Ecological restoration of semi-natural grassland from agricultural fields Status quo and future prospects

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Introduction

Semi-natural grasslands in Europe produce important ecosystem goods and services such as live-stock products (milk, meat, wool, leather), biodiversity, storage of carbon (35% of the global stock as compared to 39% in woodlands), protection against soil erosion, tourism and recreation. This is increasingly acknowledged at the European level and the conservation of high value farmland is stimulated with financial instruments. Nevertheless the area of used agricultural area decereased by over 15% in almost all EU countries between 1990 and 2003. Much less information is available on changes in the quality but the data that do exist show a sharp decrease in species richness and diversity. Major causes are the abandonment of management in less optimal conditions, intensification of use in mild climates, at good soils and close to large markets, and in some countries changes in land use such as afforestation or conversion to arable fields play also an important role.

To counteract these unwanted developments there is an increasing demand for the restoration of species-rich grasslands. The present paper attempts to give an overview of the present status quo and possible future trends.

Mowing and grazing

In the case of restoring species-rich grasslands from intensified patures nutrient availability is normally too high for the establishment of many target species and communities. The first management goal is therefore, normally the reduction of site fertility. This target has been pursued in several countries by stopping fertilisation and (re)installing traditional management regimes such as mowing and grazing. The basic idea behind this option is to remove nutrients with the hay and the fodder and deplete the soil nutrient stocks in this way.

Nitrogen budget studies have shown that mowing indeed does remove nutrients from the site. Its effect, however, depends largely on the overal landscape setting. If the site is situated in an area with high atmospheric nitrogen deposition rates such as the southern Netherlands, northern Belgium and certain parts of Germany, where deposition may be as high as 50-60 kg N * ha^{-1} * yr^{-1} the net removal may be close to zero or there may be even an increase in nitrogen. On the other hand, however, nitrate and ammonia easily leach from the soil and can be reduced to gaseous N_2 and leave the system that way. A good understanding of the nitrogen balance is therefore essential to assess the effectiveness of mowing as a tool to lower nitrogen availability.

The situation is different with respect to phosphorus. Except for the coastal region where P is supplied through the sea, phosphorus in grasslands originates mainly from antropogenic activities and the level is determined almost entirely by the fertilisation intensity. This implies, after stopping fertilisation, mowing indeed leads to a net removal of phosphorus from the system. It is not surprising that the productivity in many grassland systems that have been mown for many decades or even centuries is regulated by P-availability. Under such conditions a net P-removal immediately leads to a lowered productivity and more open vegetation. Unfortunately this is not normally the case in sites that were used agriculturally until recently. In the intensively used agricultural grasslands of NW Europe P-fertilisation was normally so high that large surplusses are stored in the soil. Bakker and Olff (1995) devised a conceptual model for the time needed to remove surplus nutrients as function of the number of years under intensive management (*Figure 1*).

Originally they devised their model for nitrogen but it is probably even better applicable for phosphorus because of the conservative behaviour of this element.

The effect of grazing under moderate stocking densities differs from that of mowing not only in its heterogeneous nature, leading to a mosaic of highly grazed patches and ungrazed sites (Mouissie et al. 2008b, Mouissie et al. 2008a),

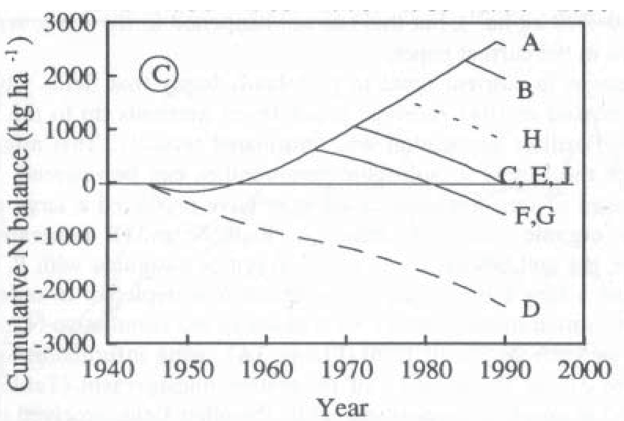

Figure 1: **Cumulative nutrient balance over time since 1945. Fields taken out of intensive agricultural use later must be subject to impoverishment schemes longer before the level of 1945 can be attained (Bakker and Olff 1995).**

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but also in its nutrient removal rates. Budget studies have shown that the net removal of nutrients through grazing herbivores is low. They mainly redistribute the nutrients over their feeding area, especially when they have latrines like horses. Most studies were carried out on husbandry animals in fenced areas, such as cows and sheep in meadows. The situation maybe somewhat different in the case of herded animals, especially when they defeacate mainly during the night. In that case there might be a net transport of nutrients from the fields to the stable.

It can be concluded that under certain conditions mowing and/or grazing can be suitable tool for the removal of excess nutrients. However, the net removal rate is low and tends to decrease under present day conditions.

Topsoil removal

Removal of the entire top layer through sod-cutting is a technique that has been used for centuries in heathlands and resulted in an extreme nutrient poverty (De Smidt 1979). Inspired by this approach nature conservation managers experimented with this method from the last decades of the 20th century onwards in an attempt to speed up nutrient removal rates and create the necessary nutrient poor conditions that are essential for the establishment and survival of the target communities.

This technique has been mainly applied in sites where the parent material underneath the cultivated layer is nutrient-poor by nature, such as sandy or calcareous soils (Verhagen et al. 2001, Van Diggelen and Marrs 2003). There are also differences in removal depth. Because there is a linear relationship between the amount of soil removed and costs, nature managers remove often only part of the cultivated layer, assuming that the great majority of the nutrients is situated in the upper layer. Soil chemical analyses showed that nitrogen and phosphorus behaved differently. Nitrogen is almost exclusively present in organic matter and, since organic matter content decreases with depth, removal of only the upper layer indeed does result in lowered nitrogen availability. Phosphorous is mainly present in inorganic form, adsorbed to soil minerals. Available P moves downward with infiltrating rainwater until free binding places are encountered. In general, therefore, P-containing soils can be seperated into two layers: an upper layer that is (almost) saturated and a layer where there is hardly any P. Unless the top soil is removed up to a depth below this P-front, there are no differences in P-availability between sites with and without topsoil removal (*Figure 2*).

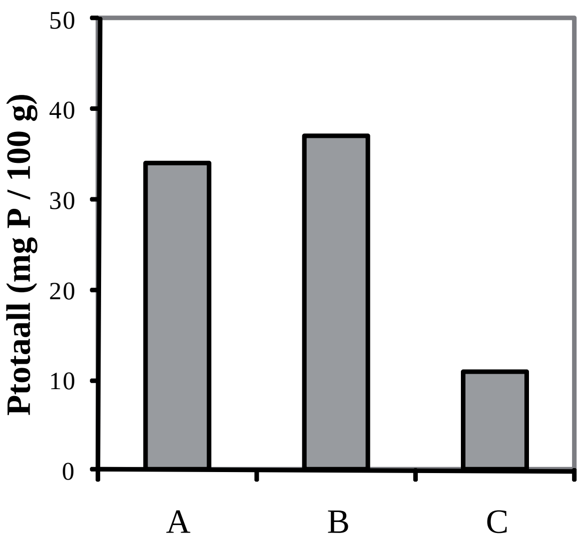

Figure 2: **Total P-content in the soil at site Eexterveld (NL) at different removal depths.**
A = before top soil removal
B = Shallow removal (ca. 15 cm)
D = Deeper removal (> 30 cm)

The productivity in the majority of ecosystems on nutrient-poor soil types is limited by the amount of available nitrogen and such systems react very sensitive to changes in air-borne nitrogen deposition, especially when other nutrients are not in short supply. Additional removal of nutrients through mowing or grazing is essential under such conditions. Ecoystems with P-limitation, either because they have a calcareous and/or iron-rich soil or because all P has been stripped away through topsoil removal, are much less sensitive in this respect.

It can be concluded that topsoil removal is a fast way to remove nutrients if it is carried out well, that is, if all nutrients have been stripped away.

Species addition

In sites where abiotic conditions had been restored and optimised for certain target plant communities species number indeed began to increase spontaneously. However, the number of species was rarely the same as from older, not restored sites with these vegetation types. The so-called saturation index (Wolters et al. 2005), a measure for the degree of similarity between the actual vegetation and an optimal developed version, typically is relatively low, thus showing that many species are lacking in the restored site. Analyses of the buried soil seedbank showed that many target species are no longer present after many years of intensive agricultural exploitation of a site. Based on measured or estimated seed longevity, compiled in Thompson et al. (1997), the fraction of target species that are likely to still have viable seeds after a certain period of exploitation can be assessed. In van Diggelen (1998) such calculations were done for wet meadows (*Table 1*) showing that many species are already lacking after a relatively short period of intensive exploitation. Especially the rarer species were no longer present. Introduction experiments (Strykstra et al. 2002) showed that lacking species often did establish on such sites, implying that the abiotic conditions are suitable for these species and that dispersal barriers prevented their actual appearance.

Large-scale experiments were then carried out (Hoelzel and Otte 2003) targeted at optimising species transfer rates, e.g. by mowing at times when most target species contain ripe seeds, in sites where they are most common, etc. etc.. They showed that under optimal conditions it was possible to transfer as much as 90% of the desired species from the donor to the receptor site. If the transfer is part of the normal mowing management, that is not optimised for species transfer rate but for agricultural efficiency, transfer rates are considerably lower but still significant (*Table 2*).

Table 1: **Seed bank characteristics of selected wetland communities**

Alliance[1]	Number of characteristic vascular plant species >4 records on seed longevity	Seed longevity at least several years[2]	Seed longevity probably several with decades[3]
Junco-Molinion	22	12	8
Calthion palustris	27	14	4
Cynosurion cristatae	19	11	7
Arrhenatherion elatioris	26	16	10
Alopecurion pratensis	13	8	3
Average	**21.4**	**12.2**	**6.4**

[1] names and species composition after Schaminée et al. (1995, 1996)
[2] longevity index cf. Bekker et al. (1998) > 0.49 or recorded seed longevity > 4 years or n (records) indicating persistent seeds > 4
[3] longevity index cf. Bekker et al. (1998) > 0.79 or recorded seed longevity > 19 years

Table 2: **Transfer efficiency of species with hay under a normal mowing management regime**

Alliance[1]	*Calthion palustris*	*Caricetum aquatilis*	Flooded *Calthion palustris*	*Nardo-Galion*	*Caricion curto-nigrae*	*Cynosurion cristatae*	*Allopecurion pratensis*
species in donor vegetation	23	18	32	14	27	27	40
species also present in hay	14	9	9	6	9	19	14
Transfer efficiency (%)	61	50	28	43	33	70	35

Another bottle neck is species establishment. When compared to the number of species with viable seeds that are transferred only about half of them actually do establish. This probably has to do partly with competition for light from other, faster growing, species (Kotowski and van Diggelen 2004) but it occurred even in sites where there were still many open gaps that could be colonised. Obviously there are also other factors that prevent establishment, even when viable seeds are present on the spot.

It can be concluded that species transfer is a technique that significantly enhances the likelihood that a target species gets established at a restored site.

Comparison of techniques

Klimkowska et al. (2007) performed a meta-analaysis on the effectiveness of several alternative techniques to restore wet grasslands, based on published results of 92 cases where different restoration techniques were applied. Interestingly they found that when only single techniques were applied topsoil removal and addition of propagules gave significantly larger increase in saturation index than rewetting. Nevertheless, rewetting is by far the most applied method and known to give sometimes very spectacular results. The authors discuss the effects that rewetting has on increasing nutrient availability in formerly fertilized meadows, thus counteracting the effects of improved hydrological conditions by degrading nutrient status.

In practical applications a combination of two or more techniques is often used. Again, combinations including topsoil removal gave the best results, especially in the case of deeper removal, where all degraded layers were certainly removed as well as all seeds of non-target species.

The largest increase in saturation index was reached under the combination of all three techniques, although the results were not significantly different from a combination of topsoil removal and propagules addition. However, the much larger deviation in the latter case indicates that the results are much less predictable than when all three techniques are applied.

Conclusions

The results presented here show that species richness does increase over time under a regime of biomass removal but, unfortunately the increase rate decreases with time since ecosystem degradation. Increase rate also decreases with increasing species richness. These results imply that it takes a long time to restore species-rich grasslands this way.

Techniques such as topsoil removal or propagule addition can potentially increase the restoration speed significantly but there are considerable differences in effects between these. Topsoil removal and addition of propagules are slightly more effective than just manipulation of abiotic conditions but especially the combination of techniques appeared most effective. In this way restoration can be reasonably fast and reasonably well-developed communities can be attained within a few years.

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