest areas (see Figure 118).

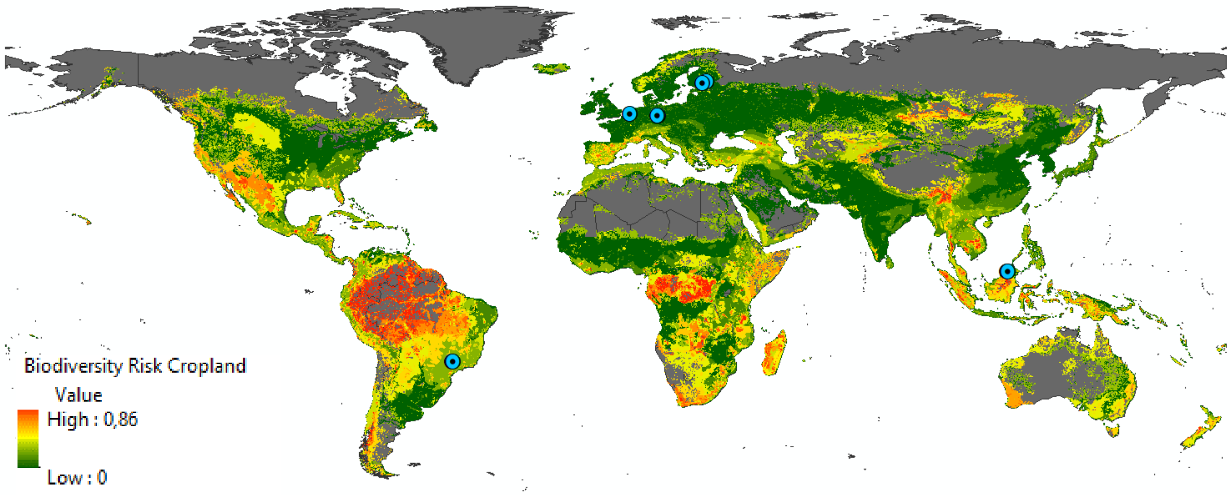


Figure 118: Production sites of biofuel crops (blue) on the UBR map for the land use type cropland

The Jenkins Index of the background database is highest for plantations in Malaysia with a value of 0.35. This indicates that in Malaysia a high proportion of plantation area is not covered by any conservation schemes, but still contains quite a high diversity of species, as well as a high proportion of threatened as well as endemic species. For the foreground database, the Jenkins value for the specific production sites of Sabah (Malaysia) and São Paulo (Brazil) is 0 since the whole area lies within biodiversity risk areas. The overall highest global risk factor is for plantations in Malaysia with a total value of 1.62 followed by the palm oil extraction plant in Malaysia with a value of 1.60. The lowest global risk factor is for the SNG plant in Finland with a value of 0.21 according to the background database. In the foreground system, the highest risks exist for the ethanol production plant in São Paulo with a value of 1.5. The lowest risks are for the HVO crude oil processing plant in South Holland which lies outside any biodiversity risk area and only has a Jenkins value of about 0.1. The values for the global biodiversity risk factor for all production processes are depicted in Table 21 for the background data and in

Table 22 for the foreground data.

Table 21: Global biodiversity risks for different land use processes – background data (country)

Fuel type	Product	Feedstock and production	Land use type	Land use subtype	Country	BR_overlay	PRA	Jenkins index	Global risk factor (BR_globe)
Bio-gas wood	Methane from wood	Pyrolysis from wood	Forestry	Secondary forest	Finland	0.273	0.084	0.067	0.423
	Methane from wood	SNG plant	Urban		Finland	0.463	0.011	0.066	0.540
Bio-diesel	Diesel from palm oil	Palm oil	Plantation	C3 perennial crop	Malaysia	0.518	0.262	0.350	1.101
	Diesel from palm oil	Extraction	Urban		Malaysia	0.485	0.185	0.347	1.016
	Diesel from palm oil	Crude palm oil HVO process	Urban		Netherlands	0.063	0.005	0.068	0.137
Bio-gas corn	Methane from corn	Anaerobic digestion from	Cropland	C4 annual crop	Germany	0.082	0.021	0.069	0.173
	Methane from corn	Biogas plant	Urban		Germany	0.021	0.020	0.069	0.11
Bio-ethanol	Ethanol from sugar-	Sugarcane	Cropland	C4 perennial crop	Brazil	0.560	0.263	0.170	0.993
	Ethanol from sugar-	Ethanol production	Urban		Brazil	0.506	0.22	0.176	0.903

Table 22: Global biodiversity risks for different land use processes – foreground data (region)

Fuel type	Product	Feedstock and production method	Land use type	Land use subtype	Country	Region	BR_overlay	PRA	Jenkins Index	Global risk factor (BR_globe)
Bio-gas wood	Methane from wood	Pyrolysis from wood	Forestry	Secondary forest	Finland	Southern Finland	0.0480	0.0582	0.0718	0.1780
	Methane from wood	SNG plant	Urban		Finland	Southern Finland	0.0358	0.0582	0.0724	0.1663
Bio-diesel	Diesel from palm oil	Palm oil	Plantation	C3 perennial crop	Malaysia	Sabah	0.9996	0.2202	0.0000	1.2199
	Diesel from palm oil	Extraction	Urban		Malaysia	Sabah	1.0000	0.2202	0.0000	1.2202
	Diesel from palm oil	Crude palm oil HVO process	Urban		Netherlands	South Holland	0.0000	0.0000	0.0761	0.0761
Bio-gas corn	Methane from corn	Anaerobic digestion from corn	Cropland	C4 annual crop	Germany	Saxony	0.1490	0.0094	0.0716	0.2300
	Methane from corn	Biogas plant	Urban		Germany	Saxony	0.0518	0.0094	0.0714	0.1326
Bio-ethanol	Ethanol from sugarcane	Sugarcane	Cropland	C4 perennial crop	Brazil	São Paulo	1.0000	0.1514	0.0000	1.1514
	Ethanol from sugarcane	Ethanol production	Urban		Brazil	São Paulo	0.9955	0.1514	0.3063	1.4532

6.4.2 Local scale: land use type and intensity specific biodiversity risks

The local biodiversity risks due to the specific land use types are highest within the background database for the land use processes urban with a biodiversity quality difference of almost 50% for the extraction plant in Malaysia as well as the ethanol extraction plant in Brazil. The lowest land use type specific risk applies to the palm oil plantation with a quality difference of about 27% compared to primary vegetation. For the background database the land use flow “intense” was assumed based on the suggestions of Uusitalo et al. [311]. Yet, by using the more specific location of the land use processes such as the region (here e.g. Saxony) or the coordinates (here e.g. São Paulo), the risks change because the land use intensity in these regions varies on the LUI maps. For example, for the land use flow forestry in Finland a biodiversity quality difference of 30% was indicated in the background database for intense forestry. By using the specific location this value decreases to only 21% suggesting that the land use intensity in this specific region might be lower than assumed. The highest difference between the background and foreground values applies to the land use flows urban in Finland as well as in Malaysia. Here, the quality difference for the production plants decreases to only 14% in Southern Finland and 19% in Sabah. For the location in Southern Finland this is due to the fact that the intensity values of the indicators of light pollution, population intensity as well as traffic intensity lies in the lowest intensity range in this region. The same is true for the location of the palm oil extraction plant where also the imperviousness intensity value is very low. The results for the impact of local biodiversity risks is depicted in Table 23 at a country level, regional level and for the coordinates of the specific production sites.

Table 23: Local biodiversity risks due to different land use types and intensities

Fuel type	Product	Feedstock and production method	Land use type	Land use subtype	Country	Region	Land use risk country (BR_locLUI)	Land use risk region (BR_locLUI)	Land use risk coordinates (BR_locLUI)
Bio-gas wood	Methane from wood	Pyrolysis from wood	Forestry	Secondary forest	Finland	Southern Finland	30.44	30.333	30.294
	Methane from wood	SNG plant	Urban		Finland	Southern Finland	44.989	44.882	16.601
Bio-diesel	Diesel from palm oil	Palm oil	Plantation	C3 perennial crop	Malaysia	Sabah	33.20	27.233	27.233
	Diesel from palm oil	Extraction	Urban		Malaysia	Sabah	49.216	22.688	19.800
	Diesel from palm oil	Crude palm oil HVO process	Urban		Netherlands	South Holland	45.412	45.412	45.412
Bio-gas corn	Methane from corn	Anaerobic digestion from corn	Cropland	C4 annual crop	Germany	Saxony	34.833	32.578	32.578
	Methane from corn	Biogas plant	Urban		Germany	Saxony	48.927	44.967	44.967
Bio-ethanol	Ethanol from sugarcane	Sugarcane	Cropland	C4 perennial crop	Brazil	São Paulo	36.165	33.265	31.922
	Ethanol from sugarcane	Ethanol production	Urban		Brazil	São Paulo	49.216	49.216	42.710

Separate land use intensity values as well as some primary data for management parameters are depicted in the next tables. For some management parameters primary data was provided by Uusitalo et al. [311]. This data was used in order to check the differences in land use intensity and thus local biodiversity impacts compared to the background database. For the palm oil production in Malaysia, for example, the background data suggest an application rate of pesticides of 5.9 kg/ha which leads to an intensity value of 0.84. Whereas the primary data indicates only a pesticide application rate of 3 kg/ha, which reduces the pesticide intensity value to 0.43. Also, the fertilizer intensity is lower by using the primary data of 95 kg nitrogen/ha (with an intensity value of 0.88) compared to the secondary data of 113.59 kg/ha with an intensity of 1. For the other management parameters no primary data was available, therefore the background data is used. In total, the background data indicates a land use intensity for palm oil production of 0.73 resulting in a biodiversity quality reduction of 29.23% whereas the primary data indicates a LUI value of 0.63 resulting in a biodiversity quality reduction of only 24.94% (see Table 24). Here the advantages of a method that can be applied both in the foreground and in the background can be demonstrated. Further specific values of the management parameters are provided in Annex VI (for the primary as well as the secondary data for all land use processes).

Table 24: Management parameter palm oil – Plantation, C3 perennial (Malaysia)

Management Parameter	Primary data*	Second-ary data	Unit	Intensity background data	Intensity pri-mary data
Fertilizer	95	113.59	kg nitrogen ha−1·year−1	1	0.883
Pesticide	3 kg/ha	5.9	kg/ha	0.84	0.43
Mechaniza-tion (tillage)	N/A*	0	No of tractor ha−1·year−1	0	N/A*
set-aside ar-eas	N/A*	5	Ratio Field size/buffer zone size [%]	0.8	N/A*
Crop rota-tion/crop di-versity	0	0	Share crop diversity per field [%]	1	1
			Total LUI	0.728	0.63
			Total BR_locLUI	29.23	24.94

* for missing primary data, the value of the background data is taken

6.4.3 Regional scale: landscape development index

The values of the BR_regLDI for the landscape matrix where the specific land use processes are embedded are depicted in Table 25. The highest BR_regLDI is for the ethanol production plant in São Paulo with a value of 0.31, followed by the HVO plant in the Netherlands for the production of bio-diesel from palm oil. The lowest value is in Malaysia, Borneo, for the palm oil plantation with a BR_regLDI of 0.12. Here, for example, more than 50% of the landscape is still comprised of primary forest under a low land use intensity. Almost 30% of the landscape is used for the plantation of palm oil trees with a medium to high land use intensity (intensity value of 0.6). The other patches of this landscape are composed of about 8% of C3 annual crop production, about 6% of secondary forests, and about 1% of C4 annual crop production and urban land use. For the production of sugar cane in Brazil with a high BR_regLDI for example about half of the land is used for C4 perennial crop production and only a very small area is comprised of primary or secondary vegetation. The values for the landscape composition as the share of land use types of all land use processes is depicted in Table 26.

6.4.4 Biodiversity impact: single point value

The characterization factor is calculated from the global (BR_globe), regional (BR_regLDI) and local biodiversity (BR_locLUI) risk values. This characterization factor is multiplied with the area required per functional unit as well as the time of occupation which is modelled here for one year. The individual biodiversity impacts are summed to a total biodiversity impact for each bio-based energy transport system. The resulting value is expressed as Potential Biodiversity Risk times area and year per functional unit (PBR *m²* year/FU). The highest characterization factor is for the ethanol production plant in São Paulo which shows a high impact at all three spatial scales. Even though the characterization factor for wood production in Finland is rather small, it has still the highest total impact. Because the pyrolysis from wood requires a very large area for wood production of about 28,000 m² per functional unit. The second highest impact is from bio-ethanol production in São Paulo, Brazil, where the area requirements are much smaller (2,442 m² per functional unit for sugarcane production) but the characterization factor is much higher. The lowest total biodiversity impact is for biogas production from corn in Germany where the area requirements for corn production are much higher than, for example, for sugarcane production (3,554 m² per functional unit) yet the characterization factor is much smaller. Upgrading from biogas to bio-methane is not included because the land requirements are negligible. Bio-diesel from palm oil production only has a marginally higher total impact than biogas production from corn even though the risks for biodiversity at the global scale are quite high. However, at the regional and local scale the biodiversity risks for corn production are much higher (see Table 27).

Table 25: Land use ratios for LDI in landscape of case study regions (coordinates)

Fuel type	Product	Feedstock and production method	Land Use type	Land use sub-type	Country	Region	BR_regLDI
Bio- diesel	Diesel from palm oil	Crude palm oil HVO process	Urban		Netherlands	South Holland	0.30
Bio- diesel	Diesel from palm oil	Extraction	Urban		Malaysia	Sabah	0.23
Bio- diesel	Diesel from palm oil	Palm oil	Plantation	C3 perennial crop	Malaysia	Sabah	0.12
Bio-ethanol	Ethanol from sugarcane	Sugarcane	Cropland	C4 perennial crop	Brazil	Sao Paulo	0.27
Bio-ethanol	Ethanol from sugarcane	Ethanol production	Urban		Brazil	Sao Paulo	0.31
Biogas corn	Methane from corn	Anaerobic digestion from corn	Cropland	C4 annual crop	Germany	Saxony	0.26
Biogas corn	Methane from corn	Biogas plant	Urban		Germany	Saxony	0.26
Biogas wood	Methane from wood	Pyrolysis from wood	Forestry	Secondary forest	Finland	Southern Finland	0.28
Biogas wood	Methane from wood	SNG plant	Urban		Finland	Southern Finland	0.29

Table 26: Landscape composition as share of land use type

Feedstock	Country	Region	C3ann	C3nfx	C3per	C4ann	C4per	Pastr	Primf	Primn	Range	Secdf	Secdn	Urban
Pyrolysis from wood	Finland	Southern Finland	1.00%	NA	NA	NA	NA	0.01%	0.00%	0.01%	0.01%	92.53%	NA	0.39%
SNG plant	Finland	Southern Finland	31.17%	0.08%	0.08%	NA	NA	0.3064%	NA	0.00%	0.00%	60.29%	NA	6.58%
Palm oil	Malaysia	Sabah	8.13%	NA	27.49%	0.15%	0.11%	0.05%	57.53%	0.00%	0.00%	5.59%	NA	0.96%
Extraction	Malaysia	Sabah	8.13%	NA	27.49%	0.15%	0.11%	0.05%	57.53%	0.00%	0.00%	5.59%	NA	0.96%
HVO process	Netherlands	South Holland	12.86%	0.36%	0.41%	4.11%	NA	10.73%	NA	0.00%	0.00%	NA	34.32%	36.68%
Anaerobic digestion	Germany	Saxony	44.72%	1.46%	0.89%	5.39%	NA	14.70%	NA	0.00%	NA	NA	32.55%	0.28%
Biogas plant	Germany	Saxony	31.83%	1.02%	0.56%	3.85%	NA	5.60%	NA	0.00%	0.00%	NA	40.77%	16.17%
Sugarcane	Brazil	São Paulo	0.80%	4.29%	1.09%	0.71%	47.56%	19.23%	1.16%	0.00%	19.77%	0.02%	NA	5.36%
Ethanol production	Brazil	Sã Paulo	0.46%	3.01%	0.78%	0.86%	35.04%	26.45%	1.63%	0.00%	5.24%	0.04%	NA	26.49%

Table 27: Total biodiversity impacts per land use process and energy system

Fuel type	Feedstock/ production method	Country	Region	Global (BR_ globe)	Local (BR_locLUI)	Regional (BR_regLDI)	CF [PBR]	CF back-ground [PBR]	Impact [PBR*m²* year]	Total impact [PBR*m²* year]
Bio-diesel	Crude palm oil HVO process	Netherlands	South Holland	0.076	0.454	0.299	0.829	0.810	0.059	
Bio-diesel	Extraction	Malaysia	Sabah	1.220	0.198	0.233	1.652	1.648	0.004	
Bio-diesel	Palm oil	Malaysia	Sabah	1.220	0.272	0.116	1.608	1.534	2763.990	2764.044
Bio-ethanol	Sugarcane	Brazil	São Paulo	1.151	0.319	0.268	1.739	1.462	4246.640	
Bio-ethanol	Ethanol production	Brazil	São Paulo	1.453	0.427	0.311	2.191	1.505	2.244	4248.880
Biogas corn	Anaerobic digestion from corn	Germany	Saxony	0.230	0.326	0.256	0.811	0.748	2747.696	
Biogas corn	Biogas plant	Germany	Saxony	0.133	0.450	0.260	0.842	0.826	0.221	2747.917
Biogas wood	Pyrolysis from wood	Finland	Southern Finland	0.178	0.211	0.284	0.673	0.991	16285.770	
Biogas wood	SNG plant	Finland	Southern Finland	0.166	0.145	0.291	0.603	1.031	0.006	16285.772

6.5 Discussion and interpretation of the case study results

The results of the case study show the area of land use are among the decisive factors for the overall biodiversity impact. Since the characterization factor is multiplied by the land area, the biodiversity impacts increase with increasing area. In general, the non-biofuel energy carriers (including the electricity grid mix) have a lower biodiversity impact. But here, especially electric energy generated from renewable sources such as wind or solar energy show the least impact. Biofuel systems, which require more land, have the highest impact which is the case here for biogas from wood production. This result is comparable to the results of Mila i Canals et al. [313] and Uusitalo et al. [311] where, when comparing the land use impacts of different products, the area under cultivation and the yields significantly influenced the results. Nevertheless, companies can still improve their impacts if they improve their management by reducing land use intensity, by selecting production sites with low biodiversity risks or by switching to resources that are more yield efficient. Furthermore, this case study reveals that biodiversity risks can be assessed more accurately by using foreground information. Additionally, the risks can be better quantified if the detailed location of the land use location is known.

Recommendations on biogas from wood production

The production location of wood is generally good, as the global biodiversity risk factor in this region of Finland is very low. Also, the regional impact on biodiversity (LDI value) in this area is relatively low since a large area of the landscape is comprised of secondary vegetation. Yet, since pyrolysis of wood for biogas production requires more land per functional unit, the overall biodiversity impact is still very high. It is therefore recommended to switch to another biomass which is more yield efficient. Alternatively, instead of using primary wood biomass a change to secondary wood biomass (for example by using wood waste or other forms of bio-waste) for the generation of biogas is recommended; this would also reduce the high land requirements. In line with these results pyrolysis from primary wood biomass is rarely practiced. Some of the existing pyrolysis plants had to be shut down because the process was uneconomical [314], or they use bio-waste and secondary wood biomass instead [315].

Recommendations on palm oil production

The location of palm oil production in Malaysia, Borneo, is decisive, as most palm oil production sites are located in areas with high biodiversity risk. Therefore, palm oil should preferably come from other palm oil exporting countries, such as Nigeria, where the risks to biodiversity are much lower. If a change of supplier is not possible, a switch to more sustainable palm oil land management should be made, as there is still room for improvement. For example, the pesticide application rate for palm oil production in this area is in the highest intensity range. Furthermore, no crop rotation and crop diversity was indicated in the background database. In order to improve the land management of these parameters the conservation evidence database recommends for plantations for example to "[...] *grow [plantation] crops under shade trees that are either native tree species that are remnants from cleared vegetation, or other crop trees (often referred to as 'agroforestry'). This approach provides a more complex habitat than conventional monoculture farming and can support higher levels of biodiversity*" [198]. In order to reduce the use of pesticides, a conservation management activity could be to cultivate plants that produce chemicals that attract natural enemies of pests. These non-crop plants may be cultivated at field edges or planted in between the main crop [198]. However, the results for palm oil also indicate that a switch to other bio fuel resources is not recommended (except for waste fats), as palm oil is quite land area efficient (it does not require much land per functional unit). This is in line with the IUCN's recommendation and the conclusion of the study of Parsons et al. [316], that assessed the impact of different feedstock oils on biodiversity and supports sustainable palm oil production over the production of other oils for biodiversity [317]. The regional biodiversity impact (BRreg_LDI) value is quite good in this production area in Borneo, so it is recommended to continue to conserve and to set-aside the remaining primary and secondary vegetation around the production sites in order to maintain the good regional impact values. This is in line with the study of Scriven et al. [318] that suggests that connectivity and the surrounding forest cover are amongst the most important factors for biodiversity in palm oil plantations. Therefore, in this land use system it is especially crucial to avoid further land transformations.

Recommendations on bioethanol from sugar cane

As the global risk factor for this region in Brazil is quite high, companies could source their sugar cane from other countries such as India (being the 2nd largest exporter of sugar cane in the world), where the global risk to biodiversity would be significantly lower. If it is not possible to change the supplier country, one should improve the BR_regLDI (which was highest for sugar cane production) by reserving more land for biodiversity around the sugar cane production sites. Furthermore, secondary vegetation within the landscape could be restored. Derived from the Conservation Evidence Database, the following management activities could be undertaken to increase the proportion of set-aside land. For set-aside land within the field, conservation headlands could be created which are not fertilized. Here, pesticides are not used even in a 6 m wide marginal strip of sown arable crops. In order to increase the proportion of secondary vegetation within the landscape, strips or blocks of native flowering plants can be planted. In addition, the former vegetation can be replanted to replace lost habitat [198].

Recommendations on bioethanol from corn

The location of corn production in Saxony, Germany, has a rather low global biodiversity impact value and can therefore be promoted. However, the local and regional impacts are quite high. The local impacts could be improved by switching to more sustainable production methods, as the quality difference value is at the upper end of the range of arable land production. Herein, especially the high amount of fertilizer application of almost 300 kg/ha and year should be reduced. The Conservation Evidence Database suggest to reduce the use of synthetic fertilizer generally and rather switch to organic fertilizer or to use a mix of synthetic and organic fertilizers [198].The regional impact could also be improved by increasing the landscape heterogeneity and by designating more secondary and primary vegetation areas around the corn fields for biodiversity or by reducing the intensity of land use in the surrounding areas as in this region the landscape is composed of many patches under high anthropogenic activity.

General recommendations and outlook

With this case study the application of the method has been demonstrated. Herein, only the occupation impacts were part of the scope of the study. Impacts of indirect land use (as they are modelled in consequential LCA), as well as transformation impacts were not investigated. Yet, the assessment of transformation impacts is crucial as Campbell et al. [319] highlight in their review of the current literature on biodiversity impacts due to biofuel production. They suggest that biodiversity impacts of biofuels very much depend on the biofuel crop that is planted, the management of the system but also on the previous land use type of the production system. For example: A transformation from intensive cropland to sustainable palm oil plantation would have positive biodiversity impacts, whereas the transformation from primary and secondary vegetation to palm oil plantations would have adverse effects [319]. In general, it can be said that non-biofuel energy carriers are more beneficial for biodiversity and here especially electric energy from renewable sources such as wind and solar or potentially also power-to-fuel pathways. With regard to the conventional energy carriers such as fossil fuels one has to bear in mind that related climate change effects are not assessed with this method. Since the effects of climate change are among the main drivers of biodiversity loss, further research is necessary to make reliable recommendations [6,13,25]. Therefore, it is recommended to also investigate the correlation between biodiversity impacts of this method and other LCA categories such as GWP or land use. With regard to the biofuels, it would be necessary to also assess transformation impacts before giving overall recommendations on which biofuel system would have the least impacts on biodiversity. Furthermore, the assessment of indirect land use aspects would also be necessary, since, for example, the production of biofuels in one area might lead to a shift in production systems in another area [320].

7 Discussion

This chapter evaluates the herein developed method for assessing biodiversity along global value chains. First, it presents the major research gaps so far and how this method addresses those gaps in order to offer a coherent and comprehensive analytical access. Afterwards, the chapter discusses the limitations and uncertainties of the method as well as needs for further development.

7.1 Major research gaps and main findings

The global loss of biodiversity is of particular importance as this crisis affects not only our societies in general but also the global economy. More and more consumers as well as producers are increasingly aware of this global crisis and want to make responsible and biodiversity-friendly decisions. Life cycle assessment is a suitable tool to support these decisions, but as has been shown in this thesis is not yet mature with regard to biodiversity, as existing methods could not provide the end user with detailed information for all decision options: e.g. where should resources be preferably cultivated or sourced from, which raw materials should be grown, which cultivation methods are biodiversity-friendly and how should the landscape be designed in order to support our life on earth. With the help of the method developed in this dissertation, this social desideratum and the academic research gap was addressed and a multi-scale method was developed that provides information about the origin of products, their cultivation methods and different management activities in the field and in the landscape, i.e. at global, regional and local scale. For each of the scales, the method quantifies potential risks to biodiversity and allows comparison with alternatives. It combines databases, concepts and methods from different disciplines. In this way, the method helps to bridge the gap between ecology and conservation science by taking into account the current state of scientific knowledge in both areas. In particular, it addresses the following aspects that were not possible with previous methods.

Firstly, it provides characterization factors that allow the evaluation of different land use types and intensities along an intensity gradient. In doing so, a research gap is addressed which has been pointed out by Souza et al. [61] in the revision of the number of land use classes and the inclusion of land management parameters. The new list and hierarchy level of land use flows developed in this thesis potentially allows the assessment of an unlimited number of land use flows. So far, available methods could only assess a limited number of flows [5,55].

Secondly, the method allows the analysis of impacts on biodiversity along different spatial and organizational scales. For each of the three spatial scales it quantifies biodiversity risks by using indicators at several organizational levels of biodiversity and allows comparison with alternatives. In this way, it supports decision makers in questions related to the choice of production sites, raw materials, cultivation methods and landscape design. Previous methods usually focused on only one or two of the scales [5,55] and could not show disaggregated results, which limited their practical application for producers.

Thirdly, one of the major challenges, as highlighted in various review studies, was the limited global applicability of the methods and the evaluation of the individual land management parameters. So far, the globally applicable methods were not capable of assessing the impacts of individual management parameters. The methods with which land management could be assessed were too data intensive to be globally applicable and were usually only specific to individual land use types or specific world regions [5]. With the help of this method both requirements are combined into one framework. It is now possible to assess both a global supply chain using the herein developed background database and primary data from companies or land owners in all steps of the assessment. This was demonstrated in this thesis by the practical application of the new characterization factors using a case study on transport energy carriers with varying global production sites both in the foreground and in the background system.

Finally, the new method presented here represents a step forward in the field of biodiversity assessment in LCA. The most important innovation is the modular structure of the method across the three spatial scales and the possibility of evaluating on the basis of primary data and background data along global supply chains. This modular framework combines the approach of Brooks et al. [76], which highlights global biodiversity risk areas, with land use maps of Hurtt et al. [186], which indicate the locations of land occupation and transformation. In addition, global land use intensity indices are calculated based on the concept of Herzog et al. [188] and Erb et al. [189] using global databases and land management statistics [190–197]. The management parameters for calculating the LUIs are determined on the basis of the Conservation Evidence database of Sutherland et al. [198], which collects scientific evidence for biodiversity-effective management activities. The LUIs are coupled with the research results of Newbold et al. [104,105] of the PREDICTS database of Hudson et al. [187] in order to translate the intensity values into biodiversity impact values. The herein provided global impact maps as well as the LUIs are then used to calculate global LDIs by adapting the approach of Brown & Vivas [199]. Furthermore, the method is consistent and embedded in the UNEP Setac framework [142], it provides characterization factors and corresponds to the standardized LCA structure [129,130]. Thus, this method provides an interdisciplinary framework by linking the state of the art and research results from the disciplines across LCA, ecology, nature conservation and geosciences. It furthermore builds upon a catalogue of crucial requirements derived from these disciplines. This bridges the often highlighted gap of previous LCA methods that do not account also for the disciplines traditionally dealing with the measurement and analysis of biodiversity impacts [61,161]. Furthermore, the hierarchical structure of the method makes it easy to integrate new research results at any spatial and organizational level, which has been highlighted as crucial for method development by the Joint Research Centre [72].

The new developments of the method of each of the three spatial scales are discussed in more detail below. The catalogue of requirements, main research gaps and how they are addressed in the BioMAPS method is summarized in Table 28 at the end of this sub chapter.

Global scale

As highlighted by nature conservationists, there are areas worldwide that are of particular importance for the conservation of global biodiversity. These areas can be divided into areas with particularly high or low vulnerability and areas with high irreplaceability. As Gordon et al. [75] emphasize, all these areas are worthy of protection, which is why for this method they are summarized in a uniform biodiversity risk map. Previous methods only focused on either proactive or reactive areas or only took into account sites with high irreplaceability. To date, no method incorporated all biodiversity risk sites in their LCA method [5]. Furthermore, for this method the uniform biodiversity risk map is compared with land use maps that indicate the location of the most important land use flows. In this way, the potential risk that a product or raw material comes from an area at risk for biodiversity is provided for each type of land use. Existing scientific research findings on global nature conservation schemes were used and harmonized to assess the risks of global production sites. As has been shown, these risks are different for each type of land use. For example, a large area of arable land in the Democratic Republic of Congo can overlap to a large extent with risk zones of biodiversity, while pasture land has a much lower risk because it tends to be located outside the risk zones. This illustrates the importance of analyzing global biodiversity risks in relation to land use locations by using global land use maps. This method can thus show more accurate risk values than existing methods, which assume a general average risk value for an ecoregion or country independent of the location of land use sites [5,140]. By incorporating the knowledge gained of global biodiversity risks into the LCA assessment, production sites could be relocated to areas with lower biodiversity risks or resources could be sourced from regions that do not overlap with biodiversity risk areas.

Local scale

The results of the calculation of land use intensity indices and predicted biodiversity risks provide important insights for the conservation of biological diversity as well as for decision support for producers and consumers and for policy makers on the local scale. This has implications for the choice of different land use types and their land management. Not only can intensively and extensively cultivated raw materials and products be compared, but also two extensively cultivated raw materials that differ in their individual management practices. This allows, for example, the comparison of products from organic farming under different labels (e.g. Demeter or Bioland) with specific criteria and cultivation standards which has been also highlighted by Armengot et al. [180].

Furthermore, another challenge in existing methods is the limited number of taxa considered and the challenges of cross-taxon surrogacy, i.e. the assumption that the effects of anthropogenic land use on one taxon can be directly transferred to other taxa. Using the PREDICTS model for this method cross-taxon surrogacy can be avoided by including all taxa for which data have been published. This includes invertebrates and soil organisms that have not been taken into account in any LCA method so far [5]. In addition, several cause-effect chains are provided for each type of land use, allowing the use of auxiliary indicators and also the measurement of positive actions. This could promote and create incentives for biodiversity-friendly measures. The selection of management parameters and indicators is based on the Conservation Evidence Database, which identifies scientifically sound management activities that are effective for the conservation of biodiversity. This approach differs from other methods, some of which do not base the selection of indicators at all or not on scientific evidence as highlighted by Michelsen & Lindner [321].

Regional scale

As has been highlighted in this thesis and by several authors [54,55,184], the regional or landscape scale has been rarely addressed by LCIA methods so far. There are few methods which focus on certain landscape aspects such as fragmentation or perforation. Consequently, this method provides an analytical framework which takes into account impacts also at the regional level by assessing the landscape composition and the intensity of the landscape matrix. For the first time, different management and conservation approaches are included, which affect both the field and the landscape. With the help of the method it is thus possible to integrate both land-sharing and land sparing strategies into the assessment and to evaluate them in terms of their impact on biodiversity.

Table 28: Summary requirements and their inclusion in BioMAPS method

Discipline	Requirements	Inclusion in the BioMAPS method
Ecology and conservation science	Development of a multi-scale assessment method for the holistic assessment of biodiversity [62].	The hierarchical structure of the method allows for the assessment of biodiversity impacts across several spatial, organizational and administrative scales.
	Inclusion of different spatial scales [62].	The BioMAPS method assesses biodiversity impacts at three different spatial scales (global, regional and local).
	Inclusion of different organizational scales and indicators at the levels of ecosystems, species and genes (basket of indicators) [73].	At the global scale it takes into account indicators at the genetic, species and ecosystem levels. At the regional and local scale it takes into account indicators at the species (richness and abundance) and ecosystem (landscape composition and intensity) levels.
	Be able to give recommendations at different administrative scales, the various end users of a LCA (e.g. governments, companies, land owners, producers and consumers) [62].	Due to the hierarchical structure of the method it is possible to give individual recommendations for different LCA end users, as has been demonstrated in the case study.
	Inclusion of ecological and conservation indicators that account for the uneven global distribution of biodiversity as well as the different levels of vulnerability and irreplaceability [75,76].	The design of the unified biodiversity risk maps allows for the assessment of vulnerability and irreplaceability criteria. Furthermore, the uneven global distribution of biodiversity is taken into account by add-

Discipline	Requirements	Inclusion in the BioMAPS method
		ing more indicators for global species richness, endemic species, and threatened species in the gap analysis.
	Inclusion of the landscape matrix enabling the assessment of land sharing and land sparing conservation strategies [57].	The calculation of global landscape development indices allows for the regional assessment of the landscape composition and the overall intensity and therefore, takes into account both land sharing and land sparing strategies.
	Inclusion of different biodiversity impacts due to the types of land use, land use intensities and land management parameters [180].	The calculation of local land use intensity indices coupled with the specific biodiversity risks derived from the PREDICTS model for each land use type allows for the assessment of impacts due to the land use types, intensities and management parameters both in foreground calculations as well as for the supply chain.
LCA discipline	Be in line with the overall LCA framework [129,130].	The BioMAPS method has been set up in line with the standardize structure of the LCA framework according to ISO distinguishing between the Life Cycle inventory and the impact assessment phases.
	Adaptation of the Life Cycle Inventory and the land use flow list [61].	A new land use flow list has been developed which can be matched to the land use flow list used in current LCA software.
	Be in line with the UNEP-SETAC framework and provide characterization factors [138].	Characterization factors are provided. The biodiversity quality due to a specific land use type is calculated compared to the reference situation primary vegetation (minimum use) for occupation and

Discipline	Requirements	Inclusion in the BioMAPS method
		transformation, which is in line with the UNEP-SETAC framework. Yet, BioMAPS does not return a result for the area of protecting of ecosystem quality in [PDF m² a].
	Development of a cause-effect chain that includes “relief” indicators that are beneficial for biodiversity [5].	For each type of land use, a cause-effect chain has been developed. Herein, “relief” indicators that account for positive biodiversity impacts are included, such as the provision of set-aside areas or the amount of deadwood for forestry.
	Provision of decision support for LCA end users regarding the location of production, the comparison of different land use types and intensities as well as, the inclusion of management parameters [5,55].	The global biodiversity risk factor enables the comparison of different locations of land use with regard to their global impacts on biodiversity. The calculation of biodiversity risks derived from the LUIs allows for the assessment and comparison of values of management parameters with the benchmark values provided in the Annex V.
	Provision of decision support for LCA end users regarding land sharing and land sparing strategies [55].	Since the BR_regLDI takes account for both the composition of the landscape and the land use intensity of individual patches, decision support can be given regarding land sharing and land sparing conservation strategies.
	Use of scientific data which can be constantly updated [66].	The method relies on the empirical data of the PREDICTS model, as well as scientific data from conservation science and NGOs. The hierarchical structure of the

Discipline	Requirements	Inclusion in the BioMAPS method
		overall method allows for the integration of new research results at all scales.
	Be globally applicable both in a foreground and background system [61].	All individual steps of the method can be assessed either in a foreground process by using primary data and following the calculation rules, or in a background process by using the background data provided in chapter 5. A decision tree for the LCA end user for the application of the BioMAPS method in a foreground or background system is provided in chapter 6.
LCIA biodiversity methods	Cross-taxon surrogacy should be avoided by including data on several taxa [61].	Since the BioMAPS method includes data for several taxa from Newbold et al. [104,105] from PREDICTS, cross-taxon surrogacy can be mitigated. Furthermore, they have shown that there is no statistically significant difference for land use impacts on different taxa [105].
	Use of land use models to assesses the location where the actual land use takes place for the regionalization [5,140].	The land use model of Hurtt et al. [186] is used to assess the location of several land use types and sub types and its overlap with biodiversity conservation schemes harmonized in the UBR map.
	Include effects of land transformation on biodiversity risk areas by using for example land use suitability maps or historical land use models [5,140].	Historical and future land use scenarios are used to assess their overlap with the UBR map for the past and prospective 10 years.

Discipline	Requirements	Inclusion in the BioMAPS method
	Definition of the 'quantification' of biodiversity, thorough, transparent and coherent while subjectivity of the results should be avoided [61].	The term biodiversity is used based on the commonly agreed upon definition of the CBD as genetic, species and ecosystem diversity [70]. For each of these levels, science-based indicators are used.
	Improvement of the classification of land use and land cover through the use of spatial information by coupling of GIS and LCA [61].	All calculation steps for the background database have been performed in a GIS environment.
	The use of more than one biodiversity metric is recommended [61].	See requirement for a basket of biodiversity indicators.
	The overestimation or underestimation of the biodiversity impacts of some land use types should be avoided [61].	The sensitivity analysis showed a quite small standard deviation for the different land use types. Only the land use type cropland had a higher standard deviation in some regions of the world. Herein, the results should be treated with care and should be discussed transparently.
	An intensity gradient instead of intensity classes which are linked to different management parameters should be included [5,55].	See requirements for the inclusion of different biodiversity impacts due to the types of land use, land use intensities and land management parameters.

7.2 Limitations and future recommendations

Despite the highlighted improvements, there remain certain limitations that need to be addressed in the future. Some of these limitations result from the use of global data sets. As noted by Borrelli et al. [322], global spatial datasets often lack information on the inherent uncertainties. Therefore, they conclude that using general propagation techniques for uncertainties model estimation does not add any value [322]. In line with the issues they have discussed in their study related to global uncertainty analysis, they propose a different approach, depicting uncertainty as a likelihood distribution by using a Bayesian modeling technique using a Monte Carlo algorithm [322]. This technique could be used in future to further highlight uncertainties of the global datasets, being aware that the uncertainties are already introduced by the input data. For example, there are data uncertainties with regard to the different sizes of the individual conservation schemes. Some conservation schemes have been identified and designed fairly accurately and precisely, while others are extensive and their locations are less accurate [75]. However, since this method aims to make production sites and their biodiversity risks globally comparable, a less accurate assessment of the background system must be made. For the foreground system, however, the proportion of overlaps with risk sites of biodiversity at the site could be calculated if it is known whether or not and to what extent the production site is located in an area that is important for global biodiversity protection. Another point is the higher weighting of global risks, since the global risk factor can reach higher values than the regional or local factor. This is based on the suggestion of conservation scientist that a global loss of biodiversity has higher impacts than a local loss of biodiversity [204]. In other words, the last remaining species or ecosystems in the world are given a higher weighting than species or ecosystems that might be threatened only in a region or country, but which still occur in other countries or regions. However, this weighting is debatable and an equal assessment of all scales would be conceivable.

Further limitations concern the use of global data sets for the calculation of LUIs, as highlighted by Kuemmerle et al. [206] and Petz et al. [301]. Global datasets already contain inherent uncertainties and estimates that depend on the type of modelling and input data. Furthermore, due to the limited availability of global data sets, the same data sources are often used for different modelling approaches. This carries the risk of autocorrelation, which was also highlighted by Kuemmerle et al. [301] and Petz et al. [206]. Although this is a legitimate concern, it is nevertheless important for the method to consider all data sets individually in order to be able to make specific recommendations for individual parameters and values to landowners and decision makers.

Further uncertainties in the calculation of LUIs result from the disaggregation of national statistics for the management parameters pesticide use and mechanization. For these two parameters only the location and area of land use are considered. Other specific characteristics such as the type of crop grown or the specific toxicity of individual pesticide types are not taken into account. Therefore, it is advisable to use this data set only in the background system and to search for primary data in order to obtain more accurate results. In addition, weighting of individual management parameters according to their specific impacts on biodiversity is recommended as soon as scientific information is available. Thus, the use of a simple additive index is a strong simplification of reality, as pointed out by several authors [5,188]. There are always trade-offs between developing a globally applicable method and maximizing accuracy. The advantage of the additive land use intensity index is that it can use individual land management parameters that can be combined into a single index, providing a continuous scale for assessing a land use intensity worldwide [5].

Another limitation relates to the use of several organizational indicators as a "basket of biodiversity indicators". Here, might be a risk of regression to the mean when using too many indicators. Therefore, the BioMAPS method proposes a maximum of five indicators per index. Furthermore, double counting by measuring impacts on biodiversity cannot always be avoided, as the different organizational scales cannot be clearly separated. The Biodiversity Risk Index for the assessment of local impacts is only based on a single additive biodiversity index with different metrics. Possible relationships between the metrics are not considered. Furthermore, by including both species richness and rarefied species richness in the index, more weight could be given to species richness as a whole. This inclusion of a basket of indicators in a single index therefore poses a great simplification of reality, as the different biodiversity metrics are simply added together.

A further uncertainty of the developed method is derived from the assumed relationship between the risks of biodiversity and the intensity of land use, which is based only on the three fixed points of minimal, light and intensive land management. As shown in the sensitivity analysis, the results are influenced by the nature of the relationship, which does not necessarily reflect the real relationships. The standard deviation between different assumptions was highest for the land use type cropland with a median deviation of about 3%. Whereas, the other land use types only showed a very small standard deviation. Thus, especially for the land use type cropland it is recommended to always carry out a sensitivity analysis before making recommendations based on the results gained. In ecological studies different types of relationships have been found. They are strongly influenced by the ecological indicators used and by the management parameters studied to measure land use intensity [182,208,213,323–325]. However, so far there is no comprehensive global assessment and no database that allows to investigate a relationship between a land use intensity gradient and biodiversity risks, therefore the most frequent relationships found in ecological research were adopted and tested within this method.

Finally, the LDI highlights landscapes with a higher proportion of primary and secondary vegetation and landscapes under lower intensities. Using average values per country or region for the background database could be misleading as the landscape is then not necessarily under the influence of the landowner. The results of the LDI should therefore be treated with special care and obtaining primary data must be a priority. However, the average country values of the LDI could still be an incentive for decision makers (e.g. governments of countries) to set-aside and maintain more areas with primary and secondary vegetation or to keep more areas under lower land use intensity.

8 Outlook and conclusion

Further research is necessary to continuously improve and update the method. For example, so far only five parameters per land use type are considered. Some management parameters that were identified as effective in the Conservation Evidence Database could not be taken into account due to the lack of global data sets or indicators to measure impacts, such as the provision of artificial nesting sites, resting or feeding areas in urban areas or measures against poaching in forests. In addition, once global data on individual pesticide types is available, specific benchmark values for each type of pesticide should be provided, taking into account individual toxicity levels. For more toxic pesticides, the benchmark value would then be correspondingly lower, so that there is a difference in the maximum intensity of the individual pesticide types. Moreover, the default values for some management parameters should be updated when data become available. The missing data for some countries and sites also need to be updated, such as for the management parameter deadwood volume or traffic intensity, where data were only available for some European countries. Furthermore, all parameters are treated equally and are not weighted for the calculation of a final LUI, even though some parameters may have more negative impacts on biodiversity than others (e.g. pesticides vs. mechanization). Such a weighting factor, which takes into account the individual importance of each parameter for biodiversity conservation, could be included in the future. For example, the effectiveness values of the Conservation Evidence Database could act as a weighting factor. Thus, a management parameter like the reduction of pesticides, which shows an effectiveness of 100 a weighting value of 1 and the parameter mechanization, which is only 70% effective, a value of 0.7 could be given.

Also, at the regional level there is still potential for further development, as the landscape development index (LDI) only takes into account the proportion of land areas and their intensity in a landscape. It constitutes a basic area averaging, which means that possible amplifying or attenuating cross patch interactions cannot be represented. For example, it does not provide information on the distribution of patches of individual land use types, the distance to patches of natural habitat for species such as primary or secondary vegetation, and the shapes of individual patches (clustered or widespread). All these features have been shown to have an impact on biodiversity [326–330]. Further indicators should therefore be included at the landscape level to take into account their specific biodiversity impacts.

Moreover, as highlighted in research the shape of the complex relationship between land use intensity and biodiversity impacts is not yet ecologically fully understood, nor are the complex relationships between the individual parameters [182]. There are studies on individual parameters and individual biodiversity metrics. However, a holistic analysis of the interactions between the individual parameters and possible trade-offs is still lacking. Since these complex interrelations and their effects on biodiversity are not yet scientifically understood, they could not be included in the method. Until then, the method is based on the recommendation of some scientists to use a simple additive index [188,189,208], which is also easy to communicate. Therefore, further research is needed to fully understand the relationship between land use intensity, individual management parameters and the associated risks to biodiversity. As soon as such research results are available, they should be updated in the method.

In conclusion of this dissertation, it can be stated that the BioMAPS method, while it should be further developed, nevertheless represents a step forward in the field of biodiversity assessment in LCA. As highlighted, socioeconomic activities have a disruptive impact on biodiversity at different spatial scales, e.g., through the design of global supply chains, field management, and landscaping. Therefore, the development of a multi-scale method for assessing the impacts on biodiversity in the LCA is of high social, economic and ecological relevance. In view of this, however, the methods used so far have not been able to provide sufficient decision support. The BioMAPS method bridges this research gap by enabling the identification and comparison of more biodiversity-friendly alternatives on different scales. In doing so, the method provides a coherent framework for companies to assess biodiversity impacts along global supply chains and thus to help mitigate negative impacts. This decision support is all the more important because progressive land use and land use changes are among the main causes of the continuing loss of biodiversity and two thirds of the global biodiversity crisis can be directly attributed to anthropogenic land use activities. Consequently, BioMAPS aims to contribute to the various political goals of the CBD by protecting biodiversity around the world.

9 References

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10 Annex

10.1 Annex I: Land use flows

Table 29: Updated land use flow list, matched to the ILCD flows

Hierarchy	Names [138]	Definitions [138]	Land use type	Parameter
0	Unspecified	Land use and cover not known	Artificial	Sealing intensity intensified Set-aside intense Traffic intense Light Ø Population Ø
0.1	Unspecified, used	Human land use and resulting land cover not known	Artificial	Sealing intensity intensified Set-aside intense Traffic intense Light Ø Population Ø
0.2	Unspecified, natural	Natural land cover not known	Primary and secondary vegetation, forest	Reference situation
1	Forest	Areas with tree cover >15%	Primary and secondary vegetation, forest	Set-aside light Tree age Ø Biomass Ø Deadwood Ø Harvesting intensity Ø
1.1	Forest, natural	Forest not used by humans	Primary vegetation, forest	Reference situation
1.1.1	Forest, primary	Forests minimally disturbed by human impact, where flora and fauna species abundance is near pristine	Primary vegetation, forest	Reference situation
1.1.2	Forest, secondary	Areas originally covered with forest or woodlands, where vegetation has been removed, forest is regrowing and is no longer in use	Secondary vegetation, forest	Set-aside minimum Tree age Ø Biomass Ø Deadwood minimum Harvesting intensity minimum
1.2	Forest, used	Forests used by humans	Secondary vegetation, forest	Average of forest intensive and forest extensive
1.2.1	Forest, extensive	Forests with extractive use and associated disturbance like hunt-	Secondary vegetation, forest	Tree age intensity reduced, harvesting intensity reduced, Set-aside light, Biomass Ø,

Hie-rarchy	Names [138]	Definitions [138]	Land use type	Parameter
		ing, and selective logging, where timber extraction is followed by re-growth, including at least three naturally occurring tree species		Deadwood light
1.2.2	Forest, intensive	Forests with extractive use, with either even-aged stands and clear-cut patches, or less than three naturally occurring species at planting/seeding	Secondary vegetation, forest	Tree age intensified, harvesting intensified, Set-aside intense, Biomass Ø Deadwood intense
3	Shrub land	Areas with shrub-dominated sclerophyllous vegetation	Primary vegetation, non forest	Reference situation
4	Grassland	Herbaceous cover, closed to open (>15%) with scattered shrubs or trees	Pasture all	Livestock intensity Ø Set-aside intense Mechanization Ø Pesticide Ø Fertilizer Ø
4.1	Grassland	Naturally grassland dominated vegetation	Pasture all	Livestock intensity Ø Set-aside intense Mechanization Ø Pesticide Ø Fertilizer Ø
4.1.1	Grassland, natural	Grassland-dominated vegetation, fauna and flora near pristine (e.g., steppe, tundra, savannah)	Primary vegetation, non forest	Reference situation
4.1.2	Grassland, for livestock grazing	Grasslands where wildlife is replaced by grazing livestock	Pasture all	Livestock intensity intensified Set-aside light Mechanization Ø Pesticide Ø Fertilizer Ø
4.2	Pasture/meadow	Areas that have been converted to grasslands for livestock grazing or fodder production	Pasture all	Set-aside intense Livestock intensity Ø Mechanization Ø Pesticide Ø Fertilizer Ø
4.2.1	Pasture/meadow, extensive	Pasture with low number of livestock or meadows mechanically harvested 2 or 3 times per year, reduced input of fertilizer	Rangeland pasture	Livestock intensity reduced; Fertilizer intensity minimum reduced; Mechanization intensity reduced Set-aside light Pesticide intensity Ø
4.2.2	Pasture/meadow, intensive	Pasture with high number of livestock or meadows mechanically harvested 3 times or	Managed pasture	Mechanization intensity intensified livestock intensity intensified; fertilizer intensity intensified, Set-aside intense

Hie-rarchy	Names [138]	Definitions [138]	Land use type	Parameter
		more per year, fertilizer applied		Pesticide Ø
5	Agriculture	Areas used for crop production	Cropland all	Fertilizer Ø Pesticide Ø Crop rotation Ø Set-aside intense Mechanization Ø
5.1	Arable	Cultivated areas regularly ploughed and generally under a rotation	Cnfx, C3 ann, C4 ann	Fertilizer Ø Pesticide Ø Crop rotation Ø Set-aside intense Mechanization Ø
5.1.1	Arable, fallow	Cropland temporarily not used (<2 years)	Secondary vegetation	Set-aside minimum Tree age Ø Biomass Ø Deadwood minimum Harvesting intensity minimum
5.1.2	Arable, non-irrigated	Annual crop production based on natural precipitation (rainfed agriculture)	Cnfx, C3 ann, C4 ann	Fertilizer intensity Ø pesticide intensity Ø Set-aside intense Mechanization Ø Crop rotation Ø
5.1.2.1	Arable, non-irrigated, extensive	Use of chemical–synthetic and organic fertilizer as well as pesticides is reduced	Cnfx, C3 ann, C4 ann	Fertilizer reduced, pesticide reduced Set-aside light Mechanization Ø Crop rotation Ø
5.1.2.2	Arable, non-irrigated, intensive	Chemical–synthetic and organic fertilizer as well as pesticides are applied	Cnfx, C3 ann, C4 ann	Fertilizer intensified, pesticide intensified Set-aside intense Mechanization Ø Crop rotation Ø
5.1.3	Arable, irrigated	Annual crops irrigated permanently or periodically, using a permanent infrastructure (irrigation channels, drainage network). Most of these crops like rice could not be cultivated without an artificial water supply.	Cnfx, C3 ann, C4 ann	Fertilizer intensity Ø pesticide intensity Ø Set-aside intense Mechanization Ø Crop rotation Ø
5.1.3.1	Arable, irrigated, extensive	Use of chemical–synthetic and organic fertilizer as well as pesticides is reduced.	Cnfx, C3 ann, C4 ann	Fertilizer reduced, pesticide reduced Set-aside light Mechanization Ø Crop rotation Ø
5.1.3.2	Arable, irrigated, intensive	Annual crops irrigated permanently or periodically, using a permanent infrastructure (irrigation channels,	Cnfx, C3 ann, C4 ann	Fertilizer intensified, pesticide intensified Set-aside intense Mechanization Ø Crop rotation Ø

Hie-rarchy	Names [138]	Definitions [138]	Land use type	Parameter
		drainage network). Most of these crops like rice could not be cultivated without an artificial water supply. Does not include sporadically irrigated land.		
5.1.4	Arable, flooded crops	Areas developed for rice cultivation. Flat surfaces with irrigation channels. Surfaces regularly flooded.	Cnfx, C3 ann, C4 ann	Fertilizer intensity Ø pesticide intensity Ø Set-aside intense Mechanization Ø Crop rotation Ø
5.1.5	Arable, greenhouse	Crop production under plastic or glass	Cnfx, C3 ann, C4 ann	Sealing intensity intense Set-aside intense Fertilizer Ø Pesticide Ø Crop rotation Ø
5.1.6	Field margins/hedgerows	Areas between fields with natural vegetation	Secondary vegetation forest	Set-aside minimum Tree age Ø Biomass Ø Deadwood minimum Harvesting intensity minimum
5.2	Permanent crops	Perennial crops not under a rotation system which provide repeated harvests and occupy the land for a long period before it is ploughed and replanted: mainly plantations of woody crops	C3 per and C4 per	Crop rotation 1 Set-aside light Fertilizer Ø Pesticide Ø
5.2.1	Permanent crops, non-irrigated	Perennial crops production based on natural precipitation (rainfed agriculture)	C3 per and C4 per	Crop rotation 1 Set-aside light Fertilizer Ø Pesticide Ø Mechanization Ø
5.2.1.1	Permanent crops, non-irrigated, extensive	Use of chemical–synthetic and organic fertilizer as well as pesticides is reduced	C3 and C4 perennial	Fertilizer intensity reduced, pesticide intensity reduced; Set-aside light Crop rotation 1 Mechanization Ø
5.2.1.2	Permanent crops, non-irrigated, intensive	Chemical–synthetic and organic fertilizer as well as pesticides are applied	C3 and C4 perennial	Fertilizer intensity intensified, pesticide intensity intensified Set-aside intense Crop rotation 1 Mechanization Ø
5.2.2	Permanent crops, irrigated	Perennial crops with artificial input of water	C3 and C4 perennial	Crop rotation 1 Set-aside intense Fertilizer Ø Pesticide Ø Mechanization Ø

Hie-rarchy	Names [138]	Definitions [138]	Land use type	Parameter
5.2.2.1	Permanent crops, irri-gated, exten-sive	+ Use of chemical–synthetic and organic fertilizer as well as pes-ticides is reduced	C3 and C4 perennial	Fertilizer intensity reduced; pesticide intensity reduced Set-aside light Crop rotation 1 Mechanization Ø
5.2.2.2	Permanent crops, irriga-ted, intensive	+ Chemical-synthetic and organic fertilizer as well as pesticides are applied	C3 and C4 perennial	Fertilizer intensity intensified, pesticide intensity intensified Set-aside intense Crop rotation 1
6	Agriculture, mosaic	Heterogeneous, agri-cultural production in-tercropped with (native) trees.	Cropland All	Set-aside light Fertilizer Ø Pesticide Ø Mechanization Ø Crop rotation Ø
7	Artificial areas	Artificial surfaces and associated area(s)	urban	Sealing intensity intensified Set-aside intense Traffic intense Light Ø population Ø
7.1.1	Urban/indust-rial fallow	Areas with remains of industrial buildings; de-posits of rubble, gravel, sand and industrial waste. Can be vege-tated	urban	Sealing intensity Ø Set-aside intense Traffic light Light Ø Population Ø
7.1.2	Urban, conti-nuously built	Buildings cover most of the land. Roads and artificially surfaced area cover almost all the ground. Non-linear areas of vegetation and bare soil are ex-ceptional. At least 80% of the total area is sea-led	urban	Sealing intensified Set-aside intense Traffic intense Light intensified Population intensified
7.1.3	Urban, dis-continuously built	Most of the land is cov-ered by structures. Buildings, roads, and artificially surfaced ar-eas associated with ar-eas with vegetation and bare soil, which occupy discontinuous but significant sur-faces. Less than 80% of the total area is sea-led	urban	Sealing intensity Ø Set-aside intense Traffic light Light Ø Population Ø
7.1.4	Urban, green areas	Areas with vegetation within urban fabric. In-cludes parks with veg-etation	urban	Sealing intensity reduced Set-aside light Traffic minimum Light reduced Population

Hie-rarchy	Names [138]	Definitions [138]	Land use type	Parameter
7.2	Industrial area	Artificially surfaced areas (with concrete, asphalt, or stabilized, e.g., beaten earth) devoid of vegetation occupy most of the area in question, which also contains buildings and/or areas with vegetation	urban	Sealing intensified Set-aside intense Traffic intense Light Ø Population Ø
7.3	Mineral extraction site	Areas with open-pit extraction of industrial minerals (sandpits, quarries) or other minerals (opencast mines). Includes flooded gravel pits, except for riverbed extraction	urban	Sealing intensity 1.0 Set-aside 1.0 Traffic intense Light Ø Population Ø
7.4	Dump site	Landfill or mine dump sites, industrial or public	urban	Sealing intensity 1.0 Set-aside 1.0 Traffic intense Light Ø Population Ø
7.5	Construction site	Areas under construction development, soil or bedrock excavations, earthworks	urban	Sealing intensity 1.0 Set-aside 1.0 Traffic intense Light Ø Population Ø
7.6	Traffic area	Areas used for traffic infrastructure	urban	Average of road, rail and embankment
7.6.1	Traffic area, road network	Motorways, including associated installations (gas stations)	urban	Sealing intensity 1.0 Set-aside 0.9 Traffic intense Light population
7.6.2	Traffic area, rail network	Railways, including associated installations (stations, platforms)	urban	Sealing intensity 0.9 Set-aside intense Traffic intense Light population
7.6.3	Traffic area, rail/road embankment	Vegetated area along motorways and railways	urban	Sealing intensity light Set-aside light Traffic intense Light population

*Note: for the forest flows the biodiversity risk impact interval of plantation forests is used. LUIs larger than 1 are capped to 1 since a further intensification is hardly possible. For all flows the maximum value per country is taken for a conservative worst case approach. If better information on the region is known the values can be calculated more accurate.
Definition: is a management practice „reduced" it is multiplied by 0.67 (which is the average minimum value divided by the light intensity value); is a management practice „intensified" it is multiplied by 1.647 (which is the intense value divided by the light intensity value). The intensity values are defined as follows:

- Minimum= 0.17
- Average minimum =0.34
- Light= 0.51
- Average intense= 0.68
- Intense =0.84

10.2 Annex II: Biodiversity risk maps – global scale

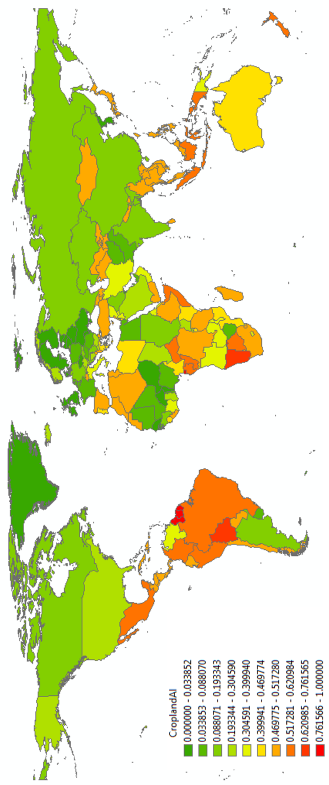

Figure 120: Proportion of land use cropland in biodiversity risk areas

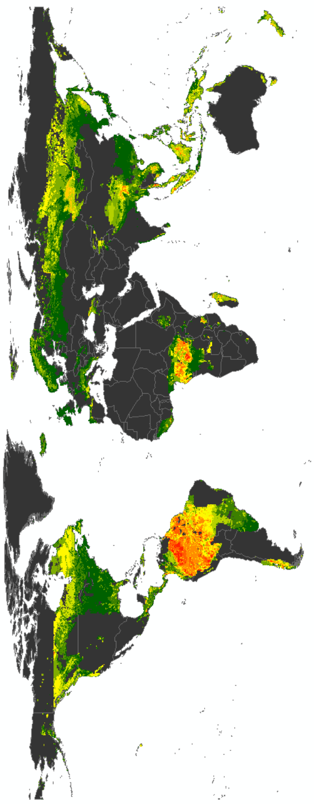

Figure 121: Biodiversity risk secondary vegetation forest foreground system

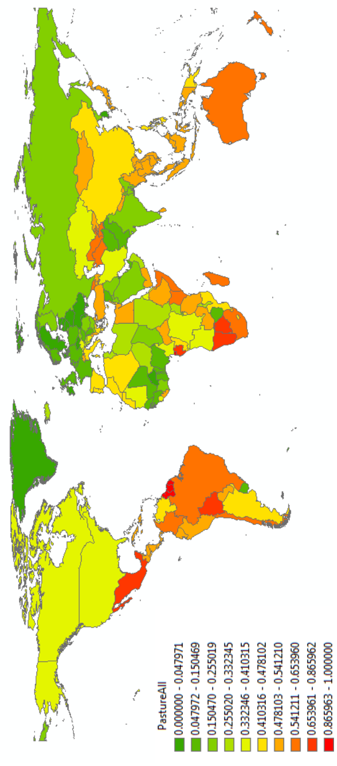

Figure 122: Proportion of land use pasture in biodiversity risk areas

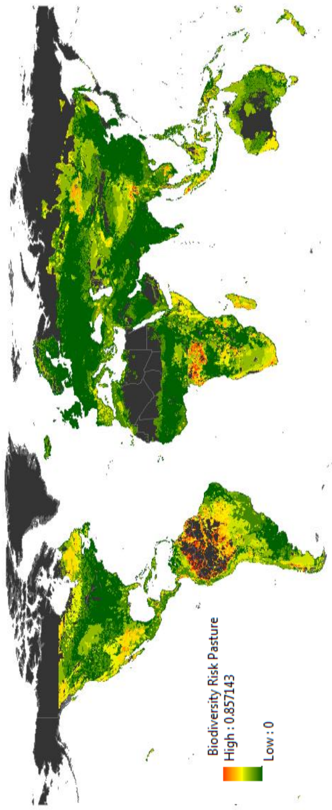

Figure 119: Biodiversity risk pasture foreground system

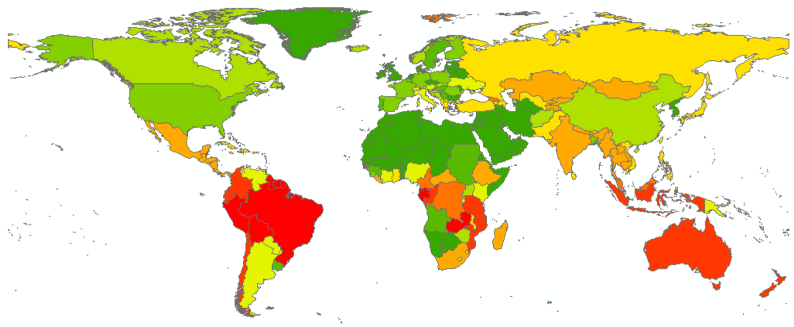

Figure 123: Proportion of land use forest in biodiversity risk areas

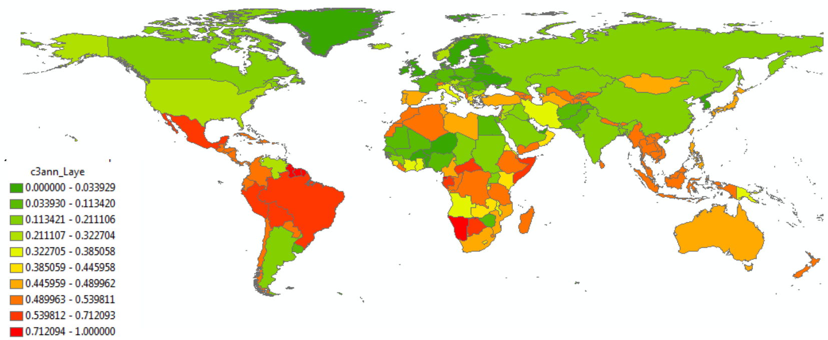

Figure 124: Proportion of C3 annual crops in proactive or reactive risk areas

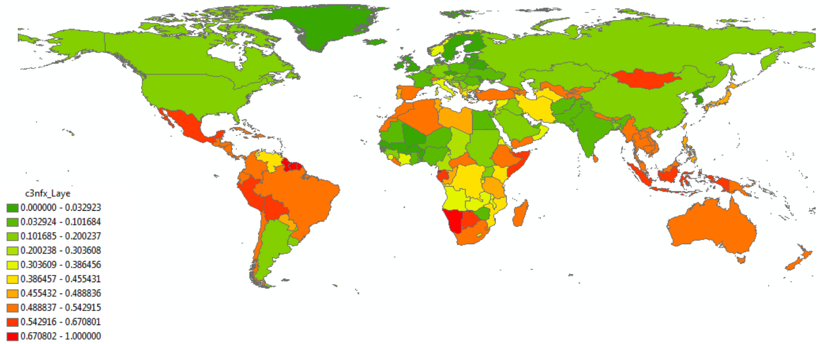

Figure 125: Proportion of C3 nitrogen-fixing crops in proactive or reactive risk areas

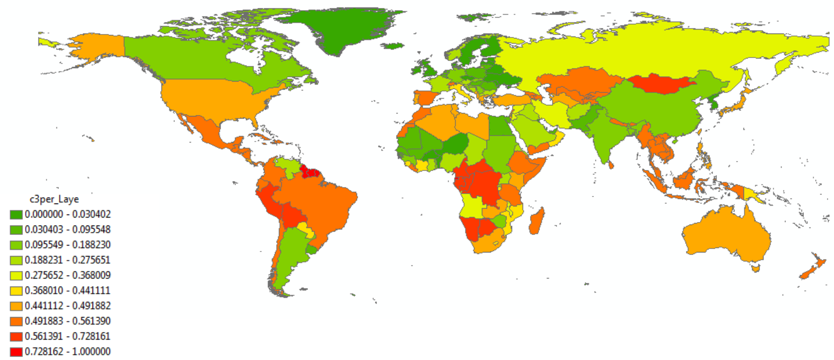

Figure 126: Proportion of C3 perennial crops in proactive or reactive risk areas

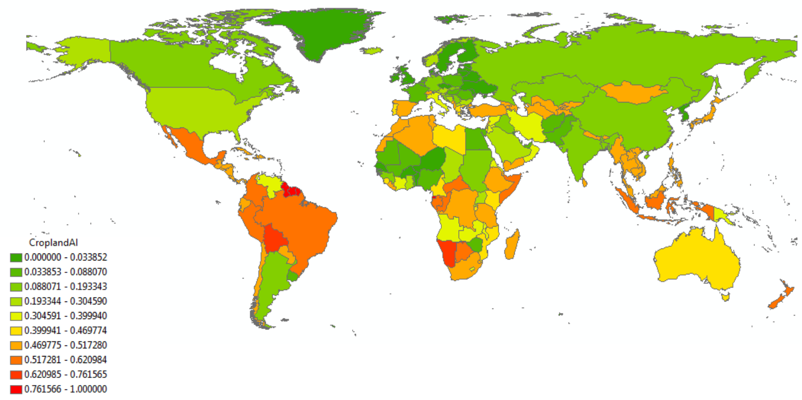

Figure 129: Proportion of cropland area in biodiversity risk areas

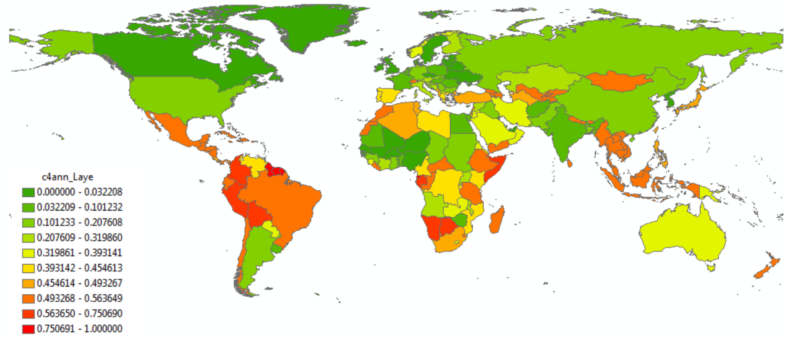

Figure 127: Proportion of C4 annual crops in proactive or reactive risk areas

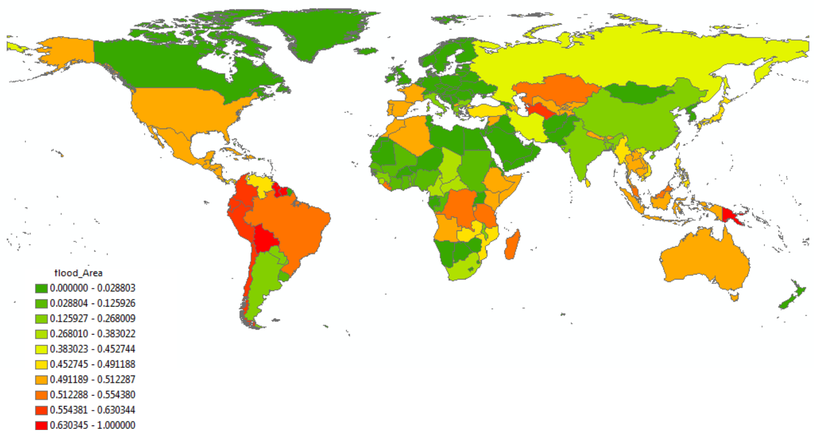

Figure 130: Proportion of flooded crop production area in biodiversity risk areas

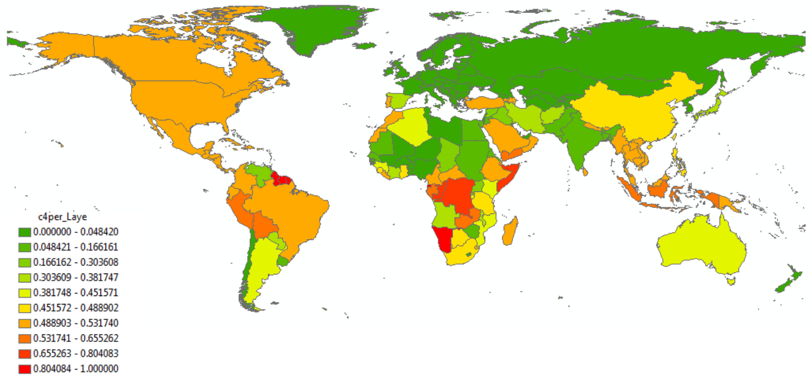

Figure 128: Proportion of C3 perennial Crops in proactive or reactive risk areas

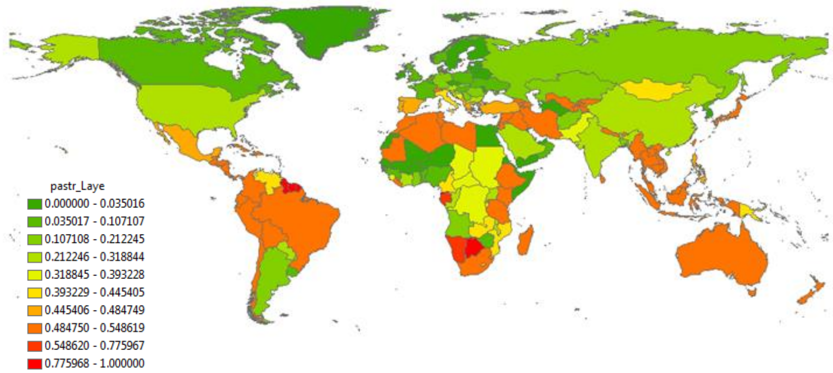

Figure 131: Proportion of managed pasture area in biodiversity risk areas

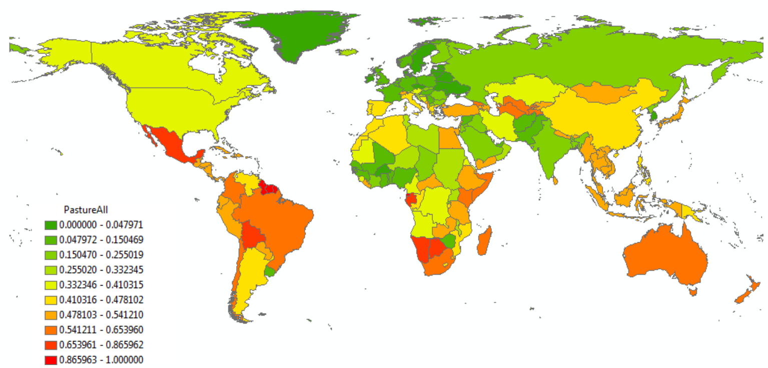

Figure 132: Proportion of pasture in biodiversity risk areas

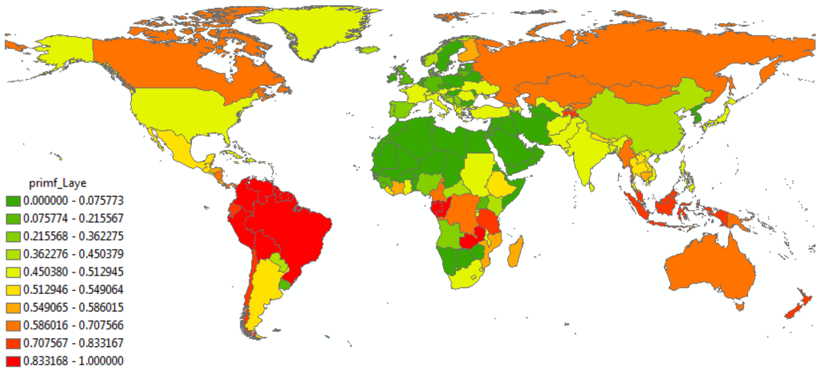

Figure 133: Proportion of primary forest in biodiversity risk areas

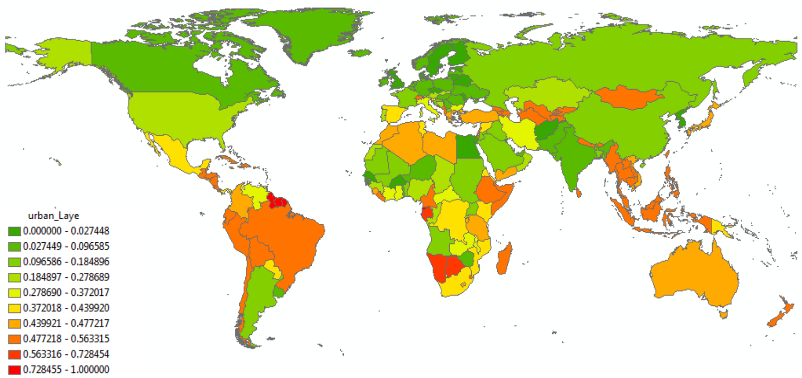

Figure 134: Proportion of urban area in biodiversity risk areas

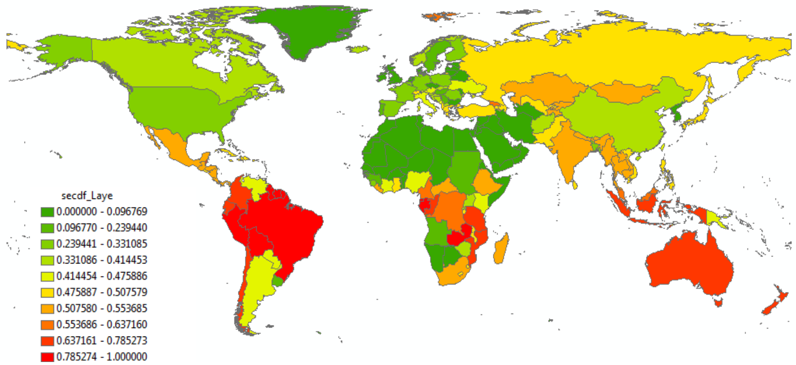

Figure 135: Proportion of secondary forest in biodiversity risk areas

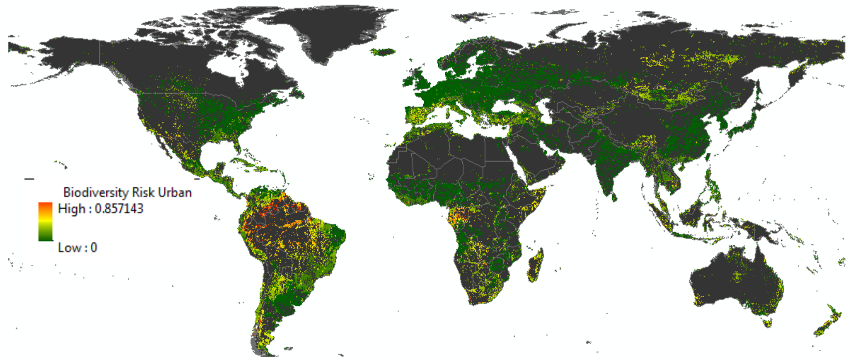

Figure 136: Biodiversity risk urban foreground system

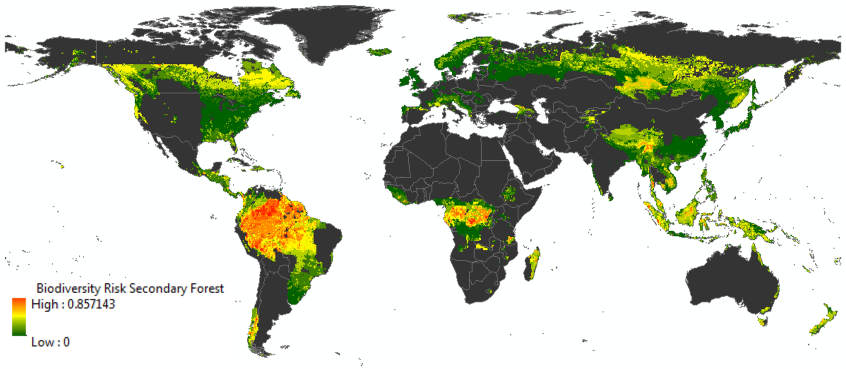

Figure 137: Biodiversity risk secondary forest foreground system

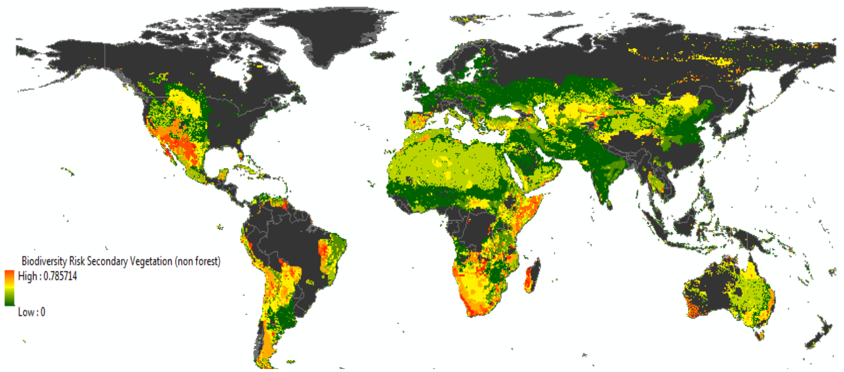

Figure 138: Biodiversity risk secondary vegetation foreground system

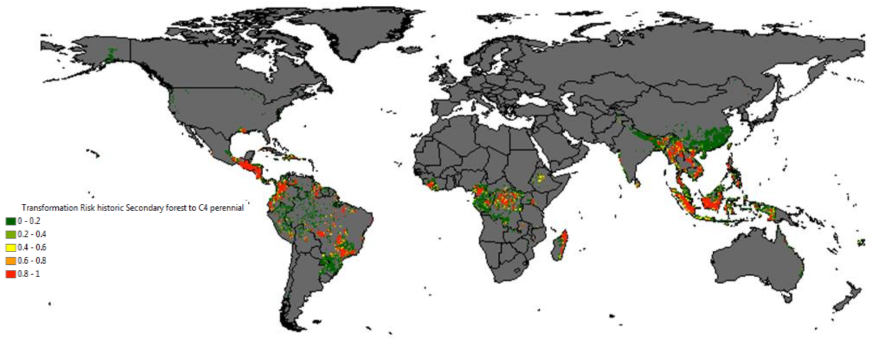

Figure 141: Biodiversity risk for transformation secondary forest to C4 perennial crops (historic)

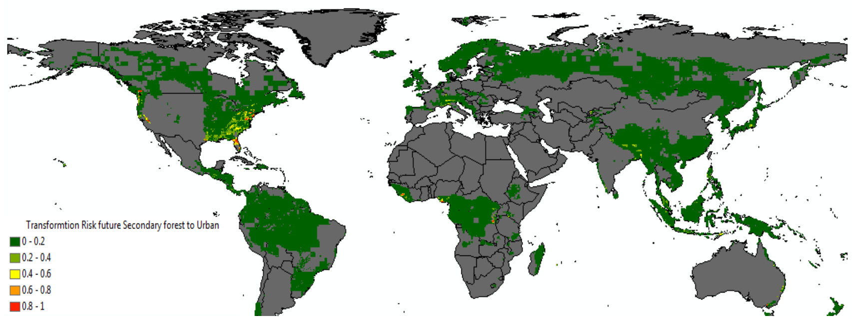

Figure 139: Biodiversity risk for transformation secondary forest to urban (future)

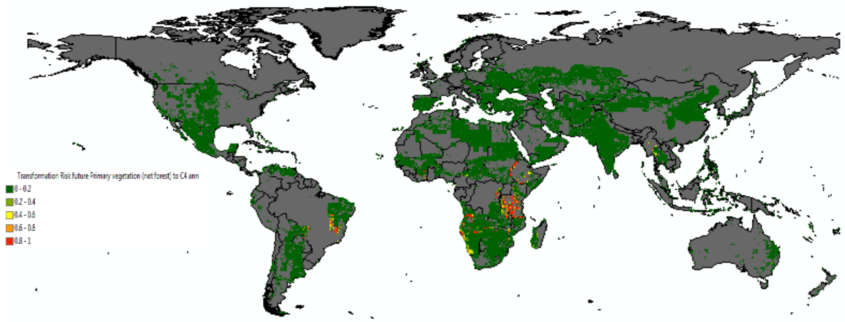

Figure 142: Biodiversity risk transformation primary vegetation (not forest) to C4 annual crops (future)

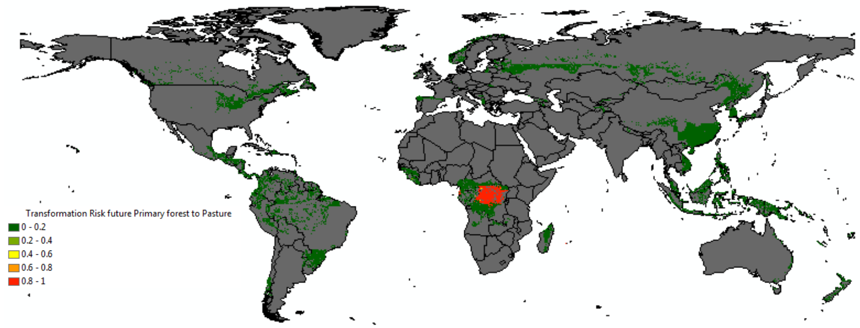

Figure 140: Biodiversity risk for transformation primary forest to urban (future)

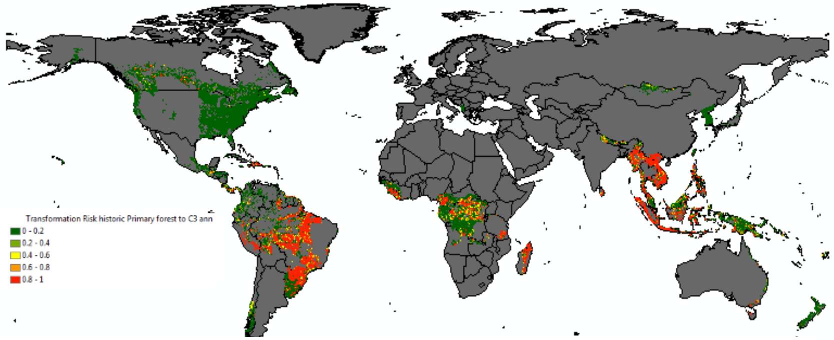

Figure 143: Biodiversity risk for transformation primary forest to C3 annual crops (historic)

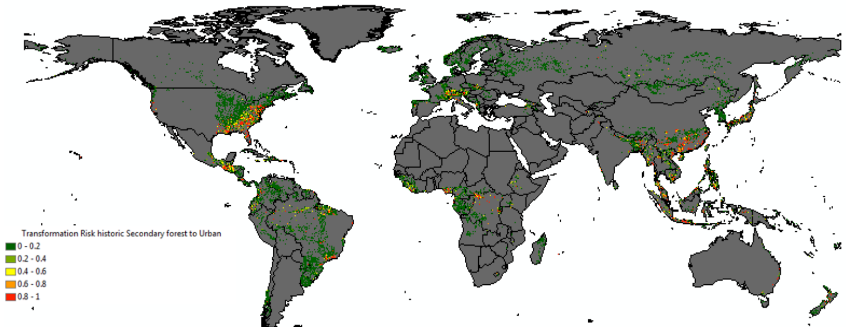

Figure 144: Biodiversity risk for transformation secondary forest to urban (historic)

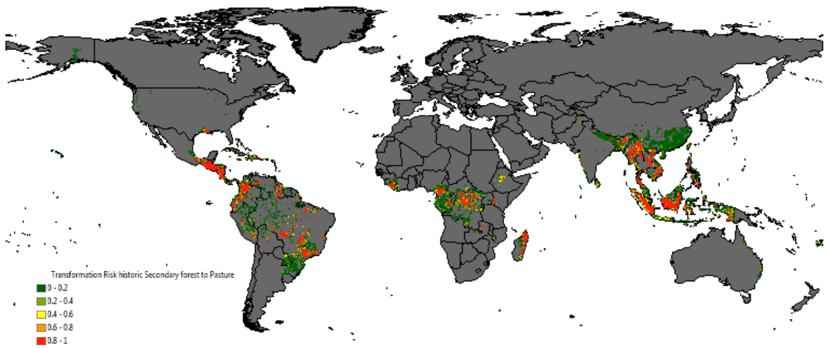

Figure 145: Biodiversity risk for transformation secondary forest to pasture (historic)

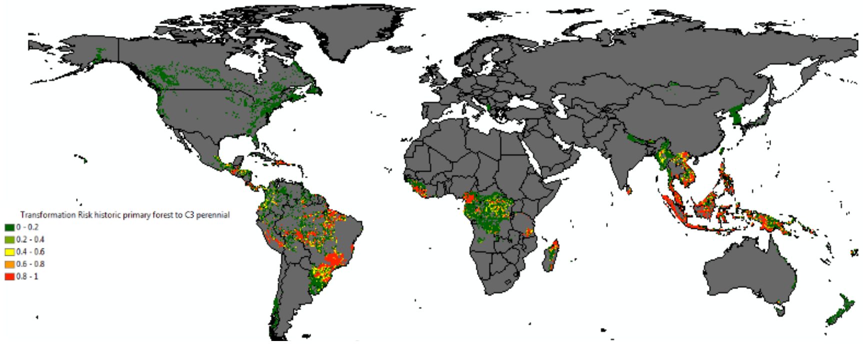

Figure 146: Biodiversity risk for transformation primary forest to C3 perennial (historic)

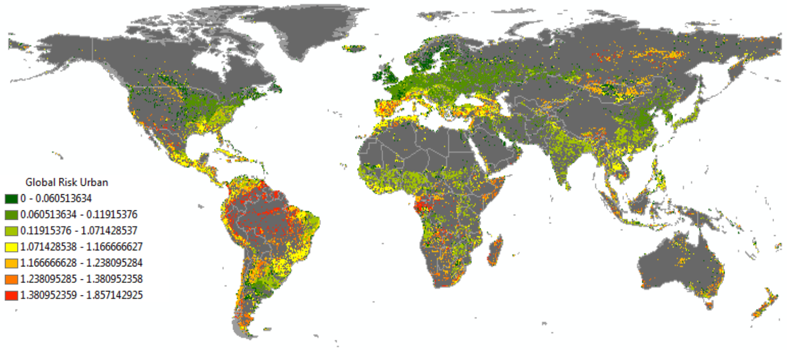

Figure 149: Global risk factor urban

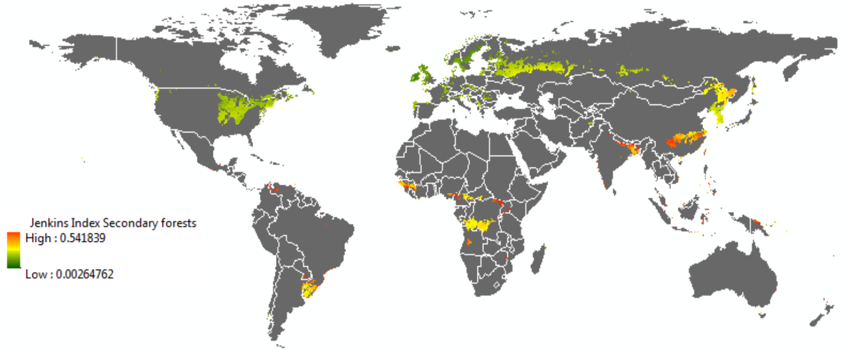

Figure 147: Jenkins Index for secondary forests

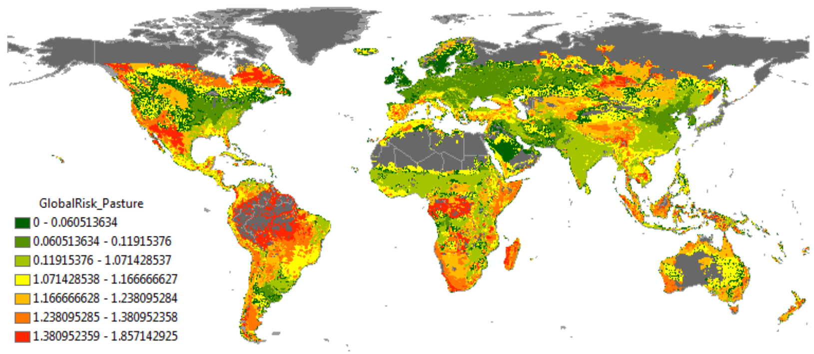

Figure 150: Global risk factor pasture

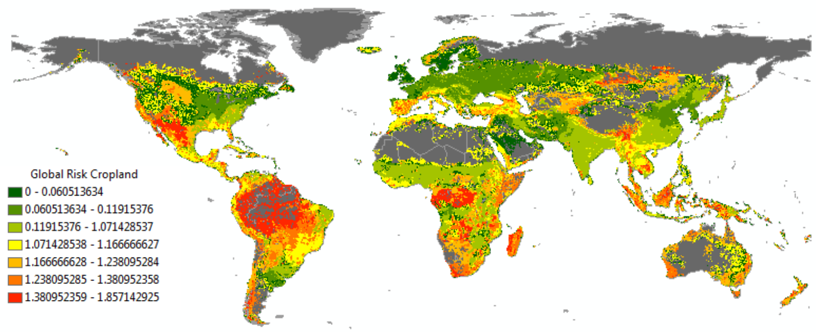

Figure 148: Global risk factor cropland

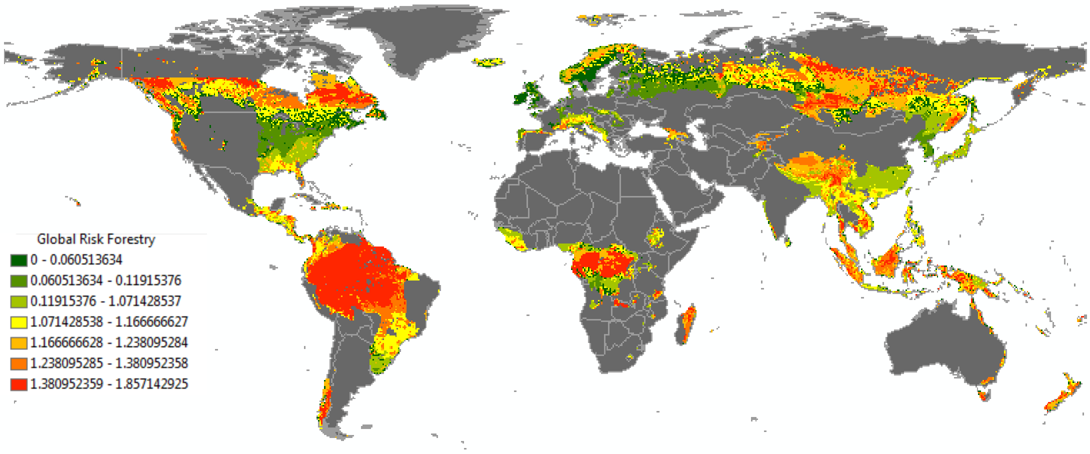

Figure 151: Global risk factor forest

10.3 Annex III: Land use specific biodiversity risks for transformation

Table 30: Biodiversity quality for local land use impact for transformation

Land use flow	Average richness	Average abundance	Average rarefaction based richness	Average similarity (Jaccard richness)	Average similarity (Jaccard abundance)	Average BR Index
Primf_to_Secdf	-6.0	-3.5	-5.8	-1.3	-3.0	-5.1
Primf_to_Secdf Minimal	-7.7	-3.9	-6.0			-5.9
Primf_to_Secdf Light/Intense use	-4.3	-3.1	-5.6			-4.3
Primf_to_Cropland	-33.8	-29.0	-24.0	-3.2	-6.6	-28.9
Primf_to_Cropland Minimal	-26.9	-10.6	-22.5			-20.0
Primf_to_Cropland Light	-38.1	-45.1	-20.9			-34.7
Primf_to_Cropland Intense	-36.3	-31.3	-28.5			-32.0
Primf_to_Pasture	-29.4	-22.5	-17.4	-5.1	-10.1	-23.1
Primf_to_Pasture Minimal	-21.8	-4.8	-10.3			-12.3
Primf_to_Pasture Light	-29.4	-27.8	-17.8			-25.0
Primf_to_Pasture Intense	-37.1	-34.9	-24.0			-32.0
Primf_to_Plantation	-28.5	-4.4	-23.3			-18.7
Primf_to_Plantation Minimal	-19.2	13.4	-11.8			-5.9
Primf_to_Plantation Light	-26.9	-22.2	-14.8			-21.3
Primf_to_Plantation Intense	-39.4	-4.3	-43.4			-29.0
Primf_to_Urban	-29.6	-41.8	-12.0	-3.1	-6.7	-27.8
Primf_to_Urban Minimal	-4.0	-18.2	9.7			-4.2
Primf_to_Urban Light	-34.7	-44.9	-16.8			-32.1
Primf_to_Urban Intense	-50.2	-62.4	-28.9			-47.2
Secdf_to_Cropland	-27.8	-25.5	-18.2	-1.9	-3.6	-23.8
SecdfMin_to_Cropland Min	-19.2	-6.7	-16.5			-14.1
SecdfMin_to_Cropland Light	-30.4	-41.2	-14.9			-28.8
SecdfMin_to_Cropland Inten	-28.6	-27.4	-22.5			-26.2
SecdfLight/Inten_to_Cropland Min	-22.6	-7.5	-16.9			-15.7
SecdfLight/Inten_to_Cropland Light	-33.8	-42.0	-15.3			-30.4
SecdfLight/Inten_to_Cropland Inten	-32.0	-28.2	-22.9			-27.7
Secdf_to_Pasture	-23.4	-19.0	-11.6	-3.8	-7.1	-18.0
SecdfMin_to_Pasture Min	-14.1	-0.9	-4.3			-6.4
SecdfMin_to_Pasture Light	-21.7	-23.9	-11.8			-19.1
SecdfMin_to_Pasture Inten	-29.4	-31.0	-18.0			-26.1
SecdfLight/Inten_to_Pasture Min	-17.5	-1.7	-4.7			-8.0

Land use flow	Average richness	Average abundance	Average rarefaction based richness	Average similarity (Jaccard richness)	Average similarity (Jaccard abundance)	Average BR Index
SecdfLight/Inten_to_Pasture Light	-25.1	-24.7	-12.2			-20.7
SecdfLight/Inten_to_Pasture Inten	-32.8	-31.8	-18.4			-27.7
Secdf_to_Plantation	-22.5	-0.8	-17.5			-13.6
SecdfMin_to_Plantation Min	-11.5	17.3	-5.8			0.0
SecdfMin_to_Plantation Light	-19.2	-18.3	-8.8			-15.4
SecdfMin_to_Plantation Inten	-31.7	-0.4	-37.4			-23.2
SecdfLight/Inten_to_Plantation Min	-14.9	16.5	-6.2			-1.5
SecdfLight/Inten_to_Plantation Light	-22.6	-19.1	-9.2			-17.0
SecdfLight/Inten_to_Plantation Inten	-35.1	-1.2	-37.8			-24.7
Secdf_to_Urban	-24.2	-38.4	-6.3	-1.8	-3.7	-23.0
SecdfMin_to_Urban Min	3.7	-14.3	15.7			1.7
SecdfMin_to_Urban Light	-27.0	-41.0	-10.8			-26.3
SecdfMin_to_Urban Inten	-45.9	-59.3	-23.3			-42.8
SecdfLight/Inten_to_Urban Min	0.3	-15.1	15.3			0.2
SecdfLight/Inten_to_Urban Light	-30.4	-41.8	-11.2			-27.8
SecdfLight/Inten_to_Urban Inten	-45.9	-59.3	-23.3			-42.8
Cropland_to_Secdf	27.8	25.5	18.2	1.9	3.6	23.8
Cropland Min_to_Secdf Min	19.2	6.7	16.5			14.1
Cropland Min_to_Secdf Light/Inten	22.6	7.5	16.9			15.7
Cropland Light_to_Secdf Min	30.4	41.2	14.9			28.8
Cropland Light_to_Secdf Light/Inten	33.8	42.0	15.3			30.4
Cropland Inten_to_Secdf Min	28.6	27.4	22.5			26.2
Cropland Inten_to_Secdf Light/Inten	32.0	28.2	22.9			27.7
Cropland_to_Pasture	4.3	6.5	6.6	-1.9	-3.5	5.8
Cropland Min_to_Pasture Min	5.1	5.8	12.2			7.7
Cropland Min_to_Pasture Light	-2.5	-17.2	4.7			-5.0
Cropland Min_to_Pasture Inten	-10.2	-24.3	-1.5			-12.0
Cropland Light_to_Pasture Min	16.3	40.3	10.6			22.4
Cropland Light_to_Pasture Light	8.7	17.3	3.1			9.7
Cropland Light_to_Pasture Inten	1.0	10.2	-3.1			2.7
Cropland Inten_to_Pasture Min	14.5	26.5	18.2			19.7
Cropland Inten_to_Pasture Light	6.9	3.5	10.7			7.0

Land use flow	Average richness	Average abundance	Average rarefaction based richness	Average similarity (Jaccard richness)	Average similarity (Jaccard abundance)	Average BR Index
Cropland Inten_to_Pasture Inten	-0.8	-3.6	4.5			0.0
Cropland_to_Plantation	4.3	22.7	0.3			9.1
Cropland Min_to_Plantation Min	7.7	24.0	10.7			14.1
Cropland Min_to_Plantation Light	0.0	-11.6	7.7			-1.3
Cropland Min_to_Plantation Inten	-12.5	6.3	-20.9			-9.0
Cropland Light_to_Plantation Min	18.9	58.5	9.1			28.8
Cropland Light_to_Plantation Light	11.2	22.9	6.1			13.4
Cropland Light_to_Plantation Inten	-10.0	23.5	-25.6			-4.0
Cropland Inten_to_Plantation Min	17.1	44.7	16.7			26.2
Cropland Inten_to_Plantation Light	9.4	9.1	13.7			10.7
Cropland Inten_to_Plantation Inten	-3.1	27.0	-14.9			3.0
Cropland_to_Urban	4.1	-12.8	12.0	0.1	-0.2	1.1
Cropland Min_to_Urban Min	22.9	-7.6	32.2			15.8
Cropland Min_to_Urban Light	-7.8	-34.3	5.7			-12.1
Cropland Min_to_Urban Inten	-23.3	-51.8	-6.4			-27.2
Cropland Light_to_Urban Min	34.1	26.9	30.6			30.5
Cropland Light_to_Urban Light	3.4	0.2	4.1			2.6
Cropland Light_to_Urban Inten	-12.1	-17.3	-8.0			-12.5
Cropland Inten_to_Urban Min	32.3	13.1	38.2			27.9
Cropland Inten_to_Urban Light	1.6	-13.6	11.7			-0.1
Cropland Inten_to_Urban Inten	-13.9	-31.1	-0.4			-15.1
Pasture_to_Secdf	23.4	19.0	11.6	3.8	7.1	18.0
Pasture Min_to_Secdf Min	14.1	0.9	4.3			6.4
Pasture Min_to_Secdf Light/Inten	17.5	1.7	4.7			8.0
Pasture Light_to_Secdf Min	21.7	23.9	11.8			19.1
Pasture Light_to_Secdf Light/Inten	25.1	24.7	12.2			20.7
Pasture Inten_to_Secdf Min	29.4	31.0	18.0			26.1
Pasture Inten_to_Secdf Light/Inten	32.8	31.8	18.4			27.7
Pasture_to_Cropland	-4.3	-6.5	-6.6	1.9	3.5	-5.8
Pasture Min_to_Cropland Min	-5.1	-5.8	-12.2			-7.7
Pasture Min_to_Cropland Light	-16.3	-40.3	-10.6			-22.4
Pasture Min_to_Cropland Inten	-14.5	-26.5	-18.2			-19.7

Land use flow	Average richness	Average abundance	Average rarefaction based richness	Average similarity (Jaccard richness)	Average similarity (Jaccard abundance)	Average BR Index
Pasture Light_to_Cropland Min	2.5	17.2	-4.7			5.0
Pasture Light_to_Cropland Light	-8.7	-17.3	-3.1			-9.7
Pasture Light_to_Cropland Inten	-6.9	-3.5	-10.7			-7.0
Pasture Inten_to_Cropland Min	10.2	24.3	1.5			12.0
Pasture Inten_to_Cropland Light	-1.0	-10.2	3.1			-2.7
Pasture Inten_to_Cropland Inten	0.8	3.6	-4.5			0.0
Pasture_to_Plantation	0.9	18.1	-6.0			4.3
Pasture Min_to_Plantation Min	2.6	18.2	-1.5			6.4
Pasture Min_to_Plantation Light	-5.1	-17.4	-4.5			-9.0
Pasture Min_to_Plantation Inten	-17.6	0.5	-33.1			-16.7
Pasture Light_to_Plantation Min	10.2	41.2	6.0			19.1
Pasture Light_to_Plantation Light	2.5	5.6	3.0			3.7
Pasture Light_to_Plantation Inten	-10.0	23.5	-25.6			-4.0
Pasture Inten_to_Plantation Min	17.9	48.3	12.2			26.1
Pasture Inten_to_Plantation Light	10.2	12.7	9.2			10.7
Pasture Inten_to_Plantation Inten	-2.3	30.6	-19.4			3.0
Pasture_to_Urban	-0.2	-19.3	5.4	2.1	3.4	-4.7
Pasture Min_to_Urban Min	17.8	-13.4	20.0			8.1
Pasture Min_to_Urban Light	-12.9	-40.1	-6.5			-19.8
Pasture Min_to_Urban Inten	-28.4	-57.6	-18.6			-34.9
Pasture Light_to_Urban Min	25.4	9.6	27.5			20.8
Pasture Light_to_Urban Light	-5.3	-17.1	1.0			-7.1
Pasture Light_to_Urban Inten	-20.8	-34.6	-11.1			-22.2
Pasture Inten_to_Urban Min	33.1	16.7	33.7			27.8
Pasture Inten_to_Urban Light	2.4	-10.0	7.2			-0.1
Pasture Inten_to_Urban Inten	-13.1	-27.5	-4.9			-15.2
Plantation_to_Secdf	22.5	0.8	17.5			13.6
Plantation Min_to_Secdf Min	11.5	-17.3	5.8			0.0
Plantation Min_to_Secdf Light/Inten	14.9	-16.5	6.2			1.5
Plantation Light_to_Secdf Min	19.2	18.3	8.8			15.4
Plantation Light_to_Secdf Light/Inten	22.6	19.1	9.2			17.0
Plantation Inten_to_Secdf Min	31.7	0.4	37.4			23.2

Land use flow	Average richness	Average abundance	Average rarefaction based richness	Average similarity (Jaccard richness)	Average similarity (Jaccard abundance)	Average BR Index
Plantation Inten_to_Secdf Light/Inten	35.1	1.2	37.8			24.7
Plantation_to_Cropland	-3.9	-26.6	2.5			-9.3
Plantation Min_to_Cropland Min	-7.7	-24.0	-10.7			-14.1
Plantation Min_to_Cropland Light	-18.9	-58.5	-9.1			-28.8
Plantation Min_to_Cropland Inten	-17.1	-44.7	-16.7			-26.2
Plantation Light_to_Cropland Min	0.0	11.6	-7.7			1.3
Plantation Light_to_Cropland Light	-11.2	-22.9	-6.1			-13.4
Plantation Light_to_Cropland Inten	3.1	-27.0	14.9			-3.0
Plantation Inten_to_Cropland Min	12.5	-6.3	20.9			9.0
Plantation Inten_to_Cropland Light	1.3	-40.8	22.5			-5.7
Plantation Inten_to_Cropland Inten	3.1	-27.0	14.9			-3.0
Plantation_to_Pasture	-0.9	-18.1	6.0			-4.3
Plantation Min_to_Pasture Min	-2.6	-18.2	1.5			-6.4
Plantation Min_to_Pasture Light	-10.2	-41.2	-6.0			-19.1
Plantation Min_to_Pasture Inten	-17.9	-48.3	-12.2			-26.1
Plantation Light_to_Pasture Min	5.1	17.4	4.5			9.0
Plantation Light_to_Pasture Light	-2.5	-5.6	-3.0			-3.7
Plantation Light_to_Pasture Inten	-10.2	-12.7	-9.2			-10.7
Plantation Inten_to_Pasture Min	17.6	-0.5	33.1			16.7
Plantation Inten_to_Pasture Light	10.0	-23.5	25.6			4.0
Plantation Inten_to_Pasture Inten	2.3	-30.6	19.4			-3.0
Plantation_to_Urban	-1.1	-37.5	11.3			-9.1
Plantation Min_to_Urban Min	15.2	-31.6	21.5			1.7
Plantation Min_to_Urban Light	-15.5	-58.3	-5.0			-26.3
Plantation Min_to_Urban Inten	-31.0	-75.8	-17.1			-41.3
Plantation Light_to_Urban Min	22.9	4.0	24.5			17.1
Plantation Light_to_Urban Light	-7.8	-22.7	-2.0			-10.8
Plantation Light_to_Urban Inten	-23.3	-40.2	-14.1			-25.9
Plantation Inten_to_Urban Min	35.4	-13.9	53.1			24.9
Plantation Inten_to_Urban Light	4.7	-40.6	26.6			-3.1

Land use flow	Average richness	Average abundance	Average rarefaction based richness	Average similarity (Jaccard richness)	Average similarity (Jaccard abundance)	Average BR Index
Plantation Inten_to_Urban Inten	-10.8	-58.1	14.5			-18.1
Urban_to_Secdf	23.6	38.3	6.2	1.8	3.7	22.7
Urban Min_to_Secdf Min	-3.7	14.3	-15.7			-1.7
Urban Min_to_Secdf Light/Inten	-0.3	15.1	-15.3			-0.2
Urban Light_to_Secdf Min	27.0	41.0	10.8			26.3
Urban Light_to_Secdf Light/Inten	30.4	41.8	11.2			27.8
Urban Inten_to_Secdf Min	42.5	58.5	22.9			41.3
Urban Inten_to_Secdf Light/Inten	45.9	59.3	23.3			42.8
Urban_to_Cropland	-4.1	12.8	-12.0	-0.1	0.2	-1.1
Urban Min_to_Cropland Min	-22.9	7.6	-32.2			-15.8
Urban Min_to_Cropland Light	-34.1	-26.9	-30.6			-30.5
Urban Min_to_Cropland Inten	-32.3	-13.1	-38.2			-27.9
Urban Light_to_Cropland Min	7.8	34.3	-5.7			12.1
Urban Light_to_Cropland Light	-3.4	-0.2	-4.1			-2.6
Urban Light_to_Cropland Inten	-1.6	13.6	-11.7			0.1
Urban Inten_to_Cropland Min	23.3	51.8	6.4			27.2
Urban Inten_to_Cropland Light	12.1	17.3	8.0			12.5
Urban Inten_to_Cropland Inten	13.9	31.1	0.4			15.1
Urban_to_Pasture	0.2	19.3	-5.4	-2.1	-3.4	4.7
Urban Min_to_Pasture Min	-17.8	13.4	-20.0			-8.1
Urban Min_to_Pasture Light	-25.4	-9.6	-27.5			-20.8
Urban Min_to_Pasture Inten	-33.1	-16.7	-33.7			-27.8
Urban Light_to_Pasture Min	12.9	40.1	6.5			19.8
Urban Light_to_Pasture Light	5.3	17.1	-1.0			7.1
Urban Light_to_Pasture Inten	-2.4	10.0	-7.2			0.1
Urban Inten_to_Pasture Min	28.4	57.6	18.6			34.9
Urban Inten_to_Pasture Light	20.8	34.6	11.1			22.2
Urban Inten_to_Pasture Inten	13.1	27.5	4.9			15.2
Urban_to_Plantation	1.1	37.5	-11.3			9.1
Urban Min_to_Plantation Min	-15.2	31.6	-21.5			-1.7
Urban Min_to_Plantation Light	-22.9	-4.0	-24.5			-17.1
Urban Min_to_Plantation Inten	-35.4	13.9	-53.1			-24.9
Urban Light_to_Plantation Min	15.5	58.3	5.0			26.3
Urban Light_to_Plantation Light	7.8	22.7	2.0			10.8
Urban Light_to_Plantation Inten	-4.7	40.6	-26.6			3.1
Urban Inten_to_Plantation Min	31.0	75.8	17.1			41.3
Urban Inten_to_Plantation Light	23.3	40.2	14.1			25.9
Urban Inten_to_Plantation Inten	10.8	58.1	-14.5			18.1

10.4 Annex IV: Management Parameter and activities for each land use type

Table 31: Management parameters for the land use type cropland

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
Artificial Nesting/resting/foraging sites	Create artificial hibernacula or aestivation sites	Likely to be beneficial	50%	*Amphibians need damp sheltered places for overwintering or aestivating during hot arid summers. Overwintering or aestivating sites, or 'hibernacula', can be created for amphibians where natural sites are limited or where these habitats have been lost, for example at newly restored sites or in gardens.*
Artificial Nesting/resting/foraging sites	Provide food for vultures to reduce mortality from diclofenac	Likely to be beneficial	60%	*Vulture 'restaurants' have therefore been proposed as a method of reducing mortality until the drug is phased out, by providing a safe, uncontaminated source of food for the remaining vulture populations.*
Artificial Nesting/resting/foraging sites	Provide supplementary food for birds or mammals	Likely to be beneficial	90%	*This intervention may involve the provision of supplementary food for birds or mammals in farmland habitats, such food typically includes seeds. Providing supplementary food for farmland wildlife may be particularly important when food resources in the wider farmed environment are scarce.*
Crop diversity/crop rotation	Grow cover crops beneath the main crop (living mulches) or between crop rows	Likely to be beneficial	65%	*Grow (winter) cover crop instead of fallow: e.g. leguminous and grass covers*
Crop diversity/crop rotation	Undersow spring cereals, with clover for example	Likely to be beneficial	60%	*This intervention involves sowing grass or clover beneath a cereal crop. The undersown crop is later ploughed in.*
Crop diversity/crop rotation	Grow cover crops when the field is empty	Beneficial	75%	*Grow (winter) cover crop instead of fallow: e.g. leguminous and grass covers*
Crop diversity/crop rotation	Use crop rotation	Beneficial	66%	*Grow and rotate different crop types (e.g. heat Triticum aestivum (replaced with canola Brassica campestris), barley Hordeum vulgare (replaced with oats Avena sativa) and alfalfa)*

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
Crop diversity/crop rotation/pesticide	Grow non-crop plants that produce chemicals that attract natural enemies	Likely to be beneficial	68%	*This intervention involves growing non-crop plants which produce volatile chemicals (quickly evaporating scents or odours) that attract natural enemies, thereby encouraging the enemies to the main crop. Non-crop plants could be grown in field margins or interspersed into the main crop. Lab studies demonstrating an attractive effect of a plant species or variety to a natural enemy are also included.*
Fertilizer (mineral)	Amend the soil using a mix of organic and inorganic amendments	Beneficial	69%	*The use of chemicals in agricultural management, such as synthetic fertilizers, may have a detrimental effect on farmland biodiversity. Organic fertilizers such as farmyard manure (including green manure or crop residues), slurry and other composts provide an alternative to synthetic or mineral fertilizers. Mineral fertilizers are manufactured preparations including nitrogen N, phosphorous P or potassium K, or 'NPK'.*
Fertilizer (mineral)	Use organic rather than mineral fertilizers	Beneficial	100%	*The use of chemicals in agricultural management, such as synthetic fertilizers, may have a detrimental effect on farmland biodiversity. Organic fertilizers such as farmyard manure (including green manure or crop residues), slurry and other composts provide an alternative to synthetic or mineral fertilizers. Mineral fertilizers are manufactured preparations including nitrogen N, phosphorous P or potassium K, or 'NPK'.*
Fertilizer, pesticide, mechanization	Use organic farming instead of conventional farming	Likely to be beneficial	40%	*Organic farming is an agricultural system that excludes the use of synthetic fertilizers and pesticides and relies on techniques such as crop rotation, compost and biological pest control. Organic standards are strictly regulated in many countries prohibiting the use of chemicals and providing recommendations for management to conserve biodiversity. Organic farming may include combinations of several separate interventions*
Habitat, set-aside areas,	Leave cultivated, uncropped margins or plots (includes 'lapwing plots')	Beneficial	100%	*uncropped, cultivated margins or plots (including one replicated, randomized, controlled trial) found benefits to some or all target farmland bird species, plants, invertebrates or mammals.*

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
Habitat, set-aside areas, connectivity, LDI	Create 'skylark plots' (undrilled patches in cereal fields)	Beneficial	100%	*Eurasian skylarks Alauda arvensis require short vegetation to nest in. Skylark plots are small (usually 4-16 m²) undrilled patches within cereal fields which provide this short vegetation, with little impact on overall productivity. They are similar to lapwing plots (see 'Leave uncropped, cultivated margins or plots (includes 'lapwing plots')) but much smaller.*
Habitat, set-aside areas, connectivity,	Leave uncut rye grass in silage fields	Likely to be beneficial	67%	*Seed-eating birds were benefited by leaving uncut (or once-cut) rye grass in fields, or that seed-eating species were more abundant on uncut plots. Seed-eating birds were more abundant on uncut and ungrazed plots than on uncut and grazed plots*
Habitat, set-aside areas, connectivity, LDI	Create beetle banks	Likely to be beneficial	80%	*Beetle banks are grassy mounds, about 2 m-wide, that run across the middle of large arable fields. They may be created using two-directional ploughing and sown with a mix of grass species (HGCA 2008). They are intended to provide habitat, especially during winter, for predatory insects such as beetles and spiders. They may also provide foraging habitats for birds and habitat for small mammals.*
Habitat, set-aside areas, connectivity, LDI	Create refuges	Likely to be beneficial	45%	*Refuge habitats can provide amphibians with microclimates to keep them at the correct temperature and prevent them from dehydrating and can protect them from predation. Many amphibians seek shelter in rocks, logs or other refuges created by tree falls and other disturbances. Refuges can be created for amphibians where natural shelter habitat is limited, or to replace these habitats where they have been lost.*
Habitat, set-aside areas, connectivity, LDI	Create uncultivated margins around intensive arable or pasture fields	Beneficial	100%	*This intervention allows the field margin vegetation to regenerate naturally, without planting, although it can involve subsequent mowing. The field margins are not fertilized and only spot-treated with herbicides if injurious weeds occur.*
Habitat, set-aside areas, connectivity, LDI	Leave refuges in fields during harvest	Likely to be beneficial	50%	*During mowing and harvesting operations, ground-nesting birds frequently remain in long grass or crops for as long as possible. If mowing/harvest occurs from the outside of the field inwards, this behaviour can leave the birds trapped in the centre of the field and killed as the last patch is harvested. However, if unharvested refuges are left in fields then it is possible that chicks and adults will remain in them and survive.*

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
Habitat, set-aside areas, connectivity, LDI	Leave uncropped cultivated margins or fallow land (includes lapwing and stone curlew plots)	Likely to be beneficial	59%	*Plots or strips are cultivated, but left undrilled. For ground-nesting birds, the plots are usually at least 2 ha in size. They are different from 'skylark plots', which are much smaller and usually created in groups. If this measure is taken in field margins (6 m strips at the edge of arable fields), no fertilizer is applied, and herbicide applications are minimal, with only spot treatment of particular weeds permitted.*
Habitat, set-aside areas, connectivity, LDI	Manage hedgerows to benefit wildlife (includes no spray, gap-filling and laying)	Likely to be beneficial	70%	*Hedges can be key habitats for farmland biodiversity, but they may need managing to maximize their value. Managing hedges to benefit wildlife involves one or more of the following management changes: reduce cutting frequency; reduce or avoid spraying; mow vegetation beneath hedgerows; fill gaps in hedges; coppice or lay to restore traditional hedge structure.*
Habitat, set-aside areas, connectivity, LDI	Plant grass buffer strips/margins around arable or pasture fields	Beneficial	90%	*This intervention involves planting field margins with a grass-rich seed mixture. It includes ‘floristically-enhanced’ grass margins available under the English Higher Level Stewardship scheme. The margins are not fertilized and only spot-treated with herbicides if necessary.*
Habitat, set-aside areas, connectivity, LDI	Plant nectar flower mixture/wildflower strips	Beneficial	100%	*Flowering plants are sown in strips or blocks, providing forage resources for bees and other flower-visiting insects. Increased insect numbers may then provide food for more birds. Nectar flower mixture may include agricultural varieties of flowering plants such as clovers.*
Habitat, set-aside areas, connectivity, LDI	Plant wild bird seed or cover mixture	Beneficial	100%	*The loss of food supplies, especially seeds, is thought to be a key driver of farmland bird declines. Plants that provide seed food and cover for wild birds include corn, sunflower and cereals. Wild bird cover crops are often planted in blocks or 6 m-wide strips and left unharvested. These are sometimes called ‘game crops’ or ‘game cover crops’. They may also provide benefits for other farmland wildlife.*
Habitat, set-aside areas, connectivity, LDI	Pollination: Plant hedgerows	Beneficial	78%	*Plant hedgerows next to fields*
Habitat, set-aside areas, connectivity, LDI	Provide (or retain) set-aside areas in farmland	Beneficial	90%	*Set-aside can be rotational (in a different place every year or two) or non-rotational (same place for 5-20 years) and fields can either be sown with fallow crops or left to naturally regenerate. Unlike fallow land left*

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
				for the benefit of ground-nesting birds or arable plants, set-aside is not ploughed or harrowed except for the purpose of sowing. *Set-aside is often managed by cutting and/or spraying. In some cases, set-aside land has had strips of wildflowers or grasses sown on it. Evidence for the effects of this management has been included under the following interventions: 'Plant nectar flower mixture wildflower strips' and 'Plant grass buffer strips margins around arable or pasture fields'.*
Habitat, set-aside areas, connectivity, LDI	Replant vegetation	Beneficial	70%	*Vegetation can be replanted to replace habitat that has been lost.*
Habitat, set-aside areas, connectivity, LDI	Restore habitat connectivity	Likely to be beneficial	75%	*Habitat destruction and fragmentation are important factors in the decline of amphibian populations. Small patches of habitat support smaller populations and if individuals are unable to move to other suitable areas, populations become isolated. This can make them more vulnerable to extinction. Restoring corridors of native vegetation between patches of suitable habitat may help to maintain amphibian populations.*
Habitat, set-aside areas, connectivity, LDI	Create/protect habitat corridors	Likely to be beneficial	65%	*Corridors are areas of natural habitat that are contiguous or isolated (i.e. linkages or stepping stones) and enable particular plant and animal species to disperse and migrate, processes which are necessary for their survival (Rouget et al. 2006).*
Habitat, set-aside areas, connectivity, LDI	Leave headlands in fields unsprayed (conservation headlands)	Beneficial	90%	*Conservation headland management involves restricted fertilizer, herbicide and insecticide spraying in a 6 m margin of sown arable crop. The prescription allows selected herbicide applications to control injurious weeds or invasive alien species.*
Mechanization	Control traffic and traffic timing	Likely to be beneficial	55%	*Control farm traffic intensities (e.g. of tractors) to reduce soil compaction and benefit below-ground biodiversity*
Mechanization	Reduce tillage	Likely to be beneficial	60%	*Conventional ploughing uses a mould-board plough, cultivating to a depth of around 20 cm. This intervention includes various methods to reduce the depth or intensity of ploughing, such as layered cultivation, non-inversion tillage and conservation tillage. It also includes stopping tillage altogether in some areas.*

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
Mechanization	Use mowing techniques to reduce mortality	Likely to be beneficial	100%	*Mowing and harvesting operations may have a negative impact on farmland wildlife. This intervention involves using different mowing machinery or mowing patterns to reduce the impact of mowing on field-dwelling animals. Adjusting mowing techniques may benefit ground-nesting birds that frequently remain in long grass or crops for as long as possible; mowing from the centre of the field outwards, rather than from the field edge inwards, may allow birds to escape the mowing machinery before the last patch of long grass is harvested.*
Mechanization/Other	Leave overwinter stubbles	Likely to be beneficial	90%	*This intervention involves leaving crop stubbles in fields until at least February-March. These stubbles may provide an important food source for seed-eating birds over the winter (Campaign for the Farmed Environment 2011). For Eurasian skylarks Alauda arvensis, approximately 0.1 km^2 of stubble/km^2 would be needed to prevent population declines. The authors also suggest that having these patches over 1 km apart would maximize winter use.*
Mechanization/Other	Relocate nests at harvest time to reduce nestling mortality	Likely to be beneficial	55%	*If nests are likely to be destroyed by machinery during harvest or mowing, it may be possible to move them and then return them after the danger has passed. If nests are extremely likely to be destroyed during harvest or mowing then it may be best to remove the chicks and hand-rear them.*
Native vegetation, plant diversity	Retain or plant native trees and shrubs amongst crops (agroforestry)	Likely to be beneficial	55%	*This intervention involves growing crops under shade trees that are either native tree species that are remnants from cleared vegetation, or other crop trees (often referred to as 'agroforestry'). This approach provides a more complex habitat than conventional monoculture farming and can support higher levels of biodiversity.*
Other	Clear vegetation	Likely to be beneficial	60%	*Vegetation can be removed to prevent natural succession where specific habitat types are desired, or where invasive species are out-competing native species for example.*
Other	Amend the soil with formulated chemical compounds	Likely to be beneficial	64%	*Ammonium-N is often added to soils as a fertilizer for crops. When added to the soil, soil bacteria convert it to nitrate (nitrification). If plants do not take up the ammonium-N immediately it gets converted to nitrate and can be lost from the soil (leaching). A nitrification inhibitor stops or slows down this conversion, and can reduce the loss of nitrate from the soil.*

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
Other	Sow crops in spring rather than autumn	Likely to be beneficial	55%	*Changes in farming practice in northern Europe have included a shift from sowing crops in spring to sowing them the preceding autumn/winter. This change is considered to have adversely affected farmland biodiversity including invertebrates and farmland birds*
Pesticides	Grow plants that compete with damaging weeds	Likely to be beneficial	70%	*This intervention involves planting species that out-compete damaging weeds, suppressing them by reducing their ground cover, growth or reproduction rate, or by increasing their mortality. The intervention is generally applied to pastureland or uncropped areas such as field margins and buffer strips.*
Pesticides	Use chemicals to attract natural enemies	Likely to be beneficial	40%	*This involves using chemicals to lure natural enemies into a crop. Communication chemicals of insects and plants (known as pheromones and volatiles, respectively) can be manufactured and deployed to manipulate invertebrates. Examples include the volatiles produced when plants are attacked by pests (e.g. methyl salicylate) and the alarm and sex pheromones of pests or natural enemies, as well as organic extracts from crop or plant leaves. Chemicals are sprayed onto crops or deployed in dispensers placed at regular intervals in the crop. Many studies have tested the efficacy of chemicals by applying them as baits in insect traps such as delta traps (plastic structures hung from branches or posts containing a sheet of sticky paper).*
Pesticides, crop diversity	Combine trap and repellent crops in a push-pull system	Beneficial	70%	*Push-pull systems involve intercropping the main crop with plants that are repellent to pests (the 'push') while also growing plants (trap crops) that are attractive to pests around the main crop (the 'pull'). This combination of repellent and attractive companion plants keeps invertebrate pests away from the crop and may provide additional benefits through improved habitat and resources for natural enemies. Push-pull systems can also be designed to suppress weeds at the same time as controlling pests.*
Pesticides, fertilizer	Reduce fertilizer, pesticide or herbicide use generally	Beneficial	100%	*Pesticide, herbicide and fertilizer applications may have a negative impact on farmland wildlife. This intervention may involve reducing or ceasing applications of pesticides (such as insecticides, fungicides), herbicides and fertilizers.*

Cropland				
Management Parameter	Management activity [198]	Category [198]	Effectivness [198]	Description from [198]
Scarers	Reduce conflict by deterring birds from taking crops: use bird scarers	Likely to be beneficial	66%	*In some parts of the world, the persecution of birds that take crops can be a serious threat to the survival of populations. Methods to reduce the damage done by birds can therefore be important in reducing the pressure on populations.*
Wetland, LDI	Manage ditches to benefit wildlife	Likely to be beneficial	40%	*Managing ditches to benefit wildlife can involve reduced or delayed cutting of vegetation on ditch banks and restricted fertilizer, herbicide or pesticide use on ditch banks or in fields adjoining ditches. 'Bundled' ditches are blocked to allow them to fill with water.*
Wetland, LDI	Regulate water levels (maintain pond water levels)	Beneficial	70%	*Drying of amphibian breeding sites before terrestrial life stages have developed can have significant detrimental effects on populations. In some cases it may be possible to maintain water levels until after metamorphosis by using a local water source or by bringing in water from an outside source. Occasional drying of breeding sites can increase diversity, as it can help control predators, non-native species or more dominant species.*
Wetland, LDI	Restore or create wetlands/ and marine habitats (coastal and intertidal wetlands)	Beneficial/Likely to be beneficial	80%/65%	*Wetland habitats are often drained or degraded during the development of agriculture or expansion of urban areas or other land uses. Restoration of these important amphibian habitats can help to increase local amphibian species richness and abundance.*
Wetland, LDI	Create scrapes and pools in wetlands and wet grasslands	Likely to be beneficial	75%	*Creating scrapes and pools in wetlands and wet grasslands can help create a heterogeneous habitat, with varying vegetation types and water levels.*

Table 32: Management parameters for the land use type pasture

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Artificial nesting/resting/foraging sites	Create artificial hibernacula or aestivation sites	Likely to be beneficial	50%	*Amphibians need damp sheltered places for overwintering or aestivating during hot arid summers. Overwintering or aestivating sites, or 'hibernacula', can be created for amphibians where natural sites are limited or where these habitats have been lost, for example at newly restored sites or in gardens.*
Artificial nesting/resting/foraging sites	Create refuges	Likely to be beneficial	45%	*Refuge habitats can provide amphibians with microclimates to keep them at the correct temperature and prevent them from dehydrating and can protect them from predation. Many amphibians seek shelter in rocks, logs or other refuges created by tree falls and other disturbances. Refuges can be created for amphibians where natural shelter habitat is limited, or to replace these habitats where they have been lost.*
Artificial nesting/resting/foraging sites	Provide food for vultures to reduce mortality from diclofenac	Likely to be beneficial	60%	*Vulture 'restaurants' have therefore been proposed as a method of reducing mortality until the drug is phased out, by providing a safe, uncontaminated source of food for the remaining vulture populations.*
Artificial nesting/resting/foraging sites	Provide supplementary food for birds or mammals	Likely to be beneficial	90%	*This intervention may involve the provision of supplementary food for birds or mammals in farmland habitats, such food typically includes seeds. Providing supplementary food for farmland wildlife may be particularly important when food resources in the wider farmed environment are scarce.*
Fertilizer	Use organic rather than mineral fertilizers	Beneficial	100%	*The use of chemicals in agricultural management, such as synthetic fertilizers, may have a detrimental effect on farmland biodiversity. Organic fertilizers such as farmyard manure (including green manure or crop residues), slurry and other composts provide an alternative to synthetic or mineral fertilizers. Mineral fertilizers are manufactured preparations including nitrogen N, phosphorous P or potassium K, or 'NPK'.*
Fertilizer Pesticide	Reduce fertilizer, pesticide or herbicide use generally	Beneficial/Likely to be beneficial	100%	*Pesticide, herbicide and fertilizer applications may have a negative impact on farmland wildlife. This intervention may involve reducing or ceasing applications of pesticides (such as insecticides, fungicides), herbicides and fertilizers.*
Fertilizer, Pesticide,	Reduce manage-	Likely to be beneficial	100%	*Reducing the intensity of grassland management involves reducing or stopping the use of fertilizers, herbicides and pesticides and delaying the mowing*

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Mechanization	ment intensity on permanent grasslands (several interventions at once)			*date until later in the summer. Studies included here have monitored the effects of carrying out two or more of these management interventions at the same time. All the studies included here monitor the effects of a change of grassland management, or of implementing a regime of reduced management intensity for the purpose of nature conservation, such as an agri-environment scheme.*
Habitat, set-aside areas, connectivity, LDI	Manage hedgerows to benefit wildlife (includes no spray, gap- filling and laying)	Likely to be beneficial	70%	*Hedges can be key habitats for farmland biodiversity, but they may need managing to maximize their value. Managing hedges to benefit wildlife involves one or more of the following management changes: reduce cutting frequency; reduce or avoid spraying; mow vegetation beneath hedgerows; fill gaps in hedges; coppice or lay to restore traditional hedge structure.*
Habitat, set-aside areas, connectivity, LDI	Leave headlands in fields unsprayed (conservation headlands)	Beneficial/Likely to be beneficial	90%	*Conservation headland management involves restricted fertilizer, herbicide and insecticide spraying in a 6 m margin of sown arable crop. The prescription allows selected herbicide applications to control injurious weeds or invasive alien species.*
Habitat, set-aside areas, connectivity, LDI	Leave refuges in fields during harvest	Likely to be beneficial	50%	*During mowing and harvesting operations, ground-nesting birds frequently remain in long grass or crops for as long as possible. If mowing/harvest occurs from the outside of the field inwards, this behaviour can leave the birds trapped in the centre of the field and killed as the last patch is harvested. However, if unharvested refuges are left in fields then it is possible that chicks and adults will remain in them and survive.*
Habitat, set-aside areas, connectivity, LDI	Restore or create species-rich, semi-natural grassland	Beneficial	100%	*Species-rich, semi-natural grasslands have declined drastically in Europe over the last 100 years (e.g. Poschlod & Bonn 1998, Eriksson et al. 2002) and conservation efforts have been directed to maintaining existing areas of these habitats. This includes hay meadows, litter meadows and other semi-natural pastures.*
Habitat, set-aside areas, connectivity,	Leave uncut rye grass in silage fields	Likely to be beneficial	67%	*Seed-eating birds were benefited by leaving uncut (or once-cut) rye grass in fields, or that seed-eating species were more abundant on uncut plots. Seed-eating birds were more abundant on uncut and ungrazed plots than on uncut and grazed plots.*

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Habitat, set-aside areas, connectivity, LDI	Plant grass buffer strips/margins around arable or pasture fields	Beneficial	90%	*This intervention involves planting field margins with a grass-rich seed mixture. It includes 'floristically-enhanced' grass margins available under the English Higher Level Stewardship scheme. The margins are not fertilized and only spot-treated with herbicides if necessary.*
Habitat, set-aside areas, connectivity, LDI	Restore or create grassland	Likely to be beneficial	45%	*species richness on restored grasslands was higher than unrestored habitats, or similar to remnant grassland, and three found that target species used restored grassland.*
Habitat, set-aside areas, connectivity, LDI	Plant nectar flower mixture/wildflower strips	Beneficial	100%	*Flowering plants are sown in strips or blocks, providing forage resources for bees and other flower-visiting insects. Increased insect numbers may then provide food for more birds. Nectar flower mixture may include agricultural varieties of flowering plants such as clovers.*
Habitat, set-aside areas, connectivity, LDI	Plant wild bird seed or cover mixture	Beneficial	100%	*The loss of food supplies, especially seeds, is thought to be a key driver of farmland bird declines. Plants that provide seed food and cover for wild birds include corn, sunflower and cereals. Wild bird cover crops are often planted in blocks or 6 m-wide strips and left unharvested. These are sometimes called 'game crops' or 'game cover crops'. They may also provide benefits for other farmland wildlife.*
Habitat, set-aside areas, connectivity, LDI	Provide or retain set-aside areas in farmland	Beneficial	90%	*Set-aside can be rotational (in a different place every year or two) or non-rotational (same place for 5-20 years) and fields can either be sown with fallow crops or left to naturally regenerate. Unlike fallow land left for the benefit of ground-nesting birds or arable plants, set-aside is not ploughed or harrowed except for the purpose of sowing.* *Set-aside is often managed by cutting and/or spraying. In some cases, set-aside land has had strips of wildflowers or grasses sown on it. Evidence for the effects of this management has been included under the following interventions: 'Plant nectar flower mixture wildflower strips' and 'Plant grass buffer strips margins around arable or pasture fields'.*
Habitat, set-aside areas, connectivity, LDI	Replant vegetation	Beneficial	70%	*Vegetation can be replanted to replace habitat that has been lost.*
Habitat, set-aside areas, connectivity, LDI	Restore habitat connectivity	Likely to be beneficial	75%	*Habitat destruction and fragmentation are important factors in the decline of amphibian populations. Small patches of habitat support smaller populations and if individuals are unable to move to other suitable areas, populations become isolated. This can*

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
				make them more vulnerable to extinction. Restoring corridors of native vegetation between patches of suitable habitat may help to maintain amphibian populations.
Habitat, set-aside areas, connectivity, LDI	Create/protect habitat corridors	Likely to be beneficial	65%	*Corridors are areas of natural habitat that are contiguous or isolated (i.e. linkages or stepping stones) and enable particular plant and animal species to disperse and migrate, processes which are necessary for their survival (Rouget et al. 2006).*
Habitat, set-aside areas, connectivity, LDI	Create uncultivated margins around intensive arable or pasture fields	Beneficial/Likely to be beneficial	100%	*This intervention allows the field margin vegetation to regenerate naturally, without planting, although it can involve subsequent mowing. The field margins are not fertilized and only spot-treated with herbicides if injurious weeds occur.*
Livestock intensity	Reduce grazing intensity	Likely to be beneficial	51%	*Livestock intensity leads to soil compaction. Overgrazing is responsible for the degradation of habitats across the world, being especially damaging in arid environments, where the removal of vegetation can quickly lead to soil erosion. Reducing grazing intensity may reduce the damage to vegetation and can also help reduce disturbance to birds and accidental loss of nests.*
Livestock intensity	Reduce number of livestock	Beneficial	65%	*Reducing grazing intensity by reducing livestock numbers may allow plants to grow taller and allow species that cannot tolerate heavy grazing to grow.*
Livestock management	Exclude or remove livestock from degraded peatlands	Likely to be beneficial	40%	*Domestic livestock directly consume peatland vegetation, destroy peatland vegetation by trampling, create bare patches of ground (e.g. repeatedly used tracks), and affect nutrient balance through excretion (Lindsay et al. 2014). Some plant groups or species, such as heather and dwarf shrubs, may be impacted more by selective grazing (Grant et al. 1987). Removing livestock could allow these species to recover to natural levels, although there is a risk that they become over-abundant when not grazed.*
Livestock management, Livestock intensity	Use fences to exclude livestock from shrublands	Likely to be beneficial	51%	*Livestock grazing can alter shrub land habitats by changing plant structure, composition, and diversity (Alkemade et al. 2013). High grazing pressure can reduce the cover of woody species in shrublands, leading to an increase in grass cover or unvegetated areas. Using fences to exclude livestock from shrublands could reduce the negative impacts of overgrazing.*

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Livestock management, Livestock intensity	Use wire fences within grazing areas to exclude livestock from specific forest sections	Likely to be beneficial	58%	*Livestock grazing changes habitats, mainly by changing soil properties, plant composition, structure and diversity (Alkemade et al. 2013). High grazing pressure can degrade understory species diversity. This is mainly due to decreasing the abundance of palatable herbaceous and low woody species. Using wire fences to excluded livestock from regularly grazed forested areas may increase species diversity (Crawley 1983).*
Mechanization	Delay mowing date on grasslands	Likely to be beneficial	60%	*Early-season, mechanised mowing is thought to be responsible for declines in the UK and elsewhere of species such as the corncrake Crex, with chicks killed and nests destroyed by mowing machinery. Delaying mowing until after chicks can escape is therefore a part of many agri-environment schemes.*
Mechanization	Relocate nests at harvest time to reduce nestling mortality	Likely to be beneficial	55%	*If nests are likely to be destroyed by machinery during harvest or mowing, it may be possible to move them and then return them after the danger has passed. If nests are extremely likely to be destroyed during harvest or mowing then it may be best to remove the chicks and hand-rear them.*
Mechanization	Use mowing techniques to reduce mortality	Beneficial/Likely to be beneficial	100%	*Mowing and harvesting operations may have a negative impact on farmland wildlife. This intervention involves using different mowing machinery or mowing patterns to reduce the impact of mowing on field-dwelling animals. Adjusting mowing techniques may benefit ground-nesting birds that frequently remain in long grass or crops for as long as possible; mowing from the centre of the field outwards, rather than from the field edge inwards, may allow birds to escape the mowing machinery before the last patch of long grass is harvested.*
Mechanization, Livestock intensity	Delay mowing or first grazing date on grasslands	Likely to be beneficial	60%	*This intervention involves delaying the first mowing or grazing date on grasslands. Early-season, mechanized mowing is thought to be responsible for declines in the UK and elsewhere of species such as the corncrake Crex, with chicks killed and nests destroyed by mowing machinery (Green & Gibbons 2000). Delaying mowing until after chicks can escape is therefore a part of many agri-environment schemes. Delaying mowing or grazing may also provide benefits to other farmland wildlife such as plants and invertebrates.*
Other	Clear vegetation	Likely to be beneficial	60%	*Vegetation can be removed to prevent natural succession where specific habitat types are desired, or where invasive species are out-competing native species for example.*

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Other	Plant cereals for whole crop silage	Likely to be beneficial	55%	*Cereal-based whole crop silage (CBWCS) is an intervention that involves growing crops, not grass, to turn into silage. This may provide seed resources for grain-eating farmland birds throughout the year.*
Other	Mark fencing to avoid bird mortality	Likely to be beneficial	65%	*Fewer birds collided with marked sections of deer fences, compared to unmarked sections.*
Pesticide	Grow non-crop plants that produce chemicals that attract natural enemies	Likely to be beneficial	68%	*This intervention involves growing non-crop plants which produce volatile chemicals (quickly evaporating scents or odours) that attract natural enemies, thereby encouraging the enemies to the main crop. Non-crop plants could be grown in field margins or interspersed into the main crop. Lab studies demonstrating an attractive effect of a plant species or variety to a natural enemy are also included.*
Pesticide	Use chemicals to attract natural enemies	Likely to be beneficial	40%	*This involves using chemicals to lure natural enemies into a crop. Communication chemicals of insects and plants (known as pheromones and volatiles, respectively) can be manufactured and deployed to manipulate invertebrates. Examples include the volatiles produced when plants are attacked by pests (e.g. methyl salicylate) and the alarm and sex pheromones of pests or natural enemies, as well as organic extracts from crop or plant leaves. Chemicals are sprayed onto crops or deployed in dispensers placed at regular intervals in the crop. Many studies have tested the efficacy of chemicals by applying them as baits in insect traps such as delta traps (plastic structures hung from branches or posts containing a sheet of sticky paper).*
Pesticide, Fertilizer	Reduce chemical inputs in grassland management	Likely to be beneficial	90%	*This intervention may involve reducing the amount of chemical inputs applied to permanent grasslands or ceasing inputs altogether. Chemical inputs to permanent grasslands may include fertilizers such as nitrogen (N), phosphorous (P), or potassium (K).*
Pesticide, Fertilizer, Mechanization	Convert to organic farming	Likely to be beneficial	40%	*Organic farming is an agricultural system that excludes the use of synthetic fertilizers and pesticides and relies on techniques such as crop rotation, compost and biological pest control. Organic standards are strictly regulated in many countries prohibiting the use of chemicals and providing recommendations for management to conserve biodiversity. Organic farming may include combinations of several separate interventions.*

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Pesticides, plant diversity	Grow plants that compete with damaging weeds	Likely to be beneficial	70%	*This intervention involves planting species that outcompete damaging weeds, suppressing them by reducing their ground cover, growth or reproduction rate, or by increasing their mortality. The intervention is generally applied to pastureland or uncropped areas such as field margins and buffer strips.*
Wetland, LDI	Restore or create traditional water meadows	Likely to be beneficial	100%	*Water meadows are areas of grazing land or hay meadow that have carefully controlled water levels to keep the soil damp. In Europe they provide valuable breeding habitats for waders and other biodiversity. The studies below describe instances when multiple interventions have been used to create water meadows. When the effects of multiple interventions, such as raising water levels and adding foot drains, can be separated, they are discussed under the relevant intervention.*
Wetland, livestock intensity	Maintain upland heath/moorland	Likely to be beneficial	90%	*This intervention involves using management techniques to maintain the conservation value of upland heath or moorland, including unenclosed rough grazing. These semi-natural habitats are predominantly found within unenclosed, extensive landscapes, their 'open' nature typically maintained by practices such as grazing, cutting and burning. Treatments tested include removing or reducing grazing, controlling the dominance of grass species by cutting or grazing by goats, and carrying out grouse moor management (rotational burning combined with predator control).*
Wetlands	Maintain traditional water meadows	Likely to be beneficial	50%	*This intervention may involve using management such as mowing or grazing to maintain plant communities and wildlife typically associated with traditional water meadows. Water meadows are areas of grazing land or hay meadow that have carefully controlled water levels to keep the soil damp. In Europe they provide valuable breeding habitats for wading birds and other biodiversity.*
Wetlands	Regulate water levels (maintain pond water levels)	Beneficial	70%	*Drying of amphibian breeding sites before terrestrial life stages have developed can have significant detrimental effects on populations. In some cases it may be possible to maintain water levels until after metamorphosis by using a local water source or by bringing in water from an outside source. Occasional drying of breeding sites can increase diversity, as it can help control predators, non-native species or more dominant species.*
Wetlands	Restore or create wetlands	Likely to be beneficial	70%	*Wetland habitats are often drained or degraded during the development of agriculture or expansion of urban areas or other land uses. Restoration of these*

Pasture				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
	and marine habitats (coastal and intertidal wetlands)			*important amphibian habitats can help to increase local amphibian species richness and abundance.*
Wetlands	Create scrapes and pools in wetlands and wet grasslands	Likely to be beneficial	75%	*Creating scrapes and pools in wetlands and wet grasslands can help create a heterogeneous habitat, with varying vegetation types and water levels.*
Wetlands, Mechanization	Manage ditches to benefit wildlife	Likely to be beneficial	40%	*Managing ditches to benefit wildlife can involve reduced or delayed cutting of vegetation on ditch banks and restricted fertilizer, herbicide or pesticide use on ditch banks or in fields adjoining ditches. 'Bundled' ditches are blocked to allow them to fill with water.*
Wetlands	Create ponds	Likely to be beneficial	80%	*Many ponds have been lost as land has been converted for agriculture or development, and with the intensification of agriculture, for example. Creation of additional breeding habitat may help to replace some of that lost and therefore help to maintain and increase amphibian populations. Different pond types can be created and some may be beneficial to certain species but not to others.*

Table 33: Management parameters for the land use type plantation

Plantation crops				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Artificial nesting/resting/foraging sites	Create artificial hibernacula or aestivation sites	Likely to be beneficial	50%	*Amphibians need damp sheltered places for overwintering or aestivating during hot arid summers. Overwintering or aestivating sites, or 'hibernacula', can be created for amphibians where natural sites are limited or where these habitats have been lost, for example at newly restored sites or in gardens.*
Artificial nesting/resting/foraging sites	Create refuges	Likely to be beneficial	45%	*Refuge habitats can provide amphibians with microclimates to keep them at the correct temperature and prevent them from dehydrating and can protect them from predation. Many am-*

Plantation crops				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
				phibians seek shelter in rocks, logs or other refuges created by tree falls and other disturbances. Refuges can be created for amphibians where natural shelter habitat is limited, or to replace these habitats where they have been lost.
Artificial nesting/resting/foraging sites	Provide food for vultures to reduce mortality from diclofenac	Likely to be beneficial	60%	*Vulture 'restaurants' have therefore been proposed as a method of reducing mortality until the drug is phased out, by providing a safe, uncontaminated source of food for the remaining vulture populations.*
Artificial nesting/resting/foraging sites	Provide supplementary food for birds or mammals	Beneficial	90%	*This intervention may involve the provision of supplementary food for birds or mammals in farmland habitats, such food typically includes seeds. Providing supplementary food for farmland wildlife may be particularly important when food resources in the wider farmed environment are scarce.*
Fertilizer	Use organic rather than mineral fertilizers	Beneficial	100%	*The use of chemicals in agricultural management, such as synthetic fertilizers, may have a detrimental effect on farmland biodiversity. Organic fertilizers such as farmyard manure (including green manure or crop residues), slurry and other composts provide an alternative to synthetic or mineral fertilizers. Mineral fertilizers are manufactured preparations including nitrogen N, phosphorous P or potassium K, or 'NPK'.*
Fertilizer, Pesticide	Reduce fertilizer, pesticide or herbicide use generally	Beneficial/Likely to be beneficial	100%	*Pesticide, herbicide and fertilizer applications may have a negative impact on farmland wildlife. This intervention may involve reducing or ceasing applications of pesticides (such as insecticides, fungicides), herbicides and fertilizers.*
Habitat, set-aside areas, connectivity, LDI	Manage hedgerows to benefit wildlife (includes no spray, gap- filling and laying)	Likely to be beneficial	70%	*Hedges can be key habitats for farmland biodiversity, but they may need managing to maximize their value. Managing hedges to benefit wildlife involves one or more of the following management changes: reduce cutting frequency; reduce or avoid spraying; mow vegetation beneath hedgerows; fill gaps in hedges; coppice or lay to restore traditional hedge structure.*
Habitat, set-aside areas, connectivity, LDI	Create uncultivated margins around intensive arable or pasture fields	Beneficial/Likely to be beneficial	100%	*This intervention allows the field margin vegetation to regenerate naturally, without planting, although it can involve subsequent mowing. The field margins are not fertilized and only spot-treated with herbicides if injurious weeds occur.*

Plantation crops				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Habitat, set-aside areas, connectivity, LDI	Leave headlands in fields unsprayed (conservation headlands)	Beneficial/Likely to be beneficial	90%	*Conservation headland management involves restricted fertilizer, herbicide and insecticide spraying in a 6 m margin of sown arable crop. The prescription allows selected herbicide applications to control injurious weeds or invasive alien species.*
Habitat, set-aside areas, connectivity, LDI	Plant nectar flower mixture/wildflower strips	Beneficial	100%	*Flowering plants are sown in strips or blocks, providing forage resources for bees and other flower-visiting insects. Increased insect numbers may then provide food for more birds. Nectar flower mixture may include agricultural varieties of flowering plants such as clovers.*
Habitat, set-aside areas, connectivity, LDI	Plant wild bird seed or cover mixture	Beneficial	100%	*The loss of food supplies, especially seeds, is thought to be a key driver of farmland bird declines. Plants that provide seed food and cover for wild birds include corn, sunflower and cereals. Wild bird cover crops are often planted in blocks or 6 m-wide strips and left unharvested. These are sometimes called 'game crops' or 'game cover crops'. They may also provide benefits for other farmland wildlife.*
Habitat, set-aside areas, connectivity, LDI	Provide or retain set-aside areas in farmland	Beneficial	90%	*Set-aside can be rotational (in a different place every year or two) or non-rotational (same place for 5-20 years) and fields can either be sown with fallow crops or left to naturally regenerate. Unlike fallow land left for the benefit of ground-nesting birds or arable plants, set-aside is not ploughed or harrowed except for the purpose of sowing.* *Set-aside is often managed by cutting and/or spraying. In some cases, set-aside land has had strips of wildflowers or grasses sown on it. Evidence for the effects of this management has been included under the following interventions: 'Plant nectar flower mixture wildflower strips' and 'Plant grass buffer strips margins around arable or pasture fields'.*
Habitat, set-aside areas, connectivity, LDI	Replant vegetation	Beneficial	70%	*Vegetation can be replanted to replace habitat that has been lost.*
Habitat, set-aside areas, connectivity, LDI	Restore habitat connectivity	Likely to be beneficial	75%	*Habitat destruction and fragmentation are important factors in the decline of amphibian populations. Small patches of habitat support smaller populations and if individuals are unable to move to other suitable areas, populations become isolated. This can make them more vulnerable to extinction. Restoring corridors of native vegetation between patches of suitable habitat may help to maintain amphibian populations.*

Plantation crops				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Habitat, set-aside areas, connectivity, LDI	Retain forested corridors in logged areas	Likely to be beneficial	55%	*This intervention involves retaining corridors of unlogged mature forest within logged areas. This may provide foraging and roosting opportunities for bats and maintain connectivity in disturbed landscapes.*
Habitat, set-aside areas, connectivity, LDI	Create/protect habitat corridors	Likely to be beneficial	65%	*Corridors are areas of natural habitat that are contiguous or isolated (i.e. linkages or stepping stones) and enable particular plant and animal species to disperse and migrate, processes which are necessary for their survival (Rouget et al. 2006).*
Habitat, set-aside areas, connectivity, LDI	Retain riparian buffer strips during timber harvest	Likely to be beneficial	50%	*Retaining forest strips along water courses or around ponds during timber harvest can help mitigate the effects of habitat loss and disturbance for forest species. They can also help sustain the microclimate and reduce potential problems such as soil erosion. Retained habitat strips also provide corridors for dispersal.*
Harvest intensity	Relocate nests at harvest time to reduce nestling mortality	Likely to be beneficial	55%	*If nests are likely to be destroyed by machinery during harvest or mowing, it may be possible to move them and then return them after the danger has passed. If nests are extremely likely to be destroyed during harvest or mowing then it may be best to remove the chicks and hand-rear them.*
Harvest intensity	Use mechanical thinning before or after planting	Beneficial	75%	*Mechanical thinning, that is removal of some trees to reduce the density, is used in restored forest areas to help the establishment of the remaining planted trees by reducing the competition for resources.*
Harvesting	Log/remove trees within forests: effects on understory plants	Beneficial	65%	*Here logging is defined as the selective removal of trees with the aim of removing tree biomass. This helps to restore natural open woodland by creating gaps and increasing light availability within the forest, which may increase the growth of the remaining vegetation.*
Harvesting	Thin trees within forests: effects on understory plants	Likely to be beneficial	58%	*Thinning is the removal of trees to control the development or enhance the future condition of a forest, by adjusting its density, structure and species composition.*
Harvesting	Thin trees within forests: effects on young trees	Likely to be beneficial	60%	*Thinning is the removal of trees to control the development or enhance the future condition of a forest, by adjusting its density, structure and species composition.*

Plantation crops				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Harvesting	Use shelterwood harvest instead of clearcutting	Likely to be beneficial	75%	*Shelterwood harvesting is a management technique designed to obtain even-aged timber without clearcutting. It involves harvesting trees in a series of partial cuttings, with trees removed uniformly over the plot, which allows new seedlings to grow from the seeds of older trees. This can help maintain characteristic forest species and increase structural diversity of stands.*
Mechanization	Prepare the ground before tree planting	Beneficial	78%	*Different soil preparation treatments are used to improve the soil before restoration planting to increase the establishment of planted tree seedlings.*
Other	Clear vegetation	Likely to be beneficial	60%	*Vegetation can be removed to prevent natural succession where specific habitat types are desired, or where invasive species are out-competing native species for example.*
Other	Reduce conflict by deterring birds from taking crops: use bird scarers	Likely to be beneficial	66%	*In some parts of the world, the persecution of birds that take crops can be a serious threat to the survival of populations. Methods to reduce the damage done by birds can therefore be important in reducing the pressure on populations.*
Pesticide	Grow non-crop plants that produce chemicals that attract natural enemies	Likely to be beneficial	68%	*This intervention involves growing non-crop plants which produce volatile chemicals (quickly evaporating scents or odours) that attract natural enemies, thereby encouraging the enemies to the main crop. Non-crop plants could be grown in field margins or interspersed into the main crop. Lab studies demonstrating an attractive effect of a plant species or variety to a natural enemy are also included.*
Pesticide	Use chemicals to attract natural enemies	Likely to be beneficial	40%	*This involves using chemicals to lure natural enemies into a crop. Communication chemicals of insects and plants (known as pheromones and volatiles, respectively) can be manufactured and deployed to manipulate invertebrates. Examples include the volatiles produced when plants are attacked by pests (e.g. methyl salicylate) and the alarm and sex pheromones of pests or natural enemies, as well as organic extracts from crop or plant leaves. Chemicals are sprayed onto crops or deployed in dispensers placed at regular intervals in the crop. Many studies have tested the efficacy of chemicals by applying them as baits in insect traps such as delta traps (plastic structures hung from branches or posts containing a sheet of sticky paper).*
Pesticide, Fertilizer, Mechanization	Convert to organic farming	Likely to be beneficial	55%	*Organic farming is an agricultural system that excludes the use of synthetic fertilizers and pes-*

Plantation crops				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
				ticides and relies on techniques such as crop rotation, compost and biological pest control. Organic standards are strictly regulated in many countries prohibiting the use of chemicals and providing recommendations for management to conserve biodiversity. Organic farming may include combinations of several separate interventions.
Pesticides	Exclude ants that protect pests	Likely to be beneficial	40%	*This involves applying adhesive substances or chemicals to the trunks of perennial crop trees, preventing pest-protecting ants from reaching the branches. Many ants form mutualistic relationships with insect pests (e.g. feeding on honeydew secreted by bugs (Hemiptera) such as aphids), defending them from predators and parasitoids. Excluding these ants may therefore increase predation and parasitism rates by beneficial invertebrates.*
Tree diversity/plant diversity/native vegetation	Retain or plant native trees and shrubs amongst crops (agroforestry)	Likely to be beneficial	55%	*This intervention involves growing crops under shade trees that are either native tree species that are remnants from cleared vegetation, or other crop trees (often referred to as 'agroforestry'). This approach provides a more complex habitat than conventional monoculture farming and can support higher levels of biodiversity.*
Tree diversity/plant diversity/native vegetation	Encourage agroforestry	Likely to be beneficial	50%	*This intervention involves growing crops under shade trees that are either native tree species that are remnants from cleared vegetation, or other crop trees (often referred to as 'agroforestry'). This approach provides a more complex habitat than conventional monoculture farming and can support higher levels of biodiversity.*
Wetland, LDI	Restore or create traditional water meadows	Likely to be beneficial	100%	*Water meadows are areas of grazing land or hay meadow that have carefully controlled water levels to keep the soil damp. In Europe they provide valuable breeding habitats for waders and other biodiversity. The studies below describe instances when multiple interventions have been used to create water meadows. When the effects of multiple interventions, such as raising water levels and adding foot drains, can be separated, they are discussed under the relevant intervention.*
Wetland, LDI	Raise water levels in ditches or grassland	Likely to be beneficial	100%	*Raising water levels increased numbers of birds, invertebrates or plants or allowed wet grassland plant species to establish more rapidly.*
Wetland, LDI	Regulate water lev-	Beneficial	70%	*Drying of amphibian breeding sites before terrestrial life stages have developed can have significant detrimental effects on populations. In*

Plantation crops				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
	els (maintain pond water levels)			*some cases it may be possible to maintain water levels until after metamorphosis by using a local water source or by bringing in water from an outside source. Occasional drying of breeding sites can increase diversity, as it can help control predators, non-native species or more dominant species.*
Wetland, LDI	Restore or create wetlands and marine habitats (coastal and intertidal wetlands)	Likely to be beneficial	70%	*Wetland habitats are often drained or degraded during the development of agriculture or expansion of urban areas or other land uses. Restoration of these important amphibian habitats can help to increase local amphibian species richness and abundance.*
Wetland, LDI	Create scrapes and pools in wetlands and wet grasslands	Likely to be beneficial	75%	*Creating scrapes and pools in wetlands and wet grasslands can help create a heterogeneous habitat, with varying vegetation types and water levels.*
Wetland, LDI	Create ponds	Likely to be beneficial	80%	*Many ponds have been lost as land has been converted for agriculture or development, and with the intensification of agriculture, for example. Creation of additional breeding habitat may help to replace some of that lost and therefore help to maintain and increase amphibian populations. Different pond types can be created and some may be beneficial to certain species but not to others.*
Wetlands, harvesting	Cut/remove/thin forest plantations	Likely to be beneficial	60%	*Only for peatlands*
Wetlands, harvesting	Cut/remove/thin forest plantations and rewet peat	Likely to be beneficial	60%	*Only for peatlands*
Wetlands, Mechanization	Manage ditches to benefit wildlife	Likely to be beneficial	40%	*Managing ditches to benefit wildlife can involve reduced or delayed cutting of vegetation on ditch banks and restricted fertilizer, herbicide or pesticide use on ditch banks or in fields adjoining ditches. 'Bundled' ditches are blocked to allow them to fill with water.*

Table 34: Management parameters for the land use type forestry

Forestry (primary and secondary vegetation)				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Anti-poaching	Conduct regular anti-poaching patrols	Likely to be beneficial	70%	*Anti-poaching patrols typically consist of a team of people that regularly patrol a pre-defined area to stop or reduce hunting. During patrols, teams may record spatial data on hunting or poaching activities and primate occurrence. Some teams may also capture and arrest illegal hunters on site, seize bush meat, and destroy hunting camps.*
Anti-poaching	Guard habituated primate groups to ensure their safety/well-being	Likely to be beneficial	60%	*A controlled, before-and-after study in 1967-2008 in tropical moist montane forest in Volcanoes-, Mgahinga-, and Virunga National Parks in Rwanda, Uganda, and the Democratic Republic of Congo found that a mountain gorilla Gorilla beringei population where individual animals were closely guarded against poachers alongside ten other interventions, increased in size over time.*
Anti-poaching	Implement community control of patrolling, banning hunting and removing snares	Likely to be beneficial	70%	*For this intervention, it is the community and not government staff, which implements all patrolling activities (including snare removals) and controls hunting bans.*
Anti-poaching	Provide better equipment (e.g. guns) to anti-poaching ranger patrols	Likely to be beneficial	50%	*If anti-poaching rangers are provided with better equipment (e.g. guns, or technical equipment, such as GPS, compass, hand-held data recording devices, binoculars, cameras, rain gear, etc.), they may be more effective at reducing hunting in the areas they patrol.*
Anti-poaching	Regularly de-activate/remove ground snares	Likely to be beneficial	60%	*This intervention involves the regular patrolling of teams to de-activate/remove ground snares.*
Artificial nesting/resting/foraging sites	Create artificial hibernacula or aestivation sites	Likely to be beneficial	50%	*Some primate species, especially larger and more terrestrial species, such as gorillas Gorilla spp. and chimpanzees Pan troglodytes, may be injured by getting caught in snares typically set out to catch animals such as duikers Cephalophus spp. and bush pigs Potamochoerus larvatus. These primate species can get their hands or feet trapped in snares while travelling through the forest, often resulting in life threatening injuries and even death.*

Forestry (primary and secondary vegetation)				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
				This intervention involves the regular patrolling of teams to de-activate/remove ground snares.
Artificial nesting/resting/foraging sites	Create refuges	Likely to be beneficial	45%	*Refuge habitats can provide amphibians with microclimates to keep them at the correct temperature and prevent them from dehydrating and can protect them from predation. Many amphibians seek shelter in rocks, logs or other refuges created by tree falls and other disturbances. Refuges can be created for amphibians where natural shelter habitat is limited, or to replace these habitats where they have been lost.*
Artificial nesting/resting/foraging sites	Provide bat boxes for roosting bats	Likely to be beneficial	30%	*Bats roost in caves, built structures, natural crevices (e.g. in rocks) and in trees. The provision of bat boxes is a widely used intervention, as a conservation measure and for research, and there is a lot of literature on the use of these structures by bats. However, the many different designs of bat box available makes it difficult to draw consistent conclusions as evidence in support of each individual design is lacking. Studies are needed that evaluate the effects of providing artificial roosts on bat populations, by observing changes in bat numbers over time, ideally in areas with and without bat boxes.*
Artificial nesting/resting/foraging sites	Provide artificial nesting sites for songbirds	Beneficial	67%	*Provide artificial nesting sites for songbirds.*
Artificial nesting/resting/foraging sites	Clean nest boxes to increase occupancy or reproductive success	Likely to be beneficial	40%	*Old nest boxes may contain parasites and so reduce breeding success in them, or they may provide a suitable base for building new nests. Cleaning them out may, therefore, have a positive or a negative impact on nest site choice and reproductive success.*
Deadwood volume	Provide deadwood/snags in forests (use ring-barking, cutting or silvicides)	Likely to be beneficial	45%	*Woody debris can be created in forests by 'ring-barking' or 'girdling', a process which removes the living tissue from a tree in a ring around the trunk. This prevents water and nutrients from reaching the leaves and upper portions of the tree, normally killing the plant, which then decays to produce a snag.*
Habitat, set-aside areas, connectivity, LDI, tree diversity	Restore or create forests	Beneficial	65%	*Some forests are the most complex terrestrial habitats, with countless species interacting. Restoring such complexity is difficult, but there is an ever-increasing amount of research and investment into the area. For example, insurance firms and shipping companies are financing a 25-year project to restore forest*

Forestry (primary and secondary vegetation)				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
				ecosystems along the Panama Canal (TEEB 2008); whilst mining companies in Australia are increasingly able to reconstruct the forests they destroyed after they have finished mining at a site (Nichols & Grant 2007).
Habitat, set-aside areas, connectivity, LDI, tree diversity	Replant vegetation	Beneficial	70%	*Vegetation can be replanted to replace habitat that has been lost.*
Habitat, set-aside areas, connectivity, LDI	Restore habitat connectivity	Likely to be beneficial	75%	*Habitat destruction and fragmentation are important factors in the decline of amphibian populations. Small patches of habitat support smaller populations and if individuals are unable to move to other suitable areas, populations become isolated. This can make them more vulnerable to extinction. Restoring corridors of native vegetation between patches of suitable habitat may help to maintain amphibian populations.*
Habitat, set-aside areas, connectivity, LDI, anti-poaching	Use wildlife refuges to reduce hunting disturbance	Likely to be beneficial	45%	*Wildlife refuges are a type of protected area where hunting is prohibited. Often situated near hunting areas, they can be effective both by reducing the number of birds shot, but also in reducing the disturbance to non-target species.*
Habitat, set-aside areas, connectivity, LDI, wetlands	Retain riparian buffer strips during timber harvest	Likely to be beneficial	50%	*Retaining forest strips along water courses or around ponds during timber harvest can help mitigate the effects of habitat loss and disturbance for forest species. They can also help sustain the microclimate and reduce potential problems such as soil erosion. Retained habitat strips also provide corridors for dispersal.*
Harvesting	Log/remove trees within forests: effects on understory plants	Beneficial	65%	*Here logging is defined as the selective removal of trees with the aim of removing tree biomass. This helps to restore natural open woodland by creating gaps and increasing light availability within the forest, which may increase the growth of the remaining vegetation.*
Harvesting	Thin trees within forests: effects on understory plants	Likely to be beneficial	58%	*Thinning is the removal of trees to control the development or enhance the future condition of a forest, by adjusting its density, structure and species composition.*
Harvesting	Thin trees within forests: effects on young trees	Likely to be beneficial	60%	*Thinning is the removal of trees to control the development or enhance the future condition of a forest, by adjusting its density, structure and species composition.*

Forestry (primary and secondary vegetation)				
Management parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Harvesting	Use selective harvesting/reduced impact logging instead of clearcutting	Likely to be beneficial	60%	*Selective logging is the removal of selected trees within a forest based on criteria such as diameter, height or species. Remaining trees are left in the stand, as opposed to clearcutting where all trees are felled.*
Harvesting	Use shelterwood harvest instead of clearcutting	Likely to be beneficial	75%	*Shelterwood harvesting increased density of trees or plant diversity, or decreased grass cover compared with clearcutting.*
Harvesting intensity, tree age, biomass density	Use patch retention harvesting instead of clearcutting	Likely to be beneficial	70%	*In forests in which trees are commercially exploited for timber, a system known as patch retention harvesting may be used as an alternative to a total clear-cut. Typically, around 10% of mature and/or immature trees are retained in patches within an otherwise, clear-cut harvest compartment, with 'prompt reforestation' subsequent to timber extraction in the other 90%. These retained patches could help maintain characteristic forest species and act as reservoirs for re-colonisation by forest dependent species.*
Harvesting intensity, tree age, biomass density	Use shelterwood harvest instead of clearcutting	Likely to be beneficial	75%	*Shelterwood harvesting is a management technique designed to obtain even-aged timber without clearcutting. It involves harvesting trees in a series of partial cuttings, with trees removed uniformly over the plot, which allows new seedlings to grow from the seeds of older trees. This can help maintain characteristic forest species and increase structural diversity of stands.*
Other	Clear vegetation	Likely to be beneficial	60%	*Vegetation can be removed to prevent natural succession where specific habitat types are desired, or where invasive species are outcompeting native species for example.*
Set-aside areas	Provide paths to limit disturbance	Likely to be beneficial	50%	*Studies have shown that visitors keep to paths when they are provided, thereby reducing disturbance to the wider habitat without the need for specific access restrictions (e.g. Pearce-Higgins & Yalden 1997).*
Set-aside areas	Use signs and access restrictions to reduce disturbance at nest sites	Likely to be beneficial	59%	*Increased numbers of breeders, higher reproductive success or lower levels of disturbance in waders and terns following the start of access restrictions or the erection of signs near nesting areas.*

Forestry (primary and secondary vegetation)				
Management parameter	Manage-ment acti-vity [198]	Cate-gory [198]	Effec-tiven-ess [198]	Description from [198]
Wetland, LDI	Restore or create tradi-tional water meadows	Likely to be benefi-cial	70%	*Water meadows are areas of grazing land or hay meadow that have carefully controlled wa-ter levels to keep the soil damp. In Europe they provide valuable breeding habitats for waders and other biodiversity. The studies be-low describe instances when multiple interven-tions have been used to create water meadows. When the effects of multiple inter-ventions, such as raising water levels and adding foot drains, can be separated, they are discussed under the relevant intervention.*
Wetland, LDI	Create scrapes and pools in wetlands and wet grasslands	Likely to be benefi-cial	75%	*Creating scrapes and pools in wetlands and wet grasslands can help create a heterogene-ous habitat, with varying vegetation types and water levels.*
Wetland, LDI	Regulate water levels (maintain pond water levels)	Benefi-cial	70%	*Drying of amphibian breeding sites before ter-restrial life stages have developed can have significant detrimental effects on populations. In some cases it may be possible to maintain water levels until after metamorphosis by us-ing a local water source or by bringing in water from an outside source. Occasional drying of breeding sites can increase diversity, as it can help control predators, non-native species or more dominant species.*
Wetland, LDI	Restore or create wet-lands and marine hab-itats (coastal and inter-tidal wet-lands)	Likely to be benefi-cial	70%	*Wetland habitats are often drained or de-graded during the development of agriculture or expansion of urban areas or other land uses. Restoration of these important amphib-ian habitats can help to increase local amphib-ian species richness and abundance.*
Wetland, LDI	Create ponds	Likely to be benefi-cial	80%	*Many ponds have been lost as land has been converted for agriculture or development, and with the intensification of agriculture, for exam-ple. Creation of additional breeding habitat may help to replace some of that lost and therefore help to maintain and increase am-phibian populations. Different pond types can be created and some may be beneficial to cer-tain species but not to others.*

Table 35: Management parameters for the land use type urban

Urban				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Artificial nesting/resting/foraging sites	Create artificial hibernacula or aestivation sites	Likely to be beneficial	50%	*Amphibians need damp sheltered places for overwintering or aestivating during hot arid summers. Overwintering or aestivating sites, or 'hibernacula', can be created for amphibians where natural sites are limited or where these habitats have been lost, for example at newly restored sites or in gardens.*
Artificial nesting/resting/foraging sites	Create alternative bat roosts within developments	Likely to be beneficial	45%	*New alternative bat roosts are often created within developments to replace original roosts that have been destroyed. This can include purpose-built bat barns, lofts or houses, bat boxes, or features can be created within existing buildings such as specially designed crevices and bat bricks.*
Artificial nesting/resting/foraging sites	Provide artificial nesting sites for songbirds	Beneficial	67%	*Provide artificial nesting sites for songbirds.*
Artificial nesting/resting/foraging sites	Provide bat boxes for roosting bats	Likely to be beneficial	60%	*Bats roost in caves, built structures, natural crevices (e.g. in rocks) and in trees. The provision of bat boxes is a widely used intervention, as a conservation measure and for research, and there is a lot of literature on the use of these structures by bats. However, the many different designs of bat box available makes it difficult to draw consistent conclusions as evidence in support of each individual design is lacking. Studies are needed that evaluate the effects of providing artificial roosts on bat populations, by observing changes in bat numbers over time, ideally in areas with and without bat boxes.*
Artificial nesting/resting/foraging sites	Provide supplementary food for songbirds to increase adult survival	Beneficial	65%	
Connectivity/Traffic intensity	Install rope or pole (canopy) bridges	Likely to be beneficial	50%	*Rope and pole bridges, or so-called 'canopy bridges', allow safe crossing of human-made barriers (e.g. roads) by arboreal primates that spend most of their time in the forest canopy.*
Connectivity/Traffic intensity	Install overpasses as	Likely to be	45%	*Overpasses (solid structures such as bridges built for pedestrians or vehicles) may help to guide bats safely over roads. This would both*

Urban				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
	road crossing structures for bats	beneficial		*reduce the number of bats killed on roads and increase the permeability of roads for bats to maintain connectivity across the landscape.*
Connectivity/Traffic intensity	Install underpasses as road crossing structures for bats	Likely to be beneficial	60%	*Underpasses may guide bats safely under roads. They have the potential to reduce the number of bats killed by traffic and increase the permeability of roads for bats to maintain connectivity across the landscape. There is evidence that an unknown proportion of bats of various species use underpasses (e.g. Bach et al. 2004, Boonman 2011, Barros 2014).*
Green space, native vegetation, plant diversity, LDI, sealing, set-aside area	Replant vegetation	Beneficial	70%	*Vegetation can be replanted to replace habitat that has been lost.*
Light pollution	Use low intensity lighting	Likely to be beneficial	65%	*Light pollution may be minimized by reducing light levels, e.g. by dimming lights or using low wattage or low intensity lights.*
Light pollution	Leave bat roosts, roost entrances and commuting routes unlit	Likely to be beneficial	80%	*Lighting in the vicinity of a bat roost may cause disturbance, altered behaviour and abandonment.*
Light pollution	Avoid illumination of bat commuting routes	Likely to be beneficial	70%	*Bat commuting routes provide essential connectivity between roosts and foraging habitats. Bats prefer to commute under the cover of darkness to avoid predation and may abandon commuting routes if they are illuminated with artificial lighting.*
Façade treatment	Use mammal safe timber treatments in roof spaces	Beneficial	90%	*Two controlled laboratory studies in the UK found commercial timber treatments (containing lindane and pentachlorophenol) to be lethal to bats, but found alternative artificial insecticides (including permethrin) and three other fungicides did not increase bat mortality. Sealants over timber treatments had varying success.*
Other	Clear vegetation	Likely to be beneficial	60%	*Vegetation can be removed to prevent natural succession where specific habitat types are desired, or where invasive species are out-competing native species for example.*
Other	Mark power lines	Beneficial	81%	*marking power lines led to significant reductions in bird collision mortalities.*

Urban				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Other	Bury or isolate power lines	Likely to be beneficial	60%	*a dramatic increase in juvenile eagle survival following the burial or isolation of dangerous power lines.*
Other	Insulate electricity pylons	Likely to be beneficial	60%	*insulating power pylons significantly reduced the number of Harris's hawks electrocuted.*
Other	Remove earth wires from power lines	Likely to be beneficial	90%	*reductions in bird collision mortalities after earth wires were removed from sections of power lines.*
Other	Use perch-deterrents to stop raptors perching on pylons	Likely to be beneficial	50%	*fewer raptors were found near perch-deterrent lines, compared to controls, but no information on electrocutions was provided.*
Scarers/Airport	Scare or otherwise deter birds from airports	Likely to be beneficial	50%	*Collisions between birds and aircraft ('bird strikes') at airports are potentially dangerous to planes, whilst also harming bird populations. Therefore making airports less attractive to birds, or scaring them off, has the potential for multiple benefits.*
Scarers/Mining	Use visual and acoustic 'scarers' to deter birds from landing on pools polluted by mining or sewage	Likely to be beneficial	50%	*Use scarers to deter birds.*
Habitat, set-aside areas, connectivity, LDI	Plant nectar flower mixture/wildflower strips	Beneficial	100%	*Flowering plants are sown in strips or blocks, providing forage resources for bees and other flower-visiting insects. Increased insect numbers may then provide food for more birds. Nectar flower mixture may include agricultural varieties of flowering plants such as clovers.*

Urban				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
Habitat, set-aside areas, connectivity, LDI	Create refuges	Likely to be beneficial	45%	*Refuge habitats can provide amphibians with microclimates to keep them at the correct temperature and prevent them from dehydrating and can protect them from predation. Many amphibians seek shelter in rocks, logs or other refuges created by tree falls and other disturbances. Refuges can be created for amphibians where natural shelter habitat is limited, or to replace these habitats where they have been lost.*
Habitat, set-aside areas, connectivity, LDI	Restore habitat connectivity	Likely to be beneficial	75%	*Habitat destruction and fragmentation are important factors in the decline of amphibian populations. Small patches of habitat support smaller populations and if individuals are unable to move to other suitable areas, populations become isolated. This can make them more vulnerable to extinction. Restoring corridors of native vegetation between patches of suitable habitat may help to maintain amphibian populations.*
Habitat, set-aside areas, connectivity, LDI	Create/protect habitat corridors	Likely to be beneficial	65%	*Corridors are areas of natural habitat that are contiguous or isolated (i.e. linkages or stepping stones) and enable particular plant and animal species to disperse and migrate, processes which are necessary for their survival (Rouget et al. 2006).*
Traffic intensity/Other	Modify gully pots and kerbs	Likely to be beneficial	80%	*Gully pots along roadside kerbs form effective traps for amphibians. Animals crossing roads reach the kerb and often move along its base, until they fall into a gully pot. Once in the gully pot amphibians cannot climb out. A study found that 63% of 636 gully pots in two areas in Scotland contained wildlife, of which 91% were amphibians (1,087 animals; Muir 2012). There are a number of ways in which the impact on amphibians could be reduced, such as moving gully pots, modifying the design of their grills, providing escape ladders or changing the shape of kerb stones (angled or indented).*
Traffic intensity	Close roads during seasonal amphibian migration	Likely to be beneficial	85%	*Road traffic can have significant effects on amphibian populations, particularly where their annual migration routes between overwintering and breeding sites cross roads. In some areas, roads can be closed to protect important migration routes.*
Wetlands/water areas, LDI	Create scrapes and pools in wetlands and	Likely to be beneficial	75%	*Creating scrapes and pools in wetlands and wet grasslands can help create a heterogeneous habitat, with varying vegetation types and water levels.*

Urban				
Management Parameter	Management activity [198]	Category [198]	Effectiveness [198]	Description from [198]
	wet grasslands			
Wetlands/water areas, LDI	Regulate water levels (maintain pond water levels)	Beneficial	70%	*Drying of amphibian breeding sites before terrestrial life stages have developed can have significant detrimental effects on populations. In some cases it may be possible to maintain water levels until after metamorphosis by using a local water source or by bringing in water from an outside source. Occasional drying of breeding sites can increase diversity, as it can help control predators, non-native species or more dominant species.*
Wetlands/water areas, LDI	Create artificial water sources	Likely to be beneficial	70%	*Artificial water sources may be created to provide foraging and drinking resources for bats in arid areas, or in areas where natural wetlands have been lost.*
Wetlands/water areas, LDI	Restore or create wetlands and marine habitats (coastal and intertidal wetlands)	Likely to be beneficial	70%	*Wetland habitats are often drained or degraded during the development of agriculture or expansion of urban areas or other land uses. Restoration of these important amphibian habitats can help to increase local amphibian species richness and abundance.*
Wetlands/water areas, LDI	Create ponds	Likely to be beneficial	80%	*Many ponds have been lost as land has been converted for agriculture or development, and with the intensification of agriculture, for example. Creation of additional breeding habitat may help to replace some of that lost and therefore help to maintain and increase amphibian populations. Different pond types can be created and some may be beneficial to certain species but not to others.*

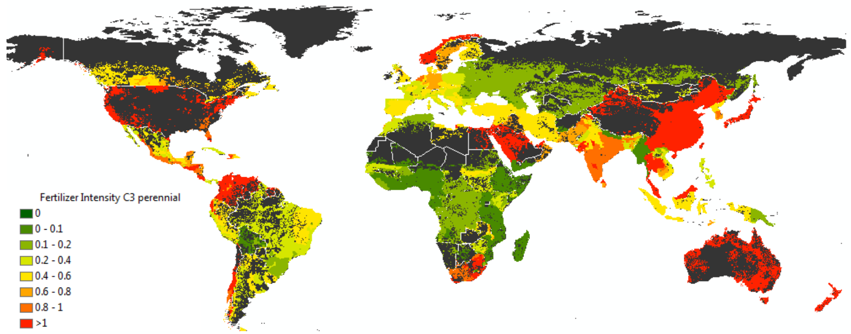

Figure 152: Fertilizer intensity of C3 perennial crop

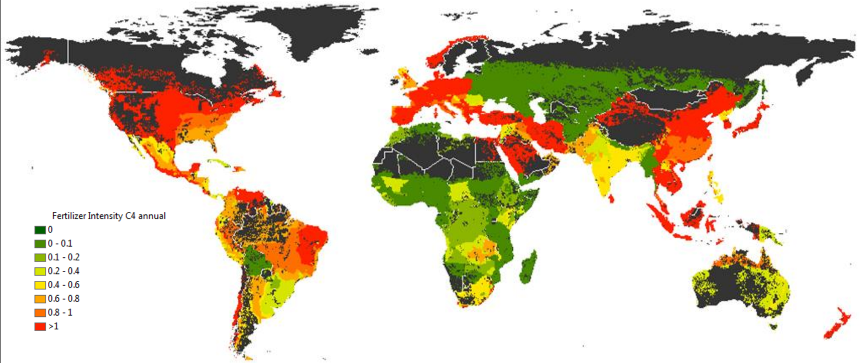

Figure 153: Fertilizer intensity of C4 annual crop

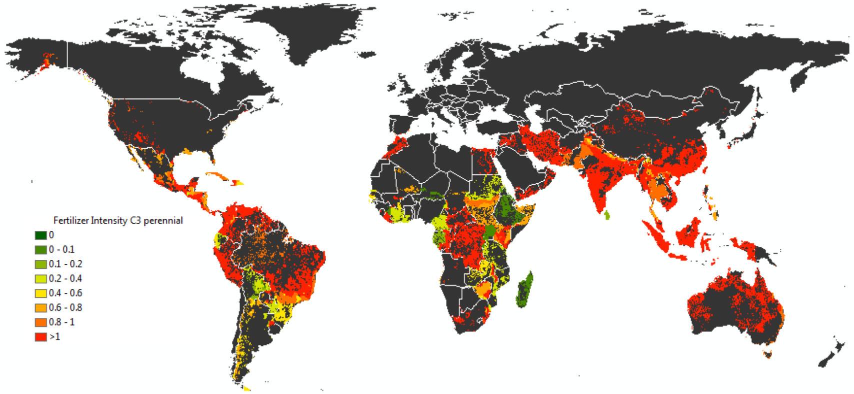

Figure 154: Fertilizer intensity of C4 perennial crop

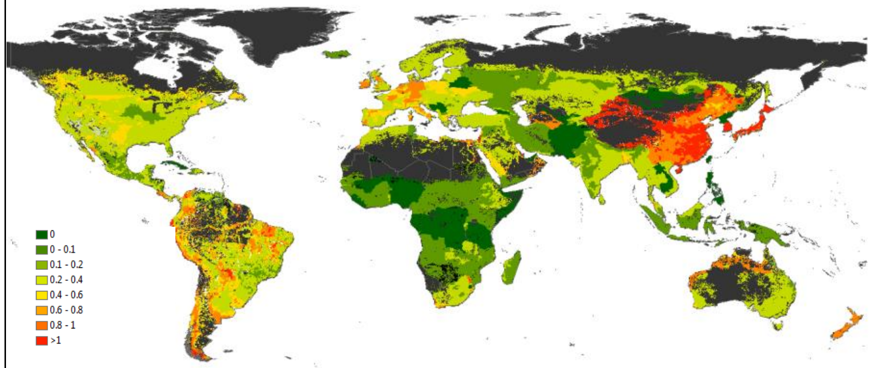

Figure 155: LUI C3 ann pesticide

10.5 Annex V: Land use intensity maps

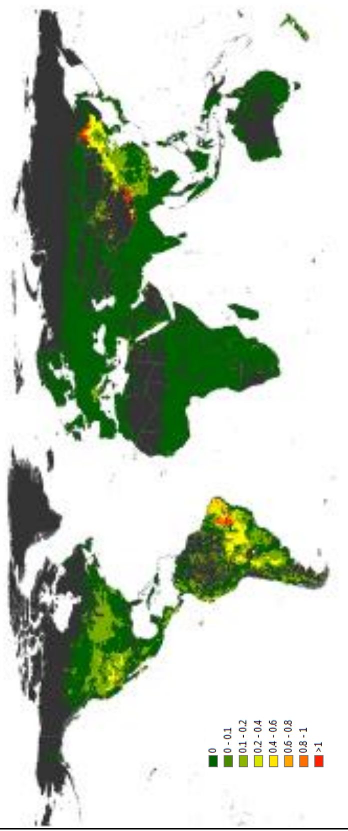

Figure 156: LUI C3 nfx pesticide

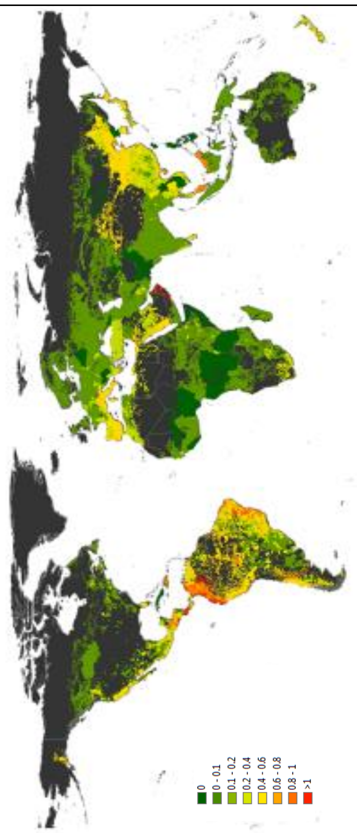

Figure 157: LUI C3 per pesticide

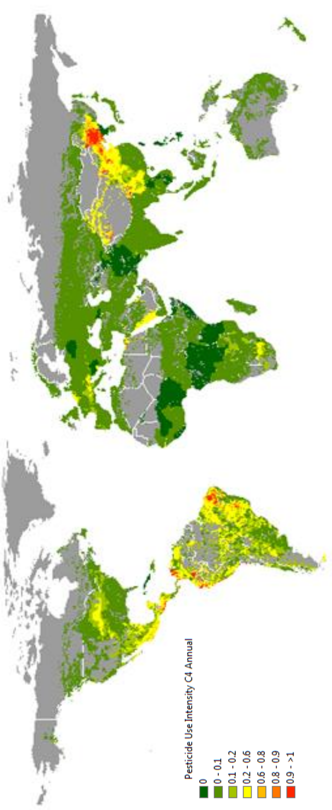

Figure 158: LUI C4 annual pesticide

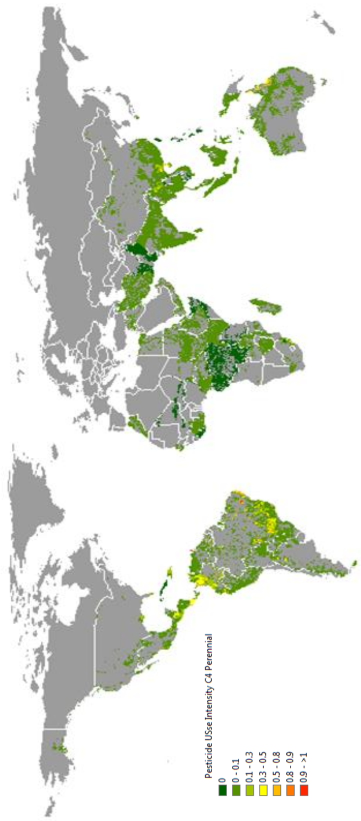

Figure 159: LUI C4 perennial pesticide

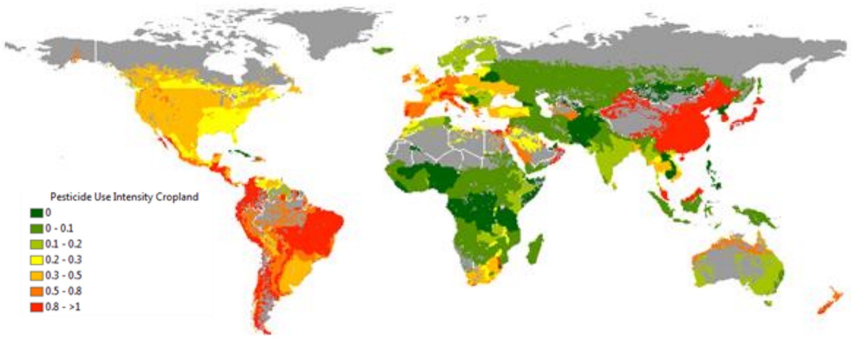

Figure 160: LUI cropland all pesticide

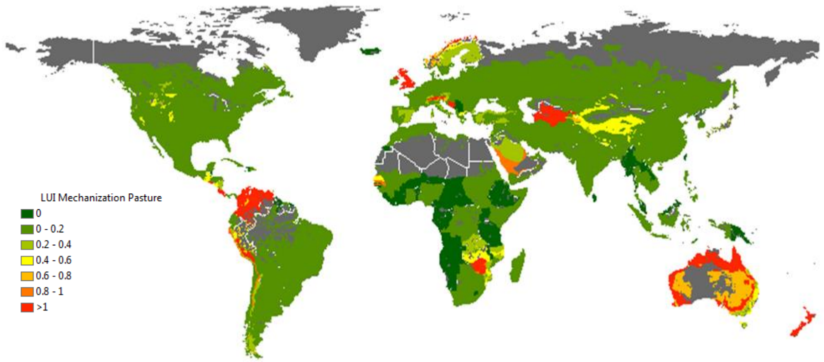

Figure 161: LUI mechanization pasture

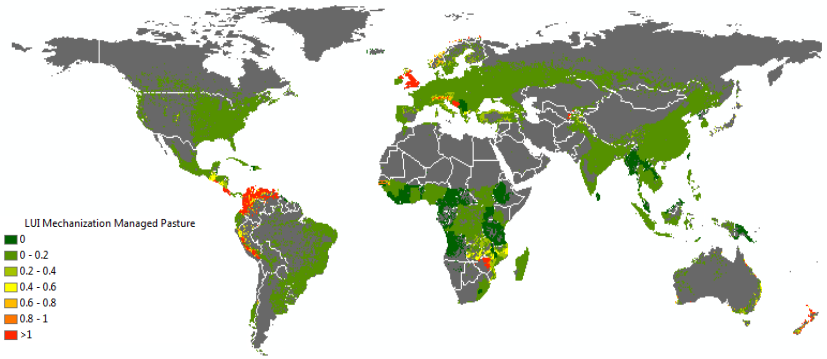

Figure 162: LUI mechanization managed pasture

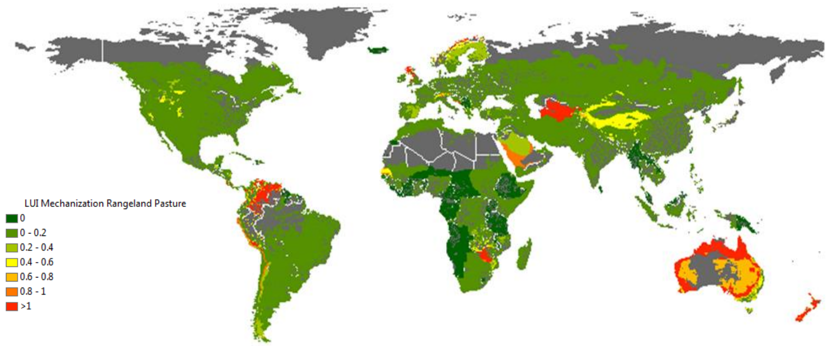

Figure 163: LUI mechanization rangeland pasture

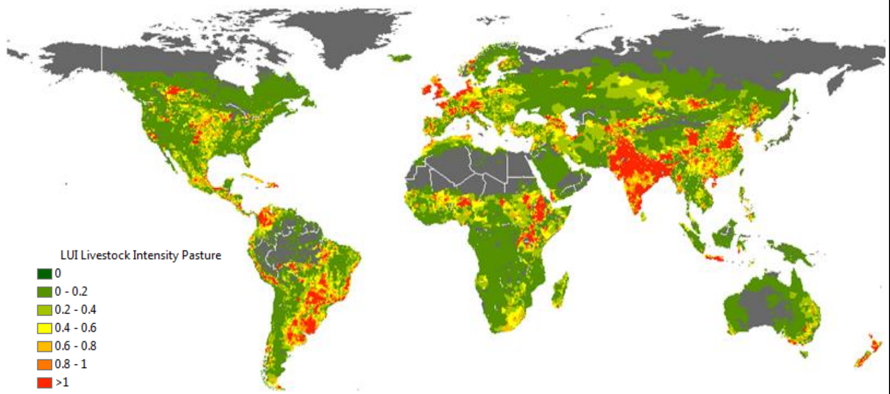

Figure 164: LUI livestock intensity pas-

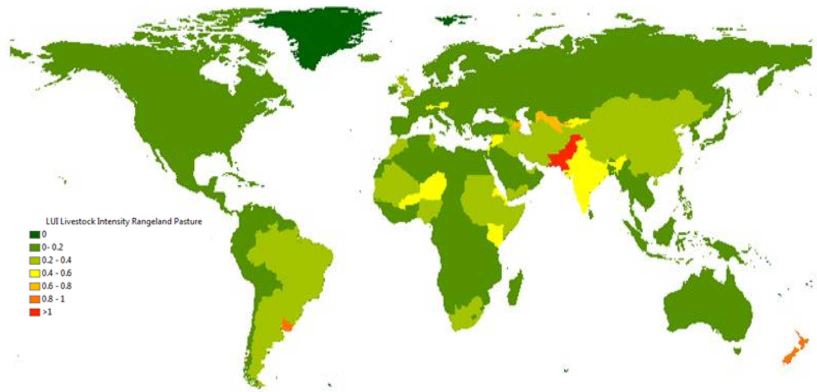

Figure 165: LUI livestock intensity rangeland pasture

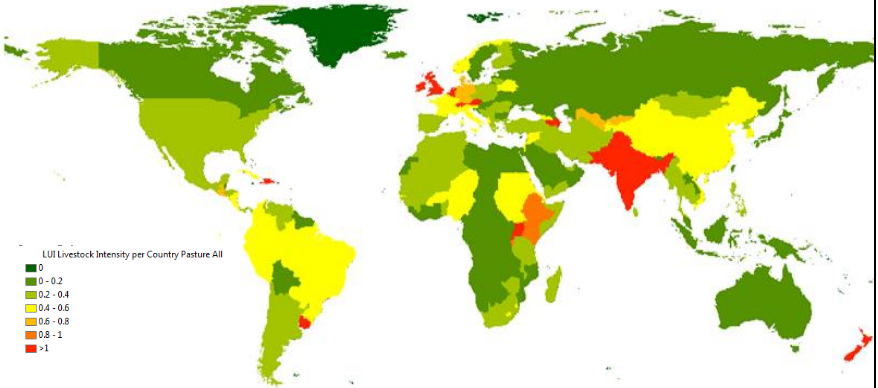

Figure 166: LUI livestock intensity per country pasture

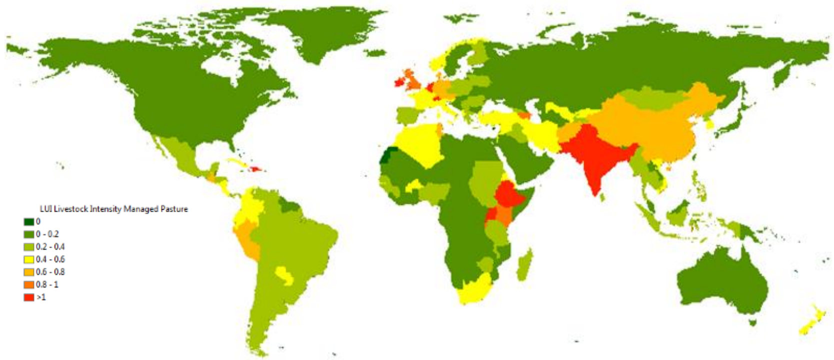

Figure 167: LUI livestock intensity managed pasture

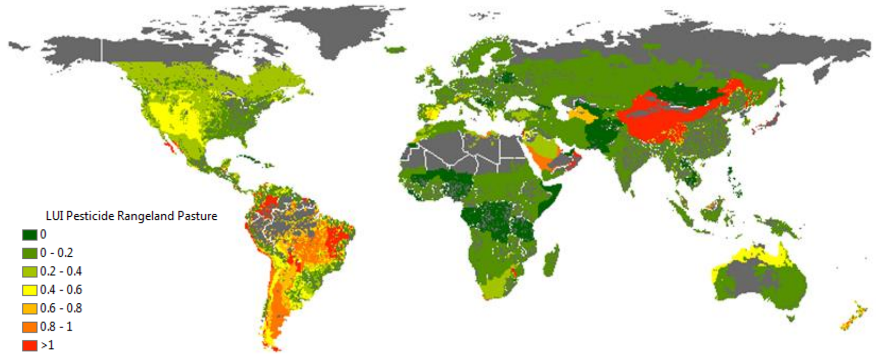

Figure 168: LUI pesticide rangeland pasture

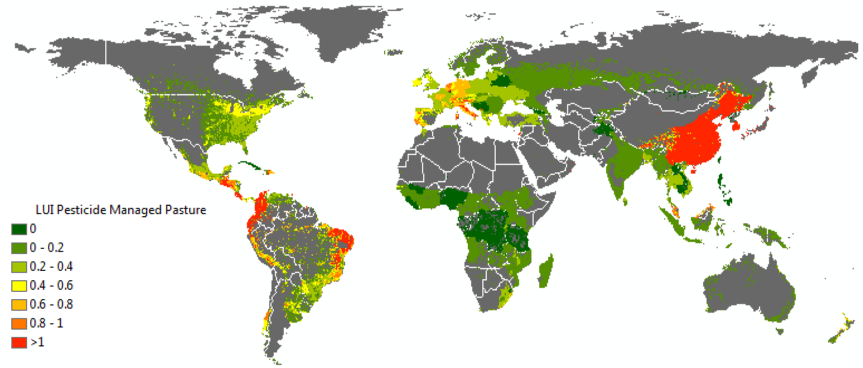

Figure 169: LUI pesticide managed pasture

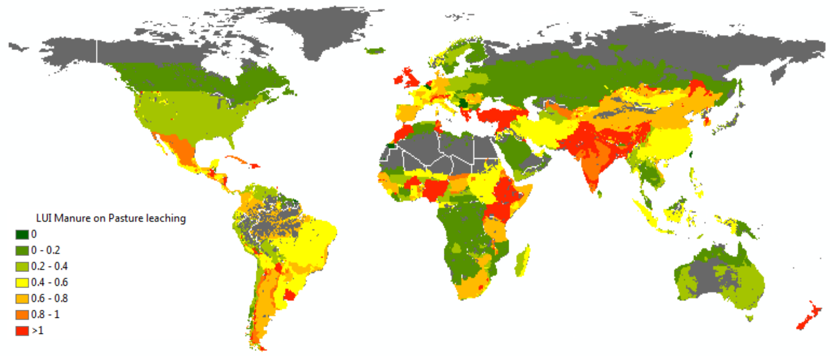

Figure 170: LUI manure on pasture leaching

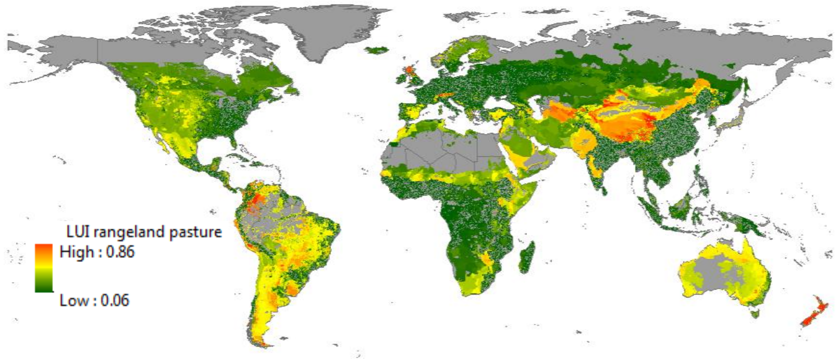

Figure 171: LUI rangeland pasture

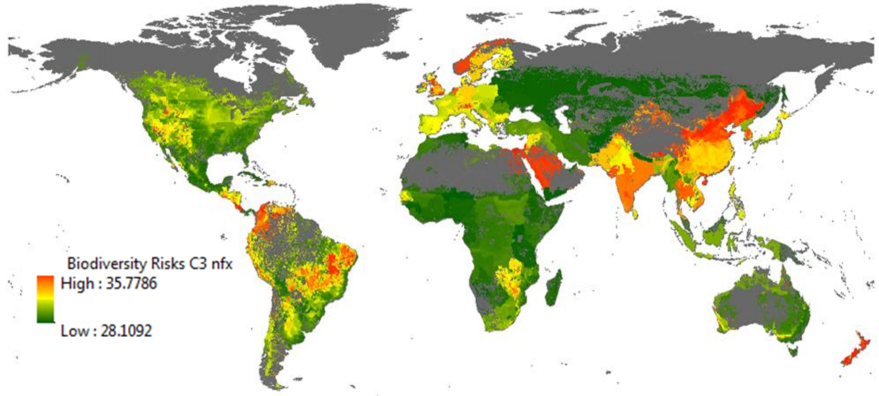

Figure 172: Biodiversity risks C3 nitrogen-fixing polynomial

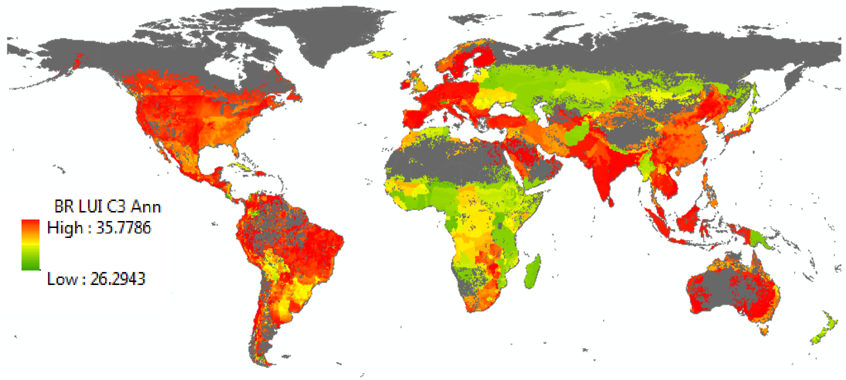

Figure 173: Biodiversity risks C3 annual polynomial fit

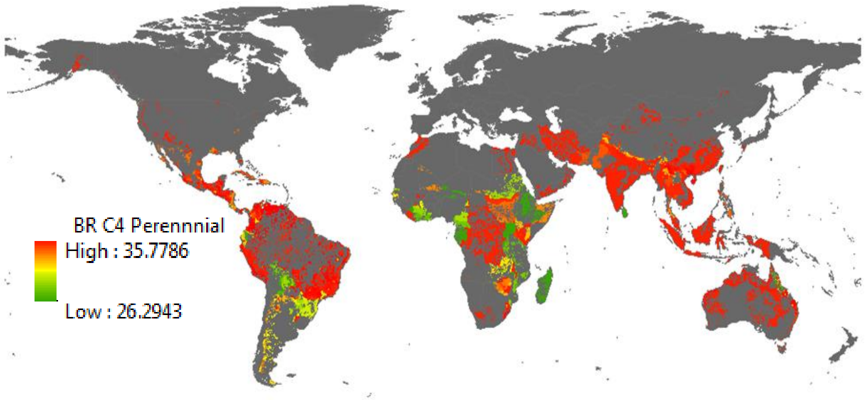

Figure 174: Biodiversity risks C4 perennial polynomial fit

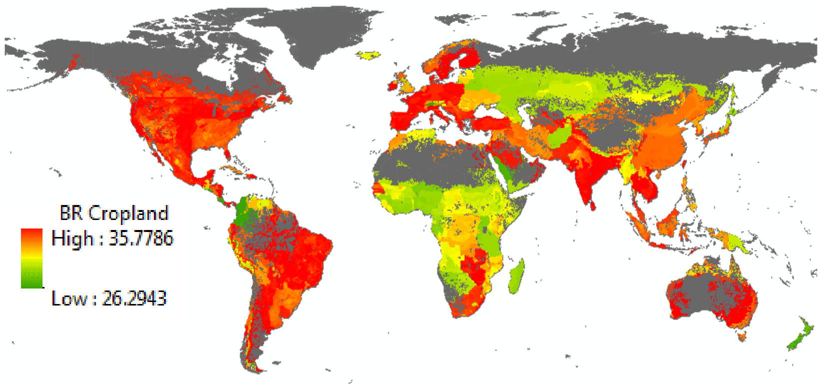

Figure 175: Biodiversity risks Cropland polynomial fit

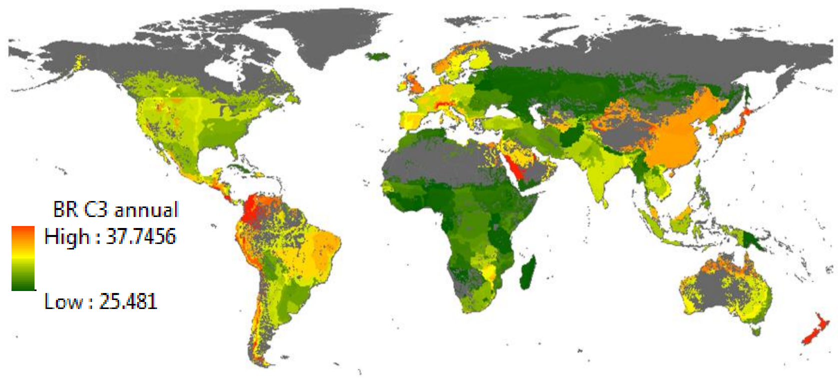

Figure 178: Biodiversity risks C3 annual crop linear fit

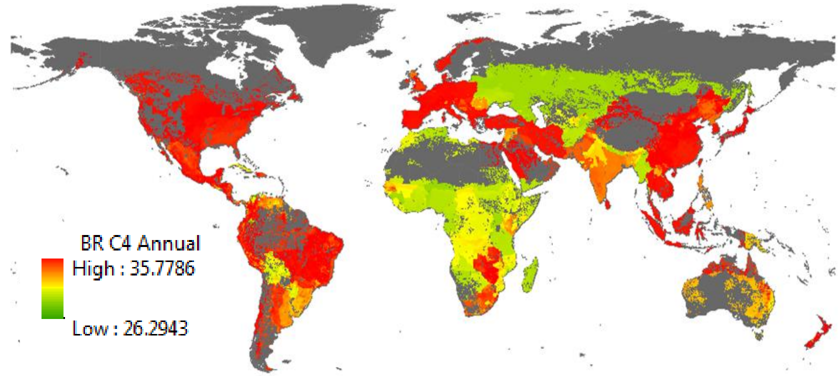

Figure 176: Biodiversity risks C4 annual polynomial fit

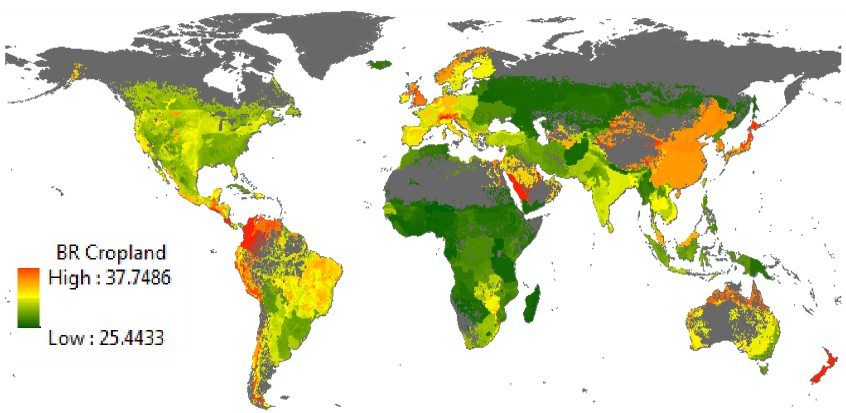

Figure 179: Biodiversity risks cropland degree 1 polynomial

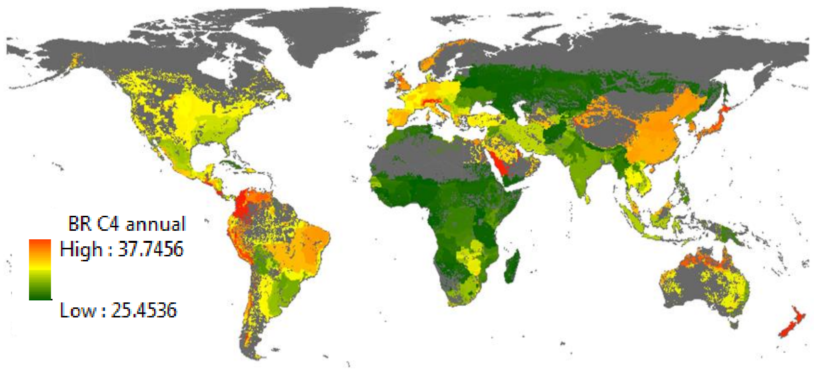

Figure 177: Biodiversity risks C4 annual crop degree 1 poly-

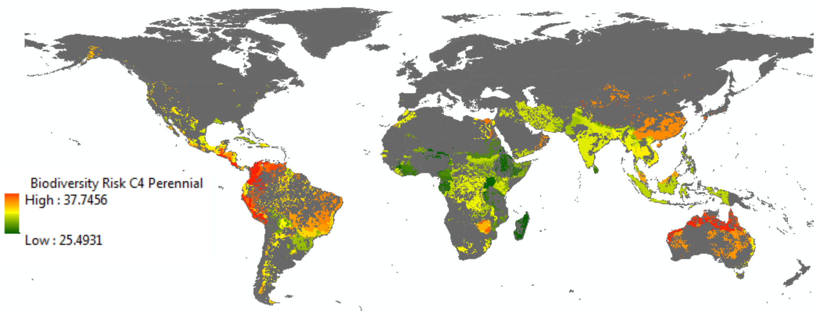

Figure 180: Biodiversity risks C4 perennial crop linear fit

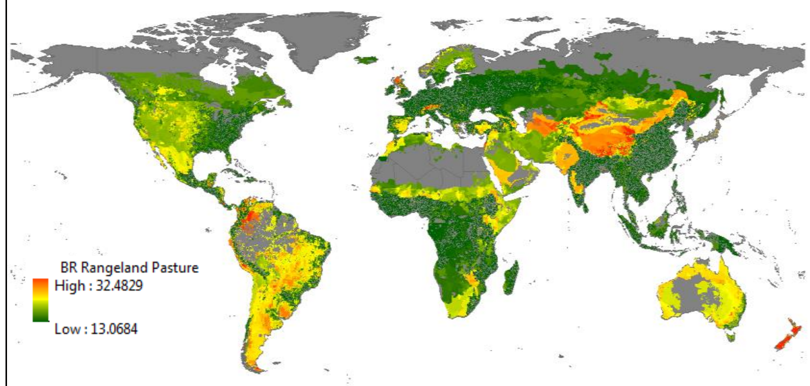

Figure 181: Biodiversity risk rangeland pasture

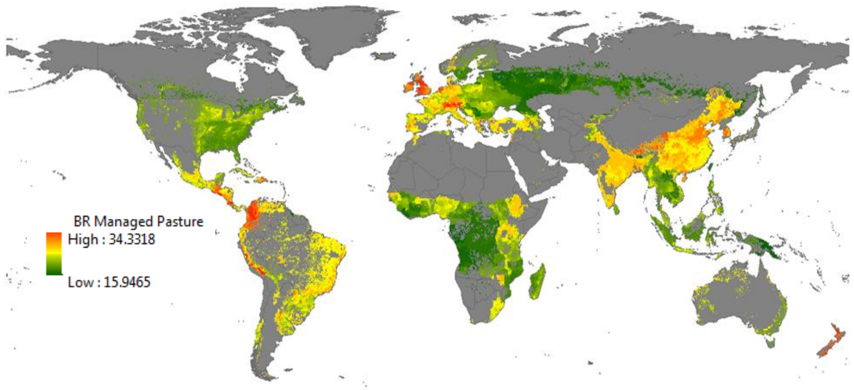

Figure 182: Biodiversity risk managed pasture

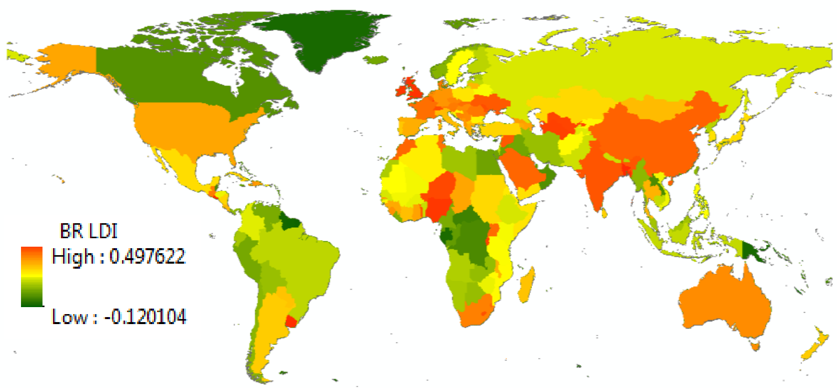

Figure 183: Biodiversity risks landscape level per country

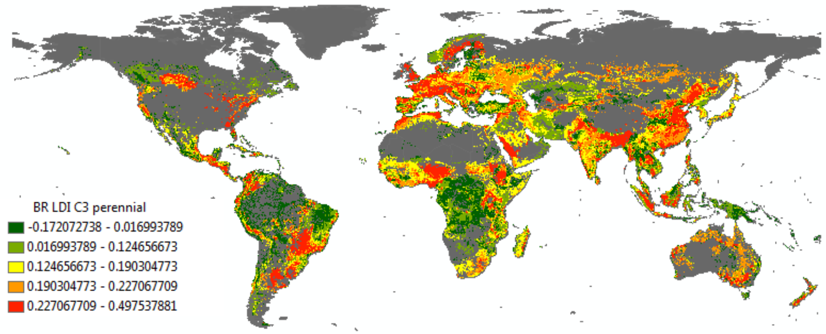

Figure 184: Biodiversity risks landscape C3 perennial

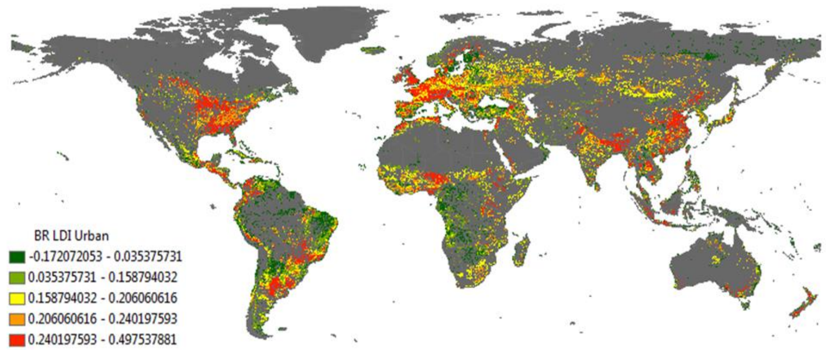

Figure 185: Biodiversity risks landscape urban areas

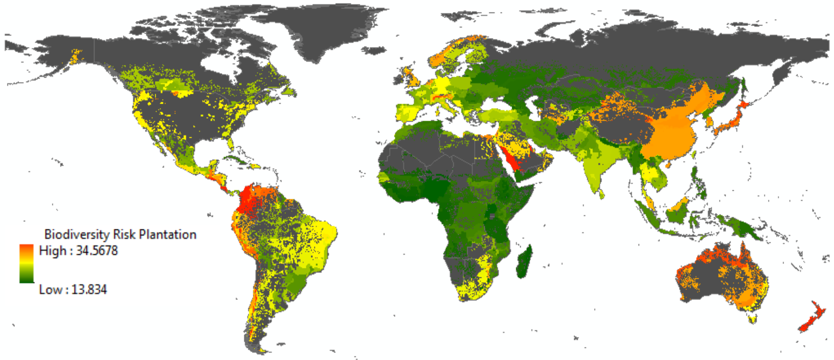

Figure 186: Biodiversity risks plantation C3 perennial

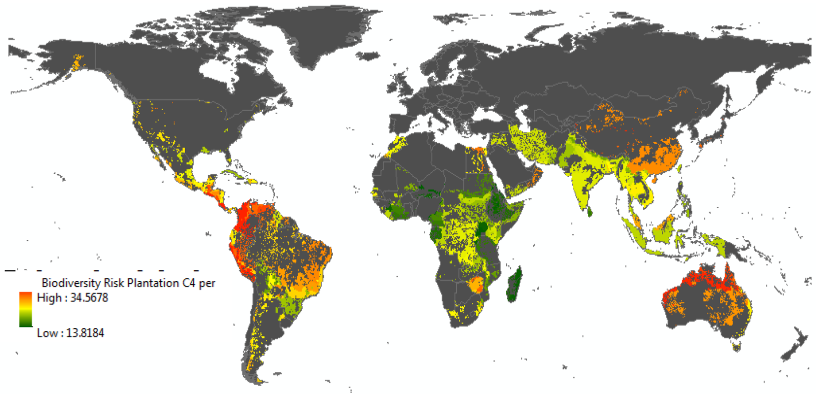

Figure 187: Biodiversity risks plantation C4 perennial

Table 36: Benchmark values for LUI cropland

	Fertilizer application kg Nitrogen/ha year					No of tractors per ha and year					Pesticide application in kg per ha and year				
GAEZ	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark
TropicalAEZ1	0.1	447.6	66.8	99.4	166.3	0	1.9	0.1	0.2	0.3	0	15.5	0.7	1.3	2.0
TropicalAEZ2	0.19	310.352	27.013	33.218	60.232	0	1.9	0.185	0.296	0.481	0	15.45	0.605	1.38	1.985
TropicalAEZ3	0.279	252.62	38.796	35.753	74.549	0	1.9	0.08	0.189	0.269	0	15.45	0.831	1.601	2.432
TropicalAEZ4	0.544	254.067	35.624	37.104	72.728	0	3.1	0.088	0.329	0.418	0	22.9	1.006	1.737	2.744
TropicalAEZ5	0.608	263.246	42.558	38.76	81.318	0	3.1	0.117	0.407	0.524	0	24.91	2.246	2.651	4.897
TropicalAEZ6	0.687	432.231	59.567	48.019	107.586	0	10	0.166	0.423	0.588	0	24.91	2.953	4.063	7.016
TemperateAEZ1	0.85	464.507	84.191	83.199	167.39	0	3.6	0.217	0.547	0.764	0	15.45	2.578	3.547	6.125
TemperateAEZ2	0.85	413.307	67.958	67.885	135.843	0	3.6	0.12	0.473	0.593	0	15.45	2.972	4.226	7.198
TemperateAEZ3	1.097	312.317	68.579	65.79	134.369	0	0.7	0.066	0.16	0.226	0	15.45	2.826	3.938	6.764
TemperateAEZ4	1.089	432.231	86.104	66.163	152.267	0	8.8	0.129	0.734	0.864	0	15.45	2.852	3.256	6.108

GAEZ	Fertilizer application kg Nitrogen/ha year					No of tractors per ha and year					Pesticide application in kg per ha and year				
	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark
TemperateAEZ5	1.405	638.282	135.6	80.311	215.912	0	8.8	0.489	1.729	2.218	0	13.17	5.356	4.477	9.833
TemperateAEZ6	4.141	432.232	140.792	98.086	238.878	0	8.8	0.288	1.157	1.445	0	13.17	6.709	4.584	11.294
BorealAEZ1	1.305	343.157	45.534	73.704	119.238	0	0.3	0.015	0.02	0.034	0	13.07	2.249	3.825	6.074
BorealAEZ2	1.089	313.776	77.381	96.557	173.938	0	3.6	0.014	0.062	0.076	0	13.07	1.798	3.323	5.122
BorealAEZ3	1.089	432.231	70.096	81.279	151.374	0	8.8	0.023	0.172	0.194	0	13.07	1.581	2.99	4.572
BorealAEZ4	2.722	432.231	177.809	109.122	286.931	0	8.8	1.266	2.739	4.004	0	13.07	5.985	4.851	10.835
BorealAEZ5	40.07	432.231	250.926	98.623	349.549	0.019	5	0.808	1.644	2.452	1.1	13.07	8.21	3.924	12.134
BorealAEZ6	24.21	245.483	190.952	77.636	268.588	0.043	0.043	0.043	0	0.043	5.69	5.69	5.69	0	5.69

Table 37: Benchmark values for LUI pasture

	Livestock intensity in TLU per km² and year					Pesticide intensity in kg per ha and year					Mechanization intensity in no per ha and year					Manure leaching intensity in kg per ha and year				
GAEZ	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark
TropicalAEZ1	0	164.61	8.42	14.77	23.2	0	15.45	0.66	1.21	1.86	0.00	1.90	0.10	0.20	0.31	0.00	11042.50	3117.60	2703.60	5821.00
TropicalAEZ2	0.12	193.28	17.65	22.38	40.03	0	15.45	0.67	1.37	2.04	0.00	1.90	0.18	0.29	0.48	0.00	13922.70	4184.31	3069.31	7254.00
TropicalAEZ3	0.01	515.82	26.27	31.49	57.76	0	15.45	0.85	1.59	2.44	0.00	1.90	0.07	0.17	0.24	0.00	13922.70	4681.09	2685.22	7366.00
TropicalAEZ4	0	279.28	17.82	29.57	47.39	0	22.9	1.01	1.74	2.76	0.00	3.10	0.09	0.33	0.42	0.00	27973.80	3546.91	2871.35	6418.00
TropicalAEZ5	0	828.14	20.04	29.82	49.86	0	24.91	2.28	2.72	5	0.00	3.10	0.11	0.39	0.50	0.00	27973.80	3368.22	3835.97	7204.00
TropicalAEZ6	0	429.24	9.32	17.53	26.85	0	24.91	2.9	4.27	7.17	0.00	10.00	0.14	0.38	0.51	0.00	27973.80	2161.84	2168.08	4330.00
TemperateAEZ1	0	299.3	5.52	15.31	20.83	0	15.45	2.71	3.69	6.4	0.00	3.60	0.22	0.57	0.79	0.00	10590.40	1786.46	1791.47	3578.00
TemperateAEZ2	0	1004.05	12.12	23.27	35.39	0	15.45	3.01	4.27	7.28	0.00	3.60	0.12	0.47	0.59	102.67	11042.50	1965.51	1773.29	3739.00
TemperateAEZ3	0	243.29	11.91	21.2	33.11	0	15.45	2.78	3.9	6.68	0.00	0.70	0.07	0.16	0.23	0.00	11042.50	1784.06	2045.66	3830.00
TemperateAEZ4	0	806.39	11.99	16.8	28.79	0	15.45	2.73	3.09	5.82	0.00	8.80	0.13	0.75	0.87	0.00	12785.50	1514.94	1631.04	3146.00

BorealAEZ6	BorealAEZ5	BorealAEZ4	BorealAEZ3	BorealAEZ2	BorealAEZ1	TemperateAEZ6	TemperateAEZ5	GAEZ	
0.05	0.05	0	0	0	0.01	0.08	0	MIN	Livestock intensity in TLU per km² and year
1.07	32.9	67.69	71.88	109.99	196.85	260.2	263.87	MAX	
0.49	4.03	8.53	1.34	2.37	2.65	21.33	20.64	MEAN	
0.3	6.17	10.05	3.65	5.26	5.09	19.64	24.72	STD	
0.79	10.2	18.58	5	7.64	7.74	40.97	45.36	Benchmark	
5.69	1.1	0	0	0	0	0	0	MIN	Pesticide intensity in kg per ha and year
5.69	13.07	13.07	13.07	13.07	13.07	13.17	13.17	MAX	
5.69	7.31	6.11	1.5	2.99	5.18	6.72	5.23	MEAN	
0	3.37	4.87	2.91	4.54	5.74	4.57	4.44	STD	
5.69	10.68	10.98	4.41	7.53	10.92	11.29	9.67	Benchmark	
0.04	0.02	0.00	0.00	0.00	0.00	0.00	0.00	MIN	Mechanization intensity in no per ha and year
0.04	5.00	8.80	8.80	3.60	0.30	8.80	8.80	MAX	
0.04	0.92	1.33	0.03	0.02	0.01	0.28	0.49	MEAN	
0.00	1.85	2.82	0.19	0.11	0.03	1.16	1.75	STD	
0.04	2.78	4.15	0.21	0.13	0.04	1.44	2.23	Benchmark	
727.81	727.81	0.00	102.67	102.67	102.67	0.00	0.00	MIN	Manure leaching intensity in kg per ha and year
727.81	12785.50	12785.50	12785.50	10590.40	10590.40	12785.50	27973.80	MAX	
727.81	3121.91	3217.83	341.47	989.51	1639.68	3132.10	2567.23	MEAN	
0.00	4369.95	3727.37	781.19	1384.94	1441.45	2664.72	2351.04	STD	
728.00	7492.00	6945.00	1123.00	2374.00	3081.00	5797.00	4918.00	Benchmark	

Table 38: Benchmark values for LUI forestry

	Mean tree age (years)					biomass density					Wood harvest rates (kg C per km² and year)				
Global Forest Zone	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark
Boreal coniferous fo-	1.00	971.60	100.96	115.71	216.67	0.03	8.40	2.79	1.85	4.64	0.00	324196.34	8866.93	38490.60	47357.53
Boreal mountain system	1.00	1140.50	95.17	130.20	225.37	0.03	9.50	1.86	1.69	3.54	0.00	237443.42	1329.38	14040.65	15370.03
Boreal tundra woodland	1.00	480.88	20.44	34.71	55.15	0.03	5.28	0.70	0.82	1.52	0.00	6281.12	6.03	172.76	178.79
Polar	1.00	1106.55	25.99	83.95	109.94	0.03	6.74	0.40	0.82	1.22	0.00	634.13	0.49	12.76	13.25
Subtropical dry forest	1.00	301.61	71.04	68.03	139.07	0.12	12.89	5.72	3.86	9.58	0.00	414134.13	81778.17	96826.33	178604.50
Subtropical humid forest	1.00	468.31	120.22	89.83	210.05	0.22	15.67	9.86	3.44	13.30	0.00	602011.75	40081.90	47047.51	87129.41
Subtropical mountain	1.00	850.84	188.78	139.63	328.41	0.06	13.26	5.58	3.19	8.78	0.00	2286885.50	43617.00	206985.41	250602.41
Subtropical steppe	1.00	89.98	39.66	21.66	61.32	0.13	9.82	5.95	3.13	9.07	0.00	262591.91	51735.81	71670.00	123405.81
Temperate continental	1.00	463.02	104.71	79.63	184.34	0.12	11.32	5.35	2.05	7.40	0.00	429303.28	10690.00	36622.66	47312.66
Temperate desert	4.58	468.51	91.29	90.11	181.40	0.16	7.27	2.81	1.72	4.53	0.00	16745.02	409.75	1749.26	2159.01

	Mean tree age (years)					biomass density					Wood harvest rates (kg C per km² and year)				
Global Forest Zone	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark	MIN	MAX	MEAN	STD	Benchmark
Temperate mountain system	1.00	780.93	98.89	90.04	188.93	0.04	12.80	4.02	2.55	6.56	0.00	491186.38	17549.04	53879.32	71428.36
Temperate oceanic forest	1.00	265.08	37.57	38.03	75.60	0.11	11.65	3.81	2.87	6.68	0.00	361559.41	47353.89	71805.18	119159.07
Temperate steppe	1.00	459.24	66.85	81.20	148.06	0.16	9.54	2.75	1.60	4.35	0.00	12183.80	80.40	533.34	613.74
Tropical dry forest	1.00	317.42	22.81	28.44	51.25	0.22	16.87	3.98	3.13	7.10	0.00	1022426.88	27686.03	95107.07	122793.09
Tropical moist forest	1.00	453.36	39.38	56.46	95.84	0.15	17.56	5.01	3.93	8.94	0.00	3652950.00	12388.52	140282.83	152671.35
Tropical mountain system	1.00	819.26	33.64	52.32	85.96	0.06	16.33	4.76	3.77	8.53	0.00	3344934.25	70457.01	351252.44	421709.45
Tropical rainforest	1.00	865.88	72.34	86.44	158.78	0.15	18.42	8.00	5.02	13.03	0.00	3962609.50	11746.88	154731.13	166478.00
Tropical shrub land	1.00	277.47	29.79	49.09	78.88	0.20	12.64	4.15	3.24	7.38	0.00	262562.66	12685.94	43395.16	56081.09

Table 39: Average deadwood volume for most of Europe [195]

Country	2000	2005	2010
Belarus	2,1	1.0	1,2
Czech Republic		11,6	11,6
Poland			5,6
Slovakia		26,2	26,2
Ukraine	25.0	26.0	27.0
Austria	13,7	17,4	20,3
Belgium	7,1	7.0	7,3
Germany	11,5	11,5	15.0
Ireland		6,6	6,3
Luxembourg	11,6	11,6	
Netherlands	4,5	8,1	9,8
Switzerland	14,5	17,9	21,3
United Kingdom	3,9	3,9	3,9
Denmark		4,9	5,1
Estonia	9,9	12,5	14,6
Finland	5,6	5,7	5,7
Latvia	6.0	17,7	17,7
Lithuania	23.0	23.0	23.0
Norway	6,8		
Sweden	6,5	7,9	8,2
Russian Federation	21,9	21,8	22.0
Croatia			14.0
Slovenia	14,9	17.0	19,1
Italy	8,3	8,7	9,1
Portugal	2000	2,8	

10.6 Annex VI: Case study data – management Parameter

Table 40: Foreground calculations for palm oil extraction plant – Urban (Malaysia), location: 114.384860 -2.897319

Management parameter	Primary data	Secondary data	Unit	Intensity
Set-aside area/green space	N/A	10	[%] green space/set-aside area per urban area	0.80
Degree of sealing	N/A	0.002	[%] imperviousness	0.04
Light pollution	N/A	0.015	artificial sky brightness [mcd/m²]	0.01
Population density	N/A	35.85	Persons per km²	0.06
Traffic intensity	N/A	1093870.09	No of million vehicle per km² and year	0.80
			Total LUI	0.34

Table 41: Foreground calculations for HVO plant – Urban (Netherlands), location: 4.595145 51.917739 decimal degrees

Management parameter	Primary data	Secondary data	Unit	Intensity
Set-aside area/green space	N/A	10	[%] green space/set-aside area per urban area	0.8
Degree of sealing	N/A	0.09	[%] imperviousness	1.0
Light pollution	N/A	50.33	artificial sky brightness [mcd/m²]	1.0
Population density	N/A	2670.62	Persons per km²	1.0
Traffic intensity	N/A	133093	No of million vehicle per km² and year	0.1
			Total LUI	0.78

Methane from corn silage

Table 42: Foreground calculation for corn production – C4 annual (Germany)

Management Parameter	Primary data	Secondary data	Unit	Intensity value back-ground data	Intensity value pri-mary data
Fertilizer	130	298.41	kg nitrogen $ha^{-1} \cdot year^{-1}$	1	0.85
Pesticide	N/A	2.06	kg/ha	0.64	
Mechaniza-tion (tillage)	N/A	0.097	No of tractor $ha^{-1} \cdot year^{-1}$	0.09	
Set-aside areas	N/A	5	Ratio Field size/buffer zone size [%]	0.8	
Crop rota-tion/crop diversity	N/A	0.01	Share crop diversity per field [%]	0.99	
			Total LUI	0.7	0.67

Table 43: Foreground calculation for biogas plant – Urban (Germany), location: 13.880334 51.153543 decimal degrees

Management para-meter	Primary data	Secondary data	Unit	Intensity
Set-aside area/green space	N/A	10	[%] green space/set-aside area per ur-ban area	0.8
Degree of sealing	N/A	0.05	[%] imperviousness	1
Light pollution	N/A	0.59	artificial sky brightness [mcd/m^2]	0.23
Population density	N/A	878.92	Persons per km^2	1.0
Traffic intensity	N/A	1093870.09	No of million vehicle per km^2 and year	0.79
			Total LUI	0.77

Methane – Pyrolysis from wood

Table 44: Foreground calculation for wood production – Secondary vegetation (forest) (Finland)

Management parameter	Primary data	Secondary data	Unit	Intensity
Mean age/tree age	N/A	65.77	years	0.89
Wood harvesting rates	N/A	944.123	units kg C	0.0199
Dead Wood volume	N/A	3.15	Average deadwood volume (m3 ha−1)	0.81
Set-aside areas/buffer zones	N/A	10	Protected forest area/total forest area [%]	0.8
biomass density	N/A	4.588	kg C/m²	0.67
			Total LUI	0.63

Table 45: Foreground calculation for SNG plant – Urban (Finland), location: 25.094550 60.390238 decimal degrees

Management parameter	Primary data	Secondary data	Unit	Intensity
Set-aside area/green space	N/A	10	Green space/set-aside area per urban area [%]	0.8
Degree of sealing	N/A		Imperviousness [%]	0.45
Light pollution	N/A	0.19	Artificial sky brightness [mcd/m²]	0.07
Population density	N/A	20.38	Persons per km²	0.034
Traffic intensity	N/A	51386	No of million vehicle per km² and year	0.04
			Total LUI	0.28

Ethanol from sugar cane

Table 46: Foreground calculation for sugarcane production – C4 perennial (Brazil)

Management parameter	Primary data	Secondary data	Unit	Intensity value background data	Intensity value primary data
Fertilizer	68.8	101.23	kg nitrogen $ha^{-1} \cdot year^{-1}$	0.94	0.64
Pesticide	4.042	4.31	kg per ha	0.61	0.58
Mechanization (tillage)	N/A	0.014	No of tractor $ha^{-1} \cdot year^{-1}$	0.02	
Set-aside areas	N/A	5	Ratio Field size/buffer zone size [%]	0.8	
Crop rotation/crop diversity	N/A	0	Share crop diversity per field [%]	1	
			Total LUI	0.68	0.61

Table 47: Foreground calculation for ethanol production plant – Urban (Brazil), location: 46.375392 23.640822 decimal degrees

Management parameter	Primary data	Secondary data	Unit	Intensity value
Set-aside area/green space	N/A	10	[%] green space/set-aside area per urban area	0.8
Degree of sealing	N/A	0.03	[%] imperviousness	0.79
Light pollution	N/A	0.4	artificial sky brightness [mcd/m^2]	0.155
Population density	N/A	659.93	Persons per km^2	1.0
Traffic intensity	N/A	1093870.095	No of million vehicle per km^2 and year	0.79
			Total LUI	0.71

Curriculum Vitae of Dipl.-Biol. M.Sc. Stephanie Maier

Experience and competence

Doctorate at the University of Stuttgart in the area of biodiversity assessments in LCA. I studied biology at the University of Tübingen and global change ecology at the University of Bayreuth. My work focuses on the modelling of land use and land use changes with Geographic Information Systems (GIS) and the analysis of impacts on biodiversity for integration into life cycle assessment. Furthermore, I develop methods for sustainability assessment based on the Sustainable Development Goals (SDGs) and life cycle analyses.

Education

02/2017 - today	Doctorate at the University of Stuttgart, Department of Life Cycle Engineering • Biodiversity assessments in global value chains
10/2013 - 04/2016	Master in Global Change Ecology in the Elite Network Bavaria (grade 1.3) • Focus on global socio-ecological change
09/2015 - 10/2015	Research stay at the GIZ Programme Renewable Energies and Energy Effi ciency, Dhaka, Bangladesh • Household interviews to gather data on social, ecological and economic indicators
09/2005 - 03/2012	Diploma in Biology at the Eberhard Karls University of Tübingen (grade 1.3) • Focus Zoology
01/2011 - 04/2011	Research stay at Programme Onchocercoses, Ngaoundéré, Cameroon • Ecological field studies on the parasite-host system of the blackfly Simulium damnosum and the nematode Onchocerca ochengi

Work experience

09/2022 – today	Researcher in the Department of Residential and Environmental Affairs, State Capital Stuttgart • SDG analysis • GIS based modelling • Environmental data analysis
02/2017 – 09/2022	Researcher in the Department of Life Cycle Engineering, University of Stuttgart in the field of Life Cycle Assessment, biodiversity assessment and SDG analysis, Stuttgart • GIS based modelling • Development of a method for the analysis of biodiversity impacts in LCA • Project management and acquisition of projects
07/2016 - 01/2017	Trainee at GIZ in the Private Sector Cooperation/ UN Global Compact (German network), Berlin • Production of publications and webinars in the fields of biodiversity and corporate responsibility • Organization of multi-stakeholder meetings and workshops
04/2015 - 11/2016	Scientific assistance in the Department of Life Cycle Engineering, University of Stuttgart • GIS analyses on land use and ecosystem services
10/2015 - 06/2016	Project assistance at the consulting institute Adelphi Consult GmbH in the fields of environment, climate and sustainable development, Berlin

- Research on the environmental impact assessment of the German energy system transformation
- Project management support

02/2014 - 04/2014 Internship at ForestFinance/ CO2OL in the field of consulting for environmental and climate protection, Bonn

- Environmental and climate consulting
- Research on climate protection in companies: best practices

08/2012 - 02/2013 Trainee at the United Nations Environment Programme - Bonn Convention on Migratory Species (UNEP-CMS), Bonn

- Research on international nature conservation projects in the field of ecological networks
- Interviews with experts and NGOs

02/2012 - 03/2012 Scientific assistant at the University of Tübingen on ecological fieldwork in Ngaoundéré, Cameroon

- Planning and implementation of ecological field studies
- Leadership of a bi-national workshop
- Preparation and implementation of zoological excursions to Cameroonian nature reserves

09/2009 - 10/2009 Internship at the NABU avian conservation centre in Mössingen

- Scientific research for the protection of the osprey (*Pandion haliaetus*) and its habitats
- Preparation of a publication on species protection in Germany

Selected publications

Uusitalo, V.; Horn, R.; **Maier, S.D.** (2022) Assessing Land Use Efficiencies and Land Quality Impacts of Renewable Transportation Energy Systems for Passenger Cars Using the LANCA® Method. Sustainability, doi:10.3390/su14106144.

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Eidesstattliche Erklärung

Hiermit versichere ich eidesstattlich, dass die vorliegende Arbeit von mir selbst, lediglich unter Benutzung der aufgeführten Literatur und ohne fremde Hilfe angefertigt worden ist.

Stuttgart, den Name

Land use and land use change are major drivers of biodiversity loss. A well-established tool for measuring such impacts throughout the life cycle of products and services is Life Cycle Assessment (LCA). Although valuable biodiversity impact assessment methodologies exist within LCA, they are still rarely used by companies and communities. This is mainly because existing methodologies are not globally applicable or provide insufficient decision support for LCA end-users.

This thesis therefore presents a new, globally operational method for analysing biodiversity impacts in LCA. The new Biodiversity Multi-Scale Assessment of Product Systems (Bio-MAPS) method is based on key ecological, conservation and technical requirements for LCA. It provides LCA end-users with a coherent framework for assessing the biodiversity impacts of product systems. Consequently, this methodology aims to mitigate negative impacts by identifying concrete actions at global, regional and local scales.

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