

1 **Title**

2 Putting species back on the map: devising a robust method for quantifying the
3 biodiversity impacts of land conversion

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29 **Running title**

30 Land conversion biodiversity footprint

31

32 **ABSTRACT**

33 **Aim:** Quantifying connections between the global drivers of habitat loss and
34 biodiversity impact is vital for decision-makers promoting responsible land-use. To
35 that end, biodiversity impact metrics should be able to report linked trends in specific
36 anthropogenic activities and changes in biodiversity state. However, for biodiversity,
37 it is challenging to deliver integrated information on its multiple dimensions (i.e.
38 species richness, endemism) and keep it practical. Here, we developed a
39 biodiversity footprint indicator that can i) capture the status of different species
40 groups, ii) link biodiversity impact to specific human activities, and iii) be adapted to
41 the most applicable scale for the decision context.

42 **Location:** Cerrado Biome, Brazil

43 **Methods:** We illustrate this globally-applicable approach for the case of soybean
44 expansion in the Brazilian Cerrado. Using species-specific habitat suitability models,
45 we assessed the impact of soy expansion and other land uses over 2,000 species of
46 amphibians, birds, mammals and plants for three time periods between 2000 and
47 2014.

48 **Results:** Overall, plants suffered the greatest reduction of suitable habitat. However,
49 among endemic and near-endemic species – which face greatest risk of global
50 extinction from habitat conversion in the Cerrado - birds were the most affected
51 group. While planted pastures and cropland expansion were together responsible for
52 most of the absolute biodiversity footprint, soy expansion via direct conversion of
53 natural vegetation had the greatest impact per unit area. The total biodiversity
54 footprint over the period was concentrated in the southern states of Minas Gerais,

55 Goiás and Mato Grosso, but the soy footprint was proportionally higher in those
56 northern states (such as Bahia and Piauí) which belong to the new agricultural
57 frontier.

58 **Main conclusions:** The ability and flexibility of our approach to examine linkages
59 between biodiversity loss and specific human activities has substantial potential to
60 better characterise the pathways by which habitat loss drivers operate.

61

62 **KEYWORDS**

63 Agriculture; Brazilian savannah; Footprint indicator; Migratory species; Suitable
64 habitat models; Soybean

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86 1 | INTRODUCTION

87 Habitat loss due to land-use change is the biggest threat facing global biodiversity
88 (Gibson et al., 2011; Joppa et al., 2016). Improved quantification of the scale of such
89 change has been essential in supporting international initiatives for better protection
90 and management of land (Ramsar, 1971; CBD, 2002; MEA, 2005). However, further
91 progress is now needed in the identification of underlying drivers of land-use change
92 and the monitoring of their associated environmental impacts to effectively articulate
93 strategies and actions to mitigate them (Han *et al.*, 2014). This need is echoed in the
94 recently approved Sustainable Development Goals (SDGs), particularly in Goals 12
95 and 15 concerning the sustainable use of land and the responsible production and
96 consumption of its derived commodities.

97 Biodiversity footprint indicators— which quantify the extent to which human
98 activities impact upon biodiversity – enable the monitoring and reporting of impacts
99 on biodiversity of specific human pressures (Sparks et al., 2011; Hoekstra &
100 Wiedmann, 2014; Hill et al., 2016). Using these to quantify associations between
101 underlying drivers, human activities and biodiversity loss not only helps track the
102 drivers of change (e.g. consumption patterns), but can also reveal the pressures
103 (e.g. agriculture expansion) and mechanisms (e.g. habitat conversion) through which
104 drivers impact biodiversity (Balmford et al., 2009). Biodiversity, however, is a
105 multidimensional concept and comprehensive measurement of changes in its state
106 still poses challenges (Souza et al., 2015). For instance, it could be assessed in
107 terms of species number (Newbold et al., 2015), rarity (Drever, Drever & Sleep,
108 2012), population density (Collen et al., 2009) and functional diversity (Cadotte,
109 Carscadden & Mirotnick, 2011). In turn, these dimensions can be variously
110 affected by different mechanisms resulting from human activities, such as habitat

111 loss (Hanski, 2011) or fragmentation (Fahrig, 2003). An increasing understanding of
112 how anthropogenic mechanisms affect different dimensions of biodiversity (Pearson
113 et al., 2014; Pfeifer et al., 2017), together with a larger methodological toolkit (Ewers,
114 Marsh & Wearn, 2010), provides an unprecedented opportunity to standardize a
115 comprehensive biodiversity impact metric. Nevertheless, most commonly-used
116 measuring techniques for the assessment of impacts on biodiversity still focus on
117 change in species richness, which does not capture the whole picture. Furthermore,
118 it is important that when assessing drivers of change, biodiversity impact metrics can
119 translate impact estimates into scales at which information on anthropogenic
120 activities is available and decisions are made (Ewers et al., 2010). When working
121 with relative species richness loss, however, the spatial variability of the impact
122 becomes difficult to scale up as the absence of species identity can lead to
123 challenges such overrepresentation of species' ranges or misrepresentation of
124 biodiversity priority areas (Veitch et al., 2017).

125 Approaches based on habitat suitability models offer great potential for
126 biodiversity footprint indicators because they can integrate spatially-explicit
127 information on anthropogenic land use and the ecology of individual species
128 (Rondinini et al., 2011; De Baan et al., 2015). Unlike approaches that estimate
129 potential regional or local loss of species richness (Newbold et al., 2015; Chaudhary
130 & Kastner, 2016), models of habitat suitability retain species-specific information
131 (Rondinini et al., 2011; de Baan, Mutel, Curran, Hellweg & Koellner et al., 2013),
132 highly relevant given the multiple dimensions of biodiversity. Specifically, they
133 quantify the relative change in the extent of suitable habitat (ESH) arising from land
134 conversion, which allows estimation of a species-specific impact metric that can be
135 associated with a particular human land-use change (Visconti et al., 2011; De Baan

136 et al., 2015). ESH is described by the intersection of a species' geographic range
137 with its environmental preferences, measured in terms of variables such as
138 vegetation cover, elevation and the location of water bodies and wetlands (Rondinini
139 et al., 2011). Changes in these variables, mainly due to habitat conversion, will
140 reduce the extent of usable habitat, affecting the persistence of local populations
141 (Mantyka-pringle, Martin, Rhodes, 2012).

142 Previous applications of habitat suitability models have tended to assume that
143 a species' persistence is directly proportional to the extent of remaining habitat,
144 failing to take into account preceding habitat loss (Buchanan, Donald & Buchart,
145 2011; van Soesbergen et al., 2017). An important consequence of this for
146 conservation is the underestimation of the impact of current habitat loss on species
147 that have lost a considerable proportion of their original habitat before the
148 assessment is performed (Groves et al., 2002; see section 2.1). Some studies have
149 addressed this limitation but have so far offered limited taxonomic coverage and
150 used projected land-use changes rather than direct observations of habitat
151 conversion (Visconti et al., 2016; Strassburg et al., 2017).

152 Here, using a non-linear and spatially-explicit approach we describe a
153 biodiversity footprint indicator designed to provide information on biodiversity impact,
154 which can be explicitly linked to specific human activities and adapted to relevant
155 contexts and scales of decision-making. The dual nature of this metric, both footprint
156 and indicator, allows us to quantify the biodiversity impact of specific human
157 activities, while reporting linked trends in pressure (e.g. agriculture expansion),
158 mechanism (e.g. land-use change) and biodiversity state. We illustrate this approach
159 using the example of the cultivation of soybean (*Glycine max*) in the Brazilian
160 Cerrado. We use species-specific habitat suitability models for four taxonomic

161 groups (amphibians, birds, mammals and plants) to illustrate three key benefits of
162 the method: i) its flexibility in capturing and incorporating various levels of ecological
163 information; ii) the scope for linking biodiversity impact to specific human activities
164 such as agricultural commodity expansion; and iii) the capacity to aggregate the
165 estimates across different spatial scales.

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167 **2 | METHODS**

168 An important attribute we have considered during the design of the method is its
169 applicability under a broad range of contexts. We therefore first describe how to
170 implement the approach in general terms (section 2.1), indicating the types of data to
171 use in each step. We then explain how this was implemented, and the specific
172 datasets used, for investigating the impacts of soy expansion in the Brazilian
173 Cerrado (section 2.2).

174 **2.1 | Method and rationale of the biodiversity footprint indicator**

175 The approach involves three steps: i) Mapping the extent of suitable habitat for each
176 species of interest, thus including species' distributions; ii) Estimating for each
177 species the reduction in their population persistence from the proportional loss of
178 ESH due to land-cover conversion. Within this same step, combining the estimates
179 across species to assess biodiversity impact, thereby considering quantity and
180 variability of species; and iii) Linking biodiversity impact to measures of specific
181 human activities.

182 Mapping the Extent of Suitable Habitat (ESH). We included all species whose
183 geographic ranges intersect the study region for which habitat information is
184 available. Every grid cell in a species' range in the region can be coded as suitable if
185 both the following conditions are met: (i) the cell is within the geographic range of the

186 species, and (ii) the local environment is within the species' known preferences (in
187 terms of land cover, elevation, etc.). The latter requires the harmonisation, in
188 consultation with experts, of the categories of the available land-cover map with
189 those used to describe species' habitat preferences. Coding of the suitability of cells
190 should be repeated in the same way for various points in time using environmental
191 data appropriate for each time. Each time period at which suitability is determined
192 should be assessed against a benchmark time. For migratory species, ESH should
193 be mapped separately for each species' resident, breeding and non-breeding
194 ranges, based on seasonal differences in their habitat preferences. This accounts for
195 seasonal variation in species' habitat requirements.

196 Estimating the marginal value of suitable habitat. The next step involves calculating,
197 for each species, the remaining proportion of its initial benchmark ESH within the
198 study area at each subsequent point in time. Changes in ESH are then used to
199 derive a non-linear persistence score, P , which captures the cumulative effect of
200 habitat loss on the likelihood of the species' persistence in the study region:

$$201 \quad P = (E)^z \quad (1)$$

202 where E is the remaining proportion of the original ESH, and z is the extinction
203 coefficient. Equation (1) is analogous (at the level of a single species; Thomas et al.,
204 2004) to the community-level species-area curve ($S=cA^z$). We propose its use here
205 based on the conjecture that the conversion of given absolute area of suitable
206 habitat (A_{loss} in Fig. 1) after a species has lost a small amount of its initial benchmark
207 ESH (a in Fig. 1), is likely to reduce the probability of the species' persistence less (c
208 in Fig. 1) than if the same area of suitable habitat was lost after much of the initial
209 ESH had already been converted (b and d in Fig. 1). As an increasing number of
210 studies have demonstrated, historical habitat loss can have important cumulative

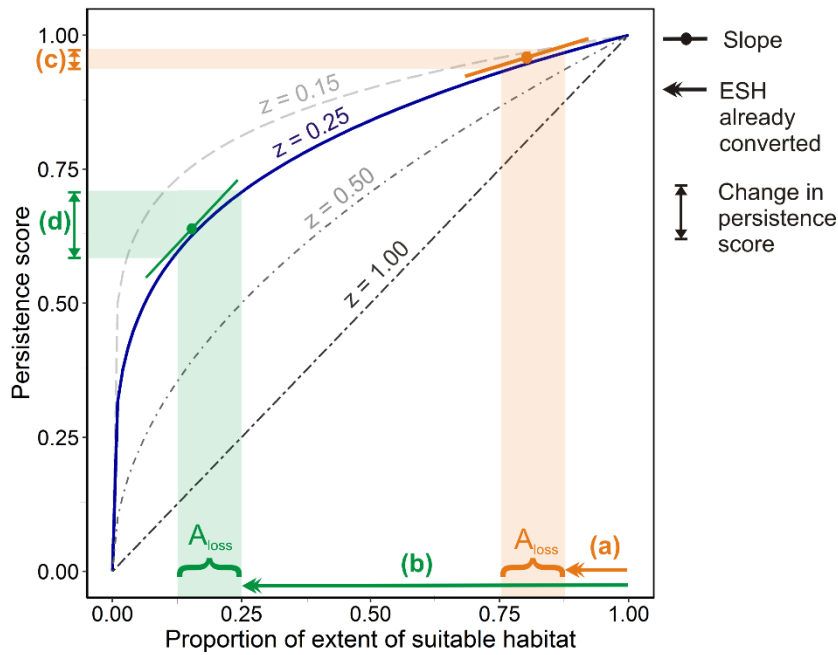
211 and delayed effects on biodiversity (Krauss et al., 2010; Wearn et al., 2012), and
212 ignoring such effects by assuming, for example, a linear relationship between habitat
213 loss and species' persistence (equivalent to a z-value of 1 in Equation 1), can result
214 in severe underestimations of biodiversity loss.

215 In addition, distribution size (here estimated by ESH), is one key factor
216 contributing to extinction risk and it is also closely correlated with population size
217 (Blackburn et al., 1997; Harris & Pimm, 2008). Therefore, reduction in species
218 distribution is expected to affect populations' persistence (IUCN, 2001). Here we
219 used proportion of ESH (and not the absolute area), as this allows assessing impact
220 across species in a standardized way while accounting for quantity and variability of
221 species. Since the initial benchmark ESH reflects the historical distribution size
222 (when probability of persistence was 1; see below for more details), the proportional
223 area loss declines at the same rate that absolute area loss relative to the
224 benchmark. Consequently, species with restricted ranges will move faster to the left
225 along the curve as the loss of one unit of absolute area means a higher proportional
226 loss than for a widespread species. It is worth noting, however, that further work is
227 required to establish empirically how the absolute and proportional area losses of
228 individual species are related to probability of persistence. As yet, there is no
229 standard method for such a calculation.

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Figure 1. Relationship between remaining extent of suitable habitat and

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species' persistence score, P , upon which the biodiversity footprint is based. If this

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relationship follows a power law (with $0 < z < 1$) the loss of a given area of suitable

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habitat (A_{loss}), when only a small proportion (a) of the original habitat has been lost

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previously, has a smaller impact (c) on score P than losing the same area when a

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much larger proportion (b) has already been lost (d). The size of this difference

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depends on the extinction coefficient z , which may well vary across taxa and regions.

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Once P has been estimated for two or more time points, the effect of

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intervening habitat loss on a species' likelihood of persistence within the study area

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can be calculated as ΔP , the corresponding difference in P -values:

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$$\Delta P = [(E_{t_0})^z - (E_{t_1})^z] \quad (2)$$

246

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

247 For migratory species, an overall ΔP_{mig} score should be calculated from ΔP
248 scores derived separately for the species' breeding and non-breeding ESH. In order
249 to estimate the total change in a migratory species' persistence score, a
250 multiplicative effect can be assumed, as previously suggested by empirical
251 (Lockwood, 2004) and theoretical studies (Iwamura et al., 2013):

$$252 \quad \Delta P_{mig} = P_{b,t0} * P_{nb,t0} - P_{b,t1} * P_{nb,t1} \quad (3)$$

253 where P_b and P_{nb} are the persistence scores within the breeding and non-breeding
254 ranges, respectively. This approach accounts for an interactive effect between
255 populations' likelihood of persistence along the migratory movements - an important
256 effect to consider in biodiversity impact quantifications (see Appendix S1 and Fig.
257 S1.1 in Supporting Information for further discussion of the implications of this
258 approach).

259 If there is interest in estimating global-level impacts but the study region itself
260 is not global, each species' ΔP -values should be weighted by the proportion of its
261 global geographic range falling within the study region. Other ways of weighting
262 different species – to reflect their ecological or evolutionary significance, for example
263 – can also be employed at this stage (see Discussion). The weighted ΔP of each
264 species (including migratory ones) will then be assigned to individual cells to derive
265 the marginal value of the loss of suitable habitat, MV , for each cell converted over a
266 given time interval. Thus, for a period of time $t_0 \rightarrow t_1$, the marginal value of the loss of
267 suitable habitat within cell j (belonging to a set of converted cells R), for the weighted
268 ΔP score of species k , $MV_{t_0 \rightarrow t_1, j, k}$, can be represented as:

$$269 \quad MV_{t_0 \rightarrow t_1, j, k} = \Delta P_k \frac{w_k}{R_k} \quad (4)$$

270 where R is the total number of cells converted from suitable to unsuitable for that
271 species in the period $t_0 \rightarrow t_1$, and w is the weight of species k (representing, for

272 example, the proportion of its geographic range falling within the study region). The
273 resulting distribution maps of marginal loss values for individual species are then
274 overlaid and values summed across species to obtain, for each cell, an aggregated
275 biodiversity impact metric. Using maps of administrative boundaries (e.g.
276 municipalities), the cell-level impact values can then be aggregated to give totals for
277 administrative units of interest.

278 Linking biodiversity impact to a human pressure. The biodiversity impact scores
279 described above can be attributed to categories of land-use conversion due to
280 different types of human use (e.g. natural vegetation to cropland). Where more
281 detailed spatial information on human activities is available, the impacts can be
282 associated even more specifically with particular production systems, which in turn
283 are related to the land-use conversion assessed above (Eq. 2).

284 **2.2 | Applying the method to soy expansion in the Cerrado**

285 We applied the approach outlined above to the specific case of the expansion of soy
286 cultivation in the Cerrado over the period 2000 – 2014. Considered one of the
287 world's most diverse savannah ecosystem, the Cerrado is severely threatened by
288 the expansion of soybean cultivation and cattle ranching (Strassburg et al., 2017).

289 Mapping ESH within the biome. After selecting all the species in our focal taxa
290 whose current ranges intersected the Cerrado boundary (IBGE 2004) and for which
291 habitat information was available, we produced habitat suitability models to obtain
292 234 ESHs for amphibians, 846 for birds and 288 for mammal species, each at 250 m
293 x 250 m resolution (the resolution of the best available land cover maps for Brazil
294 with which land-use change can be quantified consistently; IBGE 2015). Based on
295 information on habitat associations and elevation limits obtained from the IUCN

296 Habitats Classification Scheme (IUCN 2017), we refined the historical geographic
297 range (Extant, Probably Extant, Possibly Extinct, Extinct and Presence
298 Uncertain) of each vertebrate species (BirdLife 2016, IUCN 2017) using a digital
299 elevation model (USGS 2006) and land cover maps (IBGE 2004, 2011, 2014). The
300 14 categories of the land-cover map were harmonised with the 74 habitat preference
301 levels (for details see Appendix S2 and Table S2.1 in Supporting Information).

302 Multiple environmental variables define species distribution as well as a populations'
303 response to habitat loss. Yet, data on habitat preferences and altitudinal range are
304 the only species-specific variables that are available globally. Since this approach
305 aims to be globally applicable we limited the illustration of ESH mapping to these two
306 variables, although further information can be incorporated during the refinements of
307 species ranges (see Discussion).

308 We also produced habitat suitability model maps for 648 plant species whose
309 ranges intersect the Cerrado. In the absence of more detailed information on
310 species' habitat requirements, we refined their geographic ranges (Martinelli &
311 Moraes 2013) using information on vegetation types. We assumed that only those
312 vegetation categories classified as natural by the Brazilian Institute of Geography
313 and Statistics (IBGE) were potentially suitable for these species, while other semi-
314 and non-natural categories were unsuitable (Table S2).

315 We applied our approach to IBGE land cover maps for the year 2000, 2010,
316 2012 and 2014. For vertebrates, we used a map of original vegetation cover for the
317 Cerrado as our initial benchmark (c.a. 16th century; IBGE 2004), to estimate ESH
318 prior to large-scale cultivation for each species. For plants, the geographic range
319 intersecting the study region was considered to delineate its original ESH.

320 Impacts of habitat loss on the Cerrado's biodiversity. For the calculation of weighted
321 marginal values of each cell for each species (Eq. 4), we adopted a z-value of 0.25,
322 based upon its ability to predict proportions of species becoming extinct or
323 threatened as a result of habitat loss in several species-area analyses (Brooks &
324 Balmford, 1996; Brooks, Pimm & Oyugy et al., 1999). Different z-values influence the
325 effect of habitat loss on probability of persistence (Fig. 1; Eq. 2). However, our
326 qualitative conclusions concerning the relative role of human activities on different
327 groups of species and the spatial distribution of estimated biodiversity impacts are
328 not strongly dependent on the choice of a particular value of z (Table S3.2 and Fig.
329 S3.2 in Appendix 3 for the effects of plausible variation in z-coefficients). In addition,
330 when information is available different z-values can be assigned to different
331 biodiversity groups. Species' ΔP values were weighted by the proportion of their
332 global geographic range falling within the study region.

333 We also considered marginal increases due to gain of suitable cells (e.g.
334 through reversion of converted land to natural habitat). However, reversion is
335 currently on such a limited scale in the Cerrado that incorporating such gains had a
336 minor impact on the results for most of the groups, and was therefore not considered
337 in the main text (see Fig. S4.3 in Appendix S4).

338 Quantifying the biodiversity impact of soybean expansion. We used two types of
339 maps for cumulative soy expansion from Gibbs et al. (2015) for the period 2000-
340 2014: 1) direct expansion of soy into natural vegetation (where soy production
341 occurred within three years of natural vegetation conversion); and 2) expansion into
342 previously-cleared areas. Before intersecting soy-expansion maps with biodiversity
343 impact maps, we combined the former with IBGE land-conversion maps to
344 distinguish soy expansion from non-soy crop expansion (see Fig. S5.4 in Appendix

345 S5 in Supporting Information for details on land-conversion maps analysis). The
346 resulting merged layer allowed us to also assess the impact of other non-crop
347 categories such as planted pasture.

348

349 **3 | RESULTS**

350 **3.1 | Assessing the biodiversity footprint for different species groups**

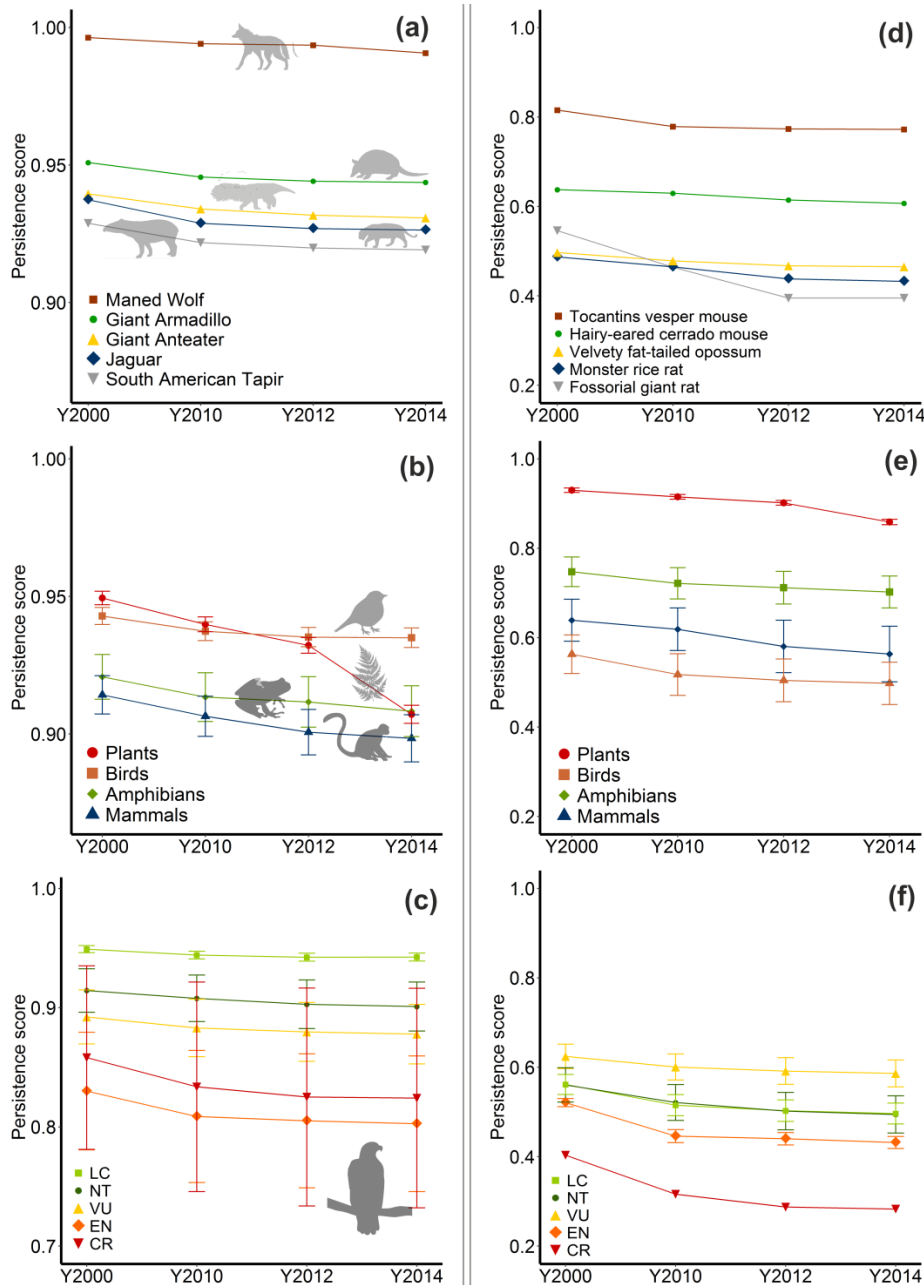
351 In order to illustrate biodiversity impact at species level we focused on five
352 conservation flagship species in the Cerrado (WWF, 2015). For the Maned Wolf
353 (*Chrysocyon brachyurus*), Jaguar (*Panthera onca*), Giant Armadillo (*Priodontes*
354 *maximus*), South American Tapir (*Tapirus terrestris*) and Giant Anteater
355 (*Myrmecophaga tridactyla*), habitat loss within the Cerrado has caused steady
356 declines in their weighted persistence scores over the 2000-2014 period (Fig. 2a;
357 declines of 0.006, 0.007, 0.009, 0.009, and 0.01, respectively). As the only species
358 for which 'Arable' and 'Pasture' are considered suitable habitats (IUCN 2017), the
359 Maned Wolf presented the smallest decline of the five species and had a markedly
360 higher score than the other species in 2014. Major losses of natural vegetation had
361 occurred by 2000 already, with Giant Armadillo losing 80% of its Cerrado ESH, Giant
362 Anteater 83%, Jaguar 88% and South American Tapir 84%. While the Jaguar
363 showed the largest reduction of its original ESH within the Cerrado, this accounts for
364 a relatively small proportion of its global range, resulting in a smaller change in its
365 persistence score than for other species (Fig. 2a).

366 When biodiversity impacts were aggregated by taxonomic group (Fig. 2b),
367 plants showed the largest impact 2000-14 (0.042 ± 0.002 ; mean \pm standard error),
368 then mammals (0.015 ± 0.007), amphibians (0.012 ± 0.008) and birds ($0.0079 \pm$
369 0.003). In the 2012-2014 period alone, plants lost on average 9.1% of their original

370 ESH within the Cerrado (0.30 y^{-1}), compared to the 7.1% lost over the 2000-2012
371 period (0.07 y^{-1}). This resulted in a sharper mean decline of plants' weighted
372 persistence score (0.025 ± 0.0032 ; $0.047/\text{y}^{-1}$), relative to the prior twelve years
373 (0.017 ± 0.0013 ; $0.006/\text{y}^{-1}$).

374 Focusing on birds, we also assessed how impacts varied across species of
375 different conservation status (Fig. 2c). Declines in persistence scores were
376 consistently greater among species in higher extinction risk categories (Fig. 2c).
377 Among Critically Endangered (CR; 0.034 ± 0.015) and Endangered (EN; $0.027 \pm$
378 0.007) species, the mean persistence score decreased more between 2000 and
379 2014 than among Vulnerable (VU; 0.014 ± 0.002), Near Threatened (NT; $0.013 \pm$
380 0.002) and Least Concern (LC; 0.006 ± 0.0004) species.

381 We also assessed endemic and near-endemic species, for which we included
382 those species with more than 70% of their global range falling within the Cerrado.
383 Overall, compared to more widely distributed species and species groups (Fig.
384 2a,b,c), endemics presented a much more severe decline in their persistence score
385 (Fig. 2d,e,f), representing an acute threat to their global persistence.



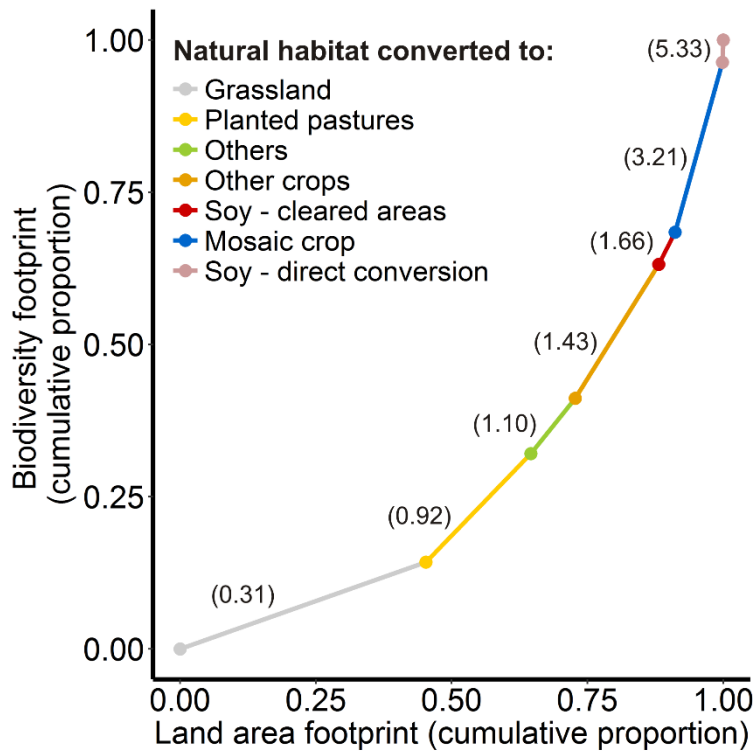
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387 **Figure 2.** Changes in persistence score due to land conversion between 2000 and
 388 2014, calculated for different levels and elements of biodiversity: (a) at species level,
 389 showing results for five flagship species; (b) for taxonomic groups, showing mean
 390 persistence scores for four vertebrate taxa; (c) grouped by IUCN Red List status,
 391 showing mean persistence scores for birds; (d) at species level, showing results for
 392 five endemic and near-endemic mammal species; (e) for taxonomic groups, showing
 393 mean persistence scores for endemic and near-endemic species only; and (f)
 394 grouped by IUCN Red List status, showing mean persistence scores for endemic
 395 and near-endemic bird species only. Upper and lower bars show one standard error.

396 **3.2 | Links between biodiversity footprint and commodity production**

397 For 2000-2014, our results revealed that conversion to grassland, whilst comprising
398 45% of the area of habitat converted, was responsible for just 14% of the total
399 biodiversity footprint of all land conversion in the region (Fig. 3). In contrast, planted
400 pastures, crops other than soybean and mosaic crops were together responsible for
401 43% of the habitat conversion but 67% of the biodiversity footprint (Fig. 3). Soybean
402 expansion into previously converted habitat was responsible for 3% of the habitat
403 conversion but 5% of the total biodiversity footprint. Lastly, whilst direct expansion of
404 soy into natural vegetation was responsible for only 0.15% of the total habitat
405 converted, it accounted for 0.8% of the biodiversity footprint.

406 We also explored the relative footprint per unit area, which can reveal land
407 use transitions with disproportionate impacts on biodiversity. To this end, we
408 calculated the ratio of proportional contribution to the total biodiversity footprint to
409 proportion of area converted (Fig. 3): higher ratios indicate land-use conversions with
410 disproportionately high biodiversity footprints. We found that, although soy expansion
411 through direct conversion of natural habitat had the smallest areal footprint, it had the
412 highest impact on biodiversity per unit area (5.3). This indicates that soy expansion
413 through direct conversion has altered a disproportional amount of ESH relative to the
414 total footprint land used. Further disaggregating this by taxa showed that mammals
415 were the most affected group (8.3), followed by birds (7.9), amphibians (2.6) and
416 plants (1.6) (see Fig. S6.5 in Supporting Information).



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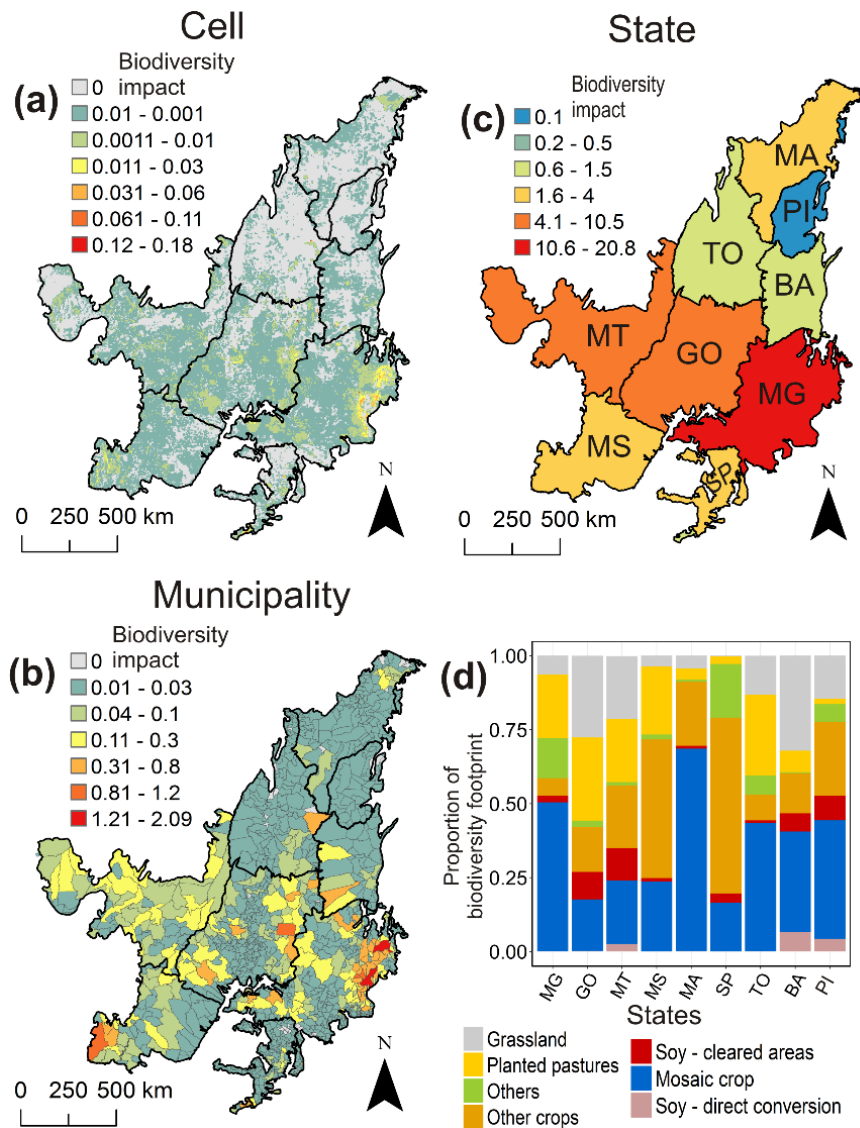
418 **Figure 3.** The proportional contribution of different land-use conversions to the total
419 biodiversity footprint 2000-14 in the Cerrado including four taxonomic groups, plotted
420 against the proportion of the total land area footprint of each land-use change. Land-
421 use conversions are plotted in order of increasing ratio of proportional contribution to
422 change in persistence score: proportional contribution to loss of ESH (with the ratios
423 shown in parentheses). Higher ratios thus indicate land-use conversions with
424 disproportionately high impacts on our biodiversity footprint metric given the area
425 converted. We aggregated IBGE land-use categories as follows: other crop (than
426 soybean), planted pasture, mosaic (mosaic-forest, mosaic-crop and mosaic-
427 shrubland), grassland, and other.

428

429 **3.4 | Adapting biodiversity footprint to scales of decision-making**

430 We designed our footprint indicator so it can be aggregated at different scales, while
431 still capturing ecological impacts of change. Each cell's score contributes
432 proportionally to the footprint (Eq. 4), so cell values can be summed across any area
433 of interest (e.g. a municipality) to reflect that area's contribution to the overall

434 footprint. In the Cerrado, aggregating the biodiversity footprint indicator across
435 municipalities and states for the 2000-14 time period revealed distinct insights at
436 different scales (Fig. 4a-c). It is possible to identify municipalities with relatively high
437 biodiversity impact within states of relatively low footprint, revealing local-scale
438 impacts that are diluted at coarser resolution. For example, the municipalities of
439 Mateiros (with a score of 0.54) and Jaborandi (0.32), fell in two states with overall
440 low values: the state of Tocantins (1.36) and Bahía (0.99), respectively. These two
441 states are part of the 'MATOPIBA' agricultural frontier and have undergone more
442 rapid habitat conversion since 2000 than other states (Fig. S7.6 in Supplementary
443 Information). Their relatively low species richness (Fig. S8.7b), however, results in
444 lower overall impact scores than in more biodiversity-rich states (Fig. S8.7b; Fig.
445 S9.8). Nevertheless our method singles-out areas that may be of particular
446 conservation concern in these states. Our method also allowed us to disentangle the
447 state-level biodiversity footprint into different types of land conversion (Fig. 4d). In
448 two states that underwent particularly extensive habitat clearance prior to 2000, Mato
449 Grosso and Goiás, subsequent soy expansion was largely into already-cleared areas
450 (Fig. 4d), and so had a relatively low footprint. In contrast, in Bahía and Piauí, two
451 states that have undergone extensive habitat clearance within the new agricultural
452 frontier, recent soy expansion is associated with a greater impact on biodiversity.



453

454 **Figure 4.** Distribution of biodiversity footprint scores due to loss of ESH 2000-14,
 455 when data are aggregated for all species at three different spatial scales and for
 456 different land/use changes. (a) Cell (0.0625 km²); (b) Municipality; (c) State; and (d)
 457 proportional contribution of different land-use conversions to the total biodiversity
 458 footprint 2000-14 at state level. [MG: Minas Gerais; GO: Goiás; MT: Mato Grosso;
 459 MS: Mato Grosso do Sul; MA: Maranhão; SP: São Paulo; TO: Tocantins; BA: Bahia;
 460 PI: Piauí].

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463

464 **4 | DISCUSSION**

465 Three criteria shaped the design of our biodiversity indicator; it should: i) be able to
466 capture the status of different components of biodiversity, while including aspects
467 such as the distribution, quantity and variability of species, ii) allow biodiversity
468 impacts to be linked to specific human activities; and iii) be scalable to inform
469 decision-making at different levels. Below we highlight strengths and limitations of
470 our approach in relation to these criteria.

471 *4.1 Capturing the status of different components of biodiversity.*

472 By combining information on land cover change, individual species' distributions and
473 habitat preferences this method identifies which biodiversity elements are most
474 affected and where the greatest impacts have occurred (Fig. 2; Fig. 4). Working at
475 the level of species allows features of the ecology of species (such as their habitat
476 specificity, endemism or migratory movements) to be incorporated, thereby
477 considering species' distribution and variability (Visconti et al., 2016). Making such
478 information spatially explicit, allows hotspots of biodiversity risk to be identified, and
479 provides information on the quantity of species that are vulnerable (Visconti et al.,
480 2011).

481 When information on habitat preferences is unavailable, as it was here for plants,
482 assumptions on habitat requirements need to be made. If such assumptions are
483 generous – such as that species can occur in a wide range of land covers including
484 anthropogenic ones - there is higher chance of incurring errors of commission
485 (assuming a species occurs where it does not) and hence of underestimating
486 species' risk of local extinction (Rondinini, Stuart & Boitani, 2005). In contrast, more
487 conservative assumptions are prone to errors of omission (incorrectly assuming that
488 a species is absent) and thus of overestimating impact (Rondinini et al., 2005).

489 Under the precautionary principle, widely adopted in assessing biodiversity risk
490 (Myers 1993; Dickson & Cooney, 2005), conservative assumptions might be more
491 appropriate.

492 As our study area covers a fraction of the global population of many species,
493 we weighted species' persistence scores by the proportion of their geographic range
494 that intersects the Cerrado. This assigns more weight to impacts on those species
495 restricted to the biome, but places less emphasis on the local loss of species with a
496 small fraction of their range intersecting the Cerrado. While such losses might have
497 limited global conservation consequences, they could nonetheless have significant
498 ecological or cultural effects. For instance the Jaguar experienced only a small
499 change in its weighted persistence score as a result of habitat loss in the Cerrado; a
500 non-weighted score shows much more extensive decline (a local decline to 0.51
501 versus a global decline to 0.92 by 2014; Fig. S10.9a). To represent losses of
502 culturally- or ecologically- important species, it would also be possible to apply
503 additional weightings when summing ΔP values across species, which could reflect
504 variation in ecological or cultural significance.

505 While the wide availability of the data used here makes our method practical
506 and accessible, we acknowledge that the variables we use cannot fully capture the
507 ecological complexity to which species respond. For instance, habitat fragmentation
508 and isolation can be important determinants of species occurrence (Ewers et al.,
509 2010) and ignoring such landscape-level information can add further error into
510 species' distribution mapping (i.e. omission and commission error, see above for
511 further discussion). Even though information on how species respond to
512 fragmentation and edge-effects is currently absent from the IUCN Red List, recent
513 studies have provided insight in how best to model this. By combining suitable

514 habitat modelling techniques and spatial layers, a continuous representation of
515 individual species' responses to fragmentation and edge-effects can be calculated
516 (Ewers et al., 2010; Pfeifer et al., 2017). Thus, combining these layers with a
517 biodiversity footprint metric, such as the one proposed in this paper, can help us
518 understand how biodiversity responds to changes in both landscape composition
519 and structure. Such an advancement will provide key insights into land management
520 and biodiversity conservation.

521 **4.2 | Linking biodiversity impact to specific human activities**

522 Our method disentangles, at different spatial scales, the effects of human activities
523 bringing about habitat loss (Fig. 3). This is essential for then tracking the pathway
524 through which underlying drivers of habitat loss operate (Moran & Kanemoto, 2017).
525 In this study we focused on soy production as the direct human activity affecting
526 habitat loss, which in turn can be influenced by remote drivers such as consumption
527 patterns (de Ruiter et al., 2017), production shortages (Godfray et al., 2010) and
528 population growth (Dasgupta & Ehrlich, 2013). As well as remote drivers, other set of
529 indirect channels can also influence the effects of a human activity on habitat loss.
530 This can be through land use displacement, a widely recognized mechanism
531 underlying indirect land use change (Lambin & Meyfroidt, 2011). Also, via the ability
532 to influence regional land markets, therefore affecting deforestation decisions
533 indirectly (Richards, 2015). Similar to previous studies (Richards, 2015), our
534 estimation of soy indirect impact (through the displacement of cattle ranches into
535 natural vegetation (see Appendix 11 and Fig.11.10 in Supplementary Information)),
536 also suggested a limited role of land use displacement in the overall impact. Thus,
537 incorporating multiple techniques to capture direct and indirect drivers, while
538 encompassing a broader time frame that allows assessing historical land-conversion

539 trends, will certainly better capture the full responsibility of assessed human
540 activities, such as the case of soy in the Cerrado.

541 **4.3 | Aggregating biodiversity impact at different spatial scales**

542 Developing tools that capture and translate the ecological scale of the problem to
543 scales where decisions are made has been suggested as a key solution to improve
544 evidence impact (Guerrero, McAllister, Corcoran & Wilson, 2013). The results
545 presented here suggest that our proposed method meets these requirements, by
546 capturing relevant ecological information such as species richness, mean historical
547 habitat losses and endemism (Fig. S8.7), which can be adapted to different scales of
548 decision-making. Metrics of impact that are adaptable to different scales of threat
549 information are also likely to be useful in evaluating causal connections between
550 biodiversity impact and human activities (see section 4.4 for more discussion on this
551 regard). Another relevant aspect is the sensitivity of the aggregated metric to its key
552 parameters (Eq. 2). Using different z-values we observed only minor changes in the
553 aggregated biodiversity footprint and the distribution of biodiversity risk hotspots (Fig.
554 S3.2e,f). As z increases, the decline of species' persistence score increases for a
555 given loss of ESH (Fig. 1). Hence at higher z, areas (e.g. states) that harbour
556 species with high historical ESH loss such as Mato Grosso (MT) and Mato Grosso
557 do Sul (MS) (Fig. S8.7c), have a higher increment in their aggregate biodiversity
558 footprint than do areas with less historical loss of ESH.

559 **4.4 | Implications of our method for the Cerrado**

560 The Cerrado example illustrates how our approach can quantify human activities
561 driving land-use change and monitor their biodiversity impacts. Although these
562 activities are well known to be in the Cerrado soy and livestock production, there
563 remains a clear need to map the underlying trade system of both commodities

564 (Garrett, Lambin & Naylor, 2013). Brazil is now the second-largest soy producer
565 worldwide, and in 2013/2014 about half (52%) of soybeans produced in Brazil came
566 from the Cerrado (INPUT, 2016). A better understanding of the highly complex
567 production-to-consumption system, comprising large numbers of trade actors (e.g.
568 producers, manufacturers, exporters), is an ongoing and challenging effort (Godar,
569 Suavet, Gardner, Dawkins, & Meyfroidt, 2016). By linking spatially-explicit
570 biodiversity risk hotspots with information on soy and livestock production and trade
571 our approach provides a platform to start disentangling the relative roles of different
572 actors.

573

574 REFERENCES

- 575 De Baan, L., Mutel, C. L., Curran, M., Hellweg, S., & Koellner, T. (2013). Land use in life
576 cycle assessment: global characterization factors based on regional and global
577 potential species extinction. *Environmental science & Technology*, *47*, 9281-9290.
- 578 de Baan, L., Curran, M., Rondinini, C., Visconti, P., Hellweg, S., & Koellner, T. (2015). High-
579 resolution assessment of land use impacts on biodiversity in life cycle assessment
580 using species habitat suitability models. *Environmental science & Technology*, *49*,
581 2237-2244.
- 582 Balmford, A., Carey, P., Kapos, V., Manica, A., Rodrigues, A. S., Scharlemann, J. P., &
583 Green, R. E. (2009). Capturing the many dimensions of threat: comment on Salafsky et
584 al. *Conservation Biology*, *23*, 482-487.
- 585 BirdLife International and NatureServe (2016) Bird species distribution maps of the world.
586 BirdLife International, Cambridge, UK and NatureServe, Arlington, USA.
- 587 Blackburn, T.M., Gaston, K.J., Quinn, R.M., Arnold, H. & Gregory, R.D. (1997) Of mice and
588 wrens: the relation between abundance and geographic range size in British mammals
589 and birds. *Philosophical Transactions of the Royal Society of London B: Biological*
590 *Sciences*, *352*, 419-427.
- 591 Brooks, T., & Balmford, A. (1996). Atlantic forest extinctions. *Nature*, *380*, 115.
- 592 Brooks, T. M., Pimm, S. L., & Oyugi, J. O. (1999). Time lag between deforestation and bird
593 extinction in tropical forest fragments. *Conservation Biology*, *13*, 1140-1150.
- 594 Buchanan, G. M., Donald, P. F., & Butchart, S. H. (2011). Identifying priority areas for
595 conservation: a global assessment for forest-dependent birds. *PloS one*, *6*, e29080.
- 596 CBD (2002) Rio+20 United Nations Conference on Sustainable Development.
- 597 Cadotte, M. W., Carscadden, K., & Mirotnick, N. (2011). Beyond species: functional

- 598 diversity and the maintenance of ecological processes and services. *Journal of Applied*
599 *Ecology*, 48, 1079-1087.
- 600 Chaudhary, A., & Kastner, T. (2016). Land use biodiversity impacts embodied in
601 international food trade. *Global Environmental Change*, 38, 195-204.
- 602 Collen, B., Loh, J., Whitmee, S., McRAE, L., Amin, R., & Baillie, J. E. (2009). Monitoring
603 change in vertebrate abundance: the Living Planet Index. *Conservation Biology*, 23,
604 317-327.
- 605 Dasgupta, P. S., & Ehrlich, P. R. (2013). Pervasive externalities at the population,
606 consumption, and environment nexus. *Science*, 340(6130), 324-328.
- 607 Dickson, B. & Cooney, R. (2005). Biodiversity and the precautionary principle : risk and
608 uncertainty in conservation and sustainable use. London, Earthscan.
- 609 Espírito-Santo, M. M., Leite, M. E., Silva, J. O., Barbosa, R. S., Rocha, A. M., Anaya, F. C.,
610 & Dupin, M. G. (2016). Understanding patterns of land-cover change in the Brazilian
611 Cerrado from 2000 to 2015. *Philosophical Transactions of the Royal Society B*, 371,
612 20150435.
- 613 Ewers, R. M., Marsh, C. J., & Wearn, O. R. (2010). Making statistics biologically relevant in
614 fragmented landscapes. *Trends in Ecology & Evolution*, 25, 699-704.
- 615 Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology,*
616 *Evolution, and Systematics*, 34, 487-515.
- 617 Garrett, R. D., Lambin, E. F., & Naylor, R. L. (2013). Land institutions and supply chain
618 configurations as determinants of soybean planted area and yields in Brazil. *Land Use*
619 *Policy*, 31, 385-396.
- 620 Gibbs, H. K., Rausch, L., Munger, J., Schelly, I., Morton, D. C., Noojipady, P., ...Walker, N.
621 F. (2015). Brazil's soy moratorium. *Science*, 347, 377-378.
- 622 Gibson, L., Lee, T. M., Koh, L. P., Brook, B. W., Gardner, T. A., Barlow, J., ... Sodhi, N. S.
623 (2011). Primary forests are irreplaceable for sustaining tropical
624 biodiversity. *Nature*, 478, 378.
- 625 Godar, J., Suavet, C., Gardner, T. A., Dawkins, E., & Meyfroidt, P. (2016). Balancing detail
626 and scale in assessing transparency to improve the governance of agricultural
627 commodity supply chains. *Environmental Research Letters*, 11, 035015.
- 628 Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., ... &
629 Toulmin, C. (2010). Food security: the challenge of feeding 9 billion
630 people. *Science*, 327, 812-818.
- 631 Groves, C. R., Jensen, D. B., Valutis, L. L., Redford, K. H., Shaffer, M. L., Scott, J. M., ...
632 Anderson, M. G. (2002). Planning for Biodiversity Conservation: Putting Conservation
633 Science into Practice: A seven-step framework for developing regional plans to
634 conserve biological diversity, based upon principles of conservation biology and
635 ecology, is being used extensively by the nature conservancy to identify priority areas
636 for conservation. *AIBS Bulletin*, 52, 499-512.
- 637 Guerrero, A. M., McAllister, R. Y. A. N., Corcoran, J., & Wilson, K. A. (2013). Scale
638 mismatches, conservation planning, and the value of social-network
639 analyses. *Conservation Biology*, 27, 35-44.

- 640 Han, X., Smyth, R. L., Young, B. E., Brooks, T. M., de Lozada, A. S., Bubb, P., ... & Turner,
641 W. R. (2014). A biodiversity indicators dashboard: Addressing challenges to monitoring
642 progress towards the Aichi biodiversity targets using disaggregated global data. *PloS*
643 *one*, 9, e112046.
- 644 Hanski, I. (2011). Habitat loss, the dynamics of biodiversity, and a perspective on
645 conservation. *AMBIO: A Journal of the Human Environment*, 40, 248-255.
- 646 Harris, G., Pimm, S.L. (2008) Range size and extinction risk in forest birds. *Conservation*
647 *Biology*, 22, 163–171.
- 648 Hill, S. L., Harfoot, M., Purvis, A., Purves, D. W., Collen, B., Newbold, T., Mace, G. M.
649 (2016). Reconciling biodiversity indicators to guide understanding and
650 action. *Conservation Letters*, 9, 405-412.
- 651 Hoekstra, A. Y., & Wiedmann, T. O. (2014). Humanity's unsustainable environmental
652 footprint. *Science*, 344, 1114-1117.
- 653 INPUT (2016) The expansion of soybean production in the Cerrado. Available:
654 [http://www.inputbrasil.org/wp-content/uploads/2016/11/The-expansion-of-soybean-](http://www.inputbrasil.org/wp-content/uploads/2016/11/The-expansion-of-soybean-production-in-the-Cerrado_Agroicone_INPUT.pdf)
655 [production-in-the-Cerrado_Agroicone_INPUT.pdf](http://www.inputbrasil.org/wp-content/uploads/2016/11/The-expansion-of-soybean-production-in-the-Cerrado_Agroicone_INPUT.pdf). Accessed: January 2018.
- 656 Instituto Brasileiro de Geografia e Estatística (IBGE). 2004. Borders of Brazilian biomes.
657 <http://maps.lapig.iesa.ufg.br/lapig.html>. Accessed August 2016.
- 658 Instituto Brasileiro de Geografia e Estatística (IBGE). 2014. COBERTURA E USO DA
659 TERRA DO BRASIL 2000, 2010, 2012, 2014.
660 [ftp://geoftp.ibge.gov.br/informacoes_ambientais/cobertura_e_uso_da_terra/mudancas/vet](ftp://geoftp.ibge.gov.br/informacoes_ambientais/cobertura_e_uso_da_terra/mudancas/vetores/)
661 [ores/](ftp://geoftp.ibge.gov.br/informacoes_ambientais/cobertura_e_uso_da_terra/mudancas/vetores/). Accessed: August 2016.
- 662 Instituto Brasileiro de Geografia e Estatística (IBGE). 2015. Mudança na Cobertura e Uso da
663 Terra 2000, 2010 and 2012. [https://biblioteca.ibge.gov.br/index.php/biblioteca-](https://biblioteca.ibge.gov.br/index.php/biblioteca-catalogo?view=detalhes&id=294724)
664 [catalogo?view=detalhes&id=294724](https://biblioteca.ibge.gov.br/index.php/biblioteca-catalogo?view=detalhes&id=294724). Accessed: August 2016
- 665 IUCN (2001) IUCN Red List categories and criteria Version 3.1. Gland, Switzerland and
666 Cambridge, UK: IUCN Species Survival Commission
- 667 IUCN 2017. IUCN Red List of Threatened Species. Version 2017-1 www.iucnredlist.org
- 668 Iwamura, T., Possingham, H. P., Chadès, I., Minton, C., Murray, N. J., Rogers, D. I., ...
669 Fuller, R. A. (2013). Migratory connectivity magnifies the consequences of habitat loss
670 from sea-level rise for shorebird populations. *Proceedings of the Royal Society of*
671 *London B: Biological Sciences*, 280, 20130325.
- 672 Joppa, L. N., O'Connor, B., Visconti, P., Smith, C., Geldmann, J., Hoffmann, M., ... &
673 Ahmed, S. E. (2016). Filling in biodiversity threat gaps. *Science*, 352, 416-418.
- 674 Krauss, J., Bommarco, R., Guardiola, M., Heikkinen, R. K., Helm, A., Kuussaari, M., ... &
675 Pöyry, J. (2010). Habitat fragmentation causes immediate and time-delayed biodiversity
676 loss at different trophic levels. *Ecology letters*, 13, 597-605.
- 677 Lambin, E. F., & Meyfroidt, P. (2011). Global land use change, economic globalization, and
678 the looming land scarcity. *Proceedings of the National Academy of Sciences*, 108,
679 3465-3472.

- 680 Lockwood, J.A. (2004). *Locust: the devastating rise and mysterious disappearance of the*
681 *insect that shaped the American frontier*. USA, Basic Books.
- 682 Mantyka-pringle, C. S., Martin, T. G., & Rhodes, J. R. (2012). Interactions between climate
683 and habitat loss effects on biodiversity: a systematic review and meta-analysis. *Global*
684 *Change Biology*, 18, 1239-1252.
- 685 Martinelli, G. and Moraes, M.A. (2013). *Livro vermelho da flora do Brasil*. Brazil, IUCN.
- 686 Moran, D., & Kanemoto, K. (2017). Identifying species threat hotspots from global supply
687 chains. *Nature Ecology & Evolution*, 1, 0023.
- 688 Myers, N. (1993). Biodiversity and the precautionary principle. *Ambio*, 74-79.
- 689 Newbold, T., Hudson, L. N., Hill, S. L., Contu, S., Gray, C.L., Scharlemann, J.P.W.,
690 ...Purvis, A. (2015). Global effects of land use on local terrestrial
691 biodiversity. *Nature*, 520, 45.
- 692 Pearson, R. G., Stanton, J. C., Shoemaker, K. T., Aiello-Lammens, M. E., Ersts, P. J.,
693 Horning, N., ... Akçakaya, H. R. (2014). Life history and spatial traits predict extinction
694 risk due to climate change. *Nature Climate Change*, 4, 217.
- 695 Pfeifer, M., Lefebvre, V., Peres, C. A., Banks-Leite, C., Wearn, O. R., Marsh, C. J., ...
696 Cisneros, L. (2017). Creation of forest edges has a global impact on forest
697 vertebrates. *Nature*, 551, 187.
- 698 Rio+20 United Nations Conference on Sustainable Development. Available:
699 <http://www.uncsd2012.org/>. Accessed July 2017
- 700 Richards, P. (2015). What drives indirect land use change? How Brazil's agriculture sector
701 influences frontier deforestation. *Annals of the Association of American*
702 *Geographers*, 105, 1026-1040.
- 703 Rondinini, C., Di Marco, M., Chiozza, F., Santulli, G., Baisero, D., Visconti, P., ... Amori, G.
704 (2011). Global habitat suitability models of terrestrial mammals. *Philosophical*
705 *Transactions of the Royal Society B: Biological Sciences*, 366, 2633-2641.
- 706 Rondinini, C., Stuart, S., & Boitani, L. (2005). Habitat suitability models and the shortfall in
707 conservation planning for African vertebrates. *Conservation Biology*, 19, 1488-1497.
- 708 Drever, R. C., Drever, M. C., & Sleep, D. J.H. (2012). Understanding rarity: A review of
709 recent conceptual advances and implications for conservation of rare species. *The*
710 *Forestry Chronicle*, 88, 165-175.
- 711 de Ruiter, H., Macdiarmid, J. I., Matthews, R. B., Kastner, T., Lynd, L. R., & Smith, P. (2017).
712 Total global agricultural land footprint associated with UK food supply 1986–
713 2011. *Global environmental change*, 43, 72-81.
- 714 van Soesbergen, A., Arnell, A. P., Sassen, M., Stuch, B., Schaldach, R., Göpel, J., ...
715 Palazzo, A. (2017). Exploring future agricultural development and biodiversity in
716 Uganda, Rwanda and Burundi: a spatially explicit scenario-based
717 assessment. *Regional Environmental Change*, 17, 1409-1420.
- 718 Sparks, T. H., Butchart, S. H., Balmford, A., Bennun, L., Stanwell-Smith, D., Walpole, M., ...
719 Collen, B. (2011). Linked indicator sets for addressing biodiversity loss. *Oryx*, 45, 411-
720 419.

- 721 Strassburg, B. B., Brooks, T., Feltran-Barbieri, R., Iribarrem, A., Crouzeilles, R., Loyola, R.,
722 ... Soares-Filho, B. (2017). Moment of truth for the Cerrado hotspot. *Nat. Ecol. Evol*, 1, 1-
723 3.
- 724 The Ramsar Convention on Wetlands. Available: <http://www.ramsar.org/>. Accessed July
725 2017
- 726 Thomas, C. D., Cameron, A., Green, R. E., Bakkenes, M., Beaumont, L. J., Collingham, Y.
727 C., ... Hughes, L. (2004). Extinction risk from climate change. *Nature*, 427, 145.
- 728 United Nations Millennium Development Goals. Available:
729 <http://www.un.org/millenniumgoals/>. Accessed July 2017.
- 730 United States Geological Survey. 2006 Shuttle Radar Topography Mission 3 arc second
731 version 2.0. See <http://www.landcover.org/data/srtm> (accessed November 2016).
- 732 Veach, V., Di Minin, E., Pouzols, F. M., & Moilanen, A. (2017). Species richness as criterion
733 for global conservation area placement leads to large losses in coverage of
734 biodiversity. *Diversity and Distributions*, 23, 715-726.
- 735 Visconti, P., Pressey, R. L., Giorgini, D., Maiorano, L., Bakkenes, M., Boitani, L., ...
736 Rondinini, C. (2011). Future hotspots of terrestrial mammal loss. *Philosophical*
737 *Transactions of the Royal Society of London B: Biological Sciences*, 366, 2693-2702.
- 738 Visconti, P., Bakkenes, M., Baisero, D., Brooks, T., Butchart, S. H., Joppa, L.,... Maiorano, L.
739 (2016). Projecting global biodiversity indicators under future development
740 scenarios. *Conservation Letters*, 9, 5-13.
- 741 World Wildlife Fund, 2015. The Big Five of the Cerrado.
742 <http://www.wwf.org.br/informacoes/english/?50242/The-Big-Five-of-the-Cerrado>,
743 Accessed August 2016.

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745 **DATA ACCESSIBILITY**

746 All data used in this study are freely available online (Please see references for more
747 details). Species ranges were obtained from: ([https://www.iucnredlist.org/resources/spatial-](https://www.iucnredlist.org/resources/spatial-data-download)
748 [data-download](https://www.iucnredlist.org/resources/spatial-data-download) - Mammals & Amphibians), (<http://datazone.birdlife.org/species/requestdis> -
749 Birds), and (<https://tinyurl.com/y7zxzxhv>- Plants). Habitat preferences for vertebrates,
750 including altitudinal ranges, we obtained from (<http://apiv3.iucnredlist.org/api/v3/docs>). Land
751 cover and boundary data for Brazil were obtained from (<https://www.lapig.iesa.ufg.br/lapig/>).
752 Digital elevation data was obtained from (https://lta.cr.usgs.gov/products_overview). Soy
753 expansion data were obtained from (Gibbs et al., 2015).

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