



HAL
open science

Effect of habitat spatiotemporal structure on collembolan diversity

Charlène Heiniger, Sébastien Barot, Jean-François Ponge, Sandrine Salmon,
Léo Botton-Divet, David Carmignac, Florence Dubs

► **To cite this version:**

Charlène Heiniger, Sébastien Barot, Jean-François Ponge, Sandrine Salmon, Léo Botton-Divet, et al.. Effect of habitat spatiotemporal structure on collembolan diversity. *Pedobiologia*, 2014, 57 (2), pp.103-117. 10.1016/j.pedobi.2014.01.006 . hal-00964875v2

HAL Id: hal-00964875

<https://hal.science/hal-00964875v2>

Submitted on 25 Mar 2014

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

1 **Effect of habitat spatiotemporal structure on collembolan diversity**

2 **C. Heiniger^{a*}, S. Barot^b, J. F. Ponge^c, S. Salmon^c, L. Botton-Divet^a, D. Carmignac^d, F. Dubs^a**

3
4 ^aIRD, UMR BIOEMCO, Centre France Nord, 93143 Bondy, France

5 ^bIRD, UMR BIOEMCO, ENS. 75006 Paris, France

6 ^cMNHN, UMR 7179, 91800 Brunoy, France

7 ^dENS, UMR BIOEMCO, ENS. 75006 Paris, France

8 Key words: Landscape structure and dynamics, collembolan diversity, colonization credit, patch age,
9 patch isolation, forest vs. agriculture.

10 **ABSTRACT**

11 Landscape fragmentation is a major threat to biodiversity. It results in the transformation of
12 continuous (hence large) habitat patches into isolated (hence smaller) patches, embedded in a matrix
13 of another habitat type. Many populations are harmed by fragmentation because remnant patches do
14 not fulfil their ecological and demographic requirements. In turn, this leads to a loss of biodiversity,
15 especially if species have poor dispersal abilities. Moreover, landscape fragmentation is a dynamic
16 process in which patches can be converted from one type of habitat to another. A recently created
17 habitat might suffer from a reduced biodiversity because of the absence of adapted species that need a
18 certain amount of time to colonize the new patch (e.g. direct metapopulation effect). Thus landscape
19 dynamics leads to complex habitat spatiotemporal structure, in which each patch is more or less
20 continuous in space and time. In this study, we define habitat spatial structure as the degree to which a
21 habitat is isolated from another habitat of the same kind and temporal structure as the time since the
22 habitat is in place. Patches can also display reduced biodiversity because their spatial or temporal
23 structures are correlated with habitat quality (e.g. indirect effects). We discriminated direct meta-
24 community effects from indirect (habitat quality) effects of the spatiotemporal structure of habitats on
25 biodiversity using Collembola as a model. We tested the relative importance of spatial and temporal

58
59 * Corresponding author. Tel. : + 33 (0) 642135945 ; fax +33 (0) 148025970
60 E-mail adress : charlene.heiniger@ird.fr (C. Heiniger).
61
62
63
64
65

26 structure of habitats for collembolan diversity, taking soil properties into account. In an agroforested
27 landscape, we set up a sampling design comprised of two types of habitats (agriculture vs forest), a
28 gradient of habitat isolation (three isolation classes) and two contrasting ages of habitats. Our results
29 showed that habitat temporal structure is a key factor shaping collembolan diversity. A reduced
30 diversity was detected in recent habitats, especially in forests. Interactions between temporal
31 continuity and habitat quality were also detected by taking into account soil properties: diversity
32 increased with soil carbon content, especially in old forests. Negative effects of habitat age on
33 diversity were stronger in isolated patches. We conclude that habitat temporal structure is a key factor
34 shaping collembolan diversity, while direction and amplitude of its effect depend on landuse type and
35 spatial isolation.

36 **1. Introduction**

37 Habitat fragmentation is well known to be a major threat to biodiversity in many
38 macroorganisms (Saunders et al., 1991; Tilman, 1994; Tilman et al., 1994; Finlay et al., 1996;
39 Stratford and Stouffer, 1999; Cushman, 2006; Mapelli and Kittlein, 2009; Krauss et al., 2010).
40 Biodiversity is not only driven by local environmental conditions, but also by spatial processes
41 (Hanski, 1994; Ettema and Wardle, 2002; Holyoak et al., 2005). It is now largely recognized that
42 ecological processes shaping communities occur at least at two distinct organisation levels (Shmida
43 and Wilson, 1985; Ricklefs, 1987; Wardle, 2006). (1) Regional processes occur since habitats within a
44 landscape are interconnected by dispersal, which gives birth to meta-community dynamics (Gilpin and
45 Hanski, 1991; Hubbell, 2001; Leibold et al., 2004). At regional scale, an increase in habitat spatial
46 connectivity increases the probability of a species to reach an unoccupied habitat and thus may
47 enhance local diversity (Bailey, 2007; Brückmann et al., 2010). (2) Local factors such as
48 environmental conditions and competition between organisms act as filters enabling species to
49 maintain a viable population in a patch of habitat (Decaëns et al., 2011; Petit and Fried, 2012) and thus
50 reduce local diversity. Within this framework, patches are defined as spatial units of habitat differing
51 from the surrounding area (Forman and Godron, 1986). Even though patches may display an internal
52 heterogeneity at finer scale, e.g. microhabitat (Leibold et al., 2004), they contain a single type of

53 habitat defined by relatively homogeneous biotic and abiotic factors such as temperature, humidity or
54 vegetation cover.

55 In fragmented landscapes, biodiversity can be locally reduced when patches become too small
56 to sustain a species or when species are not mobile enough to efficiently recolonize patches where they
57 went extinct. Characteristics of habitat patches (e.g. vegetation cover, configuration, shape and area)
58 also have various effects on biodiversity (Forman, 1995; Tanner, 2003; Davies et al., 2005) depending
59 on how the focal group of organisms perceives the surrounding landscape and on its ability to move
60 from a patch to another (Kotliar and Wiens, 1990; Ettema and Wardle, 2002; Tews et al., 2004). While
61 the effects of fragmentation are well documented for aboveground animals such as birds or
62 amphibians (Stratford and Stouffer, 1999; Cushman, 2006) they have hardly been studied in soil
63 organisms (Decaëns, 2010). However, soil fauna is the most species-rich component of ecosystems
64 (André et al., 1994), known to provide many ecosystem services (Lavelle et al., 2006) that could be
65 negatively impacted by habitat fragmentation. Soil invertebrates are known to have a low active
66 mobility because of their small body size (Finlay et al., 1996; Hillebrand and Blenckner, 2002) and
67 because it is more difficult to move within the soil than above it. For these reasons they should not
68 react to habitat fragmentation in the same way as larger aboveground animals. We tackle here these
69 general issues using Collembola as a model and focussing on the impact of habitat spatiotemporal
70 structure on their diversity. Collembola constitute a relevant model because (1) they are very abundant
71 in most soils and ecosystems, (2) many species can be found in a single location and (3) collembolan
72 species are known to differ in their dispersal abilities and their level of specialisation for different
73 habitat types (Ponge et al., 2006; da Silva et al., 2012).

74 Recent insights into the influence of landscape structure on collembolan diversity showed that
75 at patch scale, collembolan (alpha) diversity in forests may respond negatively to habitat diversity at
76 landscape scale (Ponge et al., 2003; Sousa et al., 2006). In these cases, the decrease in local or alpha
77 diversity was attributed to habitat fragmentation occurring in diverse landscapes. Indeed, patch
78 isolation which is most of the time increased in fragmented habitats, may reduce the chances of
79 colonization by species, especially if these have poor dispersal ability (Hewitt and Kellman, 2002). In
80 contrast, Querner (2013) showed that landscape heterogeneity may increase local (alpha) collembolan

1 81 diversity in oilseed rape fields (i.e. in agricultural habitats). In this case, species are thought to express
2 82 preferences for different habitat types so that regional (gamma) diversity is increased by habitat
3
4 83 heterogeneity (Vanbergen et al., 2007). Since these preferences are not strict, and species move
5
6 84 between patches, habitat heterogeneity in the neighbouring landscape would also increase diversity at
7
8 85 patch scale (alpha diversity). These results suggest that it is difficult to predict a priori the impact of
9
10 86 habitat isolation on local (alpha) species diversity and that this impact depends on the ecosystem under
11
12 87 investigation. Here, we compare the effect of patch isolation in two broad habitat types, open vs.
13
14 88 closed vegetation, within the same landscape.

17 89 Most empirical studies on metacommunity dynamics assume that local communities have
18
19 90 reached equilibrium at sampling time. However some authors suggested that the time elapsed, since
20
21 91 the first species successfully colonized a patch of habitat, is essential for the understanding of
22
23 92 observed diversity patterns (Mouquet et al., 2003). These authors assume that communities at the first
24
25 93 stages of the assembly process are unsaturated because only a subset of the regional species pool has
26
27 94 yet been able to colonize the patch. Besides spatial structure, patch temporal structure may thus also
28
29 95 influence collembolan alpha diversity. Ponge et al. (2006) showed that landscape heterogeneity may
30
31 96 come with a more dynamic patch temporal structure. They suggest that regions that include more
32
33 97 diverse habitat types may also include more patches of habitat that have experienced a recent change
34
35 98 in land use (e.g. patches that switched from forest to agriculture or the reverse, and thus are not
36
37 99 continuous through time). This may have subsequently reduced collembolan diversity at patch scale
38
39 100 (alpha diversity). In this sense, the lack of diversity observed in most heterogeneous landscape might
40
41 101 be due to patch history, i.e. to temporal discontinuity, rather than to patch spatial structure, i.e.
42
43 102 fragmentation.

48 103 Another source of complexity for understanding the influence of habitat structure on diversity
49
50 104 patterns is that patch characteristics (age, spatial isolation, land use type) may influence local
51
52 105 communities either directly or indirectly. They directly impact local communities through their effect
53
54 106 on meta-community dynamics (Driscoll et al., 2012). Patch characteristics may also impact
55
56 107 communities through complex links between landscape dynamics and local environmental properties
57
58 108 (Wu and Loucks, 1995). For example, isolation and age of a patch can impact local microclimatic
59
60
61
62
63
64
65

109 conditions (Saunders et al., 1991; Magura et al., 2003), and increased edge effects in isolated patches
110 can be responsible for changes in soil properties. In this case, patch spatial structure would be
111 responsible for changes in local conditions which would consequently affect local (alpha) diversity
112 (e.g. indirect effect). Conversely, pre-existing local conditions may impact land use changes (e.g. if the
113 forest soil is fertile, the forest is more likely to be turned into a field). Such direct and indirect effects
114 must be disentangled to determine the effects of landscape structure on local communities.

115 In the present study, we intend to disentangle the relative effects of spatial vs. temporal
116 continuity of habitats on collembolan alpha diversity in both agricultural and forest habitats. We will
117 assess the effect on diversity of 1) temporal continuity of habitats (temporal structure), 2) spatial
118 isolation of habitats (spatial structure), 3) interaction of temporal and spatial habitat structures, 4) local
119 environmental conditions (land use and soil) and whether they depend on habitat spatiotemporal
120 structure (indirect effect), and 5) forest and agricultural habitats.

121 According to the rationale above (Ponge et al., 2006), we expect (H1) stable habitats (i.e. old
122 or temporally continuous patches) to support a higher alpha diversity than habitats that have been
123 disturbed in the past decades (i.e. recent or temporally discontinuous patches). Besides being
124 considered as stable habitats, forests display a wider variety of niches than agricultural land due to the
125 quality of their soils and humus: forests have a well-developed humus layer (often including
126 fragmented OF horizons and sometimes humified OH horizons) that is absent in open or agricultural
127 habitat (Hågvar, 1983 ; Ponge, 2000). Additionally, soil carbon content and moisture are higher in
128 forest than in agricultural habitats (Batlle-Aguilar et al., 2011), thereby favouring Collembola given
129 the well-known requirements of these animals in water and organic matter (Hopkin, 1997). We thus
130 expect (H2) to find a higher diversity and a higher abundance of Collembola in forested habitats. We
131 think that vegetation structure in agricultural habitats makes dispersal easier than in forests because
132 passive dispersal vectors such as wind are more efficient in open than in closed vegetation (Morecroft
133 et al., 1998). We thus expect (H3) that spatiotemporal continuity will have a lower effect in
134 agricultural habitats when compared to forests.

135 2. Material and methods

136 2.1. Study site

137 Sampling took place in the northern part of the Morvan Regional Natural Park (Burgundy,
 138 Center-East France). The study area is located in the northern part of the Park (523 6000 – 525 2000
 139 N, 573 800 - 588 800 E ; WGS84, UTM 31N) and represents an area of 16 x 15 km. The climate is
 140 submontane-atlantic with continental influence (mean annual rainfall 1000 mm and mean temperature
 141 9°C). The bedrock is made of granite and soils are mostly acidic (Cambisols, IUSS Working Group
 142 WRB 2006). We selected this region because it displays diverse habitat spatiotemporal structures and
 143 relatively homogeneous soil conditions among all habitat of the same type. The region is rural, with
 144 intensive to extensive agriculture (55%) and forestry (45%). From the beginning of the twentieth
 145 century Douglas fir [*Pseudotsuga menziesii* Mirbel (Franco)] and Norway spruce [*Picea abies* (L.)
 146 Karst.] have been intensely planted for saw-wood production and coniferous stands have progressively
 147 replaced the formerly dominant deciduous stands. However, large areas of oak [*Quercus petraea*
 148 (Matt.) Liebl.] and beech (*Fagus sylvatica* L.) forest still subsist. Nowadays, agricultural areas consist
 149 of permanent pastures (40%), hay meadows (40%) and crops (20%). Forested areas are comprised of
 150 planted coniferous stands (45%) and deciduous stands (mostly *Ilici-Fagenion*) (55%). In this region,
 151 the landscape has experienced a dynamic period (1962 to present) due to agricultural abandonment
 152 and European subsidies that encouraged farmers to convert meadows into plantation forests. This
 153 afforestation created many recent forest patches.

154 2.2. Sampling design

155 Our sampling design was comprised of 60 sites (28 forested and 32 agricultural) classified in
 156 12 combinations of three habitat descriptors: habitat type (HT), temporal continuity (TC) and spatial
 157 isolation (SI). For each spatiotemporal combination we sampled 3 to 9 replicates (Appendix A).

158 2.2.1. Habitat type

159 Collembolan communities are likely to depend on the dichotomy between open and closed
 160 vegetation (Ponge et al., 2003; Vanbergen et al., 2007). Hence we decided to split the landscape into

161 two major habitat types: forest and agricultural land. Thus, sampled sites can either be meadows,
162 pastures, crops, or Christmas tree plantations for agricultural habitats and coniferous or deciduous
163 stands for forest habitats (Appendix A). Christmas tree plantations might at first be thought as forest
164 habitats. However, they display many characteristics of agricultural habitats: absence of litter and
165 developed humus profiles (due to low stature and density of Christmas trees), use of ploughing and
166 pesticides, i.e. same characteristics as agricultural land. They are generally constituted by no more
167 than five-year-old trees and have been shown to support collembolan communities typical of
168 agricultural habitats (Ponge et al., 2003). It is well known that the transition from deciduous to
169 coniferous stands implies an abrupt change in soil physicochemical properties (pH, humus form, etc.)
170 (Gauquelin et al., 1996; Augusto et al., 2003). However, Ponge et al. (2003) showed that in the
171 Morvan region, collembolan communities do not differ between coniferous and deciduous stands,
172 contrary to often-reported detrimental effects of coniferous plantations, mostly ascribed to changes in
173 humus form and soil acidity (Cassagne et al., 2004; Hasegawa et al., 2009). In any case, collembolan
174 communities of both forest types differ less from each other than they differ from agricultural
175 communities. The absence of pronounced reaction of collembolan communities to tree species
176 composition was also observed in similar acidic soil conditions in Germany (Salamon and Alpei,
177 2009). This is explained by the fact that on acidic bedrocks of the studied region, similar humus forms
178 with thick litter layers and strong soil acidity (moder) develop under both stand types (Ponge et al.
179 1993). It has been shown that *Collembola* mostly feed on microorganisms and animal faeces and
180 rarely consume directly leaves or needles (Ponge, 1991; Caner et al., 2004). As such, they are most of
181 all influenced by humus forms whatever the composition of forest canopies (Ponge, 1993).
182 Additionally, coniferous tree species planted in the Morvan region (Douglas fir, silver fir and more
183 rarely Norway spruce) have a more nutrient rich litter and do not acidify the soil to the same extent as
184 pines (Augusto et al., 2003). This allowed us to group both forest types into a single broader category.
185 We avoided sampling in special habitats such as humid areas or clear-cuts so as to minimize the
186 influence of particular environmental conditions within forests and agricultural land.

2.2.2. Habitat temporal continuity

188 In order to assess the temporal continuity of each habitat within the focus area, we
1 implemented a dynamic cartography (i.e. picturing both spatial and temporal continuity) using aerial
2 189 photographs. In total, about two hundred aerial photographs (IGN, France) were required from 1948 to
3 190
4 191 2008 with a photograph taken approximately every five year. We categorised each habitat into two age
5 192
6 193 classes (Appendix A). Those in place at least since 1948 were classified as old. Recent forests were
7 194
8 195 agricultural habitats until conversion to forest 30 to 40 years ago (in our classification, a change from
9 196
10 197 deciduous to coniferous stand is not a temporal discontinuity). Recent agricultural habitats were
11 198
12 199 forests until conversion to agricultural land 20 to 30 years ago. Studied local collembolan communities
13 200
14 201 were thus included in habitats that were homogeneous in type and age. In the context of this study, we
15 202
16 203 considered that a patch is not only a continuous block of the same habitat type but also a continuous
17 204
18 205 part of the same habitat type over time, meaning that we mapped four types of patch: old forest, recent
19 206
20 207 forest, old agricultural and recent agricultural.

200 2.2.3. Habitat spatial isolation

201 Using the previously mentioned cartography, we selected sampling points in both habitat types
202 (agriculture and forest) and habitat temporal continuity classes (recent and old) and then categorized
203 the landscape mosaic in a buffer zone of 300 m radius around sampling points (Appendix A). Two
204 parameters were considered to define habitat spatial isolation: edge contrast and dominant age of the
205 matrix. Edge contrast measures the magnitude of the difference between adjacent habitat types. It is
206 calculated as the percent edge of the habitat (containing the sampling point) shared with an opposite
207 habitat within a 300 m radius. Opposite habitat was forest for agricultural land and vice versa. Of
208 course, edge contrast is nil or close to nil for isolation class 0. The matrix was defined by the
209 proportion of the dominant habitat type of a given age which occupies the matrix around the sampling
210 point.

211 Isolation class 0 was comprised of large continuous habitats, i.e. larger than the scale of
212 observation around the sampling point (the 300 m buffer zone). Thus sampling points of isolation class
213 0 (whether in recent or old habitat) were entirely included in a matrix made of the same type of habitat
214 300 m around it (except for one recent forest patch that shared 11% of its edge with an old agricultural

215 patch). The matrix surrounding sampling points of isolation class 0 was mostly old (continuous or
 216 non-isolated spatial structure).

217 For isolation classes 1, criteria of spatial isolation slightly differed between forest and
 218 agricultural habitats. Agricultural patches of isolation class 1 shared a single edge with an old forest.
 219 The matrix surrounding them was mostly comprised of old (and less frequently recent) forest habitats.
 220 Forest patches of isolation class 1 corresponded to a particular situation that we repeatedly found in
 221 the studied region, e.g. some remnant forest patches in place since 1948 (i.e. old) that have been
 222 reconnected by a (recent) forest patch to another old forest patch since the last 30-40 years. Old forests
 223 of isolation class 1 shared 25 to 83 % of their edge with an old agricultural land and the rest with a
 224 recent forest. Recent forests of isolation class 1 were the “reconnecting patches”, sharing 19 to 69% of
 225 their edge with an old agricultural land and the rest with an old forest. The matrix surrounding
 226 sampling points of isolation class 1 was comprised of old and recent habitats.

227 For isolation class 2, spatial isolation also slightly differed between forest and agricultural
 228 habitats. Forest patches of isolation class 2 were remnant patches, entirely surrounded by an old
 229 agricultural matrix. They have been completely isolated since they are in place. Old and recent
 230 agricultural patches of isolation class 2 shared respectively 60 to 100 % and 45 to 80% of their edge
 231 with a forest. The matrix surrounding agricultural patches of isolation class 2 was mostly old. Isolation
 232 class 1 was considered to be less isolated than isolation class 2 because with the appearance of recent
 233 habitats in the surrounding matrix of isolation class 1, some newly created habitats were of the same
 234 type as the one located at the sampling point, originating in a less isolated context.

235 In forests as well as in agricultural land, patches of isolation classes 1 and 2 were sampled at
 236 least 10 m (but no more than 50 m) away from the opposite habitat type edge.

237 *2.3. Collection of fauna and soil data*

238 Sampling took place from June 27 to July 9, 2010. Each site was sampled for litter/soil
 239 mesofauna using a cylindrical soil corer (5 cm diameter x 7 cm depth, one sample at each sampling
 240 site). We thus sampled exactly the same volume of soil at each site, meaning that values of species
 241 density, i.e. the number of species per unit area sensu Gotelli and Colwell (2001) here presented

242 correspond to the number of species found over 0.2 dm². Litter/soil were brought back to the
 1 laboratory within a week and placed in a Berlese dry-funnel extractor for 10 days. Animals were
 2 243 collected and stored in 70% ethyl alcohol until identification. Collembola were mounted, cleared in
 3
 4 244 chloral-lactophenol and identified to species level under a light microscope (400x magnification),
 5
 6 245 according to Hopkin (2007), Potapow (2001), Thibaud et al. (2004) and (Bretfeld, 1999). A list of
 7
 8 246 species is given in Appendix B.
 9
 10 247

11
 12
 13 248 We also sampled soils (organo mineral horizon, between 0 and 10 cm) in each site in order to
 14
 15 249 characterize soil physicochemical properties at each sampling site. Three samples were taken around
 16
 17 250 soil fauna samples and were pooled together. Soils were air-dried and sieved to 2 mm before
 18
 19 251 measuring total carbon (C_{tot}) and total nitrogen (N_{tot}) contents (gas chromatography), pH (H₂O),
 20
 21 252 bioavailable phosphorus (Olsen method) and cation exchange capacity (CEC). Additionally, the top
 22
 23 253 five soil centimetres were sampled using a Burger cylinder (0.1 L volume) and immediately packed in
 24
 25 254 waterproof bags in order to determine soil moisture and bulk density. The humus form was
 26
 27 255 characterized according to Brêthes et al. (1995) and the Humus index was calculated according to
 28
 29 256 Ponge et al. (2002), and was used as a proxy of litter amount and recalcitrance to decay (Ponge et al.,
 30
 31 257 1997; Ponge and Chevalier, 2006).
 32
 33
 34
 35

36 258 *2.4. Statistical analyses*

37
 38
 39 259 The following diversity indicators were calculated for each site: species density (sensu Gotelli
 40
 41 260 and Colwell, 2001), i.e. the actual number of species found in each sample, species richness (sensu
 42
 43 261 Gotelli and Colwell, 2001) i.e. the local number of species estimated to be found in a smaller sample
 44
 45 262 containing 25 % of the mean individual density (i.e. 57 individuals), Shannon index, dominance
 46
 47 263 (relative frequency of the most abundant species), and abundance (total number of individuals).
 48
 49

50 264 These diversity indicators were analysed using linear models (type III sums of squares used
 51
 52 265 for unbalanced design and because significant interactions were expected, see Appendix B), testing for
 53
 54 266 habitat type (HT), spatial isolation (SI), and temporal continuity (TC) effects as well as effects of all
 55
 56 267 interactions between these factors. To fulfil linear model assumptions, dominance had to be log-
 57
 58 268 transformed. In order to detect possible effects of habitat spatiotemporal structure (regional factors) on
 59
 60
 61
 62
 63
 64
 65

269 its soil quality that could influence diversity indicators, we tested the effect of habitat descriptors on
1 soil physicochemical properties. Most of natural or log-transformed data fulfilled the assumptions of
2 270
3 linear models. When this was not the case, we used generalized linear models with a Gamma link
4 271
5 function or a Poisson link function. All possible correlations (Pearson) between diversity indicators
6 272
7 and soil properties were calculated and tested. We also calculated and tested these correlations in both
8 273
9 habitat types separately.
10 274

13 275 Finally, we constructed complete linear models, testing the effects of the three habitat
14 descriptors on diversity indicators and including most important soil parameters as covariates, together
15 276
16 with all their interactions. Since there are many combinations of habitat descriptors and many soil
17 277
18 parameters it was not possible to include all of them and their interactions in a single model. Therefore
19 278
20 we focussed our analysis on the two soil parameters that were the most correlated with diversity and/or
21 279
22 that were significantly affected by habitat descriptors, i.e. Ctot and pH. These two variables can be
23 280
24 considered as proxies for two main physico-chemical factors which impact collembolan communities
25 281
26 at two different scales (species or community): Ctot is a proxy for general habitat and resource
27 282
28 availability and thus determines the total abundance; and pH is a proxy for local environmental filter
29 283
30 which selects species within communities, since several collembolan species are only adapted to low
31 284
32 or high soil acidity (Ponge, 1993; Salmon, 2004). We analysed two models testing separately for the
33 285
34 effect of these two variables, the three habitat descriptors and all their interactions (two-, three- and
35 286
36 four-way interactions). Simple effects of variables and interactions that were kept in each final model
37 287
38 were selected using an automatic selection procedure based on AIC (procedure step, with backward
39 288
40 direction, Bodzogan, 1987; Posada and Buckley, 2004). Combinations of habitat descriptors were
41 289
42 compared using least square means and associated multiple comparisons of means (Tukey). All
43 290
44 statistical analyses were performed using Mass, car, vegan and Lsmeans packages of R software (R
45 291
46 Development Core Team, 2010).
47 292
48

53 293 Altogether, these analyses enabled us to discriminate between direct and indirect effects. The
54 first type of models (testing the effect of the three habitat descriptors on diversity indicators) includes
55 294
56 both direct and indirect effects of spatiotemporal structure. If the second type of models (testing the
57 295
58 effect of the three habitat descriptors on soil properties) reveals significant effects, it means that
59 296
60
61
62
63
64
65

297 indirect effects are likely to occur. Even if they do not, little differences in soil properties between
298 habitat spatiotemporal structures could still slightly influence collembolan diversity. This is why we
299 constructed a third type of models, testing for the effect of the three habitat descriptors on diversity
300 and including an influential soil property in the same model. If simple effects of habitat descriptors are
301 significant, it means that they have a direct effect on diversity. However, if simple effects of soil
302 properties are significant, indirect effects are occurring. The comparison of the first and third types of
303 models enabled us to assess whether direct or indirect effects were occurring or if both effects were
304 present at the same time (Fig. 1).

305 **3. Results**

306 *3.1. Effects of the three habitat descriptors on collembolan diversity*

307 In total, 6339 individuals were identified to 70 species. We found 3639 from 57 species in
308 forest and 2700 individuals from 39 species in agricultural habitats. In forests, 44 species were found
309 in old sites and 42 in recent sites. In agricultural land, 36 species were found in old sites and 24
310 species in recent sites. Thirteen species were present in recent forests but not in old forests and 15
311 species were present in old forests and not in recent forest. In agricultural sites, 15 species were
312 present in old but not in recent sites and only 3 species not present in old sites were found in recent
313 sites. Rarefaction curves (Fig. 2) showed that our sampling was nearly exhaustive for both habitat
314 types whatever old or recent, and that forests (both recent and old) harbored more species in total
315 (gamma diversity) than agricultural land (old > recent). Statistical models implemented on diversity
316 indicators (Model type 1 in Fig. 1, Table 1) showed that habitat type (HT) exerted an effect on species
317 density, species richness and total abundance. Species density, species richness as well as abundance
318 of Collembola were higher in forest compared to agricultural habitats (on average respectively 11.4
319 versus 8.6 for species density and 9.1 versus 7.9 for species richness). However, no effect of HT was
320 detected either on Shannon index or on dominance. Furthermore, only Shannon index, species richness
321 and species density responded to habitat temporal continuity (TC), old habitats showing higher values
322 than recent habitats (on average 10.8 versus 9.3 for species density, 1.8 versus 1.6 for Shannon index,
323 and 9.1 versus 7.5 for species richness). Multiple comparisons among means (Tukey) showed that this

324 effect was mainly due to differences between old and recent forests (on average respectively 13.1 and
 1 325 9.4 for species density, 1.9 and 1.5 for Shannon index, and 10.2 versus 7.9 for species richness),
 2 326 whereas old and recent agricultural lands showed closer values (on average respectively 9.1 versus 7.7
 3 327 for species density, 1.7 versus 1.5 for Shannon index, and 8.3 versus 7.1 for species richness) (Fig. 3).
 4 328 Both coniferous and deciduous forests displayed significantly lower values for species density, species
 5 329 richness and Shannon index in the recent age class compared to the old one (data not shown). We did
 6 330 not detect any effects of spatial isolation (SI) on diversity indicators, either in forest or in agricultural
 7 331 habitats, and no interaction between habitat descriptors (Fig. 4).

18 332 *3.2. Effects of the three habitat descriptors on soil characteristics*

21 333 Most soil properties were significantly different according to HT. Models testing the effect of
 22 334 HT on soil properties showed that CEC, Ctot, C:N and Humus index had significantly higher values in
 23 335 forest than in agricultural habitats, whereas soil density, soil moisture and pH showed opposite trends
 24 336 (Model type 2 in Fig. 1, Tables 2 and 3). On the other hand, only a few of them, i.e. pH, P and C:N,
 25 337 responded to the two other habitat descriptors (SI and TC). Soil pH, bioavailable phosphorus, and
 26 338 carbon to nitrogen ratio (C:N) showed differences between isolation classes. Soil pH was higher in
 27 339 isolation class 1 (on average 4.3) than in isolation classes 0 and 2 (on average respectively 3.7 and
 28 340 3.8), but only in forest patches, resulting in a significant interaction between HT and SI. Bioavailable
 29 341 phosphorus content was higher in isolation class 2 than in isolation classes 0 and 1 (on average
 30 342 respectively 74, 55 and 44 mg/kg). Only C:N was significantly impacted by TC and was higher in old
 31 343 habitats than in recent ones. Moreover, for C:N, many interactions between habitat descriptors were
 32 344 significant (Table 3). Multiple mean comparisons showed that these effects were mostly due to non-
 33 345 isolated old forest habitats (isolation class 0) that showed significantly higher values (on average 20.6)
 34 346 than every other forest habitats whatever their SI or TC (in old habitats: isolation 1, 15.8; isolation 2,
 35 347 16; in recent habitats: isolation 0, 14.5; isolation 1, 14.3; isolation 2, 13.8). In agricultural land, no
 36 348 effect of the spatiotemporal descriptors on C:N was detected.

349 *3.3. Effects of the three habitat descriptors and soil characteristics on collembolan diversity*

1
2 350 When implemented on data from both forest and agricultural habitats, no correlations between
3
4 351 Shannon index and soil properties were significant (Fig 5). On the other hand, many correlations
5
6 352 between species density and soil properties were significant at 5% level. Species density was
7
8
9 353 positively correlated with CEC, Ctot, C:N and Humus Index (0.37; 0.38; 0.43 and 0.44, respectively)
10
11 354 and negatively correlated with bulk density and pH (-0.3 and -0.38, respectively) (Fig 5). Only
12
13 355 positive correlations between species richness and Ctot, C:N, CEC and Humus index (respectively
14
15 356 0.30, 0.27, 0.28 and 0.33) were observed. However, most of these correlations reflected the above-
16
17
18 357 mentioned differences in soil properties between forest and agricultural habitats. When implemented
19
20 358 separately in the two HTs no correlations with species density, species richness or Shannon index were
21
22 359 significant in agricultural habitats. In forests, only Ctot, C:N (0.4 and 0.44, respectively) and pH (-
23
24 360 0.39) were significantly correlated to species density and only C:N was significantly correlated to
25
26
27 361 species richness (0.40), but none were correlated to the Shannon index (Fig. 5).

28
29 362 As C:N was highly correlated with both Ctot and pH (0.61, $p < 0.0001$ and -0.69, $p < 0.0001$
30
31 363 respectively), while Ctot and pH were also correlated (0.43, $p < 0.001$) but to a lesser extent than to
32
33 364 C:N, we decided to include only Ctot and pH as covariates in two distinct global linear models testing
34
35 365 their effect and that of habitat descriptors on species density, species richness and Shannon index and
36
37
38 366 we implemented a procedure of automatic selection based on AIC. Both models (Ctot and pH) did not
39
40 367 detect any significant effect of any variable on the Shannon index. Table 4 displays the results of
41
42 368 models analyzing the effects of habitat descriptors and either Ctot or pH on species density and species
43
44
45 369 richness. The Ctot models showed that there was no simple effect of habitat descriptors either on
46
47 370 species density or on species richness, whereas pH models showed significant effects of HT and TC
48
49 371 on species density and of TC on species richness. Additionally, pH and Ctot models revealed the effect
50
51 372 of several significant interactions. The pH model showed significant effects of the interactions
52
53 373 between HT and TC, between HT and SI and between TC and SI on species density. The Ctot models
54
55
56 374 showed the effect of interactions between HT and SI on species density, and between TC and SI on
57
58 375 both species density and species richness. Indeed, least square means for species density comparison
59
60 376 (Fig. 6a) obtained with the Ctot model showed no difference between both HTs in non-isolated
61
62
63
64
65

377 habitats (isolated class 0) whereas it showed significantly higher values in forest than in agriculture in
 1
 2 378 isolation classes 1 and 2. Moreover, comparisons of least square means for species density showed
 3
 4 379 significantly higher values in old than in recent patches for the isolation classes 1 and 2, while it
 5
 6 380 showed no difference between age classes for the continuous habitats (Fig. 6b). Similarly, least square
 7
 8 381 means for species richness comparison showed significantly higher values in old than in recent patches
 9
 10 382 for the isolation class 2, while it showed no difference between age classes for habitats of isolation
 11
 12 383 class 0 and 1 (Fig. 6c). Soil pH and Ctot did not have a significant effect on their own, neither on
 13
 14 384 species density nor on species richness. The Ctot had significant effects in interaction with HT, and
 15
 16 385 TC on species density. According to estimated model parameters, species density increased more with
 17
 18 386 Ctot in old than in recent habitats. It also increased more with Ctot in forest than in agricultural land.
 19
 20 387 The soil pH had significant effects in interaction with HT, TC and SI on species density and in
 21
 22 388 interaction with TC on species richness. Species density decreased with the pH in forests whereas it
 23
 24 389 increased with the pH in agricultural habitats. Species density and species richness also decreased with
 25
 26 390 the pH in old habitats whereas it did not change according to pH in recent habitats.
 27
 28
 29
 30

391 4. Discussion

392 4.1. Effects of habitat temporal continuity on collembolan diversity

393 The total number of species (gamma diversity) was higher in old than in recent forests.
 394 Species present in recent forests and not in old ones were all rare species (i.e. found in less than 10%
 395 of the records), except for 3 species (*Isotoma viridis*, *Lepidocyrtus cyaneus* and *Lepidocyrtus*
 396 *lignorum*). These species are known as agricultural-preferring species in the studied region (Ponge et
 397 al., 2003, 2006). Furthermore, they were found exclusively in isolation classes 1 and 2. Thus it is
 398 difficult to know whether they were present as relics of the past HT (i.e. agricultural land) or if they
 399 colonized forest patches from the neighboring agricultural matrix. However, the fact they were not
 400 present in the equivalent isolation class of old forests suggests that they benefited from the presence
 401 of a non-saturated community in recent forests. Indeed, the fact that all niches were not necessarily
 402 occupied in recent forests was certainly in favor of the persistence of agricultural-preferring species:
 403 these species might colonize spatially isolated patches but would not be able to persist in the more
 404
 405
 406
 407
 408
 409
 410
 411
 412
 413
 414
 415
 416
 417
 418
 419
 420
 421
 422
 423
 424
 425
 426
 427
 428
 429
 430
 431
 432
 433
 434
 435
 436
 437
 438
 439
 440
 441
 442
 443
 444
 445
 446
 447
 448
 449
 450
 451
 452
 453
 454
 455
 456
 457
 458
 459
 460
 461
 462
 463
 464
 465

404 diverse communities present in old forests of comparable spatial isolation. In agricultural land, the
1
2 405 total number of species was also higher in old than in recent sites. Only three species present in recent
3
4 406 sites were absent in old ones. Among the 15 species that were present in old but not in recent sites,
5
6 407 only three were common species (i.e. present in more than 10% of the records) and no common
7
8 408 species were found in recent but not in old agricultural sites. This shows that even in open habitats
9
10 409 such as agricultural land, the total number of species was reduced in recently created habitats. It is still
11
12 410 difficult to draw any conclusion on differences in the number of rare species, but the fact that some
13
14 411 common species were lacking in recent sites only, allows us to conclude that this absence is due to the
15
16 412 temporal continuity of the habitat and not to some random mechanism.
17
18

19
20 413 Our results show that habitat temporal discontinuity impacts negatively species density,
21
22 414 species richness and Shannon index but not the total abundance of Collembola. For a similar number
23
24 415 of individuals, recent habitats showed fewer species than old habitats whatever the habitat type or the
25
26 416 spatial continuity. This suggests that resources not taken by species still lacking in recent habitats are
27
28 417 used by larger populations of existing species. Among the variables under study, only the carbon to
29
30 418 nitrogen ratio reacted to temporal continuity, but this was due to the particular situation of spatially
31
32 419 continuous (non-isolated) old forests that displayed higher values than every other forest patches
33
34 420 whatever their spatiotemporal structure. All other soil properties (humus type, a proxy of litter amount
35
36 421 and quality, pH, C, N contents and P availability) were similar in old and recent habitats of the same
37
38 422 type, suggesting that habitat temporal continuity has a direct effect on collembolan diversity that does
39
40 423 not depend on soil properties. Within the limits of our analytical protocol (we did not study all
41
42 424 parameters known to influence Collembola, such as for instance the composition of litter and that of
43
44 425 soil microbial communities, nor did we study interactions between Collembola and higher trophic
45
46 426 levels) this confirms our first hypothesis (H1) predicting a lower collembolan diversity in recently
47
48 427 created habitats.
49
50

51
52
53 428 The direct effect of habitat temporal continuity on collembolan diversity is also supported by
54
55 429 the fact that both species density and species richness were impacted by temporal continuity in the pH
56
57 430 model. This means that habitat temporal discontinuity impacts diversity not only by decreasing the
58
59 431 size of the local pool of species (species density) but also by decreasing the number of species that
60
61
62
63
64
65

432 would be present if local soil conditions were strictly similar among all spatiotemporal situations
1
2 433 (species richness). This is supported by the fact that the impact of temporal continuity on Shannon
3
4 434 index must be considered as a direct effect because this index did not show any significant correlation
5
6 435 with soil physicochemical properties.
7

8
9 436 However, the temporal discontinuity of habitat could also be viewed as an index of
10
11 437 disturbance. According to the intermediate disturbance hypothesis we could expect recent habitats to
12
13 438 host low collembolan diversity, habitats of intermediate age to host a higher biodiversity, and
14
15 439 biodiversity to decrease in older habitats because of exclusive competition (Connell, 1978; Molino and
16
17 440 Sabatier, 2001; but see Fox, 2013). Our results did not support this general scheme. This might be due
18
19 441 to the fact that our study focuses on the landscape scale, so that studied habitat were not homogeneous
20
21 442 and host a high diversity of local niches at scales ranging from units to tens square meters (Ponge and
22
23 443 Salmon, 2013). Nevertheless, we cannot exclude that habitats older than the ones we studied, might
24
25 444 display a decrease in collembolan biodiversity, thus supporting the intermediate disturbance
26
27 445 hypothesis. We did not expect old most isolated habitats to display the lowest alpha diversity predicted
28
29 446 by metacommunity theories (Tilman, 1982; Hubbell, 2001) because the oldest habitats we studied
30
31 447 were certainly not old enough to support such patterns. Additionally, within-patch heterogeneity is
32
33 448 most likely to prevent collembolan alpha diversity from decreasing to such low levels.
34
35
36

37
38 449 Most empirical studies testing the effect of landscape structure on diversity patterns do not
39
40 450 take habitat temporal structure into account (Mouquet et al., 2003). We do not have knowledge of any
41
42 451 other study on Collembola that is strictly comparable to ours although collembolan species have been
43
44 452 shown to be affected by habitat successional stage (Setälä et al., 1995; Loranger et al., 2001; Scheu et
45
46 453 al., 2003; Chauvat et al., 2007; Salamon et al., 2008; Chauvat et al., 2011). While in these studies the
47
48 454 observed effects were due both to change in habitat type (succession) and to habitat age (time since the
49
50 455 habitat is in place) our results allow us to focus on the effect of habitat age within the same habitat
51
52 456 type. Temporally discontinuous (recent) habitats displayed lower diversities due to the poor
53
54 457 colonization/dispersal abilities of Collembola: some species were absent from recent habitats not
55
56 458 because the habitat was not suitable for them but because they did not have enough time to colonize it.
57
58 459 Higher values of the Shannon index in old than in recent habitats support this hypothesis: communities
59
60
61
62
63
64
65

1 460 in recent habitats are dominated by fewer species. Coined as colonization credit, this process has been
2 461 suggested to occur in plant communities (Cristofoli et al., 2010), but has so far been documented by
3
4 462 only a few empirical studies on plants (Gijbels et al., 2012) and butterflies (Cristofoli and Mahy,
5
6 463 2010). All these studies showed that specialist species are more affected than generalist species.
7
8

9 10 464 *4.2. Effect of habitat spatial isolation on collembolan diversity*

11
12 465 In this study, we tested two main possible effects of spatial isolation on diversity: the indirect
13
14 466 effect of isolation through its effect on environmental conditions and its direct effect through its
15
16 467 influence on species likelihood to colonize and persist in a patch. We detected no consistent effect,
17
18
19 468 either direct or indirect, of habitat spatial structure on collembolan diversity, among the two habitat
20
21 469 types and the two age classes here compared.

22
23 470 Habitat fragmentation is thought to decrease the local diversity of macroorganisms because
24
25 471 isolated patches (i) are too small to fulfill ecological and demographic requirements of populations
26
27 472 (Allouche et al., 2012), and (ii) are more difficult to colonize (Wamser et al., 2012). We did not find
28
29
30 473 such effect in Collembola. This is probably because our habitat patches were always large enough to
31
32 474 fulfill ecological and demographic requirements of all potentially present collembolan species due to
33
34 475 the small size of these organisms. Schneider et al. (2007) showed indeed that collembolan populations
35
36 476 in patches as small as 50 cm² did not change both in diversity and density within 16 months of
37
38
39 477 isolation. Given that our smallest patch was much larger than this restricted surface, this may explain
40
41 478 why spatial discontinuity did not affect collembolan populations in our study. Thus, even if the
42
43 479 colonization of isolated patches might be a problem for them (see section 4.3), fragmentation has
44
45 480 probably not led to local extinctions, so that colonization is not necessary to maintain diversity. Of
46
47
48 481 course, the connectivity and the size of micro-habitats inside the patches may influence collembolan
49
50 482 diversity at micro-site scale (Hertzberg, 1997; Åstrom and Bengtsson, 2011) but our sampling design
51
52 483 does not allow testing this hypothesis.
53
54
55
56
57
58
59
60
61
62
63
64
65

484 4.3. *Interacting effects of habitat temporal and spatial structure on diversity*

1
2 485 Our results showed that spatial and temporal structures of habitats had an interactive effect on
3
4 486 local diversity patterns. In the studied region, we showed that species diversity increased with spatial
5
6 487 isolation in old habitats whereas it decreased with spatial isolation in recent habitats. This resulted in
7
8
9 488 differences of diversity in isolated habitats while recent and old continuous habitats showed similar
10
11 489 values. This suggests that when in place for a long time, isolated habitats supported more colonization
12
13 490 events than spatially continuous (i.e. non-isolated) habitats, without concomitant extinction events.
14
15 491 This can be due to species from the opposite habitat type being present locally, if they do not have
16
17
18 492 strict habitat preferences, because of increased edge effects in isolated patches (Slawski and Slawska,
19
20 493 2000; Östman et al., 2009). This also means that the negative effect of spatial isolation on collembolan
21
22 494 diversity can only be unravelled if we take the habitat temporal structure into account. Our results
23
24 495 suggests that Collembola species need time to colonize recently created habitat that are isolated from
25
26 496 other habitats of the same type. This effect should be especially strong for specialist species that do not
27
28
29 497 survive in a matrix of the opposite habitat type. To our knowledge, it is the first time that such
30
31 498 interacting effects are empirically unravelled.

32
33 499 The interaction between habitat spatial and temporal continuity was not strictly similar for
34
35 500 species density and species richness. Indeed, species density was lower in recent than in old habitat of
36
37
38 501 isolation classes 1 and 2, whereas species richness was lower in recent than in old habitat of isolation 2
39
40 502 only. By definition, species density is affected by the number of individuals present in a sample, while
41
42 503 species richness is not. Species density should thus be sensitive to both local factors and landscape
43
44 504 characteristics (regional factors). Local factors determine locally the carrying capacity of the
45
46 505 environment while the number of species that can reach a local patch should be mostly determined by
47
48
49 506 landscape characteristics. This difference in sensitivity between both diversity indicators reveals that
50
51 507 old habitat of isolation class 1 should benefit from the contribution of many poorly abundant species
52
53 508 (not taken into account in the calculation of species richness) in addition to a few abundant species.
54
55 509 This results in similar values of species richness in old and recent habitats of isolation class 1, but still
56
57
58 510 different values for species density. In the other hand, old habitats of isolation class 2 benefit from an
59
60
61
62
63
64
65

511 increase in the number of species that are also more evenly distributed, which results in an increase in
1
2 512 both species richness and species density compared to recent habitat of the same isolation class.
3

4 513 These results support the novel idea that habitat fragmentation constrains diversity not only
5
6 514 through spatial but also through temporal continuity: there is a strong positive impact of temporal
7
8 515 continuity on the total number of species, in addition to the distribution of individual numbers within
9
10 516 each species. Thus, in the case of soil organisms, which probably disperse slowly at landscape scale,
11
12 517 regional processes shaping communities must be slow. That is why habitat age has to be taken into
13
14 518 account to better predict soil biodiversity in the context of habitat fragmentation.
15
16
17

18 519 *4.4. Effect of soil properties in interaction with habitat spatiotemporal structure on diversity*

19
20

21 520 Few soil characteristics responded to the spatial and /or temporal structure of habitats,
22
23 521 suggesting that indirect effect on diversity should be weak. Additionally, our results showed that both
24
25 522 species density and species richness were submitted to direct effects of habitat descriptors and their
26
27 523 interactions, whereas no indirect effects were detected since neither the Ctot nor the soil pH had
28
29 524 simple effect on diversity. However, the two parameters that were most correlated with species density
30
31 525 (soil pH and total carbon Ctot) interacted with habitat descriptors to affect diversity. Collembolan
32
33 526 diversity was more influenced by Ctot and soil pH in old than in recent patches. This result
34
35 527 corroborates the study of Dzwonko (2001), who showed that plant species indicator values for several
36
37 528 environmental factors (Ellenberg indicators) were more reliable in old than in recent forests. Species
38
39 529 can be good indicators of environmental conditions only if communities have had enough time to
40
41 530 reach equilibrium. Before equilibrium, a bias can occur because some indicator species are lacking due
42
43 531 to dispersal limitation (Seabloom and van der Valk, 2003) or because some species reflecting previous
44
45 532 habitat type still persist (French et al., 2008).
46
47

48
49 533 The introduction of Ctot in the model suppressed the significant effect of temporal continuity
50
51 534 on species density. Likewise, the introduction of the pH in the model suppressed the significant effect
52
53 535 of habitat type on species richness. This means that a part of the variability explained by habitat
54
55 536 descriptors is in fact due to a difference in soil properties. Ctot was lower (although not significantly)
56
57 537 in recent than in old habitats, especially in forest and soil pH was significantly lower in forest than in
58
59
60
61
62
63
64
65

538 agricultural land. Since the carbon content and the soil pH may influence collembolan diversity
 1
 2 539 (Chagnon et al., 2000), we cannot totally reject that a part of the observed difference in diversity
 3
 4 540 according to habitat descriptors is due to slightly different environmental conditions. This shows that
 5
 6 541 discriminating between direct and indirect effects is not straightforward, confirming that when
 7
 8 542 studying the impact of landscape structure on soil animals, soil properties should be taken into
 9
 10 543 account.

11
 12
 13 544 We also showed that effects of habitat descriptors on species density and species richness were
 14
 15 545 not strictly similar when taking into account soil properties. Fewer effects were detected on species
 16
 17 546 richness than on species density, but all factors impacting species richness also impacted species
 18
 19 547 density. This suggests that some factors impacting species density also impact species relative
 20
 21 548 abundances in a subtle way and that removing this effect on abundances through the use of species
 22
 23 549 richness suppresses the effects detected on diversity. This confirms that using an abundance-free index
 24
 25 550 of biodiversity using rarefaction curves allows a better analysis of the impact of local and regional
 26
 27 551 factors on local pools of species (Gotelli and Colwell, 2001).

31 32 552 *4.5. Differences in diversity patterns between habitat types*

33
 34 553 We defined habitat types as two very broad categories: agriculture or forest without taking the
 35
 36 554 precise habitat type (e.g. coniferous and deciduous forests) into account. Nevertheless, we were able to
 37
 38 555 reveal some significant effects of landscape characteristics on diversity and differences in patterns
 39
 40 556 between the two habitat types. This confirms that for Collembola, the land use is a fundamental factor
 41
 42 557 that determines which species can persist locally (Ponge, 1993; Hopkin, 1997; Birkhofer et al., 2012).
 43
 44 558 As hypothesized (H2), local diversity in terms of species density and species richness were higher in
 45
 46 559 forest than in agricultural habitats. This is likely to be due to differences in soil properties between
 47
 48 560 these two contrasted habitat types. (1) Forests display well developed humus profiles and a higher soil
 49
 50 561 carbon content, which provide resources for many collembolan species (Chagnon et al., 2000). The
 51
 52 562 absence of tree cover and the exportation of organic matter and associated nutrients in agricultural
 53
 54 563 land result in a reduced soil organic layer and soil moisture (Batlle-Aguilar et al., 2011) which will
 55
 56 564 consequently negatively affect diversity and abundance of soil biota. Additionally, higher soil density
 57
 58
 59
 60
 61
 62
 63
 64
 65

565 in agricultural habitats may also reduce collembolan abundance because it provides little pore space
1
2 566 for soil-dwelling populations (Vreeken-Buijs et al., 1998). It is not excluded that other biotic factors
3
4 567 such as predation contributed to drive observed diversity patterns. However, our field study did not
5
6 568 allow assessing the impact of such factor. Surprisingly, in our study, habitat type had no effect on the
7
8
9 569 Shannon index. Based on the mathematical expression of Shannon's diversity, which associates a
10
11 570 probability of occurrence to each species, this discrepancy suggests that even though the number of
12
13 571 species differed, less frequent species were most affected by change in habitat type (Gorelick, 2006).
14

15 572 The effect of temporal discontinuity was stronger in forest than in agricultural land. This
16
17 573 finding supports our third hypothesis (H3) that predicted a stronger effect of habitat spatiotemporal
18
19 574 structure on collembolan diversity in forest than in agricultural habitats. This could arise because open
20
21 575 habitats lead to more efficient dispersal either through passive or active mechanisms. First, if
22
23 576 *Collembola* have a preponderant passive dispersal and are directly or indirectly transported by wind,
24
25 577 as suggested for some species by Dunger et al. (2002) and Coulson and Midgley (2012), their
26
27 578 colonization rate might be higher in agricultural habitats since air flow is stronger in the absence of
28
29 579 tree cover (Wright et al., 2010). This result is in accordance with other findings on *Collembola* in
30
31 580 deglaciaded land (Ingimarsdottir et al., 2012). Indeed these authors showed that in such extremely open
32
33 581 habitat (i.e. with reduced to nil vegetation cover) collembolan dispersal rate was potentially high.
34
35 582 Another explanation could be that collembolan communities in agricultural land are comprised of
36
37 583 more epigeic species than in forests (Sousa et al., 2004), with morphological traits (e.g. long legs, long
38
39 584 furcula) supposed to allow higher mobility in open habitats (Salmon and Ponge, 2012). This result
40
41 585 suggests that dispersal rates and modes are habitat-dependent as shown in other microarthropod
42
43 586 groups (Lindo and Winchester, 2009).
44
45
46
47
48

49 587 Our results also showed a significant interaction between spatial isolation and habitat type.
50
51 588 With spatial isolation, species density increased in forest but decreased in agricultural habitats. This
52
53 589 could be due to better mobility and less strict habitat preferences of species found in agricultural
54
55 590 communities (Lauga-Reyrel and Deconchat, 1999; Ponge et al., 2006), likely to be present in remnant
56
57 591 forest patches. In contrast, many species from the forest communities show strict habitat preferences
58
59 592 and a reduced mobility (Auclerc et al., 2009) and are thus less prone to colonize remnant agricultural
60
61
62
63
64
65

593 patches. Our results also suggest that the forest matrix surrounding agricultural patches is more
 1
 2 594 difficult to cross for species than the agricultural matrix surrounding forests. Since passive dispersal
 3
 4 595 by wind is likely to be more efficient in agricultural lands than in forests, this suggests that largely
 5
 6 596 overlooked passive dispersal mechanisms might be influential for collembolan communities and meta-
 7
 8 597 community dynamics.

12 598 **5. Conclusion**

14 599 Our study contributes to some extent to disentangle complex effects of habitat characteristics
 15
 16 600 (i.e. spatiotemporal structure and local physicochemical conditions) on collembolan diversity. Most
 17
 18 601 importantly, we showed that habitat temporal continuity is a key factor shaping collembolan diversity
 19
 20 602 and that we must take it into account if we want to understand the effects of habitat spatial structure.
 21
 22 603 We also showed that using both species richness and species density as indicators of biodiversity
 23
 24 604 allows refining the analysis of the factors influencing collembolan diversity at different scales.
 25
 26 605 Unfortunately, it is rather impossible to experiment at landscape scale, and the limits of our study must
 27
 28 606 be clearly defined. We controlled only partly the factors thought to influence collembolan diversity:
 29
 30 607 climate and litter quality could not be controlled, as they can be in experimental conditions (Åström
 31
 32 608 and Bengtsson, 2011; Bokhorst et al., 2012). Similarly, the different levels of isolation and age were
 33
 34 609 broadly defined and not totally equivalent in forest and agricultural habitats. However, we were able to
 35
 36 610 detect several effects of our treatments (habitat type, isolation and patch age), using landscape past
 37
 38 611 dynamics as a surrogate for a strict experimental approach. Some theoretical models predicted such
 39
 40 612 patterns (Green and Sadedin, 2005; Gardner and Engelhardt, 2008), but empirical corroborating
 41
 42 613 studies are still scarce (Lemaître et al., 2012). Nevertheless, we showed that habitat fragmentation
 43
 44 614 constrains diversity not only in space but also in time. The temporal continuity of habitats is one of the
 45
 46 615 factors that have a positive effect on diversity. Soil organisms disperse slowly compared to above-
 47
 48 616 ground biota and it hence takes longer for a recent habitat to build up soil diversity especially if the
 49
 50 617 patch is very isolated.

57 618 Assembly theory is now well recognized in community ecology but still little applied in soil
 58
 59 619 ecology. This metacommunity framework predicts four models that could explain species distribution

620 at local scale (Leibold et al, 2004). These models essentially differ in the importance given to dispersal
1
2 621 process in explaining metacommunity dynamics. Two of them, ‘species sorting’ (environmental
3
4 622 constraints control species distribution) and ‘mass effect’ (dispersal redistributes species among
5
6 623 habitats independently of environmental constraints) were found relevant to collembolan communities
7
8 624 by da Silva et al. (2012) and Ingimarsdottir et al. (2012). This study also supports the idea that
9
10
11 625 dispersal plays an important role in structuring collembolan metacommunities and that the magnitude
12
13 626 and direction of its effect depend on the type of habitat considered.
14
15

16 627 *Acknowledgments*
17

18
19 628 This study was sponsored by the R2DS program of the Conseil Régional d’Ile-de-France. We would
20
21 629 like to thank the public and private owners of the sampling sites for allowing the study to be carried
22
23 630 out on their land. We would like to tank the IRD (Centre Fance Nord) and the MNHN (Brunoy) for the
24
25 631 access to their facilities and material; the Laboratoire des Moyens Analytiques (IRD, Dakar) for
26
27 632 performing the soil analysis; the IGN (France) for providing aerial photographs; Joshua Lobe for
28
29 633 correction of the English language; Thibaud Decaëns for useful advices about the use of species
30
31
32 634 rarefaction curves; anonymous reviewers for their useful comments which have helped us to increase
33
34 635 the quality of the manuscript.
35
36
37
38
39
40
41
42
43
44
45
46
47
48
49
50
51
52
53
54
55
56
57
58
59
60
61
62
63
64
65

636 **References**

- 1
2 637
3
4
5 638 Allouche, O., Kalyuzhny, M., Moreno-Rueda, G., Pizarro, M., Kadmon, R., 2012. Area-heterogeneity
6
7 639 tradeoff and the diversity of ecological communities. *Proceedings of the National Academy of*
8
9 640 *Sciences of the United States of America* 109, 17495-17500.
- 11
12
13 641 André, H.M., Noti, M.I., Lebrun, P., 1994. The soil fauna: the other last biotic frontier. *Biodiversity*
14
15 642 *and Conservation* 3, 45-56.
- 16
17
18 643 Åström, J., Bengtsson, J., 2011. Patch size matters more than dispersal distance in a mainland-island
19
20 644 metacommunity. *Oecologia* 167, 747-757.
- 22
23
24 645 Auclerc, A., Ponge, J.F., Barot, S., Dubs, F., 2009. Experimental assessment of habitat preference and
25
26 646 dispersal ability of soil springtails. *Soil Biology and Biochemistry* 41, 1596-1604.
- 27
28
29 647 Augusto, L., Dupouey, J.L., Ranger, J., 2003. Effects of tree species on understory vegetation and
30
31 648 environmental conditions in temperate forests. *Annals of Forest Science* 60, 823-831.
- 32
33
34
35 649 Bailey, S., 2007. Increasing connectivity in fragmented landscapes: an investigation of evidence for
36
37 650 biodiversity gain in woodlands. *Forest Ecology and Management* 238, 7-23.
- 38
39
40
41 651 Batlle-Aguilar, J., Brovelli, A., Porporato, A., Barry, D.A., 2011. Modelling soil carbon and nitrogen
42
43 652 cycles during land use change. A review. *Agronomy for Sustainable Development* 31, 251-
44
45 653 274.
- 46
47
48 654 Birkhofer, K., Schöning, I., Alt, F., Herold, N., Klarner, B., Maraun, M., Marhan, S., Oelmann, Y.,
49
50 655 Wubet, T., Yurkov, A., Begerow, D., Berner, D., Buscot, F., Daniel, R., Diekötter, T., Ehnes,
51
52 656 R.B., Erdmann, G., Fischer, C., Foesel, B., Groh, J., Gutknecht, J., Kandeler, E., Lang, C.,
53
54 657 Lohaus, G., Meyer, A., Nacke, H., Näther, A., Overmann, J., Polle, A., Pollierer, M.M.,
55
56 658 Scheu, S., Schloter, M., Schulze, E.D., Schulze, W., Weinert, J., Weisser, W.W., Wolters, V.,
57
58
59
60
61
62
63
64
65

- 659 Schrumpf, M., 2012. General relationships between abiotic soil properties and soil biota
1
2 660 across spatial scales and different land-use types. Plos One 7, e43292.
3
4
- 5 661 Bodzogan, H., 1987. Model selection and Akaike's Information Criterion (AIC)! The general theory
6
7 662 and its analytical extensions. Psychometrika, 52, 345-370.
8
9
- 10 663 Bokhorst, S., Phoenix, G.K., Bjerke, J.W., Callaghan, T.V., Huyer-Brugman, F., Berg, M.P., 2012.
11
12 664 Extreme winter warming events more negatively impact small rather than large soil fauna:
13
14 665 shift in community composition explained by traits not taxa. Global Change Biology 18,
15
16 666 1152-1162.
17
18
19
- 20 667 Bretfeld, G., 1999. Synopses on Palaearctic Collembola, II Symphypleona. Abhandlungen und
21
22 668 Berichte des Naturkundemuseums, Görlitz.
23
24
25
- 26 669 Brêthes, A., Brun, J.J., Jabiol, B., Ponge, J.F., Toutain, F., 1995. Classification of forest humus forms:
27
28 670 a french proposal. Annales des Sciences Forestières 52, 535-546.
29
30
31
- 32 671 Brückmann, S.V., Krauss, J., Steffan-Dewenter, I., 2010. Butterfly and plant specialists suffer from
33
34 672 reduced connectivity in fragmented landscapes. Journal of Applied Ecology 47, 799-809.
35
36
- 37 673 Caner, L., Zeller, B., Dambrine, E., Ponge, J.F., Chauvat, M., Llanque, C., 2004. Origin of the
38
39 674 nitrogen assimilated by soil fauna living in decomposing beech litter. Soil Biology and
40
41 675 Biochemistry 36, 1861-1872.
42
43
44
- 45 676 Cassagne, N., Bal-Serin, M.C., Gers, C., Gauquelin, T., 2004. Changes in humus properties and
46
47 677 collembolan communities following the replanting of beech forests with spruce. Pedobiologia
48
49 678 48, 267-276.
50
51
52
- 53 679 Chagnon, M., Paré, D., Hébert, C., 2000. Relationships between soil chemistry, microbial biomass and
54
55 680 the collembolan fauna of southern Quebec sugar maple stands. Écoscience 7, 307-316.
56
57
58
59
60
61
62
63
64
65

- 681 Chauvat, M., Titsch, D., Zaytsev, A.S., Wolters, V., 2011. Changes in soil faunal assemblages during
1
2 682 conversion from pure to mixed forest stands. *Forest Ecology and Management* 262, 317-324.
3
4
- 5 683 Chauvat, M., Wolters, V., Dauber, J., 2007. Response of collembolan communities to land-use change
6
7 684 and grassland succession. *Ecography* 30, 183-192.
8
9
- 10
11 685 Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. *Science* 199, 1302-1310.
12
13
- 14 686 Coulson, S.J., Midgley, N.G., 2012. The role of glacier mice in the invertebrate colonisation of glacial
15
16 687 surfaces: the moss balls of the Falljokull, Iceland. *Polar Biology* 35, 1651-1658.
17
18
- 19 688 Cristofoli, S., Mahy, G., 2010. Colonisation credit in recent wet heathland butterfly communities.
20
21 689 *Insect Conservation and Diversity* 3, 83-91.
22
23
- 24
25 690 Cristofoli, S., Piqueray, J., Dufrêne, M., Bizoux, J.P., Mahy, G., 2010. Colonization Credit in Restored
26
27 691 Wet Heathlands. *Restoration Ecology* 18, 645-655.
28
29
- 30
31 692 Cushman, S.A., 2006. Effects of habitat loss and fragmentation on amphibians: a review and
32
33 693 prospectus. *Biological Conservation* 128, 231-240.
34
35
- 36 694 da Silva, P.M., Berg, M.P., Serrano, A.R.M., Dubs, F., Sousa, J.P., 2012. Environmental factors at
37
38 695 different spatial scales governing soil fauna community patterns in fragmented forests.
39
40 696 *Landscape Ecology* 27, 1337-1349.
41
42
43
- 44 697 Davies, K.F., Melbourne, B.A., Margules, C.R., Lawrence, J.F., 2005. Metacommunity structure
45
46 698 influences the stability of local beetle communities. In: *Metacommunities: Spatial Dynamics*
47
48 699 *and Ecological Communities*, eds. Holyoak, M., Leibold, M.A., Holt, R.D. University of
49
50 700 Chicago Press, Chicago, pp. 170-188.
51
52
53
- 54 701 Decaëns, T., 2010. Macroecological patterns in soil communities. *Global Ecology and Biogeography*
55
56 702 19, 287-302.
57
58
59
60
61
62
63
64
65

- 703 Decaëns, T., Margerie, P., Renault, J., Bureau, F., Aubert, M., Hedde, M., 2011. Niche overlap and
1 species assemblage dynamics in an ageing pasture gradient in north-western France. *Acta*
2
3
4 705 *Oecologica* 37, 212-219.
5
6
- 7 706 Driscoll, D.A., Whitehead, C.A., Lazzari, J., 2012. Spatial dynamics of the knob-tailed gecko
8
9 707 *Nephrurus stellatus* in a fragmented agricultural landscape. *Landscape Ecology* 27, 829-841.
10
11
12
- 13 708 Dzwonko, Z., 2001. Assessment of light and soil conditions in ancient and recent woodlands by
14
15 709 Ellenberg indicator values. *Journal of Applied Ecology* 38, 942-951.
16
17
- 18 710 Dunger, W., Schulz, H.J., Zimdars, B., 2002. Colonization behaviour of Collembola under different
19
20 711 conditions of dispersal. *Pedobiologia* 46, 316-327.
21
22
23
- 24 712 Ettema, C.H., Wardle, D.A., 2002. Spatial soil ecology. *Trends in Ecology and Evolution* 17, 177-183.
25
26
- 27 713 Finlay, B.J., Esteban, G.F., Fenchel, T., 1996. Global diversity and body size. *Nature* 383, 132-133.
28
29
- 30 714 Forman, R., 1995. *Land Mosaics: The Ecology of Landscapes and Regions*. Cambridge University
31
32 715 Press, Cambridge.
33
34
35
- 36 716 Forman, R., Godron, M., 1986. *Landscape Ecology*. Wiley, New York.
37
38
- 39 717 Fox, J.W., 2013. The intermediate disturbance hypothesis should be abandoned. *Trends in Ecology*
40
41 718 and *Evolution* 28, 86-92
42
43
44
- 45 719 French, L.J., Smith, G.F., Kelly, D.L., Mitchell, F.J.G., O'Donoghue, S., Iremonger, S.E., McKee, A.-
46
47 720 M., 2008. Ground flora communities in temperate oceanic plantation forests and the influence
48
49 721 of silvicultural, geographic and edaphic factors. *Forest Ecology and Management* 255, 476-
50
51 722 494.
52
53
54
- 55 723 Gardner, R.H., Engelhardt, K.A.M., 2008. Spatial processes that maintain biodiversity in plant
56
57 724 communities. *Perspectives in Plant Ecology Evolution and Systematics* 9, 211-228.
58
59
60
61
62
63
64
65

- 725 Gauquelin, T., Gers, C., Deharveng, L., 1996. Physico-chemical heterogeneity of superficial soil
 1 layers in conifer plantations versus original broad leaf forests in Ariege (Pyrenees, France).
 2 726
 3
 4 727 Communications in Soil Science and Plant Analysis 27, 2361-2380.
 5
 6
 7 728 Gijbels, P., Adriaens, D., Honnay, O., 2012. An orchid colonization credit in restored calcareous
 8
 9 729 grasslands. *Écoscience* 19, 21-28.
 10
 11
 12
 13 730 Gilpin, M.E., Hanski, I.A., 1991. Metapopulation dynamics: Empirical and Theoretical Investigations.
 14
 15 731 Academic Press, London.
 16
 17
 18 732 Gorelick, R., 2006. Combining richness and abundance into a single diversity index using matrix
 19
 20 733 analogues of Shannon's and Simpson's indices. *Ecography* 29, 525-530.
 21
 22
 23
 24 734 Gotelli, N.J., Colwell, R.K., 2001. Quantifying biodiversity: procedures and pitfalls in the
 25
 26 735 measurement and comparison of species richness. *Ecology Letters* 4, 379-391
 27
 28
 29 736 Green, D.G., Sadedin, S., 2005. Interactions matter: complexity in landscapes and ecosystems.
 30
 31 737 *Ecological Complexity* 2, 117-130.
 32
 33
 34
 35 738 Hågvar, S., 1983. Collembola in Norwegian coniferous forest soils. II. Vertical distribution.
 36
 37 739 *Pedobiologia* 25, 383-401.
 38
 39
 40 740 Hanski, I., 1994. Patch-occupancy dynamics in fragmented landscapes. *Trends in Ecology and*
 41
 42 741 *Evolution* 9, 131-135.
 43
 44
 45 742 Hasegawa, M., Fukuyama, K., Makino, S., Okochi, I., Tanaka, H., Okabe, K., Goto, H., Mizoguchi,
 46
 47 743 T., Sakata, T., 2009. Collembolan community in broad-leaved forests and in conifer stands of
 48
 49 744 *Cryptomeria japonica* in Central Japan. *Pesquisa Agropecuária Brasileira* 44, 881-890.
 50
 51
 52
 53 745 Hertzberg, K., 1997. Migration of Collembola in a patchy environment. *Pedobiologia* 41, 494-505.
 54
 55
 56
 57 746 Hewitt, N., Kellman, M., 2002. Tree seed dispersal among forest fragments. II. Dispersal abilities and
 58
 59 747 biogeographical controls. *Journal of Biogeography* 29, 351-363.
 60
 61
 62
 63
 64
 65

- 748 Hillebrand, H, Blenckner, T., 2002. Regional and local impact on species diversity - from pattern to
1
2 749 processes. *Oecologia* 132, 479-491.
3
4
- 5 750 Holyoak, M., Leibold, M.A., Holt, R.D., 2005. *Metacommunities: Spatial Dynamics and Ecological*
6
7 751 *Communities*. University of Chicago Press Chicago.
8
9
- 10 752 Hopkin, S.P., 1997. *Biology of the Springtails (Insecta: Collembola)*. Oxford University Press,
11
12 753 Oxford.
13
14
15
- 16 754 Hopkin, S.P., 2007. *A Key to Collembola (Springtails) of Britain and Ireland*. Field Studies Council,
17
18 755 Shrewsbury.
19
20
21
- 22 756 Hubbell, S.P., 2001. *The Unified Neutral Theory of Biodiversity and Biogeography*. Princeton
23
24 757 University Press, Princeton.
25
26
- 27 758 Ingimarsdottir, M., Caruso, T., Ripa, J., Magnúsdóttir, O.B., Migliorini, M., Hedlund, K., 2012.
28
29 759 Primary assembly of soil communities: disentangling the effect of dispersal and local
30
31 760 environment. *Oecologia* 170, 745-754.
32
33
34
- 35 761 Kotliar, N.B., Wiens, J.A., 1990. Multiple scales of patchiness and patch structure: a hierarchical
36
37 762 framework for the study of heterogeneity. *Oikos* 59, 253-260.
38
39
- 40 763 Krauss, J., Bommarco, R., Gardioli, M., Heikkinen, R.K., Helm, A., Kuussaari, M., Lindborg, R.,
41
42 764 Öckinger, E., Partel, M., Pino, J., Poyry, J., Raatikainen, K.M., Sang, A., Stefanescu, C.,
43
44 765 Teder, T., Zobel, M., Steffan-Dewenter, I., 2010. Habitat fragmentation causes immediate and
45
46 766 time-delayed biodiversity loss at different trophic levels. *Ecology Letters* 13, 597-605.
47
48
49
- 50 767 Lauga-Reyrel, F., Deconchat, M., 1999. Diversity within the Collembola community in fragmented
51
52 768 coppice forests in south-western France. *European Journal of Soil Biology* 35, 177-187.
53
54
55
56
57
58
59
60
61
62
63
64
65

- 769 Lavelle, P., Decaëns, T., Aubert, M., Barot, S., Blouin, M., Bureau, F., Margerie, P., Mora, P., Rossi,
 1 J.P., 2006. Soil invertebrates and ecosystem services. *European Journal of Soil Biology* 42,
 2 770
 3 S3-S15.
 4 771
 5
 6
 7 772 Leibold, M.A., Holyoak, M., Mouquet, N., Amarasekare, P., Chase, J.M., Hoopes, M.F., Holt, R.D.,
 8
 9 Shurin, J.B., Law, R., Tilman, D., Loreau, M., Gonzalez, A., 2004. The metacommunity
 10 773
 11 concept: a framework for multi-scale community ecology. *Ecology Letters* 7, 601-613.
 12 774
 13
 14
 15 775 Lemaître, J., Darveau, M., Zhao, Q., Fortin, D., 2012. Multiscale assessment of the influence of habitat
 16
 17 structure and composition on bird assemblages in boreal forest. *Biodiversity and Conservation*
 18 776
 19 21, 3355-3368.
 20 777
 21
 22
 23 778 Lindo, Z., Winchester, N.N., 2009. Spatial and environmental factors contributing to patterns in
 24
 25 arboreal and terrestrial oribatid mite diversity across spatial scales. *Oecologia* 160, 817-825.
 26 779
 27
 28 780 Loranger, G., Bandyopadhyaya, I., Razaka, B., Ponge, J.F., 2001. Does soil acidity explain altitudinal
 29
 30 sequences in collembolan communities? *Soil Biology and Biochemistry* 33, 381-393.
 31 781
 32
 33
 34 782 Magura, T., Tothmeresz, B., Elek, Z., 2003. Diversity and composition of carabids during a forestry
 35
 36 cycle. *Biodiversity and Conservation* 12, 73-85.
 37 783
 38
 39 784 Mapelli, F.J., Kittlein, M.J., 2009. Influence of patch and landscape characteristics on the distribution
 40
 41 of the subterranean rodent *Ctenomys porteousi*. *Landscape Ecology* 24, 723-733.
 42 785
 43
 44
 45 786 Molino, J.F., Sabatier, D., 2001. Tree diversity in tropical rain forests: a validation of the intermediate
 46
 47 disturbance hypothesis. *Science* 294, 1702-1704.
 48 787
 49
 50 788 Morecroft, M.D., Taylor, M.E., Oliver, H.R., 1998. Air and soil microclimates of deciduous woodland
 51
 52 compared to an open site. *Agricultural and Forest Meteorology* 90, 141-156.
 53 789
 54
 55
 56 790 Mouquet, N., Munguia, P., Kneitel, J.M., Miller, T.E., 2003. Community assembly time and the
 57
 58 relationship between local and regional species richness. *Oikos* 103, 618-626.
 59 791
 60
 61
 62
 63
 64
 65

- 792 Östman, Ö., Mellbrand, K., Hambäck, P.A., 2009. Edge or dispersal effects - Their relative importance
1 on arthropod densities on small islands. *Basic and Applied Ecology* 10, 475-484.
2
3
4
- 5 794 Petit, S., Fried, G., 2012. Patterns of weed co-occurrence at the field and landscape level. *Journal of*
6
7 795 *Vegetation Science* 23, 1137-1147.
8
9
- 10 796 Ponge, J.F., 1991. Food resources and diets of soil animals in a small area of Scots pine litter.
11
12 797 *Geoderma* 49, 33-62.
13
14
15
- 16 798 Ponge, J.F., 1993. Biocenoses of Collembola in atlantic temperate grass-woodland ecosystems.
17
18 799 *Pedobiologia* 37, 223-244.
19
20
21
- 22 800 Ponge, J.F., 2000. Vertical distribution of Collembola (Hexapoda) and their food resources in organic
23
24 801 horizons of beech forests. *Biology and Fertility of Soils* 32, 508-522.
25
26
- 27 802 Ponge, J.F., Arpin, P., Sondag, F., Delecour, F., 1997. Soil fauna and site assessment in beech stands
28
29 803 of the Belgian Ardennes. *Canadian Journal of Forest Research* 27, 2053-2064.
30
31
32
- 33 804 Ponge, J.F., Chevalier, R., 2006. Humus Index as an indicator of forest stand and soil properties.
34
35 805 *Forest Ecology and Management* 233, 165-175.
36
37
- 38 806 Ponge, J.F., Chevalier, R., Loussot, P., 2002. Humus index: an integrated tool for the assessment of
39
40 807 forest floor and topsoil properties. *Soil Science Society of America Journal* 66, 1996-2001.
41
42
43
- 44 808 Ponge, J.F., Dubs, F., Gillet, S., Sousa, J.P., Lavelle, P., 2006. Decreased biodiversity in soil springtail
45
46 809 communities: the importance of dispersal and landuse history in heterogeneous landscapes.
47
48 810 *Soil Biology and Biochemistry* 38, 1158-1161.
49
50
- 51 811 Ponge, J.F., Gillet, S., Dubs, F., Fedoroff, E., Haese, L., Sousa, J.P., Lavelle, P., 2003. Collembolan
52
53 812 communities as bioindicators of land use intensification. *Soil Biology and Biochemistry* 35,
54
55 813 813-826.
56
57
58
59
60
61
62
63
64
65

- 814 Ponge, J.F., Salmon, S., 2013. Spatial and taxonomics correlates of species and species trait
 1
 2 815 assemblages in soil invertebrate communities. *Pedobiologia* 56, 129-136.
 3
 4
 5 816 Posada, D., Buckley, T.R., 2004. Model selection and model averaging in phylogenetics: advantages
 6
 7 817 of Akaike Information Criterion and Bayesian approaches over likelihood ratio tests.
 8
 9 818 *Systematic Biology* 53, 793-808.
 10
 11
 12
 13 819 Potapov, M., 2001. Synopses of Palearctic Collembola. III. Isotomidae. *Abhandlungen und Berichte*
 14
 15 820 *des Naturkundemuseums, Görlitz*.
 16
 17
 18 821 Querner, P., Bruckner, A., Drapela, T., Moser, D., Zaller, J.G., Frank, T., 2013. Landscape and site
 19
 20 822 effects on Collembola diversity and abundance in winter oilseed rape fields in eastern Austria.
 21
 22 823 *Agriculture, Ecosystems and Environment* 164, 145-154.
 23
 24
 25
 26 824 R Development Core Team, 2010. *R: a Language and Environment for Statistical Computing*. R
 27
 28 825 *Foundation for Statistical Computing, Vienna*.
 29
 30
 31
 32 826 Ricklefs, R.E., 1987. Community diversity: relative roles of local and regional processes. *Science* 235,
 33
 34 827 167-171.
 35
 36
 37 828 Salamon, J.A. and Alpei, J., 2009. The Collembola community of a Central European forest:
 38
 39 829 influence of tree species composition. *European Journal of Soil Biology* 45, 199-206
 40
 41
 42
 43 830 Salamon, J.A., Scheu, S., Schaefer, M., 2008. The Collembola community of pure and mixed stands of
 44
 45 831 beech (*Fagus sylvatica*) and spruce (*Picea abies*) of different age. *Pedobiologia* 51, 385-396.
 46
 47
 48 832 Salmon, S., 2004. The impact of earthworms on the abundance of Collembola: improvement of food
 49
 50 833 resources or of habitat? *Biology and Fertility of Soils* 40, 323-333.
 51
 52
 53
 54 834 Salmon, S. and Ponge, J.F., 2012. Species traits and habitats in springtail communities: a regional
 55
 56 835 scale study. *Pedobiologia* 55, 295-301.
 57
 58
 59
 60
 61
 62
 63
 64
 65

- 836 Saunders, D.A., Hobbs, R.J., Margules, C.R., 1991. Biological consequences of ecosystem
1
2 837 fragmentation: a review. *Conservation Biology* 5, 18-32.
3
4
- 5 838 Scheu, S., Albers, D., Alpehi, J., Buryr, R., Klages, U., Migge, S., Platner, C., Salamon, J.A., 2003.
6
7 839 The soil fauna community in pure and mixed stands of beech and spruce of different age:
8
9 840 trophic structure and structuring forces. *Oikos* 101, 225-238.
10
11
- 12
13 841 Schneider, K., Scheu, S., Maraun, M., 2007. Microarthropod density and diversity repond little to
14
15 842 spatial isolation. *Basic and Applied Ecology* 8, 26-35.
16
17
- 18 843 Seabloom, E.W., van der Valk, A.G., 2003. Plant diversity, composition, and invasion of restored and
19
20 844 natural prairie pothole wetlands: implications for restoration. *Wetlands* 23, 1-12.
21
22
23
- 24 845 Setälä, H., Marshall, V.G., Trofymow, J.A., 1995. Influence of micro-habitat and macro-habitat
25
26 846 factors on collembolan communities in Douglas-fir Stumps during forest succession. *Applied*
27
28 847 *Soil Ecology* 2, 227-242.
29
30
- 31
32 848 Shmida, A., Wilson, M.V., 1985. Biological determinants of species diversity. *Journal of*
33
34 849 *Biogeography* 12, 1-20.
35
36
- 37 850 Sławski, M., Sławaska, M., 2000. The forest edge as a border between forest and meadow. *Vegetation*
38
39 851 and *Collembola* communities. *Pedobiologia* 44, 442-450.
40
41
42
- 43 852 Sousa, J.P., Bolger, T., da Gama, M.M., Lukkari, T., Ponge, J.F., Simon, C., Traser, G., Vanbergen,
44
45 853 A.J., Brennan, A., Dubs, F., Ivitis, E., Keating, A., Stofer, S., Watt, A.D., 2006. Changes in
46
47 854 *Collembola* richness and diversity along a gradient of land-use intensity: a pan European
48
49 855 study. *Pedobiologia* 50, 147-156.
50
51
52
- 53 856 Sousa, J.P., da Gama, M.M., Pinto, P., Keating, A., Calhoa, F., Lemos, M., Castro, C., Luz, T., Leitao,
54
55 857 P., Dias, S., 2004. Effects of land-use on *Collembola* diversity patterns in a Mediterranean
56
57 858 landscape. *Pedobiologia* 48, 609-622.
58
59
60
61
62
63
64
65

- 1
2 860 Stratford, J.A., Stouffer, P.C., 1999. Local extinctions of terrestrial insectivorous birds in a fragmented
3 landscape near Manaus, Brazil. *Conservation Biology* 13, 1416-1423.
4
- 5 861 Tanner, J.E., 2003. Patch shape and orientation influences on seagrass epifauna are mediated by
6 dispersal abilities. *Oikos* 100, 517-524.
7
8 862
9
- 10 863 Tews, J., Brose, U., Grimm, V., Tielborger, K., Wichmann, M.C., Schwager, M., Jeltsch, F., 2004.
11 Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone
12 structures. *Journal of Biogeography* 31, 79-92.
13 864
14
15 865
16
17
- 18 866 Thibault, J.M., Schultz, H.J., da Gama, M.M., 2004. Synopses on Palearctic Collembola. IV.
19 Hypogastruridae. *Abhandlungen und Berichte des Naturkundemuseums, Görlitz*.
20 867
21
22
23
- 24 868 Tilman, D., 1982. *Resource Competition and Community Structure*. Princeton University Press,
25 Princeton.
26 869
27
28
- 29 870 Tilman, D., 1994. Competition and Biodiversity in spatially structured habitats. *Ecology* 75, 2-16.
30
31
- 32 871 Tilman, D., May, R.M., Lehman, C.L., Nowak, M.A., 1994. Habitat destruction and the extinction
33 debt. *Nature* 371, 65-66.
34 872
35
36
37
- 38 873 Vanbergen, A.J., Watt, A.D., Mitchell, R., Truscott, A.-M., Palmer, S.C.F., Ivits, E., Eggleton, P.,
39 Jones, T.H., Sousa, J.P., 2007. Scale-specific correlations between habitat heterogeneity and
40 soil fauna diversity along a landscape structure gradient. *Oecologia* 153, 713-725.
41 874
42
43 875
44
45
- 46 876 Vreeken-Buijs, M.J., Hassink, J., Brussaard, L., 1998. Relationships of soil microarthropod biomass
47 with organic matter and pore size distribution in soils under different land use. *Soil Biology*
48 and *Biochemistry* 30, 97-106.
49 877
50
51 878
52
53
- 54 879 Wamser, S., Diekötter, T., Boldt, L., Wolters, V., Dauber, J., 2012. Trait-specific effects of habitat
55 isolation on carabid species richness and community composition in managed grasslands.
56 880
57
58 881
59
60
61
62
63
64
65

- 882 Wardle, D.A., 2006. The influence of biotic interactions on soil biodiversity. *Ecology Letters* 9, 870-
1
2 883 886.
3
4
5 884 Wright, T.E., Kasel, S., Tausz, M., Bennett, L.T., 2010. Edge microclimate of temperate woodlands as
6
7 885 affected by adjoining land use. *Agricultural and Forest Meteorology* 150, 1138-1146.
8
9
10
11 886 Wu, J.G., Loucks, O.L., 1995. From balance of nature to hierarchical patch dynamics: a paradigm shift
12
13 887 in ecology. *Quarterly Review of Biology* 70, 439-466.
14
15 888
16
17
18
19
20
21
22
23
24
25
26
27
28
29
30
31
32
33
34
35
36
37
38
39
40
41
42
43
44
45
46
47
48
49
50
51
52
53
54
55
56
57
58
59
60
61
62
63
64
65

1 FIGURE CAPTIONS

2 **Figure 1:** Statistical diagram picturing steps of data analyses. Arrows represent effects of factors on
3 variables (collembolan diversity and soil properties). Direct effects are represented by continuous
4 arrows and indirect effects by dotted arrows. Model type 1 (arrow 1) stands for the linear models
5 testing the effect of the three habitat descriptors on collembolan diversity. Model type 2 (arrow 2)
6 stands for linear and generalized linear models testing the effect of the habitat descriptors on soil
7 physicochemical properties. Model type 3 (arrows 1 and 3) stands for linear models testing the effects
8 of the three habitat descriptors and a soil property on diversity indicators.

9 **Figure 2:** Individual based rarefaction curves old forests (F_old), recent forests (F_rec), old
10 agricultural land (A_old) and recent agricultural land (A_rec).

11 **Figure 3:** Mean species density (3a), Shannon index (3b) and species richness (3c) of old (grey) and
12 recent (white) agricultural and forest habitats. Letters indicate significant differences among means.
13 “A” and “B” labels represent significant differences between agricultural and forest habitats. Stars
14 show simple effect of age ($p < 0.05^*$; $p < 0.01^{**}$; $p < 0.001^{***}$). Error bars represent standard errors.

15 **Figure 4:** Mean species density (4a) and species richness (4b) according to Habitat Type, Temporal
16 Continuity (two age classes: old and recent) and Spatial Isolation (three classes: 0 black bars, 1 white
17 bars and 2 grey bars). Error bars represent standard errors.

18 **Figure 5:** Correlation coefficients (Pearson) between species density (grey bars), species richness
19 (dashed bars) or Shannon index (white bars) and soil properties for the complete data set (agricultural
20 land and forest) (5a) and for forest only (5b). Soil pH: pH, total nitrogen content: Ntot, total carbon
21 content: Ctot, carbon to nitrogen ration: CN, bioavailable phosphorus: Ph, cation exchange capacity:
22 CEC, soil moisture: moist, soil density: dens and Humus Index: Humus. Stars show significant
23 correlation ($p < 0.05^*$; $p < 0.01^{**}$; $p < 0.001^{***}$) between species density, species richness or Shannon
24 index and each soil parameters listed above.

25 **Figure 6:** Least square means for species density and species richness obtained using the model
26 testing for the effect of the three habitat descriptors as well as total carbon on species density and
27 species richness. Least square means for species density are represented according to Habitat Type
28 (agricultural, grey bars and forested, white bars) and Spatial Isolation (three isolation classes 0, 1 and

29 2) (6a), and least square means for species density and species richness are represented according to
30 Temporal Continuity (two age classes: old, dashed bars and recent, white bars) and Spatial Isolation
31 (three isolation classes 0, 1 and 2) (respectively 6b and 6c). Error bars represent standard errors. Stars
32 show significant differences between habitat types (6a) or ages (6b and 6c) within each isolation class.

Figure1
[Click here to download high resolution image](#)

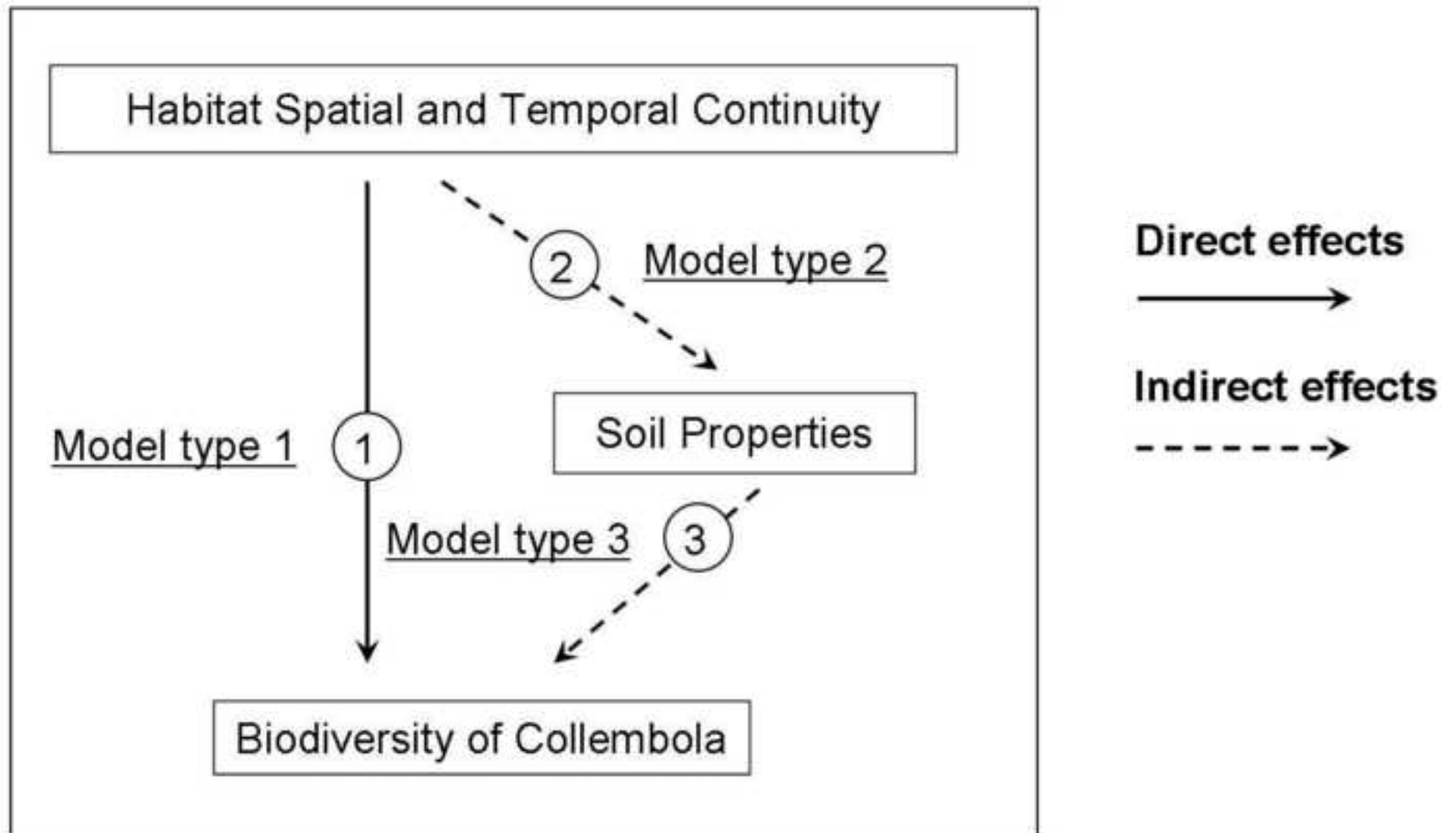


Figure2

[Click here to download high resolution image](#)

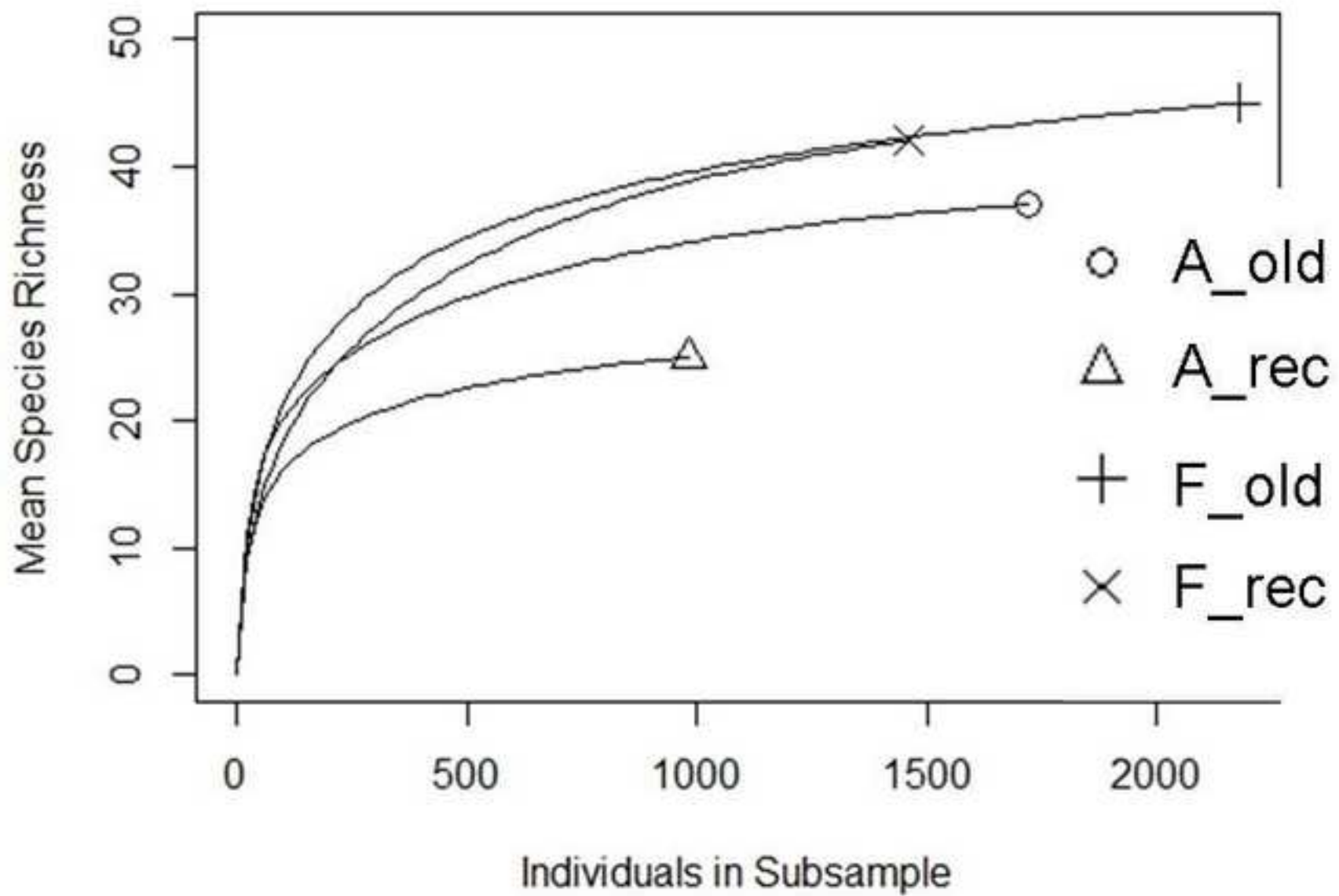
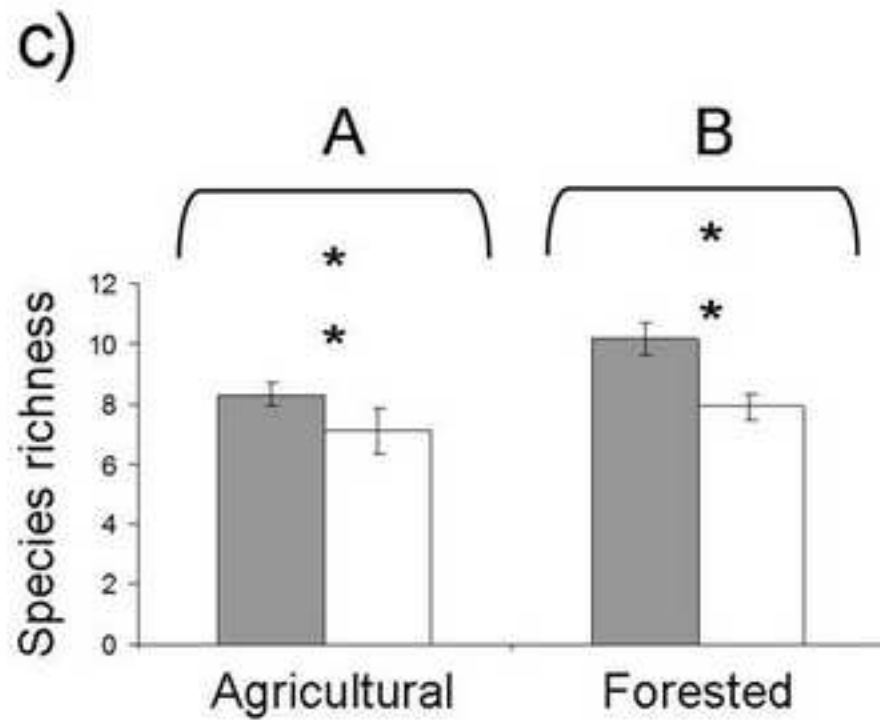
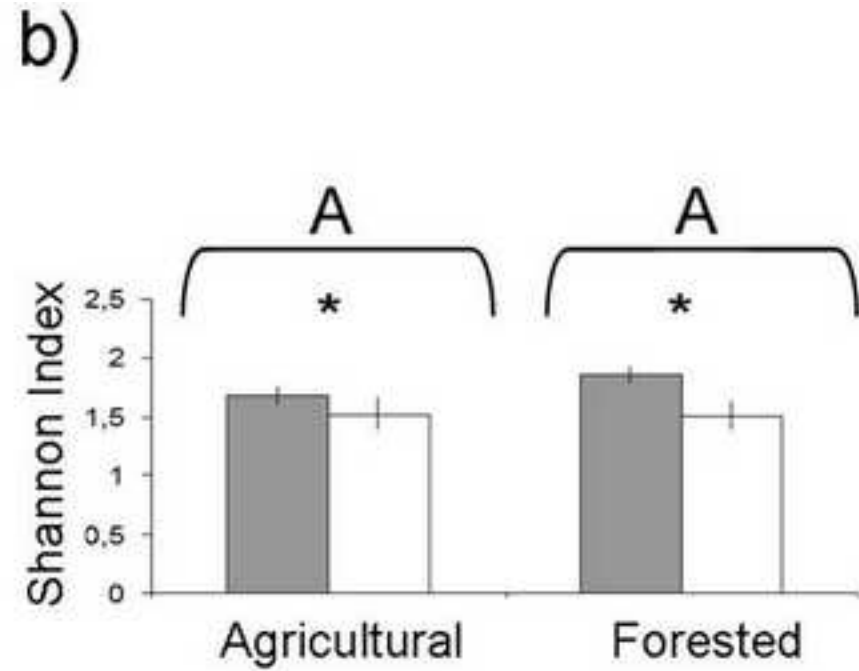
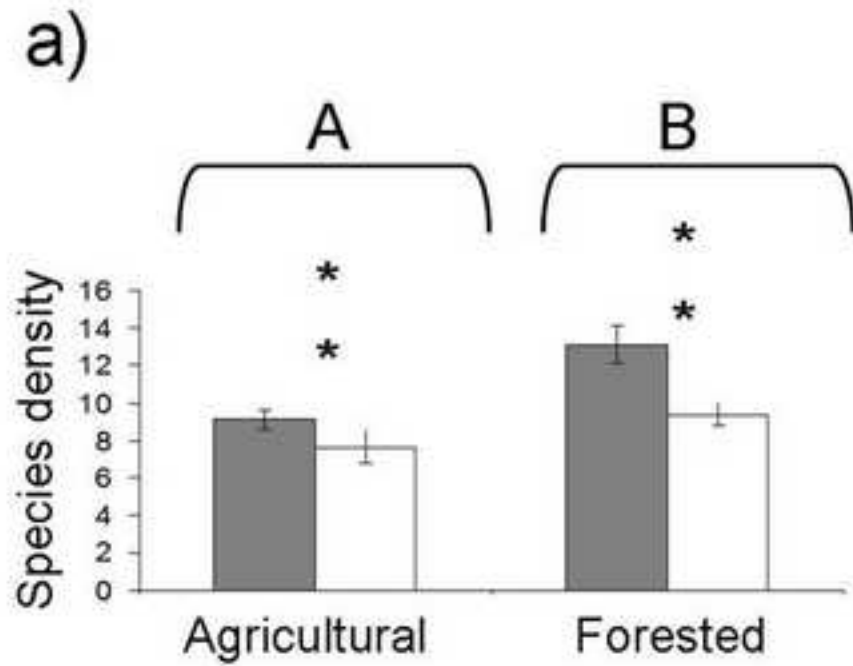


Figure3
[Click here to download high resolution image](#)



■ Old
□ Recent

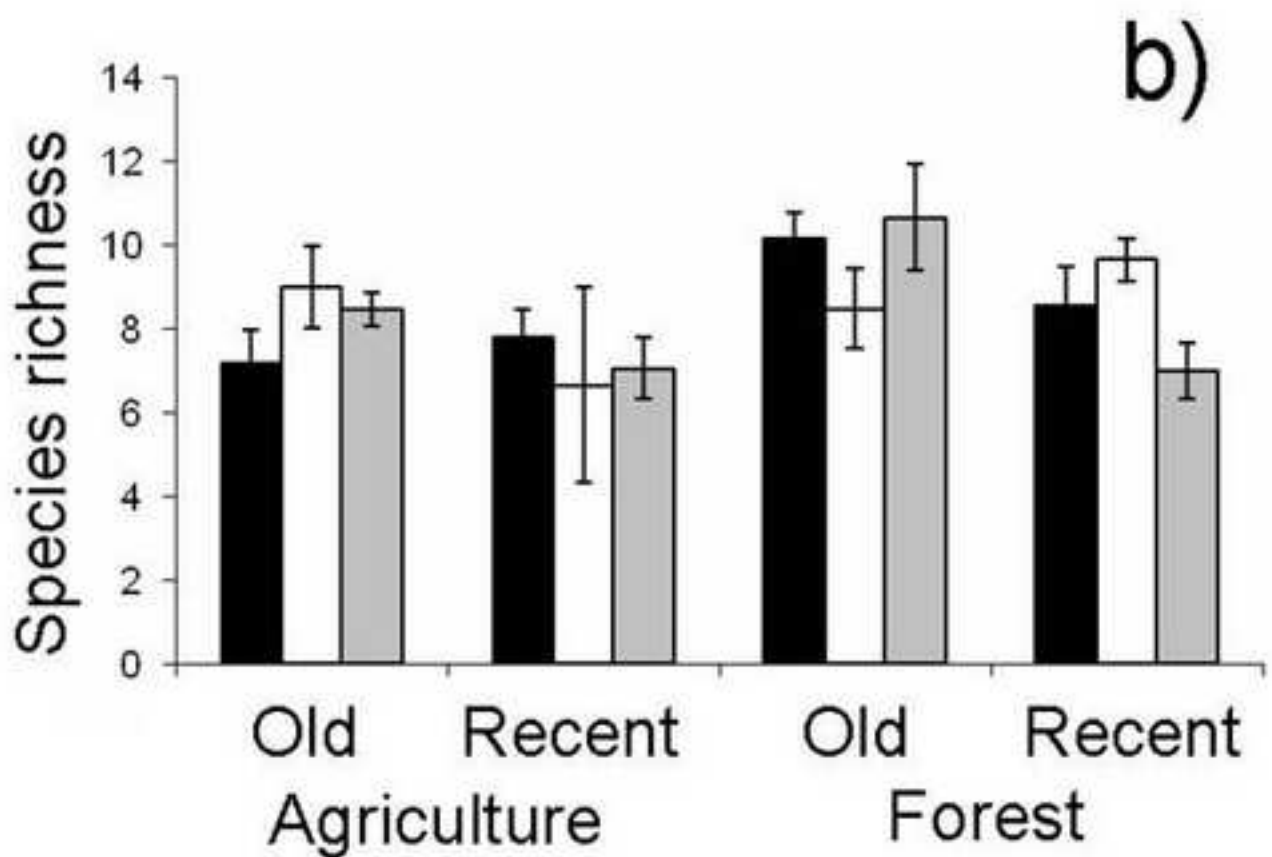
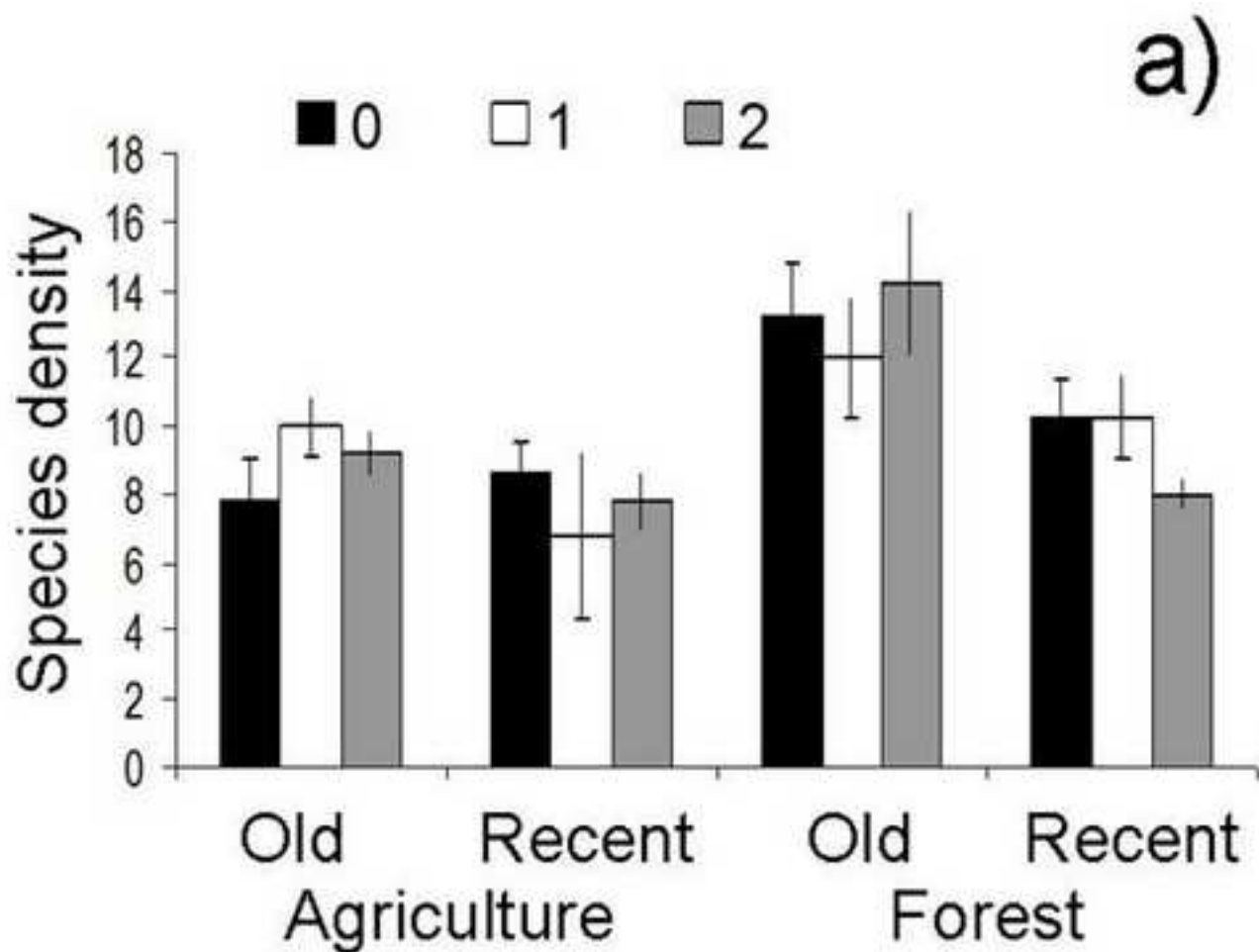


Figure5

[Click here to download high resolution image](#)

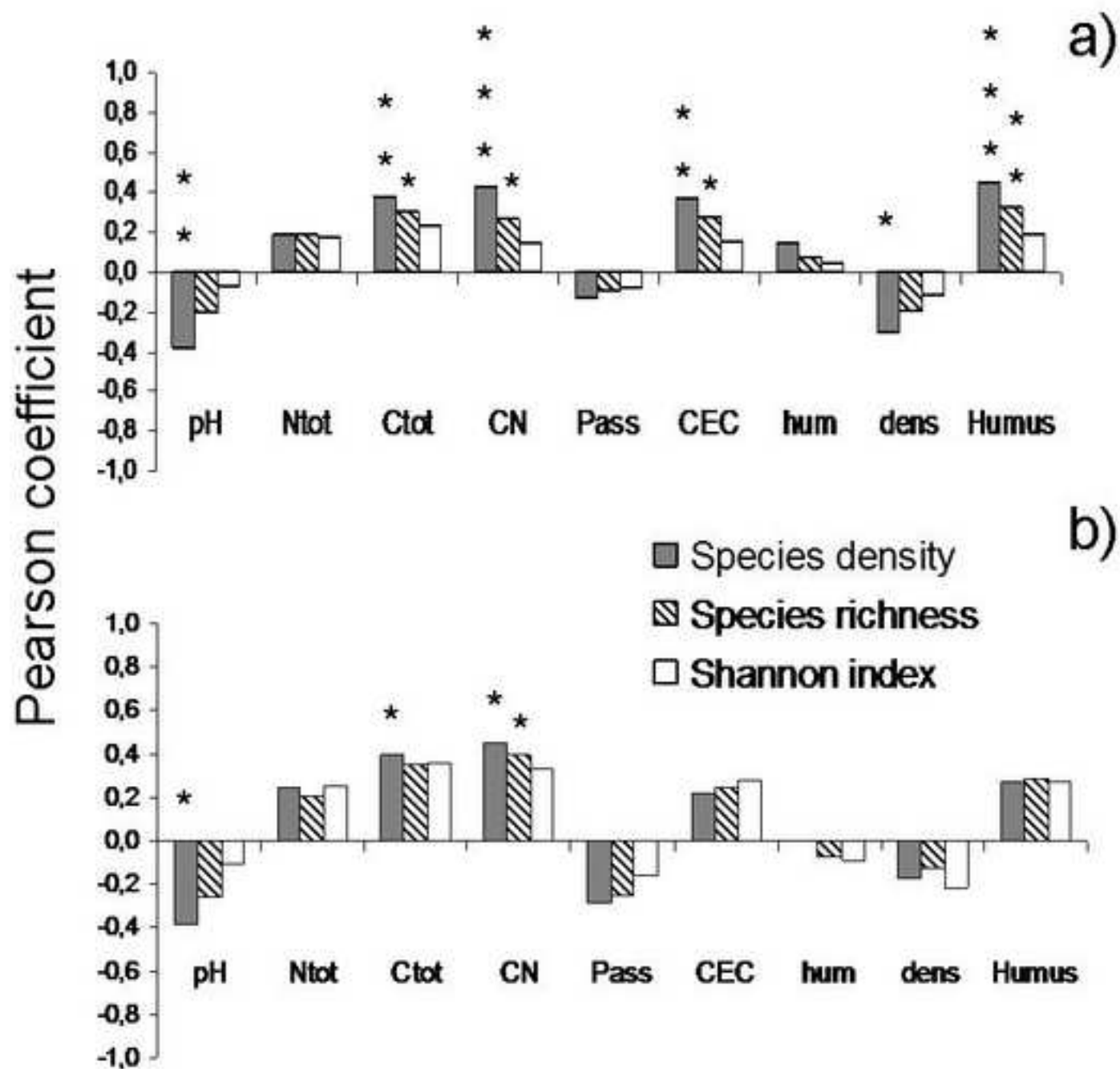
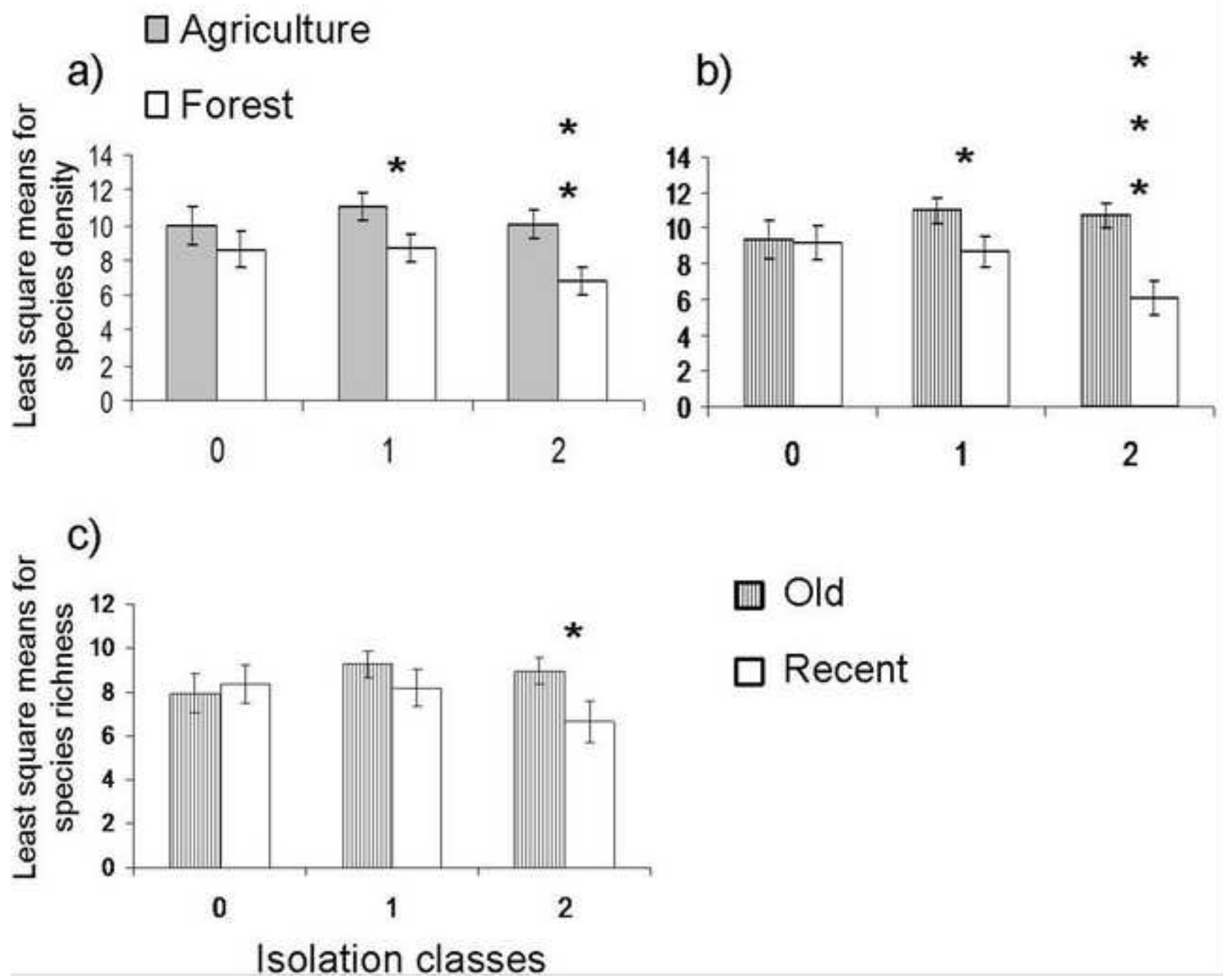


Figure 6

[Click here to download high resolution image](#)



1 **Table 1.** Effect of habitat descriptors (Habitat Type HT, Temporal Continuity TC and Spatial Isolation
 2 SI) and all their interactions on Collembola diversity indicators (Species density = total number of
 3 species collected in a sample of 0.2 dm², Rarefied species richness = number of species estimated for
 4 57 individuals, Shannon index, Dominance = relative frequency of most abundant species, Abundance
 5 = total number of individuals per sample). Transformation used on and results of linear models (F
 6 value and degrees of freedom df). Significance levels: p<0.05*; p<0.01**; p<0.001***.

7

		F value				
	df	Species density	Species richness	Shannon Index	Dominance	Abundance
Habitat type (HT)	1	14.2***	0.01*	0.54	0.83	7.09*
Temporal Continuity (TC)	1	9.84**	0.006**	5.23*	0.36	0.99
Spatial Isolation (SI)	2	0.03	0.97	0.23	0.17	0.15
HT : TC	1	2.29	0.33	0.97	0.17	0.71
HT : SI	2	0.12	0.84	0.22	0.17	0.55
TC : SI	2	1.06	0.34	0.44	0.14	0.25
HT : TC : SI	2	1.60	0.36	1.20	0.52	1.75
R ²		0.40	0.30	0.18	0.06	0.23
Transformation		none	none	none	log ₁₀	none

8

9 **Table 2.** Soil characteristics (mean values and standard deviations) showing differences between
 10 agricultural and forest habitats. Total carbon content (C_{tot}), soil pH (pH), carbon to nitrogen ratio
 11 (C/N), cation exchange capacity (CEC), soil moisture (Moisture), bulk density (Density) and Humus
 12 index (Humus). Significance levels: p-values correspond to results of linear and generalized linear
 13 models presented in Table 3 (p<0.05*; p<0.01**; p<0.001***).

14

15

16

17

18

19

20

21

22

	Agriculture	Forest	p-value
C _{tot} (%)	6.9 ± 2.41	8.74 ± 3.87	*
pH	5.14 ± 0.4	3.95 ± 0.39	***
C/N	12.55 ± 1.46	15.88 ± 2.88	***
CEC (meq%)	24.96 ± 3.82	36.57 ± 8.91	***
Moisture (%)	21.66 ± 14.37	28.78 ± 14.98	*
Density (g/cm ³)	0.9 ± 0.22	0.68 ± 0.22	***
Humus Index	1 ± 0	3.54 ± 1.55	***

23 **Table 3.** Effect of habitat descriptors (Habitat Type HT, Temporal Continuity TC and Spatial Isolation
 24 SI) and all their interactions on soil properties. Soil pH (pH), total carbon content (Ctot), total nitrogen
 25 content (Ntot), carbon to nitrogen ratio (C/N), bioavailable phosphorus (Phosphorus), cation exchange
 26 capacity (CEC), soil moisture (Moisture), bulk density (Density) and Humus index (Humus).
 27 Transformation, model type (distribution) and results of linear or generalized linear models (F
 28 value/Chi square and degrees of freedom df). Significance levels: p<0.05*; p<0.01**; p<0.001***.

		F / Chi square								
	df	pH	Ctot	Ntot	C/N	Phosphorus	CEC	Moisture	Density	Humus
Habitat type	1	185.77***	4.85*	0.12	68.45***	2.74	47.42***	4.13*	13.66***	39.56***
Temporal Continuity	1	2.34	2.16	1.05	9.43**	0.32	0.19	0.08	0.04	0.37
Spatial Isolation	2	6.28**	0.10	0.42	6.07**	3.74*	0.11	0.08	0.54	0.51
HT : TC	1	2.53	0.07	0.86	27.86***	2.69	0.72	2.69	1.14	0.36
HT : SI	2	5.60**	1.47	1.62	5.27**	2.94	1.68	0.66	1.27	0.51
TC : SI	2	1.34	1.12	1.51	1.27	2.92	0.63	0.25	0.21	0.21
HT : TC : SI	2	2.74	1.99	0.70	5.69**	0.85	0.90	0.27	0.78	0.21
R ²		0.82	0.26	0.12	0.77	0.35	0.55	0.12	0.28	0.63
Model type (distribution)		Normal	Gamma	Gamma	Normal	Normal	Normal	Gamma	Gamma	Poisson
Transformation		none	none	none	none	log ₁₀	log ₁₀	none	none	none

29

30

31 **Table 4.** Effect of habitat descriptors (Habitat Type HT, Temporal Continuity TC and Spatial Isolation
 32 SI), total soil carbon content (Ctot) (left hand side), soil pH (right hand side) and all their interactions
 33 on species density and species richness. Transformation and results of linear models (F values and
 34 degrees of freedom df), tested after variable selection based on AIC criterion). Significance levels:
 35 $p < 0.05^*$; $p < 0.01^{**}$; $p < 0.001^{***}$.

36

37

Ctot model				Soil pH model			
		Species density	Species richness			Species density	Species richness
	df	F value	F value		df		
Habitat type (HT)	1	0.79	0.15	Habitat type (HT)	1	5.98*	1.38
Temporal Continuity (TC)	1	1.9	0.05	Temporal Continuity (TC)	1	6.00*	6.56*
Spatial Isolation (SI)	2	0.2	0.02	Spatial Isolation (SI)	2	0.33	3.07
Total Carbon (Ctot)	1	0.00	0.66	pH H ₂ O (pH)	1	0.02	0.64
HT : TC	1	/	2.36	HT : TC	1	4.4*	2.34
HT : SI	2	4.14*	2.14	HT : SI	2	4.56*	1.80
TC : SI	2	10.13***	5.28**	TC : SI	2	4.04*	1.73
HT : Ctot	1	4.62*	0.02	HT : pH	1	5.23*	0.92
TC : Ctot	1	6.00*	0.03	TC : pH	1	9.84**	5.65*
SI : Ctot	2	0.07	0.07	SI : pH	2	4.89*	2.69
HT : TC : Ctot	1	/	2.07	HT : TC : pH	2	3.2	/
HT : SI : Ctot	2	5.04*	2.31	HT : SI : pH	1	2.56	1.59
TC : SI : Ctot	2	11.34***	5.74**				
R ²		0.67	0.49	R ²		0.62	0.49
Transformation		none	none	Transformation		none	none

Appendix A. List of study sites with their classification according to habitat type (agriculture vs forest), temporal continuity (old vs recent) and spatial isolation (three isolation classes), followed by % edge contrast with opposite habitat type, matrix type (old or recent or mixed), station code, land use, humus form according to Brêthes et al. (1995) and main community indices of Collembola (total abundance per sample, species density, species richness, Shannon index and dominance). The edge contrast is the % borderline which the patch with the sampling point shares with the opposite habitat in the circle of 300 m around the sampling point. The matrix type is the dominant age of the habitat surrounding the patch with the sampling point. The combination of both defines the level of isolation.

Habitat type	Temporal Continuity	Spatial Isolation	Edge contrast (%)	Matrix type	Station code	Land use	Humus form	Abundance	Species density	Species richness	Shannon index	Dominance			
Agriculture	Ancient	0	0	mostly old	A-old0 N=5	Hay meadow	EUMULL	47	9	9	1.93	0.32			
			0			Christmas tree	EUMULL	50	6	6	1.34	0.48			
			0			Crop	EUMULL	9	5	5	1.58	0.22			
			0			Pasture	EUMULL	41	7	7	1.47	0.51			
		1	both old and recent	0	A-old1 N=6	Hay meadow	EUMULL	212	12	8.89	1.83	0.33			
				25		Pasture	EUMULL	99	7	5.53	0.69	0.84			
				25		Hay meadow	EUMULL	56	8	8	1.82	0.23			
				25		Hay meadow	EUMULL	56	12	11.97	2.21	0.26			
				25		Pasture	EUMULL	75	12	10.94	2.09	0.3			
				25		Crop	EUMULL	128	11	9.75	2.05	0.23			
			2	mostly old	25	A-old2 N=9	Hay meadow	EUMULL	147	10	7.72	1.57	0.39		
					61		Hay meadow	EUMULL	101	11	9	1.32	0.66		
					74		Hay meadow	EUMULL	90	8	7.63	1.59	0.45		
					78		Hay meadow	EUMULL	112	9	7.66	1.69	0.37		
	Recent	0	mostly old	A-rec0 N=3	Hay meadow	EUMULL	244	10	7.6	1.78	0.32				
					Hay meadow	EUMULL	63	7	6.8	1.54	0.33				
					Pasture	EUMULL	25	9	9	2	0.27				
			1	both old and recent	A-rec1 N=4	23	Pasture	EUMULL	8	4	4	1.32	0.38		
						25	Christmas tree	EUMULL	16	2	2	0.23	0.94		
						32	Hay meadow	EUMULL	57	8	8	1.84	0.26		
		37				Pasture	EUMULL	60	13	12.6	1.88	0.36			
		2	mostly old	A-rec2 N=5	46	Hay meadow	EUMULL	80	8	7.59	1.59	0.3			
					46	Christmas tree	EUMULL	122	7	6.27	1.41	0.44			
					63	Hay meadow	EUMULL	11	5	5	1.55	0.27			
					80	Christmas tree	EUMULL	104	10	9.38	1.94	0.28			
		83	Hay meadow	EUMULL	191	9	6.97	1.21	0.65						
		Forest	Ancient	0	mostly old	F-old0 N=5	Deciduous	DYSMULL	65	10	9.49	1.64	0.51		
							Coniferous	DYSMODER	142	11	8.87	1.63	0.48		
Deciduous	DYMULL						227	19	12.4	2.21	0.27				
Deciduous	DYSMULL						247	14	9.44	1.79	0.43				
Deciduous	DYMODER						76	12	10.57	2.09	0.25				
1	both old and recent			F-old1 N=5	25	Deciduous	EUMODER	76	9	8.37	1.67	0.41			
					37	Coniferous	OLIGOMULL	118	13	10.31	1.87	0.41			
					51	Deciduous	DYSMULL	170	16	12.58	2.09	0.34			
					67	Deciduous	MESOMULL	78	7	6.85	1.26	0.63			
					83	Deciduous	DYSMULL	224	15	10.07	2.03	0.22			
					100	Deciduous	OLIGOMULL	115	19	14.74	2.36	0.32			
					2	mostly old	F-old2 N=5	100	Deciduous	OLIGOMULL	61	13	11.05	1.8	0.51
								100	Deciduous	DYSMULL	231	18	9.58	1.7	0.44
								100	Deciduous	DYSMULL	151	14	10.88	2.07	0.28
100	Deciduous		EUMULL	26				7	7	1.7	0.37				
Recent	0		mostly old	F-rec0 N=4	Coniferous	OLIGOMULL	86	8	7.07	1.38	0.41				
					Deciduous	OLIGOMULL	101	13	10.67	2.25	0.18				
					Coniferous	OLIGOMULL	180	9	6.78	0.99	0.75				
					Deciduous	OLIGOMULL	120	11	9.59	1.51	0.58				
	1		both old and recent	F-rec1 N=4	11	Deciduous	MESOMULL	82	10	8.52	1.72	0.32			
					19	Deciduous	OLIGOMULL	46	7	7	1.35	0.52			
					23	Deciduous	MESOMULL	165	12	8.94	1.58	0.39			
					24	Deciduous	DYSMODER	222	12	9.45	1.82	0.35			
					69	Deciduous	MESOMULL	18	9	9	2.08	0.17			
					100	Coniferous	MESOMULL	132	8	5.32	0.78	0.79			
	2		mostly old	F-rec2 N=5	100	Deciduous	OLIGOMULL	145	7	6.04	1.4	0.46			
					100	Coniferous	OLIGOMULL	92	9	8.04	1.6	0.48			
					100	Coniferous	OLIGOMULL	64	7	6.51	1.15	0.63			
		100			Coniferous	OLIGOMULL	64	7	6.51	1.15	0.63				

Appendix B. List of collembolan species in the 60 studied samples, together with total abundance and frequency (number of samples where the species was present)

Species names	Abundance	Frequency
<i>Arropalithes pygmaeus</i>	1	1
<i>Brachystomella parvula</i>	29	11
<i>Ceratophysella armata</i>	3	2
<i>Ceratophysella denticulata</i>	49	8
<i>Cryptopygus scapelliferus</i>	1	1
<i>Desoria violacea</i>	108	1
<i>Detriturus jubilaris</i>	42	6
<i>Deuteraphorura inermis</i>	53	7
<i>Deuterostminthurus pallipes</i>	11	4
<i>Entomobrya nivalis</i>	4	1
<i>Folsomia candida</i>	8	5
<i>Folsomia fimetaria</i>	4	3
<i>Folsomia listeri</i>	8	1
<i>Folsomia manolachei</i>	1	1
<i>Folsomia quadrioculata</i>	535	20
<i>Folsomia spinosa</i>	694	31
<i>Friesea claviseta</i>	3	1
<i>Friesea mirabilis</i>	6	1
<i>Friesea truncata</i>	8	1
<i>Heteromurus nitidus</i>	351	32
<i>Hypogastrura assimilis</i>	6	4
<i>Isotoma antennalis</i>	83	1
<i>Isotomurus fuscus</i>	37	7
<i>Isotomiella minor</i>	583	37
<i>Isotomurus palustris</i>	39	5
<i>Isotomiella paraminor</i>	75	6
<i>Isotomodes productus</i>	6	1
<i>Isotomodes trisetosus</i>	1	1
<i>Isotoma viridis</i>	161	23
<i>Lepidocyrtus cyaneus</i>	125	22
<i>Lepidocyrtus lanuginosus</i>	687	48
<i>Lepidocyrtus lignorum</i>	195	17
<i>Lipothrix lubbocki</i>	9	5
<i>Megalothorax minimus</i>	48	16
<i>Mesaphorura critica</i>	5	2
<i>Mesaphorura krausbaueri</i>	19	5
<i>Mesaphorura macrochaeta</i>	828	43
<i>Mesaphorura sp.</i>	14	3
<i>Mesaphorura sylvatica</i>	1	1
<i>Mesaphorura yosii</i>	76	7
<i>Micraphorura absoloni</i>	48	6
<i>Micranurida pygmaea</i>	22	8
<i>Micranurida sensillata</i>	2	1
<i>Neanura muscorum</i>	11	1
<i>Neotullbergia ramicuspis</i>	20	3
<i>Odontella lamellifera</i>	5	1
<i>Orchesella cincta</i>	2	2
<i>Orchesella sp.</i>	12	4
<i>Paratullbergia callipygos</i>	73	15
<i>Parisotoma notabilis</i>	650	42
<i>Protaphorura armata</i>	281	24
<i>Pseudosinella alba</i>	27	8
<i>Pseudanophorus binoculatus</i>	3	1
<i>Pseudosinella decipens</i>	2	1
<i>Pseudosinella maui</i>	53	15
<i>Pseudachorutes parvulus</i>	10	3
<i>Sminthurinus aureus</i>	31	12
<i>Sminthurides schoetti</i>	10	3
<i>Sminthurinus signatus</i>	20	10
<i>Sphaeridia pumilis</i>	72	19
<i>Stenaphorura lubbocki</i>	2	1
<i>Subisotoma pusilla</i>	1	1
<i>Tomocerus minutus</i>	2	1
<i>Willemia anophthalma</i>	3	2
<i>Willemia intermedia</i>	4	3
<i>Xenyllodes armatus</i>	3	2
<i>Xenylla corticalis</i>	1	1
<i>Xenylla grisea</i>	7	3
<i>Xenylla tullbergi</i>	42	6
<i>Xenylla xavieri</i>	3	2