Global land-use impacts on local terrestrial biodiversity

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Human activities, especially conversion and degradation of habitats, are causing global biodiversity declines. How local ecological assemblages are responding is less clear – a

concern given their importance for many ecosystem functions and services. We analyze a terrestrial assemblage database of unprecedented geographic and taxonomic coverage to quantify local biodiversity responses to land-use and related changes. In the worst-impacted habitats, these pressures reduce within-sample species richness by an average of 76.5%, total abundance by 39.5% and rarefaction-based richness by 40.3%. We estimate that, globally, these pressures have already slightly reduced within-sample richness (by 13.6% on average), total abundance (10.7%) and rarefaction-based richness (8.1%), with changes showing marked spatial variation. Further rapid losses are predicted under a business-as-usual land-use scenario: within-sample richness is projected to fall by a further 3.4% globally by 2100, with losses concentrated in biodiverse but economically poor countries. Strong mitigation can deliver much more positive biodiversity changes (up to a 1.9% reversal of past losses) that are less strongly related to countries' socioeconomic status.

Biodiversity faces growing pressures from human actions, including habitat conversion and degradation, habitat fragmentation, climate change, harvesting and pollution[1]. As a result, species' extinction risk increases on average while population sizes decline[1,2]. Such assessments have usually focused on data-rich vertebrates, so might not reflect broader biodiversity[3]. Furthermore, most have concentrated on the global status of species, whereas the long-term security of many ecosystem functions and services – especially in changing environments – likely depends upon local biodiversity[4–6]. Average trends in local diversity remain unclear: analyses of temporal changes in assemblages have suggested no systematic change in species richness[7,8], but the available times-series data might under-represent transitions between land-use types[9] and population time series suggest sharp declines in vertebrate populations in recent decades[3].

Spatial comparisons provide an alternative source of evidence on biodiversity impacts, assuming that differences in pressures have caused observed biodiversity differences between otherwise matched sites[10–12]. The prevalence of published spatial comparisons makes it possible to go beyond particular taxa or regions[11,12] to develop global, taxonomically representative models. Furthermore, the willingness of many researchers to share their raw data makes it possible to consider multiple aspects of biodiversity, rather than the single, simple metrics of most existing models[10] which cannot capture all key aspects of diversity[13].

We present the most geographically and taxonomically representative models to date of how several aspects of the composition and diversity of terrestrial assemblages respond to multiple human pressures. The pressures we consider most directly measure land use and infrastructure, but include proxies that might correlate[14,15] with two other important pressures – harvesting and invasive species – for which comparable high-resolution spatial data are unavailable globally. We exclude climate change effects because they are not captured well by spatial comparisons. We map global hindcasts of net changes in assemblages since 1500, and project future changes over this Century under different socioeconomic scenarios of land use. We then relate projected national changes in local biodiversity to socioeconomic variables and natural biodiversity.

Our models of local within-sample species richness (hereafter ‘richness’), rarefaction-based species richness (hereafter ‘rarefied richness’), total abundance, compositional turnover and average organism size are based on 1,130,251 records of abundance and 320,924 of occurrence or species richness at 11,525 sites (Figure 1a) in 13 of the 14 terrestrial biomes (Extended Data Figure 1). These data, from 284 publications (see Methods), represent 26,953 species – 1.4% of the number formally described[16]. Each site was scored for six putative pressures: land use[11] and use intensity[17], land-use history[18], human population density[19], proximity to roads[20], and accessibility from the nearest large town. Random effects in our models accounted for study-level differences in response variables and sampling methods, and for the within-study spatial arrangement of sites.

Human impacts on site-level diversity

Local richness, rarefied richness and total abundance were most strongly influenced by land use and land-use intensity: they were substantially lower in most other land-use types than in primary vegetation, especially in intensively-used areas (Figure 1; see Supplementary Information for statistics). These results extend those of previous, geographically or taxonomically restricted, meta-analyses (e.g. refs. 11,21). Effects of other variables were weaker, but showed stronger effects in interaction with other variables (Extended Data Figure 2) and were often significant overall (see Supplementary Information). Richness and total abundance tended to be slightly lower at the highest human population densities, and richness was lower nearer to roads and in more accessible sites (Figure 1). Differences in richness were not driven solely by differences in abundance: rarefaction-based richness[22] (see Methods for details) showed weaker but mostly similar patterns, although the effects of variables other than land use and land-use intensity were not significant (Extended Data Figure 3). Under the worst combinations of pressures, our models estimated richness, rarefied richness and total abundance to be 76.5%, 40.3% and 39.5% lower, respectively, than in minimally impacted sites. Effects of pressures on vertebrate, invertebrate and plant richness were statistically indistinguishable ($P > 0.05$; results not shown).

The importance of secondary vegetation for conservation is hotly debated[11,23,24], and is crucial because this land-use will soon become the most widespread type[25]. We find that the answer depends strongly on the secondary vegetation's maturity: early-stage communities tend to be less diverse than those in primary vegetation and are compositionally distinct, but these differences are much reduced in mature secondary vegetation (Figures 1 & 2; we caution though that not all data sources clearly distinguish mature secondary from primary vegetation). This successional rise in diversity accords with a recent meta-analysis of plant communities over time[7].

Net changes in diversity provide an incomplete view of the effects of human impacts because they ignore replacement of original species by newcomers[8]. We therefore analysed how land use affects similarity in species composition between sites. Communities under the same land use were, unsurprisingly, the most similar (Figure 2a). Across land uses, communities in primary vegetation were most like those in secondary vegetation, while plantation forest, pasture and cropland communities formed a different, human-dominated, cluster (Figure 2b).

Anthropogenic pressures can affect ecosystem functions and services more strongly than changes in species diversity would imply, if species' responses depend on their traits[26]. Large size is often linked to species' declines[27,28] and matters for some ecosystem processes[29]. We combined abundance data with species' average sizes to calculate site-level community-weighted mean plant height and animal mass. As in local studies[28], mean plant height was lower in human-dominated land uses than in primary and secondary vegetation, and tended to decline with increasing human population density (Figure 1d). Most field studies focused on particular plant taxa, so this difference does not simply reflect tree removal. Average animal mass did not change consistently with land use or human population density, but increased with proximity to roads (Figure 1d).

Models like ours that substitute space for time ignore time lags in biotic changes, which can be important[30]. We also assume that land uses are situated randomly within studies relative to sites' intrinsic suitability for biodiversity. Adding global data on other important pressures as they become available, and also incorporating climate change, will give a more complete picture of human impacts on local biodiversity.

Global impacts on local diversity to date

By applying our model for within-sample species richness – the most widely used and understood biodiversity measure – to maps of current pressure variables, we estimate the global pattern of net local changes to date in plot-level richness (we do not estimate total richness within cells; Figure 3). We estimate that human-dominated areas have lost much more local diversity than where more natural vegetation remains. The worst-affected cells showed a 31% reduction in average local richness – probably enough to impact ecosystem functioning substantially[5]. Local richness increased in 1.7% of cells (by ≤ 4.8%). Total abundance and rarefied richness showed broadly similar patterns, although less pronounced in the latter case (Extended Data Figure 4).

We applied our models to global spatial estimates of how land use and human population changed from 1500-2005[25] (see Methods) to infer the history of local biodiversity change. We focus on within-sample species richness because of its wide use and easy interpretation. Our inferences incorporate uncertainty in model parameter estimates, but not in the trajectories of the drivers themselves (which have not been assessed[31]) nor effects of changes in roads and accessibility, for which temporal estimates could not be obtained (they are treated as static).

Richness is estimated to have declined most rapidly in the 19th and 20th centuries (Figure 4), with other metrics showing similar responses (Extended Data Figure 4). By 2005, we estimate that human impacts had reduced local richness by an average of 13.6% (95% CI: 9.1 – 17.8%) and total abundance by 10.7% (95% CI: 3.8% gain – 23.7% reduction) compared with pre-impact times. Approximately 60% of the decline in richness was independent of effects on abundance: average rarefied richness has fallen by 8.1% (95% CI: 3.5 – 12.9%). Although these confidence limits omit uncertainty in the projections of land use and other pressures, there is less uncertainty in estimates of current pressure levels than in changes over time[32].

Our inferences contrast with two recent analyses of community time series[7,8], which suggested no overall trend in local diversity, and with the Living Planet Index[3] which, based on population time series, reports a much more rapid decline in abundance than we infer. Although time series potentially provide a more direct view of temporal trends than our space-for-time approach, the available data might under-represent transitions between land-use types[9]. On the other hand, our approach may underestimate additions of species through climate change and species invasion (although accessibility and proximity to roads may partly capture the latter[14,15]).

Global and national projections to 2095

Global changes in local diversity from 2005 to 2095 were projected using estimated land use and human population from the four Intergovernmental Panel on Climate Change Representative Concentration Pathway (RCP) scenarios[25], which correspond to different intensities of global climate change (Table 1). Although these estimates have limitations[31], they are the most consistent available, are widely-used[33], and are consistent with the historical estimates [25]. However, they – like all other global land-use projections – include no estimate of uncertainty; therefore, each of our projections must be viewed as the predicted biodiversity outcome under one particular set of land-use assumptions.

Projected net changes in average local diversity to 2095 vary widely among scenarios (Figure 4; Extended Data Figure 4). The scenario with the least climate change (IMAGE 2.6) yields the second-worst outcome for biodiversity, because it assumes rapid conversion of primary vegetation – especially in the tropics – to crops and biofuels[25] (Table 1, Extended Data Figure 5). These projections do not imply that low-emission scenarios must entail large losses of biodiversity, but instead reflect that scenario's mitigation strategy. Indeed, in MiniCAM 4.5 (where mitigation is through carbon markets, crop improvements and diet shifts: Table 1), average richness is projected to increase (though other diversity metrics respond more weakly: Extended Data Figure 4). The worst biodiversity outcomes arise from the scenario with most climate change – MESSAGE 8.5 – in which rapid human population growth drives widespread agricultural expansion (Table 1; Extended Data Figure 5). This scenario, which has been characterised as 'business-as-usual'[34], most closely matches recent trends in emissions[35] and performs worst even though our projections omit direct climate impacts on local assemblages.

The global projections hide wide regional and national variation (Figure 5; Extended Data Figure 6). Projections for 2095 under 'business-as-usual' (MESSAGE 8.5) are strongly inequitable, presenting serious challenges for both sustainable development and global conservation of biodiversity (Figure 5a). Under this scenario, European and North American countries – typically with a high Human Development Index (HDI), low native biodiversity and widespread historical land conversion – are mostly projected to gain in local richness by 2095. More naturally biodiverse but less economically developed Southeast Asian and especially sub-Sharan African countries, with more natural and semi-natural habitat, will suffer the greatest losses (Figure 5a; Extended Data Figure 6f).

Such globally inequitable outcomes might be avoidable: the best scenario for biodiversity (MiniCAM 4.5; Figure 4) yielded country-level outcomes that are relatively independent of HDI, native species richness (Figure 5b) and past changes (Extended Data Figure 6e). For local richness, outcomes under MiniCAM4.5 were better than MESSAGE 8.5 for 93% of countries worldwide (Figure 5c).

Under AIM 6.0, most Afrotropical countries are projected to gain in local richness but heavy losses are inferred for the Indo-Malay region (Extended Data Figure 6). Projections under IMAGE 2.6 are similar to those under MESSAGE 8.5. The land-use change caused by the biofuels-based strategy in IMAGE 2.6 is projected to have a major negative impact overall on terrestrial biodiversity (Extended Data Figure 6).

Conclusions

Many assessments of the state of biodiversity have focused on global metrics such as rates of species extinction[36], but resilient delivery of ecosystem functions and services more likely depends on local diversity[4–6]. Our models suggest land-use changes and associated pressures strongly reduce local terrestrial biodiversity, and we estimate global average reductions to date of 13.6% in within-sample species richness, 10.7% in total abundance and 8.1% in rarefaction-based species richness (Figs. 3 and 4). Climate change, which we could not include in our framework, is likely to exacerbate losses, especially under business-as-usual[37], although direct effects of climate change will increase local diversity in some regions[8].

Habitat conversion and associated changes that reduced local biodiversity had largely positive consequences for people: agricultural intensification underpinned many countries' development. However, benefits have not been shared equally among or within countries[38]. Losses of local species richness exceeding 20% are likely to substantially impair the contribution of biodiversity to ecosystem function and services, and thus to human well-being[5]. We estimate that reductions in average plot-level species richness currently exceed this level for 28.4% of grid cells, increasing to 41.5% of cells by 2095 under 'business-as-usual' (note that we do not estimate or project total richness across the cell). Importantly, our projections suggest that such widespread large losses are not inevitable. With concerted action and the right societal choices, global sustainability of local biodiversity may be an achievable goal.

Methods summary

We collated among-site comparisons of ecological assemblage composition from the literature[39]. Studies compared from 2 to 360 sites (median = 15; 82% had ≥ 5 sites); most sampled species from multiple families but fewer than half sampled multiple orders. Over 70% of sites were from studies that sampled entire communities within a taxonomic group rather than a target list of species. Removing studies having a target list did not substantially alter model coefficients (results not shown) and increased the projected global net average loss of local species richness until 2005 by 0.6%. Sites varied in the maximum linear extent sampled (median 106 m; interquartile range 50 m to 354 m). Model coefficients for the approximately 50% of studies that reported maximum linear extent were robust to its inclusion in the models (results not shown).

We computed four site-level biodiversity metrics: within-sample species richness (i.e. number of species sampled at a site), total abundance, rarefaction-based richness (i.e., average number of species found by sampling a constant number of individuals) and community-weighted mean organism size (using species data for plant height and animal mass or volume). Site land use was classified from the published data sources as primary vegetation, secondary vegetation (subdivided into young, intermediate or mature), plantation forest, cropland, pasture or urban[39] (Extended Data Table 1); these classes were chosen to map onto those in the Representative Concentration Pathways (RCP) scenarios[25]. Land-use intensity was scored as minimal, light or intense[39] (Extended Data Table 1). Land-use history was represented as the time since 30% of the 0.5° grid cell encompassing a site was converted to agricultural or urban use, above which increasingly rapid loss of biodiversity occurs[40]. Estimates were taken from the HYDE model[41]. Human population density, proximity to the nearest road and travel time to nearest population of > 50,000 (inversely proportional to accessibility to humans) for each site came from global datasets[42–44]. These variables were at most weakly inter-correlated (all $R^2 < 0.31$).

Responses of site-level diversity to pressures were fitted using mixed-effects models[45], with random effects accounting for study-level differences in diversity (fitted including random slopes) and for blocks of sites within studies. Richness and rarefied richness were fitted using Poisson models, using observation-level random effects to account for overdispersion. Residuals showed little spatial autocorrelation (Extended Data Figure 7). Publication bias, which is common in datasets of this type[3], would violate this assumption. There was some evidence that studies with fewer sites and less certainty in estimated coefficients were disproportionately likely to show significantly negative land-use impacts (Extended Data Figure 8). Excluding studies with few sites had little effect on most coefficients (Extended Data Figure 8), partly because our analyses use sites as data points, automatically giving less weight to such studies, and changed projections of richness by less than 1% (not shown). As with all studies based on data from the literature, we cannot incorporate unpublished data. Effects of land use on spatial turnover were estimated by averaging pairwise Sørensen community similarity[22] between all pairs of land uses.

To infer temporal diversity changes, we applied the model coefficients to gridded (0.5°-resolution) historical estimates and future projections of pressure variables. Land use estimates were from HYDE and the RCP scenarios[25]. Human population density estimates were available directly for HYDE and MESSAGE; for other scenarios we downscaled the associated region- and country-level population estimates conserving the current spatial pattern within countries[42]. Land-use intensity was inferred from statistical models relating current land-use intensity to the land use's prevalence within each cell and human population density. In the absence of global projections, proximity to roads and accessibility were assumed to be static over time. Uncertainty was estimated from model-estimated uncertainty around the coefficients. Cross-validation (leaving out a random 10% of studies – Extended Data Figure 3c – or one biome at a time – Extended Data Figure 3d) was used to check model robustness. To derive global averages, we weighted grid cells by land area for community-weighted mean size; by land area and current vertebrate species richness for species richness; and by potential terrestrial net primary production[46] for total abundance.

References

1. Tittensor, D. P. *et al.* A mid-term analysis of progress toward international biodiversity targets. *Science* **346,** 241–244 (2014).

2. Pimm, S. L. *et al.* The biodiversity of species and their rates of extinction, distribution, and protection. *Science* **344,** 1246752–1246752 (2014).

3. Collen, B. *et al.* Monitoring change in vertebrate abundance: the Living Planet Index. *Conserv. Biol.* **23,** 317–327 (2009).

4. Isbell, F. *et al.* High plant diversity is needed to maintain ecosystem services. *Nature* **477,** 199–202 (2011).

5. Hooper, D. U. *et al.* A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature* **486,** 105–108 (2012).

6. Cardinale, B. J. *et al.* Biodiversity loss and its impact on humanity. *Nature* **486,** 59–67 (2012).

7. Vellend, M. *et al.* Global meta-analysis reveals no net change in local-scale plant biodiversity over time. *Proc. Natl. Acad. Sci. U. S. A.* **110,** 19456–9 (2013).

8. Dornelas, M. *et al.* Assemblage time series reveal biodiversity change but not systematic loss. *Science* **344,** 296–9 (2014).

9. Cardinale, B. Overlooked local biodiversity loss. *Science* **344,** 1098 (2014).

10. Alkemade, R. *et al.* GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss. *Ecosystems* **12,** 374–390 (2009).

11. Gibson, L. *et al.* Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature* **478,** 378–381 (2011).

12. Mendenhall, C. D., Karp, D. S., Meyer, C. F. J., Hadly, E. A. & Daily, G. C. Predicting biodiversity change and averting collapse in agricultural landscapes. *Nature* **509,** 213–217 (2014).

13. Pereira, H. M. *et al.* Essential biodiversity variables. *Science* **339,** 277–278 (2013).

14. Weber, E. & Li, B. Plant invasions in China: what is to be expected in the wake of economic development? *Bioscience* **58,** 437–444 (2008).

15. Clements, G. R. *et al.* Where and how are roads endangering mammals in Southeast Asia's forests? *PLoS One* **9,** e115376 (2014).

16. Chapman, A. D. *Numbers of Living Species in Australia and the World.* (2009).

17. Phalan, B., Onial, M., Balmford, A. & Green, R. E. Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science* **333,** 1289–1291 (2011).

18. Balmford, A. Extinction filters and current resilience: the significance of past selection pressures for conservation biology. *Trends Ecol. Evol.* **11,** 193–196 (1996).

19. Newbold, T. *et al.* A global model of the response of tropical and sub-tropical forest biodiversity to anthropogenic pressures. *Proc. R. Soc. London Ser. B Biol. Sci.* **281,** 20141371 (2014).

20. Benítez-López, A., Alkemade, R. & Verweij, P. A. The impacts of roads and other infrastructure on mammal and bird populations: a meta-analysis. *Biol. Conserv.* **143,** 1307–1316 (2010).

21. Murphy, G. E. P. & Romanuk, T. N. A meta-analysis of declines in local species richness from human disturbances. *Ecol. Evol.* **4,** 91–103 (2014).

22. Magurran, A. E. *Measuring Biological Diversity*. (Wiley-Blackwell, 2004).

23. Barlow, J. *et al.* Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proc. Natl. Acad. Sci. U. S. A.* **104,** 18555–18560 (2007).

24. Dent, D. H. & Wright, S. J. The future of tropical species in secondary forests: A quantitative review. *Biol. Conserv.* **142,** 2833–2843 (2009).

25. Hurtt, G. C. *et al.* Harmonization of land-use scenarios for the period 1500–2100: 600 years of global gridded annual land-use transitions, wood harvest, and resulting secondary lands. *Clim. Change* **109,** 117–161 (2011).

26. Díaz, S. *et al.* Functional traits, the phylogeny of function, and ecosystem service vulnerability. *Ecol. Evol.* **3,** 2958–75 (2013).

27. Cardillo, M. *et al.* Multiple causes of high extinction risk in large mammal species. *Science* **309,** 1239–1241 (2005).

28. Mayfield, M. M. *et al.* Differences in forest plant functional trait distributions across land-use and productivity gradients. *Am. J. Bot.* **100,** 1356–1368 (2013).

29. Séguin, A., Harvey, É., Archambault, P., Nozais, C. & Gravel, D. Body size as a predictor of species loss effect on ecosystem functioning. *Sci. Rep.* **4,** 4616 (2014).

30. Wearn, O. R., Reuman, D. C. & Ewers, R. M. Extinction debt and windows of conservation opportunity in the Brazilian Amazon. *Science* **337,** 228–232 (2012).

31. Harfoot, M. *et al.* Integrated assessment models for ecologists: the present and the future. *Glob. Ecol. Biogeogr.* **23,** 124–143 (2014).

32. Ellis, E. C. Anthropogenic transformation of the terrestrial biosphere. *Philos. Trans. R. Soc. London Ser. A Math. Phys. Eng. Sci.* **369,** 1010–1035 (2011).

33. Mora, C. *et al.* The projected timing of climate departure from recent variability. *Nature* **502,** 183–187 (2013).

34. Burrows, M. T. *et al.* Geographical limits to species-range shifts are suggested by climate velocity. *Nature* **507,** 492–495 (2014).

35. Oldfield, F. & Steffen, W. Anthropogenic climate change and the nature of Earth System science. *Anthr. Rev.* **1,** 70–75 (2014).

36. Pereira, H. M. *et al.* Scenarios for global biodiversity in the 21st century. *Science* **330,** 1496–1501 (2010).

37. Warren, R. *et al.* Quantifying the benefit of early climate change mitigation in avoiding biodiversity loss. *Nat. Clim. Chang.* **3,** 678–682 (2013).

38. Millennium Ecosystem Assessment. *Ecosystems and Human Well-being: Biodiversity Synthesis*. (World Resources Institute, 2005).

39. Hudson, L. N. *et al.* The PREDICTS database: a global database of how local terrestrial biodiversity responds to human impacts. *Ecol. Evol.* **4,** 4701–4735 (2014).

40. Andrén, H. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* **71,** 355–366 (1994).

41. Klein Goldewijk, K., Beusen, A., Van Drecht, G. & De Vos, M. The HYDE 3.1 spatially explicit database of human-induced global land-use change over the past 12,000 years. *Glob. Ecol. Biogeogr.* **20,** 73–86 (2011).

42. Center for International Earth Science Information Network (CIESIN) Columbia University, International Food Policy Research Institute (IFPRI), The World Bank & Centro Internacional de Agricultura Tropical (CIAT). *Global Rural-Urban Mapping Project, Version 1 (GRUMPv1): Population Density Grid.* (NASA Socioeconomic Data and Applications Center (SEDAC), 2011). http://sedac.ciesin.columbia.edu/data/dataset/grump-v1-population-density (Accessed 11 July 2012).

43. Center for International Earth Science Information Network (CIESIN) Columbia University & Information Technology Outreach Services (ITOS) University of Georgia. *Global Roads Open Access Data Set, Version 1 (gROADSv1)*. (NASA

Socioeconomic Data and Applications Center (SEDAC), 2013). http://dx.doi.org/10.7927/H4VD6WCT (Accessed 18 December 2013).

44. Nelson, A. *Estimated travel time to the nearest city of 50,000 or more people in year 2000.* (2008). http://bioval.jrc.ec.europa.eu/products/gam/ (Accessed 14 July 2014).

45. Bates, D., Maechler, M., Bolker, B. & Walker, S. lme4: Linear mixed-effects models using Eigen and S4. (2013). http://cran.r-project.org/web/packages/lme4/ (Accessed 27 November 2014).

46. Haberl, H. *et al.* Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. *Proc. Natl. Acad. Sci. U. S. A.* **104,** 12942–12947 (2007).

47. Van Vuuren, D. P. *et al.* RCP2.6: exploring the possibility to keep global mean temperature increase below 2°C. *Clim. Change* **109,** 95–116 (2011).

48. Thomson, A. M. *et al.* RCP4.5: a pathway for stabilization of radiative forcing by 2100. *Clim. Change* **109,** 77–94 (2011).

49. Riahi, K. *et al.* RCP 8.5—A scenario of comparatively high greenhouse gas emissions. *Clim. Change* **109,** 33–57 (2011).

50. Rogelj, J., Meinshausen, M. & Knutti, R. Global warming under old and new scenarios using IPCC climate sensitivity range estimates. *Nat. Clim. Chang.* **2,** 248–253 (2012).

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Additional references can be found in the Supplementary Information.

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Author contributions

T.N., L.N.H., S.L.L.H., S.C., I.L., B.C., D.W.P., R.M.E., G.M.M., J.P.W.S. and A.P. designed the project and this study; T.N., L.N.H., I.L., R.A.S., L.B., J.P.W.S. and A.P. performed the analyses; T.N., L.N.H., S.L.L.H., S.C., D.B., A.C., B.C., J.D., A.D.P., S.E.-L., M.G., M.L.K.H., T.H., D.J.I., V.K., L.K., D.L.P.C., C.M., Y.P., H.R.P.P., A.R., J.S., H.J..W.

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

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Tables

Table 1. **Key features of the four Representative Concentration Pathways (RCP) scenarios** (land-use and human-population assumptions are detailed in ref. 25, energy assumptions in refs. 47–49, and climate implications in ref. 50).

Scenario	Land use (see also Extended Data Figure 5)	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.	Approximately 2.5 °C temperature increase by 2100.	9.1 billion by 2100 (UN Medium variant, 2004)

	Cropland area increases to meet food demand. Pasture area declines strongly.		
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.

Figure legends

Figure 1. Locations of sites and responses of four metrics of local diversity to human pressures. a, Sites in the dataset used to model within-sample species richness. Fitted responses of three biodiversity metrics to anthropogenic variables: **b,** richness; **c,** total abundance; and **e,** community-weighted mean organism size – $\log_{10}$ plant height (crosses) and $\log_{10}$ animal mass (triangles). Error bars show 95% confidence intervals. Land use effects: Primary = primary vegetation, YSV = young secondary vegetation, ISV = intermediate secondary vegetation, MSV = mature secondary vegetation, Plantation = plantation forest. Land-use intensity: minimal (circle), light (triangle), intense (diamond), and combined light and intense (square). Continuous effects: human population density (HPD), proximity to roads (as –log(distance to nearest road); PR), and accessibility to humans (as –log(travel time to nearest major city); ACC) are shown here as modelled marginal effects for all land uses (i.e. from a model with no interactions between continuous effects and land use) at the lowest (L), median (M) and highest (H) values in the dataset.

Figure 2. Similarity in assemblage composition as a function of land use. a, Average dissimilarity of species composition (1 – Sørenson Index) between pairs of sites within and among different combinations of land use; values are expressed proportional to the average similarity between pairs of primary-vegetation sites; blue colours indicate more similar composition, and red colours less similar; more intense colours indicate more extreme values; numbers indicate numbers of studies within which comparisons could be made. **b,** Dendrogram showing clustering of different land-use types based on average compositional dissimilarity; urban sites were excluded as their small sample size made their position unstable. Land-use labels as in Figure 1.

Figure 3. Net change in local richness from pre-impact times to 2000. Projections used an IMAGE reference scenario[10]. The pre-impact landscape was assumed to be entirely uninhabited and unused primary vegetation. Shown using a Lambert Cylindrical Equal Area projection at 0.5°resolution.

Figure 4. Projected net change in local richness from 1500 to 2095. Future projections were based on the four Representative Concentration Pathways (RCP) scenarios (Table 1). Historical (shading) and future (error bars) uncertainty is shown as 95% confidence intervals, with uncertainty rescaled to be zero in 2005 to show uncertainty in past and future change separately. The baseline for these projections is a world entirely composed of uninhabited and unused primary vegetation; thus, the value at 1500 is not constrained to be zero because non-primary land uses were present – and in some regions widespread – by that date. The global average projection for MESSAGE 8.5 does not directly join the historical reconstruction because that scenario's land use projections start in 2010 (human population estimates are available at 15-year intervals) and because human population (and thus inferred land-use intensity) and plantation-forest extent have not been harmonized among scenarios.

Figure 5. Biodiversity projections at the country level. **a-b,** Country-level projections of average net local richness change between 2005 and 2095 under the worst (**a**, MESSAGE 8.5) and best (**b**, MiniCAM 4.5) RCP scenarios for biodiversity, shown in relation to the Human Development Index (an indicator of education, life expectancy, wealth and standard of living). Colours indicate biogeographic realms; colour intensity reflects natural vertebrate species richness (more intense = higher species richness); point size is proportional to (log) country area. **c,** Correlation between projected richness changes under the MiniCAM 4.5 and

MESSAGE 8.5 scenarios, with dashed line showing equality; colours as in **a and b**; colour intensity is proportional to the HDI (more intense = higher index).

Extended Data Legends

Extended Data Figure 1. **Taxonomic and geographic representativeness of the dataset used**. **a,** The relationship between the number of species represented in our data with the number estimated to have been described[16], for 47 major taxonomic groups; lines show (from bottom to top) 0.1%, 1% and 10% representation of described species in our dataset; magenta = invertebrates, red = vertebrates, green = plants, blue = fungi, and grey = all other taxonomic groups. **b**, the relationship across biomes[403] between the percentage of global net primary production and the number of sites in our dataset; A = tundra, B = boreal forests and taiga, C = temperate conifer forests, D = temperate broadleaf and mixed forests, E = montane grasslands and shrublands, F = temperate grasslands, savannas and shrublands, G = Mediterranean forests, woodlands and scrub, H = deserts and xeric shrublands, J = tropical and subtropical grasslands, savannas and shrublands, K = tropical and subtropical coniferous forests, M = tropical and subtropical dry broadleaf forests, N = tropical and subtropical moist broadleaf forests, P = mangroves; note that the flooded grasslands and savanna biome is not represented in the dataset; grey line show 1:1 line.

Extended Data Figure 2. Detailed response of local diversity to human pressures. Modelled effects (controlling for land use) of human population density (HPD), distance to nearest road, time since 30% conversion of a landscape to human uses (TSC) and time to nearest population centre with greater than 50,000 inhabitants (**a-d**), interactions between pairs of these variables (**e**), and interactions between these variables and land use (**f-i**) on site-level diversity: **a-c, f, g**, within-sample species richness; **e, h, i**, total abundance; and **d**, community-weighted mean vertebrate body mass. Shaded polygons in **a-d** show 95% confidence intervals. For clarity, shaded polygons in **f-i** are shown as ±0.5 × SEM. Confidence intervals in **e** are omitted. Rugs along the x axes in the line graphs show the values of the explanatory variables represented in the dataset used for modelling. Only significant effects are shown. Note that distance to nearest road and travel time to major population centre measures are the raw (log-transformed) values fitted in the models rather than the proximity to roads and accessibility values (obtained as 1 minus the former values) presented in Figure 1.

Extended Data Figure 3. Robustness of modelled effects of human pressures. a, Effects of land use, human population density (HPD) and proximity to nearest road (PR) on rarefaction-based species richness. **b**, To test that any differences between these results and the results for within-sample species richness presented in the main manuscript were not because rarefied species richness could only be calculated with a smaller dataset, we also show modelled effects on within-sample species richness with the same reduced dataset. **c-d**, Cross-validated robustness of coefficient estimates for land use and land-use intensity: crosses show 95% confidence intervals around the coefficient estimates under ten-fold cross-validation, excluding data from approximately 10% of studies at a time (**c**), and under geographical cross-validation, excluding data from one biome at a time (**d**); colours, points, error bars and land-use labels are as in Figure 1 in the main text.

Extended Data Figure 4. Current, past and future projections of all metrics of local biodiversity. Net change in local diversity from pre-human times to 2000 under an IMAGE reference scenario[10]: **a**, richness; **b**, rarefied richness; **c**, total abundance; and **d**, community-weighted mean plant height. Note that the values used to divide the colours are the same in all panels, but that the maximum and minimum values are different, as indicated in the legends. Hindcast and projected net change in local diversity from 1500-2095, based on estimates of land-use, land-use intensity and human population density from the four Representative Concentration Pathways (RCP) scenarios (Table 1): **e**, richness; **f**, total abundance; and **g**, community-weighted mean plant height; historical (shading) and future (error bars) uncertainty shown as 95% confidence intervals, with uncertainty rescaled to be zero in 2005 to show uncertainty in past and future change separately; the global average projection for the MESSAGE scenario does not directly join the historical reconstruction because projections start in 2010 (human population estimates are available at 15-year intervals) and because human population (and thus land-use intensity) and plantation-forest extent have not been harmonized among scenarios. In panel **e**, the dashed line shows projected diversity change under land-use change only (i.e. without land-use intensity and human population density, the projections of which involved simplifying assumptions), and the dotted line shows projections of rarefaction-based species richness.

Extended Data Figure 5. Reconstructed and projected total global land-use areas under the Representative Concentration Pathways (RCP) scenarios. a, estimated total area of the major land-use types. **b-f**, estimated total area of secondary vegetation in different stages of recovery.

Extended Data Figure 6. **Biodiversity projections at the country level**. **a-d,** Country-level projections of net change in local richness between 2005 and 2095 under the four RCP scenarios (**a**, IMAGE 2.6; **b**, MiniCAM 4.5; **c**, AIM 6.0; and **d**, MESSAGE 8.5), shown in relation to the Human Development Index (an indicator of education, life expectancy, wealth and standard of living) in the most recent year for which data are available. **e-f**, Country-level projections of net change in local richness between 2005 and 2095 under the best- and worst-performing RCP scenarios in terms of biodiversity (**e**. MiniCAM 4.5; **f**, MESSAGE 8.5), shown in relation to past change in biodiversity from the pre-impact baseline to 2005 according to the HYDE land-use reconstruction. Colours indicate biogeographic realms (key in **b**); colour intensity reflects native vertebrate species richness (more intense = higher species richness); point size is proportional to (log) country area.

Extended Data Figure 7. Tests for spatial autocorrelation in the model residuals. For the four main modelled metrics of site-level diversity — **a**, within-sample species richness; **b**, total abundance; **c**, community-weighted mean plant-height; and **d**, community-weighted mean animal mass — histograms of P-values from sets of Moran's tests for spatial autocorrelation in the residuals of the best models for individual studies. The percentage of studies with significant spatial autocorrelation ($P < 0.05$; indicated by vertical red line) is shown.

Extended Data Figure 8. Tests of the potential for publication bias to influence the richness models and projections. Left-hand panels (**a**, **d**, **g**, **j**, **m**) show funnel plots of the relationship between the standard error around coefficient estimates (inversely related to the size of studies) and the coefficient estimates themselves for each coarse land-use type; there is evidence for publication bias with respect to some of the land-use types, as indicated by an absence of points on one or other side of zero for studies with large standard errors (but note that small studies are down-weighted in the model); red points show studies with more than 5 sites in the land use in question (10 for secondary vegetation and plantation forest because there were more sites for these land uses and some studies with between 5 and 10 sites showed variable responses); horizontal dashed lines show the modelled coefficients for each land use. Central panels (**b**, **e**, **h**, **k**, **n**) show the relationship between study size (log-transformed total number of sites) and the random slope of the land use in question with respect to study identity, from a random-slopes-and-intercepts model; where a significant relationship was detected using a linear model, fitted values and 95% confidence intervals are shown as a red dashed line and red dotted lines, respectively; conversely to what would be expected if publication bias was present, where significant relationships between study size and random slopes were detected, these were negative (i.e. larger studies detected more negative effects). Right-hand panels (**c**, **f**, **i**, **l**, **o**) show the robustness of modelled coefficients to removal of studies with few sites in a given land use (black points in the left-hand panels); left-hand error bars show coefficient estimates for all studies and right-hand error bars show coefficient estimates for studies with more than 5 sites in that land use (10 for secondary vegetation and plantation forest).

Extended Data Table 1. Land use and land-use intensity classification definitions (from ref. 39).

Extended Data Table 2. **Conversion between Global Land Systems dataset and our intensity classification for each major land-use type**. To estimate proportional coverage of each intensity class for each land-use type in the 0.5° grid cells used for projection, we calculated the number of finer-resolution Global Land Systems[401] cells with a matching intensity class for the land-use type in question, as a proportion of Global Land Systems cells matching any intensity class for the land-use type in question. For example, to calculate the proportion of urban land that is under intense use, we divided the number of cells with a Global Land Systems classification of ‘urban’ by the number of cells classified as ‘urban’ or ‘peri-urban and villages’. None of the Global Land Systems classes could inform about the intensity of plantation forest, and so we assumed that any plantation forest was composed of equal proportions under minimal, light and intense use.

Methods

Data collation

Between March 2012 and April 2014 we collated data from published studies[51-334] examining the effect of human activities on more than one named taxon. A full description of how the dataset was assembled and curated is presented elsewhere[39]. We define sites to be in the same study if they were sampled using the same methodology and the data were reported in the same paper; therefore, some publications contain multiple studies. After six months of broad searching, we targeted efforts towards under-represented taxa, habitat types, biomes and regions. We accepted data only from published or in-press papers, or data collected using a published methodology, and we required that the data providers agreed to our making their data publicly available at the end of the PREDICTS (Projecting Responses of Ecological Diversity In Changing Terrestrial Systems) project in 2015. We accepted data only where abundance, occurrence or richness had been measured at two or more sampling locations and/or times, and where all sites were sampled using the same sampling procedure and with either the same effort or site-specific data on effort. We used geographical coordinates preferentially from the paper or supplied by data providers; but where coordinates were not thus available, we georeferenced them from maps in the papers.

The great majority of listed taxa were species-level, although many could not be given explicit species identifications (e.g. morpho-species)[39]; henceforth we refer to distinct taxa in our dataset as species. We matched taxonomic names given in the source paper to the Catalogue of Life 2013 Annual Checklist (COL)[335], obtaining the full taxonomic classification. In order to relate the taxonomic names to species-level trait databases, we generated, for each taxon, a 'best-guess' Latin binomial as: a) the taxon name from COL if the COL query returned a species-level identification; b) the first two words of the text returned by the COL query if this was a sub-species designation; c) the first two words of the taxon name in the source publication if the COL query returned neither a species or sub-species name, and the taxon name in the source publication contained two or more words. Taxa that met none of these criteria were not matched to trait data, but were included in the calculation of richness and total abundance, and for estimating turnover in community composition among sites.

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

Site-level composition and diversity

Site-level metrics of diversity were calculated as follows.

Within-sample species richness was calculated as the number of differently-named taxa recorded at a given site in a standardised sampling unit (a measure also known as species density[339]). We gave precedence to the author's classification of species, even where a search of global databases revealed potential synonymies, because only certain taxonomic groups could be reliably matched to accepted taxonomies.

This measure of richness is appropriate for conservation questions but among site-differences could be due to effects on numbers of individuals as well as to changes in the shape of the species accumulation curve[336]. We therefore also calculated **rarefaction-based species richness** by taking 1000 random samples of *n* individuals from each site, where *n* is the smallest total number of individuals recorded at any site within its study, and calculating the mean species richness across samples. This index could only be calculated for sites where, in addition to the criteria above being met, abundance was recorded as number of individuals. Rarefied species richness was rounded to the nearest integer for analysis with Poisson errors.

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

Community-weighted mean organism size was calculated as the arithmetic mean of log-transformed height of plants (available for 4,235 species in our dataset) or the log-transformed body mass or volume of vertebrates, beetles and hoverflies (5,236 species) present at a site, weighted by abundance[337]. Plant height data were taken from the TRY database[338]; for 61 species where plant vegetative height data were unavailable, we estimated it from generative height from a regression across the 2,554 species with estimates of both traits ($R^2 = 0.91$). Data on vertebrate body mass were taken from the PanTHERIA database for mammals[339], from BirdLife International's World Bird Database for birds, and from a wide range of published and grey-literature sources for amphibians [340–386]. Length data for reptiles were taken from published and unpublished sources[387,388], and converted to estimates of body mass using published length-mass allometries[389,390]. Arthropod size data (beetles and hoverflies) were collated from published sources[391,392]. Beetle length and amphibian snout-vent length values were raised to the power three so that they had the same dimensionality as the other animal size measures. For both plant height and vertebrate body mass, missing values were interpolated as the average values for congeners, since both of these traits are strongly conserved phylogenetically (Pagel's $\lambda = 0.98, 0.997, 0.93, 0.89$ for plant height, vertebrate body mass, beetle body length and hoverfly thorax volume, respectively).

Human pressure data

While many human pressures can impact local biodiversity, we focus on those that can be obtained for sites around the world and for which, as far as possible, spatiotemporal data are available for 1500-2095; this focus enables us to use our statistical models as a basis for hindcasting and projecting responses through time. Each site was assigned to one of eight land-use classes based on the description of the habitat given in the source paper (see

Extended Data Table 1 for definitions): primary vegetation, secondary vegetation (subdivided into mature, intermediate or young secondary vegetation), plantation forest, cropland, pasture and urban. These classes were selected to match the land-use classification adopted in the Intergovernmental Panel on Climate Change Representative Concentration Pathways scenarios[25] in order to facilitate the projection of our models onto these scenarios. Sites were also assigned to a level of human intensity of use – minimal, light or intense – within each major land-use class, also based on the description of the habitat in the source paper (see Extended Data Table 1 for definitions). The factors that determined this level depended on the land-use class (e.g. bushmeat extraction and limited logging in primary and secondary vegetation; or stocking density and chemical inputs in pasture; Extended Data Table 1). Sites that could not be classified for land-use and use intensity were excluded from the analyses. We overlaid our sites with available global data sets to obtain site-level estimates of human population density[42], distance to the nearest road[43] and estimated travel time to nearest population centre with greater than 50,000 inhabitants[44]. For distance to nearest road, the roads map was first projected onto a Berhmann equal-area projection. These operations were carried out using Python code implemented using the arcpy Python module in ArcMap Version 10.0[393]. In the main figures, the inverses of distance to roads and travel time to major population centre (proximity to roads and accessibility) were presented so that high values corresponded to higher hypothesized human impact. To estimate the history of human use of the landscapes within which sites were located, we calculated the number of years since the 30-arc-second grid cell containing each site became 30% covered by human land uses (cropland, pasture and urban), according to the HYDE model[41]. The 30% threshold was chosen because it has been suggested as the level of landscape conversion above which increasingly rapid loss of biodiversity occurs[40]. Collinearity among variables describing anthropogenic change was low: the highest correlation was between land use and human population density (Pearson $R^2 = 0.31$).

Modelling site-level diversity, composition and turnover

The response of site-level diversity to the measures of anthropogenic change was modelled using generalized linear mixed effects models, implemented in the lme4 package Version 1.0-5[45] in R Version 3.0.2[394]. We first compared candidate random-effects structures using the full candidate fixed-effects structure[395]. Random-intercept terms considered in all models were the identity of study from which data were taken, to account for study-level differences in the response variables and sampling methods used, and – within-study – the spatial block in which the site was located, to account for the spatial arrangement of sites. For models of species richness (within-sample and rarefied), we also fitted an observation-level random effect (i.e. site identity) to account for the overdispersion present[396]. We also considered random slopes, with respect to study, of each of the main fixed effects (land use, land-use intensity, human population density, distance to nearest road, travel time to nearest major city and time since the landscape was majority converted to human uses). Random effects were retained or discarded based on AIC values.

Once the best random-effects structure had been selected, we performed backward stepwise model simplification to select the best fixed-effects structure (see ‘Full Statistical Results’)[395]. Human population density, distance to roads, travel time to nearest major city and time since major human use of the landscape were log transformed in the analyses, with a value of 1 added to human population density, travel time to nearest major city and time since major landscape conversion to deal with zero values. Human population density, distance to roads, travel time to nearest major city and time since human landscape conversion were fitted as continuous effects, with quadratic polynomials for human population, distance to roads and travel time to nearest major city, and as a linear effect for time since human landscape conversion. For variables fitted as quadratic polynomials, we also tested linear effects during the backward stepwise model selection. All continuous variables were rescaled prior to analysis so that values ranged between zero and one. Interaction terms were tested first, and then removed to test the main effects. All main effects that were part of significant interaction terms were retained in the final models regardless of their significance as main effects. For the model of community-weighted mean body mass and plant height, because the number of sites with data was smaller than for the other metrics, only land use (excluding urban sites, which were few), human population density and distance to roads, and no interactions, were fitted. The decision whether or not to retain terms was based on likelihood ratio tests. The coefficient estimates of the best models are shown in Figure 1b-d and Extended Figure 2, and the formulae and statistical results are shown in the Supplementary Information. To test for spatial autocorrelation in the residuals of the final best models, we calculated Moran's I values and associated P-values, separately for each study considered in the models, using the spdep package Version 0.5-68[397] in R; the distribution of P-values across studies was used as an indication of whether spatial autocorrelation was likely to cause a problem (Extended Data Figure 7). We used cross validation to assess the robustness of model parameter estimates, first based on dividing the studies randomly into 10 equal-sized sets and dropping each set in turn (Extended Figure 3a), and secondly based on leaving out the studies from each biome in turn (Extended Figure 3b).

Publication bias is a potential problem for any large-scale synthesis of data from many publications. In standard meta-analyses, funnel plots[398] can be used to test for any relationship between standard error and effect size, as a bias in effect sizes at high standard error toward more positive or more negative effects indicates a likely effect of publication bias. Creating funnel plots for our data was more complicated because ours was a site-level analysis of raw diversity estimates rather than a meta-analysis. We generated individual models relating diversity to land use for each study that sampled at least two sites within each of at least two land-use types. We focused on land use because: a) there were a small number of sites included in most within-study models; and 2) the original studies focused on effects of land use – not generally on land-use intensity, human population density or distance to roads – and thus any effect of publication bias would likely be seen in the land use coefficients. Funnel plots were generated by plotting, for each land-use type, the estimated model coefficients against the associated standard errors (Extended Data Figure 8). There were some indications of an effect of publication bias, with less certain coefficient estimates tending to have more negative estimates for some of the land uses (Extended Data Figure 8). On the other hand, study-level random slopes of human-dominated land uses tended to be more negative for studies that sampled more sites (Extended Data Figure 8). It is important to emphasize that in a site-level analysis like ours, studies with fewer sites have less weight in the models. Modelled coefficient estimates were generally robust to the removal of these studies (Extended Data Figure 8). Basing projections on coefficient estimates from models where small studies were excluded led to a less than 1% change in the estimated global richness values (results not shown).

To model turnover of species composition between pairs of sites, we calculated average dissimilarity[22] in the lists of present species (1 − Sørensen index) between all pairs of sites within each study. For this analysis, we were only able to consider studies with more than one site in at least one of the land-use types considered. Once compositional similarity had been calculated for every pair of sites within each study, the average compositional similarity was calculated for every pair of land-use types considered within each study (including comparisons between sites in the same land-use type). Finally, the average compositional similarity was calculated for each pair of land-use types across all studies. To visualize the clustering of different land-use types in terms of community composition, we performed a hierarchical complete-linkage cluster analysis on the compositional dissimilarity (i.e. 1 − similarity) matrix, using the hclust function in R Version 3.0.2[394]. To test whether differences in the average geographic distance between pairs of sites in different land-use combinations affected these results, we correlated average compositional similarity with average distance between sites, for all pairwise combinations of land use (including comparisons of a land-use type with itself). Correlations between average distance and average community similarity were only very weakly negative ($R^2 = 0.001$), suggesting they do not strongly distort the comparisons of community composition. However, the fact that some land uses tend to occur more closely together than others could influence the diversity patterns seen in our models, if some land uses are typically close to high-diversity habitats and so are more likely to benefit from dispersal. For example, sites in secondary vegetation and plantation forest were closer, on average, to primary vegetation sites than were those in cropland, pasture and urban (average distances to sites in primary vegetation were: other primary vegetation sites = 7.38 km, mature secondary vegetation = 4.4 km, intermediate secondary vegetation = 3.9 km, young secondary vegetation = 6.9 km, plantation forest = 4.2 km, cropland = 16.4 km, pasture = 10.1 km, and urban = 11.4 km). Accounting for distance in such already-complex models is not computationally tractable. In making the projections, we therefore implicitly assume that the average distances will not change (i.e. that secondary vegetation and plantation forests will remain closer to primary vegetation than cropland, pasture and urban habitats).

Projecting the Models onto Spatial Estimates of Anthropogenic Variables

We projected the best overall models of richness (within-sample and rarefied), abundance and community-weighted mean organism size onto estimates of land use, land-use intensity and human population density at 0.5° resolution, using historical estimates for 1500 to 2005, and four Representative Concentration Pathways scenarios of future changes (IMAGE 2.6, MiniCAM 4.5, AIM 6.0 and MESSAGE 8.5; the names refer to the models used and the numbers to the amount of radiative forcing assumed in 2100)[399].

Estimates of land use for both the historical reconstruction and the future scenarios were taken from the harmonized land-use data accompanying the scenarios[25]. Estimates of the stage of secondary vegetation (young, intermediate or mature) are not available directly in the RCP land-use data. However, these data contain estimates of the transition each year between secondary vegetation and all other land-use types. To convert this into an estimate of the proportion of secondary vegetation in each of the stages of maturity, we considered any transition to secondary vegetation to result in secondary vegetation of age zero. Each year, this age was then incremented by one. In the absence of better information, any transitions from secondary vegetation to any other land-use type were assumed to be drawn evenly from the ages currently represented. For the purposes of the projections, secondary vegetation was considered to be young until an age of 30 years, intermediate between 30 years and 100 years, and mature thereafter. We developed C# code to convert land-use transitions into estimates of the stage structure of secondary vegetation.

Gridded temporal estimates of human population density were available only for the HYDE historical scenario and MESSAGE future scenario. Human population trajectories in the MiniCAM model were resolved only to the level of United Nations regions[48]; we therefore downscaled these to grid cells assuming no temporal change in the spatial pattern of relative population density within regions compared to present day patterns[42], which is the method used in other RCP-scenario land-use models lacking human population data resolved to grid cells[25]. Gridded estimates of human population from the MESSAGE model were downloaded from http://www.iiasa.ac.at/web-apps/ggi/GgiDb/. For the scenarios for which human population projections were not available (IMAGE and AIM), we used country-level estimates from the 'medium' scenario of the United Nations population division[400], which gives the closest global predictions of future human population to those assumed by IMAGE and AIM[25]. These country-level estimates were downscaled to grid cells using the same method as for MiniCAM's regional projections.

Land-use intensity was an important explanatory variable in our models, but global maps of land-use intensity are not available. We therefore generated global estimates of current land-use intensity based on a map of 'Global Land Systems'[401], which divides coarse land-use types into sub-categories based on levels of cropland intensity, livestock densities and human population density. We mapped each Global Land Systems class onto one or more relevant combinations of our classes of land use and land-use intensity (Extended Data Table 2). The Global Land Systems dataset has a spatial resolution of 5 arc-minutes. To calculate the proportion of each 0.5° cell occupied by each land use and land-use intensity combination we calculated the proportion of 5-arc-minute cells within each 0.5° cell containing matching Global Land Systems categories (see legend of Extended Data Table 2 for details).

To generate past and future estimates of land-use intensity, we modelled the current proportion of each land-use type estimated to be under minimal, light or intense levels of intensity within each grid cell (one model for each intensity level), as a function of the prevalence of the land-use type within each cell and human population density, with the relationships allowed to vary among the 23 United Nations (UN) sub-regions (i.e. we fitted interaction terms between UN sub-region and both the prevalence of each land-use type and human population density). UN sub-region data were taken from the world borders shapefile Version 0.3 (http://thematicmapping.org/downloads/world_borders.php) and converted to a 0.5° raster using ArcMap Version 10.0[393]. The models were developed using generalized linear models with a binomial distribution of errors, implemented in the lme4 package Version 1.0-5[45] in R Version 3.0.2[394]. The resulting models explained between 30.6% and 76.7% of the deviance in estimated current levels of intensity. Past and future land-use intensities were hindcast and forecast by applying the models to the same past and future estimates of land use and human population density as above.

The scenarios gave the proportion of each grid cell estimated to be occupied by each combination of land use and land-use intensity. We did not attempt to resolve human population density within grid cells for our hindcasts or forecasts, thereby assuming it to be spatially (not temporally) constant within each cell. The coefficients from the models of site-level diversity were thus applied to each combination of land use and intensity within each cell, with the same human population density estimate across all combinations. All predictions were expressed as a percentage net change compared with a baseline, in which all land use was assumed to be primary vegetation of minimal intensity of use, and with a human population density of zero. Each cell's average value of net biodiversity change was calculated as the area-weighted mean value across all land uses and intensities. Global average values were calculated as mean values across all cells, weighted by cell area and an appropriate weighting factor to account for the fact that cells have different baseline levels of diversity. The weighting factors applied were: terrestrial vertebrate species richness in the case of richness, and net primary production (NPP) in the case of total abundance. No weighting factor was applied for projections of community-weighted mean plant height. Terrestrial vertebrate species richness was estimated by overlaying extent-of-occurrence range maps for mammals, birds, amphibians and reptiles, using Python code written by ourselves and implemented in ArcMap Version 10.0[393]. Data on Net Primary Production were estimates of potential NPP (i.e. in the absence of human impacts) from the Lund-Postdam-Jena (LPJ) Dynamic Global Vegetation Model[46].

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

To estimate average biodiversity change in individual countries, we intersected the gridded projections with the world borders shapefile (see above) using the extract function in the raster package Version 2.2-12[402] in R Version 3.0.2[394]. Mean values across the cells associated with each country were calculated, weighted by cell area. To interpret the outcomes for countries in terms of their natural biodiversity, we related the country-level projections to estimates of average natural vertebrate species richness (see above). To interpret the outcomes for countries in terms of their socio-economy, we related the projections to estimates of the Human Development Index, which is an indicator of education, life expectancy, wealth and standard of living (https://data.undp.org/).

Additional references

51. Aben, J., Dorenbosch, M., Herzog, S. K., Smolders, A. J. P. & Van Der Velde, G. Human Disturbance affects a Deciduous Forest Bird Community in the Andean Foothills of Central Bolivia. *Bird Conservation International* **18**, 363-380, doi:10.1017/s0959270908007326 (2008).

52. Adum, G. B., Eichhorn, M. P., Oduro, W., Ofori-Boateng, C. & Rodel, M. O. Two-Stage Recovery of Amphibian Assemblages Following Selective Logging of Tropical Forests. *Conservation Biology* **27**, 354-363, doi:10.1111/cobi.12006 (2013).

53. Aguilar Barquero, V. & Jiménez Hernández, F. Diversidad y distribución de palmas (Arecaceae) en tres fragmentos de bosque muy húmedo en Costa Rica. *Revista de Biología Tropical* **57**, 83-92 (2009).

54. Alberta Biodiversity Monitoring Institute (ABMI). The raw soil arthropods dataset and the raw trees & snags dataset from Prototype Phase (2003-2006) and Rotation 1 (2007-2012). (2013).

55. Alcala, E. L., Alcala, A. C. & Dolino, C. N. Amphibians and reptiles in tropical rainforest fragments on Negros Island, the Philippines. *Environmental Conservation* **31**, 254-261, doi:10.1017/s0376892904001407 (2004).

56. Alcayaga, O. E., Pizarro-Araya, J., Alfaro, F. M. & Cepeda-Pizarro, J. Spiders (Arachnida, Araneae) associated to agroecosystems in the Elqui Valley (Coquimbo Region, Chile). *Revista Colombiana De Entomologia* **39**, 150-154 (2013).

57. Ancrenaz, M., Goossens, B., Gimenez, O., Sawang, A. & Lackman-Ancrenaz, I. Determination of ape distribution and population size using ground and aerial surveys: a case study with orang-utans in lower Kinabatangan, Sabah, Malaysia. *Animal Conservation* **7**, 375-385, doi:10.1017/s136794300400157x (2004).

58. Arbeláez-Cortés, E., Rodríguez-Correa, H. A. & Restrepo-Chica, M. Mixed bird flocks: patterns of activity and species composition in a region of the Central Andes of Colombia. *Revista Mexicana De Biodiversidad* **82**, 639-651 (2011).

59. Armbrecht, I., Perfecto, I. & Silverman, E. Limitation of nesting resources for ants in Colombian forests and coffee plantations. *Ecological Entomology* **31**, 403-410, doi:10.1111/j.1365-2311.2006.00802.x (2006).

60. Arroyo, J., Iturrondobeitia, J. C., Rad, C. & Gonzalez-Carcedo, S. Oribatid mite (Acari) community structure in steppic habitats of Burgos Province, central northern Spain. *Journal of Natural History* **39**, 3453-3470, doi:10.1080/00222930500240346 (2005).

61. Azhar, B. *et al.* The influence of agricultural system, stand structural complexity and landscape context on foraging birds in oil palm landscapes. *Ibis* **155**, 297-312, doi:10.1111/ibi.12025 (2013).

62. Azpiroz, A. B. & Blake, J. G. Avian assemblages in altered and natural grasslands in the northern Campos of Uruguay. *Condor* **111**, 21-35, doi:10.1525/cond.2009.080111 (2009).

63. Baeten, L. *et al.* Early Trajectories of Spontaneous Vegetation Recovery after Intensive Agricultural Land Use. *Restoration Ecology* **18**, 379-386, doi:10.1111/j.1526-100X.2009.00627.x (2010).

64. Baeten, L., Hermy, M., Van Daele, S. & Verheyen, K. Unexpected understorey community development after 30 years in ancient and post-agricultural forests. *Journal of Ecology* **98**, 1447-1453, doi:10.1111/j.1365-2745.2010.01711.x (2010).

65. Báldi, A., Batáry, P. & Erdős, S. Effects of grazing intensity on bird assemblages and populations of Hungarian grasslands. *Agriculture Ecosystems & Environment***108**, 251-263, doi:10.1016/j.agee.2005.02.006 (2005).

66. Banks, J. E., Sandvik, P. & Keesecker, L. Beetle (Coleoptera) and spider (Araneae) diversity in a mosaic of farmland, edge, and tropical forest habitats in western Costa Rica. *Pan-Pacific Entomologist* **83**, 152-160, doi:10.3956/0031-0603-83.2.152 (2007).

67. Barlow, J. *et al.* Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences of the United States of America* **104**, 18555-18560, doi:10.1073/pnas.0703333104 (2007).

68. Bartolommei, P., Mortelliti, A., Pezzo, F. & Puglisi, L. Distribution of nocturnal birds (Strigiformes and Caprimulgidae) in relation to land-use types, extent and configuration in agricultural landscapes of Central Italy. *Rendiconti Lincei-Scienze Fisiche E Naturali* **24**, 13-21, doi:10.1007/s12210-012-0211-3 (2013).

69. Basset, Y. *et al.* Changes in Arthropod Assemblages along a Wide Gradient of Disturbance in Gabon. *Conservation Biology* **22**, 1552-1563, doi:10.1111/j.1523-1739.2008.01017.x (2008).

70. Bates, A. J. *et al.* Changing Bee and Hoverfly Pollinator Assemblages along an Urban-Rural Gradient. *Plos One* **6**, doi:10.1371/journal.pone.0023459 (2011).

71. Baur, B. *et al.* Effects of abandonment of subalpine hay meadows on plant and invertebrate diversity in Transylvania, Romania. *Biological Conservation* **132**, 261-273, doi:10.1016/j.biocon.2006.04.018 (2006).

72. Berg, A., Ahrne, K., Ockinger, E., Svensson, R. & Soderstrom, B. Butterfly distribution and abundance is affected by variation in the Swedish forest-farmland landscape. *Biological Conservation* **144**, 2819-2831, doi:10.1016/j.biocon.2011.07.035 (2011).

73. Bernard, H., Fjeldsa, J. & Mohamed, M. A case study on the effects of disturbance and conversion of tropical lowland rain forest on the non-volant small mammals in

north Borneo: Management implications. *Mammal Study* **34**, 85-96, doi:10.3106/041.034.0204 (2009).

74. Berry, N. J. *et al.* The high value of logged tropical forests: lessons from northern Borneo. *Biodiversity and Conservation* **19**, 985-997, doi:10.1007/s10531-010-9779-z (2010).

75. Bicknell, J. & Peres, C. A. Vertebrate population responses to reduced-impact logging in a neotropical forest. *Forest Ecology and Management* **259**, 2267-2275, doi:10.1016/j.foreco.2010.02.027 (2010).

76. Bihn, J. H., Verhaagh, M., Braendle, M. & Brandl, R. Do secondary forests act as refuges for old growth forest animals? Recovery of ant diversity in the Atlantic forest of Brazil. *Biological Conservation* **141**, 733-743, doi:10.1016/j.biocon.2007.12.028 (2008).

77. Billeter, R. *et al.* Indicators for biodiversity in agricultural landscapes: a pan-European study. *Journal of Applied Ecology* **45**, 141-150, doi:10.1111/j.1365-2664.2007.01393.x (2008).

78. Bóçon, R. *Riqueza e abundância de aves em três estágios sucessionais da floresta ombrófila densa submontana, Antonina, Paraná* PhD thesis, Universidade Federal do Paraná, (2010).

79. Borges, S. H. Bird assemblages in secondary forests developing after slash-and-burn agriculture in the Brazilian Amazon. *Journal of Tropical Ecology* **23**, 469-477, doi:10.1017/s0266467407004105 (2007).

80. Boutin, C., Baril, A. & Martin, P. A. Plant diversity in crop fields and woody hedgerows of organic and conventional farms in contrasting landscapes. *Agriculture Ecosystems & Environment* **123**, 185-193, doi:10.1016/j.agee.2007.05.010 (2008).

81. Bouyer, J. *et al.* Identification of ecological indicators for monitoring ecosystem health in the trans-boundary W Regional park: A pilot study. *Biological Conservation***138**, 73-88, doi:10.1016/j.biocon.2007.04.001 (2007).

82. Bragagnolo, C., Nogueira, A. A., Pinto-da-Rocha, R. & Pardini, R. Harvestmen in an Atlantic forest fragmented landscape: Evaluating assemblage response to habitat quality and quantity. *Biological Conservation* **139**, 389-400, doi:10.1016/j.biocon.2007.07.008 (2007).

83. Brearley, F. Q. Below-ground secondary succession in tropical forests of Borneo. *Journal of Tropical Ecology* **27**, 413-420, doi:10.1017/s0266467411000149 (2011).

84. Brito, I., Goss, M. J., de Carvalho, M., Chatagnier, O. & van Tuinen, D. Impact of tillage system on arbuscular mycorrhiza fungal communities in the soil under Mediterranean conditions. *Soil & Tillage Research* **121**, 63-67, doi:10.1016/j.still.2012.01.012 (2012).

85. Brunet, J. *et al.* Understory succession in post-agricultural oak forests: Habitat fragmentation affects forest specialists and generalists differently. *Forest Ecology and Management* **262**, 1863-1871, doi:10.1016/j.foreco.2011.08.007 (2011).

86. Buczkowski, G. Extreme life history plasticity and the evolution of invasive characteristics in a native ant. *Biological Invasions* **12**, 3343-3349, doi:10.1007/s10530-010-9727-6 (2010).

87. Buczkowski, G. & Richmond, D. S. The effect of urbanization on ant abundance and diversity: a temporal examination of factors affecting biodiversity. *Plos One* **7**, doi:10.1371/journal.pone.0041729 (2012).

88. Buddle, C. M. & Shorthouse, D. P. Effects of experimental harvesting on spider (Araneae) assemblages in boreal deciduous forests. *Canadian Entomologist* **140**, 437-452, doi:10.4039/n07-LS01 (2008).

89. Buscardo, E. *et al.* The early effects of afforestation on biodiversity of grasslands in Ireland. *Biodiversity and Conservation* **17**, 1057-1072, doi:10.1007/s10531-007-9275-2 (2008).

90. Cabra-García, J., Bermúdez-Rivas, C., Osorio, A. M. & Chacón, P. Cross-taxon congruence of alpha and beta diversity among five leaf litter arthropod groups in Colombia. *Biodiversity and Conservation* **21**, 1493-1508, doi:10.1007/s10531-012-0259-5 (2012).

91. Caceres, N. C., Napoli, R. P., Casella, J. & Hannibal, W. Mammals in a fragmented savannah landscape in south-western Brazil. *Journal of Natural History* **44**, 491-512, doi:10.1080/00222930903477768 (2010).

92. Cagle, N. L. Snake species distributions and temperate grasslands: A case study from the American tallgrass prairie. *Biological Conservation* **141**, 744-755, doi:10.1016/j.biocon.2008.01.003 (2008).

93. Calviño-Cancela, M., Rubido-Bará, M. & van Etten, E. J. B. Do eucalypt plantations provide habitat for native forest biodiversity? *Forest Ecology and Management***270**, 153-162, doi:10.1016/j.foreco.2012.01.019 (2012).

94. Cameron, S. A. *et al.* Patterns of widespread decline in North American bumble bees. *Proceedings of the National Academy of Sciences of the United States of America* **108**, 662-667, doi:10.1073/pnas.1014743108 (2011).

95. Carrijo, T. F., Brandao, D., de Oliveira, D. E., Costa, D. A. & Santos, T. Effects of pasture implantation on the termite (Isoptera) fauna in the Central Brazilian Savanna

(Cerrado). *Journal of Insect Conservation* **13**, 575-581, doi:10.1007/s10841-008-9205-y (2009).

96. Carvalho, A. L. d., Ferreira, E. J. L., Lima, J. M. T. & de Carvalho, A. L. Floristic and structural comparisons among palm communities in primary and secondary forest fragments of the Raimundo Irineu Serra Environmental Protection Area - Rio Branco, Acre, Brazil. *Acta Amazonica* **40**, 657-666, doi:10.1590/s0044-59672010000400004 (2010).

97. Castro, H., Lehsten, V., Lavorel, S. & Freitas, H. Functional response traits in relation to land use change in the Montado. *Agriculture Ecosystems & Environment* **137**, 183-191, doi:10.1016/j.agee.2010.02.002 (2010).

98. Castro-Luna, A. A., Sosa, V. J. & Castillo-Campos, G. Bat diversity and abundance associated with the degree of secondary succession in a tropical forest mosaic in south-eastern Mexico. *Animal Conservation* **10**, 219-228, doi:10.1111/j.1469-1795.2007.00097.x (2007).

99. Center For International Forestry Research (CIFOR). Multidisciplinary Landscape Assessment - Cameroon. (2013).

100. Center For International Forestry Research (CIFOR). Multidisciplinary Landscape Assessment - Philippines. (2013).

101. Centro Agronómico Tropical de Investigación y Enseñanza(CATIE). *Unpublished data of reptilian and amphibian diversity in six countries in Central America* (Centro Agronómico Tropical de Investigación y Enseñanza (CATIE), 2010).

102. Cerezo, A., Conde, M. & Poggio, S. Pasture area and landscape heterogeneity are key determinants of bird diversity in intensively managed farmland. *Biodiversity and Conservation* **20**, 2649-2667, doi:10.1007/s10531-011-0096-y (2011).

103. Chapman, K. & Reich, P. Land use and habitat gradients determine bird community diversity and abundance in suburban, rural and reserve landscapes of Minnesota, USA. *Biological Conservation* **135**, 527-541, doi:10.1016/j.biocon.2006.10.050 (2007).

104. Chauvat, M., Wolters, V. & Dauber, J. Response of collembolan communities to land-use change and grassland succession. *Ecography* **30**, 183-192, doi:10.1111/j.2007.0906-7590.04888.x (2007).

105. Clarke, F. M., Rostant, L. V. & Racey, P. A. Life after logging: post-logging recovery of a neotropical bat community. *Journal of Applied Ecology* **42**, 409-420, doi:10.1111/j.1365-2664.2005.01024.x (2005).

106. Cleary, D. F. R. *et al.* Diversity and community composition of butterflies and odonates in an ENSO-induced fire affected habitat mosaic: a case study from East Kalimantan, Indonesia. *Oikos* **105**, 426-446, doi:10.1111/j.0030-1299.2004.12219.x (2004).

107. Cleary, D. F. R. & Mooers, A. O. Burning and logging differentially affect endemic vs. widely distributed butterfly species in Borneo. *Diversity and Distributions* **12**, 409-416, doi:10.1111/j.1366-9516.2006.00256.x (2006).

108. Cockle, K. L., Leonard, M. L. & Bodrati, A. A. Presence and abundance of birds in an Atlantic forest reserve and adjacent plantation of shade-grown yerba mate, in Paraguay. *Biodiversity and Conservation* **14**, 3265-3288, doi:10.1007/s10531-004-0446-0 (2005).

109. Connop, S., Hill, T., Steer, J. & Shaw, P. Microsatellite analysis reveals the spatial dynamics of Bombus humilis and Bombus sylvarum. *Insect Conservation and Diversity* **4**, 212-221, doi:10.1111/j.1752-4598.2010.00116.x (2011).

110. D'Aniello, B., Stanislao, I., Bonelli, S. & Balletto, E. Haying and grazing effects on the butterfly communities of two Mediterranean-area grasslands. *Biodiversity and Conservation* **20**, 1731-1744, doi:10.1007/s10531-011-0058-4 (2011).

111. Darvill, B., Knight, M. E. & Goulson, D. Use of genetic markers to quantify bumblebee foraging range and nest density. *Oikos* **107**, 471-478, doi:10.1111/j.0030-1299.2004.13510.x (2004).

112. Davis, A. L. V. & Philips, T. K. Effect of deforestation on a southwest Ghana dung beetle assemblage (Coleoptera : Scarabaeidae) at the periphery of Ankasa conservation area. *Environmental Entomology* **34**, 1081-1088, doi:10.1603/0046-225x(2005)034[1081:eodoas]2.0.co;2 (2005).

113. Davis, E. S., Murray, T. E., Fitzpatrick, U., Brown, M. J. F. & Paxton, R. J. Landscape effects on extremely fragmented populations of a rare solitary bee, Colletes floralis. *Molecular Ecology* **19**, 4922-4935, doi:10.1111/j.1365-294X.2010.04868.x (2010).

114. Dawson, J. *et al.* Bird communities of the lower Waria Valley, Morobe Province, Papua New Guinea: a comparison between habitat types. *Tropical Conservation Science* **4**, 317-348 (2011).

115. Delabie, J. H. C. *et al.* Ants as biological indicators of Wayana Amerindian land use in French Guiana. *Comptes Rendus Biologies* **332**, 673-684, doi:10.1016/j.crvi.2009.01.006 (2009).

116. Diekötter, T., Walther-Hellwig, K., Conradi, M., Suter, M. & Frankl, R. Effects of landscape elements on the distribution of the rare bumblebee species Bombus muscorum in an agricultural landscape. *Biodiversity and Conservation* **15**, 57-68, doi:10.1007/s10531-004-2932-9 (2006).

117. Domínguez, E., Bahamonde, N. & Muñoz-Escobar, C. Efectos de la extracción de turba sobre la composición y estructura de una turbera de *Sphagnum* explotada y abandonada hace 20 años, Chile. *Anales Instituto Patagonia (Chile)* **40**, 37-45, doi:10.4067/S0718-686X2012000200003 (2012).

118. Dominguez-Haydar, Y. & Armbrecht, I. Response of ants and their seed removal in rehabilitation areas and forests at El Cerrejon coal mine in Colombia. *Restoration Ecology* **19**, 178-184, doi:10.1111/j.1526-100X.2010.00735.x (2010).

119. Dumont, B. *et al.* How does grazing intensity influence the diversity of plants and insects in a species-rich upland grassland on basalt soils? *Grass and Forage Science***64**, 92-105, doi:10.1111/j.1365-2494.2008.00674.x (2009).

120. Dures, S. G. & Cumming, G. S. The confounding influence of homogenising invasive species in a globally endangered and largely urban biome: Does habitat quality dominate avian biodiversity? *Biological Conservation* **143**, 768-777, doi:10.1016/j.biocon.2009.12.019 (2010).

121. Edenius, L., Mikusinski, G. & Bergh, J. Can Repeated Fertilizer Applications to Young Norway Spruce Enhance Avian Diversity in Intensively Managed Forests? *Ambio***40**, 521-527, doi:10.1007/s13280-011-0137-5 (2011).

122. Elek, Z. & Lovei, G. L. Patterns in ground beetle (Coleoptera : Carabidae) assemblages along an urbanisation gradient in Denmark. *Acta Oecologica-International Journal of Ecology* **32**, 104-111, doi:10.1016/j.actao.2007.03.008 (2007).

123. Endo, W. *et al.* Game vertebrate densities in hunted and nonhunted forest sites in Manu National Park, Peru. *Biotropica* **42**, 251-261, doi:10.1111/j.1744-7429.2009.00546.x (2010).

124. Faruk, A., Belabut, D., Ahmad, N., Knell, R. J. & Garner, T. W. J. Effects of oil-palm plantations on diversity of tropical anurans. *Conservation Biology* **27**, 615-624, doi:10.1111/cobi.12062 (2013).

125. Farwig, N., Sajita, N. & Boehning-Gaese, K. Conservation value of forest plantations for bird communities in western Kenya. *Forest Ecology and Management* **255**, 3885-3892, doi:10.1016/j.foreco.2008.03.042 (2008).

126. Fayle, T. M. *et al.* Oil palm expansion into rain forest greatly reduces ant biodiversity in canopy, epiphytes and leaf-litter. *Basic and Applied Ecology* **11**, 337-345, doi:10.1016/j.baae.2009.12.009 (2010).

127. Felton, A. M., Engstrom, L. M., Felton, A. & Knott, C. D. Orangutan population density, forest structure and fruit availability in hand-logged and unlogged peat swamp forests in West Kalimantan, Indonesia. *Biological Conservation* **114**, 91-101, doi:10.1016/s0006-3207(03)00013-2 (2003).

128. Fensham, R., Dwyer, J., Eyre, T., Fairfax, R. & Wang, J. The effect of clearing on plant composition in mulga (Acacia aneura) dry forest, Australia. *Austral Ecology***37**, 183-192, doi:10.1111/j.1442-9993.2011.02261.x (2012).

129. Fermon, H., Waltert, M., Vane-Wright, R. I. & Muhlenberg, M. Forest use and vertical stratification in fruit-feeding butterflies of Sulawesi, Indonesia: impacts for conservation. *Biodiversity and Conservation* **14**, 333-350, doi:10.1007/s10531-004-5054-9 (2005).

130. Ferreira, C. & Alves, P. C. Impacto da implementação de medidas de gestão do habitat nas populações de coelho-bravo (*Oryctolagus cuniculus algirus*) no Parque Natural do Sudoeste Alentejano e Costa Vicentina. (Centro de Investigação em Biodiversidade e Recursos Genéticos (CIBIO), 2005).

131. Fierro, M. M., Cruz-Lopez, L., Sanchez, D., Villanueva-Gutierrez, R. & Vandame, R. Effect of biotic factors on the spatial distribution of stingless bees (Hymenoptera: Apidae, Meliponini) in fragmented neotropical habitats. *Neotropical Entomology* **41**, 95-104, doi:10.1007/s13744-011-0009-5 (2012).

132. Filgueiras, B., Iannuzzi, L. & Leal, I. Habitat fragmentation alters the structure of dung beetle communities in the Atlantic Forest. *Biological Conservation* **144**, 362-369, doi:10.1016/j.biocon.2010.09.013 (2011).

133. Flaspohler, D. J. *et al.* Long-term effects of fragmentation and fragment properties on bird species richness in Hawaiian forests. *Biological Conservation* **143**, 280-288, doi:10.1016/j.biocon.2009.10.009 (2010).

134. Fukuda, D., Tisen, O. B., Momose, K. & Sakai, S. Bat diversity in the vegetation mosaic around a lowland dipterocarp forest of Borneo. *Raffles Bulletin of Zoology***57**, 213-221 (2009).

135. Furlani, D., Ficetola, G. F., Colombo, G., Ugurlucan, M. & De Bernardi, F. Deforestation and the structure of frog communities in the Humedale Terraba-Sierpe, Costa Rica. *Zoological Science* **26**, 197-202, doi:10.2108/zsj.26.197 (2009).

136. Garden, J. G., McAlpine, C. A. & Possingham, H. P. Multi-scaled habitat considerations for conserving urban biodiversity: native reptiles and small mammals in Brisbane, Australia. *Landscape Ecology* **25**, 1013-1028, doi:10.1007/s10980-010-9476-z (2010).

137. Gardner, T. A., Hernandez, M. I. M., Barlow, J. & Peres, C. A. Understanding the biodiversity consequences of habitat change: the value of secondary and plantation forests for neotropical dung beetles. *Journal of Applied Ecology* **45**, 883-893, doi:10.1111/j.1365-2664.2008.01454.x (2008).

138. Gheler-Costa, C., Vettorazzi, C. A., Pardini, R. & Verdade, L. M. The distribution and abundance of small mammals in agroecosystems of southeastern Brazil. *Mammalia* **76**, 185-191, doi:10.1515/mammalia-2011-0109 (2012).

139. Giordani, P. Assessing the effects of forest management on epiphytic lichens in coppiced forests using different indicators. *Plant Biosystems* **146**, 628-637, doi:10.1080/11263504.2011.654136 (2012).

140. Giordano, S. *et al.* Biodiversity and trace element content of epiphytic bryophytes in urban and extraurban sites of southern Italy. *Plant Ecology* **170**, 1-14, doi:10.1023/B:VEGE.0000019025.36121.5d (2004).

141. Golodets, C., Kigel, J. & Sternberg, M. Recovery of plant species composition and ecosystem function after cessation of grazing in a Mediterranean grassland. *Plant and Soil* **329**, 365-378, doi:10.1007/s11104-009-0164-1 (2010).

142. Gottschalk, M. S., De Toni, D. C., Valente, V. L. S. & Hofmann, P. R. P. Changes in Brazilian Drosophilidae (Diptera) assemblages across an urbanisation gradient. *Neotropical Entomology* **36**, 848-862, doi:10.1590/S1519-566X2007000600005 (2007).

143. Goulson, D. *et al.* Effects of land use at a landscape scale on bumblebee nest density and survival. *Journal of Applied Ecology* **47**, 1207-1215, doi:10.1111/j.1365-2664.2010.01872.x (2010).

144. Goulson, D., Lye, G. C. & Darvill, B. Diet breadth, coexistence and rarity in bumblebees. *Biodiversity and Conservation* **17**, 3269-3288, doi:10.1007/s10531-008-9428-y (2008).

145. Gove, A. D., Majer, J. D. & Rico-Gray, V. Methods for conservation outside of formal reserve systems: The case of ants in the seasonally dry tropics of Veracruz,

Mexico. *Biological Conservation* **126**, 328-338, doi:10.1016/j.biocon.2005.06.008 (2005).

146. Grogan, J. *et al.* What loggers leave behind: Impacts on big-leaf mahogany (Swietenia macrophylla) commercial populations and potential for post-logging recovery in the Brazilian Amazon. *Forest Ecology and Management* **255**, 269-281, doi:10.1016/j.foreco.2007.09.048 (2008).

147. Gu, W.-B., Zhen-Rong, Y. & Dun-Xiao, H. Carabid community and its fluctuation in farmland of salinity transforming area in the North China Plain: a case study in Quzhou County, Hebei Province. *Biodiversity Science* **12**, 262-268 (2004).

148. Gutierrez-Lamus, D. L. Composition and abundance of Anura in two forest types (natural and planted) in the eastern Cordillera of Colombia. *Caldasia* **26**, 245-264 (2004).

149. Hanley, M. E. *et al.* Increased bumblebee abundance along the margins of a mass flowering crop: evidence for pollinator spill-over. *Oikos* **120**, 1618-1624, doi:10.1111/j.1600-0706.2011.19233.x (2011).

150. Hanley, M. E. *Unpublished data of bee diversity in UK croplands* (2005).

151. Hanley, M. E. *Unpublished data of bee diversity in UK croplands and urban habitats* (2011).

152. Hanson, T. R., Brunsfeld, S. J., Finegan, B. & Waits, L. P. Pollen dispersal and genetic structure of the tropical tree Dipteryx panamensis in a fragmented Costa Rican landscape. *Molecular Ecology* **17**, 2060-2073, doi:10.1111/j.1365-294X.2008.03726.x (2008).

153. Hashim, N., Akmal, W., Jusoh, W. & Nasir, M. Ant diversity in a Peninsular Malaysian mangrove forest and oil palm plantation. *Asian Myrmecology* **3**, 5-8 (2010).

154. Hatfield, R. G. & LeBuhn, G. Patch and landscape factors shape community assemblage of bumble bees, Bombus spp. (Hymenoptera : Apidae), in montane meadows. *Biological Conservation* **139**, 150-158, doi:10.1016/j.biocon.2007.06.019 (2007).

155. Hawes, J. *et al.* Diversity and composition of Amazonian moths in primary, secondary and plantation forests. *Journal of Tropical Ecology* **25**, 281-300, doi:10.1017/s0266467409006038 (2009).

156. Helden, A. J. & Leather, S. R. Biodiversity on urban roundabouts - Hemiptera, management and the species-area relationship. *Basic and Applied Ecology* **5**, 367-377, doi:10.1016/j.baae.2004.06.004 (2004).

157. Hernández, L., Delgado, L., Meier, W. & Duran, C. Empobrecimiento de bosques fragmentados en el norte de la Gran Sabana, Venezuela. *Interciencia* **37**, 891-898 (2012).

158. Herrmann, F., Westphal, C., Moritz, R. F. A. & Steffan-Dewenter, I. Genetic diversity and mass resources promote colony size and forager densities of a social bee (Bombus pascuorum) in agricultural landscapes. *Molecular Ecology* **16**, 1167-1178, doi:10.1111/j.1365-294X.2007.03226.x (2007).

159. Hietz, P. Conservation of vascular epiphyte diversity in Mexican coffee plantations. *Conservation Biology* **19**, 391-399, doi:10.1111/j.1523-1739.2005.00145.x (2005).

160. Higuera, D. & Wolf, J. H. D. Vascular epiphytes in dry oak forests show resilience to anthropogenic disturbance, Cordillera Oriental, Colombia. *Caldasia* **32**, 161-174 (2010).

161. Hilje, B. & Aide, T. M. Recovery of amphibian species richness and composition in a chronosequence of secondary forests, northeastern Costa Rica. *Biological Conservation* **146**, 170-176, doi:10.1016/j.biocon.2011.12.007 (2012).

162. Hoffmann, A. & Zeller, U. Influence of variations in land use intensity on species diversity and abundance of small mammals in the Nama Karoo, Namibia. *Belgian Journal of Zoology* **135**, 91-96 (2005).

163. Horgan, F. G. Invasion and retreat: shifting assemblages of dung beetles amidst changing agricultural landscapes in central Peru. *Biodiversity and Conservation* **18**, 3519-3541, doi:10.1007/s10531-009-9658-7 (2009).

164. Hu, C. & Cao, Z. P. Nematode community structure under compost and chemical fertilizer management practice, in the north China plain. *Experimental Agriculture***44**, 485-496, doi:10.1017/s0014479708006716 (2008).

165. Hylander, K. & Weibull, H. Do time-lagged extinctions and colonizations change the interpretation of buffer strip effectiveness? - a study of riparian bryophytes in the first decade after logging. *Journal of Applied Ecology* **49**, 1316-1324, doi:10.1111/j.1365-2664.2012.02218.x (2012).

166. Hylander, K. & Nemomissa, S. Complementary Roles of Home Gardens and Exotic Tree Plantations as Alternative Habitats for Plants of the Ethiopian Montane Rainforest. *Conservation Biology* **23**, 400-409, doi:10.1111/j.1523-1739.2008.01097.x (2009).

167. Ims, R. A. & Henden, J. A. Collapse of an arctic bird community resulting from ungulate-induced loss of erect shrubs. *Biological Conservation* **149**, 2-5, doi:10.1016/j.biocon.2012.02.008 (2012).

168. Isaacs Cubides, P. J. & Urbina Cardona, J. N. Anthropogenic disturbance and edge effects on anuran assemblages inhabiting cloud forest fragments in Colombia. *Natureza & Conservacao* **9**, 39-46, doi:10.4322/natcon.2011.004 (2011).

169. Ishitani, M., Kotze, D. J. & Niemela, J. Changes in carabid beetle assemblages across an urban-rural gradient in Japan. *Ecography* **26**, 481-489, doi:10.1034/j.1600-0587.2003.03436.x (2003).

170. Jacobs, C. T., Scholtz, C. H., Escobar, F. & Davis, A. L. V. How might intensification of farming influence dung beetle diversity (Coleoptera: Scarabaeidae) in Maputo Special Reserve (Mozambique)? *Journal of Insect Conservation* **14**, 389-399, doi:10.1007/s10841-010-9270-x (2010).

171. Johnson, M. F., Gómez, A. & Pinedo-Vasquez, M. Land use and mosquito diversity in the Peruvian Amazon. *Journal of Medical Entomology* **45**, 1023-1030, doi:10.1603/0022-2585(2008)45[1023:luamdi]2.0.co;2 (2008).

172. Jonsell, M. Old park trees as habitat for saproxylic beetle species. *Biodiversity and Conservation* **21**, 619-642, doi:10.1007/s10531-011-0203-0 (2012).

173. Julier, H. E. & Roulston, T. a. H. Wild bee abundance and pollination service in cultivated pumpkins: farm management, nesting behavior and landscape effects. *Journal of Economic Entomology* **102**, 563-573, doi:10.1603/029.102.0214 (2009).

174. Jung, T. S. & Powell, T. Spatial distribution of meadow jumping mice (Zapus hudsonius) in logged boreal forest of northwestern Canada. *Mammalian Biology* **76**, 678-682, doi:10.1016/j.mambio.2011.08.002 (2011).

175. Kapoor, V. Effects of rainforest fragmentation and shade-coffee plantations on spider communities in the Western Ghats, India. *Journal of Insect Conservation* **12**, 53-68, doi:10.1007/s10841-006-9062-5 (2008).

176. Kappes, H., Katzschner, L. & Nowak, C. Urban summer heat load: meteorological data as a proxy for metropolitan biodiversity. *Meteorologische Zeitschrift* **21**, 525-528, doi:10.1127/0941-2948/2012/0361 (2012).

177. Kati, V., Zografou, K., Tzirkalli, E., Chitos, T. & Willemse, L. Butterfly and grasshopper diversity patterns in humid Mediterranean grasslands: the roles of disturbance and environmental factors. *Journal of Insect Conservation* **16**, 807-818, doi:10.1007/s10841-012-9467-2 (2012).

178. Katovai, E., Burley, A. L. & Mayfield, M. M. Understory plant species and functional diversity in the degraded wet tropical forests of Kolombangara Island, Solomon Islands. *Biological Conservation* **145**, 214-224, doi:10.1016/j.biocon.2011.11.008 (2012).

179. Kessler, M. *et al.* Tree diversity in primary forest and different land use systems in Central Sulawesi, Indonesia. *Biodiversity and Conservation* **14**, 547-560, doi:10.1007/s10531-004-3914-7 (2005).

180. Kessler, M. *et al.* Alpha and beta diversity of plants and animals along a tropical land-use gradient. *Ecological Applications* **19**, 2142-2156, doi:10.1890/08-1074.1 (2009).

181. Knight, M. E. *et al.* Bumblebee nest density and the scale of available forage in arable landscapes. *Insect Conservation and Diversity* **2**, 116-124, doi:10.1111/j.1752-4598.2009.00049.x (2009).

182. Knop, E., Ward, P. I. & Wich, S. A. A comparison of orang-utan density in a logged and unlogged forest on Sumatra. *Biological Conservation* **120**, 183-188, doi:10.1016/j.biocon.2004.02.010 (2004).

183. Kohler, F., Verhulst, J., van Klink, R. & Kleijn, D. At what spatial scale do high-quality habitats enhance the diversity of forbs and pollinators in intensively farmed

landscapes? *Journal of Applied Ecology* **45**, 753-762, doi:10.1111/j.1365-2664.2007.01394.x (2008).

184. Koivula, M., Hyyrylainen, V. & Soininen, E. Carabid beetles (Coleoptera: Carabidae) at forest-farmland edges in southern Finland. *Journal of Insect Conservation* **8**, 297-309, doi:10.1007/s10841-004-0296-9 (2004).

185. Kolb, A. & Diekmann, M. Effects of environment, habitat configuration and forest continuity on the distribution of forest plant species. *Journal of Vegetation Science***15**, 199-208, doi:10.1111/j.1654-1103.2004.tb02255.x (2004).

186. Kőrösi, Á., Batáry, P., Orosz, A., Rédei, D. & Báldi, A. Effects of grazing, vegetation structure and landscape complexity on grassland leafhoppers (Hemiptera: Auchenorrhyncha) and true bugs (Hemiptera: Heteroptera) in Hungary. *Insect Conservation and Diversity* **5**, 57-66, doi:10.1111/j.1752-4598.2011.00153.x (2012).

187. Krauss, J., Klein, A. M., Steffan-Dewenter, I. & Tscharntke, T. Effects of habitat area, isolation, and landscape diversity on plant species richness of calcareous grasslands. *Biodiversity and Conservation* **13**, 1427-1439, doi:10.1023/b:bioc.0000021323.18165.58 (2004).

188. Krauss, J., Steffan-Dewenter, I. & Tscharntke, T. How does landscape context contribute to effects of habitat fragmentation on diversity and population density of butterflies? *Journal of Biogeography* **30**, 889-900, doi:10.1046/j.1365-2699.2003.00878.x (2003).

189. Kumar, R. & Shahabuddin, G. Effects of biomass extraction on vegetation structure, diversity and composition of forests in Sariska Tiger Reserve, India. *Environmental Conservation* **32**, 248-259, doi:10.1017/s0376892905002316 (2005).

190. Lachat, T. *et al.* Arthropod diversity in Lama forest reserve (South Benin), a mosaic of natural, degraded and plantation forests. *Biodiversity and Conservation* **15**, 3-23, doi:10.1007/s10531-004-1234-6 (2006).

191. Lantschner, M. V., Rusch, V. & Hayes, J. P. Habitat use by carnivores at different spatial scales in a plantation forest landscape in Patagonia, Argentina. *Forest Ecology and Management* **269**, 271-278, doi:10.1016/j.foreco.2011.12.045 (2012).

192. Lantschner, M. V., Rusch, V. & Peyrou, C. Bird assemblages in pine plantations replacing native ecosystems in NW Patagonia. *Biodiversity and Conservation* **17**, 969-989, doi:10.1007/s10531-007-9243-x (2008).

193. Latta, S. C., Tinoco, B. A., Astudillo, P. X. & Graham, C. H. Patterns and magnitude of temporal change in avian communities in the Ecuadorian Andes. *Condor***113**, 24-40, doi:10.1525/cond.2011.090252 (2011).

194. Légaré, J.-P., Hébert, C. & Ruel, J.-C. Alternative Silvicultural Practices in Irregular Boreal Forests: Response of Beetle Assemblages. *Silva Fennica* **45**, 937-956 (2011).

195. Letcher, S. G. & Chazdon, R. L. Rapid recovery of biomass, species richness, and species composition in a forest chronosequence in northeastern Costa Rica.*Biotropica* **41**, 608-617, doi:10.1111/j.1744-7429.2009.00517.x (2009).

196. Littlewood, N. A., Pakeman, R. J. & Pozsgai, G. Grazing impacts on Auchenorrhyncha diversity and abundance on a Scottish upland estate. *Insect Conservation and Diversity* **5**, 67-74, doi:10.1111/j.1752-4598.2011.00135.x (2012).

197. Liu, Y. H., Axmacher, J. C., Wang, C. L., Li, L. T. & Yu, Z. R. Ground beetle (Coleoptera: Carabidae) assemblages of restored semi-natural habitats and intensively cultivated fields in northern China. *Restoration Ecology* **20**, 234-239, doi:10.1111/j.1526-100X.2010.00755.x (2012).

198. Lo-Man-Hung, N. F., Gardner, T. A., Ribeiro-Júnior, M. A., Barlow, J. & Bonaldo, A. B. The value of primary, secondary, and plantation forests for Neotropical epigeic arachnids. *Journal of Arachnology* **36**, 394-401, doi:10.1636/CT07-136.1 (2008).

199. López-Quintero, C. A., Straatsma, G., Franco-Molano, A. E. & Boekhout, T. Macrofungal diversity in Colombian Amazon forests varies with regions and regimes of disturbance. *Biodiversity Conservation*, 2221–2243, doi:10.1007/s10531-012-0280-8 (2012).

200. Louhaichi, M., Salkini, A. K. & Petersen, S. L. Effect of small ruminant grazing on the plant community characteristics of semiarid Mediterranean ecosystems. *International Journal of Agriculture and Biology* **11**, 681-689 (2009).

201. Lucas-Borja, M. E. *et al.* The effects of human trampling on the microbiological properties of soil and vegetation in Mediterranean mountain areas. *Land Degradation & Development* **22**, 383-394, doi:10.1002/ldr.1014 (2011).

202. Luja, V., Herrando-Perez, S., Gonzalez-Solis, D. & Luiselli, L. Secondary rain forests are not havens for reptile species in tropical Mexico. *Biotropica* **40**, 747-757, doi:10.1111/j.1744-7429.2008.00439.x (2008).

203. Luskin, M. S. Flying Foxes Prefer to Forage in Farmland in a Tropical Dry Forest Landscape Mosaic in Fiji. *Biotropica* **42**, 246-250, doi:10.1111/j.1744-7429.2009.00577.x (2010).

204. MacSwiney, M. C. G., Vilchis, P. L., Clarke, F. M. & Racey, P. A. The importance of cenotes in conserving bat assemblages in the Yucatan, Mexico. *Biological Conservation* **136**, 499-509, doi:10.1016/j.biocon.2006.12.021 (2007).

205. Maeto, K. & Sato, S. Impacts of forestry on ant species richness and composition in warm-temperate forests of Japan. *Forest Ecology and Management* **187**, 213-223, doi:10.1016/s0378-1127(03)00333-5 (2004).

206. Magura, T., Horvath, R. & Tothmeresz, B. Effects of urbanization on ground-dwelling spiders in forest patches, in Hungary. *Landscape Ecology* **25**, 621-629, doi:10.1007/s10980-009-9445-6 (2010).

207. Mallari, N. A. D. *et al.* Population densities of understorey birds across a habitat gradient in Palawan, Philippines: implications for conservation. *Oryx* **45**, 234-242, doi:10.1017/s0030605310001031 (2011).

208. Malone, L. *et al.* Observations on bee species visiting white clover in New Zealand pastures. *Journal of Apicultural Research* **49**, 284-286, doi:10.3896/ibra.1.49.3.09 (2010).

209. Marin-Spiotta, E., Ostertag, R. & Silver, W. L. Long-term patterns in tropical reforestation: Plant community composition and aboveground biomass accumulation. *Ecological Applications* **17**, 828-839, doi:10.1890/06-1268 (2007).

210. Marshall, E. J. P., West, T. M. & Kleijn, D. Impacts of an agri-environment field margin prescription on the flora and fauna of arable farmland in different landscapes. *Agriculture Ecosystems & Environment* **113**, 36-44, doi:10.1016/j.agee.2005.08.036 (2006).

211. Martin, P. S., Gheler-Costa, C., Lopes, P. C., Rosalino, L. M. & Verdade, L. M. Terrestrial non-volant small mammals in agro-silvicultural landscapes of Southeastern Brazil. *Forest Ecology and Management* **282**, 185-195, doi:10.1016/j.foreco.2012.07.002 (2012).

212. Matsumoto, T., Itioka, T., Yamane, S. & Momose, K. Traditional land use associated with swidden agriculture changes encounter rates of the top predator, the army ant, in Southeast Asian tropical rain forests. *Biodiversity and Conservation* **18**, 3139-3151, doi:10.1007/s10531-009-9632-4 (2009).

213. Mayfield, M. M., Ackerly, D. & Daily, G. C. The diversity and conservation of plant reproductive and dispersal functional traits in human-dominated tropical landscapes. *Journal of Ecology* **94**, 522-536, doi:10.1111/j.1365-2745.2006.01108.x (2006).

214. McFrederick, Q. S. & LeBuhn, G. Are urban parks refuges for bumble bees Bombus spp. (Hymenoptera : Apidae)? *Biological Conservation* **129**, 372-382, doi:10.1016/j.biocon.2005.11.004 (2006).

215. McNamara, S., Erskine, P. D., Lamb, D., Chantalangsy, L. & Boyle, S. Primary tree species diversity in secondary fallow forests of Laos. *Forest Ecology and Management* **281**, 93-99, doi:10.1016/j.foreco.2012.06.004 (2012).

216. Meyer, B., Gaebele, V. & Steffan-Dewenter, I. D. Patch size and landscape effects on pollinators and seed set of the horseshoe vetch, hippocrepis comosa, in an agricultural landscape of central Europe. *Entomologia Generalis* **30**, 173-185 (2007).

217. Meyer, B., Jauker, F. & Steffan-Dewenter, I. Contrasting resource-dependent responses of hoverfly richness and density to landscape structure. *Basic and Applied Ecology* **10**, 178-186, doi:10.1016/j.baae.2008.01.001 (2009).

218. Mico, E., Garcia-Lopez, A., Brustel, H., Padilla, A. & Galante, E. Explaining the saproxylic beetle diversity of a protected Mediterranean area. *Biodiversity and Conservation* **22**, 889-904, doi:10.1007/s10531-013-0456-x (2013).

219. Milder, J. C. *et al.* Effects of farm and landscape management on bird and butterfly conservation in western Honduras. *Ecosphere* **1**, art2, doi:10.1890/es10-00003.1 (2010).

220. Miranda, M. V., Politi, N. & Rivera, L. O. Unexpected changes in the bird assemblage in areas under selective logging in piedmont forest in northwestern Argentina. *Ornitologia Neotropical* **21**, 323-337 (2010).

221. Moreno-Mateos, D. *et al.* Effects of land use on nocturnal birds in a Mediterranean agricultural landscape. *Acta Ornithologica* **46**, 173-182, doi:10.3161/000164511x625946 (2011).

222. Muchane, M. N. *et al.* Land use practices and their implications on soil macro-fauna in Maasai Mara ecosystem. *International Journal of Biodiversity and Conservation* **4**, 500-514, doi:10.5897/IJBC12.030 (2012).

223. Mudri-Stojnic, S., Andric, A., Jozan, Z. & Vujic, A. Pollinator diversity (Hymenoptera and Diptera) in semi-natural habitats in Serbia during summer. *Archives of Biological Sciences* **64**, 777-786, doi:10.2298/abs1202777s (2012).

224. Naidoo, R. Species richness and community composition of songbirds in a tropical forest-agricultural landscape. *Animal Conservation* **7**, 93-105, doi:10.1017/s1367943003001185 (2004).

225. Nakamura, A., Proctor, H. & Catterall, C. P. Using soil and litter arthropods to assess the state of rainforest restoration. *Ecological Management & Restoration* **4**, S20-S28, doi:10.1046/j.1442-8903.4.s.3.x (2003).

226. Naoe, S., Sakai, S. & Masaki, T. Effect of forest shape on habitat selection of birds in a plantation-dominant landscape across seasons: comparison between continuous and strip forests. *Journal of Forest Research* **17**, 219-223, doi:10.1007/s10310-011-0296-z (2012).

227. Navarrete, D. & Halffter, G. Dung beetle (Coleoptera : Scarabaeidae : Scarabaeinae) diversity in continuous forest, forest fragments and cattle pastures in a landscape of Chiapas, Mexico: the effects of anthropogenic changes. *Biodiversity and Conservation* **17**, 2869-2898, doi:10.1007/s10531-008-9402-8 (2008).

228. Navarro, I. L., Roman, A. K., Gomez, F. H. & Perez, H. A. Seasonal variation in dung beetles (Coleoptera: Scarabaeidae: Scarabaeinae) from Serrania de Coraza, Sucre (Colombia). *Revista Colombiana de Ciencia Animal* **3**, 102-110 (2011).

229. Neuschulz, E. L., Botzat, A. & Farwig, N. Effects of forest modification on bird community composition and seed removal in a heterogeneous landscape in South Africa. *Oikos* **120**, 1371-1379, doi:10.1111/j.1600-0706.2011.19097.x (2011).

230. Nicolas, V., Barriere, P., Tapiero, A. & Colyn, M. Shrew species diversity and abundance in Ziama Biosphere Reserve, Guinea: comparison among primary forest, degraded forest and restoration plots. *Biodiversity and Conservation* **18**, 2043-2061, doi:10.1007/s10531-008-9572-4 (2009).

231. Nielsen, A. *et al.* Assessing bee species richness in two Mediterranean communities: importance of habitat type and sampling techniques. *Ecological Research* **26**, 969-983, doi:10.1007/s11284-011-0852-1 (2011).

232. Noreika, N. & Kotze, D. J. Forest edge contrasts have a predictable effect on the spatial distribution of carabid beetles in urban forests. *Journal of Insect Conservation* **16**, 867-881, doi:10.1007/s10841-012-9474-3 (2012).

233. Noreika, N. New records of rare species of Coleoptera found in Ukmergė district in 2004-2005. *New and Rare for Lithuania Insect Species* **21**, 68-71 (2009).

234. Norfolk, O., Abdel-Dayem, M. & Gilbert, F. Rainwater harvesting and arthropod biodiversity within an arid agro-ecosystem. *Agriculture Ecosystems & Environment***162**, 8-14, doi:10.1016/j.agee.2012.08.007 (2012).

235. Noriega, J. A., Realpe, E. & Fagua, G. Diversidad de escarabajos coprofagos (Coleoptera: Scarabaeidae) en un bosque de galeria con tres estadios de alteracion. *Universitas Scientiarum* **12**, 51-63 (2007).

236. Noriega, J. A., Palacio, J. M., Monroy-G, J. D. & Valencia, E. Estructura de un ensamblaje de escarabajos coprofagos (Coleoptera: Scarabaeinae) en tres sitios con diferente uso del suelo en Antioquia, Colombia. *Actualidades Biologicas (Medellin)* **34**, 43-54 (2012).

237. Nöske, N. M. *et al.* Disturbance effects on diversity of epiphytes and moths in a montane forest in Ecuador. *Basic and Applied Ecology* **9**, 4-12, doi:10.1016/j.baae.2007.06.014 (2008).

238. Numa, C., Verdu, J. R., Rueda, C. & Galante, E. Comparing dung beetle species assemblages between protected areas and adjacent pasturelands in a Mediterranean savanna landscape. *Rangeland Ecology & Management* **65**, 137-143, doi:10.2111/rem-d-10-00050.1 (2012).

239. O'Connor, T. G. Influence of land use on plant community composition and diversity in Highland Sourveld grassland in the southern Drakensberg, South Africa. *Journal of Applied Ecology* **42**, 975-988, doi:10.1111/j.1365-2664.2005.01065.x (2005).

240. O'Dea, N. & Whittaker, R. J. How resilient are Andean montane forest bird communities to habitat degradation? *Biodiversity and Conservation* **16**, 1131-1159, doi:10.1007/s10531-006-9095-9 (2007).

241. Ofori-Boateng, C. *et al.* Differences in the effects of selective logging on amphibian assemblages in three west African forest types. *Biotropica* **45**, 94-101, doi:10.1111/j.1744-7429.2012.00887.x (2013).

242. Oke, C. Land snail diversity in post extraction secondary forest reserves in Edo State, Nigeria. *African Journal of Ecology* **51**, 244-254, doi:10.1111/aje.12029 (2013).

243. Oke, O. C. & Chokor, J. U. The effect of land use on snail species richness and diversity in the tropical rainforest of south-western Nigeria. *African Scientist* **10**, 95-108 (2009).

244. Oliveira, D. E., Carrijo, T. F. & Brandão, D. Species composition of termites (Isoptera) in different Cerrado vegetation physiognomies. *Sociobiology* **60**, 190-197, doi:10.13102/sociobiology.v60i2.190-197 (2013).

245. Osgathorpe, L. M., Park, K. & Goulson, D. The use of off-farm habitats by foraging bumblebees in agricultural landscapes: implications for conservation management. *Apidologie* **43**, 113-127, doi:10.1007/s13592-011-0083-z (2012).

246. Otavo, S. E., Parrado-Rosselli, A. & Noriega, J. A. Scarabaeoidea superfamily (Insecta: Coleoptera) as a bioindicator element of anthropogenic disturbance in an amazon national park. *Revista De Biologia Tropical* **61**, 735-752 (2013).

247. Otto, C. R. V. & Roloff, G. J. Songbird response to green-tree retention prescriptions in clearcut forests. *Forest Ecology and Management* **284**, 241-250, doi:10.1016/j.foreco.2012.07.016 (2012).

248. Paradis, S. & Work, T. T. Partial cutting does not maintain spider assemblages within the observed range of natural variability in Eastern Canadian black spruce forests. *Forest Ecology and Management* **262**, 2079-2093, doi:10.1016/j.foreco.2011.08.032 (2011).

249. Paritsis, J. & Aizen, M. A. Effects of exotic conifer plantations on the biodiversity of understory plants, epigeal beetles and birds in Nothofagus dombeyi forests.*Forest Ecology and Management* **255**, 1575-1583, doi:10.1016/j.foreco.2007.11.015 (2008).

250. Parra-H, A. & Nates-Parra, G. Variation of the orchid bees community (Hymenoptera : Apidae) in three altered habitats of the Colombian "llano" piedmont. *Revista De Biologia Tropical* **55**, 931-941 (2007).

251. Pelegrin, N. & Bucher, E. H. Effects of habitat degradation on the lizard assemblage in the Arid Chaco, central Argentina. *Journal of Arid Environments* **79**, 13-19, doi:10.1016/j.jaridenv.2011.11.004 (2012).

252. Phalan, B., Onial, M., Balmford, A. & Green, R. Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science* **333**, 1289-1291, doi:10.1126/science.1208742 (2011).

253. Pillsbury, F. C. & Miller, J. R. Habitat and landscape characteristics underlying anuran community structure along an urban-rural gradient. *Ecological Applications* **18**, 1107-1118, doi:10.1890/07-1899.1 (2008).

254. Pineda, E. & Halffter, G. Species diversity and habitat fragmentation: frogs in a tropical montane landscape in Mexico. *Biological Conservation* **117**, 499-508, doi:10.1016/j.biocon.2003.08.009 (2004).

255. Politi, N., Hunter, M., Jr. & Rivera, L. Assessing the effects of selective logging on birds in Neotropical piedmont and cloud montane forests. *Biodiversity and Conservation* **21**, 3131-3155, doi:10.1007/s10531-012-0358-3 (2012).

256. Poveda, K., Martinez, E., Kersch-Becker, M., Bonilla, M. & Tscharntke, T. Landscape simplification and altitude affect biodiversity, herbivory and Andean potato yield. *Journal of Applied Ecology* **49**, 513-522, doi:10.1111/j.1365-2664.2012.02120.x (2012).

257. Power, E. F., Kelly, D. L. & Stout, J. C. Organic farming and landscape structure: effects on insect-pollinated plant diversity in intensively managed grasslands. *Plos One* **7**, doi:10.1371/journal.pone.0038073 (2012).

258. Power, E. F. & Stout, J. C. Organic dairy farming: impacts on insect-flower interaction networks and pollination. *Journal of Applied Ecology* **48**, 561-569, doi:10.1111/j.1365-2664.2010.01949.x (2011).

259. Presley, S. J., Willig, M. R., Wunderle, J. M., Jr. & Saldanha, L. N. Effects of reduced-impact logging and forest physiognomy on bat populations of lowland

Amazonian forest. *Journal of Applied Ecology* **45**, 14-25, doi:10.1111/j.1365-2664.2007.01373.x (2008).

260. Proenca, V. M., Pereira, H. M., Guilherme, J. & Vicente, L. Plant and bird diversity in natural forests and in native and exotic plantations in NW Portugal. *Acta Oecologica-International Journal of Ecology* **36**, 219-226, doi:10.1016/j.actao.2010.01.002 (2010).

261. Quaranta, M. *et al.* Wild bees in agroecosystems and semi-natural landscapes. 1997-2000 collection period in Italy. *Bulletin of Insectology* **57**, 11-62 (2004).

262. Quintero, C., Laura Morales, C. & Adrian Aizen, M. Effects of anthropogenic habitat disturbance on local pollinator diversity and species turnover across a precipitation gradient. *Biodiversity and Conservation* **19**, 257-274, doi:10.1007/s10531-009-9720-5 (2010).

263. Redpath, N., Osgathorpe, L. M., Park, K. & Goulson, D. Crofting and bumblebee conservation: The impact of land management practices on bumblebee populations in northwest Scotland. *Biological Conservation* **143**, 492-500, doi:10.1016/j.biocon.2009.11.019 (2010).

264. Reid, J. L., Harris, J. B. C. & Zahawi, R. A. Avian habitat preference in tropical forest restoration in southern Costa Rica. *Biotropica* **44**, 350-359, doi:10.1111/j.1744-7429.2011.00814.x (2012).

265. Reis, Y. T. & Cancello, E. M. Termite (Insecta, Isoptera) richness in primary and secondary Atlantic Forest in southeastern Bahia. *Iheringia Serie Zoologia* **97**, 229-234 (2007).

266. Rey-Velasco, J. C. & Miranda-Esquivel, D. R. *Habitat modification in Andean forest: the response of ground beetles (Coleoptera: Carabidae) on the northeastern Colombian Andes* (2012).

267. Ribeiro, D. B. & Freitas, A. V. L. The effect of reduced-impact logging on fruit-feeding butterflies in Central Amazon, Brazil. *Journal of Insect Conservation* **16**, 733-744, doi:10.1007/s10841-012-9458-3 (2012).

268. Richardson, B. A., Richardson, M. J. & Soto-Adames, F. N. Separating the effects of forest type and elevation on the diversity of litter invertebrate communities in a humid tropical forest in Puerto Rico. *Journal of Animal Ecology* **74**, 926-936, doi:10.1111/j.1365-2656.2005.00990.x (2005).

269. Robles, C. A., Carmaran, C. C. & Lopez, S. E. Screening of xylophagous fungi associated with Platanus acerifolia in urban landscapes: Biodiversity and potential biodeterioration. *Landscape and Urban Planning* **100**, 129-135, doi:10.1016/j.landurbplan.2010.12.003 (2011).

270. Rodrigues, M. M., Uchoa, M. A. & Ide, S. Dung beetles (Coleoptera: Scarabaeoidea) in three landscapes in Mato Grosso do Sul, Brazil. *Brazilian Journal of Biology***73**, 211-220 (2013).

271. Römbke, J., Schmidt, P. & Höfer, H. The earthworm fauna of regenerating forests and anthropogenic habitats in the coastal region of Paraná. *Pesquisa Agropecuaria Brasileira* **44**, 1040-1049, doi:10.1590/S0100-204X2009000800037 (2009).

272. Romero-Duque, L. P., Jaramillo, V. J. & Perez-Jimenez, A. Structure and diversity of secondary tropical dry forests in Mexico, differing in their prior land-use history. *Forest Ecology and Management* **253**, 38-47, doi:10.1016/j.foreco.2007.07.002 (2007).

273. Rosselli, L. *Factores ambientales relacionados con la presencia y abundancia de las aves de los humedales de la Sabana de Bogotá* PhD thesis, Universidad Nacional de Colombia, (2011).

274. Rousseau, L., Fonte, S. J., Tellez, O., van der Hoek, R. & Lavelle, P. Soil macrofauna as indicators of soil quality and land use impacts in smallholder agroecosystems of western Nicaragua. *Ecological Indicators* **27**, 71-82, doi:10.1016/j.ecolind.2012.11.020 (2013).

275. Safian, S., Csontos, G. & Winkler, D. Butterfly community recovery in degraded rainforest habitats in the Upper Guinean Forest Zone (Kakum forest, Ghana). *Journal of Insect Conservation* **15**, 351-359, doi:10.1007/s10841-010-9343-x (2011).

276. Sakchoowong, W., Nomura, S., Ogata, K. & Chanpaisaeng, J. Diversity of pselaphine beetles (Coleoptera : Staphylinidae : Pselaphinae) in eastern Thailand. *Entomological Science* **11**, 301-313, doi:10.1111/j.1479-8298.2008.00281.x (2008).

277. Saldana-Vazquez, R. A., Sosa, V. J., Hernandez-Montero, J. R. & Lopez-Barrera, F. Abundance responses of frugivorous bats (Stenodermatinae) to coffee cultivation and selective logging practices in mountainous central Veracruz, Mexico. *Biodiversity and Conservation* **19**, 2111-2124, doi:10.1007/s10531-010-9829-6 (2010).

278. Samnegård, U., Persson, A. S. & Smith, H. G. Gardens benefit bees and enhance pollination in intensively managed farmland. *Biological Conservation* **144**, 2602-2606, doi:10.1016/j.biocon.2011.07.008 (2011).

279. Santana, J., Porto, M., Gordinho, L., Reino, L. & Beja, P. Long-term responses of Mediterranean birds to forest fuel management. *Journal of Applied Ecology* **49**, 632-643, doi:10.1111/j.1365-2664.2012.02141.x (2012).

280. Savage, J., Wheeler, T. A., Moores, A. M. A. & Taillefer, A. G. Effects of habitat size, vegetation cover, and surrounding land use on diptera diversity in temperate nearctic bogs. *Wetlands* **31**, 125-134, doi:10.1007/s13157-010-0133-8 (2011).

281. Schmidt, A. C., Fraser, L. H., Carlyle, C. N. & Bassett, E. R. L. Does cattle grazing affect ant abundance and diversity in temperate grasslands? *Rangeland Ecology & Management* **65**, 292-298, doi:10.2111/rem-d-11-00100.1 (2012).

282. Schon, N. L., Mackay, A. D. & Minor, M. A. Soil fauna in sheep-grazed hill pastures under organic and conventional livestock management and in an adjacent ungrazed pasture. *Pedobiologia* **54**, 161-168, doi:10.1016/j.pedobi.2011.01.001 (2011).

283. Schüepp, C., Herrmann, J. D., Herzog, F. & Schmidt-Entling, M. H. Differential effects of habitat isolation and landscape composition on wasps, bees, and their enemies. *Oecologia* **165**, 713-721, doi:10.1007/s00442-010-1746-6 (2011).

284. Schüepp, C., Rittiner, S. & Entling, M. H. High Bee and Wasp Diversity in a Heterogeneous Tropical Farming System Compared to Protected Forest. *Plos One* **7**, doi:10.1371/journal.pone.0052109 (2012).

285. Scott, D. M. *et al.* The impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny forest, southern Madagascar. *Biological Conservation* **127**, 72-87, doi:10.1016/j.biocon.2005.07.014 (2006).

286. Sedlock, J. L. *et al.* Bat diversity in tropical forest and agro-pastoral habitats within a protected area in the Philippines. *Acta Chiropterologica* **10**, 349-358, doi:10.3161/150811008x414926 (2008).

287. Shafie, N. J., Sah, S. A. M., Latip, N. S. A., Azman, N. M. & Khairuddin, N. L. Diversity pattern of bats at two contrasting habitat types along Kerian River, Perak, Malaysia. *Tropical Life Sciences Research* **22**, 13-22 (2011).

288. Shahabuddin, G. & Kumar, R. Effects of extractive disturbance on bird assemblages, vegetation structure and floristics in tropical scrub forest, Sariska Tiger Reserve, India. *Forest Ecology and Management* **246**, 175-185, doi:10.1016/j.foreco.2007.03.061 (2007).

289. Sheil, D. *et al.* Exploring biological diversity, environment and local people's perspectives in forest landscapes: Methods for a multidisciplinary landscape assessment. (Center for International Forestry Research (CIFOR), Jakarta, 2002).

290. Sheldon, F., Styring, A. & Hosner, P. Bird species richness in a Bornean exotic tree plantation: A long-term perspective. *Biological Conservation* **143**, 399-407, doi:10.1016/j.biocon.2009.11.004 (2010).

291. Shuler, R. E., Roulston, T. H. & Farris, G. E. Farming practices influence wild pollinator populations on squash and pumpkin. *Journal of Economic Entomology* **98**, 790-795, doi:10.1603/0022-0493-98.3.790 (2005).

292. Silva, F. A. B., Costa, C. M. Q., Moura, R. C. & Farias, A. I. Study of the dung beetle (Coleoptera: Scarabaeidae) community at two sites: atlantic forest and clear-cut, Pernambuco, Brazil. *Environmental Entomology* **39**, 359-367, doi:10.1603/en09180 (2010).

293. da Silva, P. G. *Espécies de Scarabaeinae (Coleoptera: Scarabaeidae) de fragmentos florestais com diferentes níveis de alteração em Santa Maria, Rio Grande do Sul*MSc thesis, Universidade Federal de Santa Maria, (2011).

294. Slade, E. M., Mann, D. J. & Lewis, O. T. Biodiversity and ecosystem function of tropical forest dung beetles under contrasting logging regimes. *Biological Conservation* **144**, 166-174, doi:10.1016/j.biocon.2010.08.011 (2011).

295. Smith-Pardo, A. & Gonzalez, V. H. Bee diversity (Hymenoptera : Apoidea) in a tropical rainforest succession. *Acta Biologica Colombiana* **12**, 43-55 (2007).

296. Sodhi, N. S. *et al.* Deforestation and avian extinction on tropical landbridge islands. *Conservation Biology* **24**, 1290-1298, doi:10.1111/j.1523-1739.2010.01495.x (2010).

297. Sosa, R. A., Benz, V. A., Galea, J. M. & Poggio Herrero, I. V. Efecto del grado de disturbio sobre el ensamble de aves en la reserva provincial Parque Luro, La Pampa, Argentina. *Revista de la Asociación Argentina de Ecología de Paisajes* **1**, 101-110 (2010).

298. de Souza, V. M., de Souza, B. & Morato, E. F. Effect of the forest succession on the anurans (Amphibia: Anura) of the Reserve Catuaba and its periphery, Acre, southwestern Amazonia. *Revista Brasileira De Zoologia* **25**, 49-57 (2008).

299. Sridhar, H., Raman, T. R. S. & Mudappa, D. Mammal persistence and abundance in tropical rainforest remnants in the southern Western Ghats, India. *Current Science* **94**, 748-757 (2008).

300. St-Laurent, M. H., Ferron, J., Hins, C. & Gagnon, R. Effects of stand structure and landscape characteristics an habitat use by birds and small mammals in managed boreal forest of eastern Canada. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere* **37**, 1298-1309, doi:10.1139/xo6-295 (2007).

301. Ström, L., Hylander, K. & Dynesius, M. Different long-term and short-term responses of land snails to clear-cutting of boreal stream-side forests. *Biological Conservation* **142**, 1580-1587, doi:10.1016/j.biocon.2009.02.028 (2009).

302. Struebig, M. J., Kingston, T., Zubaid, A., Mohd-Adnan, A. & Rossiter, S. J. Conservation value of forest fragments to Palaeotropical bats. *Biological Conservation***141**, 2112-2126, doi:10.1016/j.biocon.2008.06.009 (2008).

303. Su, Z. M., Zhang, R. Z. & Qiu, J. X. Decline in the diversity of willow trunk-dwelling weevils (Coleoptera: Curculionoidea) as a result of urban expansion in Beijing, China. *Journal of Insect Conservation* **15**, 367-377, doi:10.1007/s10841-010-9310-6 (2011).

304. Sugiura, S., Tsuru, T., Yamaura, Y. & Makihara, H. Small off-shore islands can serve as important refuges for endemic beetle conservation. *Journal of Insect Conservation* **13**, 377-385, doi:10.1007/s10841-008-9185-y (2009).

305. Summerville, K. S. Managing the forest for more than the trees: effects of experimental timber harvest on forest Lepidoptera. *Ecological Applications* **21**, 806-816, doi:10.1890/10-0715.1 (2011).

306. Summerville, K. S., Conoan, C. J. & Steichen, R. M. Species traits as predictors of lepidopteran composition in restored and remnant tallgrass prairies. *Ecological Applications* **16**, 891-900, doi:10.1890/1051-0761(2006)016[0891:stapol]2.0.co;2 (2006).

307. Sung, Y. H., Karraker, N. E. & Hau, B. C. H. Terrestrial herpetofaunal assemblages in secondary forests and exotic Lophostemon confertus plantations in South China. *Forest Ecology and Management* **270**, 71-77, doi:10.1016/j.foreco.2012.01.011 (2012).

308. Threlfall, C. G., Law, B. & Banks, P. B. Sensitivity of insectivorous bats to urbanization: Implications for suburban conservation planning. *Biological Conservation***146**, 41-52, doi:10.1016/j.biocon.2011.11.026 (2012).

309. Tonietto, R., Fant, J., Ascher, J., Ellis, K. & Larkin, D. A comparison of bee communities of Chicago green roofs, parks and prairies. *Landscape and Urban Planning***103**, 102-108, doi:10.1016/j.landurbplan.2011.07.004 (2011).

310. Turner, E. C. & Foster, W. A. The impact of forest conversion to oil palm on arthropod abundance and biomass in Sabah, Malaysia. *Journal of Tropical Ecology* **25**, 23-30, doi:10.1017/s0266467408005658 (2009).

311. Tylianakis, J. M., Klein, A. M. & Tscharntke, T. Spatiotemporal variation in the diversity of hymenoptera across a tropical habitat gradient. *Ecology* **86**, 3296-3302, doi:10.1890/05-0371 (2005).

312. Vanbergen, A. J., Woodcock, B. A., Watt, A. D. & Niemela, J. Effect of land-use heterogeneity on carabid communities at the landscape scale. *Ecography* **28**, 3-16, doi:10.1111/j.0906-7590.2005.03991.x (2005).

313. Vassilev, K., Pedashenko, H., Nikolov, S. C., Apostolova, I. & Dengler, J. Effect of land abandonment on the vegetation of upland semi-natural grasslands in the Western Balkan Mts., Bulgaria. *Plant Biosystems* **145**, 654-665, doi:10.1080/11263504.2011.601337 (2011).

314. Vázquez, D. P. & Simberloff, D. Ecological specialization and susceptibility to disturbance: Conjectures and refutations. *American Naturalist* **159**, 606-623, doi:10.1086/339991 (2002).

315. Verboven, H. A. F., Brys, R. & Hermy, M. Sex in the city: Reproductive success of Digitalis purpurea in a gradient from urban to rural sites. *Landscape and Urban Planning* **106**, 158-164, doi:10.1016/j.landurbplan.2012.02.015 (2012).

316. Verdasca, M. J. *et al.* Forest fuel management as a conservation tool for early successional species under agricultural abandonment: The case of Mediterranean butterflies. *Biological Conservation* **146**, 14-23, doi:10.1016/j.biocon.2011.10.031 (2012).

317. Verdú, J. R. *et al.* Grazing promotes dung beetle diversity in the xeric landscape of a Mexican Biosphere Reserve. *Biological Conservation* **140**, 308-317, doi:10.1016/j.biocon.2007.08.015 (2007).

318. Vergara, C. H. & Badano, E. I. Pollinator diversity increases fruit production in Mexican coffee plantations: The importance of rustic management systems.

Agriculture Ecosystems & Environment **129**, 117-123, doi:10.1016/j.agee.2008.08.001 (2009).

319. Vergara, P. M. & Simonetti, J. A. Avian responses to fragmentation of the Maulino Forest in central Chile. *Oryx* **38**, 383-388, doi:10.1017/s0030605304000742 (2004).

320. Walker, T. R., Crittenden, P. D., Young, S. D. & Prystina, T. An assessment of pollution impacts due to the oil and gas industries in the Pechora basin, north-eastern European Russia. *Ecological Indicators* **6**, 369-387, doi:10.1016/j.ecolind.2005.03.015 (2006).

321. Wang, Y., Bao, Y., Yu, M., Xu, G. & Ding, P. Nestedness for different reasons: the distributions of birds, lizards and small mammals on islands of an inundated lake. *Diversity and Distributions* **16**, 862-873, doi:10.1111/j.1472-4642.2010.00682.x (2010).

322. Watling, J. I., Gerow, K. & Donnelly, M. A. Nested species subsets of amphibians and reptiles on Neotropical forest islands. *Animal Conservation* **12**, 467-476, doi:10.1111/j.1469-1795.2009.00274.x (2009).

323. Weller, B. & Ganzhorn, J. U. Carabid beetle community composition, body size, and fluctuating asymmetry along an urban-rural gradient. *Basic and Applied Ecology* **5**, 193-201, doi:10.1078/1439-1791-00220 (2004).

324. Wells, K., Kalko, E. K. V., Lakim, M. B. & Pfeiffer, M. Effects of rain forest logging on species richness and assemblage composition of small mammals in Southeast Asia. *Journal of Biogeography* **34**, 1087-1099, doi:10.1111/j.1365-2699.2006.01677.x (2007).

325. Williams, C. D., Sheahan, J. & Gormally, M. J. Hydrology and management of turloughs (temporary lakes) affect marsh fly (Sciomyzidae: Diptera) communities.

Insect Conservation and Diversity **2**, 270-283, doi:10.1111/j.1752-4598.2009.00064.x (2009).

326. Willig, M. R. *et al.* Phyllostomid bats of lowland Amazonia: Effects of habitat alteration on abundance. *Biotropica* **39**, 737-746, doi:10.1111/j.1744-7429.2007.00322.x (2007).

327. Winfree, R., Griswold, T. & Kremen, C. Effect of human disturbance on bee communities in a forested ecosystem. *Conservation Biology* **21**, 213-223, doi:10.1111/j.1523-1739.2006.00574.x (2007).

328. Woinarski, J. C. Z. *et al.* Fauna assemblages in regrowth vegetation in tropical open forests of the Northern Territory, Australia. *Wildlife Research* **36**, 675-690, doi:10.1071/wr08128 (2009).

329. Woodcock, B. A. *et al.* The potential of grass field margin management for enhancing beetle diversity in intensive livestock farms. *Journal of Applied Ecology* **44**, 60-69, doi:10.1111/j.1365-2664.2006.01258.x (2007).

330. Wunderle, J. M., Henriques, L. M. P. & Willig, M. R. Short-term responses of birds to forest gaps and understory: An assessment of reduced-impact logging in a lowland Amazon forest. *Biotropica* **38**, 235-255, doi:10.1111/j.1744-7429.2006.00138.x (2006).

331. Yoshikura, S., Yasui, S. & Kamijo, T. Comparative study of forest-dwelling bats' abundances and species richness between old-growth forests and conifer plantations in Nikko National Park, central Japan. *Mammal Study* **36**, 189-198, doi:10.3106/041.036.0402 (2011).

332. Zaitsev, A. S., Chauvat, M., Pflug, A. & Wolters, V. Oribatid mite diversity and community dynamics in a spruce chronosequence. *Soil Biology & Biochemistry* **34**, 1919-1927, doi:10.1016/s0038-0717(02)00208-0 (2002).

333. Zaitsev, A. S., Wolters, V., Waldhardt, R. & Dauber, J. Long-term succession of oribatid mites after conversion of croplands to grasslands. *Applied Soil Ecology* **34**, 230-239, doi:10.1016/j.apsoil.2006.01.005 (2006).

334. Zimmerman, G., Bell, F. W., Woodcock, J., Palmer, A. & Paloniemi, J. Response of breeding songbirds to vegetation management in conifer plantations established in boreal mixedwoods. *The Forestry Chronicle* **87**, 217-224 (2011).

335. Roskov, Y. *et al. Species 2000 & ITIS Catalogue of Life, 2013 Annual Checklist.* (2013).

336. Gotelli, N. J. & Colwell, R. K. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecol. Lett.* **4,** 379–391 (2001).

337. Violle, C. *et al.* Let the concept of trait be functional! *Oikos* **116,** 882–892 (2007).

338. Kattge, J. *et al.* TRY - a global database of plant traits. *Glob. Chang. Biol.* **17,** 2905–2935 (2011).

339. Jones, K. E. *et al.* PanTHERIA: a species-level database of life history, ecology, and geography of extant and recently extinct mammals. *Ecology* **90,** 2648 (2009).

340. Cooper, N., Bielby, J., Thomas, G. H. & Purvis, A. Macroecology and extinction risk correlates of frogs. *Glob. Ecol. Biogeogr.* **17,** 211–221 (2008).

341. AmphibiaWeb. http://amphibiaweb.org/ (2013).

342. Sunyer, J., Páiz, G., Dehling, D. M. & Köhler, G. A collection of amphibians from Río San Juan, southeastern Nicaragua. *Herpetol. Notes* **2,** 189–202 (2009).

343. Zug, G. R. & Zug, P. B. The marine toad *Bufo marinus*: a natural history resumé of native populations. *Smithson. Contrib. to Zool.* **284,** 1–58 (1979).

344. Amphibians & Reptiles of Peninsular Malaysia. http://www.amphibia.my/ (2009)

345. Shahriza, S., Ibrahim, H. J. & Shahrul Anuar, M. S. The correlation between total rainfall and breeding parameters of white-lipped frog, *Rana labialis* (Anura: Ranidae) in Kedah, Malaysia. *Trop. Nat. Hist.* **10,** 131–139 (2010).

346. Bain, R. H. & Quang Truong, N. Three new species of narrow-mouth frogs (Genus: *Microhyla* from Indochina, with comments on *Microhyla annamensis* and *Microhyla palmipes*. *Copeia* **2004,** 507–524 (2004).

347. Su, M.-Y., Kam, Y.-C. & Fellers, G. M. Effectiveness of amphibian monitoring techniques in a Taiwanese subtropical forest. *Herpetol. J.* **15,** 73–79 (2005).

348. Matson, T. O. A morphometric comparison of gray treefrogs, *Hyla chrysoscelis* and *H. versicolor*, from Ohio. *Ohio J. Sci.* **90,** 98–101 (1990).

349. Ningombam, B. & Bordoloi, S. Amphibian fauna of Loktak Lake, Manipur, India with ten new records for the state. *Zoos' Print J.* **22,** 2688–2690 (2007).

350. Lance, S. L. & Wells, K. D. Are spring peeper satellite males physiologically inferior to calling males? *Copeia* **1993,** 1162–1166 (1993).

351. Da Silva, E. T., dos Reis, E. P., Feio, R. N. & Filho, O. P. R. Diet of the invasive frog *Lithobates catesbeianus* (Shaw, 1802) (Anura: Ranidae) in Viçosa, Minas Gerais State, Brazil. *South Am. J. Herpetol.* **4,** 286–294 (2009).

352. Blomquist, S. M. & Hunter Jr., M. L. A multi-scale assessment of habitat selection and movement patterns by northern leopard frogs (*Lithobates [Rana] pipiens*) in a managed forest. *Herpetol. Conserv. Biol.* **4,** 142–160 (2009).

353. Caramaschi, U. & da Cruz, C. A. G. Redescription of *Chiasmocleis albopunctata* (Boettger) and description of a new species of *Chiasmocleis* (Anura: Microhylidae). *Herpetologica* **53,** 259–268 (1997).

354. Brasileiro, C. A., Sawaya, R. J., Kiefer, M. C. & Martins, M. Amphibians of an open cerrado fragment in southeastern Brazil. *Biota Neotrop.* **5,** BN00405022005 (2005).

355. De Almeida Prado, C. P. Estratégias reprodutivas em uma comunidade de anuros no pantanal, estado de Mato Grosso do Sul, Brasil. (PhD Thesis, Universidade Estadual Paulista, 2003).

356. De Almeida Prado, C. P., Uetanabaro, M. & Lopes, F. S. Reproductive strategies of *Leptodactylus chaquensis* and *L. podicipinus* in the Pantanal, Brazil. *J. Herpetol.* **34,** 135–139 (2000).

357. De Carvalho, T. R., Giaretta, A. A. & Facure, K. G. A new species of *Hypsiboas* Wagler (Anura: Hylidae) closely related to *H. multifasciatus* Günther from southeastern Brazil. *Zootaxa* **2521,** 37–52 (2010).

358. Heyer, W. R. & Heyer, M. M. *Leptodactylus elenae* Heyer. *Cat. Am. Amphib. Reptil.* 742.1–742.5 (2002).

359. Heyer, W. R. Variation within the *Leptodactylus podicipinus-wagneri* complex of frogs (Amphibia: Leptodactylidae). *Smithson. Contrib. to Zool.* **546,** (1994).

360. Jungfer, K.-H. & Hödl, W. A new species of *Osteocephalus* from Ecuador and a redescription of *O. leprieurii* (Dumeril & Bibron, 1841) (Anura: Hylidae). *Amphibia-Reptilia* **23,** 21–46 (2002).

361. Fouquet, A., Gaucher, P., Blanc, M. & Velez-Rodriguez, C. M. Description of two new species of *Rhinella* (Anura: Bufonidae) from the lowlands of the Guiana shield. *Zootaxa* **1663,** 17–32 (2007).

362. Lynch, J. D. A review of the leptodactylid frogs of the genus *Pseudopaludicola* in Northern South America. *Copeia* **1989,** 577–588 (1989).

363. González, C. E. & Hamann, M. I. Nematode parasites of two anuran species *Rhinella schneideri* (Bufonidae) and *Scinax acuminatus* (Hylidae) from Corrientes, Argentina. *Rev. Biol. Trop.* **56,** 2147–2161 (2008).

364. Pombal Jr., J. P., Bilate, M., Gambale, P. G., Signorelli, L. & Bastos, R. P. A new miniature treefrog of the *Scinax ruber* clade from the cerrado of central Brazil (Anura: Hylidae). *Herpetologica* **67,** 288–299 (2011).

365. Ibáñez, R., Jaramillo, C. A. & Solis, F. A. Description of the advertisement call of a species without vocal sac: *Craugastor gollmeri* (Amphibia: Craugastoridae). *Zootaxa* **3184,** 67–68 (2012).

366. Hertz, A., Hauenschild, F., Lotzkat, S. & Köhler, G. A new golden frog species of the genus *Diasporus* (Amphibia, Eleutherodactylidae) from the Cordillera Central, western Panama. *Zookeys* **196,** 23–46 (2012).

367. Goldberg, S. R. & Bursey, C. R. Helminths from fifteen species of frogs (Anura, Hylidae) from Costa Rica. *Phyllomedusa* **7,** 25–33 (2008).

368. Bennett, W. O., Summers, A. P. & Brainerd, E. L. Confirmation of the passive exhalation hypothesis for a terrestrial caecilian, *Dermophis mexicanus*. *Copeia* **1999,** 206–209 (1999).

369. Anderson, M. T. & Mathis, A. Diets of two sympatric Neotropical salamanders, *Bolitoglossa mexicana* and *B. rufescens*, with notes on reproduction for *B. rufescens*. *J. Herpetol.* **33,** 601–607 (1999).

370. McCranie, J. R. & Wilson, L. D. Taxonomic changes associated with the names Hyla spinipollex Schmidt and Ptychohyla merazi Wilson and McCranie (Anura: Hylidae). *Southwest. Nat.* **38,** 100–104 (1993).

371. Barrio-Amorós, C. L., Guayasamin, J. M. & Hedges, S. B. A new minute Andean *Pristimantis* (Anura: Strabomantidae) from Venezuela. *Phyllomedusa* **11,** 83–93 (2012).

372. Arroyo, S. B., Serrano-Cardozo, V. H. & Ramírez-Pinilla, M. P. Diet, microhabitat and time of activity in a *Pristimantis* (Anura, Strabomantidae) assemblage. *Phyllomedusa* **7,** 109–119 (2008).

373. Savage, J. M. & Myers, C. Frogs of the *Eleutherodactylus biporcatus* group (Leptodactylidae) of Central America and northern South America, including rediscovered, resurrected, and new taxa. *Am. Museum Novit.* **3357,** 1–48 (2002).

374. Simões, P. I. Diversificação do complexo *Allobates femoralis* (Anura, Dendrobatidae) em florestas da Amazônia brasileira: desvendando padrões atuais e históricos. (PhD Thesis, Instituto Nacional de Pesquisas da Amazônia, 2010).

375. Guayasamin, J. M., Ron, S. R., Cisneros-Heredia, D. F., Lamar, W. & McCracken, S. F. A new species of frog of the *Eleutherodactylus lacrimosus* assemblage (Leptodactylidae) from the western Amazon Basin, with comments on the utility of canopy surveys in lowland rainforest. *Herpetologica* **62,** 191–202 (2006).

376. Jared, C., Antoniazzi, M. M., Verdade, V. K. & Toledo, L. F. The Amazonian toad *Rhaebo guttatus* is able to voluntarily squirt poison from the paratoid macroglands. *Amphibia-Reptilia* **32,** 546–549 (2011).

377. Wollenberg, K. C., Veith, M., Noonan, B. P. & Lötters, S. Polymorphism versus species richness—systematics of large *Dendrobates* from the eastern Guiana Shield (Amphibia: Dendrobatidae). *Copeia* **2006,** 623–629 (2006).

378. Shepard, D. B. & Caldwell, J. P. From foam to free-living: ecology of larval *Leptodactylus labyrinthicus*. *Copeia* **2005,** 803–811 (2005).

379. Heyer, W. R., García-Lopez, J. M. & Cardoso, A. J. Advertisement call variation in the *Leptodactylus mystaceus* species complex (Amphibia: Leptodactylidae) with a description of a new sibling species. *Amphibia-Reptilia* **17,** 7–31 (1996).

380. Zimmermann, B. L. A comparison of structural features of calls of open and forest habitat frog species in the central Amazon. *Herpetologica* **39,** 235–246 (1983).

381. Bernarde, P. S. & Kokubum, M. N. D. C. Seasonality, age structure and reproduction of *Leptodactylus (Lithodytes) lineatus* (Anura, Leptodactylidae) in Rondônia state, southwestern Amazon, Brazil. *Iheringia Série Zool.* **99,** 368–372 (2009).

382. Campbell, J. A. & Clarke, B. T. A review of frogs of the genus *Otophryne* (Microhylidae) with the description of a new species. *Herpetologica* **54,** 301–317 (1998).

383. Kan, F. W. Population dynamics, diet and morphological variation of the Hong Kong newt (*Paramesotriton hongkongensis*). (MPhil Thesis, The University of Hong Kong, 2010).

384. Stuart, B. L., Chuaynkern, Y., Chan-ard, T. & Inger, R. F. Three new species of frogs and a new tadpole from eastern Thailand. *Fieldiana Zool. New Ser.* **111,** 1–19 (2006).

385. Ao, J. M., Bordoloi, S. & Ohler, A. Amphibian fauna of Nagaland with nineteen new records from the state including five new records for India. *Zoos' Print J.* **18,** 1117–1125 (2003).

386. Ohler, A. *et al.* Sorting out *Lalos*: description of new species and additional taxonomic data on megophryid frogs from northern Indochina (genus *Leptolalax*, Megophryidae, Anura). *Zootaxa* **3147,** 1–83 (2011).

387. Meiri, S. Evolution and ecology of lizard body sizes. *Glob. Ecol. Biogeogr.* **17,** 724–734 (2008).

388. Itescu, Y., Karraker, N. E., Raia, P., Pritchard, P. C. H. & Meiri, S. Is the island rule general? Turtles disagree. *Glob. Ecol. Biogeogr.* **23,** 689–700 (2014).

389. Meiri, S. Length-weight allometries in lizards. *J. Zool.* **281,** 218–226 (2010).

390. Feldman, A. & Meiri, S. Length-mass allometry in snakes. *Biol. J. Linn. Soc.* **108,** 161–172 (2013).

391. Edgar, M. What can we learn from body length? A study in Coleoptera. (MRes Thesis, Imperial College London, 2014).

392. Gilbert, F., Rotheray, G. E., Zafar, R. & Emerson, P. in *Phylogenetics Ecol.* 324–343 (Academic Press, 1994).

393. ESRI. ArcGIS Desktop: Release 10. (2011).

394. R Core Team. *R: A Language and Environment for Statistical Computing*. (R Foundation for Statistical Computing, 2013). http://www.r-project.org

395. Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A. & Smith, G. M. *Mixed Effects Models and Extensions in Ecology with R.* (Springer, 2009).

396. Rigby, R. A., Stasinopoulos, D. M. & Akantziliotou, C. A framework for modelling overdispersed count data, including the Poisson-shifted generalized inverse Gaussian distribution. *Comput. Stat. Data Anal.* **53,** 381–393 (2008).

397. Bivand, R. spdep: Spatial dependence: weighting schemes, statistics and models. R Package Version 0.5-68. (2013). http://cran.r-project.org/web/packages/spdep

398. Møller, A. P. & Jennions, M. D. Testing and adjusting for publication bias. *Trends Ecol. Evol.* **16,** 580–586 (2001).

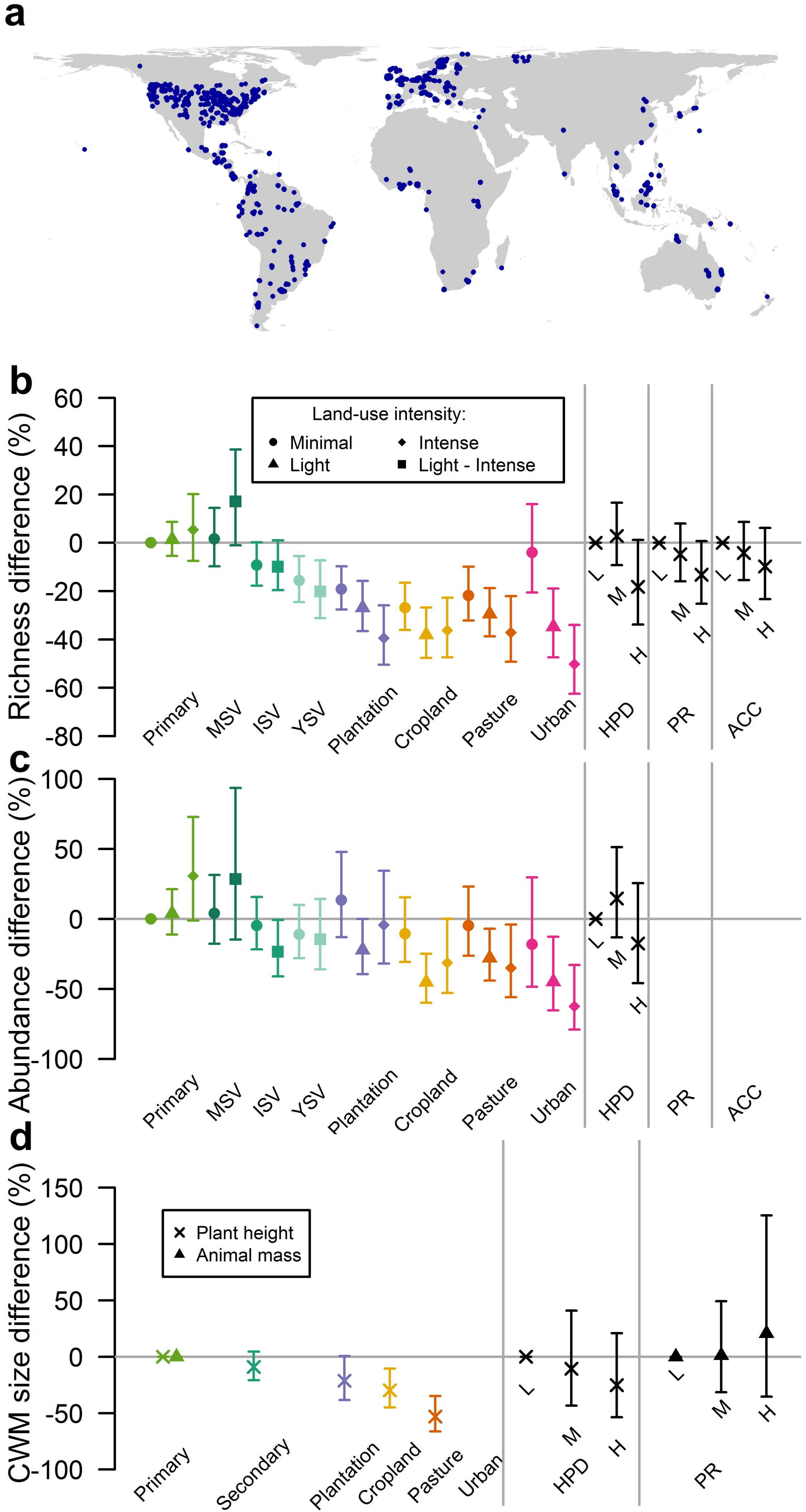
399. Van Vuuren, D. P. *et al.* The representative concentration pathways: an overview. *Clim. Change* **109,** 5–31 (2011).

400. United Nations Population Division. World Population Prospects: The 2010 Revision Population Database. *World Popul. Prospect. 2010 Revis. Popul. Database* (2011). http://www.un.org/esa/population/

401. Van Asselen, S. & Verburg, P. H. Land cover change or land-use intensification: simulating land system change with a global-scale land change model. *Glob. Chang. Biol.* **19,** 3648–67 (2013).

402. Hijmans, R. J. raster: Geographic data analysis and modeling. (2014). http://cran.r-project.org/package=raster

403. Olson, D. M. et al. Terrestrial ecoregions of the world: a new map of life on Earth. *Bioscience* **51**, 933–938 (2001).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Land-use intensity:
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YSV
Plantation
Cropland
Pasture
Urban
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Abundance difference (%)
100
50
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Primary
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YSV
Plantation
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Pasture
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CWM size difference (%)
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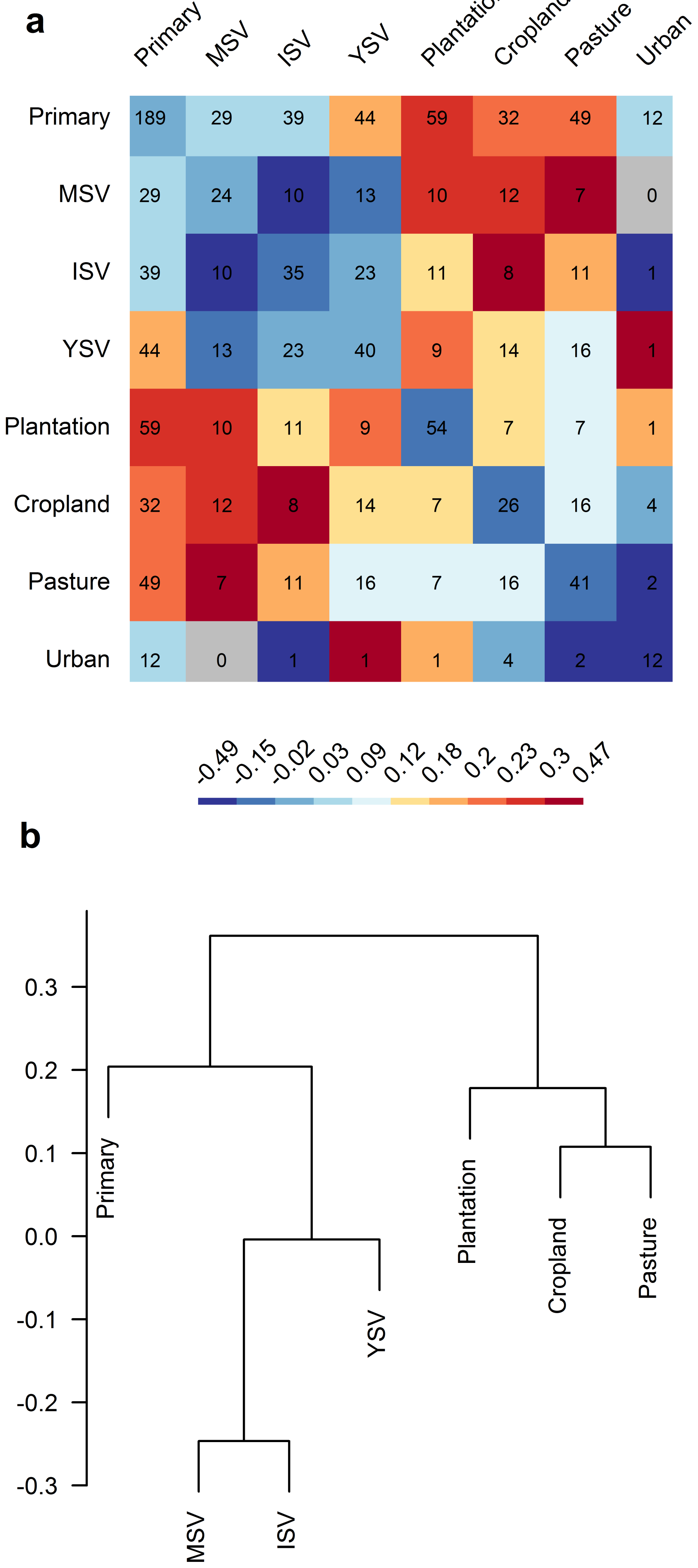

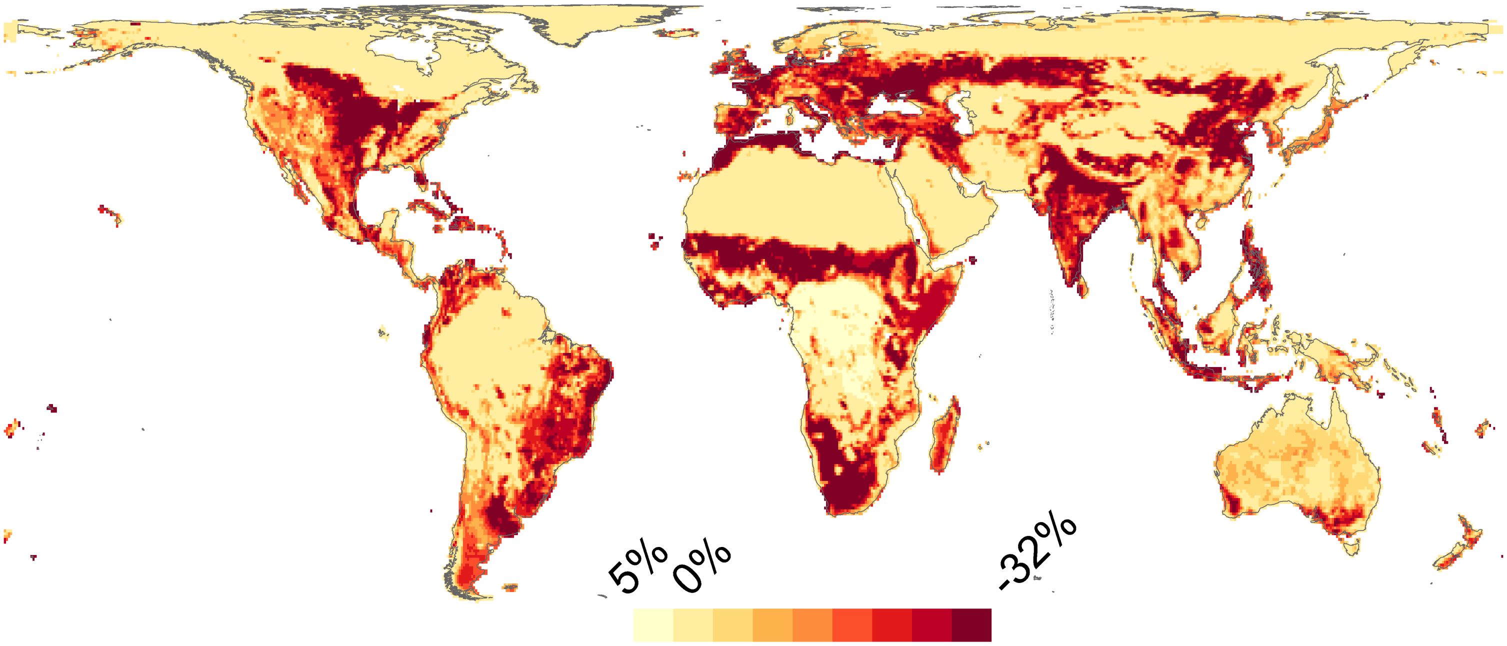
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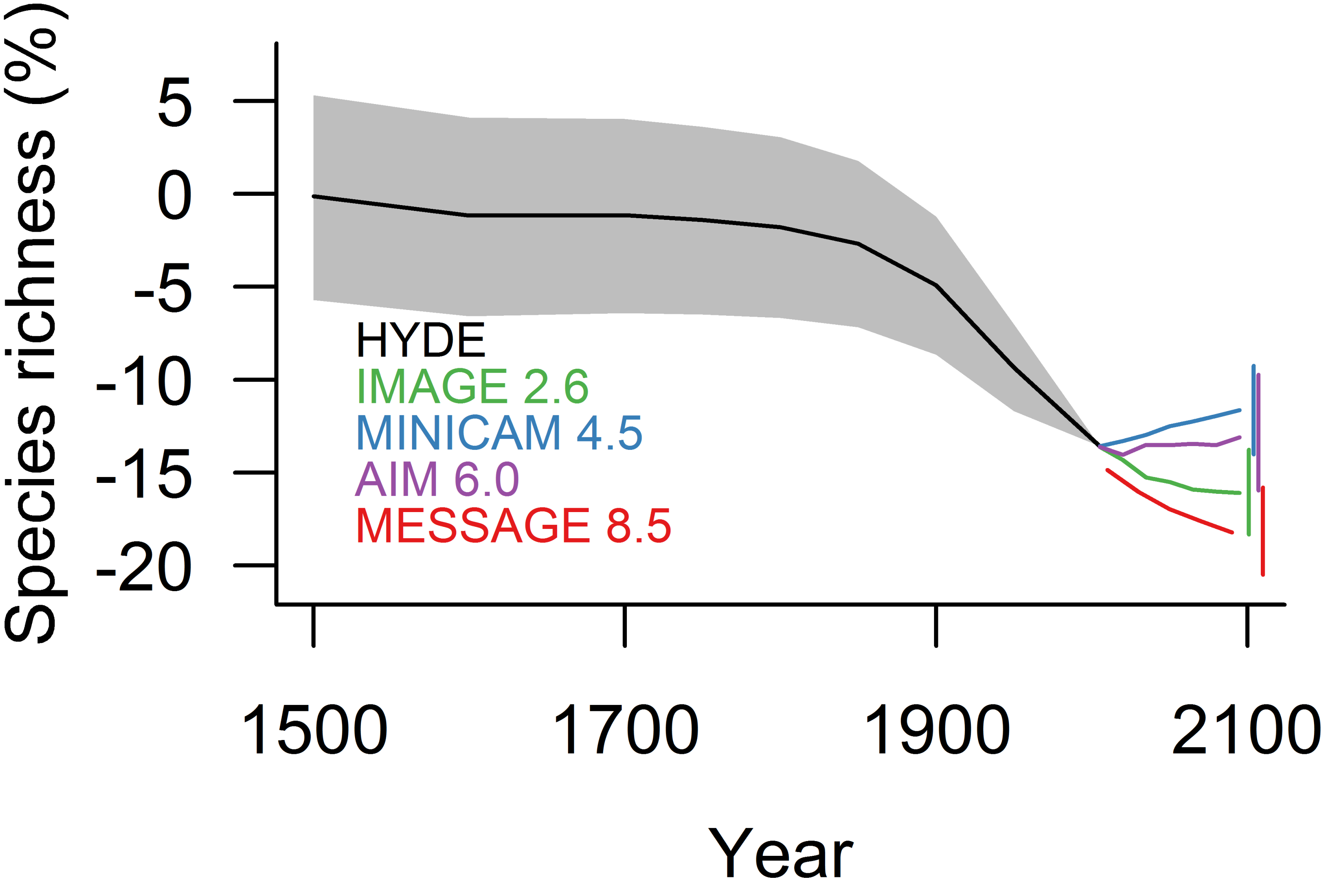
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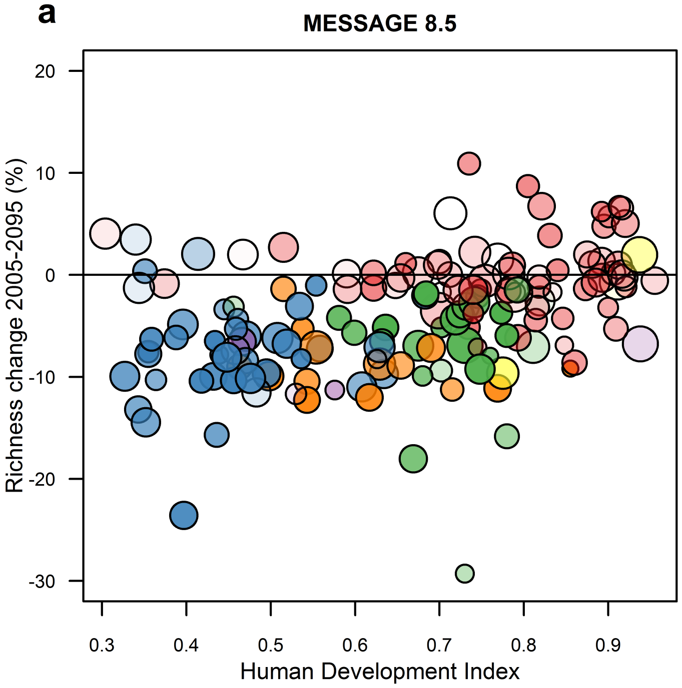

MESSAGE 8.5

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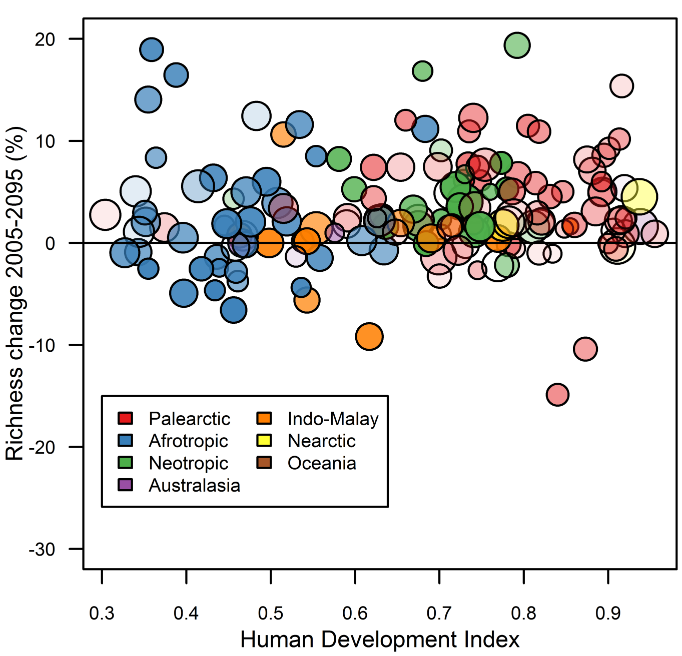

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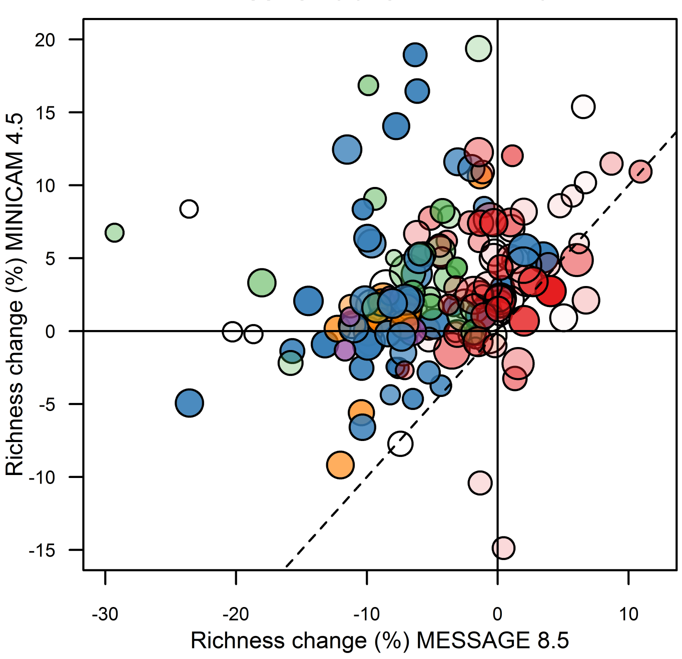

MESSAGE 8.5 vs. MINICAM 4.5