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Restoration of lowland meadows in Austria: A comparison of five techniques

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Abstract

European environmental policy mandates that biodiversity loss should be halted through restoration. However, knowledge about the efficacy of different restoration treatments for lowland meadows is still incomplete. Our study monitored two restoration projects in South-East Austria that served as compensation measures for the loss of species-rich grassland. We compared the efficacy of five restoration techniques: (1) sod transplantation, (2) natural colonization, (3) hay transfer and additions of seed mixtures for (4) wet and (5) bare soils. Over three years, we measured species richness, number of target species, Shannon diversity and similarity to reference sites. We asked: (A) What is the most effective technique for the restoration of lowland meadows? and (B) Is the applied restoration method more important than abiotic site conditions? We included 66 plots (reference and donor sites: 8 plots, restoration sites: 58 plots) in our study. We sampled data on species composition (4 m × 4 m plots) in three consecutive years since restoration initiation, estimated the slope inclination and analyzed soil parameters (K, P, pH). In general, species composition developed towards the reference vegetation for all techniques but sod transplantation produced by far the best result in terms of species richness and similarity to reference sites. By comparison, hay transfer and natural colonization produced intermediate results but performed better than seeding; the latter led to homogenous, species-poor swards. Soil preparation and abiotic site conditions played a minor role in this early stage of the restoration process, though these factors may gain importance in a longer time frame. We found sod transplantation to be a superior method for lowland meadow restoration in our study area but managers must consider its destructive nature and high costs, which might outweigh its benefits. In this light, hay transfer and natural colonization – or a combination of different techniques – could provide less destructive and more cost-effective alternatives.

Keywords: Compensation measures; Hay transfer; Seeding; Sod transplantation; Topsoil removal

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Introduction

Lowland meadows are important habitats for many plant and animal species in Europe (Habel et al. 2013). They provide significant ecosystem services, including water retention and filtration and have been a source of hay for livestock feeding for many centuries (Hopkins & Holz 2006; Stoate et al. 2009). In many European countries, hydrological perturbation and land use change have led to a decline of lowland meadow area and frequency, and have altered their species composition (Poschlod, Bakker, & Kahmen 2005; de Snoo, Nau, Verhulst, van Ruijven, & Schaffers 2012). This problem is particularly true for South-East Austria, where lowland meadows are critically endangered habitats (Essl, Egger, Karrer, Theiss, & Aigner 2004). Consequently, the restoration of lowland meadows has become a major goal in the last two decades. It aims to counteract the decline of this vegetation type (EC 1992) and to compensate for land degradation (EC 2014), e.g. when new urban areas or roads are built (Conrad & Tischew 2011).

In the last century, restoration of grasslands focused mostly on re-establishing the structure of meadows, giving little heed to site-specific species composition or local provenance (Kiehl, Kirmer, Donath, Rasran, & Hölzel 2010). Often, this led to species-poor communities with low conservation value and little similarity to traditionally managed grasslands (Conrad & Tischew 2011). Nowadays, authorities in Austria and Germany often require the creation of high nature-value grassland using regional seeds to reach a sufficient compensation effect (Molder 2015; Sengl 2015). Regional high-diversity seed mixtures can facilitate the establishment of species-rich grassland by introducing propagules of locally adapted species (Aavik, Edwards, Holderegger, Graf, & Billeter 2012). However, one important disadvantage of such seed mixtures is that they are far more expensive than commercial seed mixtures (Török, Vida, Deák, Lengyel, & Tóthmérész 2011; Mitchley, Jongepierová, & Fajmon 2012). Furthermore, like in other European countries, regional seed mixtures are still scarce in Austria (Sengl 2015). In response to these shortcomings, several studies from across Europe have shown that methods like hay transfer (Hölzel & Otte 2003; Rasran, Vogt, & Jensen 2007) can yield promising restoration outcomes. In addition, sod transplantation (Bruelheide & Flintrop 2000; Klimeš, Jongepierová, Doležal, & Klimešová 2010) was also shown to trigger a high establishment rate of target species. However, in this context we have to note that this method is destructive, because it implies that some areas with the respective habitat (i.e. donor sites) will be highly disturbed (Török et al. 2011). Furthermore, this method is very expensive and requires non-standard machinery (Scotton et al. 2012). Thus, it is only recommended in cases where the destruction of valuable plant communities (e.g. by infrastructural projects) is inevitable (Kiehl et al. 2010). By contrast, passive restoration can be a cost-effective option (Jongepierová, Mitchley, & Tzanopoulos 2007; Sengl, Wagner, & Magnes 2015) but only if diaspore sources are in the close vicinity and low risk of soil erosion and/or invasive species is expected (Kirmer et al. 2012).

In order to ensure that resources are efficiently used within the restoration project, it is essential to use the most promising techniques (Kiehl et al. 2010; Török et al. 2011). However, we have surprisingly little understanding of the general efficacy of different grassland restoration techniques (Walker et al. 2004). In Austria, the situation is particularly critical because little research has been conducted on the efficacy of lowland meadow restoration. As a consequence, practitioners must rely on publications from neighboring countries where climate, soil conditions and species communities can be quite different.

To close this gap, we evaluated five different techniques to restore lowland meadows in South-East Austria. We focused on (1) sod transplantation, (2) natural colonization, (3) hay transfer and addition of seed mixtures for (4) wet and (5) bare soils and measured restoration success through several indices (species richness, number of target species, Shannon diversity and similarity to reference sites). In particular, we asked the following questions: (A) What is the most effective technique for the restoration of lowland meadows? and (B) Is the applied restoration method more important than abiotic site conditions?

We assumed that sod transplantation, hay transfer and seeding of site-specific seed mixtures could be suitable methods for lowland meadow restoration since desired propagules are transferred to target sites (Hedberg & Kotowski 2010; Kiehl et al. 2010). Furthermore, we expected that natural colonization could also lead to a successful outcome, given that restoration sites are directly bordering on species-rich source sites (Sengl et al. 2015). In addition, we hypothesized that abiotic site conditions, i.e. soil nutrient content, would lead to differences in restoration success across restoration methods (Walker et al. 2004). Finally, since several studies reported that providing favorable site conditions can be feasible through topsoil removal (e.g. Hölzel & Otte 2003), we tested this kind of site preparation.

Materials and methods

Study area

The study area was located in South-East Austria in the province of Styria (Fig. 1). Restoration, reference, and donor sites were located in the alluvial valleys of the rivers (A) Feistritz and Lafnitz and (B) Mur. The region has a mild climate with an annual mean total precipitation of 737–827 mm, and annual mean temperatures of 9.1–9.3 °C, respectively (ZAMG 2016). Soils of the study sites are exclusively rainfed and comprise non-calcaric alluvial soils and stagnosols (Lebensministerium, 2016). The potential natural vegetation in this area is alluvial lowland forest as well as acidophilic oak forest on gravel terraces (Kilian, Müller, & Starlinger 1994). In the last centuries, lowland meadows were widespread

Table 1. Sampling design with the number of sites and the number of surveyed plots for each restoration treatment and year of observation.

Restoration treatment (soil preparation)	No of sites	No of surveyed plots		
		Year 1	Year 2	Year 3
Reference (no treatment)	8	8		
Sod transplantation (topsoil removal)	2	8	8	8
Natural colonization (ploughing)	–[a]	12	12	12
Hay transfer (ploughing)	5	6	15	15
Seeding 'wet meadow' mix (ploughing)	5	5	15	15
Seeding 'bare soil' mix (topsoil removal)	1	8	8	8
Total	21	47	58	58

[a]Natural colonization was studied next to sod transplantations.

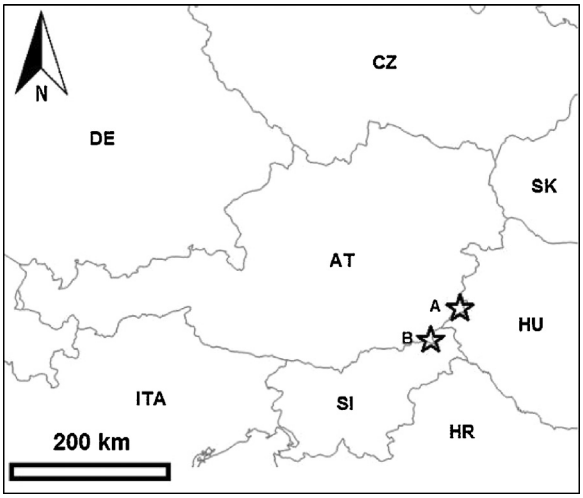

Fig. 1. Location of study areas in South-East Austria marked by stars: A (47°03′N, 16°04′E, 280 m a.s.l) and B (46°43′N, 15°56′E, 230 m a.s.l.). AT = Austria; CZ = Czech Republic; SK = Slovakia; HU = Hungary; HR = Croatia; SI = Slovenia; ITA = Italy; DE = Germany.

and used for hay production for livestock feeding or litter production for livestock bedding by the local communities (Steinbuch 1995). However, river valleys are now mostly used for the cultivation of corn crops (i.e. *Zea mays*). The study encompassed 13 restoration sites, with eight reference sites, seven donor sites for hay transfer and two donor sites for sod transplants (see Table 1 for details).

Reference and donor sites

According to the region's standard vegetation classification scheme, the reference and source vegetation for sod transplantation and hay transfer belonged to the *Molinion caeruleae* Koch 1926 alliance (alluvial meadows), and the *Arrhenatherion elatioris* Koch 1926 alliance (mesic lowland meadows) (Ellmauer & Mucina 1993). Sites were managed by mowing and subsequent biomass removal, performed mostly once or twice a year, after 15th of June. Soils were relatively acidic (pH: 4.5–5.5) and nutrient-poor (phosphorus: 15–30 mg kg^{-1}; potassium: 70–160 mg kg^{-1}). Mean total number of vascular plant species was 37.5 (±3.7 SD) on 4 m × 4 m plots (see Appendix A: Table 1). Reference and donor sites were located within the same biogeographic region as the restoration sites, but given their scarcity the distances to restoration sites reached up to 40 km. Donor sites for hay transfer were preselected considering target communities, species richness and absence of alien species and weeds. Donor sites for sod transplantation where demanded by the respective projects (road construction: project A, land consolidation: project B) and thus their translocation aimed to preserve the respective plant communities.

Restoration sites

Restoration sites encompassed former arable fields, where Styrian oil pumpkin (*Cucurbita pepo* var. *styriaca*) and corn crops had been grown. Restoration sites near Fürstenfeld (project A: $N = 12$; 8.4 ha total area) were purchased by a road construction company in order to compensate for damaged habitats during road construction. They were distributed over a length of approximately 10 km. The restoration site near Halbenrain (project B: $N = 1$; 0.31 ha total area) was obtained by a nature conservation agency in order to create a compensation for land consolidation. Sites were nearly flat or displayed only little slope (0–5° slope inclinations).

Restoration measures

Restoration measures were carried out in 2012 (treatments 3, 4 and 5 at project A; all treatments at project B) or 2013 (treatments 1 and 2 at project A), in each case directly in the year after the last crop harvest, and included:

(1) *Sod cutting and transplantation*: In April, 1 m × 1 m sod pieces, including the upper 30 cm of the soil layer and vegetation cover, were cut out with an excavator shovel and were then transferred to two receptor sites. The distance between donor and restoration sites was <1 km. The receptor sites encompassed four 15 m × 15 m patches (project A) and four approximately 10 m × 20 m patches (project B), which were prepared by removing the upper 30 cm of the topsoil.

(2) *Natural colonization*: Sites were prepared by ploughing and harrowing. They did not receive any plant material, but were directly bordering on the sod transplantation sites (see Appendix A: Figs. 1 and 2 for details of the experimental design). So, in our experiment sod transplantation sites served as source for natural colonization. Vegetation plots kept a buffer zone of one meter from the border of sod transplantations.

(3) *Hay transfer*: Five receptor sites were prepared by ploughing and harrowing. Starting in May in the year of project initiation, hay donor sites were continuously monitored for seed maturation of target species. In June, donor sites were mowed, the green hay transferred to the receptor sites (distance: 0.5 km–10 km) and distributed in a ratio of 1–2:1 between donor and receptor site. At receptor sites, hay was distributed manually in a 3–5 cm thick layer.

(4) *Seeding of a seed mixture for wet meadows (seed mix 'wet meadow')*: In May, we applied a partly regional, commercially available, seed mixture (Austrian and German provenance; 2.5 kg ha^{-1}) to five sites prepared by ploughing and harrowing. The mixtures consisted of propagules of 22 typical lowland meadow species (see Appendix A: Table 2).

(5) *Seeding of a high diversity seed mixture on bare soil (seed mix 'bare soil')*: In May, we applied a partly regional, commercially available, seed mixture (Austrian and German provenance; 2.5 kg ha^{-1}) consisting of 32 typical lowland meadow species (see Appendix A: Table 2) to one site. The site was prepared by topsoil removal (upper 30 cm removed by a bulldozer) and subsequent harrowing.

After initial treatments, all sites were mowed in late summer of the same year. In the following three years, they were mowed in early (June) and late summer (September); the cut biomass was removed.

Data sampling and preparation

We surveyed vegetation in 4 m × 4 m plots, as proposed by Chytrý and Otýpková (2003) for temperate grasslands. We estimated the cover of every vascular plant species with an extended Braun-Blanquet cover-abundance scale (Dengler, Chytrý, & Ewald 2008). Plant nomenclature followed Fischer, Oswald, and Adler (2008). We sampled one plot per reference site and placed plots in the middle of the sites to avoid edge effects. Restoration sites were sampled in May starting in the growing season following the respective restoration measures and resampled in the following two years by revisiting the plots (Table 1). Due to the limited number of sites, we sampled several plots per site but we maintained a minimum distance of 20 m between plots to avoid pseudoreplicates. Additionally, we estimated the average slope of the whole plot area (in case of uneven microtopography of the surface, we assigned 1° for the entire plot) and collected a 500–1000 g soil sample from the upper 10 cm mineral soil layer. We analyzed the content of plant-available phosphorus (mg kg^{-1}) and potassium (mg kg^{-1}) (calcium-acetate–lactate method) and pH ($CaCl_2$-solution) in the soil.

Indices for measuring restoration success

We evaluated restoration success by calculating four frequently used indices to measure restoration success for every plot (Table 2): (1) total species number, (2) number of target species (3) Shannon diversity index, (4) a Frequency–Positive–Fidelity–Index (FPFI: Tichy 2005). These indices were used for a comparison of the restoration sites with the reference sites.

Statistical analysis

We used general linear models (GLM) to analyze the relationships between restoration technique and abiotic site parameters (P, K, pH and slope) as predictors and restoration success indices as responses three years after restoration was initiated. Prior to analysis, outliers were detected with Cook's distance and removed. Models were selected based on Akaike's Information Criterion (AICc). Significant predictors were sorted in decreasing order by their relative importance (*RI*) in the model. GLM were performed using SPSS 23 (IBM Corp. Armonk, NY).

We used detrended correspondence analysis (DCA) to explore to what extent species composition in restored sites resembled that in reference sites across the three years since restoration initiation. Cover values were log-transformed and rare species were down-weighted. To increase the readability of the ordination diagram, we displayed only 12 randomly selected relevés for every method and year combination. Additionally, we created a vector graphic to indicate direction and progress of vegetation development by connecting the sample centroids of year one and three. DCA was performed using Canoco 5 (ter Braak & Šmilauer 2012).

We analyzed the effects of different restoration techniques on restoration success indices by using first, the Kruskal–Wallis-*H*-test and second, a paired Mann–Whitney-*U*-test. We chose these tests because of our low sample size and heteroscedasticity in our data. Furthermore, we used sequential Bonferroni correction (Holm 1979) to correct for multiple testing. Kruskal–Wallis-*H*-test and Mann–Whitney-*U*-test were performed using PAST 3.04 (Hammer, Harper, & Ryan 2001).

Results

Soil preparation

The overall effect of soil preparation on abiotic soil parameters was low (Table 3). Soil pH was higher on sites with topsoil removal and ploughing than on reference sites and sites with sod transplantation. Phosphorus and potassium contents were lowest on sites restored by sod transplantation

Table 2. Indices of restoration success, calculated for every plot.

Index	Abbreviation	Description
Total species number	Species (N)	Number of vascular plants per plot.
Number of target species	Target species (N)	Total number of typical lowland meadow species (i.e. diagnostic species) according to syntaxonomical standard literature of the study region (Dragulescu & Magnes 1996; Ellmauer & Mucina 1993; Oberdorfer 2001, (see Appendix A: Table 2)).
Shannon diversity index	Shannon Index	The Shannon Index (Hill 1973) is a measure of alpha-diversity that takes species richness and evenness into account. The index is increased either by having more unique species, or by having a greater evenness.
Frequency positive fidelity index	Similarity (FPFI)	The Frequency–Positive–Fidelity–Index (Tichý 2005) is a similarity index that considers both frequency and fidelity of species and thus allows a comparison of single vegetation plots to vegetation-units.

Table 3. Effect of soil preparation on soil nutrient content and pH. SD = standard deviation. Different lowercase letter within columns 'Mean' indicate significant differences ($P < 0.05$; Kruskal–Wallis-H test for overall differences among groups of soil preparation; Mann–Whitney-U test for pairwise comparisons).

	Soil pH		Phosphorus (mg kg^{-1})		Potassium (mg kg^{-1})	
Soil preparation	Mean	SD	Mean	SD	Mean	SD
Reference	4.71 a	0.16	23.63 a	5.58	98.25 a	29.74
Sod transplantation*	4.85 a	0.19	14.88 b	5.11	47.38 b	5.71
Ploughing	5.32 b	0.42	38.48 a	31.86	136.95 a	60.18
Topsoil removal**	5.5 b	0.00	23.00 a	0.00	101.00 a	0.00

*Soil nutrient content and pH were measured in sod transplants that were relocated after topsoil removal.
**Mixed sample for all plots at the topsoil removal site.

(P: 14.88 ± 5.11 mg kg^{-1}; K: 47.38 ± 5.71 mg kg^{-1}) but had no significant effect between reference sites and all other soil preparation measures.

Factors driving restoration success

GLM across all measures of restoration success (Table 4) showed that restoration treatment, rather than abiotic site conditions, was by far the most important factor ($RI > 0.8$). Overall, the relationship between restoration success and treatment was strong, with a maximum value of $R^2 = 0.804$ in model with similarity as a response variable. Models for Shannon diversity ($R^2 = 0.653$) showed the weakest relationship. In two instances, abiotic site conditions also showed a significant effect. In the models for Shannon diversity and similarity to reference sites slope had a positive correlation with $RI = 0.205$ and $RI = 0.111$, respectively.

Vegetation development across restoration techniques

In general, DCA showed that community composition at restoration sites was becoming more similar to the reference vegetation with time (Fig. 2). By the third year, sod transplantation created plant communities that had the highest similarity to reference communities followed by hay transfer. Seeding the 'wet meadow' seed mixture and natural colonization were slightly inferior. By comparison, after three years, seeding the 'bare soil' seed mixture after topsoil removal had the most dissimilar community composition. These dissimilarities were represented by Axis 1 and 2, respectively.

Except for seeding on soils prepared by topsoil removal, all restoration methods created a mean herb cover >62% in the first and >75% in the third year. By contrast, seeding on bare soil following topsoil removal failed to build up a closed vegetation cover after three years (22.5% mean cover). However, we found *Vulpia bromoides*, a ruderal species critically endangered in Austria at this site.

The success of restoration differed widely among treatments and years (Fig. 3). However, three main trends emerged. First, sod transplantation achieved the highest rates of restoration success across all indices. Second, restoration success improved over time in the remaining groups with hay transfer showing the best results. Third, the two seeding treatments had the poorest restoration outcome.

Already starting from the first year, sod transplantation created sites that did not differ significantly in total species number from reference sites (Fig. 3A). Other restoration methods had considerably fewer species per plot. Although

Table 4. Results of model selection (GLM) for every index of restoration success, in which restoration methods and abiotic site conditions were tested as predictors. Predictor values that entered model selection included sod transplantation, hay transfer, natural colonization, seeding 'wet meadow' seed mixture, seeding 'bare soil' seed mixture, slope, and soil abiotic properties (P, K, pH). Model selection was performed based on corrected Akaike's information criterion (AICc). Values with $P < 0.001$ and highest Relative Importance (*RI*) values are indicated in bold.

	Effect	Sum of squares	Df	Mean of squares	*F*	*P*	Relative importance	Corrected R^2
Species (N)								
Restoration method		2443.5	3	814.5	57.2	**<0.001**	**0.943**	
Slope	+	148.6	1	148.6	10.4	0.002	0.057	0.771
Target species (N)								
Restoration method		1631.4	3	543.8	52.2	**<0.001**	**0.968**	
Slope	+	53.7	1	53.7	5.2	0.027	0.032	0.743
Shannon Index								
Restoration method		4.01	3	1.34	28.0	**<0.001**	**0.764**	
Slope	+	1.08	1	1.08	22.6	**<0.001**	**0.205**	
Phosphorus	+	0.16	1	0.16	3.4	0.072	0.031	0.653
Similarity (FPFI)								
Restoration method		3566.300	3	1188.750	56.75	**<0.001**	**0.810**	
Slope	+	489.960	1	489.960	23.39	**<0.001**	**0.111**	
Phosphorus	+	197.42	1	197.42	9.43	0.003	0.045	
Soil pH	–	150.18	1	150.18	7.17	0.010	0.034	0.804

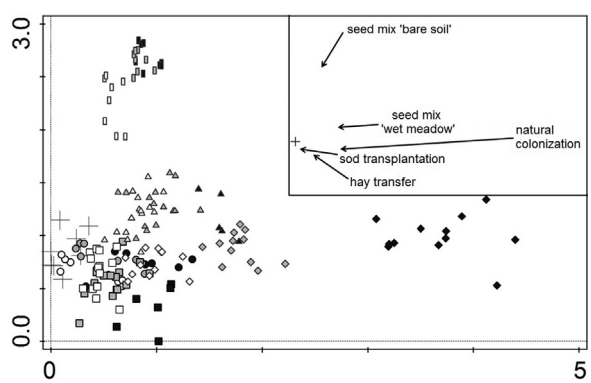

Fig. 2. Ordination diagram (DCA) of vegetation development along a three-year chronosequence since restoration initiation. Different symbols indicate restoration methods (+reference sites; ◯ sod transplantation; □ hay transfer; ◊ natural colonization; △ seeding 'wet meadow' seed mixture; ▯ seeding 'bare soil' seed mixture). Black colour = year one; grey colour = year two, white colour = year three. Cover values were log-transformed, rare species down-weighted. Eigenvalue and length of 1st axis: 0.436 and 4.30. Eigenvalue and length of 2nd axis: 0.202 and 2.85. Total inertia: 3.55. Inset plot: Arrows indicate direction and progress of vegetation development by connecting the sample centroids of year one and three.

plots restored by natural colonization had lowest species numbers in the first year, their total species numbers increased swiftly in the second year. By contrast, sites that received a seed mixture (both on sites prepared by ploughing and topsoil removal) had the lowest number of species per plot in the final year of assessment. Sod transplantation was also the most successful restoration measure in terms of the mean number of target species per plot ($N = 31.5$, third year), which were similar to reference sites and changed little across years (Fig. 3B). However, although the overall development of the number of target species across years was nearly constant, we observed a loss of some wet meadow specialists in the third year (i.e. *Poa palustris*, *Scirpus sylvaticus* and *Selinum carvifolia*). In the remaining restoration treatments, the number of target species increased over the years, with hay transfer displaying the highest values ($N = 23.9$, third year). Mean Shannon diversity was generally high across all treatments (Fig. 3C). Again, sod transplantation did not significantly differ from reference sites. Sod transplantation also outperformed other restoration measures in terms of its similarity to reference sites (Fig. 3D).

Discussion

According to European legislation, restoration is mandated to compensate for land degradation and the loss of habitat and diversity. However, restoration practice is hampered in many regions due to insufficient knowledge of effective restoration techniques. We accompanied two lowland meadow restoration projects in South-East Austria in order to test the efficacy of five techniques. Sod transplantation was by far the most effective treatment, followed by hay transfer. Natural colonization in the vicinity of sod transplantations performed similarly well as hay transfer. The two seeding approaches were less efficient. The measured abiotic factors played a negligible role during lowland meadow restoration.

Soil preparation

Contrary to our expectation topsoil removal did not lead to a lower soil nutrient content as compared with ploughed

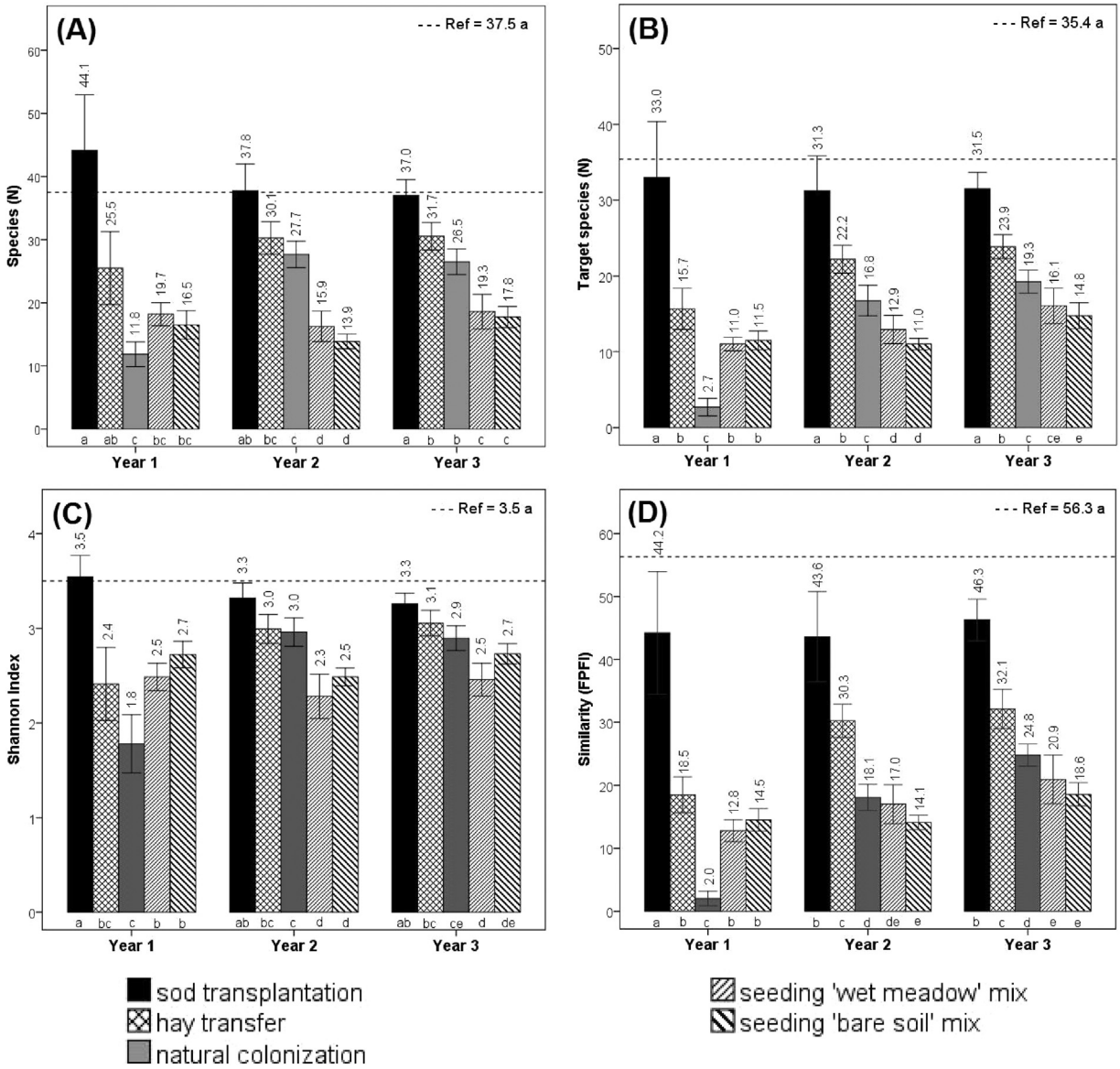

Fig. 3. Restoration success along a 3-year chronosequence and across restoration methods. Bars represent mean values among plots; error bars show standard errors (±2 SE). Mean values at reference sites are represented by the horizontal line and 'Ref'. Different lowercase letters below bars and next to 'Ref' indicate significant pairwise differences between treatments ($P < 0.05$; Kruskal–Wallis-H for overall differences among restoration methods; Mann–Whitney-U for pairwise comparisons were carried out for every year and included mean values of restoration treatments and the value for the reference site).

sites. Hölzel and Otte (2003), for example, reported that topsoil removal from former arable fields at a depth of 30 cm in a river valley in Germany, a starting point which is comparable with our study, reduced potassium content from 100 to 35 mg kg^{-1} and phosphorus content from 100 to 40 mg kg^{-1}. Other studies reported similar results (e.g. Rasran et al. 2007; Klimkowska et al. 2010). However, in our study we had no before and after comparison so it was not possible to estimate the effect size per site. In the case of ploughing and topsoil removal, soil pH was the only one factor to differ significantly from reference sites; in both cases it was higher in the respective restoration sites. This could be an effect of calcium fertilization (liming) in the past (Haynes & Naidu 1998; Walker et al. 2004). Interestingly, only sod transplants had lower nutrient contents. Most likely this can be explained by the originally low nutrient contents of the two respective donor sites.

Factors driving restoration success

Restoration method was the main explanatory factor across all measures of restoration success (Table 4). Slope was the second most important factor, while soil nutrient content and pH played a minor role. The positive effect of slope to restoration success may be linked to a higher number of available

niches for seedling establishment. Deák et al. (2015) reported that micro-topographic heterogeneity was positively related to plant diversity. This applies to the factor slope in our case, where totally flat restoration sites were opposed to sites with uneven surface or little slope.

Phosphorus content had a very weak positive impact on diversity and similarity to reference sites. This is in contrast to several previous studies (e.g. Walker et al. 2004; Fagan, Pywell, Bullock, & Marrs 2008; Marteinsdóttir & Eriksson 2014), but supported by Kardol et al. (2008), who reported that measures to decrease plant available nutrients had no effect on the establishment of sown species, but reduced biomass and weed infection of sites. Pywell et al. (2002) and Donath, Bissels, Hölzel, & Otte (2007) concluded that high soil fertility seems to be a minor constraint in grassland restoration on ex-arable fields. However, most studies that found a negative impact of soil nutrient content on restoration success reported that phosphorus content was higher at restoration sites than at reference sites. In our study, mean phosphorus and potassium content did not differ from reference sites and were in most cases below the limits for high plant diversity in grasslands (Janssens et al. 1998; Zelnik & Čarni 2013). However, contrary to several similar studies the target vegetation of our respective restoration projects were not particularly low-productive plant communities (Verhagen, Klooker, Bakker, & van Diggelen 2001).

Vegetation development across restoration techniques

The ordination (Fig. 2) and the analyses of individual indices for measuring restoration success (Fig. 3) showed that all restoration sites progressed towards the reference vegetation. Conrad and Tischew (2011) and Rydgren, Nordbakken, Austad, & Heegaard (2010) reported similar trends across different restoration methods. However, in our case, only sod transplantation fully achieved the goal within the time frame (see also: Bruelheide & Flintrop 2000). Its high success can be linked to the fact that all species of the respective donor site are introduced with this technique, including seedbank, soil biota (invertebrates, mycorrhizal fungi) and, at least partly, soil abiotic conditions (Pywell et al. 2002; De Deyn et al. 2003; Heneghan et al. 2008). However, we observed the loss of some specialist species by the third year since restoration initiation, which is in line with some previous studies (e.g. Klimeš et al. 2010). Most of these species were wet meadow specialists. Their extinction may be explained by the fact that the transplantation was followed by a particularly hot and dry summer in 2013, which was the driest in the region since 1932 (ZAMG 2016). Additionally, the respective restoration sites may have been drained in the past, and thus did not provide the same hydrological conditions as the reference sites (Pfadenhauer & Grootjans 1999). Consequently, continuing monitoring is necessary to evaluate the long-term success of this restoration technique.

Hay transfer achieved the second best results: Sites started with the second highest number of target species in the first year, significantly increased this index by the second year but stagnated by the third year (Fig. 3). Compared to sowing, most previous studies reported positive outcomes for this technique (Kiehl, Thormann, & Pfadenhauer 2006), with similar (Rydgren et al. 2010) or even higher restoration success (Kiehl et al. 2010; Baasch, Kirmer, & Tischew 2012). Moreover, the number of target species in the third year was more than 30% higher than in sown sites. Considering the hot and dry weather conditions in 2013, the mulch layer may have had a positive effect on seedlings by preventing desiccation (Eckstein, Ruch, Otte, & Donath 2012). Furthermore, hay harvest from several sites provides a much larger species pool compared to the seed mixtures, which comprised only 22 (wet mix) and 32 typical grassland species (bare soil mix), respectively (see Appendix A: Table 2).

The results of DCA ordination (Fig. 2) showed that natural colonization and seeding a 'wet meadow' mix on ploughed soils performed similarly while natural colonization made by far the greater progress. However, in this context we have to consider that, similar to the study of Klimeš et al. (2010), natural colonization was investigated in the immediate vicinity of sod transplantation, only keeping a minimum distance of one meter, with the center of the plot lying only three meters from the border of sod transplantations (see Appendix A: Figs. 1 and 2). As several studies report, passive colonization can be a successful restoration tool if dispersal distances are short (Donath, Hölzel, & Otte 2003; Albert et al. 2013; Sengl et al. 2015).

Restoration success was relatively low on sites restored by seeding a 'wet meadow' mix and especially low by seeding a 'bare soil' mix after topsoil removal (Figs. 2 and 3). The latter displayed a decrease in number of target species from the first to the second year following restoration start. Nevertheless, we observed a rising number of target species and similarity to reference sites towards the third year. We assume that the low starting point for seeding a 'bare soil' mix in the first year and its decline in the second year was due to the extreme abiotic conditions after topsoil removal (Kardol et al. 2008). It is likely that many of the plants did not overcome the seedling stage because of desiccation (Kirmer et al. 2012). Yet, as a surprising side-effect, the low cover of the site prepared by topsoil removal allowed the emergence of a rare annual plant (*Vulpia bromoides*) that is typical of disturbed soils, which had not been found in the region since 1926 (Zernig et al. 2015). Additionally, the pool of target species was lowest for sown sites, because it was restricted to the respective seed mixtures.

The presence of weeds, which can be perceived as the difference between total species number and number of target species (Fig. 3), was reported as a possible problem in several studies (e.g. Lengyel et al. 2012; Török et al. 2012a). However, in our case the presence of weeds was low after three years on all restoration sites. Overall low weed infection can be explained to some extent by proper follow-up manage-

ment which consisted of mowing twice a year (Scotton et al. 2012), and by the fast establishment of a closed vegetation cover at all sites, except the seeded site prepared by topsoil removal. In the latter case, weeds were probably mostly lacking because the largest part of the seed bank of weeds was removed with the soil (Hölzel & Otte 2003).

Conclusions and management implications

Sod transplantation led to highest restoration success across all indices. In cases where the destruction of a valuable grassland habitat is inevitable, sod transplantation should be an integral component of compensation measures in order to minimize loss of local grassland biodiversity. However, given its generally high costs, hay transfer and natural colonization can be effective alternatives, given that proper source sites are available. Seeding of partly regional seed mixtures was least successful in our study, independent of initial site preparation and should be avoided if the above-mentioned restoration techniques are applicable. Although not a part of our study, a combination of these methods (e.g. seeding and hay transfer: Török et al. 2012b) could overcome weaknesses of individual methods and lead to a fast and cost-effective recovery of grassland.

Topsoil removal led to lower total vegetation cover and allowed the emergence of a particularly rare plant species. If fostering rare ruderal plants or long-term successional stages are intended, this technique can be considered. However, since topsoil removal is very costly and vegetation development can be hampered through extreme site conditions, we do not suggest this kind of site preparation for restoring lowland meadows. Overall, abiotic site conditions of the restoration sites played a minor role for achieving restoration targets in the early stage of vegetation development, though it is possible that they gain importance in a longer time frame.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.baae.2017.08.004.

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