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14 15 **Global effects of land use on local terrestrial biodiversity**

16
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62

63 **Human activities, especially conversion and degradation of habitats, are causing global**
64 **biodiversity declines. How local ecological assemblages are responding is less clear – a**
65 **concern given their importance for many ecosystem functions and services. We analyze**
66 **a terrestrial assemblage database of unprecedented geographic and taxonomic coverage**
67 **to quantify local biodiversity responses to land-use and related changes. In the worst-**
68 **impacted habitats, these pressures reduce within-sample species richness by an average**
69 **of 76.5%, total abundance by 39.5% and rarefaction-based richness by 40.3%. We**
70 **estimate that, globally, these pressures have already slightly reduced average within-**
71 **sample richness (by 13.6%), total abundance (10.7%) and rarefaction-based richness**
72 **(8.1%), with changes showing marked spatial variation. Rapid further losses are**
73 **predicted under a business-as-usual land-use scenario: within-sample richness is**
74 **projected to fall by a further 3.4% globally by 2100, with losses concentrated in**
75 **biodiverse but economically poor countries. Strong mitigation can deliver much more**
76 **positive biodiversity changes (up to a 1.9% average increase) that are less strongly**
77 **related to countries' socioeconomic status.**

78

79 Biodiversity faces growing pressures from human actions, including habitat conversion and
80 degradation, habitat fragmentation, climate change, harvesting and pollution¹. As a result,
81 global assessments show that species' extinction risk is increasing on average while
82 population sizes are declining^{1,2}. Such assessments have usually focused on data-rich
83 vertebrates, so might not reflect broader biodiversity³. Furthermore, most have concentrated
84 on the global status of species, whereas the long-term security of many ecosystem functions
85 and services – especially in changing environments – is likely to depend upon local
86 biodiversity⁴⁻⁶. Average trends in local diversity remain unclear: analyses of temporal
87 changes in assemblages have suggested no systematic change in species richness^{7,8}, but the
88 available times-series data might under-represent transitions between land-use types⁹ and
89 population time series suggest vertebrate populations have declined sharply in recent
90 decades³.

91 Spatial comparisons provide an alternative source of evidence on how human pressures
92 affect biodiversity, assuming that differences in pressures have caused observed biodiversity
93 differences between otherwise matched sites¹⁰⁻¹². The prevalence of published spatial
94 comparisons makes it possible to go beyond particular taxa or regions^{11,12} to develop global,
95 taxonomically representative models. Furthermore, the willingness of many researchers to
96 share their raw data makes it possible to consider multiple aspects of biodiversity, rather than
97 the single, simple metrics of most existing models¹⁰ which cannot capture all key aspects of
98 diversity¹³.

99 We present the most geographically and taxonomically representative models to date of
100 how several aspects of the composition and diversity of terrestrial assemblages respond to

101 multiple human pressures. The explanatory variables in our models most directly measure
102 land use and infrastructure, but might correlate^{14,15} with two other important pressures –
103 harvesting and invasive species – for which comparable high-resolution spatial data are
104 unavailable globally. We exclude climate change effects because they are not captured well
105 by spatial comparisons. We use our models to infer past net changes in assemblages since
106 1500, project future changes over this century under different socioeconomic scenarios of
107 land use, and relate projected national changes in local biodiversity to socioeconomic
108 variables and natural biodiversity.

109 Our models of local within-sample species richness (hereafter ‘richness’), rarefaction-
110 based species richness (hereafter ‘rarefied richness’), total abundance, compositional turnover
111 and average organism size are based on among-site comparisons of ecological assemblage
112 composition collated from the literature as part of the PREDICTS project¹⁶. The dataset
113 consisted of 1,130,251 records of abundance and 320,924 of occurrence or species richness at
114 11,525 sites (2-360 sites per study, median 15; Figure 1a). These data, from 284 publications
115 (see Methods), represent 26,953 species – 1.4% of the number formally described¹⁷ – and 13
116 of the 14 terrestrial biomes (Extended Data Figure 1). Each site was scored for six putative
117 pressures: land use¹¹ and use intensity¹⁸, land-use history¹⁹, human population density²⁰,
118 proximity to roads²¹, and accessibility from the nearest large town. Random effects in our
119 models accounted for study-level differences in response variables and sampling methods,
120 and for the within-study spatial arrangement of sites.

121

122 *Effects of pressures on site-level diversity*

123

124 Local richness, rarefied richness and total abundance were most strongly influenced by land
125 use and land-use intensity: they were substantially lower in most other land-use types than in
126 primary vegetation, especially in intensively-used areas (Figure 1; see Supplementary
127 Information for statistics and coefficient estimates). These results extend those of previous,
128 geographically or taxonomically restricted, meta-analyses (e.g. refs. 11,22). Other variables
129 were weaker as main effects, but showed stronger effects in interaction (Extended Data
130 Figure 2) and were often significant overall (see Supplementary Information). Richness and
131 total abundance tended to be slightly lower at the highest human population densities, and
132 richness was lower nearer to roads and in more accessible sites (Figure 1). Differences in
133 richness were not driven solely by differences in abundance: rarefied richness²³ (see Methods
134 for details) showed weaker but mostly similar patterns, although the effects of variables other
135 than land use and land-use intensity were not significant (Extended Data Figure 3a,b). Under
136 the worst combinations of pressures, our models estimated richness, rarefied richness and
137 total abundance to be 76.5%, 40.3% and 39.5% lower, respectively, than in minimally
138 impacted sites. Effects of pressures on vertebrate, invertebrate and plant richness were
139 statistically indistinguishable ($P > 0.05$; results not shown). The modelled coefficients were
140 robust to efforts to correct for publication bias (Extended Data Figure 4). As with all studies
141 based on data from the literature, unpublished data are almost unrepresented. Coefficients
142 were also robust under cross-validation (Extended Data 3c,d), and the model residuals
143 showed little spatial autocorrelation (Extended Data Figure 5).

144 The importance of secondary vegetation for conservation is hotly debated^{11,24,25}, and is
145 crucial because this land-use will soon become the most widespread type²⁶. We find that the
146 answer depends strongly on the secondary vegetation’s maturity: early-stage communities
147 tend to be less diverse than those in primary vegetation and are compositionally distinct, but
148 these differences are much reduced in mature secondary vegetation (Figures 1 & 2; we
149 caution though that not all data sources clearly distinguished mature secondary from primary

150 vegetation). This successional rise in diversity accords with a recent meta-analysis of plant
151 communities over time⁷.

152 Net changes in diversity provide an incomplete view of the effects of human activities
153 on biodiversity because they ignore replacement of original species by newcomers⁸. We
154 therefore analysed how land use affects similarity in species composition between sites.
155 Communities under the same land use were, unsurprisingly, the most similar (Figure 2a).
156 Across land uses, communities in primary vegetation were most like those in secondary
157 vegetation, while plantation forest, pasture and cropland communities formed a different,
158 human-dominated, cluster (Figure 2b).

159 Anthropogenic pressures can affect ecosystem functions and services more strongly
160 than changes in species diversity would imply, if species' responses depend on their traits²⁷.
161 Large size is often linked to species' declines^{28,29} and is important for some ecosystem
162 processes³⁰. We combined abundance data with species' average sizes to calculate site-level
163 community-weighted mean plant height and animal mass. As in local studies²⁹, mean plant
164 height was lower in human-dominated land uses than in primary and secondary vegetation,
165 and tended to decline with increasing human population density (Figure 1d). Most field
166 studies focused on particular plant taxa, so this difference does not simply reflect tree
167 removal. Average animal mass did not change consistently with land use or human
168 population density, but increased with proximity to roads (Figure 1d).

169 Models like ours that substitute space for time ignore time lags in biotic changes,
170 which can be important³¹. We also assume that land uses are situated randomly within studies
171 relative to sites' intrinsic suitability for biodiversity. Adding global data on other important
172 pressures as they become available, and also incorporating climate change, will give a more
173 complete picture of human impacts on local biodiversity.

174
175 *Global effects on local diversity to date*

176
177 By applying our model for within-sample species richness – the most widely used and
178 understood biodiversity measure – to maps of current pressure variables, we estimate the
179 global pattern of net local changes to date in plot-level richness (Figure 3; we did not estimate
180 total richness within the 0.5° × 0.5° grid cells). Human-dominated areas are inferred to have
181 lost much more local diversity than have regions where more natural vegetation remains. The
182 worst-affected cells showed a 31% reduction in average local richness – probably enough to
183 impact ecosystem functioning substantially⁵. Local richness increased in 1.7% of cells (by ≤
184 4.8%). Total abundance and rarefied richness showed broadly similar patterns, although less
185 pronounced in the latter case (Extended Data Figure 6).

186 We applied our models to global spatial estimates of how land use and human
187 population changed from 1500-2005²⁶ (see Methods) to infer the global history of local
188 biodiversity change. We focus on within-sample species richness because of its wide use and
189 easy interpretation. Our inferences incorporate uncertainty in model parameter estimates, but
190 not in the trajectories of the drivers themselves (which have not been assessed³²) nor effects
191 of changes in roads and accessibility, for which temporal estimates could not be obtained.

192 Richness is estimated to have declined most rapidly in the 19th and 20th centuries
193 (Figure 4), with other metrics showing similar responses (Extended Data Figure 6). By 2005,
194 we estimate that land use and related pressures had reduced local richness by an average of
195 13.6% (95% CI: 9.1 – 17.8%) and total abundance by 10.7% (95% CI: 3.8% gain – 23.7%
196 reduction) compared with what they would have been in the absence of human effects.
197 Approximately 60% of the decline in richness was independent of effects on abundance:
198 average rarefied richness has fallen by 8.1% (95% CI: 3.5 – 12.9%). Although these

199 confidence limits omit uncertainty in the projections of land use and other pressures, there is
200 less uncertainty in estimates of current pressure levels than in changes over time³³.

201 Our inferences contrast with two recent analyses of community time series^{7,8}, which
202 suggested no overall trend in local diversity, and with the Living Planet Index³ which, based
203 on vertebrate population time series, reports a much more rapid decline in abundance than we
204 infer. Although time series potentially provide a more direct view of temporal trends than our
205 space-for-time approach, the available data might under-represent transitions between land-
206 use types⁹. On the other hand, our approach may underestimate additions of species through
207 climate change and species invasion (although accessibility and proximity to roads may
208 partly capture the latter^{14,15}).

209

210 *Global and national projections to 2095*

211 Global changes in local diversity from 2005 to 2095 were projected using estimated land use
212 and human population from the four Intergovernmental Panel on Climate Change
213 Representative Concentration Pathway (RCP) scenarios²⁶, which correspond to different
214 intensities of global climate change (Table 1). Although these estimates have limitations³²,
215 they are the most consistent available, are widely-used³⁴, and are consistent with the
216 historical estimates²⁶. However, they – like all other global land-use projections – include no
217 estimate of uncertainty; therefore, each of our projections must be viewed as the predicted
218 biodiversity outcome under one particular set of land-use assumptions.

219 Projected net changes in average local diversity to 2095 vary widely among scenarios
220 (Figure 4; Extended Data Figure 6). The scenario with the least climate change (IMAGE 2.6)
221 yields the second-worst outcome for biodiversity, because it assumes rapid conversion of
222 primary vegetation – especially in the tropics – to crops and biofuels²⁶ (Table 1, Extended
223 Data Figure 7). These projections do not imply that low-emission scenarios must entail large
224 losses of biodiversity, but instead reflect that scenario's mitigation strategy. Indeed, in
225 MiniCAM 4.5 (where mitigation is through carbon markets, crop improvements and diet
226 shifts: Table 1), average richness is projected to increase (though other diversity metrics
227 respond more weakly: Extended Data Figure 6). The worst biodiversity outcomes arise from
228 the scenario with most climate change – MESSAGE 8.5 – in which rapid human population
229 growth drives widespread agricultural expansion (Table 1; Extended Data Figure 7). This
230 scenario, which has been characterised as 'business-as-usual'³⁵, most closely matches recent
231 trends in emissions³⁶ and gives the worst outcomes even though our projections omit direct
232 climate impacts on local assemblages.

233 The global projections hide wide regional and national variation (Figure 5; Extended
234 Data Figure 8). Projections for 2095 under 'business-as-usual' (MESSAGE 8.5) are strongly
235 inequitable, presenting serious challenges for both sustainable development and global
236 conservation of biodiversity (Figure 5a). Under this scenario, European and North American
237 countries – typically with a high Human Development Index (HDI), low native biodiversity
238 and widespread historical land conversion – are mostly projected to gain in local richness by
239 2095. More naturally biodiverse but less economically developed Southeast Asian and
240 especially sub-Saharan African countries, with more natural and semi-natural habitat, will
241 suffer the greatest losses (Figure 5a; Extended Data Figure 8f).

242 Such globally inequitable outcomes might be avoidable: the best scenario for
243 biodiversity (MiniCAM 4.5; Figure 4) yielded country-level outcomes that are relatively
244 independent of HDI, native species richness (Figure 5b) and past changes (Extended Data
245 Figure 8e). For local richness, outcomes under MiniCAM4.5 were better than MESSAGE 8.5
246 for 93% of countries worldwide (Figure 5c).

247 Under AIM 6.0, most Afrotropical countries are projected to gain in local richness but
248 heavy losses are inferred for the Indo-Malay region (Extended Data Figure 8). Projections

249 under IMAGE 2.6 are spatially similar to those under MESSAGE 8.5. The land-use change
250 caused by the biofuels-based strategy in IMAGE 2.6 is projected to have a major negative
251 impact overall on terrestrial biodiversity (Extended Data Figure 8).

252

253 *Conclusions*

254 Many assessments of the state of biodiversity have focused on global metrics such as rates of
255 species extinction³⁷, but resilient delivery of ecosystem functions and services is more likely
256 to depend on local diversity^{4–6}. Our models suggest land-use changes and associated
257 pressures strongly reduce local terrestrial biodiversity, and we estimate global average
258 reductions to date of 13.6% in within-sample species richness, 10.7% in total abundance and
259 8.1% in rarefaction-based species richness (Figs. 3 and 4). Climate change, which we could
260 not include in our framework, is likely to exacerbate losses, especially under business-as-
261 usual³⁸, although direct effects of climate change will increase local diversity in some
262 regions⁸.

263 It is important to remember that the habitat conversion and associated changes that
264 reduced local biodiversity had largely positive consequences for people: agricultural
265 intensification underpinned many countries' development. However, benefits have not been
266 shared equally among or within countries³⁹. Losses of local species richness exceeding 20%
267 are likely to substantially impair the contribution of biodiversity to ecosystem function and
268 services, and thus to human well-being⁵. We estimate that reductions in average plot-level
269 species richness currently exceed this level for 28.4% of grid cells, increasing to 41.5% of
270 cells by 2095 under 'business-as-usual' (note that we do not estimate or project total richness
271 across the cell). Importantly, our projections suggest that such widespread large losses are not
272 inevitable. With concerted action and the right societal choices, global sustainability of local
273 biodiversity may be an achievable goal.

274

275

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385
386 **Supplementary Information** is linked to the online version of the paper at
387 www.nature.com/nature.

388

389 **Additional references** can be found in the Supplementary Information.

390

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412

413 **Author contributions**

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420 R.A.S., S.D., M.E., A.F., Y.I., J.K., M.K., S.M. and E.W. made substantial contributions to
421 the trait data used in the analyses and S.T. to the site-level environmental data; R.A.S., A.F.,
422 Y.I., S.M., and M.N. generated the maps of species richness used in the model projections;
423 T.N., L.N.H. and A.P. wrote the manuscript with contributions from G.M.M., L.B., D.W.P.,
424 R.M.E., A.D.P., H.R.P.P., S.L.L.H., R.A.S., B.C., S.D., A.F., Y.I., J.K., M.K., S.M., J.P.W.S
425 and S.T.; T.N. and L.N.H. contributed equally to the study.

426

427 **Author information**

428

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430 material should be addressed to T.N. (Tim.Newbold@unep-wcmc.org).

431

432 **Tables**

433

434

435 **Table 1. Key features of the four Representative Concentration Pathways (RCP)**
 436 **scenarios** (land-use and human-population assumptions are detailed in ref. 26, energy
 437 assumptions in refs. 40–42, and climate implications in ref. 43).

Scenario	Land use (see also Extended Data Figure 7)	Climate and energy	Human population
IMAGE 2.6	Agriculture moves from developed to developing countries. Large increase in area of biofuel plantations. Urban extent assumed constant.	Increased energy efficiency. Increased use of carbon capture and storage, nuclear, renewable energy and biofuels. Approximately 1 °C temperature increase by 2100 compared to pre-industrial.	10.1 billion by 2100 (UN Medium variant, 2010)
MiniCAM 4.5	Carbon pricing leads to preservation of primary forest and expansion of secondary forest. Crop yield increases, improved agricultural efficiency and dietary shifts lead to decreases in cropland and pasture areas. Small increase in area of biofuel plantations. Urban extent assumed constant.	Decline in overall energy use. Decreased use of fossil fuels and increase in nuclear and renewable energy, and in carbon capture and storage. Moderate increase in use of biofuels, but limited by availability of biomass. Approximately 1.75 °C temperature increase by 2100.	Peaks above 9 billion around 2065, then declines to 8.7 billion by 2095.
AIM 6.0	Urban area increases owing to human population growth. Cropland area increases to meet food demand. Pasture area declines strongly.	Approximately 2.5 °C temperature increase by 2100.	9.1 billion by 2100 (UN Medium variant, 2004)
MESSAGE 8.5	Increasing crop yields and intensification account for much of the increased production required, but area of cropland and, to a lesser extent, pasture increases rapidly. Small increase in area of biofuel plantations. Urban area increases owing to increased population.	Small improvements in efficiency leading to high demand for energy. Conventional oil and gas become scarce, leading to shift in favour of unconventional and carbon-intensive fossil fuels. Moderate increase in use of biofuels. Approximately 4 °C increase in temperature by 2100.	12 billion by 2100.

438
439

440 **Figure legends**

441

442 **Figure 1. Locations of sites and responses of four metrics of local diversity to human**
443 **pressures. a**, Sites used in the models. Responses⁴⁴ of richness, **b**, total abundance, **c**, and
444 community-weighted mean organism size – plant height (crosses) and animal mass
445 (triangles), **d**, to anthropogenic variables. Error bars show 95% confidence intervals. Primary
446 = primary vegetation, YSV = young secondary vegetation, ISV = intermediate secondary
447 vegetation, MSV = mature secondary vegetation, Plantation = plantation forest. Land-use
448 intensity: minimal (circle), light (triangle), intense (diamond), and combined light and intense
449 (square). HPD = human population density⁴⁵, PR = proximity to roads⁴⁶ (as $-\log(\text{distance to}$
450 $\text{nearest road})$), ACC = accessibility to humans⁴⁷ (as $-\log(\text{travel time to nearest major city})$)
451 are shown as fitted effects from a model with no interactions between continuous effects and
452 land use, at the lowest (L), median (M) and highest (H) values in the dataset. Sample sizes are
453 given in full in the Methods (Table S2).

454

455 **Figure 2. Similarity in assemblage composition as a function of land use. a**, Average
456 dissimilarity of species composition ($1 - \text{Sørensen Index}$) between pairs of sites within and
457 among land uses (shown relative to the similarity between pairs of primary-vegetation sites);
458 blue and red colours indicate, respectively, more and less similar composition; numbers
459 indicate numbers of studies within which comparisons could be made. **b**, Clustering of land-
460 use types based on average compositional dissimilarity; urban sites were excluded owing to
461 the small sample size. Land-use labels as in Figure 1.

462

463 **Figure 3. Net change in local richness caused by land use and related pressures by 2000.**
464 Projections used an IMAGE reference scenario¹⁰. The baseline landscape was assumed to be
465 entirely uninhabited, unused primary vegetation. Shown using a Lambert Cylindrical Equal-
466 Area projection at $0.5^\circ \times 0.5^\circ$ resolution.

467

468 **Figure 4. Projected net change in local richness from 1500 to 2095.** Future projections
469 were based on the four RCP scenarios (Table 1). Historical (shading) and future (error bars)
470 uncertainty is shown as 95% confidence intervals, rescaled to zero in 2005. The baseline for
471 projections is a world entirely composed of uninhabited, unused primary vegetation; thus, the
472 value at 1500 is not constrained to be zero because by then non-primary land uses were
473 present – and in some regions widespread. The global average projection for MESSAGE 8.5
474 does not join the historical reconstruction because that scenario's human-population
475 projections start in 2010 and because human population and plantation-forest extent have not
476 been harmonized among scenarios.

477

478 **Figure 5. Biodiversity projections at the country level. a-b**, Country-level projections of
479 average net local richness change between 2005 and 2095 under the worst (**a**, MESSAGE
480 8.5) and best (**b**, MiniCAM 4.5) RCP scenarios for biodiversity, shown in relation to the
481 Human Development Index. Colours indicate biogeographic realms; colour intensity reflects
482 natural vertebrate species richness (more intense = higher richness); point size is proportional
483 to (log) country area. **c**, Correlation between projected richness changes under the MiniCAM
484 4.5 and MESSAGE 8.5 scenarios, with dashed line showing equality; colours as in **a and b**;
485 colour intensity is proportional to the HDI (more intense = higher index).

486

487 **Methods**

488

489 *Data collation*

490

491 Between March 2012 and April 2014 we collated among-site comparisons of ecological
492 assemblage composition from published studies (or from unpublished datasets where the
493 methods have been published) examining the effect of human activities on more than one
494 named taxon. A full description of how the dataset was assembled and curated is presented
495 elsewhere¹⁶. We define sites to be in the same study if they were sampled using the same
496 methodology and the data were reported in the same paper; therefore, some publications
497 contain multiple studies. After six months of broad searching, we targeted efforts towards
498 under-represented taxa, habitat types, biomes and regions. We accepted data only from
499 published or in-press papers, or data collected using a published methodology, and we
500 required that the data providers agreed to our making their data publicly available at the end
501 of the PREDICTS (Projecting Responses of Ecological Diversity In Changing Terrestrial
502 Systems) project in 2015. We accepted data only where abundance, occurrence or richness
503 had been measured at two or more sampling locations and/or times, and where all sites were
504 sampled using the same sampling procedure and with either the same effort or site-specific
505 data on effort. We used geographical coordinates preferentially from the paper or supplied by
506 data providers; but where coordinates were not thus available, we georeferenced them from
507 maps in the papers. The final dataset came from 378 studies⁴⁹⁻³³² and two unpublished
508 datasets (M. E. Hanley, 2005 and 2011) that were collected with published methods¹⁴⁶.

509 Studies compared from 2 to 360 sites (median = 15; 82% had ≥ 5 sites); most sampled
510 species from multiple families but fewer than half sampled multiple orders. Over 70% of sites
511 were from studies that sampled entire communities within a taxonomic group rather than a
512 target list of species. Removing studies having a target list did not substantially alter model
513 coefficients (results not shown) and increased the projected global net average loss of local
514 species richness until 2005 by 0.6%. Sites varied in the maximum linear extent sampled
515 (median 106 m; interquartile range 50 m to 354 m). Model coefficients for the approximately
516 50% of studies that reported maximum linear extent were robust to its inclusion in the models
517 (results not shown).

518 The great majority of listed taxa were species-level, although many could not be given
519 explicit species identifications (e.g. morpho-species)¹⁶; henceforth we refer to distinct taxa in
520 our dataset as species. We matched taxonomic names given in the source paper to the
521 Catalogue of Life 2013 Annual Checklist (COL)³³³, obtaining the full taxonomic
522 classification. In order to relate the taxonomic names to species-level trait databases, we
523 generated, for each taxon, a 'best-guess' Latin binomial as: a) the taxon name from COL if
524 the COL query returned a species-level identification; b) the first two words of the text
525 returned by the COL query if this was a sub-species designation; c) the first two words of the
526 taxon name in the source publication if the COL query returned neither a species or sub-
527 species name, and the taxon name in the source publication contained two or more words.
528 Taxa that met none of these criteria were not matched to trait data, but were included in the
529 calculation of richness and total abundance, and for estimating turnover in community
530 composition among sites.

531 The resulting data set contained data for 26,953 species at 11,525 sites. For many high-
532 diversity taxa, the database contains data for more than 1% of the number of species thought
533 to have been formally described (Extended Figure 1a). The distribution of sites among major
534 biomes is roughly proportional to the amount of terrestrial net primary productivity (NPP)
535 fixed within each biome (Extended Figure 1b).

536

537 *Site-level composition and diversity*

538

539 We computed four site-level biodiversity metrics: within-sample species richness, total
540 abundance, rarefaction-based richness and community-weighted mean organism size. These
541 were calculated as follows.

542
543 **Within-sample species richness** was calculated as the number of differently-named taxa
544 recorded at a given site in a standardised sampling unit (a measure also known as species
545 density³³⁴). We gave precedence to the author's classification of species, even where a search
546 of global databases revealed potential synonymies, because only certain taxonomic groups
547 could be reliably matched to accepted taxonomies.

548 This measure of richness is appropriate for conservation questions but among site-
549 differences could be due to effects on numbers of individuals as well as to changes in the
550 shape of the species accumulation curve³³⁴. We therefore also calculated **rarefaction-based**
551 **species richness** by taking 1000 random samples of n individuals from each site, where n is
552 the smallest total number of individuals recorded at any site within its study, and calculating
553 the mean species richness across samples. This index could only be calculated for sites
554 where, in addition to the criteria above being met, abundance was recorded as number of
555 individuals. Rarefied species richness was rounded to the nearest integer for analysis with
556 Poisson errors.

557
558 **Total abundance** was calculated as the sum of the measures of abundance of all taxa at a
559 site; we were thus unable to estimate abundance for sites where only species occurrence or
560 overall richness or diversity had been recorded (17% of sites). Some abundance metrics –
561 those not reported as densities per unit time, distance, area or volume sampled – were
562 sensitive to sampling effort. When a study reported any of these metrics and sampling effort
563 varied among sites within a study, we corrected the raw abundance measurements for the
564 sampling effort expended at each sampling location and time; this was done by rescaling the
565 sampling efforts within each study, so that the most heavily sampled site had a value of one,
566 and then dividing the raw abundance measurements by this relative sampling effort. The
567 rescaling was performed to prevent introducing additional heterogeneity in the modelled
568 abundance values.

569
570 **Community-weighted mean organism size** was calculated as the arithmetic mean of log-
571 transformed height of plants (available for 4,235 species in our dataset) or the log-
572 transformed body mass or volume of vertebrates, beetles and hoverflies (5,236 species)
573 present at a site, weighted by abundance³³⁵. Plant height data were taken from the TRY
574 database³³⁶; for 61 species where plant vegetative height data were unavailable, we estimated
575 it from generative height from a regression across the 2,554 species with estimates of both
576 traits ($R^2 = 0.91$). Data on vertebrate body mass were taken from the PanTHERIA database
577 for mammals³³⁷, from BirdLife International's World Bird Database for birds, and from a
578 wide range of published and grey-literature sources for amphibians^{338–384}. Length data for
579 reptiles were taken from published^{385,386} and unpublished (S. Meiri & A. Feldman,
580 unpublished data) sources, and converted to estimates of body mass using published length-
581 mass allometries^{387,388}. Arthropod size data (beetles and hoverflies) were collated from
582 published sources^{389,390}. Beetle length and amphibian snout-vent length values were raised to
583 the power three so that they had the same dimensionality as the other animal size measures.
584 For both plant height and vertebrate body mass, missing values were interpolated as the
585 average values for congeners, since both of these traits are strongly conserved
586 phylogenetically (Pagel's $\lambda = 0.98, 0.997, 0.93, 0.89$ for plant height, vertebrate body mass,
587 beetle body length and hoverfly thorax volume, respectively).

588

589 *Human pressure data*

590

591 While many human pressures can impact local biodiversity, we focus on those that can
592 be obtained for sites around the world and for which, as far as possible, spatiotemporal data
593 are available for 1500-2095; this focus enables us to use our statistical models as a basis for
594 projecting responses through time. Each site was assigned to one of eight land-use classes
595 based on the description of the habitat given in the source paper (see Extended Data Table 1
596 for definitions): primary vegetation, secondary vegetation (subdivided into mature,
597 intermediate or young secondary vegetation), plantation forest, cropland, pasture and urban¹⁶.
598 These classes were selected to match the land-use classification adopted in the
599 Intergovernmental Panel on Climate Change Representative Concentration Pathways
600 scenarios²⁶ in order to facilitate the projection of our models onto these scenarios. Sites were
601 also assigned to a level of human intensity of use – minimal, light or intense – within each
602 major land-use class, also based on the description of the habitat in the source paper (see
603 Extended Data Table 1 for definitions). The factors that determined this level depended on
604 the land-use class (e.g. bushmeat extraction and limited logging in primary and secondary
605 vegetation; or stocking density and chemical inputs in pasture; Extended Data Table 1). Sites
606 that could not be classified for land-use and use intensity were excluded from the analyses.
607 The final dataset contained the following numbers of sites in each land use and land-use
608 intensity level: Primary vegetation, minimal use – 1546 (from 183 studies), light use – 860
609 (76 studies), intense use – 449 (33 studies); mature secondary vegetation, minimal use – 198
610 (52 studies), light/intense use – 213 (23 studies); intermediate secondary vegetation, minimal
611 use – 404 (55 studies), light/intense use – 269 (30 studies); young secondary vegetation,
612 minimal use – 431 (50 studies), light/intense use – 331 (34 studies); plantation forest,
613 minimal use – 356 (47 studies), light use – 402 (42 studies), intense use – 238 (29 studies);
614 cropland, minimal use – 427 (45 studies), light use – 632 (43 studies), intense use – 703 (36
615 studies); pasture, minimal use – 525 (43 studies), light use – 434 (52 studies), intense use –
616 174 (23 studies); and urban, minimal use – 174 (23 studies), light use – 244 (26 studies),
617 intense use – 195 (18 studies).

618 We overlaid our sites with available global data sets to obtain site-level estimates of
619 human population density⁴⁵, distance to the nearest road⁴⁶ and estimated travel time to nearest
620 population centre with greater than 50,000 inhabitants⁴⁷. For distance to nearest road, the map
621 of roads was first projected onto a Berhmann equal-area projection. These operations were
622 carried out using Python code implemented using the arcpy Python module in ArcMap
623 Version 10.0³⁹¹. In the main figures, the inverses of distance to roads and travel time to major
624 population centre (proximity to roads and accessibility) were presented so that high values
625 corresponded to higher hypothesized human impact. To estimate the history of human use of
626 the landscapes within which sites were located, we calculated the number of years since the
627 30-arc-second grid cell containing each site became 30% covered by human land uses
628 (cropland, pasture and urban), according to the HYDE model⁴⁸. Collinearity among variables
629 describing anthropogenic change was low: the highest correlation was between land use and
630 human population density (Pearson $R^2 = 0.31$).

631

632

633 *Modelling site-level diversity, composition and turnover*

634

635 The response of site-level diversity to the measures of anthropogenic change was
636 modelled using generalized linear mixed effects models, implemented in the lme4 package
637 Version 1.0-5⁴⁴ in R Version 3.0.2³⁹². We first compared candidate random-effects structures
638 using the full candidate fixed-effects structure³⁹³. Random-intercept terms considered in all

639 models were the identity of study from which data were taken, to account for study-level
640 differences in the response variables and sampling methods used, and – within-study – the
641 spatial block in which the site was located, to account for the spatial arrangement of sites. For
642 models of species richness (within-sample and rarefied), we also fitted an observation-level
643 random effect (i.e. site identity) to account for the overdispersion present³⁹⁴. We also
644 considered random slopes, with respect to study, of each of the main fixed effects (land use,
645 land-use intensity, human population density, distance to nearest road, travel time to nearest
646 major city and time since the landscape was majority converted to human uses). Random
647 effects were retained or discarded based on Akaike Information Criterion values.

648 Once the best random-effects structure had been selected, we performed backward
649 stepwise model simplification to select the best fixed-effects structure (see ‘Full Statistical
650 Results’)³⁹³. Human population density, distance to roads, travel time to nearest major city
651 and time since major human use of the landscape were log transformed in the analyses, with a
652 value of 1 added to human population density, travel time to nearest major city and time since
653 major landscape conversion to deal with zero values. These four variables were fitted as
654 continuous effects, with quadratic polynomials for human population, distance to roads and
655 travel time to nearest major city, and as a linear effect for time since human landscape
656 conversion. For variables fitted as quadratic polynomials, we also tested linear effects during
657 the backward stepwise model selection. All continuous variables were rescaled prior to
658 analysis so that values ranged between zero and one. Interaction terms were tested first, and
659 then removed to test the main effects. All main effects that were part of significant interaction
660 terms were retained in the final models regardless of their significance as main effects. For
661 the model of community-weighted mean body mass and plant height, because the number of
662 sites with data was smaller than for the other metrics, only land use (excluding urban sites,
663 which were few), human population density and distance to roads, and no interactions, were
664 fitted (for the model of plant height, sample sizes in each land use were: primary vegetation –
665 634 sites, secondary vegetation – 851 sites, plantation forest – 222 sites, cropland – 72 sites,
666 pasture – 412 sites; and for the model of animal mass: primary vegetation – 1728 sites,
667 secondary vegetation – 805 sites, plantation forest – 602 sites, cropland – 641 sites, pasture –
668 440 sites). The decision whether or not to retain terms was based on likelihood ratio tests.
669 The coefficient estimates of the best models are shown in Figure 1b-d and Extended Data
670 Figure 2, and the formulae and statistical results are shown in the Supplementary
671 Information. To test for spatial autocorrelation in the residuals of the final best models, we
672 calculated Moran’s I values and associated P-values, separately for each study considered in
673 the models, using the spdep package Version 0.5-68³⁹⁵ in R; the distribution of P-values
674 across studies was used as an indication of whether spatial autocorrelation was likely to cause
675 a problem. This revealed that the residuals showed little spatial autocorrelation (Extended
676 Data Figure 5). We used cross validation to assess the robustness of model parameter
677 estimates, first based on dividing the studies randomly into 10 equal-sized sets and dropping
678 each set in turn (Extended Data Figure 3c), and secondly based on leaving out the studies
679 from each biome in turn (Extended Data Figure 3d).

680 Publication bias is a potential problem for any large-scale synthesis of data from many
681 publications. In standard meta-analyses, funnel plots³⁹⁶ can be used to test for any
682 relationship between standard error and effect size, as a bias in effect sizes at high standard
683 error toward more positive or more negative effects indicates a likely effect of publication
684 bias. Creating funnel plots for our data was more complicated because ours was a site-level
685 analysis of raw diversity estimates rather than a traditional meta-analysis. Instead we
686 generated individual models relating diversity to land use for each study that sampled at least
687 two sites within each of at least two land-use types. We focused on land use because: a) there
688 were a small number of sites included in most within-study models; and 2) the original

689 studies focused on effects of land use – not generally on land-use intensity, human population
690 density or distance to roads – and thus any effect of publication bias would likely be seen in
691 the land-use coefficients. Funnel plots were generated by plotting, for each land-use type, the
692 estimated model coefficients against the associated standard errors (Extended Data Figure 4).
693 There were some indications of an effect of publication bias, with less certain coefficient
694 estimates tending to have more negative estimates for some of the land uses (Extended Data
695 Figure 4). On the other hand, study-level random slopes of human-dominated land uses
696 tended to be more negative for studies that sampled more sites (Extended Data Figure 4). It is
697 important to emphasize that in a site-level analysis like ours, studies with fewer sites have
698 less weight in the models. Modelled coefficient estimates were generally robust to the
699 removal of these studies (Extended Data Figure 4). Basing projections on coefficient
700 estimates from models where small studies were excluded led to a less than 1% change in the
701 estimated global richness values (results not shown). As with all studies based on data from
702 the literature, we under-represent unpublished data.

703 To model turnover of species composition between pairs of sites, we calculated average
704 dissimilarity²³ in the lists of present species (1 – Sørensen index) between all pairs of sites
705 within each study. For this analysis, we were only able to consider studies with more than one
706 site in at least one of the land-use types considered. Once compositional similarity had been
707 calculated for every pair of sites within each study, the average compositional similarity was
708 calculated for every pair of land-use types considered within each study (including
709 comparisons between sites in the same land-use type). Finally, the average compositional
710 similarity was calculated for each pair of land-use types across all studies. To visualize the
711 clustering of different land-use types in terms of community composition, we performed a
712 hierarchical complete-linkage cluster analysis on the compositional dissimilarity (i.e. 1 –
713 similarity) matrix, using the hclust function in R Version 3.0.2³⁹². To test whether differences
714 in the average geographic distance between pairs of sites in different land-use combinations
715 affected these results, we correlated average compositional similarity with average distance
716 between sites, for all pairwise combinations of land use (including comparisons of a land-use
717 type with itself). Correlations between average distance and average community similarity
718 were only very weakly negative ($R^2 = 0.001$), suggesting they do not strongly distort the
719 comparisons of community composition. However, the fact that some land uses tend to occur
720 more closely together than others could influence the diversity patterns seen in our models, if
721 some land uses are typically close to high-diversity habitats and so are more likely to benefit
722 from dispersal. For example, sites in secondary vegetation and plantation forest were closer,
723 on average, to primary vegetation sites than were those in cropland, pasture and urban
724 (average distances to sites in primary vegetation were: other primary vegetation sites = 7.38
725 km, mature secondary vegetation = 4.4 km, intermediate secondary vegetation = 3.9 km,
726 young secondary vegetation = 6.9 km, plantation forest = 4.2 km, cropland = 16.4 km,
727 pasture = 10.1 km, and urban = 11.4 km). Accounting for distance in such already-complex
728 models is not computationally tractable. In making the projections, we therefore implicitly
729 assume that the average distances will not change (i.e. that secondary vegetation and
730 plantation forests will remain closer to primary vegetation than cropland, pasture and urban
731 habitats).

732 733 *Projecting the Models onto Spatial Estimates of Anthropogenic Variables*

734
735 We projected the best overall models of richness (within-sample and rarefied),
736 abundance and community-weighted mean organism size onto estimates of land use, land-use
737 intensity and human population density at $0.5^\circ \times 0.5^\circ$ resolution, using historical estimates
738 for 1500 to 2005, and four Representative Concentration Pathways scenarios of future

739 changes (IMAGE 2.6, MiniCAM 4.5, AIM 6.0 and MESSAGE 8.5; the names refer to the
740 integrated assessment models used and the numbers to the amount of radiative forcing
741 assumed in 2100)³⁹⁷. In the absence of global projections, proximity to roads and accessibility
742 were omitted from our projections.

743 Estimates of land use for both the historical reconstruction and the future scenarios
744 were taken from the harmonized land-use data accompanying the scenarios²⁶. Estimates of
745 the stage of secondary vegetation (young, intermediate or mature) are not available directly in
746 the RCP land-use data. However, these data contain estimates of the transition each year
747 between secondary vegetation and all other land-use types. To convert this into an estimate of
748 the proportion of secondary vegetation in each of the stages of maturity, we considered any
749 transition to secondary vegetation to result in secondary vegetation of age zero. Each year,
750 this age was then incremented by one. In the absence of better information, any transitions
751 from secondary vegetation to any other land-use type were assumed to be drawn evenly from
752 the ages currently represented. For the purposes of the projections, secondary vegetation was
753 considered to be young until an age of 30 years, intermediate between 30 years and 100
754 years, and mature thereafter. We developed C# code to convert land-use transitions into
755 estimates of the stage structure of secondary vegetation.

756 Gridded temporal estimates of human population density were directly available for the
757 HYDE historical scenario and MESSAGE future scenario. Human population trajectories in
758 the MiniCAM model were resolved only to the level of United Nations regions⁴¹; we
759 therefore downscaled these to grid cells assuming no temporal change in the spatial pattern of
760 relative population density within regions compared to present day patterns⁴⁵, which is the
761 method used in other RCP-scenario land-use models lacking human population data resolved
762 to grid cells²⁶. Gridded estimates of human population from the MESSAGE model were
763 downloaded from <http://www.iiasa.ac.at/web-apps/ggi/GgiDb/>. For the scenarios for which
764 human population projections were not available (IMAGE and AIM), we used country-level
765 estimates from the ‘medium’ scenario of the United Nations population division³⁹⁸, which
766 gives the closest global predictions of future human population to those assumed by IMAGE
767 and AIM²⁶. These country-level estimates were downscaled to grid cells using the same
768 method as for MiniCAM’s regional projections.

769 Land-use intensity was an important explanatory variable in our models, but global
770 maps of land-use intensity are not available. We therefore generated global estimates of
771 current land-use intensity based on a map of ‘Global Land Systems’³⁹⁹, which divides coarse
772 land-use types into sub-categories based on levels of cropland intensity, livestock densities
773 and human population density. We mapped each Global Land Systems class onto one or
774 more relevant combinations of our classes of land use and land-use intensity (Extended Data
775 Table 2). The Global Land Systems dataset has a spatial resolution of 5 arc-minutes. To
776 calculate the proportion of each 0.5° cell occupied by each land use and land-use intensity
777 combination we calculated the proportion of 5-arc-minute cells within each 0.5° cell
778 containing matching Global Land Systems categories (see legend of Extended Data Table 2
779 for details).

780 To generate past and future estimates of land-use intensity, we modelled the current
781 proportion of each land-use type estimated to be under minimal, light or intense levels of
782 intensity within each grid cell (one model for each intensity level), as a function of the
783 prevalence of the land-use type within each cell and human population density, with the
784 relationships allowed to vary among the 23 United Nations (UN) sub-regions (i.e. we fitted
785 interaction terms between UN sub-region and both the prevalence of each land-use type and
786 human population density). UN sub-region data were taken from the world borders shapefile
787 Version 0.3 (http://thematicmapping.org/downloads/world_borders.php) and converted to a
788 0.5° × 0.5° raster using ArcMap Version 10.0³⁹¹. The models were developed using

789 generalized linear models with a binomial distribution of errors, implemented in the lme4
790 package Version 1.0-5⁴⁴ in R Version 3.0.2³⁹². The resulting models explained between
791 30.6% and 76.7% of the deviance in estimated current levels of intensity. Past and future
792 land-use intensities were estimated by applying the models to the same past and future
793 estimates of land use and human population density as above.

794 The scenarios gave the proportion of each grid cell estimated to be occupied by each
795 combination of land use and land-use intensity. We did not attempt to resolve human
796 population density within grid cells for our historical estimates or forecasts, thereby assuming
797 it to be spatially (not temporally) constant within each cell. The coefficients from the models
798 of site-level diversity were thus applied to each combination of land use and intensity within
799 each cell, with the same human population density estimate across all combinations. All
800 predictions were expressed as a percentage net change compared with a baseline before
801 human land-use impacts on biodiversity, in which all land use was assumed to be primary
802 vegetation of minimal intensity of use, and with a human population density of zero. Each
803 cell's average value of net biodiversity change was calculated as the area-weighted mean
804 value across all land uses and intensities. Global average values were calculated as mean
805 values across all cells, weighted by cell area and an appropriate weighting factor to account
806 for the fact that cells have different baseline levels of diversity. The weighting factors applied
807 were: terrestrial vertebrate species richness in the case of richness, and net primary
808 production (NPP) in the case of total abundance. No weighting factor was applied for
809 projections of community-weighted mean plant height. Terrestrial vertebrate species richness
810 was estimated by overlaying extent-of-occurrence range maps for mammals, birds,
811 amphibians and reptiles, using Python code written by ourselves and implemented in ArcMap
812 Version 10.0³⁹¹. Data on Net Primary Production were estimates of potential NPP (i.e. in the
813 absence of human impacts) from the Lund-Postdam-Jena (LPJ) Dynamic Global Vegetation
814 Model⁴⁰⁰.

815 The 95% confidence intervals around the projected values of biodiversity for each
816 combination of pressure variables were estimated based on uncertainty in the modelled
817 coefficients. We were unable to conduct multi-model averaging to account for uncertainty in
818 the structure of the models (i.e. projections were based only on the final best model) because
819 applying such complex mixed-effects models, based on such large datasets, to multiple
820 scenarios of human pressure at a global scale was intractable both in terms of time and
821 computer-memory requirements. We were also unable to account for uncertainty in the
822 trajectories of the human pressure variables, because uncertainty estimates are not available
823 for any of the variables considered.

824 To estimate average biodiversity change in individual countries, we intersected the
825 gridded projections with the world borders shapefile (see above) using the extract function in
826 the raster package Version 2.2-12⁴⁰¹ in R Version 3.0.2³⁹². Mean values across the cells
827 associated with each country were calculated, weighted by cell area. To interpret the
828 outcomes for countries in terms of their natural biodiversity, we related the country-level
829 projections to estimates of average natural vertebrate species richness (see above). To
830 interpret the outcomes for countries in terms of their socio-economy, we related the
831 projections to estimates of the Human Development Index, which is an indicator of
832 education, life expectancy, wealth and standard of living (<https://data.undp.org/>).

833 **Extended Data Legends**

834 **Extended Data Figure 1. Taxonomic and geographic representativeness of the dataset**
835 **used. a,** The relationship between the number of species represented in our data and the
836 number estimated to have been described¹⁷, for 47 major taxonomic groups; lines show (from
837
838

839 bottom to top) 0.1%, 1% and 10% representation of described species in our dataset; magenta
840 = invertebrates, red = vertebrates, green = plants, blue = fungi, and grey = all other taxonomic
841 groups. **b**, the relationship across biomes⁴⁰² between the percentage of global terrestrial net
842 primary production and the number of sites in our dataset; A = tundra, B = boreal forests and
843 taiga, C = temperate conifer forests, D = temperate broadleaf and mixed forests, E = montane
844 grasslands and shrublands, F = temperate grasslands, savannas and shrublands, G =
845 Mediterranean forests, woodlands and scrub, H = deserts and xeric shrublands, J = tropical
846 and subtropical grasslands, savannas and shrublands, K = tropical and subtropical coniferous
847 forests, M = tropical and subtropical dry broadleaf forests, N = tropical and subtropical moist
848 broadleaf forests, P = mangroves; note that the flooded grasslands and savanna biome is not
849 represented in the dataset; grey line show 1:1 line.

850

851 **Extended Data Figure 2. Detailed response of local diversity to human pressures.**

852 Modelled effects (controlling for land use) of human population density (HPD), distance to
853 nearest road, time since 30% conversion of a landscape to human uses (TSC) and time to
854 nearest population centre with greater than 50,000 inhabitants (**a-d**), interactions between
855 pairs of these variables (**e**), and interactions between these variables and land use (**f-i**) on site-
856 level diversity: **a-c, f, g**, within-sample species richness; **e, h, i**, total abundance; and **d**,
857 community-weighted mean vertebrate body mass. Shaded polygons in **a-d** show 95%
858 confidence intervals. For clarity, shaded polygons in **f-i** are shown as $\pm 0.5 \times \text{SEM}$.
859 Confidence intervals in **e** are omitted. Rugs along the x axes in the line graphs show the
860 values of the explanatory variables represented in the dataset used for modelling. Only
861 significant effects are shown. Note that distance to nearest road and travel time to major
862 population centre measures are the raw (log-transformed) values fitted in the models rather
863 than the proximity to roads and accessibility values (obtained as 1 minus the former values)
864 presented in Figure 1. Sample sizes are given in full in the Methods.

865

866 **Extended Data Figure 3. Robustness of modelled effects of human pressures. a**, Effects
867 of land use, human population density (HPD) and proximity to nearest road (PR) on
868 rarefaction-based species richness. **b**, To test that any differences between these results and
869 the results for within-sample species richness presented in the main manuscript were not
870 because rarefied species richness could only be calculated with a smaller dataset, we also
871 show modelled effects on within-sample species richness with the same reduced dataset. **c-d**,
872 Cross-validated robustness of coefficient estimates for land use and land-use intensity:
873 crosses show 95% confidence intervals around the coefficient estimates under ten-fold cross-
874 validation, excluding data from approximately 10% of studies at a time (**c**), and under
875 geographical cross-validation, excluding data from one biome at a time (**d**); colours, points,
876 error bars and land-use labels are as in Figure 1 in the main text. Primary, primary vegetation;
877 YSV, young secondary vegetation; ISV, intermediate secondary vegetation; MSV, mature
878 secondary vegetation; plantation, plantation forest. Sample sizes are given in full in the
879 Methods.

880

881 **Extended Data Figure 4. Tests of the potential for publication bias to influence the**
882 **richness models and projections.** Left-hand panels (**a, d, g, j, m**) show funnel plots of the
883 relationship between the standard error around coefficient estimates (inversely related to the
884 size of studies) and the coefficient estimates themselves for each coarse land-use type; there
885 is evidence for publication bias with respect to some of the land-use types, as indicated by an
886 absence of points on one or other side of zero for studies with large standard errors (but note
887 that small studies are down-weighted in the model); red points show studies with more than 5
888 sites in the land use in question (10 for secondary vegetation and plantation forest because

889 there were more sites for these land uses and some studies with between 5 and 10 sites
890 showed variable responses); horizontal dashed lines show the modelled coefficients for each
891 land use. Central panels (**b, e, h, k, n**) show the relationship between study size (log-
892 transformed total number of sites) and the random slope of the land use in question with
893 respect to study identity, from a random-slopes-and-intercepts model; where a significant
894 relationship was detected using a linear model, fitted values and 95% confidence intervals are
895 shown as a red dashed line and red dotted lines, respectively; conversely to what would be
896 expected if publication bias was present, where significant relationships between study size
897 and random slopes were detected, these were negative (i.e. larger studies detected more
898 negative effects). Right-hand panels (**c, f, i, l, o**) show the robustness of modelled coefficients
899 to removal of studies with few sites in a given land use (black points in the left-hand panels);
900 left-hand error bars show coefficient estimates for all studies and right-hand error bars show
901 coefficient estimates for studies with more than 5 sites in that land use (10 for secondary
902 vegetation and plantation forest).

903

904 **Extended Data Figure 5. Tests for spatial autocorrelation in the model residuals.** For the
905 four main modelled metrics of site-level diversity — **a**, within-sample species richness; **b**,
906 total abundance; **c**, community-weighted mean plant-height; and **d**, community-weighted
907 mean animal mass — histograms of P-values from sets of Moran's tests for spatial
908 autocorrelation in the residuals of the best models for individual studies. The percentage of
909 studies with significant spatial autocorrelation ($P < 0.05$; indicated by vertical red line) is
910 shown.

911

912 **Extended Data Figure 6. Current, past and future projections of all metrics of local**
913 **biodiversity.** Net change in local diversity caused by land use and related pressures by 2000
914 under an IMAGE reference scenario¹⁰: **a**, richness; **b**, rarefied richness; **c**, total abundance;
915 and **d**, community-weighted mean plant height. Note that the values used to divide the
916 colours are the same in all panels, but that the maximum and minimum values are different,
917 as indicated in the legends. Historical and future estimates of net change in local diversity
918 from 1500-2095, based on estimates of land-use, land-use intensity and human population
919 density from the four Representative Concentration Pathways (RCP) scenarios (Table 1): **e**,
920 richness; **f**, total abundance; and **g**, community-weighted mean plant height; historical
921 (shading) and future (error bars) uncertainty shown as 95% confidence intervals, with
922 uncertainty rescaled to be zero in 2005 to show uncertainty in past and future change
923 separately; the global average projection for the MESSAGE scenario does not directly join
924 the historical reconstruction because projections start in 2010 (human population estimates
925 are available at 15-year intervals) and because human population (and thus land-use intensity)
926 and plantation-forest extent have not been harmonized among scenarios. In panel **e**, the
927 dashed line shows projected diversity change under land-use change only (i.e. without land-
928 use intensity and human population density, the projections of which involved simplifying
929 assumptions), and the dotted line shows projections of rarefaction-based species richness.

930

931 **Extended Data Figure 7. Reconstructed and projected total global land-use areas under**
932 **the Representative Concentration Pathways (RCP) scenarios.** **a**, estimated total area of
933 the major land-use types. **b-f**, estimated total area of secondary vegetation in different stages
934 of recovery.

935

936 **Extended Data Figure 8. Biodiversity projections at the country level.** **a-d**, Country-level
937 projections of net change in local richness between 2005 and 2095 under the four RCP
938 scenarios (**a**, IMAGE 2.6; **b**, MiniCAM 4.5; **c**, AIM 6.0; and **d**, MESSAGE 8.5), shown in

939 relation to the Human Development Index (an indicator of education, life expectancy, wealth
940 and standard of living) in the most recent year for which data are available. **e-f**, Country-level
941 projections of net change in local richness between 2005 and 2095 under the best- and worst-
942 performing RCP scenarios in terms of biodiversity (**e**. MiniCAM 4.5; **f**, MESSAGE 8.5),
943 shown in relation to past change in biodiversity from a baseline with no human land-use
944 effects to 2005 according to the HYDE land-use reconstruction. Colours indicate
945 biogeographic realms (key in **b**); colour intensity reflects native vertebrate species richness
946 (more intense = higher species richness); point size is proportional to (log) country area.
947
948
949

950 **Extended Data Table 1. Land use and land-use intensity classification definitions (from**
951 **ref. 16).**
952

953 **Extended Data Table 2. Conversion between Global Land Systems dataset and our**
954 **intensity classification for each major land-use type.** To estimate proportional coverage of
955 each intensity class for each land-use type in the $0.5^\circ \times 0.5^\circ$ grid cells used for projection, we
956 calculated the number of finer-resolution Global Land Systems³⁹⁹ cells with a matching
957 intensity class for the land-use type in question, as a proportion of Global Land Systems cells
958 matching any intensity class for the land-use type in question. For example, to calculate the
959 proportion of urban land that is under intense use, we divided the number of cells with a
960 Global Land Systems classification of ‘urban’ by the number of cells classified as ‘urban’ or
961 ‘peri-urban and villages’. None of the Global Land Systems classes could inform about the
962 intensity of plantation forest, and so we assumed that any plantation forest was composed of
963 equal proportions under minimal, light and intense use.
964

965 **Additional references**

966

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