Biological Conservation 314 (2026) 111682

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Evaluating the ecological effectiveness of landscape-level collaborative conservation: A case study targeting wild bees

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ARTICLE INFO

Keywords:
Wild bees
Multi-actor
Collaborative conservation
Pollinators
Landscape ecology

ABSTRACT

Collaborative multi-actor conservation has been heralded as an effective way to address historical biodiversity loss because it makes landscape-level multi-habitat management strategies possible. However, its ecological effectiveness is not well understood. We examine a multi-actor approach in which 11 organisations collaborated to enhance wild bees in a 30 km^2 landscape in the south of the Netherlands. Using a novel study design for landscape-level conservation initiatives, we compared six-year trends in wild bee abundance and species richness and flower cover and species richness in 47 sites with bee-friendly management aimed at increasing the spatio-temporal availability of flowers in five habitats (extensive pastures, road verges, field margins, hedgerows, water retention sites) with trends in similar numbers of conventionally managed controls inside and outside the landscape. Overall, wild bee abundance and species richness increased in sites with bee-friendly management relative to controls, though effectiveness varied by habitat. Across all sites, bee-friendly management resulted in significantly more positive trends in flower cover than in control sites, yet trends in managed sites were stable rather than increasing and flower cover declined by approximately 46 % in control sites. The implementation success of bee-friendly management varied by habitat type and year, and was a key factor underlying the ecological effectiveness of said management. Our results suggest that coordinated collaborative approaches across complementary habitats can produce conservation benefits, but that success depends on effective communication with and consistent participation of actors, guidance by a coordinator, and continuous monitoring of management implementation and ecological outcomes.

1. Introduction

Despite valiant efforts to bend the curve of biodiversity decline, historical biodiversity losses due to land use change are expected to continue into the 21st century (Pereira et al., 2024). Climate change in conjunction with land use change is expected to intensify these declines. Conservation strategies that have been implemented thus far, such as protected areas and agri-environment schemes, may have halted biodiversity decline locally but have generally failed to stop regional or global biodiversity loss (Butchart et al., 2010; Kleijn et al., 2011). As conservation instruments, protected areas and agri-environment schemes represent sectorial approaches which have been criticized by conservation scientists for failing to integrate conservation actions in different parts of the landscape (e.g. in protected areas and on agricultural land; Smart et al., 2014) or lacking engagement with local communities resulting in low support for conservation actions (Kleijn et al., 2020; Reed et al., 2016). Collaborative multi-actor approaches may address these challenges and have been heralded as a more effective conservation solution, especially in the European Union (Hermoso et al., 2022).

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https://doi.org/10.1016/j.biocon.2025.111682
Received 25 July 2025; Received in revised form 16 December 2025; Accepted 19 December 2025
Available online 26 December 2025

In collaborative multi-actor approaches, different stakeholders work together towards shared objectives, pooling resources, knowledge, and expertise to achieve significant conservation outcomes. It has been argued that this will make them highly effective in reaching conservation objectives as it stimulates buy-in and allows actors to feel a sense of ownership over the conservation initiative (Reed et al., 2016). Actors can furthermore participate in the design process which improves the likelihood that the implementation of management interventions is feasible. Finally, participation in a collaborative multi-actor conservation initiative can foster a sense of community and enable actors to share knowledge and advice with each other (Prager, 2015). These features are ultimately expected to contribute to better biodiversity outcomes than traditional conservation approaches such as agri-environment schemes that subsidise individual farmers to undertake prescribed conservation actions on their land and that often show mixed biodiversity outcomes (Hasler et al., 2022; Redhead et al., 2022).

Ecological theory predicts that focusing conservation on landscapes can further improve the ecological effectiveness of conservation interventions compared to more narrow conservation approaches, such as conservation of a single habitat type or a protected area on public land (Wiens, 2009). Landscape-level conservation would allow for the implementation of coordinated and synergistic management actions on a mix of private and public land (i.e. farmland, protected areas, public space) all within the movement range of the species group of interest. In Europe, protected areas are generally small in size, with more than 60 % being less than 1 km^2 (Protected Areas in Europe – An Overview, 2012) making them highly susceptible to adverse effects of habitat fragmentation and pressures from intensively used surrounding landscapes (Laven et al., 2005). Landscape-level conservation that uses a collaborative multi-actor approach (hereafter landscape-level collaborative conservation) can facilitate conservation actions on farmland to reduce the pressures on protected areas and facilitate dispersal between them by using linear landscape elements such as field margins, road verges, and hedgerows as habitat corridors and stepping stones (Krewenka et al., 2011; Maurer et al., 2022). Furthermore, such an approach makes it possible to implement conservation actions in a larger proportion of the landscape thus making it more feasible to achieve the minimum cover of conservation management required to halt population declines (Vickery and Tayleur, 2018). Additionally landscape-level collaborative conservation could help implement conservation management in a wider range of habitats that focal species groups may use during their life cycle. At the landscape level, conservation management implemented in different sites in different habitats could add up to stronger ecological effects than the sum of their parts, for example because gaps in resource availability in one site can be overcome by dispersal to a nearby conservation site thus allowing for build-up of larger population sizes (Schellhorn et al., 2015).

Though popular in theory, there are few documented examples of the implementation of landscape-level collaborative conservation approaches in practice (Reed et al., 2016) and there is little or no evidence that it actually has positive effects on biodiversity (Koontz et al., 2020). Collaborative approaches are difficult to study, which may contribute to the lack of evidence on their ecological effectiveness. How, when, and where a management intervention is implemented can vary considerably between actors because they must balance scientifically optimal implementation with other socioeconomic priorities (Klebl et al., 2024). For example, farm layouts and willingness of farmers to implement conservation management can make it difficult to target the placement of management interventions in such a way that they have spatially independent effects. Additionally, the ultimate goal of a landscape-level collaborative conservation initiative is to enhance biodiversity at the level of the landscape. Given the time and cost involved in landscape-level collaborative conservation, there is typically only a single landscape involved meaning that independent replication of study units is difficult (Kleijn et al., 2020). Both previous points make the use of common space-for-time study designs more challenging (Christie et al., 2019; Westgate et al., 2013). Nevertheless, given increasing interest of conservation networks and policy makers in landscape-level collaborative approaches, it is critical to determine if or when these approaches are effective at promoting biodiversity.

In this study, we evaluate the ecological effectiveness of a landscape-level collaborative conservation initiative targeting wild bees using the study design presented by Kleijn et al. (2020). We compare trends in wild bee abundance and species richness, as well as flower cover and species richness, over a period of six years in sites with bee-friendly management and conventionally managed control sites within the working area of the landscape. We additionally include control sites outside the landscape that are therefore not influenced by potential spillover effects from nearby sites with conservation management. We apply this design to the conservation initiative, the "Boshommellandschap Geuldal" (i.e. Shrill carder bee landscape) in the Netherlands, that targets wild bees that provide key pollination services to both wild plant and crop species (Potts et al., 2016), but whose populations are in serious decline due to land use change, agricultural intensification, and pesticide use (Dicks et al., 2021; Rader et al., 2014). Despite the current wealth of knowledge on conservation interventions specifically designed to promote wild bees (Duque-Trujillo et al., 2023), many wild bee species are still in decline. For the conservation of mobile taxa, such as wild bees, landscape-level collaborative conservation approaches may be particularly important (Jauker et al., 2009), given that they often require different habitats for food and nesting resources (Westrich, 1996). This is exemplified by the fact that bee diversity in agricultural fields is often related to distance from source habitats such as semi-natural grasslands (Jauker et al., 2009; Klaus et al., 2021). To achieve positive trends, a more comprehensive and cohesive conservation framework may therefore be necessary, where both protected nature reserves and the landscape surrounding it are conserved in cooperation with local actors (Stout and Dicks, 2022).

In the Boshommellandschap initiative, 11 actors collaborated by implementing, monitoring, and evaluating codesigned bee-friendly management in five different habitat types: field margins and hedgerows on farmland, road verges and water retention sites on public land and extensive pastures in protected areas. Bee-friendly management differed between habitats but all measures aimed to promote flower availability and continuity and thereby wild bee abundance and diversity, for example, through staggered mowing or sowing wildflowers (see Table S2 for details). Using data from the first six years of this conservation initiative, we sought to answer the following questions: (i) do the five targeted habitat types support different species thus complementing each other at the landscape level, (ii) does bee-friendly management realised within the context of a landscape-level collaborative conservation approach result in more positive flower cover and richness trends and bee abundance and richness trends than conventional management, overall as well as per habitat type, and (iii) what are the key benefits and drawbacks of the landscape-level collaborative conservation approach as it is implemented here? We first compared floral resource and bee population trends in sites with bee-friendly management and the two types of controls (controls within the landscape and controls outside the landscape) across the five habitat types and then more specifically between different habitat types. We expected that, though the specific management interventions would show variable effectiveness between habitat types, across all habitat types bee-friendly management would add up to more positive trends in wild bee abundances and species richness compared to conventional management.

2. Methods

2.1. Study design

The data for this study was collected in the valley of the river Geul, (Zuid Limburg, the Netherlands) where the Boshommellandschap is located (Fig. 1). This area is characterised by a hilly landscape on limestone soils, which also support protected species-rich calcareous grasslands. Intensive arable farming, orchards, and dairy farming dominate this region's agricultural landscape. The Boshommellandschap is a landscape-level collaborative conservation initiative covering an area of approximately 3000 ha that began in 2018, with the aim of improving existing semi-natural habitat for wild bees (see www.boshommellandschap-geuldal.nl for details). In the conservation initiative 11 actors collaborate. The nature conservation organisations Staatsbosbeheer, Natuurmonumenten and Limburgs Landschap, Waterboard Limburg, Drinking Water Company Limburg, the municipalities Valkenburg aan de Geul and Gulpen-Wittem, the farmer collective Natuurrijk Limburg, the public benefit organisation Stichting Limburg Bloeit Op, the province Limburg, and Wageningen University & Research work together with the aim to restore bee populations (see Kleijn et al. (2020) for a more detailed overview). The term “actor” is often used interchangeably with “stakeholder”, however for the sake of consistency hereafter we use “actor” to refer to individuals, groups, or organisations with the capacity to influence the conservation effort (i.e., act) (Avelino and Wittmayer, 2016).

In 2020, bee-friendly management was introduced in 47 sites in the five different habitat types (pastures, field margins, road verges, water retention sites, and hedgerows) (Fig. 1, Table S1). Bee-friendly management generally aimed to enhance the availability and continuity of floral resources in space and time: staggered or rotational mowing/grazing in pastures, sown wildflower mixtures in field margins, adjusted mowing schedules and removal of clippings in road verges, delayed mowing, grazing in water retention sites, and reduced pruning frequencies of hedgerows. Initial suggestions for effective bee-friendly management were proposed for each habitat type by coauthors D.Kl. and I.R. These management options were discussed with the actors responsible for implementing and maintaining them, which typically resulted in modifications to make them easier to implement while still being ecologically effective (see Table S2 for implemented management per habitat type). Actors subsequently contracted out the majority of management implementation to third parties. During monitoring of flowers and bees (see below) it was noted whether habitat conditions were in line with the agreed bee-friendly management or deviated from it (e.g. mowed entirely instead of staggered). Failed management was discussed bilaterally with the responsible actor with the aim to improve implementation success in following years. Additional to the sites with bee-friendly management, conventionally managed control sites were selected for each habitat type within the Boshommellandschap area ($N = 56$). Control sites within the landscape would have a similar landscape context and experience similar environmental conditions to sites with bee-friendly management and differed only in terms of management (Kleijn et al., 2020). However, control sites within the landscape were often located in close proximity to sites with bee-friendly management (mean distance between control sites and nearest site with bee-friendly management was 394.4 ± 211.4 m). This could result in spillover from sites with bee-friendly management (Blitzer et al., 2012; Kleijn et al., 2020). We therefore also selected control sites for each habitat type outside the landscape and outside the foraging range of wild bees, but still close enough that they could be expected to experience similar environmental conditions to sites within the landscape ($N = 48$). In all sites, effect monitoring started in 2018, two years prior to the start of the bee-friendly management, resulting in a before-after-control-impact design which is generally considered a highly effective way to analyse the effects of conservation interventions (Bro et al., 2004).

2.2. Wild bee sampling

Wild bees were collected in three sampling rounds per year (May, June, and July), for a total of 18 sampling periods between 2018 and 2023. The sampling protocol followed Scheper et al. (2015). Transects of 150 × 1 m were sampled by net in 50 × 1 m increments for 5 min each, totaling 15 min of pure sampling time. Sampling only occurred during good weather conditions: temperature at or exceeding 15 °C, no rain,

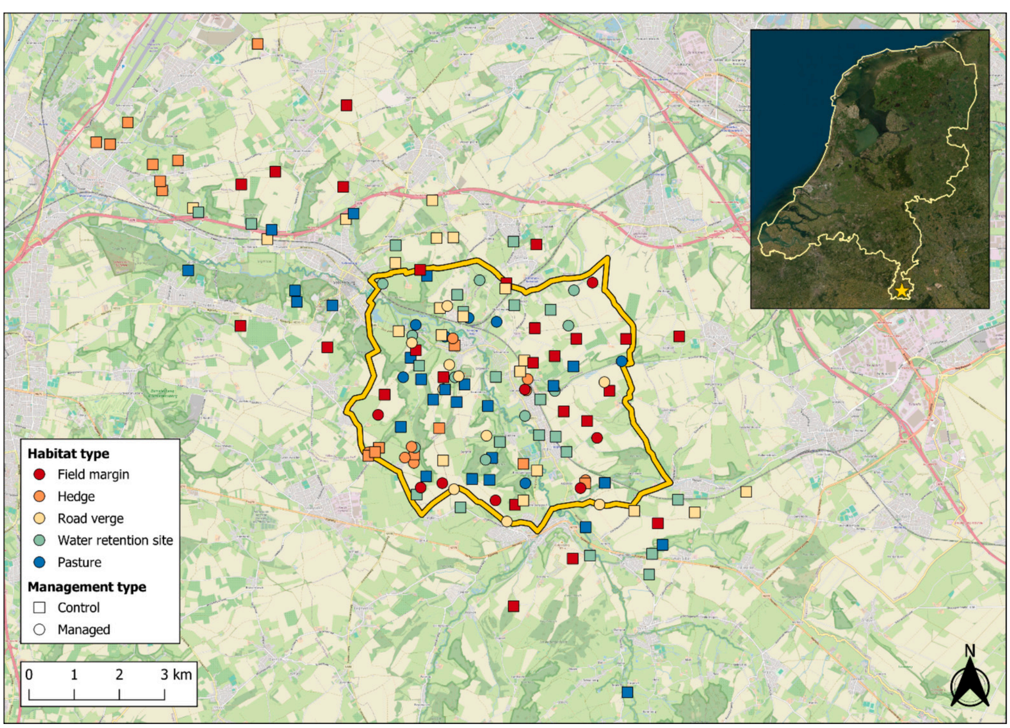


Fig. 1. All transect locations of the collaborative, landscape-level bee conservation initiative Boshommellandschap Geuldal (landscape delineated by yellow border). Transect colours denote their habitat type and shapes denote their management type. Control transects outside the yellow border are considered “outside landscape” and transects inside are considered “inside landscape”). The location of the Boshommellandschap in the Netherlands is indicated in the inset with a yellow star. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

dry vegetation, and wind levels less than Beaufort 5. Four species (*Bombus terrestris, B. lucorum, B. magnus,* and *B. cryptarum*) were counted as an aggregate, as the workers cannot be separated without using molecular methods (Murray et al., 2008). Individuals that could not be identified to the species level in the field were collected for identification in the lab using Falk (2020). A permit was not required for this.

2.3. Floral resource survey

Flower diversity and cover were estimated for each transect by counting the number of flower units (individual flower, flower head or umbel depending on species) per species. Floral surveys were done within three days of wild bee surveys on the same transect. Flower area was calculated per species by multiplying the number of floral units by average floral unit area. Transect-level flower cover was the sum of species-specific flower area divided by the total area of the transect (Scheper et al., 2015). Per-species average floral unit areas were taken from a database maintained by the Plant Ecology and Nature Conservation group (Wageningen University and Research). Flower and bee surveys were done by IR (2018–2023), RvK (2018–2023), DKl (2018–2022), JD (2018–2019), TdK (2018), MFvdS (2019), HvK (2020), MdB (2020), JannekeS (2021), DKi (2021), WO (2022), KLO (2023), FR (2023), and AH (2023).

2.4. Data analysis

Solitary bees and bumblebees were analysed together. Sampling date was converted to days since the beginning of the conservation initiative (January 1st, 2018), hereon referred to as Days Since Start. Management type was a factor with three levels: control sites within the landscape, control sites outside the landscape, and sites with bee-friendly management. Wild bee responses (abundance and species richness) were measured per transect per 150 m^2 per 15 min.

Using data from all six years, we first explored the relative importance of each habitat type to wild bee species richness, irrespective of type of management, using a generalised linear mixed-effects model, with species richness as the response, habitat type as a fixed factor, transect ID as a random effect, and using a negative binomial distribution. Pairwise comparisons between habitat type were done using package *emmeans* (Lenth, 2024) and multiple comparisons were corrected for using the Tukey method. Using data from just the four years with bee-friendly management we then examined whether specific combinations of habitat and management type supported different bee species by estimating the number of shared species by habitat and management type. This was estimated by taking a random subset of transects per habitat and management type ($N = 96$) within the Boshommellandschap, then calculating the number of species unique to that habitat, shared with one other habitat, two others, three others, and all other habitats. Ninety-six transects per habitat type was the minimum number of transects sampled from all combinations of habitat and management type within the Boshommellandschap between 2020 and 2023.

We then assessed both the overall and habitat type-specific effectiveness of bee-friendly management for flower cover and species richness and wild bee abundance and species richness using generalised linear mixed-effects models and data for all six years. Wild bee abundance and species richness were modelled with negative binomial distributions. Flower species richness was modelled with a Poisson distribution. Flower cover was modelled with a zero-inflated Gamma model with a log link, with the zero-inflation parameter applied to all observations. Days Since Start was standardised by centering and dividing by two standard deviations to aid with model convergence. All models included Management type and Days Since Start as interacting fixed factors. We assumed a linear effect of Days Since Start, based on the observed relationship between all response variables and Days Since Start. A second set of models, which additionally included the variable Habitat Type, were run with the main effects of Habitat Type, Management type, and Days Since Start, as well as all (up to three-way) interactions between Habitat Type, Management type, and Days Since Start. A significant three-way interaction would indicate that the differences between management types in the bee or flower trends were not the same for the five habitat types, and therefore potentially that the effectiveness of bee-friendly management differed between habitat types. Transect ID was included as a random effect in all models to account for repeated measurements. The significance of interactions was assessed using likelihood-ratio tests. After which, non-significant interactions, and subsequently non-significant main effects, were dropped from the models. Temporal and spatial autocorrelation were tested for (based on Durbin-Watson and Moran's I tests, respectively) to account for potential non-independence in the study design. Temporal autocorrelation was detected for all flower cover and flower species richness models. We included Ornstein-Uhlenbeck covariance structures, which can handle irregular time points, in each model to correct for this. We defined the time variable as the sampling date and group as a single dummy variable as there was only one time series. Spatial autocorrelation was detected only for the Days Since Start * Management type flower species richness model and was corrected for by including longitude and latitude as fixed factors, both standardised by centering and dividing by two standard deviations. Post-hoc testing to determine the pairwise differences in levels of Management type, plus Management and Habitat Type, as a function of Days Since Start, were done using package *emmeans* (Lenth, 2024). Multiple comparisons were corrected for using the Tukey method.

All statistical analyses were done in R version 4.4.1 (R Core Team, 2024). Packages used can be found in the Supplementary Information.

3. Results

Over six years, 21,679 wild bees from 196 species were sampled. The most common species by frequency of occurrence over all years were *Bombus lapidarius* (20.8 %), *Bombus pascuorum* (13.8 %), *Bombus terrestris/lucorum* (9.1 %), and *Lasioglossum pauxillum* (8.8 %). A total of 329 flower species were observed, of which the most common by the frequency of occurrence over all transects were *Trifolium repens* (60.4 %), *Ranunculus repens* (58.3 %), and *Cerastium fontanum* (58.2 %). See Tables S12–S13 for an overview of wild bee and flower species.

3.1. Bee communities of different habitat types

The habitat types (pastures, field margins, road verges, water retention sites, and hedgerows) differed significantly in the number of bee species they supported. Wild bee species richness in hedgerows was significantly lower than in road verges (ratio = 0.61, SE = 0.09, z(inf) = −3.34, $p = 0.007$), pastures (ratio = 0.48, SE = 0.07, z(inf) = −5.05, $p < 0.0001$), and water retention sites (ratio = 0.40, SE = 0.06, z(inf) = −6.43, $p < 0.0001$) (Fig. 2a). Field margins had significantly lower bee species richness than pastures (ratio = 0.68, SE = 0.09, z(inf) = −2.91, $p = 0.030$) and water retention sites (ratio = 0.56, SE = 0.073, z(inf) = −4.43, $p = 0.0001$) (Fig. 2a). Road verges had lower bee richness than water retention sites (ratio = 0.65, SE = 0.09, z(inf) = −3.26, $p = 0.010$). Overall, 28 wild bee species were shared between all habitat types and types of management across the four years of study (Fig. 2b). Field margins with bee-friendly management had the lowest number of unique species ($n = 1$), while water retention sites with bee-friendly management had the highest ($n = 20$). Managed hedgerows shared the fewest total species with other habitat types ($n = 71$), and control water retention sites shared the most ($n = 95$).

3.2. Effects of management on trends in flower cover and wild bee abundance

Here we present the results only for flower cover and wild bee

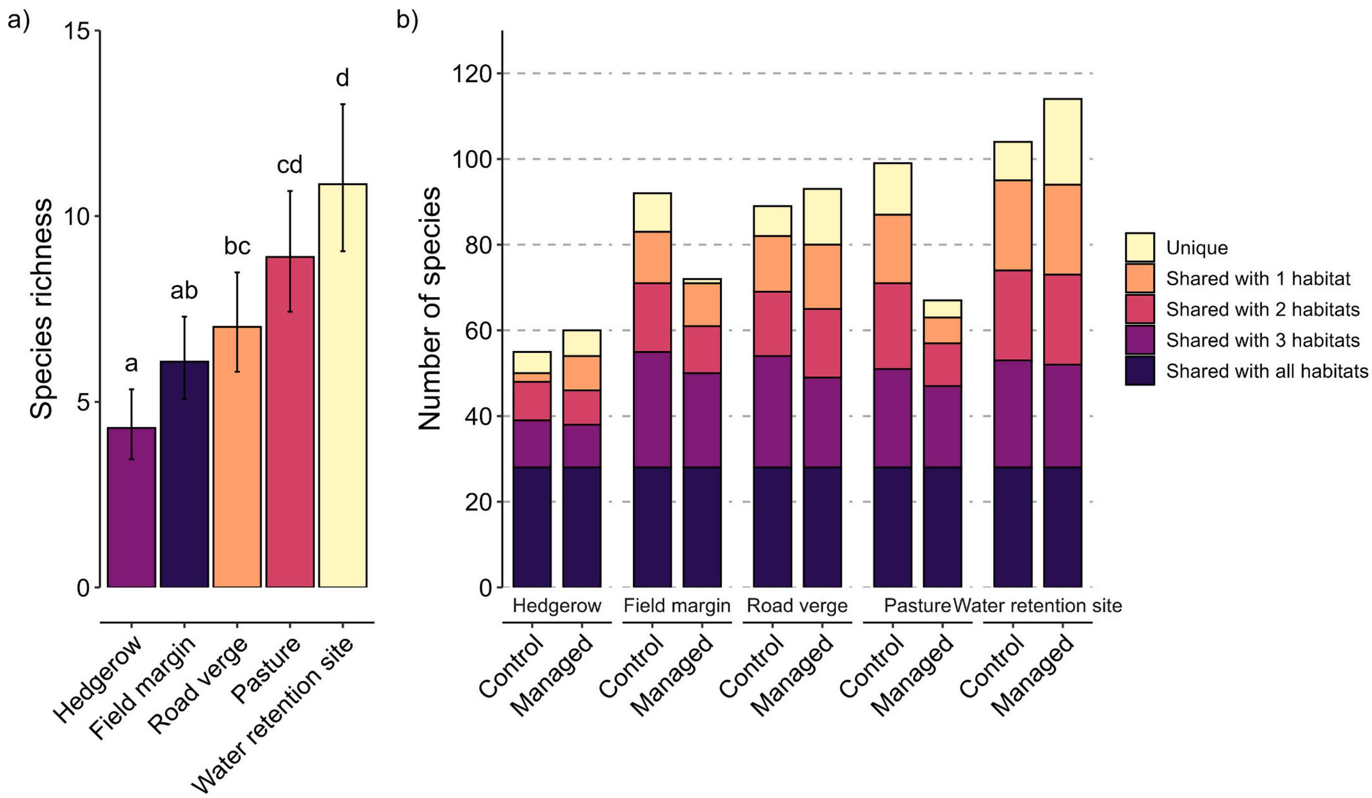


Fig. 2. a) Differences in mean species richness between habitat types across all six years of the Boshommellandschap initiative, irrespective of management. Letters indicate significant ($p < 0.05$) pairwise comparisons of the estimated marginal means of wild bee species richness by habitat type, using a z test. *P*-values are adjusted for multiple testing using the Tukey method. Error bars are the asymptotic lower and upper 95 % confidence levels. Values are on the response scale. b) The number of wild bee species for each habitat that were unique to the habitat, shared with one other habitat, two other habitats, three other habitats, or were shared with all habitats over the four-year period that bee-friendly management was introduced (2020−2023). Control category only contains control sites within the landscape.

abundance. Flower species richness followed similar patterns as flower cover and wild bee species richness trends were comparable to those of wild bee abundance and can be found in Supplementary Figs. S1 − S4 and Tables S4 − S11.

The trend in flower cover in sites with bee-friendly management was more positive than those in control sites within or outside the landscape ($\beta = 0.002$, SE = 0.0005, z(inf) = 4.03, $p < 0.001$; $\beta = 0.001$, SE = 0.0005, z(inf) = 2.19, $p = 0.072$, respectively) (Fig. 3: upper panel). We found a significant interaction between Management type and Days Since Start ($\chi^2(2) = 9.29$, $p = 0.01$). Flower cover in control sites within the landscape (slope = −0.002, SE = 0.0003, z(inf) = −6.56, $p < 0.001$) and outside the landscape (slope = −0.001, SE = 0.0003, z(inf) = −2.30, $p = 0.003$) declined significantly over time. The trend in flower cover in sites with bee-friendly management was slightly positive but not significantly different from zero (slope = 0.0003, SE = 0.0004, z(inf) = 0.60, $p = 0.55$) (Fig. 3: lower panel). Model predictions indicate that floral cover increased, on average, by 9.0 % in sites with bee-friendly management, and declined by 57.5 % in controls within the landscape and 34.8 % in controls outside the landscape.

The effect of bee-friendly management on flower cover differed between habitat types (significant three-way interaction between Management type, habitat type, and Days Since Start ($\chi^2(12) = 67.51$, $p < 0.001$)). When looking at the habitat types separately, we found that field margins with bee-friendly management showed a more positive trend in flower cover compared to both control sites within and outside the landscape ($\beta = 0.004$, SE = 0.001, z(inf) = 4.36, p < 0.001; $\beta = 0.005$, SE = 0.001, z(inf) = 4.35, p < 0.001, respectively) (Fig. 4: upper panel). Trends in flower cover did not differ significantly between sites with or without bee-friendly management in any of the other habitats. There was a significant positive trend over time in flower cover in field margins (slope = 0.003, SE = 0.001, z(inf) = 3.84, $p = 0.001$) and a significant negative trend in hedgerows (slope = −0.001, SE = 0.0003, z(inf) = −3.63, $p < 0.001$) with bee-friendly management (Fig. 4: lower panel). There were significant negative trends in field margins (slope = −0.001, SE = 0.0004, z(inf) = −2.85, $p = 0.004$; slope = −0.001, SE = 0.0005, z(inf) = −2.60, $p = 0.009$), hedgerows (slope = −0.001, SE = 0.0002, z(inf) = −3.41, $p < 0.001$; slope = −0.001, SE = 0.0002, z(inf) = −3.40, $p < 0.001$), and road verges (slope = −0.002, SE = 0.001, z(inf) = −3.53, $p < 0.001$; slope = −0.002, SE = 0.001, z(inf) = −2.93, $p = 0.003$) in controls within and outside the landscape, respectively.

Across all habitat types, bee-friendly management resulted in more positive trends in bee abundance than conventional management within and outside the landscape (significant interaction between Management type and Days Since Start ($\chi^2(2) = 11.40$, $p = 0.003$)) ($\beta = 0.35$, SE = 0.12, z(inf) = 2.86, $p = 0.012$; $\beta = 0.39$, SE = 0.13, z(inf) = 3.03, $p = 0.007$, respectively) (Fig. 5: upper panel). We found a significant positive trend over time in wild bee abundance in sites with bee-friendly management (slope = 0.26, SE = 0.09, z(inf) = 2.85, $p = 0.004$), and non-significant negative trends in control sites within the landscape (slope = −0.08, SE = 0.08, z(inf) = −1.08, $p = 0.278$) or outside the landscape (slope = −0.13, SE = 0.09, z(inf) = −1.42, $p = 0.16$) (Fig. 5: lower panel).

The effect of bee-friendly management on bee abundance differed between habitat types (significant three-way interaction between Management type, habitat type, and Days Since Start ($\chi^2(12) = 82.60$, $p < 0.001$)). Field margins with bee-friendly management had more positive trends in bee abundance compared to both control sites within and outside the landscape ($\beta = 1.38$, SE = 0.27, z(inf) = 5.06, p < 0.001; $\beta = 1.88$, SE = 0.29, z(inf) = 6.45, p < 0.001, respectively) (Fig. 6: upper panel). In road verges, controls within the landscape had a more negative trend compared to controls outside the landscape ($\beta = -0.62$, SE = 0.26, z(inf) = −2.36, $p = 0.048$). We found significant positive trends

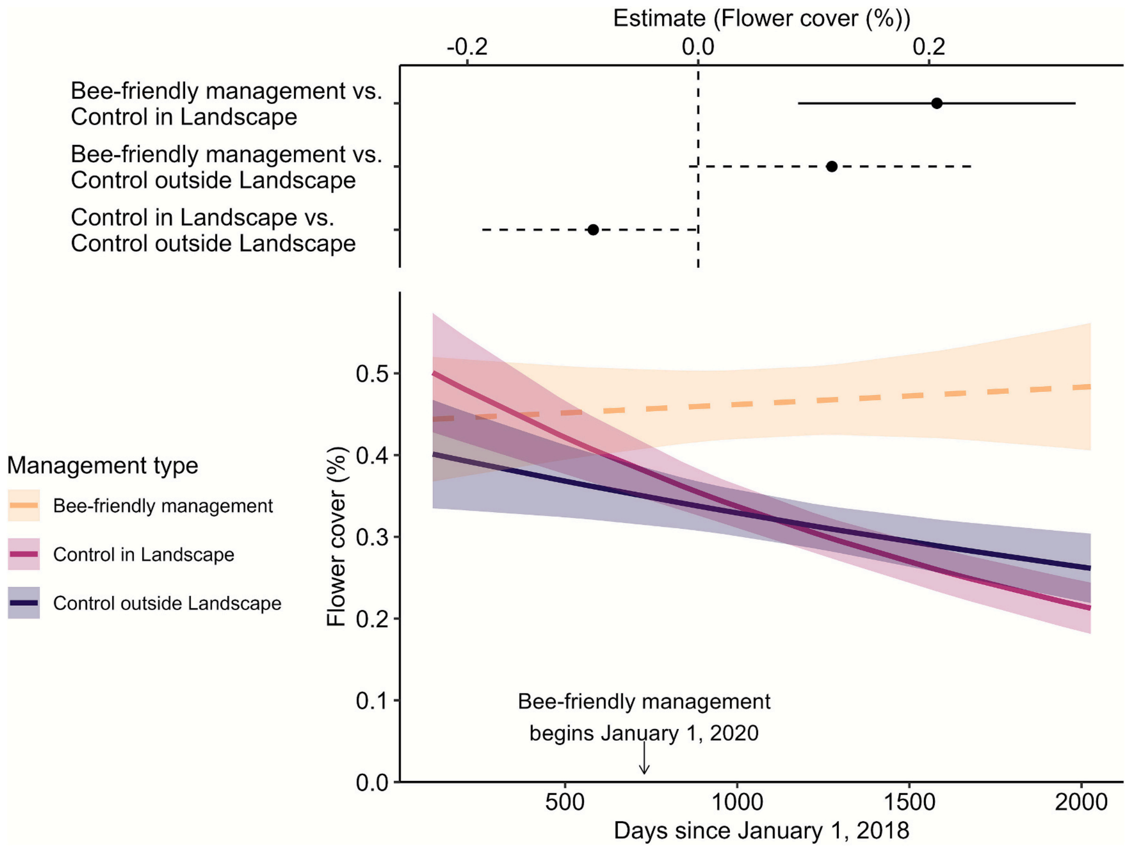


Fig. 3. Upper panel: Pairwise comparisons of the rate of change in flower cover (%) by management type. Points represent the difference in slope and lines are the 95 % confidence interval. A solid line indicates a significant difference in trends between management types ($p < 0.05$). Lower panel: Estimated marginal means of the linear trend in wild bee abundance, as a function of time (Days Since Start) and management type. A solid line indicates a trend that differs significantly from zero ($p < 0.05$).

over time in wild bee abundance in field margins with bee-friendly management (slope = 1.56, SE = 0.22, z(inf) = 7.26, $p < 0.001$) and control hedgerows within the landscape (slope = 0.67, SE = 0.21, z(inf) = 3.13, $p = 0.002$) (Fig. 6: lower panel). There were significant negative bee abundance trends in conventionally managed control pastures (slope = −0.37, SE = 0.15, z(inf) = −2.52, $p = 0.012$) and road verges (slope = −0.61, SE = 0.19, z(inf) = −3.11, $p = 0.002$) within the landscape.

3.3. *Implementation success of bee-friendly management*

In the first year of bee-friendly management, implementation success was initially poor, with successful management in only 16 of 47 sites (33 %). However, implementation improved over time in most habitat types, with successful management occurring in 34 of 47 sites (72 %) in the final year of the study (Table 1). In 2020 wildflower strips were not successfully established along eight out of nine field margins and because of farmer preferences or difficulties with implementing management, wildflower strip locations changed in subsequent years, sometimes more than once. Bee-friendly management was not successfully implemented in any of the pastures in 2020 because tenant agreements were not organised in time. However, management improved steadily in subsequent years for both habitat types and by 2023, management failed in only one pasture and one field margin. Management failed in half of the road verges in 2020 and 2021 due to the contractor of one municipality ignoring management specifications. However, all issues with mowing were resolved in 2022 and 2023 when a new contractor was hired. All water retention sites were managed successfully in 2020 and 2021, however in 2022 and 2023, management failed in two and three sites, respectively, because of miscommunication with the farmers renting the land. Bee-friendly management in all eight hedgerows failed completely between 2020 and 2023, primarily due to farmers not implementing the voluntary bee-friendly measure to not cut their hedgerows annually that had been agreed with the farmer collective.

4. Discussion

Given the rising popularity of landscape-level collaborative conservation approaches, our study contributes to an important knowledge gap: the lack of evidence supporting their ecological effectiveness. To our knowledge, this study is the first to test whether landscape-level collaborative conservation produces biodiversity benefits. A clear benefit that was identified is that our studied initiative enabled the implementation of conservation management on 47 sites, covering an area of 56.96 ha or almost 2 % of the landscape, which would otherwise not have been possible in this area. In these sites with conservation management significant overall increases in wild bee abundances and species richness were observed. Moreover, these trends were significantly more positive than those observed at conventionally managed sites (Fig. 5; Fig. S2). However, the ecological effectiveness of bee-friendly management varied considerably between the five habitat types, with sowing wildflowers along field margins showing the most pronounced positive effects and hedgerow management being the least effective. Ecological effectiveness was influenced by the extent to which bee-friendly management was successfully implemented. In general, the contrasting trends between sites with bee-friendly management and control sites appeared to be most strongly related to the increase over time in the proportion of sites with successful management (Table 1, Fig. 6). Variation in management success and its importance for

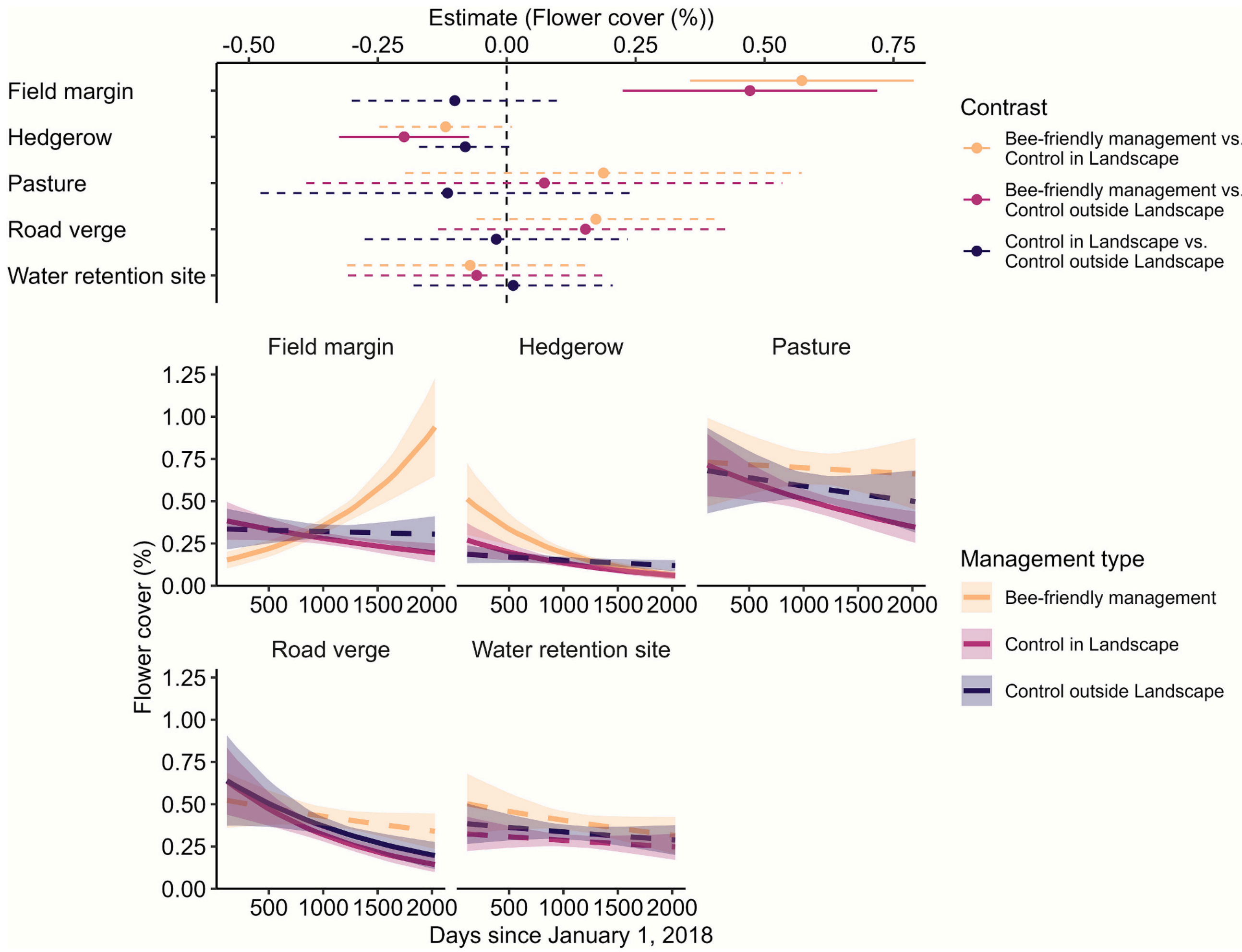


Fig. 4. Upper panel: Pairwise comparisons of the rate of change in flower cover (%) by management type and habitat type. Points represent the difference in slope and lines are the 95 % confidence interval. A solid line indicates a significant difference in trends between management types ($p < 0.05$). Lower panel: Estimated marginal means of the linear trend in wild bee abundance, as a function of time (Days Since Start) and management type. Results are faceted by habitat type. A solid line indicates a trend that differs significantly from zero ($p < 0.05$).

ecological outcomes emphasises that landscape-level collaborative approaches require ongoing monitoring and open dialogue with actors to promote the optimal biodiversity conservation results.

The variable effectiveness of bee-friendly management in the five habitat types was partly linked to the initial ability of the different actors to modify management in their sites. In the Boshommellandschap initiative, actors did not receive compensation for implementing interventions in pastures, hedgerows, road verges, or water retention sites. Consequently, measures such as staggered, rotational, or delayed mowing and grazing were chosen for their expected positive ecological effects (Buri et al., 2014), while remaining acceptable to actors—which is often the outcome of co-designed conservation actions. Field margins were the exception. Farmers received financial compensation through agri-environmental schemes for loss of income associated with establishing wildflower strips along field margins. Because conventional field margins were generally flower-poor (Fig. 4) and, when implemented successfully, sowing wildflowers introduced a vast amount of additional floral resources, this type of bee-friendly management likely created a large ecological contrast that led to positive trends significantly different from the controls (Scheper et al., 2013). As flower availability was initially high in pastures, road verges, and water retention sites, new measures did not introduce new floral resources but rather aimed to increase the continuity of already available floral resources. This resulted in a smaller ecological contrast for wild bees, which may explain why habitat-specific wild bee trends in sites with bee-friendly management were not significantly different from trends in control sites. However, floral resource continuity is still important for preventing "gaps" in food supplies over the season (Timberlake et al., 2019). Measures introduced in the Boshommellandschap, such as delayed or staggered mowing, can improve floral resource continuity (Pywell et al., 2011) and may particularly alleviate resource limitations for late-flying species (Bishop et al., 2024). On average bee-friendly management resulted in stable flower cover over time rather than the expected increase, suggesting that bee-friendly management is only sufficient to mitigate negative trends in flower cover within and outside the landscape. Across all control sites, flower cover declined by approximately 46.2 % in the six-year study period (Fig. 3). This may be due to warmer winters and the increasing number of extreme weather events. In contrast to most forbs, grasses continue growing during warm winters, especially in areas with high nitrogen deposition rates such as the Netherlands (de Vries et al., 2021), which may ultimately lead to grasses outcompeting the forbs that produce the flowers that bee forage on (Bakker et al., 2024; Kreyling et al., 2019). Extremely dry summers result in fewer flowers, as desiccated plants do not flower (Phillips et al., 2018), and extremely wet summers can promote the dominance of grasses (Morecroft et al., 2004) which, similar to warm winters, result in forbs being outcompeted. All three processes seem to constrain the persistence of forbs and keep them from producing the flowers that wild

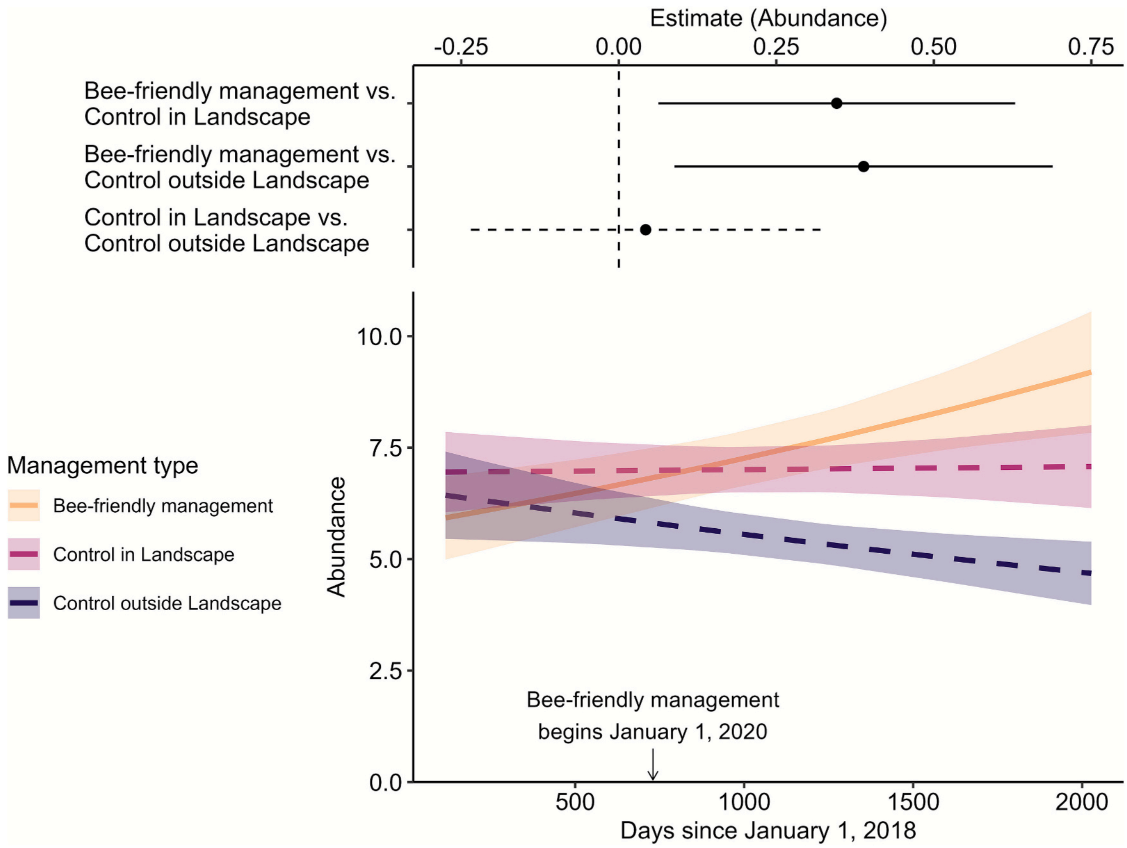


Fig. 5. Upper panel: Pairwise comparisons of the rate of change in wild bee abundance by management type. Points represent the difference in slope and lines are the 95 % confidence interval. A solid line indicates a significant difference in trends between management types ($p < 0.05$). Lower panel: Estimated marginal means of the linear trend in wild bee abundance, as a function of time (Days Since Start) and management type. A solid line indicates a trend that differs significantly from zero ($p < 0.05$).

bees rely on (Phillips et al., 2018). That we saw stronger effects for wild bees, with a clear positive trend across all habitats in sites with bee-friendly management, may be because if floral resources become more limiting, the relationship between the floral resources and wild bees becomes more pronounced (Bishop et al., 2024).

Significant increases in floral resources relative to controls were only found in field margins (flower cover; Fig. 4) and pastures (flower species richness; Fig. S2). Nevertheless, we do not think this precludes the benefits of bee-friendly management in water retention sites and road verges. Alongside pastures, road verges and water retention sites were inherently higher-quality habitats than field margins, as shown by their significantly higher average bee species richness regardless of management type (Fig. 2a). Trends in wild bee abundance and species richness were generally more positive in road verges and water retention sites with bee-friendly management than in conventionally managed sites, though these differences were small and non-significant. However, our study analysed the effects on bee densities and did not consider the area in which bee-friendly management has been implemented. Many non-significant but consistent differences in bee densities in water retention sites with bee-friendly management, which covered the second largest area (Table 1), could add up to considerable increases in the total number of bees when scaled up to the landscape level (Fijen et al., 2025). Additionally, the slopes of these artificially created depressions in the landscape offered ideal nesting conditions for a wide range of species. This includes large aggregations of *Halictus scabiosae*, *H. quadricinctus*, *Lasioglossum malachurum,* and *Andrena flavipes,* as well as their kleptoparasites. Such aggregations were seldom encountered in any of the other habitat types. Finally, many species were observed in other habitats but not in field margins (Fig. 2b). As these were mostly singletons and doubletons, it is difficult to say confidently that these species are bound to specific habitats, but it suggests that focusing on a single habitat type risks not meeting the habitat requirements of a subset of species. Different habitat types can provide diverse nesting and foraging resources both on a species-level (e.g., ground-nesting versus cavity nesting species (Antoine and Forrest, 2021)) and across the season through phenological complementarity of floral resources (Mandelik et al., 2012). The cumulative number of wild bee species observed over four years in field margins with bee-friendly management was considerably lower than in comparable control field margins (Fig. 2b), even though the average species richness was higher (Fig. S4). This indicates that the relatively uniform floral composition of managed field margins catered to a smaller number of bee species than the more sparsely occurring but more diverse community of flowers in control field margins. A similar pattern could be observed in the pastures where bee-friendly management tended to result in a more positive trend in the number of species per transect (Fig. S4), despite control pastures hosting 32 species more than pastures with bee-friendly management. This suggests that there is some level of complementarity between different habitats, supporting the usefulness of a landscape approach to wild bee conservation, though currently implemented bee-friendly management does not appear to fully capitalise on this. However, the relatively high number of singleton species in managed road verges and water retention sites shows potential for synergistic outcomes with appropriate management.

A key factor underlying the more positive trends in bees and flowers in sites with bee-friendly management was implementation success (Table 1). In the first year of implementation, bee-friendly management was implemented successfully in only one-third of the sites, which improved gradually to 79 % in the fourth year. In one habitat type, hedgerows, bee-friendly management failed completely as farmers did

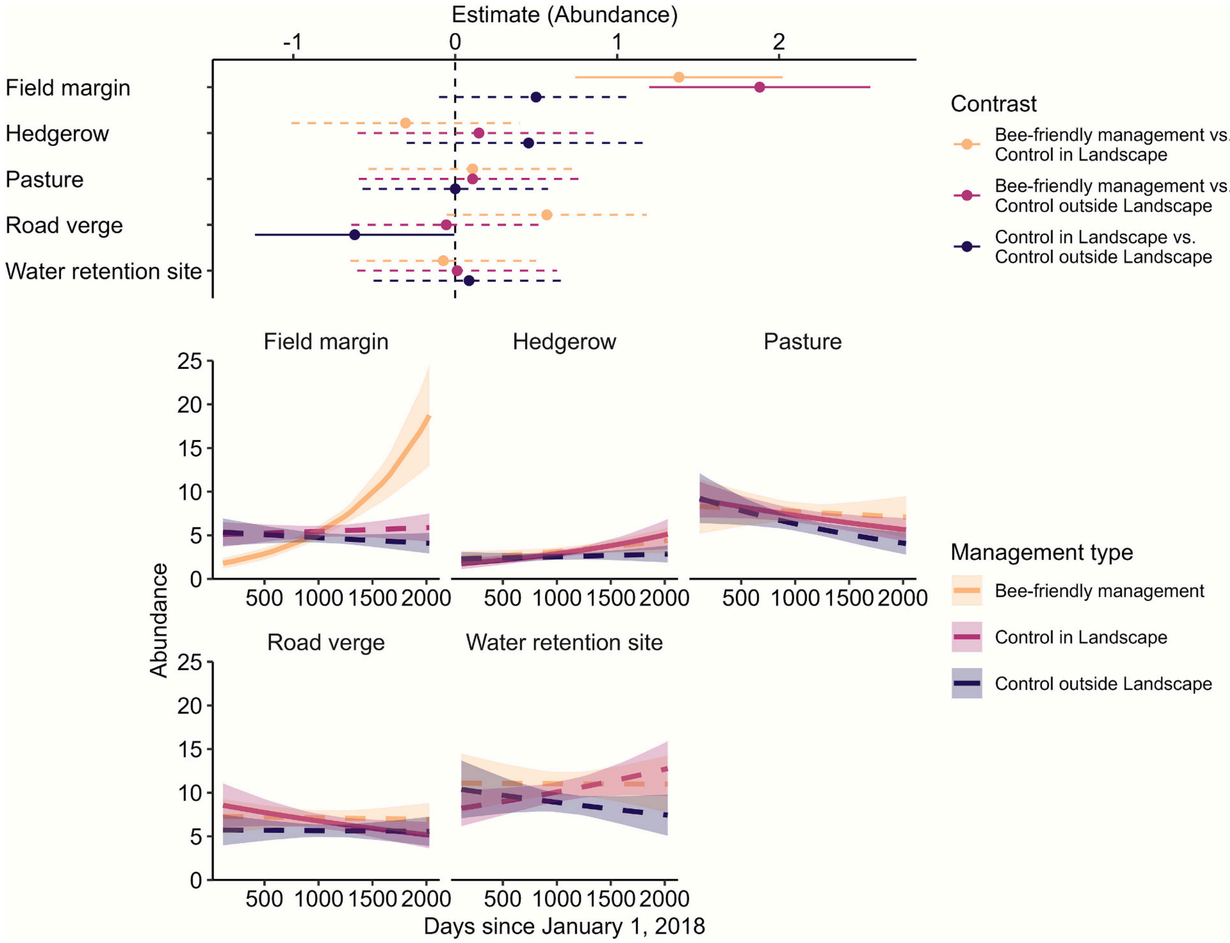


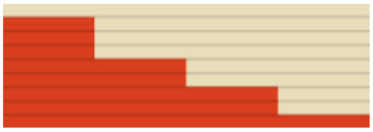
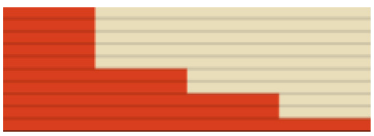
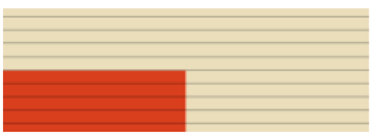
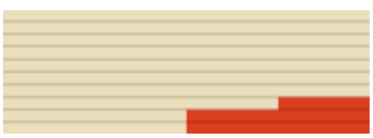
Fig. 6. Upper panel: Pairwise comparisons of the rate of change in wild bee abundance by management type and habitat type. Points represent the difference in slope and lines are the 95 % confidence interval. A solid line indicates a significant difference in trends between management types ($p < 0.05$). Lower panel: Estimated marginal means of the linear trend in wild bee abundance, as a function of time (Days Since Start) and management type. Results are faceted by habitat type. A solid line indicates a trend that differs significantly from zero ($p < 0.05$).

not allow their hedgerows to grow out, which was intended to increase floral resource availability. The most common reasons for unsuccessful implementation of bee-friendly management were (i) miscommunication between the actors responsible for managing the habitat and the tenant or contractor implementing the management, (ii) unwillingness of the contractor to modify conventional road management, and (iii) inclement weather (e.g., prolonged periods of rain; flooding) forcing land managers to modify their mowing schemes. Our study shows that ecological monitoring is pivotal to identify instances with failed management (see also Reid et al. (2007)) and by providing feedback to the responsible actors, can help improve the success rate of implementation. Further, monitoring has helped identify types of bee-friendly management that failed altogether and should be adjusted. Starting in 2024, attempts to introduce bee-friendly management in hedgerows have been discontinued. Monitoring the ecological outcome of the conservation initiative has inspired Waterboard Limburg to implement additional measures in water retention sites starting in 2024. Seeing that delayed or staggered grazing did little to enhance wild bees, but that the sloped, sparsely-vegetated sides of water retention sites were key nesting habitats, Waterboard Limburg created earth banks to increase the value of this habitat as nesting sites (Tsiolis et al., 2022). This is a good example of the adaptive management approach that the Boshommellandschap initiative set out to do and will continue to do going forward. Generally, actors may be more willing to implement costlier measures when there is evidence that the management they implement in the habitats they are responsible for is (not yet) effective in enhancing the target species.

5. Conclusion

The combined effects of bee-friendly management implemented by a range of actors in the Boshommellandschap conservation initiative added up to significant positive effects in bee abundance and richness trends, though per-habitat effects were typically small. By addressing a variety of habitat types managed by different actors (pastures, field margins, road verges, water retention sites, and hedgerows), it is likely that bee-friendly management included sites that are important for different parts of the life cycle of wild bees, such as the water retention sites for nesting. This approach also allowed for connecting the key habitats of target species (here, pastures and water retention sites) through improving the quality of linear landscape elements (road verges, field margins) in between them, which have been shown to facilitate pollinator movement through a landscape (Jauker et al., 2009). By improving the quality of both seminatural habitat and linear landscape elements, wild bee movement through the landscape may be enhanced, which could positively influence pollination (le Clech et al., 2024). However, our study highlights some key challenges of collaborative approaches. For individual actors, a collaborative conservation initiative generally represents a small proportion of their daily activities. It cannot be assumed that conservation management will be implemented in the agreed way from the outset. Further, there is often significant turnover in the people involved in collaborative conservation, such as elected government officials and third-party contractors, which can lead to a loss of institutional knowledge and familiarity with the

Table 1
Timeline of successful and failed management from 2020 (first year of implementation) to 2023. Red = failed site/year combinations, green = successful site/year combination. Number of sites and total area with bee-friendly management per habitat type given in parentheses.

	2020 2021 2022 2023	Causes for failure
Field margins (n = 9, 7.4 ha)		Agreed wildflower mixtures (Table S3) were not sown because of inclement weather; flowers sown, but wrong mixture used; flowers sown but grazing prevented them from flowering; flowers sown, but were not managed and were overgrown with grasses.
Hedgerows (*n* = 8, 0.4 ha)		Hedgerows were annually cut instead of being allowed to grow out for several years.
Pastures (*n* = 10, 139.6 ha)		Pasture completely mown or grazed due to miscommunication; flooding with polluted water requiring grass to be cut and dumped; prolonged inclement weather offering farmers few options to harvest the grass.
Road verges (n = 10, 1.3 ha)		Contractor ignoring agreed bee-friendly management – both verges cut simultaneously.
Water retention sites (n = 10, 9.5 ha)		Modified contracts with tenant not renewed on a timely basis; grazing occurred before agreed date.

project and result in poor management (Loffeld et al., 2022). With increasing number of actors involved in the implementation, the proportion of conservation management that fails to be implemented correctly will likely increase. Thus, it is important to monitor the ecological outcomes of landscape-level collaborative conservation approaches as this is often the only way to determine if management is being properly implemented. A second benefit of ecological monitoring is that it makes it possible to adapt future management to the results of past management and improve outcomes for "failed" management types. Money, time, and effort can be saved by dropping interventions that have been repeatedly shown to fail, either in terms of implementation or expected outcomes. Additionally, management can be adapted to respond to changing environmental conditions such as those most likely driving overall flower decline in our study area. These insights can be used to improve the effectiveness of bee-friendly management, for example by introducing practices that target grass suppression, or increasing grazing pressure or mowing frequency after a warm winter or conversely reducing grazing pressure and mowing frequency during droughts (Piseddu et al., 2021).

We show that a landscape-level collaborative conservation approach can be effective, however, six years of collaboration suggests the process is complex and rife with challenges surrounding communication, implementation, and long-term commitment to the project. The establishment of relationships and ongoing dialogues with and between the actors involved is central to conservation success in the long term (Reed et al., 2016; Richardson and Lefroy, 2016; Sayer et al., 2013). Implementation in the Boshommellandschap initiative improved substantially over time as trust was developed between actors and annual monitoring highlighted areas for improvement, allowing for adaptive management of the landscape (Reed et al., 2016). This eventually led to positive ecological outcomes. Consequently, we find that without ecological monitoring it is difficult to make claims about the effectiveness or the success of landscape-level collaborative conservation approaches.

Landscape-level collaborative conservation approaches are growing in popularity (Hermoso et al., 2022; Reed et al., 2016). Our findings on the variability in management implementation success emphasise the need to prioritise long-term monitoring and fostering meaningful relationships with actors to ensure that these approaches accomplish their goal of supporting biodiversity across landscapes. It is therefore worrying that the current study is, to the best of our knowledge, the first one to evaluate the ecological effects of the collaborative multi-actor conservation approach.

CRediT authorship contribution statement

Klara Leander Oh: Writing – original draft, Visualization, Formal analysis. **Ivo Raemakers:** Writing – review & editing, Investigation. **Jeroen Scheper:** Writing – review & editing, Supervision, Conceptualization. **Mats de Boer:** Writing – review & editing, Investigation. **Joan Díaz-Calafat:** Writing – review & editing, Investigation. **Anne Hage:** Writing – review & editing, Investigation. **Ruud van Kats:** Writing – review & editing, Investigation. **David Kingma:** Writing – review & editing, Investigation. **Hanna Keurhorst-van Krimpen:** Writing – review & editing, Investigation. **Thirza M. de Kruijff:** Writing – review & editing, Investigation. **Wouter G. Oe:** Writing – review & editing, Investigation. **Frank Rooijakkers:** Writing – review & editing, Investigation. **Maarten Frank van der Schee:** Writing – review & editing, Investigation. **Janneke Scheeres:** Writing – review & editing, Investigation. **David Kleijn:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This study received funding from the European Union's Horizon 2020 research and innovation programme under grant agreements No 101003476 (Safeguard) and No 862480 (Showcase); the Dutch Ministry of Agriculture, Fisheries, Food Security and Nature; Natuurrijk Limburg; the province Limburg; and Syngenta.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2025.111682.

Data availability

Data will be made available on request.

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