

Grassland restoration on former croplands in Europe: an assessment of applicability of techniques and costs

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Abstract Grasslands used to be vital landscape elements throughout Europe. Nowadays, the area of grasslands is dramatically reduced, especially in industrial countries. Grassland restoration is widely applied to increase the naturalness of the landscape and preserve biodiversity. We reviewed the most frequently used restoration techniques (spontaneous succession, sowing seed mixtures, transfer of plant material, topsoil removal and transfer) and techniques used to improve species richness (planting, grazing and mowing) to recover natural-like grasslands from ex-arable lands. We focus on the usefulness of methods in restoring biodiversity, their practical feasibility and costs. We conclude that the success of each technique depends on the site conditions, history, availability of propagules and/or donor sites, and on the budget and time available for restoration. Spontaneous succession can be an option for restoration when no rapid result is expected, and is likely to lead to the target in areas with high availability of propagules. Sowing low-diversity seed mixtures is recommended when we aim at to create basic grassland vegetation in large areas and/or in a short time. The compilation of high-diversity seed mixtures for large sites is rather difficult and expensive; thus, it may be applied rather on smaller areas. We recommend combining the two kinds of seed sowing methods by sowing low-diversity mixtures in a large area and high-diversity mixtures in small blocks to create species-rich source patches for the spontaneous colonization of nearby areas. When proper local hay sources are available, the restoration with plant material transfer can be a fast and effective method for restoration.

Keywords Grazing · Hay transfer · Mowing · Seed sowing · Spontaneous succession · Topsoil removal · Restoration cost

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Introduction

Agricultural intensification has led to the loss of area and diversity of former grasslands across Europe (Edwards et al. 2007; Pullin et al. 2009). In many regions only small fragments of grasslands have remained, which are isolated by intensively cultivated agricultural lands (Öster et al. 2009). Beyond the effects of fragmentation, the general intensification of agriculture in the landscape further threatens the native biodiversity of grassland fragments. To preserve these fragments and their biodiversity and to re-establish connections between them, the restoration of grasslands on former croplands is a high priority of nature conservation (Walker et al. 2004; Stadler et al. 2007).

In contrast with agricultural intensification, large-scale abandonment of low-productivity agricultural areas is common in certain parts of Europe and the world (Cramer et al. 2008). The rate of abandonment of croplands depends on socio-economic factors and differs greatly from west to east in Europe (Ramankutty and Foley 1999). In Western Europe, subsidy systems established under the Common Agricultural Policy have led to increased agricultural activity, further intensification and the reuse of fallows. In Central and Eastern Europe (CEE), the collapse of socialist regimes resulted in the collapse of state-owned agricultural cooperatives and led to the privatization of land in the early 1990's. Land was often privatized to their previous owners of advanced age or to farmers who could not cultivate it due to their insufficient financial background. Competition from imported agricultural goods produced by farmers heavily subsidized in Western Europe further decreased the intensity of agricultural cultivation and accelerated the rate of land abandonment (Pullin et al. 2009). For example, 600,000 ha or 10% of all croplands have been abandoned between 1990 and 2004 in Hungary, and the rate of abandonment was similar (10–20%) in four other CEE countries (Cramer and Hobbs 2007).

The restoration of grasslands on abandoned croplands offers a great opportunity to mitigate or halt the processes that damage grassland biodiversity (Stevenson et al. 1995; Hayward 2009). Thus, grassland restoration on abandoned cropland is one of the most frequently applied habitat restoration actions in Europe (Cramer and Hobbs 2007). For example, a search of a database that contains information on projects funded by the European Commission between 1992 and 2009 under the LIFE-program using the word 'restoration' (<http://ec.europa.eu/environment/life/project/Projects/index.cfm>), and filtering to 'natural and semi-natural grassland formations' returns 290 projects.

Despite the frequent application of grassland restoration in conservation practice, proportionally little attention has been given to its usefulness in biodiversity conservation. Recent reviews on grassland restoration have focused on species transfer, establishment and recovery, which topics are highly relevant for both ecological theory and practice (e.g. Kiehl et al. 2010; Hedberg and Kotowski 2010). Arising from the great interest in grassland restoration by conservation practitioners, a review of the main methods of grassland restoration on former croplands, with a special emphasis on the applicability and cost-effectiveness of the methods is warranted.

Here we present a review of current practices in grassland restoration on former croplands. We first present the frequently used restoration techniques, and accessory techniques used to improve restoration success, with a listing of pros and cons of each method. Then, we evaluate restoration success based on the reviewed studies and finally, we summarise costs and make suggestions on the concrete application of the reviewed methods.

Our review was compiled from a literature search of electronic sources (JSTOR, Science Direct, ISI Web of Knowledge and Google Scholar) using the keywords

“grassland restoration” and “grassland recovery”. Several recent articles were added based on the personal expertise of the authors. In our review we have included only those studies which focused on grassland restoration techniques that are frequently used in former croplands in Europe. For cost estimates, we gathered information from the papers (e.g. Manchester et al. 1999), but also directly from the authors of the papers included.

Restoration techniques

Spontaneous succession

Spontaneous secondary succession following the abandonment of croplands, often termed as ‘old-field succession’, is the easiest and most natural way of grassland restoration (Prach and Hobbs 2008). Old-field succession is one of the best-studied topics in ecology and the knowledge gained in such studies (for a bibliography, see Rejmánek and van Katwyk 2005) has been instrumental in the development of the field of restoration ecology (Hobbs and Walker 2007). In some cases, restoration can rely on spontaneous processes (Ruprecht 2006; Prach and Řehouňková 2008; Török et al. 2010a). In these cases, restoration is based on locally available sources of propagules, e.g. local seed bank or seed rain mediated by different seed dispersing agents from adjacent natural vegetation. In fragmented landscapes where the availability of adjacent seed sources is low and/or seed dispersing agents are missing, the regeneration of grasslands is often slow or delayed (Manchester et al. 1999; Kleijn 2003; Simmering et al. 2006; Foster et al. 2007). Furthermore, agricultural cultivation in most cases completely destroys the former grassland seed bank, resulting in the increase of seeds of weedy species in the soil (Bakker and Berendse 1999; Bossuyt and Honnay 2008; Manchester et al. 1999). The high amount of weed propagules associated with higher levels of nutrients creates perfect conditions for weeds and their seedlings (Kardol et al. 2008), which can hamper the regeneration process. Sometimes, succession stops in an early stage due to the increased dominance of a noxious competitor (Prach and Pyšek 2001). As a result, restoration by spontaneous succession in several cases can be slow or unpredictable. Thus, it is often necessary to direct vegetation changes with more active restoration measures.

Sowing seeds

Sowing seed mixtures of target species is a widely used restoration method in conservation practice (Table 1). The composition of a seed mixture is strongly influenced by the aim of restoration (e.g. target vegetation), the site conditions of receptor sites, or the availability of seed sources of potential target species. Low-diversity (LD) seed mixtures typically contain propagules of 2–8 species, which are usually the dominant grass and/or forb species of the target vegetation. High-diversity (HD) seed mixtures usually contain seeds of more than 10 species (Table 1).

The seeds for restoration can be purchased from commercial sources or collected by local harvesting. Commercial sources are appropriate if the seeds can be sourced from local populations of the target species. Seeds of rare species (characteristic grassland species usually in scattered populations, e.g. in loess grasslands—*Phlomis tuberosa*, *Thalictrum minus*, Török et al. 2010b) are often not commercially available or very expensive, and often originate from non-native source populations (Manchester et al. 1999). Thus, the compilation of a HD mixture that also contains rare species can be

Table 1 Major characteristics of grassland restorations using high-diversity (HD) and low-diversity (LD) seed mixes and spontaneous succession

Restoration technique	Accessory technique	Number of sown species	Sowing density (kg/ha)	Year of restoration and last evaluation	Establishment success (species numbers)	Reference
1 HD seed mix	Ploughing/grazing or mowing	14	20	1993–1999	13 Sown, 6 unsown species (year 1); 4/3 (year 6)	Warren et al. (2002), Fig. 1
2 Spontaneous succession	Ploughing/grazing or mowing	None	None	1993–1999	6 Unsown species (year 1); 4 sown, 14 unsown species (year 6)	Warren et al. (2002), Fig. 1
3 HD seed mix	Ploughing/mowing and grazing	25–41	25–28	1994–1998	15.8 Sown, 10.6 unsown species (year 1); 16.9/5.0 (2 year); 16.8/4.5 (year 3); 17.2/4.7 (year 4)	Pywell et al. (2002), Table 5
4 LD seed mix	Ploughing/mowing and grazing	6–17	25–31	1994–1998	6.1 Sown, 10.0 unsown species (year 1); 6.6/4.2 (year 2); 7.5/4.4 (year 3); 8.9/5.0 (year 4)	Pywell et al. (2002), Table 5
5 Spontaneous succession	Ploughing/mowing and grazing	None	None	1994–1998	1.1 Sown, 9.3 unsown species (year 1); 3.2/8.0 (year 2); 5.2/9.8 (year 3); 7.8/10.8 (year 4)	Pywell et al. (2002), Table 5
6 HD seed mix	Ploughing and harrowing/mowing	27	20	1999–2004	7 Sown grass species with 55% cover; 19 sown herbs with 30% cover	Jongepierová et al. (2007), Fig. 2
7 Spontaneous succession	Ploughing and harrowing/mowing	None	None	1999–2004	Perennial weeds 58% cover, 80 unsown grassland species with < 5% cover	Jongepierová et al. (2007), Fig. 2
8 HD seed mix	Mowing (several regimes)	18	40	1994–1996	15 Sown species; 34 or 40 non-sown species	Lawson et al. (2004), Table 3
9 HD seed mix	Harrowing/mowing	15	3,500 seeds/m ²	1996–1998	10 Unsown species (CZ); 2 (NL); 7 (SE); 5 (ES); 3 (UK)	Lepš et al. (2001), Fig. 1

Table 1 continued

Restoration technique	Accessory technique	Number of sown species	Sowing density (kg/ha)	Year of restoration and last evaluation	Establishment success (species numbers)	Reference
10 LD seed mix	Harrowing/mowing	4	3,500 seeds/m ²	1996–1998	9.5 Unsown species (CZ); 5 (NL); 7.5 (SE); 6.5 (ES); 5 (UK)	Lepš et al. (2001), Fig. 1
11 Spontaneous succession	Harrowing/mowing	None	None	1996–1998	15 Unsown species (CZ); 12 (NL); 12.5 (SE); 9 (ES); 9 (UK)	Lepš et al. (2001), Fig. 1
12 HD seed mix	Ploughing/–	16	44.6	1994–1996	12 Sown species	Owen and Marrs (2000)
13 Spontaneous succession	Harrowing/mowing	None	None	1996–2003	3 Sown, 16 unsown species (CZ); 11/18 (NL); 3/28 (SE); 2/45 (ES); 5.5/33 (UK)	Lepš et al. (2007), Fig. 2
14 HD seed mix	Harrowing/mowing	15	3,500 seeds/m ²	1996–2003	10 Sown, 12 unsown species (CZ); 12/10 (NL); 4.5/24 (SE); 3.5/29 (ES); 9.5/10 (UK)	Lepš et al. (2007), Fig. 2
15 LD seed mix	Harrowing/mowing	4	3,500 seeds/m ²	1996–2003	6 Sown, 15 unsown species (CZ); 10/12 (NL); 2.5/26 (SE); 2/30 (ES); 6.5/22 (UK)	Lepš et al. (2007), Fig. 2
16 HD seed mix	None/mowing and grazing	23	40	1993–1996	34 Target, 45 total species	Manchester et al. (1999), Table 2a
17 LD seed mix	None/mowing and grazing	4 or 11	40	1993–1996	19 Target, 28 total species (4-species mix) or 27/37 (11-species mix)	Manchester et al. (1999), Table 2a
18 Spontaneous succession	None/mowing and grazing	None	None	1993–1996	21 Target, 28 total species	Manchester et al. (1999), Table 2a
19 HD seed mix	Ploughing and harrowing/mowing	1, 2, 4, 8, 12, or 16	Sown density increasing with diversity	2000–2005	0.5–1 Target species (when 1 species sown); 1–2 (2); 3 (4); 5.5 (8); 8 (12); 6.5 (16)	Piper et al. (2007), Fig. 4 (target)

Table 1 continued

Restoration technique	Accessory technique	Number of sown species	Sowing density (kg/ha)	Year of restoration and last evaluation	Establishment success (species numbers)	Reference
20 LD seed mix	Ploughing/ mowing	4	30–35	1998–2001	4 Sown species; 25 unsown species	Vécriin et al. (2002), Fig. 1.
21 LD seed mix	Ploughing/ mowing, grazing	7	27	1991–2005	44 Or mean 33 species (year 10); 56 or mean 40 species (year 14)	Sendžikaite and Pakalnis (2006), Fig. 4 and 5
22 LD seed mix	Ploughing/ mowing	2 or 3	20–25	2005–2008	2 Or 3 sown species (years 1–3); unsown species: 33.5 (year 1); 16.5 (year 2); 13.3 (year 3)	Török et al. (2010b), Table 2
23 HD seed mix	Topsoil removal/ grazing	56	1, 4, 10, 40	1993–1994	20 Target, 12 weed species (at 1 kg/ha); 27/8 (4); 30/7 (10); 32/2.5 (40)	Stevenson et al. (1995), Fig. 4c, d
24 Spontaneous succession	Topsoil removal/ grazing	None	None	1993–1994	7 Target, 16 weed species	Stevenson et al. (1995), Fig. 4c, d

CZ Czech Republic, NL Netherlands, SE Sweden, ES Spain, UK United Kingdom

unfeasible. It is advisable to use seeds collected or sourced locally or as close to the location of the restoration as possible (Mijnsbrugge et al. 2010). Sowing seeds from local sources decreases the chances of restoration failure due to the genetic incompatibility of the sown and naturally colonising individuals of a species (Kiehl et al. 2010). Using local ecotypes also increases the chances of restoration success as such ecotypes are better adapted to the local environmental conditions and may be better competitors against local weeds (Aldrich 2002). Seeds can be collected by hand or by appropriate equipment (e.g. vacuum harvester, combine harvester; Edwards et al. 2007). Though hand collection is time consuming and costly (Stevenson et al. 1995), it is important when target species are located in scattered populations.

When seeds of certain target species are not available in the same quantity as those of other, more common species, the sowing of LD and HD seed mixtures can be combined. The HD seed mixture can be sown in small patches within a larger area sown with LD seed mixture to establish potential sources of colonising propagules. Using LD seed mixtures can lead to the restoration of basic grassland vegetation dominated by perennial grasses as fast as in 3–4 years (Török et al. 2010b). The immigration of rare herbaceous species can be very slow, so the restoration of diverse grasslands can last much longer than that of a basic grassland vegetation. The complete recovery of species-rich vegetation also requires further post-restoration management. Grassland restoration by seed sowing often but not always requires soil preparation or even topsoils removal to establish bare ground surfaces. Seeding on bare soil greatly enhances the establishment of the target species (Kiehl et al. 2010). Soil preparation is mostly done by deep or shallow ploughing or disking followed by seed bed preparation (by racking). After the sowing the cover of seeds is necessary by light racking or ring rolling. The aim of topsoil removal is to reduce the amount of weed seeds, and nutrient availability in the upper soil layers or to establish microsites favourable for germination of the target species (Coulson et al. 2001; Pywell et al. 2002; Edwards et al. 2007).

Grassland restorations vary greatly regarding the density at which seeds are sown. Many studies reported on seed sowing conducted in a few to a few hundred square meters. In these cases, sowing densities ranging from 4,000 to 13,000 seeds/m² were used. When grassland restoration is applied in large areas (e.g. at least in several hectares), sowing densities of 20–45 kg/ha were used (Table 1); some cases much higher densities are suggested (80–500 kg/ha; if rapid recovery of grass-dominated swards is needed, see Krautzer and Wittman 2006). Increasing amounts of seeds often correspond with faster establishment of the target species (Lindborg 2006), but can also lead to higher rates of competition for resources among the sown species. Stronger competitors, e.g. clonally spreading grasses, can become dominant on the cost of other sown target species. This can lead to a decreasing richness of target species (Lepš et al. 2007).

Transfer of plant material

The transfer of fresh plant material, raked litter, or hay containing the seeds of target species has been used in grassland restoration in two ways; either to start secondary succession after land abandonment, or to increase species richness of degraded grasslands (Rasran et al. 2006). Although the transfer of plant material (e.g. seed enriched barn chaff) was traditionally applied until the middle of the 20th century to improve hay meadows (Kiehl et al. 2010), here we focus only on transfers used to restore grasslands on abandoned fields. Under such conditions, the transfer of plant material is typical in restoration of

species-rich grasslands, where the compilation of a seed mixture, often containing over 50 target species, would be almost impossible (Table 2).

Important factors in the design of plant material transfers are the site conditions, the area of the receptor and the donor sites, and the timing of the collection of plant material. The ratio between the area of receptor and that of the donor site generally ranges from 1:2 to 1:10; depending on the species richness, propagule richness and quality of the vegetation in the donor site (Aldrich 2002; Edwards et al. 2007). The most appropriate time for collecting plant material is determined by the phenology of the donor community and the target species, i.e., when the seeds of most species become ripe (Edwards et al. 2007). If we want to maximise the yield of grass seeds, the most appropriate mowing time in Europe is June in dry grasslands (e.g. in alkali or sandy grasslands), between June and July in mesophilous grasslands, and usually in July to late August in wet grasslands. However, the appropriate time of harvest strongly depends on actual weather conditions. Using plant material collected later, e.g. in late September, can lead to significant loss of propagules, especially that of graminoid species (Hölzel and Otte 2003). This can be a crucial problem in the recovery of species-poor and graminoid-dominated vegetation, such as alkali grasslands (Török et al. 2010b). As an alternative, seed collections may be repeated and spread across the vegetation period to maximise the number of target species (Stevenson et al. 1995). The collected plant material can be applied immediately, i.e. up to 24 h after cutting (Pywell et al. 1995; Donath et al. 2007), or it can be dried and stored as hay (Edwards et al. 2007). The transfer of fresh plant material appears to result in a higher establishment rate of target species than that of dried material and may increase the chances of establishment of rare species (Kiehl et al. 2010), due to the loss of seeds during the drying or storing process. The fresh or dried plant material is usually spread at a thickness of 10–15 cm (Donath et al. 2007), or in a density of 1–2 kg/m² in the restored site (Kirmer and Mahn 2001; Kiehl et al. 2006). If the amount of propagules in the plant material is high, the quantity of hay can be reduced to 0.5–1 kg/m² (thickness of 3–5 cm) (Kirmer and Mahn 2001). The quantity of hay transferred is of central importance because if too much plant material is transferred per area, the thick plant layers can inhibit the germination and colonization of target species (Donath et al. 2006).

Topsoil removal and carbon addition

Some former croplands can be characterised by high nutrient loads arising from the use of chemical fertilizers on former croplands (Verhagen et al. 2001). High nutrient levels of the soil can favour weedy species after the cessation of the agricultural cultivation (Patzelt 1998; Eschen et al. 2007), which can slow down the restoration, and/or decrease its success (Patzelt et al. 2001; Hölzel and Otte 2003; Edwards et al. 2007). Two methods are the most frequently used to decrease the amount of available soil nutrients in the upper soil layers: topsoil removal and carbon addition.

The removal of the topsoil can reduce the amount of the available nutrients (Allison and Ausden 2004; Kardol et al. 2008). In addition, many of the weed propagules can be removed with the topsoil (Hölzel and Otte 2003). In most cases, the removal of the upper 25–50 cm of topsoil is enough to ensure favourable conditions for the restoration (Klimkowska et al. 2007). However large-scale topsoil removal is not recommended in fields where high deflation by wind can be foreseen and also in steep slopes exposed to soil erosion.

Another tool for decreasing soil fertility in the upper soil layer is the immobilization of nutrients, especially nitrogen, in the soil. Generally, the immobilization is executed by the

Table 2 Major characteristics of grassland restorations in former croplands using plant material transfer

Target habitat type	Accessory technique	Number of species present in plant material	Amount of transferred plant material/density of viable seed	Year of restoration and last evaluation	Establishment success (species numbers or transfer rate)	Reference
1 Heathland	Ploughing/none	Species of <i>Calluna-Ulex</i> heath land (H2c-Rodwell, 1991)	3.3 kg/m ²	1988–1991	7 Target species/year 2 10 target species/year 3	Pywell et al. (1995), Table 5
2 Wet fen meadow	Topsoil removal/none	Species of fen meadows	In a thin layer of 5–10 cm (ratio 1:1)	1991–1997	16 Target, total 57 species/6 years	Patzelt et al. (2001), Table 1
3 Species rich calcareous grassland	Topsoil removal(in some cases)/moving and sheep grazing	54–78 Potentially transferable species	231–805 g hay/m ² 349–1264 seeds/m ²	1994–1998	Transfer rate: 69–89%	Kiehl et al. (2006)
4 Species-poor <i>Corynephorion</i> (sandy grassland) <i>Armerion elongatae</i>	None/none	37 and 46	Cca. 2 kg/m ² In a thin layer of 5–10 cm	1995–1998 2000	Site I/species rich: 23–25 (year 1) 19 (year 2) 20–21 (year 3) 19–23 (year 4) 16–19 (year 6) Site II/species poor: 11–15 (year 1) 10 (year 2) 8 (year 3) 6–8 (year 4; 6)	Kirmer and Mahn (2001), Table 4
5 Species-rich flood meadow	Topsoil removal/moving	Potentially 74–97 species	in a thickness of 5–10 cm	1997–2001	Total 41–70 species	Hölzel and Otte (2003), Table 1
6 Fen meadow	Topsoil removal/ grazing	41 transferred fen meadow species	in a thickness of 5–7 cm, 740 g/m ²	2004–2007	Species richness increased from 5–10 (year 1) to 20–25 species (year 4)	Klimkowska et al. (2010a)
7 Species-rich flood meadow	Ploughing/adding species poor grass seed mix (4 species; 5 g/m ²)	–	A thickness of 10–15 cm	2001–2004 or 2002–2005	Between 23 and 95 species Total 95 species	Donath et al. (2007)

addition of various carbon sources, which alters the C:N ratio in the soil (Török et al. 2000). The higher level of carbon in the soil often restricts microbial activity; therefore, it reduces the availability of mobile nitrogen for plant uptake (Averett et al. 2004; Eschen et al. 2007). Frequently used carbon sources are mulch (Averett et al. 2004), or hay (Kardol et al. 2008). Occasionally sucrose is used (Török et al. 2000; Eschen et al. 2007). However, the effect of nutrient immobilization by carbon addition is only a short-term solution compared to topsoil removal, because of the high microbial turn-over in the soil (Reever and Seastedt 1999).

Topsoil transfer, turf transplantation and community translocation

The success of restoration can be facilitated by the transfer of the upper soil layer from native grasslands. During a topsoil transfer, the upper layer of the soil is excavated, transferred to the restored site and spread in the form of mixed soil (Bullock 1998; Skrindo and Pedersen 2004). In the case of turf transplantation, smaller turfs are cut and transferred to the restoration site (Manchester et al. 1999; Aldrich 2002). Finally, community translocation means the transfer of an entire community; it is the rescue of a whole community from complete destruction (Bullock 1998).

One advantage of turf transplantation is that small patches of the target habitat can be created in the site subject for restoration. The transplanted turves can serve as propagule sources for re-vegetation. The introduced vegetative plant parts and diaspores associated with the introduced soil fauna and micro-biota make the re-vegetation quicker than would be occurring in the spontaneous way (Kirmer and Tischew 2006).

However, topsoil transport, either with or without vegetation, is not typically recommended as a restoration method, because it damages or destroys some parts of the donor site. In addition, the collection and the transfer of turfs and topsoil require considerable manpower and intense use of machinery, which can raise costs to extremes (Table 3). The maximum distance of the transfer of turfs is often only several hundreds of meters. The excavation of soil and turves is the simplest in loose and sandy soils, but the excavated material can easily fall apart during the transport. In heavy soils the excavation is much harder and only small turves can be transported because of high specific weight of the soil. In most cases, high specific survival rates were reported: 54–90% of species were successfully survived the transfer (Bullock 1998; Vécirin and Muller 2003). But in some cases high mortality of the transferred plants was reported. For example, only 16% of the grassland species survived in a translocation project in NW Hungary where $4 \times 1 \times 0.6$ -m blocks of topsoil were transferred (Takács, G., personal communication).

Accessory techniques used to improve restoration success

These techniques are widely used to increase the availability of seeds of target species either by planting entire plant individuals or introducing seeds, and to help plant establishment by grazing animals or by mowing.

Planting

Species richness in the restored grasslands can be improved with the plantation of entire plant individuals, or belowground parts (e.g. rhizomes, bulbs) of plant individuals. In addition, the selective planting of late-successional species in early successional stages can

Table 3 Direct costs of grassland restoration techniques

Restoration technique	Targeted grassland type	Cost/ha in Euros	Reference	Region
Seed sowing				
2–3 Species, 25 kg/ha	Alkali dry grasslands	225–465	Török et al. (2010b), and unpublished	HU
6–7 Species, 25–31 kg/ha	Species-rich neutral grassland	548	Pywell et al. (2002), and personal communication	UK
6–17 Species, 20–25 kg/ha	Species-rich calcareous grassland	354	Pywell et al. (2002), and personal communication	UK
6 Species, 25 kg/ha	Species-rich acid grassland	326	Pywell et al. (2002), and personal communication	UK
4 Species, 40 kg/ha	Lowland wet meadows	185	Manchester et al. (1999)	UK
11 Species, 40 kg/ha	Lowland wet meadows	745	Manchester et al. (1999)	UK
23 Species, 40 kg/ha	Lowland wet meadows	1341	Manchester et al. (1999)	UK
15 Species, 25 kg/ha	Loess dry grasslands	375–520	Deák and Kapocsi personal communication	HU
39 Species, 28 kg/ha	Species-rich neutral grassland	1,300	Pywell et al. (2002) and personal communication	UK
41 Species, 28 kg/ha	Species-rich calcareous grassland	1,927	Pywell et al. (2002) and personal communication	UK
25 Species, 25 kg/ha	Species-rich acid grassland	1,165	Pywell et al. (2002) and personal communication	UK
30 Species, 20 kg/ha	Dry calcareous hay meadows	800–1,000	Jongepierová et al. (2007), and personal communication	CZ
Plant material transfer				
Plant material to be purchased	Loess dry grasslands	3,240–3,300	Deák and Kapocsi personal communication	HU
Own harvest, applied after storage of 3 months	Loess dry grasslands	595–650	Deák and Kapocsi, personal communication	HU
Own harvest without storage (fresh material)	Loess dry grasslands	250–300	Deák and Kapocsi personal communication	HU
Own harvest without storage	Species rich floodplain meadow	6,020–7,763	Harmish and Donath unpublished	GE
Own harvest without storage, one mulching incl.	Species rich floodplain meadow	1,800–8,800	Donath et al. (2007)	GE
Own harvest without spreading cost	Fen meadow	425	Klimkowska et al. (2010a)	PL
Own harvest and transfer of heather shoots	Lowland heath/acid grassland	545	Pywell et al. (1995), and personal communication	UK
Transfer of threshed hay	Various type of grasslands	6,000–16,000	Kirmer and Tischew (2006)	GE

Table 3 continued

Restoration technique	Targeted grassland type	Cost/ha in Euros	Reference	Region
Topsoil removal				
Topsoil removal and soil transport	Fen meadow (on organic soil)	16,700	Klimkowska et al. (2010b)	PL
Topsoil removal without transport	Fen meadow (on organic soil)	13,000	Klimkowska et al. (2010b)	PL
Topsoil removal without transport	Various type of grasslands	4,000–6,000	Klimkowska, personal communication	NL
Community translocation				
With shallow soil layers	Lowland heath/acid grassland	14,982	Pywell et al. (1995) and personal communication	UK
With deep soil layers	Wet moorlands and grasslands	19,00,000–30,00,000	Kirmer and Tischew (2006)	GE

The cost/ha values include costs for (i) plant material/seed, (ii) seed collection/plant harvest/soil collection or soil removal, (iii) storage and transfer costs, (iv) site preparation (ploughing, disking, seed bed preparation), (v) sowing, spreading, (vi) post-restoration management (e.g. mowing, mulching etc.)
HU Hungary, *UK* United Kingdom, *CZ* Czech Republic, *GE* Germany, *PL* Poland, *NL* Netherlands

greatly accelerate restoration success (Du et al. 2007). The planting of belowground parts is used for species with good vegetative reproduction and establishment capabilities (Kirmer and Tischew 2006). Planting, however, is a cost-intensive method. It is recommended only in cases when the immediate translocation of endangered plant populations is necessary. This technique is often used to improve species richness or the availability of propagules in seed sowing or hay-transfer restored fields.

Grazing and mowing

Once basic grassland vegetation is established, mowing and/or grazing can be used to facilitate the colonisation of further characteristic species and increase plant diversity. Grazing and mowing generally accelerate the restoration, but they can also create conditions under which the restoration process is hampered. Both grazing and mowing have an extensive published literature (e.g. Bakker 1989); here we focus only on those studies in which some practical aspects for post-restoration are discussed. The most important effect of grazing and mowing is the reduction of aboveground biomass (Diemer et al. 2001; Bonanomi et al. 2006; Billeter et al. 2007). A common phenomenon in grassland restorations is that an accumulation of plant biomass and litter of sown grasses can be observed from the first year onwards. The accumulated litter can often be several times of what is typical for target grasslands (e.g. Török et al. 2010b) and can hamper the establishment and immigration of further target species. The removal of accumulated litter by grazing and mowing can be highly beneficial to post-restoration processes as litter removal can open up niches for further colonisation of target herbs from adjacent vegetation (Bissels et al. 2006). However, gaps may also provide physical space for the germination and establishment of weedy species present in the seed bank, especially in that of former croplands. An evaluation of the seed bank is essential to ensure that ecological processes progress towards the target status and not towards a weed-dominated phase (Török et al. 2009). Gaps originating from grazing can also favour the immigration of non-grazed noxious and poisonous invaders (e.g. *Asclepias syriaca* in degraded sand grasslands, Csontos et al. 2009).

Grazing has several advantages for grassland restoration compared to mowing. First, grazing can be more efficient in introducing propagules of the target species. Grazing animals bring in propagules of the target species from target-state grasslands through their guts or on their fur (Fischer et al. 1996; Mouissie et al. 2005; Mann and Tischew 2010). Mowing is less likely to have such an advantageous effect, although it is possible that propagules attached to the mowing machinery can be introduced into the receptor sites (Bakker et al. 1996). Second, grazing pressure is more likely to vary in space, especially when grazing is conducted by horses, which may create a more mosaic-like habitat structure. In contrast, mowing by heavy machinery often leads to homogenization of the plant community locally and at the landscape level (Zechmeister et al. 2003). In addition, mowing by large machinery also increases the compactness of the soil (Schäffer et al. 2007), which can reduce the success of colonization of target species. When mowing is done by hand, it can mimic the uneven distribution of disturbance necessary to create species-rich grasslands (Bissels et al. 2006). However, post-restoration management by hand mowing is costly or unfeasible in larger areas.

Livestock grazing is highly selective, which could negatively affect the target species. For example, most of grazing animals (e.g. cattle and sheep) usually avoid thorny or woody plants, which may lead to an increase of such plants and decrease the abundance of target native species (Hayes and Holl 2003). Furthermore, trampling and overgrazing of livestock

over grasslands enhance species that well tolerate such disturbances, but detrimental to species that are sensitive to trampling (Belsky and Blumenthal 1997). The intensity of grazing and the frequency and timing of mowing are crucial in influencing the success of restoration (Dostálek and Frantík 2008).

For grazing, the identity of livestock is also important. Cattle provide a rather even distribution of grazing pressure; however, their selective grazing benefits the establishment of thorny or woody forbs and trees (Hayes and Holl 2003). To avoid the invasion of thorny and woody species goat grazing may be a useful solution (Celaya et al. 2010). Grazing by sheep can result in a homogeneously low vegetation height, but may also damage plants of conservation importance, especially in wetter habitat types.

Mowing is generally more cost-effective and more readily available as a management option than grazing, which needs to encompass infrastructural and manpower investments (fencing, shepherds) over longer periods. However, mowing can result in high direct mortality and lower densities of invertebrates, whereas grazing is less directly damaging to invertebrates and can provide additional niches for invertebrates, e.g. to decomposers of animal droppings (Humbert et al. 2009). In general, a more complex food-web is expected in grazed areas than in mown areas (Wang et al. 2006).

Restoration success

The overall success of each restoration technique discussed above is difficult to assess for several reasons. First, each restoration technique may be suitable for certain starting conditions and target states and a comparison of very different techniques can be misleading. Second, even when similar techniques are used, there is high variation in the technical details of how restoration was carried out (Table 1). Finally, the measure of restoration success reported often varies among the studies, which makes a direct comparison difficult. Evaluation of the success of each technique to restore target grasslands is meaningful only for those techniques which are similar enough with regard to starting conditions, main methods and measure of success. Restorations based on seed sowing and transfer of plant material offer a possibility for such a comparison. Furthermore, spontaneous succession, particularly when monitored along with seed sowing or hay transfer in the same study, offers a useful reference against which to judge the success of the more active restoration technique.

Restoration success after seed sowing and spontaneous succession

Data for a comparison of relative restoration success were available from ten studies reporting on the effects of sowing HD seed mixtures, seven studies on LD mixtures and seven studies on spontaneous succession (Table 1). Especially valuable are those studies that monitored several treatments simultaneously, under the same settings (seed mix diversity, spontaneous succession, Table 1).

A detailed analysis of establishment success suggests that sowing HD seed mixtures leads to faster establishment of the species targeted by the restoration than an LD mixture or spontaneous succession (Table 1). The richness of established species increased with the number of sown species in most restorations and in the two experiments directly testing the effect of sown species richness on establishment success (Manchester et al. 1999; Piper et al. 2007). However, in some cases, even sowing HD seed mixtures cannot guarantee fast establishment success. For example, both Stevenson et al. (1995) and Lawson et al. (2004)

reported very poor establishment of the sown species. The establishment of sown species is often delayed until after year 1 following restoration (Vécrin et al. 2002). In other cases, establishment was highly successful even in year 1 (Manchester et al. 1999). Even though sowing HD mixture increases established species richness compared to sowing LD mixture or spontaneous succession, sowing HD seed mix may also constrain the colonisation of late-successional species (Lepš et al. 2007). In studies conducted over 6–7 years, species richness in HD sowing decreased below that in LD sowing. Lepš et al. (2007) found slightly fewer established species in HD restorations compared to LD restorations in each of the five European countries studied (Table 1). Early observations in that experiment showed that the best LD seed mixture can approach the species richness obtained by HD seed mixture, but that usually LD seed mixture led to less diverse communities (Lepš et al. 2001). However, over longer time scales, more unsown species colonized LD restorations than HD restorations (Lepš et al. 2007, Table 1).

Other observations also confirm these contrasting patterns. For example, species richness often decreases in sown plots and increases in naturally re-vegetating plots with time, sometimes even surpassing species richness in the sown plots (Jongepierová et al. 2007). These observations suggest that the sown grass community reduces the possibility of further colonization by late-successional species by competitive exclusion or litter accumulation (Lepš et al. 2007; Török et al. 2010b). Therefore, the initial floristic composition hypothesis (Egler 1954), which predicts that early processes largely determine the outcome of restoration, may explain short-term changes following grassland restorations, but is not supported by longer-term experiments (Table 1).

We identified three shortcomings of grassland restoration experiments carried out with seed sowing. First, generally little attention is paid to some starting conditions (e.g. seed content of the soil) and site history. For example, previous crop type on the abandoned arable land is only rarely given in the studies reviewed here, even though crop type and the corresponding cultivation (use of fertilizers, pesticides etc.) can greatly determine restoration success. For instance, some crop types require more fertilizers/pesticides than others, which may influence both the amount of nutrients available for restoration and the insects that may provide important ecological services such as pollination (Stoate et al. 2001). Moreover, some crops can produce allelopathic chemicals, which could hamper the establishment of target species especially in the early years (e.g. sunflower, Leather 1987). Site history is neglected even in studies focusing exclusively only on spontaneous processes (but see Molnár and Botta-Dukát 1998).

A second problem is that high concentrations of soil phosphorous or other nutrients on abandoned cropland can limit the success of colonization by several target species due to the increased competition from a few dominant grasses caused by high nutrient availability (Gough and Marrs 1990). The majority of the studies reviewed here did not typically report initial nutrient loads and did not consider this effect during the design of restorations (but see Van der Putten et al. 2000; Lepš et al. 2001; Pywell et al. 2002). As a remedy, Marrs (1985) suggested topsoil removal, deep ploughing or post-restoration management such as regular harvesting of biomass to reduce high nutrient loads or the high cover of the few dominants. However, Marrs et al. (1998) showed that the amount of available nutrients was not significantly changed in a 7 year study of cropping, suggesting that this method may work only in the long run (likely more than 10 years, see also Hrevušová et al. 2009).

Finally, there is also a general technical problem in such experiments. The sown species often colonize non-sown control plots (Pywell et al. 2002; Warren et al. 2002; Lepš et al. 2007). Some experiments even had to be terminated after year 2 because the sown species invaded non-sown control plots (Stevenson et al. 1995). One way to get around this

problem is to sow buffer zones consisting of commercial grass seed mixtures between experimentally treated plots, as was shown by Jongepierová et al. (2007). However, commercial seed mixtures are often composed of competitor grasses and forbs, which can also invade the treatment plots. A solution is to use a wide unsown buffer around the plots.

Restoration success after plant material transfer

Data for a comparison of relative restoration success following plant material transfer in former croplands were available from seven studies (Table 2). The success of plant material transfer depends on the availability of high-quality donor sites. If suitable initial conditions are met (e.g. high-quality of plant material, low nutrient availability), the application of plant material transfer provides rapid establishment of the target species (Patzelt et al. 2001; Kiehl et al. 2006; Donath et al. 2007). Measures of establishment success are available only when the seed content of plant material or the species composition of the donor sites are studied (Hölzel and Otte 2003; Kiehl et al. 2006; Rasran et al. 2006; Donath et al. 2007). The transfer rate of species depends strongly on the vegetation type and site conditions of the donor site and varies between 20 and 80%. The number of transferred species can be enhanced by the combination of early and late-harvested plant materials (Kiehl et al. 2006). An advantage of this method is that the transferred plant material may effectively suppress weeds. Conversely, the germination of the transferred species was neither hampered nor facilitated by plant material cover (Kirmer and Mahn 2001; Hölzel and Otte 2003; Donath et al. 2007). In one of the most comprehensive long-term study of plant material transfer, Kirmer and Mahn (2001) found that the number of established target species rapidly increased during the first 5 years; later on this process slowed down, even some of the transferred poor competitor species disappeared; the disappearance was caused by the increased dominance of a strong competitor (Kiehl et al. 2006).

We found several general shortcomings which can constrain the use of plant material in restoration. First, the seed content and species composition of hay is difficult to determine. In most cases, only the species composition of the used plant material and/or the vegetation of the donor site is determined. The propagulum content of the transferred plant material is not frequently studied (but see Hölzel and Otte 2003; Rasran et al. 2006; Donath et al. 2007). Second, the vegetative propagules (e.g. tillers and rhizomes) of asexually reproducing species (e.g. several *Carex* species, Hölzel and Otte 2003) cannot be transferred in the form of cut plant material; therefore, the direct seeding or transplanting of individuals of these species is necessary (Pywell et al. 1995). Third, the use of this method requires donor sites of appropriate size and quality. This constraint strongly limits the area that can be restored. Two- to ten-times larger area of donor sites are required than that of the receptor site to effectively transfer target species with plant material (Aldrich 2002; Kirmer and Tischew 2006; Edwards et al. 2007). The use of low-quality plant material (e.g. originating from species-poor or weedy grasslands) can shift vegetation changes to unsuitable directions, and can result in a weed domination (Hölzel and Otte 2003; Donath et al. 2007). So, the upper limit of the area that can be restored by this method is often a few hectares, especially where only small fragments of suitable donor sites are available.

Cost of restoration

In restoration planning it is crucial to know the costs of restoration actions. Most studies included in this review did not provide cost estimates, except for Manchester et al. (1999). Direct contact with several authors of previous studies as well as our own experience in

grassland restoration (e.g. <http://life2004.hnp.hu>) enabled us to summarise direct costs of various restoration actions from several European countries (Table 3). We considered the following direct costs: (i) plant material/seed, (ii) seed collection/plant material harvest/soil collection/soil removal, (iii) storage and transfer costs, (iv) site preparation (ploughing, disking, seed bed preparation), (v) sowing, spreading, (vi) post-restoration management (e.g. mowing, mulching or additional techniques in the first year after restoration).

The least expensive method is spontaneous succession where only some costs of regular mowing or grazing can be foreseen to improve immigration of target species and decrease the cover of weeds. The least expensive active restoration measure is the sowing of LD seed mixtures. Based on our query, a cost of several hundred euros per hectare can be expected (185–548 €/ha, Table 3). The use of HD seed mixtures involves costs that are at least two-times higher (around 1,000 €/ha). These costs strongly depend on sowing density and the species richness of the used mixture. The cost for plant material transfer ranged from several hundred €/ha (Table 3, Hungary and Poland) to several thousand €/ha (Germany). These costs strongly depended on (i) country, (ii) grassland type, (iii) size of donor area (small patches are more costly to harvest) and (iv) thickness of transferred plant material. Topsoil removal and community transfer are even more costly. The costs of topsoil removal can exceed 10,000 €/ha, which can be partly recovered by selling the removed topsoil (Marrs et al. 1998; Klimkowska et al. 2010b). In community translocation, the costs can be astronomical if we try to transfer also the deeper soil layers (Table 3). During the planning of grassland restoration it should be taken into account that there are several national agri-environmental schemes and/or EU funding programmes (e.g. LIFE+) which can cover the cost of grassland restoration actions entirely or partly.

Conclusions and implications for practitioners

As a first step in restoration planning the target state of the habitat or ecosystem of interest needs to be clarified, for which past observations, historical accounts, old maps or aerial images, local stakeholder knowledge can all be good sources.

After we have chosen the proper target grassland to be restored the restoration method should be chosen in accordance also with the financial background, manpower and other investments needed. The spatial and temporal scales of the project need to be clarified. These scales will fundamentally determine the financial background, manpower and other investment (e.g. post-restoration management) necessary to implement the restoration. All restorations should have strong spatial component (e.g. layout of what, where and when is implemented).

Spontaneous grassland recovery can be recommended when restoration is planned for a small area and/or when no rapid result is expected and when sources of propagules of target species are readily available in the vicinity of the area to be restored. This technique is the least expensive of all reviewed techniques, but the recovery of grassland vegetation in spontaneous succession compared to technical restoration is often slower and often unidirectional.

When the area to be restored is isolated from potential propagule sources sowing seed mixtures of local origin in a sowing density up to 40 kg/ha can be a feasible alternative to spontaneous recovery. The use of a LD seed mixture is recommended when the aim is to recover grassland vegetation in a relatively short time and introduction of rare species has only minor importance compared to the recovery of vegetation cover at the first stage (e.g. risks of soil erosion or weed infestation). Because the compilation of the proper amount of

a HD seed mixture for a large area is rather difficult, HD mixtures may be applied rather on comparatively smaller sites.

When the area to be restored is large (several tens of hectares and up), the combination of sowing low and high-diversity seed mixtures is recommended. The matrix of restored grassland should be sown at densities of 20–25 kg/ha with low-diversity mixture (usually basic grasses and forbs), and small patches within the matrix should be sown at 40 kg/ha with high-diversity mixtures that consist several additional species characteristic to the target grassland type. The high-diversity patches should be designated and maintained to maximise the chances of successful colonisation of species of interest.

In all restorations using seed sowing, seeds of species of local origin should be used, and manual or machine-assisted collection of seeds in natural grasslands should be preferred over obtaining seeds from commercial sources to reduce the risks of failure arising from the presence of non-local species (e.g. genetic incompatibility, out breeding depression, invasion).

In cases when species-rich grasslands are targeted (e.g. fen meadows or other grasslands with more than 30–40 species) and when proper sources of high-quality plant material (donor sites) are available, plant material transfer is likely to have high chances of success.

Planning should also incorporate post-restoration management (at the appropriate time scale, see above) because regular human interventions (mowing, grazing) are generally necessary, although the general aim should be to create self-maintaining natural-like ecosystems. Post-restoration management should be designed to facilitate the establishment of subordinate species characteristic to the target vegetation.

We suggest that further, carefully designed studies are needed to make a more precise evaluation of the applicability of each technique. First, different restoration techniques should be applied in similar site conditions and circumstances to compare their relative success. Second, the applicability of each technique should be tested in the restoration of several grassland types with the same study design. Finally, it is advisable for further studies to report carefully detailed information about the implementation of restoration and make conclusions about restoration success. At least the major restoration actions, but preferably, the entire restoration programme should be followed up by a monitoring system that has a proper sampling design and an adequate sampling effort. Sampling should be designed in advance; with the basic requirement that monitoring should be able to detect changes we expect to be able to show as significant. All steps in restoration planning and implementation need to be documented thoroughly, i.e. in written form, for future reference.

Funding the restoration programme can represent challenges. However, in many European countries, national agri-environmental schemes are incorporating grassland restoration as one way to reduce the effect of agricultural intensification of landscapes. European funding sources that aim to implement the Habitats Directive (e.g. the LIFE + programme) are also potential sources of co-funding for grassland restoration actions.

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