communications earth & environment

ARTICLE

https://doi.org/10.1038/s43247-022-00360-6 OPEN

Relocating croplands could drastically reduce the environmental impacts of global food production

Robert M. Beyer[1,2✉], Fangyuan Hua[3], Philip A. Martin[1,4], Andrea Manica[1,8] & Tim Rademacher[5,6,7,8]

Agricultural production has replaced natural ecosystems across the planet, becoming a major driver of carbon emissions, biodiversity loss, and freshwater consumption. Here we combined global crop yield and environmental data in a ~1-million-dimensional mathematical optimisation framework to determine how optimising the spatial distribution of global croplands could reduce environmental impacts whilst maintaining current crop production levels. We estimate that relocating current croplands to optimal locations, whilst allowing ecosystems in then-abandoned areas to regenerate, could simultaneously decrease the current carbon, biodiversity, and irrigation water footprint of global crop production by 71%, 87%, and 100%, respectively, assuming high-input farming on newly established sites. The optimal global distribution of crops is largely similar for current and end-of-century climatic conditions across emission scenarios. Substantial impact reductions could already be achieved by relocating only a small proportion of worldwide crop production, relocating croplands only within national borders, and assuming less intensive farming systems.

[1] Department of Zoology, University of Cambridge, Cambridge, UK. [2] Potsdam Institute for Climate Impact Research (PIK), Potsdam, Germany. [3] Institute of Ecology, College of Urban and Environmental Sciences, Peking University, Beijing, China. [4] Basque Centre for Climate Change (BC3), Leioa, Spain. [5] Harvard Forest, Harvard University, Petersham, USA. [6] School of Informatics, Computing and Cyber Systems and Center for Ecosystem Science and Society, Northern Arizona University, Flagstaff, USA. [7] Institut des Sciences de la Forêt Tempérée, Université du Québec en Outaouais, Ripon, Canada. [8] These authors contributed equally: Andrea Manica, Tim Rademacher. ✉email: robert.beyer@pik-potsdam.de

The conversion of nearly half of the world's ice-free land area to agricultural areas[1] has contributed to three of humanity's most pressing environmental challenges[2,3]: (i) agriculture is a major source of anthropogenic greenhouse gas emissions[4–6], largely from the release of carbon stored in natural vegetation and soils[7,8]; (ii) agriculture is the main driver of habitat loss, the greatest threat to terrestrial biodiversity[9–12]; and (iii) agriculture is responsible for about 70% of global freshwater consumption[13,14], leading to shortages of potable water in many arid parts of the world[15,16]. Rising global demands for animal products[17] reduce hopes that the benefits of societal dietary shifts to decrease the environmental footprints of food production[2,3,18] can be fully realised in the near future. Yield increases through more resource-efficient practices, technological advancements, and genetically enhanced crop varieties are promising[2,3,19]; however, a growing human population and increasing per-capita consumption[20,21] threaten to offset the potential of these developments without complementary measures.

Ecosystem restoration has been identified as a key strategy for achieving large-scale carbon sequestration and reducing pressures on terrestrial biodiversity[22–24]. Indeed, carbon stocks and biodiversity lost from land conversion can often rapidly approach predisturbance levels when agriculturally degraded areas are allowed to regenerate[25–30] (Supplementary Note 1). However, if total food production levels are to remain constant, the restoration of land currently used for agriculture will require production to be intensified or spatially expanded elsewhere. Whilst previous studies have identified priority areas for ecological restoration[22–24], it is less clear how agricultural production should be spatially redistributed to maximise long-term environmental benefits without compromising food security. In addition to carbon and biodiversity gains, optimally relocating croplands could also substantially reduce the water footprint of agriculture if new areas were established where sufficient rainfall obviates the need for irrigation[31]. Importantly, relocating agricultural areas may not only represent an environmental opportunity, but may become a necessity for maintaining global food security as changing precipitation and snowmelt patterns are threatening crop water supply[32,33] whilst shifting temperature regimes are reducing productivity[34–36] across large parts of the world.

Here we determined the optimal distributions of global croplands that minimise carbon and biodiversity footprints whilst obviating the need for systematic irrigation, under current and future climatic conditions. We used global maps of the current growing areas of 25 major crops[37] (see the "Methods" section), which between them account for 77% of croplands worldwide. For each crop we assessed the carbon impact associated with cultivation in an area as the difference between natural and crop-specific local carbon stocks in vegetation and soils[8] (see the "Methods" section). Similarly, biodiversity impacts were estimated as the difference between local biodiversity under natural vegetation and under cropland[38]. For our main analysis, we measured local biodiversity in terms of range rarity, given by the sum of the inverse natural range sizes of locally occurring species (see the "Methods" section), a metric advocated as particularly meaningful for conservation planning[39], given the strong relationship between species' range sizes and their vulnerability to extinction[40,41]. The same methods allowed us to predict the potential carbon and biodiversity impacts of crop production in locations that are currently not cultivated (see the "Methods" section). We then combined these impact estimates with a global dataset of agro-ecologically attainable yields, available for the same 25 crops, across both currently cultivated and uncultivated areas, and for current and projected future climatic conditions[42]. We used potential yield estimates for three alternative management scenarios, representing the range from traditional, subsistence-based organic farming systems to advanced, fully mechanised production with high-yielding crop varieties and optimum fertiliser and pesticide application[42] (see the "Methods" section). All estimates were derived assuming only rainfed water supply[42]; thus, any configuration of croplands based on them represents a scenario in which no systematic irrigation is required. Using these potential yields, we considered all possible spatial distributions of rainfed croplands for which the total global production of each individual crop was the same as at present (see the "Methods" section). In this step, the area assumed to be potentially available for agriculture in a grid cell was defined as the area not currently covered by water bodies, land unsuitable due to soil and terrain constraints, urban areas and infrastructure, crops not included in our analysis, pasture lands, and protected areas (see the "Methods" section). Finally, among all such distributions of croplands, we identified those for which global carbon and biodiversity impacts were minimal. On the 20 arc-minute (0.33°) grid used here, this required solving a ~1-million-dimensional linear optimisation problem (see the "Methods" section).

Results and discussion

Optimal transnational relocation with high-input crop management. The optimal distribution of croplands determined here depends on how carbon and biodiversity are weighted relative to each other in the impact minimisation, and we consider this trade-off later in our analysis; the following results are based on an optimal weighting between carbon and biodiversity impacts designed to minimise trade-offs (see the "Methods" section). We first consider the scenario of high-input crop management on relocated croplands, and examine the effects of less intensive farming practices later on. For this scenario, we estimate that a complete optimisation of the spatial distribution of croplands would simultaneously reduce the current carbon and biodiversity impacts of global crop production by 71% and 87%, respectively, whilst eliminating the need for irrigation altogether (Fig. 1). The total worldwide area used for agriculture in the optimised scenario would be less than half (48%) of the current area. The amount of carbon sequestered in vegetation and soils would be equivalent to 20 years of the current annual net increase of atmospheric CO_2 of 5.1 Pg carbon per year[43]. Pressures on terrestrial species would be drastically lower than they are today given the major role of agricultural habitat destruction for global biodiversity loss[10]. In particular, optimally sited croplands overlap with only 0.2% of areas classified as tropical forest biomes[44], representing a reduction of 98% in the tropical forest area that is currently being used for the crops in our analysis. Particularly pronounced spatial clusters of optimal cropping locations include growing areas of barley, cotton, rapeseed, soybean, sunflower, and wheat around the corn belt in the Midwestern USA; of cotton, maize, soybean, millet, and rice along a longitudinal band south of the Sahel zone in Sub-Saharan Africa; of maize, rice, and soybean in northeast Argentina and neighbouring regions; and of rapeseed and soybean in northeast China (Supplementary Movie 1), driven by high potential rainfed yields combined with relatively low environmental impacts in these areas (Supplementary Movie 2, Supplementary Fig. 1a–c).

Optimal national relocation with high-input crop management. Moving agricultural production, and thus labour and capital, across national borders poses political and socioeconomic challenges that may be difficult to resolve in the near future. We therefore repeated our analysis whilst requiring that, in the optimised scenario, the total production of each crop in each country remain identical to the one at present (see the

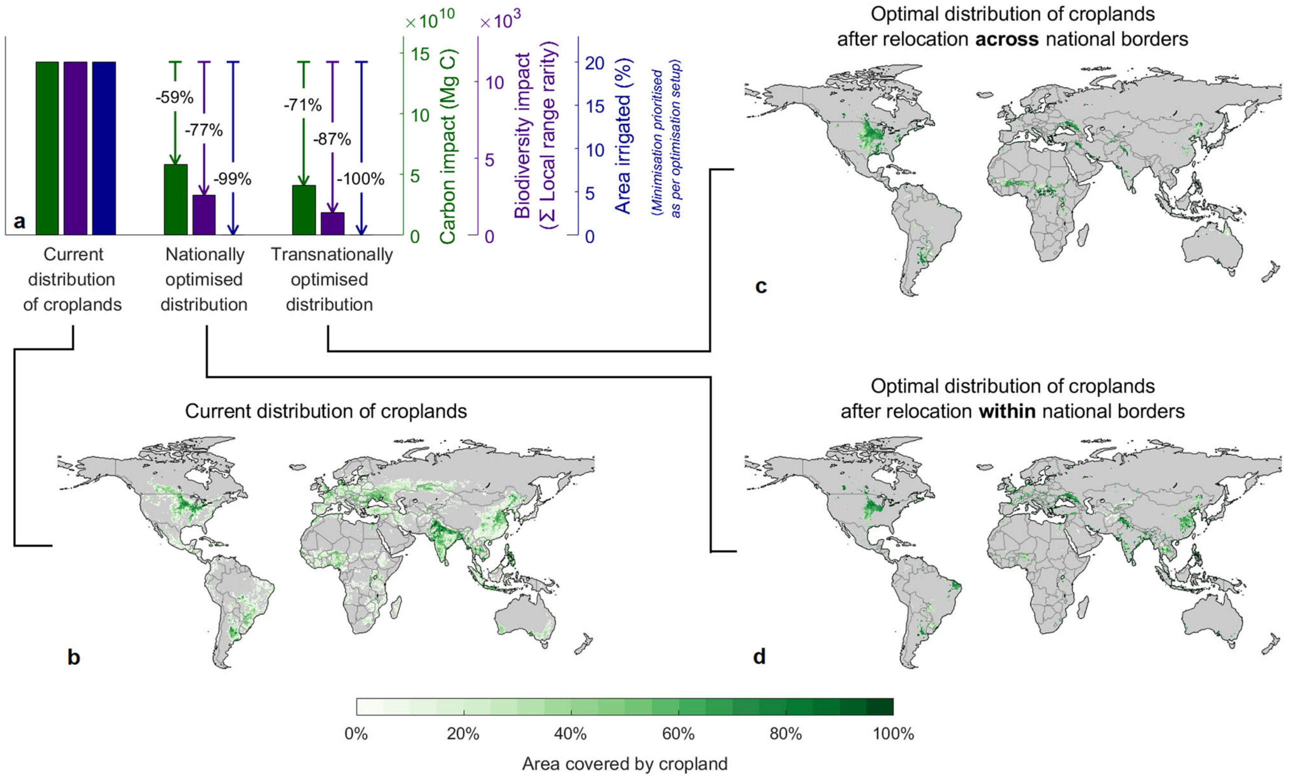

Fig. 1 Current and environmentally optimal distributions of global croplands. The estimated reduction potentials in **a** are based on an optimal trade-off between carbon and biodiversity impacts (see the "Methods" section). Potential yield data used here are based on current climatic conditions and assume high-input crop management and rainfed water supply, so that relocated areas are, by design of the approach, not irrigated. The global production levels of individual crops for optimally distributed areas are identical to current levels; in the scenario of national relocation, this is additionally the case for national production levels. For visualisation purposes, in **b**–**d**, the 25 crops were grouped together; maps of the optimal distribution of individual crops for across- and within-border relocation are shown in Supplementary Movie 1.

"Methods" section). Assuming high-input crop management on relocated sites, we estimate that if each country independently optimised its distribution of croplands, the resulting worldwide carbon and biodiversity impacts would be 59% and 77% lower than they are currently (Fig. 1). The vast majority of production, corresponding to 99.4% of global croplands, could be nationally relocated so that rainfall provides sufficient water supply; however, some countries produce crops for which natural agro-ecological conditions within their borders are not suitable, and thus some irrigation or greenhouse cultivation remains needed to maintain the current national production levels of each crop (see the "Methods" section).

Optimal cropland distribution under future climate scenarios. The optimal global distribution of growing areas providing the same total production of each crop as at present based on current climatic conditions is to a substantial degree similar for climatic conditions at the end of the century, irrespective of the specific climate change scenario. 73%, 73%, 70%, and 63% of the optimal areas for RCP 2.6, 4.5, 6.0, and 8.5 climate in 2071–2100 (Fig. 2a–d), respectively, overlap with those associated with the current climate in Fig. 1c. In particular, the four aforementioned major spatial clusters largely remain in place; though, parts the band south of the Sahel zone become somewhat less suitable under increased warming, whilst optimal sites in Argentina expand. Some smaller areas emerge as optimal locations as the result of climate change, including parts near the northwest American coast and east of the Ural Mountains in Russia (Fig. 2). Overall, our analysis suggests that if the global demand for the crops considered here does not decrease below current levels in the coming decades, then croplands established in optimal locations now would largely remain optimally sited in the future. The lack of suitable long-term projections of future global crop production levels (which are either too short-term[45] or not specific enough[46]) prevents us from determining the optimal distributions of croplands for specific future scenarios of global food production; however, given an anticipated continued growth of the global population, total production levels of major crops are predicted to increase at least over the next decade[45]. In principle, large-scale dietary shifts could lead to a decrease in the future demand for some of the crops considered here, in which case some areas identified in our analysis may no longer represent optimal cropping locations; however, whilst such shifts could in theory generate nutritional and environmental benefits[3,47–49], recent trends and future projections dampen expectations about a swift implementation[46,50].

Partial relocation of production and lower-input crop management. Thus far, we have considered the scenario in which all crop production is optimally relocated and in which newly established croplands are managed according to best practice. How do the potential environmental benefits of crop relocation change if these assumptions are altered? Across crops, the environmental impacts attributed to the production of one unit of produce vary greatly across current growing areas: half of the global carbon and biodiversity impact attributed to each crop is currently caused on areas accounting for only 23% and 5% of the crop's total production, respectively (Supplementary Fig. 2). This suggests that prioritising the relocation of production away from these areas–where potential carbon stocks and biodiversity are

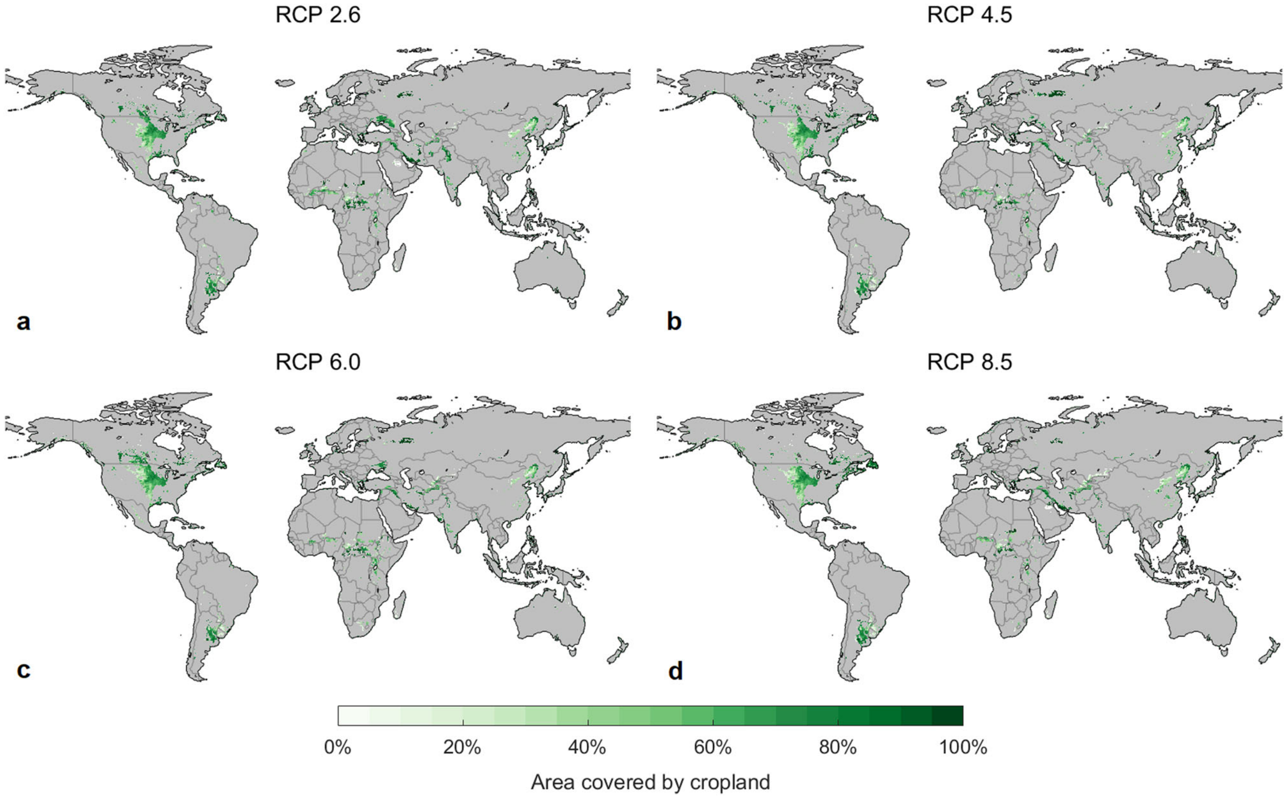

Fig. 2 Optimal distributions of global croplands for end-of-century climate. The maps show the equivalents of Fig. 1c based on potential yields projected for 2071–2100 climate under four alternative emission scenarios: RCP **a** 2.6, **b** 4.5, **c** 6.0, **d** 8.5. In each scenario, the production levels of individual crops are identical to current ones, given the lack of suitable future projections (see text). Maps of the distribution of the 25 individual crops for each climate scenario are shown in Supplementary Movie 3. Optimal cropland sites for relocation within national borders (i.e., equivalents of Fig. 1d) were not estimated, as the optimisation problem in this case is ill-defined (see the "Methods" section).

high and yields are low–could generate particularly large environmental benefits. We repeated our analysis of the optimal distribution of croplands across and within national borders, but assumed that only a certain proportion of the current total production of each crop is being moved, beginning with current croplands where local impacts are highest in relation to local production (see the "Methods" section). As before, yields on newly established areas assume only rainfall water supply. For high-input management on new croplands, optimally relocating only around 15% of the production of each crop across borders, or around 25% within borders, could already generate around half of the potential carbon and biodiversity benefits previously estimated for a complete redistribution of areas (Fig. 3a, b). Even relocating the least agro-environmentally efficient areas accounting for as little as 5% of production to optimal locations could produce considerable environmental gains (Fig. 3a, b).

Potential impact reductions estimated under the assumption of high-input crop management on newly established croplands are necessarily higher than for less intensive farming practices, given the higher land requirements in the latter case. An important aspect in the context of lower-input crop management on new sites is that relocation beyond a certain proportion of production can increase environmental impacts, rather than reduce them. This occurs when crop production is moved from existing areas, on which yields are high as the result of intensive farming, to locations where crops are managed less intensively; this step may require an additional area that is so large that it offsets the environmental benefits of the new site. For medium- and low-input crop management, respectively, on newly established croplands, we estimate the threshold beyond which transnational or national relocation is no longer beneficial in terms of reducing carbon and biodiversity impacts at around 75% and 20–30% of production (Fig. 3c–f). At these levels, and assuming medium-input management on new croplands, current carbon and biodiversity impacts of global crop production would be simultaneously reduced by an estimated 54% and 80%, respectively, for an optimal relocation across national borders (Fig. 3c, black marker), and by 35% and 63%, respectively, for an optimal relocation within countries (Fig. 3d, black marker). For low-input management on new areas, carbon and biodiversity impacts, respectively, would be reduced by 27% and 65% for transnational relocation (Fig. 3e, black marker), and by 8% and 37% for national relocation (Fig. 3f, black marker). These results reaffirm that closing yield gaps is important for reducing the environmental footprint of agriculture[2,3,18–20,51,52]; at the same time, even for less intensive farming practices, carbon and biodiversity impacts on optimally distributed growing areas are substantially lower than they are at present (Fig. 3c–f), demonstrating that substantial benefits could be achieved by cropland relocation alone. This is particularly relevant given that implementing high-input crop management can be difficult, due to short-term marginal economic returns of mechanising production, using high-yielding crop varieties, and improving fertiliser and pesticide or due to local socio-economic, political, or infrastructural constraints[53]. Governmental and intergovernmental efforts to support farmers in sustainably intensifying production are key for realising the environmental benefits associated with closing agricultural yield gaps[54].

Carbon and biodiversity impact reductions associated with cropland relocation are disproportionally high in relation to the

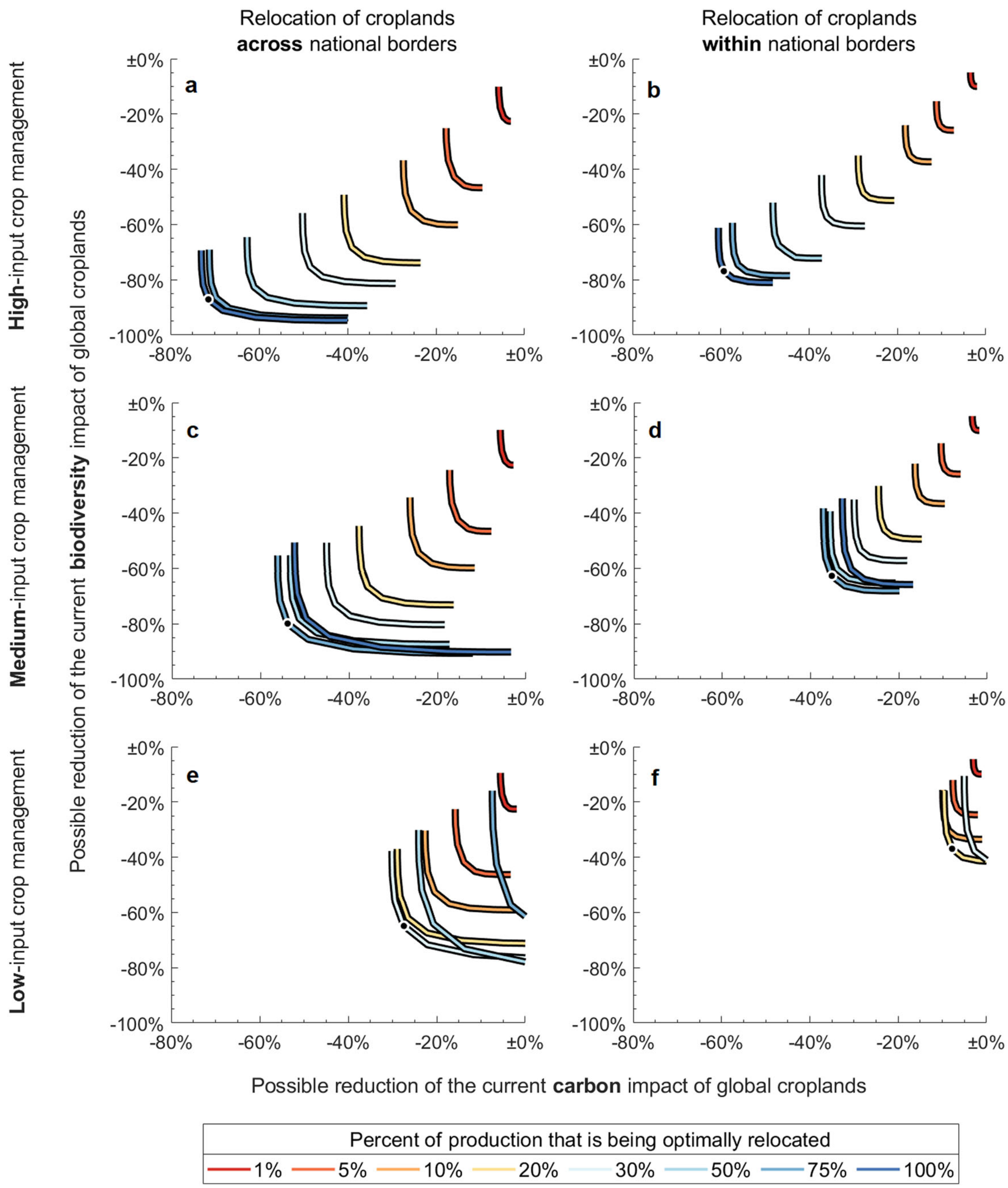

Fig. 3 Reductions of carbon and biodiversity impacts for different relocation levels and management scenarios. Each coloured line represents the set of simultaneously achievable carbon and biodiversity impact reductions, relative to current impacts, which vary according to the weight given to them in the optimisation framework (see the "Methods" section). As per the data used, relocated croplands are rainfed; the resulting percentage of irrigated croplands in each scenario (ranging between 21% for the current distribution, and 0% and 0.6% in the scenario of full relocation across and within national borders, respectively) is not displayed. Black markers represent the optimal trade-off between carbon and biodiversity impacts (see the "Methods" section); those in **a** and **b** correspond to the cropland maps in Fig. 1c, d, respectively, and those in **c**–**f** to the maps in Supplementary Fig. 3a–d. In all scenarios, the global production levels of individual crops are identical to current ones; in the scenarios of national relocation, this is additionally the case for national production levels. All estimates are based on current climatic conditions.

change in total cropland area (Supplementary Fig. 4). For high-input crop management (Supplementary Fig. 4a), this demonstrates that the estimated benefits are not merely a consequence of the decrease in total area due to higher yields. This fact becomes even clearer in the case of low-input management, where considerable impact reductions can be achieved despite an increase in total cropland area caused by low yields on newly established areas (Supplementary Fig. 4c).

Trade-off between carbon and biodiversity impact reduction. The trade-off between reducing carbon versus biodiversity impacts is very small, as shown by the high convexity of the lines representing sets of simultaneously achievable carbon and biodiversity impact reductions in Fig. 3. Optimising the distribution of croplands for each of the two impact measures independently allows for reduction potentials of up to 73% and 94%, respectively, in the case of a transnational relocation of areas, and up to

61% and 81%, respectively, in the case of national relocation, assuming high-input management on new croplands (Fig. 3a, b, end points of dark blue lines). These reductions are only slightly higher than those obtained simultaneously for the scenario of an optimal weighting considered hitherto (Fig. 3a, b, black markers). Results are similar for low- and medium-input crop management (Fig. 3c–f).

Sensitivity to biodiversity metric. The strong relationship between species' range sizes and their vulnerability to extinction[40,41] makes range rarity a particularly relevant biodiversity metric for conservation[39]; however, like any indicator, it captures only certain aspects of biological diversity[55]. To assess the effect of using an alternative metric, we repeated our analysis in Fig. 1, but measured biodiversity in terms of species richness (Methods). We estimated that for a globally optimised distribution of croplands and high-input rainfed yields, carbon and species richness impacts of crop production would be 70% and 77% lower, respectively, than they are at present, whilst reductions of 60% and 67%, respectively, could be achieved by an optimal relocation within national borders (Supplementary Fig. 5). The locations of optimal cropping sites in these scenarios are overall similar to those identified based on range rarity, with 69% and 86% of optimal areas for across- and within-border relocation, respectively, overlapping with the respective areas in Fig. 1c, d. In particular, in the case of transnational relocation, optimal areas overlap in the Midwestern US corn belt, northeast Argentina, northeast China, and north of the Black Sea, though, the longitudinal band south of the Sahel zone is no longer optimal in terms of species richness (Supplementary Fig. 5b), due to the relatively high number of (generally not small-ranged) species in this area (Supplementary Fig. 1c, d). The slightly lower potentials for reducing biodiversity impacts, compared to our results based on range rarity, are due to the more uniform spatial distribution of species richness compared to that of range rarity (Supplementary Fig. 1c, d).

Carbon payback times after crop relocation. The restoration of carbon stocks and biodiversity on abandoned agricultural areas is a long-term process, whereas the environmental impacts of relocating crop production to currently uncultivated optimal locations would be quasi-instantaneous. How long would it take for these new impacts to be offset by the gradual regeneration of abandoned areas? Assuming conservative estimates of the carbon recovery trajectory on agriculturally degraded land (see the "Methods" section) and high-input management on new croplands, we estimated that the net carbon impact of optimally relocating all current crop production within national borders would break even within a decade, and in less than half of that time for relocation across borders (Supplementary Fig. 6). For the optimal relocation of smaller shares of production that are currently located in areas with the highest restoration potentials (see the "Methods" section), break-even points would be reached disproportionally sooner (Supplementary Fig. 6) as the result of the unequal distribution of impacts and production across current areas (Supplementary Fig. 2). These short time scales again illustrate the considerable difference between current and minimum achievable carbon impacts. We did not attempt to estimate break-even timings for biodiversity, given the lack of recovery estimates with suitable spatial coverage.

Policy aspects of cropland relocation. How can the relocation of croplands, at small or large scales, be implemented in practice? A number of national and supranational set-aside schemes, aimed at retiring agricultural land for environmental benefits, can offer useful templates for how payments for ecosystem services can reduce current impacts in socio-economically sustainable ways[56–58]. As a case example, China's Grain for Green programme, the world's largest national scheme of payments for ecosystem services, achieved the regeneration of 15 million hectares of farmland to forests between 1999 and 2010, with overall positive economic outcomes for the 124 million people involved[59]. The programme generated substantial carbon and biodiversity benefits, despite being primarily aimed at reducing soil erosion[59,60]. It also facilitated an effective relocation of cropland from southern to northern China[61]. Designing incentives to encourage the abandonment and regeneration of the least agro-environmentally efficient areas will be crucial for achieving benefits most effectively. International climate funds can support countries lacking the financial means for payments to farmers in implementing durable set-aside schemes[62]. In addition, in many parts of the world, agricultural subsidies prevent the abandonment of agricultural land that would otherwise occur naturally[63]; reducing subsidies in areas with high potential carbon stocks and biodiversity therefore represents a particularly cost-efficient strategy for generating environmental benefits[64]. A number of financial, infrastructural, and policy measures, ranging from land-use zoning to strategic agronomic support and certification schemes[54], can incentivise the establishment of new agricultural land in optimal target areas. Simultaneously, strong legal and policy frameworks[65–68] are needed to ensure an effective protection of regenerating abandoned croplands, and promote active restoration to support the recovery process when necessary. Finally, it is crucial that such measures accommodate the social equity dimension of agricultural land abandonment and relocation[69]. Acknowledging that financial compensation cannot replace social and cultural assets, and that relocation of farmland must be consensual, needs to be a cornerstone of any programme design.

Uncertainties. Uncertainties in our analysis are linked to those of the relevant input data, specifically the global maps of carbon and biodiversity impacts and actual and potential yields; though, the lack of uncertainty estimates for these datasets does not allow us to formally quantify uncertainties in our analysis. Country- and crop-specific data used to generate global maps of observed yields and harvested areas used here differ in quality; however, the derived maps have been curated and validated extensively based on independent regional and national datasets and local expert opinion[37]. Estimates of potential crop yields are constrained by the availability and quality of climatic, ecological, and agricultural data required to calibrate yield models; spatially heterogeneous information on global soil properties in particular has been noted as a relevant limitation[42]. However, the derived estimates have undergone extensive ground-truthing based on local statistical records across the world[42]. Potential natural carbon stocks, used here to assess carbon impacts, were derived based on established methodologies that achieves robustness by providing estimates that are generally specific to a given combination of ecosystem, climate, and geographical region, and which may therefore underrepresent finer spatial heterogeneities[8]. Expert species range maps, used to estimate biodiversity impacts, are not without uncertainties and inaccuracies[70,71]; however, they allow for a consistent and transparent estimation of the effects of agricultural land conversion on natural biodiversity on currently cultivated and uncultivated areas worldwide that would not be trivial for alternative approaches to achieve[38,72].

Future perspectives. Considering rainfed potential yields has allowed us to identify cropland distributions that obviate the need for systematic irrigation; however, in several parts of the world, local water resources are at present[73], or will be in future[74], abundant enough to allow for sustainable irrigation, which can substantially increase yields[53]. This highlights the need for potential yield estimates assuming irrigation-based water supply, which are currently not available to the same extent as the rainfed yields used here[42]. Such estimates, combined with crop water use models[31], would allow for a rigorous assessment of the three-way trade-offs between carbon, biodiversity, and water impacts.

We examined three different crop management scenarios at the global scale; in reality, farming intensity and yield gap closure are highly spatially heterogeneous[53], depending on local socio-economic conditions. Estimates of the likely yield levels of crops in locations where they are currently not grown, given the specific current or projected future local circumstances, would be valuable for determining optimal relocation strategies in local contexts.

Here we did not attempt to also optimise the spatial distribution of global pastures (but assumed these to be unavailable for new cropland), due to a lack of appropriate maps of potential grass growth and the higher complexity of livestock compared to crop production processes. However, their immense size suggests tremendous potential for reducing environmental impacts through optimal relocation, which deserve examining. In this context of animal-based products, as well as that of crops, it will be highly informative to explore the potential benefit of a combined strategy of dietary shifts and spatial optimisation of agricultural areas, in which environmental impacts are minimised not whilst maintaining the global production levels of individual crops, as done here, but whilst meeting an appropriate production of nutrients.

Conclusions

Spatial reallocation of agricultural production has tremendous potential for reducing the environmental impacts of global food production. Importantly, cropland relocation need not be implemented at full scale in order to generate substantial benefits; even a redistribution of small parts of production, across smaller spatial scales, and without fully optimised management on new sites, would have considerable positive effects. Whilst the political and socio-economic challenges associated with this strategy are undeniable, a range of proven policy measures are available for facilitating relevant steps in a sustainable manner. Their implementation will be crucial for realising the environmental potential of moving agricultural areas, providing gains that are badly needed if we are to reverse the ongoing degradation of global climate, biodiversity, and water under an ever-increasing demand for food.

Methods

We use the notation in Table 1.

Current crop production and areas, $P_i(x)$, $H_i(x)$. We used 5-arc-minute maps of the fresh-weight production $P_i(x)$ (Mg year^{-1}) and cropping area $H_i(x)$ (ha) of 25 major crops (Table 2) in the year 2010[37]. These represent the most recent spatially explicit and crop-specific global data[75]. Separate maps were available for irrigated and rainfed croplands, allowing us to estimate the worldwide proportion of irrigated areas as 21% of all croplands.

Agro-ecologically attainable yields $\widehat{Y}_i(x)$. We used 5-arc-minute maps of the agro-ecologically attainable dry-weight yield (Mg ha $^{-1}$ year^{-1}) of the same 25 crops on worldwide potential growing areas (Supplementary Movie 3) from the GAEZ v4 model, which incorporates thermal, moisture, agro-climatic, soil, and terrain conditions[42]. These yield estimates were derived based on the assumption of rainfed water supply (i.e., without additional irrigation) and are available for current climatic conditions and, assuming a CO_2 fertilisation effect, for four future (2071–2100 period) climate scenarios corresponding to representative concentration pathways (RCPs) 2.6, 4.5, 6.0, and 8.5[76] simulated by the HadGEM2-ES model[77]. Potential rainfed yield estimates for current climatic conditions were available for a low- and a high-input crop management level, representing, respectively, subsistence-based organic farming systems and advanced, fully mechanised production using high-yielding crop varieties and optimum fertiliser and pesticide application[42]. We additionally considered potential yields representing a medium-input management scenario, given by the mean of the relevant low- and high-input yields. Future potential yields were available only for the high-input management level. Thus, we considered a total of 175 ($=25\times3$ present $+$ 25×4 future) potential yield maps. Potential dry-weight yields were converted to fresh-weight yields, $\widehat{Y}_i(x)$, using crop-specific conversion factors[42,78].

Both current and future potential rainfed yields from GAEZ v4 were simulated based on daily weather data, and therefore account for short-term events such as frost days, heat waves, and wet and dry spells[42]. However, the estimates represent averages of annual yields across 30-year periods; thus, whilst the need for irrigation on cropping areas identified in our approach during particularly dry years may in principle be obviated by suitable storage of crop production[79], in practice, ad hoc irrigation may be an economically desirable measure to maintain productivity during times of drought, which are projected to increase in different geographic regions due to climate change[80,81].

Table 1 Notation used in the description of the optimisation framework.

x	Index representing a spatial grid cell
i	Index representing a crop
$A(x)$	Area of grid cell x (ha)
$P_i(x)$	Current production of crop i in grid cell x (Mg year^{-1})
$H_i(x)$	Area covered by crop i in grid cell x (ha)
$\widehat{Y}_i(x)$	Agro-ecologically attainable yield of crop i in grid cell x (Mg ha $^{-1}$ year^{-1})
$C_i(x)$	Carbon impact of crop i in grid cell x (Mg carbon ha^{-1})
$B_i(x)$	Biodiversity impact of crop i in grid cell x (local range rarity loss)
$V(x)$	Area potentially available for agriculture in grid cell x (ha)

Table 2 Crops included in the analysis.

Banana	Cotton	Oil palm	Rice	Sweet potato
Barley	Cowpea	Peal millet	Soybean	Tea
Cassava	Green bean	Plantain	Sugar beet	Tobacco
Coconut	Groundnut	Potato	Sugar cane	Wheat
Coffee	Maize	Rapeseed	Sunflower	Yams

Carbon impact $C_i(x)$. Following an earlier approach[8], the carbon impact of crop production, $C_i(x)$, in a 5-arc-minute grid cell was estimated as the difference between the potential natural carbon stocks and the cropland-specific carbon stocks, each given by the sum of the relevant vegetation- and soil-specific carbon. The change in vegetation carbon stocks resulting from land conversion is given by the difference between carbon stored in the potential natural vegetation, available as a 5-arc-minute global map[8] (Supplementary Fig. 1a), and carbon stored in the crops, for which we used available estimates[8,78]. Regarding soil, spatially explicit global estimates of soil organic carbon (SOC) changes from land cover change are not available. We therefore chose a simple approach, consistent with estimates across large spatial scales, rather than a complex spatially explicit model for which, given the limited empirical data, robust predictions across and beyond currently cultivated areas would be difficult to achieve. Following an earlier approach[8], and supported by empirical meta-analyses[82–86], we assumed that the conversion of natural habitat to cropland results in a 25% reduction of the potential natural SOC. For the latter, we used a 5-arc-minute global map of pre-agricultural SOC stocks[7] (Supplementary Fig. 1b). Thus, the total local carbon impact (Mg C ha^{-1}) of the production of crop i in the grid cell x was estimated as

$$C_i(x) = C_{\text{potential vegetation}}(x) + 0.25 \cdot C_{\text{potential SOC}}(x) - C_{\text{crop}}(i) \quad (1)$$

where $C_{\text{potential vegetation}}(x)$ and $C_{\text{potential SOC}}(x)$ denote the potential natural carbon stocks in the vegetation and the soil in x, respectively, and $C_{\text{crop}}(i)$ denotes the carbon stocks of crop i (all in Mg C ha^{-1}). By design, the approach allows us to estimate the carbon impact of the conversion of natural habitat to cropland regardless of whether an area is currently cultivated or not.

In our analysis, we did not consider greenhouse gas emissions from sources other than from land use change, including nitrous emissions from fertilised soils and methane emissions from rice paddies[87]. In contrast to the one-off land use change emissions considered here, those are ongoing emissions that incur continually in the production process. We would assume that the magnitude of these emissions in a scenario of redistribution of agricultural areas, in which the total production of each crop remains constant, is roughly similar to that associated with the current distribution of areas. We also did not consider emissions associated with transport; however, these have been shown to be small compared to other food chain emissions[88] and poorly correlated with the distance travelled by agricultural products[89].

Biodiversity impact $B_i(x)$. Analogous to our approach for carbon, we estimated the biodiversity impact of crop production, $B_i(x)$, in a 5-arc-minute grid cell as the difference between the local biodiversity associated with the natural habitat and that associated with cropland. For our main analysis, we quantified local biodiversity in terms of range rarity (given by the sum of inverse species range sizes; see below) of mammals, birds, and amphibians. Range rarity has been advocated as a biodiversity measure particularly relevant to conservation planning in general[39,90–93] and the protection of endemic species in particular[39]. In a supplementary analysis, we additionally considered biodiversity in terms of species richness.

We used 5-arc-minute global maps of the range rarity and species richness of mammals, birds, and amphibians under potential natural vegetation (Supplementary Fig. 1c, d) and under cropland land cover[94]. The methodology used to generate these data[38] combines species-specific extents of occurrence (spatial envelopes of species' outermost geographic limits[40]) and habitat preferences (lists of land cover categories in which species can live[95]), both available for all mammals, birds, and amphibians[96,97], with a global map of potential natural biomes[44] in order to estimate which species would be present in a grid cell for natural habitat conditions. Incorporating information on species' ability to live in croplands, included in the habitat preferences, allows for determining the species that would, and those that would not, tolerate a local conversion of natural habitat to cropland. The species richness impact of crop production in a grid cell is then obtained as the number of species estimated to be locally lost when natural habitat is converted to cropland. Instead of weighing all species equally, the range rarity impact in a grid cell is calculated as the sum of the inverse potential natural range sizes of the species locally lost when natural habitat is converted; thus, increased weight is attributed to range-restricted species, which tend to be at higher extinction risk[40,41].

As in the case of carbon, the approach allows us to estimate the biodiversity impact of crop production in both currently cultivated and uncultivated areas.

Land potentially available for agriculture, $V(x)$. We defined the area $V(x)$ (ha) potentially available for crop production in a given grid cell x, as the area not currently covered by water bodies[42], land unsuitable due to soil and terrain constraints[42], built-up land (urban areas, infrastructure, roads)[1], pasture lands[1], crops not considered in our analysis[37], or protected areas[42] (Supplementary Fig. 1e). In the scenario of a partial relocation of crop production, in which a proportion of existing croplands is not moved, the relevant retained areas are additionally subtracted from the potentially available area, as described further below.

Optimal transnational relocation. We first consider the scenario in which all current croplands are relocated across national borders based on current climate (Fig. 3a, dark blue line). For each crop i and each grid cell x, we determined the local (i.e., grid-cell-specific) area $\widehat{H}_i(x)$ (ha) on which crop i is grown in cell x so that the total production of each crop i equals the current production and the environmental impact is minimal. Denoting by

$$\bar{P}_i = \sum_x P_i(x) \tag{2}$$

the current global production of crop i, any solution $\widehat{H}_i(x)$ must satisfy the equality constraints

$$\sum_x \widehat{H}_i(x) \cdot \widehat{Y}_i(x) = \bar{P}_\mathrm{i} \text{ for } \quad \text{each crop } i \tag{3}$$

requiring the total production of each individual crop after relocation to be equal to the current one. A solution must also satisfy the inequality constraints

$$\sum_i \widehat{H}_i(x) \le V(x) \text{ for } \quad \text{each grid cell } x\,, \tag{4}$$

ensuring that the local sum of cropping areas is not larger than the locally available area $V(x)$ (see above). Given these constraints, we can identify the global configuration of croplands that minimises the associated total carbon or biodiversity impact by minimising the objective function

$$\sum_x \widehat{H}_i(x) \cdot C_i(x) \to \ \min \quad \text{or} \quad \sum_x \widehat{H}_i(x) \cdot B_i(x) \to \ \min \tag{5}$$

respectively. More generally, we can minimise a combined carbon and biodiversity impact measure, and examine potential trade-offs between minimising each of the two impacts, by considering the weighted objective function

$$\sum_x \widehat{H}_i(x) \cdot (\alpha \cdot C_i(x) + (1-\alpha) \cdot B_i(x)) \to \ \min \tag{6}$$

where the weighting parameter α ranges between 0 and 1.

Considering all crops across all grid cells, we denote by

$$\bar{C} = \sum_i \sum_x H_i(x) \cdot C_i(x) \tag{7}$$

the global carbon impact associated with the current distribution of croplands, and by

$$\hat{C}(\alpha) = \sum_i \sum_x \hat{H}_i(x) \cdot C_i(x) \tag{8}$$

the global carbon impact associated with the optimal distribution $\{\widehat{H}_i(x)\}_{i,x} (= \{\widehat{H}_i^\alpha(x)\}_{i,x})$ of croplands for some carbon-biodiversity weighting $\alpha \in [0,1]$. The relative change between the current and the optimal carbon impact is then given by

$$\hat{c}(\alpha) = 100\% \cdot \frac{\hat{C}(\alpha) - \bar{C}}{\bar{C}} \tag{9}$$

Using analogous notation, the relative change between the current and the optimal global biodiversity impact across all crops and grid cells is given by

$$\widehat{b}(\alpha) = 100\% \cdot \frac{\widehat{B}(\alpha) - \bar{B}}{\bar{B}} \tag{10}$$

The dark blue line in Fig. 3a visualises $\widehat{c}(\alpha)$ and $\widehat{b}(\alpha)$ for the full range of carbon-biodiversity weightings $\alpha \in [0,1]$, each of which corresponds to a specific optimal distribution $\{\widehat{H}_i(x)\}_{i,x}$ of croplands. We defined an optimal weighting α_opt, meant to represent a scenario in which the trade-off between minimising the total carbon impact and minimising the total biodiversity impact is as small as possible. Such a weighting is necessarily subjective; here, we defined it as

$$\alpha_\mathrm{opt} = \arg\min_{\alpha \in [0,1]} \left| \frac{\frac{\partial \hat{c}(\alpha)}{\partial \hat{b}(\alpha)}}{\hat{c}(\alpha)} \cdot \frac{\frac{\partial \hat{b}(\alpha)}{\partial \hat{c}(\alpha)}}{\hat{b}(\alpha)} \right| \tag{11}$$

Each of the two factors on the right-hand side represents the relative rate of change in the reduction of one impact type with respect to the change in the reduction of the other one as α varies. Thus, α_opt represents the weighting at which neither impact type can be further reduced by varying α without increasing the relative impact of the other by at least the same amount. Scenarios based on this optimal weighting are shown in Figs. 1, 2, and Supplementary Figs. 3–6, and are represented by the black markers in Fig. 3.

Our approach does not account for multiple cropping; i.e., part of a grid cell is not allocated to more than one crop, and the assumed annual yield is based on a single harvest. Allowing for multiple crops to be successively planted in the same location during a growing period would increase the dimensionality of the optimisation problem substantially. However, given that only 5% of current global rainfed areas are under multiple cropping[98], this is likely not a strong limitation of our rainfed-based analysis. As a result of this approach, our results may even slightly underestimate local crop production potential and therefore global impact reduction potentials.

Optimal national relocation. In the case of areas being relocated within national borders, the mathematical framework is identical with the exception that the sum over relevant grid cells x in Eqs. (2) and (4) is taken over the cells that define the given country of interest, instead of the whole world. In this way, the total production of each crop within each country for optimally distributed croplands is the same as for current areas. The optimisation problem is then solved independently for each country.

Optimal partial relocation. When (either for national or transnational relocation) only a certain proportion $\lambda \in [0,1]$ of the production of each crop (of a country or the world) is being relocated rather than the total production, Eq. (3) changes to

$$\sum_x \widehat{H}_i(x) \cdot \widehat{Y}_i(x) = \lambda \cdot \bar{P}_i \text{ for each crop } i\,. \tag{12}$$

In addition, the area potentially available for new croplands, $V(x)$, (see above) is reduced by the area that remains occupied by current croplands accounting for the proportion $(1-\lambda)$ of production that is not being relocated. We denote by $H_i^\lambda(x)$ the area that continues to be used for the production of crop i in grid cell x in the scenario where the proportion λ of the production is being optimally redistributed. In particular, $H_i^0(x) = H_i(x)$ and $H_i^1(x) = 0$ for all i and x. For a given carbon-biodiversity weighting $\alpha \in [0,1]$ in Eq. (6), $H_i^\lambda(x)$ is calculated as follows. First, all grid cells in which crop i is currently grown are ordered according to their agro-environmental efficiency, i.e., the grid-cell-specific ratio between the environmental impact attributed to the production of the crop and the local production,

$$E_i^\alpha(x) = \frac{H_i(x) \cdot (\alpha \cdot C_i(x) + (1-\alpha) \cdot B_i(x))}{P_i(x)}. \tag{13}$$

Let $x_1 (= x_1(i,\alpha))$ denote the index of the grid cell in which crop i is currently grow for which E_i^α is smallest among all grid cells in which the crop is grown. Then let x_2 be the index for which E_i^α is second smallest (or equal to the smallest), and so on. Thus, the vector $(x_1, x_2, x_3, \ldots)$ contains all indices of grid cells where crop i is currently grown in descending order of agro-environmental efficiency. The area

Table 3 Assumed times required for carbon stocks on abandoned cropland to reach pre-disturbance levels.

Ecosystem	Assumed carbon recovery time, T_{carbon}
Tropical forest	150 years
Temperate forest, boreal forest, woodland	150 years
Boreal forest	150 years
Savanna	75 years
Grassland, shrubland	50 years
Desert	25 years

$H_i^\lambda(x_n)$ retained in some grid cell x_n is then given by

$$H_i^\lambda(x_n) = \begin{cases} H_i(x_n) & \text{if } \sum_{m=1}^{n} P_i(x_m) \le (1-\lambda)\cdot \bar{P}_i \\ 0 & \text{else} \end{cases} \quad (14)$$

Thus, cropping areas in a grid cell x_n are retained if they are amongst the most agro-environmentally efficient ones of crop i on which the combined production does not exceed $(1-\lambda)\cdot\bar{P}_i$ (which is not being relocated). Growing areas in the remaining, less agro-environmental efficient grid cells are abandoned and become potentially available for other relocated crops. Note that H_i^λ depends on the weighting α of carbon against biodiversity impacts. Finally, instead of Eq. (4), we have, in the case of the partial relocation of the proportion λ of the total production,

$$\sum_i \widehat{H}_i(x) \le V(x) - H_i^\lambda(x) \quad \text{for each grid cell } x\,. \quad (15)$$

Solving the optimisation problem. All datasets needed in the optimisation (i.e., $A(x)$, $P_i(x)$, $H_i(x)$, $C_i(x)$, $B_i(x)$, $\widehat{Y}_i(x)$, $V(x)$) are available at a 5 arc-minute (0.083°) resolution; however, computational constraints required us to upscale these to a 20-arc-minute grid (0.33°) spatial grid. At this resolution, Eq. (6) defines a 1.12×10^6-dimensional linear optimisation problem in the scenario of across-border relocation. The high dimensionality of the problem is in part due to the requirement in Eq. (3) that the individual production level of each crop is maintained. Requiring instead that, for example, only the total caloric production is maintained[31,99] reduces Eq. (6) to a 1-dimensional problem. However, in such a scenario, the production of individual crops, and therefore of macro- and micronutrients, would generally be very different from current levels, implicitly assuming potentially drastic dietary shifts that may not be nutritionally or culturally realistic.

The optimisation problem in Eq. (6) was solved using the dual-simplex algorithm in the function *linprog* of the Matlab R2021b Optimization Toolbox[100] for a termination tolerance on the dual feasibility of 10^{-7} and a feasibility tolerance for constraints of 10^{-4}.

In the case of a transnational relocation of crop production, the algorithm always converged to the optimal solution, i.e., for all crop management levels, climate scenarios, and proportions of production that were being relocated. For the relocation within national borders, this was not always the case. This is because some countries produce small quantities of crops which, according to the GAEZ v4 potential yield estimates, could not be grown in the relevant quantities anywhere in the country under natural climatic conditions and for rainfed water supply; these crops likely require greenhouse cultivation or irrigation can therefore not be successfully relocated within our framework. Across all countries, this was the case for production occurring on 0.6% of all croplands. When this was the case for a certain country and crop, we excluded the crop from the optimisation routine, and a country's total carbon and biodiversity impacts were calculated as the sum of the impacts of optimally relocated crops plus the current impacts of non-relocatable crops.

This issue is linked to why determining the optimal distribution of croplands within national borders is not a well-defined problem for future climatic conditions. Under current climatic conditions, if a crop cannot be relocated within our framework, then its current distribution offers a fall-back solution that provides the current production level and allows us to quantify environmental impacts. Different climatic conditions in the future mean that the production of a crop across current growing locations will not be the same as it is today, and therefore the fall-back solution available for the present is no longer available, so that a consistent quantification of the environmental impacts of a non-relocatable crop is not possible.

Carbon and biodiversity recovery trajectories. Our analysis in Supplementary Fig. 6 requires spatially explicit estimates of the carbon recovery trajectory on abandoned croplands. Whilst carbon and biodiversity regeneration have been shown to follow certain general patterns, recovery is context-specific (Supplementary Note 1) in that, depending on local conditions, the regeneration in a specific location can take place at slower or faster speeds than would typically be the case in the broader ecoregion. Here, we assumed that these caveats can be accommodated by using conservative estimates of recovery times and by assuming that local factors will average out at the spatial resolution of our analysis. The carbon recovery times assumed here are based on ecosystem-specific estimates of the time required for abandoned agricultural areas to retain pre-disturbance carbon stocks[82]. Aiming for a conservative approach, we assumed carbon recovery times equal to at least three times these estimates, rounded up to the nearest quarter century (Table 3). Independent empirical estimates from specific sites and from meta-analyses are well within these time scales (Supplementary Note 1).

Applying the values in Table 3 to a global map of potential natural biomes[44] provides a map of carbon recovery times. We assumed a square root-shaped carbon recovery trajectory across these regeneration periods[101]; similar trajectories, sometimes modelled by faster-converging exponential functions, have been identified in other studies[25,27,30,102–105]. Thus, the carbon stocks in an area of a grid cell x previously used to grow crop i were assumed to regenerate according to the function

$$\begin{cases} C_{\mathrm{agricultural}}(x) + \sqrt{\frac{t}{T_{\mathrm{carbon}}(x)}}\cdot\left(C_{\mathrm{potential}}(x) - C_{\mathrm{agricultural}}(x)\right) & \text{if } t < T_{\mathrm{carbon}} \\ C_{\mathrm{potential}}(x) & \text{if } t \ge T_{\mathrm{carbon}} \end{cases} \quad (16)$$

where, using the same notation as further above

$$\begin{aligned} C_{\mathrm{potential}}(x) &= C_{\mathrm{potential\ vegetation}}(x) + C_{\mathrm{potential\ SOC}}(x) \\ C_{\mathrm{agricultural}}(x) &= C_i(x) + 0.75\cdot C_{\mathrm{potential\ SOC}}(x) \end{aligned} \quad (17)$$

Reporting summary. Further information on research design is available in the Nature Research Reporting Summary linked to this article.

Data availability

Data associated with this study are available on the Open Science Framework (https://doi.org/10.17605/OSF.IO/MHS9K).

Code availability

Code associated with this study is available on the Open Science Framework (https://doi.org/10.17605/OSF.IO/MHS9K).

Received: 15 January 2021; Accepted: 27 January 2022;
Published online: 10 March 2022

References

1. Ellis, E. C. et al. Anthropogenic transformation of the biomes 1700 to 2000. *Glob. Ecol. Biogeogr.* **19**, 589–606 (2010).
2. Foley, J. A. et al. Solutions for a cultivated planet. *Nature* **478**, 337–342 (2011).
3. Springmann, M. et al. Options for keeping the food system within environmental limits. *Nature* **562**, 519–525 (2018).
4. Vermeulen, S. J., Campbell, B. M. & Ingram, J. S. I. Climate change and food systems. *Annu. Rev. Environ. Resour.* **37**, 195–222 (2012).
5. Tubiello, F. N. et al. The contribution of agriculture, forestry and other land use activities to global warming, 1990–2012. *Glob. Change Biol.* **21**, 2655–2660 (2015).
6. Crippa, M. et al. Food systems are responsible for a third of global anthropogenic GHG emissions. *Nat. Food* **2**, 198–209 (2021).
7. Sanderman, J., Hengl, T. & Fiske, G. J. Soil carbon debt of 12,000 years of human land use. *Proc. Natl. Acad. Sci.* **114**, 9575–9580 (2017).
8. West, P. C. et al. Trading carbon for food: Global comparison of carbon stocks vs. crop yields on agricultural land. *Proc. Natl. Acad. Sci.* **107**, 19645–19648 (2010).
9. Sala, O. E. et al. Global Biodiversity Scenarios for the Year 2100. *Science* **287**, 1770–1774 (2000).
10. IPBES. *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (IPBES Secretariat, 2019).
11. Dudley, N. & Alexander, S. Agriculture and biodiversity: a review. *Biodiversity* **18**, 45–49 (2017).
12. Benton, T. G., Bieg, C., Harwatt, H., Pudasaini, R. & Wellesley, L. *Food system impacts on biodiversity loss: Three Levers Food System Transformation in Support of Nature* (Chatham House, 2021).
13. Postel, S. L., Daily, G. C. & Ehrlich, P. R. Human appropriation of renewable fresh water. *Science* **271**, 785–788 (1996).
14. Gleick, P. H. et al. *The World's Water*, Vol. 7: *The Biennial Report on Freshwater Resources* (Island press, 2012).

15. Rosegrant, M.W., Cai, X. & Cline, S. A. *World water and food to 2025: dealing with scarcity* (International Food Policy Research Institute, 2006).
16. Wada, Y., van Beek, L. P. H. & Bierkens, M. F. P. Nonsustainable groundwater sustaining irrigation: a global assessment. *Water Resour. Res*. **48**, W00L06 (2012).
17. Thornton, P. K. Livestock production: recent trends, future prospects. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* **365**, 2853–2867 (2010).
18. Tilman, D. et al. Future threats to biodiversity and pathways to their prevention. *Nature* **546**, 73–81 (2017).
19. Clay, J. Freeze the footprint of food. *Nature* **475**, 287–289 (2011).
20. Tilman, D., Balzer, C., Hill, J. & Befort, B. L. Global food demand and the sustainable intensification of agriculture. *Proc. Natl. Acad. Sci.***108**, 20260–20264 (2011).
21. Myers, N. & Kent, J. New consumers: the influence of affluence on the environment. *Proc. Natl. Acad. Sci. USA* **100**, 4963–4968 (2003).
22. Bastin, J.-F. et al. The global tree restoration potential. *Science* **365**, 76–79 (2019).
23. Strassburg, B. B. N. et al. Strategic approaches to restoring ecosystems can triple conservation gains and halve costs. *Nat. Ecol. Evol.* **3**, 62–70 (2019).
24. Strassburg, B. B. N. et al. Global priority areas for ecosystem restoration. *Nature* **586**, 724–729 (2020).
25. Poorter, L. et al. Multidimensional tropical forest recovery. *Science* **374**, 1370–1376 (2021).
26. Jones, H. P. & Schmitz, O. J. Rapid recovery of damaged ecosystems. *PLoS ONE* **4**, e5653 (2009).
27. Poorter, L. et al. Biomass resilience of Neotropical secondary forests. *Nature* **530**, 211–214 (2016).
28. Silver, W. L., Ostertag, R. & Lugo, A. E. The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. *Restor. Ecol.* **8**, 394–407 (2000).
29. Chazdon, R. L. et al. The potential for species conservation in tropical secondary forests. *Conserv. Biol. J. Soc.* **23**, 1406–1417 (2009).
30. Gilroy, J. J. et al. Cheap carbon and biodiversity co-benefits from forest regeneration in a hotspot of endemism. *Nat. Clim. Change* **4**, 503–507 (2014).
31. Davis, K. F., Rulli, M. C., Seveso, A. & D'Odorico, P. Increased food production and reduced water use through optimized crop distribution. *Nat. Geosci.* **10**, 919–924 (2017).
32. Elliott, J. et al. Constraints and potentials of future irrigation water availability on agricultural production under climate change. *Proc. Natl. Acad. Sci.***111**, 3239–3244 (2014).
33. Qin, Y. et al. Agricultural risks from changing snowmelt. *Nat. Clim. Change* **10**, 459–465 (2020).
34. Teixeira, E. I., Fischer, G., van Velthuizen, H., Walter, C. & Ewert, F. Global hot-spots of heat stress on agricultural crops due to climate change. *Agric. For. Meteorol.* **170**, 206–215 (2013).
35. Zhao, C. et al. Temperature increase reduces global yields of major crops in four independent estimates. *Proc. Natl Acad. Sci.***114**, 9326–9331 (2017).
36. Rosenzweig, C. et al. Assessing agricultural risks of climate change in the 21st century in a global gridded crop model intercomparison. *Proc. Natl. Acad. Sci.***111**, 3268–3273 (2014).
37. Yu, Q. et al. A cultivated planet in 2010 – Part 2: The global gridded agricultural-production maps. *Earth Syst. Sci. Data* **12**, 3545–3572 (2020).
38. Beyer, R. M. & Manica, A. Historical and projected future range sizes of the world's mammals, birds, and amphibians. *Nat. Commun.* **11**, 5633 (2020).
39. Guerin, G. R. & Lowe, A. J. 'Sum of inverse range-sizes' (SIR), a biodiversity metric with many names and interpretations. *Biodivers. Conserv.* **24**, 2877–2882 (2015).
40. Gaston, K. J. & Fuller, R. A. The sizes of species' geographic ranges. *J. Appl. Ecol.* **46**, 1–9 (2009).
41. Staude, I. R., Navarro, L. M. & Pereira, H. M. Range size predicts the risk of local extinction from habitat loss. *Glob. Ecol. Biogeogr.* **29**, 16–25 (2020).
42. Fischer, G. et al. *Global Agro Ecological Zones v4 Model Documentation* (FAO, 2021).
43. Friedlingstein, P. et al. Global carbon budget 2020. *Earth Syst. Sci. Data* **12**, 3269–3340 (2020).
44. Ramankutty, N. & Foley, J. A. Estimating historical changes in global land cover: croplands from 1700 to 1992. *Glob. Biogeochem. Cycles* **13**, 997–1027 (1999).
45. OECD/FAO. *OECD-FAO Agricultural Outlook 2021–2030* (OECD Publishing, 2021).
46. Bodirsky, B. L. et al. The ongoing nutrition transition thwarts long-term targets for food security, public health and environmental protection. *Sci. Rep.* **10**, 19778 (2020).
47. Kim, B. F. et al. Country-specific dietary shifts to mitigate climate and water crises. *Glob. Environ. Change* **62**, 101926 (2020).
48. Behrens, P. et al. Evaluating the environmental impacts of dietary recommendations. *Proc. Natl. Acad. Sci.***114**, 13412–13417 (2017).
49. Aleksandrowicz, L., Green, R., Joy, E. J. M., Smith, P. & Haines, A. The impacts of dietary change on greenhouse gas emissions, land use, water use, and health: a systematic review. *PLoS ONE* **11**, e0165797 (2016).
50. Swinburn, B. A. et al. The global obesity pandemic: shaped by global drivers and local environments. *The Lancet* **378**, 804–814 (2011).
51. Fader, M., Gerten, D., Krause, M., Lucht, W. & Cramer, W. Spatial decoupling of agricultural production and consumption: quantifying dependences of countries on food imports due to domestic land and water constraints. *Environ. Res. Lett.* **8**, 014046 (2013).
52. Kummu, M. et al. Bringing it all together: linking measures to secure nations' food supply. *Curr. Opin. Environ. Sustain.* **29**, 98–117 (2017).
53. Mueller, N. D. et al. Closing yield gaps through nutrient and water management. *Nature* **490**, 254–257 (2012).
54. Phalan, B. et al. How can higher-yield farming help to spare nature? *Science* **351**, 450–451 (2016).
55. Harper, J. L. & Hawksworth, D. L. Biodiversity: measurement and estimation. Preface. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.* **345**, 5–12 (1994).
56. Buskirk, J. V. & Willi, Y. Enhancement of farmland biodiversity within set-aside land. *Conserv. Biol.* **18**, 987–994 (2004).
57. OECD. *Environmental Indicators for Agriculture,* Vol. 3. *Methods and Results* (OECD Publishing, 2001).
58. Sutherland, W. J., Dicks, L. V., Petrovan, S. O. & Smith, R. K. *What Works in Conservation 2021* (Open Book Publishers, 2021).
59. Delang, C. O. & Yuan, Z. *China's Grain for Green Program: A Review of the Largest Ecological Restoration and Rural Development Program in the World* (Springer International Publishing, 2015).
60. Hua, F. et al. Opportunities for biodiversity gains under the world's largest reforestation programme. *Nat. Commun.* **7**, 12717 (2016).
61. Liu, J. et al. Spatiotemporal characteristics, patterns, and causes of land-use changes in China since the late 1980s. *J. Geogr. Sci.* **24**, 195–210 (2014).
62. Bishop, J. & Hill, C. *Global Biodiversity Finance: The Case for International Payments for Ecosystem Services* (Edward Elgar Publishing Ltd, 2014).
63. Li, S. & Li, X. Global understanding of farmland abandonment: a review and prospects. *J. Geogr. Sci.* **27**, 1123–1150 (2017).
64. Myers, N. & Kent, J. *Perverse Subsidies: How Tax Dollars Can Undercut the Environment and the Economy* (Island Press, 2001).
65. Lamb, D., Erskine, P. D. & Parrotta, J. A. Restoration of degraded tropical forest landscapes. *Science* **310**, 1628–1632 (2005).
66. Chazdon, R. L. Beyond deforestation: restoring forests and ecosystem services on degraded lands. *Science* **320**, 1458–1460 (2008).
67. Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F. & Rey-Benayas, J. M. Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends Ecol. Evol.* **26**, 541–549 (2011).
68. Chazdon, R. L. & Guariguata, M. R. Natural regeneration as a tool for large-scale forest restoration in the tropics: prospects and challenges. *Biotropica* **48**, 716–730 (2016).
69. Benayas, J., Martins, A., Nicolau, J. & Schulz, J. Abandonment of agricultural land: an overview of drivers and consequences. *CAB Rev. Perspect. Agric. Vet. Sci. Nutr. Nat. Resour*. **2**, 1–14 (2007).
70. Akçakaya, H. R. et al. Making consistent IUCN classifications under uncertainty. *Conserv. Biol.* **14**, 1001–1013 (2000).
71. Hughes, A. C., Orr, M. C., Yang, Q. & Qiao, H. Effectively and accurately mapping global biodiversity patterns for different regions and taxa. *Glob. Ecol. Biogeogr.* **30**, 1375–1388 (2021).
72. Jetz, W., Wilcove, D. S. & Dobson, A. P. Projected impacts of climate and land-use change on the global diversity of birds. *PLoS Biol.* **5**, e157 (2007).
73. D'Odorico, P. & Rodriguez-Iturbe, I. Sustaining water resources. In *Health of People, Health of Planet and Our Responsibility: Climate Change, Air Pollution and Health* (eds Al-Delaimy, W. K., Ramanathan, V. & Sánchez Sorondo, M.) 149–163 (Springer International Publishing, 2020).
74. Rosa, L. et al. Potential for sustainable irrigation expansion in a 3 °C warmer climate. *Proc. Natl. Acad. Sci.***117**, 29526–29534 (2020).
75. Kim, K.-H. et al. A review of global gridded cropping system data products. *Environ. Res. Lett.* **16**, 093005 (2021).
76. Vuuren, D. P. et al. The representative concentration pathways: an overview. *Clim. Change* **109**, 5–31 (2011).
77. Jones, C. D. et al. The HadGEM2-ES implementation of CMIP5 centennial simulations. *Geosci. Model Dev.* **4**, 543–570 (2011).
78. 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).
79. Mbow, C. et al. Food Security. In *Climate Change and Land: an IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse Gas Fluxes in Terrestrial Ecosystems* 437–550 (Intergovernmental Panel on Climate Change, 2019).
80. Dai, A. Increasing drought under global warming in observations and models. *Nat. Clim. Change* **3**, 52–58 (2013).

81. Trenberth, K. E. et al. Global warming and changes in drought. *Nat. Clim. Change* **4**, 17–22 (2014).
82. Houghton, R. A. The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. *Tellus B* **51**, 298–313 (1999).
83. Guo, L. B. & Gifford, R. M. Soil carbon stocks and land use change: a meta analysis. *Glob. Change Biol.* **8**, 345–360 (2002).
84. Murty, D., Kirschbaum, M. U. F., Mcmurtrie, R. E. & Mcgilvray, H. Does conversion of forest to agricultural land change soil carbon and nitrogen? a review of the literature. *Glob. Change Biol.* **8**, 105–123 (2002).
85. Don, A., Schumacher, J. & Freibauer, A. Impact of tropical land-use change on soil organic carbon stocks—a meta-analysis. *Glob. Change Biol.* **17**, 1658–1670 (2011).
86. Laganière, J., Angers, D. A. & Paré, D. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. *Glob. Change Biol.* **16**, 439–453 (2010).
87. Carlson, K. M. et al. Greenhouse gas emissions intensity of global croplands. *Nat. Clim. Change* **7**, 63–68 (2017).
88. Edwards-Jones, G. et al. Testing the assertion that 'local food is best': the challenges of an evidence-based approach. *Trends Food Sci. Technol.* **19**, 265–274 (2008).
89. Coley, D., Howard, M. & Winter, M. Food miles: time for a re-think? *Br. Food J.* **113**, 919–934 (2011).
90. Williams, P. et al. A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. *Conserv. Biol.* **10**, 155–174 (1996).
91. Lamoreux, J. F. et al. Global tests of biodiversity concordance and the importance of endemism. *Nature* **440**, 212–214 (2006).
92. Albuquerque, F. & Beier, P. Rarity-weighted richness: a simple and reliable alternative to integer programming and heuristic algorithms for minimum set and maximum coverage problems in conservation planning. *PLoS ONE* **10**, e0119905 (2015).
93. Roberts, C. M. et al. Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science* **295**, 1280–1284 (2002).
94. Beyer, R. & Manica, A. Global and country-level data of the biodiversity footprints of 175 crops and pasture. *Data Brief* **36**, 106982 (2021).
95. Commission, I. S. S. *IUCN Red List Categories and Criteria: Version 3.1*. (Union Internationale pour la Conservation de la Nature et de ses Ressources, 2001).
96. IUCN & NatureServe. *The IUCN Red List of Threatened Species* https://www.iucnredlist.org/. (2022).
97. BirdLife International. *Bird Species Distribution Maps of the World* http://datazone.birdlife.org/species/requestdis. (2022).
98. Waha, K. et al. Multiple cropping systems of the world and the potential for increasing cropping intensity. *Glob. Environ. Change* **64**, 102131 (2020).
99. Johnson, J. A., Runge, C. F., Senauer, B., Foley, J. & Polasky, S. Global agriculture and carbon trade-offs. *Proc. Natl. Acad. Sci. USA* **111**, 12342–12347 (2014).
100. The MathWorks, Inc. *MATLAB and Optimization Toolbox Release 2021b* (The MathWorks, Inc., 2021).
101. Johnson, C. M., Zarin, D. J. & Johnson, A. H. Post-disturbance aboveground biomass accumulation in global secondary forests. *Ecology* **81**, 1395–1401 (2000).
102. Fu, Z. et al. Recovery time and state change of terrestrial carbon cycle after disturbance. *Environ. Res. Lett.* **12**, 104004 (2017).
103. Rappaport, D. I. et al. Quantifying long-term changes in carbon stocks and forest structure from Amazon forest degradation. *Environ. Res. Lett.* **13**, 065013 (2018).
104. Yang, Y., Luo, Y. & Finzi, A. C. Carbon and nitrogen dynamics during forest stand development: a global synthesis. *New Phytol.* **190**, 977–989 (2011).
105. Martin, P. A., Newton, A. C. & Bullock, J. M. Carbon pools recover more quickly than plant biodiversity in tropical secondary forests. *Proc. R. Soc. B Biol. Sci.* **280**, 20132236 (2013).

Acknowledgements
R.M.B. and A.M. were supported by ERC Consolidator Grant 647797 "LocalAdaptation". This work benefited from conversations with América P. Durán, Catherine Tayleur, Sharon E. Brooks, David Coomes, Paul F. Donald, and Fiona J. Sanderson during a separate research project.

Author contributions
R.M.B. designed the project, conducted the analysis, and drafted the manuscript. R.M.B., F.H., P.A.M., A.M., T.R. interpreted the results and revised the manuscript.

Competing interests
The authors declare no competing interests.

Additional information
Supplementary information The online version contains supplementary material available at https://doi.org/10.1038/s43247-022-00360-6.

Correspondence and requests for materials should be addressed to Robert M. Beyer.

Peer review information *Communications Earth & Environment* thanks the anonymous reviewers for their contribution to the peer review of this work. Primary Handling Editors: Clare Davis. Peer reviewer reports are available.

Reprints and permission information is available at http://www.nature.com/reprints

Publisher's note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.