Thesis of doctoral (PhD) dissertation

EXPLORING THE POTENTIAL FOR RECULTIVATION OF DEGRADED SALINE GRASSLAND AND THE CHARACTERISTICS OF A TUSSOCK MEADOW AROUND KARCAG

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1. BACKGROUND AND OBJECTIVES OF THE DOCTORAL DISSERTATION

European grasslands are important biodiversity hot-spots (Hönigová et al., 2012), provide a range of provisioning, regulatory and cultural ecosystem services (Dengler et al., 2014) and serve as an important fodder base for livestock production (Bengtsson et al., 2019). These critical habitats are threatened by a number of factors. Despite the large number of grasslands used for agricultural purposes, many of them are not properly managed (Sartorello et al., 2020). First, the proportion of underutilised areas is significant, especially in mountainous areas where lack of management leads to afforestation (Valkó et al., 2018). This tendency is typical in many parts of Central and Eastern Europe and occurs in about 20% of Hungarian grasslands (Tasi et al., 2014). Secondly, overuse is also a typical phenomenon, especially in terms of over-intensive grazing (Sartorello et al., 2020). In Europe, permanent and temporary grassland cover 39% of agricultural land, which is a strong base for the livestock sector (De Vliegher et al., 2014). Between 1990 and 2019, we can track changes in the size of Europe's grassland area through the World Food and Agriculture Organization (FAO) statistical census, which shows that compared to 1990, the area of grassland increased by 1.09% in 2019. In Hungary, the agricultural statistics census of the Hungarian Central Statistical Office (KSH) goes back much earlier, to 1853, so we can track the changes in the grassland area of our country. Before the regime change, the area under grassland cultivation was 1 209 900 hectares, which decreased to almost 1 million hectares by the beginning of the millennium, and in 2010 the area under grassland cultivation was only 762 600 hectares. Grassland could in principle provide an extremely large fodder base, but as the importance of grazing-based ruminant farming has declined, its income-generating capacity and prestige have also been in crisis. In addition, the share of grassland in our country has been steadily declining (Harcsa et al., 2011). The decline in grassland area has also been attributed to the loss of land to set-aside (e.g., for building) (Vinczeffy, 1993). The success of grassland management is also increasingly affected by increasingly extreme meteorological conditions (Halász et al, This is due to the fact that the evapotranspiration (water consumption) coefficient of grassland is very high (Barcsák et al., 1978, Szemán, 2006A). At the same time, grasslands are predominantly under some kind of nature conservation classification (Molnár and Csízi, 2015), As a consequence, agrotechnical and utilization technologies should be applied in grassland management practices that result in a yield surplus that compensates for nutrient replenishment (Bajnok et al., 2011) and yield losses due to the abandonment of irrigation (Dér et al., 2003). The technological elements of professional grassland management were already developed in the second half of the last century (Barcsák-Kertész, 1986; Nagy, 1993), but with the liquidation of large farms and the 'retreat' of the pastoral system to national parks, it has become a marginal segment of the agricultural sector. The severe shortage of animal caretakers has led to an increase in the construction of pasture gardens on the pastures that are still in use. The pasture directly adjacent to the livestock farm, where animals graze on a 'self-service' basis, inevitably 'spoils' the time-strapped keeper, who, by not allowing the grassland time to regenerate (he should graze further away from the 'foot'), predictably overloads it. Valuable grassland components are selected out by grazing livestock, the soil becomes highly compacted and grass yields are minimal (Vinczeffy, 1993). Paradoxically, grasslands far from the livestock farm are at risk of underutilisation, which also leads to a loss of biodiversity through overgrazing. Grassland that is 'left standing' can be a hotspot for wildfires and scrub encroachment, and value is lost. Among the many problems facing the sector, my research aims to clarify the possibilities for improving the plant structure and increasing yields of extensive grasslands degraded by over- and undergrazing, and is therefore managed by the Karcag Research Institute. The results of these experiments can serve as a model for other grassland farmers with similar conditions.

In the light of the above, I have set the following objectives for my research:

- To clarify the possibilities of recultivation of an overgrazed ancient grassland with saline soil conditions and to investigate the fluctuation of the vegetation structure.
- To investigate the impact of different land-use types on underutilised saline grassland with solonyec soils and to instrumentally monitor carbon-dioxide-emissions and soil moisture changes.
- Clarifying and creating a database of the morphometric data of the tussock meadow, which has been developing for more than three decades as a result of the microrelief conditions and underutilization.

2. MATERIAL AND METHOD

I set up two different experiments and one study at a site representative of the site conditions, the University of Debrecen,Institutes for Agricultural Research and Agricultural Farming, Karcag Research Institute (hereinafter referred to as Karcag Research Institute), in a grassland area: an overgrazed area, an underutilized area, and a tussock area. The altitude of the experiments ranges from 82 to 83 m above sea level. The soil type of all three experimental sites is meadow solonyec soil. The experimental areas investigated belong to the Pannonian flora region, the Tiszántúli flora region of the lowland flora region (Hortobágyi and Simon, 2000). The study areas are classified in the *Achilleo-Festuceteum pseudovinae* and *Artemisio santonici-Festucetum pseudovinae* transitional grassland and Agrosti-Alopecuretum pratensis grassland associations. The study areas belong to the Natura 2000 network (Šefferová Stanová et al., 2018) and have been included in the National Agri-environmental Programme in several rotations.

The designated grassland in the overgrazed area is under the use of the Karcag Research Institute. The overgrazed area was excluded from sheep grazing in spring 2017. To set up the experiment, I designated 9 plots in three replicates, each 4×5 m (20m2) with 0.5 m paths between them. Treatments are labelled F for top seeding with English grass seed (0; 20; 40 kg/ha at standard rates) and K for nutrient replenishment with sheep manure-based Terrasol biocompost (permit number 02.5/48/7/2008) (0; 20; 40 t/ha at standard rates). Treatments labelled T/L1-3 are plots of the treatment still overgrazed by sheep (25 sheep/ha). Plots in the overgrazed area were top-seeded with Karcagi's eelgrass at 0 kg/ha, 20 kg/ha and 40 kg/ha rates. I applied to the plots of the overgrazed area a bio-graded compost made of deep-digested sheep manure of the brand "Terrasol", patented by the University of Debrecen, at a rate of 0 t/ha, 20 t/ha and 40 t/ha. The plants were recorded using the Balázs quadrat method (Balázs, 1949). The classification of plant species names was based on Király (2009). The economic classification of the recorded plant species was made according to the guidelines of Barcsák (2004), and after the cenological recording, each plant species was classified according to its ecological status into the categories of Social Behaviour Types (SBT) of Borhidi (1993). To determine the degree of degradation (Degree of Degradation - Dʄ) in the experiment, I used the ratio of the cover of species indicative of degradation to the cover of species indicative of natural conditions based on Borhidi's SBT categories, without taking into account the extent of uncovered areas. In each plot of the experiment, I mowed the grassland with a grasscutter, of which I kept 250 g of plant samples. These samples were tested in the accredited laboratory of the Research Institute in Karcag.

In 2009, an experiment was set up in the grassland area of the Karcag Research Institute to clarify the effects of changes in plant structure on underutilised natural grassland. Since 1987, the rest of the grassland area has been subject to extensive meadow management (1 mowing per year, followed by cattle grazing). I joined the research in 2017 and the results reported cover the period 2017-2020. At the start of the experiment, in 2009, the following 4 treatments were set up in 3 replicates, with a net area of 20 m2 (10.4m×2m) per replicate plot: zero tillage treatment: since 2009 the area is not used (designated A/Z); mulch treatment: since 2009, the area is left fallow in the 3rd decade of May and the mulch is left on the field (designated A/M); mowing treatment: once a year, on the 3rd of May. Mowing only in the 3rd decade of May, with removal of phytomass (designation: A/K); Grassland management: Mowing in the 3rd decade of May, haying, in August, sheep grazing on the heel (designation: A/R). The plants were recorded using the Balázs quadrat method (Balázs, 1949). The plant species names were classified according to Király (2009). The economic classification of the recorded plant species was made according to the guidelines of Barcsák (2004), and after the cenological recording, each plant species was classified according to its ecological status into the SBT categories of Borhidi (1993), and into the water demand (WB) and nitrogen demand (NB) categories. Furthermore, I calculated the degree of degradation based on the SBT values. Plant height was measured before utilization in 3 replicates per plot along an imaginary diagonal. Soil CO2 emissions were measured using a framework method developed at the Karcag Research Institute (Kovács, 2014). To measure CO_2 concentrations, I used a Testo 535 infrared gas analyser. For soil moisture and soil temperature measurements, I used an SMT-100 instrument, which measures the dielectric conductivity of the soil and calculates the moisture content, expressed as a percentage by volume.

The study area is located in the "Papere" tussocky area managed by the Karcag Research Institute. The so-called 'tussock meadow', which is the deepest part of the area with the lowest water table, has not been exploited since 1987. I randomly selected a 2×2 m quadrat in ten replicates in the area, in a location that was representative of the area. Within each quadrat I recorded the number of tussocks and measured the height and girth size of each tussock to the nearest centimeter. In addition to this, I calculated the number of tussocks per hectare for each quadrat.

The data collected in the experiments were recorded and summarised, and the results were processed and evaluated using Microsoft® Office Excel. For data analysis, I used one-factor analysis of variance (ANOVA). Analysis of variance is used to determine whether there is a significant difference between the means of two groups. It is important to note, however, that statistical analysis does not show where the difference between the means of the two groups lies. For the statistical evaluation, I used the p-value of the elements of the analysis of variance ("SS" is the sum of the squares of the variance of the factors, "DF" is the degree of freedom, "MS" is the variance, "F" is the calculated F-value, "p-value" is the probability associated with the calculated F-value, "F crit" is the critical F-value) at a 5% significance level.

3. RESULTS

3.1. Exploring the potential for recultivation of overgrazed grassland

3.1.1. Results of cenological surveys

During the cenological recording I found that in the fenced area, which had been converted to mowing, the dominant plant was the lean fescue (Festuca pseudovina), while in the overgrazed area the dominant plant was the mouse-grass (Hordeum murinum).

Furthermore, I found that the meadow area was the most diverse in terms of plant species, with an average of 18 species recorded, compared to an average of 13 in the area converted to mowing.

During recultivation, over-seeding with Lolium perenne was unsuccessful in plots coded F20-F40 in the experimental area. This highly stress-tolerant grass species was unable to find a habitat in the natural grassland of the highly saline, shallow soil with a natural grassland association. The failure of overgrowth confirms Vinczeffy's (1993) finding on the stability of the plant species structure of ancestral grasslands, that new species are difficult to "let in" to the dominant species of these grassland associations. However, I was able to measure treatment effects in plots of treatments coded K20-K40 where biocompost was applied.

When including Poaceae, I found that their cover rate increased over the study area (p-value of plots receiving 0 t/ha of compost was 1.105E-09, p-value of plots receiving 20 t/ha of compost was $1.990E^{-12}$, and p-value of plots receiving 40 t/ha of compost was $1.182E^{-11}$). These statistical values indicate that my results are significant. From my results, it can be concluded that, on all replicates of the treatments excluded from grazing (F0K0/1-F40K40/3), there was an average increase in Poaceae cover from 2017 to 2020, regardless of the amount of compost spread on the field. One year after the cessation of overgrazing, Elymus repens appeared, which, as a wheat-grass species, survived the strong species selection during overgrazing in a 'latent' state. It can also be seen that, on average, the overgrazing treatment continued to reduce the cover of grasses, with a value of 14.06-17.19% in 2020. In these plots, the average reduction in Poaceae cover between 2017 and 2020 was 50.30%. The analysis of variance also showed a significant result for the period under study (p-value: 0.011).

During the experiment, compared to the treatments excluded from grazing, Poaceae cover was higher in the plots receiving compost. Statistical analysis showed a statistically verifiable relationship only when comparing plots receiving 0 t/ha and 40 t/ha of compost in 2019 (p-value: 0.030) and plots receiving 0 t/ha and 20 t/ha of compost in 2020 (p-value: 0.048). However, the proportion of Poaceae was lower in the area still overgrazed than in the areas excluded from grazing, which can be explained by the selection of grazing animals (0 t/ha p-value: $8.950E^{-05}$ - 2017; 0.002 - 2018; $1.380E^{-06}$ - 2019; $1.280E^{-08}$ - 2020. 20 t/ha p-value: 0.011 - 2017; 0.0001 - 2018; $4.270E^{-06}$ - 2019; $1.100E^{-11}$ - 2020. 40 t/ha p-value: 0.005 - 2017; 0.0001 - 2018; $9.796E^{-06}$ - 2019; $4.100E^{-10}$ - 2020). In plots receiving compost, the 40t/ha treatment plots had higher Poaceae cover than plots receiving 20 t/ha compost, which showed no statistically verifiable correlation in the analysis. The cover of Poaceae could be due to the abandonment of overgrazing.

When recording the Fabaceaes, I found a p-value of 8.842E-06 for plots receiving 0 t/ha of compost, 0.0004 for plots receiving 20 t/ha of compost, and 0.184 for plots receiving 40 t/ha of compost. These statistical values indicate that the results for plots receiving 0 t/ha and 20 t/ha of compost are significant, i.e., there is a reduction in Fabaceae cover in this area. However, no such correlation was found in fields containing 40 t/ha of compost. From my results, it can be concluded that Fabaceae cover decreased in all replicates of the treatments excluded from grazing (F0K0/1-F40K40/3) and in the area still overgrazed from 2017 to 2020, regardless of the amount of compost applied. Also in the plots of the area that continued to be grazed, Fabaceae cover decreased by an average of 66.67% between 2017 and 2020. This phenomenon can be explained by the lack of rainfall in 2018-2020. Analysis of variance showed no significant results (p-value: 0.184) over the period studied.

The area still overgrazed was compared with the area under treatments that had been converted to mowing. The areas converted to mowing had higher Fabaceae cover than the area that remained overgrazed, although they showed a decreasing trend. Statistical analysis showed no significant difference between the treatments receiving 0 t/ha and 40 t/ha compost during the experimental years, but a significant difference was found when comparing the treatment with 20 t/ha compost with the treatment that continued to be overgrazed from 2018 onwards (2018 p-value: 0.041; 2019 p-value: 0.033; 2020 p-value: 0.033). In the areas converted to mowing, statistical analysis showed a statistical difference in 2018 when comparing areas receiving 0 t/ha and areas receiving 40 t/ha of compost, i.e. statistically verifiable increase in butterfly cover in areas receiving 40 t/ha of compost in 2018 compared to areas receiving 0 t/ha of compost. When comparing the area receiving 0 t/ha and the area receiving 20 t/ha of compost, the analysis showed no such correlation. The switch to mowing has resulted in the elimination of the manure effect, but also in the elimination of plant selection by grazing. This beginning of a new grassland structural equilibrium after the radical change is reflected in the measured results.

I used analysis of variance to verify how the conditional weed cover changed in the areas converted from grazing to mowing over the study period. Plots receiving 0 t/ha compost had a p-value of 0.590, plots receiving 20 t/ha compost had a p-value of 0.0002, and plots receiving 40 t/ha compost had a p-value of 0.0004. These statistical values show that the results for the plots with 20 t/ha and 40 t/ha compost are significant, i.e. the conditional weed cover was reduced in this area, whereas no such correlation was found in the plots with 0 t/ha compost. From my results, I found that all replications of the treatments excluded from grazing (F0K0/1-F40K40/3), from 2017 to 2020, showed a decrease in the cover of conditional weeds, regardless of the amount of compost applied. The reason for this is explained by the increasing cover of turfgrass species during this period, increasing their habitat at the expense of the conditional weeds. In the plots of the area that continued to be grazed, the average reduction in conditional weed cover between 2017 and 2020 was 27.09%. In this treatment, however, the increase in cover of unconditional weeds with high vigour could have been a competitor. The analysis of variance showed no significant results (p-value: 0.255) over the period studied.

The area still overgrazed was compared with the area under treatments that had been converted to mowing. The continued overgrazed areas had a higher proportion of conditional weed cover than the area converted to mowing, but showed a decreasing trend across the experiment. In the still overgrazed area, there is a statistically verifiable decrease in unconditional weed cover year on year (presumably due to lack of Poaceae and Fabaceae preceding sheep grazing), as well as in the fenced treatments, probably due to the increase in Poaceae cover (0 t/ha: 2017 p-value: 0.0002; 2018 p-value: 0.0008; 2019 p-value: 0.0001; 2020 p-value: 0.002. 20 t/ha: 2017 p-value: 0.005; 2018 p-value: 2.97E-05; 2019 p-value: 1.12E-06; 2020 p-value: 7.7E-07. 40 t/ha: 2017 p-value: 0.0003; 2018 p-value: 1.43E-05; 2019 p-value: 5.31E-07; 2020 p-value: 3.23E-06). When comparing the treatments in the fenced area, I found that there might have been some effect of compost in 2018, as the statistical analysis showed a strong correlation between the areas receiving 0 t/ha and 20 t/ha and between the areas receiving 0 t/ha and 40 t/ha of compost (20 t/ha p-value: 0.020; 40 t/ha: p-value: 0.003). In the other cases, the statistical analysis showed no correlation.

I also verified how the cover of uncontrolled weeds changed in the areas converted from grazing to mowing over the period, with a p-value of 2.745E-11 for plots receiving 0 t/ha of compost, 2.666E-12 for plots receiving 20 t/ha of compost, and 8.530E-07 for plots receiving 40 t/ha of compost. From my results, I found that all replicates of the treatments excluded from grazing (F0K0/1-F40K40/3) from 2017 to 2020 showed an average reduction in uncontrolled weed cover (especially Hordeum murinum and Bromus hordaeceaus), regardless of the amount of compost applied. This can be explained by the fact that the economically valuable grass species, free from grazing selection, have undergone significant biotope recovery. However, the area that remains overgrazed has seen an average increase in uncontrolled weed cover of 107.60% between 2017 and 2020. Analysis of variance showed a significant result (p-value 3.574E-05) over the period studied.

The area still overgrazed was compared with the area under treatments that had been converted to mowing. The remaining overgrazed areas had a higher percentage of uncontrolled weed cover than the area converted to mowing. In the overgrazed area, the cover of unconditioned weeds increased year by year, statistically verifiable (explained by sheep avoiding them, they were free to spread), while in the areas converted to mowing, it decreased, probably due to the cutting to one-crop, (0 t/ha: 2018 p-value: 8.81E-08; 2019 p-value: 1.06E-09; 2020 p-value: 1.8E-11. 20 t/ha: 2017 p-value: 0.037; 2018 p-value: 1.69E-08; 2019 p-value: 4.77E-10; 2020 p-value: 5E-12. 40 t/ha: 2017 p-value: 0.039; 2018 p-value: 2.62E-05; 2019 p-value: 3.84E-11; 2020 p-value:9.86E-13). When comparing areas still overgrazed and areas converted to mowing, no statistically significant correlation was found for areas 0 t/ha in the first year (2017) (p-value: 0.124). When comparing treatments within the fence, no correlation was found in the statistical analysis.

The Prunus domestica subsp. syriaca, belonging to the group of trees and shrubs, was displaced from the study area in 2019, which can be explained by the emergence of mechanical mowing.

3.1.2. Degradation during recultivation of overgrazed area

Degree of degradation was calculated on the basis of the cover of plants classified according to SBT.

Between 2017 and 2020, the degradation rate decreased by 63.41% on average in plots receiving 0 t/ha of compost, while the degradation rate decreased by 57.34% on average in plots receiving 20 t/ha of compost and by 58.39% on average in plots receiving 40 t/ha of compost.

Analysis of variance showed that the degradation rate decreased in the areas converted from grazing to mowing over the period studied. Plots receiving 0 t/ha of compost had a p-value of 6.62E-10, plots receiving 20 t/ha of compost had a p-value of 1.24E-07, and plots receiving 40 t/ha of compost had a p-value of 7.02E-08. These statistical values indicate that my results are significant, i.e., degradation has decreased in this area. From my results, it can be concluded that for all replicates of the treatments excluded from grazing use (F0K0/1-F40K40/3), there was a statistically verifiable reduction in degradation from 2017 to 2020, regardless of the amount of compost spread on the area.

It can also be seen that, at the same time, the degree of degradation increased steadily for the still overgrazed treatment, reaching a Dʃ value of 3.43-5.00 in 2020. In these plots, the degradation rate increased by 182.99% on average between 2017 and 2020. The analysis of variance also showed a significant result (p-value: 0.004) over the period.

My results suggest that a change from grazing to once-yearly mowing, i.e. a change in the land use pattern, could help in the reclamation of the area. Comparing the mowed (0 t/ha) and overgrazed plots, a significant difference was found over the experimental years (p-value 2017: 0.0004; p-value 2018: 5.58E-05; p-value 2019: 2.94E-07; p-value 2020: 7.61E-08). Comparing the plots treated with 20 t/ha and 40 t/ha with the plots still overgrazed, the statistical analysis showed a similar significant correlation. Within the fenced area, there was no detectable correlation between treatments.

3.1.3. Results of dry matter yield measurements

The dry matter content (m/m%) was measured at the Central Laboratory of the Research Institute in Karcag, from which the dry matter yield (kg/ha) was calculated.

The dry matter yield values in the still overgrazed area were higher than the dry matter yields tested in the area converted to mowing, with the exception of 2017, when the raw protein yields of the plots receiving 20 t/ha and 40 t/ha of compost were higher. In the analyses, the difference in 2017 was not significantly detected (p-value 20 t/ha: 0.950; p-value 40 t/ha: 0.909).

In the 0 t/ha, 20 t/ha and 40 t/ha treatments, the average dry matter yield decreases from year to year, except for the 20 t/ha treatment in 2018. The decrease in dry matter yield of treatments from year to year was demonstrated by analysis of variance, with results significant only for the 0 t/ha and 20 t/ha treatments in the 2019-2020 vintage (0 t/ha p-value: 0.007; 20 t/ha p-value: 0.026).

In any case, the crop composition should be taken into account when evaluating yield results. In the still overgrazed area, the high cover values of unconditionally unforageable but high phytomass weeds, such as thistles, determined the yield values.

3.1.4 Results of measurements of crude protein yields

Results of crude protein content (m/m%) measured in the experiment, from which crude protein yields (kg/ha) were calculated.

Although the crude protein yield of plants is determined by dry matter yield, my analysis showed an interesting trend. In the still overgrazed area, the values are higher than the crude protein yields tested in the area converted to mowing. Using analysis of variance, I verified that the crude protein yield in the still overgrazed area was higher than the results found in the 0 t/ha treatment, with statistical analysis showing a significant correlation (2017 p-value: 0.001; 2018 p-value: 1.28E-05; 2019 p-value: 6.49-E-07; 2020 p-value: 6.2E-10). Compared to plots that continued to be overgrazed and received compost, the difference in 2017 was not statistically verified (20 t/ha p-value: 0.123; 40 t/ha p-value: 0.920).

In the treated plots, the crude protein yield was lower in 2018 than in 2017, higher in 2019 than in 2018, and decreased again in 2020. In the still overgrazed area, the crude protein yield was higher in 2018 and 2019 than in 2017 and decreased again in 2020. I verified my claim by analysis of variance. I found a positive correlation between treatments 40 t/ha (p-value: 0.006) in 2017-2018, 0 t/ha (p-value: 0.010), 40 t/ha (p-value: 0.044), T/L (p-value: 0.043) in 2018-2019, and 0 t/ha (p-value: 0.0009), 20 t/ha (p-value: 0.015) in 2019-2020.

The results can be explained by the unconditional weed density of the plots that were still overgrazed.

3.2. Impact of land use practices on underutilised grassland

3.2.1. Results of coenological surveys

During the coenological recordings, it was found that the mulched treatment area (A/M) has undergone a change of control plants in 2019. Alopecurus pratensis Poa pratensis subsp. angustifolia was replaced by as the dominant grass species. Rosa canina appeared in the zero use area (A/Z). The meadow use area (A/R) also saw a change of lead plant in 2018.

Alopecurus pratensis was replaced by a predominance of Festuca pseudovina cover. Alopecurus pratensis was able to maintain its dominance under the influence of underutilisation as a stubborn stitchgrass in mown, mulched and unutilised experimental plots. In these areas only the main crop is exploited. However, the fenced grassland outside the meadow, which is used for the meadow grass treatment and protects the other treatments, is used twice a year, after the main crop is mown in May, and in August, the field is used for cattle grazing. It is likely that the grazing to low stubble height and the trampling and dung effects are increasing the proportion of undergrowth cover, mainly of Festuca pseudovina.

Furthermore, I found that the meadow-utilisation area was the most diverse, with an average of 21 species found in the area, while the zero-utilisation area had the fewest plant species (6).

I measured similar cover data for Poaceae in each plot. When comparing treatments, the cover of Poaceae in the meadow areas was lower than in the fenced, underutilised areas. When comparing the mulched and meadowed plots, analysis of variance showed a strong correlation in each year of the experiment (2017 p-value: 0.016; 2018 p-value: 0.010; 2019 p-value: 0.027; 2020 p-value: 0.011). Furthermore, statistical analysis also showed a strong correlation when comparing mowed and grassland areas (2017 p-value: 0.009; 2018 p-value: 0.019; 2019 p-value: 0.025; 2020 p-value: 0.012). However, it can be concluded that the cover value of a grass species varied depending on the degree of overgrowth, with almost only the fibrous grasses being able to successfully outgrow the thick avar plate, similar to the research findings of Nagy (2001). In the zero utilisation area, I looked at the cover ratio of undergrowth and fibre grasses, and the results showed that fibre grasses had a higher cover ratio in this area, but I could only confirm this relationship in 2017 (p-value: 0.002).

When recording Fabaceae, the highest values were recorded in the meadow area. The cover values of the light-demanding Fabaceae flowered grassland species decreased from meadow use to zero use. This phenomenon shows similarities with the results of Da Ronch (2002) and Tóth et al. (2002) in their studies on natural grasslands, although I did not find a verifiable correlation in the analysis of variance. When comparing mulch and zero tillage sites, I found higher Fabaceae cover in the mowing treatment in 2019 (p-value: 0.007). Comparing mulch and meadow tillage sites showed a statistically verifiable relationship in 2017 (p-value: 0.031) and 2020 (p-value: 0.006). When comparing mowing and meadow use treatments (2017 p-value: 0.041, 2019 p-value: 0.025, 2020 p-value: 0.006) and comparing no-till and meadow use treatments (2017 p-value: 0.031, 2019 p-value: 0.006, 2020 p-value: 0.003), I found that meadow use treatments had higher Fabaceae cover. Analysis of variance showed a positive correlation in all years of the experiment except 2018.

The highest cover values for the grassland components classified as conditional weeds were recorded in the meadow area. In addition to light-demanding, medicinal grass species such as Achillea collina, I also included pannonic endemic species such as Plantago schwarzenbergiana. Based on the perspective of the cover data, I compared the cover of conditional weeds in the zero utilization and meadow utilization areas, which confirmed that the meadow utilization areas have a higher cover of conditional weeds. When comparing the mulch and zero tillage treatments, I found that the mulch treated area had higher cover of conditional weeds than the zero tillage area. I statistically verified my result in 2019 using analysis of variance (p-value: 0.047). When comparing mowing and meadowland treatments, statistical analysis showed a positive correlation in all experimental years except 2019 (2017 p-value: 0.0005; 2018 p-value: 0.017; 2020 p-value: 0.04). When comparing mulch and grassland treatments, statistical analysis showed a positive correlation throughout the experiment (2017 p-value: 0.005; 2018 p-value: 0.013; 2019 p-value: 0.006; 2020 p-value: 0.005). When comparing zero tillage and meadow tillage treatments, analysis of variance showed a positive correlation throughout the experiment (2017 p-value: 0.0005; 2018 p-value: 0.010; 2019 p-value: 0.004; 2020 p-value: 0.004).

The highest cover values for uncontrolled weed species were also recorded in the meadow area. When comparing the treatments, I found that the meadow-utilised areas had a higher cover of uncontrolled weeds than the underutilised areas. When comparing mulched and meadow-utilized areas, statistical analysis showed a strong correlation in 2020 (p-value: 0.024). When comparing mowing and meadow-utilized treatments, analysis of variance showed a strong correlation in 2018 (p-value: 0.016). When comparing zero tillage and meadow use, analysis of variance showed statistically valid results in 2018 (p-value: 0.025) and 2020 (p-value: 0.033).

In the mulch-treated (A/M) plots, woody plants were recorded during the study period, exclusively Rosa canina, a species of prickly pear, avoided by animals, with an overstory of 3.13%. In the zero-tillage (A/Z) area, the woody plants varied in cover values from 18.75-34.38% in 2017, 12.50-31.25% in 2018, and 25.00-37.50% in 2019, with Balázs dominance values. The high woody stem cover values in areas that have not been used for a decade support the finding of Barcsák et al. (1978) that "natural succession processes are revived in abandoned grassland", i.e. the scrub formation process has started, confirming the "green desert" theory of Perevolotsky and Seligman (1998) and the results of Szentes et al. (2012) and Bajor et al. (2016). No woody stem plants were recorded in the mowed (A/K) and meadow (A/R) areas, probably due to mechanical mowing. The explanation for the high presence of grasses measured at zero utilisation is that in 1989 the Research Institute planted forest strips in the pasture where my experiment was carried out, in order to protect the sheep from the wind. The shrub layer of the woodland strips consists of turfgrass, the seeds of which were dispersed by the droppings of wild birds (ornitochoria) that regularly landed on the marker cones of the experiment. I found no sense in detecting a correlation in the statistical analyses, since without turfgrass prapagulum the reproduction of the experiment is doubtful.

3.2.2. Evolution of the degradation rate under different types of recovery

Degree of degradation was calculated on the basis of the cover of plants classified according to SBT. The highest degradation was measured in the zero utilization treatment in 2020 (Df: 0.568). The lowest degradation was measured in the mulch utilization treatment in 2017 (Df: 0.157). Degradation increased by 3.81% in the mulch utilization treatment between 2017-2018, increased by 49.59% between 2018-2019 and decreased by 16.37% between 2019-2020. In the mowing treatment, the degradation rate decreased by 4.59% between 2017-2018, increased by 123.93% between 2018-2019, and decreased by 10.57% between 2019-2020. In the zero tillage treatment, degradation increased by 12.23% between 2017-2018, 45.52% between 2018-2019 and 19.31% between 2019-2020. Degradation in the meadow use treatment increased by 37.64% between 2017-2018, 12.37% between 2018-2019 and 8.57% between 2019-2020. It can be concluded that the degradation rate increases year by year in the treatments, the statistical analysis of which showed no correlation. When comparing treatments, analysis of variance did not show a correlation for any of the treatments. When evaluating the results, it should be noted that the plant species diversity of all the treatments studied could have been negatively affected by the dry-semiarid climate index values that determined the experimental period.

3.2.3 Results of measurements of carbon dioxide concentration, soil moisture and soil temperature

The determination of soil carbon dioxide emission, soil moisture and soil temperature were measured once in 2017 and 2018 (29.06.2017 and 06.06.2018), while in 2019 and 2020, due to the perspective results and the better access to the measuring instrument, the measurements were performed every two weeks (28.03.2019 - 10.01.2019 and 07.04.2020 - 10.09.2020).

In 2017, soil temperature values were higher in the mowing and meadow use areas and soil moisture values were found to be higher in the mulch and zero use areas. Carbon emission values were highest in the zero tillage area. When comparing the carbon emission values of the treatments, a significant difference was found between the zero tillage and mowing treatments (p-value: 0.009).

In 2018, the "mowing" treatment and the "meadow use" area have higher soil moisture values than the "mulch" and "zero use" treatments. These data, which are inconsistent with those of the previous year, can be explained by the fact that a total of 20.6 mm of precipitation fell in the week before sampling, after a prolonged dry period, so that the leafy phytomass absorbed part of the precipitation in the "mulch" and "zero tillage" treatments, while in the plots of the other treatments (mowing, meadow) the lack of leafy phytomass allowed precipitation to infiltrate into the soil. However, the mowing treatment and the meadow treatment had lower soil temperature values, probably due to precipitation cooling the soil at the measurement level. When carbon emissions were determined, the following was observed: the "mulch" and "zero use" treatments had higher values than the "mowing" treatment and the "meadow use" treatment. Similar results were obtained by Kovács et al. (2013) in saltmarsh grasslands. Carbon dioxide emissions are influenced by soil organic matter content. In areas covered with mulch or straw, microbial activity is higher, as the mulch or straw decomposes and is released as organic matter. Significant differences were found when comparing the carbon emission values of the soils of the treatments between the zero tillage and mowing treatments (p-value: 0.005), between the zero tillage and meadow tillage treatments (p-value: 0.003), between the mulch and mowing treatments (p-value: 0.004) and between the mulch and meadow tillage treatments (p-value: 0.002).

In my 2019 study, the highest values of carbon dioxide emissions, for all treatments, were obtained between 23 May and 06 June.

In the zero and mulched treatments, the soil surface is covered by a crust of dead phytomass tissue, and in the mowed treatment, the dense leafy growth, which is not suitable for mowing, can reach a shade height of ~ 15 cm by July. However, in meadow utilization, the leaf litter was grazed by sheep, so the drying effect of the sapping solar heat may have been more pronounced, and the living conditions for carbon dioxide-producing microbes may have been less favourable, and they were least likely to have access to dead organic matter sources in meadow utilization. Based on the data from our experiment, it is likely that grassland under grazed (meadow) use also contributes less carbon dioxide to the atmosphere than grassland left fallow. When comparing the carbon emission values of the treatments, the following results were obtained. At measurement 11 (15 August 2019), when comparing mulch and meadow cover treatments and mulch and mowing treatments, the carbon emission values of the mulch treatment were higher (p-value: 0.005 and 0.003, respectively). At the same time of measurement, when comparing zero tillage and meadow tillage treatments, and zero tillage and mowing treatments, the carbon emission values of the zero tillage treatment were higher (p-value: 0.023 and 0.009, respectively). When comparing meadow tillage and mowing treatments, the carbon emission values of the meadow tillage treatment were higher (p-value: 0.038). In the 13th measurement (11 September 2019), when comparing mulch and meadow cover treatments and mulch and mowing treatments, the carbon emission values of the mulch treatment were higher (p-value: 0.001 and 0.028, respectively). At the same time of measurement, when comparing zero tillage and meadow tillage treatments and zero tillage and mowing treatments, the carbon emission values of the zero tillage treatment were higher (p-value: 0.007 and 0.025, respectively). When comparing the mowing treatment to the meadow use treatment, the carbon emission values of the mowing treatment were higher (p-value: 0.013). At measurement date 14 (1 October 2019), when comparing the meadow use treatment to the mowing treatment, the carbon emission values of the mowing treatment were higher (p-value: 0.046).

I measured the soil moisture and temperature of the treatments at 0-10 cm depth at the same time as the carbon dioxide measurements. I measured soil moisture of 16.23% on 8 May 2019, which is the moisture content in the mowing treatment plots, as 74.7 mm of rain had fallen in the two weeks prior to that date. Due to the results of mulch technologies spreading in the agricultural field due to climate change, I predicted parallel ranging values for carbon dioxide emissions and soil moisture (Zsembeli et al. 2015). But due to the accumulation of grass clippings for the 10th year and unusually long periods without precipitation due to climate change, the results of our measurements did not follow the "paper form" on several occasions.

The extremely long period of drought in the summer months was interrupted by a 22 mm rainfall in the first decade of September, before our measurement date of 11 September. The highest values of carbon dioxide emissions were then measured for zero and mulched grassland use, the lowest for mowing and meadow use. The soil moisture values, on the other hand, show the opposite order. The accumulation of avarnemez in the zero and mulched grassland systems, after a prolonged period of aridity, retains the amount of precipitation that falls, up to a certain limit, from reaching the soil. Nevertheless, the avarnemez, which continuously feeds soil bacteria, can result in high soil biological activity that produces high levels of carbon dioxide even at lower soil moisture values, as shown by Kovács et al. (2013). Thus, a similar phenomenon is observed with the results of the June 2018 measurements.

In 2020, I was not able to measure any data on 2 occasions during the measurement period, as a total of 206.1 mm of precipitation fell between 18 June and 27 July 2020, which could not be leached into the soil.

At the same time as the carbon dioxide measurement dates, I measured the temperature and moisture content of the soil in the treatments at a depth of 0-10 cm, similar to 2019. Due to the high rainfall in summer, I measured the highest soil moisture data in the mulch treatment: 21.30% (28 July 2020). Soil temperature data were broadly even throughout the year's measurement period, with the highest temperature data in the zero tillage plots (39.77°C) measured on 3 June.

When comparing the carbon emission values of the treatments, the following results were obtained: in the 2nd measurement (21 April 2020), when comparing mulch and meadow-applied treatments and zero tillage and meadow-applied treatments, the carbon emission values of the meadow-applied treatment were higher (p-value: 0.016 and 0.009, respectively). At the same time of measurement, when comparing zero tillage and mowing treatments and mulch and mowing treatments, the carbon emission values of the mowing treatment were higher (p-value: 0.024 and 0.039, respectively). At the 5th measurement (3 June 2020), when comparing the zero tillage treatment with the meadow mowing treatment and the zero tillage treatment with the mowing treatment and the zero tillage treatment with the mulch treatment, the carbon emission values of the zero tillage treatment were higher (p-value: 0.004, p-value: 0.012, and p-value: 0.0007, respectively). At measurement 9 (28 July 2020), when comparing zero tillage and mowing treatments and zero tillage and mulch treatments, the carbon emission values of the zero tillage treatment were higher (p-value: 0.033, p-value: 0.021).

The results of my tests confirmed the results of Kovács and Szőllősi (2008); Zsembeli et al. (2015); and Birkás, (2017) in field studies, which showed that the carbon emission was higher in mulch-covered areas (average carbon emission rate between 2017 and 2020: mulch (A/M) treatment: 0.60 g×m-2×h-1, zero use (A/Z) treatment: 0.68 g×m-2×h-1).

To summarise, the key to the reclamation of underutilised grassland with similar site conditions is the removal of excess phytomass from the site. Mowing (average carbon emission rate from 2017-2020: 0.58 g×m-2×h-1) or meadow use (average carbon emission rate from 2017-2020: 0.57 g×m-2×h-1), the CO2 emissions are lower than in the case of fallow grassland (average carbon emission rate from 2017-2020: 0.68 g×m-2×h-1), thus reducing the carbon emission rate and preserving the vegetation structure of the association. Statistical analysis comparing the treatments over the study period (2017-2020) showed no significant difference.

3.3. Investigation of the properties of underutilised solonyec-type tussocks

3.3.1. Number of tussocks per unit area

As a first step in the creation of a surrogate database, in 2019 I estimated the number of tussocks per sampling quadrat. The number of coleoptiles varied between 9 and 19 per survey quadrat, with an average of 14.60 tussocks per quadrat. Based on these data, I determined the possible number of tussocks per hectare in each quadrat. I determined the number of tussocks per hectare to be between 22500 and 47500, with an average of 36500 tussocks per hectare (Table 1).

Table 1: Number of tussocks per plot and per hectare (Karcag, 2019)

Area	Number/4m^2	Number/ha
Tussock 1	14	35000
Tussock 2	11	27500
Tussock 3	14	35000
Tussock 4	9	22500
Tussock 5	13	32500
Tussock 6	16	40000
Tussock 7	16	40000
Tussock 8	17	42500
Tussock 9	17	42500
Tussock 10	19	47500
Average	**14,6**	**36500**

3.3.2. Results of morphometric recording of tussocks deposits

The dimensional parameters (height and waist size) of the tussocks were investigated in 2019 and 2020. Based on the data from 2019, I found that the waist size of the tussocks ranged from 52.00 cm to 160.00 cm during the measurements in 2019, with an average value of 130.63 cm. Their height varied from 12 cm to 40 cm, with an average of 26.72 cm. In 2020, I determined that the height of the tussocks measured varied from 14.00 cm to 44.00 cm, with an average of 27.83 cm. The waist size of the tussocks ranged from 52.00 cm to 169.00 cm, with an average value of 134.40 cm. From 2019 to 2020, the size parameters of the tussocks increased, both in height and waist size. On average, the height of the tussocks increased by 4.01%, while their waist size increased by 2.81%. I used a one factor analysis of variance to detect the change in both cases. The results show that no significant result was found in either case (p-value for height: 0.606 and p-value for waist size: 0.699).

4. NEW SCIENTIFIC RESULTS OF THE DISSERTATION

1. 1. I found that the four-year exclusion of grazing and the switch to mowing resulted in positive changes in the rehabilitation of the vegetation structure of the overgrazed Achilleo setaceae–Festucetum pseudovinae saline grassland. The cover of Poaceae increased by 53.75% (from an average of 52.26% to 80.03%) between 2017 and 2020 due to the change of use.
2. 2. the overgrazing of the overgrazed Achilleo setaceae–Festucetum pseudovinae saline grassland vegetation structure may pose a serious threat to the grazing sheep flock, due to the increase in the cover of Hordeum murinum, which is a serious threat to animal health, by 40.91% (from 13.54% to 22.92% on average) between 2017 and 2020.
3. 3. I found that in the overgrazing Achilleo setaceae–Festucetum pseudovinae saline grassland community, changing the grassland use (from grazing to mowing) could be the reclamation solution to stop degradation, as the degree of degradation of grassland converted to mowing decreased by 63.41% (from 0.533 to 0.188) between 2017 and 2020, stably below the critical threshold (1.0).
4. 4. the degradation rate of the overgrazed Achilleo setaceae–Festucetum pseudovinae saline grassland association increased by 182.99% (from 1.53 to 3.95) between 2017 and 2020, which is four times the critical threshold (1.0) in the 7th year of overgrazing.
5. The dominant grasses of the Agrosti-Alopecuretum pratensis saline grassland have a high cover even 11 years after the abandonment of the use, indicating the stability of this semi-natural grassland association.In the period 2017-2020, the average cover of Alopecurus pratensis was 50.78% and that of Poa pratensis subsp. angustifolia was 19.01%.
6. Saline grassland with meadow utilisation contributes less carbon dioxide to the atmosphere (average 0.57 g×m-2×h-1 in summer 2017-2020) than unutilized grassland (average 0.68 g×m-2×h-1 in summer 2017-2020).
7. I recorded in a database the numerical and size characteristics of a tussock meadow field of the meadow-solonyec soil type, which has been developing for more than three decades on a tussocky meadow association. I have found that the average number of tussocks per hectare in the tussock meadow I studied is 36500.

5. PRACTICAL RESULTS

1. The results of my thesis demonstrate the necessity of an optimal choice of grassland utilisation methods in grassland areas with similar site conditions to avoid both over- and under-utilisation.

2. In order to maintain the grassland as a crop in the long term, a periodic change of use may be justified, as was the case with the recultivation of the overgrazed grassland in my thesis. The preservation of the vegetation structure of our natural grasslands is a prerequisite for practical grassland management.

3. The occasional introduction of mowing in permanently grazed areas can, according to my results, also be effective in the control of weed species that are a threat to livestock.

4. The accumulation of grassland phytomass has, among other negative effects on plant stand structure and increases greenhouse carbon dioxide emissions. My results support the view that it is dangerous to maintain accumulated duff on grassland used for farming for many years.

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