



Developing characterisation factors for land fragmentation impacts on biodiversity in LCA: key learnings from a sugarcane case study

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Abstract

Purpose Habitat change was identified by the Millennium Ecosystem Assessment as the main direct driver of biodiversity loss. However, while habitat loss is already implemented in Life Cycle Impact Assessment (LCIA) methods, the additional impact on biodiversity due to habitat fragmentation is not assessed yet. Thus, the goal of this study was to include fragmentation effects from land occupation and transformation at both midpoint and endpoint levels in LCIA.

Methods One promising metric, combining the landscape spatial configuration with species characteristics, is the metapopulation capacity λ , which can be used to rank landscapes in terms of their capacity to support viable populations spatially structured. A methodology to derive worldwide regionalised fragmentation indexes based on λ was used and combined with the Species Fragmented-Area Relationship (SFAR), which relies on λ to assess a species loss due to fragmentation. We adapted both developments to assess fragmentation impacts due to land occupation and transformation at both midpoint and endpoint levels in LCIA. An application to sugarcane production occurring in different geographical areas, more or less sensitive to land fragmentation, was performed.

Results and discussion The comparison to other existing LCIA indicators highlighted its great potential for complementing current assessments through fragmentation effect inclusion. Last, both models were discussed through the evaluation grid used by the UNEP-SETAC land use LCIA working group for biodiversity impact assessment models.

Conclusions Midpoint and endpoint characterisation factors were successfully developed to include the impacts of habitat fragmentation on species in LCIA. For now, they are provided for bird species in all forest ecoregions belonging to the biodiversity hotspots. Further work is required to develop characterisation factors for all taxa and all terrestrial ecoregions.

Keywords Biodiversity loss · Habitat fragmentation · LCIA characterisation factors · Metapopulation capacity · Species-fragmented area relationship (SFAR) · Sugarcane production

1 Introduction

Life Cycle Assessment (LCA) is a standardised and globally applied decision-support tool used to assess the environmental

impacts of a product or service according to both a life cycle and a multi-criteria perspective. Over decades, LCIA methods have been developed to cover many different impact categories (e.g. climate change, toxicity, or eutrophication) and to convert each inventory flow (i.e. emissions or resources) into a potential impact according to relevant environmental mechanisms (Hauschild et al. 2013). Among them, many assess the land use impacts on biodiversity (Curran et al. 2011; Souza et al. 2015), because alongside climate change or pollution, habitat change and fragmentation were identified by the Millennium Ecosystem Assessment as one of the five direct drivers of biodiversity loss (Millennium Ecosystem Assessment 2005). These impacts can occur from two types of human intervention, i.e. (1) occupation, the use of a certain land area during a given duration of time, and (2)

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transformation, a land area change for a new category of land cover (Milà i Canals et al. 2007). It is common practice that these human interventions can lead to two kinds of impacts on habitat structure, i.e. (1) physical changes to the vegetation cover and land structure and (2) physical changes to soil (e.g. erosion). According to the impact pathway framework for land use impacts on biodiversity, these impacts on vegetation cover and soil can in turn affect ecosystems and lead to damages on ecosystem quality. Some LCIA characterisation models rely on land use flow inventory indicators (e.g. ReCiPe 2008 midpoint (Goedkoop et al. 2013), with inventory flows aggregated into three classes), whereas others are modelling environmental mechanisms, providing ecosystem (Michelsen 2008; Coelho and Michelsen 2013) or species impact indicators (e.g. species composition (de Baan et al. 2013a, b; Curran 2014; Chaudhary et al. 2015), or functional diversity (de Souza et al. 2013), among others). To assess the local and regional land use impacts on species richness, most of them rely on species area relationship (SAR) models that establish a relationship between a habitat area and the number of species found within that area (Teixeira 2014; de Souza et al. 2015). De Schryver et al. (2010) used the classic SAR (Rosenzweig 1995), whereas de Baan et al. (2013b) and Chaudhary et al. (2015) rely on enhanced SAR models to take into account habitat heterogeneity with matrix effects and species adaptation to human converted habitats, respectively, the Matrix SAR (Koh and Ghazoul 2010) and the Countryside SAR (Pereira and Daily 2006). Nonetheless, the UNEP-SETAC land use working group recently pointed out that habitat composition and configuration, including fragmentation, seemed to be still under-represented in impact assessment models and should be considered, particularly when dealing with SARs (Teixeira et al. 2016). Although several land use impact pathway frameworks propose to include fragmentation as a human intervention leading to biodiversity loss (Koellner et al. 2013), there is no characterisation model that represents this issue. In other words, if we consider two area losses of equal size, one with little and one with much habitat fragmentation, the assessed impacts on biodiversity will be the same with available characterisation models, which could prove drastically inaccurate (Chaplin-Kramer et al. 2015). There has been one attempt to include fragmentation into the Life Cycle Inventory (LCI). Jordaan et al. (2009) introduced the fragmentation concept in the LCI by considering an additional buffer zone, leading to a larger area than the direct land use area to consider in the LCI. The problem with this proposal is that it artificially inflates the land use area in the LCI but does not represent the landscape fragmentation effects on biodiversity.

Outside the LCA field, in ecology and conservation biology, thousands of studies focused on showing the impacts of patch area, edge effects, patch shape complexity, isolation, and landscape matrix contrast on population density, species

richness, community dynamics, and ecosystem functioning (Didham 2010). However, even if fragmentation is commonly defined as a process in which “a large expanse of habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original” (Wilcove et al. 1986), the concept has been, and still is, extensively debated. During the 1970s and the 1980s, Simberloff and Abele (1976) raised the SLOSS (Single Large Or Several Small) debate that questioned if one large reserve was preferable to several smaller ones of the same total area for conservational strategies. More recently, the discussion has focused on the breaking apart (also called “fragmentation per se”) relative importance on species distribution and abundance compared to habitat loss. Although theoretical studies tend to predict a strong negative fragmentation effect, empirical studies are more contrasted. This has led to two opposite trends in the ecology community. Some authors suggest that fragmentation effects on biodiversity are negligible compared to the change in total habitat area effects, i.e. the “Habitat Amount Hypothesis” (Fahrig 2013), whereas other authors support the idea that the spatial configuration and land use patterns, e.g. agricultural expansion (Chaplin-Kramer et al. 2015), can have dramatic impacts on the biodiversity loss rate (Hanski 2015). In particular, the fragmentation consideration is strongly defended by authors that promote the extinction debt existence (Kuussaari et al. 2009). Even if in some cases the immediate impacts may seem small, fragmentation can also cause time-delayed biodiversity loss at different trophic levels (Krauss et al. 2010; Alofs et al. 2014). The main problem appears to lie in the intimate link between habitat loss and breaking apart separated processes. In reality, the loss is rarely simply a removal of contiguous areas and is always accompanied by more or less fragmentation (Whitmore and Sayer 1992). Hence, it may not be possible to isolate the “fragmentation per se” effect from habitat loss (Fahrig 2003; Didham 2010). In line with this, some authors recently discussed the question of habitat loss and habitat fragmentation relationship (Didham 2010) and interdependence among the various sub-components of habitat fragmentation (Didham et al. 2012). To overcome this problem, recent approaches propose to discuss *where* habitat configuration should matter. These approaches intend to determine for which remaining available habitat configuration effects are most likely to be observed (Villard and Metzger 2014). Several studies suggest that fragmentation effects should not be ignored when remaining available habitat falls below a critical threshold, generally ranging from 20 to 30% depending on authors (Andren 1994; Schneider 2001; Flather and Bevers 2002; Pardini et al. 2010; Hanski 2015). Given this threshold assumption, fragmentation impacts could be underestimated in highly converted landscapes, especially with current SAR models (Whitmore and Sayer 1992; Fattorini and Borges 2012; Haddad et al. 2017). With this in mind, Hanski et al. (2013) and Rybicki and

Hanski (2013) recently proposed to adapt the SAR in order to include fragmentation effects along with area loss. They defined the species-fragmented area relationship (SFAR) that uses the metapopulation capacity λ to describe the landscape fragmentation degree (Hanski and Ovaskainen 2000). This model has been identified as a promising approach to account for habitat fragmentation in LCIA (Teixeira et al. 2016) and a first approach based on the metapopulation capacity to derive worldwide forest fragmentation potential (FFP) indicators at the ecoregion scale has been proposed by Larrey-Lassalle et al. (2018).

The goal of this study was to adapt the FFP indicators and the SFAR model in order to assess fragmentation effects from land occupation and transformation at midpoint and endpoint level in the LCIA phase. Aiming (1) to explore the feasibility of including these models in LCIA and (2) to discuss their interest and added value compared to existing models, the objectives were as follows: (1) to develop midpoint and endpoint characterisation factors (CFs) for regional fragmentation impacts on biodiversity using the metapopulation capacity concept and the SFAR model; (2) to apply the CFs to a case study of sugarcane production occurring in geographical areas with different sensitivities to land fragmentation, in order to test the capability of the new indicators to provide additional, complementary information to land use LCIA results; and (3) to discuss the current challenges to fully integrate fragmentation into LCIA methods, as well as future research needs for this important issue.

2 Materials and methods

2.1 Using the metapopulation capacity λ to derive an LCIA midpoint indicator for habitat fragmentation

Hanski and Ovaskainen (2000) defined the metapopulation as a group of spatially separated populations of the same species interconnected by dispersal. In particular, the metapopulation capacity λ measures how a given landscape spatial configuration contributes to the long-term persistence of a particular species structured as metapopulation (Hanski and Ovaskainen 2000). Formally, in the single-species metapopulation theory, the metapopulation capacity λ depends on landscape structure (i.e. number of fragments, fragment areas, distances between fragments) and species characteristics (i.e. dispersal distance, and extinction, immigration, and emigration rates). For more details on the calculation see Online Resource 1, section 1-A of the Electronic supplementary material.

Schnell et al. (2013) recently adapted the original metapopulation model by adding a self-colonisation component that considers the potential of survivors in large patches to repopulate the rest of the patches after an extinction event. Building on that, Larrey-Lassalle et al. (2018) developed a

methodology to derive worldwide regionalised fragmentation indexes based on the metapopulation capacity with self-colonisation λ_{self} and applied it to all forest ecoregions in the *biodiversity hotspots*. In accordance with their definition, their forest fragmentation potential $FFP_{g,j}$ indexes could be directly used as CFs to assess the decrease in “land quality” defined by Milà i Canals et al. (2007) as the difference of an indicator of land quality from a reference state to a given land use i . Consequently, the fragmentation impact $FI_{\text{occ},g,i,j}$ for taxon g due to land occupation by land use i ($A_{\text{occ},i}$ during $t_{\text{occ},i}$) in ecoregion j (in potential fragmented area · year) is calculated as follows (Eq. (1)):

$$FI_{\text{occ},g,i,j} = A_{\text{occ},i} \times t_{\text{occ},i} \times CF_{\text{occ},g,i,j} \text{ with } CF_{\text{occ},g,i,j} = FFP_{g,j}. \quad (1)$$

If $FFP_{g,j}$ is null in ecoregion j , a 1 m² use will correspond to 0 m² having a potential impact on species persistence due to fragmentation, i.e. no additional impact due to fragmentation. Conversely, in an ecoregion with a high fragmentation potential, e.g. $FFP_{g,j} = 0.9$, a 1 m² use will correspond to 0.9 m² having a potential impact on species persistence due to fragmentation. In addition, the metapopulation model for the fragmentation potential does not differentiate between land use types as it only looks at remaining original habitat (here, forest) in a dichotomous classification (i.e. forest/no forest), regardless of species affinity to the matrix surrounding the native forest fragments. Thus, the fragmentation indexes $FFP_{g,j}$ are the same for any land use type i . Finally, the level of spatial aggregation for the fragmentation index $FFP_{g,j}$ can be adapted to the spatial resolution of the LCI (see Table S1 in the Electronic supplementary material, Online Resource 1).

To determine the forest transformation fragmentation impacts, the current approach in LCIA (Milà i Canals et al. 2007) is to assume a linear and full recovery, by multiplying the occupation impact by half the regeneration time, i.e. the time needed for the transformed area to restore to a land use type with a similar diversity as the reference (Curran et al. 2014). The fragmentation impact $FI_{\text{trans},g,i,j}$ for taxon g due to original forest transformation ($A_{\text{trans},i}$) in ecoregion j (in potential fragmented area · year) for all land use types i is given by Eq. (2):

$$FI_{\text{trans},g,i,j} = A_{\text{trans},i} \times CF_{\text{trans},g,i,j} \text{ with } CF_{\text{trans},g,i,j} = 0.5 \times t_{\text{reg},i} \times FFP_{g,j}. \quad (2)$$

For transformation impacts, as we only consider the changes to original habitat (here, native forest), the transformation of any land use type but native forest in ecoregion j will not

generate any additional fragmentation impact. Conversely, the original forest transformation will have a fragmentation impact, more or less important depending on whether the land transformation occurs in an already highly converted forest zone or a preserved forest zone.

Finally, the midpoint indicator $FI_{g,i,j}$, for the habitat fragmentation impact due to land use in LCIA is (Eq. (3)):

$$FI_{g,i,j} = FI_{occ,g,i,j} + FI_{trans,g,i,j} \tag{3}$$

2.2 Using the SFAR model to derive an LCIA endpoint indicator for land use impacts on biodiversity including fragmentation effects

The SFAR model (Hanski et al. 2013) predicts the number of species S in the reduced and fragmented natural area A , where c , z , and b are fitted constants, and $P(\lambda)$ is the fraction of species that are expected to persist when the degree of fragmentation of A is given by λ (Eq. (4)).

$$S = c \times A^z \times P(\lambda), \text{ with } P(\lambda) = e^{-\frac{b}{\lambda}} \tag{4}$$

The approach proposed by de Baan et al. (2013b) and adapted by Chaudhary et al. (2015) was used to calculate the SFAR-based CFs. The predicted species loss of taxon g due to cumulative land use in region j , including fragmentation effects $S_{lost,g,j,reg}$ is given by:

$$S_{lost,g,j,reg} = S_{org,g,j} - S_{new,g,j} \\ = S_{org,g,j} \left[1 - \left(\frac{A_{new,j}}{A_{org,j}} \right)^{z_{j,SFAR}} \times P(\lambda_{new,g,j}) \right] \tag{5}$$

where $S_{org,g,j}$ is the number of species in the original continuous natural area $A_{org,j}$, $S_{new,j}$ the new number of species in the remaining reduced and fragmented natural area $A_{new,j}$ with fragmentation degree $\lambda_{new,g,j}$ for taxon g , and $z_{j,SFAR}$ the constant rate at which species accumulate with increasing area.

$$P(\lambda_{new,g,j}) = e^{-\frac{b_g}{\lambda_{new,g,j}}} = e^{-b_g^* \times \frac{\lambda_{new,g,j}}{\lambda_{new,g,j}}} \tag{6}$$

where b is the sensitivity of taxon g to fragmentation, and b_g^* is the sensitivity of taxon g to fragmentation, independent of the unit and calculation choices of λ (see methodology in Online Resource 1, section I-C). $\lambda_{new,g,j}$ is the harmonic mean of $\lambda_{new,g,j}$ of all ecoregions.

Chaudhary et al. (2015) proposed average and marginal CFs. The marginal damage function was obtained by the first derivative of the average damage function S_{lost} by the area lost (i.e. $\frac{\partial S_{lost}}{\partial A_{lost}}$). As $\lambda_{new,g,j}$ is not a constant, because it depends on the remaining original habitat area $A_{new,j}$ and the spatial configuration in the ecoregion j , the derivative cannot be easily

obtained. Thus, we focused on average CFs representing regional species lost per square meter (Eq. (7)).

$$CF_{average,occ,reg,g,i,j} = \frac{S_{lost,g,j} \times a_{i,j}}{A_{i,j}}, \text{ with } : A_{i,j} \\ = p_{i,j} \times A_{lost,j} \tag{7}$$

where $a_{i,j}$ is the species loss allocation to the land conversion in the ecoregion j , which is based on the current areas $A_{i,j}$ of land use types i in ecoregion j , their relative area share $p_{i,j}$ in the total converted land area A_{lost} and the taxon g sensitivity to each land use type i (details in section 1-D, in Online Resource 1 of the Electronic supplementary material).

Regional CFs for land transformation were calculated by multiplying $CF_{reg,occ,g,i,j}$ with half the regeneration time (Eq. (8)). The unit of CF_{trans} is regional species lost-year per square meter of transformed land.

$$CF_{trans,reg,g,i,j} = 0.5 \times t_{reg,g,i,j} \times CF_{occ,reg,g,i,j} \tag{8}$$

The links between land occupation and transformation LCIA flows and endpoint CFs can be found in Table S2 in Online Resource 1 of the Electronic supplementary material.

The SFAR-based CFs were compared with CFs based on the classic SAR model and the countryside SAR model (Chaudhary et al. 2015), which is the current model proposed as interim recommendation for the CF calculation (for application in hotspot analyses in LCA) by the UNEP-SETAC LCIA Guidelines (UNEP SETAC Life Cycle Initiative 2016). An alternative CF, based on a ‘‘countryside SFAR’’ (a blend between countryside SAR and SFAR) is proposed to combine both positive countryside effects (i.e. species adaptation and survival in human-modified habitats) and negative fragmentation effects. Theoretical equations used to calculate S_{lost} based on four SAR models (classic SAR, countryside SAR, SFAR, and countryside-SFAR) are summarised in Table S3 in Online Resource 1 of the Electronic supplementary material.

2.3 Evaluation of CFs

The models proposed at midpoint and endpoint level were evaluated against the five evaluation categories for LCIA models and indicators used by the European Commission (2010) and adapted by the UNEP-SETAC Life Cycle Initiative for biodiversity impact assessment models (Curran et al. 2016), i.e. completeness of scope, biodiversity representation, impact pathway coverage, scientific quality, and stakeholder acceptance. Completeness of scope and scientific quality are presented in the ‘‘Discussion’’ section while the three other criteria are discussed in Online Resource 1. This analysis facilitates the comparison between the proposed indicators and other existing approaches used to assess land use impacts on biodiversity in LCIA.

2.4 Case study

The different CFs developed above are applied to a case study to discuss their applicability and interest for supporting decision-making. Sugarcane has been chosen because large amounts are produced each year all around the world, about 210 million metric t/year, for a total area occupation of around 29 million ha (FAO 2014). Out of more than 100 sugarcane-producing countries, only 16 provide 90% of the global production, and in 2005 seven already devoted more than 50% of their land area to cane cultivation (WWF 2005). To that end, considerable, biodiversity-rich habitat areas, such as tropical rain forests, have been cleared. Since sugarcane production is still expected to increase by 10% in the next 10 years (FAO 2016), concerns are raised on land expansion needed to meet future sugarcane demand (Marin et al. 2016). As land use impacts will strongly depend on the region in the world where it is grown, we assess as a case study the land fragmentation impacts of 1 kg of sugarcane produced in three contrasted parts of the world, i.e. in the *Neotropical*, *Indomalayan*, and *Australasian* biogeographic realms, respectively, in Brazil, Indonesia and Thailand, and Australia. Brazil is the largest world sugarcane producer with around 37% of global production in 2014, while Thailand, Australia, and Indonesia rank respectively fourth, eighth, and ninth globally, with around 5.2, 1.5, and 1.4% of global production (Table S4 in Online Resource 1 of the Electronic supplementary material). The prevailing geographic zones for sugar cane growing were identified for each country, and corresponding ecoregions were selected (Table S5 in Online Resource 1 of the Electronic supplementary material). The LCA food database Agri-footprint® (version 2.0, October 2015) was used for the LCIs of the four scenarios (Blonk Consultants 2015a). Land use and land use change estimations were based on FAO statistics and IPCC calculation rules, following the PAS 2050-1 methodology (Blonk Consultants 2015b). Details on land use inventories are available in Table S6 in Online Resource 1 of the Electronic supplementary material. The input parameters for midpoint CFs and endpoint CFs calculated for these ecoregions are described in Online Resource 1 (respectively in sections 2-C and 2-D), and resulting CFs are reported in excel file Online Resource 3 of the Electronic supplementary material.

3 Results

3.1 An additional midpoint indicator to consider fragmentation effects

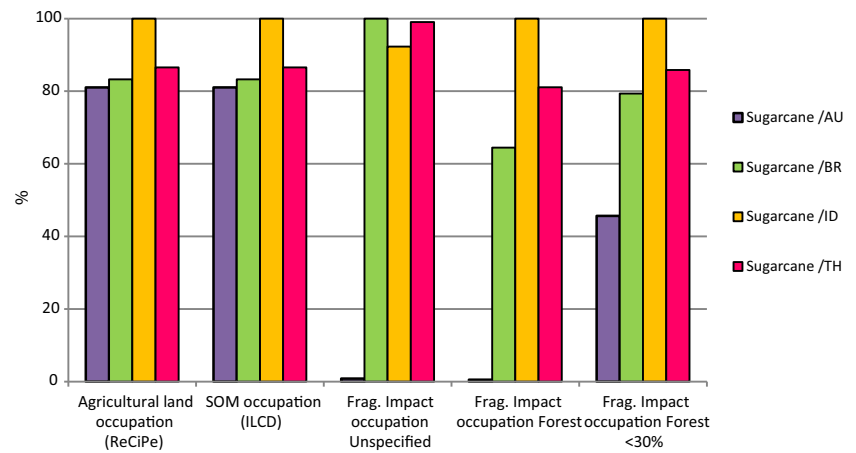
The proposed midpoint indicator for fragmentation effects is no substitute for other midpoints and has to be considered as additional information to current indicators.

The results are compared with (1) ReCiPe midpoint indicators and (2) ILCD midpoint indicators for land use. ReCiPe midpoint indicators are LCI flows aggregated into three classes: *agricultural land occupation*, *urban land occupation*, and *natural land transformation*. Sugarcane processes in the Agri-footprint database exclude infrastructure processes; thus, there is no impact on urban land occupation for such agricultural products. The ILCD midpoint indicator is based on changes in soil organic matter (SOM) (Milà i Canals et al. 2007). Note that the soil organic carbon (SOC) indicator (Brandão and Milà i Canals 2013) could be an interesting alternative to the SOM indicator because the CFs based on SOC are spatially differentiated at a climate region level and provided for ten climate regions worldwide.

The land use flow for occupation is 1 ha for all countries (Table S5 in Online Resource 1, Electronic supplementary material). The slight differences between countries for the ReCiPe indicator (first indicator in Fig. 1) are linked to the yields (Table S5 in the Electronic supplementary material). The smaller the yield, the greater are the impacts for 1-kg sugarcane. Indonesian sugarcane production has the highest impacts, followed by Thai, Brazilian, and Australian sugarcane. The maximum difference between countries is around 20%. The SOM indicator (second indicator in Fig. 1) gives the exact same ranking for countries as the ReCiPe indicator, because the same CF is applied to the LCI occupation flow irrespective of the sugarcane cultivation location. The ranking is also the same when assessing the fragmentation effects due to land occupation when spatial information is known. However, results are much more contrasted, showing the interest of this new indicator to discriminate different alternatives. It is particularly true for Australia, where fragmentation impacts are null even if the occupation surface is the same as for the other countries, except when the occupation takes place in highly converted forest areas. For occupation impacts in an unspecified location within the ecoregion (third indicator in Fig. 1), the ranking between countries slightly changes compared to other indicators. Brazilian sugarcane is causing most fragmentation impacts, followed by Thai and Indonesian sugarcane, but differences range from 1 to 8% for these three countries. Australian sugarcane has almost no fragmentation impact. At the ecoregion scale, the indicator seems to be more dominated by the total forest share, i.e. the forest areas/non-forest areas ratio, than by the fragmentation degree itself. These results highlight the weakness of this site-generic indicator and the interest to consider spatial variability in impact assessment whenever relevant.

For transformation impacts, the LCI flows vary considerably from one country to another, e.g. transformed forest area per sugarcane hectare is 100 times higher for Brazil than for Australia (see Table S6 in Online Resource 1 of the Electronic supplementary material). Consequently, even if in theory CFs distinguish well between countries, in this case, the

Fig. 1 Midpoint land use impacts due to occupation (ReCiPe, ILCD, and fragmentation impact for birds) associated with 1 kg of sugarcane production in four countries (AU: Australia; BR: Brazil; ID: Indonesia; TH: Thailand)



transformation results are mainly driven by the LCI flows (Fig. S2 in Online Resource 1 of the Electronic supplementary material). Nevertheless, the fragmentation indicator better balances the results and the differences become less pronounced when country fragmentation impacts are taken into account, e.g. from 74% with LCI only to 50 or 59% difference between Indonesia and Brazil, depending on the transformed forest cover.

In conclusion, the fragmentation indicator provides valuable additional information to the current land use midpoint indicators and allows for better differentiation between the four compared scenarios.

3.2 Improving the environmental relevance of current endpoint indicators with fragmentation impacts on biodiversity

The proposed SFAR-based endpoint indicators add the fragmentation effects to the SAR models. As shown in Fig. 2, for all ecoregions, SFAR-based CFs are more conservative (negative fragmentation effect) than countryside-based CFs (positive countryside effect). Moreover, for the ecoregions considered, the countryside effect contribution is higher than the fragmentation effect contribution. Nevertheless, it is important to consider that the fragmentation effect computation at the ecoregion scale tends to hide the high spatial variability throughout the ecoregion and smoothes the results. Fragmentation effects can be very important locally, e.g. there is almost no fragmentation effect for Australia at the ecoregion scale, even if fragmentation effects can be considerable in Australian most converted forest areas. CFs for smaller spatial scales (e.g. ecoregion subdivisions) might indicate higher species loss, but as of now, that is dependent on data availability on species and on land use occupation at such scales. Nevertheless, current results show great potential for combining both countryside and fragmentation

effects, and the use of countryside-SFAR based CFs could already substantially improve the comprehensiveness and environmental relevance of land use LCIA. The same conclusions can be drawn for transformation CFs, as they are directly proportional to occupation CFs (see Eq. (8)).

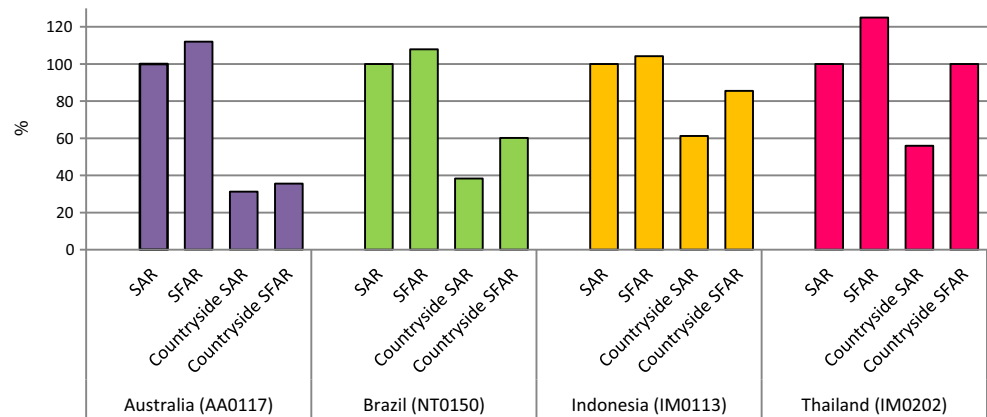
Regarding the sugarcane case study results shown in Fig. 3, damage on biodiversity is higher when considering fragmentation using the countryside SFAR for Indonesia, Thailand, and Brazil, while the results remain almost similar for Australia. For damage due to land occupation, the overall ranking changes between countries, i.e. Indonesian sugarcane production has the greatest impact on biodiversity, whereas Australian production has the higher impact without considering fragmentation effects. More generally, taking fragmentation effects into account, which have a high variability between (and within) countries, allows for a better differentiation between the four scenarios when assessing impacts on biodiversity incurred by land use.

3.3 Toward practical integration of the endpoint indicator into current LCIA methods

The additional term $P(\lambda)$ in the SFAR model is provided for all forest ecoregions included in the biodiversity hotspots (Fig. 4 and excel file Online Resource 4, Electronic supplementary material).

At the ecoregion scale, i.e. using λ values aggregated for the ecoregion, even if $P(\lambda)$ frequently approaches 1 (pale yellow in Fig. 4), fragmentation hotspots can be still be located (deep red in Fig. 4). The same taxon sensitivity to fragmentation was used for all ecoregions, but if specific values were available, differences between ecoregions would be even more pronounced. Nevertheless, the new distribution (i.e. negative exponential function) already gives different distributions compared with metapopulation capacity λ maps (Larrey-Lassalle et al. 2018).

Fig. 2 Regional land occupation CFs calculated using four models (classic SAR, SFAR, countryside SAR, and countryside SFAR), expressed as a percentage of the classic SAR-CFs, for birds and the land use type *permanent crop* in four contrasted ecoregions



4 Discussion

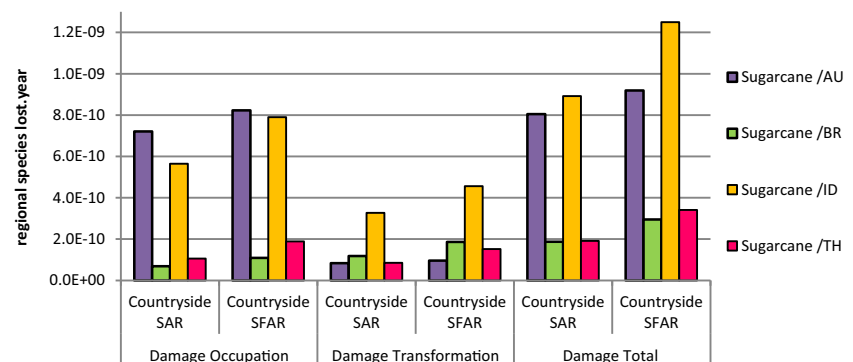
4.1 Completeness of scope

For the midpoint fragmentation indicator, land use classes are all covered by the occupation CF. Since the model deals with the spatial arrangement of original habitat remnants, it is not intended to differentiate land use types and intensities, i.e. what replaces original habitat. The *FFP* indexes are provided for all biogeographic realms except Antarctic (i.e. Afrotropic, Australasia, Indo-Malay, Neotropic, Nearctic, Oceania, and Palearctic), with a focus on forest biodiversity hotspots (i.e. 283 out of 814 terrestrial ecoregions). The spatial resolutions of assessment are the sub-ecoregion (more or less converted forest areas within the ecoregion) and the ecoregion. Based on data compiled by de Baan et al. (2013c) and Chaudhary et al. (2015), 482 ecoregions have a remaining original habitat cover smaller than 30% (directly derived from the ratio $\frac{A_{new}}{A_{org}}$ for all ecoregions). Two hundred ninety-two of them are forest ecoregions; thus, these particular ecoregions seem to be well-covered by this preliminary assessment. Yet, there is room for improvement to include the 190 other non-forest ecoregions where less than 30% of the original habitat remains, e.g. grasslands, savannas, or mangroves, which also host considerable species diversity. Furthermore, some ecoregions can have a total remaining original cover higher

than 30% at the ecoregion scale but still contain specific zones highly sensitive to fragmentation, which can be captured by the different levels of spatial aggregation defined for the fragmentation indexes. The landscape analysis scale (i.e. 100-km² landscapes) would make it possible to implement the indicator at smaller spatial resolutions. For now, the taxonomic coverage is one taxon (birds), but the methodology could easily be applied to other taxa (e.g. mammals, reptiles, amphibians, and vascular plants) based on their dispersal abilities used for calculating λ . The choice of a dispersal distance representative of a whole taxon is crucial. One simplistic approach is to take a median dispersal distance based on those available in literature (as we did here for birds). A more elaborated approach involves the selection of umbrella species, i.e. species assumed to properly represent the ecosystem in which they live (Baguette et al. 2013). Finally, the most conservative approach consists in favouring specialist species with low dispersal rates which are expected to be more affected by changes in landscape connectivity than generalist or highly mobile species (Soga and Koike 2013; Matthews et al. 2014).

For the endpoint fragmentation indicator, as the proposed indicator is an extension to already existing countryside SAR-based CFs, the coverage is the same as Chaudhary et al. (2015), i.e. six land use types and difference in intensity for forestry only. The damage to biodiversity assessment is also carried out at the ecoregion level. Intended as a proof of

Fig. 3 Biodiversity impacts on birds due to land use (occupation, transformation, and total) associated with 1 kg of sugarcane production in four countries (AU: Australia; BR: Brazil; ID: Indonesia; TH: Thailand), calculated using the countryside model, including fragmentation impacts (countryside SFAR) or not (countryside SAR)



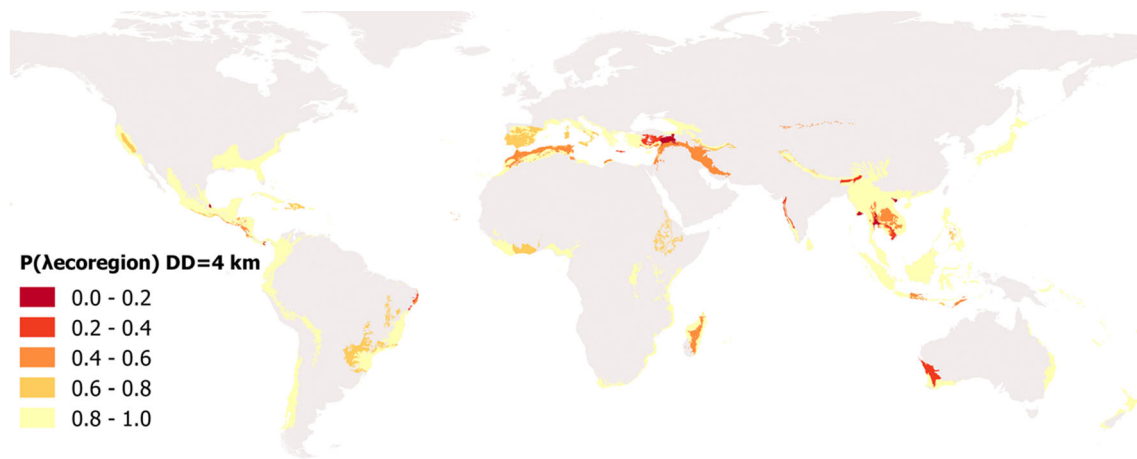


Fig. 4 Fragmentation term $P(\lambda)$ in the SFAR model for all forest ecoregions included in the biodiversity hotspots (coloured areas) calculated for a birds dispersal distance (DD) of 4 km

concept, the CF calculation has been conducted on four ecoregions, but further development would involve widening the scope of application to more ecoregions and to lower spatial resolutions (e.g. sub-ecoregions). Likewise, the feasibility of calculating and applying the new indicator has been demonstrated on one taxon (birds), but the methodology could be applied to other taxa, based on species dispersal distances (affecting λ calculation) and on species sensitivity to fragmentation (b parameter). The latter could be estimated using several trait combinations (i.e. population size, population fluctuation and storage effects, disturbance and competition sensitive traits, habitat specialisation and matrix use, rarity, and biogeographic position) as predictors/proxies of species sensitivity to fragmentation (Henle et al. 2004).

4.2 Scientific quality

Regarding scientific robustness, the midpoint model is a *bottom-up approach* linking the habitat configuration to the species persistence in a fragmented landscape based on explicit metapopulation models. The endpoint model uses the SFAR model, which is, as all other SAR models, a *top-down approach* based on observed empirical data, which links area (and landscape configuration for the SFAR) and species richness. The interest of the metapopulation theory for studying the landscape fragmentation impacts on species is widely recognised in ecology. Whenever possible, the most recent data available were used, e.g. Globcover 2009 maps used for landscape analysis, are among the most detailed, reliable, and up-to-date global land cover maps (Bontemps et al. 2011). However, as landscape structure is continually evolving, the λ calculation should be performed on updated maps as soon as they become available. Regarding primary data underpinning the SFAR model, there is room for improvement. More observed data should be gathered to conduct a meta-analysis in fragmented landscapes exclusively and obtain robust z and b

parameters, which are currently derived from a specific empirical dataset. Several values should be provided for different types of ecoregions and taxa. Furthermore, the proposed countryside SFAR model only partially considers the fact that some species can survive in non-original habitats (i.e. more “remaining habitat area” in the calculation of S_{lost}). For completeness, this survival should also be taken into account in the spatial analysis conducted to calculate λ which is then used to calculate the countryside SFAR CFs. Including the sensitivity of each taxon to these new land use types in the connectivity analysis raises some methodological challenges, in particular regarding the additional spatial treatment necessary to consider other land cover classes as new habitat for each taxon in each ecoregion, or to qualify the matrix between the habitat fragments (e.g. by incorporating a matrix resistance parameter which is expected to vary among species (Ricketts 2001), and which will artificially weigh the distances between the fragments). Another important perspective following this first investigation of land fragmentation in LCA would be the assessment of uncertainty bounds. This seems feasible in the long term for the factor λ in the light of its sensitivity analysis to input parameters discussed by Larrey-Lassalle et al. (2018) using also the bird dispersal distance variability available in Online Resource 1 (Fig. S1, Electronic supplementary material). On the other hand, the question of quantifying the uncertainty of the other parameters used for the endpoint CFs calculation is much more difficult. While some parameters ($S_{\text{org}, g, j}$, $A_{\text{new}, j}$, $A_{\text{org}, j}$, $a_{i, j}$, $P_{i, j}$, $CF_{\text{loc}, g, i, j}$, $z_{\text{org}, j}$) and their associated uncertainties are provided by (Chaudhary et al. 2015), the specific SFAR key parameters $z_{\text{new}, j}$ and b_g^* have been calibrated on experimental observations available for a single dataset. Therefore, the challenge to perform a quantitative uncertainty analysis based on the variability of those parameters is related to the availability of multiple field observation datasets. Moreover, major uncertainties are associated to species richness data, i.e. numbers of species on Earth today

(S_{new} , in A_{new}) and before human land use changes (S_{org} , in A_{org}), which are crucial inputs to assess species loss due to human activities. The use of the WildFinder Database (World Wildlife Fund 2006) to estimate the original number of species S_{org} is debatable. This database gives a global animal species distribution based on the WWF terrestrial ecoregion maps. Species presence/absence data from WWF are based on the ranges of *extant* species and exclude globally extinct species. When available, historic ranges of species (i.e. their approximate distribution 500 years ago) were used instead of current distributions, but such maps were only available for approximately 200 species out of more than 26,000. Generally, the time gap between the rise of modern anthropogenic pressures on biodiversity and most of available biodiversity data, which started to be collected in the second half of the twentieth century, has been identified as a major cause of underestimation of impacts (Mihoub et al. 2017). In addition and more generally, even if the SFAR model is strengthened and the absolute uncertainty of the fragmentation CF is known, this uncertainty represents the contribution of the parameters of this CF to the total CF's or the impact score's overall parametric uncertainty, but it will not capture the fact that the impact score is more complete (and therefore more representative for the modelled environmental problem) if habitat surface and configuration are considered together. Including fragmentation may decrease the unquantifiable uncertainty of relevance or representativeness of the result, while it may increase parametric uncertainty in cases where fragmentation impacts dominate the land-use biodiversity impact score. This bias should always be kept in mind when analysing the uncertainty of the CF based on the parametric uncertainty only. Lastly, the fragmentation indexes used for the midpoint indicator and the SFAR model used for the endpoint indicator are well documented (Hanski et al. 2013; Larrey-Lassalle et al. 2018). Model assumptions and data sources are transparently reported in SI. The matrix calculation program (Python file) and details on GIS processing (Larrey-Lassalle et al. 2018) are also available upon request, to ensure the study reproducibility and to encourage new CF calculation.

5 Conclusions and perspectives

New methodological developments have been proposed to include the impacts of habitat fragmentation on species in LCIA in order to fill an important gap in the LCA land use framework. The resulting CFs quantified at both midpoint and endpoint levels provide additional information to existing LCIA indicators for land use impacts on biodiversity. The case study results show that land use impacts in LCA are currently underestimated in fragmented areas, and an endpoint indicator combining habitat loss effects and habitat fragmentation

effects (i.e., based on the countryside SFAR) provides a more comprehensive assessment.

However, further developments are required to improve the spatial and ecological coverage of these new indicators. For now, the CFs are provided for all forest ecoregions included in biodiversity hotspots and for birds species only. More data and additional methodological developments should be proposed to apply these indicators to all ecoregions and for a wider set of taxa. Furthermore, in the same way as for current countryside-SAR based CFs, the SFAR-based CFs operationalisation will also depend on LCA software progress in terms of regionalised LCI (i.e. inventory flows at the ecoregion or sub-ecoregion scales) and LCIA modelling.

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