



Gardens benefit bees and enhance pollination in intensively managed farmland

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ABSTRACT

The recent loss of pollinating insects and out-crossing plants in agricultural landscapes has raised concern for the maintenance of ecosystem services. Wild bees have been shown to benefit from garden habitats in urban and suburban areas. We investigated the effects of distance from garden habitats on wild bees and seed set of a native out-crossing plant *Campanula persicifolia*, in intensively managed agricultural landscapes in Southern Sweden. Bee abundance and species richness, as well as plant seed set, were higher closer to gardens (<15 m) than further away (>140 m). This highlights private gardens as a landscape wide resource for pollinators but also the lack of sufficient pollination of wild plants in contemporary agricultural landscapes.

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1. Introduction

Agricultural intensification, resulting in loss and fragmentation of natural habitats, has caused large-scale losses of farmland biodiversity in general (Krebs et al., 1999). Widespread declines of pollinators have received particular attention because of the risk to the ecosystem service they provide (Kremen et al., 2002; Potts et al., 2010; Steffan-Dewenter et al., 2005). In fragmented landscapes, a main threat to wild plant reproduction is pollination failure, caused by lack either of mates or of pollinators (Wilcock and Neiland, 2002). In fact, large-scale losses of pollinators have been paralleled by losses of out-crossing plant species (Biesmeijer et al., 2006; Gabriel and Tschardtke, 2007).

Semi-natural habitats are known to positively affect pollinators in the surrounding agricultural landscape (e.g. Öckinger and Smith, 2007; Ricketts et al., 2008) presumably through contributing both nest sites and forage resources. Other non-crop areas such as field margins may also be beneficial provided that they are rich in flower resources (Kleijn and van Langevelde, 2006). Linear landscape elements are also known to be important for bumblebee nesting (Lye et al., 2009; Osborne et al., 2008; Svensson et al., 2000). Non-crop, semi-natural areas add heterogeneity to otherwise, in many aspects, simplified agricultural landscapes (Benton et al.,

2003). Another type of non-crop areas is domestic gardens situated in the countryside. In heavily cultivated surroundings gardens can be assumed to enhance floral abundance and diversity, as well as three-dimensional structure (i.e. habitat complexity). Lately attention has been drawn to the positive impact of urban gardens and allotment gardens on pollinators (Ahrné et al., 2009; Goddard et al., 2010) and on the process of pollination (Cussans et al., 2010). Gardens often provide a continuous supply of nectar and pollen which bees can utilise (Fussell and Corbet, 1992). Suburban gardens have been shown to increase growth of experimental colonies of *Bombus terrestris* compared to rural areas (Goulson et al., 2002). In urban gardens, habitat complexity and diversity of flowering plants have been shown to be positively related to bumblebee and solitary bee diversity (Smith et al., 2006). Gardens can provide suitable habitats for bees to nest and have been shown to contain higher densities of bumblebee nests than grasslands and woodlands in arable landscapes (Osborne et al., 2008). Hence, gardens may promote pollinator abundance and species richness also in agricultural landscapes.

However, measures promoting pollinators may not necessarily benefit pollination of wild plants, because species may vary in their effectiveness as pollinators (Klein et al., 2003). Species may for example vary in rates of removal and deposition of pollen (Wilson and Thomson, 1991) and also in their degree of flower constancy (Goulson, 1999). Another example of a more indirect effect on pollination is large-scale cultivation of oilseed rape, *Brassica napus* L. This mass flowering crop may be beneficial for some early emerging and short-tongued bumblebee species, but results in reduced abundance of long-tongued bumblebees, which are in turn important

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pollinators of plants with deep corollas (Diekötter et al., 2010). Thus, it is important to determine the effect of gardens not only on the general abundance of pollinators, but also on different groups of pollinators and pollination *per se*. It has also been shown that the decline with distance (from natural or semi-natural habitats) of native pollinator visits to crops is steeper than the decline of pollinator richness with distance (Ricketts et al., 2008), which again highlights the importance of studies including the pollination service itself. Foraging ranges of bees are positively related to body size (Greenleaf et al., 2007). In the region of this study, bee body size correlates with sociality in that social bees (*Bombus* species and *Apis mellifera*) are larger than solitary bees.

Our aim was to investigate whether gardens in landscapes highly dominated by agriculture can act as sources of pollinators and subsequently benefit pollination of wild out-crossing plants. To this end we investigated whether species richness and abundance of bees were higher close to gardens than further away, whether the abundance of two groups of pollinators (large social and small solitary bees) were differently affected by distance and whether, because of improved pollination, plant seed set for a native out-crossing plant was higher close to gardens. We focused on bees (Hymenoptera: Apoidea) as they are an important group of pollinators (Winfree et al., 2008). Bees can benefit from gardens for both nesting and foraging but as they are central place foragers with restricted foraging ranges (Goulson, 2003) they may also be negatively affected by distance between nests and forage sites and thus indirectly allow detection of their source of origin.

2. Material and methods

2.1. Study set up

The study was carried out in southernmost Sweden (approx. 56°N, 13°30'E, Fig. 1). We selected nine landscape sectors (squares of 2.5 × 2.5 km) situated in a region largely dominated by agriculture. All landscapes were similar regarding the percentage area of sectors under agricultural land use which was on average 81.7 ± 10.7% (mean ± stdv). Of this area annual crop fields comprised 91.2 ± 5.9% and leys 5.9 ± 5.5% (mean ± stdv). The total area of permanent pastures was 1 ± 1.5% (mean ± stdv). Within each landscape sector two isolated domestic gardens were identified and inspected to ensure reasonable similarity with respect to features important to pollinator abundance and diversity (Osborne et al., 2007; Smith et al., 2006). All gardens had an area of at least 500 m² and included all the following features: planted flowers, native herbaceous plant species, trees, flowering bushes and sections with tall grass. The majority of gardens also included hedges and a compost heap.

Along a road verge bordering a non-flowering crop field and reaching out from the gardens, either phytometers (plants used to estimate pollination [cf. Albrecht et al., 2007], in this case *Campanula persicifolia*) or a set of three pan-traps were placed. Phytometers and traps were placed at two different distances, either “proximate” within 15 m from the edge of the garden or “distant” approximately 140 m away. Since the study was made up of paired potted plants and traps, respectively, separated by only ca. 125 m, factors related to land-use and management as well as vegetation and soil properties were largely controlled for. We did not use the same garden for both phytometers and traps because of the risk of pollinator depletion due to the traps. Which of the two gardens within a landscape received phytometers and traps, respectively, was randomly selected. One landscape also had a third garden with phytometers bagged in fine mesh. These plants served as pollinator free controls and were placed in the field to ensure similar weather and wind conditions to experimental plants.

The pan-traps were sets of three traps: one yellow, one white and one blue plastic cup, (6 cm deep, Ø 15 cm) sprayed with the corresponding fluorescent colour (Sparvar, Leuchtfarbe). Traps were placed directly on the ground and filled with 50% propylene glycol. Each phytometer consisted of two plants of peach-leaved bellflower, *C. persicifolia*, a wild, self-incompatible flower native to Sweden (Nyman, 1992). The plants were purchased from a local garden centre at the beginning of May 2009 and were immediately replanted in 7.5 l pots with commercial garden soil. Both the phytometers and traps were kept in the field for 3 weeks, from the end of June until mid July, and were visited and watered twice a week. To be able to determine in which order inflorescences had bloomed, we marked all inflorescences that had started to bloom since the last visit (i.e. every 3rd to 4th day) with coloured thread and used a unique colour for each visit to keep track of the order of flowering.

C. persicifolia was present in one of the gardens where phytometers were placed. Lack of other suitable gardens made us unable to remove this garden from the study, but we do not believe that this will lead to any bias since the study design focuses on pollination in relation to distance from gardens rather than on pollination inside gardens. Both proximate and distant phytometers should benefit from a pollen source inside the garden. Insects caught in traps were collected and stored in 70% ethanol. After the field study all plants were transferred to a greenhouse.

2.2. Data collection

All capsules from *C. persicifolia* marked in the field, except those marked at the last visit, were harvested between 30 July and 20 August when ripe ($n = 233$). Seeds were weighed and we used

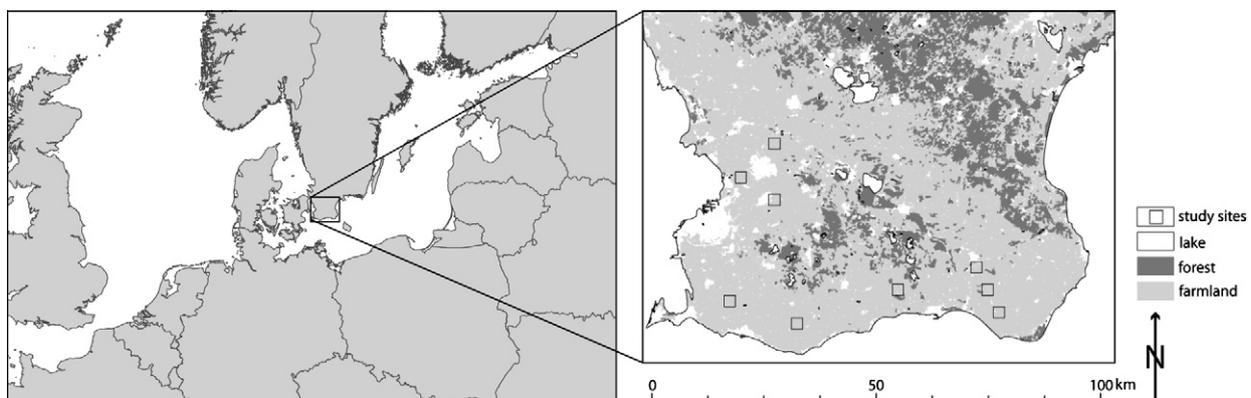


Fig. 1. Map of the study region in southernmost Sweden, which is largely dominated by agriculture. The nine landscape sectors (2.5 × 2.5 km quadrates) used in the study are drawn.

the weight of each capsule's seeds as a proxy for seed set. To estimate plant size we noted total number of flowers per plant. Capsules hosting seed eating weevils ($n = 68$) were excluded from analysis. In two landscapes plants had all flowers and capsules eaten by slugs, leaving us with six complete pairs of phytometers and one with only distant plants. Analyses were run with and without honey bees ($n = 18$) since their origin is determined not only by habitat quality but also by where bee keepers place hives. For some analyses we distinguished between social and solitary species to evaluate if body size has an effect on pollinator foraging distance at the distances under study. Nine proximate and nine distant sets of three pan traps were used in the field. One landscape's traps had to be excluded from the analyses because of continuous dust accumulation in the traps preventing colour reflection and thereby attraction of bees, resulting in samples from eight landscapes. Traps were emptied twice a week for the 3 weeks that the experiment lasted, i.e. there were a maximum of five samples from each trap. In three landscapes, traps got damaged and were therefore emptied only four times. If a trap had been damaged, neither the sample from this nor other traps in that landscape and date were used.

2.3. Statistical analysis

To account for the pair-wise design with two distances, we used mixed models with Landscape as a random factor and the Distance from the gardens as a fixed factor. Seed set was analysed using general mixed model (SAS Proc Mixed) assuming normal distribution, whereas counts were analysed using generalised linear models (SAS Proc Glimmix) assuming Poisson distribution, while accounting for over dispersion by using an extra-dispersion scale parameter. Denominator degrees of freedom were estimated using the Satterthwaite approximation.

For analyses of seed set (i.e. total seed weight) we calculated the mean value of the capsules' seed set at each plant followed by mean value per distance. To test if the size of plants had any effect, we also analysed data at the plant level, including Distance nested within Landscape as a second random factor and Plant size as a covariate. To test if sequential order of inflorescences (i.e. the order in which individual inflorescences bloomed on a plant) had any effect on seed set we analysed data at the capsule level including Plant nested within Distance and Landscape as a third random factor and with Order of inflorescence and Plant size as covariates. In these latter two analyses, there was a negative covariance component between seed weight from the two plants at the same Distance, possibly because of competition, which was accounted for by the random structure in the analysis. Tests were run with and without data from capsules without any seeds but not clearly attacked by weevils ($n = 18$), which we suspected were from seed abortion or damage to the plant. Pollinator data was summarised at each Distance within a Landscape. For comparison of the effect of distance on social and solitary bees, the abundances of groups were summarised separately and Distance nested within Landscape included as a second random factor.

3. Results

In total, 244 bees of 28 species and eight genera were sampled in the 16 triplets of pan-traps (Table 1). The most abundant social bee was *B. terrestris* (27% of social bees) and the most abundant solitary bee was *Andrena nigroaenea* (29% of solitary bees). Abundances of bees were significantly higher proximate than distant to gardens ($F_{1,7.46} = 21.02$, $P = 0.0021$). Over the 3 weeks of the study, on average 23.75 ± 6.79 (mean \pm SEM) bees were sampled per set of three proximate traps and 7.25 ± 1.42 bees per set of

Table 1

Total number of individuals and species per genus of social and solitary bees collected in pan-traps.

Genus	No. individuals	No. species
<i>Bombus</i>	148	12
<i>Apis</i>	18	1
<i>Andrena</i>	40	6
<i>Lasioglossum</i>	23	4
<i>Halictus</i>	11	2
<i>Colletes</i>	2	1
<i>Hylaeus</i>	1	1
<i>Osmia</i>	1	1

three distant traps. Social and solitary bee abundances were not differently affected by distance ($F_{1,21.61} = 1.19$, $P = 0.29$), and social bees were significantly more abundant in proximate traps even when honey bees were excluded ($F_{1,7.66} = 11.75$, $P = 0.0096$). Species richness of solitary bees was significantly higher close to gardens (3.28 ± 0.96) than farther away (1.13 ± 0.30), ($F_{1,14} = 5.79$, $P = 0.0305$). Bumblebee species richness was only marginally significantly higher in proximate traps (5.00 ± 0.93) compared to distant traps (3.25 ± 0.65 ; $F_{1,7.62} = 4.88$, $P = 0.060$).

The mean seed weight per capsule was significantly higher on proximate (32.55 ± 2.67 mg) compared to distant phytometers (17.78 ± 1.83 mg), ($F_{1,5.01} = 12.27$, $P = 0.017$; effect size 1.95), Fig. 2. The result held true both when excluding the garden containing *C. persicifolia* ($F_{1,4.23} = 9.12$, $P = 0.037$) and when capsules without seeds were excluded ($F_{1,5.14} = 7.86$, $P = 0.037$). Plant size and sequential order of flowering did not explain any additional variance ($P = 0.52$ and $P = 0.17$ respectively). The control plants bagged in the field ($n = 11$ capsules) did not set any seeds, confirming that *C. persicifolia* is self-incompatible and dependent on animal pollination (Nyman, 1992).

4. Discussion

We found evidence that gardens acted as a source of pollinating bees for a native out-crossing plant in landscapes dominated by agriculture. Both abundance and species richness of bees were higher close to gardens than further away. Furthermore, seed set of *C. persicifolia* was higher close to gardens, suggesting that the presence of gardens indeed enhanced pollination. Our results also strengthen the notion that modern agricultural landscapes are lacking in pollinator services. They further point to the value of other habitat types than the natural or semi-natural ones, which are commonly considered in these circumstances and most often constitute the focus of both scientific studies and management actions. To our knowledge there are no previously published studies

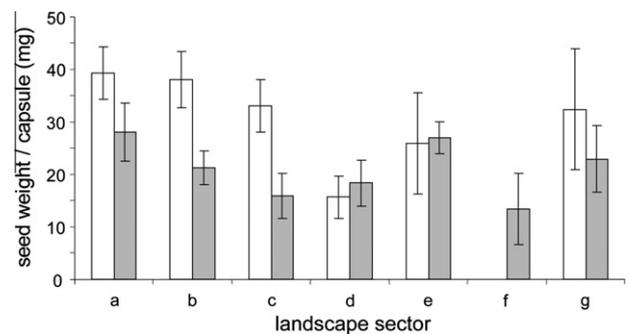


Fig. 2. Mean (and SEM) in seed weight per capsule of *Campanula persicifolia* at proximate (white bars) and distant (grey bars) locations. The proximate location is missing from landscape f, see text.

from Scandinavia regarding the value of domestic gardens for pollination or the lack of full pollination of native plants growing in highly productive farmland landscapes.

It remains to be shown to what extent our results generalise to other plant species. In a similar study of an agricultural crop, *Trifolium pratense*, we could not detect any effect of gardens on seed set because of heavy seed predation (Samnegård, 2010). Likewise, Albrecht et al. (2007) could not detect any effect of distance (<200 m) from restored meadows on either decline of large sized pollinators or seed set of three insect pollinated plants species (*Raphanus sativus*, *Hypochaeris radicata* and *Campanula glomerata*). On the other hand, small sized pollinators did show clear declines (Albrecht et al., 2007) and visitation to and seed set of *Centaurea jacea* showed a negative relation with distance from meadows (Albrecht et al., 2009). Steffan-Dewenter and Tschardt (1999) demonstrated declines in seed set of *Sinapis arvensis* and *R. sativus* related to declines in bee visits with distance from grasslands and Cussans et al. (2010) reported on increased seed production of *Glechoma hederacea* and *Lotus corniculatus* when grown in suburban gardens compared to adjacent farmland fields. In other words, whether proximity to semi-natural or other flower enriched and complex non-crop habitats indeed benefits pollination of a particular species depends on characteristics of the pollinator community involved as well as the reproductive system of the plant. Also factors related to plant population size and density may affect both pollinator visitation frequencies and seed set (Dauber et al., 2010).

Solitary bees are known to forage close to their nests, whereas many bumblebee species cover greater distances (reviewed in Zurbuchen et al., 2010). We therefore used sociality as a proxy for body size and foraging distance; social bumblebees constituting the “large size and long distance”-group and solitary bees the “small and short”-group. However, we did not find any difference in how abundances of solitary and social bees declined with distance from gardens. Distant sites were however only 140 m away from gardens, a foraging distance which may be achieved also by many solitary bees (Zurbuchen et al., 2010). Furthermore, sample sizes of solitary and social bees separately were small, resulting in low statistical power. Another study on distance from conservation grasslands has demonstrated a difference between small and large pollinators (Albrecht et al., 2007).

The fact that distant plants had a lower seed set than proximate ones in the present as well as in other studies (e.g. Albrecht et al., 2009; Ricketts et al., 2008; Steffan-Dewenter and Tschardt, 1999) may suggest a shortage of pollination of wild plants in intensively managed landscapes. A shortage of pollinators can in turn, through a decrease in the pollination service they provide, affect plant community structures (Biesmeijer et al., 2006). Interestingly, organic farming has been found to benefit both pollinators (Holzschuh et al., 2008; Rundlöf et al., 2008) and insect-pollinated plants (Gabriel and Tschardt, 2007). Likewise, domestic gardens may promote persistence of insect-pollinated wild plants in intensively farmed landscapes because the resource rich habitats they constitute act as refuges for pollinators; habitats which have so far largely been overlooked in agricultural ecosystems (Goddard et al., 2010).

Despite the relatively low sample size, we found 12 out of the 17 species of true bumblebees potentially found in southern Sweden (Holmström, 2007). The majority of the species not found are either locally extinct or extremely rare (Holmström, 2007). Thus, a quite diverse species pool may still exist in pockets of beneficial habitat, even in intensively managed agricultural regions in Sweden; possibly partly because of the presence of gardens (cf. Osborne et al., 2008). This implies that pollinator conservation in this region may actually pay off quite quickly, since at least there are remnant populations to build on.

Earlier studies on the impact of domestic gardens on pollinators have focused on urban or suburban environments (Ahrné et al., 2009; Cussans et al., 2010; Fetridge et al., 2008; Goddard et al., 2010; Goulson et al., 2002; Matteson et al., 2008; but see Osborne et al., 2008) or on pollinators in urban parks (McFrederick and LeBuhn, 2006). We have shown that gardens can contribute to the ecosystem service of pollination also in agricultural landscapes. Since gardens often include features beneficial for many bee species; e.g. a diversity of nesting substrates and continuous supply and diversity of nectar and pollen (Fussell and Corbet, 1992; Osborne et al., 2008), they may complement more “natural” habitats for pollinators in otherwise impoverished environments. However, establishing more gardens in agricultural landscapes is of course not a realistic conservation measure. Instead we propose that by making the importance of gardens known, awareness of ecosystem services can be spurred and improvements of existing gardens can be made by an interested general public. Also, acknowledging garden habitats as a resource for biodiversity not only in cities, could lead to domestic gardens being included in conservation planning situations (Goddard et al., 2010) also outside the urban environment. The position and management of gardens could for example be considered one way to increase connectedness of isolated (semi)-natural habitat fragments. The relatively high species richness of bees found in proximity to gardens also demonstrates the importance of not overlooking gardens (and other recently man-made habitats) when studying biodiversity, especially in otherwise species poor environments. Most importantly however, the lack of pollination (even only 140 m from gardens) found here calls for more directed measures to aid pollinators in agricultural landscapes.

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