Ref. Ares(2024)8518671 - 29/11/2024

ForestPaths
Co-designing holistic forest-based policy pathways for climate change mitigation

D3.3 Biodiversity and ecosystem services modelling

28/11/2024

Authors:
Hanneke van ‘t Veen, Rob Alkemade (PBL)
Joanna Raymond, Calum Brown, Mark Rounsevell, Yongchao Zeng, Mohamed Byari, Almut Arneth (KIT)

Contributors to model development:
Alexandra Marques, Aafke Schipper, Michel Bakkenes, Paul Giesen, Martijn van der Marel (PBL)
Elizabeth Diaz-General, Reinhard Prestele, Ankita Saxena, Thomas Schmitt, Maximilian Tschol, Karina Winkler (KIT)

Funded by
the European Union

This project receives funding from the European Union’s Horizon Europe Research and Innovation Programme (ID No 101056755), as well as from the United Kingdom Research and Innovation Council (UKRI). Views and opinions expressed are those of the author(s) only and do not necessarily reflect those of the European Union or the European Commission. Neither the EU nor the EC can be held responsible for them.

Prepared under contract from the European Commission and the United Kingdom Research and Innovation Council.
Grant agreement No. 101056755
EU Horizon Europe Research and Innovation Action

Project acronym:	**ForestPaths**
Project full title:	**Co-designing Holistic Forest-based Policy Pathways for Climate Change Mitigation**
Project duration:	01.09.2022 – 28.02.2027 (54 months)
Project coordinator:	Dr. Hans Verkerk, European Forest Institute (EFI)
Call:	HORIZON-CL5-2021-D1-01
Deliverable title:	Biodiversity and ecosystem services modelling
Deliverable n°:	D3.3
WP responsible:	WP 3
Nature of the deliverable:	Report
Dissemination level:	Public
Lead partner:	PBL
Recommended citation:	van ‘t Veen, H., Raymond, J., Brown, C., Rounsevell, M., Zeng, Y., Byari, M., Arneth, A., Alkemade, R. (2024). D3.3 Biodiversity and ecosystem services modelling. ForestPaths project deliverable D3.3.
Due date of deliverable:	Month 27
Actual submission date:	28.11.2024

Deliverable status:

Version	Status	Date	Author(s)
1.0	Draft	01.11.2024	Hanneke van ‘t Veen, Joanna Raymond, Calum Brown, Mark Rounsevell, Yongchao Zeng, Mohamed Byari, Almut Arneth, Rob Alkemade
	Review	18.11.2024 22.11.2024	Hans Verkerk Cornelius Senf
2.0	Final	28.11.2024	Hanneke van ‘t Veen, Joanna Raymond, Calum Brown, Mark Rounsevell, Yongchao Zeng, Mohamed Byari, Almut Arneth, Rob Alkemade

Table of contents

Key takeaway messages

- We empirically assessed the effects of forest management and restoration on biodiversity with the aim to parameterize the GLOBIO model, a global biodiversity model for policy support, and the CRAFTY model, an agent-based model that is part of LandSyMM. Additionally, we improved ecosystem services components of the GLOBIO and CRAFTY models. We linked the GLOBIO and CRAFTY models by matching their respective forest management and land use classes to prepare for the simulation of exploratory scenarios and policy pathways.
- At a global scale, we find larger negative effects of forest plantations on biodiversity than of selective cutting, relative to natural forests. Additionally, we find larger negative effects of perennial tree crop plantations on biodiversity than of agroforests, relative to natural forests. Hence, at a global scale, we find that some types of forest management are more biodiversity-friendly than others.
- At the European scale, we find generally similar effects of plantations and selective cutting on biodiversity, in particular for animal biodiversity. Hence, we were unable to identify forest management types or practices that are more biodiversity-friendly than others at European scale, based on the available data. This highlights a need to move away from business as usual in Europe and consider broader ranges of management types and practices to achieve biodiversity goals.
- At a global scale, we find that biodiversity generally recovers faster in less degraded areas, such as former croplands and pastures, than in heavily degraded areas, such as old mining sites, irrespective of whether the forest is planted or regenerates naturally.
- In Europe, we find that animal biodiversity generally does not recovery after forest restoration and afforestation. Although we find that plant biodiversity recovers slowly upon natural forest regeneration in Europe, we do not observe biodiversity recovery upon forest restoration through planting. Hence, it appears that an inhibiting factor prevents biodiversity from recovering to reference levels in Europe.
- To further improve ecosystem services modelling, we created an ecosystem services module in GLOBIO to simulate changes in recreation opportunities across Europe based on the simulation of landscape aesthetic quality and accessibility. This has prepared us to simulate changes in recreation opportunities in response to changes in forest cover across policy pathways.
- The CRAFTY and GLOBIO models have been linked by matching the land use and land management classes distinguished in the respective models through careful examination of their definitions. Additionally, tests were conducted to use CRAFTY modelling output as input for the GLOBIO model to assess biodiversity responses to other drivers of biodiversity change, including climate change, nitrogen deposition, fragmentation and road disturbance. By establishing the link between the CRAFTY and GLOBIO models, we have prepared the models to simulate exploratory scenarios and policy pathways in which synergies and trade-offs between climate change mitigation, biodiversity conservation and the provisioning of ecosystem services can be assessed.

Summary

This deliverable presents improvements made in forest biodiversity and ecosystem services modelling in the GLOBIO and LandSyMM-EU models. Additionally, the deliverable presents an approach to link the GLOBIO and LandSyMM-EU models to facilitate the simulation of changes in biodiversity and ecosystem services in response to forest management and restoration across policy pathways developed in the ForestPaths project.

First, effects of forest management and restoration on biodiversity relative to the biodiversity of reference forests were determined through statistical analysis of biodiversity data. Global effects of forest management and restoration on biodiversity were determined using existing databases (e.g., PREDICTS, GLOBIO and a database from Kuipers et al. (2023)). In addition, a European-scale meta-analysis was conducted to provide a more detailed regional assessment and facilitating comparisons between global and European biodiversity impacts of forest management and restoration.

The global results on forest management revealed that biodiversity is generally less affected under low intensity forest management (e.g., selective cutting) than under high intensity forest management (e.g., timber plantations). Yet, the results also indicate that biodiversity can recover over time under certain types of intensive forest management, indicating that these systems can become more biodiversity-friendly over time. On the European scale, biodiversity responses were similar across both intensive and less intensive management types, particularly for animals and to a certain extent also plants. This highlights a need to move away from business as usual in Europe and consider broader ranges of management types and practices to achieve biodiversity goals.

For forest restoration, we find that biodiversity generally increases after afforestation and reforestation at the global scale, while in Europe animal biodiversity does not recover after afforestation and reforestation, while we find that plant biodiversity only recovers to a limited extent after reforestation through natural forest regeneration. This indicates that there are inhibiting factors that prevents biodiversity from recovering to levels found in natural old-growth forests in Europe. There is a need to further investigate these inhibiting factors and assess the potential to introduce additional measures to support species adapted to natural reference forests (e.g., assisted migration or active restoration practices that mimic natural regeneration) to foster biodiversity recovery in restoring forests.

Second, the biodiversity estimates for different types of forest management and restoration were linked to the land management/use classes distinguished in the LandSyMM-EU model, primarily its CRAFTY model component. To do so, the definitions of CRAFTY and GLOBIO land use/management classes were matched during an interactive workshop. We decided to fully integrate the effects of land management/use, as well as climate on biodiversity in the CRAFTY model. This allows CRAFTY to simulate effects of land management/use and climate change on biodiversity. Additionally, we decided to use CRAFTY land use/management output as input for the GLOBIO model to determine effects of additional drivers on biodiversity, including atmospheric nitrogen deposition, fragmentation and road disturbance.

Third, and finally, improvements were made in ecosystem services modelling. The recreational value component of GLOBIO was improved and tested. In CRAFTY, the range of services that agents provide was expanded to include flood control, soil erosion, landscape heterogeneity, rural employment and a set of cultural ecosystem services including recreation. Combined, the GLOBIO recreation model and CRAFTY annual ecosystem service supply can be used to assess the influence of changes in forest cover on recreation opportunities, and ecosystem service provision more broadly, in Europe and worldwide.

In summary, biodiversity and ecosystem service components of both the GLOBIO and LandSyMM models have been improved in this deliverable, and the models are linked and ready to simulate changes in biodiversity across scenarios developed in the ForestPaths project.

List of abbreviations

AFT	Agent functional type
BA	Basal area
CBD	Convention on Biological Diversity
CA	Cropland abandonment
CBD	Convention on Biological Diversity
CBS	Climate- and biodiversity-smart forestry and wood use
CC	Clear cutting and regrowth
CCo	Canopy cover
CH	Canopy height
DBH	Diameter at breast height
ESF	Ecosystem functions
ESS	Ecosystem services
EU	European Union
FC	Foliage cover
GDP	Gross domestic product
IN	Biodiversity intactness
LLC	Leaf litter cover
LLD	Leaf litter depth
MSA	Mean Species Abundance, an indicator for biodiversity intactness
NOAA	National Oceanic and Atmospheric Administration
PA	Pasture abandonment
Rcl	Active reclamation
Rco	Active reconstruction
PL	Forest plantations
PT	Perennial tree crop plantations
RI	Reduced-impact logging
RT	Retention forestry
SB	Abandoned slash-and-burn cropland
SC	Selective cutting
SI	Similarity
SP	Silvopasture
SR	Species richness
SS	Selection systems
TA	Total abundance
TD	Tree density
TR	Tree species richness
UC	Understory cover
UD	Understory density
UH	Understory height

1 Introduction

Forests and other wooded land cover about 31% of the world's land area (FAO 2020b). They play a crucial role in addressing the coupled climate and biodiversity crises. Forests capture carbon (Nunes et al. 2020) and simultaneously harbour a rich diversity of plant and animal species, as well as fungi and microorganisms (Muys et al. 2022). They provide habitat for many endangered and threatened species and provide a range of important ecosystem services, such as carbon sequestration and recreation (Betts et al. 2017).

Forest biodiversity is influenced by multiple environmental pressures, such as land use changes, forest management practices, landscape fragmentation, climate change, and pollution from substances like nitrogen and other acidifying compounds (EEA 2020). Projected shifts in temperature and precipitation patterns may alter ecosystem dynamics, habitat connectivity, species composition, and seasonal biological events (EEA 2020). Therefore, it is crucial to promote forest-based climate change mitigation, enhance forest adaptation to climate change, and ultimately protect and restore forest biodiversity.

Current global extraction of roundwood and wood fuel along with the conversion of forests into agricultural land drives deforestation (FAO 2020b) and forest degradation (Matricardi et al. 2020), posing serious threats to biodiversity (IPBES 2019). Only protecting biodiversity in conservation areas is not enough to reverse the global trend of biodiversity loss (Leclère et al. 2020). Therefore, it is essential to adopt more sustainable practices for the production and consumption of natural resources outside these protected areas (Laurance et al. 2012). For this reason, the Kunming-Montreal Global Biodiversity Framework, established by the Convention on Biological Diversity, aims to ensure that all land, including forested land, is managed using biodiversity-friendly practices by 2030 (CBD 2022).

In Europe, forests have been increasing since the 1950s (Fuchs et al. 2013). Between 1992 and 2015, European forests expanded by 1.4% due to (natural) expansion of forest on abandoned agricultural lands and afforestation (Palmero-Iniesta et al. 2021) and currently forests cover around 38% of the land in Europe (Ceccherini et al. 2020). This increase in forested area has not led to significant gains in biodiversity (EEA 2020). Across all land use types in Natura 2000 areas, only 50% of bird species and 27% of non-bird species have a favourable conservation status (EEA 2020), and biodiversity overall declined significantly in Europe (Watt et al. 2007). Part of the decline of biodiversity in European forests can be attributed to the loss of old-growth forests and primary forests, which now only account for approximately 3% and 0.7% of Europe's forested land, respectively (Alessia et al. 2024; Sabatini et al. 2018, 2020). Additionally biodiversity loss in European forests is linked to forest management, which has suppressed natural dynamics in forest ecosystems, has simplified forest structures, enhanced the share of relatively young forests, and promoted the use of relatively few tree species (Paillet, Bergès, HjÄltén, et al. 2010). Even though a gradual rise in biodiversity-friendly management practices has been observed in Europe, such as increases in the amount of deadwood and diversification in tree species, (FOREST EUROPE 2020), degrative wood extraction practices continue contribute to biodiversity loss in Europe (Paillet, Bergès, HjÄltén, et al. 2010). Therefore, the European Union Forest Strategy was constructed, which seeks to enhance biodiversity conservation through the promotion of sustainable forest management and protection of old-growth forests (EU 2021).

In addition to the impacts of forest management practices, biodiversity is also threatened by land degradation on areas that were once forested in Europe and around the globe (Pacheco et al. 2018). This land degradation results in a significant loss of biodiversity and ecosystem services across one to six billion hectares of degraded land worldwide (Gibbs and Salmon 2015). To address and mitigate this issue, the Kunming-Montreal Global Biodiversity Framework has established a global target to restore 30% of degraded ecosystems by 2030 (CBD 2022). Furthermore, nations and regions have committed to restoring degraded lands within their territories through the Bonn Challenge, which aims to rehabilitate 150 million hectares of land globally (Dave et al. 2019).

In Europe, over 80% of habitats in the Natura 2000 network are in poor condition, which severely impacts biodiversity (EEA 2020). To restore land and prevent further habitat degradation, the European Union has introduced the Nature Restoration Law. This legislation sets specific targets for ecosystem restoration to boost biodiversity and forests are among the ecosystems prioritized for restoration under this law (EU 2024). The plan includes a target to plant 3 billion additional trees by 2030 to improve biodiversity and restore ecosystems and urges member states to restore at least 80% of degraded lands by 2050. These measures aim to enhance species populations, particularly of indicator bird species (EU 2024).

Forests are not only important for biodiversity, but also provide multiple environmental, societal and cultural benefits, including as a source for renewable raw materials, erosion and flood control, and as a place for recreation (Forest Europe 2014). How forests are managed and the condition they are in also influence these benefits, or ecosystem services (Millennium Ecosystem Assessment 2005). For example, extension of rotation periods between harvests has been shown to have a positive effect on soil protection, while extensive management can provide better conditions for water and recreational opportunities (Puettmann, Coates, and Messier 2012; Başkent and Kašpar 2022). There can be trade-offs and synergies between the provisioning of different types of ecosystem services, as well as trade-offs and synergies between ecosystem services and biodiversity (Zhang et al. 2023). For example, trade-offs have been observed between carbon sequestration and recreation, as landscape aesthetic quality is not always high in fast growing forests with high carbon sequestration potential (Iversen et al. 2019). Additionally, trade-offs occur between biodiversity and carbon sequestration in intensive pine and eucalyptus plantations, while synergies occur between carbon storage in old-growth forests and biodiversity (Onaindia et al. 2013). Careful consideration is required to balance the ecosystem service benefits of specific management practices with biodiversity.

The aim of this deliverable is to improve the modelling of forest biodiversity and ecosystem services in response to forest management and restoration to facilitate the simulation of changes in forest biodiversity and ecosystem services over space and time in response to policy pathway scenarios. Although effects of forest management and restoration on biodiversity and ecosystem services has been unravelled in previous global (Chaudhary et al. 2016; Crouzeilles et al. 2016; Onaindia et al. 2013) and European (Paillet, Bergès, HjÄltén, et al. 2010) studies, these have not yet been included in global and European scale models, such as the GLOBIO and LandSyMM models used in the ForestPaths project. In Chapter 2 and 3 of this report, we first describe effects of forest management and restoration on biodiversity (i.e., species diversity and abundances) and ecosystem services to improve parameterization of the GLOBIO and LandSyMM models. In Chapter 4, we describe the complete set of models that will be used to analyse biodiversity and ecosystem services. In Chapter 5 we describe the way in which the models will be linked together to simulate changes in biodiversity and assess trade-offs and synergies with climate change mitigation and ecosystem services provisioning.

The next stages of the project will focus on developing policy pathways that explore trajectories of European forest management. Simulation of policy pathways within the linked GLOBIO-LandSyMM integrative modelling framework will reveal how the forest-based sector can contribute to climate change mitigation and adaptation while also conserving forest biodiversity and maintaining ecosystem services. To comprehensively understand how the forest-based sector can achieve these objectives, it is essential to examine the effects of forest management and restoration on biodiversity both within European forests and globally.

2 Methods used for model improvements

In Section 2 we describe the methodology used to adapt and improve the models that will be used to simulate changes in biodiversity and ecosystem services in response to policy pathways developed in WP5. The models will be linked to evaluate consequences of policy options for biodiversity and ecosystem services across different scenarios. Extensive model development has been undertaken to include effects of different types of forest management and forest restoration on biodiversity. Hereby, we focussed on the effects of forest management and restoration on species diversity and abundances of species; hence, we did not consider other important aspects of biodiversity, such as genetic diversity, functional diversity and beta-diversity. We performed both a global meta-analysis and a meta-analysis specific for the European context to estimate effects of forest management and restoration on biodiversity. We used the results of these studies to incorporate the effects of different types of forest management on biodiversity in the LandSyMM model, specifically the CRAFTY module.

2.1 Forest management and biodiversity

We evaluated the effects of forest management on biodiversity, measured as species diversity and abundances of species, through a two-step process. First, we merged three existing global biodiversity databases and supplemented the resulting combined database with information on forest management, which we derived from the underlying data sources (e.g., research papers and reports). We then categorized this data into generic global forest management systems, allowing us to estimate biodiversity levels in managed forests relative to those of natural reference sites, as well as track changes in biodiversity over time. Second, we performed a meta-analysis specifically focused on Europe to address a data gap identified in our global dataset. This European-scale meta-analysis provided us more accurate estimates of the effect of common forest management systems on biodiversity in Europe.

2.1.1 Global effects of forest management on biodiversity

2.1.1.1 Data preparation

We sourced data from three established biodiversity databases: PREDICTS (Hudson et al. 2017), a database developed by Kuipers et al. (2023) and GLOBIO (Alkemade et al. 2009). These databases provided information on the populations and assemblages of animal species (amphibians, birds, insects, mammals, and reptiles) and plant species across various forest management systems, compared to natural reference sites—defined as forests showing no evident signs of human activity for at least 80 years. Our definition of natural reference sites is in line with the FAO definition of primary forests, i.e., “naturally regenerating forest of native tree species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed” (FAO 2020a). Each database contained abundance data at the species or genus level, derived from individual empirical studies conducted worldwide. Each entry in the databases represents a distinct source, which may encompass multiple studies differentiated by location and/or the year of data collection. This abundance data enabled us to perform pairwise comparisons between the biodiversity of managed forests and that of reference sites.

We enhanced the three biodiversity databases by incorporating additional data on the forest management systems from which the original data were collected (Appendix A). Specifically, we revisited the original sources to supplement the databases with detailed information on various management practices, such as cutting strategies. We also included the duration of management, on which data was available for about half of the animal biodiversity studies. We classified the forest management systems into seven categories: (i) agroforests (AG), (ii) clear-cutting and unassisted natural regeneration of forests (CC), (iii) perennial tree crop plantations (PC), (iv) forest plantations (PL), (v) reduced-impact logging (RI), (vi) selective cutting (SC), and (vii) silvopasture plantations (SP) (see Table 1 for detailed descriptions). This classification was developed to offer a broad, yet detailed, overview of various forest management approaches for both wood and food production, while considering data availability. Forest management systems for wood production differ mainly in differ in the way forests regenerate (i.e., natural succession or planting) and the degree to which canopy is removed in a single cutting event. We expect both forest regeneration and the degree to which canopy is removed to affect biodiversity, as this was observed in previous studies (Crouzeilles et al. 2017; Chaudhary et al. 2016) .The management systems were adapted from Chaudhary et al. (2016) and Crouzeilles et al. (2016), who identified the most prevalent timber and non-timber forest management practices. Reduced-impact logging, selective cutting, and clear-cutting were included because of their widespread use in wood production globally (Chaudhary et al. 2016). Forest plantations were added because their global prevalence is increasing (Brockerhoff et al. 2008) and comparing biodiversity in natural versus plantation forests helps to understand the effects of transitioning from natural to planted forests. We also considered two types of agroforestry, agroforests and silvopasture, as well as perennial tree crop plantations because of their growing global extent (Fitzherbert et al. 2008; Zomer et al. 2022), which raises questions about their potential to reduce biodiversity loss in agricultural landscapes. Moreover, these systems are frequently classified as forests under many definitions, such as the FAO definition of forests (FAO 2020a).

Table 1. This study categorizes various forest management systems, drawing from those identified by Chaudhary et al. (2016) and Crouzeilles et al. (2016). The description of "silvopasture" has been adapted from Roellig et al. (2016), where it is referred to as "wood-pasture". The main characteristics indicate the main points on which the systems differ from each other.

Management type		**Main characteristics**	**Description**
AG	*Agroforests*	- Crops growing underneath tree canopy	Agroforests are a sub-type of agroforestry. Agricultural land where crops are cultivated beneath or alongside trees that provide wood, food, energy, or other non-timber forest products, or enhance crop growth. Agroforest systems can be established in two main ways: either by selectively harvesting from naturally regenerated forests with diverse native tree species to allow crops to grow beneath the remaining trees, or by planting or seeding trees, whether native or exotic, on agricultural land to develop a new agroforest. The density and diversity of trees in agroforest systems vary widely globally, as do the management practices employed to regulate tree growth, such as thinning and cutting.
CC	*Clear cutting and regrowth*	- Removal of all trees in a forest area - Natural regeneration of the forest	The complete removal of all trees within a forest area for wood production, either for material or energy uses, after which the forest regenerates naturally. As forests regenerate naturally, seeds are provided by surrounding forests and through wind and animals. This leads to a uniform, even-aged forest stand structure. It is applied in naturally regenerating forests, predominantly composed of native tree species. The rotation periods for this practice can vary widely, ranging from 20 to over 150 years.
PC	*Perennial tree crop plantation*	- Trees that produce crops for food or energy production are planted on agricultural land	Land where one or a few species of trees, either native or exotic, are intentionally planted or seeded to produce food and/or energy, such as oil palm or banana. This practice creates a uniform, even-aged stand structure. Management activities, including the application of fertilizers and irrigation, are commonly employed to support and enhance crop growth.
PL	*Forest plantations*	- Trees are planted - Replating occurs after plantations are harvested	Land where one or multiple species of trees, whether native or exotic, are planted or seeded specifically for wood production, either for material or energy uses. Trees are harvested through clear-cutting, after which the area is replanted. This approach results in a uniform, even-aged forest stand structure. Rotation periods for these plantations range from 10 years to over 150 years. During the rotation, management practices such as

			thinning and fertilizer application may be implemented, though this input can vary.
RI	*Reduced-impact logging*	• Selective harvesting of trees, resulting in an uneven forest structure • Forest regenerates naturally • Less trees are harvested than under selective cutting	Carefully managed and controlled harvesting of trees for wood production, either for material or energy uses. This practice is conducted over extended rotation periods in naturally regenerating forests with diverse native tree species. The primary goal is to sustain a near-natural forest environment and preserve biodiversity, while minimizing environmental impacts.
SC	*Selective cutting*	• Selective harvesting of trees, resulting in an uneven forest structure • Forest regenerates naturally • More trees are harvested than under reduced-impact logging	The extraction of trees for wood production, either for material or energy uses, based on specific harvesting guidelines and preferences. This method is applied in naturally regenerating forests predominantly composed of native tree species. Compared to reduced-impact logging, selective cutting involves harvesting at higher intensities and intervals.
SP	*Silvopasture*	- Livestock grazing underneath the canopy	Silvopasture is a sub-type of agroforestry. Land where one or more tree species, either native or exotic, are planted or seeded for wood and/or energy production, while livestock such as cows or sheep graze between the trees. Harvesting involves the complete removal of all trees in an area, followed by replanting or natural regeneration through coppicing to establish a new stand. This practice creates a uniform, even-aged forest structure. Rotation periods typically range from 20 years to over 150 years. During the rotation, management practices, such as thinning and the application of fertilizers, may be employed, although their use can vary.

We retrieved IUCN Red List data for all taxonomic groups included in this study from the official IUCN Red List website (https://www.iucnredlist.org/, accessed July 19th, 2024). This dataset encompasses, among other details, the binomial scientific names of threatened species and their assessed risk of global extinction, ranging from "Least Concern" to "Extinct" (IUCN 2024). To integrate this information in our biodiversity database, we matched the binomial names of species in our database with those listed in the IUCN Red List.

The references of all data sources in the databases can be found in Appendix A.

2.1.1.2 Biodiversity indicators

We assessed local animal and plant biodiversity using four indicators to evaluate how species assemblages in managed sites compare to those in reference sites: (i) intactness (IN), (ii) relative richness (SR), (iii) compositional similarity (SI), and (iv) relative total abundance (TA). The selection of these four indicators was informed by previous research, which suggests that a combination of species richness, abundance, and indicators related to intactness is essential for capturing changes in biodiversity in response to anthropogenic pressures (Kuipers et al. 2023; Santini et al. 2017; Vačkář et al. 2012). Intactness assesses whether species specifically adapted to reference sites are lost or if their abundances decline. Compositional similarity helps determine if a loss of intactness is due to a reduction in species specifically adapted to reference forests or a decrease in their abundance. Relative richness is compared with intactness and similarity to discern if species adapted to reference sites are being replaced by other species, or if forest management leads to a general loss of species. Relative total abundance is evaluated alongside intactness to identify whether declines in abundance are limited to species specifically adapted to reference sites or if there is a broader reduction in species abundances upon a transition from reference forests to managed forests.

We computed biodiversity intactness using the mean species abundance (MSA) indicator (Alkemade et al. 2009; Kuipers et al. 2023; Schipper et al. 2020):

$$R_{itp} = \frac{G_{it}}{G_{ip}} \qquad \text{(1a)}$$

$$IN_i = \frac{\sum R_{itp,1}}{N_p} \qquad \text{(1b)}$$

where, *G* are species or genera level abundances in managed sites (*t*) and reference sites (*p*) of those species or genera that occur in reference sites, *R* is the ratio of the abundances of species or genera of managed sites (*t*) divided by those of reference sites (*p*), *IN* is intactness expressed as the mean species abundance of study *i*, and *N* is the number of species or genera present in the reference sites (Alkemade et al. 2009; Schipper et al. 2020). We truncated *R*-values at 1 to assure that potentially high abundances of generalist or opportunistic species did not inflate intactness values to prevent an overestimation of intactness (Alkemade et al. 2009; Schipper et al. 2020). Finally, we calculated the arithmetic mean over the *R*-values of all species for each study to produce one intactness value per forest management system per study (Alkemade et al. 2009; Schipper et al. 2020).

We assessed effects of forest management on species richness by calculating a log-response ratio of the species richness of management systems and reference sites:

$$SR_i = \ln\left(\frac{S_{ti}}{S_{pi}}\right) \qquad \text{(2)}$$

where, *SR* is the log-response ratio of species richness in study *i*, and *S* is the number of species or genera in managed sites (*t*) or in reference sites (*p*). We used the log-response ratio to adjust for the right-skewed distribution of the richness data.

We calculated species similarity between managed and natural reference sites using the Sørensen Similarity Index (Sorensen 1948):

$$SI_i = \frac{2 \cdot T_{tpi}}{(2 \cdot T_{tpi} + F_{tpi})} \qquad \text{(3)}$$

where, *SI* is the Sørensen Similarity Index of study *i*, *T* is the number of species or genera that are the same across reference (*p*) and managed sites (*t*), and *F* is the number of species or genera that are different between reference and managed sites.

Finally, we evaluated the impact of forest management on relative total abundance by calculating the log-response ratio between the total abundance of all species in managed sites and the total abundance of all species in reference sites:

$$TA_i = \ln\left(\frac{\sum_S^M A_{tsi}}{\sum_S^N A_{psi}}\right) \qquad \text{(4)}$$

where, *TA* represents the log-response ratio of summed abundances for study *i*, and *A* denotes the total abundance of species *s* in managed sites (*t*) or reference sites (*p*). *M* and *N* are the numbers of species in the restoration (*t*) and reference (*p*) sites, respectively. We used the log-response ratio to adjust for the right-skewed distribution of the abundance data.

2.1.1.3 Effects of forest management on biodiversity over time

To evaluate the effect of forest management on biodiversity and its temporal changes across various taxonomic groups and geographical regions, we employed generalized linear mixed-effects modelling.

2.1.1.3.1 Response variables

We constructed generalized linear mixed-effect models to analyse the effects of forest management on biodiversity using the four biodiversity indicators (i.e., intactness, relative richness, similarity, and relative total abundance) as response variables. Initially, we created models combining all animal taxa and all plant taxa, respectively. Subsequently, we developed separate models for insect, bird, herpetofauna, and mammal diversity to explore taxonomic differences in responses to forest management. Amphibians and reptiles were analysed together as herpetofauna because of to their similar ecological roles and data constraints. Additionally, we examined threatened species by building linear mixed-effect models with the relative richness, similarity, and relative total abundance for IUCN Red List animal species as response variables. Different combinations of IUCN categories were included as response variables to determine if forest management effects varied across these groups. The combinations included (i) “Least Concern” to “Critically Endangered,” (ii) “Near Threatened” to “Critically Endangered,” and (iii) “Vulnerable” to “Critically Endangered.” Due to data limitations, further categorization was not performed and threatened plant species were excluded from this analysis.

2.1.1.3.2 Fixed effects

Our models incorporated several fixed effects: (i) forest management systems, (ii) time since management initiation, (iii) continent, and (iv) taxonomic group. Initially, we assessed the overall effect of different forest management systems on biodiversity. We then analysed the effect of time since the start of management on biodiversity for each management system separately, defining it as “time since harvesting” for selective and clear cutting, and “time since establishment” for other systems. Due to limited temporal data, we excluded silvopastoral and reduced-impact logging systems from our temporal analyses (<5 data points). We focused on overall animal biodiversity due to insufficient data for plant and threatened species and insufficient data to distinguish changes over time for different animal taxonomic classes. For perennial tree crop plantations, we removed the outlier of 150 years post-establishment to address data skewness. Time since establishment for agroforest systems was log-transformed to account for observed non-linear biodiversity changes. We also built models with both forest management systems and animal taxonomic class as fixed effects to test if taxonomic class significantly explains the variation on biodiversity besides management. Finally, we built models with both forest management and continent as fixed effects, to assess whether spatial variations affected biodiversity responses besides forest management.

2.1.1.3.3 Model type

We built two types of generalized mixed-effects models: (i) beta-regression models with a logit-link function for intactness and similarity, using the *glmmTMB* function from the Temperate Model Builder R package (Brooks et al. 2023), and (ii) Gaussian linear mixed-effects models for relative richness and total abundance, employing the *lmer* function from the lme4 R package (Bates et al. 2015). We used beta-regression models to estimate intactness and similarity because the predicted estimates of these indicators cannot be lower than 0 or higher than 1. Beta-regression does not allow for values of 0 and 1 (intactness and similarity can be 0 or 1), thus we transformed intactness and similarity to values between 0 to 1 using the Smithson-Verkuilen transformation (Smithson and Verkuilen 2006). We used Gaussian linear mixed-effect models to estimate relative richness and total abundance because the predicted estimates of these indicators can be higher than one and we determined that the distribution of the response variables was Gaussian.

2.1.1.3.4 Random effects and weights

For models estimating overall animal diversity, we included nested random effects of source-study combinations (1|Source/Study) per management system to account for potential differences among datasets and non-independence of data. For plant, insect, bird, herpetofauna, mammal, and threatened diversity models, we included only the source (1|Source) as a random effect because sources typically included only one study. We applied the square root of the total number of species per study in reference forests as weights on the biodiversity indicators, hereby increasing the influence of studies with a higher species count and reducing the impact of those with fewer species. Square-root transformation of the number of species was used to mitigate data skewness.

2.1.1.3.5 Model selection

To determine if incorporating taxonomic class as an additional fixed effect improved model performance, we compared Bayesian Information Criteria (BIC) using ANOVA. Models with the lowest BIC values were considered the most effective. This approach helped us assess whether taxonomic diversity contributed to explaining variations in biodiversity beyond forest management. Similarly, we evaluated whether adding continent as an additional fixed effect improved model performance, indicating whether spatial variability explained additional variation in biodiversity and whether global modelling results could be generalized across geographical regions.

2.1.1.3.6 Model evaluation and visualization

We assessed model assumptions using residual plots, histograms, and the Shapiro-Wilk test to check for normality. Omnibus tests and likelihood-ratio chi-square tests were conducted to evaluate if explained variance exceeded unexplained variance. Results were presented by back-transforming log-response ratios, as well as back-transforming estimates for intactness and similarity based on logit-link distributions.

2.1.1.3.7 Post-hoc tests

To evaluate differences between forest management systems for wood and food production, we conducted Tukey post-hoc tests using the *glht* function from the multcomp R package (Hothorn et al. 2024). We used post-hoc tests to assess the difference between forest management systems in which wood is extracted from natural forests (i.e., selective cutting, reduced-impact logging and clear cutting) and systems in which wood is extracted from plantations (i.e., forest plantations). We repeated this to assess the difference between forest management systems in which food is produced under the canopy (i.e., agroforests and silvopasture) and those in which food is produced in plantations (i.e., perennial tree crop plantations). We did not compare management systems for wood production with those for food production because these systems do not fulfil the same function and hence are not interchangeable.

2.1.1.3.8 Sensitivity analyses

We tested the sensitivity of models with intactness, relative richness, similarity or relative total abundance of overall animal or plant biodiversity as response variables and the seven management as fixed effects to variations in the data pool. We did this by randomly removing the data from 10 to 40% of the sources in the database twenty times and calculating the variance over the predicted biodiversity indicator estimates for all indicators. Larger variations in predicted biodiversity indicator estimates indicated higher sensitivity of models to variations in the data pool. Finally, we conducted principal component analyses (PCA) and produced Spearman correlation matrices to assess correlations between biodiversity intactness, similarity, relative richness and total abundance for animals and plants overall, as well as for insect, bird, herpetofauna and mammals separately. The outcome of these analyses provided an indication of the complementarity and redundancy of our biodiversity indicators.

Data analyses were conducted using R version 4.3.1 (R 2023), and spatial data visualization was performed with QGIS (QGIS 2023).

2.1.2 Effects of forest management on biodiversity in Europe

2.1.2.1 Meta-analysis

We conducted a meta-analysis to assess the effects of forest management on biodiversity in Europe. First, we conducted a search in the search tool Scopus (https://www.scopus.com/home.uri, last accessed 26-07-2024) to find relevant papers on forest management in Europe. To do so, we created a search string that included keywords about (i) biodiversity (e.g., “abundance”), (ii) natural reference forest sites (e.g., “pristine” and “old-growth”), (iii) forest management (e.g., “plantation”), and (iv) European countries (e.g., “Spain”) (see Appendix C1 for the full search string). Second, we screened the relevance of all scientific papers Scopus provided us for inclusion. To be included, publications needed to provide (i) species-level abundance data for a natural reference (i.e., forests with no clearly visible indications of human activities for at least 80 years) and a managed forest site in Europe, and (ii) a clear description of the forest management system. Our definition of natural reference sites is in line with the FAO definition of primary forests, i.e., “naturally regenerating forest of native tree species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed” (FAO 2020a). Third, we extracted data on species-level abundances from the publications that fulfilled our requirements and added it to a database. We also added time since the start of forest management to this database. We included each publication in the database as a single source. Each source may contain one or more studies, which were distinguished from each other based on their location and/or the year in which data was gathered. Fourth, we categorized the forest management data into four systems that differ in the manner in which forests regenerates (i.e., natural succession or planting) and the degree to which canopy is removed in a single cutting event: (i) clear cutting and regrowth (CC), (ii) forest plantations (PL), (iii) retention forestry (RT), and (iv) selection systems (SS) (see Table 2 for descriptions of all systems). Both the manner of forest regeneration (Crouzeilles et al. 2017) and the degree to which canopy (Chaudhary et al. 2016) is removed has been shown to affect biodiversity in previous studies. The final set of management systems emerged out of an effort to provide a generalized, yet comprehensive overview of different forest management types and data availability. We adapted the clear cutting and regrowth, retention forestry and selection systems descriptions from the management system descriptions of Chaudhary et al. (2016) for temporal and boreal regions. We adapted the multipurpose plantation description from the combined objective forestry description of Duncker et al. (2012) but adjusted it so it always includes a form of tree planting. There are other systems that could have been considered, such as shelterwoods, but unfortunately insufficient data was found in these systems to assess the effect these systems have on biodiversity.

The references of the sources from which we gathered data can be found in Appendix C.

Table 2. Names and descriptions of forest management systems distinguished in this study. The main characteristics indicate the main points on which the systems differ from each other. The descriptions of the clear

cutting and regrowth, retention forestry and selection systems were adapted from Chaudhary et al. (2016) and the description of the multipurpose plantations system was adapted from Duncker et al. (2012).

Management type		Main characteristics	Description
CC	*Clear cutting and regrowth*	- Removal of all trees in a forest area - Natural regeneration of the forest	Systematic removal of all trees within a forest area for wood production (either for material or energy uses), after which forests regenerates naturally, leading to a uniformly aged forest stand. As forests regenerate naturally, seeds are provided by surrounding forests and through wind and animals. Harvesting occurs in forests that naturally regenerate, with rotation periods varying from 20 years to over 150 years. The main difference between clear cutting and regrowth and multipurpose plantations is that the forest regenerates naturally in the former and is planted in the latter.
PL	*Forest plantations*	- Trees are planted - Replanting occurs after plantations are harvested	Land where one or more tree species, whether native or exotic, are planted or seeded for wood production, either for material or energy uses. Trees are harvested by clearing all trees from an area, followed by replanting. This creates a regularly structured, even-aged forest stand. Rotation periods can range from 20 to over 150 years. Management practices, including thinning and fertilizer application, may be applied throughout the rotation. These plantations are often utilized for additional purposes, such as recreation and foraging. The main difference between clear cutting and regrowth and forest plantations is that the forest regenerates naturally in the former and is planted in the latter.
RT	*Retention forestry*	- A small number of individual trees or a group of trees are left after cutting - Remaining trees are clear cut - Natural regeneration of the forest	Retention forestry is similar to clear cutting and regrowth (see the definition of CC) but instead of removing all trees from a site, either individual trees or a small group of trees are left to conserve understory vegetation, maintain some of the forest structure, and/or to assure that seed sources are provided so that forests can regenerate naturally. Retention forestry differs from selective logging because more trees are removed in a single cutting event.
SS	*Selection systems*	• Selective harvesting of trees, resulting in an uneven forest structure • A combination of natural regeneration and tree planting occurs	Harvesting of trees for wood production is conducted according to established rules or preferences aimed at maintaining an uneven-aged forest structure. This practice occurs in forests predominantly composed of native tree species where trees are planted but also some natural regeneration of forests may occur. Trees are selectively removed, either as individual specimens or in groups, which results in a heterogeneous stand structure.

2.1.2.2 Biodiversity indicators

See Section 2.1.1.2 for a description of the biodiversity indicators used to calculate effects of forest management on biodiversity in Europe.

2.1.2.2.1 Rationales for the use of natural forests as reference in Europe

Even though European forests have been managed for centuries and very few areas of primary forest remain (Sabatini et al. 2018), we used natural forests as reference in our European scale study (i.e., forests with no clearly visible indications of human activities for at least 80 years). We decided not to change the definition of our reference sites for several reasons. First, the only alternative to taking natural forests as reference is taking a type of managed forest as reference or forests that have recently been managed. Forest disturbances resulting from management of forests are variable (e.g., varying rates of tree harvesting, varying intensities of thinning) and affect biodiversity in varying ways (Paillet, Bergès, Hjältén, et al. 2010). Therefore, taking a (recently) managed forest as reference would not have allow us to clearly distinguish effects of management on biodiversity across Europe, as the results would not only have been influenced by variations in the effects of management on biodiversity in the managed sites but also by those in the reference sites (as biodiversity in those sites would also have been influenced by management). Second, even though it may be unlikely that large areas of forests in Europe return to their natural state in the near future, the comparison to the natural state of ecosystems is useful because it shows us the extent to which biodiversity has been lost in response to forest management over the past centuries. By transitioning from a natural forest as reference to a managed forest as a reference, we would shift the baseline from biodiversity levels under natural conditions to biodiversity levels under anthropogenic conditions. This would result in an underestimation of the true magnitude of the effect humans have had on forests in Europe, potentially resulting in a “shifting baseline” under which a downgrade of perceived normal biodiversity conditions occurs (Jones et al. 2020). Arguably, this could result in a loss of incentive to protect the small area of remaining primary and old-growth forests in Europe. Third, as there are abundant natural reference forests outside of Europe, it would be undesirable to shift from natural forests as reference to (previously) managed forests as reference to measure effects of forest management on biodiversity in those areas. Using (previously) managed forests as a reference in Europe and primary forests outside of Europe as reference sites would likely result in a lower perceived effect of forest management on biodiversity in Europe in comparison to the rest of the world. Fourth, and finally, although it may be unlikely that large areas of forests in Europe return to their primary state in the near future, we may still aspire to restore forests to their natural state and increase the area of primary forests in Europe on the long term. If we shift the baseline from natural to managed forests, we may potentially risk losing incentive for such aspirations.

2.1.2.3 Effects of forest management on biodiversity over time

To estimate the effect of forest management on biodiversity and evaluate how these effects evolve over time, we utilized generalized linear mixed-effects modelling. Data analysis was performed using R4.3.1 (R 2023), and spatial data was visualized using QGIS (QGIS 2023).

2.1.2.3.1 Response variables and fixed effects

We built generalized linear mixed-effect models with intactness, relative richness, similarity and relative total abundance as response variables, respectively, to assess effects of forest management on animal and plant biodiversity. We built models with the same response variables to assess the effects of forest management on insect biodiversity. Not enough data was available to analyse effects of forest management on other animal classes (data was only available for birds and reptiles for forest plantations). As fixed effects we considered: (i) the forest management systems, and (ii) the time since the start of management. First, we built models with only the forest management systems as a fixed effect to estimate overall effects of each of the forest management systems on biodiversity. For animal diversity not enough data was available (1 study) on retention forestry, thus we only assessed effects of the remaining tree systems on animal biodiversity. Second, we built models with time since the start of management as a fixed effect for each forest management system separately. We only assessed effects over time for clear cutting and multipurpose plantation systems, because not enough data was available (<5 data points) for the remaining systems. We defined time since the start of management as “time since logging” for clear cutting and “time since establishment” for the multipurpose plantation systems. Due to data limitations, we were unable to assess effects of biodiversity over time in selective cutting and retention systems. We built separate models per management system with time since the start of management as fixed effect because we wished to understand and compare the different responses of biodiversity to the varying types of forest management over time. We only studied changes in overall animal biodiversity over time because of insufficient time points for plants per management system.

2.1.2.3.2 Model type

We built two different types of generalized mixed-effect models: (i) beta-regression models with a logit-link function for intactness and similarity response variables using the Temperate Model Builder R-function *glmmTMB* (Brooks et al. 2023), and (ii) Gaussian linear mixed-effect models for relative richness and total abundance using the *lmer* function of the R-package *lme4* (Bates et al. 2015). We used beta-regression models to estimate intactness and similarity because the predicted estimates of these indicators cannot be lower than 0 or higher than 1. Beta-regression does not allow for values of 0 and 1 (intactness and similarity can be 0 or 1), thus we transformed intactness and similarity to values between 0 to 1 using the Smithson-Verkuilen transformation (Smithson and Verkuilen 2006). We used Gaussian linear mixed-effect models to estimate relative richness and total abundance because the predicted estimates of these indicators can be higher than one and their distribution was Gaussian.

2.1.2.3.3 Random effects and weights

We included sources (1|Source) per management system as random effect to account for potential differences among datasets and data non-independence. The study was not included as a random effect because typically only one study was conducted per source. Biodiversity indicator values were weighted by the square root of the total number of species per study in reference forests, which enhances the influence of studies with a higher number of species and diminishes the impact of those studies with fewer species. The square-root transformation of species counts was used to address data skewness.

2.1.2.3.4 Model evaluation and visualization

Normality was assessed through residual plots, histograms, and the Shapiro-Wilk test. Omnibus tests were conducted to estimate whether explained variance surpasses unexplained variance. The log-response ratios of species richness and the estimates of intactness and similarity, based on logit-link distributions, were back-transformed for result presentation.

2.1.2.3.5 Correlations between biodiversity indicators

Principal component analyses (PCA) and Spearman correlation matrices were employed to examine correlations between biodiversity indicators such as intactness, similarity, relative richness, and total abundance for both animals and plants. This analysis provided insights into the complementarity and redundancy of the biodiversity indicators used.

2.2 Forest restoration and biodiversity

To evaluate how biodiversity changes in response to forest restoration, we followed a two-step approach that was similar to our analysis of forest management effects on biodiversity. First, we merged two existing global biodiversity databases and supplemented the resulting combined database with information on forest restoration, which we derived from the underlying data sources (e.g., research papers and reports). This dataset was then classified into generic global forest restoration types, enabling us to estimate biodiversity levels of restoring forests, as well as changes in biodiversity in response to forest restoration over time. Second, we conducted a European-scale meta-analysis specifically. This European-scale meta-analysis provided more precise estimates of the biodiversity levels of restoring forests and changes in biodiversity in response to forest restoration across the European continent.

2.2.1 Global changes in biodiversity under forest restoration

2.2.1.1 Database integration and expansion

We obtained data from two existing global biodiversity databases (i.e., PREDICTS (Hudson et al. 2017), GLOBIO (Alkemade et al. 2009)) that contained data on animal (i.e., amphibians, birds, insects, mammals, and reptiles) and plant species populations or assemblages in sites subjected to forest restoration as compared to natural reference sites (i.e., forests with no clearly visible indications of human activities for at least 80 years). Our definition of natural reference sites is in line with the FAO definition of primary forests, i.e., “naturally regenerating forest of native tree species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed” (FAO 2020a). Each database aggregates species- or genus-level abundance data from numerous empirical studies conducted globally. Each paper in these databases represents a single source, which may contain multiple distinct studies, defined by their geographical location or the year in which the data was collected. The species abundance data enabled direct pairwise comparisons between biodiversity levels in restoring forests and natural reference forests, which were defined as native forests free from significant human disturbance for at least 80 years.

To further enhance these global databases, we revisited the original studies and incorporated additional details specific to forest restoration in the databases (Appendix E). This included data on factors such as previous land use and the time since the start of restoration activities. We identified five main types of forest restoration: (i) cropland abandonment (CA), (ii) pasture abandonment (PA), (iii) reclamation (Rcl), (iv) reconstruction (Rco), and (v) abandoned slash-and-burn cropland (SB) (see Table 3 for a detailed description of each type). The restoration types mainly differ in prior land use and the manner of afforestation or forest regeneration, whether through active tree planting or natural regeneration; previous studies showed that both mechanisms can affect biodiversity (Crouzeilles et al. 2017; Meli et al. 2017). The set of restoration types emerged out of an effort to provide a generalized, yet comprehensive overview of different forest restoration types and data availability. Cropland abandonment, pasture abandonment, and abandoned slash-and-burn agriculture represent passive restoration, where forests regenerate naturally without human intervention. In contrast, reclamation and reconstruction involve active restoration efforts, including tree planting and other human-led initiatives (Stanturf, Palik, and Dumroese 2014). These categories were adapted from the classification proposed by Stanturf et al. (2014), who outlined four primary active forest restoration strategies based on a systematic review. Due to data limitations, we were unable to include assessments of biodiversity in the rehabilitation (restoring ecosystem composition and function) and replacement (introducing new species better suited to climate change) categories.

The references of all data sources in the databases can be found in Appendix E.

Table 3. Names and descriptions of forest restoration types distinguished in this study. The main characteristics indicate the main points on which the systems differ from each other. The set of restoration types emerged out of an effort to provide a generalized, yet comprehensive overview of different forest restoration types and data availability. We included cropland abandonment, pasture abandonment and abandoned slash-and-burn cropland because of sufficient data on prior land use for these restoration types. We adapted the active forest restoration types, i.e., active reclamation and active reconstruction, from Stanturf et al. (2014).

Restoration type		Main characteristics	Description
CA	Cropland abandonment	• Natural regeneration of forests on cropland	Passive colonization of abandoned cropland by native and/or exotic tree species along successional stages. Natural forest regeneration occurs without human interference and regenerating forests are not used or exploited.
PA	Pasture abandonment	• Natural regeneration of forests on pastureland	Passive colonization of abandoned pasture by native and/or exotic tree species along successional stages. Natural forest regeneration occurs without human interference and regenerating forests are not used or exploited.
Rcl	Active reclamation	• Active planting of forests on severely degraded land	Active restoration of severely degraded land with very limited vegetation cover. Land has been degraded by heavy resource extraction or erosion, such as through

			mining. Multiple native tree species are planted or seeded according to a predefined restoration plan, with the aim to re-establish the forest ecosystem that was present prior to land degradation.
Rco	Active reconstruction	• Active planting of forests on limitedly or non-degraded land	Active restoration of land that was priorly used and managed to extract resources, such crops or livestock. Multiple native tree species are planted or seeded according to a predefined restoration plan with the aim to re-establish a native forest that resembles the one prior to land conversion.
SB	Abandoned slash-and-burn cropland	• Natural regeneration of forests on cropland that was derived through burning of forested land	Passive colonization of abandoned cropland that was created through the cutting and subsequent burning of forested land by native and/or exotic tree species along successional stages. Natural forest regeneration occurs without human interference and regenerating forests are not used or exploited.

2.2.1.2 Biodiversity indicators

We assessed local biodiversity of animals and plants using four key indicators that quantify how species assemblages in restoration sites compare to those in reference sites: (i) intactness (IN), (ii) relative richness (SR), (iii) compositional similarity (SI), and (iv) relative total abundance (TA). These indicators collectively reflect the differences between biodiversity in restoration sites and reference sites. The selection of these four metrics is grounded in previous research, which shows that biodiversity responses to human-induced changes are best captured by combining indicators related to species richness, abundance, and intactness (Kuipers et al. 2023; Santini et al. 2017; Vačkář et al. 2012). Intactness indicates whether species that are specifically adapted to reference sites occur in restoring forests and whether the abundances of these species are similar among restoring forests and reference sites. A comparison between intactness and similarity reveals whether a loss of intactness is mainly due to a loss of species specifically adapted to reference forests, or rather a reduced abundance of those species. Comparing intactness and similarity with relative richness helps us identify whether those species that are specifically adapted to reference sites are replaced by other species in restoration sites. Finally, a comparison between intactness and relative total abundance reveals whether only the abundances of species specifically adapted to reference sites are lower in restoring forests than in pristine sites or whether species abundances in restoring forests are lower overall.

We computed biodiversity intactness using the mean species abundance (MSA) indicator (Alkemade et al. 2009; Kuipers et al. 2023; Schipper et al. 2020):

$$R_{itp} = \frac{G_{it}}{G_{ip}} \qquad (1a)$$

$$IN_i = \frac{\sum R_{itp},1}{N_p} \qquad (1b)$$

where, *G* are species or genera level abundances in restoring sites (*t*) and reference sites (*p*) of those species or genera that occur in reference sites, *R* is the ratio of the abundances of species or genera of restoration sites (*t*) divided by those of reference sites (*p*), *IN* is intactness expressed as the mean species abundance of study *i*, and *N* is the number of species or genera present in the reference sites (Alkemade et al. 2009; Schipper et al. 2020). We truncated *R*-values at 1 to assure that potentially high abundances of generalist or opportunistic species did not inflate intactness values to prevent an overestimation of intactness (Alkemade et al. 2009; Schipper et al. 2020). Finally, we calculated the arithmetic mean over the *R*-values of all species for each study to produce one intactness value per forest restoration type per study (Alkemade et al. 2009; Schipper et al. 2020).

We assessed effects of forest restoration on species richness by calculating a log-response ratio of the species richness of restoration types and reference sites:

$$SR_i = \ln\left(\frac{S_{ti}}{S_{pi}}\right) \quad (2)$$

where, *SR* is the log-response ratio of species richness in study *i*, and *S* is the number of species or genera in restoration sites (*t*) or in natural reference sites (*p*). We used the log-response ratio to adjust for the right-skewed distribution of the richness data.

We calculated species similarity between restoration and natural reference sites using the Sørensen Similarity Index (Sorensen 1948):

$$SI_i = \frac{2 \cdot T_{tpi}}{(2 \cdot T_{tpi} + F_{tpi})} \quad (3)$$

where, *SI* is the Sørensen Similarity Index of study *i*, *T* is the number of species or genera that are the same across reference (*p*) and restoring sites (*t*), and *F* is the number of species or genera that are different between natural reference and restoring sites.

Finally, we evaluated the impact of forest restoration on relative total abundance by calculating the log-response ratio between the total abundance of all species in restoring sites and the total abundance of all species in reference sites:

$$TA_i = \ln\left(\frac{\sum_S^M A_{tsi}}{\sum_S^N A_{psi}}\right) \quad (4)$$

Where *TA* represents the log-response ratio of summed abundances for study *i*, and *A* denotes the total abundance of species *s* in restoring sites (*t*) or reference sites (*p*). *M* and *N* are the numbers of species in the restoration (*t*) and reference (*p*) sites, respectively. We used the log-response ratio to adjust for the right-skewed distribution of the abundance data.

2.2.1.3 Effects of forest restoration on biodiversity over time

We applied generalized linear mixed-effect models to estimate biodiversity in forests undergoing restoration and to evaluate how biodiversity evolves over time during the restoration process. All analyses were conducted using R version 4.3.1 (R 2023), and spatial data were visualized using QGIS (QGIS 2023).

2.2.1.3.1 Response variables and fixed effects

We developed generalized linear mixed-effect models to evaluate animal and plant biodiversity using intactness, relative richness, similarity, and relative total abundance as response variables. The fixed effects in our models included: (i) forest restoration type, (ii) time since restoration began, (iii) taxonomic group, and (iv) continent. Initially, we created models with only forest restoration type as a fixed effect to estimate overall biodiversity differences between restoration types and natural reference sites. Next, we built separate models for each restoration type, incorporating time since restoration began as a fixed effect. For active restoration, we defined this as “time since the start of restoration,” while for passive restoration, we defined it as “time since abandonment.” When both minimum and maximum times since restoration were provided, we used the maximum time to avoid underestimating the actual duration of restoration. These models were developed to compare biodiversity changes in response to different restoration types over time. Subsequently, we included both forest restoration type and animal taxonomic group as fixed effects in models to examine if variations in taxonomic responses to restoration significantly affected model outcomes. Finally, we incorporated both forest restoration type and continent as fixed effects to assess whether spatial differences influenced the relationship between biodiversity and forest restoration.

2.2.1.3.2 Model type

We built two different types of generalized mixed-effect models: (i) beta-regression models with a logit-link function for intactness and similarity response variables using the R-function *glmmTMB* of the Temperate Model Builder package (Brooks et al. 2023), and (ii) Gaussian linear mixed-effect models for relative richness and total abundance using the *lmer* function of the R-package lme4 (Bates et al. 2015). We used beta-regression models to estimate intactness and similarity because the predicted estimates of these indicators cannot be lower than 0 or higher than 1. Beta-regression does not allow for values of 0 and 1 (intactness and similarity can be 0 or 1), thus we transformed intactness and similarity to values between 0 to 1 using the Smithson-Verkuilen transformation (Smithson and Verkuilen 2006). We used Gaussian linear mixed-effect models to estimate relative richness and total abundance because the predicted estimates of these indicators can be higher than one and their distribution was Gaussian.

2.2.1.3.3 Random effects and weights

We added nested random effects of sources (1|Source) per restoration type to allow for potential differences among datasets and non-independence of data. We did not include the studies within the sources as random effect because usually only one study per source was conducted. We weighted biodiversity indicator values by the square-root of the total number of species per study that occurred in reference sites to increase the weight of studies with a high number of species and reduce the weight of those with a limited number of species. We square-root transformed the number of species to mitigate skewness in the data.

2.2.1.3.4 Model evaluation and visualization

We assessed normality by examining residual plots, histograms, and performing the Shapiro-Wilk test. To evaluate the model fit, we conducted Omnibus tests and used the likelihood-ratio chi-square test to determine whether the explained variance significantly exceeded the unexplained variance. For presenting our results, we back-transformed the log-response ratios for species richness, as well as the intactness and similarity estimates, from logit-link distributions.

2.2.1.3.5 Post-hoc test

We performed Tukey post-hoc tests with Bonferroni correction to evaluate differences between forest restoration types, utilizing the *glht* function from the R package *multcomp* (Hothorn et al. 2024).

2.2.1.4 Correlations between biodiversity indicators

We performed principal component analyses (PCA) and generated Spearman correlation matrices to evaluate the relationships between biodiversity indicators—intactness, similarity, relative richness, and total abundance—for both animals and plants. This analysis helps to identify the complementarity and redundancy among the biodiversity indicators.

2.2.2 Changes in biodiversity under forest restoration in Europe

2.2.2.1 Meta-analysis

We conducted a meta-analysis to evaluate changes in biodiversity in response to forest restoration in Europe. We began with a search in the Scopus database (https://www.scopus.com/home.uri, accessed on 26-07-2024) to identify relevant studies. We used a search string incorporating keywords related to (i) biodiversity (e.g., “abundance”), (ii) natural reference forest sites (e.g., “pristine,” “old-growth”), (iii) forest restoration (e.g., "restoration"), and (iv) European countries (e.g., "Spain") (see Appendix G1 for the complete search string). Following the search, we screened the identified papers for relevance. To be included, studies had to provide (i) species-level abundance data for both a natural reference site and a restoration site in Europe, and (ii) a detailed description of the forest restoration type. We defined natural forest reference sites as areas of forests with no visible signs of human activity for at least 80 years. Our definition of natural reference sites is in line with the FAO definition of primary forests, i.e., "naturally regenerating forest of native tree species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed" (FAO 2020a). We then extracted species-level abundance data from the qualifying publications and compiled it into a database. We also recorded the time since the start of forest restoration in the database. Each publication was entered as a single source, which could include one or more studies, differentiated by location and/or the year data were collected. Lastly, we categorized forest restoration into two types based on human involvement: (i) active forest restoration, involving the planting of trees and other vegetation to aid forest growth and recovery, and (ii) passive forest restoration, where forests naturally expand on abandoned land without human intervention, such as on former cropland or pasture.

The references of the sources from which we gathered data can be found in Appendix G.

2.2.2.2 Biodiversity indicators

See Section 2.2.1.2 for a description of the biodiversity indicators used to calculate changes in biodiversity in response to forest restoration in Europe.

2.2.2.3 Effects of forest restoration on biodiversity over time

We applied generalized linear mixed-effect models to estimate biodiversity in forests undergoing restoration and to evaluate how biodiversity changes over time during the restoration process. All analyses were conducted using R version 4.3.1 (R 2023), and spatial data were visualized using QGIS (QGIS 2023).

2.2.2.1.1 Response variables and fixed effects

We developed generalized linear mixed-effect models using intactness, relative richness, similarity, and relative total abundance as response variables to evaluate animal and plant biodiversity across two forest restoration types, compared to reference sites. The fixed effects in our models included: (i) forest restoration type and (ii) time since restoration began. Initially, we constructed models with only forest restoration types as a fixed effect to assess overall differences in biodiversity between restoration types and reference sites. Next, we created separate models for each restoration type, incorporating time since restoration began as a fixed effect. We defined time since the start of restoration as "time since the start of restoration" for active restoration and "time since abandonment" for passive restoration. In case a minimum and maximum time since restoration was provided, we decided to include the maximum time that passed since the start of restoration because calculating an average across minimum and maximum times often resulted in an underestimation of the time passed in most restoration sites. These separate models allowed us to explore and compare how biodiversity changes in response to different types of forest restoration over time.

2.2.2.1.2 Model type

We employed two types of generalized mixed-effect models: (i) beta-regression models with a logit-link function for intactness and similarity, using the *glmmTMB* function from the R package TMB (Brooks et al. 2023), and (ii) Gaussian linear mixed-effect models for relative richness and total abundance, utilizing the *lmer* function from the R package lme4 (Bates et al., 2015). We used beta-regression models to estimate intactness and similarity because the predicted estimates of these indicators cannot be lower than 0 or higher than 1. Beta-regression does not allow for values of 0 and 1 (intactness and similarity can be 0 or 1), thus we transformed intactness and similarity to values between 0 to 1 using the Smithson-Verkuilen transformation (Smithson and Verkuilen 2006). We used Gaussian linear mixed-effect models to estimate relative richness and total abundance because the predicted estimates of these indicators can be higher than one and their distribution was Gaussian.

2.2.2.1.3 Random effects and weights

We included nested random effects of source-study combinations (1|Source) for each restoration type to account for potential differences among datasets and to address data non-independence. We did not include the Study as a random effect because typically only one study per source was conducted. To adjust for varying numbers of species, we weighted biodiversity indicator values by the square root of the total number of species recorded in reference forests for each study. This approach increased the influence of studies with a larger number of species and reduced the influence of those with fewer species. We applied a square-root transformation to the number of species to reduce skewness in the data.

2.2.2.1.4 Model evaluation and visualization

We assessed normality by analysing residual plots, histograms, and performing the Shapiro-Wilk test. To evaluate model fit, we conducted Omnibus tests and a likelihood-ratio chi-square test to determine if the explained variance significantly surpassed the unexplained variance. For presentation, we back-transformed the log-response ratios for species richness, as well as the intactness and similarity estimates, from logit-link distributions.

2.2.2.1.5 Post-hoc test

We performed Tukey post-hoc tests with Bonferroni correction to evaluate differences between forest restoration types, utilizing the *glht* function from the R package *multcomp* (Hothorn et al. 2024).

2.2.2.1.6 Correlations between biodiversity indicators

We performed principal component analyses (PCA) and generated Spearman correlation matrices to evaluate the relationships between biodiversity indicators—intactness, similarity, relative richness, and total abundance—for both animals and plants. This analysis helps to identify the complementarity and redundancy among the biodiversity indicators.

3 Results of model improvements

In Section 3 we describe the results of the statistical analyses to estimate biodiversity under different forest management systems and forest restoration types, which will be used to parameterize the LandSyMM model, in particular the CRAFTY module (described in Section 4).

3.1 Effects of forest management on biodiversity

Here we discuss the results of the global and European scale assessments of effects of forest management on biodiversity. The intactness estimates produced in these assessments were used to parameterize the LandSyMM model, in particular the CRAFTY module. First, we discuss the global outcomes on effects of forest management on biodiversity. Hereafter, we discuss the European outcomes on effects of forest management on biodiversity. Finally, we shortly discuss the difference between the global- and European-scale biodiversity outcomes.

3.1.1 Global effects of forest management on biodiversity

In total, the three databases included 860 animal and plant studies of which 602 from PREDICTS, 146 from GLOBIO and 171 from Kuipers et al. 2020. Out of these 860 studies, 182 studies sourced from 123 publications contained data on forest management systems. Of these 182 studies, 157 focused on animal diversity, while 25 examined plant diversity. In total, we compiled 159,148 abundance records for 9,240 animal species or genera and 153,305 abundance records for 4,917 plant species or genera. Some studies included information on various management systems and different taxa. Most studies came from South America (animals = 53, plants = 6), followed by Asia (animals = 33, plants = 9), Africa (animals = 34, plants = 3), and North America (animals = 25, plants = 1, with a focus on Central America). Europe (animals = 6, plants = 5) and Oceania (animals = 6, plants = 1) were less represented (Fig. 1). Hence, there are more studies on tropical forests than temperate and boreal forests in the database. All seven forest management systems were represented in South American studies, while North America, Asia, and Africa had studies on six management systems. Oceania and Europe had studies on three and two systems, respectively (Fig. 1). Plantation systems were the most represented systems in South America, Asia, Oceania, and Europe, with selective cutting systems also prevalent in South America and Asia. Agroforest systems were the most studied in North America, while Africa had more studies on selective cutting systems. The distribution of studies varied by taxon: insects had the most studies (63), followed by mammals (56), birds (38), plants (30), amphibians (16), and reptiles (15) (Appendix Fig. B1). Animal data covered at least five management systems across all taxonomic classes. For plants, data was more limited, with only five management systems, namely agroforests (AG), clear-cutting and regrowth (CC), perennial tree crop plantations (PT), forest plantations (PL), and selective cutting (SC). The most frequently represented plant classes were Liliopsida, Magnoliopsida, and Polypodiopsida.

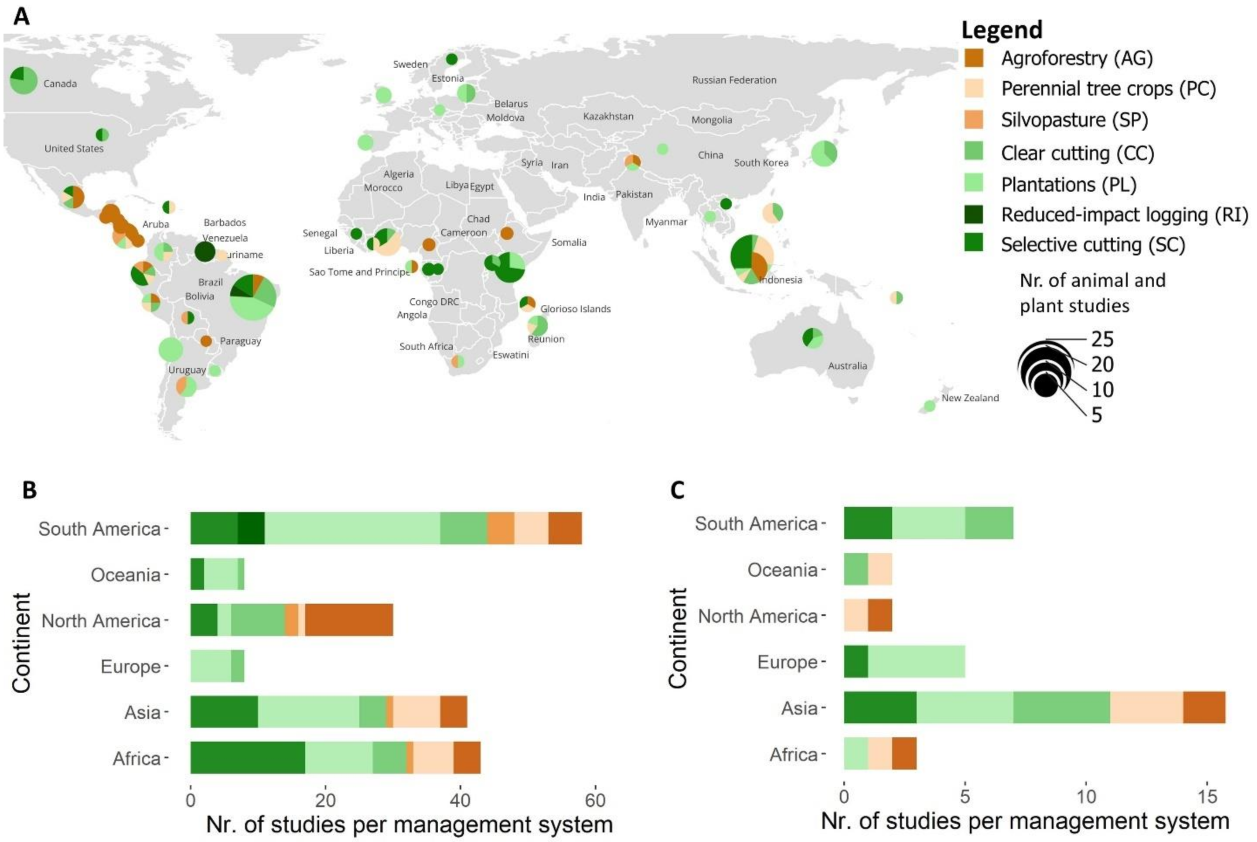


Figure 1. Spatial distribution and study counts. (A) Global Distribution of Empirical Biodiversity Studies: This map shows the global distribution of studies categorized by forest management system. The size of the pie charts represents the number of studies per country included in the databases, with the scale indicated by the black circle in the lower right corner of Panel A. (B) The distribution of animal biodiversity studies across continents, categorized by forest management system. (C) The distribution of plant biodiversity studies across continents, also categorized by forest management system. Each study may encompass multiple management systems and cover various taxonomic groups.

3.1.1.1 Effects of forest management on biodiversity

3.1.1.1.1 Intactness (IN)

We observed significantly lower intactness (i.e., all values < 1) for both animals and plants across all forest management systems compared to natural reference sites (see Fig. 2-3; Appendix Table A1). Overall, perennial tree crops and forest plantations had the lowest levels of intactness, whereas reduced-impact logging, clear-cutting, and selective cutting exhibited relatively higher intactness compared to reference sites. Among animal groups, herpetofauna and insects had the lowest intactness, while mammals generally showed the highest levels (refer to Appendix Fig. B2-B4). In the context of food production systems, animal intactness was significantly higher in agroforest systems compared to perennial tree crop plantations (P = 0.014) (Appendix Table A2).

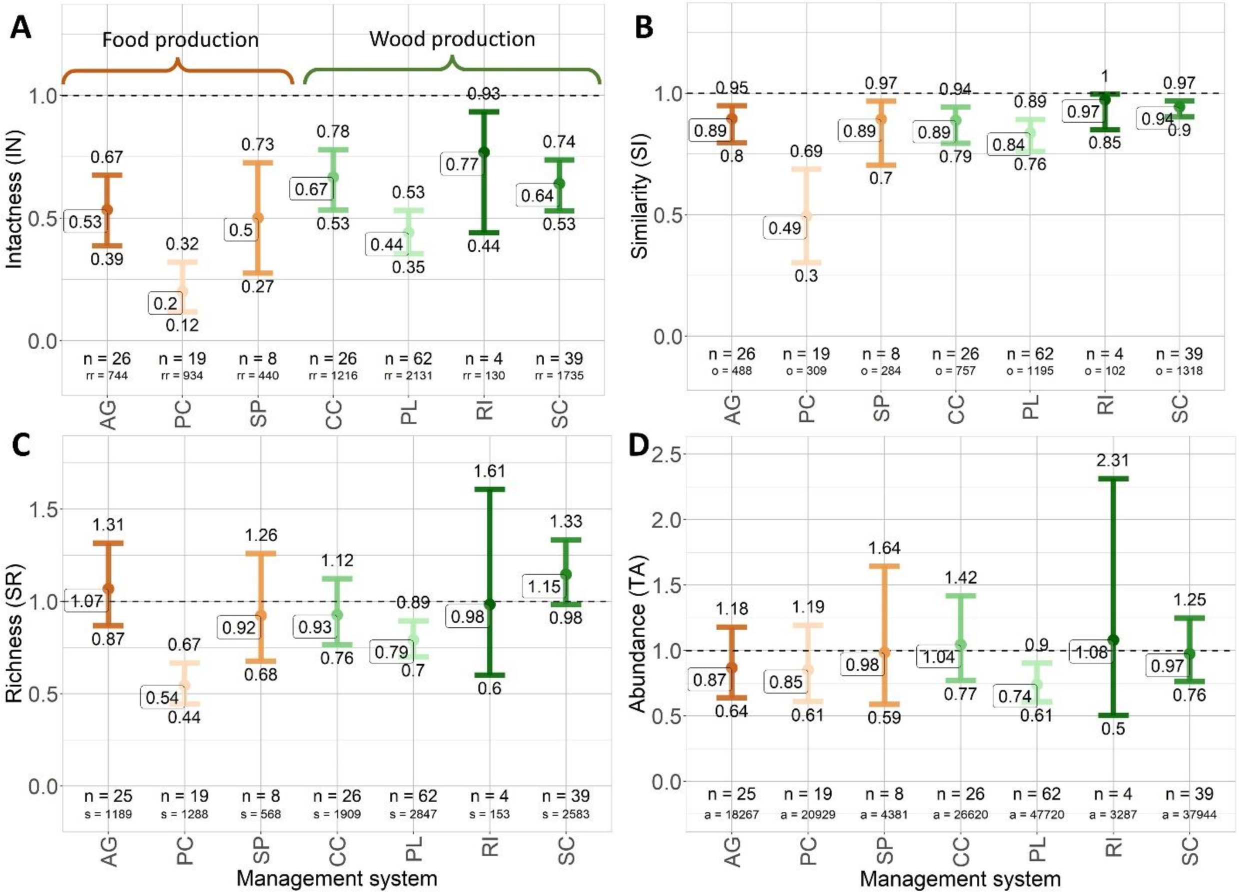


Figure 2. Impacts of forest management on animal biodiversity. The dotted line represents the reference level biodiversity. Biodiversity levels in managed sites are similar to those of reference sites if the estimates are close to the dotted line, and the confidence intervals overlap with the dotted because in this case the ratio between the biodiversity of managed and reference sites is

1 or close to 1.The effects of forest management on animal biodiversity are presented using four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). The seven forest management systems analysed are: (i) agroforests (AG), (ii) clear-cutting and regrowth (CC), (iii) perennial tree crop plantations (PC), (iv) forest plantations (PL), (v) reduced-impact logging (RI), (vi) selective cutting (SC), and (vii) silvopasture (SP). Each study may include multiple management systems and cover various taxonomic classes. For animal biodiversity, distinct values were calculated for each class. n = number of studies included per management type. rr = number of response values used to compute intactness. o = number of overlapping species between managed and reference forest sites. s = number of species occurring in both managed and reference sites. a = number of abundance values used to compute relative total abundance

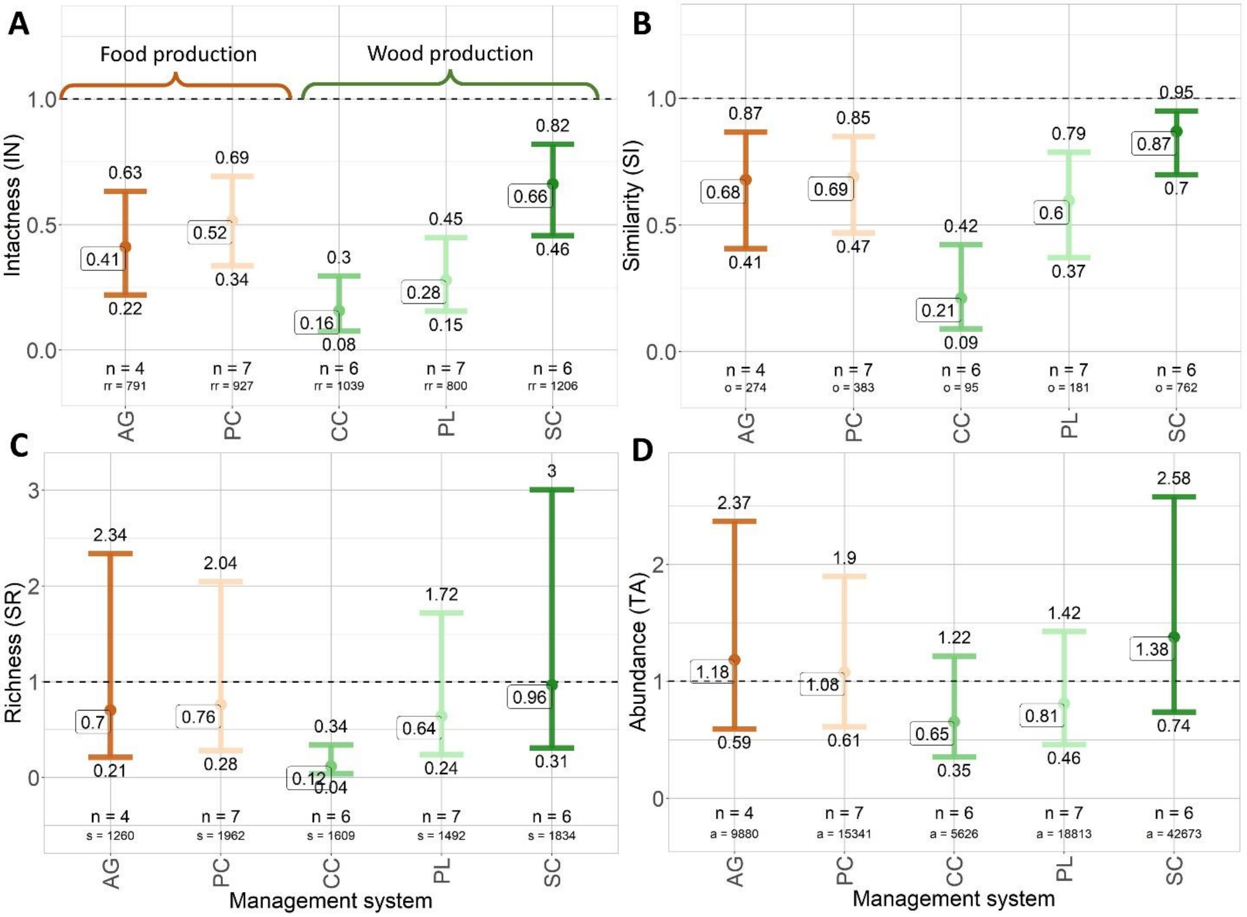


Figure 3. Impacts of forest management on plant biodiversity. The effects of forest management on plant biodiversity were assessed using four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). The analysis included five forest management systems: (i) agroforests (AG), (ii) clear-cutting and regrowth (CC), (iii) perennial tree crop plantations (PC), (iv) forest plantations (PL), and (v) selective cutting (SC). Each study may cover multiple management systems and taxonomic classes. n = number of studies included for each management type. rr = number of response values used to calculate intactness. o = number of overlapping species between managed and reference forest sites. s = number of species found in both managed and reference sites. a = number of abundance values used to compute relative total abundance.

3.1.1.1.2 Relative richness (SR)

We observed significantly lower relative richness of animals in perennial tree crop and forest plantation systems compared to reference sites, while richness was comparable to reference sites in all other management systems (see Fig. 2; Appendix Table A1). For plants, a similar pattern was noted for perennial tree crop systems, but not for forest plantations (see Fig. 3; Appendix Table A1). In particular, bird, herpetofauna, and insect species exhibited significantly reduced richness in perennial tree crop systems relative to reference sites (refer to Appendix Fig. B2-B4). Insects and mammals also showed significantly lower richness in forest plantations relative to reference sites. Additionally, perennial tree crop plantations had significantly lower richness of vulnerable to critically endangered animal species compared to reference sites (see Fig. 4; Appendix A3). For wood production systems, animal richness was significantly lower in forest plantations compared to selective cutting systems (P < .001). In terms of food production systems, agroforest systems were associated with significantly higher animal richness compared to perennial tree crop plantations (P < .001) (Appendix Table A2).

3.1.1.1.3 Similarity (SI)

We observed generally similar levels of animal and plant similarity across most forest management systems. However, perennial tree crop plantations exhibited substantially lower similarity compared to all other systems when compared to reference sites (see Fig. 2-3; Appendix Table A1). This pattern was also evident for mammals, and birds (refer to Appendix Fig. B2-B4). Additionally, we found significantly lower similarity of vulnerable to critically endangered animal species in reference sites (see Fig. 4; Appendix Table A3). For wood production systems, animal similarity was significantly lower in forest plantations compared to selective cutting systems (P = 0.009). In food production systems, agroforests demonstrated significantly higher animal similarity compared to perennial tree crop plantations (P = 0.002) (Appendix Table A2).

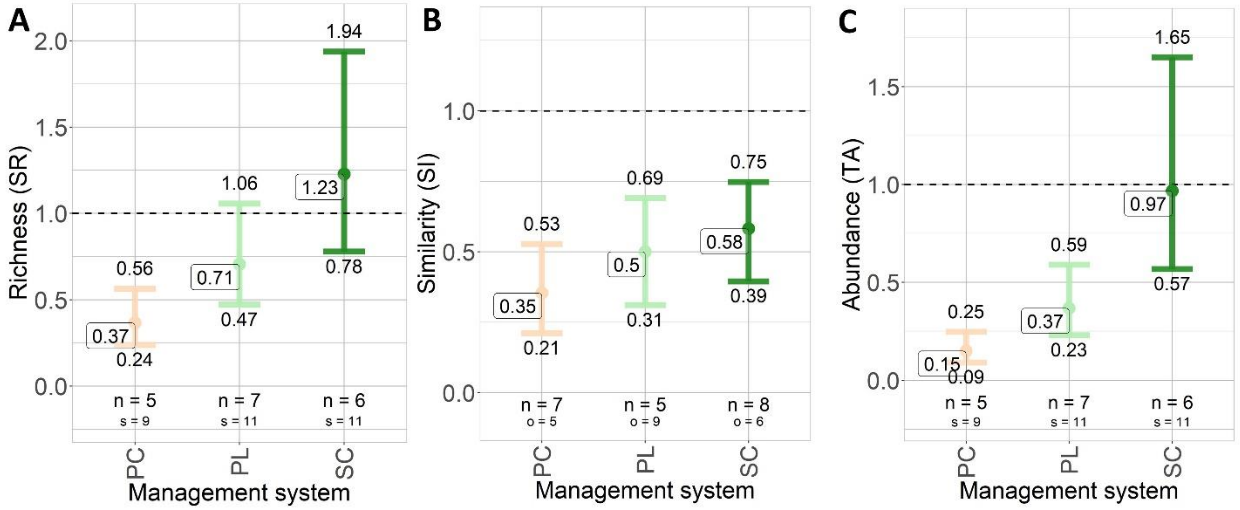


Figure 4. Impacts of forest management on threatened animal biodiversity. The effects of forest management on threatened animal biodiversity, categorized by IUCN Red List statuses ranging from Vulnerable to Critically Endangered, are assessed using four indicators: similarity (SI) (A), relative richness (SR) (B), and relative total abundance (TA) (C). The analysis focuses on three forest management systems: (i) perennial tree crop plantations (PC), (ii) forest plantations (PL), and (iii) selective cutting (SC). Each study may cover multiple management systems and taxonomic classes. n = number of studies included for each management type. o = number of overlapping species between managed and reference forest sites. s = number of species found in both managed and reference sites. a = number of abundance values used to compute relative total abundance.

3.1.1.1.4 Relative total abundance (TA)

We found that the relative total abundance of all animal species was generally comparable between managed sites and reference sites, except for forest plantations, which exhibited a significantly lower total abundance compared to reference sites (see Fig. 2-3; Appendix Table A1). This pattern was also observed in mammal and insect biodiversity (Appendix Fig. B4-B4). For bird diversity, we noted a significantly lower total abundance in perennial tree crop plantations compared to reference sites (Appendix Fig. B2). Additionally, the total abundance of vulnerable to critically endangered animal species was significantly lower in both perennial tree crop and forest plantation systems compared to reference sites (see Fig. 4; Appendix Table A3). No significant differences were observed in the total abundances of animals and plants across different wood or food production systems (Appendix Table A2).

3.1.1.2 Effects of forest management on biodiversity over time

We found varying impacts of forest management on animal biodiversity over time and across biodiversity indicators (Fig. 5; see Appendix Table A4 for statistics). We found a significant increase in the intactness and similarity of agroforest systems of ± 0.25 between 0 and 40 years after establishment (P-value >0.05). Richness showed a sharper and more curved increase of ± 0.5 over this period in agroforest systems. In forest plantation systems, we found a curved increase in intactness and similarity of ± 0.7 and ± 0.25, respectively, over a period of 80 years after establishment, whereas we found no significant increase in richness over this period. In clear cutting systems, we observed a significant negative relationship between biodiversity and time since logging for intactness and similarity over a period of 70 years, while we found no significant change in richness. Of all forest management systems, we found the smallest changes in biodiversity over time in selective cutting systems; only intactness slightly but significantly increased over 50 years after logging.

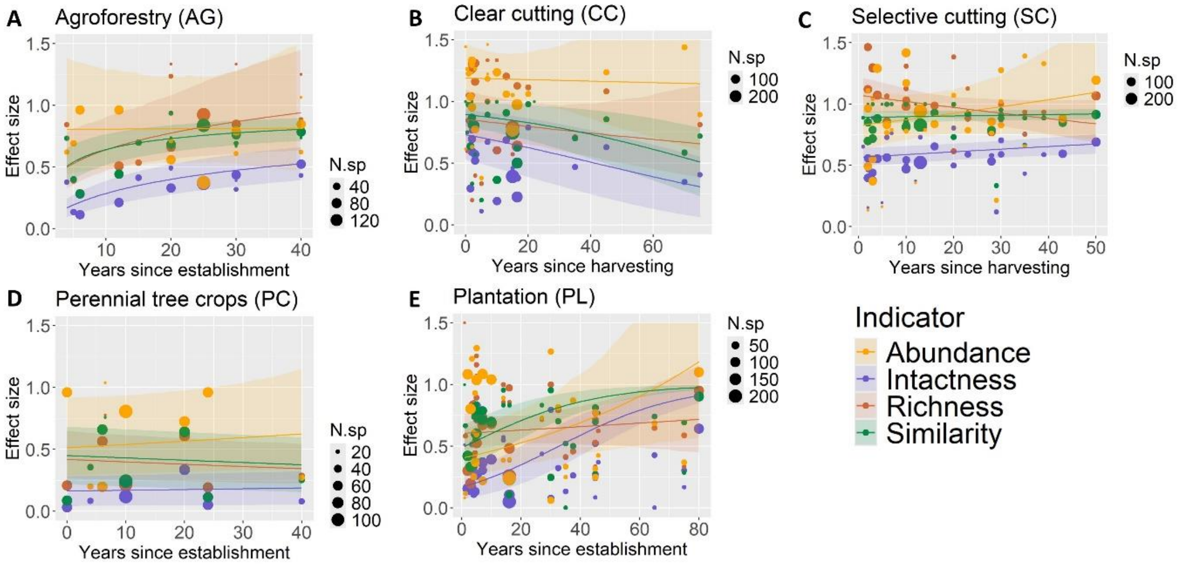


Figure 5. Relationships between time since management and animal biodiversity. We examined the relationships between the duration of management and animal biodiversity, measured through four indicators that all provide effect sizes of biodiversity change in response to management over time: intactness (IN), relative richness (SR), similarity (SI), and relative total abundance (TA). The indicator values are directly reflected in the relationships, they can be read from the y-axis, as all indicators represent effect sizes. This analysis was conducted for management systems with sufficient data to build generalized linear mixed-effect models (refer to Table 1 for definitions of these systems). N.sp represents the number of species used to calculate biodiversity metrics or

response ratios for each time point since the start of management. In the model for agroforests, a logarithmic relationship with time was incorporated to account for the non-linear nature of the observed positive trend.

3.1.1.3 Stability of effect size and model performance

We found minimal variability in effect sizes across indicators when randomly removing 10 to 40% of the data sources from the database, with variance remaining below 0.04 for animal biodiversity and below 0.4 for plant biodiversity (Appendix Fig. B5). Among animals, abundance was most sensitive to random source removal, while for plants, richness was the most affected. Models that included taxonomic class as an additional explanatory variable, alongside management systems, showed better performance for intactness and similarity compared to models with management systems alone. However, this improvement was not observed for richness and abundance (Appendix Table A5). Adding continents as an additional explanatory variable did not significantly enhance the performance of animal biodiversity models beyond what was achieved with management systems alone (Appendix Table A5). All biodiversity indicators were positively correlated, with correlations ranging from 0.1 to 0.7 ρ for animals and from 0.2 to 0.9 ρ for plants, indicating that the indicators are correlated with each other, yet that there are differences between the indicators. Additionally, the principal components did not overlap (Appendix Fig. B6-B7). Therefore, animal intactness, richness, and similarity differed from one another and could be considered distinct indicators of biodiversity, each capturing their own specific aspects of biodiversity.

3.1.1.4 Discussion of the results

Our analysis reveals that the effect of forest management system on biodiversity varies depending on the chosen indicators and taxonomic groups. We observe the strongest differences in intactness between managed forests and reference sites, followed by similarity, richness, and abundance. This pattern suggests that forest management primarily leads to a loss of species specifically adapted to reference sites, while also significantly reducing the abundance of species shared between managed and natural forests. Despite this, the similar levels of species richness and total abundance between most managed forests and reference sites indicate that the loss of species typical of reference sites is often offset by an influx of different species. This observation aligns with studies indicating that species richness recovers more rapidly than species composition in managed forests (Xu et al. 2015). Our findings also suggest that taxonomic groups respond differently to forest management. For instance, herpetofauna and insects show a more pronounced reduction in intactness between managed and natural reference sites compared to birds and mammals. This reduction is consistent with previous research (Chaudhary et al. 2016; López-Bedoya et al. 2022) and may be due to herpetofauna and insects being more sensitive to changes in forest structure and composition than birds and mammals.

When examining specific management systems, we find that naturally regenerating, managed forests tend to have biodiversity that is more similar to reference sites than forest plantations. This finding is consistent with global studies (Chaudhary et al. 2016; Crouzeilles et al. 2016). The small differences observed between the biodiversity of natural reference forests and selective cutting likely stem from the minimal impact of selective harvesting on forest vegetation structure and heterogeneity (Addo-fordjour, Sarfo, and Oppong 2022; Von Oheimb and Härdtle 2009; Robinson and Robinson 1999). Clear cutting affects forest structure and heterogeneity more than selective cutting (Addo-fordjour, Sarfo, and Oppong 2022; Gatti et al. 2015), yet natural forest regeneration post-harvest supports biodiversity through natural succession (Taki et al. 2013). Plantation systems skip early successional stages that would naturally occur because threes are planted. This leads to younger, less structurally complex forests, which likely explains the lower animal richness and similarity in these systems compared to reference sites (Donato, Campbell, and Franklin 2012). Additionally, the lower total abundance of vulnerable to critically endangered species in forest plantations highlights the potential risks of expanding plantations on natural forest land.

For forest management systems in which predominantly food is produced, we observe variable impacts on biodiversity relative to natural reference sites. Generally, we find small differences in biodiversity between reference sites and agroforests and, to a certain extent, silvopastural systems. This is likely related to the complex canopy layers and diverse tree and shrub/crop species in these systems, which provide habitat for a variety of species (Bohada-Murillo, Castaño-Villa, and Fontúrbel 2020; Schroth, Harvey, and Vincent 2004; Torralba et al. 2016). Therefore, the introduction of agroforests and silvopasture on agricultural land may enhance animal and plant richness at landscape scale, because they provide refuge for complementary species that do not occur in reference sites. Conversely, perennial tree crop plantations consistently show lower biodiversity compared to natural reference sites and agroforests systems, with reduced species richness and total abundance of vulnerable to critically endangered animals. The limited biodiversity in perennial tree crop plantations is likely due to their often simple, even-aged stand structure and sparse understory vegetation (Fitzherbert et al. 2008). This indicates that, unlike agroforests systems, expansion of perennial tree crop systems would not enhance biodiversity in agricultural areas.

Our results suggest that the effects of forest management systems on biodiversity may change over time. In forest management systems for wood production, we find that intactness and similarity increase over time in forest plantation systems over 80 years following establishment. In line with previous studies (Tudge et al. 2023), our results indicate that forest plantations may become more biodiversity-friendly over time, potentially because their vegetation structure and composition become more similar to that of undisturbed forests, hereby increasingly providing habitats for species that occur in reference sites. In contrast, we only find slow increases in intactness over 50 years after logging in selective cutting systems. This indicates that, although selective cutting and clear cutting have low impacts on biodiversity compared to other management systems, impacts remain persistent for decades. This is in line with previous findings (Crouzeilles et al. 2016) and may relate to persistent changes in forest structure (Gatti et al. 2015). In contrast with our expectations, we find a decrease in intactness and similarity in clear cutting systems over a period of 70 years after tree harvest. Although clear cutting often affects forest vegetation structure and heterogeneity more than selective cutting (Gatti et al. 2015; Addo-fordjour, Sarfo, and Oppong 2022), forests regenerate naturally after harvesting. Therefore, we would have expected positive changes in biodiversity over time upon natural succession, as found in previous studies (Taki et al. 2013). Potentially, the negative changes in intactness and similarity over time in clear cutting systems may relate to generally high biodiversity at early successional stages (Swanson et al. 2011), after which biodiversity declines (Hilmers et al. 2018). More research is necessary to explore effects of clear cutting and regrowth over time further, to assess whether these results are potentially driven by limited data 20 years after clear cut harvesting.

Sensitivity analyses reveal that the effects of forest management on animal biodiversity are generally robust, while plant biodiversity results are more sensitive to variations in the data pool (Appendix Fig. B6). This sensitivity may be due to the lower number of studies available for plants compared to animals. High variability in reduced-impact logging data reflects the limited number of studies for this system, resulting in wide confidence intervals. Caution is advised in interpreting results for agroforests and silvopasture systems, as most data are from tropical regions, which may not be representative of other regions (Schroth, Harvey, and Vincent 2004). Furthermore, data availability is skewed towards years close to harvesting, which may obscure the effects of important ecological processes such as biodiversity recovery along successional stages (Hilmers et al. 2018). Adding continents as a fixed effect in models did not improve animal biodiversity models, suggesting that our results are broadly applicable globally, despite spatial biases in data availability. In the future, differences between ecoregions, biomes and realms could be explored.

3.1.2 Effects of forest management on biodiversity in Europe

In total, our search term provided 1,128 hits for publications that were of potential interest. After screening, we ended up with 20 publications that met our criteria. From these publications, we derived data for 20 animal and 11 plant studies, derived from 13 and 7 publications, respectively. A single study may contain data on multiple management systems and multiple taxa. In total the database included 2,340 abundance values of 1,460 animal species and 3,231 abundance values of 421 plant species. Studies were spread across the European continent, with most studies carried out in the North of Europe, in particular for clear cutting and selection systems (Fig. 6). Variations in data availability occurred among taxa, with the highest number of studies included for insects (15), followed by plants (11), birds (4) and reptiles (1) (Appendix Fig. D1). For reptiles and birds, the database included only data on forest plantations. Hence, there is a strong overrepresentation of insects in the animal data pool.

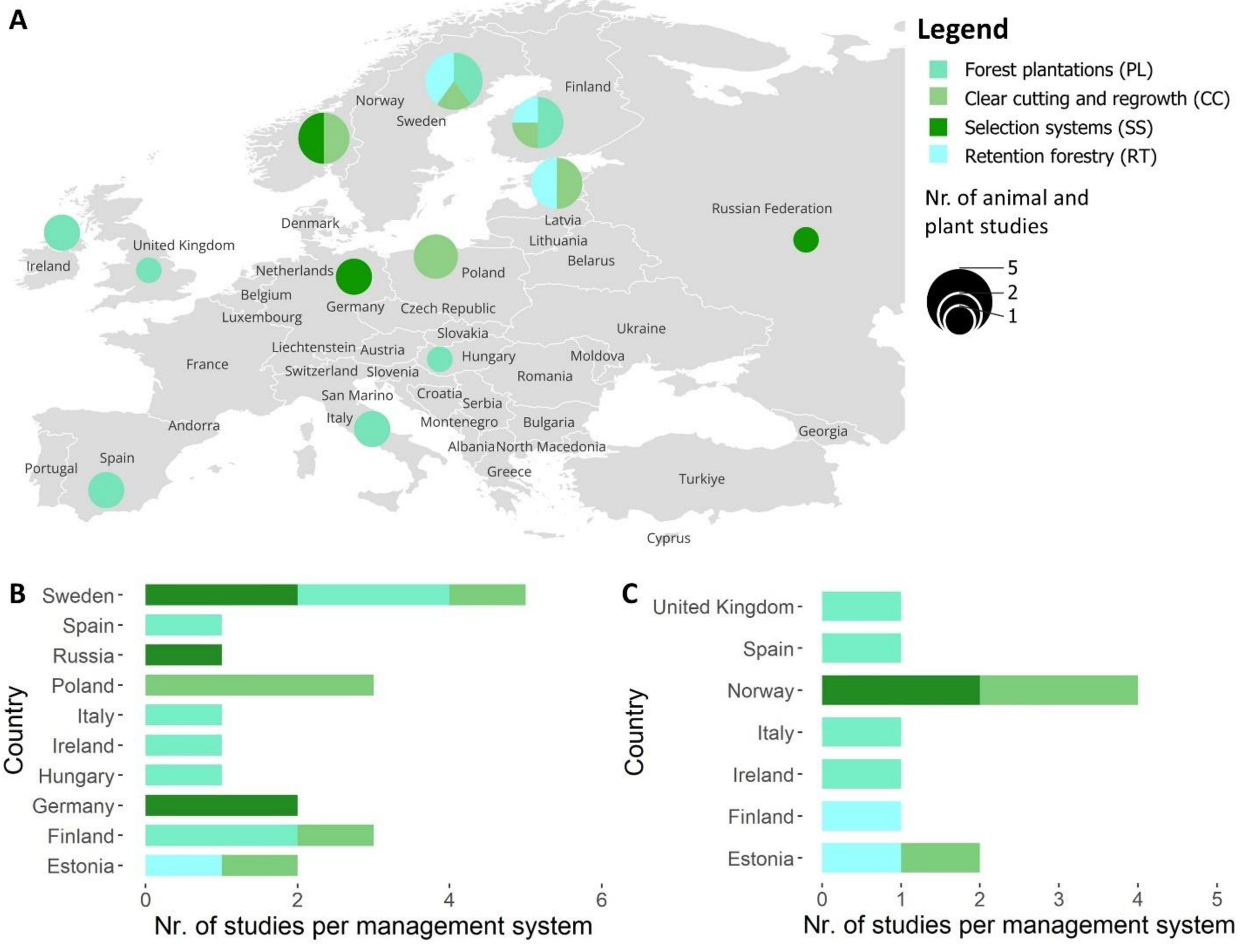


Figure 6. Spatial distribution and quantity of included studies: (A) The distribution of empirical biodiversity studies across Europe, categorized by forest management system. The sizes of the pie charts represent the number of studies per country included in the databases, with the scaling indicated by the black circle in the bottom right corner of Panel A. (B) Distribution of animal biodiversity studies by country and forest management system. (C) Distribution of plant

biodiversity studies by country and forest management system. Each study may encompass multiple management systems and taxonomic groups.

3.1.2.1 Effects of forest management on biodiversity

We found significantly lower animal and plant intactness (i.e., all values < 1) and similarity in all forest management systems than in reference sites (Fig. 7-8, Appendix Table C1). Effects of forest management on intactness were approximately similar across all systems for both animals and plants. We found similar animal and plant relative richness and total abundance in reference sites as in forest management systems. Effects of forest management on similarity, relative richness and total abundance were approximately similar across all systems for both animals altogether and plants. For insects, we found similar results to that of animals overall but found a lower relative total abundance in selection systems than in reference sites (Appendix Fig. D2).

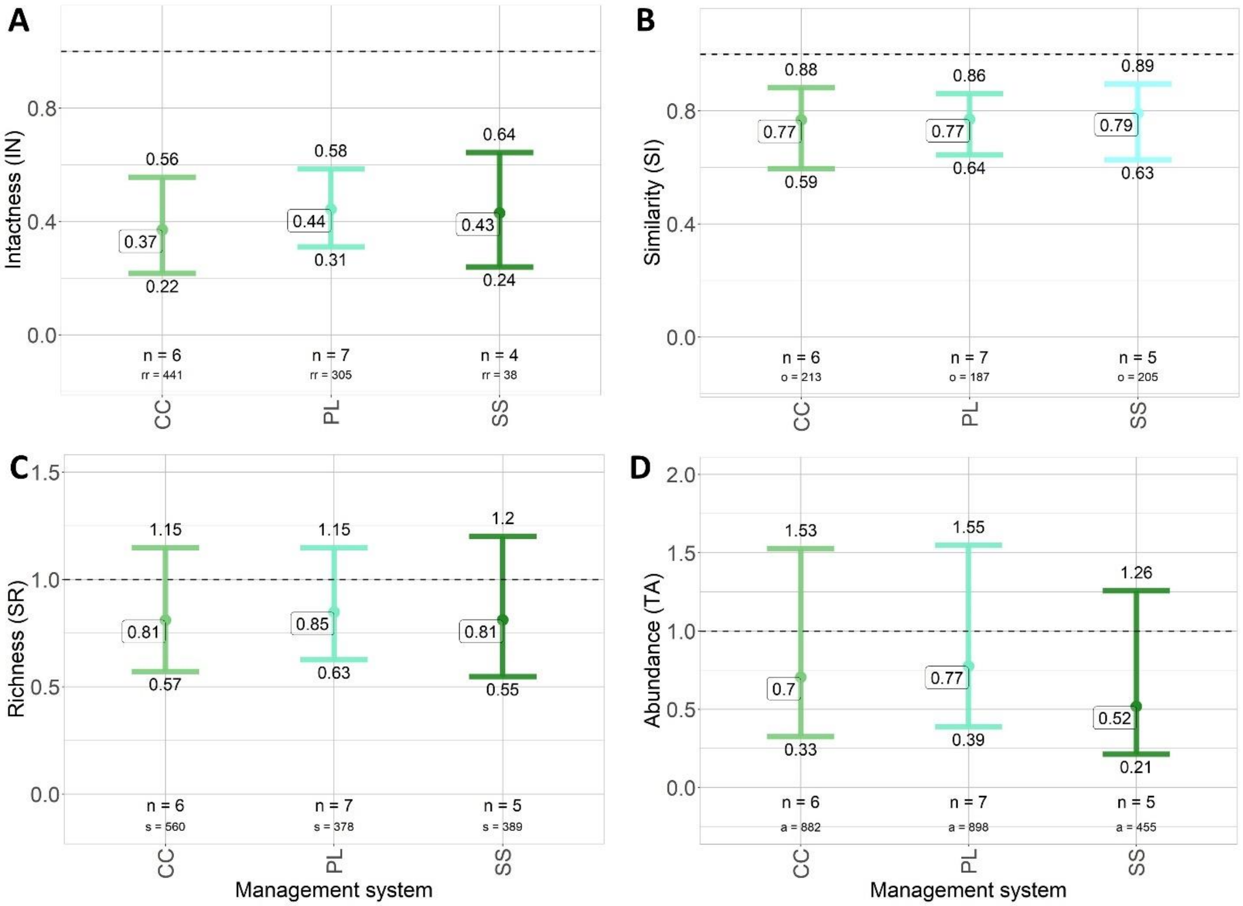


Figure 7. The impacts of forest management on animal biodiversity were analysed using four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We identified three forest management systems: (i) clear cutting and regrowth (CC), (ii) forest plantations (PL), and (iii) selection systems (SS). In this context, *n* denotes the number of studies included for each management type, *rr* represents the number of response values used to calculate intactness, *o* refers to the number of species shared between managed and reference forest sites, *s* indicates the number of species present across both managed and reference sites, and *a* stands for the number of abundance values used to compute relative total abundance.

3.1.2.2 Effects of forest management on biodiversity over time

We found varying effects of forest management on animal biodiversity over time and across indicators (Fig. 9; Appendix Table C2). In multipurpose plantation systems, we found a curved increase in intactness and similarity of ± 0.1 and ± 0.5, respectively, over a period of 80 years after establishment, whereas we found no significant change in richness and abundance over this period. In clear cutting systems, we observed no significant changes in biodiversity over time since harvesting over a period of 15 years.

3.1.2.3 Correlations between biodiversity indicators

We found that all biodiversity indicators positively related to each other with correlations between 0.5 and 0.9 ρ for animals and 0.4 and 0.8 ρ for plants, indicating that the indicators are correlated with each other, yet also differ from each other. Additionally, the principal components do not overlap. Therefore, the biodiversity indicators differ from each other and could be considered distinct indicators of biodiversity, each capturing their own specific aspects of biodiversity (Appendix Fig. D3).

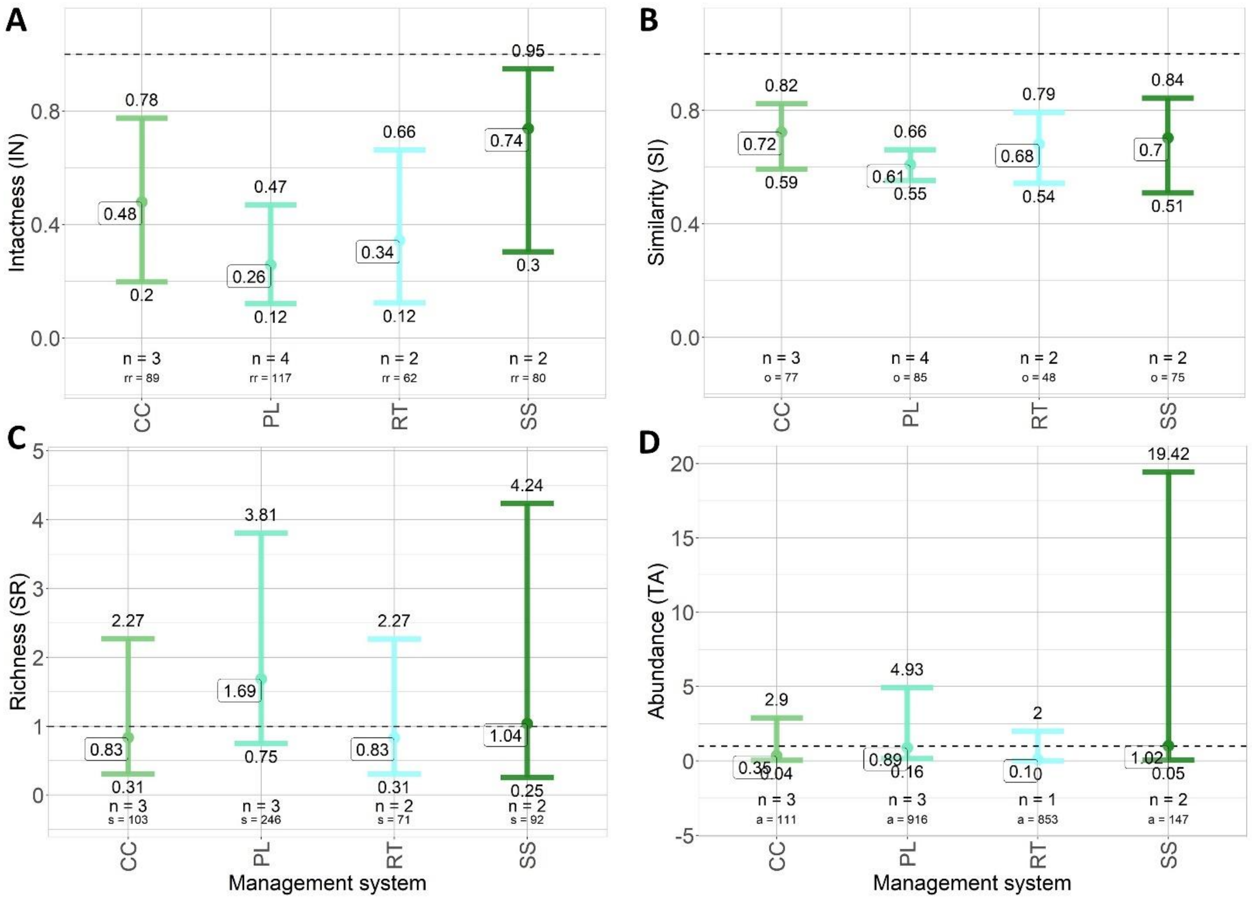


Figure 8. The effects of forest management on plant biodiversity were assessed using four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We classified four forest management systems: (i) clear cutting and regrowth (CC), (ii) forest plantations (PL), (iii) retention forestry (RT), and (iv) selection systems (SS). Here, n represents the number of studies included for each management type, *rr* denotes the number

of response values used to calculate intactness, *o* indicates the number of species shared between managed and reference forest sites, *s* refers to the number of species present across both managed and reference sites, and *a* signifies the number of abundance values used to compute relative total abundance.

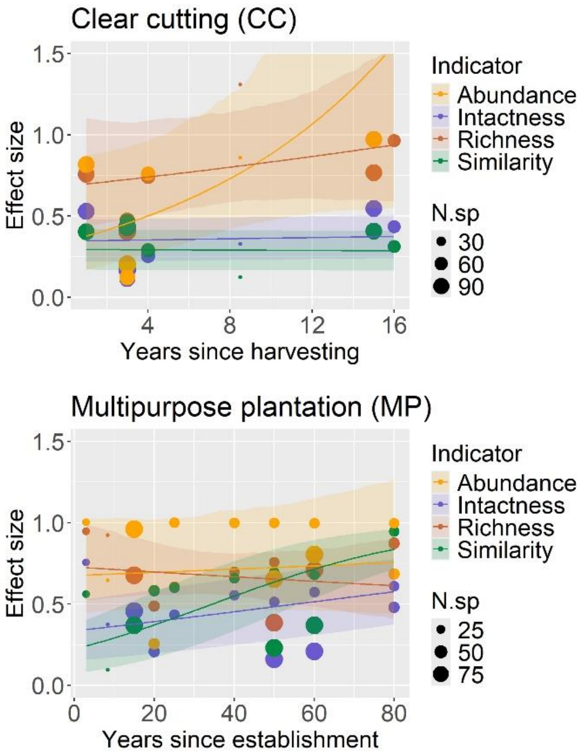


Figure 9. The relationships between time since the initiation of forest management and animal biodiversity were examined using four indicators: intactness (IN), relative richness (SR), similarity (SI), and relative total abundance (TA). This analysis was conducted for management systems with sufficient data to construct generalized linear mixed-effect models (refer to Table 2 for system definitions). The variable N.sp represents the number of species used to calculate biodiversity metrics and response ratios for each time since management.

3.1.2.4 Discussion of the results

We find that the biodiversity-friendliness of forest management systems depends on indicator choice. More specifically, we find the largest differences in intactness between managed forests and reference sites followed by similarity, richness and abundance. This suggests that management first and foremost causes a loss of species that specifically occur in reference sites and not in managed forest. Additionally, management causes a substantial reduction in the abundances of those species that are present in both managed forests and reference sites. Yet, similar richness and total abundance between managed forests and reference sites indicate that the loss of species occurring in natural reference sites is always compensated for by an influx of different species that do not occur in these sites. This finding is in line with previous studies that show limited effects of forest management on species richness and abundance (Dieler et al. 2017).

When zooming into management systems, we find that differences between reference sites and managed forests are highly similar across forest management systems for animals and plants in Europe. For plants, extensive management in the form of selective cutting appears to affect intactness less, yet confidence intervals overlap almost completely with those of the other systems. This finding contrasts with studies that show a larger decline in biodiversity in response to clear cutting than selective cutting in Europe (Paillet, Bergès, Hjältén, et al. 2010). Additionally, the finding contrasts with studies that find significant differences in biodiversity between management systems, such as between retention and clear cutting systems (Fedrowitz et al. 2014). Therefore, our findings reveal that, even though there may be differences in biodiversity between forest management systems (Dieler et al. 2017; Fedrowitz et al. 2014), a transition from natural reference sites to forest management results in similar changes of biodiversity composition across management sites. As the differences between the systems, such as differences in forest regeneration and the extent to which the canopy is disturbed, are often subtle we may not have been able to identify small differences in the effect of different forest management systems on biodiversity. On top of this, data limitations and the resulting large confidence intervals may have prevented us from differentiating the effect of the subtle differences in biodiversity responses to the different forest management systems.

Our results suggest that the biodiversity-friendliness of forest management systems may change over time and that such changes vary with indicator choice. We find that intactness and similarity increase over 80 years following establishment in multipurpose plantation systems. In line with previous studies (Tudge et al. 2023), our results indicate that forest plantations may become more biodiversity-friendly over time, potentially because their vegetation structure and composition become more similar to that of undisturbed forests, hereby increasingly providing habitats for species that occur in reference sites. In contrast, we find no significant changes in biodiversity over a period of 15 years in clear cutting. Therefore, our findings do not show signs of biodiversity recovery in clear cutting systems on the short term, which contrasts with findings that show increases in richness over time in these systems (Paillet, Bergès, Hjältén, et al. 2010). Potentially, the period in which we study changes in biodiversity are too short for clear cutting systems to observe any changes. Because of the large difference in time since the start of management or harvesting that could be assessed due to data limitations (i.e., 80 versus 15 years), we are unable to compare the results of biodiversity changes over time of plantations with those of clear cutting and regrowth. Therefore, no conclusion can be derived on whether plantations have smaller effects on biodiversity than clear cutting over time.

Limited differences found in the effects of forest management on biodiversity relative to the biodiversity of reference sites may result from the relatively low number of studies and observations found per management system, which results in relatively wide and overlapping confidence intervals. Higher numbers of observations and studies and more balanced distribution of studies across Europe may have enhanced the robustness of our assessment and reduced confidence intervals. This indicates the need for more studies that compare the biodiversity in reference sites with that of managed forests. Additionally, large confidence intervals may have resulted from differences in management approaches, such as differences in the intensity of cutting in selection systems, the extent area of forest that was cut for clear cutting systems, the number of trees or size of patches that were left in retention tree systems, the diversity of trees and age structure of multipurpose plantation systems and the types of tree species occurring in the forests and plantations. Future studies may attempt to account for these effects by creating further sub-categories of forest management systems in Europe. To achieve this, more data, and in particular, more detailed descriptions of forest management systems studies in source papers are needed. Besides insufficient sub-categorization of management systems and data limitations, large confidence intervals and limited differences across systems could also have occurred due to unobserved or unreported anthropogenic influences in reference sites. As most of Europe's forests are managed and only a small proportion of primary forests remain, it is likely that the biodiversity in those forests has been substantially influenced by anthropogenic disturbances, such as recreation and fragmentation. Additionally, biodiversity recovery in naturally regenerating forests may have been impeded, as observed in Section 3.2.2. Altogether, this may have resulted in the comparison of biodiversity in managed forests with biodiversity in forests that were affected by a range of unreported or unobserved anthropogenic disturbances. Besides this, unaccounted for effects of landscape diversity may have overshadowed effects of management (Brotons et al. 2003). For instance, reference sites and managed forest sites vary in size and border varying land uses, such as cropland, natural reference forests or urban land. Finally, there is a bias towards insect biodiversity in our animal biodiversity dataset. Therefore, effects of forest management on animal biodiversity mainly reveal effects on insect biodiversity, even though effects of forest management on biodiversity have been shown to vary across taxonomic groups in global-scale analyses (Chaudhary et al. 2016). Hence, there is a need for future studies to differentiate taxon-specific responses to forest management.

3.1.3 Comparing global and European management effects

We find contrasting patterns in the effect forest management has on biodiversity at global and European scale. At a global scale we find that less intensive forest management systems, such as selective cutting, generally foster higher biodiversity than found in intensive forest management systems, such as forest plantations, relative to reference sites. In contrast, at a European scale, forest management systems that are deemed to be less intensive, such as selection systems, have generally similar effects on biodiversity than systems that are deemed more intensive, such as forest plantations, in particular for animals. For example, for animal biodiversity, intactness estimates at European scale are similar to those for forest plantations at a global scale across management systems. For plants, we derived slightly higher estimates for intactness in selective cutting systems, yet confidence intervals across systems mostly overlap. Thus, while at global scale we point out forest management systems that are generally more biodiversity-friendly than others, it is generally not possible to identify a management system that is more biodiversity-friendly than others at European scale, based on the data available. This outcome highlights two things: (i) limited existing options to mitigate forest management effects on biodiversity in Europe, given currently available management practices; and (ii) the challenge of data limitations. Future studies could explore the potential for introducing different forest management systems, such as those more resembling selective cutting systems at global scale, as options for biodiversity-friendly forest management to better conserve biodiversity in Europe. More biodiversity studies are necessary at European scale that compare biodiversity in reference sites to that of managed sites to assess whether the limited differences found in biodiversity estimates across management systems in Europe are due to data limitations. Hereby, special attention should be paid to collecting data for a wide range of animal taxonomic classes.

3.2 Responses of biodiversity to forest restoration

Here we discuss the results of the global and European scale assessments of biodiversity recovery under forest restoration. In the future, the outcomes of these studies can be used to further improve the parameterization of the GLOBIO and LandSyMM models, in particular the CRAFTY module. First, we discuss the global outcomes on changes in biodiversity in response to forest restoration. Hereafter, we discuss the European outcomes on changes in biodiversity in response to forest restoration. Finally, we shortly discuss the observed difference in the global- and European-scale response of biodiversity to forest restoration.

3.2.1 Global changes in biodiversity in response to forest restoration

In total, the three databases included 748 animal and plant studies of which 602 from PREDICTS, 146 from GLOBIO. Of all 748 studies, 52 studies, derived from 46 publications, included data on the forest restoration types we distinguished of which 44 on animal and 8 on plant diversity, with in total 53,060 abundance values of 3,224 animal species or genera and 30,688 abundance values of 2,080 plant species or genera. A single study may contain information on multiple restoration types and multiple taxa. Most studies originated from Asia ($N_{animals}$ = 9, N_{plants} = 4), followed by South America ($N_{animals}$ = 10, N_{plants} = 2), North America ($N_{animals}$ = 9, N_{plants} = 2) and Oceania ($N_{animals}$ = 11, N_{plants} = 0), Africa ($N_{animals}$ = 5, N_{plants} = 0), and Europe ($N_{animals}$ = 0, N_{plants} = 1) (Fig. 10). Variations in data availability occurred among taxa, with the highest number of studies included for insects (22), followed by amphibians (10), and mammals (8), birds (8), plants (8) and reptiles (8) (Appendix Fig. F1). For animals, data was available for at least four forest restoration systems for all taxonomic classes. Less data was available for plants – only three restoration types could be distinguished, including cropland abandonment (CA), pasture abandonment (PA) and abandoned slash-and-burn cropland (SB). Most abundant plant classes were Magnoliopsida, Polypodiopsida and Liliopsida.

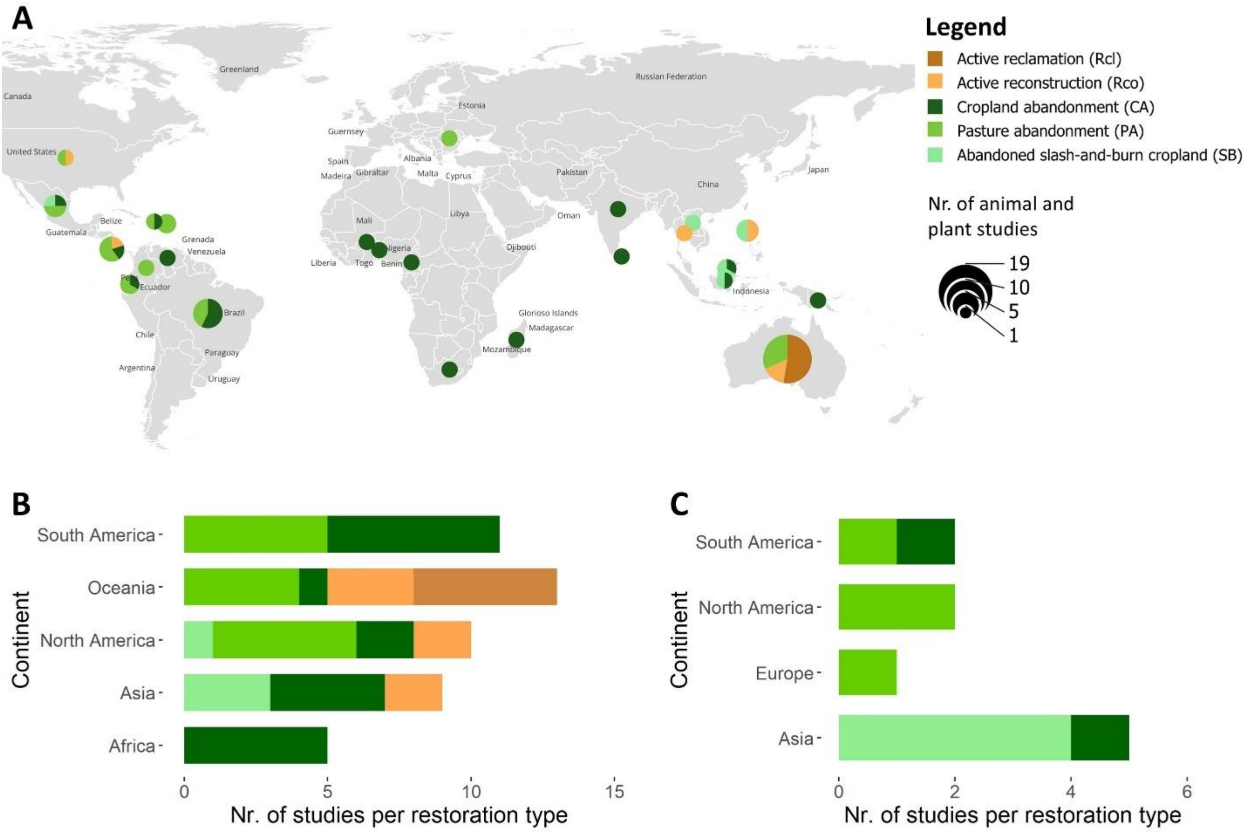


Figure 10. Spatial distribution and numbers of included studies. (A) Global distribution of empirical biodiversity studies categorized by forest restoration type. The sizes of the pie-charts indicate the number of studies per country included in the databases, the scaling of which is shown in the black circle on the right bottom of Panel A. (B) Animal biodiversity studies per country categorized by forest restoration type. (B) Plant biodiversity studies per country categorized by forest restoration type. Each study can contain multiple restoration types and cover multiple taxonomic classes.

3.2.1.1 The biodiversity of restoring forests

We found significantly lower animal and plant intactness and similarity in all forest restoration types than in reference sites (Fig. 11-12, Appendix Table E1). Intactness and similarity were similar across the different restoration types. We found similar relative total abundance in restoring forests as in natural reference sites. We found similar relative animal richness under cropland abandonment (CA), pasture abandonment (PA) and reclamation (Rcl), while animal richness was lower under reconstruction (Rco) and abandoned slash-and-burn cropland. For plants, we found similar relative plant richness under pasture abandonment and slash-and-burn cropland, but lower richness under cropland abandonment. We found no significant differences between the biodiversity estimates of forest restoration types (Appendix Table E2).

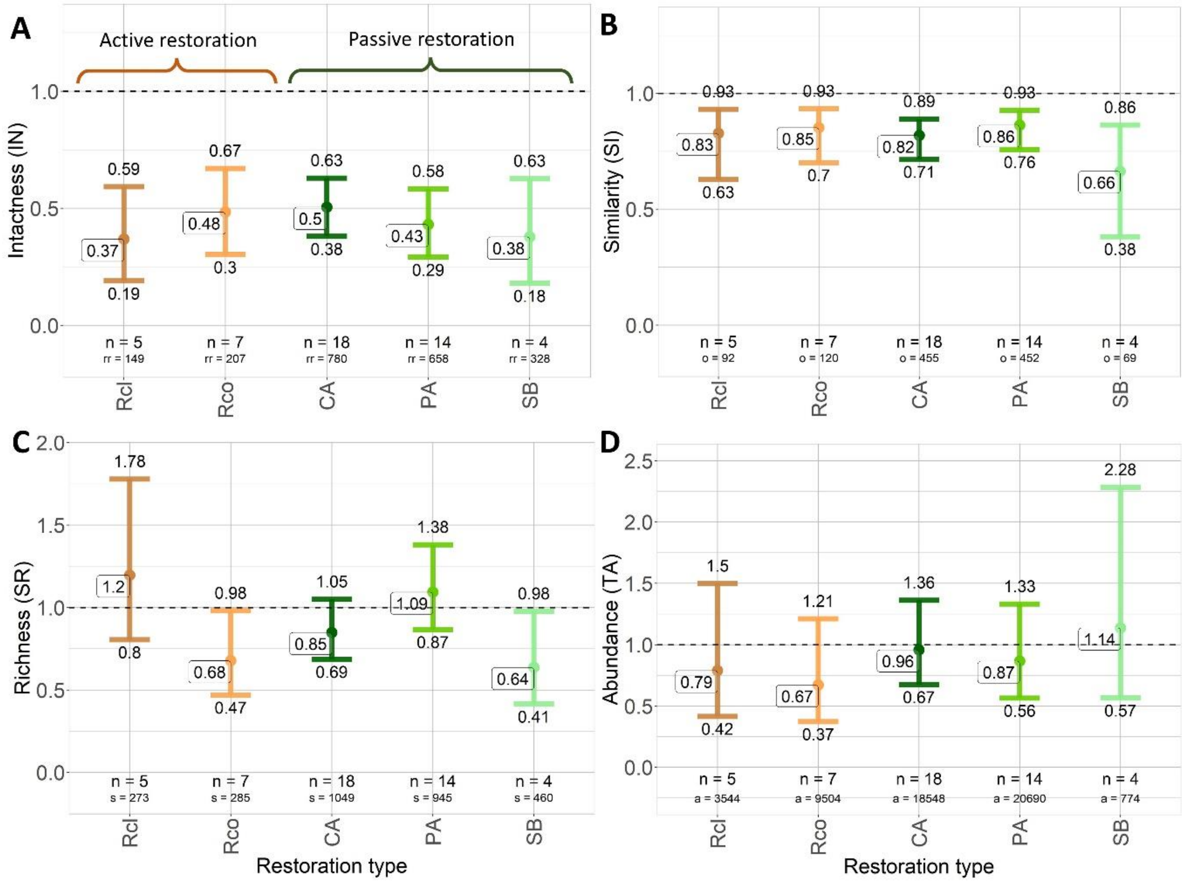


Figure 11. Animal biodiversity in forest restoration sites, relative to that of natural reference sits. Animal biodiversity was expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish five forest restoration types: (i) cropland abandonment (CA), (ii) pasture abandonment (PA), (iii) reclamation (Rcl), (iv) reconstruction (Rco), (v) abandoned slash-and-burn cropland (SB). n = number of studies included per restoration type, rr = number of response values used to compute intactness, o = number of overlapping species between restoration and reference forest sites, s = number of species occurring across the restoration and reference sites, a = number of abundance values used to compute relative total abundance.

3.2.1.2 Biodiversity changes over time in restoring forests

We found varying changes in animal and plant biodiversity over time and across indicators in response to forest restoration (Fig. 13; Appendix Table E3). We found significant increases in intactness, similarity and relative richness for cropland abandonment over a period of over 75 years. The increases range for < 0.2 and > 0.5 for richness. For pasture abandonment and active reconstruction, we find significant increases in intactness and similarity of ± 0.1 and of ± 0.5 over a period of 50 and 17 years, respectively. In contrast, we find a decline of ± 0.2 in intactness for both active reclamation and abandoned slash-and-burn cropland over a period of 17 and 60 years, respectively. For abandoned slash-and-burn cropland, we also find ± 0.2 declines in relative richness and similarity.

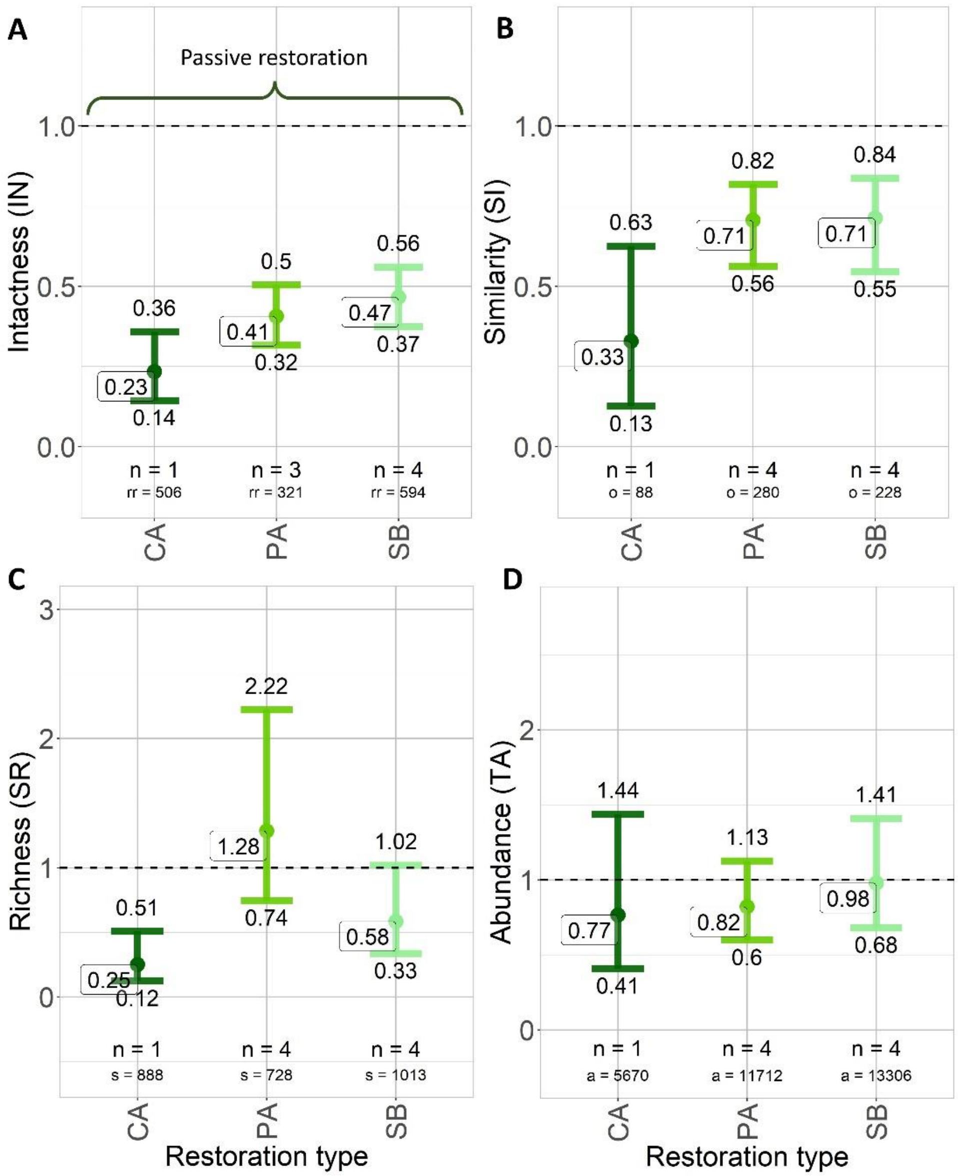


Figure 12. Plant biodiversity in forest restoration sites, relative to that of reference sites. Plant biodiversity was expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish two forest restoration types: (i) cropland abandonment (CA), (ii) pasture abandonment (PA), (iii) reclamation (Rcl), (iv) reconstruction (Rco), (v) abandoned slash-and-burn cropland (SB). n = number of studies included per restoration type, rr = number of response values used to compute intactness, o = number of overlapping species between restoration and reference forest sites, s = number of species occurring across the restoration and reference sites, a = number of abundance values used to compute relative total abundance.

3.2.1.3 Correlations between biodiversity indicators

Yet, we found that all biodiversity indicators positively related to each other with correlations between 0.2 and 0.7 ρ for animals and 0.1 and 0.9 ρ for plants, indicating that the indicators are correlated with each other, yet that there are differences between them. Additionally, the principal components do not overlap (Appendix Fig. F2). Therefore, the biodiversity indicators differ from each other and could be considered distinct indicators of biodiversity, each capturing their own specific aspects of biodiversity.

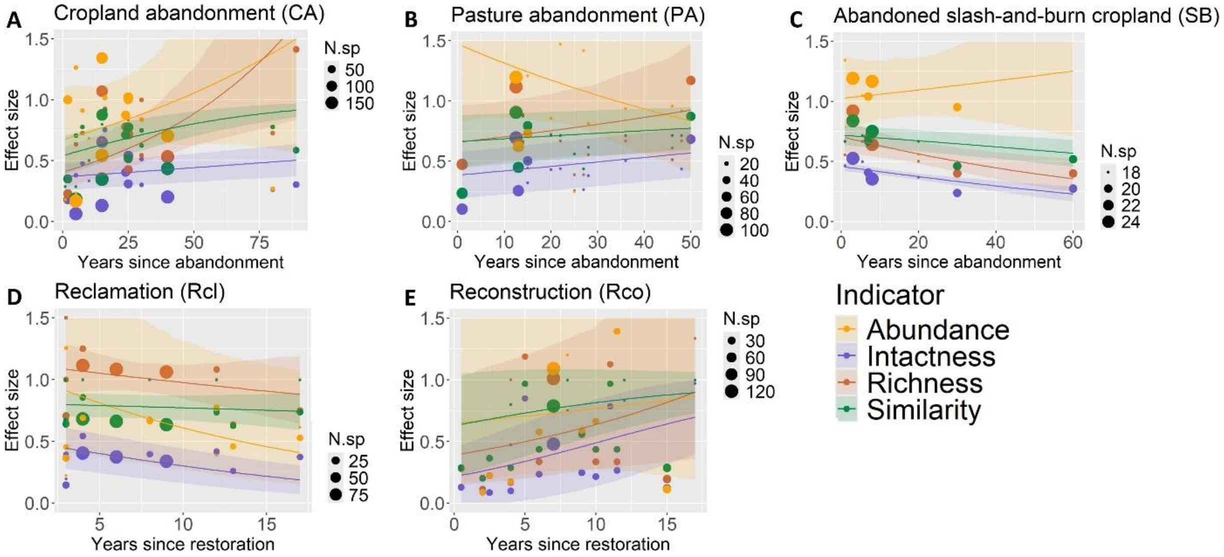


Figure 13. Relationships between time since the start of restoration or since abandonment and animal biodiversity expressed as intactness (IN), relative richness (SR), similarity (SI) and relative total abundance (TA). N.sp = the number of species that are included to calculate biodiversity metrics / response ratios per time since restoration or abandonment.

3.2.1.4 Robustness analyses

We detected only minor variations in effect sizes across indicators when randomly removing 10 to 40% of the data sources from the animal biodiversity database (variance below 0.1) (see Appendix Fig. F3). The random removal of sources had the greatest impact on relative total abundance, while it had the least effect on intactness. Models incorporating taxonomic class as an additional explanatory variable, alongside restoration types, were more effective for intactness and similarity than models using only restoration types; however, they did not show improved performance for richness and abundance (Appendix Table E4). Furthermore, including continents as an additional explanatory variable along with management systems did not significantly improve the animal biodiversity models (Appendix Table E4). In fact, models that excluded continent as an explanatory variable demonstrated better performance in terms of relative richness, similarity, and total abundance.

3.2.1.5 Discussion of the results

We find that differences between reference sites and restoring forests are highly similar across forest restoration types for animals and plants. This finding is in line with studies that show similar biodiversity levels between actively and passively restored forests (Cruz-Alonso et al. 2019; Staples et al. 2020), but it contrasts with previous findings of higher biodiversity in passive than actively restored forests (Crouzeilles et al. 2017), and with findings that show significant effects of prior land use on biodiversity in restoring forests (Meli et al. 2017). Similar biodiversity effects across active and passive restoration types suggest that large financial investments in active reconstruction following cropland or pasture abandonment are likely unnecessary. This is in line with previous studies that discourage large financial investments in tree planting in areas where forests can regenerate by themselves (Chazdon 2008; Lamb, Erskine, and Parrotta 2005).

Instead, tree planting investments could shift to heavily degraded areas (e.g., old mining sites), where reclamation of forests is only possible through tree planting (Holl and Aide 2011). As we show that active reclamation may foster comparative richness and similarity compared to active reconstruction and land abandonment, our results suggest that investments to reclaim heavily degraded land could pay off, in particular to restore animal species richness.

When assessing changes in biodiversity in response to forest restoration over time, our results suggest that biodiversity recovery in restoring forests depends on prior land use and varies across biodiversity indicators. Our finding that biodiversity recovers fastest in abandoned croplands, abandoned pastures and active reconstruction systems across biodiversity indicators indicates that biodiversity recovery is fastest when less degraded land, such as croplands and pastures, is being restored than when more degraded land, such as mines (i.e., under active reclamation) and burned cropland are being restored. This corroborates previous findings that show slower forest restoration on land that has been very intensively used and, hence, is heavily degraded, than on land that is less intensively used (Jakovac et al. 2021). Our findings further contrast with previous studies that highlight that biodiversity recovers faster under passive than active restoration (Crouzeilles et al. 2017). These previous findings could have potentially shown effects of prior land use, as it is more likely that studies on restoration through tree planting were carried out on heavily degraded land than studies on natural regeneration of forests, as the latter may be inhibited in severely degraded areas.

Finally, we find that the level of biodiversity that can be found in restoring forests depends on indicator choice. More specifically, we find the largest differences in intactness between restored forests and reference sites followed by similarity, richness and abundance. This suggests that forests that are being restored first and foremost have different species compositions than those of natural reference sites. Yet, similar richness and total abundance between restored forests and reference sites indicates that restoring forests provide habitats for different types of species than those of natural reference sites. This finding may potentially relate to the different successional state of restoring forests compared to reference sites, where restoring forests may inhabit early to mid-successional species and reference sites late successional species. This finding indicates that although restoring global forests has significant potential to enhance biodiversity it will likely take a long time before habitats of species that are specifically adapted to old-growth forests return. This is in line with previous literature that shows effects of land use on forest structure and composition remain present for many decades (Jakovac et al. 2021).

The limited differences observed between biodiversity in actively and passively restoring forests compared to reference sites may stem from the relatively small number of studies and observations available for each restoration type, leading to broad and overlapping confidence intervals. An increased number of studies and observations could have improved the reliability of our assessment and narrowed the confidence intervals. Thus, more research is needed to compare biodiversity in reference sites with that of restoring forests. Additionally, large confidence intervals might reflect variations in initial conditions before forest restoration, such as differences in land use intensity (e.g., fertilizer and pesticide application) and land degradation. The impact of passive versus active restoration could have been obscured by these prior land use effects. For example, similar biodiversity levels in actively restored areas and abandoned cropland or pasture might result from either similar pre-restoration land use intensities or variations in degradation levels before active restoration. Future research should aim to disentangle the effects of active versus passive forest restoration and the influence of prior land use to better understand their relative impacts. Our findings for active reclamation systems should be interpreted cautiously as they are based solely on data from Australia, specifically old mining areas, which limits the variability in prior land use in our dataset. Data from diverse global regions and different types of highly degraded areas, such as severely degraded agricultural lands, might have affected our results differently. Similarly, our findings for abandoned slash-and-burn cropland should be interpreted with care due to considerable uncertainties and sensitivities observed in this system, consistent with studies that highlight the highly variable impacts of abandoned slash-and-burn practices on biodiversity and forest structure (Mukul and Herbohn 2016).

Including continents as an additional fixed effect in the models, alongside restoration systems, did not enhance the performance of the animal biodiversity models. This suggests that our findings are broadly applicable across different geographical regions for animal biodiversity; yet it may also suggest that there is too little data to detect differences. In the future, continental differences in biodiversity effects of forest managment could be further explored. Additionally, differences between ecoregions, biomes and realms could be explored. Despite significant spatial biases in data availability, the impact values observed in this study can likely be generalized globally. Conversely, the results may be influenced by the specific taxonomic groups evaluated for intactness and similarity, underscoring the need for future research to explore variability in biodiversity responses across different taxa. Currently, our combined database lacks sufficient studies to conduct such an analysis, highlighting the need for additional data to more accurately differentiate the impacts of various restoration types on different taxonomic classes. Additionally, biodiversity data for some restoration types are available over extended time periods, while for others, the data cover only shorter durations. This discrepancy limits our ability to fully understand the dynamics of biodiversity recovery over time and to make comprehensive comparisons across systems. For example, the observed reduction in intactness over a 17-year period of active reclamation might be affected by the limited time available for biodiversity to recover, preventing us from assessing the long-term outcomes of restoration. Moreover, the effects of landscape diversity might overshadow the impacts of restoration efforts (Brotons et al. 2003). For instance, natural reference sites and restoring forest sites differ in size and are bordered by various land uses, such as cropland, natural reference forests, or urban areas. Finally, there are other restoration types that are not considered in this study, such as peatland restoration, which have been shown to affect biodiversity (Ramchunder, Brown, and Holden 2012). Future studies could assess biodiversity changes in response to such restoration types.

3.2.2 Changes in biodiversity in response to forest restoration in Europe

In total, our search term provided 1,128 hits for publications that were of potential interest. After screening, we ended up with 29 publications that met our criteria. From these publications, we derived data for 12 animal and 28 plant studies, derived from 12 and 17 publications, respectively. A single study may contain data on multiple restoration types and multiple taxa. In total the database included 15,543 abundance values of 853 animal species and 11,462 abundance values of 1,379 plant species. Studies were spread relatively evenly across the European continent (Fig. 14). Variations in data availability occurred among taxa, with the highest number of studies included for plants (28), followed by arthropods (8), birds (3) and mammals (1) (Appendix Fig. H1).

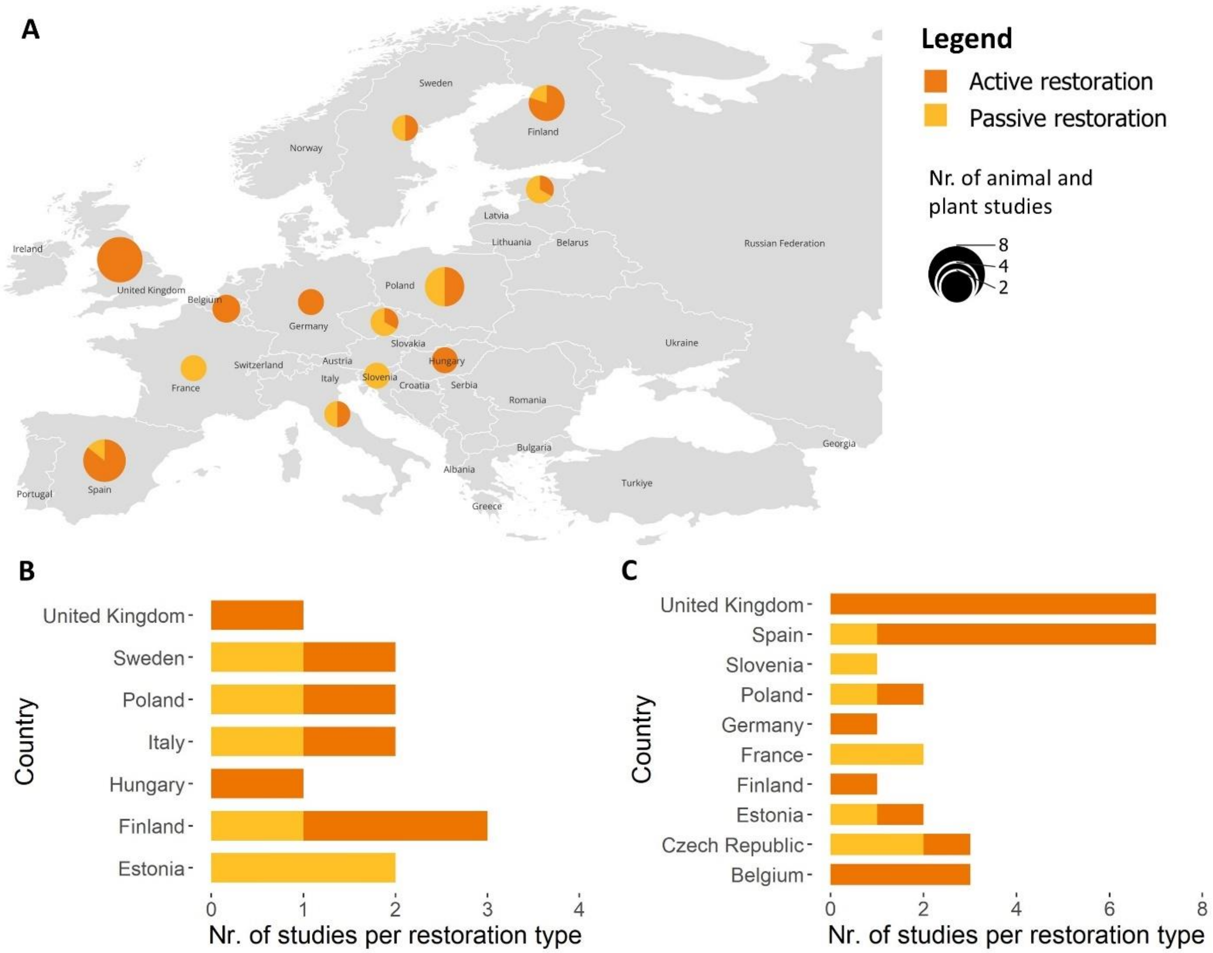


Figure 14. Spatial distribution and numbers of included studies. (A) European distribution of empirical biodiversity studies categorized by forest restoration type. The sizes of the pie-charts indicate the number of studies per country included in the databases, the scaling of which is shown in the black circle on the right bottom of Panel A. (B) Animal biodiversity studies per country categorized by forest restoration type. (B) Plant biodiversity studies per country categorized by forest restoration type. Each study can contain multiple restoration types and cover multiple taxonomic classes.

3.2.2.1 The biodiversity of restoring forests

We found significantly lower animal and plant intactness and similarity in all forest restoration types than in natural reference sites (Fig. 15-16, Appendix Table G1). Effects of forest restoration on intactness were approximately similar across all types for both animals and plants. We found similar animal relative richness in natural reference sites as in forest restoration types. The relative total abundance of active restoration sites is slightly but significantly lower than that of natural reference sites. For plants we find significantly higher richness in active restoration sites than in natural reference sites, while total abundance of active restoration sites is lower than that of natural reference sites. We find a significantly lower total abundance but higher similarity in active restoration sites than in passive restoration sites.

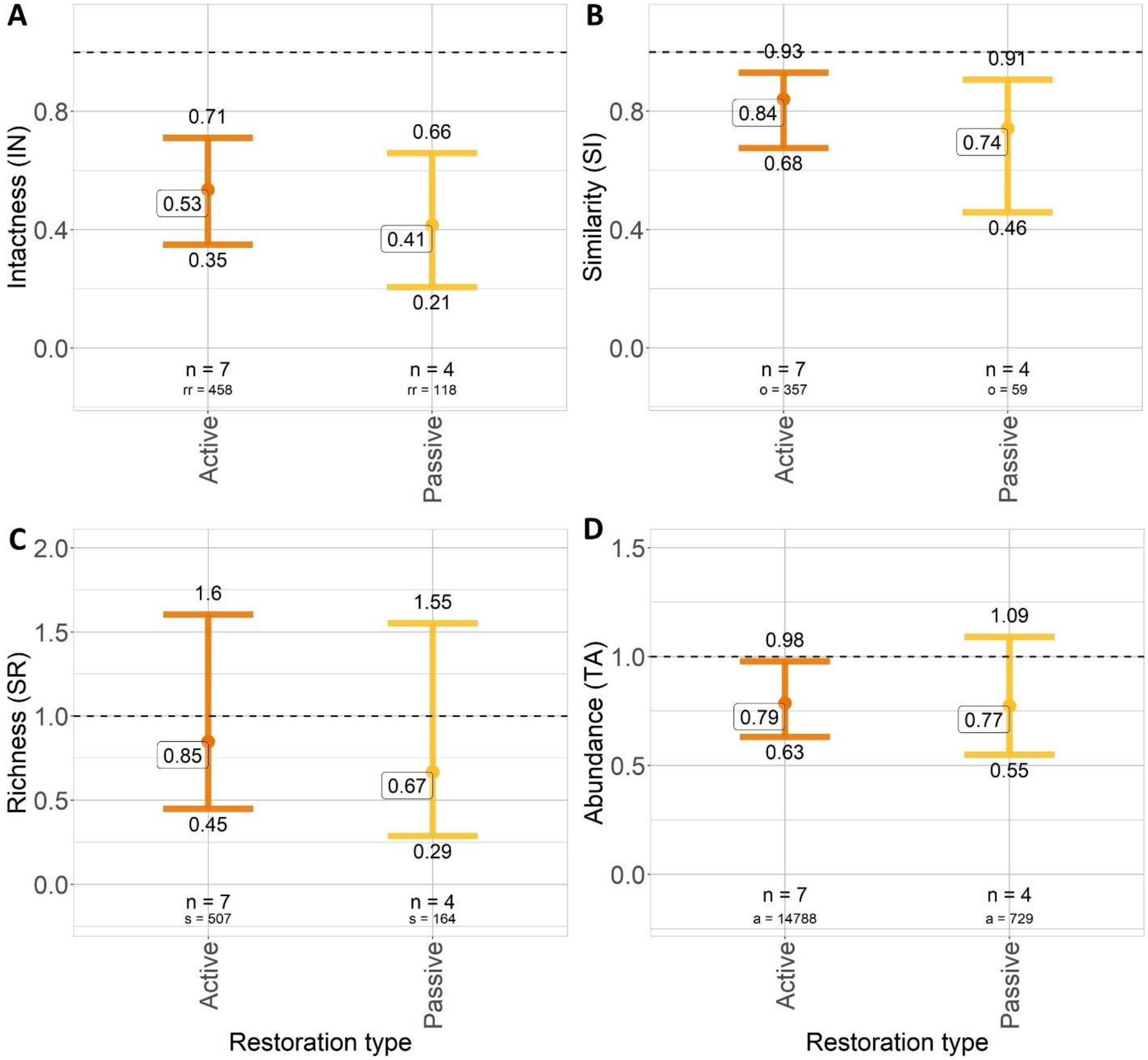


Figure 15. Animal biodiversity in forest restoration sites, relative to that of natural reference sits. Animal biodiversity was expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish two forest restoration types: (i) active and (ii) passive restoration. n = number of studies included per restoration type, rr = number of response values used to compute intactness, o = number of overlapping species between restoration and reference forest sites, s = number of species occurring across the restoration and reference sites, a = number of abundance values used to compute relative total abundance.

3.2.2.2 Biodiversity changes over time in restoring forests

We found varying changes in animal and plant biodiversity over time and across indicators in response to forest restoration (Fig. 17; Appendix Table G2). For passive restoration, we find small (< 0.1) but significant increases in animal intactness and similarity over a period of 68 years and a small (<0.1) significant increase in richness over a period of 160 years. In contrast, for active restoration we find small (± 0.1) but significant declines in animal intactness and similarity over a period of 179 years and a small significant decrease in plant intactness (± 0.2) over a period of 230 years.

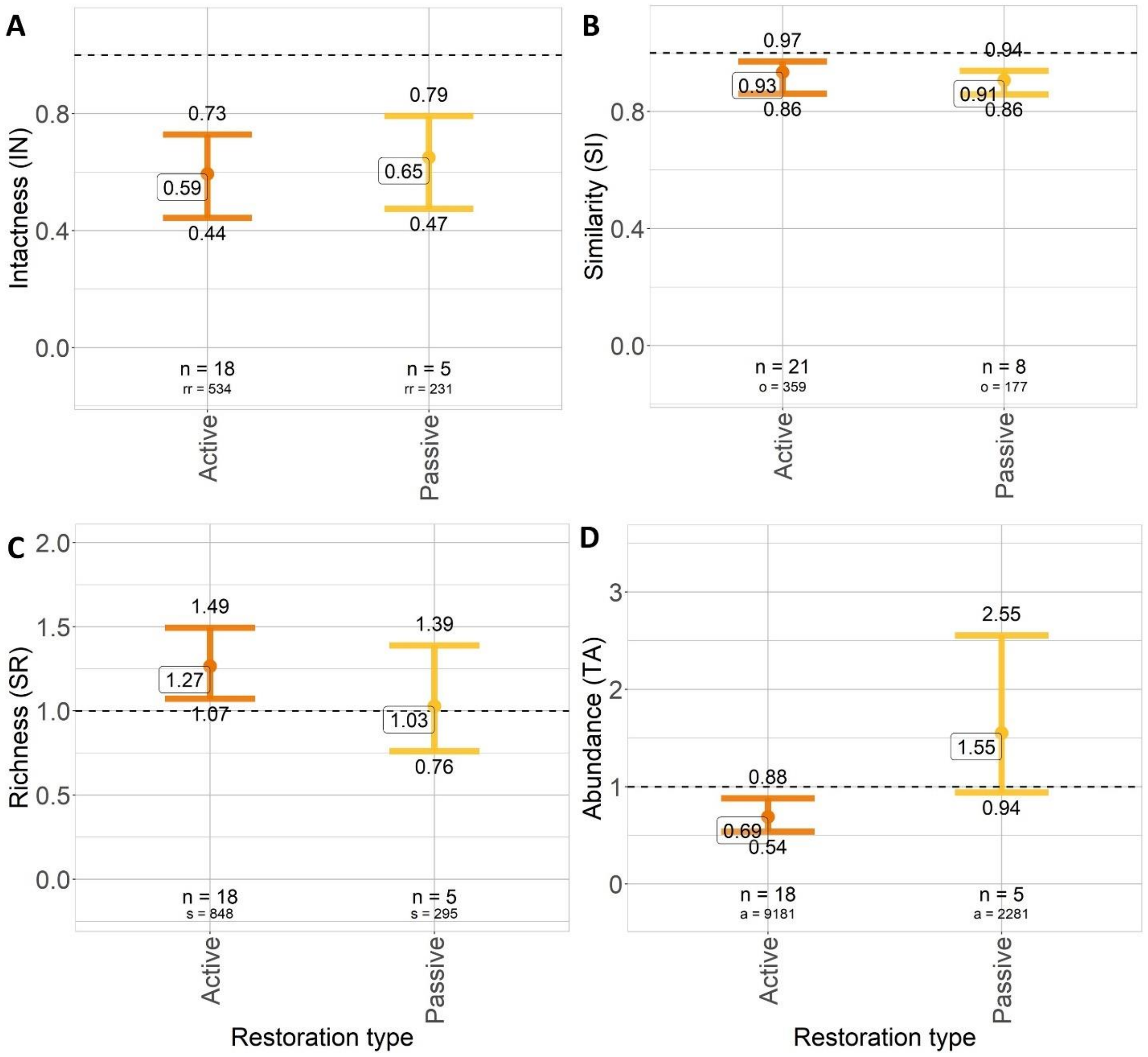


Figure 16. Plant biodiversity in forest restoration sites, relative to that of natural reference sits. Plant biodiversity was expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish two forest restoration types: (i) active and (ii) passive restoration. n = number of studies included per restoration type, rr = number of response values used to compute intactness, o = number of overlapping species between restoration and reference forest sites, s = number of species occurring across the restoration and reference sites, a = number of abundance values used to compute relative total abundance.

3.2.2.3 Correlations between biodiversity indicators

We found that most biodiversity indicators positively related to each other with correlations between 0.3 and 0.9 ρ for animals and -0.1 and 0.9 ρ for plants, indicating that the indicators are generally correlated with each other, yet that there are differences between them. Additionally, the principal components do not overlap (Appendix Fig. H2). Therefore, the biodiversity indicators differ from each other and could be considered distinct indicators of biodiversity, each capturing their own specific aspects of biodiversity.

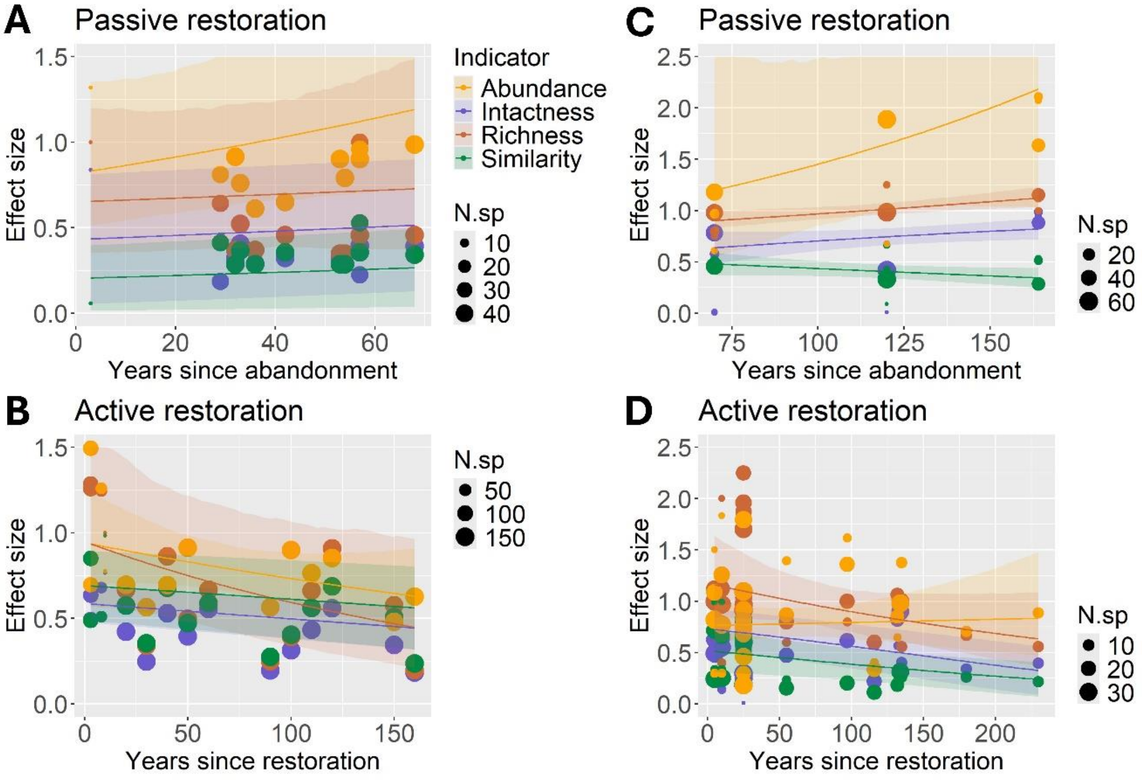


Figure 17. Relationships between time since the start of restoration or since abandonment and animal (A-B) and plant (C-D) biodiversity expressed as intactness (IN), relative richness (SR), similarity (SI) and relative total abundance (TA). N.sp = the number of species that are included to calculate biodiversity metrics / response ratios per time since restoration or abandonment.

3.2.2.4 Discussion of the results

When zooming into restoration types, we find that differences between natural reference sites and restored forests are highly similar across forest restoration types for animals and plants in Europe. This finding is in line with studies that show similar biodiversity between actively and passively restored forests (Cruz-Alonso et al. 2019; Staples et al. 2020), but contrasts with previous findings that indicate higher biodiversity in passive than actively restored forests (Crouzeilles et al. 2017). Yet, we also find that biodiversity recovers faster over time in passively than actively restored forests across indicators. This may indicate that over time, passive restoration benefits biodiversity more than active restoration, which is in line with previous findings (Crouzeilles et al. 2017). Planting forests disrupts natural processes, often resulting in lower or different functional diversity (Chazdon and Guariguata 2016) and, as a result, less habitats and food and shelter than naturally regenerating forests over time (Chazdon et al. 2009).

We find that the level of biodiversity that can be found in restoring forests depends on indicator choice. More specifically, we find the largest differences in intactness between restored forests and natural reference sites followed by similarity, richness and abundance. This suggests that forests that are being restored first and foremost have different species compositions than those of natural reference sites. Yet, similar richness and total abundance between restored forests and natural reference sites indicates that restoring forests provide habitats for different types of species than those of natural reference sites. This finding may potentially relate to the different successional state of restoring forests compared to natural reference sites, where restoring forests may inhabit early to mid-successional species and natural reference sites late successional species.

Our finding that passive forest restoration fosters similar levels of biodiversity as active forest restoration indicates that financial investments in tree planting may not be necessary in areas where forests can naturally regenerate. Tree planting investments could instead shift to heavily degraded areas (e.g., old mining sites), where forest restoration is only possible through tree planting (Holl and Aide 2011). Our findings contrast with policies and targets of the EU Nature Restoration Law to plant an additional 3 billion trees by 2030 to restore European forests (EU 2024). This suggests a need to carefully revise the target to plant 3 billion trees to prevent large financial investments that may have been more effectively spend for other restoration purposes, such as the restoration of heavily degraded areas or a transition to more biodiversity-friendly forest management.

Limited differences found between the biodiversity in active and passively restoring forests relative to the biodiversity of natural reference sites may result from the relatively low number of studies and observations found per restoration type, which results in relatively wide and overlapping confidence intervals. Higher numbers of observations and studies could have enhanced the robustness of our assessment and reduced confidence intervals. Hence, there is need for more studies that compare the biodiversity in natural reference sites with that of restoring forests in the future. Additionally, large confidence intervals may have resulted from differences in starting conditions prior to forest restoration, such as differences in land use and land degradation. For example, forest recovery is often slower on degraded land (e.g., mines) than on non-degraded land (e.g., non-degraded pastures), which may result in slower recovery of biodiversity in regenerating forests on degraded land (Meli et al. 2017). As active restoration is often necessary to restore forests on degraded land (Stanturf and Madsen 2002), the lack of recovery of biodiversity we observe upon active restoration may relate to higher levels of land degradation prior to active restoration. Future studies may attempt to account for these effects by creating further sub-categories of forest restoration types in Europe, e.g., using the categorization of Stanturf et al. (2002). To achieve this, more data and more detailed descriptions of forest restoration in source papers are needed. Besides limited data, the effects of landscape diversity may have overshadowed the effects of restoration on biodiversity (Brotons et al. 2003). For instance, natural reference sites and restored forest sites vary in size and border varying land uses, such as cropland, natural reference forests or urban land.

3.2.3 Comparing global and European restoration effects

We find contrasting patterns in biodiversity restoration through passive and active afforestation and reforestation at global and European scale. While biodiversity generally recovers over time upon both passive and active forest restoration at global scale, we show limited signs of biodiversity recovery in Europe over relatively long time scales up to 150 years since the start of restoration. Hence, there appears to be an inhibiting factor that prevents biodiversity from recovering to reference levels in Europe, in particular for intactness. One potential explanation could be that there are insufficient reference forests that function as seed sources due to the very small cover of reference forests in Europe (only approximately 3% of the European forests is considered old-growth (O’Brien et al. 2021)). Consequently, forests may not resemble reference forests over time but instead resemble managed forests in the neighbourhood, such as multifunctional forest plantations. Another potential explanation could be that there is too much fragmentation in the European landscape, inhibiting species from migrating from reference forests to afforested and reforested areas. Since the intactness of managed forests is generally low, in particular for animal species, influx of species from managed forests likely does not enhance the intactness of restoring forests. These findings are concerning, and we highly recommend looking further into biodiversity recovery under forest restoration in Europe to better understand our findings.

4 Models for biodiversity and ecosystems services

We use two models to simulate biodiversity and ecosystem services: (i) the GLOBIO model, a biodiversity model for policy support (Schipper et al. 2020), and (ii) the LandSyMM model, in particular the CRAFTY module (Brown, Seo, and Rounsevell 2019). Here we provide a description of the two models. For GLOBIO, we firstly describe the GLOBIO terrestrial biodiversity model that will be linked with CRAFTY and run to help assess trade-offs between biodiversity, climate change mitigation and ecosystem services under WP5 scenarios. For GLOBIO, we secondly describe the GLOBIO ecosystem services model that will be used to help assess trade-offs between biodiversity, climate change mitigation and ecosystem services under WP5 scenarios. Finally, we describe updates that were done to ecosystem services modules of the GLOBIO and LandSyMM models. Both the LandSyMM and GLOBIO model will be used to simulate changes in ecosystem services across the different scenarios developed in WP5 of the ForestPaths project. GLOBIO will use LandSyMM land use output to model changes in ecosystem services, such as recreation across scenarios.

4.1 GLOBIO terrestrial

The GLOBIO model is a global biodiversity model for policy support (https://www.globio.info/, last accessed 28.11.2024). The GLOBIO terrestrial model is used to simulate the effect of drivers of biodiversity loss on terrestrial biodiversity (i.e., land use/management, atmopsheric nitrogen deposition, climate change, road disturbance, fragmentation, and hunting) into the future in response to different scenarios (Schipper et al. 2020). The GLOBIO terrestrial model is described in Appendix I.

4.2 GLOBIO-ES

The GLOBIO ecosystem services (ES) model is used to simulate changes in ecosystem services provisioning and demand (e.g., carbon sequestration, pollination) into the future in response to different scenarios (Schulp et al. 2012). The GLOBIO-ES model is described in Appendix J.

4.2.1 New recreation module in GLOBIO-ES

We added a new ecosystem services module in GLOBIO that can be used to simulate recreation potential (i.e., the opportunity for recreation in a given area) across Europe. The module on recreation is based on two elements: (i) the appreciation of landscapes by people as an indicator of recreation potential, referred to as landscape aesthetic quality, and (ii) the accessibility of recreational areas and landscapes (see Appendix J for a description of GLOBIO-ES). The combination of these two elements allows for the computation of a recreation opportunity spectrum map, which shows the opportunity for recreation in a given area. We based our approach for calculating recreation opportunity on the approach of Paracchini et al. (2014). The advantage of basing our approach to recreation modelling on the approach of Paracchini et al. (2014) is that we were able to compare our recreation opportunity output to that published in a peer reviewed article.

4.2.1.1 Landscape aesthetic quality

The approach for mapping landscape aesthetic quality is developed by Tisma et al. (under review). Landscape preference is influenced by both cognitive and physical landscape attributes (Walz and Stein 2018; Hedblom et al. 2020). Based on an extensive review of literature on landscape appreciation by people and prior modelling and mapping approaches, Tisma et al. (under review) decided to capture these cognitive and physical landscape attributes by combining indicators for three different landscape preference categories: (i) scenic beauty, (ii) visual diversity, and (iii) tranquillity.

- **Scenic beauty:** Scenic beauty was defined as the biophysical attractiveness of the landscape in the eye of the recreationist, which is determined by features in the landscape, such as land use types (Tisma et al., under review). Scenic beauty is subjective, yet various landscape attributes, such as the naturalness of the vegetation, the presence of water and relief have been found to enhance scenic beauty for most observers. Tisma et al. (under review) measured scenic beauty using the following three indicators:
    - *Naturalness*: Landscapes that include large areas of natural vegetation, in particular forests, are experienced to be more visually attractive than landscapes with limited natural vegetation, such as urban areas and croplands (Paracchini et al. 2014; Walz and Stein 2018). Naturalness was determined using a land use layer as input for the aesthetic value module (Tisma et al., under review). As land use/management input an ESA-CCI map, GLOBIO land allocation output or CRAFTY AFT layers can be used. In the current module, naturalness was determined by scoring land uses from 0 to 4 (see Table J2 for the scoring approach).
    - *Presence of water and coast*: Landscapes in which water is present are experienced to be more visually attractive than landscapes without water (Herbst, Förster, and Kleinschmit 2009; Paracchini et al. 2014). Presence of water was determined using a land cover layer as input for the aesthetic value module (Tisma et al., under review). As land cover input an ESA-CCI map, or GLOBIO land allocation output can be used. In the current module, the presence of water and coast was scored from 0 to 3 (see Table J2 for the scoring approach).
    - *Relief*: Landscapes with higher levels of relief, defined as the heterogeneity and variability in geomorphology, are experienced to be more visually attractive than landscapes with low levels of relief (Aminzadeh and Ghorashi 2007). Relief was determined using elevation layers as input for the aesthetic value module, such as data from Digital Elevation Models. Areas with much relief, such as mountains, score high because they are visible from a distance (Tisma et al., under review). In the current module, relief was scored from 0 to 2 (see Table J2 for the scoring approach).
- **Visual diversity:** Visual diversity was defined as the heterogeneity and richness of visual stimuli from the landscape, which was determined by the presence of a combination of elements, such as water and different types of vegetation, as well as the heterogeneity of land use (Tisma et al., under review). An extensive review was conducted to identify the most common approaches to estimate visual diversity. The following three most commonly used indicators were adopted:
    - *Shannon diversity index*: This is a commonly used index to derive landscape quality in terms of its visual diversity (Schüpbach et al. 2016). This index determines the number of land use classes and their proportional distribution, where more diverse landscapes are assumed to be more preferred by people than less diverse landscapes. In the current module, Shannon diversity scores were rated from 0 to 4 (see Table J3 for the scoring approach).
    - *Shannon evenness index*: This is a commonly used index to derive landscape quality in terms of its visual diversity (Schüpbach et al. 2016). This index

determines the distribution of area among land-cover classes, where areas with an even distribution of land use classes are assumed to be more preferred by people than areas with an uneven distribution of land use classes. In the current module, Shannon evenness scores are rated from 0 to 4 (see Table J3 for the scoring approach).

  - *Aggregation index*: This is a commonly used index to landscape quality in terms of the weighted proportion different land use classes take up (Fang et al. 2024). The higher the value of the aggregation index the more aggregated the distribution of land use classes. In the current module, aggregation scores are rated from 0 to 4 (see Table J3 for the scoring approach).
- **Tranquillity:** Landscape tranquillity was defined as the peacefulness and calmness of the landscape to the different senses of humans, in particular sight and hearing, which is, among others, determined by light, noise and being alone/away from others (Tisma et al., under review). Tisma et al. (under review) measurekd tranquillity using the following three indicators:
  - *Darkness*: Absence of light pollution enhances the experience of peace and calmness (Hewlett et al. 2017). The less light pollution the more tranquil the landscape is assumed to be. This indicator was measured using the nighttime light data (average DMSP F16 values 2009) from the National Oceanic and Atmospheric Administration (NOAA), which is available under licence at their data hub (http://lossofthenight.blogspot.com/2014/10/last-nights-lunar-eclipse-viewed-from.html, last accessed 27.11.2024). In the current module, darkness was scaled from 0 to 2 (see Table J4 for the scoring approach).
  - *Silence*: Silence is measured as the absence of noise disturbance (Hedblom et al. 2020). The method of the European Environment Agency was adopted to estimate noise disturbance (European Environment Agency 2014), using GRIP road data as input (Meijer et al. 2018). See the European Environment Agency report on Noice in Europe for a detailed description of the methodology (European Environment Agency 2014). The lower the noise pollution, the more tranquil the landscape was assumed to be. In the current module, silence was scaled from 0 to 2 (see Table J4 for the scoring approach).
  - *Solitude*: Solitude was defined as the extent to which people are secluded (i.e., away from others) (Hedblom et al. 2020). Solitude was measured using population density derived as number of people per km^2, using data derived from the Gridded Population of the World dataset (Deichmann, Balk, and Yetman 2001). In the current module, solitude was scaled from 0 to 2 (see Table J4 for the scoring approach).

All indicators of aesthetic quality were combined by first scoring each sub-indicator value between 0 and 1 by dividing the indicator values by the maximum value obtained for that indicator. These indicator values were used to compute an average scenic beauty, visual diversity and tranquillity, respectively. The final values for the three aspects of aesthetic quality were summed together, where scenic beauty counted twice because of the high number of indicators used to compute this landscape aesthetic quality aspect, resulting in aesthetic quality values between 0 and 4. To visualize the aesthetic quality values, all values were summed with 1; hence, the aesthetic quality indicator ranges between 1 and 5. Unlike the recreation potential indicator of Paracchini et al. (2014), the landscape aesthetic value only provides an indication of the recreation potential of terrestrial areas, not of salt water and freshwater areas. All types of land cover and management types were considered to estimate aesthetic quality. Forests play an important role in the module, as these ecosystems are experienced by recreationists to be more visually attractive (Paracchini et al. 2014; Walz and Stein 2018). Forest cover and management affect aesthetic quality in two ways. First, forest cover and management affect scenic beauty because if influences the naturalness of the environment. Second, forest management may also affect visual diversity because it can influence the variation of land uses in the landscape.

4.2.1.2 Landscape accessibility

To determine accessibility of recreational areas and landscapes, we adopted the approach of Paracchini et al. (2014). Paracchini et al. (2014) determined the accessibility of areas based on two variables: (i) distance to roads and (ii) distance to urban areas. Paracchini et al. (2014) parameterized accessibility across Europe based on expert opinions on maximum and preferred travel times to recreation areas. To implement the approach, we first computed Euclidean distances from the roads of the GRIP database (Meijer et al. 2018). Besides this, we computed the Euclidean distance from areas classified as urban areas by GLOBIO. We then combined the two outputs together to scale accessibility between 1 and 5, using the approach of Paracchini et al. (2014) (Table 4). The value 1 indicates that the recreation area is close by, meaning highly accessible, due to low distances from urban areas (<5 km) and low distances from the road (<1km). The value 5 indicates that the recreation area is far away meaning very remote, because of both high distances to urban areas (>50 km) and high distances from the road (>10 km).

We combined the aesthetic quality and accessibility output to derive a recreation opportunity spectrum map. We first normalized the aesthetic quality values between 0 and 1, using Equation 5:

$$I_i = \frac{\sum_i^n S_i}{n} \tag{5}$$

where *I* is the average aesthetic quality indicator value, *S* is the value of the indicator *i* and *n* is the number of indicators. Hereafter, we determined the quantile distribution of the aesthetic quality map and used these quantiles to determine low (lower quantile), high (upper quantile) and medium (between lower and upper quantile) provisioning of landscape aesthetic quality. Table 5 shows how the recreation opportunity spectrum was calculated using the accessibility map and the landscape aesthetic quality map.

Table 4. Classification of accessibility, using distance from the road and distance from urban areas as indicators (this table is copied from Table 1 of Paracchini et al. (2014)). The color coding matches the legend of Fig. 19.

		Distance from road (km)			
		<1	**1-5**	**5-10**	**>10**
Distance from urban areas (km)	**<5**	1	2	2	4
	5-10	2	2	2	4
	10-25	3	3	3	4
	25-50	3	4	4	4
	>50	4	4	4	5

Table 5. Classification of accessibility, using distance from the road and distance from urban areas as indicators (this table is copied from Table 2 of Paracchini et al. (2014)). Low indicates low provisioning, medium indicates medium provisioning and high indicates high provisioning of aesthetic quality. The color coding matches the legend of Fig. 20.

		Aesthetic quality		
		< 0.64 (low)	**0.64-0.75 (medium)**	**>0.75 (high)**
	1 Neighborhood	1	4	7

Accessibility	**2 Proximity**	1	4	7
	3 Far	2	5	8
	4 Remote	3	6	9
	5 Very remote	3	6	9

4.2.1.3 Testing the recreation module and future use

We tested the recreation module with preliminary baseline GLOBIO land use data as input, which prepares us to analyze changes in recreation opportunity in Europe across policy pathways developed in the ForestPaths project. We created three test maps for Europe: (i) a map for landscape aesthetic quality (Fig. 18), (ii) a map for accessibility (Fig. 19), and (iii) a map reflecting the recreation opportunity spectrum (Fig 20). The intention of producing these maps was not to study recreation patterns across Europe just yet, but to determine the validity of our approach. To assess the validity of our approach to recreation modelling, we qualitatively compared preliminary baseline accessibility and recreation opportunity spectrum maps for the year 2015 with those of Paracchini et al. (2014). We compared the accessibility and recreation opportunity maps to output of Paracchini et al. (2014) to qualitatively assess the validity of our approach (see Fig. 4 of Paracchini et al. (2014)). We found similar patterns in accessibility across Europe. Additionally, we found generally similar patterns in recreation opportunity across Europe (see Fig. 5 of Paracchini et al. (2014)).

Once policy pathways have been developed within the ForestPaths project, we will assess changes in recreation opportunity in response to the different scenarios, with land management/use maps from CRAFTY and GLOBIO as input. The output of these scenarios will reveal the effects of changes in forest related land use and management on recreation potential.

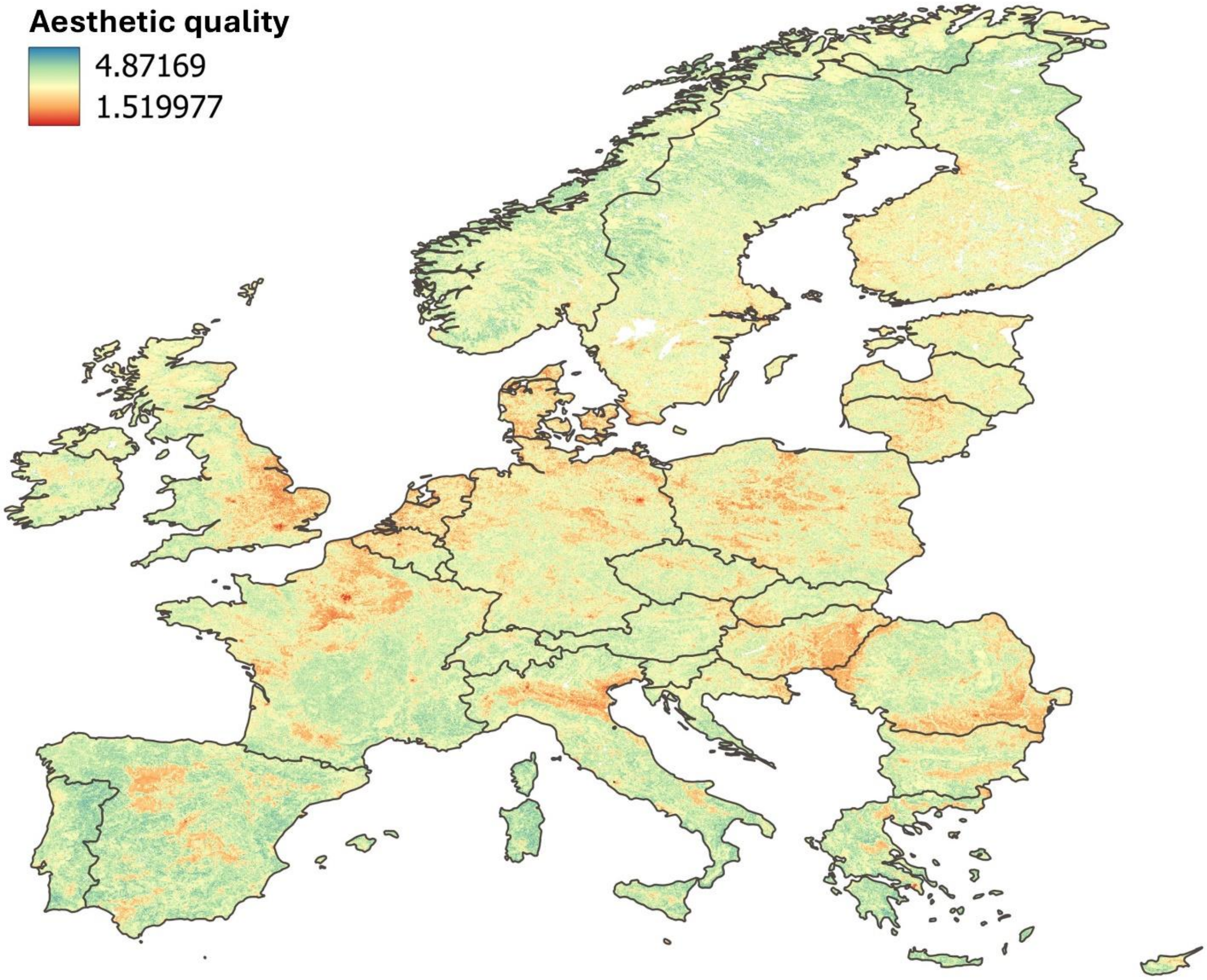


Figure 18. Preliminary aesthetic quality patterns in Europe across all ecosystems derived using the approach of Tisma et al. (in review). The intention of producing this map was not to study aesthetic value patterns across Europe just yet, but to determine the validity of our approach. The aesthetic quality indicator ranges between 1 and 5, where 1 indicates lowest aesthetic quality and 5 the highest aesthetic quality. Aesthetic quality is assessed across ecosystems; hence this map does not only refer to the aesthetic quality of forests. As aesthetic quality is influenced by all land use and management types together, as well as their spatial composition, it should be interpreted and examined holistically across land uses.

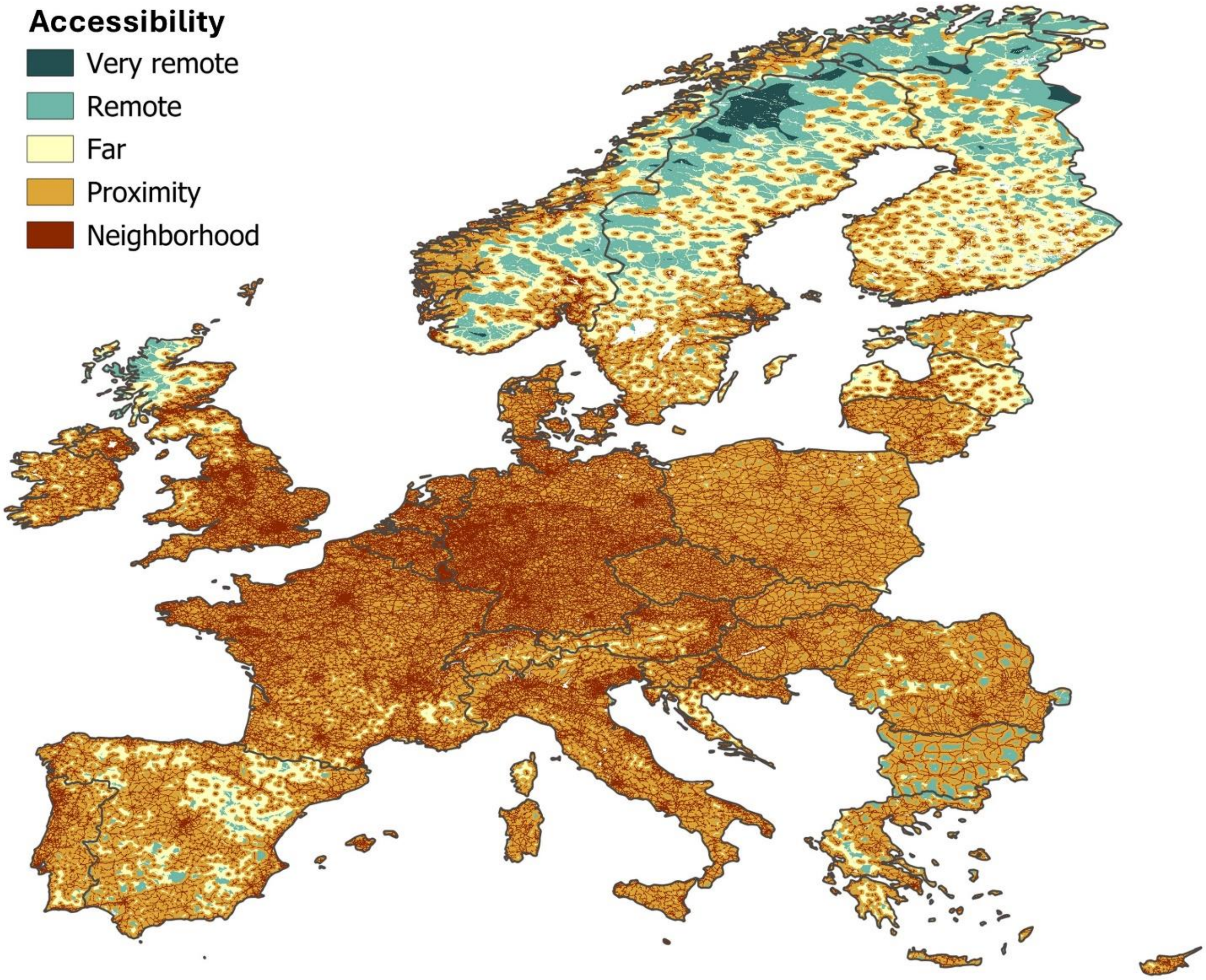


Figure 19. Patterns in accessibility of areas in Europe derived using the approach of Paracchini et al. (2014). The figure is modelled after Fig. 4 of the paper of Paracchini et al. (2014), to allow for comparison. The intention of producing this map was not to study accessibility patterns across Europe just yet, but to determine the validity of our approach.

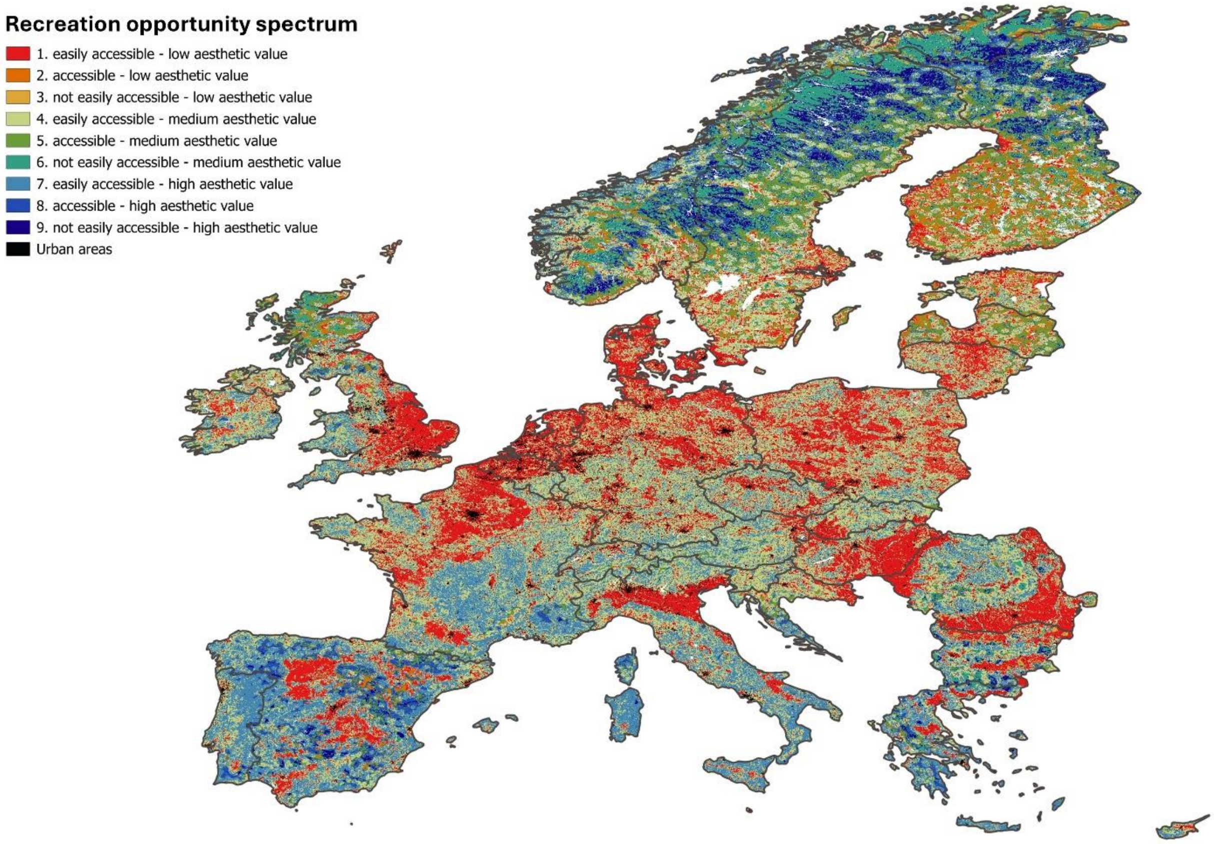


Figure 20. Patterns in the recreation opportunity spectrum in Europe derived by combining the aesthetic value and accessibility preliminary basemaps using the approach of Paracchini et al. (2014). The intention of producing this map was not to study recreation opportunity value patterns across Europe just yet, but to determine the validity of our approach. The figure is modelled after Fig. 5 of the paper of Paracchini et al. (2014), to allow for comparison. Aesthetic quality is assessed across ecosystems; hence the recreation opportunity spectrum map does not only refer to the aesthetic quality of forests. As aesthetic quality is influenced by all land use and management types together, as well as their spatial composition, it should be interpreted and examined holistically across land uses.

4.3 LandSyMM-EU ecosystem services

LandSyMM (see https://landsymm.earth/) is a sub-national to global model of the land system, that incorporates modules on ecosystem processes, decision-making, trade and land use change. LandSyMM-EU is the European-scale version of the model. In LandSyMM-EU, the CRAFTY model is an important module. In the CRAFTY model, agents produce ecosystem services in response to demand for them. How much they produce is based on their access to natural and socioeconomic capitals, their sensitivity to the capitals and their overall ability to produce each service. Ecosystem services are then aggregated across space to enable an evaluation of the demand-supply gap in ecosystem service provision. CRAFTY-EU is an EU27+3 (Norway, Switzerland and UK) application of the CRAFTY agent-based modelling framework, which simulates large-scale land use change (Murray-Rust et al. 2014). It has a 1 km resolution and builds off the previous 5 km CRAFTY-EU version (Brown, Seo, and Rounsevell 2019). Land users are categorised into Agent Functional Types (AFTs) based on their behaviours and what they produce. Agents compete for the land and use a range of resources, known as capitals, to produce ecosystem services. Natural capital inputs, and demands for food and timber, are provided by the LPJ-GUESS, and PLUM, modules of LandSyMM, respectively. The demands for non-material goods, such as recreation, are defined using scenario narratives. The model is currently parameterised for the European Shared Socioeconomic Pathways (SSP) (Kok et al. 2019) for 2020 until 2100.

One of the major updates to the new CRAFTY-EU model was the increase and improvements in the range of provisioning, regulating and cultural ecosystem services that agents in CRAFTY-EU can produce. The set of services are described individually in Table 6, along with how they have been calibrated, using empirical and modelled evidence where available. Previously, the production of each service was dependent on the Cobb-Douglas function by default, used to combine optimal production levels with dependence on each capital to give service productivity. For non-material services such as flood control, soil erosion, and rural employment, these are now produced using a set of equations which account for other factors, such as the AFT, resource availability, topography etc.

Table 6. Services supplied by agents within the CRAFTY models. The status indicates whether this is a new service, whether the mechanisms for calibration and/or supply have been updated, or if it remains as in the original CRAFTY-EU version in Brown et al. (2019).

Service	Status	Explanation	Source and method of service production
Food crops	Original	Crops for human consumption: C3 cereals, starchy roots, oil crops, pulses, fruit and veg, C4 crops	The supply of these services comes directly from LPJ-GUESS, based on the natural productivity, as well as the access and sensitivity an agent has to the socioeconomic capitals
Fodder crops	Original	Crops for livestock	
Grass-fed meat	Original	Pasture-fed (ruminant) meat	
Grass-fed milk	Original	Pasture-fed (intensive) milk	
Bioenergy fuel	New	First generation – food crops e.g. maize, Second generation - non-food crop e.g. miscanthus	
Wood	Updated	Divisible into softwood and hardwood	
Sustainable production	Updated	Abstract service as proxy for organic demand	Abstract attribute of sustainable/extensive/ organic systems represent scenario-based demand for those management practices (as opposed to specific services).
Solar energy	New	Energy provided by solar power installations	Agents leverage solar capital, derived from the photovoltaic potential of Saxena et al. (2023) to produce solar energy.
Carbon sequestration	Updated	Potential GHG sequestration by ecosystems (climate regulation)	CRAFTY has a basic ranking of land uses for carbon sequestration. For more detailed calculation of carbon sequestration, the annual AFT maps for the model runs are fed back to LPJ-GUESS and used a seamless historical future set-up to compute carbon cycle impacts in crops, forests and pastures (set-up under development)
Flood control	New	Flood control expressed by the potential runoff generated by rainfall.	Calculation of curve number with dynamic land use adjustment. Approach adjusted from Vallecillo et al. (2020).
Soil erosion	New	Soil loss due to water run off i.e., (inter)-rill-erosion.	RUSLE equation with dynamic land use adjustment. Approach adjusted from Borrelli et al. (2017).
Pollination & seed dispersal	New	Potential diversity & abundance of animal pollinators & seed dispersers	Ranking approach based on literature evidence and Species Distribution Model results.

Landscape heterogeneity	Updated	Diversity within and/or between land uses in a landscape.	Shannon Diversity Index calculated for each AFT per country, using EUNIS habitat level 2 data (EEA 2019).
Rural employment	New	Potential employment in forestry and agriculture	Relationship between number of forestry agents and forest employment derived from Eurostat employment in forestry and forestry-related jobs dataset for each country. Agricultural employment will be derived using statistical methods to obtain estimates for agricultural employment per square kilometre for each aft using the FADN data of agricultural employment.
Cultural ecosystem services	Updated New New New	**Recreation**: Characteristics of living systems that enable activities promoting health or enjoyment through active or immersive interactions **Cultural landscapes**: cultural landscape index **Identity**: stability of landcover **Experience/inspiration**: Related to recreation but focused on the effects of it; e.g., spirituality, connection or contact with nature, relaxation, reflection, aesthetic experience, inspiration by nature	A popular indicator for assessing cultural ecosystem services are geo-tagged, crowd-sourced photos from online photo-sharing platforms such as Flickr (Figueroa-Alfaro and Tang 2017; Lee et al. 2019). Based on the content displayed on the photos, a transformer-based zero-shot classification is used to categorise the content of the photos. This analysis is combined with secondary data to categorise recreation, cultural landscapes (e.g. using Tieskens et al. (2017)), experience and inspiration, and identity (using the HILDA+ dataset (Winkler et al. 2021). The analysis is currently under development and subject to modifications.

5 Linking and applying improved models

We linked GLOBIO with LandSyMM, specifically the CRAFTY module, to enable trade-off analyses between biodiversity, climate change mitigation, and ecosystem services for WP5. We fully integrated the effects of two drivers of biodiversity loss on biodiversity in CRAFTY: (i) land use/management and (ii) climate change. The land use/management effects on biodiversity were derived from a combination of estimates from a European meta-analysis on effects of forest management on biodiversity and a global analysis of forest management effects on biodiversity (see Section 2.1), as well as the GLOBIO 4 model land use impact assessment (see Section 4.1 and Appendix I). Relationships between biodiversity and climate change were directly derived from the GLOBIO 4 model (Section 4.1 and Appendix I). By integrating effects of land use/management and climate change on biodiversity in CRAFTY, we are able to simulate feedbacks between AFTs and biodiversity over time to assess trade-offs between biodiversity and ecosystem services for WP5. To this end, land use output of CRAFTY will be used as input for GLOBIO to assess the effects of remaining drivers (i.e., N deposition, fragmentation and road disturbance) on biodiversity over time. Finally, we explored potential links between the GLOBIO and LPJ-GUESS model to assess whether output of LPJ-GUESS could be used to assess effects of forest management on biodiversity in GLOBIO.

5.1 Integrating effects of land management and climate change on biodiversity into the CRAFTY module

We fully integrated the effects of two drivers of biodiversity loss on biodiversity in CRAFTY, (i) land use/management and (ii) climate change.

5.1.1 Integrating effects of land management on biodiversity in CRAFTY

Both GLOBIO and CRAFTY model land use and land management by categorizing them into different classes (e.g., agroforestry, plantations) and assigning biodiversity and ecosystem services values to them. We introduced the results of Section 3 in the CRAFTY model by carefully matching the land management classes for which we derived biodiversity estimates with CRAFTY AFTs during a two-day workshop at KIT in December 2023. We matched any remaining CRAFTY AFTs with GLOBIO 4 land use/management classes derived from Schipper et al. (2019). Table K1 of Appendix K provides an overview of the AFTs of CRAFTY and the matching land use/management classes of GLOBIO. Once all CRAFTY AFTs were matched with GLOBIO land management and use classes, corresponding MSA values were assigned to each AFT. Table K2 provides an overview of the CRAFTY AFTs and corresponding GLOBIO MSA values integrated into CRAFTY.

Although we matched the CRAFTY and GLOBIO land use/management classes to the best of our abilities, mismatches occurred between some of the classes. These mismatches were caused by different levels of detail at which land use/management types are distinguished and defined in CRAFTY and GLOBIO, as well as a complete absence of certain land use and/or management classes in CRAFTY or GLOBIO. For instance, GLOBIO does not yet distinguish broadleaf from needleleaf forests and does not include solar PV on pastureland, while CRAFTY does not include perennial tree crops and the types of selective cutting systems that occur outside of Europe, where forests are selectively cut and allowed to regenerate naturally without the planting of trees or occurrence of other management activities. The latter challenges the modelling of more biodiversity-friendly management because we find that selective cutting and reduced-impact logging have the lowest effect on biodiversity, relative to natural forests (Section 2 and 3), as well as in prior studies (Chaudhary et al. 2016).

5.1.2 Integrating effects of climate change on biodiversity in CRAFTY

GLOBIO includes climate change, measured as increases in temperature, as a driver of biodiversity loss. Effects of climate change on biodiversity were determined by Nunez et al. (2019) through meta-analysis during which biodiversity data was derived from studies that assess the effects of temperature changes on biodiversity using species distribution models (see Appendix I for more information). Nunez et al. (2019) conducted mixed modelling to determine the relationship between temperature increase and biodiversity globally. The regression coefficients of this relationship were reported, and the regression could, therefore, be directly introduced in the CRAFTY model. Temperature increase data can be used as an explanatory variable to simulate biodiversity loss in response to climate change. In the case of CRAFTY, temperature data is provided by LPJ-GUESS. Hence, CRAFTY can now simulate changes in biodiversity in response to temperature change in exploratory scenarios and policy pathways using LPJ-GUESS temperature data as input.

5.2 Using CRAFTY output to model effects of atmospheric N deposition, fragmentation and road disturbance in GLOBIO

Land use output of CRAFTY will be used as input for GLOBIO to assess the effects of atmospheric N deposition, fragmentation and road disturbance on biodiversity over time once policy pathways have been developed.

5.2.1 Using CRAFTY output to model effects of atmospheric N in GLOBIO

Like for climate change, effects of atmospheric N deposition on biodiversity were determined through a meta-analysis, during which biodiversity data was derived from studies that assess the effects of atmospheric N deposition on biodiversity in natural lands (see Appendix I for more information on the GLOBIO N deposition module) (Midolo et al. 2019). Mixed modelling was conducted to determine the relationship between atmospheric N deposition increase and biodiversity globally (Midolo et al. 2019). The regression coefficients of this relationship are included in the GLOBIO model and can be used to predict changes in biodiversity in response to atmospheric N deposition in natural land (Schipper et al. 2020). Atmospheric N deposition does not include N deposition from fertilizer use, which are considered to be embedded in the land use/management effects on biodiversity (Schipper et al. 2020). Hence, atmospheric N deposition only affects biodiversity in natural lands, such as forests and natural grasslands.

GLOBIO uses land use/management as main input to simulate effects of atmospheric N on biodiversity (see Appendix I for more information on the GLOBIO N deposition module). In addition, atmospheric N deposition data is needed. We linked the GLOBIO and CRAFTY model in such a way that CRAFTY AFT output can be used as natural land input for the GLOBIO model. To test this approach, we used preliminary CRAFTY AFT outputs for the year 2020 as input for the GLOBIO atmospheric N deposition module. We considered the CRAFTY AFTs “Conserved Woodland”, “Unmanaged” and “Very extensive pastoral” to be natural land that could be affected by atmospheric N deposition (see Table K1-2). In addition, we derived atmospheric N deposition data from the IMAGE model. The spatial resolutions of IMAGE and CRAFTY do not match (5 arc-minutes, versus 1 km resolution). Therefore, we first disaggregated the N deposition data of IMAGE to 1 km resolution. We entered the disaggregated N deposition data as explanatory variable in the mixed model to predict the response of biodiversity to atmospheric N deposition in those CRAFTY AFTs considered to be natural land. Hence, GLOBIO can now simulate changes in biodiversity in response to atmospheric N deposition changes in exploratory scenarios and policy pathways, using CRAFTY AFT input to determine the area of natural land that is affected by atmospheric N deposition.

5.2.2 Using CRAFTY output to model effects of fragmentation in GLOBIO

Like for atmospheric N deposition, effects of fragmentation on biodiversity were determined through a meta-analysis of studies that assess effects of fragmentation, measured as patch size, on biodiversity in natural lands (see Appendix I for more information on the GLOBIO fragmentation module) (Schipper et al. 2020). Mixed modelling was conducted to determine the relationship between fragment size and biodiversity globally (Schipper et al. 2020). The regression coefficients of this relationship are included in the GLOBIO model and can be used to predict changes in biodiversity in response to fragmentation of natural land (Schipper et al. 2020).

GLOBIO uses land use/management as main input to simulate effects of fragmentation on biodiversity in natural land (see Appendix I for more information on the GLOBIO fragmentation module). Additionally, road data is used to delineate fragments. We linked the GLOBIO and CRAFTY model in such a way that CRAFTY AFT output can be used as natural land input for the GLOBIO model. To test this approach, we used preliminary CRAFTY AFT outputs for the year 2020 as input for the GLOBIO fragmentation module. We considered the CRAFTY AFTs “Conserved Woodland”, “Unmanaged” and “Very extensive pastoral” to be natural land that could be affected by fragmentation (see Table K1-2). In addition, we derived road data from the Global Roads Inventory Project (GRIP) dataset (Meijer et al. 2018). This dataset includes five different road types, from highways to local roads (Meijer et al. 2018). The spatial resolutions of GRIP and CRAFTY do not match (10 arc-seconds, versus 1 km resolution). Therefore, we first disaggregated the CRAFTY natural land cover layer to 10 arc-second resolution. Then, we masked the cells covered by roads by overlaying the natural land cover layer with the GRIP road layer. Hereafter, we determined the patch size of all fragments of natural land and entered the patch size data as explanatory variable in the mixed model to predict the response of biodiversity to fragmentation. Hence, GLOBIO can now simulate changes in biodiversity in response to fragmentation changes in exploratory scenarios and policy pathways, using CRAFTY AFT input to determine the area of natural land that is affected by fragmentation.

5.2.3 Using CRAFTY output to model effects of road disturbance in GLOBIO

Like for fragmentation, effects of road disturbance on biodiversity were determined through a meta-analysis of studies that assess effects of road disturbance, measured as distance to major roads and highways, on biodiversity (see Appendix I for more information on the GLOBIO road disturbance module) (Ana Benítez-López, Alkemade, and Verweij 2010). Mixed modelling was conducted to determine the relationship between the distance to roads and biodiversity globally (Ana Benítez-López, Alkemade, and Verweij 2010). The regression coefficients of this relationship are included in the GLOBIO model and can be used to predict changes in biodiversity in response to road disturbance.

GLOBIO uses GRIP road data (level 1 to 3) to determine the distance from major roads and highways (see Appendix I for more information on the GLOBIO road disturbance module). Road type 1 to 3 are considered to influence biodiversity due to noise and deaths from collision with cars in the GLOBIO 4 model (Schipper et al. 2020). Hence, local forest roads are not considered to affect biodiversity in GLOBIO 4 (Schipper et al. 2020). We used the distance to roads as explanatory variable in the mixed model to predict the response of biodiversity to road disturbance at 10 arc-resolution. Then, we aggregated the resulting biodiversity layer to match the extent and resolution of the CRAFTY AFT output (1 km resolution).

5.3 Exploring links between GLOBIO and LPJ-GUESS

We explored relationships between biodiversity and the vegetation structural indicators LPJ-GUESS (can) model(s) to explore opportunities to link GLOBIO and LPJ-GUESS to simulate the effects of forest management and restoration on biodiversity. For this purpose, we added any continuous vegetation structure data we could find in the different sources to the global databases described under Section 2.1.1 and Section 2.1.2. In total, we added data on 12 different vegetation structure indicators (Table 7). We calculated response ratios between the vegetations structure of managed or restored forests and the vegetation structure of reference sites to allow for a comparison of vegetations structure data across different studies around the world. Management and restoration affect vegetation structure in their own specific way and multiple aspects of vegetations structure are affected simultaneously, each with their own relationship to biodiversity. Therefore, it is important to determine effects of forest vegetation structure on biodiversity per management system or restoration type that we distinguished under Section 2.1 and 2.2. There were too few data on vegetation structure per forest management and restoration system in the global database to determine robust relationships between biodiversity and vegetation structural variables that could be used to link the GLOBIO and LPJ-GUESS model. Therefore, we decided not to link GLOBIO and LPJ-GUESS based on these relationships just yet. We did, however, qualitatively identify potential promising relationships between biodiversity and vegetation structure that may be interesting to explore further in the future.

Table 7. Vegetation structural indicators on which data was added to the database used to assess global effects of forest management and restoration on biodiversity (Section 2.1.1).

CCo	Canopy cover	Proportion of soil covered by tree crowns
CH	Canopy height	Height of trees from base to crown
UC	Understory cover	Proportion of soil covered by understory vegetation
UD	Understory density*	Closeness of understory vegetation
UH	Understory height	Height of understory vegetation from base to tip
FC	Foliage cover	Proportion of soil covered by vertical projection of leaves from woody vegetation
LLC	Leaf litter cover	Proportion of soil covered by dead leaves
LLD	Leaf litter depth*	Depth of dead leaves covering the soil
BA	Basal area*	Area of tree stems measured at breast height
TD	Tree density*	Closeness of trees
TR	Tree species richness	Number of tree species

DBH	Diameter at breast height	Diameter of tree stems measured at breast height

Table 8 provides an initial overview of the vegetation structure indicators and their qualitatively explored relationship to MSA per management or restoration type described under Section 2.1 and 2.2. Although there are many data gaps, several potential relationships between biodiversity and vegetation structure indicators that could be of interest to explore further were identified for different management systems.

Table 8. Overview of the vegetation structure and composition indicators for which data is available in the GLOBIO combined database (for animals only), as well as the CBS management practices to which these indicators related, the GLOBIO management and restoration types for which data was available and the number of MSA values per indicator (Nr. obs) per GLOBIO management or restoration type (see Table 1 and 2 for the management and restoration type definitions). Besides this, the general direction of the relationship between the vegetation structure and composition indicators with biodiversity is described.

Indicator	CBS management	GLOBIO management	Nr. obs	Taxa	Relationship
Canopy cover (CCo)	Related to harvesting practices – gaps in the forest theory	Cropland abandonment (CA)	10	Insects and mammals	Slightly positive linear relationship, outliers
		Agroforests (AG)	54	Amphibians, birds and insects	Negative linear relationship
		Clear cutting (CC)	2	Insects	Too few observations
		Perennial tree crops (PC)	7	Insects	Negative linear relationship
		Forest plantations (PL)	7	Insects, birds and mammals	No relationship
		Reduced-impact logging (RI)	1	Birds	Too few observations
		Active reclamation (Rcl)	18	Amphibians and reptiles	No relationship
		Active reconstruction (Rco)	9	Amphibians, birds, mammals and reptiles	Positive relationship
		Abandoned slash-and-burn cropland (SB)	2	Insects and mammals	Too few observations
		Selective cutting (SC)	97	Amphibians, insects and mammals	Negative linear relationship
		Silvopasture (SP)	2	Mammals	Too few observations
Canopy height (CH)	Related to harvesting, rotation length	Agroforests (AG)	1	Birds	Too few observations
		Pasture abandonment (PA)	3	Birds	Too few observations
		Clear cutting (CC)	1	Birds	Too few observations
		Forest plantations (PL)	5	Insects, birds and mammals	No relationship
		Active reclamation (Rcl)	18	Amphibians and reptiles	Negative relationship
		Active reconstruction (Rco)	6	Birds, insects and mammals	No relationship
		Abandoned slash-and-burn cropland (SB)	2	Mammals	Too few observations
		Selective cutting (SC)	6	Birds and mammals	No relationship

Tree density (TD)	Related to planting, harvesting, thinning	Cropland abandonment (CA)	5	Insects and mammals	Positive relationship
		Agroforests (AG)	2	Insects	Too few observations
		Clear cutting (CC)	2	Insects and birds	Too few observations
		Forest plantations (PL)	7	Birds, insects and mammals	No relationship
		Reduced-impact logging (RI)	2	Mammals	Too few observations
		Active reclamation (Rcl)	18	Amphibians and mammals	Negative linear relationship
		Active reconstruction (Rco)	4	Mammals	Too few observations
		Selective cutting (SC)	5	Insects and mammals	Too few observations
Understory height (UH)	Thinning, planting, forest age	Silvopasture (SP)	208	Birds	Positive relationship
Understory cover (UC)	Thinning, planting, forest age	Pasture abandonment (PA)	1	Amphibia	Too few observations
		Clear cutting (CC)	5	Insects and mammals	No relationship
		Forest plantations (PL)	3	Insects and mammals	Too few observations
		Reduced-impact logging (RI)	1	Birds	Too few observations
		Active reclamation (Rcl)	8	Amphibians and reptiles	Slight negative relationship
		Active reconstruction (Rco)	4	Mammals	Positive relationship
		Abandoned slash-and-burn cropland (SB)	2	Mammals	Too few observations
		Selective cutting (SC)	3	Insects and mammals	Too few observations
		Silvopasture (SP)	210	Birds and reptiles	No relationship
Understory density (UD)	Thinning, planting, forest age	Cropland abandonment (CA)	1	Insects	Too few observations
		Agroforests (AG)	2	Insects	Too few observations
		Active reconstruction (Rco)	5	Birds and mammals	Too few observations
		Silvopasture (SP)	11	Birds and reptiles	Positive relationship
		Active reconstruction (Rcl)	18	Amphibians and reptiles	Slight negative relationship
		Selective cutting (SC)	6	Insects and mammals	Positive relationship
Tree basal area (BA)	Forest age, tree planting, thinning	Afforestation of cropland (AC)	1	Insects	Too few observations
		Agroforests (AG)	28	Amphibians and insects	Linear positive relationship
		Afforestation of pasture (AP)	36	Amphibians	No relationship

		Clear cutting (CC)	5	Insects	Too few observations
		Forest plantations (PL)	4	Birds, insects and mammals	No relationship
		Silvopasture (SP)	232	Birds and mammals	
		Selective cutting (SC)	99	Amphibians, insects and mammals	Slight linear positive relationship
Tree species richness (TR)	Restoration measures, planting	Cropland abandonment (CA)	4	Mammals	Too few observations
		Pasture abandonment (PA)	22	Amphibians	Positive linear relationship
		Clear cutting (CC)	19	Insects	No relationship
		Perennial tree crops (PC)	1	Insects	Too few observations
		Forest plantations (PL)	14	Birds, insects and mammals	No relationship
		Abandoned slash-and-burn cropland (SB)	2	Insects	Too few observations
		Silvopasture (SP)	61	Birds	
		Selective cutting (SC)	2	Birds and insects	Too few observations
Leaf litter depth (LLD)	Site preparation	Cropland abandonment (CA)	6	Insects	No relationship
		Agroforests (AG)	12	Insects	Slight positive relationship
		Pasture abandonment (PA)	23	Amphibia	No relationship
		Clear cutting (CC)	1	Reptiles	Too few observations
		Perennial tree crops (PC)	7	Insects and reptiles	Slight negative relationship
		Multiculture plantations (PMu)	24	Birds, insects and reptiles	No relationship
		Active reconstruction (Rco)	1	Insects	Too few observations
		Selective cutting (SC)	6	Insects and mammals	Positive relationship
Leaf litter cover (LLC)	Site preparation	Clear cutting (CC)	1	Mammals	Too few observations
		Forest Plantations (PL)	5	Birds, insects and mammals	Too few observations
		Active reclamation (Rcl)	10	Amphibians and reptiles	Slightly negative relationship
		Silvopasture (SP)	2	Reptiles	Too few observations
Foliage cover (FC)	Thinning	Clear cutting (CC)	2	Birds and reptiles	Too few observations
		Perennial tree crops (PC)	1	Reptiles	Too few observations
		Active reconstruction (RS)	1	Insects	Too few observations
		Silvopasture (SP)	225	Birds and reptiles	Slight negative relationship
	Forest age	Clear cutting (CC)	1	Birds	Too few observations

Diameter at breast height (DBH)		Forest plantations (PL)	3	Birds, insects and mammals	Too few observations
		Active reconstruction (Rco)	1	Insects	Too few observations

Both GLOBIO and LPJ-GUESS have specific indicators on which data is available or on which model output can be produced. A match between the empirical relationships GLOBIO identifies and the model output of LPJ-GUESS needed to be facilitated to assess the potential of simulating effects of specific management or restoration practices on biodiversity using vegetations structure relationships with biodiversity. Per management or restoration type, we picked different promising potential relationships that could be explored in the future to simulate effects of management and restoration practices on biodiversity per management system or restoration type. We identified three thresholds that should be considered in the future before a vegetation structure or composition indicator can be considered to simulate impacts of management practices on biodiversity: (i) GLOBIO needs sufficient empirical data to create models that reflect relationships between biodiversity and vegetation structure indictors per management system, (ii) the vegetations structure indicators should match with modelling output of LPJ-GUESS, and (iii) the vegetation structure indictors should be indicative of relevant management and restoration practices for which biodiversity impacts should be quantified. Table 9 provides an overview of the indicators of GLOBIO and their match with LPJ-GUESS output, as well as notes on data availability and the qualitatively explored relationships. Based on all this information, we the potential relevance of each vegetation structure indicator to link GLOBIO with LPJ-GUESS in the future once sufficient data becomes available to robustly do so.

Table 9. Overview of the vegetation structure and composition indicators of GLOBIO and their match with LPJ-GUESS output as well as notes on data availability, the relationships that are qualitatively interpreted to be of potential interest in the future, and interpretation of the potential relationship. Only (slightly) positive or negative relationships between biodiversity and vegetation structure indicators are included. **Green** = relevant, **yellow** = likely relevant, **orange** = potentially relevant and **red** = irrelevant.

GLOBIO	LPJ-GUESS	Data availability	Promising relationship		Relevance and interpretation	
Canopy cover (CCo)	Proportion of the soil covered by canopy	**High**; especially for agroforests (AG), selective cutting (SC), and active reclamation (Rcl)	Agroforests (AG)	Negative linear relationship	🟩	Indicates potential trade-offs between carbon storage (high tree coverage) and biodiversity in agroforest systems. Perhaps useful to explore in this context.

Canopy height (CH)	95th percentile of tree height	Low; data mainly available for active reclamation	Active reclamation (Rcl)	Negative linear relationship	🟩	The negative relationship between MSA and canopy cover is surprising because we would expect canopy height to relate to forest age (O'Brien et al. 1995); the older the forest the higher biodiversity (Hilmers et al. 2018). Yet, early successional stages are often more biodiversity rich, after which there is a small decline and rise again in old growth forests (Hilmers et al. 2018). In active reclamation systems it likely takes time for forests to restore; hence the relationship might reflect the process from moving from early successional stages of the forest to later stages. Like for plantations it may potentially reflect tree density effects when trees are planted to closely together and compete; potentially resulting in a homogeneous vegetation structure. Needs to be better understood.
Tree density (TD)	Stem number by size class	Low; data mainly available for active reclamation	Active reclamation (Rcl)	Negative linear relationship	🟩	Potentially, trees are planted close together in active reclamation systems, potentially resulting in a homogeneous vegetation structure. Needs to be better understood.
Understory height (UH)		Medium; quite some data available for silvopasture systems but not for other systems	Silvopasture (SP)	Positive linear relationship	🟩	High understory cover may relate to gaps in the forests or light being able to reach the forest floor, which increases forest structural complexity and hence positively affects biodiversity (Fahey and Puettmann 2008). LPJ-GUESS is currently unable to model understory vegetation; yet is able to simulate gaps in the forest and determine how much light penetrates to the forest floor. Hence, potentially understory vegetation cover and/or density could be modelled in the future.
Understory cover (UC)		Medium; quite some data available for silvopasture systems but not for other systems	Silvopasture (SP)	No relationship	🟥	Based on our results, understory cover may not be a good indicator for vegetation complexity.
Understory density (UD)		Medium; some data available active reclamation	Active reclamation (Rcl)	Slight negative linear relationship	🟥	This result is challenging to interpret.

		and silvopasture systems	Silvopasture (SP)	Indication of a positive relationship but outliers	🟧	Dense understory cover may relate to gaps in the forests or light being able to reach the forest floor, which increases forest structural complexity and hence positively affects biodiversity (Fahey and Puettmann 2008). LPJ-GUESS is currently unable to model understory vegetation; yet is able to simulate gaps in the forest and determine how much light penetrates to the forest floor. Hence, potentially understory vegetation cover and/or density could be modelled in the future.
Tree basal area (BA)	Basal area by size class	**High**; especially for agroforests (AG), pasture abandonment (PA), silvopasture (SP), and selective cutting (SC)	Agroforests (AG)	Linear positive relationship	🟩	Reflects tree densities and also tree ages, where larger trees contribute more to the basal area than smaller ones (Poage and Tappeiner 2002). Hence, either basal area reflects positive relationships between agroforests and tree density or a positive effect of large trees (old trees). Interestingly this finding contrasts with the negative relationship found with canopy cover in agroforest systems, since often positive relationships between basal area and canopy cover are found due to increases in tree densities (Brūmelis et al. 2020). Perhaps the positive impacts on biodiversity found for basal area are, hence, related to the presence of large trees rather than high tree densities. Basal area also positively relates to forest age (Peña-Claros 2003), and hence positive effects on biodiversity may indirectly reflect impacts of forest age.
			Silvopasture (SP)	Linear positive relationship	🟩	Reflects tree densities and tree ages (Poage and Tappeiner 2002; Peña-Claros 2003), where larger trees contribute more to the basal area than smaller ones. Hence, either basal area reflects positive relationships between agroforests and tree density or a positive effect of large trees (old trees). Basal area also positively relates to forest age (Peña-Claros 2003), and hence positive effects on biodiversity may indirectly reflect impacts of forest age.

			Selective cutting (SC)	Positive linear relationship with outliers	🟨	Here it is likely that basal area reflects either selective cutting intensity or forest age. Higher intensity cutting may results in lower basal areas with negative implications for biodiversity, although basal area may also increase due to the growth of large numbers of young trees (Rendón-Carmona et al. 2009).
Tree species richness (TR)	Tree species	Medium; relatively abundant data available for clear cutting (CC), pasture abandonment (PA), forest plantations (PL) and silvopasture (SP)	Pasture abandonment (PA)	Positive linear relationship	🟩	Tree species richness has been found to positively influence biodiversity (Wang et al. 2019).
			Silvopasture (SP)	Positive linear relationship	🟩	Tree species richness has been found to positively influence biodiversity in plantation systems (Wang et al. 2019).
Leaf litter depth (LLD)	Leaf litter that turns into soil the next year	Medium; mainly data on pasture abandonment (PA), forest plantations (PL) and agroforests (AG)	Agroforests (AG)	Slight positive linear relationship	🟧	Leaf litter is positively related to insect richness (Silva et al. 2011) and also to tree species richness (Scherer-Lorenzen, Luis Bonilla, and Potvin 2007), which may indirectly promote other types of biodiversity than insects (Wang et al. 2019). LPJ-GUESS models leaf litter that turns into soil the next year. Potentially an improved module could be developed for leaf litter.
Leaf litter cover (LLC)	Leaf litter that turns into soil the next year	Very low; limited data	Active reclamation	Slight negative relationship	🟥	This result is challenging to interpret and potentially related to limited data.
Foliage cover (FC)		Medium; quite some data available for silvopasture systems but not for other systems	Silvopasture (SP)	Negative relationship	🟧	This result is challenging to interpret. Potentially dense foliage cover prevents light from reaching the soil. Birds prefer open canopy (Beedy 1981) (mostly bird data found for foliage cover). Not yet included in LPJ-GUESS, likely challenging to model.
Diameter at breast		Very low; limited data	-	-	🟥	No relationships were found as result of limited data.

height (DBH)					🟥	

Table 9 shows that the most promising vegetation structure and composition indicators that could be explored within the ForestPaths project are (i) tree basal area, (ii) tree species richness and (iii) canopy cover. Tree basal area may be used as a proxy for forest age in case it is difficult to extract biodiversity-forest age relationships for specific management and restoration systems. For instance, in the case of selective cutting, relationships between biodiversity and basal area could be used to distinguish selective cutting in old forests from that in younger ones (Fig. 21). For silvopasture, tree basal area (Fig. 21) may reflect increases in biodiversity with plantation age and may inherently also relate to other vegetation structure effects that affect biodiversity, such as tree densities. Increases in intactness with tree species richness for pasture abandonment and silvopasture is in line with literature that shows increases in biodiversity in response to enhanced tree diversity (Wang et al. 2019). Observed declines in intactness with canopy cover in agroforest systems indicates potential trade-offs between carbon storage (high tree coverage) and biodiversity, which have been observed in previous studies (Zomer et al. 2022). The declines in biodiversity in response to canopy cover could potentially relate to a reduction of gaps in the canopy, which can positively influence biodiversity (Ren et al. 2021).

Relevant indicators for which potentially feasible adaptations in LPJ-GUESS could be made in the future are (i) understory height. On GLOBIOs side effort could be made to gather more data to derive (i) tree species richness, (ii) leaf litter depth, and (iii) understory height relationships. All promising relationships are shown in Fig. 21. A big gap in data occurs for dead wood, which provides important habitat for specialist insect species (Brumelis et al. 2011).

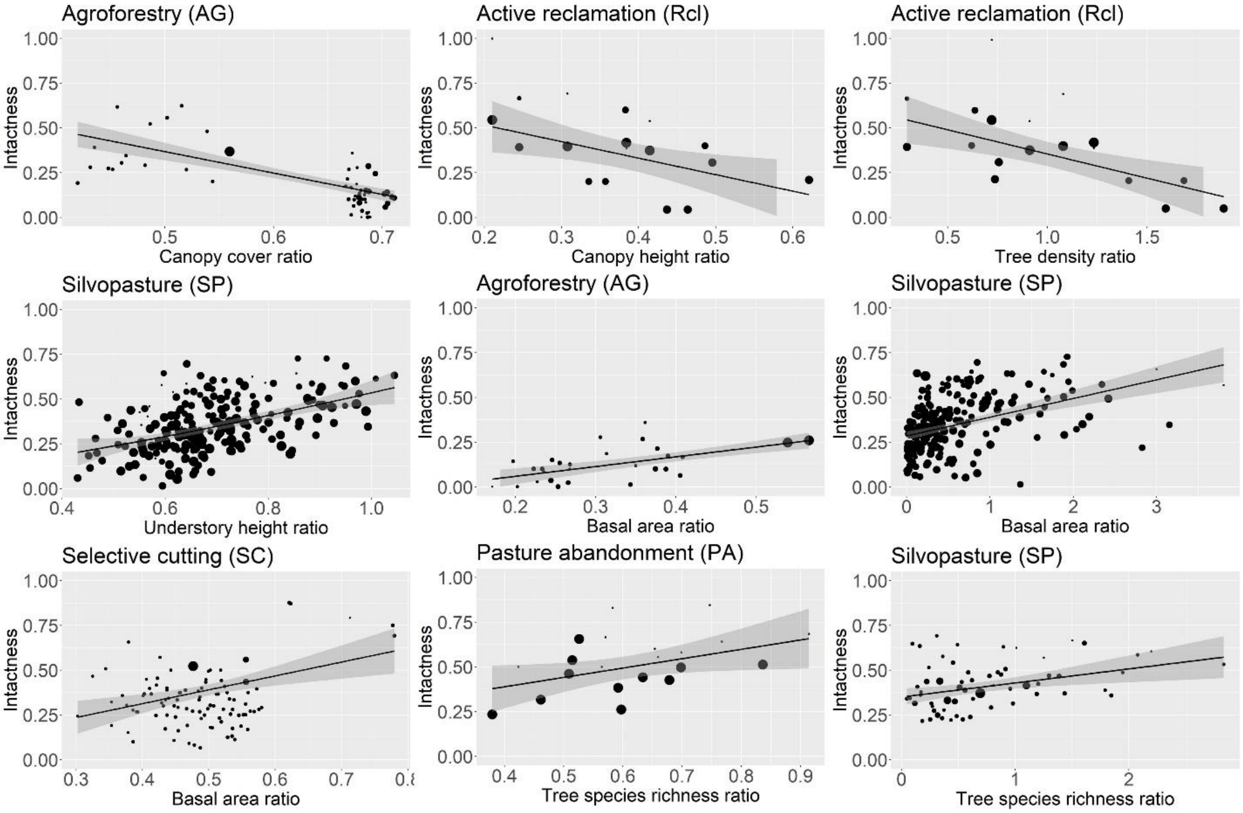


Figure 21. Potentially promising qualitatively interpreted relationships between MSA and vegetation structure and composition indicators for specific management systems. In the future, LPJ-GUESS output could be used to predict intactness for the specific management systems and

restoration types to assess impacts of specific management practices within the systems on biodiversity, such as practices related to thinning, planting and species selection.

GLOBIO could explore whether a meta-analysis on dead wood could be conducted in the future to shed light on effects of dead wood in the forest on biodiversity under different forest management regimes. Yet, due to data limitations, GLOBIO will not yet directly be linked with the LPJ-GUESS model to simulate biodiversity in response to management-related changes in vegetation structure.

References

Abernethy, K. A., L. Coad, G. Taylor, M. E. Lee, and F. Maisels. 2013. “Extent and Ecological Consequences of Hunting in Central African Rainforests in the Twenty-First Century.” *Philosophical Transactions of the Royal Society B: Biological Sciences* 368 (1625). https://doi.org/10.1098/rstb.2012.0303.

Addo-fordjour, Patrick, Isaac Sarfo, and Jennifer Oppong. 2022. “Selective and Clear-Cut Logging Have Varied Imprints on Tree Community Structure in a Moist Semi-Deciduous Forest in Ghana.” *Heliyon* 8 (April): e11393. https://doi.org/10.1016/j.heliyon.2022.e11393.

Alessia, Bono, Alberti Giorgio, Berretti Roberta, Curovic Milic, Dukic Vojislav, and Motta Renzo. 2024. “The Largest European Forest Carbon Sinks Are in the Dinaric Alps Old-Growth Forests: Comparison of Direct Measurements and Standardised Approaches.” *Carbon Balance and Management* 19 (1): 1–12. https://doi.org/10.1186/s13021-024-00262-4.

Alkemade, Rob, Mark Van Oorschot, Lera Miles, Christian Nellemann, Michel Bakkenes, and Ben Ten Brink. 2009. “GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss.” *Ecosystems* 12 (3): 374–90. https://doi.org/10.1007/s10021-009-9229-5.

Aminzadeh, B., and S. Ghorashi. 2007. “Scenic Landscape Quality and Recreational Activities in Natural Forest Parks, Iran.” *Int. J. Environ. Res. 1* 1 (1): 5–13.

Anttila, Pia, and Timo Salmi. 2006. “Characterizing Temporal and Spatial Patterns of Urban PM10 Using Six Years of Finnish Monitoring Data.” *Boreal Environment Research* 11 (6): 463–79.

Assessment, Millennium Ecosystem. 2005. “Millenium Ecosystem Assessment.” *Ecosystems and Human Well-Being*. Vol. Synthesis.

Başkent, Emin Zeki, and Jan Kašpar. 2022. “Exploring the Effects of Management Intensification on Multiple Ecosystem Services in an Ecosystem Management Context.” *Forest Ecology and Management* 518 (May): 1–15. https://doi.org/10.1016/j.foreco.2022.120299.

Bates, Douglas, Martin Mächler, Benjamin M. Bolker, and Steven C. Walker. 2015. “Fitting Linear Mixed-Effects Models Using Lme4.” *Journal of Statistical Software* 67 (1). https://doi.org/10.18637/jss.v067.i01.

Batjes, N H. 1996. “Global Assessment of Land Vulnerability to Water Erosion on a 1/2◦ by 1/2◦ Grid.” *Land Degrad Dev* 7 (4): 353–365.

Beedy, A C. 1981. “Bird Communities and Forest Structure in the Sierra Nevada of California.” *Journal of the Cooper Ornithological Society* 83 (2): 97–105.

Benítez-López, A., R. Alkemade, A. M. Schipper, D. J. Ingram, P. A. Verweij, J. A.J. Eikelboom, and M. A.J. Huijbregts. 2017. “The Impact of Hunting on Tropical Mammal and Bird Populations.” *Science* 356 (6334): 180–83. https://doi.org/10.1126/science.aaj1891.

Benítez-López, Ana, Rob Alkemade, and Pita A. Verweij. 2010. “The Impacts of Roads and Other Infrastructure on Mammal and Bird Populations: A Meta-Analysis.” *Biological Conservation* 143 (6): 1307–16. https://doi.org/10.1016/j.biocon.2010.02.009.

Benítez-López, Ana, Luca Santini, Aafke M. Schipper, Michela Busana, and Mark A.J. Huijbregts. 2019. “Intact but Empty Forests? Patterns of Huntinginduced Mammal Defaunation in the Tropics.” *PLoS Biology* 17 (5): 1–18. https://doi.org/10.1371/journal.pbio.3000247.

Betts, Matthew G., Christopher Wolf, William J. Ripple, Ben Phalan, Kimberley A. Millers, Adam Duarte, Stuart H.M. Butchart, and Taal Levi. 2017. “Global Forest Loss Disproportionately Erodes Biodiversity in Intact Landscapes.” *Nature* 547 (7664): 441–44. https://doi.org/10.1038/nature23285.

Bohada-Murillo, Mauricio, Gabriel J. Castaño-Villa, and Francisco E. Fontúrbel. 2020. “The

Effects of Forestry and Agroforestry Plantations on Bird Diversity: A Global Synthesis." *Land Degradation and Development* 31: 646–54. https://doi.org/10.1002/ldr.3478.

Borrelli, Pasquale, David A. Robinson, Larissa R. Fleischer, Emanuele Lugato, Cristiano Ballabio, Christine Alewell, Katrin Meusburger, et al. 2017. "An Assessment of the Global Impact of 21st Century Land Use Change on Soil Erosion." *Nature Communications* 8 (1): 1–13. https://doi.org/10.1038/s41467-017-02142-7.

Bouwman, A. F., T. Kram, and K. K. Goldewijk. 2006. "Integrated Modelling of Global Environmental Change: An Overview of IMAGE 2.4." *Netherlands Environmental Assessment Agency (MNP)* Bilthoven: 261–76.

Brockerhoff, E. G., H. Jactel, J. A. Parrotta, C. P. Quine, and J. Sayer. 2008. "Plantation Forests and Biodiversity: Oxymoron or Opportunity?" *Biodiversity and Conservation* 17 (5): 925–51. https://doi.org/10.1007/s10531-008-9380-x.

Brooks, Mollie, Ben Bolker, Kasper Kristensen, Martin Maechler, Arni Magnusson, Maeve McGillycuddy, Hans Skaug, et al. 2023. *Package 'GlmmTMB.'*

Brotons, L, M Mönkkönen, E Huhta, A Nikula, and A Rajasärkkä. 2003. "Effects of Landscape Structure and Forest Reserve Location on Old-Growth Forest Bird Species in Northern Finland." *Landscape Ecology* 18 (4): 377–93. https://doi.org/10.1023/A:1026148825138.

Brown, Calum, Bumsuk Seo, and Mark Rounsevell. 2019. "Societal Breakdown as an Emergent Property of Large-Scale Behavioural Models of Land Use Change." *Earth System Dynamics* 10 (4): 809–45. https://doi.org/10.5194/esd-10-809-2019.

Brumelis, G, B G Jonsson, J Kouki, T Kuuluvainen, and E Shorohova. 2011. "Forest Naturalness in Northern Europe: Perspectives on Processes, Structures and Species Diversity." *Silva Fennica* 45 (5): 807–21. https://doi.org/10.14214/sf.446.

Brūmelis, Guntis, Iluta Dauškane, Didzis Elferts, Linda Strode, Tatjana Krama, and Indrikis Krams. 2020. "Estimates of Tree Canopy Closure and Basal Area as Proxies for Tree Crown Volume at a Stand Scale." *Forests* 11 (11): 1–9. https://doi.org/10.3390/f11111180.

Brye, K. R., J. M. Norman, L. G. Bundy, and S. T. Gower. 2000. "Water-Budget Evaluation of Prairie and Maize Ecosystems." *Soil Science Society of America Journal* 64 (2): 715–24. https://doi.org/10.2136/sssaj2000.642715x.

CBD. 2022. "Post-2020 Global Biodiversity Framework."

Ceccherini, Guido, Gregory Duveiller, Giacomo Grassi, Guido Lemoine, Valerio Avitabile, Roberto Pilli, and Alessandro Cescatti. 2020. "Abrupt Increase in Harvested Forest Area over Europe after 2015." *Nature* 583 (7814): 72–77. https://doi.org/10.1038/s41586-020-2438-y.

CEIP, Centre on Emission Inventories and Projections. 2011. "Centre on Emission Inventories and Projections – Convention on Long-Range Transboundary Air Pollution."

Chaudhary, A., Z. Burivalova, L. P. Koh, and S. Hellweg. 2016. "Impact of Forest Management on Species Richness: Global Meta-Analysis and Economic Trade-Offs." *Scientific Reports* 6: 1–10. https://doi.org/10.1038/srep23954.

Chazdon, R. L. 2008. "Beyond Deforestation: Restoring Degraded Lands." *Communities* 320 (2008): 1458–60. https://doi.org/10.1126/science.1155365.

Chazdon, Robin L., and Manuel R. Guariguata. 2016. "Natural Regeneration as a Tool for Large-Scale Forest Restoration in the Tropics: Prospects and Challenges." *Biotropica* 48 (6): 716–30. https://doi.org/10.1111/btp.12381.

Chazdon, Robin L., Carlos A. Peres, Daisy Dent, Douglas Sheil, Ariel E. Lugo, David Lamb, Nigel E. Stork, and Scott E. Miller. 2009. "The Potential for Species Conservation in Tropical Secondary Forests." *Conservation Biology* 23 (6): 1406–17. https://doi.org/10.1111/j.1523-1739.2009.01338.x.

Crouzeilles, Renato, Michael Curran, Mariana S. Ferreira, David B. Lindenmayer, Carlos E.V. Grelle, and José M. Rey Benayas. 2016. "A Global Meta-Analysis on the Ecological Drivers of Forest Restoration Success." *Nature Communications* 7 (May): 1–8.

https://doi.org/10.1038/ncomms11666.

Crouzeilles, Renato, Mariana S. Ferreira, Robin L. Chazdon, David B. Lindenmayer, Jerônimo B.B. Sansevero, Lara Monteiro, Alvaro Iribarrem, Agnieszka E. Latawiec, and Bernardo B.N. Strassburg. 2017. “Ecological Restoration Success Is Higher for Natural Regeneration than for Active Restoration in Tropical Forests.” *Science Advances* 3 (11): 1–8. https://doi.org/10.1126/sciadv.1701345.

Cruz-Alonso, Verónica, Paloma Ruiz-Benito, Pedro Villar-Salvador, and José María Rey-Benayas. 2019. “Long-Term Recovery of Multifunctionality in Mediterranean Forests Depends on Restoration Strategy and Forest Type.” *Journal of Applied Ecology* 56 (3): 745–57. https://doi.org/10.1111/1365-2664.13340.

Dave, Radhika, Carole Saint-Laurent, Lara Murray, Gabriel Antunes Daldegan, Rens Brouwer, Carlos Alberto de Mattos Scaramuzza, Leander Raes, et al. 2019. “Second Bonn Challenge Progress Report: Application of the Barometer in 2018.” *Gland, Switzerland: IUCN*. https://doi.org/10.2305/iucn.ch.2019.06.en.

Deichmann, Uwe, Deborah Balk, and Greg Yetman. 2001. “Transforming Population Data for Interdisciplinary Usages: From Census to Grid.” *CIESIN, Columbia University Working Paper*, no. October: 1–19. http://sedac.ciesin.org/gpw-v2/GPWdocumentation.pdf.

Deutsch, Curtis A., Joshua J. Tewksbury, Raymond B. Huey, Kimberly S. Sheldon, Cameron K. Ghalambor, David C. Haak, and Paul R. Martin. 2008. “Impacts of Climate Warming on Terrestrial Ectotherms across Latitude.” *Proceedings of the National Academy of Sciences of the United States of America* 105 (18): 6668–72. https://doi.org/10.1073/pnas.0709472105.

Dieler, Jochen, Enno Uhl, Peter Biber, Jörg Müller, Thomas Rötzer, and Hans Pretzsch. 2017. “Effect of Forest Stand Management on Species Composition, Structural Diversity, and Productivity in the Temperate Zone of Europe.” *European Journal of Forest Research* 136 (4): 739–66. https://doi.org/10.1007/s10342-017-1056-1.

Dinerstein, Eric, David Olson, Anup Joshi, Carly Vynne, Neil D. Burgess, Eric Wikramanayake, Nathan Hahn, et al. 2017. “An Ecoregion-Based Approach to Protecting Half the Terrestrial Realm.” *BioScience* 67 (6): 534–45. https://doi.org/10.1093/biosci/bix014.

Donato, Daniel C., John L. Campbell, and Jerry F. Franklin. 2012. “Multiple Successional Pathways and Precocity in Forest Development: Can Some Forests Be Born Complex?” *Journal of Vegetation Science* 23 (3): 576–84. https://doi.org/10.1111/j.1654-1103.2011.01362.x.

Duncker, Philipp S., Susana M. Barreiro, Geerten M. Hengeveld, Torgny Lind, William L. Mason, Slawomir Ambrozy, and Heinrich Spiecker. 2012. “Classification of Forest Management Approaches: A New Conceptual Framework and Its Applicability to European Forestry.” *Ecology and Society* 17 (4): 51. https://doi.org/10.5751/ES-05262-170451.

EEA, European Environment Agency. 2011. “AirBase – the European Air Quality Database.” *Copenhagen (Denmark): EEA*.

EEA, European Environment Agency. 2019. “Ecosystem Types of Europe.” Https://Sdi.Eea.Europa.Eu/Catalogue/Srv/Api/Records/573ff9d5-6889-407f-B3fc-Cfe3f9e23941. 2019.

EEA, European Environment Agency. 2020. *State of Nature in the EU: Results from Reporting under the Nature Directives 2013-2018*. *EEA Report*.

EFI, European Forest Institute. 2007. “Long-Term European Forest Resources Assessment.” *Helsinki (Finland): EFI*. http://www.efi.int/%0Adatabases/ltfra.

EU, EU Parliament. 2024. “REGULATION OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL on Nature Restoration and Amending Regulation (EU).”

EU, European Commission. 2021. “New EU Forest Strategy for 2030.”

Europe, Forest. 2014. “Expert Group and Workshop on Valuation of Forest Ecosystem Services.” *Group of Expert (2012-2014) & Begrade Workshop (Republic of Serbia)*

September (Final report): 7–91.

EUROPE, FOREST. 2020. “State of Europe’s Forests 2020.” *State of Europe’s Forests*, 1–394.

European Environment Agency. 2014. *Noise in Europe*. *Publications Office of the European Union*.

Eurostat. 2011. “European Statistical Database – Arrivals in Tourist Accommodation Establishments – Regional Annual Data.”

Fahey, Robert T., and Klaus J. Puettmann. 2008. “Patterns in Spatial Extent of Gap Influence on Understory Plant Communities.” *Forest Ecology and Management* 255 (7): 2801–10. https://doi.org/10.1016/j.foreco.2008.01.053.

Fang, Ya Nan, Jian Tian, Aihemaiti Namaiti, Shaoqiang Zhang, Jian Zeng, and Xuemei Zhu. 2024. “Visual Aesthetic Quality Assessment of the Streetscape from the Perspective of Landscape-Perception Coupling.” *Environmental Impact Assessment Review* 106 (May): 107535. https://doi.org/10.1016/j.eiar.2024.107535.

FAO, Food and Agriculture Organization. 2020a. “Global Forest Resources Assessment 2020: Terms and Definitions.” *Forest Resources Assessment Working Paper*. Vol. 188.

FAO, Food and Agriculture Organization. 2020b. “Global Forest Resources Assessment 2020 - Key Findings.” *Global Forest Resources Assessment 2020*. Vol. Rome. https://doi.org/10.4060/ca8753en.

FAO, Food and Agriculture Organization. 2008. “Harmonized World Soil Database (Version 1.0).” Rome (Ital.

FAO, Food and Agriculture Organization. 2011. “Fishery Statistical Collections - Global Capture Production.” *Rome (Italy): FAO*.

Fedrowitz, Katja, Julia Koricheva, Susan C. Baker, David B. Lindenmayer, Brian Palik, Raul Rosenvald, William Beese, et al. 2014. “Can Retention Forestry Help Conserve Biodiversity? A Meta-Analysis.” *Journal of Applied Ecology* 51 (6): 1669–79. https://doi.org/10.1111/1365-2664.12289.

Figueroa-Alfaro, Richard Wagner, and Zhenghong Tang. 2017. “Evaluating the Aesthetic Value of Cultural Ecosystem Services by Mapping Geo-Tagged Photographs from Social Media Data on Panoramio and Flickr.” *Journal of Environmental Planning and Management* 60 (2): 266–81. https://doi.org/10.1080/09640568.2016.1151772.

Fischer, Paul H., Bert Brunekreef, and Erik Lebret. 2004. “Air Pollution Related Deaths during the 2003 Heat Wave in the Netherlands.” *Atmospheric Environment* 38 (8): 1083–85. https://doi.org/10.1016/j.atmosenv.2003.11.010.

Fitzherbert, Emily B., Matthew J. Struebig, Alexandra Morel, Finn Danielsen, Carsten A. Brühl, Paul F. Donald, and Ben Phalan. 2008. “How Will Oil Palm Expansion Affect Biodiversity?” *Trends in Ecology and Evolution* 23 (10): 538–45. https://doi.org/10.1016/j.tree.2008.06.012.

Fohrer, N., S. Haverkamp, and H. G. Frede. 2005. “Assessment of the Effects of Land Use Patterns on Hydrologic Landscape Functions: Development of Sustainable Land Use Concepts for Low Mountain Range Areas.” *Hydrological Processes* 19 (3): 659–72. https://doi.org/10.1002/hyp.5623.

Fuchs, R., M. Herold, P. H. Verburg, and J. G.P.W. Clevers. 2013. “A High-Resolution and Harmonized Model Approach for Reconstructing and Analysing Historic Land Changes in Europe.” *Biogeosciences* 10 (3): 1543–59. https://doi.org/10.5194/bg-10-1543-2013.

Gallai, Nicola, Jean Michel Salles, Josef Settele, and Bernard E. Vaissière. 2009. “Economic Valuation of the Vulnerability of World Agriculture Confronted with Pollinator Decline.” *Ecological Economics* 68 (3): 810–21. https://doi.org/10.1016/j.ecolecon.2008.06.014.

Gatti, R. C., S. Castaldi, J. A. Lindsell, D. A. Coomes, M. Marchetti, M. Maesano, A. Di Paola, F. Paparella, and R. Valentini. 2015. “The Impact of Selective Logging and Clearcutting on Forest Structure, Tree Diversity and above-Ground Biomass of African Tropical Forests.” *Ecological Research* 30 (1): 119–32. https://doi.org/10.1007/s11284-014-1217-3.

Gibbs, H. K., and J. M. Salmon. 2015. “Mapping the World’s Degraded Lands.” *Applied Geography* 57: 12–21. https://doi.org/10.1016/j.apgeog.2014.11.024.

Hall, Derek R. 1998. “Tourism Development and Sustainability Issues in Central and South-Eastern Europe.” *Tourism Management* 19 (5): 423–31. https://doi.org/10.1016/S0261-5177(98)00039-9.

Hedblom, M., H. Hedenås, M. Blicharska, S. Adler, I. Knez, G. Mikusiński, J. Svensson, S. Sandström, P. Sandström, and D. A. Wardle. 2020. “Landscape Perception: Linking Physical Monitoring Data to Perceived Landscape Properties.” *Landscape Research* 45 (2): 179–92. https://doi.org/10.1080/01426397.2019.1611751.

Herbst, H., M. Förster, and B. Kleinschmit. 2009. “Contribution of Landscape Metrics to the Assessment of Scenic Quality - the Example of the Landscape Structure Plan Havelland/Germany.” *Landscape Online* 10 (1): 1–17. https://doi.org/10.3097/LO.200910.

Hewlett, Denise, Lisa Harding, Tom Munro, Ainara Terradillos, and Keith Wilkinson. 2017. “Broadly Engaging with Tranquillity in Protected Landscapes: A Matter of Perspective Identified in GIS.” *Landscape and Urban Planning* 158: 185–201. https://doi.org/10.1016/j.landurbplan.2016.11.002.

Hilmers, T., N. Friess, C. Bässler, M. Heurich, R. Brandl, H. Pretzsch, R. Seidl, and J. Müller. 2018. “Biodiversity along Temperate Forest Succession.” *Journal of Applied Ecology* 55 (6): 2756–66. https://doi.org/10.1111/1365-2664.13238.

Holl, K. D., and T. M. Aide. 2011. “When and Where to Actively Restore Ecosystems?” *Forest Ecology and Management* 261 (10): 1558–63. https://doi.org/10.1016/j.foreco.2010.07.004.

Hootsmans, R, A F Bouwman, R Leemans, and G J J Kreileman. 2001. “Modelling Land Degradation in IMAGE 2.” *RIVM Report 481508009* 787: 33.

Hothorn, Torsten, Frank Bretz, Peter Westfall, Richard M Heiberger, Andre Schuetzenmeister, and Susan Scheibe. 2024. *Package ‘ Multcomp .’*

Hudson, Lawrence N., Tim Newbold, Sara Contu, Samantha L.L. Hill, Igor Lysenko, Adriana De Palma, Helen R.P. Phillips, et al. 2014. “The PREDICTS Database: A Global Database of How Local Terrestrial Biodiversity Responds to Human Impacts.” *Ecology and Evolution* 4 (24): 4701–35. https://doi.org/10.1002/ece3.1303.

Hudson, Lawrence N., Tim Newbold, Sara Contu, Samantha L.L. Hill, Igor Lysenko, Adriana De Palma, Helen R.P. Phillips, et al. 2017. “The Database of the PREDICTS (Projecting Responses of Ecological Diversity In Changing Terrestrial Systems) Project.” *Ecology and Evolution* 7 (1): 145–88. https://doi.org/10.1002/ece3.2579.

IPBES. 2019. “Global Assessment Report on the Interngovernmental Science-Policy Platform on Biodiversity and Ecosystem Services,” 1–1029.

IUCN. 2024. “The IUCN Red List of Threatened Species.” *Version 2024-1* https://ww (Accessed on 19-07-2024).

Iversen, Kidal Endre, Kristine Grimsrud, Henrik Lindhjem, and Jette Bredahl Jacobsen. 2019. “Trade-Offs between Carbon Sequestration, Landscape Aesthetics and Biodiversity in a Cost-Benefit Analysis of Land Use Options in Norway.”

Jakovac, Catarina C., André B. Junqueira, Renato Crouzeilles, Marielos Peña-Claros, Rita C.G. Mesquita, and Frans Bongers. 2021. “The Role of Land-Use History in Driving Successional Pathways and Its Implications for the Restoration of Tropical Forests.” *Biological Reviews* 96 (4): 1114–34. https://doi.org/10.1111/brv.12694.

Johnston, C A, N E Detenbeck, and G Niemi. 1990. “The Cumulative Effect of Wetlands on Stream Water Quality and Quantity. A Landscape Approach.” *Biogeochemistry* 10: 105–41.

Jones, Lizzie P., Samuel T. Turvey, Dario Massimino, and Sarah K. Papworth. 2020. “Investigating the Implications of Shifting Baseline Syndrome on Conservation.” *People and Nature* 2 (4): 1131–44. https://doi.org/10.1002/pan3.10140.

Kienast, F, J Bolliger, M Potschin, R S De Groot, P H Verburg, I Heller, D Wascher, and R Haines-Young. 2009. “Assessing Landscape Functions with Broad-Scale Environmental

Data: Insights Gained from a Prototype Development for Europe.” *Environmental Management* 44 (6): 1099–1120. https://doi.org/10.1007/s00267-009-9384-7.

Klein, Alexandra Maria, Bernard E. Vaissière, James H. Cane, Ingolf Steffan-Dewenter, Saul A. Cunningham, Claire Kremen, and Teja Tscharntke. 2007. “Importance of Pollinators in Changing Landscapes for World Crops.” *Proceedings of the Royal Society B: Biological Sciences* 274 (1608): 303–13. https://doi.org/10.1098/rspb.2006.3721.

Kok, Kasper, Simona Pedde, Marc Gramberger, Paula A. Harrison, and Ian P. Holman. 2019. “New European Socio-Economic Scenarios for Climate Change Research: Operationalising Concepts to Extend the Shared Socio-Economic Pathways.” *Regional Environmental Change* 19 (3): 643–54. https://doi.org/10.1007/s10113-018-1400-0.

Kuipers, K.J.J., S. Sim, J.P. Hilbers, S.K. van den Berg, M.M.J. de Jonge, K. Trendafilova, M.A.J. Huijbregts, and A.M. Schipper. 2023. “Land Use Diversification May Mitigate On-Site Land Use Impacts on Mammal Populations and Assemblages.”

Lamb, David, Peter D. Erskine, and John A. Parrotta. 2005. “Restoration of Degraded Tropical Forest Landscapes.” *Science* 310 (5754): 1628–32. https://doi.org/10.1126/science.1111773.

Laurance, W. F., C. D. Useche, J. Rendeiro, Margareta Kalka, Corey J.A. Bradshaw, Sean P. Sloan, Susan G. Laurance, et al. 2012. “Averting Biodiversity Collapse in Tropical Forest Protected Areas.” *Nature* 489 (7415): 290–93. https://doi.org/10.1038/nature11318.

Leclère, David, Michael Obersteiner, Mike Barrett, Stuart H.M. Butchart, Abhishek Chaudhary, Adriana De Palma, Fabrice A.J. DeClerck, et al. 2020. “Bending the Curve of Terrestrial Biodiversity Needs an Integrated Strategy.” *Nature* 585 (7826): 551–56. https://doi.org/10.1038/s41586-020-2705-y.

Lee, Heera, Bumsuk Seo, Thomas Koellner, and Sven Lautenbach. 2019. “Mapping Cultural Ecosystem Services 2.0 – Potential and Shortcomings from Unlabeled Crowd Sourced Images.” *Ecological Indicators* 96 (December 2017): 505–15. https://doi.org/10.1016/j.ecolind.2018.08.035.

López-Bedoya, Pablo A., Erika Alejandra Cardona-Galvis, José Nicolás Urbina-Cardona, Felicity A. Edwards, and David P. Edwards. 2022. “Impacts of Pastures and Forestry Plantations on Herpetofauna: A Global Meta-Analysis.” *Journal of Applied Ecology* 59 (12): 3038–48. https://doi.org/10.1111/1365-2664.14299.

MA. 2005. “Millennium Ecosystem Assessment.” *Ecosystems and Human Well-Being: Synthesis* Washington. https://doi.org/10.4337/.M.48.

Matricardi, E. A. T., D. L. Skole, O. B. Costa, M. A. Pedlowski, J. H. Samek, and E. P. Miguel. 2020. “Long-Term Forest Degradation Surpasses Deforestation in the Brazilian Amazon.” *Science* 369 (6509): 1378–82. https://doi.org/10.1126/SCIENCE.ABB3021.

Meijer, Johan R., Mark A.J. Huijbregts, Kees C.G.J. Schotten, and Aafke M. Schipper. 2018. “Global Patterns of Current and Future Road Infrastructure.” *Environmental Research Letters* 13 (6): 064006. https://doi.org/10.1088/1748-9326/aabd42.

Meli, Paula, Karen D. Holl, José María Rey Benayas, Holly P. Jones, Peter C. Jones, Daniel Montoya, and David Moreno Mateos. 2017. “A Global Review of Past Land Use, Climate, and Active vs. Passive Restoration Effects on Forest Recovery.” *PLoS ONE* 12 (2): 1–17. https://doi.org/10.1371/journal.pone.0171368.

Midolo, Gabriele, Rob Alkemade, Aafke M. Schipper, Ana Benítez-López, Michael P. Perring, and Wim De Vries. 2019. “Impacts of Nitrogen Addition on Plant Species Richness and Abundance: A Global Meta-Analysis.” *Global Ecology and Biogeography* 28 (3): 398–413. https://doi.org/10.1111/geb.12856.

Mukul, Sharif A., and John Herbohn. 2016. “The Impacts of Shifting Cultivation on Secondary Forests Dynamics in Tropics: A Synthesis of the Key Findings and Spatio Temporal Distribution of Research.” *Environmental Science and Policy* 55: 167–77. https://doi.org/10.1016/j.envsci.2015.10.005.

Murray-Rust, D., C. Brown, J. van Vliet, S. J. Alam, D. T. Robinson, P. H. Verburg, and M. Rounsevell. 2014. “Combining Agent Functional Types, Capitals and Services to Model Land Use Dynamics.” *Environmental Modelling and Software* 59: 187–201. https://doi.org/10.1016/j.envsoft.2014.05.019.

Muys, B., P. Angelstam, J. Bauhus, L. Bouriaud, H. Jactel, H. Kraigher, J. Müller, et al. 2022. “Forest Biodiversity in Europe. From Science to Policy 13.” *European Forest Institute*. https://doi.org/https://doi.org/10.36333/fs13.

Nelson, A. 2008. “Estimated Travel Time to the Nearest City of 50,000 or More People in Year 2000.”

Nowaczek, A M K, and D A Fennel. 2002. “Ecotourism in Post-Communist Poland: An Examination of Tourists, Sustainability and Institutions.” *Tourism Geographies* 4 (4): 372–95. https://doi.org/10.1080/14616680210158137.

Nunes, Leonel J.R., Catarina I.R. Meireles, Carlos J.Pinto Gomes, and Nuno M.C.Almeida Ribeiro. 2020. “Forest Contribution to Climate Change Mitigation: Management Oriented to Carbon Capture and Storage.” *Climate* 8 (2). https://doi.org/10.3390/cli8020021.

Nunez, Sarahi, Eric Arets, Rob Alkemade, Caspar Verwer, and Rik Leemans. 2019. “Assessing the Impacts of Climate Change on Biodiversity: Is below 2 °C Enough?” *Climatic Change* 154 (3–4): 351–65. https://doi.org/10.1007/s10584-019-02420-x.

O’Brien, Lyla, Andreas Schuck, Cecilia Fraccaroli, Elisabeth Pötzelsberger, Georg Winkel, and Marcus Lindner. 2021. “Protecting Old-Growth Forests in Europe - a Review of Scientific Evidence to Inform Policy Implementation,” no. June: 104.

O’Brien, S. T., S. P. Hubbell, P. Spiro, R. Condit, and R. B. Foster. 1995. “Diameter, Height, Crown, and Age Relationships in Eight Neotropical Tree Species.” *Ecology* 76 (6): 1926–39. https://doi.org/10.2307/1940724.

Oheimb, Goddert Von, and Werner Härdtle. 2009. “Selection Harvest in Temperate Deciduous Forests: Impact on Herb Layer Richness and Composition.” *Biodiversity and Conservation* 18 (2): 271–87. https://doi.org/10.1007/s10531-008-9475-4.

Onaindia, Miren, Beatriz Fernández de Manuel, Iosu Madariaga, and Gloria Rodríguez-Loinaz. 2013. “Co-Benefits and Trade-Offs between Biodiversity, Carbon Storage and Water Flow Regulation.” *Forest Ecology and Management* 289: 1–9. https://doi.org/10.1016/j.foreco.2012.10.010.

Oosterbaan, A, A Van de Berg, T J A Gies, and A J Griffioen. 2009. “Een Nieuwe Landschapsdienst: Landschapselementen Voor Een Goede Luchtkwaliteit. Alterra-Rapport 1949,” 29p.

Pace, T G. 2003. “A Conceptual Model to Adjust Fugitive Dust Emis- Sions to Account for near Source Dust Removal in Grid Model Applications.” *Joint Meeting of WRAP Emissions & Fugitive Dust Forums* Las Vegas,.

Pacheco, Fernando António Leal, Luís Filipe Sanches Fernandes, Renato Farias Valle Junior, Carlos Alberto Valera, and Teresa Cristina Tarlé Pissarra. 2018. “Land Degradation: Multiple Environmental Consequences and Routes to Neutrality.” *Current Opinion in Environmental Science and Health* 5: 79–86. https://doi.org/10.1016/j.coesh.2018.07.002.

Paillet, Y., L. Bergès, J. HjÄltén, P. Ódor, C. Avon, Markus Bernhardt-Römermann, Rienk Jan Bijlsma, et al. 2010. “Biodiversity Differences between Managed and Unmanaged Forests: Meta-Analysis of Species Richness in Europe.” *Conservation Biology* 24 (1): 101–12. https://doi.org/10.1111/j.1523-1739.2009.01399.x.

Paillet, Y, L Bergès, J Hjältén, P Ódor, C Avon, M Bernhardt-Römermann, R.-J. Bijlsma, et al. 2010. “Compromises in Data Selection in a Meta-Analysis of Biodiversity in Managed and Unmanaged Forests: Response to Halme et Al.” *Conservation Biology* 24 (4): 1157–60. https://doi.org/10.1111/j.1523-1739.2010.01543.x.

Palmero-Iniesta, Marina, Joan Pino, Lluís Pesquer, and Josep Maria Espelta. 2021. “Recent Forest Area Increase in Europe: Expanding and Regenerating Forests Differ in Their

Regional Patterns, Drivers and Productivity Trends.” *European Journal of Forest Research* 140 (4): 793–805. https://doi.org/10.1007/s10342-021-01366-z.

Paracchini, Maria Luisa, Grazia Zulian, Leena Kopperoinen, Joachim Maes, Jan Philipp Schägner, Mette Termansen, Marianne Zandersen, Marta Perez-Soba, Paul A. Scholefield, and Giovanni Bidoglio. 2014. “Mapping Cultural Ecosystem Services: A Framework to Assess the Potential for Outdoor Recreation across the EU.” *Ecological Indicators* 45 (2014): 371–85. https://doi.org/10.1016/j.ecolind.2014.04.018.

Peña-Claros, Marielos. 2003. “Changes in Forest Structure and Species Composition during Secondary Forest Succession in the Bolivian Amazon.” *Biotropica* 35 (4): 450–61. https://doi.org/10.1111/j.1744-7429.2003.tb00602.x.

Poage, Nathan J., and John C. Tappeiner. 2002. “Long-Term Patterns of Diameter and Basal Area Growth of Old-Growth Douglas-Fir Trees in Western Oregon.” *Canadian Journal of Forest Research* 32 (7): 1232–43. https://doi.org/10.1139/x02-045.

Post, Eric, Mads C. Forchhammer, M. Syndonia Bret-Harte, Terry V. Callaghan, Torben R. Christensen, Bo Elberling, Anthony D. Fox, et al. 2009. “Ecological Dynamics across the Arctic Associated with Recent Climate Change.” *Science* 325 (5946): 1355–58. https://doi.org/10.1126/science.1173113.

Puettmann, K J, K D Coates, and C C Messier. 2012. “A Critique of Silviculture: Managing for Complexity.” *Island Press*.

QGIS, Development Team. 2023. “QGIS Geographic Information System. Open Source Geospatial Foundation Project.” http://qgis.osgeo.org.

R, Core Team. 2023. “R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing” Vienna. https://www.r-project.org/.

Ramchunder, S. J., L. E. Brown, and J. Holden. 2012. “Catchment-Scale Peatland Restoration Benefits Stream Ecosystem Biodiversity.” *Journal of Applied Ecology* 49 (1): 182–91. https://doi.org/10.1111/j.1365-2664.2011.02075.x.

Redford, Kent H. 2009. “The Empty of Neotropical Forest Where the Vegetation Still Appears Intact.” *SciencesNew York* 42 (6): 412–22.

Ren, Siyuan, Arshad Ali, Heming Liu, Zuoqiang Yuan, Qingsong Yang, Guochun Shen, Shuangshuang Zhou, and Xihua Wang. 2021. “Response of Community Diversity and Productivity to Canopy Gap Disturbance in Subtropical Forests.” *Forest Ecology and Management* 502 (April): 119740. https://doi.org/10.1016/j.foreco.2021.119740.

Rendón-Carmona, Humberto, Angelina Martínez-Yrízar, Patricia Balvanera, and Diego Pérez-Salicrup. 2009. “Selective Cutting of Woody Species in a Mexican Tropical Dry Forest: Incompatibility between Use and Conservation.” *Forest Ecology and Management* 257 (2): 567–79. https://doi.org/10.1016/j.foreco.2008.09.031.

Ripple, William J., Katharine Abernethy, Matthew G. Betts, Guillaume Chapron, Rodolfo Dirzo, Mauro Galetti, Taal Levi, et al. 2016. “Bushmeat Hunting and Extinction Risk to the World’s Mammals.” *Royal Society Open Science* 3 (10). https://doi.org/10.1098/rsos.160498.

Robinson, W. D., and S. K. Robinson. 1999. “Effects of Selective Logging on Forest Bird Populations in a Fragmented Landscape.” *Conservation Biology* 13 (1): 58–66. https://doi.org/https://doi.org/10.1046/j.1523-1739.1999.97226.x.

Roellig, M, L M E Sutcliffe, M Sammul, H von Wehrden, J Newig, and J Fischer. 2016. “Reviving Wood-Pastures for Biodiversity and People: A Case Study from Western Estonia.” *Ambio* 45 (2): 185–95. https://doi.org/10.1007/s13280-015-0719-8.

Roman, M. De, and E. Boa. 2004. “Collection, Marketing and Cultivation of Edible Fungi in Spain.” *MICOLOGIA APLICADA INTERNATIONAL* 16 (2): 25–33.

Russell, Andrew. 2007. “Anthropology and Ecotourism in European Wetlands: Bubbles, Babies and Bathwater.” *Tourist Studies* 7 (2): 225–44. https://doi.org/10.1177/1468797607083504.

Sabatini, F. M., S. Burrascano, W. S. Keeton, C. Levers, M. Lindner, F. Pötzschner, Pieter Johannes Verkerk, et al. 2018. “Where Are Europe’s Last Primary Forests?” *Diversity and*

Distributions 24 (10): 1426–39. https://doi.org/10.1111/ddi.12778.

Sabatini, F. M., W. S. Keeton, M. Lindner, M. Svoboda, P. J. Verkerk, J. Bauhus, Helge Bruelheide, et al. 2020. “Protection Gaps and Restoration Opportunities for Primary Forests in Europe.” *Diversity and Distributions* 26 (12): 1646–62. https://doi.org/10.1111/ddi.13158.

Santini, Luca, Jonathan Belmaker, Mark J. Costello, Henrique M. Pereira, Axel G. Rossberg, Aafke M. Schipper, Silvia Ceaușu, et al. 2017. “Assessing the Suitability of Diversity Metrics to Detect Biodiversity Change.” *Biological Conservation* 213: 341–50. https://doi.org/10.1016/j.biocon.2016.08.024.

Saxena, Ankita, Calum Brown, Almut Arneth, and Mark Rounsevell. 2023. “Modelling the Global Photovoltaic Potential on Land and Its Sensitivity to Climate Change.” *Environmental Research Letters* 18 (10): 104017. https://doi.org/10.1088/1748-9326/acf86f.

Scherer-Lorenzen, Michael, José Luis Bonilla, and Catherine Potvin. 2007. “Tree Species Richness Affects Litter Production and Decomposition Rates in a Tropical Biodiversity Experiment.” *Source: Oikos* 116 (12): 2108–24.

Schipper, Aafke M., Jelle P. Hilbers, Johan R. Meijer, Laura H. Antão, Ana Benítez-López, Melinda M.J. de Jonge, Luuk H. Leemans, et al. 2020. “Projecting Terrestrial Biodiversity Intactness with GLOBIO 4.” *Global Change Biology* 26 (2): 760–71. https://doi.org/10.1111/gcb.14848.

Schroth, Götz, Celia A Harvey, and Grégoire Vincent. 2004. “Complex Agroforests: Their Structure, Diversity and Potential Role in Landscape Conservation.” *Agroforestry and Biodiversity Conservation in Tropical Landscapes* 10: 537.

Schulp, Catharina J.E., Rob Alkemade, Kees Klein Goldewijk, and Katalin Petz. 2012. “Mapping Ecosystem Functions and Services in Eastern Europe Using Global-Scale Data Sets.” *International Journal of Biodiversity Science, Ecosystem Services and Management* 8 (1–2): 156–68. https://doi.org/10.1080/21513732.2011.645880.

Schüpbach, B., X. Junge, P. Lindemann-Matthies, and T. Walter. 2016. “Seasonality, Diversity and Aesthetic Valuation of Landscape Plots: An Integrative Approach to Assess Landscape Quality on Different Scales.” *Land Use Policy* 53: 27–35. https://doi.org/10.1016/j.landusepol.2015.01.032.

Silva, P. S.D., A. G.D. Bieber, M. M. Corrêa, and I. R. Leal. 2011. “Do Leaf-Litter Attributes Affect the Richness of Leaf-Litter Ants?” *Neotropical Entomology* 40 (5): 542–47.

Smithson, Michael, and Jay Verkuilen. 2006. “A Better Lemon Squeezer? Maximum-Likelihood Regression with Beta-Distributed Dependent Variables.” *Psychological Methods* 11 (1): 54–71. https://doi.org/10.1037/1082-989X.11.1.54.

Sorensen, Thorvald. 1948. “A Method of Establishing Groups of Equal Amplitude in Plant Sociology Based on Similarity of Species Content and Its Application to Analyses of the Vegetation on Danish Commons.” *Biologiske Skrifter* 5: 1–34.

Stanturf, J A, and P Madsen. 2002. “Restoration Concepts for Temperate and Boreal Forests of North America and Western Europe.” *Plant Biosystems* 136 (2): 143–58. https://doi.org/10.1080/11263500212331351049.

Stanturf, John A., Brian J. Palik, and R. Kasten Dumroese. 2014. “Contemporary Forest Restoration: A Review Emphasizing Function.” *Forest Ecology and Management* 331: 292–323. https://doi.org/10.1016/j.foreco.2014.07.029.

Staples, Timothy L., Margaret M. Mayfield, Jacqueline R. England, and John M. Dwyer. 2020. “Comparing the Recovery of Richness, Structure, and Biomass in Naturally Regrowing and Planted Reforestation.” *Restoration Ecology* 28 (2): 347–57. https://doi.org/10.1111/rec.13077.

Steffan-Dewenter, Ingolf, and Teja Tscharntke. 1999. “Effects of Habitat Isolation on Pollinator Communities and Seed Set.” *Oecologia* 121 (3): 432–40. https://doi.org/10.1007/s004420050949.

Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. B. Lindenmayer, and F. J. Swanson. 2011. “The Forgotten Stage of Forest Succession: Early-Successional Ecosystems on Forest Sites.” *Frontiers in Ecology and the Environment* 9 (2): 117–25. https://doi.org/10.1890/090157.

Taki, H., I. Okochi, K. Okabe, T. Inoue, H. Goto, T. Matsumura, and S. Makino. 2013. “Succession Influences Wild Bees in a Temperate Forest Landscape: The Value of Early Successional Stages in Naturally Regenerated and Planted Forests.” *PLoS ONE* 8 (2). https://doi.org/10.1371/journal.pone.0056678.

Tieskens, Koen F., Catharina J.E. Schulp, Christian Levers, Juraj Lieskovský, Tobias Kuemmerle, Tobias Plieninger, and Peter H. Verburg. 2017. “Characterizing European Cultural Landscapes: Accounting for Structure, Management Intensity and Value of Agricultural and Forest Landscapes.” *Land Use Policy* 62: 29–39. https://doi.org/10.1016/j.landusepol.2016.12.001.

Tisma, A., P. Giesen, A. Marques, E. van der Zanden, S. Okayasu, and R. Alkemade. n.d. “A Global Model of Landscape Aesthetic Quality.” *(Under Review)*, 1–23.

Torralba, Mario, Nora Fagerholm, Paul J. Burgess, Gerardo Moreno, and Tobias Plieninger. 2016. “Do European Agroforestry Systems Enhance Biodiversity and Ecosystem Services? A Meta-Analysis.” *Agriculture, Ecosystems and Environment* 230: 150–61. https://doi.org/10.1016/j.agee.2016.06.002.

Tsachalidis, Efstathios Pan, and Eleftherios Hadjisterkotis. 2008. “Wild Boar Hunting and Socioeconomic Trends in Northern Greece, 1993-2002.” *European Journal of Wildlife Research* 54 (4): 643–49. https://doi.org/10.1007/s10344-008-0190-y.

Tudge, Sophie Jane, Zoe M. Harris, Richard J. Murphy, Andy Purvis, and Adriana De Palma. 2023. “Global Trends in Biodiversity with Tree Plantation Age.” *Global Ecology and Conservation* 48 (August): e02751. https://doi.org/10.1016/j.gecco.2023.e02751.

UN, United Nations Framework Convention on Climate Change. 1997. “United Nations Framework Convention on Climate Change.”

UNstats, United Nations Statistics Division. 2011. “Millennium Development Goals Indicators.” http://mdgs.un.org/unsd/mdg/SeriesDetail.%0Aaspx?srid=749&crid=.

Vačkář, David, Ben Ten Brink, Jonathan Loh, Jonathan E.M. Baillie, and Belinda Reyers. 2012. “Review of Multispecies Indices for Monitoring Human Impacts on Biodiversity.” *Ecological Indicators* 17: 58–67. https://doi.org/10.1016/j.ecolind.2011.04.024.

Vallecillo, Sara, Georgia Kakoulaki, Alessandra La Notte, Luc Feyen, Francesco Dottori, and Joachim Maes. 2020. “Accounting for Changes in Flood Control Delivered by Ecosystems at the EU Level.” *Ecosystem Services* 44 (April): 101142. https://doi.org/10.1016/j.ecoser.2020.101142.

Walz, U., and C. Stein. 2018. “Indicator for a Monitoring of Germany’s Landscape Attractiveness.” *Ecological Indicators* 94 (July 2017): 64–73. https://doi.org/10.1016/j.ecolind.2017.06.052.

Wang, Xiaoyang, Fangyuan Hua, Lin Wang, David S. Wilcove, and Douglas W. Yu. 2019. “The Biodiversity Benefit of Native Forests and Mixed-Species Plantations over Monoculture Plantations.” *Diversity and Distributions* 25 (11): 1721–35. https://doi.org/10.1111/ddi.12972.

Watt, A. D., R. H.W. Bradshaw, J. Young, D. Alard, T. Bolger, D. Chamberlain, F. Fernández-González, et al. 2007. “Trends in Biodiversity in Europe and the Impact of Land-Use Change.” *Issues in Environmental Science and Technology* 25 (25): 135–60. https://doi.org/10.1039/9781847557650-00135.

Winkler, Karina, Richard Fuchs, Mark Rounsevell, and Martin Herold. 2021. “Global Land Use Changes Are Four Times Greater than Previously Estimated.” *Nature Communications* 12 (1): 1–10. https://doi.org/10.1038/s41467-021-22702-2.

Xu, H., Y. Li, S. Liu, R. Zang, F. He, and J. R. Spence. 2015. “Partial Recovery of a Tropical

Rain Forest a Half-Century after Clear-Cut and Selective Logging.” *Journal of Applied Ecology* 52 (4): 1044–52. https://doi.org/10.1111/1365-2664.12448.

Zhang, Hongjuan, Zhicheng Zhang, Kang Liu, Chunbo Huang, and Guanpeng Dong. 2023. “Integrating Land Use Management with Trade-Offs between Ecosystem Services: A Framework and Application.” *Ecological Indicators* 149 (March): 110193. https://doi.org/10.1016/j.ecolind.2023.110193.

Zomer, Robert J., Deborah A. Bossio, Antonio Trabucco, Meine van Noordwijk, and Jianchu Xu. 2022. “Global Carbon Sequestration Potential of Agroforestry and Increased Tree Cover on Agricultural Land.” *Circular Agricultural Systems* 2 (1): 1–10. https://doi.org/10.48130/cas-2022-0003.

Appendix A

Table A1. Biodiversity estimates and confidence intervals (CI) for intactness (IN), relative richness (SR), similarity (SI) and total abundance (TA) (computed by back-transforming the estimates) for animal (al taxa at the same time), plant, brid, herpetofauna, insect and mammal species. A significant p-value (<0.05) indicates that there is a significant difference between the impacts of different forest management systems on biodiversity. Coloring of the cells reflect the coloring used in the main text.

	Management systems	Nr. Studies	IN - estimate	IN - lower CI	IN - upper CI	p-value	SR - estimate	SR - lower CI	SR - upper CI	p-value	SI - estimate	SI - lower CI	SI - upper CI	p-value	TA - estimate	TA - lower CI	TA - upper CI	p-value
Animals	Agroforestry (AG)	26	0.53	0.38	0.68	1.15E-05	1.07	0.87	1.31	2.09E-10	0.89	0.80	0.95	< 2.2e-16	0.87	0.65	1.17	0.01
	Perennial tree crops (PC)	19	0.20	0.12	0.32		0.54	0.45	0.66		0.49	0.31	0.68		0.85	0.62	1.17	
	Silvopasture (SP)	8	0.50	0.27	0.73		0.92	0.68	1.25		0.89	0.71	0.97		0.98	0.60	1.60	
	Clear cutting (CC)	26	0.66	0.53	0.78		0.93	0.77	1.12		0.89	0.80	0.94		1.04	0.78	1.40	
	Forest plantations (PL)	51	0.42	0.32	0.52		0.75	0.65	0.85		0.81	0.71	0.87		0.65	0.53	0.81	
	Reduced-impact logging (RI)	4	0.77	0.44	0.93		0.98	0.61	1.59		0.97	0.85	1.00		1.08	0.52	2.24	
	Selective cutting (SC)	39	0.64	0.53	0.74		1.15	0.99	1.33		0.94	0.90	0.97		0.98	0.77	1.23	
Plants	Agroforestry (AG)	4	0.41	0.22	0.63	1.03E-04	0.70	0.21	2.34	0.002	0.68	0.41	0.87	0.0002	1.18	0.59	2.37	0.54
	Perennial tree crops (PC)	6	0.16	0.08	0.30		0.12	0.04	0.34		0.21	0.09	0.42		0.65	0.35	1.22	
	Clear cutting (CC)	7	0.52	0.34	0.69		0.76	0.28	2.04		0.69	0.47	0.85		1.08	0.61	1.90	
	Forest plantations (PL)	7	0.28	0.15	0.45		0.64	0.24	1.72		0.60	0.37	0.79		0.81	0.46	1.42	
	Selective cutting (SC)	6	0.66	0.46	0.82		0.96	0.31	3.00		0.87	0.70	0.95		1.38	0.74	2.58	
Birds	Agroforestry (AG)	4	0.64	0.48	0.78	4.88E-05	1.14	0.83	1.57	1.31E-05	0.90	0.80	0.95	< 2.2e-16	0.74	0.45	1.22	0.19
	Perennial tree crops (PC)	4	0.18	0.10	0.30		0.43	0.32	0.58		0.43	0.26	0.62		0.55	0.34	0.88	
	Silvopasture (SP)	4	0.45	0.30	0.62		0.88	0.66	1.19		0.81	0.67	0.90		1.08	0.68	1.72	
	Clear cutting (CC)	5	0.66	0.47	0.81		1.00	0.71	1.41		0.89	0.77	0.95		1.22	0.72	2.08	
	Forest plantations (PL)	8	0.51	0.39	0.62		1.00	0.80	1.26		0.88	0.81	0.93		0.86	0.60	1.23	
	Reduced-impact logging (RI)	3	0.64	0.41	0.82		0.98	0.63	1.52		0.88	0.71	0.96		1.17	0.59	2.32	
	Selective cutting (SC)	10	0.64	0.51	0.75		1.07	0.84	1.37		0.89	0.82	0.94		1.05	0.72	1.53	
Herpetofauna	Agroforestry (AG)	7	0.43	0.33	0.53	1.08E-02	0.86	0.42	1.78	0.33	0.82	0.57	0.94	3.54E-07	0.64	0.40	1.02	0.39
	Perennial tree crops (PC)	4	0.30	0.19	0.43		0.51	0.27	0.99		0.56	0.30	0.79		0.70	0.30	1.66	
	Silvopasture (SP)	2	0.54	0.35	0.73		0.89	0.36	2.23		0.91	0.67	0.98		0.65	0.24	1.76	
	Clear cutting (CC)	2	0.25	0.12	0.43		0.45	0.18	1.13		0.40	0.12	0.75		0.60	0.19	1.92	
	Forest plantations (PL)	3	0.43	0.28	0.60		1.05	0.49	2.27		0.81	0.55	0.94		1.15	0.45	2.93	
	Selective cutting (SC)	6	0.52	0.41	0.64		1.05	0.62	1.79		0.86	0.71	0.94		0.78	0.40	1.51	
Insects	Agroforestry (AG)	7	0.35	0.20	0.54	4.18E-09	1.09	0.77	1.55	1.42E-05	0.76	0.54	0.90	1.23E-09	1.14	0.60	2.16	0.14
	Perennial tree crops (PC)	8	0.15	0.08	0.25		0.59	0.43	0.80		0.45	0.25	0.66		0.80	0.45	1.44	
	Clear cutting (CC)	12	0.53	0.38	0.68		1.01	0.78	1.32		0.84	0.70	0.92		1.02	0.65	1.60	
	Forest plantations (PL)	21	0.24	0.16	0.34		0.61	0.48	0.77		0.57	0.41	0.72		0.58	0.40	0.84	
	Selective cutting (SC)	11	0.58	0.43	0.72		1.19	0.92	1.54		0.89	0.79	0.95		0.88	0.57	1.35	
Mammals	Agroforestry (AG)	8	0.66	0.45	0.83	1.06E-03	1.07	0.70	1.62	0.06	0.87	0.76	0.93	< 2.2e-16	1.00	0.52	1.91	0.06
	Perennial tree crops (PC)	3	0.36	0.15	0.64		0.63	0.37	1.08		0.60	0.38	0.79		2.15	0.94	4.93	
	Clear cutting (CC)	8	0.77	0.60	0.88		0.94	0.66	1.34		0.89	0.80	0.94		1.21	0.66	2.19	
	Forest plantations (PL)	19	0.53	0.41	0.64		0.74	0.59	0.92		0.81	0.74	0.86		0.58	0.41	0.83	
	Reduced-impact logging (RI)	3	0.84	0.55	0.96		1.00	0.52	1.93		0.96	0.87	0.99		1.05	0.36	3.02	
	Selective cutting (SC)	13	0.68	0.55	0.79		1.23	0.95	1.59		0.92	0.88	0.95		0.99	0.65	1.52	

Table A2. Tukey post-hoc tests with Bonferroni correction outcomes show whether siginficant differences in animal and plant biodiversity occur between forest management types (p-value < 0.05).

		Intactness (IN)				Richness (SR)				Similarity (SI)				Total abundance (TA)			
	Combination	**Estimate**	**Std. Error**	**z value**	**p-value**	**Estimate**	**Std. Error**	**z value**	**p-value**	**Estimate**	**Std. Error**	**z value**	**p-value**	**Estimate**	**Std. Error**	**z value**	**p-value**
Animals	PC - AG	-1.5134	0.4467	-3.3880	0.0148	-0.6765	0.1444	-4.6840	0.0001	-2.1616	0.5571	-3.8800	0.0022	-0.0231	0.2203	-0.1050	1.0000
	SP - AG	-0.1310	0.5865	-0.2230	1.0000	-0.1490	0.1866	-0.7990	1.0000	-0.0234	0.7324	-0.0320	1.0000	0.1217	0.2888	0.4210	1.0000
	SP - PC	1.3824	0.5962	2.3190	0.4286	0.5275	0.1849	2.8520	0.0911	2.1382	0.7420	2.8810	0.0831	0.1448	0.2957	0.4900	1.0000
	PL - CC	-1.0111	0.3578	-2.8260	0.0991	-0.2182	0.1179	-1.8500	1.0000	-0.6474	0.4459	-1.4520	1.0000	-0.4663	0.1824	-2.5560	0.2220
	RI - CC	0.5086	0.7945	0.6400	1.0000	0.0578	0.2628	0.2200	1.0000	1.5037	0.9877	1.5220	1.0000	0.0354	0.3971	0.0890	1.0000
	SC - CC	-0.1083	0.3752	-0.2890	1.0000	0.2115	0.1222	1.7300	1.0000	0.7274	0.4643	1.5670	1.0000	-0.0665	0.1896	-0.3510	1.0000
	RI - PL	1.5196	0.7676	1.9800	1.0000	0.2760	0.2543	1.0860	1.0000	2.1511	0.9554	2.2510	0.5114	0.5017	0.3841	1.3060	1.0000
	SC - PL	0.9028	0.3132	2.8830	0.0827	0.4297	0.1027	4.1840	0.0006	1.3748	0.3902	3.5230	0.0089	0.3998	0.1605	2.4910	0.2680
	SC - RI	-0.6168	0.7755	-0.7950	1.0000	0.1536	0.2563	0.6000	1.0000	-0.7763	0.9651	-0.8040	1.0000	-0.1020	0.3875	-0.2630	1.0000
Plants	PC - AG	-1.3226	0.6217	-2.1270	0.3339	-1.7995	0.7761	-2.3190	0.2041	-2.0641	0.7693	-2.6830	0.0729	-0.5928	0.4496	-1.3190	1.0000
	PL - CC	-1.0168	0.5382	-1.8890	0.5888	-0.1699	0.6746	-0.2520	1.0000	-0.4089	0.6644	-0.6150	1.0000	-0.2857	0.3875	-0.7370	1.0000
	SC - CC	0.6049	0.5752	1.0520	1.0000	0.2388	0.7280	0.3280	1.0000	1.0933	0.7131	1.5330	1.0000	0.2473	0.4085	0.6050	1.0000
	SC - PL	1.6217	0.5753	2.8190	0.0482	0.4086	0.7267	0.5620	1.0000	1.5022	0.7117	2.1110	0.3480	0.5330	0.4083	1.3050	1.0000

Table A3. Biodiversity estimates and confidence intervals (CI) for relative richness (SR), similarity (SI) and total abundance (TA) of vulnerable to critically endangered animal species (computed by back-transforming the estimates). A significant p-value (<0.05) indicates that there is a significant difference between the impacts of different forest management systems on biodiversity. Coloring of the cells reflect the coloring used in the main text.

	Management systems	**Nr. Studies**	**SR - estimate**	**SR - lower CI**	**SR - upper CI**	**p-value**	**SI - estimate**	**SI - lower CI**	**SI - upper CI**	**p-value**	**TA - estimate**	**TA - lower CI**	**TA - upper CI**	**p-value**
Animals	Perennial tree crops (PC)	7	0.35	0.21	0.53	8.33E-05	0.37	0.24	0.56	0.23	0.15	0.09	0.25	0.04
	Forest plantations (PL)	7	0.50	0.31	0.69		0.71	0.47	1.06		0.37	0.23	0.59	
	Selective cutting (SC)	8	0.58	0.39	0.75		1.23	0.78	1.94		0.97	0.57	1.65	

Table A4. Regression coefficients for models with time as an fixed effect and a biodiversity indicator as the response variable. Biodiversity indicators include biodiversity intactness (IN), calculated as Mean Species Abundance (MSA), richness (SR), expressed as the relative number of species, similarity (SI), calculated as the Sørensen Similarity Index and total abundance (TA), calculated as the response ratio between the summed abundance of all species between managed and reference studies. Under the columns $\mathrm{Coef}_{\mathrm{Interc}}$ and $\mathrm{Coef}_{\mathrm{Ind}}$ coefficients are provided for the intercept, the ecological driver. SE = Standard error, X^2 = Chi-squared test result, p-value = p-value of Chi-squared test, N = the number of data points in the model, and df = degrees of freedom.

Indicator	**Management system**	$\mathrm{Coef}_{\mathrm{Interc}}$	$\mathrm{Coef}_{\mathrm{Ind}}$	$\mathrm{SE}_{\mathrm{inter}}$	$\mathrm{SE}_{\mathrm{ind}}$	**p-value**	X^2	**N**	**df**
Intactness (IN)	Agroforestry (AG)	-2.616	0.738	0.328	0.084	< 2.2E-16	77.196	15	1
	Perennial tree crops (PC)	-1.612	0.004	0.387	0.001	4.09E-04	12.492	9	1
	Clear cutting (CC)	0.998	-0.024	0.523	0.005	4.67E-07	25.396	35	1
	Forest plantations (PL)	-1.600	0.049	0.350	0.003	< 2.2E-16	260.770	42	1
	Selective cutting (SC)	0.257	0.009	0.159	0.004	1.16E-02	6.379	37	1
Richness (SR)	Agroforestry (AG)	-1.086	0.278	0.374	0.130	3.25E-02	4.575	15	1
	Perennial tree crops (PC)	-0.848	0.001	0.267	0.004	1.54E-01	0.695	9	1
	Clear cutting (CC)	-0.155	-0.004	0.129	0.004	4.06E-01	0.692	35	1
	Forest plantations (PL)	-0.504	0.002	0.127	0.004	5.88E-01	0.294	42	1
	Selective cutting (SC)	0.070	-0.005	0.070	0.003	8.92E-02	2.889	37	1
Similarity (SI)	Agroforestry (AG)	-0.760	0.561	0.345	0.088	1.98E-10	40.482	15	1
	Perennial tree crops (PC)	0.075	-0.006	0.412	0.001	2.28E-07	26.776	9	1
	Clear cutting (CC)	1.860	-0.026	0.427	0.004	3.67E-11	43.781	35	1
	Forest plantations (PL)	-0.025	0.043	0.336	0.003	< 2.2E-16	161.670	42	1
	Selective cutting (SC)	1.904	0.006	0.206	0.004	1.78E-01	1.813	37	1
Abundance (TA)	Agroforestry (AG)	-0.482	0.097	0.593	0.200	6.27E-01	0.236	15	1
	Perennial tree crops (PC)	-0.626	0.006	0.291	0.003	4.61E-02	3.977	9	1
	Clear cutting (CC)	0.173	-0.001	0.109	0.005	9.08E-01	0.013	35	1
	Forest plantations (PL)	-0.909	0.013	0.242	0.005	7.27E-03	7.205	42	1
	Selective cutting (SC)	-0.193	0.006	0.142	0.006	3.39E-01	0.913	37	1

Table A5. We used ANOVA to test whether models with both management systems (M) and taxonomic class (TC) or continent (C) outperformed those with only management for animal biodiversity for models estimating intactness, richness, similarity and total abundance, respectively. The models with the lowest BIC values were considered the best performing ones. p-value = significance of the difference between models with only management systems and with management systems and taxonomic class or continent as explanatory variables. If p-values were > 0.05 the models did not differ significantly from each other in terms of performance. Significantly best performing models are indicated in bold.

	Indicator	Fixed effects	BIC	p-value
Class	Intactness (IN)	M	1633.5	
		M + TC	-3523.4	$< 2.2e^{-16}$
	Richness (SR)	M	252.96	
		M + TC	271.59	0.6971
	Similarity (SI)	M	-4275.8	
		M + TC	-4826.5	$< 2.2e^{-16}$
	Abundance (TA)	**M**	469.32	
		M + TC	477.9	0.0155
Continent	Intactness (IN)	**M**	1633.5	
		M + C	1639.5	0.03456
	Richness (SR)	M	252.96	
		M + C	270.92	0.1511
	Similarity (SI)	M	-4275.8	
		M + C	-4257.3	0.1869
	Abundance (TA)	**M**	469.32	
		M + C	483.59	0.0379

Table A6. List of references from which biodiversity and forest management data was derived.

Title	Authors	Year	Journal	DOI	Volume	Issue	Page numbers
Biodiversity of man-made open habitats in an underused country: a class of multispecies abundance models for count data	Yamaura, Y.;Royle, J. A.;Shimada, N.;Asanuma, S.;Sato, T.;Taki, H.;Makino, S.	2012	Biodiversity and Conservation	10.1007/s10531-012-0244-z	21	6	1365-1380
Effects of habitat and landscape characteristics on medium and large mammal species richness and composition in northern Uruguay	Andrade-Núñez, M.J. and Aide, T.M.	2010	Zoologia (Curitiba)	10.1590/S1984-46702010000600012	27		909-917
Effect of deforestation on a southwest Ghana dung beetle assemblage (Coleoptera: Scarabaeidae) at the periphery of Ankasa conservation area	Davis, A. L. V.;Philips, T. K.	2005	Environmental Entomology	10.1603/0046-225x(2005)034[1081:eodoas]2.0.co;2	34	5	1081-1088
Diversity and abundance of terrestrial ants along a gradient of land use intensification in a transitional forest-savannah zone of CÃ´te d'Ivoire	Kone, Mouhamadou;Konate, Souleymane;Yeo, Kolo;Kouassi, Philippe Kouassi;Linsenmair, K E	2010	Journal of Applied Biosciences		29		1809-1827
Biodiversity and ecosystem function of tropical forest dung beetles under contrasting logging regimes	Slade, Eleanor M.;Mann, Darren J.;Lewis, Owen T.	2011	Biological Conservation	10.1016/j.biocon.2010.08.011	144	1	166-174
From forest to farmland: butterfly diversity and habitat associations along a gradient of forest conversion in Southwestern Cameroon	Bobo, K.S.	2006	Journal of Insect Conservation	10.1007/s10841-005-8564-x	10		29-42
Elephant- and human-induced changes to dung beetle (Coleoptera: Scarabaeidae) assemblages in the Maputaland Centre of Endemism	Botes, A.	2006	Biological Conservation	10.1016/j.biocon.2006.01.020	130	4	573-583
Effects of reduced impact logging on bat biodiversity in terra firme forest of lowland Amazonia	Castro-Arellano, I.	2007	Biological Conservation	10.1016/j.biocon.2007.04.025	138	1-2	269-285
Connecting sustainable agriculture and wildlife conservation: Does shade coffee provide habitat for mammals?	Caudill, S.A., DeClerck, F.J. and Husband, T.P.	2015	Agriculture, Ecosystems & Environment	10.1016/j.agee.2014.08.023	199		85-93
Effects of fragmentation of the Atlantic forest on mammal communities in south-eastern Brazil	Chiarello, A. G.	1999	Biological Conservation	10.1016/S0006-3207(98)00130-X	89	1	71-82
The diversity of beetle assemblages in different habitat types in Sabah, Malaysia	Chung, A.Y.C.	2000	Bulletin of Entomological Research	10.1017/S0007485300000602	90	6	475-496
Logging Concessions Can Extend the Conservation Estate for Central African Tropical Forests	Clark, C.J.	2009	Conservation Biology	10.1111/j.1523-1739.2009.01243.x	23	5	1281-1293

Butterfly community recovery in degraded rainforest habitats in the Upper Guinean Forest Zone (Kakum forest, Ghana)	Safian, S.;Csontos, G.;Winkler, D.	2011	Journal of Insect Conservation	10.1007/s10841-010-9343-x	15	2-Jan	351-359
The role of remnants of Amazon savanna for the conservation of Neotropical mammal communities in eucalyptus plantations.	Coelho, M., Juen, L. and Mendes-Oliveira, A.C.	2014	Biodiversity and conservation	10.1007/s10531-014-0772-9			
Bats like vintage: managing exotic eucalypt plantations for bat conservation in a M editerranean landscape	Cruz, J., Sarmento, P., Rydevik, G., Rebelo, H. and White, P.C.L.	2016	Animal Conservation	10.1111/acv.12216	19	1	53-64
Responses of diurnal tree squirrels to selective logging in western Arunachal Pradesh	Datta, A. and Goyal, S.P.	2008	Current Science				895-902
A comparison of orang-utan density in a logged and unlogged forest on Sumatra	Knop, E.;Ward, P. I.;Wich, S. A.	2004	Biological Conservation	10.1016/j.biocon.2004.02.010	120	2	183-188
Species richness and community composition of songbirds in a tropical forest-agricultural landscape	Naidoo, R.	2004	Animal Conservation	10.1017/s1367943003001185	7		93-105
The impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny forest, southern Madagascar	Scott, D. M.;Brown, D.;Mahood, S.;Denton, B.;Silburn, A.;Rakotondraparany, F.	2006	Biological Conservation	10.1016/j.biocon.2005.07.014	127	1	72-87
Oil palm expansion into rain forest greatly reduces ant biodiversity in canopy, epiphytes and leaf-litter	Fayle, T. M.;Turner, E. C.;Snaddon, J. L.;Chey, V. K.;Chung, A. Y. C.;Eggleton, P.;Foster, W. A.	2010	Basic and Applied Ecology	10.1016/j.baae.2009.12.009	11	4	337-345
Abundance responses of frugivorous bats (Stenodermatinae) to coffee cultivation and selective logging practices in mountainous central Veracruz, Mexico	Saldana-Vázquez, R.A., Sosa, V.J., Hernández-Montero, J.R. and López-Barrera, F.	2010	Biodiversity and Conservation	10.1007/s10531-010-9829-6	19	7	2111-2124
The distribution and abundance of small mammals in agroecosystems of southeastern Brazil	Gheler-Costa, C.;Vettorazzi, C. A.;Pardini, R.;Verdade, L. M.	2012	Mammalia	10.1515/mammalia-2011-0109	76	2	185-191
Terrestrial non-volant small mammals in agro-silvicultural landscapes of Southeastern Brazil	Martin, P. S.;Gheler-Costa, C.;Lopes, P. C.;Rosalino, L. M.;Verdade, L. M.	2012	Forest Ecology and Management	10.1016/j.foreco.2012.07.002	282		185-195
The influence of agricultural system, stand structural complexity and landscape context on foraging birds in oil palm landscapes	Azhar, B.;Lindenmayer, D. B.;Wood, J.;Fischer, J.;Manning, A.;McElhinny, C.;Zakaria, M.	2013	Ibis	10.1111/ibi.12025	155	2	297-312
Biodiversity and land-use change: understanding the complex responses of an endemic-rich bird assemblage	de Lima, R. F.;Dallimer, M.;Atkinson, P. W.;Barlow, J.	2013	Diversity and Distributions	10.1111/ddi.12015	19	4	411-422
Avian responses to fragmentation of the Maulino Forest in central Chile	Vergara, P. M.;Simonetti, J. A.	2004	Oryx	10.1017/s0030605304000742	38	4	383-388
Presence and abundance of birds in an Atlantic forest reserve and	Cockle, K. L.;Leonard, M. L.;Bodrati, A. A.	2005	Biodiversity and Conservation	10.1007/s10531-004-0446-0	14	13	3265-3288

adjacent plantation of shade-grown yerba mate, in Paraguay							
Human disturbance affects a deciduous forest bird community in the Andean foothills of central Bolivia	Aben, Job;Dorenbosch, Martijn;Herzog, Sebastian K.;Smolders, Alfons J. P.;Van Der Velde, Gerard	2008	Bird Conservation International	10.1017/s0959270908007326	18	4	363-380
Ants as biological indicators of Wayana Amerindian land use in French Guiana	Delabie, Jacques H. C.;Cereghino, Regis;Groc, Sarah;Dejean, Andrea;Gibernau, Marc;Corbara, Bruno;Dejean, Alain	2009	Comptes Rendus Biologies	10.1016/j.crvi.2009.01.006	332	7	673-684
Diversity and abundance of orchid bees (Hymenoptera: Apidae, Euglossini) in a tropical rainforest succession	Rasmussen, Claus	2009	Neotropical Entomology	10.1590/s1519-566x2009000100006	38	1	66-73
Small off-shore islands can serve as important refuges for endemic beetle conservation	Sugiura, S.;Tsuru, T.;Yamaura, Y.;Makihara, H.	2009	Journal of Insect Conservation	10.1007/s10841-008-9185-y	13	4	377-385
Riqueza e abundância de aves em três estágios sucessionais da floresta ombrófila densa submontana, Antonina, Paraná.	Boçon, R.	2010					
Plant and bird diversity in natural forests and in native and exotic plantations in NW Portugal	Proença, V.M., Pereira, H.M., Guilherme, J. and Vicente, L.	2010	Acta Oecologica-International Journal of Ecology	10.1016/j.actao.2010.01.002	36	2	219-226
Differences in the effects of selective logging on amphibian assemblages in three West African forest types	Ofori-Boateng, C.;Oduro, W.;Hillers, A.;Norris, K.;Oppong, S. K.;Adum, G. B.;Rodel, M. O.	2013	Biotropica	10.1111/j.1744-7429.2012.00887.x	45	1	94-101
Medium to large-sized mammals in agricultural landscapes of south-eastern Brazil	Dotta, G. and Verdade, L.M.	2011		10.1515/MAMM.2011.049			345-352
Responses of five arboreal marsupials to recent selective logging in tropical Australia	Laurance, W. F.;Laurance, S. G. W.	1996	Biotropica	10.2307/2389195	28	3	310-322
Phyllostomid bats of a fragmented landscape in the north-eastern Atlantic forest, Brazil	Faria, D.	2006	Journal of Tropical Ecology	10.1017/S0266467406003385	22	5	531-542
The value of primary, secondary and plantation forests for Amazonian birds	Barlow, Jos;Mestre, Luiz A. M.;Gardner, Toby A.;Peres, Carlos A.	2007	Biological Conservation	10.1016/j.biocon.2006.11.021	136	2	212-231
A case study on the effects of disturbance and conversion of tropical lowland rain forest on the non-volant small mammals in north Borneo: management implications	Bernard, Henry;Fjeldsa, Jon;Mohamed, Maryati	2009	Mammal Study	10.3106/041.034.0204	34	2	85-96
Terrestrial wildlife responses to logging and fire in a Bolivian tropical humid forest	Fredericksen, N.J.	2002	Biodiversity and Conservation	10.1023/A:1014065510554	11	Jan	27-38

Lizard diversity and agricultural disturbance in a Caribbean forest landscape	Glor, R.E.	2001	Biodiversity and Conservation	10.1023/A:1016665011087	10		711-723
Effects of forest disturbance on the structure of ground-foraging ant communities in central Amazonia	Vasconcelos, H. L.	1999	Biodiversity and Conservation		8	3	409-420
Responses of ants to selective logging of a central Amazonian forest	Vasconcelos, H. L.;Vilhena, J. M. S.;Caliri, G. J. A.	2000	Journal of Applied Ecology	10.1046/j.1365-2664.2000.00512.x	37	3	508-514
Diversity of pselaphine beetles (Coleoptera: Staphylinidae: Pselaphinae) in eastern Thailand	Sakchoowong, Watana;Nomura, Shuhei;Ogata, Kazuo;Chanpaisaeng, Jariya	2008	Entomological Science	10.1111/j.1479-8298.2008.00281.x	11	3	301-313
Bat diversity in tropical forest and agro-pastoral habitats within a protected area in the Philippines	Sedlock, Jodi L.;Weyandt, Sarah E.;Cororan, Laura;Damerow, Marin;Hwa, Shi-Hsia;Pauli, Benjamin	2008	Acta Chiropterologica	10.3161/150811008x414926	10	2	349-358
Hunting for sustainability in tropical secondary forests	Parry, L.;Barlow, J.;Peres, C. A.	2009	Conservation Biology	10.1111/j.1523-1739.2009.01224.x	23	5	1270-1280
Diversity and similarity of butterfly communities in five different habitat types at Tam Dao National Park, Vietnam	Vu, L. V.	2009	Journal of Zoology	10.1111/j.1469-7998.2008.00498.x	277	1	15-22
Mousedeer densities in a tropical rainforest: The impact of selective logging	Heydon, M.J.	1997	Journal of Applied Ecology	10.2307/2404892	34		484-496
Orangutan population density, forest structure and fruit availability in hand-logged and unlogged peat swamp forests in West Kalimantan, Indonesia	Felton, A. M.;Engstrom, L. M.;Felton, A.;Knott, C. D.	2003	Biological Conservation	10.1016/s0006-3207(03)00013-2	114	1	91-101
Diversity and community composition of butterflies and odonates in an ENSO-induced fire affected habitat mosaic: a case study from East Kalimantan, Indonesia	Cleary, D. F. R.;Mooers, A. O.;Eichhorn, K. A. O.;van Tol, J.;de Jong, R.;Menken, S. B. J.	2004	Oikos	10.1111/j.0030-1299.2004.12219.x	105	2	426-446
Species diversity and habitat fragmentation: frogs in a tropical montane landscape in Mexico	Pineda, E.;Halffter, G.	2004	Biological Conservation	10.1016/j.biocon.2003.08.009	117	5	499-508
Short-term responses of birds to forest gaps and understory: an assessment of reduced-impact logging in a lowland Amazon forest	Wunderle, J. M.;Henriques, L. M. P.;Willig, M. R.	2006	Biotropica	10.1111/j.1744-7429.2006.00138.x	38	2	235-255
Influence of anthropogenic disturbance on birds of tropical dry forest: the role of vegetation structure	Shahabuddin, G.;Kumar, R.	2006	Animal Conservation	10.1111/j.1469-1795.2006.00051.x	9	4	404-413
Effects of extractive disturbance on bird assemblages, vegetation structure and floristics in tropical scrub forest, Sariska Tiger Reserve, India	Shahabuddin, Ghazala;Kumar, Raman	2007	Forest Ecology and Management	10.1016/j.foreco.2007.03.061	246	3-Feb	175-185

Effects of rain forest logging on species richness and assemblage composition of small mammals in Southeast Asia	Wells, Konstans;Kalko, Elisabeth K. V.;Lakim, Maklarin B.;Pfeiffer, Martin	2007	Journal of Biogeography	10.1111/j.1365-2699.2006.01677.x	34	6	1087-1099
The value of primary, secondary and plantation forests for fruit-feeding butterflies in the Brazilian Amazon	Barlow, Jos;Overal, William L.;Araujo, Ivanei S.;Gardner, Toby A.;Peres, Carlos A.	2007	Journal of Applied Ecology	10.1111/j.1365-2664.2007.01347.x	44	5	1001-1012
Quantifying the biodiversity value of tropical primary, secondary, and plantation forests	Barlow, J.;Gardner, T. A.;Araujo, I. S.;Avila-Pires, T. C.;Bonaldo, A. B.;Costa, J. E.;Esposito, M. C.;Ferreira, L. V.;Hawes, J.;Hernandez, M. M.;Hoogmoed, M. S.;Leite, R. N.;Lo-Man-Hung, N. F.;Malcolm, J. R.;Martins, M. B.;Mestre, L. A. M.;Miranda-Santos, R.;Nunes-Gutjahr, A. L.;Overal, W. L.;Parry, L.;Peters, S. L.;Ribeiro-Junior, M. A.;da Silva, M. N. F.;Motta, C. da Silva;Peres, C. A.	2007	Proceedings of the National Academy of Sciences of the United States of America	10.1073/pnas.0703333104	104	47	18555-18560
Conservation value of forest plantations for bird communities in western Kenya	Farwig, Nina;Sajita, Nixon;Boehning-Gaese, Katrin	2008	Forest Ecology and Management	10.1016/j.foreco.2008.03.042	255	11	3885-3892
Understanding the biodiversity consequences of habitat change: the value of secondary and plantation forests for neotropical dung beetles	Gardner, Toby A.;Hernandez, Malva I. M.;Barlow, Jos;Peres, Carlos A.	2008	Journal of Applied Ecology	10.1111/j.1365-2664.2008.01454.x	45	3	883-893
Tolerance of frugivorous birds to habitat disturbance in a tropical cloud forest	Gomes, Laurens G. L.;Oostra, Vicencio;Nijman, Vincent;Cleef, Antoine M.;Kappelle, Maarten	2008	Biological Conservation	10.1016/j.biocon.2008.01.007	141	3	860-871
Invasion and retreat: shifting assemblages of dung beetles amidst changing agricultural landscapes in central Peru	Horgan, Finbarr G.	2009	Biodiversity and Conservation	10.1007/s10531-009-9658-7	18	13	3519-3541

Alpha and beta diversity of plants and animals along a tropical land-use gradient	Kessler, Michael;Abrahamczyk, Stefan;Bos, Merijn;Buchori, Damayanti;Putra, Dadang Dwi;Gradstein, S. Robbert;Hoehn, Patrick;Kluge, Juergen;Orend, Friederike;Pitopang, Ramadhaniel;Saleh, Shahabuddin;Schulze, Christian H.;Sporn, Simone G.;Steffan-Dewenter, Ingolf;Tjitrosoedirdjo, S.;Tscharntke, Teja	2009	Ecological Applications	10.1890/08-1074.1	19	8	2142-2156
Vertebrate population responses to reduced-impact logging in a neotropical forest	Bicknell, Jake;Peres, Carlos A.	2010	Forest Ecology and Management	10.1016/j.foreco.2010.02.027	259	12	2267-2275
Do riparian reserves support dung beetle biodiversity and ecosystem services in oil palm-dominated tropical landscapes?	Gray, C. L.;Slade, E. M.;Mann, D. J.;Lewis, O. T.	2014	Ecology and Evolution	10.1002/ece3.1003	4	7	1049-1060
Impacts of forestry on ant species richness and composition in warm-temperate forests of Japan	Maeto, K.;Sato, S.	2004	Forest Ecology and Management	10.1016/s0378-1127(03)00333-5	187	3-Feb	213-223
Comparative study of forest-dwelling bats' abundances and species richness between old-growth forests and conifer plantations in Nikko National Park, central Japan	Yoshikura, S.;Yasui, S.;Kamijo, T.	2011	Mammal Study	10.3106/041.036.0402	36	4	189-198
Habitat use by carnivores at different spatial scales in a plantation forest landscape in Patagonia, Argentina	Lantschner, M. V.;Rusch, V.;Hayes, J. P.	2012	Forest Ecology and Management	10.1016/j.foreco.2011.12.045	269		271-278
Effects of habitat degradation on the lizard assemblage in the Arid Chaco, central Argentina	Pelegrin, N.;Bucher, E. H.	2012	Journal of Arid Environments	10.1016/j.jaridenv.2011.11.004	79		13-19
The effects of habitat disturbance on lemurs at Ranomafana National Park, Madagascar	Herrera, J. P.;Wright, P. C.;Lauterbur, E.;Ratovonjanahary, L.;Taylor, L. L.	2011	International Journal of Primatology	10.1007/s10764-011-9525-8	32	5	1091-1108
Small rodent populations in selectively felled and mature tracts of Kibale forest, Uganda	Isabirye-Basuta, G.	1987	Biotropica	10.2307/2388345	19		260-266
The scale of community structure: habitat variation and avian guilds in tropical forest understory	Pearman, P. B.	2002	Ecological Monographs	10.2307/3100083	72	1	19-39
Distribution and abundance of small mammals in relation to habitat in Pureora Forest Park	King, C.M., Innes, J.G., Flux, M., Kimberley, M.O., Leathwick, J.R. and Williams, D.S.	1996	New Zealand Journal of Ecology				215-240
An evaluation of the contribution of cultivated allspice (Pimenta dioca) to vertebrate biodiversity conservation in Nicaragua.	King, D. I., Hernandez-Mayorga, M. D., Trubey, R., Raudales, R., & Rappole, J. H.	2007	Biological Conservation	10.1007/s10531-006-9068-z	16		1299-1320

Effects of anthropogenic environmental changes on amphibian diversity in the rain forests of eastern Madagascar	Vallan, D.	2002	Journal of Tropical Ecology	10.1017/s026646740200247x	18		725-742
The consequences of selective logging for Bornean lowland forest birds	Lambert, F.R.	1992	Philosophical Transactions: Biological Sciences	10.1098/rstb.1992.0036	335	1275	443-457
Effects of mahogany (Swietenia macrophylla) logging on small mammal communities, habitat structure, and seed predation in the southeastern Amazon Basin	Lambert, T.D.	2005	Forest Ecology & Management	10.1016/j.foreco.2004.11.012	206	1-3	381-398
The Conservation biology of the Leopard Panthera pardus in Gabon: status, threats and strategies for conservation	Henschel, Philipp	2008					
Bats of the Chilean temperate rainforest: patterns of landscape use in a mosaic of native forests, eucalyptus plantations and grasslands within a South American biodiversity hotspot	Meynard, C.N., Soto-Gamboa, M., Heady, P.A. and Frick, W.F.	2014	Biodiversity and conservation	10.1007/s10531-014-0697-3	23		1949-1963
Effect of land-use heterogeneity on carabid communities at the landscape scale	Vanbergen, A. J.;Woodcock, B. A.;Watt, A. D.;Niemela, J.	2005	Ecography	10.1111/j.0906-7590.2005.03991.x	28	1	16-Mar
Effects of exotic conifer plantations on the biodiversity of understory plants, epigeal beetles and birds in Nothofagus dombeyi forests	Paritsis, Juan;Aizen, Marcelo A.	2008	Forest Ecology and Management	10.1016/j.foreco.2007.11.015	255	6-May	1575-1583
Unpublished data of reptilian and amphibian diversity in six countries in Central America	Centro AgronÃ³mico Tropical de InvestigaciÃ³n y EnseÃ±anza (CATIE)	2010					
Bird species richness in a Bornean exotic tree plantation: a long-term perspective	Sheldon, FH;Styring, A;Hosner, PA	2010	Biological Conservation	10.1016/j.biocon.2009.11.004	143	2	399-407
Reconciling food production and biodiversity conservation: land sharing and land sparing compared	Phalan, B;Onial, M;Balmford, A;Green, RE	2011	Science	10.1126/science.1208742	333	6047	1289-1291
Bird species richness and abundance in different forest types at Kakamega Forest, western Kenya	Munyekenye, FB;Mwangi, EM;Gichuki, NN	2008	Ostrich	10.2989/ostrich.2008.79.1.4.361	79	1	37-42
Habitat disturbance reduces seed dispersal of a forest interior tree in a fragmented African cloud forest	Lehouck, V;Spanhove, T;Colson, L;Adringa-Davis, A;Cordeiro, NJ;Lens, L	2009	Oikos	10.1111/j.1600-0706.2009.17300.x	118	7	1023-1034
Amphibian community along elevational and habitat disturbance gradients in the Taita Hills, Kenya	Malonza, Patrick K.;Veith, Michael	2012	Herpetotropicos		7	1--2	7--16
Two-stage recovery of amphibian assemblages following selective logging of tropical forests	Adum, G. B.;Eichhorn, M. P.;Oduro, W.;Ofori-Boateng, C.;Rodel, M. O.	2013	Conservation Biology	10.1111/cobi.12006	27	2	354-363
Population density and habitat preferences of forest duikers in	Nakashima, Y;Inoue, E;Akomo-Okoue, EF	2013	African Zoology	10.3377/004.048.0212	48	2	395-399

Moukalaba-Doudou National Park, Gabon							
Human disturbance and the activity patterns and temporal overlap of tapirs and jaguars in reserves of NW Belize.	Monette, V.D., Kelly, M.J. and Buchholz, R.	2020	Biotropica	10.1111/btp.12834	52	6	1262-1274
Native forest replacement by exotic plantations triggers changes in prey selection of mesocarnivores	Moreira-Arce, D., Vergara, P.M., Boutin, S., Simonetti, J.A., Briceño, C. and Acosta-Jamett, G.	2015	Biological Conservation	10.1016/j.biocon.2015.09.015	192		258-267
Effects of forest fragmentation on small mammals in an Atlantic Forest landscape	Pardini, R.	2004	Biodiversity and Conservation	10.1023/B:BIOC.0000048452.18878.2d	13		2567-2586
Correlates of amphibian diversity in an altered landscape of Amazonian Ecuador	Pearman, P.B.	1997	Conservation Biology	10.1046/j.1523-1739.1997.96202.x	11		1211-1225
Low leopard populations in protected areas of Maputaland: a consequence of poaching, habitat condition, abundance of prey, and a top predator	Ramesh, T., Kalle, R., Rosenlund, H. and Downs, C.T.	2017	Ecology and evolution	10.1002/ece3.2771	7	6	1964-1973
Small mammals of Maulino forest remnants, a vanishing ecosystem of south-central Chile	Saavedra, B. and Simonetti, J.A.	2005		10.1515/mamm.2005.027			337-348
Effects of timber harvest on forest Lepidoptera: community, guild and species responses	Summerville, K. S.;Crist, T. O.	2002	Ecological Applications	10.1890/1051-0761(2002)012[0820:eothof]2.0.co;2	12	3	820-835
Conservation Comores 2005: biodiversity and resource-use assessment and environmental awareness	Doulton, H.;Marsh, C.;Newman, A.;Bird, K.;Bell, M.	2007					110
Spatio-temporal dymanics of snake diversity in four habitats with different degrees of anthropogenic disturbance in the Gorgona Island National Natural Park in the Colombian Pacific	URBINA-CARDONA, JOSÉ NICOLÁS, MARIA CECILIA LONDOÑO-MURCIA, and DANIEL GUILLERMO GARCÍA-ÁVILA	2008	Caldasia		30	2	479-493
Effects of anthropogenic activities on lizard communities in northern Madagascar	D'Cruze, N.;Kumar, S.	2011	Animal Conservation	10.1111/j.1469-1795.2011.00459.x	14	5	542-552
Lizard assemblages in a fragmented landscape of central Chile	Rubio, Andre V.;Simonetti, Javier A.	2011	European Journal of Wildlife Research	10.1007/s10344-010-0434-5	57	1	195-199
Forest-land use complementarity modifies community structure of a tropical herpetofauna	Kurz, David J.;Nowakowski, A. Justin;Tingley, Morgan W.;Donnelly, Maureen A.;Wilcove, David S.	2014	Biological Conservation	10.1016/j.biocon.2013.12.027	170		246-255
Cross-taxon congruence of alpha and beta diversity among five leaf litter arthropod groups in Colombia	Cabra-García, J., Bermúdez-Rivas, C., Osorio, A.M. and Chacón, P.	2012	Biodiversity and Conservation	10.1007/s10531-012-0259-5	21	6	1493-1508

Unpublished data of the response of ground beetles (Coleoptera: Carabidae) in the northeastern Colombian Andes to habitat modification	Rey-Velasco, Juan Carlos;Miranda-Esquivel, Daniel Rafael	2012					
Adult butterfly communities in coffee plantations around a protected area in the Western Ghats, India	Dolia, J.;Devy, M. S.;Aravind, N. A.;Kumar, A.	2008	Animal Conservation	10.1111/j.1469-1795.2007.00143.x	11	1	26-34
Assessing rodent community responses in disturbed environments of the Chilean Patagonia	Garcia, K.P., Zapata, J.C.O., Aguayo, M. and D'Elia, G.	2013	Mammalia	10.1515/mammalia-2011-0134	77	2	195-204
Diversity and abundance of some ground-dwelling invertebrates in plantation vs. native forests in Tasmania, Australia	Bonham, K. J.;Mesibov, R.;Bashford, R.	2002	Forest Ecology and Management	10.1016/s0378-1127(00)00717-9	158	3-Jan	237-247
Shrew species diversity and abundance in Ziama Biosphere Reserve, Guinea: comparison among primary forest, degraded forest and restoration plots	Nicolas, V;Barriere, P;Tapiero, A;Colyn, M	2009	Biodiversity and Conservation	10.1007/s10531-008-9572-4	18	8	2043-2061
The effects of management systems on ground-foraging ant diversity in Costa Rica	Roth, D. S.;Perfecto, I.;Rathcke, B.	1994	Ecological Applications	10.2307/1941947	4	3	423-436
Bird communities in logged and unlogged compartments in Budongo Forest, Uganda	Owiunji, I.;Plumptre, A. J.	1998	Forest Ecology and Management	10.1016/s0378-1127(98)00219-9	108	2-Jan	115-126
From forest to farmland: butterfly diversity and habitat associations along a gradient of forest conversion in Southwestern Cameroon	Bobo, K. S.;Waltert, M.;Fermon, H.;Njokagbor, J.;Muhlenberg, M.	2006	Journal of Insect Conservation	10.1007/s10841-005-8564-x	10	1	29-42
The effects of various forest uses on small mammal communities in Sarawak, Malaysia	Nakagawa, Michiko;Miguchi, Hideo;Nakashizuka, Tohru	2006	Forest Ecology and Management	10.1016/j.foreco.2006.05.006	231	3-Jan	55-62
High sensitivity of montane bird communities to habitat disturbance in Peninsular Malaysia	Soh, M. C. K.;Sodhi, N. S.;Lim, S. L. H.	2006	Biological Conservation	10.1016/j.biocon.2005.10.030	129	2	149-166
Bat diversity and abundance associated with the degree of secondary succession in a tropical forest mosaic in south-eastern Mexico	Castro-Luna, A. A.;Sosa, V. J.;Castillo-Campos, G.	2007	Animal Conservation	10.1111/j.1469-1795.2007.00097.x	10	2	219-228
How resilient are Andean montane forest bird communities to habitat degradation?	O'Dea, N.;Whittaker, R. J.	2007	Biodiversity and Conservation	10.1007/s10531-006-9095-9	16	4	1131-1159
Secondary rain forests are not havens for reptile species in tropical Mexico	Luja, VH;Herrando-Perez, S;Gonzalez-Solis, D;Luiselli, L	2008	Biotropica	10.1111/j.1744-7429.2008.00439.x	40	6	747-757
Bat diversity in the vegetation mosaic around a lowland dipterocarp forest of Borneo	Fukuda, Daisuke;Tisen, Oswald Braken;Momose, Kuniyasu;Sakai, Shoko	2009	Raffles Bulletin of Zoology		57	1	213-221
Habitat preferences of medium/large mammals in human disturbed forests in Central Japan	Tsujino, R., & Yumoto, T.	2014	Ecological research	10.1007/s11284-014-1159-9	29		701-710

Analysis of the richness of Staphylinidae (Coleoptera) on different scales of a sub-Andean rural landscape in Colombia	Vásquez-Vélez, L.M., Bermúdez, C., Chacón, P. and Lozano-Zambrano, F.H.	2010	Biodiversity and conservation	10.1007/s10531-010-9812-2	19		1917-1931
Effects of stand structure and landscape characteristics on habitat use by birds and small mammals in managed boreal forest of eastern Canada	St-Laurent, M. H.;Ferron, J.;Hins, C.;Gagnon, R.	2007	Canadian Journal of Forest Research-Revue Canadienne de Recherche Forestiere	10.1139/xo6-295	37	8	1298-1309
New records of rare species of Coleoptera found in UkmergÄ— district in 2004-2005	Noreika, Norbertas	2009	New and Rare for Lithuania Insect Species		21		68-71
Spatial distribution of meadow jumping mice (Zapus hudsonius) in logged boreal forest of northwestern Canada	Jung, T. S.;Powell, T.	2011	Mammalian Biology	10.1016/j.mambio.2011.08.002	76	6	678-682
Alternative silvicultural practices in irregular boreal forests: response of beetle assemblages	Légaré, J.P., Hébert, C. and Ruel, J.C.	2011	Silva Fennica		45	5	937-956
Distribution of carabid beetles among regenerating and natural forest types in Southwestern China	Yu, X.D., Luo, T.H. and Zhou, H.Z.	2006	Forest Ecology and Management	10.1016/j.foreco.2006.05.043	231	1-3	169-177
Long-term changes of small mammal communities in heterogenous landscapes of Central Europe.	Zárybnická, M., Riegert, J., Bejček, V., Sedláček, F., Šťastný, K., Šindelář, J., Heroldová, M., Vilímová, J. and Zima, J.	2017	European Journal of Wildlife Research	10.1007/s10344-017-1147-9	63		1-12

Appendix B

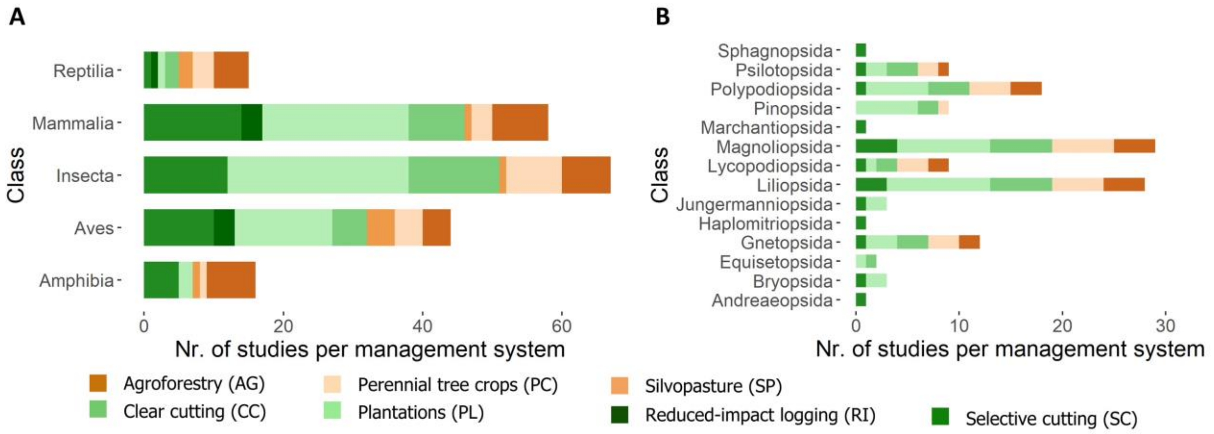


Figure B1. Number of studies carried out per animal and plant class for the different forest management systems. (A) Animal biodiversity studies per class categorized by forest management system. (B) Plant biodiversity studies per class categorized by forest management system.

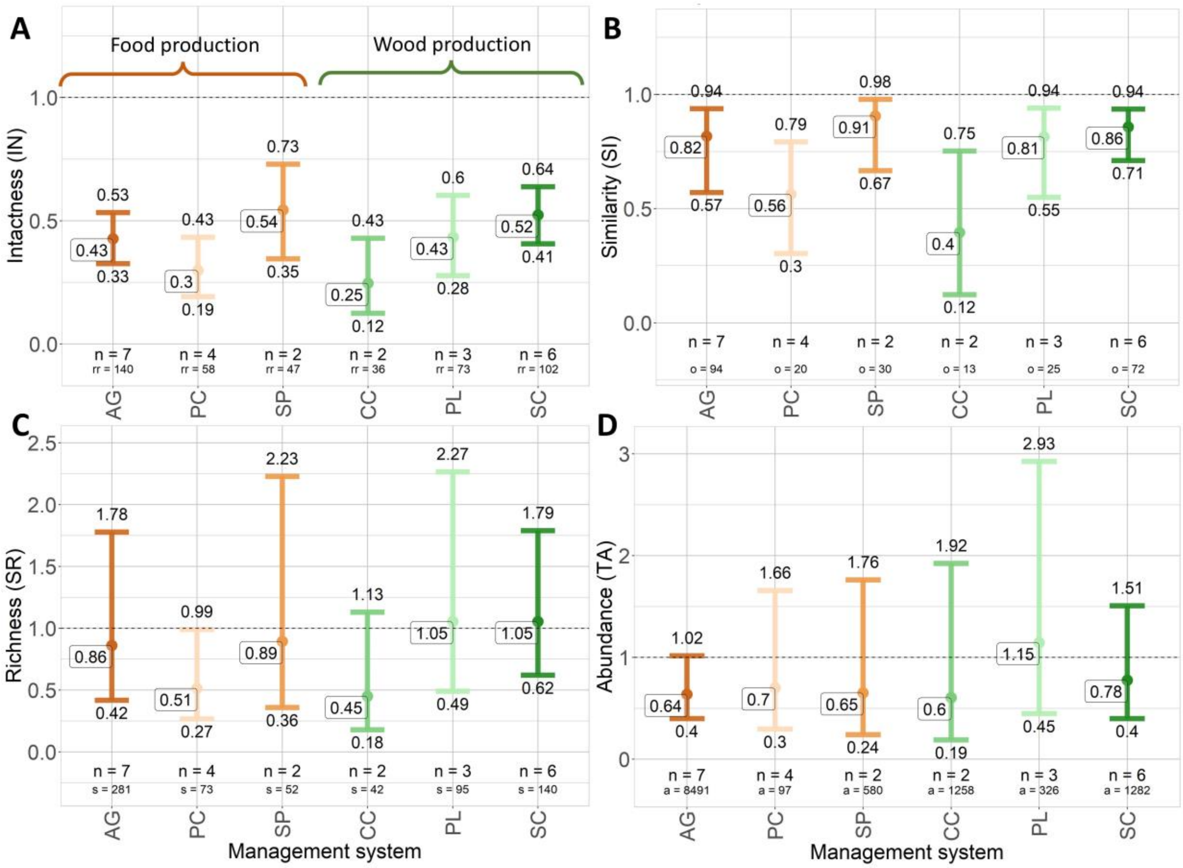


Figure B2. Impacts of forest management on herpetofauna biodiversity, expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish six forest management systems: (i) agroforests (AG), (ii) perennial tree crop plantations (PC), (iii) silvopasture (SP), (iv) clear cutting and regrowth (CC), (v) forest plantations (PL), (vi) selective cutting (SC). The wood production systems are in green and the food production systems in browns. n = number of studies included per management type, rr = number of response values used to compute intactness, o = number of overlapping species between managed and reference forest sites, s = number of species occurring across the managed and reference sites, a = number of abundance values used to compute relative total abundance. The dotted line indicates the reference level biodiversity.

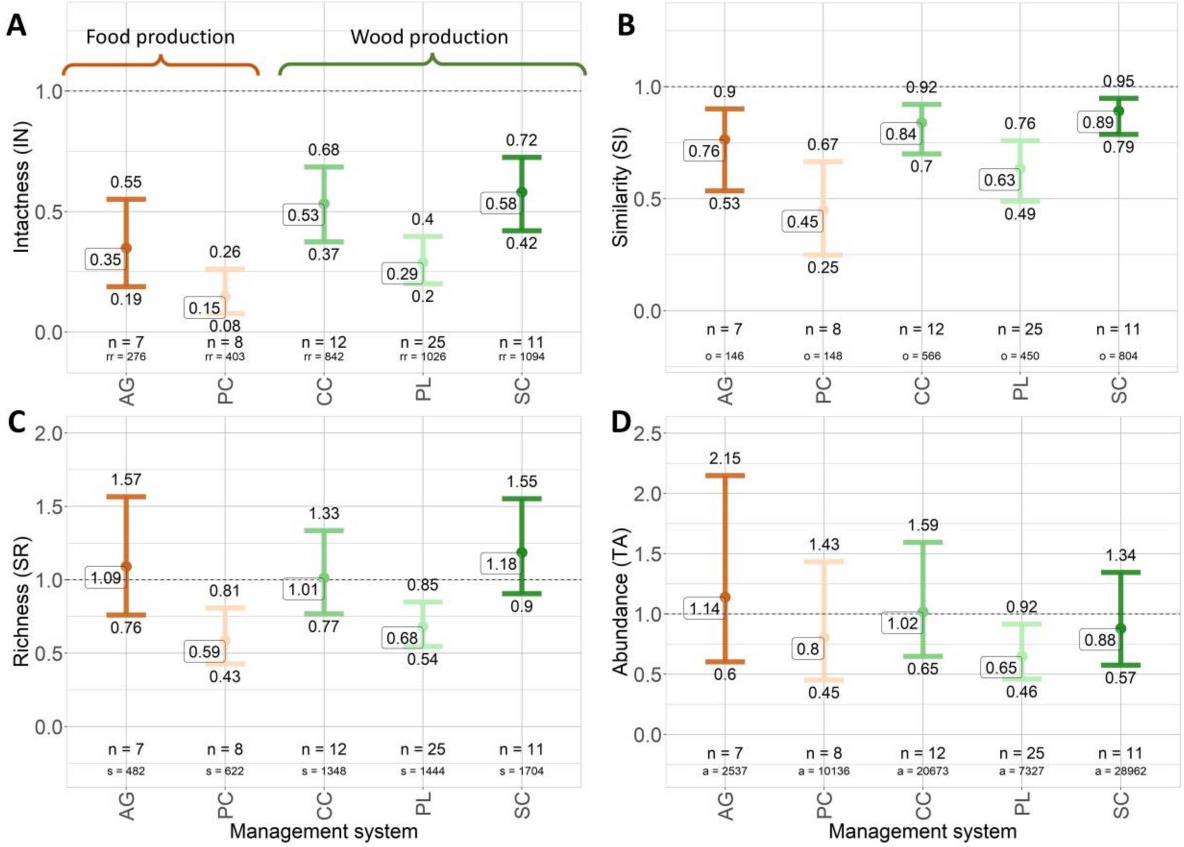


Figure B3. Impacts of forest management on insect biodiversity, expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish five forest management systems: (i) agroforests (AG), (ii) perennial tree crop plantations (PC), (iii) clear cutting and regrowth (CC), (iv) forest plantations (PL), (v) selective cutting (SC). The wood production systems are in green and the food production systems in browns. n = number of studies included per management type, rr = number of response values used to compute intactness, o = number of overlapping species between managed and reference forest sites, s = number of species occurring across the managed and reference sites, a = number of abundance values used to compute relative total abundance. The dotted line indicates the reference level biodiversity.

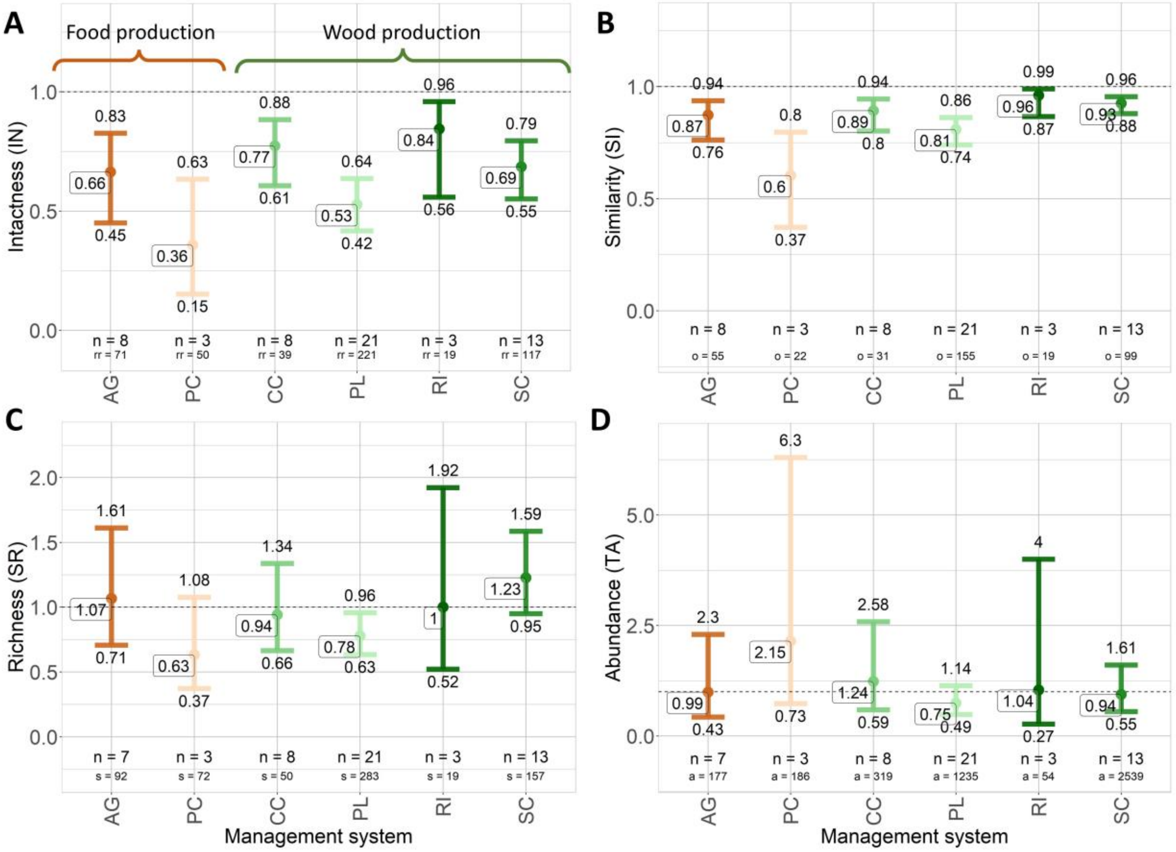


Figure B4. Impacts of forest management on mammal biodiversity, expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish six forest management systems: (i) agroforests (AG), (ii) perennial tree crop plantations (PC), (iii) clear cutting and regrowth (CC), (iv) forest plantations (PL), (v) reduced-impact logging (RI), (vi) selective cutting (SC). The wood production systems are in green and the food production systems in browns. n = number of studies included per management type, rr = number of response values used to compute intactness, o = number of overlapping species between managed and reference forest sites, s = number of species occurring across the managed and reference sites, a = number of abundance values used to compute relative total abundance. The dotted line indicates the reference level biodiversity.

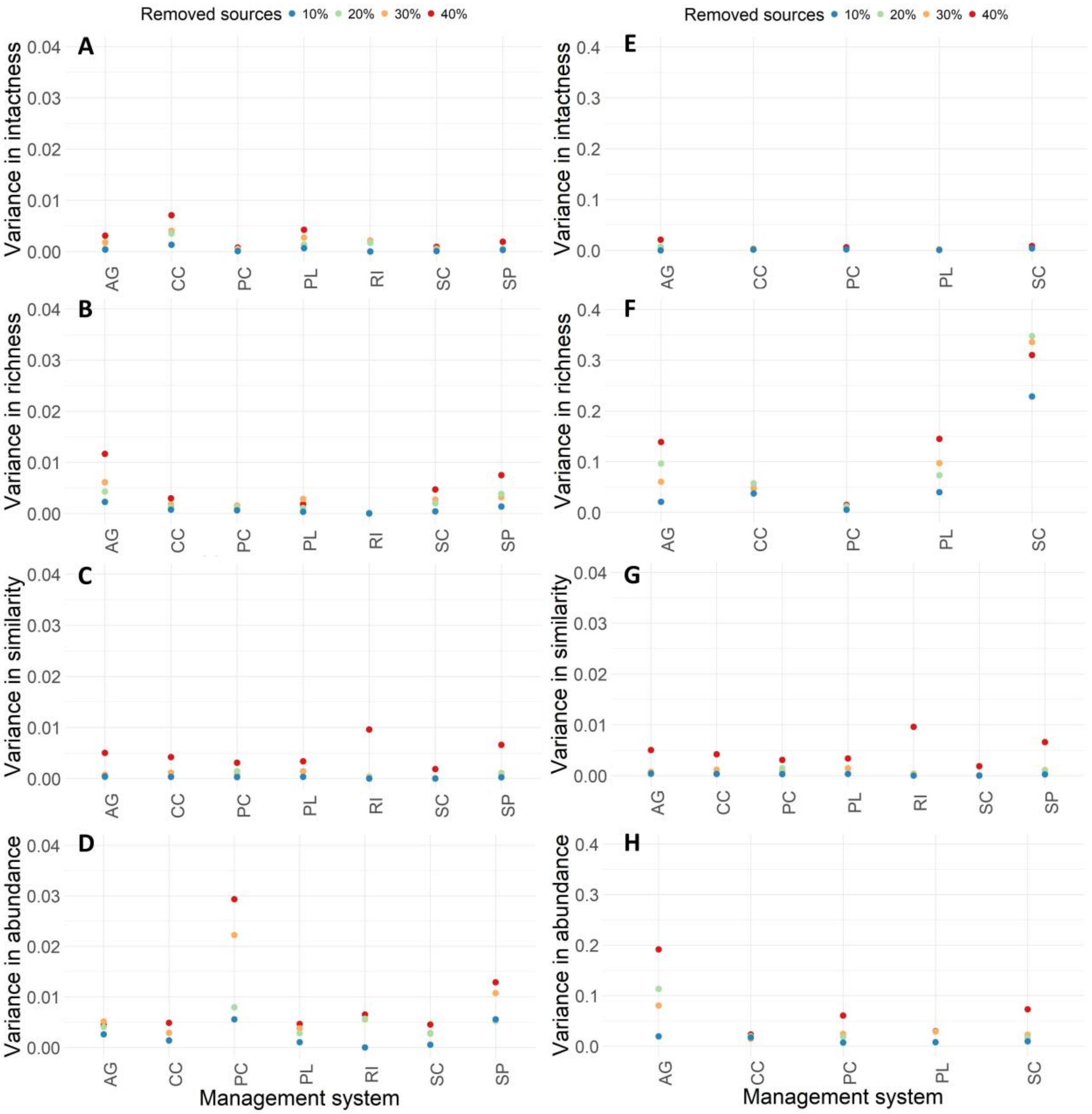


Figure B5. Sensitivity of biodiversity estimates to variations in source selection. We selected respectively 10-40% of the sources from the total source pool per management system twenty times and calculated the variance in outcomes for the four biodiversity indicators. (A-D) Sensitivity of overall animal biodiversity mixed-model estimates to variations in source selection. For animals, larger variances were observed in abundance and richness than in intactness or similarity. (E-H) Sensitivity of plant biodiversity mixed-model estimates to variations on source selection. Plants showed larger variances in biodiversity estimates in response to source selection than animals for all indicators.

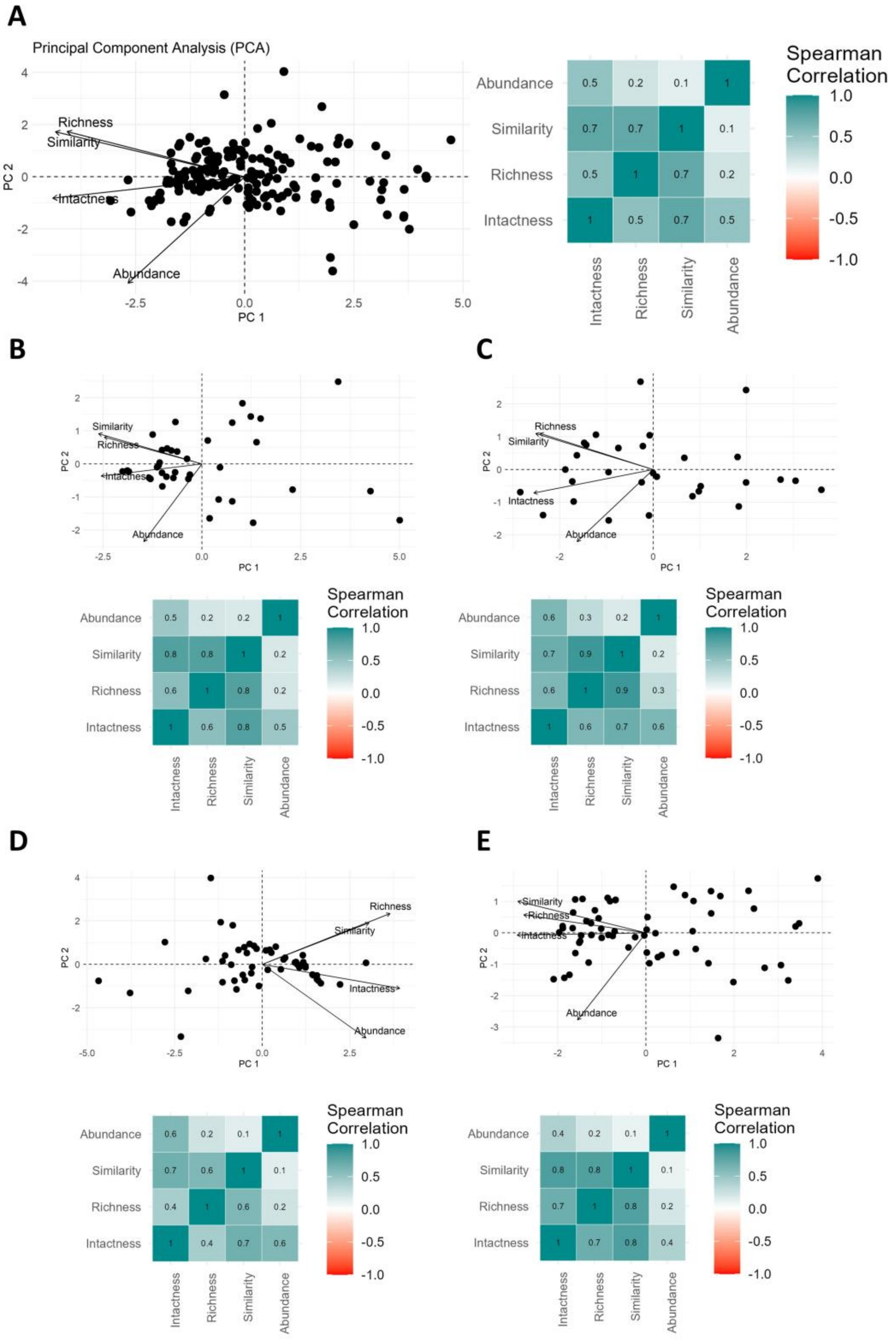


Figure B6. Principal component analyses and Spearman correlation plots for all response ratios of all management systems for animals altogether (A), birds (B), herpetofauna (C), mammals (D) and insects (E). The results indicate that intactness, relative richness, similarity and relative total abundance capture distinct, yet positively correlated aspects of animal biodiversity across taxonomic groups.

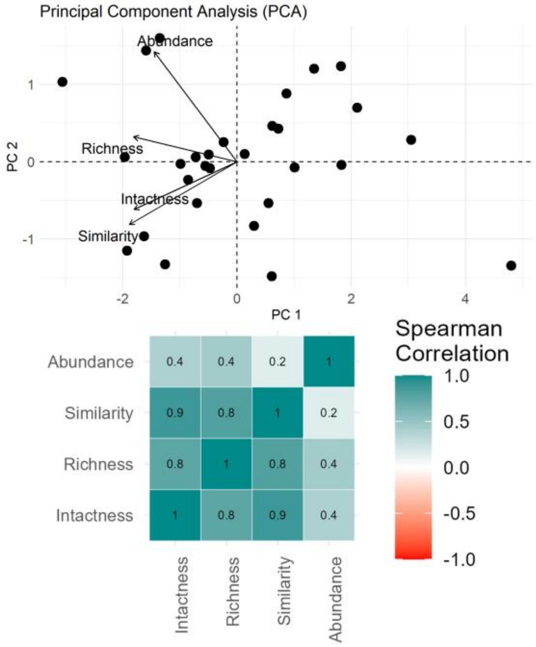


Figure B7. Principal component analyses and Spearman correlation plots for all response ratios of all management systems for plant biodiversity. The results indicate that intactness, relative richness, similarity and relative total abundance capture distinct, yet positively correlated aspects of plant biodiversity.

Appendix C

Search string used to find relevant forest management studies in the Scopus database

TITLE-ABS-KEY (“species diversity” OR biodiversity OR richness OR abundance AND pristine OR primary OR undisturbed OR original OR virgin OR “old-growth” OR “old growth” OR remnant* OR ancient* AND forest* OR woodland AND “selective log*” OR “selective cut*” OR “selective-cut*” OR “clear-cut*” OR “clear cut*” OR “clear fell*” OR “reduced impact logging” OR “reduced-impact logging” OR agroforest* OR silvopastur* OR plantation* OR retention OR silvicultur* OR restoration OR rewilding OR regenerat* OR revegetat* OR abandon* OR "natural succession" OR "natural dispersal" OR reforest* OR afforest* OR reclamat* OR rehabilit* OR reconstruct* OR replace* OR renew* OR transform* OR "re-colonization" OR "re-colonisation" OR "re-establishment" OR conversion OR reintroduct* AND Europe OR Albania OR Andorra OR Armenia OR Austria OR Azerbaijan OR Belarus OR Belgium OR Bosnia OR Herzegovina OR Bulgaria OR Croatia OR Cyprus OR Czechia OR Denmark OR Estonia OR Finland OR France OR Georgia OR Germany OR Greece OR Hungary OR Iceland OR Ireland OR Italy OR Kazakhstan OR Kosovo OR Latvia OR Liechtenstein OR Lithuania OR Luxembourg OR Malta OR Moldova OR Monaco OR Montenegro OR Netherlands OR Macedonia OR Norway OR Poland OR Portugal OR Romania OR Russia OR Serbia OR Slovakia OR Slovenia OR Spain OR Sweden OR Switzerland OR Turkey OR Ukraine OR “United Kingdom” OR Scotland OR England OR Wales OR Czechoslovakia OR Yugoslavia) AND (LIMIT-TO (DOCTYPE, “ar”))

Table C1. Overall biodiversity estimates and confidence intervals (CI) for intactness (IN), relative richness (SR), similarity (SI) and total abundance (TA) (computed by back-transforming the estimates). A significant p-value (<0.05) indicates that there is a significant difference between the impacts of different forest management systems on biodiversity. Coloring of the cells reflect the coloring used in the main text.

	Management systems	Nr. Studies	IN - estimate	IN - lower CI	IN - upper CI	p-value	SR - estimate	SR - lower CI	SR - upper CI	p-value	SI - estimate	SI - lower CI	SI - upper CI	p-value	TA - estimate	TA - lower CI	TA - upper CI	p-value
Animals	Clear cutting (CC)	6	0.37	0.22	0.56	0.40	0.81	0.57	1.15	0.22	0.77	0.59	0.88	2.59E-07	0.70	0.33	1.53	0.24
	Forest plantations (PL)	7	0.44	0.31	0.58		0.85	0.63	1.15		0.77	0.64	0.86		0.77	0.39	1.55	
	Selection systems (SS)	5	0.43	0.24	0.64		0.81	0.55	1.20		0.79	0.63	0.89		0.52	0.21	1.26	
Insects	Clear cutting (CC)	6	0.37	0.23	0.55	0.19	0.81	0.53	1.23	0.34	0.76	0.59	0.87	3.08E-05	0.97	0.52	1.83	0.12
	Forest plantations (PL)	3	0.35	0.19	0.55		0.79	0.48	1.32		0.73	0.52	0.87		0.78	0.37	1.61	
	Selection systems (SS)	5	0.43	0.25	0.63		0.81	0.51	1.30		0.79	0.63	0.89		0.48	0.26	0.90	
Plants	Clear cutting (CC)	3	0.48	0.20	0.78	0.13	0.83	0.31	2.27	0.45	0.72	0.59	0.82	< 2.2e-16	0.35	0.04	2.90	0.07
	Forest plantations (PL)	4	0.26	0.12	0.47		1.69	0.75	3.81		0.61	0.55	0.66		0.89	0.16	4.93	
	Retention forestry (RT)	2	0.34	0.12	0.66		0.83	0.31	2.27		0.68	0.79	0.54		0.89	0.16	4.93	
	Selection systems (SS)	2	0.74	0.30	0.95		1.04	0.25	4.24		0.70	0.51	0.84		1.02	0.05	19.42	

Table C2. Regression coefficients for models with time as an independent variable and a biodiversity indicator as the dependent variable. Biodiversity indicators include biodiversity intactness (IN), calculated as Mean Species Abundance (MSA), richness (SR), expressed as the relative number of species, similarity (SI), calculated as the Sørensen Similarity Index and total abundance (TA), calculated as the response ratio between the summed abundance of all species between management and reference studies. Under the columns Coef$_{Interc}$ and Coef$_{Ind}$ coefficients are provided for the intercept, the ecological driver. SE = Standard error, X^2 = Chi-squared test result, p-value = p-value of Chi-squared test, N = the number of data points in the model, and df = degrees of freedom.

Indicator	**Management system**	**Coef$_{Interc}$**	**Coef$_{Ind}$**	**SE$_{inter}$**	**SE$_{ind}$**	**p-value**	**X^2**	**N**	**df**
Intactness (IN)	Clear cutting (CC)	-0.639	0.008	0.292	0.005	0.09108	2.8552	7	1
	Forest plantations (PL)	-0.686	0.012	0.42	0.0006	< 2.2e-16	327.12	11	1
Richness (SR)	Clear cutting (CC)	-0.38	0.02	0.25	0.023	0.388	0.7453	7	1
	Forest plantations (PL)	-0.317	-0.002	0.174	0.004206	0.6075	0.2639	11	1
Similarity (SI)	Clear cutting (CC)	-0.753953	-0.00233	0.252669	0.00437	0.5947	0.2831	7	1
	Forest plantations (PL)	-1.142354	0.032916	0.411699	0.001888	< 2.2e-16	303.82	11	1
Abundance (TA)	Clear cutting (CC)	-1.07096	0.09456	0.47726	0.0495	0.05609	3.6495	7	1
	Forest plantations (PL)	-0.393022	0.00141	0.221583	0.002554	0.5807	0.305	11	1

Table C3. References from which biodiversity data was derived to assess impacts of forest management on biodiversity in Europe. The sources were found through Scopus.

Source_ID	**Authors**	**Title**	**Year**	**Source title**	**Volume**	**Issue**	**Page start**	**Page end**
63	Ónodi G.; Botta-Dukát Z.; Winkler D.; Rédei T.	Endangered lowland oak forest steppe remnants keep unique bird species richness in Central Hungary	2022	Journal of Forestry Research	33	1	343	355
50	Amo L.; López P.; Martín J.	Natural oak forest vs. ancient pine plantations: Lizard microhabitat use may explain the effects of ancient reforestations on distribution and conservation of Iberian lizards	2007	Biodiversity and Conservation	16	12	3409	3422
152	Cooper A.; McCann T.	Early stage plant species assemblage on harvested sitka spruce plantation over ancient woodland and hedged agricultural grassland	2013	Biology and Environment	113 B	3	1	18
65	Danise T.; Innangi M.; Curcio E.; Fioretto A.	Covariation between plant biodiversity and soil systems in a European beech forest and a black pine plantation: the case of Mount Faito, (Campania, Southern Italy)	2022	Journal of Forestry Research	33	1	239	252
75	Durska E.	Effects of disturbances on scuttle flies (Diptera: Phoridae) in Pine Forests	2013	Biodiversity and Conservation	22	9	1991	2021
142	Felton A.; Hedwall P.O.; Lindbladh M.; Nyberg T.; Felton A.M.; Holmström E.; Wallin I.; Löf M.; Brunet J.	The biodiversity contribution of wood plantations: Contrasting the bird communities of Sweden's protected and production oak forests	2016	Forest Ecology and Management	365		51	60
88	Helle P.	Bird community dynamics in a boreal forest reserve: the importance of large-scale regional trends.	1986	Annales Zoologici Fennici	23	2	157	166
70	Hjältén J.; Joelsson K.; Gibb H.; Work T.;	Biodiversity benefits for saproxylic beetles with uneven-aged silviculture	2017	Forest Ecology and Management	402		37	50

	Löfroth T.; Roberge J.-M.							
101	Ivanov A.V.; Ivanova E.V.; Gamaeva S.V.	Changes in the Diversity of Conifer–Broadleaf Forests of Southern Primorye Resulting from Selective Logging and Fires	2022	Russian Journal of Ecology	53	2	83	90
66	Joelsson K.; Hjältén J.; Gibb H.	Forest management strategy affects saproxylic beetle assemblages: A comparison of even and uneven-aged silviculture using direct and indirect sampling	2018	PLoS ONE	13	4	NA	NA
64	Kim S.; Axelsson E.P.; Girona M.M.; Senior J.K.	Continuous-cover forestry maintains soil fungal communities in Norway spruce dominated boreal forests	2021	Forest Ecology and Management	480		NA	NA
69	Kraut A.; Liira J.; LÃµhmus A.	Beyond a minimum substrate supply: Sustaining saproxylic beetles in semi-natural forest management	2016	Forest Ecology and Management	360		9	19
73	Lõhmus A.; Kull T.	Orchid abundance in hemiboreal forests: Standscale effects of clear-cutting, green-tree retention, and artificial drainage	2011	Canadian Journal of Forest Research	41	6	1352	1358
56	Økland T.; Rydgren K.; Økland R.H.; Storaunet K.O.; Rolstad J.	Variation in environmental conditions, understorey species number, abundance and composition among natural and managed Picea abies forest stands	2003	Forest Ecology and Management	177	3-Jan	17	37
35	Oldén A.; Ovaskainen O.; Kotiaho J.S.; Laaka-Lindberg S.; Halme P.	Bryophyte species richness on retention aspens recovers in time but community structure does not	2014	PLoS ONE	9	4		
159	Otto R.; García-del-Rey E.; Méndez J.; Fernández-Palacios J.M.	Effects of thinning on seed rain, regeneration and understory vegetation in a Pinus canariensis plantation (Tenerife, Canary Islands)	2012	Forest Ecology and Management	280		71	81
72	Pedley S.M.; Martin R.D.; Oxbrough A.; Irwin S.; Kelly T.C.; O'Halloran J.	Commercial spruce plantations support a limited canopy fauna: Evidence from a multi taxa comparison of native and plantation forests	2014	Forest Ecology and Management	314		172	182
26	Roth N.; Doerfler I.; Bässler C.; Blaschke M.; Bussler H.; Gossner M.M.; Heideroth A.; Thorn S.; Weisser W.W.; Müller J.	Decadal effects of landscape-wide enrichment of dead wood on saproxylic organisms in beech forests of different historic management intensity	2019	Diversity and Distributions	25	3	430	441
91	Sferlazza S.; Londi G.; La Mela Veca D.S.; Maetzke F.G.; Vinciguerra S.; Spampinato G.	Close-to-Nature Silviculture to Maintain a Relict Population of White Oak on Etna Volcano (Sicily, Italy): Preliminary Results of a Peculiar Case Study	2023	Plants	12	10		
60	Sippola A.-L.; Siitonen J.; Punttila P.	Beetle diversity in timberline forests: A comparison between old-growth and regeneration areas in Finnish Lapland	2002	Annales Zoologici Fennici	39	1	69	86
62	Summers R.W.; Mavor R.A.; MacLennan A.M.; Rebecca G.W.	The structure of ancient native pinewoods and other woodlands in the Highlands of Scotland	1999	Forest Ecology and Management	119	3-Jan	231	245

Appendix D

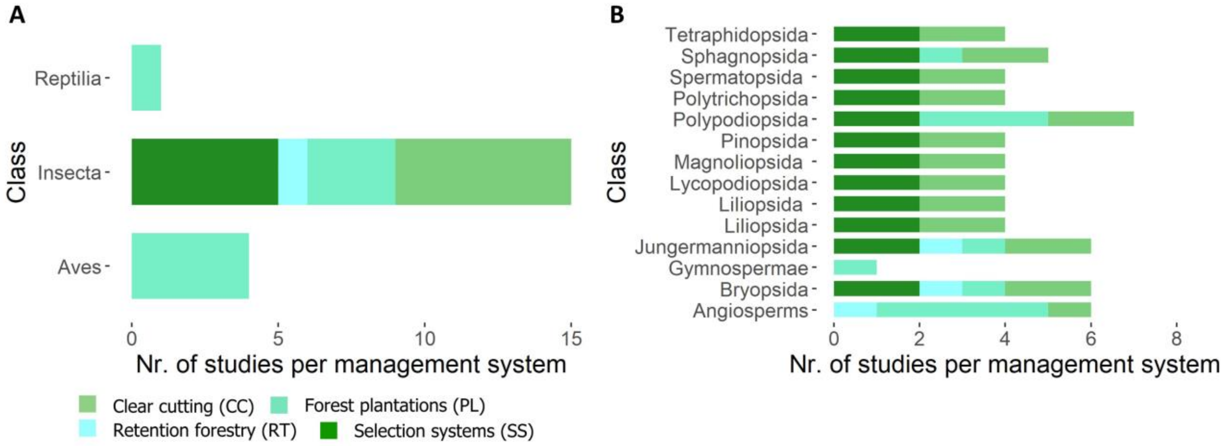


Figure D1. Number of studies carried out per animal and plant class for the different forest management systems. (A) Animal biodiversity studies per class categorized by forest management system. (B) Plant biodiversity studies per class categorized by forest management system.

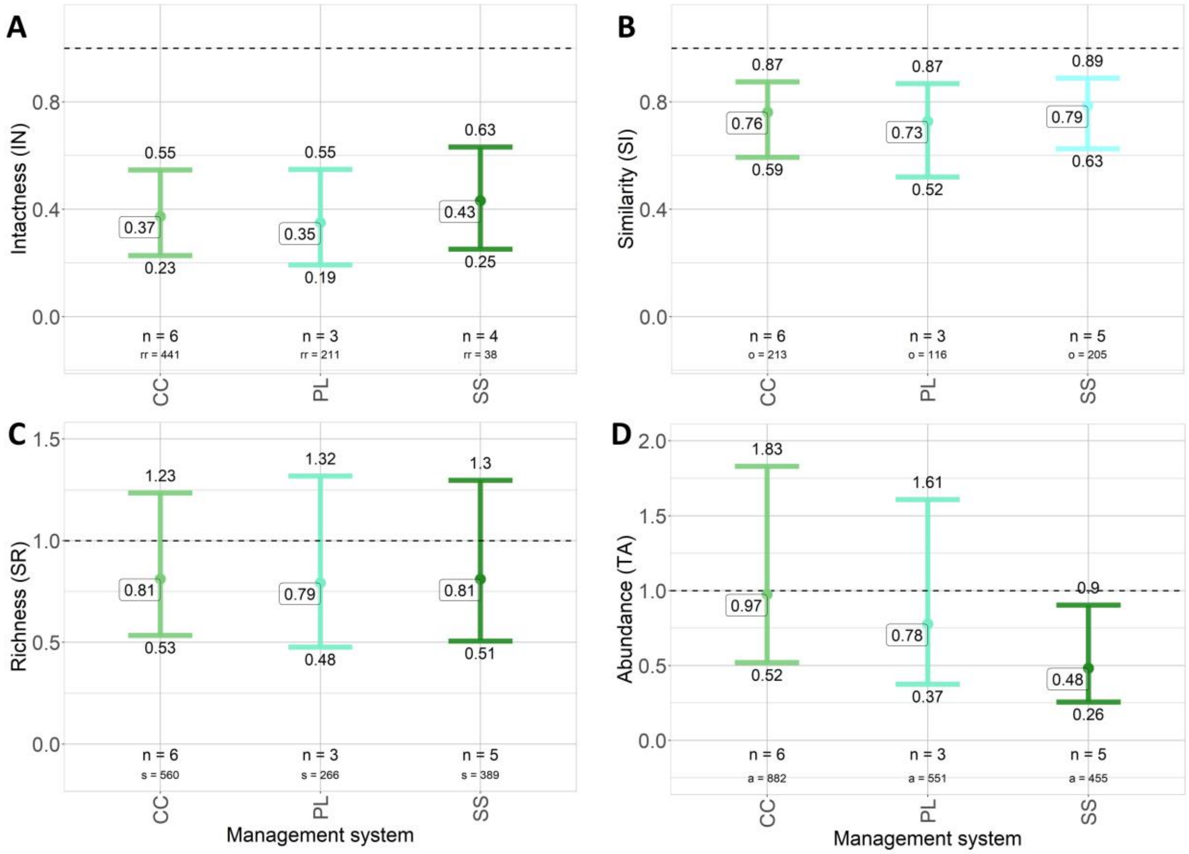


Figure D2. Impacts of forest management on insect biodiversity, expressed in four indicators: intactness (IN) (A), similarity (SI) (B), relative richness (SR) (C), and relative total abundance (TA) (D). We distinguish three forest management systems: (i) clear cutting and regrowth (CC) (AG), (ii) forest plantations (MP), and (iii) selection systems (SS). n = number of studies included per management type, rr = number of response values used to compute intactness, o = number of overlapping species between managed and reference forest sites, s = number of species occurring across the managed and reference sites, a = number of abundance values used to compute relative total abundance.

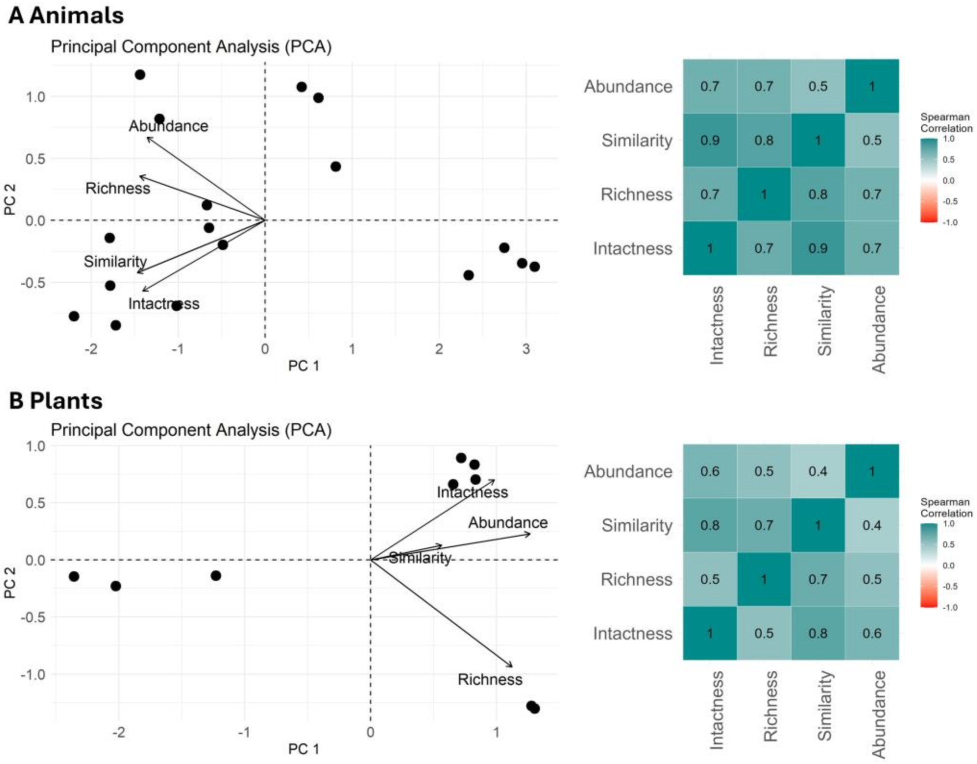


Figure D3. Principal component analyses and Spearman correlation plots for all response ratios of all management systems for animals (A) and plants (B). The results indicate that intactness, relative richness, similarity and relative total abundance capture distinct, yet positively correlated aspects of animal and plant biodiversity.

Appendix E

Table E1. Biodiversity estimates and confidence intervals (CI) for intactness (IN), relative richness (SR), similarity (SI) and total abundance (TA) (computed by back-transforming the estimates) for animal (al taxa at the same time), plant, brid, herpetofauna, insect and mammal species. A significant p-value (<0.05) indicates that there is a significant difference between the biodiversity of different restoration types. Coloring of the cells reflect the coloring used in the main text.

	Restoration types	Nr. Studie	IN - estim	IN - lower	IN - uppe	p-value	SR - estim	SR - lowe	SR - uppe	p-value	SI - estim	SI - lower	SI - upper	p-value	TA - estim	TA - lowe	TA - uppe	p-value
Animals	Reclamation (Rcl)	5	0.37	0.19	0.59	0.6858	1.20	0.80	1.78	0.02464	0.83	0.63	0.93	6.55E-15	0.79	0.42	1.50	0.69
	Reconstruction (Rco)	7	0.48	0.30	0.67		0.68	0.47	0.98		0.85	0.70	0.93		0.67	0.37	1.21	
	Cropland abandonment (CA)	18	0.50	0.38	0.63		0.85	0.69	1.05		0.82	0.71	0.89		0.96	0.67	1.36	
	Pasture abandonment (PA)	12	0.43	0.29	0.58		1.09	0.87	1.38		0.86	0.76	0.93		0.87	0.56	1.33	
	Abandoned slash-and-burn cropland (SB)	4	0.38	0.18	0.63		0.64	0.41	0.98		0.66	0.38	0.86		1.14	0.57	2.28	
Plants	Cropland abandonment (CA)	18	0.31	0.19	0.46	5.23E-03	0.29	0.14	0.62	2.39E-05	0.69	0.36	0.90	0.0550	0.85	0.58	1.27	0.22
	Pasture abandonment (PA)	12	0.37	0.28	0.47		1.28	0.70	2.35		0.71	0.48	0.86		0.82	0.62	1.08	
	Abandoned slash-and-burn cropland (SB)	4	0.48	0.36	0.59		0.58	0.32	1.08		0.73	0.47	0.89		0.98	0.71	1.35	

Table E2. Tukey post-hoc tests with Bonferroni correction outcomes show whether siginficant differences in animal and plant biodiversity occur between forest restoration types (p-value < 0.05).

		Intactness (IN)				Richness (SR)				Similarity (SI)				Total abundance (TA)			
	Combination	**Estimate**	**Std. Error**	**z value**	**p-value**	**Estimate**	**Std. Error**	**z value**	**p-value**	**Estimate**	**Std. Error**	**z value**	**p-value**	**Estimate**	**Std. Error**	**z value**	**p-value**
Animals	PA - CA	-0.30	0.41	-0.73	1.00	0.25	0.16	1.62	1.00	0.34	0.47	0.73	1.00	-0.10	0.28	-0.36	1.00
	Rcl - CA	-0.56	0.53	-1.05	1.00	0.34	0.22	1.54	1.00	0.07	0.61	0.11	1.00	-0.19	0.36	-0.53	1.00
	Rco - CA	-0.08	0.47	-0.18	1.00	-0.22	0.21	-1.06	1.00	0.25	0.55	0.45	1.00	-0.36	0.34	-1.04	1.00
	SB - CA	-0.52	0.58	-0.89	1.00	-0.29	0.24	-1.21	1.00	-0.82	0.66	-1.23	1.00	0.17	0.39	0.44	1.00
	Rcl - PA	-0.26	0.56	-0.46	1.00	0.09	0.23	0.40	1.00	-0.27	0.64	-0.43	1.00	-0.09	0.38	-0.24	1.00
	Rco - PA	0.21	0.50	0.43	1.00	-0.48	0.22	-2.20	0.28	-0.09	0.58	-0.15	1.00	-0.25	0.36	-0.70	1.00
	SB - PA	-0.22	0.61	-0.37	1.00	-0.54	0.24	-2.23	0.26	-1.16	0.70	-1.67	0.96	0.27	0.41	0.67	1.00
	Rco - Rcl	0.47	0.61	0.78	1.00	-0.57	0.27	-2.11	0.35	0.18	0.70	0.26	1.00	-0.16	0.43	-0.37	1.00
	SB - Rcl	0.04	0.70	0.06	1.00	-0.63	0.29	-2.18	0.30	-0.89	0.80	-1.11	1.00	0.36	0.47	0.78	1.00
	SB - Rco	-0.44	0.65	-0.67	1.00	-0.06	0.28	-0.22	1.00	-1.07	0.75	-1.42	1.00	0.53	0.45	1.16	1.00
Plants	PA - CA	0.26	0.38	0.68	1.00	1.47	0.37	3.95	0.00	0.07	0.86	0.08	1.00	-0.04	0.19	-0.22	1.00
	Rcl - CA	0.70	0.40	1.76	0.23	0.68	0.38	1.82	0.21	0.15	0.90	0.17	1.00	0.14	0.20	0.68	1.00
	Rco - CA	0.44	0.32	1.35	0.53	-0.79	0.34	-2.35	0.06	0.08	0.75	0.11	1.00	0.18	0.17	1.06	0.86

Table E3. Regression coefficients for models with time as an fixed effect and a biodiversity indicator as the response variable. Biodiversity indicators include biodiversity intactness (IN), calculated as Mean Species Abundance (MSA), richness (SR), expressed as the relative number of species, similarity (SI), calculated as the Sørensen Similarity Index and total abundance (TA), calculated as the response ratio between the summed abundance of all species between restoration and reference studies. Under the columns $Coef_{Interc}$ and $Coef_{Ind}$ coefficients are provided for the intercept, the ecological driver. SE = Standard error, X^2 = Chi-squared test result, p-value = p-value of Chi-squared test, N = the number of data points in the model, and df = degrees of freedom.

Indicator	Restoration types	$Coef_{Interc}$	$Coef_{Ind}$	SE_{inter}	SE_{ind}	p-value	X^2	N	df
Intactness (IN)	Reclamation (Rcl)	0.040	-0.088	0.359	0.020	0.00001	20.450	16	1
	Reconstruction (Rco)	-1.297	0.125	0.727	0.013	< 2.2e-16	99.502	16	1
	Cropland abandonment (CA)	-0.547	0.006	0.242	0.002	0.00350	8.527	20	1
	Pasture abandonment (PA)	-0.471	0.015	0.414	0.004	0.00006	16.028	15	1
	Abandoned slash-and-burn cropland (SB)	-0.165	-0.017	0.075	0.004	0.00000	22.679	8	1
Richness (SR)	Reclamation (Rcl)	0.125	-0.015	0.131	0.016	0.34340	0.898	16	1
	Reconstruction (Rco)	-0.944	0.049	0.500	0.026	0.05922	3.559	16	1
	Cropland abandonment (CA)	-0.901	0.016	0.191	0.005	0.00112	10.618	20	1
	Pasture abandonment (PA)	-0.420	0.007	0.207	0.005	0.19590	1.673	15	1
	Abandoned slash-and-burn cropland (SB)	-0.345	-0.011	0.100	0.004	0.00265	9.034	8	1
Similarity (SI)	Reclamation (Rcl)	1.173	-0.023	0.236	0.020	0.25730	1.283	16	1
	Reconstruction (Rco)	0.229	0.090	0.727	0.011	< 2.2e-16	71.313	16	1
	Cropland abandonment (CA)	0.169	0.022	0.304	0.002	< 2.2e-16	82.665	20	1
	Pasture abandonment (PA)	0.601	0.010	0.454	0.004	0.00966	6.696	15	1
	Abandoned slash-and-burn cropland (SB)	0.806	-0.010	0.171	0.002	0.00001	20.532	8	1
Abundance (TA)	Reclamation (Rcl)	0.077	-0.057	0.446	0.032	0.07114	3.256	16	1
	Reconstruction (Rco)	-0.434	0.017	0.749	0.065	0.79350	0.069	16	1
	Cropland abandonment (CA)	-0.397	0.009	0.252	0.007	0.19410	1.687	20	1
	Pasture abandonment (PA)	0.387	-0.011	0.305	0.011	0.31440	1.012	15	1
	Abandoned slash-and-burn cropland (SB)	0.023	0.003	0.150	0.006	0.57990	0.306	8	1

Table E4. We used ANOVA to test whether models with both restoration types (R) and taxonomic class (TC) or continent (C) outperformed those with only restoration for animal biodiversity for models estimating intactness, richness, similarity and total abundance, respectively. The models with the lowest BIC values were considered the best performing ones. p-value = significance of the difference between models with only restoration types and with restoration types and taxonomic class or continent as explanatory variables. If p-values were > 0.05 the models did not differ significantly from each other in terms of performance. Significantly best performing models are indicated in bold.

	Indicator	Fixed effects	BIC	p-value
Class	Intactness (IN)	R	-602.16	
		R + TC	-652.77	1.16E-13
	Richness (SR)	R	99.182	
		R + TC	111.624	0.4758
	Similarity (SI)	R	-867.09	
		R + TC	-1031.68	< 2.2e-16
	Abundance (TA)	R	142.62	
		R + TC	155.18	0.4943
Continent	Intactness (IN)	R	-602.16	
		R + C	-597.32	0.09434
	Richness (SR)	**R**	99.182	
		R + C	102.797	0.01499
	Similarity (SI)	**R**	-867.09	
		R + C	-863.29	0.01621
	Abundance (TA)	**R**	142.62	
		R + C	147.94	0.031

Table E5. Sources from which biodiversity data was derived to determine biodiversity levels under forest restoration.

Source ID	**Authors**	**Title**	**Year**	**Journal**	**DOI**	**Volume**	**Issue**	**Page numbers**
Castro-Arellano, I.	Castro-Arellano, I.	Effects of reduced impact logging on bat biodiversity in terra firme forest of lowland Amazonia	2007	Biological Conservation	10.1016/j.biocon.2007.04.025	138	1-2	269-285
Glor, R.E._2001	Glor, R.E.	Lizard diversity and agricultural disturbance in a Caribbean forest landscape	2001	Biodiversity and Conservation	10.1023/A:1016665011087	10	10	711-723
Grimbacher, P.S.	Grimbacher, P.S.	Responses of ground-active beetle assemblages to different styles of reforestation on cleared	2007	Biodiversity and Conservation	10.1007/s10531-006-9146-2	16		2167-2184

		rainforest land						
Raman, T.R.S.	Raman, T.R.S.	Impact of shifting cultivation on diurnal squirrels and primates in Mizoram, northeast India: A preliminary study	1996	Current Science		70		747-750
Vester, H.F.M._2007	Vester, H.F.M.	Land change in the southern Yucatan and Calakmul Biosphere Reserve: effects on habitat and biodiversity	2007	Ecological Applications	10.1890/05-1106	17	4	989-1003
CM1_2012__Hilje	Hilje, B.;Aide, T. M.	Recovery of amphibian species richness and composition in a chronosequence of secondary forests, northeastern Costa Rica	2012	Biological Conservation	10.1016/j.biocon.2011.12.007	146	1	170-176
DI1_2003__Nakamura	Nakamura, Akihiro;Proctor, Heather;Catterall, Carla P.	Using soil and litter arthropods to assess the state of rainforest restoration	2003	Ecological Management & Restoration	10.1046/j.1442-8903.4.s.3.x	4		S20-S28
DI1_2005__Gove	Gove, A. D.;Majer, J. D.;Rico-Gray, V.	Methods for conservation outside of formal reserve systems: the case of ants in the seasonally dry tropics of Veracruz, Mexico	2005	Biological Conservation	10.1016/j.biocon.2005.06.008	126	3	328-338
DI1_2011__Dawson	Dawson, J.;Turner, C.;Pileng, O.;Farmer, A.;McGary, C.;Walsh, C.;Tamblyn, A.;Yosi, C.	Bird communities of the lower Waria Valley, Morobe Province, Papua New Guinea: a comparison between habitat types	2011	Tropical Conservation Science		4	3	317-348
DI1_2011__Neuschulz	Neuschulz, E. L.;Botzat, A.;Farwig, N.	Effects of forest modification on bird community composition and seed	2011	Oikos	10.1111/j.1600-0706.2011.19097.x	120	9	1371-1379

		removal in a heterogeneous landscape in South Africa						
DI1_2012__Reid	Reid, J. L.;Harris, J. B. C.;Zahawi, R. A.	Avian habitat preference in tropical forest restoration in southern Costa Rica	2012	Biotropica	10.1111/j.1744-7429.2011.00814.x	44	3	350-359
DL1_2007__Reis	Reis, Yana T.;Cancello, Eliana M.	Termite (Insecta, Isoptera) richness in primary and secondary Atlantic Forest in southeastern Bahia	2007	Iheringia Serie Zoologia		97	3	229-234
DL1_2009__Woinarski	Woinarski, J. C. Z.;Rankmore, B.;Hill, B.;Griffiths, A. D.;Stewart, A.;Grace, B.	Fauna assemblages in regrowth vegetation in tropical open forests of the Northern Territory, Australia	2009	Wildlife Research	10.1071/wr08128	36	8	675-690
DL1_2010__Bocon	Boçon, R.	Riqueza e abundância de aves em três estágios sucessionais da floresta ombrófila densa submontana, Antonina, Paraná.	2010					
DL1_2010__Silva	Silva, F. A. B.;Costa, C. M. Q.;Moura, R. C.;Farias, A. I.	Study of the dung beetle (Coleoptera: Scarabaeidae) community at two sites: atlantic forest and clear-cut, Pernambuco, Brazil	2010	Environmental Entomology	10.1603/en09180	39	2	359-367
HB1_1999__Vasconcelos	Vasconcelos, H. L.	Effects of forest disturbance on the structure of ground-foraging ant communities in central Amazonia	1999	Biodiversity and Conservation		8	3	409-420
HB1_2008__Sakchoowong	Sakchoowong, Watana;Nomura, Shuhei;Ogata, Kazuo;Chanpaisaeng, Jariya	Diversity of pselaphine beetles (Coleoptera: Staphylinidae: Pselaphinae) in eastern Thailand	2008	Entomological Science	10.1111/j.1479-8298.2008.00281.x	11	3	301-313

HB1_2009__Parry	Parry, L.;Barlow, J.;Peres, C. A.	Hunting for sustainability in tropical secondary forests	2009	Conservation Biology	10.1111/j.1523-1739.2009.01224.x	23	5	1270-1280
HP1_2004__Cleary	Cleary, D. F. R.;Mooers, A. O.;Eichhorn, K. A. O.;van Tol, J.;de Jong, R.;Menken, S. B. J.	Diversity and community composition of butterflies and odonates in an ENSO-induced fire affected habitat mosaic: a case study from East Kalimantan, Indonesia	2004	Oikos	10.1111/j.0030-1299.2004.12219.x	105	2	426-446
JD1_2004__Alcala	Alcala, E. L.;Alcala, A. C.;Dolino, C. N.	Amphibians and reptiles in tropical rainforest fragments on Negros Island, the Philippines	2004	Environmental Conservation	10.1017/s0376892904001407	31	3	254-261
KS1_2002__Vallan	Vallan, D.	Effects of anthropogenic environmental changes on amphibian diversity in the rain forests of eastern Madagascar	2002	Journal of Tropical Ecology	10.1017/s026646740200247x	18		725-742
KS1_2012__Pethiyagoda	Pethiyagoda, Rohan S., Jr.;Manamendra-Arachchi, Kelum	Endangered anurans in a novel forest in the highlands of Sri Lanka	2012	Wildlife Research	10.1071/wr12079	39	7	641-648
KS1_2013__Gould	Gould, Rachelle K.;Pejchar, Liba;Bothwell, Sara G.;Brosi, Berry;Wolny, Stacie;Mendenhall, Chase D.;Daily, Gretchen	Forest restoration and parasitoid wasp communities in montane Hawai'i	2013	PLOS ONE	10.1371/journal.pone.0059356	8	3	e59356
SE1_2011__Arbelaez	ArbelÃ¡ez-CortÃ©s, Enrique;RodrÃ-guez-Correa, Hernando A.;Restrepo-Chica, Manuela	Mixed bird flocks: patterns of activity and species composition in a region of the Central Andes of Colombia	2011	Revista Mexicana de Biodiversidad		82	2	639-651
SE2_2005__Moir	Moir, M. L.;Brennan, K. E. C.;Koch, J. M.;Majer, J. D.;Fletcher, M. J.	Restoration of a forest ecosystem: the effects of vegetation and dispersal capabilities on the reassembly of	2005	Forest Ecology and Management	10.1016/j.foreco.2005.06.012	217	3-Feb	294-306

		plant-dwelling arthropods						
SE2_2009__Craig	Craig, Michael D.;Grigg, Andrew H.;Garkaklis, Mark J.;Hobbs, Richard J.;Grant, Carl D.;Fleming, Patricia A.;Hardy, Giles E. St J.	Does habitat structure influence capture probabilities? A study of reptiles in a eucalypt forest	2009	Wildlife Research	10.1071/wr09014	36	6	509-515
SE2_2012__Craig	Craig, Michael D.;Hardy, Giles E. St J.;Fontaine, Joseph B.;Garkakalis, Mark J.;Grigg, Andrew H.;Grant, Carl D.;Fleming, Patricia A.;Hobbs, Richard J.	Identifying unidirectional and dynamic habitat filters to faunal recolonisation in restored mine-pits	2012	Journal of Applied Ecology	10.1111/j.1365-2664.2012.02152.x	49	4	919-928
SE2_2014a_Craig	Craig, Michael D.;Grigg, Andrew H.;Hobbs, Richard J.;Hardy, Giles E. St J.	Does coarse woody debris density and volume influence the terrestrial vertebrate community in restored bauxite mines?	2014	Forest Ecology and Management	10.1016/j.foreco.2014.01.011	318		142-150
SE2_2014b_Craig	Craig, Michael D.;Stokes, Vicki L.;StJ. Hardy, Giles E.;Hobbs, Richard J.	Edge effects across boundaries between natural and restored jarrah (Eucalyptus marginata) forests in south-western Australia	2015	Austral Ecology	10.1111/aec.12193	40	2	186-197
SH1_2001__Floren	Floren, A.;Freking, A.;Biehl, M.;Linsenmair, K. E.	Anthropogenic disturbance changes the structure of arboreal tropical ant communities	2001	Ecography	10.1111/j.1600-0587.2001.tb00489.x	24	5	547-554
TN1_1994__Roth	Roth, D. S.;Perfecto, I.;Rathcke, B.	The effects of management systems on ground-foraging ant diversity in Costa Rica	1994	Ecological Applications	10.2307/1941947	4	3	423-436
TN1_2005__Fermon	Fermon, H.;Waltert, M.;Vane-Wright, R. I.;Muhlenberg, M.	Forest use and vertical stratification in fruit-feeding butterflies of Sulawesi, Indonesia: impacts for conservation	2005	Biodiversity and Conservation	10.1007/s10531-004-5054-9	14	2	333-350
TN1_2006__Bobo	Bobo, K. S.;Waltert, M.;Fermon,	From forest to farmland: butterfly	2006	Journal of Insect	10.1007/s10841-005-8564-x	10	1	29-42

	H.;Njokagbor, J.;Muhlenberg, M.	diversity and habitat associations along a gradient of forest conversion in Southwestern Cameroon		Conservation				
TN1_2006__Nakagawa	Nakagawa, Michiko;Miguchi, Hideo;Nakashizuka, Tohru	The effects of various forest uses on small mammal communities in Sarawak, Malaysia	2006	Forest Ecology and Management	10.1016/j.foreco.2006.05.006	231	3-Jan	55-62
TN1_2007__Bouyer	Bouyer, J.;Sana, Y.;Samandoulgou, Y.;Cesar, J.;Guerrini, L.;Kabore-Zoungrana, C.;Dulieu, D.	Identification of ecological indicators for monitoring ecosystem health in the trans-boundary W Regional park: a pilot study	2007	Biological Conservation	10.1016/j.biocon.2007.04.001	138	2-Jan	73-88
TN1_2007__CastroLuna	Castro-Luna, A. A.;Sosa, V. J.;Castillo-Campos, G.	Bat diversity and abundance associated with the degree of secondary succession in a tropical forest mosaic in south-eastern Mexico	2007	Animal Conservation	10.1111/j.1469-1795.2007.00097.x	10	2	219-228
TN1_2007__ODea	O'Dea, N.;Whittaker, R. J.	How resilient are Andean montane forest bird communities to habitat degradation?	2007	Biodiversity and Conservation	10.1007/s10531-006-9095-9	16	4	1131-1159
TN1_2009__Fukuda	Fukuda, Daisuke;Tisen, Oswald Braken;Momose, Kuniyasu;Sakai, Shoko	Bat diversity in the vegetation mosaic around a lowland dipterocarp forest of Borneo	2009	Raffles Bulletin of Zoology		57	1	213-221
TN1_2009__Furlani	Furlani, Dario;Ficetola, Gentile Francesco;Colombo, Giorgio;Ugurlucan, Murat;De Bernardi, Fiorenza	Deforestation and the structure of frog communities in the Humedale Terraba-Sierpe, Costa Rica	2009	Zoological Science	10.2108/zsj.26.197	26	3	197-202

Appendix F

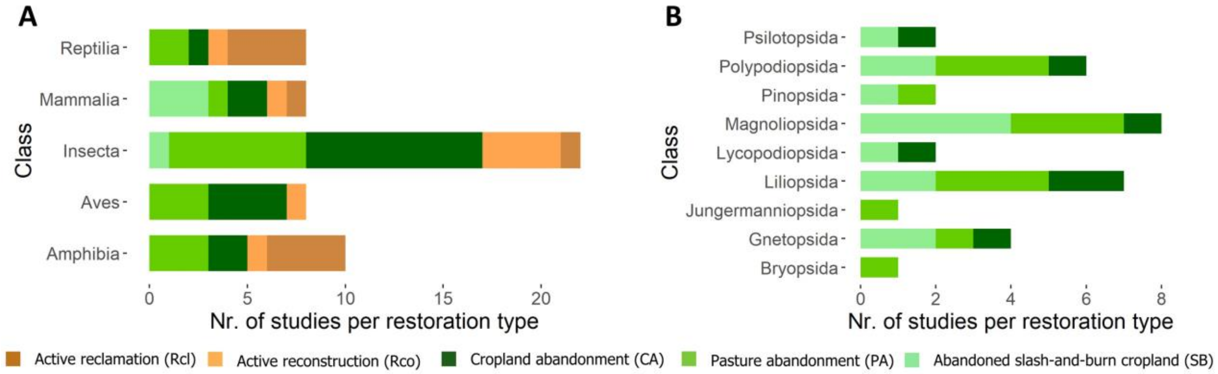
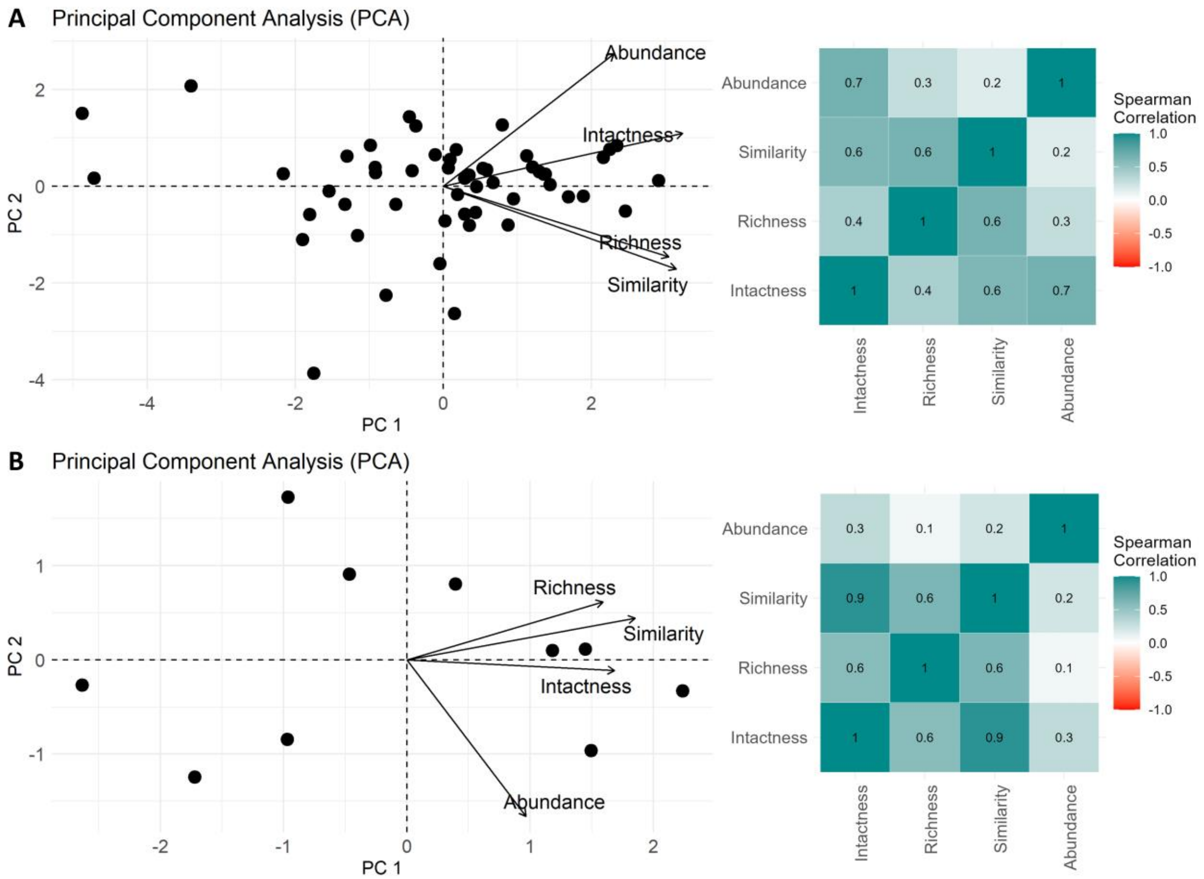


Figure F1. Number of studies carried out per animal and plant class for the different forest restoration types. (A) Animal biodiversity studies per class categorized by forest restoration type. (B) Plant biodiversity studies per class categorized by forest restoration type.



A Principal Component Analysis (PCA)

	Intactness	Richness	Similarity	Abundance
Abundance	0.7	0.3	0.2	1
Similarity	0.6	0.6	1	0.2
Richness	0.4	1	0.6	0.3
Intactness	1	0.4	0.6	0.7

B Principal Component Analysis (PCA)

	Intactness	Richness	Similarity	Abundance
Abundance	0.3	0.1	0.2	1
Similarity	0.9	0.6	1	0.2
Richness	0.6	1	0.6	0.1
Intactness	1	0.6	0.9	0.3

Figure F2. Principal component analyses and Spearman correlation plots for all response ratios of all restoration types for animals (A) and plants (B). The results indicate that intactness, relative richness, similarity and relative total abundance capture distinct, yet all positively correlated aspects of animal and plant biodiversity.

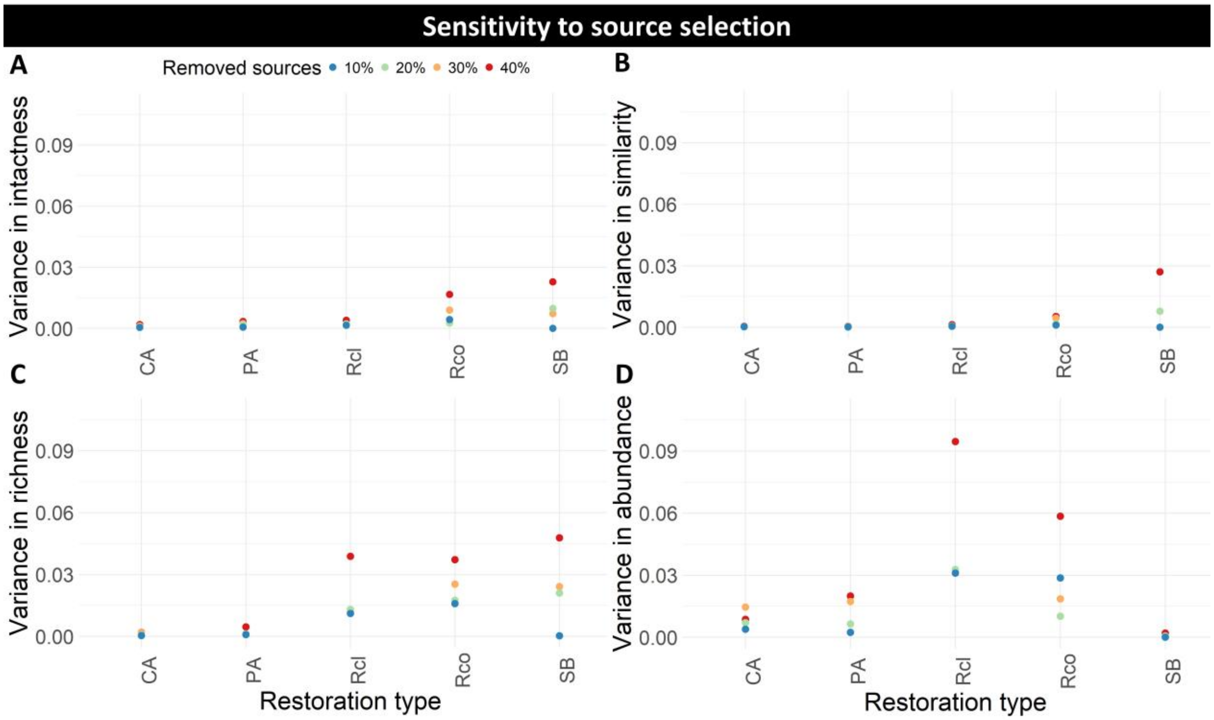


Figure F3. Sensitivity of animal biodiversity estimates to variations in source selection. We selected respectively 10-40% of the sources from the total source pool per restoration type twenty times and calculated the variance in outcomes for the four biodiversity indicators. For animals, larger variances were observed in abundance and richness than in intactness or similarity. We were unable to conduct the sensitivity analysis for plants because of insufficient data.

Appendix G

Search string used to find relevant forest restoration studies in the Scopus database

TITLE-ABS-KEY (“species diversity” OR biodiversity OR richness OR abundance AND pristine OR primary OR undisturbed OR original OR virgin OR “old-growth” OR “old growth” OR remnant* OR ancient* AND forest* OR woodland AND “selective log*” OR “selective cut*” OR “selective-cut*” OR “clear-cut*” OR “clear cut*” OR “clear fell*” OR “reduced impact logging” OR “reduced-impact logging” OR agroforest* OR silvopastur* OR plantation* OR retention OR silvicultur* OR restoration OR rewilding OR regenerat* OR revegetat* OR abandon* OR "natural succession" OR "natural dispersal" OR reforest* OR afforest* OR reclamat* OR rehabilit* OR reconstruct* OR replace* OR renew* OR transform* OR "re-colonization" OR "re-colonisation" OR "re-establishment" OR conversion OR reintroduct* AND Europe OR Albania OR Andorra OR Armenia OR Austria OR Azerbaijan OR Belarus OR Belgium OR Bosnia OR Herzegovina OR Bulgaria OR Croatia OR Cyprus OR Czechia OR Denmark OR Estonia OR Finland OR France OR Georgia OR Germany OR Greece OR Hungary OR Iceland OR Ireland OR Italy OR Kazakhstan OR Kosovo OR Latvia OR Liechtenstein OR Lithuania OR Luxembourg OR Malta OR Moldova OR Monaco OR Montenegro OR Netherlands OR Macedonia OR Norway OR Poland OR Portugal OR Romania OR Russia OR Serbia OR Slovakia OR Slovenia OR Spain OR Sweden OR Switzerland OR Turkey OR Ukraine OR “United Kingdom” OR Scotland OR England OR Wales OR Czechoslovakia OR Yugoslavia) AND (LIMIT-TO (DOCTYPE, “ar”))

Table G1. Overall biodiversity estimates and confidence intervals (CI) for intactness (IN), relative richness (SR), similarity (SI) and total abundance (TA) (computed by back-transforming the estimates). A significant p-value (<0.05) indicates that there is a significant difference between the impacts of different forest restoration types on biodiversity. Coloring of the cells reflect the coloring used in the main paper.

	Restoration type	Nr. Studies	IN - estimate	IN - lower CI	IN - upper CI	p-value	SR - estimate	SR - lower CI	SR - upper CI	p-value	SI - estimate	SI - lower CI	SI - upper CI	p-value	TA - estimate	TA - lower CI	TA - upper CI	p-value
Animals	Active restoration	7	0.53	0.35	0.71	0.75	0.85	0.45	1.60	0.45	0.84	0.68	0.93	5.02E-04	0.79	0.63	0.98	0.01
	Passive restoration	4	0.41	0.21	0.66		0.67	0.29	1.55		0.74	0.46	0.91		0.77	0.55	1.09	
Plants	Active restoration	18	0.59	0.44	0.73	0.12	1.27	1.07	1.49	0.01	0.93	0.86	0.97	< 2.2e-16	0.69	0.54	0.88	0.00
	Passive restoration	5	0.65	0.47	0.79		1.03	0.76	1.39		0.91	0.86	0.94		1.55	0.94	2.55	

Table G2. Regression coefficients for models with time as an independent variable and a biodiversity indicator as the dependent variable. Biodiversity indicators include biodiversity intactness (IN), calculated as Mean Species Abundance (MSA), richness (SR), expressed as the relative number of species, similarity (SI), calculated as the Sørensen Similarity Index and total abundance (TA), calculated as the response ratio between the summed abundance of all species between restoration and reference studies. Under the columns $\text{Coef}_{\text{Interc}}$ and Coef_{Ind} coefficients are provided for the intercept, the ecological driver. SE = Standard error, X^2 = Chi-squared test result, p-value = p-value of Chi-squared test, N = the number of data points in the model, and df = degrees of freedom.

Taxa	**Indicator**	**Restoration type**	**$\text{Coef}_{\text{Interc}}$**	**Coef_{Ind}**	**SE_{inter}**	**SE_{ind}**	**p-value**	**X^2**	**N**	**df**
Animals	Intactness (IN)	Active restoration	0.3485	-0.0036	0.2326	0.0008	1.14E-05	19.2570	15	1
		Passive restoration	-0.2811	0.0049	0.7868	0.0014	0.0004	12.5520	11	1
	Richness (SR)	Active restoration	-0.0528	-0.0047	0.2676	0.0026	0.0764	3.1390	15	1
		Passive restoration	-0.4302	0.0016	0.3284	0.0045	0.7143	0.1340	11	1
	Similarity (SI)	Active restoration	0.6965	-0.0034	0.4062	0.0009	0.0003	13.2740	15	1
		Passive restoration	-1.2023	0.0047	0.4995	0.0012	0.0001	15.9850	11	1
	Abundance (TA)	Active restoration	-0.0596	-0.0025	0.1445	0.0017	0.1328	2.2595	15	1
		Passive restoration	-0.2013	0.0055	0.2587	0.0035	0.1185	2.4375	11	1
Plants	Intactness (IN)	Active restoration	0.9890	-0.0072	0.4915	0.0026	5.23E-03	7.7998	32	1
		Passive restoration	-0.1397	0.0101	0.6924	0.0056	0.0737	3.1990	11	1
	Richness (SR)	Active restoration	0.1194	-0.0025	0.1694	0.0016	0.1273	2.3254	32	1
		Passive restoration	-0.2658	0.0023	0.0909	0.0007	0.0016	9.9555	11	1
	Similarity (SI)	Active restoration	-0.0950	-0.0054	0.2848	0.0023	0.0210	5.3273	32	1
		Passive restoration	0.2428	-0.0052	0.3943	0.0032	0.1057	2.6169	11	1
	Abundance (TA)	Active restoration	-0.2703	0.0004	0.1345	0.0016	0.8127	0.0562	32	1
		Passive restoration	-0.2649	0.0064	0.8035	0.0065	0.3259	0.9651	11	1

Table G3. Sources from which biodiversity data was derived to assess biodiversity levels in restoration forests in Europe. The sources were found through Scopus.

Source ID	**Authors**	**Title**	**Year**	**Source title**	**Volume**	**Issue**	**Page start**	**Page end**	**DOI**
1	Fuentes-Montemayor E.; Watts K.; Sansum P.; Scott W.; Park K.J.	Moth community responses to woodland creation: The influence of woodland age, patch characteristics and landscape attributes	2022	Diversity and Distributions	28	9	1993	2007	10.1111/ddi.13599
2	Šipek M.; Ravnjak T.; Šajna N.	Understorey species distinguish late successional and ancient forests after decades of minimum human intervention: A case study from Slovenia	2023	Forest Ecosystems	10				10.1016/j.fecs.2023.100096
7	Pass E.; Kont R.; Lõhmus A.	Spruce (Picea abies L.) planting leads post-clearcut bird assemblages to a novel successional pathway—a comparative study in hemiboreal mixed forests	2022	Annals of Forest Science	79	1			10.1186/s13595-022-01138-8
11	Alsila T.; Elo M.; Hakkari	Effects of habitat restoration on	2021	Restoration Ecology	29	1			10.1111/rec.13304

	T.; Kotiaho J.S.	peatland bird communities							
13	Mannu R.; Pantini P.; Sassu A.; Verdinelli M.	A multidiversity approach to investigate the impact of mining exploitation on spider diversity in the abandoned mine district of Montevecchio-Ingurtosu (Sardinia, Italy)	2020	Environmental Science and Pollution Research	27	26	32615	32627	10.1007/s11356-020-09553-y
14	Morel L.; Barbe L.; Jung V.; Clément B.; Schnitzler A.; Ysnel F.	Passive rewilding may (also) restore phylogenetically rich and functionally resilient forest plant communities	2020	Ecological Applications	30	1			10.1002/eap.2007
21	Versluijs M.; Eggers S.; Hjältén J.; Löfroth T.; Roberge J.-M.	Ecological restoration in boreal forest modifies the structure of bird assemblages	2017	Forest Ecology and Management	401		75	88	10.1016/j.foreco.2017.06.055
24	Haapalehto T.; Juutinen R.; Kareksela S.; Kuitunen M.; Tahvanainen T.; Vuori H.; Kotiaho J.S.	Recovery of plant communities after ecological restoration of forestry-drained peatlands	2017	Ecology and Evolution	7	19	7848	7858	10.1002/ece3.3243
27	Horváth G.F.; Tóth D.	Abundance of bank vole (Myodes glareolus Schreb.) as an indicative factor of different forest structure and management in the Drava plain region	2018	Sumarski List	142	3-Apr	161	170	10.31298/sl.142.3-4.4
28	Táborská M.; Procházková J.; Lengyel A.; Vrška T.; Hort L.; Ódor P.	Wood-inhabiting bryophyte communities are influenced by different management intensities in the past	2017	Biodiversity and Conservation	26	12	2893	2909	10.1007/s10531-017-1395-8
30	Paal T.; Kütt L.; Lõhmus K.; Liira J.	Both spatiotemporal connectivity and habitat quality limit the immigration of forest plants into wooded corridors	2017	Plant Ecology	218	4	417	431	10.1007/s11258-017-0700-7
31	Naaf T.; Kolk J.	Colonization credit of post-agricultural forest patches in NE Germany remains 130-230years after reforestation	2015	Biological Conservation	182		155	163	10.1016/j.biocon.2014.12.002
32	Palo A.; Ivask M.; Liira J.	Biodiversity composition reflects the history of ancient semi-natural woodland	2013	Ecological Indicators	34		336	344	10.1016/j.ecolind.2013.05.020

		and forest habitats - Compilation of an indicator complex for restoration practice							
34	Matuszkiewicz J.M.; Kowalska A.; Kozłowska A.; Roo-Zielińska E.; Solon J.	Differences in plant-species composition, richness and community structure in ancient and post-agricultural pine forests in central Poland	2013	Forest Ecology and Management	310		567	576	10.1016/j.foreco.2013.08.060
38	Vojta J.; Drhovská L.	Are abandoned wooded pastures suitable refugia for forest species?	2012	Journal of Vegetation Science	23	5	880	891	10.1111/j.1654-1103.2012.01399.x
47	Napierała A.; Błoszyk J.; Bruin J.	Communities of uropodine mites (Acari: Mesostigmata) in selected oak-hornbeam forests of the Wielkopolska region (Poland)	2009	Experimental and Applied Acarology	49	4	291	303	10.1007/s10493-009-9262-2
49	Brunet J.; Isacsson G.	Restoration of beech forest for saproxylic beetles-effects of habitat fragmentation and substrate density on species diversity and distribution	2009	Biodiversity and Conservation	18	9	2387	2404	10.1007/s10531-009-9595-5
51	Sciama D.; Augusto L.; Dupouey J.-L.; Gonzalez M.; Moares Domínguez C.	Floristic and ecological differences between recent and ancient forests growing on non-acidic soils	2009	Forest Ecology and Management	258	5	600	608	10.1016/j.foreco.2009.04.027
53	De Keersmaeker L.; Martens L.; Verheyen K.; Hermy M.; De Schrijver A.; Lust N.	Impact of soil fertility and insolation on diversity of herbaceous woodland species colonizing afforestations in Muizen forest (Belgium)	2004	Forest Ecology and Management	188	3-Jan	291	304	10.1016/j.foreco.2003.07.025
99	Fuentes-Montemayor E.; Park K.J.; Cordts K.; Watts K.	The long-Term development of temperate woodland creation sites: From tree saplings to mature woodlands	2022	Forestry	95	1	28	37	10.1093/forestry/cpab027
119	Trilikauskas L.A.; Novgorodova T.A.	Spiders and harvestmen (Arachnida: Aranei, Opiliones) of the revegetating ash dumps of a combined heat and power plant in Novosibirsk	2020	Arthropoda Selecta	29	4	499	510	10.15298/arthsel.29.4.14

		(Russia, West Siberia)							
139	Noreika N.; Kotiaho J.S.; Penttinen J.; Punttila P.; Vuori A.; Pajunen T.; Autio O.; Loukola O.J.; Kotze D.J.	Rapid recovery of invertebrate communities after ecological restoration of boreal mires	2015	Restoration Ecology	23	5	566	579	10.1111/rec.12237
140	Lacasella F.; Zapparoli M.	Effects of grasslands and conifer reforestations on centipedes (Chilopoda): Barriers, semi-permeable matrices or secondary habitats?	2015	Insect Conservation and Diversity	8	6	525	537	10.1111/icad.12133
141	Punttila P.; Autio O.; Kotiaho J.S.; Kotze D.J.; Loukola O.J.; Noreika N.; Vuori A.; Vepsäläinen K.	The effects of drainage and restoration of pine mires on habitat structure, vegetation and ants	2016	Silva Fennica	50	2			10.14214/sf.1462
157	Dekoninck W.; Hendrickx F.; Dethier M.; Maelfait J.	Forest Succession Endangers the Special Ant Fauna of Abandoned Quarries along the River Meuse (Wallonia, Belgium)	2010	Restoration Ecology	18	5	681	690	10.1111/j.1526-100X.2008.00499.x
160	Escribano-Avila G.; Sanz-Pérez V.; Pías B.; Virgós E.; Escudero A.; Valladares F.	Colonization of Abandoned Land by Juniperus thurifera Is Mediated by the Interaction of a Diverse Dispersal Assemblage and Environmental Heterogeneity	2012	PLoS ONE	7	10			10.1371/journal.pone.0046993
169	Bossuyt B.; Heyn M.; Hermy M.	Seed bank and vegetation composition of forest stands of varying age in central Belgium: Consequences for regeneration of ancient forest vegetation	2002	Plant Ecology	162	1	33	48	10.1023/A:1020391430072
179	Dzwonko Z.; Gawroński S.	The role of woodland fragments, soil types, and dominant species in secondary succession on the western Carpathian foothills	1994	Vegetatio	111	2	149	160	10.1007/BF00040334

180	Bossuyt B.; Hermy M.; Deckers J.	Migration of herbaceous plant species across ancient-recent forest ecotones in central Belgium	1999	Journal of Ecology	87	4	629	638	10.1046/j.1365-2745.1999.00379.x
15	Blandino C.; Fernández-Pascual E.; Bueno Á.; Pritchard H.W.	Plant reproductive traits in old and recently-restored temperate forest understories	2021	Forest Ecology and Management	496				10.1016/j.foreco.2021.119385

Appendix H

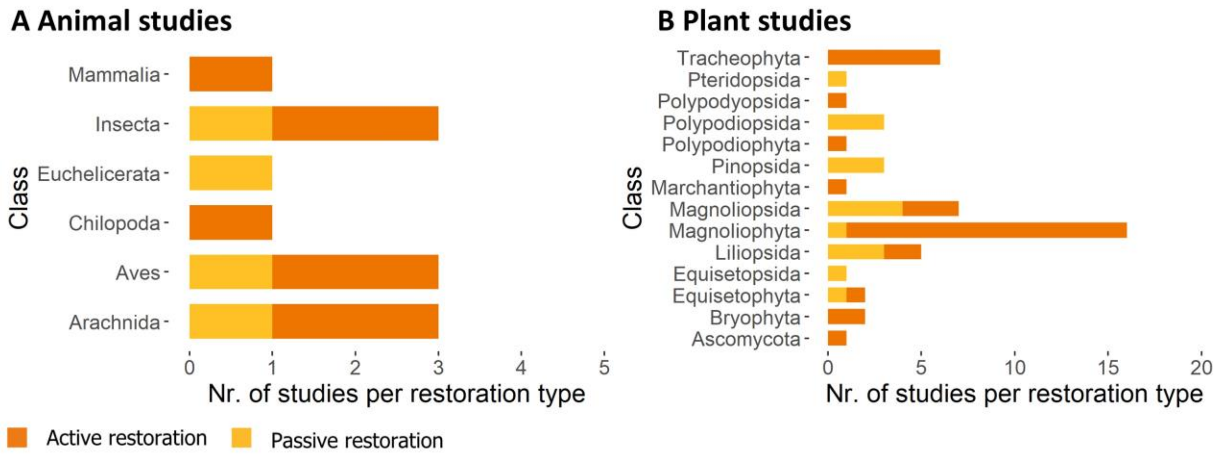


Figure H1. Number of studies carried out per animal and plant class for the different forest restoration types. (A) Animal biodiversity studies per class categorized by forest restoration type. (B) Plant biodiversity studies per class categorized by forest restoration type.

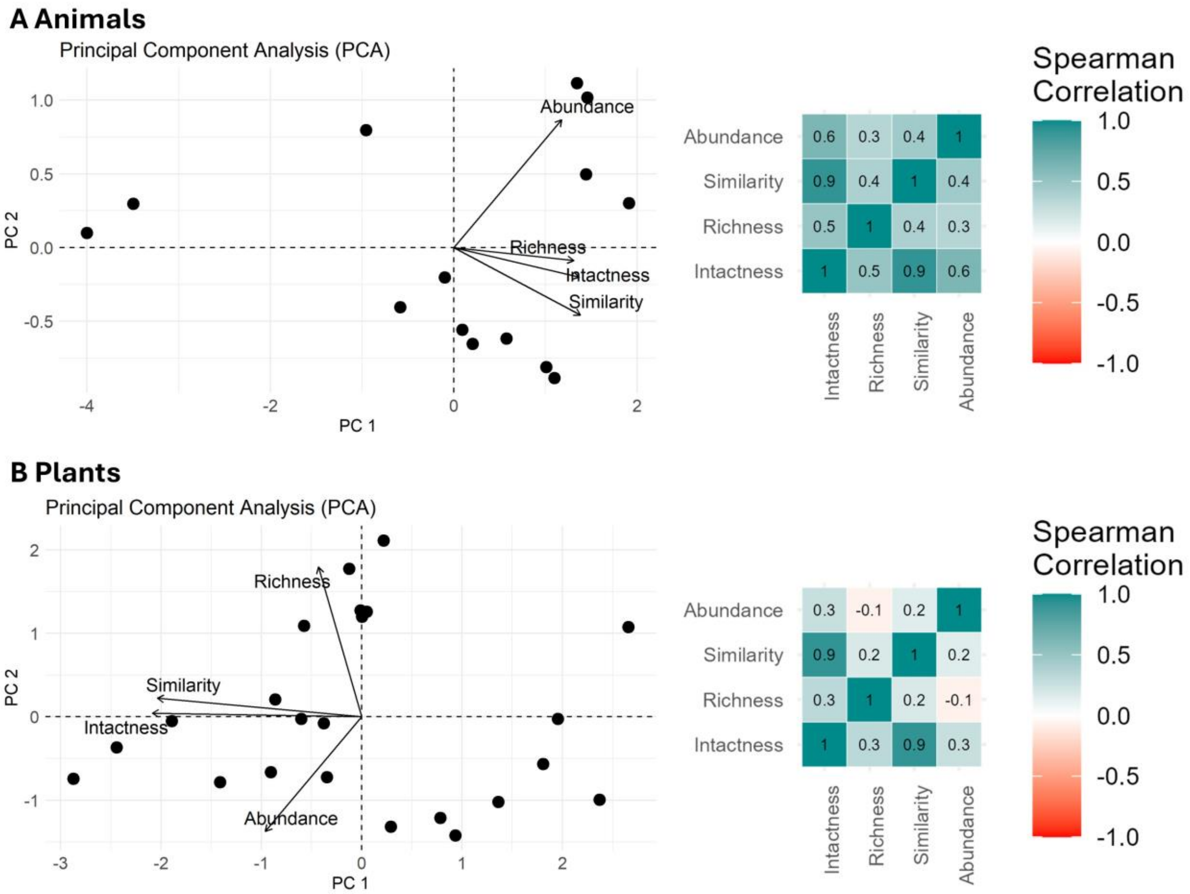


Figure H2. Principal component analyses and Spearman correlation plots for all response ratios of all restoration types for animals (A) and plants (B). The results indicate that intactness, relative richness, similarity and relative total abundance capture distinct, yet mostly positively correlated aspects of animal and plant biodiversity.

Appendix I

<COPIED FROM THE MAIN PAPER AND SUPLEMENTARY INFORMATION of Schipper et al., 2019 (GCB)>

1 General approach

The core of the GLOBIO model is a set of quantitative relationships that assess the impacts of anthropogenic pressures on biodiversity. Pressures included in GLOBIO are climate change, land use, roads, atmospheric nitrogen deposition and hunting. Impacts are quantified based on the mean species abundance, MSA, metric, which is an indicator of local biodiversity intactness. The metric is quantified based on data that describe changes in community composition in relation to particular pressures. MSA values are retrieved by dividing the abundance of each species found in relation to a given pressure level by its abundance found in an undisturbed situation within the same study, truncating the values at 1, and then calculating the arithmetic mean over all species present in the reference situation (Alkemade et al. 2009; Santini et al. 2017). Increases in individual species abundance from reference to impacted situation are truncated to avoid the indicator being inflated by opportunistic or generalist species that benefit from habitat disturbance. The GLOBIO model combines the pressure–impact relationships with maps of the pressures (i.e. climate change, land use, roads, atmospheric nitrogen deposition and hunters' access points) resulting in maps with impact-specific MSA values. Fragmentation impacts are quantified based on the size of natural habitat patches, calculated based on the land use and roads maps. Maps of impact-specific MSA values are then combined in order to calculate an overall MSA. GLOBIO includes two approaches to integrate MSA values across the impacts, whereby the choice depends on the land use. If the land use impact is expected to dominate over other impacts, the overall MSA value equals the MSA value of the land-use class. For example, it is assumed that there are no additional impacts of atmospheric nitrogen deposition in croplands, which are typically fertilized, and that there are no additional impacts or roads within urban areas. Alternatively, it is assumed that (a) pressures act independently, that is, an organism is lost from the community if at least one of the pressures is higher than its tolerance limit; (b) organisms' tolerances to different pressures are uncorrelated and (c) pressure–impact relationships are based on representative, random samples of the community. Furthermore, the contribution of each pressure to the MSA loss in a given grid cell is calculated relative to the sum of the losses across all pressures and then rescaled to the total loss in the grid cell. Subsequently, the cell-specific MSA losses and pressure contributions can be aggregated across the grid cells 1 to n in any larger region of interest, calculated as mean value weighted by the area of the grid cells.

2 Impacts of climate change

To establish pressure-impact relationships for climate change, we used the database recently published by Nunez et al. (2019), which contains information on local biodiversity intactness retrieved from 97 bioclimatic envelope modelling studies. From this dataset we selected data on the fraction of remaining species (FRS), which represents the ratio between the number of species remaining after climate change and the original number of species in a location (typically a grid cell in the underlying bioclimatic envelope modelling study). Thus, FRS is a relative index between 0 (no original species present) and 1 (all original species present), which we considered a proxy for MSA. Per study, region and climate scenario, FRS values were provided as average across the locations within a region (Nunez et al. 2019). In total, 135 FRS values from 31 studies were available for plants and 141 FRS values from 19 studies for warm-blooded vertebrates. Because the underlying studies differed in the climatic variables selected, we used the global mean temperature increase corresponding with each climate scenario as common explanatory variable, following Nunez et al. (2019). Per taxonomic group, we established a mixed beta regression model to relate FRS to the global mean temperature increase (°C), using study as random intercept. Because responses to climate change are expected to differ between tropical, temperate and Arctic regions (Deutsch et al. 2008; Post et al. 2009), we tested for an interaction with climate zone. To that end, we assigned each observation to either the tropics or to non-tropics (because of a lack of data to distinguish Arctic regions separately), delineating the tropics as the biomes 'Tropical and subtropical moist broadleaf forests', 'Tropical and subtropical dry broadleaf forests', 'Tropical and subtropical coniferous forests', and 'Tropical and subtropical grasslands, savannas, and shrublands', as distinguished by Dinerstein et al. (2017). However, the overall model had a lower BIC value.

3 Impacts of atmospheric nitrogen deposition

We retrieved a pressure-impact relationship for nitrogen deposition ($kg \cdot ha \cdot yr^{-1}$) based on a data from field experiments that quantified responses of natural and semi-natural terrestrial plant communities to nitrogen addition. We used the database published by Midolo et al. (2019), which contains observations from 115 nitrogen addition experiments reported in 85 studies. Of these, 37 studies reported ratios of individual species abundance at a given level of N addition relative to the abundance in a control plot, which we used to calculate 89 MSA values. We used these MSA values as input to a mixed beta regression model with dataset within study as nested random intercept and the $\log_{10}$-transformed amount of nitrogen addition ($kg \cdot ha \cdot yr^{-1}$) as fixed effect. We tested for interactions with mean annual temperature, following Midolo et al. (2019). However, the model including only nitrogen addition had the lowest BIC value.

4 Impacts of land use

We quantified relationships between MSA and land use based on monitoring data from the 2016 release of the PREDICTS database (https://data.nhm.ac.uk/dataset/the-2016-release-of-the-predicts-database), which contains spatial comparisons of species' assemblages in particular land-use types and intensity levels (Hudson et al. 2017). PREDICTS categorizes each record into one of nine predominant land-use (LU) types: 'Primary vegetation', 'Mature secondary vegetation', 'Intermediate secondary vegetation', 'Young secondary vegetation', 'Secondary vegetation of indeterminate age', 'Plantation forest', 'Cropland', 'Pasture', or 'Urban'. Furthermore, each record falls into one of four land-use intensities (UI): 'Minimal use', 'Light use', 'Intense use', and 'Cannot decide'. Details on the land-use classification are provided by Hudson et al. (2014). We extracted observations from studies that reported abundances of plant and warm-blooded vertebrate species in 'Primary vegetation – minimal use', which we used as controls, and at least one other land use type. We then assigned each record from the selected studies a GLOBIO land-use category ('Secondary vegetation', 'Plantation', 'Cropland - Minimal use', 'Cropland - Intense use', 'Pasture - Minimal use', 'Pasture - Intense use' and 'Urban'). We excluded observations from PREDICTS land-use categories that we could not match with a GLOBIO category.

Typically, multiple sites were sampled in a given LU category within a dataset. We averaged the abundance records per species, dataset and LU category over the sample sites. We then calculated for each species within a LU type within a dataset an abundance ratio by dividing its abundance in the disturbed habitat by its abundance in the reference site, and we retrieved MSA values by averaging the truncated abundance ratios per LU type, species group, and dataset. This resulted in 55 MSA values from 32 studies for plants and 85 MSA values from 48 studies for warm-blooded vertebrates, which we used as input to mixed beta regression models with dataset within study as nested random intercept and the GLOBIO land-use classes as fixed effects.

5 Impacts of habitat fragmentation

To quantify the impact relationship for habitat fragmentation, we used data from the PREDICTs database (i.e., the same database as used for quantifying the land-use impacts). Because fragmentation impacts in GLOBIO are confined to warm-blooded vertebrates and natural vegetation, we selected from PREDICTS observations for birds and mammals from the categories "Primary vegetation – minimal use" and "Mature secondary vegetation – minimal use" with quantitative information available on the patch size. Some patches had size denoted as '-1', meaning that the patch was so large it was not measured (Hudson et al. 2014). To these patches we assigned a size slightly larger than the largest value reported in PREDICTS. We then selected datasets that included multiple patches with at least one patch of 10,000 ha or larger in size and used the largest patch as undisturbed reference (control). Thus, we assumed that fragmentation impacts are absent in patches of 10,000 ha or more, due to a shortage of datasets including larger patches. We then calculated per species, patch and dataset an abundance ratio by dividing its abundance in the smaller patch by its abundance in the control patch and obtained MSA values by averaging the truncated abundance ratios per patch size and dataset, which yielded 39 MSA values from seven studies. We used these MSA values as input to mixed beta regression models with dataset nested in study as random intercept and the $\log_{10}$-transformed patch size (ha) as fixed effect.

6 Impacts of road disturbance

In GLOBIO, impacts of road disturbance are quantified based on a relationship between MSA and the distance to the closest road (m). To quantify the relationship, we used the database collected by Benítez-López, Alkemade, and Verweij (2010) as a starting point and extended this with additional observations from more recent literature. The studies included in the database focused on the impacts of road construction and use on wildlife populations in the areas adjacent to roads. Noise, movements and exhausts of cars, for example, disturb habitat adjacent to roads and create road-effect zones that extend beyond the road itself. Papers addressing secondary impacts of road development (i.e. human encroachment and hunting) were discarded from the database, thereby precluding any overlap with studies included in the hunting database (see below). Per dataset within the database, we calculated MSA values based on the abundance of species measured at a given distance or within a distance interval from the road relative to their abundance at a larger control distance where road impacts were presumably absent. The extended database included 204 MSA values from 34 studies, which we used as input to mixed beta regression models with dataset within study as nested random intercept and the $\log_{10}$-transformed distance to the road (m) as fixed effect.

7 Impacts of hunting

In GLOBIO, impacts of hunting in the tropics are quantified based on the distance to hunters' access points (i.e., small settlements) within tropical biomes. Based on the central-place foraging hypothesis, hunting intensity is generally higher in the proximity of hunters' access points (Abernethy et al. 2013; Benítez-López et al. 2017), generating gradients of increasing species densities up to a distance where no effect is observed (i.e., species depletion distances). Although a proxy, distance to a settlement has been identified as the main factor explaining animal abundance declines due to hunting pressure, even after accounting for other factors such as human population density, poverty levels or accessibility to urban markets (Ana Benítez-López et al. 2019). We quantified the relationship between MSA and distance to settlement based on a database with mammal and bird abundances measured in hunted versus sites where hunting did not occur (Abernethy et al. 2013; Benítez-López et al. 2017). For this database, only studies that assessed the impact of hunting on wildlife abundance were included. Specifically, the studies had to report species abundance in at least one hunted area and one control area where hunting does not take place, and at increasing distance from access points. Studies with potential confounding effects due to other disturbances (e.g., hunted and logged area versus unlogged area where hunting takes place) were discarded. Also, we did not include studies that reported hunting on managed wild populations (i.e., with release of farm-reared individuals or subjected to culling) and/or populations used for recreational or trophy hunting. As MSA is supposed to represent overall assemblage-level intactness, we included observations of species irrespective of whether they are hunted or not. Overall, approximately 20% of the mammal species in our dataset have a body mass < 1 kg, hence are usually not hunted (Ripple et al. 2016). As for birds, 38% of the species in the dataset belong to families that are usually not hunted (Redford 2009).

Per dataset within the database, we calculated MSA values based on the abundance of species measured at a given distance or within a distance interval from an access point relative to their abundance at a larger control distance where hunting impacts were presumably absent. This resulted in 465 MSA values from 125 studies that we used as input to mixed beta regression models with dataset within study as nested random intercept, country as crossed random intercept (to account for possible cultural differences in hunting; Benítez-López et al. 2019) and the $\log_{10}$-transformed distance to hunters' access point (km) as fixed effect. Following Benítez-López et al. (2019), we tested for a possible effect of human population density, using $\log_{10}$-transformed population density values from the Gridded Population of the World dataset (Deichmann, Balk, and Yetman 2001) matched as closely as possible to the location and year of the study. However, the model with distance to settlements as only predictor had a lower BIC value.

Appendix J

The model includes two agricultural production ecosystem services (crop production and grass and fodder production) and six regulating services (pest control, pollination, carbon sequestration, erosion control, natural water purification, and the health and recreational value of lakes). GLOBIO-ES uses primarily output from the IMAGE model as input to calculate changes in ecosystem services (Schulp et al. 2012). Below we provide an overview of all ecosystem services that are modelled within GLOBIO-ES. Adjustments to the recreation potential module of GLOBIO-ES are shortly described under Section 2.3.2 of the main deliverable text and initial results are shown under Section 3.3.1 of the main deliverable text. The final choice for the types of ecosystem services that will be assessed within the ForestPaths project, will be determined in WP5. Thus, not all ecosystem services described in this Section may be analysed in the ForestPaths project.

<COPIED FROM THE MAIN PAPER of Schulp et al., 2012 (International Journal of Biodiversity Science, Ecosystem Services & Management)>

For each service, a conceptual model of the relationship between ecosystem properties and the ESF (Ecosystem Functions; the capacity of an ecosystem to deliver a service) was developed based on published equations or data, including online databases. ESSs (Ecosystem Services; the actual use of ESFs by humans and the contribution of the ecosystem to human well-being) were derived from ESFs by including the use of ESFs by humans in the model. For this, the demand of the function has been quantified, by, for example, calculating the yield of crops that depend on animal pollinators, or the fraction of the land where human use requires protection against erosion or floods. An overview of the data used in this study is presented in Table J1. For each ESF and ESS model, spatial calculations were done at the input resolution of the ecosystem property data and aggregated to 0.5◦ × 0.5◦ grid cells by calculating an area-weighted average value over each output grid cell. We simulated the provisioning services, such as food crop yield; regulating services, such as carbon sequestration, protection against natural hazards including erosion and floods, pollination and air quality; cultural services, such as tourism; and the provision of wild food which is both a cultural and a provisioning service (MA 2005).

1 Food crop yield

Food crop yield (Mg/km^2 per growth cycle in fresh weight) was simulated with the IMAGE framework (Bouwman, Kram, and Goldewijk 2006). The ESF for food crop yield was defined as the potential yield a location can provide, that is, the maximum of the potential yields of all crops included in IMAGE (Table J1). Potential yields were calculated as a function of climate, soil and relief conditions. The ESS for food crop yield was the actual yield. This was calculated from the potential yield by including the actual crop cover and the management factor (Table J1).

2 Wild food

Collection of wild food can provide an important part of the diet and can be seen as a sport or be important in local traditions (Russell 2007). The ESF for wild food was defined as the annual availability of game, fish, berries and mushrooms (kg/km^2). Based on the national and international hunting statistics (EFI 2007; FAO 2011), average availability of game, fish, berries and mushrooms per square kilometre for each land-cover type was calculated and coupled to the land-cover map. The ESS for wild food was calculated as the amount of wild food accessible to people within the maximum amount of time that people spend for collecting wild food. In this study, for game, a threshold of 2 hours was assumed; for fish 1 hour and for mushrooms and berry collection 0.5 hours (De Roman and Boa 2004; Tsachalidis and Hadjisterkotis 2008). Travel time from villages was calculated by applying the methods described by Nelson (2008) to the data described in Table J1.

Table J1. Overview of the data used in this study. The content of this table was directly copied from Table 1 in Schulp et al., 2012.

Ecosystem property	Unit	Description
Land cover		GlobCover global land cover map at a 250 m resolution
Elevation	m	Gtopo30 global DEM at 1km resolution
Precipitation sum	mm	Annual precipitation sum, 0.5◦ resolution
Precipitation surplus	mm	Annual precipitation sum minus annual evapotranspiration, 0.5◦ resolution
Precipitation distribution	%	% of annual precipitation per month, 0.5◦ resolution
Wet day frequency	#	Number of rain days per year
Temperature	◦C	Annual mean temperature, 0.5◦ resolution
Rivers		Location and hierarchy of rivers
Coasts		Land-sea boundary
Soil characteristics	%, cm, g/cm^3	Clay, silt and sand content; rooting depth, bulk density, from the Harmonized World Soil Database version 1.0, at 30" resolution
Population density	$\#/km^2$	Number of people per pixel at 1 arc second resolution
Crop fraction	%	For each crop included in IMAGE, the percentage of agricultural land in each IMAGE grid cell covered by this particular crop. Crops included are: Cereals (temperate/tropical), rice, maize, pulses, roots and tubers, oil crops
GDP	Euro/capita	Gross Domestic Product per country and NUTS2 region
Roads		Location and type of roads, from GRIP
Management factor	-	Management intensity (IMAGE); crop specific and region specific

Notes: DEM, Digital Elevation Model; GRIP, Global Road Inventory Project (Meijer et al. 2018).

3 Carbon sequestration

Sequestration of carbon in soil and vegetation is seen as a means for mitigating climate change (UN 1997). The ESF for carbon sequestration was defined as the net ecosystem productivity (NEP) (Mg C/km^2 per year), which is the difference between net primary productivity (NPP) and respiration, which have both been simulated with the IMAGE framework (Bouwman, Kram, and Goldewijk 2006). NPP was considered a function of climate, soil, atmospheric CO_2 concentration, altitude, land cover and land-cover history. Respiration depends on the carbon stocks in different soil compartments, turnover rates, soil water availability and temperature. The ESS for carbon sequestration was defined as the climate regulation by capturing CO_2 in soil and vegetation and calculated as the percentage of the annual country total CO_2 emission (UNstats 2011) that is captured by the ecosystem.

4 Protection against erosion

We calculated the erosion protection ESF as the decrease of erosion risk by vegetation, using indices ranging from 0 to 1 for the protective effects of each land-cover type as provided by Hootsmans et al. (2001). The ESS for erosion protection was defined as the decrease of erosion risk by vegetation in utilized areas with a high erosion risk. Erosion risk due to soil and landscape characteristics and rainfall intensity was mapped using a 0–1 index based on the Universal Soil

Loss Equation (USLE) (Batjes 1996). To calculate the ESS for erosion protection, the erosion risk index was multiplied with data from the erosion ESF map (Batjes 1996). Utilized areas are croplands and urban areas and were defined using the land-cover map (Table J1).

5 Flood protection

Floods can occur due to accumulation of run-off, river flooding or flooding from the sea while vegetation cover and soils can retain run-off and thus protect against floods (Fohrer, Haverkamp, and Frede 2005). To map the ESF for flood protection, first, the retention capacity of the landscape (%) was calculated, using retention capacities as a function of land cover (Johnston, Detenbeck, and Niemi 1990; Brye et al. 2000) and soil (FAO 2008) and maps of land cover and soil characteristics (Table J1). Then, sensitive areas, defined as areas close to rivers or coasts with a low elevation difference with the river or coast, or areas that receive a lot of run-offs were mapped. Third, the ESF was calculated as the retention capacity in areas that are sensitive to floods. Finally, the ESS for flood risk (percentage of grid cell with flood risk) was calculated as the ESF in areas that are sensitive to floods due to utilization of the land for crop production and urban land (Table J1).

6 Pollination

Several food crops depend on animal pollinators for pollination (Gallai et al. 2009). The pollination ESF was defined as the percentage yield loss due to diminished pollination (yield reduction fraction – YRF). In this study, the YRF was calculated for pulses and oil crops only. Other crops that dependent on pollinators are not included in the IMAGE model. For the pollinator-dependent crops, the YRF is set at 100% at a zero distance to nature and decreases upon increasing distance to nature. At a distance to nature >1200 m, the YRF is 40% (Steffan-Dewenter and Tscharntke 1999; Klein et al. 2007). The ESS for pollination was calculated as the additional yield (Mg/km^2) of pulses and oil crops due to wild pollination, based on the YRF and the food crop yield (Section 1 on food crop yield).

7 Air quality

As ESF for air quality, we calculated the capacity of the landscape to capture dust particles <10 µm (PM_{10}) (%). Atmospheric PM10 concentrations are influenced by vegetation that captures PM_{10}, rainfall and temperature (Fischer, Brunekreef, and Lebret 2004; Anttila and Salmi 2006). The percentage captured by vegetation was derived from Oosterbaan et al. (2009) and Pace (2003). The percentage PM_{10} removed from the atmosphere due to precipitation and temperature was calculated based on PM_{10} concentrations in 2000 from air quality monitoring stations throughout Europe (EEA 2011) and weather characteristics at the monitoring locations derived from the data described in Table J1. The ESS for air quality was considered as the amount of PM_{10} actually captured (g/km^2). This was calculated by multiplying atmospheric PM_{10} concentrations for Pan-Europe (CEIP 2011) with the ESF for air quality.

8 Tourism and recreation

The ESF for tourism was defined as the capacity of landscapes to supply attractive areas for tourism and recreation. We estimated indices for the attractiveness based on landscape features attractive for tourists and holidaymakers. Each index ranges from 0 (unattractive) to 1 (attractive). Landscape features included were the presence of coasts, relief (low mountainous areas are attractive), land cover (varied land cover, with low amounts of urban area and arable land are particularly attractive) and the presence of protected natural areas (Hall 1998; Russell 2007; Kienast et al. 2009). The indices were quantified using the data described in Table 7 and the data from Eurostat (2011). Finally, an average index was calculated. The ESS for tourism was defined as the suitability of attractive areas. People spend more time for holiday and recreation in richer regions (Nowaczek and Fennel 2002) and areas need to be accessible to attract tourists or holidaymakers. To map the tourism ESS, therefore, the tourism ESF was supplemented with an index for GDP and the accessible areas were identified using the travel time map (see Section 2 on wild food).

9 Combining services and visual presentation

To interpret the results and demonstrate the spatial relationships between ESFs, ESSs and landscape characteristics, we calculated landscape characteristics (% nature, % agricultural land, number of land-cover types per 0.5◦ × 0.5◦ grid cell, degree of mixing of land-cover types) based on the land-cover map (Table J1). Then, correlations between ESFs, ESSs and the landscape characteristics were calculated. All ESF and ESS maps were linearly normalized towards a 0–1 scale. Then, all normalized maps are added up and normalized again to provide a general overview of the availability of ESFs and supply of ESSs. The same was done for all categories (provisioning, regulating, cultural) separately. For the provisioning services, the non-normalized maps were added up and then normalized to give a realistic weight to the contribution of total food provision of both food crop production and wild food harvest.

Table J2. Scoring approach for each indicator of scenic beauty. The approach is flexible, hence can be adapted to different type of land use input, such as ESA-CCI maps and land use/management output of models. This table is copied from Tisma et al. (under review).

		Scores
Naturalness	% Agricultural land	
	< 40	2
	40-90	1
	90-100	0
	% urban area	
	Less than 5	3
	5-25	2
	25-90	1
	90-10	0
	% forest	
	0	0
	1 - 40	2
	40 - 90	3
	90-100	1
	% other natural land cover	
	0	0
	1- 40	2
	40 - 90	3
	90-100	1
Presence of water and coast	No water and 80 - 100% water	0
	1 -40	3
	40 - 390	2
	90-100	1
Relief	< 100m	0
	100-250m	1
	>250m	2

Table J3. Scoring approach for each indicator of visual diversity. The approach is flexible, hence can be adapted to different type of land use input, such as ESA-CCI maps and land use/management output of models. This table is copied from Tisma et al. (under review).

Visual diversity scores	SDI	SEI	AI
0	0 & 4	0 & 4	0 & 4
1	0 - <1	>0 - <2	0 - <1
2	>=3 - <4	>=3 - <4	>=3 - <4
3	>=1 - <2	>=1 - <2	>=1 - <2
4	>=2 - <3	>=2 - <3	>=2 - <3

Table J4. Scoring approach for each indicator of tranquillity. This table is copied from Tisma et al. (under review).

Darkness	Absence of stable night-time lights at a given place.	2 – dark (80-100% dark) 1 – partly dark (60-80% dark) 0 – light (0 – 60% dark)
Silence	Quietness, absence of noise disturbance. Computed following the method of EEA (2014) using roads from GRIP.	2 – quiet 1 – moderately quiet 0 - noisy
Solitude	Seclusion or "away from others". Measured as population density in number of people per km2.	2 – lowest population density 1 - intermediate 0 – highest population density

Appendix K

Table K1. CRAFTY land use classes and matching GLOBIO land use and land management classes. For GLOBIO classes highlighted in green completely match the definitions of the CRAFTY AFT classes, while GLOBIO classes highlighted in orange partially mismatch with CRAFTY categories.

AFT	**AFT category**	**AFT description**	**GLOBIO category**	**GLOBIO description**
IntC3C	Intensive C3 cereal crops	Intensive cereal cultivation (wheat, rye, barley, etc.) for food production (high fertilization <250 kg N/ha)	Cropland – intense use	Land on which crops are grown with N fertilizer application > than 100 kg N-input/ha
IntC3oil	Intensive C3 oil crops	Intensive oil crops (rapeseed, sunflower, linseed, soybeans) for food production (high fertilization <250 kg N/ha)	Cropland – intense use	Land on which crops are grown with N fertilizer application > than 100 kg N-input/ha
IntC3fruitveg	Intensive C3 fruit and vegetables	Intensive fruit or vegetable crops (leeks, onion, carrots, strawberries, grape etc.) for food production (high fertilization <250 kg N/ha)	Cropland – intense use	Land on which crops are grown with N fertilizer application > than 100 kg N-input/ha
IntC3star	Intensive C3 starchy roots	Intensive starchy root crops (potatoes, sugarbeet) for food production (high fertilization <250 kg N/ha)	Cropland – intense use	Land on which crops are grown with N fertilizer application > than 100 kg N-input/ha
IntC4	Intensive C4 crops	Intensive maize cultivation for food production (high fertilization <250 kg N/ha)	Cropland – intense use	Land on which crops are grown with N fertilizer application > than 100 kg N-input/ha
IntFodder	Intensive C3/C4 fodder crops	Intensive crops (cereals, starchy roots, maize) cultivated for feeding domestic livestock (high fertilization <250 kg N/ha)	Cropland – intense use	Land on which crops are grown with N fertilizer application > than 100 kg N-input/ha
ExtC3C	Extensive C3 cereal crops	Extensive cereal cultivation (wheat, rye, barley, etc.) for food production (low fertilization <100 kg N/ha)	Cropland – minimal use	Land on which crops are grown with N fertilizer application of up and including 100 kg N-input/ha
ExtC3oil	Extensive C3 oil crops	Extensive oil crops (rapeseed, sunflower, linseed,	Cropland – minimal use	Land on which crops are grown with N fertilizer application of up and including 100 kg N-input/ha

		soybeans) for food production (low fertilization <100 kg N/ha)		
ExtC3fruitveg	Extensive C3 fruit and vegetables	Extensive fruit or vegetable crops (leeks, onion, carrots, strawberries, grape etc.) for food production (low fertilization <100 kg N/ha)	Cropland – minimal use	Land on which crops are grown with N fertilizer application of up and including 100 kg N-input/ha
ExtC3puls	Extensive C3 pulses	Extensive pulses cultivation (peas, lentils, chickpeas, beans) for food production (low fertilization <100 kg N/ha)	Cropland – minimal use	Land on which crops are grown with N fertilizer application of up and including 100 kg N-input/ha
ExtC3star	Extensive C3 starchy roots	Extensive starchy root crops (potatoes, sugarbeet) for food production (low fertilization <100 kg N/ha)	Cropland – minimal use	Land on which crops are grown with N fertilizer application of up and including 100 kg N-input/ha
ExtC4	Extensive C4 crops	Extensive maize cultivation for food production (low fertilization <100 kg N/ha)	Cropland – minimal use	Land on which crops are grown with N fertilizer application of up and including 100 kg N-input/ha
ExtFodder	Extensive C3/C4 fodder crops	Extensive crops (cereals, starchy roots, maize) cultivated for feeding domestic livestock (low fertilization <100 kg N/ha)	Cropland – minimal use	Land on which crops are grown with N fertilizer application of up and including 100 kg N-input/ha
IntP	Intensive pastoral	Intensive pasture for livestock (maximum fertilization < 250 kg N/ha)	Pasture – intense use	Land on which livestock is kept and where N application > than 100 kg N-input/ha
ExtP	Extensive pastoral	Extensive pasture for livestock (moderate fertilization <50 kg N/ha)	Pasture – minimal use	Land on which livestock is kept and where N application reaches up and including 100 kg N-input/ha
VExtP	Very extensive pastoral	Very extensive pasture for livestock (minimum fertilization 0 kg N/ha, e.g. mountain areas)	Pristine	Naturally regenerating forest, grasslands or shrublands with no clearly visible indications of human activities for at least 80 years. The forests, grasslands or shrublands consist of multiple native tree species.
AF	Agroforestry	Agro-forestry systems with a mix of broadleaf trees and crop	Agroforests	Agricultural land on which crops are grown underneath or besides trees used for wood, food, energy and/or other non-timber forest products, or to promote crop growth.

		cultivation (cereals, fruit and vegetables, oil crops)		Naturally regenerated forest consisting of multiple mainly native tree species are either partially cut in order to grow crops underneath and besides remaining trees or native or excotic trees are planted or seeded on agricultural land to create the agroforest from scratch. The density and diversity of trees in agroforests around the world varies substantially, as do management practices to control tree growth, such as thinning and cutting.
IntBF	Broadleaf forest (intensive)	Broadleaf trees with intensive forest management. 40% Of trees are thinned every 30 years.	Multipurpose plantation	Land on which one or multiple tree species, either native or exotic, are planted or seeded for wood and/or energy production. Trees are harvested by removing all trees in an area, after which trees are planted or seeded again, or the previously planted trees coppice to produce a new stand. This results in a regular, even-aged forest stand structure. Rotation lengths range between 20 to over 150 years. Management practices, such as thinning and fertilizer application may or may not take place over the course of a rotation period. These plantations are often also used for other purposes, such as recreation and foraging.
IntCF	Conifer forest (intensive)	Conifer trees with intensive forest management. 40% Of trees are thinned every 30 years.	Multipurpose plantation	Land on which one or multiple tree species, either native or exotic, are planted or seeded for wood and/or energy production. Trees are harvested by removing all trees in an area, after which trees are planted or seeded again, or the previously planted trees coppice to produce a new stand. This results in a regular, even-aged forest stand structure. Rotation lengths range between 20 to over 150 years. Management practices, such as thinning and fertilizer application may or may not take place over the course of a rotation period. These plantations are often also used for other purposes, such as recreation and foraging.
ExtBF	Broadleaf forest (extensive)	Broadleaf trees with extensive forest management. 10% Of trees are thinned every 30 years.	Selection systems	Harvesting of high value trees for wood and/or energy production according to (predefined) harvesting rules and/or preferences with the purpose to maintain an uneven forest stand. Harvesting takes place in naturally regenerating forests consisting of mainly native tree species. Single trees or groups and/of mature trees are removed, resulting in a heterogeneous stand structure.
ExtCF	Conifer forest (extensive)	Conifer trees with extensive forest management. 10% Of trees are thinned every 30 years.	Selection systems	Harvesting of high value trees for wood and/or energy production according to (predefined) harvesting rules and/or preferences with the purpose to maintain an uneven forest stand. Harvesting takes place in naturally regenerating forests consisting of mainly native tree species. Single trees or groups and/of mature trees are removed, resulting in a heterogeneous stand structure.

MW	Multifunctional mixed woodland (extensive)	Mixed trees with extensive forest management. 10% Of trees are thinned every 30 years.	Selection systems	Harvesting of high value trees for wood and/or energy production according to (predefined) harvesting rules and/or preferences with the purpose to maintain an uneven forest stand. Harvesting takes place in naturally regenerating forests consisting of mainly native tree species. Single trees or groups and/of mature trees are removed, resulting in a heterogeneous stand structure.
CW	Conserved Woodland	Protected forests with no/low management	Pristine	Naturally regenerating forest, grasslands or shrublands with no clearly visible indications of human activities for at least 80 years. The forests, grasslands or shrublands consist of multiple native tree species.
BioenergyG1	Bioenergy Gen. 1 (intensive)	Food crops (rapeseed, wheat, barley, maize, sugar beet) cultivated for bioenergy production (high fertilization <250 kg N/ha)	Cropland – intense use	Land on which crops are grown with N fertilizer application > than 100 kg N-input/ha
BioenergyG2	Bioenergy Gen. 2 (intensive)	Non-food crops or grasses used in the production of biofuels or biomass (e.g. miscanthus, Short Rotation Coppice willow)	Perennial tree crops	Land on which one or a few tree species, either native or exotic, are planted or seeded for food and/or energy production (e.g., oil palm or banana). This results in a regular, even-aged stand structure. Management practices to promote crop growth, such as fertilizer application and irrigation, often occur.
Solar	Dedicated solar farm	Large-scale installation of solar panels for solar energy production	Urban area	Urban land.
Agrovoltaic	Agrovoltaic farm	Combination of cropland (cereals, maize, starchy roots or fruit and vegetables) and solar energy production	Pasture – minimal use	Land on which livestock is kept and where N application reaches up and including 100 kg N-input/ha
UL	Unmanaged	Unlabelled, sparsely or un-managed land (mining areas, wetlands, shrubs, barren land, rocks and ice)	Pristine	Naturally regenerating forest, grasslands or shrublands with no clearly visible indications of human activities for at least 80 years. The forests, grasslands or shrublands consist of multiple native tree species.
Urban	Urban	Urban land	Urban area	Urban land.
WATER	Water	Water areas		

Table K2. Matched CRAFTY AFTs and GLOBIO land use/land management classes and corresponding MSA values for vertebrate, plant, and animal biodiversity, as well as the sources from which MSA values were derived. MSA vertebrates provides MSA values calculated for vertebrates only (i.e., amphibians, birds, mammals, reptiles). MSA plants provides MSA values calculated for plants only and MSA animals provides MSA values calculated for vertebrates and insects together.

AFT	AFT category	GLOBIO category	MSA vertebrates	MSA plants	MSA animals	MSA source
IntC3C	Intensive C3 cereal crops	Cropland – intense use	0,356	0,131		GLOBIO 4 (Schipper et al. 2020)
IntC3oil	Intensive C3 oil crops	Cropland – intense use	0,356	0,131		GLOBIO 4 (Schipper et al. 2020)
IntC3fruitveg	Intensive C3 fruit and vegetables	Cropland – intense use	0,356	0,131		GLOBIO 4 (Schipper et al. 2020)
IntC3star	Intensive C3 starchy roots	Cropland – intense use	0,356	0,131		GLOBIO 4 (Schipper et al. 2020)
IntC4	Intensive C4 crops	Cropland – intense use	0,356	0,131		GLOBIO 4 (Schipper et al. 2020)
IntFodder	Intensive C3/C4 fodder crops	Cropland – intense use	0,356	0,131		GLOBIO 4 (Schipper et al. 2020)
ExtC3C	Extensive C3 cereal crops	Cropland – minimal use	0,542	0,129		GLOBIO 4 (Schipper et al. 2020)
ExtC3oil	Extensive C3 oil crops	Cropland – minimal use	0,542	0,129		GLOBIO 4 (Schipper et al. 2020)
ExtC3fruitveg	Extensive C3 fruit and vegetables	Cropland – minimal use	0,542	0,129		GLOBIO 4 (Schipper et al. 2020)
ExtC3puls	Extensive C3 pulses	Cropland – minimal use	0,542	0,129		GLOBIO 4 (Schipper et al. 2020)
ExtC3star	Extensive C3 starchy roots	Cropland – minimal use	0,542	0,129		GLOBIO 4 (Schipper et al. 2020)
ExtC4	Extensive C4 crops	Cropland – minimal use	0,542	0,129		GLOBIO 4 (Schipper et al. 2020)
ExtFodder	Extensive C3/C4 fodder crops	Cropland – minimal use	0,542	0,129		GLOBIO 4 (Schipper et al. 2020)
IntP	Intensive pastoral	Pasture – intense use	0,504	0,188		GLOBIO 4 (Schipper et al. 2020)
ExtP	Extensive pastoral	Pasture – minimal use	0,355	0,251		GLOBIO 4 (Schipper et al. 2020)
VExtP	Very extensive pastoral	Pristine	1	1	1	
AF	Agroforestry	Agroforests		0,41	0,53	Global analysis (Section 2.1)

IntBF	Broadleaf forest (intensive)	Multipurpose plantation		0,27	0,44	European meta-analysis (Section 2.1)
IntCF	Conifer forest (intensive)	Multipurpose plantation		0,27	0,44	European meta-analysis (Section 2.1)
ExtBF	Broadleaf forest (extensive)	Selection systems		0,74	0,43	European meta-analysis (Section 2.1)
ExtCF	Conifer forest (extensive)	Selection systems		0,74	0,43	European meta-analysis (Section 2.1)
MW	Multifunctional mixed woodland (extensive)	Selection systems		0,74	0,43	European meta-analysis (Section 2.1)
CW	Conserved Woodland	Pristine	1	1	1	GLOBIO 4 (Schipper et al. 2020)
BioenergyG1	Bioenergy Gen. 1 (intensive)	Cropland – intense use	0,356	0,129		GLOBIO 4 (Schipper et al. 2020)
BioenergyG2	Bioenergy Gen. 2 (intensive)	Perennial tree crops		0,16	0,2	GLOBIO 4 (Schipper et al. 2020)
Solar	Dedicated solar farm	Urban area	0,26	0,308		GLOBIO 4 (Schipper et al. 2020)
Agrovoltaic	Agrovoltaic farm	Pasture – minimal use	0,355	0,251		GLOBIO 4 (Schipper et al. 2020)
UL	Unmanaged	Pristine	1	1	1	
Urban	Urban	Urban area	0,26	0,308		GLOBIO 4 (Schipper et al. 2020)
WATER			NA	NA	NA	