No significant effects of insecticide use indicators and landscape variables on biocontrol in field margins

L. Bakker a, *, W. van der Werf b, F.J.J.A. Bianchi a

a Farming Systems Ecology Group, Wageningen University and Research, Wageningen, the Netherlands
b Centre for Crop Systems Analysis, Wageningen University and Research, Wageningen, the Netherlands

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A B S T R A C T
Insecticides are used widely to manage pests, but they may cause mortality in natural enemy populations and non-target prey. On the other hand, resources provided by the landscape – often associated with semi-natural habitat – may enhance natural enemy communities. While there is substantial information on how insecticide use and semi-natural habitats influence biocontrol in crop fields, little is known about how local insecticide use interacts with landscape variables to govern natural biocontrol in field margins. In this 2-year study we assessed predation and parasitism rates of eggs of the cabbage moth (Mamestra brassicae) on potted cabbage plants in field margins adjacent to 20 organically managed fields and 18 conventionally managed fields. We analysed egg predation and parasitism rates in field margins using three predictors for insecticide use: (i) farming system (organic or conventional), (ii) frequency of insecticide use in the adjacent focal fields, and (iii) estimated insecticide use quantity within a radius of 1 km from the sampling sites. Insecticide use included foliar or soil-applied applications, including mineral oils. Landscape variables were used as covariables to account for landscape effects. Mean predation and parasitism rates were 14% and 6%, respectively. There were no significant associations between predation and parasitism rates in field margins with insecticide use in the adjacent field or the wider landscape. Our findings suggest that field margins can be valuable habitats for natural enemies in agricultural landscapes, irrespective of the insecticide use intensity at the local and landscape scale.

1. Introduction
Agricultural intensification has been associated with declines in biodiversity and regulating ecosystem services (Kleijn et al., 2019). Intensive management practices, including insecticide use, can impact biological pest control in arable crops (Bianchi et al., 2006; Pretty, 2018; Tooker et al., 2020). In contrast, resources provided by the landscape – often associated with semi-natural habitat – may enhance natural enemy communities (Harvey et al., 2020). While we have a fairly good understanding of how insecticide use and semi-natural habitat influence biocontrol in crop fields, little is known about how local insecticide use interacts with landscape variables to govern natural biocontrol in field margins.

Insecticides cause mortality in natural enemy populations and their prey in crop fields (Busch et al., 2020; Sponsler et al., 2019; Wagner, 2020; Woodcock et al., 2016b), but also affect beneficial insect populations in surrounding non-target habitats (Geiger et al., 2010; Marshall and Moonen, 2002). Non-target habitats, such as field margins, are often in close proximity to arable fields and are widespread in agricultural landscapes. These habitats may have an important function for beneficial insects within the agroecosystem, e.g. by providing overwintering sites, food resources due to presence of alternate prey or flower resources, reproduction sites, or dispersal corridors (Dover, 2019; Li et al., 2020; Marshall and Moonen, 2002; Pywell et al., 2015). Drift or overspray of insecticides impacts plant community structure and plant diversity (Schmitz et al., 2014), and affects non-target insects in field margins through (i) close contact to polluted areas, (ii) eating of contaminated food, and (iii) uptake through mouth and nose airways (Botías et al., 2019; Marshall and Moonen, 2002). However, impacts of insecticides on arthropod communities can reach further than would be expected based on drift or direct overspray alone (EFSA Scientific Committee, 2016), as arthropod populations are mobile and may give rise to ‘action at a distance’ effects (Sromberg et al., 1998). For instance, insecticide-inflicted mortality in sprayed fields may turn these fields into sink habitats, thereby also reducing population densities elsewhere in the landscape (EFSA Scientific Committee, 2016;
Resources associated with (semi-natural) habitats may enhance the abundance and diversity of natural enemies and the associated biological control services (Chaplin-Kramer et al., 2011; Rusch et al., 2016; Tscharntke et al., 2005). Spill-over of natural enemies from source habitats can be influenced by patch size, patch shape, shared edges, or through complementation of resource availability across habitats (Bianchi et al., 2013b; Haan et al., 2020). In diversified landscapes with high levels of natural enemy activity, insecticide-treated areas may be readily recolonized by natural enemies from source habitats once the insecticide residue has declined (EFSA Scientific Committee, 2016; Schellhorn et al., 2014; Sromberg et al., 1998). In these cases, spill-over of natural enemy populations from semi-natural habitats to agricultural fields may counterbalance non-target impacts of insecticides on natural enemies in fields (Roubos et al., 2014; Tscharntke et al., 2016). Whereas spill-over effects of natural enemy populations from field margins to agricultural fields have been relatively well studied (e.g. Dainese et al., 2017; Holzschuh et al., 2007; McHugh et al., 2020; Woodcock et al., 2016a), the impacts of insecticides on ecosystem services in field margins have received little attention.

Here, we explore the effects of insecticide use on biocontrol services (egg predation and parasitism of *Mamestra brassicae*) in field margins. Our study focuses on the assessment of the biocontrol potential of flying natural enemies. We assess whether insecticide use in adjacent fields or the surrounding landscape affects biological control in field margins. Furthermore, we study whether the proportion of semi-natural habitats in the surrounding landscapes counteracts potential impacts of insecticide use on the biocontrol potential of natural enemies in field margins. We hypothesize that (i) biocontrol services in field margins adjacent to conventionally managed fields are lower than in organically managed fields due to drift of insecticides in conventionally managed fields, and ‘action at a distance’ mechanisms; (ii) biocontrol services in field margins are positively associated with the proportion of semi-natural habitat at the landscape scale because semi-natural habitats may act as source habitats for natural enemies, and (iii) the proportion of semi-natural habitat at landscape level counterbalances the impacts of insecticide use on biocontrol services in field margins.

2. Materials and methods

2.1. Experimental design

Thirty-eight field margins (sites) were selected across twenty landscape settings, ranging from structurally simple to structurally complex. Sites were located adjacent to organically or conventionally managed arable fields (Fig. 1), and paired within landscapes to account for possible effects of the surrounding landscape. This resulted in 18 paired organic and conventional sites, while there were two non-paired organic sites in addition. The mean distance between sites was 6.75 ± 4.22 km (range 1.72–18.1 km). Field margins consisted of mown grass-dominated vegetation, unmown grass-dominated vegetation, or flower strips. All fields were managed according to farmers’ practice, and cultivated crops included potatoes (21 fields over 2 years), cereal crops (18), cover crops (4), corn (3), herbs (2), tulips (1), sugar beets (5), beans (3), cabbage (2), and other vegetables (17). Field data were collected in June and July 2017 and 2018 at each of the 38 sites to obtain replication over time and assess the consistency of effects and relationships across early and late season and years.

2.2. Field experiment

At each site we placed four plastic pots with a single 5-weeks old cabbage plant (*Brassica oleracea*) in the field margin for 48 h. The four plants were placed in sets of two, at approximately 50 m from each other. To quantify predation and parasitism, we pinned a card with an egg batch of the cabbage moth (*Mamestra brassicae*) on each plant on the lower side of a newly mature leaf. Egg batches were obtained from a laboratory colony of cabbage moth and were not older than one day. We chose this species because it is a prominent pest in cabbage crops and predated by a range of generalist predators, including spiders, staphylinids and carabids (Bianchi et al., 2005; Pfiffner et al., 2009). We assumed that ground-dwelling arthropods were not able to climb the pots in which the plants with egg cards were placed. The exposure and retrieval of egg batches was completed in a period of 3 weeks during each sampling round, and four sampling rounds were conducted in two years; weeks 25–27 and weeks 30–32 in 2017, and weeks 24–26 and weeks 29–31 in 2018. After a two day-exposure period, all egg cards

![Fig. 1. Locations of the 38 sampling sites in the Netherlands. Pies show the composition of the landscape at 1 km radius around the focal fields. Map of the Netherlands downloaded from GADM (https://gadm.org/index.html).](image-url)
were collected. Egg predation was assessed by taking photographs of the egg card before and after exposure in the field, and counting the number of eggs per egg card before and after field exposure. Empty, damaged or missing eggs were considered as destroyed or removed by predators (‘predation’). Parasitism was assessed in the laboratory. The collected egg cards were incubated for at least one week at 22 °C to allow egg parasitoids larvae to develop within the moth egg, which then turns brown or black as the larvae pupate. After one week the number of darkened, parasitized eggs were counted (‘parasitism’).

2.3. Land use variables

Land use data were extracted from a publicly available land-use database (TOP10NL) and crop field registrations (BRP2017 and BRP2018) (PDOK, 2018). A third database consisted of organic and conventional crop field registrations in 2017. A polygon map, based on these three databases, was developed to derive landscape parameters within a 1 km radius area around each sampling site. The 1 km spatial extent was chosen because this captures the functional spatial scale for a range of flying natural enemies (Bianchi et al., 2008; Thies et al., 2003). Six land use categories were used to describe land cover: i) organic agriculture (including orchards); ii) conventional agriculture (including orchards); iii) herbaceous semi-natural habitat (extensive perennial grassland, nature areas/reserves, flower strips, grassy field margins); iv) wooded habitats (hedgerows and forests); v) pasture, both permanent and annual grasslands (either organic or conventionally managed); and vi) urban area. These land-use categories were chosen to account for potential effects of crop management intensity (organic versus conventional) and habitat (agriculture versus semi-natural habitat) on biological control (Karp et al., 2018; Zou et al., 2020). At the 1 km radius scale, the proportion of herbaceous semi-natural habitats ranged from 0 to 13 %, the proportion of woody habitats ranged from 0 to 60 %, the proportion of organic farming ranged from 0 to 40 %, and the proportion of conventional farming ranged from 5% to 87 % in the landscape circles (Appendix A).

2.4. Quantification of insecticide use

Insecticide use intensity in the adjacent field of the sampled field margin was expressed in terms of Treatment Frequency Index of applied insecticides (TFI) (Gravesen, 2003). TFI is an indicator to express insecticide dependency of cropping systems (Lêchenet et al., 2014). TFI is calculated using as follows:

$$\text{TFI} = \sum (\text{AD} / \text{MD})$$  \hspace{1cm} (1)

where AD is the applied dosage of formulated insecticide product $i$ (kg per ha$^{-1}$), and MD is the maximum label-recommended dosage for this product (kg per ha$^{-1}$). Summing over all applications, Information on the product and the dose was provided by farmers. In 2017 we received insecticide use information of 14 conventionally managed fields, and 20 organically managed fields, while in 2018 we received information of 9 conventionally managed fields, and 20 organically managed fields. Farmers used soil-applied and foliar insecticide applications, including mineral oils, and TFI ranged from 0 to 20. Due to unavailability of data we were not able to include seed coatings as insecticide treatment. TFI at most organic sampling sites was zero, but some organic farmers used organically approved insecticides and these treatments were included in the calculation of TFI. Information on maximum label-recommended dosages were derived from the online database of the Dutch Board for the Authorisation of Plant Protection Products (CTGB, 2019).

Since there are no open data bases of pesticide use, we estimated the quantity insecticides applied (QIA in kg active ingredient ha$^{-1}$) at the landscape scale as the total input of active ingredient per crop species. First, we extracted insecticide use information (kg active ingredient ha$^{-1}$) for 43 crops from the national pesticide data censuses of 2016 (CBS, 2020) (Appendix B). Third, we extrapolated the 2016 insecticide use information to our crop-area data. Thus, QIA amounts to the total estimated quantity of insecticides applied per hectare within each 1-km landscape circle around the focal field margin.

2.5. Data analysis

The relationship between biological control potential, insecticide use and landscape composition was analysed with generalized linear mixed models (GLMM). Data on egg predation and egg parasitism on the four plants per site were pooled, resulting in 145 observations over the 38 sites and four sampling periods. Seven observations were missing due to the inability to visit all sites in sampling round three, and fields for which we did not receive TFI information were considered to have missing data for the insecticide indicator TFI. Egg predation and egg parasitism were analysed as separate response variables (‘A’ and ‘B’ models, respectively; Table 2 and 4).

To estimate the effect of insecticide use on egg predation and parasitism, we used the following variables as insecticide indicators: (i) management of the field adjacent to the margin (management; organic vs. conventional), (ii) insecticide use intensity in the field adjacent to the margin (TFI), and (iii) estimated quantity insecticide applied (QIA) per ha in a 1-km circle around the focal field margin. We accounted for variation in field margins by categorizing field margins in three types: mown grass-dominated vegetation, unmown grass-dominated vegetation, or flower strips. This categorical explanatory variable was included as a fixed factor in all models. Landscape variables were entered as continuous covariables to account for differences in landscape composition: woody semi-natural habitat (‘woodySNH’), herbaceous semi-natural habitat (‘herbSNH’), proportion of land under organic management (‘propORG’), and proportion of land under conventional management (‘propCONV’). The proportion of pastures was omitted due to colinearity with other landscape variables. Interactions between insecticide use indicators and ‘woodySNH’ were added as we expected the effect of insecticide use on biocontrol services to differ in landscapes with different proportions of woody semi-natural habitat. All continuous explanatory variables were standardized prior to analysis.

‘Farm pair’ was included as a random factor to account for possible effects associated with the landscape context in which the paired farms were embedded. We included ‘sampling period’ (with four levels) as random effect to account for the effects of variation in weather conditions in interaction with phenological processes, even though we found no main effects of season (June vs. July) or year (2017 vs. 2018) for parasitism, and no main effect of season for predation. We tested the effect of crop type on predation and parasitism, but there was no significant relationship, and therefore we decided to not include this factor.

While the number of eggs offered was finite (86 ± 44 eggs; mean ± SD), the actual number of eggs parasitized or removed by predation was well below this maximum. We therefore used count models without a maximum to model the stochastic variation in egg predation and parasitism, and explored two error distributions: the Poisson and the negative binomial error distribution. Here we report the analysis with the negative binomial error distribution because it provided the best fit to the data as judged by Akaike’s information criterion (AIC).

Because we did not find strong support for a single ‘best’ model, based on corrected Akaike’s criterion (AICC), we used model averaging among the set of ‘best’ models to interpret model results (Burnham and Anderson, 2007; Feld et al., 2016; Grueber et al., 2011). We compared two criteria to select the set of top models for averaging: (i) all ‘top’ models based on $\Delta AICc < 2$, and (ii) the cumulative weight of all top models is 95 %. Model averaging estimates were obtained using the zero-method, in which zero substitution of parameter estimates is applied to those models from which a parameter is absent (Grueber et al., 2011). Deviance residuals of the ‘top’ models were visually assessed using the model validation protocol of Zuur et al. (2009) to
ensure that model assumptions were met.

All analyses were performed in R (R Development Core Team, 2019), using the lme4 package (Bates et al., 2015) and MASS package (Venables and Ripley, 2002). Model selection and averaging was conducted with the MuMIn package (Barton, 2019). Figures were made using ggplot2 (Wickham, 2016) and mnormal (Pante and Simon-Bouhet, 2013) (Fig. 2).

3. Results

Across 2017 and 2018, we placed a total of 576 egg batches containing 49,979 eggs at 38 sites during four sampling periods. After 48 h exposure in the field, the predation rate was 14 % ± 14 % (mean ± SD; range 0–65 %), and the parasitism rate 6 % ± 12 % (mean ± SD; range 0–67 %). Average predation was 48 ± 47 (mean ± SD) eggs, and average parasitism was 16 ± 34 (mean ± SD) eggs per site per sampling period. We occasionally observed larvae of Chrysopa oculata feeding on eggs, but in most cases predators were not identified. Most parasitism was caused by Trichogramma spp.

3.1. Egg predation and parasitism

The univariate model with woodySNH was the most parsimonious model for egg predation for all three insecticide indicator models (management, TFI and QIA; Table 1). However, the marginal variance explained (marginal R²) by this model was only 5–7 % (Table 1). Competing models that included an insecticide use indicator (management or QIA) explained 8 % and 7 %, respectively (marginal R²; Table 1). For egg parasitism, the null model (intercept only) was the most parsimonious for all three insecticide indicator models (management, TFI and QIA; Table 3).

The model averaging procedure for both egg predation and parasitism provided no support for the hypothesis that predation and parasitism rates are associated with landscape variables and insecticide use (Table 2 and 4; Fig. 2). Model 2A, analysing effects of insecticide indicator TFI on egg predation, showed a significant negative effect of proportion of woody habitat on egg predation (Table 2). However, the sample size was smaller in this model due to missing values for TFI. Local vegetation of the field margins (‘field margin type’) did not come up as a variable in any of the averaged models. All confidence intervals for the parameters estimates included zero, indicating that there is no statistical support for concluding that any of the variables on insecticide use or landscape were associated with egg parasitism and predation (Tables 2 and 4).

Table 1

| Insecticide indicator | Top models | df | NLL | AICc | weight | marginal R² | conditional R² |
|-----------------------|------------|----|-----|------|---------|-------------|---------------|
| Management            |            |    |     |      |         |             |               |
| woodySNH              | 5          | −700.01 | 0.00 | 0.10 | 0.05    | 0.19        |               |
| woodySNH * management | 7          | −698.00 | 0.36 | 0.08 | 0.08    | 0.20        |               |
| woodySNH + propORG    | 6          | −699.45 | 1.04 | 0.06 | 0.06    | 0.19        |               |
| woodySNH + propORG    | 6          | −699.59 | 1.34 | 0.05 | 0.05    | 0.19        |               |
| woodySNH + herbSNH    | 6          | −699.80 | 1.75 | 0.04 | 0.06    | 0.20        |               |
| woodySNH + propCONV   | 6          | −699.83 | 1.82 | 0.04 | 0.05    | 0.19        |               |
| TFI                   |            |    |     |      |         |             |               |
| woodySNH              | 5          | −578.58 | 0.00 | 0.15 | 0.07    | 0.13        |               |
| woodySNH + herbSNH    | 6          | −577.51 | 0.06 | 0.15 | 0.09    | 0.16        |               |
| woodySNH              | 5          | −700.02 | 0.00 | 0.12 | 0.05    | 0.19        |               |
| woodySNH + propORG    | 6          | −699.60 | 1.34 | 0.06 | 0.05    | 0.19        |               |
| QIA                   |            |    |     |      |         |             |               |
| woodySNH * QIA        | 7          | −698.50 | 1.36 | 0.06 | 0.07    | 0.19        |               |
| woodySNH + herbSNH    | 6          | −699.80 | 1.75 | 0.05 | 0.06    | 0.20        |               |
| woodySNH + propCONV   | 6          | −699.84 | 1.82 | 0.05 | 0.05    | 0.19        |               |

NLL: log-likelihood of the model.
AICc: difference in corrected Akaike’s Information Criterion (AICc) value between the best ranked models.
Weight: can be interpreted as the probability being the ‘best’ model (Field et al., 2016).
Marginal R²: variance explained by fixed factors in the model.
Conditional R²: variance explained by fixed and random factors in the model.

4. Discussion

In this two-year study we assessed the effects of insecticide use at a local and landscape scale on biocontrol services in agricultural field margins. We report three key findings. First, there were no significant associations between the predation and parasitism rates of M. brassicae eggs and indicators for insecticide use at the field and landscape scale. Second, there was an indication that woody habitat is negatively associated with egg predation in field margins. Finally, while it has been postulated that more diverse landscapes can compensate for impacts of insecticides on natural enemies (Roubos et al., 2014; Tscharntke et al., 2016), this premise was not supported by our data collected in field margins. Nevertheless, egg predation and parasitism did happen in field margins. Thus, the results suggest that field margins may provide refuge supporting natural enemies of pests, irrespective of the surrounding landscape and local management practices.

Our first hypothesis, postulating that insecticide use in adjacent fields impacts biocontrol services in field margins, was not supported by the data. Specifically, we found that indicators of insecticide use at the local field level were not significantly associated with predation or parasitism rates in field margins. Similarly, Gagic et al. (2019) did not find effects of insecticide spraying on predation of eggs of cotton bollworm in Australian cotton field margins. However, several studies on biocontrol in agricultural fields report that disturbances associated with local farm management (e.g. pesticide use) impacted on-field biocontrol. For example, Jonsson et al. (2012) found that the intensity of insecticide treatments had a negative effect on parasitism rates of two important pests (aphids and Plutella xylostella) in a brassica cropping system in New Zealand. Furthermore, a European scale study by Geiger et al. (2010) showed that local pesticide application had persistent negative effects on biological pest control. Muneret et al. (2019) found that increasing pesticide use decreased removal rates of moth eggs in vineyards, and Yang et al. (2019) conclude that ladybeetle abundance in wheat fields decreased with increasing insecticide use at the local scale. The combination of negative effects of pesticides on-field, but no effects off-field, could imply that in landscapes with high land-use intensities (e.g. more pesticide applications) field margins may be important refuges for natural enemies (Li et al., 2020). Our findings suggest that effects of direct insecticide overspray or drift to field margins are limited or at least below the detection limit of the methods used.

Our second hypothesis, in which we postulated that complex landscapes (with a relatively high proportion of semi-natural habitat) provide better biocontrol than more simple landscapes, was also not
Table 2

| Model | Insecticide indicator | Random effects | Variable | Estimate | Adj SE | t-value | Pr(>|z|) | relative importance |
|-------|-----------------------|----------------|----------|----------|--------|---------|----------|-------------------|
| Farm pair | Management | Sampling period | intercept | 3.82 | 0.19 | 20.20 | 0.00 | 1 |
| woodySNH | -0.14 | 0.19 | 0.73 | 0.46 | 0.38 |
| management | -0.07 | 0.14 | 0.55 | 0.59 | 0.38 |
| propORG | -0.01 | 0.04 | 0.26 | 0.80 | 0.14 |
| herbSNH | 0.01 | 0.03 | 0.18 | 0.86 | 0.11 |
| propCONV | 0.01 | 0.04 | 0.17 | 0.87 | 0.11 |
| Sampling period | woodySNH | Farm pair | -0.26 | 0.09 | 2.86 | <0.01 | 1 |
| herbSNH | 0.06 | 0.09 | 0.69 | 0.49 | 0.18 |
| Farm pair | TFI | Sampling period | intercept | 3.73 | 0.15 | 24.91 | 0.00 | 1 |
| woodySNH | 0.17 | 0.14 | 1.21 | 0.22 | 0.18 |
| propORG | -0.01 | 0.04 | 0.30 | 0.76 | 0.18 |
| QIA | 0.02 | 0.06 | 0.32 | 0.75 | 0.18 |
| QIA + woodySNH | 0.05 | 0.14 | 0.39 | 0.70 | 0.18 |
| herbSNH | 0.01 | 0.04 | 0.21 | 0.83 | 0.15 |
| propCONV | 0.01 | 0.04 | 0.20 | 0.84 | 0.14 |

Table 3

| Insecticide indicator | ‘Top’ model | df | NLL | ΔAICc | weight | marginal R² | conditional R² |
|-----------------------|-------------|----|-----|-------|--------|-------------|----------------|
| Management | Null | 4 | -341.73 | 0.00 | 0.12 | 0.00 | 0.01 |
| woodySNH | 5 | -341.21 | 1.09 | 0.07 | 0.01 | 0.02 |
| management | 5 | -341.36 | 1.40 | 0.06 | 0.01 | 0.02 |
| TFI | Null | 4 | -305.65 | 0.00 | 0.14 | 0.00 | 0.01 |
| woodySNH | 5 | -305.25 | 1.36 | 0.07 | 0.01 | 0.01 |
| propORG | 5 | -305.44 | 1.74 | 0.06 | 0.01 | 0.01 |
| QIA | Null | 4 | -341.70 | 0.00 | 0.13 | 0.00 | 0.01 |
| woodySNH | 5 | -341.18 | 1.12 | 0.07 | 0.01 | 0.02 |
| QIA | 5 | -341.36 | 1.47 | 0.06 | 0.01 | 0.02 |

NLL: log-likelihood of the model.
ΔAICc: difference in corrected Akaike’s Information Criterion (AICc) value between the best ranked models.
Weight: can be interpreted as the probability being the ‘best’ model (Feld et al., 2016).
Marginal R²: variance explained by fixed factors the model.
Conditional R²: variance explained by fixed and random factors in the model.

supported by our data. There was one model that indicated that egg predation was significantly negatively associated with the proportion of woody habitat in the landscape, but for parasitism this relationship was non-significant. Inconsistent effects of woody natural habitats on predation and parasitism have been reported previously (Karp et al., 2018; McHugh et al., 2020) found that herbaceous habitat was positively associated with predation of aphids, while woody linear habitat was negatively associated with predation of aphids. There are also studies reporting positive associations between biocontrol rates and woody elements in the landscape (Bianchi et al., 2008, 2005; Dainese et al., 2017; Dover, 2019). Thus, these effects vary across studies and appear contextual (Karp et al., 2018). The effects of landscape composition on natural enemy communities are species-specific and dependent on species’ traits, such as dispersal capacity, resource requirements, and overwintering sites (Bianchi et al., 2006; Chaplin-Kramer et al., 2011; Rusch et al., 2016; Tscharntke et al., 2005).

Our data do not support the hypothesis that a more diverse landscape can compensate for insecticide-mediated impacts on services provided by beneficial insects (Blitzer et al., 2012; Gabriel et al., 2010; Tooker et al., 2020; Tscharntke et al., 2016). Most of our locations (33 out of 38) were embedded in agricultural landscapes with relatively low proportions of semi-natural habitat (< 5%). In these ‘simple’ landscapes it could be expected that these small amounts of semi-natural habitat would have relatively little effect on the level of biological control (Ekström and Ekblom, 2011; Roubo et al., 2014), let alone counterbalance any negative effect of insecticide use. There are several studies that report on interactions between pesticide use and landscape factors. Yang et al. (2019) found that beneficial insect abundance were low in areas that were sprayed with a frequent insecticide sprays, regardless of the landscape context, while Ricci et al. (2019) reported that negative effects of local pesticide use intensity on beneficial insects was only sometimes modified by landscape characteristics. Gagic et al. (2019) showed that the recovery time of natural enemies after insecticide spraying in cotton fields was not compensated for by the presence of semi-natural field margins. Thus, the majority of the studies report detrimental effects of insecticide use on beneficial insects, although the differences in insecticide impacts in...
Field populations of *M. brassicae* can be controlled by a range of natural enemies, e.g. staphylinids, carabids and parasitoids (Bianchi et al., 2005; Piffner et al., 2009). Our study focussed on the biocontrol potential of flying natural enemies, as ground-dwelling arthropods were most likely not able to climb the pots in which the plants with egg cards were placed. While low predation rates were anticipated due to exclusion of ground-dwelling predators, parasitism rates were lower than expected (6 ± 12%; mean ± SD). Compared to our results, Piffner et al. (2009) report higher levels of parasitism of *M. brassicae* in flower strips with a range between 14%–52%. Rates of parasitism assessed in arable fields were more similar to our findings. For instance, Bianchi et al. (2005) found parasitism rates of egg batches of *M. brassicae* of 14% in cabbage fields, and Balmer et al. (2014) found *M. brassicae* parasitism rates between 5–6.5% in cabbage fields close to flower strips. Many factors affect parasitism levels, e.g. habitat quality, seasonality, and weather (Stilmait et al., 2008). It is not possible to ascertain why the parasitism rate in our study is relatively low when compared to some of these other studies. Furthermore, a limitation of using sentinel prey is that natural enemies could be discouraged by our set up of using potted plants and egg cards, potentially missing chemical cues from the plants. Still, important natural enemy groups of *M. brassicae*, including staphylinids, ladybeetles, lacewings, and spiders, were also found using sweepnet and yellow sticky trap sampling (Bakker et al., 2020).

One of our study limitations was missing information on seed coatings with insecticides. Seed coatings with neonicotinoids were allowed in the EU for non-flowering crops, such as sugar beets, and have been banned for use in the open field as of 2019 in the EU (EU, 2018). Our study was conducted before this ban. We were not able to quantify the use of seed coatings in the focal fields, because this information was not provided by farmers. The impacts of insecticide use on beneficial arthropod communities may depend on many factors, such as application rates, the effectiveness of the active ingredients, their toxicity, and the environment in which products are used (Kniss, 2017), but a systematic overview of data on the use of insecticides is not available or incomplete (Eurostat, 2019; FAO, 2019). This hampers assessment of insecticide use impacts on the environment. There is substantial evidence that systemic insecticides, such as neonicotinoids, have adverse impacts on the environment and beneficial insects (Furlan et al., 2018; Giorio et al., 2017; Pisa et al., 2017). Better accessibility of insecticide use data would therefore enable an improved assessment of effects of insecticide use on biocontrol services.

The absence of an effect of insecticide use on biocontrol services could be explained in at least two very contrasting ways. First, drift and/or overspray of field margins may have been negligible and field margins may therefore have been very little exposed to insecticides. In addition, natural enemies in field margins may have suffered little from insecticide applications in the adjoining field and wider surroundings when their movement between sprayed fields and field margins has been limited. Second, the wide-spread use of insecticides could have impacted natural enemy communities at a scale beyond 1 km radius, masking relationships between insecticide use intensity on biocontrol at the scales considered in our study. Regarding the first explanation, limited insecticide drift to field margins would indicate compliance to good agricultural practices, such as using crop-free buffer strips and usage of spray-drift-reducing nozzles (van Eerdt et al., 2014). Concerning the second explanation, it could be that pesticide drift from conventional farms to non-target areas, including organic farms, may have disrupted biocontrol in these non-target areas, and diluted potential effects of insecticide use on biocontrol services. In support of this explanation, de Jong et al. (2008) estimated that 11% of the natural areas within agricultural landscapes were affected by herbicide drift in the Netherlands in 2005.

Considering the wide-spread use of agrochemicals and potential pathways for run-off to non-target areas (Chagnon et al., 2015; de Jong et al., 2008; Goulson, 2013), it could be that organic agricultural fields might have been impacted by practices of conventional agriculture, masking the effects we expected to find comparing margins of conventional and organic fields. While drift or overspray may be limited, insecticide residues may still be present in the soil and water for a period of time, e.g. half-lives of neonicotinoids in soils can be longer than 1000 days (Bonmatin et al., 2015). Sublethal toxic effects and reduced availability of food resources may affect reproduction and survival of non-target species (Desneux et al., 2007; Devine and Furlong, 2007). These can be subtle effects that are nevertheless relevant on population level processes, but are not captured by short-term experiments (Woodcock et al., 2016b), and could possibly have masked effects of insecticide use on non-target organisms in field margins.

Modelling studies indicate that insecticide use in fields affect natural enemy populations in non-sprayed fields because insecticide treated fields act as sink habitats and diminish the population size of mobile natural enemies at the landscape scale (Bianchi et al., 2013a; Topping et al., 2015). Thus, the impacts of insecticide use may occur at larger scales than what we considered in our study. Indeed, recent analyses of dramatic insect declines in protected nature areas, on a national and global scale, associated these negative trends to land use intensification at the landscape scale (Hallmann et al., 2017; Seibold et al., 2019; van Klink et al., 2020). Therefore, while we do not find evidence for
relationships between indicators for insecticide usage at field and landscape level and biological control in field margins, our data cannot exclude the possibility that such relationships may exists.

5. Conclusion

This study aimed to unravel the effects of (local) insecticide use and landscape complexity on biocontrol potential, but in contrast to our expectations we did not find consistent effects of insecticide use or landscape composition on egg predation and parasitism of *M. brassicae*. These findings imply that field margins could act as habitats supporting natural enemies in agricultural landscapes, regardless of the land-use intensity at the local and landscape scale. This suggests that with ongoing intensification of agricultural landscapes, field margins could become an increasingly important habitat for natural enemies, e.g. as refuge habitat or corridors in simplified landscapes. Furthermore, field margins should be protected from adverse agricultural practices, e.g. exposure to agrochemical drift or frequent mowing. Data on insecticide
use in landscapes are required to improve environmental impact assessments but are not available for the study area because they are considered private even though the potential impacts affect public goods. Studies on insecticide usage and impacts might benefit from a landscape perspective because spill-over mechanisms are likely, particularly due to the movement of insects at a landscape scale. Finally, our study did not confirm broadly supported hypotheses, and highlights that interactions between landscape and insecticide use are complex and merit further research to obtain a better understanding of the underlying mechanisms and relevant spatial scales.

Declaration of Competing Interest

The authors report no declarations of interest.

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Supplementary data

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