Quantifying the impact of the food we eat on species extinctions

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Abstract

Agriculturally-driven habitat degradation and destruction is the biggest threat to global biodiversity, yet the impacts on extinctions of different types of food and where they are produced and the mitigation potential of different interventions remain poorly quantified. Here we link the LIFE biodiversity metric – a high-resolution global layer describing the marginal impact of land-use on extinctions of ~30K vertebrate species – with food consumption and production data and provenance modelling. Using an opportunity cost framing we discover that the impact of producing one kilogram of different food commodities on species extinction risks varies widely both across and within foods, in many cases by more than an order of magnitude. Despite marked differences in per-capita impacts across countries, there are consistent patterns that could be leveraged for mitigating harm to biodiversity. In particular, we find that animal products and commodities grown in the tropics are generally much more impactful than staple crops and vegetables grown elsewhere. We anticipate the approach and results outlined here could inform decision-making across many levels, from national policies to individual dietary choices.

Introduction

The production of food to meet the needs of people is inextricably reliant on and at the same time the most salient threat to nature and biodiversity (IPBES, 2019). Alongside being one of the most significant contributors to anthropogenic greenhouse gas emissions (Crippa, 2021), food production and in particular terrestrial agriculture is the leading driver of recent and projected future losses of biodiversity (IUCN, 2024; Jaureguiberry, 2022; Tilman, 2020; IPBES, 2019). Although agriculture harms biodiversity in many ways, land-use change and habitat destruction are the most damaging mechanisms, with their impacts projected to increase substantially as global demand for food increases (Winkler, 2021; Tilman, 2017). Indeed, almost a third of the global land surface has been altered for agriculture in the past six decades, and the expansion of arable and grazing land is ongoing, and in some cases accelerating (Potapov, 2022; Williams, 2021; Zalles, 2021).

Halting biodiversity loss and especially extinctions arising from agriculture is thus a key policy concern (Pettorelli, 2021; Scherer, 2020). Effective mitigation hinges on the robust quantification of the impacts of different foods, how these vary spatially, and how they might be reduced by changes in consumption patterns, provenance, or production methods. Spatial layers and summary metrics can inform the efficacy of candidate interventions across diverse scales and actors: from individuals to governments, and from personal dietary choices to national trade policies. Yet to date, there is no agreed-upon method for assessing the biodiversity impacts of food (Damiani, 2023) and despite concerted efforts, attempts to quantify the impacts of agricultural commodities have been limited in spatial and methodological scope. A common approach to capture biodiversity impacts in life-cycle analysis (LCA) involves detailed estimation of specific aspects of food production that might affect biodiversity. But this approach is highly subjective, being dependent on the selection of system boundaries and characterisation factors, with no generally accepted set of criteria or way of translating findings into harmonised results which allow impacts to be compared or summed across regions or products (Bromwich, 2024; Damiani, 2023). Drawbacks of important current land-based biodiversity metrics include subjective prescription of factors based on threat categories (Verones, 2022; Mair 2021) or limitations in species habitat preferences and spatial granularity (Chaudhary, 2015). Green (2019) assessed the impact of soy using state of the art provenance modelling and a pre-cursor to the more comprehensive metric used in this paper (Duran, 2020), but the study was limited to the Brazilian Cerrado. Schwarzmueller (2022) linked consumption to production impacts via trade and the Species Habitat Index (CBD Secretariat, 2021), but because this uses a year-2000 baseline, it only captures historic damage rather than ongoing impacts. Scarborough (2023) links biodiversity and other outcomes on a commodity-specific basis but forgoes any element of spatiality and hence is unable to capture provenance as a lever for mitigation.

To address these shortcomings in quantifying the biodiversity impacts of the food system, here we bring together best-available national data on the consumption and provenance of 140 food types with the LIFE (Land-cover change Impacts on Future Extinctions) metric (Eyres, 2025). LIFE integrates information on species' habitat preferences and ranges in the absence of people from the IUCN Red List (IUCN, 2024) with estimates of potentially restorable habitat derived from Jung (2020) to map (at 1.8km^2 resolution) the marginal impact of land-cover change on the medium-term extinction risk of ~30,000 individually assessed terrestrial vertebrate species – all terrestrial vertebrates for which the Red List has range and habitat preference data, excluding exclusively aquatic or cave dwelling species (Eyres, 2025).

The marginal impact of current food consumption on biodiversity can be viewed as the forgone opportunity to restore biodiversity arising through ongoing agricultural land use. By linking LIFE with spatial crop and pasture distributions (FAO & IIASA, 2024; Klein-Goldewijk, 2017), consumption, production, and trade data (FAO, 2024), and the provenance modelling approach taken by Schwarzmueller et al. (2022), we quantify the opportunity cost to biodiversity of producing or consuming one kilogram of each FAO-aligned food commodity in 174 countries, taking the feed and grazing requirements for animal products into account. We elected to use the FAO database, in which post-production food waste and on-farm losses are embedded in consumption and production data respectively. Our analyses are based on LIFE 'scores'– changes in the expected number of extinctions (ΔE) summed across all ~30K species – which we estimate for one unit of production of each commodity in each country (see Eyres, 2025 for more information on LIFE units). Using these data we first quantify the mass-specific impact on expected extinctions of producing each commodity and how this varies globally. We then examine per-capita consumption impacts of exemplar countries with disparate trade, production, and consumption profiles. Last, we illustrate the power of our approach for quantifying the mitigation potential of interventions by estimating the potential impact of simple dietary changes.

Results

Variation in the opportunity cost of food production

Animal products generally have substantially greater impacts on species' extinction risk than staple vegetal products (Figure 1). This is a result of the inherently inefficient nature of these products (Halpern, 2022; Poore, 2018): producing a unit of animal product requires grazing land and/or cropland for feed production, which when combined with the intrinsic feed conversion efficiency of animals leads to high land use and hence extinction impacts. Ruminant meat, for example, has a weighted global median opportunity cost on species extinctions ~340 times greater than that of grains, by mass. These findings are broadly consistent when expressed instead in terms of functional units of consumption (e.g. impact per serving; Fig S1), and are unsurprising, given that more than three quarters of the human-appropriated land-surface is dedicated to the production of animal products while providing only 17% of global calories (Ritchie, 2019).

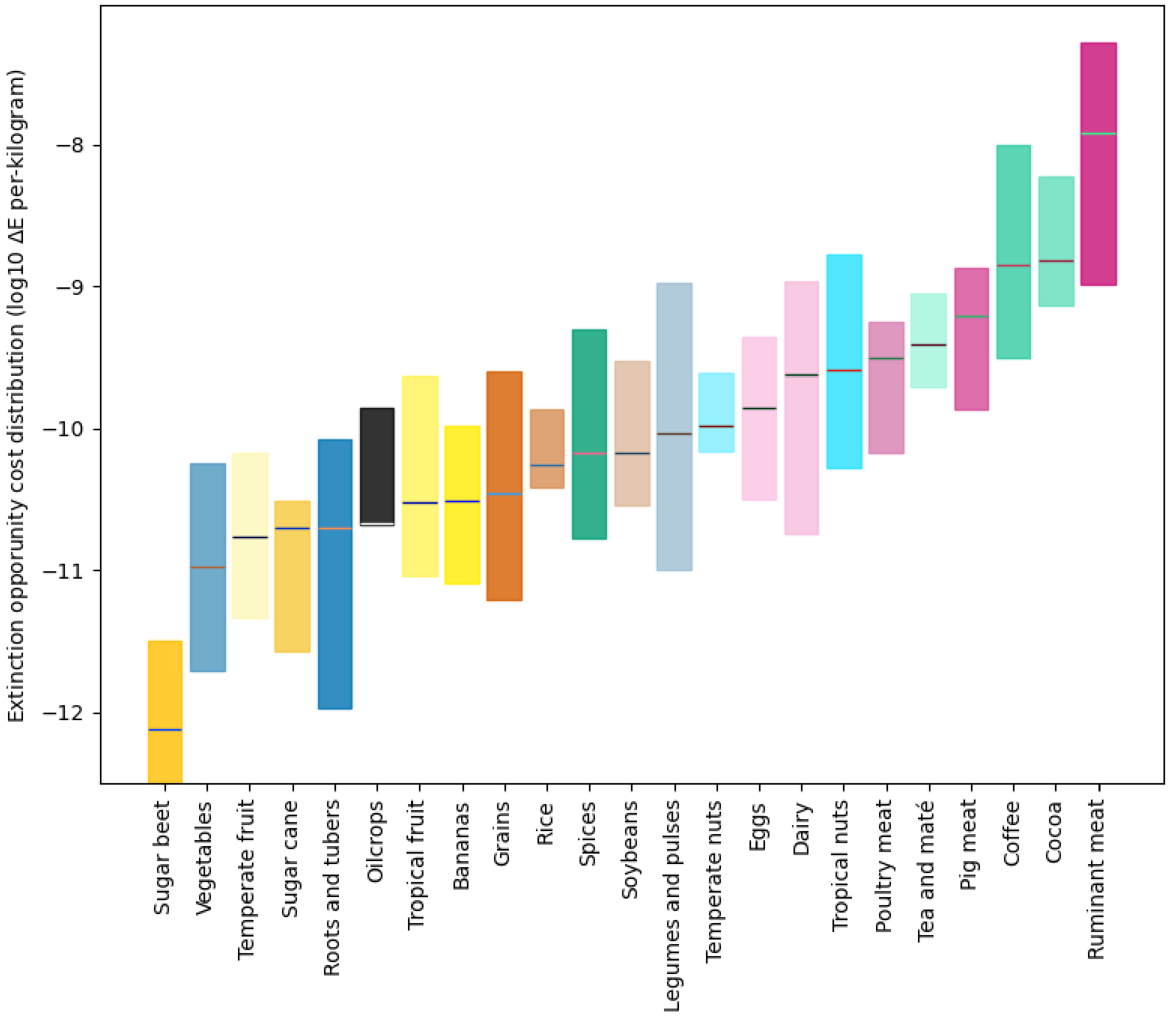

Figure 1. Global variation across and within commodities in the expected extinction impact of producing one kilogram of agricultural commodity or commodity group. The lower and upper boundaries of the boxes represent the production-mass weighted 10th and 90th percentiles respectively. Horizontal lines represent the weighted global median (50th percentile). Where commodities are grouped, the per-kilogram extinction risk values of each constituent commodity are weighted by their contribution (by mass) to the total global production of that group. See Fig S1 for a version of this analysis based on functional units of consumption (e.g. by protein content or serving size) rather than mass.

In contrast staple crops have relatively low impacts on the opportunity cost to species extinctions (ΔE); grains (excluding rice), vegetables, roots, oil crops, and fruits all sit between 10^-10 ΔE/kg and 10^-11 ΔE/kg. Conversely 'luxury' crops (those with little to no calorific benefit but generally commanding a high price) such as coffee, cocoa, tea, and spices are all toward the higher end of the impact distribution per-kilogram – although of course these commodities are typically consumed in relatively modest amounts. Sugar beet has the lowest extinction opportunity cost impact per-kilogram; it is high-yielding compared with most other crops (FAO, 2024), and its production is concentrated in northern and central Europe and northern USA, where the opportunity cost to biodiversity of agricultural land-use is relatively low.

The extinction impact of one kilogram of production varies within most commodities by nearly an order of magnitude, with some varying significantly more. Exceptions to this general pattern include commodities whose production is dominated by a handful of nations. For example, India and China alone produce around 50% of the world's rice with relatively similar per-kilogram impacts, and so there is relatively low variation in the impacts of rice production. The opposite is true of commodity groups that are produced in many different locations – grains, legumes and pulses and dairy products, for example, are produced in significant quantities on every continent and vary in their per-kilogram impacts by almost two orders of magnitude. Likewise impacts vary widely with provenance for coffee. Coffee produced in South America and Sub-Saharan Africa has an impact approximately 10 times greater than that produced in southeast Asia, explained by lower per-area LIFE opportunity costs and higher yields of *robusta* coffee, which dominates Asian production (Bunn, 2015), compared with *arabica*, which dominates production elsewhere.

Generally, commodities that are produced in tropical or sub-tropical regions have higher per-kilogram impacts than those from temperate regions (evident from the distinction between temperate and tropical fruit and nuts, for example). This is unsurprising, given that tropical regions often house exceptional levels of biodiversity and endemism, greatly increasing the impacts of agriculture on global extinction. In the next section we explore the issue of location in more depth by comparing the extinction impacts of food consumption across six countries with widely differing dietary and sourcing profiles.

Across-country differences in the impacts of food consumption and provenance

We selected six countries as examples for comparing consumption impacts - the USA, Japan, and the United Kingdom as Global North nations with high, moderate, and low levels of agricultural self-sufficiency, respectively (Beltran-Peña 2020; DEFRA, 2023a); Brazil as a largely self-sufficient, highly productive tropical country which is a globally important producer of many commodities (soy, corn, sugar cane, cattle meat, fruits, and nuts; Valdes, 2022); Uganda as one of several nations in Sub-Saharan Africa that, whilst not reliant on imports, nevertheless faces widespread malnutrition (The Economist, 2022; Development Initiatives, 2018); and India as a largely self-sufficient country and a major exporter of rice, sugarcane, and tea (Beltran-Peña, 2020).

Our results reveal the very substantial contribution of ruminant meat consumption to the per-capita extinction impact of food consumption in every one of these countries (Figure 2). The impact profiles of the three nations in the global north are driven strongly by their ruminant meat intake, suggesting that even small changes to diet composition would have proportionally large consequences (see next section). In the case of the USA, cattle production is concentrated in southern states, where the extinction impact of a unit area of agricultural land-use is much higher than the north (Eyres, 2025). While the UK and Japan consume comparable quantities of ruminant meat per-capita, the UK's ruminant footprint arises largely from imported cattle and sheep meat from Australia and New Zealand (approximately 25% of consumption) and soy imported from South America. On the other hand, roughly half of all ruminant meat consumed in Japan is imported from high-impact regions (in particular, the USA, Australia and New Zealand, and Mexico) and so on-average the per-kilogram impact of consumption is far greater. Even in India, where approximately one third of the population are lacto-vegetarian and cattle slaughter is banned in many states (Devi 2014), ruminant meats (mainly sheep and goat meat) contribute of 40% of the impact of people's diets on species extinctions.

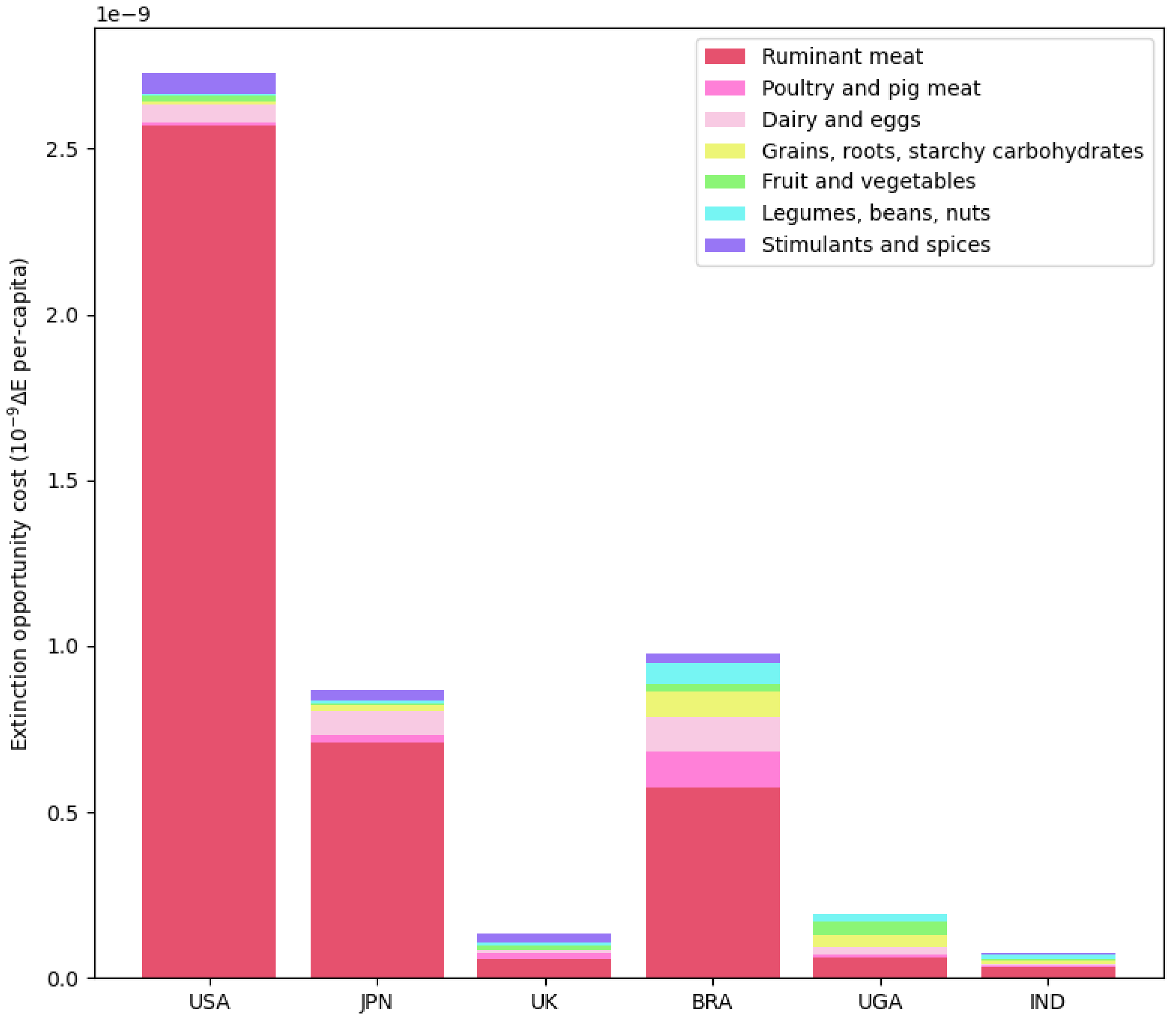

Figure 2. The mean daily specific extinction impact for consumption within the USA, Japan (JPN), the United Kingdom (UK), Brazil (BRA), Uganda (UGA), and India (IND). Note that due to complex and opaque supply chains, sugar impacts have been excluded from these consumption analyses. A version of this figure with commodity detail broken down further can be found in the supplementary materials (Figure S2).

Unpacking the issue of provenance further, Figure 3 shows the proportion of the impact of consuming one kilogram of a commodity that arises from imported versus domestically produced food. The extinction opportunity costs of food consumption within the United Kingdom and Japan are both driven almost entirely by imports. Although these countries produce 60% and 38% of their food domestically (DEFRA, 2023a; MAFF 2023), 95% and 98% of their impacts respectively come from imported commodities. In terms of species extinction risks, for both nations almost all of the impacts of food they consume are accrued in other nations. Given their size and population densities, some reliance on imported food is perhaps inevitable, but this result suggests that sustainably intensifying domestic production and investing in less damaging production overseas should be key policy concerns for these countries. Conversely, current trends in both countries towards promoting low-yielding domestic agriculture (DEFRA, 2023b; USDA, 2021) – which may increase reliance on imports from higher-impact regions – are a cause for concern (Balmford, 2025). These countries might still take steps to reduce the localised impact of domestic agriculture (for example through some regenerative practices) whilst at the same time reducing their exported impacts through trade policies or more efficient use of their available land, for example through dietary shifts or sustainable intensification. In addition, for the UK, the effects of post-Brexit trade deals on the extent to which they increase imports of ruminant products from Australia and New Zealand should also be carefully scrutinised (Roberts, 2024; Bateman, 2023; Fuchs, 2020).

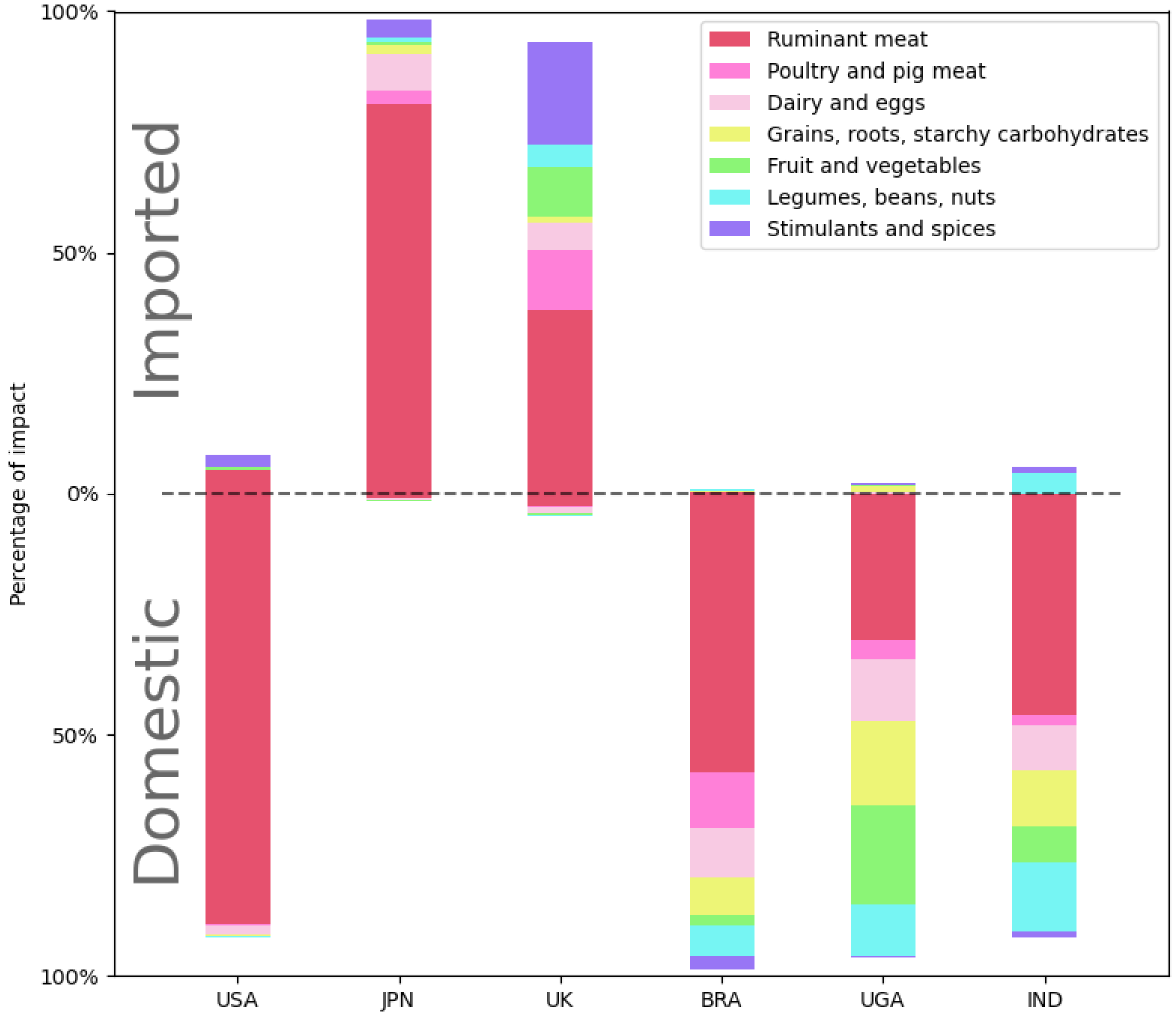

Figure 3. The percentage of consumption-driven extinctions arising from imported (above the dashed line) and domestically produced (below the dashed line) food commodities, estimated for USA, Japan (JPN), the United Kingdom (UK), Brazil (BRA), Uganda (UGA), and India (IND). Once again sugar is excluded from these analyses. A version of this figure with commodity detail broken down further can be found in the supplementary materials (Figure S3).

In contrast, the extinction impact of food consumed in Brazil, Uganda, and India arises very largely from domestic production, with the impact of imports being just 2%, 9%, and 4% respectively. We suggest reducing the impacts of the food people consume in these countries will often best be addressed by sustainably increasing yields and thus increasing production to meet rising demand without clearing remaining areas of natural habitat (Bateman, 2023; Balmford, 2021). The USA also has a low proportion of import-driven impacts (~10%) - it produces the bulk of its food domestically, although it is nevertheless reliant on imports of commodities such as coffee and bananas – but yields in the USA are already relatively high (at least, for vegetal products), so conservation policies may be more effective here if directed at dietary shifts (Williams, 2021), as we explore in the next section.

Dietary impact in the USA

Last, turning from consumption and provenance, we illustrate for the USA how our approach can be used to compare the impact of different diets. We modelled the extinction opportunity cost of baseline individual consumption in the USA using FAOSTAT data (FAO, 2024). We then compare it to the hypothetical impact of three reference diets, firstly, that of the EAT-Lancet planetary health diet as consumed by an individual in the USA (Willett, 2019); this is designed to be consistent with both health and planetary boundaries. Whilst the micronutrient profile of the EAT-Lancet diet as a globally suitable diet has seen some criticism (Beal, 2023), we selected it as a reference diet with a theoretically healthy macronutrient profile that allowed for the consumption of some ruminant meat, where many other references diets preclude ruminant meat in favour of other protein sources. We also constructed simple hypothetical diets which modify the relative consumption of food groups based on vegetarian and plant-based Eatwell guides (PBHP.UK, 2023; NHS, 2022a, 2022b; Sustain, 2018), shown in Table 1. These 'diets' are idealised, and thus in theory contain sufficient macronutrient quantities for an average healthy adult. The total calorific contribution of the modified groups was scaled to meet the calorific value of the products that are replaced. In reality, vegetarian and plant-based consumers are likely to have lower total calorific intakes, so we probably underestimate the mitigation effects of shifting to such diets (Scarborough, 2023). Each diet (including the baseline) also includes calories lost to food wasted at the consumer level; these calories are included in FAOSTAT consumption statistics, and the production impacts of food are blind to its ultimate usage. Stimulants and spices do not meaningfully contribute to calorific intake, so we elected to set them as constant. In each of these examples, food is assumed to be sourced from the same places that the USA currently sources food. This means, for example, that the impact of the vegetarian diet is specific to the USA: the vegetarian diet applied elsewhere might have different impacts. In these examples, in which we are considering the marginal impact of individual consumption patterns, changes in demand are sufficiently small that market price effects would not come into play.

Table 1. Contributions to total calories of different food groups in our baseline US and hypothetical diets as consumed in the USA. Each hypothetical diet has the same overall calorific intake as the baseline. Sugar calories are omitted from the impact calculation as described in the discussion.

	Fruit and vegetables	Legumes, beans, nuts	Grains, roots, starchy carbohydrates	Dairy and eggs	Ruminant meat	Poultry and pig meat	Sugar and other
Baseline	6%	12%	35%	13%	4%	10%	20%
EAT-Lancet planetary health	12%	23%	43%	8%	1%	5%	8%
Vegetarian (Eatwell)	40%	10%	35%	15%	0%	0%	0%
Plant-based (Eatwell)	40%	25%	35%	0%	0%	0%	0%

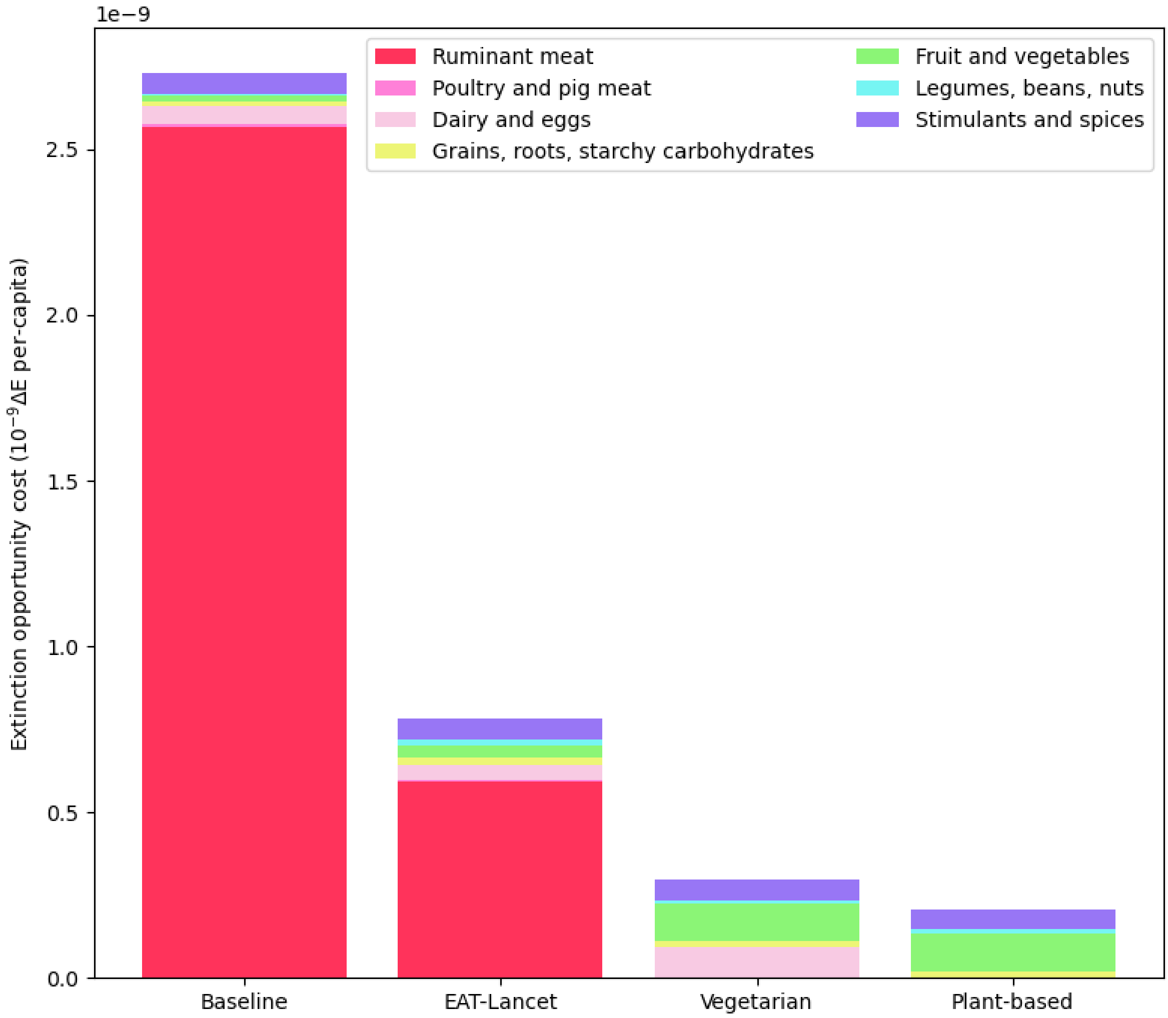

Figure 4. The extinction impacts of average daily per-capita food consumption in the USA for consumption in 2021 ("baseline"; FAO, 2024) and three hypothetical diets (see Table 1). The intake of spices, coffee, cocoa, tea and maté remains constant across diets. The EAT-Lancet diet is the 'Planetary Health Diet', designed to be consistent with health requirements and planetary boundaries (Willett, 2019). The vegetarian and vegan diets are based on 'Eatwell' plates, with constituent commodities consumed in the same ratios that they currently are within each group (PBHP.UK, 2023; NHS, 2022a, 2022b; Sustain, 2018). Each reference diet is scaled to have the same number of calories as the baseline, which in-theory accounts for food waste and overconsumption, likely overestimating their impacts.

Our results underscore the greatly disproportionate extinction impacts of eating animal products, especially ruminant meat. Reducing the prevalence of ruminant meat from the baseline 4% of energy intake down to 1% in the EAT-Lancet diet reduces the total extinction opportunity cost by almost three quarters, despite a necessary increase in the consumption of fruit, vegetables, legumes and nuts. The removal of ruminant meat in the vegetarian and vegan diets leads to a further reduction in extinction opportunity cost of more than half, despite these diets having crop impacts which are higher than the baseline (by 50% and 10% respectively) due to the approximately 4-fold increase in the intake of fruits and vegetables. Given that land use is central to the extinction opportunity costs we have calculated here it is important to note that not all land is necessarily suitable for the same purposes. Monogastric livestock rely almost entirely on feed produced on land that might otherwise be used to produce human-edible commodities. van Zanten (2016) found that it was approximately twice as land-efficient (per unit of human-edible protein) to produce crops for direct human consumption than feed for poultry and pigs. This effect is generally more pronounced for products from ruminant animals (including dairy products), whose feed conversion efficiency is much lower than that of monogastric animals, and whose land-use can include both grazing and feed-producing land. Of course in some systems ruminant animals utilise land that is unsuitable for crop production, but even in these circumstances grazing still imposes an opportunity cost to biodiversity (albeit typically less than arable farming). We thus see that there is very significant scope for reducing the harm to nature by changing US diet patterns, driven largely by the prominence of ruminant meat impacts. Mitigating impacts in a healthy way beyond reducing ruminant consumption is likely to be more complex, and subject to both provenance and methods of production. We also conducted this analysis for the United Kingdom (see supplementary materials, Figure S4), finding that replacing animal-derived calories with entirely plant-based calories leads to just over 50% reduction in impact – notably a similar finding to that of Scarborough (2023), assuming that baseline consumption is comparable with the medium meat-eater group in that study.

Discussion and caveats

The impact on species extinctions of producing food varies dramatically, in many cases 10-100 times per kilogram, both within and between commodities. Unsurprisingly, the per-capita impact of consumption also varies significantly between countries, again by over an order of magnitude. Animal products, in-particular ruminant meat and 'luxury' commodities like coffee and cocoa have consistently high extinction opportunity costs, with median per-kilogram values being over 100 times greater than those of grains and roots. Moreover, together with dairy, pig and poultry products they form the majority of per-capita consumption impacts for all countries assessed in this study. Given that agriculture is and will continue to cause greater harm to biodiversity than any other sector, recognising, understanding, and addressing these patterns will be critical in informing decisions at national, subnational, and individual scales aimed at curbing the global extinction crisis.

The combination of methodologies used in this paper have furnished us with a powerful apparatus for further research, with the capacity to explore the effects of several levers to effect change. Future work might examine the effects of reducing food waste and loss, of changing methods of agricultural production, or of shifting trade policy toward sourcing from less-impactful regions.

Our analyses are subject to three important sets of caveats. First, some important commodities are not included: aquatic foods, oil palm, and in some analyses, sugar. The LIFE metric is driven by the conversion of terrestrial habitat types, and thus any assessment of the impact of fish production or consumption on species extinctions would be limited to the cropland used for aquaculture feed. It might in future be possible to derive estimates of the impacts of habitat damage or bycatch from fisheries, to facilitate comparison of fish with terrestrial products, although terrestrial impacts generally outweigh those in aquatic systems (Halpern, 2022). Importantly, sugar is only partly included in our analyses. Sugar beet and cane have very different per-kilogram extinction opportunity costs (Fig. 1) but supply chain data do not distinguish which of them traded sugar is derived from, so we were unable to include sugar reliably in our consumption analyses. The reported use of palm oil as food appears to be inconsistent. For many countries in the FAOSTAT database the stated 'food' value for palm oil is zero (including for the UK and Canada). The reason for these inconsistencies is unclear but perhaps arises from the numerous non-food uses of palm oil or reporting at the point of processing rather than consumption. Whilst it might be possible to perform an analysis for palm oil via accounting methods or other data sources, it would necessarily require a great deal of inference, and we wished here to avoid treating any one commodity in a bespoke manner to minimise biases and hence omitted palm oil from this study. The national consumption, production, and statistics from the FAO are not always empirical, and often rely on estimation, imputation, or modelling, which may be a contributing factor to anomalous entries such as this one.

Second, the results presented here rely on best-available but still relatively coarse species and agricultural data: the crop and livestock distributions from GAEZ and Gridded Livestock of the World have resolutions of 5 arcminutes (or ~9km at the equator), which fail to capture nuances in more heterogenous production landscapes. Currently, we do not consider the intensity of agricultural land-use; this is particularly relevant for grazed lands, since the ecological implications of intensely grazed and managed pasture differ significantly from extensively grazed rangelands. Our representation of grazing livestock is limited by the quality and availability of data: global, spatial grazing productivity data (i.e. 'yield' for pasture at the commodity level) do not to our knowledge currently exist, but would be invaluable in improving our results. Moreover, it is important to make clear that the opportunity costs presented in our analyses should not be interpreted as meaning there would be instantaneous reductions in extinction risk by restoration of natural habitat. Such restoration may not take place, and it can take many years and varying degrees of human intervention to restore anthropogenic land-cover to natural habitat. Whilst the LIFE metric is a powerful new tool for assessing the impact of land-cover on species extinction risk, it is also fairly coarse in its current iteration – it relies on species distribution and habitat preference data from the IUCN, subject to varying degrees of estimation. The LIFE metric includes all comprehensively assessed taxa, which includes birds, mammals, amphibians, and reptiles, but currently excludes plants and invertebrates. For some less well-documented species only very basic estimated range polygons are available, whilst the ranges of others are known far more precisely. We also acknowledge that the species coverage of the Red List is biased and not necessarily representative of an equal and global picture of biodiversity (Bachman, 2019). Despite these limitations, LIFE has been designed around an incremental pipeline, so, as taxonomically broader, better quality, and higher resolution data become available, LIFE values can easily be recalculated, and analyses updated.

A third caveat is around marginality. The production and trade data can be used to indicate the opportunity cost of producing a unit of production of a commodity, but clearly economic factors such as price effects and substitution mean such values cannot be extrapolated to estimate overall impacts at national or global scales. The LIFE metric itself also of necessity takes a marginal approach: it is based on the understanding that extinction risk changes in a non-linear fashion with the fraction of a species' potential area of habitat which is currently available to it (Eyres et al. 2025). This non-linearity means that the aggregate impact of global food production and consumption cannot be estimated by extrapolating from marginal effects. Moreover, the LIFE metric has significant scope for improvement – in particular through the introduction of habitat quality and degradation: habitats are treated as either suitable or not (which is why LIFE does not yet capture the nuances of land-use and production intensity). Nonetheless, ceteris paribus the per-capita and per-kilogram scale results presented in this paper are valuable and provide a robust approach for further research and development.

Finally, this study is focused on species extinction risk mediated by the use of land for agriculture, which we selected due to its primacy as a driver of historical and current biodiversity loss. However, it is important to note that there are other ways in which the cultivation of land to produce food might harm biodiversity. Examples include the use of fertilisers and pesticides, which may have localised consequences for wildlife such as nutrient run-off or damage to invertebrate populations (Ortiz, 2021), the emission of greenhouse gases which are likely to drive further habitat deterioration through climate change (Segan, 2016), or soil erosion (Pahalvi 2021).

Despite these limitations, we believe that this study represents a significant improvement on previous efforts to link food production and consumption with global extinctions at the national and commodity level. Two findings in particular stand out. Whilst we recognise that malnutrition remains a significant problem in many parts of the world, clearly governments, corporations, individuals, and others in wealthier nations concerned with averting the extinction crisis cannot do so without serious consideration of steps to dramatically reduce consumption of animal products, especially ruminant meat. In some such areas - North America, Australia and New Zealand, and much of Europe – the current contribution of animal products to total caloric intake is as high as 40-48% (FAO, 2023). Significant reductions in the impacts of animal products might be achieved through intensification and hence land-use reduction – but often this comes at the cost of higher localised emissions or resource uses that might affect biodiversity, or a potential reduction in animal welfare (Bartlett, 2024; Broom, 2019). A second clear conclusion is that great care should be taken by richer countries in relatively low-biodiversity part of the world to avoid exacerbating the overseas impacts of their food consumption through land-use and trade policies which will increase the offshoring of production of the food they eat to more biodiverse parts of the world. While these may increase biodiversity domestically it seems very likely that at global scale they will cause net biodiversity harm (Zhong et al., 2024; Balmford et al., 2025). It seems implausible that these two core findings will change as better data become available.

Methods

LIFE estimates the current probability of extinction (relative to that in the absence of anthropogenic land-cover change) for ~30,000 terrestrial vertebrates assessed on the IUCN Red List by relating proportional area of habitat compared to a human-absent scenario via a power law (Eyres et al. 2025). LIFE then calculates the change in their probabilities of extinction following a further land-cover change and thus change in their area of habitat. The metric is calculated by sequentially calculating this change in extinction risk on a pixel-by-pixel basis, then aggregating across all species present in the pixel to get the LIFE score. This produces a global map of summed changes in extinction risk due to marginal land-cover changes. If a species has a very narrow range, then a change in its area of habitat the size of a pixel may well result in a significant portion of its habitat being lost, and hence would contribute strongly to the score of that pixel. Conversely, a species with a very wide range would be less affected by single-pixel changes, and thus contributes a much smaller amount to each of those pixels. All species are treated equally regardless of their threat status, so the LIFE metric scales with species richness, endemism, and habitat loss to date. LIFE consists of two layers, one being a conversion of natural habitat to arable land, and the other being the restoration of agricultural lands (pasture and arable) to potential natural habitat. It is the 'restore' layer that we use in this study.

Our approach allows us to estimate the extinction impact of both the production and consumption of food commodities in terms of the forgone opportunity to reduce extinction risks through habitat restoration because of continued agricultural land-use. In the case of production of vegetal products, impact values are derived from the weighted median national LIFE score (per unit area) for the spatially explicit locations in which that commodity is produced. To derive these estimates, we intersect a partially modified version of the LIFE 'restore' layer (Eyres 2025, see supplementary information) with crop production layers from the Global Agro-Ecological Zones (GAEZ) project (FAO & IIASA, 2024). For animal products, the impacts depend firstly on the LIFE scores associated with grazing land, which we obtain by following Alexander (2017) and using Gridded Livestock of the World data (Gilbert, 2018) to weight cell contributions; and secondly cropland used to produce any feed consumed by the animal. This feed may is often imported, so our framing allows us to capture the impact of South American soy used in the production of animal products in northern Europe, for example.

Note that LIFE is a metric of marginal change – the certainty of results reduces as cell values are aggregated – however, modelling analysis conducted in Eyres (2025) finds that this deviation remains very low (<10%) for land-cover changes up to around 1000km^2, larger than per-kilogram or per-capita analyses would precipitate.

The global distribution of impacts for each commodity is calculated as a production mass-weighted distribution of the median values for each commodity across all countries. This means that, for example, if country *a* produces 100 bananas, and country *b* produces 3 bananas, the weighted median (50th percentile) impact of bananas would be much closer to that of *a* than of *b*. To calculate the consumption-side impacts of countries, we use the method described by Schwarzmueller (2022) to estimate the provenance portfolio of commodities consumed within each country, the distribution of which we use to weight the previously calculated impact values for that commodity in that country. See the supplementary information for more detail on these calculations.

For our simple representation of the impacts of current and potential alternative diets in the USA, we used a baseline mean daily caloric intake of 3911 kcal per capita (FAO, 2024). This daily intake also includes food loss and waste, which for the purposes of this analysis is important, as the impacts of agricultural land use are blind to whether food is eaten by people or not. For the EAT-Lancet, vegetarian, and vegan dietary diets, the total calories available in the baseline were allocated to each of the food groups via the proportions derived from the EAT-Lancet planetary health diet (Willett, 2019), and from vegetarian and vegan 'Eatwell' plates from the Vegetarian Society and Public Health England respectively (PBHP.UK, 2023; Sustain, 2018).

References

Alexander, P., Brown, C., Arneth, A., Finnigan, J., Moran, D. and Rounsevell, M.D., 2017. Losses, inefficiencies and waste in the global food system. Agricultural systems, 153, pp.190-200.

Bachman, S. P., Field, R., Reader, T., Raimondo, D., Donaldson, J., Schatz, G. E., & Lughadha, E. N. (2019). Progress, challenges and opportunities for Red Listing. *Biological Conservation*, 234, 45-55.

Balmford, A., Bateman, I. J., Eyres, A., Swinfield, T., & Ball, T. S. (2025). Sustainable high-yield farming is essential for bending the curve of biodiversity loss. *Philosophical Transactions B*, 380(1917), 20230216.

Balmford, A., 2021. Concentrating vs. spreading our footprint: how to meet humanity's needs at least cost to nature. Journal of Zoology, 315(2), pp.79-109.

Bartlett, H., Zanella, M., Kaori, B., Sabei, L., Araujo, M. S., de Paula, T. M., ... & Balmford, A. (2024). Trade-offs in the externalities of pig production are not inevitable. *Nature Food*, 1-11.

Bateman, I. and Balmford, A., 2023. Current conservation policies risk accelerating biodiversity loss. Nature, 618(7966), pp.671-674.

Beal, T., Ortenzi, F., & Fanzo, J. (2023). Estimated micronutrient shortfalls of the EAT–Lancet planetary health diet. *The Lancet Planetary Health*, 7(3), e233-e237.

Beltran-Peña, A., Rosa, L., & D'Odorico, P., 2020. Global food self-sufficiency in the 21st century under sustainable intensification of agriculture. Environmental Research Letters, 15(9), 095004.

Bromwich, T., White, T., Bouchez, A., Hawkins, I., zu Ermgassen, S., Bull, J. W., ... & Milner-Gulland, E. J. (2024). Navigating uncertainty in LCA-based approaches to biodiversity footprinting (No. th8j6). *Center for Open Science*.

Broom, D.M., 2019. Animal welfare complementing or conflicting with other sustainability issues. *Applied Animal Behaviour Science*, 219, p.104829.

Bunn, C., Läderach, P., Ovalle Rivera, O. and Kirschke, D., 2015. A bitter cup: climate change profile of global production of Arabica and Robusta coffee. Climatic change, 129(1), pp.89-101.

CBD Secretariat, 2021. CBD/WG2020/3/INF/6. 24 August 2021, Montreal. https://www.cbd.int/doc/c/2397/5133/3ce87fa6c735a7bf1cafb905/wg2020-03-inf-06-en.pdf

Chaudhary, A. and Brooks, T.M., 2019. National consumption and global trade impacts on biodiversity. World Development, 121, pp.178-187.

Chaudhary, A., Verones, F., De Baan, L., & Hellweg, S. (2015). Quantifying land use impacts on biodiversity: combining species–area models and vulnerability indicators. *Environmental science & technology,* 49(16), 9987-9995.

Crippa, M., Solazzo, E., Guizzardi, D., Monforti-Ferrario, F., Tubiello, F. N., & Leip, A. J. N. F., 2021. Food systems are responsible for a third of global anthropogenic GHG emissions. Nature Food, 2(3), 198-209.

Damiani, M., Sinkko, T., Caldeira, C., Tosches, D., Robuchon, M. and Sala, S., 2023. Critical review of methods and models for biodiversity impact assessment and their applicability in the LCA context. Environmental Impact Assessment Review, 101, p.107134.

DEFRA, 2023a. Theme 2: UK Food Supply Sources (Updated 2023). United Kingdom Food Security Report 2021. www.gov.uk/government/statistics/united-kingdom-food-security-report-2021/united-kingdom-food-security-report-2021-theme-2-uk-food-supply-sources [accessed 20-03-2024].

DEFRA, 2023b. Environmental Land Management (ELM) update: how government will pay for land-based environment and climate goods and services. www.gov.uk/government/publications/environmental-land-management-update-how-government-will-pay-for-land-based-environment-and-climate-goods-and-services/environmental-land-management-elm-update-how-government-will-pay-for-land-based-environment-and-climate-goods-and-services [Accessed 28-02-2024].

Development Initiatives, 2018. 2018 Global Nutrition Report: Shining a light to spur action on nutrition. Bristol, UK: Development Initiatives.

Devi, S. M., Balachandar, V., Lee, S. I., & Kim, I. H., 2014. An outline of meat consumption in the Indian population-A pilot review. Korean journal for food science of animal resources, 34(4), 507.

Durán, A.P., Green, J.M., West, C.D., Visconti, P., Burgess, N.D., Virah-Sawmy, M. and Balmford, A., 2020. A practical approach to measuring the biodiversity impacts of land conversion. Methods in Ecology and Evolution, 11(8), pp.910-921.

Eyres, A., Ball, T. S., Dales, M., Swinfield, T., Arnell, A., Baisero, D., Durán, A. P., Green, J. M. H., Madhavapeddy, A. & Balmford, A. (2025). LIFE: A metric for mapping the impact of land-cover change on global extinctions. Phil. Trans. R. Soc. B380: 20230327.

FAO and IIASA, 2024. Global Agro Ecological Zones version 4 (GAEZ v4). www.fao.org/gaez [accessed 20-03-2024].

FAO, 2024. FAOSTAT: Food and agriculture data. www.fao.org/faostat/en/#data [accessed 20-03-2024].

FAO. 2023. *Contribution of terrestrial animal source food to healthy diets for improved nutrition and health outcomes – An evidence and policy overview on the state of knowledge and gaps*. Rome, FAO. . https://doi.org/10.4060/cc3912en

Fuchs, R., Brown, C. and Rounsevell, M., 2020. Europe's Green Deal offshores environmental damage to other nations. Nature, 586(7831), pp.671-673.

Gilbert, M., Nicolas, G., Cinardi, G., Van Boeckel, T.P., Vanwambeke, S.O., Wint, G.R. and Robinson, T.P., 2018. Global distribution data for cattle, buffaloes, horses, sheep, goats, pigs, chickens and ducks in 2010. Scientific data, 5(1), pp.1-11.

Green, J.M., Croft, S.A., Durán, A.P., Balmford, A.P., Burgess, N.D., Fick, S., Gardner, T.A., Godar, J., Suavet, C., Virah-Sawmy, M. and Young, L.E., 2019. Linking global drivers of agricultural trade to on-the-ground impacts on biodiversity. Proceedings of the National Academy of Sciences, 116(46), pp.23202-23208.

Halpern, B.S., Frazier, M., Verstaen, J., Rayner, P.E., Clawson, G., Blanchard, J.L., Cottrell, R.S., Froehlich, H.E., Gephart, J.A., Jacobsen, N.S. and Kuempel, C.D., 2022. The environmental footprint of global food production. Nature Sustainability, 5(12), pp.1027-1039.

Hayek, M.N., Harwatt, H., Ripple, W.J. and Mueller, N.D., 2021. The carbon opportunity cost of animal-sourced food production on land. Nature Sustainability, 4(1), pp.21-24.

IPBES, 2019. Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.3553579 [accessed 20-03-2024].

IUCN, 2024. The IUCN Red List of Threatened Species. Version 2023-1. https://www.iucnredlist.org [accessed 20-03-2024].

Jaureguiberry, P., Titeux, N., Wiemers, M., Bowler, D.E., Coscieme, L., Golden, A.S., Guerra, C.A., Jacob, U., Takahashi, Y., Settele, J. and Díaz, S., 2022. The direct drivers of recent global anthropogenic biodiversity loss. Science advances, 8(45), p.eabm9982.

Jung, M., Dahal, P.R., Butchart, S.H., Donald, P.F., De Lamo, X., Lesiv, M., Kapos, V., Rondinini, C. and Visconti, P., 2020. A global map of terrestrial habitat types. Scientific data, 7(1), p.256.

Klein Goldewijk, K., Beusen, A., Doelman, J. and Stehfest, E., 2017. Anthropogenic land use estimates for the Holocene–HYDE 3.2. Earth System Science Data, 9(2), pp.927-953.

Laroche, P.C., Schulp, C.J., Kastner, T. and Verburg, P.H., 2020. Telecoupled environmental impacts of current and alternative Western diets. Global Environmental Change, 62, p.102066.

MAFF, 2023. Summary of the Annual Report on Food, Agriculture and Rural Areas in Japan – Ministry of Agriculture, Forestry, and Fisheries. https://www.maff.go.jp/e/data/publish/attach/pdf/index-224.pdf [accessed 09-04-2024].

Mair, L., Bennun, L. A., Brooks, T. M., Butchart, S. H., Bolam, F. C., Burgess, N. D., ... & McGowan, P. J. (2021). A metric for spatially explicit contributions to science-based species targets. *Nature Ecology & Evolution*, 5(6), 836-844.

NHS, 2022a. The vegetarian diet. www.nhs.uk/live-well/eat-well/how-to-eat-a-balanced-diet/the-vegetarian-diet/ [accessed 20-03-2024].

NHS, 2022b. The vegan diet. www.nhs.uk/live-well/eat-well/how-to-eat-a-balanced-diet/the-vegan-diet/ [accessed 20-03-2024].

Ortiz, A. M. D., Outhwaite, C. L., Dalin, C., & Newbold, T. (2021). A review of the interactions between biodiversity, agriculture, climate change, and international trade: research and policy priorities. *One Earth*, *4*(1), 88-101.

Pahalvi, H. N., Rafiya, L., Rashid, S., Nisar, B., & Kamili, A. N. (2021). Chemical fertilizers and their impact on soil health. *Microbiota and Biofertilizers, Vol 2: Ecofriendly tools for reclamation of degraded soil environs*, 1-20.

PBHP.UK, 2023. The Plant-Based Eatwell Guide. plantbasedhealthprofessionals.com/wp-content/uploads/Plant-Basted-Eatwell-Guide-A4.pdf [accessed 20-03-2024].

Pettorelli, N., Graham, N.A., Seddon, N., Maria da Cunha Bustamante, M., Lowton, M.J., Sutherland, W.J., Koldewey, H.J., Prentice, H.C. and Barlow, J., 2021. Time to integrate global climate change and biodiversity science-policy agendas. Journal of Applied Ecology, 58(11), pp.2384-2393

Poore, J. and Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science, 360(6392), pp.987-992.

Potapov, P., Turubanova, S., Hansen, M. C., Tyukavina, A., Zalles, V., Khan, A., .. & Cortez, J., 2022. Global maps of cropland extent and change show accelerated cropland expansion in the twenty-first century. Nature Food, 3(1), 19-28.

Ritchie, H. and Roser, M., 2019. Land use. Our world in data. ourworldindata.org/land-use [accessed 20-04-2024].

Roberts, E., 2024. UK beef and lamb trade update: exports supported by tighter EU market, while sheep meats imports remain firm. Agriculture and Horticulture Development Board. https://ahdb.org.uk/news/uk-beef-and-lamb-trade-update [accessed 09-04-2024].

Scarborough, P., Clark, M., Cobiac, L., Papier, K., Knuppel, A., Lynch, J., Harrington, R., Key, T. and Springmann, M., 2023. Vegans, vegetarians, fish-eaters and meat-eaters in the UK show discrepant environmental impacts. Nature Food, 4(7), pp.565-574.

Scherer, L., Svenning, J.C., Huang, J., Seymour, C.L., Sandel, B., Mueller, N., Kummu, M., Bekunda, M., Bruelheide, H., Hochman, Z. and Siebert, S., 2020. Global priorities of environmental issues to combat food insecurity and biodiversity loss. Science of the Total Environment, 730, p.139096.

Schwarzmueller, F., & Kastner, T., 2022. Agricultural trade and its impacts on cropland use and the global loss of species habitat. Sustainability Science, 17(6), 2363-2377.

Segan, D. B., Murray, K. A., & Watson, J. E. (2016). A global assessment of current and future biodiversity vulnerability to habitat loss–climate change interactions. *Global Ecology and Conservation*, 5, 12-21.

Sustain, 2018. A new vegetarian Eatwell guide. www.sustainweb.org/news/oct18_vegetarian_eatwell/ [accessed 20-03-2024].

The Economist, 2022. Global Food Security Index 2022. impact.economist.com/sustainability/project/food-security-index/ [accessed 20-03-2024].

Tilman, D. & Williams, D. R., 2020. Preserving global biodiversity requires rapid agricultural improvements. royalsociety.org/news-resources/projects/biodiversity/preserving-global-biodiversity-agricultural-improvements [accessed 20-03-2024].

Tilman, D., Clark, M., Williams, D. R., Kimmel, K., Polasky, S., & Packer, C., 2017. Future threats to biodiversity and pathways to their prevention. Nature, 546(7656), 73-81.

USDA, 2021. MAFF Releases Interim Report on Green Food System Strategy. United States Department of Agriculture, Report JA2021-0048. USDA and Global Agricultural Information Network.

Valdes, C., 2022. Brazil's Momentum as a Global Agricultural Supplier Faces Headwinds. Economic Research Service. U.S. Department of Agriculture. ers.usda.gov/amber-waves/2022/september/brazil-s-momentum-as-a-global-agricultural-supplier-faces-headwinds/ [accessed 20-03-2024].van Zanten, H. H., Mollenhorst, H., Klootwijk, C. W., van Middelaar, C. E., & de Boer, I. J. (2016). Global food supply: land use efficiency of livestock systems. The International Journal of Life Cycle Assessment, 21, 747-758.

Verones, F., Kuipers, K., Núñez, M., Rosa, F., Scherer, L., Marques, A., Michelsen, O., Barbarossa, V., Jaffe, B., Pfister, S. and Dorber, M., 2022. Global extinction probabilities of terrestrial, freshwater, and marine species groups for use in Life Cycle Assessment. Ecological Indicators, 142, p.109204.

Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., ... & Murray, C. J. (2019). Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. *The lancet*, 393(10170), 447-492.

Williams, D.R., Clark, M., Buchanan, G.M., Ficetola, G.F., Rondinini, C. and Tilman, D., 2021. Proactive conservation to prevent habitat losses to agricultural expansion. Nature sustainability, 4(4), pp.314-322.

Winkler, K., Fuchs, R., Rounsevell, M., & Herold, M., 2021. Global land use changes are four times greater than previously estimated. Nature communications, 12(1), 2501.

Zalles, V., Hansen, M. C., Potapov, P. V., Parker, D., Stehman, S. V., Pickens, A. H., .. & Kommareddy, I., 2021. Rapid expansion of human impact on natural land in South America since 1985. Science Advances, 7(14), eabg1620.

Zhong, H., Li, Y., Ding, J. et al. (2024). Global spillover effects of the European Green Deal and plausible mitigation options. *Nature Sustainability* 7, 1501–1511 https://doi.org/10.1038/s41893-024-01428-1

Acknowledgements
TSB was funded by UK Research and Innovation's BBSRC through the Mandala Consortium (grant number BB/V004832/1). AE and MD were supported through grants from the Tezos Foundation and Tarides to the Cambridge Centre for Carbon Credits (grant code NRAG/719). JG was supported by UK Research and Innovation's Global Challenges Research Fund (UKRI GCRF) through the Trade, Development and the Environment Hub project (project number ES/S008160/1).

Data Availability
All modelling and analysis in this manuscript was carried out by TB using the Python programming language (www.python.org/), relying only on public libraries. All relevant code can be found on GitHub (https://github.com/thomasball42/food_v0). Processing of geospatial data (GAEZ and LIFE) was performed with GDAL (gdal.org/en/stable/). Data generated by these analyses are available in a persistent Zenodo repository: *10.5281/zenodo.11161331*. All underlying data are publicly available. The LIFE species extinction opportunity cost data here: *10.5281/zenodo.14188449*, nationally aggregated agricultural production and consumption statistics from the FAO here: *www.fao.org/faostat/en/#data/scl*, and spatial crop production and yield data from GAEZ here: *gaez.fao.org/*.