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Pesticides and land cover heterogeneity affect functional group and taxonomic diversity of arthropods in rice agroecosystems

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Abstract

Biodiversity can be characterised across several dimensions, which are crucial for the evaluation of ecosystem services. Functional diversity, a key aspect of biodiversity, provides a more realistic characterisation of the functioning of ecological communities than only studying their taxonomic diversity. The relevance of functional ecology studies has steadily increased in agroecosystems. However, the combined effects of pesticides and land cover heterogeneity on the taxonomic and functional diversity of arthropod communities have been studied less frequently. We sampled arthropods during the dry season in 19 rice fields located in two different regions of Northern Vietnam. We assorted the arthropods into functional groups corresponding to different feeding habits and calculated the taxonomic and functional group diversities. Finally, we analysed the impacts of pesticide applications and land cover heterogeneity on both diversity measures. Taxonomic and functional group diversity measures were highly correlated. In turn, both diversity measures responded similarly to land cover heterogeneity and pesticides. Land cover heterogeneity had positive effects on taxonomic and functional group diversity, mainly at the early stage of rice crops. Conversely, the impact of pesticide application on both diversity measures was strongly negative. Our results suggest that rice agroecosystems can be more sustainable by increasing landscape heterogeneity and a reduced pesticide use. Such schemes may help to maintain higher levels of biodiversity that ensure ecosystem functioning, which will be therefore likely beneficial to provide ecosystem services in agroecosystems.

Keywords: Arthropod communities, diversity dimensions, land use intensity, natural enemies, Northern Vietnam, rice agroecosystems

1. Introduction

Multiple hierarchical assembly processes have been shown to influence biodiversity patterns across spatial scales. At broad spatial scales, for instance, dispersal among habitat patches is key to the maintenance of diversity patterns in a metacommunity context (Hassan et al., 2016; Oliver et al., 2010). At more local scales, abiotic filtering and biotic interactions play an important role (De Bello et al., 2009; Gianuca et al., 2017; Kraft et al., 2015). Increasing human impacts may alter such community assembly processes across scales and may lead to significant biodiversity changes (Naeem et al., 2012). For instance, reduced connectivity in fragmented landscapes influences dispersal rates among populations and communities. Likewise, environmental degradation imposes a strong local filter on species traits and may result in depauperated local communities (Brudvig et al., 2015; Haddad et al., 2015). Due to well-established universal scale-dependence of biodiversity processes and patterns, it is ever more recognised that community level analyses benefit from being conducted at multiple spatial scales, especially if we want to correctly inform management and conservation decisions (Hendrickx et al., 2007).

Diversity partitioning techniques have traditionally been applied to taxonomic-based approaches, which are based on species identities but disregard their functional differences (Hooper et al., 2002). It has been proposed that accounting for functional differences among different ecological groups (Cardoso et al., 2014; Naeem et al., 2012) can provide a more direct link between organisms and ecosystem processes (Cadotte, 2017; McGill et al., 2006). In addition, phylogenetically distantly related organisms can have similar functionality due to convergent evolution (Díaz et al., 2013). Therefore, focusing directly on functional diversity instead of taxonomic or phylogenetic diversity can provide deeper insights into community assembly (Cardoso et al., 2014; Hooper et al., 2002) and ecosystem functioning (De Bello et al., 2010).

Land cover heterogeneity can support different ecological mechanisms, which have positive effects on community structure (Oliver et al., 2010; Papanikolaou et al., 2017a, 2017b). For example, higher landscape heterogeneity provides more opportunities for niche partitioning

(Amarasekare, 2003; Yang et al., 2015). Especially in agroecosystems, diverse land cover types can act for different organisms as refuge (Gurr et al., 2017), food source (Westphal et al., 2015) and overwintering site (Duflot et al., 2015). Habitat heterogeneity may consequently increase both taxonomic and functional diversity.

Land-use intensification is amongst the major drivers of biodiversity change (Foley et al., 2005; Laliberté et al., 2010). It mostly results in a simplification of landscapes (e.g. large amount of single, homogeneous types) and high external inputs (e.g. pesticides) (Foley et al., 2005; Laliberté et al., 2010). This, in turn, leads to a loss of species in important groups, like arthropods in agroecosystems (Hendrickx et al., 2007; Lingbeek et al., 2017), that provide essential ecosystem services like biocontrol (Gurr et al., 2012, 2011). Yet, some studies have demonstrated uncoupled responses of taxonomic and functional diversity to common environmental drivers (e.g. De Palma et al., 2017), but it is not entirely clear how land-use intensification, pesticide application, and habitat homogenization simultaneously influence these two biodiversity measures (Mayfield et al., 2010; Peco et al., 2012). Some studies have suggested that land-use intensification can lead to a reduction of functional group richness, whereas taxonomic diversity can be more resilient against external disturbances (Schweiger et al., 2007). This may happen when species with unique functional traits get replaced by functionally redundant species (Ernst et al., 2006; Teresa and Casatti, 2012; Villéger et al., 2010). Consequently, measuring functional diversity in addition to taxonomic diversity can help to understand the full dimension of how anthropogenic land use and its change impacts biodiversity (Ernst et al., 2006).

Here, we analyse how pesticides and land cover heterogeneity impacts both arthropod taxonomic and functional group diversity. We focus on areas in Vietnam, where the intensification of rice cultivation constantly increased during the last decades (Schreinemachers et al., 2015). So far, the effects of pesticides and land cover heterogeneity on taxonomic and functional group diversity of arthropods in rice agroecosystems in Vietnam remain elusive and are poorly understood. Given the likely strong impacts of land-use intensification on biodiversity and the potential uncoupled patterns of taxonomic and functional group diversity, we hypothesise that increasing pesticide usage negatively influences functional group diversity, although it may have a smaller effect on taxonomic diversity. Furthermore, we hypothesise that higher land cover heterogeneity positively affects taxonomic and functional group diversity.

2. Material and Methods

2.1. Study area

This study was part of the research project LEGATO (Land-use intensity and Ecological enGineering-Assessment Tools for risks and Opportunities in irrigated rice based production systems; Settele et al., 2018) and was carried out in 19 rice fields located in two rice dominated lowland regions along the Red River Delta in Northern Vietnam. In these areas, rice is the main crop (Global Rice Science Partnership, 2013). The Red River Delta is characterised by a warm, humid, and subtropical climate (Klotzbücher et al., 2015) with a distinct seasonality with two growing seasons per year. The first season ranges from February to May and the second from July to October (Klotzbücher et al., 2015).

The first region, Hai Duong (LEGATO region VN1: 21°00'N 106°23'E), is situated 60 km east of Hanoi. The region is heavily industrialised and dominated by intensively farmed rice fields. We originally selected ten rice fields but in one of these fields the cropping system changed during the investigation period and we removed it from the analysis.

The second region, Vinh Phuc (LEGATO region VN2: 21°20'N 105°43'E), is located 35 km northwest of Hanoi. Similarly to Hai Duong, the landscape is dominated by rice fields but is industrialised to a lesser extent (Burkhard et al., 2015). In this region, we selected ten rice fields.

The average distance between rice fields was 338 m within a region. The size of the rice fields was on average 491 m^2 ranging from 97 - 1883 m^2 (Appendix: Table A.1). All investigated rice fields were sprayed with pesticides and fertilised using chemically produced NPK (Nitrogen, Phosphorus, Potassium) fertiliser during our study (Klotzbücher et al., 2015). In Hai Duong, farmers use rice varieties that are highly productive, whereas in Vinh Phuc farmers use traditional varieties with higher genetic diversity (Burkhard et al., 2015). Similar to Wilby et al. (2006) and Dominik et al. (2017), all observations and investigations were implemented in real agricultural settings without controlling external factors. The decisions about agricultural practices, like fertiliser use, weeding, pesticide application, and the choice of rice varieties were left to the farmers.

2.2. Arthropod sampling and assignment

We collected arthropods during the dry season from March to April in 2015 using blow vac and sweep net. Both are highly effective standard methods to sample arthropods or specific taxa in rice agroecosystems (Bambaradeniya et al., 2004; Gangurde, 2007; Ghahari et al., 2008; Schoenly et al., 2010). We sampled arthropods during the vegetative stage of the rice plant at two points in time: 35 and 50 days after the rice seedlings were transplanted into the fields (days after transplanting = DAT). Sampling times are in accordance with the overall LEGATO sampling design in order to standardise methodologies. For each sampling time, we took five replicates per method in each rice field. For sweep net, we sampled arthropods at two locations along the rice bunds and at three locations within the centre of each rice field. For each sample location, we sampled an area of 30 m^2 while walking with a speed of approximately 0.5 m/sec and performing 30 sweeps. If the size of a rice field was too small to locate five sample units, we took the remaining units in the immediate vicinity (i.e. in the neighbouring field). Sample units did not overlap with one another. With the blow vac method, we sampled an area of 0.25 m^2 in five randomly chosen locations within a rice field using a square plastic enclosure fitted with nylon net on top to prevent arthropods from escaping.

Arthropods were counted, identified to family level, and assigned to functional groups, which are based on similar functional behaviour and food acquisition strategies. We used the following functional groups defined after Shepard et al. (1995, 1987) and Heong et al. (1991): predators, parasitoids, herbivores, decomposers (detritivores and scavengers), and fungivores. Arthropod samples that could not be assigned to one of these functional groups, due to the samples' poor condition, or could be assigned to more than one group (a total of 0.001% of all collected arthropods) were excluded from further analysis.

2.3. Predictor variables

2.3.1. Land cover heterogeneity

To investigate the relationship between land cover heterogeneity and arthropod taxonomic and functional group diversity, we recorded and classified the land cover within a 300 m buffer around the centre of the rice field. Land cover was classified based on digital habitat mapping by satellite images following Burkhard et al. (2015). Land cover was classified into the following ten types: bare soil, forest, fruit, meadow/grassland, rice field, vegetable, water, crops, compacted surface and sealed surface (see Appendix: Table A.1 and A.2 for more information). We calculated the Shannon diversity (H') index to measure land cover heterogeneity based on the proportion of land cover types (in %) within the 300 m buffer.

2.3.2. Pesticides

To investigate the relationship between pesticides and arthropod functional and taxonomic groups, we assessed the number of all pesticide applications combined and of insecticide applications separately by interviewing farmers. Farmers used herbicides, fungicides, molluscicides, rodenticides and insecticides which are here summarised as pesticides. Detailed information about the used pesticides can be found in Sattler et al. (2018). Farmers sprayed pesticides on average four times (ranging from four to six) during the cropping season and all farmers sprayed pesticides immediately after transplanting the rice plants (Sattler et al., 2018).

2.4. Data analysis

For statistical analysis, we used abundance data of either the functional groups or the taxonomic units of arthropod communities of each sample unit per rice field from both sweep net and blow vac sampling (five replicates per method, field and at 35 and 50 DAT). Diversity indices were calculated using Shannon entropy (H', Shannon-Wiener index). Due to the difficult interpretation of most standard indices (Jost, 2007; Marcon and Hérault, 2015), we converted the Shannon entropy to the 'effective number of species' by taking its exponential (Jost, 2007, 2006), which is referred to as 'effective number of functional or taxonomic groups'. For simplification, the term 'diversity' will be used instead of 'effective number of functional groups or taxonomic groups'.

We calculated the local (= alpha) diversity of each rice field (Marcon and Hérault, 2015). Alpha diversity was defined as the diversity of functional or taxonomic groups of all specimens found in one rice field. We also partitioned diversity of arthropod communities at the landscape scale, whereby gamma was defined as the diversity of functional or taxonomic groups of all specimens found in one region, alpha as the local communities within a rice field (i.e. aggregated across the sampling units) of one region and beta as the variation of taxonomic and functional compositions among the rice fields within a region. Scale definitions (local and landscape scale) are according to Willis and Whittaker (2002).

Data of taxonomic and functional group diversity were normally distributed (tested with the Shapiro-Wilk test; Royston 1982). To analyse the relationship of pesticide applications (for both all pesticides and insecticides only) and land cover heterogeneity with the calculated taxonomic and functional group diversity we used linear mixed effects models. To control for potential non-independence of the data points, we included rice fields nested within region as random effects. The number of pesticide applications and land cover heterogeneity were not collinear (Pearson correlation $r=0.12$) and thus included as fixed effects in the models. For these analyses, only the alpha diversity at the local scale was used because of too few data points at the landscape scale for beta and gamma diversity (only one data point) per region. Model selection followed a multimodel inference approach relying on the second-order Akaike information criterion (AICc; Burnham and Anderson 2002) and only the best model was selected. Candidate models included as predictor variables: number of pesticide applications and land cover heterogeneity. The variable importance was computed based on the sum of AICc weights for each model in which the predictor variable appeared.

In total, we performed eight linear mixed effects models separated by methods (sweep net and blow vac), sampling days (35 and 50 DAT), and for both functional and taxonomic measures. We calculated for each model the marginal and conditional R-squared values. To compare the effect size (calculated marginal R^2) of pesticides on taxonomic and functional group diversity, we used the Paired Student's t test as data were normally distributed (tested with the Shapiro-Wilk test). Pairs were separated into methods and sampling days. Paired Student's t test was also used to compare the taxonomic and functional group diversity between the two sampling days (35 and 50 DAT). All analyses were performed in the statistical environment R for Windows (Version 3.2.4; R Core Team 2016) using the packages 'lmer4' (Bates et al., 2015), 'MuMIn' (Bartoń, 2016), 'vegan' (Oksanen et al., 2017), 'ggplot2' (Wickham, 2015), 'entropart' (Marcon and Hérault, 2015), and 'lmerTest' (Kuznetsova et al., 2017).

3. Results

Overall, we collected 164,671 arthropod specimens belonging to 17 orders and 77 identified families. In all functional groups the highest number of specimens was sampled by sweep netting, with 151,585 specimens. Blow vac sampling yielded a total of 13,086 specimens. We collected 73 different families with the sweep netting method and 57 families with the blow vac method. A comparison of collected functional groups by the two sampling methods can be found in Fig. 1. Arthropods were dominated by the class Insecta with 99.8%. The class of Arachnida was present with 0.2% and one specimen of Diplopoda was found. Decomposers were the most abundant functional group with 132,662 specimens followed by herbivores with 19,697 specimens, predators with 7,321 specimens, parasitoids with 4,720 specimens, and fungivores with 58 specimens. In total, 213 specimens could not be assigned into a specific functional group and were excluded from further analysis. A list of all collected arthropods can be found in the Appendix (Table A.3).

Generally, taxonomic and functional group diversity were highly correlated (Pearson correlation r = 0.94). On the landscape scale, we found very low beta diversity among the rice fields for both taxonomic and functional group diversity (see Table 1 for landscape scale).

We found higher functional group diversity at 50 DAT compared to 35 DAT for both sampling methods (t-test; Blow vac: t = -3.87, df = 18, p-value = 0.001; sweep net: t = -4.04, df = 17, p-value = 0.0008). A similar result was found for taxonomic diversity (t-test; Blow vac: t = -2.87, df = 18, p-value = 0.01; sweep net: t = -2.52, df = 18, p-value = 0.02).

3.1. Relationship of land cover heterogeneity and pesticides with functional and taxonomic group diversity at local scale

Given the high correlation between functional and taxonomic diversity, the impacts of pesticides and land cover heterogeneity were highly similar for both diversity measures. Alpha diversity decreased with an increasing number of pesticide applications consistently across sampling methods and dates (Fig. 2a, b, c, e; Fig. 3a, b, c, e). However, an effect of land cover heterogeneity on alpha diversity was only evident for communities sampled with sweep nets at 35 DAT where diversity increased with increasing land cover heterogeneity (Fig. 2d; Fig 3d). Linear mixed effects models at local scale can be found in Table 2 and Table 3. The global models which include all predictor variables can be found in the Appendix (Table A.4, Table A.5 and Figure A.1).

The effect size (marginal R^2) of pesticides on taxonomic and functional group diversity showed no differences based on the Paired Student's t test ($p>0.05$). Therefore, the relationship of pesticides with both diversity dimensions (taxonomic and functional group diversity) was negative to a similar extent. The number of insecticide applications alone did not result in significant effects for both taxonomic and functional group diversity. Results of global models of insecticide applications and land cover heterogeneity can be found in the Appendix (Table A.6 and A.7).

Table 1 Taxonomic and functional group diversity (mean of alpha, beta, and gamma diversity) at landscape scale. Functional and taxonomic alpha (α), beta (β), and gamma (γ) diversity for Hai Duong (VN1) and Vinh Phuc (VN2), separated for blow vac (BV) and sweep net (SN) as well as for sampling days (35 and 50 days after transplanting).

Method	Reg	α (functional)	β (functional)	γ (functional)	α (taxonomic)	β (taxonomic)	γ (taxonomic)
BV 35	VN1	2.16	1.02	2.20	4.14	1.35	5.57
	VN2	2.04	1.05	2.15	3.46	1.32	4.58
BV 50	VN1	2.82	1.08	3.06	5.65	1.41	7.98
	VN2	2.48	1.18	2.94	4.93	1.54	7.61
SN 35	VN1	1.83	1.02	1.87	2.47	1.07	2.65
	VN2	1.72	1.05	1.81	2.24	1.09	2.45
SN 50	VN1	3.18	1.05	3.32	5.53	1.18	6.52
	VN2	2.08	1.03	2.13	2.86	1.10	3.15

Table 2 Linear mixed effects models of functional group diversity at local scale (alpha diversity). The best model was selected among the candidate models following a multimodel inference approach separately for blow vac (BV) and sweep net (SN) as well as sampling days (35 and 50 days after transplanting). R–squares are shown as marginal R–squared values (R^2m) and conditional R–squared values (R^2c).

Model	Response variable	Predictor variable	p-value	Variable importance	R^2m	R^2c
BV 35	alpha diversity	Number of pesticide applications	0.005	0.59	0.31	0.52
BV 50	alpha diversity	Number of pesticide applications	0.02	0.43	0.26	0.26
SN 35	alpha diversity	Number of pesticide applications	0.003	1	0.46	0.46
		Land cover heterogeneity	0.05	1		
SN 50	alpha diversity	Number of pesticide applications	0.05	0.29	0.18	0.55

Table 3 Linear mixed effects models of taxonomic diversity at local scale (alpha diversity). The best model was selected among the candidate models following a multimodel inference approach separately for blow vac (BV) and sweep net (SN) as well as sampling days (35 and 50 days after transplanting). R–squares are shown as marginal R–squared values (R^2m) and conditional R–squared values (R^2c).

Model	Response variable	Predictor variable	p-value	Variable importance	R^2m	R^2c
BV 35	alpha diversity	Number of pesticide applications	0.01	0.69	0.27	0.49
BV 50	alpha diversity	Number of pesticide applications	0.03	0.71	0.25	0.25
SN 35	alpha diversity	Number of pesticide applications	0.008	0.68	0.37	0.5
		Land cover heterogeneity	0.09	0.3		
SN 50	alpha diversity	Number of pesticide applications	0.05	0.3	0.05	0.89

4. Discussion

4.1. Effects of pesticides and land cover heterogeneity on taxonomic and functional group diversity

Beta diversity at the regional scale was consistently low for both sampling methods and sampling days. This indicates that arthropod communities do not differ much among the different rice fields of a region and that local arthropod diversity is largely defined by the regional species pool. However, the observed level of variation among the rice fields in a region was strongly related to the number of pesticide applications indicating that both taxonomic and functional group diversity of arthropod communities decrease with pesticide applications in the rice fields. Insecticide applications alone did not affect taxonomic and functional group diversity, which shows that the combination of all applied pesticides have stronger effects on the taxonomic and functional group diversity than insecticide applications only. This was similarly reviewed by Wu et al. (2020), who pointed out that insecticides and the combination of pesticides can lead to two different effects: acute vs. chronic pest resurgence. The acute resurgence is caused by a higher sensitivity of natural enemies to insecticides compared to pest species. Consequently, the higher mortality of natural enemies stimulates pest reproduction. Chronic resurgence of pests, on the other hand, emerges if a combination of pesticides has smaller effects on natural enemies but positively induces pest reproduction at longer latency. Since we did not find evidence for an insecticide-only effect but arthropod diversity rather reacted to all pesticides in combination, arthropod communities are likely more affected by chronic pest resurgence in our study region.

Heterogeneous land cover in the areas surrounding rice fields was associated with an increase in functional and taxonomic arthropod diversity. However, this effect was only found in the early rice growth stage and only when using the sweep net sampling method.

Taxonomic and functional diversity can be closely connected but do not necessarily need to correlate (Cardoso et al., 2014; Mayfield et al., 2010). Flynn et al. (2009) studied the effect of land-use intensity on mammals, birds, and plants. Similar to our study, both species richness and functional diversity declined with land-use intensity. However, Peco et al. (2012) studied the effect of grazing abandonment on functional and taxonomic diversity of grasslands and found a loss of functional diversity rather than species richness. In a study by Schweiger et al. (2007), increasing land-use intensity led to decreasing functional richness of hoverfly communities rather than affecting species richness. Villéger et al. (2010) showed contrasting responses of biodiversity in aquatic ecosystems influenced by habitat degradation: functional diversity of fish was negatively affected whereas fish species richness increased. In our study, taxonomic and functional group diversity were highly correlated, which is reflected by similar responses to pesticides and land cover heterogeneity. Taxonomic and functional group diversity consistently showed a negative response to the number of applied pesticides regardless of the sampling method. Our hypothesis that increasing pesticide use negatively affects functional group diversity can be confirmed. However, we cannot confirm our hypothesis that increasing pesticide usage would have a smaller effect on taxonomic diversity than on functional group diversity, since the effects were similarly strong for both taxonomic and functional group diversity. This might be either because each species is broadly functionally unique or because of the non-independence in our two diversity metrics. Despite of the potential non-independence of the two diversity metrics, we expected different results if a single or few functional groups had been lost but compensated by an increasing number of species from different taxonomic groups, therefore reducing the impact on taxonomic diversity while still affecting functional group diversity. This mechanism was shown by Ernst et al. (2006), who found a negative effect of forest degradation on functional diversity of amphibians but no effect on taxonomic diversity.

Furthermore, we hypothesised that land cover heterogeneity would increase taxonomic and functional group diversity. In general, we only found effects of land cover heterogeneity with sweep net data at 35 DAT. A study by Wilby et al. (2006), which focused on similar questions, found under 'real' agricultural conditions that arthropod species diversity in rice fields generally decreases with a decrease in structural diversity in the surroundings, which is in line with our results for 35 DAT using the sweep net sampling method. One reason that we did not find a similar effect for blow vac data could be that the probabilities of sampling particular species groups likely differ between both methods. Indeed, sweep net data contained a higher abundance of different taxonomic and functional groups compared to blow vac samples. This might increase the likelihood to cover communities that are more responsive to habitat heterogeneity.

Taxonomic and functional group diversity and abundance can change with crop age (Wilby et al., 2006) and these different communities might respond differently to land cover heterogeneity depending on whether they migrate from somewhere else in the landscape at a given point in time. For instance, early arriving arthropod groups mainly immigrate into the rice fields from the surroundings (Settle et al., 1996; Wilby et al., 2006), and thus fine-scale land cover heterogeneity seems to benefit these groups, as our results have shown. Dominik et al. (2018) showed that the effects on a very local scale can be important: bunds (levee of terrestrial area surrounding the fields) build an extensive network connecting the rice fields. Such bunds often have sparse vegetation that can potentially offer alternative food resources or refuge to natural enemies (Way and Heong, 1994) and likely facilitate the ability of rice arthropods to move through the rice agroecosystem (Sigsgaard, 2000; Yu et al., 1996; see Settele and Settle, 2018 for further discussion).

Nevertheless, such an effect might have diminished at a later stage of rice plants (50 DAT) when arthropod composition changes. Dominik et al. (2017) found no effect of fine-scale landscape heterogeneity on assemblage structure of arthropod communities in rice fields in the Philippines. They argue that regional-scale effects like climate conditions, elevation and landscape structure at broader scale might be more important than fine-scale effects. Another reason for the diminishing effect of land cover heterogeneity on arthropod diversity at the later stage might be an increasing pesticide application in the surroundings. Many of the rice fields in our study were surrounded by fruit and vegetable fields which can suffer from even higher pesticide applications compared to rice fields (Hoi et al., 2016; Van Mele et al., 2002). This means that not only pesticide application within rice fields influences the arthropod communities in the rice fields, but also the application of pesticides in the surrounding non-rice habitats may counteract the positive effects of land cover heterogeneity. Thus, management practices in the surrounding land-use types might be important drivers of the diversity in rice agroecosystems; an important research question for future studies.

4.2. Outlook and Conclusion

Rice ecosystems depend on multiple functions related to multiple ecosystem services, such as pest control by predators and parasitoids or nutrient cycling mediated by decomposers (Schmidt et al., 2016). A decline of functional diversity can lead to a loss of ecosystem services (Villéger et al., 2010). Also, a change in taxonomic composition can impact ecosystem processes as even single species can hold key functions necessary for a stable ecosystem (Chapin et al., 2000; Hooper et al., 2002). Therefore, the maintenance of taxonomic and functional group diversity is important for rice agroecosystems. However, our study showed that high levels of pesticide applications lead to a reduction of the two diversity dimensions, while land cover heterogeneity can have a positive effect. Gurr et al. (2011) showed that land cover heterogeneity in rice-based landscapes can be improved by ecological engineering. For instance, when rice bunds are planted with flowering plants they can provide additional food sources, such as nectar and pollen, as well as shelter for arthropods from various functional groups (Hassan et al., 2016). Further studies might focus on whether land cover heterogeneity and pesticides have an effect on single functional groups and if there is a shift in rice yield when reducing pesticides and increasing landscape heterogeneity. Previous studies successfully introduced ecological engineering to farmers in the Mekong Delta (e.g. Heong et al. 2014; Le 2014), but field studies are mostly local and there is no law enforcement to implement ecological engineering by farmers. To obtain more sustainable farming practises in rice agroecosystems, approaches like ecological engineering should be more in the focus of research studies.

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Fig. 1. Total abundance of functional groups. Total abundance (log-transformed) of decomposers, herbivores, parasitoids, and predators sampled by using blow vac (BV) and sweep net (SN) at 35 a) and 50 b) days after transplanting (DAT).

Fig. 2. Functional group diversity: Relationship between the number of pesticide applications (a-c, e) and land cover heterogeneity (d) for blow vac (BV) and for sweep net data (SN) at 35 and 50 days after transplanting. Alpha diversity is based on the exponential Shannon entropy and expressed as effective numbers of functional groups. Pesticides were measured as number of pesticide applications. Land cover heterogeneity is based on the proportion of habitat types and was calculated with the Shannon index (H').

Fig. 3. Taxonomic diversity: Relationship between the number of pesticide applications (a-c, e) and land cover heterogeneity (d) for blow vac (BV) and for sweep net data (SN) at 35 and 50 days after transplanting. Alpha diversity is based on the exponential Shannon entropy and expressed as effective numbers of taxonomic groups. Pesticides were measured as number of pesticide applications. Land cover heterogeneity is based on the proportion of habitat types and was calculated with the Shannon index (H').