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3 Earthworms, spiders and bees as indicators of habitat quality and management in
4 a low-input farming region – a whole farm approach

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26 **Abstract**

27 The benefits of low input farming on biodiversity and ecosystem services are already well-
28 established, however most of these studies focus only on the focal field scales. We aimed to
29 study whether these benefits exist at the whole farm scale, to find the main environmental
30 driving effects on biodiversity at the whole farm scale in farms of different grassland grazing
31 intensity, applying three well-known species diversity indicator groups of different ecological
32 traits.

33 Edaphic (earthworms), epigeic (spiders) and flying (bees) taxa were sampled in each
34 identified habitat type within eighteen low-input farms in Central Hungary, 2010. The number
35 of habitat types, the number of grassland plots, the cumulative area of grasslands and habitat
36 type had an effect on the species richness and abundance of spiders, while grassland grazing
37 intensity influenced the species richness of bees. Both bees and spiders were sensitive to
38 vegetation and weather conditions, resulting in more bees on flower-rich farms and those
39 having higher temperature; and more spiders on farms with more heterogeneous vegetation
40 structure and in low-wind areas. Relatively few earthworms were found in the whole study,
41 and their abundance was not influenced by any of the farm composition and management
42 variables.

43 We conclude that local field management (grazing intensity of grassland patches) can
44 have a farm scale effect, detectable on species diversity indicators that have high dispersal
45 ability and strong connection to grasslands as important foraging sites (bees). However, other
46 farmland biota (spiders) is also strongly determined by farmland composition and habitat
47 diversity, therefore the maintenance of a mosaic within-farm habitat structure is strongly
48 recommended. The application of earthworms as farmland composition or management
49 indicators is strongly restricted because of their special needs of soil conditions.

50

51 **Keywords:** extensive farming; farmland biodiversity; grazing intensity; invertebrates;

53 **1. Introduction**

54 Farmland habitats, including arable fields and grasslands are the dominant land use types all
55 over Europe, and are very important for several open landscape-related species. The
56 management of these habitats directly effects biodiversity at field and landscape scales
57 (Donald et al., 2001; Stoate et al., 2009). Intensive agricultural management, especially the
58 use of inorganic fertilisers and pesticides as well as increased land use intensity resulted in
59 landscape homogenisation, and caused declining population trends of several plant,
60 invertebrate and vertebrate taxa (Benton et al., 2003; Geiger et al., 2011). In the last decades,
61 an emerging demand to halt the loss of farmland biodiversity and ecosystem degradation
62 resulted in increased implementation of “low input farming systems”, among them organic
63 farming and subsidised agri-environment schemes in the EU (Kleijn and Sutherland, 2003;
64 Kleijn et al., 2011).

65 The benefits of low input farming systems were addressed in several former studies
66 (Hole et al., 2005; Knop et al., 2006), but their overall effectiveness are mixed (Kleijn et al.,
67 2006). One reason for this could be that there are no generally accepted indicators. Several
68 studies include popular and easy-to-study taxa, like birds or plants, and simple descriptors as
69 species richness and/or abundance data (Chamberlain et al., 1999; Gabriel et al., 2005;
70 Verhulst et al., 2004). Additionally, studies usually compared a focal field under low input
71 management with a nearby, conventionally managed field (Bengtsson et al., 2005; Hole et al.,
72 2005; Kleijn et al., 2006, 2011; Pacini et al., 2003), thus on a restricted spatial scale, although
73 landscape scale effects may interact with local scale management (Batáry et al., 2011).
74 However, landscape scale is rather vaguely defined, and not operative in a socio-economic
75 context (Gabriel et al., 2010). Whole farm scale is appropriate to assess larger spatial scale
76 effects, and is the real scale for management decisions.

77 In the present study, we applied a “whole farm approach”, sampling all kind of major
78 habitat types to assess the main drivers of biodiversity in a low input farming system,

79 including 1) the effects of habitat composition within the farm, 2) effects at the whole farm
80 scale of grazing intensity in grassland patches, 3) effects of local vegetation composition and
81 structure and 4) the influence of selected weather conditions. This approach enabled us to
82 investigate the importance on farm-level biodiversity of habitats that are not directly managed
83 by farmers but are influenced by farming practices (linear habitat features, forest patches and
84 wetlands). Such habitats are often neglected in studies that usually focus on the effects of
85 management procedures on cultivated fields, yet they are important for most taxa living in
86 agricultural landscapes (Hendrickx et al., 2007; Hof and Bright, 2010; Sunderland and Samu,
87 2000). Considering management effects, we focused on grassland grazing intensity, as this is
88 probably the most important habitat for agricultural biodiversity (Duelli and Orbist, 2003) and
89 was the dominant land use type within the study region. Vegetation structure and species
90 composition usually has strong influence on arthropods, while the activity of most animal taxa
91 might be fundamentally affected by weather conditions (Schmidt and Tscharnke, 2005a;
92 Willmer et al., 2004).

93 Most of the published studies on farmland biodiversity were conducted in Western and
94 Northern European countries, and there is much less knowledge from Central and Eastern
95 Europe (CEE) on the relationship of agricultural practices and farmland biodiversity (Báldi
96 and Batáry, 2011; Tryjanowski et al., 2011). CEE countries have different economic and
97 agricultural history, with different biogeographical and climatic conditions. These differences
98 highlight the urgent need for research evidence in the CEE countries because the applied
99 conservation strategies based on knowledge of farmland ecology in Western Europe cannot
100 simply be adopted in the CEE region (Báldi and Batáry, 2011; Hartel et al., 2010).

101 To assess farm composition and grassland management effects within the studied low-
102 input systems, proper species diversity indicators are needed, which are relatively easy to
103 monitor, provide relevant information on environmental conditions and environmental
104 changes, are generic at wider scale and provide useful and easily understandable information

105 for stakeholders. As a first step, three animal taxa were chosen to represent the endogeic
106 (earthworms, Lumbricidae), epigeic (spiders, Araneae) and flying (bees, Apoidea)
107 macroinvertebrate fauna, and we examined the farming effects on these selected groups,
108 representing both below- and above-ground biodiversity. These groups also provide important
109 ecosystem services. Earthworms have an essential role in the productivity of organic and low-
110 input farming systems through recycling and composting soil nutrients, enhancing soil fertility
111 and enhancing decomposition processes (Jouquet et al., 2006). Both physical (e.g. ploughing,
112 trampling) and chemical (fertiliser and pesticide use) agricultural practices affect soil
113 conditions and earthworm assemblages. Therefore they are suggested to be suitable indicators
114 of soil structure, tillage practice and grassland management (Chan, 2001). Spiders have an
115 important role in biological control as natural enemies of invertebrate pests in agro-
116 ecosystems (Marc et al., 1999; Riechert and Lockley, 1984; Schmidt et al., 2003). Spiders are
117 broadly distributed in agricultural and semi-natural habitats (Schmidt and Tschardtke, 2005b)
118 and are sensitive to arable crop (Batáry et al., 2008b; Samu, 2003) and grassland (Batáry et al.,
119 2008a,b) management and weather conditions (Schmidt and Tschardtke, 2005a), making them
120 widely used environmental indicators. Wild bees are the most important pollinators of arable
121 crops and wild plant species, especially in Europe (Biesmejer et al., 2006; Klein et al., 2007).
122 They are highly sensitive to the presence of flowering plants as foraging resources (Ebeling et
123 al., 2008; Freund et al., 2010), require several special conditions for nesting, such as bare soil,
124 dead wood or plant stems (O'Toole and Raw, 1991), and therefore show usually direct
125 response to habitat management and landscape compositional effects (Holzschuh et al., 2007;
126 Kovács-Hostyánszki et al., 2011; Steffan-Dewenter et al., 2002).

127 We hypothesized that (1) farmland composition has an effect on the species richness
128 and abundance of all the selected below- and above-ground indicator taxa, showing higher
129 values in the more natural habitats (grasslands, wetlands) than in the managed patches (crops).
130 However, species richness and abundance of the different invertebrate groups can be

131 determined by the presence and size of different habitat types within a farm; (2) they will
132 show various response to the intensity of grassland management at the whole farm scale due
133 to their different life history and mobility, predicting stronger effect on flying organisms
134 (bees) compared to ground-dwelling arthropods (spiders); (3) local factors such as vegetation
135 structure will mostly influence the species richness and abundance of spiders, while bees are
136 better predictors in changes of plant species richness; (4) the above effects will be modulated
137 by local weather conditions through influence on activity and therefore the applicability of the
138 selected species diversity indicators.

139

140 **2. Material and methods**

141 2.1. Study sites

142 The study was conducted in 2010 in the Homokhátság (“Sand Ridge”), an alluvial plain
143 covered with Aeolian, sand-based low fertility solonchak-solonetz plains in the Kiskunság
144 region, Central-Hungary (Appendix). The region contains a mosaic of slightly undulating,
145 semi-fixed sandhillocks and flat areas of fixed sand, is extensively managed in general. Due to
146 the poor conditions and low economic power of the local land-holders; the major difference
147 between low-input and organic farms is only certification; the management was rather similar
148 on all farms (see Appendix). The major habitats of the region are unimproved semi-natural
149 grasslands and arable fields. Agro-chemicals are not applied on the grasslands, stocking rates
150 are very low (0.15-1.75 LU/ ha grassland). Zero or low inputs of fertilisers (15-50 t/4 year
151 solid cattle manure or 20-30 kg N/ha/year inorganic fertiliser) and one or two pesticide
152 applications are usual on the arable fields. We selected 18 low-input farms; that contained a
153 mosaic of fields under agricultural management and adjacent, non-managed landscape
154 elements that might be affected by farming practices. All habitats at each farm were mapped
155 and classified according to a European scale standard habitat mapping procedure developed in
156 the BioHab project (Bunce et al., 2008), based on a generic system of habitat definition, the

157 General Habitat Categories (GHC). Areal, linear and point habitat features were characterized
158 with respect to their ecological quality, farming and soil properties (see details in Bunce et al.,
159 2008 and the BioBio project website, <http://www.biobio-indicator.org/deliverables/D22.pdf>).
160 On each mapped farm, one plot from each previously identified habitat category, but a
161 maximum of 15 different habitat types were randomly selected (152 plots in total).

162

163 2.2. Zoological sampling

164 2.2.1. Earthworms

165 Three soil samples of 30 cm×30 cm×20 cm deep were taken in each of the sampling plots in
166 May, 2010. The three samples were located 20 m from the edge of the plot and 10 m apart
167 from each other. We extracted earthworms first by using an expellant solution; after this, the
168 samples were hand-sorted to find all remaining earthworms. The expellant was prepared by
169 allyl-isothiocyanate (AITC) diluted with ethanol 70% to give a 5 g/l solution, shortly before
170 going into the field to prevent loss of irritating activity. This was diluted with water to reach a
171 concentration of 0.1 g/l in the field prior to application. Metal frames (30×30 cm) were
172 installed at each sampling locations in depth of approximately 1-2 cm to prevent the solution
173 from running off. We cleaned the sampling site from vegetation or leaves and poured two
174 doses per sampling site of 2 l of AITC solution at 5 min intervals. Emerging specimens were
175 washed with cold water. After 10 min, the soil was dug up from the metal frame to a depth of
176 20 cm. This extracted soil was put on a white plastic sheet and hand sorted for 20 minutes.
177 Earthworms were cleaned in cold water and kept in 70% ethanol solution. In the laboratory,
178 each individual was identified to species level.

179

180 2.2.2. Bees

181 Bees were sampled three times during May, June and August, taking one sample per plot on
182 each of the three sampling dates. Each plot was surveyed by walking along 100 m long and 2

183 meter wide transect over 15 min. All individual bees seen within the transect were caught with
184 an insect net, transferred into a killing jar with ethyl acetate, and identified in the laboratory.
185 Easily identifiable bumble bee species and domestic honey bees, were recorded and released
186 in the field (Móczár, 1957, 1958, 1960, 1967; Schmid-Egger and Scheuchl, 1997). Sampling
187 was carried out on dry and warm days with minimal wind, between 09.00 and 18.00 o'clock,
188 which covers the daily maximum activity regime of the flying insects. During each sampling
189 session, vegetation height and cover of flowering plants (in 1-5 scale) were estimated and the
190 flowering plant species were recorded along a transect.

191

192 2.2.3. Spiders

193 Spiders were caught with a D-VAC sampler, also three times in May, June and August within
194 a period of 10 days to avoid the effect of seasonal succession of spider species (following the
195 approach described by Schmidt et al., 2005). A 50 cm long, tapering gauze bag (mesh < 0.5
196 mm) was inserted into the 11 cm diameter intake nozzle to intercept the spiders. On each of
197 three sampling dates, five suction samples were taken in each of the selected habitat plots,
198 located 20 m from the border and 10 m from each other. In linear elements, the samples were
199 taken along a line in the middle of the habitat and 10 m apart. Each suction sample was taken
200 for 30 seconds within a sample tube with 35.7 cm internal diameter and 40 cm height pre-
201 installed on the target vegetation. The five suction samples were kept separate. Sampling was
202 carried out during dry, warm weather, between 09.00 and 18.00. Specimens were identified to
203 the species level, if possible (Heimer and Nentwig, 1991; Loksa, 1969, 1972; Nentwig et al.,
204 2012). Vegetation height was recorded during each of the three sampling times as average
205 minimum and maximum vegetation height on the site.

206

207 2.3. Data analysis

208 Due to the extremely rainy weather in May 2010 and the consequent constrained sampling,

209 data only from the second and third sampling occasions were analysed in the case of bees and
210 spiders. In the case of earthworms, we have had 489 zero samples from the 660 soil samples,
211 and we pooled the samples at plot level for further analyses.

212 To get species accumulation curves and measure the habitat use of bees and spiders,
213 and the effectiveness of the sampling, we conducted rarefaction analyses. The species richness
214 of bees and spiders was estimated in each habitat type with the Chao estimator (Chao 1987)
215 using the *vegan* package (Oksanen et al., 2011) in R programme. The standard deviations
216 were generated from 10,000 reshufflings of the sample order.

217 General linear mixed-effect models were used to study the relationship between the
218 assumed explanatory variables and the abundance and species richness of earthworms, bees
219 and spiders (GLMM, Bolker et al., 2009). We added a nested spatial random effect to account
220 for the spatial structure among the sampling plots: 1. farm - farm, where the sample was
221 taken; 2. plot - identification code of the sampling plots (eight plots per farm were sampled).
222 In the case of earthworms, only farm was applied as random factor. We did not consider the
223 samples from the same farm as independent ones because of the potential ownership effects,
224 even though they were true spatial replicates. Most of the cases the farm denoted a spatial unit
225 as well (in four cases few fields were spatially apart from the rest of the farm). Plot and farm
226 were considered nested variables for random effect terms in the analyses. In order to avoid the
227 heterogeneity in variance caused by the different sampling intensity (i.e. different number of
228 samples per habitat types per farm), the log-transformed sample number was added to the
229 linear predictor as a known coefficient (1). In the models the response variables were $\log(x+1)$
230 transformed to fulfil the normality requirement for the model residuals. We used the following
231 explanatory variables in the evaluated models:

232 *1. farm model:* habitat (factor with eight levels: canal, forest, crop, grassland, linear habitat,
233 shrubland, trees, wetland), number of habitat types per farm (numerical), total area of the farm
234 (ha), arable area (ha) , grassland area (ha), number of arable fields (within a farm), number of

235 grassland fields (within a farm);
236 2. *management model*: grazing type (factor with four levels: cattle, cattle-sheep, horse-cattle,
237 horse-cattle-sheep), total number of grazing animals (per farm), LU (livestock unit/farm area),
238 LU/grassland (livestock unit/grassland area of the farm, ha);
239 3. *environmental model*: cloud (cover, based on a 1-5 scale), wind (Beaufort scale),
240 temperature (C°), minimum vegetation height (cm), maximum vegetation height (cm), flower
241 cover (1-5 scale), number of flowering species (per field).

242 Environmental variables were measured on the field during the sampling periods; data
243 on farm attributes were reported by the owner in a questionnaire.

244 The differences among the levels of the tested factors (habitat, grazing type) were
245 evaluated by multiple comparisons (with Tukey computed contrast matrices for several
246 multiple comparisons procedures) after a single argument ANOVA for the tested model. The
247 model estimates were obtained using a maximum likelihood method and diagnostics included
248 the Akaike Information Criterion and the model residuals. We estimated the model parameters
249 by using the *nlme* (Pinheiro et al. 2011) and *gplots* packages (Warnes, 2011) for graphical
250 outputs in R 2.14.0 (R Development Core Team, 2011).

251

252 **3. Results**

253 3.1. Species richness

254 Only seven earthworm species were collected, thus we did not apply the rarefied species
255 richness curves for earthworms. For bees, the value was not stable for any of the habitat types,
256 but it rose continuously as the number of samples increased (Fig. 1). For spiders, the values
257 were stable for grassland at 100 samples ($46.94 \text{ species} \pm 1.3 \text{ S.D.}$), as well as for linear
258 elements and “woodlands”. The species richness estimations were approximately stable for
259 grassland at 85 and 100 samples, respectively ($43.91 \pm 0.29 \text{ species}$, and $46.94 \pm 0.22 \text{ species}$,
260 respectively, Fig. 2).

261

262 3.2. Earthworms

263 We collected 551 individuals of seven species in total, with 93% of the individuals belonging
264 to three *Aporrectodes* species (*A. caliginosa*, *A. georginii*, *A. rosea*). Other species were
265 represented by only a few individuals. Consequently, we analysed only the abundance of
266 earthworms, which was not influenced by any of the studied habitat or environmental
267 variables (Table 1).

268

269 3.3. Bees

270 Although the 1135 individuals belonged to 85 bee species, most of the collected bees were
271 honeybees (*Apis mellifera* L.). The species richness and abundance of bees did not show any
272 significant response to the explanatory variables included in the farm model (Table 1). The
273 value of livestock unit in grasslands had a significant, positive effect on bee species richness
274 ($t_{11}= 2.34$, $p=0.03$; Table 1, Fig. 3). According to the environmental model, cloud cover had a
275 negative effect on the abundance of bees ($t_{126}= -2.38$, $p=0.01$), while the air temperature had a
276 slight positive effect on species richness and abundance; post-hoc comparisons revealed that it
277 was not significant (species richness: $t_{126}= 0.42$, $p=0.66$; abundance: $t_{126}= 0.27$, $p=0.78$; Table
278 1). Flower cover had positive effect on bee abundance ($t_{126}= 10.23$, $p<0.001$) (Fig. 4). The
279 flower cover ($t_{126}= 7.86$, $p<0.001$) and the number of flowering species seemed to be the most
280 important environmental variables influencing bee species richness (Table 1, Fig. 4).

281

282 3.4. Spiders

283 In total of 4222 individuals of 199 spider species were collected. The abundance and species
284 richness of spiders increased by the number of habitat types per farm ($t_{11,634}=2.39$, $p=0.035$;
285 $t_{11,634}=2.03$, $p=0.067$ respectively). Both species richness ($t_{119}= -2.80$, $p=0.005$) and the
286 abundance ($t_{119}= -2.48$, $p= 0.01$) of spiders were lower in the crop fields than in the other

287 habitat types (Table 1, Fig. 5). The area of grassland had a negative effect on the species
288 richness ($t_{12} = -4.65$, $p < 0.001$) and abundance ($t_{12} = -5.27$, $p < 0.001$), while the species richness
289 ($t_{12} = 2.61$, $p = 0.02$) and abundance ($t_{12} = 2.47$, $p = 0.02$) of spiders were higher in farms with
290 more grassland patches (Table 1). Patterns of spider species richness and abundance were
291 better explained by the environmental model than either the farm or the management models
292 (Table 1). Wind intensity negatively influenced both species richness ($t_{508} = -4.89$, $p < 0.001$)
293 and abundance of spiders ($t_{508} = -5.00$, $p < 0.001$). The minimum and maximum vegetation
294 height had a positive impact on spiders' species richness ($t_{508} = 2.74$, $p = 0.006$ and $t_{508} = 2.38$,
295 $p = 0.01$ respectively) and abundance ($t_{508} = 3.14$, $p = 0.001$ and $t_{508} = 2.81$, $p = 0.005$) (Fig. 6).

296

297 **4. Discussion**

298 4.1. Farm composition effects

299 The rarefied species richness revealed that the spiders were numerous in the grasslands, linear
300 habitats and tree groups, in descending order. However, reliable estimation of species richness
301 required >80 samples. The high number of spider species in the linear elements and tree
302 groups highlighted the importance of these marginal habitats as sources for spill-over to
303 croplands where they contribute to biological control (Rand et al., 2006). We found similar
304 trends for bees, but the estimations were unstable. This phenomenon could be explained by
305 the high mobility of bees, which may reduce their beta diversity (Marini et al., 2011).

306 Habitat type and farmland composition had an effect on the species richness and
307 abundance of spiders, while the species richness and abundance of earthworms and bees were
308 not affected by the presence, area or number of the various habitat types. Earthworms form
309 the greatest biomass of soil fauna in temperate grasslands; in these communities the number
310 of earthworm species ranges usually 1-15 species, but they contain mostly only 3-6 species
311 (Edwards and Bohlen, 1996). The low number of species and individuals found in our study
312 (most of the individuals belonged only to three species, *Aporrectodea rosea* Savigny, 1826,

313 *Aporrectodea caliginosa* Savigny, 1826, *Aporrectodea georgii* Michaelsen, 1890) and the lack
314 of difference among different habitat types can be explained by the frequent occurrences of
315 sandy soils that provides inappropriate habitat for burrowing earthworms due to the lack of
316 texture (Bardgett, 2005; Edwards and Bohlen, 1996; van Diepeningen et al., 2006). Further,
317 rainy weather during the sampling resulted in extremely high soil moisture even in the
318 normally dry habitats, when most terrestrial earthworm species are expected to emigrate from
319 the flooded soil (Edwards and Bohlen, 1996).

320 Bees did not show any significant difference among the habitat types and were not
321 influenced by the area or number of grassland and arable field patches on the farm. Semi-
322 natural habitats are usually the main drivers of bee diversity in an agricultural landscape, due
323 to the considerable flower resources they provide (Sjödín et al., 2008; Steffan-Dewenter et al.,
324 2002). However, the loss of semi-natural grasslands does not necessarily cause a decline in
325 species richness or abundance of wild bees at the farm level (Carré et al., 2009). The available
326 habitats, especially nesting and foraging resources influence the number of species and
327 individuals, and the species composition of bee communities, but habitat compositional
328 change within the farm may not necessarily cause change in the total abundance or species
329 richness (Carré et al., 2009). The lack of habitat effect on bee species richness and abundance
330 suggests that at the farm scale, low-input farms in Hungary offer appropriate foraging
331 resources (flowers) and nesting conditions even when the proportion of grasslands is lower.

332 The species richness and abundance of spiders were the lowest in the crop fields, and
333 were enhanced by the number of grassland fields in the farm, but decreased by the increase in
334 total grassland area. We suggest that as the number of grassland patches increase, so does
335 habitat heterogeneity at the farm level, which contributes to the increase of spider richness
336 and abundance (Batáry et al., 2008a; Benton et al., 2003). Arable fields are preferred less by
337 spiders than perennial grasslands (Ratschker and Roth, 2000; Schmidt and Tschardtke, 2005),
338 possibly due to the negative effects of management (e.g. fertilizer and pesticide use) and the

339 less heterogeneous vegetation structure (Batáry et al., 2008b). The complexity of landscapes
340 including perennial non-crop habitats is key to preserve or restore high levels of spider
341 diversity (Schmidt et al., 2005).

342

343 4.2. Management effects

344 Grassland management had no effect on the species richness and abundance of earthworms at
345 the farm scale. The generally unfavourable soil and weather conditions overwrote the effects
346 of management. Grazing intensity had a positive effect on the species richness of bees at farm
347 level. Grazing on the grasslands could have important role in the maintenance of high flower
348 diversity, preventing the dominance of few perennial species and/or succession into
349 shrublands (Bakker, 1998; Hansson and Fogelfors, 2000). In the more intensively grazed
350 grasslands the increased diversity and amount of flowers provides more foraging resources for
351 bees, explaining their higher species richness on these farms (Sjödin et al., 2008; Batáry et al.,
352 2010). Nevertheless, it should be noted that grazing intensity in our study sites (see methods)
353 was still much lower than several Western-European countries, where grazing has significant
354 negative effect on pollinators (Sároszpataki et al., 2009; Batáry et al., 2010).

355 We found no direct effect of grazing intensity on spiders. As only high grazing
356 intensity affects spiders, especially the number of vegetation-dwelling species due to the
357 changes in the vegetation structure (Batáry et al., 2008a,b; Dennis et al., 2001; Horváth et al.,
358 2009). Our results show that even under 1.75 LU/ ha grazing intensity, grasslands provide
359 valuable habitats for species rich spider assemblages. It is also likely that in our low-input
360 farms, the potential negative effects of grazing are buffered by the presence of semi-natural
361 habitats and landscape heterogeneity at the farm scale (Benton et al., 2003; Schmidt and
362 Tschamntke, 2005).

363

364 4.3. Vegetation structure and weather effects

365 Bees and spiders were both influenced by vegetation structure and weather conditions. For
366 bees, flower resources were the most important driving factors; this is more important than
367 habitat type or management (Bardgett, 2005; van Diepeningen et al., 2006). The species
368 richness of flowering plants enhanced bee species richness, while flower cover had a positive
369 effect on both their species richness and abundance (Ebeling et al., 2008; Holzschuh et al.,
370 2007; Sároszpataki et al., 2009). Bee species differ in their floral requirements, and a higher
371 number of flowering plant species increases nectar and pollen resource heterogeneity, which
372 enhances attractiveness for many pollinators and increases their species richness (Potts et al.,
373 2003).

374 We found a strong relationship between vegetation structure and spider assemblages.
375 Both the minimum and maximum vegetation height increased both species richness and
376 abundance. Vegetation as the most important local habitat characteristic influencing spiders is
377 well documented (Batáry et al., 2008a; Dennis et al., 2001; Gibson et al., 1992). Web-building
378 spiders demand higher stems at different heights to construct their nets, whereas hunting
379 species are associated with patches of low vegetation so that they can see and pursue their
380 prey. Therefore, a more complex vegetation structure supports more spiders, both species and
381 individuals (Dennis et al., 2001).

382 Cloud cover had a negative effect on bee abundance, while air temperature had a slight
383 positive effect on bee species richness and abundance. Thermal constraints limit the foraging
384 activity of bees, defining a microclimatic range of each species within which foraging flight
385 can be sustained (Corbet et al., 1993), which explains the enhanced number of bees under
386 conditions of lower cloud cover and the consequent higher temperatures (Stone and Willmer,
387 1989). Higher cloud cover and wind had negative effects on spider species richness and
388 abundance. Wind prevents spiders from successfully foraging and destroys spider webs,
389 which decreases their activity and forces them to hide in the lower levels of vegetation or in
390 the litter layer, decreasing the possibility of their capture even by suction.

391

392 **5. Conclusion**

393 Low-input Eastern European farmlands are traditional, extensive management systems,
394 characterized by rich biodiversity. The study of eighteen Hungarian farms showed that
395 heterogeneous habitat composition and moderate grazing (1.75 LU/ha) intensity in grasslands
396 have considerable importance to maintain the richness of spider and bee assemblages at the
397 farm scale. We conclude that the farm scale effect of local field management, i.e. grazing
398 intensity of grasslands is more detectable on species diversity indicators of higher dispersal
399 ability and strong connection to grasslands. However, because of the strong connection
400 between flower resources and their pollinator bees, bees were related to only a few, more
401 flower-rich habitat types, such as grasslands and linear habitat elements, while spiders were
402 more sensitive to habitat diversity and within-farm habitat types, which made spiders better
403 indicators of farm compositional effects. The distribution of earthworms was strongly
404 restricted by soil and humidity; therefore we suggest their applicability as farmland
405 management and composition indicators only at spatially restricted scales, in the case of
406 appropriate soil conditions. Weather conditions, such as temperature and cloud cover for bees
407 or cloud cover and wind for spiders, have significant influence on activity of these taxa. This
408 should be taken into account during their application as environmental indicators.

409

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421

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633 Table1. Results of general linear mixed-effect models relating farm, management and environmental variables to log-transformed abundance and
 634 species richness of earthworms (abundance only), bees and spiders. Significances are marked in bold.

	Earthworms		Bees				Spiders			
	log(abundance)		log(abundance)		log(species richness)		log(abundance)		log(species richness)	
	F (d.f.)	p	F (d.f.)	p	F (d.f.)	p	F (d.f.)	p	F (d.f.)	p
Farm model										
Habitat	0.701 (6,148)	0.648	0.83 (7,128)	0.55	0.93 (7,128)	0.48	2.28 (7,119)	0.03	2.7 (7,119)	0.01
Habitat types/farm							14.143 (11,634)	0.003 +	9.53 (11,634)	0.01 +
Total area (ha)	2.43 (1,11)	0.146	0.32 (1,12)	0.57	0.89 (1,12)	0.36	0.11 (1,12)	0.74	0.21 (1,12)	0.65
Arable area (ha)	0.008(1,11)	0.928	2.47 (1,12)	0.14	0.92 (1,12)	0.35	2.77 (1,12)	0.12	2.88 (1,12)	0.11
Grassland area (ha)	1.141 (1,11)	0.308	0.19 (1,12)	0.66	0.14 (1,12)	0.7	23 (1,12)	<0.001 -	16.91 (1,12)	0.001 -
No. arable plots	1.213 (1,11)	0.294	0.14 (1,12)	0.7	0.04 (1,12)	0.82	0.21 (1,12)	0.65	0.43 (1,12)	0.52
No. grassland plots	0.336 (1,11)	0.57	0.9 (1,12)	0.36	0.78 (1,12)	0.39	6.08 (1,12)	0.02 +	6.8 (1,12)	0.02 +
sd (random effect's residuals)		0.46		0.99		0.53		0.74		0.53
Management model										
Grazing type	1.027 (3,10)	0.421	2.12 (3,11)	0.15	2.33 (3,11)	0.12	0.79 (3,11)	0.52	0.56 (3,11)	0.65
Total no. grazing animals	0.003 (1,10)	0.957	3.09 (1,11)	0.1	0.006 (1,11)	0.93	0.006 (1,11)	0.93	0.03 (1,11)	0.85
LU	0.07 (1,10)	0.796	0.29 (1,11)	0.59	0.02 (1,11)	0.86	0.46 (1,11)	0.51	0.98 (1,11)	0.34
LU/grassland	0.542 (1,10)	0.478	0.93 (1,11)	0.35	5.49 (1,11)	0.003 +	1.6 (1,11)	0.23	1.16 (1,11)	0.3
sd (random effect's residuals)		0.46		0.99		0.53		0.74		0.53
Environment model										
Cloud cover (1-5 scale)			9.18 (1,111)	0.003 -	2.23 (1,111)	0.13	3.65 (1,508)	0.05	5.85 (1,509)	0.01 -
Wind (Beaufort scale)			2.43 (1,111)	0.12	1.67 (1,111)	0.19	30.08 (1,508)	<0.001 -	28.68 (1,509)	<0.001 -
Temperature (°C)			2.11 (16,111)	0.01 +	2.14 (16,111)	0.01 +	1.09 (1,508)	0.29	1.65 (1,509)	0.19
Min. vegetation height (cm)			0.11 (1,111)	0.73	0.42 (1,111)	0.51	33.97 (1,508)	<0.001 +	24.93 (1,509)	<0.001 +
Max. vegetation height (cm)			0.44 (1,111)	0.5	1.43 (1,111)	0.23	8.37 (1,508)	0.004 +	6.19 (1,509)	0.01 +
Flower cover (1-5 scale)			165.5 (1,111)	<0.001 +	113.3 (1,111)	<0.001 +				
No. flowering species			0.81 (1,111)	0.36	8.18 (1,111)	0.005 +				
sd (random effect's residuals)				0.72		0.42		0.56		0.79

635 Figure legends

636

637 Fig. 1: The species richness estimation for bees according to Chao estimators (with 95 %
638 confidence intervals) based on the habitat types on 18 low-input farms in Central
639 Hungary.

640

641 Fig. 2: The species richness estimation for spiders according to Chao estimators (with 95 %
642 confidence intervals) based on the habitat types on 18 low-input farms in Central
643 Hungary.

644

645 Fig. 3: The relationship between grazing intensity (livestock unit/ ha grassland) and species
646 richness of bees on 18 low-input farms in Central Hungary. The smoothed line was fitted
647 by local polynomial regression techniques (lowess; alpha was set to one to fit closely
648 linear regression).

649

650 Fig. 4: Relationship between the most influential environmental variables and the abundance
651 (A) and species richness (B) of bees on 18 low-input farms in Central Hungary. The
652 smoothed line was fitted by local polynomial regression techniques (lowess; alpha was
653 set to one to fit closely linear regression).

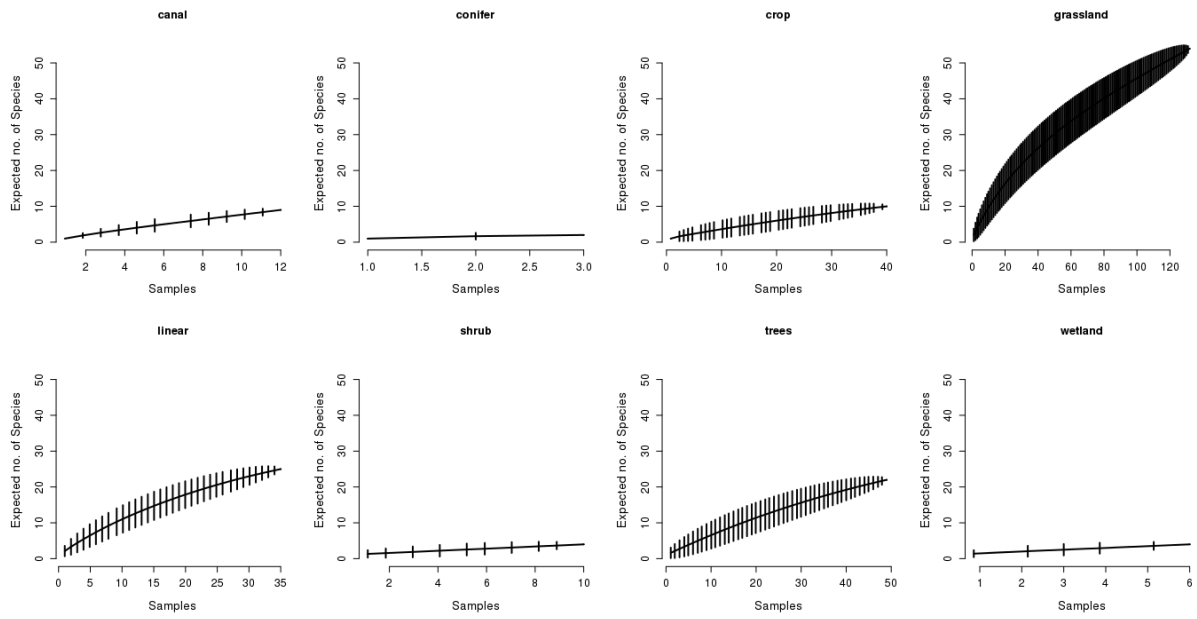
654

655 Fig. 5: Distribution of abundance (A) and species richness (B) of spiders according to the
656 studied habitat types.

657

658 Fig. 6: Relationship between the spiders' abundance and species richness and the most
659 influential environmental variables such as wind (A and B respectively) and minimum
660 vegetation height (C and D respectively). The smoothed line was fitted by local

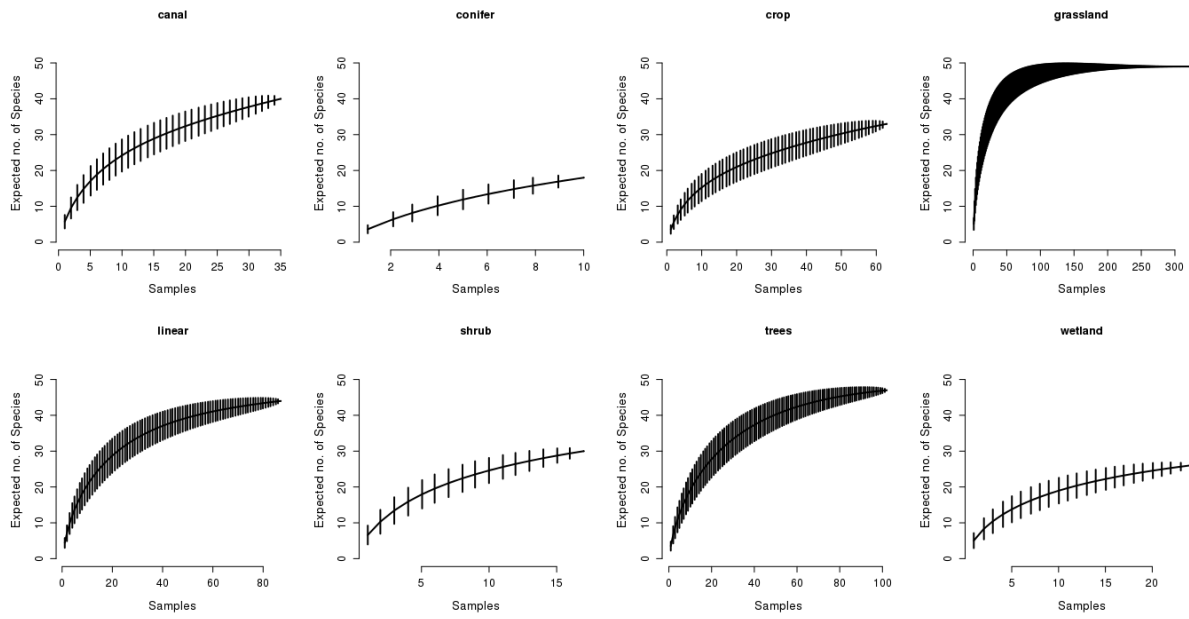
661 polynomial regression techniques (lowess; alpha was set to one to fit closely linear
662 regression).
663



664

665 Fig. 1: The species richness estimation for bees according to Chao estimators (with 95 %
 666 confidence intervals) based on the habitat types on 18 low-input farms in Central
 667 Hungary.

668



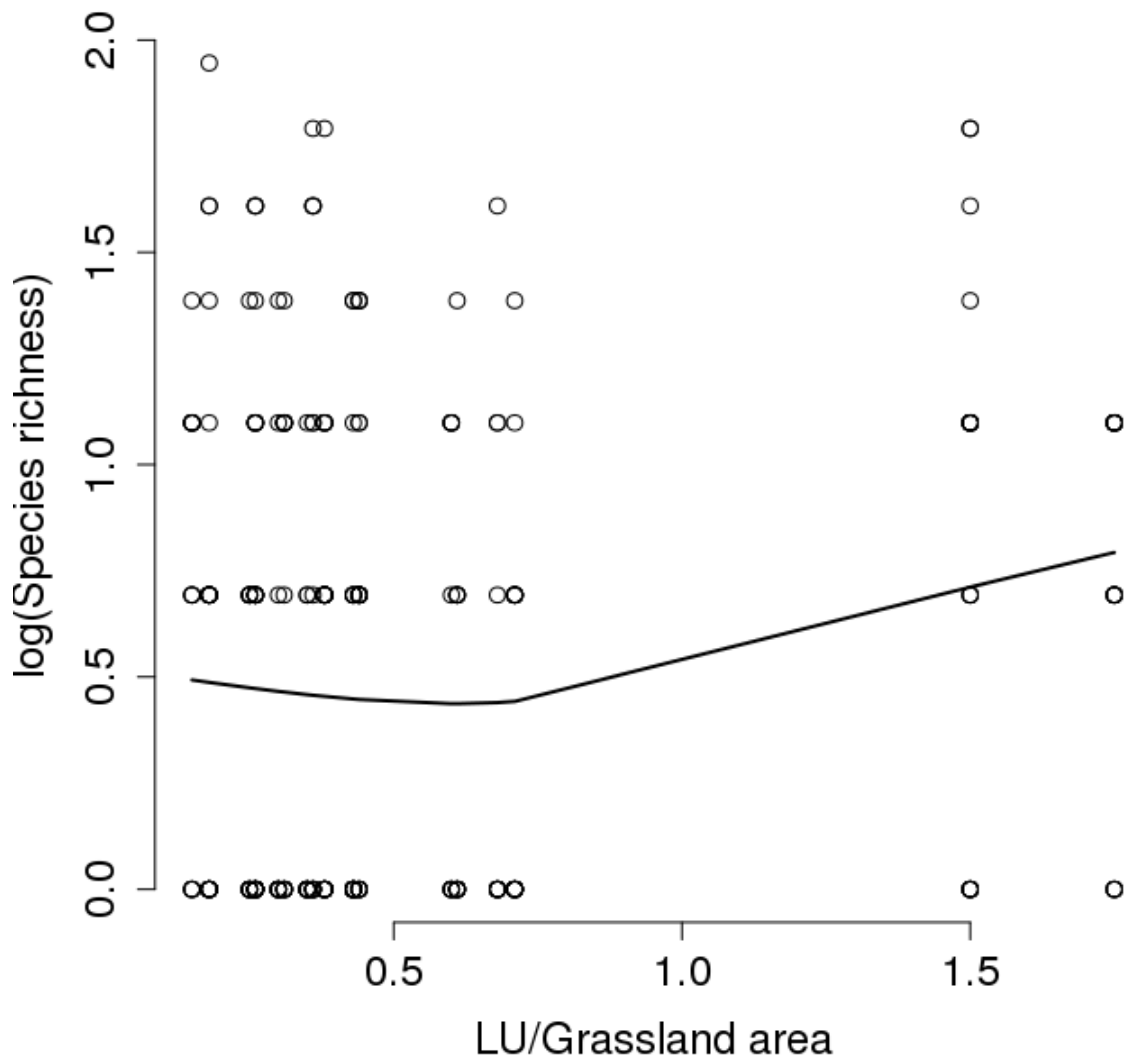
669

670 Fig. 2: The species richness estimation for spiders according to Chao estimators (with 95 %

671 confidence intervals) based on the habitat types on 18 low-input farms in Central

672 Hungary.

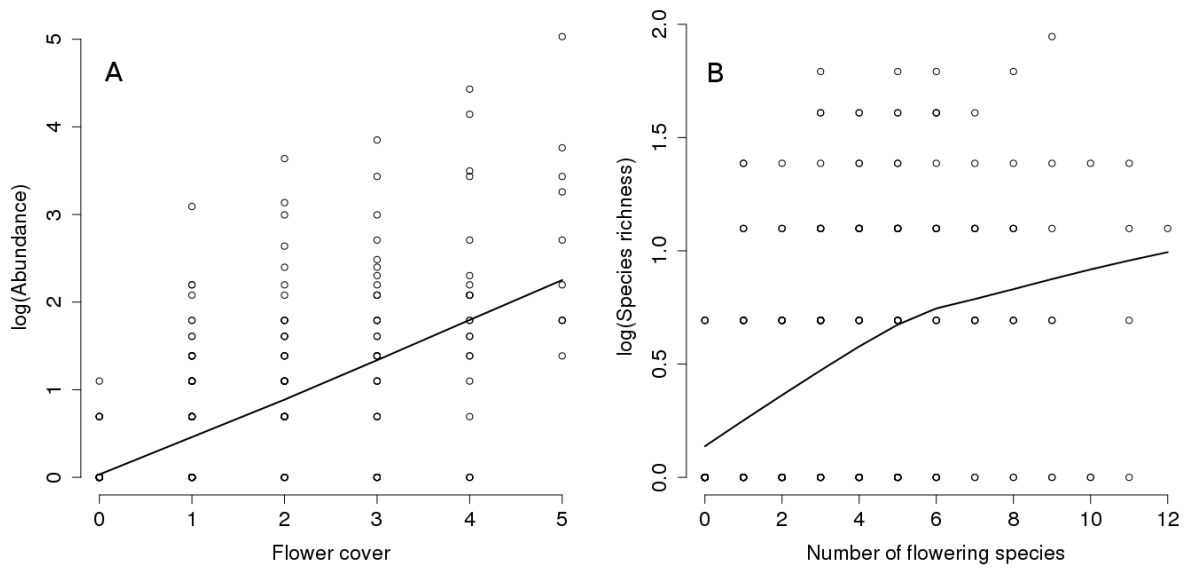
673



674

675 Fig. 3: The relationship between grazing intensity (livestock unit/ ha grassland) and species
 676 richness of bees on 18 low-input farms in Central Hungary. The smoothed line was fitted
 677 by local polynomial regression techniques (lowess; alpha was set to one to fit closely
 678 linear regression).

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680

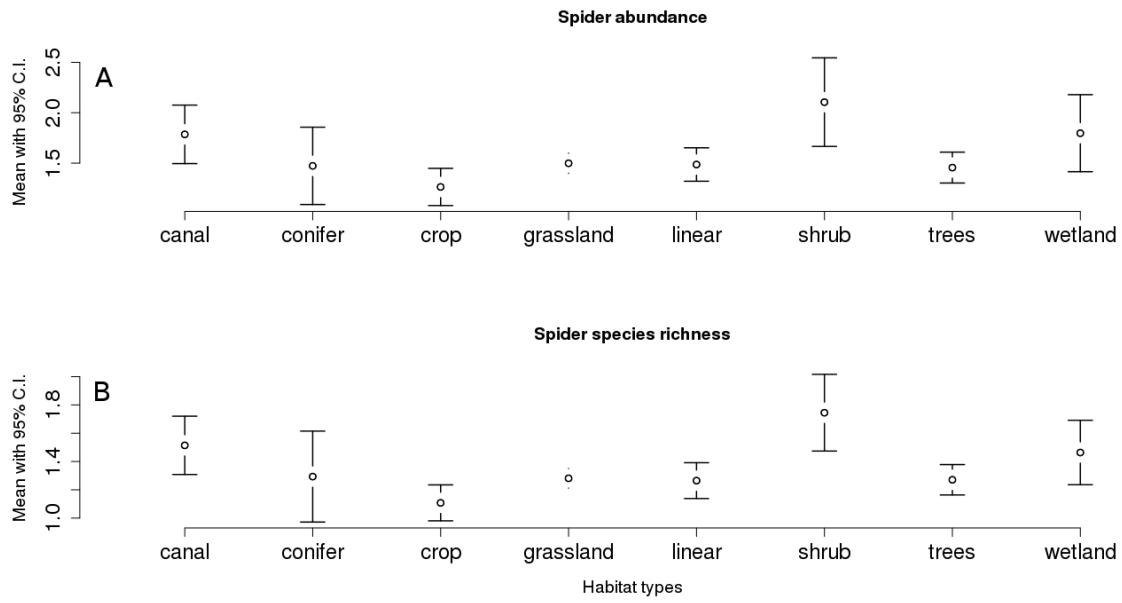
681 Fig. 4: Relationship between the most influential environmental variables and the abundance

682 (A) and species richness (B) of bees on 18 low-input farms in Central Hungary. The

683 smoothed line was fitted by local polynomial regression techniques (lowess; alpha was

684 set to one to fit closely linear regression).

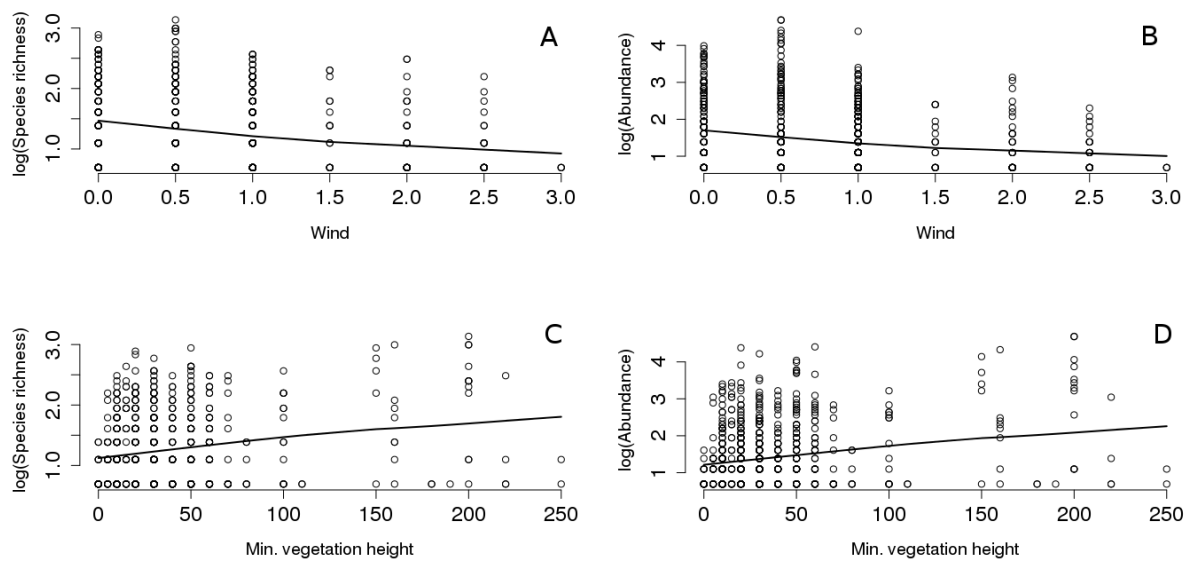
685



686

687 Fig. 5: Distribution of abundance (A) and species richness (B) of spiders according to the
 688 studied habitat types.

689



690

691 Fig. 6: Relationship between the spiders' abundance and species richness and the most
 692 influential environmental variables such as wind (A and B respectively) and minimum
 693 vegetation height (C and D respectively). The smoothed line was fitted by local
 694 polynomial regression techniques (lowess; alpha was set to one to fit closely linear
 695 regression).