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List of abbreviations

ANOVA:	Analysis of Variance
BDM:	Biodiversity Monitoring Switzerland
BDP:	Biodiversity Damage Potential
CF:	Characterization Factor
CurBm:	Current Biomass Scenario
DALY:	Disability Adjusted Life Years lost
DF:	Degrees of Freedom
EEA:	European Environmental Agency
EPIC:	Environmental Policy Integrated Climate
FAO :	Food and Agriculture Organization
FF :	Fate Factor
G4M:	Global Forestry Model
GHG:	Green House Gas
GLC2000:	Global Land Cover for the year 2000
GPS:	Global Positioning System
GWP:	Global Warming Potential
HRU:	Homogeneous Response Units
HWSD:	Harmonized World Soil Database
IIASA:	International Institute for Applied Systems Analysis
ILCD:	International Reference Life Cycle Data System
IPCC :	Intergovernmental Panel on Climate Change
ISO:	International Standards Organization
LCA:	Life Cycle Assessment
LCIA:	Life Cycle Impact Assessment
LCI:	Life Cycle Inventory
LU:	Land Use
MBF:	Moist Broadleaf Forest
MSA:	Mean species abundance of original species
NPP:	Net Primary Productivity
NPPD:	Net Primary Productivity Depletion
PAF :	Potentially Affected Species
PDF :	Potentially Disappeared Species
PNV:	Potential Natural Vegetation
PNOF :	Potentially not occurring fraction of plant species
(R)USLE:	(Revised) Universal Soil Loss Equation
SETAC:	Society of Environmental Toxicology and Chemistry
SIMU:	Simulation Units
SOC:	Soil Organic Carbon
SOM:	Soil Organic Matter
UNEP :	United Nation Environmental Program
USDA:	United States Department of Agriculture
VMS:	Vessel Monitoring System
WWF:	World Wide Fund for Nature

Definitions

Biodiversity: means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (Definition from Convention on Biological Diversity).

Characterization factor: Quantitative expression of the impact on the environment caused by an intervention in the ecosystem

Carbon sequestration: Net additions, for a period of time, to a stock or pool of carbon

Ecoregions: ecologically and evolutionary defined area, with distinct environmental conditions, such as climate, species patterns, etc.

Fishing effort: A measure of the time spent fishing, sometimes measured in hours a gear is under water, sometimes in fishing days, sometimes in combination with the power of the engine (e.g. kW*days).

Functional diversity: is the component of biodiversity, related to the functions of an organism in communities and ecosystems.

Functional traits: inherent measurable organism features/characteristics or an organisms, such as morphological (e.g. seed size), ecophysiological (e.g. foraging habitat) and life history characteristics.

Meta-analysis: analysis based on the combination of different studies, in which contrast and/or similar patterns are identified.

Potential natural vegetation (NPP_0): the anticipated state of mature vegetation in the absence of human intervention

Species richness: the number of different species present in a particular area.

Species abundance: to the relative representation of a species in a particular ecosystem. It thus takes into account the number of individual organisms of a species

WWF biomes: represent 14 major habitat types of the world. The classification is based on regions with similar climatic conditions and similar ecosystem types.

Executive Summary

The goal of Task 1.1 is to develop operational and scientifically sound methods for the assessment of land use on ecosystem services, biodiversity and human health. Characterization factors for land use impacts have been developed with specific attention for ecosystem services as well as biodiversity which have hardly been assessed within the framework of Life Cycle Assessments. The work reported here focuses on results obtained in relation to new and novel methods that have been developed for the assessment of land use impacts on biodiversity (**Chapters 1, 2** and **3**), global, spatially-explicit life cycle impact assessment of forest and agricultural management quantifying the impact of forest management on the forest carbon balance with implications for global warming potential and subsequent damage to biodiversity and health (**Chapter 4**), and agricultural management with implications for erosion and damage to soil (**Chapters 5** and **6**), including a specific case-study in relation to energy crops in Spain (**Chapter 6**), as well as a preliminary assessment of impacts of fishing on seafloor disturbance (**Chapter 7**). All of the Chapters are characterized by methods that add a well-developed spatial dimension to the Life Cycle Assessment framework.

Chapter 1 presents a first approach to quantify land use impacts on biodiversity across different world regions and highlights uncertainties and research needs. This method is based on the UNEP/SETAC land use assessment framework and focuses on occupation impacts, quantified as a Biodiversity Damage Potential. Data on multiple species groups were derived from a global quantitative literature review and national biodiversity monitoring data from Switzerland. The presented characterization factors for Biodiversity Damage Potential can approximate land use impacts on biodiversity in LCA studies that are not intended to directly support decision-making on land management practices. **Chapter 2** presents a spatially explicit approach to assess the impacts of land use on biodiversity at regional and global scales. Potential regional species loss due to total accumulated land use activities within all global WWF ecoregions were modelled and used to calculate characterization factors for LCIA. A distinction was made between potentially reversible impacts (i.e. regionally extinct, non-endemic species) and irreversible impacts (i.e. global extinction of endemic species) to calculate land occupation and transformation impacts, and permanent impacts respecitively for five taxonomic groups. Uncertainties of the characterization factors were quantified with Monte Carlo simulations. **Chapter 3** examines habitat change as a direct driver of biodiversity loss. The novelty of this Chapter is the use of a new biodiversity indicator based on functional diversity within mammals, birds and plants as opposed to species richness to describe the impacts of land use on ecosystem functions and services. Significant differences have been found between characterization factors for species richness and functional diversity. Across all taxa, characterization factors differ significantly between land use types. **Chapter 4** used output from an operational Global Forest Model with different management options in relation to wood extraction with implication for forest carbon balance to develop a global, spatial-explicit life cycle impact assessment method to quantify the effects of changes in forests biomass extraction for bioenergy with implications for global warming potential and subsequent damage to biodiversity and health. **Chapter 5** used output from a global implementation of the EPIC crop model with different management intensities to calculate spatially explicit characterization factors for erosion regulation due to crop cultivation for the world. Damage is expressed in terms of extra costs due to soil loss caused by crop production. The methodology presented in Chapter 4 will be extended to include spatial characterization factors for impacts on carbon stocks due to crop cultivation for several crop types as specified in DoW and this will be reported in the final deliverable of the project.

Chapter 6 describes the development of a globally applicable, spatially differentiated Life Cycle Impact Assessment method to account for land occupation impacts in LCA, with a focus on soil erosion. The LCA model was applied to a case study in Spain focusing on soil erosion-related impacts resulting from substituting traditional crops for enery crop rotation. **Chapter 7** reports on Life Cycle Assessment methods to assess the impact of fisheries on the seafloor disturbance and presents the state of current knowledge on seafloor impacts of fishing and present a number of alternative approaches that can be taken to advance the current state of the art indicator that has been used in seafood LCAs to data, swept seafloor area. A step-wise guide for LCA practitioners is included on how to assess whether seafloor impacts are relevant in a study or not, and if they are how they can be assessed quantitatively.

The ILCD currently does not recommend an endpoint method for life cycle impact assessment of land use. Therefore research efforts of D1.6 mainly focused on impacts on biodiversity, human health damage and additional costs. Direct as well as indirect pathways were assessed. The indirect pathways, via carbon stock changes have the advantage to provide the possibility to come up with midpoint characterization factors (i.e. change in carbon stock due to forestry land use) that are compatible with the endpoint characterization factors reflecting changes in biodiversity and human health damage. This compatibility is currently not there and seen as one of the drawbacks of existing land use LCIA methods. The endpoint models focus on expanding global coverage, including more data to make a more robust method as well as providing more information on the endpoint indicator.

1. Land use impacts on biodiversity in LCA: a global approach[‡]

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1.1. Introduction

During the last decades, global biodiversity loss has become a major environmental concern. One of the main drivers of current and projected future biodiversity loss is habitat change or land use (Alkemade et al. 2009; Millennium Ecosystem Assessment 2005; Pereira et al. 2010; Sala et al. 2000). Within life cycle impact assessment (LCIA) attempts have been made to quantify the impacts of land use and other important drivers of biodiversity loss, such as climate change and pollution (for a review see Curran et al. 2011). Several approaches on how to quantify land use related biodiversity impacts have been proposed (Achten et al. 2008; Geyer et al. 2010; Kyläkorpi et al. 2005; Koellner 2000; Koellner et al. 2004; Koellner and Scholz 2007; Lindeijer 2000a, b; Michelsen 2008; Müller-Wenk 1998; Penman et al. 2010; Schenck 2001; Schmidt 2008; De Schryver et al. 2010; van der Voet 2001; Vogtländer et al. 2004; Weidema and Lindeijer 2001), of which some have been operationalized in LCA software for broad use by LCA practitioners (e.g. Goedkoop and Spriensma 1999; Goedkoop et al. 2008).

Although the environmental relevance of assessing land use impacts on biodiversity in LCIA is widely recognized, the task remains difficult. Biodiversity is a complex and multifaceted concept, involving several hierarchical levels (i.e. genes, species, ecosystems), biological attributes (i.e., composition, structure, function; Noss 1990) and a multitude of temporal and spatial dynamics (see e.g. Rosenzweig 1995). Biodiversity assessments therefore have to simplify this complexity into a few facets, which are quantifiable with current knowledge and data. Existing land use LCIA methods were mainly developed for one specific region (often Europe) using species richness of vascular plants as an indicator (e.g. Koellner and Scholz 2008; De Schryver et al. 2010). Weidema and Lindeijer (2001) proposed a first approach to assess land use impacts on biodiversity on a global scale, quantifying the biodiversity value of reference habitat of different biomes based on vascular plant species richness, ecosystem scarcity and ecosystem vulnerability. However, the reduction of the biodiversity value of different land use types was estimated based on assumptions by the authors and was not supported by empirical data (see Weidema and Lindeijer 2001, p. 37). To quantify land use impacts across global value chains more accurately a regionalized global method is needed, based on a broader

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[‡] This chapter has been published in a similar form in The International Journal of Life Cycle Assessment (de Baan, Alkemade, Koellner, 2012)

taxonomic coverage. This is required due to the spatial heterogeneity of biodiversity and due to the non-uniform and variable reactions of ecosystems and species to disturbances such as land use. Although plants are important components of terrestrial ecosystems, they only make up an estimated 2% of all species (Heywood and Watson 1995) and their reaction to land use is not necessarily representative for the impacts on other species groups.

In this paper we propose a first attempt to quantify biodiversity impacts in LCIA in different world regions based on empirical data, focusing on the facet of species composition. We illustrate how global quantitative analysis of peer-reviewed biodiversity surveys can be combined with national biodiversity monitoring data to assess land use impacts across multiple taxonomic groups and world regions, using a set of species-based biodiversity indicators. The indicator relative species richness is used to calculate characterization factors for occupation impacts of terrestrial ecosystems expressed as a Biodiversity Damage Potential (BDP).

1.2. Methods

This study is based on the framework for Life Cycle Impact Assessment (LCIA) of land use, developed by the UNEP/SETAC Life Cycle Initiative working group (LULCIA; Milà i Canals et al. 2007; Koellner et al. 2012b), which distinguishes three types of land use impacts: transformation impacts (caused by land use change), occupation impacts (occurring during the land use activity) and permanent impacts (i.e. irreversible impacts on ecosystems, which occur when an ecosystem cannot fully recover after disturbance). For calculating transformation and permanent impacts, reliable data on regeneration success and times of the world's ecosystems is required, which were not available for this study. Therefore, we only focused on occupation impacts and, for modeling purposes, neglected the temporal dynamics of biodiversity by assuming that we can assign a constant "biodiversity score" to occupied land (i.e. no reduction in biodiversity over time) and to a (semi)-natural reference habitat. The impact of land use on biodiversity was assessed by comparing the relative difference of biodiversity of a land use *i* with a (semi-) natural reference situation. Spatial aspects were considered by using a site-specific reference situation and by calculating impacts per biogeographic region. As proposed in Koellner et al. (2012a) biomes defined by the World Wide Fund For Nature (WWF; see Olson et al. 2001) were used as spatial unit for biogeographic differentiation, which represent the world's 14 major terrestrial habitat types. Land use types were classified based on the UNEP/SETAC LULCIA proposal (Koellner et al. 2012a).

1.2.1. Calculation of characterization factors

Characterization factors of occupation impacts, CF_{Occ}, were calculated according to the UNEP/SETAC framework (Milà i Canals et al. 2007; Koellner et al. 2012b), which is graphically illustrated in Figure 8.1.1, Annex. CF_{Occ} are given as the difference between the ecosystem quality of a reference situation ref (defined as 100% = 1) and a land use type LU_i per region j. In this study, ecosystem quality was expressed as biodiversity, measured as relative species richness S_{rel} (see section 1.2.4).

$$CF_{Occ,LU_{i,j}} = S_{rel,ref,j} - S_{rel,LU_{i,j}} = 1 - S_{rel,LU_{i,j}} \quad \text{Eq. 1.1}$$

The numerical value of CF_{Occ} is normally between 0 and + 1 (representing a damaging impact on biodiversity), but negative values are also possible (denoting a beneficial impact). To calculate impact scores for land use occupation, CF_{Occ} is multiplied by the land use occupation flows from a life cycle inventory (given as time t_{Occ} and area A_{Occ} required for a certain land use activity).

$$\text{Occupation Impact} = (\text{Inventory flow land occupation}) * CF_{Occ} = (A_{Occ} * t_{Occ}) * CF_{Occ} \quad \text{Eq. 1.2}$$

Transformation impacts scores would be calculated following Eq. 1.3. Here, the inventory flow is given as a transformed area A_{Trans} and the characterization factor CF_{Trans} is calculated based on equation 1.4, with t_{reg} being the time required for an ecosystem to recover after a disturbance.

$$\text{Transformation Impact} = (\text{Inventory flow land transformation}) * CF_{Trans} = A_{Trans} * CF_{Trans} \quad \text{Eq. 1.3}$$

$$CF_{Trans,LU_{i,j}} = 0.5 * (S_{rel,ref,j} - S_{rel,LU_{i,j}}) * t_{reg,LU_{i,j}} = 0.5 * CF_{Occ,LU_{i,j}} * t_{reg,LU_{i,j}} \quad \text{Eq. 1.4}$$

As no reliable data on region and land use type specific regeneration times of biodiversity t_{reg} were available for different world regions, CF_{Trans} were not calculated in this study.

1.2.2. Reference situation

Ecosystems and biodiversity are changing over time due to population, succession and evolutionary dynamics, but also due to intended and unintended human impacts. To quantify land use impacts on biodiversity on a global scale, a temporal baseline or reference situation for biodiversity has to be defined, which lies either in the past, present or future. In addition, one can either use the biodiversity occurring in a region on all different land use types, or the biodiversity occurring on specific land use types (e.g. semi-natural habitat) as a reference. Any choice of such a temporal reference involves different degrees of human impacts for different world regions, as the human land use history varies from region to region (see e.g. Ramankutty and Foley 1999). Here we chose the current, late succession habitat stages as reference, which are widely used as target for restoration ecology and serve as a proxy for the Potential Natural Vegetation (PNV), i.e., hypothetical future ecosystems that would develop if all human activities would be removed at once (Chiarucci et al. 2010). Such late succession habitat stages can have experienced different degrees of natural or human disturbances in the past. In many tropical world regions, the past human influence was low, so the chosen reference is to a large extent undisturbed by humans, whereas in many temperate regions few or no undisturbed habitat exists. In Europe, for example, forests currently cover 35% of the surface (SOER Synthesis 2010), whereas the natural post-glacial forest cover (i.e. without human land use) is estimated to be 80-90% (Stanners and Philippe 1995). Of the remaining forest area, only 5% is considered as undisturbed forest (SOER Synthesis 2010). Thus, as the reference habitats chosen in this study not necessarily represent pre-human, natural habitats, we use the term “(semi)-natural” to refer to the reference situation. More detail on the data used for quantifying biodiversity of the reference habitat is given in the next section.

1.2.3. Data sources

Two data sources were combined in this study to quantify biodiversity of different land use types and reference situations for different world regions: the GLOBIO3 database, which is based on a quantitative review of literature (Alkemade et al. 2009), and national biodiversity monitoring data of Switzerland (BDM, 2004). The GLOBIO3 database was compiled for the GLOBIO3 model, which aims at assessing impacts of multiple drivers of biodiversity loss at regional and global scales (Alkemade et al. 2009). The database contains datasets extracted from peer-reviewed empirical studies that compare biodiversity of different land use types with an undisturbed or little disturbed reference situation within the same study site. Depending on data provided in each study, the impact of land use is recorded as relative change in species richness or abundance of a range of different taxonomic groups. For each study, we additionally extracted the geographical coordinates of the study site to assign it to the corresponding WWF biome and ecoregion. A total of 195 publications, providing 644 data points on different land use types and 254 data points on reference situations from a total of 9 out of 14 biomes were included here (de Baan, Alkemade, Koellner, 2012), but the data was unevenly distributed. Due to publication bias and lack of undisturbed reference habitats in regions with long and intense human land use history, the database contains many studies conducted in tropical regions and less data in temperate and none in boreal zones (for geographical distribution of data see Figure 8.1.2 and Table 8.1.1, Annex). We therefore complemented our analysis with national biodiversity monitoring data of Switzerland (BDM 2004) used in earlier land use LCIA methods (Koellner and Scholz 2008). The used BDM indicator “species diversity in habitats (Z9)” is based on a grid of 1’600 sampling points evenly distributed over Switzerland, covering two biomes (Temperate Broadleaf & Mixed Forests and Temperate Coniferous Forests). In each of the 10 m2 sampling points, species richness of vascular plants, moss and mollusks and the corresponding land use type are recorded. To make this dataset comparable to the GLOBIO3 data we first reclassified the land use type of each sampling point based on Koellner and Scholz (2008) into broader land use classes (see Table 8.1.6, Annex). We then grouped all sampling points into ecologically similar regions to define regional (semi-)natural reference situations. We split the 10 biogeographic regions of Switzerland defined in BDM (2004) into three altitudinal zones (colline: below 800 m a.s.l.; montane: 800-1300 m; subalpine: 1300-2000 m; see Baltisberger 2009) and excluded the high elevation plots (alpine and nival: above 2000m). This resulted in 26 regions j across Switzerland, as not all altitudinal zones occur in every biogeographic region. For each of the 26 regions and for each of the three sampled species groups, the average species richness of all sampling points per land use type was calculated, resulting in totally 186 averaged data points for different land use types (see also Table 8.1.1, Annex). All sampling points in (semi-)natural habitats (forests, grasslands, wetlands, bare areas and water bodies) were assigned as regional reference situation (for more detail see Table 8.1.6, Annex). As for the land use types, the average species richness per region and species group was calculated for the reference, resulting in 72 data points for the reference. To test the sensitivity of choice of reference situations, results were recalculated using an alternative reference habitat containing only forest sampling points.

1.2.4. Indicator selection and calculation

As a primary indicator for biodiversity impacts we chose relative changes in observed species richness S_{rel} between a (semi-)natural reference and a specific land use type i. For each taxonomic group g and region j the species richness of the reference S_{ref} was divided by the species richness of each land use type *i*, S_{LUi} , (Eq. 1.5). For the BDM dataset, the regionally averaged species richness of the land use types and the reference were used for calculating the relative species richness.

$$S_{rel,LU_{i,j,g}} = \frac{S_{LU_{i,j,g}}}{S_{ref_{j,g}}} \qquad \text{Eq. 1.5}$$

The selected indicator species richness is a simple and widely applied indicator recording the number of species in a habitat (also referred to as α-diversity or within habitat diversity; Hayek and Buzas 2010) and data availability is rather high compared to other biodiversity indicators. The disadvantage of using species richness as a proxy for biodiversity is that it only contains limited information on the many facets of biodiversity. It only records the presence or absence of species within a sampling area and gives equal weight to all species recorded in a sample, no matter how abundant or biologically distinct they are (i.e., 10 individuals of an endemic species and 1 individual of an invasive species are both recorded as 1 species). Species richness neither provides information on between-habitat diversity, i.e., species turnover or β-diversity (see Koellner et al. 2004). This indicator is in addition affected by undersampling: the species richness of an ecosystem is often underestimated as the number of species recorded highly depends on sampling efforts.

Besides species richness, a wide range of diversity measures have been developed, each quantifying other aspects of biodiversity (see e.g. Hayek and Buzas 2010; Purvis and Hector 2000). To analyze the influence of choice of indicator on the results, we calculated four additional, commonly used species based biodiversity indicators: Fisher's α, Shannon's entropy H, Sørensen S_s and Mean species abundance of original species (MSA) (see formulas in Table 1.1). Fisher's α (Fisher et al. 1943) is an indicator that corrects for incomplete sampling: it estimates "true" species richness from a sample, fitting the observed values of species richness Sobs and total number of individuals N_{obs} to a theoretical (empirically derived) relationship between "true" species richness S and "true" number of individuals N. Shannon's entropy H (Shannon 1948) combines information on species abundance and richness in one number and reaches a maximum when all species occurring in a sample are equally abundant. Sørensen S_s (Sørensen 1948) and mean species abundance of original species (MSA, Alkemade et al. 2009) both compare the species composition of two samples (here the reference and a land use type i). Sørensen reports how many reference-habitat species occur in the land use type *i* and reaches a maximum value of 1 if all of them occur in the land use type *i* and a minimum value of 0 if none of the reference-habitat species occur in the land use type i. MSA, which has been developed for the GLOBIO3 model (Alkemade et al. 2009), assesses changes in abundance of each reference-habitat species and thus reports changes in species composition earlier than Sørensen, which only indicates a complete absence of a species from a site.

Besides the number of species S, these indicators all require additional information such as species identity (i.e. checklist of species present) and/or abundance (number of individual organisms n_k, per species k or total individual organisms N per sample). This additional information complicates the process of data collection and was only available in parts of the studies in the GLOBIO3 database. We therefore performed this indicator comparison with a subset of the data: we chose all those studies from the biome (Sub-) Tropical and Moist Broadleaf Forest (i.e.“tropical rain forest”) in which a full species list indicating the abundance of each species in different land use types and a (semi-)natural reference was provided. The species abundance lists of these studies were extracted to Microsoft Excel to calculate the selected biodiversity indicators (see Table 1.1). Two indicators (Mean species abundance MSA and Sørensen S_s) directly calculate the relative change between a land use type *i* and a reference, for the other three indicators (species richness, Shannon’s entropy and Fisher’s α), the relative values per land use type LU_i and taxonomic group g within each study j were calculated as follows:

$$I_{rel,LU_{i,j,g}} = \frac{I_{LU_{i,j,g}}}{I_{ref_{j,g}}} \qquad \text{Eq. 1.6}$$

The numerical values range from 0-1 for the two indicators MSA and Sørensen S_s, whereas I_{rel} of the other three indicators species richness, Shannon’s entropy and Fisher’s α allow values above 1. For studies containing data from several reference situations, relative indicators were calculated for all possible combinations of references and land use types and also within references, giving an additional estimate of uncertainty. Hence, the reference situation was not fixed at 1 as was the case for the data on S_{rel} from the full dataset (BDM and GLOBIO3 database), where multiple reference plots per study site were averaged before the calculation of the relative indicator. This resulted in a final number of 168 (pairwise) data points for the reference and a total of 337 for all land use types.

Table 1.1: Biodiversity indicators calculated for a subset of studies from the biome (Sub-) Tropical Moist Broadleaf Forest. For the indicators marked with $^{(*)}$, the presented formulas are for calculating the *absolute* indicator values. *Relative* values are derived from Eq. 1.5.

Indicator type	**Name and reference**	**Data requirement**	**Formula**
Alpha diversity	Species richness *S* $^{(*)}$	Species numbers	n.a.
Sampling corrected alpha diversity	Fisher's α $^{(*)}$ (Fisher et al. 1943)	Species numbers and total number of individuals	$\frac{N}{S}=\frac{\left(e^{\frac{S}{\alpha}}-1\right)}{\frac{S}{\alpha}}$
Diversity measure	Shannon's entropy *H* $^{(*)}$ (Shannon 1948)	List of species and their rel. abundance	$H=-\sum p_k * \ln p_k$
Abundance measure	Mean species abundance of original species (*MSA*) (Alkemade et al. 2009)	List of species, original species and their rel. abundance	$MSA=\frac{1}{S_{ref}}\sum_k \frac{p_{k,\,LUi} * N_{LUi}}{p_{k,\,ref} * N_{ref}}$, for all species $k \in ref$ 1
Dissimilarity measure	Sørensen S_s (Sørensen 1948)	List of species	$S_s=\frac{2c}{S_{LUi}+S_{ref}}$

LU$_i$ = land use type *i*; *ref* = (semi-)natural reference; *S* = Number of species; *c* = number of shared species between two land use types; *N* = total number of individuals; n_k= number of individuals of species *k*; $p_k = n_k / N$ = relative abundance of species *k*.

* If abundance in land use type *i* was higher than in the reference habitat, MSA values were truncated at 1 (see Alkemade et al. 2009).

1.2.5. Statistical analysis

Analysis of Variance (ANOVA) was used to analyze the differences in mean relative species richness S_{rel}, depending on the four factors land use type (LU), taxonomic group (Taxa), biogeographic region (Biome), and data source (i.e. GLOBIO or BDM; Data), including the interaction of factors. Following model structure was tested:

$$S_{rel} = f(\mathrm{LU; Biome; Taxa; Data; LU \times Biome; LU \times Taxa; Biome \times Taxa; LU \times Data;}\\ \mathrm{Biome \times Data; Taxa \times Data; LU \times Biome \times Taxa; LU \times Biome \times Data; LU \times Taxa \times Data}) \qquad \text{Eq. 1.7}$$

As the data did not follow the assumption of normal distribution, we additionally applied the Kruskal–Wallis Test to test the difference of medians of S_{rel} of the four factors (without interaction). Mann–Whitney U Test was conducted for pairwise comparison of median S_{rel} of different land use types.

For each of the five indicators I_{rel} (see Table 1.1 and Eq. 1.6) calculated for a subset of data, the differences in means for the three factors LU, Taxa and biogeographic region (Realm) and their interactions were assessed with ANOVA with following model structure:

$$I_{rel} = f(\mathrm{LU; Taxa; Realm; LU \times Taxa; LU \times Realm; Taxa \times Realm; LU \times Taxa \times Realm}) \qquad \text{Eq. 1.8}$$

As with the total dataset, robustness of results was assessed with nonparametric Kruskal-Wallis Tests and Mann-Whitney U Tests. In addition, Pearson's correlation between indicators was calculated. All data analysis was carried out using R statistical package v2.11 (R Development Core Team 2011).

1.3. Results

1.3.1. Land use impacts on biodiversity

Characterization factors of land occupation CF_{Occ} for BDP were calculated according to equation (1.1) and are shown in Table 1.2 and in the Annex (see Table 8.1.1, Annex). For easier interpretation of results, the biodiversity indicator relative species richness S_{rel} is chosen for graphical display (Figs. 1.1 to 1.3). The CF can be derived by subtracting the median S_{rel} from 1 (see equation 1.1).

Averaged across all regions and taxa, relative species richness S_{rel} of all land use types was significantly lower than in the reference, but results varied strongly from negative impacts ($S_{rel} < 1$) to positive impacts ($S_{rel} > 1$) (see Fig. 1.1). The strongest negative impact was found in annual crops, where S_{rel} was reduced by 60%, followed by permanent crops and artificial areas (40% decreased S_{rel}). In pastures the reduction of S_{rel} was around 30%, in secondary vegetation, used forests and agroforestry around 20%. A pairwise comparison of the difference of median S_{rel} of different land use types is given in Table 8.1.2 (Annex).

A significant effect on S_{rel} of land use (LU), taxonomic group (Taxa) and biogeographic region (Biome) and a non-significant effect of the source of data (GLOBIO or BDM) were found for the full dataset both in ANOVA (Table 1.3) and Kruskal-Wallis Test (results not shown). In the ANOVA, land use effects on S_{rel} differed significantly between biomes (LU x Region) and taxa (LU x Taxa), but not between data source (LU x Data). The latter was supported by Mann-Whitney U Tests, which did not show any significant difference ($p<0.05$) in S_{rel} between the two data sources for any land use type (results not shown).

Table 1.2: World average and regionalized characterization factors CF (median) and their uncertainties (1. and 3. quartiles) for Biodiversity Damage Potential (BDP) per land use type. CF of four selected biomes are displayed, a full list of CF per biomes and taxonomic groups can be found in the Annex, Table 8.1.1. For land use types with less than 5 data points (N), no CF is provided

		Forest, not used	**Secondary vegetation**	**Forest, used**	**Pasture/ meadow**	**Annual crops**	**Permanent crops**	**Agro-forestry**	**Artificial areas**
Total world average	**Median**	**0**	**0.18**	**0.18**	**0.33**	**0.60**	**0.42**	**0.20**	**0.44**
	1.Quartile	0	-0.03	-0.05	0.00	0.31	0.06	0.01	-0.01
	3.Quartile	0	0.37	0.50	0.55	0.79	0.70	0.48	0.62
	N	326	272	148	133	96	52	76	53
Biome 1 (Sub-)Tropical Moist Broadleaf Forest	**Median**	**0**	**0.22**	**0.13**	**0.45**	**0.54**	**0.42**	**0.18**	**-**
	1.Quartile	0	0.00	-0.09	0.31	0.36	0.18	-0.02	-
	3.Quartile	0	0.43	0.45	0.75	0.72	0.70	0.44	-
	N	173	172	79	26	46	40	70	1
Biome 4 Temperate Broadleaf Forest	**Median**	**0**	**0.08**	**0.22**	**0.52**	**0.76**	**0.02**	**-**	**0.40**
	1.Quartile	0	-0.26	-0.09	-0.35	0.46	-0.11	-	-0.10
	3.Quartile	0	0.33	0.43	0.67	0.86	0.69	-	0.58
	N	46	20	35	33	24	9	0	24
Biome 5 Temperate Coniferous Forest	**Median**	**0**	**0.17**	**0.15**	**0.24**	**0.54**	**-**	**-**	**0.50**
	1.Quartile	0	-0.22	0.02	-0.64	-0.15	-	-	-0.05
	3.Quartile	0	0.30	0.33	0.38	0.87	-	-	0.71
	N	45	15	7	27	8	3	0	21
Biome 7 (Sub-)Tropical Grassland & Savannah	**Median**	**0**	**0.00**	**0.01**	**0.12**	**0.65**	**-**	**-**	**-**
	1.Quartile	0	-0.17	0.00	0.02	0.02	-	-	-
	3.Quartile	0	0.15	0.06	0.27	0.80	-	-	-
	N	21	27	6	8	9	0	0	0

Table 1.3: Results of ANOVA testing the difference in mean Srel of the full dataset depending on land use type (LU), taxonomic group (Taxa), biogeographic region (Region=Biome or Realmb) and data sources (Data) and their interactions. Model of full dataset, see equation (7); model for subsets (GLOBIO, BDM and Tropical data), see equation (8). MBF: Moist broadleaf forest; Df : degrees of freedom ; *** p-values<0.001; ** p-values<0.01; * p-values<0.05; (*) p-values<0.1; ns p-values > 0.1

	Full dataset		GLOBIO data		BDM data		Subset: Biome (Sub)-Tropical MBF data					
	Df	S_{rel}	Df	Srel	Df	Srel	Df	Srel	MSA	Sørensen	Shannon (rel)	Fishers
LU	7	***	7	***	6	***	7	***	***	***	***	***
Region[1]	8	***	8	***	1	ns	4	***	***	***	***	***
Taxa[2]	3	***	3	***	2	***	2	*	**	***	(*)	*
Data	1	ns	-	-	-	-	-	-	-	-	-	-
LU x Region	38	***	33	**	6	*	20	***	***	***	*	***
LU x Taxa	20	***	19	***	12	***	11	(*)	**	***	ns	ns
Region x Taxa	17	(*)	16	***	2	ns	5	ns	ns	ns	ns	ns
LU x Data	4	ns	-	-	-	-	-	-	-	-	-	-
Region x Data	1	(*)	-	-	-	-	-	-	-	-	-	-
Taxa x Data	1	ns	-	-	-	-	-	-	-	-	-	-
LU x Region x Taxa	45	(*)	34	*	12	ns	8	ns	ns	ns	ns	ns
LU x Region x Data	1	ns	-	-	-	-	-	-	-	-	-	-
LU x Taxa x Data	2	ns	-	-	-	-	-	-	-	-	-	-

[1] Biomes were used as the factor for testing regional differences for all datasets, except for the subset of data from the biome (Sub-)Tropical Moist Broadleaf forest, where biogeographic realms (i.e. continents) were used for regionalization.

[2] For BDM data, the factor levels of Taxa were vascular plants, moss and mollusks. For the other datasets, the levels were plants, vertebrates, arthropods, and other invertebrates.

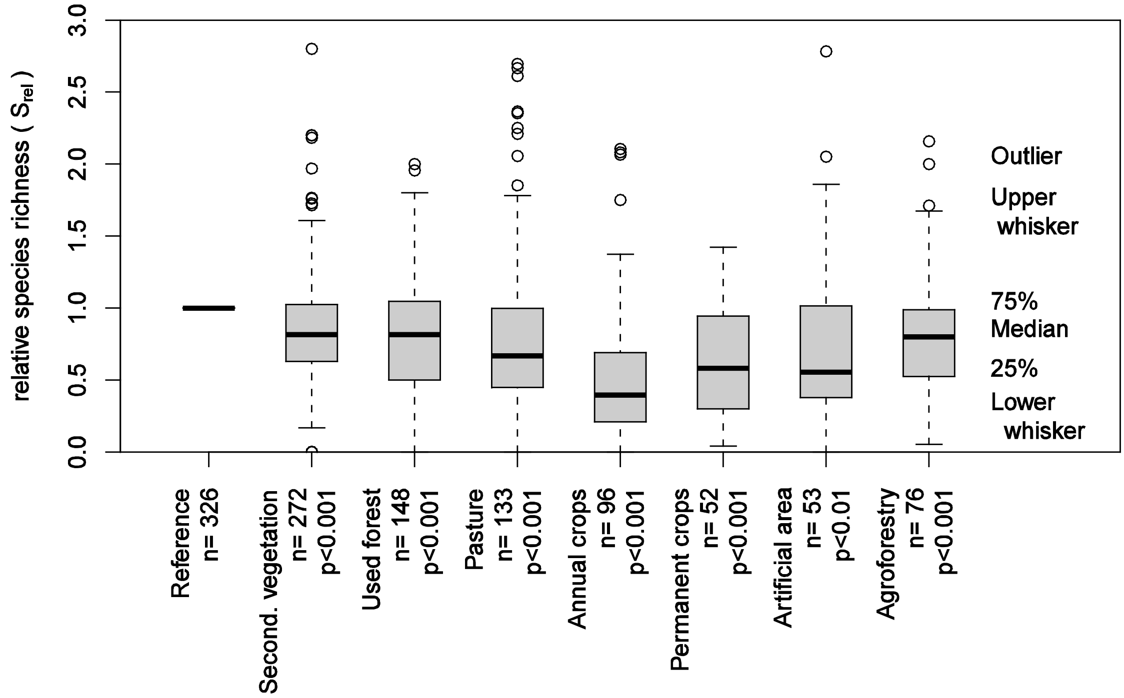

Figure 1.1: Box and whisker plot of relative species richness per land use type, number of data points n per land use type, and test statistics (1-sided Mann-Whitney U Test) of pairwise comparison of each land use with the reference for the full dataset (global averages across all biomes and taxonomic groups)

1.3.2. Regionalization

Data from nine biomes were included in the analysis, but the majority of studies provide data on land use of biome Tropical Moist Broadleaf Forests (Table 8.1.1, Annex). For many combinations of land use types and biomes, no or too little data was available to draw conclusions. Due to the inclusion of Swiss BDM data, the number of data points for two temperate biomes (Broadleaf & Mixed Forests and Coniferous Forests) was considerably improved. The relative species richness of the four biomes with the highest data availability is displayed in Fig. 1.2. A significantly different land use impact across biomes was only found for three land use types (secondary vegetation, used forest and pasture; Kruskal-Wallis Test, $p<0.05$). All land use types in all biomes showed a median negative land use impact ($S_{rel} < 1$), with one exception. Pastures in the biome "Deserts & Xeric Shrublands" showed a slight positive median land use impact (S_{rel} increased by 8%, Table 8.1.1, Annex), but the small number of data points (n=5) and the large variation in data does not allow for strong conclusions here. In general, large within biome variations were observed.

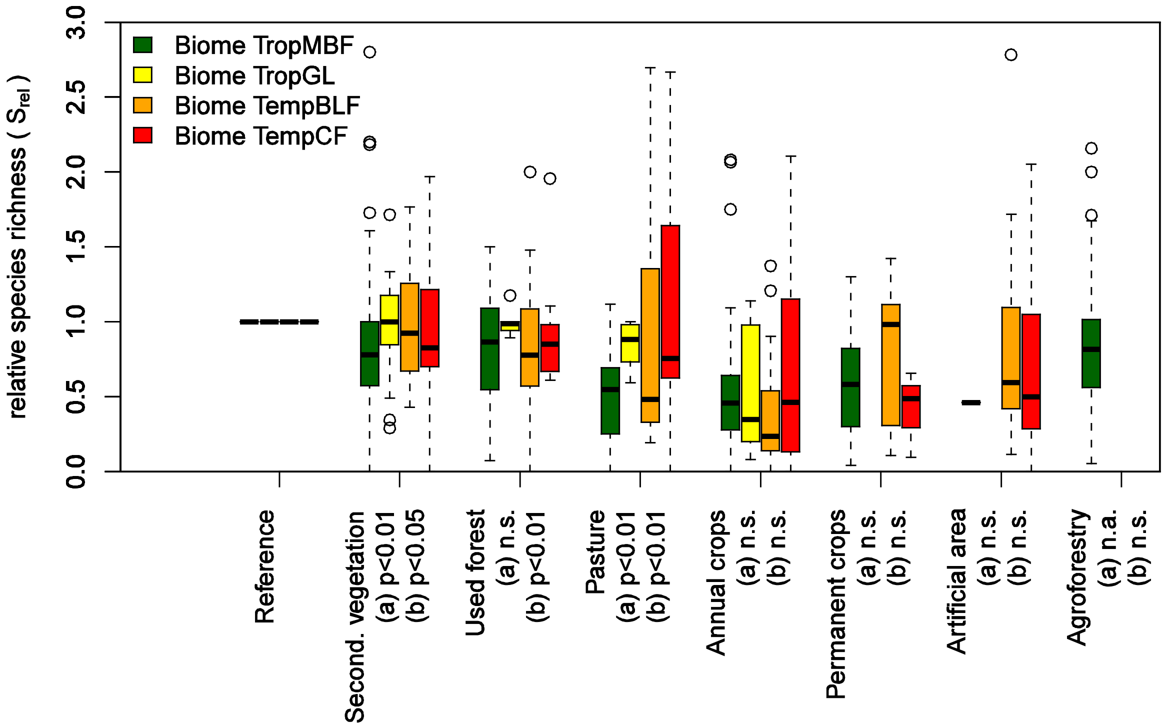

Figure 1.2: Box and whisker plot of relative species richness per land use type and four selected biomes and test statistics of Kruskal-Wallis Test (S_{rel} = f(LU x Biome)) for (a) displayed biomes and (b) all biome in full dataset. n.s.: not significant; n.a.: not applicable (just one biome per land use type); TropMBF: (Sub-)Tropical Moist Broadleaf Forest; TropGL: (Sub-)Tropical Grassland, Savannas and Shrublands; TempBLF: Temperate Broadleaf &Mixed Forest; TempCF: Temperate Coniferous Forest

1.3.3. Indicator comparison

Comparing impacts across taxonomic groups

Data on different species were aggregated into broad taxonomic groups to get enough data points per group and land use type. The global dataset from GLOBIO3 contains a broad range of taxonomic groups, whereas the Swiss BDM dataset only contains data on plants (vascular plants and moss) and invertebrates (mollusks, see Table 8.1.5, Annex). To achieve more or less globally averaged results, we further aggregated groups that were mainly consisting of data from the Swiss BDM. Finally, four classes of taxonomic groups were distinguished: plants, arthropods, other invertebrates and vertebrates. The impacts of agroforestry on different taxonomic groups were significantly (p<0.001) different and slightly different (p<0.1) for used forest (see Fig. 1.3). Overall, plants and invertebrates (excluding arthropods) showed a slightly stronger land use effect than arthropods and vertebrates, but this pattern was not found across all land use types. The variation within taxonomic group was considerable. Therefore, we further separated two groups, neglecting the over-representation of the Swiss data: plants were split into moss and vascular plants and vertebrates were split into birds and other vertebrates, resulting in a total of 6 taxonomic groups (Figure 8.1.3 and Table 8.1.1, Annex). With this more refined classification, significantly different (p<0.05) land use impacts across taxonomic groups were found for all land use types except for secondary vegetation, where only a slight difference (p<0.1) between taxonomic groups was found (Figure 8.1.3, Annex). Here, other invertebrates and moss showed the strongest land use impacts, with roughly a 50-90% median reduction in S_{rel} in pasture, annual crops and artificial area. A strong positive land use impact (42% increased S_{rel}) on vascular plants was found in artificial areas. The underlying data (n=16) was purely based on the Swiss BDM, and data from very heterogeneous land use types were included (from dump sites to urban green areas). Therefore, we further split the land use type artificial area in the Swiss BDM data into low and high intensity (Figure 8.1.4, Annex). However, no significant differences in S_{rel} between the high and low intensity artificial areas were found for the BDM data with a Mann–Whitney U Test. Within the Swiss BDM data, vascular plants were generally less affected by land use than moss and mollusks (Figure 8.1.4, Annex), and showed an increased median relative species richness in pasture, permanent crops and artificial areas. Moss and mollusks showed a decreased relative species richness in all land use types.

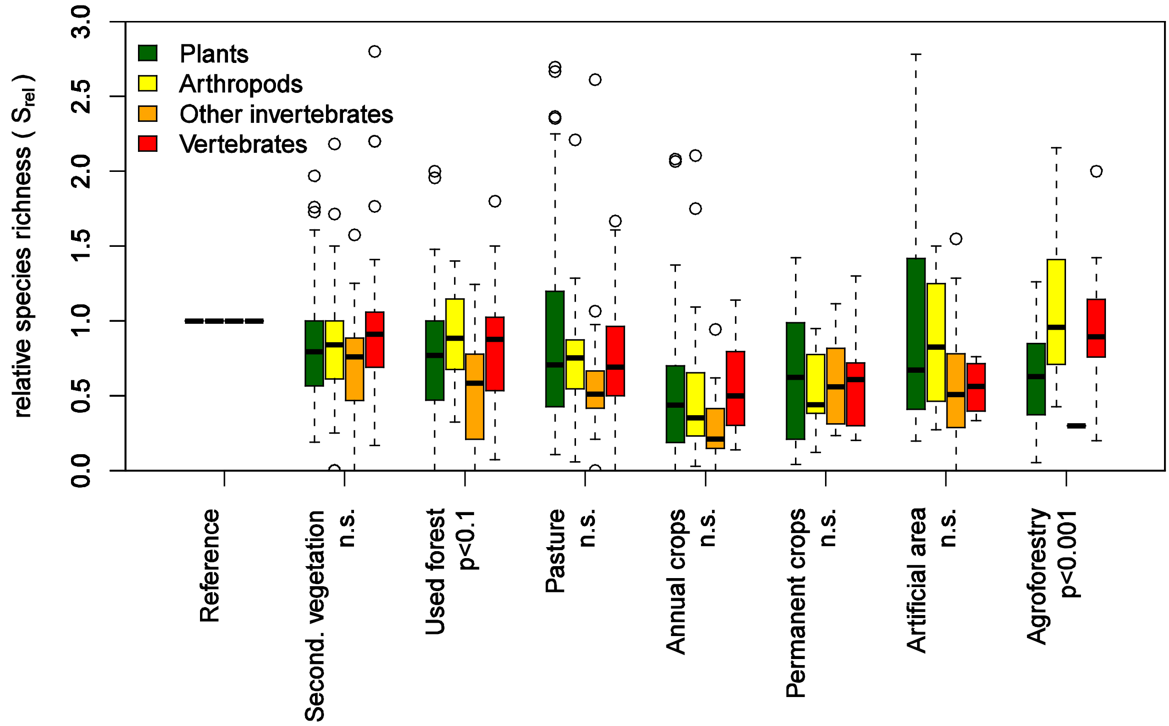

Figure 1.3: Box and whisker plot of relative species richness per land use type and taxonomic group and test statistics of Kruskal-Wallis Test (S_{rel} = f(LU x Taxa)) for full dataset. n.s.: not significant

Comparing impacts across biodiversity indicator

For a subset of data from the biome (Sub-)Tropical Moist Broadleaf forest, four additional indicators were calculated: Fisher's α, Shannon's entropy H, Sørensen S_s, and Mean species abundance of original species (MSA, see Table 1.1). For all land use types, the impacts varied significantly across indicator (Fig. 1.4). Relative species richness was highly correlated with relative Shannon's H (Pearson's r = 0.79) and relative Fisher's α (Pearson's r = 0.83, see also Table 8.1.3, Annex). This group of indicators showed less negative (or even positive) land use impacts compared to a second group of indicators, Sørensen's S_s and mean species abundance MSA, which were also highly correlated (Pearson's r=0.81). In Fig. 1.4, the reference situation shows a considerable within study variation, calculated as the relative difference in biodiversity indicators of multiple reference situations given for individual studies. This variation was not calculated for the full dataset (see Figs. 1.1 to 1.3), where the average of multiple references were used to calculate relative changes in species richness.

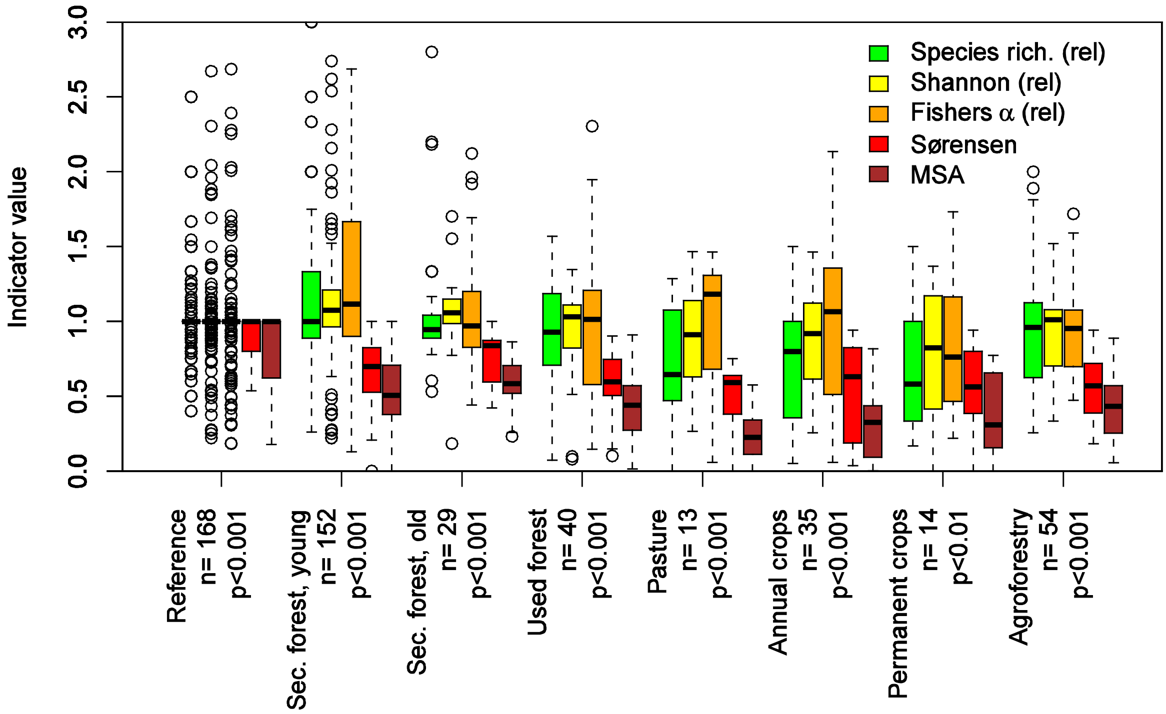

Figure 1.4: Box and whisker plot of different indicators per land use type and test statistics of Kruskal-Wallis Test (I_{rel} = f(LU x Indicator)) for a subset of data from biome (Sub-)Tropical Moist Broadleaf Forest. Secondary forest was divided into young (<30 years) and old growth forest (>30 years). n: Number of data points (pairwise comparisons) per land use type and indicator

1.4. Discussion

Biodiversity is a multi-faceted concept and it is difficult to express product related impacts of land use in a single indicator value. Our analysis illustrated the variability of results, ranging from positive to negative impacts of land use, but we also found an overall negative median impact on relative species richness across all analyzed land use types. Land use impacts differed significantly across taxonomic groups and biogeographic regions, but could not be determined for all world regions due to data limitations. The choice of indicator also strongly influenced the outcome, where relative species richness was less sensitive to land use impacts than MSA or Sørensen. In the following sections we want to highlight the uncertainties, limitations and opportunities for improvements of this first attempt to quantify land use impacts on biodiversity in LCIA on a global scale based on empirical data. We discuss the suitability of different indicators for use in land use LCIA, appropriate coverage and classification of taxonomic groups, land use types and biogeographic regions and finally address general uncertainties of the presented approach.

1.4.1. Choice of indicator

From a practical point of view, species richness might be the indicator of choice for biodiversity assessments on species level: data are relatively readily available, as the data requirements are low compared with other indicators, which need additional information on abundance and/or species composition. However, from a theoretical point of view, the indicator species richness has many shortcomings. Here, we discuss four alternative indicators analyzed in this study which partly overcome some of the disadvantages of species richness, such as (i) high dependence on sampling effort, (ii) missing information on abundance, (iii) no link to conservation targets, and (iv) missing information on species turnover.

First, species richness is highly dependent on sampling size, whereby a non-linear relationship between area sampled and species richness has been observed (due to a species sampling and a species area relationship; Arrhenius 1921; Dengler 2009). To compare absolute species numbers of different land use types, species richness has to be standardized to the same sampling effort (Koellner and Scholz 2008; Schmidt 2008). This proves to be difficult or even impossible when dealing with different taxonomic groups that are surveyed with very different sampling methods (e.g., visual observations of birds along 50m transects; 20 pitfall-traps of arthropods; or plant counts on $10m^2$ plots). Therefore, we divided absolute species numbers of each taxon of every land use type *i* by the absolute species numbers of a regional reference to obtain relative species richness (given that both absolute numbers were obtained with a similar sampling effort). This approach partly circumvents effects of sampling bias. However, a bias remains in cases where the land use type *i* and the reference show very different species turnover (e.g. homogeneous species composition of arable field vs. heterogeneous rain forest). In such cases, the relative land use impact is underestimated with small sample size, as most species of the arable field are captured, but only a small share of the species richness of the rain forest is recorded.

Several biodiversity indicators have been developed that correct for incomplete sampling (see e.g. Beck and Schwanghart 2010). In our analysis we applied Fisher's α (Fisher et al. 1943, see formula in Table 1.1) and found a high correlation between land use impacts measured as relative species richness and as relative Fisher's α (see Fig. 1.4). This finding is supported by the empirical study of Kessler et al. (2009), which did not find a strong influence of sampling incompleteness on land use impacts. This indicates that correcting for undersampling might not be the most important aspect to reduce overall uncertainty of biodiversity related land use LCIA.

A second shortcoming of species richness is the missing information on abundance. Shannon's entropy H, derived from information theory, expresses abundance and richness in one number (see formula in Table 1.1) and reaches a maximum value when all species occur equally abundant. As in our study relative Shannon's entropy H was highly correlated with relative species richness (see Fig. 1.4), the latter might be preferred as indicator, as it is easier to communicate to LCA users or the general public.

Mean species abundance (MSA), an indicator developed for the GLOBIO3 model (Alkemade et al. 2009), is correcting the second and third shortcoming of species richness as it includes abundance and is linked to conservation targets. MSA compares the abundance of "original" species occurring in natural, undisturbed habitats, in their primary "original" habitat with their abundance in secondary habitats (i.e., on the land use type i). As expected, our results showed that land use impacts were stronger (more damaging) when measured with MSA than with S_{rel} (see Fig. 1.4), indicating that the original species adapted to undisturbed habitats are more susceptible to land use changes than species adapted to disturbance. MSA is therefore suitable to report land use impacts in regions, where conservation targets are mainly focusing on protection of primary habitats. In areas such as central Europe, where conservation is mainly targeting at protecting species adapted to traditional land use practices, the definition of "original" species needs to be extended to these species. To base future land use LCIA methods on MSA, the habitats or species relevant for conservation have to be defined for all world regions, whereby the value choices unavoidably involved in this definition need to be critically discussed.

Similar to MSA, Sørensen S_s can measure the similarity of the species composition of a land use type and a reference situation but without considering abundance. As they were calculated in this study, both MSA and Sørensen reached a maximum value of 1, when the land use type had a maximum similarity (i.e., the same species composition as the reference) and the two indicators were therefore highly correlated (see Fig. 1.4). As expected, the land use impacts measured with Sørensen were smaller than with MSA, as MSA already reports a decreasing abundance of species, whereas Sørensen only reports if a species is not present anymore. Sørensen S_s can also be used to calculate similarity of species composition within a land use type or reference, i.e. giving information of species turnover (or β-diversity). In that case, a maximum β-diversity score would be reached within a land use class or reference with minimum average similarity between samples, indicating high rates of turnover. This would require data on species composition of multiple plots of the same land use and multiple plots of a reference within one study site or studies directly reporting β-diversity. As β-diversity can play a key role in biodiversity conservation (Gardner et al. 2010), this information is increasingly available and might open the way to use this indicator in future biodiversity LCIA.

Biodiversity impacts can be assessed in relative or absolute terms, which finally represent different value choices: if we assess absolute impacts, all species are equally weighted, if we assess relative impacts, all ecosystems get equal weight. The indicators calculated in this study all assess relative impacts. As explained above, this was required to standardize the data from a multitude of studies with different sampling design and species groups. As a consequence, a 40% decrease of relative species richness in a species rich ecosystem (e.g., with 100 species) and in a more species poor ecosystem (e.g., 10 species) are weighed evenly, although the absolute reduction in species richness is much higher in the species rich ecosystem (40 vs. 4 species). To account for regional differences in absolute species richness, a weighing system of land use could be applied as suggested by Weidema and Lindeijer (2001). Global conservation priorities could help to develop such a weighting scheme, using weighting factors such as regional species richness, irreplaceability and vulnerability of ecosystems (Brooks et al. 2006).

To conclude, we think that—in view of current data availability—relative species richness, as an indicator for α-diversity, is a suitable indicator for biodiversity related land use LCIA. If future research progress allows quantifying land use related impacts on β-diversity or changes in abundance of species important for conservation, indicators such as MSA or Sørensen should be preferred. To also account for regional differences of absolute species richness, a weighting of the presented CFs is required. Here, only a few facets of biodiversity were considered, with a focus on species composition. Including other facets of ecosystem quality, for example land use impacts on ecosystem functioning (see e.g. Michelsen 2008; Wagendorp et al. 2006) or on ecosystem services (Müller-Wenk and Brandão 2010; Saad et al. 2011; Brandão and Milà i Canals 2012), would be an important complement of this method.

1.4.2. Taxonomic coverage

Attractive species groups, such as mammals, birds or butterflies are often used as indicator taxa in biodiversity assessments, with the underlying hope “that the known biodiversity is a good surrogate for the unknown” (Rodrigues and Brooks 2007, p. 714). Data availability is therefore biased towards some well-studied species groups. Existing land use LCIA mostly focused on vascular plant species richness as an indicator (e.g. Koellner and Scholz 2008; Schmidt 2008; De Schryver et al. 2010). This makes a method very transparent, but the potential to generalize results from one well-studied species group to biodiversity as whole is questionable (Purvis and Hector 2000). Empirical studies from different world regions found little predictive power of one species group for other species groups (e.g. Billeter et al. 2008; Kessler et al. 2009; Wolters et al. 2006).

In this study, we combined data from global literature review, covering a range of taxonomic groups (see Table 8.1.5, Annex), with data from Swiss biodiversity monitoring BDM, containing data on vascular plants, moss and mollusks. Although being more representative than previous LCIA studies, a publication bias towards some well studied species groups remained. For the averaged results, species groups with more data points got a higher weight than groups with fewer data points. If we compare the share of species groups in our analysis with their estimated global species richness (Heywood and Watson 1995; see Table 8.1.4, Annex), we find that vertebrates (26% of data) and plants (43%) were strongly overrepresented, as they only make up 0.4% and 2% respectively of estimated global species richness. With 20% of data points, arthropods were underrepresented in this study as they make up an estimated 65% of global species richness. Some species groups, such as bacteria (7% of estimated global species richness) or fungi (11%) were not at all represented in the used dataset.

Ideally, the impact of different land use types on each target species group in each biogeographic region should be separately assessed. This could later be aggregated into characterization factors for archetypical groups of species, regions and land use types showing similar land use effects. In this study, we present one possible classification, but due to limited data availability we could not make a thorough analysis of different classification options nor recommend an optimal classification, where the variation within each characterization factors is minimized (i.e., representing a homogeneous group). We first divided data into four very broad taxonomic groups (plants, arthropods, other invertebrates and vertebrates), and then further subdivided plants into moss and vascular plants and vertebrates into birds and other vertebrates. A further subdivision was not possible, as too little data points were then given for each land use type. Especially for plants, the separation into moss and vascular plants revealed very diverging impacts of these subgroups (Table 8.1.1 and Figure 8.1.3, Annex), suggesting that these groups should be assessed separately. However, it is unclear to which extent this conclusion is valid for other world regions, as for many land use types the data on moss is purely derived from the Swiss BDM (see Table 8.1.5, Annex).

Here we chose a classification based on phylogenetic relationship, but closely related species not necessarily show homogeneous reactions to land use (see e.g. Attwood et al. 2008; Blaum et al. 2009; Anand et al. 2010). To find an optimal representation of impacts across species groups other classification criteria, such as functional traits (e.g. morphological, ecophysiological and life history characteristics, see e.g. Vandewalle et al. 2010) or feeding-guilds (see e.g. Scherber et al. 2010) should be tested as alternative grouping factor for species groups.

As data on all species will probably never be available, we need to find the optimal taxonomic coverage for land use LCIA. This requires a clear definition of the goals of including biodiversity into LCIA (i.e., why we want to conserve biodiversity; see also Michelsen 2011). If we aim at conserving biodiversity due to its intrinsic value or due to its potential future economic value (e.g. as medicine), threatened species should get higher weights and species groups selected for LCIA should be proportional to their total richness. If the target is to sustain ecosystem services, we need to conserve functional diversity (and assess land use impacts on species important for ecosystem functioning). However, this requires a more sound understanding of the underlying ecosystem processes, e.g. on how ecosystems react if a certain species occurs more or less abundantly. In addition, better knowledge on vulnerability and potential tipping-points of ecosystems (i.e., non-linear reactions of ecosystems after certain levels of accumulated multiple disturbances, see e.g. Holling 2001; Scholz 2011) is required. Resolving the important normative question of setting appropriate targets for biodiversity assessments within LCA and of finding the right proxy for it remains a challenge for future research.

1.4.3. Land use classification and regionalization

As outlined above, characterization factors ideally should represent archetypical land use impacts on species groups, but also of land use classes and regions showing similar impacts. In our study, very broad land use types were classified showing considerable within class variation of effects. Including further data points would allow to separate intensive and extensive land use (e.g., for agriculture) and could potentially reduce this variation and improve the validity of the characterization factors. However, in the case of artificial area in Switzerland, no significant differences between high and low intensive artificial area were found (Figure 8.1.4, Annex). Caution should be taken with extrapolating the findings for artificial area, which are largely based on the Swiss BDM data, to other world regions.

The question of appropriate classification also applies to regionalization. Here, we chose WWF biomes as spatial units as a coarse regionalization scale with ecologically distinct regions. Due to limited data availability, it was not possible to have a more finely scaled regionalization of relative impacts. However, a weighting of these relative impacts, as suggested above, could be done on ecoregion level, using for example data on species richness of different taxa (see data of Olson et al. 2001; Kier et al. 2005). As significant differences in land use impacts were not only found across biomes (full dataset), but also across biogeographic realms (subset (Sub-)Tropical Moist Broadleaf Forest, see Table 1.3), a further distinction of biomes across realms might better reflect differences in relative impacts. The analysis of the Swiss BDM data, covering two biomes, showed no significant difference between their reactions to land use. This suggests that not only the broad ecosystem type is important to determine land use impacts, but also the geographical proximity or similarity of land use history. Of course, aspects of practicality also need to be considered when choosing an optimal scale of regionalization. Finally, to assess land use impacts in LCA, not only the characterization factors have to be regionalized, but also the inventory data. How the presented CFs can be applied is illustrated in a case study on margarine by Milà i Canals et al. (2012).

As for taxonomic groups, data availability of land use impacts on biodiversity is biased towards some biogeographic regions, with data dominantly derived from a few well-studied research stations in tropical regions (see Gardner et al. 2009, 2010). In addition, some ecosystem types, such as grassy ecosystems, received less attention of researchers than forest ecosystems (Bond and Parr 2010). The uneven regional distribution is also visualized in the data distribution of this study (see Figure 8.1.2, Annex). Very little or no data were available for following five out of fourteen biomes: (Sub)-Tropical Coniferous Forests, Boreal Forests/Taiga, Flooded Grasslands & Savannas, Tundra, and Mangroves (see Table 8.1 and Figure 8.1.2, Annex). For three biomes, enough data were only provided for pastures (Temperate Grassland & Savannah, Mediterranean Forests, Woodlands & Scrub, and Deserts & Xeric Shrublands). For permanent crops, agroforestry and artificial areas, data were only available from two biomes. In general, the biome (Sub)-Tropical Moist Broadleaf Forest had the highest data availability. The two temperate biomes Mixed & Broadleaf and Coniferous Forest also showed a reasonable amount of data, but as this was mainly derived from Swiss BDM data, results are highly biased towards the European context. To which extent these results are valid for temperate forest biomes in other world regions remains a question for future research.

1.4.4. Data limitations and uncertainties

In this study we combined global literature data with national biodiversity monitoring data. Both datasets have different sources of uncertainties. Summarizing data from multiple studies involves consideration of within and between study variance (Gurevitch and Hedges 1999). As it was beyond the scope of this study to perform a full statistical meta-analysis, only between study variation was considered. Therefore, the overall assessment on relative species richness suggests no variation of the reference habitat (see Fig. 1.1 to 1.3), which does not reflect reality. For the subset of data used to compare biodiversity indicators, the within-study variance was included when studies reported data on multiple reference habitats. The considerable variation of indicator values of reference habitats observed in the subset of data (see Fig. 1.4) suggests that variation of results (including within-study variance) of the full dataset was underestimated.

For the Swiss BDM data, the main sources of uncertainties are the definition of ecologically similar regions (see section 2.3) and the definition of reference habitat. It was beyond the scope of this study to test the sensitivity of results to choice of boundary of regions. However, for the definition of reference habitat per region, the sensitivity of results to selection of two different reference situations was tested. We compared the outcomes for using (i) a combination of all potential natural habitats (n=305 monitored sampling points in forests, grasslands, wetlands, bare areas and water bodies) and (ii) only forest sampling points (n=221). No significantly different result of any land use type was found between the two alternative reference situations. Although there is a large overlap of data points between the two alternatives, it indicates that results are not very sensitive to choice of reference habitat. However, in both alternatives the reference habitat experienced considerable past (and present) human disturbance, as no pristine areas exist in Switzerland, whereas more pristine reference habitat was included for other biomes in the GLOBIO3 database. However, this inconsistency is unavoidable when a globally valid reference situation has to be defined, as different world regions show different land use history.

In our study, we found a median reduced relative species richness across all globally averaged land use types. However, we cannot rule out that other factors, such as changes in overall landscape composition or pollution might also have contributed to the result. A meta-study across multiple taxonomic groups in the Western Ghats, India, for example found no significant effect of land use on species richness, but a significant effect of native forest cover within the landscape (Anand et al. 2010). Besides the necessity to understand cause-effect chains of biodiversity loss, this illustrates the importance of spatial context of land use (i.e. in what landscape a land use occurs). Despite their importance, it was beyond the scope of this study to include spatial and temporal effects. To improve the assessment of biodiversity loss related to land use or other drivers of biodiversity loss, better concepts including these temporal and spatial aspects are required for LCIA (see also Curran et al. 2011).

1.5. Conclusion and recommendations

Although uncertainties and data and knowledge gaps are considerable, human impacts on biodiversity are ongoing. Decisions how to adapt production towards being less harmful for biodiversity need to be taken urgently, and cannot wait until all data and knowledge gaps are filled. Based on empirical data, this study provides a first attempt to quantify land use impacts on biodiversity within LCA across world regions to support such decisions. Due to the mentioned challenges to quantify biodiversity impacts, the presented characterization factors (CF) should be used with caution and remaining uncertainties should be considered when LCA results are interpreted and communicated. In LCA studies, where the “user may not directly decide on the land management practices” (Milà i Canals et al. 2007, p. 13), our CF can serve as a first screening of potential land use impacts across global value chains. For LCA studies aiming to support decisions of specific land management, a more detailed, site-dependent assessment, including additional region- or site-specific data, is indispensable (see e.g. Geyer et al. 2010).

In this paper, occupation impacts of a range of land use types in many world regions could be assessed, but some data gaps remain. Research priorities should be set to first close data gaps for environmentally important land use activities (such as agri- and silviculture, construction, mining and land filling) in economically important world regions (e.g. by using regionalized global inventories such as the inventory of global crop production from Pfister et al. 2011). To assess total land use impacts on biodiversity, we need to complement the presented CF of occupation with regionalized global estimates of transformation impacts. This requires more reliable information on regeneration times of ecosystems across the world, as transformation impacts (calculated according to the UNEP/SETAC framework; Milà i Canals et al. 2007; Koellner et al. 2012b) are highly sensitive to this parameter and currently available estimates vary considerably (Schmidt 2008). Estimates of regeneration times should ideally be based on empirical data, for example derived through meta-analysis of ecosystem regeneration studies.

In view of current data availability, the applied indicator relative species richness is suitable for biodiversity related global land use LCIA. As ecological research evolves, LCIA methods should be complemented with indicators measuring other facets of biodiversity, such as conservation value, species abundance or turnover. This applies not only to land use impacts, but also to other drivers of biodiversity loss, such as climate change, eutrophication, acidification or ecotoxicity. To inform decision-makers about potential trade-offs of different drivers of biodiversity loss along the life cycle, indicators need to be comparable across impact pathways (see also Curran et al. 2011). Finding a measure to quantify impacts of concurrent multiple drivers of biodiversity loss in a globally applicable and spatially differentiated way will be a challenge for future LCA research. As the importance of halting global biodiversity loss is increasingly recognized in research, industry and policy (e.g. formulated as the 2020 targets of the Convention on Biological Diversity; CBD 2010), increased research efforts are made to close some of the mentioned knowledge and data gaps. This will also open the way to improve the accuracy of biodiversity assessments within LCA and allow for more robust and credible decision-support.

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2. Land use impacts in LCA: Global characterization factors based on regional and global potential species extinctions[§]

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2.1. Introduction

Decision-makers in business and governmental organizations increasingly consider the environmental consequences of economic activities. Life cycle assessment (LCA) is a methodology that can inform their decisions by quantifying various environmental impacts over the entire life cycle of products. LCA was originally developed for the industrial sector, but is also used to support decision-making in other sectors, such as the agricultural. Here, an assessment of the impacts of land use on biodiversity is crucial, as agriculture currently occupies about one third of the global terrestrial surface (Lambin and Meyfroidt 2011), and this large-scale modification of land has been identified as one of the main drivers of global biodiversity loss (Sala et al. 2000; Millennnium Ecosystem Assessment 2005; Alkemade et al. 2009; Pereira et al. 2010). However, no consensus exists on how these impacts could be meaningfully quantified within LCA. This hampers a comprehensive assessment of environmental impacts in land intensive sectors, and reduces the credibility of LCA results in these sectors.

Biodiversity is a relatively young and multifaceted concept. The term “biodiversity” was coined only in the mid 1980s (Wilson and Peter 1988). Biodiversity encompasses different hierarchical levels of life (genes, species, populations, ecosystems) and their various attributes (composition, function and structure; Noss 1990). The proposed indicators for measuring impacts on biodiversity in LCA are thus manifold. Many methods focus on the composition of species, but some studies also include other aspects, such as ecosystem scarcity and vulnerability (Michelsen 2008; Weidema and Lindeijer 2001); see also the reviews of Lindeijer (2000b) and Curran et al. (2011). Globally applicable life cycle impacts assessment (LCIA) methods for quantifying bio-physical impacts of land use on ecosystem services were developed recently (Brandão and Milà i Canals 2013; Müller-Wenk and Brandão 2010; Saad et al. subm.). For assessing impacts on species diversity, most methods were developed for specific world regions, such as Central and Northern Europe (De Schryver et al. 2010; Jeanneret et al. 2009; Koellner 2000; Koellner and Scholz 2007, 2008; Schmidt 2008; Vogtländer et al. 2004),

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[§] This chapter has been submitted in a similar form to Environmental Science and Technology (de Baan, Mutel, Curran, Hellweg, Koellner, subm.)

Malaysia/Indonesia (Schmidt 2008), Japan (Itsubo and Inaba 2012) or California (Geyer et al. 2010). Extrapolating results from one region to others poses a challenge, as biodiversity varies strongly across regions, and the required input data is often only available for one or a few countries or regions. Data availability also limits the range of taxonomic groups that are assessed within LCA. Although results have been shown to be dependent on the choice of the assessed taxonomic groups (de Baan et al. 2013; Koellner and Scholz 2008), most methods are still based on a single taxon, mostly vascular plants (De Schryver et al. 2010; Itsubo and Inaba 2012; Koellner 2000; Lindeijer 2000a; Schmidt 2008; Vogtländer et al. 2004; Weidema and Lindeijer 2001), but also vertebrates (Geyer et al. 2010). Only a few authors proposed globally applicable land use LCIA methods. Two early studies are based on global species richness maps of vascular plants, estimating the biodiversity value of different land use types (Weidema and Lindeijer 2001) or extrapolating data from one country to the rest of the world (Lindeijer 2000a). One recent study is based on empirical data of multiple taxonomic groups, calculating the relative difference in species richness of different land use types and regional reference habitat for different biomes (de Baan et al. 2013). However, this previous study only considers land use (occupation) but not land use change (transformation), has a quite coarse spatial resolution (biomes) and only considers local impacts.

Here, we focus on species extinction – the aspect of biodiversity loss that is most prominently discussed in public debates and easy to communicate to decision-makers. Land use activities can contribute to local, regional or even global species extinction, which show different degrees of reversibility. For example, converting one hectare of rain forest to cropland can lead to a local displacement of about 50% of all species (de Baan et al. 2013). If viable populations of these locally displaced species still exist in adjacent rain forest areas they can gradually recolonize the cleared field after it is abandoned. In this case, local impacts are (to a large extent) reversible, although decades or centuries might be needed for full biodiversity recovery (Curran et al. subm.). If only a little rain forest habitat remains within the region, the risk for regional extinction of some species increases (Swift and Hannon 2010). Reintroducing regionally extinct species might still be possible, if sufficient suitable habitat can be provided in the region and species have the ability to recolonize the area. However, if the region contains the full geographic range of a species (i.e. species are endemic), there is a high risk that the species will become globally extinct (Purvis et al. 2000), which is fully irreversible.

When using species extinction as an indication of biodiversity loss in LCA, it is therefore essential to clearly define the spatial scale of impacts. The existing land use LCIA methods differ in the spatial scales at which they assess impacts. While some studies address both local and regional impacts (Koellner 2000; Schmidt 2008; De Schryver et al. 2010; Weidema and Lindeijer 2001; Vogtländer et al. 2004), others just consider local (Koellner and Scholz 2007, 2008; de Baan et al. 2013; Lindeijer 2000a) or regional impacts (Geyer et al. 2010; Itsubo and Inaba 2012). To our knowledge, no existing land use LCIA method assesses global species extinction although this is fully irreversible and of high political relevance. Avoiding global extinction is a declared target of international agreements (Convention on Biological Diversity 2012).

Here, we present a spatially explicit approach to assess the impacts of land use on biodiversity at regional and global scales. We model the potential regional species loss due to total accumulated land use activities within all global WWF ecoregions (Olson et al. 2001) and use this as a basis for calculating characterization factors for LCIA. We distinguish between potentially reversible impacts (i.e. regionally extinct, non-endemic species) to calculate land occupation and transformation impacts, and irreversible impacts (i.e. global extinction of endemic species) to calculate permanent impacts. We calculate characterization factors for five taxonomic groups (mammals, birds, amphibians, reptiles and plants). Finally, we analyze and quantify the uncertainties of the characterization factors with Monte Carlo calculations.

2.2. Methods

2.2.1. *Modeling species extinction*

A species-area relationship (SAR) model was used to assess the number of species that might be driven to extinction due to land use. This model is derived from island biogeography theory (MacArthur and E.O.Wilson 1963), describing a power relationship between the area *A* of an ecosystem and the number of species *S* it contains, where *c* and *z* are constants (Arrhenius 1921).

$$S = cA^z \tag{1}$$

SAR models are commonly used to predict species extinction due to habitat loss (Brooks et al. 2002; Millennnium Ecosystem Assessment 2005; Pimm et al. 2006). The number of species S_{new} of an area A_{new} is then calculated as a function of the species S_{org} occurring in the original habitat area A_{org} (Koh and Ghazoul 2010).

$$\frac{S_{new}}{S_{org}} = \left(\frac{A_{new}}{A_{org}}\right)^z \tag{2}$$

A shortcoming of the SAR model is that it traditionally focuses on natural habitats and assumes that no species survive on human-modified land (the so called “matrix”), although in reality this land provides habitat for some species groups (e.g. farmland birds; Pereira and Daily 2006; de Baan et al. 2013). We therefore used an adapted, matrix calibrated species-area relationship model (hereafter, matrix SAR) developed by Koh and Ghazoul (2010). This model predicts lower species extinction risks of habitat conversion, when the converted region contains suitable habitat for species. Mathematically, it lowers the curve of the species-area relationships (Koh and Ghazoul 2010) by adapting the *z* value (eq 3, Koh and Ghazoul 2010) of the power model (eq 2). The suitability of the matrix is dependent on the sensitivity *σ* of species to all *n* land use types *i* and on the composition of the matrix, expressed as the relative area share *p* of each land use type *i* from the total converted land area.

$$z' = z\sum_i^n p_i \sigma_i \tag{3}$$

The sensitivity *σ* is quantified as the relative decrease in species richness (*S)* between a land use type *i* (S_i) and a (natural) reference habitat (S_{ref}). This equals local land use characterization factors CF_{loc} as calculated by de Baan et al. (2013) per WWF biomes.

$$\sigma_i = CF_{loc,Occ,i} = \frac{S_{ref} - S_i}{S_{ref}} \tag{4}$$

The species lost S_{lost} per taxonomic group g due to all land use in one ecosystem is thus given by substituting eq 3 and 4 into eq 2 (Koh and Ghazoul 2010).

$$S_{lost,g} = S_{org,g} - S_{new,g} = S_{org,g} - S_{org,g} * \left(\frac{A_{new}}{A_{org}} \right)^{z \sum_{i}^{n} p_i CF_{loc,i}} \tag{5}$$

We chose WWF ecoregions (Olson et al. 2001) as spatial units for calculating species loss due to land use. Ecoregions contain distinct assemblages of species communities and their boundaries approximate the original extent of natural communities prior to major land use change (Olson et al. 2001).

As the $CF_{loc,Occ,i}$ range from positive to negative values (representing a higher species richness on used land than the reference habitat; de Baan et al. 2013), the exponential term in eq 5 could turn negative, resulting in a regional species gain (negative S_{lost}). Although human land use can sometimes increase the regional species pool (Zobel 2001; Zobel 1997), the matrix SAR was not developed to model this aspect. We thus did not allow regional species gain (negative S_{lost}) by setting $\Sigma p_i CF_i < 0$ to $\Sigma p_i CF_i = 0$ resulting in $S_{lost,g} = 0$.

2.2.2. *Calculation of characterization factors*

According to the UNEP/SETAC Life Cycle Initiative (Milà i Canals et al. 2007; Koellner et al. 2013), three types of land use impacts can be distinguished in LCA. First, land is transformed (*transformation impact*) to prepare for the actual land use (*occupation impact*). During land occupation, a natural regeneration of biodiversity is prevented, but as soon as the land is abandoned a recovery process begins. Occupation and transformation impacts are thus considered fully reversible given large enough time horizons. However, *permanent impacts* can occur if the regeneration potential of an ecosystem is irreversibly compromised. On the species level, we interpret this as inferring that species cannot recolonize the area in question because they are globally extinct.

We used eq 5 to calculate the total number of non-endemic species lost per ecoregion j and taxonomic group g due to all land use within each region. This total regional damage was then allocated to the different land use types according to the area share they occupy and their habitat quality. The allocation factor a is then calculated for each land use type i and ecoregion j as follows (6):

$$a_{i,j} = \frac{p_{i,j} CF_{loc,i,j}}{\sum_{i}^{n} p_{i,j} CF_{loc,i,j}} \tag{6}$$

Regional characterization factors for occupation of each land use type, $CF_{Occ,reg,i}$, were calculated by multiplying the potentially lost non-endemic species per region with the corresponding allocation factor $a_{i,j}$ and dividing this by the area occupied by the land use type, A_i. This finally gives us [potentially lost non-endemic species / m^2] as a unit.

$$CF_{reg,occ,i,j,g} = \frac{S_{lost,nonend,j,g} * a_{i,j}}{A_{i,j}} \tag{7}$$

Characterization factors for transformation were calculated as a multiplication of CF_{Occ} with half the regeneration time (Milà i Canals et al. 2007; Koellner et al. 2013), t_{reg} (eq 8). Here, the unit is [potentially lost non-endemic species *years / m^2].

$$CF_{reg,trans,i,j,g} = 0.5 * t_{reg,i,j,g} * CF_{reg,occ,i,j,g} = 0.5 * t_{reg,i,j,g} \frac{S_{lost,nonend,j,g} * a_{i,j}}{A_{i,j}} \tag{8}$$

Permanent impacts were calculated based on the total number of potentially lost endemic species per ecoregion *j* and taxonomic group *g*, due to all the accumulated land use within the region. This total damage was allocated to the different land use types within the region (multiplication with $a_{i,j}$) and divided by the area of each land use. As global extinction is fully irreversible, and the impact potentially continues during an infinite time, we calculated the impacts only over a modeling period, $t_m = 10^6$ years, representing the estimated time for a new species to evolve (Smith et al. 2012; Weir and Schluter 2007). The unit is [potentially lost endemic species *years / m^2].

$$CF_{reg,perm,i,j,g} = t_m * \frac{S_{lost,end,j,g} * a_{i,j}}{A_{i,j}} \tag{9}$$

For application in an LCA study, the CF_{Occ} are multiplied with the inventory flow of occupation, given in [m^2*years], and for calculating transformation and permanent impacts a multiplication with the inventory flow of transformation, in [m^2], is performed. Ultimately, the three impacts could be summed up into a total regional Biodiversity Depletion Potential for each taxonomic group g (rBDPg) expressed in the unit [potentially lost species * year / m^2]. By choosing the modeling time for the permanent impacts, a weighing can be performed on how damaging a global species loss is considered compared to a regional species loss. A proposal on how to aggregate CFs across taxa and how to derive world average CFs for land use flows with unspecified location is given in the Supporting Information (Annex).

2.2.3. *Input data for model parameters*

For each of the aforementioned parameters, we used globally available data and, where possible, also created an uncertainty distribution for each parameter (see overview in Annex Table 8.2.1). CF_{loc} (de Baan et al. 2013; Alkemade et al. 2009)*, z (Drakare et al. 2006)* and t_{reg} (Curran et al. subm.) were derived from global meta-studies and data was subset into various configurations based on data availability and relevance. The CF_{loc} were shown to differ significantly across biome and land use types (de Baan et al. 2013), thus we split the data into CF_{loc} specific per land use type and biome. If less then 5 data points per land use type and biome were available, world average CF_{loc} were used for the land use type. *z*-values strongly differed between broad habitat types (Drakare et al. 2006) and were calculated specifically for islands, forests and non-forest ecoregions (see Annex for how ecoregions were assigned to these habitat types and Annex Table 8.2.2 for applied *z*-values).

Recovery times t_{reg} were estimated based on a meta-analysis by Curran et al. (subm.). This study reviewed 109 peer-reviewed publications that compared the diversity of old growth (OG) and secondary growth (SG) habitat of different age, using the occurrence based Sørenson similarity index as a measure for diversity. Curran et al. (subm.) used generalized linear models to predict the time it would take for SG-OG similarity to reach average OG-OG values based on within-study comparisons. Predictors of recovery included habitat age, taxon, latitude, altitude, previous disturbance intensity, biogeographic realm and a simplified biome classification. Based on the model parameters for these predictors, we calculated land use-, taxon- and region-specific recovery times for 520 archetypical situations (parameter combinations): recovery after “intensive” (agriculture and urban land) or “extensive” (pasture and managed forests) land use for each taxonomic group (plants, birds, mammals and herpetofauna) in 65 world regions (WWF biomes per WWF realms). For each region, the median distance from equator, median elevation, and biome type (forest/non-forest) were specified and used to calculate recovery times (see Annex Table 8.2.3 for input parameters and resulting median recovery times).

Data on original species richness and endemism of mammals, birds, amphibians and reptiles were derived from the WWF database (Olson et al. 2001). As no data on plant endemism is available per ecoregion, (Holger Kreft, pers. comm.), permanent impacts could not be calculated for plants. Total plant species richness from Kier et al. (2005) was used for calculating occupation and transformation impacts.

Several global land cover/use maps are available, but their agreement on cover types and distribution is limited (Fritz and See 2008; Fritz et al. 2010; Tchuenté et al. 2011). Most maps do not distinguish between natural, managed or inhabited forests or grassland (Bartholomé and Belward 2005; European Space Agency 2009). We thus chose two maps, LADA (Land Degradataion Assessment in Drylands; 2008) and Anthromes (Ellis and Ramankutty 2008) for deriving land use shares per ecoregion, which combine remote sensing data with statistical data on human activities. Five broad land use types were distinguished (agriculture, pasture, managed forests, urban area, natural habitat) in our model. To get an estimate of parameter uncertainty, we calculated the area shares of each land use type per ecoregion in a GIS separately for each land use map. The maps were first transformed from WGS1984 projection to equal-area projections, using seven globally applicable equal-area projections. This resulted in n = 2 x 7 = 14 different estimates of land use shares for all global ecoregions.

2.2.4. *Uncertainty assessment*

Parameter uncertainty was propagated into characterization factors using Monte-Carlo simulation (1000 iterations). For each parameter, a distribution was directly derived from the data using non-parametric Kernel Density Estimation or, if only data ranges were available, a triangular distribution was assumed (see Annex Table 8.2.1). Except for plants, there was no uncertainty information available for species richness and endemism, thus these parameters were modeled without uncertainty. Median, upper and lower 95% confidence intervals were calculated for each characterization factor.

To assess the influence of each parameter on the uncertainty of characterization factors, their contribution to variance (CTV; Geisler et al. 2004) was calculated. In this method, the Spearman's rank-order correlation coefficient (ROCC) of each parameter with the characterization factor results is calculated for the set of Monte Carlo iterations. The CTV is calculated as

$$CTV_i = \frac{ROCC_i^2}{\sum_i^n ROCC_i^2} \tag{10}$$

where *i* is the calculated parameter, and *n* is the set of all parameters (see also Mutel et al. subm.).

2.2.5. Validation of species extinction

To test the validity of the model, we compared our prediction of global extinction of endemic species ($S_{lost,end}$), with observed numbers of extinct and threatened species (Koh and Ghazoul 2010). As it can take decades or centuries for a species with unviable populations to completely disappear (extinction dept; Brooks et al. 1999; Tilman et al. 1994), we considered the following species as "condemned to extinction": all species classified by the International Union for Conservation of Nature (IUCN) as "vulnerable," "endangered," "critically endangered," or "extinct". Data per ecoregion was extracted from the WWF wildfinder database (2006) for endemic mammals, birds, amphibians and reptiles. No validation could be performed for regional extinction of non-endemic species ($S_{lost,nonend}$), as only information on *global* extinction is available.

2.2.6. Sensitivity analysis

The above-presented model to derive land use characterization factors calculates the *average* impacts of past land use changes and is thus *retrospective*. Alternatively, *prospective* impacts can be calculated as *marginal* changes (Huijbregts et al. 2011; Weidema 2012): i.e. the impact of one additional m^2 of future land use change. To illustrate the sensitivity of the model to these model choices, we calculated average and marginal impacts for both retro- and prospective assessments for all forest ecoregions of the Amazon (n=19). This region was selected because no future land use scenarios for all global ecoregions were readily available, and the Amazon contains some relatively undisturbed ecoregions, which are expected to be converted for human use in the near future. Best- (good governance) and worst-case (business-as-usual) land use scenarios per ecoregion were derived from Soares-Filho et al. (2006) for the year 2050 and used to calculate prospective CFs. Equations for calculating marginal CFs and more detail on the applied method and scenarios are given in the Annex.

2.3. Results

2.3.1. Regional characterization factors

Regional CFs for occupation, transformation and permanent impacts were calculated per land use type for all WWF Ecoregions for mammals, birds, amphibians and reptiles, and occupation and transformation CFs for plants. Median values were used as default CFs, upper and lower 95% confidence intervals as measure for uncertainty (data in Annex).

For all three impact types and for all five taxa, the regions with high median CFs largely overlapped (see correlation analysis in Annex Table 8.2.5-8.2.6) and corresponded to regions that have been heavily converted in the past (see Figures 2.1 and 2.2 for a selection and Annex Figures 8.2.1-8.2.4 for all CFs). CFs were very low in regions with large shares of undisturbed habitat. Across all ecoregions, median CF values ranged over several orders of magnitude, and also showed large differences within biomes, indicating that a finer resolution than biomes is required for regionalized biodiversity assessments. For most ecoregions, the median CFs of different land use types were within the same order of magnitude (Figure 2.2 and

Annex). Thus, to determine land use CFs, the region where a land use is taking place seemed to be more important than the type of land use. For most ecoregions, agriculture had the highest median CFs, but the ranking of impacts of the other three land use types (pasture, urban areas, and forestry) was not consistent across taxa, impact type and ecoregion. For all but one ecoregion (PA0505), CFs differed significantly across land use type (at $p < 0.05$, evaluated with Kruskal-Wallis tests), for both occupation and transformation impacts. In general, the CFs were highest for plants, the most diverse taxon, followed in decreasing order by birds, mammals, reptiles and amphibians. Strong correlations of CFs were found across taxa (Annex Table 8.2.5).

Transformation CFs were half a regeneration time (median 5 - 199 years, Annex Table 8.2.3) larger than occupation CFs (eq 8). When endemic species were present, permanent CFs were mostly larger than transformation CFs, but results largely depend on the choice of the modeling period for permanent CFs (here, an estimated speciation time of 10^6 years was chosen). As permanent CFs represent global species losses, and transformation and occupation CFs represent regional species loss, they can not be directly compared.

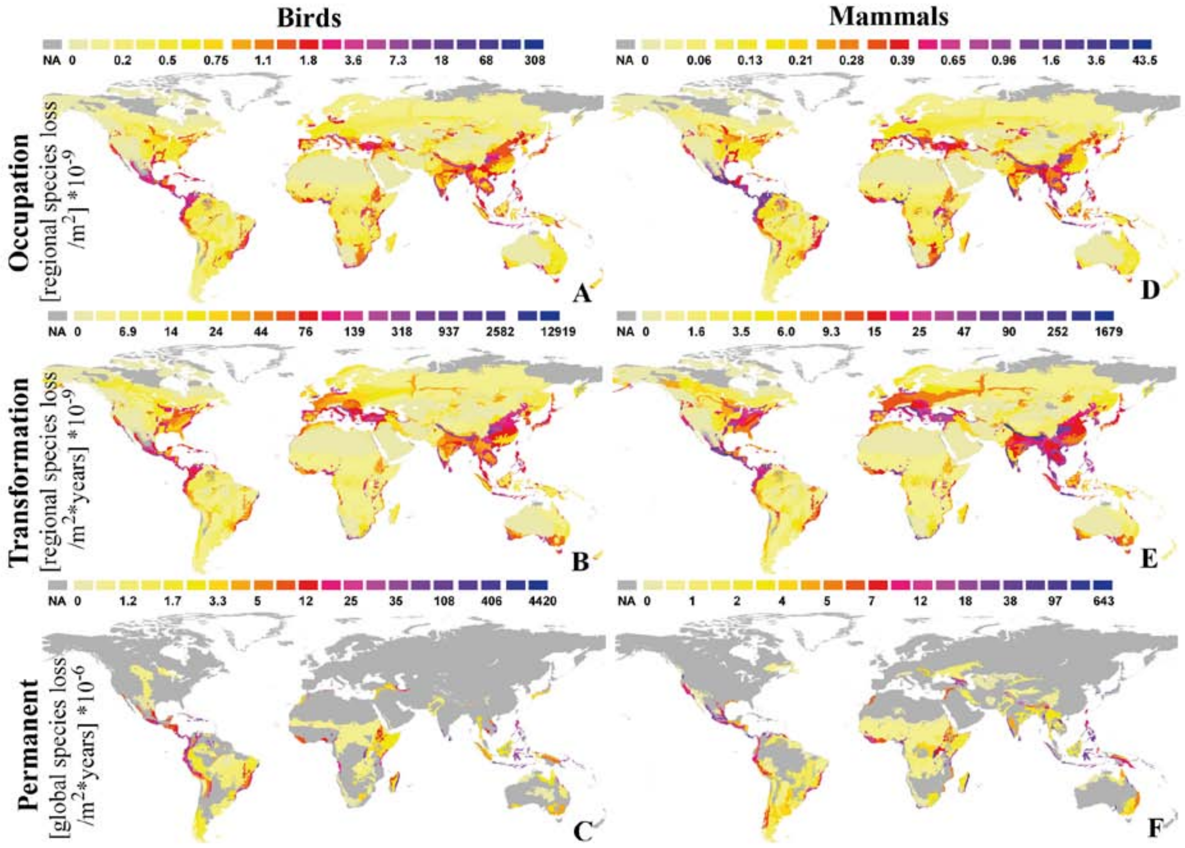

Figure 2.1. Median characterization factors of agricultural land, based on birds (left) and mammals (right), for occupation (top), transformation (middle) and permanent impacts (bottom). NA: No data available. Data for other land use types and taxa (amphibians and reptiles) is given in the Annex Figures 8.2.1-8.2.4.

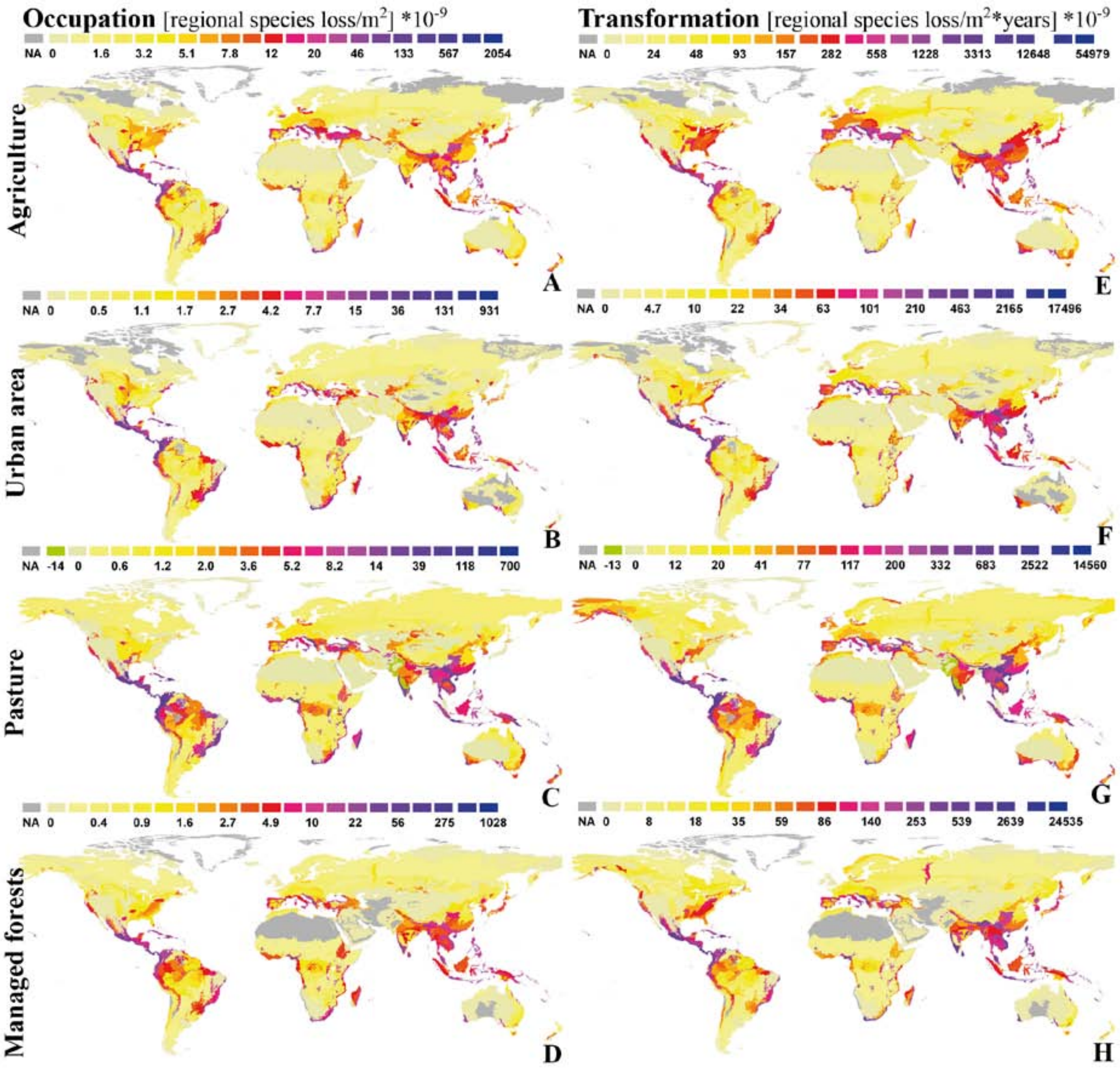

Figure 2.2. Median characterization factors based on plant species, for agriculture (top), urban (second), pasture (third) and managed forests (bottom), for occupation (left) and transformation impacts (right). Permanent CFs could not be calculated for plants. NA: No data available. Negative CFs = beneficial impacts on biodiversity.

2.3.2. Contribution to uncertainty

Uncertainty of the CFs was considerable. Most regional CFs ranged from positive to negative (i.e. beneficial) values (see Annex). The local characterization factors $CF_{loc,i}$ contributed most to the variance of regional CFs of occupation (67%-97%) and transformation (44% - 90%, Annex Table 8.2.7-8.2.8). This can be attributed to the value range of the local CFs, with span both positive and negative values (de Baan et al. 2013), i.e. both damaging and beneficial effects on biodiversity. Regeneration times were an important contribution to variance for the transformation CFs of agriculture (21%-31%), but a bit less for the other land use types (4%-16%, Annex Table 8.2.7). *z*-values (<0.2%) and the parameters for area (original area A_{org}, remaining natural habitat A_{new}, and area of each land use type A_i) did not contribute highly to the overall uncertainty of CFs (0.1%-4.4%).

2.3.3. Model evaluation

To evaluate our predictions of global extinction ($S_{lost,end}$), we compared them with observed numbers of threatened and extinct endemic species. The observed numbers were mostly within the uncertainty ranges of our predictions, but our median values were mostly smaller than the observed extinction (Annex Figure 8.2.8). This result seemed realistic, as observed species extinction is caused by multiple drivers, and should thus be higher than the modeled values for just a single driver.

2.3.4. Sensitivity analysis

Using future scenarios for 2050 (Soares-Filho et al. 2006), we calculated prospective characterization factors for 19 ecoregions in the Amazon. For the worst-case scenario, the median CFs increased only for ecoregions with large projected land use changes: for small increases in land use no changes in median CFs could be observed (Annex Figure 8.2.7). A maximum increase of median CFs of 65% was observed in ecoregions with a projected land use change from 60% remaining habitat to 20%.

Using a marginal impact calculation, the CFs did not change considerably at low levels of habitat conversion, but at high levels, impacts were even larger using a marginal approach, turning to infinity when the remaining natural habitat would become very small (Annex Figures 8.2.5 and 8.2.7).

2.4. Discussion

2.4.1. Modeling choices

The CFs of occupation and transformation of different taxa and land use types showed a relatively clear and consistent picture of regions with high potential land use impacts. High CFs occurred in highly vulnerable ecoregions where most natural habitat has been converted in the past, showing a strong overlap with Biodiversity Hotspots (Myers et al. 2000), classified by Brooks et al. (2006) as a reactive global conservation priority scheme. Ecoregions with low CFs largely overlapped with proactive conservation priority schemes such as of the Last of the Wild(Sanderson et al. 2002), which identifies large, undisturbed areas such as the Amazon or boreal forests with high conservation potential. The presented approach is clearly *retrospective*, only considering past accumulated land use changes. The matrix SAR model can in principle also be applied to future scenarios of land use change, and provide a *prospective* assessment, which we illustrated in the example of 19 forest ecoregions of the Amazon. In ecoregions with high projected future land use change, such a prospective assessment might be more appropriate to represent the potential future loss of species. However, global scenarios of future land use composition per ecoregion would be needed for a consistent comparison of land use impacts, adding another dimension of uncertainties to the CFs.

Based on the matrix SAR, both *average* and *marginal* impacts can be calculated. In the sensitivity analysis conducted for 19 exemplary ecoregions, the marginal CFs were similar to the average CFs at low levels of habitat conversion, but became much higher at high levels of habitat conversion. Thus, the marginal approach results in larger differences between ecoregions at low and high levels of converted habitat. We thus do not recommend using a marginal approach in combination with a retrospective assessment, as it ignores the conservation concerns in intact regions with high current or expected future rates of habitat conversion, such as the Amazon. For LCA studies where the main land use takes place in regions with high projected future land use change, a prospective assessment in combination with a marginal approach might provide more relevant results than the presented retrospective CFs.

2.4.2. Data availability and uncertainty

In this study we provided global CFs, which can be applied to a range of LCA studies. We thereby relied on available global land use and biodiversity data, such as the WWF database (Olson et al. 2001). As the latter does not distinguish between ecoregions with missing data or zero species richness or endemism, we treated both cases as missing data. This resulted in missing data for many ecoregions, particularly for permanent impacts, and a reduced applicability of this impact type in standard LCAs. Data on amphibians and reptiles was less complete than on mammals and birds. If better data on these species groups or new data on additional species groups such as arthropods become available, the presented CFs should be updated. As an interim suggestion, average CF values for neighboring ecoregions of the same biome could supplement regions that lack data.

For most input parameters (e.g. local CFs, *z*-values), the available data was not taxa- or ecoregion-specific, but rather aggregated across larger spatial units (e.g. biomes or globally). As expected, the resulting regional CFs were highly uncertain, often ranging from positive (detrimental) to negative (beneficial) values. The parameter dominating uncertainty of regional CFs were the local CFs. These were derived from a quantitative review of comparative biodiversity surveys of human-modified land (Alkemade et al. 2009; de Baan et al. 2013) and range between positive and negative values (the latter representing a higher species richness on the used land compared to a natural reference habitat; de Baan et al. 2013). Better data on taxa- and region-specific habitat suitability for different land management types is required to reduce the uncertainties for assessing the impacts of land use on biodiversity. In addition, the use of detailed and accurate global land use classification maps, such as currently developed by Fritz et al. (2012) or van Asselen and Verburg (2012), could further reduce uncertainties. Here, we could only distinguish between four very broad land use classes, each of them containing a broad range of land management practices. For a comparison of management practices (e.g. organic vs. conventional farming), the presented CFs would have to be further refined, as area requirements as well as biodiversity impacts of different management practices differ considerably (Müller et al. subm.).

The uncertainty of transformation CFs was also strongly influenced by the uncertainty of the biodiversity recovery times. This is the first global land use LCIA study that does not use recovery times based on expert estimates (van Dobben et al.

1998), but applies values derived from a meta-analysis of empirical data (Curran et al. subm.). The recovery of biodiversity following disturbance is a complex process, and the predicted recovery times are highly uncertain and represent best-case scenarios. The recovery time estimates implicitly assume that adequate amounts of old growth habitat exist in a region to act as a reservoir for species to recolonize secondary growth areas. In addition, the available empirical data covers a relatively short time horizon of up to 150 years. This is particularly relevant for young and currently dominating land use types (i.e. industrial agriculture, urbanization), for which predictions of ecosystem recovery may not apply, given the considerable physical and chemical impacts on biotic integrity (e.g. soil compaction, topsoil loss and erosion, chemical contamination). Additionally, in regions with predicted median recovery times of several hundred years, it may be doubtful to assume that transformation impacts will reverse at all, as a constant non-use of these areas over the entire recovery time is highly unlikely. Given these caveats, we believe that this study represents a first and important attempt to include some degree of reversibility into land use LCIA methods to be improved with future research. Here, we modeled permanent impacts based on endemism. Future LCIA studies modeling permanent impacts should attempt to include additional risk factors for global species extinction, such as global species rarity or vulnerability (Verones et al. subm.).

2.4.3. Model validity

Traditional species-area relationships (see eq 1) are commonly used models to assess species extinction due to habitat change, but their validity has been questioned in recent years as they tend to overestimate species extinction (He and Hubbell 2011). Koh and Ghazoul (2010) adapted the traditional SAR model to incorporate the moderating effect on extinction risk from habitat loss of land with high habitat value for species. They showed that the matrix SAR better predicted extinction of endemic birds than the traditional SAR model in 20 tropical forest biodiversity hotspots (Koh and Ghazoul 2010). However, the matrix SAR has not yet been tested outside of the tropics, for non-endemic species and other taxonomic groups than birds (Garcia-Ulloa et al. 2012; Koh and Ghazoul 2010) or mammals (Koh et al. 2010). In our analysis, the matrix SAR tended to underestimate global species extinction (Annex Figure 8.2.8). This might be partly due to the optimistic modeling assumption that endemic and non-endemic species have the same habitat suitability (i.e. local CFs). In reality, endemic species might be more sensitive to anthropogenic disturbances as they are often habitat specialists relying on natural habitat. In addition, the observed numbers of extinct and threatened endemic species are a result of several drivers contributing to global biodiversity loss (e.g. land use/change, invasive species, pollution, or global warming; Millennnium Ecosystem Assessment 2005). It is thus plausible, that predicted species losses due to land use are smaller than the observed numbers. The validity of regional extinction of non-endemic species could not be assessed, as no data on observed regional extinction are available. Therefore, it is important for the interpretation of the CFs that the results represent modeled *potential* regional or global species extinction and not predictions of true species extinction.

For non-forest habitats, such as grassland, savanna, tundra or deserts, it is unclear how well the species extinction estimates represent true extinction risk, as the ecological reactions to habitat change can be quite different than in forest ecosystems (Bond and Parr 2010). For these regions, the CFs should be interpreted with caution, especially for desert biomes. For the latter, the local CF of pasture showed a median negative value (i.e. higher median richness on pastures than on unused land for the available 5 data points), and as pasture was the dominant land use type in most desert ecoregions, the power term of the damage model (eq 5) became negative and was capped at zero (see Methods). Therefore, the median species loss was zero, resulting in regional CF values of zero for all land use types and taxa. Although local increases in species richness might occur due to, for example, irrigation (Wenninger and Inouye 2008), a beneficial regional effect is not guaranteed, as negative impacts due to reduced water availability might occur elsewhere in the watershed.

The presented approach is static, and land use dynamics and history are not considered. We compared the current land use composition with a potential “pre-human” situation, assuming that each ecoregion once consisted of a homogeneous ecosystem without any human influence (Olson et al. 2001). This might induce a bias for regions with a long land use history, (e.g. temperate regions of Europe have settlement histories spanning thousands of years), where the regional species pool might contain a broad range of species adapted to human-influence. Here, the main conservation concerns are more related to semi-natural habitat loss and fragmentation due to land intensification or abandonment of traditional agricultural practices (Krauss et al. 2010; Donald et al. 2001). However, even in Europe, the effects of accumulated land use change and fragmentation are still visible in current spatial diversity patterns (Krauss et al. 2010) indicating that the full effects of past change have yet to play out.

2.4.4. Applicability

To apply and test the presented CFs in case studies, spatially explicit land use inventory data are required, as announced by ecoinvent for the coming version v3.0 (Weidema et al. 2012), but global average CFs for land use in unknown locations are also provided in the Annex. As impacts differ strongly across regions, efforts to determine the region of production in life cycle inventories on ecoregion instead of country level are worthwhile. Otherwise, probabilities of potential production ecoregions can be used to calculate impacts (Mutel et al. 2012). For easier applicability of CFs, we also calculated aggregated CFs across all five taxonomic groups (see Annex), by first normalizing by the median species richness per ecoregion of each taxonomic group (giving equal weight to each taxonomic group). Alternatively, CFs could be simply summed up across taxonomic groups (giving equal weight to each species). Finally, we recommend to not aggregate reversible (occupation and transformation impacts) and irreversible (permanent) impacts into one land use indicator, as they assess regional and global species extinction, respectively.

2.4.5. Implications

In this paper, we present an approach how to derive globally applicable land use characterization factors from a species extinction model. Our approach allows a more complete assessment of land use impacts in LCA than previous methods: we provide global CFs for occupation, and, for the first time, transformation and permanent impacts including uncertainties for nearly all world regions for five taxonomic groups and four broad land use types. We regionalize the CFs to the ecologically relevant scales of ecoregions. With these three impact types, we provide decision-makers with information on the impacts of the actual land use, of land use changes, and of the risk of irreversible damages. The approach also aims to assess more relevant aspects than previous methods: We calculate impacts at the regional instead of the local scale, which is more relevant for assessing species extinction risk and assess absolute instead of relative species losses. The resulting unit for occupation and transformation impacts, potential regional species loss per m^2 times year, might be more intuitively understandable for LCA users than the currently prevalent unit potentially disappeared fraction of species (PDF). The unit PDF additionally has the disadvantage that the scale of impacts is not clearly defined, i.e. a value of 0.5 PDF could both refer to a locally disappeared fraction of 50% of all locally present species (e.g. on an arable field) and a 50% loss of all global species. This unclear definition of units results in a misleading aggregation of biodiversity impacts of different impact pathways (e.g. land use, climate change and eutrophication) modeled at different spatial scales (Curran et al. 2011). A reevaluation of meaningful endpoint units for biodiversity loss in LCA would thus be desirable.

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3. Land use impacts on biodiversity: proposal of characterization factors based on functional diversity[5]

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3.1. Introduction

The Millennium Ecosystem Assessment (MA) published in 2005 provided a detailed review on the assessment of the consequences of ecosystem changes in the world. On the local scale, within the interaction of biodiversity, ecosystem services and human well-being, land use is pointed as one of the main direct drivers of change, affecting living organisms. According to the Millennium Ecosystem Assessment, the rate of conversion of natural areas on Earth has increased dramatically in the last 50 years especially in tropical and subtropical areas, and many biomes have undergone up to 50% change. The direct consequence of such actions is, among others, the loss of biological diversity, observed by measures such as the reduction in population size, increase in homogeneity of species distribution and increase in species extinction (MA 2005).

The focus of land use impact modeling, based on biodiversity indicators, not only in Life Cycle Impact Assessment (LCIA), has been exclusively on taxonomic measures (Vandewalle et al. 2010), such as species richness (Achten et al. 2008, Koellner 2003, Mueller-Wenk 1998, Schmidt 2008, Weidema & Lindeijer 2001). Species richness and other classical ways of measuring biological diversity, have mainly treated all species in a relatively similar way, no matter the role they play in their habitat (Mouchet et al. 2010), i.e. species are assigned an equal weight, regardless of their functional characteristics. However, the loss of species also implies a change in the ecosystem related functions. Therefore, current studies emphasize functional diversity (FD) as a more appropriate indicator for ecosystem functioning, in comparison to taxonomic ones (Díaz & Cabido 2001, Flynn et al. 2009, Flynn et al. 2011, Mouchet et al. 2010, Petchey & Gaston 2006).

Functional diversity (FD) is a reflection of the range and value of the quantifiable aspects of species which are measurable at the level of the individual (Petchey et al. 2009), which if

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[5] This chapter has been published in a similar form in The International Journal of Life Cycle Assessment (Maia de Souza, Flynn, Declerk, Rosenbaum, de Melo Lisboa and Koellner, 2013)

chosen appropriately can act as indicators of the roles species play and the rules by which they assemble into communities (Vandewalle et al. 2010), and how they influence the way ecosystems operate (Tilman 2001). The basis for calculation of functional diversity is a set of functional traits, namely the morphological physiological, or behavioral characteristics of organisms (e.g. seed size, leaf area, wood density or plant size, for plants) which if chosen correctly can reveal their ability to respond to environmental pressures or to cause effects on ecosystem processes (Harrington et al. 2010). Functional traits are inherent measurable organism features or characteristics (e.g. seed size, leaf area, wood density or plant size, for plants) which reveal their ability to respond to environmental pressures or to cause effects on ecosystem processes (Harrington et al. 2010)

According to Hooper et al. (2002), ecosystem processes are rather affected by functional diversity among organisms than their taxonomic identity, such as species richness. Species richness change does not mean a direct impact on ecosystem processes: a loss of species can be, for example, compensated for by other species exerting similar functions. It is necessary to evaluate other aspects, such as redundancy or complementarity of species functions, and this cannot be captured by species counting – such as currently done in LCIA - in which species are been treated as equal, without taking into account their role in the ecosystem complexity. McLaren (2006) adds that ecosystem processes are an effect of both (i) change of functional effects of organisms in an ecosystem, due to loss of species; and (ii) functional response of other organisms, to that change and simple taxonomic measures, such as species richness, is not able to explain the impacts of these changes.

For example, measures of functional diversity have shown how species assemblages change across natural, semi-natural and human-dominated habitats (Mayfield et al. 2006). Any change on species composition has consequences on the quantity, range and combination of functional traits and therefore has a direct influence on the regulation of processes in an ecosystem (Chapin III et al. 2000) depending on the relationship between traits and ecosystem services (de Bello et al. 2010).

There are basically two ways to quantify FD: (i) discontinuous measures, i.e., group/guild-based, such as the classification of species traits according to functional group richness; and (ii) continuous measures, i.e., non-group measures, in which species are not divided among functional groups. (Petchey & Gaston 2006, Petchey et al. 2009). Continuous measures all start with calculations of the multivariate distance between each pair of species in an assemblage. Discontinuous group-based measures are based on a priori assignment of species to functional groups by expert decision and ad-hoc and subjective process (Petchey et al. 2009). Such group-based measures might not be very appropriate for all ecosystem processes and a large amount of decisions and assumptions are required, such as where to place the boundaries of each group and the number of groups to include. Most importantly, substantial functional differences among species are likely to be disregarded in some cases when forming these functional groups. The second approach consists in calculating the distances between species based on their trait values, and then summarizing these distances by hierarchical clustering into a dendrogram (Petchey & Gaston 2006), or taking the product of each distance pair and the abundance of the species Rao`s quadratic entropy (Botta-Zukat 2005). Univariate measures based on the abundance-weighted range of each trait have also been developed (Mason et al. 2003, Mason et al. 2005). Here we apply the widely used dendrogram-based approach proposed by Petchey & Gaston (2002).

In order to assess the potential contribution of functional diversity measures, this paper proposes the use of functional diversity indicators for the calculation of characterization factors in land use impact modeling in LCIA. This study therefore assesses differences between the characterization factors based on functional diversity and species richness for different taxonomic groups and across land use types. The answers will be helpful also for the critical analysis of existing LCIA land use models which usually use taxonomic measures for assessing biodiversity loss in different land use types.

3.2. Methods

The model proposed is based on data compiled by Flynn et al. (2009), who analyzed data on species richness and functional diversity across land use intensification gradients. Three taxonomic groups have been included in this study: (i) mammals, (ii) birds and (iii) plants, mainly due to data availability.

The spatial resolution unit for modeling the biogeographical differences are ecoregions, as defined by Koellner et al. (in review) and Olson et al. (2001). However, it is important to emphasize that for this initial analysis, data on a relatively small scale (plot scale) has been used as representative for ecoregions. The calculation of the FD index for each study area has been done according to Petchey & Gaston (2002). No abundance data has been included on the calculation.

Under the framework for Life Cycle Impact Assessment of land use two types of land interventions can be distinguished, land use (occupation) and land use change (transformation). However, in this model, only characterization factors for occupation impacts are calculated.

3.2.1. Data sources

The meta analysis done for mammals comprised eight studies carried out in Peru, Costa Rica, Mexico, United States of America and Canada. Bird data (Table 8.3.1, Annex) were gathered in Brazil, Costa Rica, Mexico and United States of America. Plant data (Table 8.3.2, Annex) ranged from Coto Brus, Costa Rica to eastern Canada. Fig. 3.1 presents the spatial location of the data among different ecoregions. Further, Table 8.3.1, (Annex), presents more detailed information on the ecoregions and land use types where the locations of data collection as well as the world ecoregions where they are located and the types of land use included in each of the studies.

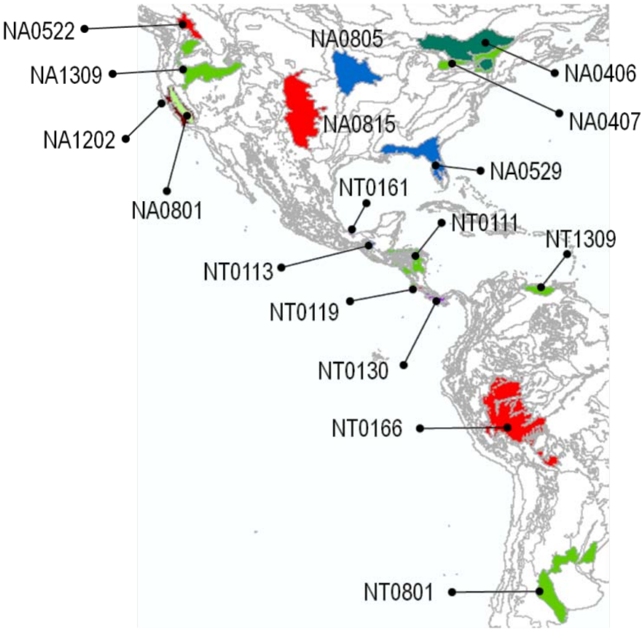


Figure 3.1: Ecoregions for which data on species richness have been sampled. The map shows North and Central America and the northern part of South America

3.2.2. Land use classes

The different land use types present in the studies analyzed have been aligned with the land use types classification of the UNEP/SETAC LULCIA proposal (Table 3.1), following Koellner et al. (submitted). The reference situation chosen was the most natural or close-to-natural state present in each of the studies included in the meta analysis, i.e. primary or secondary forest. The studies on mammals, plants and birds included 13, 13 and 15 different land use types, respectively. For details, see the supplementary Tables 8.3.1, 8.3.2 and 8.3.3 (Annex).

Table 3.1 Land use and cover classification for LCA (Koellner et al. in review)

Code	Land use type classification	Code	Land use type classification
[1]	Forest	**[5.1]**	Agriculture, arable
[1.1]	Forest, natural	**[5.1.1]**	Agriculture, arable, fallow
[1.1.1]	Forest, primary	**[5.1.2]**	Agriculture, arable, non-irrigated
[1.1.2]	Forest, secondary	**[5.1.2.1]**	Agriculture, arable, non-irrigated, extensive
[1.2.1]	Forest, extensive	**[5.1.2.2]**	Agriculture, arable, non-irrigated, intensive
[2.2]	Wetlands, inland	**[5.1.6]**	Field margins/Hedgerow
[3]	Shrubland	**[5.1.8]**	Agriculture, arable, intensive
[4]	Grassland	**[5.2.1]**	Agriculture, permanent crop, non-irrigated
[4.1.1]	Grassland, not used	**[6]**	Agriculture, mosaic
[4.1.2]	Grassland, for livestock/grazing	**[7.6.3]**	Traffic area, railroad, embankment
[4.2]	Pasture/meadow	**[10.1.2]**	Rivers, artificial
		[10.2.2]	Lakes, artificial

3.2.3. Functional trait data

The functional traits included in the analysis of mammals include mass, feeding guild, food type, activity, nesting and litter size (Flynn et al. 2009). For birds, they were mass, feeding guild, food type, foraging location and foraging habitat. Plants traits taken into consideration were leaf area, height, fruit type, fruit length, foliage, growth form and the characteristic of being leguminous or not leguminous (Flynn et al. 2009). Table 3.2 displays a summary of the taxonomic groups and respective traits, considered on the calculation of the FD index.

Table 3.2 Functional traits used for functional diversity calculation (Flynn et al. 2009)

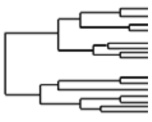
Taxonomic Group	Trait	Categories
Birds	Mass	-
	Feeding guild	Carnivore, herbivore, insectivore, omnivore
	Food type	Invertebrates, small fruits, seeds, nectar, fish, generalist
	Foraging location	Ground, upper canopy, shrub layer, mid canopy, forage throughout, aquatic
	Foraging habitat	Ground, leaves, perch-and-attack, stems, aerial, water, hover, soar-and-attack, other
Mammals	Mass	-
	Feeding guild	Carnivore, herbivore, insectivore, omnivore
	Food type	Invertebrates, fruit, seeds, vertebrates, vegetation
	Activity	Diurnal, nocturnal, either
	Nesting	Aquatic, arboreal, burrows, multiple, terrestrial
Plants	Leaf area	-
	Height	-
	Fruit type	Fleshy, not fleshy
	Fruit length	-
	Foliage	Deciduous, evergreen
	Growth form	Tree, shrub, tall herb, low herb, grass
	Leguminous	Legume, not legume

As observed on Table 3.2, the main traits chosen for this analysis were (i) *resource use and behavioral traits* for mammals and birds; and (ii) *morphological and anatomical traits* for plants, related to the species capacity to capture resources. These traits are important on the analysis of a possible loss of species and the consequent change in ecosystem functions (Flynn et al. 2009).

3.2.4. Functional Diversity calculation

Petchey & Gaston's FD index was used in this study (Petchey & Gaston 2002, Petchey et al. 2004, Petchey et al. 2009). Due to the lack of species abundance data in some of the studies, abundance-weighted measures of functional diversity could not be calculated here. Four steps are involved in the calculation of FD: (i) construction of a matrix containing species' traits values (Fig 3.2, a); (ii) calculation of the multivariate distances between species, using these trait values (Fig 3.2, b); (iii) hierarchical clustering of the distance matrix into a dendrogram, which is a tree diagram, used to depict the arrangement of clusters (Fig 3.2, c); (iv) calculation of FD values based on the total branch length of the dendrogram, for the species present in a particular community.

$$T = \begin{pmatrix} v_{1,1} & v_{1,2} & \cdots & v_{1,t} \\ v_{2,1} & v_{2,2} & \cdots & v_{2,t} \\ \cdots & \cdots & \cdots & \cdots \\ v_{s,1} & v_{s,2} & \cdots & v_{s,t} \end{pmatrix} \quad D = \begin{pmatrix} 0 & d_{1,2} & \cdots & d_{1,s} \\ d_{2,1} & 0 & \cdots & d_{2,s} \\ \cdots & \cdots & \cdots & \cdots \\ d_{s',1} & d_{s',2} & \cdots & 0 \end{pmatrix}$$



(a) (b) (c)

Figure 3.2: Elements involved on the calculation of Petchey & Gaston's FD Index: (a) trait matrix, with trait values *v* corresponding to species *s*, and trait *t*; (b) distance matrix, with the pair-wise distances between species; and (c) dendrogram, from which the FD values are calculated.

The distance matrix is used to estimate the similarity among organisms, in terms of the differences in the trait values. Gower's distance was the metric used to calculate the distance between species based on their traits, as this distance metric allows the use of multiple data types, including categorical and binary data (Podani & Schmera 2006).

After obtaining FD values, in order to evaluate FD and SR differences within studies, these values were normalized to a value of 1 for the reference land use type, in each study as in Eq. (1) and (2).

$$FD_N = FD_i / FD_{ref} \qquad (1)$$

$$SR_N = SR_i / SR_{ref} \qquad (2)$$

where FD_N and SR_N are the normalized values for FD and SR, respectively. FD_i corresponds to the calculated functional diversity value for a certain land use type ***i*** and FD_{ref} is the functional diversity value for the reference land use type. Sr_i and SR_{ref} are the corresponding values for species richness.

The reference chosen for the normalization procedure was the natural or most close-to-natural land use type (Table 8.3.1 for mammals; Table 8.3.2, for plants; and Table 8.3.3, for birds, see Annex). However, when more than one natural or close-to-natural land use types were available within one study, a sensitivity analysis was done in order to verify if a change in the results would be observed.

3.2.5. Calculation of Characterization Factors

In the calculation of characterization factors for different land use types, the natural reference values for FD and SR were set at one, by means of normalization of FD and SR values. Characterization factors have been calculated for FD (CF_{FD}) and SR (CF_{SR}) as follows in Eq. (3) and (4), respectively:

$$CF_{FD} = 1 - FD_N \qquad (3)$$

$$CF_{SR} = 1 – SR_N \qquad (4)$$

for which FD_N and SR_N are the normalized values for FD and SR, respectively (Table 8.3.1 for mammals; Table 8.3.2, for plants; and Table 8.3.3, for birds, see Annex). It is possible to verify that for the natural reference, in which the characterization factor equals to zero, the damage is also zero.

An illustration on how the characterization factor would be applied in LCIA is represented in Eq. (5) and (6):

$$SI = CF_{FD} \times A_{occ} \times t_{occ} \qquad (5)$$

$$SI = CF_{SR} \times A_{occ} \times t_{occ} \qquad (6)$$

where SI corresponds to the resulting impact score, A_{occ} represents the area of occupation and t_{occ} the time of occupation.

3.2.6. Statistical analysis

Statistical analysis have been performed in order to analyze if significant differences exist (i) between the characterization factors (CFFD x CFSR) for each taxonomic group; and (ii) among the land use types included in all studies. Pairwise t-tests were used to compare characterization factors for species richness (CFSR) and for functional diversity (CFFD) for the three taxonomic groups, within each study. Non-parametric Wilcoxon Matched Pairs test was also applied, due to the small sample sizes, in order to assess whether deviation from normality affected the results. It was not possible to run a comparison between the CFFD and CFSR for different land use types within one specific taxonomic group CFFD for mammals x CFSR for mammals) due to the statistical population size.

Furthermore, aiming to verify for significant differences among the characterization factors for each land use type, an analysis of variance (ANOVA) was performed for all data, followed by a post-hoc test, in order to check which groups of data were particularly different from each other. Fisher's Least Significant Difference (LSD) test was chosen for post hoc comparisons among groups. Statistica 9.0 (StatSoft Inc. 2009) was used to perform the statistical analyses.

3.3. Results

3.3.1. Land use impacts on biodiversity

In the calculation of characterization factors for different land use types, the natural reference for each study was chosen taking into account the natural or closest to natural type. The references used and the calculated values can be seen from Table 8.3.1 to Table 8.3.3. These Tables present the raw values for SR and calculated FD, the normalized values and the calculated characterization factors based on species richness data (CF_{SR}) and functional diversity (CF_{FD}).

3.3.2. Analysis of characterization factors (CFSR and CFFD) for each taxonomic group

Paired t-tests were applied in order to assess differences between CF_{SR} and CF_{FD} within each study for all three taxonomic groups, when not considering the different land use types. Significant differences have been found for birds and plants (Table 3.3), with larger impacts detected for characterization factors based on FD. However, the same was not found for mammals. Due to the small sample size, it was not possible to perform a test to check for significant differences between the characterization factors for a certain land use type.

Table 3.3 Results for the t-tests between characterization factors for functional diversity CF_{FD} and species richness CF_{SR} for dependent samples (marked differences are significant at $p < 0.05$).

	CF_{FD}		CF_{SR}		CF_{FD} versus CF_{SR}					
	Mean	**Std. Dv**	**Mean**	**Std. Dv**	**Dif**	**Std. Dv. Diff**	**t**	**df**	**p**	**N**
Birds	0.3763	0.2702	0.4193	0.2773	-0.0429	0.0799	-3.6431	45	**0.0007**	46
Mammals	0.2287	0.2529	0.2369	0.2849	-0.0082	0.1006	-0.4869	35	0.6293	36
Plants	0.0141	0.2378	-0.1553	0.4528	0.1694	0.3810	2.9466	40	**0.00694**	41

Similar results could be observed when the non-parametric Wilcoxon's matched pairs test is applied: mammals were the only taxonomic group to not present significant differences between the characterization factors for FD and SR.

3.3.3. Analysis of characterization factors for different land use types

Analysis of significance using parametric tests

In order to evaluate the differences existing among characterization factors for different land use types, ANOVA was performed, taking into consideration all data. After, in order to compare the means of different land use types, a Fisher LSD *post-hoc* test was used.

The ANOVA Univariate test of significance (Table 3.4) showed that there were significant differences among the CF_{FD} and CF_{SR} values for different land use types. The significance value of 5% (0.05) has been used for the test. Once the test statistic is larger than the critical values obtained in F tables for both CF_{FD} and CF_{SR}, it can be concluded that there is a significant difference among the population means, at p levels of 0.0026 and 0.0001, respectively.

Table 3.4 Univariate tests of significance for CF_{SR} and CF_{FD}.

		Sum of Squares (SS)	**Degree of freedom**	**MS**	**F**	**p**
	Intercept	1.6365	1	1.6365	23.206	0.000005
CF_{FD}	**Land use types**	3.5913	22	0.1632	2.315	0.002607
	Error	7.0523	100	0.0705		
	Intercept	0.7025	1	0.7025	5.324	0.023088
CF_{SR}	**Land use types**	8.6639	22	0.3938	2.984	0.000110
	Error	13.1941	100	0.1319		

On the results of the *post-hoc* Fisher LSD test, significant differences for CF_{FD} among the different land use types have been found for the cases shown on Table 3.5, mainly among natural and close-to-natural land use types, such as "forest" and "forest, primary" and more managed ones, such as "agriculture, mosaic", "agriculture, arable, non-irrigated, intensive" and "agriculture, permanent crop, non-irrigated, intensive". No significant differences have been observed among forest and agriculture types (e.g. between "forest primary" and "forest secondary" or between "agriculture arable, non-irrigated, intensive" and "agriculture, permanent crop, non-irrigated"). "Forest primary" has shown to be a closer-to-natural environment, while, on the other side, higher characterization factors have been calculated for "agriculture, arable, non-irrigated, intensive". The results also show some differences between the characterization factors for functional diversity (CF_{FD}) and species richness (CF_{SR}).

Table 3.5 Groups of land use types for which significant differences have been found for CF_{FD} and CF_{SR} values when Fisher LSD test was applied. . Land uses in **bold** are those which differ between CF_{FD} and CF_{SR}.

	Land use types which presented significant differences towards natural and close-to-natural land use types	
Natural and close-to-natural	**Results for CF_{FD}**	**Results for CF_{SR}**
Forest, Forest, primary Forest, secondary	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Grassland Pasture/Meadow Wetlands, inland	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Grassland Pasture/Meadow Wetlands, inland **Traffic area, railroad, embankment**
Forest extensive	Agriculture, arable, non-irrigated, intensive Grassland	Agriculture, arable, non-irrigated, intensive Grassland **Agriculture, mosaic** **Agriculture, permanent crop, non-irrigated** **Pasture/Meadow** **Wetlands, inland**
Field margins/Hedgerows	Agriculture, arable, non-irrigated, intensive **Agriculture, permanent crop, non-irrigated** **Grassland**	Agriculture, arable, non-irrigated, intensive **Lakes, artificial** **Rivers, artificial**
Shrubland	Agriculture, arable, non-irrigated, intensive **Grassland**	Agriculture, arable, non-irrigated, intensive **Lakes, artificial**
Rivers, artificial	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Grassland Pasture/Meadow Wetlands, inland	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Grassland Pasture/Meadow Wetlands, inland **Field margins/ Hedgerows** **Grassland for livestock/grazing** **Traffic area, railroad, embankment**

Daily et al. (2001), Daily et al. (2003), Hilty & Merenlender (2004) and Sullivan & Sullivan (2006) observed that when primary forest was taken as a reference land use type, other land uses such as “forest, secondary”, “field margins/hedgerows” and all the other agriculture land use types had positive characterization factors. “Shrubland” has also presented one of the lowest characterization factors. As expected, managed landscapes should present higher biodiversity loss and a higher impact in ecosystem processes.

In the studies in which “shrubland” and “forest, primary” were present, the smallest characterization factors have been found for shrubland: Sullivan & Sullivan (2006). However, this is a unique study in which both land use types occurred together and a more extended analysis should be done in order to evaluate the real magnitude of these characterization factors.

On the other side, higher characterization factors have been calculated for “pasture/meadow” and “agriculture, permanent crop, non-irrigated” and “agriculture, arable, non-irrigated, intensive”, in Estrada & Coates-Estrada (1997a, 2005a), Horvath et al. (2001a), Jobin et al.(1996), Smukler et al. (2010). Regarding “pasture/meadow”, the same behavior could not be observed in Sanchez Merlos et al. (2005a, 2005b), mainly due to the presence of arboreal cover in the “pasture/meadow” land use type. However, “pasture/meadow” with low arboreal cover presented higher characterization factors (more impact), for both FD and SR, in comparison to “pasture/meadow” with high arboreal cover. In a similar manner, the study from Hilty and Merenlender (2004) showed that “permanent crop” areas near natural areas have smaller characterization factors than those isolated, meaning those isolated are more impacted than those near natural areas.

For all bird studies and the majority of mammals' studies, the characterization factors for SR are bigger than the respective characterization factors for FD. However, the same was not observed for plants, for which the CF_{SR} were usually smaller than the CF_{FD}. Therefore, no conclusion could be drawn for the differences in magnitude for both characterization factors, associated with the various land use types.

Analysis of significance using non-parametric tests

Similar results to those obtained for parametric ANOVA have been obtained when running non-parametric tests for the whole set of characterization factors, aiming to check for significant differences among land use types. A clear difference can be seen between managed landscapes and natural or close-to-natural land covers. When the sample size was N=1, the Mann-Whitney U test could not be performed and the comparison could not be done. For plants, the comparison pointed "pasture/meadow" and "lakes, artificial" important changes. At last, for the taxonomic group birds, the differences were mainly "forest, secondary", in relation to "pasture/meadow" and "agriculture, arable, non-irrigated, intensive". However, when the taxonomic groups and studies are "set aside" and a pairwise comparison is done with all data available, the result is what can be seen in Table 3.6.

Table 3.6 Results for non-parametric pairwise comparison test for ALL GROUPS with $p<0.01$ (**) and $0.01<p<0.05$ (*).

		Forest	**Forest, primary**	**Forest, secondary**	**Wetlands, inland**	**Shrubland**	**Grassland**	**Pasture/ Meadow**	**Agriculture, arable, fallow**	**Agriculture, arable, non-irrigated, intensive**	**Field margins/ Hedgerow**	**Agriculture, permanent crop, non-irrigated**	**Agriculture, mosaic**	**Rivers, artificial**	**Lakes, artificial**
Forest	CF_{FD}						*	*		*		**	*		
	CF_{SR}						*			*		**	*		
Forest, primary	CF_{FD}				*		**	**		**		**	**		
	CF_{SR}				*		**	**	*	**	*	**	**		*
Forest, secondary	CF_{FD}				*		*	*		**		**	**		
	CF_{SR}				*		*	*		**		**	**		
Wetlands, inland	CF_{FD}		*	*											
	CF_{SR}		*	*											
Shrubland	CF_{FD}						*			*		*	*		
	CF_{SR}						*			*		*	*		
Grassland	CF_{FD}	*	**	*		*					*				
	CF_{SR}	*	**	*		*									
Pasture/ Meadow	CF_{FD}	*	**	*										*	
	CF_{SR}		**	*										*	*
Agriculture, arable, fallow	CF_{FD}														
	CF_{SR}		*												
Agriculture, arable, non-irrigated, intensive	CF_{FD}		**	**		*									
	CF_{SR}	*	**	**		*								*	*
Field margins/ Hedgerow	CF_{FD}						*								
	CF_{SR}		*												*
Agriculture, permanent crop, non-irrigated	CF_{FD}	**	**	**		*								*	
	CF_{SR}	**	**	**		*								*	*
Agriculture, mosaic	CF_{FD}	*	**	**		*								*	
	CF_{SR}	*	**	**		*								*	*
Rivers, artificial	CF_{FD}							*				*	*		
	CF_{SR}							*		*		*	*		
Lakes, artificial	CF_{FD}														
	CF_{SR}		*					*		*	*	*	*		

The analysis was done here taking into account two confidence levels: $p < 0.05$ (*) and $p < 0.01$ (**). It can be observed that such as for mammals, the land uses “agriculture, permanent crop, non-irrigated”, “agriculture, arable, non-irrigated, intensive” and “agriculture, mosaic” have shown more significant results (**), in comparison to “forest”, “forest, primary” and “forest, secondary”. For those, the null hypothesis (H_o) of no difference between the land use types was rejected at $p < 0.01$. Values in the range $0.01 < p < 0.05$ have been considered as less significant (*). For all the other values in the Table, no differences have been found.

Box-Whisker Plot of all characterization factors for different land use types

Box and whisker plots for the all characterization factors, per land use types, for both species richness and functional diversity values are shown Fig. 3.3.

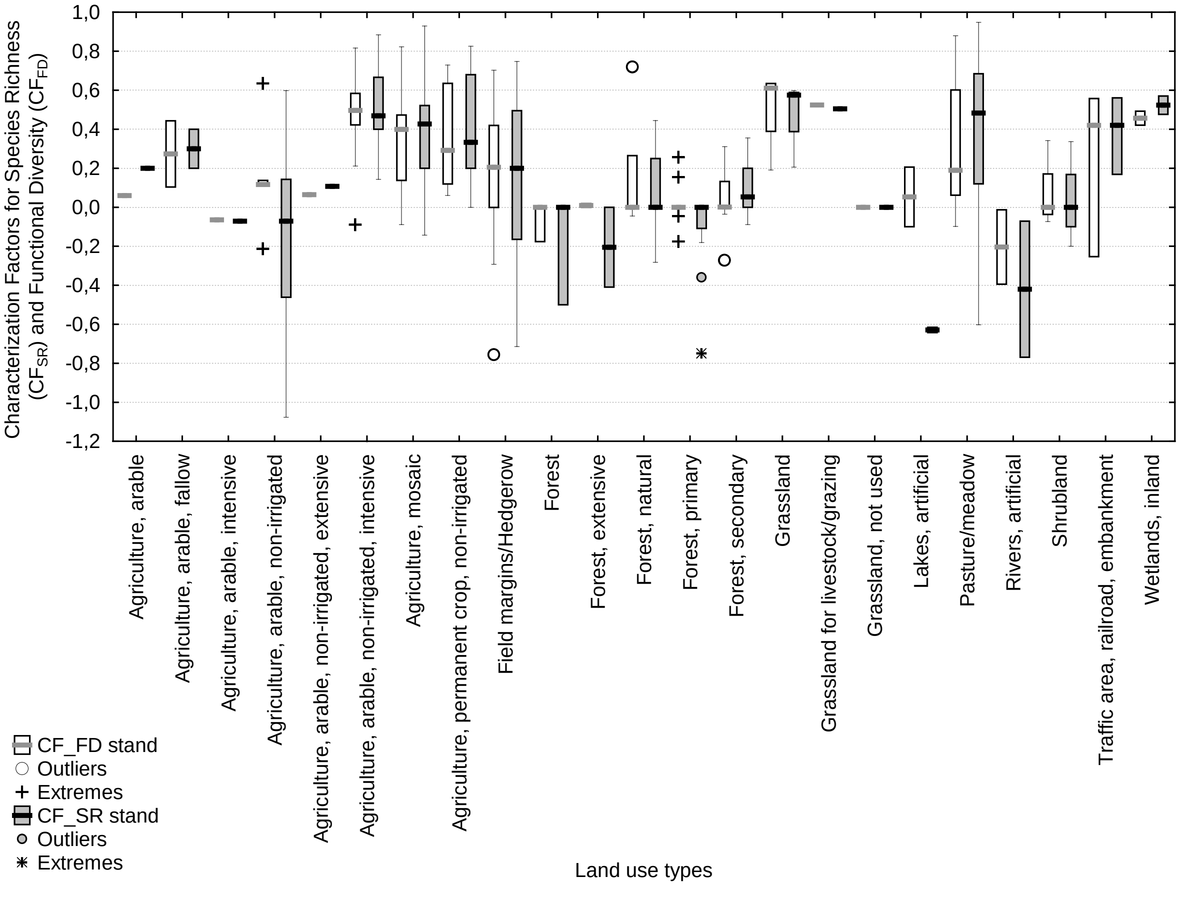


Figure 3.3: Box-Whisker Plot of CF_{SR} and CF_{FD} grouped by land use types for all taxonomic groups (mammals, birds and plants).

As observed in Table 3.6 significant differences among land use types have been observed between natural and close-to-natural covers, such as “forest”, “forest, primary” and “forest, secondary” and managed land for agriculture purposes, such as “agriculture mosaic”, “agriculture, permanent crop, non-irrigated” and “agriculture, arable, non-irrigated, intensive”. This trend has been mainly observed for mammals, although the contribution for the overall combined data has been quite balanced: mammals (29%), plants (33%) and birds (38%). A within-taxonomic group analysis of characterization factors for different land use types could not be performed, due to small sample sizes.

3.3.4. Sensitivity analysis for the choice of reference land use types

Aiming to evaluate the sensitivity of the data towards the choice of the reference land use type, a sensitivity analysis has been carried out, by changing the reference land use type for those studies which contained more than one natural or close-to-natural LU types. Table 3.7 shows the new reference land use types applied in the sensitivity analysis.

Table 3.7 Other reference land use types which have been used in order to run the sensitivity analysis.

Taxonomic group	Study	Previous Reference LU type	Reference LU type applied in the new analysis
Mammals	Husband (2009)	Forest (data 2005)	Forest (data 2006)
Plants	Margie (Mayfield et al. 2006)	Forest, primary (forested riverbank)	Forest, primary (tree-fall gap in forested area) Forest, primary (understory in forested area)
	(Sanchez Merlos et al. 2005b)	Forest, natural	Forest, secondary
	(Middleton & Merriam 1983)	Forest, extensive	Forest, primary
	(Sanchez Merlos et al. 2005a)	Forest, natural	Forest, secondary
Birds	(Best et al. 1995)	Floodplain forest	Grassland (Upland forest)

Results for the analysis of sensitivity for characterization factors (CFSR x CFFD) for each taxonomic group

There was no significant change in the reference land use type for the studies shown on Table 3.7. The results for the t-tests applied for the sensitivity analysis are displayed on Table 3.8. It can be seen that no significant differences between the means of the characterization factors for FD and SR have been found for mammals. On the other side, they have been again found different for birds and plants.

Table 3.8 Results for the sensitivity t-tests between characterization factors for functional diversity CF_{FD} and species richness CF_{SR} for dependent samples (marked differences are significant at p <0.05).

	CF_{FD}		CF_{SR}		CF_{FD} versus CF_{SR}					
	Mean	**Std. Dv**	**Mean**	**Std. Dv**	**Dif**	**Std. Dv. Diff**	**t**	**Df**	**p**	**N**
Birds	0.3233	0.2678	0.3621	0.2826	0.0388	0.0837	3.1479	45	**0.0029**	46
Mammals	0.2428	0.2447	0.2711	0.2635	0.0283	0.0916	1.8534	35	0.0723	36
Plants	0.0036	0.2581	0.1067	0.3752	0.1104	0.3032	2.3310	40	**0.0249**	41

However, it can be seen that the ***p*** values have varied from the original ones (see Table 3.7), especially for mammals.

Results for the analysis of sensitivity for characterization factors for different land use types

In a similar way as shown on Table 3.4, the results of the ANOVA pointed significant differences among CF_{FD} and CF_{SR} values for different land use types. But when Fisher LSD Test was applied, some differences between the CF_{FD} and CF_{SR} before and after the sensitivity analysis have though been observed. They are shown on Table 3.9. However, for "forest, primary", "forest, secondary", "agriculture, arable, non-irrigated, intensive", "agriculture, permanent crop, non-irrigated", "grassland" and "rivers, artificial", the results seem to remain similar, even with the changes in the reference state.

Table 3.9 Sensitivity Analysis: groups of land use types for which significant differences have been found for CF_{FD} and CF_{SR} values when Fisher LSD test was applied. Land uses in **bold** are those which differ from original results (see Table 3.5)

	Land use types which presented significant differences towards natural and close-to-natural land use types	
Natural and close-to-natural	**Results for CF_{FD}**	**Results for CF_{SR}**
Forest, Forest, primary Forest, secondary	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Grassland Pasture/Meadow	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Grassland Pasture/Meadow **Lakes, artificial**
Forest extensive	Agriculture, arable, non-irrigated, intensive Agriculture, mosaic Agriculture, permanent crop, non-irrigated Grassland	**Lakes, artificial**
Field margins/Hedgerows	Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated	Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Lakes, artificial Rivers, artificial **Agriculture, mosaic** **Pasture/Meadow**
Shrubland	Agriculture, arable, non-irrigated, intensive Grassland	Agriculture, arable, non-irrigated, intensive Lakes, artificial **Agriculture, mosaic** **Agriculture, permanent crop, non-irrigated**
Rivers, artificial	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Grassland Pasture/Meadow Wetlands, inland Grassland for livestock/Grazing	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated Field margins/ Hedgerows Grassland Grassland for livestock/grazing Pasture/Meadow Traffic area, railroad, embankment Wetlands, inland

3.4. Discussion

The results suggest that differences in species richness and functional diversity between broad land use categories, such as forest and agriculture are pronounced and therefore characterization factors should be differentiated. However, within the same category, such as "forest, primary" and "forest, secondary", significant differences were not found. Statistical tests, such as ANOVA, have been applied in order to analyze if significant differences among characterization factors for distinct land use types could be obtained. Aiming to identify where the changes particularly occurred, *post-hoc* test Fisher LSD has been used. On the evaluation of the differences between characterization factors for FD and SR for each taxonomic group, t-tests were applied. Significant results were obtained for plants and birds.

3.4.1. Characterization factors for different land use types

Results showed significant differences among different land use types for all the taxonomic groups. Highly significant differences (mainly $p < 0.01$) were found between three agriculture land use types (“agriculture, arable, non-irrigated, intensive”, “agriculture, permanent crop, non-irrigated”, “agriculture, mosaic”) and three forest land use types (“forest”, “forest, primary” and “forest, secondary”). A similar situation has also been found between open field areas - such as “pasture/meadow” and “grassland” – and forest land use types (“forest”, “forest, primary” and “forest, secondary”). However, no significant differences were found within agriculture land use types (i.e. “agriculture mosaic and “agriculture, arable, non-irrigated, intensive), neither among the mentioned forest land use types (i.e. “forest” and “forest, primary”), nor among the “open field areas”.

According to the results obtained, Table 3.10 shows eight different land use type categories for which significant differences have been observed. It also presents the land use sub-categories, for which no differences have been found and for which distinct characterization factors do not apply. For other land use types mentioned in Table 3.1, but not present in Table 3.10, no conclusions could be drawn, mainly due to small data sets.

Table 3.10 Land use types whose calculated characterization factors could be aggregated, according to the results obtained*.

Land use type category	Land use type subcategories
1. Agriculture, intensive	Agriculture, mosaic Agriculture, arable, non-irrigated, intensive Agriculture, permanent crop, non-irrigated
2. Agricultural (other)	Agriculture, arable
3. Forest	Forest Forest, natural Forest, primary Forest, secondary
4. Grassland and Pasture/Meadow	Grassland Pasture/Meadow
5. Lakes	Lakes, artificial
6. Rivers	Rivers, artificial
7. Shrubland	Shrubland
8. Wetlands	Wetlands, inland

** Land use types for which no conclusions could be drawn are (i) “agriculture, arable”, (ii) “agriculture, arable, fallow”, (iii) “agriculture, arable, non-irrigated”, (iv) “agriculture, arable, non-irrigated, extensive”, (v) “forest extensive”, (vi) “forest, natural”, (vii) “grassland, not used”, (viii) “grassland for livestock/grazing” and (ix) “traffic area, railroad, embankment”.*

However, it is important to emphasize that the results have been obtained through the analysis of all data, taking into account different taxa. Resulting characterization factors (see Tables 8.3.1, 8.3.2 and 8.3.3, Annex) have therefore not been aggregated in this study. Future work may address on aggregated characterization factors.

The resulting characterization factors’ values were as expected for both FD and SR indicators: they were higher for managed land use types, such as “agriculture, mosaic” and lower or negative for close-to-natural or natural land use type. However, both indicators did not behave equally: this is due to the non-linear differences on the impact caused by biodiversity loss and the corresponding effect on ecosystem processes, especially for birds and plants. It is thus recommended that FD indicators, as a closer link to ecosystem processes, be used as land use impact indicators, in order to get a better understanding and linkage between biodiversity loss and resulting impacts in ecosystem functioning.

Classification of land use types

The first consideration in interpreting these results is the classification of land use types to specific categories in each study (see Table 3.1). This involves certain subjectivity on the associations made and relies on the description of land use types by the authors of the studies. In some of them, only few details were given on the specific land covers. The category “forest”, for example, is used, when not much information is obtained for classifications such as “forest, primary” and requires only that “an area should be covered more than 15% by trees”. Differences between “forest, primary” – “forests minimally disturbed by human impact, where flora and fauna species abundance are near pristine” (Koellner et al. 2012) - and “forest, secondary” – “areas originally covered with forest or woodlands, where vegetation has been removed, forest is re-growing and is no longer in use” (Koellner et al. 2012) – have not always been clear in the studies. For other forest types “forest, natural” and “forest, extensive”, few data points have been found within all data. Therefore, the results have not shown to be representative in the analysis. Again, the differentiation among them and other forest types relied on information gathered in the studies and was shown to be small.

Sample sizes

The second consideration for these results is the relatively small sample sizes for the number of communities in each land use type. Ecosystem processes result from the interactions among species and the higher the sample size, the better it is to define the difference between the simple presence or absence of a species versus the functional change in ecosystem characteristics (Chapin III et al. 2000). Small sample sizes might not be adequate to verify the variation, especially to what concerns SR x FD measures. The analysis of land use types has been done across different taxa and ecoregions and might have influenced on the results. A separate analysis could, however, not be performed, due to lack of data within different land use types and ecoregions.

Functional traits

One single set of functional traits was used for the calculation of the functional diversity values. Those traits have been considered by Flynn et al. (2009) as correlated to critical ecosystem functions, which would help to predict the differences in results between loss of functional diversity and species richness. However, in order to better evaluate the magnitude of change due to these functional characteristics, an analysis with different sets would be possible in the future. One example of another trait for mammals is the “offspring size for each species” - characteristic that could be drastically changed in an event of expressive species drop. For plants, the growth rate would also be an important aspect to be considered. Functional differences – redundancy or complementarity – are evaluated according to a certain set of species functional traits and variation is a reflection of diversity of those specific species characteristics (Hooper et al. 2002, Petchey & Gaston 2002). Therefore, redundancy of different ecosystem functions might not have been well represented, due to relatively small sample sizes and/or to the traits chosen. However, there is still low availability of such data, for different species, in different land use types, within the same study or different studies in the same area.

3.4.2. Functional Diversity vs. Species Richness

Significant differences between characterization factors for FD and SR have been observed for two taxonomic groups: plants and birds. However, in order to further analyze the influence of different land use types in the results, it would be necessary to have larger data availability and in this study a conclusion on which land use might have played a role on the resulting differences could not be drawn.

Plants are the taxonomic group most commonly used as proxy indicator in biodiversity modeling, mainly due to higher data availability (Hooper et al. 2002, Schmidt 2008) and relatively straight-forward sampling procedure, when compared to sampling for animal communities. However, as verified in the results, there are variations among the results obtained for distinct taxonomic groups and it's advisable to have data from other taxa to evaluate the impact on different groups.

3.4.3. Choice of reference state

The sensitivity analysis has shown that no significant changes in the results have been obtained for the confidence level of $p>0.05$. Although, it could be observed that a change in the reference state caused the resulting p levels for birds and plants to increase and for mammals to decrease. However, for each study, from which data has been used, the main close-to-natural or natural land use type was mainly classified within the "forest category" and, as shown in the results, no representative distinction has been made among its subcategories. An interesting sensitivity analysis could result from studies containing multiple choices for reference land use types, classified in distinct land use categories, such as shrubland, forest or grasslands.

3.4.4. Regional differences

Unfortunately, due to the small samples sizes, it was not possible to run a statistical analysis to verify the significance of differences among the characterization factors for different land use types in each of the ecoregions included in the studies. All data collected has been used on the analysis of land use types, no matter where they were located. This certainly adds uncertainty to the calculation of characterization factors. In principle, the range of data was collected in different spatial and temporal scales, with diverse sampling techniques. However, this could be reverted by a larger sample size. Moreover, in the studies from which data has been gathered there is a large variation in species richness in three different taxonomic groups within different land use types located in many ecoregions. The data collection for such a model is highly data demanding and more effort would therefore lead to more representative characterization factors in function of land use type, ecoregion, and taxonomic group.

3.5. Conclusions

Currently, biodiversity indicators used are mainly based on taxonomic measures, such as species richness. Further, they do not take into account important factors such as species endemism, abundance or species vulnerability status, such as identified in the IUCN Red List (IUCN 2001) categories classification (Maia de Souza 2010). These might be crucial for the definition of species loss in a local, regional, or global scale and also on the capacity of a certain species to recover in a certain environment. Although, functional diversity does not necessarily include data on these factors, it represents an improvement towards a better association between species loss and ecosystem processes.

Based on the results obtained and the previous discussion, there are significant differences between the land use impact assessment approaches using species richness and functional diversity indicators, hence the use of FD characterization factors is justified as it establishes a more accurate link between biodiversity loss and ecosystem processes. Clearer results would likely be obtained with a larger sample size of communities within each land use type. The provision of characterization factors for more detailed land use types in LCIA, such as forest types and agriculture types can still be a difficult issue, and big efforts need to be done on data collection. The level of detail in the definition of different land use categories for LCIA does not correspond to biodiversity data availability, especially in current models. More data collection is needed, in order to enable the definition of characterization factors for a longer list of different land use types.

For both indicators, FD and SR, the differences between land use types have been significant, especially for agricultural and forest land use types, although differences could not be deeply detailed. Further, the subjectivity on the classification of land use types, according to certain categories, following Koellner et al. (submitted) also adds variability to the results. The choice of a reference land use type has also proven to be another source of variability.

In comparison with current methodologies, this paper has demonstrated that for both approaches, there was a difference among the results obtained for each taxonomic group. For now, in LCIA, plants are usually taken as proxy, without analyzing the impact of such choice. As a further improvement, we suggest to analyze data within a certain area, in order to check for sources of variability among various taxonomic groups, although this is very data demanding.

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4. Climate change related impacts from wood extraction for bio-energy on a global scale

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4.1. Introduction

Biofuels are a potentially important source of energy for our society in an era of increasing demand for energy, progressive depletion of fossil fuels, and increasing societal risks from climate change. Wood is the main and oldest form of renewable energy (EIA 2011). Global wood usage for both heat and electric energy production is increasing (FAO 2010), and advances have been made towards increased production of cellulose-based biofuels, such as bioethanol from wood biomass (Nieminen *et al.* 2012). This increased focus on renewable energy sources has encouraged a strong debate on how to derive a scientifically valid greenhouse gas (GHG) life cycle balance related to the use of bio-energy (Schlamadinger *et al.* 1997; Lal 2004a; Davidson & Janssens 2006; Cherubini 2010; Cherubini *et al.* 2011; Cherubini & Strømman 2011). Several GHG sources and sinks need to be considered for bio-energy production, including changes in the sequestered stock of carbon due to biomass extraction, contribution of land use and land use change related emissions as well as emissions from use of fossil fuels during planting, maintenance and harvesting of the biomass.

Common practice in life cycle assessment (LCA) of bioenergy has been to assume that any carbon dioxide (CO_2) emission related to biomass combustion equals the amount of CO_2 absorbed in biomass, thus assuming a carbon neutral system with no climate change impacts (Cherubini *et al.* 2011). Therefore, most LCA studies on biofuels do not include impacts of wood extractions nor emissions of biogenic CO_2 (van der Voet *et al.* 2010; Cherubini & Strømman 2011). Lal (2004b) and Ostle and Ward (2012), however, showed that biogenic CO_2 emissions are a significant flow in the carbon cycle e.g. CO_2 emissions from changes in soil carbon due to biogenic crop cultivation. Next to the fact that CO_2 release from biogenic sources may not be taken up at the original site of release, CO_2 spends time in the atmosphere before being captured by biomass regrowth, which can possibly lead to climate change related impacts.

Some recent studies have proposed ways to improve bioenergy GHG accounting. Müller-Wenk and Brandão (2010) proposed a method to estimate the CO_2 emissions due to transformation of land from one use to another. They focused on the potential carbon stock of a specific land use type compared to the other and estimated the net release to air and the average duration of carbon stay in the atmosphere before taken up elsewhere (e.g. other land and oceans). Cherubini *et al.* (2011) determined biogenic global warming potentials (GWP) based on the modelled atmospheric decay of biogenic CO_2. Their method is world generic and includes clear cutting of forest followed by regrowth that captures the same amount of CO_2 that was initially released. Changes in wood extraction rates due to an increase in wood biomass production processes require a change in rotation time, which was not included in the study from Cherubini *et al*. (2011). Kilpeläinen *et al.* (2011) developed an LCA tool to calculate the net carbon exchange of forest bioenergy production taking into account uptake of carbon into biomass, the decomposition of litter and humus, emissions from forest management operations and carbon released from the combustion of biomass and degradation of wood-based products. The tool has been applied to Finnish case studies (Kilpeläinen *et al.* 2012; Routa *et al.* 2012) for which many data are available. Subsequently, Kilpeläinen *et al.* (2012) determined the radiative forcing associated with net CO_2 emissions in Finnish boreal conditions. To the best of our knowledge, there is currently no study that quantifies the potential climate change related impacts on humans or the environment, resulting from biogenic CO_2 emissions from spatially-explicit forest wood extraction on a global scale.

The goal of this paper is to develop a global, spatially-explicit method to quantify the impacts on human health and ecosystem quality per amount of wood extracted for life cycle impact assessment (LCIA). For this, effects on the carbon balance due to increases in forest wood extraction for bio-energy through changes in forest rotation time are determined. We link this change in forest carbon sequestration to a change in carbon release to the atmosphere, which in turn alters atmospheric temperature through changes in CO_2's concentration and radiative forcing capacity. The resultant change in global mean temperature leads to impacts on human health and ecosystem quality (De Schryver *et al.* 2009). The significance of including biogenic emissions due to wood extraction in life cycle assessment is tested for the case of heat production from wood extracted from specific countries, compared to heat production from anthracite coal.

4.2. Methods

4.2.1. Framework

In LCIA, characterization factors are determined that quantify the impact of emissions to impacts to selected areas of protection such as human health and ecosystem quality. We followed the cause-effect pathway outlined in Figure 1 to determine the effects of changes in wood resource extraction on carbon stock changes and subsequent impacts on human health and terrestrial ecosystems.

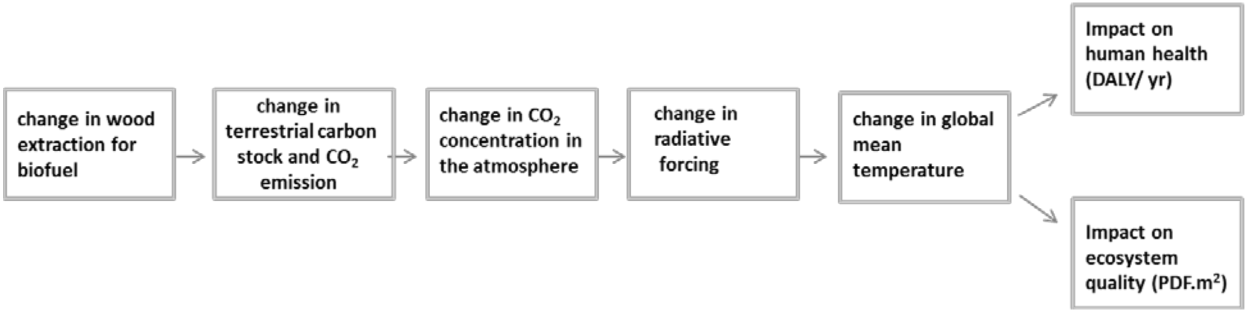

Fig 1 Cause-effect pathway used in determining the effect of increased wood resource extraction on human health and terrestrial ecosystem quality (adapted from De Schryver *et al.* (2009))

When there is an increase in wood extraction for biofuel use, the amount of above ground carbon stock in the forest changes, followed by a change in the quantity of atmospheric carbon. Changes in the related CO_2 concentration will cause changes in the radiative forcing capacity, which results in global mean temperature changes. Changes in global mean temperature cause impacts on human health, which can be expressed as Disability Adjusted Life Years (DALY), and on terrestrial ecosystem quality in terms of potentially disappeared fraction of species (PDF). This pathway translates into a characterisation model that quantifies the impact per unit of extraction in terms of a characterisation factor (CF).

$$CF_e = \frac{-\Delta C_{forest}}{\Delta R} \cdot \frac{\Delta T}{-\Delta C_{forest}} \cdot \frac{\Delta I_e}{\Delta T} \tag{1}$$

where ΔR is the change in wood resource extraction ($m^3 \cdot yr^{-1}$), $-\Delta C_{forest}$ is the reduction in the stock of carbon in the forest (kg C), ΔT is the change in global mean temperature (^oC) and ΔI_e is the change in impact at endpoint *e*, i.e. humans or ecosystems. CFs were determined for a change in PDF of terrestrial species, including birds, butterflies, mammals and plants, caused by a change in wood extraction in managed forests ($PDF \cdot yr \cdot m^2 \cdot m^{-3}$ of wood extracted) and the change in impacts on human health in DALYs related to a number of climate sensitive diseases and health risks (malaria, malnutrition, drowning, diarrhoea, and cardio-vascular diseases) ($DALY \cdot m^{-3}$ of wood extracted).

We followed the cause effect pathway according to three model options, based on cultural perspective theory (Hofstetter 1998). Cultural perspective theory can be used to distinguish value choices in modelling by presenting various perspectives. Following De Schryver et al. (2009), we included three cultural perspectives namely the Individualistic, the Hierarchist, and the Egalitarian perspective. The Individualist coincides with the view that mankind has a high adaptive capacity through technological and economic development; therefore the present (short-term) is more important than the future. Another characteristic is that only well- understood impacts are taken into account in the decision making process. The egalitarian coincides with the view that nature is strictly accountable. Egalitarians consider present and future effects equally important. This perspective represents the most precautionary approach, both for time frame, as well as including impacts. For the hierarchical perspective, scientific consensus is taken as a starting point for including/excluding information in the assessment. This coincides with the view that impacts can be avoided with proper management, and uses the most common policy principles regarding most issues, including the time-frame (Thompson *et al.* 1990; Hofstetter 1998).

4.2.1. Forest carbon stock

To determine the change in carbon stock due to a change in wood resource extraction ($-\Delta C_{forest} / \Delta R$), the Global forestry model (G4M) was applied. G4M simulates the processes of growth and management of forests on a global scale (Gusti *et al.* 2008; Kindermann *et al.* 2013). It calculates above ground forest biomass on a 0.5° x 0.5° resolution in a yearly time step. Model inputs include global land cover, which outlines the forest area and site productivity. Net primary productivity (NPP) data from Cramer *et al.* (1999) were used to estimate the site productivity of the forest. The current biomass quantity was determined based on the forest biomass, which is reported by FAO statistics as described by Kindermann *et al.* (2008). The forests modelled by G4M consist of generic trees, i.e. individual species are not distinguished. For the forest carbon stock we considered the change in above ground biomass carbon only, as was done by Cherubini *et al.* (2011). G4M simulations were performed for 20 year, 100 year, and infinite (steady state) time horizons which coincide with the Individualist, Hierarchist, and Egalitarian perspective, respectively.

In G4M, extraction rates depend on availability of extractable wood, which is largely determined by the forest rotation time. The rotation time is in turn dependent on the site productivity of the forest. Fast growing stands typically have a shorter rotation length, while slow growing sites have a longer one (Kindermann *et al.* 2006). For this study, we simulated a change in wood extraction in every grid by employing a 5% change in rotation time of current managed forests. This resulted in either an increase or decrease in wood extraction per grid. Since from an LCA perspective, we are interested in an increase in wood extraction from the forest, we proceeded with the grids in which we observed an increase in wood extraction, which is either cause by a decrease in rotation time, or an increase. The change in forest carbon stock due to this increase in wood extraction was quantified as kilograms of carbon per m^3 of wood harvested, aggregated over a yearly time step ($kg\ C \cdot yr \cdot m^{-3}$). These calculations were performed and analysed using ArcGIS 9.2 and presented at 0.5°x0.5° grid resolution. Since most process inventory data (e.g. wood extracted per source location) are available on country level, country-specific changes in carbon stock due to a change in wood extraction were determined as well. For this, grid-specific factors were aggregated based on the G4M simulated amount of wood resource available for extraction.

4.2.2. Environmental impacts

We assumed that the change in above ground forest biomass (stocking biomass) equals to a change in atmospheric carbon, thus directly linking it to global mean temperature change, ($\Delta T/-\Delta C_{forest}$), via three subsequent steps. First, the carbon released to the atmosphere ($-\Delta C_{forest}$) was converted into ppb CO_2 via the conversion factor $5.17 \cdot 10^{-10}$ ppb $CO_2 \cdot kgC^{-1}$ (De Schryver *et al.* 2009). Second, a change in concentration results in a change in the radiative forcing ($W.m^{-2}$) by a factor of $1.4 \cdot 10^{-5}$ $W \cdot m^{-2} \cdot ppb^{-1}$ (Forster *et al.* 2007). Third, the change in radiative forcing causes a change in the global mean temperature of respectively 0.34, 0.48 or 0.67 $^{o}C \cdot W^{-1} \cdot m^{2}$ for a 20, 100, and infinite time horizon as calculated by De Schryver *et al.* (2009) using the IMAGE model (Eickhout 2004).

For the change in impacts on human health and terrestrial ecosystems caused by a change in global mean temperature ($\Delta I_e/\Delta T$), we directly used the so-called damage factors from De Schryver *et al.* (2009). For human health impacts, value choices regarding a future scenario, certainty about climate-related health impacts, age weighting, and discount rate in the Disability Adjusted Life Year (DALY) calculations are of importance. For the Individualist perspective, only a limited number of (proven certain) impacts were included. An optimistic future scenario with age weighting and a discount rate of 3% was applied, resulting in a $\Delta I_e/\Delta T$ of $1.35 \cdot 10^{6}$ $yr \cdot yr^{-1} \cdot {}^{o}C^{-1}$. The Egalitarian perspective follows a worst case scenario that coincides with a pessimistic future scenario, no age weighting, and no discount rate in the DALY calculations. All impacts, irrespective of their certainty of occurrence, were included, resulting in a $\Delta I_e/\Delta T$ of $2.95 \cdot 10^{7}$ $yr \cdot yr^{-1} \cdot {}^{o}C^{-1}$. For the Hierarchical perspective a baseline future scenario with no age weighting and discount rate at 3% was applied in the DALY-calculations. An intermediate number of diseases was included, most dominantly malnutrition, resulting in a $\Delta I_e/\Delta T$ of $6.12 \cdot 10^{6}$ $yr \cdot yr^{-1} \cdot {}^{o}C^{-1}$. For ecosystem impacts, the species' resilience to climate change according to dispersal ability and level of protection based on the IUCN Red list classification (IUCN 2001) are of importance. For the Egalitarian perspective only red list species were considered and no dispersal assumed, resulting in a $\Delta I_e/\Delta T$ of 0.2 PDF $^{o}C^{-1}$.

Inclusion of impacts to all species and a dispersal of species resulted in a $\Delta I_e/\Delta T$ of 0.06 PDF·$^{o}C^{-1}$ for both the Individualist and Hierarchist perspectives. Cultural theory was therefore applied in all stages of the CF calculation and the value choices per perspective used in this paper, are listed in the supporting information Tables S1and S2.

4.2.3. Normalization

Total damage due to bio-energy production was assessed by multiplying the country-specific CFs with the total wood fuel production quantity (m^3) per country in 2011 from the FAO (http://faostat.fao.org/site/626/default.aspx#ancor).

4.2.4. Case study

We performed a case study on heat production to (i) test the significance of including the biogenic carbon emissions due to wood extraction in life cycle assessment, and (ii) compare the impacts of using wood biomass for heat production to the impact of using fossil fuel, in this case coal anthracite. We calculated human health and ecosystem quality impacts for a functional unit (FU) of "1 MJ heat production". The systems compared were a 6kW wood heater with 75% efficiency and a 5-15kW anthracite coal stove with 70% efficiency. The energy content of the wood was estimated to be ~9500 MJ·m^{-3} based on values from Werner *et al.* (2007). The impact of biogenic carbon emissions due to wood sourcing were determined for wood sourced from the most wood fuel producing country in each continent and from the European Union. These countries are the United States of America (USA), Russia, Germany, India, Ethiopia, Australia and Brazil and were identified from the FAO database's latest (i.e. year 2011) figures for total wood fuel production (http://faostat.fao.org/site/626/default.aspx#ancor).

Characterisation factors for greenhouse gas emissions from fossil energy sources were taken from De Schryver *et al.* (2009). Other impacts included in the case study were, for human health impacts (DALY·kg^{-1}), fine particulate matter formation, photochemical oxidant formation, human toxicity, ozone depletion and ionising radiation. Impacts to terrestrial ecosystems (PDF·m^2·yr·kg^{-1} or PDF·m^2·yr·m^{-2}·yr^{-1}) included were acidification, land use (change), and ecotoxicity. These endpoint CFs were taken from the ReCiPe methodology (Goedkoop *et al.* 2009).

4.3. Results

4.3.1. Characterisation factors

Fig 2 and Fig 3 show grid-specific characterisation factors for impacts on human health (DALY ·m^{-3}) and ecosystem quality (PDF·m^2·yr·m^{-3}) due to increased wood extraction, respectively. Fig S1 and Fig S2 in the supporting information show the country average CFs. Grids in grey are areas where wood extraction is at its maximum and therefore an increase is assumed not realistic. These areas vary per perspective due to the differences in time horizon per perspective. Grids without CF values, for example in northern Africa, do not contain forest cover.

Both positive and negative CF values were obtained. For the Individualist perspective, only a few grids, e.g. in Ethiopia, showed negative CFs. In the rest of the grids there was a decrease in the forest stock when the rotation time was changed to allow increased extraction, therefore positive CFs were observed. In the Hierarchist and Egalitarian perspectives, positive CFs were observed in the tropical climate regions (e.g. Central Africa), and in parts of Argentina, Canada, United States, and Eastern Europe. Negative CF values were mostly found in Southern Canada, Eastern Africa, Australia, parts of Russia and parts of Europe, such as Germany and Austria.

The magnitude of CFs varied depending on the cultural perspective. In the Individualist perspective, the 95% confidence range was between $+2.40\cdot10^{-5}$ and $+6.70\cdot10^{-5}$ DALY/m^3 wood, and $+1.15\cdot10^{2}$ and $+3.20\cdot10^{2}$ PDF·m^2·yr/m^3 wood. For the Hierarchist, the 95% confidence range was between $-2.40\cdot10^{-2}$ and $+3.10\cdot10^{-2}$ DALY/m^3 wood for impacts on human health and $-2.50\cdot10^{4}$ and $+3.30\cdot10^{4}$ PDF·m^2·yr/m^3 wood for impacts on ecosystem quality. The largest spread in CFs was observed in the Egalitarian perspective with a 95% confidence range between $-6.10\cdot10^{-2}$ and $+4.90\cdot10^{-2}$ DALY/m^3 wood for impacts on human health and $-4.50\cdot10^{4}$ and $+3.60\cdot10^{4}$ PDF·m^2·yr/m^3 for ecosystem quality.

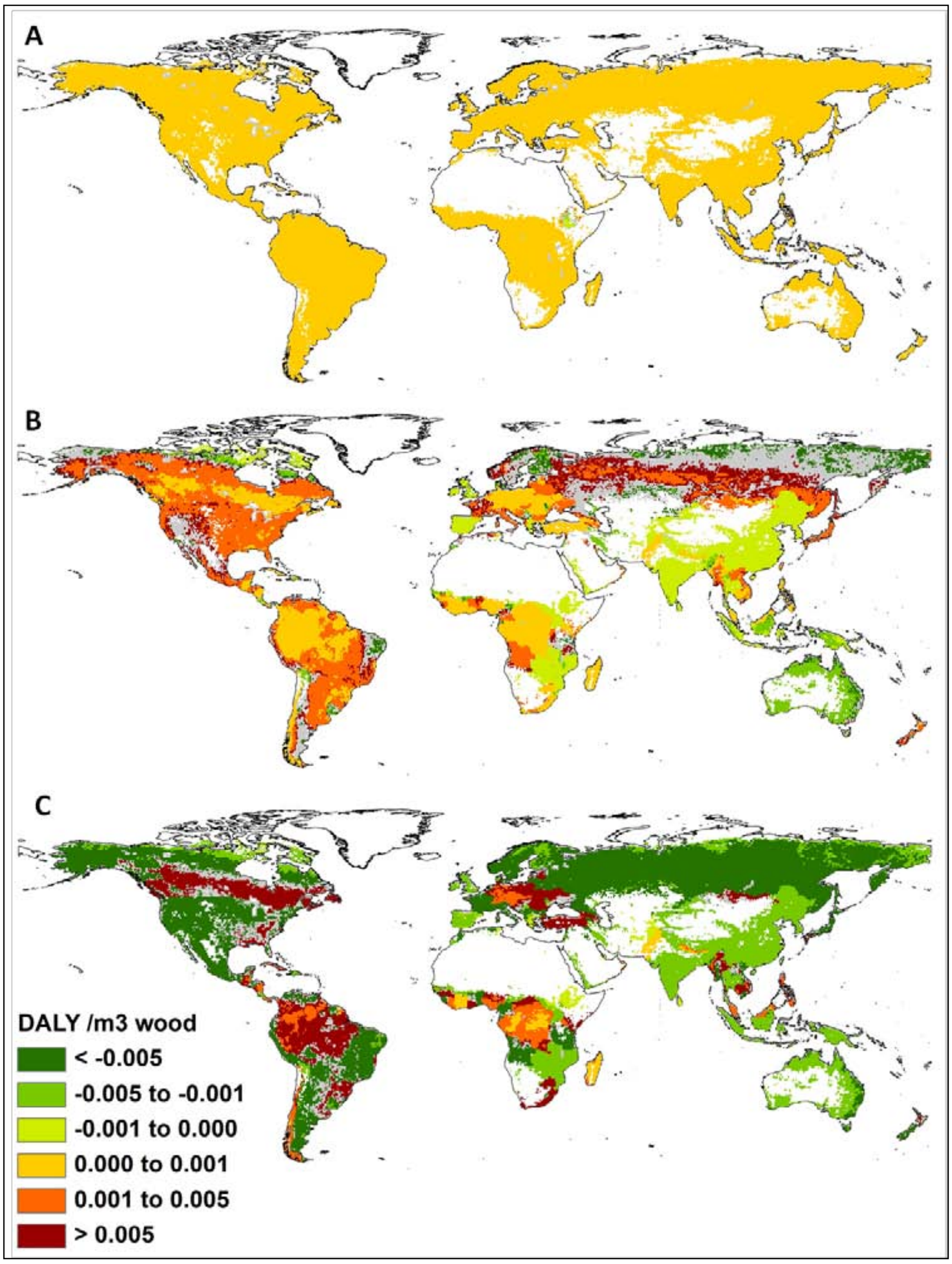

Fig 2 Characterisation factors for impacts on human health due to forest wood extraction for bio-energy production (DALYm $^{-3}$ of wood) at grid level for (A) Individualist, (B) Hierarchist and (C) Egalitarian perspective. Grids in grey are areas where wood extraction is already close to its optimum with the current rotation time therefore changes in rotation time only result in decreased harvest. Grids with no data, shown in white, represent areas with no forest cover

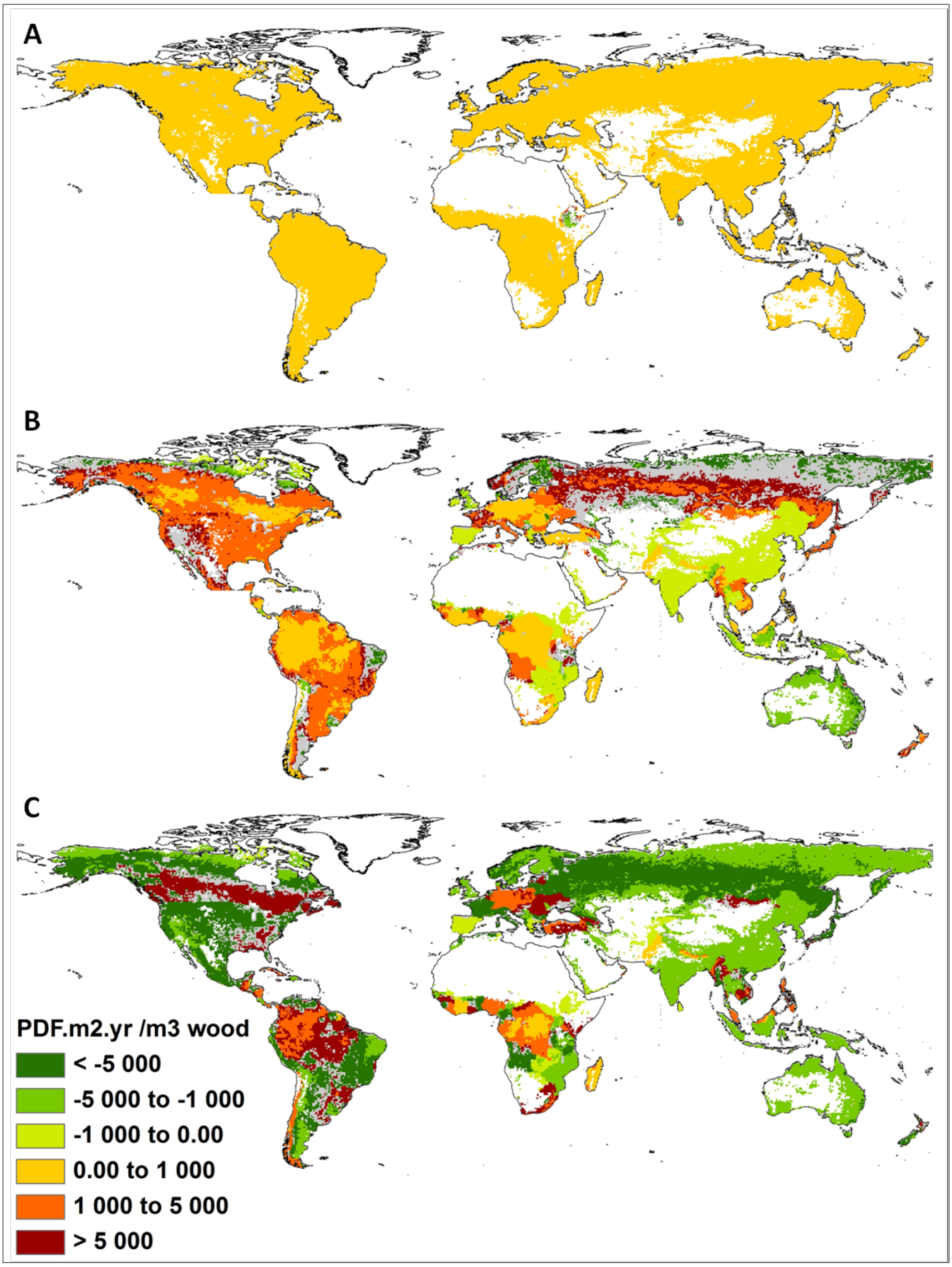

Fig 3 Characterisation factors for impacts on ecosystem quality due to forest wood extraction for bio-energy production PDF·m^2·yr·m^{-3} of wood) at grid level for (A) Individualist, (B) Hierarchist and (C) Egalitarian perspective. Grids in grey are areas where wood extraction is already close to its optimum with the current rotation time therefore changes in rotation time only result in decreased harvest. Grids with no data, shown in white, represent areas with no forest cover

4.3.2. Normalization factors

Normalization shows the same results for human health impacts as well as ecosystem impacts. In India the largest impacts (individualist perspective: ·1509 $^{-4}$ PDF·m^2/ 1.3·10 $^{-4}$ DALY/yr) or benefits (hierarchist: -2.2·10^{-3} PDF·m^2/ -2.3·10^5 DALY/yr, and egalitarian:

$-5.0\cdot10^{-3}$ PDF·m^{2}/ $-7.3\cdot10^{5}$ DALY/yr) occur due to wood production for fuel use, because of the largest wood production worldwide ($3.1\cdot10^{8}$ m^{3} in 2011). China, Brazil and Ethiopia also produced more than 100 Mm^{3} of wood in 2011, leading to large normalization factors for Brazil, and large negative factors for Ethiopia for all three perspectives. For China, large impacts are obtained for the individualist perspective and large benefits for the hierarchist and egalitarian perspective.

Impacts due to heat energy production

Fig 4 shows the contribution of each impact category to the total impact on human health (DALY/MJ) caused by the production of 1MJ of heat following the Individualist, Hierarchist, and Egalitarian perspectives, respectively. For the Individualist perspective, biogenic climate change had a relatively low contribution to the total human health impact (between+3% and +6%) of burning wood in most countries. Only for Ethiopia a negative impact was observed, which was considered as a benefit related to CO_2 emissions. The largest impact of wood combustion was caused by particulate matter in the Individualistic perspective. Relative to coal anthracite, the use of wood for heat production had higher overall climate change related impacts on human health.

In the Hierarchist perspective, we observed the largest contribution for biogenic climate change impacts in the wood combustion case. The largest contribution was observed in Russia where the biogenic climate change impacts determined were about 85% of the total impact. The relative contribution of the impacts of particulate matter to the total impacts was less than in the Individualist perspective while the relative contribution of human toxicity increased. Unlike in the Individualist perspective, the total impact due to heat production using coal anthracite was higher than the impacts resulting from use of wood biomass from India, Ethiopia and Australia. The biogenic climate change impacts were negative for these countries. However, for the other countries, wood use continued to give higher impacts on climate change related human health than use of coal anthracite for heat production.

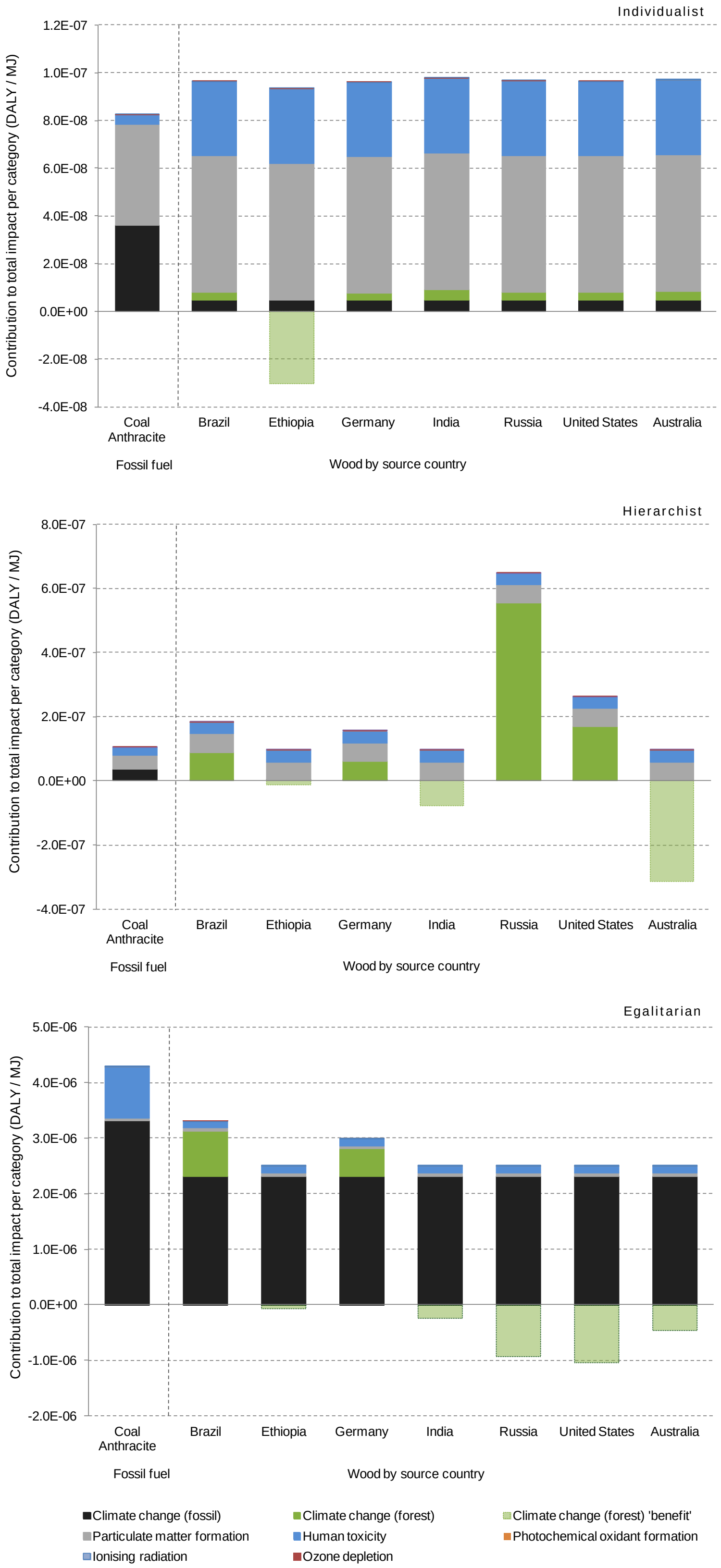

Fig 4 Impact on human health due to production of 1MJ heat energy (DALY /MJ) using coal anthracite or wood sourced from the indicated countries

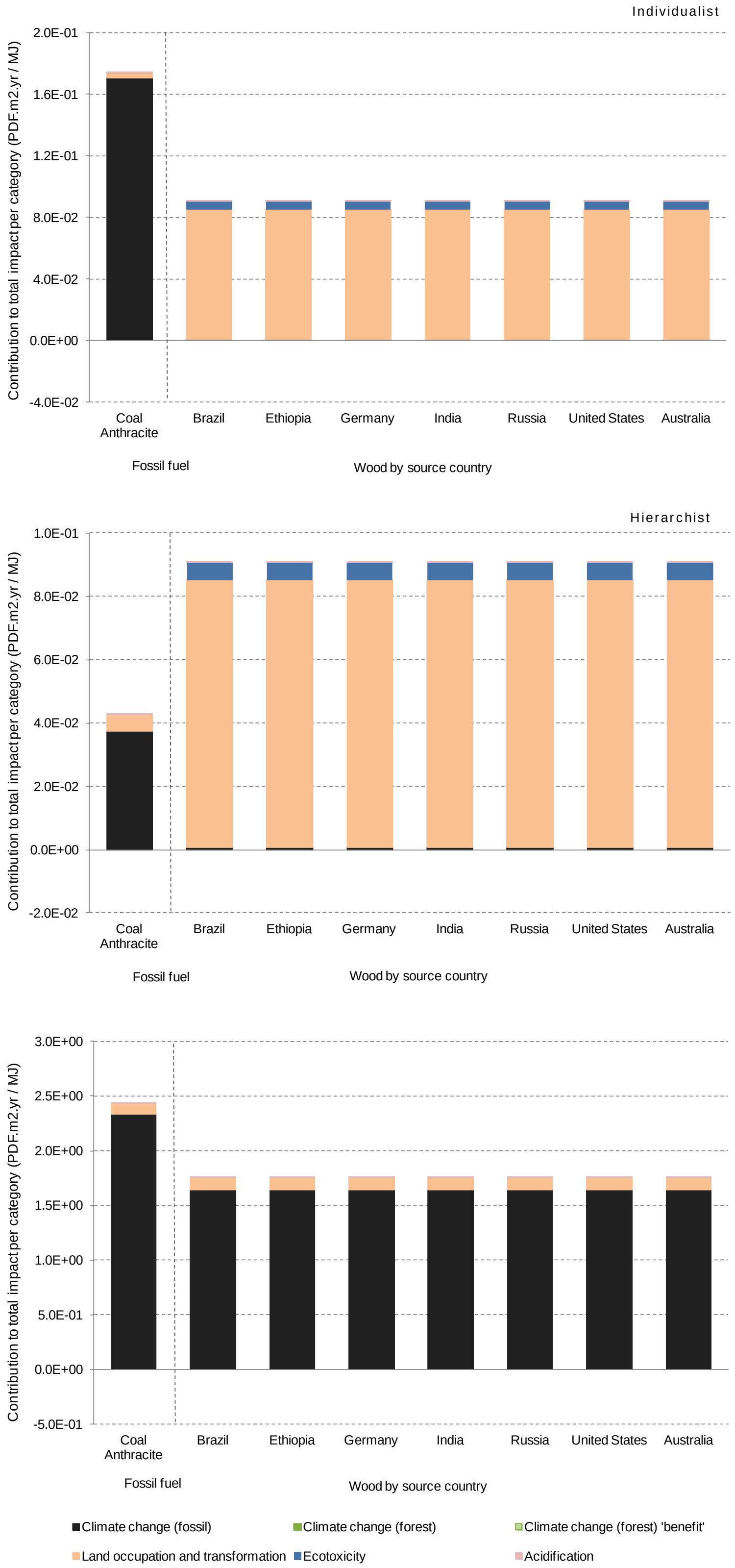

Fig 5 Impact on ecosystem quality due to production of 1MJ heat energy (PDF m^2·yr /MJ) using coal anthracite or wood sourced from the indicated countries

In the Egalitarian perspective, climate change due to fossil fuel use was the most dominant contributor to human health impacts due to wood heat production. Biogenic climate change impacts were positive in Brazil and Germany only. In all the other countries, (United States, Russia, Australia, India, and Ethiopia), the biogenic climate change impacts were negative; therefore climate change related benefits were realised from the use of wood for heat production. The total human health impacts from heat production using coal anthracite exceeded that of wood fuel use in all countries. Thus, in the long term, it is more beneficial to use wood than to use the fossil coal anthracite for heat production.

Fig 5 shows the contribution per impact category to the total impact on ecosystem quality caused by production of 1MJ of heat following the Individualist, Hierarchist, and Egalitarian perspectives. Unlike in the case of human health, the contribution of biogenic climate change impacts to the total impact on ecosystem health was < 1% for the wood combustion case. In the Individualist and Hierarchist perspectives, land occupation and transformation impacts were the most dominant impacts, contributing more than 90% to the total impacts regardless of the wood's source country. However, for coal anthracite use, the climate change fossil impacts were the greatest contributor and these were always higher for coal anthracite use than for wood fuel use. In the Egalitarian perspective, more than 90% of total impacts on ecosystem quality due to heat production were from climate change due to fossils, for both types of fuel. Total impacts from coal anthracite use were higher relative to use of wood from any of the studied countries.

4.4. Discussion

We discuss here the main limitations in our global scale modelling of carbon changes in forests and subsequent climate change related impacts on human health and terrestrial ecosystems. We then provide an interpretation of our presented results followed by a discussion on their application in LCAs of bioenergy production.

4.4.1. *Limitations and sources of uncertainty*

There are a number of limitations and uncertainties in the modelling approach we took. First, we modelled only above ground biomass carbon changes due to changes in rotation time without considering potential changes in other pools of forest carbon, such as below-ground biomass, soil organic carbon (SOC), and dead wood. The carbon content in below ground living biomass such as roots is much lower than the carbon content in above ground biomass and although SOC is a significant component of total forest carbon, various studies showed that harvesting activities in forests and changes in rotation length generally do not affect stable soil carbon stocks (Kirschbaum 2000; Johnson & Curtis 2001; Liski *et al.* 2001; Kaipainen *et al.* 2004; Lal 2004b; Davidson & Janssens 2006).

Second, the forest modelled in G4M is a generic forest, identical for the whole globe, modelled with an assumed equal share area for all age classes. In reality, forests differ in terms of e.g. tree species or wood density. Some studies showed the effect of tree species type on carbon stock dynamics. Kaipainen *et al.* (2004), for example, found that growth of spruce tree is more age-dependent than growth of pine tree. Also the yield tables from Marschall and Forstverein (1975) suggest, that even for the same species in a relatively small area, the growth dependency with age varies. This shows that region-specific differences can even be larger than indicated in our simulations. Decisions on forestry management activities such as fertilisation and harvesting technique also differ between or within regions (González-García *et al.* 2009; Michelsen *et al.* 2012). However, including more region-specific detail of the forest is currently not possible on a global scale due to data limitations.

Third, we assumed that the change in above ground forest biomass (stocking biomass) equals to a change in atmospheric carbon. A carbon increase/decrease in above ground biomass will also affect other pools than atmospheric carbon. However, the direct and the major changes will occur in these two compartments when the changes in above-ground biomass are moderate.

4.4.2. *Interpretation*

Our CFs quantify the contribution to global warming and resultant impact on human health and ecosystem quality experienced worldwide due to forest wood extraction for bioenergy. Negative characterization factors for impacts on human health should not be interpreted as an addition of years to human life. Similarly, the negative CFs for impacts on ecosystem quality do not mean introduction of new species in the other parts. They only indicate that the applied management activity, in this case increased wood extraction due to changes in rotation time, results in an increase in carbon sequestration in the forest. This is because the extraction increase is reached by creating a larger rotation time to allow for new growth in order to reach a new steady state. Longer rotation times allow for older forests which have a larger carbon stock than young ones (Davidson & Janssens 2006; Hergoualc'h & Verchot 2011; Zhang *et al.* 2012). An increase in the forest carbon stock implies a decrease in atmospheric carbon pool, thereby slowing down global mean temperature rise and the resultant impact. In a case study like our LCA of heat production, benefits due to wood extraction can only be substracted from carbon emission and not from other impacts to the environment.

The spatial variation in the magnitude and sign of CF values per perspective is due to the location specific characteristics of the forest and its local environment, i.e. current rotation time, site productivity, and the FAO based current biomass statistics used as input for the G4M model. The magnitude of the CFs also varied among the different perspectives with an increase in the CF spatial variability being observed from Individualist, to the Hierarchist, and finally the Egalitarian perspective. Obtaining positive CFs for the vast majority of the grids in the Individualist perspective, in contradiction to the large share of negative CFs in the Egalitarian perspective, shows that 20 years is too short a time length to observe significant forest carbon increase. This implies that in the first 20 years, increased wood extraction leads to reductions in amounts of sequestered carbon. While it may be theoretically justifiable to consider a 20 year time frame in line with the Individualist perspective, forest wood extraction is optimized to longer rotation time, and obtained results for the 20 years perspective might in this sense be less realistic. The positive CFs observed in the Hierarchist perspective indicate that in about 2/3rd of the forest grids, steady-state was also not reached in the 100 year time frame. For the Egalitarian perspective which assumed an infinite time horizon, there was an increase in the amount of negative CFs indicating increased amounts of sequestered carbon.

Another major cause of the difference in magnitude of CFs from one perspective to another is the climate change impact factors ($\Delta I_e / \Delta T$) applied per perspective. For the Egalitarian perspective, they were an order of magnitude greater than for the individualist perspective, while values for the hierarchist and individualist perspectives remained in the same order of magnitude.

4.4.3. *Application*

Impacts caused by forest wood extraction are dependent on a number of site-specific aspects, such as biomass stock, rotation times, and climatic conditions. Site-specific contributions to global warming are important in the determination of the environmental impacts of a product taking into account the origin of the wood. We recommend to apply the

CFs on the smallest scale possible, to capture the detailed information the CFs are based on. However, the chosen resolution for the CFs needs to be applicable in LCA, i.e. it needs to match up the spatial scale of the inventory data. Our starting point was the smallest scale currently possible, 0.5°x0.5°. These grid level CFs can be aggregated to a larger spatial scale, e.g. a country, when necessary. Such manual aggregations however result in loss of spatial detail as we observed in this study, as countries such as Canada and Brazil ultimately had a positive country CF value despite the presence of grids with negative CFs. Mutel *et al.* (2012) proposed the application of geostatistical analysis tools such as autocorrelation analysis towards defining an optimum spatial scale for impact characterisation. However, most life cycle inventory data are currently available at country level only.

Results from the heat production case study depict the influence of spatial factors in determining the effect of global wood combustion. This is evidenced by the differences in the sign and magnitude of the biogenic emissions due to wood extraction from one country to another. The results show that in current LCAs of bioenergy, climate change impacts due to heat production from wood are miscalculated when biogenic emissions due to wood extraction are disregarded, especially when quantifying impacts on human health.

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5. Spatially explicit characterization factors for damage costs of erosion due to agricultural land occupation on a global scale

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5.1. Introduction

Characterization factors (CFs) are used in the life cycle impact assessment (LCIA) of products to determine the impact that a stressor causes to humans and ecosystems. One of these stressors is land use, an intensive human activity that aims at exclusive use of land for certain purposes and adapting the properties of land areas in view of these purposes. Milà i Canals (2007a) provided a framework to address land use impacts in LCIA, outlining that impacts on biodiversity (Michelsen 2008), biotic production (Brandão & Milà i Canals 2012), as well as on regulating and life support functions (Lindeijer 2000, Milà i Canals et al. 2007b) are of importance. In order to quantify land use impacts in the LCA framework, land occupation and land transformation are considered basic land use activities (Lindeijer et al. 2002). All basic land use activities result in either damage to or benefits for ecosystem quality. Ecosystem quality can be expressed as the capacity to provide certain ecosystem services, that are generally divided into provisioning, regulating, supporting, and cultural services (MA 2005). Over the years the idea has been evolving that life cycle oriented methods should encompass ecosystem services, since these form the basis of planetary activities and human well-being (MA 2005). A recent review shows that comprehensive accounting for ecosystem services in LCA requires, among others, greater integration among existing methods, and greater understanding of the role of ecosystems in supporting human activities (Zhang et al. 2010). There are many important ecosystem services that should be addressed in LCIA (Zhang et al. 2010), among other regulating services is soil erosion, which increases with unsustainable land use. Poor land management accelerates soil erosion. A main cause is agriculture, as the cultivation of crops requires nutrient use and irrigation, contributing to soil loss, which on its turn degrades arable land and eventually renders it unproductive (Adhikari & Nadella 2011, Pimentel et al. 1995). In the framework of Life Cycle Assessment impacts of erosion were recently assessed by Saad et al. (2011) and Núñez et al. (2013). Saad et al. (2011) adopted a calculation tool that needs site-specific input data to derive CFs (ton ha·a^{-1}) for erosion regulation of land occupation and land transformation on various spatial levels for the Canadian context. Núñez et al. (2013) developed a regionalized approach for the world to quantify damage of erosion due to land occupation on the available soil resource stock and on ecosystem's NPP, linking soil organic carbon loss to biomass productivity drop using literature data. Their approach was soil type specific and applied to various crop rotation systems with food and energy crops in Spain. To apply their factors, the amount of soil loss needs to be known.

5.2. Methods

Characterization factors. Characterization Factors (CF) on an endpoint level for cultivation of crop *x* under management scenario *s* are defined as the additional costs due to erosion under cultivation compared to erosion in the reference situation. The potential natural vegetation state was selected as the reference situation. CFs are obtained per grid cell *i*:

$$CF_{x,s,i} = \frac{\Delta ER}{\Delta Y} \cdot \frac{\Delta C}{\Delta ER} = \frac{1}{Y_{x,s,i}} \cdot \left(ER_{x,s,i} - ER_{0,i} \right) \cdot C$$

where $Y_{x,s}$ is the yield of crop *x* (kg crop·m^{-2}·a^{-1}) under scenario *s*, $ER_{x,s}$ is the erosion rate of crop *x* (kg soil·m^{-2}·a^{-1}) under scenario *s*, and C are the costs associated to erosion regulation ($/kg soil).

Agricultural land use model. To simulate erosion rates, ($ER_{x,s,i}$) a global implementation of the EPIC (Environmental Policy Integrated Climate) model was used for three different agricultural management intensities. EPIC is a deterministic biophysically based agro-environmental simulation model operating on a daily time step (Williams et al. 1984). EPIC is driven by spatially explicit information on climate, weather, topography, soil and agricultural management practices. The major processes simulated in EPIC include biomass production, water, nutrient and carbon cycling, soil erosion, and crop growth. Model output includes information on e.g. crop yields, hydrological variables (evapotranspiration, runoff, and percolation), sediment transport, nitrogen leaching, green house gas emissions, and soil carbon sequestration (Izaurralde et al. 2006, Williams et al. 1984). Erosion regulation was determined as the sediment yield (kg·m^{-2}·a^{-1}), calculated with the revised universal soil loss equation (RUSLE, Renard et al. 1997). 10-year EPIC simulation results are averaged to provide the soil sediment rates in each grid cell i and crop yield estimates ($Y_{x,s,i}$) for each scenario.

Input data. A detailed description of the global EPIC input data which was collected during the GEO-BENE project can be found in Skalský et al. (2008). Historical monthly climate ERA-40 statistics from 1957 to 2002 were obtained from the Tyndall Centre for Climate Change Research of University of East Anglia (UK, see Mitchel et al, 2004). From the monthly statistics on solar radiation, precipitation, maximum and minimum temperature and relative humidity daily weather was generated by the EPIC built-in weather generator. A global soil dataset was created by combining data on global soil distribution (DSMW) with soil profile data (WISE) obtained from the International Soil Reference and Information Centre (ISRIC, NL) and described in Batjes et al., 2006. Digital terrain information was derived GTOPO sources (Global 30 Arc Second Elevation Data; http://eros.usgs.gov).

To organize the spatial data in the global database a global grid was created as the primary geographical reference. To avoid redundant model runs, model grid cells with homogeneous input data were aggregated following a two-step approach. First, Homogeneous Response Units (HRU) were spatially delineated as zones of the global grid based on topography and soil characteristics that remain relatively stable over time. The HRU's were defined on the basis of 5 altitude classes, 5 soil textural classes (course, medium, heavy, stony, peat), and 7 slope (degrees) classes (Skalský et al. 2008). Daily weather files were generated from monthly Tyndall ERA-40 statistics from 1957 to 2002 and interpolated and geographically referenced on a 30` resolution grid. In a second step, Simulation Units were created to form the spatial basis for running the global EPIC simulations. Simulation Units (SIMU) were defined as intersections of HRU's with the 30` weather data grid cells, and administrative country level delineations, resulted in individual landscape units with homogenous topography, soil and weather conditions, resulting in 212,000 unique SIMU's. The maximal area of a SIMU is equal to an area of one 30` spatial resolution grid pixel and its area decreases from about 300,000 ha on the equator to about 30,000 ha in high latitudes; the minimal area of a SIMU is equal to an area of one 5` spatial resolution grid pixel and its value decrease from about 8500 ha at the equator to about 950 ha in high latitudes.

Cropland management. In this analysis, 4 crops i.e. cassava, rapeseed, sunflower, wheat have been simulated with EPIC worldwide, except in current wetlands, or ice, rocks, and desert areas. Crop calendars with information on planting and harvesting dates were collected from a variety of sources including FAO, USDA, European (MARS) and national agricultural institutes. Three input systems have been distinguished for the EPIC simulations:

i. Subsistence farming; no fertilization, no irrigation (subsistence scenario).
ii. High input; with 90% of crop nitrogen requirements satisfied to a maximum application of 200 N kg ha^{-1} a^{-1}, without irrigation (rainfed scenario).
iii. Irrigation systems; 90% of crop nitrogen requirements satisfied to a maximum application of 200 N kg ha^{-1} a^{-1}, and 90% of crop irrigation requirements satisfied to a maximum application of up to 300 mm a^{-1} (irrigated scenario).

Potential natural vegetation state. The reference situation was simulated with EPIC as well to provide consistency in model outcomes. Rangeland and trees were simulated around the world as an estimation of the potential vegetation when there would be no crop cultivation. Ramankutty and Foley (2010) published a dataset representing potential natural vegetation before human alteration on a 0.5°x0.5° scale. We ascribed the 17 potential land cover classes to either tree, rangeland, or not applicable (see SI). Subsequently we ascribed either rangeland or tree simulated erosion ranges from EPIC to each grid to come to a baseline scenario of erosion rates in a potential natural vegetation state.

Costs. Pimentel et al. (1995) and Crosson (1995) estimate the onsite costs of water applied and lost due to erosion of 2 \$ t^{-1} and nutrients lost of 3 \$ t^{-1}. The latest reviews regarding erosion costs show that the numbers of Pimentel et al. (1995) still hold today (Kuhlman et al. 2010, Santos Telles et al. 2011). Kuhlman et al. (2010) estimate the offsite costs to be around 1.8 times lower than onsite costs, which matches the estimate by Pimentel et al. (1995) of 3 \$ t^{-1}. We therefore estimate the total costs of erosion to be 8 \$ t^{-1} in 1995. Correcting this number for inflation to 2012 leads to 13 \$ t^{-1} of erosion costs $^{-1}$.

Spatial aggregation. For land use, spatial differentiation is very important as the impacts of land use are dependent on local conditions (climate, soil composition, vegetation, etc.). To be applicable in LCA, however, information is generally not available on such a small scale. Therefore, CFs were aggregated on a country level based on the total amount produced per grid.

5.3. Results

Characterization factors. Grid and country-specific CFs for wheat for the subsistence scenario (no fertilizer use nor irrigation) are shown in Figure 1. Grid-specific CFs for the subsistence scenario have a 95% range between $6.7 \cdot 10^{-4}$ to 2.1. CFs for the irrigation and the rainfed scenario show smaller CFs due to larger yields, having a 95% range of $2.4 \cdot 10^{-7}$ to $1.8 \cdot 10^{-3}$, and $4.8 \cdot 10^{-7}$ to $7.3 \cdot 10^{-2}$, respectively. Largest CFs were found around and below the equator. For some grids, the potential natural vegetation showed larger erosion rates than wheat cultivation, hence negative characterization factors were derived. When aggregating to country level (yield-weighted), the negative factors showed no influence due to the insignificant yields in these grids compared to the overall country yield.

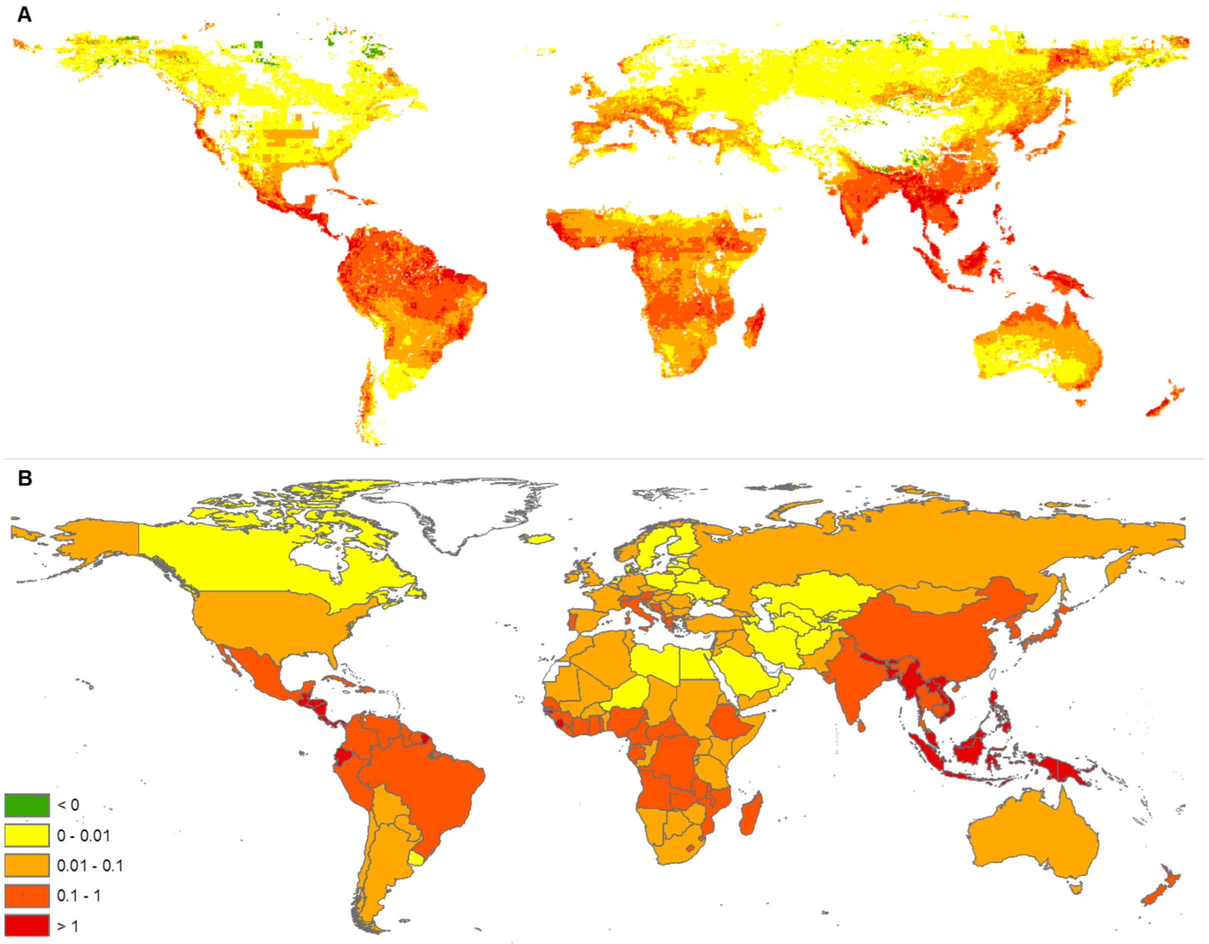

Figure 1. Grid (A) and country (B) -specific characterization factors ($ kg^{-1} crop) for wheat over the world for the subsistence scenario without fertilizer use and irrigation. The grids and countries shown in white include areas not suitable to grow crops (i.e. wetlands, deserts, snow and ice).

Figure 2A shows that larger CFs are obtained when management without input of nutrients or irrigation is performed on wheat. The small yields obtained without fertilizer and irrigation lead to higher costs per kg crop due to erosion than when fertilizer and irrigation are applied. For sunflower (Fig 2B) and rapeseed (Fig 2D) the same trend is observed although slightly less pronounced. For Cassava (Fig 2C), the rainfed and subsistence scenario lead to comparable country-specific CFs, while for the irrigated scenario various countries show smaller CFs than for the other scenario's.

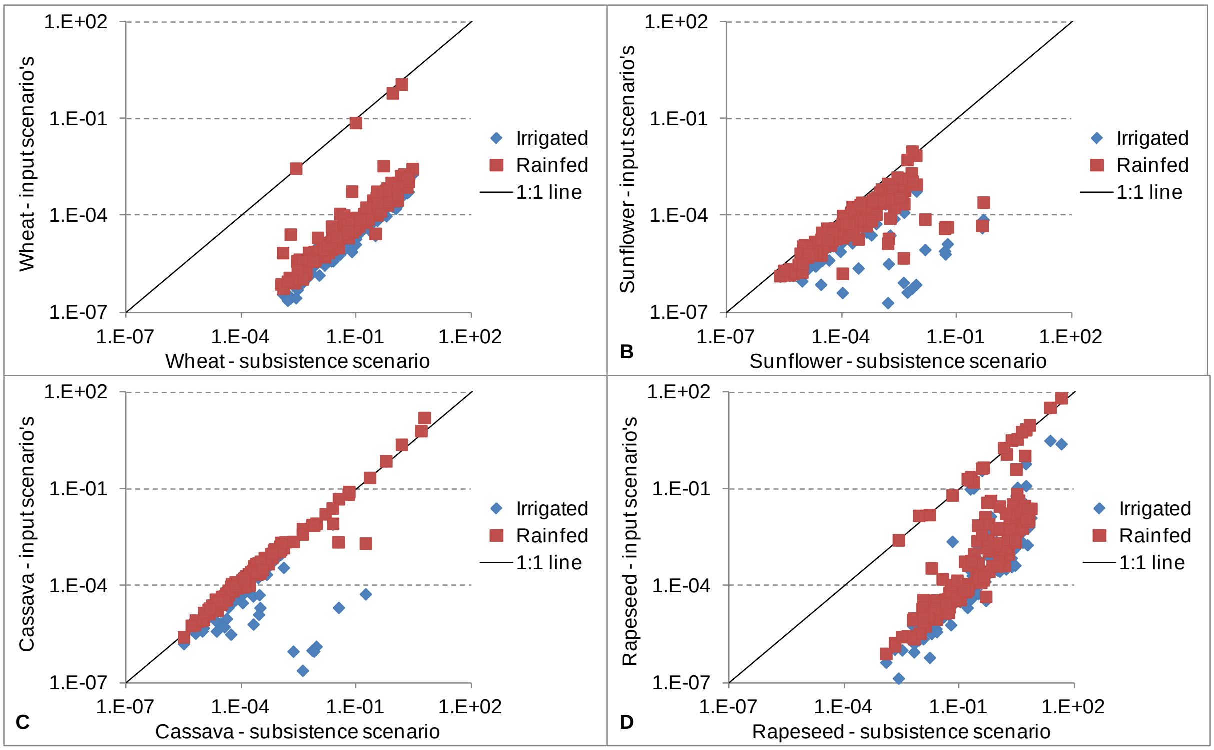

Figure 2. Country-specific characterization factors ($ kg^{-1} crop) for (A) wheat, (B) Sunflower, (C) Cassava, and (D) Rapeseed. The rainfed and irrigated management scenario's are compared to the subsistence scenario.

Figure 3 shows the country-specific CFs ($ kg^{-1} crop) for sunflower, cassava, and rapeseed compared to the CF for wheat for all 3 management scenario's. A positive trend can be seen for the CFs of each crop compared to the CFs of wheat. However, CFs can substantially differ from each other. CFs for sunflower are a factor of 920 lower than for wheat for the subsistence scenario, and a factor of 313 lower for the rainfed scenario, while the CFs for the irrigation scenario are on average equal for these two crops. For cassava, CFs are on average 242 and 297 times lower than the CFs for wheat for the subsistence and rainfed scenario's respectively, while for the irrigation scenario are on average equal again for these two crops. CFs for rapeseed are almost all larger than the CFs for wheat for all 3 management scenario's, but for subsistence only up to a factor of 4, while for the rainfed and irrigated scenario's they can be up to 797 times larger. These differences show that crop-specific CFs are of importance.

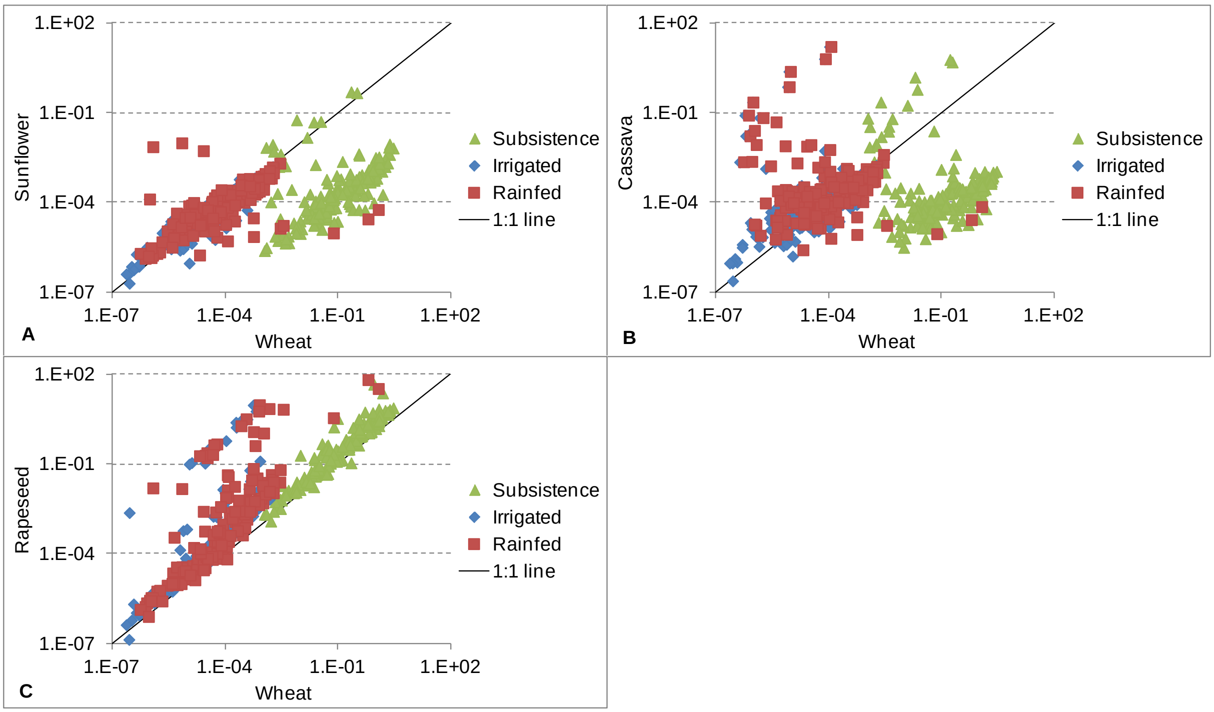

Figure 3. Country-specific characterization factors ($ kg^{-1} crop) for sunflower, cassava, and rapeseed compared to the characterization factor for wheat for all three management scenario's.

Model performance. EPIC was originally developed at the field scale and is therefore a relatively detailed model and thus data intensive. EPIC has been extensively tested, calibrated and validated against observation under various conditions at the field scale (e.g. Billen et al. 2009, Cabelguenne et al. 1990, Rosenberg et al. 1992, Williams et al. 1989). The EPIC simulations presented here combine a wealth of global data and as such represent the current state of the art in global crop modeling. Nevertheless, quality and validation status of such large-scale applications is continuously being improved (see e.g. Balkovič et al. 2012, Van der Velde et al. 2010, Wriedt et al. 2009). Improvements of input data and validation status (e.g. comparisons against reported yields) will improve model simulations and consequently any CFs derived from this.

Characterization factors that may be derived with less data intensive approaches may be compared to our results to better understand which level of detail is necessary to derive meaningful CFs. This will of course be very dependent on the level of aggregation and the need for e.g. region or watershed specific CFs or information on the within country variability in CFs.

5.4. Conclusions

We derived grid-specific characterization factors expressing the damage due to soil erosion worldwide when cultivating various agricultural crops. The damage is expressed as the additional costs per kg of crop, i.e. wheat, rapeseed, sunflower and cassava. Characterization factors are larger when crops are cultivated with no fertilizer nor irrigation input than with these inputs due to low crop yields. Differences between rainfed and irrigation systems are relatively low. Total environmental impacts will, however, include toxicity due to pesticide use, eutrophication due to fertilizer use, and impacts due to water withdrawals in irrigated systems as well. Decisions on the management type to use therefore do not only depend on impacts due to erosion only. Differences between crops can be several orders of magnitude, showing the importance of differentiating between crop types. We show that our method is applicable for various crop types and management scenario’s. It leads to crop and management type specific CFs.

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6. Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain

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6.1. Introduction

Life cycle assessment (LCA) aims to provide a general picture of the environmental impacts of resource consumption and emissions during the entire cycle of products and systems. As a first approach, LCA uses spatially and temporally independent linear impact assessment (LCIA) models to provide this holistic perspective. Site-specific conditions are therefore not taken into account when the environmental impacts of the product are evaluated. The impacts of the different LCA categories have consequences for the environment and human welfare on different spatial scales. This has nothing to do with the importance of the categories, but with a need for spatial differentiation for some impact categories. Since economic processes are spread worldwide, local impacts have a global extension as well (UNEP, 2003).

Despite the fundamental role of ecosystem goods and services in sustaining human activities, there is no harmonized and internationally agreed method for including them in LCA. According to a key framework for land use impact assessment in LCA (Milà i Canals et al., 2007a), ecosystem goods and services that should be integrated within LCA are impacts on biodiversity and, at least, impacts on the following five major ecosystem services: biotic production potential, carbon sequestration potential, freshwater regulation potential, water purification potential and erosion regulation potential. Operational characterization factors and methods covering impacts on biodiversity (De Schryver et al., 2010; Koellner and Scholz, 2008) and on ecosystem services (Brandão and Milà i Canals, 2012; Müller-Wenk and Brandão, 2010; Saad et al., 2011) have been recently proposed. Impacts of erosion were addressed by Saad et al. (2011), who focused on the degradation of the erosion regulation function due to land transformation and land occupation using different spatial scales. Characterization factors (CFs) were developed within a Canadian context and therefore they are not globally applicable.

Because an activity's land use impacts depend on local conditions, a conventional site-independent LCA methodology might not be very accurate. Methods focusing on these impacts should therefore include geospatial information in both the inventory (LCI) and LCIA phases. Different levels of regionalization (e.g., countries, ecoregions, biomes) and ecological unit classifications (e.g., life zones by Holdridge, 1947, ecoregions by Olson et al., 2001) in the LCIA are presently used without a clear recommendation on a standardized approach to address spatial differentiation. Although the use of ecological or geographical units instead of administrative borders provides better estimates of the site-dependency of land-use impacts, especially in countries with a high degree of variability, it is generally easier to find information at the country scale.

Our objective was to go one step further toward the integration of ecosystem services in LCA by developing a globally applicable and spatially resolved method to include land occupation impacts on the erosion regulation. Indicators of the impact category were defined on the endpoint level, which means that were modeled up to the entities described by the areas of protection (AoP), i.e. soil resources, ecosystem quality and human health. The case study conducted to demonstrate the applicability of the method focused on the impacts of agricultural rotations with energy crops in Spain as compared to the cultivation of traditional crops.

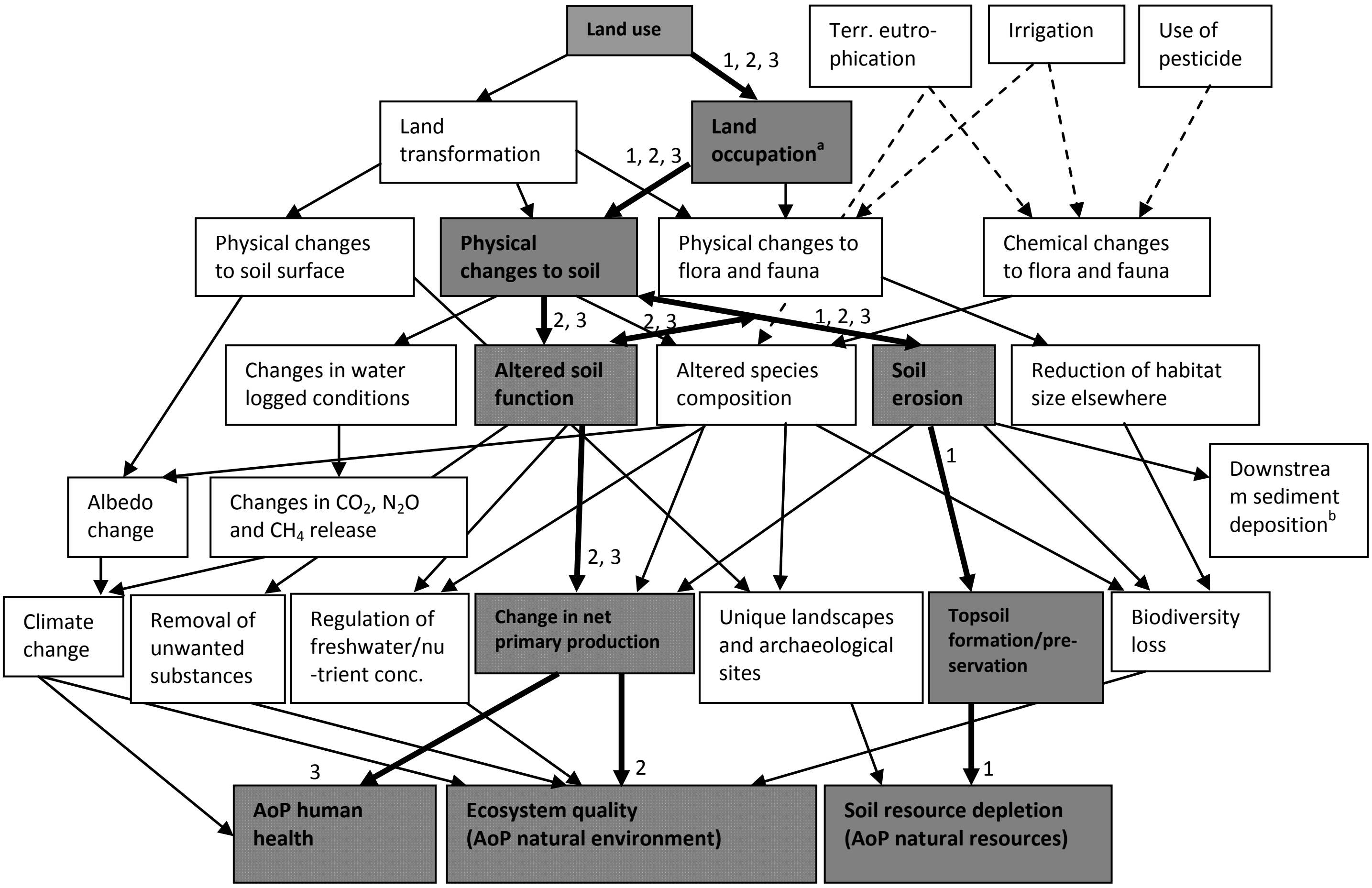
6.2. Methods

Figure 6.1 illustrates the general land-use impact mechanism and shows the three impact pathways studied, as follows:

Land occupation leads to soil erosion and this leads to loss of topsoil reserves, which leads to soil resource depletion (impact pathway 1 in Figure 6.1).

Land occupation leads to soil erosion and altered soil function, which affects net primary production and leads to damage to the ecosystem quality (impact pathway 2 in Figure 6.1).

Land occupation leads to soil erosion and altered soil function, which affects net primary production and leads to damage to human health (impact pathway 3 in Figure 6.1).



[a] Land occupation does not “cause changes” but contributes to prolong altered conditions.

[b] Benefits from downstream deposition of eroded material disregarded in the paper, as sediments cross the system boundary.

Figure 6.1: main impact pathways related to land use. The pathways discussed in the paper are shown with cross-hatching and thicker arrows (adapted from the International Reference Life Cycle Data System handbook, ILCD, JRC, 2010).

6.2.1. Resource-depletion impact pathway

This AoP is concerned with the removal of resources from the environment (impact pathway 1 in Fig. 6.1). Annually, humans cause the loss of 50 to 75 billion metric tons of soil (Harvey and Pimentel, 1996). Agricultural land accounts for 75% of the soil erosion worldwide, though it also occurs in other human-modified ecosystems, such as during the construction of roads and buildings. More than 75% of the arable soils of the world suffer from moderate to very high soil losses (Reich et al., 2001), generally ranging from 10 to 100 t $ha^{-1}y^{-1}$ (Pimentel et al., 1987), which is at least tenfold higher than the average rate of soil formation of 0.5-1 t $ha^{-1}y^{-1}$ (Mann et al., 2002). Current soil losses due to land use (land occupation) reduce soil availability as a future resource.

Life cycle inventory data requirements: the type of land use has a determining role in the quantity of soil loss, as specific direct physical interventions are often related to land management. In LCA, ecoinvent database v3.0 (Weidema et al., 2011), based on Global Land Cover 2000 (Bartholomé and Belward, 2005) and ecoinvent v2.0 (Frischknecht et al.,2007), identifies ten main types of land-use and land-cover classes in the first level (e.g., agriculture, forest), and provides more detailed information on land intensity and management in the following levels (e.g., arable non-irrigated, arable non-irrigated extensive). This tiered structure allows for different levels of detail in the LCI, depending on the quality requirements of the LCA study and the information available on the product under study. We chose this flexible classification system to record the land occupation type in the LCI. Yet, ecoinvent v3.0 does not distinguish between cultivated crops and specific crop management practices (e.g., type and timing of tillage operations, use of residue mulches), which are key factors controlling soil erosion on agricultural lands. In order to carry out an accurate soil erosion assessment in agricultural LCA studies, ecoinvent v3.0 should be further refined by registering the type of land use as specific as possible.

Soil losses due to the land occupation must also be included in the LCI as soil loss mass (g). There are many estimation models that can be used to this end, such as PESERA (Gobin and Govers, 2003), INRA (Le Bissonais et al., 2002) or USLE (Wischmeier and Smith, 1978). From all of them, we recommended the universal soil loss equation erosion model (USLE) to register soil losses in the LCI. USLE takes into account the effect of a particular land-use type on water erosion. There is a consensus that the USLE equation and its update (revised USLE, RUSLE, Renard et al., 1997) are valid methods to estimate soil losses by water at the inventory stage (Beck et al., 2010; Muys and García Quijano, 2002). The use of (R)USLE to predict soil erosion losses has spread worldwide thanks to the growing availability of and accessibility to climatic, edaphologic and land-use and land-cover data at the local and regional level. This spatial information can be increasingly found in a geo-referenced format, thus allowing for data processing and visualization in geographic information system (GIS) software. Furthermore, there are many biophysical models, such as EPIC (Williams et al., 1984) or APEX (Williams and Izaurralde, 2005), which already have algorithms to simulate soil erosion with (R)USLE for many crops and on different spatial resolutions. Results from scenario simulations from these models can be incorporated in the inventory stage to register soil erosion.

Soil formation was not contrasted to soil erosion because factors other than human land use, namely climate and the soil parent material, are generally recognized as the most important aspects governing soil formation (Jenny, 1994).

As land occupation impacts are recorded in area per time (e.g., m^2 y) the area as well as the duration required for the production of a certain amount of products and services have to be gathered in the LCI.

The georeferenced location of land use (longitude/latitude) should also be included if available. Failing that, a broader resolution (e.g., region, country) can be used, although this reduces the quality of the LCI data by increasing uncertainty due to the high variability of spatial conditions.

In summary, the following elementary flows need to be accounted in LCI: type of land use, soil erosion by water, time of occupation, area of occupation and location of the activity.

Impact assessment model: in line with other endpoint methods (Ecoindicator 99, EI99, Goedkoop and Spriensma, 2001), damage to resources is expressed as surplus energy needed to make the resource available at some point in the future. This is a suitable unit to evaluate soil depletion, which indicates the anticipated energy removal from nature to provide a unit of soil eroded during land occupation. Instead of using energy units (MJ-equivalents), such as in EI99, we used emergy units (MJ-solar equivalents). Unlike the energy metric, emergy accounts for quality differences of the energy used to generate a product or service by converting raw units (e.g., kg soil, m^3 water) to a common basis, i.e., units of solar energy. The advantage of emergy compared with other energy and exergy units is that it not only accounts for energy carriers (e.g., gas) and non-energetic materials (e.g., minerals), but also assigns an exergy (useful energy) value to land use (Rugani et al., 2011). Emergy therefore evaluates the (solar) energy the natural system is deprived of to yield the new stock of soil lost during land use. We assigned an average energy quality to soil (called the resource's transformity in the emergy literature (Odum, 1996) and the solar energy factor (SEF) in LCA (Rugani et al., 2011)) for all soil types and locations: 23.9 $MJ_{se}\ g^{-1}$ soil loss (Odum, 1996). The effect of soil erosion on soil resource depletion(ΔR) is expressed as follows with units of MJ-solar equivalent soil loss per unit of area and time of land occupation:

If soil loss = 0, $\Delta R = 0$

$$\text{If soil loss} > 0,\ \Delta R = \underbrace{A \times t \times \text{Soil loss}}_{\text{LCI}} \times \underbrace{\frac{SD_{ref} - SD_i}{SD_{ref}} \times SEF_{soil}}_{\text{CF}} = MJ_{se}\ m^2 y \qquad \text{Equation 6.1}$$

This endpoint indicator combines the inventory flow (i.e., soil loss) with the local available soil reserves (SD_i, soil depth in the specific location i) on a spatial resolution of 5 arc-minutes (approximately 10×10 km^2, FAO/UNESCO, 2007), and with the solar energy factor of soil (SEF_{soil}) as the characterization factor. Soil loss mass is weighted with soil depth, as the environmental significance of soil loss depends on the soil stock size at the specific location. Twenty-one soil-depth classes were distinguished, from very shallow soils (0.05 m deep) to very deep soils (2.25 m deep). These classes resulted from the combination of five major soil-depth categories. Local soil reserves were normalized with a reference soil depth (SD_{ref}). We selected the upper limit of the very deep soil-depth category as the reference (SD_{ref} = 3m) as an indicator of the potential soil quality. This would only occur if the whole of the grid-cell area had this maximum soil depth, which we assumed was not possible once the first soil particle was eroded. Note therefore that SD_i can never be equal to SD_{ref}, so the characterization factor always takes positive values and any amount of soil loss will have an environmental impact. Choosing the maximum soil depth as the reference was judged to be a more representative correction of the site dependency of the characterization factor, rather than the soil reserves of a particular region (e.g., Swiss lowlands for ecosystem-quality assessment in the EI99 methodology, Goedkoop and Spriensma, 2001). Larger damage factors were assigned to thinner soils by calculating the difference between SD_{ref} and SD_i before normalization with the soil reference. This means that, based on the same amount of soil loss, thinner soils are more vulnerable than thicker soils. The indicator relates impact assessment to biogeographical conditions in each grid cell i (SD_i) without any further aggregation of land-use type or land-use cover. For a land-use activity under study, the lower the indicator result, the less soil resource depletion and the less damage to the environment.

6.2.2. Ecosystem-quality impact pathway

This AoP is concerned with negative effects on the function and structure of natural ecosystems (impact pathway 2 in Fig. 6.1). In the erosion process, soil quality declines, essential plant nutrients are lost and soil depth is reduced. As a result, biomass productivity diminishes. Ultimately, this can adversely affect overall biodiversity and ecosystem quality. Numerous positive correlations between plant biomass productivity (NPP) and vascular plant species diversity and richness have been established (Costanza et al., 2007; Flombaum and Sala, 2008), though overall biodiversity does not always correlate with productivity (e.g., Mediterranean hotspots and intensively managed agricultural lands).

One of the methods used in LCA for measuring environmental impacts on ecosystem functions is based on the soil organic matter (SOM) content (Milà i Canals et al., 2007b). We assessed the effects of soil erosion on the terrestrial ecosystem quality by, in a first step (damage factor), linking soil loss to soil organic carbon (SOC) loss, and, in a second step (effect factor), linking SOC loss to biomass production drop. Thus, ecosystem biomass production was modeled as a function of soil quality, which is indicated by the soil organic carbon content of the soil lost. We assessed ecosystem biomass production as a function of the net primary production of potential natural vegetation (NPP_0, i.e., the anticipated state of mature vegetation in the absence of human intervention).

Life cycle inventory data requirements: Soil organic carbon losses must be registered in the LCI. Such as soil erosion, SOC losses can be directly derived using biophysical models like EPIC (Williams et al., 1984) or APEX (Williams and Izaurralde, 2005). Alternatively, SOC losses may be calculated by, in a first step, determining the topsoil OC content and, in a second step, multiplying the already estimated soil losses by the percentage of topsoil OC content. The quantity of SOC in the topsoil can be determined by direct measurements. In the absence of plot-level soil data, SOC content can be determined using already existing spatial data layers (e.g., map of organic carbon in topsoils in Europe, Jones et al., 2005; Harmonized World Soil Database, HWSD, FAO et al., 2009), though inventory data quality will be affected. On the other hand, while site-specific measurements are more accurate, this is not likely to be achievable in most LCA studies (Milà i Canals et al., 2007b).

The soil unit where the activity is developed should also be included in the LCI using the most recent FAO classification (FAO et al., 1990), which identifies 28 different soil units (soil types), each with harmonized soil parameters. The soil unit can be identified by direct measurements on the occupied land under analysis. Otherwise, it may be approximated using literature or spatial databases such as the Harmonized World Soil Database (FAO et al., 2009). Again, direct measurements are more accurate, although not always available in LCA studies.

As a rough estimate of the SOC in each soil unit, we determined the content of the 28 soil units in the FAO classification system (Table 6.1). This was done by averaging the topsoil OC content of over 16,000 soil mapping units in the HWSD (FAO et al., 2009), which holds, with a resolution of 30 arc-seconds (approximately 1×1 km^2), information on selected soil parameters of soil units in the entire land area of the world.

The 28 soil units were grouped into seven major categories (Table 6.1). In 24 out of 28 soil units there was less than 2% SOC, the threshold selected by the European Commission for defining soils in phase of pre-desertification (COM, 2002).

As in the case of the inventory for the soil-resource depletion indicator, information on the type and intensity of land use, area size (m^2), duration (y) and the location of the occupation should be recorded.

Impact assessment model: According to the International Reference Life Cycle Data System (ILCD) Handbook (JRC, 2010), which is a series of technical documents that provide detailed guidance on all the steps required to conduct a LCA study, species diversity is the recommended indicator to be implemented in endpoint LCA methodologies when modeling damage to ecosystem quality. Effects on species diversity are usually quantified in terms of the Potentially Disappeared Fraction of vascular plant species (PDF, Koellner, 2000), such as in the EI99 (Goedkoop and Spriensma, 2001), ReCiPe 2008 (Goedkoop et al., 2009) and IMPACT2002+ (Jolliet et al., 2003) methods. However, according to ILCD, function-related parameters, such as the biomass production of the ecosystem that we used here, might also be good endpoint indicators.

A very limited number of studies have focused on accounting for current or potential NPP losses caused by soil erosion, due to the complex connection between the two factors. In addition to soil properties, NPP depends on many other physical environmental aspects, such as leaf area index, precipitation, atmospheric CO_2 concentration and temperature (Melillo et al., 1993). NPP is therefore commonly used as an indicator to reflect ecosystem response to climate change. Change in climate may decrease NPP (lower precipitation or cloudiness) or increase it (photosynthesis enhancement). Due to the many relationships between factors, estimates of the isolated linkage between NPP and soil erosion are scarce and highly uncertain.

Of the studies performed on the relationship between soil loss and NPP/NPP_0 loss, most estimate productivity losses according to qualitative degrees of erosion (light/slight, moderate, strong/severe, extreme/very extreme erosion) and limit the scope to the local or regional level (Mann et al., 2002; Mokma and Sietz, 1992). We developed a first approach to convert soil loss-NPP0 loss qualitative relationships at the global level found in the literature (Dregne and Chou, 1992; FAO/UNEP, 1984; Zika and Erb, 2009) into approximate quantitative linear relationships. This was done for each soil unit group in Table 6.1. Using this method, soil loss was first related to the loss of soil organic carbon as a measure of soil quality and, finally, to ecosystems' loss of biomass productivity.

In the impact assessment model, the inventory flow (i.e., SOC losses) should be used as an input parameter in the equation in Table 6.1 for the soil unit where the land-use activity is developed. Note that for SOC losses greater than or equal to a predetermined threshold for each soil unit group (when NPPD=100), NPP_0 is completely lost and the soil is unlikely to be able to recover. Soils with low SOC content, such as those in arid and semi-arid areas, are less resilient than soils rich in SOC, which are usually found in wet regions. These SOC loss thresholds are equivalent to a soil depletion of approximately 65 t $ha^{-1}y^{-1}$, which means extreme/very extreme soil loss. Most agricultural land in the world loses soil at a rate of between 13 and 40 t $ha^{-1}y^{-1}$ (Pimentel and Kounang, 1998) whereas losses of more than 100 t $ha^{-1}y^{-1}$ only occur in extreme events (Morgan, 1992).

Table 6.1: topsoil organic carbon (% weight) of the 28 soil units in the Soil Map of the World (FAO et al. 1990), according to the HWSD (FAO et al., 2009), and linear equations used in the impact model. Soil units within the same soil category are arranged by increasing SOC content. SOC_{loss} in the $\%NPP_0$ depletion (NPPD) equations is expressed as g C loss (in a square meter and year).

Soil units HWSD	Topsoil organic carbon (% weight)	$\%NPP_0$ depletion equations
Gypsisols Arenosols Calcisols Solonchaks	0 - < 0.5	If $SOC_{loss} = 0$, $NPPD = 0$ If $SOC_{loss} > 0$, $NPPD = 4.09 \times SOC_{loss} + 2.66$ for $SOC_{loss} < 23.80$ g $NPPD = 100$ for $SOC_{loss} \geq 23.80$ g
Lixisols Luvisols Solonetz Plinthosols Planosols Fluvisols Regosols Leptosols	0.5 - < 1.0	If $SOC_{loss} = 0$, $NPPD = 0$ If $SOC_{loss} > 0$, $NPPD = 1.96 \times SOC_{loss} + 2.66$ for $SOC_{loss} < 49.66$ g $NPPD = 100$ for $SOC_{loss} \geq 49.66$ g
Acrisols Vertisols Cambisols Anthrosols Kastanozems Ferralsols	1.0 - < 1.5	If $SOC_{loss} = 0$, $NPPD = 0$ If $SOC_{loss} > 0$, $NPPD = 1.32 \times SOC_{loss} + 2.66$ for $SOC_{loss} < 73.74$ g $NPPD = 100$ for $SOC_{loss} \geq 73.74$ g
Greyzems Podzoluvisols Alisols Nitisols Phaeozems Chernozems	1.5 - < 2.0	If $SOC_{loss} = 0$, $NPPD = 0$ If $SOC_{loss} > 0$, $NPPD = 0.88 \times SOC_{loss} + 2.66$ for $SOC_{loss} < 110.61$ g $NPPD = 100$ for $SOC_{loss} \geq 110.61$ g
Gleysols Podzols	2.0 - < 2.5	If $SOC_{loss} = 0$, $NPPD = 0$ If $SOC_{loss} > 0$, $NPPD = 0.69 \times SOC_{loss} + 2.66$ for $SOC_{loss} < 141.07$ g $NPPD = 100$ for $SOC_{loss} \geq 141.07$ g
Andosols	4.86	If $SOC_{loss} = 0$, $NPPD = 0$ If $SOC_{loss} > 0$, $NPPD = 0.31 \times SOC_{loss} + 2.66$ for $SOC_{loss} < 314.00$ g $NPPD = 100$ for $SOC_{loss} \geq 314.00$ g
Histosols	34.60	If $SOC_{loss} = 0$, $NPPD = 0$ If $SOC_{loss} > 0$, $NPPD = 0.04 \times SOC_{loss} + 2.66$ for $SOC_{loss} < 2433.50$ g

$$\text{NPPD} = 100 \text{ for } SOC_{loss} \geq 2433.50 \text{ g}$$

The effects of soil erosion on ecosystem quality (ΔEQ) are expressed using a growth-based value: NPPD (potential net primary production depletion). For an occupation of 1 m^2 and 1 year, NPPD ranges from 0 to 1:

$$\text{If } SOC_{loss} = 0, \quad \Delta EQ = 0$$

$$\text{If } SOC_{loss} > 0, \quad \Delta EQ = \underbrace{A \times t \times \frac{aSOC_{loss} + b}{100}}_{\text{LCI}} \times \underbrace{\frac{NPP_{0,i}}{NPP_{0,ref}}}_{\text{CF}} = \text{NPPD } m^2y \qquad \text{Equation 6.2}$$

This endpoint model combines the inventory flow (a×SOC_{loss}+b, if $SOC_{loss} > 0$, i.e., mass of SOC losses transformed into %NPP_0 losses) with NPP0 values spatially resolved for each grid cell ($NPP_{0,i}$) at 5 arc-minutes (approximately 10×10 km^2, Haberl et al., 2007) to obtain the absolute biomass productivity drop at the specific location. Values were then normalized with an NPP_0 value corresponding to that of the ecosystem with the highest biotic productivity worldwide ($NPP_{0,ref}$ = 1496 g C $m^{-2}y^{-1}$) as a representative reference of the potentiality of the ecosystem. Larger impact factors were allocated to the most productive soils (higher $NPP_{0,i}$). From an ecosystem services viewpoint, we made the general assumption that ecosystems with higher NPP values are considered more valuable, as NPP is a scarce resource on earth (Pfister et al., 2011). The most productive lands can also be used for a greater diversity of purposes. Unlike Equation 6.1, $NPP_{0,i}$ is normalized without first calculating the difference between $NPP_{0,ref}$ and $NPP_{0,i}$. This implies that the CFs of both AoP are not at all comparable and do not have to be, as they represent different environmental concerns. Note that, for complete losses of NPP_0 (a×SOC_{loss}+b=100), damages only depend on the $NPP_{0,i}/NPP_{0,ref}$ ratio. As for the resource-depletion indicator, regionalization was at the grid-cell level, without aggregating values on broader scales. For a land-use activity under study, the lower the indicator result, the less of a drop in biomass production and the less damage to the environment.

6.2.3. Case study on energy crops grown in Spain

The proposed method was applied to agricultural plots of food and energy crops in Spain. This was selected as a representative case study because water erosion is one of the main causes of land degradation in the country (EEA, 2005) and, more importantly for validating the methodology, because soil erosion is unevenly distributed, with a gradient from north to south and from the Atlantic to the Mediterranean coast (MERMA, 2012). Because an increase in energy-crop production is forecast in Spain (EEA, 2006), it is necessary to assess the environmental cost of this alternative, given the effect of agriculture on the environment. To illustrate the outlined method, we analyzed the environmental impacts of growing five three-year agricultural rotations, two of them with energy crops, on the erosion regulation. The analysis includes 120 agricultural plots covering the main Spanish water basins (Figure 6.2). The functional unit selected for the assessment was a land occupation of 1m^2 during 1 year (m^2y), considering the average soil and SOC loss of a complete crop rotation system. We used the attributional framework so that any potential additional consequence of planting the rotations on other parts of the economy was not included. Nor did we include past impacts on soils from historical land use. The five crop production systems assessed were rotations with food and energy purposes. Three of these were traditional rainfed rotations of annual crops grown in the Mediterranean region: i) winter barley-winter wheat-rye, ii) winter barley-winter wheat-pea, and iii) winter barley-winter wheat-unseeded fallow. Another was a rainfed rotation where a bioenergy crop was introduced: iv) winter barley-winter wheat-oilseed rape; and finally, a deficit-irrigated short rotation coppice of a perennial crop: v) poplar-poplar-polar. All are extensive systems, as the economic income from non-irrigated and bioenergy agriculture is low.

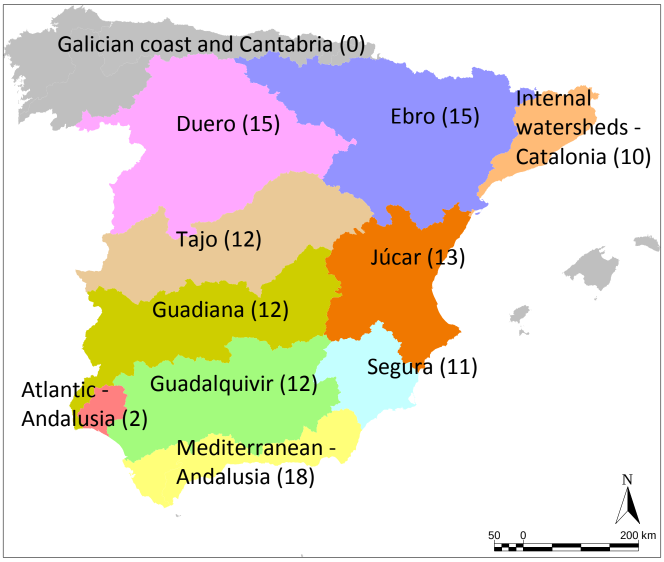


Figure 6.2: geographical distribution of the plots in the Spanish water basins. The number in brackets indicates the quantity of plots studied within the watershed.

6.3. Results

6.3.1. LCIA characterization factors

Regional characterization factors for the resource-depletion and ecosystem-quality impact pathways of the erosion regulation ecosystem service are shown in Figure 6.3. For resource depletion, the ratio is $((SD_{ref} – SD_i)/ SD_{ref})*SEF_{soil}$. Lower soil depths and therefore higher damage factors are found at high northern latitudes and over wide areas of Asia. For ecosystem quality, Figure 6.3 shows the $NPP_{0,i}/NPP_{0,ref}$ ratio. Higher biomass productivities and therefore damage factors are found at the low latitudes of the tropics. For both the resource-depletion and ecosystem-quality indicators, lower CF values indicate less sensitivity of the ecosystem to potential land-use impacts.

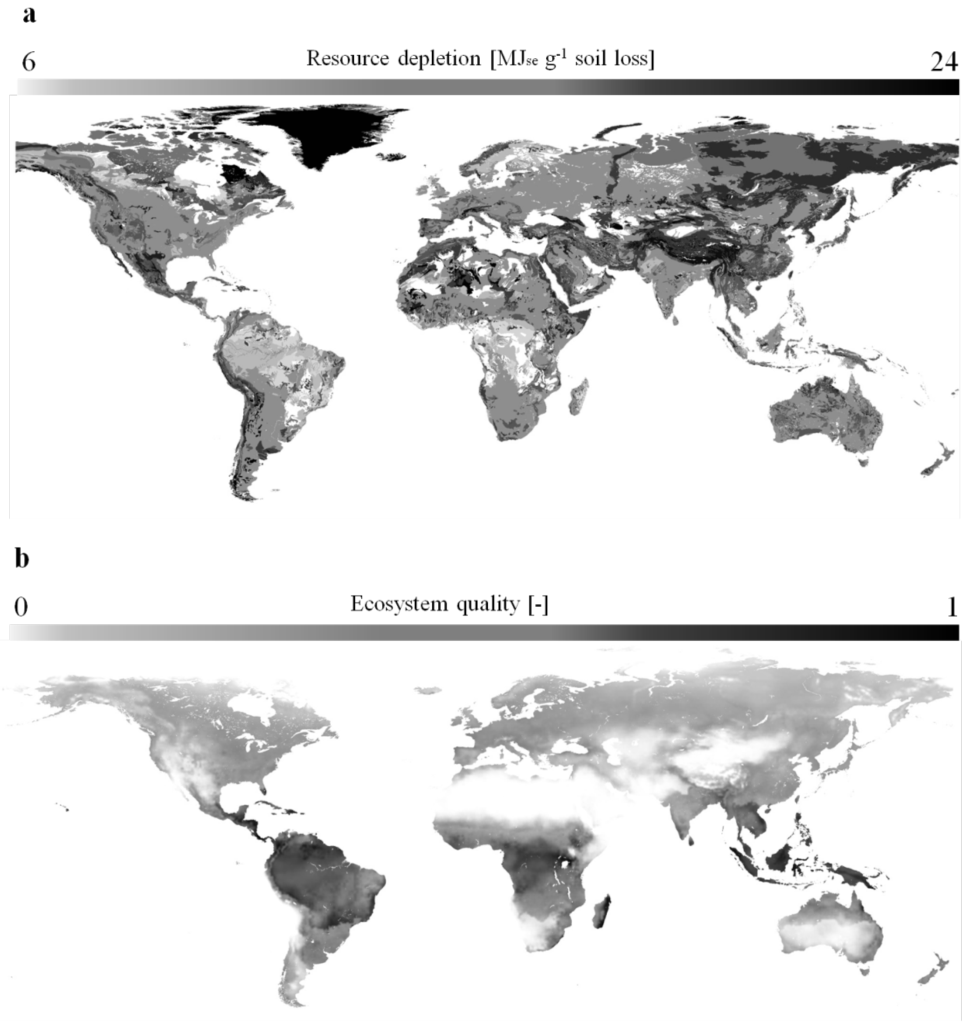


Figure 6.3: characterization factors for (a) resource depletion and (b) ecosystem quality.

6.3.2. Case study on energy crops grown in Spain

Both soil and organic carbon losses and impact factors vary as a function of location, thus leading to considerable differences in the environmental damage from soil erosion in different water basins.

In the LCI, the crop rotation with the greatest erosion rate (Table 6.2) was when the field lay fallow in the last year (winter barley - winter wheat - unseeded fallow). These soil losses were around 10 times higher than the crop rotation with the lowest erosion rate, the energy profitable poplar short-forestry rotation (poplar - poplar - poplar). For annual cereal (winter barley - winter wheat - rye), legume (winter barley - winter wheat - pea) and energy-crop (winter barley - winter wheat - oilseed rape) rotation systems, similar soil losses were recorded, with rates about 40% lower than for fallow rotation. Similar differences between rotations were found in the soil erosion LCIA.

The hydrological basins in Spain with the lowest water erosion rates and environmental damage, thus making them the most appropriate for rotating poplar and oilseed rape energy crops, are the Duero (Northern Spain) and the Guadiana (Central Spain), while those with the highest water erosion rates and environmental impact are the internal watersheds of Catalonia (Northeast Spain) and the Júcar basins (Eastern Spain).

Table 6.2: life cycle inventory and life cycle impact assessment results per m^{2}y of land occupation.

	Internal watersheds-Catalonia	Ebro	Duero	Júcar	Tajo	Guadiana	Segura	Guadalquivir	Mediterranean-Andalusia	Atlantic-Andalusia
LCI – soil erosion [10^{3}g]										
B-W-R[a]	1.58	0.43	0.27	1.13	0.56	0.31	0.39	0.68	0.69	0.59
B-W-P[b]	1.63	0.45	0.29	1.16	0.58	0.31	0.39	0.68	0.68	0.58
B-W-F[c]	2.44	0.70	0.46	1.78	0.90	0.50	0.60	1.07	1.01	0.85
B-W-OR(*)[d]	1.51	0.41	0.24	1.03	0.54	0.28	0.38	0.63	0.63	0.56
PP(*)-PP(*)-PP(*)[e]	0.28	0.09	0.05	0.20	0.10	0.06	0.07	0.11	0.09	0.08
Resources [MJ$_{se}$ m^{-2}y^{-1}]										
B-W-R	2.5 E+04	6.4 E+03	4.1 E+03	1.9 E+04	9.0 E+03	5.0 E+03	5.8 E+03	9.9 E+03	9.7 E+03	9.9 E+03
B-W-P	2.6 E+04	6.6 E+03	4.4 E+03	2.0 E+04	9.3 E+03	5.2 E+03	5.8 E+03	9.8 E+03	9.7 E+03	9.9 E+03
B-W-F	3.9 E+04	1.0 E+04	7.0 E+03	3.0 E+04	1.4 E+04	8.2 E+03	9.2 E+03	1.6 E+04	1.5 E+04	1.4 E+04
B-W-OR(*)	2.4 E+04	6.0 E+03	3.6 E+03	1.8 E+04	8.7 E+03	4.6 E+03	5.6 E+03	9.2 E+03	9.2 E+03	9.4 E+03
PP(*)-PP(*)-PP(*)	4.5 E+03	1.3 E+03	7.5 E+02	3.4 E+03	1.6 E+03	9.0 E+02	9.7 E+02	1.6 E+03	1.4 E+03	1.3 E+03
Ecosystem quality [NPPD m^{-2}y^{-1}]										
B-W-R	0.12	0.03	0.02	0.07	0.04	0.03	0.02	0.05	0.04	0.05
B-W-P	0.12	0.04	0.02	0.07	0.04	0.03	0.02	0.05	0.04	0.05
B-W-F	0.18	0.05	0.03	0.11	0.06	0.04	0.03	0.07	0.06	0.07
B-W-OR(*)	0.11	0.03	0.02	0.07	0.04	0.02	0.02	0.05	0.04	0.05
PP(*)-PP(*)-PP(*)	0.03	0.02	0.01	0.02	0.01	0.01	0.01	0.02	0.02	0.02

[a] winter barley-winter wheat-rye
[b] winter barley-winter wheat-pea
[c] winter barley-winter wheat-unseeded fallow
[d] winter barley-winter wheat-oilseed rape
[e] poplar-poplar-poplar
Asterisks indicate crops for energy use

6.4. Discussion

6.4.1. Case study on energy crops grown in Spain

The results of the case study showed that the implementation of the poplar energy-crop rotation system in Spain can potentially reduce erosion rates and related environmental impacts per area-time unit compared to traditional cereal and legume crop rotations in the country. Apart from erosion, other aspects such as biodiversity impacts or replacement of other crops which then need to be imported, would need to be considered to assess if planting poplar in Spain is really an environmentally-friendly decision.

Although the area-time unit does not relate to energy crops' function of producing biobased electricity or fuels, the framework is still meaningful when ranking crop rotations by their impact on soil replacement energy and lost productivity. Sustainable land-management practices reduce impact intensity in a given area. However, intensive agricultural production can diminish soil quality and jeopardize the preservation of productive agricultural land. While impact per unit of output energy would have been a meaningful indicator to compare energy crops, this is not applicable when energy and food crops are being compared. The average results of Table 6.2 are highly variable within a water basin due to the disparity of soils, climates and ecosystem biomass productivities. There is a need for statistical analyses combining basins and geospatial features to show to what extent the average results can be extrapolated across the watershed. Aggregation at the water-basin level was not specific enough to reflect a common trend, though it is a useful reference area to compare to and/or combine with results of a water-use impact assessment. Apart from taking into account environmental impacts from soil erosion, other impact category indicators should be also accounted for in the selection of the most adequate watersheds for growing energy crops in Spain. For water consumption and environmental damages, it has been reported (Núñez et al. 2012) that the most suitable locations in Spain for energy-crop rotations are basins in the northeast of the country, while they should not be cultivated in some southeast basins. According to these results, there is not a specific water basin capable of minimizing both water consumption and soil erosion impacts at the same time.

6.4.2. Soil erosion impact assessment model

Analysis of the method followed the general evaluation criteria and specific sub-criteria relevant for land-use impacts in ILCD Handbook (JRC, 2010). The aim of the analysis was to qualitatively address model uncertainty, facilitate comparison with other soil-erosion impact assessment models, and identify the strengths and weaknesses of our method.

Completeness of scope: two relevant impact pathways leading to AoP soil resources and ecosystem quality were addressed at the endpoint level. We expressed ecosystem damages using a growth-based value (NPPD), while the majority of endpoint methods quantify ecosystem impacts on PDF (Goedkoop and Spriensma, 2001; Goedkoop et al., 2009; Jolliet et al., 2003). A significant correlation has been found (Pfister et al., 2009) between vascular plant species biodiversity and net primary productivity of the actual vegetation, which led the authors to select NPP as a proxy for ecosystem quality. The same proxy could be applied to transform NPPD to PDF, taking into consideration that we used potential instead of actual vegetation. Globally, the NPP_0 to NPP_{act} ratio is between 0.9 and 1.2 in 81% of the terrestrial area.

Similarly, for the indicator of soil resource depletion, we used emergy units, as in the work by Rugani et al., (2011), whereas most endpoint resource demand indicators are expressed in energy units (MJ), as in the EI99 (Goedkoop and Spriensma, 2001) and IMPACT2002+ (Jolliet et al., 2003) methods. In reality, any energy measure for reconstructing a renewable resource such as soil lacks meaning. To allow for comparisons with other methods and impact categories, energy values can be transformed into emergies using the resource-specific solar equivalent factor. When this is done, the results can be compared to or aggregated with surplus emergy demands of energy-carriers (e.g., crude oil: 0.091 MJ_{se} g^{-1}) as well as non-energetic resources (e.g., water supply: 0.203 MJ_{se} g^{-1}, Zhang et al., 2010). We assumed an SEF_{soil} of 23.9 MJ_{se} g^{-1} soil loss (Odum, 1996), as this is the only available estimate. This global value does not distinguish between different soil types, land uses and world regions, so it should be further refined.

One main soil erosion-related impact pathway for human health was identified (impact pathway 3 in Fig. 6.1): soil erosion affects productivity of agricultural and pasture lands, leading to a reduction in food availability. This, in turn, results in one of two scenarios, depending on the regional context: (1) increase in malnutrition or undernutrition in the so-called deficiency scenarios (i.e., developing countries) or (2) food importation or changes in food production in the so-called compensation scenarios (i.e., wealthy countries), using the same terminology agreed for water use in LCA (Bayart et al., 2010). Deficiency scenarios are unable to adapt to productivity losses and this generates an impact on human health, which results in a loss of quality of life or longevity, usually measured in endpoint methods with the unit of disability-adjusted life years (DALY). In contrast, compensation scenarios are wealthy enough to offset a lack of food, so human health impacts due to soil erosion are avoided. Many regional socio-economic parameters may influence definition as a compensation or deficiency scenario, such as the gross domestic product (GDP), the percentage of malnutrition and the human development index. The connection between soil erosion and human health damages is highly complex and dependent on many regional conditions that are difficult to reflect in the LCA methodology. We therefore left the modeling of this AoP for future research. Existing LCA methods (Motoshita et al., 2011; Pfister et al., 2009), in which human damages due to water consumption are partially covered, are a helpful starting point to model soil erosion-related human health impacts.

The characterization model and factors are globally applicable and spatially defined, taking into account ecosystem biomass productivity and soil characteristics at grid level. How to aggregate these grid-cell-specific factors on a wider and still accurate scale (e.g., ecoregions, land cover, water basins) is a complex unresolved issue due to the huge variability of soil types even at the landscape scale. This variability of soil types makes it difficult to identify a standardized approach to address spatial differentiation.

Environmental relevance: the method focuses on the assessment of land occupation impacts (i.e., the use of a land area for a specific human purpose) for any type of human activity, whenever soil erosion rate of the assessed land use activity (e.g., industrial, mining) is available or estimated by the LCA practitioner. However, the method discounts impacts due to land transformation (i.e., change of a land area to make it suitable for a specific use). For occupation, it is debatable what period of occupation should be considered in the agricultural LCA inventory (i.e., duration of the crop, duration of the crop plus the fallow period, crop rotation). For both transformation and occupation, the choice of the reference situation to measure the magnitude of the change and the time needed after occupation to recover this reference situation are two fundamental issues to be agreed upon to properly assess impacts from both land use interventions. So far, the recommended reference situation in attributional LCA is the so-called potential natural vegetation after land occupation (Milà i Canals et al., 2007a), which differs from the natural situation because nature rarely returns to its original state after being disturbed.

Of the overall cause-effect chain for land use (Figure 6.1), we focused on the soil-erosion impact mechanism by quantifying changes in topsoil preservation and changes in the NPP_0 due to altered soil function. For a complete evaluation of land-use impacts, the soil erosion assessment must be complemented with indicators that measure effects on climate change, biodiversity loss, water and nutrient regulation and unique landscapes.

Scientific robustness and certainty: the two indicators reflect the cause-effect chain from the interventions to the latest environmental damages. For resource depletion, the impact can be expressed at the midpoint level if soil losses of the LCI are weighted by the available reference-corrected soil depth without further transformation to emergy units. The cause-effect chain for ecosystem quality directly links soil erosion to the NPP_0 depletion indicator, which can be further modeled to PDFs.

The geographical differentiation of the model has good potential for being improved and further developed when more detailed global maps of soil properties and biomass productivity of ecosystems become available. The current large-scale maps do not fully resolve the real diversity of soil and ecosystems.

The indicator for ecosystem quality can be partially verified against monitoring data by measuring the loss of NPP at different degrees of soil erosion. Note that we used potential NPP, so the model can be tested in areas with potential or near potential vegetation. The indicator for resources cannot be verified (MJ_{se}), as emergy is an abstract concept.

Model uncertainties were qualitatively evaluated using the set of criteria listed in the ILCD Handbook (JRC, 2010). Most statistical and decision rule uncertainties still have to be estimated. Considerable statistical uncertainties are attached to the ecosystem-quality impact-assessment model. Our approach of linking soil erosion to NPP loss is based on very limited and highly uncertain studies about the effect of soil erosion on ecosystem's biomass production. This adds substantial uncertainty to our model. Moreover, we allocated quantitative ranges of soil loss to qualitative classes of soil loss and then calculated a linear regression with the average of each range. Choosing other quantitative ranges or a value different from the average of each range would have given different NPPD equations. Another source of uncertainty arises from the type of relationship established between SOC losses and NPP_0 losses. We assumed both variables are linearly related. The linear model was used instead of non-linear models because there is insufficient knowledge of the type of relation between the two variables and the possible interference of other variables (e.g., climate change).

Further statistical uncertainty that affects input data in the LCI and the characterization factors is the uncertainty arising from the resolution of soil data. Also, for the resource-depletion indicator, the use of a site-dependent solar energy factor would reduce uncertainty. Uncertainty also comes from the type of spatial aggregation of the LCI and characterization factor results. These are key future research directions to improve the reliability of the model.

Documentation, transparency and reproducibility: the documentation used for the model is published and readily accessible. The maps used to derive characterization factors are available on line. This availability of input data allows third parties to further develop and improve the impact factors and the model.

Applicability: the characterization factors are applicable by general LCA practitioners. Incorporation in current LCA software will require adaptation of the software to tackle spatially explicit data in GIS format. Estimates of the soil erosion needed as LCI data and measured with USLE require some knowledge of soil sciences and experience in the application of the equation.

Stakeholder acceptance: for the ecosystem-quality indicator, results are easily interpretable and understandable by non-LCA experts. In contrast, a possible barrier hindering acceptance of the soil resources indicator is the use of surplus emergy units, which may be deemed too complex and meaningless for most people. The absence of soil and climate data availability can also hamper the applicability and acceptance of the two indicators.

6.5. Conclusions

We developed a globally applicable, spatially differentiated LCIA method to account for land occupation impacts in LCA, focusing on the aspect of soil erosion. LCI data required and data sources and models that can be used to obtain the inventory flows were also identified. Spatially explicit damage factors on a grid-cell level resolution (approximately 10×10 km^2) for the entire world were provided for resources and ecosystem quality endpoints. The LCA model was successfully applied to agricultural plots in Spain to compare soil erosion-related environmental impacts that may have substituting traditional food for energy crop rotations. Results from the case study show that the lowest erosion rates and environmental damage occur when rotating poplar short rotation coppice in Northern and Central water basins in Spain.

Further research should focus on improving reliability of the model by reducing statistical uncertainty arising from the key sources discussed in the article.

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7. Assessment of seafloor impacts in seafood LCAs – A desk study and stepwise guide[6]

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7.1. Background

General principles

Seafloor impact of fishing is gaining increased attention both by scientists and in public debate. It is remarkable how little is known about the distribution of trawl fishing effort compared to industrial activities on land. While it is clear that some types of fishing such as bottom trawling (Fig.1) does impact benthic communities, as has been described by a considerable body of literature consisting of local and regional case studies (Jones 1992; Collie et al. 2000; Kaiser et al. 2006; FAO 2005). Bottom trawls are dragged over, close to or on the seafloor, depending on the target species and there are also pelagic species targeting species that do not live close to the seafloor (but higher in the water column) and pelagic trawling is therefore not considered to cause any seafloor impacts. Bottom trawls and dredges (the latter used to fish e.g. scallops and other types of mussels) inflict initial mortality to many benthic species and the question is where the threshold values are and how much of the damage will be sustained over long time(Kaiser et al. 2006). Models of the heavy trawled North Sea however suggest that the long-time impacts can be substantial (Hinz et al. 2009; Frid et al. 2009). Some argue that trawling can be compared to ploughing an acre and that it will only increase the production of the seafloor (by making nutrients and prey available), others compare trawling to forest clear-cutting (Watling and Norse 1998). There is a body of research showing clear impacts in local to regional experimental trawl studies or comparisons between trawled and untrawled sites, although unevenly distributed. It seems to be clear that both production and biomass in seafloor communities decrease after trawl disturbance (Hiddink et al. 2006), which threatens both structural and functional biodiversity (Thrush and Dayton 2002) and could alter the carrying capacity of the ecosystem (Shephard et al. 2010). It has also been stated that, even if the comparison with ploughing would hold, we would never dream of allowing ploughing to be done in such a random way as demersal trawling and dredging is currently is (Hiddink et al. 2006). The Millennium Ecosystem Assessment considered benthic impacts to be a major driver of loss of marine biodiversity, although exceeded by overfishing of fish stocks (MEA 2005). There is a gowing need for quantitative indicators of benthic impacts both to evaluate progress in relation to the Marine Strategy Framework Directive and, more generally, as part of ecosystem-based fisheries management.

This work is planned to be undertaken during the fall 2012 as originally planned, it will be reported in the final deliverable of the project in 2013 and will be subject to internal and external review then. The text here is a description of the planned approach for this methodology.[6]

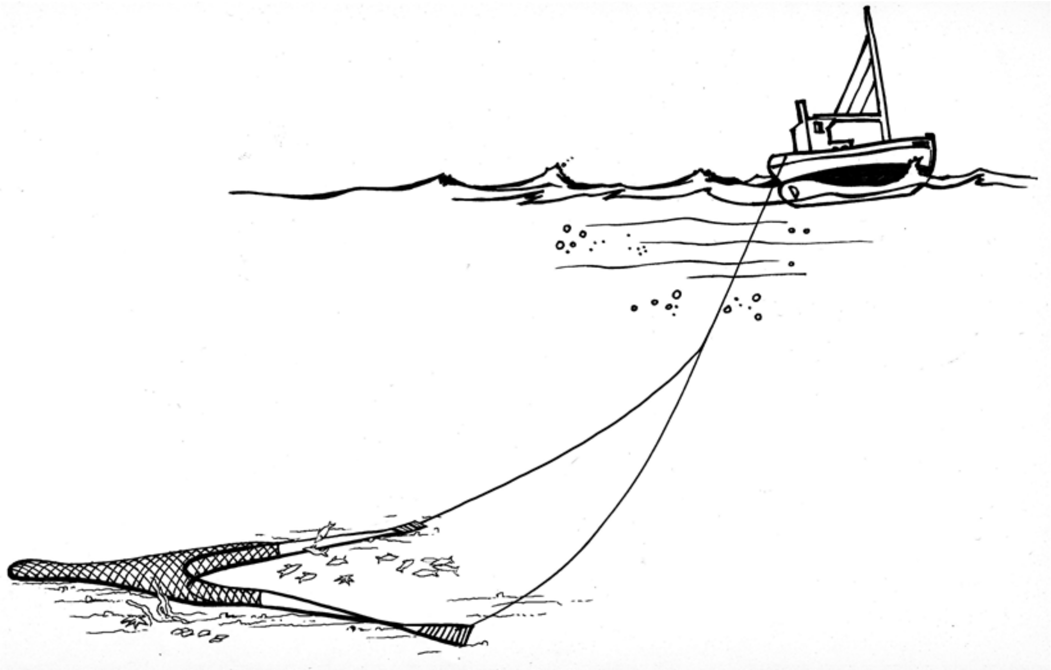


Figure 1. A bottom (or demersal) trawl with iron trawl doors, keeping it open while trawling. Pelagic trawls target species living higher in the water column and are not in contact with the seafloor.

Some attempts have been made to draw more general conclusions from these individual, local studies, both by reviewing them (Collie et al. 2000), by developing indicators on fishing impacts from them (Hiddink et al. 2006) (Piet and Hintzen 2012) and by meta-analysing them (Kaiser et al. 2006). The studies build on each other to various extents and even though over 1700 experimental manipulations (combinations of seafloor habitats or species assemblages and specific fishing pressures) have been identified in published literature, these are not independent from each other as they result from only 68 studies and they do not represent a balanced experimental setup of independent samples to allow proper statistical analysis. Rather, some combinations of gear and habitat are overrepresented while others are underrepresented or completely missing (Table 1). Moreover, results for sensitivity and/or recovery rates of species/phyla/communities/habitats are somewhat inconsistent between studies, indicating that there are also interactions, making generalisation for use in e.g. management or LCAs difficult.

Seafloor impact in LCA

Life Cycle Assessment of seafood products is a rapidly growing research field, and today over 100 published case studies of products originating either in capture fisheries, aquaculture or feed production systems exist (Parker 2012). The amount of peer-reviewed articles is considerably lower, and only some of them incorporated fishery-specific impact categories to include issues of large relevance for fisheries (Pelletier et al. 2007) (Vázquez-Rowe et al. 2013), some examples of such studies are described below.

Early approaches of integrating marine seafloor impact in LCA assessed the swept area corresponding to the amount landed (i.e. a functional unit) first introduced in case studies of Baltic and Norwegian cod fisheries (Ziegler et al. 2003, Ellingsen et al. 2006) and trawling for

Norway lobster (Ziegler and Valentinsson 2008) and Senegalese shrimps (Ziegler et al. 2009, 2011). More recent applications have been Spanish mackerel (Ramos et al. 2011) and octopus fisheries (Vazquez-Rowe et al. 2012), and a study comparing selective and non-selective trawling for Norway lobster (Hornborg et al. 2012). In a currently ongoing case study (Emanuelsson et al. 2013), the seafloor impact is modeled using high resolution data on fishing positions to measure the average trawl speed. Ziegler et al. (2003) specified the part of the total area swept affecting areas with anoxic conditions, based on the assumption that trawling anoxic areas would lead to less impact on benthic communities as no higher forms of life can survive in these areas.

In some cases, only the fishing effort in terms of trawl hours has been available without positioning data (Ziegler et al. 2011) Early case studies (Ziegler et al. 2003, Nilsson and Ziegler 2007) were based on logbook data with only the set position of the trawl, while today the increased implementation of satellite based Vessel Monitoring Systems (VMS), in many fisheries give much higher resolution on fishing positions, often providing a position every hour. The basic principle is however still the same, that a swept area can be estimated by the average trawl speed, fishing effort (hours trawled) and effective width of gear and is attributed to the landings during the same period.

Aim of this desk study

The aim of this study is to summarize the state of knowledge regarding seafloor impacts of fishing and present a number of approaches that could be used in seafood LCAs or in fisheries management. It can represent the starting point of the future development of a more quantified seafloor impact assessment methodology.

This is achieved by presenting a hierarchy of different approaches that could be taken to handle seafloor impacts in seafood LCAs from a midpoint towards an endpoint basis. The overview also visualizes current trade-offs between LCA impact assessment framework, data availability and scientific robustness. The first and simplest of the approaches included is the state of the art indicator swept area (section 2). A practical guide to seafloor impact assessment explaining in which situations seafloor impacts of fishing are at all important to consider and whether one approach or the other should be chosen, is provided in section 3.

7.2. Overview of alternative approaches to seafloor impact assessment

State of the art today for including seafloor impacts in seafood LCAs is as mentioned above to calculate the seafloor area swept by trawls or dredges per functional unit. While this is not easy in itself, since it requires data collection on specific trawl width, speed and landings per unit of fishing effort in each specific fishery, it fails to account for the fact that different gear types lead to different levels of impact because of technical characteristics and the way they are operated (some are dragged closer to the seafloor than others and some even in the seafloor). At least equally important as the gear type used is the type of habitat impacted, as it has been widely documented that different habitats have a different sensitivity to fishing disturbance and the frequency of disturbance, i.e. the intensity of fishing pressure (NAS 2004, Kaiser et al. 2006, Peat and Hintzen 2012).

One broadly distributed perception is the fact that sediment particle size is correlated to habitat sensitivity, the coarser the particles, the lower the sensitivity. Another commonly held view is that exposure to natural disturbance, which is often correlated with depth (the deeper a habitat, the less it is influenced by natural wind and wave exposure) and sometimes with particle size, also explains the sensitivity of a habitat to fishing disturbance. This view is based on the reasoning that a habitat whose inhabitants are adapted to frequent natural disturbance of a similar kind as that caused by trawling, will also be more resistant to fishing impacts (NAS 2004) . A third view that has been presented is that it is the complexity of the habitat that makes it more sensitive to fishing impacts, indicating that all types of three-dimensional structures (e.g. coral and polychaete reefs, maerl and mussel beds) are the most sensitive habitats (Auster 1998).

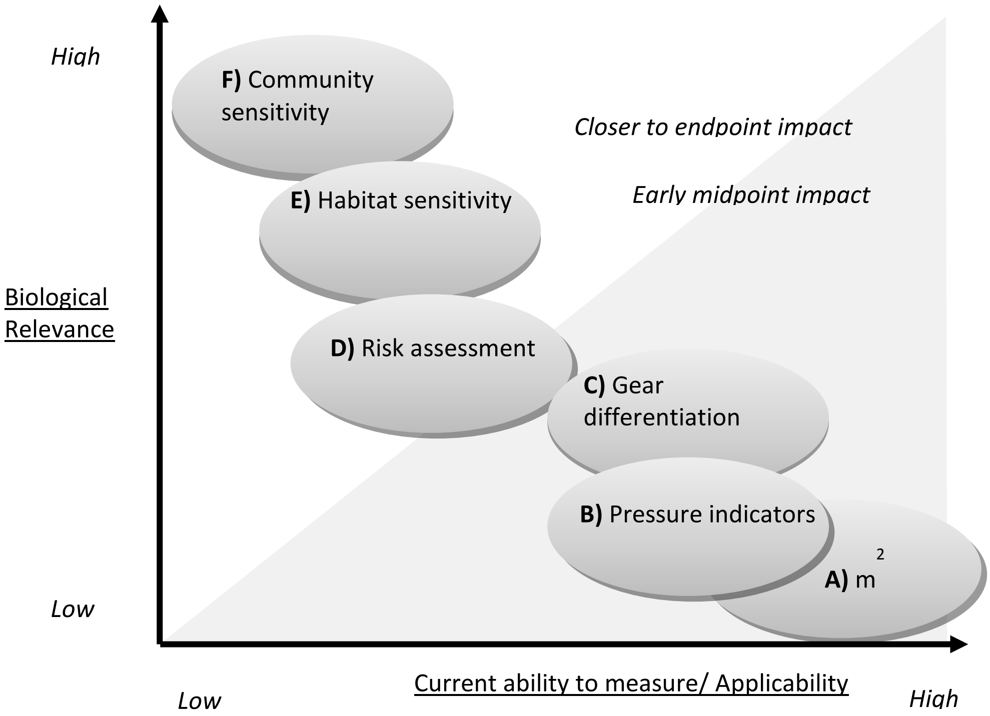
However, the conclusion in literature today is that it is not as simple as that. One cannot follow one single measure (particle size, depth or habitat complexity) as there seem to be important interactions between these factors and the gear type. Therefore, they cannot alone be used as proxies for habitat sensitivity and/or recovery rate straight away.

In the process of reviewing the literature, we identified a number of different possible approaches to quantifying seafloor impacts of fishing and we started off by placing them in a schematic graph with LCA compatibility on the Y axis and accuracy or robustness on the X axis, as we found them to differ with regard to these aspects (Fig. 2).

Figure 2. Schematic graph over the different approaches A-F considered in this overview ranked according to LCA compatibility and current applicability/scientific robustness.

A) Swept area

The swept area is a relatively simple and entirely technical indicator to calculate, independent of biological data and spatial and temporal variability. However, guidelines on how to calculate this indicator would be helpful, since assumptions and methodological choices made will influence the result and trawl designs are not very standardised, rather it is quite difficult to identify a "typical" trawl in a fishery. A guideline has been discussed in relation to using swept area an proxy for fishing mortality (Kotwicki et al. 2011). Attempts have been done to relate vessel size and gear type to models of gear opening that in the future could be used as proxies for swept area measurements (Eigaard et al. 2011). Modern industrialised fisheries such as the Swedish Baltic cod fishery are typically equipped with sensors to measure the trawl opening which could provide an accurate approximation of the effective width, while in other cases rough approximations can be made by fishermen or the LCA practitioner based on width of trawl opening, length of trawl boards, wire length and angles (Prat et al. 2008), Fig. 3.



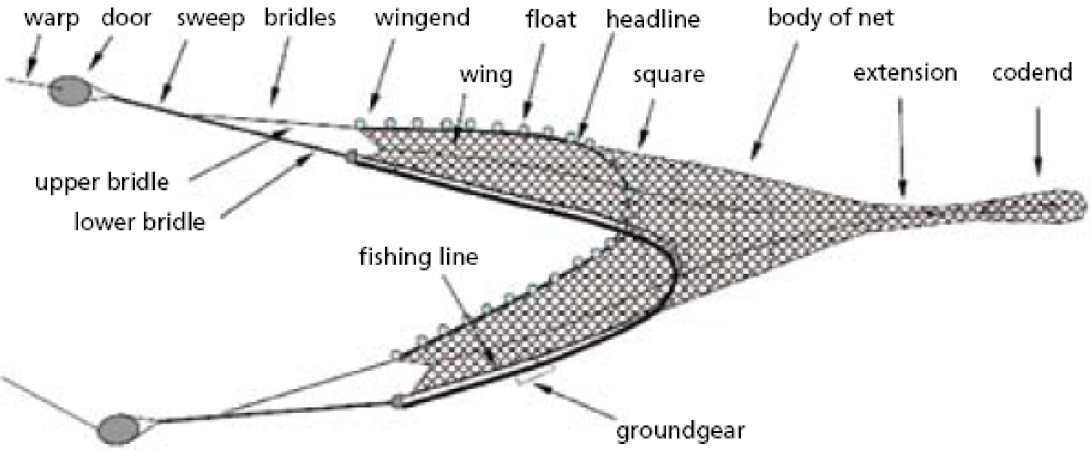


Figure 3. Technical terminology for trawls
For more details on definitions of fishing gear, see also the FAO technical guide:
ftp://ftp.fao.org/docrep/fao/010/a1466e/a1466e02.pdf

Drawbacks of this measure include the fact that an area is swept by a trawl says little about the impact this disturbance causes, trawling a sandy area at a certain frequency will e.g. have an entirely different impact compared to trawling a coral reef. The data required to estimate the swept area per kilo landed are fishing effort (hours trawled), effective gear width in seafloor contact, trawling speed and landings per unit of effort (LPUE in kg/h) and these are combined in a way shown in section 3.

An additional problem is that there are almost as many trawl designs as there are trawlers and as already mentioned, standardisation is almost non-existing. Although fisheries are highly regulated, it is more the outcome, i.e. the catches, that are regulated than how the gear should be designed or operated (except for certain technical regulations on mesh sizes and selectivity devices) Since there are no limitations for how wide a trawl can be, it is very difficult to determine typical trawl widths (Thörn 2012).

B) Pressure indicators

Several authors have suggested indicators to evaluate trawl impacts, e.g. Hiddink et al. (2006) and most recently Piet & Hintzen (2012), primarily for use in fisheries and marine ecosystem management, but also to follow up the EU Marine Strategy Framework Directive (MSFD). They advocate pressure indicators as opposed to state indicators.

By pressure indicators they mean using a measure that indirectly indicates the level of impacts, more specifically the spatial distribution and aggregation of fishing effort (from VMS) on a defined scale. This requires data treatment to obtain an average fishing pressure from VMS signals. The choice of spatial scale will influence the outcome (Lambert et al. 2012, Piet and Quirijns 2009). Pressure indicators could be e.g. proportion of area impacted by fishing or proportion not impacted (suggested indicator of the MSFD).

By state indicators the authors mean measures of the actual biological impact in terms of reductions of biomass, production, species or phyla or biodiversity indices. It is very resource intensive to quantify these impacts and the evaluation through these indicators is too slow to evaluate changes in the management system. The main drawback of pressure indicators is that they still not take into account the intensity of the disturbance or which communities and habitats are actually hit by the impact, although Piet and Hintzen (2012) suggest defining an “intensity threshold” depending on the benthic recovery capacity. In addition, limiting the area of study becomes crucial when determining proportions affected and not affected. For example, if a fishery takes place in a limited part of the Mediterranean and the total area impacted by the fishery is calculated from VMS data, should the indicator then, when transformed into a proportion, relate to the entire area of the Mediterranean or only a part of it?

C) Gear differentiation

As mentioned above, a square meter of swept area, does not cause the same impact in different habitats and biological communities. Moreover, even in the same habitat, different gear types will lead to different impacts depending on the technical properties of the gear, often in relation to what type of target species they are designed to catch. Shrimp and otter trawls for example have rather light seafloor contact as they catch species that live close to the seafloor, but not on it. Norway lobster (*Nephrops*) trawls and dredges targeting scallops or other types of mussels, however, target species that live in the seafloor and are designed to dig up the catch. Beam trawls, often used in flatfish fisheries in the North Sea, have tickler chains intended to scare up the flatfish from the seafloor into the trawl. Other inventions include the so called rock-hopper-trawl using rubber bobbins to overcome minor boulders or rocks and making it possible to trawl previously inaccessible areas for trawlers. Recent developments also include demersal fish trawling with pelagic trawl doors, i.e. trawl doors that are not in contact with the seafloor. This change potentially reduces the seafloor impact of such a trawl substantially, since the trawl doors are the part of the trawl often causing most direct impact. Differentiating the swept area measure with regard to these gear characteristics would improve the measure as an indicator of impact considerably.

The “digging” of trawls could in theory increase the cycling of both nutrients and toxic compounds buried in sediment layers by the mechanical turbation caused by this type of trawling and dredging, which could be an interesting feedback loop to the impact categories eutrophication and aquatic toxicology.

D) Risk assessment

Although all of the marine coverage of trawling can be hard to relate quantitatively, some special structures are more sensitive to the passing of a trawl or dredge such as reef structures and corals compared with a sandy habitat (Auster 1998), although the exact quantitative impact pathways are hard to determine. A recent example are deep sea mounts that represent new fishing grounds for demersal gear with potentially large consequences for communities we don't know much about except that their growth rates are much lower than those of communities on the continental shelf (Althaus et al. 2009).

Since detailed maps are only available for few marine areas (and are sometimes classified for military use), and trawling is quite imprecise - a potential risk factor r could be established from the probability p and the potential loss L, which is the fundamental part of a risk assessment. Environmental risk assessment is a part of industrial ecology that traditionally has been separated from LCA, and might be better suited for analysing more site specific consequences, while the goal of LCA impact assessment methods is typically global coverage. Some aspects of modern impact assessment are already approaching risk assessment especially in terms of toxicity endpoint assessment and risk assessment has also been suggested to become more integrated into LCA (Owens 1997). One example where risk assessment has been used to examine the risk of not achieving management goals, such as following an ecosystem based management, are Australian off shore fisheries (Hobday et al. 2011).

Other aspects that in theory could be aggregated with a risk based characterization could be the distance to cable, gas pipes, ship wrecks or other hazardous objects. Other by society defined valuable areas, such as European Natura 2000 sites[7] established under the Habitats Directive, could also provide potential geographical input data sets for risk assessment.

Although practically useful in management and many specific cases, a characterization based on risk would require large amount of data and typically also involve subjective choices of an expert panel and would be hard to execute in a uniform way. Depending on how the elements of risk are defined, this might as well be a characterization of type E see below. An important property of this approach is the crucial need for probabilistic modelling.

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[7] http://ec.europa.eu/environment/nature/natura2000/index_en.htm

E) Habitat sensitivity, defined by sediment type

As mentioned previously, one variable that would be desirable to incorporate into a quantitative measure of seafloor impacts is the type of habitat impacted, defined by the type of sediment and related to how common that particular habitat is, since it is clear that the impact of trawling at a certain intensity and with a certain type of gear, will be very different in different habitats due to different sensitivity. Sediment data are more easily available than data on the distribution of marine communities and as mentioned previously, several authors suggest a mechanism where high natural variability makes sandy sediment types less sensitive than more stable habitats such as muds and coral reefs (NAS 2004). The sensitivity of different habitats and recovery times from fishing impacts are parameters quantified and used. However, the review and meta-analyses done to date conclude that there is a heavy bias in the studies performed and that some combinations of gear and habitats are overrepresented (otter trawling in sandy areas), while others are underrepresented (otter trawling in muddy sand) or completely lacking (scallop dredging in muddy areas, Table 1). In some cases, this is due to the target species, beam trawling e.g. is rarely done in muddy areas because plaice, witch and other target species live in more sandy areas. The unbalanced sample, however, makes it difficult to analyse studies statistically. It also seems clear that both gear and habitat matter and that there are interactions between these two variables. Collie et al. (2000) for example found that otter trawling caused the most long-term impacts in sandy and biogenic (i.e. reef-building organisms) habitats while scallop dredging caused the most serious impacts in gravel habitats. Beam trawling, often pointed at as one of the most destructive fishing methods, had a surprisingly short recovery time both in sands and gravels. When disregarding the gear type however, muds and sands on average recovered more quickly than gravel and biogenic habitats (Collie et al.2000). Hiddink et al. also conclude a high sensitivity for fishing impacts of gravel habitats and much lower for sandy and muddy sediments. The fact that muddy sediments turn out as fairly resistant to fishing impacts and gravels as sensitive shows that the "particle size" relationship is too simplistic. Kaiser et al. (2006) also identify interactions between fishing pressure, habitats and gear types and the most severe impacts occurred after scallop dredging in biogenic habitats.

Table 1. Number of studies looking at impacts of each combination of gear type and habitats. From Kaiser et al. 2006

Habitat type: **Fishing gear:**	**Sand**	**Muddy sand**	**Mud**	**Gravel**	**Biogenic**
Scallop dredge	8	4	0	5	1
Otter trawl	13	1	8	1	3
Beam trawl	3	2	0	2	0

F) Community sensitivity

The physical habitat, although indicating what type of organisms can live in it, can be inhabited by more or less sensitive species and communities, and the quantitative response of a biological community to a certain level of fishing pressure represents the highest possible resolution and the biologically most relevant level. In LCA terms, such a measure would be close to an endpoint. However (for the reasons mentioned in B), these studies are very resource-demanding to perform and therefore only sparse data exist from experimental studies performed in various regions of the world. This type of data will not in the near (or far) future be sufficiently available to allow basing a quantitative impact function on it. The optimal measure would be a relation of the current to the pristine state, which is complicated by the fact that most of the worlds fishing grounds are affected by trawling and have been so for many years and it is therefore not trivial to define a pristine state. The measure could be in the form of a recovery time to the pristine state or a recovery of biomass or production as indicated by Hiddink et al. (2006).

An online database published by the Marine Life Network (MarLIN) could potentially be used for this purpose and is promising as it could apply various maritime activities (different types of fishing and construction) and included detailed data on the sensitivity and recovery of marine communities typical around the British Isles, which are common throughout most of the North Atlantic and have been used previously in the assessment of seafloor impacts of fishing (Nilsson and Ziegler 2007). For several reasons, this idea was not pursued further, though, e.g. the fact that the online database clearly states that the quantified values are not recommended for further use in other applications (as do most of the authors of reviews of seafloor impact studies). In relation to the fish case study of the LC-IMPACT project, a method building on North Atlantic marine communities or habitats would not have been fully applicable. This is due to the fact that the four studied fisheries all took place in the Baltic Sea, a brackish sea with highly unique environmental characteristics. Therefore both seafloor by-catch impacts differ considerably from those in more truly marine areas.

7.3. Best practice to assess seafloor impact in LCA

Our recommendation for best practise in incorporating seafloor disturbance in LCA , based on current knowledge in this field, is that: 1) seafloor impacts should be assessed in any seafood LCA involving towed gear (bottom trawls and dredges) 2) a stepwise procedure for how to do this is suggested and 3) every type of assessment will start with and estimation of the seafloor area swept per kilo landed.

The approach taken to assess seafloor impacts in the case study of Baltic cod and herring fisheries (Emanuelsson et al. 2013) was a development of the methodology to estimate the swept seafloor area. It was performed by using high resolution VMS data on fishing effort to estimate trawl speed. A survey was performed among fishermen and trawl designers regarding trawl dimensions to find representative data on both the theoretical and effective width of trawls in the fisheries studied.

In addition to this we encourage further cross cutting methodological development in and outside the LCA community to increase the knowledge for spatial seafloor planning accordingly to an ecosystem based management of fisheries.

A step-wise description of these procedure that we recommend seafood LCA practitioners to take based on our review follows.

7.4. Stepwise procedure for best practise assessment of seafloor impacts and guide for future methodological development

The first thing to determine for a seafood LCA practitioner thinking about whether and how to calculate seafloor impact is whether there are any fisheries involved in the LCA to be performed that actually cause seafloor impact, i.e. is involve the use of either demersal otter trawls, beam trawls or scallop dredges (there are also fishing gear for raking mussels in shallow waters etc. but we believe that they do not represent landings that typically will be subject of seafood LCA calculations). If not, with present knowledge, the seafloor impact is zero. For descriptions of fishing methods, see:

ftp://ftp.fao.org/docrep/fao/010/a1466e/a1466e02.pdf

A relatively simple calculation of the swept area per kilo landed will be sufficient in two cases, if the focus is to get the comparison right rather than on the absolute numbers: When only one of the systems to be compared involves either of the three fishing methods mentioned (if e.g. gillnet fishing and otter trawling for cod are the systems to be compared) or if all systems to be compared involve the same fishing methods and are used in the same seafloor habitats (e.g. different scenarios for beam trawling for plaice in a limited part of North Sea with sandy bottoms). Adding resolution of differences due to gear types and/or habitat in such cases would not change the comparison, only the absolute values for seafloor impact.

STEP 1: Differentiating which type of gear

The first step of the stepwise qualitative or quantitative assessment is therefore to determine which type of fishing gear is being used. FAO classifies all types of fishing gear into 11 categories where essentially bottom trawls and dredges are the ones to consider. However, surrounding nets such as purse seines and some other gear types that are not typically in seafloor contact, can cause seafloor impacts when used under certain conditions, e.g. very close to the coast.

STEP 2: Calculating the swept area

The swept area of the three towed gear types (otter trawls, beam trawls and dredges) is calculated by finding out the width of the demersal trawl gear in use (i.e. the effective width of the trawl or dredge that has seafloor contact including chains and otter boards when applicable, Fig. 3). It is hard to generalise as there is little standardisation of trawl dimensions and regulations (except for mesh size and some mandatory selectivity devices) only apply to their resulting selectivity performance.

Therefore, it is important to find out the trawl width in each case. In addition to the trawl width (W), the average speed (S) during trawling (in knots) is needed. These two variables together with the average landing per unit of effort (LPUE) in kilos/hour trawled will provide the swept area (SA) per kilo landed according to:

$SA = (W*S*1852)/LPUE \ (m^2/kg)$

One knot represents a speed of 1 nautical mile/h which is 1852 m/h, so the constant 1852 is to translate the distance trawled during one hour from nautical miles into meters.

If however, different types of seafloor habitats or bottom types are impacted, then further assessment is needed to illustrate that different habitats have different sensitivity and ability to recover from fishing disturbance.

STEP 3: Including the type of habitat impacted

The next step in that case is to find out which types of seafloor are impacted by the fishery and in what proportions. As no detailed global mapping of the seafloor at that resolution exists, this will depend on the availability of regional benthic maps often from geological surveys. A suggested differentiation based on current literature would be to determine the proportions of Muds, Muddy sands, Sands, Gravel and Biogenic habitats in the area impacted by the fishery. Other aspects could be areas of high preservatory value such as Natura 2000 habitats. At present, we recommend to stop here if different habitats and gear types are involved, but seafloor impact is not the very focus of a study.

STEP 4: Including fishing intensity

Additional factors that determine fishing impacts on benthic ecosystems (in addition to gear and habitat) are the frequency with which a habitat is impacted in relation to its sensitivity and recovery rate and the proportion of a habitat that is impacted by fishing at least once. It is important whether a habitat is trawled more frequently than its recovery time or not. If it is, fishing will keep it in a permanently altered condition. The distribution of fishing effort can be found from logbooks on a crude level (gear set positions) , but also at much higher resolution from vessel monitoring systems (VMS) which are today mandatory on vessels from 12m and up within the European Union. These data show the position of a vessel every hour and are stored by the authority managing each countries fishery and should be available for research projects. At present this represents the highest possible level of analysis of seafloor impacts at large-scale and it is motivated when seafloor impacts are a central part of a study involving different fishing practices and habitat types impacted by them.

STEP 5: Including the sensitivity of the biological community

Since several different types of biological communities can inhabit e.g. sandy seafloors with highly different sensitivity to fishing disturbance, the most relevant, but also the most data intensive, way of quantifying the seafloor impact would be to take into account the actual communities impacted based on its sensitivity, the area impacted, the frequency of fishing impacts and the proportion of the total distribution of an community in a particular area. Although there is information on the sensitivity of marine species and communities to fishing disturbance from certain regions, at this point there is too much information missing on the spatial distribution of different communities and their sensitivity to be able to build a large-scale seafloor impact model on it, but the approach using the online MarLIN database could be worth exploring further.

Due to the lack of standardisation of trawl design and operation, the lack of quantitative studies on fishing impacts on benthic ecosystems and a conclusive outcome of those that have been done, it is at this point still very difficult to develop a quantitative method for seafloor impact assessment. In terms of the case study, the use of VMS data with a reported position every hour represents an important improvement in terms of accuracy compared to the use of logbook data with one single position per haul. Nevertheless, for the general seafood LCA practitioner, the issue of seafloor impacts is a challenge, as neither VMS nor logbook data for entire fisheries are readily available. If only a small number of boats are involved, the fishing effort, landings, gear dimensions and fishing areas should be more readily available directly from the boats. However, this reflects maybe one important conclusion of this work, which is the question of whether this type of impacts is suitable for global generalisation or rather should be looked at and quantified locally.

Considering the current lack of data to establish a robust model of seafloor impacts of fishing based on fishing intensity, gear type, target species and habitat impacted, to compare it with biodiversity loss on land due to other activities (such as agriculture, forestry or mining) is even further away. Marine ecosystems differ from terrestrial ecosystems in many ways, one of them being that many species are dispersed either actively (migration) or passively by currents to a much larger extent than on land. This leads to completely different patterns of biodiversity impacts of activities on land compared to in the sea. A large proportion of the animals living in the seafloor are killed by each passing trawl directly and another proportion is killed due to being exposed to predators. The species that migrate least and grow slowest (such as e.g. the bivalve *Arctica islandica* and burrowing corals such as *Virgularia mirabilis*) will be more severely hit by trawl impact and take longer to recover than other, more mobile and fast-growing species, such as some species of polychaetes. Scavenging species, such as echinoderms (sea stars), whelks and other polychaetes can actually benefit from the increased food availability in trawl tracks. As some species will increase and others decrease, a measure measuring how many species have disappeared in an area, which is common in the assessment of landbased biodiversity impacts, is not useful in the marine context and in addition the problem of limitation of the area arises again (as with pressure indicators).

The stepwise procedure suggested in this study helps the seafood LCA practitioner to decide whether and how to assess seafloor impacts and we believe it takes seafloor impact assessment further than the current state of the art, without actually representing a robust LCIA methodology for seafloor impacts.

7.5. Conclusions

- Seafloor impacts of fishing are relevant to assess when towed fishing gear are used, both in seafood LCAs, in fisheries management and in seafood certification.
- When performing seafood LCAs, it should be assessed whether it is relevant to describe seafloor impacts or not.
- At present, the knowledge about quantitative fishing impacts of seafloor habitats does not easily translate into a general vulnerability function
- Biased data samples, data(un)availability, and inconclusive review studies represent additional challenges for establishing a damage function
- Several approaches from an early midpoint and towards an endpoint level are presented, but a clear trade- off situation exists between current applicability and biological relevance.
- A refined way to calculate the swept seafloor area by taking into account different impacts of different gear types and using VMS data is suggested.
- When swept seafloor area is used as an early midpoint category, fishing effort (hours trawled) and average landing per effort represents inventory data together with the trawling speed and effective trawl width.

- Perhaps the most ecosystem-relevant indicator is based on a few types of fishing gear (otter trawl, beam trawl and dredges) and broad habitat types (mud, sand, muddy sand, gravel and biogenic habitats).
- Practical guidelines for best practise in seafood LCA are provided.

7.6. References

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8. Annex

8.1. Annex to Chapter 1

Table 8.1.1: Biodiversity damage potential (BDP) characterization factors (median) of occupation per land use type, 1. and 3. quartile and number of data points (n). For n<5, no characterization factors are provided.

		1.1.1./4.1.1 Forest/Grassland, not used	1.1.2. Secondary Vegetation	1.2. Forest, used	4.2. Pasture/meadow	5.1 Annual crops	5.2. Permanent crops	6. Agriculture, mosaic (Agroforestry)	7. Artificial areas
Characterization factors, global									
Total world average	**Median**	**0**	**0.18**	**0.18**	**0.33**	**0.60**	**0.42**	**0.2**	**0.44**
	1. quartile	0	-0.03	-0.05	0.00	0.31	0.06	0.01	-0.01
	3. quartile	0	0.37	0.50	0.55	0.79	0.70	0.48	0.62
	n	326	272	148	133	96	52	76	53
Characterization factors, per biome									
Biome 1 (Sub-)tropical moist broadleaf forest	**Median**	**0**	**0.22**	**0.13**	**0.45**	**0.54**	**0.42**	**0.18**	-
	1. quartile	0	0.00	-0.09	0.31	0.36	0.18	-0.02	-
	3. quartile	0	0.43	0.45	0.75	0.72	0.70	0.44	-
	n	173	172	79	26	46	40	70	1
Biome 2 (Sub-)tropical dry broadleaf forest	**Median**	**0**	**0.17**	**0.58**	**0.48**	-	-	-	-
	1. quartile	0	-0.04	0.34	0.17	-	-	-	-
	3. quartile	0	0.19	0.68	0.69	-	-	-	-
	n	8	5	8	8	3	0	1	2
Biome 4 Temperate broadleaf forest	**Median**	**0**	**0.08**	**0.22**	**0.52**	**0.76**	**0.02**	-	**0.40**
	1. quartile	0	-0.26	-0.09	-0.35	0.46	-0.11	-	-0.10
	3. quartile	0	0.33	0.43	0.67	0.86	0.69	-	0.58
	n	46	20	35	33	24	9	0	24
Biome 5 Temperate coniferous forest	**Median**	**0**	**0.17**	**0.15**	**0.24**	**0.54**	-	-	**0.50**
	1. quartile	0	-0.22	0.02	-0.64	-0.15	-	-	-0.05
	3. quartile	0	0.30	0.33	0.38	0.87	-	-	0.71
	n	45	15	7	27	8	3	0	21
Biome 7 (Sub-)tropical grassland & savannah	**Median**	**0**	**0.00**	**0.01**	**0.12**	**0.65**	-	-	-
	1. quartile	0	-0.17	0.00	0.02	0.02	-	-	-
	3. quartile	0	0.15	0.06	0.27	0.80	-	-	-
	n	21	27	6	8	9	0	0	0
Biome 8 Temperate grassland & savannah	**Median**	**0**	-	-	**0.23**	-	-	-	-
	1. quartile	0	-	-	0.07	-	-	-	-

	3. quartile	0	-	-	0.39	-	-	-	-
	n	7	4	2	10	1	0	0	0
Biome 10 Montane grassland & shrublands	**Median**	**0**	**0.21**	**0.55**	**0.33**	**-**	**-**	**0.39**	**-**
	1. quartile	0	0.10	0.34	0.13	-	-	0.29	-
	3. quartile	0	0.38	0.71	0.37	-	-	0.57	-
	n	13	25	11	5	2	0	5	0

Table 8.1.1 (continued)

		1.1.1./4.1.1 Forest/Grassland, not used	1.1.2. Secondary Vegetation	1.2. Forest, used	4.2. Pasture/meadow	5.1 Annual crops	5.2. Permanent crops	6. Agriculture, mosaic (Agroforestry)	7. Artificial areas
Characterization factors, per biome (continued)									
Biome 12 Mediterranean forests & shrublands	**Median**	**0**	-	-	**0.24**	-	-	-	-
	1. quartile	0	-	-	0.11	-	-	-	-
	3. quartile	0	-	-	0.42	-	-	-	-
	n	8	3	0	11	3	0	0	4
Biome 13 Deserts & xeric shrublands	**Median**	**0**	-	-	**-0.08**	-	-	-	-
	1. quartile	0	-	-	-0.50	-	-	-	-
	3. quartile	0	-	-	0.17	-	-	-	-
	n	5	1	0	5	0	0	0	1
Characterization factors, per taxonomic group									
Arthropods	**Median**	**0**	**0.16**	**0.11**	**0.25**	**0.65**	**0.56**	**0.04**	-
	1. quartile	0	0.00	-0.15	0.13	0.35	0.23	-0.41	-
	3. quartile	0	0.39	0.32	0.45	0.77	0.62	0.29	-
	n	68	64	34	21	20	7	20	4
Other invertebrates	**Median**	**0**	**0.24**	**0.41**	**0.49**	**0.79**	**0.44**	-	**0.49**
	1. quartile	0	0.12	0.22	0.33	0.59	0.18	-	0.22
	3. quartile	0	0.53	0.79	0.58	0.85	0.69	-	0.71
	n	33	16	14	21	14	7	1	15
All vertebrates	**Median**	**0**	**0.09**	**0.12**	**0.31**	**0.50**	**0.39**	**0.11**	-
	1. quartile	0	-0.06	-0.03	0.04	0.20	0.28	-0.14	-
	3. quartile	0	0.31	0.47	0.5	0.70	0.70	0.24	-
	n	99	75	42	25	23	19	19	4
Birds	**Median**	**0**	**0.07**	**0.00**	**0.20**	**0.53**	**0.62**	**0.22**	-
	1. quartile	0	-0.06	-0.08	-0.08	0.21	0.38	0.07	-
	3. quartile	0	0.26	0.35	0.47	0.70	0.73	0.32	-
	n	53	39	17	14	17	12	11	3
Other vertebrates	**Median**	**0**	**0.15**	**0.14**	**0.33**	**0.45**	**0.27**	**-0.14**	-
	1. quartile	0	-0.06	0.00	0.16	0.20	-0.01	-0.37	-
	3. quartile	0	0.39	0.56	0.50	0.68	0.38	0.05	-
	n	46	36	25	11	6	7	8	1

Table 8.1.1 (continued)

		1.1.1./4.1.1 Forest/Grassland, not used	1.1.2. Secondary Vegetation	1.2. Forest, used	4.2. Pasture/meadow	5.1 Annual crops	5.2. Permanent crops	6. Agriculture, mosaic (Agroforestry)	7. Artificial areas
Characterization factors, per taxonomic group (continued)									
All plants	**Median**	**0**	**0.21**	**0.23**	**0.29**	**0.56**	**0.38**	**0.37**	**0.33**
	1. quartile	0	0.00	0.00	-0.20	0.30	0.01	0.15	-0.42
	3. quartile	0	0.43	0.53	0.57	0.81	0.79	0.63	0.59
	n	126	117	58	66	39	19	36	30
Vascular plants	**Median**	**0**	**0.21**	**0.27**	**0.11**	**0.42**	**0.28**	**0.37**	**-0.42**
	1. quartile	0	0.02	0.04	-0.37	0.17	-0.04	0.15	-0.79
	3. quartile	0	0.45	0.57	0.40	0.61	0.63	0.63	0.15
	n	97	109	44	48	28	15	32	15
Moss	**Median**	**0**	**0.03**	**0.05**	**0.65**	**0.87**	-	-	**0.58**
	1. quartile	0	-0.19	-0.16	0.29	0.73	-	-	0.41
	3. quartile	0	0.24	0.28	0.75	0.9	-	-	0.68
	n	29	8	14	18	11	4	4	15
Characterization factors, per data source									
GLOBIO data	**Median**	**0**	**0.19**	**0.17**	**0.33**	**0.57**	**0.42**	**0.20**	**0.34**
	1. quartile	0	0.00	0.00	0.12	0.35	0.18	0.01	0.12
	3. quartile	0	0.38	0.50	0.49	0.74	0.70	0.48	0.60
	n	254	248	121	79	72	40	76	8
Swiss data (BDM)	**Median**	**0**	**0.07**	**0.22**	**0.32**	**0.81**	**0.43**	-	**0.45**
	1. quartile	0	-0.19	-0.17	-0.38	0.26	-0.08	-	-0.05
	3. quartile	0	0.31	0.48	0.62	0.90	0.77	-	0.62
	n	72	24	27	54	24	12	0	45

Table 8.1.2: Results of 2-sided Mann-Whitney U test testing the difference of median Srel of all combinations of land use types (full dataset). *** p-values<0.001; ** p-values<0.01; * p-values<0.05; (*) p-values<0.1; ns p-values > 0.1

Land use type	Reference	Secondary vegetation	Used forest	Pasture	Annual crop	Permanent crop	Agroforestry
Secondary vegetation	***						
Used forest	***	ns					
Pasture	***	**	ns				
Annual crops	***	***	***	***			
Permanent crops	***	***	**	*	(*)		
Agroforestry	***	ns	ns	ns	***	**	
Artificial area	***	*	ns	ns	**	ns	ns

Table 8.1.3: Peason's correlation coefficients r between indicators for a subset of data from biome (sub-) tropical moist broadleaf forest

	Srel	MSA	Sørensen's Ss	Shannon's H (rel)	Fisher's α (rel)
Srel	1	0.41	0.17	0.79	0.83
MSA		1	0.81	0.19	0.06
Sørensen's Ss			1	0.03	0.20
Shannon's H (rel)				1	0.87
Fisher's α (rel)					1

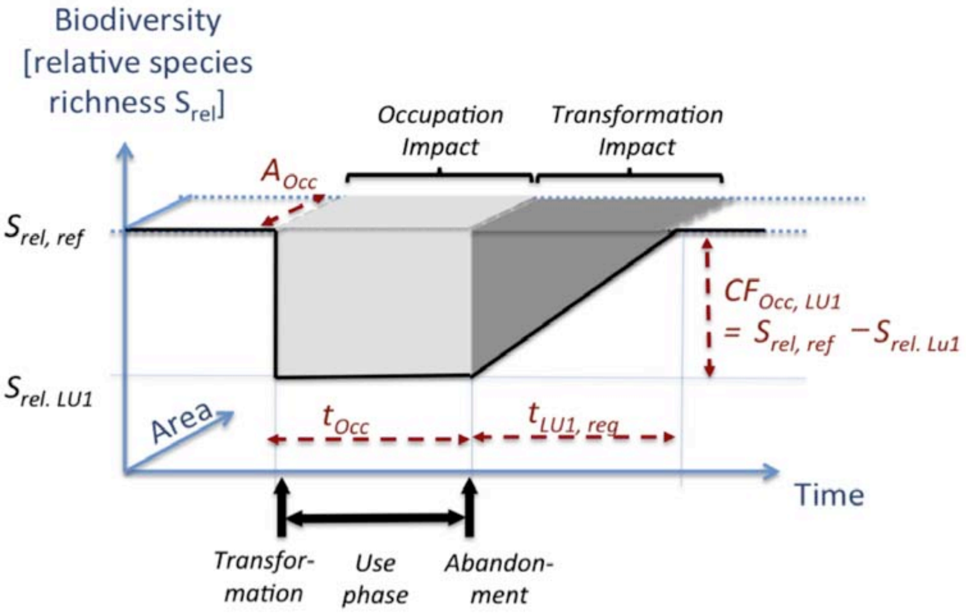

Figure 8.1.1: UNEP/SETAC framework for calculating land use occupation and transformation impacts (adapted from Milà i Canals et al., 2007 and Koellner et al., 2012-b).

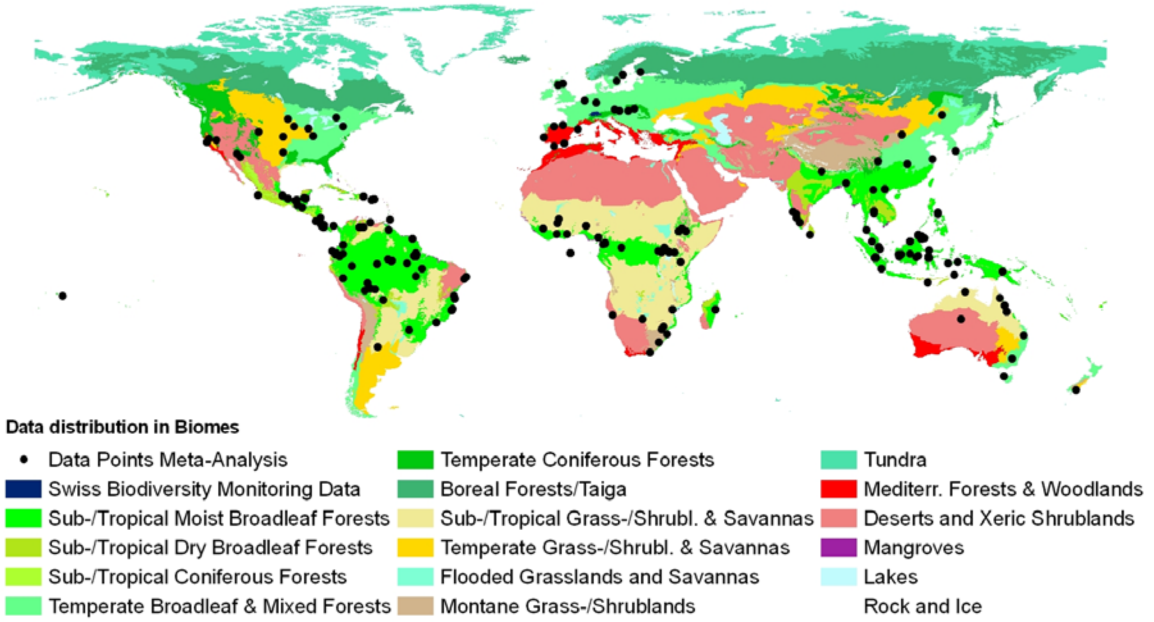

Figure 8.1.2: Map of geographic location of studies included in this study (indicated as black dots). The colors indicate different WWF biomes

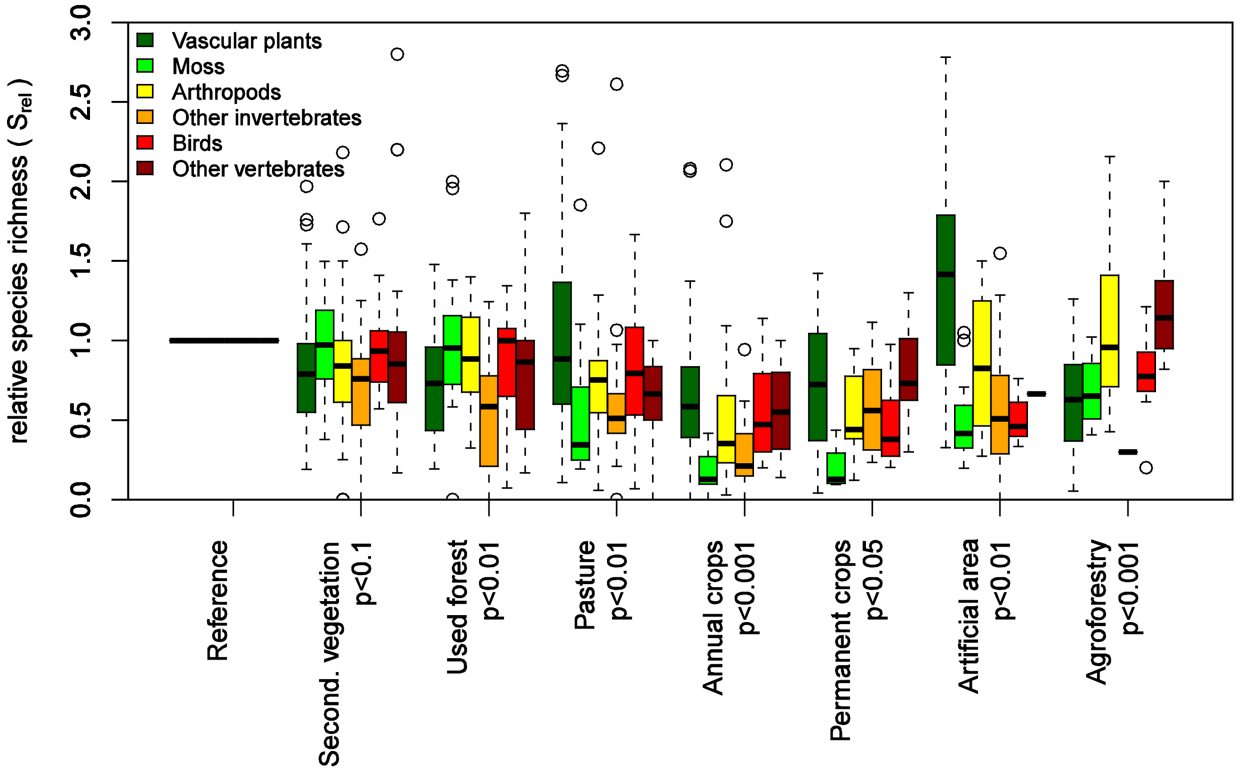

Figure 8.1.3: Box and whisker plot of relative species richness per land use type and taxonomic group and test statistics of Kruskal-Wallis test (Srel = f(LU x taxa)) for full dataset. In contrast to the results presented in Fig 3 (main article), the taxonomic groups 'plants' were split into 'vascular plants' and 'moss' and 'vertebrates' were split into 'birds' and 'other vertebrates'(see also Table 5 for an overview of the underlying data sources).

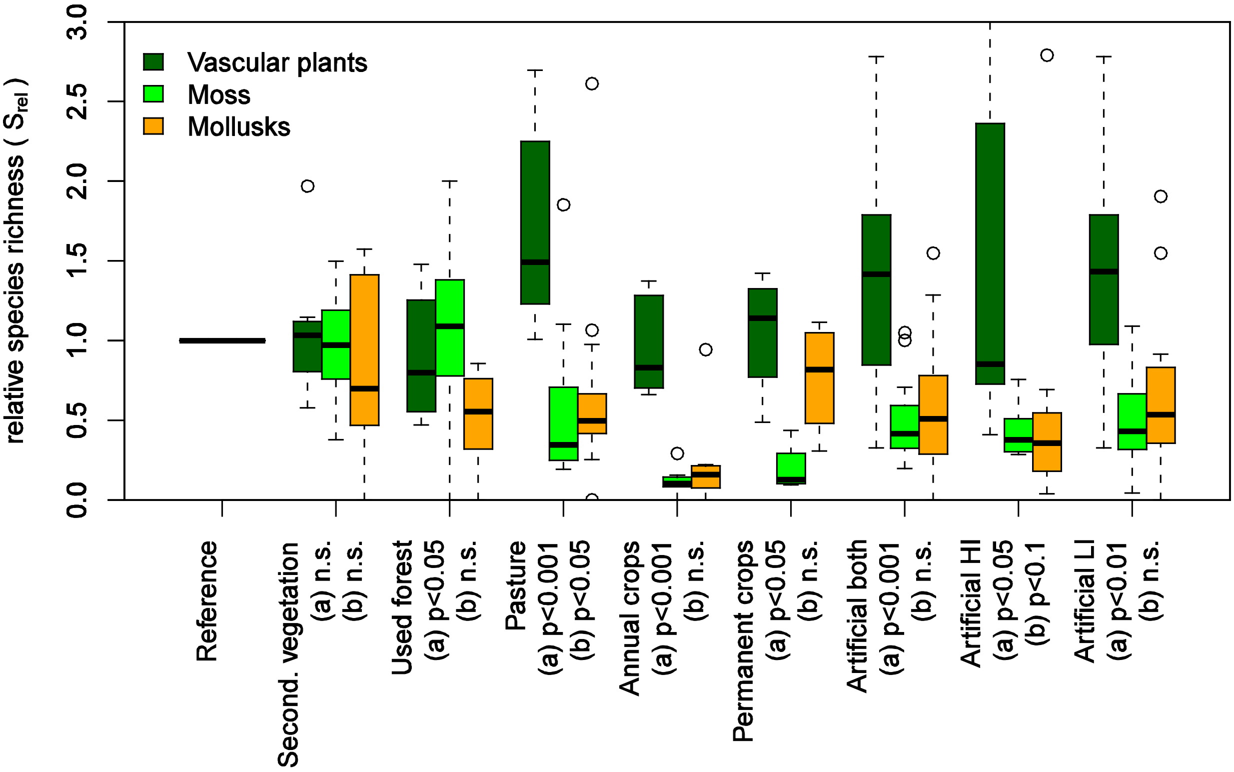

Figure 8.1.4: Box and whisker plot for the Swiss BDM data of relative species richness per land use type and taxonomic group. (a) test statistics of Kruskal-Wallis test Srel = f(LU x taxa); (b) Comparison of the LU effect across the two biomes temperate broadleaf forest and temperate coniferous forest (test statistics of Kruskal-Wallis test Srel = f(LU x biome)). Results for all artificial area ('Artificial both') were split into high (HI: land use types '8.1.2 Urban, continuously built'; '8.3 Industrial area'; '7.4 Dump site') and low intensity (LI: land use types '8.1.3 Urban, discontinuously built'; '7.6 Traffic area'; '8.1.4 Urban, green areas'). There was no significant difference between Artificial HI and Artificial LI for the overall data nor for any taxonomic group (Mann-Whitney U test, not shown). ns not significant.

Table 8.1.4: Representation of species groups in this study compared to global estimates of terrestrial species richness

	Data representation of this study	Estimated species numbers (Heywood and Watson 1995)
Invertebrates	31%	70%
Arthropodes	20%	65%
Insects	19%	59%
Coleoptera	6%	
Hymenoptera	6%	
Lepidoptera	5%	
Other insects	2%	
Other arthropods	2%	7%
Other Invertebrates	11%	
Mollusks	8%	1%
Other invertebrates	3%	3%
Vertebrates	26%	0.4%
Birds	14%	
Other vertebrates	12%	
Amphibians	4%	
Mammals	8%	
Reptiles	0%	
Plant	43%	2%
Vascular plants	34%	
Herbaceous plants	13%	
Trees	13%	
Mosses	9%	
Virus		3%
Bacteria		7%
Fungi		11%
“Protozoa”		1%
“Algea”		3%
Others		2%
Total	100%	100%

Table 8.1.5: Number of data points included in this study per species group, land use type and dataset (GLOBIO and BDM).

	Reference		Secondary vegetation		Used forest		Pasture		Annual crops		Permanent crops		Agroforestry		Artificial area		tal		
	GLOBIO	BDM	GLOBIO	BDM	GLOBIO	BDM	GLOBIO	BDM	GLOBIO	BDM	GLOBIO	BDM	GLOBIO	BDM	GLOBIO	BDM	GLOBIO	BDM	Total
Invertebrates	77	24	72	8	39	9	24	20	26	8	10	4	21	0	4	16	273	89	285
Arthropodes	68	0	64	0	34	0	21	0	20	0	7	0	20	0	4	0	238	0	170
Insects	60	0	57	0	31	0	20	0	20	0	7	0	19	0	3	0	217	0	157
Coleoptera	16	0	13	0	15	0	10	0	3	0	1	0	7	0	0	0	65	0	49
Hymenoptera	20	0	22	0	6	0	4	0	11	0	5	0	7	0	0	0	75	0	55
Lepidoptera	17	0	19	0	7	0	0	0	4	0	1	0	4	0	2	0	54	0	37
Other insects	7	0	3	0	3	0	6	0	2	0	0	0	1	0	1	0	23	0	16
Other arthropods	8	0	7	0	3	0	1	0	0	0	0	0	1	0	1	0	21	0	13
Other Invertebrates	9	24	8	8	5	9	3	20	6	8	3	4	1	0	0	16	35	89	115
Mollusks	2	24	0	8	2	9	0	20	0	8	0	4	0	0	0	16	4	89	91
Other invertebrates	7	0	8	0	3	0	3	0	6	0	3	0	1	0	0	0	31	0	24
Vertebrates	99	0	75	0	42	0	25	0	23	0	19	0	19	0	4	0	306	0	207
Birds	53	0	39	0	17	0	14	0	17	0	12	0	11	0	3	0	166	0	113
Other vertebrates	46	0	36	0	25	0	11	0	6	0	7	0	8	0	1	0	140	0	94
Amphibians	17	0	12	0	9	0	7	0	1	0	3	0	1	0	0	0	50	0	33
Mammals	28	0	24	0	16	0	4	0	5	0	4	0	6	0	1	0	88	0	60
Reptiles	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	2	0	1
Plant	78	48	101	16	40	18	30	40	23	16	11	8	36	0	0	32	319	178	419
Vascular plants	73	24	101	8	35	9	30	20	20	8	11	4	32	0	0	16	302	89	318
Herbaceous plants	34	0	38	0	13	0	26	0	11	0	8	0	16	0	0	0	146	0	112
Trees	39	0	63	0	22	0	4	0	9	0	3	0	16	0	0	0	156	0	117
Mosses	5	24	0	8	5	9	0	20	3	8	0	4	4	0	0	16	17	89	101
Total	254	72	248	24	121	27	79	60	72	24	40	12	76	0	8	48	898	267	911

Table 8.1.6: Overview of land use classification of BDM (2004) data as applied in Koellner and Scholz (2008) and classification applied in this study (based on Koellner et al., 2012-a).

CORINE Plus classification (Koellner & Scholz, 2008)	CORINE Plus description (type of intensity)	Simplified classification used in this paper	UNEP/SETAC classification (Koellner et al, 2012-a)
1 Artificial surfaces			
10 Built up land			
11 Urban fabric			
111 Continuous urban fabric	Buildings cover most of the land. Roads and artificially surfaced area cover almost all the ground. Non-linear areas of vegetation and bare soil are exceptional. At least 80% of the total area is sealed. (artificial_hi)	7 Artificial area (High intensity)	7.1.2 Urban, continuously built
112 Discontinuous urban fabric	Most of the land is covered by structures. Buildings, roads and artificially surfaced areas associated with areas with vegetation and bare soil, which occupy discontinuous but significant surfaces. Less than 80% of the total area is sealed. (artificial_hi)	7 Artificial area (Low intensity)	7.1.3 Urban, discontinuously built
12 Industrial, commercial and transport			
121 Industrial or commercial units	Artificially surfaced areas (with concrete, asphalt, tamacadam, or stabilized, e.g., beaten earth) devoid of vegetation occupy most of the area in question, which also contains buildings and/or areas with vegetation. (artificial_hi)	7 Artificial area (High intensity)	7.2 Industrial area
122 Road and rail networks and associated land	Motorways, railways, including associated installations (stations, platforms, embankments). Minimum width to include: 100 m.	7 Artificial area (Low intensity)	7.6 Traffic area
132 Dump sites	Landfill or mine dump sites, industrial or public. (artificial_hi)	7 Artificial area (High intensity)	7.4 Dump site
14 Artificial, non-agricultural areas with vegetation			
141 Green urban areas	Areas with vegetation within urban fabric. Includes parks and cemeteries with vegetation. (artificial_li)	7 Artificial area (Low intensity)	7.1.4 Urban, green areas
142 Sport and leisure facilities	Camping grounds, sports grounds, leisure parks, golf courses, racecourses, etc. Includes formal parks not surrounded by urban zones. (artificial_li)	7 Artificial area (Low intensity)	7.1.4 Urban, green areas
2 Agricultural areas			
21 Arable land			
211 Non-irrigated arable land	Cereals, legumes, fodder crops, root crops and fallow land. Includes flower and tree (nurseries) cultivation and vegetables, whether open field, under plastic or glass (includes market gardening). Includes aromatic, medicinal and culinary plants. Excludes permanent pastures.	5.1 Annual crop	5.1.2 Arable, non-irrigated
211b Integrated	Chemical-synthetic and organic fertilizer as well as pesticides are applied. However, the input of these substances is reduced. 21121 Wheat, 21122 Maize (agri_hi)	5.1 Annual crop	5.1.2.2 Arable, non-irrigated, intensive
211e Agricultural fallow	Agricultural fallow. (non-use)	5.1 Annual crop	5.1.1 Arable, fallow
211f Artificial meadow	Artificial meadow in rotation system. (agri-hi)	4.2 Pasture/meadow	4.2 Pasture/meadow
22 Permanent crops			
221 Vineyards	Areas planted with vines.	5.2 Permanent crop	5.2 Permanent crops

CORINE Plus classification (Koellner & Scholz, 2008)	CORINE Plus description (type of intensity)	Simplified classification used in this paper	UNEP/SETAC classification (Koellner et al, 2012-a)
222 Fruit trees and berry plantations	Parcels planted with fruit trees or shrubs: single or mixed fruit species, fruit trees associated with permanently grassed surfaces. Includes chestnut and walnut groves.	5.2 Permanent crop	5.2 Permanent crops
222a Intensive orchards	Orchards with small growing fruit trees. (agri_hi)	5.2 Permanent crop	5.2.4 Permanent crops, intensive
222b Organic orchards	Orchards with meadows and large fruit trees. (agri_li)	5.2 Permanent crop	5.2.3 Permanent crops, extensive
23 Pastures and meadows			
231 Pastures and meadows	Dense, predominantly graminoid grass cover, of floral composition, not under a rotation system. Mainly used for grazing, but the fodder may be harvested mechanically. Includes areas with hedges (bocage), e.g., oat grass meadow (Arrhenatherion, Polygono), fertilized moist meadow (Calthion).	4.2 Pasture/meadow	4.2 Pasture/meadow
231a Intensive pasture and meadows	Meadows mechanically harvested 3 times or more per year, fertilizer applied, perhaps on former arable land. (agri_hi)	4.2 Pasture/meadow	4.2.2 Pasture/meadow, intensive
24 Heterogeneous agricultural areas			
244 Agroforestry areas	Annual crops or grazing land under the wooded cover of forestry species. (agri_li)	6 Agroforestry	6 Agriculture, mosaic
3 Forests and semi-natural areas			
31 Forests			
311 Broad-leafed forest	Vegetation formation composed principally of trees, including shrub and bush understories, where broad-leafed species predominate. (Presence of conifers 0-10%)	1.11 Forest, **Reference**	1 Forest
312 Coniferous forest	Vegetation formation composed principally of trees, including shrub and bush understories, where coniferous species predominate. (Presence of conifers 91-100%)	1.11 Forest, **Reference** (for montane and subalpine regions) 1.22 Used Forest (for colline regions)	1 Forest
312a Coniferous plantations	Plantations of fast growing tree species like Picea abies. (forest_hi)	1.22 Used Forest	1.2.2 Forest, intensive
312b Semi-natural coniferous forests	Natural or semi-natural forests, where coniferous species predominate. (forest_li)	1.11 Forest, **Reference**	1.1.1 Forest, primary
313 Mixed forest	Vegetation formation composed principally of trees, including shrub and bush understories, where broad-leafed and coniferous species co-dominate. (forest_li)	1.11 Forest, **Reference**	1 Forest
313b Mixed coniferous forest	Forests with presence of conifers 51-90%. (forest_li)	1.22 Used Forest	1.2.2 Forest, intensive
32 Shrub and/or herbaceous vegetation associations			
321 Semi-Natural grassland	Low productivity grassland. Often situated in areas of rough uneven ground. Frequently includes rocky areas, briars, and heath land., e.g., xeric grassland (Xero-Bromium), mesoxeric grassland (Meso-Bromium), mat-grass pasture (Violion), moor-grass meadow (Molinion). (agri_li)	4.11 Grassland, **Reference**	4.1.1 Grassland, natural
322 Moors and heath land	Vegetation with low and closed cover, dominated by bushes, shrubs and herbaceous plants (heath, briars, broom, gorse, laburnum, etc.). (non-use)	3 Shrub land, **Reference**	3 Shrub land

CORINE Plus classification (Koellner & Scholz, 2008)	CORINE Plus description (type of intensity)	Simplified classification used in this paper	UNEP/SETAC classification (Koellner et al, 2012-a)
324 Transitional woodland/shrub	Bushy or herbaceous vegetation with scattered trees. Can represent either woodland degradation or forest regeneration/colonization. (non-use)	1.12 Secondary forest	1.1.2 Forest, secondary
33 Open spaces with little or no vegetation			
331 Beaches, dunes, and sand plains	Beaches, dunes and expanses of sand or pebbles in coastal or continental areas, including beds of stream channels with torrential regime. (non-use)	8 Bare area, **Reference**	8 Bare area
332 Bare rock	Scree, cliffs, rocks and outcrops. (non-use)	8 Bare area, **Reference**	8 Bare area
333 Sparsely vegetated areas	Includes steppes, tundra and badlands. Scattered high-attitude vegetation. (non-use)	8 Bare area, **Reference**	8 Bare area
4 Wetlands			
41 Inland wetlands			
411 Inland marshes	Low-lying land usually flooded in winter, and more or less saturated by water all year round. (non-use)	2.2 Wetlands, **Reference**	2.2 Wetlands, inland
412 Peat bogs	Land consisting mainly of decomposed moss and vegetable matter. May or may not be exploited. e.g., raised bog (Sphagnetum), intermediate mires (e.g., Scheuchzerietum) and fens (Caricetum). (non-use)	2.2 Wetlands, **Reference**	2.2 Wetlands, inland
51 Inland waters			
511 Water courses	Natural or artificial watercourses serving as water drainage channels. Includes canals. Minimum width to include: 100 m.	10 Water bodies, **Reference**	10 Water bodies

References to Annex 8.1 (Land use impacts on biodiversity)

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Milà i Canals L, Bauer C, Depestele J, Dubreuil A, Freiermuth Knuchel R, Gaillard G, Michelsen O, Müller-Wenk R, Rydgren B. 2007. Key Elements in a Framework for Land Use Impact Assessment Within LCA. International Journal of Life Cycle Assessment 12(1):5-15.

8.2. Annex for Chapter 2

Table 8.2.1. Overview of input data, assumed distribution and variable specificity

Variables		Data source	**Assumed distribution**	**Variable specificity**
CF_{loc}	Local characterization factors	de Baan et al. (2013)	Non-parametric Kernel Density Estimation	Independent variables for each biome (n=14) and land use type (n=4)
z	z-values	Drakare et al. (2006)	Triangular distribution, based on mode, and lower and upper bounds	Independent variables for each habitat type (n=3; islands, forest, non-forest ecoregions).
$A_{org,j}$, $A_{new,j}$, $A_{i,j}$	Areas (original habitat, remaining habitat, area per land use type)	LADA (2008), Anthromes (Ellis and Ramankutty 2008) using 7 geographic projections for calculating areas	Non-parametric Kernel Density Estimation	$A_{org,j}$, $A_{new,j}$: Independent variables for each ecoregion (n=804) $A_{i,j}$: Independent variables for each land use type (n=4; see Table 8.2.4 for classification) and ecoregion (n=804)
S_{org}	Species numbers per ecoregion(*)	Olson et al. (2001) (mammals, birds, amphibians, reptiles), Kier et al. (2005) (plants)	Plants: triangular distribution based on working figure, upper and lower ranges. Other groups: no uncertainty assumed	Independent variables for each ecoregion (n=804) and taxa (n=5; mammals, birds, amphibians, reptiles, plants)
$S_{org, end}$	Endemism numbers per ecoregion(*)	Olson et al. (2001) (mammals, birds, amphibians, reptiles), plants: no data.	No uncertainty assumed	Independent variables for each ecoregion (n=804) and taxa (n=4; mammals, birds, amphibians, reptiles)
t_{reg}	Regeneration time	Curran et al. (subm.)	Non-parametric Kernel Density Estimation	520 different regeneration times, based on all combinations of Realm x Biome (n = 65), land use intensity (n=2), taxonomic group (n=4)

(*) Species richness numbers from WWF are based on the ranges of extant species. Species that are introduced, present as human commensals, vagrants, or passage migrants are not recorded. Where available, historic ranges of species (i.e., approximate distribution at 1500 AD) were used to compile the database (World Wildlife Fund 2006).

Table 8.2.2. z-values applied in this study (derived from Drakare et al. 2006), Fig. 1e, for both average and nested SAR)

		island	forest	non-forest
lower	Confidence	0.242	0.307	0.185
mean		**0.258**	**0.344**	**0.211**
upper	Confidence	0.282	0.384	0.247

Table 8.2.3. Median regeneration times in years (based Curran et al. subm.)

Region name			Input parameters			Median regeneration times (in years)							
Realm	Biome	Realm_Biome	median Latitude [° S or ° N]	Altitude (median) [m.a.s.l]	Ecosystem type	Birds Ext.	Int.	Mammals Ext.	Int.	Herpeto-fauna Ext.	Int.	Plants Ext.	Int.
AA	01	AA01	10	300	Forest	36	36	36	36	40	39	27	27
	02	AA02	10	400	Forest	37	37	37	36	41	40	28	27
	04	AA04	40	400	Forest	78	77	83	82	85	84	56	55
	07	AA07	20	300	Non-forest	12	12	13	13	14	14	7	7
	08	AA08	40	200	Non-forest	21	21	24	24	25	25	12	12
	10	AA10	40	1300	Non-forest	29	30	34	36	33	33	15	16
	11	AA11	60	200	Forest	128	127	138	137	143	144	86	84
	12	AA12	40	200	Forest	73	73	79	79	83	82	53	52
	13	AA13	30	400	Non-forest	16	16	18	19	19	19	9	9
	14	AA14	10	100	Forest	35	34	34	34	38	37	26	26
AN	11	AN11	65	1000	Forest	176	177	185	184	199	198	113	113
AT	01	AT01	10	500	Forest	35	35	35	35	39	39	27	26
	02	AT02	20	300	Forest	44	43	43	42	48	47	33	33
	07	AT07	10	600	Non-forest	9	9	10	10	10	10	5	5
	08	AT08	30	700	Non-forest	17	17	19	20	19	20	10	10
	09	AT09	20	500	Non-forest	11	11	13	13	14	14	7	7
	10	AT10	20	1600	Non-forest	16	16	18	18	19	19	9	8
	12	AT12	40	500	Forest	76	76	80	80	83	83	53	52
	13	AT13	30	1000	Non-forest	19	19	22	22	22	22	10	10
	14	AT14	10	100	Forest	33	33	33	32	36	36	25	25
IM	01	IM01	20	300	Forest	51	51	52	52	56	56	39	38
	02	IM02	20	300	Forest	51	51	52	52	56	56	39	38
	03	IM03	30	1400	Forest	83	82	85	85	92	91	59	58
	04	IM04	30	1800	Forest	88	87	92	91	98	98	62	62
	05	IM05	40	2000	Forest	120	120	126	124	134	132	83	83
	07	IM07	30	200	Non-forest	20	20	21	21	22	22	11	10
	09	IM09	30	100	Non-forest	19	20	20	20	22	22	10	10
	10	IM10	10	1000	Non-forest	13	13	14	14	16	16	7	7
	13	IM13	30	200	Non-forest	20	20	21	21	22	22	11	10
	14	IM14	20	100	Forest	49	49	50	50	54	53	38	37
NA	02	NA02	30	400	Forest	60	59	60	60	65	65	44	44
	03	NA03	30	2000	Forest	78	77	85	83	89	89	58	58
	04	NA04	50	300	Forest	98	97	103	103	109	110	66	65
	05	NA05	50	1100	Forest	119	118	124	124	130	130	80	80
	06	NA06	60	400	Forest	132	130	140	139	147	147	82	82
	07	NA07	30	100	Non-forest	14	14	15	15	16	16	8	8
	08	NA08	50	700	Non-forest	34	34	40	40	40	41	19	19
	11	NA11	65	300	Forest	143	143	159	159	166	166	91	91
	12	NA12	40	500	Forest	79	78	83	82	85	85	55	55
	13	NA13	40	1500	Non-forest	30	30	34	35	34	34	16	16
	14	NA14	30	100	Forest	56	56	58	57	62	61	42	41
NT	01	NT01	10	200	Forest	35	35	35	34	39	38	26	26
	02	NT02	20	400	Forest	46	45	47	47	51	51	35	34
	03	NT03	20	1400	Forest	55	55	61	60	63	63	41	41
	04	NT04	50	500	Forest	109	109	113	112	119	119	71	71
	07	NT07	20	300	Non-forest	12	12	13	14	14	14	7	7
	08	NT08	40	300	Non-forest	23	23	25	25	26	26	13	13
	09	NT09	30	100	Non-forest	15	15	17	17	18	18	9	9
	10	NT10	30	2000	Non-forest	25	26	30	30	31	31	13	13
	12	NT12	40	600	Forest	83	83	86	86	90	90	59	58
	13	NT13	20	500	Non-forest	12	12	15	15	15	15	7	7
	14	NT14	20	100	Forest	43	43	44	44	48	47	33	33
OC	01	OC01	20	300	Forest	46	45	46	45	50	50	34	34
	02	OC02	20	300	Forest	46	45	46	45	50	50	34	34
	07	OC07	30	2000	Non-forest	24	24	28	28	29	29	13	13
PA	01	PA01	30	1300	Forest	63	62	65	64	69	68	47	47
	04	PA04	50	200	Forest	84	83	84	83	94	92	59	59
	05	PA05	50	1300	Forest	106	106	111	111	117	115	73	70
	06	PA06	65	300	Forest	120	119	127	125	134	133	81	81

08	PA08	50	400	Non-forest	25	24	26	26	28	27	15	15
09	PA09	50	100	Non-forest	23	23	24	24	25	25	14	14
10	PA10	40	2000	Non-forest	26	25	28	28	29	28	14	15
11	PA11	65	200	Forest	118	117	124	122	132	130	80	79
12	PA12	40	600	Forest	72	71	72	70	79	79	52	51
13	PA13	40	500	Non-forest	18	18	19	19	21	20	11	11

Table 8.2.4. Land use classification of LADA and Anthromes

Land use class used for this comparison	LADA	Anthrome_v2
Reference (=primary habitat) **Reference (Natural forest, shrub, grassland, sparse vegetation, wetland, bare area)**	Forest - virgin Forest - protected Shrubs - unmanaged Shrubs - protected Grasslands - unmanaged Grasslands - protected Sparsely vegetated areas - unmanaged Sparsely vegetated areas - protected Wetlands - unmanaged Wetlands - protected Wetlands - mangrove Bare areas - unmanaged Bare areas - protected	Wild woodlands Remote woodlands Wild treeless and barren lands
Managed forest (human modified forest) **Used Forest, incl Agroforest and young secondary forest (<10 years regrowing)**	Forest - with agricultural activities Forest - with moderate or higher livestock	Populated woodlands Residential woodlands
Pasture	Grasslands - low livestock density Grasslands - moderate livestock density Grasslands - high livestock density Shrubs - low livestock density Shrubs - moderate livestock density Shrubs - high livestock density Sparsely vegetated areas - with low livestock Sparsely vegetated areas - mod.or high livestock Bare areas - with low livestock density Bare areas - with mod. livestock density	Residential rangelands Populated rangelands Pastoral villages Remote rangelands Inhabited treeless and barren
Agriculture **Cropland (flooded, irrigated, rainfed, other)**	Wetlands - with agricultural activities Agriculture - large scale Irrigation Rainfed crops (Subsistence/Commercial) Agriculture - protected Crops and mod. intensive livestock density	Rice villages Irrigated villages Residential irrigated cropland Populated croplands Rainfed villages Residential rainfed croplands Remote croplands

Land use class used for this comparison	LADA	Anthrome_v2
	Crops and high livestock density Crops, large-scale irrig., mod. or higher livestock	
Urban area	Urban land	Urban Mixed settlements
No Data	No data Undefined	
Excluded land cover types Reference Natural (water body)	Open Water - inland Fisheries Open Water - unmanaged Open Water - protected	

Table 8.2.5. Pearson's correlation coefficients of median CFs across taxonomic groups

		Occupation					**Transformation**					**Permanent**				
		Amphibians	Mammals	Reptiles	Birds	Plants	Amphibians	Mammals	Reptiles	Birds	Plants	Amphibians	Mammals	Reptiles	Birds	Plants
Agriculture	Amphibians	1					1					1				
	Mammals	0.71	1				0.57	1				0.50	1			
	Reptiles	0.54	0.45	1			0.45	0.24	1			0.86	0.54	1		
	Birds	0.44	0.58	0.79	1		0.43	0.45	0.90	1		0.55	0.26	0.54	1	
	Plants	0.56	0.57	0.88	0.89	1	0.49	0.40	0.92	0.95	1	-	-	-	-	1
Urban	Amphibians	1					1					1				
	Mammals	0.54	1				0.50	1				0.33	1			
	Reptiles	0.94	0.52	1			0.95	0.46	1			0.84	0.37	1		
	Birds	0.28	0.85	0.29	1		0.35	0.76	0.33	1		0.41	0.33	0.46	1	
	Plants	0.31	0.71	0.30	0.79	1	0.25	0.60	0.24	0.81	1	-	-	-	-	1
Forestry	Amphibians	1					1					1				
	Mammals	0.68	1				0.60	1				0.30	1			
	Reptiles	0.79	0.66	1			0.79	0.58	1			0.87	0.33	1		
	Birds	0.48	0.79	0.50	1		0.49	0.76	0.52	1		0.65	0.23	0.66	1	

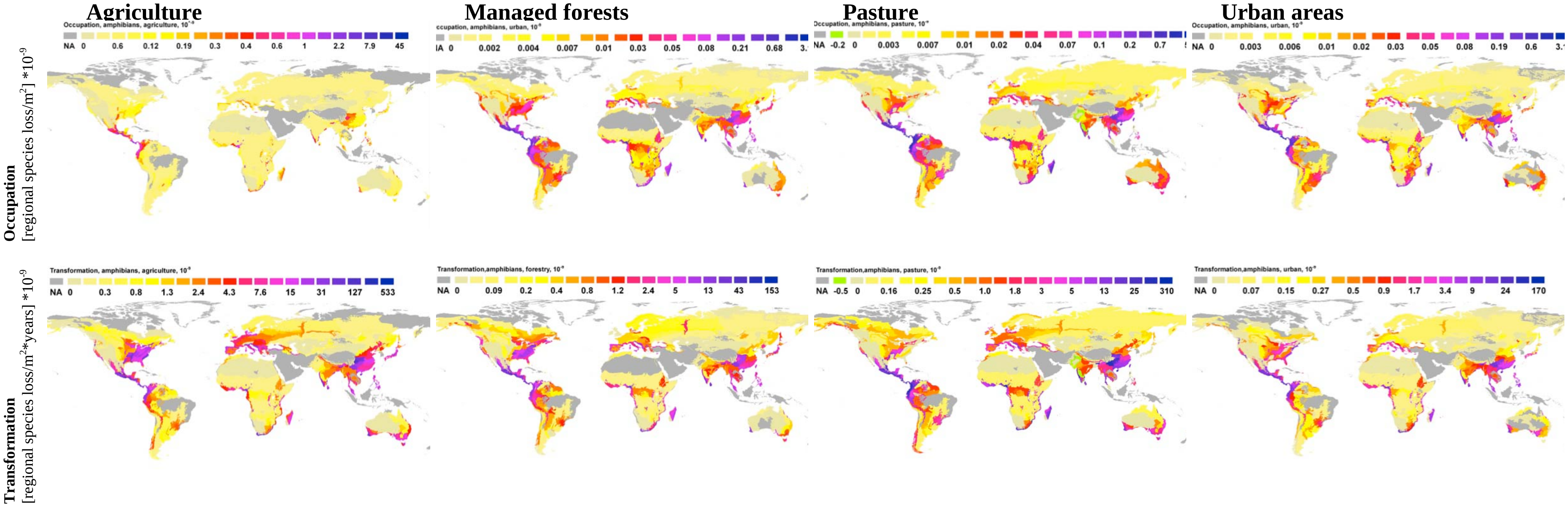
	Plants	0.51	0.69	0.51	0.68	1	0.44	0.57	0.45	0.66	1	-	-	-	-	1
Pasture	Amphibians	1					1					1				
	Mammals	0.74	1				0.75	1				0.48	1			
	Reptiles	0.90	0.72	1			0.91	0.70	1			0.73	0.62	1		
	Birds	0.70	0.85	0.70	1		0.79	0.87	0.80	1		0.37	0.40	0.46	1	
	Plants	0.73	0.82	0.70	0.85	1	0.79	0.81	0.75	0.91	1	-	-	-	-	1

Table 8.2.6. Pearson's correlation coefficients of median CFs across land use types

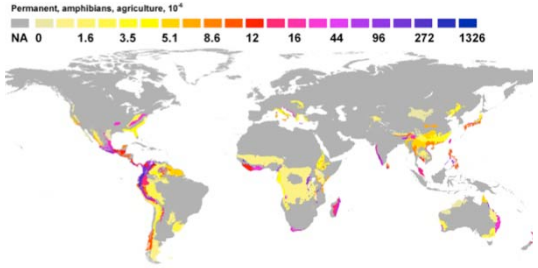
		Occupation				**Transformation**				**Permanent**			
		Agriculture	**Urban**	**Forestry**	**Pasture**	**Agriculture**	**Urban**	**Forestry**	**Pasture**	**Agriculture**	**Urban**	**Forestry**	**Pasture**
Mammals	Agriculture	1				1				1			
	Urban	0.54	1			0.67	1			0.42	1		
	Forestry	0.85	0.61	1		0.79	0.64	1		0.86	0.50	1	
	Pasture	0.65	0.27	0.62	1	0.52	0.24	0.53	1	0.44	0.42	0.44	1
Amphi-bians	Agriculture	1				1				1			
	Urban	0.48	1			0.41	1			0.89	1		
	Forestry	0.94	0.41	1		0.87	0.32	1		0.93	0.90	1	
	Pasture	0.43	0.46	0.44	1	0.42	0.37	0.42	1	0.61	0.48	0.51	1
Reptiles	Agriculture	1				1				1			
	Urban	0.34	1			0.18	1			0.88	1		
	Forestry	0.60	0.56	1		0.47	0.36	1		0.94	0.91	1	
	Pasture	0.23	0.47	0.53	1	0.14	0.33	0.49	1	0.75	0.43	0.59	1
Birds	Agriculture	1				1				1			
	Urban	0.44	1			0.27	1			0.69	1		
	Forestry	0.60	0.75	1		0.40	0.68	1		0.96	0.69	1	
	Pasture	0.36	0.13	0.41	1	0.21	0.13	0.32	1	0.16	0.02	0.11	1
Plants	Agriculture	1				1				-			
	Urban	0.27	1			0.19	1			-	-		
	Forestry	0.47	0.37	1		0.33	0.28	1		-	-	-	
	Pasture	0.31	0.21	0.38	1	0.23	0.15	0.31	1	-	-	-	-

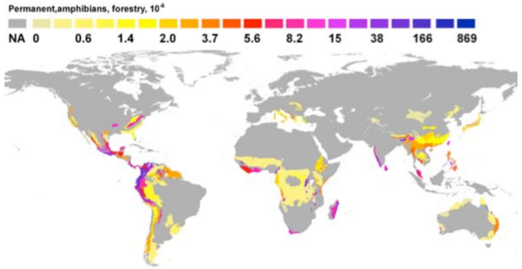
8.2.1. Maps of median characterization factors

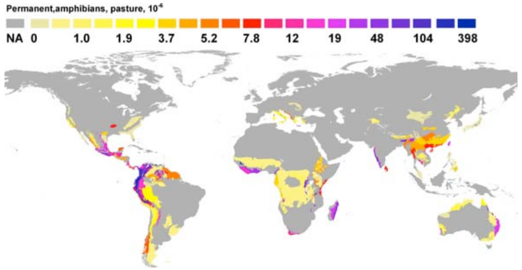
Figure 8.2.1. Median characterization factors amphibians. NA: No data available.



Permanent
[global species loss/m^2*years]







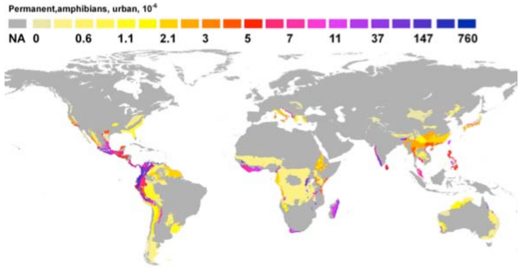


Figure 8.2.2. Median characterization factors birds. NA: No data available.

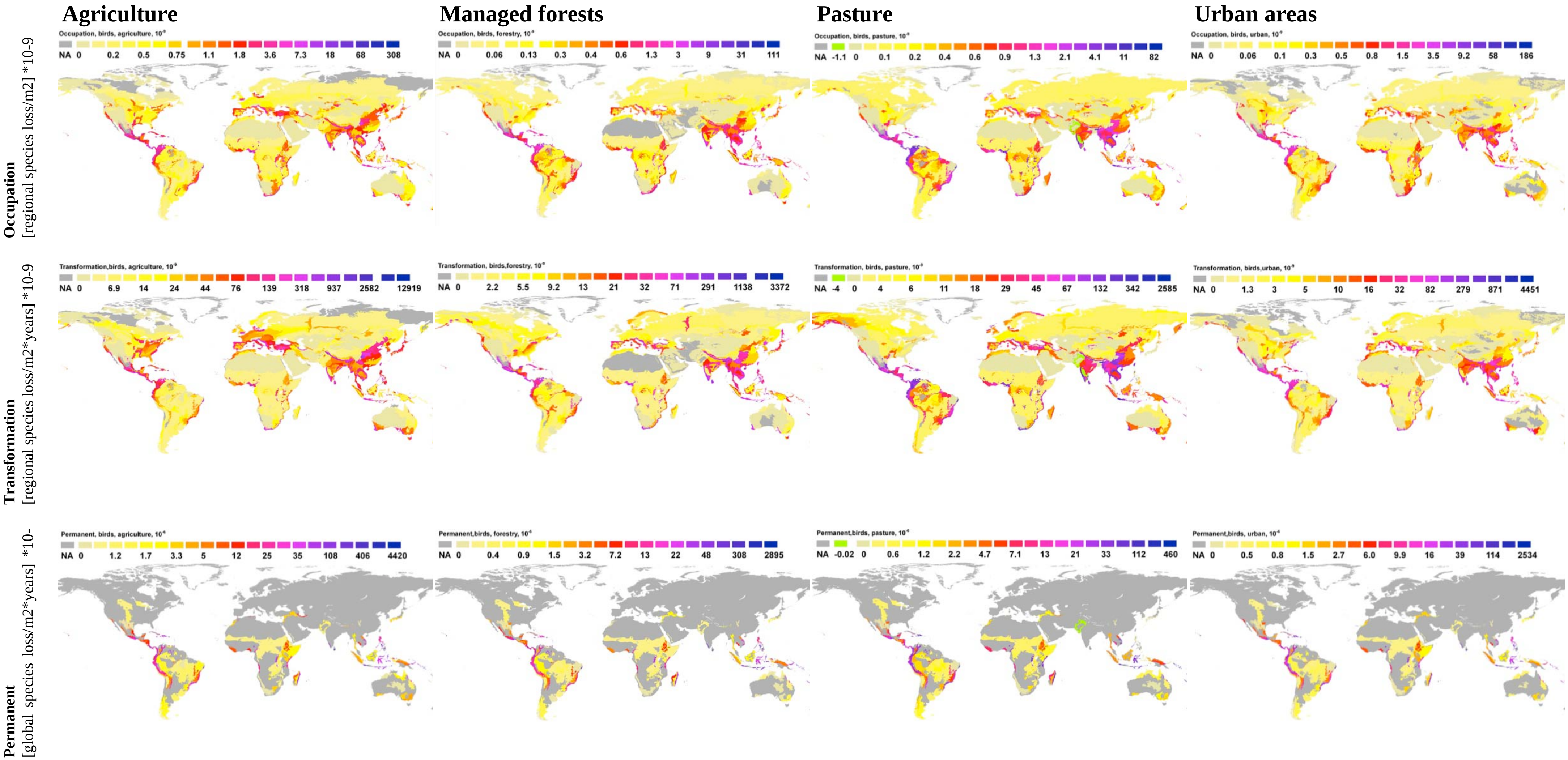


Figure 8.2.3. Median characterization factors mammals. NA: No data available.

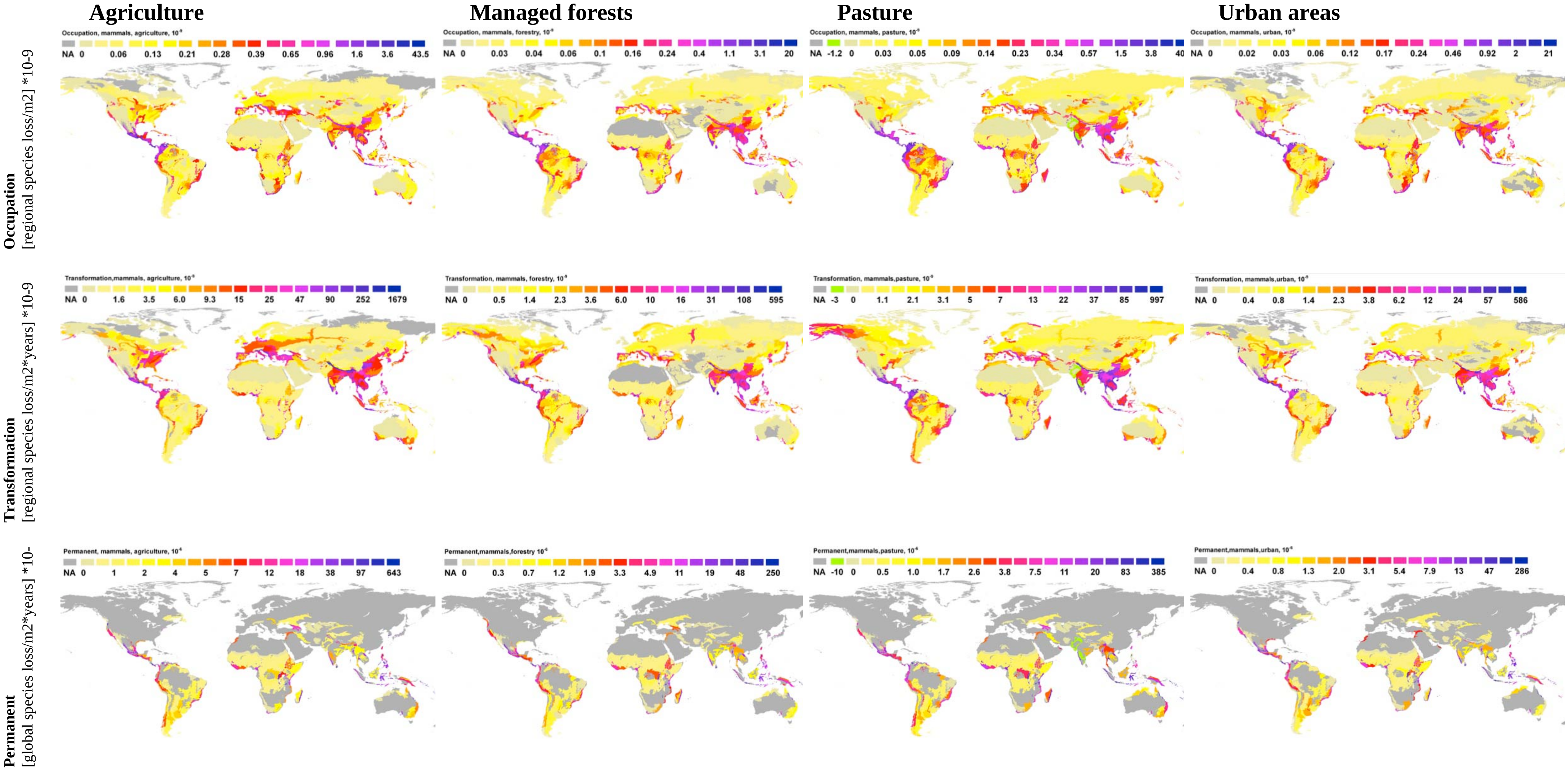
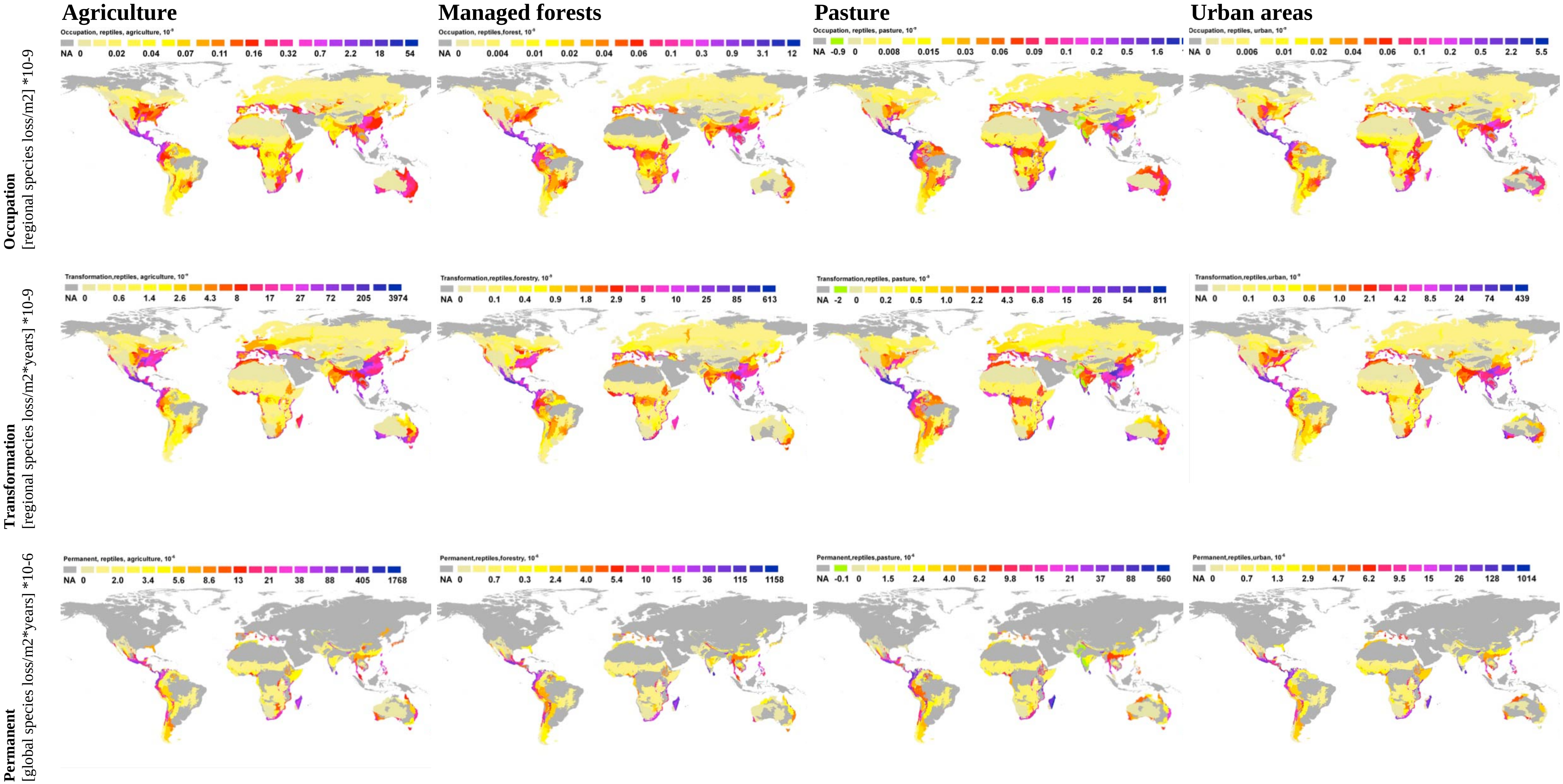


Figure 8.2.4. Median characterization factors reptiles. NA: No data available.



Sensitivity analysis

The model presented in the manuscript is calculating *average* impacts based on past land use changes and is thus *retrospective*. Alternatively, impacts can be calculated as *marginal* changes (Huijbregts et al. 2011; Weidema 2012), i.e. the impact one additional m^2 of land use would cause. Impacts can also be calculated for future land use changes, i.e. *prospective*. To illustrate the sensitivity of the model to these model choices, we calculated average and marginal impacts for both retro- and prospective assessments for all forest ecoregions (at least 90% original forest cover) of the Amazon (n=19), for plants, mammals and birds. This region was selected as no future land use scenarios for all global ecoregions were readily available and the Amazon contains some currently little disturbed ecoregions, which are expected to be converted for human use in the near future.

Methods for the prospective assessment

Land use scenarios for 2050 were derived from Soares-Filho et al. (2006), which provide several deforestation scenarios for each ecoregion in the Amazon. To get the range of possible results, we selected a best- (Governance) and worst-case (Buisness-as-usual, BAU) land use scenario. The BAU scenario assumes that the recent deforestation trends will continue, currently planned highways will be paved, legislation requiring forest reserves will remain low and no new protected areas will be created (Soares-Filho et al. 2006). The Governance scenario assumes that Brazilian environmental legislation is implemented across the Amazon basin, including current experiments on frontier governance (mandatory forest reserves, protected area network, agro-ecological zoning of land use; Soares-Filho et al. 2006). As these scenarios only consider deforestation and do not specify the type of land use established on the converted land, we assumed that the current land use mix remains the same (i.e. p_i is constant, but A_{new} changes, see 'Methods' section of manuscript). In addition, Soares-Filho et al. only considered deforestation and no other types of land use (e.g. selective logging, sparse settlement, etc.) and used other land cover data with a higher resolution than in this study. Therefore, their estimates of deforested land cannot be directly compared to the share of human-modified land (A_{lost}/A_{org}) as derived in our study. We used the future deforestation rate r_j per ecoregion j (Eq S1) as an proxy of the land use change rate. To get future shares of human-modified land ($A_{lost,2050}/A_{org}$), we multiplied the current share of human-modified land by the land use change rate r, eq S2. The resulting remaining share of natural habitat (A_{new}/A_{org}) of the original scenarios and our adaptations are illustrated in Figure 8.2.6.

$$r_j = \frac{A_{deforest,j,2050}}{A_{deforest,j,current}} \quad \text{(S1)}$$

$$\frac{A_{lost,j,2050}}{A_{org,j}} = r_j * \frac{A_{lost,j,current}}{A_{org,j}} \quad \text{(S2)}$$

Methods for the marginal assessment

Marginal damage function (eq S4) is given by the first derivative of the average damage function (eq S3) by the lost area (converted area):

$$S_{lost,g} = S_{org,g} - S_{new,g} = S_{org,g} - S_{org,g} * \left(\frac{A_{new}}{A_{org}} \right)^{z\sum_{i}^{n} p_i CF_{loc,i}} \tag{S3}$$

$$\frac{\partial S_{lost,g,j}}{\partial A_{lost,g,j}} = \frac{S_{org,g,j}}{A_{org,g,j}} * \gamma \sum_{i}^{n} p_{i,j} CF_{loc,i,j} * \left(1 - \frac{A_{lost,g,j}}{A_{org,g,j}} \right)^{\left(\gamma \sum_{i}^{n} p_{i,j} CF_{loc,i,j}\right)-1}$$

$$= \frac{S_{org,g,j}}{A_{org,g,j}} * \gamma \sum_{i}^{n} p_{i,j} CF_{loc,i,j} * \left(\frac{A_{new,g,j}}{A_{org,g,j}} \right)^{\left(\gamma \sum_{i}^{n} p_{i,j} CF_{loc,i,j}\right)-1} \tag{S4}$$

A_{org} is assumed constant, only A_{lost} is changing (no new reference). Also p_i and CFi are assumed as constants.

The CFs are then calculated as a marginal species loss $\Delta S_{lost,marg}$ due to a marginal increase in human used area $\Delta A_{lost,\ marg} = 1\text{m}^2$.

$$CF_{reg,occ,i,j,g,mar} = \frac{\Delta S_{lost,nonend,j,g} * a_{i,j}}{\Delta A_{lost,j} * p_{i,j}} \tag{S5}$$

$$CF_{reg,trans,i,j,g,mar} = 0.5 * t_{reg,i,j,g} * CF_{reg,occ,i,j,g,mar} \tag{S6}$$

$$CF_{reg,perm,i,j,g,mar} = t_m * \frac{\Delta S_{lost,end,j,g} * a_{i,j}}{\Delta A_{lost,j} p_{i,j}} \tag{S7}$$

A graphical illustration is provided in Figure 8.2.5, for an invented example.

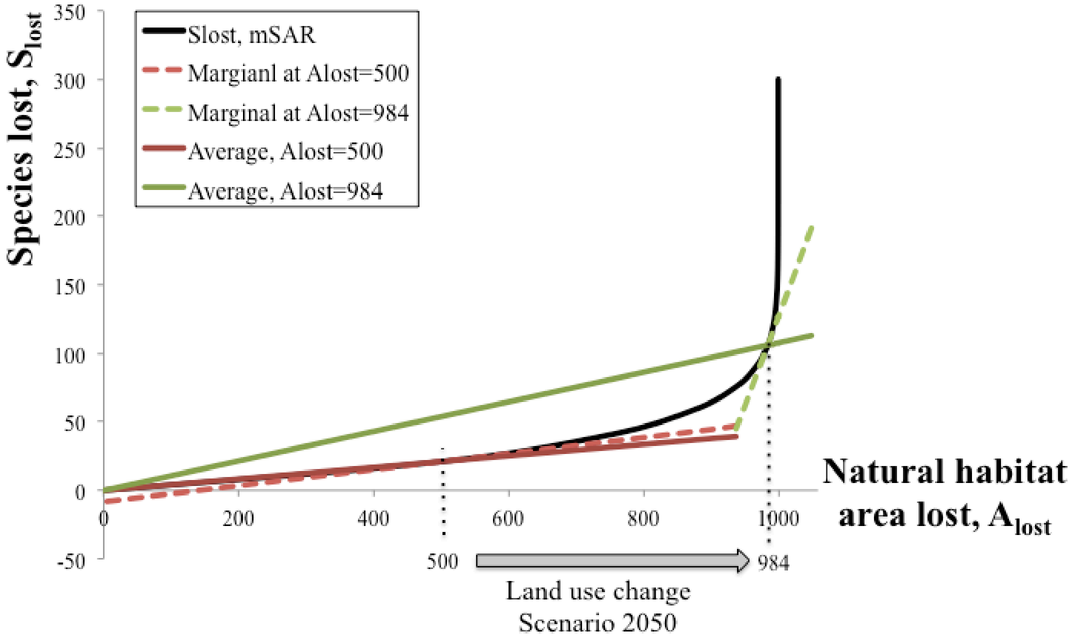


Figure 8.2.5. Illustration of differences between average and marginal damages. At low levels of remaining habitat (A_{lost}=984, green, full lines): marginal damage much higher than average. At intermediate levels of remaining habitat (A_{lost}=500, red dotted lines): differences are small. (Example numbers: A_{total} = 1000, S_{total} =300)

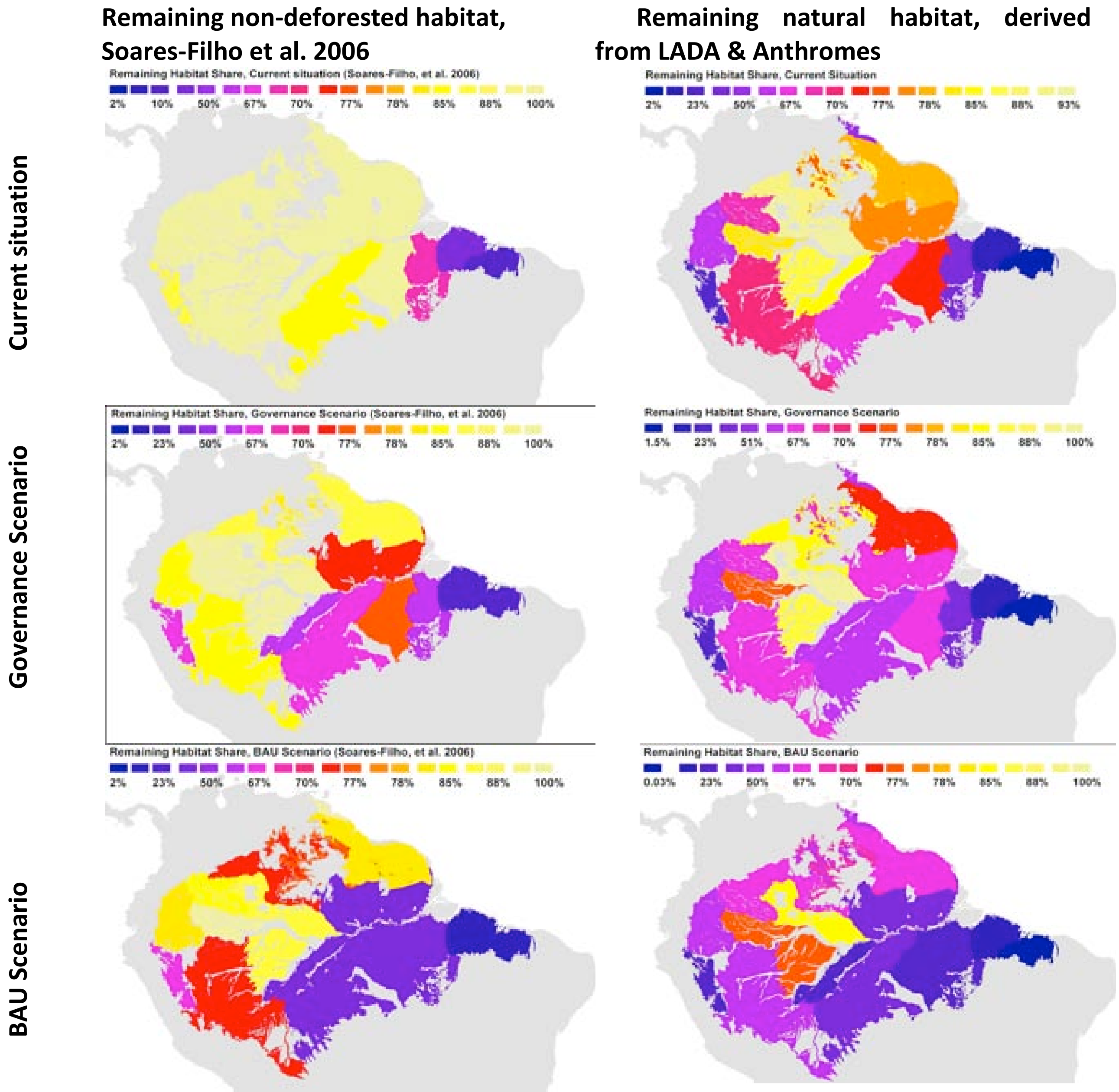


Figure 8.2.6: Share of remaining habitat (A_{new}/A_{org}) for the current situation (top) and for the two scenarios Governance (middle) and Business as Usual, (BAU, bottom) 2050. Left: share of non-deforested area (data from Soares-Filho et al. 2006). Right: own calculations, extrapolated scenarios with the LADA and Anthromes land use shares. Light yellow = undisturbed; dark blue = heavily disturbed.

Results marginal and prospective assessment

Results for both prospective assessment and marginal approach are illustrated in Figure 8.2.7. For the worst-case scenario, the median *prospective* CFs increased only for ecoregions with large projected land use changes, for small increases in land use no changes in median CFs becomes apparent (SI, Fig S7). A maximum increase of median CF of 65% was observed in ecoregions with a projected land use change from 60% remaining habitat to 20%.

Using a *marginal* impact calculation, the CFs did not change considerably at low levels of habitat conversion, but at high levels, impacts were even more pronounced using a marginal approach, turning to infinity when the remaining natural habitat would become very small (Fig S5 and S7). Thus, the marginal approach gives even higher weight to highly vulnerable areas and is even more reactive as the average approach.

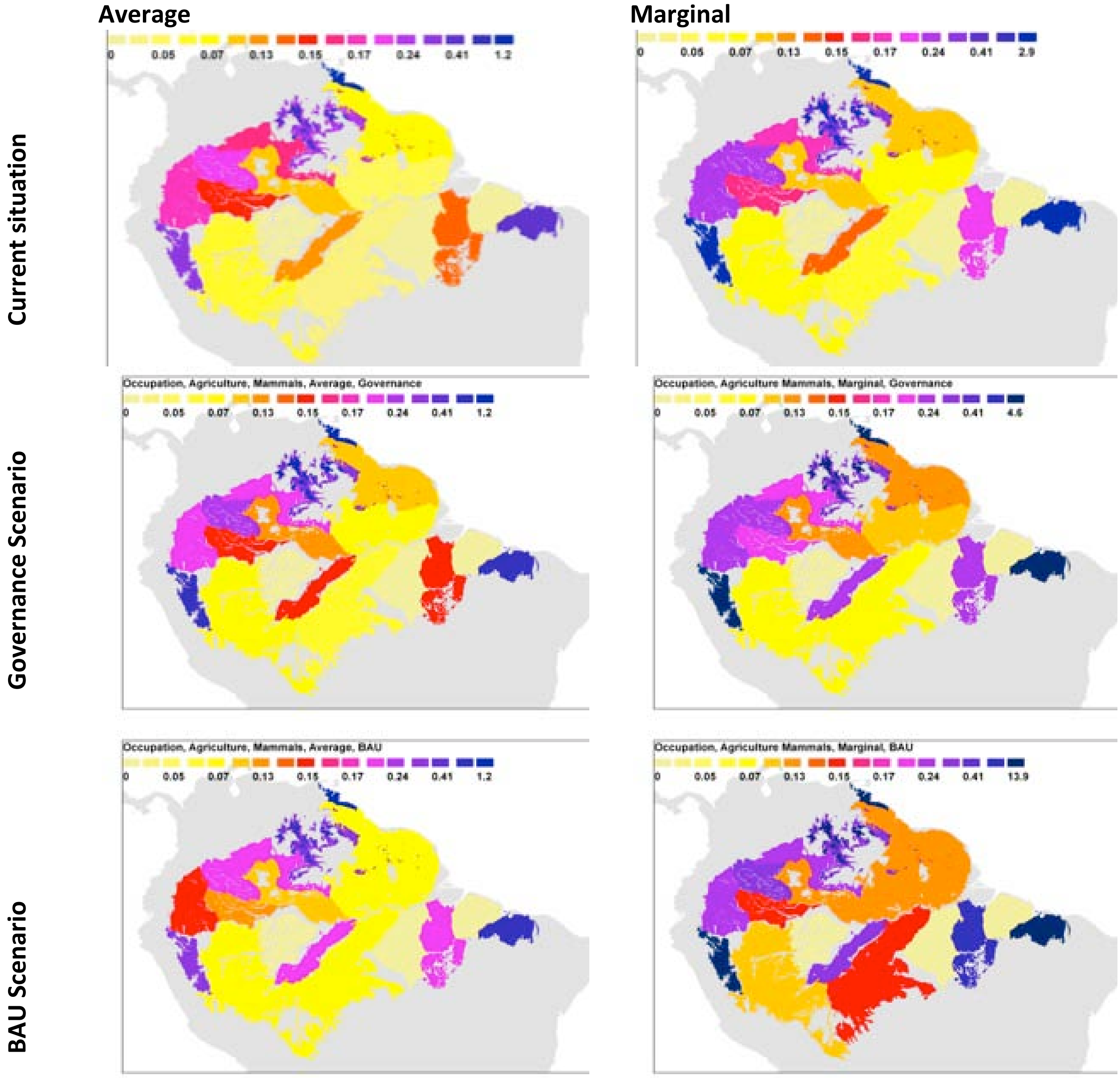


Figure 8.2.7. Characterization factors, occupation, mammals: Average (left) and marginal (right) CFs for the current situation (top) and for the two scenarios Governance (middle) and Business as Usual, (BAU, bottom) for 2050. Numbers displayed as [regional species loss / m^2]*10^{-9}.

Table 8.2.7. Median contribution to variance (CTV) scores (of 1000 Monte Carlo iterations) for regional *transformation* characterization factors (per land use type and taxonomic group)

Parameter	Taxa	Agriculture	Pasture	Urban area	Managed forest
$CF_{loc,i,j}$ (local characterization factors)	Amphibians	44%	53%	87%	77%
	Reptiles	45%	51%	87%	74%
	Mammals	52%	71%	89%	83%
	Birds	51%	70%	90%	83%
	Plants	53%	69%	90%	83%
$A_{org,j}$ (original natural habitat area)	Amphibians	0.33%	0.30%	0.07%	0.13%
	Reptiles	0.33%	0.32%	0.07%	0.13%
	Mammals	0.97%	0.93%	0.26%	0.37%
	Birds	1.09%	0.89%	0.24%	0.28%
	Plants	1.19%	0.87%	0.28%	0.37%
$A_{new,j}$ (remaining natural habitat)	Amphibians	0.48%	0.36%	0.08%	0.10%
	Reptiles	0.42%	0.36%	0.08%	0.09%
	Mammals	1.36%	1.14%	0.22%	0.38%
	Birds	1.48%	1.09%	0.20%	0.36%
	Plants	1.49%	1.07%	0.24%	0.47%
$A_{i,j}$ (area of land use type *i*)	Amphibians	0.37%	0.39%	0.07%	0.12%
	Reptiles	0.36%	0.42%	0.07%	0.11%
	Mammals	1.17%	1.22%	0.22%	0.41%
	Birds	1.33%	1.23%	0.22%	0.40%
	Plants	1.38%	1.29%	0.26%	0.48%
$t_{reg\ i,j,g}$ (Regeneration time)	Amphibians	21%	6%	4%	4%
	Reptiles	21%	5%	4%	4%
	Mammals	30%	15%	6%	9%
	Birds	31%	16%	6%	9%
	Plants	31%	16%	5%	9%
z (power term of SAR model)	Amphibians	0.03%	0.04%	0.02%	0.02%
	Reptiles	0.03%	0.04%	0.02%	0.02%
	Mammals	0.08%	0.09%	0.07%	0.03%
	Birds	0.06%	0.09%	0.07%	0.02%
	Plants	0.06%	0.10%	0.07%	0.05%

Table 8.2.8. Median contribution to variance (CTV) scores (of 1000 Monte Carlo iterations) for regional *occupation* characterization factors (per land use type and taxonomic group)

Parameter	**Taxa**	**Agriculture**	**Pasture**	**Urban area**	**Managed forest**
$CF_{loc,i,j}$ (local characterization factors)	Amphibians	67%	85%	93%	90%
	Reptiles	68%	84%	93%	89%
	Mammals	81%	92%	97%	96%
	Birds	81%	92%	97%	96%
	Plants	81%	92%	97%	96%
$A_{org,j}$ (original natural habitat area)	Amphibians	0.7%	0.5%	0.1%	0.2%
	Reptiles	0.7%	0.5%	0.1%	0.2%
	Mammals	2.6%	1.4%	0.4%	0.5%
	Birds	2.7%	1.6%	0.4%	0.5%
	Plants	2.8%	1.7%	0.4%	0.6%
$A_{new,j}$ (remaining natural habitat)	Amphibians	1.4%	0.6%	0.2%	0.2%
	Reptiles	1.3%	0.7%	0.1%	0.2%
	Mammals	4.2%	2.1%	0.5%	0.8%
	Birds	4.3%	2.1%	0.5%	0.8%
	Plants	4.4%	2.1%	0.5%	0.8%
$A_{i,j}$ (area of land use type *i*)	Amphibians	1.1%	0.7%	0.2%	0.2%
	Reptiles	1.1%	0.8%	0.2%	0.2%
	Mammals	3.6%	2.2%	0.4%	0.7%
	Birds	3.8%	2.3%	0.4%	0.7%
	Plants	3.9%	2.5%	0.5%	0.8%
$t_{reg\ i,j,g}$ (Regeneration time)	Amphibians	0.03%	0.02%	0.03%	0.01%
	Reptiles	0.03%	0.02%	0.03%	0.01%
	Mammals	0.05%	0.04%	0.07%	0.03%
	Birds	0.06%	0.04%	0.03%	0.03%
	Plants	0.05%	0.07%	0.08%	0.05%
z (power term of SAR model)	Amphibians	0.03%	0.07%	0.03%	0.03%
	Reptiles	0.03%	0.07%	0.03%	0.02%
	Mammals	0.08%	0.14%	0.08%	0.06%
	Birds	0.08%	0.15%	0.08%	0.06%
	Plants	0.08%	0.15%	0.08%	0.06%

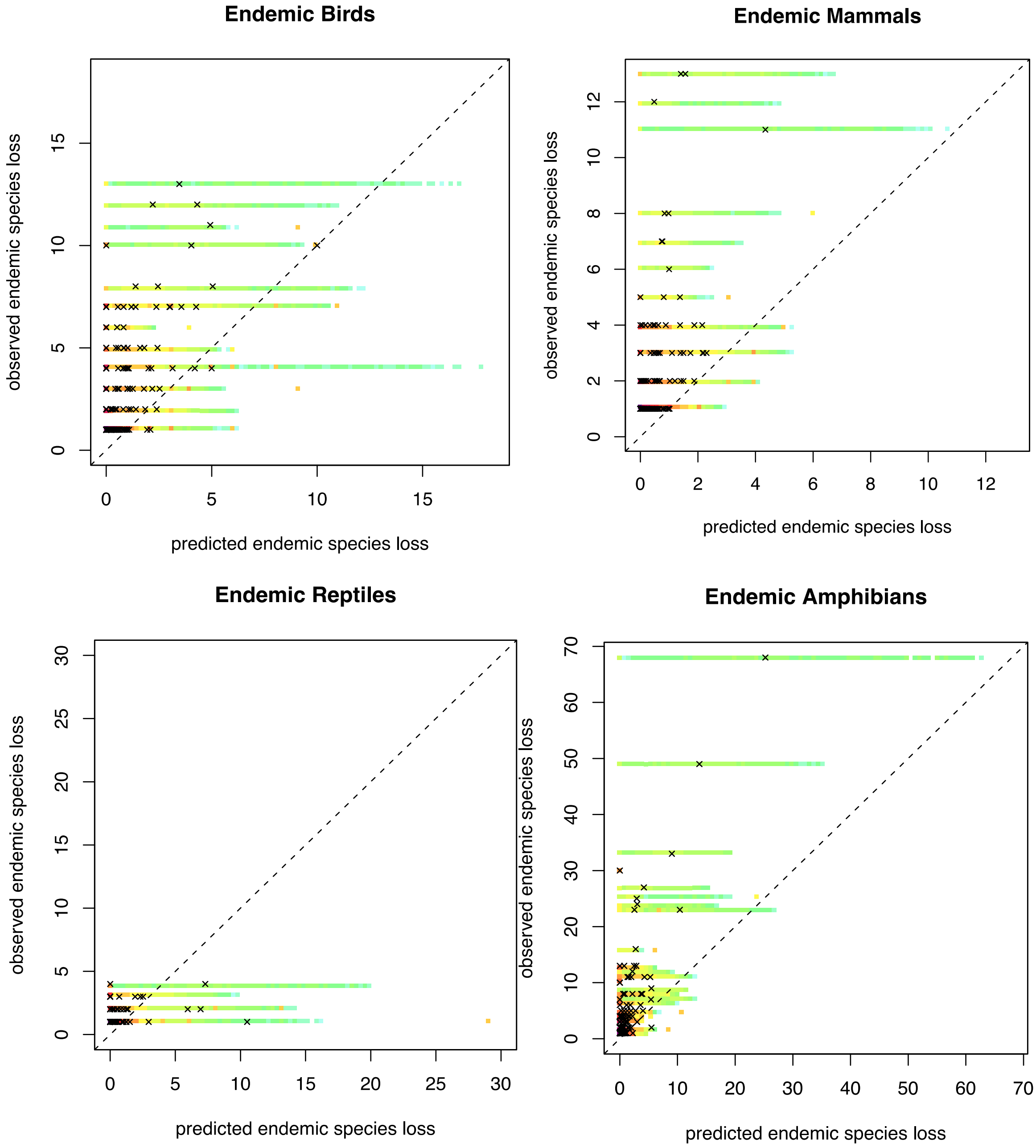


Figure 8.2.8. Comparison of predicted and observed globally threatened and extinct endemic birds, mammals, reptiles and amphibians. For each ecoregion, all extinction estimates (n=1000 Monte Carlo iterations) are displayed. Colors represent data density (green = low density, dark red = highest density); black crosses represent median species loss per ecoregion; dotted line represents 1:1 (complete overlap of predicted and observed).

8.2.2. Aggregation of characterization factors across taxa

LCA practitioners might be overwhelmed by 5 characterization factors specific per taxonomic group. We therefore suggest a way to aggregate these factors. As we present absolute impacts (species losses), simply adding CFs across taxon would give equal weight to e.g. one plant species lost and one mammal species lost. In this case, the impacts would be strongly dominated by plants, by far the most species rich taxon assessed here. To also capture impacts on other taxa, we suggest a weighing of the CFs per taxon. The weighing factor was calculated based on the median species richness per taxa of all global ecoregions, $S_{g,med}$. We then standardized these values by the median richness of mammals $S_{mam,med}$. The weighing factor w per taxonomic group g is then given as, (see Table 8.2.9):

$$w_g = \frac{S_{mam,med}}{S_{g,med}} \qquad \text{(S8)}$$

Table 8.2.9. Median species richness $S_{g,med}$ per taxa *g* of all global ecoregions and resulting weighing factors w_g.

Taxon	Mammals	Birds	Plants	Amphibians	Reptiles
$S_{g,med}$	69	230	1400	7	17
w_g (rounded)	1	0.3	0.05	10	4

The aggregated CFs were then calculated as a weighted average across all taxonomic groups *g*.

$$CF_{reg,occ,i,j} = \frac{\sum_g^n w_g * CF_{reg,occ,i,j,g}}{n_j} \qquad \text{(S9)}$$

$$CF_{reg,trans,i,j} = \frac{\sum_g^n w_g * CF_{reg,trans,i,j,g}}{n_j} \qquad \text{(S10)}$$

(n_j= total number of taxonomic groups *g* that contain data (i.e. not only zero values) per ecoregion *j*).

Characterization factors for permanent impacts should indicate, if a land use activity might cause irreversible impacts. Therefore, we suggest to not include a weighing or averaging, but simply summing up all permanent impacts.

$$CF_{reg,perm,i,j} = \sum_g^n CF_{reg,perm,i,j,g} \qquad \text{(S11)}$$

8.2.3. Calculation of world average characterization factors

In many LCA studies, the geographic location of background processes and their land use is unknown. For these cases, we calculated world average CFs by weighing the CF of each ecoregion by their global area share (see Excel Table in SI). Alternatively, worst-case (highest values) or median CFs could be chosen.

8.2.4. Normalization

As a reference system for normalization, we used the average land use impact per person in the year 2005. To get the total land use impact per ecoregion, we first multiplied the CFs for occupation by the area occupied by each land use type within each ecoregion $A_{Occ(i)}$. The total global land use impact per taxonomic group was derived as the sum of all land use impacts of all ecoregions. Finally, the result was devided by the world human population in the year 2005, $P_{glob,2005}$. A total world population of 6.51 billion people was assumed for the year 2005 (official estimate of the United Nations).

$$NF_{Occ,g} = \frac{\sum_j \sum_i \left(CF_{Occ,i,j,g} * A_{Occ(i),j} \right)}{P_{glob,2005}} * 1year$$

If data for one taxa was missing for one ecoregion, aggregated CFs across taxa were used instead. This was especially the case for amphibians and reptiles, where data was missing for many ecoregions. The resulting factors are given in Table 8.2.10.

Table 8.2.10: Normalization factors for land occupation.

		Mammals	Birds	Plants	Amphibians	Reptiles	Aggregated
Total global impact	[pot. lost non-endemic sp. / year]	1.44E+04	4.76E+04	3.57E+05	5.91E+03	9.43E+03	2.22E+04
Normalization factors $NF_{Occ,g}$	[pot. lost non-endemic sp. / year / person]	2.22E-06	7.31E-06	5.48E-05	9.09E-07	1.45E-06	3.41E-06

For transformation and permanent impacts, normalization factors would be calculated as follows:

$$NF_{Trans,g} = \frac{\sum_j \sum_i \left(CF_{Trans,i,j,g} * A_{trans(nat->i),j} \right)}{P_{glob,2005}} * 1year$$

$$NF_{Perm,g} = \frac{\sum_j \sum_i \left(CF_{Perm,i,j,g} * A_{trans(nat->i),j} \right)}{P_{glob,2005}} * 1year$$

Here, the area of land transformed from natural to each land use type *i*, $A_{trans(nat->i),j}$, is needed per ecoregion. As this information is not readily available, we could not calculate normalization factors for transformation and permanent impacts. Yearly land use statistics are available on a country level from FAOSTAT, and could be used to calculate net land use change per country. However, this data (i) does not inform about which land use type was transformed into which other type, (ii) only records a change in forests ecosystem area, but not of other natural ecosystem types such as grassland ecosystems and (iii) is not available on a ecoregion level needed for this project.

8.2.5. References

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8.3. Annex to Chapter 3

8.3.1 Species richness (SR) and Functional Diversity (FD) data for mammals' studies.

Location	Source	Land use types	Intensity	Ecoregion	FD	SR	FD_N	SR_N	CF_{FD}	CF_{SR}
Peru (Tambopata Province)	(Naughton-Treves et al. 2003)	**Forest, primary**	**Natural**	Southwest Amazon Moist Forests (NT0166)	**4.335**	**10**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, arable, fallow (*regenerating field*)	Low		3.884	8	0.896	0.800	0.104	0.200
		Agriculture, arable *(rice, maize, yucca)*	High		4.075	8	0.940	0.800	0.060	0.200
Costa Rica (Coto Brus)	(Daily et al. 2003)	**Forest, primary**	**Natural**	Isthmian-Pacific Moist Forests (NT0130)	**8.204**	**25**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, mosaic (*forest remnants*)	Low		7.144	20	0.871	0.800	0.129	0.200
		Pasture/meadow (*open areas*)	High		6.942	19	0.846	0.760	0.154	0.240
Costa Rica (San Pedro)	(Husband et al. 2009)	**Forest (*data from 2005*)**	**Natural**	Costa Rican Seasonal Moist Forests (NT0119)	**4.364**	**10**	**1.000**	**1.000**	**0.000**	**0.000**
		Forest (*data from 2006*)	Natural		5.132	15	1.176	1.500	-0.176	-0.500
		Agriculture, mosaic (*shaded coffee*)	Low		4.013	8	0.920	0.800	0.080	0.200
		Agriculture, permanent crop, non-irrigated (*sun coffee*)	High		1.304	4	0.299	0.400	0.701	0.600
Mexico ((Los Tuxtlas, Veracruz)	(Estrada et al. 1994)	**Forest, secondary**	**Natural**	Sierra de los Tuxtlas (NT0161)	**11.042**	**38**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, mosaic (*cacao*)	High		4.935	16	0.447	0.421	0.553	0.579
		Agriculture, mosaic (*mixed*)	High		4.274	13	0.387	0.342	0.613	0.658
		Agriculture, mosaic (*coffee*)	High		3.959	13	0.359	0.342	0.641	0.658
		Field Margins/Hedgerows	Low		3.680	11	0.449	0.440	0.551	0.560
		Agriculture, permanent crop, non-irrigated (*citrus*)	High		3.517	8	0.318	0.211	0.682	0.789
		Agriculture, permanent crop, non-irrigated (*allspice*)	High		2.993	7	0.271	0.184	0.729	0.816
		Pasture/meadow	High		2.187	4	0.198	0.105	0.802	0.895
Mexico (Chiapas)	(Horváth et al. 2001b)	**Shrubland, ecotone**	**Natural**	Montane Forests (NT0113)	**3.572**	**9**	**1.000**	**1.000**	**0.000**	**0.000**
		Forest, natural	Low		2.627	5	0.735	0.556	0.265	0.444
		Agriculture, arable, non-irrigated (*intensive*)	High		1.485	3	0.416	0.333	0.584	0.667
United States (Northern California)	(Hilty & Merenlender 2004)	**Forest, primary**	**Natural**	California interior chaparral and woodlands (NA1202)	**2.761**	**9**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, permanent crop, non-irrigated (*isolated*)	High		1.955	7	0.708	0.778	0.292	0.222
		Agriculture, permanent crop, non-irrigated (*near natural area*)	High		2.525	6	0.915	0.667	0.085	0.333
United States (southern Wyoming)	(Olson & Brewer 2003)	**Grassland, not used**	**Natural**	Western Short Grasslands (NA0815)	**2.478**	**5**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, arable, non-irrigated, intensive (*winter wheat*)	High		1.955	4	0.789	0.800	0.211	0.200
		Agriculture, arable, non-irrigated, intensive (*oat*)	High		1.379	3	0.556	0.600	0.444	0.400
		Agriculture, arable, non-irrigated, intensive (*corn*)	High		1.208	3	0.487	0.600	0.513	0.400
		Agriculture, arable, fallow	Low		1.379	3	0.556	0.600	0.444	0.400
Canada (British	(Sullivan &	**Forest, primary**	**Natural**	Okanagan Dry	**2.550**	**5**	**1.000**	**1.000**	**0.000**	**0.000**

Columbia)	Sullivan 2006)	Shrubland	Natural	Forests (NA0522)	2.736	6	1.073	1.200	-0.073	-0.200
		Field Margins/Hedgerows	Low		2.550	5	1.000	1.000	0.000	0.000
		Pasture/meadow	Low		2.550	5	1.000	1.000	0.000	0.000
		Agriculture, permanent crop, non-irrigated (*convent. orchard*)	High		2.245	5	0.880	1.000	0.120	0.000
		Field Margins/Hedgerows (*riparian*)	Natural		2.026	4	0.795	0.800	0.205	0.200
		Agriculture, permanent crop, non-irrigated (*dwarf orchard*)	High		2.026	4	0.795	0.800	0.205	0.200

8.3.2 Species richness (SR) and Functional Diversity (FD) data for plant studies.

Location	Source	Land use types	Intensity	Ecoregion	FD	SR	FD_N	SR_N	CF_{FD}	CF_{SR}
Costa Rica (Coto Brus)	(Mayfield et al. 2006)	**Forest, primary**, (*riverbank in forested area*)	**Natural**	Isthmian-Pacific Moist Forests (NT0130)	**15.566**	**83**	**1.000**	**1.000**	**0.000**	**0.000**
		Pasture/meadow	High		17.096	133	1.098	1.602	-0.098	-0.602
		Forest, extensive (*riverbank in deforested area*)	High		15.276	117	0.981	1.410	0.019	-0.410
		Forest, primary (*tree-fall gap in forested area*)	Natural		18.310	98	1.176	1.181	-0.176	-0.181
		Forest, primary, (*understory in forested area*)	Natural		13.182	86	0.847	1.036	0.153	-0.036
		Traffic area, rail/road embankment	Low		19.510	69	1.253	0.831	-0.253	0.169
Nicaragua, (Matiguas)	(Sanchez Merlos et al. 2005b)	**Forest, natural** (*dry forest*)	**Natural**	Central American Atlantic Moist Forests (NT0111)	**7.932**	**85**	**1.000**	**1.000**	**0.000**	**0.000**
		Forest, secondary	Natural		7.357	68	0.927	0.800	0.073	0.200
		Field Margins/Hedgerows (*riparian forest*)	Natural		10.254	99	1.293	1.165	-0.293	-0.165
		Field Margins/Hedgerows (*live fence*)	Low		5.310	51	0.669	0.600	0.331	0.400
		Pasture/meadow (*pasture with high arboreal cover*)	High		6.948	62	0.876	0.729	0.124	0.271
		Pasture/meadow (*pasture with low arboreal cover*)	High		5.310	51	0.669	0.600	0.331	0.400
Nicaragua, (Rivas)	(Sanchez Merlos et al. 2005a)	**Forest, natural** (*dry forest*)	**Natural**	Central American Dry Forests (NT0209)	**8.512**	**75**	**1.000**	**1.000**	**0.000**	**0.000**
		Forest, secondary	Natural		8.813	67	1.035	0.893	-0.035	0.107
		Field Margins/Hedgerows (*riparian forest*)	Natural		9.149	88	1.075	1.173	-0.075	-0.173
		Field Margins/Hedgerows (*live fence*)	Low		4.941	34	0.580	0.453	0.420	0.547
		Pasture/meadow (*pasture with high arboreal cover*)	High		6.591	39	0.774	0.520	0.226	0.480
		Pasture/meadow (*pasture with low arboreal cover*)	High		7.310	38	0.859	0.507	0.141	0.493
United States (Washington)	(Quinn 2004)	**Shrubland**	**Natural**	Snake-Columbia shrub steppe (NA1309)	**2.824**	**14**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, mosaic	High		3.075	16	1.089	1.143	-0.089	-0.143
		Agriculture, arable, intensive	High		3.006	15	1.064	1.071	-0.064	-0.071
United States	(Smukler et al. 2010)	**Forest, primary**, *data from 2005*	**Natural**	California Central Valley Grasslands (NA0801)	**2.100**	**14**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, arable, non-irrigated	Low		1.855	12	0.883	0.857	0.117	0.143
		Field margins/Hedgerow	High		3.687	24	1.756	1.714	-0.756	-0.714
		Lakes, artificial	High		2.309	23	1.100	1.643	-0.100	-0.643
		Rivers, artificial	High		2.128	15	1.013	1.071	-0.013	-0.071
		Agriculture, arable, non-irrigated	High		1.859	15	0.885	1.071	0.115	-0.071
		Forest, primary, *data from 2006*	**Natural**		**2.579**	**13**	**1.000**	**1.000**	**0.000**	**0.000**
		Lakes, artificial	High		3.129	27	1.213	2.077	-0.213	-1.077
		Field margins/Hedgerow	High		3.596	23	1.394	1.769	-0.394	-0.769
		Rivers, artificial	High		2.228	19	0.864	1.462	0.136	-0.462
		Agriculture, arable, non-irrigated	Low		2.047	21	0.794	1.615	0.206	-0.615
		Agriculture, arable, non-irrigated	High		2.225	19	0.862	1.462	0.138	-0.462
Canada (St. Lawrence Region)	(Middleton & Merriam 1983)	**Forest, extensive** (*Mark S.Burnham Provincial Park*)	**Natural**	Eastern Great Lakes Lowland Forests (NA0407) and Eastern Forest-Boreal	**2.457**	**12**	**1.000**	**1.000**	**0.000**	**0.000**
		Forest, secondary (*wood remnants in farmlands*)	High		3.127	32	1.273	2.667	-0.273	-1.667
		Forest, primary (*Gatineau Park*)	Natural		1.826	21	0.743	1.750	0.257	-0.750
		Forest, natural (*Shaw Woods*)	Natural		0.691	9	0.281	0.750	0.719	0.250

				Transition (NA0406)						
Canada (St. Lawrence Region)	(Jobin et al. 1996)	**Pasture/meadow, setaside**	**Natural**	Eastern Great Lakes Lowland Forests (NA0407)	**3.016**	**39**	**1.000**	**1.000**	**0.000**	**0.000**
		Forest, natural* (*woodlots*)	Low		3.151	50	1.045	1.282	-0.045	-0.282
		Forest, primary (*woodland edges*)	Low		3.157	53	1.047	1.359	-0.047	-0.359
		Field Margins/Hedgerows	High		3.018	48	1.001	1.231	-0.001	-0.231

8.3.2 Species richness (SR) and Functional Diversity (FD) data for bird studies.

Study	**Source**	**Land use types**	**Intensity**	**Ecoregion**	**FD**	**SR**	**FD_N**	**SR_N**	**CF_{FD}**	**CF_{SR}**
Costa Rica (Coto Brus, Las Cruces Forest Reserve)	(Daily et al. 2001)	**Forest, primary**	**Natural**	Isthmian-Pacific Moist Forests (NT0130)	**12.115**	**45**	**1.000**	**1.000**	**0.000**	**0.000**
		Forest, secondary	Low		10.205	49	0.842	1.089	0.158	-0.089
		Forest, secondary	Low		10.511	34	0.868	0.756	0.132	0.244
		Forest, secondary	High		12.081	39	0.997	0.867	0.003	0.133
		Forest, secondary	High		8.354	29	0.690	0.644	0.310	0.356
Mexico (Los Tuxtlas, Veracruz), 1997	(Estrada & Coates-Estrada 1997b)	**Forest, secondary*** (*forest fragment*)	**Natural**	Sierra de los Tuxtlas (NT0161)	**13.205**	**46**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, arable, non-irrigated, intensive (*non-arboreal crops*)	High		6.852	18	0.519	0.391	0.481	0.609
		Agriculture, mosaic	High		11.278	44	0.854	0.957	0.146	0.043
		Agriculture, permanent crops, non-irrigated	High		12.404	38	0.939	0.826	0.061	0.174
		Field Margins/Hedgerows (*live fence*)	Low		10.164	30	0.770	0.652	0.230	0.348
		Pasture/meadow	High		2.836	7	0.215	0.152	0.785	0.848
Mexico (Los Tuxtlas, Veracruz), 2005	(Estrada & Coates-Estrada 2005b)	**Forest, secondary***	**Natural**	Sierra de los Tuxtlas (NT0161)	**32.433**	**155**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, arable, non-irrigated, intensive (*coffee*)	High		18.747	85	0.578	0.548	0.422	0.452
		Agriculture, arable, non-irrigated, intensive (*jalapeno*)	High		5.950	18	0.183	0.116	0.817	0.884
		Agriculture, mosaic (*banana*)	High		5.757	11	0.178	0.071	0.822	0.929
		Agriculture, mosaic (*citrus*)	High		19.396	69	0.598	0.445	0.402	0.555
		Agriculture, mosaic (*mixed*)	High		19.472	73	0.600	0.471	0.400	0.529
		Agriculture, permanent crops, non-irrigated (*allspice*)	High		17.413	58	0.537	0.374	0.463	0.626
		Agriculture, permanent crops, non-irrigated (*cacao*)	High		24.973	108	0.770	0.697	0.230	0.303
		Agriculture, permanent crops, non-irrigated (*corn*)	High		8.773	27	0.271	0.174	0.729	0.826
		Field Margins/Hedgerows (*live fence*)	Low		21.973	86	0.678	0.555	0.322	0.445
		Pasture/meadow	High		3.923	8	0.121	0.052	0.879	0.948
United States of America (North-central Florida)	(Jones et al. 2005)	**Field margins/Hedgerow (organic field margin)**	**Low**	Southeastern conifer forests (NA0529)	**13.781**	**56**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, arable, non-irrigated, extensive (*organic field*)	High		12.890	50	0.935	0.893	0.065	0.107
		Field Margins/Hedgerows (*conventional field margin*)	Low		15.095	53	1.095	0.946	-0.095	0.054
		Agriculture, arable, non-irrigated, intensive (*conventional field*)	High		15.022	48	1.090	0.857	-0.090	0.143
United States of America (Iowa, northeastern Kansas, eastern Nebraska, southern Minnesota, southwestern Wisconsin, Illinois and Northern Missouri)	(Best et al. 1995)	**Forest (Floodplain forest)**	**Natural**	Central tall grasslands (NA0805)	**24.000**	**107**	**1.000**	**1.000**	**0.000**	**0.000**
		Agriculture, arable, non-irrigated (*alfafa hayfield*)	High		8.752	43	0.365	0.402	0.635	0.598
		Agriculture, arable, non-irrigated, intensive (*small grains*)	High		7.458	31	0.311	0.290	0.689	0.710
		Agriculture, arable, non-irrigated, intensive (*tilled row crops*)	High		11.475	55	0.478	0.514	0.522	0.486
		Agriculture, mosaic (*farmstead shelterbelt*)	Low		15.445	67	0.644	0.626	0.356	0.374
		Agriculture, mosaic (*wooded farmstead*)	Low		12.796	52	0.533	0.486	0.467	0.514
		Grassland	Natural		19.416	85	0.809	0.794	0.191	0.206
		Grassland (*grass hayfield*)	High		8.752	43	0.365	0.402	0.635	0.598
		Grassland (*Conservation Reserve Program*)	Low		8.773	45	0.366	0.421	0.634	0.579
		Grassland (*prairie*)	Natural		9.913	46	0.413	0.430	0.587	0.570
		Grassland for livestock/grazing	Low		11.410	53	0.475	0.495	0.525	0.505
		Field Margins/Hedgerows (*grassed waterway*)	Low		10.716	44	0.447	0.411	0.553	0.589
		Field Margins/Hedgerows (*herbaceous fencerow*)	Low		7.126	27	0.297	0.252	0.703	0.748

	Field Margins/Hedgerows (*wooded fencerow*)	Low	13.432	54	0.560	0.505	0.440	0.495
	Pasture/meadow (*oldfield set aside*)	Natural	12.745	55	0.531	0.514	0.469	0.486
	Shrubland	Natural	15.789	71	0.658	0.664	0.342	0.336
	Traffic area, railroad, embankment (*herbaceous roadside*)	Low	10.625	47	0.443	0.439	0.557	0.561
	Traffic area, railroad, embankment (*railroad*)	Low	13.918	62	0.580	0.579	0.420	0.421
	Wetlands, inland (*natural marsh*)	Natural	13.893	56	0.579	0.523	0.421	0.477
	Wetlands, inland (*restored marsh*)	Natural	12.182	46	0.508	0.430	0.492	0.570