

EFFECT OF GRAZING AND MOWING ON THE MICROCOENOLOGICAL COMPOSITION OF A SEMI-ARID GRASSLAND IN HUNGARY

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Abstract. Grasslands play an important role in carbon sequestration. Different management regimes may change the species composition of plants in grasslands, thus may affect the carbon sequestration potential. Therefore, we investigated if short-term changes in species composition occurred in response to shift in grazing to mowing in a semi-arid grassland in Hungary (Bugac). Species abundances were recorded as frequencies of species in contiguous 5 cm × 5 cm quadrates along six (5 meters long) permanent transects at adjacent grazed and mowed sites in the spring and fall of 2012. High variation of vegetation characteristics was found within the management units at patch scale (at few m scale). The species richness, relative abundance of species, species density, Shannon diversity index, and forage quality differed among transects; however, no differences were observed between the two management regimes at site scale (at hectare scale) (ANOVA, $p > 0.05$). The average species abundance of the grazed and mowed sites were very similar (*Poa* spp.: 13.48±5.34%, 14.67±6.9%; *Carex* spp.: 12.46±8.75%, 14.24±10.16%; *Cynodon dactylon*: 10±8.52%, 16.01±3.49%, respectively). Our results suggest that species composition differences can not be the main factors explaining short-term differences in carbon sequestration in natural semi-arid grasslands.

Keywords: *diversity, management, vegetation structure, species composition changes, species abundance*

Introduction

Due to climate change the greenhouse gas (GHG) emission should be reduced in all sectors of economics including agriculture (IPCC 2007). Therefore, new adaptation and mitigation technologies are being developed in this sector to reduce GHG's emission (Smith et al., 2008). The "AnimalChange" project of the EU's 7th framework program (<http://www.animalchange.eu/>) is taking a major part in achieving these goals. Our research group has participated to this project, with a special focus on the effect of management (grazing vs. mowing) on the carbon dioxide balance of grasslands.

Grasslands cover approximately 11% of Hungary and are an important sink for carbon dioxide (Haszpra et al., 2011, Nagy et al., 2011). However, changes in the management usually result in changes in carbon sequestration potential (NIR, 2010) and changes in species composition of grasslands (Moog, 2002, Jantunen, 2003). For example decreasing carbon storage capacity coupled with degradation of species

composition was reported when intensive grassland management (fertilized and irrigated) in Hungary changed to extensive management (decreased stocking rate) (NIR, 2010).

Changing management affects several structural and functional attributes in vegetation. Therefore, it is hard to separate the direct effects of species composition change from other related effects (changes of leaf area index, soil bulk density, and soil water holding capacity) on carbon sequestration (Klumpp et al., 2011). Management regime shifts from grazing to mowing could have a significant, but varying (negative vs. positive) effect on species composition (Fritch et al., 2011, Jacquemyn et al., 2011, Socher et al., 2012). In general grasslands mostly benefit from light grazing and mowing and degrade in terms of species composition from frequent mowing and intense grazing but the direction of change also depends on the initial status of the grassland (Jantunen, 2003, Socher et al., 2012). However, comparative studies on the effects of mowing and grazing in the same areas are rare (Jantunen, 2003) and studies are mostly focusing on the changes in species richness (Jantunen, 2003, Pykälä, 2005, Socher et al., 2012), rather on fine scale structural changes in the grassland, which could early indicate the direction of changes (Virágh et al., 2000, Bartha et al., 2008).

Our initial question was whether management change from grazing to mowing influences carbon sequestration potential. Our hypothesis is that management change has an effect on carbon sequestration. If the species composition of the grazed and mowed sites differs then it could be a potential driving factor for carbon sequestration differences. Therefore, to answer the initial question, first we needed to investigate the coenological composition of grazed and mowed areas. We assumed that the two sites have similar species composition, because we wanted to start the mowed and grazed experiment on the carbon balance on sites having similar species composition. We predicted that if the two sites will have similar species composition then the species richness, the abundance of species, species density, Shannon diversity index, and forage quality will be similar at the two sites. In this paper we investigated the species composition of the grazed and mowed sites, which provides a baseline to further microcoenological and eco-physiological surveys of mowed and grazed sites.

Materials and methods

Microcoenological investigation

To investigate the effect of grazing and mowing on the species composition of grassland we performed repeated microcoenological surveys in a semi-arid sandy grassland of Hungary (Bugac, N46°41'28", E19°36'42"). The grassland (500 ha) has been grazed by cattle for a minimum of 20 years (Nagy et al., 2011). The average annual precipitation at the area is 562 mm and the annual mean temperature is 10.4°C (Nagy et al., 2011). The grazing period of cattle (1.02 cattle/ha) on the grazed site in the previous year of the present survey lasted from June to July and from October to December in 2011. In 2012, the grazing period of cattle (1.87 cattle/ha) lasted from the middle of June to August and from September to November. The mowed area (1 ha) was fenced from the grazed area in March 2011 (Figure 1). The first mowing occurred on August 10th, 2011. The second mowing was on June 22th, 2012 (for both years with a mowing height of 6 centimeters above ground). After drying for a few days the hay was removed from the site. No other treatments were applied on the sites (e.g. watering or fertilization).

The microcoenological surveys occurred on June 13-14th and October 11-13th, 2012. The microcoenological surveys consisted of 6 permanent transects of 5 meters starting at 40 meters from a climate station and where separated by 60 degrees (*Figure 1*). 5 × 5 centimeter contiguous quadrates were spaced by 5 cm distance along each transects. Within each transect the presence of all rooting plant species was recorded in the 100 contiguous microquadrats to determine species abundances and composition. Species nomenclature followed Király (2009).

The experiment was designed for the carbon sequestration survey. Carbon sequestration is studied through biomass sampling at the vicinity of the climate stations and through carbon balance measurement based on eddy covariance methods at the climate stations on both sites. Therefore, we tried to match the heterogeneity of both sites surrounding the climate stations in a 1 ha area where the carbon balance and biomass data are originating.

We used permanent transects for microcoenological survey as it provides an objective and precise baseline record to follow fine-scale changes in species abundances and composition (Bartha, 2008). Microquadrates were used to avoid subjective bias from cover estimate and also because vegetation appeared to be very compact and dense though line-intercept method could not have been used.

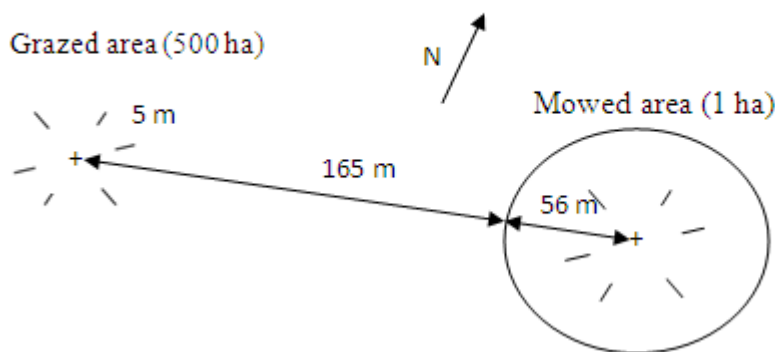


Figure 1. The experimental design of the microcoenological survey conducted at the grassland of Bugac, Hungary (climate station: +; microcoenological transects: -) (N46°41', E19°36')

Statistical analysis of microcoenological indicators

Grasslands were characterized by species richness, relative species abundance, species density, Shannon diversity index, and by forage quality. Identification to species level was not always possible due to the lack of flowers, seeds or to the lack of the main part of stem; therefore for the analysis we combined certain species at genus level (e.g. *Achillea* spp., *Carex* spp., *Poa* spp. see in Appendix 1.). Relative abundance of species was calculated by equitation (1):

$$A_i = \frac{a_i}{T} * 100 \quad (\text{Eq. 1})$$

A_i is the relative abundance of species i (%) where a_i is the number of presences of species i along the transects, T is the total number of presences of all species along the

transect. To calculate the average species abundance for a given species at site level the average of the six transects were calculated. Coefficient of variation (CV) was calculated for all species based on the six transects, and the average of all species level coefficients was compared between sites and seasons (Equation 2). The coefficient of variation of species abundance at site level shows the relation of the standard deviation of species abundance to the mean. If it is higher than one than the standard deviation is higher than the mean.

$$CV_a = \sum_{i=1}^n CV_i/n \quad (\text{Eq. 2})$$

Where CV_a is the average coefficient of variation of species abundance for species i at site level, CV_i is the coefficient variation of species i , and n is the total species number encountered at the site.

Species density was calculated as the average number of species found within a microquadrant per transect (Equation 3). To calculate the average species density at site level the average of the six transects were calculated.

$$SD = \sum_{i=1}^{n=100} s_i/100 \quad (\text{Eq. 3})$$

Where SD is the average species density (species per microquadrant), s_i is the species number within one quadrat.

To estimate the quality of forage at site level the average of transect's forage quality was calculated. Even if the two sites has similar species abundance, still the forage quality could differ as species with high forage quality values weight more in calculating the total quality of forage. Thus, little differences in the abundance of valuable forage species could lead to larger differences in total forage quality between sites. The average quality of forage was estimated at site level by a modified equation of Klapp (1953) (Equation 4). The quality values for species were from Balázs (1960). Equation 4 (modified from Klapp et al., 1953):

$$Q = \sum_{i=1}^n \frac{f * A}{100} * \bar{A} \quad (\text{Eq. 4})$$

Where Q is the quality value of the forage (dimensionless index) at site level, f is the forage quality of species i , A is the abundance of the species i , and \bar{A} is the average abundance of all species, instead of the total cover of plants (%) as by Klapp et al. (1953), because the total abundance here would be always 100%.

Differences in species richness, relative species abundance, species density, Shannon diversity index, and forage quality between grazed and mowed sites and between spring and fall were compared by the average values of the six transect's with ANOVA (post

hoc analysis with Tukey test, where random effects were the managements regimes and seasons), and also with paired t-test between spring to fall for the same managements.

To demonstrate the overall level of similarity between mowed and grazed sites and to show the dissimilarity of individual transects cluster analysis was used (single linkage, Euclidean distance). All statistics were performed with Minitab Inc (2006).

Results

Species richness, relative abundance of species

We found 67 species on the grazed site, while 62 species on the mowed site (for a total of 83 different species) (Appendix 1). Out of these 21 species were found only on the grazed site with a very low average abundance ($0.15 \pm 0.43\%$). The total abundance of these 21 rare species on the grazed site was 3.34%. The most abundant among these rare species on the grazed site were *Inula britannica* ($0.64 \pm 1.51\%$, found at the same spot during spring and fall), *Cerastium* spp. ($0.62 \pm 1.57\%$, found only during fall), and *Trifolium repens* ($0.40 \pm 0.99\%$, found at the same spots during spring and fall) (Appendix 1). Conversely, 16 species were not found on the grazed site, but rather on the mowed site with an average abundance of $0.08 \pm 0.23\%$. The total abundance of these 16 rare species on the mowed site was 1.36%. The most abundant among these rare species on the mowed site were *Veronica* seedlings ($0.5 \pm 1.37\%$, found both during spring and fall), *Verbascum phoeniceum* ($0.15 \pm 0.37\%$, found both during spring and fall), and *Buglossoides arvensis* ($0.14 \pm 0.26\%$, found only during spring) (Appendix 1).

The most abundant species found at the grazed site during spring and fall were *Poa* spp. ($13.45 \pm 5.28\%$, 13.45 ± 4.94), *Festuca pseudovina* ($12.44 \pm 7.14\%$, $9.14 \pm 4.32\%$), *Carex* spp. ($12.38 \pm 8.64\%$, $12.54 \pm 7.79\%$) and *Cynodon dactylon* ($11.38 \pm 9.94\%$, $8.64 \pm 5.55\%$) (Appendix 1). At the mowed site the average species abundance for spring and fall was very similar; *Cynodon dactylon* ($17.51 \pm 4.45\%$, 14.5 ± 2.52), *Poa* spp. ($15.56 \pm 7.77\%$, 13.69 ± 5.02), *Carex* spp. ($15.35 \pm 11.35\%$, $13.14 \pm 7.69\%$), and *Festuca pseudovina* ($9.43 \pm 9.36\%$, $8.85 \pm 6.6\%$) (Appendix 1). The species abundance was highly variable among the six transects within the grazed and mowed sites as shown by the large standard deviations (Appendix 1) and by the large coefficients of variation for species abundance (Table 1). Coefficients of variation for species abundance showed that the ratio of the standard deviation to the mean is higher than one, which indicated a high variability of species abundance between transects. Differences among transects resulted from the mosaics' of different vegetation patches. Transects crossed patches where the species richness was low and a few frequent species dominated the patch (*Festuca pseudovina* and *Cynodon dactylon*), whereas other transects crossed patches having high species richness with more variable dominant species (*Poa* spp., *Carex* spp., *Elymus* spp., *Medicago falcata*). The two sites appeared to be equally patchy as coefficient of variation in species abundance was not significantly different between sites (Table 1). Also, relative species abundance differed only in a few instances between the grazed and mowed sites. Strong significant differences observed in species abundance only in case of *Arenaria serpyllifolia* in spring ($p < 0.05$, where species abundance was higher at grazed site), and a weak difference in case of *Cynodon dactylon* in fall ($p < 0.1$, where species abundance was higher at the mowed site) (Figure 3). For all of the rest of the species the abundance did not differ significantly between the grazed and mowed sites (for the ten most abundant species see Figure 2 and for all species see Appendix 1). The total abundance of ten most abundant species at the

grazed and mowed site at spring ($84.01 \pm 30.36\%$, $82.74 \pm 28.44\%$) and at fall (73.30 ± 27.49 , $81.31 \pm 30.89\%$, respectively) was similar.

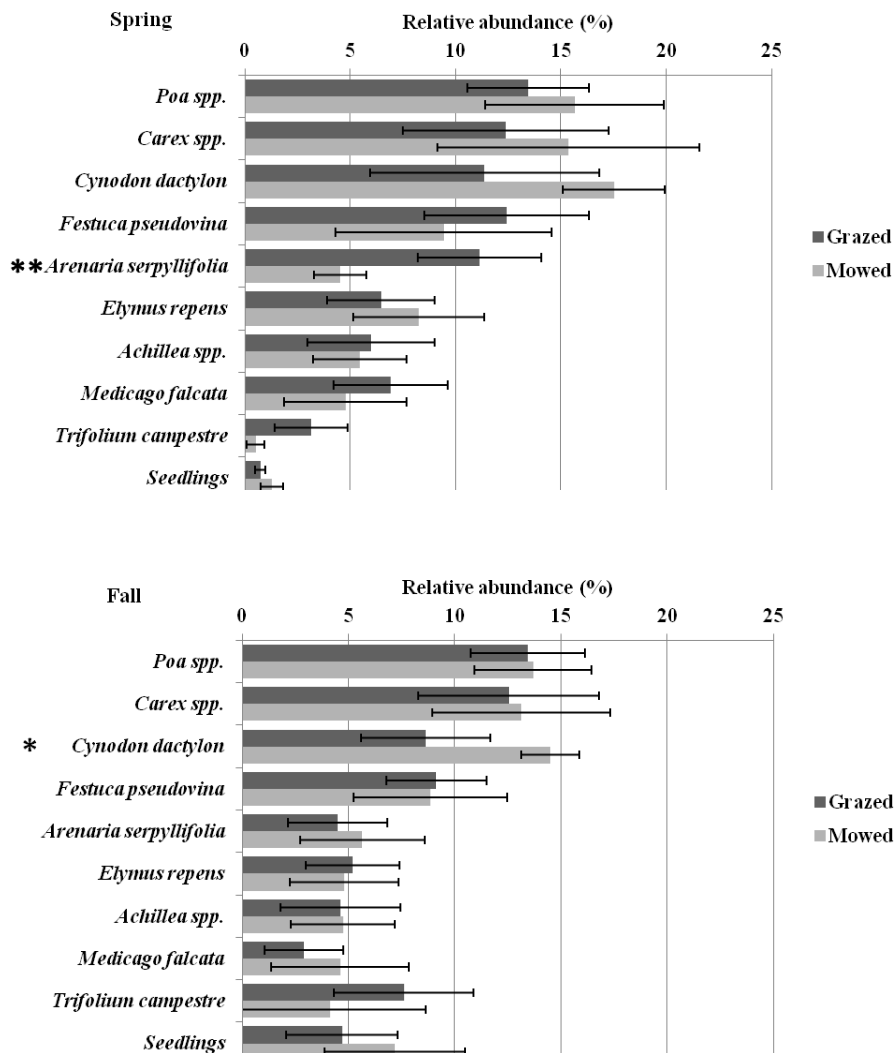


Figure 2. Relative species abundance (%) of grazed and mowed sites at the grassland of Bugac at spring and fall (** indicates significant differences between grazed and mowed site at $p < 0.05$ and * at $p < 0.1$ level, t-test)

Between spring and fall we found significant differences ($p < 0.05$) only at two cases. At the grazed site the abundance of *Medicago minima* and the *Geranium pusillum* was higher during fall compared to spring (Appendix 1). The abundance of *Medicago minima* was also higher on the mowed site at fall compared to spring, but the difference was weak ($p < 0.1$). The abundance of seedlings was also higher during fall for both sites compared to spring at low significance ($p < 0.1$).

Species density, Shannon diversity index and forage quality

The average species density found within one microquadrant ($5 \text{ cm} \times 5 \text{ cm}$) did not differ between grazed and mowed site and between spring and fall (ANOVA, post hoc

Tukey test). Between season difference was also tested by paired t-test, which did not reveal differences as well in species density. The Shannon diversity index and forage quality did not differ as well between grazed and mowed sites and between seasons (Table 1). Since very little differences observed between the abundance of species between the grazed and mowed sites, therefore it had no effect on differentiating the average values of forage quality between the two sites and seasons (Table 1).

Table 1. Mean and standard deviation (std in parenthesis) of species richness, species density, Shannon diversity and average forage quality of grazed and mowed sites at the grassland of Bugac (no significant differences observed among groups, ANOVA, Tukey test, $p > 0.05$)

| Management | Spring | | Fall | | p value |
|----------------------------------|-----------------|------------------|-----------------|-----------------|---------|
| | Grazed | Mowed | Grazed | Mowed | |
| Species number | 21.17 (6.49) | 22.17 (8.21) | 23.83 (4.49) | 18.33 (3.61) | 0.46 |
| Species density | 3.85 (0.78) | 3.46 (0.52) | 3.84 (0.50) | 3.74 (0.72) | 0.69 |
| CV for species abundance (Eq. 2) | 1.78 (0.69) | 1.83 (0.68) | 1.80 (0.67) | 1.87 (0.67) | 0.48 |
| Shannon diversity index | 2.31 (0.50) | 2.14 (0.56) | 2.43 (0.31) | 2.23 (0.44) | 0.73 |
| Forage quality (Eq. 4) | 15.71 (5.60) | 17.31 (12.12) | 13.93 (6.43) | 16.87 (4.13) | 0.87 |
| n (transects) | 6 | 6 | 6 | 6 | |

Overall similarity between management regimes

The average relative species abundance was very similar for the grazed and mowed sites at spring and at fall for all species (ANOVA, Tukey, $p > 0.05$) (Table 2).

Table 2. Species abundance differences between the grazed and mowed sites (p values for ANOVA)

| | Spring | | Fall | |
|------------------|--------|-------|--------|-------|
| | Grazed | Mowed | Grazed | Mowed |
| Grazed vs. mowed | 0.75 | | 0.44 | |
| Among all groups | | | 0.88 | |

Overall similarity between the grazed and mowed sites was proved by cluster analysis, as treatments did not separate into groups of grazed and mowed sites (Figure 3). The main dissimilarity was due to the variability between transects (Figure 3). A few transects differentiated from the rest of the transects e.g. transect number 1 and 5 at the grazed site was distinct from other transects during spring and fall. These transects were characterized with high species richness.

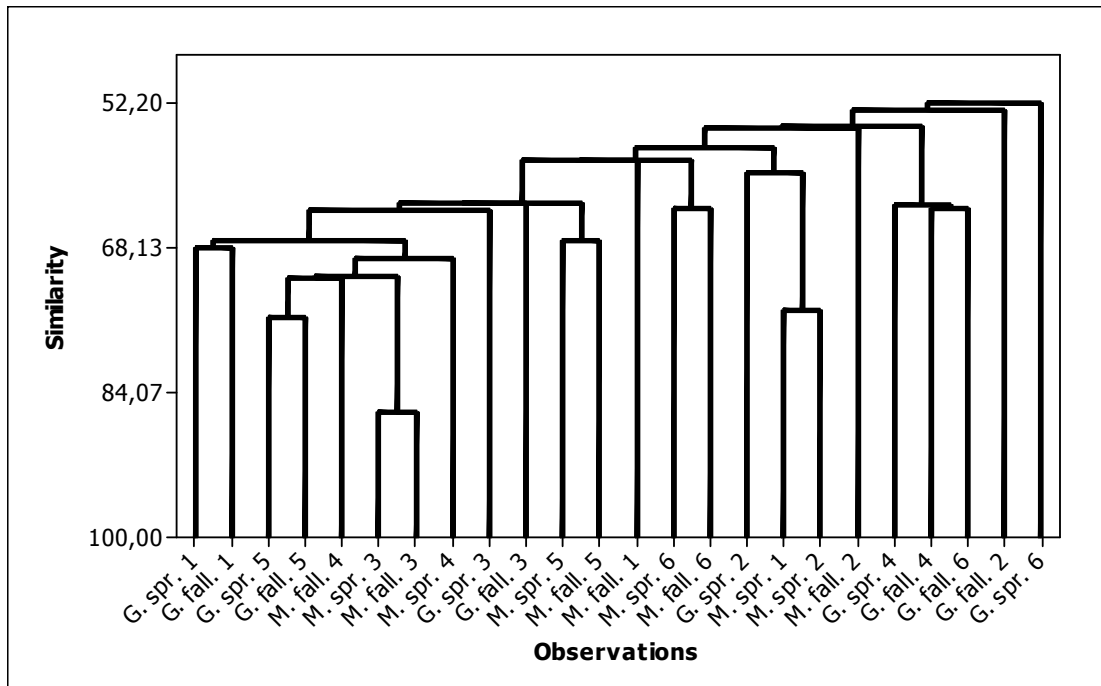


Figure 3. Classification of transects (1-6) on grazed (G.) and mowed (M.) sites during fall and spring (spr.), 2012 at the grassland of Bugac. (Cluster analysis, single linkage, Euclidean distance) (2006 Minitab Inc.)

Discussion

Present survey provides a baseline to follow the changes in species composition due to management change from grazing to mowing in a long-term experiment. Management change could alter the species composition and carbon sequestration potential of grasslands (Moog, 2002, NIR, 2010). Therefore, we investigated the vegetation attributes of adjacent grazed and mowed sites in semi-arid grassland of Hungary (Bugac) parallel with carbon sequestration survey (unpublished results). Species composition was investigated on both sites with microcoenological methods and described by the species richness, relative abundance of species, species density, Shannon diversity index and by the average forage quality. Within both sites the variability of these microcoenological indicators was high, which indicated the patchiness of the vegetation. Also, the grazed and mowed sites had a number of different, unique species. As these species were rare in our observations, we might have increased the abundance of these species if we had used more transects. Interestingly, the vegetation at fall was similar to spring, which was maybe due to rainy fall in 2012. Therefore, many species was germinating and growing at the both site during fall (e.g. *Geranium pusillum*, *Arenaria serpyllifolia*, *Veronica arvensis* and also seedlings), which are usually present at spring. We assumed that due to rainy fall in 2012 many species grew again (seedlings and other species usually present at spring) filling the gaps in the structure of the grassland and increased the species density to the same level as it was at spring.

In contrast to the considerable heterogeneity observed within sites, no differences were found in community level attributes when the analyses were performed at site level (comparing average attributes between sites). Species richness, relative abundance

of species, species density, Shannon diversity index and forage quality did not differ significantly between the grazed and mowed sites. Cluster analysis also supported the similarity of the two sites.

In summary we can assume that the management change from grazing to mowing had no effect on the vegetation patterns of the mowed site, and also we can conclude that the two sites were similar in species composition. Had there been a difference in species composition between the two sites then it was assumed to be either due to the initial differences of the two sites or to the effect of management. Depending on management intensities grazing and mowing found to change the species composition in long-term studies.

Due to management change species composition could either remain unchanged or change into a negative (loss of species or increase of unfavourable species condition in terms of conservation purposes) or positive direction as found in other studies. Depending on stocking densities and grazer species, more unpalatable species might benefit from grazing while others are suppressed (Jacquemyn et al., 2011). In a study under continuous grazing no change was observed in species richness (Jacquemyn et al., 2011). In contrast, grazing was found to increase (due to light grazing of cattle, Fritch et al., 2011) or decrease (due to overgrazing by horses, Penksza, 2005) the species richness compared to the mowing treatment (Fritch et al., 2011, Penksza, 2005, respectively). Mowing found to decrease the abundance of disturbance sensitive species, while increased the disturbance-tolerant ruderal species by frequent destruction of above-ground plant organs (Socher et al., 2012). Enrichment of species or recovery after mowing might be due to enhanced seedling germination by the removal of litter and above-ground biomass, which creates gaps that act as suitable micro-sites for the establishment of small-seeded species (Socher et al., 2012). The species richness could change even during a short period of three years of mowing (once per year), however with unfavourable effects on the original dry grassland community (Jantunen, 2003). Still, grazing and mowing found to be better management regime than leaving the site abandonment as species richness was higher compared to the abandonment site (Hejzman et al., 2010). A 25 year study of light mowing and grazing from an initial homogenous state also showed that mowing and grazing were the most suitable management regimes than abandonment for conservation purposes (Moog, 2002). As species composition changes due to management change it could affect carbon sequestration potential. In a similar study contrasted management (intense vs. extensive grazing of heifers) led to distinct plant communities, which had an indirect effect on carbon fluxes as different communities respond differently to precipitation and soil drying in terms of CO₂ exchange (Klumpp et al., 2011). However, species composition change is usually a consequence of management change and it is a hard to identify the main factor responsible for changes in carbon sequestration (Klumpp et al., 2011). Besides species composition drivers could be differences in topography, phenology, direct and indirect effect of management (biomass removal, vegetation recovery, lack of fertilization on mowed sites) or even local soil properties. Our study provided evidence that if there is a difference in carbon sequestration between the grazed and mowed sites at the studied area then species composition differences can not be a leading factor for differences in carbon sequestration.

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Appendices

Appendix 1. Mean and standard deviation (std) of relative species abundance (%) on the grazed and mowed sites during spring and fall at the grassland of Bugac

| # Species | Spring | | Fall | | Grazed | | Mowed | |
|---|--------|------|------|------|--------|------|-------|------|
| | % | Std | % | Std | % | Std | % | Std |
| 1. <i>Achillea</i> spp.; <i>A. setacea</i> , <i>A. collinea</i> , <i>A. pannonica</i> | 5.99 | 5.52 | 5.47 | 4.05 | 4.63 | 5.12 | 4.74 | 4.49 |
| 2. <i>Alopecurus pratensis</i> | 0 | 0 | 0 | 0 | 0.34 | 0.56 | 0 | 0 |
| 3. <i>Alyssum alyssoides</i> | 0.51 | 0.72 | 0 | 0 | 0 | 0 | 0 | 0 |
| 4. <i>Anchusa officinalis</i> | 0 | 0 | 0.04 | 0.09 | 0 | 0 | 0 | 0 |
| 5. <i>Anthemis ruthenica</i> | 0.07 | 0.15 | 0.11 | 0.24 | 0 | 0 | 0 | 0 |
| 6. <i>Arenaria serpyllifolia</i> | 11.15 | 5.37 | 4.51 | 2.28 | 4.50 | 4.27 | 5.65 | 5.35 |
| 7. Asteraceae spp. | 0 | 0 | 0 | 0 | 0.14 | 0.31 | 0 | 0 |
| 8. <i>Astragalus cicer</i> | 0.15 | 0.33 | 0 | 0 | 0 | 0 | 0 | 0 |
| 9. <i>Astragalus onobrychis</i> | 0 | 0 | 0.04 | 0.09 | 0 | 0 | 0 | 0 |
| 10. <i>Berteroa incana</i> | 0.10 | 0.22 | 0.23 | 0.52 | 0.23 | 0.51 | 0 | 0 |
| 11. <i>Brachythecium</i> spp. | 0.65 | 1.46 | 0 | 0 | 0.88 | 1.69 | 0.12 | 0.27 |
| 12. <i>Bromus hordeaceus</i> | 0.46 | 0.50 | 1.02 | 1.12 | 0.42 | 0.53 | 1.14 | 1.86 |
| 13. <i>Bromus inermis</i> | 0 | 0 | 0.11 | 0.24 | 0 | 0 | 0 | 0 |

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|---|-------|------|-------|-------|-------|------|-------|------|
| 14. Bryophytes | 0.86 | 1.44 | 1.95 | 3.61 | 0.09 | 0.20 | 0 | 0 |
| 15. <i>Buglossoides arvensis</i> | 0 | 0 | 0.29 | 0.30 | 0 | 0 | 0 | 0 |
| 16. <i>Capsella bursa-pastoris</i> | 0 | 0 | 0.10 | 0.23 | 0 | 0 | 0 | 0 |
| 17. <i>Carduus nutans</i> | 0 | 0 | 0.15 | 0.24 | 0 | 0 | 0 | 0 |
| 18. <i>Carex</i> spp.; <i>Carex stenophylla</i> , <i>C. divisa</i> , <i>C. liparicarpos</i> , <i>C. Praecox</i> | 12.38 | 8.94 | 15.35 | 11.35 | 12.54 | 7.79 | 13.14 | 7.63 |
| 19. Caryophyllacea spp. | 0 | 0 | 0 | 0 | 0.04 | 0.08 | 0.89 | 1.98 |
| 20. <i>Centaurea</i> spp. | 0 | 0 | 0 | 0 | 0.09 | 0.12 | 0 | 0 |
| 21. <i>Cerastium</i> spp.; <i>C. pumilum</i> , <i>C. semidecandrum</i> | 0 | 0 | 0 | 0 | 1.25 | 1.95 | 0 | 0 |
| 22. <i>Chenopodium album</i> | 0 | 0 | 0.05 | 0.12 | 0 | 0 | 0 | 0 |
| 23. <i>Cirsium</i> spp. Seedling | 0 | 0 | 0 | 0 | 0.11 | 0.25 | 0 | 0 |
| 24. <i>Convolvulus arvensis</i> | 0.84 | 0.87 | 0.76 | 1.00 | 0.39 | 0.31 | 0.20 | 0.46 |
| 25. <i>Conyza Canadensis</i> | 0.04 | 0.08 | 0 | 0 | 0.09 | 0.20 | 0 | 0 |
| 26. <i>Cruciata pedemontana</i> | 0.45 | 0.73 | 2.18 | 2.31 | 0.80 | 1.78 | 0.04 | 0.09 |
| 27. <i>Cynodon dactylon</i> | 11.38 | 9.94 | 17.51 | 4.45 | 8.64 | 5.55 | 14.50 | 2.52 |
| 28. <i>Dactylis glomerata</i> | 0.24 | 0.37 | 0.17 | 0.37 | 0.21 | 0.22 | 0.16 | 0.36 |
| 29. <i>Descurainia sophia</i> | 0 | 0 | 0 | 0 | 0.09 | 0.20 | 0 | 0 |
| 30. <i>Draba nemorosa</i> | 0.11 | 0.16 | 0 | 0 | 0 | 0 | 0 | 0 |
| 31. <i>Elymus repens</i> | 6.45 | 4.65 | 8.25 | 5.66 | 5.20 | 4.06 | 4.79 | 4.66 |
| 32. <i>Erodium cicutarium</i> | 0 | 0 | 0.37 | 0.72 | 0.85 | 1.91 | 2.32 | 4.72 |
| 33. <i>Eryngium campestre</i> | 0.69 | 0.56 | 1.27 | 2.21 | 0.81 | 1.27 | 1.21 | 1.97 |
| 34. <i>Erysimum diffusum</i> | 0.04 | 0.08 | 0.19 | 0.23 | 0 | 0 | 0 | 0 |
| 35. <i>Euphorbia cyparissias</i> | 0.04 | 0.09 | 0 | 0 | 0 | 0 | 0 | 0 |
| 36. <i>Falcaria vulgaris</i> | 0.44 | 0.99 | 0.32 | 0.52 | 0.25 | 0.56 | 0.31 | 0.60 |
| 37. <i>Festuca pseudovina</i> | 12.44 | 7.14 | 9.43 | 9.36 | 9.14 | 4.32 | 8.85 | 6.60 |
| 38. <i>Galium verum</i> | 2.56 | 2.95 | 1.43 | 1.71 | 3.34 | 4.02 | 1.75 | 1.95 |
| 39. <i>Geranium pusillum</i> | 0.21 | 0.28 | 0.58 | 1.03 | 1.64 | 1.22 | 1.05 | 1.67 |
| 40. <i>Herniaria incana</i> | 0.16 | 0.37 | 0 | 0 | 0.04 | 0.09 | 0 | 0 |
| 41. <i>Inula Britannica</i> | 0.63 | 1.40 | 0 | 0 | 0.67 | 1.50 | 0 | 0 |
| 42. <i>Knautia arvensis</i> | 0.04 | 0.09 | 0 | 0 | 0 | 0 | 0 | 0 |
| 43. <i>Koeleria cristata</i> | 0.16 | 0.37 | 0 | 0 | 0.14 | 0.20 | 0.17 | 0.37 |
| 44. <i>Lamium amplexicaule</i> | 0 | 0 | 0 | 0 | 0.13 | 0.28 | 0.09 | 0.20 |
| 45. <i>Lamium purpureum</i> | 0 | 0 | 0 | 0 | 0.09 | 0.20 | 0 | 0 |
| 46. <i>Leontodon hispidus</i> | 0 | 0 | 0.08 | 0.17 | 0 | 0 | 0 | 0 |
| 47. <i>Marrubium peregrinum</i> | 0.15 | 0.33 | 0 | 0 | 0.08 | 0.19 | 0 | 0 |
| 48. <i>Medicago falcate</i> | 6.92 | 4.95 | 4.77 | 5.30 | 2.90 | 3.39 | 4.62 | 5.91 |
| 49. <i>Medicago lupulina</i> | 0 | 0 | 0.17 | 0.37 | 0.39 | 0.77 | 0 | 0 |
| 50. <i>Medicago minima</i> | 0.31 | 0.44 | 0.17 | 0.37 | 2.72 | 1.64 | 2.10 | 2.29 |
| 51. <i>Medicago</i> spp. | 0 | 0 | 0 | 0 | 0.04 | 0.09 | 0.64 | 1.43 |
| 52. <i>Medicago</i> spp. Seedling | 0 | 0 | 0 | 0 | 1.02 | 2.16 | 0.27 | 0.61 |

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|---|-----------|------|-----------|------|-----------|------|-----------|------|
| 53. <i>Plantago lanceolata</i> | 0.04 | 0.08 | 0.78 | 1.36 | 0.08 | 0.19 | 0.63 | 0.79 |
| 54. <i>Poa</i> spp.; <i>P. angustifolia</i> , <i>P. bulbosa</i> , <i>P. humilis</i> | 13.45 | 5.28 | 15.65 | 7.77 | 13.45 | 4.94 | 13.69 | 5.02 |
| 55. <i>Podospermum canum</i> | 0 | 0 | 0.16 | 0.24 | 0.07 | 0.17 | 0 | 0 |
| 56. <i>Polygonum aviculare</i> agg. | 0 | 0 | 0.20 | 0.35 | 0 | 0 | 0 | 0 |
| 57. <i>Polygonum bellardii</i> | 0 | 0 | 0 | 0 | 0 | 0 | 0.03 | 0.08 |
| 58. <i>Potentilla arenaria</i> | 0.20 | 0.36 | 0.12 | 0.28 | 0.41 | 0.91 | 0.22 | 0.49 |
| 59. <i>Potentilla argentea</i> agg. | 0.37 | 0.82 | 0.59 | 1.21 | 0.12 | 0.27 | 0.37 | 0.84 |
| 60. <i>Potentilla</i> spp. | 0.92 | 2.06 | 0 | 0 | 0.42 | 0.83 | 0.04 | 0.09 |
| 61. <i>Scirpoides holoschoenus</i> | 0 | 0 | 0.08 | 0.19 | 0 | 0 | 0.04 | 0.09 |
| 62. <i>Securigera varia</i> | 0 | 0 | 0 | 0 | 0.34 | 0.75 | 0 | 0 |
| 63. Seedlings | 0.73 | 0.46 | 1.28 | 0.96 | 4.69 | 4.78 | 7.18 | 6.04 |
| 64. <i>Silene alba</i> | 0.11 | 0.25 | 0.40 | 0.69 | 0.58 | 0.74 | 0.93 | 1.62 |
| 65. <i>Silene otites</i> | 0 | 0 | 0.05 | 0.12 | 0 | 0 | 0 | 0 |
| 66. <i>Stellaria media</i> | 0 | 0 | 0 | 0 | 0.55 | 1.23 | 0 | 0 |
| 67. <i>Taraxacum laevigatum</i> agg. | 0.03 | 0.08 | 0.12 | 0.19 | 0.08 | 0.11 | 0.04 | 0.10 |
| 68. <i>Taraxacum officinale</i> | 0.07 | 0.15 | 0 | 0 | 0 | 0 | 0.66 | 0.81 |
| 69. <i>Teucrium chamaedrys</i> | 0 | 0 | 0 | 0 | 0 | 0 | 0.11 | 0.24 |
| 70. <i>Thlaspi perfoliatum</i> | 0.04 | 0.09 | 0 | 0 | 0 | 0 | 0 | 0 |
| 71. <i>Tortula ruralis</i> | 1.39 | 2.07 | 0.96 | 1.23 | 3.68 | 6.61 | 0.05 | 0.10 |
| 72. <i>Trifolium arvense</i> | 0 | 0 | 0 | 0 | 0.14 | 0.31 | 0 | 0 |
| 73. <i>Trifolium campestre</i> | 3.13 | 3.18 | 0.51 | 0.75 | 7.61 | 5.98 | 4.15 | 8.18 |
| 74. <i>Trifolium repens</i> | 0.77 | 1.24 | 0 | 0 | 0.04 | 0.09 | 0 | 0 |
| 75. <i>Trifolium</i> spp. | 0.03 | 0.08 | 0.05 | 0.11 | 0 | 0 | 1.73 | 3.31 |
| 76. <i>Verbascum phoeniceum</i> | 0 | 0 | 0.20 | 0.44 | 0 | 0 | 0.11 | 0.24 |
| 77. <i>Veronica arvensis</i> | 0.46 | 0.68 | 0.79 | 0.83 | 1.31 | 1.58 | 0 | 0 |
| 78. <i>Veronica prostrata</i> | 0.97 | 0.85 | 0.65 | 0.64 | 0.31 | 0.44 | 0.23 | 0.33 |
| 79. <i>Veronica</i> spp. | 0.18 | 0.41 | 0.04 | 0.09 | 0.63 | 1.40 | 0.12 | 0.18 |
| 80. <i>Veronica</i> spp. seedling | 0 | 0 | 0.22 | 0.49 | 0 | 0 | 0.78 | 1.75 |
| 81. <i>Vicia angustifolia</i> | 0.39 | 0.56 | 0.05 | 0.12 | 0.55 | 0.86 | 0.09 | 0.13 |
| 82. <i>Vicia</i> spp. | 0 | 0 | 0 | 0 | 0 | 0 | 0.05 | 0.10 |
| 83. <i>Viola arvensis</i> | 0.11 | 0.18 | 0.04 | 0.09 | 0 | 0 | 0 | 0 |
| Total | 50 | | 51 | | 56 | | 44 | |