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Manipulating hedgerow quality: Embankment size influences animal biodiversity in a peri-urban context

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ABSTRACT

Hedgerowsare important features within urban, peri-urban, and agricultural habitats because they shelter most of the biodiversity in a landscape dominated by infrastructures or a monoculture. Hedges are characterized by their vegetative cover but also by their base, notably the breadth of the embankment and the various microhabitats made by stones, coarse woody debris, and leaf litter. These features determine the availabilities of arboreal and ground refuges. Their respective roles on biodiversity remain poorly explored. We experimentally manipulated the size of the embankment in newly-constructed hedges in a peri-urban context. We used non-lethal rapid biodiversity assessments and functional indices (accounting for body mass, trophic level, and metabolic mode) to monitor the presence of a wide range of animal taxa. We observed a positive effect of embankment size on animal biodiversity. Various elements of the fauna (e.g. arthropods, reptiles) rapidly colonized newly-constructed hedges provided with an embankment. Guidelines to restore hedgerows should consider embankment size and quality. Both of these features can be improved by simply retaining the materials that are extracted when establishing agricultural plots such that a diversity of microhabitats and ground refuges become available.

1. Introduction

Urban sprawl and transport infrastructure expansion are leading causes of forest fragmentation and habitat alteration, and the concomitant loss of biodiversity (Wilcox and Murphy, 1985; Savard et al., 2000; Seto et al., 2012). Furthermore, conversion of forest habitats to agricultural use has yielded more than 1.5 billion ha that are currently cultivated, representing > 10% of the surface of the planet and more than 36% of the land surface (Bruinsma, 2003). It has been estimated that an additional 2.7 billion ha of forests might be progressively converted for crop production in the coming decades (Van Vliet et al., 2017).

Certain types of anthropogenic modifications of the landscape can be beneficial to the wildlife (Fahrig et al., 2011; Pe'er et al., 2014). Natural or managed forests offer refuges for many organisms in highly altered urban and agricultural landscapes (Savard et al., 2000; Alvey, 2006). Yet the space available is strongly constrained by infrastructures (buildings, roads, etc.). Many urban forests are linear, bordering roads, parks or rivers (Faiers and Bailey, 2005). The benefits of urban forests to wildlife inhabitants depend on the connectivity among patches; corridors shelter more biodiversity compared to isolated parcels (Mörtberg and Wallentinus, 2000). Linear forests provide essential systems of exchange between peri-urban areas and inner zones of cities, especially alongside rivers and railways (Varet et al., 2013). The benefits for biodiversity and human welfare that stem from promoting urban forest networks connected to surrounding habitats are now implemented into urban planning strategies (Goddard et al., 2010).

Hedgerows (i.e. linear forests) shelter most of the biodiversity in agricultural and urban landscapes, and they contribute to spatial and structural heterogeneity (Burel, 1996). Trees are the most salient part of hedgerows, but previous investigations of the value of this habitat also considered bordering herbaceous strips and connectivity with other habitats (Hinsley and Bellamy, 2000; Moonen and Marshall, 2001; Bailey, 2007). Little attention has been paid to the base of the hedges however, especially the embankment: stones, coarse woody debris, tree roots form a complex matrix of burrows and refuges (Lecq et al., 2017). These structures offer microhabitats for a wide range of organisms and substantially contribute to species richness (Lecq et al., 2017). Moreover, complex interactions exist among species and many animals routinely shuttle between ground shelters and the tree cover above (Ctifl, 2000). Unfortunately, the parameters of the embankment are typically not accounted for in planting or management guidelines for

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hedgerows.

We focused on the embankments of hedgerows. Large trees have broad bases that provide abundant ground refuges and, as such, the contributions of tree cover to the biodiversity of a given hedgerow are not easily dissociated from embankment size. For this reason, our experiment manipulated the basal structure of hedges independently from vegetative cover. Hedgerows were installed in a meadow connected to an agricultural landscape and a small city. Three types of embankments were created and we sampled animal biodiversity during two years. To examine the influence of embankment quality on biodiversity, we used a non-lethal survey method to limit the impact of sampling on our dependent variable, both for ethical reasons and to encompass a wide range of taxa (i.e., from small invertebrates to mammals; Lecq et al., 2015).

The primary goal of our study was to examine specific features that can improve hedgerow quality with minimal impacts on managers and farmers in terms of costs and labor. We addressed two questions: (1) Does the presence of an embankment promote animal biodiversity following the installation of a hedgerow? (2) Does embankment size contribute to that measure of diversity?

2. Material and methods

2.1. Study site

The experiment took place in western central France (46°07′21″N, 0°21′24″W) in a typical peri-urban landscape that has been modified extensively in the past several decades (Baudry and Jouin, 2003). Traditional agriculture has been replaced by intensive practices while urbanization developed along with an expanding infrastructure. At the study site, approximately 50% of the hedges have been removed (Fig. 1) and most residential areas were constructed recently (e.g., within the last 25 years; almost all properties containing dwellings that are visible in Fig. 1).

2.2. Experimental design

In February 2011, nine hedges were created in a rectangular grass meadow (115 × 80 m) with the main axis oriented ∼30 °N. The southern margin of the meadow was bordered with a 900 m-long hedge (35 m width) connected to a village in the east and to a forest in the west (600 m distant). The northern margin was adjacent to a working area used for gravel storage and to a road. The two other margins were adjacent to a cultivated meadow (west) and to an athletic field (east). Prior the experiment, the meadow was regularly mowed and no trees or bushes were present.

The nine hedges were oriented west-east in order to present one side to maximal sun exposure. They were regularly spaced (10 m) and each measured 60 m in length. Each hedge was planted with 61 young trees (< 1.5 m in total height) representing species that occur locally (e.g. *Carpinus betulus, Corylus avellana, Prunus spinosa, Crataegus monogyna* in alternation). We constructed each hedge using one of three types of embankment size: minimal, small and large. The minimum base (MB; n = 3 hedges) lacked any sort of embankment; thus, the surface was level with the existing grade. Each tree was planted directly in the ground. Each hedge having a small base (SB; n = 3 hedges) included a small (1.00 m wide, 0.75 m high) embankment constructed using earth and small stones. We planted the trees on the top of the ridge formed by the embankment. Hedges having a large base (LB; n = 3 hedges) differed only in the size of the embankment (1.50 × 1.20 m), and trees were again planted at the top of the embankment ridge. The volume of material used to construct the LB hedgerows was twofold greater than the amount needed for the SB type. In addition, we placed several stones (∼40 × 40 cm) on the south slope in order to cover ∼5% of the ground surface of the LB hedges. The embankments were not compacted. The three types of embankment correspond to the most widespread and traditional basal structures found among the hedgerows in the study area. Overall, we adopted a simple and realistic approach by selecting easily-built structures. In practice, the dimensions of the SB and LB base types corresponded to the amount of material that can be excavated with a backhoe loader during replanting programs. In areas of our study site where the arable soil is relatively thin, the underlying marly-calcareous layers must be broken up before plantation. The three types of hedges were placed in semi-random order, avoiding a configuration that placed two of the same hedge type next to each other (Fig. 2). This design enabled us to focus on the effect of the size of the embankment.

Following the construction of the hedges (February 2011), the area was not managed nor did we monitor the herbaceous vegetation. The purpose was to monitor colonization of the hedges by various animal species. The proximity of the forest, and the connection of the meadow with large hedges, provided a putative means by which non-flying species could colonize the hedges within the meadow (Alignier and Deconchat, 2013). For example, many organisms such as woodlouses, myriapods, cryptic spiders, reptiles or small mammals avoid crossing open areas and follow corridors. To contrast the constructed hedgerows with those bordering the meadow, the former type is hereinafter termed the experimental hedges.

2.3. Biodiversity sampling

Many studies addressing the ecological impacts of agricultural practice have successfully used birds as an index of animal biodiversity (e.g., Pe'er et al., 2014). Others have focused on a particular taxon (e.g., Cole et al., 2002). Approaches that integrate a more accurate examination of biodiversity are preferred, however, because using only a few taxa as surrogates of overall diversity provides unreliable assessment that might not reflect all ecological processes occurring in that habitat (Van Jaarsveld et al., 1998; Andelman and Fagan, 2000; Verissimo et al., 2011). Therefore, we did not focus on a given taxonomic group and instead attempted to sample all macroscopic animals.

The macrofauna was sampled using five complementary versions of a protocol developed to visually identify morphospecies: non-lethal rapid biodiversity assessment (NL-RBA; Lecq et al., 2015; Książkiewicz-Parulska and Gołdyn, 2017). Individuals were not collected, but directly identified and/or photographed in the field. Precise species identification was not always possible; instead observed specimens were assigned to different taxonomic categories, from the species- (fine resolution) to the order-level (coarse resolution; Oliver and Beattie, 1996). To limit observer bias, pictures of common and difficult species were used as "reference specimens;" moreover pictures were taken randomly or when the observer was uncertain. This limitation of the NL-RBA approach was offset by the absence of environmental or ethical concerns, the ability to sample a wide range of taxa, and a high cost/efficiency ratio that enabled us to accumulate a large data set (Lecq et al., 2015).

Three versions of the NL-RBA protocol (rapid visual transect, slow visual transect and focal observation) were relatively similar as they relied on visual searching of the fauna using different walking speeds. The two other versions (active searching and cover objects [five corrugated slabs of cement were placed along each experimental hedge]) targeted cryptic fauna that typically associate with ground refuges. Natural shelters such as stones, leaf litter, or artificial shelters were lifted during these survey variations (see Lecq et al. (2015) for details on each version of the protocol). These two last survey methods attempted to detect hidden animals and, as such, could produce an encounter even at times outside of the activity period for a given species. By combining the different versions of the NL-RBA, the methods were designed to include the relatively cryptic species that depend on the availability of ground refuges (e.g. arthropods, mollusks, reptiles). Yet, many individuals belonging to not-cryptic species were also counted (e.g., a pair of wagtails successfully nested in one experimental hedge).

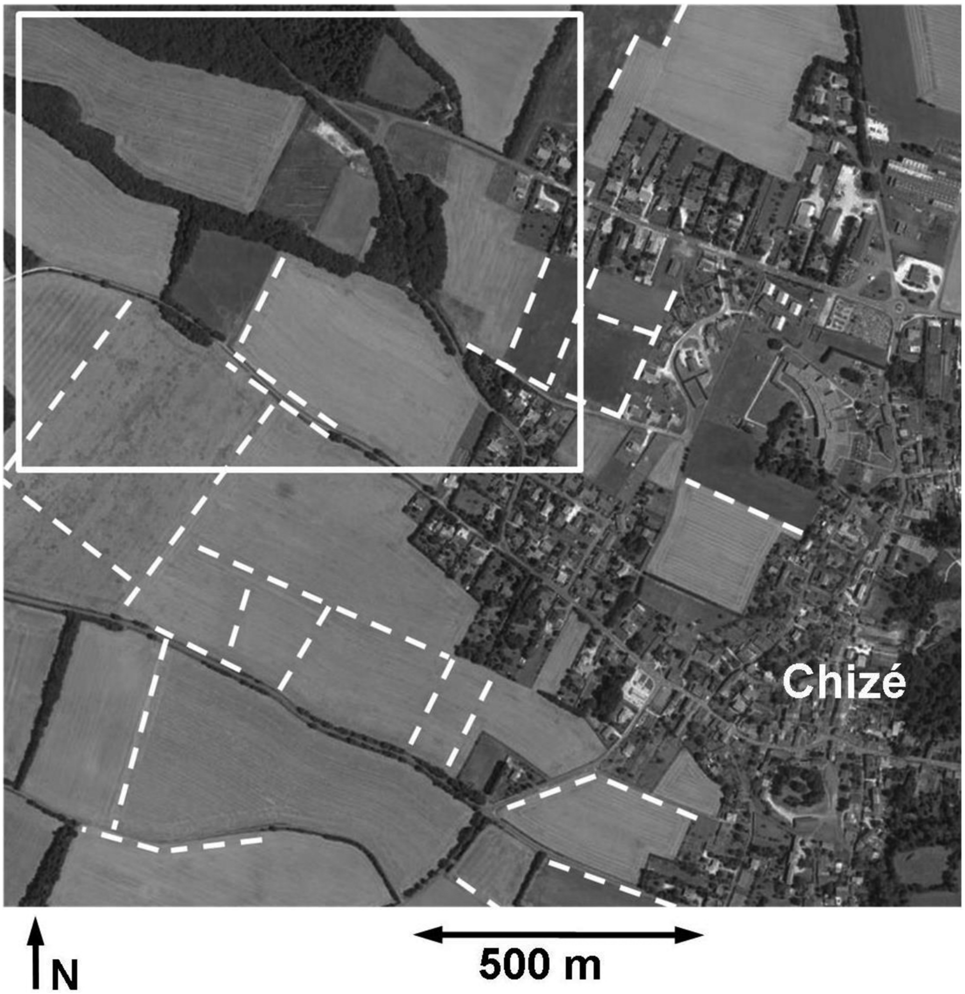

Fig. 1. Aerial view of the broad context of the study (Google-Earth 2015), situated within in a typical peri-urban landscape where agricultural fields are partly bordered by hedgerows. The city of Chizé is in the right side of the image. The white square indicates the enlarged portion of the Fig. 2. The dashed white lines show the past position of hedgerows destroyed during the last decades. Likely, a greater number of hedges have been removed than indicated.

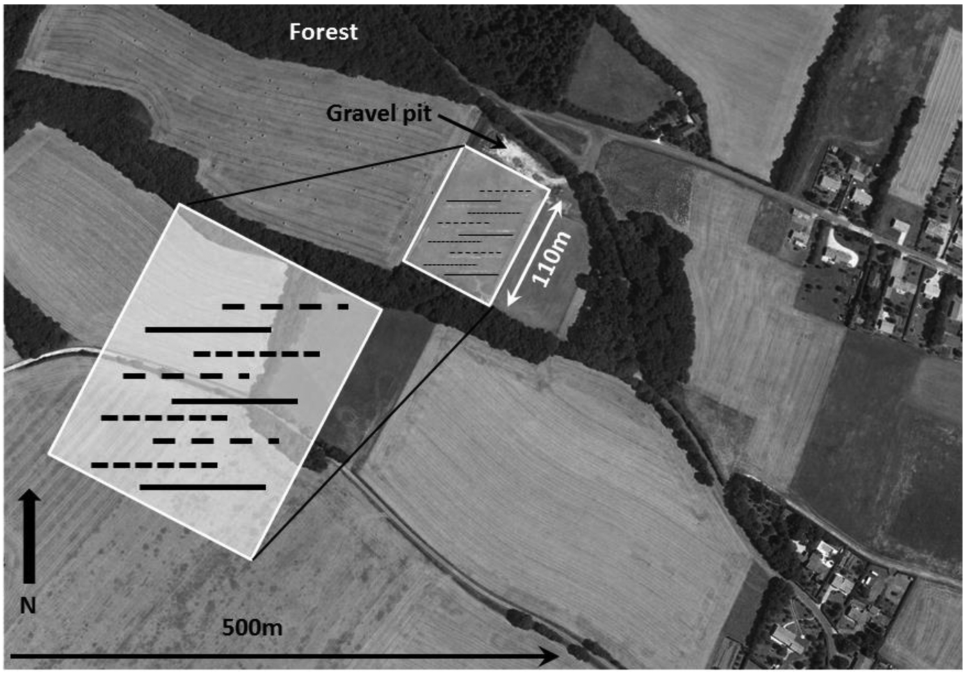

Fig. 2. Aerial view of the experimental study site. Nine hedges (black lines) were built in a meadow (white rectangle). The enlargement shows the three types of experimental hedges: short dashed line indicates lack of embankment, the medium dashed line indicates small embankment (1 m wide × 0.75 m high), and the continuous line shows large embankment (1.50 × 1.20 m). The southern limit of the meadow was adjacent to an existing hedgerow. A gravel pit, a hedge and a road form the northern limit. The other two borders were adjacent to relatively open habitat, notably a stadium on the east side (see text for details). Nearest estates are less than 300 m away.

For simplicity, each occasion that a hedge was surveyed was called a monitoring session regardless of the protocol used.

The monitoring period extended from spring 2011 (April) to summer 2012 (August), but winter (October–March) and drought periods (in July) were excluded. The experimental hedges were colonized relatively quickly (e.g., after only 2 month of maturation). Each of the five versions of the protocol was performed 5 to 8 times in each experimental hedge. A total of 328 transects was surveyed, and each experimental hedge was surveyed an average (± 1 SD) of 7.3 ± 0.8 times. The different types of transects were equally distributed across each hedge (Lecq et al., 2017).

2.4. Analyses

All observed specimens were assigned to a morphospecies. We compared the number of morphospecies between the different types of experimental hedges (a proxy of the diversity of morphospecies that colonized the hedges). Individuals were not marked (and not killed), and thus some of them might have been observed more than once. A single individual counted more than once, however, cannot belong to more than one morphospecies and, as such, our analyses were not subject to pseudoreplication.

A limitation to using the number of morphospecies as a metric of diversity is that the ecological roles of each morphospecies are not identical. For example, a large predator (> 100 g) should not be directly compared to a small primary consumer (1 g) because the former depends on a more complex trophic chain, and therefore a larger amount of resources, than the latter. Our data set includes such a wide range of morphospecies (e.g. large colubrid snakes *versus* grasshoppers). In order to account for the respective weight of the different morphospecies observed, we used a simple functional index that accounts for body mass, trophic level, and metabolic mode (Lecq et al., 2017). The criteria used in this index to characterize, and thus rank the morphospecies, were conservative. For each morphospecies the index was calculated using the following equation:

$$\text{Ecological rank} = \log(\text{body mass}) \times \text{trophic level} \times \text{metabolic mode}$$

Details for the estimation of the parameters are provided in Lecq et al. (2017). As an example, the shift from ectothermy to endothermy (metabolic mode) was associated with a factor of 2 even though, on average, endothermic animals exhibit a metabolism 10 times greater compared to ectothermic species.

For each hedgerow, we used the sum of the ecological ranks of the different morphospecies observed divided by the number of monitoring sessions. The value obtained was named ‘hedgerow biodiversity score’ (HB). This score differed from the numbers of morphospecies observed in the hedges – it provided an index of the complexity of the trophic chain instead of an index of morphospecies richness.

Predators depend on underlying trophic levels and members of this group are considered as a useful surrogate for biodiversity (Sergio et al., 2009). For this reason, we compared the proportion of predator morphospecies to the total number of morphospecies. In our sample, most predators were represented by various species of invertebrates (e.g. spiders, carabids). Using NL-RBA, predators cannot be confused with other consumers: for example, even if identified incorrectly at the species level, a spider will be correctly assigned to spiders. For this specific analysis, possible pseudoreplicates were included. However, considering the duration of the monitoring period (1.5 year) and because the time elapsed between sessions was usually greater than one month, the likelihood of counting the same individual several times was limited. Moreover, the effect of pseudoreplicated data should apply equally among the three hedges types and thus did not impede comparisons among them.

Our analyses also considered the number of reptiles. Three species were observed in the experimental hedges (two colubrid snakes and one lacertid lizard) and individuals were individually marked (enabling us to discard possible pseudo-replicates). They are all predators, but the two snakes (*Hierophis viridiflavus* and *Zamenis longissimus*) are among the largest predatory species (> 1.3 m, > 400 g) in the region. They feed on vertebrates and their diet includes other predatory species (e.g. lizards, shrews, birds; Lelièvre et al., 2012). They exhibit sedentary habits and they depend on ground refuges (Bonnet et al., 1999; Lelièvre et al., 2010) especially in hedgerow systems (Bonnet et al., 2016; Lecq et al., 2017). More generally, the herpetofauna offers suitable surrogate taxa to estimate local biodiversity, and as such they are considered as useful bio-indicators (Beaupre and Douglas, 2009; Lewandowski et al., 2010). Although reptiles were included in the two other analyses (i.e., numbers of observations and HB score), their contribution was limited (in reptiles, $N = 23$ observations and $N = 3$ morphospecies, compared to the arthropods where $N > 7000$ observations and $N > 200$ morphospecies). Thus, the use of reptile number was considered as a relatively independent index.

The distribution of the variables tested did not deviate from normality (e.g. HB, Shapiro-Wilk normality test $W = 0.98$, $P = 0.26$), thus we used analyses of variance (ANOVA) for our comparisons. None of the hedges provided disproportionate amount or lack of data; instead each set of three hedges delivered relatively homogenous signal as indicated by the relatively modest standard deviations (see Fig. 3; no outlier effect). Proportional values were arsine-transformed prior analyses.

3. Results

3.1. General trend

We obtained a total of 13,776 observations during the study period that comprised a total of 237 morphospecies (Table 1). The results consistently indicated a positive effect of embankment size on the three indices (Fig. 3). Thus, the shelters provided by the embankment *per se*, and by the greater amounts of refuges (e.g. stones) influenced the likelihood of observing a greater number of morphospecies, notably large predators.

3.2. Number of morphospecies

Embankment size influenced the mean number of morphospecies observed per hedge (ANOVA, $F_{2,6} = 6.71$, $P < 0.03$; Fig. 3). A Kruskal-Wallis test returned a similar result ($H = 5.96$, $P = 0.05$). Post-hoc tests indicated a difference in the number of morphospecies between only the MB and LB embankment types (Bonferroni post-hoc test, $P = 0.03$; Fig. 3).

3.3. Hedgerow biodiversity score (HB)

The mean HB score differed among the three types of experimental hedges (ANOVA, $F_{2,\ 6} = 72.98$, $P < 0.001$; Kruskal-Wallis $H = 7.20$, $P = 0.03$; Fig. 3). Bonferroni post-hoc tests indicated differences in HB scores among all hedgerow types ($0.010 < P < 0.001$).

3.4. Proportion of predator morphospecies

The proportion of predatory species differed across the three hedge types (ANOVA, $F_{2,\ 6} = 28.35$, $P < 0.001$; Kruskal-Wallis $H = 5.96$, $P = 0.05$; Fig. 3). These values were greater in the SB and LB hedge types than in the hedges without any embankment (Bonferroni post-hoc test $P < 0.01$). However, differences in this response variable were not apparent between SB and LB hedge types ($P = 0.27$).

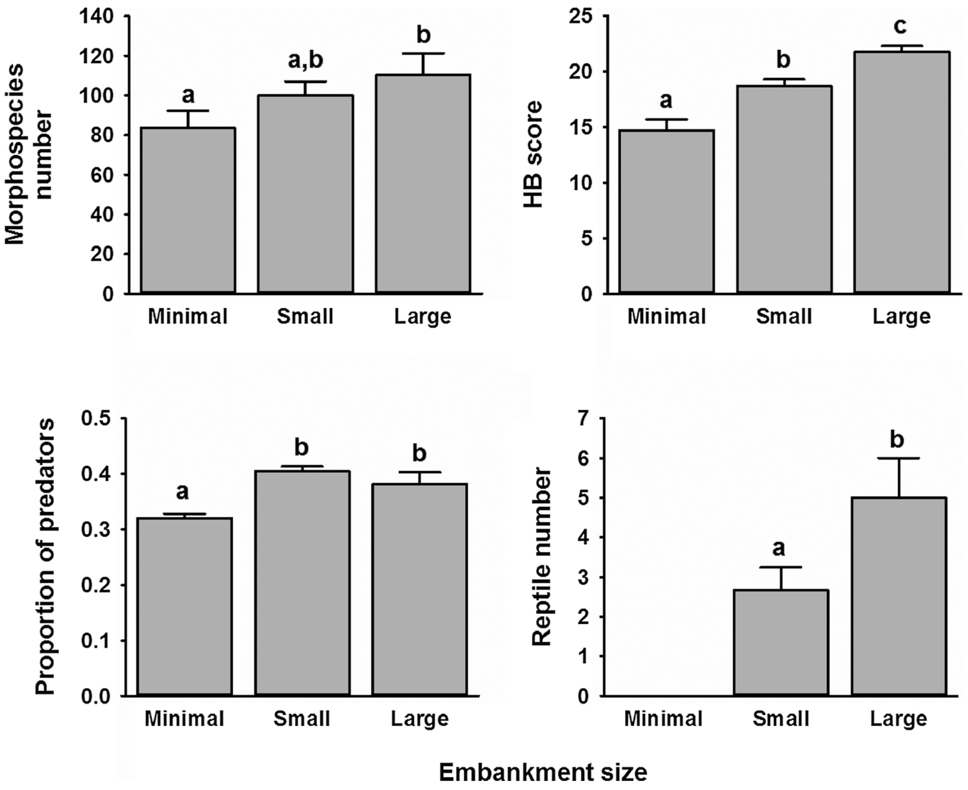

Fig. 3. The effect of different embankments on different indices of macrofaunal diversity in hedgerows: no embankment (= 'minimal'), small, and large embankments (see text for details). The number of morphospecies represents various taxa of invertebrates and vertebrates. HB scores account for the size, diet and metabolic mode of the morphospecies. Values are shown as means + 1 standard deviation. Letters above bars denote significant differences (post hoc tests) among hedge types.

3.5. *Number of reptiles*

We found no reptile in the experimental hedge with no embankment. The mean number of reptiles increased with embankment size (ANOVA: $F_{2,\ 6} = 42.25$, $P < 0.001$; Kruskal-Wallis $H = 9.00$, $P = 0.01$; Fig. 3).

4. Discussion

To our knowledge, this study is the first to focus specifically on the microhabitats at the basal areas of hedges: the embankments. Although framed in a peri-urban context, this study has broader implications. Large amounts of hedgerow habitat have been removed from agricultural areas during the past several decades leading to a marked homogenization of the landscapes, especially in Europe (e.g., Baudry and Jouin, 2003). Detrimental effects on biodiversity, with a noticeable loss of ecosystem services, have been observed (Le Coeur et al., 2002; Batáry et al., 2015). This study examines the relative importance of local management in order to propose cost effective recommendations to limit the biodiversity loss caused by landscape modifications. The consistent signal across all response variables was that changes in the basal structure of the hedge influenced the occurrence of morphospecies. Indeed, each of the different indices for the presence of macroscopic fauna exhibited a positive relationship with the size of the embankment base. This result was expected because many animal species depend on the presence of appropriate refuges, notably cryptic predators belonging to various taxa (e.g. arachnids, carabids, reptiles). Furthermore, the positive relationship between the occurrence of the embankments and the diversity of morphospecies could be driven by more abundant or higher-quality ecotone habitat being available (Perea et al., 2011; Klar and Crowley, 2012). We next review the main methodological limits of this study, and then discuss results and perspectives.

The main limitation of our study is represented by the relatively small number of experimental hedges per treatment (e.g. 3 hedges per size-group, 5 added cover objects per hedge). Despite this small number of replicates, confidence intervals remained narrow (Fig. 3). Thus, the consistency within groups was associated with the notable differences

Table 1
Number and occurrences of morpho-species observed in the hedgerows.

	Morpho-species				Occurrence			
	Large Base	Small Base	Minimum Base	Total	Large Base	Small Base	Minimum Base	Total
Arachnida	30	34	23	40	523	495	315	1333
Aves	1	1	1	2	5	16	11	32
Chilopoda	3	3	1	4	6	4	1	11
Clitellata	1	0	1	1	1	0	2	3
Crustacean	1	1	1	1	317	256	232	805
Diplopoda	1	0	0	1	2	0	0	2
Gastropoda	8	7	7	8	2185	2525	1782	6492
Insecta	124	111	106	171	2000	1756	1178	4934
Mammalia	2	3	5	6	20	18	39	77
Reptilia	3	1	0	3	15	8	0	e
Nothing	NA	NA	NA	NA	5	6	48	59
Total	**174**	**161**	**145**	237	**5079**	**5084**	**3608**	**13771**

among groups, suggesting that our results were robust. A possible lack of independence among the hedges attributable to the short distance between them did not influence the interpretation of our results. During colonization, individuals had the capacity to navigate across the meadow. Indeed, with the exception of sessile life-history stages (e.g. lepidopteran chrysalis) almost all the morphospecies studied had the physical ability to cross the distance between adjacent hedges (10 m). This means that they had ample opportunities to select settlement or foraging places, especially considering the long time periods elapsed between consecutive sampling events. Thus, a greater number of morphospecies (notably, large predators) actually preferred hedges with a large embankment, where shelter availability was higher.

The relative homogeneity of structure within each of the three hedge types (i.e., similar age and species composition of planted trees) indicates that embankment size was more important than the exact position of the hedges within the meadow. Finally, limitations associated with the use of slightly different NL-RBA protocols were offset by advantages. Our approach incorporating multiple survey modes enabled us to sample a wide range of taxa, and to collect a large data set in a non-destructive way. Lethal methods remove important numbers of individuals and might hinder establishment in newly-constructed habitat, especially for species or populations that are sensitive to periodic takings. In practice, identification errors were limited to those species that are morphologically similar in size and in shape. Further, three of the four indices do not rely on taxonomic accuracy and did not present major identification errors (Lecq et al., 2015; Książkiewicz-Parulska and Gołdyn, 2017).

Overall, our conclusions that increasing embankment size promotes animal biodiversity, augments the proportion of predators, and creates attractive conditions for reptiles are likely robust. During the study period, the planted trees played a minor role (if any) because they had insufficient time for any appreciable above-ground growth. Precipitation levels were low during the experiment (MeteoFrance) and almost all trees remained small (< 50 cm in height). Continued growth of the trees and shrubs, and the associated production of leaf litter, should further enhance biodiversity on the local scale (Zanaboni and Lorenzoni, 1989). Long-term experiments are required to examine this issue, however, because the typical vegetation along hedgerows requires at least 20 years to reach maturity.

Reptiles were observed only in the hedges provided with an embankment (SB and LB treatments). Lizards and snakes use ground refuges intensively, and thus were likely attracted by the shelters offered by the embankment and stones (Bonnet et al., 1999, 2013). Our experimental hedges were built in a set-aside meadow adjacent to a large hedgerow (Fig. 2). Reptiles were probably present in nearby habitats at the beginning of the study. The rapid colonization of the hedges was enhanced by the connectivity with surrounding favorable habitats (e.g. large hedge, forest) that represented a source population of individuals. Thus, our results might not be transposable to large industrial and intensively managed urban-ecosystems devoid of source of colonizers. This does not diminish the potential importance of embankments for those species that can cross large distances, notably aerial dispersers or flying species. Hedgerows with large embankments might be useful to reinforce the corridors and connection among structures (Mauremooto et al., 1995) and to promote the other beneficial roles of hedgerows (Xiao et al., 2010). Regarding reptiles specifically, a long-term experiment set up in a peri-urban park demonstrated that promoting snake populations through habitat management was possible and well accepted by residents (Bonnet et al., 2016). The initial conditions in that study were particularly challenging for several reasons: 1) hedgerows were created through tree-cutting (people prefer large trees over shrubs), 2) the main snake species involved (*Vipera aspis*) was venomous, potentially dangerous, and thus undesired in public areas, and 3) the park was amidst a vast urban area and was thus relatively isolated from other natural habitats. The current experiment did not involve these degrees of difficulties. Convincing managers and people to accept and to promote complex structures at the basement of hedgerows (e.g. large embankments) that might potentially attract small invertebrates and reptiles should represent an easier objective.

Our study provides information that offers practical outcomes for habitat management and restoration. During planting programs, for example, recommendations can be proposed to improve the biodiversity and carrying capacity of hedges. For aesthetic reasons, material excavated from a site is typically deposited elsewhere. We suggest that retaining this material on site to build uncompacted embankments will create microhabitats that are attractive to those organisms that are desired - notably predators of plant pest species like parasitoids (Ctifl, 2000; Langer, 2001; Tscharntke et al., 2005; Batáry et al., 2010). Specific educational programs might be useful. The relatively rapid colonization (< 1.5 year) by more than 200 different morphospecies suggests that including embankment structures in construction guidelines for hedgerows might present a cost-effective method that favors animal biodiversity. Although not investigated in this study, plant biodiversity might also respond positively to the presence of embankments, a process that could further increase animal diversity (Lawton, 1983).

5. Conclusions

This study provides an experimental support to a recent empirical investigation that suggested a positive influence of embankment size on animal biodiversity in agricultural landscapes (Lecq et al., 2017). Our study design involved the construction of realistic embankments - the three types of experimental hedges used were representative of the situations encountered in peri-urban areas and agricultural landscape of the region in which we conducted the study. It would be interesting to replicate this experiment in different contexts (e.g. parks, rivers, gardens). Furthermore, it could be useful to augment features of the hedge with shelters (like stone heaps) or plant debris to further bolster biodiversity. The loss of ecological heterogeneity at multiple spatial and temporal scales is a universal consequence of the intensification of anthropogenic activities. Future research should develop integrative policy frameworks and management solutions that re-establish that heterogeneity. Implementing habitats and microhabitats into large scale restoration programs to promote the conservation of biodiversity has been identified as a priority (Gren et al., 2014; Crouzeilles et al., 2015). Practical solutions to reach these objectives, one of which has been examined in this study, should imply a high efficiency/cost ratio to be applicable.

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