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Short-Term Effects of Contrasting Management Regimes on Plant Species Diversity, Herbage Production and Soil CO_2 Emissions in Species-Rich Grasslands

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Received: 21 January 2025 | **Revised:** 1 November 2025 | **Accepted:** 20 July 2026

Keywords: above-ground biomass | biodiversity | *Bromus erectus* | greenhouse gas | mowing | soil organic carbon

ABSTRACT

Semi-natural grasslands are among the most plant-rich ecosystems worldwide, providing multiple ecosystem services but requiring site-specific management for their conservation. This study assessed short-term dynamics of plant biodiversity, herbage production, pastoral value and soil CO_2 emissions under contrasting management regimes in *Bromopsis erecta*-dominated grasslands (2016–2018). A randomised complete block design tested three regimes: customary management (CST, two cuts per year), intensive utilisation (INT, four to six cuts per year) and abandonment (ABN, no cuts). Changes in species composition relative to CST were mainly associated with ABN, which increased the cover of *Brachypodium rupestre*, *Dactylis glomerata* and *Cerastium arvense*, while leading to a general decline in species associated with mowing. Diversity indices showed significant interactions with year and management: target species number differed only in the third year between INT and ABN, while richness and H′-index were lower in INT than in CST and ABN in the first year, reversing by the last year. Herbage production in ABN reached 6.8 t dry matter ha^{-1} in the third year (+180% vs. CST; +117% vs. INT), whereas pastoral value ranged broadly (13.4–57.1) with no significant differences among regimes. Cumulative soil CO_2 emissions were highest in the first year (31.12 t CO_2 ha^{-1} $year^{-1}$) and declined to 17.5–19.3 t CO_2 ha^{-1} $year^{-1}$ in subsequent years, without significant management effects. Overall, short-term intensive utilisation does not negatively affect herbage production or soil CO_2 emissions, although slight increases in annual mesophilic species may signal a shift towards nutrient-rich communities. Even short-term abandonment shows no major trade-offs, suggesting flexible management can support both biodiversity and farmer livelihoods while maintaining key ecosystem services.

1 | Introduction

Grasslands are among the most widespread ecosystems globally, covering more than a third of the Earth's surface and present on every continent except Antarctica (Blair et al. 2014; Milazzo et al. 2023). Among grasslands, semi-natural ones occur in areas that were once natural forests cleared of trees. They have been maintained through traditional practices such as mowing and grazing since the onset of agriculture (Hejcman et al. 2013). Semi-natural grasslands are considered among the most plant-rich ecosystems in the world (Wilson et al. 2012) and in particular those found on calcareous substrates (Dengler et al. 2020). These grasslands, present across almost the entire European continent, are among the most species-rich plant communities in Europe and host a large number of rare and endangered species (Habel et al. 2013) and their high ecological value is recognised by the European Union through the designation of specific and priority habitat type 6210* 'semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (*important orchid sites)' (EU Habitats Directive 92/42/EEC Annex I).

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Semi-natural grasslands, with their high plant and animal biodiversity, provide a wide range of provisioning, regulating and cultural ecosystem services, including forage and livestock product quality, water and soil regulation, carbon sequestration, wildfire mitigation and preservation of traditional cultural landscapes (Dean et al. 2024; D'Ottavio et al. 2018; Labadessa et al. 2023). However, semi-natural grasslands have undergone decades of biodiversity loss due to the abandonment of traditional management practices (Tälle et al. 2018), driving vegetation changes (Allegrezza et al. 2016), altering historical landscapes (Tesei et al. 2025) and highlighting the need for site-specific management tailored to local ecological and management conditions to conserve their biodiversity (Toderi et al. 2017).

The long-standing management of semi-natural grasslands encompasses a range of regimes, such as grazing, mowing and fertilisation, applied at varying intensities, each exerting distinct influences on the plant species diversity (Lanta et al. 2009; Melts et al. 2018; Tesei et al. 2020), herbage production and quality (Castillo-Garcia et al. 2022; Scotton et al. 2014; Stybnarova et al. 2016) and soil carbon cycle (D'Ottavio et al. 2023; Li et al. 2024; Liu et al. 2023). However, the progressive abandonment of semi-natural grasslands, which is already occurring in various locations across Europe, is leading to vegetation dynamics with short-term increases in forage availability but a decline in plant biodiversity (Bonanomi et al. 2013; Tesei et al. 2020). In this case, the increase in above-ground biomass could correspond to a rise in root activity and potentially higher CO_2 emissions (Francioni, D'Ottavio, et al. 2019; Francioni, Trozzo, et al. 2019). Among the few farmers who still manage semi-natural grasslands through grazing and/or mowing, some may seek to adopt temporary intensive use beyond what is permitted under regulations that can be overly restrictive and insufficiently aligned with the current context of climate change (Toderi et al. 2017). In such cases, more intensive use than customary management may trigger short-term shifts in floristic composition and modify herbage production. These changes can alter nutrient allocation patterns, reducing root activity and consequently lowering soil CO_2 emissions. Such emissions result from the sum of autotrophic respiration (roots) and heterotrophic respiration (soil microorganisms responsible for organic matter mineralisation) (D'Ottavio et al. 2023; Francioni, D'Ottavio, et al. 2019; Francioni, Trozzo, et al. 2019).

Although several studies have examined the effects of either abandonment or management intensification of semi-natural grasslands on multiple ecosystem components (i.e., plant diversity, herbage yield and quality and soil CO_2 emissions), these drivers have rarely been investigated in combination, leaving open the question of potential trade-offs. Despite the high ecological value of *Bromopsis erecta*-dominated grasslands for biodiversity conservation and their crucial role in sustaining local economic activities, comprehensive evaluations in these systems remain scarce.

This study aims to assess the short-term effects of both intensive use and abandonment, compared with customary management, on a *B. erecta*-dominated grassland. Specifically, it examines changes in plant species diversity, herbage production, pastoral value and soil CO_2 emissions. It is hypothesised that (i) an intensive herbage use would not reduce forage quality or induce shifts in floristic composition, leaving soil CO_2 emissions unaffected and (ii) abandonment may increase above-ground biomass but decrease forage quality and plant diversity, while potentially enhancing soil CO_2 emissions through increased root activity.

With the ultimate goal of providing guidance to protected area managers and policymakers, this study aims to explore whether semi-natural grasslands—commonly found in these areas—can be managed under less restrictive regulations (i.e., intensive use) while still supporting economic activities and conserving priority habitats.

2 | Materials and Methods

2.1 | Study Area

The study area is located in Monte San Vicino (Central Apennines, Marche Region, Italy) within Natura 2000 Special Protection Area IT5330025 'Monte San Vicino e Monte Canfaito'. The bedrock of the study area is calcareous, and the climate is temperate oceanic, with a mean annual temperature of 13°C and an average annual precipitation of 865 mm, peaking in autumn and spring with a minimum in summer (Allegrezza 2003). The monthly mean air temperature and precipitation during the study period (January 2016–December 2018) and long-term period (1982–2012) are shown in Figure 1. Precipitation observed from June–September 2016 (374.2 mm) was approximately double that recorded in 2017 (191.8 mm) and 2018 (197.8 mm).

2.2 | Study Site

The soil at the study site is classified as Mollisol according to the USDA Soil Taxonomy system (Soil Survey Staff 2014). The soil texture at 0–20 cm depth is clay loam, with 7% gravel, a pH of 6.79, a total organic C of 5.92% and a total N content of 0.53%. It has a field capacity of 46.2% and a wilting point of 32.1%. The site is characterised as a species-rich semi-natural *B. erecta*-dominated grassland, comprising the following species: *Bromus hordeaceus, Centaurea ambigua, Cynosurus cristatus, Festuca circummediterranea, Lolium perenne, Plantago lanceolata, Poterium sanguisorba* subsp. *balearicum, Rhinanthus minor, Teucrium chamaedrys, Trifolium incarnatum* subsp. *molinierii* and others. This grassland belongs to the association *Brizo mediae–Brometum erecti* Bruno in Bruno et Covarelli 1968 corr. Biondi et Ballelli 1982 and is included in priority habitat code 6210* 'Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco-Brometalia*) (*important orchid sites)' (Council Directive 92/43/EEC) (Biondi et al. 2009). The investigated grassland has been managed through mowing and aftermath grazing, a practice commonly observed in most similar calcareous semi-natural grasslands of the Central Apennines (Caballero et al. 2009). During the last 30 years, the management in the study site comprised cutting at the end of June–beginning of July and subsequent grazing of the regrowth grass by sheep and cattle until the end of October. The hay obtained is used by the same

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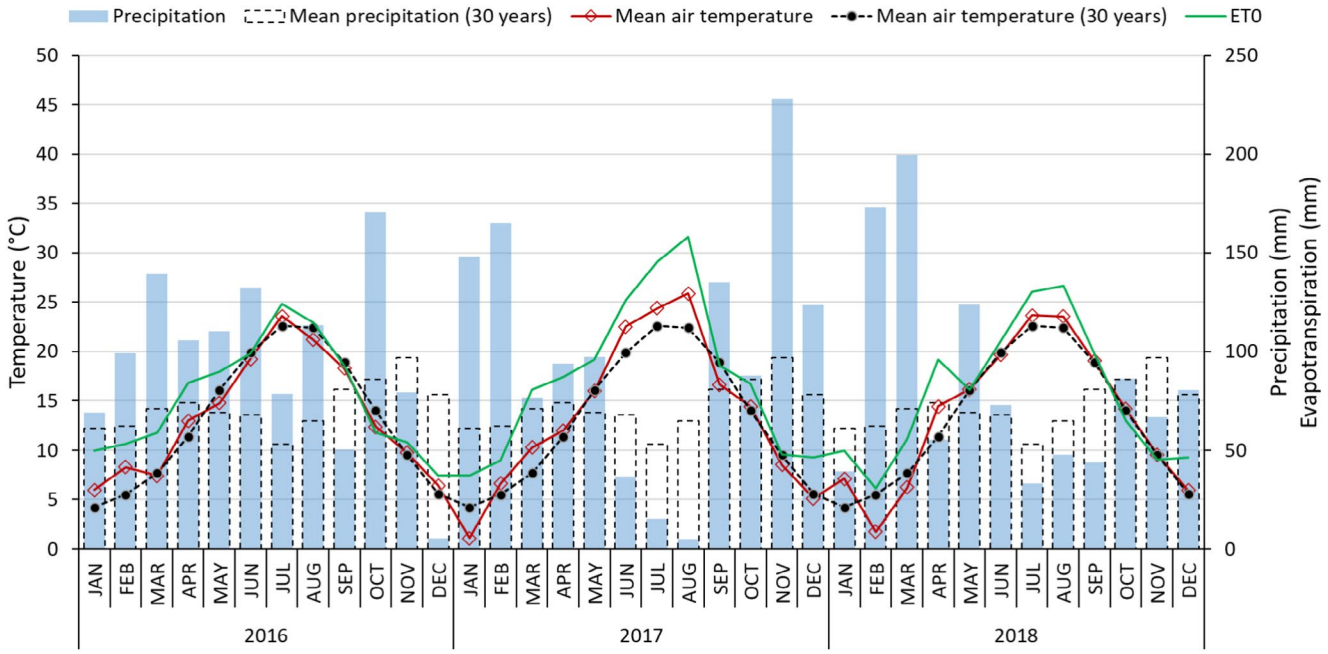


FIGURE 1 | Monthly mean air temperature and precipitation during the monitoring (January 2016–December 2018) and long-term (1982–2012) periods for Poggio San Vicino (Macerata province, Italy). The monthly cumulative reference evapotranspiration (ET_0) was calculated using the FAO Penman-Monteith method. *Source:* Sistema Informativo Regionale Meteo-Idro-Pluviometrico, Servizio Protezione Civile, Regione Marche.

farm to feed livestock during the winter non-grazing period. The estimated stocking rate in the study area is approximately 0.2 livestock units ha^{-1} for a grazing period of 120 days.

2.3 | Experimental Design

At the study site, an area of approximately 20 m × 20 m with homogeneous soil, vegetation, and topographic conditions (N 43°21′13.4″, E 13°03′31.6″; 900 m a.s.l.; NE aspect; 10% slope) was identified and fenced off to prevent disturbance in November 2015.

Three contrasting management regimes were applied to 2.0 m × 2.0 m plots using a randomised complete block design with three replicates from 2016 to 2018: (i) Customary management (CST), where herbage cutting and removal were performed during the first week of July and by the end of October each year (control, business-as-usual treatment); (ii) Abandonment (ABN), where no herbage removal was carried out throughout the study period, and (iii) Intensive utilisation (INT), where continuous cutting and removal of herbage were conducted from April until the end of October each year, whenever herbage production exceeded 0.5 Mg ha^{-1} of dry matter (i.e., six, four, and four times in 2016, 2017 and 2018, respectively). Herbage was cut using a bar mower set at a cutting height of 5 cm. To avoid potential mulching effects, the cut herbage from both INT and CST was immediately collected and removed after each cut.

The use of mowing rather than grazing is acknowledged as a potential limitation, since grazing selectively affects floristic composition, while mowing removes all species uniformly, including unpalatable ones. However, given the small plot size and the need to control spatial variability, mowing was chosen as the most practical method for biomass removal.

2.4 | Data Collection and Analysis

2.4.1 | Plant Species Diversity

Vegetation surveys were conducted each year in each experimental unit before the first herbage mowing (i.e., in April for INT and in June for CST and ABN). Data were collected from a 1.0 m × 1.0 m quadrat that was randomly located within each plot. For each quadrat, a list of rooted vascular plant species was compiled and their abundance was estimated according to Braun-Blanquet (1928). All species were identified according to taxonomic nomenclature, following the Italian Flora Checklist (Bartolucci et al. 2018). Plant species diversity in each quadrat was estimated using the number of target species of habitat code 6210*, species richness and Shannon diversity index (H′).

The target species associated with habitat code 6210* (Table S1) were determined according to the Italian Interpretation Manual of Directive 92/43/EEC Habitats (Biondi et al. 2009). *Brachypodium rupestre* was excluded from the list of target species because it is associated with progressive vegetation dynamics following grassland underuse or abandonment (Angelini et al. 2016; Allegrezza et al. 2016). H′ was calculated according to the Equation (1):

$$H' = -\sum_{i=1}^{i=n} p_i \log_2 p_i \quad (1)$$

where p_i is the proportional abundance of species i.

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2.4.2 | Herbage Production and Pastoral Value

Each year, herbage dry matter production was assessed at monthly intervals from April to October for INT, twice per year (i.e., at the end of June and October) for CST and at the end of October for ABN. On each sampling date, the above-ground biomass was harvested from a 0.5 m × 0.5 m quadrat, randomly placed in each plot. To account for the cumulative effect of no herbage use throughout the study period for ABN, production assessments were conducted in different subplots each year. Herbage samples were oven-dried at 65°C to a constant weight. Annual cumulative dry matter production was calculated by summing the herbage yield recorded on each sampling date, excluding periods with no herbage growth, typically observed during summer (primarily from July to September) in Mediterranean mountain grasslands (Caballero et al. 2009; Cavallero et al. 1992).

The pastoral value (Equation 2) of the grassland was assessed using the data collected from vegetation surveys (Section 2.4.1) and the specific indices (*Si*: 0–5) reported by Roggero et al. (2002), which indicate the forage value of each species contributing to CSP_i (i.e., the percentage contribution of each species to plant cover abundance) (Table S1), according to Daget and Poissonet (1971).

$$\text{Pastoral value} = 0.2 \sum_{i=1}^{i=n} CSP_i \times Si_i \quad (2)$$

where CSP_i was calculated as the percentage ratio between the species abundance assigned to each species and the sum of the abundances of all species, multiplied by the total plant cover.

2.4.3 | Soil CO_2 Emissions

The cumulative soil CO_2 efflux over the study period (April–November in 2016, 2017 and 2018) was calculated using linear interpolation and numerical integration via the trapezoidal rule between consecutive sampling dates (Toderi et al. 2022). The data used for the calculation were obtained from Francioni, D'Ottavio, et al. (2019) and Francioni, Trozzo, et al. (2019) and an excerpt is available in Figure S1. During the same study period, soil CO_2 efflux was measured at 2–3-week intervals in each experimental plot using a portable infrared gas analyser equipped with a closed chamber (EGM-4; PP-Systems, Hitchin, UK).

2.5 | Statistical Analysis

To test and visualise the effects of the management regimes over time, vascular plant composition data were analysed using the principal response curve (PRC) multivariate method (van den Brink and Ter Braak 1999). This method is based on the partial redundancy analysis (RDA) ordination technique, a constrained form of principal component analysis (PCA). PRC allows for the partitioning of variation (Borcard et al. 1992) into variance attributed to time (where sampling time is treated as a covariate) and variance attributed to management, with interactions between sampling time and management treated as explanatory variables. In the PRC diagram, the *Y*-axis displays the first principal component of the variance explained by management differences over time, while the *X*-axis indicates time (sampling dates). Additionally, a secondary *Y*-axis (on the right) represents species as variables, showing each species' affinity for the management effect depicted in the PRC diagram (van den Brink et al. 2009). The significance of the PRC diagrams was tested using Monte Carlo permutation tests (9999 permutations). The 'surveys × species cover-abundance values' matrix was first converted to the van der Maarel (1979) ordinal scale and was then Hellinger-transformed, whereas plant diversity was standardised. For the statistical analysis of PRC, the 'vegan' package (Oksanen et al. 2022) of the R software (R Core Team 2020) was used.

Plant diversity indicators (the number of target species of 6210* habitat, species richness and H′), herbage production, pastoral value, soil organic carbon, and cumulative soil CO_2 emissions were analysed using a two-way repeated measures ANOVA to assess the effect of time (year of monitoring), management (i.e., INT, CST and ABN) and their interaction over the study period. Subsequently, a one-way repeated measures ANOVA was conducted within each management regime. Tukey's HSD test was used to separate means of all the studied variables. Data were tested for normal distribution using Shapiro–Wilk's test. Homoscedasticity was tested using Levene's test and sphericity in repeated measures was tested using Mauchly's test. Data were Box–Cox transformed to meet all previous assumptions. ANOVA and post hoc tests were performed using SPSS Statistics (version 25.0; SPSS Inc., IBM, Chicago, IL, USA).

To explore the relationships between plant composition in the plots and the associated variables, the 'surveys × species cover-abundance values' matrix was first converted to the ordinal scale described by van der Maarel (1979) and subjected to Hellinger transformation. Subsequently, a PCA, a linear unconstrained ordination method, was conducted. The variables were then used in a multiple regression analysis, with the first two ordination axes as explanatory variables. The regression coefficients obtained were normalised and multiplied by the coefficient of determination (R^2) to create vectors plotted on the ordination diagram. To test the significance of the relationship between the variables and the ordination axes, a Monte Carlo permutation test was used. For this analysis, the 'vegan' package (Oksanen et al. 2022) of the R software (R Core Team 2020) was used.

3 | Results

3.1 | Plant Community

The PRC analysis attributed 15% of the total variation to time (i.e., year of monitoring) and 30% to management; 43% of the management-related variation was explained by the PRC1 axis; $p = 0.0004$, (Figure 2). Compared with CST, INT did not show any significant effect on vegetation in any of the three study years, with only a slight response detected in 2017 and 2018 (treatment interaction coefficients ranged from −0.001 in 2016 to 0.003 in 2018). In contrast, ABN differed from CST already in 2016, with a progressive divergence in 2017 and the greatest separation observed in 2018 (treatment interaction coefficients ranged from

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−0.12 in 2016 to −0.33 in 2018) (Figure 2). Changes in species composition relative to CST were mainly associated with ABN, particularly through an increase in the cover of *B. rupestre, Dactylis glomerata* and *Cerastium arvense* subsp. *suffruticosum*, with a lesser effect on *B. erecta*, *Vicia angustifolia* and *Galium verum*. In contrast, ABN was associated with reduced cover of several species, including *Trifolium striatum* subsp. *striatum, R. minor, T. incarnatum* subsp. *molinerii, Cynosurus echinatus, P. lanceolata, Ranunculus neapolitanus, Galium corrudifolium, Taraxacum officinale, Linum usitatissimum* subsp. *angustifolium, Ranunculus acris* and *C. cristatus*.

3.2 | Plant Diversity, Herbage Production, Pastoral Value and Soil CO_2 Emissions

A significant interaction between management regime and monitoring year was observed for all diversity indices (Table 1). In the first monitoring year, both richness and the H′-index showed lower values in INT compared with CST, whereas the number of target species in the 6210* habitat did not differ among management regimes (Figure 3A–C). In the second monitoring year, no differences emerged among management regimes. In the third monitoring year, the number of target species, species richness, and the H′-index were significantly higher in INT than in ABN, while CST showed intermediate values and did not differ significantly from either treatment.

A significant interaction between management regime and year emerged for herbage production but not for pastoral value (Table 1). In the first and second monitoring years, no differences were detected among management regimes (Figure 3D,E), although herbage production in ABN showed a marked increase from the second year onwards (Figure 3D), which, however, did not reach statistical significance

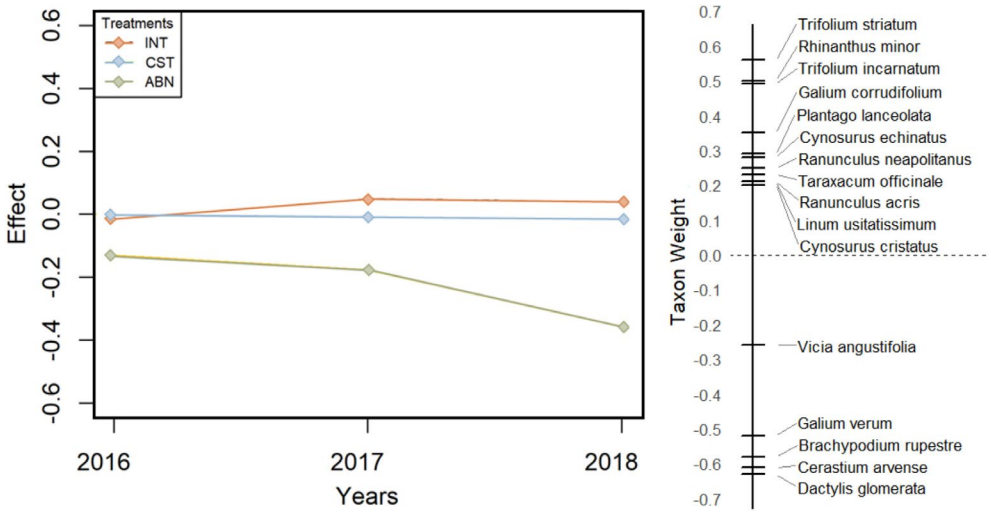


FIGURE 2 | Principal response curve diagram of the treated vegetation data from 2016 to 2018. Approximately 15% and 30% of the total variation in the vegetation were attributed to time and treatment, respectively. On the right the species are ordered by their weights in the PRC. Positive weights (top) indicate species that increase where PRC scores are high, negative weights indicate species that increase where PRC scores are low and weights near zero indicate little association with the PRC pattern. Treatments: intensive utilisation (INT), customary management (CST) and abandonment (ABN).

TABLE 1 | Two-way repeated measures ANOVA results for the studied variables.

Source	DF	TS	Richness	H′	DM	PV	Soil CO_2
Within-subject effects							
Year	2	0.712	0.009	0.007	0.022	0.005	0.001
Year × management	4	0.035	0.001	0.001	0.001	0.095	0.110
Error (*A*)	8						
Between-subject effects							
Block	2	0.233	0.656	0.667	0.719	0.122	0.793
Management	2	0.829	0.687	0.715	0.066	0.366	0.081
Error (*B*)	4						

Note: Degrees of freedom (df) are reported in the first column, while the *p*-values are presented in the other columns. The complete ANOVA tables are available in Tables S2–S4.
Abbreviations: DM = dry matter (herbage production); PV = pastoral value; TS = target species.

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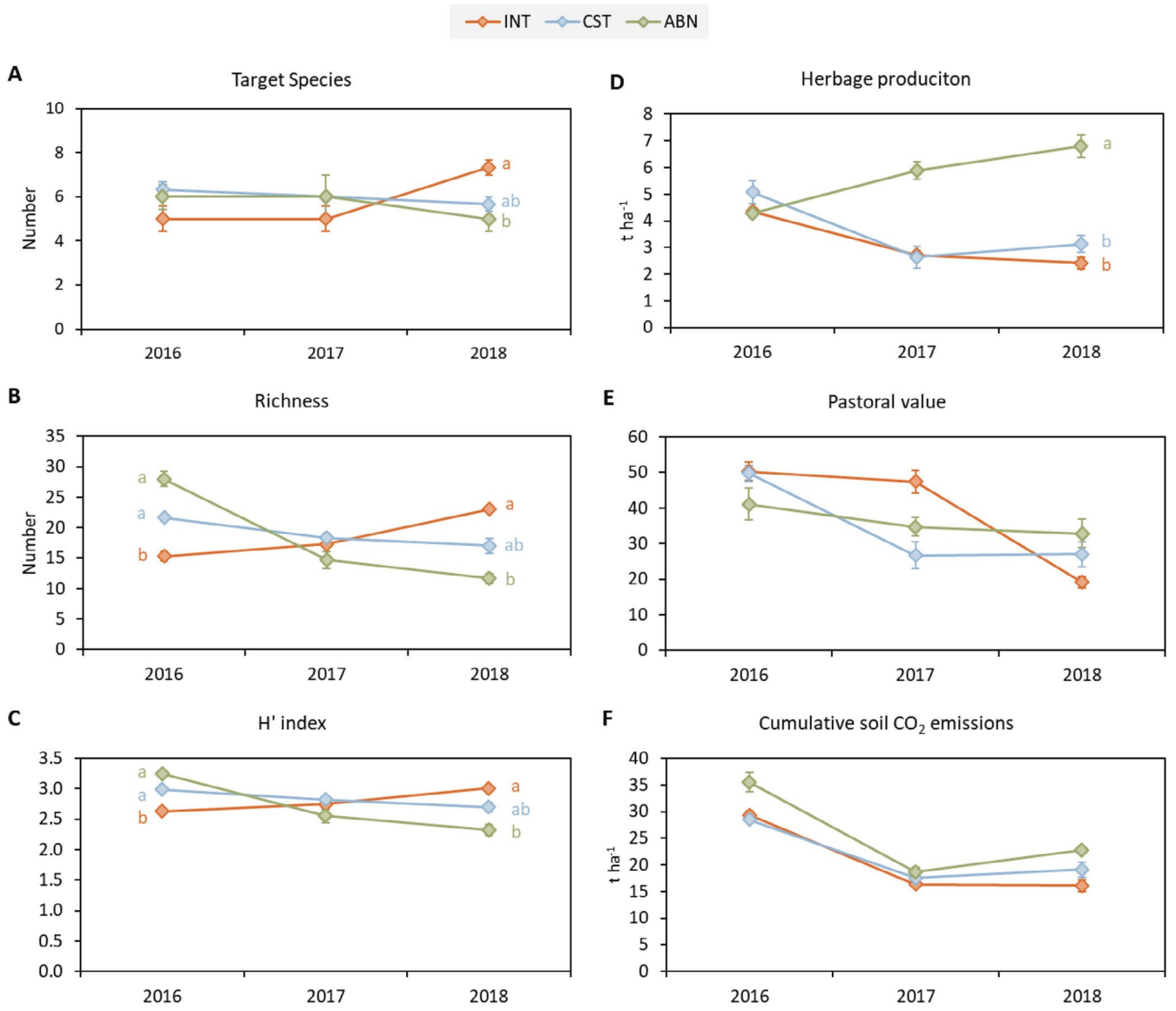


FIGURE 3 | Plant diversity indices (A–C), herbage production (D), pastoral value (E) and soil CO_2 emissions (F) under intensive utilisation (INT), customary management (CST) and abandonment (ABN). The means were separated at $p < 0.05$ according to Tukey's HSD. Lowercase letters indicate differences among management regimes within the same year. The bars represent the standard error of the mean.

($p = 0.065$). In the third year, herbage production was significantly higher in ABN than in the other two management regimes. Pastoral value was not affected by management regime but varied significantly with monitoring year, following a decreasing trend with significant differences between the first and the last year of monitoring (Figure 3E).

Cumulative soil CO_2 emissions varied significantly over the years, but no significant interaction with the management regimes was observed (Table 1). In the first year of monitoring, cumulative soil CO_2 emissions were significantly higher under ABN compared to CST; however, these differences disappeared in 2017 and 2018 (Figure 3F). In both INT and ABN, cumulative soil CO_2 emissions were significantly higher in 2016 compared to 2017 and 2018 (Figure 3F; Table 1).

3.3 | Relationship Among Plant Diversity, Herbage Production, Pastoral Value and Soil CO_2 Emissions

PCA ordination revealed variation trends in species composition (Figure 4). Along the PC1 axis, a clear distinction emerged between the 2016 observations, primarily associated with *L. perenne*, *Trifolium nigrescens* and *C. cristatus,* and the 2018 observations. In contrast, along the PC2 axis, the ABN plots were distinctly separated, characterised by species such as *B. rupestre*, *D. glomerata* and *C. arvense*. The variation in species composition along these two main axes was significantly correlated with all three variables used ($p < 0.01$): pastoral value and soil CO_2 emissions were correlated with the PC1 axis with the highest values recorded in the 2016 observations, while herbage production was correlated with the PC2 axis, showing the highest values in ABN plots.

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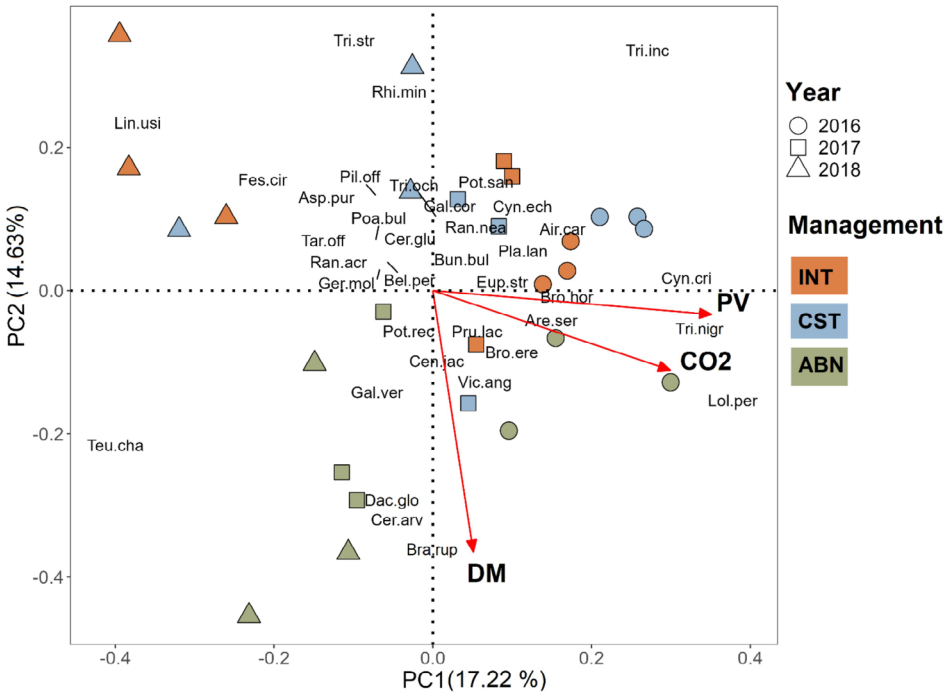


FIGURE 4 | Biplot of the principal component analysis showing the position of species, management regimes, year of monitoring (circles, squares and triangles) and response variables (red arrows). CO_2 = soil CO_2 emission, DM = dry matter (herbage production), PV = pastoral value. Only the species with −0.04 < score < 0.04 are shown in the biplot. The percentages of variance explained by each constrained axis are indicated in brackets. The abbreviation of the species is listed in Table S1.

4 | Discussion

4.1 | Plant Diversity

The short-term effects of ABN did not significantly differ from those of CST with respect to diversity indices, including target species, richness and Shannon diversity. However, ABN favoured the cover of the most competitive grass species, such as *B. rupestre* and reduced the abundance of species that are typically associated with mowing (e.g., *C. cristatus*, *R. minor*) (Bonanomi et al. 2009). In the Apennine grasslands, *B. rupestre* is an indicator of the early progressive vegetation dynamics that can lead to a grassland plant diversity reduction (Allegrezza et al. 2016; Biondi et al. 2015), probably by limiting light penetration at the soil surface (Bonanomi et al. 2006). When *B. rupestre* exceeds 80% cover, it can lead to the loss of grassland habitats (EU 'Habitat' Directive 92/43/EEC) through the replacement of grassland communities by edge communities belonging to the *Trifolio-Geranietea* class (Allegrezza et al. 2016) that are not ascribed to any habitat type. In the study area, the non-significant effects of ABN on diversity may be attributed to the cover of competitive species, particularly, *B. rupestre*, which does not exceed 50%. These findings suggest that a longer period (> 3 years) may be required for perennial grasses to increase in abundance and consequently reduce species diversity, as reported by Bonanomi et al. (2009). Compared with CST, the short-term effects of INT on diversity indices (Target species, Richness, Shannon) were not significant. However, INT promoted the cover of short-lived species such as *L. usitatissimum* and *T. striatum*, which are common in regularly mown mesophilous meadows of the class *Molinio-Arrhenatheretea* (Biondi et al. 2014) and are not ascribed to any habitat type (Directive 92/43/CEE) (Biondi et al. 2009).

Short-term studies lasting 3–6 years on the effects of different management regimes on the floristic diversity of semi-mesophilous Apennine grasslands of habitat 6210* have demonstrated that the outcomes are strongly influenced by the type and duration of previous management. Grasslands that had been abandoned for a long period (25–30 years) and subsequently invaded by *B. rupestre*, with high cover values (> 80%), showed a marked response, with a rapid recovery of floristic diversity already after the first year of mowing (Bonanomi et al. 2006, 2009, 2013; Tardella et al. 2018). By contrast, the grassland in the study area had been continuously managed through grazing and mowing for 30 years, and the application of ABN and INT produced no significant short-term effects on floristic diversity, but only slight shifts in the abundance of some species that may, over time, lead to substantial changes in species composition and habitat structure. Therefore, in the long term, ABN could lead to a *Brachypodium*-dominated grassland with reduced diversity and habitat loss, whereas INT could result in a transformation towards mesophilous grasslands, altering floristic composition and leading to the loss of habitat 6210*. These findings suggest that a significant vegetation response to management changes in grasslands with a long history of grazing and mowing requires longer-term experimentation, as also highlighted by Bonanomi et al. (2009).

4.2 | Herbage Production and Pastoral Value

In this study the variations in herbage production are mainly attributable to weather patterns, particularly the amount and distribution of precipitation, rather than mowing frequency (Dibari

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et al. 2021; D'Ottavio et al. 2023; Stybnarova et al. 2016). These results align with those reported by other authors in different contexts. For example, Koncz et al. (2015) observed a higher above-ground biomass for a semi-arid grassland under a once-yearly mowing regime than under continuous grazing. However, these differences emerged in specific years of monitoring, when early mowing was possible and rainy autumn conditions allowed biomass to persist for a longer period. Stybnarova et al. (2016) monitored a mowed grassland belonging to *Lolio perennis-Cynosuretum cristati* association for 7 years and observed that low-intensity management (i.e., mowing twice per year) resulted in higher above-ground biomass compared to high-intensity management (i.e., mowing thrice or four times per year) over a short term (i.e., in the first 3 years of monitoring). In the case of abandonment, as expected, a progressive accumulation of biomass was recorded as reported in other studies (Bonanomi et al. 2009).

On the other hand, it is well known that different mowing frequencies can substantially affect not only above-ground biomass but, in association with floristic composition changes, also grassland pastoral value. Some studies observed that mowing two times a year enhanced the species richness and pastoral value of semi-natural grasslands encroached by *B. rupestre* in the short term (Tardella et al. 2020). In the case of cessation of mowing, even over a short period, the cover of a few species with low forage value may increase (e.g., *B. rupestre*) (Bonanomi et al. 2009). In this case study, patterns in pastoral value appeared to be more strongly influenced by temporal changes than by differences among management regimes within individual years. The trends observed are consistent with floristic shifts driven by the cumulative effects of the three-year management regimes. Under INT, an increase in plant species richness was accompanied by a higher prevalence of species with low pastoral value and a concurrent decline in those with high pastoral value. In contrast, under ABN, the species that became dominant over time included taxa with intermediate *Si* values, which may have contributed to the absence of a decline in pastoral value over the 3 years of monitoring. Despite the relatively low productivity and pastoral value observed in the study, INT proved to be a viable option for maintaining high levels of biodiversity in *B. erecta* secondary grasslands of the Apennines. This suggests that, in these contexts, relatively intensive use does not pose an immediate threat to plant species diversity, at least in the short term. Therefore, in the short term, all management options, including abandonment, appear compatible with biodiversity conservation and may be alternated according to different needs, including economic considerations.

4.3 | Soil CO_2 Emissions

Generally, grasslands store large amounts of carbon in their topsoil (Austrheim et al. 2016; Bagella et al. 2020; Francioni, D'Ottavio, et al. 2019; Francioni, Trozzo, et al. 2019). However, the ability of grasslands to store carbon is not perpetual, as grasslands tend to reach 'carbon stock equilibrium' after several decades (Smith 2014). This suggests that the presence of grasslands does not automatically guarantee a carbon sink, as their capacity to store carbon depends on both soil carbon saturation and management, which can be highly context-dependent (Arca et al. 2021; Cao et al. 2024; Gilmullina et al. 2023). The reduction of soil CO_2 emissions by mowing has been widely observed previously in grassland ecosystems and was mainly explained by a reduced supply of photosynthetic assimilates from above-ground plant parts to roots and by lower availability of labile carbon substrates for rhizosphere microorganisms (Gavrichkova et al. 2010; Wei et al. 2016). In contrast, Francioni, D'Ottavio, et al. (2019), Francioni, Trozzo, et al. (2019) and D'Ottavio et al. (2023) demonstrated that different cutting intensities showed a limited impact on soil CO_2 emissions in semi-natural grasslands of the Apennines. In these ecosystems the rates of soil CO_2 emissions are in fact mainly constrained by the soil water content and temperature. In our study, no differences in seasonal cumulative soil CO_2 emissions were observed among treatments. A significant difference was only detected in the first year, when all treatments showed higher emissions compared with the second and third years. This pattern can be explained by the marked differences in meteorological conditions between 2016 and the following 2 years, with cumulative rainfall during the vegetative seasons of approximately 375, 192 and 198 mm in 2016, 2017 and 2018, respectively (Figure 1). Thus, soil CO_2 emissions generated from such grasslands were more affected by weather conditions than management regimes, at least in the short term (D'Ottavio et al. 2023; Francioni, D'Ottavio, et al. 2019; Francioni, Trozzo, et al. 2019).

There appears to be a positive relationship between floristic diversity and soil CO_2 emissions across diverse ecosystems (Chen and Chen 2019). In grasslands, this can be attributed to more diverse plant communities enhancing productivity, which may increase soil CO_2 emissions (Dias et al. 2010). However, this short-term study, which yielded contrasting results regarding floristic diversity, is likely too brief to draw definitive conclusions about the relationship between plant diversity and soil CO_2 emissions. Nevertheless, it raises important questions that should be addressed through longer-term research, once different management regimes have reached their equilibrium.

5 | Conclusion

The results suggest that, in the short term, more frequent utilisation of *B. erecta*-dominated grasslands—performing two to four additional cuts beyond the standard spring mowing and a second autumn harvest—would be a feasible management option. This is because intensive use did not show detrimental effects on herbage production or soil CO_2 emissions. However, further long-term studies are required to confirm these trends, particularly regarding the relationship between plant diversity and soil CO_2 emissions, a key component of the global carbon cycle. Under intensive utilisation, the slight short-term increase observed in annual species typical of mesophilic grasslands may indicate a shift in plant composition towards communities characteristic of nutrient-rich environments. This could serve as a cautionary signal, as such changes diverge from the typical features of priority habitat 6210*. On the other hand, even short-term abandonment does not appear to result in similar trade-offs, suggesting that occasional or reduced use of these grasslands could also be a viable option compared with complete abandonment.

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Given the general decline in European pastoral systems, we believe the priority should be to conserve biodiversity while also supporting the farmers who maintain these grassland ecosystems. If this is the objective, adopting a more flexible management approach, such as using permissive utilisation calendars or implementing more intensive management practices, would not lead to significant trade-offs among plant diversity, herbage production and soil CO_2 emissions—well-recognised key ecosystem services. Based on these results, there is a need to reflect on the adequacy of certain instruments, such as prescriptions or regulations at different administrative levels, that govern pastoral activities (e.g., grazing calendars or stocking limits) in many areas of the Central Apennines. These regulatory tools are often too rigid, outdated or not site-specific, both inside and outside protected areas, such as Natura 2000 sites. In this context, it would be valuable to study the impact on biodiversity and ecosystem services resulting from changes in timing (e.g., early meadow mowing and extended grazing) and stocking levels (e.g., the minimum level needed for grassland conservation). Such assessments should also consider global change factors, such as climate change and declining stocking rates, and may help encourage innovative management approaches. This becomes crucial if we consider *B. erecta*-dominated grasslands as a source of income for farmers, who are key players in the conservation of biodiversity, responsible for maintaining these important ecosystems.

Acknowledgements

We would like to thank late Marco Toderi for his considerable help and inspiration, without whom this research would have never been possible. We thank Loris Barbetti, Andrea Papaveri and Lucia Foresi for their assistance in the field. We thank the farm 'Azienda agricola biologica Valdicastro' for providing the necessary space and the 'Unione Montana Potenza Esino Musone' for providing the support. Open access publishing facilitated by Universita Politecnica delle Marche, as part of the Wiley - CRUI-CARE agreement.

During the preparation of this work, the authors used ChatGPT for linguistic revision in English. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

Funding

This study was carried out with the support of project PACTORES (PAstoral ACTORs, ES and Society as key elements of agro-pastoral systems in the Mediterranean), ERANETMED 'EURO-MEDITERRANEAN Cooperation through ERANET joint activities and beyond'—Joint Transnational Call 2016—Environmental challenges and solutions for vulnerable communities (ERANETMED2-72-303).

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Table S1:** List of plant taxa. **Table S2:** Two-way repeated measures ANOVA for plant species diversity indices. **Table S3:** Two-way repeated measures ANOVA for herbage production. **Table S4:** Two-way repeated measures ANOVA for soil CO_2 efflux. **Figure S1:** Seasonal variations of soil CO_2 emissions during the study period (April 2016 to October 2018) for intensive utilisation (INT), customary management (CST) and abandonment (ABN).

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