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Abstract

Coffee is one of the most important globally traded commodities from the developing world. While it is widely believed that agroforestry grown coffee is less damaging to biodiversity than monoculture coffee, increasing demand has resulted in an intensification of growing practice. Despite awareness of the contribution of coffee to biodiversity loss, there is no global study on the role of agricultural management practice in driving biodiversity loss. Here, I build a dataset on species richness change between forest and different management intensities of coffee production across the world. I then analyse the association of the different management practices with extent of species loss. I found monoculture coffee systems experienced the greatest species loss while diverse coffee agroforestry experienced minimal species loss. My results showed that agroforestry coffee was not better at conserving species than the monoculture unless it was diverse and retained native tree species. This suggests that efforts to promote biodiversity-friendly coffee, such as through sustainability certifications, should favor coffee grown in diverse agroforestry systems with native trees rather than any shaded coffee system. However, as existing data on the effect of coffee production on biodiversity is limited, further studies are needed that consider other aspects of biodiversity such as species composition and consider proximity to native forest to account for potential spillover of species into the coffee plantations.

Introduction

Terrestrial vegetation biomass has halved since the start of agriculture—the main identified threat of 86% of species at risk of extinction (Bradshaw et al., 2021; Díaz et al., 2019; Benton et al., 2021). While increasing agricultural yields have slowed the rate of land conversion, the resulting intensification has reduced biodiversity within the existing agricultural landscapes (Millennium Ecosystem Assessment, 2005). With current agricultural systems covering ~43% of Earth's ice and desert-free land it remains a challenge to balance productive agricultural land use and biodiversity conservation (Poore & Nemecek, 2018; Tscharntke et al., 2012). This is an issue in the biodiversity-rich tropics, where the majority of land use change for agriculture is occurring (Foley et al., 2011; Jenkins et al., 2013). However, the trade-off of environmental and economic goals, upon which people's livelihoods depend, complicate the ability to enact progressive change. This is because agricultural systems with the greatest profit involve greater modification and disruption of natural ecosystems (Phalan et al., 2011).

Coffee is an important tropically grown crop, with high and fast-growing demand (FAOSTAT, 2022). It is one of the most valuable legally traded commodities from the developing world, spanning around 80 tropical and subtropical countries with an estimated 25 million people in the production alone (Damatta et al., 2018; Donald, 2004; FAO et al., 2014) Global coffee production has nearly doubled in the last thirty years with 9.9 billion kg produced in 2020 (ICO, 2021). To meet increasing demands, coffee production is being introduced into new regions of the world (Fig.1) (Harvey et al., 2021). The growing demand for coffee has also been met with intensification of coffee-growing practices, intensification being one of the largest drivers of tropical biodiversity loss (Jha et al., 2014; Tscharntke et al., 2011). This therefore explains the association of coffee production and substantial biodiversity loss for which growing concern is reflected in the rise of eco-certification programs such as Rainforest Alliance, Fairtrade, Organic, and Smithsonian Bird Friendly, which regulate agrochemical application and shade tree planting (Harvey et al., 2021; Millard, 2011). Over half of global coffee production in 2020 was certified by voluntary sustainability standards (Panhuysen & Pierrot, 2020).

Coffee originates in Ethiopia where it grows in the rainforest understory (Damatta et al., 2018). It is not economically viable to harvest wild coffee from forests as the high levels of shade result in low yields and low coffee-bush density (Hirons et al., 2018). Therefore, traditional coffee growing practices involve planting additional coffee bushes in the rainforest understory – a type of agricultural practice called agroforestry (Moguel & Toledo, 1999). This practice is most common in Ethiopia and several Latin American countries (Harvey et al., 2021; Hirons et al., 2018). However, coffee agroforestry systems are shifting from retaining natural overstory to replacing it with non-native trees that provide added economic benefits such as fruits, timber, or nitrogen fixation (Moguel & Toledo, 1999). Furthermore, coffee varieties have been bred to better withstand heat stress and can thus be grown in a shade-less monoculture, known as sun coffee (Perfecto et al., 1996). The expansion of coffee production to new regions, primarily in southeast Asia, occurs with adoption of high intensity management systems such as modified agroforestry systems or coffee monocultures (Fig. 1) (Jha et al., 2014). Most coffee comes from Latin America where there too has been a shift to more intensively managed coffee (ICO, 2021; Harvey et al., 2021). As a result of their high yields, coffee monocultures have become the dominant method of global coffee production (Jezeer et al., 2017).

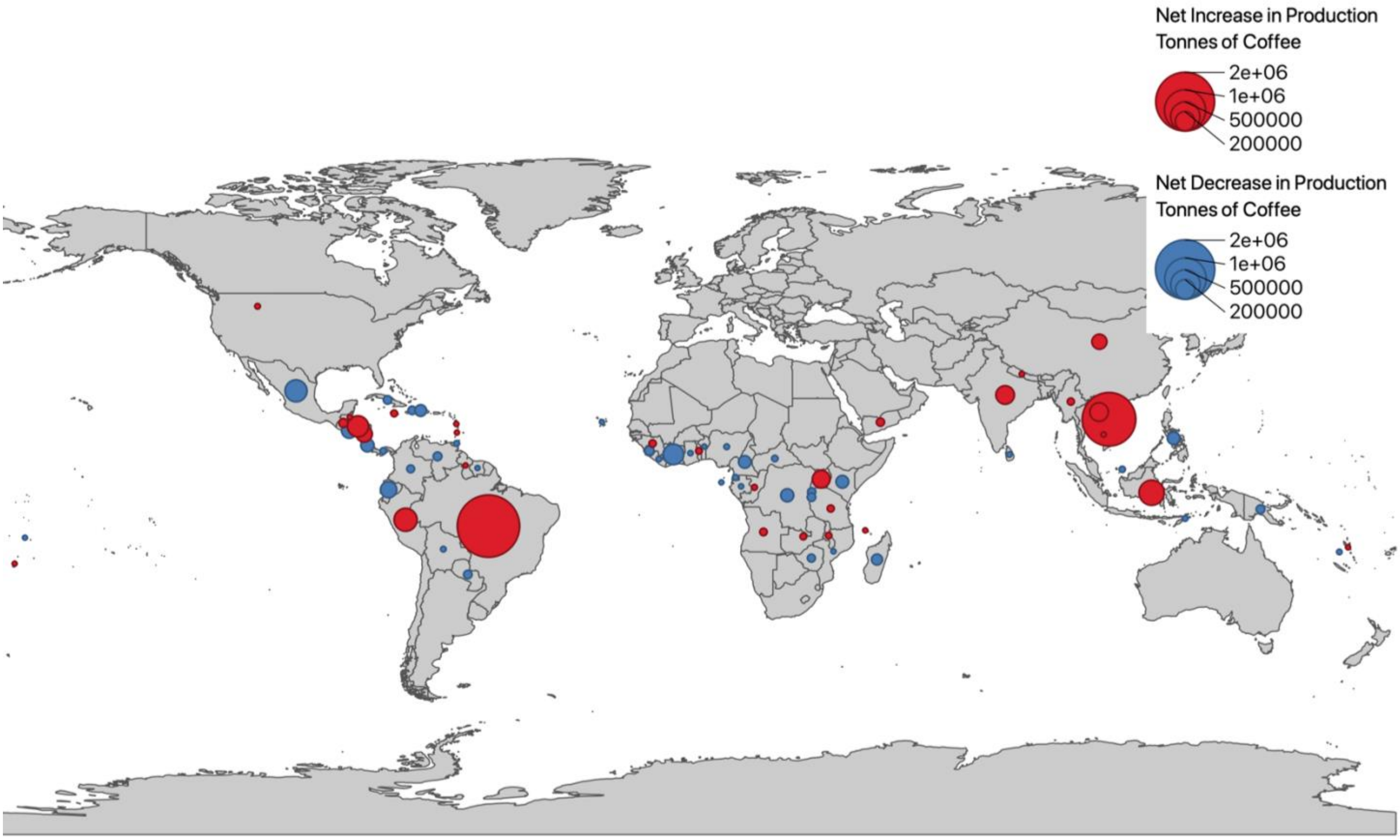

Figure 1. World map showing change in tonnes of coffee production per country from 1990 to 2020 where larger circles represent greater change. Red represents increase in production and blue represents decrease. The overall trend is large increases in production. Coffee data from FAOSTAT (2021) and shapefile from https://www.naturalearthdata.com/downloads/

There is a gap in our current understanding of the environmental impact of coffee as previous studies are often restricted in scope and scale. Most studies on coffee's effect on biodiversity consider impacts of the land use on one taxon in one region, leaving limited room to extrapolate to wider impacts. There is some cross-continental research on the value of shade coffee systems for biodiversity conservation (de Beenhouwer et al., 2013), but an absence of studies also considering the effect of coffee monocultures. This is a large gap our knowledge as coffee monocultures represent the largest share of coffee area and are dominant in regions where coffee production has recently been introduced (Jezeer et al., 2017). To date there exists no representative global assessment of the effects of the full range of coffee system management intensities on biodiversity.

Understanding the extent to which different coffee growing systems drive biodiversity loss would better enable promotion of environmentally friendly production methods. This understanding is important for not just the wider environmental impacts but for the long-term viability of coffee production which itself benefits from forest ecosystem services (Jha et al., 2014). Such ecosystem services include pollination, pest control, and erosion control (Wardle et al., 2011). While coffee, a climate sensitive crop, benefits from the microclimate of shade and farmers get diversified income sources from fruit or timber trees, the extent of biodiversity conservation from shade trees is not always clear (Ahmed et al., 2021; Bunn et al., 2015; Pham et al., 2019). It is generally asserted that the lower intensity the coffee agroforestry system, the less the associated biodiversity loss (Greenberg et al., 1997; Moguel & Toledo, 1999). However, studies do not always find significant species richness declines with intensifying coffee management practices (Pinkus-Rendón et al., 2006; Ricketts et al., 2001).

An improved understanding of the role of management practices on biodiversity loss could better inform sustainability certifications and their environmental criteria. Sustainability certifications are often vague in their requirements for biodiversity conservation, promoting planting of native shade tree species in the coffee system (Fairtrade International, 2021; Rainforest Alliance, 2020; Smithsonian Bird Friendly, 2022). Apart from Smithsonian Bird Friendly, the environmental sustainability criteria of the primary sustainability certifications only suggest or advise incorporation of native tree species (Fairtrade International, 2021; Rainforest Alliance, 2020; Smithsonian Bird Friendly, 2022). This makes it possible for coffee monocultures to receive these sustainability certifications with claims of higher environmental standards. To inform certification that adequately supports species conservation, the effect of monoculture and the varying types of shade coffee systems on native biodiversity must be better understood.

This thesis presents an analysis of the effects of coffee management type on biodiversity based on the current literature. It includes all relevant coffee management types on a global scale. It additionally considers the role of geographic and climatic factors in modulating the effect of coffee management on biodiversity loss. This study therefore aims to provide information that would support recommendations on methods of producing coffee at a lower environmental cost. It also aims to assess where gaps exist in current literature on the biodiversity impact of coffee systems and highlight underreported or under-researched areas.

Methods

Overview

To determine the aspects of coffee production that drive biodiversity loss, I combined data from existing reviews with data collected by conducting a systematic review of the literature. Using species richness as a measure for biodiversity, I considered the effects of management practice, region, and taxon of study on the level of species loss recorded. Species loss was calculated using an effect size measure (Hedges' G) to standardize the difference between species richness in a coffee site and a paired forest site. Positive values of Hedges' G represent species gain and negative values represent species loss.

Coffee Management Practice

Coffee systems can be broadly categorized by the way in which they are managed. While there are many variations on how categories are set, Moguel and Toledo (1999) outlined five types of management that encapsulate the range of intensities under which coffee is grown today. Throughout this thesis I will use five categories of coffee management which are defined as follows (Fig. 2):

1. Rustic coffee – removal only of lower forest strata for planting of coffee bushes
2. Traditional Polyculture – sophisticated management of coffee alongside other plants of use to people with addition of domestic species to the natural forest
3. Commercial Polyculture – complete removal of original forest canopy with introduction of shade trees of either commercial use or benefit to the coffee crop
4. Shaded monoculture – complete removal of original forest canopy with introduction of a monospecific shade system above a monospecific coffee understory
5. Sun coffee – complete removal of original forest canopy with coffee grown in a monoculture exposed to direct sunlight

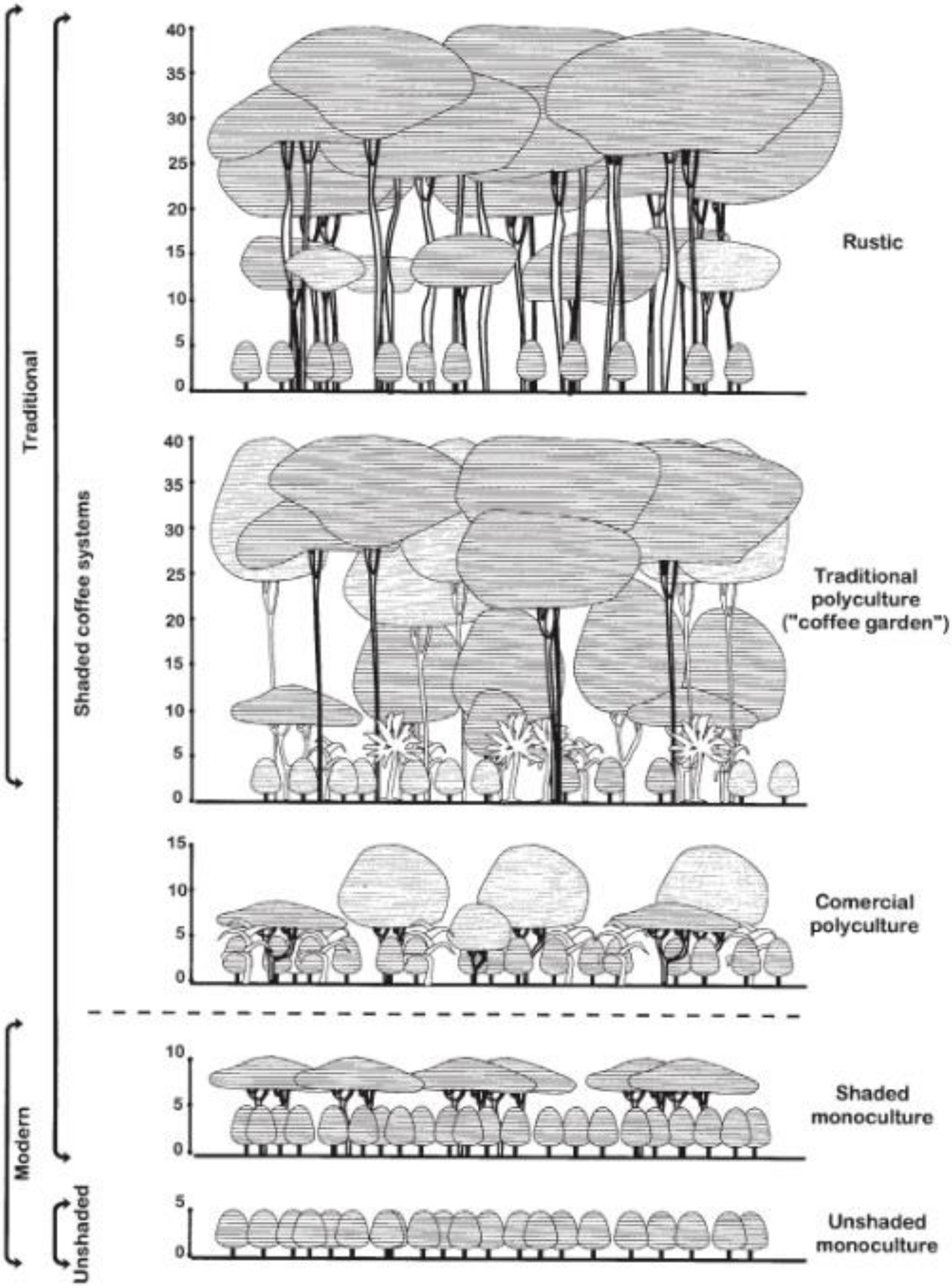

Figure 2. The five coffee management types depicting canopy height and vegetational complexity (Moguel and Toledo, 1999)

Dataset construction

Analysis of existing data

From the scoping review of the literature, I found a meta-analysis by De Beenhouwer et al (2013) containing a dataset with 161 pairwise comparisons of species richness of either coffee or cocoa systems and nearby forests. I obtained the data for each observation point related to coffee through personal communications with the lead author. I then reassessed all the primary literature used in the De Beenhouwer et al (2013) meta-analysis herein refered to as De Beenhouwer. De Beenhouwer conducted pairwise comparisons of either forest to agroforestry coffee or agroforestry coffee to plantation coffee. This looked at the extent of species loss for both conversion of land and intensification, however I was interested in the effect of coffee management practice on native biodiversity. Therefore, I reanalyzed the comparisons of agroforestry against plantation coffee to compare native forest against plantation coffee. Where no forest site was measured, I excluded the observation of plantation coffee. From each primary source I re-categorized coffee systems (which had been classified by De Beenhouwer as either agroforestry or plantation) to the more informative 5 types outlined by Moguel and Toledo (1999). I either used the description of the coffee systems from the primary text or obtained the information on management category by communicating with the lead authors where the category was ambiguous. When I was not able to categorize the coffee system, I still included the observation in the dataset but excluded it from analyses considering the effect of management practice.

De Beenhouwer excluded observations of sun coffee systems for unspecified reasons but as these have become the dominant production method (Jha et al., 2014), I added all observations of sun coffee from the primary literature for comparisons to forest. Where observations of species richness on a coffee site were aggregated across different management types, I split them according to the Moguel and Toledo (1999) categories if sufficient data were present. I also retrieved coordinate, altitude, precipitation, temperature, and slope data from the primary literature. Where coordinate data was not given, I derived coordinates from the nearby towns or farms mentioned, and where several coordinates where given, I averaged them. I then used the coordinates to fill in the gaps for precipitation and temperature by uploading the coordinates to the https://hestia.earth data platform. This platform automatically retrieved the data from a long-term annual mean for each site with dates set from 1960-2020. This information was chosen as it may explain some variation in level of species loss across coffee plantations.

Additional pairwise comparisons of coffee to forest sites were obtained from (Gibson et al., 2011), a meta-analysis of tropical crops, and PREDICTS, an open-source database of biodiversity loss associated with land use https://data.nhm.ac.uk/dataset/the-2016-release-of-the-predicts-database (Hudson et al., 2017). I reassessed the primary coffee literature from these two sources in accordance with my re-analysis of De Beenhouwer. Where data were only presented in a figure containing mean species richness and error bars, I used WebPlotDigitizer to extract the values. Where information on management type was vague, data were aggregated across different management types, or data were presented as total rather than mean species richness, I emailed the lead authors requesting the missing information and appropriate raw data. If there was no communication on management type the data were still included excepting for analysis of management type. If there was no raw data to convert from total to mean species richness then the study was excluded, as there was insufficient information to calculate the effect size measure.

Systematic review to extend dataset

As the three initial databases included primary sources only published until 2012, I conducted a systematic review in February 2022 with dates set from 2012 to present. I used the search terms used by De Beenhouwer but added a term to specify forest site as De Beenhouwer included sources comparing coffee systems to each other without a forest comparison. I also simplified the search terms to only use one search string due to time constraints. The search terms used were: “coffee” and “biodiversity” and “forest”. I screened all results for potential relevance and then accessed the full text for the remaining articles to assess for eligibility. The criteria for eligibility were:

a) Land use = coffee and comparable forest sites
b) Metric = mean species richness in the different sites and their variance
c) Geography = retrievable coordinates for the area of study

Eligible studies were then screened for information on which of the five coffee management types the systems fell under, along with altitude, precipitation, temperature, and slope. Where information on management type was missing or insufficient to categorize into one of the five management practices, I emailed the lead authors to request this information. I also included all non-English studies that came up which were either in Spanish or Portuguese. A flow diagram depicting the source of data can be found in (Fig. 3).

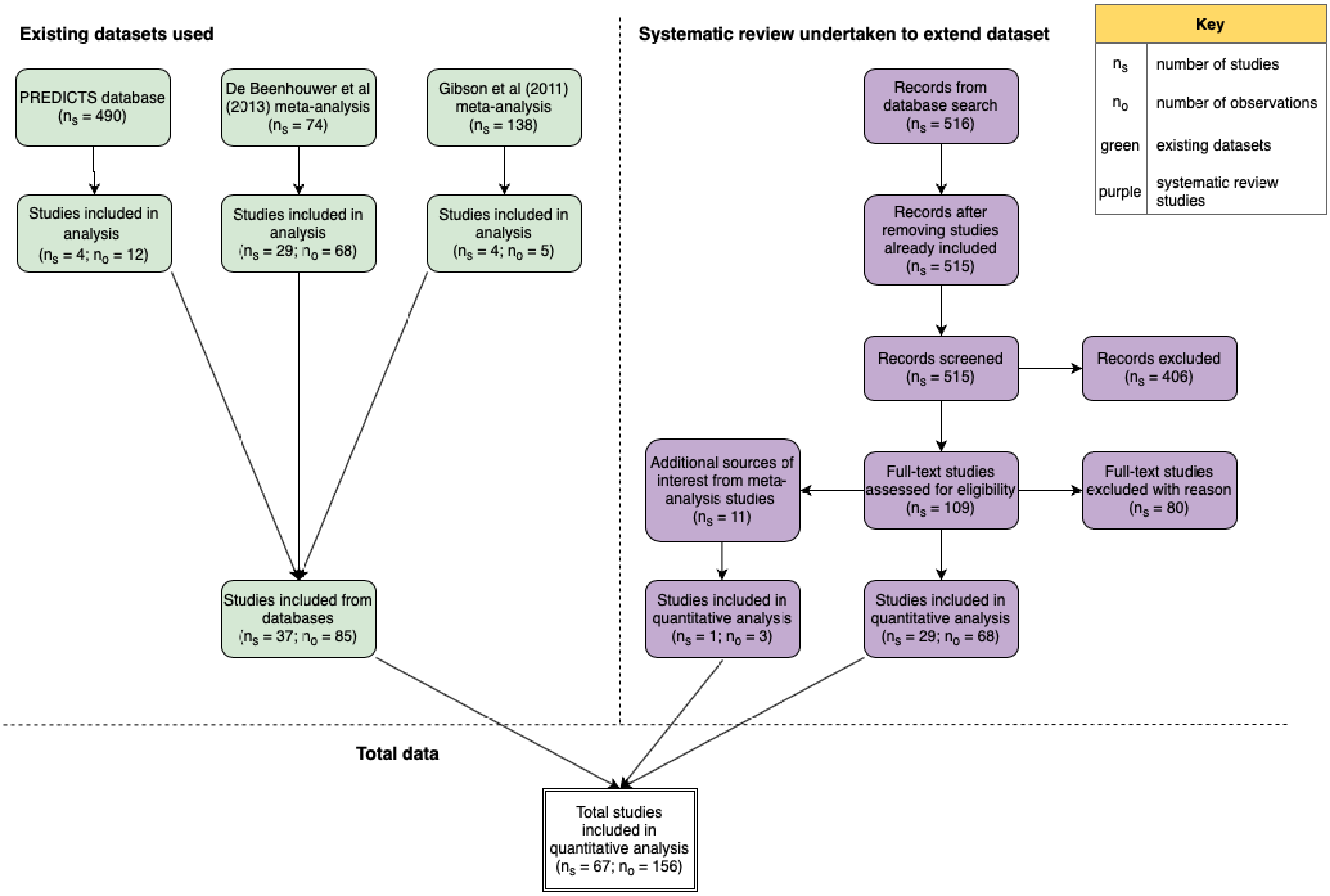

Figure 3. Flow diagram depicting where data was obtained from and how many studies came from each source

Data Analysis

Overview of analysis

I used the analysis methods outlined by De Beenhouwer in which Hedges' G (an effect size measure) was calculated for each paired forest-coffee site. I analyzed the data in R version 4.1.1. I used GQIS to map the location of each observation in my dataset over an EarthStat raster file mapping areas of coffee production (plotted at 5x5minute resolution) across the world map from https://www.naturalearthdata.com/downloads/ as an indicator of the global representativeness of my dataset.

Replication of De Beenhouwer methods

Using raw data obtained through personal communications with Matthias De Beenhouwer, I recreated the output for Fig. 2 from De Beenhouwer. This was done to replicate the model used and ensure my model and analysis was consistent with that of De Beenhouwer. This was a mixed effect model with study included as a random effect and inverse variance weighting. I ran a model of Hedges' G against land use comparison (either forest to agroforest, or agroforest to plantation) with the inverse variance weighting and was able to recreate the values in Fig. 2 of De Beenhouwer (Fig S1 & S2). I also ran the model without the weighting and the implication of the results changed as did the mean values, suggesting that use of weighting has a large impact on how the data is interpreted (Fig S3). The unweighted model closer met the assumptions of normal distributions of residuals than the inverse variance weighted model (Fig S4 and S5). Inverse variance weighting assumes that an individual study with a large sample size is more representative of the population or study system than a study with a smaller sample size by giving more weight to studies of a large sample size. However, the studies in this meta-analysis are not uniform in regards to factors that might influence biodiversity loss, being taxon of study, geographic region, and agricultural practice. Therefore, one study with a large sample size is not necessarily a better representation of global coffee systems than a study considering different factors with a smaller sample size. Some areas naturally experience higher levels of biodiversity so will experience greater loss due to land use, some taxa are more vulnerable to change therefore will suffer more (Olson et al., 2001; Stork et al., 2009). My dataset includes a large range of regions, taxa, and management practices of coffee systems. For these reasons, in my own analysis I considered models with and without inverse variance weighting and compared the results.

Application of updated De Beenhouwer methods

I checked for normality of the data using quantile-quantile plots in addition to Shapiro-Wilk tests and Kolmogorov-Smirnoff tests on Hedges' G and on the residuals of each linear model (Figure S6 and S7).

I conducted mixed effect models using the lme4 package in R, with study included as a random effect. This recognizes that different research methods may affect results between studies and is in line with the analysis structure of De Beenhouwer. I first ran univariate mixed effect models for Hedges' G against management type. When analyzing effect size against management type I excluded all observations in which management type data were not retrievable. I then ran another model of effect size against management type where all observations reporting tree species richness were excluded. This is because the two most intensive coffee management systems represent two types of monoculture systems without native trees, rendering measures of tree species richness irrelevant with the results of the model including trees found in (Table S1). Finally, I conducted a multivariate mixed effects model including all the variables with sufficient data. This meant I excluded altitude and slope as several studies did not report it and coordinate data were not sufficiently precise to retrieve these values. I separated plants into trees, epiphytes, and other vascular plants. This separation is important as trees are cleared and planted in accordance with the different management intensities, epiphytes are removed due to traditional belief that they reduce coffee yield, and other vascular plants such as shrubs are cleared with increasing understory density of coffee (Cruz-Angón & Greenberg, 2005). In the analysis including continent, I aggregated Central and South America into Latin America due to geographic proximity and connectedness. This model was conducted to see if the effect of management type remained after other variables potentially affecting biodiversity loss were included.

Tests of normality of model residuals

For each mixed effect model the normal quantiles plot of the residuals had a slight left tail in the unweighted model and a strong tail in the weighted model (Fig S6 and S7). The results of the Shapiro-Wilk normality tests were consistent in that the residuals were considered non-normally distributed regardless of whether the model was weighted or unweighted ($p < 0.05$). The Kolmogorov-Smirnov normality test consistently showed that residuals were normally distributed in the unweighted models but non-normal in the weighted models ($p > 0.05$ unweighted, $p < 0.05$ weighted). In the unweighted mixed effect model of Hedges' G against all the fixed effects, the Shapiro-Wilk test showed a near-normal distribution of residuals ($p = 0.0465$). As a result of the strong skew in all the weighted models, and the lack of clear benefit of using inverse variance weighting for this study, I report only the results of the unweighted models.

Test of publication bias

Finally, I checked for publication bias following the methods from De Beenhouwer. I assessed symmetry of studies published using a funnel plot of effect size against standard error. Asymmetrical funnel plots suggest a relationship between study size and effect size where small effect sizes are less likely to be published. Normal quantiles plots were visualized to further assess possible publication bias. If the points almost co-occur with the Y=X line then the distribution is considered normal.

Results

Dataset Structure

The dataset comprises 67 total studies providing 156 pairwise comparisons (Fig 2, Tables 1-3). Reassessment of the primary sources used by De Beenhouwer resulted in the addition of 7 pairwise comparisons not included in the initial analysis. Overlaying global coffee production with the sites of each observation in the dataset highlights the bias towards studies in Latin America, in spite of high levels of coffee production in Africa and Asia (Fig. 4).

Table 1. Number of observations for each management practice

Management practice	
Category	n_o
Rustic coffee	11
Traditional Polyculture	40
Commercial Polyculture	47
Shaded Monoculture	32
Sun Coffee	14

Table 2. Number of observations for each taxon

Taxa	
Category	n_o
Amphibians	6
Arthropods	65
Birds	21
Epiphytes	9
Fungi	6
Mammals	14
Trees	31
Other Vascular Plants	4

Table 3. Number of observations for each continent

Continent	
Category	n_o
Latin America	129
Africa	16
Asia	11

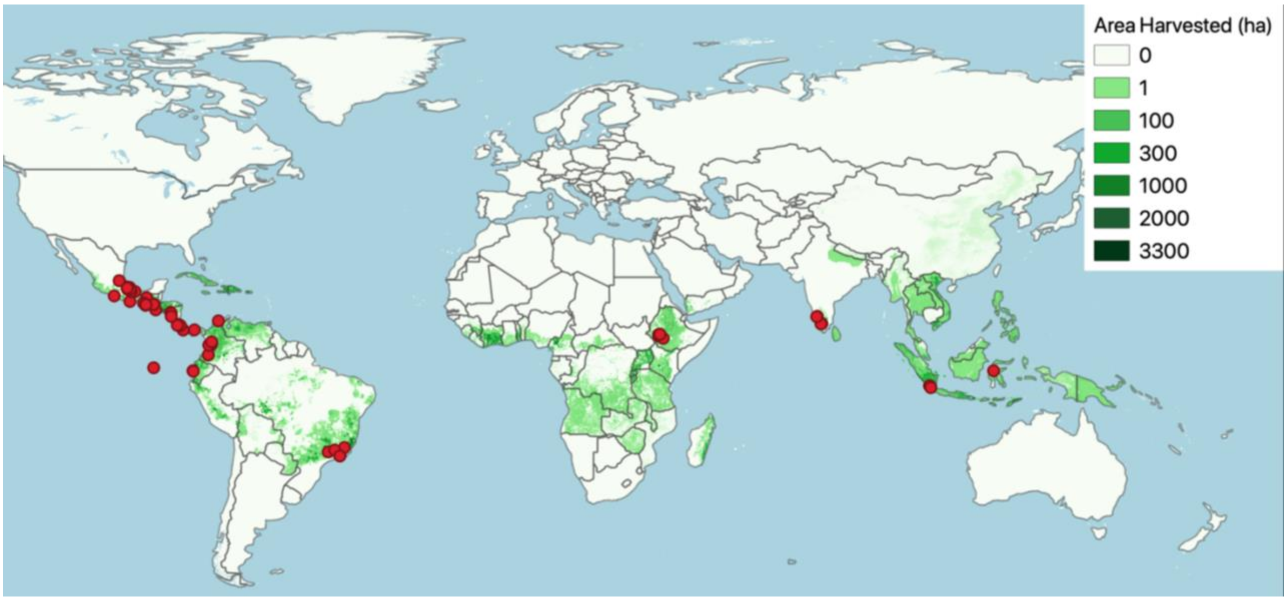

Figure 4. World map with areas of coffee production in green (darker green represents more coffee) with observations of my dataset mapped over it (red circles). The key shows number of hectares of coffee production per 5x5min grid.

General trends

Overall, coffee production causes species loss which is represented by a negative Hedges' G (Mean Hedges' G: -1.163 ± 0.148 SE). When observations on tree species richness were excluded, coffee production caused species loss to a smaller degree (mean Hedges' G: -0.853 ± 0.153 SE). The funnel plot showed lots of scatter and some asymmetry for the dataset both with and without tree species (Fig. 5 and Fig. 6). The normal quantiles plot of Hedges' G for both versions of the dataset appear roughly normal with a slight tail.

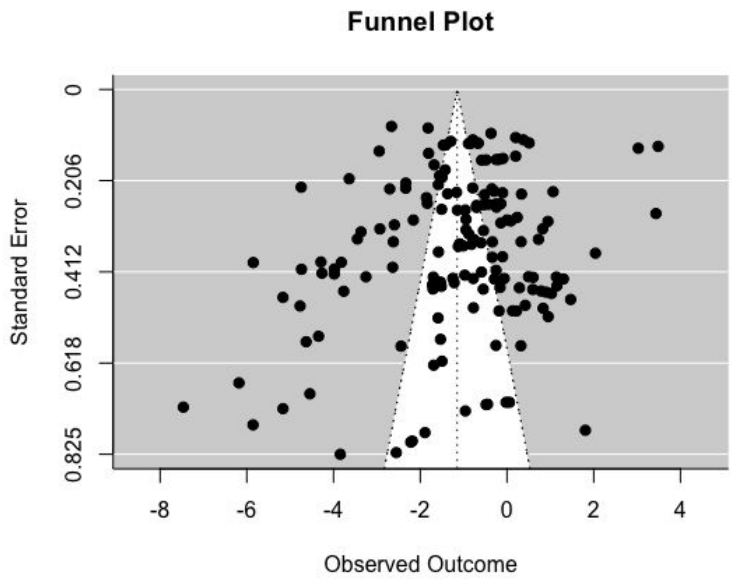

Figure 5. Funnel plot of dataset (including tree species) plotting standard error against Hedges' G

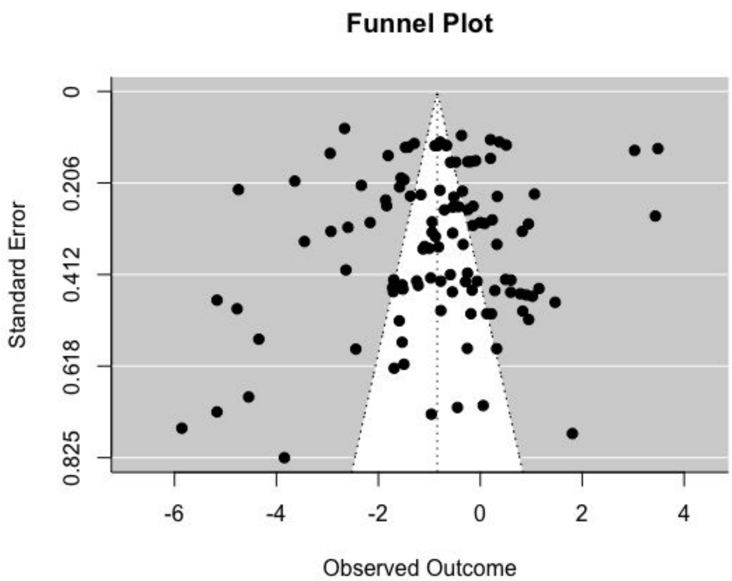

Figure 6. Funnel plot of dataset (excluding tree species) plotting standard error against Hedges' G

Linear mixed effect model analysis

Association of management practice on Hedges' G

In the univariate model with Hedge's G related to management practice, there is a clear trend of greater species loss in coffee sites compared to paired forest sites as management practices become more intensive (Fig. 7, Table 5). Sun coffee experiences the greatest species loss while rustic coffee is associated with little change in species richness, even reporting small gains (Table 5). Rustic coffee had significantly less species loss than sun coffee while traditional polyculture, commercial polyculture, and shaded monoculture systems did not differ significantly from sun coffee (Table 5). There is variation in the effect size for each management type but there is an overall trend of higher intensity management types correlating with higher species loss (Fig 7). As tree species were excluded from this model along with observations where management type wasn't specified there were 114 total observations in this model.

Table 5.
Mixed effect model output for Hedges' G with management practice as fixed effect in data without tree species richness. P-values are of difference from sun coffee. Values in bold represent a significant effect.

Management practice	Mean Hedges' G	Standard error	*p-value*
Sun coffee	-1.3263	0.6292	
Shaded monoculture	-1.0442	0.5401	0.549
Commercial polyculture	-0.9449	0.5440	0.423
Traditional polyculture	-0.4247	0.5591	0.074
Rustic coffee	0.2477	0.5099	**0.0141**

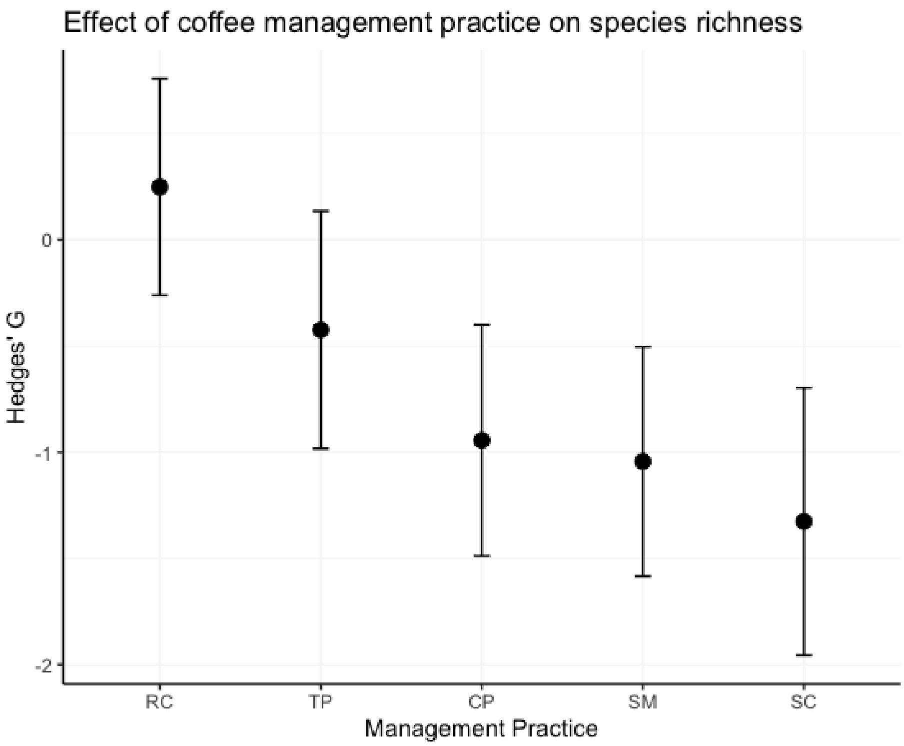

Figure 7. Plot of mixed effect model estimated mean Hedges' G values by management type. Error bars are standard error. RC - rustic coffee, TP - traditional polyculture, CP - commercial polyculture, SM - shaded monoculture, SC - sun coffee

Combined effects of management and other covariates on Hedges' G

In a multivariate model with all relevant fixed effects, only traditional polyculture and rustic coffee had significantly lower species loss than sun coffee which was represented as the Intercept in (Table 6). Both shaded monoculture and commercial polyculture coffee had lower species loss than sun coffee but to a small degree (Table 6). An ANOVA of the model revealed that management, taxon, and rainfall had significant effects on Hedges' G (ANOVA: p 0.0197, p 0.0298, p 0.0062). The other fixed effects (continent and temperature) had no significant effect on Hedges' G. Taxon ranged in extent of species loss in which plants had the greatest species loss while mammals, birds, and fungi experienced the greatest species gains in that order (Table 6). Ants were presented separately from other arthropods as they responded differently, and the model performed better with them separate. There were also enough observations to justify this separation with 31 observations of ants and 28 remaining observations of other (fewer than the total 64 due to observations removed from lacking management practice information). Continent of coffee growing did not significantly differ from each other in associated Hedges' G (Table 6). Of the climatic variables, higher rainfall correlated with more species loss while temperature had the same trend but no significant correlation (Table 6; Fig. 8).

Table 6. Mean Hedges' G of all categorical fixed effects in the multivariate mixed effect model or gradients for rainfall and temperature as they are continuous variables. The intercept represents mean Hedges' G for sun coffee, amphibians, Africa and the y-intercept for rainfall and temperature. P values are in bold if significant (meaning <0.05) and represent difference from intercept if categorical, or significant trend if continuous.

Fixed Effect	Category of Fixed Effect	Mean Hedges' G	Standard error	P value
	(Intercept)	-0.718	1.479	
Management practice	Shaded monoculture	-0.394	0.467	0.490
	Commercial polyculture	-0.132	0.4661	0.212
	Traditional polyculture	0.615	0.5226	**0.0125**
	Rustic coffee	0.742	0.6124	**0.019**
Taxon	Ants	0.214	0.918	0.315
	Arthropods	0.0218	0.935	0.432
	Birds	1.166	0.938	**0.0498**
	Epiphytes	-1.388	1.023	0.516
	Fungi	0.844	1.26	0.228
	Mammals	1.184	0.999	0.063
	Other plants	-1.425	1.462	0.632
Continent	Asia	-1.74	1.257	0.421
	Latin America	-0.085	0.904	0.480
Rainfall	Rainfall	-0.00045	0.00015	**0.0062**
Temperature	Temperature	-0.060	0.0702	0.396

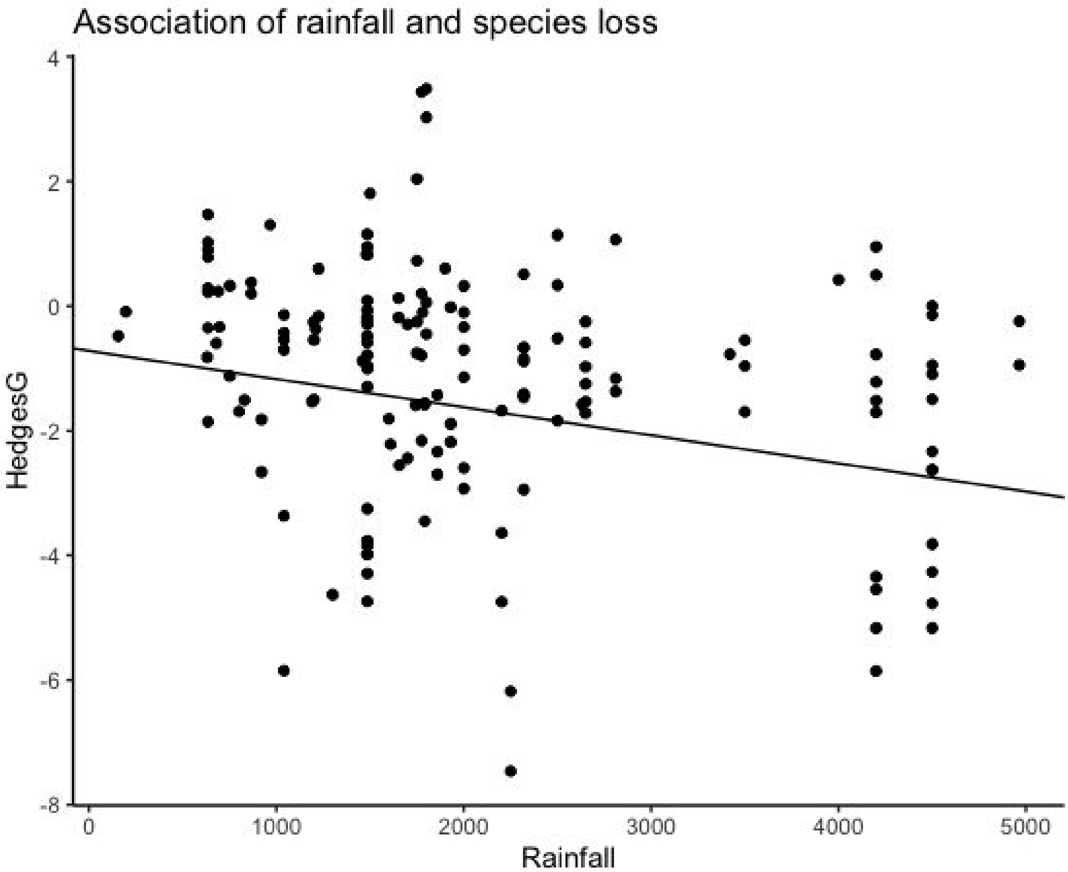

Figure 8. Scatter plot of raw values of mean annual rainfall (mm/year) against Hedges' G with gradient and intercept from multivariate mixed effect model.

Discussion

Effect of management type on biodiversity loss

This analysis suggests that the only coffee systems that are better for biodiversity conservation than sun coffee (the high intensity monoculture system) are rustic coffee and traditional polycultures. These are the only two systems in which complete deforestation is not necessary in order to grow coffee. Surprisingly, commercial polyculture, a mixed agroforestry system that involves deforestation and replanting of a select polyculture of tree species, is not notably different from sun coffee in associated species loss. This is despite commercial polyculture having greater plant diversity than monoculture coffee.

Sustainability certification schemes promote planting of native shade trees in coffee plantations as a method of conserving biodiversity (Fairtrade International, 2021; Rainforest Alliance, 2020; Smithsonian Bird Friendly, 2022). My results challenge the conventional view that commercial shade coffee is a biodiversity-friendly alternative to monoculture sun coffee. Rainforest Alliance and Fairtrade coffee promote the planting of native shade trees and creation of polycultures. However, it appears that only traditional polyculture and rustic coffee systems are better for biodiversity conservation than sun coffee monocultures. While the dataset needs to be better balanced in terms of regions and taxa studied, the consistency of results and the expected patterns of greater species loss with higher rainfall, a known correlate of biodiversity, suggests that the dataset is nonetheless reliable (Antonelli & Sanmartin, 2011; Pearson & Carroll, 1998). Smithsonian Bird Friendly Coffee is the only certification system with robust enough criteria for their coffee to be better for biodiversity than sun coffee as they state that coffee should be grown under traditional polyculture-style management (Smithsonian Bird Friendly, 2022). Voluntary sustainability certifications often have a trade-off of either benefiting the environment or the economic state of the farmers (Vanderhaegen et al., 2018). This research may support the idea of such a trade-off as it suggests that the sustainability certifications that aim to address both social and environmental sustainability (Fairtrade and Rainforest Alliance) are too weak in their environmental criteria to conserve biodiversity.

It can be understood why rustic coffee and traditional polyculture coffee are the only two systems with clear biodiversity conservation benefits over sun coffee as they are the systems that most resemble native forest (Moguel & Toledo, 1999). While commercial polyculture systems have multiple different tree species forming the canopy they are not representative of the native forest, instead offering either secondary sources of income or benefiting the growth of coffee bushes (Moguel & Toledo, 1999). Shaded monoculture and sun coffee are both variations on monoculture crop systems. Although shaded monoculture has an overstory comprising of one dominant tree species it is still a highly modified habitat, explaining its similar levels of species loss to sun coffee. It is widely reported in existing literature that agroforestry systems, and more specifically shade coffee, is an important land sharing technique in which crops can be grown profitably while also conserving biodiversity (Mcneely, 2004; Perfecto et al., 2007). The results of this study challenge this belief and suggest a need for more nuance in discussing the role of shade in promoting species conservation in coffee systems. This is because the diversity of the agroforest and the extent to which it is native appear to determine its success in conserving biodiversity, rather than the presence of shade trees themselves.

Other correlates of biodiversity loss

Studies of biodiversity are complicated by the many factors that influence biodiversity and the taxonomic differences in response to ecological disruption (Olson et al., 2001; Stork et al., 2009). It is for this reason that this study captured as wide a range of taxa across geographical locations as possible to best represent the response of ecosystems to coffee management.

Taxon of study explained some differences in species loss where plants consistently experienced highest levels of species loss while animals varied in their response but primarily experienced loss. There were few studies on fungi but they all experienced species gain. It is unclear if this is due to insufficient data but has been suggested that as fungal diversity is regionally distinct the results from one site should not inform on the overall perceptions of how fungi respond (Karun & Sridhar, 2016). Therefore further studies on fungal diversity in coffee systems are necessary to draw any conclusions on their response to land use change. It is expected that plants would suffer the greatest levels of species loss due to conversion of forest to coffee systems, as they are actively removed to create the coffee systems. Animals suffer more indirectly as they may experience loss of habitat due to the land use change, with certain taxa suffering more from the change than others. Birds and mammals experienced species gain from coffee management. This trend of increasing bird species richness has been seen in the least intensive coffee systems, rustic coffee, which may be explained by the added vegetational complexity of coffee bushes in the native forest (Petit et al., 2003). Their resilience to change can be explained by their ability to disperse more easily than non-flying animals. It is possible mammals experienced some species gain as generalist species such as rodents and marsupials might increase in prevalence (Magioli et al., 2021). It is also possible that the relatively high diversity of birds and mammals is explained by spill over of species from forest if the coffee site is adjacent to a forest (Komar, 2006). This hypothesis is supported as similar levels of species richness have been found in coffee sites as their nearby forest fragments (Tejeda-Cruz & Sutherland, 2004). To account for this, a further study that measures distance of forest from coffee would have to be conducted. However, this was insufficiently reported to include in my analysis.

Higher rainfall and higher temperature are both associated with areas of higher biodiversity (Antonelli & Sanmartin, 2011; Pearson & Carroll, 1998). Therefore, land conversion in high rainfall regions is likely to experience more species loss than in an area of lower rainfall. The fact that this trend is revealed in this analysis suggests that despite the dataset being small it is still capturing expected trends in biodiversity. This is an indication that the results from this dataset are in line with expected species responses. While there was a trend of higher temperature correlating with higher species loss, it was not significant. It is possible this trend was not strong as the range of temperatures in this study was too small, given that all study sites were in similar latitudinal ranges and coffee grows in a narrow temperature range (Pham et al., 2019).

The continent in which coffee is grown did not have any robust or consistent effect on species richness which is likely because within each of the continents included there is a wide range of ecosystem types and local biodiversity levels. This is too large a regional measure to explain variation in biodiversity, let alone explain different levels of species loss (Olson et al., 2001). It would be more informative to consider the ecoregion level however the dataset was too small for this analysis (Olson et al., 2001). Additional studies would be needed for this further analysis.

Importance of Modelling Methods

The results of this study also revealed important issues to consider when modelling ecological data. The models in this study were a continuation of the modelling conducted by De Beenhouwer which I expanded and improved upon. Validating their methods against their own raw data revealed that the assumptions of their models were not met. Mixed effect models assume normal distribution of residuals (Schielzeth et al., 2020). However, when inverse variance weighting is applied to the model the residuals deviate from the normal distribution, while in the unweighted model they closer follow a normal distribution. When the unweighted model was used in the assessment of management intensity against species loss the reported significance was lost. While mixed effect models are robust to deviations from assumptions (Schielzeth et al., 2020) the unweighted model was a better fit than the weighted model. Furthermore, the assumptions of inverse variance weighting itself are not necessarily met by this dataset. We cannot assume that a study with a larger sample size is more representative than a study with a smaller sample size when each study consists of a different combination of agricultural, geographic, and taxonomic factors. It is for this reason that my analysis deviates from the modelling outlined by De Beenhouwer, as I chose to use unweighted models.

The funnel plot suggested slight bias in studies included in the analysis. This can likely be explained by the unbalanced dataset in which certain taxa, such as arthropods, dominated the dataset as did sites in Latin America. The slight bias in the funnel plot likely reflects the regional and taxonomic publication bias in the literature.

Areas for further study

An extension to this analysis should increase the size and range of the dataset by searching for grey literature, increase the number of search strings used to retrieve sources, and search for non-Western literature, potentially in different languages, to better represent coffee production in South Asia and Africa. Furthermore, as the dataset is unbalanced, a weighting system setting proportional representation to coffee growing regions, taxon, and coffee management practices could be added to make the results representative of global coffee production. This form of weighting for taxon of study is utilized in the LC-Impact method of estimating biodiversity loss from geography and land use type (Chaudhary et al., 2015), and weighting by prevalence of agricultural practice has been applied in the (Poore & Nemecek, 2018) meta-analysis of environmental impacts of food systems. A continuation of this analysis would involve use of Bayesian statistics in which the slight deviation from a normal distribution could be addressed as these statistics are more flexible than the conventional frequentist statistics (Dunson, 2001). As this dataset has many variables that are expected to influence species loss, prior assumptions, such as different taxonomic responses to land use change or different associations of species loss with management practices, could be incorporated into the analysis with a Bayesian approach.

Species richness was used as a measure of biodiversity because it is the most widely reported and simplest way of describing regional biodiversity (Gotelli & Colwell, 2001). While other measures of biodiversity such as species evenness or species composition might be more informative, they were much less prevalent in the literature and therefore insufficiently reported for study in this large-scale analysis. Species composition was studied by De Beenhouwer. However, I did continue this analysis of the effect of land use change on forest specialist versus generalist species because De Beenhouwer almost exclusively considered all tree species as a forest specialist. This is a biologically uninteresting method of measuring effect on forest specialists as it is implicit that in a more intensive agroforestry system there are fewer tree species. A better analysis of species composition would consider the effect on forest specialist species that are indirectly affected by management practice, however there was insufficient reporting of such species composition in the literature.

There was also a lack of studies reporting both species loss and associated yield at a given site, therefore it is not possible to comment on the economic or market viability of coffee systems that better conserve biodiversity. This separation of yield reporting in life cycle analysis literature and species richness reporting in biodiversity literature needs to change to give conservation a holistic perspective. At the present, a handful of studies consider the relationship between level of shade and yield in coffee systems but even fewer consider the direct relationship between species richness and yield (Gordon et al., 2007; Méndez et al., 2009; Souza et al., 2012). This limits the ability to make practical policy-oriented suggestions that both consider social and environmental needs.

Given the growing demand for coffee it is imperative that further research on their environmental impacts considers not just species richness but the effects on vulnerable species. It is also important that species richness measurements on coffee site account for proximity to native forest as it is possible that dispersing species are regularly passing through from the forest but not able to survive in the plantation, thereby inflating the perceived biodiversity (Komar, 2006). A wider range of taxa need to be studied to understand the extent of species loss at a broader scale, and more research should be done on the impact of sun coffee, which is the dominant production system but made up just under 10% of the dataset. It is also crucial to consider the economic viability of the different coffee management practices and the role of certification schemes in equating profitability with environmental sustainability.

Efforts to quantify biodiversity impacts of agriculture are rapidly expanding with global assessment platforms like the Biodiversity Intactness Index (BII) and Life Cycle Impact (LC-Impact) methodology which uses species-area curves to estimate land use-associated biodiversity loss (Chaudhary et al., 2015; Hudson et al., 2017). These enable large scale estimates of anthropogenic biodiversity loss due which is impractical to carry out with species richness measures. However, these estimates are highly approximative due to their degree of removal from actual species data. Furthermore, the LC-Impact model of estimating biodiversity loss is too far derived from the baseline species richness values that the model cannot be tested against raw biodiversity data to check for accuracy (Chaudhary et al., 2015). While coffee, a globally important commodity, is known to cause biodiversity loss, the limitations of existing large scale biodiversity impact estimators and lack of sufficient raw data on coffee associated biodiversity loss mean it is not yet possible to make a globally valid estimate of the biodiversity impact of coffee. This is part of a wider problem of needing to address the biodiversity impact of our food systems with limited knowledge of the extend and drivers of those impacts.

Conclusion

The finding that shade systems, unless resembling natural forests, are not better suited to conserving biodiversity than sun coffee systems suggests that sustainability certifications should increase the rigor of their environmental sustainability criteria. Of the major sustainability certifications only Smithsonian Bird Friendly Coffee requires agroforestry standards rigorous to make a difference in biodiversity conservation compared to sun coffee. In the light of the biodiversity crisis and the threat of the 6^{th} mass extinction (Barnosky et al., 2011), it is important that measures are put in place to assist smallholder farmers to promote biodiversity-friendly agricultural practices. To best support improvements to the environmental sustainability of coffee, data gaps urgently need to be filled, particularly on the ecological impact of Asian and African coffee systems, and in the form of more nuanced measures of biodiversity than species richness. More broadly, incomplete information on the biodiversity impact of agriculture is hindering the urgent need to promote environmentally sustainable agriculture.

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Management Report

I first met with Joseph Poore to discuss doing my masters project on issues of food sustainability in Hilary Term 2021 (henceforth HT21). I decided I wanted to focus on coffee as I knew little of its environmental impact before the start of the project in spite of it being grown in some of the most biodiverse regions of the world. This was an opportunity for me to learn not only about the effects of my own consumption habits but to understand the methods behind deriving large-scale estimates of agricultural impacts such as Joseph's work in Poore and Nemecek (2018) which I was inspired by.

We met again in TT21 to discuss what method I would take to quantify the biodiversity impact of coffee and I was given some reading on methods of estimating large-scale biodiversity impact using life-cycle analysis tools. We discussed deriving a global estimate of coffee-associated biodiversity loss using one of these existing methods in addition to assessing the validity of these methods against trends in raw biodiversity data from coffee sites. I continued reading about methods of estimating biodiversity loss from land use, focusing on methods using species-area relationships to derive these estimates.

Over the summer I had my first meeting with both E.J. Milner-Gulland and Joseph Poore where we discussed my plans of both assessing the validity of species-area relationship models for estimating species loss due to land use, and of using these methods to derive a global estimate of coffee's biodiversity impact. Over summer I continued reading about biodiversity impact estimating methods in order to understand the methodology. During MT21 I began reading about coffee systems and their associated biodiversity impacts. From my reading I found the meta-analysis source on which my methods were later built off, along with other databases and sources on biodiversity impacts of agriculture. I spent the rest of MT21 continuing with my reading and working to understand how the species-area model of estimating species loss from land use (Chaudhary et al., 2015) could be compared and validated against the raw biodiversity data I had found from coffee sites.

As I was struggling to find a way of comparing raw biodiversity data (in the form of species richness data) to the output estimates of Chaudhary et al (2015)'s model, I got into contact with Abisheck Chaudhary in HT22 to discuss how this would work. We discussed my work over email and came to the conclusion that his model of estimating biodiversity loss was too far derived from raw biodiversity data to be compared or validated against it. This resulted in a shift in focus of my project away from using large-scale methods of estimate biodiversity impact. At this point I discussed with Joseph how to adapt my project in the remaining time. We came to the conclusion that it would be best to focus on the existing analyses of species richness loss associated with coffee production. I spend the rest HT22 restructuring and combining existing sources of data on coffee associated biodiversity loss, measured through species richness. I then extended the existing sources with a systematic search in order to update the combined dataset which only extended to 2012. This extended my dataset by another 30 studies. Once I'd finished the systematic search and collated all existing data I began my analysis which was based of the methods of the first and largest coffee meta-analysis I'd found during my initial reading on coffee's biodiversity impact.

The analysis entailed replicating the methods of the meta-analysis I'd extended my dataset from which was achieved using the raw data sent to me from the lead author through personal communications. After replicating results from their paper, I used the same modelling method to analyze my own extended version of the dataset which was not only larger but considered a wider range of covariates. This analysis was finished at the end of easter break of 2022, at which point I began my write up.

During TT22 I wrote up my results, starting with the methods and introduction as I was still testing out variations of my models to determine the best, most robust modelling method. Once I'd set on my model I wrote up my results followed by my discussion and had the first draft of my dissertation in by the end of 2nd week, TT22. However, Joseph, who was acting as my primary supervisor, took leave from the start of May so was unable to read my dissertation draft. I was still able to send my draft to E.J. who gave me feedback in 3rd week TT22. After this I edited the draft and conducted final edits and formatting before submitting at the end of 4th week TT22.

Supplementary Materials

S.1 Replication of meta-analysis results

De Beenhouwer et al (2013) results replication for model validation. These figures (Fig 1 and Fig 2) represent mean Hedges' G of all species for the comparison of forest to agroforestry systems and then of agroforestry to plantation systems in an inverse variance weighted model. This was recalculated using the raw data obtained through personal communications with M. De Beenhouwer.

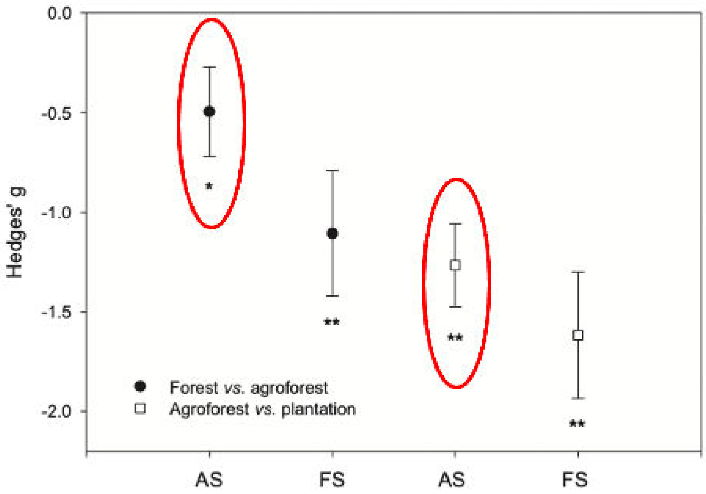

*Figure S1. The figure in De Beenhouwer et al (2013) I recreated with their raw data to validate my model in which the AS (all species) bars circled in red are relevant. Bars are standard errors and the stars indicate values differed significantly from zero (*p < 0.05 and **p<0.01)*

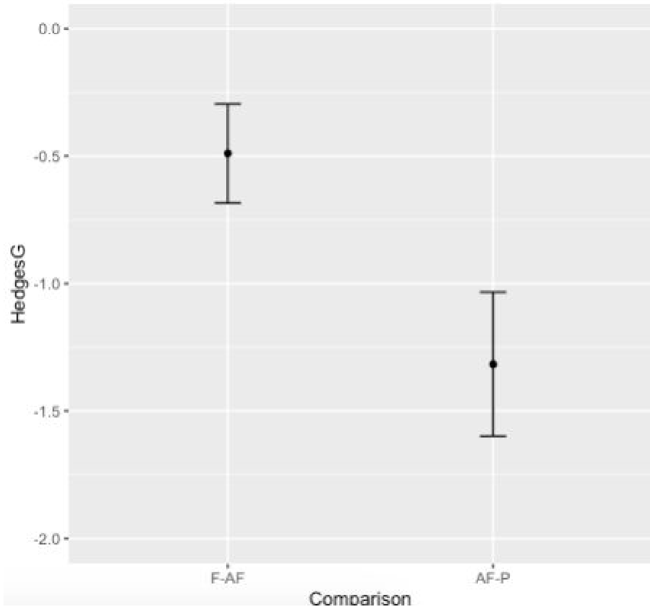

Figure S2. The recreated version on the De Beenhouwer et al (2013) figure in which I was able to replicate the values when using an inverse-variance weighted mixed effect model. F-AF represents forest vs. agroforest comparison and AF-P represents agroforest vs plantation comparison

S.2 Testing validity of replicated models

The residuals of my recalculated model were tested for normality and found to deviate in the normal quantiles plot and the Shapiro-Wilk and Kolmogorov-Smirnov normality tests (p-value < 0.0001 and p value = 0.013) (Fig 3). The model was then run without weighting and the residuals in the normal quantiles plot was closer to a normal distribution however the significance changed (Fig 4). The normal quantiles plot of the unweighted model had smaller tails than the weighted model (Fig 5) and the residuals were still non-normal according to the Shapiro-Wilk test but normal according to the Kolmogorov-Smirnov normality test (p value = 0.0004 and p value = 0.126).

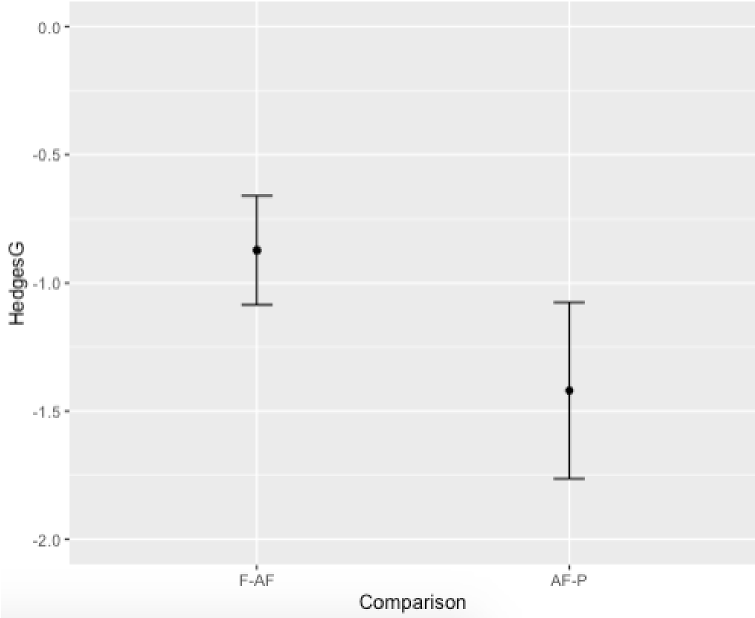

Figure S3. Replicated Fig 2 of De Beenhouwer without inverse variance weighting, showing greater species loss for F-AF and less difference between F-AF and AF-P.

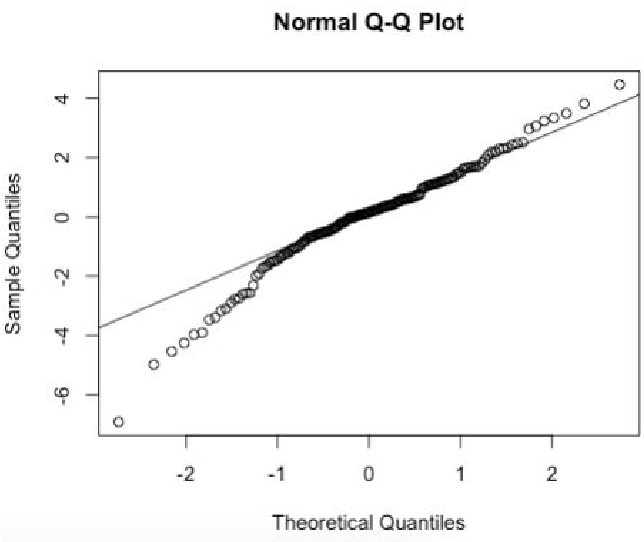

Figure S4. Normal quantiles plot of the residuals of the unweighted version of the replicated figure for De Beenhouwer et al (2013) figure 2.

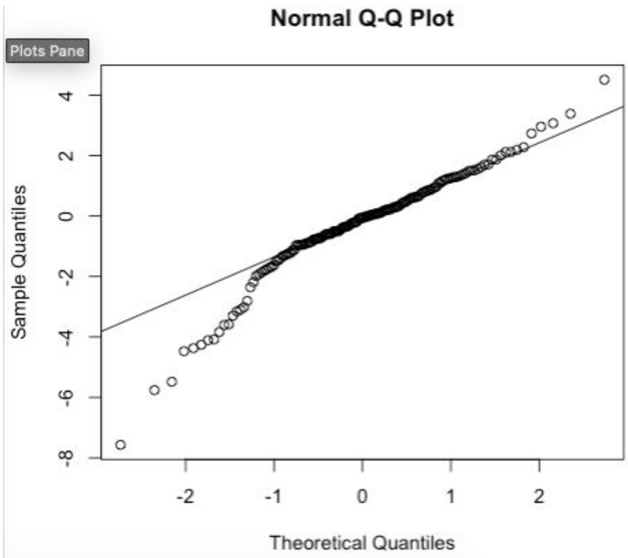

Figure S5. Normal quantiles plot of the residuals of the replicated De Beenhouwer figure 2 with the inverse variance weighting. Shows stronger deviation of residuals from a normal distribution than in the unweighted model

S.3 Model selection for analysis

Further reason to use unweighted models in my analysis than the inverse-variance assumptions not necessarily applying to the nature of my dataset is in the normal quantiles plot of residuals of the unweighted and inverse variance weighted model. There is a strong tail in the normal quantiles plot of the inverse variance weighted model that is weaker in the unweighted model, therefore the unweighted model is a better fit. This is shown for the multivariate mixed effect models with study as the random effect and the following fixed effects: management practice, taxa, continent, rainfall, and temperature.

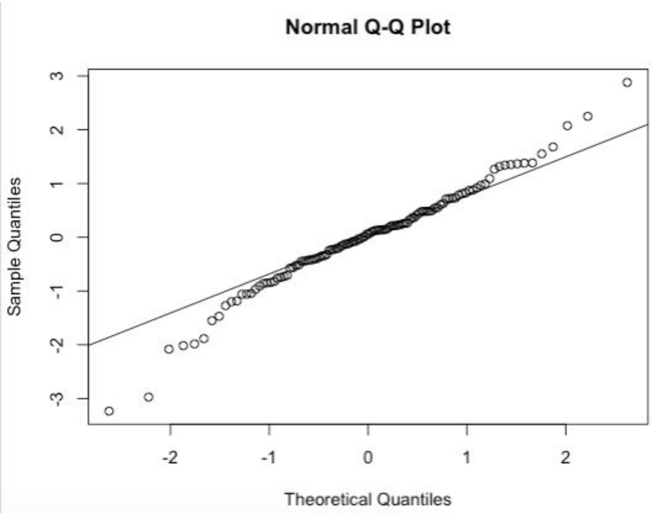

Figure S6. Normal quantiles plot of the unweighted multivariate mixed effect model depicting a light tail but relatively normal distribution of residuals. Shapiro-Wilk test suggests residuals are approaching normal distribution and Kolmogorov-Smirnov test suggests residuals are normally distributed (p = 0.047 and p = 0.50)

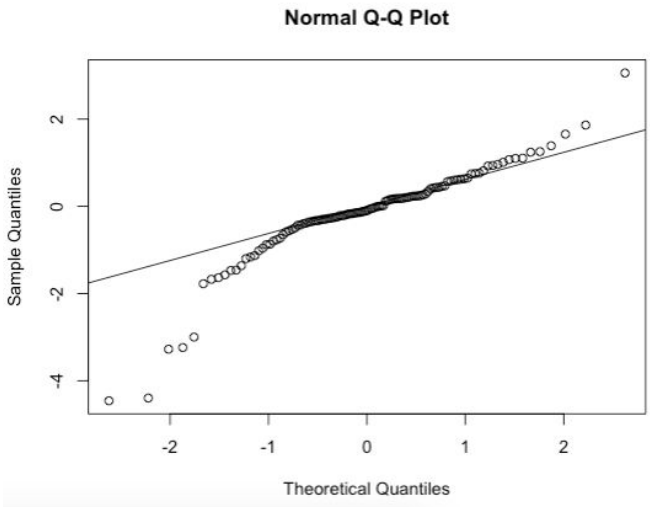

Figure S7. Normal quantiles plot of the residuals of the inverse variance weighted multivariate mixed effect model depicting stronger tails than the unweighted model. Shapiro-Wilk and Kolmogorov-Smirnov normality test results were both non-normal (p < 0.0001 and p = 0.0048)

S.4 Effect of Management Including Tree Species

A univariate model of Hedges' G against management type in which tree species were included in the dataset reveals the same trend of results as seen in the univariate and multivariate model without trees. (Table S1) presents the values from this model which was not reported in the main test as measurement of tree species richness is only relevant in the diverse agroforestry systems of coffee, not the two types of coffee monoculture.

Table S1. Mixed effect model output for Hedges' G with management practice as fixed effect (dataset with tree species richness) P-values are of difference from sun coffee. Values in bold represent a significant effect

Management practice	Mean Hedges' G	Standard error	*p-value*
Sun coffee	-1.4322	0.4999	
Shaded monoculture	-1.3008	0.5534	0.817
Commercial polyculture	-1.5921	0.5485	0.771
Traditional polyculture	-0.8097	0.5664	0.274
Rustic coffee	0.0294	0.6948	**0.037**