The impacts of tropical agriculture on biodiversity: a meta-analysis

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Abstract

Biodiversity underpins all food production and strengthens agricultural resilience to crop failure. However, agricultural expansion is the primary driver of biodiversity loss, particularly in the tropics where crop production is increasing and intensifying rapidly to meet a growing global food demand. It is therefore crucial to ask, how do different crops and crop production systems impact biodiversity? Here we show the increasing intensification of tropical agriculture since 1961, along with a sharp rise in harvested area. Using meta-analysis, we find that crop type, rotation time and agricultural intensity, are important determinants of biodiversity assemblages. Perennial tropical crops that are grown in shaded plantations or agroforests (e.g., banana and coffee) support higher alpha-diversity, while those cultivated in unshaded and often homogeneous plantations (e.g., maize, sugarcane, and oil palm) have impoverished biodiversity communities, particularly annual crops. These findings inform our understanding of changes in the ecological contribution of biodiversity to tropical agriculture.

Introduction

Biodiversity underpins all food production and strengthens agricultural resilience to crop failure due the ecological functions that animals provide (Bélanger & Pilling 2019). However, many of the species that perform these functions are disappearing, in part due to the intensification of agricultural systems (Bélanger & Pilling 2019; Foley *et al.* 2005; Figure S1). With human resource demands predicted to double by 2050 (Springmann *et al.* 2018), food security is an increasingly global issue (Rosegrant & Cline 2003). It is therefore important to consider how different crop production systems impact biodiversity communities.

Agricultural expansion is a major driver of habitat loss (Curtis *et al.* 2018; Foley *et al.* 2005; Phalan *et al.* 2013) and one of the most detrimental disturbances to biodiversity assemblages (Gibson *et al.* 2011; Green, 2005; Newbold *et al.* 2014). In the next three decades, to meet a growing food demand, it is expected that up to 10 million km^2 of uncultivated land will be required (Tilman *et al.* 2011), mostly in tropical regions, where land for crop production often comes at the expense of natural habitats (Newbold *et al.* 2015; Tilman *et al.* 2011). Over the last sixty years, the production of different tropical crops has increased by varying degrees (Phalan *et al.* 2013). Much of the food produced in tropical regions is exported internationally. Thus, a large proportion of impacts on tropical biodiversity are remotely incurred by industrialised countries (Chaudhary & Kastner 2016; Green *et al.* 2019; Lenzen *et al.* 2012).

The tropics are extremely biodiverse, with tropical forests alone containing more than two-thirds of the world's terrestrial biodiversity (Giam 2017). The presence of wildlife in ecosystems is important due to the ecological functions and ecosystem services that they provide, such as pollination, seed dispersal, nutrient cycling, energy flow through trophic levels, and pest control (Bélanger & Pilling 2019; Mathieu *et al.* 2005; Valencia-Aguilar *et al.*

2013; Willig *et al.* 2007). Therefore, the promotion of biodiversity in agricultural systems, alongside appropriate management, can provide these benefits in addition to high crop yields (Bélanger & Pilling 2019; Clough *et al.* 2011). In some taxa, particularly birds and bats, agricultural conversion influences the proportions of functional groups. Insectivorous and carnivorous species that provide pest control services often decline, whilst the proportion of frugivores, nectarivores and granivores may increase, depending on food availability within the cropland (Curtis *et al.* 2018; Mtsetfwa *et al.* 2018; Willig *et al.* 2007). With these changes, so do changes occur in the ability of biodiversity communities to perform functions important to food production, particularly pollination and pest control (Bélanger & Pilling 2019).

The magnitude to which agriculture affects biodiversity varies greatly between different crops and agricultural management practices. For example, rice fields are generally less biodiverse than the natural forests or wetlands that they replace (Mathieu *et al.* 2005; Tscharntke *et al.* 2008). However, well-managed rice fields can maintain biodiversity and provide important foraging and breeding grounds for some birds, including rare species (Elphick *et al.*, 2010). Forest conversion for oil palm is the one of the greatest threats to biodiversity in Southeast Asia, characterised by the loss of high conservation value species, and overall, harbouring fewer species than natural forests (Fitzherbert *et al.* 2008; Wilcove & Koh 2010). Crops such as coffee and cacao, when grown in shaded plantations, support a greater diversity than those grown in open monocultures, since they provide arboreal habitats and are more structurally similar to natural forests (Estrada, *et al.* 1997; Zermeño-Hernández *et al.* 2016). In addition to the ecological conditions of croplands, crop rotation times (e.g., perennial or annual), proximity to natural habitats, fragmentation, and connectivity are other major factors that influence the capacity for agricultural areas to support biodiversity (Haddad *et al.* 2015; Şekercioğlu *et al.* 2019).

Despite numerous studies on the impacts of tropical food crops on biodiversity, most are limited to certain crops, taxa, and geographic regions. Therefore, a global analysis to identify and compare the impacts of different tropical food crops on biodiversity assemblages is needed. Here we explore trends in crop production in the tropics between 1961 and 2019, identifying the crops which have expanded the most. We then present a meta-analysis to assess the impacts of tropical agriculture on biodiversity. We investigate whether biodiversity impacts vary between different crops, agricultural intensities, crop rotation times, taxonomic groups, and geographic regions. We expected that agricultural systems that are structurally complex, or similar to natural counterparts, would maintain biodiversity (alpha- and beta-diversity, and assemblage composition) closer to natural levels, whilst crop sites that are homogeneous and structurally simple would harbour impoverished biodiversity assemblages. Quantifying the impacts of different food crops and their cultivation approaches on biodiversity, can inform our understanding of changes to the ecological contribution of biodiversity in tropical agricultural landscapes. In turn, this may inform potential improvements to agricultural practices, and the long-term sustainability of tropical food production.

Materials and Methods

Quantifying tropical crop expansion

In order to quantify crop expansion in the tropics, following Phalan *et al.* (2013), we defined tropical countries as those with at least one-third of their land area situated within the tropics. We used this definition because data on crop cultivation were only available as totals per country for each crop. Although the FAOSTAT dataset has some caveats, as outlined by Phalan *et al.* (2013), it is the best global crop harvesting data available. We used data from FAOSTAT (fao.org/faostat/) on the production and area harvested for all food crops in 115 tropical countries for the years 1961-2019. The harvested area of each of the 137 crops was totalled in each year to compute pan-tropical estimates for each crop's total harvested area per year, and changes in harvested area.

Literature search to quantify agricultural impacts on biodiversity

To quantify the relative impacts of different tropical crops on biodiversity, we first conducted a rapid evidence assessment (REA) to search for peer-reviewed studies measuring biodiversity in both food crops sites and natural reference sites, based on inclusion and exclusion criteria (described below). We used Web of Science to search for studies published prior to 9th June 2020.

After trialling various search strings, we finally conducted our search using the query: TS = (*tropic* AND (agricultur* OR farm* OR plantation* OR crop* OR agroforest*) AND (biodiversity OR wildlife OR *fauna* OR bird* OR mammal* OR bat* OR reptil* OR amphibia* OR insect* OR invertebrate*) AND (abundance* OR *diversit* OR richness* OR communit*)). We restricted search results to journals within the subject areas: ecology, environmental sciences, biodiversity conservation, entomology, forestry, multidisciplinary sciences, agriculture multidisciplinary, zoology, and ornithology. We limited our search to English language studies, with no restrictions on the date of publication. This search returned 3,900 results (Figure S2).

The retrieved studies were subsequently screened for relevance based on the title, abstract, and text of the articles. Studies that met our inclusion criteria: (a) reported vertebrate or macroinvertebrate species richness, density, or abundance within both an area cultivated for food crops and a paired natural landscape of any size with little or no disturbance - yielding us a pairwise comparison for the calculation of effect sizes in the meta-analysis, (b) were located within the tropics, and (c) provided or allowed us to calculate the mean, standard deviation, and sample size, from which we could compute an effect size. We were unable to calculate effect sizes for pairwise comparisons where the standard deviation was zero or the sample size was one, therefore they were excluded. We also excluded pairwise comparisons where food crops were mixed with other anthropogenic land uses (e.g., pasture). Studies that measured biodiversity in aquatic ecosystems within agricultural and reference sites (e.g., streams, irrigated croplands or wetlands) were included.

Our screening process resulted in 194 studies (Figure S2; Table S1) which contributed to our final dataset, amounting to a total of 1,364 pairwise comparisons from 34 countries (Figure S3), spanning five geographic regions: Africa ($N_{studies}$=38, $N_{comparisons}$=281), Asia ($N_{studies}$=55, $N_{comparisons}$=432), Central America ($N_{studies}$=48, $N_{comparisons}$=371), South America ($N_{studies}$=52, $N_{comparisons}$=278), and Oceania ($N_{studies}$=1, $N_{comparisons}$=2). Brazil, Malaysia, Mexico, and Indonesia were the most well-studied countries, comprising more than 50% of all studies (Figure S3). Macroinvertebrates were the most well-represented group ($N_{comparisons}$=613), followed by birds ($N_{comparisons}$=428), mammals ($N_{comparisons}$=248), herpetofauna ($N_{comparisons}$=65), and fish ($N_{comparisons}$=10).

Meta-analysis

To conduct our meta-analysis, for each pairwise comparison, we extracted the mean and standard deviation of the biodiversity data. Where studies reported median values, we used these directly (Higgins *et al.*, 2019). We converted standard error, interquartile ranges and confidence intervals to standard deviation. Data were extracted from tables, figures or the text of each study. For those that presented data graphically, we used WebPlotDigitiser (https://apps.automeris.io/wpd/) to extract the data. Where studies provided multiple pairwise comparisons (e.g., different crops, taxonomic groups, or geographic locations) we recorded each separately. We considered sample sizes as the number of independent sites within a study. For each pairwise comparison, we also recorded the taxonomic group (birds, fish, herpetofauna, invertebrates, or mammals), geographic region (Africa, Asia, Central America, South America, or Oceania), crop, agricultural intensity, and crop rotation time. We divided data into three different categories for agricultural intensity. We define 'shaded plantations' as those characterised by natural or planted shade trees. 'Unshaded plantations' contained crops grown in open land with sparse or no shade trees. Finally, 'plantations with some vegetation' included those which the authors stated had moderate levels of shade trees, understory vegetation, or something to a similar effect. We calculated an effect size for individual crops if there were at least four studies reporting data for that crop. For single crops with fewer than four studies, we grouped these and reported them as 'other tropical crops' (e.g., 'brazil nut', or 'pineapple'). When biodiversity values were provided for sites that did not distinguish between multiple different crops, we reported them as 'mixed tropical crops' (e.g., 'annual crops', or 'sugarcane, pineapple, and banana'). We divided data into four categories for crop rotation time, classified as annual, perennial, mixed, or unknown if the crops were not specified.

To assess the magnitude of the impact of tropical agriculture upon biodiversity, we calculated the Hedges' *g* effect size of the standardized mean difference between agricultural and natural reference sites. Some studies provided multiple pairwise comparisons with a common control (natural reference) site, so we accounted for the potential non-independence of these by nesting them within study, computing a mean for each study (Borenstein *et al.*, 2009). We used a random-effects model, which weighted each comparison by the inverse of within-study variance and between-study variance (Borenstein *et al.* 2009; Koricheva *et al.* 2013).

In cases where data were extracted from figures, and the variance was so small that it was indiscernible from the mean, we recorded the variance as 0.001 so that an effect size could be computed. The effect direction was reported as positive for cases where the biodiversity value was more favourable in the reference site than the agricultural site, and negative for cases where the biodiversity value was less favourable in the reference site than agricultural site. In cases where there was a greater abundance and/or diversity of invasive species in the agricultural site, this was deemed negative. Therefore, a negative effect size indicates that the agricultural site had an impoverished biodiversity community, and a positive effect size indicates that the agricultural site supported higher levels of biodiversity than the reference site. We considered effect sizes to be significant if the confidence interval did not overlap zero (Koricheva *et al.* 2013).

We calculated the mean effect size for the overall dataset, and the mean effect size for each of the moderator variables (crop type, agricultural intensity, crop rotation time, taxonomic group, geographic region, and biodiversity metric – richness or abundance). Where fewer than four studies were used for each category, they contributed to the calculation of the overall effect size, but were otherwise not displayed separately in Figure 2.

To test for publication bias, we followed Nakagawa *et al.* (2017). As such, we plotted funnel plots of standard error and precision for Hedges' *g* (Figure S4), and calculated the Classic Fail-safe N. The Classic Fail-safe N was 5,151, which means that we would need to locate and include 5,151 null studies in order to overturn the significance of our results (Borenstein *et al.* 2009; Koricheva *et al.* 2013). The symmetry of the funnel plots and high Fail-safe N suggest that publication bias is minimal or non-existent in our dataset. We conducted all meta-analyses in the Comprehensive Meta-analysis v3.0 software (Borenstein *et al.* 2013).

Results

Crop expansion

In 2019, tropical croplands covered at least 7.21 million km^2 (Figure 1), equivalent to 5.5% of global land area (i.e., approximately the size of the Australian continent). The top ten crops by harvested area in tropical countries in 2019 were rice, maize, soybeans, wheat, sorghum, beans, millet, oil palm, cassava, and groundnuts, which together accounted for two-thirds (67%) of total harvested area (Figure 1a). Across the tropics, the total area of cultivated land has more than doubled between 1961 and 2019 (Figure 1b), with production increasing at a greater rate than cultivation area (Figure S1), showing the overall increasing intensification of tropical food production.

Between 1961 and 2019, soybeans were the most rapidly expanding crop both in terms of absolute area, increasing by 0.54 million km^2 (Figure 1c), and percentage, increasing by 4,597% (Figure 1d). After soybeans, maize, rice, and oil palm expanded most in absolute area, while oil palm, cow peas, and sugarcane increased by the greatest percentage.

Biodiversity impacts

Overall, food crop cultivation reduced biodiversity in tropical regions, although the direction and magnitude of the impact depends on the crop, agricultural intensity, rotation time, taxonomic group, and geographic region. The overall effect of tropical agriculture upon biodiversity is negative and significantly different from zero (Figure 2; mean Hedges' g [± 95% CI] =-0.59 [-0.67 to -0.51], p<0.001; Table S2).

Exploring the data by crops, we found that the biodiversity assemblages were significantly impoverished in maize, oil palm, sugarcane, 'other tropical crops', tea, rice, cacao, and 'mixed tropical crop' sites, compared with natural habitats (Figure 2a; Table S2). Biodiversity responses were in general negative but not significant in citrus, allspice, and coffee plantations, while banana and mixed cacao and coffee plantations sometimes supported greater levels of biodiversity, providing strong evidence that many tropical crops can be cultivated with relatively minimal impacts to biodiversity assemblages. Examining our results by agricultural intensity, we found that all categories had negative effects on biodiversity, while shaded and unshaded plantations showed a significant difference from zero (Figure 2b; Table S2). There was a large variation in the effect size of plantations with some natural vegetation, with some studies demonstrating that these crop management strategies can support biodiversity similar to reference sites, while others do not. We find that crop rotation time is another important determinant of impacts, with annual (temporary) crops resulting in more impoverished communities compared with perennial crops that have longer rotation periods (Figure 2c). Exploring the results by taxonomic group, we found that overall, bird, herpetofauna, and invertebrate assemblages were significantly impoverished in agricultural treatments, while mammal responses were mostly negative but not significant (Figure 2d; Table S2). Examining our results by geographic region, we found there was a significantly negative effect of agriculture on biodiversity in all tropical regions (Figure 2e; Table S2). Finally, comparing by biodiversity metric, effect sizes for both richness and abundance are negative and significantly different from zero, with richness showing the strongest response to agriculture (Table S2).

Discussion

Agriculture has been identified as one of the largest anthropogenic disturbances to biodiversity (Gibson *et al.* 2011). Previous research has reported the expansion of tropical crops, identifying trends in production and land conversion (Curtis *et al.* 2018; Phalan *et al.* 2013). There have also been numerous studies highlighting the adverse impacts of agriculture upon animal assemblages including alpha- and beta-diversity, as well as the varying responses of different taxa (Arenas-Clavijo & Armbrecht 2019; Chapman *et al.* 2019; Ocampo-Ariza *et al.* 2019; Ramamonjisoa *et al.* 2020). In general, it is known that crop systems support widespread, common, and generalist species, while more specialist, disturbance-sensitive, endemic, and threatened species are likely to be absent (Gallmetzer & Schulze 2015; Şekercioğlu 2012). However, our meta-analysis is the first to compare the magnitude and direction of the impacts of different food crops across the whole of the tropics, and demonstrates that agricultural conversion across a range of ecosystems has an effect on biodiversity, depending on the type of crop and intensity of land use. We also demonstrate the sheer scale of tropical crop expansion (Figure 1), and that intensification is increasing year-on-year due to production increases out-accelerating area increases (Figure S1). Intensification is particularly concerning because there is increasing evidence that croplands with impoverished biodiversity communities can produce lower yields, and require higher levels of chemical inputs (Bélanger & Pilling 2019). This is therefore due in part to intensification undermining the pollination and other services provided by biodiversity, because of the impact intensification has on biodiversity assemblages as illustrated herein.

Overall, our results show that agricultural conversion negatively affects tropical biodiversity. Unshaded plantations result in the most impoverished biodiversity communities, however, the effects varied greatly depending on the crop species. Impoverished biodiversity in agricultural sites could be associated with reduced structural complexity, the removal of understory vegetation, destructive land management practices (Bohada-Murillo *et al.* 2020; Castaño-Villa *et al.* 2014; Zermeño-Hernández *et al.* 2016), use of agrochemicals (Smith *et al.*, 2016; Zermeño-Hernández *et al.* 2016), reduced food availability (Mang & Brodie, 2015), changes in soil quality and communities (Franco *et al.* 2019; Smith *et al.* 2016), and an increase in pest or invasive species (Paini *et al.* 2016; Suzán *et al.* 2008). Crops grown in systems that are structurally complex or similar to natural ecosystems, such as agroforests (e.g., some cacao, coffee, and banana plantations), harbour biodiversity closer to natural levels (Estrada *et al.* 1997; Zermeño-Hernández *et al.* 2016). The substantially smaller impact of shaded plantations than unshaded plantations highlights the potential for improving agricultural practices to reduce biodiversity loss. The wide confidence intervals for plantations with some vegetation could be due to fewer studies, or variation in the capacity for different types of vegetation to support biodiversity (e.g., plantations with scattered shade trees provide a different habitat from those with an intact understory). We also show that crops that are harvested on an annual basis, such as maize, sugarcane and rice, result in the greater biodiversity impacts when compared with crops that have longer rotation periods, such as coffee, tea, citrus, allspice, cacao and banana. However, oil palm (a perennial with ~25-year rotation cycles) which has significant impacts on biodiversity, does not follow this trend. This may be due to oil palm often being planted within large-scale, high-yield monocultures, but also the fact that 80% of oil palm is produced in the highly biodiverse Southeast Asia biodiversity hotspot, much of this replacing tropical forests (Fitzherbert *et al.* 2008).

While our findings provide insights into the impacts of different crops on biodiversity, there is a distinct lack of data for most crops. Of the top ten crops in terms of harvested area in the tropics, our REA only returned enough studies for rice, maize, and oil palm to be analysed individually. The large negative effect size of the other tropical crops category highlights the need for more research on understudied crops to identify their individual impacts. Despite soybeans being the most rapidly expanding crop in recent decades, we only found one study reporting biodiversity in soybean sites with data that met our criteria for the meta-analysis (Moura *et al.* 2013). Soybean expansion is well documented, particularly in Brazil. It has been responsible for large areas of deforestation of the Amazon and habitat loss in the globally important Cerrado biome (Kastens *et al.* 2017; Soterroni *et al.* 2019), though there is evidence that soybeans usually replace previously deforested land (Barona *et al.*, 2010). Nonetheless, the biodiversity impacts of soybeans are understudied compared with other tropical crops such as cacao, coffee, and oil palm, which account for considerably less cultivated land area (fao.org/faostat/). Many lesser-known crops are grown by small-scale subsistence farmers and are less likely to gain attention from conservationists than industrially produced crops that are traded internationally (Balmford *et al.* 2012).

In our analysis, birds showed the greatest negative response to agricultural conversion while mammals displayed the most tolerance, reflecting the findings of Gibson *et al.* (2011). It has been suggested that large-bodied mammals are often extirpated due to habitat loss, whereas small nonflying mammal and bat populations can thrive in agricultural habitats (Daily *et al.* 2003; Gibson *et al.* 2011; Wearn *et al.* 2017). Bird feeding guilds show drastic differences between agricultural and natural habitats. Frugivores, granivores, and generalist species abundance and diversity generally increase in cropland, due to food abundance, while insectivorous and carnivorous species decline, though there are exceptions (Mang & Brodie 2015; Tscharntke *et al.* 2008). Likewise, bird specialist species are more sensitive than generalists as well as migratory birds (Philpott *et al.* 2008; Şekercioğlu 2012).

Our REA also showed some geographic bias in the papers we found. In the Neotropics, research appears concentrated in Brazil, Mexico, Costa Rica, Colombia, and Peru, and in Asia the majority of studies come from Malaysia, Indonesia, and India (Figure S3). Most other countries provided few or no studies; research in tropical Oceania is particularly limited.

In many studies used in our meta-analysis, reference sites were fragmented landscapes. Evidence suggests that due to fragmentation, 70% of global forest lies within 1 km of the forest edge (Haddad *et al.* 2015). Agricultural land can have adverse impacts upon biodiversity at considerable distances into natural habitats (Hurst *et al.* 2013; Scriven *et al.* 2018). Therefore, biodiversity levels in reference sites would be influenced by factors such as proximity to agricultural land, patch size, connectivity, edge effects, and the intensity of land use in the surrounding matrix. Consequently, the true effects of agricultural conversion are likely to be greater than our estimates, when considering the additional impacts of fragmentation (Haddad *et al.* 2015). Nonetheless, the relative differences between the impacts of different crops are likely to remain largely the same.

The results of this meta-analysis detail the negative impact that agriculture has on biodiversity across the tropics, increasing in magnitude as agricultural intensity increases. Crops grown in structurally complex systems that mimic, to a degree, natural habitats are able to maintain biodiversity, whilst crops that are usually grown in often annual, homogeneous systems, such as maize, oil palm and sugarcane, have significant negative effects. Responses of different taxonomic groups vary greatly. Bias towards certain crops and limited data of others highlights a major gap in knowledge.

It is crucial to understand the consequences of food cultivation on biodiversity in the tropics. This can help to identify areas for improvements to agricultural practices, and consequently, minimise further adverse impacts, but also, potentially influence consumer choice. The knowledge gained from this study could also be incorporated into the modelling of future agricultural expansion scenarios (e.g., Chaplin-Kramer *et al.* 2015), helping to identify areas for crop expansion with minimal adverse impacts on biodiversity. Most of all though, our findings may serve as a warning sign for agricultural systems that rely on the ecological functions provided by dispersers, pollinators, and pest predators to maximise their yields. This is crucial, because with an ever-increasing global food demand, yield deficits could result in further expansion to the area footprint of tropical agriculture.

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Figures

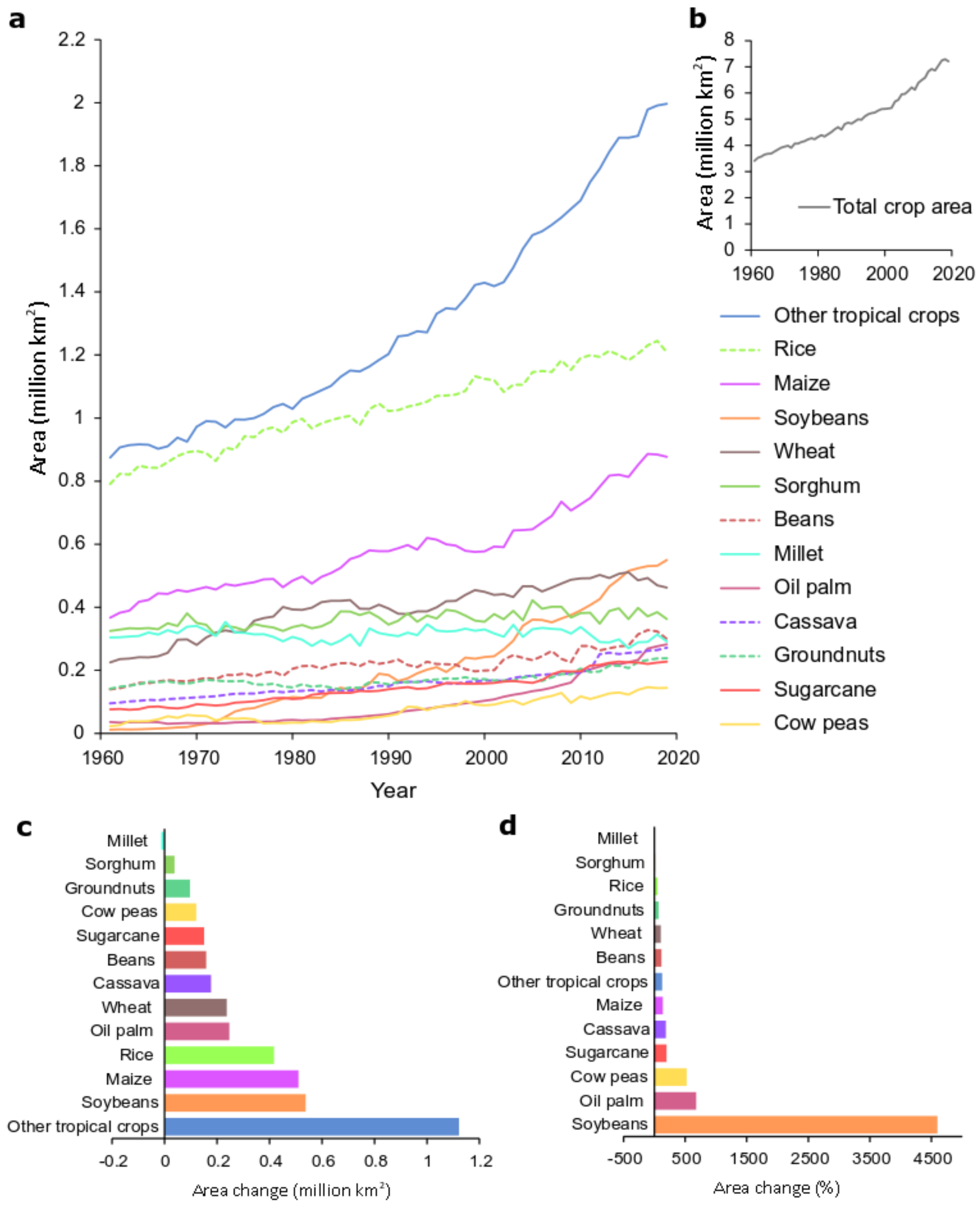

Figure 1. Changes in harvested area of tropical crops from 1961-2019. (a) Harvested area of individual food crops. (b) Total harvested area of food crops. (c) Increase in harvested area of food crops by absolute area and (d) by percentage, in tropical countries from 1961-2019. The top ten tropical crops by area in 2019 are shown. Additionally, sugarcane and cow peas, which were in the top ten by area increase, are also shown. The harvested areas of ‘other tropical crops’ were combined. Data from: FAOSTAT.

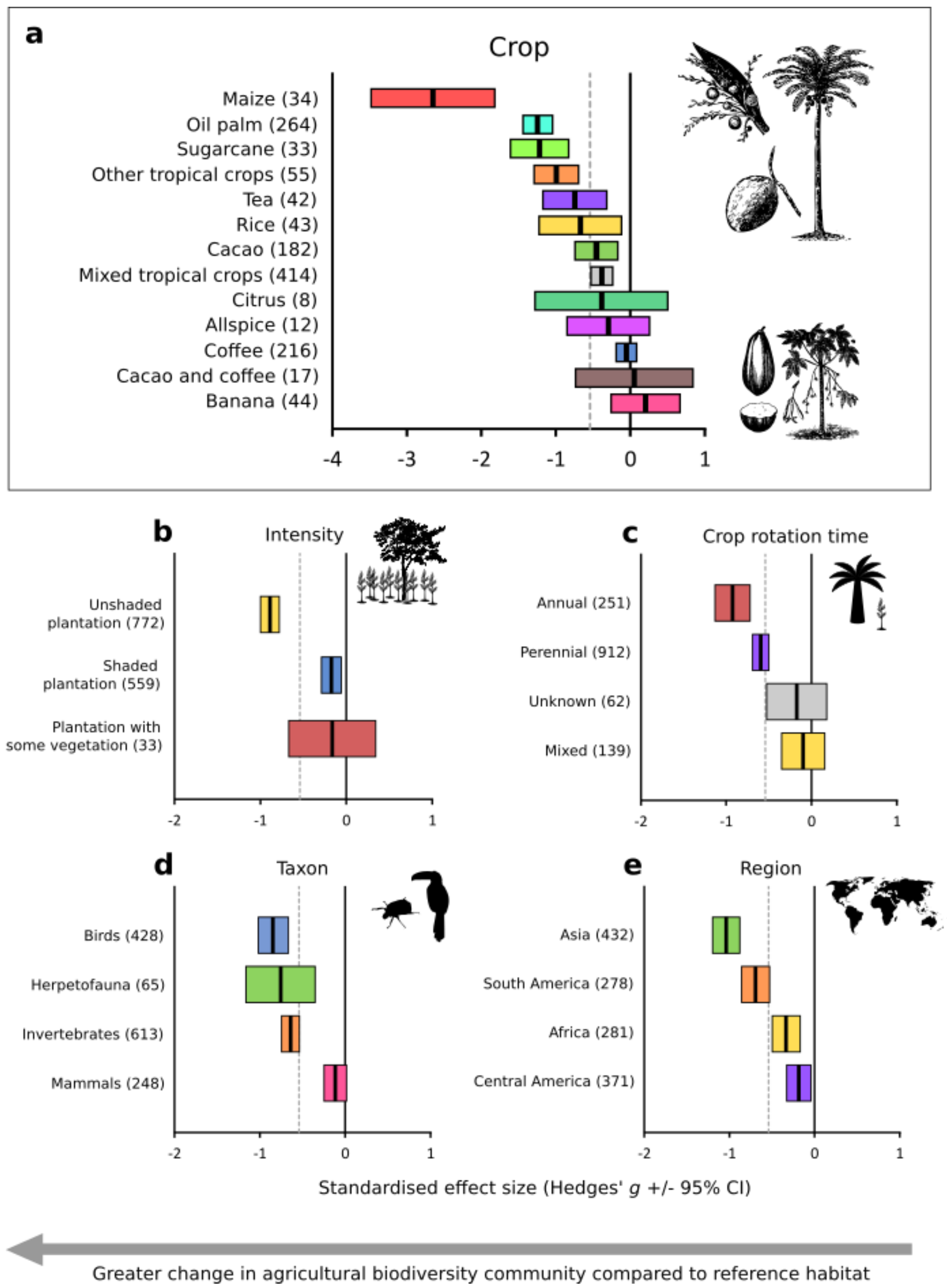

Figure 2. Effect sizes of agricultural impacts on biodiversity by (a) crop, (b) intensity, (c) crop rotation time, (d) taxonomic group (omitting fish $N_{studies}$=3), and (e) geographic region (omitting Oceania $N_{studies}$=1). The number of pairwise comparisons between agricultural and reference sites per category is reported in parentheses. The black vertical lines show the mean standardised effect size (Hedges' *g*), and 95% CI are indicated by the width of the boxes. Effect sizes are significant if the confidence intervals do not overlap zero. The tall vertical black lines and grey dashed lines represent an effect size of zero and mean overall effect size respectively.

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Conflicting interests

The authors declare no conflicting interests.