IMPACTS OF FIELD EDGE FLOWER PLANTINGS ON POLLINATOR CONSERVATION AND ECOSYSTEM SERVICE DELIVERY: A META-ANALYSIS

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A thesis submitted in partial fulfillment
of the requirements for the degree of

Master of Science

(Agroecology and Entomology)

at the

UNIVERSITY OF WISCONSIN - MADISON
2020

Abstract

Planting flowers along crop field edges is an increasingly common management strategy for addressing pollinator declines and improving crop pollination. However, no systematic review has been published that summarizes the efficacy of this specific management practice and how it impacts wild pollinators and pollinator-dependent crops. We conducted a systematic review and meta-analysis to determine whether such plantings 1) increase pollinator abundance or richness within field edges, 2) increase the abundance or richness of pollinators visiting crop flowers, or 3) improve crop yields. Our analysis of 54 studies suggests that field-edge flower plantings are highly effective at increasing pollinator richness and abundance in field edges and that plantings become more effective as they mature. However, the influence of field-edge plantings on crop pollination and yield is inconsistent. Planting size and richness did not change these results. Our analysis emphasizes that field-edge flower plantings consistently increase pollinator abundance and richness, but that there is a critical need for research on when and how plantings can improve ecosystem service provision and delivery. Determining if field-edge plantings affect pollinator pollination growth may uncover a mechanism for how plantings could improve crop pollination, while factors such as landscape context and crop-type may define when this happens.

Keywords: *pollinator plantings, agri-environment schemes, floral enhancements, wildflower strips, pollinator conservation, bee pollinators, ecosystem service, crop pollination, sustainable agriculture*

Introduction

Limited flower availability is believed to be one of the leading causes of bee declines worldwide (Brown & Paxton, 2009; Potts et al., 2003; Scheper et al., 2014). In rural areas, decreases in flower availability are driven primarily by policy and economic incentives that promote the expansion of monoculture agriculture (Lark et al., 2015; Otto et al., 2016). Monocultures replace more diverse farms and floral-rich natural area with vast expanses of single crops that bloom for only a limited time. In more heterogenous landscapes, flower diversity creates phenological diversity in floral resources, providing consistent food availability for bees over the course of the season. In contrast, monocultures often create pulses in flower availability, and depending on the crop, may provide only low-quality pollen and nectar resources (Di Pasquale et al., 2016). The exception to this rule may be mass-flowering monocultures such as oilseed rape, which have been shown to provide important (though still temporally limited) resources for bees (Holzschuh et al., 2013; Jauker et al., 2012; Westphal et al., 2003). However, these crops likely compliment rather than substitute for the floral resources provided by diverse landscapes and natural area (Holzschuh et al., 2013) and may have detrimental effects on the pollination of co-flowering crops or wild plants (Holzschuh et al., 2011, 2016).

In addition to concerns about the status of wild bees, there is also growing recognition of the valuable role wild bees play in pollinating crops (Garibaldi et al., 2013; Klein et al., 2007; Losey & Vaughan, 2006; Winfree et al., 2007). Honey bees are widely used for crop pollination, but the benefits of wild bees along with increasing hive rental costs (Bond et al. 2014) have heightened farmers' interest in utilizing wild bees for crop pollination.

Farmers are frequently encouraged to plant herbaceous flowers or flowering hedges along the edges of crop fields as a means of addressing wild bee declines and attracting wild bees to crop fields for pollination. By providing additional floral and nesting resources, these plantings are thought to increase bee populations, providing long-term conservation benefits. The purported benefits of these plantings to crops may come from "spillover" effects, where bees attracted to the planting also forage on the crop, or from population growth which should increase bee abundance in the broader landscape including crop fields. In addition to these benefits, field-edge plantings are an attractive management strategy because they are usually smaller in size than other restoration practices (e.g. prairie restoration) necessitating less commitment of land and up-front investment, and because they are often installed on unused field margins, avoiding the need to remove land from production. For these reasons, field-edge plantings are widely promoted by conservation groups (Vaughan et al. 2015), highlighted in scientific literature (e.g., Garibaldi et al., 2014; Isaacs et al., 2017; Sidhu & Joshi, 2016), and incentivized through pollinator conservation policy (e.g., USDA Conservation Stewardship Program practices 327 and 386 and the Conservation Reserve Program State Acres for Wildlife Enhancement in the US, and Agri-environment schemes in Europe such as ELS EF4 or EK21 in the English “Environmental Stewardship” scheme).

Despite the increasing popularity of field-edge pollinator plantings as well as a growing body of empirical studies on their utility, no meta-analysis has been published summarizing the efficacy of this specific management practice and how it impacts wild pollinators and pollinator-dependent crops. While some studies have found that field-edge plantings can significantly increase wild bee abundance, richness, and/or crop yields (Blaauw & Isaacs 2014a.; Morandin & Kremen, 2013; Scheper et al., 2015), others have shown more mixed results (Nicholson et al., 2020; Sardiñas & Kremen, 2015; Wood et al., 2018). A meta-analysis on the effects of field-edge plantings is thus helpful in determining the contexts in which they are most useful.

Multiple literature reviews have focused on pollinator habitat restoration broadly, combining effects of field-edge plantings with other types of pollinator management practices (Haaland et al., 2011; Holland et al., 2017; Scheper et al., 2013; Venturini et al., 2017a), or with fallow, unplanted field margins (Uyttenbroeck et al., 2016). While these reviews help assess the effectiveness of habitat restoration in a general sense, they combine the effects of multiple management practices in the same analyses making it difficult to determine if field-edge plantings, specifically, will have desired effects.

A new analysis by Albrecht et al., (in press), which is more similar to our review, focuses on field-edge plantings and evaluates their impact on crops. However, our review differs from Albrecht et al. in both methodology and the specific response variables measured. First, Albrecht et al. used raw data from studies contributed by collaborators, while we conducted a meta-analysis of a larger number of studies, providing a valuable comparison between the general patterns generated with these two methods. Second, we assess more aspects of pollination service provision than Albrecht et al. by evaluating pollinator visitation metrics such as pollinator richness and abundance on crop flowers and further subdividing some of these analyses by different groups of bees. Finally, we evaluate more outcomes than Albrecht et al., focusing not only on ecosystem service provision to crops but also on bee conservation.

We chose to focus this review predominantly on wild bees. Although other taxa contribute to crop pollination (Rader et al., 2016), we concentrate on wild bees because they contribute most to pollinating many crops (Delaplane et al., 2000; Klein et al., 2007). Likewise, while declines of other insect taxa are a critical issue (Klink et al., 2020; Sánchez-Bayo & Wyckhuys, 2019; Wagner, 2020), many conservation efforts explicitly target wild bees and thus it is important to understand bee-specific outcomes.

In this review we ask three primary questions: (1) Do field-edge plantings affect wild bee conservation (e.g. increase wild bee abundance, richness, or fecundity in field-edge plantings)? (2) Do field-edge plantings increase ecosystem service (ES) provision (e.g. increase the abundance, richness, or visitation rate of wild bees to crop flowers)? (3) Do field-edge plantings improve ES delivery (crop yields)? We conducted a systematic review, and for the subset of studies that provided sufficient data, we performed a meta-analysis to ascertain the magnitude of the effect of plantings on these outcomes. We also evaluated the influence of several factors that might influence planting efficacy including planting maturity, richness, size, and landscape context.

Methods

Systematic Review

We used the Web of Science Database to identify studies that evaluated impacts of field-edge plantings on wild pollinators. Specifically, we conducted a keyword search which pulls from terms in article titles, abstracts, keywords, and KeyWords Plus.® We limited the search to English language articles and evaluated all records from 1900 through February of 2020 (Appendix II, Figure S1). Search terms included: ("planting*" OR "enhancement*" OR "hedgerow*" OR "strip*" OR "reservoir*" OR "flower border*" OR "flower margin*" or "floral border*" or "floral margin*" or "sown margin*") AND ("pollin*" OR "bee" or "bees" OR "bombus" OR "apidae"). This produced 2406 citations. We identified additional citations via ad-hoc searches of studies referenced in highly-cited papers and through the "view related records" tool in Web of Science, which generates a list of papers that share citations with a selected paper (Appendix II). These additional searches produced 905 citations.

Articles were screened initially by reading titles and abstracts. Any articles that did not evaluate the effects of intentionally planted field-edge flowers on pollinators within an agricultural context were excluded from further review. After this filtering, 92 articles remained in the pool of potential studies. The remaining papers were read in full to determine if they met further criteria for exclusion. This included the stipulation that articles must report data, be peer-reviewed, include a field component, and compare planted edges to a control area that was not planted. The type of control varied across studies and included unplanted, unmanaged field edges; unplanted, managed field edges (e.g., herbicide or mowing); grass strips; bare ground; and crop fields with no edge. Finally, to be included, studies needed to report a response variable relevant to wild bees. Some studies focused on broader pollinator communities rather than just wild bees. These were accepted as long as they included wild bees in their pollinator surveys. Of the studies read in full, 38 met these exclusion criteria (Appendix II, Figure S1a). This left a grand total of 54 studies which were included in the review.

Meta-analysis

Inclusion criteria

On a subset of studies for which adequate data were available, we conducted a meta-analysis to estimate the relative effect size of field edge plantings on bee conservation and ES metrics. Studies in the meta-analysis had to include a minimum sample size of 3 replicates in both the treatment and control group; provide a spatial replicate; and report mean, sample size, and a measure of variance (standard deviation, standard error, standard error of the mean, confidence interval, or interquartile range), in the text, tables, or figures (Appendix 1, Figure S1). Minor exceptions to these rules are outlined in Appendix II.

Categories and response variables

We organized analyses into three categories based on our questions around bee conservation and ES delivery. The first consisted of studies that measured pollinators in field-edge plantings and controls, hereafter "field edges" (conservation outcome). The second included studies that measured pollinators visiting crop flowers in fields adjacent to plantings and controls (ES provision), and the third included studies that measured crop yield in fields adjacent to plantings and controls (ES delivery) (Table 1). For all analyses, hedgerows and forb plantings were considered together as we did not have enough hedgerow studies in any group to separate their effects. For each of the three categories, we evaluated the effect of plantings on all pollinators as a single group. This included studies for which the entire pollinator community was analyzed (including bee and non-bee pollinators), studies for which the entire bee community was analyzed, as well as studies that focused only on *Bombus* and/or solitary bees. In addition, within the studies that measured bees in field edges, we conducted a subgroup analysis with *Bombus* and solitary bees as separate groups.

Within each category or subgroup, papers were broken down by response variable (Table 1). A given response variable had to be assessed in at least 4 studies ($k \geq 4$) to be included in analyses. Therefore, the results we report include only the categories for which there were at least 4 studies for the given response. Pollinator response variables included abundance, density, visitation rate, and richness. Abundance, density, and visitation rate were considered together in a single "abundance" category because density and visitation rate are calculated as abundance divided by area or time, respectively. Richness was considered as its own category. For crop yields, responses included fruit or seed weight per plant, branch, or area and percent or proportion seed set. When both weight and proportion measures were provided, weight was used in analyses. While fruit quality is another important measure of ES delivery, few studies collected this data precluding analysis.

Table 1. Number of studies (k) in review, meta-analyses, and meta-regressions by categories of response variable. In the review k represents the total number of studies, in the meta-analyses and meta-regressions, k represents the study-location combinations that are included as independent replicates in analysis. NA indicates that we did not have enough data to run the

		Abundance					Richness				
Category	**Groups measured**	**Studies in review**	**Studies in meta-analysis**	**Studies in meta-regressions**			**Studies in review**	**Studies in meta-analysis**	**Studies in meta-regressions**		
				Planting Maturity[1]	**Planting Richness**	**Planting size**			**Planting maturity**	**Planting Richness**	**Planting Size**
Pollinators in plantings (conservation metric)	All pollinators	k=27	k=29	k=9, n=26	k=13	k=19	k=19	k=20	k=6, n=20	NA	k=11
	Bombus only	k=14	k=17	k=10, n=23	NA	k=12	k=6	k=7	NA	NA	NA
	Solitary bees	k=5	k=8	k=6, n=15	NA	NA	k=1	k=4	NA	NA	NA
Pollinators in field (ES Provision)	All pollinators	k=18	k=12	NA	NA	NA	k=7	k=6	NA	NA	NA
	Bombus only	k=4	NA	NA	NA	NA	k=1	NA	NA	NA	NA
	Solitary only	k=0	NA	NA	NA	NA	k=0	NA	NA	NA	NA
Crop yield (ES Delivery)	Yield	k=17	k=7	NA	NA	NA					

Notes:

[1] For meta-regressions of planting maturity, we used studies of forb plantings that reported results for multiple years within a study and calculated within-study effect sizes across years. n represents the number of within-study reps, while k represents the number of studies.

Calculating effect size

We used Hedges' unbiased weighted standardized mean difference (Hedges' *d*) to estimate effect sizes. All estimates were calculated using the metafor package in R (Viechtbauer, 2010). For studies that broke results into multiple years, we used the final year as the best measure of a mature planting. For studies that included analyses in multiple regions or crops, each region or crop was treated as an independent replicate and the study was included multiple times in analyses. One study included separate sites with forb and hedgerow plantings, each with their own set of controls. For this study, results for forb and hedgerow plantings were also considered independent replicates.

We used a number of methods to avoid pseudo-replication depending on how data was aggregated and reported (Appendix II). These approaches led to conservative results because we chose the most conservative controls and because averaging across levels of some variables obscured more nuanced variability that authors were able to account for. As a result, ~40% of studies that reported significant, positive effects of plantings showed non-significant, positive effects of plantings in our meta-analysis.

Analyses

We performed all meta-analyses using random effects models fit with REML in the metafor package in R (Viechtbauer, 2010). Before performing these analyses, we assessed the normality of effect size estimates, and evaluated publication bias and heterogeneity. No groups showed strong non-normality or publication bias, and most groups showed only moderate heterogeneity (Appendix II). For all groups with outliers (i.e. confidence interval did not overlap the overall confidence interval), we conducted analyses with and without the outlying study (Appendix II, Table S3 and Table S4). As outliers did not significantly impact results, we report results from the full dataset.

We ran several meta-regressions to examine how planting size, richness, and maturity influence bee responses and crop yield. We evaluated the effects of planting size and richness by including each as covariates in separate meta-regressions for all groups with $k \geq 10$. To investigate the question of planting maturity, we used studies of forb plantings that reported results for multiple years within a study from which we could calculate within-study effect sizes across years. To account for the non-independence of within-study effect sizes, we ran multi-level meta-regressions that included a random term for study. For this analysis, we restricted our analyses to groups with $k \geq 6$. The data structure for studies that evaluated landscape context precluded quantitative analysis, so our discussion of this factor is based on a qualitative assessment of the studies in our review.

Finally, to determine if study region influenced our results, we ran subgroup analyses that compared studies across broad regions (US and Europe) for all groups with $k \geq 4$ in both regions (Appendix II, Table S5). We also ran analyses (for groups with $k \geq 4$) that included only studies that met cutoffs for study rigor based on sample size, number of years of data, independence of sites, and type of control (Appendix II, Table S6).

Results

Systematic review:

Of the 54 studies that met the criteria of our systematic review (Appendix I, Table S1 and Table S2), 80% reported results for bees in plantings (conservation metric), 37% reported results for bees in the crop field (ES provision), and 26% reported results for crop yield (ES delivery). All but 3 studies investigated bee abundance or a similar metric such as visitation rate or density. In contrast, only about half the studies (n=24) evaluated bee species richness. Just 5 studies split results into separate groups for *Bombus* and solitary bees, while 14 focused exclusively on *Bombus*.

Forb plantings were the most represented planting type (85% of studies), while 20% of studies evaluated hedgerow plantings. For forb plantings that reported plant composition, 28% of plantings were composed of annual species, while 66% were perennial or a mix of annual and perennial species. Of these, 28.5% did not report planting maturity in the body of the paper. However, for those that did and did not use transplanted plants (e.g., seedlings or plugs), the average age of plantings for the last year of data collected was 3 years for perennial plantings, and 1.7 years for mixed annual and perennial plantings. The average number of years studies ran after plantings were one year old (the minimum many perennial plants need to reach maturity) was 2.1 years. Planting size ranged widely from 16m^2 (Balzan, 2017) to ~ 5 ha (Kleijn et al. 2018), although the majority were ≤ 500m^2. The crops grown in fields adjacent to plantings also varied widely. Twenty-one different crops were represented with the most common being cereal and/or oilseed (n=8), strawberry (n=5), blueberry (n=4), and apple, melon, cherry, or tomato (n=2) (Appendix III, Table S7).

All but two studies (from New Zealand and South Africa) were conducted in the US (33.3% of studies) or Europe (63% of studies) (Appendix III, Figure S6). This is undoubtedly due to research and language bias, but also tracks regions with highly intensive agriculture—the type of landscape to which management practices like pollinator plantings are targeted. While the oldest study was published in 2002, nearly three-quarters of all studies were published in 2015 or after (Appendix III, Figure S5).

<u>Meta-analysis</u>

Pollinator conservation (bee abundance and richness in field edges)

For studies that could be used in a quantitative meta-analysis, pollinator plantings had strong and significant positive effects on overall pollinator abundance (k=29, p<0.001) and richness (k=20, p<0.001) in field edges adjacent to crops (Figure 1). This was also true for studies that reported responses of *Bombus* or solitary bees separately (Figure 1). Effects of plantings were relatively similar for *Bombus* and solitary bees and for abundance versus richness although overall there was a trend toward stronger effects on abundance and *Bombus* than richness or solitary bees. Effect sizes were very similar between the US and Europe and subset analyses using only the most rigorous studies made observed effects stronger (Appendix II, Table S5 and S6). Overall, the results of this quantitative meta-analysis are consistent with the results authors reported across studies in our full systematic review, all of which reported significant, positive effects of plantings on pollinators in field edges for at least one of the response variables measured (Appendix I, Table S1 and Table S2).

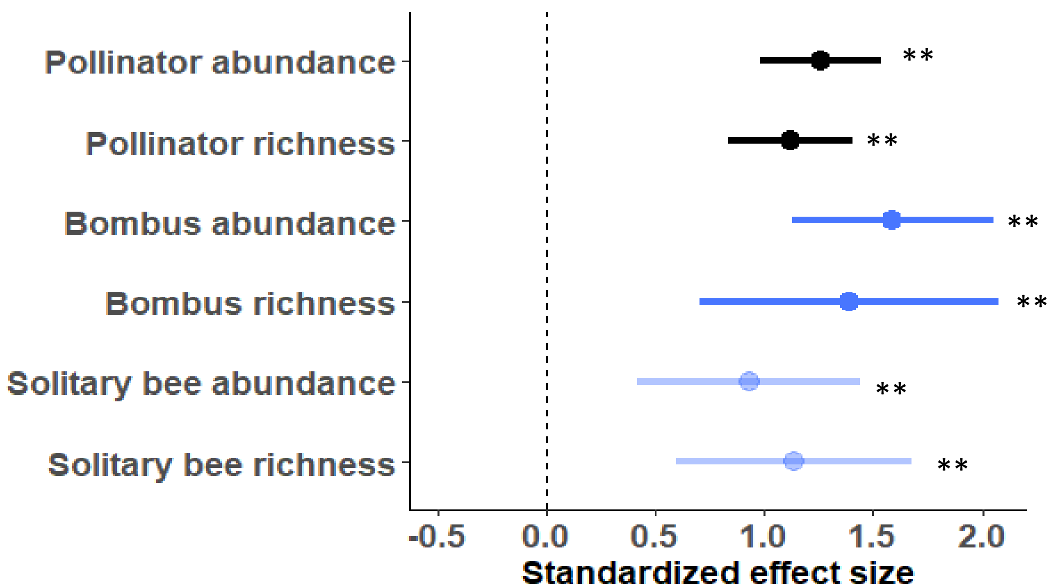

Figure 1. The effects of field edge pollinator plantings on overall pollinator abundance and richness, Bombus abundance and richness, and solitary bee abundance and richness in field edges. Mean effect sizes (Hedge's *d*) ± 95% CI are illustrated. Double asterisks denote significance at α <0.001.

Ecosystem service provision (pollinators in crop fields) and delivery (crop yields)

Pollinator plantings did not alter overall pollinator abundance (k=12, p=0.44) or richness (k=6, p=0.94) in crop fields, or crop yields (k=7, p=0.18) (Figure 2). Neither study region nor rigor changed the significance of results for pollinator abundance, the only group with enough data for subgroup analyses (Appendix II, Table S5 and S6). In both the review and the meta-analysis few studies showed significant negative results while a modest number showed significant positive results (Appendix II, Figure S3 and S4).

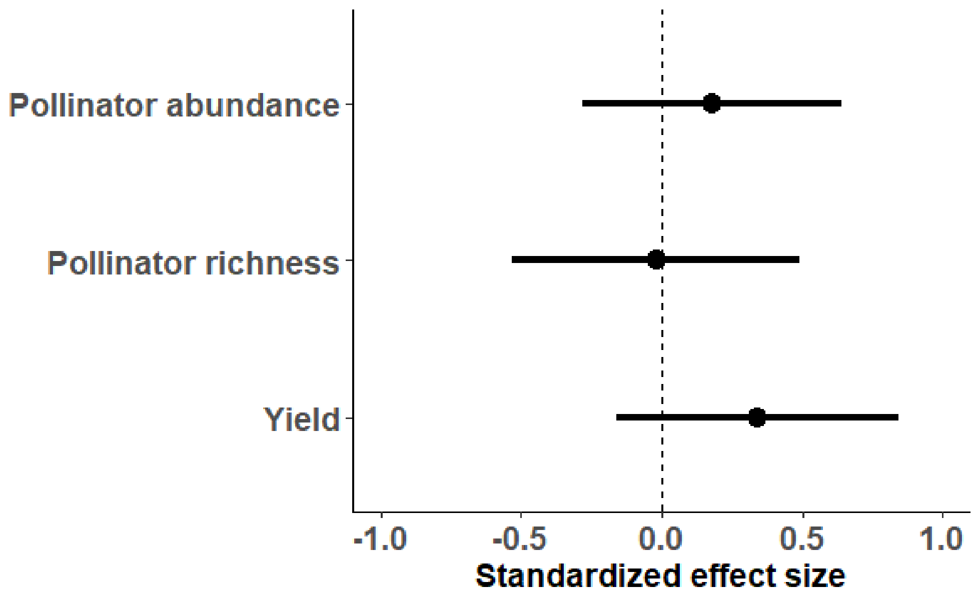

Figure 2. Effects of field-edge pollinator plantings on pollinator abundance and richness in crop fields (ES provision) and crop yields (ES delivery). Mean effect sizes (Hedge's *d*) ± 95% CI are illustrated. No groups were significant at α <0.05.

Factors modulating planting effects

In addition to our primary research questions, we were also interested in how planting maturity, richness, size, and landscape context might modify the effect that plantings have on pollinator abundance, richness, and crop yields. We did not have enough data to evaluate the effect of any of these factors on ES metrics (Table 1), so all results are reported for bees in field edges only.

Planting maturity (i.e. the number of years after planting establishment) was significantly and positively associated with overall pollinator abundance (p<0.001) as well as *Bombus* (p<0.001) and solitary bee abundance (p<0.002) (Figure 3). Maturity was also positively related to overall pollinator richness (p<0.001) (Figure 3). Planting richness was not associated with pollinator abundance (p=0.37).

Similarly, planting size was not associated with pollinator abundance (p=0.59) or *Bombus* abundance (p=0.77), nor was there was a relationship between planting size and overall pollinator richness (p=0.12).

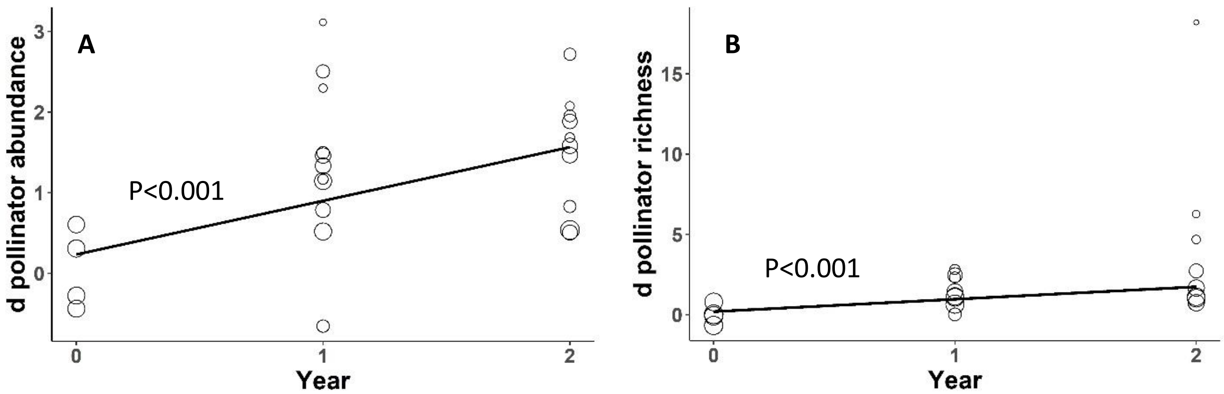

Figure 3. Relationship between planting maturity and a) pollinator abundance and b) pollinator richness for studies included in meta-regressions. Point size = 1/var. Year 0 represents the year plantings were established.

Because the way data was presented, we could not quantitatively evaluate the effect of landscape context. However, 8 of the 11 studies in our review that evaluated the interaction between plantings and the landscape (Carvell et al., 2011; Carvell et al., 2015; Grab et al., 2018, 2019; Herbertsson et al., 2018; Jönsson et al., 2015; Kleijn et al., 2018; Krimmer et al., 2019; Rundlöf et al., 2018; Scheper et al., 2015; Sutter et al., 2017) found a significant or nearly significant planting by landscape interaction for at least one response variable. That is, the amount or configuration of existing habitat around plantings significantly altered the effect that plantings had on pollinators.

Discussion

Pollinator conservation (pollinator abundance and richness in field-edge plantings)

Our systematic review and meta-analysis found that field-edge plantings can have strong positive effects on pollinator abundance and richness within plantings themselves. This is consistent with other reviews that found benefits of a variety of types of floral enhancements on bee abundance and richness (Haaland et al., 2011; Scheper et al., 2013; Uyttenbroeck et al., 2016; Holland et al., 2017; Venturini et al., 2017a). In addition, we found that planting maturity significantly influenced the strength of these results. Our meta-regression showed that overall pollinator abundance and richness, as well as *Bombus* and solitary bee abundance and richness, increased within plantings as they matured. We found similar effects in our systematic review. Seventy five percent of the studies in our review that reported results over the course of at least 3 years found that plantings increased abundance, richness, and/or crop yield over time. This increase was either gradual enough or had high enough variability that most studies did not find a significant difference in pollinator richness or abundance relative to controls until plantings were at least 2-3 years old.

There are two reasons that planting maturity may increase bee abundance and richness. First, mature plantings generally have more flowers, which in turn can attract more bees. Because many plantings are composed entirely or partially of perennial species, they often mature slowly, producing few flowers for at least a year and sometimes much longer. Since bees preferentially forage on dense floral displays (Rowe et al., 2020; Williams et al., 2015), they are likely more attracted to mature plantings with greater floral cover. Second, plantings may increase pollinator abundance and richness via population growth (Venturini et al., 2017b). However, abundance and richness can only reliably provide evidence of population growth in longer-duration studies of mature plantings (Kremen et al., 2007). To build pollinator populations, plantings must first mature enough to effectively attract bees, and after this, studies must run for long enough to detect population trends. Because bees are univoltine, these trends may not be evident in studies that run for less than three years after plantings mature. This is an issue because determining whether plantings promote population growth is important for evaluating conservation outcomes.

To accurately determine whether field-edge plantings increase pollinator populations will require both longer duration studies and studies that assess more direct metrics of population growth. A small number of studies in our review evaluated more direct growth metrics including colonization and persistence (M'Gonigle et al., 2015), body size (as a proxy for immune function and fitness) (Grab et al., 2019), production of *Bombus* reproductives (males and gynes) (Holland et al., 2015), and nesting success (Sardiñas et al., 2016; Wood et al., 2015). Of these, all but one found positive effects of plantings. While these data are limited, they provide promising evidence for the benefits of field-edge plantings for promoting population increases.

<u>Ecosystem service provision (pollinators in crop fields) and delivery (crop yields)</u>

In contrast to the effects of plantings on pollinators in field edges, our analyses indicated that plantings do not consistently alter pollinator visits to crop flowers or crop yields, a finding that agrees with the review by Albrecht et al. (in press). While this does not support the hypothesis that plantings increase ES provision or delivery, it also shows that plantings do not decrease it. This is important because there is a concern among some farmers that plantings may draw pollinators away from the crop field resulting in lower crop pollination (Kremen et al. 2019; Lundin et al., 2017). If this were the case, we would expect to see a significant negative impact of plantings on pollinators in the crop, an observation that was not supported by our analyses.

However, in order to benefit the crop, plantings must increase the number of pollinators foraging on the crop, a pattern that was not strongly supported in this study. Field-edge plantings could attract pollinators to adjacent crop fields in two ways: either pollinators attracted by the planting might "spillover" into the crop to also forage on crop flowers, and/or by promoting pollinator population growth, plantings would eventually increase pollination in the landscape around plantings, including in the crop field. Both of these factors could explain why we did not find consistent impacts of plantings on ES. First, the lack of consistency in crop type across studies in our analysis may have obscured treatment effects. We know that bees preferentially forage for nutritive content and that crops are known to vary in pollen and nectar quality (Dufour et al., 2020; Toshack & Elle, 2019). Thus, the probability that bees will spill over into the crop field is likely driven by the attractiveness of the adjacent crop (Kremen et al., 2007). The papers in our review spanned 21 different crops, over 2/3 of which were represented by a single study. This made it impossible to detect any crop-specific effects of spillover. Second, the studies in our meta-analysis were likely too short in duration to reliably detect increases in population growth and determine its impact on crop pollination. For studies included in our meta-analysis of bees in crop fields, the average study duration was 2.3 years—not long enough to detect population trends. Among the five studies in our review that were potentially long enough in duration (>3 years) (Blaauw & Isaacs, 2014a; Grab et al., 2018; Korpela et al., 2013; Schulte et al., 2017; Wood et al., 2018)), 3/5 reported significant positive effects of plantings on bees in crop fields. This indicates that positive effects of field-edge plantings on crop pollination may be more evident in longer-duration studies of mature plantings.

Planting richness

Planting richness may be important for promoting pollinator diversity because flower morphology dictates which flowers pollinators can forage on (Harder, 1983), and because oligolectic species require specific flowers. Accordingly, previous studies found significant, positive effects of planting richness on bee richness and abundance (Blaauw & Isaacs, 2014b; Gill et al., 2014), and the review by Albrecht et al. (in press) found a significant impact of planting richness on pollination service provision.

In contrast, our meta-regression indicated that planting richness was not associated with pollinators in field edges. Similarly, the six studies in our systematic review that evaluated richness as an explicit part of their study design (Barbir et al., 2015; Carvell et al., 2007; Pontin et al., 2006; Pywell et al., 2011; Venturini et al., 2017b; Williams et al., 2015) found neutral or even negative relationships between planting richness and pollinator responses. This may be because separating the effects of flower abundance from flower diversity is challenging and it is possible that richness effects were confounded with abundance or cover (Williams et al., 2015). It may also be that simpler plantings in these studies included highly attractive species while complex plantings did not contain as many attractive species overall. When many studies in our review took place less research had been published on the optimal composition of planting mixes, making it difficult to maximize the quality of diverse plantings. However, the number of publications on this topic has increased dramatically in the past few years. This work has focused on evaluating the attractiveness of plant species to a range of bee taxa (e.g., Russo et al., 2013; Sutter et al., 2017; Nichols et al., 2019), as well as on optimizing species selection based on a variety of goals (M'Gonigle et al., 2017; Williams & Lonsdorf, 2018). These studies support the idea that richness may be less important than the inclusion of key species (Warzecha et al., 2018), and that species that are optimal are goal-dependent.

Planting size

Based on well-established species-area relationships, some studies support positive associations between bee abundance or richness and patch size (Bommarco et al., 2010; Blaauw & Isaacs, 2014b). Our meta-analysis, however, found no significant effect of planting size on overall pollinator abundance or richness, or *Bombus* abundance in field edges. This was also true of the five studies in our systematic review that evaluated the effect of planting size as part of their study design (Carvell et al., 2011; Carvell et al., 2015; Kleijn et al., 2018; Krimmer et al., 2019; Rundlöf et al., 2018), only one of which found a marginally significant positive impact of size (Krimmer et al., 2019). This finding may suggest that the size of a patch may be less critical than the total amount of habitat present in the landscape surrounding plantings as many bees can disperse to nearby patches (Fahrig, 2013; Kremen et al., 2007).

While our analyses indicate that patch size may not strongly influence pollinator abundance or richness, some research has shown it may have an important influence on the abundance and richness of nesting bees (Steffan-Dewenter, 2003). This indicates that planting size could be an important factor for increasing pollinator populations and thus exporting bees to crop fields. No studies in our review evaluated the effect of planting size on colonization, persistence, or nesting, and only three studies evaluated the impact of planting size on pollinators outside of plantings. While two of these three found a significant benefit of size on pollinators in the adjacent landscape (Carvell et al., 2015; Krimmer et al., 2019), more data is needed to determine if this is a consistent pattern.

Landscape context

A final factor that may influence the effectiveness of field-edge plantings is landscape context (Kennedy et al., 2013). Understanding the influence of landscape is critical because it could be used to strategically locate plantings where they will have the greatest effect. Albrecht et al. (in press) found that the interaction between field-edge plantings and landscape did not significantly influence pollination service provision, but another review found the opposite result for bee abundance and richness within plantings (Scheper et al., 2013). In accordance with Scheper et al. (2013), a majority of studies in our review that evaluated the interaction between plantings and landscape-scale habitat found a significant or near-significant planting by landscape interaction. In other words, the effect that field-edge plantings had on pollinator responses varied depending on the composition of the surrounding landscape. While studies in our review did not present data that allowed for quantitative analysis, upon visualizing these studies, plantings appear to have a greater positive impact in landscapes with intermediate amounts of existing pollinator-friendly habitat (e.g., semi-natural areas in the landscape) than in landscapes with either larger or small amounts (Table 2). The general "hump-shape" of this relationship is consistent with Intermediate Complexity Hypothesis proposed by Tscharntke et al. (2005), and is supported by some empirical studies (Grab et al., 2018; Scheper et al., 2013). However, Tscharntke et al. (2005) theorized that plantings would have the greatest impact in intermediate landscapes with 1-20% natural area, while the studies in our review indicate that this range may be as broad as 10% to 50% natural area.

Table 2. Effects of plantings on bee abundance across a landscape-scale habitat gradient for six studies that reported data comparing plantings vs controls at the endpoints of or along their landscape gradients. Shading correspond to the number of studies that fall within each bin, letters correspond to specific studies detailed in notes.

	Percent landscape-scale natural area[1]									
	0-10	10-20	20-30	30-40	40-50	50-60	60-70	70-80	80-90	90-100
Greater[2] effect		[a, b, c][3]	[d, e]	[f, e]	[e]					
Lesser[2] effect	[f, d, c]	[e]				[e]	[e]	[a, b]		

Notes:

[1] If studies reported % cropland, they were converted to % natural area as (1 - % cropland) as natural area and cropland are often roughly inversely correlated. Rundlof et al. 2018 reported results in terms of 14-76% cropland but indicated a range of natural area from 1-18%. 18% natural area was thus assumed to correspond to 14% cropland and 1% natural area to 76% cropland.

[2] Most studies reported the effect of plantings at the low end vs. high ends of their gradients, therefore each study appears twice to represent the relative effect of plantings on bee abundance at gradient endpoints. Grab et al. 2018 appears six times because they indicate positive or negative effects of plantings along their entire gradient.

[3] Studies: [a] Carvell et al. 2011, [b] Carvell et al 2015, [c] Rundlof et al. 2018, [d] Sutter et al. 2016, [e] Grab et al. 2018, [f] Krimmer et al. 2019.

Research needs

We found that field-edge flower plantings consistently increase pollinator abundance and richness within plantings themselves and that the effects of plantings increase as they mature. However, additional research is required to determine whether observed increases in abundance and richness are the result of plantings promoting pollinator population growth or whether plantings are simply attracting pollinators from the surrounding landscape. Likewise, more research is necessary to determine the circumstances under which plantings may improve crop pollination. The key to answering both these questions may be in conducting longer-duration studies of mature plantings (particularly how they impact pollinators in crop fields adjacent to plantings), as well as studies that focus on how plantings impact direct metrics of population growth such as fecundity and nesting success. Such studies would help determine both the capacity of field-edge plantings to influence population growth and the effect that population growth may have on crop pollination. As planting size has been implicated as a factor that could alter the degree to which field-edge plantings impact population growth, more studies on how planting size affects persistence, nesting success, and spillover are also critical. Additionally, the landscape surrounding plantings may affect both plantings (Kleijn et al., 2011; Kleijn & Sutherland, 2003;

Tscharntke et al., 2005) and the importance of planting size (Kremen et al., 2007). Thus, understanding how habitat in the landscape interacts with plantings may help locate plantings in the places where they will have the greatest impact. For this reason, future studies should also account for the influence of landscape context or habitat to crop ratio.

Research on two additional factors might also help evaluate how field-edge plantings influence crop pollination. First, repeated studies in the same crop types, or studies designed explicitly to evaluate the effect of plantings on crop pollination across multiple crops would be useful in determining if planting effects are crop specific. Second, evaluating other crop response metrics such as fruit quality would clarify if plantings can influence economically important factors other than crop yield.

A few additional considerations may also have influenced the conclusions we drew about field-edge plantings. The studies in our review were biased toward *Bombus* over solitary bees and toward pollinator abundance over richness (Table 1). Thus, we know little about how field-edge plantings might differentially impact specific bee taxa or influence pollinator community composition. Determining how plantings in general, or factors such as planting maturity, size, richness, or landscape context might impact important crop specialists or species of conservation concern would help tailor plantings to specific conservation and ES goals. In addition, over half of the studies on forb plantings in our review did not report data on floral richness or floral cover in plantings versus controls. Including this information would be helpful in parsing the importance of flower species richness versus flower cover, as well as assessing overall planting establishment.

Conclusions

This review focuses on the impacts of field-edge plantings on pollinators and crops. However, pollinator restorations can be beneficial for numerous reasons including habitat provisioning for pest predators and other organisms, preventing soil erosion and runoff, and beautifying agricultural landscapes (Haaland et al., 2011; Holland et al., 2017; Wratten et al., 2012), and there is even potential for them to provide supplemental income from seed production (Delphia et al. 2019). Adopting a more multifunctional approach to evaluating the benefits of pollinator restorations will both help justify them and allow for an approach to farm management that is more attuned to the farmers' complex decision-making (Sidhu & Joshi, 2016).

Field-edge plantings are just one of a broad range of restoration and agroecological farming practices, and the greatest impacts for both pollinator conservation and ES outcomes will likely arise from a combination of field, farm, and landscape-scale diversification and restoration strategies that increase flower availability at multiple spatial scales (Garibaldi et al., 2014; Kremen et al., 2007). In addition to field-edge plantings, these could include the expansion of polyculture, inter-cropping, catch-cropping, and flowering cover crops; increasing flowering ground cover in orchards and other perennial crops; and developing landscape-scale diversification and restoration strategies. While additional ecological research on these practices and how they interact is important, it is critical to acknowledge that many of them have been used for centuries by farming communities around the world (Vandermeer & Perfecto, 2012). Recognizing farmers' knowledge of these practices as well as integrating this knowledge with ecological research is thus key to managing agriculture in a way that benefits pollinators, people, and the environment as a whole.

Acknowledgements

I owe a huge thank you to my committee, Claudio Gratton, Russ Groves, and Rufus Isaacs for reviewing and editing this manuscript. I'd also like to thank my friends; parents, Madeline Barnes and Gerry Lowe; and partner, Patrick Monari, for ongoing support and encouragement, and the graduate students in the University of Wisconsin-Madison Agroecology Program for pushing me to more fully integrate across disciplines and imagine what truly just and sustainable agriculture could look like. Finally, I owe a thank you to Maria Kamenetsky on the UW College of Agriculture and Life Sciences statistical consulting team, Shinichi Nakagawa, and Jeroen Scheper for answering questions on meta-analysis methods. The UW Entomology R.K. Chapman Fellowship and the University of Wisconsin Hatch Grant (WIS01988) in part funded this research.

The University of Wisconsin-Madison sits on land taken from the Ho-Chunk Nation and its wealth is built in part from additional parcels of Indigenous land seized under the Morrill Act. While this statement acknowledges our history, it is also a matter of the present. These wrongs are perpetuated today in our science and institutions which continue to privilege Western science and appropriate Indigenous ecological knowledge. In the past, I have not done enough to actively incorporate these issues into my work. Moving forward, I challenge myself to invest more in Indigenous knowledge and to partner with Indigenous communities to uphold their ongoing science and management practices alongside my own work. I challenge my readers to do the same.

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Appendix I: Studies in review

Table S1. Studies identified via our systematic review evaluating the effect of field-edge plantings on bees within field edges (conservation outcome). All results are those reported by authors rather and are reported for all pollinators (which sometimes included non-bee flower visitors) unless a specific bee group is specified. Throughout, NR=not reported.

Study features				Planting features				Reported results	Additional info	
Reference	Location	Study design[1]	n_t, n_c[2]	Planting type[3]	Richness	Size (m^2)	Maturity (years)[4]		Other responses	Other predictors
In meta-analysis										
Blaauw & Isaacs, 2014	US (MI)	Paired design, 175-400m apart w/in pairs, ≥9.5km between sites; control=unplanted edges	5,5	P	15	600-11,000	0,1,2,3	+ abundance	cost/benefit analysis	
Campbell et al., 2017a	England	Paired design, 100m apart w/in pairs, all sites within a 15,000x15,000m area; 2 treatment, 1 control (n=4 ea); control=grass margin mowed every 2 weeks	4,4	A, P	11-14	40	0,1	+ pollinator richness + *Bombus* abundance + solitary bee abundance (after yr 1 in planting)	natural enemy abundance, richness; aphid pests; predation on sentinel moth eggs	
Campbell et al., 2019	England	Paired design, 500m apart w/in pairs, ≥3000m between sites; control=unplanted edges	8,8	A,P	9	1000	1,2	+ abundance + richness	herbivorous arthropod and natural enemy abundance	
Carvell et al., 2004	England	Paired design, all treatments on same edge, distance between sites NR; 2 treatment, 3 control (n=3 ea); control=unplanted edge (natural regeneration), grass, crop	3,3	P	26	432 (divided btwn. 5 treatments)	0,1,2	+ *Bombus* abundance		
Carvell et al., 2015	England	Paired design, 3000m apart w/in pairs, distance between sites NR; 3 treatment, 1 control (n=8 ea); control=unplanted edges	8,8	A/P	2	2500, 5000, 10,000	2,3,4	+ *Bombus* abundance (males and gynes) Landscape x planting: (-) *Bombus* abundance Spillover: 0 *Bombus* abundance Size: + for some spp.		Landscape (% cropland), planting size, planting floral density

								Size x spillover: + effect of size on *Bombus* abundance in surrounding landscape		
Holland et al., 2015	England	Block design, distance w/in blocks and between sites NR; 2 treatments, 3 control (n=28 ea); control=wild bird seed mix, unplanted margins (natural regeneration), crop	28,28	P	NR	NR (varied)	1/2,3,4	+ *Bombus* abundance + solitary bee abundance		floral cover, surrounding wildlife habitat, organic mangement
Jönsson et al., 2015	Sweden	Both paired and unpaired design, 50-800m apart w/in site, distance NR between sites; control=unplanted edges	9,9 (1,2 w/in sites)	NR (varied)	NR (varied)	2-20m wide, 35-2900m long	NR (varied, existing plantings)	+ *Bombus* abundance (w/in sites and between sites) 0 solitary bee abundance (w/in sites and between sites)		Landscape (% natural area), distance from flower strip
Kleijn et al., 2018	Netherlands	Paired design, 2400m apart w/in pairs, distance between pairs NR; control=unplanted edges	10,10	NR	NR	4000-49000, 24000 avg	1,2	+ *Bombus* abundance 0 solitary bee abundance Spillover: + *Bombus* and solitary bee abundance in landscape (extrapolated)		Landscape (% natural area), planting size
Kohler et al., 2007	Netherlands	Unpaired design, ≥4000m between sites; control=unplanted edges	5,5	P	17	100	≥15	+ abundance + density	life history traits (reproductio n, dispersal, etc.)	flower abundance, distance from plantings
Kremen et al., 2018	US (CA)	BACI design unpaired, ≥1000m between sites; control=unplanted edges	5,10	H	NR	1050-2100	0,1,2,3,4,5,6,7,(8)	+ richness + diversity + functional dispersion 0 evenness		Landscape (% natural area), in-planting floral diversity
Lundin et al., 2017	US (CA)	Paired design, 200m apart w/in pairs, 400m between sites; control=unplanted edges	3,3	A	7	4800	0/1,2	+ abundance		
Meek et al., 2002	England	Paired design, all treatments on same edge, distance between sites NR; 2 treatment, 3 control (n=4 ea); control=unplanted edges (natural regeneration), grass, crop	4,4	P	26	432 (divided btwn. 5 treatments)	1	+ *Bombus* abundance		
Morandin & Kremen, 2013	US (CA)	Unpaired design, min 1000m, mean 3000m between sites; control=unplanted edges	6,8	H	NR	305-550 long	≥10	+ abundance + richness + diversity	specialist vs. generalist spp., nesting	

								+ abundance of uncommon species + beta diversity		
Nicholson et al., 2020	US (MI, OR, CA)	Unpaired design: MI, OR, distance between sites NR, paired design: CA, min 400m, most >1000 w/in pairs, distance between pairs NR; control=unplanted edges	15,15 (all regions)	NR	NR	405-5260 (avg 2023)	NR	+ abundance + richness + diversity + functional diversity		
Pywell et al., 2005	England	Paired design, distance w/in and between pairs NR; 1 treatment, 3 control; control=unplanted edges (natural regeneration), unplanted edges (cereal field margin), crop (conservation headland)	28,92	NR	NR	NR	NR	+ *Bombus* abundance + *Bombus* richness		
Pywell et al., 2006	England	Paired design, all treatments in same edge, distance between sites NR; 2 treatments, 3 controls (32 ea); control=grass, crop	32,32	A/P, P	4, varied	NR	NR	+ *Bombus* abundance + *Bombus* richness	Short vs. long tongued bumblebees	landscape heterogenei ty, local floral abundance, planting richness
Pywell et al., 2011	England	Paired design, all treatments in same edge, distance between sites NR; 2 treatments, 3 control (3 ea); control= unplanted edges (natural regeneration), grass, crop	3,3	NR	17	432 (divided btwn. 5 treatments)	NR	+ abundance + richness		
Pywell et al., 2012	England	Unpaired design, distance between sites NR; 1 treatment, 2 controls; control=grass, crop	38,76	NR	NR	NR	NR	+ *Bombus* abundance + *Bombus* richness	plants, birds	
Pywell et al., 2015	England	Block design, distance w/in and between blocks NR; 2 treatment, 2 control; control=grass, crop	5,5	P	29	3-8% of crop area	NR	+ abundance + richness	cost-benefit, natural enemies	
Rosas-Ramos et al., 2019	Spain	Unpaired design, distance between sites NR; control=unplanted edges	3,3 hedge 3,3 forb	H, P	NR	NR	NR	0 abundance 0 richness		
Sardiñas & Kremen, 2015	US (CA)	Unpaired design, 900-5409m between sites; control=unplanted edges	9,9	H	NR	750-1800	NR	+ abundance + richness + sunflower specialists	Bee spp. in planting vs. crop, common vs rare spp., specialists	distance from field edge, landscape (% natural area)
Scheper et al., 2015	Germany, Sweden,	BACI design, unpaired ≥2000m between sites; control=unplanted edges (mowed 1-3x/yr)	8,8	P	12	300	-1,1,2	+ *Bombus* abundance (by yr 2)		Land-use intensity, landscape

	Netherlands, UK							+ *Bombus* richness (some locations) + solitary bee abundance + solitary bee richness		composition /complexity, landscape-wide flower availability
Williams et al., 2015	CA, MI, FL	Paired design, all treatments on same edge, ≥900m between sites; 6 treatments, 1 control (9 ea); control=unplanted margins	9,9	A, P	4-5, 9-11	270	1,2,(3)	+ abundance + richness + *Bombus* abundance Planting richness: 0 abundance, richness, *Bombus* abundance		Annual vs perennial plantings, flower mix phenology
Wood et al., 2018	US (MI)	Unpaired design, distance between sites NR; control=unplanted edges	5,5 (ea blueberry, cherry)	NR	NR	NR	2,3,4	+ abundance + richness	pollen diet, short vs long-season bees	
Not in meta-analysis										
Buhk et al., 2018	Germany	Paired design, 2x50ha study areas at each site, distance between sites NR; control=unplanted edges	2,2	A,P	NR	10% of land area	-1,0,1,2,3,4	+ abundance + richness + diversity + oligolectic species richness + oligolectic species abundance	Butterfly richness	
Carvell et al., 2007	England	Paired design, all treatments on same edge, distance between sites NR; 2 treatment 4 control (6 ea); control=unplanted edge (natural regeneration), grass, crop (normal management and conservation headland)	6,6	A, P	21, 4	NR	1,2,3	+ *Bombus* abundance + *Bombus* richness 0 abundance of rare species		planting richness and composition
Carvell et al., 2011	England	Paired design, 3000m apart w/in pairs, distance between sites NR; control=unplanted edge	8,8	P	3	2500, 5000, 10,000	2,3,4	+ *Bombus* density + *Bombus* richness		landscape (composition, floral cover), patch size
Feltham et al., 2015	Scotland	Paired design, 500m apart w/in pairs; control=unplanted edges	6,6	A	NR	300	0 or 1	+ abundance	costs	
Grab et al., 2019	US (MI, NY)	NY: Paired design, 200m apart w/in pairs, distance between sites NR; MI: unpaired design ≥2000m between sites; control=unplanted edges	NY: 17,17; MI: 13,23	P	NR	NR	NR (5 yrs data)	+ plantings buffered against decreases in bee body size as natural area decreased		Landscape (% natural area)

Haaland & Gyllin, 2010	Sweden	Unpaired design, distance between sites NR; control=grass strips mowed a few x/yr	1,3	NR	NR	Unclear	≥15	+ *Bombus* abundance		
Kremen & M'Gonigle, 2015	US (CA)	Unpaired design, ≥1000m between sites; control=unplanted edges	36,36	H	NR	1050-2100	1,2,3,4,5,6,7	+ abundance + occurrence + above-ground nesting bees + oligolects	trait-based analyses (e.g. diet, nesting, size)	
Krimmer et al., 2019	Germany	Unpaired design, ≥2100m between sites; control=calcereous grassland	23,4	NR	NR	2900-30000	≥1	+ abundance Size: marginal effect of planting size on abundance Size x landscape: visits to small plantings increased with natural habitat, large plantings not affected Bees in landscape: large, old flower fields greatest spillover into crop		Landscape (% natural area), size, planting age
M'Gonigle et al., 2015	US (CA)	BACI design unpaired, 1000 min, 3000m mean between sites; control=unplanted edges	5,10	H	NR	1050-2100	0,1,2,3,4,5,(6)	+ richness + persistence + colonization of specialist species	specialist vs generalist bees	
Ponisio et al., 2019	US (CA)	Unpaired design, 2000 min, 15,000m mean between sites; control=unplanted edges	21,24	H	NR	1050-2100	1 to >10	+ beta diversity (mature hedges only) + trait uniqueness (mature hedges only) + trait diversity (mature hedges only) + trait evenness (mature hedges only)		hedgerow plant community, hedgerow maturity, nesting resources
Pontin et al., 2006	New Zealand	Block design, 15-30m apart w/in blocks, distance between blocks NR; 4 treatment, 1 control (4 ea); control=crop	4,4	A	1, 2, 7	24	0	+ *Bombus* abundance		Planting richness
Sanchez et al., 2019	Spain	Paired design, 20m apart w/in pairs, distance between sites NR; control=bare dirt	2,2	A/P, H	11	75 (divided btwn. 3 treatments)	1,2	+ abundance + diversity		
Sardiñas et al., 2016	US (CA)	Unpaired design, ≥1000m between sites; control=unplanted edges	8,8	H	NR	NR	5+	'- ground nesting bee abundance - ground nesting bee richness (emergence trap data)	nest site quality	

Schulte et al., 2017	US (IA)	Block design, ≥36m apart w/in blocks, distance between blocks NR; control=crop	9,3	NR	≥32	NR	2,3,4	+ abundance (data combined between planting and crop?)	runoff, perennial cover, all insects, birds	Planting area (as % of crop)
Wood et al., 2015	US (MI)	Unpaired design, ≥5400m between sites; control=unplanted margins	8,9	P	NR	NR (varied)	NR (established previously)	+ *Bombus* abundance + *Bombus* nest density		Number of colonies by spp., foraging range by spp., pollen preference
Single-site studies										
Quinn et al., 2017	US (MI)	Block design, 40-50m apart w/in blocks; control=crop	1	A	1	NR	0,1	+ abundance		sampling location (distance from plantings)

Notes:

[1] If a study had more than 2 types of plantings the number of treatments and controls is reported in the study design column. “Treatments” are defined as flower plantings, while all non-flower treatments are listed as controls regardless of what the authors considered them. For studies with multiple controls listed, reported results compare the planting or planting treatment(s) to unplanted edges rather than other potential controls like grass or crop, if possible.

[2] n_t = sample size for treatment, n_c = sample size for control

[3] Planting type: A = annual; H=hedge; P=perennial; A/P = mixed annual and perennial; A, P = both annual and perennial

[4] Planting maturity: 0 = the year the planting was established, 1 = one year post-establishment …

Table S2. Studies identified via our systematic review evaluating the effect of field-edge plantings on bees in crop fields adjacent to plantings and crop yields (ecosystem service provision and delivery). All results are those reported by authors rather and are reported for all pollinators (which sometimes included non-bee flower visitors) unless a specific bee group is specified. Throughout, NR=not reported.

Study features				Planting features					Reported results	Additional info	
Reference	Location	Crop	Study design[1]	n_t, n_c[2]	Planting type[3]	Richness	Size (m^2)	Maturity (years)[4]		Other responses	Other predictors
In meta-analysis											
Blaauw & Isaacs, 2014a	US (MI)	Blueberry	Paired design, 175-400m apart within pairs, 9.5km min between sites; control=unplanted edges	5,5	P	15	600-11,000	0,1,2,3	+ abundance + visitation rate + yield	cost/benefit analysis	
Campbell et al., 2017a	England	Apple (cider)	Paired design, 100m apart w/in pairs, all sites within a 15,000x15,000m area; 2 treatment, 1 control (n=4 ea); control=grass margin mowed every 2 weeks	4,4	A, P	11-14	40	0,1	0 fruit weight, size, number	natural enemy abundance, richness; aphid pests; predation on sentinel moth eggs	
Campbell et al., 2017b	England	Apple (cider)	Unpaired design, 500m between sites; control=unmanaged edges	4,4	NR	NR	500	2	+ visitation rate (marginal) + *Bombus* visitation rate 0 andrenid visitation rate 0 fruit set	Foraging behavior	dandelion abundance, distance to natural area
Ganser et al., 2018	Switzerland	Strawberry	Unpaired design, 3000m between sites; control=regularly mowed grass	12,7	A/P	8 (avg)	480	0,1	0 seed set		distance from field edge
Feltham et al., 2015	Scotland	Strawberry	Paired design, 500m between sites; control=unplanted edges	6,6	A	NR	300	0 or 1	+ abundance	costs	
Morandin & Kremen, 2013	US (CA)	Tomato (processing)	Unpaired design, 1000m min, mean 3000m between sites; control=unplanted edges	6,8	H	NR	305-550 long	≥10	+ abundance + richness	specialist vs. generalist spp., nesting	
Nicholson et al., 2020	US (MI, OR, CA)	sour cherry (MI), blueberry (MI, OR), watermelon (CA)	Unpaired design: MI, OR, distance between sites NR, paired design: CA, ≥400m (most >1000m) apart w/in pairs; control=unplanted edges	15,15 (all region s)	NR	NR	405-5260 (avg 2023)	NR	0 abundance 0 richness 0 evenness 0 functional diversity		
Rundlöf et al., 2018	Sweden	Red clover	Unpaired design, 2500m between sites; control=NR	22,22	A	1	125-2000	0	0 *Bombus* density + *Bombus* richness	pest (weevil) abundance, parasitism rate	Landscape (% cropland),

									(at 30% cropland in landscape but not at 60% and only in 2009) 0 seed yield		planting size, crop field size
Sardiñas & Kremen, 2015	US (CA)	Sunflower	Unpaired design, 900-5409m between sites; control=unplanted edges	9,9	H	NR	750-1800	NR	0 abundance 0 richness 0 visitation rate + sunflower specialists 0 seed set	Bee spp. in planting vs. crop, common vs rare spp., specialists	distance from field edge, landscape (% natural area)
Sutter et al., 2018	Switzerland	Oilseed rape	Unpaired design, distance between sites NR; control=unplanted edges	12,6	H, P	NR	NR	NR	0 pollinator visitation + proportion seed set bagged vs unbagged 0 yield Landscape x planting: + (marginal) interaction between planting and landscape	predation, parasitism, ground-dwelling arthropods	landscape (% “greening measures” e.g., restored prairie, flower strips, forest edges, cover crops)
Venturini et al., 2017b	US (ME)	Blueberry	Paired design, ≥1500m apart w/in pairs, distance between sites NR; control=unplanted edges	3,3	A/P	11	500	0,1,2	+ visitation (yr 2 only) 0 richness 0 diversity 0 evenness + fruit set (yr 2 only)	*Bombus* pollen load, economics	planting richness
Wood et al., 2018	US (MI)	Blueberry, tart cherry	Unpaired design, ≥5400m between sites; control=unplanted margins	5,5 (ea blueberry, cherry)	NR	NR	NR	2,3,4	0 abundance 0 richness	pollen diet, short vs long-season bees	
Not in meta-analysis											
Carvalheiro et al., 2012	South Africa	Mango	Unpaired design, 50-250m between sites; control=unplanted edges	4,32	P	2	25	0 (transplant)	+ abundance + richness (possibly both + far from natural area only but unclear) + fruit production	costs	distance to natural habitat, organic vs. conventional

Grab et al., 2018	US (NY)	Strawberry	Paired design ≥200m apart w/in pairs, distance between sites NR; control = regularly mown edge	12,12	P	9	40	1,2,3	+ visitation + yield	biocontrol, pest abundance, crop damage	Landscape (% natural area)
Herbertsson et al., 2018	Sweden	Field bean, woodland strawberry	Unpaired design, ≥800m between sites; control=unplanted edges	7,7 straw berry, 9,9 bean	A/P	12, strawberr y; NR bean	300 strawberry ; varied bean	0-2	+/- yield (depended on landscape: + in homogenous, - in hetereogeneous)		Landscape (hetereogen eity)
Korpela et al., 2013	Finland	Canary grass	Block design, 25m apart w/in blocks, distance between blocks NR; 2 treatment, 2 control (6 ea); control= unplanted edges, crop (cereal and reed canary grass)	6,6	P	5	250	0,1,2,3	0 *Bombus* abundance		Planting location (edge/cente r), planting shape
Krimmer et al. 2019	Germany		Unpaired design, ≥2100m between sites; control=calcereous grassland	23,4	NR	NR	2900-30000	≥1	+ / - visitation rate: significantly lower next to large, new plantings; significantly higher next to older plantings Size: marginal effect of planting size on abundance Size x landscape: visits to small plantings increased with natural habitat, large plantings not affected Bees in landscape: large, old flower fields greatest spillover into crop		
Phillips & Gardiner, 2015	US (OH)	Pumpkin	Unpaired design, ≥4250m between sites; control=grass mowed 2x per month	8,6	A, P	perennial planting: 23; alyssum monocult ure	36	perennial: 2, Alyssum: 0	0 visitation rate - *Bombus* visitation rate 0 pollen deposition		landscape composition , planting richness
Pywell et al., 2015	England	Wheat, field beans, rape	Block design, distance w/in blocks and between sites NR; 2 treatment, 2 control; control=grass, crop	10,5	P	29	3-8% of crop area	NR	+ yield	cost-benefit, natural enemies	
Single site studies											
Azpiazu et al., 2020	Spain	Melon	Block design (3 reps of each treatment), 10m apart w/in blocks; control=unplanted edges	1	A	11	10	0,1	+ abundance		Sampling location (distance

											from plantings)
Barbir et al., 2015	Spain	Coriander	Paired design (3 reps of each treatment), 20-30m apart w/in pairs, 200m between pairs; control=bare soil	1	A, A/P	1, 6	10.5	0	+ visitation rate + yield		
Balzan et al., 2016	Spain	Tomato	Block design (4 reps of each treatment), 8-15m apart w/in blocks; control=bare soil	1	A/P	3, 6, 9	8	0,1,2	+ abundance (only in most diverse treatment in yr 2) + yield		Planting functional diversity
Quinn et al., 2017	US (MI)	Cucumber	Block design (6 reps of each treatment), 40-50m apart w/in blocks; control=crop	1	A	1	NR	0,1	0 abundance + yield (yr 1 only) 0 cucumber grade		sampling location (distance from plantings)
Schulte et al., 2017	US (IA)	Corn, soy	Block design (3 reps of each treatment), ≥36m apart w/in blocks; control=crop	1	NR	≥32	NR	2,3,4	0 yield	runoff, perennial cover, all insects, birds	Planting area (as % of crop)

Notes:

[1] If a study had more than 2 types of plantings the number of treatments and controls is reported in the study design column. “Treatments” are defined as flower plantings, while all non-flower treatments are listed as controls regardless of what the authors considered them. For studies with multiple controls listed, reported results compare the planting or planting treatment(s) to unplanted edges rather than other potential controls like grass or crop, if possible.

[2] n_t = sample size for treatment, n_c = sample size for control

[3] Planting type: A = annual; H=hedge; P=perennial; A/P = mixed annual and perennial; A, P = both annual and perennial

[4] Planting maturity: 0 = the year the planting was established, 1 = one year post-establishment …

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Appendix II: Methods and analysis details

SYSTEMATIC REVIEW

Literature search and screening

In addition to our Web of Science database search (detailed in Methods), we evaluated several highly-cited papers (Blaauw & Isaacs, 2014; Scheper et al., 2015; Wood et al., 2018) and 5 related reviews (Haaland et al., 2011; Holland et al., 2017; Scheper et al., 2013; Uyttenbroeck et al., 2016; Venturini et al., 2017) for additional references that were not captured in the initial search. Thirteen papers were added from these references. Additionally, abstracts from the new papers were evaluated for search terms that would have ensured these papers were captured in the initial search. Using these new search terms, we conducted two additional literature searches. The first included (the original phrase of planting terms) AND (“bumblebee” or “bumble*”). The second search included (“floral supplement*” or “greenway*” or “colourful fallow*” or “improved field margin*” or “biodiversity fallow*”) AND ([the original phrase of bee related search terms] or “bumblebee” or “bumble*”). Together these searches returned 217 results, only 1 of which was deemed relevant. Finally, we searched for papers using the “view related records” tool in Web of Science, which generates a list of papers that share citations with a selected paper. We used this tool on Blaauw and Isaacs (2014a), one of the most cited papers from the original search and limited our review of related records to those that shared at least 5 citations with the original paper. The related records search yielded 675 results including 3 papers which were added to the review. In total, these additional searches added 17 papers.

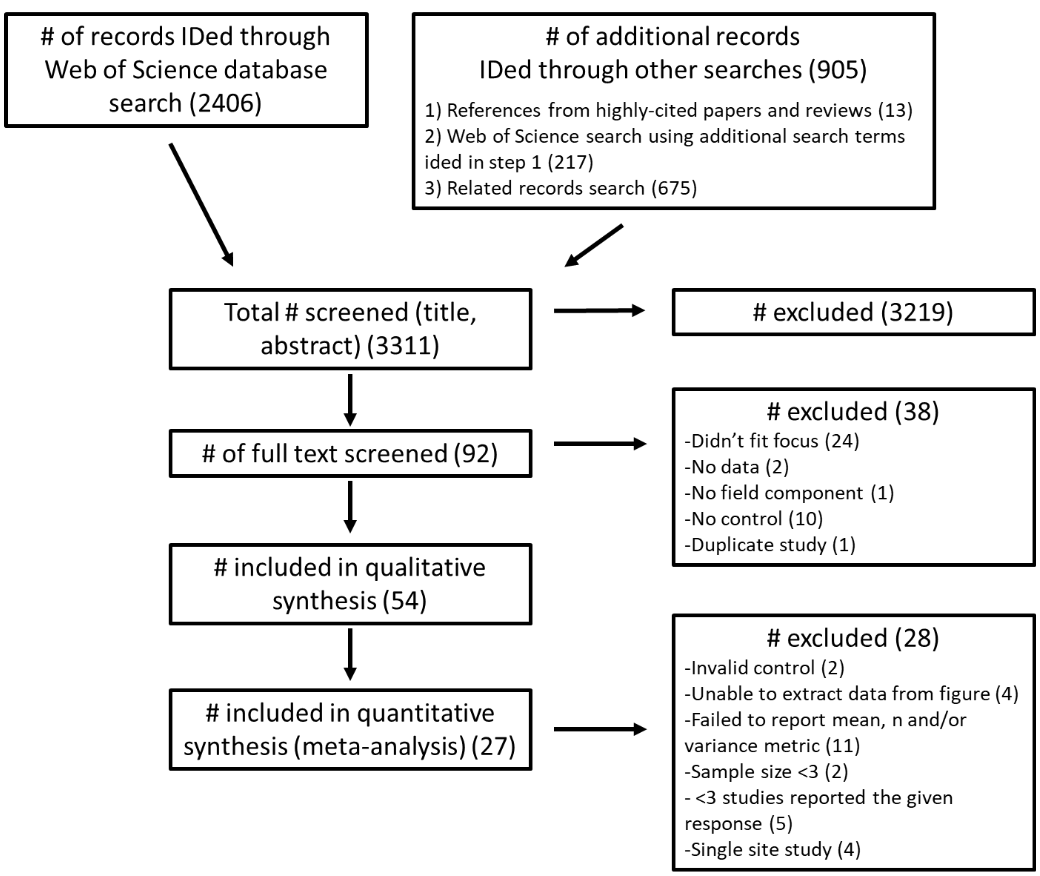


Figure S1. PRISMA flow diagram (Liberati et al. 2009) detailing the systematic review process

META-ANALYSIS

Inclusion criteria – exceptions to our rules

We made an exception to the rule that all studies must include a spatial replicate for one study (Lundin et al., 2017) because their site locations within a single study site were separated by ≥400m (greater than the minimum distance for several of the "multi-site" studies). Additionally, we excluded two studies from our analyses of bees in field edges because they compared bees in planted margins to bees in a control that might cause us to over- or underestimate the effect of the planting. These controls included calcareous grassland (a high-quality habitat type that could under-estimate planting effectiveness relative to the most common control, unplanted crop field edges) and the crop itself (over-estimate planting effectiveness relative to unplanted crop field edges). Such controls were considered acceptable for analyses of pollinators in crop fields or crop yields.

Methods for avoiding pseudo-replication

We used a number of methods to avoid pseudo-replication depending on how data was aggregated and reported. For studies of perennial plantings that reported results separately for multiple years of data, we used the most recent year as the best measure of the effect of a mature planting. For studies with annual plantings that reported multiple years of data we calculated a within-study Hedges' d and variance estimate for each year and averaged them. This produced an estimated pooled mean within-study effect size and pooled within-study variance which were used in the overall analysis (Scheper et al. 2013). We took the same pooled-d approach if a study reported results separately for *Bombus* and solitary bees or for different amounts of surrounding natural area. For studies that included multiple types of non-planted margins (e.g. grass margins, crop fields, conservation headlands, unplanted edges), we chose the treatment that most closely resembled a fallow, unplanted margin. This was both the most conservative and common type of control across studies, as well as the one we considered the most appropriate comparison to planted field edges. If a study included multiple forb planting treatments (e.g., multiple planting sizes or flower mixtures) we averaged the planting treatment means and calculated a pooled SD. We took the same approach for studies that reported results separately for multiple months, multiple locations within a crop field (e.g. transects increasing in distance from the field edge), rare and common bees, or multiple bee species within the same genus (e.g. multiple species of *Bombus*).

The use of multilevel models has been proposed as an alternative to averaging techniques such as these (Nakagawa et al. 2017). However, multi-level models were inappropriate in this case because the nature of categories across which within-study data was aggregated (year, treatment, species, etc.) were very different study-to-study, precluding our ability to accurately calculate the study-level random effects term that would be included in a multi-level model.

Other notes on data extraction

Many studies reported data in figures only. For these studies data was extracted using WebPlotDigitizer (Rohatgi 2012). For studies that reported confidence intervals we used a t-value appropriate for the sample size (Higgins et al. 2019) to back-calculate the standard deviation. If a study reported the median and interquartile range (IQR), we used an online calculator based on methods developed by Luo et al. (2018) and Shi et al. (2020) which are accurate for skewed data, to back-calculate an estimate of the mean and standard deviation. The vast majority of papers showed bar charts with means and standard errors. Therefore, if papers included bar charts that showed the mean but did not expressly state what error bars represented, they were assumed to represent standard error. Conversely, if a paper included a bar chart that showed standard error but did not state what the bar height represented, it was assumed to represent the mean.

Normality, publication bias, and heterogeneity

Before performing these analyses, we assessed the normality of effect size estimates using normal quantile-quantile plots (Wang and Bushman 1998) and evaluated publication bias using contour-enhanced funnel plots for groups with k≥10 (Peters et al., 2008). No groups showed strong non-normality or publication bias. We assessed between-study heterogeneity using Cochran's Q and Higgin's and Thompson's I. Heterogeneity was high for a few groups: up to 82% for the *Bombus* subgroups in the field-edge analyses. However, for all other groups, heterogeneity was <55% which is moderate, particularly within an ecological context where studies may average up to 92% heterogeneity (Nakagawa et al., 2017).

Outliers

A study was considered an outlier if its confidence interval did not overlap the overall confidence interval. For all groups that included outliers, we conducted analyses with and without the outlier. Results from these analyses are reported below.

Analyses of pollinators in plantings

Holland et al. 2015 was an outlier that showed strong positive effects of plantings on pollinator abundance within plantings as well as *Bombus* abundance within plantings for the *Bombus* subgroup analysis (Figure S2). Likewise, Williams et al. 2015 was an outlier for pollinator richness in plantings as well as *Bombus* abundance for the *Bombus* subgroup analysis, while Scheper et al. 2015 (UK) was a positive outlier for the *Bombus* richness analysis (Figure S2). We ran analyses excluding these outliers (results below). Since none of the outliers changed the significance of results all results reported in the body of the paper are for the full dataset.

Table S3. Model estimates for the effect of plantings on pollinator abundance and richness in plantings. For each group model estimates are presented with and without outliers. Results are from random effects models fit with REML.

Response	k	% variability unexplained by sampling error (I^2)	Estimate	CI	P-value
Pollinator abundance	29	40.94%	1.26	(0.97, 1.54)	<0.001*
Pollinator abundance w/o Holland et al. 2015	28	38.13%	1.22	(0.94, 1.49)	<0.001*
Pollinator richness	20	28%	1.12	(0.83, 1.41)	<0.001*
Pollinator richness w/o Williams et al. 2015 (CA)	19	27.93%	1.11	(0.82, 1.39)	<0.001*
Bombus abundance	17	76.28%	1.59	(1.02, 2.05)	<0.001*
Bombus abundance w/o Williams et al. 2015 (CA) and Holland et al. 2015	15	72.95%	1.49	(0.96, 2.02)	<0.001*
Bombus richness	7	82.13%	1.39	(0.69, 2.08)	<0.001*
Bombus richness w/o Scheper et al. 2015 (UK)	6	59.11%	1.08	(0.62, 1.54)	<0.001*

<u>Analyses of pollinators in crop fields</u>

Sardiñas and Kremen 2015 was an outlier for our analysis on pollinator abundance in crop fields (Figure S3). This study was the only study in our review that showed significant negative impacts of plantings. Removing this study made effect sizes approximately twice as positive as they were originally negative (see below), however it did not change the significance of results.

Table S4. Model estimates for the effect of plantings on pollinator abundance and richness within crop fields. For each group, model estimates are presented with and without Sardiñas and Kremen 2015. Results are from random effects models fit with REML.

Response	k	% variability unexplained by sampling error (I^2)	Estimate	CI	P-value
Pollinator abundance	12	55.65%	0.18	(-0.28, 0.64)	0.44
Pollinator abundance w/o Sardiñas and Kremen 2015	11	19.70%	0.26	(-0.09, 0.61)	0.15

Subgroup analyses (study region and rigor)

Table S5. Model estimates for the effect of plantings on pollinators from subgroup analyses comparing studies conducted in the US to studies conducted in Europe. Analyses were limited to groups with k>3. Results are from random effects models fit with REML.

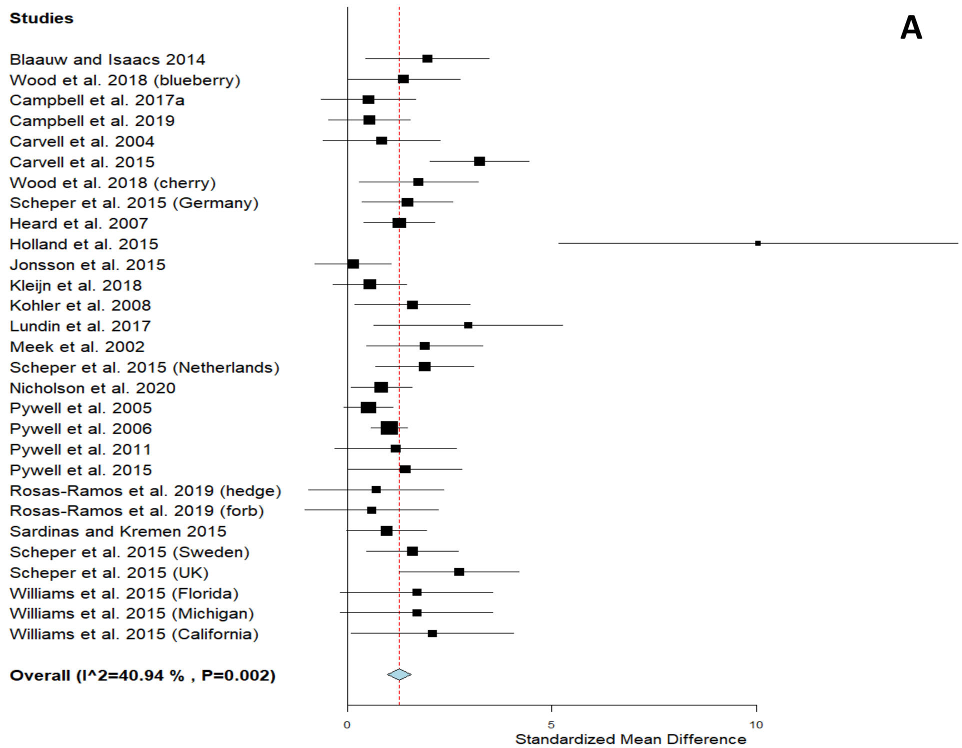
Response	k	% variability unexplained by sampling error (I^2)	Estimate	CI	P-value
Pollinators within plantings					
Pollinator abundance (US)	9	2.20%	1.32	(0.87, 1.76)	<0.001*
Pollinator abundance (Europe)	20	53.95%	1.22	(0.85, 1.58)	<0.001*
Pollinator richness (US)	9	40.41%	1.49	(0.82, 2.17)	<0.001*
Pollinator richness (Europe)	11	37.91%	1.04	(0.68, 1.39)	<0.001*
Pollinators in crop fields					
Pollinator abundance (US)	7	75.38%	0.26	(-0.64, 1.15)	0.57
Pollinator abundance (Europe)	5	0%	0.07	(-0.32, 0.47)	0.71

Table S6. Comparison of model estimates of the effect of plantings on pollinator and bee groups using the full data set versus a subset of data containing only the most rigorous studies.[1] Analyses were limited to groups with k>3. Results are from random effects models fit with REML.

Response	k	% variability unexplained by sampling error (I^2)	Estimate	CI	P-value
Pollinators within plantings					
Pollinator abundance	29	40.94%	1.26	(0.97, 1.54)	<0.001*
Pollinator abundance subset	15	37.61%	1.53	(1.14, 1.93)	<0.001*
Pollinator richness	20	28%	1.12	(0.83, 1.41)	<0.001*
Pollinator richness subset	13	25.99%	1.43	(0.99, 1.87)	<0.001*
Bombus abundance	17	76.28%	1.59	(1.02, 2.05)	<0.001*
Bombus abundance subset	10	63.12%	1.97	(1.27, 2.66)	<0.001*
Bombus richness	7	82.13%	1.39	(0.69, 2.08)	<0.001*
Bombus richness subset	4	72.59%	1.97	(0.80, 3.14)	<0.001*
Solitary bee abundance	8	47.26%	0.93	(0.41, 1.44)	<0.001*
Solitary bee abundance subset	5	39.31%	1.05	(0.45, 1.64)	<0.001*
Pollinators in crop fields					
Pollinator abundance	12	55.65%	0.18	(-0.28, 0.64)	0.44
Pollinator abundance subset	7	71.6%	-0.007	(-0.70, 0.68)	0.98

[1]Studies included in the subset analysis had a minimum sample size of 5 in both the treatment and control group, included a minimum of 2 years of data, and compared plantings to a fallow, unplanted field edge (the most common, conservative, and appropriate control to a planted margin). In addition, all but one was an unpaired design with a minimum of 1000m between sites (all sites were independent). The one study with a paired design had a minimum of 9.5km between pairs.

Figure S2. Effects of field-edge plantings on a) pollinator abundance and b) pollinator richness in field edges by study. Results are from random effects models including all pollinator groups. Mean effect sizes (Hedge's *d*) +/- CI are illustrated. Overall effect sizes are represented by the blue diamonds.



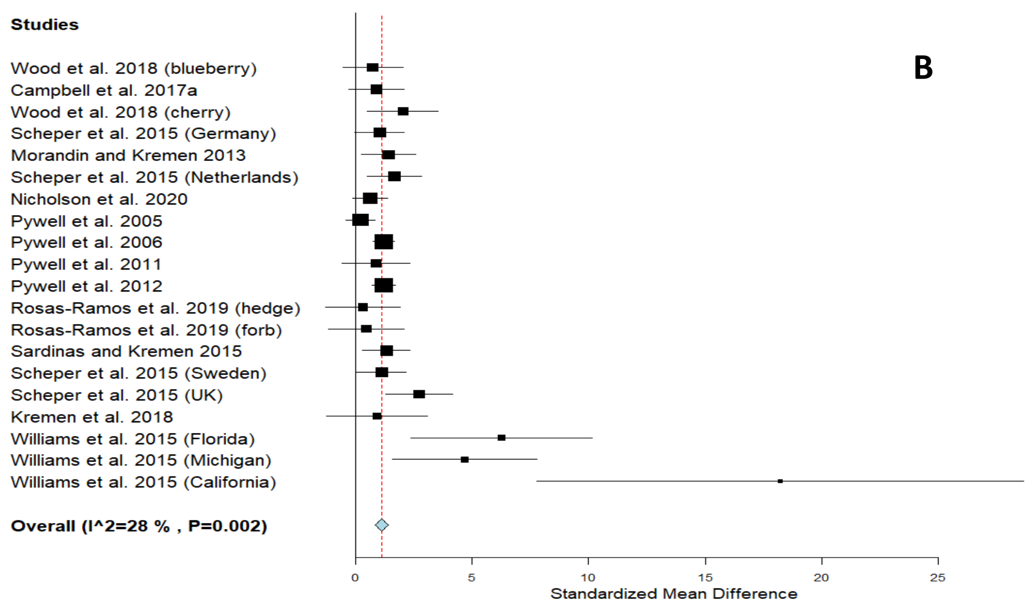


Figure S3. Effects of field-edge plantings on a) pollinator abundance and b) pollinator richness in crop fields by study. Results are from random effects models including all pollinator groups. Mean effect sizes (Hedge's *d*) +/- CI are illustrated. Overall effect sizes are represented by the blue diamonds.

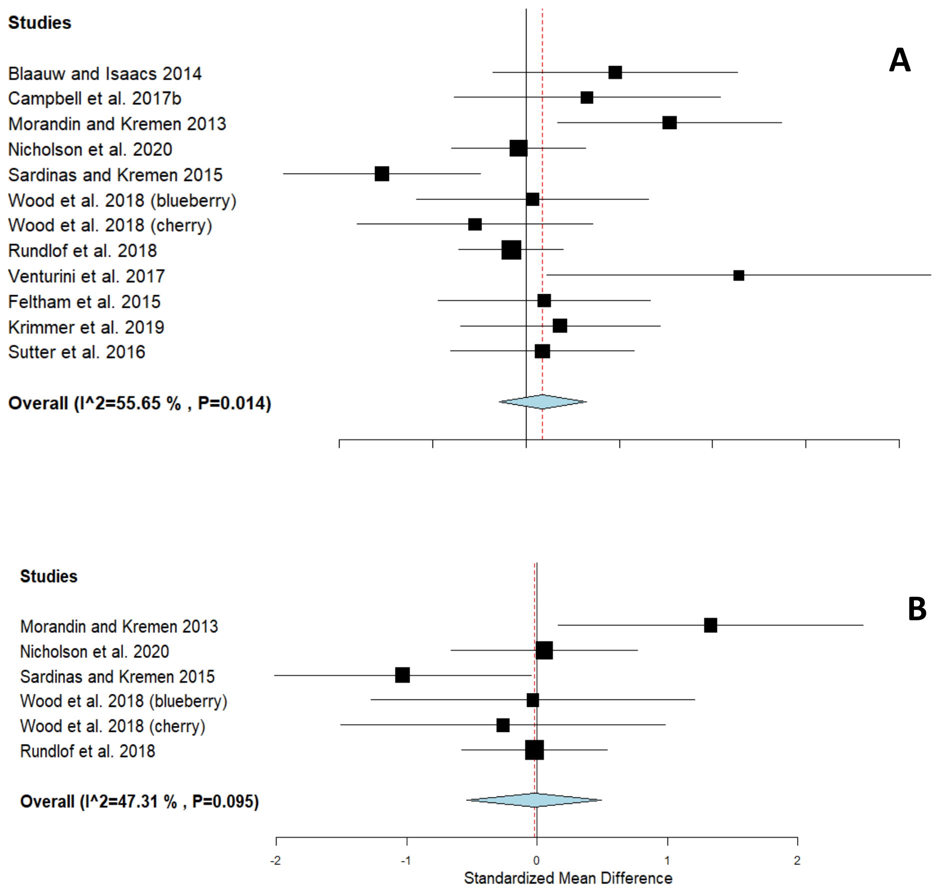
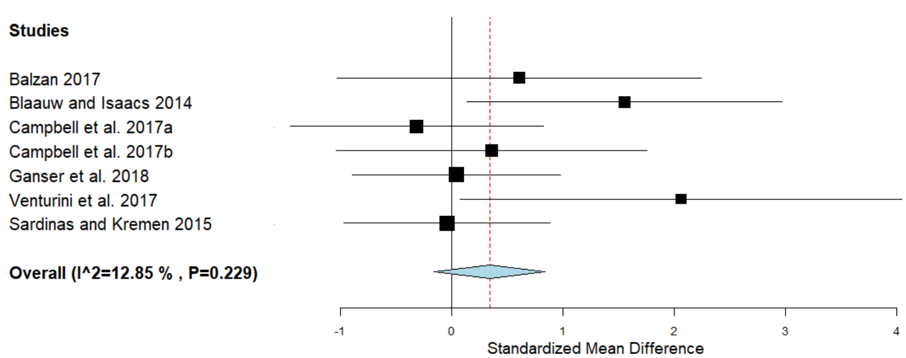


Figure S4. Effects of field-edge plantings on crop yields by study. Results are from random effects models including all pollinator groups. Mean effect sizes (Hedge's *d*) +/- CI are illustrated. Overall effect sizes are represented by the blue diamonds.



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Appendix III: Supporting figures

Figure S5. Histogram of studies in the systematic review by year of publication.

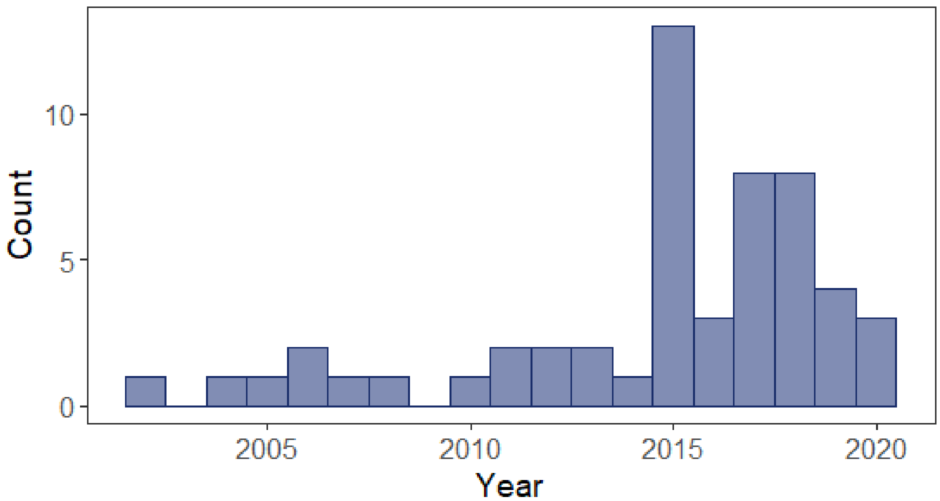
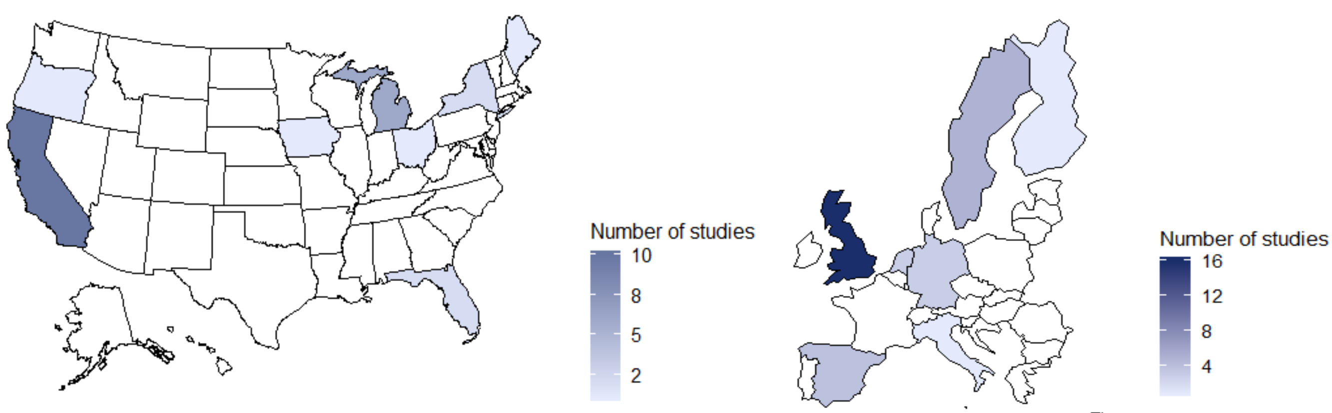


Figure S6. Map of study locations minus a single study conducted in New Zealand and one conducted in South Africa. Shading represents the number of studies in each location



Studies conducted in the US represented all regions except for the Southwest. 10 studies were conducted on the West Coast, with all but one of these in California. Eight studies were conducted in the Midwest (primarily in Michigan), 3 were conducted in New England, and 2 were conducted in Florida. The European studies were strongly biased toward the UK (16 studies). Eight additional countries (all in Western Europe and Scandinavia) made up the remaining 20 studies.

Table S7. Crop by study. If studies evaluated multiple crops they are included multiple times. ES=ecosystem service.

Citation	Crop type	Measured ES provision or delivery?
Lundin et al. 2017	Almond	N
Campbell et al. 2017a	Apple (cider)	Y
Campbell et al. 2017b	Apple (cider)	Y
Blaauw and Isaacs 2014	Blueberry	Y
Venturini et al. 2017	Blueberry	Y
Wood et al. 2018	Blueberry	Y
Nicholson et al. 2020	Blueberry	Y
Pontin et al. 2006	Broccoli	N
Korpela et al. 2013	Canary grass	N
Carvell et al. 2007	Cereal	N
Meek et al. 2002	Cereal	N
Wood et al. 2015	Cereal/oilseed rape	N
Carvell et al. 2004	Cereal/oilseed rape	N
Pywell et al. 2011	Cereal/oilseed rape	N
Pywell et al. 2015	Cereal/oilseed rape	Y
Krimmer et al. 2019	Oilseed rape	Y
Sutter et al. 2016	Oilseed rape	Y
Nicholson et al. 2020	Cherry (sour)	Y
Wood et al. 2018	Cherry (tart)	Y
Babir et al. 2015	Coriander	Y
Schulte et al. 2017	Corn/Soy	Y
Quinn et al. 2017	Cucumber	Y
Herbertsson 2018	Field bean	Y
Pontin et al. 2006	Lucrene (*Medicago sativa*)	N
Carvalhiero et al. 2012	Mango	Y
Azpiazu et al. 2020	Melon	Y
Nicholson et al. 2020	Melon (watermelon)	Y
Phillips and Gardiner 2015	Pumpkin	Y
Rundlof et al. 2018	Red clover	Y
Sanchez et al. 2020	Spinach	N
Grab et al. 2019	Strawberry	N
Herbertsson 2018	Strawberry	Y
Feltham et al. 2015	Strawberry	Y
Ganser et al. 2018	Strawberry	Y
Grab et al. 2018	Strawberry	Y
Sardinas and Kremen 2015	Sunflower	Y
Balzan 2017	Tomato	Y

Balzan et al. 2016	Tomato	Y
Morandin and Kremen 2013	Tomato (processing)	Y
Rosas-Ramos et al. 2019	Vineyards	N
Buhk et al. 2018	NR	N
Kremen and M'Gonigle, 2015	NR	N
Kremen et al. 2018	NR	N
Pywell et al. 2012	NR	N
Sardinas et al. 2016	NR	N
Scheper et al. 2015	NR	N
Ponsino et al. 2017	NR	N
Carvell et al. 2011	NR	N
Haaland and Gyllin 2010	NR	N
Kohler et al. 2008	NR	N
Pywell et al. 2005	NR	N
Pywell et al. 2006	NR	N
Carvell et al. 2015	NR	N
Kleijn et al. 2018	NR	N
Holland et al. 2015	NR	N
Williams et al. 2015	NR	N
Campbell et al. 2019	NR	N
Jonsson et al. 2015	NR	N
M'Gonigle et al. 2015	NR	N