Increasing impacts of land-use on biodiversity and carbon-sequestration driven by population growth, consumption and trade

Alexandra Marques[1,2,3], Inês S. Martins[2,3], Thomas Kastner[4,5], Christoph Plutzar[5,6], Michaela C. Theurl[5], Nina Eisenmenger[5], Mark A.J. Huijbregts[7], Richard Wood[8], Konstantin Stadler[8], Martin Bruckner[9], Joana Canelas[2,3,10], Jelle Hilbers[7], Arnold Tukker[1,11], Karlheinz Erb[5], Henrique M. Pereira[2,3,12]

[1] Institute of Environmental Sciences (CML), Leiden University, P.O. Box 9518, 2300 RA/Einsteinweg 2, 2333 CC Leiden, The Netherlands

[2] German Centre for Integrative Biodiversity Research (iDiv), Halle-Jena-Leipzig, Deutscher Platz 5e, 04103 Leipzig, Germany

[3] Institute of Biology, Martin Luther University Halle-Wittenberg, Am Kirchtor 1, 06108 Halle (Saale), Germany

[4] Senckenberg Biodiversity and Climate Research Centre (SBiK-F), Senckenberganlage 25, 60325 Frankfurt am Main, Germany

[5] Institute of Social Ecology (SEC), University of Natural Resources and Life Sciences, Vienna, Schottenfeldgasse 29, A-1070 Vienna, Austria

[6] Division of Conservation Biology, Vegetation Ecology and Landscape Ecology, University of Vienna, Rennweg 14, 1030 Vienna, Austria

[7] Institute for Water and Wetland Research, Department of Environmental Science, Radboud University Nijmegen, Heyendaalse 135, 6500 GL Nijmegen, The Netherlands

[8] Industrial Ecology Programme, Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), Trondheim, Norway.

[9] Institute for Ecological Economics, Vienna University of Business and Economics, Welthandelsplatz 1/D5, 1020 Vienna, Austria

[10] Durrell Institute of Conservation and Ecology (DICE), School of Anthropology and Conservation, University of Kent, Canterbury CT2 7NZ, United Kingdom

[11] Netherlands Organisation for Applied Scientific Research TNO, Van Mourik Broekmanweg 6, 2628XE Delft, The Netherlands

[12] Cátedra Infraestruturas de Portugal-Biodiversidade, CIBIO/InBIO, Universidade do Porto, Campus Agrário de Vairão, 4485-661 Vairão, Portugal

Biodiversity and ecosystem services losses driven by land use are expected to intensify as a growing and more affluent global population requires more agricultural and forestry products. In addition, teleconnections in the global economy lead to increasing remote environmental responsibility[1,2]. Here we provide an assessment of the impacts of the economy on bird diversity and carbon sequestration, and their dynamics in the last decade, by combining global biophysical and economic models[3–6]. Between 2000 and 2011, despite gains in efficiency (i.e. reduction of land –use impacts per unit GDP), overall population and economic growth resulted in increasing total impacts on bird diversity and carbon sequestration globally and in most world regions. The exceptions were North America and Western Europe, where the 2007-2008 financial crisis led to an actual reduction of forestry and agriculture impacts on nature. Biodiversity losses occurred predominantly in Central and Southern America, Africa and Asia with international trade as an important and growing driver. In 2011, 33% of Central and Southern America and 26% of Africa's biodiversity impacts were driven by consumption in other world regions. In contrast, impacts on carbon sequestration were more homogenously distributed globally. Overall, cattle farming is the major driver of biodiversity loss, but oil seeds production showed the largest increases in biodiversity impacts during the analysed period. Forestry activities exerted the highest impact on

carbon sequestration, much higher than any agricultural activity including deforestation, and also showed the largest growth in carbon impacts. Our results suggest that to address the biodiversity crisis, governments should take an equitable approach recognizing remote responsibility. Environmental policies should be tailored for each world region, promoting a shift of economic development towards activities with low biodiversity impacts and increase of consumer awareness to promote sustainable consumption. In addition they should take into account the importance of the Sustainable Development Goals (SDGs) in addressing population growth[7].

MANUSCRIPT:

Agriculture and forestry activities are major drivers of biodiversity loss and ecosystem degradation[8–10]. Population growth and economic development will continue to increase the demand for agricultural and forestry products, and shift consumption patterns towards products with higher overall environmental burdens[8,11]. If unchecked, such strong demand-side drivers will cause higher pressures on biodiversity and ecosystems and put future well-being at risk[12]. Ensuring sustainable production and consumption patterns, by decoupling economic growth from natural resource use and environmental impacts, is fundamental to sustainable development[13]. However, teleconnections between world regions through international trade lead to an increasing disconnect between production and consumption, resulting in complex causal interrelationships, hampering straightforward analyses and resulting in governance challenges[1,2,9,14–17]. In this study we systematically analyse the global impacts of agricultural and forestry activities on biodiversity and a key ecosystem service, the sequestration of atmospheric carbon in ecosystems, taking these complex production-consumption interlinkages into account. We quantify the magnitude and dynamics of these pressures from agriculture, forestry and the consumption of biomass products between 2000 and 2011 and analyse the role of underlying drivers such as population growth, economic development and technological progress.

Assessing the impacts of socioeconomic activities on biodiversity and ecosystem services is complex due to their multidimensional nature[18,19]; this work covers one dimension of biodiversity and one ecosystem service. To assess the biodiversity impacts we focus on bird species richness, the species group best characterized in terms of responses to land-use activities[9]. We estimated, for each year, impending bird extinctions (i.e. number of species that would become extinct if land-use activities would be maintained in the long run) based on the number of endemic bird species in each biogeographical region (Methods, Supplementary Methods 1 and Supplementary Tables 1-2) and the amount and type of land being used for agriculture and forestry activities in each country or region (Methods and Extended Data Fig. 1-2). To assess the impacts on ecosystem services, we focused on net carbon sequestration, a key ecosystem service for climate change mitigation[20]. We estimated the biomass carbon sequestration lost each year, by calculating the potential additional carbon that would be sequestered if current land use ceased and natural vegetation were allowed to regrow (Supplementary Tables 3-4). In order to quantify the consumption drivers we linked the two impact indicators to a multi-regional input-output (MRIO) model based on EXIOBASE 3, a new time series of MRIO tables (Methods)[6].

Globally, between 2000 and 2011 we found increasing impacts of agriculture and forestry on biodiversity and ecosystem services; the number of bird species with impending extinction due to land-use activities increased 3 to 7% (from 69 to 74 in our conservative estimate, and from 118 to 121 in our non-conservative estimate, Supplementary Tables 1-2 and 6-7), and the amount of carbon sequestration lost increased 6% (from 3.2GtC to 3.4GtC/year, Supplementary Tables 3-4). As a comparison, 140 bird species are estimated to have been lost since the beginning of the 16th century from all drivers combined[21], and in the period 2002 – 2010, global carbon emissions were estimated at 8 ± 2 GtC/year (30 ± 8 $GtCO_2$/year)[22].

Our estimates show that cattle farming had the highest impact on biodiversity, contributing to approximately 28% of total impending extinctions in 2011, mostly in Central and South America and in Africa (Fig. 1a). The production of oil seeds (including soy beans) was the activity with the highest contribution to the increase in impacts on biodiversity from 2000 to 2011 (Fig. 1b). The expansion of oil seeds production typically occurs at the expense of tropical forests[23] rich in biodiversity. Forestry activities, i.e. the use of forests for timber and woodfuel extraction, had the highest impact on carbon sequestration, contributing approximately 30% of the total carbon sequestration lost (Fig. 1a), and contributed most to the increasing losses from 2000 to 2011, albeit a strong reduction of forestry impacts occurred in North America (Fig. 1b).

Increasing impacts have occurred despite improvement in land-use economic efficiency, i.e. reduction of biodiversity or carbon sequestration impacts per unit GDP (Fig. 2a-b). This happened because combined economic and population growth exceeded these efficiency gains both for biodiversity and carbon sequestration (Fig. 2a-b). We found consistent improvements in land-use economic efficiency in all world regions (Fig. 2c-d and Extended Data Fig. 3-4); in Africa, Asia and Pacific, Central and South America and Eastern Europe these were not sufficient to enable a reduction of the impacts caused by increased production. The overall decrease of the production impacts in Western Europe, Middle East and North America could indicate a decoupling of biodiversity and carbon sequestration impacts from economic growth. However, analysing decoupling trends only by assessing impacts from production activities taking place within a region might be misleading; a region may effectively import the environmental impacts from another region ("displacement effects")[24]. Therefore, we used a MRIO model to assess the impacts from consumption activities.

The comparison between per capita impacts from a production and consumption perspective for the different world regions shows that the consumption patterns of an average citizen in North America, Western Europe, Eastern Europe and Middle East is driving biodiversity impacts elsewhere, i.e. consumption impacts are up to an order of magnitude greater than the production impacts for those regions, (Fig. 3a), and the same happens for carbon sequestration except for Eastern Europe (Fig. 3b). Interestingly, between 2000 and 2011, per capita consumption impacts decreased in North America, Western Europe, Africa and Central and South America (Fig. 3a-b). In contrast, in Eastern Europe, Asia and Pacific and Middle East consumption impacts per capita increased (Fig. 3a-b), reflecting the recent rapid economic expansion of these regions.

Our land-use economic efficiency analysis from both a production and consumption perspective shows that decoupling between economic growth and impacts occurs in Western Europe and North America, but not in the Middle East (Extended Data Fig. 3-4). While the decoupling in production impacts is expected, due to decreases in land use in both regions during the period analysed (Supplementary Table 5), the decoupling in per capita consumption impacts is surprising and requires a reduction of consumption and/or an increase of the efficiency in the regions exporting to Western Europe and North America. In Western Europe, the consumption impacts on biodiversity and carbon sequestration decreased between 2007 and 2009 and in North America between 2006 and 2009. After 2009 there is again an increase in impacts for biodiversity, although by 2011 they were still below their 2001 levels. These results reflect the financial crisis and consequent decrease in consumption that occurred in these regions. The decreases of the biodiversity impacts associated with agricultural activities are mainly due to decreases of food consumption in hotels and restaurants and a decrease in clothing purchases by consumers, both in Western Europe and North America (Extended Data Fig. 5a-6a). These sectors are amongst those whose consumption was most affected during the financial crisis[25]. The decreases of the biodiversity and carbon sequestration impacts associated with forestry activities are mainly due to decreases in the manufacturing, construction and products of forestry sectors (Extended Data Fig.

5b-6b). Such findings reflect the reduction of the activity of the construction sector in both regions as a direct consequence of the financial crisis[26,27].

In any case, consumption based on internationally traded goods was driving 25% and 21% of the global impacts on biodiversity and carbon sequestration in 2011, representing a 3% and 1%, increase in relation to 2000, respectively (Fig. 4 and Extended Data Table 1-2). In 2000, Western Europe and North America were responsible for 69% and 58%, of the biodiversity and carbon sequestration impacts transferred through international trade; in 2011 these shares were reduced to 48% in the case of biodiversity impacts and 41% in the case of the carbon sequestration impacts (Fig. 4). In contrast the shares of other regions were increasing fast: for example, Asia and Pacific drove 13% in 2000 and 23% in 2011 of the biodiversity impacts embodied in international trade; and 20% in 2000 and 29% in 2011 of the carbon sequestration impacts embodied in international trade (Fig. 4 and Extended Data Table 1-2).

A complex analysis as the one presented here has several associated uncertainties, some of which we discuss in the Methods section, particularly those related with the forest areas under active management and the affinity parameter values of the countryside species-area relationship. In addition, it is particularly important to highlight that our analysis does not fully account for the effects of agriculture intensification (e.g., the response of biodiversity to different intensification levels of farmland was not discriminated in our calculations). Therefore, our estimates of impending extinctions due to land-use activities can be considered a lower bound for the likely range of values. As some of the recent trends in land-use change have been on intensifying levels of production (i.e. yields per ha of farmland use) we may also overestimate the gains in land-use impact economic efficiency of the last decade[28,29]. In addition, the decomposition of the impacts into the product of population growth component, economic growth, and efficiency change has been criticized for not considering other driving forces and for ignoring more complex interactions between these three components[30]. Nevertheless, we believe that our main results are robust to these uncertainties.

Decoupling economic development and population growth from environmental impacts and natural resource use, e.g. via technological progress, is often seen as the solution to the current sustainability challenges[13,31]. Our analysis highlights several intricacies related to such a perspective. In developed regions, a relative decoupling is observed, however it occurred mostly due to the financial crisis. In developed regions more than 90% of the biodiversity impacts from consumption as well as 40% of the carbon sequestration impacts from consumption, on average between 2000 and 2011, were outsourced (Extended Data Table 1-2). This is of particular concern in terms of global equity. The upcoming discussion of the parties to the Convention on Biological Diversity on the post-2020 biodiversity strategy should consider remote responsibility in an equitable way. Policies need to be tailored for each region and biodiversity and ecosystem services need to be mainstreamed into specific sectors. For developing regions, continuous population growth and rapid economic development outweigh any efficiency increase. In these regions biodiversity issues might co-benefit from the progress towards other SDG goals which might attenuate population growth[7]. For developed regions and emerging economies, policies need to address the increasing teleconnection through designing policies based on consumption-based accounting to avoid any biodiversity and ecosystem services impact leakage. Our work supports recent calls for changes in production and consumption patterns[32,33], and it shows the importance of taking into account time trends as well as all economic sectors' processes to properly identify the drivers of increasing impacts on biodiversity and ecosystem services.

References

1. Lenzen, M. *et al.* International trade drives biodiversity threats in developing nations. *Nature* **486,** 109–112 (2012).
2. Wilting, H. C., Schipper, A. M., Bakkenes, M., Meijer, J. R. & Huijbregts, M. A. J. Quantifying Biodiversity Losses Due to Human Consumption: A Global-Scale Footprint Analysis. *Environ. Sci. Technol.* **51,** 3298–3306 (2017).
3. Hudson, L. N. *et al.* The database of the PREDICTS (Projecting Responses of Ecological Diversity In Changing Terrestrial Systems) project. *Ecol. Evol.* **7,** 145–188 (2017).
4. Chaudhary, A., Verones, F., de Baan, L. & Hellweg, S. Quantifying Land Use Impacts on Biodiversity: Combining Species–Area Models and Vulnerability Indicators. *Environ. Sci. Technol.* **49,** 9987–9995 (2015).
5. Erb, K. H. *et al.* Biomass turnover time in terrestrial ecosystems halved by land use. *Nat. Geosci.* **9,** 674–678 (2016).
6. Stadler, K. *et al.* EXIOBASE 3: Developing a Time Series of Detailed Environmentally Extended Multi-Regional Input-Output Tables. *J. Ind. Ecol.* n/a-n/a (2018). doi:10.1111/jiec.12715
7. Abel, G. J., Barakat, B., Kc, S. & Lutz, W. Meeting the Sustainable Development Goals leads to lower world population growth. *Proc. Natl. Acad. Sci.* **113,** 14294–14299 (2016).
8. Venter, O. *et al.* Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. *Nat. Commun.* **7,** 12558 (2016).
9. Newbold, T. *et al.* Global effects of land use on local terrestrial biodiversity. *Nature* **520,** 45–50 (2015).

10. MA. *Millenium Ecosystem Assesment - Ecosystems and human well-being*. (Island Press, 2005).

11. West, P. C. *et al.* Leverage points for improving global food security and the environment. *Science* **345,** 325–328 (2014).

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

13. UN. *Transforming our World: the 2030 Agenda for Sustainable Development.* (United Nations, 2015).

14. Verones, F., Moran, D., Stadler, K., Kanemoto, K. & Wood, R. Resource footprints and their ecosystem consequences. *Sci. Rep.* **7,** 40743 (2017).

15. Laurance, W. F., Sayer, J. & Cassman, K. G. Agricultural expansion and its impacts on tropical nature. *Trends Ecol. Evol.* **29,** 107–116 (2014).

16. 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).

17. Erb, K.-H., Krausmann, F., Lucht, W. & Haberl, H. Embodied HANPP: Mapping the spatial disconnect between global biomass production and consumption. *Ecol. Econ.* **69,** 328–334 (2009).

18. Pereira, H. M. *et al.* Essential Biodiversity Variables. *Science* **339,** 277–278 (2013).

19. Reyers, B., Stafford-Smith, M., Erb, K.-H., Scholes, R. J. & Selomane, O. Essential Variables help to focus Sustainable Development Goals monitoring. *Curr. Opin. Environ. Sustain.* **26–27,** 97–105 (2017).

20. Pan, Y. *et al.* A Large and Persistent Carbon Sink in the World’s Forests. *Science* **333,** 988–993 (2011).

21. Ceballos, G. *et al.* Accelerated modern human–induced species losses: Entering the sixth mass extinction. *Sci. Adv.* **1,** e1400253 (2015).
22. Blanco, G. *et al.* Drivers, Trends and Mitigation. in *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds. Edenhofer, O. et al.) (Cambridge University Press, 2014).
23. Vijay, V., Pimm, S. L., Jenkins, C. N. & Smith, S. J. The Impacts of Oil Palm on Recent Deforestation and Biodiversity Loss. *PLOS ONE* **11,** e0159668 (2016).
24. Ward, J. D. *et al.* Is Decoupling GDP Growth from Environmental Impact Possible? *PLOS ONE* **11,** e0164733 (2016).
25. Eurostat. Archive:Household consumption expenditure - national accounts - Statistics Explained. (2013). Available at: http://ec.europa.eu/eurostat/statistics-explained/index.php/Archive:Household_consumption_expenditure_-_national_accounts. (Accessed: 22nd March 2018)
26. Eurostat. *EU-27 construction activity falls by 16 % from its precrisis high by the second quarter of 2011*. (2011).
27. USDA. *U.S. Forest Resource Facts and Historical Trends*. (United States Department of Agriculture, 2014).
28. Seppelt, R., Manceur, A., Liu, J., Fenichel, E. & Klotz, S. Synchronized peak-rate years of global resources use. *Ecol. Soc.* **19,** (2014).
29. Gerstner, K., Dormann, C. F., Stein, A., Manceur, A. M. & Seppelt, R. EDITOR’S CHOICE: REVIEW: Effects of land use on plant diversity – A global meta-analysis. *J. Appl. Ecol.* **51,** 1690–1700 (2014).
30. Alcott, B. Impact caps: why population, affluence and technology strategies should be abandoned. *J. Clean. Prod.* **18,** 552–560 (2010).

31. UNEP. *Decoupling natural resource use and environmental impacts from economic growth, A Report of the Working Group on Decoupling to the International Resource Panel.* (UNEP, 2011).

32. Godfray, H. C. J. *et al.* Meat consumption, health, and the environment. *Science* **361,** eaam5324 (2018).

33. Poore, J. & Nemecek, T. Reducing food's environmental impacts through producers and consumers. *Science* **360,** 987–992 (2018).

Supplementary Information is linked to the online version of the paper.

Acknowledgements Authors would like to thank the financial support provided by EU-FP7 project DESIRE (FP7-ENV-2012-308552). K.H.E and T.K gratefully acknowledge fundings from the Austrian Science Fund Project GELUC (P29130) and ERC-2010- 263522 LUISE. TK acknowledges support from the Swedish Research Council Formas (grant number 231-2014-1181). M.A.J.H was supported by the ERC project (62002139 ERC – CoG SIZE 647224).

Author Contributions: All authors provided input into the final manuscript. A.M., I.S.M, M.B., M.A.J.H, K.H.E, H.M.P designed the study. A.M., I.S.M, C.P, M.T, N.E., K.H.E., R.W., K.S. contributed data. A.M., I.S.M. and T.K performed the analysis.

A.M. and H.M.P wrote the paper with help from all the authors.

Author Information: Reprints and permissions information is available at www.nature.com/reprints. The authors declare no competing financial interests. Readers are welcome to comment on the online version of the paper. Correspondence and requests for materials should be addressed to A.M. (alexandra.penedo@gmail.com).

Figures

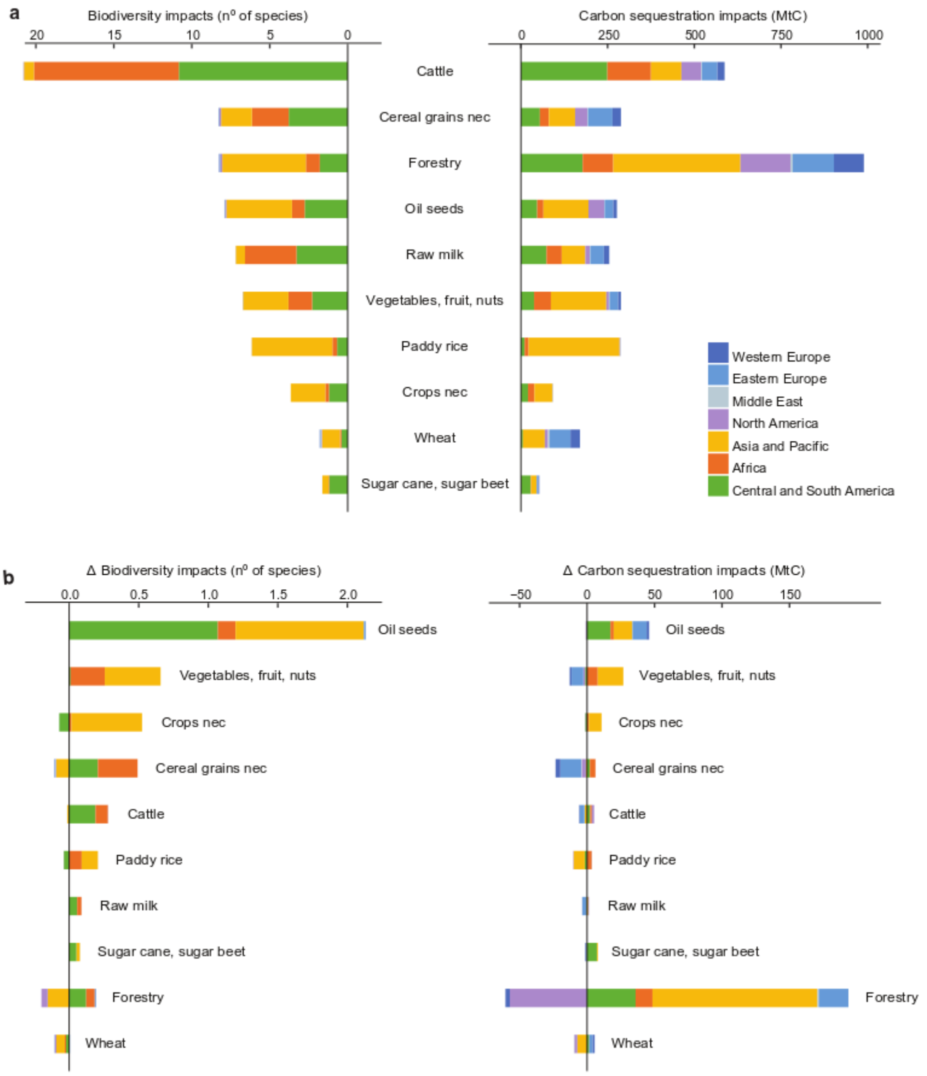


Figure 1 – Production impacts on biodiversity and carbon sequestration per economic sectors. **a**, Impacts in absolute terms for the year 2011; **b**, the difference between the impacts in 2011 and 2000. Negative values imply a decrease of their impacts by 2011. The left side are represents impending global bird extinctions (number of species) and on the right side carbon sequestration lost (MtC per year). Results are sorted by decreasing biodiversity impacts from production activities. The impacts associated with plant-based fibers, pigs, poultry and meat animals nec account for less than 1% each and are not represented. Nec stands for not elsewhere classified.

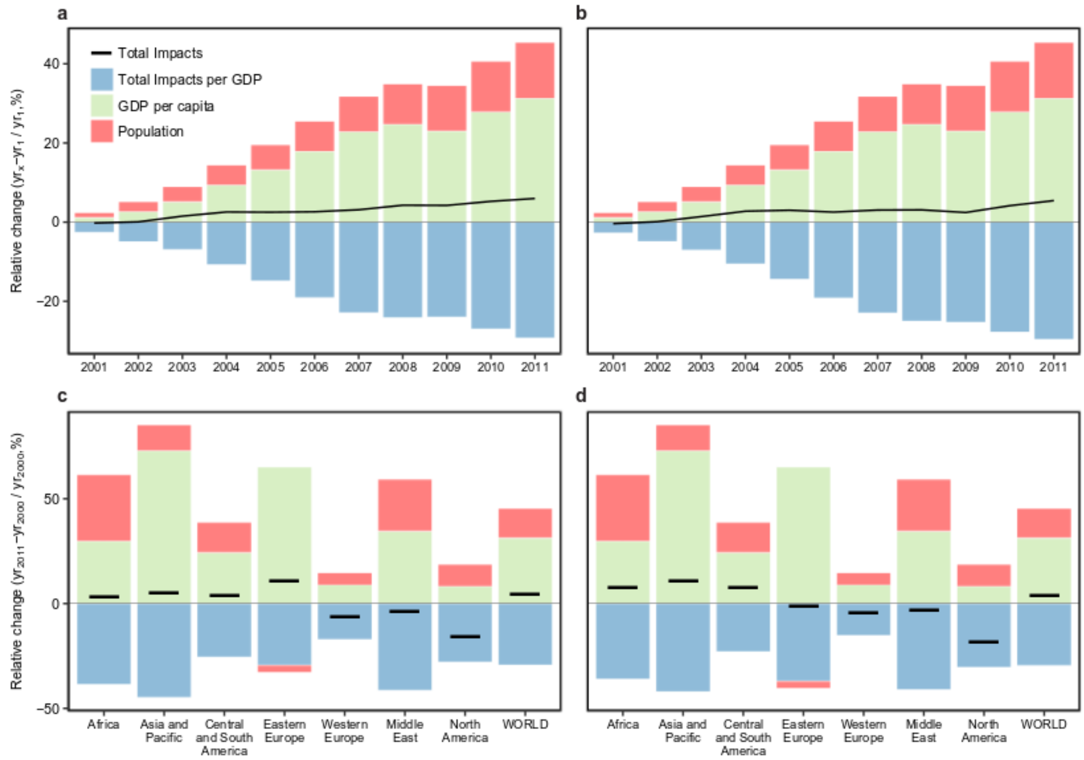


Figure 2 – Decomposition of changes in impacts of agriculture and forestry on biodiversity and carbon sequestration into the contribution of the changes in population, GDP per capita and impact per GDP. Biodiversity impacts are measured in terms of impending global bird extinctions, and ecosystem services impacts in terms of carbon sequestration lost. Impacts can be decomposed as (Methods): Δ Impacts = Δ Population × Δ GDP per capita (i.e. affluence) × Δ Impacts per GDP (i.e. land-use efficiency). Annual changes in production impacts relative to 2000 (Δ) at the global level for biodiversity (**a**) and ecosystem services (**b**), overall changes between 2000-2011 for different world regions for biodiversity (**c**) and ecosystem services (**d**).

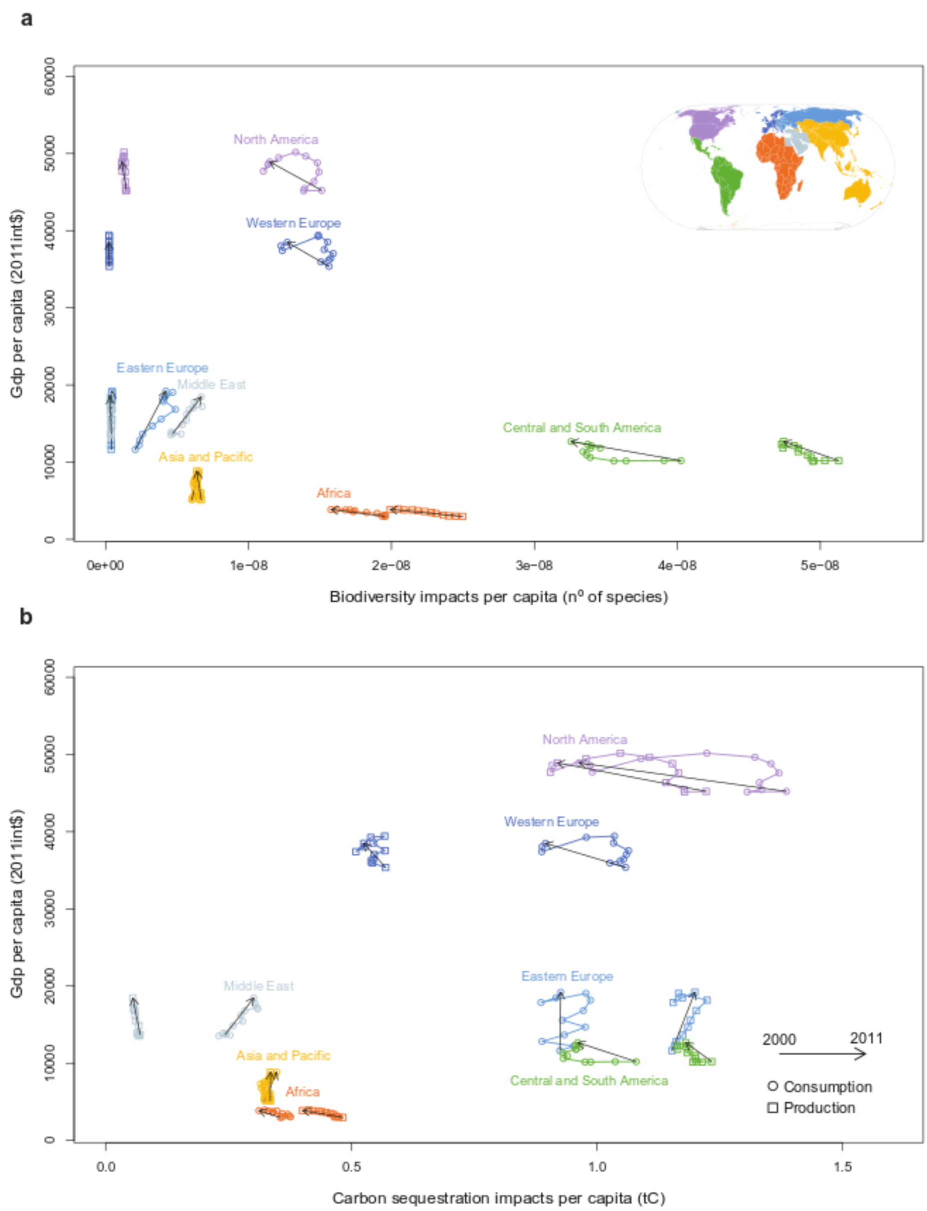


Figure 3- GDP per capita (in constant 2011 international$) and per capita impacts on biodiversity and carbon sequestration, per world region. Consumption and production impacts on biodiversity (**a**) as global impending bird extinctions (number of species per capita and year) and ecosystem services (**b**) as carbon sequestration lost (tC per capita and year).

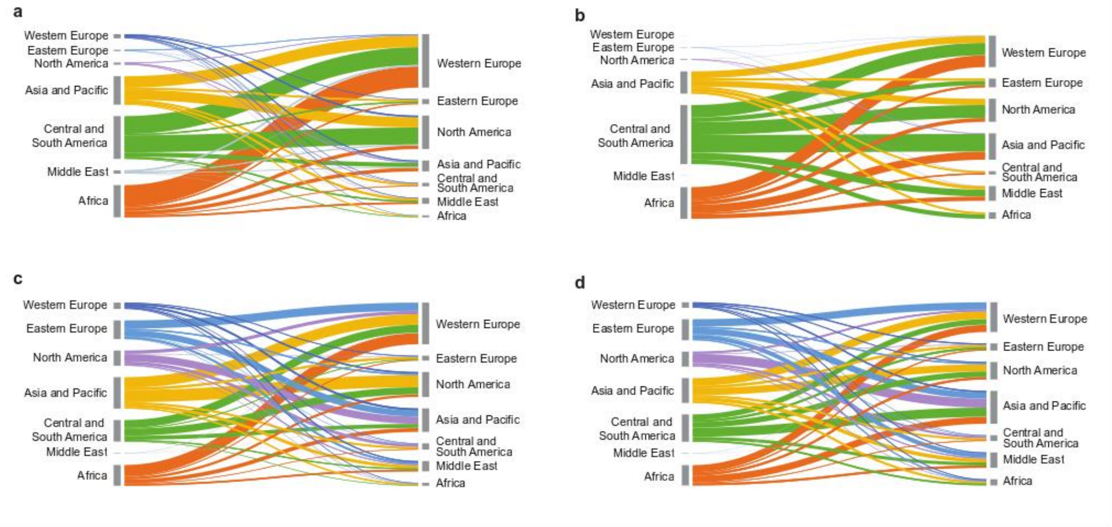


Figure 4 –Biodiversity (a,2000; b,2011) and carbon sequestration (c,2000; d,2011) impacts embodied in international trade. On the left is the region where the impacts occur and on the right is the region whose consumption is driving the impacts. The width of the flows represents the magnitude of the impacts. Exact values can be found in Extended Data Tables 1-2. Impacts arising from domestic production and consumption are not included in this figure. The visualized impacts represent 22%, 25%, 19% and 21% of the yearly global totals, respectively for biodiversity and carbon sequestration lost.

Methods

The starting point for the quantification of the drivers of biodiversity and ecosystem services loss was a spatially-explicit land-use dataset, with information on 14 categories of land-use activities which cover all the agricultural and forestry production reported in authoritative international databases (FAOSTAT). This enabled determining the impacts to biodiversity and ecosystem services per km^2 of land-use activity (the so-called characterization factors). The characterization factors together with a time series of land-use data for 49 countries/world regions was used to determine the total impacts on biodiversity and ecosystem services, for the period 2000-2011. We referred to these as the supply side drivers of biodiversity and ecosystem services loss; these are the impacts driven by the production activities. To determine the consumption patterns driving biodiversity and ecosystem services loss we coupled the impacts from production activities to a multi-regional input-output model. We used the IPAT identity to distinguish the influence of population growth (P), economic development (A) and technological progress (T) on the evolution of the drivers of biodiversity loss and ecosystem degradation. The results were aggregated into 7 world regions, using EXIOBASE's world regions and the United Nations regional groups[34]. In the following sections the methods are presented in detail.

Land-use spatially explicit dataset

A spatially explicit land-use dataset for the year 2000, matching the sectoral resolution (for land-use activities) of the EXIOBASE dataset (see below **Multi-regional input-output analysis** and Supplementary Methods 2), was developed to assess the biodiversity impacts as well as carbon sequestration foregone due to agriculture and forestry activities[6]. The starting point of the assessment was the construction of a consistent and comprehensive set of layers at the spatial resolution of 5 arc minutes. We followed a previously published approach[35] and used a series of recent datasets for the year 2000 (restricted to this year by the availability of comprehensive cropland maps which currently are only available for the year 2000) to create the individual layers. A cropland layer[36] was adjusted to reproduce newly published national statistics for cropland area for the year 2000 (based on the regular updates by FAO[37] and data on cropland distribution[36]). The cropland layer was split into nine sub-layers (corresponding to crop-categories in EXIOBASE) using the distribution of major crop groups[38]: (a) paddy rice, (b) wheat, (c) cereals, grains nec (not elsewhere classified) (d) vegetables, fruit and nuts, (e) oil seeds, (f) sugar cane, sugar beet (g) plant-based fibres, (h) crops nec such as herbs and spices and (i) fodder crops (Extended Data Fig. 1-2 and Supplementary Methods 2). Next, a recent global forest map was integrated into the dataset[39]. This dataset is based on the integration of recent high-resolution tree cover maps and a validation procedure through citizen science approaches, and applies a single definition of "forest" globally. Compared to FAO data this leads to a lower global forest cover estimate (32 Mkm^2 vs 42 Mkm^2). Individual input data and maps for the construction of the land-use dataset origin from different sources. The resulting inconsistencies have been solved the following way: in grid cells where the sum of all allocated layers (cropland, built-up and infrastructure, and the forest layer) exceeded 100%, the forest layer was capped so that all land-use types fill 100% of the grid cell. Information on intact forests[40] was used to identify unused forests. The layer of permanent pastures was derived from[36] and added to the grid, also here capping the pasture layer at 100% total land use coverage in each grid cell. The permanent pasture dataset is largely consistent with FAO statistics for permanent pastures, but uses national and subnational statistics and corrects the FAO data based on top-down considerations (e.g., on the maximum extent of grazing activities, or outlier correction based on statistical approaches) and plausibility checks, e.g. with remote sensing data[36]. In consequence, the total sum for permanent pastures is 27Mkm^2 (in contrast to 35Mkm^2 in FAO). By taking non-productive areas (aboveground NPP below 20gC m^{-2} yr^{-1}) into account[35], permanent pasture land was further reduced to 23km^2. This reduction occurs mainly in dryland areas of Australia and central Asia and assumes that permanent pastures at a very low productivity do not contribute to grazing. Fodder crops were split into five separate layers (raw milk, cattle meat, pig meat, poultry and other meat), and permanent pastures into three layers (raw milk, cattle meat, other meat)[41], matching the available livestock sectors in EXIOBASE (Extended Data Fig. 1-2). The remaining areas can be considered under extensive, sporadic use, mainly for temporary livestock grazing and wood fuel collection. However, no biodiversity or ecosystem service impacts were allocated to them due to large uncertainties about the dimension and nature of the impacts of land use on these lands.

Correction of forest areas for quantification of biodiversity impacts

The approach described above gives an estimate of all forest areas not considered wilderness. In many contexts it will, however overestimate the amount of forests actively managed for forestry. To account for this, we used an alternative approach to estimate the area of managed forests: we first estimated the forest area that would have to be cleared to produce the harvest volumes (section **Characterization factors for ecosystem services impacts** for details on how biomass harvest data were assessed), assuming clear-cut regimes. To convert the estimates of harvest volumes into areas we assumed that biomass stocks at the time of harvest equal the average national potential biomass stocks (i.e. the stock that would prevail without land use but under current climatic conditions; from refs.[5,42]). In order to determine an estimate of forest area actively managed, we multiply the amount of clear cut area by the estimates of typical rotation times[43,44] (Supplementary Methods Table 3). Following this procedure yearly correction coefficients for each country were determined (Supplementary Methods Table 4).

In general, this estimate should give areas smaller or similar to the area calculated via the spatially explicit land-use datasets. In a few cases (Supplementary Methods Table 4) the numbers were higher, owing to uncertainties in all the data involved. To arrive at a conservative estimate, we use the smaller number of the two approaches as the area of managed forests considered in the biodiversity impact assessment, with the affinity parameter of the countryside species area relationship set for intensive forestry use (see **Characterization factors for biodiversity impacts**). We have also computed the biodiversity impacts associated with the higher non-conservative estimates of forest area under active management, for these estimates the affinity parameter of the countryside species-area relationship was set as the average value between the affinities for intensive and extensive forest use. (Extended Data Table 3). The results are reported in Supplementary Tables 6-7.

Characterization factors for biodiversity impacts

In order to quantify potential global bird species extinctions due to different land-use activities, we started by computing characterization factors (CFs) for each land-use activity (number of birds potentially extinct per km^2 of area used by land-use activity), based on the land-use dataset described in the previous section. To compute the extinctions associated to each individual land-use activity we used the countryside species-area relationship (cSAR)[45,46]. Species-area relationship models have been classically used to assess species extinctions after habitat loss, however this approach has a number of limitations. One issue is assuming that the number of species is mainly determined by habitat area, and that the habitat is uniform and continuous[47,48]. Another issue, that we believe to be even more prevalent, is that the classic SAR only captures the species richness response to changes in native habitat area, overlooking the diversity of species responses to changes in habitat composition. The countryside species-area relationship[45] describes the use of both human-modified and natural habitats by different functional species groups. Consider a completely natural landscape where habitat conversion takes place and only a single functional group of species is present. Then, according to the cSAR, the proportion of species remaining $\left(\frac{S^1}{S^0}\right)$ after habitat conversion is[46]

$$\frac{S^1}{S^0} = \left(\frac{\sum_j^n h_j A_j^1}{h_1 A_1^0}\right)^z, \tag{1}$$

where *n* is the number of habitat types, h_j is the affinity of species to non-natural habitat *j* (hereafter called land-use activity *j*), h_1 is the affinity of species to the natural habitat, A_j is the area occupied by the different land-use activities *j*, A_1 the area of natural habitat before conversion takes place and *z* is a constant indicating the rate at which species richness increases with area. The superscript 0 indicates the natural state, and the superscript 1 indicates the modified state (i.e. after land-use change occurred). We used a value of $z = 0.20$, as it is an appropriate value for the spatial scales used in this work (biogeographical region)[49,50]. We assumed that species have maximum affinity for the natural habitat ($h_1 = 1$) For human-modified habitats we calculated affinities as[46]:

$$h_j = \left(1 - \sigma_j\right)^{1/z}, \tag{2}$$

where σ_j is the mean sensitivity of the species to each land-use activity j. Sensitivity values (σ) were retrieved from previously published global databases[4,51,52] of studies of biodiversity responses to human-modified landscapes (Supplementary Methods 5). From these databases, we selected studies that provided data on bird species richness on both natural habitat and at least one human-modified habitat (i.e. land-use activity), as σj is the difference between the plot scale species richness found in the modified habitat of type j and the species richness in the native habitat (i.e. the proportion of species disappearing at the plot-scale in modified habitats), which led to a total of 319 pairwise comparisons. The data was subset into four land use classes based on the description of the habitat given in the source dataset: managed forest (extensive and intensive use), cropland, permanent crops and pastures; and two major biomes, tropical and temperate (Supplementary Methods 5). From these σ_j values and h_j were computed (see Supplementary Methods 5 and Extended Data Table 3). The correspondence between the habitats types used for the computation of the h_j values and the categories in our land-use dataset can be found in Supplementary Methods 2.

Using ArcGIS version 10.2[53], we overlaid the land-use layers (see previous section for details on the spatially explicit land-use dataset), with a biogeographic region layer[54] to derive the current share of each of the fourteen land-use activities (13 agricultural types and forestry), A_j, per biogeographic region g, $A_{g,j}$. We used equation (1) to calculate the proportion of endemic species remaining after land-use change in each of the 19 biogeographical regions, with A_1^0 as the area of the biogeographic region g. Bird species' distribution maps[55] were used to derive the number of endemic species present in each of the biogeographic regions (S_g), 1295 endemic bird species were identify across all biogeographic regions (Supplementary Methods 1), which represents approximately 12% of the total number of bird species reported in ref.[55]. The total number of endemic species lost in each biogeographic region, ΔS_g, was calculated as:

$$\Delta S_g = \left(1 - \frac{S^1}{S^0}\right) \times \ S_g \ , \tag{3}$$

where S_g is the number of endemic species in a biogeographic region as determined through bird species distribution maps[55]. Then, the total number of species lost per land-use activity j in each biogeographic region g was computed as follows,

$$\Delta S_{g,j} = \frac{w_j A_{g,j}}{\sum_j^n w_j \, A_{g,j}} \times \ \Delta S_g \ , \tag{4}$$

where $w_j = (1 - h_j)$ is a weight that reflects the impacts of the different land-use activities and n the number of land-use activities considered. For each biogeographic region g, the number of species lost due to each land-use activity j in each country i was then determined by taking into account the area of each land-use activity in each country that crosses the biogeographic region, $A_{g,i,j}$:

$$\Delta S_{g,i,j} = \Delta S_{g,j} \times \frac{A_{g,i,j}}{A_{g,j}} \ . \tag{5}$$

If a country contained more than one biogeographic region, the impacts across several regions were summed:

$$\Delta S_{i,j} = \sum_{g=1}^{G_i} \Delta S_{g,i,j} \, , \tag{6}$$

where G_i is the number of different biogeographic regions in country i. The biodiversity characterization factors, CFs, were then determined by dividing the $\Delta S_{i,j}$ by the area of each land-use activity j in each country i:

$$CF_{i,j} = \frac{\Delta S_{i,j}}{A_{i,j}}\,. \tag{7}$$

The biodiversity CFs (bird species potentially lost per km^2 of land use) were multiplied by the land-use data time series (see **Multi-regional input-output analysis**) to obtain the impending birds extinctions in every year. All calculations were performed using Python[56].

Previous studies[4,57], applying the countryside species area relationship at the global level, determined that the parameter associated with the responses of species to habitat changes was the one contributing the most to the uncertainty of the characterization factors. This is mostly a result of the broad range of values reported for species response to habitat changes spanning from positive to negative (i.e. from a detrimental effect to a beneficial one) and a heterogeneous distribution of the data in terms of taxa and biogeographical regions covered. In this study we focused on the birds group, the one which is best covered in terms of number of studies assessing their response to land-use change[9]. Despite limiting the uncertainty of our results by covering just one species group, it is still important to mention that the range of the values and the unbalanced geographical distribution (Extended Data Fig. 7) (e.g., for temperate biogeographical regions there are 82 data points whereas for tropical there are 237 data points) are still important sources of uncertainty in the determination of the characterization factors. By using birds as a single functional group, we assume that all bird species respond equally to land use and habitat loss, also by considering broad geographic areas we ignore the effects of the particular characteristics of habitats[47].

Characterization factors for carbon sequestration impacts

Ecosystems store large amounts of carbon in living biomass providing a crucial climate regulation service. Globally, the largest amounts of biomass carbon are stored in forest systems[42]. Agricultural activities replace these natural ecosystems with agro-ecosystems (cropland and pasture) that provide higher amounts of biomass flows useful for society, but massively reduce vegetation carbon stocks. Forestry lowers biomass carbon stocks through wood harvests, even if practiced sustainably, as forestry operations optimize the annual wood increment, which leads to lower biomass carbon stocks compared to forests not under harvest regimes[42,58]. When agricultural and forestry practices cease, systems can regenerate towards a more natural state. We estimated the biomass carbon sequestration potential on land currently under use that would prevail in the absence of land use, the carbon sequestration potential lost. It is important to note that this potential is expressed as annual flow, but these flows cannot be expected continue infinite as biomass carbon stocks in ecosystem without land use will saturate at some point. Thus, the indicator reflects short-to-medium term conditions only. This assumption, however, allows to unambiguously link carbon stock impacts and current land-use activities, irrespective of the long legacy effects of past land uses on biomass carbon stocks[42,59,60], and thus avoids incorrect attributions.

For agricultural land use, we assign the effect of land conversion (i.e. clearing of forests to agricultural fields) to the agricultural sectors in EXIOBASE (Supplementary Methods 2). We based our calculations on the land-use maps described in the land-use dataset section (see **Land-use spatially explicit dataset)** and combine them with a map of the biomass carbon stocks in the potential natural vegetation[5] (i.e. the vegetation that would prevail without human land use). Due to large uncertainties relating to biomass carbon stocks of non-forest ecosystems we perform the assessment only for agricultural land on potentially forested areas. These sites were identified by combining three biome maps[61–63], and assuming potential forest cover where two of the three maps report a forest biome. Because of the omission of lands without potential forest cover, our estimate on the impact of agriculture on biomass carbon stocks should be considered conservative.

We assume that in absence of agricultural land use, vegetation would grow back to 75% of the potential natural carbon stock value within 50 years[59]. The calculations are performed on a global grid with a resolution of five arc minutes. The annual carbon sequestration lost (ΔC) in agricultural land-uses activities j, per grid cell m is calculated as:

$$\Delta C_{m,j} = \left(0.75 \times \frac{C_m^o}{50}\right) \times A_{m,j}, \tag{8}$$

where C_m^o is the potential biomass carbon stock per unit area in the grid cell m and $A_{m,j}$ is the area of agricultural land-use activity j in the grid cell m. In equation (8) we implicitly assume that the biomass stock of agricultural land is negligible compared with the potential carbon stock[42]. To link the indicator to the multi-regional input-output model an indicator per country i and land-use activity j was computed:

$$\Delta C_{i,j} = \sum_{m=1}^{M_i} \Delta C_{m,j}, \tag{9}$$

where $\Delta C_{i,j}$ represents the amount of carbon sequestration lost due to each land-use activity j in each country i, and M_i is the number of grid cells per country i.

For forestry a different approach was required to account for the effect of forest management on biomass carbon stocks. The difference between potential biomass carbon stocks and current biomass carbon stocks is not a good proxy for this effect, as this difference is largely influenced by land-use histories and not solely by present use[42]. To unambiguously account for the effect of forestry on biomass carbon socks, we focus on wood harvest, the main purpose of forestry activities. We assume that, at the national level, annual carbon sequestration lost due to forestry equals the biomass removed by wood harvest (industrial roundwood and fuelwood) activities in a given year[60]. For this we convert annual wood harvest quantities from ref.[37] into carbon, taking into account bark and other biomass destroyed in the harvest process, but not removed from the forests, correcting for the fact that part of this biomass was foliage and would not have contributed to long term carbon sequestration (factors from ref.[64]). Part of the harvested wood is stored in long lived products, representing a form of carbon sequestration. We account for this, by deducting amount of industrial roundwood that ends up in such products (about 20% of harvested industrial roundwood globally, based on ref.[65]). The national level data for annual carbon sequestration lost due to forestry, $\Delta C_{i,forestry}$, were aggregated where necessary to match EXIOBASE's regional resolution (Supplementary Methods 6) .This approach disregards ecosystem effects such as compensatory growth and thus only holds for a short term perspective, but gives an indication on how forestry practices currently lower the potential sink function of biomass in ecosystems[58,66,67].

The ecosystem services characterization factors, CFs, were then determined by dividing the $\Delta C_{i,j}$ by the area of each land-use activity *j* in each country *i*:

$$CF_{i,j} = \frac{\Delta C_{i,j}}{A_{i,j}} \,. \tag{10}$$

Similarly to the biodiversity CFs, the ecosystem services CFs (carbon sequestration lost per km^2 of land use) were multiplied by the land-use data time series (see **Multi-regional input-output analysis**) to obtain carbon sequestration lost in every year.

Multi-regional input-output analysis

Multi-regional input-output (MRIO) analysis has been increasingly used to identify the consumption drivers of environmental impacts. Environmental impacts analysed within a MRIO framework include emissions of pollutants, appropriation of natural resources and loss of biodiversity[1,68,69]. Environmentally-extended MRIO (EEMRIO) models are particularly suited to track the spatial disconnection between environmental pressures from production processes and the consumption drivers behind them as they cover the world economy and the international trade relations between different countries and sectors. In this work we followed the standard Leontief model to compute the biodiversity and ecosystem services impacts from consumption activities. The standard environmentally extended Leontief pull model is formulated as follows[70]:

$$\mathbf{E} = \mathbf{f}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{Y} \tag{11}$$

Where (for *i* countries and *m* economic sectors):

- **E** is the (1 x *i*) matrix of environmental impacts associated with final demand of each country.
- **f** is a (1 x *i.m*) direct intensity vector, which gives the environmental pressures (biodiversity and ecosystem services losses) associated with 1€ of production of the economic sectors. Since in this work we quantified the biodiversity and ecosystem services losses associated with land-use activities this vector will be a sparse vector only populated in the entries for land-use activities. The biodiversity and ecosystem services losses are calculated by multiplying the previously determined characterization factors (CFs) by the amount of land used in each year by a given land-use activity. The amount of annual land used was extracted from the MRIO database used (see below for more details).

- **A** is the (*i.m* x *i.m*) matrix of technical coefficients, which gives the amount of inputs that are required to produce 1€ of production.

- **Y** is the (*i.m* x *i*) matrix of final demand in monetary terms.

- **I** is the (*i.m* x *i.m*) identity matrix.

- The matrix inversion is represented by the exponent $^{-1}$.

More details on the calculations underlying environmental input-output analysis can be found elsewhere [2,71,72].

The MRIO database used in this work was EXIOBASE 3; this database provides a harmonized time series of MRIO tables and environmental extensions ranging from 1995 to 2011[6], sectoral disaggregation of 200 products and 49 regions/countries (Supplementary Methods 6 and 7). Particular important to this work and for the time-series calculation of the biodiversity and ecosystem services are the land-use accounts, developed consistently to the spatial explicitly land-use data set[6].

MRIO models are top-down models that assume a linear relationship between a unit of demand, and the production (and, in this case) land use required to produce goods and services along the supply chain. Accuracy of MRIO analysis is estimated to be in the order of 10-20% at the national level[73,74], given a consistent coverage of the account for the environmental pressure (in this case, land use). High sector detail helps to reduce this uncertainty[75,76], and the EXIOBASE MRIO model provides the highest harmonized sector detail available[77]. Regional aggregation affects results in a similar way to product aggregation[78]. Whilst many comparative MRIO studies find quantitative differences between databases, they also point to robust trends for consumption based accounts observed in all EEMRIO studies such that qualitative conclusions from the quantitative data are reliable[73–80].

IPAT Identity

We used the IPAT identity[81] to distinguish the influence of population growth (P), economic development (A) and technological progress (T) on the evolution of the drivers of biodiversity loss and ecosystem degradation through time:

$$\mathrm{I} = \mathrm{P} \times \frac{I}{A} \times \frac{A}{P} \qquad (13)$$

I refers to impacts (on biodiversity and ecosystem services), in this work the absolute amount of impacts was determined from a supply side perspective, by multiplying the CFs with land-use data, and from a demand side perspective through multi-regional input-output analysis. P refers to population. A refers to affluence measured as Gross Domestic Product (GDP). $\mathrm{I}/_{A}$ is a metric of technological progress and it measures the impacts per unit of GDP. The higher the value less efficient is the economic as more impacts are generated per unit of GDP. $\mathrm{A}/_{P}$ is the metric of affluence in per capita terms. Population data was retrieved from ref.[82] and GDP data was collected in 2011 international dollars (corrected for purchasing power parity) from ref.[83].

References

34. UN. *United Nations regional groups of member states.* (United Nations, 2014).

35. Erb, K.-H. *et al.* A comprehensive global 5 min resolution land-use data set for the year 2000 consistent with national census data. *J. Land Use Sci.* **2,** 191–224 (2007).

36. Ramankutty, N., Evan, A. T., Monfreda, C. & Foley, J. A. Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. *Glob. Biogeochem. Cycles* **22,** GB1003 (2008).

37. FAOSTAT. *Statistical Databases*. http://faostat.fao.org/ (FAO, 2014).

38. Monfreda, C., Ramankutty, N. & Foley, J. A. Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. *Glob. Biogeochem. Cycles* **22,** GB1022 (2008).

39. Schepaschenko, D. *et al.* Development of a global hybrid forest mask through the synergy of remote sensing, crowdsourcing and FAO statistics. *Remote Sens. Environ.* **162,** 208–220 (2015).

40. Potapov, P. *et al.* The last frontiers of wilderness: Tracking loss of intact forest landscapes from 2000 to 2013. *Sci. Adv.* **3,** e1600821 (2017).

41. Müller, M., Pérez Dominguez, I. & Gay, S. H. *Construction of Social Accounting Matrices for EU27 with disaggregated Agricultural Sectors (agroSAM)*. (2009).

42. Erb, K.-H. *et al.* Unexpectedly large impact of forest management and grazing on global vegetation biomass. *Nature* **553,** 73–76 (2018).

43. *Planted forests: uses, impacts and sustainability*. (CABI Publishing, 2009).

44. Penna, I. *Understanding the FAO's 'Wood Supply from Planted Forests' Projections*. (Centre for Environmental Management, University of Ballarat, 2010).

45. Pereira, H. M. & Daily, G. C. Modeling biodiversity dynamics in countryside landscapes. *Ecology* **87,** 1877–1885 (2006).

46. Pereira, H. M., Ziv, G. & Miranda, M. Countryside Species–Area Relationship as a Valid Alternative to the Matrix-Calibrated Species–Area Model. *Conserv. Biol.* **28,** 874–876 (2014).

47. Hanski, I., Zurita, G. A., Bellocq, M. I. & Rybicki, J. Species–fragmented area relationship. *Proc. Natl. Acad. Sci.* **110,** 12715–12720 (2013).

48. Rybicki, J. & Hanski, I. Species–area relationships and extinctions caused by habitat loss and fragmentation. *Ecol. Lett.* **16,** 27–38 (2013).

49. Rosenzweig, M. L. *Species diversity in space and time*. (Cambridge Univ Pr, 1995).

50. Storch, D., Keil, P. & Jetz, W. Universal species–area and endemics–area relationships at continental scales. *Nature* **488,** 78–81 (2012).

51. Sodhi, N. S., Lee, T. M., Koh, L. P. & Brook, B. W. A Meta-Analysis of the Impact of Anthropogenic Forest Disturbance on Southeast Asia's Biotas. *Biotropica* **41,** 103–109 (2009).

52. Hudson, L. N., Newbold, T., Contu, S. & L L Hill, S. *The 2016 release of the PREDICTS database*. (Natural History Museum Data Portal, 2016).

53. ESRI. *ArcGIS*. (Environmental Systems Resource Institute, 2009).

54. Holt, B. G. *et al.* An Update of Wallace's Zoogeographic Regions of the World. *Science* **339,** 74–78 (2013).

55. BirdLife International & NatureServe. *Bird species distribution maps of the world*. (BirdLife International and NatureServe, 2014).

56. Python Software Foundation. Python Language Reference, version 2.7.

57. de Baan, L., Mutel, C. L., Curran, M., Hellweg, S. & Koellner, T. Land Use in Life Cycle Assessment: Global Characterization Factors Based on Regional and Global Potential Species Extinction. *Environ. Sci. Technol.* **47,** 9281–9290 (2013).

58. Holtsmark, B. Harvesting in boreal forests and the biofuel carbon debt. *Clim. Change* **112,** 415–428 (2011).

59. Houghton, R. A. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850-2000. *Tellus B* **55,** 378–390 (2003).

60. Kastner, T., Erb, K.-H. & Nonhebel, S. International wood trade and forest change: A global analysis. *Glob. Environ. Change* **21,** 947–956 (2011).

61. FAO. *Global Ecological Zoning for the Global Forest Resources Assessment, 2000*. (Food and Agriculture Organization of the United Nations, 2001).

62. Ramankutty, N. & Foley, J. Estimating historical changes in global land cover: croplands from 1700 to 1992. *Glob. Biogeochem. Cycles* **13,** 997–1027 (1999).

63. Olson, D. M. *et al.* Terrestrial Ecoregions of the World: A New Map of Life on Earth A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. *BioScience* **51,** 933–938 (2001).

64. Krausmann, F., Erb, K.-H., Gingrich, S., Lauk, C. & Haberl, H. Global patterns of socioeconomic biomass flows in the year 2000: A comprehensive assessment of supply, consumption and constraints. *Ecol. Econ.* **65,** 471–487 (2008).

65. Lauk, C., Haberl, H., Erb, K.-H., Gingrich, S. & Krausmann, F. Global socioeconomic carbon stocks in long-lived products 1900–2008. *Environ. Res. Lett.* **7,** 034023 (2012).

66. Schlesinger, W. H. Are wood pellets a green fuel? *Science* **359,** 1328–1329 (2018).
67. Pingoud, K., Ekholm, T., Sievänen, R., Huuskonen, S. & Hynynen, J. Trade-offs between forest carbon stocks and harvests in a steady state – A multi-criteria analysis. *J. Environ. Manage.* **210,** 96–103 (2018).
68. Davis, S. J. & Caldeira, K. Consumption-based accounting of CO2 emissions. *Proc. Natl. Acad. Sci.* **107,** 5687–5692 (2010).
69. Wiedmann, T. O. *et al.* The material footprint of nations. *Proc. Natl. Acad. Sci.* **112,** 6271–6276 (2015).
70. Kitzes, J. An Introduction to Environmentally-Extended Input-Output Analysis. *Resources* **2,** 489–503 (2013).
71. Kanemoto, K., Lenzen, M., Peters, G. P., Moran, D. D. & Geschke, A. Frameworks for Comparing Emissions Associated with Production, Consumption, And International Trade. *Environ. Sci. Technol.* **46,** 172–179 (2012).
72. Miller, R. & Bair, P. *Input-Ouput Analysis. Foundations and Extensions*. (Cambridge University Press, 2009).
73. Lenzen, M., Wood, R. & Wiedmann, T. Uncertainty Analysis for Multi-Region Input–Output Models – a Case Study of the Uk's Carbon Footprint. *Econ. Syst. Res.* **22,** 43–63 (2010).
74. Moran, D. & Wood, R. Convergence Between the Eora, Wiod, Exiobase, and Openeu's Consumption-Based Carbon Accounts. *Econ. Syst. Res.* **26,** 245–261 (2014).
75. de Koning, A. *et al.* Effect of aggregation and disaggregation on embodied material use of products in input–output analysis. *Ecol. Econ.* **116,** 289–299 (2015).

76. Lenzen, M. Aggregation Versus Disaggregation in Input–Output Analysis of the Environment. *Econ. Syst. Res.* **23,** 73–89 (2011).

77. Wood, R., Hawkins, T. R., Hertwich, E. G. & Tukker, A. Harmonising National Input—Output Tables for Consumption-Based Accounting — Experiences from Exiopol. *Econ. Syst. Res.* **26,** 387–409 (2014).

78. Stadler, K., Steen-Olsen, K. & Wood, R. The ‘Rest of the World’ – Estimating the Economic Structure of Missing Regions in Global Multi-Regional Input–Output Tables. *Econ. Syst. Res.* **26,** 303–326 (2014).

79. Owen, A., Wood, R., Barrett, J. & Evans, A. Explaining value chain differences in MRIO databases through structural path decomposition. *Econ. Syst. Res.* **28,** 243–272 (2016).

80. Steen-Olsen, K., Owen, A., Hertwich, E. G. & Lenzen, M. Effects of Sector Aggregation on Co2 Multipliers in Multiregional Input–Output Analyses. *Econ. Syst. Res.* **26,** 284–302 (2014).

81. Ehrlich, P. R. & Holdren, J. P. Impact of Population Growth. *Science* **171,** 1212–1217 (1971).

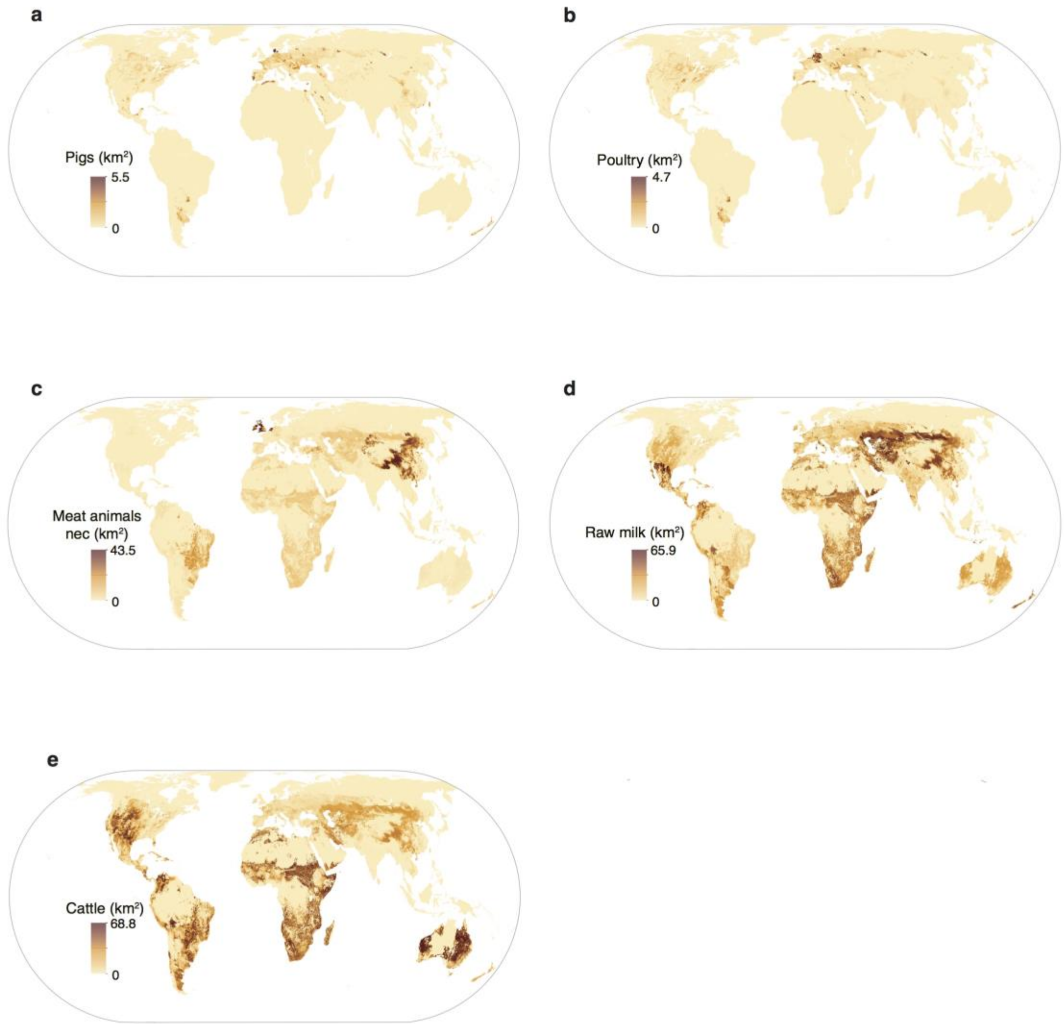
82. World Bank. *World Development Indicators, SP.POP.TOTL Population, total.* (World Bank, 2015).

83. World Bank. *World Bank. 2015b. World Development Indicators, NY.GDP.MKTP.PP.KD, GDP, PPP (constant 2011 international $)*. (World Bank, 2015).

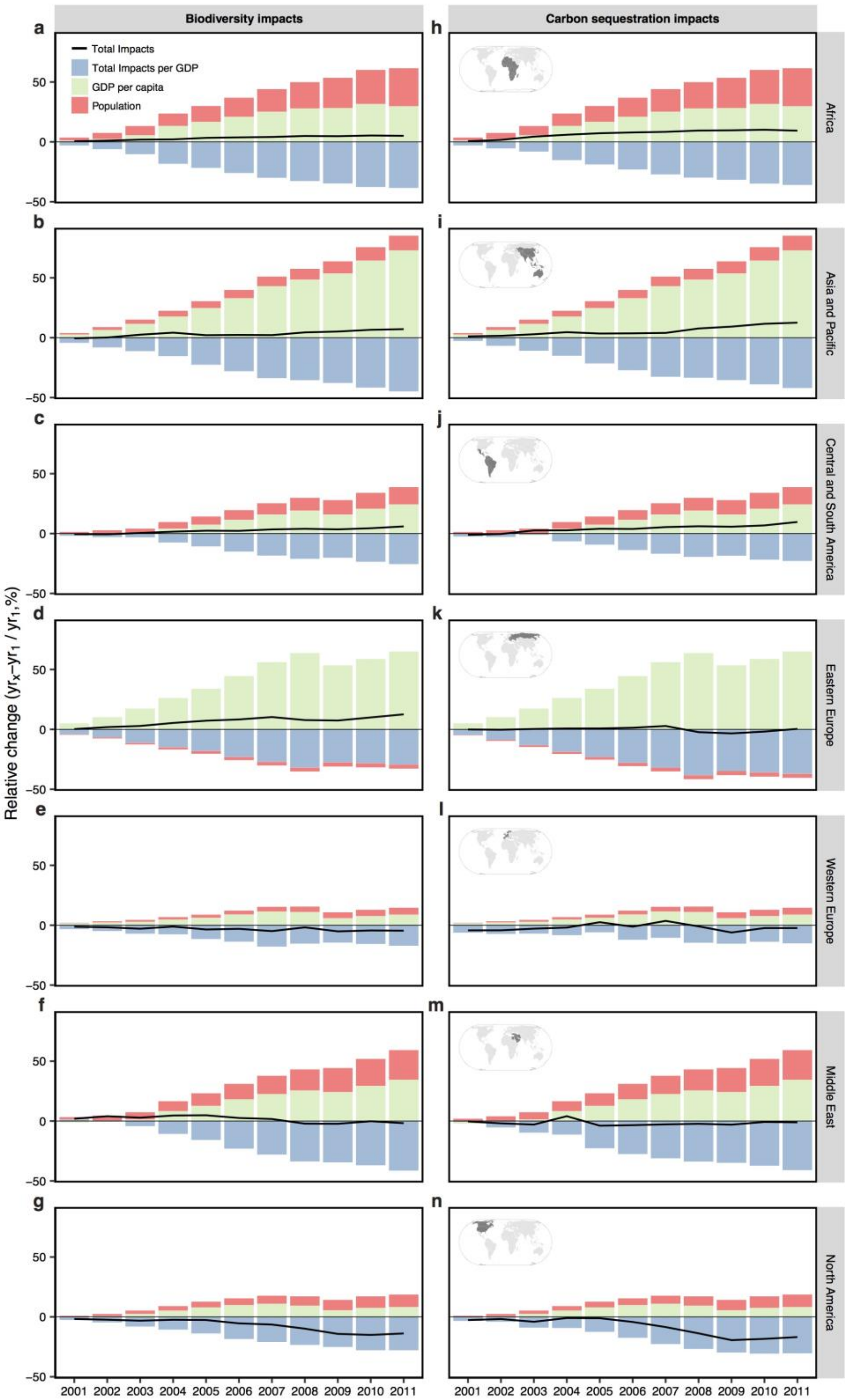
Extended Data

ED Figure 1 – Land-use maps (**a**-**h**), in km^2, for the non-fodder crops layers at 5 arc min resolution (nec = not elsewhere classified).

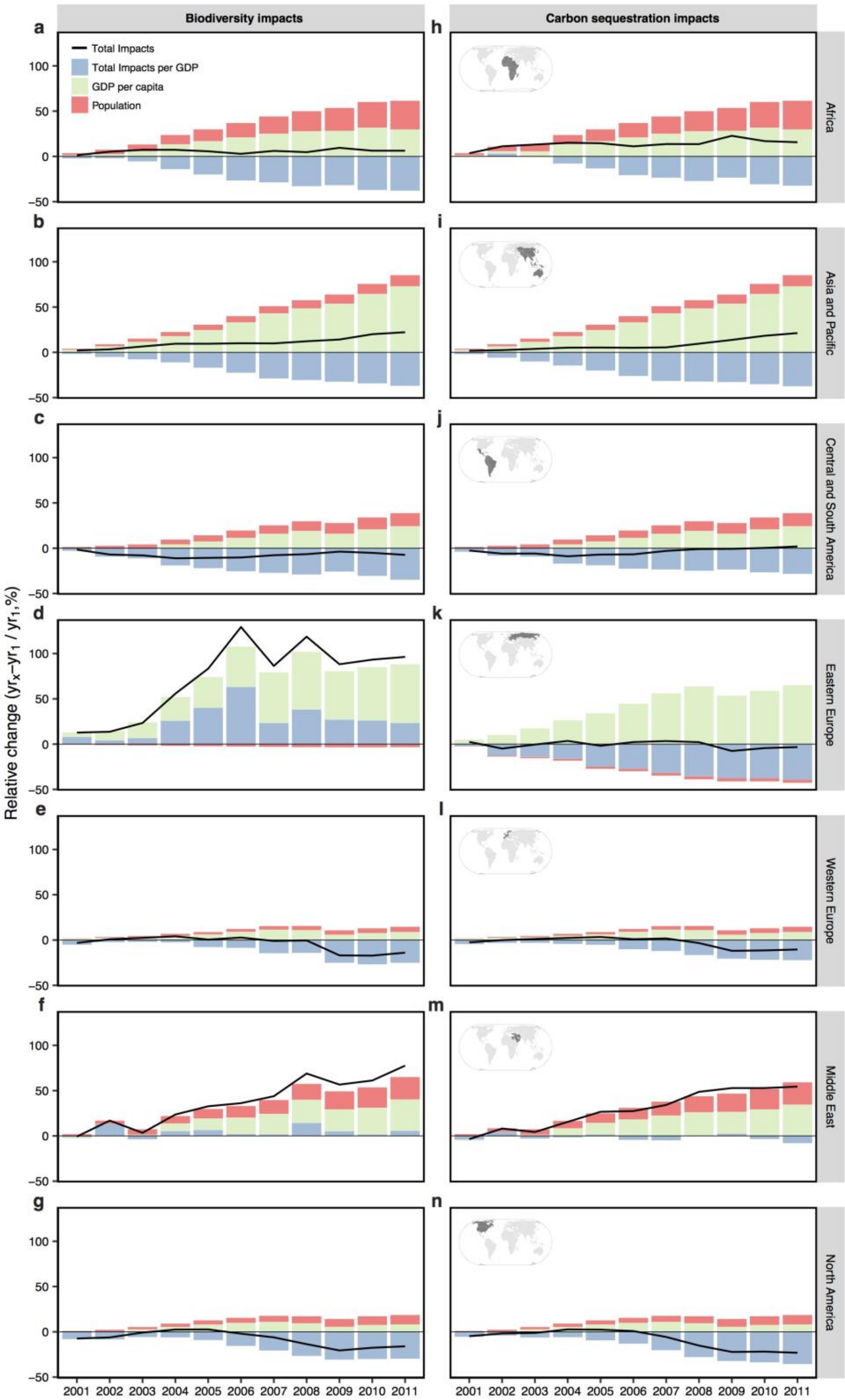
ED Figure 2 – Land-use maps (**a**-**e**), in km^2, for the fodder crops (raw milk, cattle meat, pig meat, poultry and other meat), and permanent pastures (raw milk, cattle meat, other meat) at 5 arc min resolution (nec = not elsewhere classified).



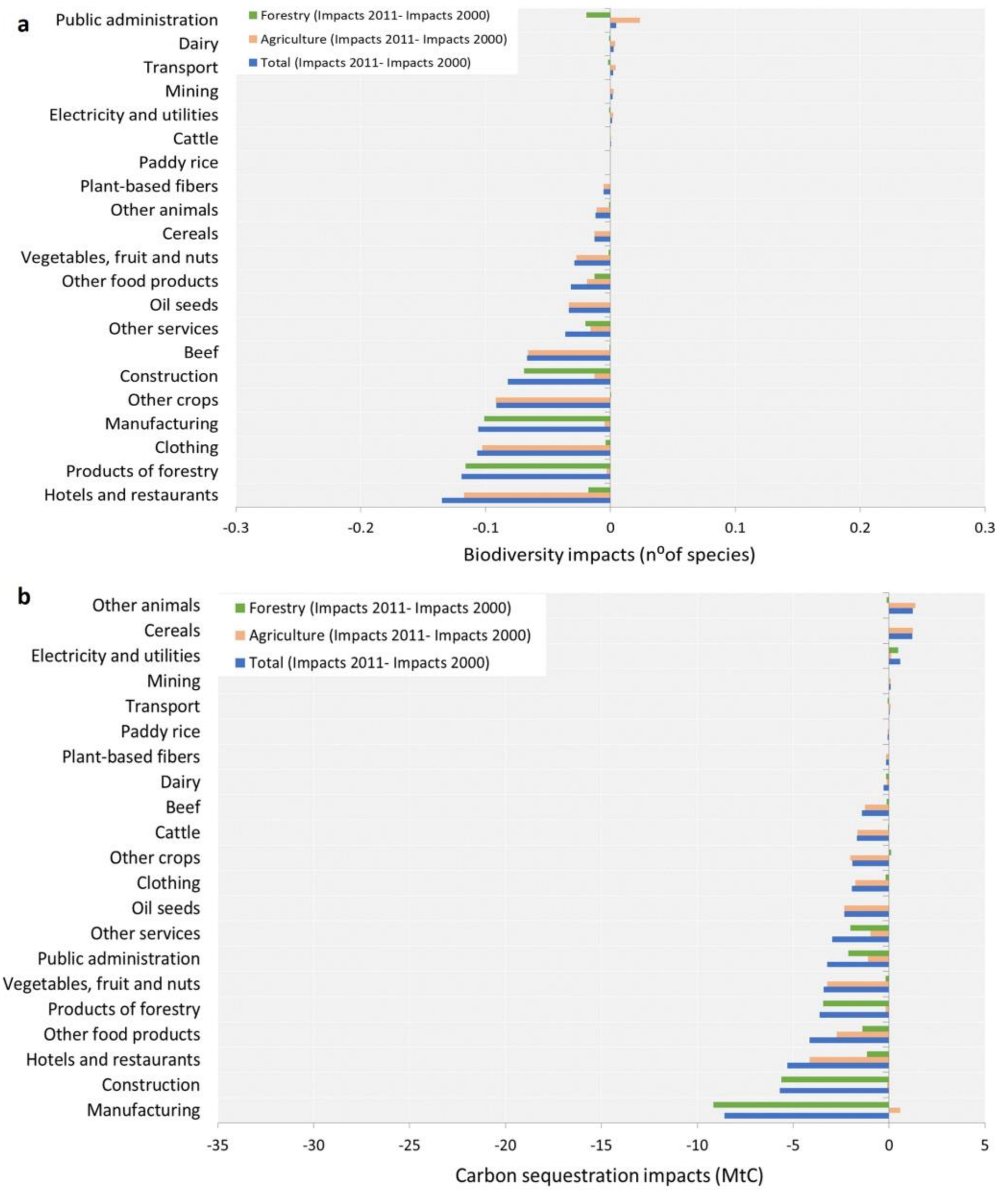
ED Figure 3: Decomposition of impacts from agricultural and forestry production activities on biodiversity (**a**-**g**) and carbon sequestration (**h**-**n**) into their immediate drivers for 7 world regions.



ED Figure 4: Decomposition of impacts from consumption activities on biodiversity (**a-g**) and carbon sequestration (**h-n**) into their immediate drivers for 7 world regions.



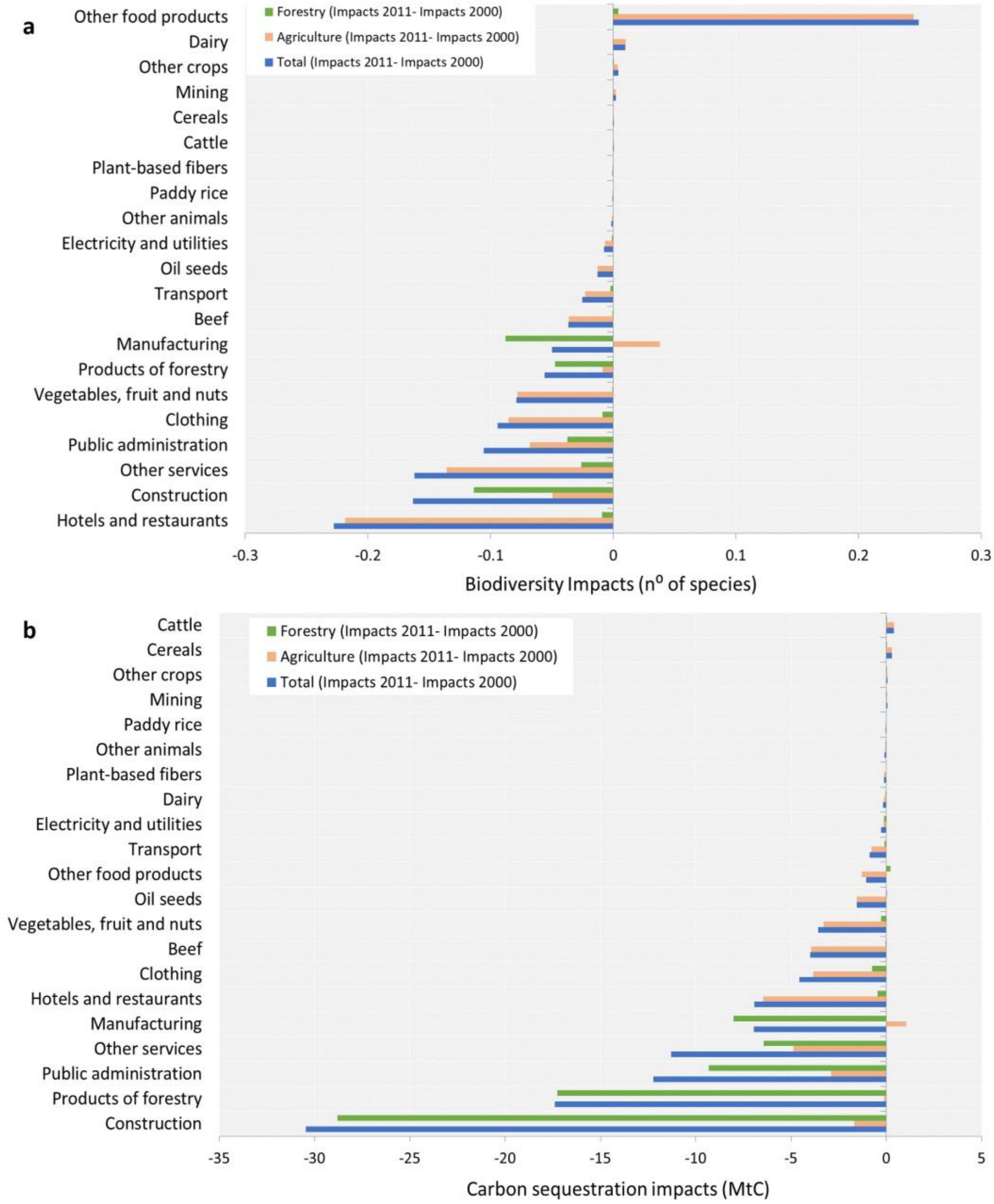
ED Figure 5: Sectoral disaggregation of the change in impacts between 2011 and 2000 on a) biodiversity (**a**; number of bird species) and carbon sequestration (**b**; MtC per year) in Western Europe.



807 ED Figure 6: Sectoral disaggregation of the change in impacts between 2011 and

808 2000 on a) biodiversity (**a**; number of bird species) and carbon sequestration (**b**, MtC

809 per year) in North America.



810

ED Table 1: Impending bird extinctions (species numbers) due to domestic consumption and international trade between world regions, in 2000 and 2011. The grey cells indicate the impacts associated with domestic consumption. In the rows the impacts associated with the exports to other world regions are represented and in the columns the impacts associated with the imports from each region. Summing over the rows provides the total production impacts of a region, summing over the columns the total consumption impacts of a region.

	Western Europe	Eastern Europe	Middle East	North America	Asia and Pacific	Africa	Central and South America
2000							
Western Europe	0.090	0.001	0.002	0.004	0.003	0.001	0.001
Eastern Europe	0.018	0.091	0.006	0.003	0.014	0.001	0.001
Middle East	0.010	0.001	0.093	0.004	0.005	0.002	0.001
North America	0.024	0.002	0.010	0.335	0.055	0.004	0.027
Asia and Pacific	1.460	0.299	0.439	1.642	19.022	0.145	0.238
Africa	2.315	0.191	0.417	0.563	0.711	14.137	0.150
Central and South America	2.083	0.215	0.428	2.179	1.127	0.179	20.733
2011							
Western Europe	0.084	0.003	0.002	0.003	0.004	0.002	0.001
Eastern Europe	0.019	0.082	0.019	0.005	0.019	0.005	0.001
Middle East	0.008	0.003	0.089	0.003	0.007	0.004	0.001
North America	0.016	0.003	0.012	0.253	0.080	0.005	0.025
Asia and Pacific	1.119	0.319	0.570	0.999	21.332	0.296	0.272
Africa	1.902	0.323	0.699	0.630	1.303	14.331	0.234
Central and South America	1.996	0.746	1.089	2.080	2.836	0.738	19.065

ED Table 2: Carbon sequestration lost (MtC) due to international trade between world regions, in 2000 and 2011. The grey cells indicate the impacts associated with domestic consumption. In the rows the impacts associated with the exports to other world regions and in the columns the impacts associated with the imports from each region. Summing over the rows provides the total production impacts of a region, summing over the columns the total consumption impacts of a region.

	Western Europe	Eastern Europe	Middle East	North America	Asia and Pacific	Africa	Central and South America
2000							
Western Europe	185.549	4.374	6.280	8.013	9.816	2.790	2.205
Eastern Europe	43.526	293.921	10.516	7.994	33.127	1.528	1.644
Middle East	1.287	0.186	18.139	0.412	0.538	0.180	0.108
North America	17.751	1.269	4.924	302.099	38.704	1.354	16.062
Asia and Pacific	56.056	11.511	16.702	64.446	998.190	7.134	9.769
Africa	59.098	4.234	9.140	13.034	20.568	247.921	3.202
Central and South America	41.811	3.892	6.585	37.594	21.003	2.556	534.759
2011							
Western Europe	179.731	7.245	5.325	5.443	8.847	4.982	1.688
Eastern Europe	45.229	266.102	26.211	8.630	38.507	7.232	2.740
Middle East	1.022	0.409	17.800	0.295	0.676	0.335	0.088
North America	10.914	2.341	6.393	226.177	55.311	2.281	14.375
Asia and Pacific	47.700	13.915	23.023	43.643	1158.846	12.286	11.569
Africa	43.620	6.802	13.283	13.883	41.665	266.447	4.894
Central and South America	33.224	12.901	19.607	34.793	56.344	10.748	543.413

ED Table 3: Affinity values (*h*) computed for the countryside species area relationship model used in the quantification of biodiversity impacts. Affinity values can be interpreted as the proportion of area of modified habitat that can be effectively used by a particular species group.

	Tropical	Temperate
Cropland	0,062	0,091
Permanent crops	0,077	0,731
Pastures	0,273	0,970
Managed Forest (intensive use)	0,247	0.196
Managed Forest (intensive and extensive use)	0,409	0,239

ED Figure 7– Local scale sensitivity (σ) of species to the full conversion of native habitat into the human-modified habitat (i.e. the proportion of species disappearing at the plot-scale in human-modified habitats) in tropical and temperate regions. **a**, Distribution of σ found in the literature. **b**, range of σ values to the different land-use activities. Error bars in **b** indicate standard errors.