1 **Toward the next CAP reform: determinants of avian communities in hay-meadows reveal current**
2 **policies' inadequacy for biodiversity conservation in grassland**
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13 **Summary**

14 1. Semi-natural grasslands are among the richest European ecosystems in biodiversity, although they have

15 been severely impacted by farming intensification and land abandonment, both exacerbated by European

16 Union's Common Agricultural Policy (CAP). The last CAP included a 'greening' measure dedicated to

17 grassland conservation, presumed to be beneficial for biodiversity; however, scientific evidence about its

18 effectiveness is still very scarce. In the Alps, hay-meadows underwent dramatic management changes in

19 the recent decades; here, we used a comprehensive community ecology approach to highlight how the

20 multi-scale and interacting effects of such changes are impacting on birds, with the aim of providing

21 knowledge to support improvements to the CAP.

22 2. Birds were surveyed at 63 landscape units in NE Italy, equally subdivided into areas dominated by i)

23 extensive or ii) intensive hay-meadows and iii) areas formerly dominated by meadows but partially

24 converted into other agricultural land-use. This environmental gradient mirrors in space the temporal

25 gradient of the agricultural changes occurred in recent decades in the Alps.

26 Community composition, species richness and the number of meadow specialist species were analysed

27 according to environmental predictors (i.e. landscape, meadow management, and topography), and to

28 spatial factors, aiming at disentangling the exclusive and joint fraction of variation explained by each of

29 them.

30 3. Meadow conversion, allowed by the CAP in force, created a shift in community composition toward

31 assemblages dominated by generalist species at the expense of meadow specialists.

32 The cover of intensive meadows was negatively correlated with species richness, whereas the number of

33 meadow specialists was negatively correlated with the cover of early mown (i.e. within the third week of

34 June) meadows. Mowing date was in turn related to elevation, with meadows at higher elevations mown

35 later in the season, and to meadow intensification (the use of external inputs determines an increase in the

36 number of cuts per year).

37 4. *Policy implications.* Our study confirms the concerns about effectiveness of the CAP grassland measure in

38 conserving biodiversity. We suggest rethinking the CAP environmental prescriptions to account for the

39   importance of Alpine meadow management in determining biodiversity patterns. Finally, market-based

40   conservation strategies are discussed as complementary approaches for preserving grassland biodiversity.

41

42   **Keywords -** avian assemblages, Alps, Pillar One, CAP greening, manure, meadow specialist, mowing.

43 **Introduction**

44

45 The support towards agriculture provided by the Common Agricultural Policy (CAP) is the most expensive

46 part of the European Union's budget (40% of the total EU expenses; European Commission 2013). CAP has

47 been widely acknowledged as the major driver of agricultural intensification and abandonment in Europe

48 (at least form the Seventies onward), with strong negative impacts on biodiversity (Bignal 1998; Renwick et

49 al. 2013).

50 These effects led to a 2013 CAP reform (in force until 2020), that was announced to be "greener" with 30%

51 of direct payments to farmers dependent on "greening measures" aimed at halting biodiversity loss. One

52 such measure was targeted at counteracting permanent grassland reduction (Pe'er et al. 2014).

53 Most European grasslands are semi-natural habitats characterized by native plants, but they were created

54 to sustain livestock through maintenance by mowing and/or grazing. Semi-natural grasslands are among

55 the most iconic and biodiversity-rich European landscapes (Pykälä 2000; Veen et al. 2009), covering c.8% of

56 the continent and 35% of the utilized agricultural areas (Smit et al. 2008).

57 Permanent grassland has decreased in the EU by 6.4% between 1993 and 2011, and by 11.8% in countries

58 that joined the EU by 2004 (Pe'er et al. 2014), due to conversion to other land-use or abandonment

59 (MacDonald et al. 2000; Laiolo et al. 2004). In several regions, remaining grasslands have been strongly

60 intensified (Humbert et al. 2016). These two opposite processes have been worsened by the CAP (Donald *et*

61 *al.* 2002; Souchère *et al.* 2003) and have had major negative impacts on grassland biodiversity (Vickery et

62 al. 2001; Monteiro et al. 2011).

63 The last CAP greening measure dedicated to permanent grassland (enacted in 2013) is unlikely to have

64 widespread positive effects on grassland biodiversity because: i) it allows a further 5% reduction in

65 grassland extent at a national/regional scale by 2020, an amount higher than the current loss rate in several

66 regions; ii) it has an obligation to maintain the overall grassland area, but not the individual parcels,

67 allowing farmers to plough, reseed and relocate them, iii) it does not distinguish among grassland

68 typologies, so that grasslands of any natural value contribute the same to the overall grassland quota; iv)

69 further degradation by management intensification is allowed due to the lack of targeted environmental

70 prescriptions (Pe'er et al. 2014, 2017; Dicks et al. 2014).

71 Scientific evidence of the measure's effectiveness is still very scarce, given the short period of

72 implementation, and its impacts on biodiversity can only be estimated (Pe'er et al. 2017). Notwithstanding

73 this, discussion on the next reform of the CAP has advanced (European Commission 2017), and appropriate

74 evaluations on grassland measures are urgently required to halt biodiversity loss and to address

75 management strategies and policy for this ecosystem, which is pivotal for biodiversity conservation in

76 Europe.

77 Hay-meadows represent the most emblematic and biodiversity-rich traditional agroecosystem in the Alpine

78 region (Kampmann et al. 2008; Agnoletti 2013) and represent an urgent conservation issue due to the

79 ongoing deep changes in traditional farming (Fischer et al. 2008; Henle et al. 2008). They are permanent

80 semi-natural grasslands, maintained by mowing and only occasionally grazed. Their widespread occurrence

81 in the Alps is related to an extensive form of livestock farming, in which cattle (mainly cows) spend the

82 summer in mountain pastures and the other seasons in stables, fed with the fodder obtained from

83 meadows (Marini et al. 2011; Monteiro et al. 2011).

84 From the second half of the 20th Century, and more markedly in the last c.40 years, the dairy sector in the

85 Alps underwent deep changes, with a concentration of livestock in a few, much larger farms. These farms

86 are highly specialized and breed larger and more productive cows (e.g. Holstein Friesian), largely fed with

87 concentrated cereal feedstuff. This results in a reduction of summer grazing in upland pastures and in a

88 higher production of organic fertilizer, often deposited on meadows as liquid manure (Marini et al. 2011;

89 Graf et al. 2014). Simultaneously, marginal, less productive grasslands were abandoned or converted into

90 more profitable cropland (Zimmermann et al. 2010; Monteiro et al. 2011). This transformation of the dairy

91 sector in the Alps was due to several reasons (i.e. social and cultural changes, local investments in

92 mountain development, technological innovation and policy), although the CAP played a crucial role in

93 sustaining this process (Marini *et al.* 2011; Tappeiner *et al.* 2003). Those agricultural changes impacted (and

94 are still heavily impacting) plant communities (Marini et al. 2008b; Niedrist et al. 2009; Pierik et al. 2017)

95 and invertebrate assemblages (Marini et al. 2008a; Andrey et al. 2014). However, it is uncertain how they

96   are affecting the upper level of food webs. Although several bird species are sensitive to meadow

97   management (Britschgi et al. 2006; Sergio et al. 2009), and especially to mowing (Broyer 2009; Pedrini et al.

98   2012; Strebel et al. 2015), to the best of our knowledge, studies on the multi-faceted effects of modern

99   hay-meadow management on avian communities had never been performed in the Alps.

100   Here we investigate the effects of recent grassland transformation on hay-meadow bird communities,

101   considering broad- (e.g. landscape) and fine-scale (e.g. management) drivers in the Alps. We aimed at

102   disentangling the role in structuring avian assemblages of: i) meadow conversion into more productive

103   crops; ii) landscape-scale intensification, and iii) in-field meadow intensification (increase of external inputs

104   and changes in mowing regimes).

105   Our ultimate goal is to identify best options for biodiversity conservation in this iconic agroecosystem and

106   to provide recommendations for improving the forthcoming CAP reform.

107

108   **Materials and methods**

109

110   ***Study area, design and bird surveys***

111   The study was conducted in the Trento province (NE Italy; 45.67-46.51° N; 10.51-11.96° E; Fig. 1a). Here,

112   anthropogenic grasslands are scattered above 250m and are more widespread between 800–2,000m asl.

113   Permanent hay-meadows cover roughly 200 km^2, representing 3.3% of the province surface and 14.8% of

114   the Utilised Agricultural Area (ISTAT, 2010).

115   In 2010, 54927 Livestock Units occurred in Trentino, distributed over c.1400 farms (La Notte et al. 2015).

116   From 1990 to 2010, the overall surface of hay-meadows in this region almost halved (Provincia Autonoma

117   di Trento 2017), leading to a considerable increase in stocking rate (Scotton et al. 2012). Rural

118   abandonment was lower than in other Alpine areas, partly due to specific incentives in the framework of

119   the Rural Development Programs, and meadow reduction was primarily due to conversion into other crops

120   (Streifeneder et al. 2007; Marini et al. 2011).

121   Birds were surveyed along 63 200m-long linear transects, scattered over nine areas along an altitudinal

122   gradient (310-1565m asl) encompassing the entire belt in which meadows are found in the study area (Fig.

123   1b).

124   Bird surveys and environmental variable collections were performed within a 100m-buffer around the

125   transect; these 7.15-ha Landscape Units (LU) became the sampling units of the study and were selected

126   according to a stratified design: 22 units were dominated by extensive hay-meadows, 20 units by intensive

127   hay-meadows, and 21 by hay-meadows partly converted into other crops.

128   These three grassland typologies (extensive, intensive, converted) approximate well the modifications that

129   occurred in the last decades in hay meadow-dominated landscapes in our study area, and more in general

130   in several areas in the Alps. In other words, we studied the avian communities found at a specific time in

131   multiple sites chosen along a gradient of hay meadow intensification, assuming this one-year 'snapshot' as

132   a proxy of the community changes occurred in the last c.40 years in relation to the widespread changes

133   from extensive to intensive and then converted grassland. This approach, known as space-for-time

134   substitution, is commonly adopted in ecological studies when long-term data are unavailable and assumes

135   that changes in space reflect those in time, i.e. that there are no important interactions with other factors

136   such as climate or period. This is a simplification of the patterns occurring in real systems, but can still

137   prove useful to understand the impacts of environmental changes on biodiversity(Pickett 1989; Bennett et

138   al. 2006). In our study case, we are confident, thanks both to personal experience and botanical surveys ,

139   that current grasslands mirror the main characteristics of past grasslands of the same type.

140   We censused birds during three visits in the 2017 breeding season (12-24.05; 13-23.06; 2-12.07), surveying

141   six/seven transects per morning (see Assandri et al. 2017c and SI for further details).

142

143   ***Avian and environmental variables***

144   From bird surveys, we derived three community variables for each LU: community composition, breeding

145   species richness (number of breeding species), and the number of meadow specialist species. Before

146   computing community variables, we removed records related to juveniles, overflying birds, species not

147   breeding in the study area, and species observed only once.

148   Composition assessment was based on the maximum abundance of each species recorded across the three

149   surveys, which were entered into a site (i.e. LU) by species matrix. Late season high counts (i.e. those

150   exceeding of 10 units the mean of the previous session counts in the same LU) of early-breeding species,

151   which display gregarious habits at the end of the breeding seasons (e.g. flocks of finches, sparrows and

152   corvids) were excluded to avoid abundance over-estimation (Jakobsson and Lindborg 2017). We considered

153   as meadow specialists the species which: i) are known to depend on grassland habitats in the study region

154   (Pedrini et al. 2005), and ii) with >50% of their precise locations falling within meadow. Those include *Crex*

155   *crex*, *Coturnix coturnix*, *Saxicola rubetra*, *Anthus trivialis*, *Alauda arvensis*, *Sylvia nisoria*, *Lanius collurio*,

156   *Turdus viscivorus*, *Emberiza citrinella. Pica pica* and *Corvus corone*, which are habitat generalists in the

157   region, satisfy only the second condition and thus were discarded.

158   At each LU, we measured three types of environmental variables: landscape, management, topography,

159   and built a set of spatial variables based on LU coordinates.

160   Landscape variables were measured on aerial photographs that had been validated and updated in the

161   field. We calculated the relative cover of meadows, urban areas, woodlands, traditional high-stem

162   orchards, shrublands/fallows (including small wetlands), and converted meadows (former meadows

163   recently converted into other arable crops, intensive orchards, vineyards, or greenhouses). Based on these,

164   we calculated a land-cover H' Shannon diversity index (Laiolo 2005). The length of hedgerows (with and

165   without trees) and tree rows, and the number of isolated trees and shrubs were also assessed.

166   Management variables were evaluated at the parcel scale (i.e. a meadow with a defined plant community,

167   spatial arrangement, and management characteristics; 882 parcels identified); the percentage surface of

168   highly intensive and that of early mown meadows occurring in each LU were used as management

169   predictors. Highly intensive meadows were defined as species-poor meadows, highly fertilized (85-420 kg N

170   ha-1 year-1) and mown 2-3 times/year in contrast with low intensive meadows (species-rich meadows, not

171   fertilized or poorly fertilized (0-150 kg N ha-1 year-1), and subject to only 1 or, rarely, 2 cuts per year).

172   Early mown meadows were defined (in contrast with late mown) as meadows mown before the end of the

173   third week of June (see SI for further details).

174 Topographic variables (mean elevation and slope) were derived from a 1-m resolution digital elevation

175 model.

176 Spatial variables were built by means of Moran's eigenvector maps (MEMs) (Dray et al. 2006), a method

177 that produces flexible spatial predictors starting from sample plot coordinates, and capturing spatial effects

178 at multiple spatial scales that can be used in regression and ordination to account for spatial

179 autocorrelation (Borcard et al. 2011). Further details are available in SI.

180

181   *Analyses*

182 All the analyses were performed with R version 3.4.1 (R Core Team, 2017). We scrutinized the three groups

183 of environmental predictors to avoid common statistical problems, following Zuur et al. (2010). We left out

184 meadow cover and landcover diversity from subsequent analyses due to high collinearity with converted

185 meadow area, both pairwise (Spearman's Rho: meadow cover=-0.75; p<0.001; landcover diversity=-0.90;

186 p<0.001) and multivariate (GVIF: converted meadow cover=29.91; meadow cover=40.39; landcover

187 diversity=11.83). Elevation and slope showed a low collinearity (Rho < 0.5), and both were retained. We

188 applied a log+1 transformation to hedgerows, tree rows, and slope to reduce the weight of outliers. All the

189 explanatory variables were standardized before analysis (Schielzeth 2010; Cade 2015).

190 We separately tested the effect of each group of environmental and spatial (MEM) predictors on the three

191 response variables. Community composition was analysed by means of redundancy analysis (RDA)

192 performed with the package *vegan* (Oksanen et al. 2017). The abundance site-by-species matrix was first

193 Hellinger-transformed to make it appropriate for linear analyses (Legendre and Gallagher 2001). Global

194 significance ($p < 0.05$) was assessed by means of ANOVA-like permutation (N=999) tests. Overall, significant

195 RDAs were obtained by using forward selection (performed in *adespatial* package; Dray et al. 2017), which

196 retained significant variables by applying a double-stopping criterion (Blanchet et al. 2008), which reduces

197 type I errors and the overestimation of explained variance.

198 We built Poisson Generalized Linear Models (GLMs) to evaluate predictor effects on species richness and

199 the number of meadow specialists by adopting an information-theoretic approach (Burnham and Anderson

200 2002) and ranking all possible models for each set of explanatory variables separately according to the

201 relative value of Akaike's Information Criterion corrected for small sample size (AICc). The most

202 parsimonious models (ΔAICc ≤ 2) were selected and averaged within each group of predictors weighing by

203 model weights, and obtaining model-averaged coefficients, their relative SEs, and the relative variable

204 importance (Johnson and Omland 2004) for each explanatory variable applying the "zero-method" (*sensu*

205 Grueber et al. 2011). 'Uninformative parameters' (Arnold 2010), i.e. the variables which, when included,

206 determined an increase of the model's AICc value, were discarded (Richards 2008; Richards et al. 2011).

207 The inclusion of more models (ΔAICc ≤ 6) led to substantially similar results. For subsequent variation

208 partitioning (VP) analyses, we retained only the variables with confidence intervals of parameter estimates

209 not encompassing zero.

210 VP was applied to disentangle the unique and joint fraction of variation among the response variables

211 explained by the four sets of predictors. We performed VP on parsimonious models (simplified models

212 resulting from model selection; Peres-Neto and Legendre 2010) (see SI for details).

213

214 **Results**

215 We obtained 3037 bird records referring to 88 species; we removed records related to 13 species observed

216 only overflying LUs, 10 species exclusively migrants and 6 species observed only once (Table S2).

217 The final dataset included 59 species and 2040 individuals (adults). Ten species alone accounted for c.59%

218 of the data (see SI).

219

220 ***Community composition***

221 The community composition was significantly structured by eight environmental variables (Table S3).

222 *Saxicola rubetra*, *Anthus trivialis*, *Emberiza citrinella*, *Coturnix coturnix,* and *Pica pica* were grouped

223 together and were negatively correlated with the cover of converted meadows (and with orchard and

224 hedgerows, which were in turn positively correlated with the agricultural landscapes that were different

225 from meadows). This group of species also included at its margins, *Turdus viscivorus* (positively related to

226 shrubs and woodland) and *Lanius collurio* (Fig. 2a). All these species but the latter were also positively

227 associated with elevation and secondarily, slope and negatively with early mown grassland. Variables

228   indicating low-elevation agroecosystems different from meadows were positively associated with *Chloris*

229   *chloris*, *Serinus serinus*, *Passer montanus*, *Turdus merula*, *Passer italiae* and less strongly with *Linaria*

230   *cannabina*, *Jynx torquilla*, *Picus viridis* (these three favoured by hedgerows) and *Muscicapa striata*, *Turdus*

231   *philomelos*, *Cyanistes caeruleus*, *Sylvia atricapilla*, *Parus major*, *Columba palumbus*, which were related to

232   more forested landscapes. The woodland specialists found within sampled LUs (which are meadow-

233   dominated) were all clearly aligned according to woodland cover, with several also favoured by slope (Fig.

234   2a-c).

235   Ten MEMs were retained in the parsimonious spatial RDA (Table S3).

236   RDAs had an overall low explanatory power (Fig. 4), as is commonly found in ecological studies (Borcard et

237   al. 2011).

238

239   ***Species richness and meadow specialist species***

240   The most supported models on the effect of environmental and spatial predictors on species richness and

241   the number of meadow specialists are summarized in Tables S4 and S6.

242   Species richness was positively affected by woodland cover, the length of woody hedgerows and tree rows,

243   and slope, and negatively by the cover of highly intensive meadows and elevation (Fig. 3a-f). The positive

244   effects of urban cover and hedgerow length were weaker, since their estimates had confidence intervals

245   encompassing zero. Two MEMs were retained after model selection, but only MEM12 had confidence

246   intervals not encompassing zero (Table S5).

247   The number of meadow specialists was negatively affected by the cover of converted meadows and of

248   early mown meadows, plus the length of tree rows and orchard cover (although the latter had confidence

249   intervals encompassing zero), positively by elevation plus the number of shrubs (although its confidence

250   intervals encompassed zero) (Fig. 3g-i). Nine MEMs were retained after model selection, but only MEM3

251   and MEM9 had confidence intervals for the estimate not encompassing zero (Table S7).

252

253   ***Variation partitioning***

254   Variation partitioning highlighted a complex contribution of different sets of predictors in explaining

255   variation in the response variables; shared fractions were often conspicuous, indicating that multiple

256   factors co-explain the observed patterns (Fig. 4). For community composition and the number of meadow

257   specialists, the intersection of all four components was particularly relevant.

258   For community composition, the spatial component had the main role in explaining the observed variation,

259   and 41% of this component was unique (i.e. not joint), followed by landscape and topography components,

260   which were for three quarters interrelated to other components. Management (cover of early mown

261   meadows) explained a reduced quota of variation and was always interrelated with other components.

262   For species richness, variation was mainly explained by landscape (35% uniquely) and topography

263   components (mostly jointly with landscape), whereas management (cover of highly intensive meadows)

264   and spatial components had a limited explanatory power.

265   Vice-versa, the number of meadow specialists was mainly jointly explained by management (cover of early

266   mown meadows) and topography (elevation); about half of the variation explained jointly by management

267   and topography was also explained by the landscape component (cover of converted meadows). The

268   contribution of the spatial component was mostly interrelated with that of other components.

269

270   **Discussion**

271   We used a broad and comprehensive community ecology approach to show the impacts of the multi-scale

272   and interacting trajectories of Alpine grassland changes on birds, which here occupy the highest levels of

273   the trophic web. Mountains have been reported to act as refuges for lowland farmland species, which

274   strongly declined at lower elevations in response to agricultural mechanisation and intensification (Schmid

275   et al. 1998). A study in the French Alps showed that this is still true for farmland generalists, but that

276   farmland specialists (which match quite well with our meadow specialists) also declined at elevations

277   higher than 1000 m (Archaux 2007). Coherent patterns had been reported from the Swiss Alps (Korner et

278   al. 2017) and suggest that recent agricultural transformations are also impacting these species in mountain

279   'refuges', which are rapidly losing their conservation potential for farmland birds.

280   These negative transformations are at least partly linked with the European CAP. The greening measure

281   specifically included in the current CAP for permanent grassland, aimed at halting grassland biodiversity

282   loss, is unlikely to invert this negative trend (Pe'er et al. 2014). One of the main shortcomings of this

283   measure is that it allows a further 5% grassland reduction at national and regional scales. Our study

284   highlighted that one of the major causes of grassland reduction in the Alps, the conversion of hay-meadows

285   into other crops, does not drive an increase on overall avian biodiversity (species richness) in a meadow

286   dominated landscape. Such an increase in species richness could have been expected, because different

287   crops co-occurring at a landscape scale could increase overall landscape heterogeneity, which is widely

288   recognised to be positively correlated with the diversity of many taxa (Benton et al. 2003; Fahrig et al.

289   2015). Results showed that meadow conversion caused a shift in the community composition towards

290   assemblages dominated by generalist species, especially Turdidae and Fringilladae, which are known to

291   adapt to intensive permanent monocultures (i.e. apple orchards and vineyards; Brambilla et al. 2015;

292   Assandri et al. 2017a), which usually replace semi-natural grassland in the Alps. These changes occurred at

293   the expense of the meadow specialists, which disappear with the increase of these crops. The conversion of

294   hay-meadows into other crops thus has a clear negative effect on meadow specialists, which are mostly

295   decreasing species at the European level.

296   The cover of highly intensive meadows was negatively correlated with overall species richness. This pattern

297   could be due both to the fact that intensive meadows tend to occur in more agriculturally intensive

298   landscapes (which in most cases host a lower biodiversity than extensive ones; e.g. Verhulst *et al.* 2004),

299   and to a negative effect of meadow intensification *per se* on bird assemblages, via an effect on generalist

300   species too. These latter species, which are not obligate meadow-dwellers (e.g. doves, woodpeckers,

301   raptors, wagtails, corvids, thrushes, finches, starlings, and sparrows), often use meadows for feeding.

302   This negative effect did not emerge for the community composition and meadow specialists, mainly

303   affected by the mowing regime. Early mown meadows (i.e. those mown within the third week of June) have

304   considerably fewer meadow specialists than meadows mown later. The impact of (intensive) hay making on

305   birds determines high rates of nest destruction and nestling mortality (Schekkerman et al. 2009;

306   Buckingham et al. 2015), finally resulting in lowering reproductive success (Müller et al. 2005; Broyer 2009),

307   modifying behaviour (Grüebler et al. 2015) and even mating systems, sexual selection, and consequently,

308   evolutionary processes (Perlut et al. 2008), turning once favourable habitats into ecological traps (Broyer et

309   al. 2012). Also in our study area, modern hay-making is resulting in a drastic decline (sometimes eventually

310   leading to local extinction) of once very common meadow birds breeding on the ground (e.g. *Coturnix*

311   *coturnix, Crex crex, Alauda arvensis, Anthus trivialis, Saxicola rubetra*, and *Emberiza citrinella*; Pedrini et al.

312   2005, Brambilla and Pedrini 2013; this study).

313   The date of mowing, as shown by variation partitioning (Figure 4c), is strictly related to elevation, with

314   meadows at higher elevations mown later in the season (see also Fig. S3). This is mirrored by the positive

315   effect of elevation on meadow specialists, opposite to the effect on species richness in our study area and,

316   in general, in mountain ranges worldwide (Nogués-Bravo et al. 2008; McCain and Grytnes 2010). This

317   confirms the key importance of meadow management and its interaction with seasonal progression (which

318   is delayed at higher elevation) for grassland birds in areas with elevation gradients (Brambilla and Rubolini

319   2009; Brambilla and Pedrini 2011).

320   Recent intensification in meadow management also determines an earlier mowing, since the two are

321   intimately connected in a cyclic positive feedback: the use of external inputs (water/fertilizer

322   supplementation) allows advanced mowing (and then increased number of cuts/year) and to sustain a

323   larger number of (more productive) cattle, which produce more manure that is usually deposited on

324   meadows, further increasing their productivity (Scotton et al. 2014). In-field meadow intensification thus

325   also negatively affects meadow specialists. It also promotes new vegetation types dominated by a few

326   species to the point that, after some years, only a few nitrophilous species of low forage value occur, and

327   the semi-natural meadows have to be ploughed and re-sown with industrial seed mixtures (Marini et al.

328   2008b; Andrey et al. 2014; Humbert et al. 2016). This is also a widespread practice in our study areas,

329   where 9% of meadows are reseeded, and 10% were dominated by weeds (e.g. Apiaceae), i.e. those near to

330   being ploughed and reseeded (see SI). This causes a shift from a long-established ecosystem dominated by

331   autochthonous plants and harbouring structured animal communities (Humbert et al. 2009), into a new,

332   artificial and temporary one, with few (and often allochthonous) species. However, according to the current

333   grassland CAP measure, these degraded meadows, as well as the intensively managed ones, contribute the

334   same as the biodiversity-rich, permanent and unimproved grassland in reaching the grassland quota

335   needed to access the greening requirement under the CAP (Pe'er et al. 2014).

336   Previous studies highlighted that steepness better predicts the occurrence of meadows with high diversity

337   of plants (Marini et al. 2008b) and invertebrates (Marini et al. 2011) than elevation. This occurs because

338   steeper slopes prevent meadow intensification (e.g. irrigation and fertilization are difficult to achieve);

339   additionally, the more extreme microclimates of these grasslands are disadvantageous to more competitive

340   plant species while favouring less competitive ones, allowing them to co-exist and to increase diversity

341   (Marini et al. 2008b). Comparably, in our study areas, less intensified meadows and richer plant

342   communities are found on steeper slopes (Fig. 5). However, slope has a positive effect on overall species

343   richness, but not on meadow specialists. The community composition analysis indeed suggests that the

344   positive effect of slope on species richness is most likely related to the fact that the steepest slopes are

345   associated with abandoned areas invaded by shrubs and first-stage successional woods and this, in a

346   landscape dominated by meadows, allows some species (e.g. *Prunella modularis*, *Troglodytes troglodytes*,

347   *Phylloscopus collybita*, *Erithacus rubecula*) to occur, increasing the overall species number (Laiolo et al.

348   2004). This is also confirmed by the observed positive effect of woodland cover (and also of hedgerow and

349   tree row length) on overall species richness.

350   Landscape linear elements, such as hedgerows and tree rows, often favours birds in agroecosystems

351   (Baudry et al. 2000; Hinsley and Bellamy 2000); however, hedgerow networks (or other forms of tree and

352   shrub restoration) may negatively affect grassland specialists, via habitat fragmentation for open-habitat

353   species (Besnard and Secondi 2014; Assandri et al. 2017b). In our study, tree rows do not have any effect

354   on meadow specialists, whereas woody hedgerows have a negative (non-significant) effect. Conversely,

355   isolated shrubs are positively related with most of these species, whereas (shrubby) hedgerows are

356   positively related with species found in mixed farmland (e.g. *Picus viridis, Jynx torquilla, Sturnus vulgaris,*

357   *Passer italiae*) and locally with two species commonly found in grassland, *Lanius collurio* and *Sylvia nisoria*.

358   In areas in which meadows were converted to other crops, the maintenance or creation of hedgerows with

359   shrubs could sustain richer avian communities and declining farmland species, whereas in grassland-

360 dominated areas, where hedgerows are not part of the traditional landscape, their creation is not

361 recommended.

362

363 **Conclusion**

364 With the next CAP reform expected soon, recommendations for major amendments to current measures

365 are particularly timely and are required to reverse the negative trend of farmland species and, in particular,

366 of avian grassland specialists, which still do not show any recovery (Inger et al. 2014; Gamero et al. 2017).

367 If the conservation of grassland biodiversity is a priority for the European Union, the CAP greening measure

368 referring to permanent grassland must be rethought. Our study confirms the concerns about its low

369 expected effectiveness for biodiversity conservation for birds (Pe'er et al. 2014). The further 5% grassland

370 reduction at the national and regional scale means -in most cases- further conversion to other crops, with

371 potentially severe consequences for biodiversity. Additionally, the lack of distinction between intensive,

372 low-natural value grasslands and traditional, high-natural value grasslands, could lead to further

373 biodiversity losses, since vast extents of permanent grassland could be transformed into non-permanent

374 grassland regulated by ploughing and reseeding, which are allowed by the measure. If the 5% threshold had

375 to be maintained, it would be essential to avoid the conversion of unimproved/low-intensive meadows.

376 This implies that further efforts and resources should be allocated to map unimproved meadows and to

377 design high tier agri-environment schemes to compensate farmers for the income loss due to, e.g., mowing

378 being delayed after the third week of June.

379 This considered, agri-environmental schemes specifically thought to halt the decline of grassland birds and

380 to maintain grassland surface resulted in mixed effects (Broyer et al. 2014; Batáry et al. 2015; Darnhofer *et*

381 *al.* 2017); thus complementary strategies are desirable to address this conservation issue from a different

382 perspective.

383 In the U.S.A., sustainable market-based conservation models were suggested as the best opportunity to

384 conserve grassland bird populations. These models assume that consumers will pay more for a product if its

385 sustainability, healthiness, and quality are evident (Perlut 2014). In an Alpine perspective, these models

386 could take the form of self-sustaining dairy micro-economies, based on the promotion of the local specific

387 characteristics ("buy local"), which can enhance product quality, while promoting the maintenance of

388 traditional landscapes, which in turn favour tourism and other recreational activities (Lindemann-Matthies

389 et al. 2010; Assandri et al. 2018), and, hopefully, grassland biodiversity. These initiatives should be

390 recognised, sustained (by e.g. dedicated measures in the framework of Rural Development Programmes)

391 and controlled by public authorities and guaranteed by means of dedicated quality brand and certification

392 for the producers, highlighting the support to biodiversity and mountain traditional agriculture given by a

393 product, which in turn justifies its higher cost.

394 Meadow management, although mediated by the topographic, landscape and spatial context, played a

395 fundamental role in allowing meadow specialists to persist in grassland landscapes, thus conservation

396 strategies for grassland should necessarily include well-focused management prescriptions and should be

397 shared and discussed with farmers and other stakeholders in order to develop sustainable and effective

398 solutions.

399

400 **Authors' contributions**

401 GA, MB, and PP conceived the idea; GA carried out fieldwork and led the analyses, helped by MB; PP

402 acquired funding; GA and MB wrote a first draft of the paper; GB supervised the research development; all

403 authors contributed critically to the drafts.

404

405 **Data accessibility**

406 Data available from the Figshare repository http://dx.doi.org/ 10.6084/m9.figshare.7296905 (Assandri et al.

407 2019).

408

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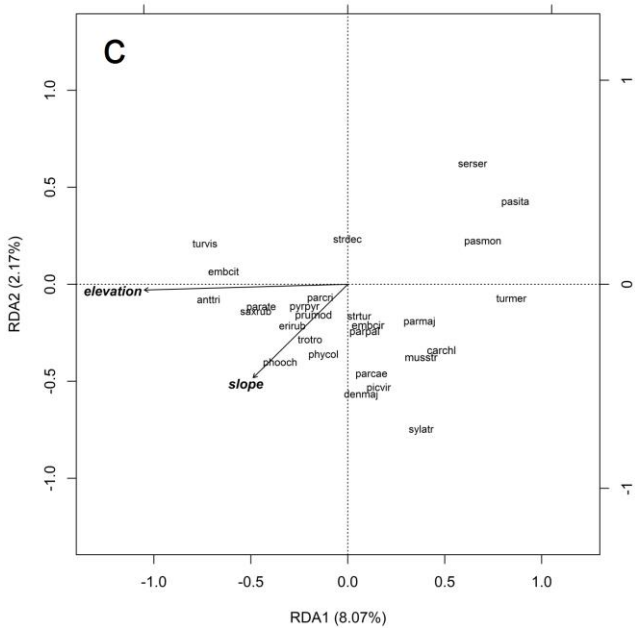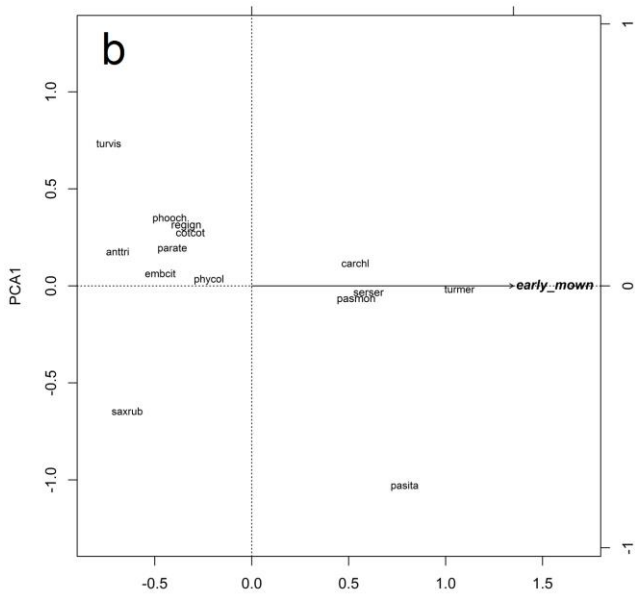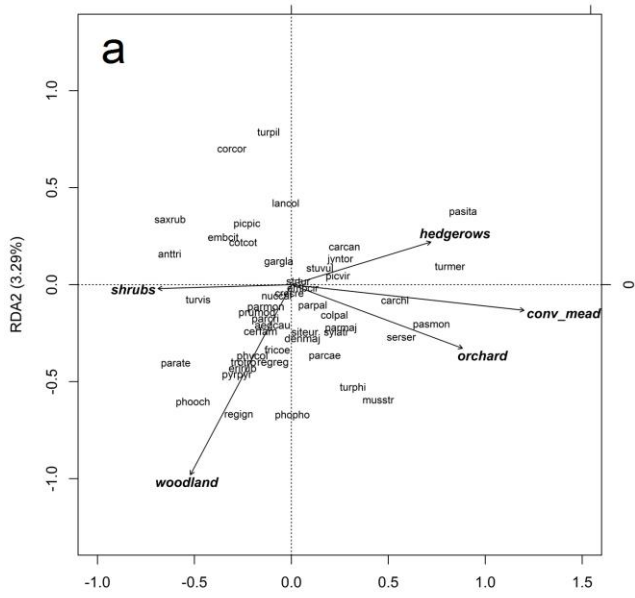
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643

Fig. 1. Study area. (a) Location of the study area in the Alpine region (Trentino in orange). (b) The 63 landscape units investigated are shown in black and hay-meadow cover is in green. (c) Detail on one landscape unit showing the field mapping of the landscape and management variables. Base maps: Natural Earth; Ortofoto 2011 ©AGEA – Agenzia per le Erogazioni in Agricoltura, Roma.
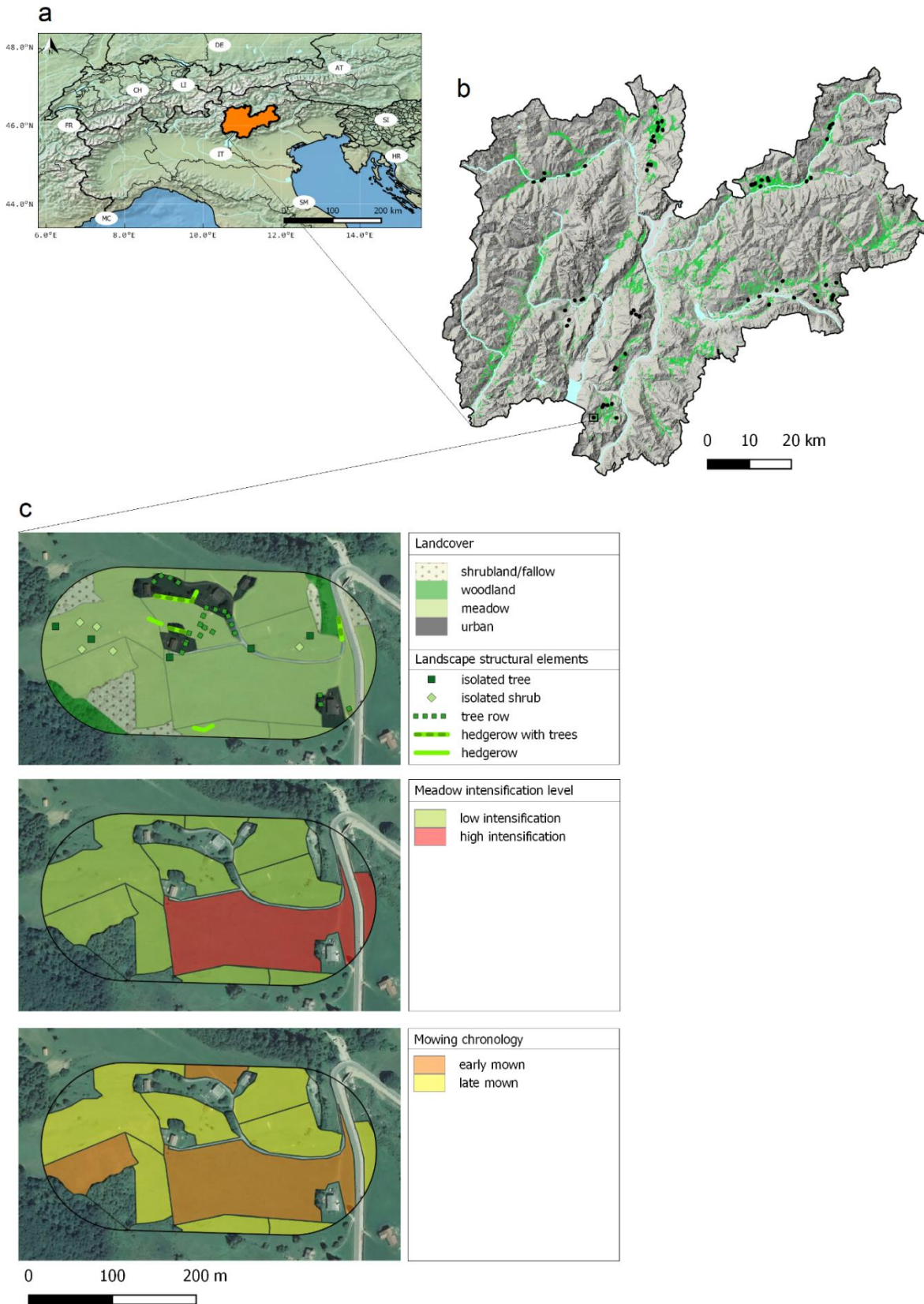
651 **Fig. 2.** Distance biplots of the effects of environmental predictors (a. landscape; b. management; c.
652 topographic) on avian community composition according to parsimonious RDA analysis. Selected variables
653 are represented by arrows. Species are abbreviated with 6 letters (initials of genus and species; see Table
654 S2). Only species with goodness-of-fit > 0.10 are shown. Angles between species and predictors reflect their
655 correlation (angle<90°: positive correlation; angle>90° negative correlation; angle=90°: no correlation).
656 Values in parentheses give the percentage of total variance explained by each canonical axis. N=63.

a

RDA2 (3.29%)

turpil
corcor
lancol
saxrub
picpic
embcit
cotcot
anttri
carcan
gargla
jyntor
stuvul
picvir
hedgerows
pasita
turmer
shrubs
turvis
nucste
parpal
carchl
colpal
conv_mead
parmon
parcri
siteur
parmaj
pasmon
cenfam
phycol
emibr
fricoe
denmaj
sylatr
serser
orchard
parate
pyrpyr
regreg
parcae
turphi
musstr
phooch
regign
phopho
woodland

b

PCA1

turvis
phooch
regign
cotcot
anttri
parate
carchl
early_mown
embcit
phycol
serser
pasmon
turmer
saxrub
pasita

c

RDA2 (2.17%)

serser
pasita
turvis
strdec
pasmon
embcit
elevation
anttri
parate
parcri
turmer
saxrub
pyrpyr
prumod
strtur
embcir
parmaj
erirub
parpal
trotro
phycol
carchl
musstr
slope
phooch
parcae
picvir
denmaj
sylatr

RDA1 (8.07%)

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Fig. 3. Graphical representation of the effect of environmental predictor on species richness (a-f) and number of meadow specialists (g-i) as predicted by averaged models. Only variables for which confidence intervals of estimates did not include zero are shown. Other predictors included in the models are kept constant at their mean value. Variables in subfigures c and e were log+1 transformed for the analyses (untransformed values are displayed to facilitate understanding). 95% C.I. of the mean are shown in grey. N=63.
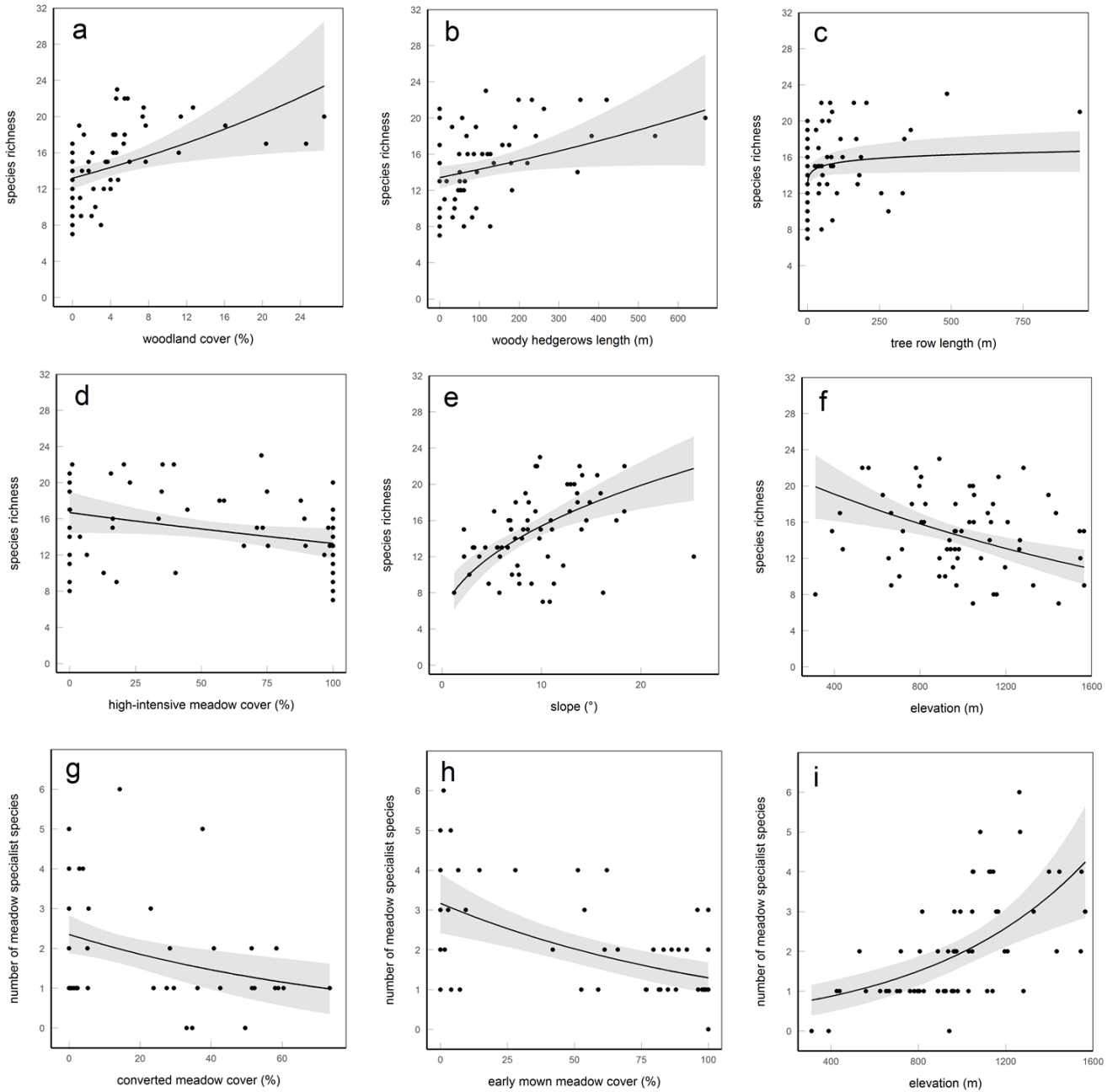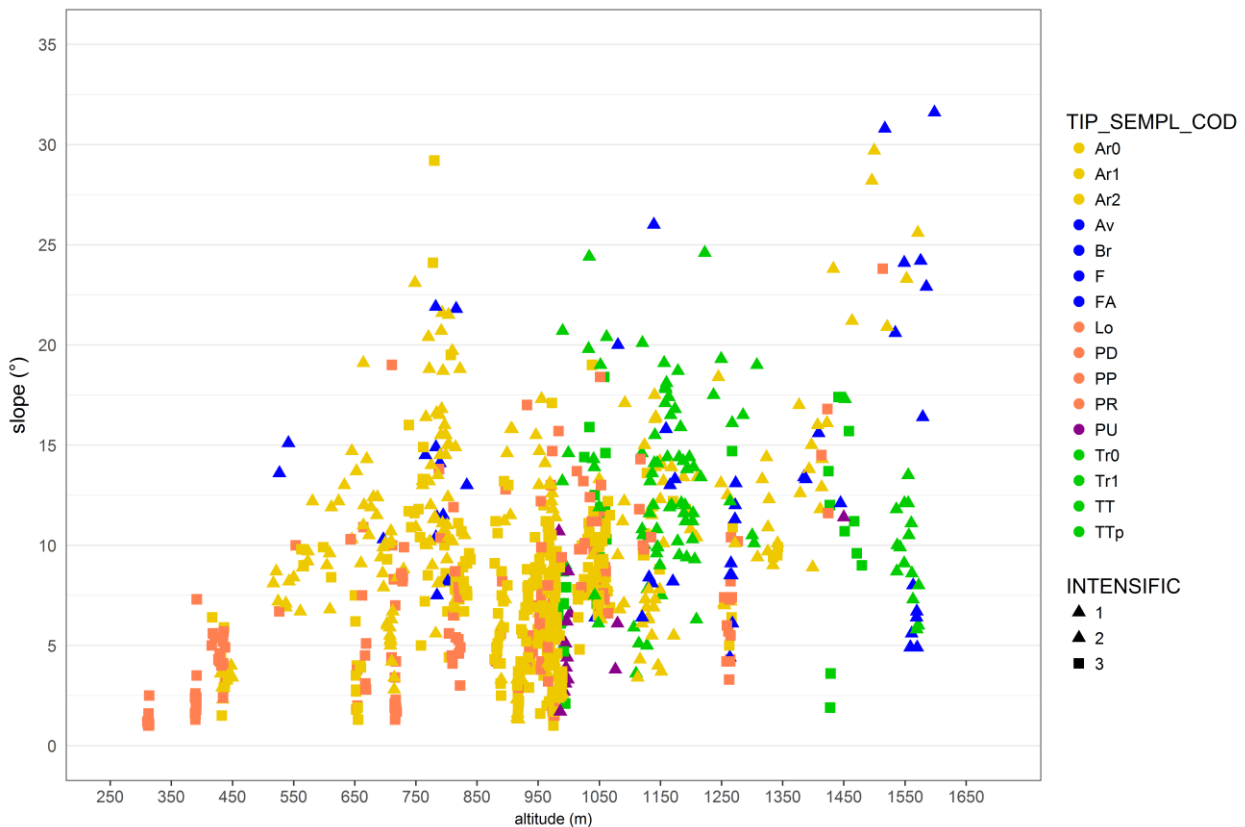
668 **Fig. 4.** Venn diagrams for variation partitioning showing the percentage contribution of landscape,
669 management, topography, and spatial components in explaining a) community composition, b) species
670 richness, and c) the number of meadow specialists in the 63 landscape units. Circle areas is roughly
671 proportional to the percentage of variation explained. Areas within overlapping circles indicate
672 approximate percent variation shared by different components. Numbers in brackets inside component
673 labels refer to the percentage of overall variation explained by a component (including shared quotas).
674 Values < 0 not shown.
675
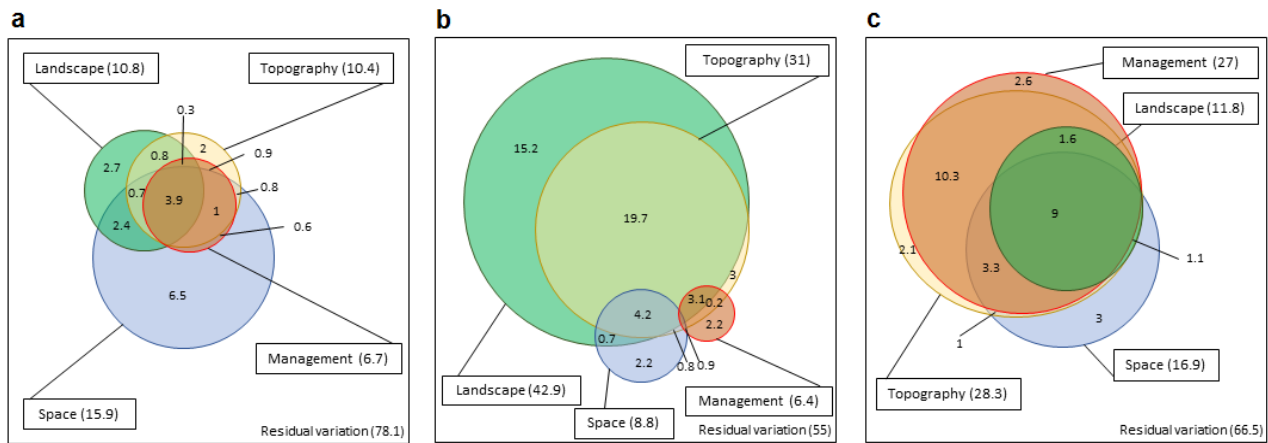
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679 **Fig. 5.** Joint effect of slope and elevation on meadow typology and intensification level. Each point refers to
680 a meadow parcel (N=882). The 16 meadow typologies found in the study area are grouped (by colours) in 5
681 categories: yellow: *Arrhenateretum* meadows; blue: meadow rich of species typical of non-fertilized soils
682 (dominated by *Bromus*, *Festuca*, *Avenula*, *Agrostis*); orange: highly fertilized, disturbed, and reseeded
683 meadows; violet: wet meadows; green: *Trisetetum* meadows. The meadow intensification level is shown by
684 symbols (see SI for further details).
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