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Agri-environmental payments drive the conservation and forage value of semi-natural grasslands by modifying fine-scale grazing intensity

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ABSTRACT

In Europe, the conservation of extensively grazed semi-natural grasslands is addressed by agricultural policies whose effectiveness is questioned. We studied sub-xerophilous *Bromus erectus* semi-natural grasslands to analyse the interactions among: i) agri-environmental payments, ii) grazing regimes, iii) environmental conditions, iv) habitat conservation state, and v) forage yield and quality.

We sampled 98 plots across 19 farms and unmanaged control areas in five regions encompassing Italy and Switzerland. We fitted two piecewise structural equation models (SEM) to infer direct and indirect effects of agri-environmental payments, grazing regimes and environmental conditions on proxies of habitat conservation state, (i.e., the number and cover of diagnostic species), and forage yield and quality (i.e., specific leaf area - SLA, leaf dry-matter content - LDMC, sward height and pastoral value).

Agri-environmental payments contributed to maintain grazing management and in turn to preserve the habitat biodiversity and functions. Payments did not affect stocking rates, but determined a more even distribution of grazing intensity, with positive effects on habitat conservation state and negative outcomes for LDMC. Conversely, LDMC increased with stocking rates. Among environmental condition, elevation and soil carbonates content had a positive effect on the habitat conservation state, while slope exerted only indirect effects on forage quality and diagnostic species by reducing fine-scale grazing intensity. Overall, the effectiveness of payments largely depended on the scale of measures' implementation. Farm-level grazing contracts and periodic field monitoring would allow to influence the fine-scale grazing intensity and to implement a result-oriented approach towards the objectives of the post-2020 CAP.

1. Introduction

European semi-natural grasslands have been maintained since the Mesolithic–Neolithic transition through agro-pastoral practices (Hejcman et al., 2013). They are among the most species-rich ecosystems worldwide (Wilson et al., 2012; Habel et al., 2013; Dengler et al., 2014), and are important for provisioning, regulating, and cultural ecosystem services (Bengtsson et al., 2019). In the European lowlands and hilly regions extensively grazed semi-natural grasslands are relevant mainly for biodiversity conservation and support to pollinators; while in mountain regions they also play an important economic role for local societies through provisioning services (Mengist et al., 2020).

Despite their recognized relevance, European semi-natural grasslands have one of the worst conservation states among all terrestrial ecosystems (EEA, 2010). Unbalanced grazing is among the most widespread drivers of degradation (EEA, 2016). On the one hand, abandonment and undergrazing result in the rapid accumulation of litter and soil organic matter (Zou et al., 2016), sward height and coarse grasses increase (Cislaghi et al., 2019) and species diversity declines towards woody species-encroached communities (EEA, 2016). On the other

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hand, overgrazing leads to the dominance of few species adapted to high nutrient concentration and intense trampling and defoliation (e.g., species with spines, high content of lignin or toxic compounds), often associated to soil compaction, nitrification and erosion (Mysterud, 2006; Díaz et al., 2007). Under- and overgrazing cause the reduction of the semi-natural grassland extent, and the modification of their species composition, with detrimental effects on ecosystem conservation state and services (Bengtsson et al., 2019).

Maintaining semi-natural habitats and environment-friendly rural activities is highly prioritized in the European agenda through the implementation of key economic policies, e.g., the EU Biodiversity Strategy for 2030 (European Commission, 2021a), the Farm to Fork Strategy (European Commission, 2021b). The Common Agricultural Policy (CAP) is the most important EU policy for the agricultural sector to which 37.8% of the total EU budget, corresponding to 362.8 billion €, was allocated in 2014–2020 (Hristov et al., 2020). Rural Development Programmes (RDPs), the so-called 2nd Pillar of the CAP, are national or regional (e.g., in Italy) multi-annual programmes that aim at combining the economic and environmental sustainability of farming. For the 2014–2020 period, RDPs included agri-environmental payments for farmers adopting specific management practices to enhance species and habitat conservation. The RDPs also included payments to compensate for the low profitability of traditional farming in areas with natural constraints, including mountain areas, to prevent abandonment, and thus the loss of semi-natural grasslands. Also in Switzerland, several measures have been defined for the enhancement of biodiversity in semi-natural habitats and for maintaining open rural landscapes. In general, these payments are designed to promote a balanced grazing pressure since it enhances biodiversity and ecosystems functions (Schils et al., 2022). Despite their common aim, measures may vary greatly across states. In Switzerland, considerable amounts of funds have been allocated to farm-level grazing plans that account for local biodiversity and site conditions; farmers have been required to have a qualification in agriculture to access direct payments, which are result-based (i.e., grassland species composition is regularly surveyed). In contrast, in Italy, measures have been defined at a regional scale, farmers have not been required to have a specific qualification, and a result-oriented approach has not been implemented.

Despite their wide application, the effectiveness of agri-environmental payments in counteracting biodiversity decline is questioned (Kleijn et al., 2011), since they resulted, at best, in modest increases in richness or abundance of common species in intensively used areas, with scarce information on extensively managed grasslands (Batáry et al., 2015). In some cases, policy payments were even found to increase land-use intensity (e.g., by financing structural improvements facilitating the access to extensively farmed areas), causing habitat and species loss (Gubler et al., 2020; Hristov et al., 2020). This lack of success has led to criticisms of past and current agricultural policies and to the demand for innovative approaches to improve biodiversity conservation, particularly in species-rich semi-natural grasslands. Such demands are accounted for in the post-2020 CAP, which aims to improve the management of natural resources and enhance the conservation of biodiversity in agricultural systems (De Castro et al., 2021).

Sub-xerophilous *Bromus erectus* semi-natural grasslands (hereafter semi-natural grasslands) hold world records for plant species richness at fine spatial grains (Wilson et al., 2012), and host several species of conservation concern. They are widespread in the European continent (Preislerová et al., 2022) but their maintenance largely depends on extensive grazing under specific environmental conditions (Olmeda et al., 2019).

Here we focus on this habitat to assess the interactions among i) agri-environmental and marginal areas payments, ii) grazing regimes, iii) environmental conditions, iv) conservation state, and v) forage yield and quality. We gathered an original dataset across a wide latitudinal and altitudinal range in Italy and Switzerland, in areas under different environmental and socio-economic contexts. We tested the direct and indirect influence of agri-environmental payments (hereafter also including payments for marginal areas) on grazing regimes, and in turn on the habitat conservation state and forage yield and quality. We assumed a better conservation state of the habitat and higher forage yield and quality in grazed areas receiving payments (Johansen et al., 2019). We hypothesized that payments shaped the grazing regime (i.e. stocking rate, grazing system and livestock species) and, in turn, grassland composition and functions (Hristov et al., 2020). In particular, we assumed that payments favored low stocking rates and controlled grazing systems (i.e. rotational grazing systems and/or shepherding) with positive effects on the habitat conservation state and forage yield and quality (Perotti et al., 2018; Pittarello et al., 2019). We expected environmental conditions to exert substantial effects on these variables (Burrascano et al., 2013; Napoleone et al., 2021), either negative, i.e., slope and elevation on stocking rate, fine-scale grazing intensity and forage, or positive, i.e., soil carbonates on the number and cover of the habitat diagnostic species (Giarrizzo et al., 2017).

2. Materials and methods

2.1. Study area

Together with other semi-dry secondary grasslands, *Bromus erectus* semi-natural grasslands are considered a habitat of conservation concern in Europe (Habitat 6210* of EU Habitats Directive 94/93/ECC; Habitat 4.2.2/4.2.4 in the Swiss habitat inventory). Estimates for the EU report this habitat as widespread (Willner et al., 2019), about 2% of the European grasslands (Olmeda et al., 2019; Squires et al., 2018), ranging from lowlands to mountain areas, on calcareous to neutro-alkaline substrates and well-drained low nutrient soils. These grasslands are generally assigned to the phytosociological order *Brachypodietalia pinnati* of the class *Festuco-Brometea* (Mucina et al., 2016), and in the sampled areas they are generally referred to the alliances *Bromion erecti* and *Polygalo mediterraneae-Bromion erecti* (Preislerová et al., 2022).

We sampled 98 plots of 16 m^2 across 19 farms (Table 1) in nine sites (Fig. 1) across five administrative regions (Table 2), during the spring-summer of 2018 and 2019. Plots were sampled before the grazing period at the flowering phenological stage for the dominant graminoids, after checking for homogeneous vegetation composition and structure. Year-round grazed sites were sampled during the same period. Sites were selected among habitat areas representative of different environmental and management conditions and for which reliable information on payments and grazing management could be collected through direct interviews with farmers, local authorities and agricultural advisors. Farms that received constant payments for at least seven years (eight in Switzerland) and were managed similarly for at least 10 years were chosen. Although we did not have quantitative information about past grazing activities, these have been constant for a long period of time, as it is often the case in durable family-centered farms.

Since virtually all the surveyed farmers received agri-environmental payments, we had no alternative but to use previously grazed areas known to be currently unmanaged as control areas. We used aerial photographs of late eighties to identify these areas in the close vicinity of managed farms, but lying out of the area reported as grazed. Such areas were available for seven out of nine sites, and were sampled by 20 plots. Plots were randomly distributed over the grazed and ungrazed area at a minimum distance of 500 m from each other, and aiming at an even representation of the existing management regimes through an a priori GIS procedure.

2.2. Grazing management and agri-environmental payments

Grazing management regimes and agri-environmental payments were assessed for each farm. The number of heads belonging to different animal species (i.e. cattle, sheep and horses) were converted into livestock units, i.e., reference units to aggregate livestock from various

Table 1
Environmental and management features of the 19 farms within the nine study sites. The number of heads belonging to different livestock species were converted into livestock units (LU).

Site name	Site ID	Farm ID	N. plots	Elevation (m a.s.l.)		Slope (%)		Area (ha)	Stocking rate ($LU \cdot ha^{-1} \cdot year^{-1}$)	Controlled grazing (% LU)	Cattle (LU)	Sheep (LU)	Horses (LU)
				Mean	s.d.	Mean	s.d.						
Mt. Prenestini	PRE	PRE.1	4	1053.75	23.20	15.00	9.13	91	0.18	60.18	10.40	20.25	3.00
		PRE.2	4	1104.75	25.85	12.25	6.13	228	0.76	5.74	221.00	15.00	25.00
Mt. Ernici	ERN	ERN.1	3	1129.00	48.51	10.67	9.02	349	0.22	55.75	50.00	63.00	0.00
Mt. Simbruini	SIM	SIM.1	6	1411.50	26.22	15.00	5.48	400	0.31	31.19	105.60	66.00	40.00
Marsica	MAR	MAR.1	3	1056.67	51.23	30.00	8.66	137	0.59	100.00	0.00	121.50	0.00
		MAR.2	4	1290.25	70.65	17.50	9.57	259	0.14	67.68	10.00	55.50	16.50
East Gran Sasso	EGS	EGS.1	4	1520.00	14.14	1.25	2.50	251	0.37	100.00	0.00	225.00	0.00
		EGS.2	4	1585.00	12.91	3.50	4.43	265	0.08	100.00	0.00	52.50	0.00
		EGS.3	4	1702.75	80.60	16.25	11.09	430	0.09	100.00	0.00	90.00	0.00
		EGS.4	4	1612.50	20.62	10.00	5.77	84	0.10	100.00	0.00	19.50	0.00
West Gran Sasso	WGS	WGS.1	4	1359.25	73.94	15.25	11.98	227	0.27	100.00	0.00	105.00	0.00
Mugello	MUG	MUG.1	5	870.00	59.16	18.00	5.70	170	0.11	0.00	43.00	0.00	0.00
		MUG.2	6	897.50	31.27	21.67	11.69	100	0.46	100.00	110.80	0.00	0.00
Val di Susa	SUS	SUS.1	4	1685.00	179.87	9.00	8.12	201	0.36	100.00	174.60	0.00	0.00
		SUS.2	4	1894.50	21.30	14.00	5.89	323	0.30	100.00	176.80	59.40	0.00
		SUS.3	3	1613.00	58.81	13.00	4.36	47	0.15	100.00	6.00	166.95	0.00
Canton Vaud	VAU	VAU.1	6	1110.50	57.73	18.83	6.65	13	0.53	100.00	16.60	0.00	0.00
		VAU.2	3	733.00	45.97	13.67	9.81	5	0.50	100.00	60.00	0.00	0.00
		VAU.3	3	641.33	18.15	13.33	2.89	4	0.42	100.00	0.00	3.00	7.00

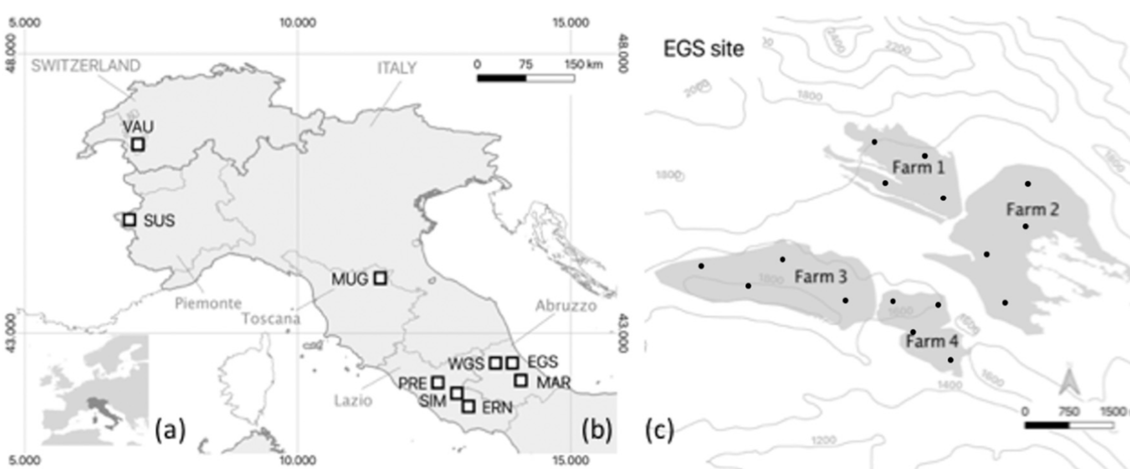

Fig. 1. Location of the study area in Europe (a) and of the three spatial scales mentioned in the study. We represented the nine study sites (site-scale) distribution in Italy and Switzerland (b) (coordinate system: WGS84 datum), and an example of the farms' (farm-scale) and plots' (plot-scale) distribution within a site (c). Site acronyms and characteristics are reported in Table 1.

species and ages by means of specific coefficients defined by the Commission Implementing Regulation (EU) No 808/2014. Then, stocking rate (*SR*), which is the main parameter commonly used to assess grazing intensity, was calculated as:

$$SR = LU/A \cdot m/12$$

where LU is the number of livestock units, A is the grazed area in hectares, and m indicates the duration of the grazing period expressed in months (Table 1).

The grazing system was calculated as the percentage of livestock units managed through controlled grazing techniques, i.e., rotational grazing systems and/or shepherding, as opposed to free ranging.

Agri-environmental payments included those aimed at biodiversity conservation and support for agriculture in marginal areas (Table 2). To prevent biases due to highly different socio-economic contexts, payments were scaled as:

$$Payments = f/lp$$

where f were the payments received from farmers per hectare, and lp was the labour productivity of the farmers in the region, i.e., the ratio between economic output to the labour input required for the production, an economic parameter closely linked with the concept of income (Federal Statistical Office of Switzerland, 2021). Data derived from Italian regional RDPs for 2014–2020, and the Swiss direct agricultural payments and Federal Statistical Office website of Switzerland for 2019 (Table 2). Since payments mostly did not bound farmers to specific management actions, but rather to a continuation of farming activities, the farming techniques could not be individually linked to payments but varied widely depending on the environmental and the socio-economic context.

2.3. *Environmental conditions and fine-scale grazing intensity*

Environmental conditions and fine-scale grazing intensity, i.e., the in-field estimate of the local grazing pressure, were measured for each plot. We recorded elevation, slope and soil carbonates level ($CaCO_3$).

Table 2

Measures and payments for semi-natural grasslands included in Rural Development Programs (RDPs) for the Italian regions, and in Swiss Direct Payments (DPs). Biodiversity measures (B) refer to measure 10 in Italian RDPs and to Biodiversity DPs in Switzerland; marginal areas measures (M) refer to measure 13 in Italian RDPs and to rural landscape DPs in Switzerland. Payments and labour productivity are expressed in €/ha and M€/AWU for Italian RDPs, and in CHF/ha and MCHF/AWU for Switzerland. AWU refers to annual working unit.

Site	Accepted measures	Region/ country	Labour productivity	Biodiversity measures		Marginal areas measures	
				Payments	Conditions	Payments	Conditions
PRE SIM ERN	M M M	Lazio	26.60	0	–	150–300	• Max 30 ha of pasture area • Stocking rate min = 0.2 $LU \cdot ha^{-1} \cdot year^{-1}$ • Increases as pasture area decreases
MAR EGS WGS	B; M B; M B; M	Abruzzo	15.21	100	• Stocking rate = 0.4–0.6 $LU \cdot ha^{-1} \cdot year^{-1}$ • Rotational grazing • Ban on pesticides and fertilizers • Removal of invasive plants	160	Stocking rate min = 0.2 $LU \cdot ha^{-1} \cdot year^{-1}$
MUG	M	Toscana	31.73	100	• Stocking rate = 0.4–0.6 $LU \cdot ha^{-1} \cdot year^{-1}$ • Ban on pesticides and fertilizers • Removal of invasive plants	30–150	Increases as pasture area decreases
SUS	B; M[a]	Piemonte	19.08	110	• Stocking rate = 0.2–0.5 $LU \cdot ha^{-1} \cdot year^{-1}$ • Rotational grazing • Grazing period >180 $days \cdot year^{-1}$ • Ban on pesticides and fertilizers • Removal of invasive plants	0–280	Increases as elevation and slope increase, and pasture area decreases
VAU	B; M	Switzerland	55.66	1350[b]	• To be registered to the national inventory on dry grasslands • Qualification in agriculture • Farm-scale contracts with specific measures • Ban on pesticides and fertilizers • Removal of invasive plants • Periodical field surveys • Occurrence of nutrient-poor pasture species	100–390 410–1000	Increases as elevation increases Increases as slope increases

[a] Two-thirds of the farmers did not receive any payment.
[b] Swiss payments are the sum of the payments received for biodiversity conservation quality level I, II and an additional fund allocated by the Canton for the management of grasslands which are registered in the national inventory of dry grasslands.

The latter was measured along a 5-classes ordinal scale based on the visible and audible reaction of a soil sample from 5 to 10 cm depth to a solution of 1-normal hydrochloric acid (1 N HCl) (Ditzler et al., 2017). The large extent of the study area hampered the possibility of monitoring fine-scale grazing intensity directly (e.g., through GPS collars for grazing animals). Thus, this variable was assessed indirectly by means of several plot-level proxies of animal grazing activity, deriving from the literature. We measured the number of dungs of different animal species not fully degraded since the previous year, which is strongly correlated with animal activities (Turner, 1998; Stumpp et al., 2005; Manthey and Peper, 2010), assuming constant patterns of livestock excreta among years (Schnyder et al., 2010) and decomposition rate across our study areas given the same habitat conditions, i.e., relatively dry and temperate climate. We measured the percentage of trampling, i.e., highly compacted and eroded soil created by livestock paths/resting areas that is related to grazing pressure through the reduction of plant cover and the degradation of soil structure (Lai and Kumar, 2020; Teague and Kreuter, 2020). We measured litter height, i.e., the average height across five randomly located points within the plot, and litter cover, i.e., visual estimate of litter percentage cover, as indicators of the degree of herbage removal by livestock (Elias et al., 2018; Mapfumo et al., 2002). In year-round grazed sites where vegetation and soil were impacted by recent animal activity, the number of fresh dungs and the cover of recent trampling and bite signs were recorded as well. Bite was expressed on an ordinal scale of values (from 0 to 3) indicating the abundance of evident signs of eaten phytomass (Orlandi et al., 2016). All the proxies were combined in a principal component analysis (PCA) to summarize the fine-scale grazing intensity ('Statistical analyses').

2.4. *Habitat conservation state and forage yield and quality*

Habitat conservation state and forage yield and quality were assessed for each plot. We recorded the occurrence and cover value of vascular plant species, as the visual estimate of the vertical projection of the species individuals on the ground, expressed through an ordinal percentage scale (Mueller-Dombois and Ellenberg, 1974). The interpretation manuals of European Union habitats (European Commission, 2013), Italian habitats (Biondi et al., 2009) and natural habitats in Switzerland (Delarze et al., 1998) were used to identify diagnostic species (Supplemental Table 1). We calculated the number of habitat diagnostic species (hereafter 'number of diagnostic species') that proved to be an effective indicator of the conservation state of *Bromus erectus* semi-natural grasslands (Carli et al., 2018), and the percentage cover of diagnostic species (hereafter 'cover of diagnostic species').

Because of the priority state of the habitat and the difficulty of carrying out direct measurements in mountain pastures (Redjadj et al., 2012), forage yield and quality were estimated through non-destructive methods, based on indirect measures of biomass productivity and forage features. Since single measurement may contain a margin of inaccuracy, as already done for fine-scale grazing intensity, we combined multiple complementary proxies. Sward height was used to proxy forage yield (e.g. Grigulis and Lavorel, 2020) by applying the sward stick method (Stewart et al., 2001) as the average value of 10 measurements of the distance from the ground to the highest photosynthetic tissues of the most abundant species, evenly distributed in the plot. We also used the Specific Leaf Area index (SLA), which is a functional trait generally correlated with greater photosynthetic capacity and productivity (Pérez-Harguindeguy et al., 2013). Forage quality was estimated by the proxy of Leaf Dry-Matter Content (LDMC), a functional trait indirectly related to forage appetence (Pauler et al., 2020), forage fiber and lignin content

(Khaled et al., 2006) and digestibility (Pontes et al., 2007; Gardarin et al., 2014; Tasset et al., 2018). Trait data were extracted from TRY database (Kattge et al., 2020) (data release date 17/12/2019). To ensure a coherent geographical context, according to the information provided by the data custodians, we selected measures collected in Italy (Chelli et al., 2019) and Switzerland and, only if these were not available, those collected in Central and Southern Europe. Only mature and healthy plants growing in natural environments were taken into account. Since the ecosystem functioning is largely determined by the trait values of dominant species ('mass ratio hypothesis', Grime, 1973), we calculated the community weighted mean (CWM) trait values for each plot considering all the herbaceous species with a relative cover higher than 5% in at least one plot. The species relative cover, i.e., the percentage ratio between the cover of each species and the total cover of all species, was assessed as:

$$SRC_i = \frac{SC_i}{\sum_i^n SC_i} \cdot 100(\%)$$

where SC_i is the cover of species *i*.

Trait CWMs were assessed as:

$$CWM = \sum_{i=1}^{S} SRC_i \cdot trait_i$$

where S is the species richness, SRC_i is the relative cover of species *i*, and $trait_i$ is the trait value of species *i* (Garnier et al., 2004).

For each plot, we calculated the Pastoral Value (*PV*), a synthetic index derived from grassland species composition summarizing forage yield, quality, and palatability for livestock, ranging from 0 to 100 (Daget and Poissonet, 1969; Pittarello et al., 2018, 2020). Each recorded species was associated to the Index of Specific Quality (ISQ_i) of Cavallero et al. (2007) and Roggero et al. (2002) which summarized information on preference, morphology, structure, and productivity of the species in different geographical contexts and was expressed with a discrete scale of increasing palatability and forage yield and quality ranging from 0 (ungrazed/toxic species) to 5 (excellent forage species) (Bagella and Roggero, 2004). Then, the PV was calculated as (Daget and Poissonet, 1971):

$$PV = \sum_{i=1}^{n} (SRC_i \cdot ISQ_i) \cdot 0.2$$

2.5. Statistical analyses

All statistical analyses were conducted in R (R Core Team, 2020). Since for unmanaged areas quantitative management information equaled zero or were lacking, the associated plots could not be included in quantitative models. The habitat conservation state and forage yield and quality of managed and unmanaged areas were compared through ridgeline plots and the significance of results was measured using the non-parametric Wilcoxon rank sum test (*wilcox.test* function, *stats* package v3.6.2).

We used a principal component analysis - PCA (*prcomp* function, *stats* package v3.6.2) to convert the *LU* per animal species per farm into a single gradient (first axis explaining 50.72% of variance) to be included in the SEM (Supplemental Fig. 1a). A second PCA was based on dungs, bite, trampling, litter height and cover, and returned a gradient of fine-scale grazing intensity (first axis explaining 47.99% of variance) (Supplemental Fig. 1b). Using a priori knowledge based on the scientific literature, we built a network of causal relationships among all measured variables (Supplemental Fig. 2a, b) and fitted two piecewise structural equation models (SEM, *piecewiseSEM* package v2.1.0, (Lefcheck, 2016)) to infer direct and indirect effects of agri-environmental payments, grazing regimes, environmental conditions on: i) number and cover of diagnostic species, ii) CWM-SLA, CWM-LDMC, PV, and sward height while accounting for site effect. Since piecewise structural equation models can handle a limited number of variables and links, we carefully selected them based on the study aims and context. Piecewise SEMs were built using linear mixed-effect models (*nlme* package v3.1.142, (Pinheiro et al., 2019)) with 'site' as random factor. Number of diagnostic species was square root-transformed, vegetation height was log-transformed, and the CWM-SLA and CWM-LDMC were reciprocal-transformed to respect assumptions of normality and homoscedasticity of the model residuals. The fit of the piecewise SEMs was evaluated using Shipley's test of d-separation through Fisher's C statistic, and the Akaike information criterion corrected for small sample size (AICc) was used to reduce the number of variables. The final optimized model was the one with statistically non-significant chi-square ($p > 0.05$) and lowest AICc with most variables included (Grace et al., 2010). We reported the standardized coefficient for each path from each component model, and the R^2 values calculated for each mixed-effect model.

3. Results

3.1. Managed vs. unmanaged

The number of diagnostic species and the proxies of forage yield and quality differed significantly between managed and unmanaged grasslands (Fig. 3). Specifically, in managed areas the number of diagnostic species, CWM-SLA and pastoral value were higher, while CWM-LDMC and sward height lower. Although generally higher in managed areas, the cover of diagnostic species did not vary significantly between the two management groups since the spread of *Brachypodium rupestre* maintained a high occurrence of the habitat diagnostic species in unmanaged areas.

3.2. Habitat conservation state

The number and cover of diagnostic species were positively correlated and were both directly and positively affected by agri-environmental payments (Fig. 2a). The positive effect of payments on conservation state was also exerted indirectly, through their negative effect on fine-scale grazing intensity. The number of diagnostic species was also positively influenced by cattle dominance, elevation and soil carbonate level, whereas the effect of slope was exerted indirectly by limiting fine-scale grazing intensity.

3.3. Forage yield and quality

Overall, policy, environmental and management variables impacted only weakly the CWM-SLA and CWM-LDMC, sward height and pastoral value (Fig. 2b). As expected, the CWM-LDMC and CWM-SLA were negatively correlated to each other. CWM-LDMC was directly and positively affected by the stocking rate, and indirectly and negatively affected by payments and slope through reduction of fine-scale grazing intensity.

4. Discussion

4.1. Habitat conservation and forage value depends on grazing management

Agri-environmental payments were a necessary condition to maintain grazing management and turned out as relevant for the conservation of the habitat and of the forage yield and quality. In unmanaged areas coarse grasses and woody species encroachment phased out several habitat diagnostic species, although their cover was kept relatively constant by the spread of *Brachypodium rupestre*. Although being listed among habitat diagnostic species, this graminoid species has a key role in successional dynamics and is recognized as a threat to the habitat species diversity (Bonanomi et al., 2006, 2009; Catorci et al., 2011), and forage value (Vitasović Kosić et al., 2014). Also *Juniperus communis* was more abundant in unmanaged areas, and its needles, cones and wood

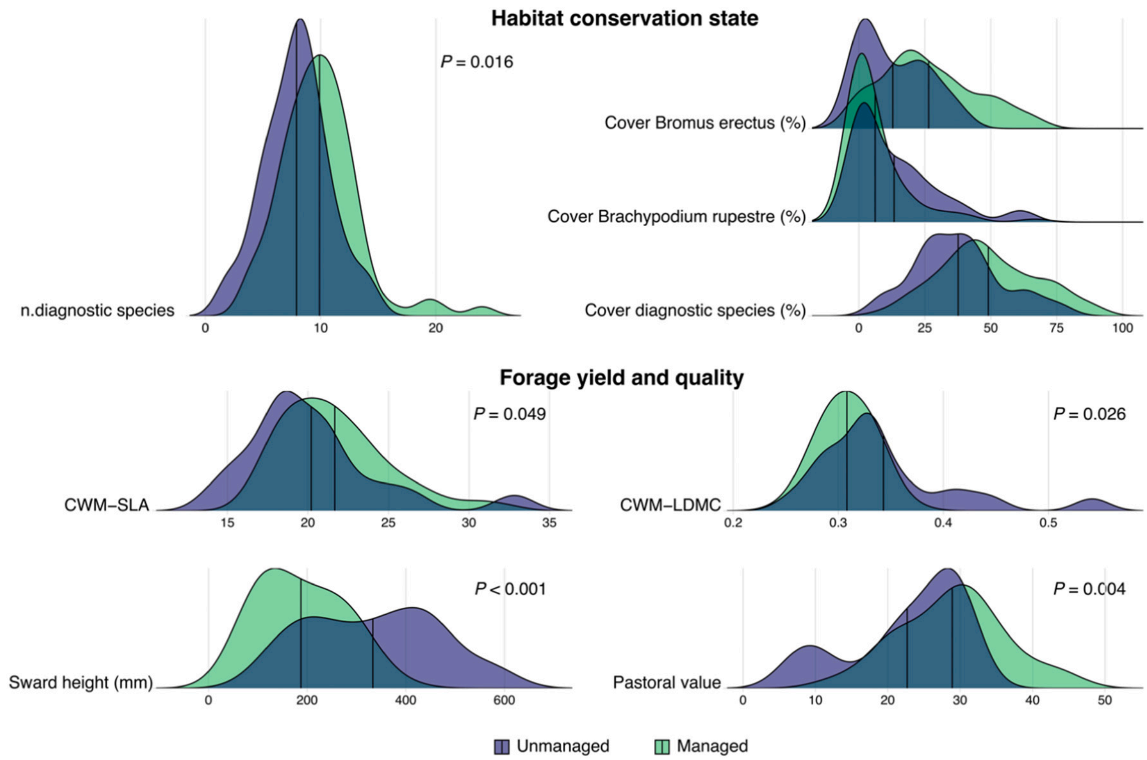

Fig. 2. Ridgeline plots and post -hoc Wilcoxon tests ($p < 0.05$) comparing proxies of habitat conservation state and forage yield and quality in managed and unmanaged plots.

contain oils that combined with the species spinescent needles make the plant virtually unpalatable (Thomas et al., 2007). Although the role of grazing in preserving biodiversity is currently debated (Queiroz et al., 2014), our results were in line with studies asserting that maintaining appropriate management practices are crucial to preserve the biodiversity and functions of semi-natural grasslands (Bengtsson et al., 2019; Johansen et al., 2019; Schils et al., 2022). In this context, promoting adequate agri-environmental policies represents one of the major tools for conservation-oriented management of mountain pastures.

4.2. Agri-environmental payments affect fine-scale grazing intensity rather than stocking rates

Higher agri-environmental payments did not affect farm-scale stocking rate as we hypothesized, but they rather resulted in lower fine-scale grazing intensity, suggesting that farmers receiving higher payments acted towards an even distribution of grazing pressure. At the sites where payments were higher (i.e., Switzerland), they were not bound to a given regional stocking rate, but to farm-scale grazing contracts, whose implementation was verified through periodic field surveys. Furthermore, higher payments allowed investments in grazing management infrastructures (e.g., increasing the number of water troughs) that may have promoted an even livestock distribution.

Our findings are in line with criticisms blaming the RDPs for the lack of target-oriented measures for habitats and species, and for the inability to prevent intensification and abandonment of agricultural practices (Pe'er et al., 2014, 2019, 2020). The efficacy of agri-environmental payments was already demonstrated to be highly context-specific, depending on climate, species pool, landscape configuration, and land-use intensity (Merckx et al., 2009; Tscharntke et al., 2012). Hence, agri-environmental payments should be designed as locally as possible to improve habitat conservation state. Among the proposals to implement the CAP, our results strongly support those suggesting that RDP payments should not depend on farm size, but rather on clear and measurable context-specific targets that contribute to biodiversity conservation. In this regard, the support to farmers with ecological expertise should be increased and account for context-specific biodiversity and farm conditions, as it is for instance in Switzerland (Chevillat et al., 2017; Ravetto Enri et al., 2020). This confirms the effectiveness of farm-scale grazing management plans to enhance pasture yield and quality while increasing plant diversity in species-rich extensive mountain pastures (Probo et al., 2014; Perotti et al., 2018; Pittarello et al., 2019).

4.3. Habitat biodiversity and forage yield and quality depend primarily on fine-scale grazing and environmental conditions

Except for CWM-LDMC, which increased with stocking rates, both diagnostic species and forage indicators were influenced by fine-scale grazing intensity. According to the 'Intermediate disturbance hypothesis' (Grime, 1973; Connell, 1978) and the subsequent 'land use-moderated conservation effectiveness hypothesis' (Kleijn and Sutherland, 2003; Kleijn et al., 2011), low rates of disturbance decrease interspecific competition and allow many species to occupy different available niches, increasing the species diversity of semi-natural grasslands. Moderate grazing pressure ensures a sustainable consumption of forage and a balanced provision of nutrients, maintaining a relevant proportion of palatable species available for livestock. Conversely, overgrazing and undergrazing both lead to a loss of desirable species as forage, and in turn to a decrease in forage quality (Pittarello et al., 2019, 2020). Under heavy stocking rate, LDMC increased whereas SLA decreased, likely in relation to the increase of species with leaves more resistant to herbivory and trampling (Pontes et al., 2007; Targetti et al., 2013; Gardarin et al., 2014; Tasset et al., 2018; Pauler et al., 2020).

While many studies suggest that controlled grazing techniques (i.e., rotational grazing systems and/or shepherding) promote higher levels of biodiversity and enhance productivity (Jacobo et al., 2006; Ravetto Enri et al., 2017; Perotti et al., 2018; Pittarello et al., 2019), this was not the case in our study. In contrast to our original hypothesis, controlled grazing systems did not affect our proxies of habitat conservation state

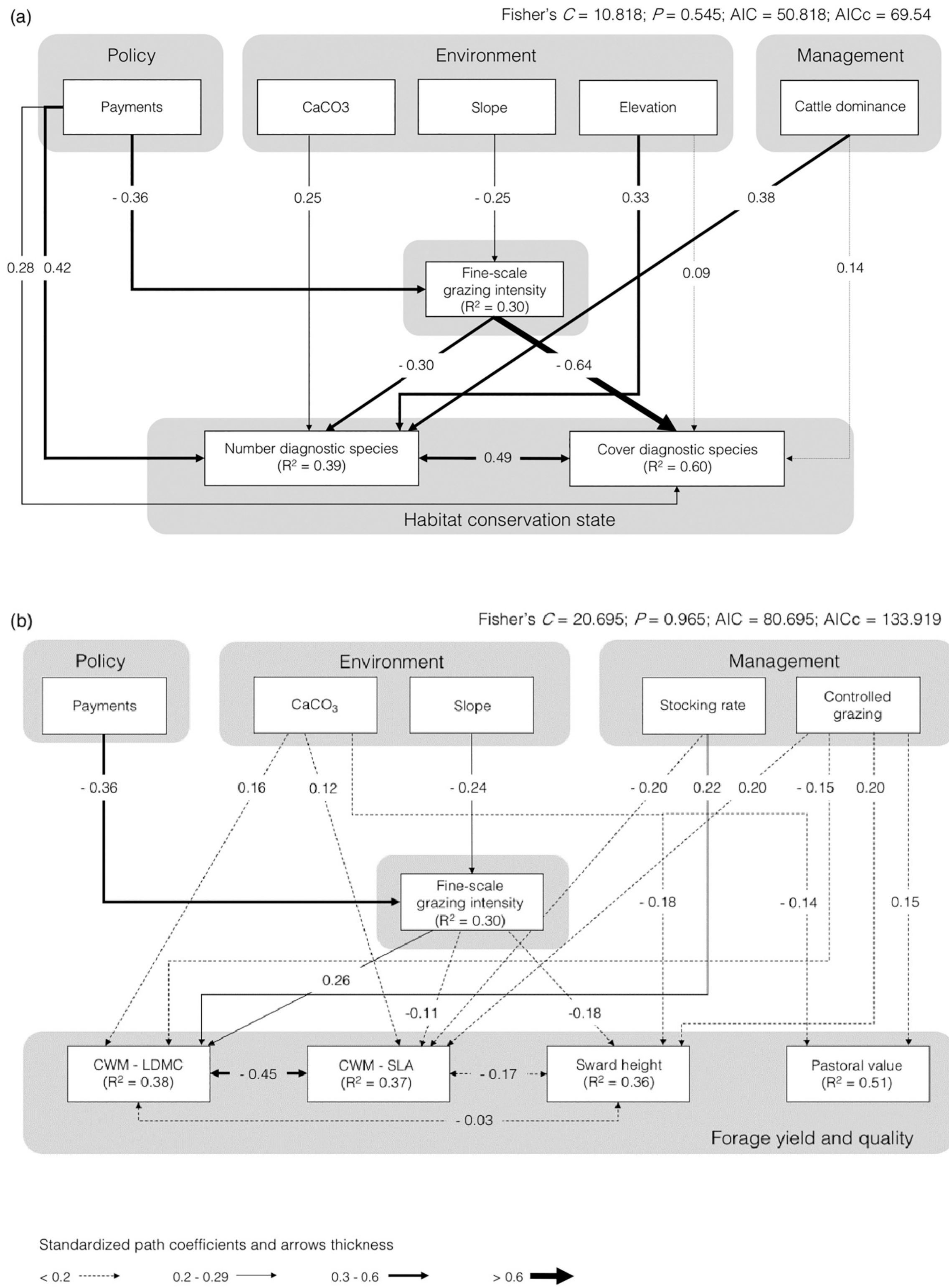

Fig. 3. Structural equation models (SEM) with policy, management and environmental variables as predictors of (a) number and cover of diagnostic species and (b) community-weighted mean (CWM) traits value of specific leaf area (SLA) and leaf dry matter content (LDMC), sward height and pastoral value. Solid arrows represent significant paths ($p < 0.05$), dashed arrows represent non-significant ones ($p > 0.05$). Path coefficients are standardized. Fisher's C statistic, Akaike information criterion (AIC) and corrected AIC (AICc) are reported in the upper part of the graphs.

and had weak positive effects on forage yield and quality. This may derive from the generally low grazing pressure at our study sites that may had limited the effect of these techniques on the use of resources by grazing animals.

Interestingly, livestock species had an important role in preserving the habitat since grasslands mainly grazed by cattle were associated with a higher number of diagnostic species. Indeed, cattle use their tongue to sever forage and are therefore less selective than sheep and horses, which graze closer to the ground with narrower mouths and more flexible lips (Calaciura and Spinelli, 2008). This also implies that cattle preferably feed on large clumps of dominant high grasses, causing a decrease in interspecific competition, and allowing for the coexistence of several habitat diagnostic species.

As expected, environmental features had an important role in determining habitat conservation state and forage yield and quality (Argenti et al., 2020; Pittarello et al., 2020). The number of diagnostic species increased with elevation, since it facilitates the competitive exclusion of thermophilous and ruderal species (Sundqvist et al., 2013). The number of diagnostic species also increased with carbonate content in line with the definition of the habitat (Bohner et al., 2019) and the theory that the pool of the Central European grassland flora is strictly linked to very base-rich and calcareous soils (Ewald, 2003). Finally, forage quality was favored by well-drained steep slopes, where fine-scale grazing intensity was reduced. Our results were in line with previous studies on Apennine semi-natural grasslands that found the greatest deviation from the original habitat species composition at low elevation and flat sites (Giarrizzo et al., 2017).

4.4. Study limitations and further research needs

Although we showed that a low grazing intensity is desirable for the plot-scale habitat conservation state, whether this is in line with the vegetation carrying capacity should be investigated through direct measures of stocking rates and non-linear statistical methods complying with the 'Intermediate disturbance hypothesis' (Grime, 1973; Connell, 1978).

Our ability to highlight the full range of management effects on forage, however, may have been hampered by its indirect assessment through species-level functional traits rather than by individual-based local measurements and by the use of proxies for fine-scale management intensity. On the one hand, the wide extent of our study area hampered a detailed analysis of animal movements and may have determined a certain margin of inaccuracy in fine-scale grazing patterns, which was however partly limited by the combination of multiple variables. On the other hand, the broad scale of our study, allowed us to highlight differences related to different payment schemes that may inform future policies, which local studies fail to detect. The same is true for using a single animal species gradient rather than distinguishing the individual effects of their different grazing behaviors. However, we can state that horses occurred in low numbers and only in few farms, and that the gradient we investigated is mainly a cattle-sheep gradient.

It should also be taken into account that the direct positive effect of agri-environmental payments on the habitat diagnostic species can be interpreted as the result of payment-influenced factors other than those we measured in our study. Indeed, farmers received payments for the adoption of further management practices designed to improve the conservation value of pastures (e.g., grazing period, shrub removal) that we did not include in our analyses. Overall, long-lasting programs may result in long-term positive outcomes by increasing farmers' awareness of environmental priorities and reshaping management habits towards more sustainable practices (Riley, 2016), although the contribution of management activities of previous decades was not assessed. Following an inverse line of thought, the influence of payments may derive from an inverse effect due to areas with greater relevance for biodiversity conservation receiving higher payments, e.g., payments for biodiversity mainly targeted farms in areas with a favorable habitat conservation state, either assessed through the occurrence of indicator species (in Switzerland), or recognized from site descriptions and protection status (in Italy, e.g., the areas within the Gran Sasso National Park). While interesting, this inverse effect could not be verified directly in the model since piecewise SEM cannot disentangle cyclic relationships (Lefcheck, 2016).

5. Conclusions

Our findings indicate that the spatial scale at which agricultural measures are designed and applied is crucial for the conservation of *Bromus erectus* semi-natural grasslands. Management guidelines should be implemented as locally as possible to account for the distinctive features of stakeholders as well as for local environmental conditions. CAP should progressively increase the application of result-based agri-environmental payments (e.g., the persistence of certain species typical for relevant habitats) defined at the farm level and assessed through periodic field-surveys. A wide implementation of these systems could provide direct control of the payment environmental impact, and a win-win increase of the environmental awareness of farmers who would progressively take up voluntary measures to support biodiversity and ecosystem services (Russi et al., 2016). This would go in the direction of the 'new architecture' of the post-2020 CAP, through the New Delivery Model (NDM) which facilitates the transition from a rule-focused to a result-oriented approach, and implies diverse national measure within a common general framework of biodiversity protection and provision of ecosystem services (De Castro et al., 2021).

CRediT authorship contribution statement

Francesca Napoleone: Conceptualization, Methodology, Investigation, Formal analysis, Writing - original draft. **Massimiliano Probo**: Methodology, Investigation, Formal analysis, Writing - original draft. **Pierre Mariotte**: Formal analysis, Writing - Review & Editing. **Simone Ravetto Enri**: Investigation, Writing - Review & Editing. **Michele Lonati**: Investigation, Writing - Review & Editing. **Giovanni Argenti**: Methodology, Investigation, Writing - Review & Editing. **Sabina Burrascano**: Conceptualization, Methodology, Investigation, Formal analysis, Writing - original draft.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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