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Ecological assessment of different riverbank revitalisation measures to restore riparian vegetation in a highly modified river

Ökologische Bewertung unterschiedlicher Uferstrukturierungsmaßnahmen zur Renaturierung der Ufervegetation eines anthropogen geprägten Flusses

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Abstract

Since anthropogenic activities have become concentrated along rivers, river regulations have strongly reduced the lateral connectivity by separating rivers from their floodplains. Consequently riparian habitat heterogeneity and the related species diversity are degrading, especially in highly modified prealpine rivers. Riverbank revitalisation measures aim at mitigating this degradation and river restoration projects have become widespread. Nonetheless, little knowledge exists about their specific outcome, as standardised monitoring programs are missing. The aim of this study is to systematically compare vegetation change in response to three contrasting measures of bank diversification, i.e. embankment removal, sand input or gravel addition. Moreover, the influence of these measures on adjacent vegetation is studied. Conclusions were drawn on the basis of three common goals of restoration projects: (i) improvement of vegetation structure, (ii) increase of species diversity, and (iii) characteristic species composition. The field work was done along River Inn northeast of Munich. Vegetation structure, species identity and cover as well as selected habitat variables were recorded in a stratified randomised sampling design; variation between measures was analysed using uni- and multivariate statistics. We detected great differences in the effect of the three measures two years after implementation. Embankment removal initiated highly dynamic habitats where plant establishment was difficult. The input of sand led to a rather homogenous species composition, at least partly because the habitats were productive and therefore most likely will develop to tall reed stands or riparian forests. After gravel addition the restored sites remained relatively open, while riparian pioneer species could colonise. Vegetation structure and composition of adjacent reed stands were positively affected. The results indicate how restoration outcomes can vary depending on the specific measures chosen. This confirms the need for careful consideration of the pursued goals and site-specific conditions prior to implementation as well as long-term monitoring after implementation.

Keywords: embankment removal, floodplain, gravel addition, monitoring, restoration, sand input

Erweiterte deutsche Zusammenfassung am Ende des Artikels

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1. Introduction

River ecosystems are among the biologically most diverse ecosystems in the world (WARD et al. 1999), and they provide important ecosystem services like flood mitigation, nutrient exchange, drinking water supply and habitat provisioning (SCHOLZ et al. 2012). At the same time river biodiversity and the associated ecosystem processes are negatively affected (WARD et al. 1999), since anthropogenic activities have become concentrated in riparian zones (HABERSACK & PIÉGAY 2008). River regulations alter natural longitudinal and lateral dynamics, which sustain high riverine habitat heterogeneity and species diversity (WARD et al. 1999). Yet, as demonstrated by the Flood Pulse Concept (JUNK et al. 1989), strong hydrological and ecological interconnections of rivers and their floodplains are the basis for the functioning of riverine ecosystems.

Riparian habitats and communities are particularly affected by these changes, as rivers and their floodplains have become separated by embankments and straightening (STAMMEL et al. 2012). The riparian species richness generally decreases as habitat diversity gets lost (WARD et al. 1999). Specialists like riparian pioneers are outcompeted by woody species (KARRENBERG et al. 2002) or herbaceous perennials (ELLENBERG & LEUSCHNER 2010), when connectivity and disturbance by floods are missing (TOCKNER et al. 1999). This often results in a 'single-channel' morphology and straight bank lines with a dominance of monotonous tall reed stands dominated by herbaceous perennials, while there is a loss of diverse open communities dominated by annual pioneers, which are the most threatened species in floodplains (VON HEßBERG 2003).

In order to counteract these degradations, river restoration projects have increased exponentially in the past years, including a large number of actions on riverbanks (BERNHARDT et al. 2005). Nevertheless, although there is a large diversity of restoration measures, little is known about their impact on flora, fauna and the surrounding landscape (BERNHARDT et al. 2005). This is mainly due to a lack of monitoring programs, which meet statistical and scientific demands (CHAPMAN & UNDERWOOD 2000, BERNHARDT et al. 2005). Moreover, the studies which fit scientific requirements focus mostly on the evaluation of the effectiveness of single types of restoration measures in comparison to control sites (MÜLLER et al. 2014), like removing bank fixation (JÄHNIG et al. 2009, JANUSCHKE et al. 2011), erosion control by gravel addition (BARITEAU et al. 2013), or creation of new floodplain channels (SCHAICH et al. 2010, STAMMEL et al. 2012). The publications, which compare the effects of different types of restoration, focus mostly on aquatic organisms like fish and macroinvertebrates (MÜLLER et al. 2014), or specific hydraulic aspects like the effect of embankment techniques on biodiversity (CAVAILLÉ et al. 2013). In contrast, comparative studies take rarely into account riparian vegetation, although it is a suitable indicator for restoration success (JÄHNIG et al. 2009, JANUSCHKE et al. 2011). Plant species respond relatively fast to riverbank restoration, due to effective dispersal or persistence in the soil seed bank (JÄHNIG et al. 2009), and they are the first to indicate changes in abiotic conditions (ELLENBERG et al. 1992). As our investigations as well as many monitoring programs are carried out only a few years after restoration, riparian vegetation is a suitable indicator showing early developments.

Besides systematic comparative studies of contrasting types of restoration measures, it is important to evaluate whether there are positive effects on adjacent vegetation and habitat types. Even if some studies exist concerning the influence of land use on riparian forests (cf. FERREIRA et al. 2005, FERNANDES et al. 2011), none of them deals with the specific effects of restoration measures. However, such knowledge is of great practical relevance, since it answers the question of possible positive effects of small-scale restoration measures on larger areas. This is particularly important in highly modified rivers where space for restoration measures is limited (DE NOOIJ et al. 2006).

The analysis of the restored communities often focuses on the presence of target species, rare species or habitat specialists, and the absence of alien plants (HOELZEL & OTTE 2001, JÄHNIG et al. 2009, CAVAILLÉ et al. 2013, JANUSCHKE 2014). However, there is a problem with defining target species in riparian vegetation, and a potential conflict between total species diversity and presence of rare species, especially in ruderal sites of restored rivers.

Here we focus on the prealpine River Inn as suitable study system to better understand the effects of contrasting measures that enhance natural dynamics and structural diversity of riparian vegetation. The removal of embankment as well as the addition of gravel or sand aim to (i) improve structural diversity, (ii) increase species diversity, and (iii) create a characteristic species composition. The aim of our study is to compare these different revitalisation treatments (removal of embankment, sand input or gravel addition) by analysing vegetation change as indicator of restoration success. In addition, the influence of revitalisation measures on adjacent vegetation types is investigated. We ask the following specific questions: (1) Is the removal of the embankment or sediment input better suited for increasing the riparian diversity? (2) Does vegetation diversity (structure, small-scale variability, species richness) in prealpine rivers respond differently when gravel or sand is added? (3) Can an impact of the revitalisation measures be observed in adjacent vegetation?

2. Material and methods

2.1 Study area and revitalisation measures

The study area is located in the upper valley of the Bavarian Inn, ca. 60 km east from Munich, between the barrages Feldkirchen in the south (RW 4511890, HW 5313609, 440 m a. s. l.) and Gars in the north (RW 4523404, HW 5335074, 412 m). The River Inn originates at about 2,500 m altitude in Switzerland and flows into the Danube in Passau (300 m). Two thirds of its total length of 500 km is located in the Alps resulting in a flow regime that is highly influenced by mountains.

In the study area, River Inn crosses the prealpine Inn-Chiemsee hilly region (*Inn-Chiemsee-Hügelland*, MEYNEN et al. 1962), the end moraine and the Lower Inn Valley (*Isar-Inn-Schotterplatten*, MEYNEN et al. 1962). The studied river section is situated in the braided reach. It has a slope of 0.5-1.0‰ and its floodplain can be attributed to the group of (pre)alpine floodplains (*gefällereiche Flussaue der Alpen / Voralpen*; KOENZEN et al. 2005). The natural substrate type of the river and its floodplain is gravel (MEYNEN et al. 1962, KOENZEN 2005). According to ECKELMANN et al. (1996) this grain fraction is round shaped and >2 mm. The mean discharge (1965–2012) is 221 $m^3 s^{-1}$ in winter and 490 $m^3 s^{-1}$ in summer in Wasserburg, which represents the centre of the investigated river section (LFU 2015). As many other European rivers the Inn is nowadays highly modified. A total of 24 barrages were constructed; four of them located in the study area (Feldkirchen, Wasserburg, Teufelsbruck, Gars).

The barrages and other intensive hydro-engineering activities led to a loss of the natural river dynamics. The straightening caused a lowering of the riverbed and of the groundwater level (CONRAD-BRAUNER 1994). Due to the construction of hydropower plants, the river carries almost no gravel anymore, but mostly silt, sand and organic material (< 2 mm; CONRAD-BRAUNER 1994). The mean daily transport of suspended solid materials is 4.5 kg s^{-1}. During flooding these fine sediments are deposited along the riverbank, resulting in a strong reduction of floodplain dynamics, little turnover of sand or gravel bars (CONRAD-BRAUNER 1994), and a disconnection between river and floodplain (HABERSACK & PIÉGAY 2008).

In order to counteract these degradation processes, the revitalisation along River Inn aimed at improving the ecological quality of the river and its floodplains as well as at increasing the natural dynamics. Therefore, several restoration and revitalisation measures were carried out in 15 sites within a ca. 35 km long section of the river in winter 2012/13. For the purpose of this study only the riverbank diversification measures were considered and solely their impact on riparian vegetation was studied. They aimed at creating more diverse riparian habitats by adding or removing sediment. All sediments originated from the study area to avoid introduction of non-native species and genotypes. In particular, three measures can be distinguished: (i) embankment removal, (ii) sand input, or (iii) gravel addition.

i. Embankment removal consisted in the removal of constructed reinforcements or opening of naturally fixed embankments by using excavators. The resulting open sites should facilitate subsequent erosion. The substrate type favoured by this measure was mostly sand.
ii. Sand input was done in front of the bank lines. The sand originated from the digging of oxbow lakes in the direct surroundings. It was deposited in form of piles of variable height (1.0–2.5 m) and became eroded by river dynamics. The gravel cover in these sites was 0–10%.
iii. Gravel addition was also done in front of the straight or opened bank; the material was available from local deposit. In most cases, gravel was introduced in form of flat groynes. The slope of these sites is half as steep as the one of the two other measures. Gravel cover ranged from 70–100% in these sites.

Three representative study sites were chosen for each of the three measure types. If more than three sites were available, the choice was determined by accessibility. For the study of the adjacent vegetation types five locations were chosen, and reference sites were selected as a control for each of these five sites. The references represent the regulated pre-restoration state and are located upstream from each restored section. Thus, they are similar in terms of habitat conditions and exposed to similar disturbance regimes, albeit without being affected by the restoration measure.

2.2 Study design and vegetation sampling

For the evaluation of floristic patterns, the survey was conducted on two spatial scales. On a smaller scale, a map of riparian habitat types was obtained by studying aerial photographs of summer 2012. As most of these pictures were taken prior to restoration, information was completed by more recent pictures from GoogleEarth (GOOGLE INC. 2012) and verified in the field. Thus, homogenous structures were delimited and assigned to three different categories. A predefined standard was established for the mapping in order to prevent interpretation errors and to enable the reproducibility of the mapping:

- *Reed*: herbaceous vegetation, which can be distinguished from shrubs by a different shade of green and missing shadows;
- *Bare soil with pioneer vegetation*: sparsely vegetated or vegetation free areas with clear characteristics of exposed sediments;
- We further distinguished between sand and gravel dominated *bare soil* habitats on aerial photographs if possible and verified it in the field.

On a larger scale, vegetation and abiotic variables were studied in the field using a stratified-randomised sampling design. The different strata corresponded to the habitat types delineated on the aerial photographs and described above. In each stratum five units were randomly selected at each location. The randomisation was carried out in two steps for each stratum. Firstly the placement on the longitudinal axis was randomly selected. Secondly the plot was again placed randomly on a transverse line through the first selected point. In narrow vegetation stripes, the plots were located in the middle of these strata, in order to avoid edge effects. The plot size was adapted to each habitat type in order to meet the requirements of the minimum area (MÜLLER-DOMBOIS & ELLENBERG 1974). In reed zones the plots measured 1.4 x 1.4 m, in pioneer vegetation 1.0 x 1.0 m.

We chose three locations for each measure type (embankment removal, addition of gravel, introduction of sand) and five for reed and control plots, thus investigating 15 plots per measure type (3 x 5 plots) and 25 plots per treatment (5 x 5 plots). Thus, the study design resulted in a total of 95 sampling plots.

The fieldwork was carried out between mid-June and early August 2014, when riparian vegetation was most developed. The order of the study sites was random in order to compensate for phenological differences between early and late sampling. Vegetation was recorded according to the BRAUN-BLANQUET (1964) method, which was slightly modified to fit our purposes. The vegetation cover was sampled instead of the abundance and number of individuals. The cover of each species, the proportion of vegetation, moss, litter and bare ground were estimated, and the mean vegetation height was measured. Nomenclature of plant species followed BUTTLER & HAND (2008).

In addition, soil pH, distance to the riverbank and the slope of the riverbank were recorded. The pH value of the soil was measured using pH-indicator strips following the protocol of ECKELMANN et al. (1996). The slope of the riverbank was studied with a level instrument (Theis Tecomat 5/8''); because of the limited visibility, it could not be studied in reed zones. An orthogonal transect from the waterfront towards the plot until the forest or reed edge was established for each plot.

2.3 Conservation status and habitat specialism

Plants were identified as target species for pioneer vegetation or reed zones according to the classification of ELLENBERG et al. (1992). They were described as target species for pioneer vegetation on bare soil habitats when attributed to one of the following phytosociological classes: *Isoëto-Littorelletea* Br.-Bl. Et Vlieger in Vlieger 1937, *Scheuchzerio-Caricetea nigrae* Tüxen 1937, *Isoëto-Nanojuncetea* Br.-Bl. et Tx. ex Westhoff et al. 1946, *Bidentetea tripartitae* Tx. et al. ex von Rochow 1951 or *Agrostietea stoloniferae* Oberdorfer et al. 1967 (= *Polygono-Potentilletalia anserinae* Tx. 1947).Species were defined as target species for reed-zones when they were characteristic for the phytosociological class *Phragmito-Magnocaricetea* Klika in Klika et Novák 1941, the orders *Calystegietalia sepium* Tx. ex Moor 1958 or *Molinietalia cearuleae* W. Koch 1926 or the alliance *Filipendulion ulmariae* Segal ex Lohmeyer in Oberd. et al. 1967. Species were called threatened when they were assigned to the categories 1, 2, 3 or V (V = likely to become endangered in the near future) in the Red List of Germany (LUDWIG & SCHNITTLER 1996) or Bavaria (LFU 2003). Alien species were determined according to the black list of invasive species (NEHRING et al. 2013), and the list of neophytes from the BFN (2014).

The definition of moisture indicating species was based on Ellenberg indicator values (ELLENBERG et al. 1992). We calculated the median of the moisture value over all species to compare the average moisture value of different vegetation units. In order to investigate whether the strategies of the riparian vegetation differed between treatments, life-forms according to the classification of Raunkiaer (MÜLLER-DOMBOIS & ELLENBERG 1974) were analysed. The amount of therophytes indicates the disturbance regime, whereas the abundance of phanerophytes shows the regeneration of riparian forest.

2.4 Data analyses

Alpha-diversity was analysed directly via the number of species per plot as well as by the determination of the evenness index, in order to get information about the relative abundance of species. It was calculated using the following equation:

$$E = \frac{Hs}{lnS}$$

where Hs is the alpha-diversity according to SHANNON (1948) and S is the total number of species.

Non-metric multidimensional scaling (NMDS) based on Bray-Curtis dissimilarities was computed to visualise variation within and among types of measure. It aimed to investigate whether species communities differ depending on the revitalisation measures. The appropriate number of dimensions was determined by evaluating a scree plot choosing the number of axes, beyond which stress values do not reduce considerably anymore (MCCUNE & GRACE 2002). Abiotic variables were subsequently overlaid in the NMDS graph as explicating variables. Species used for emphasising results and statements could clearly be assigned to one measure type on the ordination graphs. In order to test whether there are significant differences in species composition PERMANOVA (permutational multivariate analysis of variance; ANDERSON (2001) based on Bray-Curtis dissimilarities was applied.

Univariate statistics were investigated to test significant differences in the distribution of the measures and to support the results of the multivariate analyses. The normality of the data and the homogeneity of variances were tested using the Shapiro-Wilk test and the Levene test. In case of normality and homogeneity of variances a t-test was computed for pairwise comparisons and one-way ANOVA for more than two variables (*a posteriori* Tukey-HSD). Yet the Mann-Whitney U test for pairwise comparisons or the Kruskal-Wallis test (*a posteriori* Tamhane-T2) for multiple comparisons was chosen if data were not normally distributed. Non-parametric Spearman-rank correlations (rho) were calculated to test relations between different variables, since normality and homogeneity of variances could not be detected for all data.

The univariate statistics were carried out using the IBM SPSS Statistics 22 (IBM® 2013) software. For multivariate analysis the statistical software program PCOrd (McCune & Mefford 2011) was used.

3. Results

3.1 Structural diversity

A total of 45 plots were analysed to compare the three types of riverbank revitalisation measures: (i) embankment removal, (ii) sand input, and (iii) gravel addition. Vegetation structure differed significantly between the three types of measure. The removal of the embankment resulted in a significantly higher cover of bare soil ($p < 0.001$) in comparison to sand input, whereas gravel addition showed intermediate values and no significant differences to the other types of measure. Vegetation cover and height were significantly lower on plots with gravel than on those with sand (Fig. 1; $p < 0.01$, $p < 0.001$) and intermediate on embankment removal. These results were also supported by the negative correlation of gravel cover with vegetation height ($rho = -0.68$, $p < 0.01$) and vegetation cover ($rho = -0.64$, $p < 0.01$), as well as the positive correlation with the cover of bare soil ($rho = 0.74$, $p < 0.01$).

3.2 Species diversity

In total 117 plant species were recorded. The median of the species number per plot after embankment removal was 9 compared to 13 on sand and 10 on gravel. The species number was significantly lower, when the embankment was removed than when sandy sediment was added (Fig. 2; $p < 0.05$), even if the calculation of the evenness showed no significant differences between the three types of measure ($p > 0.05$). Both diversity indicators showed a higher variance of values for the sites restored by embankment removal, while the other two measures were more homogenous.

3.3 Species composition

The NMDS ordination showed clear differences in the species composition of the three types of measure (Fig. 3, Supplement E1) and PERMANOVA revealed that these differences are significant ($F = 3.80$, $p < 0.01$). The NMDS ordination showed a clear distinction between sand and gravel addition plots, where the species composition was mostly homogenous. However, the plots recorded after removal of the embankment largely overlapped with the other two studied riverbank revitalisation measures. Their species composition was more variable.

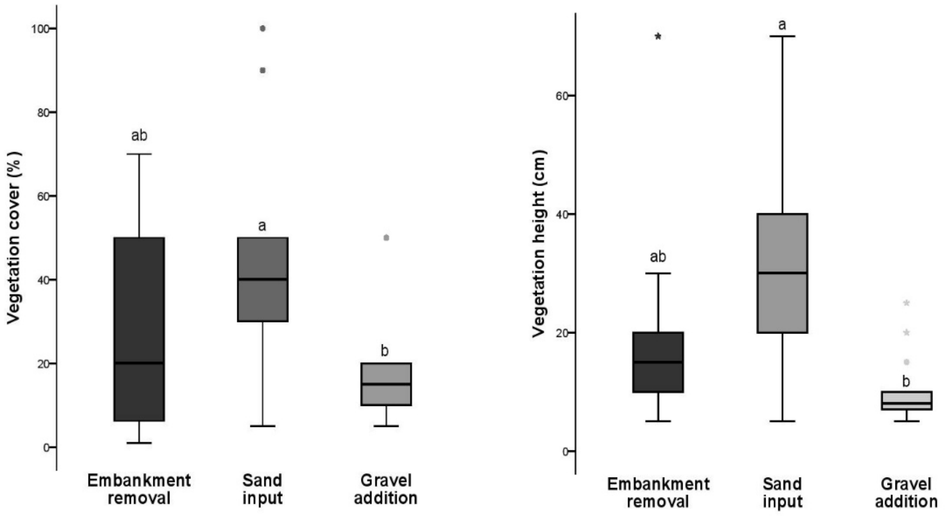

Fig. 1. Comparison of vegetation cover (left) and vegetation height (right) for the three different types of measure: embankment removal (black), sand input (dark grey) and gravel addition (light grey). Gravel cover ranged 70–100% on sites with gravel addition, while it was between 0–10% on sites with sand input and between 0–5% on sites with embankment removal. Different letters indicate significant differences ($p < 0.05$; median; box, 25–75%; whiskers, min–max).

Abb. 1. Vergleich der Vegetationsdeckung (links) und der Vegetationshöhe (rechts) der drei Maßnahmentypen (vlnr): Uferanbruch (schwarz), Sandzugabe (dunkelgrau) und Kieszugabe (hellgrau). Die Kiesdeckung betrug 70–100 % auf Flächen mit Kieszugabe, 0–10 % auf Flächen mit Sandzugabe und 0–5 % nach Uferanbruch. Unterschiedliche Buchstaben verweisen auf signifikante Unterschiede (p < 0,05; Median; Box: 25–75 %; Whisker: Min–Max).

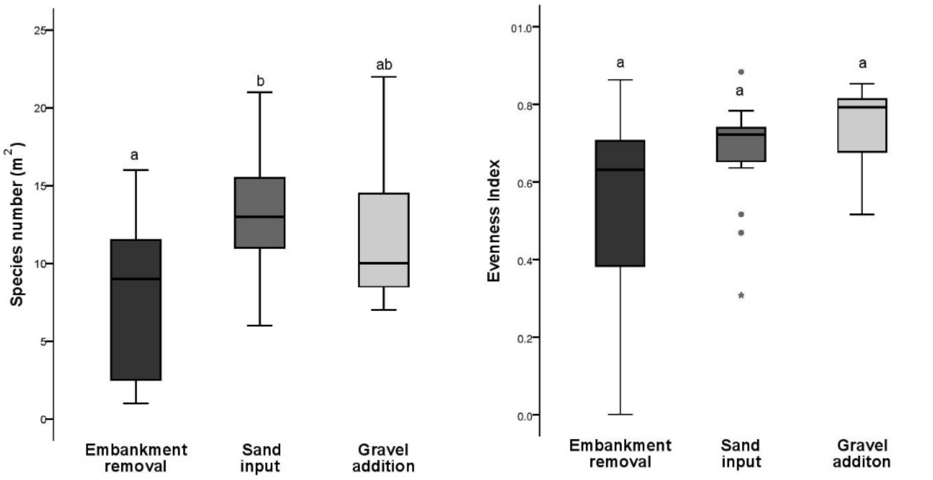

Fig. 2. Comparison of species number (left) and evenness (right) for the three different types of measure: embankment removal (black), sand input (dark grey) and gravel addition (light grey). Different letters indicate significant differences ($p < 0.05$).

Abb. 2. Vergleich der Artenzahl (links) und des Evenness Index (rechts) der Maßnahmentypen Uferanbruch (schwarz), Sandzugabe (dunkelgrau) und Kieszugabe (hellgrau). Unterschiedliche Buchstaben verweisen auf signifikante Unterschiede (p < 0,05).

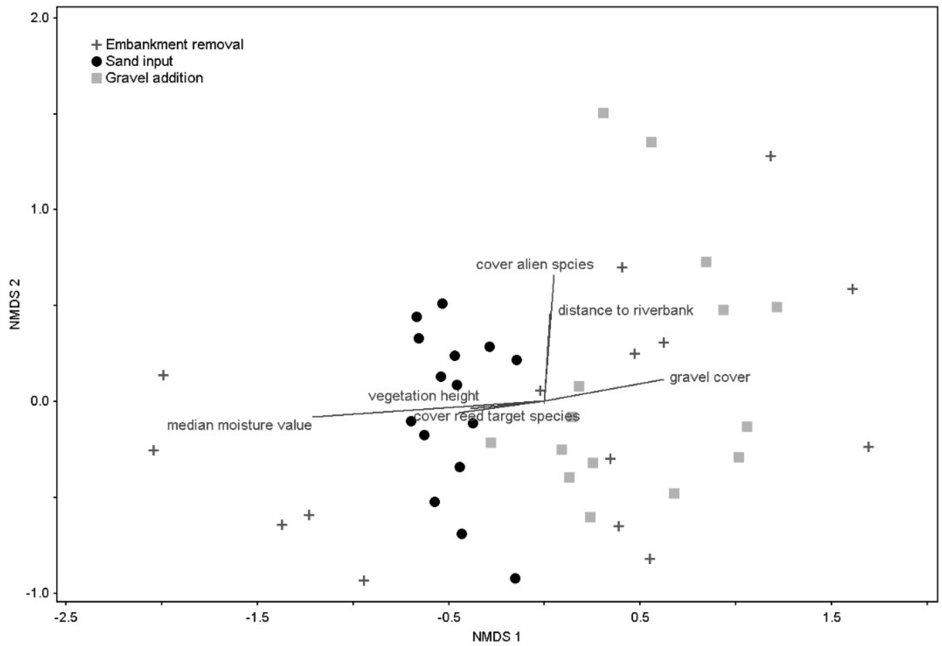

Fig. 3. NMDS ordination based on Bray-Curtis dissimilarity for the comparison of species composition of 15 plots sampled on three sites with embankment removal (grey crosses), 15 plots sampled on three sites with sand input (black circles) and 15 plots sampled on three sites with gravel addition (black quadrats). It is based on the cover of 117 plant species and is visualised as a joint plot with environmental gradients represented as black vector lines (cutoff $r^2 = 0.15$; stress based on two dimensions: 0.18; Monte-Carlo: $p < 0.001$). The vector lines of 'vegetation height' and 'median moisture value' as well as 'distance to riverbank' and 'cover alien species' overlap.

Abb. 3. NMDS Ordination basierend auf Bray-Curtis Unähnlichkeiten. Es wird die Artenzusammensetzung von je 15 Plots und je drei Standorten nach Uferanbruch (dunkelgraue Kreuze) nach Sandzugabe (schwarze Kreise) und nach Kieszugabe (hellgraue Quadrate) verglichen. Die Ordination basiert auf der Deckung von 117 Pflanzenarten und ist als ‚Joint Plot' dargestellt, zusammen mit Umweltgradienten, die hier als schwarze Vektorlinien angezeigt werden (cutoff $r^2 = 0{,}15$). Stresswert basierend auf zwei Dimensionen: 0,18; Monte-Carlo: $p < 0{,}001$. Die Vektorlinien ‚Vegetationshöhe' und ‚Median Feuchtewert' sowie ‚Abstand zum Ufer' und ‚Deckung Neophyten' überschneiden sich.

Besides the variable heterogeneity, the three treatments could be distinguished by different plant species. There was a significantly higher cover of therophyte species like *Arabidopsis arenosa*, *Arenaria serpyllifolia* and *Herniaria glabra* on sites, where gravel was added, compared to sites, where sand was added ($p < 0.05$). In addition, typical riparian plant species like *Clematis vitalba*, *Humulus lupulus* and *Solanum dulcamara* were recorded on sites with gravel addition (Fig. 4). Univariate statistics showed no significant difference in the amount of pioneer target species ($p > 0.05$). On the NMDS graph (Fig. 4) all species classified as target species for pioneer habitats were found in the centre of the graph on plots with gravel and sand input. Sites after restoration by sand input were characterised by the presence of pioneer species like *Alopecurus aequalis*, *Isolepis setacea* and *Juncus inflexus* and those after gravel addition by species like *Carex hirta*, *Juncus articulatus* and *Rumex obtusifolius*.

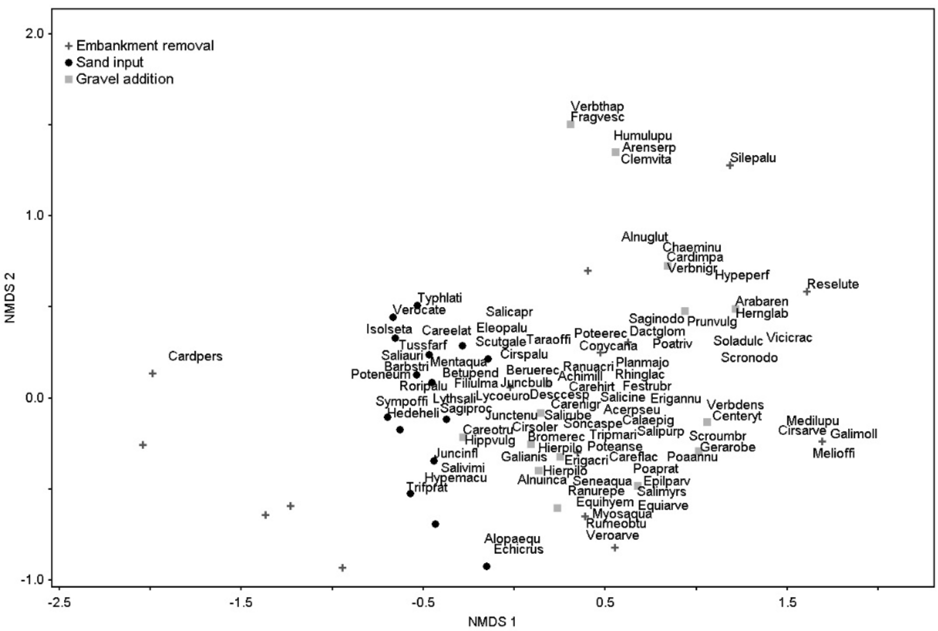

Fig. 4. NMDS ordination based on Bray-Curtis dissimilarity for the comparison of species composition of 15 plots sampled on three sites with embankment removal (grey crosses), 15 plots sampled on three sites with sand input (black circles) and 15 plots sampled on three sites with gravel addition (black quadrats). It is based on the cover of 117 plant species (stress based on two dimensions: 0.18; Monte-Carlo: $p < 0.001$). Frequent species wich an occurence on more than 10 plots are not shown.

Abb. 4. NMDS Ordination basierend auf Bray-Curtis Unähnlichkeiten. Es wird die Artenzusammensetzung von je 15 Plots und je drei Standorten nach Uferanbruch (dunkelgraue Kreuze) nach Sandzugabe (schwarze Kreise) und nach Kieszugabe (hellgraue Quadrate) verglichen. Die Ordination basiert auf der Deckung von 117 Pflanzenarten. Stresswert basierend auf zwei Dimensionen: 0,18; Monte-Carlo: $p < 0{,}001$. Häufige Arten, die auf mehr als 10 Plots vorkommen, werden nicht angezeigt.

Abbreviations: Acerpseu: *Acer pseudoplatanus*; Achimill: *Achillea millefollium* agg.; Alnuglut: *Alnus glutinosa*; Alnuinca: *Alnus incana*; Alopaequ: *Alopecurus aequalis*; Arabaren: *Arabidopsis arenosa*; Arenserp: *Arenaria serpyllifolia*; Barbstri: *Barbarea stricta*; Beruerec: *Berula erecta*; Betupend: *Betula pendula*; Bromerec: *Bromus erectus*; Calaepig: *Calamagrostis epigejos*; Cardimpa: *Cardamine impatiens*; Cardpers: *Carduus personata*; Careelat: *Carex elata*; Careflac: *Carex flacca*; Carehirt: *Carex hirta*; Carenigr: *Carex nigra*; Careotru: *Carex otrubae*; Centeryt: *Centaurium erythraea*; Chaeminu: *Chaenorhinum minus*; Cirsarve: *Cirsium arvense*; Cirsoler: *Cirsium oleraceum*; Cirspalu: *Cirsium palustre*; Clemvita: *Clematis vitalba*; Conycana: *Conyza canadensis*; Dactglom: *Dactylis glomerata*; Desccesp: *Deschampsia cespitosa* agg.*;* Echicrus: *Echinochloa crus-galli*; Eleopalu: *Eleocharis palustris*; Epilparv: *Epilobium parviflorum*; Equiarve: *Equisetum arvense*; Equihyem: *Equisetum hyemale*; Erigacri: *Erigeron acris*; Erigannu: *Erigeron annuus*; Festrubr: *Festuca rubra* agg.; Filiulma: *Filipendula ulmaria*; Fragvesc: *Fragaria vesca*; Galianis: *Galium anisophyllon*; Galimoll: *Galium mollugo agg.*; Gerarobe: *Geranium robertianum*; Hedeheli: *Hedera helix*; Hernglab: *Herniaria glabra*; Hierpilo: *Hieracium pilosella*; Hierpilo: *Hieracium piloselloides*; Hippvulg: *Hippuris vulgaris*; Humulupu: *Humulus lupulus*; Hypemacu: *Hypericum maculatum*; Hypeperf: *Hypericum perforatum*; Isolseta: *Isolepis setacea*; Juncbulb: *Juncus bulbosus*; Juncinfl: *Juncus inflexus;* Junctenu: *Juncus tenuis*; Lycoeuro: *Lycopus europaeus*; Medilupu: *Medicago lupulina*; Melioffi: *Melilotus officinalis*; Mentaqua: *Mentha aquatica*; Myosaqua: *Myosoton aquaticum*; Planmajo: *Plantago major* subsp. *major*; Poaannu: *Poa annua*; Poaprat: *Poa pratensis*; Poatriv: *Poa trivialis*; Poteanse: *Potentilla anserina*; Poteerec: *Potentilla erecta*; Poteneum: *Potentilla neumanniana*; Prunvulg: *Prunella vulgaris*; Ranuacri: *Ranunculus acris*; Ranurepe: *Ranunculus repens*; Reselute: *Reseda lutea*; Rhinglac: *Rhinanthus glacialis*; Roripalu: *Rorippa palustris*; Rumeobtu: *Rumex obtusifolius*; Saginodo: *Sagina nodosa*; Sagiproc: *Sagina procumbens*; Saliauri: *Salix aurita*; Salicapr: *Salix caprea*; Salicine: *Salix cinerea*; Salimyrs: *Salix myrsinifolia*; Salipurp: *Salix purpurea*; Salirube: *Salix rubens*; Salivimi: *Salix viminalis*; Scronodo: *Scrophularia nodosa*; Scroumbr: *Scrophularia umbrosa*; Scutgale: *Scutellaria galericulata*; Seneaqua: *Senecio aquaticus*; Silepalu: *Silene palustris/ latifolia*; Soladulc: *Solanum dulcamara*; Soncaspe: *Sonchus asper*; Sympoffi: *Symphytum officinale*; Taraoffi: *Taraxacum officinale* agg.; Trifprat: *Trifolium pratense*; Tripmari: *Tripleurospermum maritimum* agg.; Tussfarf: *Tussilago farfara*; Typhlati: *Typha latifolia*; Verbdens: *Verbascum densiflorum*; Verbnigr: *Verbascum nigrum*; Verbthap: *Verbascum thapsus*; Veroarve: *Veronica arvensis*; Verocate: *Veronica catenata*; Vicicrac: *Vicia cracca.*

Plots with sand addition showed a significant higher median of moisture value than plots with gravel input, revealing more moisture indicating species ($p < 0.001$). This result is supported by the clear gradient in direction of 'sand plots' in the ordination graph (Fig. 3). Furthermore, the amount of target species for reed stands was significantly higher on these sites ($p < 0.001$) in comparison to the other measures. The NMDS ordination showed that those are species like *Carex elata*, *Lythrum salicaria* and *Phragmites australis*. None of the studied species groups could be exclusively attributed to the heterogeneous plots after embankment removal. Thus, they were characterized by a significantly lower amount of phanerophytes ($p < 0.05$).

There were no significant differences in univariate statistics in regard to the amount of alien species ($p > 0.05$). Yet, the NMDS (Fig. 3) showed a distinct gradient of the cover of alien species in the direction of two 'gravel plots' outlying the cluster. These two plots had a comparatively higher cover (10 and 15%) of *Impatiens glandulifera* in proportion to their main vascular plant cover (10 and 20% total cover).

3.4 Effect of restored riverbanks on adjacent reed stands

In order to detect the effects of structural bank diversification measures, 25 plots close to restored riverbanks were compared to 25 plots in monotonous reed stands of regulated areas. Although no significant differences in structural parameters could be found, there were significant differences in species diversity. The species number per plot in restored areas had a median of seven, which is significantly higher than in non-restored areas (four spp.; $p < 0.001$). The evenness also revealed a significant higher diversity in restored areas ($p < 0.001$).

In addition to differences in the species diversity, species composition varied significantly between impact and control sites (PERMANOVA: $F = 5.98$, $p < 0.001$). The restored areas were characterised by a significantly higher number of endangered species ($p < 0.01$) and target species for reed stands ($p < 0.01$) like *Carex elata, Equisetum palustre*, *Poa palustris* and *Valeriana officinalis* (Fig. 5). Although alien species (*Conyza canadensis, Erigeron annuus, Impatiens glandulifera*) were found on both restored and unrestored sites, the cover of alien plants was significantly higher on the control site ($p < 0.05$).

4. Discussion

4.1 Structural diversity

Our results showed that the removal of embankments and the addition of gravel are suitable measures to create open, sparsely vegetated areas. Our data revealed a lower vegetation cover and height on these sites compared to those with addition of sand. It can be supposed that these bare soil zones cannot be sustained on sites with sand input in absence of regular natural disturbance or management. A high cover of reed species on these sites indicates fast succession. This can have two explanations: Firstly, the sand used for creating the artificial bank originated from the floodplain, so it can be suggested that it is nutrient-rich (FRIESE et al. 2000) inducing high productivity. Secondly, the sand contains seeds that get activated during disturbance (cf. HOELZEL & OTTE 2001); these processes lead to dense and high vegetation cover, which is not suitable for riparian pioneer species.

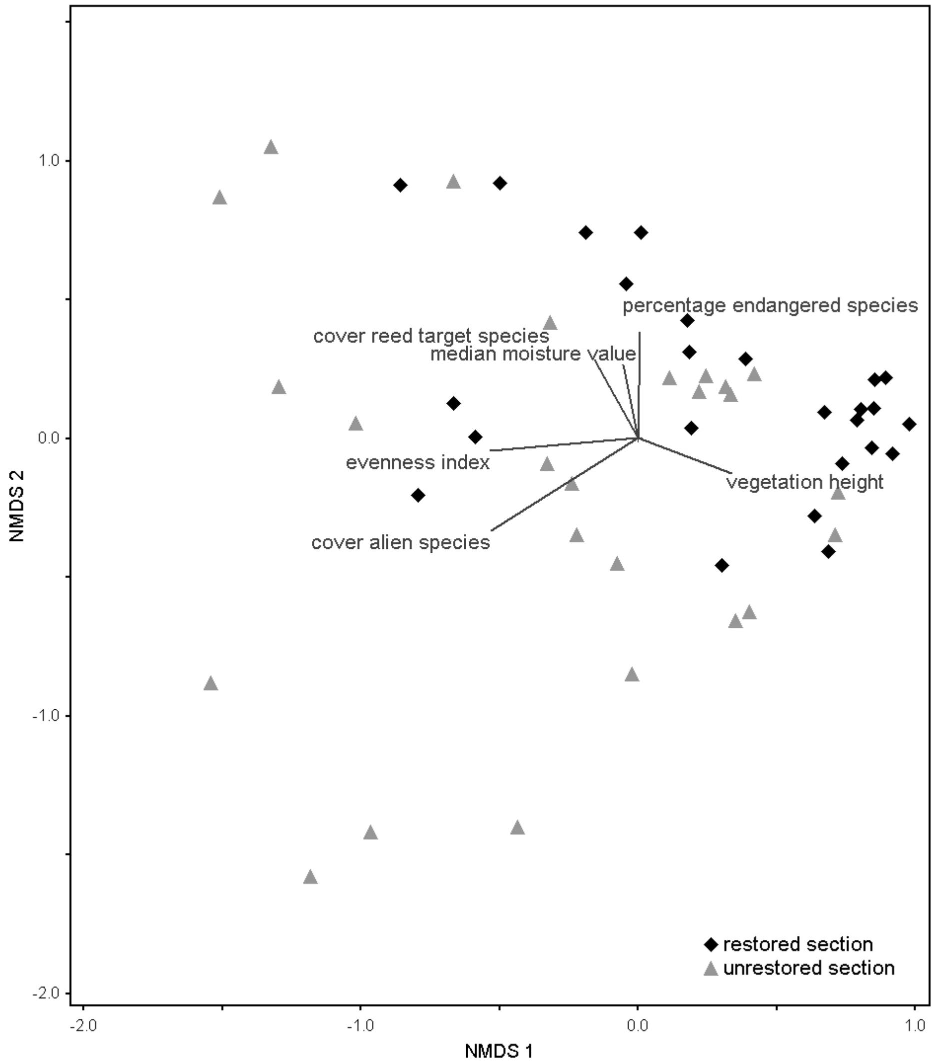

Fig. 5. NMDS ordination based on Bray-Curtis dissimilarity for the comparison of species composition of 25 plots sampled on five sites close to restored sites (black diamonds) and 25 plots sampled on five non-restored control sites (grey triangles). It is based on the cover of 55 plant species and is visualised as a joint plot with environmental gradients represented as red vector lines (cutoff $r^2 = 0.15$; stress based on three dimensions: 0.10; Monte-Carlo: $p < 0.05$).

Abb. 5. NMDS Ordination basierend auf Bray-Curtis Unnähnlichkeiten. Es wird die Artenzusammensetzung von 25 Plots von fünf Standorten im direkten Umgriff der Renaturierungsmaßnahmen (schwarze Rauten) verglichen mit 25 Kontrollflächen von fünf Standorten nicht-renaturierter Flussabschnitte (graue Dreiecke). Die Ordination basiert auf der Deckung von 55 Pflanzenarten und wird als ‚Joint Plot' dargestellt, zusammen mit Umweltgradienten, die hier als rote Vektorlinien angezeigt werden (cutoff $r^2 = 0{,}15$; Stresswert basierend auf drei Dimensionen: 0,10; Monte-Carlo: $p < 0{,}05$).

Two years after restoration, the highly productive and dominant species can only hardly settle under the circumstances created by gravel addition or embankment removal. The removal of embankment structures leads to the development of instable banks with a dynamic bare soil surface. It can be assumed that in general plant establishment is difficult due to these sediment dynamics (WARD et al. 1999). This is shown by the significantly lower amount of phanerophytes on these sites, for example. Though, even if in terms of habitat creation these sites provide open spaces, it appears to be less suited for the development of riparian vegetation.

Gravel introduction represents an intermediate situation with moderate species establishment leading to environmental conditions suitable for early-successional species, i.e. nutrient-poor conditions, occasional inundation and frequent drought (VON HEßBERG 2003, ELLENBERG & LEUSCHNER 2010). Thus, it increases the structural diversification in prealpine rivers by the establishment of another more open habitat type. Since gravel bars have largely disappeared in the past decades, structural restoration measures should focus on their reestablishment.

While considerable achievements could be obtained directly on measure sites for the open structure, none of the studied measures revealed significant structural changes in the adjacent reed zones. Nonetheless, this was not expected, because reed zones are naturally characterised by a high and homogenous vegetation cover and height (ELLENBERG & LEUSCHNER 2010). This implicates, that structural parameters are not a suitable indicator for the monitoring of restoration impacts in these areas.

4.2 Species diversity

The results of our study indicated that sand and gravel input are more suitable measures for an increase of species diversity than embankment removal. These results are following the conclusions of JÄHNIG et al. (2009), who found out that gravel bars and loamy habitats greatly increased species richness. Plots with embankment removal differ greatly in terms of species diversity compared to the two other treatments. The reason for this might be the difficult abiotic conditions only few species can deal with.

Concerning the regional diversity, ROSENZWEIG (1995) remarked a positive correlation between the biotic diversity and habitat heterogeneity. As our results revealed very different species compositions depending on the measures, we suggest that each measure leads to another habitat type. In this case, implementing different revitalisation measures can increase β-diversity. These findings follow the statements of WARD et al. (1999), who noticed that regional diversity not only depends on the number of species per habitat, but also on the number of habitats and the turnover between habitats. Compared to gravel input and embankment removal the species composition of sand input is rather homogenous. Therefore, if the creation of different habitats is defined as a goal and only one type of measure is implemented, the input of sand cannot be recommended. To sum up, sand and gravel input are most suitable for the enhancement of local species diversity, whereas the embankment removal is more likely to increase the habitat diversity due to more heterogeneous species compositions on these sites.

Furthermore, the results of this study showed that the implementation of structural diversification measures enhances the species diversity of direct measure sites, but also the one of the adjacent reed zones. Like JANUSCHKE et al. (2011) we recorded an increased α-diversity in reed zones close to restoration sites in comparison to control sites. This could indicate positive restoration effects by an easier colonisation due to a shorter distance to a diverse species pool in restored areas (KAREIVA 1990). Nonetheless, this assumption cannot be exclusively confirmed. Sites for revitalisation measures may have been chosen according to an existing good conservation potential, whereas the control sites are situated upstream to the revitalisation measures outside those better-preserved areas. Furthermore, the sites could have been affected by disturbances during the implementation works and therefore support a higher diversity. Intermediate disturbance events lead to a temporarily increase in species diversity, because they create open spaces where other species can colonise, corresponding to the well-known Intermediate Disturbance Hypothesis (CONNELL 1978).

4.3 Species composition

In our study, species composition greatly differs between all treatments as a result of a successful creation of different habitat conditions. Moreover, our results showed that different types of measures promote different species. The introduction of gravel leads to the development of riparian pioneer species (therophytes and pioneer target species) shortly after restoration. These results support the findings of JÄHNIG et al. (2009), who also described that short-living taxa on gravel bars benefit from floodplain restoration. In floodplains, where natural sediment deposition patterns are missing, gravel addition is an efficient means to create open spaces that are suitable for many riparian species (RICHARDSON et al. 2007)

Nonetheless, the vegetation composition after gravel input can vary considerably. In our results two plots differed greatly from the others due to a particularly high cover of *Impatiens glandulifera*. This may be problematic as invasive species can affect native species communities if they reach high abundances (RICHARDSON et al. 2007), though HEJDA & PYŠEK (2006) argued that *I. glandulifera* does not considerably affect native species; it merely influences the proportional cover of other dominant species like *Urtica dioica.* According to this assumption, the occurrence of *I. glandulifera* could be neglected and thus rather be seen as one component of a partially new system (RICHARDSON et al. 2007).

Compared to the gravel introduction and embankment removal the input of sand was more homogenous. Over all studied plots, the species composition was quite similar with few outliers. In comparison with embankment removal, the level of disturbance was lower due to more gentle slopes on these plots. Moisture indicating plants as well as pioneer species cover was increased on these plots. HOELZEL & OTTE (2001) suggested that riparian species increase after seed bank mobilisation due to sediment relocation. Nonetheless, as we also recorded a high number of phanerophytes and reed species, the succession of these sites cannot be predicted two years after restoration. In the absence of sediment relocating floods, some of these sites will develop into riparian forest or into reed stands. The fine sediment with a high capacity of water retention provides good conditions for germination of riparian tree species (KARRENBERG et al. 2002), at least when the abundance of competitive herbs is low. Otherwise, the sites are more likely to support reed stands in the near future, especially when competitive reed species are found in the direct surroundings. The measure of embankment removal leads to a highly variable species composition. It increases dynamics, which causes a high variability of habitats and species. Nonetheless, our results also indicate that this measure is not suitable, if quick establishment of riparian pioneer species or the development of riparian forest is aimed.

The species composition in the surroundings of restored river sections is positively affected by revitalisation measures. Our results show a higher conservation value in these areas, which is indicated by a higher amount of endangered species and target species for reed stands on these plots. This observation can be due to a different species pool in the seed bank or the immediate surroundings of restored and degraded river sections (HOELZEL & OTTE (2001). It could also be due to the reflection of methodological problems in the sampling design. Indeed, plots may differ in other factors than restoration impact when using a time for space substitution design. Indeed, HEJDA & PYŠEK (2006) discussed similar patterns with the same kind of spatial design.

Finally, one must note, that our results represent only are a 'snap shot' of the restoration state two years after measure implementation. After some years, the conservation value of sites in the surrounding of restored river sections may diminish again. Sites after embankment removal might flatten, facilitating species establishment on less dynamic habitats. Succession towards dominant reed stands, indicated by the presence of *Phalaris arundinacea* and *Phragmites australis,* may also be observed on the now more open sites after gravel input and embankment removal, as natural disturbances are missing in highly modified rivers like the Inn. Though, fluvial dynamics are crucial for the maintenance of different successional stages enhancing the overall species diversity (WARD et al. 1999). Therefore, JANUSCHKE (2014) argues that a monitoring should not only evaluate, whether the desired habitat types and species are initially present, but also verify, if they are maintained after a longer time period. If then undesired vegetation change is observed, adaptive restoration measures or management have to be considered.

4.4 Perspectives and implications for management

The three studied types of revitalisation measures for riverbank diversification greatly differ in their development since implementation. For future projects, this means that different measures should be accomplished according to the predefined goals. Gravel introduction leads to open habitat structures where riparian pioneer species can establish. Due to high nutrient levels and suitable moisture conditions the input of sand causes a comparatively fast succession into either desirable riparian forest or monotonous reed stands. The removal of embankment leads to the creation of highly dynamic habitats. This may be difficult for plants, but can benefit reptiles, ground beetles or kingfishers.

In our study we showed that rare species, pioneer target species and other desirable riparian plants increase with the creation of open soil areas, which nowadays are also the most threatened habitats in floodplains (VON HEßBERG 2003). Therefore, management efforts need to guarantee the conservation of these early successional sites, as there is a lack of dynamics in highly modified rivers. Our results show that succession is particularly fast on fine sediments, whereas restored gravel bars or reopened embankment are slower in succession and can therefore be considered more sustainable. Possible management strategies to counteract succession are soil-disturbance and biomass reduction, initiated for instance by grazing (SCHAICH et al. 2010). As outcomes vary considerably depending on the type of measure, our results emphasise how important careful planning, evaluation of site-specific conditions and post-implementation monitoring and management can be.

Erweiterte deutsche Zusammenfassung

Einleitung – Fließgewässer und ihre Auen gehören zu den artenreichsten Ökosystemen der Welt (WARD et al. 1999). Mit der Intensivierung der anthropogenen Nutzung in Flussauen wurden Flüsse zunehmend verbaut und ihre Gewässerstruktur sowie die natürliche Fließgewässerdynamik beeinträchtigt (HABERSACK & PIÉGAY 2008). Dabei sind das Auftreten regelmäßiger Störungen und eine intakte laterale Konnektivität notwendig für den Artenreichtum der Aue (WARD et al. 1999). Besonders in den stark verbauten Voralpenflüssen wird die Revitalisierung im Sinne einer Strukturverbesserung zur Dynamisierung der Auen zu einer dringenden Notwendigkeit. Eine Studie von BERNHARDT et al. (2005) belegt eine starke Zunahme von Renaturierungsprojekten an Flüssen. Standardisierte, langjährige Monitoringprogramme sind dabei bisher selten (CHAPMAN & UNDERWOOD 2000), aber unbedingt nötig, um den Erfolg der Maßnahmen bewerten und zukünftige Projekte zielgerichteter durchführen zu können. Ziel dieser Arbeit ist ein systematischer Vergleich unterschiedlicher Uferstrukturierungsmaßnahmen und die Untersuchung des Einflusses dieser Maßnahmen auf die Vegetation. Die abschließende Bewertung erfolgte anhand von drei in Renaturierungsprojekten häufig verfolgten Zielen: (1) Verbesserung der Vegetationsstruktur, (2) Erhöhung der Artenvielfalt und (3) charakteristische Artenzusammensetzung.

Material und Methoden – Das Untersuchungsgebiet liegt an einem stark anthropogen überprägten Fluss des bayerischen Voralpenlandes, dem Inn südlich und nördlich von Wasserburg. Dort wurden mit dem Ziel der Dynamisierung des Flusses die Ufer umstrukturiert. Die durchgeführten Maßnahmen lassen sich in drei Gruppen einteilen: Uferanbruch (Entsteinung, Aufbrechen), Kies- und Sandzugabe. In einem stratifiziert-randomisierten Aufnahmedesign wurden gezielt die Vegetationsstruktur, die Deckung der Arten und ausgewählte abiotische Größen (pH-Wert, Uferentfernung, Uferneigung) erhoben. Als Straten dienten die vorgefundenen Strukturtypen ‚Röhricht' und ‚Pionierfluren'. Die Datenauswertung erfolgte anhand uni- und multivariater Analysen.

Ergebnisse – Alle drei Maßnahmentypen unterschieden sich bezüglich der Vegetationsstruktur. Die Rohbodendeckung war maximal nach Revitalisierung durch Uferanbruch. Die Vegetationshöhe und -deckung am Flussufer waren am höchsten nach Zugabe von Sand. Die Probeflächen mit Uferanbruch wiesen eine starke Streuung der Artenzahl auf, und auch die Diversität der angrenzenden Röhrichte war erhöht. Alle drei Maßnahmen unterscheiden sich deutlich in ihrer Artenzusammensetzung, wobei erneut die Maßnahme des Uferanbruchs eine sehr heterogene Entwicklung bewirkte. An Renaturierungsflächen angrenzende Röhrichte zeigten eine veränderte Artenzusammensetzung im Vergleich mit Kontrollflächen.

Diskussion – Zwei Jahre nach der Durchführung führten die drei untersuchten Maßnahmen zu großen Unterschieden in der Vegetationsentwicklung. Durch die Entfernung der Uferbefestigung entstehen dynamische Habitate, auf denen sich nur wenige Pflanzen etablieren. Wie auch in der Studie von JÄHNIG et al. (2009) führt die Kieszugabe zu einer vergleichsweise offenen Vegetationsstruktur, in der sich auentypische Pionierarten ansiedeln. Auf Sandflächen stellt sich eine homogene Artenzusammensetzung ein; diese lässt sich zumindest teilweise durch die hohe Produktivität der Standorte erklären, die sich vermutlich entweder in Richtung Röhricht (FRIESE et al. 2000) oder Auwald (KARRENBERG et al. 2002) entwickeln werden, jedenfalls wenn Störungsereignisse ausbleiben. Die beobachteten Entwicklungen zeigen die kurzfristige Reaktion der Auepflanzen auf die Uferstrukturierungsmaßnahmen. Für eine abschließende Beurteilung der Wirksamkeit der Maßnahmen ist jedoch ein längerfristiges Monitoring nötig (JANUSCHKE 2014). Wenn die natürliche Dynamik des Flusses nicht ausreicht, müsste über geeignete Managementstrategien zur Offenhaltung von Pionierstandorten nachgedacht werden.

Perspektiven für die Renaturierung – Die Ergebnisse zeigen, wie unterschiedlich die Wirkungen einzelner Renaturierungsmaßnahmen sein können. Dies unterstreicht, wie wichtig es ist, vor den Maßnahmen abzuwägen, welche Renaturierungsziele unter den gegebenen lokalen Bedingungen verfolgt werden sollen. Nach Maßnahmendurchführung ist ein Langzeitmonitoring wünschenswert.

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Supplements

Additional supporting information may be found in the online version of this article.

Zusätzliche unterstützende Information ist in der Online-Version dieses Artikels zu finden.

Supplement E1. List of species and frequency of occurrence on all surveyed plots.

Anhang E1. Artenliste und Frequenz des Vorkommens auf allen aufgenommenen Flächen.

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Supplement E1. List of species and frequency of occurrence on all surveyed plots.
Anhang E1. Artenliste und Frequenz des Vorkommens auf allen aufgenommenen Flächen.

Scientific name	German name	Total	Gravel addition	Sand input	Embankment removal	Reed - restored	Reed - control
Acer pseudoplatanus L.	Berg-Ahorn	3	2	0	1	0	0
Achillea millefollium agg.	Gewöhnliche Schafgarbe	1	1	0	0	0	0
Alnus glutinosa (L.) Gaertn.	Schwarz-Erle	3	2	0	1	0	0
Alnus incana (L.) Moench	Grau-Erle	7	1	2	0	4	0
Alopecurus aequalis Sobol.	Rotgelbes Fuchsschwanzgras	2	0	2	0	0	0
Alopecurus pratensis L.	Wiesen-Fuchsschwanz	1	0	0	0	0	1
Arabidopsis arenosa (L.) Lawairée	Sand-Schmalwand	1	1	0	0	0	0
Arenaria serpyllifolia L.	Quendel-Sandkraut	1	1	0	0	0	0
Barbarea stricta Andrz.	Steifes Barbarakraut	1	0	1	0	0	0
Berula erecta (Huds.) Coville	Schmallblättriger Merk	1	1	0	0	0	0
Betula pendula Roth	Hänge-Birke	6	1	4	0	1	0
Bromus erectus Huds.	Aufrechte Trespe	1	1	0	0	0	0
Calamagrostis pseudophragmites	Ufer-Reitgras	18	4	4	4	5	1
Calystegia sepium agg.	Gewöhnliche Zaunwinde	3	0	0	0	1	2
Cardamine impatiens L.	Spring-Schaumkraut	1	1	0	0	0	0
Carduus personata (L.) Jacq.	Ketten-Distel	6	1	0	1	1	3
Carex acutiformis Erh.	Sumpf-Segge	2	0	0	0	2	0
Carex elata All.	Steife Segge	9	1	6	0	2	0
Carex flacca Schreb	Blaugrüne Segge	5	4	1	0	0	0
Carex hirta L.	Behaarte Segge	3	2	0	1	0	0
Carex nigra (L.) Reichard	Braun-Segge	2	2	0	0	0	0
Carex otrubae Podp.	Hain-Segge	1	1	0	0	0	0
Carex pseudocyperus L.	Scheinzypergras-Segge	33	2	5	2	16	8
Carlina vulgaris agg.	Kratz Distel	1	0	0	0	1	0
Centaurium erythraea Rafn	Echtes Tausendgüldenkraut	1	1	0	0	0	0
Chaenorhinum minus (L.) Lange	Kleines Leinkraut	2	2	0	0	0	0
Circaea lutetiana L.	Großes Hexenkraut	3	0	0	0	3	0
Cirsium arvense (L.) Scop.	Acker-Kratzdistel	8	0	2	2	4	0
Cirsium oleraceum (L.) Scop.	Kohl-Kratzdistel	6	0	2	2	2	0
Cirsium palustre (L.) Scop.	Sumpf-Kratzdistel	3	0	2	0	1	0
Clematis vitalba L.	Gewöhnliche Waldrebe	1	1	0	0	0	0
Conyza canadensis L.	Kanadisches Berufkraut	9	4	2	2	1	0
Corylus avellana L.	Gewöhnliche Hasel	1	0	0	0	1	0
Dactylis glomerata L.	Gewöhnliches Knauelgras	1	0	0	1	0	0
Deschampsia cespitosa agg.	Rasen-Schmiele	5	2	1	1	1	0
Echinochloa crus-galli (L.) P. Beauv.	Gewöhnliche Hühnerhirse	2	1	1	0	0	0
Eleocharis palustris (L.) Roem & Schult.	Echte Sumpfsimse	1	0	1	0	0	0
Epilobium hirsutum (L.)	Behaartes Weidenröschen	15	0	7	0	8	0
Epilobium parviflorum Schreb.	Kleinblütiges Weidenröschen	2	1	0	0	1	0
Epilobium tetragonum L.	Vierkantige Weidenröschen	16	2	9	0	5	0
Equisetum arvense L.	Acker-Schachtelhalm	9	1	0	2	5	1
Equisetum hyemale L.	Winter-Schachtelhalm	5	0	1	1	1	2
Equisetum palustre L.	Sumpf-Schachtelhalm	22	3	10	3	6	0
Equisetum telmateia Ehrh.	Riesen-Schachtelhalm	1	0	0	0	0	1
Equisetum variegatum Schlech. Ex. Weber & D. Mohr	Bunter Schachtelhalm	2	0	0	0	0	2
Erigeron acris L.	Scharfes Berufskraut	2	1	0	1	0	0
Erigeron annuus (L.) Pers.	Feinstrahl-Berufkraut	9	4	0	3	2	0
Festuca pratensis Huds.	Wiesen-Schwingel	1	0	0	0	1	0
Festuca rubra agg.	Gewöhnlicher Rot-Schwingel	3	1	0	0	1	1
Filipendula ulmaria (L.) Maxim.	Echtes Mädesüß	3	2	1	0	0	0
Fragaria vesca L.	Wald-Erdbeere	1	1	0	0	0	0
Galium anisophyllon Vill.	Ungleichblättriges Labkraut	2	2	0	0	0	0
Galium aparine L.	Kletten-Labkraut	8	0	0	0	3	5
Galium mollugo agg.	Wiesen-Labkraut	6	0	0	1	5	0
Geranium robertianum L.	Stinkender Storchschnabel	1	1	0	0	0	0
Hedera helix L.	Gewöhnlicher Efeu	1	0	1	0	0	0
Herniaria glabra L.	Kahles Bruchkraut	1	1	0	0	0	0
Hieracium pilosella L.	Kleines Habitchtskraut	6	4	0	2	0	0
Hieracium piloselloides Vill.	Florentiner Habichtskraut	1	1	0	0	0	0
Hippuris vulgaris L.	Gewöhlicher Tannenwedel	1	1	0	0	0	0
Humulus lupulus L.	Gewöhnlicher Hopfen	2	1	0	0	1	0
Hypericum maculatum Crantz.	Geflecktes Johanniskraut	4	0	2	1	1	0
Hypericum perforatum L.	Echtes Johanniskraut	2	2	0	0	0	0
Hypericum tetrapterum Fr.	Geflügeltes Johanniskraut	1	0	0	0	1	0
Impatiens glandulifera Royle	Drüsiges Springkraut	48	2	4	3	17	22
Impatiens noli-tangere L.	Großes Springkraut	1	0	0	0	1	0
Iris pseudacorus L.	Wasser-Schwerlilie	1	0	0	0	0	1
Isolepis setacea (L.) R. Br.	Borstige Schuppensimse	2	0	2	0	0	0
Juncus articulatus L.	Glieder-Binse	27	9	11	5	1	1
Juncus bulbosus L.	Zwiebel-Binse	1	0	0	1	0	0
Juncus effusus L.	Flatter-Binse	31	9	15	2	4	1
Juncus inflexus L.	Blaugrüne Binse	1	0	1	0	0	0
Juncus tenuis Willd	Zarte Binse	5	1	2	2	0	0
Lathyrus pratensis L.	Wiesen-Platterbse	1	0	0	0	1	0
Lycopus europaeus L.	Ufer-Wolfstrapp	5	4	1	0	0	0
Lysimachia vulgaris L.	Gewöhnlicher Gilbweiderich	2	0	0	0	2	0
Lythrum salicaria L.	Gewöhnlicher Blutweiderich	13	1	8	1	2	1
Medicago lupulina L.	Hopfenklee	3	1	0	2	0	0
Melilotus officinalis (L.) Lam.	Gelber Steinklee	2	0	0	2	0	0
Mentha aquatica L.	Wasser-Minze	3	2	1	0	0	0
Myosoton aquaticum (L.) Moench.	Wasserdarm	1	0	0	1	0	0
Phalaris arundinacea L.	Rohr-Glanzgras	59	4	9	6	21	19
Phragmites australis (Cav.) Trin. Ex Steud.	Gewöhnliches Schilf	68	1	12	5	25	25
Plantago major subsp. Major L.	Breit-Wegerich	6	3	1	0	2	0
Poa annua L.	Einjähriges Rispengras	7	4	1	2	0	0
Poa palustris L.	Sumpf-Rispengras	14	3	0	1	9	1
Poa pratensis L.	Wiesen-Rispengras	2	2	0	0	0	0
Poa trivialis L. s. l.	Gewöhnliches Rispengras	2	1	0	1	0	0
Populus nigra L.	Schwarz-Pappel	15	5	4	6	0	0
Potentilla anserina L.	Gänse-Fingerkraut	1	0	0	1	0	0
Potentilla erecta (L.) Raeusch.	Blutwurz	1	0	0	1	0	0
Potentilla neumanniana Rchb.	Niedriges Fingerkraut	1	0	1	0	0	0
Prunella vulgaris L.	Gewöhnliche Braunelle	2	1	0	1	0	0
Ranunculus acris L.	Scharfer Hahnenfuß	1	1	0	0	0	0
Ranunculus repens L.	Kriechender Hahnenfuß	3	0	1	2	0	0
Reseda lutea L.	Gelber Wau	1	0	0	1	0	0
Rhinanthus glacialis Personnat	Begrannter Klappertopf	1	1	0	0	0	0
Rorippa palustris (L.) Besser	Gewöhnliche Sumpfkresse	2	1	1	0	0	0
Rubus caesius L.	Kratzbeere	19	6	0	6	5	2
Rumex obtusifolius L.	Sumpfblättriger Ampfer	1	0	0	1	0	0
Sagina nodosa L. Fenzl	Knotiges Mastkraut	1	0	0	1	0	0
Sagina procumbens L.	Niederliegendes Mastkraut	2	0	2	0	0	0
Salix alba L.	Silber-Weide	34	12	15	7	0	0
Salix aurita L.	Ohr-Weide	8	0	5	0	3	0
Salix caprea L.	Sal-Weide	4	1	2	1	0	0
Salix cinerea L. s. l.	Grau-Weide	3	2	0	1	0	0
Salix myrsinifolia Salisb.	Schwarz-Weide	5	4	0	1	0	0
Salix purpurea L.	Purpur-Weide	5	4	0	1	0	0
Salix rubens Schrank	Hohe Weide	9	1	4	4	0	0
Salix viminalis L.	Korb-Weide	6	1	5	0	0	0
Scrophularia nodosa L.	Knoten-Braunwurz	4	2	0	0	2	0
Scrophularia umbrosa Dumort.	Geflügelte Braunwurz	7	1	2	0	3	1
Scutellaria galericulata L.	Sumpf-Helmkraut	6	0	1	0	5	0
Senecio aquaticus Hill s. str.	Wasser-Greiskraut	1	1	0	0	0	0
Silene palustris/ latifolia Mill.	Weiße Lichtnelke	1	0	0	1	0	0
Solanum dulcamara L.	Bittersüßer Nachtschatten	4	2	1	1	0	0
Sonchus asper (L.) Hill	Raue Gänsedistel	7	4	2	1	0	0
Symphytum officinale L. s. str.	Gewöhnlicher Beinwell	3	0	1	0	2	0
Taraxacum officinale agg.	Löwenzahn	5	1	3	0	1	0
Trifolium pratense L.	Rot-Klee	1	0	1	0	0	0
Tripleurospermum maritimum agg.	Geruchlose Kamille	1	0	0	1	0	0
Tussilago farfara L.	Huflattich	5	1	4	0	0	0
Typha latifolia L.	Breitblättriger Rohrkolben	1	0	1	0	0	0
Urtica dioica L	Große Brennessel	34	3	1	0	12	18
Valeriana officinalis L	Arznei-Baldrian	1	0	0	0	1	0
Verbascum densiflorum Bertol.	Großblütige Königskerze	2	1	0	0	1	0
Verbascum nigrum L.	Schwarze Königskerze	1	1	0	0	0	0
Verbascum thapsus L.	Kleinblütige Königskerze	1	1	0	0	0	0
Veronica arvensis L.	Feld-Ehrenpreis	1	0	0	1	0	0
Veronica beccabunga L.	Bach-Ehrenpreis	13	5	5	3	0	0
Veronica catenata Pennell	Roter Waser-Ehrenpreis	1	0	1	0	0	0
Vicia cracca L.	Vogel-Wicke	2	1	0	1	0	0