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## RESEARCH ARTICLE

# The impacts of tropical agriculture on biodiversity: A meta-analysis

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Email: [josephoakley94@gmail.com](mailto:josephoakley94@gmail.com)**Handling Editor:** Maria Fernanda Peñaflor**Abstract**

1. 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?
2. We first use the FAO database of harvested crop area to explore temporal changes in crop area and intensification across the entire tropical realm. We show that the harvested area of tropical crops has more than doubled since 1961, with ever-increasing intensification. The harvested area in 2019 was 7.21 million km<sup>2</sup>, equivalent to 5.5% of global ice-free land area, or 11.5% of land area in the tropics.
3. Second, we conducted a meta-analysis of 194 studies and 1364 pairwise comparisons to assess the impact of tropical agriculture on biodiversity, comparing biodiversity values in food crop sites versus natural reference habitats.
4. Our meta-analysis shows that crop type, rotation time and level of shading are important determinants of biodiversity assemblages. Perennial tropical crops that are grown in shaded plantations or agroforests (e.g. banana and coffee) support higher biodiversity, while crops cultivated in unshaded and often homogeneous croplands (e.g. maize, sugarcane and oil palm), and particularly annual crops, have impoverished biodiversity communities.
5. *Policy implications.* Our findings highlight the increasing agricultural expansion and intensification over the last 60 years and inform our understanding of how different crops and crop production systems impact biodiversity. Furthermore, they provide insight into the long-term sustainability of tropical food production and may serve as a warning sign for agricultural systems that rely on the ecosystem services provided by biodiversity.

**KEYWORDS**

agriculture, biodiversity, crops, ecosystem services, food systems, intensification, meta-analysis, tropical

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## 1 | INTRODUCTION

Biodiversity underpins all food production and strengthens agricultural resilience to crop failure due to the ecological functions that animals provide (Bélanger & Pilling, 2019; Foley et al., 2005; Figure S1). With the demand for food predicted to double by 2050 from 2010 levels (Springmann et al., 2018), food security is an increasingly important 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). Over the last 60 years, the production of different tropical crops has increased by varying degrees (Phalan et al., 2013). In the next three decades, to meet a growing demand for food, it is predicted that agricultural expansion will continue to increase. This is expected to occur mostly in poorer countries throughout the tropics, where land for crop production often comes at the expense of natural habitats (Tilman et al., 2011).

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 wild animals 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 affects the relative composition of functional groups. Insectivorous and carnivorous species that provide pest control services often decline, while the proportion of frugivores, nectarivores and granivores may increase, depending on food availability within the cropland (Mtsetfwa et al., 2018; Tschardt et al., 2008; Willig et al., 2007). These changes affect 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; Tschardt 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 geographical regions. Therefore, a global analysis to identify and compare the impacts of individual 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 animal diversity. We investigate whether biodiversity impacts vary between different crops, shading levels, crop rotation times, taxonomic groups and geographical regions. We expected that agricultural systems that are structurally complex, or similar to natural counterparts (e.g. shaded crops), would maintain biodiversity closer to natural levels, while crop sites that are homogeneous and structurally simple (e.g. unshaded crops) would harbour impoverished biodiversity assemblages. Furthermore, we hypothesised that perennial crops such as coffee, cacao and banana would better support biodiversity than annual crops such as maize and sugarcane. We also expected to see differences in agricultural impacts between different geographical regions, due to the variation in crop species and agricultural practices in different parts of the world. 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.

## 2 | MATERIALS AND METHODS

### 2.1 | Quantifying tropical crop expansion

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 harvesting were only available as totals per country for each crop. We used data from FAOSTAT (2021) 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.

While the FAO provides some of the best available data on crop harvesting, it must be acknowledged that it has some limitations, so caution must be taken when interpreting the data. Where annual

crops are harvested in rotation on the same land multiple times a year, they are all counted towards crop harvesting data, so may lead to overestimations of the true harvested land area. Conversely, underestimations may also occur since crop harvesting data excludes areas where crops were planted but not harvested due to natural calamities or economic reasons. Additionally, there are discrepancies in the reporting of data between countries, with some reporting the entire cultivated area of perennial crops, while others report only the productive area (FAO, 2011). Therefore, there may be some discrepancies between the reported and true harvested areas of crops, but the results are likely to be indicative of trends.

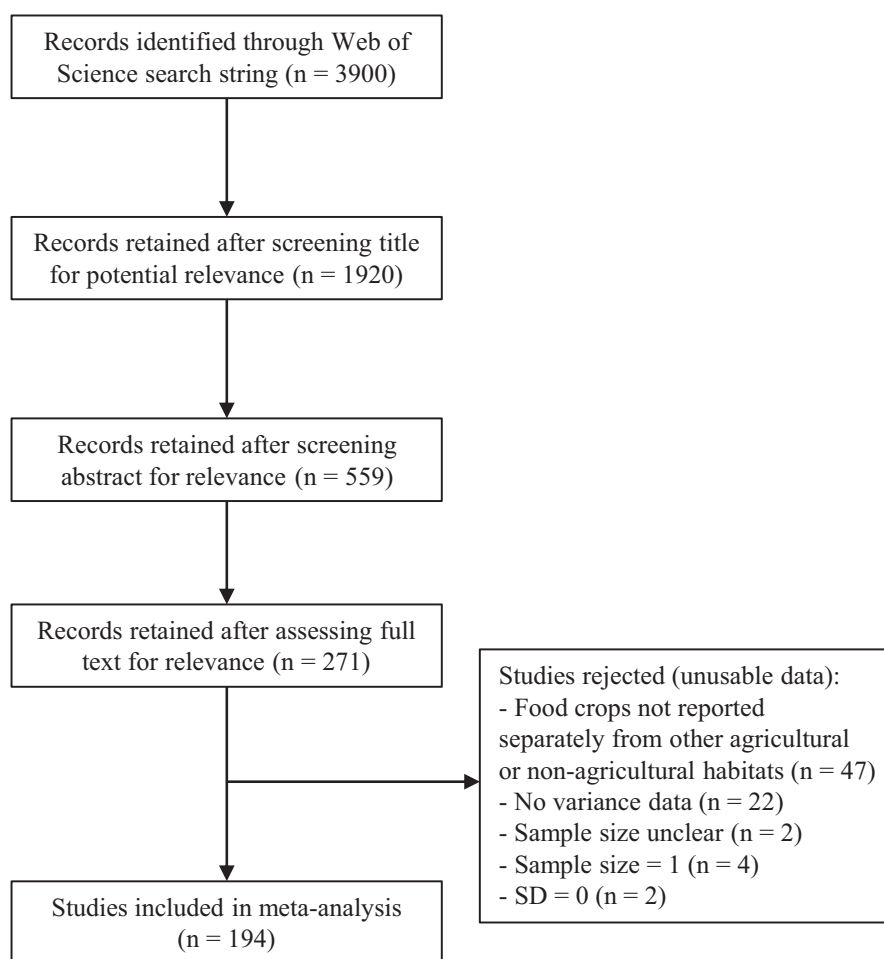
## 2.2 | Literature search to quantify agricultural impacts on biodiversity

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 crop 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 3900 results (Figure 1).

The lead author (JLO) subsequently screened the retrieved studies for relevance based on the title, abstract and full text of the articles. A conservative approach was taken in the inclusion of papers during the title and abstract screening to reduce errors of omission. Both authors screened a subset of studies independently and assessed the level of agreement using Cohen's kappa statistic (Cohen, 1960), scoring 0.8, to ensure the inclusion criteria were applied consistently. 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



**FIGURE 1** Prisma diagram of the number of studies included during each filtering stage of the rapid evidence assessment. See methods for inclusion criteria.

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 1; Table S1) which contributed to our final dataset, amounting to a total of 1364 pairwise comparisons for 13 crop categories (Table S2), from 34 countries (Figure S2), spanning five geographical regions (Tables S1 and S2): Africa ( $N_{\text{studies}} = 38$ ,  $N_{\text{comparisons}} = 281$ ), Asia ( $N_{\text{studies}} = 55$ ,  $N_{\text{comparisons}} = 432$ ), Central America ( $N_{\text{studies}} = 48$ ,  $N_{\text{comparisons}} = 371$ ), South America ( $N_{\text{studies}} = 52$ ,  $N_{\text{comparisons}} = 278$ ) and Oceania ( $N_{\text{studies}} = 1$ ,  $N_{\text{comparisons}} = 2$ ). Brazil, Malaysia, Mexico and Indonesia were the most well-studied countries, comprising more than 50% of all studies (Figure S2). Macroinvertebrates were the most well-represented group ( $N_{\text{comparisons}} = 613$ ), followed by birds ( $N_{\text{comparisons}} = 428$ ), mammals ( $N_{\text{comparisons}} = 248$ ), herpetofauna ( $N_{\text{comparisons}} = 65$ ) and fish ( $N_{\text{comparisons}} = 10$ ).

### 2.3 | Data extraction and meta-analysis

The data that met the inclusion criteria were extracted by JLO with a second opinion from JEB where necessary. 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 geographical locations), we recorded each separately. For those that provided separate pairwise comparisons for food crops and other agricultural or anthropogenic habitats with reference sites, we only extracted the food crop comparison. 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), geographical region (Africa, Asia, Central America, South America or Oceania), crop rotation time and level of shading. We divided shading into three categories: 'Shaded crops' were those characterised by natural or planted shade trees above the crop in question. This was most common in cacao and coffee; 'Unshaded crops' contained crops grown in open land with sparse or no shade trees. In the case of large crops, for example, oil palm, where the mature trees create shade we considered these unshaded crops because they were not being shaded by a second vegetation type; finally, 'Crops with some vegetation' included those which the authors stated had moderate levels of shade trees, understorey vegetation or something to a similar effect. These were classified to the best of our ability with the

information available in the papers. We calculated an effect size for individual crops if there were at least four studies reporting data for that crop. For single crops represented by fewer than four studies, we grouped these and reported them as 'all 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 standardised 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, shading, crop rotation time, taxonomic group, geographical 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 3.

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 S3), and calculated the Classic Fail-safe  $N$ . The Classic Fail-safe  $N$  was 5151, which means that we would need to locate and include 5151 null studies 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). Our study did not require ethical approval.

### 3 | RESULTS

#### 3.1 | Crop expansion

According to the FAO data, the summed harvested area of crops in tropical countries in 2019 was 7.21 million km<sup>2</sup> (Figure 2), equivalent to 5.5% of global ice-free land area, or 11.5% of land area in the tropics (i.e. approximately equivalent to the size of the Australian continent). The top 10 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 2a). Across the tropics, the total area of harvested land has more than doubled between 1961 and 2019 (Figure 2b). The mean annual rate of expansion has accelerated in the past two decades, almost doubling in 2000–2019 compared to that of 1980–1999. Production has increased at a greater rate than harvested 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<sup>2</sup> (Figure 2c), and percentage, increasing by 4597% (Figure 2d). After soybeans, maize, rice and oil palm expanded most in absolute area, while oil palm, cow peas and sugarcane increased by the greatest percentage.

#### 3.2 | Biodiversity impacts

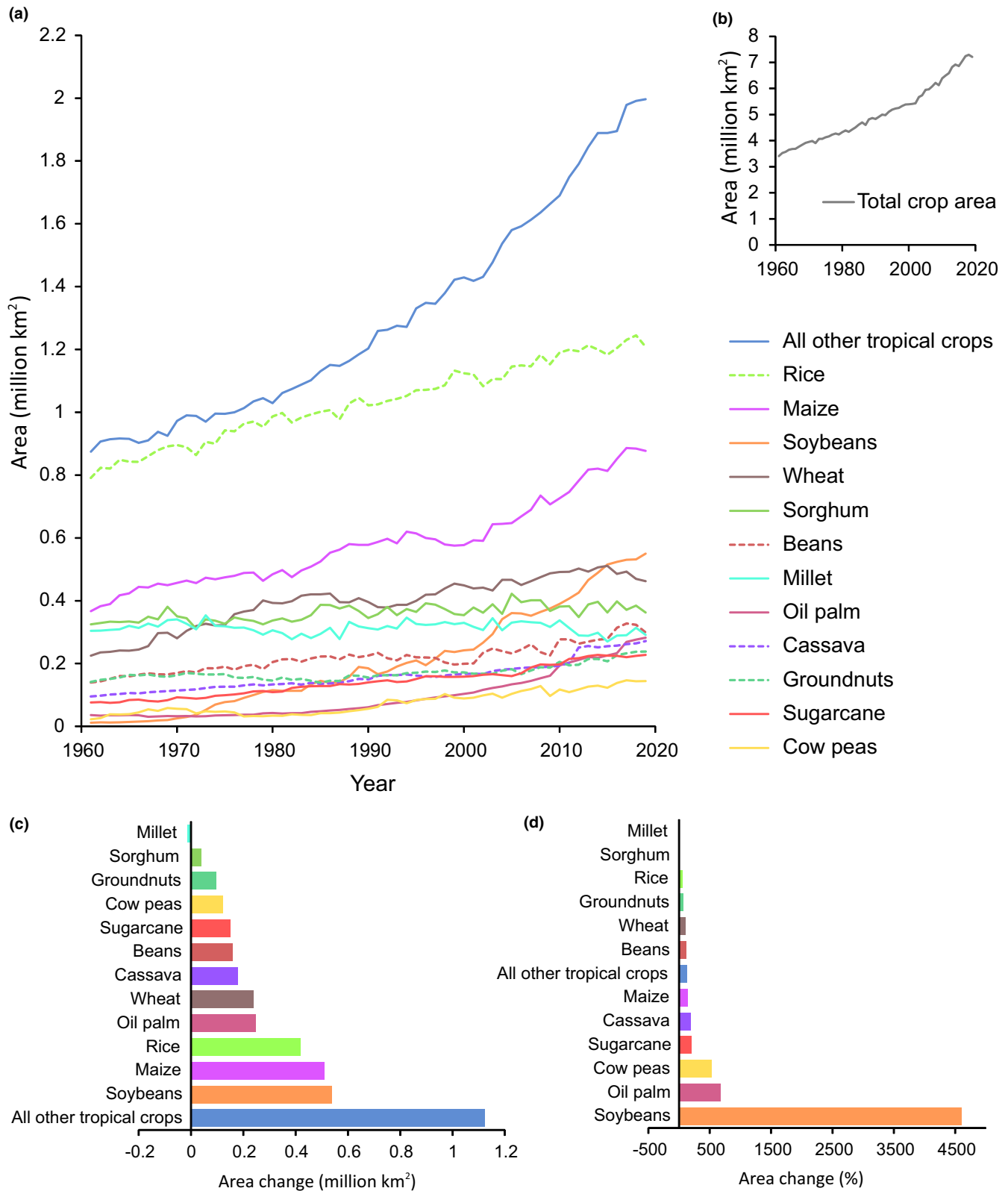
Our results suggest that, overall, food crop expansion has contributed towards biodiversity loss in tropical regions, although the direction and magnitude of the impact depends on the crop, level of shading, rotation time, taxonomic group and geographical region. The overall effect of tropical agriculture upon biodiversity is negative and significantly different from zero (Figure 3; mean Hedges' *g* [ $\pm 95\%$  CI] =  $-0.59$  [ $-0.67$  to  $-0.51$ ],  $p < 0.001$ ; Table S3).

Exploring the data by crops, we found that effect sizes were negative and significantly different from zero in maize, oil palm, sugarcane, 'all other tropical crops', tea, rice, cacao, and 'mixed tropical crop' sites, compared with natural habitats (Figure 3a; Table S3). Biodiversity responses were in general negative but not significant in citrus, allspice and coffee plantations, while banana and mixed cacao and coffee plantations showed a positive effect size, though not significantly different from zero. Examining our results by level of shading, we found that for shaded and unshaded crops, biodiversity showed a negative and significant difference from zero, with unshaded crops having a considerably greater negative effect size than shaded crops (Figure 3b; Table S3). However, we did not find a significant effect for crops with some vegetation where the confidence intervals were particularly wide and overlapped zero. We found that crop rotation time is an important determinant of impacts, with annual crops showing a greater negative response than perennial crops that have longer rotation periods, though both categories had a significantly negative effect size (Figure 3c). Effect sizes were negative

but not significantly different from zero for croplands with 'mixed' annual and perennial crops, and for 'unknown' where studies did not specify whether crops were annual or perennial. Exploring the results by taxonomic group, we found that bird, herpetofauna and invertebrate assemblages showed significantly negative effect sizes of similar magnitudes in response to agricultural treatments, while mammal responses were negative but not significant (Figure 3d; Table S3). Examining our results by geographical region, we found there was a significantly negative effect of agriculture on biodiversity in all tropical regions (Figure 3e; Table S3). Asia showed the greatest negative response, followed by South America, Africa, and lastly, Central America. Finally, comparing by biodiversity metric, effect sizes for both richness and abundance were negative and significantly different from zero, with richness showing the strongest response to agriculture (Figure S4). Breaking this down by crop, in all cases, the effect sizes for richness were consistently more negative than those for abundance, more likely to be significant, and in the case of coffee, there was evidence of a positive effect on abundance (Figure S4).

### 4 | DISCUSSION

Our study supports the existing literature highlighting the adverse impacts of tropical agriculture upon animal assemblages (Chapman et al., 2019; Gibson et al., 2011; Ocampo-Ariza et al., 2019; Ramamonjisoa et al., 2020). Adding to this, 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 2), with our findings of increased acceleration of crop expansion over the past two decades corroborating those of Potapov et al. (2022), which are based on remote sensing data. Potapov et al. (2022) further emphasises the magnitude of crop expansion in the tropics, showing that globally, conversion of natural vegetation to croplands was proportionately largest in Africa, Southeast Asia and South America. Our results demonstrate 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. Indeed, 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), and along with them, their specific functions lost.



**FIGURE 2** Changes in harvested area of tropical crops from 1961 to 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 to 2019. The top 10 tropical crops by area in 2019 are shown. Additionally, sugarcane and cow peas, which were in the top 10 by area increase, are also shown. The harvested areas of 'all other tropical crops' were combined. Data: FAOSTAT (2021).

A particularly important finding from our study is the relative impacts from different crop production systems. We show that unshaded crops 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 resource 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 crops than unshaded crops highlights the potential for improving agricultural practices to reduce biodiversity loss, and this may explain why abundance in some coffee plantations can increase. The wide confidence intervals for crops with some vegetation could be due to fewer studies, or variation in the capacity for different types of vegetation to support biodiversity (e.g. croplands with scattered shade trees provide a different habitat from those with an intact understory). Better measures of agricultural intensity could include chemical inputs, monoculture vs polyculture, and weed richness and cover; however, these details were often not reported in studies. We also show that crops that are harvested on an annual basis, such as maize, sugarcane and rice, result in 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 10 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 'all 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). Nonetheless, the biodiversity impacts of soybeans are understudied compared with other tropical crops such as cacao, coffee and oil palm, which account

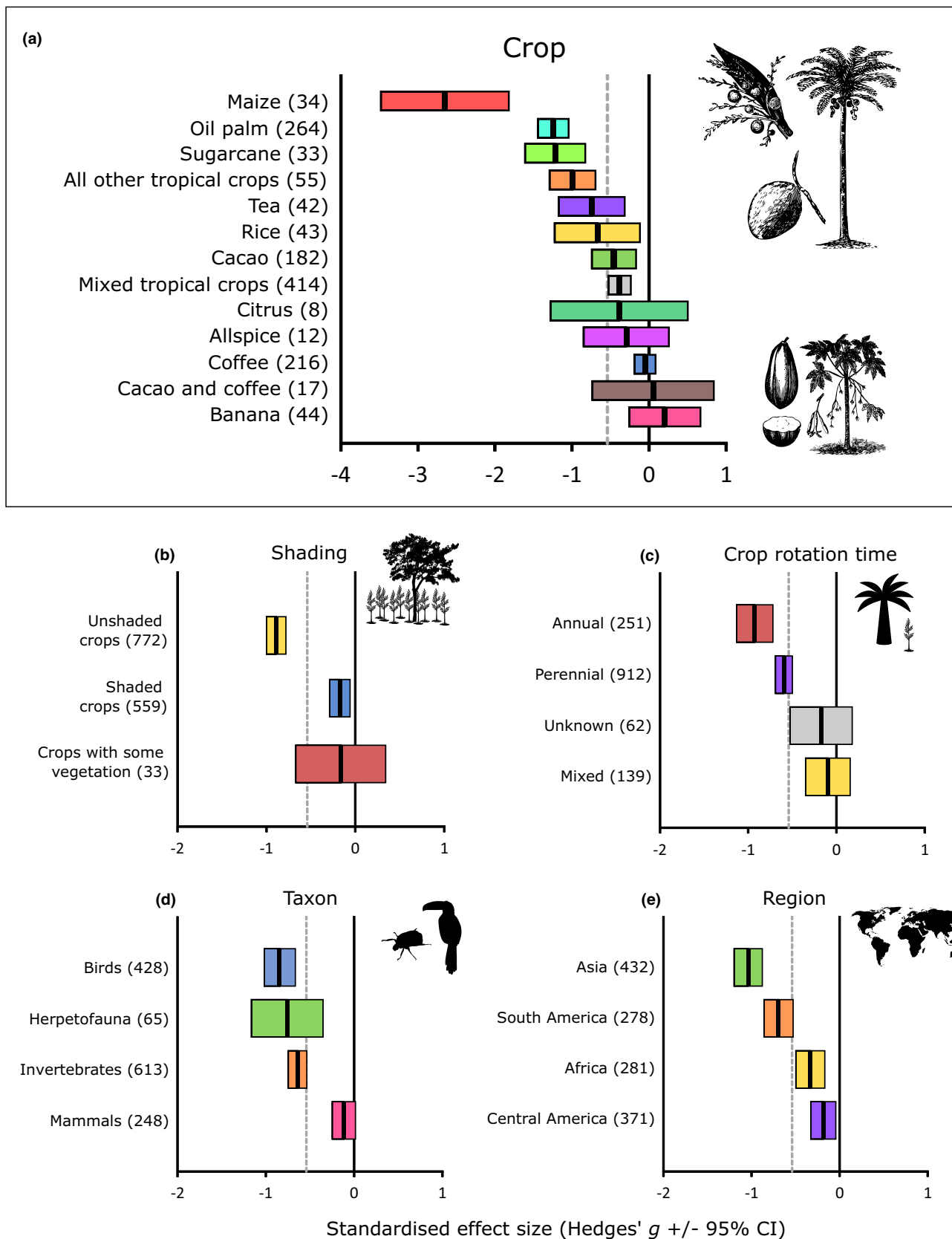
for considerably less harvested land area (FAOSTAT, 2021). Many lesser-known crops are grown by small-scale subsistence farmers and are less likely to gain attention from conservationists than industrially produced crops that are traded internationally (Balmford et al., 2012). Our REA also showed some geographical bias in the papers we found. In the Neotropics, research is concentrated in Brazil, Mexico, Costa Rica, Colombia and Peru, and in Asia the majority of studies come from Malaysia, Indonesia and India (Figure S2). Most other countries provided few or no studies; research in tropical Oceania is particularly limited. Gaps in our dataset may be due in part to our restricting of the literature search to English language studies. It must therefore be acknowledged that language bias presents a limitation to our study. The inclusion of non-English languages could provide further data and consequently potentially alter effect sizes (Konno et al., 2020). However, as demonstrated by our assessments of publication bias (described in the Methods), any changes to effect sizes as a result of missing articles are highly unlikely to overturn the conclusions of the study.

In our analysis, richness metrics declined more than abundance metrics. Both are concerning, because there is an abundant literature to show that, in general, the first species to be lost under habitat conversion (and therefore reduce richness) are the most sensitive species that are typically of conservation concern (e.g. Newbold et al., 2015 and studies therein). On the other hand, reductions in abundance metrics (and richness metrics) indicate potential declines in the provisioning of ecosystem functions performed by key groups. As such, birds which are important mobile seed dispersers and pest controllers 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).

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 (Prugh et al., 2008). 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.

Understanding the consequences of food cultivation on biodiversity can help to identify improvements to agricultural practices and influence consumer choice. Since much of the food produced in tropical regions is exported internationally, a large proportion of impacts on tropical biodiversity are remotely driven by industrialised countries





← More impoverished biodiversity community in agricultural area compared to reference habitat

FIGURE 3 Legend on next page

**FIGURE 3** Effect sizes of agricultural impacts on biodiversity by (a) crop, (b) intensity, (c) crop rotation time, (d) taxonomic group (omitting fish  $N_{\text{studies}} = 3$ ) and (e) geographical region (omitting Oceania  $N_{\text{studies}} = 1$ ). 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. For single crops represented by fewer than four studies, we grouped these and reported them as 'all other tropical crops'. When biodiversity values were provided for sites that did not distinguish between multiple different crops, we reported them as 'mixed tropical crops'.

(Green et al., 2019; Lenzen et al., 2012). This study therefore provides us with food for thought regarding the positive and negative environmental impacts caused by our food choices. It is particularly pertinent as we are trying to improve the transparency of food supply chains and connecting consumer markets to habitat destruction and biodiversity loss through projects such as Trase (<http://www.trase.earth>). 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 biodiversity 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.

#### AUTHOR CONTRIBUTIONS

Joseph L. Oakley conducted the REA, collected the data, analysed the data and led the writing of the manuscript. Jake E. Bicknell conceived the study, analysed the data and co-wrote the manuscript. Both authors contributed critically to the drafts and gave final approval for publication.

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#### CONFLICTS OF INTEREST

The authors declare no conflicts of interests.

#### DATA AVAILABILITY STATEMENT

Data available from the Dryad Digital Repository: <https://doi.org/10.5061/dryad.8w9ghx3qn> (Oakley & Bicknell, 2022).

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## SUPPORTING INFORMATION

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