



## Short-term impact of river restoration and grazing on floodplain vegetation in Luxembourg

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### ABSTRACT

In 2003, the Syr River was redirected from a channel running along the valley edge into a slightly reshaped riverbed on the valley floor, after which, in 2004, a low intensity (1.0 livestock units ha<sup>-1</sup>), year-round grazing system with Galloway cattle was introduced on a 24 ha floodplain pasture. 200 permanent plots were randomly distributed on the floodplain pasture to study vegetation development between 2004 and 2006 under the influence of the combined rewetting and grazing regime. The effect of grazing was investigated separately in 16 grazing exclosures (comprising 64 grazed and 64 control plots) in different habitat types during 2005–2007. At each plot, vegetation composition was surveyed using a decimal scale to determine cover estimates, and vegetation structure and grazing impact indicators were determined. Non-metric multidimensional scaling revealed that the restoration measures triggered a marked short-term shift in vegetation composition. The intensity and direction of this shift varied clearly along a topographical gradient in the floodplain. Species richness increased significantly, and typical wetland species and Red Data Book species were favoured in the riparian area. Grazing hindered rapid species turnover since the turnover rate in grazed plots (from 2006 to 2007: 0.29) was significantly ( $p < 0.05$ ) lower than in control plots (0.35). Species richness differed significantly ( $p < 0.001$ ) between 15.8 species per plot in the grazed to 10.1 in the control treatment (in 2007). The short-term trial suggests that river restoration and moderate grazing in combination may represent a promising strategy for the restoration of floodplain wetlands.

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### 1. Introduction

It is widely acknowledged that river restoration measures are needed to restore healthy and diverse floodplain landscapes. The re-establishment of natural river dynamics is the key factor for enhancing floodplain ecology and functionality (Schiemer et al., 1999; Rohde et al., 2006). The abiotic conditions of semi-natural river floodplain systems provide niches for typical wetland species: flood events physically form bare ground habitats, and reconnection of river to floodplain water raises the groundwater table (Cooper et al., 2003; Van Dijk et al., 2007). Flooding is widely regarded as significant vector for plant distribution on floodplains and may therefore enrich plant biodiversity (Bissels et al., 2004). However, many restored floodplain grasslands lose their open character and specific biodiversity as a result of the prevalent land use, mechanised cutting, being abandoned because it is no longer economically viable or possible (Jensen and Schrautzer, 1999).

Moderate grazing can mimic the disturbance regime created by traditional livestock herding practices in floodplains, and is thought to enhance species diversity while maintaining a predominantly open landscape. While grazing and rewetting effects on grasslands are considerable, the combination of moderate grazing and river restoration to promote plant diversity and restore traditional wet grasslands on floodplains has rarely been studied. Furthermore, studies of restoration activity mostly have compared restored sites with abandoned and managed sites elsewhere under similar abiotic conditions. Yet comparisons of restoration schemes should be made at the same site to increase the meaningfulness of the statistical analysis (Legg, 2006; Öckinger et al., 2006).

In this paper, we analyse the short-term effects of the Syr Valley Restoration Project (SVP) on vegetation composition, richness and structure on floodplain grasslands. Since the 19th century, the floodplain of the Syr Valley has been drained by the channelization of the Syr River and the excavation of ditches, and the grasslands have been used as hay meadows mainly in a two cut system. The SVP combined the restoration of the water course by redirecting the Syr River from the channel along the valley edge to the valley bottom in 2003 and introducing a low intensity, year-round grazing system with Galloway cattle in 2004.

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**Table 1**  
Description and area of habitat types in the Syr Valley study area in 2004.

Habitat type	Description, vegetation groups and relative landscape position	Dominant plants	ha
Mesophilic grasslands	Coarse grasslands with mesic to dry moisture levels, <i>Arrhenatherion</i> alliance, valley edges	<i>Alopecurus pratensis</i> L., <i>Arrhenatherum elatius</i> (L.) Beauv. ex J. & C. Presl., <i>Festuca pratensis</i> Huds., <i>Lolium perenne</i> L.	12.8
Ruderal grasslands	Ruderal and nitrophytic communities, <i>Aegopodion podagrariae</i> and <i>Polygonion avicularis</i> alliance, valley edges and cattle attraction sites	<i>Plantago spec.</i> , <i>Cirsium vulgare</i> (Savi) Ten., <i>Urtica dioica</i> L., <i>Rumex obtusifolius</i> L.	1.4
Marsh and tall forb grasslands	Wet grasslands and tall forb meadows, <i>Filipendulion</i> and <i>Calthion</i> alliance, influenced by groundwater, between high water zone and valley edge	<i>Filipendula ulmaria</i> (L.) Maxim., <i>Persicaria amphibia</i> (L.) S.F. Gray, <i>Scirpus sylvaticus</i> L., <i>Equisetum palustre</i> L.	2.4
Flood meadows	Alternating wet and dry grasslands, <i>Agropyron-Rumicion</i> alliance, high water zones	<i>Agrostis stolonifera</i> L., <i>Ranunculus repens</i> L., <i>Carex hirta</i> L., <i>Mentha aquatica</i> L.	3.2
Large sedge swamps	Seasonally flooded and riparian sedge swamps, <i>Magnocaricion</i> alliance, watersides and flood troughs	<i>Carex acutiformis</i> Ehrh., <i>Carex riparia</i> Curt., <i>Carex acuta</i> L., <i>Carex disticha</i> Huds.	2.3
Reeds	Deep or riparian marsh dominated by <i>Phragmition</i> , <i>Magnocaricion</i> alliance, water course and watersides	<i>Phragmites australis</i> L., <i>Phalaris arundinacea</i> L.	2.1
Total			24.2

The SVP aimed to enhance biodiversity on the floodplain, maintain a semi-open landscape character and restore typical wetland species composition. Thus SVP does not aim to restore the 18th century floodplain ecosystem – where historical maps show the floodplain as an open wetland with the Syr River course at the valley bottom which supported unregulated livestock grazing – but to return the floodplain to a healthy and vigorous state, which, according to Bradshaw (1997), is the purpose of ecological restoration. The current definition of restoration as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” by the Society for Ecological Restoration (2004) implies that mimicking traditional land use systems and states constitutes successful restoration when the resulting ecosystems contain enough biotic and abiotic resources to sustain themselves structurally and functionally. However, SVP restoration measures create “new” cultural landscapes with unique elements, which also need to be evaluated in terms of their cultural sustainability (Schaich, 2009).

After river restoration in 2004, the vegetation in the SVP consisted mainly of mesophilic grasslands (*Arrhenatheretalia*) and, to a lesser degree, marsh and tall forb grassland (*Calthion*, *Filipendulion*), flood meadows (*Agropyron-Rumicion*), large sedge swamps (*Magnocaricion*) and reeds (*Phragmition*). Due to river restoration and infrastructural works on the pasture, ruderal and nitrophilous grasslands (*Polygonion avicularis*, *Aegopodion podagrariae*) are found in several small patches. In the absence of further anthropogenic disturbance the climax vegetation in the Syr Valley would consist of willow-alder forest (*Salicion albae*, *Almion glutinosae*) in the riparian areas and hardwood alluvial forests (*Pruno-Fraxinetum*, *Stellario-Carpinetum*) in the upper floodplain areas. On the few steeper slopes at the valley edge, the thermophilic *Pruno-Rubion* alliance would be expected.

The aim of our study was to evaluate whether the combination of river restoration and grazing in the SVP represents a suitable strategy for increasing plant diversity, fostering typical wetland vegetation and preserving an open landscape character from the outset, and to make recommendations for the future management of restored floodplains. The specific research questions were:

- What are the short-term effects of combined river restoration and grazing on vegetation composition, species richness, vegetation structure and site characteristics?

- What are the differences in vegetation composition and species richness between grazed and abandoned sites on this floodplain where the river course has been restored?

## 2. Materials and methods

The study area is situated in the upper part of the Syr floodplain in South-eastern Luxembourg (49°38'N, 6°17'E). The Upper Syr Valley is located in the undulating foothills of the Luxembourg sandstone plateau, which mainly consists of marl and alluvial formations. Floodplain soils are predominantly nutrient rich, loamy gleys (Administration des Eaux et Forêts, 1995). A temperate, western European climate prevails with a mean annual precipitation between 700 and 750 mm and a mean annual temperature between 9 and 9.5 °C. The study area belongs to the 375 ha 'Vallée de la Syre de Moutfort à Roodt/Syre' Special Protected Area, designated in January 2004 under the EU Conservation of Wild Birds Directive and forms part of the Natura 2000 network.

The Syr Valley Restoration Project (SVP) was established along a 2.5 km section of the Syr floodplain in Luxembourg. To redirect the Syr River from the channel at the valley edge into the riverbed on the valley floor, a 400 m long channel was dug through mesic grasslands in 2003, and 1.5 m high berms constructed on each side of the river. Near the valley floor, construction was limited to raising the existing drainage ditch at specific locations; no river berms were present for the next 1.5 km. At the boundary of the restoration area, the river was captured by two constructed meanders 180 m long before flowing back into the riverbed channel. Overall, 4750 m<sup>3</sup> soil was moved. At mean height of the River Syr, approximately 9 ha were rewetted, i.e. 20–120 m on each side of the river – depending on topography – is directly affected by the restoration. The area has been grazed by Galloway cattle year-round since 2004, supplemented by hay from adjacent meadows in winter. The stocking densities varied between 0.8 and 1.0 livestock units ha<sup>-1</sup>.

### 2.1. Sampling design

In 2004, 189 permanent plots of 2 m × 2 m were established randomly along 66 transects on the floodplain pasture. The transects were placed randomly in a 50 m strip perpendicular to the waterway on each side of the valley. Depending on transect length, between two and six plots were placed on each. Plot number was

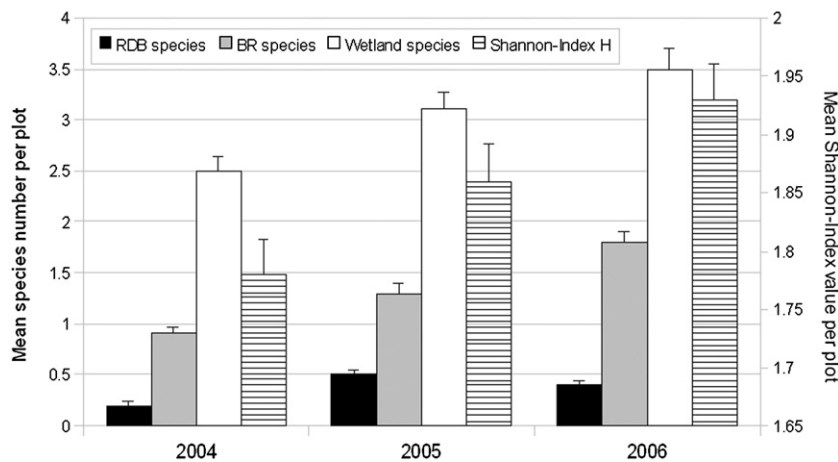


Fig. 1. Shannon-Index H, number of Red Data Book (RDB) and Luxembourg Biodiversity Regulation (BR) species, and characteristic wetland species by permanent plot (standard error as error bars); numbers between all years vary significantly (Shannon-Index and RDB species at  $p < 0.01$ ; BR species and wetland species at  $p < 0.001$ ).

expanded to 200 plots in 2005. Vascular plant cover was recorded in each plot from June to July in the three consecutive years from 2004 to 2006, using a decimal scale (Londo, 1976). Nomenclature of the identified plants followed Lambinon et al. (2004). Additionally, proportion of bare ground, grass cover and herbaceous plant cover was recorded. Mean vegetation height at each 1 m<sup>2</sup> sector of the plot was determined by dropping a 0.25 m<sup>2</sup> styrofoam quadrat (180 g) and measuring the height of the quadrat above the ground when it stopped falling (Stammel et al., 2003). Four such height measurements were used to determine mean height at each of the 4 m<sup>2</sup> plots. The grazing impact and trampling impact were estimated visually using a five and six point ordinal scale, respectively, for the proportion of plants <15 cm and of trampling damage in each permanent plot (Lederbogen et al., 2004). The distance to the main river channel was measured for each permanent plot and the slope determined using a clinometer. In 2005, 16 grazing exclosures were established in different habitat types to assess the effects of grazing compared to abandonment (Sarr, 2002). Patches of vegetation types were delimited by a GPS/GIS tool in the field based on dominant plant species (>25% of total vegetation cover), physiognomic vegetation and site characteristics. Physiognomically similar vegetation types were pooled into habitat types to study vegetation development on a higher spatial scale (Schaich et al., 2010). Based on the 2004 survey of vegetation types, every exclosure was placed in one of the six homogenous habitat types identified (see Table 1) and comprised four sampling plots of 2 m × 2 m inside the barbed-wire fence and a further four plots outside, resulting in a total of 128 plots. This 'paired plot' concept allowed a direct comparison between the two land use alternatives because the site conditions and past land uses were similar (Hayes and Holl, 2003). Cover estimates, plant heights and environmental parameters were recorded similarly to the permanent plots in three consecutive summers from 2005 to 2007.

## 2.2. Data analysis

Species richness, Shannon-Index H and evenness index  $E_{1/D}$  were determined as indicators of vegetation diversity for all permanent and exclosure plots. Mean vegetation height, bare ground proportion and grass or herb cover values were calculated for both plot categories, while mean ordinal grazing and trampling impact were only determined for permanent plots. The number of species listed in the Red Data Book (Colling, 2005) and in the Biodiversity Regulation of Luxembourg (Grand-Duché de Luxembourg, 2002) were calculated for all plots as potential target values for nature

conservation. Plants were defined as wetland species when they were characteristic of syntaxa belonging to the phytosociological classes *Phragmitetea* or *Agrostiotea stoloniferae*, or the order *Molinietalia* (Ellenberg et al., 1992). Cover-weighted means of Ellenberg indicator values were calculated from the permanent plot vegetation data for soil moisture (EIV-M), acidity (EIV-R) and nitrogen (EIV-N) (Ellenberg et al., 1992). Landolt's indicator value for dispersity (LIV-D) was used to identify any lack in aeration in the top soil layer, which is related to the duration of standing groundwater on the floodplain (Landolt, 1977). The general linear model or the non-parametric Friedman-Test was applied to compare the development of all parameter means determined for the permanent plots over three consecutive years (2004–2006). The same parameters were compared for the management alternatives of the paired exclosure plots using the *t*-test, or the Mann-Whitney *U*-test and Wilcoxon-test when the data were not normally distributed. Turnover rates were calculated as an indicator of  $\beta$ -diversity using presence and absence data for each plot by comparing the relevés from one year to another. The turnover rate varied between 0 and 1 and was calculated by the formula:

$$\text{Turnover} = \frac{(A_i + B_i)}{(A_i + B_i + C_i)}$$

where  $A_i$  is the number of species observed in the first relevés exclusively,  $B_i$  is the number of species observed only in the consecutive relevés and  $C_i$  is the number of common species in plot  $i$  during the observation period. Mean turnover rates were determined for the permanent plots and the exclosure plots, and used the paired *t*-test to compare the time periods 2004–2005 and 2005–2006. A one-way ANOVA was carried out to assess differences in mean turnover rates between exclosures of different habitat types. The SPSS 15.0 software was used for all statistical analyses.

Non-metric multidimensional scaling (NMDS) ordination was applied to analyse spatial and temporal gradients in species composition of all permanent plots from 2004 to 2006 with the MASS and the vegan package (version 1.15) of the R programme (version 2.8.0). Prior to the NMDS ordination, the square root of the percentage species cover values was obtained. Wisconsin double standardisation was applied to the NMDS and the Bray-Curtis index was used as the distance measure. NMDS runs were repeated 20 times with random starting configurations in one to five dimensions each. The appropriate number of dimensions (axes) was determined by plotting final stress values against the number of dimensions on a scree plot. A scree plot shows stress as a function of dimensionality of the gradient model. We chose a number

of dimensions beyond which reductions in stress are small (the 'elbow' in the scree plot) (McCune and Grace, 2002). The stress value of the NMDS ordination showed the steepest descent, shifting from a one (48.1) to a two dimensional solution (22.1) (the three and four dimensional solutions had stress values of 17.0 and 13.5, respectively), so a two dimensional solution was used for the analysis of the permanent plot data set. Environmental parameters and indicator values were overlaid onto the NMDS ordination diagram as fitted vectors. *p*-values of the environmental vectors were based on 1000 permutations and only vectors with *p* < 0.1 were plotted.

### 3. Results

A total of 187 vascular plant species was recorded on the permanent plots over the three years of observation (2004: 139 plant species, 2005: 158 and 2006: 162; see Appendix A). Mean species number per plot differed significantly (*p* < 0.001) during the study period, varying between 12.6 (SD: 4.5) in 2004, 14.0 (5.3) in 2005 and 15.1 (5.7) in 2006. Except for evenness  $E_{1/D}$  values, all other indicators of vegetation diversity, rare, and typical wetland species differed significantly over this short monitoring period (Fig. 1). While 119 plant species occurred in all three years, 35 species were newly recorded in 2005 and 13 in 2006. Twelve species, recorded in 2004, were not detected again. A further 8 species were recorded in 2004 and 2006, but not in 2005. In 2004, 12 Red Data Book (RDB) species were recorded, whereas, in 2005 and 2006, 18 RDB species were observed. The mean number of characteristic wetland species amounted to ca. 22% on a permanent plot in 2006. Species of the phytosociological class *Phragmitetea* and the order *Molinietalia* contributed significantly to the increasing pool of wetland species over the years. In 2004 there were three permanent plots with woody species, whereas in 2005 and 2006 woody species were growing in seven and nine plots respectively. Two woody species were recorded (*Prunus spinosa*, *Quercus robur*) in 2004, five in 2005 (species from 2004 plus *Crataegus monogyna*, *Prunus padus*, *Rubus idaeus*) and seven in 2006 (species from 2005 plus *Salix alba*, *Betula pendula*). The mean vegetation height per permanent plot differed significantly, from 50.7 cm (SD 24.0) in 2004 to 32.6 cm (26.2) in 2005 and 35.4 cm (28.7) in 2006 ( $Z=97.35$ , *p* < 0.001). The mean herb and grass cover percentages did not vary significantly over the years and ranged from about 69% for grass, and 24% for

**Table 2**

Mean weighted indicator values per 4 m<sup>2</sup> permanent plot (standard deviation in parentheses): EIV after Ellenberg et al. (1992) and LIV after Landolt (1977).

	2004	2005	2006	F	<i>p</i> <sup>a</sup>
EIV-moisture	6.70 (0.87)	6.83 (0.94)	6.93 (0.99)	8.22	***
EIV-acidity	6.43 (0.73)	6.46 (0.60)	6.51 (0.61)	1.97	ns
EIV-nitrogen	6.04 (0.73)	5.88 (0.45)	5.86 (0.39)	13.08	***
LIV-dispersity	4.42 (0.29)	4.30 (0.34)	4.30 (0.32)	8.98	***

<sup>a</sup> ns, not significant, \**p* < 0.05; \*\**p* < 0.01; \*\*\**p* < 0.001.

herbaceous species. The short-term variation in site characteristics, the indicator values for moisture, nitrogen and soil dispersity differed significantly between the three consecutive sample years (see Table 2).

#### 3.1. Vegetation change and species composition

Given the high number of objects, with 589 plots included in the NMDS ordination, the stress value (22.1) of the two dimensional solution is still acceptable. The *r*<sup>2</sup> values for the non-metric and linear fit of the two dimensions were 0.95 and 0.81 respectively. The indicators of soil moisture and water availability (EIV-M, distance from the river) and the grazing impact (grazing impact, trampling impact) correlated strongly with the first dimension of NMDS (NMDS1) (Table 3). The indicator of nitrogen availability (EIV-N) correlated with the second dimension (NMDS2) as did parameters 'plant height' and 'slope'.

The results of the NMDS ordination of the 2004, 2005 and 2006 permanent plot data indicated a significant short-term variation in vegetation composition (Table 4). Three groups of permanent plots were distinguished in the ordination diagram: riparian plots lay close to the river in the lowest parts of the valley <24 m from the river; floodplain plots were located in the lower and regularly inundated parts of the floodplain between 24 and 48 m from the river; and, finally, plots at the valley edge were situated higher up the valley >48 m from the river. The three groups represent the valley topographical gradient, and also represent different hydrological conditions of groundwater levels and inundation frequencies. The detailed analysis of the groups revealed clear differences in the direction of the vegetation development between the permanent plots situated along this topographical gradient. The riparian and

**Table 3**

Correlation of environmental variables, vegetation parameters and indicator values with NMDS ordination dimensions.

Variables	xNMDS1	yNMDS2	<i>r</i> <sup>2</sup> -value <sup>a</sup>
Distance from the river (DR)	-0.920	-0.391	0.158***
Slope (SL)	-0.694	0.720	0.384***
Plant height (PL)	0.891	0.455	0.437***
Grazing impact (GI)	-0.999	-0.024	0.127***
Trampling impact (TI)	-0.943	-0.332	0.022**
Open ground cover (OGC)	-0.943	0.333	0.003 ns
Ellenberg indicator value – moisture (EIV-M)	0.994	-0.106	0.844***
Ellenberg indicator value – acidity (EIV-R)	0.999	0.042	0.220***
Ellenberg indicator value – nitrogen (EIV-N)	0.104	0.995	0.029***
Landolt indicator value – dispersity (LIV-D)	-0.882	0.470	0.003 ns

<sup>a</sup> *p*-value based on 1000 permutations: \*\*\* significant at the *p* < 0.001 level, \*\* significant at the *p* < 0.01 level, \* significant at the *p* < 0.05 level, ns not significant.

**Table 4**

Mean position values of the permanent plots (standard deviation in parentheses) along both axes of the NMDS in 2004 and 2006.

	xNMDS1				yNMDS2			
	2004	2006	Z	<i>p</i>	2004	2006	Z	<i>p</i>
All plots	-0.064 (0.605)	0.035 (0.638)	5.73	<0.001	-0.018 (0.596)	0.016 (0.443)	2.87	0.004
Riparian <sup>a</sup>	0.284 (0.611)	0.390 (0.659)	4.47	<0.001	0.042 (0.601)	0.062 (0.481)	0.47	0.630
Floodplain <sup>a</sup>	-0.239 (0.525)	-0.126 (0.511)	2.71	0.007	0.021 (0.654)	0.016 (0.489)	1.91	0.056
Valley edge <sup>a</sup>	-0.312 (0.467)	-0.232 (0.525)	2.86	0.004	-0.118 (0.533)	-0.036 (0.344)	2.51	0.012

<sup>a</sup> Riparian: plots situated <24 m from the river; floodplain: plots situated >24 and <48 m from the river; valley edge: plots situated >48 m from the river.

**Table 5**  
Vegetation diversity indices, rare species indices and characteristic wetland species and weighted mean indicator values per 4 m<sup>2</sup> permanent plot along the river valley gradient (standard deviation in parentheses).

	Riparian <sup>a</sup>			Floodplain <sup>a</sup>			Valley edge <sup>a</sup>		
	2004 (n = 73)	2006 (n = 76)	Z <sup>b</sup>	2004 (n = 52)	2006 (n = 58)	Z <sup>b</sup>	2004 (n = 64)	2006 (n = 66)	Z <sup>b</sup>
Species richness per plot	12.5 (4.84)	14.4 (6.01)	2.21*	13.6 (4.59)	15.7 (5.85)	2.26*	12.1 (3.088)	15.0 (4.83)	4.03***
Shannon-Index H	1.80 (0.46)	1.80 (0.51)	0.27 ns	1.81 (0.36)	2.02 (0.39)	3.39**	1.73 (0.39)	2.00 (0.32)	4.43***
Evenness E <sub>1,D</sub>	0.73 (0.12)	0.69 (0.12)	2.22*	0.71 (0.10)	0.75 (0.08)	2.40*	0.71 (0.11)	0.75 (0.07)	2.76**
RDB species per plot	0.2 (0.44)	0.6 (0.77)	3.46**	0.3 (0.57)	0.4 (0.88)	0.67 ns	0.2 (0.39)	0.2 (0.47)	0.73 ns
BR species per plot	0.9 (0.93)	2.0 (1.32)	4.92***	1.2 (1.29)	1.9 (1.66)	2.85**	0.8 (0.97)	1.4 (1.48)	2.54*
Wetland species per plot	2.5 (1.93)	4.2 (2.33)	4.69***	2.7 (2.36)	3.3 (3.00)	1.39 ns	2.4 (1.68)	2.6 (2.62)	0.46 ns
EIV-moisture	6.75 (0.97)	7.40 (1.01)	3.80***	6.59 (0.95)	6.74 (0.88)	1.44 ns	6.61 (0.66)	6.55 (0.83)	0.11 ns
EIV-nitrogen	5.98 (0.37)	5.90 (0.42)	1.65 ns	6.00 (0.44)	5.81 (0.40)	2.26*	6.13 (0.49)	5.87 (0.34)	3.09**
LIV-dispersy	4.39 (0.30)	4.34 (0.37)	0.51 ns	4.41 (0.31)	4.26 (0.30)	2.69**	4.46 (0.28)	4.31 (0.28)	2.84**

<sup>a</sup> Riparian: plots situated <24 m from the river; floodplain: plots situated >24 and <48 m from the river; valley edge: plots situated >48 m from the river.

<sup>b</sup> ns, not significant, \*p < 0.05; \*\*p < 0.01; \*\*\*p < 0.001.

the floodplain plots moved significantly only along the first dimension of the NMDS (NMDS1), with the riparian plots exhibiting a larger displacement than the plots on the floodplain (Table 4). The direction of the mean displacements showed development between both groups towards wetter conditions and more basic soil and species composition responses. The valley edge plots shifted significantly along the NMDS1 and the NMDS2. The direction of the shift along the NMDS1 was equivalent to that of the riparian and floodplain groups. The mean relocation of the plots along the NMDS2 indicated a slight tendency towards nitrophilous species.

The turnover rates differed significantly among the three permanent plot groups for the periods 2004–2005 ( $F = 7.186$ ,  $df = 2$ ,  $p = 0.001$ ) and 2005–2006 ( $F = 5.244$ ,  $df = 2$ ,  $p = 0.006$ ). Compared to the period 2005–2006 (all: 0.38, riparian: 0.41, floodplain: 0.34), the mean turnover rates were significantly higher for all permanent plots ( $p < 0.001$ ), the riparian ( $p = 0.001$ ) and the floodplain group ( $p = 0.002$ ) in the 2004–2005 period (all: 0.43, riparian: 0.48, floodplain: 0.41). Only the mean turnover rates of the permanent plots at the valley edge did not vary significantly ( $p = 0.202$ ) between both periods (2004–2005: 0.39, 2005–2006: 0.37). The short-term development of the diversity indices and species compositions again

revealed differences among the three groups (Table 5). All groups showed significantly higher numbers of species per plot in 2006 than in 2004. The dominance of few single species in the riparian group diminished from 2004 to 2006, whereas the dominance of a few species (mostly grass species) increased in the other groups. Species of high interest for nature conservation were significantly more abundant per plot mainly in the riparian zone.

Most of the species that disappeared from the permanent plots were characteristic of managed semi-dry grasslands (*Campanula patula*, *Knautia arvensis*, *Potentilla argentea*, *Ranunculus bulbosus*) and mesic grasslands (*Anthoxanthum odoratum*, *Cardamine hirsuta*, *Lolium multiflorum*), or ruderal and eutrophic grasslands of wet sites (*Chenopodium polyspermum*, *Eupatorium cannabinum*, *Matricaria recutita*, *Trifolium fragiferum*). *Ranunculus repens*, characteristic of wet grasslands and flood meadows, was the only species that exhibited significantly lower cover over time during the study period in all three groups (Table 6). Grass species characteristic for intensively managed grasslands (e.g. *Festuca rubra*, *Phleum pratense*) showed lower cover over time in the riparian groups, but higher values in the other groups. Most of the 21 species with higher cover values over the study period belonged to characteristic wet-

**Table 6**  
Plant species for which the observed cover values within the permanent plots changed significantly between 2004 and 2006 ( $p < 0.05$ , Wilcoxon-Test): percentage mean cover.

		Riparian <sup>a</sup>		Floodplain <sup>a</sup>		Valley edge <sup>a</sup>	
		2004	2006	2004	2006	2004	2006
Increased	<i>Agrostis stolonifera</i>	12.3	25.1	12.5	29.8	15.8	22.4
	<i>Alopecurus pratensis</i>	9.4	11.6	9.1	14.5	7.8	16.9
	<i>Atriplex prostrata</i>	0.01	0.1	0	0.07	0	0.09
	<i>Callitriche palustris</i>	0	2.5	0	0	0	0
	<i>Carex acutiformis</i>	4.9	17.2	4.7	16.3	3.3	3.6
	<i>Carex hirta</i>	1.1	2.6	0.92	3.1	1.4	1.6
	<i>Cerastium fontanum</i>	0.14	0.19	0.11	0.36	0.16	0.27
	<i>Cirsium arvense</i>	0.14	0.21	0.24	1.8	0.01	0.05
	<i>Epilobium hirsutum</i>	0.64	1.1	0.13	0.09	0.08	0.22
	<i>Epilobium parviflorum</i>	0	0.03	0	0.06	0	0.14
	<i>Epilobium tetragonum</i>	0.06	0.26	0.03	0.12	0.03	0.26
	<i>Equisetum palustre</i>	0.14	1.4	0.05	0.69	1.9	2.7
	<i>Galeopsis tetrahit</i>	0	0.04	0	0.03	0	0.02
	<i>Lycopus europaeus</i>	0	0.11	0	0	0	0.01
	<i>Lythrum salicaria</i>	0.34	0.96	0.1	0.22	0.03	0.21
	<i>Poa trivialis</i>	6.6	20.7	8.3	16.7	10.6	24.9
	<i>Persicaria amphibia</i>	1.1	1.7	0.63	1.1	0.68	1.2
	<i>Ranunculus acris</i>	0.05	5.9	0.26	2.7	0.38	1.0
	<i>Rumex crispus</i>	0.40	1.2	0.53	0.43	0.49	1.5
<i>Sonchus asper</i>	0.02	0.04	0	0.03	0	0.05	
<i>Veronica beccabunga</i>	0	1.5	0.01	0.01	0	0	
Decreased	<i>Ranunculus repens</i>	12.9	6.1	14.0	7.9	18.5	6.9
Decreased (riparian) and Increased (floodplain & valley edge)	<i>Festuca pratensis</i>	7.5	7.5	4.2	16.2	9.2	18.7
	<i>Festuca rubra</i>	1.5	0.13	0.27	0.31	0.10	0.09
	<i>Phleum pratense</i>	3.3	1.7	1.6	2.6	3.0	4.5

<sup>a</sup> Riparian: plots situated <24 m from the river; floodplain: plots situated >24 and <48 m from the river; valley edge: plots situated >48 m from the river.

**Table 7**Vegetation diversity indices in 16 exclosures each with four ungrazed and grazed plots of 4 m<sup>2</sup> (standard deviation in parentheses).

	2005			2006			2007		
	Ungrazed	Grazed	Sig. <sup>a</sup>	Ungrazed	Grazed	Sig. <sup>a</sup>	Ungrazed	Grazed	Sig. <sup>a</sup>
Total species richness	101	101	–	99	113	–	88	108	–
Mean species richness per plot	14.41 (4.55)	15.13 (5.14)	ns t	11.83 (4.84)	16.23 (5.77)	*** t	10.09 (3.83)	15.83 (6.14)	*** U
Mean species richness per exclosure	22.31 (6.92)	23.63 (7.26)	ns t	19.00 (6.97)	25.40 (8.74)	* t	16.19 (6.48)	24.63 (9.05)	** t
Shannon-Index H	1.79 (0.49)	1.88 (0.42)	ns U	1.67 (0.53)	1.97 (0.43)	** t	1.64 (0.42)	2.02 (0.44)	*** t
Evenness $E_{1/D}$	0.67 (0.14)	0.70 (0.10)	ns U	0.69 (0.14)	0.72 (0.09)	ns t	0.73 (0.11)	0.75 (0.09)	ns t
Total number of RDB species	9	9		9	11		8	12	
Mean number of RDB species per plot	0.45 (0.81)	0.56 (0.88)	ns t	0.36 (0.72)	0.42 (0.79)	ns t	0.36 (0.62)	0.51 (0.83)	ns t
Mean number of BR per plot	1.28 (1.30)	1.53 (1.36)	Ns t	1.20 (1.04)	1.61 (1.19)	* t	1.06 (.88)	1.53 (1.21)	ns t

<sup>a</sup> ns, not significant, \* $p < 0.05$ ; \*\* $p < 0.01$ ; \*\*\* $p < 0.001$ ; t, student's  $t$ -test, U, Mann-Whitney  $U$ -Test.

land habitat types: wet grasslands (*Agrostis stolonifera*, *Cerastium fontanum*, *Equisetum palustre*, *Poa trivialis*, *Persicaria amphibia*, *Ranunculus acris*), marsh and tall forb grasslands (*Epilobium* spp., *Galeopsis tetrahit*, *Lycopus europaeus*, *Lythrum salicaria*), large sedge swamps (*Carex acutiformis*, *Carex hirta*) and semi-aquatic habitats (*Callitriche palustris*, *Veronica beccabunga*).

### 3.2. Response of vegetation diversity to grazing

Analysis of the grazing exclosures revealed considerable differences between grazed and abandoned plots over the study period (Table 7). In 2006 and 2007 a larger number of species was observed in the grazed plots, and species richness and the Shannon-Index value were significantly higher than in 2005, immediately after the exclusion of the cattle. The total number of Red Data Book species was also higher under the grazing treatment. However, the number of endangered species per plot did not vary significantly between the management alternatives, possibly due to the enormous variability in values for the RDB species.

The number of exclosure plots with woody species varied from 3 (ungrazed) and 4 (grazed) in 2005 (*C. monogyna*, *P. spinosa*, *Q. robur*), to 6 (ungrazed) and 3 (grazed) in 2007 (species from 2005 plus *Amelanchier ovalis*). The mean vegetation height, an indicator of vertical vegetation structure, did not differ significantly ( $Z = 1.840$ ,  $p = 0.066$ ) between the management alternatives in 2005, with 35.5 cm (SD 19.4) in the ungrazed and 29.4 cm (SD 17.4) in the grazed plots. In 2007 the vegetation was significantly taller ( $t = 5.245$ ,  $p < 0.001$ ) in the abandoned plots (53.0 cm, SD 28.0) than in the grazed plots (30.0 cm, SD 21.0). The mean percentage of bare ground did not vary significantly ( $Z = 2.258$ ,  $p = 0.24$ ) in 2005, but exhibited a significant difference ( $Z = 4.500$ ,  $p < 0.001$ ) in 2007 (ungrazed: 1.3%, SD 3.1; grazed: 5.6, SD 8.1). The mean percentage of grass and herb cover did not differ in 2005, but there was a significant difference ( $t = 3.277$ ,  $p = 0.001$ ) between grass cover in the pasture (74.7%, SD 20.3) and the abandoned sites (61.4%, SD 25.3) in 2007.

The turnover rate differed significantly between the pasture and the abandoned areas in the periods 2005–2006 ( $p = 0.006$ ) and 2006–2007 ( $p = 0.014$ ) with higher turnover rates in the ungrazed (2005–2006: 0.39; 2006–2007: 0.35) than in the grazed treatment (0.33; 0.29). When the grazed, and ungrazed exclosure plots were pooled for analysis, a significant difference in the turnover rates was also detected between the six habitat types in 2005–2006 ( $F = 2.644$ ,  $df = 5$ ,  $p = 0.026$ ) and 2006–2007 ( $F = 5.500$ ,  $df = 5$ ,  $p < 0.001$ ). This indicated that the habitat types responded differently to the river restoration measure in terms of quality and speed.

## 4. Discussion

This study revealed major short-term changes in the vegetation composition of alluvial meadows in the first three years after

floodplain restoration. The intensity and direction of these changes clearly depended on the topographical gradient on the floodplain. These major shifts were related primarily to the restoration of the hydrological regime, as the decreasing turnover rates along the valley slope indicated. The turnover rate in the riparian zone was higher and differed significantly between the first two time periods. Such drastic changes to the vegetation composition due to fluctuations in the water level have also been reported by Leyer (2005) and Hausman et al. (2007). They identified height to the water table as the primary factor controlling vegetation composition in wetland restoration activities. In the SVP we identified many wetland species that increased their cover due to higher groundwater levels like e.g. *C. hirta* or *L. salicaria* or more frequent flood events like *C. palustris*. However, grazing also played a vital role in influencing the vegetation and topsoil structure, and thus influencing the competitive interaction between species (Moran et al., 2008). The plant and structural diversity of the grazed plots were significantly higher in the SVP, and several plant species increased their cover over the three-year study period, profiting either from grazing impact or the cattle's selective disregard (*C. arvensis*, *E. palustre*, *R. acris*, *R. crispus*). The change in management, from mowing in most parts of the former floodplain to a moderate grazing system characterised by heterogeneous spatial effects, may also have contributed to the shift detected by diversity indicators and turnover rate. The adverse effects of the change in the management regime on grassland vegetation detected in other studies (Fischer and Wipf, 2002; Matejková et al., 2003) was not confirmed.

The indirect measurements of site characteristics via indicator values revealed increasing soil moisture (EIV-M) as the decisive factor for changes in the riparian area, and partly also in the floodplain area. By contrast, the shifts at the valley edge and in the floodplain areas were mostly related to decreasing nutrient levels (EIV-N) and a higher oxygen supply (indicated by the lower value of LIV-D). This contrasted with the higher nutrient levels observed by Van Dijk et al. (2007) as a result of rewetting on peat meadows. The analysis of vegetation composition and indicator values in our study revealed no significant increase in nutrients as a result of the restored floodplain dynamics and inundations. In areas close to the river, this finding might be explained by the restricted availability of nitrogen (denitrification, limited nitrogen mineralisation) due to increased soil moisture (Berendse et al., 1994). A further explanation for the restriction of N in the valley edge areas could be the occurrence of rare extreme inundations over the study period, as well as the translocation of the riverbed from the valley edge to the valley bottom. Nutrient shifts from the riparian to the valley edge zone due to cattle grazing and defecation behaviour further balanced the nutrient distribution in the SVP area (Schaich et al., 2010). The relocation of the river may also have caused the lower dispersity values, indicating higher soil permeability and aeration in the floodplain and valley edge zone.

#### 4.1. Effects on vegetation diversity

The species number grew and the species richness, as well as the number of endangered species and typical wetland plants, increased under the management of the restored floodplain site in contrast to studies using only rewetting to restore wetland areas, which have reported negative effects on vegetation diversity (Van Dijk et al., 2007; Vercoutere et al., 2007). The low intensity grazing system appears to provide the necessary disturbances in grassland cover and creates the regeneration niches necessary for new species, thereby fostering species diversity in wet grasslands. Similar conclusions have been drawn for fens (Stammel et al., 2003; Ausden et al., 2005) and coastal grasslands (Loucougaray et al., 2004).

The increase in, and recovery of the numbers of rare species, as indicated by the occurrences of species listed in the Red Data Book and the Luxembourg Biodiversity Regulation appears unrelated to the grazing system, and was concentrated mainly on the riparian parts of the floodplain. This fact, although reported in other studies (Pykälä, 2003), must be viewed as a preliminary finding, as the recovery of rare species is known to take a long time, especially in nutrient rich ecosystems. However, the moderate grazing pressure did not restrict the establishment of rare and endangered species in the riparian zone, which are especially prone to trampling like *V. beccabunga* or *Iris pseudacorus*. Thus a carefully designed, low-intensity grazing regime may be less destructive in riparian areas and on floodplains than traditional intensive grazing regimes reported by Fleischner (1994) or Clary and Kinney (2002).

The riparian zone exhibited a less even distribution of species and cover than the floodplain and valley edge zones, which may be related to the greater disturbances adjacent to the water course (e.g. by flooding or stream bank creations) followed by a higher habitat heterogeneity and the provision of new habitats. At the same time, the increase in dominance of a few species in the outer parts of the floodplain is due to gains in grass species cover (*A. stolonifera*, *Alopecurus pratensis*, *P. trivialis*, and in the outer parts especially *Festuca pratensis* and *P. pratense*), which is – as the results of the enclosure analysis also demonstrated – mainly due to their clonal reproduction and resprouting capacity in response to herbivory. Paine and Ribic (2002) and Pavlu et al. (2007) also observed higher proportions of grass species under grazing, but according to Van Dijk et al. (2007), this may also be partly related to rewetting and the higher groundwater table.

#### 4.2. Grazing as a 'new' paradigm in the restoration of floodplain grasslands?

Grazing is not necessarily beneficial to grassland vegetation and diversity, but, in this case study, the moderate grazing regime seems to overcome the problem of limited recruitment in the vegetation cover. However, Houlahan et al. (2006) demonstrated that adjacent land use practices and intensities are of critical importance to species richness and composition of wetlands, and consequently to their conservation and restoration. Therefore, dispersal limitation could possibly be deemed less problematic in the Syr Valley than in other areas – the headwaters of the river were not heavily modified and are also part of the Natura 2000 network, and may have contributed to the positive development in plant diversity (Paine and Ribic, 2002).

Oppermann and Merenlender (2000) reported that the exclusion of livestock along streams in California was insufficient to promote alluvial forests, because deer herbivory effectively hindered the recovery of woody riparian species. An early successional study of grasslands in Sweden found no differences in species richness or abundance between the management alternatives abandonment and grazing (Öckinger et al., 2006). Therefore, 'aban-

donment' may be considered a cost-efficient management option in regions with a sufficiently high abundance of wild herbivores to preserve semi-open floodplain landscapes. The results of this study showed, however, that, in the short-term, species diversity and also structural diversity in terms of variation of vegetation height and bare ground proportion is much lower than for a managed grazing system. Other studies of different grassland types confirmed these findings and pointed to the suppression of strong competitors and the higher numbers of habitat niches for small and specialised plant species under grazing systems of management (Smith and Rushton, 1994; Ausden et al., 2005).

Changes in the management of grassland ecosystems may result in negative effects on species diversity and composition (Smith and Rushton, 1994; Jantunen, 2003). Such problems could not be detected in our trial, as the grassland diversity and the number of typical wetland plant species increased significantly. When compared to the effects of abandonment, mowing, as a method to prevent succession, also tends to increase species diversity in semi-natural grasslands. However, the raised groundwater levels brought about by the river restoration measures hamper mechanised cutting in winter, and, even in summer, the mowing of the entire riparian and floodplain area would not be possible without damaging the top soil layers considerably. A key factor of the effect of low intensity, year-round grazing on species diversity seems to be the correct adjustment of stocking densities. The grazing system should foster spatial variability in grazing pressure so that even species adversely affected by trampling, browsing or defecation of livestock can persist in areas less intensively used by the herbivores (Ausden et al., 2005). The effects of river restoration and moderate grazing in combination appear to enable principally those species to expand that spread vegetatively via rhizomes or stolons (e.g. *Carex* spp., *A. stolonifera*) and species that profit from selective grazing (e.g. *R. crispus*, *E. palustre*) and/or regular flooding (e.g. *V. beccabunga*, *P. amphibia*). Adaptive management of livestock stocking rates and the restoration measures applied should be backed by long-term, consistent monitoring programmes of such restoration projects on floodplains.

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#### Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.agee.2010.07.012.

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