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RESEARCH ARTICLE

Complementing urban agriculture and green spaces is important for ecosystem functions and biodiversity in cities: A systematic review and meta-analysis

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Abstract

1. As cities expand, they encroach on agricultural land, impacting food production and natural habitats. While urban agriculture could help address these issues, the impact of increased food production on ecosystem functions—particularly with regards to soil conservation, climate regulation and biodiversity—remains poorly understood.
2. To fill this knowledge gap, we conducted a meta-analysis to quantify the effects of urban agriculture on ecosystem functions (i.e. soil quality and climate regulation) and biodiversity (i.e. plants, birds, mammals, arthropods and insect pollinators). We estimated ecosystem functions and biodiversity in urban agriculture by comparing them with other urban green spaces (e.g. parks, residential gardens, green roofs) and conventional rural farms.
3. Our results overall showed that urban agriculture is in an intermediate state between the other urban green spaces and conventional rural farms in terms of environmental impacts. Urban agriculture had a positive effect on the studied ecosystem functions relative to conventional rural farms (+25%, CI_{95}: +12% to +39%), but was equivalent in its provision of these functions to other green spaces. Urban agriculture also had a positive effect on biodiversity compared to conventional rural farms (+38%, CI_{95}: −10 to +111%), but a negative effect compared to green spaces (−12%, CI_{95}: −29 to +8%). Specifically, urban agriculture

had 39% (CI_{95}: −59 to −10%) lower plant diversity and abundance than that seen in other green spaces.

4. *Practical implications*. Our results suggest that urban agriculture's contribution to urban sustainability requires complementarity with other green spaces to effectively support ecosystem functions and biodiversity in cities. We also detected substantial issues with the classification and characterisation of urban agriculture plots. Improving conceptual and methodological consistency will continue to be a crucial challenge for urban agriculture research.

KEYWORDS
sustainability, urban agriculture, urban biodiversity, urban ecology, urban green spaces, urban parks

1 | INTRODUCTION

Urban areas have experienced a net expansion of 80% over the last 30 years, with an estimated 70% of this directly encroaching on agricultural land, and so impacting food security (Liu et al., 2020; Mulya et al., 2023). This change in land use may actively drive intensification of surrounding agricultural areas to support food production (Foley et al., 2011). This poses risks to ecosystem functions and habitat conservation, leading to a chain of negative impacts on biodiversity and human well-being (Tscharntke et al., 2012). Urban renewal and development plans increasingly prioritise strategies that maximise both ecological and socio-ecological benefits (Langemeyer et al., 2021). Such strategies include designing, planning and managing green infrastructure like parks and roadside verges, private gardens, vacant lots, rooftop gardens and remnant woodland patches (Priyadarshana et al., 2025). In particular, urban agriculture represents an alternative green infrastructure that emphasises the production of crops and livestock integrated into urban economies and ecosystems (Mougeot, 2000). Unlike conventional rural agriculture, urban agriculture emphasises both sustainability and multifunctionality (Lovell, 2010; Pradhan et al., 2023). This makes it relevant to 11 of the 17 UN sustainable development goals (SDGs) (Viana et al., 2022), especially contributing to mitigating the loss of ecosystem functions caused by urban expansion.

Recent research suggests that urban agriculture is a highly diverse conglomerate of practices that, while achieving relatively high yields, can in some cases be less sustainable than conventional rural agriculture (Dorr et al., 2021, 2023; Goldstein et al., 2016; Hawes et al., 2024; McDougall et al., 2019). Additionally, while urban agriculture's contribution to food security and carbon footprint is a well-studied topic, there is a significant knowledge gap in terms of its effects on ecosystem functions and biodiversity (Nicholls et al., 2020). Existing qualitative reviews suggest positive effects of urban agriculture on biodiversity, particularly in plants and insect pollinators, which are associated with the planting of local species and high floral diversity within these systems (Clucas et al., 2018; Lin et al., 2015). Research investigating trade-offs between ecosystem functions has shown that urban agricultural systems (e.g. community gardens) interact with natural landscape elements to promote habitat connectivity with benefits for invertebrates providing pest control and pollination (Jha et al., 2023). Conversely, previous studies have highlighted the importance of traditional green infrastructure such as urban parks in enhancing ecosystem functions, particularly soil, water and air quality (Elmqvist et al., 2016; Evans et al., 2022), resulting in human health benefits (Fischer, 2016). Currently, a quantitative synthesis that compares the effects of urban agriculture on ecosystem functions and biodiversity relative to urban green infrastructures and conventional rural agriculture is lacking. The meta-analysis provided in this paper aims to address this knowledge gap and to test the following two hypotheses:

> **Hypothesis 1.** Urban agriculture may have a negative impact on ecosystem functions compared to green infrastructure, but it could offer positive outcomes when compared to conventional rural agriculture.

High vegetation cover typical of green infrastructure compared to urban agriculture could effectively improve climate regulation and soil conservation (Elmqvist et al., 2016; Evans et al., 2022).

> **Hypothesis 2.** Urban agriculture could increase biodiversity compared to both urban green infrastructures and conventional rural agriculture.

High floral diversity in urban agriculture could provide benefits particularly to plants and insect pollinators, whereas more homogeneous vegetation cover typical green infrastructure and conventional rural agriculture may negatively impact biodiversity (Borysiak et al., 2017; Clucas et al., 2018; Lin et al., 2015; Royer et al., 2023).

2 | MATERIALS AND METHODS

2.1 | Literature search and inclusion criteria

We employed the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) guidelines to screen articles on the effects of urban agriculture on biodiversity and ecosystem functions, published in the past 20 years (Shamseer et al., 2015). We selected studies according to three criteria: (1) studies focusing on farms or gardens located in an urban area and using soil or other substrate to grow their crops, including hydroponic or other growing systems using an inert substrate, (2) the farm or garden was partly or wholly planted with crops, that is, excluding farms or gardens that only raise livestock and (3) the study focused on a comparative approach, by which urban agriculture is compared to either green infrastructure and/or conventional rural farms. We conducted the literature search in the Scopus (scopus.com/) and Web of Science (webofscience.com/) databases using the following search terms: "Urban agriculture" OR "Urban farming" OR "Agroecology" OR "Ecosystem services" OR "Biodiversity" OR "Birds" OR "Plants" OR "Mammals" OR "Insects" OR "Invertebrates" OR "Vertebrate" OR "Herbivores". These search terms were applied to the titles, abstracts and keywords of articles in English language from 2000 to May 2024. After removing duplicates, we retrieved 1288 potential studies for data compilation. From these, 21 studies matched our inclusion criteria, which yielded 243 effect sizes to be included in the final meta-analysis. See Supporting Information File S1 for more details.

2.2 | Data compilation

To assess the impacts of urban agriculture on ecosystem functions and biodiversity relative to green infrastructure and conventional rural farms, we categorised studies into two main themes based on whether they focused on (i) 'ecosystem functions', or (ii) 'biodiversity'. We labelled this variable as 'topic'. To categorise ecosystem functions, we used 'The Economics of Ecosystems and Biodiversity' (TEEB) as the principal ecosystem service framework and supplemented this with additions from the 'Millennium Ecosystem Assessment' (MEA) framework (Evans et al., 2022). The recorded comparative studies only assessed two functions, climate regulation and soil conservation (e.g. by measuring soil quality, temperature; see Table S1 for more details). To assess the impact of urban agriculture on biodiversity, we retrieved information on biodiversity metrics collected in these comparative studies. Studies quantifying the effects on biodiversity measured activity levels, species abundance or species richness. As our goal was to determine the effects on different taxa, we classified these different biodiversity metrics (abundance, activity, diversity and richness) into five broad taxa: plants, birds, arthropods, mammals and insect pollinators (Table S1). We also characterised the reference systems for comparison with urban agriculture. We classified studies into two classes depending on their comparison system, that is, (i) 'conventional rural farms', also referred to as traditional farms or industrial agriculture, which are large scale, high-input, intensive agricultural systems, located in rural areas adjacent to urban areas (Le Campion et al., 2020; Misra & Ghosh, 2024); and (ii) 'green infrastructure', which refers to natural or semi-natural green spaces within built grey infrastructure, such as green spaces, parks, residential gardens, green walls and roofs, vacant spaces, roadsides and pathways and remaining patches of natural vegetation (EEA, 2011; Evans et al., 2022).

We also assessed to what extent different types of urban agriculture impacted ecosystem functions and biodiversity. We recorded three main urban agriculture types based on their cultivation and management approaches (Dorr et al., 2023; Evans et al., 2022; Royer et al., 2023): (i) 'commercial farms', defined as productive spaces led by farmers with multiple goals, including food production as well as social and environmental functions, where a portion or all the food produced is sold; (ii) 'collective gardens', characterised by non-commercial purposes on land cultivated by the community or institutions such as Universities or Schools; and (iii) 'individual (or private) gardens', characterised by non-commercial purposes with their land divided into plots managed by individual gardeners, including allotment plots and home gardens. Gardens or farms growing plants in urban areas, whose characteristics did not clearly fit into the above three categories, were classified as 'other farms'. We did not formulate specific predictions about these three types of urban agriculture as they are often loosely defined and previous studies have highlighted that these systems are highly variable in terms of productivity and environmental impacts (Hawes et al., 2024; Royer et al., 2023). However, our focus was on assessing whether these different types of urban agriculture produced clearly different patterns when compared to green infrastructure and conventional rural farms.

To calculate the effect sizes, the means, standard deviations (or standard errors), and sample sizes, were extracted for each contrasting treatment (i.e. urban agriculture vs. green infrastructure vs. conventional rural farms). We extracted the data from text, tables or from supplementary files, and used 'Getdata Graph Digitizer' version 2.26 (getdata-graph-digitizer.software.informer.com/) when these data were extracted from graphs. Any studies with missing data were excluded from the analysis. Finally, we also extracted bibliographic information and geographical context (city/country) for each study. A complete list of these data types extracted, including a short description, type of data, and an example, can be found in Supporting Information File S2. We also intended to record plot size to examine its effect on the average effects estimated by our models, but this was not feasible due to the lack of consistent data across many studies.

2.3 | Effect sizes calculation

We calculated the log transformed response ratio of means (log RRs) and their variances to quantify the effects of urban agriculture on ecosystem functions and biodiversity components relative to the comparison systems (i.e. green infrastructure and conventional rural farms) across each study (Hedges et al., 1999). These effects sizes were computed using the 'escalc' function in the 'metafor' package v4.4-0 (Viechtbauer, 2010). As we used mean measures from diverse quantitative metrics, the ratio between urban agriculture and comparison systems means measures could be interpreted as positive or negative depending on the used metric. For example, if mean CO_2 emissions were higher in conventional rural farming than in urban agriculture, the computed effect size would be negative, but the net effect of urban agriculture on regulating ecosystem services would be positive. We thus multiplied the output effect sizes by −1 when the direction of the effect size needed to be inverted to accurately represent either positive or negative effects of urban agriculture.

2.4 | Statistical analysis

To test our hypotheses, we utilised a multilevel mixed-effects modelling approach with the 'rma.mv' function from the 'metafor' package (Viechtbauer, 2010). As multiple effect sizes were reported within each study, we used a random structure consisting of effect size ID nested within study ID, which allowed us to account for both within- and between-study variances (e.g. plot size, taxa) when estimating the mean effect (Raudenbush, 2009; Viechtbauer, 2007). Effect size estimates were deemed significantly different from zero if their 95% confidence intervals did not encompass zero.

We first fitted a null model using the full dataset and with no moderators to obtain the overall mean effects and to ascertain sources of heterogeneity linked to our random structure, and other using sampling variances as a moderator. These models were non-significant, suggesting no significant publication bias (Table S2). We also ran a model using the full dataset and with 'topic' as the moderator (see above) to assess overall effects of urban agriculture on ecosystem functions and biodiversity regardless of the comparison system (see Section 3). We then ran a set of models including different moderators to explain the heterogeneity in the estimated true effect and to test our hypotheses (Tables S3 and S4). We estimated the effects of urban agriculture on ecosystem functions and biodiversity effects separately, as they cannot be compared directly. All the analyses were performed in R statistical environment (r-project.org/; R version 4.3.0).

2.4.1 | H1. Effects of urban agriculture on ecosystem functions

To test this hypothesis, we ran a model only using data from studies focusing on ecosystem functions and with comparison system as a moderator (green infrastructure or convention rural farms). The sample size for comparative studies reporting ecosystem functions was relatively small (number of studies, $n=3$; and effect sizes, $k=27$), so the model did not converge well due to the low number of studies (see Supporting Information File S3). Moreover, studies on climate regulation have focused solely on comparing urban agriculture with green infrastructure, whereas studies on soil conservation have looked at urban agriculture in relation to both green infrastructure and conventional rural farms. Consequently, we could not distinguish the effects on functions from those associated with the comparison systems. Still, we ran an additional model using function type (climate regulation and soil conservation) as a moderator, yet with the same convergence issue (Supporting Information File S3).

2.4.2 | H2. Effects of urban agriculture on biodiversity

To test this hypothesis, we ran a model only using data from studies focusing on biodiversity with the comparison as a moderator system (green infrastructure or conventional rural farms) ($n=18$, $k=216$). We were also interested in examining these effects for the different taxa, so we ran an additional model including 'taxa' (five levels: plants, birds, arthropods, mammals and pollinators) as a moderator. We used this approach (i.e. running two separate models rather than running directly a model with the interaction between these two variables), because the sample size for certain combinations of taxa and comparison systems was small. Still, we ran a model including the interaction between taxa and comparison system as moderators to determine if any overall effects on the different taxa were dependent on the comparison system. We ran two additional models, one with 'urban agriculture type' (four levels: commercial farms, collective gardens, individual gardens and other farms) as a moderator to determine if these different systems have differential effects on biodiversity, and another with 'continent' as a moderator (four levels: Asia, Europe, North America and South America) to determine if the effects of urban agriculture on biodiversity vary across geographical regions.

We calculated the estimated marginal means for each level of the moderators and performed Tukey post-hoc pairwise comparisons between them using the 'emmeans' package v1.10.2 (Lenth, 2017). We finally extracted and converted the mean estimated log response ratios to percentage change for each factor level to facilitate interpretation of our results (Hedges et al., 1999).

2.5 | Publication bias analysis and sensitivity analysis

We created a 'funnel plot' using the null model (see above) to visually check for publication bias (Viechtbauer, 2010). To test the 'funnel plot' asymmetry statistically, we ran a meta-regression using standard errors (SEs) as a moderator (Egger et al., 1997; Thompson & Sharp, 1999). We found no evidence for publication bias using either approach. We also checked for potential outliers influencing our results in each model by examining the Cook's distances (Viechtbauer & Cheung, 2010). A single influential effect size was identified in the biodiversity model (Cook's Distances >1) (Chatterjee & Hadi, 1986). We excluded this effect size and reran the model; however, the results did not significantly change between the models. Therefore, we reported the results from the full dataset, as it provides more statistical power. Finally, we profiled restricted log-likelihood to assess model performance. The log-likelihood function measures how well the model explains the observed data, given certain parameter values, with a marked peak in the profile plot indicating precise estimates (Pawitan, 2001). All the models displayed optimal log-likelihood profiles (see Supporting Information File S3 for diagnostic plots).

3 | RESULTS

The studies included here spanned all continents apart from Australia and Antarctica (Figure 1). Most studies were conducted in North America (n=12, k=123) and Europe (n=4, k=89), but our sample also contained studies from Global South countries, especially from Asia (n=4, k=28). The Southern Hemisphere was underrepresented. Our database included records from Mexico (n=1, k=4), Hungary (n=1, k=26) and South Korea (n=1, k=9). However, the sample size was relatively small for all countries except the United States (n=10, k=107) and China (n=2, k=16).

3.1 | H1. Effects of urban agriculture on ecosystem functions

We did not observe overall significant effects of urban agriculture on ecosystem functions (Table S5). Yet, we found a positive impact of urban agriculture on ecosystem functions compared to conventional rural farms, improving its performance by 25% (CI_{95}: +12% to +39%) on average, while it did show negligible effects compared to green infrastructure (Figure 2A; Table S6). These effects of urban agriculture on ecosystem functions were significantly different

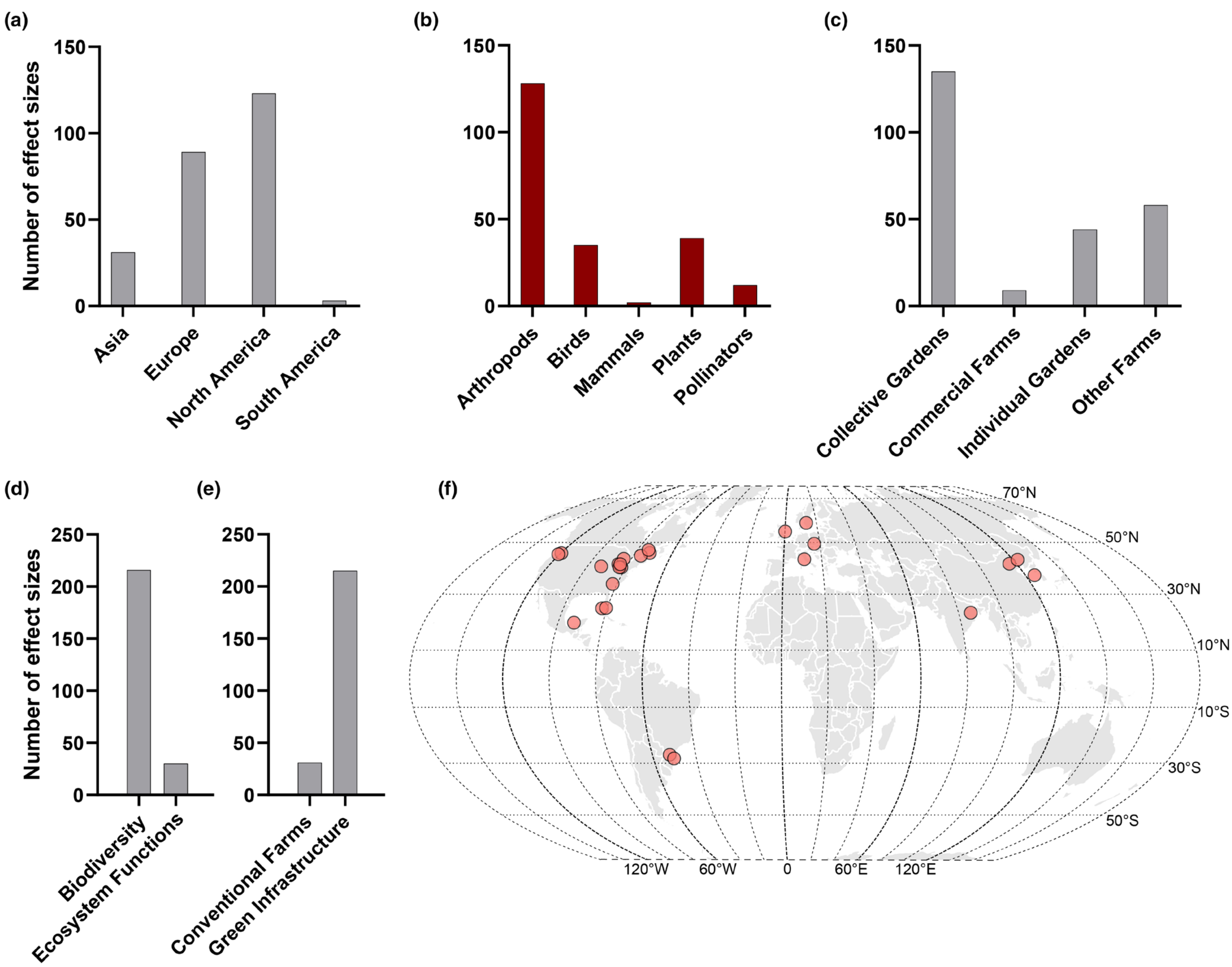

FIGURE 1 The number of observations across different contexts. (a–e) Summary of the number of observations (that is, effect sizes) in the database according to: Continents (a), study taxa (b), urban agriculture types (c), the main study topics (biodiversity or ecosystem functioning) (d) and the main study system types used for comparisons with urban agriculture (e). (f) The locations of field studies show broad global coverage of studies included in the database.

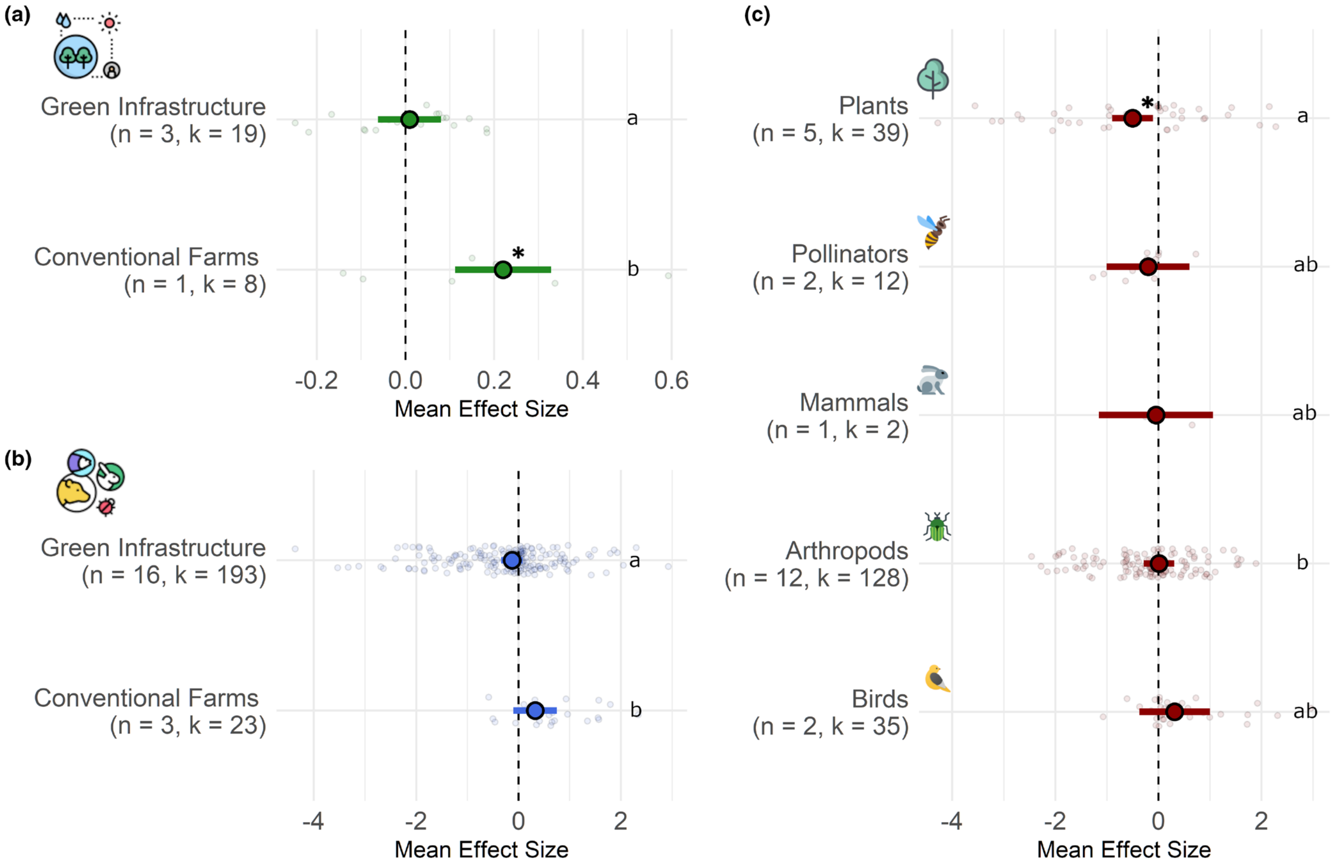

FIGURE 2 (A–C) Mean effect sizes (response ratios, RRs) of urban agriculture across the different themes assessed in this study: Overall effects on ecosystem functioning according to comparison system type (A), overall effects on biodiversity according to comparison system type (B) and effects across five main taxa categories (C). The numbers of studies (*n*) and effect sizes (*k*) for each driver are shown in parentheses. The displayed points and error bars represent the mean predicted values and the 95% confidence intervals, respectively, from a meta-analytical model with separate random intercepts for study. Asterisks denote significant effects. The letters denote which moderator levels were significantly different between them.

when compared to conventional rural farms or green infrastructure (Table S7). Regarding the analysed ecosystem functions, urban agriculture had a marginally significant positive effect on soil conservation (with effect sizes from both conventional rural farms and green infrastructure) and negligible effects on climate regulation (with effect sizes only from green infrastructure) (Table S8).

3.2 | H2. Effects of urban agriculture on biodiversity

We did not find overall significant effects of urban agriculture on biodiversity (Table S5), nor significant differences in the effects of urban agriculture on biodiversity compared to both conventional rural farms and green infrastructure (Figure 2A; Table S9). However, the effects on comparative systems were significantly different, meaning that urban agriculture reduced biodiversity by 12% (CI_{95}: −29 to +8%) compared to green infrastructure, but increased by 38% (CI_{95}: −10% to +111%) compared to conventional rural farms (Table S10). We recorded a negative effect of urban agriculture on plants compared to green infrastructure and conventional rural farms, decreasing their diversity and abundance by 39% (CI_{95}: −59% to −10%) (Figure 2B; Table S11). The effects on other taxa (arthropods, birds, mammals, or insect pollinators) were non-significant. We also recorded a significant difference between the effects on plants and arthropods (Table S12). No other taxa showed significant differences between them. This negative effect of urban agriculture on plants was only apparent compared to green infrastructure but not compared to conventional rural farms (Table S13). Finally, we did not record any significant effects on biodiversity depending on the different urban agriculture types (Table S14), nor across continents (Table S15).

4 | DISCUSSION

Our study is the first to quantitatively analyse at a global scale the effects of urban agriculture on ecosystem functions and biodiversity in relation to both green infrastructure (e.g. amenity parks) and conventional rural farms. In line with our predictions, urban agriculture represented a mid-point between green infrastructure and conventional rural farms in terms of their impacts, with

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conventional rural farms showing the most negative environmental effects for both biodiversity and ecosystem functions. We did not record, however, a significant positive effect of urban agriculture on biodiversity compared to green infrastructure. More specifically, we recorded a negative effect of urban agriculture on plants, decreasing its abundance and diversity by 39%, particularly compared to green infrastructure. We recorded no other relevant effects of urban agriculture on other taxa (i.e. arthropods, birds, mammals), nor groups that provide crucial ecosystem functions for agriculture such as insect pollinators. Overall, this suggests that, to fully achieve its sustainability goals, urban agriculture likely needs to be combined with other green infrastructure types to exploit synergies between these systems (Jha et al., 2023; Qiu et al., 2024). This would ensure that urban agriculture positively contributes to ecosystem functioning and biodiversity conservation in urban environments (Evans et al., 2022; Langemeyer et al., 2021).

4.1 | Urban agriculture as a mid-point between green infrastructure and conventional farms in terms of biodiversity conservation

We found that the overall effects of urban agriculture on biodiversity were intermediate compared to green infrastructure and conventional farms. Intuitively, urban agriculture might represent a less extractive land use type than conventional farms, although there is great variability in how intensively land is managed by individual farmers (e.g. Dorr et al., 2021; Hawes et al., 2024; McDougall et al., 2019). In our sample, when combining all available biodiversity data, we found evidence of this pattern, with conventional farms displaying the most negative effects on biodiversity. Still, green infrastructure had a more positive effect on biodiversity compared with urban agriculture. This finding challenges the notion that urban agriculture might have a particularly positive effect on biodiversity, especially for certain taxa such as insect pollinators, when compared to green infrastructure (Clucas et al., 2018; Lin et al., 2015). However, no study had assessed urban agriculture's environmental performance compared to conventional farms and green infrastructure. Previous studies have suggested that high floral diversity characteristic of urban agriculture, combined with low impervious cover, may effectively attract multiple insect species and significantly contribute to protecting urban pollinators (Baldock, 2020; Bennett & Lovell, 2019). Our study fills this gap and shows that urban agriculture can be integrated with green infrastructure to optimise its benefits for biodiversity (Hackett et al., 2024). Urban agriculture's primary focus is food production, and as such provides complementary social benefits outside of the purely amenity value of green infrastructure. However, our results show that the impacts of urban agriculture on ecosystems and biodiversity are less negative than those of conventional farms, while also simultaneously reducing supply chain distances for food production in cities. Further refinements in its management and design practices, similar to how agri-environmental schemes aim to green conventional agriculture, could enable urban agriculture to make a significant contribution to urban sustainability (Qiu et al., 2024).

4.2 | Urban agriculture has relatively positive effects on ecosystem functions

We also recorded more positive effects of urban agriculture on ecosystem functions (soil conservation and climate regulation) compared to conventional farms. When examining these functions separately, this positive effect was particularly evident with regard to soil conservation, suggesting that urban agriculture may involve improved soil management compared to conventional farms (Edmondson et al., 2014). Low sample size for this analysis meant that we were unable to disentangle the effects of these two anthropogenic green space types on ecosystem functions with confidence. Still, our results suggest potential, and possibly significant, differences between urban agriculture and (especially) conventional farms in terms of ecosystem impacts. There are multiple studies highlighting the economic relevance of both urban agriculture and green infrastructure in terms of climate regulation (e.g. Clinton et al., 2018; Xu & Zhao, 2021). Here, we did not detect substantial differences between these systems in that regard. A highly dense vegetation cover characteristic of green infrastructure has been suggested to significantly contribute to temperature regulation and soil protection (Bowler et al., 2010; Evans et al., 2022). However, at least in our sample, these features do not seem to entail a clear advantage over less vegetated systems such as urban agriculture. Also, the provision of ecosystem services is dependent on landscape context and seasonality. For instance, green infrastructure and urban agriculture provide stronger temperature reduction effects compared to conventional farms and are more effective at climate regulation during hot summer months (Hamada & Ohta, 2010). Still, more comprehensive studies that directly compare the effects of each of these anthropogenic green space types on ecosystem functions are needed to confirm these hypotheses.

4.3 | Urban agriculture negatively impacts plant diversity and abundance

Urban vegetation plays a key role in supporting nutrient cycling and water regulation and natural pest control by limiting natural enemies of crops and plants hosting these pests (Dassou & Tixier, 2016; Letourneau et al., 2011; Molina et al., 2023; Priyadarshana et al., 2023; Ratnadass et al., 2012). Moreover, high crop and floral diversity has been consistently linked to positive effects on ecosystem functions and biodiversity in agroecological systems and urban areas (Baldock, 2020; Isbell et al., 2017; Priyadarshana et al., 2024, 2025). The negative effects of urban agriculture on plants recorded in this study may extend to other taxa. Taxa that rely on plant materials for food and shelter, including most arthropods, would likely be negatively impacted by urban agriculture if it replaces more floristically diverse forms of green infrastructure. This sharply contrasts with the suggestions made by preliminary qualitative reviews on the effect of urban agriculture on biodiversity (Clucas et al., 2018; Lin et al., 2015). Overall, our findings regarding to plant diversity may also be related to how land is utilised across different urban agriculture systems. For instance, recent research has shown that while individual gardeners (i.e. those associated with private domiciles) tend to use all available land for ornamental and crop plants, community or collective garden projects focus more exclusively on food production but also often leave large areas uncultivated (Dorr et al., 2023). Urban agriculture crops likely have fewer field margins than conventional farms due to limited land availability in urban areas, especially in regions where agri-environmental schemes are active for conventional farming (Batáry et al., 2015). This limitation may reduce the capacity of urban agriculture to support biodiversity, as the lack of vegetated margins could limit animals movements—the food resources found in field margins play a key role in attracting arthropods and birds (Priyadarshana et al., 2024). The lack of these margins may also reduce the ecological resilience of urban agriculture systems, akin to the effects of field margins loss in intensively farmed areas (e.g. Burel et al., 1998). In this regard, leaving uncultivated areas (e.g. semi-natural and natural habitats) and broad field margins wherever possible could enhance biodiversity (Lee et al., 2022; Priyadarshana et al., 2024; Stanton et al., 2018).

4.4 | Limitations

Research into urban spaces remains in its relative infancy, such that limited sample sizes hindered our ability to detect complex effects on biodiversity. Small sample sizes likely limited our ability to make strong inferences across the different typologies of the urban agriculture and green infrastructure systems studied here. A more general issue relates to the nomenclature in this field of research, with the definition of urban agriculture and green infrastructure types typically being loosely defined, overlapping or sometimes referring to completely different systems. Conceptual and terminological refinements are crucial to advance this field by enhancing reproducibility and consistency. Additionally, key information was missing from many studies. For example, it is highly likely that both urban agriculture plots are relatively small compared to other urban green spaces (e.g. McDougall et al., 2019), a factor we were unable to quantify from the current data sources. Green space area is a main driver of urban biodiversity patterns and ecosystem service provision (Beninde et al., 2015; Jaganmohan et al., 2016), so that the intrinsic capacity of urban agriculture to sustain these services may be more limited than that of green infrastructure and conventional farms. While we were unable to investigate this, the studies we reviewed employed a comparative approach, which means the effects of area were systematically accounted for within each study.

5 | CONCLUSIONS

Our results suggest that, compared to other green infrastructure, current urban agriculture practices do not benefit biodiversity, nor do they represent an improvement in terms of ecosystem functioning. In other words, urban agriculture is far from being a regenerative form of agriculture in urban settings (Schulte et al., 2022). While the primary function of urban agriculture is food production and social benefits, there remains room for improvement if we aim to meet the UN sustainable development goals (SDGs) (Qiu et al., 2024; Viana et al., 2022). There is a need to promote a balanced approach to quantifying socio-ecological benefits through urban planning and design (United Nations, 2015). For instance, enhancing consistency in the metrics used to characterise biodiversity is crucial, prioritising functional and evolutionary diversity metrics over single-species abundance metrics (Morelli et al., 2017; Núñez-Florez et al., 2019). Furthermore, more consistent reporting of key environmental data, such as plot sizes, is needed. It has been suggested that consistent strategies promoting the regenerative use of the available cultivation space—such as enhancing biodiversity and ecosystem functions through management and design—could mitigate these negative impacts (Buckton et al., 2023). A paradigm shift is needed to address knowledge gaps and the apparent limitations of current strategies for implementing urban agriculture. Exploring solutions to enhance plant diversity and abundance in urban agriculture landscapes seems a suitable starting point.

AUTHOR CONTRIBUTIONS

Conceptualization: Emilio Pagani-Núñez, Zeyu Li, Tharaka S. Priyadarshana and Ben A. Woodcock. *Methodology*: Emilio Pagani-Núñez, Zeyu Li, Tharaka S. Priyadarshana and Ben A. Woodcock. *Investigation*: Zeyu Li, Sihao Chen and Emilio Pagani-Núñez. *Data curation*: Zeyu Li and Sihao Chen. *Writing—original draft*: Zeyu Li and Emilio Pagani-Núñez. *Writing—review and editing*: Zeyu Li, Sihao Chen, Tharaka S. Priyadarshana, Myung-Bok Lee, Lingyun Xiao, Thomas B. Fischer, Xueqing He, Ben A. Woodcock and Emilio Pagani-Núñez. *Visualization*: Sihao Chen and Emilio Pagani-Núñez.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

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DATA AVAILABILITY STATEMENT

Data available from the Dryad Digital Repository: https://doi.org/10.5061/dryad.2bvq83c31 (Li et al., 2025).

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

File S1. Keywords, data extraction indicators, PRISMA flow diagram, search string, and retrieved references from the systematic review.

File S2. Extracted data from the systematic review, including all analysed variables with corresponding descriptions and classifications.

File S3. Diagnostic plots for fitted models, including funnel plots (publication bias), profile likelihood plots (parameter uncertainty), and Cook's distance (influence of outliers).

Table S1. Our dataset was divided into two main topics, biodiversity and ecosystem functioning.

Table S2. Statistical testing for publication bias in the dataset based on two models: a null model with no moderators, and another model using sampling variances as moderator (sei).

Table S3. Moderator variables used in this meta-analysis.

Table S4. A summary of the fitted models.

Table S5. Statistics related to the Model 1 (for Topic).

Table S6. Statistics related to the Model 2a (for CWS on Ecosystem functions).

Table S7. Statistics of pairwise comparisons related to the Model 1 and 2a.

Table S8. Statistics related to the Model 2b (for Ecosystem functions-CAT).

Table S9. Statistics related to the Model 3a (for CWS on Biodiversity).

Table S10. Statistics of pairwise comparisons related to the Model 3a (for Biodiversity).

Table S11. Statistics related to the Model 3b (for Biodiversity-CAT).

Table S12. Statistics of pairwise comparisons related to the Model 3b (for Biodiversity-CAT).

Table S13. Statistics related to the Model 3c (for Biodiversity-CWS:CAT).

Table S14. Statistics related to the Model 4 (for UAT).

Table S15. Statistics related to the Model 5 (for Continent).

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