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A five-year rotational grazing changes the botanical composition of sub-alpine and alpine grasslands

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Abstract

Aim: The implementation of Grazing Management Plans (GMP), a specific policy and management tool, aimed at enhancing farm productivity, while preserving plant diversity, soil, and landscape. The GMP are based on rotational grazing systems (RGS) with animal stocking rate adjusted to keep it balanced with grassland carrying capacity. The aim was to test the five-year effects produced by GMP implementation on botanical composition, plant diversity, and soil nutrient content on sub-alpine and alpine pastures.

Location: Val Troncea Natural Park, western Italian Alps.

Methods: A total of 199 vegetation transects was carried out in summer 2011 and repeated in summer 2016. The botanical composition was recorded and plant diversity indexes, i.e. species richness and Shannon diversity (H' index) were computed. Moreover, the mean soil nutrient content was indirectly estimated through the computation of Landolt N indicator value (N index) for each transect.

Pair-sample statistical tests and PERMANOVA were perfomed at different levels: on the whole vegetation dataset, on vegetation communities (i.e. vegetation types and ecological groups), and considering functional pools of species.

Results: Considering the whole vegetation dataset, species richness, H' index, and N index significantly increased from 2011 to 2016. Moreover, species richness significantly increased in almost all the vegetation ecological groups, with the highest increase within mesotrophic one. The H' index significantly increased in eutrophic, pre-forest, and thermic groups, while N index increased in all the vegetation groups, except in the eutrophic and snow-bed ones. A significant difference in botanic composition was detected within oligotrophic, mesotrophic, and thermic groups. The number and cover of nitrogen-poor high-elevation species increased in all groups and this result could be probably related to the effects produced by livestock, which promoted seed transportation and increased connectivity amongst different communities. The meso-eutrophic species number and cover significantly increased within thermic, mesotrophic, and pre-forest groups, suggesting a greater use of such areas by livestock under RGS than continuous grazing system.

Conclusions: The implementation of a RGS with stocking rate adjustments proved to be an effective and a sustainable management tool to enhance botanical composition and plant diversity of sub-alpine and alpine grasslands over a five-year span.

Keywords

Alps; Cattle grazing; High-elevation; Livestock management; Meso-eutrophic species; Pasture; Shannon diversity index; Species richness

Nomenclature

Pignatti (1982) for plant species; Aeschimann et al. (2004) for plant communities

Abbreviations: CGS = Continuous Grazing System; RGS = Rotational Grazing System; GMP = Grazing Management Plan

Introduction

Permanent grasslands, both meadows and pastures, are spread worldwide and cover about 25% of the earth's surface (FAOSTAT 2015, http://www.fao.org/faostat/en/#data/RL; accessed on 15 Dec 2017). They host a wide range of animal and plant species (Wilson et al. 2012; Dengler et al. 2014), amongst which many exclusively dependent on these open habitats (Reitalu et al. 2008; Schmid et al. 2017). Moreover, they provide essential functions and services to human and ecosystem health, such as animal production, carbon storage, nutrient cycling, pollination, soil protection and water conservation, wild-fire risk mitigation, and tourism opportunities (Silva et al. 2008; Harrison et al. 2010; Conant et al. 2017). However, permanent grasslands are among the main threatened ecosystems, particularly the most extensively managed ones. Indeed, in the last century, a massive decline of grasslands due to land-use changes has occurred (Dong et al. 2011; Gillet et al. 2016), resulting in a loss of biodiversity (Wesche et al. 2012).

The role of grazing in preserving and improving grassland ecosystems has been broadly studied (Collins et al. 1998; Adler et al. 2001; Sebastià et al. 2008; Li et al. 2017). Livestock management influences plant species composition (Olff & Ritchie 1998), nutrient redistribution (Malo & Suarez 1995; Dai 2000; Gaujour et al. 2012; Lonati et al. 2015), biomass removal (Borer et al. 2014), and soil and plant species cover by grazing, trampling, excreta deposition, and seed transportation (Pittarello et al. 2016 a, b; Probo et al. 2016). For all these reasons, it can modify intraspecific and interspecific competition dynamics among plant species, often favouring meso-eutrophic species dominance within regularly grazed grasslands (Niu et al. 2016; Nervo et al. 2017).

A dramatic decline in the overall area covered by grasslands has been also recorded for the Italian Alps (Bätzing 2005; Orlandi et al. 2016). Alps are a biodiversity hotspot, with many rare and endemic species, often relics of ice ages (Stehlik 2003). In the last decades, alpine semi-natural grasslands, shaped by millennia of extensive human activities and land management, have undergone a process of agro-pastoral abandonment due to socio-economic changes (Probo et al. 2013). As a consequence, undergrazing has occurred over large areas, i.e. often the steepest and most marginal ones, resulting in widespread vegetation cover and composition changes. Oligotrophic herbaceous species, shrubs and trees have encroached large areas of semi-natural grasslands, decreasing plant and animal diversity, meso-eutrophic species cover, forage mass and quality, and grassland carrying capacity (Probo et al. 2014). The carrying capacity has been defined by Allen et al. (2011) as the maximum livestock stocking rate achieving a target level of animal performance, in a specified grazing system, that can be applied over a defined time without deterioration of the grazing land.

For all these reasons, European Union latest policies aim to safeguard the remaining grasslands and to restore the degraded ones (Mikkonen & Moilanen 2013; Mihók et al. 2017).

Specifically, in the western Italian Alps (Piedmont Region) the implementation of Grazing Management Plans (GMP) has been promoted. The GMP are a management tool aiming at enhancing farm productivity, while preserving biodiversity, soil, and landscape through the application of farm-specific and sustainable grazing management actions (Lombardi et al. 2011; Argenti & Lombardi 2012). To achieve these objectives, GMP implemented rotational grazing systems (RGS) with animal stocking rate adjusted to keep it balanced with corresponding grassland carrying capacity (Probo et al. 2014). The RGS improved the spatial distribution of grazing cattle on rough sub-alpine (i.e. the ones located below the treeline) and alpine pastures by homogenizing the selection of different vegetation communities by livestock when compared to continuous grazing systems (CGS) (Probo et al. 2014). Indeed, free-roaming cattle under CGS, generally with a stocking rate lower than grassland carry capacity, resulted in a more uneven livestock distribution, with extensive undergrazing and overgrazing situations determined by the exacerbated selection for preferred vegetation communities. However, the effects produced by the implementation of GMP on plant diversity and botanical composition were not taken into consideration in previous research. Actually, some studies assessed the effects produced by RGS on plant and animal biodiversity, soil properties, vegetation cover, animal productivity, and forage selection and quality (Morris et al. 2005; Jacobo et al. 2006; Wrage et al. 2011; Teague et al. 2013; Ravetto Enri et al. 2017 a), but they were not conducted on high-elevation grasslands. No research, to our knowledge, has been carried-out on the medium-term effects (e.g. five years) of RGS on botanical composition, biodiversity indexes, and soil nutrient content on sub-alpine and alpine grasslands.

With this goal, a research was conducted on sub-alpine and alpine pastures of Val Troncea Natural Park (north-western Italy), which were exploited by a beef cattle herd, managed under the implementation of a GMP for five consecutive years (2011-2015). To test the effectiveness of GMP, the effects produced on plant diversity (species richness, Shannon diversity index, and beta-diversity indexes), soil nutrient content, estimated indirectly by Landolt N indicator value for soil nutrient, plant composition and species cover were assessed. It was hypothesized that after five-year of GMP implementation: (1) species richness and Shannon diversity index would enhance, (2) soil nutrient content would increase and homogenize across different vegetation communities, and (3) meso-eutrophic species number and cover would increase.

Methods

Study area and grazing management

The study was conducted within Val Troncea Natural Park (Piedmont, north-western Italy, latitude 44°57’N and longitude 6°57’E), which is included within the Site of Community Interest and the Special Protection Area (code id: IT1110080) of Nature 2000 network (92/43/EEC and 2009/147/EC directives) (Figure 1).

Dominant soils, originated from calcareous parent rock, were gravelly and nutrient-poor. According to the meteo station located at 2150 m a.s.l. (latitude 44°98’N and longitude 6°94’E), average annual temperature is about 4°C (February: -3.8°C; July: 12.6°C) and annual average precipitation is 703 mm (mean from 2003 to 2015).

Within the Park boundaries, one study area of 448 ha was selected (Figure 1). The study area was dominated by sub-alpine and alpine grasslands and shrublands, ranging from about 1900 to about 2820 m a.s.l. Grasslands were mainly dominated by *Festuca curvula* Gaudin, *Carex sempervirens* Vill., *Festuca nigrescens* Lam. non Gaudin, *Agrostis tenuis* Sibth., and *Poa alpina* L., while the shrub layer was predominantly composed by *Rhododendron ferrugineum* L., *Juniperus nana* Willd., *Vaccinium myrtillus* L., and *Vaccinium gaultherioides* Bigelow.

Until 2010, the study area was traditionally grazed by one free-ranging cattle herd of about 80 animal units (*sensu* Allen et al. 2011), composed by cows predominantly of Piedmontese breed. From 2011 to 2015, with the implementation of a specific GMP (Probo et al. 2014), the area experienced a change in grazing management, since it was subdivided into four paddocks, which were grazed under a RGS (starting in paddock 1 and ending in paddock 4, Figure 1), from the beginning of July to the end of September, by 105 animal units, according to grassland carrying capacity. The average annual carrying capacity of the four paddocks was calculated using the method defined in Daget & Poissonet (1971), based on the multiplication of the grazable area with pastoral value, which is a synthetic index of sward forage quality (Pittarello et al. 2018), and altitudinal and slope coefficients.

Vegetation transects

The study area was subdivided into 150 x 150-m grid cells and a linear vegetation transect was established in the centre of each grid cell using the vertical point-quadrat method (Daget & Poissonet 1971; Pittarello et al. 2017). A total of 199 vegetation transects was carried out in summer 2011 and was repeated in summer 2016 to measure vegetation changes after five years of GMP implementation. Each transect, 10-m long, was pinpointed thanks to a GPS device (Stonex® S3). Vegetation transects were carried out from June to August, before livestock grazing and at the flowering phenological stage of the dominant graminoids. At 50-cm interval along each transect, plant species touching a steel needle were identified and recorded. Since rare species are often missed by this method, a complete list of all other plant species included within a 1-m buffer area around the transect line, was also recorded (Probo et al. 2017).

Data analysis

The frequency of occurrence of each plant species recorded (f_i = number of occurrences/20 points of vegetation measurement), which is an estimate of species canopy cover, was calculated for each transect and converted to percentage cover (Pittarello et al. 2016 a). Species relative abundance (SRA_i) was determined in each transect and used to detect the proportion of different species according to the equation of Daget & Poissonet (1971):

$$SRA_i = \frac{f_i}{\sum_{i=1}^{n} f_i} \times 100$$

where SRA_i and f_i are species relative abundance and frequency of occurrence of species *i*.

Vegetation diversity was expressed according to two indexes: species richness and Shannon diversity index. Shannon diversity index (H') was calculated for each transect according to the equation (Magurran 1988):

$$\mathrm{H}' = -\sum_{i=1}^{i=n} \left\{ \frac{SRA_i}{100} \times \log_2 \left(\frac{SRA_i}{100} \right) \right\}$$

where SRA_i is the species relative abundance of species *i*.

Moreover, beta-diversity indexes (Sørensen, Horn and Morosita-Horn) were computed for each paddock, considering the multiple-community dissimilarity between transects (Chao et al. 2008; Chao et al. 2012). The indexes differ in the weights attributed to SRA_i (Magurran & McGill 2011), ranging from 0 (set of identical communities) to 1 (set of communities that share no species).

Soil nutrient content was estimated indirectly by Landolt N indicator value for soil nutrient (hereafter N index). Indeed, the N index can properly characterize an area (Tölgyesi et al. 2014) and it is well correlated to the supply of several nutrients, such as nitrogen, phosphorous, and potassium (Diekmann 2003). Each plant species was associated to the corresponding N value (Landolt et al. 2010) and the mean N index was calculated for each transect, by averaging species values weighted on their SRA (Ravetto Enri et al. 2017 b).

In order to identify different functional pools of species, which are characterized by similar ecological needs, each plant species was classified according to its phytosociological optimum at the class level, as identified by Aeschimann et al. (2004). Moreover, species belonging to phytosociological classes having physiognomic, ecological, and floristic similarity were pooled according to Theurillat et al. (1995; Appendix S1). Six functional species pools out of the ten detected were retained for further analyses, in order to focus on the ones more directly affected by grazing management, i.e. excluding typical forest and rocky species.

Statistical analyses

Changes between 2011 and 2016 in plant diversity, soil nutrient content, and botanical composition were assessed at different levels: on whole vegetation dataset, on vegetation communities (i.e. vegetation types and ecological groups), and considering the functional pools of species identified in Appendix S1.

Vegetation transects were classified into vegetation types and ecological groups, according to Cavallero et al. (2007) and Probo et al. (2014) by hierarchical cluster analysis (Appendix S2). The classification variable was SRA, the cluster method was Pearson correlation coefficient, and the between group linkage was the resemblance coefficient.

To detect differences in plant diversity indexes and N index between 2011 and 2016, on the whole dataset and for each vegetation ecological group, pair-sample statistical tests were perfomed. Vegetation variables were tested for normality and homogeneity of variance using Shapiro-Wilk and Levene's test, respectively. When assumptions were not verified, data were log-transformed and normality and homogeneity of variance were tested on log-transformed data. Paired-sample *t*-tests and non-parametric Wilcoxon signed-rank tests (Sokal & Rohlf 1995) were used, depending if normality and homogeneity of variance were verified or not, respectively. The same analysis was carried out to test the differences in beta-diversity between 2011 and 2016, considering paddock as the experimental unit.

A permutational analysis of variance (PERMANOVA), considering species percentage cover as main variable, was performed using 9999 permutations to assess differences in botanical composition for the whole dataset and for each vegetation ecological group identified, using Bray-Curtis dissimilarity as measure of the vegetation changes between 2011 and 2016.

Moreover, changes in the total number and cover of species belonging to different functional pools between 2011 and 2016 were tested by paired-sample *t*-tests and non-parametric Wilcoxon signed-rank tests for each ecological group, except for the rocky one, since it proved to be not significantly different between surveyed years.

All stastistical analyses were performed using IBM SPSS Statistics 24.0 (IBM Corp. Released 2016. IBM SPSS Statistics for Windows, Version 24.0. Armonk, NY: IBM Corp.), except the PERMANOVA analysis, which was performed with Past software (PAST 3.16, Hammer et al. 2001) and beta-diversity computed using R 3.4.1 (R Core Team, 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/) with "SpadeR" package (Chao et al. 2016).

Results

Effects on plant diversity and N index

A total of 275 and 281 plant species was recorded, in 2011 and in 2016, respectively. The complete species list and corresponding Landolt N indicator values, phytosociological optima, and functional species pool attribution are available in Appendix S3. Vegetation transects recorded in 2011 were classified into 18 vegetation types and seven ecological groups (Appendix S2). The thermic ecological group was the most widespread in the study area, followed by oligotrophic and pre-forest and shrub-encroached ones (hereafter called pre-forest group), with eutrophic and mesotrophic groups together accounting for 22.7% of the total area. Considering the whole vegetation dataset, species richness, Shannon diversity index, and N index significantly increased from 2011 to 2016 (Table 1). Conversely, Sørensen, Horn and Morosita-Horn indexes of beta-diversity did not show significant differences between the two years (Table 1).

Species richness significantly increased in almost all the vegetation ecological groups except within the rocky one, with the highest increase detected within mesotrophic group (+ 11.5 species, on average). H’ index significantly increased in eutrophic, thermic, and pre-forest groups, while N index increased in all the vegetation groups except in eutrophic and snow-bed ones (Table 2).

Effects on botanical composition

The PERMANOVA performed on the whole dataset did not show a significant difference between 2011 and 2016 (*P*-value = 0.262), while differences in species composition were detected within mesotrophic, oligotrophic, and thermic groups (Table 3).

Considering the whole vegetation dataset, a significant increase in the number of species was detected for all functional pools of species (Table 4). Conversely, species cover significantly changed only for nitrogen-poor high-elevation and meso-eutrophic species pools.

The number and cover of species belonging to different pools highlighted predominant changes within oligotrophic, thermic, and pre-forest ecological groups, while less pronounced changes were detected within other groups (Table 5). More in detail, in all the ecological groups the number of nitrogen-poor high-elevation species (e.g. *Myosotis alpestris* f.w. Schmidt, *Festuca violacea* Gaudin, *Agrostis alpina* Scop., etc.) significantly increased. Moreover, some species belonging to the same pool significantly increased their cover (e.g. *F. violacea*, *A. alpina*, *Carex rosae* (Gilomen) Hess et Landolt, etc.) in almost all the vegetation ecological groups. Conversely, the number of meso-eutrophic species (e.g. *Poa alpina* L., *Achillea* gr. *millefolium*, *Carum carvi* L., etc.) increased only within thermic groups, while they increased their cover within mesotrophic, pre-forest, and snow-bed ones. The number of ruderal species (e.g. *Cerastium arvense* L., *Gagea fistulosa* (Ramond) Ker.-Gawl., *Cirsium eriophorum* (L.) Scop., etc.) increased in oligotrophic, thermic, and pre-forest groups, while their cover increased in the mesotrophic one. Snow-bed species number (e.g. *Salix herbacea* L., *Sibbaldia procumbens* L., *Luzula alpino-pilosa* (Chaix) Breistr.) mainly increased within the snow-bed group. The total number of species belonging to the fringe and tall herb pool (e.g. *Poa chaixii* Vill., *Pulmonaria australis* (Murr) Sauer, *Cruciata glabra* (L.) Ehrend., *Geranium sylvaticum* L., etc.) significantly increased only within pre-forest group, while the cover of these species increased within the eutrophic one.

Discussion

The five-year implementation of GMP provided very promising results for biodiversity conservation in sub-alpine and alpine pastures. The implementation of a RGS with stocking rate adjustments proved to be an effective management tool to enhance botanical composition and plant diversity of sub-alpine and alpine grasslands. Indeed, beyond the improvement of grazing spatial distribution on rough areas and the homogenization of the selection of vegetation communities by cattle herds since the very beginning of their application (Probo et al. 2014), it also significantly changed these communities during a five-year span. A more homogeneous exploitation of sub-alpine and alpine grasslands by livestock led to an overall increase of species richness, H' index, and soil nutrient content, as well as to remarkable changes in plant species composition and cover, despite the well documented evidence of slow vegetation dynamics at high elevation (Tasser & Tappeiner 2002; Körner 2003). Mesotrophic, oligotrophic, and thermic groups showed significant changes in botanical composition. This result is partly consistent with Pavlů et al. (2003), who studied the effects of different grazing practices on the vegetation of a poorly improved upland pasture. These authors observed, in five years, a significant distinction in plant species composition and structure between rotational grazing and continuous stocking, but they did not detect a significant effect on species richness. Also Jacobo et al. (2006) evaluated the effects of RGS compared to the traditional CGS on floristic composition dynamic in a study conducted in the Flooding Pampa region, over a period of 4 years. The authors did not detect plant species diversity changes due to grazing management, while other variables did, e.g. functional group cover, and they ascribed this result to a new balance in the vegetation community typical of temperate grasslands, where the increase of some species led to a decrease of others. Conversely, in the present study, the change in grazing regime led also to an increase in plant species richness within all the considered ecological groups, except in the rocky one, which was little interested by grazing, due to its topographical and vegetation constraints (Probo et al. 2014). Moreover, beta-diversity remained stable, highlighting that the increase in species richness was homogeneously distributed across vegetation transects within paddocks.

Both H’ and N indexes showed the same increasing trend, considering the whole dataset, thus confirming that grazing enhances species diversity and soil nutrient, homogenizing it across different vegetation communities and supporting the first two experimental hypotheses. The increase in the mean N value from 2.3 to 2.4 and the consequent biodiversity enhancement measured are consistent with Pittarello et al. (2018), who found that species richness and H’ index peaked at an average N index value of 2.5 along an “humped-shape” curve in alpine pastures. The highest increases of N value were recorded within two vegetation ecological groups normally avoided and undergrazed by free-ranging animals, i.e. thermic and pre-forest ones (Probo et al. 2014), highlighting an increased exploitation of these communities.

The third hypothesis was confirmed as well, i.e. meso-eutrophic species significantly increased both in their number and cover. Particularly, they increased within mesotrophic, thermic, and pre-forest groups, suggesting a greater use of such areas by livestock under RGS than CGS. This is also supported by the increase of N index value in these groups, which was probably related to the increase of soil nutrient content produced by the increase of excreta deposition by livestock (Aarons et al. 2004; Güsewell et al. 2005). The increased nutrient availability and cattle trampling favoured the establishment of meso-eutrophic more competitive species (i.e. *A.* gr. *millefolium, C. carvi,* etc.), whose main dispersal mechanisms are endochory and epichory (Traba et al. 2003; Cosyns et al. 2005).

The effect of the change in grazing management on botanical composition can be further demonstrated by the increase both in number and in cover of plant species belonging to the nitrogen-poor high-elevation pool within all vegetation ecological groups. The increase of these species at lower elevation could be probably related to the seed transportation operated by livestock. Indeed, the RGS implementation allowed livestock to homogenise the selection for different areas within the paddock, even the most inaccessible, steep and high-altitude ones (Bailey et al. 1998; Probo et al. 2014). Therefore, animals may have promoted seed transportation and increased connectivity amongst different communities. The increase in ruderal species number within communities generally avoided by free-ranging animals can also be connected to the “disturbance” effect of the grazing livestock (Bullock et al. 2001; Teuber et al. 2013). Although many of these species, characterized by relative short life, light-demanding and which produce a large amount of seeds, are important for pollinators, some of them (i.e. *Cirsium eriophorum* (L.) Scop., *Cirsium spinosissimum* (L.) Scop.) should be monitored in the coming years to avoid their negative effect on forage quality and palatability (Pittarello et al. 2018). However, the ecological impact of these species is limited, since they are all autochthonous species and any alien species was recorded in the study area due to high altitude constraints (Marini et al. 2009).

To conclude, RGS with stocking rate adjustments was an effective and a sustainable management tool to protect semi-natural grassland diversity threatened by the abandonment, even in high-altitude grasslands. However, to assess the long-term effects produced by this policy and management tool implementation, a longer monitoring of the vegetation and a deeper assessment of other ecological variables and animal productivity would be advisable for future research. Specifically, the possibility that RGS promote a homogenization of botanical composition across different vegetation communities within paddocks due to the loss of peculiar ecological niches, should be further investigated on the long term.

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Supplementary material

Appendix S1. Functional pools of species recorded in the study area with the corresponding phytosociological optimum at the class level. * = groups taken into account for statistical analyses.

Appendix S2. Vegetation ecological groups and types (labelled according to dominant species names), corresponding phytosociological community, and area covered by each of them.

Appendix S3. List of the plant species recorded in the botanical transects, with the corresponding phytosociological optimum and class, functional pool of species, and N index (Landolt et al. 2010).

Figure 1. Location of the study area in Val Troncea, Piedmont, Italy (UTM zone 32 N, WGS84 datum).

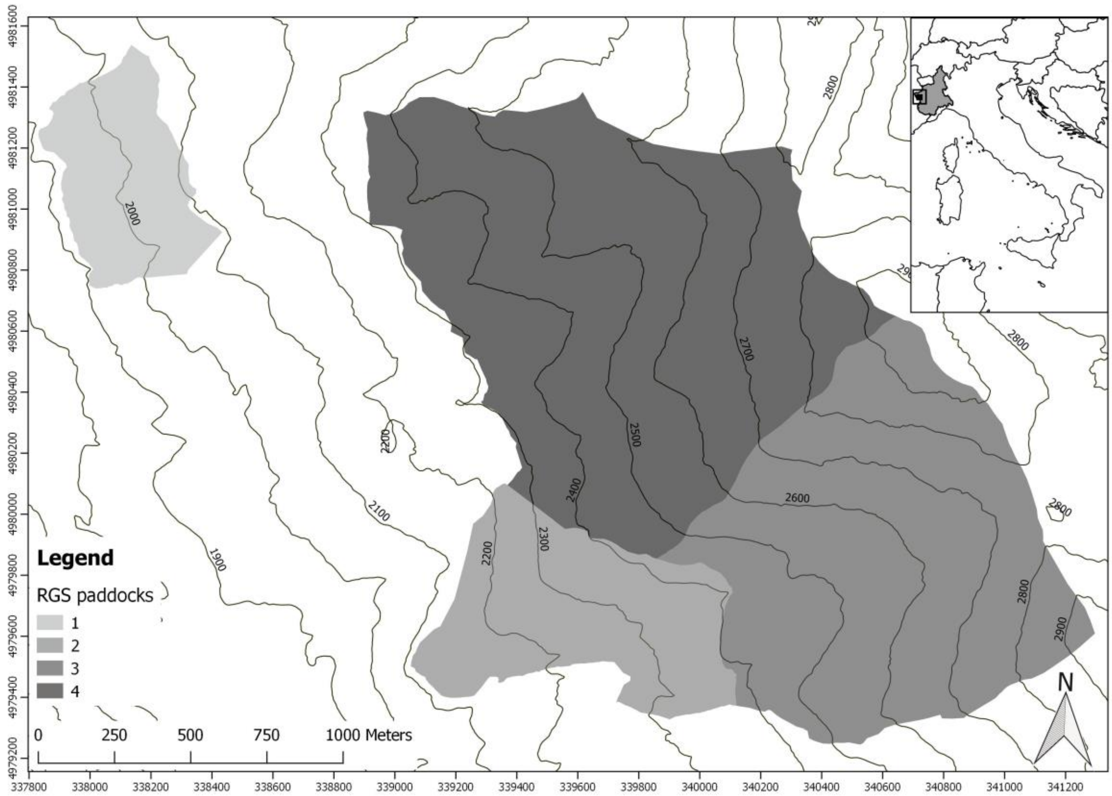

Table 1. Mean species richness, Shannon diversity index (H' index), Landolt N indicator (N index) values, and beta-diversity (Sørensen, Horn and Morosita-Horn indexes) in 2011 and 2016 for the whole vegetation dataset. Asterisks represent the statistical significance level of differences between 2011 and 2016: *** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; ns, not significant ($P > 0.1$).

	2011			2016			
Variables	Mean		SE	Mean		SE	*P*-value
Sp. richness	28.72	±	0.622	36.11	±	0.814	***
H' index	3.35	±	0.040	3.51	±	0.042	***
N index	2.32	±	0.020	2.40	±	0.025	***
Sørensen index	0.22	±	0.011	0.22	±	0.013	ns
Horn index	0.45	±	0.006	0.46	±	0.007	ns
Morosita-Horn index	0.75	±	0.009	0.76	±	0.008	ns

Table 2. Mean species richness, Shannon diversity index (H' index) and Landolt N indicator value (N index) values in 2011 and 2016 vegetation transects. Asterisks represent the statistical significance level of differences between 2011 and 2016: *** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; ns, not significant ($P > 0.1$).

		2011			2016			
Vegetation ecological group	Variables	Mean		SE	Mean		SE	*P*-value
1. Eutrophic group	Sp. richness	28.14	±	1.133	34.43	±	1.606	***
	H' index	3.51	±	0.069	3.68	±	0.094	*
	N index	2.70	±	0.054	2.70	±	0.089	ns
2. Mesotrophic group	Sp. richness	34.50	±	3.864	46.00	±	5.213	*
	H' index	3.73	±	0.105	3.78	±	0.126	ns
	N index	2.63	±	0.057	2.83	±	0.064	**
3. Oligotrophic group	Sp. richness	26.53	±	0.908	33.50	±	1.533	***
	H' index	3.41	±	0.062	3.51	±	0.077	ns
	N index	2.28	±	0.027	2.33	±	0.031	*
4. Thermic group	Sp. richness	30.53	±	1.165	38.27	±	1.319	***
	H' index	3.36	±	0.062	3.59	±	0.064	***
	N index	2.14	±	0.026	2.24	±	0.033	**
5. Pre-forest group	Sp. richness	29.16	±	1.505	36.56	±	1.943	***
	H' index	3.14	±	0.134	3.38	±	0.114	**
	N index	2.22	±	0.024	2.31	±	0.042	*
6. Snow-bed group	Sp. richness	26.75	±	1.851	35.08	±	2.155	**
	H' index	3.10	±	0.177	3.11	±	0.215	ns
	N index	2.39	±	0.065	2.52	±	0.089	ns
7. Rocky group	Sp. richness	20.50	±	2.172	22.50	±	2.952	ns
	H' index	2.83	±	0.244	2.70	±	0.202	ns
	N index	2.32	±	0.089	2.09	±	0.074	*

Table 3. Results of the PERMANOVA analysis for the ecological groups comparing botanical composition in 2011 and 2016 (Permutation number = 9999). Asterisks represent the statistical significance level of differences between 2011 and 2016: *** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; ns, not significant ($P > 0.1$). SST = Total sum of squares; SSW = Within-group sum of squares.

Vegetation ecological group	SST	SSW	F	*P*-value
1. Eutrophic group	17.8	17.6	0.83	ns
2. Mesotrophic group	9.9	9.45	1.89	*
3. Oligotrophic group	18.1	17.6	2.35	**
4. Thermic group	41.9	41.2	1.83	*
5. Pre-forest and shrub-encroached group	24.0	23.7	0.93	ns
6. Snow-bed group	6.8	6.47	1.08	ns
7. Rocky group	2.6	2.4	0.97	ns

Table 4. Effects produced by Grazing Management Plan implementation on the total number and cover of species having the phytosociological optimum belonging to different functional species pools for the whole vegetation dataset. Asterisks represent the statistical significance level of differences between 2011 and 2016: *** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; "+" $P < 0.1$; ns, not significant ($P > 0.1$).

	Species number							Species cover						
	2011			2016				2011			2016			
Functional species pool	Mean		SE	Mean		SE	*P*-value	Mean		SE	Mean		SE	*P*-value
Ruderal plant species	0.8	±	0.05	1.1	±	0.06	***	2.9	±	0.44	3.2	±	0.39	ns
Snow-bed species	0.5	±	0.08	0.7	±	0.10	**	2.5	±	0.69	3.2	±	0.84	ns
Nitrogen-poor high-elevation species	15.7	±	0.42	20.4	±	0.50	***	99.1	±	3.30	120.4	±	3.91	***
Fringe and tall herb species	0.6	±	0.10	0.7	±	0.12	+	9.2	±	1.50	9.8	±	1.53	ns
Dry species	2.4	±	0.21	2.8	±	0.27	*	29.4	±	2.27	31.8	±	2.73	ns
Meso-eutrophic species	3.0	±	0.18	3.4	±	0.23	**	32.4	±	2.56	42.9	±	3.84	***

Table 5. Effects producted by Grazing Management Plan implementation on the total number and cover of species having the phytosociological optimum belonging to different functional species pools for each vegetation ecological group identified. Asterisks represent the statistical significance level of differences between 2011 and 2016: *** P < 0.001; ** P < 0.01; * P < 0.05; "+" P < 0.1; ns, not significant (P > 0.1).

		Species number							Species cover						
		2011			2016				2011			2016			
Vegetation ecological group	Functional species pool	Mean		SE	Mean		SE	*P*-value	Mean		SE	Mean		SE	*P*-value
1. Eutrophic group	Ruderal species	1.1	±	0.16	1.4	±	0.17	+	8.9	±	2.23	6.5	±	1.26	ns
	Snow-bed species	1.4	±	0.37	2.1	±	0.49	ns	3.2	±	1.48	2.9	±	1.31	ns
	Nitrogen-poor high-elevation species	17.4	±	0.89	21.6	±	1.30	*	78.9	±	8.58	102.7	±	7.65	**
	Fringe and tall herb species	0.1	±	0.10	-	±	-	ns	2.3	±	0.97	7.5	±	2.55	*
	Dry species	1.0	±	0.30	0.8	±	0.28	ns	35.5	±	8.16	42.9	±	10.67	ns
	Meso-eutrophic species	2.6	±	0.17	2.8	±	0.30	ns	72.8	±	8.15	81.9	±	14.48	ns
2. Mesotrophic group	Ruderal species	1.4	±	0.46	2.1	±	0.23	ns	1.2	±	0.67	3.7	±	1.32	+
	Snow-bed species	0.3	±	0.25	-	±	-	ns	0.0	±	0.00	0.0	±	0.00	ns
	Nitrogen-poor high-elevation species	12.6	±	1.15	19.8	±	1.35	**	111.5	±	9.28	138.7	±	11.88	ns
	Fringe and tall herb species	2.9	±	0.90	3.3	±	1.10	ns	36.2	±	7.46	36.3	±	7.93	ns
	Dry species	4.6	±	1.12	5.6	±	1.51	ns	29.8	±	4.28	30.5	±	9.40	ns
	Meso-eutrophic species	8.8	±	1.15	10.8	±	2.08	ns	82.3	±	10.82	117.1	±	17.78	*
3. Oligotrophic group	Ruderal species	0.7	±	0.08	1.2	±	0.10	***	2.9	±	0.70	3.4	±	0.82	ns
	Snow-bed species	0.3	±	0.11	0.6	±	0.15	*	0.4	±	0.21	1.4	±	1.07	ns
	Nitrogen-poor high-elevation species	17.9	±	0.57	22.2	±	0.92	***	148.4	±	5.00	160.7	±	5.93	+
	Fringe and tall herb species	0.1	±	0.05	0.1	±	0.06	ns	0.0	±	0.00	0.5	±	0.53	ns
	Dry species	1.3	±	0.25	1.4	±	0.27	ns	12.9	±	2.15	14.5	±	3.09	ns
	Meso-eutrophic species	2.2	±	0.16	2.3	±	0.18	ns	18.7	±	3.11	21.8	±	3.19	ns
4. Thermic group	Ruderal species	0.8	±	0.07	1.1	±	0.09	*	2.5	±	0.69	3.4	±	0.75	ns
	Snow-bed species	0.3	±	0.09	0.3	±	0.11	ns	0.9	±	0.41	0.9	±	0.52	ns
	Nitrogen-poor high-elevation species	16.4	±	0.80	21.5	±	0.87	***	104.9	±	4.51	125.2	±	6.28	**
	Fringe and tall herb species	0.4	±	0.12	0.4	±	0.13	ns	4.7	±	1.64	2.4	±	0.97	+

	Dry species	3.5	±	0.36	4.2	±	0.45	*	46.9	±	4.11	48.5	±	4.59	ns
	Meso-eutrophic species	2.8	±	0.26	3.6	±	0.30	**	19.2	±	2.60	26.9	±	3.75	*
5. Pre-forest and shrub-encroached group	Ruderal species	0.6	±	0.13	0.9	±	0.16	*	0.7	±	0.35	1.6	±	0.70	ns
	Snow-bed species	0.0	±	0.04	0.2	±	0.10	ns	0.0	±	0.00	1.0	±	0.76	ns
	Nitrogen-poor high-elevation species	13.4	±	0.96	17.6	±	1.24	***	62.2	±	6.20	85.3	±	9.75	**
	Fringe and tall herb species	1.5	±	0.32	2.0	±	0.36	+	22.1	±	5.15	25.3	±	5.19	ns
	Dry species	2.6	±	0.59	3.5	±	0.87	*	25.1	±	5.68	30.3	±	6.18	ns
	Meso-eutrophic species	3.1	±	0.37	3.1	±	0.35	ns	17.4	±	3.01	26.9	±	5.61	+
6. Snow-bed group	Ruderal species	0.8	±	0.21	0.8	±	0.21	ns	2.3	±	1.08	0.4	±	0.38	+
	Snow-bed species	1.8	±	0.34	2.7	±	0.36	*	25.0	±	7.70	28.1	±	8.94	ns
	Nitrogen-poor high-elevation species	14.8	±	1.67	20.4	±	1.53	*	69.6	±	8.96	106.9	±	17.96	+
	Fringe and tall herb species	-	±	-	-	±	-	ns	0.9	±	0.92	1.5	±	1.54	ns
	Dry species	0.8	±	0.30	0.4	±	0.15	ns	6.0	±	3.55	1.5	±	0.67	ns
	Meso-eutrophic species	2.9	±	0.29	3.4	±	0.31	ns	31.1	±	6.94	55.4	±	14.75	*

Appendix S1. Functional pools of species recorded in the study area with the corresponding phytosociological optimum at the class level. * = groups taken into account for statistical analyses. Syntaxonomic nomenclature follows Aeschimann et al. 2004.

Functional species pool	Phytosociological class
Dry species*	*Festuco-Brometea*
Meso-eutrophic species*	*Molinio-Arrhenatheretea*
Snow-bed species*	*Salicetea herbaceae*
Nitrogen-poor high-elevation species (acidic and calcareous)*	*Juncetea trifidi,*
	Nardetea strictae,
	Carici rupestris-Kobresietea bellardii,
	Elyno-Seslerietea variae
Fringe and tall herb species*	*Epilobietea angustifolii,*
	Filipendulo-Convolvuletea,
	Mulgedio-Aconitetea,
	Trifolio-Geranietea sanguinei
Ruderal species*	*Agropyretea intermedii-repentis,*
	Artemisietea vulgaris,
	Polygono-Poetea annuae,
	Stellarietea mediae
Boreal forest species	*Erico-Pinetea,*
	Pyrolo-Pinetea,
	Vaccinio-Piceetea excelsae
Boreal shrubland species	*Betulo carpaticae-Alnetea viridis,*
	Crataego-Prunetea,
	Roso pendulinae-Pinetea mugo
Broad-leaved forest species	*Carpino-Fagetea,*
	Quercetea robori-sessiliflorae
Rocky species	*Asplenietea trichomanis,*
	Thlaspietea rotundifolii

Appendix S2. Vegetation ecological groups and types (labelled according to dominant species names), corresponding phytosociological community, and area covered by each of them. Vegetation ecological groups and types follows Cavallero et al. (2007), species nomenclature follows Pignatti et al. 1982, syntaxonomic nomenclature follows Aeschimann et al. 2004.

Vegetation ecological group	Vegetation type	Phytosociological plant community	Area (ha)	Area (%)
1 - Eutrophic group			58.5	13.1
	Dactylis glomerata	*Polygono-Trisetion*	22.5	5.0
	Poa alpina	*Crepido-Festucetum commutatae (=Poetum alpinae)*	36.0	8.0
2 - Mesotrophic group			42.8	9.6
	Festuca gr. *rubra* and *Agrostis tenuis*	*Nardo-Agrostion tenuis*	42.8	9.6
3 - Oligotrophic group			85.6	19.1
	Nardus stricta	*Nardion strictae*	58.5	13.1
	Carex sempervirens	*Caricion curvulae*	6.8	1.5
	Trifolium alpinum and *Carex sempervirens*	*Caricion curvulae*	20.3	4.5
4 - Thermic group			139.6	31.2
	Carex rosae	*Seslerion variae*	13.5	3.0
	Elyna myosuroides	*Oxytropido-Elynion*	13.5	3.0
	Festuca quadriflora	*Caricion firmae*	22.5	5.0
	Helianthemum nummularium	Transition between *Seslerion variae* and *Caricion ferrugineae*	78.8	17.6
	Helianthemum oelandicum	*Seslerion coeruleae* (transition towards *Caricion ferrugineae)*	4.5	1.0
	Sesleria varia	*Seslerion variae*	6.8	1.5
5 – Pre-forest and shrub-encroached group			78.8	17.6
	Calamagrostis villosa	*Calamagrostion villosae*	4.5	1.0
	Juniperus nana	*Junipero-Arctostaphyletum*	33.8	7.5
	Vaccinium gaultherioides	*Loiseleurio-Vaccinion*	40.5	9.0
6 - Snow-bed group			29.3	6.5
	Plantago alpina	Transition between *Salicion herbaceae* and *Nardion strictae*	18	4.0
	Salix herbacea	*Salicion herbaceae*	11.3	2.5
7 - Rocky group			13.5	3.0
	Saxifraga oppositifolia	*Thlaspion rotundifolii*	13.5	3.0
			448.1	100