

1 **Could plant diversity metrics explain climate-driven vegetation changes on**
2 **mountain summits of the GLORIA network?**

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30 **Abstract**

31 High-elevation habitats host a large number of plant species and are characterized by high biodiversity. The vegetation
32 dynamics in these cold adapted ecosystems are difficult to predict, being affected by global warming, especially in the
33 last decades. With the aim to promote a better understanding of climate-driven changes of alpine vegetation, we
34 investigated the variation in species richness, α -diversity, β -diversity, and total cover of plant functional types over a
35 time lapse of 15 years, relying on multiple re-surveys of mountain summit vegetation in 2001, 2008 and 2015. The
36 study area, included in the long term global observation network GLORIA, was at the boundary between temperate and
37 Mediterranean mountains of S-Europe (northern Apennines, Italy). We identified a trend of loss in biodiversity and
38 signals of biotic homogenization using multiple diversity metrics, despite the overall species richness increment
39 observed in the study area. Cold-adapted and rare species declined while dominant species like shrubs and graminoids
40 increased. Our results highlights that long-term vegetation monitoring activities paired with multiple measures of
41 diversity are required to properly assess biodiversity and to obtain useful indications for future conservation activities in
42 alpine environments. The methods here presented could be applied in all GLORIA sites to quantify biodiversity changes
43 over time, obtaining comparable results for biodiversity monitoring in high-elevation habitats from all over the world.

44

45 **Keywords**

46 GLORIA, biodiversity loss, vegetation dynamics, biotic homogenization, plant functional types, Hill numbers

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60 **Introduction**

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62 Mountains are regarded as biodiversity hotspots (e.g. Körner et al. 2017), and are experiencing higher rates of climate
63 change than other regions of the world (IPCC 2014; Mountain Research Initiative EDW Working Group 2015). In
64 particular, alpine ecosystems are especially sensitive to global warming, as biological and chemical processes are more
65 temperature-sensitive in cold environments (Kirschbaum 1995; Körner 2005).

66 As reported by several studies (Grabherr et al. 1994; Walther et al. 2005; Rixen et al. 2014), the observed
67 climate warming directly affected alpine plant communities, causing changes in distribution, physiology and phenology
68 of a wide range of taxa, and promoting both species richness increase and cryophilic species extinction (e.g. Sala et al.
69 2000; Parmesan 2006; Bellard et al. 2012; Pauli et al. 2014; Steinbauer et al. 2018). These concerns have been
70 confirmed by recent studies, indicating that variation in temperature and precipitation along with extreme climatic
71 events (e.g. heat waves) represent one of the major **threats** for biodiversity of the alpine ecosystems (Abeli et al. 2012a;
72 Emakovich et al. 2014; Orsenigo et al. 2014).

73 Increased temperatures result in direct abiotic pressure for resident alpine species by altering their physiology
74 and phenology (e.g. Fu et al. 2015; Carbognani et al. 2016), and, indirectly, the rise of temperatures is opening “new
75 ground” for invaders from lower elevations. This process, known as upward migration, is causing an increase in species
76 richness in alpine and other cold environments (Grabherr et al. 1994; Walther et al. 2005, Steinbauer et al. 2018), with
77 predicted local extinctions of those plants that already live close to the upper margins of elevational gradients
78 (Theurillat and Guisan 2001; Colwell et al. 2008; Pauli et al. 2012).

79 Resurvey of historical sites and long-term monitoring studies provided evidences of the occurring vegetation
80 changes at local and regional scale, reporting upward shifts of plant species often associated with an increase in species
81 richness (e.g. Parolo and Rossi 2008; Lenoir et al. 2008; Steinbauer et al. 2018). According to the data collected in 2001
82 and 2008 following the sampling design proposed by the long-term monitoring project GLORIA (Global Observation
83 Research Initiative in Alpine environments; www.gloria.ac.at) on 66 different mountain peaks displaced all over
84 Europe, species gains seemed to be particularly evident in boreal and temperate European mountains, whereas most of
85 the Mediterranean summits had, in contrast, species losses (Pauli et al. 2012). Both gains and losses were more
86 pronounced on lower summits, suggesting that changes in species richness are mostly driven by leading edge
87 expansions in boreal and temperate mountains and by rear edge retractions in the Mediterranean ones (Pauli et al.
88 2012). Considering that rear edges retract as much as leading edges expand (Rumpf et al. 2019), these regional-specific
89 patterns suggest that different processes could drive the vegetation dynamics in different mountain regions. However, it

90 is not known if these trends will remain constant over longer time-scales, in particular in mountain ranges straddling
91 different climatic regimes.

92 **Dullinger et al. (2012)** introduced a model indicating that the opposing effects of delayed local population
93 extinctions and lagged migration rates will result in less severe twenty-first-century range reductions of alpine plants
94 than expected from static, niche-based model predictions, and provided insights into the transient dynamics that are
95 likely to dominate the range responses of plants to climate warming over the next century. In addition, Alexander et al.
96 (2018) developed a process-based and dynamic community model to understand how dispersal lags and competitive
97 interactions could influence expectations for range shifts and rates of community turnover along elevation under climate
98 change.

99 Up to now, many studies on the impact of changing climate on plant diversity in mountain summits were
100 focused on species richness (see Stanisci et al. 2005; Walther et al. 2005; Holzinger et al. 2008; Pauli et al. 2012; Wipf
101 et al. 2013; Steinbauer et al. 2018). Nonetheless, species richness alone is a weak indicator as it does not take into
102 account the abundance of species, which is another fundamental component of the biological diversity of plant
103 assemblages. Also within the GLORIA network, a comprehensive review on diversity changes capable of going beyond
104 species richness is still missing. Diversity is a complex multidimensional property of a vegetation stand and many
105 indices have been developed to quantify α -diversity. However, methods integrating the sensitivity of diversity estimates
106 to species abundances, such as the diversity profiles based on Hill numbers (Hill 1973), remain currently underused in
107 plant ecological research. Hill numbers are a parametric family of diversity indices, and have several advantages over
108 other single diversity metrics (Chao et al. 2014). They are expressed by a clear unit of measure (*effective number of*
109 *species*), and can depict simultaneously the values of several diversity indices with different sensitivity to species
110 richness and evenness. As such, they could provide new insight for comparing and interpreting vegetation diversity.

111 Among the plant functional types, forbs are the major contributor to species diversity in alpine and arctic
112 communities (Körner 1995). However, shrubs and graminoids are commonly regarded as the most responsive groups to
113 warming in cold ecosystems (e.g. Bigelow et al. 2003, Elmendorf et al. 2012; Stanisci et al. 2014). Nevertheless, the
114 influence of climate change on plant functional types **seems region-specific** (Elmendorf et al. 2012). For instance,
115 warming trends lead to shrubification in tundra ecosystems (e.g. **Bigelow et al. 2003; Myers-Smith et al. 2011; Myers-**
116 **Smith et al. 2015**) with a shift from herbaceous to woody communities (e.g. Walker et al. 2006). Differently, in less
117 extreme habitats, shrubs can be outcompeted by graminoids (Thompson et al. 1995), probably due to the greater
118 response of graminoids to soil nutrient availability (Dormann and Woodin **2002**). Nitrogen soil concentration in natural
119 environmental is related to temperature (Rustad et al. 2001), it is constantly increasing in alpine and subalpine

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120 environments (Rogora et al. 2006), and it has already been recognized as a driver for species composition changes,
121 often associated with increments in nutriphilous species abundance and biodiversity loss (Bobbink et al. 2010).
122 Consequently, our knowledge of warming-induced vegetation dynamics in mountain areas at the intersection of
123 different climatic regimes and their consequences for plant diversity are still incomplete.

124 To fully describe the processes that are taking place in plant communities, scientific investigations regarding the
125 influence of global change on alpine ecosystems should include variation of both α - and β -diversity, since the increase
126 of α -diversity on mountain summits can be accompanied by decreasing β -diversity, leading to a vegetation
127 homogenization over time (Jurasinski and Kreyling 2007; Britton et al. 2009; Ross et al. 2012).

128 In this context, we propose here a multi-faceted approach to analyse the changes in vascular plant diversity
129 (using multiple α - and β -diversity measures) and cover of the main functional types (graminoids, forbs, shrubs)
130 recorded through multiple re-surveys. The proposed methodology could be used in other long-term vegetation
131 monitoring studies, especially within the global observation network GLORIA.

132 The methods proposed here have been used to assess the diversity changes occurred in the IT-NAP GLORIA Target
133 Region corresponding to the northern Apennines, a mountain system lying at the boundary between northern and central
134 Italy. The spatial position of the northern Apennines is important from a phytogeographical point of view, as it stands at
135 the southernmost boundary of the temperate region and at the forefront of the Mediterranean region. In 2008, this
136 Target Region shared the same general increasing trend in species richness recorded in other boreal and temperate
137 Target Regions within the European GLORIA network. In contrast, the Central Apennines Target Region (about 350
138 km southeast) and the other Mediterranean Regions showed a different trend of species richness remaining stable or
139 decreasing (Pauli et al. 2012; Stanisci et al. 2014).

140 In particular, applying the methods presented in this paper, we aimed to address the following questions: (1)
141 Did changes in species richness from 2001 to 2015 confirm the magnitude and the “temperate-boreal” trend of changes
142 observed in the period 2001-2008? (2) Did α -diversity vary between 2001 and 2015 and, if so, were the changes
143 consistent among different summits? (3) Did the cover of plant functional types vary over time? (4) Was the shift in
144 species composition leading to a homogenization and biodiversity loss of the mountain summits?

145

146 **Materials and Methods**

147

148 **Study area**

149 In 2001, four summits were selected in the GLORIA Target Region “IT-NAP” (Figure 1a), following the multi-summit
150 approach proposed by Pauli et al. (2015). The four summits, Mt. Casarola, Alpe di Mommio, Cima di Pian Cavallaro
151 and Cima di Foce a Giovo, will be hereafter labelled respectively as CAS, MOM, PCA and FOG. Long term climatic
152 features of the four summits, along with geographical information, are reported in Table S1 in Supplementary Material.
153 The four study sites, located on the top of the four summits, lay above the treeline, which stands at around 1700m a.s.l.
154 (Tomaselli and Rossi, 1994), and are disposed along an elevational gradient that ranges from 1722m to 1978m a.s.l.
155 The summits were selected **in such a way** to represent the vegetation patterns occurring in the summit areas of the
156 Target Region according to the requirements reported in the GLORIA field manual (Pauli et al. 2015). Moreover, all the
157 sampled areas were in recent past and are also presently **completely** free from grazing or other anthropogenic impacts.
158 Geologically, the summits consist of thick-bedded turbiditic sandstones and marlstones corresponding to the Macigno
159 formation (Dallan Nardi and Nardi 1974).

160 Climatic data recorded by the Meteorological Observatory of Mt. Cimone, located about 1 km from PCA
161 summit, indicates a significant warming trend of summer air temperatures for the period 1970-2017 (as shown by the
162 linear regression model fitted to the summer temperature data against years; $R^2 = 0.613$, $t = 8.539$, $F_{1-46} = 72.92$, p
163 **<0.001**) with an average increment of 0.094 K year⁻¹ (Figure 1b).

164 In the study area, summit vegetation consists of dwarf shrub heaths dominated by *Vaccinium* species, mostly
165 occurring on the northern slopes, and of secondary grasslands dominated by *Brachypodium genuense* or by *Nardus*
166 *stricta*, prevailing over the southern slope (Tomaselli 1994; Ferrari and Piccoli 1997).

167

168 **Field work**

169 Three vegetation surveys were carried out in the summer of 2001, 2008 and 2015, following the sampling design of
170 GLORIA (Pauli et al. 2015; <https://gloria.ac.at/downloads/manual>). At each summit of the Target Region the area
171 investigated was defined as a polygon with four corners fixed at each cardinal direction, placed 10 vertical metres below
172 the highest summit point (HSP) of the summit, and delimited by a perimeter called 10m contour line. Each summit’s
173 polygon was then divided into four sectors, following cardinal directions facing N, S, E and W. Each sector was further
174 subdivided into 0-5 and 5-10-m zones below the HSP, by a line called 5m contour line, obtaining eight summit area
175 sections (referred as “SAS” hereafter) for each summit. The group of four SAS lying above the 5m contour line and
176 group of four SAS lying below it will be referred hereafter as “upper belt” and “lower belt”. In total, 32 SAS were
177 included in the Target Region. For each SAS, a complete list of vascular plant species was recorded, along with a
178 visually estimated coverage expressed by the following classes of abundance: r! = very rare, r = rare, s = scattered, c =
179 common, d = dominant, following GLORIA 2015 guidelines (Pauli et al. 2015) (see Table S2 in Supplementary

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180 materials). Species nomenclature follows Tutin et al. (1964-1993). Species richness values recorded during the three
181 surveys for the Target Region, summit and belt levels are reported in Table S3.

182 The areas of each SAS were estimated by trigonometry on the basis of angles and lengths measured in the field (see
183 Table S4 in supplementary materials for SAS polygon area).

184

185 Data processing

186 First, to avoid pseudo-absence or pseudo-presence signals, following the indications of Vittoz et al. (2010a) and Pauli et
187 al. (2012), all the species that were found only in one SAS and just once among the three surveys were excluded from
188 the original species list (14 species, see Table S5 in Supplementary Material). Potential mistakes in species
189 identifications, or changes in nomenclature, were corrected by a scrupulous check of possible synonymies and by
190 aggregating the records of pairs of species with frequent confusions into the same taxon. Second, the five qualitative
191 abundance classes defined by the GLORIA protocol were converted to cover percentage approximations, on the basis of
192 the GLORIA classes descriptors and their percentage breakpoints. The values used for the conversion were assigned
193 using the cover percentage approximations (e.g. the mid values of the Braun-Blanquet cover/abundance scale
194 transformation) used for the numerical treatment of phytosociological data (Braun-Blanquet 1964; Van der Maarel
195 1979). Converted values are reported in Table S2 in Supplementary Materials. Finally, to consider the differences in
196 SAS areas due to summit micro-topography, species cover was standardized by means of a correction factor determined
197 by the ratio between the SAS area where a species occurred and the cumulative SAS area.

198

199 Species richness

200 To detect if ~~a significant variation in~~ species richness ~~varied occurred~~ through time at the Target Region level, taking
201 into account the species richness – area relationship, we firstly calculated the residuals of a linear mixed-effect
202 (LMM) fitted on the 32 SAS species richness values in 2001, 2008 and 2015, with “SAS area” as fixed effect and with
203 “SAS ID” (identification code of each SAS, ranging from 1 to 32) and “Summit ID” (identification code of each
204 summit) as nested random effects (to consider the repeated measure structure of the dataset, and the spatial relationship
205 of the SAS), and with “SAS area” as fixed effect. Subsequently, to eliminate the effect of SAS area on species richness,
206 we fitted a linear model (LM) on the obtained LMM residuals, similarly to the approach ~~already~~ proposed by Vittoz et
207 al. (2010b). The LM was performed including “Time” (3-level factor: 2001, 2008 and 2015), “Summit” (4-level factor:
208 CAS, MOM, PCA_FOG), and “Belt” (2-level factor: upper and lower belt) as fixed effect, to test if the species richness
209 changed over time in the Target Region, and if there were ~~significant~~ differences among the summits and belts. The

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210 interactions between the factors (~~Time × Summit, Time × Belt, Summit × Belt and~~ Time × Summit × Belt) were also
 211 included in the model, to test if species richness varied disproportionately within different Target Region areas. We
 212 choose this approach as the different abiotic conditions and vegetation of the 4 summits and belts could have had
 213 different effects on the species richness changes (e.g. we did expect higher rates of new species recruitment in the lower
 214 summits and ~~in the lower~~ SAS belts, as a consequence of upward migration ~~process~~). The same approach was then
 215 repeated at summit level (8 SAS for each summit), ~~firstly~~ calculating with a LMM the ~~residuals of the~~ SAS species
 216 richness residuals on summits' SAS areas (with "SAS ID" as a random factor), and then by fitting a LM on the obtained
 217 residuals with "Time", ~~and "Belt"~~ ~~and as fixed factors, along with their interaction (Time × Belt) term as fixed factors.~~
 218 LMM and LM were performed by means of the packages lme4 (Bates et al. 2015), ~~and stats V3.6.0 of the statistical~~
 219 ~~suite R (R Core Team 2017)~~. F-statistics and *p*-values were obtained using the function "Anova" by means of "Car"
 220 package (Fox et al. 2012), while Post Hoc tests were performed with the "TukeyHSD" function of the package stats
 221 V3.6.0 of the statistical suite R (R Core Team 2017).

222 Finally, plant species that disappeared entirely (i.e. as species sampled in 2001, and then not found again ~~in~~
 223 ~~2008 or 2015~~) from the Target Region, or ~~that disappeared~~ in at least one summit, were included in two lists that could
 224 be useful for plant conservation activities.

226 Assessment of α -diversity

227 To assess α -diversity changes, we calculated the diversity profiles in 2001 and 2015 for the whole Target Region, as
 228 well as for the four summits separately. The diversity profiles were obtained by calculating the Hill Numbers (qD sensu
 229 Hill 1973) of each assemblage, using the SAS standardized cover of species. Hill numbers have been recently
 230 reintroduced into the analyses of ecological data by Jost (2006), as they have the possibility to describe different aspects
 231 of diversity of one site at once, allowing to rank different assemblages from high to low diversity (Tóthmérész 1998;
 232 Chao et al. 2014). Hill numbers are a mathematically unified family of diversity indices expressed in units of *effective*
 233 *numbers of species* (i.e. the number of equally abundant species that would be needed to give the same value of a
 234 diversity measure; Hill, 1973), calculated from the frequencies of each component species, and a scale parameter,
 235 defined as:

$$236 \quad {}^qD = \left(\sum_{i=1}^S p_i^q \right)^{1/(1-q)}$$

237 where p_i is the relative frequency of the i th species, S is the total number of species and q is the scale parameter.

238 This last parameter determines the index sensitivity to species relative abundance: at small values of q , Hill numbers are
 239 less influenced by species abundance (being, ~~in fact, the~~ qD values at $q = 0$ equal to species richness), while as q

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240 increases, their values are increasingly influenced by the abundance of species. The H' values of an assemblage can be
241 plotted against the scale parameter to obtain a decreasing function, ~~referred in literature as called~~ diversity profile (Patil
242 and Taillie 1979, 1982). A diversity profile is ~~thus~~ composed by several diversity indices that have different sensitivity
243 to the presence of rare and abundant species, and using them as a tool to evaluate assemblages' diversity allows us to
244 overcome the limits of a single numerical definition. The value of a single index, ~~in fact~~, cannot adequately summarize
245 biodiversity and community structure, as it is well-known that different diversity indices may rank sets of communities
246 in different ways (Pielou 1975), depending on how it is influenced by ~~species abundance different aspects of the~~
247 ~~partition of abundance among species~~ (Hill 1973). For this reason, diversity profiles can be used ~~also~~ to compare the
248 diversity of different sites ~~more~~ consistently (or, as in this study, to compare diversity of the same ~~assemblage~~ SAS in
249 different years). If two diversity profiles do not intersect each other, the assemblage with the diversity profile that runs
250 above the other one is unequivocally more diverse. However, if the profiles cross each other, the two assemblages
251 cannot be unequivocally ordered, as one assemblage is more diverse for the rare species while another is more diverse
252 for the frequent species (Tóthmérész, 1998). In order to ~~overcome this limitation, and~~ rank diversity profiles even when
253 they cross each other, we calculated the underlying area of the diversity profile function, that will be referred as
254 "surface area" hereafter, as suggested by Di Battista et al. (2017), using the trapezoid method (Atkinson 2008). ~~The~~
255 ~~surface area~~ is directly proportional to both ~~the~~ species evenness and ~~the~~ species richness, and it does not attribute
256 more importance to one or another aspect, ~~as because~~ it is calculated while considering the entire domain. From an
257 ecological point of view, a higher value of the surface area denotes a greater community biodiversity. ~~Moreover, to~~
258 gather further information for interpreting ~~and ranking~~ diversity variation, we calculated the following indices: (1)
259 Shannon index (Shannon 1948), (2) Sheldon evenness (Sheldon 1969), and (3) Dominance index, expressed as $1 -$
260 Simpson index (Simpson 1949). Diversity profiles and ~~the above mentioned~~ indices were calculated and plotted with
261 the software PAST 3.21 (Hammer et al. 2001).

262

263 **Cover changes of plant functional types**

264 To analyse vegetation dynamics over time, we grouped the species in the following plant functional types classified by
265 growth forms: forbs, graminoids (including grasses, sedges and rushes), and shrubs. This simplified classification is
266 currently adopted in the studies concerning predictions of vegetation responses to global change in Arctic and boreal
267 ecosystems (see Chapin et al. 1996; Wullschleger et al. 2014; Thomas et al. 2018). Ferns were excluded from this
268 analysis due to their low abundance on the summits investigated. For each functional type, we calculated the sum of
269 standardized cover of all species occurring in each SAS at the Target Region level (32 SAS included). To assess if
270 significant differences in the cover of functional types occurred between 2001 and 2015, we fitted LMMs to log-

271 transformed data by means of the lme4 package (Bates et al. 2015), with “Time” (2-level factor: 2001 and 2015) as
272 fixed effect and “Summit ID” and “SAS ID” as nested random effect, in order to take into account both spatial and
273 temporal relationship within the data. Finally, for the functional types showing significant variation over time at the
274 Target Region level, further LMMs were performed at the summit level (including 8 SAS each included for each
275 model). In these models the log-transformed standardized cover of functional types was the response variable, whereas
276 “Time” and “SAS ID” were considered as the fixed and random effect, respectively. F-statistics and *p*-values were
277 obtained using the function “Anova” by means of “Car” package (Fox et al. 2012).

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279 Assessment of β -diversity

280 To evaluate if a process of biotic homogenization is currently undergoing ongoing in the study area, we calculated β -
281 diversity for 2001 and 2015, as suggested in several papers other researches (Jurasinski and Kreyling 2007; Britton et
282 al. 2009; Ross et al. 2012). First, we used the dissimilarity among all the SAS of the Target Region as a descriptor of the
283 spatial patterns of diversity. The Sørensen multiple-site dissimilarity measure described in Baselga (2010) was thus
284 calculated for the Target Region level (32 SAS included) for 2001 and 2015. This measure, calculated based on
285 presence/absence data, is dependent on the proportion of species shared between different assemblages, and it is related
286 to species turnover, nestedness and heterogeneity (Koleff et al. 2003; Baselga 2010). It has several advantages
287 compared to other dissimilarity metrics: (1) This measure can be it can be partitioned in the two separate indices
288 “multiple-site β_{SM} ” and “multiple-site β_{SNE} ” (accounting for the dissimilarity among sites due to turnover and
289 nestedness, respectively), (2) it shows a linear relationship with the original β diversity measure firstly proposed by
290 Whittaker in 1972 (Diserud and Ødegaard 2007); and (3) it is specifically designed to quantify the overall heterogeneity
291 of a pool of sites (Baselga 2013). In particular, we focused on the changes of the “multiple-site β_{SM} ”, the multiple-site
292 version of the Simpson β -diversity index (*sensu* Baselga 2010), which is regarded as one of the presence-absence β -
293 diversity indices that perform better under different conditions and is less biased by sample size and nestedness of the
294 quadrant than other β diversity indices (Koleff 2003; Barwell 2015). In addition, we calculated the “pairwise β_{sim} ” index
295 (*sensu* Baselga 2010) between each SAS and all the others within the same period, testing differences between 2001 and
296 2015 with the Wilcoxon test for paired data. Finally, these analyses were repeated for the upper and lower belts
297 (including 16 SAS each) to understand if there are potential differences in terms of β diversity due to upward
298 migration and species extinction. The β -diversity analyses were performed by means of the “Betapart” (Baselga and
299 Orme 2012) and the “PairedData” packages (Champely 2018) of the statistical suite R (R Core Team 2017).

300

301 **Results**

302

303 **Species richness**

304 In the three surveys, a total of 169 vascular plant species were recorded at least once in the study area. The number of
305 vascular plants in the Target Region varied over time (Table S3), as testified by the significant effect of the factor Time
306 on species richness (Table 1a). In particular, the number of species recorded in each SAS increased significantly from
307 2001 to 2008 (mean difference \pm 95% CI: $+4.1 \pm 1.83$, $p < 0.001$) and to 2015 ($+4.3 \pm 1.83$, $p < 0.001$), whereas the
308 species richness recorded during the two later surveys was not significantly different ($+0.15 \pm 1.83$, $p = 0.977$). The
309 number of colonizers decreased from 11 species between 2001 and 2008 to 3 species between 2008 and 2015, whereas
310 disappearances increased from 1 to 8.

311 Species richness variation over time was not consistent among the four summits (significant Time \times Summit
312 interaction term in Table 1a). In particular, the effect of “Time” on Species Richness was significant for both the two
313 lower summits PCA and FOG (mean difference: $+12.12 \pm 4.06$, $p < 0.001$, and $+5.12 \pm 3.63$, $p = 0.005$, respectively),
314 whereas it had no significant effect in ~~the two higher summits~~: CAS ($+0.25 \pm 4.98$, $p = 0.99$) and in MOM (-0.37 ± 3.19 ,
315 $p = 0.95$), ~~the two higher summits~~ (see Table 1b). The colonization prevailed over disappearance between 2001 and
316 2015 in FOG and PCA with a net increase in total number of species recorded ~~of (6)~~ on both summits, while
317 disappearance prevailed on colonization in MOM, with a net decrement of 6 species. At the level of upper and lower
318 belt, species richness increased (with significant or marginally significant change) over time in three belts: in the two
319 lower belts of FOG ($+6.25 \pm 6.38$, $p = 0.057$) and PCA ($+15.75 \pm 7.16$, $p < 0.001$), and in the upper belt of PCA ($+8.5$
320 ± 7.16 , $p = 0.01$), the second lowest summit (as highlighted by the Tukey HSD post-hoc test for “Time \times Belt”
321 interaction, not reported). The highest increments were recorded on the lower belts of lower summits (+17.5% and
322 +15.9% species richness increment in PCA and FOG, respectively).

323 With regard to the species that disappeared from the study area, overall 9 species disappeared from the Target Region,
324 while 19 species disappeared from at least one summit. These species are reported in Table S6 and S7 in Supplementary
325 Materials respectively.

326

327 **Assessment of α -diversity**

328 The Shannon index and the Sheldon evenness values observed in 2015 in the Target Region decreased compared to
329 2001, whereas the value of the Dominance index increased (Table 2a). The diversity profiles of the Target Region were
330 very similar in 2001 and 2015 (Figure 2) and the pronounced decline of the curves indicated an uneven distribution of

331 relative abundances among plant species, in particular in 2015. However, the two profiles did intersect, thus an
332 immediate ranking of diversity level could not be performed. On the other hand, focussing on the variation of the
333 Surface area underlying the diversity profiles in 2001 and 2015, it could be stated that, despite the species richness
334 increment in 2015, the overall diversity of the Target Region decreased (Figure 2 and Table 2a).

335 Two different patterns of diversity profiles were observed within the four summits investigated (Figure 3): (1)
336 an overall diversity loss was detected in the two higher summits (CAS and MOM) along with stagnating species
337 richness and (2) diversity variations (both increments and decrements, depending on the considered summit)
338 accompanied by an increase in species richness were observed in the two lower summits (PCA and FOG). In particular,
339 for the two higher summits the diversity profiles and surface area values in 2015 were lower to the 2001 ones (Figure
340 3a, 3b and Table 2b), indicating diversity loss. Moreover, both Shannon index and Sheldon evenness calculated for
341 these two summits decreased over the years, whereas the Dominance index increased (Table 2b). With respect to the
342 two lower summits, PCA showed higher Hill numbers in 2015 for all values of the scale parameter q (Figure 3c) and
343 consistently, Shannon index, Sheldon evenness and Surface area increased indicating a diversity increment, while the
344 Dominance index decreased (Table 2b). In FOG, despite the increase in species richness over time (Table 1b), diversity
345 decreased in 2015 and thus the profiles crossed each other (Figure 3d). In this summit, the Shannon index remained
346 unchanged, but the **decreasing Sheldon** evenness and Surface area, along with increasing Dominance index, indicated
347 an overall diversity loss.

348

349 **Cover changes of plant functional types**

350 The total cover of graminoids and shrubs increased between 2001 and 2015 at the Target Region level, showing
351 however different patterns among the four summits studied (Table 3a). ~~Conversely~~~~At the contrary~~, forbs did not show
352 any significant variation in space or over time. The magnitude of changes observed for graminoids was higher in
353 comparison with the ones observed for shrubs at the Target Region level (Figure 4). **At the summit level, graminoids**
354 **cover increased significantly on the two higher summits** (Figure 5, Table 3b), but no significant differences were
355 detected on the lower summits. Shrubs and forbs did not show any significant change at the summit level.

356

357 **Assessment of β -diversity**

358 The “multiple-site β_{SIM} ” index values (indicating the turnover component of β -diversity) were considerably higher in
359 comparison with the ones of the “multiple-site β_{SNE} ” (indicating the nestedness component of β -diversity) across the
360 years, with β_{SIM} index values always > 0.8 and β_{SNE} index values always < 0.05 , showing that the β -diversity patterns in

361 the studied area are mainly influenced by the species turnover. At the Target Region level, the “multiple-site β_{SIM} ”
362 index showed a decrease between 2001 and 2015, with a stronger loss of dissimilarity among SAS in the upper belt
363 compared to the lower one (Table 4). The “multiple-site β_{SIM} ” values were higher in the upper belt compared to the
364 lower belt ones both in 2001 and 2015, but a stronger decrease in dissimilarity was found in the upper ones. Similar to
365 the “multiple-site β_{SIM} ”, expressed as an overall dissimilarity among all the 32 SAS, the “pairwise β_{sim} ” index was lower
366 in 2015 compared to 2001 at the Target Region level, whereas the changes at the belt levels were not significant (Figure
367 6).

368

369 Discussion

370 The results of this study showed that, despite the relatively short time span of the investigations, significant changes
371 occurred in the northern Apennines vegetation along elevation above tree-line. The integration of different biodiversity
372 metrics has provided useful data for understanding the processes that have led to a general loss of diversity in these
373 mountain summits. We have hence confirmed that a range of biodiversity metrics can give an accurate picture of change
374 and allows assessment of biodiversity trends to support conservation and management decision making.

375

376 Species richness

377 The number of vascular species increased from 2001 to 2015 in the Target Region as a whole (Table 1a, Table S3).
378 When the data were split into summits and belts, a substantial increase was observed only on the two lower summits
379 (PCA and FOG) and especially in their lower SAS. The asymmetry of the changes was due to the marked prevalence of
380 colonization on loss events at these SAS. This result suggests the possibility that an increase in species richness may be
381 supported by an upward migration. Species upward migration is consistent with expectations based on climate warming
382 scenarios, and it was clearly detected by many recent observational studies and reviews from temperate and boreal
383 mountains across Europe (Pauli et al. 2012; Grytnes et al. 2014; Steinbauer et al. 2018 among others). Taking into
384 account that the four summits selected for the GLORIA Target Region are (and were) free from antropogenetic impacts
385 (such as grazing or trampling), a possible role of climatic drivers for the upward migration can be hypothesized, as also
386 suggested by the occurrence of several understory species of beech woodlands and their fringes (*Dactylorhiza fuchsii*,
387 *Dryopteris filix-mas*, *Hieracium murorum*, *Luzula pedemontana* and *Silene nutans*) and of species of montane
388 xerophytic grasslands (*Knautia purpurea* var. *calabrica* and *Asperula aristata* subsp. *oreophila*) and meadows (*Trollius*
389 *europaeus*), in the 2015 survey, in both PCA and FOG.

390 The lack of significant variation in species richness between 2008 and 2015, along with the increment in
391 species loss detected and the decrement of new species recorded in 2015 with respect to 2008, may indicate that not
392 only the magnitude, but also the temperate-boreal direction of changes observed in the northern Apennine summits
393 between 2001 and 2008 (see Pauli et al. 2012) were not confirmed in 2015. This suggests that a trend to species loss is
394 possibly taking place since 2008 in the GLORIA summits of the northern Apennines. In the future, this could make the
395 vegetation dynamics of the northern Apennine mountain summits more similar to the ones occurring in the Central
396 Apennines, where a “Mediterranean” trend of species losses was already recorded between 2001 and 2008 on the higher
397 summits (Pauli et al. 2012). This may lead to a northward shift of the border between temperate-boreal and
398 Mediterranean summits across the Italian peninsula. Further support for this hypothesis is provided by the variation of
399 species richness at the summit level. In the Target Region investigated, species richness showed no significant variation
400 from 2001 to 2015 on the two higher summits and involved the loss of some of the most cold-adapted species occurring
401 in the Target Region (*Arenaria moehringioides*, *Asplenium trichomanes-ramosum*, *Sagina glabra* and *Viola biflora*), as
402 reported in the lost species list of our study sites (Table S6 and S7).

403 Starting from these two species lists, conservation actions such as seed collection (to be performed following the
404 international standards by ENSCONET 2009) for *ex situ* conservation or plant translocation for *in situ* conservation
405 could be planned in the next future. Successful seeds collection activities have already been carried out on the Alps by
406 the Alpine Seed Conservation and Research Network (Müller et al. 2017) in which the Plant Germplasm Bank of the
407 University of Pavia had an active role. A similar approach could be replicated in the Apennines and other mountain
408 systems, taking into account the vegetation dynamics assessed within the GLORIA Target Regions, in order to collect
409 seeds not only of already endangered species, but also of species that could be threatened in a long-term scenario (e.g.
410 species with a constant decreasing abundance trend detected within a Target Region).

411 The role of global warming as a possible driver of this trend is supported by the climatic data documenting the
412 combination of rising summer temperature and decreasing of summer precipitations in the northern Apennines (Antolini
413 et al. 2017) and by a significant increase of summer temperature recorded at the Meteorological Observatory of Mt.
414 Cimone, close to PCA summit (Figure 1b). Moreover, Abeli et al. (2012a, b) and Orsenigo et al. (2015) already reported
415 how warmer temperatures, heat waves, drought and reduced snow cover persistence can strongly affect both biomass
416 production and reproduction of plant populations in the northern Apennines. However, some caution must be adopted
417 when interpreting the variation of species richness trends over a time span of only seven years (between 2008 and
418 2015), also considering that the disappearance of species observed may either be the final stage of a population decline
419 or the result of an unsuccessful species establishment, which are difficult to disentangle (Grytnes et al. 2014).

420

421 **Assessment of α -diversity**

422 Until about the beginning of this decade, it was predicted that the vegetation in the alpine life zone could resist long
423 enough to climate change, showing a relative inertia in species composition, since slow growth rates, clonal abilities
424 and longevity could delay the disappearance of species from habitats becoming climatically unsuitable (Theurillat and
425 Guisan, 2001; Grabherr, 2003; Körner 2003; Dullinger et al. 2012). However, recent studies have showed that even
426 closed alpine and subalpine communities may be considerably more dynamic than had previously thought, with changes
427 in species richness and diversity over 25-50 years or shorter timescales (Ross et al. 2012; Cannone and Pignatti 2014;
428 Carbognani et al. 2014; Matteodo et al. 2016 and others). When species richness and diversity were simultaneously
429 analysed, declines in diversity indices, alternatively combined with increase or decrease of species richness, were found
430 and differently explained by several authors (Britton et al. 2009; Ross et al. 2012).

431 Diversity profiles provide a comprehensive and concise view of various facets of α -diversity of plant
432 assemblages, and their variation over time reflects the balance among different on-going processes driving the
433 vegetation dynamics, such as upward migration, local extinction, competition and gap filling. Overall, greater
434 vegetation diversity, in term of species richness, was found on the higher summits, where, however, the greatest loss of
435 α -diversity is occurring. These diversity trends are consistent with a positive balance between species gains and losses
436 in lower elevation summits, and a null or negative balance in higher elevation ones, affecting diversity profiles at low q
437 values. In addition, the decrease in time of the profile slopes indicate an overall decrease in evenness among species,
438 due to the combined effects of the appearance of new species (e.g. by upward migration) (Steinbauer et al. 2018; Wipf
439 et al. 2013) and the expansion of abundant species (e.g. by gap filling) (Cannone and Pignatti 2014; Rumpf et al. 2018),
440 with the latter process affecting diversity profiles at high q values, and more markedly in the higher summits.

441 Our results show the peculiarity that different patterns of species richness and diversity changes can occur
442 within the same mountain range, ~~even among summits with small differences in terms of elevation along a relatively~~
443 ~~narrow elevation gradient~~ (of about 250 m or less). This can be explained by the fact that this gradient approximately
444 embraces the whole alpine life zone of the northern Apennines, characterised by a high degree of local microhabitat
445 diversity and heterogeneity in vegetation composition and structure within a relatively narrow surface (see Tomaselli
446 1994, Ferrari and Piccoli 1997; Tomaselli et al. 2000; Petraglia and Tomaselli 2007; Gennai et al. 2014).

447

448 **Cover changes of plant functional types**

449

Commentato [F10]: Q2-8

450 We tried to understand the reason underlying the different patterns of α -diversity observed in the Target Region and in
451 the four summits starting from the consideration that where α -diversity decreased (e.g. in term of diversity profile
452 surface area), dominance regularly increased. Following this rationale, we observed that, among the functional type
453 growth form, only graminoids species experienced a significant increase of their total cover on the two higher summits,
454 where dominance increased the most, while both graminoids and shrubs significantly increased at the Target Region
455 level. Among graminoids, the endemic grass *Festuca riccerii* showed the highest increase in total cover in both CAS
456 and MOM; other graminoids that increased on these summits were *Deschampsia flexuosa*, *Festuca rubra* subsp.
457 *commutata* and *Juncus trifidus* (Figure S1 in Supplementary Material). A significant increase of graminoid species
458 cover in upland heaths and grasslands was already reported by Ross et al. (2012) for the northwest Highlands of
459 Scotland and by Cannone and Pignatti (2014) and Stanisci et al. (2014) for the Italian Alps and central Apennines.
460 Differently, other authors demonstrated a high stability of species cover in siliceous grasslands from the northern Alps
461 (Vittoz et al. 2009; Windmaißer and Reisch 2013; Matteodo et al. 2016). We must consider that the habitat conditions
462 affecting the vegetation on the summits of our Target Region resemble those occurring on the windy ridges of the Alps,
463 from where also Matteodo et al. (2016) reported significant changes in species composition, and of Scottish Highlands.
464 On the other hand, Elmendorf et al. (2012) and Walker et al. (2006) suggested that graminoids could expand also in
465 several Arctic, alpine and tundra sites as a consequence of experimental climate warming. Warmer temperatures can
466 stimulate the mineralization of nitrogen in the soil (e.g. MacDonald et al. 1995; Rustad et al. 2001), increasing the
467 nutrient availability for plant growth. Based on a 3-year experiment of nutrient addition in a summit heathland lying in
468 the northern Apennines, Gerdol et al. (2000) proved that fertilization did not increase the aboveground biomass of
469 ericaceous shrubs but, conversely, increase the one of graminoids. As suggested by Bobbink et al. (2010), due to
470 increasing nitrogen concentration in the soil, vegetation changes have to be expected in natural environments, and as
471 highlighted by Rumpf et al. (2018), thermophilic and nitrophilous species (as the graminoids experiencing abundance
472 increments in our Target Region) are expected to become more abundant. Our results seem to fit with these findings, in
473 particular when referring to the higher summits, where shrub cover was relatively stable, while graminoids species
474 became more abundant (Fig. 4), also by colonizing open areas (average SAS vegetation cover increased over bare
475 ground/rock cover by 1.8% in CAS and by 5.125% in MOM). At the two lower summits (PCA and FOG), instead, the
476 diversity profile surface area decreased and Dominance index increased only in FOG, where the abundance of the
477 thermophilous shrub *Juniperus communis* subsp. *alpina* increased the most (Figure S1 in Supplementary Material).
478 PCA summit was the only one where both species richness and diversity increased, likely due to the particular feature
479 of this summit, characterised by widespread open areas, with a very low vegetation cover, and no clear trend in term of
480 plant functional type dynamics was observed.

481

482 **Assessment of β -diversity**

483 The higher absolute values of “multiple-site β_{SIM} ” in comparison with the ones of “multiple-site β_{SNE} ” indicate that in
484 the northern Apennines summits the species turnover (i.e. replacement) has a prevailing importance over nestedness
485 (i.e. species loss) in causing β -diversity patterns of vegetation. This result is likely due to the marked differences in
486 vegetation composition among SAS with different aspect; as highlighted by Winkler et al. 2016, different aspect
487 conditions can heavily influence species richness and species colonization, especially in temperate alpine summits, and
488 thus could result as a key driver for summits β -biodiversity. The predominance of species turnover over nestedness
489 suggests, once more, a “Mediterranean” pattern of vegetation diversity in the northern Apennines, consistent with the
490 findings of Baselga (2010), indicating that β -diversity of longhorn beetle fauna in southern Europe is almost completely
491 caused by species replacement only, whereas in northern Europe both turnover and nestedness contribute to β -diversity
492 patterns.

493 Biotic homogenization as a consequence of upward migrated species arrival, has been recently documented- in
494 subalpine and alpine vegetation of several European mountain systems (Jurasinski and Kreyling 2007; Britton et al.
495 2009; Ross et al. 2012; Cannone and Pignatti 2014; Matteodo et al. 2016). The analysis of β -diversity trends in our
496 study area between 2001 and 2015 showed that a process of homogenization is on-going also within ~~our Target~~
497 ~~Region~~northern Apennines, with a decrease of both multiple-site β_{SIM} and pairwise β_{sim} indices, indicating similarity
498 increase. The substantial asymmetry of changes resulting from the other biodiversity metrics analysis was confirmed
499 also for the homogenization process. Dissimilarity loss is stronger in the upper belts, as they were less affected by the
500 upward migration of new species. Nevertheless, the results obtained from the pairwise β_{sim} index calculation were not
501 significant at the belt level. This indicates a weak homogenization signal, which is, however, more difficult to detect
502 with fewer data, especially in the short time span of only ~~fourteen~~ ~~fifteen~~ years (Jurasinski and Kreyling 2007).

503

504 **Concluding remarks**

505

506 Our results confirmed the effectiveness of a multi-faceted approach for assessing plant diversity changes, capable of
507 deepening the way and entity of the modification processes in the summits investigated. Overall, the diversity of the
508 study site decreased over time, despite the increased species richness recorded in the period 2001-2015. In particular,
509 there were evidences of a spatial asymmetry of both species richness and diversity variation along an elevational
510 gradient: species enrichment was more effective on lower summits (and lower SAS belts) due to upward migration,

Commentato [SO11]: Hai appena detto our study area

Commentato [SO12]: Abbiamo sempre parlato di quindici anni

511 whereas in higher summits species richness remained unvaried. Cover of graminoids and shrubs increased, especially
512 on the higher summits, causing a reduction of the vegetation assemblages evenness and diversity. These findings, along
513 with the species turnover and species loss recorded at the study site, suggest the presence of an on-going process of
514 homogenization and biodiversity loss. Moreover, as the study area is located at the southern border of the “temperate”
515 mountains in Europe, the temporal trends in variation of species richness observed, could be interpreted as an alarming
516 signal, suggesting that a potential northward shift of the species richness dynamics observed in 2012 within the
517 Mediterranean GLORIA Target Regions characterized by species loss (Pauli et al. 2012), could be taking place.
518 Consequently, a transformation of the vegetation could be expected in the next decades also in the northern Apennines,
519 and conservation activities, such as *ex/in situ* conservation actions are needed in the very next future, especially for
520 plant species that are already declining in the studied mountain summits.

521 As a final consideration, we can say that the multi-faceted approach proposed in this study for the assessment
522 of biodiversity over time could be effectively applied to the long-term vegetation data collected in the other GLORIA
523 Target Regions. The diversity measures used in this case study are responsive to the vegetation changes recorded over
524 time, despite the not so long period of monitoring considered. They can give valuable information about diversity
525 dynamics of a site, detecting potential diversity loss and ongoing homogenization processes, that could be overlooked.
526 The potential application of these methods within the GLORIA network (that include more than 120 sites), could thus
527 result in a useful tool to highlight the areas most threatened by plant diversity loss in high elevation habitats all over the
528 world.

529

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535

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825

826 **Tables**

827

828 **Tab.1 Results of the linear models for species richness in relation to Time, Summit and Belt for: a) Target Region (32**
 829 **SAS), b) Summits (8 SAS for each of the four summits: CAS, MOM, PCA and FOG); Degrees of freedom (Df), F**
 830 **statistic and significance level are shown. *p*-values below 0.05 are given in bold (the analysis were run on the residual**
 831 **of species richness versus SAS area)**

Scale level	Factor	Df	F	<i>p</i> -value
a) Target Region				
	Time	2	20.07	< 0.001
	Summit	3	0.094	0.96
	Belt	1	2.25	0.14
	Time × Summit	6	8.32	< 0.001
	Time × Belt	2	0.36	0.70
	Time × Summit × Belt	6	1.74	0.12
	Summit × Belt	3	0.13	0.94
b) Summits				
CAS	Time	2	0.18	0.83
	Belt	1	0.05	0.82
	Time × Belt	2	1.28	0.30
MOM	Time	2	2.09	0.15
	Belt	1	0.14	0.71
	Time × Belt	2	0.45	0.64
PCA	Time	2	31.63	< 0.001
	Belt	1	4.49	0.049
	Time × Belt	2	2.71	0.09
FOG	Time	2	10.69	< 0.001
	Belt	1	1.82	0.19
	Time × Belt	2	0.31	0.73

833

834 **Tab.2** Shannon diversity, Sheldon evenness and Dominance index values calculated for 2001 and 2015 at two different
 835 levels: a) Target Region (32 SAS), b) Summits (8 SAS for each of the four summits: CAS, MOM, PCA and FOG).
 836 Surface area represents the underlying area of the diversity profile functions (see Figure 2 and 3)

Scale level	Shannon index		Sheldon evenness		Dominance index		Surface area	
	2001	2015	2001	2015	2001	2015	2001	2015
a) Target Region	4.19	4.16	0.43	0.40	0.022	0.025	231.1	218.8
b) Summits								
CAS	4.15	4.03	0.59	0.53	0.019	0.026	227.2	190.0
MOM	3.95	3.69	0.54	0.44	0.025	0.033	187.7	140.3
PCA	3.84	3.94	0.50	0.53	0.028	0.027	166.6	178.6
FOG	3.65	3.65	0.50	0.46	0.032	0.036	136.3	133.2

837

838 **Tab.3** Results of linear mixed-effect models performed on the sum of standardized cover of plant functional types in
 839 relation to Time for: a) Target Region (32 SAS included) and b) Summits (8 SAS included for each of the four summits:
 840 CAS, MOM, PCA and FOG). Degrees of freedom (Df), F statistic and significance level are shown. *p*-values below
 841 0.05 are given in bold

Scale level	Functional type	Factor	Df	F	<i>p</i> -value	
a) Target Region	Forbs	Time	1-31	0.59	0.448	
	Graminoids	Time	1-31	5.84	0.022	
	Shrubs	Time	1-31	4.40	0.044	
b) Summits	CAS	Graminoids	Time	1-7	11.06	0.013
		Shrubs	Time	1-7	1.08	0.334
	MOM	Graminoids	Time	1-7	36.42	< 0.001
		Shrubs	Time	1-7	3.36	0.109
	PCA	Graminoids	Time	1-7	1.15	0.319
		Shrubs	Time	1-7	0.83	0.393
	FOG	Graminoids	Time	1-7	0.96	0.361
		Shrubs	Time	1-7	0.12	0.738

842

843 **Tab.4** β -diversity multiple-site Simpson dissimilarity index (β_{SIM} *sensu* Baselga 2010) at the Target Region (32 SAS
 844 included) and at upper and lower belt level (each including 16 SAS) in 2001 and 2015

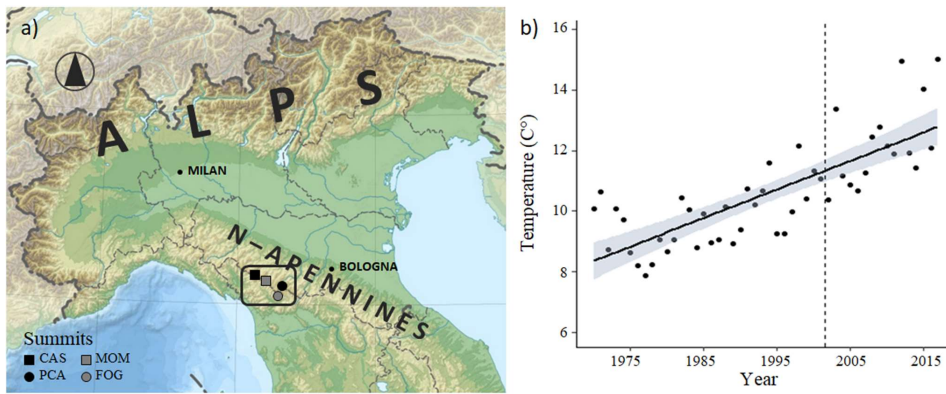
Scale level	β_{SIM} index 2015	
	2001	2015
Target Region	0.834	0.829
Upper belt	0.736	0.722
Lower belt	0.722	0.720

845

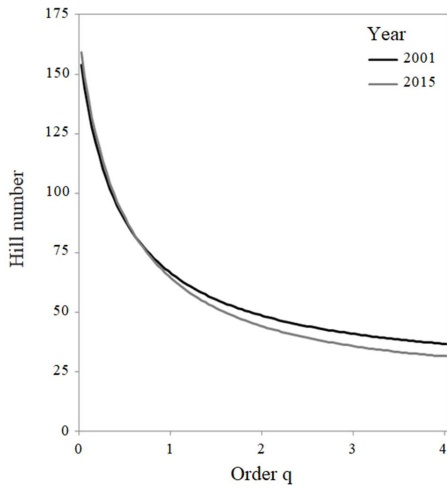
846 Figures

847

848 **Fig.1** a) Schematic map of the study area with geographic position of the sampled summits. CAS: Mt. Casarola; MOM:
 849 Alpe di Mommio; PCA: Cima di Pian Cavallaro; FOG: Cima di Foce a Giovo. b) Average summer temperatures (June
 850 to August) recorded at the Meteorological Observatory at Mt. Cimone (northern Apennines) from 1970 to 2017.
 851 Starting year of the GLORIA monitoring activity is highlighted by the dotted line (slope = 0.094, $R^2 = 0.613$, $p = 4.8e^{-11}$)
 852 ¹¹⁾

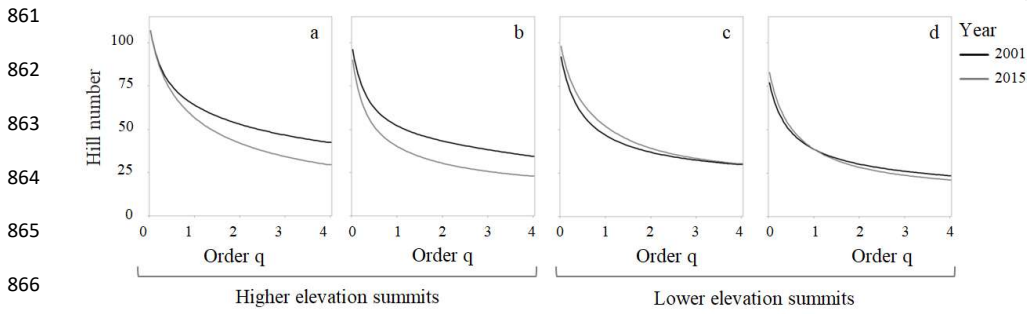


853
 854
 855 **Fig.2** Diversity profiles obtained by calculating the Hill Numbers for the Target Region using the relative species cover
 856 percentage dataset of 2001 and 2015



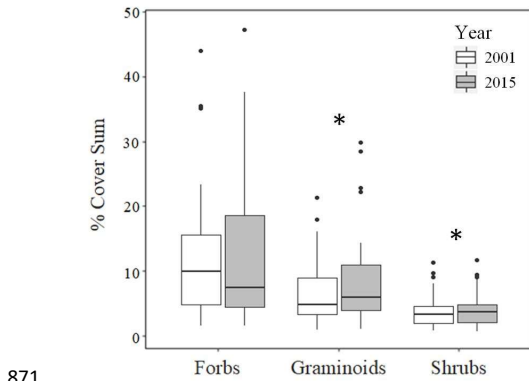
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859 **Fig.3** Diversity profiles obtained by calculating the Hill Numbers for the four summits investigated (a: CAS, b: MOM,
 860 c: PCA and d: FOG) using the relative species cover percentage dataset of 2001 and 2015

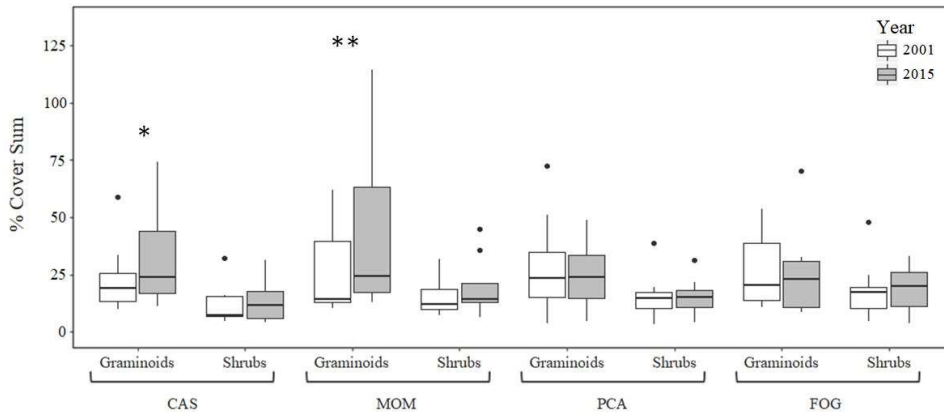


867

868 **Fig.4** Sum of standardized cover of plant functional types in 2001 and 2015 at the Target Region level. Horizontal bold
 869 line is the median, and the boxes are limited to the 1st and 3rd quartiles. Asterisks indicate significant changes between
 870 the two years (*: $p < 0.05$)



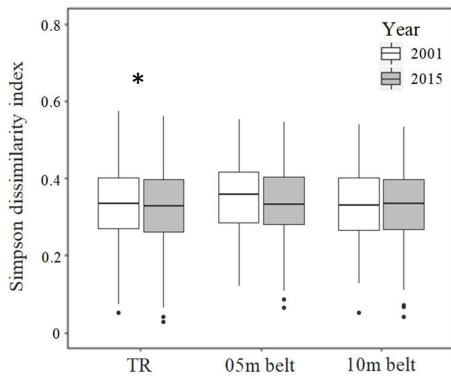
873 **Fig.5** Sum of standardized cover of plant functional types in 2001 and 2015 at the summit level. Horizontal bold line is
 874 the median, and the boxes are limited to the 1st and 3rd quartiles. Asterisks indicate significant changes between the two
 875 years (*: $p < 0.05$; **: $p < 0.01$).



876

877

878 **Fig.6** Pair-wise Simpson dissimilarity index (β_{sim} *sensu* Baselga, 2010) at the Target Region level and at the upper
 879 (05m) and lower (10m) belt levels in 2001 and 2015. Horizontal bold line is the median, and the boxes are limited to the
 880 1st and 3rd quartiles. Asterisks indicate significant changes between the two years (*: $p < 0.05$).



881

882

883 **Supplementary Material**

884

885 **Tab.S1** Elevation, geographic coordinates and climatic features (1970-2000) and vegetation belt of the four selected
 886 summits. CAS: Mt. Casarola; MOM: Alpe di Mommio; PCA: Cima di Pian Cavallaro; FOG: Cima di Foce a Giovo.
 887 Long-term climatic features of the four summits were extracted from WorldClim 2 (interval 1970-2000; Fick and
 888 Hijmans 2017) at 30 seconds spatial-resolution.

	CAS	MOM	PCA	FOG
Elevation (m a.s.l.)	1978	1855	1815	1722
Latitude (N)	44°19'53"	44°16'32"	44°12'09"	44°07'07"
Longitude (E)	10°12'37"	10°14'40"	10°41'33"	10°36'43"
Mean annual temperature (°C)	4.8	5.7	5.1	5.1
Mean annual precipitation (mm)	1055	1269	992	1065
Vegetation belt	Lower alpine	Higher subalpine	Lower subalpine	Treeline

889

890 **Tab.S2** Gloria abundances classes codes and descriptors (Pauli et al. 2015), along with the converted cover
891 approximation proposed following Braun-Blanquet

GLORIA abundance codes	GLORIA abundance classes descriptor	Converted % Cover approximation	Corresponding Braun-Blanquet scale codes
r! – very rare	Few individuals within the SAS	0.1%	+
r – rare	Some individuals at several locations that can hardly be overlooked in a careful observation	5%	1
s – scattered	Widespread within the SAS. Can't be overlooked but its presence could be not obvious at first glance	17.5%	2
c – common	Occurs frequently and is widespread within the section (cover < 50%)	37.5%	3
d - dominant	Very abundant, making up a high portion of the phyto mass (cover > 50%)	75%	4-5

892

893 **Tab.S3** Dataset composition and relative number of vascular species recorded in the three years of monitoring in the a)
894 Target Region, b) Summit and c) belt level. "SAS" indicates the number of summit area section included for each level,
895 whereas 05m and 10m indicate the upper and lower belt of each summit, respectively

Scale level	SAS	Number of species			Difference 2001-2015
		2001	2008	2015	
a) Target Region	32	154	164	159	+5
b) Summit					
CAS	8	107	104	107	0
MOM	8	96	98	90	-6
PCA	8	92	95	98	+6
FOG	8	77	85	83	+6
c) belt					
CAS 05m	4	89	88	93	+4
CAS 10m	4	102	99	98	-4
MOM 05m	4	83	81	77	-6
MOM 10m	4	81	86	79	-2
PCA 05m	4	87	87	88	+1
PCA 10m	4	80	90	94	+14
FOG 05m	4	65	69	67	+2
FOG 10m	4	69	78	80	+11

897

898 **Tab.S4** Area (expressed in m²) of all the 8 SAS of each summit (two per aspect) included in the Target Region IT-NAP.

899 Upper and lower SAS are indicated by “5” and “10” respectively

Summit	East 5	East 10	North 5	North 10	South 5	South10	West 5	West 10
CAS	155.36	392.09	229.1	567.71	171.04	264.38	214.73	809.53
MOM	139.5	155.52	131.15	441.31	169.07	212.29	534.05	597.93
PCA	304.55	725.26	53.75	133.06	225.23	465.38	398	394.24
FOG	72.63	504.98	109.15	198.07	84.00	215.37	290.12	254.48

900

901 **Tab.S5** List of the species removed from the database

Species

Astragalus purpureus Lam.
Bellardiochloa violacea (Bellardi) Chiov.
Carex pilulifera L.
Colchicum alpinum DC.
Gymnadenia conopsea (L.) R.Br.
Helianthemum nummularium (L.) Miller subsp. *nummularium*
Koeleria macrantha (Ledeb.) Schult.
Leucanthemum adustum (W.D.J. Koch) Grelli
Leucanthemum atratum (Jacq.) DC. subsp. *ceratophylloides* (All.) Horvatic
Pedicularis cenisia Gaudin
Polygonum viviparum L.
Scorzonera rosea Waldst. & Kit.
Sorbus aria (L.) Crantz
Unidentified *Hieracium* species

902

903 **Tab.S6** List of the species disappeared from the Target Region.

Species

Crocus vernus (L.) Hill subsp. *vernus*
Deschampsia cespitosa (L.) P.Beauv. subsp. *cespitosa*
Maianthemum bifolium (L.) F.W.Schmidt
Rumex nebroides Campd.
Sagina glabra (Willd.) Fenzl
Taraxacum officinale Weber agg.
Cuscuta epithymum (L.) L. subsp. *epithymum*
Anemone nemorosa L.
Lilium bulbiferum L. subsp. *bulbiferum* var. *croceum* (Chaix) Pers.

904

905 **Tab.S7** List of the species disappeared from at least one summit within the Target Region

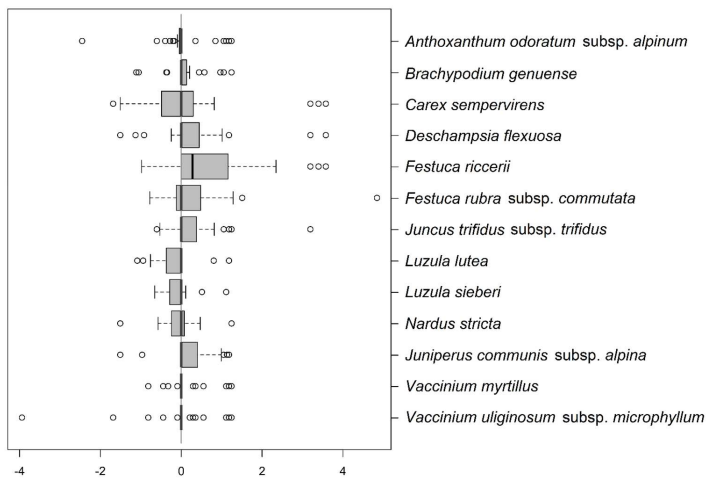
Species

Alchemilla transiens (Buser) Buser
Allium senescens subsp. *montanum* (F.W.Schmidt) Holub
Arenaria moehringioides Murr
Asplenium trichomanes-ramosum L.
Aster alpinus L.

Biscutella laevigata L. subsp. *laevigata*
Carduus carlinifolius Lam.
Galium anisophyllum Vill.
Galium x centroniae Cariot
Geranium sylvaticum L. subsp. *sylvaticum*
Hieracium lactucella Wallr.
Hieracium pilosella L.
Huperzia selago (L.) Bernh. ex Schrank & Mart. subsp. *selago*
Linum alpinum Jacq. subsp. *gracilius* (Bertol.) Pignatti
Myosotis alpestris F.W.Schmidt
Phyteuma orbiculare L.
Scabiosa lucida Vill.
Sedum monregalense Balb.
Viola biflora L.

906

907 **Fig.S1** Difference in standardized cover of the most abundant graminoids and shrubs in the summit studied.



908

909