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RESEARCH ARTICLE

A practical approach to measuring the biodiversity impacts of land conversion

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Abstract

1. Further progress in reducing biodiversity loss relies on the improved quantification of the connections between drivers of habitat loss and subsequent biodiversity impacts. To this end, biodiversity impact metrics should be able to report linked trends in specific human activities and changes in biodiversity state, accounting for both the ecology of different species and the cumulative effects of historical habitat losses. These characteristics are not currently captured within a single metric.

2. Here, we develop a globally applicable methodological framework that uses freely and publicly available datasets to quantify the relative impacts of anthropogenic activities on biodiversity. We use species-specific habitat suitability models to link specific land uses to ensuing changes in the likelihood that local populations of those species will persist. To illustrate our approach, we assess the impacts of soy expansion and other land uses within the Brazilian Cerrado on over 2,000 species of amphibians, birds, mammals and plants for three periods between 2000 and 2014.

3. Our results showed that mammals and plants suffered the greatest overall reduction of suitable habitat. However, among endemic and near-endemic species— which face greatest risk of global extinction from habitat conversion in the Cerrado—birds and mammals were the most affected groups. While conversion of natural vegetation to grassland and planted pastures were together responsible for most of the biodiversity impact of recent changes, soy expansion (via direct conversion of natural vegetation) had the greatest impact per unit area. The total biodiversity impact of recent land-use change was concentrated in the southern

states of the Cerrado—Minas Geráis, Goiás and Mato Grosso—but the impact on biodiversity of production of soy was greatest within the agricultural frontiers of Bahía and Piauí.

4. The flexibility of our approach to examine linkages between biodiversity loss and specific human activities has clear potential to better characterize the pathways by which habitat loss drivers operate. Its capacity to incorporate species-specific ecological needs, through a globally applicable methodology, can improve the tangibility of biodiversity loss assessments.

1 | INTRODUCTION

Metrics that allow the impact of human activities on biodiversity to be monitored and reported are essential tools in global conservation efforts. They have revealed unprecedented rates of species extinctions (Butchart et al., 2010), the extent of local population losses (Hill et al., 2016) and the contribution of different sectors (such as agriculture) towards these impacts (Hoekstra & Wiedmann, 2014; Moran & Kanemoto, 2017). With such evidence, habitat loss from land-use change is now recognized as the biggest contributor to biodiversity decline and this information supports multiple international initiatives for the protection of biodiversity on land and at sea (CBD, 2012; Millennium Ecosystem Assessment, 2005; Ramsar, 1971). Yet, while quantifying the scale of global biodiversity loss remains vital, more work is needed to develop tools for identifying and tracking the drivers of land-use change that underlie such trends.

Attributing biodiversity losses to specific drivers has proved challenging. Actionable information requires that biodiversity impact estimates can be derived at scales at which information on anthropogenic activities and drivers are available, and decisions are made (Guerrero, McAllister, Corcoran, & Wilson, 2013). Biodiversity, however, is a multifaceted construct and difficulties in measuring changes in its state still limit the effective guidance of conservation efforts (Sparks et al., 2011). Most common techniques for measuring biodiversity changes provide an estimate of species richness loss, which poses important shortcomings for biodiversity conservation (Hillebrand et al., 2018). Of particular importance is the inability to predict extinction risk for individual species. Due to the absence of species identity, richness metrics fail to incorporate species-specific information such as their distribution and ecological requirements. The absence of this information limits the ability of the metric to be adapted to different spatial scales (Veach, Di Minin, Pouzols, & Moilanen, 2017). This is required to assess drivers of biodiversity loss as it demonstrates the linkages between changes in the state of biodiversity and specific human activities (Green et al., 2019). Furthermore, conservation decisions rely on metrics that permit the

user to quantitatively differentiate levels of extinction risk among individual species. This requires that the cumulative and nonlinear effects of historical habitat loss are accounted for.

Approaches based on species' area of habitat (AOH; previously known as extent of suitable habitat—ESH; Brooks et al., 2019) show promise in the development of biodiversity impact metrics because they can integrate spatially explicit information on the ecology of individual species with data on the distribution of anthropogenic land use (De Baan et al., 2015; Rondinini et al., 2011). Unlike approaches that estimate potential regional or local loss of species richness, AOH maps are also adaptable to different spatial scales while retaining species-specific information (de Baan, Mutel, Curran, Hellweg, & Koellner, 2013; Rondinini et al., 2011). Specifically, they can quantify the relative change in AOH arising from land conversion, which allows species-specific impacts associated with a particular human land-use change to be calculated (De Baan et al., 2015). AOH is described by the intersection of a species' geographical range with its environmental preferences, measured in terms of variables such as vegetation cover, elevation and proximity to water bodies (Rondinini et al., 2011). AOH is a key determinant of species extinction risk (Blackburn, Gaston, Quinn, Arnold, & Gregory, 1997; Harris & Pimm, 2008), with reductions in AOH affecting the persistence of local populations (Mantyka-Pringle, Martin, & Rhodes, 2012).

An important benefit for applied work of using species-specific AOH in the assessment of biodiversity impact is that it can allow for incorporation of the cumulative, nonlinear effects of habitat loss on species persistence. Impacts of habitat loss are not typically linear because as the area of habitat diminishes the effect of losing each additional hectare of habitat increases (Kitzes & Harte, 2014). Failure to account for this cumulative effect will underestimate the impacts of current habitat loss on species that have suffered historical habitat loss prior to the land-use change in question. This issue has been addressed by a handful of studies that have considered a non-proportional relationship between the extent of remaining habitat and species' persistence (De Baan et al., 2015; Strassburg et al., 2017). However, these have, so far, had limited taxonomic coverage, and use projected land-use changes rather than direct

observations of habitat conversion. They have also been focused at particular spatial scales; given that land-use data and decisions operate across a range of scales, it would be helpful to develop these approaches so that they are applicable across multiple scales.

Here, using a nonlinear and spatially explicit approach, we describe a biodiversity impact metric designed to provide information on changes in local population persistence, which can be both linked to specific human activities and adapted to scales relevant to different datasets and levels of decision-making. We use freely and publicly available datasets to generate AOH models and develop them in four important ways. First, we account for various levels of information on the ecology of individual species, including the estimated importance of breeding and non-breeding ranges. Second, we incorporate historical losses to estimate the cumulative and nonlinear impacts of further losses. Third, we calculate the marginal value of spatial

units of species' habitat so that the impact metric can be adapted across different spatial scales. Fourth, we use crop-specific as well as general land-use maps to estimate the most important drivers of loss. We illustrate the details and power of our approach using the example of the cultivation of a globally important crop, the soybean *Glycine max*, in the Brazilian Cerrado, a globally important savanna that hosts 5% of the world's species (Strassburg et al., 2017). We use species-specific AOH models for 2,009 species from four taxonomic groups: amphibians, birds, mammals and plants.

2 | MATERIALS AND METHODS

Our approach to calculating a biodiversity impact metric involves four main steps (Figure 1): (a) mapping species' AOH at different time

Method stage	Inputs	Analyses	Outputs
1. Mapping species' area of habitat (AOH) at different points in time	- Geographical ranges - Species habitat preferences - Altitudinal preferences - Land-cover and land-use maps for each point in time - Digital elevation model	- For vertebrates, clip geographical ranges by habitat and altitudinal preferences for different points in time, including baseline - For plants, clip geographical ranges by natural land-cover categories	- For each species, a map of AOH for each point in time
2. Calculating proportional loss of baseline AOH, and estimating reduction in species' likelihood of persistence	- For each species, a map of AOH for each point in time	- Calculate proportional loss of AOH by overlaying AOH maps for each point in time against baseline map - Calculate changes in local population persistence (Δp-values) based on remaining proportional AOH (Equations 1,2 and 3)	- Δp-values for each species and time interval (e.g. 2000–2010) - Maps of AOH-losses for each species and point in time
3. Mapping the marginal value of land-use change and its biodiversity impacts across different scales	- Δp-values for each species and time interval. - Maps of AOH-losses for each species and point in time	- Using Δp-values, calculate for every cell within a species' AOH its marginal contribution to that species' continued persistence (MV; Equation 4) - Assign resulting cell-level marginal value of land-use change to AOH-loss maps for each species and point in time - Combine resulting 'biodiversity impact' maps across species - Aggregate combined biodiversity impact map across different scales using administrative boundaries	- Maps of the marginal value of land-use change for each species and point in time - Maps of aggregated biodiversity impact across species and scales
4. Measuring the relative contribution of specific human activities to biodiversity impacts .	- Maps of the marginal value of land-use change for each species and point in time - Maps of aggregated biodiversity impact across species and scales - Land-cover and land-use maps	- Map land-conversion by overlaying corresponding pairs of land-cover maps thus capturing the type of land transition, including soybean expansion - Overlay resulting land conversion maps with species-specific marginal value maps to calculate relative roles for each land-use category in biodiversity impacts	- Estimates of relative biodiversity impacts of land-use change - Maps of biodiversity impacts associated with specific land uses

FIGURE 1 Schematic of method stages specifying input data required, analyses to be performed and resulting outputs

points of interest; (b) calculating the proportional losses in AOH due to land conversion, and estimating the resulting change in likelihood of persistence for each species; (c) mapping these marginal changes in AOH, and aggregating them across species and at different scales and (d) measuring the relative contribution of specific human activities to species impacts. Below we detail these steps for soy production in the Brazilian Cerrado.

2.1 | Mapping species' area of habitat at different points in time

We produced AOH maps for all amphibians, birds and mammals whose geographical ranges intersect the Cerrado boundary (IBGE, 2004), and for which habitat preferences were available. Following Rondinini et al. (2011), we clipped these ranges to exclude unsuitable habitat, based on their habitat preferences as coded against the IUCN habitats classification scheme (IUCN, 2018). For each species, we retained only the habitats listed in level 2 of the scheme that were coded as 'suitable'. To calculate and map AOH for each species, we used national land-cover maps for the years 2000, 2010, 2012 and 2014 (IBGE, 2014; 250-m resolution). These were matched to the IUCN habitat classification scheme following a conversion table that allows the reclassification of habitat preferences into land cover categories ('crosswalk'; see Supporting Information Section 1 and Table S1). Then, using altitudinal preferences from the IUCN Red List and the Shuttle Radar Topography Mission elevation model (USGS, 2006), we excluded land at unsuitable altitudes within species ranges. The elevation map was produced by resampling (averaging) to 250 m the elevation model, originally at 1 arc second resolution (approximately 90 m). For migratory species, AOH was mapped separately for resident, breeding and non-breeding ranges, to reflect seasonal differences in species' habitat requirements. This resulted in a total of 234 AOH maps for amphibians, 846 for birds and 288 for mammals, each at 250-m resolution (the resolution of the best available land cover maps for Brazil with which land-use change can be quantified consistently; IBGE, 2015).

We also calculated AOH for 641 plant species whose ranges intersect the Cerrado. AOH maps for each year were generated by clipping plant geographical ranges (from Martinelli & Moraes, 2013) so they only included those land-cover categories classified as natural by the Brazilian Institute of Geography and Statistics (IBGE, 2014), and excluded other semi- and non-natural categories (Table S1).

For each species, we produced baseline AOH maps against which subsequent proportional losses of AOH were assessed. For the vertebrate groups, we used a map of original vegetation cover for the Brazilian Cerrado, which showed the distribution of natural vegetation prior to large-scale cultivation (c. 16th century; IBGE, 2004). This reclassification was also performed using a conversion table (Table S1) and was restricted to each species' geographical range.

For plants, the entire geographical range intersecting the study region was considered its baseline AOH.

2.2 | Calculating proportional loss of species' AOH, and estimating changes in species' local likelihood of persistence

For each species, we calculated the proportion of its baseline AOH remaining at each subsequent point in time—years 2000, 2010, 2012 and 2014. We translated the change in AOH between years into an associated change in likelihood of persistence. The impact on population persistence of losing a given amount of AOH increases as total AOH decreases, resulting in a concave relationship between remaining AOH and local persistence (Kitzes & Harte, 2014). Although the detailed form of this relationship has yet to be investigated, we followed other studies (Balmford, Green, Onial, Phalan, & Balmford, 2019; Strassburg et al., 2017; Thomas et al., 2004) in converting changes in AOH into changes in population persistence using a power-law function with an exponent <1. Remaining proportions of AOH were used to derive a nonlinear 'persistence score', P, which captures the cumulative effect of habitat loss on the likelihood of the species' persistence in the study region:

$$P = (E)^z, \qquad (1)$$

where E is the remaining proportion of the original AOH, and z is the extinction coefficient. While Equation (1) is one way of capturing the cumulative effects of historical habitat loss on biodiversity persistence, other functions could be used once further work is done to establish the form of the relationship between the persistence of populations and habitat loss.

The change in a species' likelihood of persistence between two points in time was then calculated as ΔP, the corresponding difference in persistence score values:

$$\Delta P = \left[\left(E_{t0} \right)^z - \left(E_{t1} \right)^z \right], \qquad (2)$$

where E_{t0} and E_{t1} are the remaining proportions of AOH at t_0 and t_1, respectively.

For migratory species, an overall ΔP_{mig} score was calculated from ΔP scores derived separately for the species' breeding and non-breeding AOH. To estimate the overall change in a migratory species' persistence score, we assumed a multiplicative effect of changes in both parts of a species' range, as previously suggested by empirical (Lockwood, 2004) and theoretical studies (Iwamura et al., 2013):

$$\Delta P_{mig} = P_{b,t0} * P_{nb,t0} - P_{b,t1} * P_{nb,t1}, \qquad (3)$$

where P_b and P_{nb} are the persistence scores within the breeding and non-breeding ranges, respectively. For further discussion of the

conservation implications of this approach, see Supporting Information Section 2 and Figure S2.

We used a range of *z*-values to assess the effects of plausible variation in the extinction coefficient (see Supporting Information Section 3 and Figure S3). Our qualitative conclusions concerning the relative role of human activities on estimated biodiversity impacts were not dependent on the choice of a particular value of *z*. We therefore adopted a *z*-value of 0.25 in the main text, based upon its ability to predict proportions of species becoming extinct or threatened as a result of habitat loss (Brooks & Balmford, 1996; Brooks, Pimm, & Oyugi, 1999).

We also considered increases in species' persistence scores due to gain of suitable grid cells (e.g. through reversion of converted land to natural habitat). However, reversion is currently on such a limited scale in the Cerrado that incorporating these gains had a minor impact on the results for most of the groups, and was therefore not considered in the main text (Figure S4).

2.3 | Mapping the marginal value of land-use change and its biodiversity impacts across different scales

Although our assessment is not global, we wanted to capture the global conservation consequences of Cerrado habitat loss. Therefore, species' ΔP-values were weighted by the proportion of their global geographical range that fell within the study region. This assigns more weight to impacts on those species restricted to the biome.

To make the weighted ΔP spatially explicit, we next divided this by the number of cells converted over a given time interval (e.g. 2012–2014), thus capturing the marginal value of the loss of suitable habitat for each cell for that specific time period. The marginal value of land-use change permits estimation of the contribution of one spatial unit of AOH to a given species' likelihood of persistence. By estimating the marginal value of each spatial unit of land-use change, we can then map this as a continuous metric in a gridded landscape and assess the spatial distribution of the overall impact of habitat loss on species persistence. The resulting maps have the flexibility to be combined across species and to be aggregated across any scale of interest. Thus, for a period of time $t_0 \rightarrow t_1$, the marginal value of the loss of AOH within cell *j* (belonging to a total set of converted cells *R*), for the weighted persistence score of species *k*, $\mathrm{MV}_{t0 \rightarrow t1,j,k}$, can be represented as follows:

$$\mathrm{MV}_{t0 \rightarrow t1,j,k} = \Delta P_k \frac{w_k}{R_k}, \qquad (4)$$

where *R* is the total number of cells converted from suitable to unsuitable for that species in the period $t_0 \rightarrow t_1$, and *w* is the weight of species *k*. MV-values were then assigned to a gridded map, detailing the cells that turned into unsuitable habitat within the corresponding time interval. The resulting distribution maps of marginal loss values

for individual species were then overlaid and values summed across species to obtain, for each cell, an aggregated biodiversity impact metric of land-use change. Using maps of administrative boundaries (e.g. municipalities, states), cell-level impact values were then aggregated to give totals for administrative units of interest.

2.4 | Measuring the relative contribution of specific human activities to biodiversity impacts

We overlaid the aggregated biodiversity impact maps with land-use conversion maps to attribute impact estimates to soy cultivation and non-soy land-uses. To this end, we combined the soy-expansion maps with IBGE land-conversion maps to distinguish soy expansion from non-soy expansion (see Supporting Information Section 5 and Figure S5 for details on land-conversion maps analysis). We mapped both direct expansion of soy into natural vegetation (where soy production occurred within 3 years of natural vegetation conversion); and expansion into previously cleared areas, for the period 2000–2014 (from Gibbs et al., 2015).

3 | RESULTS

3.1 | Assessing species-level impacts

To illustrate changes in the persistence score at species level, we focused on five conservation flagship species in the Cerrado (WWF, 2015). For the Maned Wolf *Chrysocyon brachyurus*, Jaguar *Panthera onca*, Giant Armadillo *Priodontes maximus*, South American Tapir *Tapirus terrestris* and Giant Anteater *Myrmecophaga tridactyla*, habitat loss within the Cerrado has caused steady declines in their weighted persistence scores over the 2000–2014 period (Figure 2a; declines of 0.009, 0.07, 0.05, 0.05 and 0.04, respectively). Interspecific variation in such declines reflects differences in species-specific attributes captured by our approach, including habitat preferences, historical losses and geographical distributions. For instance, as the only species for which the human land-use categories 'Arable' and 'Pasture' are considered suitable habitats (IUCN, 2018), the Maned Wolf exhibits the smallest decline of the five species and had a markedly higher score than the other species in 2014. In addition, major losses of natural vegetation had already occurred by 2000 and these differed among species, with Maned Wolf losing 3.4% of its Cerrado AOH, Jaguar 88%, Giant Armadillo 80%, South American Tapir 84% and Giant Anteater 83%. Our metric accounted for such differences and so captured the varying effect of further losses of AOH on species' probabilities of persistence (Figure 2a). Finally, while the Jaguar showed the largest reduction of its original AOH within the Cerrado, this accounts for a relatively small proportion of its global range, resulting in a smaller change in its global persistence score than for other species (Figure 2a). We illustrate this by comparing weighted (Figure 2a) and non-weighted (Figure 2b) persistence scores. When no weighting was applied, all
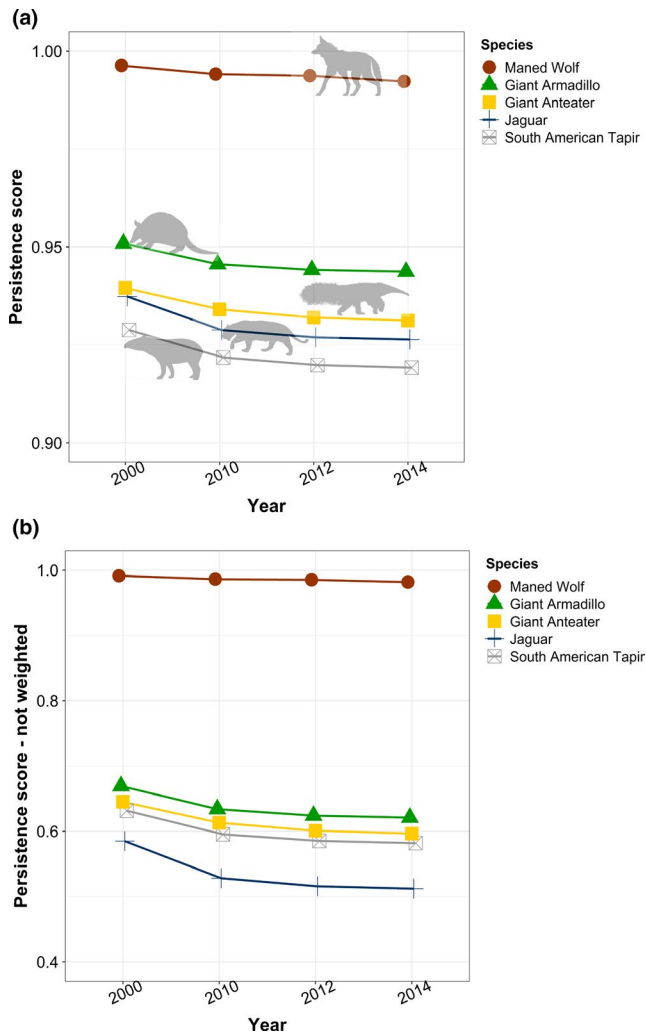
(a)

(b)

FIGURE 2 Changes in persistence score due to land conversion between 2000 and 2014, calculated at species level for five flagship species: (a) scores weighted by the proportion of species global range that fall within the study area and (b) scores not weighted

species had lower persistence scores—with the jaguar showing the lowest (0.58; Figure 2b).

3.2 | Capturing biodiversity impact variation across taxonomic groups

On average, the impact of Cerrado conversion 2000–2014 was greatest on mammals (0.074 ± 0.006; $M ± SE$), then plants (0.069 ± 0.003), birds (0.051 ± 0.002) and amphibians (0.037 ± 0.003; Figure 3). This variation was further explored by stratifying trends by species' level of endemism within the Cerrado (Figure 3). Overall, endemic and near-endemic groups—species with ≥70% of their global range falling within the Cerrado—showed the largest impact (for 2000–2014: 0.066 ± 0.004), and endemic birds (0.59 ± 0.042) and mammals (0.55 ± 0.065) were the two groups with the lowest scores by 2014 (Figure 3). In the 2012–2014 period alone, endemic plants lost on average 9.7% of their original AOH within the Cerrado, compared to the 7.8% lost over the 2000–2012 period. This resulted in a sharper mean decline of plants' weighted persistence score (0.038 ± 0.0032), relative to the prior 12 years (0.027 ± 0.0015).

To explore the conservation consequences of the estimated persistence scores, we also assessed how the latter varied across species with different sizes of area of occupancy (AOO); one of three criteria used by the IUCN RedList in criterion B2 to assess extinction risk (i.e. AOO < 10 km^2 can justify a classification of 'Critically Endangered'; IUCN, 2017a, 2017b). Our results show that, on average, lower persistence score values are associated with AOO sizes that meet the criteria for higher threat categories (see Supporting Information Section 6 and Figure S6). Thus, where there is further interest to explore the level of threat of species assessed under our methodology, the AOO criterion can be combined with our methodological framework.
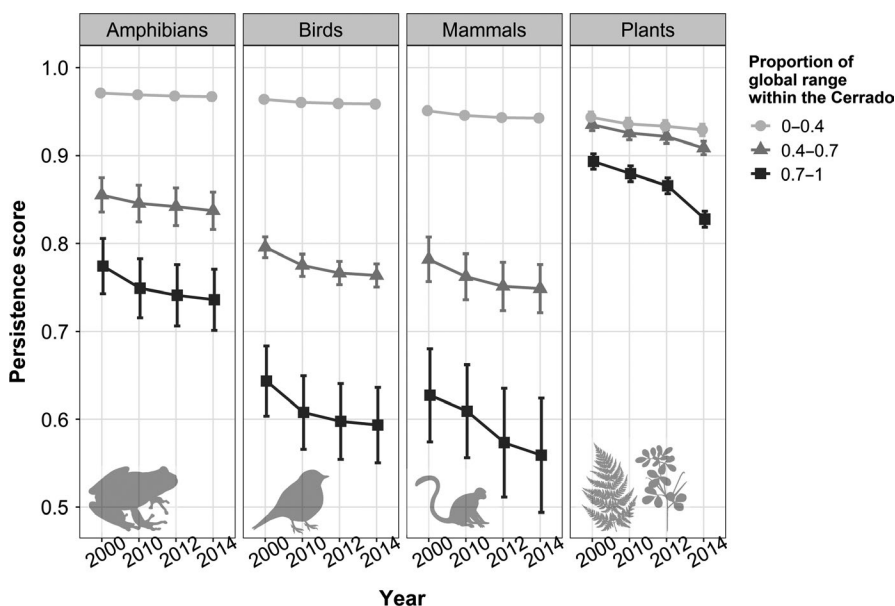

FIGURE 3 Changes in persistence score due to land conversion between 2000 and 2014, calculated for four taxonomic groups: amphibians, birds, mammals and plants. Trends show mean persistence scores stratified by the proportion of species global range that fall within the study area

3.3 | Mapping the impacts of land-use change on biodiversity across different scales

By deriving estimates for the marginal value of AOH loss for each cell in our analysis, our approach enables the biodiversity impacts of habitat conversion to be aggregated at different scales. Because each cell's score contributes proportionally to the aggregated biodiversity impact metric (Figure 4a; Equation 4), cell values can be summed across any area of interest (e.g. a municipality) to reflect that area's contribution to the overall biodiversity impact. In the Cerrado, aggregating the marginal values of AOH loss across municipalities and states for the 2000–2014 time period revealed distinct insights at different scales (Figure 4a–c)—identifying, for example, municipalities with relatively high biodiversity impact within states of relatively low aggregated total score (Figure 4b,c). As an illustration, the municipalities of Mateiros (with a score of 0.53; identified with a black star in the map) and Jaborandi (0.33; black pentagon) make up a substantial proportion of their states' impacts: Tocantins (TO: 1.23) and Bahía (BA: 0.51), respectively.

Areas of particular conservation concern can be further explored by disentangling mapped impacts for different taxonomic groups (Figure 4d,e). For example, for amphibians and birds, their aggregated biodiversity impact scores are concentrated in different states, with ~60% of the total impact for amphibians concentrated in only two states, Minas Geráis (MG), Goiás (GO), while for birds the same percentage is largely concentrated elsewhere: Mato Grosso du Sud (MS), Mato Grosso (MT) and Goiás (GO).

3.4 | Attributing biodiversity impacts to specific land uses

For 2000–2014, our results show that different types of land conversion vary in their impacts across taxonomic groups (Figure 5). For instance for birds, mammals and plants, while conversion to grassland comprised on average ~50% of their converted AOH, this was responsible for <25% of the total biodiversity impact for each group (grey line—Figure 5). In contrast, conversion to planted
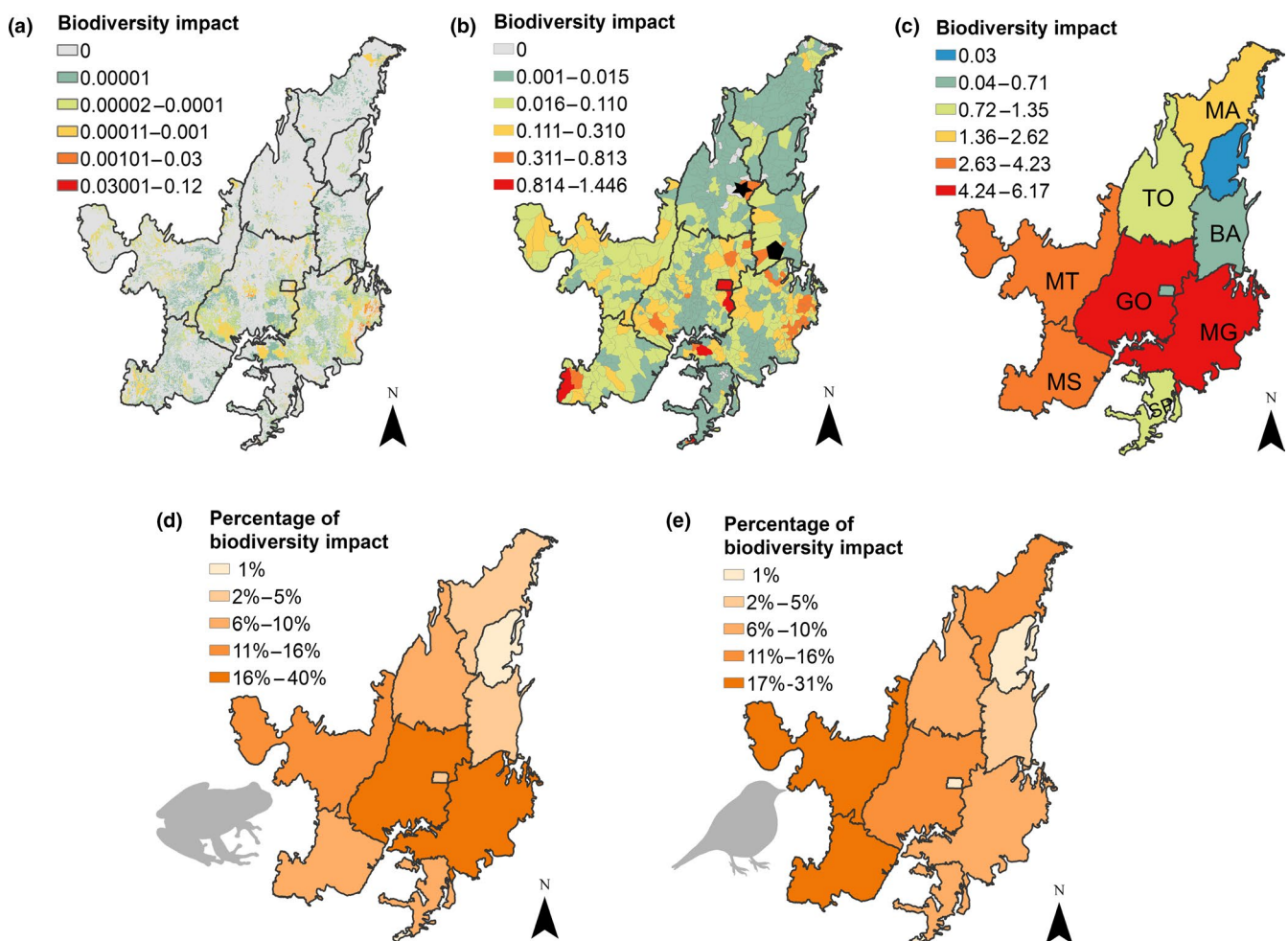
FIGURE 4 Maps of biodiversity impacts due to the losses of AOH between 2000 and 2014, when data are aggregated for all species at three different spatial scales. (a) Cell (0.0625 km²); (b) Municipality and (c) State. Also, distribution of percentage of persistence score for two taxonomic groups at state level: (d) Amphibians and (e) Birds. (MG: Minas Geráis; GO: Goiás; MT: Mato Grosso; MS: Mato Grosso do Sul; MA: Maranhão; SP: São Paulo; TO: Tocantins; BA: Bahia; PI: Piauí)

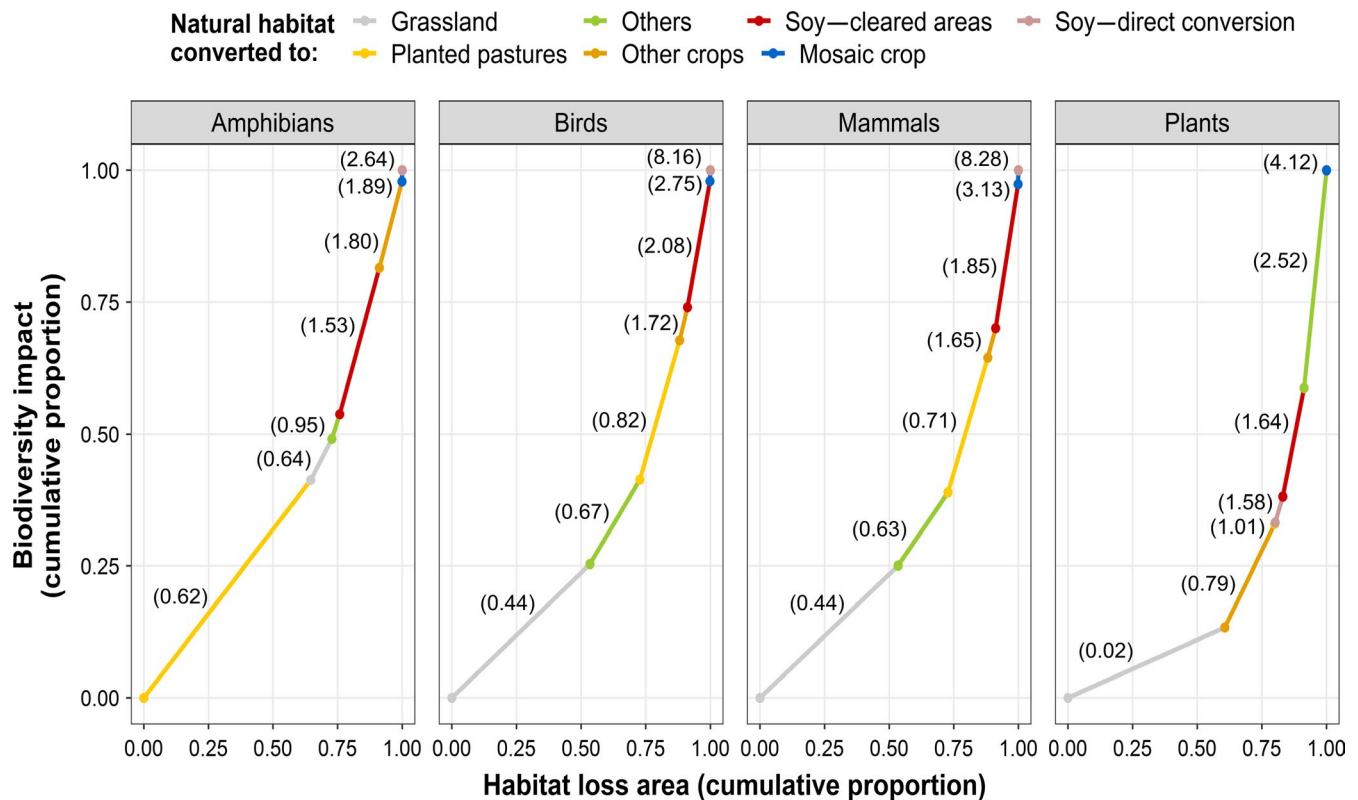**FIGURE 5** The proportional contribution of different land-use conversions to the total biodiversity impact 2000–2014 in the Cerrado for four taxonomic groups, plotted against the proportion of the total habitat loss area of each land-use change. Land-use conversions are plotted in order of increasing ratio of proportional contribution to change in persistence score: proportional contribution to loss of AOH (with the ratios shown in parentheses). Higher ratios thus indicate land-use conversions with disproportionately high impacts on our biodiversity footprint metric given the area converted. This is also reflected in a steeper slope. We aggregated IBGE land-use categories as follows: other crop (than soy), planted pasture, mosaic (mosaic-forest, mosaic-crop and mosaic-shrubland), grassland and other

pastures was responsible for up to 70% of the conversion of amphibian species' AOH and 45% of the associated biodiversity impact (yellow line—Figure 5). Interestingly, impacts driven by soy expansion demonstrated a similar pattern across taxonomic groups. Soy expansion into previously converted habitat was responsible for 3% of habitat conversion but 5% of total biodiversity impact. Direct expansion of soy into natural vegetation was responsible for only 0.15% of the total habitat converted, and 0.2% of the biodiversity impact for plants, 0.4% for amphibians and 1.2% for birds and mammals (Figure 5).

We also explored the relative biodiversity impact per unit area, which can reveal land-use transitions with disproportionate impacts on biodiversity. To this end, we calculated the ratio of proportional contribution to the total biodiversity footprint to proportion of area converted (Figure 5): higher ratios indicate land-use conversions with disproportionately high biodiversity footprints. We found that, although soy expansion through direct conversion of natural habitat had the smallest areal footprint, it had the highest impact on biodiversity per unit area for amphibians (2.64), birds (8.16) and mammals (8.28). This suggests that soy expansion through direct conversion has disproportionally reduced AOH (given the area involved). For plants, however, the mosaic crop category had the highest impact per unit area (4.12).

4 | DISCUSSION

Four criteria shaped the design of our biodiversity impact metric. We required it to (a) be able to capture the status of different components of biodiversity, while including species-specific aspects such as the distribution and ecological needs, (b) have the ability to account for historical habitat losses to estimate the cumulative impacts of further losses, (c) be scalable so it can be adapted to resolutions at which information on human activities and drivers is available and (d) allow biodiversity impacts to be linked to specific human activities. Below we highlight strengths and limitations of our approach in relation to both these criteria and existing biodiversity impact metrics.

4.1 | Capturing the status of different components of biodiversity

Our results illustrate that species' AOH maps provide a practical way to incorporate species-specific information in the assessment of biodiversity loss. Such accounting is an important first step in capturing biodiversity's multiple dimensions within a single metric. Here, we have focused on the ecological dimension, from which we have incorporated two key ecological attributes: global distribution and

habitat preferences. These are particularly relevant as their inclusion captures variation in individual species responses to different land-conversion types.

By combining information on land cover change, individual species' distributions and habitat preferences, this method identifies which biodiversity elements are most affected and where the greatest impacts have occurred (Figures 3 and 4). When information on habitat preferences is unavailable, as it was here for plants, assumptions on habitat requirements need to be made. If such assumptions are generous—such as that species can occur in a wide range of land covers including anthropogenic ones—there is higher chance of incurring errors of commission (assuming a species occurs where it does not) and hence of underestimating species' risk of local extinction (Rondinini, Stuart, & Boitani, 2005). In contrast, more conservative assumptions are prone to errors of omission (incorrectly assuming that a species is absent) and thus of overestimating impact (Rondinini et al., 2005). Under the precautionary principle, widely adopted in assessing biodiversity risk (Dickson & Cooney, 2005; Myers, 1993), conservative assumptions might be more appropriate.

As our study area covers a fraction of the global population of many species, we weighted species' persistence scores by the proportion of their geographical range that intersects the Cerrado. This assigns more weight to impacts on those species restricted to the biome, but places less emphasis on the local loss of species with a small fraction of their range intersecting the Cerrado. While such losses might have limited global conservation consequences, they could nonetheless have significant ecological or cultural effects. For instance, the Jaguar experienced only a small change in its weighted persistence score because of habitat loss in the Cerrado; a non-weighted score shows much more extensive decline (Figure 2). To represent losses of culturally or ecologically important species, it would also be possible to apply additional weightings when summing ΔP values across species, which could reflect variation in species' ecological or cultural significance.

While the wide availability of the data used here makes our method practical and accessible, we acknowledge that the variables we use cannot fully capture the ecological complexity of species' responses to habitat changes. For instance, habitat fragmentation and isolation can be important determinants of species occurrence (Rosa, Purves, Carreiras, & Ewers, 2014) and ignoring such landscape-level information can add further error into species' distribution mapping. Even though information on how species respond to fragmentation and edge-effects is currently absent from the IUCN Red List, recent studies have provided insight in how best to model this (Ewers, Marsh, & Wearn, 2010; Pfeifer et al., 2017). Moreover, direct habitat loss is not the only impact of land-use change, which can, in turn, lead to the expansion of invasive species, over-harvesting and pollution. Where land-use change exacerbates these other threats our impact estimates will again be conservative. However, the key characteristics of our metric computations (e.g. being highly spatially explicit,

working at species level and across different scales) could provide a flexible framework for incorporating such indirect impacts into this approach in the future.

4.2 | Accounting for historical habitat losses

Most studies that assess the impact of land-use change on biodiversity can be considered snap-shots of habitat loss patterns which do not account for historical losses. However, a 50% loss of current habitat for species A and B is not equivalent if species A has lost 20% more of its AOH than species B prior to the assessment. This is because (a) species that have lost a high proportion of their habitat prior to assessment will already have a higher local extinction risk and (b) the marginal value of each unit of remaining habitat increases as the total AOH remaining declines. The method we present here accounts for these two observations by assessing proportional habitat loss against species-specific AOH baselines. In addition, it estimates the cumulative effects of different habitat loss through a nonlinear relationship (power-law function; Equation 1) between proportional habitat loss and population persistence: the smaller the remaining AOH, the higher its value for the survival of species and consequently for conservation. While Equation 1 does not capture all the intricacies of the relationship between AOH and population persistence, it does reflect the increasing marginal value of remaining AOH. Moreover, this framework is amenable to revision as understanding of the form of the relationship between loss of AOH and the probability of local persistence improves.

4.3 | Aggregating biodiversity impact at different spatial scales

To bring these issues into decision-making processes, tools are needed to capture and translate ecological information to the scales at which decisions are made (Guerrero et al., 2013). The results presented here suggest that our proposed method meets these requirements, by capturing relevant ecological information such as species richness, mean historical habitat losses and endemicity (Figure S7), which can be adapted to different scales of decision-making. Metrics of impact that are adaptable to different scales of threat information are also likely to be useful in evaluating causal connections between biodiversity impact and human activities. We argue that such adaptability to different scales is one of the major shortcomings of biodiversity impact metrics that estimate species richness loss.

4.4 | Linking biodiversity impact to specific human activities

Globally, applicable biodiversity impact metrics have traditionally assessed anthropogenic impacts at sector level—in particular agriculture,

harvesting, transport, fishery and mining. While this helps identify sectors to target with conservation efforts, it does not provide sufficient detail to design detailed plans to tackle underlying drivers of biodiversity loss (IPBES, 2019). These drivers operate through specific human activities (Moran & Kanemoto, 2017) and, to mitigate their impacts, it is essential to quantify and map the connections between the consumption that drives habitat loss and its biodiversity impact (Lambin & Meyfroidt 2011). In this study, we focused on soy production as the proximate cause of habitat loss, which is influenced by remote drivers such as consumption patterns (Croft, West, & Green, 2018; de Ruiter et al., 2017), production shortages elsewhere (Godfray et al., 2010) and population growth (Dasgupta & Ehrlich, 2013).

The expansion of many worldwide agricultural commodities and their effects on biodiversity through land conversion are determined by local activities and processes, driven by international policies, trade agreements and consumption patterns. The methodological flexibility of our metric will facilitate the assessment of such drivers, and consequently the support of different decision-making contexts. Indeed, the different taxonomic and spatial resolution at which this approach works makes it suitable for assessing links between biodiversity impacts and specific human activities with data of different spatial resolution. For instance, if spatial information on specific human activities is only available at a higher administrative scale (e.g. state), aggregated changes in species persistence can still be linked to the human activity considering appropriate proportional relationship between the extent of both the human activity and the habitat converted. The transparency of our method permits its adaptation according to available data while still delivering applicable and practical information on changes in biodiversity state.

AUTHORS' CONTRIBUTIONS

A.P.D., J.M.H.G., C.D.W. and A.B. conceived the idea; A.P.D., J.M.H.G., P.V. and A.B. designed the analysis, A.P.D. conducted the analysis; and all the authors contributed to the writing, which was led by A.P.D.

DATA AVAILABILITY STATEMENT

Full data have not been archived according to the BES data archiving policy due to restrictions from the data owners. Below we indicate where and how to obtain data that were not archived. Species ranges for mammals and amphibians can be downloaded from https://www.iucnredlist.org/resources/spatial-data-download. Plant polygons are available in the repository https://tinyurl.com/y7zxzxhv. Bird distribution ranges have to be requested at http://datazone.birdlife.org/species/requestdis. Habitat preferences for vertebrates, including altitudinal ranges, can be downloaded from http://apiv3.iucnredlist.org/api/v3/docs after requesting a token in http://apiv3.iucnredlist.org/api/v3/token. The rest of the data used in this study (land cover and boundary data for Brazil, digital elevation map, soy expansion data) and R code are available at https://doi.org/10.5061/dryad.nvx0k6dpd (Durán et al., 2020).

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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