

Do local landscape features affect wild pollinator abundance, diversity and community composition on Canterbury farms?

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Abstract: Pollination is an essential ecosystem service that can be affected by habitat features in the immediate environment, termed here 'local landscape features'. This study tested how five local landscape features (bare ground, native biodiversity plantings, homestead gardens, shelterbelts, and control areas of pasture) affect local pollinator communities on Canterbury farms. We also compared two sampling methods (flower visitation to native potted plants vs sticky traps) to determine if the sampling method affects the results of landscape-feature comparisons. We recorded 928 pollinators of 17 taxa on the potted plants and 791 pollinators of 16 taxa on the sticky traps. There were significant differences in pollinator abundances between the landscape features and the control areas. Both sampling methods recorded fewer pollinators overall at shelterbelts and bare ground sites than control sites, although morning-evening fly was more common at bare ground sites. However, the methods gave contrasting results for biodiversity plantings and gardens: the flower method recorded significantly more pollinators in biodiversity plantings and gardens than it did at control sites, whereas the sticky trap method recorded significantly fewer. As the flower method showed higher pollinator abundances near biodiversity plantings and gardens, planting native insect-pollinated plants on farms could boost populations of wild pollinators, which may improve crop pollination.

Keywords: agriculture; bees; biodiversity; habitat; insects; pollination

Introduction

Pollination is essential for the reproduction and survival of many wild plant populations, cultivated gardens and agricultural systems (De Groot et al. 2002). In fact, 75% of global crops depend on insect pollination (Klein et al. 2007). The intensification of food crop production is likely increasing the requirements for pollination services per land area and simultaneously reducing local and regional pollinator biodiversity (Ghazoul, 2005).

There are global concerns about pollinator decline (Aizen & Harder 2009) and what this could mean for future cropping systems. Managed pollinators, mainly *Apis mellifera* (honey bee), are heavily relied on to supplement the natural pollination of agricultural crops (Potts et al. 2010). However, *A. mellifera* populations have been threatened by a parasitic mite (*Varroa destructor*) that has spread throughout nearly all major beekeeping countries (Iwasaki et al. 2015). Declines in *A. mellifera* are likely driven by pests, pathogens and management choices (Smith et al. 2013) and the current state of wild pollinators is gaining importance (Potts et al. 2010).

The importance of wild pollinators for pollinating agricultural cropping systems has become increasingly documented. Garibaldi et al. (2013) found that fruit set increased twice as strongly with visitation from wild pollinators than by *A. mellifera* and increased significantly in all of the insect-pollinated cropping systems surveyed. Further evidence shows that high native bee diversity is correlated with better pollination services and fruit set (Kremen et al. 2002; Klein et al. 2003) and non-bee pollinators are able to provide levels of pollination services similar to those provided by bees (Rader et al. 2016). Ghazoul (2005) suggested that regional and local declines of pollinators are likely affecting crop pollination.

Therefore, local management decisions could be vital for wild pollinators and, in turn, insect-pollinated cropping systems.

Habitat features in local environments can have varying impacts on wild pollinator communities. Garibaldi et al. (2014) suggest that key habitat for wild pollinators should include semi-natural areas, habitat heterogeneity, nesting resources and hedgerows. The proximity to and area of semi-natural and natural vegetation has been associated with higher abundances and diversity of bee species (Kremen et al. 2002; Klein et al. 2003; Ricketts et al. 2008; Garibaldi et al. 2011; Blaauw & Isaacs 2014). Agricultural intensification has been shown to reduce appropriate nesting resources for some pollinators (Kremen et al. 2002). However, some pollinating taxa can cope with moderate habitat loss and some agricultural systems can provide nesting sites (Tylianakis et al. 2006; Potts et al. 2010). In agriculturally dominated landscapes, patches of non-crop vegetation such as hedgerows may be important for sustaining pollinator populations. Hedgerows planted with insect-pollinated plants have been found to provide resources for some pollinating species (Morandin & Kremen, 2013). Yet, in New Zealand, hedgerows are typically replaced by shelterbelts of trees planted predominately with *Pinus radiata* and *Cupressus macrocarpa* (wind-pollinated exotic species). Davidson and Howlett (2010) found that only two pollinating taxa (*A. mellifera* and *Bombus* spp.) were associated with these tree species.

There is increasing interest in the state of wild pollinator communities both in urban parks (Webber et al. 2012) and in agro-ecosystems in New Zealand. Two New Zealand studies investigated *A. mellifera* and wild pollinator richness, and compared efficiency and effectiveness of pollinators in agricultural systems. Rader et al. (2014) found that pollinator richness was lower in high intensity systems, and Rader et al.

(2009) found that some common wild pollinator species were as efficient as *A. mellifera*, suggesting that promoting wild pollinator communities might result in services similar to those provided by *A. mellifera*. Although exotic pollinator species dominated highly modified agro-ecosystems, they cannot replace the functional composition of native pollinators, which could have serious implications for pollination services (Stavert et al. 2017). While differences in pollinator diversity and efficiency across farming systems are informative, management to promote wild pollinator populations is likely to be at a local scale and involve only minor modifications to the landscape.

Our first research aim was to investigate how local landscape features affected wild pollinator abundance and species diversity on farming properties in the Canterbury Region, New Zealand. Four local landscape features reflecting contrasting histories of cultivation, species composition and vegetation structure were chosen and compared to control sites (open pasture, which is the dominant landscape element in Canterbury). This research could provide information on how to manage local landscapes to enhance pollinator communities in order to sustain pollination services in the face of managed pollinator declines and increased pollination demands, particularly in land uses that negatively influence pollinator populations. Our second aim was to compare two different sampling methods used to measure pollinator abundances and richness (sticky traps and flower visitation). Sticky traps have previously been used to measure pollinator community composition by Plant and Food Research Ltd in Canterbury. We also recorded insect visitation to flowers on potted plants placed at the field sites. The two methods were compared to determine if choice of sampling method affects the results of pollinator studies.

Methods

Pollinator measurements were carried out in February 2015 on two farming properties in Canterbury. By completing all sampling during a single month, we avoided confounding effects of seasonal changes in pollinating taxa. The first property (43°54.8' S, 172°6.2' E) was located approximately 18 km southeast of the Rakaia Township near Dorie (study site approximately 308 ha), and the second property (43°33.7' S, 171°40.1' E) was located approximately 35 km northwest of the Rakaia Township near the Rakaia Gorge (study site approximately 210 ha). The properties were mainly used for a range of cropping systems, with some pasture used for grazing of stock.

Five local landscape features were studied. (1) Control sites: typically flat areas located in permanent pasture at least 50 m from any of the other landscape features. Pasture was used as the control because it was the dominant habitat between local landscape features. (2) Bare ground: areas at least 2 m² in size, where the ground was bare within a paddock or at a fence line with minimal grass. Bare ground was used as a feature because some native solitary bees use bare ground for nesting sites (Donovan 2007). (3) Biodiversity planting: located in plantings undertaken by staff of Plant & Food Research Ltd during 2013, using a variety of native species to promote pollinator abundances. (4) Garden: located in the gardens surrounding the main farmhouses. There were two of these farmhouses on each property, and in each garden there was a range of native and exotic plants. (5) Shelterbelt: located

beside shelter belts at least 5 m in height and 30 m in length, that were planted with pine or macrocarpa. On each property, measurements were taken at two replicates of each of the five landscape features, resulting in a total of four replicates for each feature and 20 sites across both properties. There was a minimum distance of 50 m between sites.

Two sampling methods were used at all 20 sites. The flower method used potted plants of *Veronica catarractae* (syn. *Parahebe catarractae*) that were approximately 20 cm in height with at least 40 open white flowers per plant. This species was used both because it was producing high numbers of flowers and because the flowers were easily accessible by pollinators. Four potted *Veronica* plants were placed at each site and then observed for 10 min periods to record insect visitation to flowers. Six to eight recording periods were carried out for each site when the temperature was between 15°C and 28°C with wind speeds no greater than 11 km h⁻¹. Sampling at each site occurred at different times of the day over multiple days and there was a minimum of 15 minutes between sampling periods at the same site to reduce the chance of the same pollinators being counted twice. During observation periods, insects that landed on the *Veronica* flowers were noted as potential pollinators and counted. These insects were identified visually, in some cases to species (e.g. *Melanostoma fasciatum*), sometimes to genus (e.g. *Lasioglossum* species) and sometimes to higher taxa (e.g. Tachinidae species). All taxa represent one species with the exception of *Lasioglossum* (two species), *Bombus*, *Pollenia* and Tachinid which may contain a number of species. This may create some bias when analysing species richness. However, we do not believe that this would be significant, and were unable to identify all taxa to species level without trapping all pollinators. Where possible, unknown insects were identified by taking photographs, to later compare to an unpublished collection of photos of local pollinators prepared by Plant and Food Research Ltd.

For the sticky trap method an insect trap was erected at each site and left in the field for 15 days. These traps were made using 4 L paint tins (80 mm diameter and 82 mm height) wrapped in yellow plastic with a yellow base. The cylinder was wrapped in clear plastic (approximately 90 mm × 570 mm), covered in “Tangle-Trap” to create a coating to which small insects would stick. These traps were set up on stakes approximately 1 m off the ground. At the end of the 15 days the plastic strips were taken into the laboratory and the pollinator specimens were identified, sometimes to species level, sometimes to genus level and sometimes to a higher taxonomic level using an unpublished entomology pollinator guide from Plant and Food Research Ltd. Insects smaller than *Delia platura* (< 4 mm in length) were not counted, nor were non-pollinator species (as advised by Plant & Food Research Ltd staff).

Statistical Analysis

All statistical testing used the statistical programme R, version 3.2.1 (R Development Core Team 2015) and R packages lme4 and vegan versions 1.1-10 and 2.4-1, respectively (Bates et al. 2015; Oksanen et al. 2016).

Each dataset (flower and sticky trap) was analysed using Generalised Linear Mixed-effects Models (GLMMs) to test whether landscape feature (a factor with 5 levels) significantly affected the abundance of each pollinating taxon. To test this, the glmer function was run with “property” as the random effect and “landscape feature” as the fixed effect. Poisson error distributions and their canonical log link function were used.

All taxa with an abundance of 20 or higher were run through this analysis separately. Additionally, three new variables were created for each dataset, one for total native insect abundance, one for total insect abundance and one for number of taxa (richness), again run using the `glmer` function with Poisson distributions. To control for varying numbers of observation periods in the flower method while retaining Poisson (count) data, the response variable was the total number of each taxon seen across all intervals at a site, then we corrected for variable numbers of 10-minute observation intervals per site using an “offset” argument in the `glmer` function. To remove the effect of differences in observation time at different landscape features, in Table 1 we present mean visitors per hour.

The effects of individual landscape features (levels of this factor) were tested using the model coefficients. To investigate whether landscape feature as a whole provided significant explanatory power, the analyses were re-run without landscape feature. The two models were then compared using a likelihood ratio test (the `ANOVA` function in R). When the likelihood ratio test was non-significant we chose the simplest model as the best fit.

A large number of GLMM tests were run, particularly on taxon level data, which can result in an increased risk of Type I error. To determine the likely influence of this problem on the results, a Bernoulli process was used to calculate the exact probability of getting the number of significant results found in this study, given the number of trials, following Moran (2003). The data were further analysed using ordinations to test whether there was any overall change in pollinator community composition with regard to sampling method and landscape feature. This community analysis may reveal overall changes in the whole community, in addition to the “single taxon at a time” approach used in the GLMMs.

The metaMDS function from the `vegan` package was chosen to perform non-metric multidimensional scaling (NMDS). This function was run in R using the Hellinger distance metric because it is recommended for species abundance data (Rao, 1995; Legendre & Gallagher 2001). Three ordinations were run using this method. The first was run on the entire data set to compare the sampling methods and then a separate ordination was run on each of the flower and sticky trap datasets to compare landscape features under constant sampling methods.

Additionally, permutation multivariate analyses of variance (PERMANOVAs) were run on all three of the ordinations using the `adonis` function in the `vegan` package within R. The entire dataset PERMANOVA compared the difference in community composition as a consequence of sampling method, and the flower and sticky trap PERMANOVAs compared the difference in community composition as a consequence of landscape feature. This method measures distances of each site to their centroid in multivariate community composition space, then uses a permutation procedure to determine statistically if there was a difference in community composition. We used the Bray-Curtis distance metric, which includes an abundance weighting. Thus, it takes into consideration which taxa are present and how abundant they are.

Because the PERMANOVA for the sticky trap data was non-significant, further analyses were run to explore whether the presence of rare species was skewing the results. First, we re-ran the analysis using the Jaccard distance metric, which tests for only presence or absence (does not provide weighting for abundance) and secondly we re-ran the analysis after removing taxa with very low abundance (< 3 individuals)

from the dataset. These analyses were also non-significant and are therefore not reported in more detail.

Results

Using the flower method, 17 taxa and 928 individual pollinators were recorded, which was an average of 900 insect visits per hour across all taxa (Table 1). Eight of the taxa had counts of 20 or above and were analysed separately. The sticky trap method caught 16 taxa across 791 individual pollinators (Table 2). Twenty or more individuals were recorded for seven taxa, including five taxa that were common in the flower dataset.

For both sampling methods, landscape feature was a poor predictor of pollinator richness (number of taxa). The effects on pollinator richness were small (Tables 1, 2) and rarely significant (Appendix S1, S2 in Supplementary Materials). However, landscape feature was a significant factor for total number of insects and total number of native insects. There were significantly fewer native insects caught at each landscape feature compared to the control feature in sticky traps. However, more native insects were observed at the biodiversity planting and garden sites for the flower dataset. For total pollinator abundance, fewer were found at shelterbelt stations for both methods, the other three landscape features had contrasting results depending on sampling method (Table 1 & 2).

At the taxon level, all significant differences were in the direction of fewer pollinators at shelterbelt sites compared with control sites for both sampling methods (Table 1 & 2). The results for the bare-ground feature were similar, with fewer pollinators found, except that morning-evening fly (*Delia platura*) increased (Table 1 & 2). The results for biodiversity planting and garden sites were distinctive depending on sampling method. There tended to be greater abundances per taxon using the flower method (Table 1) than with the sticky trap method (Table 2). This result suggests that the two sampling methods are affected differently by nearby floral resources (see Discussion); Appendix S1 gives full details of the GLMM analyses.

According to the Bernoulli process used to address the risk of type I error, the probability of finding this number of significant effects (8 of 11 for the flower dataset and 8 of 10 for the sticky trap dataset) by chance was lower than 0.001 for both datasets, indicating that these are likely to be real effects.

The ordination for the entire data set (stress = 0.17) separated out the sampling methods in ordination space and the PERMANOVA showed there was significantly different community compositions measured by the two methods ($F_{1,38} = 9.50$, $R^2 = 0.20$, $P < 0.001$) (Fig. 1a). The ordination run on the flower dataset (stress = 0.17) showed substantial overlap between different landscape features in ordination space (Fig. 1b), but the PERMANOVA showed that there were significant systematic changes in pollinator community composition among the different landscape features ($F_{4,15} = 1.59$, $R^2 = 0.30$, $P = 0.002$). The ordination run on the sticky trap dataset had even less distinction between landscape features. The PERMANOVA run on the sticky trap dataset (not shown) was non-significant, indicating that there was no difference between pollinator communities in different landscape features when considering all taxa at once.

Table 1. The number of insects per hour visiting potted flowers at each landscape feature for each taxon. Taxa highlighted in grey had 20 or more individuals and were used in further analysis; within those rows, features in grey were significantly different from Controls (in direction shown by symbols). For statistical tests see Appendix S1. *Exotic species

Taxon	Control	Bare ground	Bio-diversity	Garden	Shelter belt	Total
Orange hoverfly (<i>Melanostoma faciatum</i>)	51.0	▼27.5	56.0	▲78.9	▼14.0	227.4
*Striped thorax fly (<i>Oxysarcodexia varia</i>)	37.0	▲60.2	▲59.0	▼20.1	▼6.0	182.4
<i>Lasioglossum</i> species	15.0	14.0	▲100.0	▲46.7	0.0	175.7
*Morning evening fly (<i>Delia platura</i>)	9.0	▲49.3	9.0	3.4	15.0	85.7
*Honeybee (<i>Apis mellifera</i>)	2.0	3.8	▲39.0	▲11.3	0.0	56.0
Black hoverfly (<i>Melangyna novae-zealandiae</i>)	10.0	4.0	13.0	9.1	7.0	43.1
Drone fly (<i>Eristalis tenax</i>)	5.0	3.8	▲15.0	5.1	1.0	29.9
*Bronze thorax fly (<i>Pollenia</i> species)	3.0	2.0	9.0	6.1	1.0	21.1
Green soldier fly (<i>Odontomyia</i> species)	1.0	0.0	15.0	1.0	1.0	18.0
*Tachinid species	0.0	5.0	9.0	2.0	0.0	16.0
*Brown blowfly (<i>Calliphora stygia</i>)	6.0	2.8	0.0	2.0	0.0	10.6
*Euro-green blowfly (<i>Lucilia sericata</i>)	2.0	1.0	7.0	0.9	0.0	10.9
* <i>Eumerus funeralis</i>	0.0	0.0	7.0	0.0	0.0	7.0
*Euro-blue blowfly (<i>Calliphora vicina</i>)	0.0	0.0	1.0	2.6	1.0	4.6
Blue hoverfly (<i>Helophilus hochstetteri</i>)	2.0	0.0	3.0	0.0	0.0	5.0
*Bumblebee (<i>Bombus terrestris</i>)	0.0	1.8	1.0	0.0	1.0	3.6
*Other <i>Bombus</i> species	0.0	0.0	1.0	1.7	0.0	2.7
Native pollinator abundance	84.0	▼ 49.3	▲ 202.0	▲ 140.8	▼ 23.0	499.1
Total pollinator abundance	143.0	175.0	▲ 344.0	▲ 190.9	▼ 47.0	899.7
Pollinator diversity	12	12	16	14	9	17

Table 2. The total number of insects caught in sticky traps at each landscape feature for each taxon. Taxa are in the same order as in Table 1; grey highlighting and symbols as for Table 1, with statistical tests in Appendix S2. *Exotic species

Taxon	Control	Bare ground	Bio-diversity	Garden	Shelter belt	Total
Orange hoverfly (<i>Melanostoma faciatum</i>)	51	▼ 3	▼ 8	▼ 29	▼ 3	94
*Striped thorax fly (<i>Oxysarcodexia varia</i>)	3	5	0	0	2	10
<i>Lasioglossum</i> species	51	▼ 11	▼ 8	▼ 29	0	99
*Morning evening fly (<i>Delia platura</i>)	75	▲ 116	61	▲ 104	▼ 34	390
*Honeybee (<i>Apis mellifera</i>)	1	2	0	2	0	5
Black Hoverfly (<i>Melangyna novae-zealandiae</i>)	8	2	3	5	2	20
Drone fly (<i>Eristalis tenax</i>)	0	1	0	0	0	1
*Bronze thorax fly (<i>Pollenia</i> species)	9	8	4	▲ 20	2	43
Green soldier fly (<i>Odontomyia</i> species)	5	0	4	0	2	11
*Tachinid species	16	10	9	24	16	75
*Euro-green blowfly (<i>Lucilia sericata</i>)	1	0	0	0	0	1
* <i>Eumerus funeralis</i>	1	0	0	1	0	2
*Euro-blue blowfly (<i>Calliphora vicina</i>)	0	0	0	1	0	1
Blue hoverfly (<i>Helophilus hochstetteri</i>)	4	0	0	4	0	8
*Three-spotted fly (<i>Anthomyia punctipennis</i>)	22	▼ 3	▼ 3	▼ 2	0	30
<i>Hylaeus</i> species	0	0	0	1	0	1
Native pollinator abundance	119	▼ 17	▼ 23	▼ 68	▼ 7	234
Total pollinator abundance	247	▼ 161	▼ 100	222	▼ 61	791
Pollinator diversity	13	10	8	12	7	16

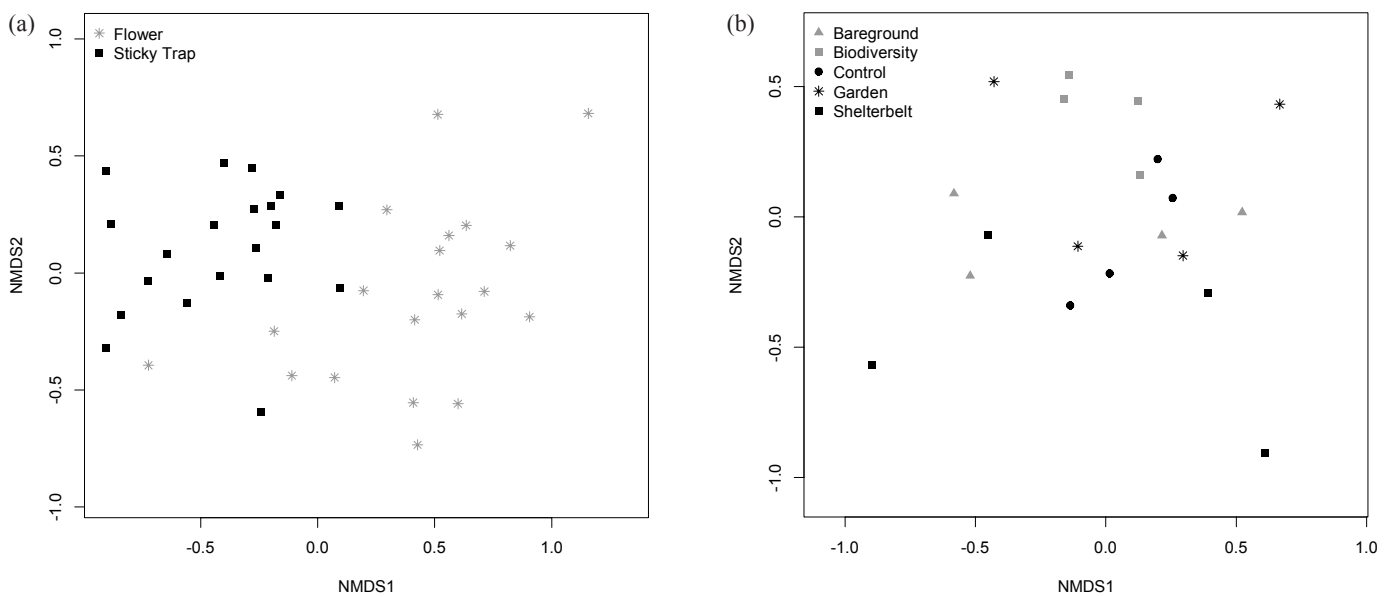


Figure 1. A non-metric multidimensional scaling ordination, using the Hellinger distance metric on (a) the entire dataset by sampling method and on (b) potted flower visitation by landscape feature.

Discussion

Landscape features significantly altered both the measured abundance of pollinators and pollinator community composition, with different taxa responding to varying degrees; the results also varied according to the sampling method.

For landscape feature effects, in the flower dataset biodiversity plantings and gardens had higher richness and greater pollinator abundance compared to the other landscape features. Adjacent areas with high pollinator abundance and diversity can lead to greater crop pollination services and fruit set (Garibaldi et al. 2011; Caralheiro et al. 2012; Blaauw & Isaacs 2014). Increasing the heterogeneity and abundance of flowering plants, particularly native species, will likely increase the wild pollinator abundance and diversity. In turn, this could increase pollination services on Canterbury farms.

The low numbers of pollinators at shelterbelts indicate that pine and macrocarpa shelterbelts (both of which are wind-pollinated) provide poor habitat for insect pollinators, and are potentially reducing the local abundance of wild pollinators on Canterbury farms. Improving habitat for insects in hedgerows can be an important conservation tool for pollinator enhancement (Morandin & Kremen, 2013). Thus, shelterbelts could be planted with floral understory targeted towards wild pollinators, and new hedgerows could be planted predominately with native insect-pollinated species. However, hedgerows can act as a barrier for pollinators between adjacent areas (Wratten et al. 2003; Klaus et al. 2015), which should be considered in the conservation of pollinator communities and crop pollination.

Lasioglossum was the only genus recorded in this study that uses bare ground as a nesting resource. There was no significant difference between *Lasioglossum* abundance at bare ground sites compared with control sites using the flower method. Donovan (2007, p. 134) notes that *Lasioglossum sordidum* nests are usually reported as being in bare ground. However, a study in the Christchurch Botanic Gardens found them nesting in lawns among grass (Bennet et al. 2018). Therefore, there

could be abundant nesting resources in an agricultural matrix for ground-nesting pollinators, suggesting that the bare ground sites are not essential as a nesting resource.

The two methods were apparently measuring different aspects of the pollinator communities, with the two datasets showing disparate community composition in the ordination, and sometimes opposite effects in the taxon-level analyses. In particular, the sampling methods gave opposite results for the garden and biodiversity planting features. One possible explanation could be that the sticky traps are visited (landed on) less often when there are real floral resources in close proximity. This difference in visitation could be due to colour not being the sole attractant for pollinators and other factors influencing pollinator behaviour (Gumbert 2000; Campbell et al. 2010). Perhaps sticky traps are able to attract certain pollinators from distances (being large and bright yellow), but when pollinators get closer they are more interested in real floral resources nearby with more 'attractive' attributes. Likewise, in the gardens there was a diverse range of floral resources (varying in colours, shapes, and sizes), which could also have reduced the appeal of both, the flower and trap methods, for some pollinator species. For example, *Bombus* species were often observed visiting *Lavandula angustifolia* in one of the homestead gardens, but only low numbers were observed visiting the experimental potted *Veronica* flowers and none were caught in the sticky traps. We consider the flower method to be a better measure of local pollinator community composition for this research. More pollinators were recorded using the flower method with most taxa found at higher abundances. The flower results also matched casual observations at the sites showing more pollinator activity around biodiversity plantings and gardens (not less, as the sticky traps found).

The flower method measured the actual flower visitors to small, white native flowers. Thus, this sampling method provided targeted monitoring for pollinators of a specific native plant. The flower method required approximately 60 hours of sampling time to gather the field data. The sticky

trap method catches insects in the vicinity of the trap because either: (1) the insects are blown into the traps by the wind or (2) the insects choose to land on the trap, perhaps attracted to the large yellow shape. The glue used to trap insects is less effective on large insects (\geq honeybees); thus, sticky traps are biased towards smaller insects. The sticky trap method requires less time in the field (approximately 25 hours to place traps out and retrieve them), but does require more time in the lab to separate, identify and count insects. The flower method could be easily adapted to attract a wider range of pollinators by creating a portable planter box with species that vary in flower colour, size and structure.

The flower method provided evidence that biodiversity plantings attract greater numbers of pollinators. These results indicate that targeted native planting is a useful tool to enhance wild pollinator communities on farms in Canterbury. If native plantings are used in farm management, future research could investigate three questions. (1) Can native plantings provide an increase in crop yields in adjacent fields? (2) How close do these plantings need to be to enhance pollinators and crop yields? (3) Do the size and shape of these plantings have an effect on pollinators and crop yields? Carvalho et al. (2012) found that areas of native plantings as small as 25 m² could enhance pollinator-dependent crop production if combined with other land management. Biodiversity plantings are a tool that is already used on farmland to support ecosystem services. For example, native riparian plantings are used to improve water quality (Collins et al. 2013) and other native plantings have been used to support natural biological control (Walton & Isaacs 2011). The results from this study further support biodiversity plantings as a tool on farmlands to enhance species providing pollination services.

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Supplementary Material

Additional supporting information may be found in the online version of this article:

Appendix S1. Summary of Generalised Linear Mixed Model (GLMM) results, using the data from potted flower visitation, testing abundance of each taxon against landscape feature, with property as the random effect.

Appendix S2. Summary of Generalised Linear Mixed Model (GLMM) results, using the data from pollinators caught in sticky traps, testing abundance of each taxon against landscape feature, with property as the random effect.

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