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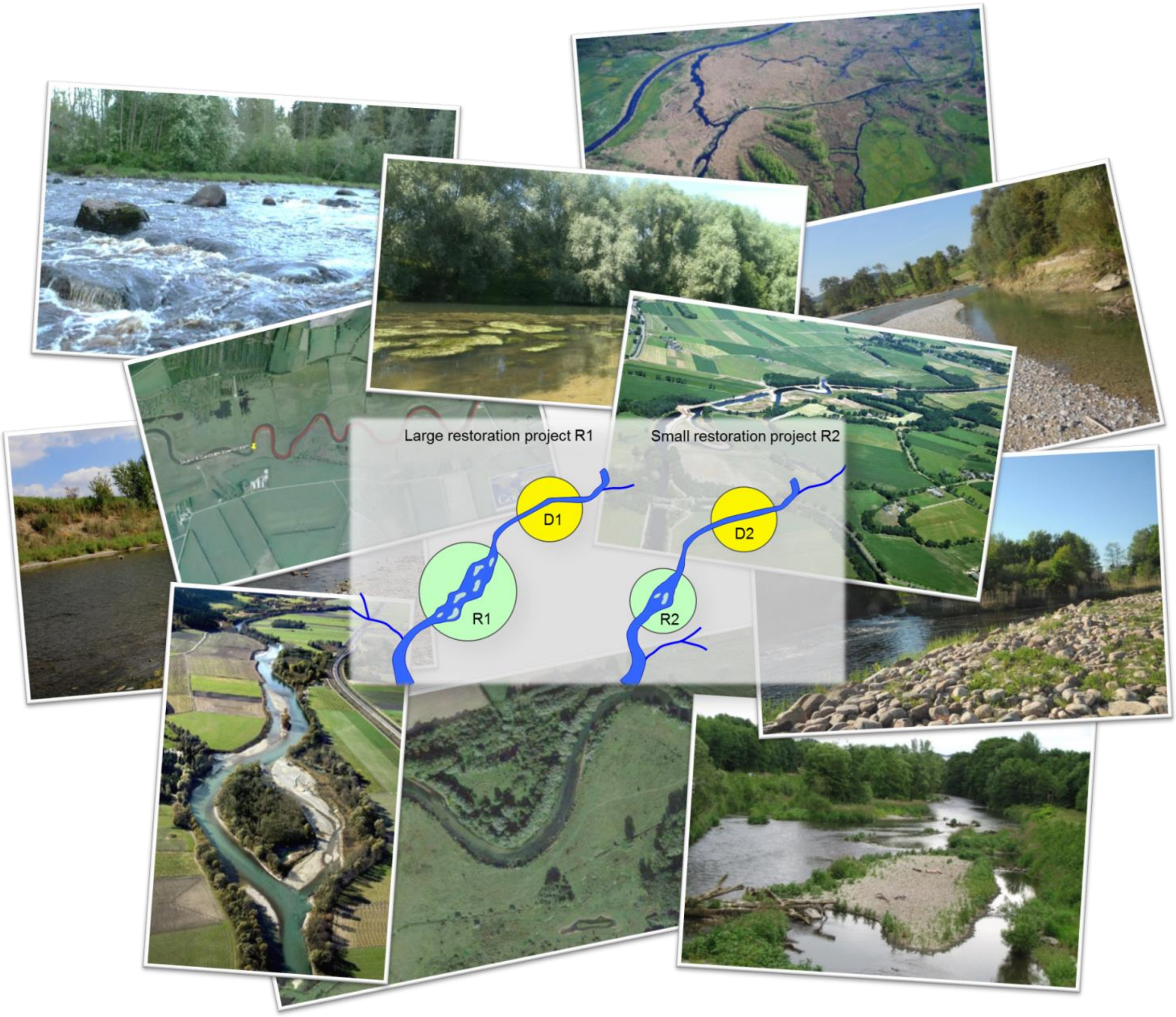

Deliverable	D4.3 Results of the hydromorphological and ecological survey
Title	Effects of large- and small-scale river restoration on hydromorphology and ecology
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Summary

An increasing number of river sections have been restored in the past few decades but only a small number of these projects have been monitored. The few monitoring studies mainly investigated single organism groups, reported contrasting results, and rarely did investigate the influence of catchment, river or project characteristics. In this study, we compiled a harmonized dataset on the effects of hydromorphological river restoration measures on biota based on a standardized monitoring design to minimize scatter due to methodological differences. A broad range of response variables was recorded to draw conclusions on the effect of restoration on biota in general, including habitat composition in the river and its floodplain, three aquatic and two floodplain-inhabiting organism groups, as well as food web composition and aquatic land interactions as reflected by stable isotopes. Additional data on factors potentially constraining or enhancing the effect of restoration were compiled to identify conditions which favour restoration success. The main focus was dedicated to investigate the effect of restoration extent (as indicated by restored section length and restoration intensity).

Ten pairs of one large and a similar but small restoration project were investigated to address the role of restoration extent for river restoration effects. The restoration effect was quantified by comparing each of the 20 restored river sections to a nearby non-restored, i.e. still degraded section. The large restoration projects were representing good-practice examples in different European regions either targeting medium-sized lowland rivers or medium-sized mountain rivers. Many of the mountain rivers investigated were restored by removing bed and bank fixation, flattening river banks, and partly widening the cross-section (referred to as widening in the following). In the lowland rivers, remeandering and reconnecting oxbows were the most prominent measures besides increasing groundwater levels for restoring wetlands. Moreover, instream measures like large wood and boulder placement have been applied.

We found a significant effect on the number of ground beetle species and on richness and diversity of macrophytes, a moderate effect on fish, and a low effect on macroinvertebrates and floodplain vegetation. This is consistent with the findings of other studies on single organism groups, except for floodplain vegetation, which usually benefits from restoration but restoration effects were constrained by agricultural land use in our study. Since the effect of restoration was generally higher on terrestrial and semi-aquatic organism groups, we recommend that they are considered in the monitoring and assessment of river restoration projects.

In general, the effect of restoration on community structure, traits, and functional indicators was more pronounced compared to the effects on species number and diversity. These changes in community structure indicate specific functional changes caused by river restoration and can be used to increase our understanding how restoration measures affect aquatic ecosystems, investigate causal relationships, and identify sustainable, (cost-) effective restoration measures. Therefore, we recommend that future restoration projects and monitoring studies should focus more on functional aspects (e.g. species traits, community structure) to investigate how river restoration affects river hydromorphology and biota, which would offer a great opportunity to make fundamental advances in restoration ecology and management.

The factors potentially constraining or enhancing the effect of restoration were partly correlated, which made it difficult to infer causal relationships (e.g. most old projects

were located in gravel-bed rivers where mainly widening was the main restoration measure applied, and catchment land use was less intensive). Nevertheless, it was possible to draw some first conclusions on the conditions favouring restoration success:

It has been widely stated that **large-scale pressures** like water quality and fine sediment loads might constrain the effect of restoration. However, in this study, catchment land use did only affect restoration success for floodplain vegetation, and restoration effect might have been rather constrained by the limited **species poo**l available for re-colonization and dispersal since the organism groups which did benefit most from restoration also have relatively high dispersal abilities (ground beetles, macrophytes). This topic clearly merits further investigation since a limited re-colonization potential would need a completely different restoration strategy compared to reach-scale habitat improvements.

Restoration extent (length of restored section, restoration intensity) was not the main factor determining restoration effects. Most probably, the restoration projects investigated were simply too small to benefit from possible positive effects of restoration extent, which is also supported by other recent studies. Furthermore, **project age** (time between implementation of the measures and monitoring) only had a positive effect on the aquatic habitat conditions but not on any of the organism groups investigated, possibly due to the young age of most projects investigated. In contrast, project age was identified as one of the most important variables affecting restoration success in the REFORM deliverable D 4.2, stressing the need to further investigate the effect of restoration over time in future studies.

Widening was applied in 11 of the projects investigated and had a significantly larger effect on hydromorphology and several organism groups (e.g. ground beetles, macrophytes) compared to other measures (among others instream measures), which is consistent with the findings of the REFORM deliverable D 4.2, and the widely endorsed assumption that restoring geomorphological processes has a higher effect compared to other measures. Since widening includes a set of measures, it was not possible to investigate the contribution of single measures. Since the positive effect on ground beetles was mainly due to the creation of open pioneer habitats covered by sparse woody vegetation, flattening river banks might already suffice but this has to be further investigated. Moreover, these results do not question the use of **instream measures** since transferability is limited due to the relatively low number of instream projects investigated in this study, and results of several other studies showing that instream measures generally have a positive effect on different aquatic organism groups.

The results indicated that future restoration projects should aim at increasing and monitoring **habitat diversity** at spatial scales which are ecologically relevant for the targeted organism groups. Although we found enhanced macro- and mesohabitats, which often is visually appealing, the measures often failed at increasing microhabitat diversity, which in turn was correlated with the effect of restoration on macroinvertebrates. Furthermore, it is not necessarily most important to increase the mere number of habitat types (e.g. habitat diversity) but to restore specific habitats which are of special importance. For ground beetles, the positive effect of widening was mainly due to the strong relationship between ground beetle richness and a specific habitat type: the open pioneer stage covered by sparse woody vegetation, but not to the mere number of habitat types.

The following more specific conclusions can be drawn for the single organism groups and river hydromorphology:

Overall, restoration increased habitat diversity through changes in channel morphology. The dominance of the main channel was significantly reduced, while other channel features such as islands, banks and bars became more frequent. The effect of restoration on **hydromorphology** was not higher in larger restoration projects compared to smaller projects. The effect of restoration was high for macro- and mesohabitat divsersity but low for microscale substrate composition. Key indicators for identifying restoration success should include parameters at larger spatial scales such as channel adjustments. There is a need to develop terrestrial parameters to assess the lateral dimension of restoration.

In line with other restoration studies no effects of restoration on **macroinvertebrates** were detected. However, macroinvertebrate richness and diversity was correlated with microhabitat diversity. While restoration projects like widening are visually appealing and increase macro- and mesohabitat diversity, they apparently rarely increase microhabitat diversity relevant for macroinvertebrates and species diversity.

Fish respond in a consistent way to hydromorphological restoration measures by an increase of rheophilic and a decrease of eurytopic fish. The restoration effect increases with habitat quality and length of restored river sections. Future restoration should focus on more dynamic, self-sustaining habitat improvements extending over several kilometres.

Restoration had an overall positive effect on richness and diversity of specific **macrophytes** (so-called helophytes, emergent plants rooting under water or in wetted soils) but not on emergent and submerged aquatic plants (hydrophytes). Restoration effects were especially high in widening projects located in mountain rivers.

An increase in total **ground beetles** species richness and richness of habitat specialists could be achieved primarily by creating pioneer patches, for example by river widening, which result in more open banks. Suitable restoration measures should aim on a strong lateral connection between the river and its floodplain. Further research should focus on determining optimal conditions of such pioneer habitats.

Responses of **floodplain vegetation** were related to changes in trait composition, while general effects on diversity were limited (small restoration projects) or absent (large restoration projects). Few general responses to restoration could be detected because species and trait composition and plant diversity varied substantially between the European regions.

For stable isotopes results supported our hypotheses that trophic length (indicated by Δ15N) as well as diversity of assimilated food sources (indicated by Δ13C) increase with restoration. Δ13C was significantly larger in large restoration projects compared to the corresponding degraded sections, suggesting that macroinvertebrates were feeding from more diverse sources. The results underlined the necessity to limit comparisons to sections within a region, as large-scale differences possibly masked the effects of restoration.

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1. Introduction

1.1 Background

Over the last decades, enhancing the hydromorphological and biological state of degraded rivers has become a widely accepted ecological and social objective in developed nations (Shields et al. 2003, Bernhardt et al. 2005). An increasing number of rivers have been restored in the past few decades but only a small number of these projects have been monitored, and hence, the knowledge on the effect of river restoration projects is limited (Bash and Ryan 2002, Bernhardt et al. 2005).

The few studies investigating the effect of restoration on hydromorphology and biota reported contrasting results. Several studies showed that the ecological effect of river restoration projects has been small even if local river morphology and habitat conditions have substantially improved (Lepori et al. 2005, Jähnig et al. 2010, Palmer et al. 2010). In contrast, other studies found a significant positive effect of river restoration on specific organism groups (Lorenz et al. 2012, Schmutz et al. 2014). The few narrative reviews that compiled information on a larger number of restoration projects also found highly variable restoration effects (Roni et al. 2002, 2008). Moreover, two recent quantitative meta-analysis showed that restoration generally has a positive effect on the diversity and abundance of different aquatic organism groups (fish, macroinvertebrates, macrophytes) but variability of restoration effect was high and a substantial part of the projects showed no or even a negative effect (Miller et al. 2010, Kail and Angelopoulos 2014). The high variability is probably partly due to real differences in the effectiveness of the restoration measures applied, as well as other catchment, river, and project characteristics which either enhance or constrain restoration effect. For example, Kail and Angelopoulos (2014) reported that nearly half of the variance in restoration effect was due to differences in characteristics like project age, river size, catchment land use, organism groups, river type, and the biological metric considered as well as the restoration measures applied. The substantial unexplained variance might be partly due to missing information on factors enhancing or constraining restoration effect (Roni et al. 2008) but also caused by the large methodological differences in respect to monitoring design, field sampling, and data analysis, which limits comparability of results. Therefore, scatter in the dataset could possibly substantially be reduced and the prediction of restoration effect could be enhanced by using a standardized monitoring and sampling design as well as data analysis, resulting in a harmonized dataset.

Besides the limitations due to the high variability, it is presently difficult to draw general conclusions on the effect of restoration on biota since most studies were restricted to one or few organism groups, mainly to fish and invertebrates (Lepori et al. 2005, Jähnig et al. 2010, Palmer et al. 2010, Schmutz et al. 2014, Miller et al. 2010). There are some few studies on the effect of restoration on macrophytes (e.g. Lorenz et al. 2012) and ground beetles (Januschke et al. 2011), but comparative studies on several organism groups are rare (Jähnig et al. 2009, Januschke et al. 2011, Haase et al. 2013, Kail and Angelopoulos 2014), and studies comprising aquatic, semi-terrestrial, and terrestrial biota are virtually missing (but see Jähnig et al. 2009, Januschke et al. 2011).

While most of the studies mentioned above quantified the effect of different restoration measures on different organism groups, only few studies tried to identify catchment, river or project characteristics which either constrain or enhance restoration effect and to identify conditions which favour restoration success (Miller et al. 2010, Kail and Angelopoulos 2014). A variety of reasons for limited biotic effects of morphological restoration measures has been suggested, including (i) stressors acting at larger scales such as water quality, those associated with intensive landuse and hydrological alterations in the catchment (Palmer et al. 2010, Lorenz and Feld 2013; Sundermann et al. 2013), (ii) the inadequate restoration of hydromorphological processes (Jähnig et al. 2009), (iii) minor changes in relevant microhabitats (Lepori et al. 2005), and a limited re-colonization potential due to a lack of source populations and a large number of migration barriers (Stoll et al. 2014, Tonkin et al. 2014). Several authors suggest a hierarchy of stressors, with water quality parameters, in particular oxygen depletion caused by organic pollution, acting as an overarching stressor which may mask the effects of habitat enhancement (Sundermann et al. 2011, Wahl et al. 2013). In principal, other water quality parameters, such as pesticides, can act similarly (Malaj 2014). Moreover, there is overwhelming evidence that stressors acting at larger spatial scales (catchment, subcatchment, sections of several kilometres in length) strongly determine aquatic assemblage composition (Kail and Hering 2009, Lorenz & Feld 2013, Marzin et al. 2013, Verdonschot et al. 2013). The pathways are manifold (Feld et al. 2011) and include, in addition to water quality, alteration of water temperature (Kiffney et al. 2003) and fine sediment entry (Teufl et al. 2013). All these can significantly influence assemblage composition in a restored section and thus limit restoration effects.

Many of these parameters, which potentially limit the effects of habitats enhancement, may be mitigated in large restoration projects where restored sections are relatively long and/or restoration actions have been intense. Accordingly, restoration effect possibly depends on restoration extent. Hydromorphological processes are scale dependent, including the formation of meanders and braided patters and of riffle-pool sequences (Richards et al. 2002). Similarly, water quality parameters may differ between short and long restored river sections: the effect of riparian forests on water temperature is depending on the length of a shaded river section (Kiffney et al. 2003); self-purification depends on the length of a section with near-natural morphology. Assuming similar large-scale pressures, short restored sections are likely to be more strongly impacted by stressors acting at the catchment scale, e.g. fine sediment entry. Viable populations of aquatic organisms require a minimum area of suited habitats. Finally, the effect of natural channel features like large wood or boulders on habitat conditions and biota largely depends on the amount present (Fausch and Northcote 1992). A strong correlation between the restoration extent and the biological effects can therefore be assumed.

In this study we compiled a harmonized dataset on the effects of hydromorphological river restoration measures on biota based on a standardized monitoring and sampling design to minimize scatter due to methodological differences. A broad range of response variables was investigated to draw conclusions on the effect of restoration on biota in general, including habitat composition in the river and its floodplain, three aquatic organism groups, two floodplain-inhabiting organism groups, as well as food web composition and aquatic land interactions as reflected by stable isotopes. Additional data on factors potentially constraining or enhancing the effect of restoration were compiled to identify conditions which favour restoration success and we designed the study to especially investigate the effect of restoration extent.

1.2 General study design

Restored sections and study reaches

We investigated ten pairs of one large and a similar but small restoration project to address the role of restoration extent for river restoration effects. The restoration effect was quantified by comparing each restored river section to a nearby non-restored, i.e. still degraded section (space for time substitution, Figure 1-1).

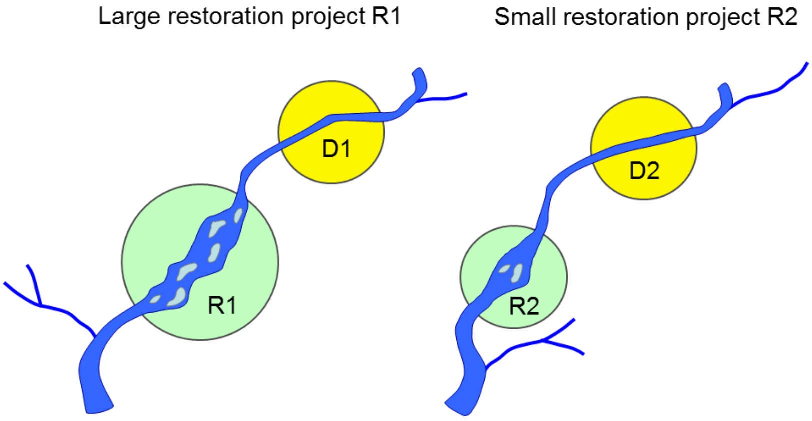

Figure 1-1: General study design of the paired restored sections.

The large restoration projects were representing good-practice examples in Northern Eastern and Central Europe either targeting medium-sized lowland rivers or medium-sized mountain rivers and were located in Finland, Sweden, Denmark, the Netherlands, Germany (lowlands), Germany (mountains), Poland, the Czech Republic, Austria and Switzerland (Figure 1-2). One study reach was selected in the downstream part of each of these large restoration projects R1 to consider potential mitigating effects of restoration extent like the reduction of fine sediment loads in the downstream part caused by deposition of fines in the upstream part. A second study section still degraded (D1) was selected some few hundred meters upstream of the restored section. For each of the ten large restoration projects, a second restoration project was selected in a river of comparable size and character. In contrast to the R1-sections, these restored sections were shorter and/or restoration has been performed with less intensity. Similarly to the large restoration projects, one study reach was selected in the small restoration projects (R2) and one in a degraded section some few hundred meters upstream (D2). Virtually each of the 40 reaches was sampled for all of the response variables: hydromorphological variables, three aquatic organism groups (fish, benthic invertebrates, aquatic macrophytes), two floodplain-inhabiting organism groups (ground beetles and floodplain vegetation) and stable isotopes.

While restoration projects in a given region were selected to differ just in restoration intensity and were comparable in terms of river size, catchment land use and altitude, there was nevertheless inevitable variation between regions. First, we tested if there were general differences in restoration effect between the two groups of large and small restoration projects (despite regional differences). Second, to account for these regional differences, we limited direct comparisons of large and small restoration projects to the corresponding pairs and their degraded control sections (R1/D1 compared to R2/D2 following Kenobi et al. 1980), i.e. we mainly used the pairwise difference of corresponding large and small projects (R1 and R2).

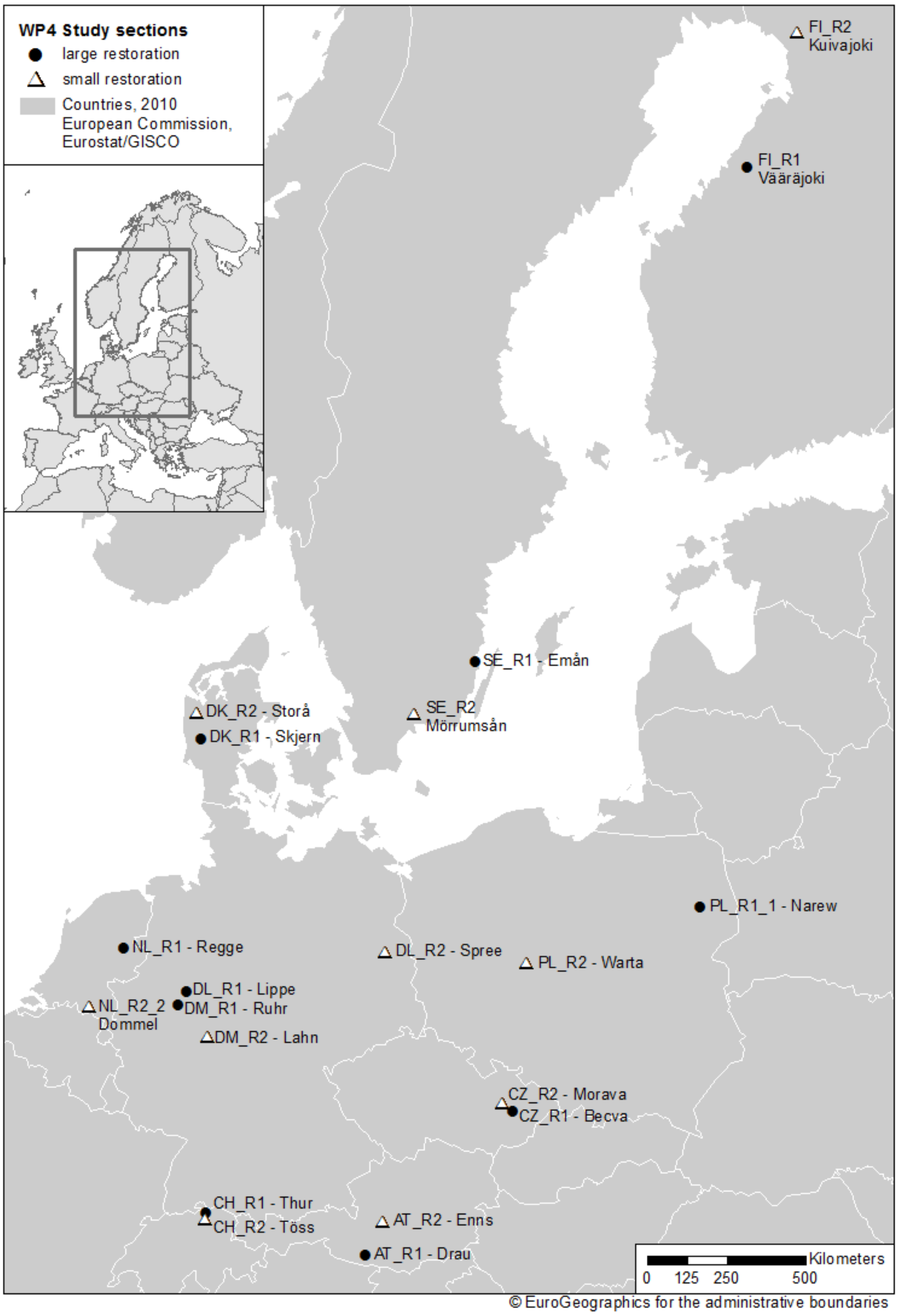

Figure 1-2: Location of the large (R1) and small (R2) restoration projects. Abbreviations consist of the country code, restoration extent code, and river name.

Quantifying restoration effect

The effect of restoration on the different response variables was measured using different variables and in different units. For example, variables used range from ordinal scaled assessment scores for the hydromorphological state to different biological metrics used to assess the biological state (e.g. richness, diversity, number of sensitive taxa), and different units were used to quantify species abundance (e.g. number of fish individuals, abundance classes of invertebrates). Therefore, it was necessary to standardize the different state variables and units using a single effect size to allow for meaningful comparisons of the restoration effect.

We used two different approaches to quantify restoration effect, and a total of three different effect sizes:

First, for a first overview analysis (Chapter 3), we quantified the difference between the restored (R) and corresponding degraded control reaches (D) for each response variable using the Bray-Curtis dissimilarity index, which ranges from 0 to 1 and quantifies the dissimilarity between the restored reaches. The similarity index is lowest (0) if the same objects occur in both sections having the same value (e.g. all species occur in both sections with the same abundance, all channel features occur in the same number), it is highest (1) if the sections have no objects in common (e.g. sections do not share any species). Assuming that the less similar the restored and degraded sections are, the higher the effect of restoration, dissimilarity between sections was used as an effect size, i.e. a high Bray-Curtis dissimilarity index is indicating a high effect size.

Second, for the more detailed analysis of each response variable (Chapter 4 to 10), we quantified restoration effect on different variables (e.g. species diversity or abundance, number of microhabitats) using two different effect sizes:

(i) Absolute values (subtracting values of the degraded sections from the restored sections R - D), with positive values denoting an increase and negative values a decrease of the variable. This effect size is easy to interpret since it gives the absolute change but variables measured in different units or different variables cannot be compared.

(ii) The response ratio of Osenberg et al. (1997):

$$\Delta r = \ln\left(\frac{\bar{X}_T}{\bar{X}_C}\right)$$

with $\bar{X}_T$ and $\bar{X}_C$ being the means of the treatment (restored) and control (degraded), values > 0 denoting a positive effect (e.g. increase in species diversity), and negative values a negative effect. According to Osenberg et al. (1997), an exponential model is assumed by using a logarithmic function, i.e. a fast increase of the variables in the first years and a smaller increase in the following years until equilibrium is reached. The response ratio is dimensionless (standardized) since $\bar{X}_T$ is divided by $\bar{X}_C$ and hence, the effect of restoration on different variables describing the hydromorphological, biological, and isotope conditions can be compared.

Hypothesis on restoration effect and the role of restored section length

Based on the results of previous studies and the potential constraining effect of large scale stressors (see Chapter 1.1), it was hypothesized that restoration effect differs between the response variables investigated (Figure 1-3, y-axis). We expected that floodplain-related variables (e.g. floodplain vegetation, ground beetles, floodplain and riparian habitats) respond more strongly, and variables related to the river itself (e.g. fish, benthic invertebrates, substrate diversity) respond weakly, as they are more strongly influenced by catchment-scale stressors, e.g. through water quality.

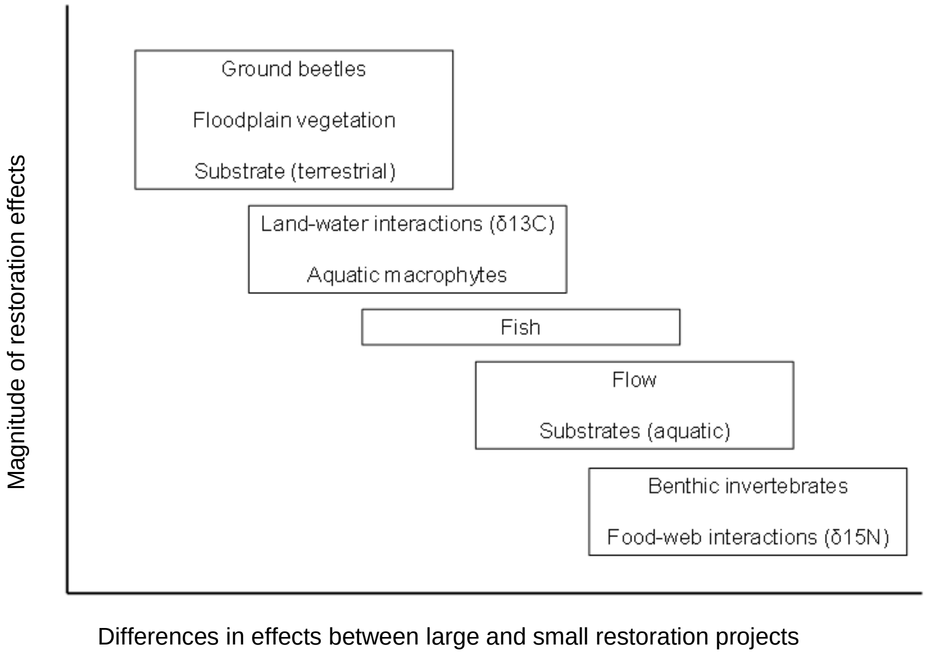

Figure 1-3: Conceptual diagram reflecting the magnitude of restoration effect (y axis) and the additional effect of restoration extent (x axis) for the response variables investigated.

In terms of variables potentially constraining or enhancing restoration effect, we focused on the mitigating effect of restoration extent for the reasons outlined in Chapter 1.1, a strong correlation between the restoration extent (e.g. restored section length, restoration intensity) and the restoration effects were assumed. However, the effect of restoration extent may differ between individual organism groups, hydromorphological and functional response variables. Primarily, strong effects can be assumed for organism groups which are most impacted by large-scale stressors (e.g. benthic invertebrates), which depend on hydromorphological processes requiring a certain section length (several instream habitats) and have a larger home range (fish). In more detail, we expected the following effects of restoration extent (Figure 1-3, x-axis):

- the weakest effect of restoration extent was expected for floodplain biota and habitats, as strong effects of restoration have been documented already for small restoration projects;
- a stronger effect of restoration extent was expected for land-water interactions, which depends on floodplain habitats, and for aquatic macrophytes, for which restoration effects have been documented for small restoration projects, but which are generally influenced by large-scale effects such as water quality;
- an even stronger effect of restoration extent was expected for fish as large organisms, which require a certain restored section length for sufficient population size;
- followed by aquatic microhabitats and flow patterns, which are generated by the restoration measure per se but are jeopardized by catchment influences, e.g. fine sediment entry, which will decrease with restoration extent;

- the strongest effect of restoration extent was expected for benthic invertebrates, which strongly depend on factors acting at larger scales; and for aquatic food web interactions, which depend on both aquatic habitats and benthic invertebrates.

In summary, we expected that restoration extent has a minor effect on response variables which strongly react to restoration (such as floodplain habitats), while it will boost the effect of restoration on variables generally responding poorly to restoration (Figure 1-3).

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2. Methods

2.1 *Selection of case study rivers and sections*

The selection of case study restoration projects and study sections was directed to cover two main river types, gravel-bed mountain rivers and sand-bed lowland rivers. The restoration projects comprised a wide range of hydromorphological restoration measures.

Basically, rivers and sections which met those selection criteria have been nominated by the project partners. An additional criterion was the availability of already existing monitoring data of these rivers and their catchments as well as the accessibility of the sections in the field. For the final selection also an even geographical distribution has been considered.

A more detailed description of the restoration projects and the applied measures as well as information on the catchment, river and project characteristics of the study sections are given in Annex B.

2.2 *Conceptual locations and dimension of sampling areas*

Almost all degraded sections were located upstream of the corresponding restored sections and with a sufficient distance to prevent mutual interferences. Within each degraded and restored section a representative sampling/mapping reach was selected. The restored sampling reach was located in the downstream part of the restored section to consider potential mitigating effects of restoration extent like the reduction of fine sediment loads in the downstream part caused by deposition of fines in the upstream part.

The lengths of sampling reaches depended on wetted channel width and the response variable (Table 2-1). Sampling reaches for recording hydromorphological transects and sampling of ground beetles, floodplain vegetation and stable isotopes were 200 or 500 m in length. For macroinvertebrates and macrophytes, the length of sample reaches was 200 m irrespective of wetted channel width.

Table 2-1: Length of sampling reaches (m) (wcw = wetted channel width).

	Reach length (m) for wcw < 50 m	Reach length (m) for wcw > 50 m
Hymo - survey	4x100(200)*	4x500
Hymo - transect method	200	500
Macroinvertebrates	200	200
Fish	10 to 20 times wcw (min. 100 m)	10 to 20 times wcw
Macrophytes	200	200
Ground beetles	200	500
Stabile isotopes	200	500
Floodplain vegetation	200	500

*wcw<20m – length of sampling reach is 100m; wcw=20-50m – length of sampling reach is 200m, Hymo = hydromorphological

The lateral boundaries of sampling areas and sampling seasons also differed between the response variables (Table 2-2).

Table 2-2: Sampling area (lateral boundaries)

	Recording/Sampling area	Recording/Sampling season
Hymo - survey	4x wetted channel width	Low flow in summer
Hymo - transect method: Channel features	The whole flood-prone area including aquatic, transient and terrestrial parts; in restored sections terrestrial area comprises the bankfull discharge area, in degraded sections the area of high-water level (debris lines); maximum width of 200m	Low flow in summer
Hymo - transect method: Microhabitats	Aquatic area	Low flow in summer
Macroinvertebrates	Aquatic area without oxbow lakes	Low flow in early summer (June to July)
Fish	Aquatic area	Late summer/early autumn
Macrophytes	Aquatic area	Maximum growth in low flow conditions (mid-summer)
Ground beetles	Strip of the river bank with a maximum width of 10 m	Late June (Mediterranean sites) to early August (Scandinavian sites)
Floodplain vegetation	The whole flood-prone area including aquatic, transient and terrestrial parts; in restored sections terrestrial area comprises the bankfull area, in degraded sections the area of high-water level (debris lines); maximum width of 200m	Maximum growth in low flow conditions
Stable isotopes	Aquatic, transient and terrestrial area; terrestrial area comprises the whole flood-prone area + a strip across the edges of embankment for sampling of non-riparian beetles	Maximum of biomass

2.3 *Hydromorphology*

The hydromorphological conditions of the restored and degraded sections were assessed by (1) mapping of the general hydromorphological state of the river and its surrounding floodplain area using a CEN compliant hydromorphological survey method and (2) more detailed mapping of the meso- and microhabitats, i.e. habitat composition of the river and the floodplain along transects.

CEN compliant hydromorphological survey method

For the hydromorphological field assessment a standard Austrian survey method (NOEMORPH - Hydromorphological Mapping of Selected Running Waters in Lower Austria; freiland umweltconsulting) has been selected and further developed to meet the requirements of the WFD and to be CEN compliant. We chose this method instead of the CEN norm itself as it is principally similar to several other national survey methods applied by default in Europe (see Rinaldi et al, 2013, Reform Del.1.1; Belletti et al., 2014).

Following the review of Belletti et al. (2014) these methods can be categorized as surveys for characterizing and evaluating physical river conditions. Overall parameters such as channel geometry or flow dynamics help to identify and assess main hydromorphological conditions of the river. These survey parameters are described and finally evaluated. Basically, the underlying evaluation follows the concept of reference conditions (WFD 2000). Applying this methodological approach the deviation of the current status of the river from the "type-specific status" - the target state that is represented by "reference conditions" - is analysed.

For the field assessment, the hydromorphological survey was conducted in four single reaches in the degraded and restored section, respectively (Figure 2-1), each reach 100, 200 or 500 m in length, depending on the wetted channel width (see Table 2-1). Hydromorphology of these four reaches has been characterised as well as assessed and evaluated separately. In restored sections with a length larger than the survey section (consisting of four reaches each 100, 200, or 500m in length), the survey section was located in the most downstream part of the restored section.

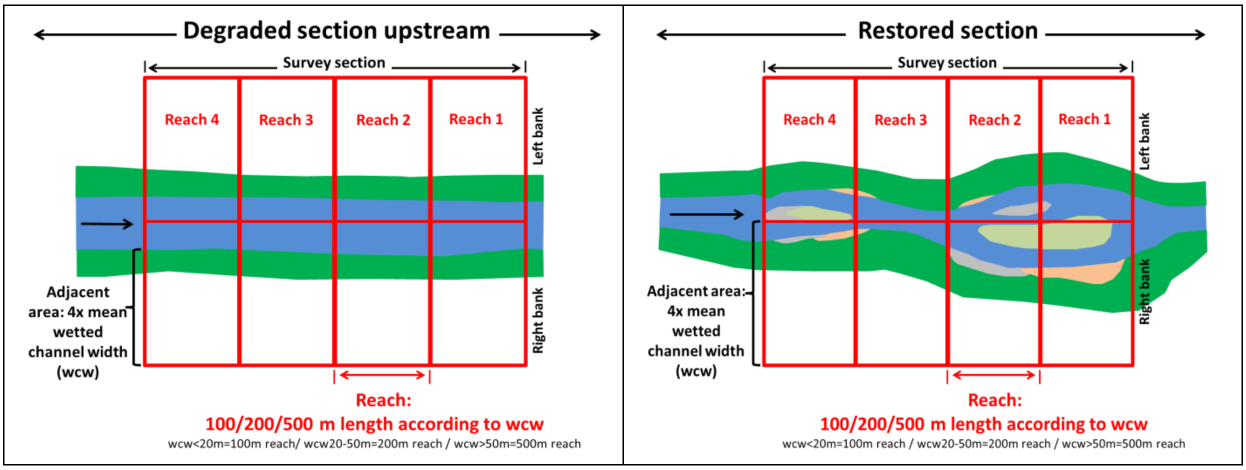

Figure 2-1: Delineation of survey sections and reaches for the field assessment in the degraded (left) and restored (right) sections.

All descriptive attributes listed in Table 2-3 were mapped and used to assess all evaluation parameters within five main survey parameters on a five-point ordinal scale ranging from 1 (undisturbed, hymo status "high") to 5 (totally disturbed, hymo status "bad"). Three of the main parameters describe the conditions in the active channel (channel geometry and flow characteristics, riverbed, water – land transition zone) whereas two main parameters are used to assess the riparian and floodplain area and hence, are recorded separately for the left and right side of the river (bank and riparian structures, vegetation of the adjacent area). In addition, dams and weirs, impoundments and water abstraction have been recorded, as well as basic information with regard to the geomorphological character, the vegetational zone and the morphological river type (Muhar et al. 2000, Jungwirth et al. 2003) of the river reach. To finally assess the hydromorphological status of each reach, we calculated the mean of all five main parameters.

Table 2-3: Descriptive attributes and evaluation parameters for the main hydromorphological survey parameters.

Main survey parameter	Descriptive Attributes	Evaluation parameter
Channel geometry and flow characteristics	**Current:** mean flow velocity in m/s **Flow character**: slow, uniform (homogenous anthropogenically caused), swirled/heterogeneous, turbulent	Channel Geometry Flow Pattern River Dynamics
Riverbed	**Depth:** maximum and minimum **Depth variability:** high, moderate, low, none **Riverbed stabilization:** absent, present covered by substrate, continuous stabilization structures, local stabilization measures **Type of riverbed stabilization:** concrete, asphalt, pavement, pavement grouted, cobbles **Choriotopes:** abiotic (ÖNORM 1997), biotic	Substrate characteristics Riverbed relief Hyporheic interstitial
Water – land transition zone	**Width variability:** high, moderate, low, none **Shoreline Stabilization:** absent, single, partly, continuous **Stabilization type:** biological engineering measures, combined, pilotage, riprap, stone pitching facing, stone pitching tightly packed, concrete **Important woody debris accumulation(s)** **Important bedload accumulation(s):** gravel banks, sand banks, silt banks	Connectivity Structures
River bank / riparian zone	**Cross-sections of longitudinal course:** variable, uniform; trapeze, double-trapeze, arc **Embankment** **Bank gradient:** vertical, steep (>30° - 1:1,6 and steeper), moderate (10-30° - 1:5 to 1:1,6), plain (<10° - 1:5 and less) **Bank protection:** absent, single, partly, continuous **Dimension of the bank** protection: < 1/3 of the bank, 1/3 of the bank, 3/4 of the bank, up to top edge of the bank **Type of bank protection:** biological engineering measures, combined, pilotage, riprap, stone pitching facing, stone pitching tightly packed, concrete, grass **Vegetation coverage of the river bank:** +/- 100%, >50%, <50%, absent	Bank characteristics Species composition of vegetation

Table continued

Main survey parameter	Descriptive Attributes	Evaluation parameter
River bank / riparian zone	**Vegetation coverage of the river bank:** +/- 100%, >50%, <50%, absent **Canopy/Shadowing of the water body:** complete, predominant, partly, absent **Vegetation Types:** vegetation types are assigned to one of four frequency classes for banks, vegetation types see below (vegetation of the adjacent area)	Riparian vegetation cover and age
Vegetation of the adjacent area	**Total width of woody riparian vegetation zone:** >15 m, multi-row 5-15 m, single-row 2-5 m, single-row interrupted, isolated woods/absent **Coverage of riparian woods:** +/- 100 %, >50 %, <50 %, absent	Buffer zone total
	Vegetation Types: (vegetation types are assigned to one of four frequency classes for the adjacent area) herbaceous pioneer vegetation, cane brake, tall herb fringe, nitrophilous fringe, invasive herbaceous species, woody pioneer plants, soft wood floodplain forest, hard wood floodplain forest, wetlands/bogs	Species composition of vegetation of surroundings
	pasture, fallow land, grassland extensive, grassland intensive, lawn, field, deciduous forest, mixed forest, coniferous forest, invasive woody species, no vegetation/ sealing	Vegetation cover and age of surroundings

Assessment of hydromorphological micro-/mesohabitats (transect method)
Within each restored and degraded section, we selected a sampling reach, 200 or 500 m in length depending on the wetted channel width (see Table 2-1).

First, common parameters were recorded in each sampling reach by counting the number of the following morphological characteristics:

- unvegetated bars and islands,

- bars and islands with herbaceous and with woody vegetation,

- woody debris and deadwood trunks,

- standing water bodies and sidearms.

Second, we divided each sampling reach into 10 transects spanning the flood-prone area from one side to the other comprising aquatic, transient and terrestrial zones. In restored sections it was the area of bankfull discharge, in degraded sections the area of high-water level. The bankfull width and height were measured for each transect. The transect method for recording hydromorphology at meso- and microscale comprised two steps:

1. recording of channel features in the flood-prone area,

2. recording of aquatic microhabitats.

Along each transect, the lengths of channel features, classified according to Jähnig et al. (2008) and Januschke et al. (2009), were measured (Table 2-4).

Table 2-4: Channel features (modified after Jähnig et al. 2008 and Januschke et al., 2009).

	Channel feature	Description
Aquatic	Main channel	Hydrological dynamic water body, most important runoff channel
	Secondary channel	Hydrological dynamic water body, connected with the main channel at both ends, less water runoff
	Connected sidearm	Water bodies lacking unidirectional current, connected only at the downstream or upstream end
	Disconnected sidearm	No connectivity with the main channel
	Permanent standing water body	On the floodplains, fed by high water levels and groundwater, no signs of drying
	Temporally standing water body	On the floodplains, fed by high water levels, dries out quite shortly, puddle-like
Transient	Bank with woody vegetation	Woody aquatic-terrestrial transient zone with an inclination <30°
	Bank with herbaceous vegetation	Herbaceous aquatic-terrestrial transient zone with an inclination <30°
	Side bar	Unvegetated bar close-by the shoreline either at the floodplain or at an island
	Midchannel bar	Unvegetated bar in the middle of main or secondary channel
Terrestrial	Island with woody vegetation	Large woody bar, separating main and secondary channel(s)
	Island with herbaceous vegetation	Large herbaceous bar, separating main and secondary channel(s)
	Artificial embankment	Artificially created area e.g. with trapezoidal or rectangular profile, often built of blocks as bank fixation
	Embankment with woody vegetation	Woody area with an inclination >30°, confines bankfull discharge area
	Embankment with herbaceous vegetation	Herbaceous area with an inclination >30°, confines bankfull discharge area
	Steep (unvegetated) embankment	Steep brim at riparian area with an inclination >50°; if inclination is 90°, it is mapped with length=0
	Floodplain area	Within bankfull discharge area, area prone to flooding

Within each channel feature we recorded the dominant substrate using the classification according to Hering et al. (2003) (Table 2-5). Transects were marked in an aerial picture and geographic coordinates (longitude, latitude, WGS84) of transect 1 and 10 were recorded in each sampling reach. Furthermore, we took pictures of at least transect 1, 5 and 10.

Table 2-5: Substrates for instream microhabitat recording according to multi-habitat sampling protocol (Hering et al. 2003); substrates marked green are also used for recording of channel features.

Substrate name	Description	Type	Grain size (mm)
Mega-/Macrolithal	Large cobbles, boulders and blocks, bedrock; coarse blocks, head-sized cobbles, with a variable percentages of cobble, gravel and sand	mineral	>200
Mesolithal	Fist to hand-sized cobbles with a variable percentage of gravel and sand	mineral	>60-200
Microlithal	Coarse gravel (size of a pigeon egg to child's fist) with variable percentages of medium to fine gravel	mineral	>20-60
Akal	Fine to medium-sized gravel	mineral	>2-20
Psammal	Sand	mineral	>0.006–2
Argyllal	Silt, loam, clay (inorganic)	mineral	<0.006
Technolithal	Artificial blocks often used as bank fixation in degraded sections	mineral	>200
Xylal	Tree trunks, dead wood, branches, roots	biotic	
CPOM	Deposits of coarse particulate organic matter, e.g. fallen leaves	biotic	
FPOM	Deposits of fine particulate organic matter, e.g. mud und sludge (organic)	biotic	
Algae	Filamentous algae, algal tufts	biotic	
Submerged macrophytes	Submerged macrophytes, including moss and Characeae	biotic	
Emergent macrophytes	Emergent macrophytes, e.g. *Typha*, *Carex*, *Phragmites*	biotic	
LPTP	Fine roots, floating riparian vegetation	biotic	

Aquatic microhabitats were recorded at 10 survey points along each transect, with the distance between the survey points being (width of water surface – 20 cm)/9, since survey point 1 and 10 have a respective distance of 10 cm from the left/right bank. At each survey point we recorded water depth, dominant substrate (Table 2-5) and flow velocity class (Table 2-6).

Table 2-6: Classification of flow velocity classes.

Flow velocity class	Description	Flow velocity (m/s)
0	stagnant	0
1	slow	<0,3
2	rippled	0,3-0,5
3	swirled	0,5-1
4	turbulent	>1

2.4 Macroinvertebrates

The sampling of macroinvertebrates followed an EU Water Framework Directive (WFD) compliant sampling protocol (e.g. Haase et al. 2004). We performed the multihabitat sampling standardized in the AQEM and STAR projects, which reflects the proportion of the microhabitat types (substrate types according to Hering et al., 2003) that are present with > 5 % cover. Samples were taken from a 200 m long reach (Figure 2-1) in early summer (June to July, see Table 2-1) prior to the emergence period of many Trichoptera and Ephemeroptera species.

Based on the microhabitat list given in the AQEM field protocol, the coverage of all microhabitats with at least 5 % cover was recorded to the nearest 5 % interval, the presence of other microhabitats (< 5 % cover) was indicated (by "X") but not quantified.

In each reach sampled, 20 individual macroinvertebrate samples (sample units) were taken with a hand-net/shovel sampler or a surber sampler with a mesh size of 500 µm. The recommended area is 25 x 25 cm each, resulting in 1.25 m^2 of river bottom being sampled. A 'sampling unit' is a stationary sampling accomplished by positioning the net and disturbing the substrate for a distance that equals the square of the frame width upstream of the net (0.25 x 0.25 m). The 20 sampling units were distributed according to the share of microhabitats. For example, if 50 % of the channel bed sampling reach was covered with sand (psammal), half of the sampling units (10 out of 20) have to be taken on sand.

In the field, the 20 samples of each sampling reach were pooled and preserved with ethanol (96 %). In the laboratory, the subsampling method based of Caton (1991) was used to reduce the effort required for sorting and identification, to provide an unbiased representation of a large sample, to provide a more accurate estimate of time expenditure and to reduce costs for the process of macroinvertebrate samples. Therefore, a minimum amount of 1/6th of the material has to be subsampled, containing a minimum number of 350 individuals. The subsampled individuals were sorted according to Haase et al. (2004). Species were identified to the lowest possible level as suggested by Haase et al. (2006).

In addition to the standardized multihabitat sampling, we took samples in lentic habitats. In most of the case-study pairs, one of the main differences between restored and morphologically degraded sections is the configuration of the bank structure. Restoration measures created shallow and slow flowing areas at the river banks. Therefore, to account for this difference, we investigated the macroinvertebrate communities at the river margins/banks by taking 5 sample units per sampling reach with the shovel sampler in the lentic zones. The lentic zone is characterized by flow velocities between 0 and 30 cm/s and a water depth between 1 and 30 cm. If there were different microhabitats present in the lentic zone, the 5 samples were allocated accordingly. The 5 sample units were pooled and subsequently sorted completely in the lab. Identification of the organism was done at the same level as in the standard composite sample.

2.5 Fish

The sampling of fish followed an EU Water Framework Directive (WFD) compliant sampling protocol (EFI+ Consortium, 2009). Therefore, we used standardised electric fishing procedures that are precisely described in the CEN directive, "Water Analysis – Fishing with Electricity (EN 14011; CEN, 2003) for wadable and non-wadable rivers".

According to the CEN-standard, the main purpose of the standardised sampling procedure is to record information concerning fish composition and abundance; therefore, no sampling period is defined (according to CEN). However, the EFI+ approach recommends to sample in late summer/early autumn, except for intermittent Mediterranean rivers where spring samples may be more appropriate.

Electric fishing in each study reach was conducted over a river length of 10 to 20 times river width, with a minimum length of 100 m to cover all habitats and fish communities present, and to accurately characterise the fish assemblage. However, in large and shallow rivers (width > 15 m, water depth < 70 cm) where electric fishing by wading can be used, several single sites were sampled with a total area of at least 1000 m^2 and a total length of 10 to 20 times river width, covering all types of mesohabitats present in the sampling section (partial sampling method).

As a general rule, one anode per 5 m of wetted width was used for sampling in wadable rivers. The operators fished upstream so that water and sediment disturbed by wading did not affect efficiency. Operators moved slowly, covering the habitat with a sweeping movement of the anodes and attempt to draw fish out of hiding. To aid effective fish capture in fast flowing water, the catching nets were held in the wake of the anode. Each anode was generally followed by one or two hand-netters (hand net: mesh size of 6 mm maximum) and one suitable vessel for transporting fish.

In large rivers, water depth (> 70 cm) and habitat diversity hindered sampling of the entire channel area. Therefore, a partial sampling procedure was applied covering all types of habitats to obtain a representative sample of the site. Qualitative and semi-quantitative information was obtained by using conventional electric fishing with hand held electrodes in the river margins and delimited areas of habitat. Alternatively, where resources exist, capture efficiency was improved by increasing the size of the effective electric field relative to the area being fished by increasing the number of catching electrodes (electric fishing boats with booms). Arrays comprising many pendant electrodes were mounted on booms attached to the bows of the fishing boat. The principal array was entirely anodic with separate provision being made for cathodes. Depending upon water conductivity, the current demands of multiple electrodes were high and large generators and powerful control boxes were needed.

Each collected specimen was identified to species level by external morphological characters. The total number of specimens per species was recorded and the total length of all fish captured was measured.

2.6 Macrophytes

Aquatic macrophytes were surveyed during the peak of the growing season by using an EU Water Framework Directive (WFD) compliant sampling protocol (Schaumburg et al., 2004).

Table 2-7: Growth forms of macrophytes.

Growth form	Definition	Example
Ceratophyllids	Free-floating plants with large, finely divided submerged leaves	*Ceratophyllum* spec., *Utricularia* spec.
Elodeids	Submerged plants with whorled stems	*Elodea* spec., *Hippuris vulgaris*
Equisetids	Horse tails	*Equisetum* spec.
Haptophyts	Mosses, red and green algae, lichen	*Fontinalis* spec.
Helodids (Helophytes)	Emergent plants	*Typha* spec., *Phalaris arundinacea*
Hydrocharids	Free-floating plants with rosettes of specialised floating leaves	*Hydrocharis morsus-ranae*
Isoetids	Submerged plants (and filamentous algae) with short shoots/stems and a rosette of stiff radical leaves	*Isoëtes* spec., *Littorella uniflora, Cladophora spec.*
Juncids	Submerged plants with simple, narrow, margin entire, with septate leafs (rush)	*Juncus* spec.
Lemnids	Free-floating plants with small leaf-like thalli	*Spirodela polyrhiza, Lemna* spec.
Magnopotamids	Submerged plants with oblong to lanceolate submerged leaves	*Potamogeton polygonifolius, Potamogeton crispus*
Myriophyllids	Submerged plants with leafs at stem, feather-like leafs	*Myriophyllum* spec., *Ranunculus* spec.
Nymphaeids	Plants with longly petiolated floating leaves	*Nuphar lutea, Persicaria amphibia*
Parvopotamids	Entirely submerged plants with linear to oblong leaves	*Zannichellia palustris, Potamogeton berchtoldii*
Peplids	Plants with oblong and spatulate leaves, the upper ones forming floating rosettes	*Callitriche* spec.
Vallisnerids	Submerged plants with a short stem and a rosette or bundle of long, linear, floating leaves, rooted in the soil	*Sparganium* spec., *Vallisneria spiralis*

Macrophyte sampling was done in the main growing season (July to mid-September). One 200 m reach (Figure 2-1, Table 2-1) was sampled in each of the restored and degraded sections by wading in a zigzag manner across the channel and walking along the riverbank. In non-wadable areas, the river bottom was raked with a rake (on a long pole or at the end of a rope) to reach the macrophytes. All macrophyte species were recorded and identified to species level, except for *Callitriche* stands without fruits, which were identified to genus level. The survey included all submerged, free-floating, amphibious and emergent angiosperms, liverworts and mosses. In addition, plants were recorded which were attached or rooted in parts on the river bank that were likely to be submerged for more than 85% of the year. The abundance of each species was recorded according to the 5-point NOVANA scale: 1= 1-5 %; 2= 5-25 %; 3= 25-50 %; 4= 50-75 %; 5= 75-100 %. Additionally, the growth form of each species was recorded according to Den Hartog & Van der Velde (1988) and Wiegleb (1991). The growth forms (Table 2-7) comprise different plant species that realized the same or comparable phenotypical adaptations to the aquatic environment.

2.7 Ground beetles

Ground beetles were investigated in one reach of each restored and degraded section, with the length of the reaches (200 or 500 m) depending on the wetted channel width (Table 2-1). Sampling season was late June to early August at conditions of low discharge; in the Scandinavian sites ideally August, in the Mediterranean sites ideally late June. As there is no standard method for ground beetles, we developed a mesohabitat-specific sampling procedure similar to the multihabitat sampling of benthic invertebrates (Haase et al. 2004).

The sampling area comprised max. 10 m wide strips of all riparian areas including river banks left and right of the river channel and mid-channel bars. If width of the river banks was less than 10 m (common in degraded reaches), the sampling area only included the area below the high-water level. If the banks of degraded sections were made up of riprap, we positioned the traps in the embankment, preferably in the shortest distances to the area of high-water level.

The coverage of different riparian mesohabitats (Table 2-8) was estimated in each sampling strip in 10 %-steps; mesohabitats with coverage of < 10 % were recorded and marked with an “x”. If available, aerial photographs and/or transect data of recorded channel features were additionally used.

Only mesohabitats with a coverage of at least 10 % were sampled. Each 10 % of total habitat coverage accounted for one riparian beetle sample; so, according to the mesohabitat composition 10 samples were taken per sampling reach.

Vegetated mesohabitats were sampled by using pitfall traps (diameter 4 cm, depth 8.5 cm, volume 200 ml) filled with 100 ml Renner-solution (40 % ethanol, 20 % glycerine, 10 % acetic acid, 30 % water) and a detergent to reduce surface tension. The pitfall traps were secured from rain and falling leaves by a petri dish (9 cm diameter) as a roof. Traps were exposed for one week. After collecting the traps, larger animals that are not part of the epigeic arthropod fauna were removed; mice were preserved separately. All other animals were placed in vials (1 vial per pitfall trap) and preserved with 96 % ethanol.

Open bars (mesohabitats with < 25 % vegetation coverage) were sampled by 'hand sampling' at sunny days. Organisms were collected with an exhaustor in an area of 1 m^2 per sample. A wooden quadratic frame (50 x 50 cm = 0.25 m^2) was used to delineate the surface area to be sampled. For one sample, four 0.25 m^2 areas were sampled parallel to and in direction of the shoreline. Each area was scanned for a maximum of 10 minutes by turning over all mineral and organic substrates to collect riparian beetles, which hide or live in the underground. Afterwards, water was poured over the area to drive organism hidden in the interstitial to the surface. All organisms were sucked in with the exhaustor, killed using some drops of ethylacetate and afterwards preserved with 96 % ethanol. The 10 individual samples per sampling reach were kept separate. For each sample, we recorded the sampled mesohabitat and the type of sample in the field protocol. Ground beetle species were identified to the species level according to Müller-Motzfeld (2004).

Table 2-8 Classification of mesohabitats used for the sampling of ground beetles

Mesohabitats for carabid sampling	Description
Riparian forest	> 25 % coverage of woody riparian vegetation; trees cover the area
Pasture	Gras land (no tree cover)
Other herbaceous vegetation	Riparian herbaceous vegetation (no tree cover)
Vegetated swamp	Very moist (muddy) vegetated patches
Steep (unvegetated) embankment	Steep brim at riparian area with an inclination >50°
Open gravel bar	< 25 % vegetation coverage, dominated by gravel
Open sand bar	< 25 % vegetation coverage, dominated by sand
Open mud bar	< 25 % vegetation coverage; dominated by mud

2.8 Floodplain vegetation

Floodplain vegetation was sampled in summer (June-July) in one reach of each restored and degraded section, with the length of the reaches (200 or 500 m) depending on the wetted channel width (Table 2-1). We chose three of the transects that were surveyed for hydromorphology (transect method), one at the lower, middle and upper end of the sampling reach.

First, the length of vegetation units, classified according to Oberdorfer (1983, 1992) and Ellenberg (1996) to the order level (Table 2-9), were measured along transects to determine the proportion of vegetation units per reach.

Second, each of the transects was divided into three subzones of equal length on each side of the main channel, with subzone 1 located nearest to the waterline of the main channel. If for example the total width of the floodplain at one site of the main channel is 100 m, then each sub-zone will be 33 m. A total of 12 sample plots (size 0.5 m x 0.5 m) per transect was established (six on each side of the main channel) differently distributed within the subzones following a randomized and stratified sampling approach. At each side of the channel, three sample plots per transect were placed evenly distributed in the first subzone, two sample plots were placed in the second subzone and finally one sample plot was placed in the third subzone. The total number of sample plots per reach was 18 sample plots in subzone 1, 12 sample plots in subzone 2 and six sample plots in subzone 3. In case of a narrow floodplain (e.g. just 5 meters of each side of the channel), in which the sample plots could not be distributed along transects, nine sample plots were placed close to the waterline of the main channel, three at the margins of waterline and six in between. Within the sample plots, plant species and their abundance were recorded by estimating their coverage following the classification of Braun-Blanquet: (0-1 %, 1-5 %, 5-25 %, 25-50 %, 50-75 %, 75-100 %) and values were transformed into the Ord% scale (1, 2, 8.5, 35, 70, 140) following Van der Maarel (2007).

Table 2-9: Classification of vegetation units according to Oberdorfer (1983, 1992) and Ellenberg (1996) adjusted to particular 'new' units.

Name of vegetation unit	Description
Aegopodion	Nitrophilous stands dominated by *Urtica dioica*, *Aegopodium podagraria* or *Galium aparine*
Afforestation with non-native or atypical species	Embankment afforestations with *Salix-, Alnus* or *Fraxinus*-species (atypical or non-native species)
Afforestation with *Populus* sp.	Afforestation with *Populus*-species
Agropyro-Rumicion	Grassland in frequently flooded areas dominated by *Alopecurus geniculatus*
Alno-Padion	Most frequent floodplain-forests in low-mountain regions dominated or characterized by *Alnus glutinosa* (tree layer) and *Stellaria nemorum* in the herb layer

Table continued

Name of vegetation unit	**Description**
Arrhenatherion - fragment association	Mown (or grazed) grassland dominated by *Arrhenatherum elatius* and other meadow-species like *Trifolium pratense*, *T. repens*, *Alopecurus pratensis* or *Leucanthemum vulgare*, as well as species poor stands composed of *Arrhenatherum elatius* and a few other species (e.g. *Dactylis glomerata*, *Taraxacum officinalis* agg.) frequently abandoned
Artemisietea fragments	Stands dominated by Elytrigia repens or with high cover values of *Cirsium arvense*
Bidention - fragment association	Species poor and not well developed stands dominated or characterized by *Bidens*-species
Calthion	Moist, species poor grassland dominated by *Scirpus sylvaticus* (and *Juncus effusus)*
Calthion elements	Moist, species poor grassland dominated of or only comprising *Juncus effusus*
Calthion-Filipendulion	Embankment edges dominated by *Mentha aquatica* and others
Calystegion - fragment association	Nitrophilous stands dominated by *Impatiens glandulifera*
Calystegion sepi	Nitrophilous stands dominated by *Calystegia sepium, Convolvulus, Galium aparine* (and *Urtica dioica)*
Calystegion sepi - fragment association_Heracleum	Nitrophilous stands dominated by or only comprised of *Heracleum mantegazzianum*
Calystegion sepi - fragment association_Fallopia	Stands dominated by *Fallopia* ssp.
Calystegion sepi - fragment association_Solidago	Nitrophilous stands dominated by or only comprised of *Solidago*
Carpinion	Forests characterized by *Carpinus* ssp. and *Quercus robur* in the tree layer, *Stellaria holostea* and *Poa nemoralis* in the herb layer
Dauco-Melilotion_diverse	Dry ruderal stands dominated or characterized by *Daucus carota, Melilotus* ssp. or *Echium vulgare*
Dauco-Melilotion_Tanacetum	Stands dominated by *Tanacetum vulgare*
Epilobion fleischeri	Open gravel banks dominated by *Salix purpurea*, *Myricaria germanica* and different weeds

Table continued

Name of vegetation unit	Description
Fagion	Forests dominated by *Fagus sylvatica*
Glycerion_Sparganium	Stands of *Sparganium* ssp. in running water bodies with low current
Glycerion_Glyceria	Stands dominated by *Glyceria fluitans* or *G. plicata*
Glycerion_Veronica	Stands dominated by *Veronica beccabunga*
Lemnion	Stands of floating *Lemna* ssp.
Magnocaricion	Stands of tall sedges like *Carex gracilis, C. acutiformis*
Mixture of Sysimbrion-Chenopodium-Dauco-Melilotion on gravel bars	Sparse vegetation on open gravel banks comprising a species-mixture from many different units, frequently characterized by predominantly dry-ruderals like *Daucus, Melilotus, Sisymbrium, Echium* or other ruderals like *Arctium, Saponaria, Alliaria*
Nymphaeion	Flooding stands of *Myriophyllum spicatum*
Phalaridion	Reeds of *Phalaris arundinacea*
Phragmition_Phragmites	Stands of *Phragmites australis*
Phragmition_Typha	Stands dominated by *Typha latifolia*
Plantaginetalia fragments	Stands of agricultural managed grasslands with high covers of *Lolium perenne* or *Agrostis stolonifera* not part of the *Arrhenaterion* vegetations
Potamogetonion_Elodea	Standing water bodies dominated by *Elodea*-species
Potamogetonion_diverse	Stands of floating species like *Nymphaea, Nuphar, Potamogetum* etc.
Potamogetonion - Pot_Glyc	Stands of *Potamogetum*-species in pools or in water bodies with low current, frequently mixed with *Glyceria* ssp.
Pruno-Rubion-fruticosi / Calystegion sepi - fragment association	Shrub patches dominated or characterized by *Rubus fruticosus* agg. or *R. caesius*
Quercion	Woods and forest on acidous soils dominated by *Quercus petraea*
Ranunculion / Nymphaeion	Stands in pools or in water bodies with low current dominated or characterized by *Callitriche* ssp.
Ranunculion fluitantis	Flooding stands of *Ranunculus fluitans*
Rubo-Prunion	Shrub patches dominated or characterized by *Prunus spinosa* or *Crataegus* ssp.

Table continued

Name of vegetation unit	**Description**
Salicion albae_1	Frequently flooded woods and forests dominated by *Salix alba* or *S. fragilis* (and hybrids)
Salicion albae_2	Floodplain-woods characterized by *Salix viminalis, S. cinerea or S. triandra*
Salicion eleagni	Pioneer-vegetation on gravel banks; *Myricaria germanica, Salix purpurea*
Sambuco-Salicion_Betula	Wood and shrubland of early successional stages dominated by *Betula pendula*
Sambuco-Salicion_Sambucus_Salix	Open woods in early successional stages dominated by *Sambucus ssp., Salix caprea*
Senencion Union fluviatilis	Stands dominated by *Impatiens glandulifera, Solidago canadensis* and *Urtica dioica*

2.9 *Stable isotopes (N and C)*

Stabile isotopes were investigated at time of maximum biomass in summer 2012 or 2013 in one sub-reach of each restored and degraded section, with the length of the reaches (200 or 500 m) depending on the wetted channel width (Table 2-1). The sampling procedure and the stable isotope analysis supported the investigation of effects of river restoration on ecosystem functioning. It aimed to show the effect of hydromorphological restoration on aquatic terrestrial linkages and on the complexity of food webs by comparing restored and degraded reaches across Europe. Therefore, the sampling was done in aquatic, riparian and terrestrial areas aiming to cover the dominant taxa of each component to gain a representative and comparable overview of the trophic structure. The following components of the food web were sampled: fine and coarse particulate organic sediment (POM), periphyton, dominant aquatic and riparian plants, dominant benthic invertebrates, and predatory riparian and terrestrial arthropods (beetles and spiders).For the sampling of benthic invertebrates we further categorized into the following functional feeding types: predators, shredders, grazers, collector-filterers and collector-gatherers and we aimed to cover at least the dominant taxa for each of these types.

Samples of stream bed organic sediment (POM) were taken with a sediment corer in ten different POM deposition zones per reach. The upper 1-2 cm of the sediment core were transferred to a sample bucket. The samples were pooled in a bucket per reach. In the laboratory, fine and coarse POM was separated by sieving and benthic invertebrates were removed.

Periphyton was brushed from randomly selected plants and/or stones into stream water and filtered through Whatman GF/F filters. Aquatic and riparian plants were sampled by taking at least one sample of the dominant species along the section. Plants were identified to genus level. For each sample, different plants/stands of the same species were collected to achieve a representative composite sample.

Benthic invertebrates were taken from different habitats along the section using a shovel sampler (mesh size 500 µm) and hand net. The sampling aimed to collect late-instar larvae of major taxa representing the following functional feeding types:

- Predators (e.g. *Rhyacophila* sp., *Sialis* sp.),
- Grazer (e.g. *Baetis* sp., *Rhithrogena* sp.),
- Shredders (e.g. *Gammarus* sp., *Asellus* sp., *Nemoura* sp.),
- Collector-gatherers (e.g. Oligochaeta,)
- Collector-filterers (e.g. *Hydropsyche* sp., *Simuliidae* sp.).

In the field, individuals were pre-sorted to genus level, counted and kept separated by functional feeding groups to avoid contact between predators and prey. For each feeding group at least one composite sample was taken. Each sample consisted of several individuals to obtain sufficient material for analysis.

Riparian and terrestrial ground-beetles (Carabidae) and spiders (Araneae) were sampled using an exhaustor. They were collected within 1 m of the stream edge, terrestrial arthropods across the top edge of the embankment. Sampling locations were randomly selected along the sample sections. Each composite sample consisted of several individuals. All samples were placed in a frost box in the field.

Fish sampling was optional and depended on fishing restrictions within the countries. In case fish were sampled, they were identified to species level; the length and weight of each fish was measured. For each sample, tissues (liver and dorsal muscle) were taken.

In the laboratory, benthic invertebrates, riparian and terrestrial arthropods were kept individually for 12 to 24 hours to allow for gut evacuation. In case of benthic invertebrates they were kept in filtered stream water. Afterwards, specimen were identified to the lowest feasible level.

To prepare samples for stable isotope analysis they were ground with mortar and pestle and freeze-dried afterwards until all water has been removed. According to the amount of sample material, four subsamples of each component were loaded into tin capsules (species ~0.8 mg and sources 2-15 mg). Content of carbon (C) and nitrogen (N) and stable isotopes of C and N were analysed with an elemental analyser (CE Instruments – EA 1110 CHNS) connected to a Thermo Finnigan MAT 253 isotope ratio mass spectrometer at University of Duisburg-Essen’s Stable Isotope Facility (Instrumental Analytical Chemistry Department).

Data of the stable isotope analysis are expressed as relative difference between ratios of samples and standards (VPDB for $\delta^{13}C$ and air for $\delta^{15}N$). The analytical precision over all measurements (standard deviation from 791 in-house standards) was 0.08‰ for $\delta^{13}C$ and 0.19‰ for $\delta^{15}N$.

2.10 Database

To enable the investigation of the effects of hydromorphological river restoration measures on river habitats and biota, task 4.2 and 4.3 required the collection of various variables for each dataset. Thus, comparable data on hydromorphology, pressures,

restoration measures, land use and numerous biotic key variables, potentially constraining or enhancing restoration success were collected for all cases study catchments. Data were compiled from current field sampling (see Chapters 2.3-2.9) as well as already existing monitoring data of the case study sites. Additionally data from national databases of the WP4 partner countries supplemented the data records.
The database for the collected information on all WP4 case study catchments consists of several sheets for the various key subjects, tables with ID and taxa lists as well as a detailed description of all required variables. Approximately 600 parameters were defined to describe the following key subjects:

- five abiotic (site information, hydromorphology, pressure types, restoration measure types, physico-chemical data),
- five biotic (fish, invertebrates, macrophytes, riparian beetles, floodplain vegetation) and
- seven catchment related subjects (BQE status, colonization sources, hydromorphology, hydromorphological pressures, pressure point/diffuse sources, physico-chemical data, additional parameter).

Table 2-10 gives an overview of the database’s content. More detailed tables and descriptions of all variables and parameters of the database can be found in Annex A.

Table 2-10: Content overview of the WP4 database.

Table	Content	No. of entries / variables
Abiotic data (according to task 4.2)		
SiteInfo	General information on case study site	53
Hydromorph	Information on hydromorphology of the site	38
Pressure	Information on pressure types of the site	28
RestorMeasures	Information on restoration measures of the site	81
PhysChemic	Information on physic-chemical parameters	21
Biotic data (according to task 4.3)		
Fish		
Fish Site	General info on fish sampling reach	8
FishSample	Specific info on fish sampling (date, method, etc.)	27
Fish Catch	Specific info on fish catches (taxa, etc.)	9
Invertebrates		
InvSite	General info on invertebrates' sampling reach	6
InvSample	Specific info on invertebrates' sampling (date, method, etc.)	8
InvCatch	Specific info on invertebrates' catches (taxa, etc.)	4
Macrophytes		
MacrophSite	General info on macrophytes' sampling reach	5
MacrophSample	Specific info on macrophytes' sampling (date, method, etc.)	7
MacrophCatch	Specific info on macrophytes' catches (taxa, etc.)	6
Riparian beetles		
BeetSite	General info on sampling points of riparian beetles	5
BeetSample	Specific info on sampling of riparian beetles (date, method, etc.)	16
BeetCatch	Specific info on riparian beetles' catches (taxa, etc.)	7
Floodplain Vegetation		
VegSite	General info on vegetation sampling reach	5
VegSample	Specific info on vegetation sampling (date, etc.)	6
VegTransUnit	Info on transects, vegetation orders and units	7
VegTaxa	Specific info on vegetation taxa and coverage	5
Catchment data (according to task 4.2 and 4.3)		
BQE_status	Info on Biological Quality Classes for different buffers	35
Colonization sources		52
Hydromorph	General info on hydromorphology of the catchment	15
Pressure_hydromorph	Info on hydromorphological pressures in the catchment	63
Pressure_sources	Info on point / diffuse sources of pressure in the catchment	26
PhysChemic	Info on physico-chemical parameters in the catchment	75
Additional parameter	General info about the catchment (GDP, population density, etc.)	9

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PL:

Hydrological data are obtained from the IMGW (Institute of Meteorology and Water Management – National Research Institute. Some other data (slopes, longitude, latitude, altitude etc.) obtained from own (WULS) measurements.

SE:

All data comes from GIS analyses made at the Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences (SLU). Except:

Mean discharge in River Emån: SMHI öppna data http://opendata-catalog.smhi.se/explore/, Swedish Hydrological and Meteorological Institute SMHI

-Mean discharge in River Mörrumsån: Raw data supplied by E.ON Vattenkraft Sverige AB, discharge calculations made at the the Department of Aquatic Sciences and Assesment, Swedish University of Agricultural Sciences (SLU).

3. Overview analysis

3.1 Introduction

Worldwide, rivers are being restored at an increasing rate to enhance overall biodiversity, re-create fish habitats and to increase attractiveness and ecosystem services provision (Bernhardt *et al.* 2005; Strayer & Dudgeon 2010). River restoration is a business worth billions of dollar / Euro and driven by societal demands and respective legislation, such as the EU Water Framework Directive (WFD) or the Clean Water Act in the US (Baron *et al.* 2002; Sondergaard & Jeppesen 2007; Palmer 2009). In Europe, recent inventories highlight the pivotal role of river hydromorphology for river biota and ecological status: The hydromorphology of about 50% of European river water bodies is degraded; in Central European countries such as Germany and the Netherlands almost all river sections are affected (EEA 2012). Consequently, the majority of river restoration measures will in future need to target hydromorphological improvements.

In sharp contrast to the demand and investments in restoration little is known about restoration effects and factors responsible for success or failure. The majority of measures, both in Europe and North America, have hardly been subjected to monitoring and evaluation (Bernhardt *et al.* 2005, Jähnig *et al.* 2011). However, recently a growing body of literature deals with river restoration effects on hydromorphological and biotic response variables. The majority of studies report minor effects on benthic invertebrates (Harrison *et al.* 2004; Jähnig *et al.* 2010; Haase *et al.* 2013; Friberg *et al.* 2014), and minor to medium effects on macrophytes (Pedersen *et al.* 2007; Lorenz *et al.* 2012) and fish (Roni, Hanson & Beechie 2008; Lorenz *et al.* 2013; Schmutz *et al.* 2014; Stoll *et al.* 2014), while direct effects on hydromorphology and effects on floodplain biota are stronger (Woolsey *et al.* 2007; Jähnig *et al.* 2009; Januschke *et al.* 2014).

A variety of reasons for limited biotic effects of hydromorphological restoration measures has been suggested, including stressors acting at larger scales, such as catchment land use, water quality, and hydrological alterations; the insufficient restoration of hydromorphological processes; minor changes in relevant microhabitats; lack of recolonization potential and blocked recolonization pathways.

Viewed in more detail, water quality parameters, in particular oxygen depletion caused by organic pollution, acts as an overarching stressor and may mask the effects of habitat enhancement (Sundermann *et al.* 2011; Wahl*,* Neils & Hooper 2013). In principal, other water quality parameters, such as pesticides, can act similarly (Rasmussen *et al.* 2012; Malaj *et al.* 2014). There is overwhelming evidence that stressors acting at larger spatial scales (catchment, sub-catchment, reaches of several kilometre lengths) strongly determine aquatic assemblage composition (Kail & Hering 2009; Lorenz & Feld 2013; Marzin, Verdonschot & Pont 2013; Verdonschot *et al.* 2013). The pathways are manifold (Feld *et al.* 2011) and include, in addition to water quality, alteration of water temperature (Kiffney, Richardson & Bull 2003) and fine sediment entry (Teufl *et al.* 2013). All these can significantly influence assemblage composition in a restored reach and thus limit restoration effects. The generation and establishment of habitats relevant for aquatic biota requires hydromorphological processes addressing the overall morphological character at a larger spatial scale (e.g. initiating a braiding system) and therewith the generation of gravel bars, pools and riffles and supply of large wood. In many projects, these processes are not being restored and biological effects are therefore vanishing once the restored habitats have been subject to hydromorphological and biological processes (Hughes, Colston & Mountford, 2005). In other cases the habitats relevant for sensitive biota are not being generated to a sufficient degree. This particularly concerns benthic invertebrates, which are often depending on few key habitats such as wood or gravel, which are either not resulting from restoration (Jähnig *et al.* 2009) or covered by fine sediment soon after restoration (Lorenz, Jähnig & Hering 2009). Even if habitats have been generated, there is no guarantee that they are being colonized by aquatic species sensitive to anthropogenic disturbance and that measures are initiating changes in the composition of aquatic assemblages. Due to long lasting river degradation and pollution, populations of sensitive species have been eradicated from entire catchments and source populations from which restored sections could be recolonized are scares (Harding *et al.* 1998; Hughes 2007). Recolonization may further be obstructed by barriers in the river (most relevant for fish; Roberts, Angermeier & Hallermann 2013) and the surrounding landscape (most relevant for dispersing adults of aquatic insects; Dedecker *et al.* 2007). In conclusion, restored river reaches might be impacted by a mix of stressors with manifold and complex interactions that will retain them in an unfavourable condition.

Many of these parameters, which potentially limit the effects of habitats enhancement, may be mitigated in large restoration projects where restored sections were relatively long and/or restoration actions have been intense, and hence, restoration effect possibly depends on restoration extent. Hydromorphological processes are scale dependent, including the formation of meanders and braided patters and of riffle-pool sequences (Richards et al. 2002). Similarly, water quality parameters may differ between short and long restored river sections: the effect of riparian forests on water temperature is depending on the length of a shaded river section (Kiffney et al. 2003); self-purification depends on the length of a section with near-natural morphology. Short restored sections are relatively stronger impacted by stressors acting at the catchment scale, e.g. fine sediment entry. Viable populations of aquatic organisms require a minimum area of suited habitats. Finally, the effect of natural channel features like large wood or boulders on habitat conditions and biota simply depends on the amount present. A strong correlation between the restoration extent and the biological effects can therefore be assumed.

In this study we analysed the effects of hydromorphological river restoration measures on different response variables. We investigated ten pairs of one large (R1) and one similar but small (R2) restoration project to address the role of restoration extent for river restoration effects. The restoration effect was quantified by comparing each restored river section (R1 and R2) to a nearby non-restored degraded section (D1 and D2) (space for time substitution). We addressed a large number of response variables, including habitat composition in the river and its floodplain, three aquatic organism groups, two floodplain-inhabiting organism groups, as well as food web composition and aquatic land interactions reflected by stable isotopes (see Chapter 1.2 for a more detailed description of the study design).

In this overview analysis, we quantified the difference between the restored (R) and corresponding degraded control reaches (D) for all response variable using the Bray-Curtis dissimilarity index, while the more detailed analysis of the single response variables is described in the following Chapters.

3.2 *Material and methods*

Study sections and sampling methods

The study sections and reaches as well as sampling methods for the response variables are described in Annex B and Chapter 2.3 to 2.9.

Data analysis

For each response variable, Bray-Curtis dissimilarity between the restored (R) and the nearby no-restored, degraded section (D) was calculated. The dissimilarity expresses the "effect size", i.e. a high Bray-Curtis dissimilarity is indicating a high effect size. For the biotic response variables we used species-station tables indicating the (relative) abundance. In case of floodplain vegetation species coverages in each plot were transformed into the Ord% scale, which is an appropriate transformation before conducting numerical analyses on vegetation data (van der Maarel & Franklin 2013). For floodplain mesohabitats and instream microhabitats we used tables indicating the relative coverage of habitats in the river sections. For flow velocity patterns a table with the number of aquatic survey points meeting the individual flow velocity classes were used. For stable isotopes (C and N) tables indicating the relative content of δ13C and δ15N (in ‰), respectively, in the individual sampled components were used. In contrast to all other parameters the stable isotope tables had no missing values, as always the same set of components was sampled.

First, we tested if the general effect of restoration in the 20 restored sections differed between the response variable, i.e. if the mean effect sizes of the response variables were significantly different using a one-way ANOVA. Second, we tested if restoration had a larger effect on the different response variables in large (R1) compared to the small (R2) restored sections, i.e. if effect sizes were significantly different in the ten large (R1) compared to the corresponding ten small (R2) restoration projects using a Wilcoxon Matched Pair Test. Third, we re-grouped the sections based on the analysis of the hydromorphological data survey and compared sites with larger changes in substrate and habitat composition (S1) to those with smaller changes (S2). For each of the ten pairs of restoration projects (i.e. region), the restored section with the strongest difference in substrate composition compared to the corresponding unrestored, degraded section was labelled as S1, while the other restored section was labelled S2. The resulting grouping based on the instream substrates (relevant for benthic invertebrates, macrophytes and fish) were always equivalent to the grouping resulting from the floodplain habitats (relevant for floodplain vegetation and ground beetles). For six of the ten pairs / regions the S1 sections were identical with the large restoration projects (R1), but in four regions (Switzerland, German mountain area, Denmark, Netherlands) the stronger changes in instream habitats occurred in the smaller restoration projects (R2). For each response variable, we tested if effect sizes were significantly different in the ten S1 compared to the corresponding ten S2 restoration projects using a Wilcoxon Matched Pair Test.

Bray-Curtis dissimilarity was calculated with a self-written Excel macro, all other analyses were performed in R (Version 3.0.2, http://www.r-project.org/).

3.3 *Results*

Overall effect of restoration on response variables (R1 and R2 pooled)

In general, restoration effect differed between the response variables despite the regional differences in river and project characteristics (e.g. river size, restoration measures). Considering all restoration projects regardless of restoration extent (R1 and R2 sections pooled), Bray-Curtis dissimilarity significantly differed between the response variables (Figure 3; one-way ANOVA, $F_{9/185}=35.21$, $p<0.01$).

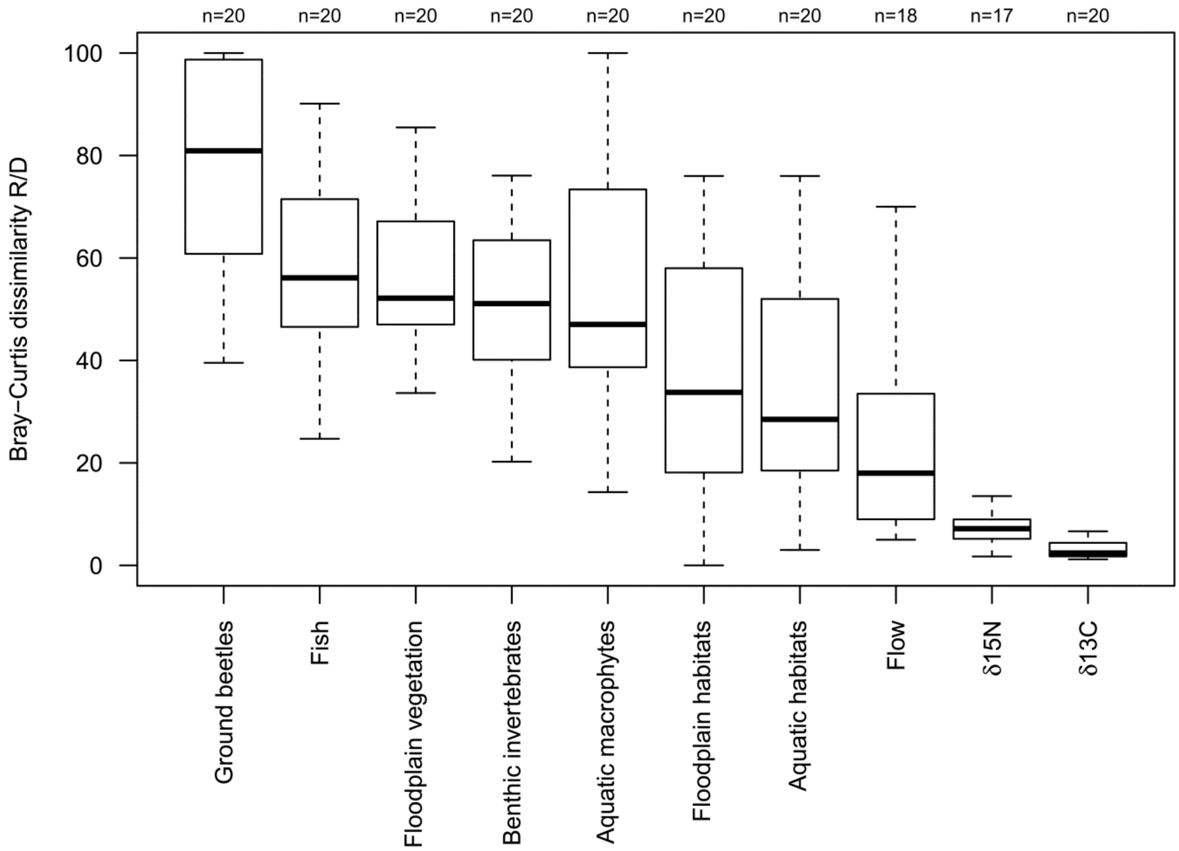

Figure 3-1: Figure 3: Restoration effects on response variables.

Floodplain biota (ground beetles and floodplain vegetation) were among the variables most strongly responding to restoration. The general order of response variables according to the effect sizes shows comparatively strong effects on aquatic biota (macrophytes, benthic invertebrates and fish) and weak effects on floodplain habitats (Figure 3-1).

Differences of restoration effect in large and small projects (R1 vs. R2)

Restoration effect did not differ between the large (R1) and small (R2) restoration projects (Figure 3-2). Positive values for the difference of the Bray-Curtis dissimilarity of a large (R1) minus a small (R2) restoration project indicates a larger restoration effect of R1 sections. Median difference of all ten pairs of restoration sections (R1 vs. R2) was indeed positive for all response variables, except for ground beetles and fish. However, in contrast to our expectations, restoration effects of large and small restoration projects were not significantly different ($p > 0.17$), except for the food web interactions (δ15N) (Wilcoxon Matched Pairs test, n=10, $p < 0.05$).

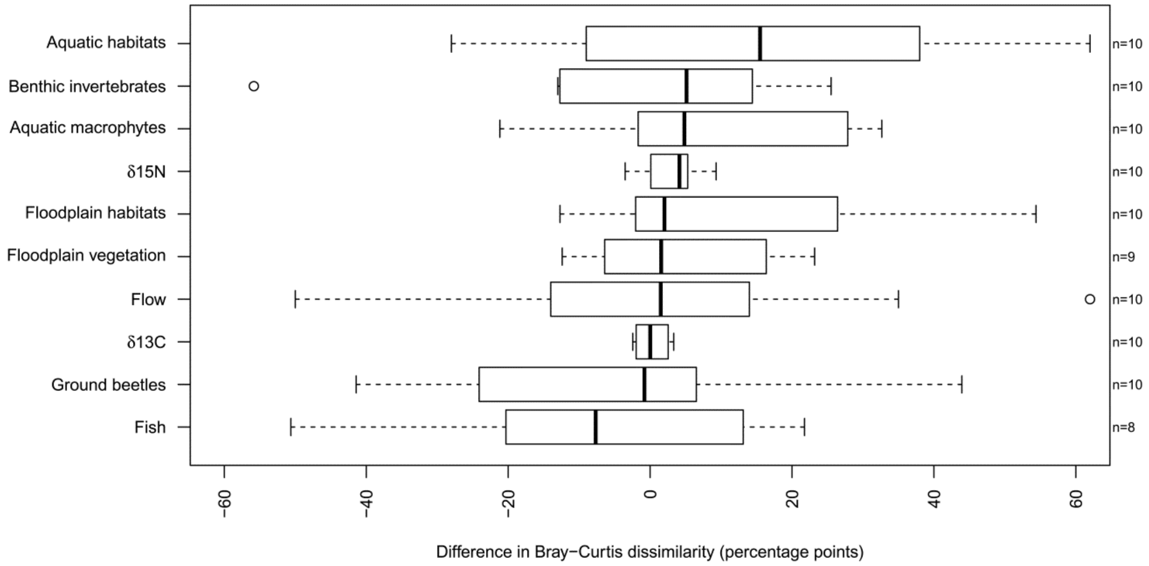

Figure 3-2: Difference between the restoration effects (Bray-Curtis dissimilarity) of the large (R1) and small (R2) restoration projects (i.e. R1 minus R2 values) for morphological and biological response variables. Median values, quartiles, and non-outlier range of all ten pairs are shown.

Effects of substrate diversity

Restoration effect was generally larger in those restoration projects where changes in aquatic substrate conditions were more pronounced compared to the corresponding restoration projects with smaller changes. Median difference of the ten pairs of restoration sections (S1 vs. S2) was positive for all response variables (Figure 3-3) indicating a larger restoration effect in S1 restoration projects to the corresponding S2 projects. Moreover, restoration effect sizes of S1 projects were significantly larger for most response variables: benthic invertebrates, aquatic macrophytes and all recorded morphological response variables (flow diversity, floodplain habitats) (Wilcoxon Matched Pairs test, n = 9-10, $p < 0.05$). Though differences between S1 and S2 were not significant for fish (probably due to the small sample size), the differences were positive, larger than between R1 and R2, and nearly significant ($p = 0.08$).

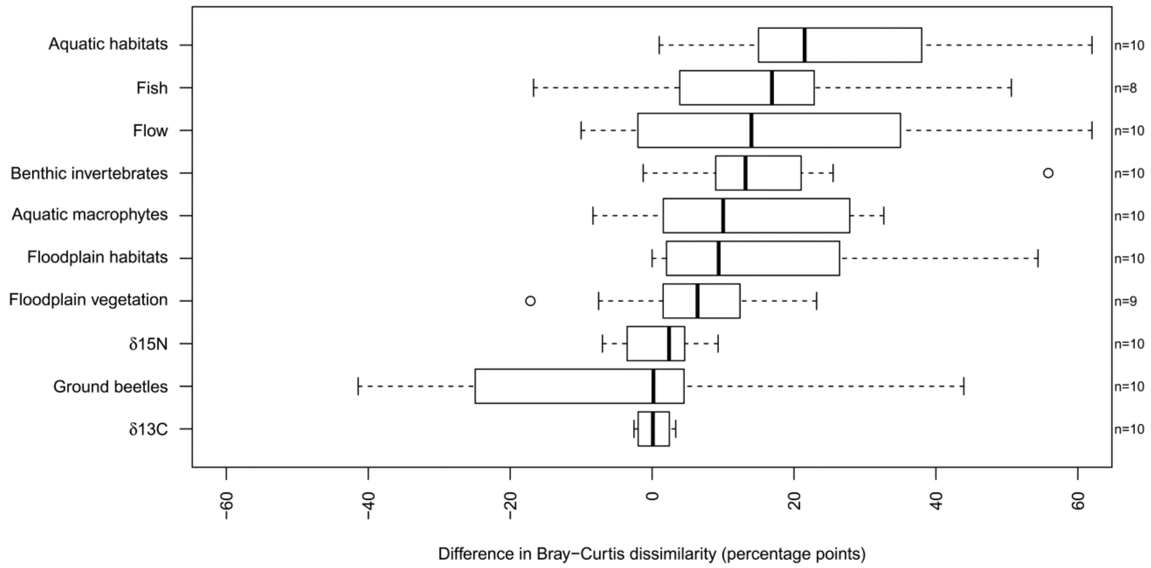

Figure 3-3: Difference between the restoration effects (Bray-Curtis dissimilarity) of the restored sections with higher changes in substrate conditions (S1) and the corresponding restored sections with smaller changes (S2) (i.e. S1 minus S2 values) for morphological and biological response variables. Median values, quartiles, and non-outlier range of all pairs are shown.

3.4 *Discussion*

Effects of restoration on different response variables

In line with the results of Jähnig *et al.* (2009, 2011), we expected a ranking of response variables in terms of restoration effects (Figure 3-1), with strong effects on floodplain habitats, floodplain biota and land-water interactions, and only minor effects on aquatic organism groups such as benthic invertebrates and macrophytes. This hypothesis was partly confirmed; though restoration effects on floodplain biota were strongest, we also observed compositional changes of fish, benthic invertebrates and aquatic macrophytes, while changes of land-water interactions reflected by carbon isotope signatures were minor. Possible reasons for these observations, which differ from what has recently been published, include factors related to both the investigated restoration measures and our data analysis strategy.

The observed strong restoration effects on floodplain biota are in line with several publications (Rohde *et al.* 2005; Lambeets *et al.* 2008; Jähnig *et al.* 2009; Meyer *et al.* 2010; Januschke *et al.* 2011). Hydromorphological restoration, even relatively small measures, tend to create habitat types close to the land-water interface (such as gravel and sand bars), which are almost completely lacking in degraded sections. Such habitat types are rapidly colonized by riparian ground beetles and, to a lesser degree, by specialized floodplain vegetation. Both organism groups have a comparatively high dispersal ability (Bates, Sattler & Fowles 2006; Johansson & Nilsson 1996; Soons 2006).

We also suppose effects on land-water interactions, as particularly riparian ground beetle species feed on aquatic organisms, which emerge close to the shoreline or are washed ashore (Paetzold, Schubert & Tockner 2005). For the overall composition of carbon isotope signatures, however, these alterations do not suffice, as they just affect single components of the food web.

The relatively strong effect of restoration on aquatic biota (fish, benthic invertebrates, macrophytes) corresponds to the significant effects of restoration found in recent meta-analyses (Miller, Budy & Schmidt, 2010; Kail and Angelopoulos 2014) but differs from the small or missing effect reported in many other studies (e.g. Lepori *et al.* 2005; Jähnig *et al.* 2010; Palmer, Menninger & Bernhardt 2010). In parts, this may be due to our type of data analysis. In contrast to several other publications we did not compare metrics but compared taxa lists using a dissimilarity coefficient. The Bray-Curtis Index can be applied to a wide array of response variables, including all (semi)quantitative taxa lists and also (semi)quantitative lists of habitat composition. This is an advantage over metrics such as feeding type composition or indices used for bioassessment, which are specific for individual organism groups and can therefore not be used to compare different response variables. However, the Bray-Curtis Index has its limitations. First, dissimilarity is not necessarily related to quality nor to successional stage: strong dissimilarities between assemblages (such as ground beetles) of restored and degraded sections may be caused by various reasons, including natural variability, increasing or decreasing environmental quality. Second, the number of observed species or habitats influences the results; in case of fish or floodplain habitats it is more likely that all species or habitats present in the section have been recorded, while in case of benthic invertebrates some species might have been overlooked. Third, the result is not just determined by abundances but also by the number of species, habitats or components both sections have in common; this explains the always very high similarities in case of carbon and nitrogen isotopic composition, as, in each case, the same set of components has been sampled. The effect size *per se* is therefore hardly comparable between the isotopic composition and the other parameters, while the size effect or the effects of habitat alterations can be compared.

Despite these methodological limitations, the results reflect surprisingly strong restoration effects. One reason is the representation of extensively restored "flagship projects" and restored sections with substrate compositions greatly differing from degraded sections. These are much more likely to yield positive results on aquatic biota than those reported in the majority of recent publications, which is supported by a meta-analysis of Kail and Angelopoulos 2014 showing that the variability of restoration effect is high.

Restoration extent

We expected stronger restoration effects in case of large restoration projects. Furthermore, we expected a strong effect of restoration extent on those response variables, which generally respond poorly to restoration. In these cases restoration effects will be most strongly masked by catchment influences, which decrease with restoration extent. These hypotheses were rejected, as effect sizes of only one response variables (food web interactions; $\delta 15N$) significantly differed between the large (R1) and small (R2) restoration projects.

For the floodplain biota this observation is in line with the general ranking of response variables: in particular ground beetles responded in all cases strongly to restoration (in particular to river widening), even in small restoration projects and there is thus no additional effect of restoration extent. Based on the species-area relationship we would expect an increase in local biodiversity with an increase in area (i.e. restoration extent). As the restored habitats are young most probably we are still only seeing pioneer species, and time is needed before more developed stages evolve.

In case of the aquatic biota, most likely the majority of the large restoration projects (R1) were still too small to cause significant effects of restoration extent. With the exception of Skjern (Denmark) and Narew (Poland) all restored sections of the large restoration projects were shorter than, or equalling, 2 km, possibly not a sufficient length to initiate additional geomorphic processes or support viable populations of additional sensitive species. In the cases of the Skjern (compare Kristensena *et al.* 2014) and the Narew, however, the restoration measures were mainly affecting the floodplains, in which large flood prone areas were created, while there were relatively little changes in stream bottom substrates. In contrast, in the small restoration project Stora (Denmark) the focus was on instream measures, which directly generated habitats for aquatic organisms. This example shed light on an intriguing result of our study, namely the strong effect of habitat alterations on aquatic biota, which overrules possible effects of restoration extent.

Based on these results, one should not conclude that it is sufficient to restore short river sections and implement small restoration projects. The majority of the large restoration projects (R1) were still too small to cause significant differences compared to the smaller projects (R2). For example, restored section length was less than or equalling 2 km, except for two restoration projects (see Annex B). This is consistent with the results of Kail and Angelopoulos (2014) who also concluded that the missing effect of restored section length on restoration success was most probably due to the short length of most restored sections investigated (< 2.6 km). Moreover, it is in line with the results of Schmutz *et al.* (2014), who observed a higher effect of restoration on the number of rheophilic fish species in long restored as compared to short restored sections but only at length greater than 3.8 km.

Effects of substrate diversity

The magnitude of changes in aquatic substrate conditions directly impacted benthic invertebrates, macrophytes and morphological variables and also initiated taxonomic changes of fish assemblages, though not significant. Ground beetle assemblages and carbon stable isotope signatures, however, were not related to substrate conditions of the restored sites but responded already to slight changes in habitat composition.

In case of floodplain biota a similar rationale as for the missing effect of restoration extent can be assumed: already the relatively minor substrate alteration in the S2 sections caused significant effects, and in case of larger substrate changes no additional effects were generated.

For the aquatic biota, in particular for benthic invertebrates, our results differ from the majority of published studies. For example, Jähnig *et al.* (2008) observed Bray-Curtis similarities of 69-77% between benthic invertebrate assemblages of restored (braided) and nearby degraded mountain stream sections with only very few taxa specific for the restored sections. Based on an extensive literature analysis, Kail and Angelopoulos (2014) recently summarized restoration effects on macrophytes, benthic invertebrates and fish: there are minor to medium effects of measures enhancing substrate diversity (such as widening or addition of large wood); the mean species numbers and abundances in restored sections are between 1.13 and 1.2 higher than in control sections. It should be noted that these results are not necessarily comparable to dissimilarity indices, as used in our study.

While there is an overall rationale for the coherence of changes in bottom substrates and changes in aquatic assemblages (i.e. the provision of habitat), there are often no biological effects of respective measures. We assume that in many published cases the measures did not result in significant and sustainable instream habitat changes and that therefore the response of biota is minor. When relating measures and quantified habitat composition to biota, such as in our study, the coherence is much more obvious.

3.5 Conclusion

For effects on aquatic or floodplain biota, restoration extent was not directly relevant, maybe as even the large restoration projects investigated in our study are still too small for an additional positive effect based on project size. The study by Schmutz *et al.* (2014) suggests, however, the existence of a "size effect". Habitat composition has an impact on both floodplain and aquatic biota. In case of the floodplain assemblages, in particular ground beetles, already minor restoration effort results in significant effects, obviously as small habitat patches are already sufficient. In case of aquatic biota, larger substrate changes are required, as revealed by the differences in effect sizes between projects leading to smaller and larger substrate changes. In conclusion, the effects of hydromorphological restoration measures on aquatic and floodplain biota strongly depend on the generation of habitats for aquatic and riparian organisms, which were not present, or not sufficiently so, prior to restoration. These positive effects on habitats are not necessarily related to restoration extent.

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4. Hydromorphology

4.1 *Introduction*

River restoration is a key issue in River Basin Management. Over the last decades a variety of restoration measures have been conducted over different river types and spatial extents of restoration, starting from local and experimental projects to a broader scale implementation (Kondolf et al. 2007, Roni et al. 2008, RESTORE 2011 Failure or success of the implemented measures were documented for only few restoration projects, and long-term monitoring programs have rarely been carried out (Pander & Geist 2013, Smith et al. 2013, Kail & Angelopoulos 2014). In the context of the European Water Framework Directive (WFD) it is crucial to implement the most effective restoration measures and identify the most suitable indicators for assessing restoration effectiveness.

In many restoration projects hydromorphological parameters have been used to assess restoration success (Morandi et al. 2014, Kurth & Schirmer 2014), often based on the assumption that the restored natural habitat heterogeneity would support good ecological condition. Although the relationships between habitat diversity and the response of specific organism groups (fish, macroinvertebrates, macrophytes) have been widely studied (Palmer et al. 1997 and 2010, Lepori et al. 2005, Vaughan et al. 2009, Friberg 2010, Miller et al. 2010, Elosegi et al. 2011, Sundermann et al. 2011, Paillex et al. 2013), the biological indicators often have not shown response to restoration despite an enhancement in hydromorphological condition or heterogeneity. The restored habitat conditions might still not be favourable for the establishment of specific biological groups. However, at least three other factors not related to the restored habitat conditions can potentially explain a small biological response to hydromorphological restoration: Source populations might be missing (Schmutz et al. 2013), the post-restoration time might be insufficient to enable recolonization of biota (Hering et al. in prep.), or large-scale pressures might affect local biota in the restored reaches (Sundermann et al. 2011, Bernhard & Palmer 2011, Verdonschot et al. 2013). Therefore, hydromorphological variables should always be used in addition to biotic monitoring as basic parameters within restoration monitoring schemes to assess restoration success. This calls for choosing adequate parameters that portray habitat condition, habitat diversity and, optimally, processes within the monitored reach (Brierley et al. 2010) integrating effects from up- and downstream.

Many indicator systems and protocols have been developed for determining the morphological condition of aquatic systems (Belletti et al. 2014). Such hydromorphological assessment methods, however, are not necessarily suited to evaluate restoration effect. Most of the physical habitat assessment methods focus on the reach scale describing meso- and/or microhabitat conditions and characteristics (Belletti et al. 2014, LAWA 2002). At the reach scale, changes in habitat diversity due to restoration can be measured, but the integration of large-scale influences may be insufficient (Brierley et al. 2010). Morphological assessment methods (Belletti et al. 2014) use larger river units such as river segments (REFORM 2014) and investigate parameters at wider spatial scales integrating dynamic processes (Brierley & Fryirs 2005, Rinaldi et al. 2013). Assessing the effectiveness of river restoration requires identifying those hydromorphological parameters which are best suited to detect the main changes caused by the specific restoration measures. These parameters potentially vary among river types.

There is broad agreement that effects of reach-scale restoration are potentially constrained by catchment influences (Palmer et al. 2005, 2010, Beechie et al. 2010). Several studies indicated large impacts, especially on biota such as invertebrates, macrophytes and fish (Hering et al. 2006, Miller et al. 2010, Bernhardt & Palmer 2011, Lorenz et al. 2012, Kail et al. 2012, Trautwein et al. 2013, Schmutz et al. 2014). Such impacts have also been described on hydromorphological conditions. For example, changes in flow regime or sediment yield can affect restored reaches far downstream (Kondolf 1998, Ibisate et al. 2011, Schinegger et al. 2013). Restoration effect may thus depend on restoration extent (e.g. restored reach length) because negative effects from upstream pressures (e.g. fine sediment input) might be mitigated more effectively by larger restoration projects.

Restoration measures restore natural channel dynamics by removing bank fixation and offering extended space for the river. It is intuitively appealing to assume that this It is intuitively appealing to assume that this in turn increases mesohabitat diversity due to the formation of bars, islands, secondary channels or standing water zones, and ultimately also enhances microhabitat conditions due to more diverse flow and substrate patterns. Accordingly, large river restoration projects that enhance macrohabitat conditions should also improve meso- and microhabitats. Moreover, large restoration projects enhancing natural channel dynamics potentially also have a larger effect on meso- and microhabitat conditions compared to smaller projects. There is, however, limited empirical evidence for this assumption.

The main objective of this study was to investigate the effectiveness of restoration on different hydromorphological parameters in 20 restored sections across Europe. We specifically explored (i) whether the restoration effect depends on restoration extent described by the restored reach length and intensity as well as on the restoration measures applied, (ii) whether enhancing macrohabitat conditions in turn also improves meso- and microhabitats, i.e. restoration effects on habitats at larger scales are associated with effects at smaller scales, and (iii) which hydromorphological indicators are best suited to quantify restoration effects.

We hypothesized that

(i) the effect of restoration on hydromorphology increases with restoration extent and is higher in larger restoration projects compared to smaller projects because dynamic processes are enhanced and large-scale pressures mitigated,, independent of which type of river has been restored,

(ii) enhancing macrohabitat conditions in turn also improves meso- and microhabitats, i.e. the effect of restoration on hydromorphology at the river section scale is associated with effects on meso- and microscale habitat conditions (but not vice versa),

(iii) hydromorphological parameters that portray the re-establishment of dynamic processes are best suited to identify restoration effects.

4.2 Methods

Study sections and sampling methods

The study sections and reaches as well as the hydromorphological mapping methods are described in Annex B and Chapter 2.3.

Study design to investigate the effect of restoration extent

The extent of restoration was described by the restored reach length and restoration intensity. In addition, differences between the restoration measures applied and river types were considered.

Restoration extent: The classification of the ten pairs of restoration projects was based on restored reach length and restoration intensity, with the longer sections or more intense restoration projects denoted as R1 and the shorter sections or less intense restoration projects denoted R2 (see Chapter 1.2).

Restoration measures: Each restored section was assigned to one main restoration measure type based on the information available on the measures applied (see Annex B). As some main measures were only rarely represented by the case studies, we compared river "widening" (n=9) with all other restoration measure types (n=11). Concerning the length of the restored sections assigned to the main restoration type "widening" comprised five long restored sections (R1) and four short sections (R2).

River types: The restoration measures applied differed between river types. In gravel-bed rivers the restoration measures included removing river bed stabilization, restructuring and/or widening the riverbed and improving the lateral connectivity by reconnecting wetlands. In sand-bed rivers reconnecting old side arms and remeandering were the most significant restoration measures focusing on the main channel. In some cases the groundwater level was raised by constructing weirs and wetlands or oxbows were reconnected. Moreover, the study sections represented two main European river types: mid-sized gravel-bed rivers and mid-sized sand-bed rivers. The effect of river types on restoration effect was investigated because river types differed in respect to the main measures applied, and because the effect of restoration on river hydromorphology potentially differs between gravel-bed and sand-bed rivers.

Study design to investigate the effect of restoration on hydromorphology at different spatial scales (macro-, meso-, microscale)

Hydromorphology was mapped and assessed at different scales to investigate if enhancing macrohabitat conditions also improves meso- and microhabitats, i.e. if the effect of restoration on hydromorphology at the river section scale is associated with effects on meso- and microscale habitat conditions. Two different assessment methods were applied at two different spatial scales (see Chapter 2.3 for details): (i) a CEN compliant hydromorphological survey at the river section scale (Poppe et al. 2012) and (ii) a detailed mapping of meso- and microhabitat characteristics at the reach scale according to Jähnig et al. (2008) and Januschke et al. (2009).

The basic spatial unit for physical river habitat assessment is the reach scale (Belletti et al. 2014, REFORM 2014). This is commonly a few hundred meters in length, and in our study depended on the wetted channel width. The field survey was conducted at a larger spatial scale. These survey sections comprise several homogeneous river reaches and should reflect processes at larger spatial scales more adequately. The range of the surveyed sections is at the lower end of the segment scale which is defined in REFORM (2014) to be 1-100 km in length. Accordingly, we refer to the spatial extent of the surveys as "section scale".

Selection of hydromorphological indicators / parameters

The following indicators of the section-scale and reach-scale mapping methods were selected as parameters for the analysis presented in this chapter (Table 4-1):

Section-scale mapping: Based on the data from the section scale mapping (Poppe et al. 2012), the mean of all 14 evaluation parameters was calculated as an indicator for the hydromorphological state at the section scale (Mean_hymo – "mean hymo survey evaluation", Table 4-1). In addition, three indicators were used to describe the hydromorphological state following Jähnig et al. (2009), Kristensen et al. (2011) and Feld et al. (2014): (i) a five point ordinal-scale assessment of the occurrence of sediment depositions (gravel/sand/silt) and large woody debris (parameter "dynamic feature class"); (ii) channel width variability and (iii) the width of the riparian vegetation. These indicators were used as proxies for describing dynamic processes within the river channel, river banks and within the adjacent area of the floodplain at a wider spatial extent.

Reach-scale mapping: Based on the data from the reach-scale mapping (Jähnig et al. 2008, Januschke et al. 2009), the following indicators were used to describe the meso- and microhabitat conditions: For mesohabitat analyses we used the "Number of natural channel features" (NMchanfeat_nat), the "Number of natural dominant substrates" (NMsubstr_nat), the "Share of main channel width on total transect length (%)" (Mainchan_share), the "Shannon–Wiener Index" (SWI) as well as the "Spatial Diversity Index" (SDI; Fortin et al. 1999), the latter incorporating spatial occupancy patterns. Both diversity indices were calculated to detect channel features diversity at mesohabitat level, excluding non-natural elements for the SWI ('Artificial embankment' from channel features and 'Technolithal' from dominant substrates), and including natural and artificial ones for the SDI (Table 4-1). For microhabitat analyses five parameters were calculated following Jähnig et al. (2008): the "Number of natural microhabitats" (all natural substrate types excluding 'Technolithal') and "SWI of natural microhabitats" were used to describe substrate diversity, "SDI_micro" was applied to describe the spatial distribution of substrate diversity within transects. In addition the "Coefficients of variation of depth and flow" (CV_depth, CV_flow) were calculated.

Table 4-1 Hydromorphological parameters and indicators used in the following analyses

Abbreviation	Full name of parameter
Section scale - Morphological survey	
1_Chan_geom	Channel geometry
1_Flow_patt	Flow pattern
1_Riv_dyn	River dynamics
2_Substrate	Substrate characteristics
2_Rbed_relief	Riverbed relief
2_Hyporh_int	Hyporheic interstitial
3_Connect	Connectivity
3_Structures	Structures
4_Bank	Bank characteristics
4_Species_veg	Species composition of vegetation
4_Rip_veg	Riparian vegetation cover and age
5_Buffer	Width of riparian buffer zone
5_Species_surr	Species composition of vegetation of surroundings
5_Veg_surr	Vegetation cover and age of surroundings
Mean_hymo	Mean hymo survey evaluation (mean of 14 above parameters)
Dyn_feature_class	Dynamic feature class (occurrence of sediment deposits and/or large wood)
Width_variab	Channel width variability
Rip_veg_width	Riparian vegetation width
Reach scale - mesohabitats (occurrence of mesohabitats along transects)	
NMchanfeat_nat	Number of natural channel features
NMsubstr_nat	Number of natural dominant substrate types
Mainchan_share	Share of main channel width of total transect length in %
SWI_chanfeat	Shannon-Wiener Diversity Index (SWI) of natural channel features
SWI_substrate	Shannon-Wiener Diversity Index (SWI) of natural substrate classes
SDI_chanfeat	Spatial Diversity Index (SDI) of channel features
SDI_substrate	Spatial Diversity Index (SDI) of substrate
Reach scale - microhabitats (10-point measurements along transects)	
NMhabnat	Number of natural microhabitats
ShanMhabnat	Shannon-Wiener Diversity Index (SWI) of natural microhabitats
SDI_micro	Spatial Diversity Index of substrate
CV_depth	Coefficient of variance of depth
CD_flow	Coefficient of variance of flow

Statistical analyses

Most statistical analyses were performed in IBM SPSS Statistics Version 21. Some of the analyses (included in Annex C) were run in R version 3.1.1 and Statistica 8 software from StatSoft. All analyses were done using the response ratio (Osenberg et al. (1997) to quantify the effect of restoration (see Chapter 1.2). It was necessary to use this standardized effect size (instead of absolute values) to compare the effect of restoration on the different hydromorphological parameters because they were measured on different scales and in different units. Positive values denote a positive restoration effect, negative values a negative effect.

We performed one sample Wilcoxon signed-rank Tests to identify hydromorphological parameters with median values significantly larger than zero, indicating a general positive effect of restoration. Mann Whitney U-Tests were used for any group comparisons and Wilcoxon-Matched pairs Tests for pairwise comparisons of large (R1) and small (R2) restoration projects. For all statistical analyses a significance level of $p<0.05$ was used. Boxplots were generated to illustrate differences in effect sizes between the hydromorphological parameters. Boxes indicate the interquartile range with whiskers to one quarter of the sample. Outliers (outlying more than one-and-a-half box length) are visualized with a circle, extreme values (beyond three box lengths) with an asterisk.

First, we pooled R1 and R2 sections to identify general positive effects for the whole dataset regardless of restoration extent. We screened restoration effect values of all parameters investigated (Table 4-1) across all spatial scales (section scale, meso-, and microscale) to identify hydromorphological parameters with median values larger than zero, indicating an overall positive restoration effect.

Second, we tested our first hypothesis (restoration effect on hydromorphological parameters is higher in larger vs. smaller restoration projects) by comparing effect values of large and small restored sections. We tested for group and pairwise differences of R1 vs. R2 sections, for group differences of different main restoration measure types ("widening"; n=9 to "all other measure types"; n=11), and for group differences of sand-bed (n=8) vs. gravel-bed rivers (n=12; restored sections grouped according to the dominant substrate types). Moreover, we identified hydromorphological parameters which had median effect sizes significantly larger than zero, indicating a positive restoration effect for either R1 or R2 sections. We assumed that large restoration projects enhance dynamic processes within the river bed and riparian zone and are detectable in several floodplain features.

Third, we tested our second hypothesis that enhancing macrohabitat conditions also improves meso- and microhabitats, i.e. the effect of restoration on hydromorphology at the river section scale is associated with effects on meso- and microscale habitat conditions. For this purpose effect sizes were analysed for each pair of restoration sections (R1 vs. R2). We started with pairwise comparisons of all parameters using Mann-Whitney U-Tests analysing if there was a significant difference between restored and degraded sections. We performed correlation analyses (Spearman`s rank coefficient ρ) of all effect values to identify correlations of hydromorphological parameters across the whole data set.

Fourth, we tested our third hypothesis stating that hydromorphological parameters which portray the re-establishment of dynamic processes are best suited to identify restoration effects. Within the survey data set we analysed all those survey evaluation results per restored section (n=8 per restored section - four survey reaches at left and right bank side; see Chapter 2.3) that led to a total number of 160 for the survey data set. Within the whole hydromorphological data set the survey data set was overbalanced with 14 single parameters. Therefore, Spearman`s rank correlation coefficient ρ was calculated for the effect sizes of survey parameters. As the effect sizes showed high correlation and the Kaiser-Meyer-Olkin Test stated a significant relationship between all parameters (KMO-Index = 0.824), we performed a normed principal component analyses (PCA on correlation matrix) to determine main groups of hydromorphological survey parameters.

The PCA (rotated VARIMAX – IBM SPSS Statistics Version 21) was calculated based on eigenvalues greater than 1, which led to three components. In a first run the third component was dominated by only a single parameter. The factor scores and the scree plot were used to state main components for variance explanation. In a second run we fixed the extracted factors to two. Reasoning forward from the results of the PCA, Spearman`s rank correlation coefficient ρ was calculated for these two PCA components to determine relationships to the 14 evaluation parameters and to fix key parameters within the survey data set.

Additionally, two explanatory parameters (restoration length, years after restoration implementation) were correlated by Spearman`s rank correlation analyses to the PCA components 1 and 2 to identify possible relations.

The general pattern of the PCA components (first and second PCA axis) to the explanatory parameters was visually checked and illustrated using scatter plots. A linear regression line was added to the plots to investigate the relationship of the components to these parameters. PCA results were illustrated by component plots and scatter plots.

4.3 *Results*

Overall effect of restoration on hydromorphology (R1 and R2 pooled)

Overall (pooling short and long restored sections), restoration had a positive effect on most survey and mesohabitat parameters but only on one parameter describing microhabitat conditions (Figure 4-1, one sample Wilcoxon signed-rank test p<0.05). Effect sizes varied considerably between survey, mesohabitat and microhabitat data, i.e. between the section, meso- and microscale. Restoration had the largest effect on the mesohabitat parameter "Share of main channel width of total transect length in % - Mainchan_share" (median 0.41, 25 % percentile -0.2). Within the survey data "Width of the riparian vegetation – Rip_veg_width" (median 0.35, 25 % percentile 0.1) showed the highest positive effect, whereas effect values of microhabitat data generally showed lower effect sizes. Within this group the effect size of the parameter "Coefficient of depth variance – CV_depth" showed highest values (median 0.18, 25 % percentile 0.00). The effect of restoration was lowest on microscale substrate conditions, which are especially important for macroinvertebrates (NMhabnat, ShanMhabnat, SDI_micro).

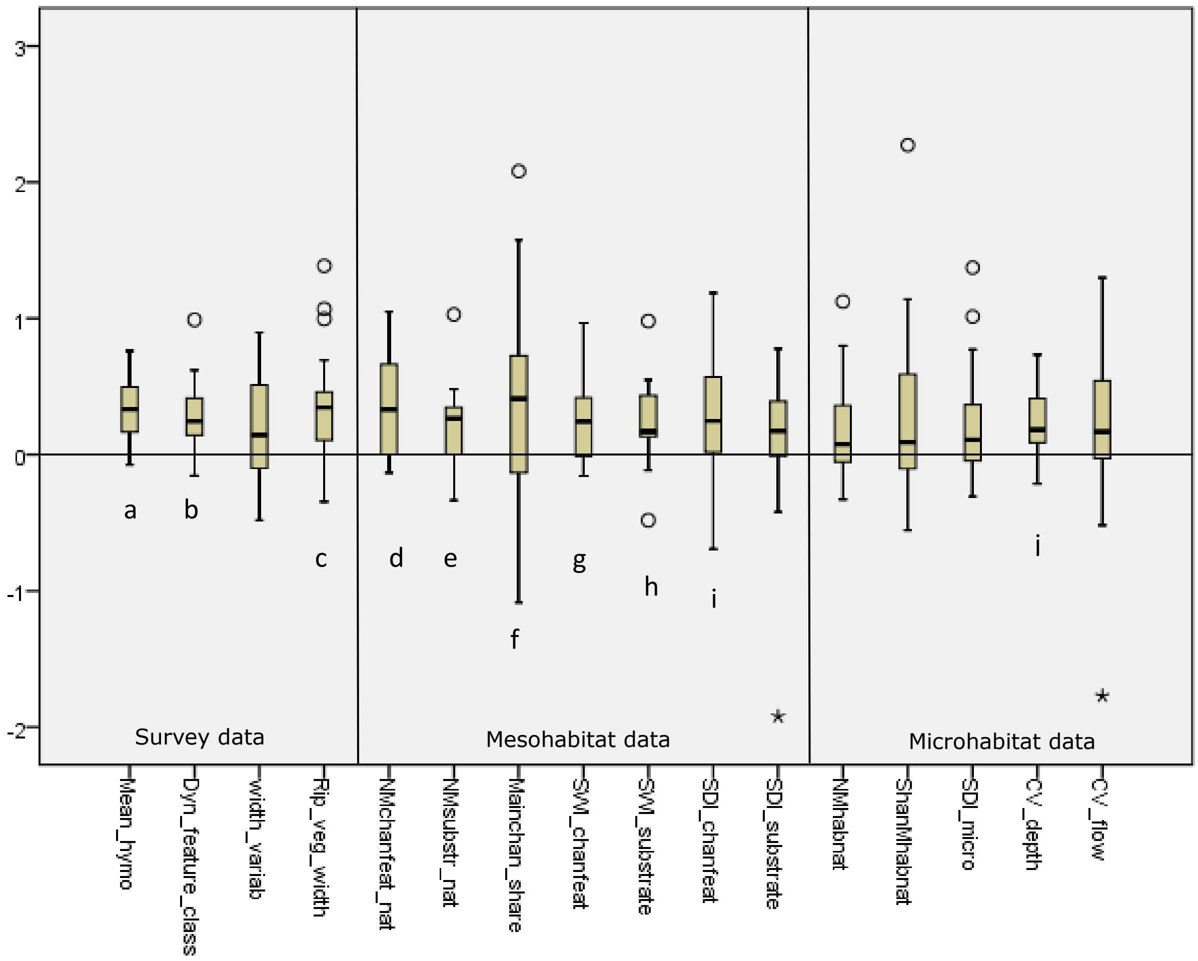

Figure 4-1 Effect sizes (ln(R/D)) of all hydromorphological parameters of the 20 case study catchments. Median values significantly larger than zero are indicated by different letters (One sample Wilcoxon signed-rank test; p<0.05).

Differences of restoration effect in large and small projects (R1 vs. R2)

There were no statistically significant differences between large (R1) and small (R2) restoration projects for none of the hydromorphological parameters investigated, neither within survey parameters nor within mesohabitat or microhabitat data (Wilcoxon-Matched Pairs test, p>0.05). The box-plots, however, revealed a tendency for a higher effect of restoration on survey and mesohabitat parameters in large compared to the small restoration projects. Differences between large and small restoration projects were not detectable or minimal for the microhabitat parameters.

Restoration had a greater effect on the following survey parameters in the large restoration projects (R1) compared to the small projects (R2), but the differences were not significant: the overall "mean of the hymo survey evaluation - Mean_hymo" and the parameter indicating "dynamic river features - Dyn_feature_class". The effect sizes of the "width of the riparian vegetation – Rip_veg_width" did not differ between R1 and R2-sections, whereas the effect sizes of the parameter "Channel width variability – width_variab" were higher in the R2 sections (Figure 4-2). One sample Wilcoxon signed-rank tests showed median values significantly different to zero (p<0.05) for three survey parameters, which proved a positive restoration effect.

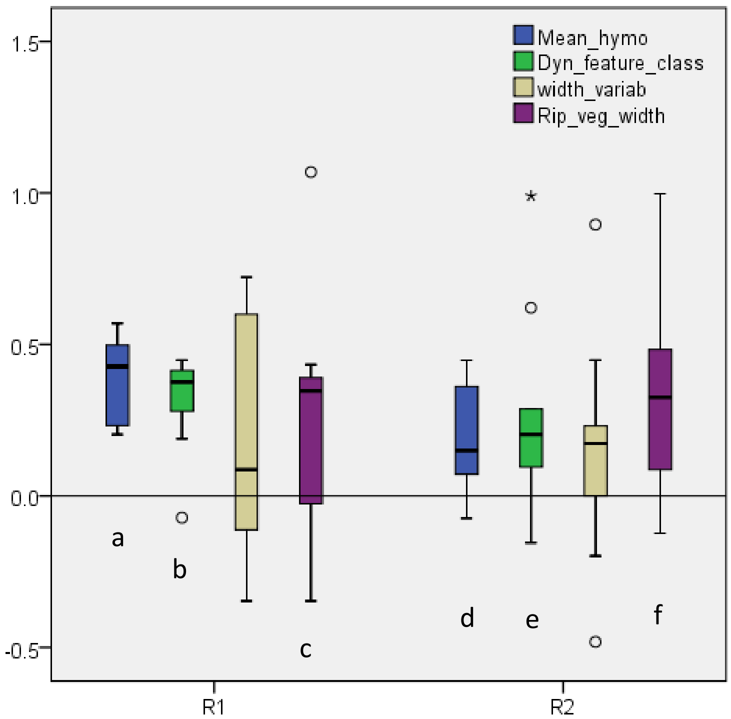

Figure 4-2 Effect sizes (ln(R/D)) of the hydromorphological survey parameters of the large (R1) and small (R2) restoration projects. Median values significantly larger than zero are indicated by different letters (One sample Wilcoxon signed-rank test; p<0.05).

At the mesohabitat level almost all parameters showed a tendency for higher restoration effect sizes in large restoration projects (R1; Figure 4-3). The sole exception was higher effect sizes of the "Shannon-Wiener Index of natural substrates - SWI_substrate" in R2 sections. The maximum value was identified in large restoration projects for the parameter "Share of main channel width of total transect length in % - Mainchan_share" (median 0.59, 25 % percentile 0.27). Six out of seven mesohabitat parameters showed median values significantly different from zero based on one sample Wilcoxon signed-rank tests ($p<0.05$) in small restoration projects. Two parameters with median values significantly different to zero were identified at R1 sections.

Remarkably, at the microhabitat level, only one parameter ("Coefficient of variance of depth – CV_depth") showed higher effect sizes in larger restoration projects (R1) compared to small projects. This microhabitat parameter showed median values significantly different from zero based on one sample Wilcoxon signed-rank tests ($p<0.05$) for R1 and R2 sections. Additionally, differences between R1 and R2 sections were less pronounced within the microhabitat data.

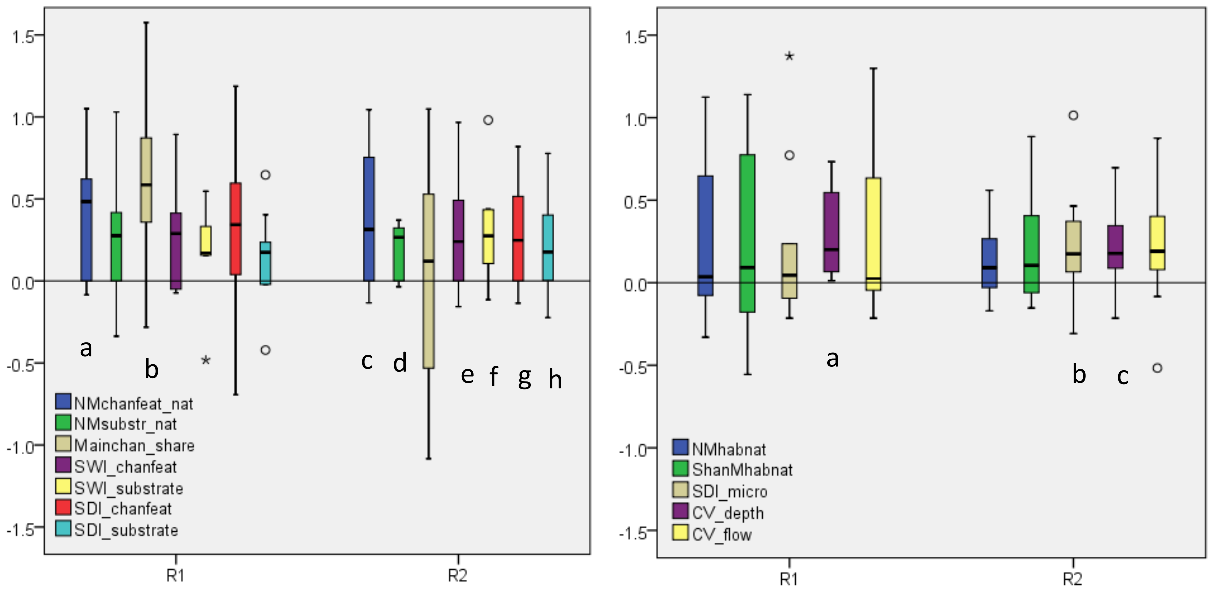

Figure 4-3 Effect sizes (ln(R/D)) of the hydromorphological parameters at mesohabitat level (left) and microhabitat level (right) of the large (R1) and small (R2) restoration projects. Median values significantly larger than zero are indicated by different letters (One sample Wilcoxon signed-rank test; $p<0.05$).

Restoration effect of different restoration measures

We found significant differences (Mann-Whitney U-Test; $p<0.05$) between the two restoration measure type groups ("widening"; $n=9$ and "all other types"; $n=11$) for many survey and mesohabitat parameters. In contrast, none of the microhabitat parameters showed significant differences (Mann-Whitney U-Test; $p>0.05$) in effect sizes between the two groups.

Figure 4-4 illustrates a higher restoration effect for all four survey parameters for the restoration measure type "widening" compared to the other restoration measure types. Three out of four survey parameters differed significantly between the restoration measure groups, whereas the parameter ("Mean hymo"), which is simply the mean of 14 evaluation parameter, did not reflect a significantly larger restoration effect in widening projects.

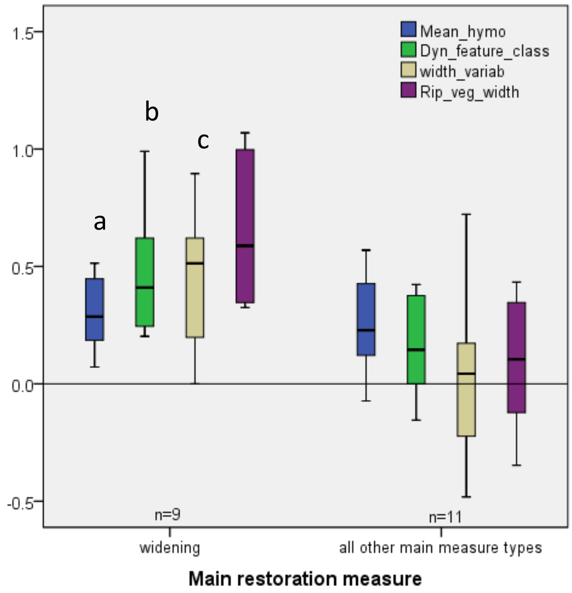

Figure 4-4 Effect sizes (ln(R/D)) of the hydromorphological survey parameters differentiated by main restoration measure type. Significant differences between the groups are indicated by different letters - Mann-Whitney U-Test (p<0.05).

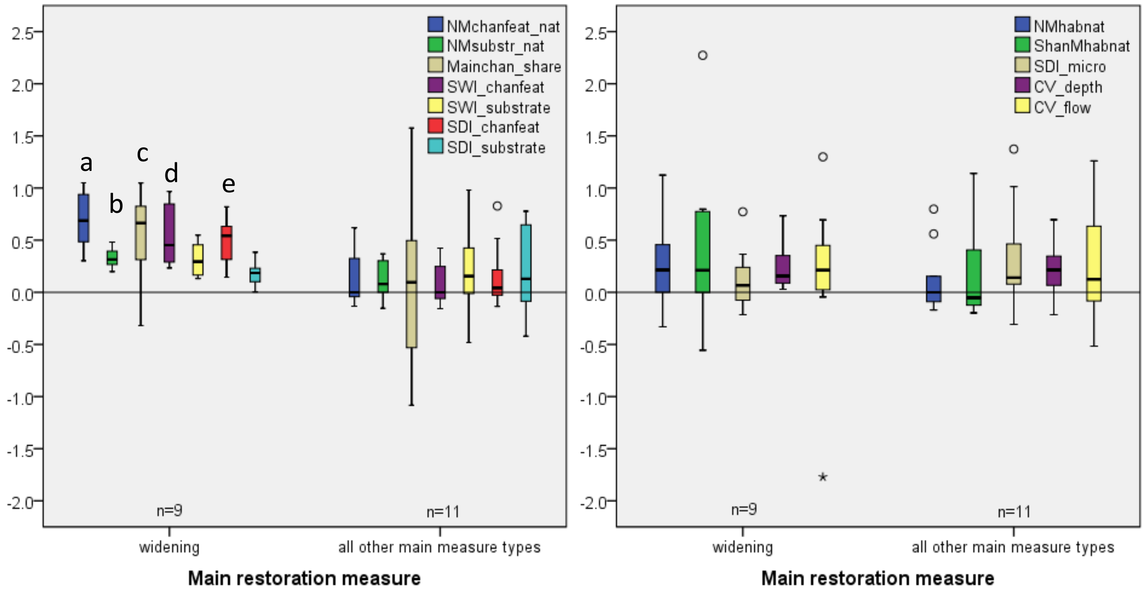

Figure 4-5 Effect sizes (ln(R/D)) of the hydromorphological parameters at mesohabitat level (left) and microhabitat level (right) by main restoration measure type. Significant differences between the groups are indicated by different letters - Mann-Whitney U-Test (p<0.05).

At the mesohabitat level, five parameters ("number of natural channel features", "number of natural substrate types", "share of main channel width", "SWI of channel features" and "SDI of channel features"; for all $p<0.05$) out of seven showed a significant difference based on Mann-Whitney U-Tests between the two measure groups. Boxplots (Figure 4-5) illustrated higher restoration effects for restoration type "widening" and lower effect values for sections where other measure types were implemented.

None of the five microhabitat parameters showed a significant difference between the two restoration type groups based on Mann-Whitney U-Tests ($p>0.05$). The boxplots (see Figure 4-5) visualized higher effect sizes for the parameters "Number of natural microhabitats – Nmhabnat" and the "Shannon-Wiener Index of natural microhabitats – ShanMhabnat" for the restoration type "widening". Both parameters indicated no restoration effect or even a slightly negative effect at sections where all other restoration measures were set. No trends were detectable for the other microhabitat parameters.

Differences of restoration effect in gravel vs. sand-bed rivers

If the investigated sections were grouped according to the dominant substrate type (gravel-bed rivers n=12, sand-bed rivers n=8), we identified significant differences (Mann-Whitney U-Test, $p<0.05$) in restoration effects between gravel-bed and sand-bed rivers for some mesohabitat parameters.

The boxplots in Figure 4-6 illustrated a higher restoration effect for gravel-bed rivers than for sandy-ones for all survey parameters, but none of the differences were statistically significant (Mann-Whitney U-Test, $p>0.05$). Highest effect sizes (median 0.45, 25% percentile -0.11) were documented for the survey parameter "width variability" for gravel-bed rivers.

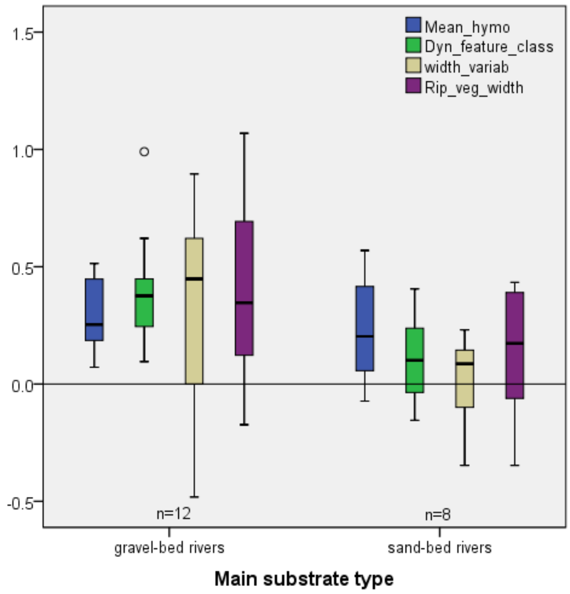

Figure 4-6 Restoration effect (ln(R/D)) on four survey parameters for different river types (gravel-bed vs. sand-bed rivers).

At the mesohabitat level, we identified significantly different effect sizes of four parameters ("Number of natural channel features – Nmchanfeat_nat", "Number of natural substrate– Nmsubstr_nat", "Shannon-Wiener Diversity Index of natural channel features - SWI_chanfeat", "Spatial Diversity Index of channel features - SDI_chanfeat") between gravel-bed and sand-bed rivers, which were mainly triggered by negative effect sizes at sand-bed rivers (Mann-Whitney U-Tests, $p<0.05$).

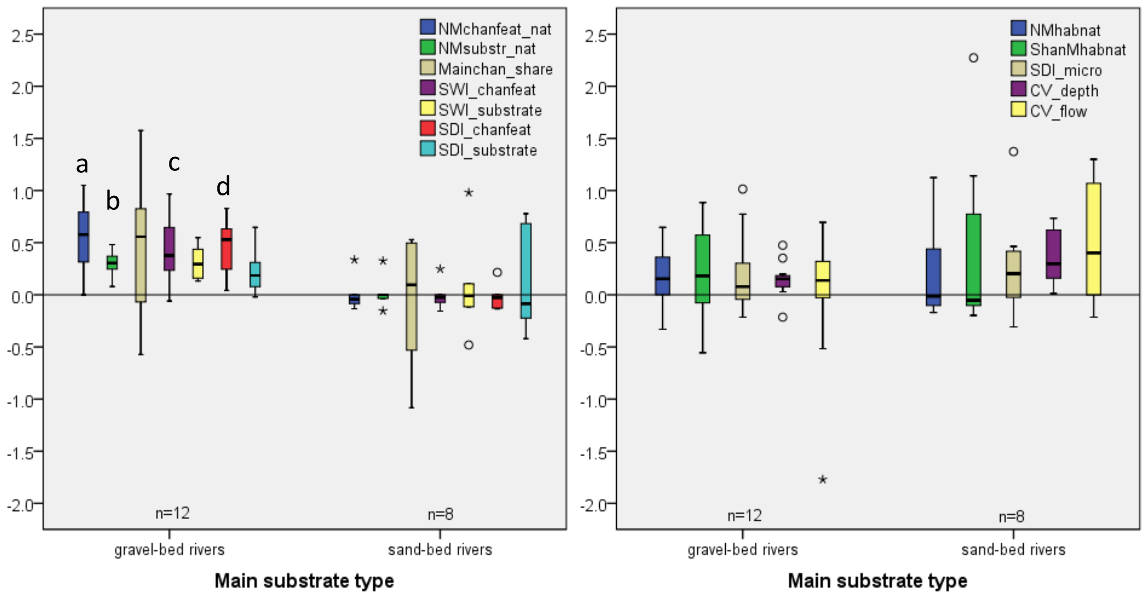

Figure 4-7 Effect sizes (ln(R/D)) of the hydromorphological parameters at mesohabitat level (left) and microhabitat level (right) differentiated by main substrate type. Significant differences between the groups are indicated by different letters - Mann-Whitney U-Test (p<0.05).

Across all microhabitat data there was no significant difference (Mann-Whitney U-Tests, p>0.05) in restoration effects between gravel-bed rivers and sandy ones.

Interestingly, the boxplots in Figure 4-7 (right) illustrated higher effect sizes of the microhabitat parameters "Coefficient of variance of depth" (CV_depth) and "Coefficient of variance of flow" (CV_flow) within sand-bed rivers.

Are section-scale hydromorphological effects associated with effects on meso- and microscale habitat conditions?

The results indicated that enhancing section-scale macrohabitat conditions also improved mesohabitat conditions but had only limited effect on microhabitats. This only partly supported our hypothesis that the effect of restoration on hydromorphology at the river section scale is associated with effects on meso- AND microscale habitat conditions. The missing relationship between section scale and microscale hydromorphological parameter does not imply that restoration had no effect on microhabitat conditions. It does, however, indicate that section-scale improvements were not necessarily associated with microscale habitat enhancement.

Jointly analysing all 20 restored sections showed that several survey parameters describing section-scale hydromorphological conditions were correlated with reach-scale mesohabitat parameters. In contrast, there was only one significant relation between survey and microhabitat parameters (Table 4-2). Moreover, none of the microhabitat parameters was correlated to one of the mesohabitat parameters.

Table 4-2 Spearman`s correlation coefficient ρ of the effect sizes (ln(R/D)) of the main hydromorphological parameters with ** p<0.01 and * p<0.05 and sample size n ranging from 16-20 depending on the availability of hydromorphological data at the 20 restored sections.

	Survey data				Mesohabitat data							Microhabitat data				
	Mean_hymo	Dyn_feature_class	Width_variab	Rip_veg_width	NMchanfeat_nat	NMsubstr_nat	Mainchan_share	SWI_chanfeat	SWI_substrate	SDI_chanfeat	SDI_substrate	NMhabnat	ShanMhabnat	SDI_micro	CV_depth	CV_flow
Mean_hymo																
Dyn_feature_class	.23															
Width_variab	.15	.29														
Rip_veg_width	.06	.38	.58*													
NMchanfeat_nat	.06	.48*	.48	.43												
NMsubstr_nat	.19	.42	.54*	.44	.79**											
Mainchan_share	.45*	.16	.27	.18	.56*	.54*										
SWI_chanfeat	.06	.41	.49	.39	.98**	.74**	.53*									
SWI_substrate	.06	.23	.26	.51*	.48	.87**	.22	.46								
SDI_chanfeat	.23	.46	.61*	.30	.84**	.67**	.54*	.84**	.27							
SDI_substrate	.16	.11	.15	-.03	.38	.64**	.54*	.40	.53*	.36						
NMhabnat	.17	.51*	.18	.29	.39	.28	.23	.40	.28	.43	.11					
ShanMhabnat	.14	.38	.25	.23	.31	.19	.22	.34	.12	.38	.12	.93**				
SDI_micro	-.18	-.18	-.10	.05	.15	.05	-.06	.20	.26	.04	.00	.61**	.63**			
CV_depth	-.31	.09	.23	.08	.19	.07	.16	.21	-.12	.19	.03	.23	.30	.31		
CV_flow	-.07	-.03	.22	.00	.07	-.17	.13	.07	-.48	.18	-.20	.34	.31	.25	.61**	

More specifically:

- A better overall hydromorphological state ("Mean-hymo") at the section scale was related to a lower share of the main channel on total transect length ("Mainchan_share").
- A higher number of dynamic channel features ("Dyn_feat_class") such as sediment deposits and large wood at the section scale increased the number of natural features and habitats at the meso- ("NMchanfeat_nat") and microscale ("NMhabnat").
- A higher width variability ("Width_variab") at the section scale increased the number of natural substrates ("NMsubstr_nat") and the spatial diversity of channel features ("SDI_chanfeat") at the mesoscale.
- Wider riparian buffers ("Rip_veg_width") at the section scale were related to a higher substrate diversity ("SWI_substrate") at the mesoscale.

Table 4-3 Effect of restoration on the hydromorphological parameters for the 20 restored sections. Significant differences on the distribution of the parameters of the restored sections compared to the corresponding degraded sections are shown (Mann-Whitney U-Test, p <0.05). Significantly higher values (+) or significantly lower values (-) are indicated.

	Survey data				Mesohabitat data							Microhabitat data					Summary		
	Mean_hymo	Dyn_feature_class	Width_variab	Rip_veg_width	NMchanfeat_nat	NMsubstr_nat	Mainchan_share	SWI_chanfeat	SWI_substrate	SDI_chanfeat	SDI_substrate	NMhabnat	ShanMhabnat	SDI_micro	CV_depth	CV_flow	Suvrey data	Mesohabitat data	Microhabitat data
AT_R1	+	+	+		+	+	+	+	+	+		+	+	+	+	+	+	+	+
CH_R1	+			+	+		+	+		+							+	+	
CZ_R1	+				+	+	+	+	+			-	-				+	+	-
DK_R1	+									-					+	+	+		+
DL_R1	+		+		+	+	-	+	+	+		+	+		+	+	+	+	+
DM_R1	+				+	+	+	+		+							+	+	
FI_R1	+			-													+		
NL_R1	+	+					+				-	+	+	+	+	+	+	+	+
PL_R1	+						+									+	+	+	+
SE_R1	+		+		+	+	+	+		+					+		+	+	+
AT_R2				+	+	+		+	+	+		+			+	+	+	+	+
CH_R2	+				+	+	+	+		+						+	+	+	+
CZ_R2	+	+	+	+	+	+	+	+		+							+	+	
DK_R2																			
DL_R2							+							-	+			+	+
DM_R2	+				+	+	-	+		+		+	+				+	+	+
FI_R2							+								+	+		+	+
NL_R2					+		-	+	+	+	-			+	+	+		+	+
PL_R2	+						+				+				+	+	+	+	+
SE_R2	+	+										+	+	+			+		+

The detailed analysis of the 20 single restored sections revealed that microscale habitat diversity was significantly improved in 11 out of the 16 restored sections where section-scale conditions were enhanced. However, substrate composition, which is especially important for macroinvertebrates, was significantly improved in only 6 out of the 16 restored sections and in only 7 out of the total 20 restored sections investigated (Table 4-3).

Detailed results of all case study sections are given in the Annex C.

Identifying key hydromorphological parameters

Section scale

The effect sizes of the 14 hydromorphological survey parameters were highly correlated (Table 4-4). This hampered identifying single parameters that are best suited to quantify restoration success. Nonetheless, a Principal Component Analysis revealed that two groups of parameters can be distinguished which quantify the effect of restoration on aquatic and riparian/terrestrial hydromorphology, respectively (Figure 4-8). The two components explain more than half (57.2%) of the total variance in effect size values:

- Component 1: Effect sizes of aquatic habitat parameters within the river channel (parameter groups P1_ - P3_: factor loading to Component 1 ranges from 0.849 to 0.717).
- Component 2: Effect sizes of terrestrial vegetation and habitat parameters describing river banks and adjacent floodplain areas (parameter group P4_ and parameter group P5_: factor loading to Component 2 ranges from 0.787 to 0. 631).

The survey parameter "bank characteristics" (4_Bank) was dedicated to the aquatic parameter group but weakly loaded, visually isolated from the other aquatic parameters (Figure 4-8; factor loading to Component 1 of 0.667 and to Component 2 of 0.443).

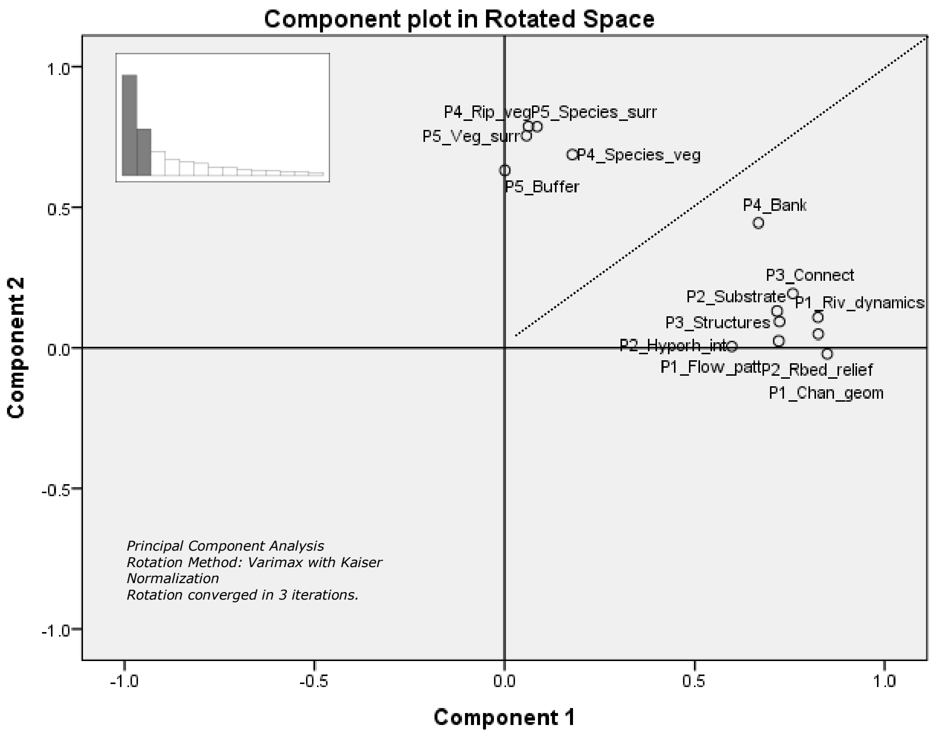

Figure 4-8 Component Plot of PCA on restoration effect values of 14 survey parameters (Parameter groups 1_ - 5_ – see Table 4-1). All parameters transformed to effect sizes (ln(R/D)). The small insert bar chart shows the corresponding eigenvalues of the analysis with the main axes` eigenvalues indicated in grey.

The hydromorphological survey parameters which were related best to the two components, and hence which are best suited as a single parameter to quantify restoration effect, were river bed relief (2_Rbed_relief) for the aquatic component 1 and riparian vegetation cover and age (4_Rip_veg) for the riparian/terrestrial component 2 (Table 4-4).

Table 4-4 Spearman`s correlation coefficient ρ of the effect sizes (ln(R/D)) of all 14 hydromorphological survey parameters as well as from PCA generated Component 1 (PCA extracted "Aquatic habitat parameter") and Component 2 (PCA extracted "Terrestrial habitat parameter") with ** p<0.01 and * p<0.05.

	Survey parameters														PCA axis	
	1_Chan_geom	1_Flow_patt	1_Riv_dyn	2_Substrate	2_Rbed_relief	2_Hyporh_int	3_Connectvty	3_Structures	4_Bank	4_Species_veg	4_Rip_veg	5_Buffer	5_Species_surr	5_Veg_surr	Component1 (aquatic)	Component 2 (terrestrial)
1_Chan_geom																
1_Flow_patt	.47**															
1_Riv_dyn	.52**	.67**														
2_Substrate	.54**	.40**	.46**													
2_Rbed_relief	.75**	.48**	.55**	.51**												
2_Hyporh_int	.34**	.18*	.36**	.50**	.46**											
3_Connectvty	.44**	.50**	.64**	.30**	.54**	.27**										
3_Structures	.53**	.50**	.59**	.34**	.50**	.22**	.64**									
4_Bank	.52**	.44**	.55**	.43**	.53**	.26**	.59**	.56**								
4_Species_veg	.16*	.07	.16	.30**	.13	.24**	.21*	.15	.47**							
4_Rip_veg	.15	.13	.15	.22**	.08	-.01	.14	.21*	.45**	.68**						
5_Buffer	-.03	.11	.14	.12	.04	-.03	.16	.09	.31**	.41**	.39**					
5_Species_surr	.18*	.16	.19*	.17*	.11	-.06	.15	.09	.26**	.30**	.42**	.38**				
5_Veg_surr	.09	.17*	.22**	.08	.12	.04	.18*	.10	.24**	.25**	.38**	.36**	.71**			
Component 1 (aquatic)	.81**	.71**	.80**	.65**	.83**	.54**	.72**	.74**	.68**	.17*	.10	.02	.11	.10		
Component 2 (terrestrial)	.03	.12	.17*	.20*	.03	.00	.19*	.13	.47**	.68**	.78**	.70**	.71**	.71**	.05	

Besides identifying those hydromorphological parameters which are best suited to quantify restoration success, we also identified project characteristics which affect restoration success by investigating relationships of both components with explanatory parameters (Table 4-5). The effect of restoration on aquatic and terrestrial habitat parameters increased with project age (Figure 4-9). Restoration length showed no effect either on aquatic or on terrestrial parameter groups.

Table 4-5 Spearman`s correlation coefficients ρ of Component 1 ("aquatic habitat parameter") and Component 2 ("terrestrial habitat parameter") performed on two explanatory parameters with ** p<0.01 and * p<0.05.

	Component 1 (aquatic habitat parameter)	Component 2 (riparian/terrestrial habitat parameter)
Restoration length	**0.091**	**0.066**
Years after restoration implementation	**0.273 ** **	**0.193 * **

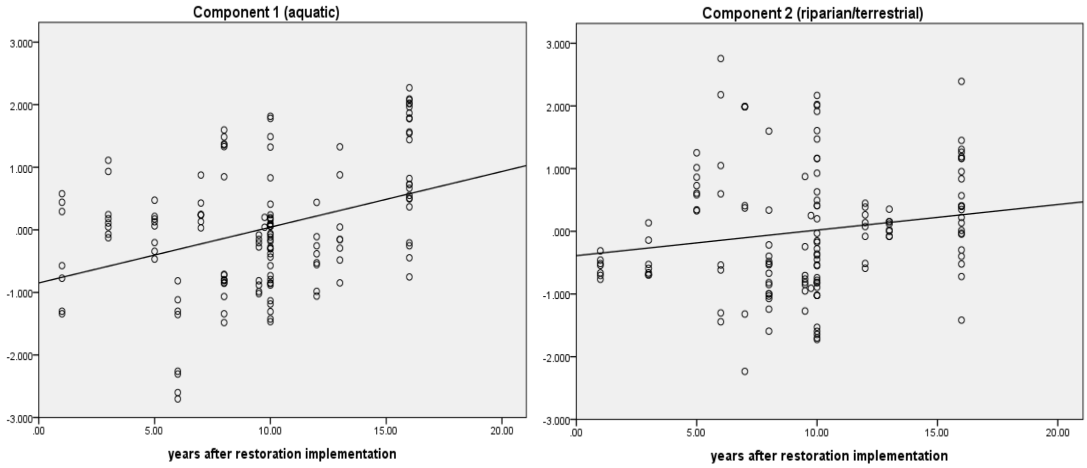

Figure 4-9 Correlation of Component 1 (left - aquatic habitat parameter - Spearman`s ρ=0.273 p=0.00; $R^2_{(lin)}$=0.13) and Component 2 (right – riparian/terrestrial habitat parameter - Spearman`s ρ=0.193 p=0.02; $R^2_{(lin)}$=0.03) to the explanatory parameter "years after restoration implementation"

Reach scale

The reach-scale correlation matrix (Table 4-2) revealed high correlation within the mesohabitat and microhabitat data sets. Nonetheless, analyses of single restored sections showed a considerable variation of effect sizes within the data sets (Table 4-3). At the mesohabitat scale the parameter "Share of main channel width of total transect – Mainchan_share" as well as the "Number of natural channel features – NMchanfeat_nat" proved positive effects in many cases. Within the microhabitat parameters, restoration effects were less obvious, but the "Coefficients of variances of depth and flow – CV_depth and CV_flow") revealed positive effects. Especially for sand-bed rivers, positive restoration effects were determined within the microhabitat data set additionally to the other hydromorphological parameters.

4.4 Discussion

The effect of restoration on hydromorphology was mapped and assessed using the same methods in all 20 restoration projects investigated. Nevertheless the twenty catchments in nine European countries differed in respect to their own restoration history, and were individually affected by large-scale pressures. At most restored sections, the channel and floodplain heterogeneity were initially increased by specific measures.

Although this makes drawing a general conclusion difficult, we were able to identify some general trends Overall, we found that restoration increased habitat diversity through changes in channel morphology (compared to the degraded sections upstream).Nonetheless, we identified considerable differences in restoration effect sizes between sections, restoration measures, river types and spatial scales.

Differences in restoration effect

Restoration effect sizes showed highest values within the survey and mesohabitat parameters, while the effect of restoration on aquatic microhabitats was less pronounced. The "Number of natural microhabitats" showed the lowest effect sizes, similar to the findings of Jähnig et al. (2009). Moreover, the effect of restoration was also especially low on microscale substrate composition, which is of special importance for macroinvertebrates (diversity of microhabitats and spatial diversity of substrates). This might explain why we found no effect of restoration on aquatic macroinvertebrates (see Chapter 5).

We had to reject our first hypothesis that the effect of restoration on hydromorphology increases with restoration extent and is higher in larger restoration projects compared to smaller projects by enhancing dynamic processes and mitigating large-scale pressures. Overall, the effect of restoration on hydromorphology did not significantly differ between large (R1) and small (R2) restoration projects for any of the hydromorphological parameters investigated. This result is consistent with the findings of the overview analysis for the aquatic and floodplain-inhabiting organism groups (Chapter 3, Hering et al. in prep). However, there was a tendency for higher effect sizes of the survey and mesohabitat parameters. The effect sizes of three out of four analysed survey parameters were higher in large restoration projects, but the difference was not significant. At the mesohabitat scale, all parameters showed higher restoration effect in large restoration projects, indicating a higher effect on mesohabitat diversity. Especially the dominance of the main channel was significantly reduced in large restoration projects. Other channel features such as islands, banks and bars became more frequent, and the restoration measures also increased heterogeneity along the cross section of the river. At the microhabitat scale, only one out of five parameters showed higher effect sizes in larger restoration projects.

Differentiating the data set according to the main restoration measure types revealed a decrease of restoration effect from the restoration measure type "widening" as opposed to all other restoration measure types. The measure type "widening" comprises mainly the removal of bank enforcement and the creation of secondary channels which initialize dynamic processes and enables higher diversity of flow velocities and depths. These processes were statistically proven with a high correlation to the occurrence of unvegetated sediment banks and / or islands with early successional stages of vegetation as well as woody debris within the reaches (survey parameter "dynamic features class"). Effect sizes of survey and mesohabitat parameters differed significantly between the restoration type riverbed "widening" and the other measure types. This result supports the call for restoration measures at larger extent (Bernhardt & Palmer 2011, Mueller et al. 2014, Schmutz et al 2014) and going beyond the instream scale.

When we grouped the whole data set according to the dominant substrate type, the box-plots illustrated a higher restoration effect on survey and mesohabitat parameters for gravel-bed rivers than for sandy ones. The differences were significant at the mesohabitat level. Some meso- and microhabitat parameters showed negative restoration effect sizes for sand-bed rivers and led to a further question: did we measure appropriate parameters for this river type? Brierley et al. (2010) stated that, in gravel-bed systems, heterogeneity is shaped by complex sediment and flow interactions, which are more strongly reflected in the applied assessment methods. In contrast, heterogeneity in sand-bed rivers is far more dependent upon riparian vegetation and the presence of wood. Remarkably, the effect sizes for sand-bed rivers were highest at the microhabitat scale for the parameter "variances of flow". This result supports additionally the urgent need for carefully designed restoration and monitoring programs for sand-bed rivers.

Physical habitats are most strongly determined by processes at larger spatial scales (Frissel et al. 1986, Brierley & Fryirs 2005, Habersack & Piegay 2007). Changes in sediment and hydrological regime should be included in restoration monitoring. This implies comprehensive catchment data sets, which were not available in our study. Temporally changing hydrological conditions due to flow regime variability between years must be considered (Brierley & Fryirs 2005, Palmer et al. 2010, Belletti et al. 2014) and should be incorporated in future monitoring designs. Habitat heterogeneity in the floodplain, for example, is induced by floods, and the maintenance of dynamic floodplain ecosystems over time depends on sediment relocation by floods (Tockner et al. 2009, Habersack & Kreisler 2013).

The time between the implementation of restoration measures and the field sampling in 2012/2013 ranged from 1-16 years. There was a discernible time effect in the data set. This demonstrates larger effect sizes for the aquatic as well as for the terrestrial habitat parameters at sections with higher restoration project age.

Overall, our findings revealed that restoration effects on hydromorphological parameters gradually decreased from section scale to reach scale and from mesohabitat to microhabitat level.

Relation of effect sizes of hydromorphological parameters

Our second hypothesis (enhancing macrohabitat conditions in turn also improves meso- and microhabitats) was only partly supported by detailed analyses of each restoration project.

Positive restoration effects were related across the hydromorphological data set, indicating that enhanced dynamic processes could be identified by parameters at the section scale and further proven by positive effects on mesohabitat parameters. This was not always the case with microhabitat parameters.

As stated above, restoration in general had the lowest effect on microhabitat parameters but there were large differences between the restoration projects. For example, restoration indeed had a positive effect on microhabitat parameters in two large restoration projects in sand-bed rivers (NL_R1, DL_R1) but this was only partly related to significant effects on survey or mesohabitat parameters.

Those restored sections where only in-stream measures were implemented showed the lowest restoration effects on hydromorphological parameters at all analysed scales. In one of these projects, restoration had no effect on none of the hydromorphological parameters at any of the spatial scales considered (DK_R2 – Denmark/Stora - sand-bed river- main restoration measure: gravel introduction). This is consistent with findings of Mueller et al. (2014), who reported only immediate short-term improvements of habitat condition if only minor or single measures were implemented (for example if addition of gravel was the only restoration technique at restored reaches).

Key hydromorphological parameters

Our third hypothesis (hydromorphological parameters mirroring the re-establishment of dynamic processes are best suited to identify restoration effects) was tested mainly by statistical analyses on the survey parameter set. The 14 morphological evaluation parameters investigated in the survey are commonly used and CEN-compliant (CEN 2002). Very similar approaches can be found throughout Europe and worldwide (Raven et al. 1997, NERI 1999, LAWA 2000, Parsons et al. 2004, EPA 2004, Belletti et al. 2014).

The correlation matrix revealed that most of the survey evaluation parameters were highly correlated to each other. Using PCA we identified two main components within the whole data set: Component 1 of aquatic habitat parameters and Component 2 of terrestrial riparian and floodplain vegetation parameters.

Especially Component 1 revealed high restoration effects for many restored sections that were not identified with the "Mean hymo survey evaluation" (Mean_hymo), which is simply the mean of all 14 parameters.

Based on our findings, hydromorphological key indicators for identifying restoration success should include parameters at larger spatial scales that consider or reflect processes such as bank erosion and channel adjustments.

A further descriptive morphological parameter - "dynamic features class" - which incorporates channel patterns showed high correlation to the effect sizes of aquatic habitat parameters. We analysed unvegetated dynamic patches and/or bars with early successional stages of vegetation that prove renewing successional processes were present. But still, we used the occurrence or absence of specific features in a static visual assessment as an indicator of processes. The elaboration of parameters mirroring dynamic processes is still essential.

Corresponding to the needs of a morphological assessment (Belletti et al. 2014), our study went beyond investigating and analysing only the river channel and the riparian zones. We also included parameters within the riparian buffer zone and the adjacent area.

In many of the restored sections the "width of riparian vegetation" was a good indicator of a high positive restoration effect. However, terrestrial habitat parameters of the adjacent floodplain showed significant restoration effects in only a few cases.

There is a further need to develop terrestrial parameters to assess restoration effects adequately towards the lateral dimension. The survey parameters on riparian and terrestrial vegetation used in this study did not reflect restoration effects sufficiently within the floodplain. For this purpose a restoration monitoring should incorporate the use of digital maps, remotely sensed data and GIS analyses. These approaches allow larger spatial scales of analyses focusing on habitat features and the vegetation composition of the floodplain (Smith et al. 2013).

At the mesohabitat scale, the parameter "Share of main channel width of total transect – Mainchan_share" as well as the "Number of natural channel features – "NMchanfeat_nat" proved positive effects in many cases. Diversity Indices such as "Shannon-Wiener Index or Spatial diversity index" additionally reflect positive effects.

Within the microhabitat parameters, restoration effects were less obvious, but the "Coefficients of variances of depth and flow" (CV_depth and CV_flow) revealed positive effects. Especially for sand-bed rivers, we determined positive restoration effects within the microhabitat data set beyond the other hydromorphological parameters.

Additional to parameters at larger spatial scales, diversity indices and parameters related to the occurrence of natural channel features and their extent, as well as variances of flow and depth at the microhabitat level, reflect morphological diversity at the reach scale.

These results revealed the need to incorporate adequate hydromorphological parameters at different scales for restoration monitoring and highlighted the demand to consider different river types as well as measure types. These findings support the conclusions of Pander & Geist (2013), who underlined that parameters at all scales should be included in post-restoration monitoring schemes.

Morphological characteristics and conditions help explain organism distributions and play a key role for understanding ecosystem functioning. Accordingly, our analyses provide the basis for establishing links between morphology, ecological condition and communities.

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5. Macroinvertebrates

5.1 *Introduction*

Many river restoration projects have been carried out in Europe, aiming at restoring natural flow patterns and enhancement of habitat heterogeneity to increase biodiversity. Nonetheless, after decades of restoration their remains a lack of evidence for strong and long-term positive ecological effects of these measures on macroinvertebrates (e.g. Palmer et al. 2010; Feld et al. 2011; Friberg et al. 2014). Even where the ecological effects of hydromorphological restorations have been scientifically assessed, effects on invertebrates often seem weak. This could be the result of, amongst other factors, an insufficient extent of restoration, or a mismatch between the measures applied and the requirements of the targeted organism groups. The scale of most restoration projects carried out to date has been small in comparison to total catchment size, generally not exceeding a river length of several kilometers. The type of restoration measures applied in these reaches varies considerably, ranging from measures aiming at instream habitat improvements to channel widening and floodplain reconnection.

To improve understanding of the effectiveness of hydromorphological restoration measures on macroinvertebrates, a standardized field study was carried out in catchments of mid-sized lowland and mountain rivers throughout Europe. We investigated ten pairs of one large (R1) and a similar but small (R2) restoration project. In contrast to the R1-sections, the R2 sections were shorter, and/or restoration was performed with less “intensity” (i.e. a lower intensity of restoration effort, fewer parameters addressed, etc.). The restoration effect was quantified by comparing each restored river section to a nearby non-restored, i.e. still degraded section (see Chapter 1.2 for more information on the general study design). Multiple metrics characterising macroinvertebrate community diversity, functional traits and taxonomic composition were assessed, including total taxon richness and diversity, richness of flow indicators (rheophiles) and indicators of habitat heterogeneity (diversity of macroinvertebrate habitat preferences). These metrics were related to (i) changes in habitat availability and/or quality as a result of restoration, (ii) differences in restoration type and extent.

It was expected that if hydromorphological river restoration results in either a more stable flow regime or an increase in the number or heterogeneity of habitat types, this will have positive effects on macroinvertebrate indicators of flow and habitat heterogeneity. Because total richness and diversity are more general measures, which could be influenced by a variety of environmental or biological factors, we did not expect effects for these metrics. Furthermore, we expected that the specific restoration measures applied, the time since restoration, and the size or extent of the restored section would all influence the magnitude macroinvertebrate responses.

5.2 *Methods*

Study sections and sampling methods

The study sections and reaches as well as sampling methods for the macroinvertebrates are described in Annex B and Chapter 2.4. Macroinvertebrate samples were taken in 19 degraded and 19 restored sections out of the 20 paired degraded / restored sections.

Diversity indices, ecological preferences and effect sizes

As not all macroinvertebrate specimens collected were identified to the same taxonomic level (for example early instars of insects), an adjustment procedure was applied (e.g. Vlek et al., 2004). This procedure reduced bias in the subsequent analyses due to differences in taxonomic resolution by grouping to a higher taxonomical level (Schmidt-Kloiber & Nijboer, 2004).

Total macroinvertebrate taxon richness and Shannon Wiener diversity (Shannon & Weaver, 1949) were calculated for each section sampled based on the adjusted data. Current and habitat preferences were derived from the freshwaterecology.info database (Schmidt-Kloiber & Hering, 2012). For each sample the number of taxa classified as 'rheophilic' or 'rheobiont', was counted. Furthermore, the diversity of habitat preferences in a section was calculated based on the Shannon Wiener diversity of the sum of all taxon preference scores per sample. Finally, to determine overall community change, the Euclidian distance between each of the paired restored-degraded sections was calculated.

To quantify the effects of restoration on macroinvertebrate metrics, the effect sizes for total richness and diversity, number of taxa preferring a high current velocity and diversity of habitat preferences were calculated. We used (i) the pairwise calculation of the difference between each pair of restored and degraded section, and (ii) a modified version of the response ratio Δr developed by Osenberg et al. (1997). The original formula given by Osenberg et al. (1997) is:

$$\Delta r = \ln\left(\frac{X_R}{X_D}\right),$$

whereas X_R is the species richness or diversity of the restored section and X_D of the non-restored section. Thereby, values > 0 denote a positive effect (e.g. increase of richness or diversity), and negative values a negative effect. This formula was not appropriate for our data (e.g., for diversity or the proportion of species with habitat preferences) as we had 0-values for the degraded sections and could, therefore, not calculate the response ratio. Instead, we calculated a modified response ratio Δr_m according to the following formula:

$$\Delta r_m = \ln\left(\frac{(1+X_R)}{(1+X_D)}\right).$$

Environmental variables

Several environmental variables related to river, habitat and restoration project characteristics were used (Table 5-1). River characteristics comprised the altitude of the restored reach, slope of the restored channel, mean discharge, mean channel width and overall bed coarseness based on the dominant substrate of the riverbed. Project characteristics were the extent of restoration (large vs. small restoration projects and the type of restoration measure applied). Two groups of restoration measures were distinguished: measures which primarily aimed at widening (usually affecting aquatic, semi-terrestrial, and terrestrial areas) and projects which applied other, less extensive measures mainly affecting the river channel itself (instream measures, flow restoration, remeandering, anastomosing). Habitat characteristics included the mean current velocity of the river section, the number of number of natural substrates present (excluding technolithal) and its diversity based on the Shannon-Wiener index. Besides the substrate diversity its spatial arrangement was included by calculating the Spatial Diversity Index (SDI; Fortin et al., 1999, Jähnig et al. 2008).

Table 5-1 Environmental variables classified according to river, project and habitat characteristics of the 200-m river sections sampled.

Variable class	Variable
River characteristics	Altitude (m above sea-level)
	Slope (%)
	Discharge (m^3/s)
	River width (m)
	Bed coarseness (cobbles-gravel or sand bed)
Project characteristics	Restoration extent (large vs. small restoration projects)
	Restoration type / measure (widening, other)
	Time since restoration (year)
Habitat characteristics	Current velocity (mean value m/s)
	Presence of natural substrates (total number)
	Natural substrate diversity (Shannon-Wiener index)
	Spatial distribution of substrate diversity (Spatial Diversity Index)

Data analysis

First, it was tested if there was an overall positive effect of restoration on macroinvertebrate taxon richness and diversity by comparing the richness and diversity of all restored (R) and all degraded (D) river sections (group and pairwise comparison of R vs. D). Second, it was tested if the effect of restoration depended on restoration extent by comparing richness and diversity of all large (R1) and all small (R2) restoration projects using absolute values (group and pairwise comparison of R1 vs. R2). Furthermore, effects sizes based on richness and diversity were compared, expressed as the absolute difference between the restored and degraded sections as well as the response ratio modified after Osenberg et al. (1997). Both an overall comparison of effect sizes and a comparison taking differences in river type into account were carried out. Third, we tested if effect sizes differ between projects which mainly aimed at river widening (usually affecting aquatic, semi-terrestrial, and terrestrial areas) and projects which applied other, less extensive measures mainly affecting the river channel itself (instream measures, flow restoration, remeandering, anastomosing, similar to the grouping of measures in Chapter 8 on ground beetles). Significance testing was carried out in IBM SPSS for Windows (version 19) using Mann Whitney U tests, t-tests, Kruskal-Wallis tests and One-Way ANOVAs.

The next step was to investigate in more detail which restoration and habitat characteristics, alone or in combination, were best explaining the variation in effect sizes for all metrics used in the study. To parameterize the typological differences among European rivers, we combined river characteristics (bedtype, slope, altitude, discharge, width) as superordinated variables into one parameter. Thereby, we extracted a composite descriptor using principal components analysis (PCA) which was used for further analyses (Table 5-2). Principal components that explained a significant non-random part of the variation were retained (broken-stick model; Jackson 1993), which in this case was only principal component 1 (eigenvalue = 13.7, Broken-stick eigenvalue = 10.2, 61.3% of total variance explained). Correlations of each parameter with the first principal component (PC-1) were calculated to derive its main descriptors, which turned out to be a combination of coarseness of the riverbed, altitude and slope. Sample scores of the sites on the significant principal component were used as a new quantitative variable in the subsequent ordinations, which here could be defined as the hydraulic gradient, ranging from coarse-bed, high gradient rivers to low gradient rivers with a sand bed. Habitat characteristics were represented by their Osenberg response ratios.

Table 5-2 Results of the principal component analysis. Based on the loadings of each variable on the significant principal components (PC) expressed as Pearson correlation coefficients, it main descriptors (r > 0.8; in bold) were determined; significance of the principal components: * significant, n.s. not significant.

River characteristics and PC parameters	Pearson correlation coefficient (r)		
	PC-1	PC-2	PC-3
Altitude (m above sea-level)	**-0.8**	-0.5	0.2
Slope (%)	**-0.8**	0.6	-0.0
Discharge (m^3/s)	-0.5	0.3	0.7
Channel width (m)	-0.5	0.3	0.6
Bed coarseness (gravel vs. sand-bed)	**-0.8**	-0.1	-0.5
Eigenvalue	13.7*	4.4^{ns}	2.7^{ns}
Broken-stick eigenvalue	10.2	5.7	3.5
% of total variance explained	61.3	19.7	11.9

Spearman rank order correlation was used to investigate bivariate relationships between the response variables and different predictors. Subsequently, the relationships between the effect sizes of the macroinvertebrate metrics and a selection of river, habitat and restoration project characteristics (Table 5-3) were analysed using redundancy analysis (RDA). To be able to determine which part of the variation in effect sizes can uniquely be attributed to changes in certain environmental variables and which part is shared with other variables, variance partitioning was applied. This is important, since it enables us to show to what extent the effects of the different groups of variables are related to each other. Forward selection (Monte Carlo permutation test, 9999 permutations, P values Holm corrected) was used to retain only those variables which significantly contributed to the variance explained by each of the groups. Ordinations were carried out using Canoco 5.03 (Ter Braak & Šmilauer, 2012).

Table 5-3 Classification and description of parameter types used in the redundancy analyses to analyse the relationship between macroinvertebrate metrics (response variables) and environmental characteristics (explanatory variables); R = restored reach, D = degraded reach; response ratio refers to the modified response ratio after Osenberg et al. (1997).

		Parameter type	Parameter description	Value calculated as
Response variables	Macroinvertebrate metrics	Richness	Total taxon richness	Response ratio
		Diversity	Shannon-Wiener index value	Response ratio
		Community composition	Taxon composition samples	Euclidian distance between R and D
		Flow preference	Number of rheophilic + rheobiont taxa	Response ratio
		Habitat preference	Habitat preference diversity (Shannon–Wiener index)	Response ratio
Explanatory variables	River characteristics	River characteristics	PC-1 (hydraulic gradient)	R
	Restoration characteristics	Restoration	Restoration extent (large/small)	R
		Restoration	Restoration measure (widening/other)	R
		Restoration	Time since restoration (year)	R
	Habitat characteristics	Flow	Mean current velocity	Response ratio
		Habitat richness	Number of natural substrates (#)	Response ratio
		Habitat diversity	Natural substrate diversity (Shannon–Wiener index)	Response ratio
		Habitat diversity	Spatial distribution of substrate diversity (Spatial Diversity Index)	Response ratio

5.3 Results

Overall effect of restoration on macroinvertebrates (R1 and R2 pooled)

Overall (pooling large and small restoration projects), there was no significant difference between restored and degraded sections in respect to taxon richness or diversity (Mann-Whitney U test $p > 0.41$, $n = 19$). Mean richness was about 35 taxa and mean Shannon-Wiener diversity about 2.3 in both degraded and restored sections (Table 5-4).

Moreover, restoration had no overall positive effect on richness or diversity when restored sections were compared to the corresponding degraded sections for both methods used to quantify restoration effect size (the difference of the 19 pairs of restored and corresponding degraded sections based on absolute values as well as the relative response ratios), i.e. mean effect sizes were not different from zero (t-test, $p > 0.27$). Variability was especially high for macroinvertebrate richness, demonstrating that some projects indeed increased the number of taxa but other even lead to a substantial decrease in species richness.

Table 5-4 Macroinvertebrate richness and diversity in restored and degraded sections of rivers, for all rivers combined and for rivers which differ in restoration extent.

	Taxon richness		Shannon-Wiener diversity		n
	Mean	SD	Mean	SD	
R1 and R2 pooled					
R	35.3	8.3	2.24	0.35	19
D	35.1	11.0	2.36	0.48	19
Large projects					
R1	34.1	9.3	2.30	0.34	10
D1	33.4	11.4	2.25	0.55	10
Small projects					
R2	36.6	7.3	2.17	0.37	9
D2	36.9	10.9	2.49	0.37	9

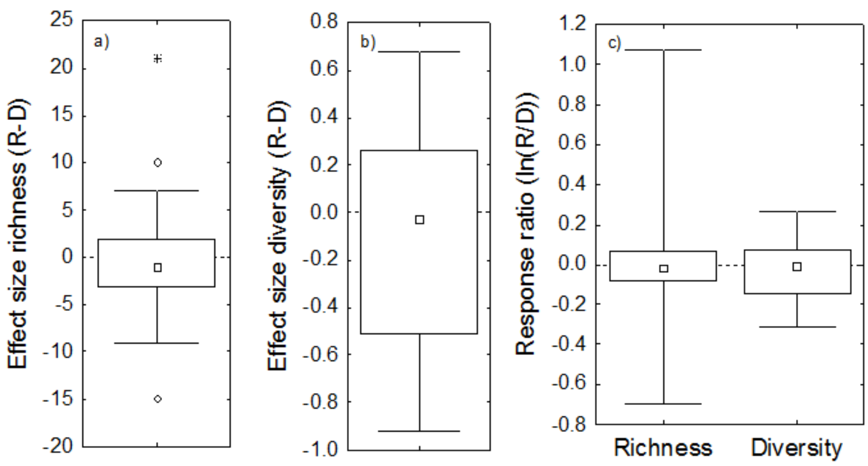

Figure 5-1: Effect of restoration on macroinvertebrate richness and diversity. a.) and b.) absolute values of difference R minus D, c.) relative response ratio ln(R/D).

Differences of restoration effect in large and small projects (R1 vs. R2)

Group comparison did not reveal significant differences of richness and diversity between the four groups of R1 (large restoration projects), R2 (small restoration projects), the degraded sections (D1 and D2) (One-way ANOVA, richness $F_{3,37} = 2.95$, $p = 0.829$; diversity $F_{3,37} = 0.889$, $P = 0.456$, Table 5-4).

Similarly, pairwise calculated effect sizes, expressed as the absolute difference between the restored and degraded sections and the relative difference (Osenberg response ratio) showed no significant effect of restoration on richness and diversity, i.e. mean values were not significantly different from zero, neither for the large nor for the small restoration projects (t-tests, $p > 0.05$, n=10 and n=9, respectively), except for the significant negative effect of restoration on diversity in the small restoration projects ($p < 0.05$, Figure 5-2b). Moreover, effect sizes of richness and diversity were not significantly different between large and small restoration projects, neither for comparing the two groups R1 and R2 (Mann-Whitney U test, $p > 0.07$, n=19), nor for a paired comparison (R1 compared to corresponding R2 section, Wilcoxon-Matches Pairs test, $p > 0.14$, n= 9).

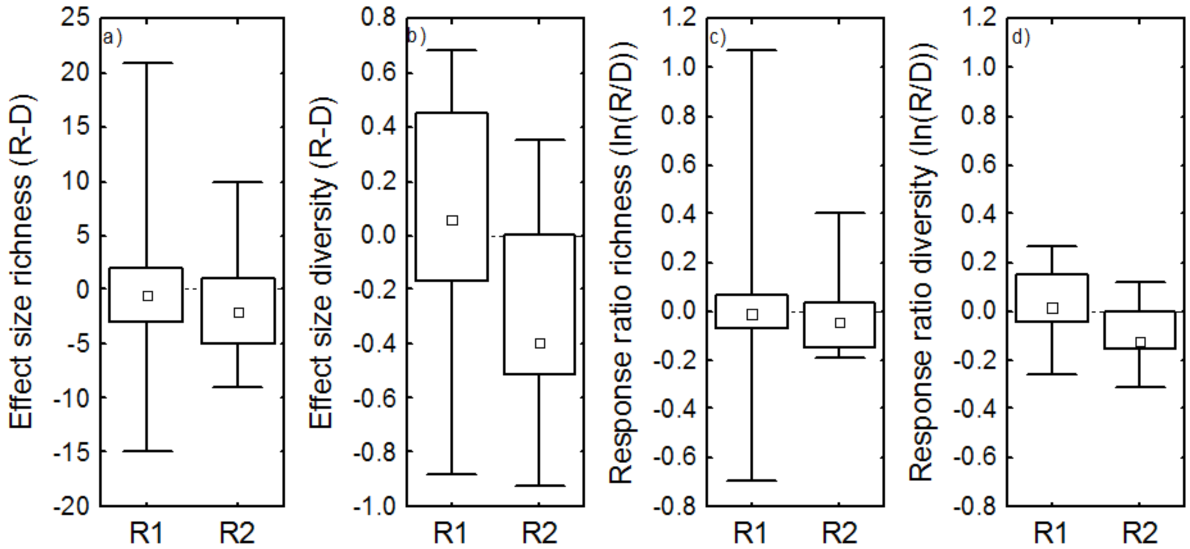

Figure 5-2: Effect of restoration on macroinvertebrate richness and diversity for large (R1) and small (R2) restoration projects,. a.) and b.) absolute values of difference R minus D, c.) d.) relative response ratio ln(R/D).

Differences of restoration effect in river types (gravel- vs. sand-bed rivers)

There were no significant differences of macroinvertebrate richness and diversity between gravel-bed and sand-bed rivers (Mann-Whitney U test, $p > 0.50$). However, especially richness effect sizes showed a tendency for a larger effect of restoration in gravel-bed rivers (n=12) compared to sand-bed rivers (n=7), i.e. low versus high gradient rivers (Figure 5-3a).

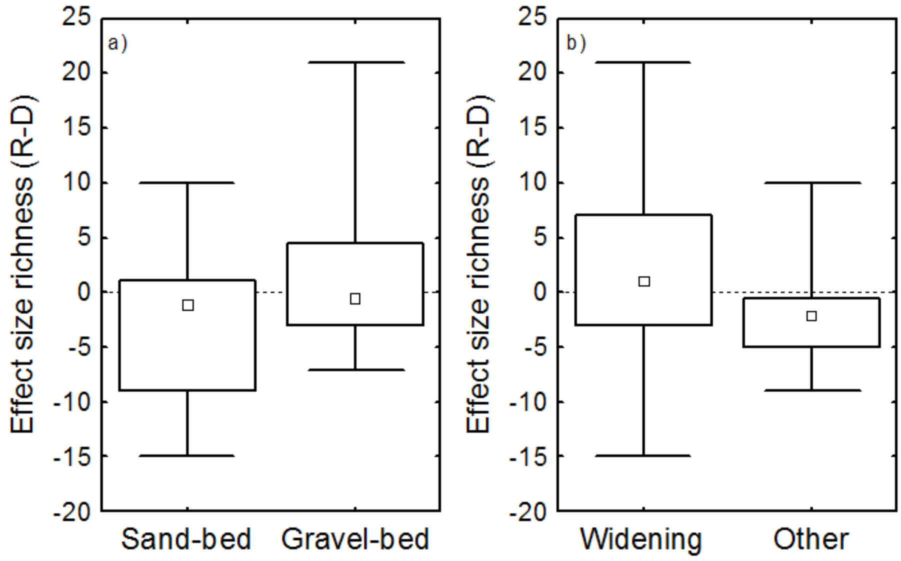

Figure 5-3: Effect of restoration on macroinvertebrate richness in sand-bed vs. gravel-bed rivers (a) and widening vs. other projects (b) using the absolute difference between richness values of restored and degraded sections as effect size.

General relationship of macroinvertebrate richness and diversity and the type of restoration measures

There were no significant differences of macroinvertebrate richness and diversity between restoration projects which mainly applied river widening as a main measure and other measures (Mann-Whitney U test, $p > 0.22$). However, richness effect sizes showed a tendency for a larger effect of restoration in projects which aimed at river widening (n=11) compared to projects which mainly applied other, less extensive measures mainly affecting the river channel itself (instream measures, flow restoration, remeandering, anastomosing, n=8).

Relationship between biological metrics and environmental variables

No significant correlations between the effect sizes for total richness and diversity, number of rheophilic species and diversity of habitat preferences within the assemblage were found (Table 5-5). Furthermore, the variation in effect sizes of neither the macroinvertebrate metrics nor the Euclidian distance representing the change in community composition could be explained by any of the environmental variables investigated (Table 5-6). This indicated that the direction and magnitude of the differences in the macroinvertebrate metrics between the restored and degraded sections was not related to differences in the environmental variables measured.

Table 5-5 Correlation coefficients (Spearman rank order) between the effect size (Osenberg-ratio; n =19) of macroinvertebrate variables (total macroinvertebrate taxon richness and Shannon-Wiener diversity) and predictor variables. None of the correlations was significant (P<0.05). Richness = total taxon richness, Diversity = Shannon-Wiener diversity, Rheo = number of rheophilic + rheobiont taxa, Habpref = Habitat preference diversity.

Predictor group	Predictor	Richness	Diversity	Rheo	Habpref
River typology	Altitude	0.269	0.011	-0.101	-0.254
	Slope	0.212	0.042	-0.229	-0.370
	Discharge	0.297	0.183	-0.082	-0.447
	Width	0.393	0.181	-0.162	-0.499
	River type (gravel-sand)	0.120	0.159	-0.172	-0.279
	Typology PC1*	-0.291	-0.167	0.185	0.367
Restoration	Restoration extent	0.096	0.404	0.127	-0.116
	Restoration age	0.332	0.151	-0.109	-0.004
	Restoration length	0.175	0.261	0.317	0.050
Water quality	PO4	-0.155	-0.228	0.234	0.081
	NO3	0.137	-0.054	-0.092	0.228
	NH4	-0.278	-0.232	0.302	0.269
Catchment land use cover	Artificial surfaces	0.186	0.175	-0.063	0.299
	Agricultural areas	0.061	-0.142	0.145	0.185
	Forest and seminatural areas	-0.011	0.161	-0.137	-0.239
	Wetlands	-0.342	-0.101	-0.105	-0.204
	Waterbodies	-0.236	-0.075	0.137	-0.109
Microhabitat characteristics	Current velocity	0.356	0.195	0.144	0.096
	Number of natural habitats	0.318	0.206	0.019	-0.151
	Natural substrate diversity	0.444	0.288	-0.026	-0.158
	Spatial distribution of substrate diversity	0.196	0.232	-0.144	-0.312

*see Table 5-2

Table 5-6 Significance testing of macroinvertebrate metrics based on redundancy analyses with all environmental variables included.

Metric	F	P
Taxon richness	1.1	0.306
Taxon diversity	<0.1	0.959
Assemblage composition	1.2	0.456
Number of rheophilic + rheobiont taxa	2.2	0.160
Habitat preference diversity	0.6	0.438

Effect of hydromorphological changes on macroinvertebrates

The effect on macroinvertebrate richness was significantly higher in projects where restoration measures increased microhabitat diversity (Spearman rank correlation, $p < 0.05$, $n = 19$). Moreover, excluding one single outlier resulted in significant correlations between the effect of restoration on macroinvertebrate richness as well as diversity and its effect on two parameters describing substrate conditions at the microhabitat scale (response ratio of Shannon-Wiener diversity of microhabitats and total number of microhabitats, see results on hydromorphology in Chapter 4) (Spearman rank correlation, $p < 0.05$, $n = 18$).

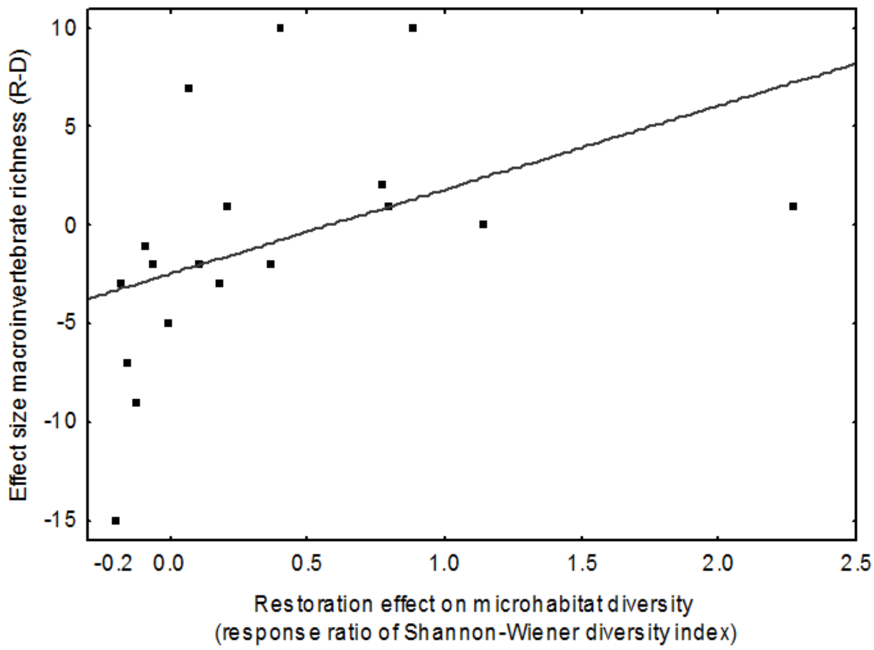

Figure 5-4: Correlation between the effect of restoration on macroinvertebrate richness and its effect on microhabitat diversity (response ratio of the Shannon-Wiener diversity index of natural microhabitats).

The strongest correlation was found between the effect size of macroinvertebrate richness and the response ratio of the Shannon-Wiener diversity of microhabitats (Spearman rank correlation, $\rho = 0.74$, $p < 0.001$, $n = 18$, Figure 5-4). No such correlation was found between macroinvertebrate metrics and hydromorphological parameters describing macro- or mesohabitats.

5.4 Discussion

No effects of restoration on the macroinvertebrate metrics were detected based on the pairwise comparison of restored and upstream non-restored river sections throughout Europe; neither on total richness, diversity or assemblage composition, nor on a subset of assemblage which should be, at least in theory, indicative of rivers of a good ecological quality. No effects could be detected for rheophiles, which are regarded as indicators of natural flow regimes and based on the diversity of habitat preferences within the assemblage, indicative of high habitat diversity and or heterogeneity. These results are in line with other restoration studies, which have already indicated that hydromorphological restoration measures which increase structural heterogeneity or restore natural flow regimes do not necessarily promote macroinvertebrate biodiversity, even when habitat changes are large (Lepori et al., 2005; Haase et al. 2013; Friberg et al. 2014).

However, macroinvertebrate richness and diversity was correlated with microhabitat diversity (Figure 5-4). Since microhabitat conditions were not significantly improved in the restored sections investigated in this study, and the effect of restoration was especially low on substrate diversity (see Chapter 4), the low effect of restoration on macroinvertebrates might mainly reflect the low effect of the studied restorations on microhabitat diversity (see results on hydromorphological effects in Chapter 4). Accordingly, while restoration projects like widening are visually appealing and increase macro- and mesohabitat diversity (Chapter 4), they may generally not increasing microhabitat diversity relevant for macroinvertebrates and species diversity. Overall, in contrast to other studies which concluded that restoring habitat diversity does not promote invertebrate diversity (e.g. Palmer et al. 2010), our results indicated that reach-scale restoration can indeed increase species richness and diversity, but that this is dependant on creation of ecologically relevant microhabitats. The scatterplot (Figure 5-4) suggested that decreasing microhabitat diversity had a negative effect and even slightly increasing might have a strong positive effect on macroinvertebrate richness. However, further increasing microhabitat diversity did not further increase richness, which indicated that other factors might constrain the effect of restoration (e.g. depleted species pools for re-colonization, low water-quality).

Such factors which might have constrained the effect of restoration include the impact of landscape-level stressors not mitigated by the restoraton measures applied (Palmer et al. 2010; Haase et al. 2013), or local stressors which interfered with the paired design of our study. The scale at which such stressors operate can range from microhabitat (e.g. clogging of interstitial spaces of coarse substrate by silt, missing habitat components such has dead wood), to mesohabitat (large water temperature fluctuations because of lack of shading by riparian trees) to catchment scale (e.g. eutrophication, impact of pesticides or other harmful substances). It is important to note that stressors can also affect specific habitat needs of the terrestrial adult life stage of aquatic insects (e.g. riparian trees; Hoffmann, 2000), something which appears to be often overlooked in river restoration.

In this study there was no difference in the effect of restoration on macroinvertebrates between large and small restoration projects. More important is the composition of the regional species pool and the distance to the nearest populations of target species or species which are otherwise related to water of a good ecological quality (Sundermann et al., 2011). Many species have been lost from catchments as a result of, amongst others, habitat degradation and pollution. As a consequence, there are no source populations left which could act as a starting point for recolonization of the restored river sections (Haase et al., 2013), resulting in species being currently absent at sites which are suited based on the hydromorphology, physical and chemical conditions and biology. For several restored sections investigated in this study (13 out of 20), data were available on the total length of the water bodies 0-1 and 1-5 km upstream of the restored sections being in a high or good ecological status. However, the effect of restoration on macroinvertebrate richness and diversity did not depend on this proxy for the species pool available for re-colonization (Spearman rank correlation, $p > 0.21$), indicating that the depleted regional species pools was not the main reason for the missing effect of restoration on macroinvertebrates in our study.

Given the increasing number of studies finding no or only minor effects of hydromorphological restoration on macroinvertebrates, it is very important to identify the main reason for the lack of success, which amongst others might be (i) a low effect of restoration on the relevant microhabitat conditions despite a high effect on meso- and macrohabitats, (ii) the impact of other, large-scale stressors, and (iii) depleted regional species pools and dispersal limitations. If the latter is true and the reason in mainly biological, project age will become very important as a factor determining success. Unfortunately, dispersal in macroinvertebrates is a rather understudied topic in freshwater ecological research (Bilton et al. 2001); we know that long-distance dispersal takes place, but it is not known on what time-scale this process operates. If there are good reasons to assume that colonization is nearly impossible, translocation of organisms can be considered as an option, especially when the species in question play an important functional role within the ecosystem (IUCN/SSC, 2013). The results of the present study indicated that it is crucial to restore physical habitat conditions which are ecologically relevant like substrate diversity at the microhabitat scale for invertebrates and many restoration projects might have had a low effect on macroinvertebrates due to a low effect on microhabitat diversity.

5.5 References

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6. Fish

6.1 *Introduction*

Rivers are among the most heavily degraded ecosystems on earth. In Europe, 64% of 1.17 million river kilometres have been reported not in good ecological status (EEA 2012). Hydromorphological pressures and altered habitats have been identified as significant pressure for 48.2% and 42.7% of the rivers, respectively (Fehér et al. 2012). Similarly, in the United States, 44% of 0.9 million river and stream kilometres have been reported as impaired (USEPA 2009). Habitat alteration occurred in 23.2% of the impaired rivers, and flow alteration in 9.7%. Therefore, besides improving water quality, which is still a significant pressure in European rivers, hydromorphological river restoration has become a key objective in river basin management (Schinegger et al. 2012). Here "river restoration" is used as a general term for any improvement of ecological conditions in rivers, including pressure mitigation, habitat and flow enhancement, or continuity reestablishment.

Studies on river restoration showed site or river-specific responses of biota to restoration of hydromorphological pressures (Jungwirth et al. 1995; Lamouroux et al. 2006; Muhar et al. 2007; Zitek et al. 2008; Schmutz et al. 2013). Fish has been identified as a key indicator to reflect biotic response to river restoration (Haase et al. 2012). Only few studies have compared the response of restoration measures across multiple rivers (Haase et al. 2012; Lorenz & Feld 2012; Januschke et al. 2014). Most of the multi river comparisons were limited to specific regions, thus preventing general conclusions for larger areas or different bioregions.

Restoration measures may affect only specific species, life stages, or functional groups before the entire community reacts. However, specific metrics, e.g. juvenile fish, have rarely been investigated (Lorenz et al. 2013). Information about fish changes post restoration are important today to value the success of restoration project, and also to guide future restoration programmes.

Beside direct response of fish assemblages to hydromorphological changes, it is likely that the length of the restored river section and the time after restoration would also have an effect on fish communities. There is evidence that the dimension of restoration measures plays a critical role in the likely effects on biota (Schmutz et al. 2013).

Moreover, fish communities have been shown to change with time. Long recovery periods (i.e. 10-20 years) were supposed to result with strong effects on fish (Jones & Schmitz 2009). However, so far just few of those factors have been tested in the restoration context across a large range of restored rivers in Europe.

For European rivers, the Water Framework Directive (WFD) aims at achieving good ecological status or potential. The challenge is to predict how biota will respond to restoration and what management actions are best suited. However, here is a lack of empirical data on relevant geographical and long-term scales required for assessing restoration / rehabilitation success (Hering et al. 2010).

This study was part of a larger approach to analyse the response of biota to hydromorphological restoration within the EU-project REFORM. In addition to the effects on fish, responses of habitat, macrophytes, benthic invertebrates, floodplain vegetation, ground beetles, and stable isotopes were analysed in a common framework (Chapter 1).

The objective of the study was to test if there is a consistent change in fish assemblages in response to hydromorphological restoration measures in 20 European restoration projects. We compared assemblage-based metrics with functional metrics and tested if restoration extent (restored section length and restoration intensity), its hydromorphological quality, and project age (time between implementation of measures and monitoring) affect restoration success.

6.2 Methods

Study sections

The study sections and reaches as well as sampling methods for fish are described in Annex B and Chapter 2.5. Fifteen out of the 20 restoration projects were selected for this study which were located in seven regions, covering a latitudinal gradient from Central to Northern Europe (latitude range 46-65°). The restoration projecs were located in Austria (n=2), Switzerland (n=2), Czech Republic (n=1) Germany (n=4), Denmark (n=2), Sweden (n=2), and Finland (n=2) and vary in terms of river type, altitude, slope, and size (Annex B).

Attributes of fish assemblages

For this study, the length measured during fish sampling (see Chapter 2.5 for details) was used to discriminate between small (≤15 cm body length) and large (>15 cm) fish.

The catch data were standardised by dividing the number of sampled fish by the sampled area (ind ha^{-1}). We calculated (1) the total number of species, (2) the proportional densities of species (pi) and (3) the total density per hectare for all species and habitat traits (rheophilic, limnophilic, and eurytopic species). The proportional abundance of species, and the fish densities were divided into small (≤15cm) and large (>15 cm) fish.

In total 13 metrics were considered in the analyses. We assigned all species to habitat traits according to the EFI+ classification (EFI+Consortium 2009) and discriminated between salmonid and non-salmonid species. In order to assess the potential influence of the sampling intensity on the number of species, we regressed the sampling area against the number of species. Furthermore, we calculated the Shannon Wiener diversity index $H = - \sum (pi * \ln (pi))$. Relation among fish communities of different sections were analysed using ordination techniques, i.e. multidimensional scaling (MDS). MDS takes a set of dissimilarities and returns a set of points such that the distances between the points are approximately equal to the dissimilarities. Euclidian distances were computed using relative species composition with the R function “dist”. The R function “cmdscale” was used to perform the MDS.

Effect size and restoration success

As the restored sections vary in terms of species composition and abundance due to natural differences, we used an effect size as a standardised metric for comparing the restored and the corresponding degraded sections. We calculated the effect size as the value of restored sections minus the values of degraded sections (R - D). An effect size of zero indicates no change, a positive value represents an increase, and a negative value a decrease. First, effect sizes were tested for being different from zero using Student's t-test and Bonferroni correction for multiple testing ($p= 0.05/13 = 0.00385$). Second, highly correlated metrics (Pearson: $|r|>0.8$) were removed in an iterative way. Metrics with the highest number of correlations with other metrics were removed, and the procedure was repeated until only uncorrelated metrics remained. Significant positive change was considered as a restoration success for species richness, densities and diversity except for eurytopic fish where a decrease was indicative for restoration success. The difference between restoration effect in large vs. small restoration projects was tested for non-redundant metrics using Student's t-test.

Factors affecting restoration success

We analysed the following factors potentially affecting restoration success: (i) length of the restored river section, (ii) time after restoration (project age) and (iii) hydromorphological quality of restoration. The length of the restored river section (km) was measured from the uppermost to the lowermost part of the restored river section. The time after restoration is the number of years passed since the implementation of the restoration measures. The hydromorphological quality of restoration was assessed using four types of attributes related to (i) channel geometry and flow characteristics (flow velocity and character), (ii) riverbed (water depth, bed stabilisation, substrate), (iii) water-land transition zone (river width, stabilisation, woody debris, bedload accumulation), (iv) riparian zone (cross section, bank protection, vegetation), and floodplain vegetation (extent and type). Each attribute was classified from 1 (high status) to 5 (bad status) following the WFD principle of status classification. Finally, an overall hydromorphological index was calculated by first averaging all attributes of an attribute type, followed by averaging the four attribute types. For more details on the hydromorphological monitoring methods see Chapter 2.3. Correlations among potential factors affecting restoration success were tested using Spearman's rank correlations.

We used classification and regression trees (CRT), a recursive partitioning method, to model fish metrics as a function of (i) length of the restored river stretch (km), (ii) time after restoration (years), and (iii) hydromorphological quality of restoration (index). Only significant fish metric were used for the tree models. CRT methods were available in the package rpart for R-library (R-project CRAN). Tree methods encompass several advantages, nonparametric basis, no implicit assumption of linearity, simplicity of results for interpretation, and ability of predictive classification for new observations. Trees were first developed with single factors (restored length, hydromorphology, time), and second with all factors combined. All analyses were computed using R version 3.1.1.

6.3 Results

Fish were sampled in the years 2011-2014. A total of 43 species and 25,746 individuals were sampled, encompassing 20 rheophilic species, 15 eurytopic and 8 limnophilic species (Annex D). Due to the low number of limnophilic species, this trait was not considered in further analyses.

Regressing the number of species against the sampling area revealed a significant response ($F=11.08$, $p= 0.003$), however, this relationship was triggered only by one river (DE_Lippe, 21 species) and, therefore, was not considered influential for the further analyses.

MDS revealed closer relationships between restored and corresponding upstream degraded sections than among restored sections at different locations (Figure 6-1). Fifteen sections were dominated by non-salmonid and the same number (15) by salmonid species. Eleven restored sections remained in the same type of fish community after restoration compared to the corresponding degraded section. One restored section changed from salmonid to non-salmonid (CH_Thur_R1) and three from non-salmonid to salmonid communities (DK_Storaa_R2, SE_Morrum_R2, SE_Eman_R1).

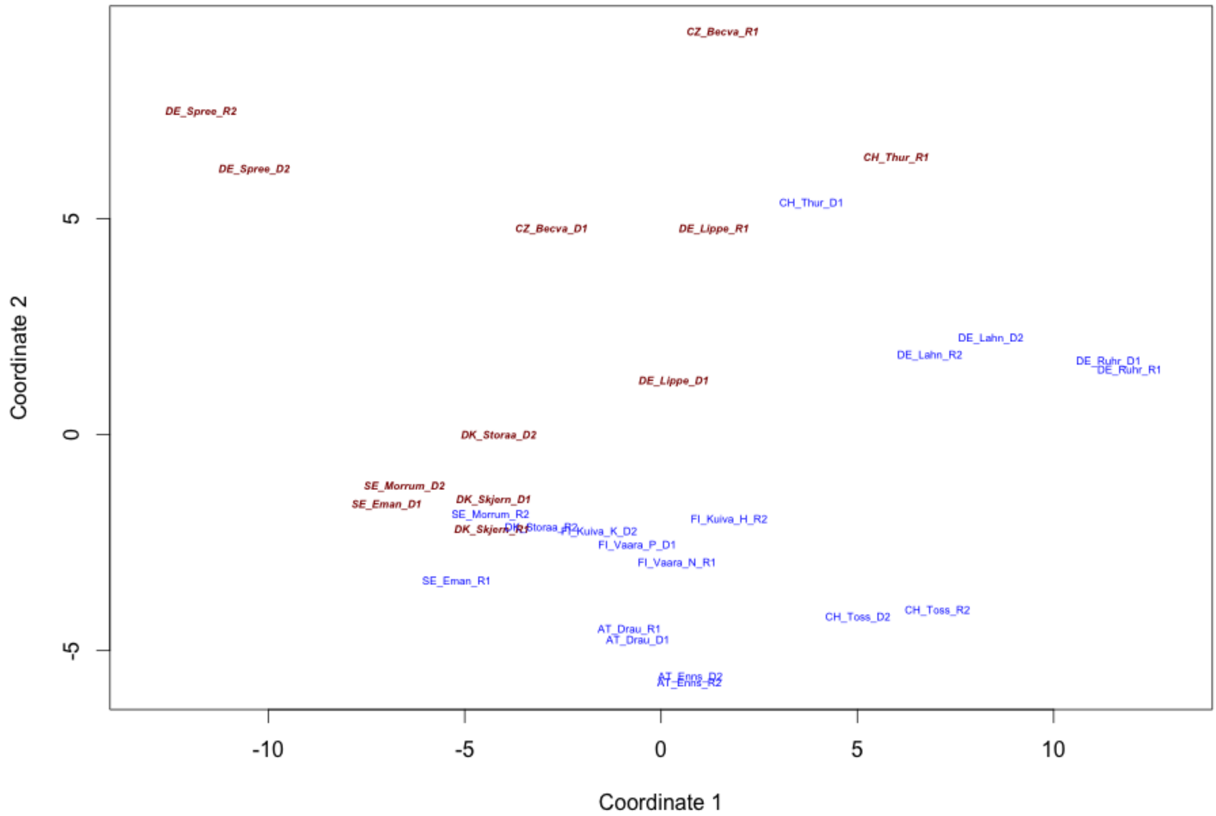

Figure 6-1: Multidimensional scaling (MDS) of fish communities of degraded and restored sections. Sections are coded with country names, restoration exent (R1= large, R2= small) and river names (e.g. AT_ Drau_R1: Austria-large restoration River Drau). Blue italics: salmonid dominated sections, red: non-salmonid dominated sections.

Restoration had a significant effect ($p<0.05$) on 5 out of the 13 metrics investigated, with significant changes and no redundancy with other metrics (Table 6-1, Figure 6-2). Mean species richness increased by approximately one species, which can be attributed mainly to an increase of rheophilic species. The density of rheophilic fish and of small rheophilic fish and the proportion of density of small rheophilic fish increased. However, only the proportion of density of small rheophilic fish (increase of 24%) was significant when considering the Bonferroni corrected p-value ($p = 0.00385$). Eurytopic fish decreased or changed only slightly, however, they were either not significant or redundant to other metrics. Neither total density nor Shannon-Wiener diversity increased significantly. No difference between small and larger restoration projects were found when using the most significant metric, i.e. proportion of small rheophilic fish ($p = 0.8689$, Figure 6-3).
The hydromorphological index of restored sections ranged from 1.4 to 2.5 (median 1.9), indicating "high" to "good" hydromorphological status. Restored sections were monitored in the years 2011-2014, 1 to 17 years (median 7 years) after completion of restoration measures, and the length of restored sections covered a wide range between 0.2 and 26.0 km (median 0.9 km). Correlations among potential factors affecting restoration success were low and not significant ($r< |0.26|$, Table 6-2).

Table 6-1: Effect size measured for 13 metrics based on fish. P-values, significance level and redundancy are given for each fish metric. Bold metrics indicate significant metrics considering Bonferroni correction.

Fish metric	Unit	Mean effect size	p-value	Significance level	Redundancy
Species richness	number	1.07	0.03310	>0.00385	not redundant
Richness rheophilic	number	1.00	0.02700	>0.00385	not redundant
Density rheophilic	number per ha	301.33	0.03660	>0.00385	not redundant
Density rheophilic small	number per ha	213.63	0.02080	>0.00385	not redundant
Proportion density rheophilic small	**percentage**	**24.11**	**0.00350**	**<0.00385**	**not redundant**
Total density	number per ha	313.07	0.41690	>0.00385	---------
Shannon diversity	index	0.14	0.14190	>0.00385	---------
Richness eurytopic	number	0.00	0.97500	>0.00385	---------
Density eurytopic	number per ha	-2.19	0.20610	>0.00385	---------
Density eurytopic small	number per ha	-84.02	0.09380	>0.00385	---------
Proportion density rheophilic	percentage	19.88	0.01560	>0.00385	redundant
Proportion density eurytopic	**percentage**	**-19.76**	**0.00330**	**<0.00385**	**redundant**
Proportion density eurytopic small	**percentage**	**-17.19**	**0.00080**	**<0.00385**	**redundant**

Table 6-2: Correlations among three factors potentially affecting restoration success

	Hydro-morphological index	Length of restored section
Length of restored sections	-0.26 (p=0.34)	-
Years after restoration	-0.14 (p=0.62)	0.17 (p=0.55)

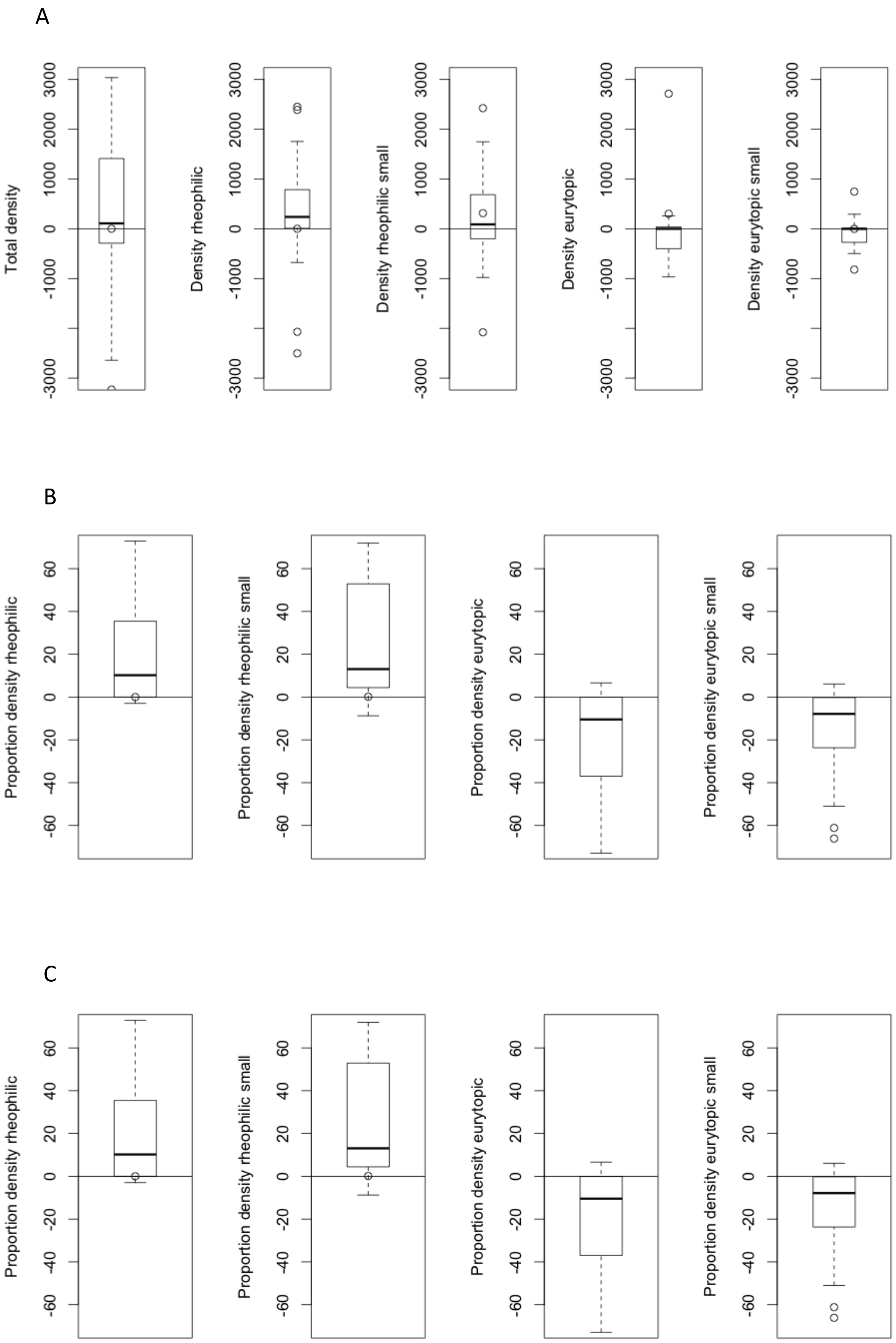

Figure 6-2: Effects sizes of 13 analysed metrics related to (A): species richness and diversity, (B): density (ind ha^{-1}) and (C): proportion of density.

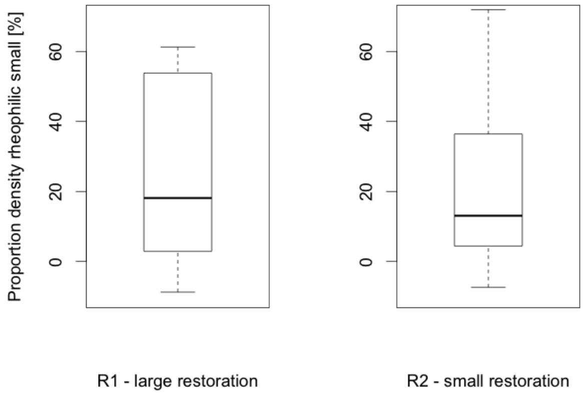

Figure 6-3: Effect size of the proportion of small rheophilic fish in large (R1) and small (R2) restoration projects.

Based on single factor regression tree analyses, using the only significant and non redundant metric proportion of small rheophilic fish as independent variable, sections with a length > 1.95 km revealed stronger responses to restoration than shorter sections. This was driven by three restoration projects (DE_Lippe_R1, DK_Skjern_R1, SE_Morrum_R2). Sections with hydromorphological indices <2.14 showed higher effect sizes than those with indices ≥ 2.14 (AT_Enns_R2, DE_Lahn_R2, DE_Ruhr_R1, DE_Spree_R2). Restored sections which were monitored before three years or after 12.5 years (CZ_Becva_R1, DE_Lahn_R2, DE_Lippe_R1, DK_Storaa_R2, SE_Eman_R1 SE_Morrum_R2) showed stronger restoration effects than those monitored between 3 and 12.5 years. When considering all three factors simultaneously, short-term effects were most important for high effect sizes (DK_Storaa_R2, SE_Eman_R1, SE_Morrum_R2). In addition, time effects and hydromorphological index interacted in a way that, when excluding the short-term effects, sections with a very high hydromorphological index (indices < 1.57) responded more strongly (CZ_Becva_R1, DE_Lippe_R1, DK_Skjern_R1, FI_Kuiva_H_R2) than others (Figure 6-4).

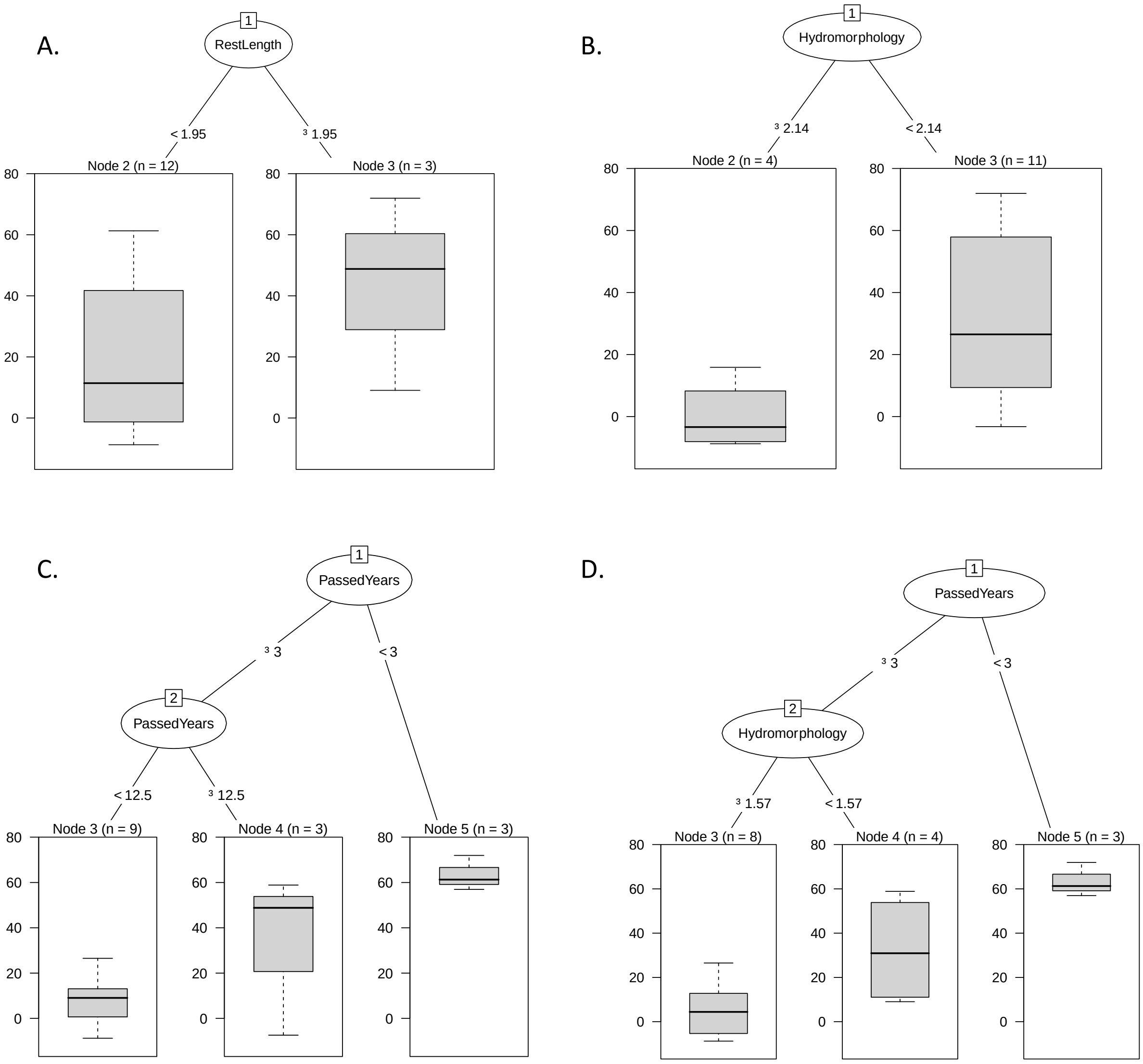

Figure 6-4: Response of proportion of small rheophilic fish to (A) length of restored section („RestLength", km), (B) hydromorphological index ("Hydromorphology", Index 1-5) and (C) number of years passed after restoration („PassedYears"), using single factors (A-C)) and all factors (D).

6.4 Discussion

The hydromorphology of lotic ecosystems is being increasingly modified worldwide by damming, fragmentation, flow regulation and channel modification. Serious threats to riverine biodiversity are suspected (Collen et al. 2014), yet available field data are few and rarely address the various taxonomic, functional and phylogenetic components of biodiversity (Feld et al. 2014). At the same time, public awareness has increased and political masterplans (e.g. the EU-Water Framework Directive) try to counteract the ecological degradation. Particularly in Europe and the U.S. large numbers of river restoration measures are realised (Bernhardt et al. 2005). Assessing the outcome of river restoration projects is vital for adaptive management, evaluating project efficiency, optimising future programmes and gaining public acceptance (Woolsey et al. 2007). Although the effectiveness of river restoration has been analysed for many years, clear and detailed results are scarce (Bernhardt et al. 2005). For example, despite locating 345 studies on effectiveness of stream rehabilitation by Roni et al. (2008), firm conclusions about restoration techniques were difficult to make first due to the limited information provided on physical habitat, water quality, and biota and second, due to the short duration and limited scope of most published evaluations. Therefore, more in-depth studies on river restoration are required to provide the scientific basis for effective restoration programmes in future.

Only few studies compared the response of restoration measures across multiple rivers (Haase et al. 2012; Lorenz & Feld 2012; Januschke et al. 2014). Most of the multi river comparisons were limited to specific regions, preventing general conclusions for larger areas or different bioregions. For example, the study of Stoll et al. (2013) was restricted to lower mountain ranges of Germany, Schmutz et al. (2014) analysed the effect of restoration measures in the Austrian Danube. Only few studies compared restoration effects across different regions (Feld et al. 2014).

In this study, fish data of 15 restored sections were sampled and analysed, covering a large latitudinal gradient from Central to Northern Europe. Species richness, species diversity and fish density showed only weak or no response to restoration, while habitat traits, i.e. rheophilic and eurytopic fish, reacted in a consistent way across the restoration projects investigated. Fish assemblages showed changes with hydromorphological restoration while other biological groups in other studies revealed less consistent results indicating that stressors other than hydromorphological degradation might affect the biota in restored sections (Haase et al. 2012). Weak diversity responses to hydromorphological alteration were found for macroinvertebrates in lowland rivers (Feld et al. 2014). Their results suggested that taxonomic and trait replacement with hydromorphological alteration is not followed by changes in whole-community diversity. Morandi et al. (2014) analysed 37 restoration projects and found that in 76 % community structure was the most often monitored metric, used more often than species richness (57 %). Mueller et al. (2014) found that fish community composition only changed significantly in 50% of the restored rivers, depending on the occurrence of species sensitive to the structures introduced by the restoration treatments. A change in fish assemblage structure but not in biomass has also been detected in lake restoration (Gao et al. 2013).

These examples are consistent with our findings that restoration projects – as practised today – do not change species richness and diversity but rather community structure, in our case expressed as increase of rheophilic and decrease of eurytopic fish. One reason could be that in headwaters (salmonid dominated communities) species diversity is low even under natural conditions. However, in lowland rivers (non-salmonid dominated communities), which naturally have a higher species richness and diversity, this had to be due to other reasons like water pollution, migration barriers or poor colonization sources. Stoll et al. (2013) attributed weak restoration response to impoverished regional species pool as nearly all fish species occurring in restored reaches were present in reaches within a distance of 5 km up- or downstream of the restored reach. They concluded that the limited success in establishing natural fish assemblages in restored reaches was attributed to spatial limitation (e.g. due to fragmentation) and an impoverished regional species pools from which restored reaches recruit. Future restoration efforts and studies should also incorporate the effects of nearby barriers, temporal patterns in species dispersal, and minimum effective size of potential founder populations (Radinger & Wolter 2014).

We found that the proportion of rheophilic fish increased after restoration. Similar change was also observed in the Danube after implementation of rehabilitation measures (Schmutz et al. 2013). Mueller et al. (2014) demonstrated that besides lithophilic and invertivorous species, rheophilic fishes benefited from restoration measures. In our study, small rheophilic fish showed a stronger reaction than all rheophilic fish. Likewise, Woolsey et al. (2007) proposed to use age structure besides guilds (species traits) as metrics for monitoring restoration success. YOY lithophilic fish - also strongly associated with riverine conditions - was the reproduction guild with the highest increase in a similar study (Lorenz et al. 2013). As expected, the increase of rheophilic fish was accompanied by a decrease of eurytopic fish given the fact that total density did not change as a result of restoration. Restoration measures applied in our study, i.e. river widening, creation of instream structures, flow enhancement, re-meandering and side-channel reconnection recreated mesohabitats important for rheophilic fish species particularly for early life history, i.e. gravel bars as spawning and nursery habitats.

Beside hydromorphological quality, our results showed that the response of fish was stronger within the first three years and after 12 years post restoration, and less pronounced in the mid-term range (3-12 years). This seems to contradict the expectation that longer recovery periods would result in stronger effects. Jones & Schmitz (2009) reviewed 240 recovery studies across terrestrial and aquatic ecosystems and identified mean recovery times of 10 to 20 years for freshwater, brackish and marine systems. In our study, the median time frame between restoration and monitoring was seven years, representing only one to three generations depending on fish species. Short recovery effects might be due to the creation of local gravel bars providing spawning and nursery habitats for rheophilic fish. This is in accordance with studies on artificial redd constructions. Pulg et al. (2013) found that in the first two years after artificial redd construction, highly suitable conditions were maintained, with a potential egg survival of more than 50% for brown trout (*Salmo trutta*). Afterwards, the sites offered moderate conditions, indicating an egg survival of less than 50%. Conditions unsuitable for reproduction were expected to be reached five to six years after restoration. Otherwise, mid-term recovery might be hampered by the restricted spatial extent of restoration measures and lack of dynamic rejuvenation of created habitats. Finally, a mean increase of only one species in our restoration sections indicates that even longer recovery periods than 10 years might be necessary.

Muhar et al. (2007) showed a clear relationship between restoration effect and spatial extent of restoration measures, but even a re-establishment of 94% of aquatic habitats compared with reference conditions did not guarantee good ecological status sensu WFD if other factors limited recovery processes. While in our study sections with a restored length over 1.95 km showed stronger responses, the highest positive restoration response in the Danube was observed for measures larger than 3.9 km (Schmutz et al. 2013). It seems that a minimum extent of restoration measures is required to enable fish recovery, but thresholds might depend on river size, type of fish community and source populations in the surrounding (Stoll et al. 2013).

6.5 Conclusions

Our study demonstrates that fish respond in a consistent way to hydromorphological restoration measures by an increase of rheophilic and a decrease of eurytopic fish. Restoration effects are more pronounced within the first years after restoration than later. The restoration effect increases with habitat quality and length of restored river sections. However, current restoration practice and technique do not allow comprehensive recovery of lost species and population densities. The reasons for that are probably manifold. The length of current restoration measures is short (mostly < 1km) limiting the amount and diversity of provided habitats. The quality of habitat improvement has to receive more attention. Therefore, future restoration should focus on more dynamic, self-sustaining habitat improvements extending over several kilometres and should be coupled with other measures such as restoring river continuity and species reintroductions.

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7. Macrophytes

7.1 *Introduction*

Macrophytes are important for the structure and functioning of aquatic systems in general and river systems in particular. Amongst others they regulate river processes (Tabacchi et al. 1998), provide important habitat and food for many different organism groups including macroinvertebrates and fish (Heck & Crowder 1991), and function as ecosystem engineers (Asaeda, Rajapakse & Kanoh 2010; O'Hare et al. 2011).

The degradation of river ecosystems has resulted in the partial loss of macrophytes and to it related functions (Steffen et al. 2013). River restorations are expected to reverse these adverse effects. Comparisons between degraded and non-degraded stream reaches indicate that river restoration should favour vegetation typical for non-degraded reaches. Indeed, previous studies have shown such positive restoration effects Lorenz et al. (2012). These effects were evident in different life forms including helophytes, elodeids and lemnids (Lorenz et al. 2012).

Helophytes are an important growth form in the riparian and littoral zone of rivers. Hence, restoration measures such as removal of bank fixation, re-meandering, and widening should favour helophytes whereas for example flow restoration should favour submerged hydrophytes. Time after restoration is another important predictor of macrophyte responses to restoration (Baattrup-Pedersen et al. 2000) and any potential response might be blurred by too short time span between restoration and follow-up study.

Here, we examine the response of macrophytes to restoration in 10 large and 10 small restoration projects. We expect that river restoration results in increased species diversity of macrophytes compared to degraded systems and that the response is more pronounced in large restoration projects compared to small restoration projects. In addition, we expect that responses vary among life forms due to the large variation in studied restoration measures.

7.2 *Methods*

Study sections and sampling methods

The study sections and reaches as well as sampling methods for macrophytes are described in Annex B and Chapter 2.6.

Data analyses

To detect general patterns in the species data, we performed a non-metric multi-dimensional scaling (MDS) based on the number of different life forms per reach. Differences in species richness and diversity between degraded and restored sites were tested with the Mann-Whitney *U*-test. Paired comparisons, e.g. between short and long restorations were performed with Wilcoxon Matched pairs test. If effect sizes were significantly higher than zero was tested with one-sided paired t-test. Spearman rank order correlation was used to test for the relationship between the effect size of macrophyte variables and predictor variables. To reduce the hydromorphological predictor variables to a few essential components, we used principal component analysis (PCA) (Jongman, ter Braak & van Tongeren 1995).

7.3 Results

Macrophyte life forms in the dataset and regional differences

A total of 148 macrophyte, i.e. non-terrestrial species were found in the sampling reaches. The species richness of all life forms except hydrophytes and helophytes per reach was low. Haptophyds (bryophytes) were represented with a maximum number of 10 species at site D1 in Austria but were absent from 43% of the study reaches. Nymphaeids were absent from 45% of the study reaches and the remaining life forms except helophytes and hydrophytes were absent from >50% of the study reaches.

The macrophyte communities showed small regional differences in respect to the life forms, and similarity was also high between degraded and restored sites, except for three degraded sites in the Czech Republic and Germany (Figure 7-1).

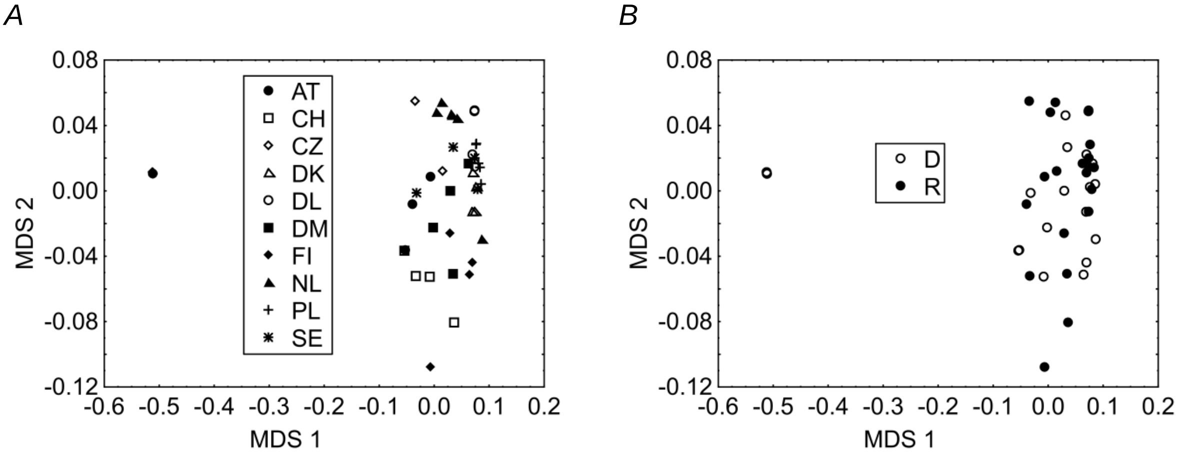

Figure 7-1: Biplots of non-metric multi-dimensional scaling (MDS) axes (1, 2) by reach. The non-metric MDS was based on the number of species per life form using the Sørensen index as similarity measure. Symbols represent the different reaches per country (A) and site (B) where D=degraded and R=restored. DL and DM are the two German sites Lippe/Spree and Ruhr/Lahn, respectively. The symbols to the left of the legends represent degraded systems in CZ (n=2) and DL (n=1) (symbols overlayed).

Due to the low number of representatives per life forms except for hydrophytes and helophytes, all further analyses were based on hydrophytes, helophytes and their combination, i.e. macrophytes. Hydrophytes comprise emergent and submerged aquatic plants. Helophytes are emergent plants rooting under water or in wetted soils, with a gradual transition from hydrophytes to helophytes and terrestrial plants.

Overall effect of restoration on macrophytes (R1 and R2 pooled)

Overall (pooling large and small restoration projects), comparing the two groups of restored and degraded sections did reveal significant differences in species richness and diversity for helophytes, only (Figure 7-2). Species richness and diversity of helophytes was significantly higher in the restored sections (n=20) compared to the group of degraded sections (n=20, species richness: $U=117$, $p<0.05$, diversity: $U=94$, $p<0.01$).

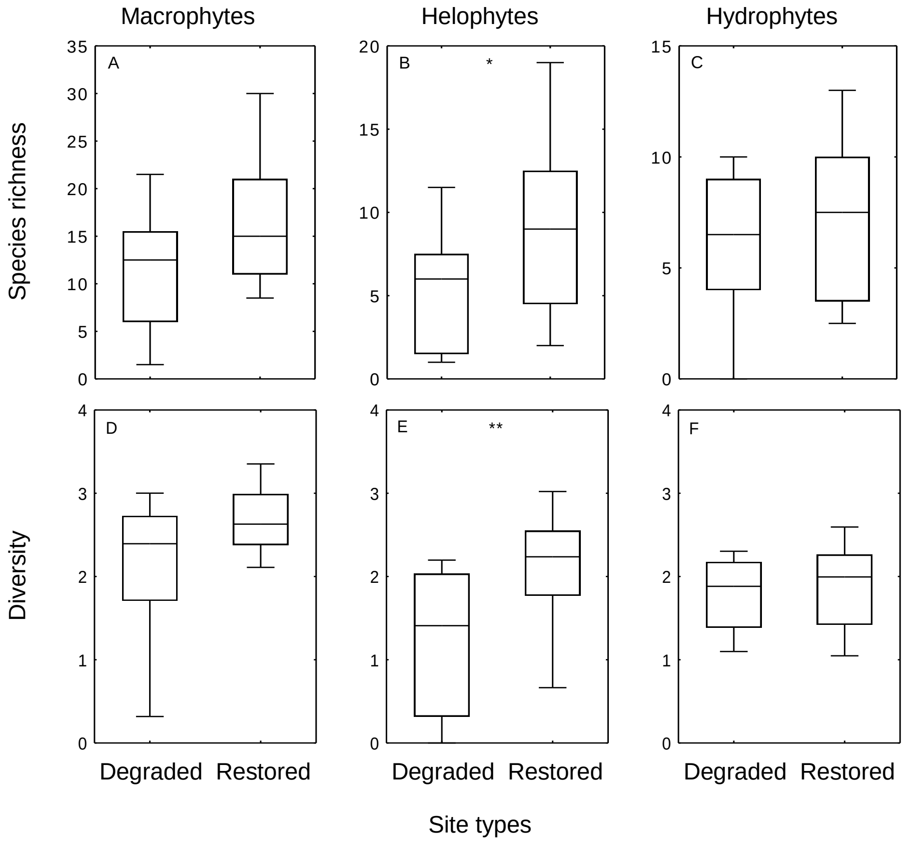

Figure 7-2: Median species richness and diversity (Shannon index) in the degraded and restored sections divided by species group. Macrophytes consist of helophytes and hydrophytes. Boxes represent 25 and 75 % percentiles and whiskers 10 and 90 % percentiles. Asterisks denote significant differences between degraded and restored sections (* p<0.05, ** p<0.01).

In addition, if the restored sections were compared to the corresponding degraded sections (pairwise comparison) by calculating the response ratio according to Osenberg et al. (1997), the mean restoration effects for overall macrophyte richness and diversity were significantly larger than zero (t-test, n=20, $p<0.01$ and $p<0.05$, respectively). In particular, mean restoration effects for helophyte richness and diversity were significantly larger than zero (t-test, n=20, $p<0.001$ and $p<0.01$, respectively), whereas restoration had no overall positive effect on species richness and diversity of hydrophytes (t-test, n=20, $p>0.05$ and $p>0.05$, respectively).

Differences of restoration effect in large and small projects (R1 vs. R2)

Neither group wise nor pairwise comparisons revealed differences in the effect size of species richness and diversity between large and small restoration projects, for none of the two life forms and for macrophytes in general (group wise: Mann Whitney U-test; macrophytes: species richness U=40.0, $n_1=10$ and $n_2=10$, $p>0.05$, diversity U=34.5, $n_1=10$ and $n_2=10$, $p>0.05$; helophytes: species richness U=31.5, $n_1=10$ and $n_2=10$, $p>0.05$, diversity U=35.0, $n_1=10$ and $n_2=10$, $p>0.05$; hydrophytes: species richness U=45.0, $n_1=10$ and $n_2=10$, $p>0.05$, diversity U=33.0, $n_1=9$ and $n_2=8$, $p>0.05$; pairwise:

Wilcoxon paired-sample test; macrophytes: species richness T=20, n=10, p>0.05, diversity T=14, n=9, p>0.05; helophytes: species richness T=14, n=10, p>0.05, diversity T=11, n=8, p>0.05; hydrophytes: species richness T=21, n=10, p>0.05, diversity T=16, n=8, p>0.05, Figure 7-3).

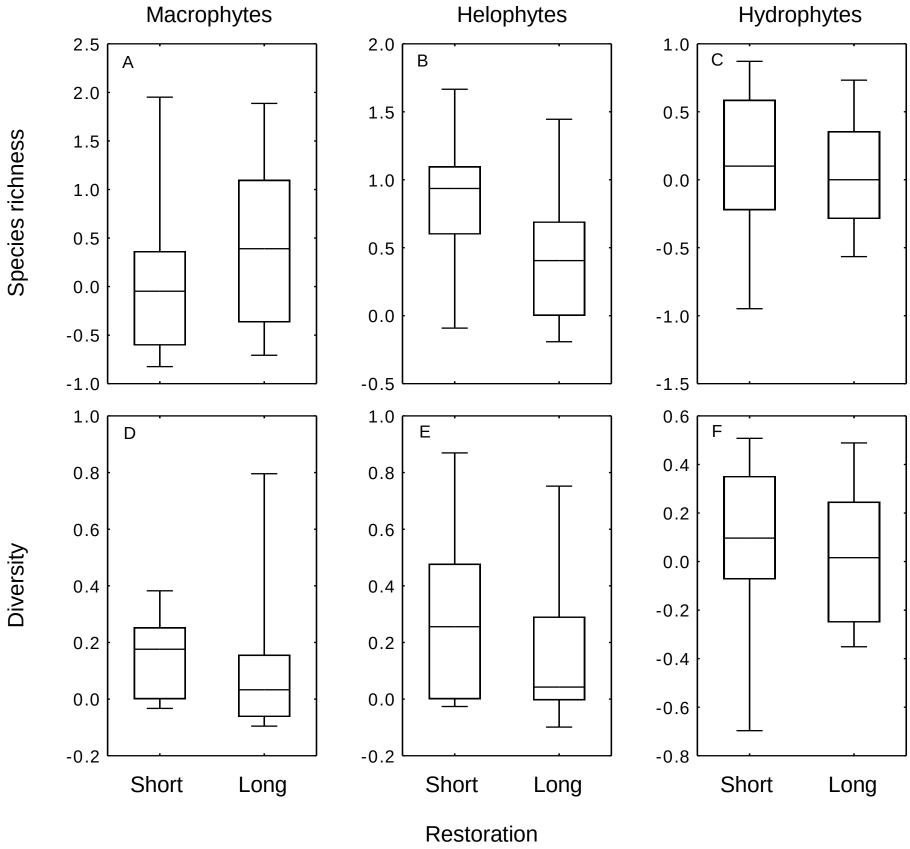

Figure 7-3: Median effect size (ln[restored/degraded]) of species richness and diversity (Shannon index) for large and small restoration projects for different life forms. Boxes represent 25 and 75 % percentiles and whiskers 10 and 90 % percentiles.

Relationship between biological metrics and environmental variables

Only the predictor variable altitude was correlated with the effect size of species richness of helophytes but this single correlation was non-significant after p-value adjustment for multiple comparisons (Table 7-1). All other predictor variables didn't show any correlation with species richness or diversity even prior to p-value adjustment for multiple comparisons (Table 7-1). The effect size of helophyte richness differed between restored sections in mountain and lowland streams (Table 7-2). Other richness and diversity effect sizes didn't differ within the predictor groups, i.e. countries, river types, substrate types and type of main measure (Table 7-2). The median effect size of helophyte richness was especially high in restoration projects which mainly applied widening measures compared to other restoration measures. Indeed, species richness of helophytes was significantly higher in restored sections with widening restoration measures compared to those with other restoration measures (Mann Whitney test, U=18.5, n_1=9 and n_2=11, p<0.05). We only considered principal components (PCs) that explained at least 10 % variation among the variables. For the interpretation of the PCs, we used factor loadings >|0.7|.

Table 7-1: Correlation coefficients (Spearman rank order) between the effect size (ln[R/D]; n=20) of macrophyte variables and predictor variables. The significance level was Bonferoni-adjusted (α'=0.0036).

Predictor	Macrophytes		Helophytes		Hydrophytes	
	Richness	Diversity	Richness	Diversity	Richness	Diversity
Altitude	0.06	0.08	0.47	0.12	0.04	0.10
Discharge	-0.14	0.35	0.22	0.26	0.32	0.45
Slope	0.24	-0.11	0.21	0.09	-0.04	0.03
Restoration length	-0.04	0.14	-0.20	0.07	0.19	0.17
Project size	-0.15	0.04	-0.19	-0.05	0.01	-0.00
Time after restoration	-0.03	-0.32	0.28	-0.04	-0.20	-0.22
Land cover						
Artificial surface	-0.09	0.15	-0.12	-0.20	0.06	0.04
Agricultural areas	-0.30	0.16	-0.20	-0.07	0.10	0.15
Forest, semi natural areas	0.26	-0.02	0.35	0.22	0.01	-0.03
Wetlands	0.25	-0.16	-0.44	-0.01	0.02	-0.02
Water bodies	0.28	-0.06	-0.40	-0.24	0.26	0.27
Hymo PC1[1]	0.23	-0.02	-0.06	-0.07	-0.07	-0.12
Hymo PC2[1]	0.07	0.19	0.35	0.12	0.28	0.22
Hymo PC3[1]	0.05	-0.24	-0.20	0.06	-0.21	-0.31

[1] Hymo PCs represent the principal components of the assessed hydromorphological predictors. PC1 explained 30.7, PC2 17.5 and PC3 11.6 % of all variance in the hydromorphological variables. PC1 was dominated (factor loadings >|0.7|) by variables of the hydromorphological survey, PC2 represented hydromorphological variables at the scale of the mesohabitat and PC3 represented hydromorphological at the microscale.

Table 7-2: Median effect size (ln[R/D]) of species richness and diversity (Shannon index) for different predictor variables (country, river types, substrate type and main measure). 25 and 75 % percentiles are given in parentheses. Differences within predictor groups were tested with Kruskal-Wallis ANOVA by ranks. Significant differences (p<0.05) are indicated by bold median values.

Predictor	Macrophytes		Helophytes		Hydrophytes	
	Richness	Diversity	Richness	Diversity	Richness	Diversity
Country						
AT (n=2)	0.11 (-0.88-1.1)	0.66 (-0.02-1.34)	1.45 (1.39-1.5)	1.12 (1.07-1.17)	-0.60 (-1.2-0)	-0.70 (-0.7--0.7)
CH (n=2)	2.87 (2.64-3.09)	0.16 (-0.08-0.4)	0.35 (0.00-0.69)	0.00 (0.00-0.00)	0.20 (-0.51-0.92)	0.08 (-0.35-0.51)
CZ (n=2)	-0.57 (-0.77--0.37)	0.00 (0.00-0.00)	1.67 (1.39-1.95)	0.00 (0.00-0.00)	0.00 (0.00-0.00)	
DK (n=2)	-0.37 (-0.41--0.34)	0.06 (-0.04-0.16)	0.11 (-0.18-0.41)	0.14 (-0.05-0.34)	-0.05 (-0.22-0.12)	0.03 (-0.06-0.12)
DL (n=2)	0.63 (0.12-1.13)	0.07 (-0.06-0.21)	0.46 (-0.18-1.1)	0.29 (-0.08-0.67)	0.05 (-0.15-0.26)	0.02 (-0.09-0.13)
DM (n=2)	-0.48 (-0.96-0.00)	0.25 (0.13-0.37)	0.55 (0.00-1.10)	0.04 (0.00-0.09)	0.47 (0.36-0.59)	0.25 (0.15-0.35)
FI (n=2)	0.75 (0.69-0.81)	0.04 (-0.02-0.10)	0.59 (0.41-0.77)	0.27 (0.18-0.36)	-0.31 (-0.62-0.00)	-0.15 (-0.22--0.07)
NL (n=2)	-0.26 (-0.61-0.09)	0.07 (-0.11-0.25)	0.45 (-0.2-1.1)	0.18 (-0.11-0.48)	-0.49 (-0.69--0.29)	-0.48 (-0.68--0.28)
PL (n=2)	-0.28 (-0.46--0.1)	0.2 (0.14-0.25)	0.65 (0.61-0.69)	0.29 (0.29-0.29)	0.45 (0.20-0.69)	0.29 (0.10-0.49)
SE (n=2)	0.54 (0.36-0.72)	0.14 (0.07-0.22)	0.30 (0.00-0.60)	0.11 (0.00-0.22)	0.80 (0.77-0.83)	0.36 (0.34-0.38)
River type						
Mountain (n=10)	0.35 (-0.77-1.1)	0.05 (-0.02-0.37)	**0.94** (0.41-1.39)	0.04 (0-0.36)	0.00 (-0.51-0.36)	-0.07 (-0.35-0.35)
Lowland (n=10)	0.00 (-0.41-0.36)	0.15 (-0.04-0.22)	**0.50** (-0.18-0.69)	0.25 (-0.05-0.34)	0.16 (-0.22-0.69)	0.11 (-0.09-0.34)
Substrate type						
Gravel (n=12)	0.53 (-0.57-0.95)	0.08 (-0.01-0.29)	0.73 (0.20-1.39)	0.04 (0.00-0.29)	0.00 (-0.26-0.68)	0.15 (-0.22-0.35)
Sand (n=8)	-0.22 (-0.43-0.10)	0.15 (-0.05-0.23)	0.51 (-0.18-0.90)	0.29 (-0.07-0.41)	-0.02 (-0.26-0.23)	0.02 (-0.18-0.12)
Main measure						
Widening (n=9)	0.00 (-0.77-1.10)	0.13 (0.00-0.37)	1.10 (0.69-1.39)	0.00 (0.00-0.67)	0.00 (0.00-0.36)	0.14 (-0.35-0.35)
Remeandering (n=3)	-0.41 (-0.61-1.13)	-0.04 (-0.06-0.25)	-0.18 (-0.18-1.10)	-0.05 (-0.08-0.48)	-0.22 (-0.69--0.15)	-0.09 (-0.68--0.06)
Instream measures (n=4)	0.39 (-0.12-0.75)	0.04 (-0.06-0.13)	0.41 (0.10-0.59)	0.26 (0.04-0.35)	-0.14 (-0.45-0.06)	-0.15 (-0.25-0.02)
Anastomosing (n=1)	-0.10	0.14	0.69	0.29	0.20	0.10
Floodplain reconnection (n=1)	-0.46	0.25	0.61	0.29	0.69	0.49
Flow restoration (n=2)	0.54 (0.36-0.72)	0.14 (0.07-0.22)	0.30 (0.00-0.60)	0.11 (0.00-0.22)	0.80 (0.77-0.83)	0.36 (0.34-0.38)

7.4 Discussion

Nutrient input and impoverishment in hydromorphology properties have been suggested as important drivers of species and diversity loss of macrophytes in streams (Steffen et al. 2013). Restoration measures are hence expected to reverse this process. In contrast to our hypothesis and Lorenz et al. (2012), restoration did generally not result in higher species richness and diversity of macrophytes when comparing degraded and restored sections. However, the life form showing a positive response to restoration was helophytes. Helophytes, corresponding to the life form emergent hydrophytes, with representatives such as *Phragmites australis*, *Alisma plantago-aquatica* and *Caltha palustris*, grow in the riparian and littoral zone on exposed or submerged soils (Mäkirinta 1978). Hence, restoration measures targeting e.g. removal of bank fixation as done in Lorenz et al. (2012) or widening and remeandering as done in several restoration projects investigated in this study should favour helophytes. Indeed, in our study, widening was the restoration measure that had a significant effect on the effect size of helophytes. Our study confirmed the importance of stream type (lowland versus mountain) for the effect size of helophyte response. In accordance with Lorenz et al. (2012), the effect size was higher in mountain compared to lowland streams. However, this might also be due to the fact that most widening projects were located in mountain rivers.

The response of hydrophytes depends most likely on the type of restoration measures. Whereas instream measures such as boulder placement (Finland) showed a negative effect size of both hydrophyte richness and diversity, flow restoration (Sweden) showed the opposite effect. The non-significant hydrophyte effect sizes of richness and diversity might hence be due to the range of different restoration measures performed in combination with a lack of replicates per stream type.

Time after restoration is an important predictor of macrophyte responses to restoration (Baattrup-Pedersen et al. 2000). Our study was performed on average 10 years (range 3-16 yrs.) after restoration. This time period was on average 5 years in Lorenz et al. (2012) that found significant restoration responses of several macrophyte life forms. Hence, time after restoration can most likely not explain the low macrophyte response in our restored sections.

The effect of local and reach-scale restoration measures might be overruled by upstream and non-restoration related river characteristics (Lorenz & Feld 2013). In our study, macrophyte-related effect sizes were not related to upstream land use. Also, and in contrast to earlier findings (Baattrup-Pedersen & Riis 1999), effect sizes were in our study not substrate dependent.

In conclusion, we suspect that any potential further responses of macrophytes to the here studied restoration measures were masked by the diversity of performed measures. Different restoration types could even have opposite effects on macrophytes. Remeandering and widening could potentially increase the effect size for lemnids (floating macrophytes) due to lowering stream flow, whereas flow restoration targeting an increase in flow should have a negative effect on lemnids. Indeed, however based on low species number, the restored sections DL_R1 (widening) and DK_R1 (remeandering) showed positive effect sizes for leminds whereas the Swedish site where an entire hydropower dam was removed showed a negative effect size for this life form.

7.5 References

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8. Ground beetles

8.1 Introduction

Many species of ground beetles are found in riparian areas worldwide. These species often have special adaptations to the specific environmental conditions associated with these habitats, for example, to be able to withstand flooding. In general, ground beetles strongly respond to changes in microhabitat conditions (Rainio and Niemelä, 2003; Lambeets et al., 2009), especially vegetation density or substrate composition (Lambeets et al., 2008) as they mainly live in the soil or above ground. In terrestrial habitats, ground beetles are well-known indicators for management and disturbance (Kotze et al., 2011), e.g. habitat changes in agricultural areas and forests (Lövei & Sunderland, 1996; Kromp, 1999; Niemelä et al., 2007) and hydrological conditions in grasslands (Gerisch et al., 2006; Follner & Henle, 2006). For riparian habitats, several studies point out the importance of near-natural flooding disturbance for the presence of characteristic carabid assemblages (e.g., Van Looy et al., 2007; Lambeets et al., 2008).

However, riparian and aquatic habitats have been altered by man since the Middle Ages to benefit from provisioning, regulatory and cultural services, e.g., navigation, waste water treatment and recreation (Millennium Ecosystem Assessment, 2005). Particularly in densely populated areas, such as Central Europe, most rivers have suffered from straightening, bed and bank fixation, the loss of lateral and longitudinal connectivity and altered flow and sediment regimes. More than 50% of European river are affected by hydromorphological pressures (EEA, 2012), and 90% of floodplain forests have disappeared, whereas remaining fragments are often in a critical condition (UNEP-WCMC, 2000). Negative effects on riparian communities have been detected (Greenwood et al. 1991; Godreau et al. 1999; Tockner et al. 2008). For example, for German floodplains, agricultural landuse in the floodplain, the construction of dikes, river training and impounding inhibit natural flooding dynamics and are considered as major threats for carabid beetles (Müller-Motzfeld, 2000; Reißmann et al., 2005).

The European Water Framework Directive (WFD; Directive 2000/60/EC) aims to improve the ecological status of all ground and surface waters in the European Union according to chemical, hydromorphological and biological conditions. Thereby, the improvement of river hydromorphology is one of the top measures (EEA, 2012). This has led to a large increase in the number of restoration projects, a large number of restoration projects have been implemented in recent years (e.g. in North America, Europe, Japan and Australia) (Lake et al., 2007, Feld et al., 2011) and this number is still increasing.

Although the number of empirical studies increased over the last 20 years, studies dealing with effects of restoration on riparian communities are rare (Wortley et al., 2013). It may also due to the fact that the WFD focuses exclusively on aquatic organism groups. These studies mainly act on reach-scale (Jähnig et al., 2009, Januschke et al., 2014) or on single rivers (Lambeets et al., 2008, Günther & Assmann (2005). Although they suggest general responses of carabid beetles to restoration, e.g. increased species richness and the presence or a higher number of riparian specialists, investigations at larger spatial scales, e.g. comparing several rivers are nearly missing. Studies about the relationship between restoration effects on carabid beetles and the type of restoration measures applied (e.g., widening vs. instream measures) and the effect of the length of restored sections are still missing.

Most probably, ground beetles benefit from restoration measures which create favourable riparian habitats like open gravel bars. Riparian carabid species are well adapted to dynamic flood-prone areas and have a strong flight and, therefore, dispersal ability (Desender, 2000) which makes them fast colonizers (Lambeets et al., 2008). Therefore, measures that include river widening and the creation of flood-prone riparian areas should generally have strong positive effects on ground beetles as flooding dynamics and disturbances will re-create the pioneer habitats. In contrast, measures that focus on the improvement of instream habitats, remeandering of the watercourse or reconnecting existing waters such as oxbows may not have effects on carabids, as the channels are still fixed and flood-prone riparian areas or erosional zones are missing.

Furthermore, it could be suggested that large restoration projects with high channel dynamics are more effective in generating these specific habitats compared to small projects where natural channel dynamics are restricted. The larger area of suitable habitats may contain more viable populations of different species.

A comparative analysis of hydromorphological restoration measures and restoration effects on ground beetles at the European scale is still missing, although general patterns of positive effects can be derived from the performed studies. Therefore, we investigated the ground beetle assemblage compositions collected in riparian zones of 20 paired restored and degraded reaches of rivers throughout Europe (see detailed description in Chapter 1.2 and Annex B). We tested, if changes in total species richness, richness of riparian specialists, Shannon Wiener diversity and community composition could be related to differences in river characteristics, restoration type and extent and habitat availability.

We expect that:

• In general, morphological river restoration increases richness and diversity of ground beetle assemblages,

• restoration measures which aim at widening and create pioneer habitats are more successful in increasing ground beetle richness and diversity than other measures,

• restoration measures in gravel-bed rivers naturally characterized by high hydraulic power, which creates and maintains pioneer habitats, are more successful in increasing ground beetle richness and diversity than measures in sand-bed rivers,

• ground beetle assemblages were mainly influenced by habitat characteristics (e.g. the presence of open bars) and restoration project characteristics (e.g. restoration type, age of restored sections) and to a lesser extent by river characteristics (e.g. altitude).

8.2 *Methods*

Study sections and sampling methods

The study sections and reaches as well as sampling methods for the ground beetles are described in Annex B and Chapter 2.7.

Calculation of parameters

We calculated species richness and Shannon Weaver diversity (Shannon & Weaver, 1949) of ground beetles for each sample section. For all recorded species, we compiled information about their ecological preference and counted the number of species with a preference for sparsely vegetated river banks and shores, wetlands or wet to moist forests. Preferences were derived from the carabids.org database (Homburg et al., 2013).

To quantify effects of restoration on ground beetles, we calculated two types of effect sizes for richness, Shannon diversity and the number of species with defined habitat preferences. We used: (1.) Pairwise calculation of the difference between each pair of restored and degraded section, and (2.) a modified version of the response ratio Δr developed by Osenberg et al. (1997).

The original formula given by Osenberg et al. (1997) is:

$$\Delta r = \ln\left(\frac{X_R}{X_D}\right),$$

whereas X_R is the species richness or diversity of the restored section and X_D of the non-restored section. Thereby, values > 0 denote a positive effect (e.g. increase of richness or diversity), and negative values a negative effect. This formula was not appropriate for our data (e.g., for diversity or the proportion of species with habitat preferences) as we had 0-values for the degraded sections and could, therefore, not calculate the response ratio. Instead, we calculated a modified response ratio Δr_m according to the following formula:

$$\Delta r_m = \ln\left(\frac{(1+X_R)}{(1+X_D)}\right).$$

Environmental parameters

We chose a set of environmental variables related to river, habitat and restoration project characteristics (Table 8-1).

River characteristics contained altitude of the restored reach, slope of the restored channel, mean discharge, mean channel width and overall bed coarseness. Project characteristics were the extent of restoration (large and small restoration projects differing in respect to restored reach length and/or restoration intensity), the type of restoration measure (widening or others, e.g. flow restoration, remeandering, instream measures) and the time since restoration in years. Habitat characteristics included the number of mesohabitats present in the sampling section and their proportional cover. Cover was estimated in a maximally 10 m wide strip of all riparian zone. If the bank's width was smaller, sampling area only spanned the area of the high-water level. Originally, also data including the whole wetted width of the sampled sections based on hydromorphological survey was included (transect method), but this gave the same results as using the data for the riparian zone only. Therefore, only the former is used in the analyses.

Table 8-1: Environmental variables classified according to river, project and habitat characteristics (10 m wide strip of riparian area).

Variable class	Variable
River characteristics	Altitude (m above sea-level)
	Slope (%)
	Discharge (m^3/s)
	River width (m)
	Bed coarseness (cobbles-gravel or sand bed)
Project characteristics	Restoration extent (large vs. small restoration projects)
	Restoration type / measure (widening, other)
	Time since restoration (year)
Habitat characteristics	Mesohabitat presence (total number)
	Sparsely vegetated mineral bars and banks (%)
	Woodland (%)
	Herbaceaous vegetation (%)

Data analyses

First, we tested if there was an overall positive effect of restoration on ground beetle richness and diversity by comparing richness and diversity of all restored (R) and all degraded (D) sample sections (group and pairwise comparison of R vs. D). Second, we tested, if restoration effects depend on restoration extent by comparing richness and diversity of all large (R1) and all small (R2) restoration projects using absolute values (group and pairwise comparison of R1 vs. R2). Additionally, we analysed effects sizes based on richness and diversity in terms of differences between values (R2-R1) and using the response ratio modified after Osenberg et al. (1997) calculated for each pair of restored and degraded section. Third, we tested if effect sizes differ between projects which mainly aimed at river widening (usually affecting aquatic, semi-terrestrial, and terrestrial areas) and projects which applied other, less extensive measures mainly affecting the river channel itself (instream measures, flow restoration, remeandering, anastomosing, similar to the grouping of measures in Chapter 5 on invertebrates). Fourth, we investigated in more detail which habitats should be restored and which biological ground beetle metrics benefit or are suitable to assess restoration effect. To parameterize the typological differences among European rivers, we combined river characteristics (bed type, slope, altitude, discharge, width) as super-ordinated variables into one parameter. Thereby, we extracted a composite descriptor using principal components analysis (PCA), which was used for further analyses. Principal components that explained a significant non-random part of the variation were retained (broken-stick model; Jackson 1993). Correlations for each variable with Principal Component 1 were calculated to derive its main descriptors. Sample scores of the sections on the significant principal component were used as a new quantitative variable in the subsequent ordinations. Subsequently, we analysed the relationship between the effect sizes for the biological metrics (Table 8-2) and environmental variables (river, habitat and restoration project characteristics) using redundancy analysis (RDA). As biological metrics we chose commonly applied metrics, e.g. richness, diversity, community composition, and metrics related to habitat preferences as we expected restoration benefits for species specialized on river bank, wetlands and wet woodland. To determine which part of the variation in effect sizes can uniquely be attributed to changes in certain environmental variables and which part is shared with other variables, variance partitioning was applied to test if the different groups of variables are related to each other. Forward selection (Monte Carlo permutation test, 9,999 permutations, P values Holm corrected) was used to retain only those variables which significantly contributed to the variance explained by each of the groups. Ordinations were carried out using Canoco 5.03 (Ter Braak & Šmilauer, 2012).

Table 8-2: Classification and description of parameter types used in the redundancy analyses to analyse the relationship between biological metrics (response variables based on ground beetles) and environmental characteristics (explanatory variables); R = restored reach, D = degraded reach.

		Parameter type	Parameter description	Value calculated as
Response variables	Biological metrics	Richness	Total species richness	Response ratio modified after Osenberg et al. (1997)
		Diversity	Shannon Wiener index value	Response ratio modified after Osenberg et al. (1997)
		Community composition	Species composition samples	Euclidian distance between R and D
		Habitat preference	Number of river bank specialists	Response ratio modified after Osenberg et al. (1997)
		Habitat preference	Number of wetland specialists	Response ratio modified after Osenberg et al. (1997)
		Habitat preference	Number of wet woodland specialists	Response ratio modified after Osenberg et al. (1997)
Explanatory variables	River characteristics	River characteristics	PC-1 (hydraulic gradient)	R
	Restoration characteristics	Restoration	Restoration extent (large/small)	R
		Restoration	Restoration measure (widening/others)	R
		Restoration	Time since restoration (year)	R
	Habitat characteristics	Habitat richness	Number of mesohabitats in riparian area	R – D
		Habitat composition	Sparsely vegetated mineral bars and banks (%)	R – D
		Habitat composition	Woodland (%)	R – D
		Habitat composition	Herbaceous vegetation (%)	R – D

8.3 Results

Overall effect of restoration on ground beetles (R1 and R2 pooled)

In total, we found 130 ground beetle species; species richness per sample section varied between one and 25 species. Overall (pooling large and small restoration projects), simply comparing the two groups of restored and degraded sections did not reveal any difference (Figure 8-1). Mean species richness was 8 species per section and mean diversity about 1.7 in both degraded and restored sections.

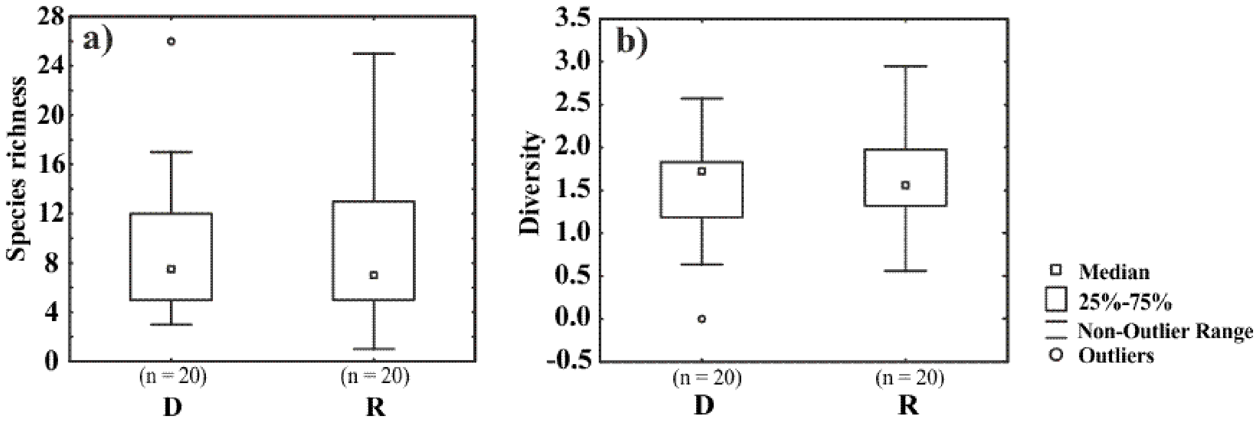

Figure 8-1: Comparison of a) species richness and b) diversity of ground beetles in degraded (= D) and restored (= R) sections.

However, if the restored sections were compared to the corresponding degraded sections (pairwise comparison), mean effect sizes for ground beetle richness were significantly larger than zero (t-test, $p<0.05$). This pattern applies for both methods used to quantify restoration effect size (the difference of the 20 pairs of restored and corresponding degraded sections based on absolute values as well as the relative response ratios). Restoration increased ground beetle richness by about 3 species (max 12). In contrast, restoration had no overall positive effect on diversity.

Differences of restoration effect in large and small projects (R1 vs. R2)

Group comparison did not reveal significant differences between the small (R2) and large (R1) restoration projects and the degraded sections (D1, D2) in respect to species richness and diversity (Figure 8-2).

However, a paired comparison (calculating effect sizes of restored sections compared to the corresponding degraded sections) showed that mean effect sizes were significantly larger than zero (t-test, $p<0.05$) for the large but not for the small restoration projects (Figure 8-1 a). This was not a general pattern as species richness decreased in some restored sections (expressed as negative effect sizes). In case of the small restoration projects (R2), ground beetle richness decreased in four out of the ten sampling sections. Moreover, differences of effect sizes between large and small restoration projects were not statistically significant, neither for comparing the two groups R1 and R2 (Mann-Whitney U test, $p = 0.52$, $n = 20$), nor for a paired comparison (R1 compared to corresponding R2 section, Wilcoxon-Matches Pairs test, $p = 0.55$, $n= 10$). In contrast to species richness, restoration did not increase diversity in none of the subsets of large and small restoration projects. We observed both, an increase and decrease in the single restoration projects and mean values for large and small restoration projects were not different from zero (Figure 8-1b). Moreover, the range of richness and diversity changes did not differ.

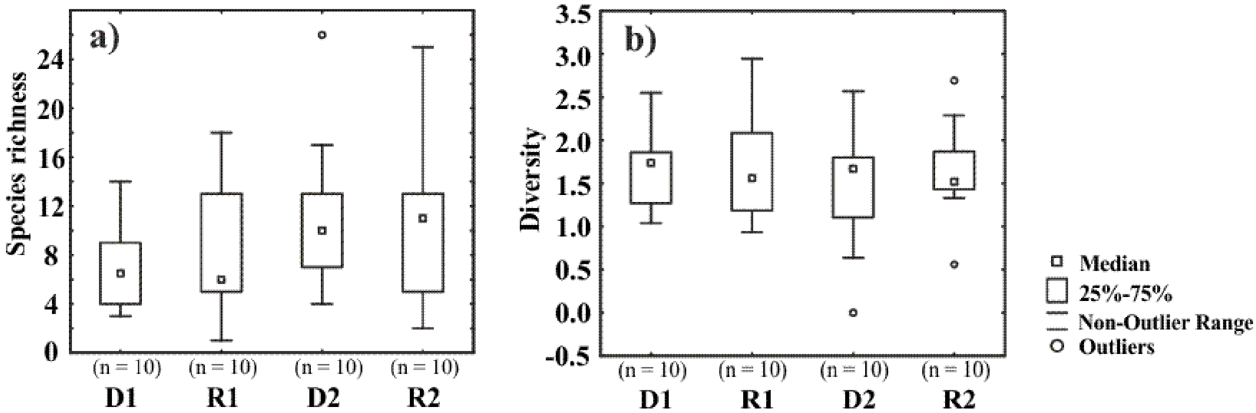

Figure 8-2: a) species richness and b) diversity of ground beetles in large and small restoration projects and paired degraded sections (R1 = large projects, D1 = degraded sections belonging to R1; R2 = small projects, D2 = degraded sections belonging to R2).

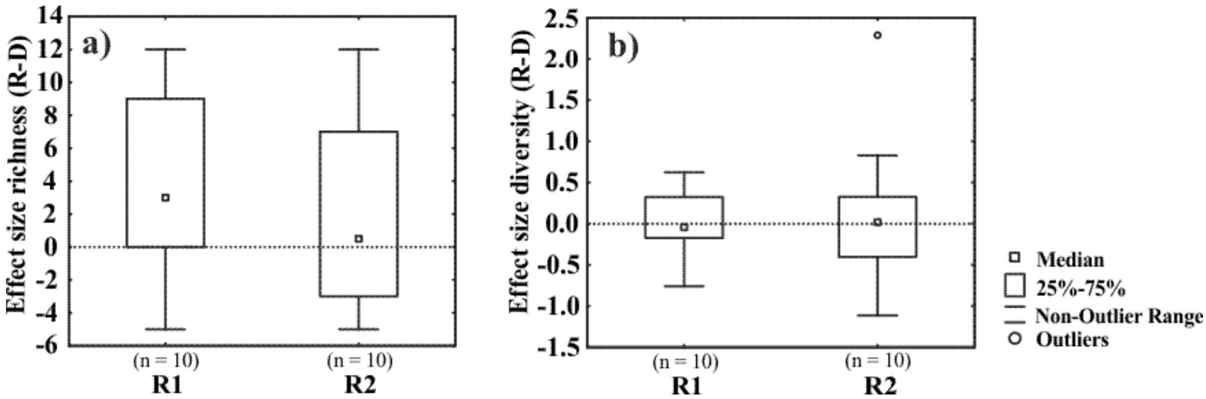

Figure 8-3: Comparison of effect sizes based on a) species richness and b) diversity of ground beetles in large (R1) and small (R2) restoration projects and paired degraded sections; effect sizes were pairwise calculated as the difference between restored and degraded (R1-D1 and R2-D2).

The effect of restoration on richness and diversity was standardized using the response ratio (Osenberg et al. 1997), which allows to compare the resulting relative values between metrics (in contrast to the absolute differences, see Figure 8-4). Restoration effect on richness was larger compared to diversity and differences were significant for the large restoration projects (Mann-Whitney U test, $p<0.05$) but none of the mean response ratios was larger than zero (t-test, $p>0.07$).

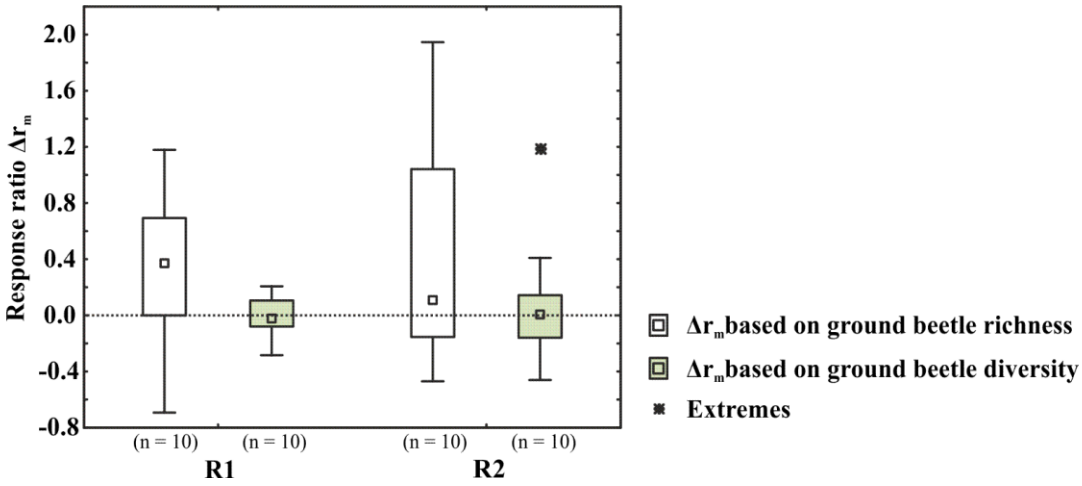

Figure 8-4: Comparison of response ratios modified after Osenberg et al. (1997) based on species richness and diversity of ground beetles in large and small restoration projects and paired degraded sections (R1 = large projects; R2 = small projects); effect sizes were pairwise calculated.

General relationship of ground beetle richness and diversity and the type of restoration measures

Effect sizes based on ground beetle richness and diversity differed significantly between restoration measures (Kruskal-Wallis test, $p < 0.01$), which aimed at widening, and other restoration measures, e.g. improvement of instream habitats, flow restoration and remeandering (Figure 8-5).

In all restored sections where widening was applied as a restoration measure, species richness was increased by around seven species, and in most of the sections diversity was increased as well (t-test, $p<0.05$, $n=11$). In contrast, other restoration measures led predominantly to decreased ground beetle richness and diversity.

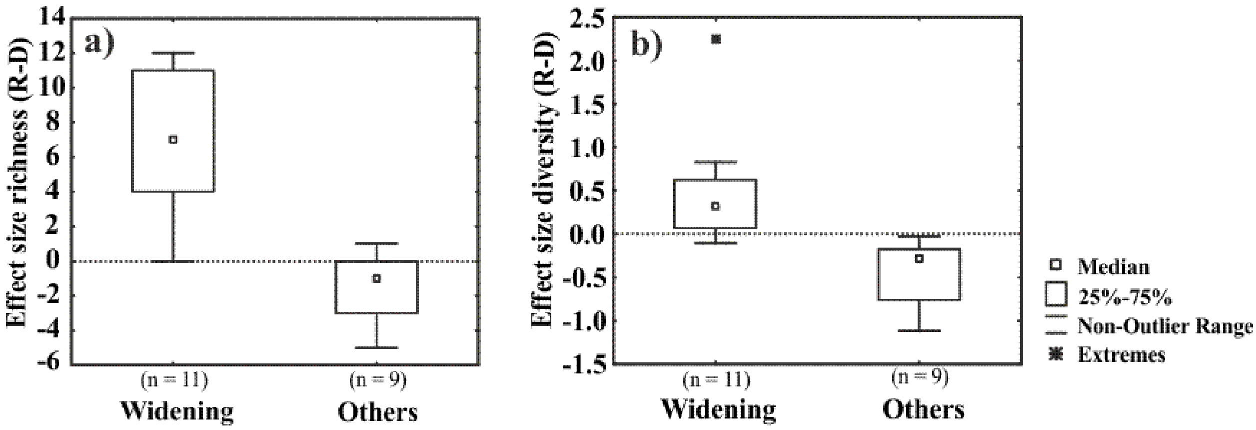

Figure 8-5: Comparison of effect sizes based on a) species richness and b) diversity of ground beetles in restored sections with widening and restored sections with other measures (e.g., improvement of instream habitats); effect sizes were pairwise calculated as the difference between restored and degraded (R1-D1 and R2-D2).

A comparison of the response ratios modified after Osenberg et al. (1997) showed that effects of widening on ground beetle richness were strong, whereas effects on diversity were comparatively low (Figure 8-6).

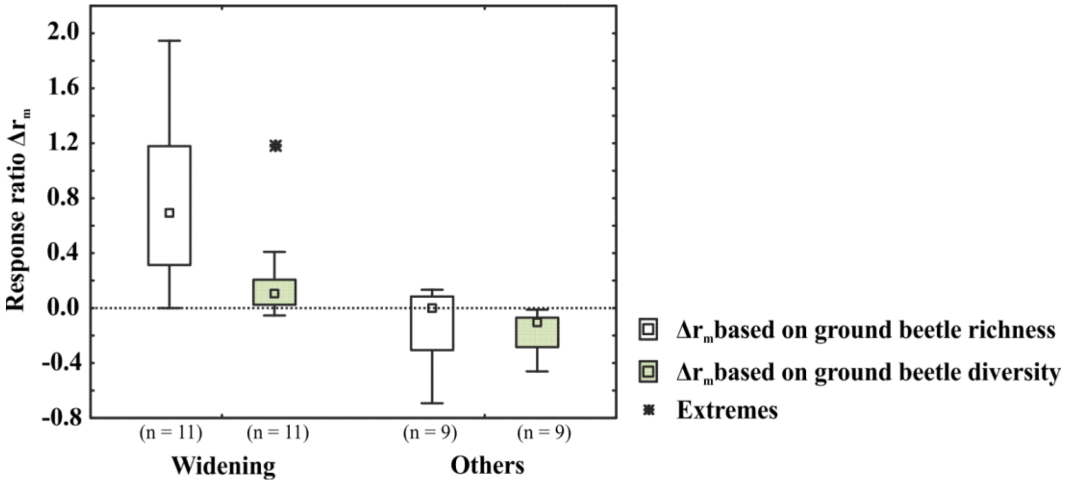

Figure 8-6: Comparison of response ratios modified after Osenberg et al. (1997) based on species richness and diversity of ground beetles in restored sections with widening and restored sections with other measures (e.g., improvement of instream habitats).

Moreover, response ratios differ between river types (gravel vs. sand bed river, Figure 8-7) with patterns similar to differences between restoration measures. Restoration measures in gravel-bed rivers mainly increased ground beetle richness, whereas there were no clear effects in sand-bed rivers. Effects on diversity were low, both in gravel- and sand-bed rivers, whereas richness tend to decrease in restored sections of sand-bed rivers.

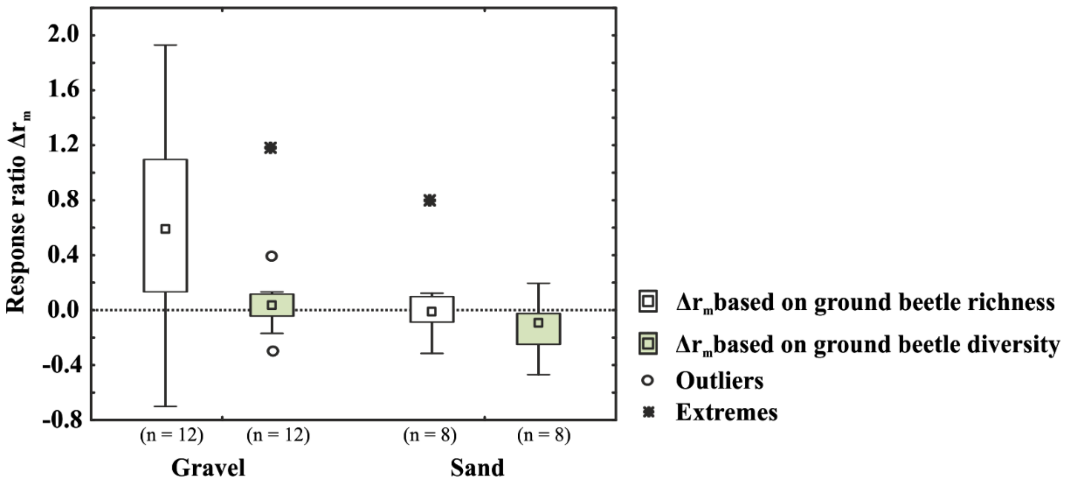

Figure 8-7: Comparison of response ratios modified after Osenberg et al. (1997) based on species richness and diversity of ground beetles in restored sections of gravel- and sand-bed rivers.

Relationship between biological metrics and environmental variables

The main descriptors for typological differences among European rivers, calculated by correlations of river characteristics (bed type, slope, altitude, discharge, width) with Principal Component 1, were a combination of coarseness of the riverbed, altitude and slope (Table 3). Sample scores of the sites on the significant principal component were used as a new quantitative variable in the subsequent ordinations, which we defined as hydraulic gradient, ranging from coarse-bed, high gradient rivers to low gradient rivers with a sand bed.

Table 8-3: Results of the principal component analysis. Based on the loadings of each variable on the significant principal components (PC) expressed as Pearson correlation coefficients, it main descriptors (r > 0.8; in bold) were determined; significance of the principal components: * significant, n.s. not significant.

River characteristics and PC parameters	Pearson correlation coefficient (r)		
	PC-1	PC-2	PC-3
Altitude (m above sea-level)	**-0.8**	-0.5	0.2
Slope (%)	**-0.8**	0.6	-0.0
Discharge (m^3/s)	-0.5	0.3	0.7
Channel width (m)	-0.5	0.3	0.6
Bed coarseness (gravel vs. sand-bed)	**-0.8**	-0.1	-0.5
Eigenvalue	13.7*	4.4^{ns}	2.7^{ns}
Broken-stick eigenvalue	10.2	5.7	3.5
% of total variance explained	61.3	19.7	11.9

Richness

There was a significant relationship between the variation in effect sizes for total richness and the environmental variables (F = 5.4, P = 0.006). Based on forward selection the variability in the effect size for total ground beetle richness was explained best by the application of widening as a restoration measure (category "restoration"), the difference in proportion of woody vegetation along the river banks between the restored and degraded reach (category "habitat") and the hydraulic gradient (category "river type"). Variance partitioning showed that widening accounted for 21.4% of the effect size variability, which was not shared with the other variables (Figure 8-8 a). The difference in proportion of woody vegetation accounted for another 10.9%. Shared variance between these two variables accounted for 21.7% of the variability in effect size; another 21.7% was shared by all three variables. No significant unique contribution of the hydraulic gradient was detected; variability explained by this parameter was in the first place shared with the other two variables and to a lesser extent complementary. In total, 26% of the variation remained unexplained. Highest effect sizes for total species richness were obtained in high-gradient rivers, where widening as a restoration measure was applied and where the proportional cover of woody vegetation is lower in the restored reach in comparison to the degraded reach (Figure 8-8 b).

Diversity and community composition

Variation in the effect of restoration on species diversity (floodplain scale: $F = 1.0$, $P = 0.475$; river bank scale $F = 1.4$, $P = 0.305$) could not be explained by the differences in environmental variables between the restored and degraded sections. No environmental variables explained the variation in the differences between community composition in the degraded and restored sections on floodplain scale ($F = 1.5$, $P = 0.270$) and river bank scale ($F = 1.8$, $P = 0.190$).

Species with specific habitat preferences

There was a significant relationship between the variation in effect sizes for those ground beetle species preferring sparsely vegetated river banks and the environmental variables ($F = 7.7$, $P = 0.002$). Forward selection resulted in the same set of variables as for total richness, except that habitat characteristics was represented by the difference in proportion of sparsely vegetated banks with coarse substrate between the restored and degraded sections. Variance partitioning showed that widening accounted for 20.0% of the effect size variability which was not shared with the other variables; the unique contribution of the other two variables was not significant (Figure 8-8 c). Nonetheless, variance shared between all variables was 30.2%, and widening shared another 9.6% with sparsely vegetated banks. Therefore, the variability in effect size for river bank specialists was explained by the variables in a similar way. In total, 36.3% of the variation remained unexplained. For riparian specialists, highest effect sizes are again obtained in high-gradient rivers, where widening as a restoration measure was applied and where the proportional cover of sparsely vegetated banks is higher in the restored reach in comparison to the degraded reach (Figure 8-8 d). The effect sizes for the number of species preferring either wetlands or wet to moist forests were not related to any of the environmental variables (wetland preference $F = 0.9$, $P = 0.537$; wet to moist forest preference $F = 0.6$, $P = 0.774$).

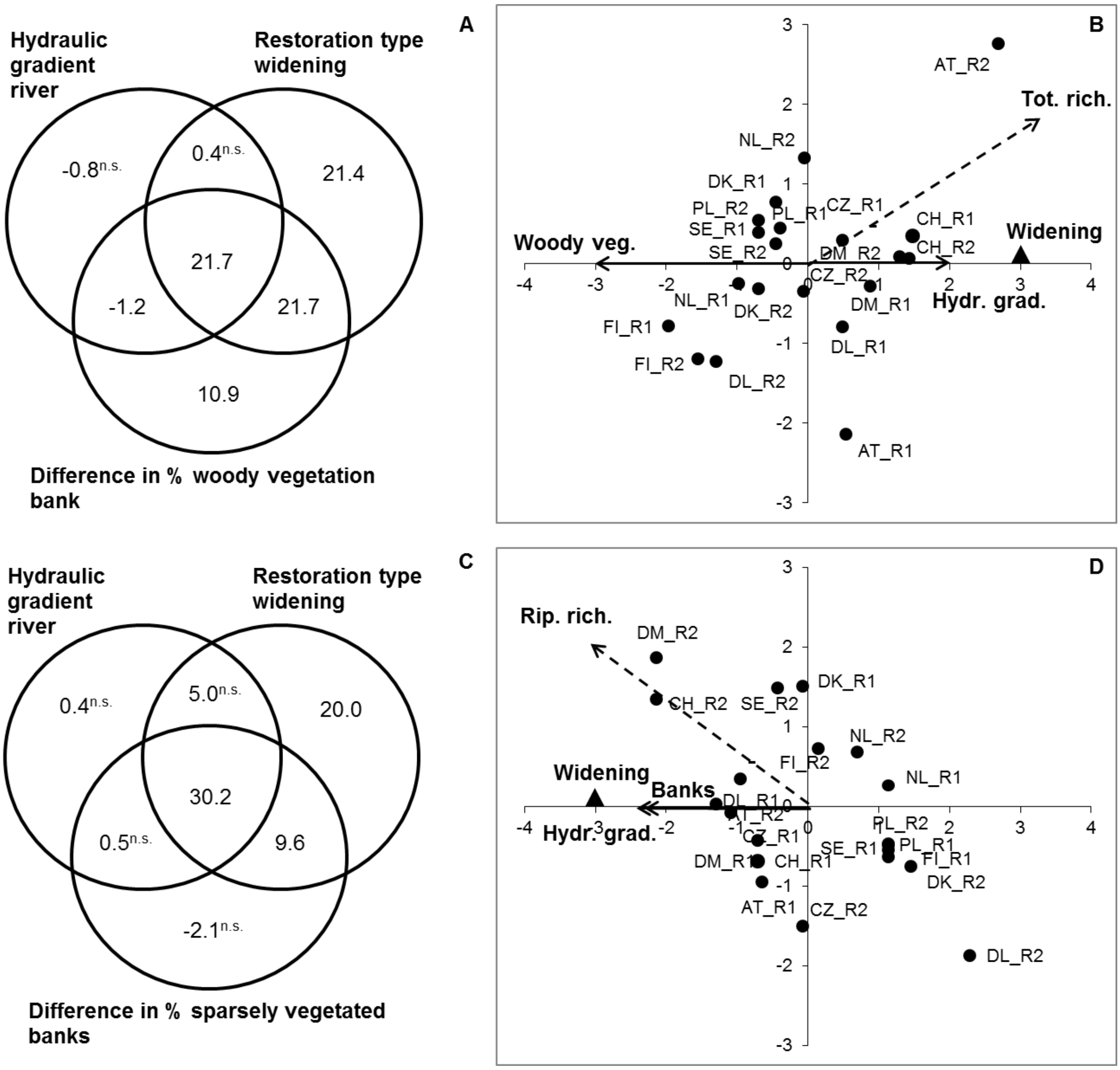

Figure 8-8: Results of redundancy analysis performed with response ratios modified after Osenberg et al. (1997) for total ground beetle richness: A. variance decomposition for river, project and habitat characteristics; B. triplot of significant environmental variables, response ratios and sample scores on axis 1 and 2 and richness of species with a preference for sparsely vegetated river banks (= riparian richness) (C., D.) and environmental data for paired restored and degraded sections.

8.4 Discussion

General restoration effects on ground beetle richness and diversity

Habitat diversity in riparian areas is a precondition for high ground beetle richness mainly due to the presence of primary habitats as open sand and gravel bars (Bonn et al., 2002; Van Looy et al., 2005) as also shown in our study. Accordingly, we found a strong relationship between species richness and a specific habitat type, the open pioneer stage covered by sparse woody vegetation. Thereby increased species richness was not related to the number of habitat types. Restoration increased species richness using pairwise comparisons of degraded and restored sections, thus supporting our first hypothesis. However, we did not find a common pattern of higher richness in restored sections compared to all degraded sections contrasting the results of Januschke et al. (2011), who found low ground beetle richness in degraded and high richness in restored sections for 24 hydromorphological restored sites in three Federal States of Germany. In our case studies, riparian areas of degraded sections differed in their morphological status. Although most of them were characterized by fixed embankments, some degraded sections had shallow vegetated banks which were affected by flooding. These additional habitats offer more niches for ground beetles than the fixed embankments in most of the degraded sections.

The importance of restoration project, habitat and river characteristics

Ground beetle assemblages were mainly influenced by the restoration type (widening), habitat characteristics (the presence of open bars) and to a lesser extent by river characteristics (high hydraulic gradient).

As expected, widening was an effective restoration measure leading to strong increases of ground beetle richness, and to a lesser extent of diversity, in all investigated sampling sections. This measure creates lateral connectivity between the river and its floodplain. At best, it leads to a habitat mosaic of different successional stages containing open bars and shallow vegetated banks at the shoreline and higher elevated and less flooded banks with woody vegetation. As ground beetle assemblages contain many species with selective habitat preferences according to vegetation density, substrate and moisture conditions (Van Looy et al., 2005), the created habitat mosaic offers many niches for them. Increased species richness of ground beetles due to river bank widening was also found by Van Looy et al. (2005), Zulka (2008), Jähnig et al. (2009) and Januschke et al. (2014).

Similar to investigations of Günther & Assmann (2005) and Sadler et al. (2004), dynamic habitats at the shoreline, e.g. open gravel and sand banks, were crucial habitats enhancing species richness by increasing the number of riparian specialists. Species with a strong preference of open banks are well-adapted to dynamic riparian areas underlying flood disturbance because of their small body size, flattened bodies and well-developed wings and flight-muscles (Desender & Turin, 1989). Due to their high dispersal ability (Desender, 2000), they can colonize newly generated habitats rapidly and intensively. Thereby, main factors for a successful dispersal of riparian ground beetle species and their colonization of new habitats are flooding disturbance, increasing the rate of dispersal, and a natural distribution of appropriate habitat patches (Bates et al., 2006). Accordingly, effects of restoration on ground beetles were independent from the longitudinal extent or the age of restoration. The presence of pioneer-habitats is more important than their area, as apparently dispersing individuals are able to detect these small patches very fast and to complete their life cycle there.

For both, species richness and richness of riparian specialists, highest effect sizes were obtained in high-gradient rivers where widening as a restoration measure was applied and where, due to restoration, woody banks were decreased (in case of species richness) or open bars were increased (in case of riparian specialists). Thus, the hydraulic gradient of rivers has an additional effect in combination with the restoration measure type and habitat characteristics. Mountain rivers are naturally characterized by high hydromorphological dynamics which lead to sediment erosion and deposition and therefore to a shifting habitat mosaic due to flooding (Ward et al., 2002). Therefore, river widening sets a starting point for self-reinforcing processes in direction of a habitat mosaic and the maintenance of open bars.

For low-gradient rivers (lowland rivers), we did not detect restoration effects on ground beetles, neither on richness and diversity nor on species with specific habitat preferences. Two reasons could be supposed. First, many lowland rivers in Europe were straightened for agricultural landuse in the floodplain (EEA, 2012). The loss of natural floodplains implies impoverished source populations of wetland and wet forest species, which would be mainly typical for lowland floodplains (Bonn et al., 2002; Gerisch et al., 2006), due to the fact that well-developed marshes dominated by sedges or old wet forests are very rare. Second, investigated measures, which are applied in lowland rivers, may not create suitable habitats such as wetlands or wet forests with required inundation frequency as they were mainly instream measures or aimed at flow restoration. However, the presence of wetland species strongly depends on hydrological parameters such as inundation and groundwater depth (Gerisch et al., 2006) and particularly lowland rivers are naturally characterized by a strong lateral connection between the river and its floodplain. Moreover, it is well-known that habitat turnover in terms of bank erosion and lateral migration of the channel takes longer time spans in lowland than in mountain rivers (Richards et al. 2002). Hence, the development of near natural habitat conditions in restored lowland rivers such as open mud and sand bars, wetlands and wet forests may need more restoration effort than the initiation of self-reinforcing processes.

However, implications of our study are limited as we focused on riparian areas on a 10m-strip and did not investigate whole floodplains. Therefore, it could also be supposed that carabids typical for wetlands were not captured as they probably occur further away from the river channel, in marshes surrounding backwaters in the floodplain.

8.5 Conclusions

For ground beetles restoration success in terms of an increase in total species richness and richness of habitat specialists could be achieved primarily by measures creating pioneer patches, for example by river widening, which result in more open banks. For instream fauna, shading is regarded an important factor increasing ecological quality of the river, because it dampens temperature fluctuations, provides food and offers habitat structure. As a result, the development of woody riparian vegetation is a commonly applied restoration measure. Our study shows that this measure could be counterproductive for the specialist riparian carabid fauna, which requires open habitat. If the purpose of restoration includes enhancing the conditions for floodplain biota, some open areas should remain present. This could be well combined with providing enough shade for the instream fauna, because our results also indicate that the mere presence of open habitats is more important than its area. Further research should focus on determining optimal conditions of such pioneer habitats, e.g. the maximum vegetation cover tolerated. We used 25% cover as the maximum proportion to which the habitat was regarded as 'open', but we do not know what the optimal proportion is. Also plant species composition could be important. Woody vegetation encroachment resulted in a decrease in species richness, but maybe there is also an effect of different species of herbaceous vegetation.

Widening is an appropriate measure in mountain rivers as flooding maintains created habitat mosaics and characteristic dynamic riparian areas. For lowland rivers, we suggest that the creation of shallow riparian patches is of particular importance as habitat turnover and the development of habitat diversity takes longer timespans due to less power of the river. Suitable restoration measures should aim on a strong lateral connection between the river and its floodplain to guarantee inundation frequency and low distance to groundwater to create habitats which are important for typical ground beetles for wetlands and wet forests. Thereby, longer time spans for recolonization should be mentioned as catchments of lowland rivers are often highly degraded implying impoverished species pools of typical species.

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9. Floodplain vegetation

9.1 *Introduction*

Streams and rivers are used by humans for many different purposes (e.g. for hydroelectric power, agriculture, recreation, industry) with negative consequences for stream biota due to their vulnerability to flow modifications, habitat degradation and water pollution (Poff et al. 1997, Malmqvist and Rundle 2002, Dudgeon et al. 2006). To reverse the negative effects of habitat degradation on stream ecosystem structure and function, a large number of restoration projects have been implemented in recent years (e.g. in North America, Europe, Japan and Australia) (Lake et al. 2007, Feld et al. 2011). Restoration measures vary from small restoration projects aiming at improving specific instream conditions, e.g. by introducing small riffle habitats, to large projects aiming to re-establish features characterizing natural systems, e.g. by reintroducing meanders or removing dikes which increases the intensity of processes operating in the land-water ecotone, (Palmer et al. 2010) and restorations that target disturbances at the catchment level, e.g. by minimizing sediment inputs originating from forest harvesting activities across the catchment (Bohn and Kershner 2002).

Small restoration projects are, by far, the most common practice (Bond and Lake 2003, Palmer et al. 2010) but a majority of these projects have not led to recovery of biodiversity. While restoring local habitat structures is a prerequisite for species to establish at a site, factors that operate at larger spatial scales may also constrain restoration success (Palmer et al. 1997, Poff 1997, Bond and Lake 2003). For example, large scale disturbances such as past and present landuse in a region or catchment may limit the regional species pool available for locally restored sites, and thus, the desired effects of restorations may be absent (Harding et al. 1998, Bohn and Kershner 2002, Lake et al. 2007, Palmer et al. 2010). Likewise, the dispersal of organisms is a regional process influenced by the natural hydrological regime of streams (i.e. the timing, duration and magnitude of flow and the rate of change in flow), which may control the distribution and abundance of species in restored sites (Poff et al. 1997). Such large scale factors may be of overriding importance for the success of local restorations implemented at individual sites or reaches (Bond and Lake 2003). Moreover, there is a natural time lag between restoration and recolonization which depends on factors such as dispersal abilities of the organisms and distance to source populations from the restored site (Gore and Milner 1990, Mitsch and Wilson 1996, Huxel and Hastings 1999).

In the present study, we have a unique opportunity to examine how plant communities in European floodplains respond to restoration measures of different extent. We expect that restoration extent will be particularly important for structural and functional characteristics of the floodplain plant community since species living here are adapted to and dependent on a variety of large-scale processes (e.g. flooding and sedimentation) that occur under natural variations in flow regime (Gregory et al. 1991). Repeated waterlogging and flooding of river banks create and sustain high habitat heterogeneity and may also lead to the development of distinct vegetation belts according to hydrological gradients ranging from wet to dry conditions as one move further away from the shoreline (Gregory et al. 1991). The hydrologic regime is furthermore considered to be of overriding importance for the transport and deposition of plant propagules (Mahoney and Rood 1998, Merritt and Wohl 2002) that may establish in the areas. As many as >100 million propagules can be transported in free-flowing reaches in a single growing season and some disperse long distances (hundreds of km) with the water before being deposited further downstream (Nilsson and Grelsson 1990, Andersson et al. 2000, Merritt and Wohl 2006). However, the dispersal and retention of propagules depend both on the presence and characteristics of flooding events (Boedeltje et al. 2004, Gurnell et al. 2007) and whether structures of the stream reach and local habitat allows propagules and sediments to be deposited at river banks (Engström et al. 2009). Considering that the flow regime is of great importance for the existence of highly diverse riparian communities, targeting large-scale features of stream channels that can influence flood-related processes are likely to have a larger effect on riparian plant communities than restoration measures which only target local instream habitat structures.

Specifically, we examine and compare the structural and functional response of the floodplain vegetation to large restoration projects (i.e. the reconstruction of meanders and removal of dikes) vs. small projects (i.e. the reintroduction of coarse substrates into the stream channel; Chapter 2 and Annex B) and investigate to what extent restoration outcomes are influenced by the underlying stream or river typology (e.g. altitude and discharge), catchment land use and time since restoration. Large restoration projects are likely to mediate more intense and diverse hydrological interactions across the land-water ecotone (e.g. by flooding and sedimentation processes) that will improve conditions for dispersal and establishment of diverse floodplain communities. We therefore expect a greater effect of restoration on species richness, trait diversity and trait composition in floodplains where long river sections have been restored compared to floodplains of short restored sections. Additionally, we expect that there will be a time-dependency in the recovery of the biological communities (Lake et al. 2007) that may be prolonged when recolonization occurs from available source communities in the landscape (e.g. Mitsch and Wilson 1996, Lake et al. 2007, Nilsson et al. 2014).

9.2 Materials and methods

Study sections and sampling methods

The study sections and reaches as well as sampling methods for the floodplain vegetation are described in Annex B and Chapter 2.8. The sampling of the floodplains situated in the Netherlands (NL) differed from the standard sampling procedure described above as a greater plot size (3 m^2 instead of 0.25 m^2) was used with only one plot per observed vegetation type (following Oberdorfer, 1983, 1992; Ellenberg 1991). However, the vegetation in NL sites was highly homogeneous making the area of the sample plot less important and we therefore consider the data comparable to those from the other European regions.

Table 9-1: Short explanation, abbreviations (Abbrev) and references of the 18 traits used.

Trait	Categories (if any)	Abbrev	Expl	Unit/Range	Ref
Leaf dry matter content		LDMC		mg/g	1
Canopy height		CH		m	1
Seed mass		SM		mg	1
Specific leaf area		SLA		mm^2/mg	1
Buoyancy		BYC		1-100	1
Seed number per plant		SNP		N/plant	1
Ellenberg light		GBEL	A	1-9	2, 3
Ellenberg moisture		GBEF	A	1-12	2, 3
Ellenberg nutrients		GBEN	A	1-9	2, 3
Ellenberg temperature		ET	A	1-9	2
Grime's competitiveness		GC	B	0-1	4, 5
Grime's stress tolerance		GS	B	0-1	4, 5
Grime's ruderality		GR	B	0-1	4, 5
Dispersal type	Autochor	DLT_AU_p	C	0-1	1
	Hemerochor	DLT_HE_p	C	0-1	1
	Meteorochor	DLT_ME_p	C	0-1	1
	Nautochor	DLT_NA_p	C	0-1	1
	Zoochor	DLT_ZO_p	C	0-1	1
Plant growth forms	Chamaephyte	PGF_CH_pa	D	0/1	1, 6
	Geophyte	PGF_GE_pa	D	0/1	1, 6
	Hemicryptophyte	PGF_HE_pa	D	0/1	1, 6
	Hydrophyte	PGF_HY_pa	D	0/1	1, 6
	Phanerophyte	PGF_PH_pa	D	0/1	1, 6
	Therophyte	PGF_TH_pa	D	0/1	1, 6
Age of first flowering	1-5 years	AFF1_B15	D^1	0/1	1
	> 5 years	AFF1_O5	D^1	0/1	1
	≤ 1 year	AFF1_W1	D^1	0/1	1
Plant life span	Annuals	PLS1_A	D^2	0/1	1
	Perennials	PLS1_P	D^2	0/1	1
Seed bank type	Long-term persistent	SBT1_LTP	D^2	0/1	1
	Short-term persistent	SBT1_STP	D^2	0/1	1
	Transient	SBT1_T	D^2	0/1	1

Explanations (Expl): A = indicator value, B = functional signature value, C = proportional expression of trait category, D = presence/absence of trait expression with superscript 1 = several categories were reported for a species, the lowest reported trait category was used, and superscript 2 = several categories were reported for a species, the longest reported trait category was used

References (Ref): 1 = Kleyer et al. (2008), 2 = Ellenberg et al. (1991), 3 = Hill et al. (2000), 4 = Grime et al. (2007), 5 = Hunt et al. (2004), 6 = Raunkiaer (1934).

Diversity indices and community weighted means

All diversity and trait indices were calculated based on average Ord% values per study reach (see Chapter 2.8). We calculated taxon richness and Shannon diversity as indices of local taxonomical diversity. A number of traits and species indicator values were allocated to the encountered species (Table 9-1) and used to calculate trait-diversity and community weighted means (CWMs) for each individual trait. These included morphological traits (e.g. specific leaf area, seed mass, canopy height), Ellenberg indicator values (light, moisture, nutrients and temperature), plant life strategies (competitiveness, ruderality, stress tolerance), life history traits (e.g. age of first flowering and life span) and dispersal traits such as buoyancy and means of dispersal (e.g. autochor, zoochor, nautochor) (Table 9-1). For the categorical traits, we used two different approaches when assigning trait values. A proportional trait expression per category was calculated if species were known to commonly express more than one trait category (e.g. means of dispersal). This was done by dividing each trait expression with the sum of all reported trait expressions per species giving species a proportional value between 0-1 for each trait category. For all other categorical traits, for which species was not to the same extent expected to express more than one category, we instead assigned a value of 0 (i.e. absent) or 1 (i.e. present) depending on if the trait was expressed for that specific species (Table 9-1).

Trait diversity and CWMs were then calculated using R package *FD* (Laliberté and Shipley 2011). The CWMs were calculated with the function *functcomp* as:

$$CWM = \sum_{i=1}^{n} p_i \times trait_i$$

where p_i is the relative contribution of species i to the community, and $trait_i$ is the trait value of species *i* (e.g. Lavorel et al. 2008).

Trait diversity (functional dispersion or FDis) was calculated with the function *fdisp* for each trait individually. FDis is a multidimensional functional diversity index that is weighted by species abundances (Laliberté and Legendre 2010). Thus, the most dispersed communities are composed of evenly distributed dissimilar trait categories (categorical traits) or trait values (numerical traits).

Finally, a response ratio (Δr) (Osenberg et al. 1997) for each diversity and trait metric was calculated per floodplain as:

$$\Delta r = ln\left(\frac{Nr}{Nd}\right)$$

where *Nr* is the metric value for the restored reach and *Nd* is the metric value at the control (degraded) reach. Response ratios allowed us to combine and compare the results from the different floodplains and identify general pattern in the relative change in the metric values (i.e. the metric values for degraded reaches relative to the restored reaches) across floodplains.

Statistical analyses

To determine whether species composition differed between the European regions, Non Metric Multidimensional Scaling (NMDS) followed by analysis of similarities (ANOSIM, based on both Bray-Curtis and Sørensen dissimilarities) were performed in statistical software PAST (Hammer et al. 2001). NMDS is an ordination method based on ranked distances between samples which is highly suitable for ecological data that typically contain numerous zero values (Minchin 1987). NMDS was based on the algorithm by Taguchi and Oono (2005). ANOSIM is a non-parametric test of significant difference between two or more groups, based on any distance measure (Clarke 1993).

For each metric and restoration type we tested whether Δr was significantly different from zero (i.e. higher or lower than zero) using one sample t-tests. A significant result was interpreted as a consistent and detectable change in the metric value in degraded vs. restored reaches across floodplains. The response ratios were also regressed against the four predictor variables altitude, discharge, % agriculture in the catchment and time elapsed after restoration. Predictor variables were $\log_{10}$ transformed before analyses to approximate normal distribution if necessary.

Response ratios were compared between short and long restored sections by means of two sample t-tests to elucidate whether the response of specific metrics differed and restoration effect did depend on restored reach length.

9.3 Results

General regional differences between paired restored sections (R1, R2)

The NMDS ordination showed clear differences in species community structure between the European regions where the ten paired long restored (R1) and short restored (R2) river sections were located (compare Chapter 1.2) both when ordinations were based on Bray-Curtis dissimilarities (ANOSIM; R=0.76, p<0.05) and Sørensen dissimilarity index (ANOSIM; R=0.91, p<0.05) (Figure 9-1).

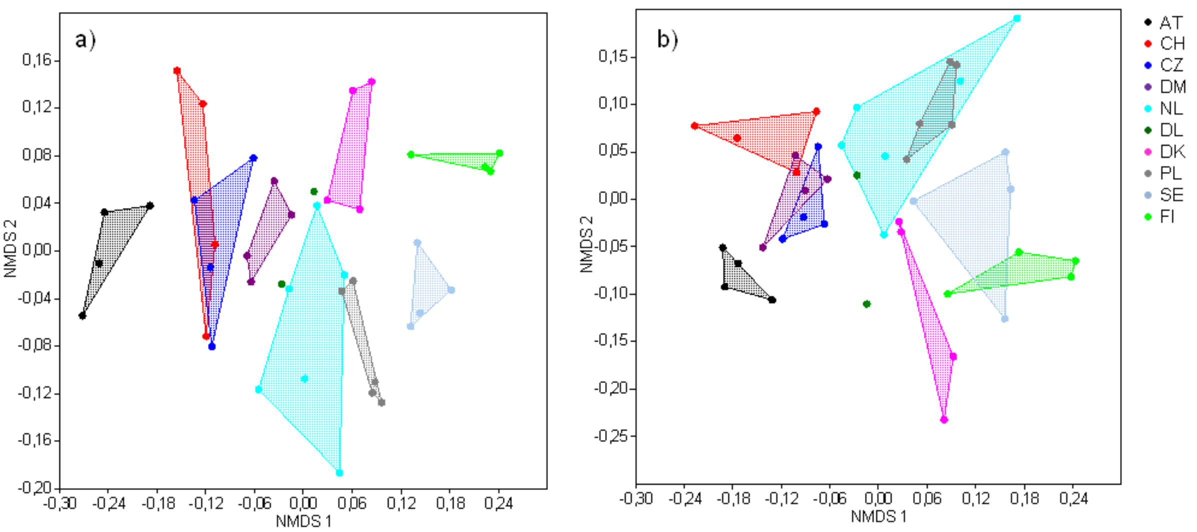


Figure 9-1: NMDS ordination plots showing differences in plant community composition between the 10 European regions in which the paired R1/R2 restored sections are located. The ordinations are based on a) Sørensen dissimilarity index (stress: 0.23) and b) Bray-Curtis dissimilarity index (stress: 0.24).

Overall effect of restoration on floodplain vegetation (R1 and R2 pooled)

Overall (pooling large and small restoration projects), restoration had a significant effect on floodplain vegetation as indicated by the mean response ratios of several metrics/traits which were significantly different from zero (Table 9-2). Restoration had a small negative effect on leaf dry matter content. Moreover, the share of short lived species was higher (significantly larger share of annual species and therophytes) in the restored reaches compared to the corresponding unrestored degraded sections. These species benefit from disturbances, indicating that restoration increased the frequency of disturbances like flooding. However, there was no overall positive effect of restoration on species diversity.

Table 9-2: Significant effects of restoration on the floodplain vegetation metrics investigated when data on large and small restoration projects were pooled.

Trait / metric	p-value	Mean response ratio
Leaf dry matter content (LDMC)	0.040	-0.04
Therophytes (PGF_TH_pa)	0.001	+0.82
Annuals (PLS1_A)	0.019	+0.94
Perennials (PLS1_P)	0.002	-0.05

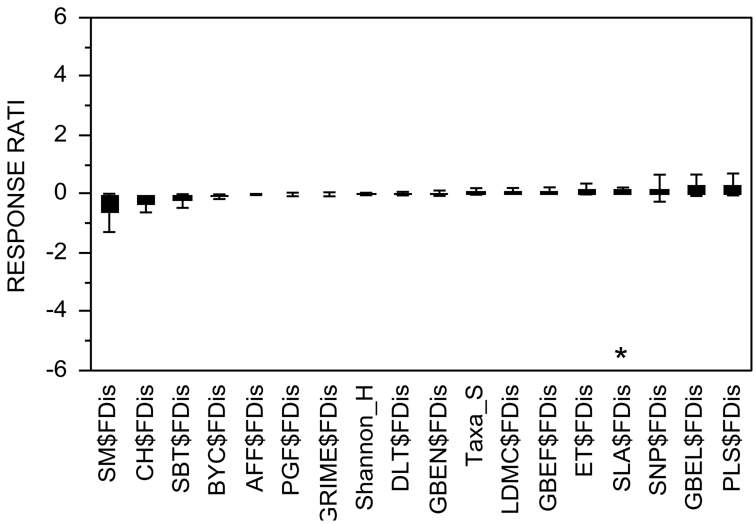
Differences of restoration effect in large and small projects (R1 vs. R2)

There was no significant difference in the response ratios between large and small restoration projects on any of the diversity indices or CWMs used (two-sample t-test; $p>0.05$). Besides this group comparison (to test if there were general differences in restoration effect between the two groups of large and small restoration projects despite regional differences), we originally planned to account for regional differences by limiting direct comparisons to the corresponding pairs of large and small restoration projects (see Chapter 1.2). However, data on floodplain vegetation was not available for one of the small restoration projects, and hence hampering a pairwise comparison.

Responses of diversity and trait composition to restoration in large and small restoration projects (R1 and R2 analysed separately)

We only found very limited effects of restoration analysing response ratios describing diversity characteristics of the floodplain vegetation. In the small restoration projects, restoration had a significant effect on the dispersion of specific leaf area (SLA) and the mean response ratio was significantly larger than zero (mean $\Delta r = 0.18$; one-sample t-test; $p<0.05$) (Figure 9-2), whereas no significant differences were found for any diversity indices in the large restoration projects. In contrast, we found that some of the response ratios of CWM's changed significantly depending on restoration extent (Figure 9-3) suggesting that trait composition responded to restoration and that the responses were dependent on restoration extent. We found a significant and relatively large decrease in chamaephyte-CWM (mean $\Delta r = -1.05$) and a relatively large increase in therophyte-CWM (mean $\Delta r = +1.17$) in large restoration projects (one sample t-test; $p<0.05$) (Figure 9-3 B). In contrast, we detected a small decrease in perennial-CWM (mean Δr = -0.05) and a relatively large increase in annual-CWM (mean Δr = +1.2) in small restoration projects (Figure 9-3 A).

A: R2 (small restoration projects)



B: R1 (large restoration projects)

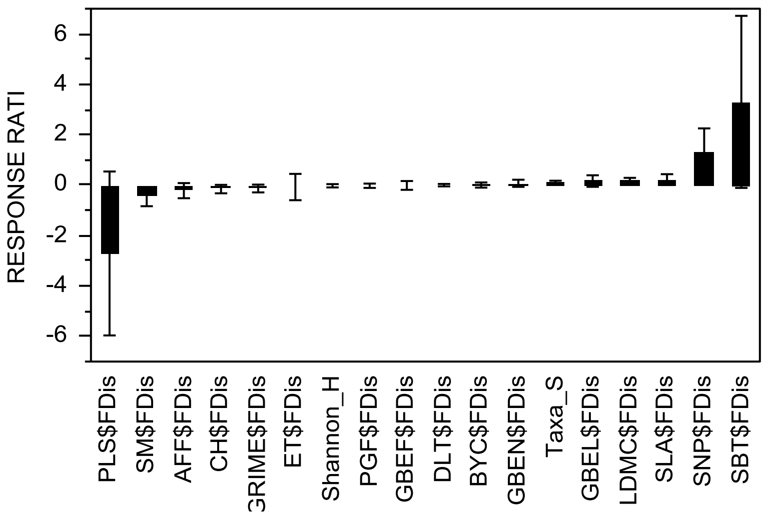
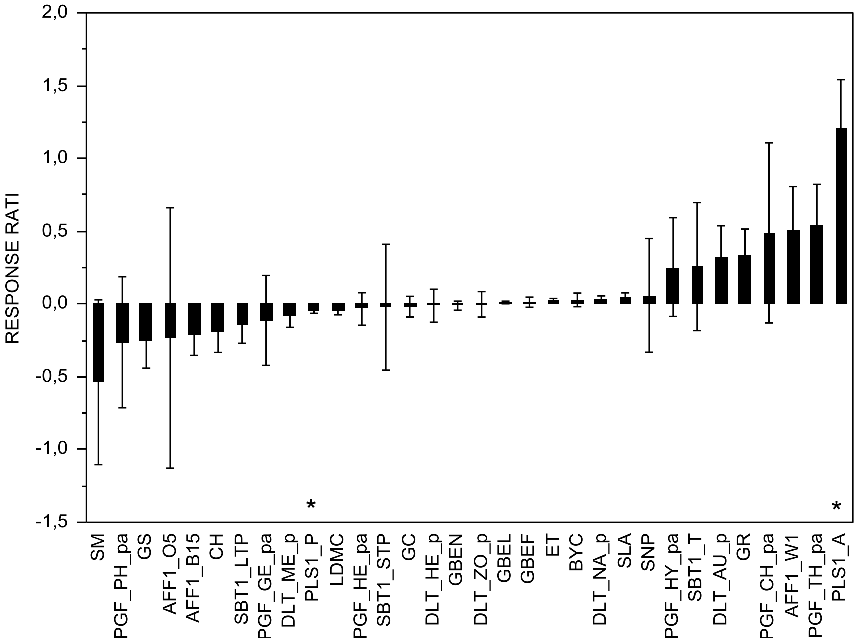


Figure 9-2: Mean (± 1SE) response ratios of diversity metrics across floodplains of small (R2; N=11) (a) and large restoration projects (R1; N=10) (b). Taxa_S = taxon richness, Shannon H = Shannon diversity, X$FDis = functional diversity for trait X. *= mean response ratio of the metric is significantly different from zero (p<0.05).

A: R2 (small restoration projects)



B: R1 (large restoration projects)

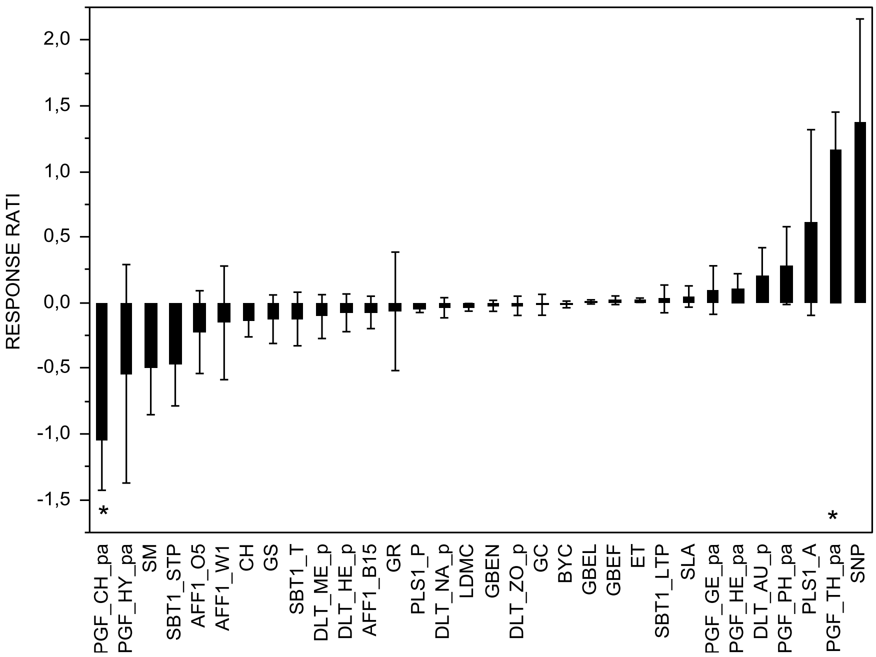


Figure 9-3: Mean (± 1SE) response ratios of trait CWMs across floodplains of small (R2; N=11) (a) and large restoration projects (R1; N=10) (b). *= mean response ratio of the metric is significantly different from zero (p<0.05).

Environmental drivers of responses of diversity and trait composition to restoration

The typology (altitude and discharge) of the stream/river played a significant role for the outcome of the restoration (Table 9-3). Here we focus mainly on strong relationships being those with Adj R^2 > 0.40 and/or relationships that were significant for both, small and large restoration projects. In both, small and large restoration projects, we found that response ratios of the dispersion of Ellenberg moisture values and plant life strategies were positively related to altitude. This suggests that, as altitude increases, restoration is more likely to promote the coexistence of plants with different moisture preferences and life strategies (competitors, stress-tolerants and ruderals). Additionally, altitude was strongly and positively related to the response ratio of phaenerophyte-CWM in small restoration projects (Adj R^2 =0.50), whereas response ratios of geophyte-CWM and long-term persistent seed bank-CWM were strongly and positively related to altitude in large restoration projects (Adj R^2 = 0.45 and Adj R^2 = 0.40, respectively).

Table 9-3: Adjusted R^2 values of correlations between environmental variables related to typology (altitude and discharge) and response ratios of community weighted means (CWM) and diversity indices in small (R2) and large (R1) restoration projects. Only traits with at least one significant correlation are shown. Light grey cells indicate a significant negative relationship and dark grey cells indicate a significant positive relationship. Shannon H = Shannon diversity, X$FDis = functional dispersion for trait X.

		Altitude		Discharge	
		R2	R1	R2	R1
CWM	BYC				0.48
	DLT_AU_p		0.33		
	DLT_ZO_p		0.37		
	ET			0.34	
	GBEF			0.29	
	GR				0.34
	GS				0.49
	SBT1_LTP		0.40		
	PGF_GE_pa		0.45		
	PGF_PH_pa	0.50			
	PLS1_A	0.39			
	SM			0.41	
	SLA		0.38		
Diversity	Shannon_H			0.33	0.59
	BYC$Fdis				0.40
	GBEF$Fdis	0.31	0.57		
	GRIME$Fdis	0.54	0.69		
	SM$Fdis			0.47	
	SBT$Fdis				0.40

In both, small and large restoration projects, the response ratio of Shannon diversity was consistently and positively related to discharge, suggesting that there is a higher probability of a positive effect of restoration on taxon diversity with an increasing discharge (stream size) (Table 9-3). In small restoration projects, we also found that discharge was strongly and positively related to the dispersion and CWM of seed mass (Adj R^2 = 0.47 and Adj R^2 = 0.41, respectively). In large restoration projects, we found that response ratios of stress tolerance-CWM, Shannon diversity and the dispersion of buoyancy was strongly and positively related to discharge, whereas response ratios of buoyancy-CWM and the dispersion of seed bank types were negatively related to discharge (Table 9-3).

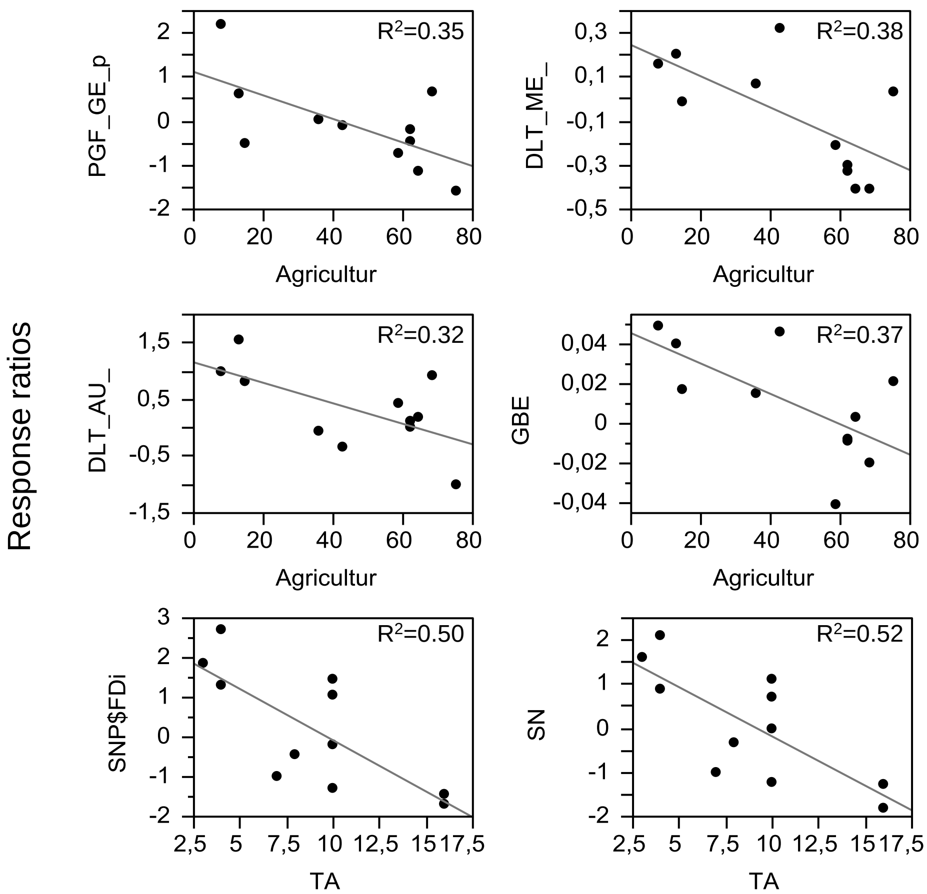


Figure 9-4: Significant correlations between predictor variables (% agriculture in the catchment and time after restoration (TAR)) and response ratios of community weighted means and functional dispersion (FDis) in short restored sections (R2). Adjusted R^2 values (R^2) are reported in the upper or lower right hand corner of each figure.

Besides typology, we also found that time after restoration (TAR) and agricultural land use in the catchment played a significant role for the outcome of the restoration projects expressed in terms of diversity and trait composition of the floodplain communities and, additionally, that the significance and strengths of the relationships differed between restoration types. In small restoration projects, response ratios of both CWM and dispersion of seed numbers per plant were strongly and negatively related to TAR (Adj R^2=0.52 and Adj R^2=0.50, respectively; Figure 9-4). Significant relationships between %

agriculture and response ratios of autochor-, meteorochor-, Ellenberg light- and geophyte-CWM's were also detected in small restoration projects (Figure 9-4). In large restoration projects, stress tolerant-CWM was strongly and negatively related to TAR (Adj R^2=0.63) (Figure 9-5) and we also found moderate-strong positive relationships between TAR and response ratios of taxon richness (Adj R^2=0.26; P=0.078), Ellenberg N-CWM (Adj R^2=0.27, P=0.069) and competitor-CWM (Adj R^2=0.28, P=0.067). Similarly to the pattern observed in small restoration projects, the response ratio of geophyte-CWM was negatively correlated with % agriculture in large restoration projects (Adj R^2=0.41; Figure 9-4 and Figure 9-5). We did not find any significant correlations between % agriculture and diversity indices in either small or large restoration projects (Figure 9-4 and Figure 9-5).

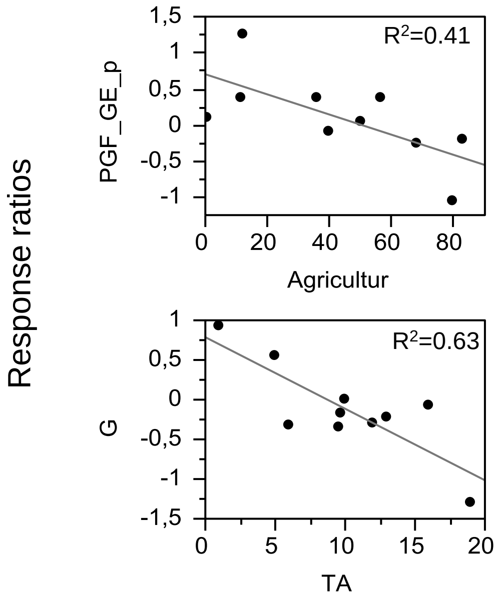


Figure 9-5: Significant correlations between predictor variables (% agriculture in the catchment and time after restoration (TAR)) and response ratios of community weighted means in long restored sections (R1). None of the diversity indices were significantly correlated with TAR and % agriculture and thus not shown. Adjusted R^2 values (R^2) are reported in the upper or lower right hand corner of each figure.

9.4 Discussion

Overall effect of restoration and differences between large and small restoration projects

In this study, we aimed to identify general patterns in community responses to restoration across large environmental gradients in European floodplains and to compare responses in floodplains where restoration targeted local scale factors such as instream substratum composition vs. large scale factors such as channel morphology features. Our results showed that not only species composition (Figure 9-1), but also the responses of plant diversity and trait composition to restoration varied substantially between the European regions. As a consequence of this high variability across the floodplains, only a few general responses to restoration could be detected. For example, we found a significant and relatively large decrease in the abundance of chamaephytes and a relatively large increase in the abundance of therophytes in large restoration projects compared to corresponding non-restored control sections. The Ranunkiær life forms indicate the position and degree of protection of the growth points (buds) from where the plant can regrow and is considered to be a good indicator of disturbance (e.g. Van der Maarel and Franklin 2013). Indeed, previous studies have found differences in life form composition between floodplains exposed to varying disturbance (flooding) regimes (Glaeser and Wulf 2009, Wang et al. 2014). For example, phaenerophytes have been shown to decrease while short lived species such as therophytes benefit from disturbances and thus increases in response to flooding (Glaeser and Wulf 2009, Wang et al. 2014). Further, Wang et al. (2014) investigated the effect of winter flooding following dam regulation in the Three Gorges Reservoir (China) and found the lowest proportion of chamaephytes in low elevation sites, which were greatly affected by flooding, and the highest proportions in high elevation sites, which were little affected by flooding. Thus, it is possible that the detected changes in the relative abundances of chamaephytes and therophytes are an effect of increased flooding disturbance of riverbanks in the large restoration projects.

We also observed a decrease in the abundance of perennials and an increase in the abundance of annuals in the small restoration projects. Annual growth form in plants is often associated with highly disturbed environments (e.g. due to fast growth rates and early and prolific seed set) while perennials usually dominate along more stable channels (e.g. Grime 1979, Pettit et al. 2001). Small restoration projects were not to the same extent as large restoration projects expected to mediate more intense and diverse hydrological interactions across the land-water ecotone (e.g. by flooding and sedimentation processes) and the higher abundance of annuals in response to small scale restoration was therefore unexpected. However, the observed response might reflect that reconstruction of instream habitat structures (e.g. riffles) have disturbed also the river banks and created open space for the establishment of annual plants.

Differences between river types

In our study, the response of Shannon diversity was positively correlated with discharge in both small and large restoration projects suggesting that high discharge may facilitate species establishment in restored sections independent of the extent of restoration. Discharge was also positively related to the response ratio of buoyancy dispersion and, at the same time, negatively related to the response ratio of buoyancy-CWM in large restoration projects. This suggests that (i) as discharge increase, species with contrasting floating capacity are more likely to colonize the restored sections and (ii) the apparent trait divergence in the restored sections is largely due to an increased abundance of species with low floating capacity in restored sections of larger rivers. This finding is in line with previous studies showing that mean and maximum dispersal distances of propagules increase with an increasing discharge (Nilsson et al. 2010) thereby facilitating the overall passive dispersal of plants by water. Consequently, the probability of encountering a community with wide buoyancy range (including species with low floating capacity) is likely to increase with increasing discharge provided that suitable habitats are available for colonization and establishment (e.g. created by restoration). Additionally, our findings indicate that a low proportion of riparian species with a high floating capacity establish in large restored rivers. Other studies have attributed similar effects to an impoverished sediment seed pool in agricultural landscapes (Baattrup-Pedersen et al. 2013a, Baattrup-Pedersen et al. 2013b). However, our study suggests that such effects are equally likely to be dependent on factors related to stream size (e.g. discharge).

The response ratios of Ellenberg moisture and plant life strategy dispersion increased with increasing altitude indicating that species with different moisture preferences and life strategies are more likely to coexist in restored, high altitude rivers. While it is difficult to assess the exact cause behind this result, we suggest two explanations to the observed altitudinal patterns that are not necessarily mutually exclusive. First, local environmental characteristics may be more diverse in high altitude restored sections. In this study, restoration projects in high altitude river sections mainly aimed at widening the stream channels that may have resulted in greater habitat diversity compared to instream and remeandering measures mainly applied in low altitude restoration projects and additionally, flashier hydrological regimes and greater stream flow in mountain areas can lead to faster habitat turnover with subsequent effects on the riparian communities. Second, greater regional diversity in high altitude river sections may allow for colonization of species with wider ranges in moisture preferences and life strategies. This could indicate that anthropogenic disturbances is generally less intense in high altitude regions (i.e. at larger geographical extents than individual catchments) which allows for higher regional species diversity and, hence, more incoming species to the areas (e.g. Fischer and Lindenmayer 2007).

Constraining effect of catchment pressures

Large scale anthropogenic disturbances and landscape fragmentation can limit the species pool available for local river sections (e.g. Fischer and Lindenmayer 2007) and weaken biological responses to local environmental conditions. Consequently, factors such as regional or catchment land use can be important determinants of restoration "success" (e.g. Palmer et al. 1997, Bond and Lake 2003, Lake et al. 2007). Here, we show that an increased agricultural intensity in the catchment directly affects the response of trait composition of the floodplain community (i.e. growth form, dispersal strategies and light preferences) suggesting that any changes in these traits that may develop in response to restoration risk to be masked in catchments with high agricultural intensity. Of particular interest we found that response ratios of geophyte-CWM decreased with increasing agriculture, which may reflect that geophytes respond negatively to grazing and in particular phosphorous availability - factors which are both highly associated with agricultural intensity (Dorrough and Scroggie 2008).

In contrary to our expectations, responses of all the diversity metrics used to characterize the floodplain vegetation were unrelated to catchment scale disturbances. However, disturbances occurring at larger spatial scales (stream network, bioregion) (Poff 1997) or even historical disturbances may more strongly affect present day diversity than present day disturbances (Harding et al. 1998, Lindborg and Eriksson 2004). Thus, while we provide some evidence that catchment scale disturbances influence community response when analysing trait composition, it is possible that expected effects on diversity are masked by factors operating at much larger spatial scales and/or over longer time periods (e.g. Nilsson et al. 2014).

Time lag of biological response

A time lag in the response of biota to restoration is always expected, since time is required for species to recolonize the restored sections. The time required may, however, differ between species (Trexler 1995) due to differences in dispersal potential or between target sections due to factors such as source community proximity and connectivity in the landscape (e.g. Lake et al. 2007).

Our study shows that time after restoration was able to predict changes in trait composition, but to a lesser extent changes in diversity. None of the diversity indices increased significantly over time in the restored sections even though a positive relationship of moderate strength (Adj R^2=0.26) with taxon richness was detected ($P<0.1$) in large restoration projects. However, while positive relationships between riparian plant richness and time after restoration have been observed in some systems, the time scale which we were investigating (1-20 years) may be far below what is needed for full community recovery (Nilsson et al. 2014). Further, effects on diversity may be delayed by time-lags in the recovery of environmental conditions that the restoration measures were unable to target (Hamilton 2012).

For example, in this study, we observed a strong decreased response of stress tolerance-CWM over time and a moderate-strong positive response of competitor- and Ellenberg N-CWM over time – a pattern which was not observed in small restoration projects. This suggests that an eutrophication of the restored sections in the large restoration projects may have occurred over time, possibly delaying or hindering the expected positive effects on plant diversity by increasing the dominance of productive taxa that may inhibit the establishment of others. Importantly, this pattern was not observed along the agricultural gradient suggesting that the restoration itself can lead to higher concentrations of nutrients over time.

While this observation may seem counterintuitive, the observed pattern can be explained by an altered hydrology. Internal releases of nutrients can be induced by an increased flooding of previously nutrient rich dry soils and sediments (Hamilton 2012) that may increase the productivity of the community following restoration. Moreover, nutrients can be stored in groundwater reservoirs for long time periods and thus, time lags in the response of stream water chemistry to restoration may be as long as decades (Hamilton 2012) which can further delay responses of the riparian vegetation. Another complementary explanation to this pattern is that diverse habitats might have been successfully created early after the restoration but that these habitats were not maintained over time. For example, discharges might not have been strong enough to sustain dynamic patches (due to e.g. flow regulations of adjacent river sections in the catchments or upstream river conditions) and thus, later successional stages (i.e. more competitive species) were gradually becoming more dominant in the restored reaches.

9.5 Conclusions

Our results showed that not only species composition but also the responses of plant diversity and trait composition to restoration varied substantially between the European regions. As a consequence of this high variability across the floodplains only a few general responses to restoration could be detected. These responses were related to changes in trait composition, while general effects on diversity were limited (small restoration projects) or absent (large restoration projects). Interestingly, the detected responses were specific to restoration extent (small vs. large restoration projects) and included changes in the relative abundance of traits previously known to respond to disturbance (e.g. plant growth form and life span). The apparent high variability in response ratios could be attributed to factors related to river typology (discharge and altitude), catchment scale disturbance and time after restoration which were strongly and significantly related to the plant community response to restoration. These strong relationships may partly explain why no general effects of restoration on diversity indices were detected. However, communities may also need considerable more time to establish and an increase in diversity may not be seen within the time frame investigated here. Finally, it is likely that additional confounding environmental and/or spatial factors that operate at much larger spatial scales than what was considered in this study further delays/masks expected effects on diversity.

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10. Stable isotopes

10.1 Introduction

Rivers are being restored worldwide with the aims to enhance biodiversity and ecosystem services (Pander & Geist 2013). Effects of river restoration can principally be monitored with a wide range of variables. Currently, parameters used to assess success or failure of restoration projects are mainly of structural nature, e.g. the composition of biological assemblages. In the context of the EU Water Framework Directive fish, phytoplankton and benthic fauna and flora are most commonly investigated and the response of these assemblages to hydromorphological restoration is well understood (Lepori *et al.* 2005, Jähnig *et al.* 2010, Sundermann *et al.* 2011, Lorenz *et al.* 2012, Friberg *et al.* 2014). Functional components, even though widely applied in ecological studies dealing with aquatic systems (e.g. vander Zanden & Rasmussen 1999, Hieber & Gessner 2002, Dudgeon *et al.* 2006 Fischer *et al.* 2005, Friberg *et al.* 2009, Gücker *et al.* 2009), are less commonly used for monitoring the effects of river restoration.

Implicitly, hydromorphological restoration of rivers aims at enhancing habitat diversity and aquatic-terrestrial linkages. Therefore, significant alterations of food web structure and trophic relationships can be assumed: A higher diversity of niches can contribute to more complex food webs, as a higher variety of resources is available to consumers enabling more trophic linkages. A stronger connection of river and floodplain, e.g. caused by a more shallow profile, will increase inundation frequency and thus the matter flow from land to water might be increased, as inundations may wash terrestrial nutrients into the river. A shallow profile will also enhance the transport of organic matter from the river to its floodplain. At the same time, it will also make aquatic prey more easily accessible to riparian predators.

Stable isotope composition of carbon and nitrogen ($\delta^{15}N$ and $\delta^{13}C$) are commonly used to study food web structure as they provide information on the material assimilated by organisms. $\delta^{15}N$ trophic fractionation changes about +3‰ between trophic levels (Minagawa & Wada 1984, Post 2002, McCutchan *et al.* 2003). Thus, it is generally used to calculate the trophic position of an organism. $\delta^{13}C$ trophic fractionation is less, changing only 0-1‰ from source to consumer (DeNiro & Epstein 1978, McCutchan et al. 2003). $\delta^{13}C$ also varies between different producers, thus it is often used as an indicator for sources within a food web, e.g. to identify if consumers are feeding on allochthonous or autochthonous sources. Hence, stable isotopes of carbon and nitrogen provide information on assimilated sources and trophic relations, which integrate spatial and temporal scales. Recently, a number of community-wide metrics have been introduced by Layman *et al.* (2007a) to gain more quantitative information from stable isotope data at the species or community level. These metrics have been used to quantify niche width and study the effects of impacts such as ecosystem fragmentation (Layman *et al.* 2007b).

We applied stable isotope analysis of ^{15}N and ^{13}C in context of river restoration to quantitatively characterize patterns in trophic structure. We sampled different components of food webs on paired restored and degraded sections of rivers in 20 different catchments throughout Europe. Two types of restoration projects were investigated; comprehensive large projects where a large restoration effort was put in place and smaller projects relying on mainly single restoration measures. The restored sections were compared to degraded "control sites" that are located upstream of the restored sections. The sampling included elements of the resource base (particulate organic matter, most abundant aquatic and riparian plant material, periphyton), macroinvertebrates comprising at least the dominant taxa within different functional feeding groups as well as predatory riparian and non-riparian arthropods. In the study presented here, we focused on macroinvertebrate communities as commonly applied indicators of ecosystem health. We used two of the metrics introduces by Layman *et al.* (2007a): Nitrogen-range ($\Delta^{15}N$) and carbon-range ($\Delta^{13}C$) of the dominant feeding types of macroinvertebrate communities to quantify changes in trophic structure between restored and degraded sections.

We assumed that the complexity of a food web increases with restoration as a consequence of habitat diversity. We further assumed that large restoration projects have a stronger effect on food web composition compared to small restoration projects, as they increase habitat diversity more strongly. These assumptions are based on the consideration that habitat diversity corresponds to the availability of autochthonous food sources and to more diverse assimilated resources, e.g. based on more intense interconnections of water and land. For instance, restoration might result in shallower profiles and thus enhance the availability of allochthonous material.

Specifically, we tested the following hypotheses

- Trophic length of the macroinvertebrate community increases with habitat complexity and hence the degree of restoration (reflected by $\Delta^{15}N$ of macroinvertebrate feedings types).
- The diversity of basal resources increases with the degree of restoration, making a greater range of carbon sources available to macroinvertebrates (reflected by $\Delta^{13}C$ of macroinvertebrate feedings types).

10.2 *Material and methods*

Study design

In ten regions across Europe we sampled four river sections: one river section of a large restoration project (R1), one section of a small restoration project (R2) and non-restored, degraded sections directly upstream of the restored sections (D1, D2, see Chapter 1.2 for more detailed information on the general study design). Representative samples of the food web components were collected to identify effects of restoration on patterns in trophic structure. Samples contained elements of the resource base (particulate organic matter, aquatic and riparian plant material, periphyton), the most abundant macroinvertebrates representing different functional feeding groups as well as riparian and non-riparian arthropods. We tested for large-scale, general patterns influencing ^{15}N-enrichment and used isotope-biplots to visually describe differences between restored and degraded sections. Here, we initially focused on macroinvertebrate communities and calculated nitrogen- and carbon range ($\Delta^{15}N$ and $\Delta^{13}C$). First, all community members with their corresponding $\delta^{13}C$- and $\delta^{15}N$-values were considered. Second, we classified macroinvertebrates into functional feeding groups and used average values of this *a priori* grouping to calculate $\Delta^{15}N$ and $\Delta^{13}C$.

Study sections and sampling methods

The study sections and reaches as well as sampling methods and laboratory analysis for the stable isotopes are described in Annex B and Chapter 2.9.

Samples from Switzerland and the Netherlands were affected by sampling and conservation errors and delivered no reliable results in respect to our analysis approach presented here. They were omitted from further analysis.

Data analysis

To visually test for large-scale impacts (latitude, altitude, geology and land use intensity) influencing carbon and nitrogen enrichment, we plotted $\delta^{13}C$ and $\delta^{15}N$ of all components (food sources, macroinvertebrates as well as riparian and non-riparian arthropods) against latitude.

To analyse isotopic composition of macroinvertebrates, the data was plotted in carbon-nitrogen-biplot-space. To test for restoration effects on macroinvertebrate communities, we used two community-wide metrics introduced by Layman *et al.* (2007a): Nitrogen-range ($\Delta^{15}N$) was calculated with maximum (δN) – minimum (δN) and carbon-range ($\Delta^{13}C$) with maximum (δC) – minimum (δC). First, these two metrics were calculated considering all community members with their corresponding $\delta^{13}C$- and $\delta^{15}N$-values (subsequently referred to as absolute values). Second, we classified the macroinvertebrates into five feeding groups (predators, shredders, grazers, collector-filterers, collector-gatherers) based on Schmidt-Kloiber & Hering (2012) and used the corresponding average values of the feeding groups to calculate $\Delta^{15}N$ and $\Delta^{13}C$ again (subsequently referred to as mean values).

Metrics were compared between restored and degraded sections (R vs. D) as well as between large and small restoration projects compared to the corresponding degraded sections (R1 vs. D1 and R2 vs. D2). We then calculated the difference between the metric values of each restored (R) and corresponding degraded (D) section; these differences were then compared between the large (R1/D1) and small (R2/D2) restoration projects. Differences were tested with the Wilcoxon matched pairs test.

For the analysis we used the following software: We visualized large-scale patterns using OriginPro 9.0. For the visualization in isotope-biplots as well as the calculation of community-wide metrics we used the package Stable Isotope Analysis in R (SIAR: Parnell *et al.* 2008, 2010) in R (R Development Core Team, 2007). The further statistical analyses for pairwise comparison were run using Statistica 8 software from StatSoft.

10.3 Results

General patterns

Enrichment of ^{15}N within the given dataset showed clear differences between countries (Figure 10-1), with higher $\delta^{15}N$-enriched samples in mid-latitudes (Germany, Netherlands, Czech Republic, Sweden, Denmark and Poland). Samples taken in alpine regions (Austria and Switzerland) as well as those in high latitudes (Finland) were less enriched in $\delta^{15}N$. There was also a corresponding difference in $\delta^{13}C$ enrichment, though less obvious.

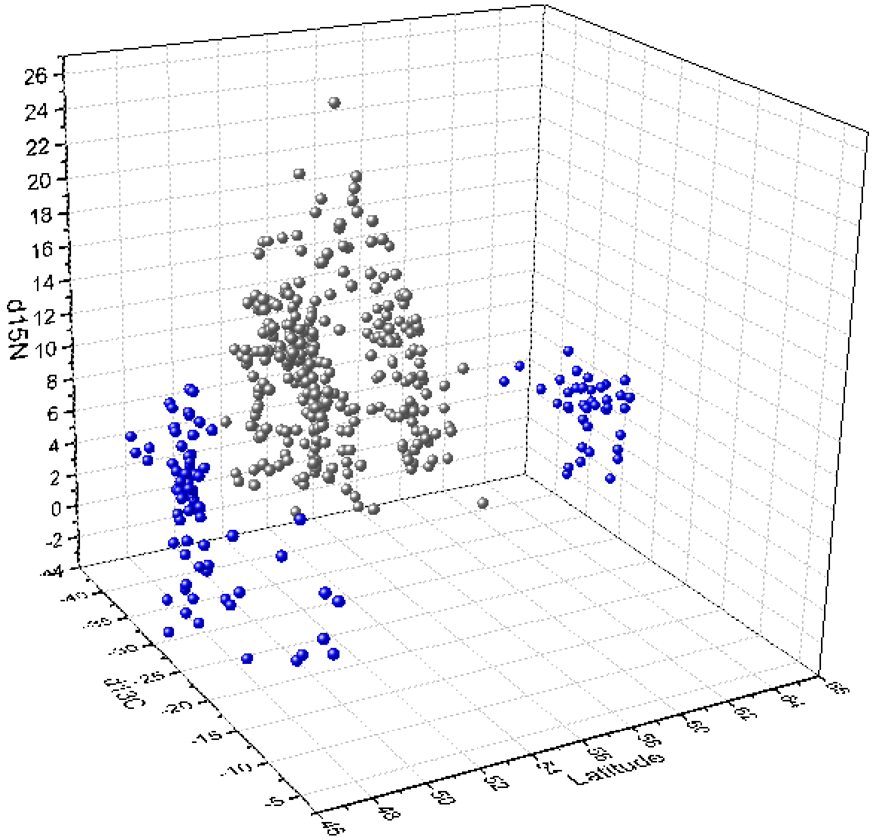

Figure 10-1: $\delta^{13}C$ and $\delta^{15}N$ of dataset plotted against latitude (grey: areas with high $\delta^{15}N$-enriched samples; blue: areas with less $\delta^{15}N$-enriched samples).

Trophic structure

As an example for isotopic composition of food web components in restored and degraded sections, the results for the river Drau in Austria (R1 and D1) are shown in Figure 10-2. $\Delta^{15}N$ and $\Delta^{13}C$ of the respective macroinvertebrate communities were calculated for both sections: $\Delta^{15}N_{restored}$ was higher (4.56‰) than $\Delta^{15}N_{degraded}$ (3.23‰) suggesting that the trophic length of the macroinvertebrate community was higher in the restored compared to the degraded section. Furthermore $\Delta^{13}C_{restored}$ was higher (3.93‰) than $\Delta^{13}C_{degraded}$ (1.67‰) suggesting that macroinvertebrates in the restored section were using a wider spectrum of basal sources.

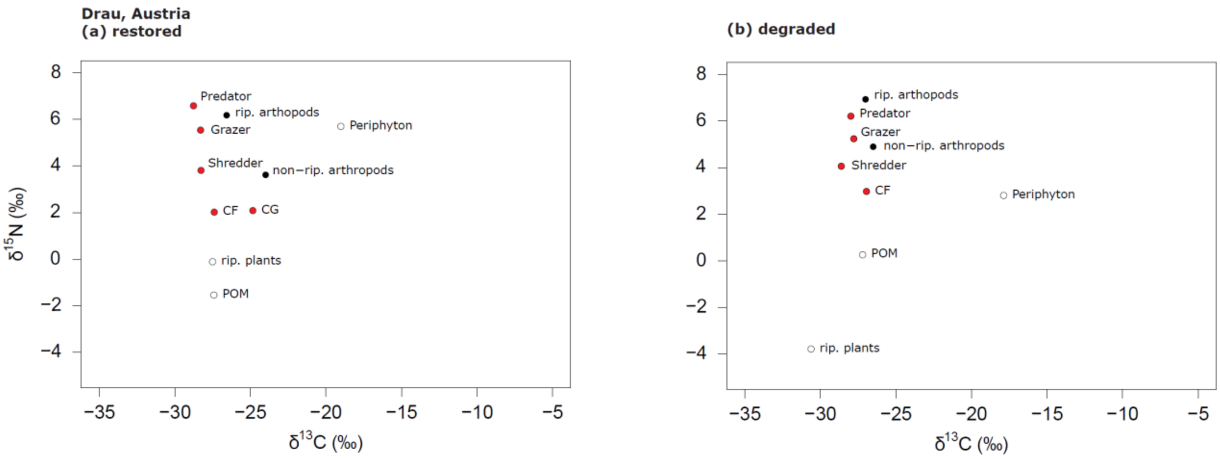

Figure 10-2: $\delta^{13}C$ and $\delta^{15}N$ of producers (white symbols; mean), riparian and non-riparian arthropods (black symbols; mean) and macroinvertebrates (red symbols; mean) for a) restored and b) degraded sections of river Drau (Austria). Macroinvertebrates were classified to feeding-types: predator, shredder, grazer, collector-filterers and collector-gatherers. Collector-gatherer were not present at degraded site. All values shown are means of several samples.

Effects of river restoration on isotopic composition of macroinvertebrates

The pairwise comparison of macroinvertebrate communities between restored (R) and degraded (D) sections showed minor differences for both absolute and mean values of $\Delta^{15}N$ and $\Delta^{13}C$ (Figure 10-3a,b, Figure 10-4a,b). Differences between restored and degraded sections were not significant (Wilcoxon Matched pairs test).

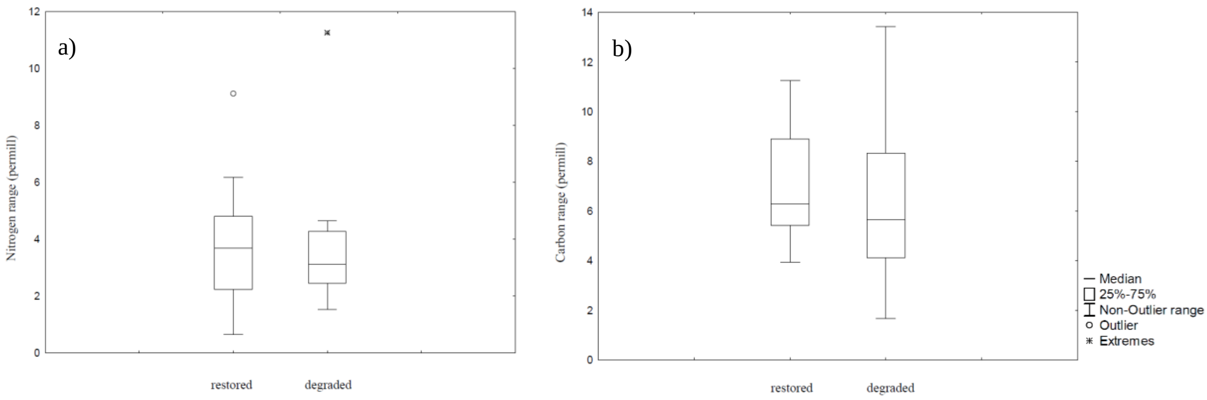

Figure 10-3: Comparison of a) $\Delta^{15}N$ and b) $\Delta^{13}C$ of macroinvertebrate communities for restored and degraded sections (n = 16) based on absolute values (i.e. all community members with their corresponding $\delta^{13}C$- and $\delta^{15}N$-values are considered; not grouped into feeding types).

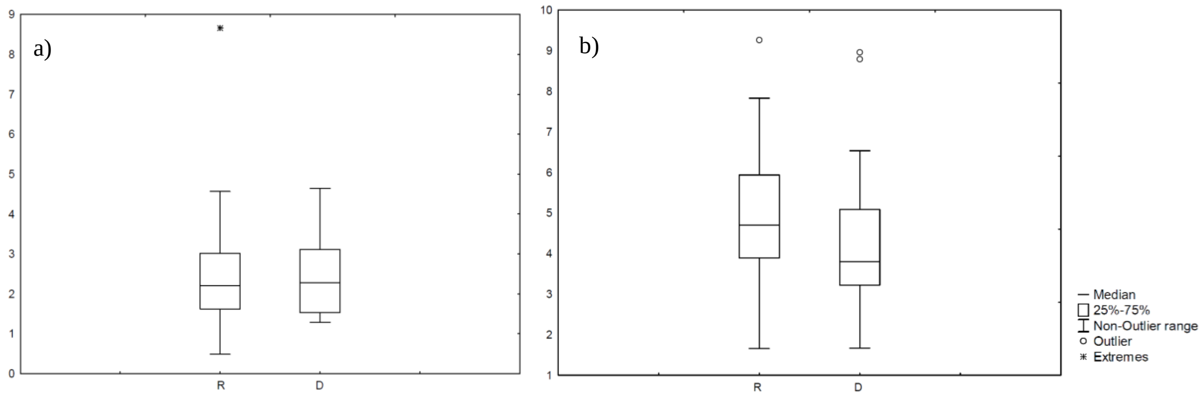

Figure 10-4: Comparison of a) $\Delta^{15}N$ and b) $\Delta^{13}C$ of macroinvertebrate communities for restored and degraded sections (n = 16) based on mean values (i.e. macroinvertebrates were grouped into five feeding types and the corresponding mean values of the feeding types were used).

Effects of large and small restoration projects on isotopic composition of macroinvertebrates

The pairwise comparison between the four groups of sections (large restoration projects: R1; corresponding degraded sections: D1; small restoration projects: R2; corresponding degraded sections: D2) showed minor differences for $\Delta^{15}N$ (Figure 10-5a). In contrast, $\Delta^{13}C$ differed significantly between R1 and D1 (Wilcoxon Matched pairs test, $p < 0.05$) with larger $\Delta^{13}C$ for R1 when considering the absolute values of macroinvertebrates (Figure 10-5b). When considering mean values of macroinvertebrates grouped into feeding types and comparing $\Delta^{13}C$ of the feeding types, there was no significant differences in $\Delta^{13}C$ (Figure 10-6b).

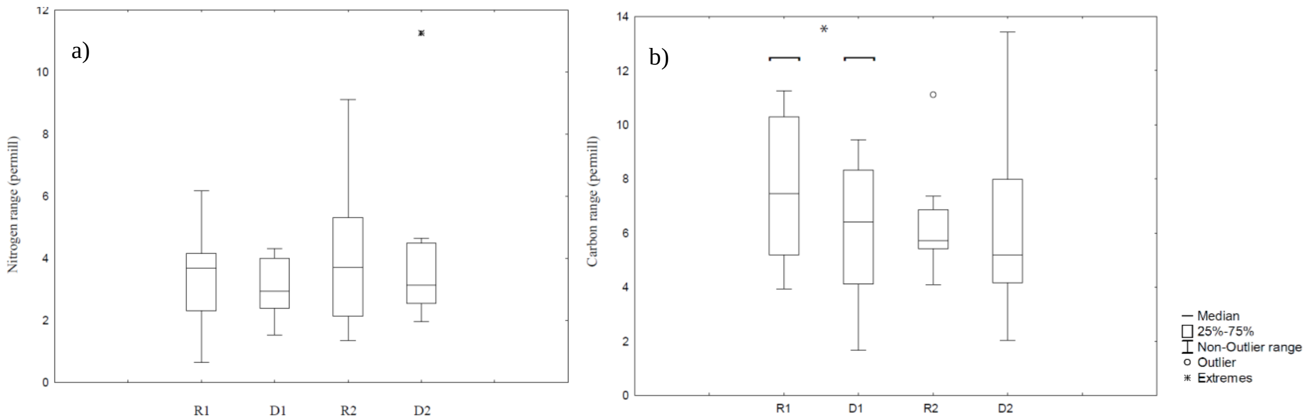

Figure 10-5: a) $\Delta^{15}N$ and b) $\Delta^{13}C$ of macroinvertebrate communities for sample sets of R1, D1, R2 and D2 (n=8). Pairwise comparison of R1/D1 and R2/D2 using Wilcoxon Matched pairs test (* p<0.05). The analysis is based on absolute values (i.e. all community members with their corresponding $\delta^{13}C$- and $\delta^{15}N$-values are considered; not grouped into feeding types).

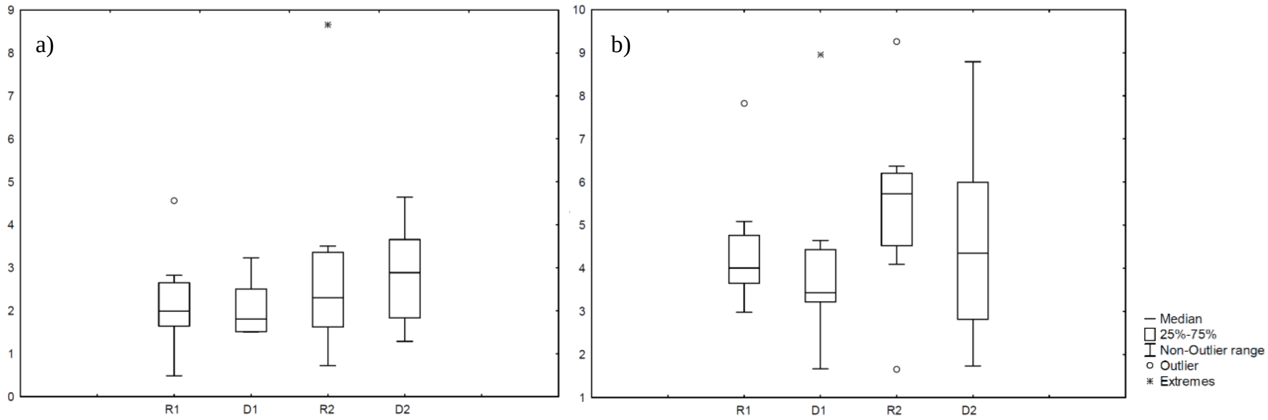

Figure 10-6: a) $\Delta^{15}N$ and b) $\Delta^{13}C$ of macroinvertebrate communities for sample sets of R1, D1, R2 and D2 (n=8). The analysis is based on mean values (i.e. macroinvertebrates were grouped into five feeding types and the corresponding mean values of the feeding types were used).

As expected, the pairwise calculated differences between the metric values of R1 minus D1 and R2 minus D2 showed similar patterns (Figure 10-7, Figure 10-8). Here, values above zero indicated enhanced $\Delta^{15}N$ respectively $\Delta^{13}C$ in trophic structure. Most obvious was the larger $\Delta^{13}C$ in R1 (Figure 10-7b). However, differences between R1 and R2 were not significant (Wilcoxon Matched pairs test, $p = 0.89$ for $\Delta^{15}N$, $p = 0.12$ for $\Delta^{13}C$).

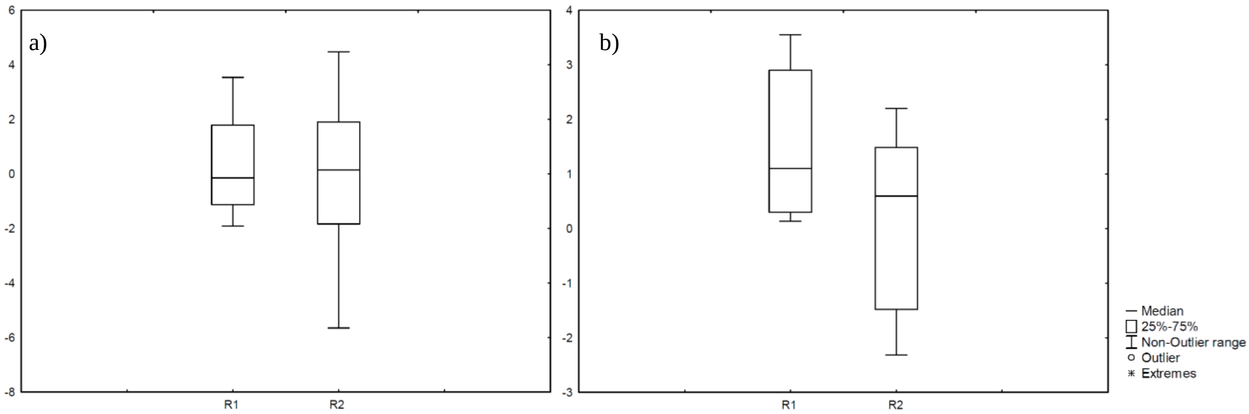

Figure 10-7: Comparison of differences between the long (R1/D1) and short (R2/D2) restored sections for a) $\Delta^{15}N$ and b) $\Delta^{13}C$ of macroinvertebrate communities; difference was pairwise calculated between restored and corresponding degraded sections (R1-D1 and R2-D2) based on absolute values (i.e. all community members with their corresponding $\delta^{13}C$- and $\delta^{15}N$-values are considered; not grouped into feeding types).

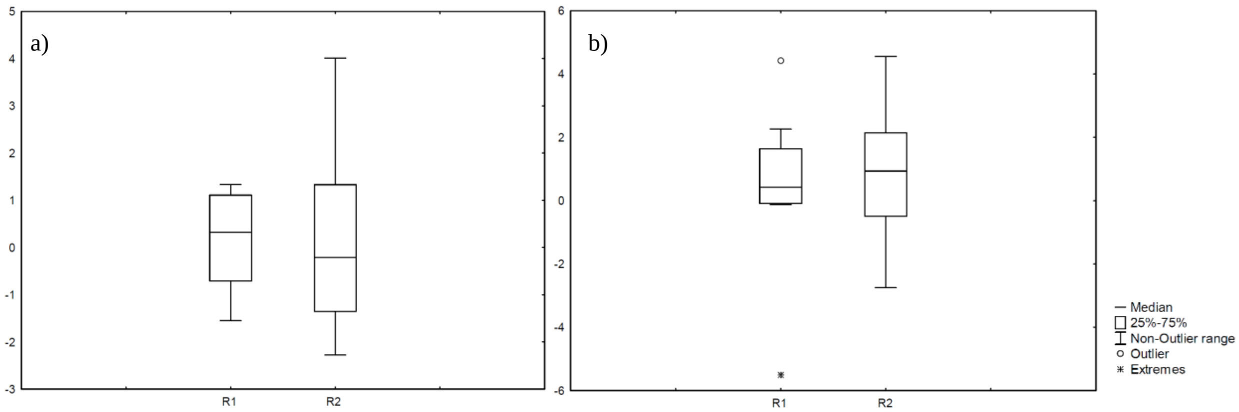

Figure 10-8: Comparison of differences between the large (R1/D1) and small (R2/D2) restoration projects for a) Δ^{15}N and b) Δ^{13}C of macroinvertebrate communities; difference was pairwise calculated between restored and corresponding degraded sections (R1-D1 and R2-D2) based on mean values (i.e. macroinvertebrates were grouped into five feeding types and the corresponding mean values of the feeding types were used).

10.4 Discussion

The dataset was subject to large-scale patterns on a European level. Samples taken in mid-latitudes (Germany, the Netherlands, Sweden, Denmark, Poland and Czech Republic) were more enriched in ^{15}N than samples from alpine regions (Austria and Switzerland) and from high latitudes (Finland). Different land-use intensity (e.g. fertilizer application) might have been a reason for the higher ^{15}N enrichment, as reflected by the high values in the Netherlands and Germany. In this study, we did not analyse the large-scale patterns in detail as this will be done in upcoming studies. The results, however, underlined the necessity to limit comparisons to sites within a region, as large-scale differences possibly masked the effects of restoration.

The river Drau (Figure 10-2) showed higher Δ^{15}N and Δ^{13}C for the macroinvertebrate communities at the restored section compared to the degraded section. This supported our hypotheses that trophic length (indicated by Δ^{15}N) as well as diversity of assimilated food sources (indicated by Δ^{13}C) increase with restoration. However, an increased trophic length (Δ^{15}N) only appeared in single cases. The overall difference between restored and degraded sections, however, was not significant neither using absolute nor mean values of macroinvertebrate communities. Our first hypothesis was therefore rejected. In both restored and degraded sections, the Δ^{15}N of the macroinvertebrate communities was almost within the limits of a single trophic level.

When using absolute values (i.e. considering all community members and not grouping them into feeding types), Δ^{13}C was significantly larger in large restoration projects (R1) compared to the corresponding degraded sections (D1), suggesting that macroinvertebrates were feeding from more diverse sources. The comparison of Δ^{13}C between R2 and D2 showed almost no difference. This implies that diversity at the resource base was positively related to restoration extent, thus confirming our second hypothesis.

The results of our analysis were partly determined by the type of data used: Significant differences in $\Delta^{13}C$ between R1 and D1 were only obtained with absolute values. Mean values of the feeding types possibly reduced the corresponding nitrogen and carbon ranges since outliers were less influential, while absolute values (i.e. original values, not grouped into feeding types) consider outliers more strongly, resulting in larger ranges. For instance, we sampled only few large predators. Averaging the corresponding isotope concentrations with values from a larger number of small and medium sized predators results in relatively low values. In fact, the outliers might reflect a higher diversity of the resource base, as stated in our second hypothesis. Consequently, outliers might be a result of restoration as the corresponding macroinvertebrates assimilated sources that were only present at the restored sections.

We limited sampling to the dominant taxa of each component to gain a representative overview of food web structure and aquatic-terrestrial interaction and to test for differences between restored and degraded sections. For a detailed construction of the food webs using mixing models (Brauns et al. 2011) a more extensive sampling, considering all potential food sources and consumers would be necessary (Brauns *et al.* 2011). But even a comprehensive sampling could be subject to sampling errors, as in mid-sized rivers external factors such as drift are important and need to be considered. For the construction of detailed food webs a habitat-specific approach should be used.

10.5 Outlook

In the next set of analysis we will focus on large-scale patterns within our dataset. For example we will test for correlation between land use (especially usage of fertilizers) and ^{15}N enrichment in the food webs. Also effects of different geological and/or soil characteristics on ^{13}C will be tested.

Furthermore, we will test which environmental parameters impact the trophic structure of macroinvertebrate communities, e.g. river characteristics like altitude, discharge or substrate and restoration characteristics like restoration size, time since restoration and type of restoration. In addition, we will also take the other sampled components of the food web into account (food sources, riparian arthropods and non-riparian arthropods). When considering riparian arthropods, we will also test for effects of restoration on trophic interaction between the river and its riparian zone as well as general patterns in feeding preference of riparian arthropods. Paetzold *et al.* (2005) already described that beetles in riparian zones feeding on aquatic prey and Collier *et al*. (2002) described similar effects for spider predation. So we would like to investigate if restoration increases the content of aquatic prey in the diet of these riparian arthropods due to a stronger connection of the river and its riparian zone (e.g. caused by a more shallow profile that will increase inundation frequency and thus the transport of organic matter from the river to its floodplain. A shallow profile will also make aquatic prey more easily accessible to riparian predators).

The Bayesian approach as described by Layman *et al.* (2007a) will be applied on our data as well. Furthermore, we will test other community-wide metrics introduced by Layman *et al.* (2007a) to describe trophic structure: Total area (TA), mean distance to centroid (CD), mean nearest neighbour distance (MNND) and standard deviation of nearest neighbour distance (SDNND).

10.6 References

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11. Summary and conclusions

11.1 Study objectives

The main objectives of the study were:

(i) Quantify and compare the effect of restoration on different response variables (habitat composition in the river and its floodplain, three aquatic organism groups, two floodplain-inhabiting organism groups, food web composition and aquatic land interactions as reflected by stable isotopes). Comparing different response variables allows to draw conclusions on the general effect of restoration on habitats and biota. We hypothesized that floodplain-related variables (e.g. floodplain vegetation, ground beetles, floodplain and riparian habitats) respond more strongly to restoration, and variables related to the river itself (e.g. fish, benthic invertebrates, substrate diversity) respond weakly, as they are more strongly influenced by catchment-scale stressors, e.g. through water quality.

(ii) Identify variables which either constrain or enhance the effect of restoration, i.e. identify conditions which favour restoration success. We especially focused on the potential positive mitigating effect of restoration extent since longer sections or more intense restoration might buffer negative impacts from upstream large-scale stressors (e.g. fine sediment input), provide the necessary minimum area for viable populations, and allow habitats to be created by natural channel dynamics, i.e. sustainable habitat creation. Therefore, we assumed strong effects of restoration extent for organism groups which depend on large-scale stressors (e.g. benthic invertebrates), depend on hydromorphological processes requiring a certain section length (several instream habitats), and have a larger home range (fish). In contrast, weakest effect of restoration extent was expected for riparian and floodplain biota and habitats, as strong effects of restoration have been documented already in small restoration projects.

11.2 *Summary of results*

Comparing dissimilarity of restored and degraded sections for different response variables

- Quantify restoration effect:
    - Ground beetles were most strongly responding to restoration, followed by fish, floodplain vegetation, benthic invertebrates and aquatic macrophytes. Floodplain and aquatic habitats as well as stable isotope signatures differed less strongly between the restored and corresponding unrestored degraded sections.
- Conditions favouring restoration success:
    - Restoration extent: There was no significant difference in the response to restoration between the large and small restoration projects for none of the response variables, except for the food web interactions.
    - Substrate diversity: The responses of benthic invertebrates, aquatic macrophytes and all recorded morphological response variables (flow diversity, floodplain habitats) were greater in restored sections with larger changes in substrate composition as compared to those with smaller changes, and differences were nearly significant for fish.
- Conclusions:
    - We conclude that restoration extent was still too small (e.g. restored reach length was usually shorter than 2 km) for effects of restoration extent on biota. Changes in substrate composition, however, significantly affected aquatic organism groups, while small changes in substrate composition were already sufficient for ground beetles.

Hydromorphology

- Quantify restoration effect:
    - There was no overall significant difference between different spatial scales but results indicated that the effect of restoration on hydromorphology was largest at the section and reach scale, i.e. on macro- and mesohabitat diversity Effects on aquatic microhabitats were less pronounced and especially small on substrate diversity, which is of special importance for macroinvertebrates. Only in few restoration projects substrate diversity was significantly increased due to restoration.

- Conditions favouring restoration success:
  - Restoration extent: The effect of restoration on hydromorphology was not significantly higher in large compared to small restoration projects. However, there was a tendency for a higher effect of restoration on macro- and mesohabitats in large restoration projects. Especially the effect on mesohabitat diversity was higher in larger restoration projects: channel features like islands, banks and bars were more frequent and increased heterogeneity along the cross section, while the river bed itself remained the dominant channel feature in small restoration projects. In contrast, large and small restoration projects did not differ in respect to microhabitat diversity.
  - Restoration measures: The effect of restoration did not significantly differ between the restoration measures. However in general, widening had a higher effect on macro- and mesohabitats compared to remeandering and instream measures. This was possibly due to the natural channel dynamics restored by the removal of bank fixation and creation of secondary channels. In contrast, there was no such tendency for different effects of restoration measures on microhabitat diversity.
  - River type: There was a general trend for a higher effect of restoration on macro- and mesohabitats in gravel-bed compared to sand-bed rivers, but differences were not statistically significant. Moreover, restoration even had a significant negative effect in some restored sections if restoration projects were analysed separately. This could be partly due to the focus of the survey method on channel features and substrate conditions typical for gravel-bed rivers. In contrast, the habitat diversity in sand-bed rivers, which is far more dependent upon riparian vegetation and large wood, is possibly not adequately considered by the survey method.
  - Project age (time between implementation of the measures and monitoring): The effect of restoration on aquatic habitat conditions increased with project age. This might be due to restored channel dynamics increasing habitat quality over time. However, higher aquatic habitat quality might also simply be due to the fact that old projects were mainly located in gravel-bed rivers and applied widening as the main restoration measure, which generally had a higher effect.
- Conclusions:
  - Overall, we found only few general significant effects and differences between the large and small scale measures. However, results indicated that restoration increased macro- and mesohabitat diversity but had a limited effect on microhabitat conditions, especially on substrate diversity. There was a tendency for widening measures, projects in gravel-bed rivers, and older projects having a higher effect on macro- and mesohabitats but differences were not significant. Furthermore, the results revealed the need to consider adequate hydromorphological parameters for monitoring sand-bed rivers in the future.

Macroinvertebrates

- Quantify restoration effect:
    - Restoration had no overall positive effect on macroinvertebrate richness and diversity. Variability of restoration effect was especially high for macroinvertebrate richness, demonstrating that some projects indeed increased the number of taxa but other lead to a substantial decrease in species richness.
    - The effect on macroinvertebrate richness and diversity was significantly higher in projects increasing microhabitat diversity, indicating that the overall low effect of restoration on macroinvertebrates was mainly due to the low effect on microhabitat diversity. However, a small increase in microhabitat diversity already had a relatively high effect on richness and further increasing it did not further increase richness, which indicated that other factors might have constrained the effect of restoration.
- Conditions favouring restoration success:
    - Restoration extent: The effect of restoration on macroinvertebrates was not significantly higher in large compared to small restoration projects. Moreover, restoration had no significant positive effect on richness and diversity, neither for the large nor for the small restoration projects.
    - Restoration measures: There was a tendency for a higher effect of restoration on macroinvertebrate richness in widening projects compared to projects which applied less intensive measures but differences were not statistically significant.
    - River type: There was a tendency for a higher effect of restoration on macroinvertebrate richness in gravel-bed compared to sand-bed rivers but differences were not statistically significant.
- Conclusions:
    - Apparently, the extent and measures applied in the restoration projects investigated was not sufficient to enhance macroinvertebrate communities. In the restored sections investigated, the most probable reason was the low effect of restoration on the microhabitat. Therefore, future projects should aim at increasing and monitoring habitat diversity at the microhabitat scale, which is most relevant for macroinvertebrates.
    - Even if microhabitat diversity is improved, other factors seem to constrain the effect of restoration (similar to Liebig’s law of the minimum). Potential constraining factors are the impact of large-scale stressors and a depleted regional species pool. It is essential to identify the main reasons for the low effect of restoration on macroinvertebrates in future studies since the different reasons involve completely different restoration strategies.

Fish

- Quantify restoration effect:
    - Restoration had a significant but weak positive effect on fish species richness, with a mean increase of about 1 species mainly attributed to an increase of rheophilic species. Moreover, there was a tendency for a higher total density and diversity but restoration effect on these two metrics was not significant.
    - Restoration had the largest effect on small rheophilic fish compared to all rheophilic fish or other guilds, especially on the proportion density of small rheophilic fish.
- Conditions favouring restoration success:
    - Restoration extent: There was no overall effect of restoration extent on all fish species but restoration effect on the proportion of small rheophilic fish was higher in restored sections with a length of about 2 km compared to shorter sections.
    - Project age: Similarly, there was no overall effect of project age on all fish species but restoration effect on the proportion of small rheophilic fish was higher for rather young and old projects (< 3 and >12.5 years) and lowest in projects of an intermediate age.
- Conclusions:
    - Species richness, species diversity and fish density showed only weak or no response to restoration, while habitat traits reacted in a consistent way across the restoration projects by an increase of rheophilic and a decrease of eurytopic fish. This is consistent with the results of other studies showing that restoration – as practised in the past – does not change species richness and diversity but rather community structure.
    - Restoration effects were more pronounced within the first years after restoration than later. The restoration effect increased with habitat quality and length of restored river sections. However, current restoration practice does not allow comprehensive recovery of lost species and population densities. The reasons for that are manifold. The length of current restoration measures is short (mostly < 1km) limiting the amount and diversity of provided habitat and re-colonization is hampered by limited species pools and migration barriers. Future restoration should focus on more dynamic, self-sustaining habitat improvements extending over several kilometres and should be coupled with other measures such as restoring river continuity and species reintroductions.

Macrophytes

- Quantify restoration effect:
    - Restoration had an overall significant positive effect on richness and diversity of helophytes (emergent plants rooting under water or in wetted soils) but not on hydrophytes (emergent and submerged aquatic plants).
- Conditions favouring restoration success:
    - Restoration extent: The effect of restoration on macrophytes was not significantly higher in large compared to small restoration projects. Moreover, restoration had no significant positive effect on richness and diversity, neither for the large nor for the small restoration projects.
    - Restoration measures: The effect of restoration on helophyte richness was significantly higher in widening projects compared to projects which applied less intensive measures.
    - River type: For helophytes, the effect of restoration was higher in mountain compared to lowland streams.
    - Restoration effect did not depend on any other catchment, river or project characteristic investigated (e.g. restored reach length, project age).
- Conclusions:
    - Restoration had a significant effect on helophytes, especially in widening projects and mountain rivers. Since most widening projects were located in mountain rivers, it was difficult to deduce causal relationships. However, it is reasonable that widening projects, which usually create shallow low-velocity habitats at the river banks, had a positive effect on helophytes which are adapted to these semi-aquatic habitats.

Ground beetles

- Quantify restoration effect:
    - Overall (pooling large and small restoration projects), restoration had a significant positive effect on ground beetle richness but not on diversity for both effect sizes used (difference between and ratio of restored vs. degraded sections).
- Conditions favouring restoration success:
    - Restoration extent: Restoration had a significant positive effect on ground beetle richness in the large but not in the small restoration projects. However, this was only true if the difference between the restored and degraded sections was used to quantify restoration success and not for the ratio of restored and degraded sections. Since results did depend on the choice of the effect size, these results only partly confirmed that restoration extent favoured restoration success.

- Restoration measures: In contrast to restoration extent, differences between restored sections were much more pronounced if projects were grouped according to the main measure applied (widening vs. instream, flow restoration, remeandering). Widening had a significant positive effect on ground beetle richness and a smaller but still significant effect on diversity, whereas other measures even tended to have a negative effect.
- River type: There was a general trend for a higher effect of restoration on ground beetle richness in gravel- compared to sand-bed rivers. For riparian specialists, strongest effects were obtained in high-gradient rivers, where widening as a restoration measure was applied.
- Habitat types: The positive effect of widening was mainly due to the strong relationship between ground beetle richness and a specific habitat type: the open pioneer stage covered by sparse woody vegetation, but not to the mere number of habitat types.

- Conclusions:
  - For ground beetles' species richness, diversity, and richness of riparian specialists, measures creating pioneer patches are crucial, for example by river widening, which result in more open banks. For instream fauna like fish and invertebrates, riparian trees are regarded an important factor increasing ecological quality of the river, because it dampens temperature fluctuations by shading, provides food and offers habitat structure. As a result, the development of woody riparian vegetation is a commonly applied restoration measure. Our study shows that this measure could be counterproductive for the specialist riparian carabid fauna, which requires open habitat. If the purpose of restoration includes enhancing the conditions for floodplain biota, some open areas should remain present – in this case the restoration goals addressing aquatic biota and floodplain biota are not contradictory. This could be well combined with providing enough shade for the instream fauna because our results also indicate that the mere presence of open habitats is more important than its area.
  - Suitable restoration measures should aim on a strong lateral connection between the river and its floodplain to guarantee frequent inundation which creates and maintains pioneer patches over time. This is particularly important for lowland rivers as habitat turnover and the development of habitat diversity takes longer timespans due to less power of the river. A strong lateral connection will also help to promote the colonization of typical ground beetles for wetlands and floodplain forests.

Floodplain vegetation

- Quantify restoration effect:
    - Overall (pooling large and small restoration projects), restoration had a significant effect on floodplain vegetation as indicated by the mean response ratios of several metrics/traits which were significantly different from zero. Restoration had a small negative effect on leaf dry matter content. Moreover, the share of short lived species was higher (significantly larger share of annual species and therophytes) in the restored reaches. These species benefit from disturbances, indicating that restoration increased the frequency of disturbances like flooding. However, there was no overall positive effect of restoration on species diversity.
- Conditions favouring restoration success:
    - Restoration extent: There were no significant differences between the large and small restoration projects for none of the diversity and trait metrics investigated (e.g. richness, plant growth form). However, the following metrics were significantly different from zero for only either large or small restoration projects, indicating that restoration extent had an influence on the response of floodplain vegetation:
        - For the small restoration projects, we observed a decrease in the abundance of perennials and an increase in the abundance of annuals, which is often associated with highly disturbed environments. This was unexpected since small restoration actions were thought to disturb riparian and floodplain areas to a lesser extent compared to large restoration actions. However, the observed response might reflect that reconstruction of instream habitat structures have disturbed also the river banks and created open space for the establishment of annual plants.
        - For the large restoration projects we observed a decrease of the relative abundances of chamaephytes and an increase of therophytes compared to the corresponding degraded sections. These results indicated that large restoration increased flooding disturbance of riverbanks.
    - Altitude: As altitude increases, restoration is more likely to promote the coexistence of plants with different moisture preferences and life strategies. While it is difficult to assess the exact cause, possible reasons are (i) that restoration in high altitude sections mainly aimed at widening the stream channels which may have resulted in greater habitat diversity compared to instream and remeandering measures applied in low altitude sites, in addition to flashier hydrological regimes and greater stream flow in mountain areas leading to a faster habitat turnover, and (ii) a generally less intense anthropogenic disturbance in high altitude regions resulting in greater regional diversity and colonization of restored section by species with a wider range in moisture preferences and life strategies.

- Discharge: There is a higher probability of a positive effect of restoration on taxon diversity with an increasing discharge (stream size) independent of the restoration extent, most probably because dispersal distances of vegetation propagules generally increase with an increasing discharge.
- Agricultural land use: An increased agricultural intensity in the catchment affected the response of trait composition of the floodplain community (i.e. growth form, dispersal strategies and light preferences) suggesting that any changes in these traits due to restoration are at risk to be masked in catchments with high agricultural intensity. Of particular interest is the observation that the presence and abundance of geophytes (perennial plant surviving part of their life cycle as a dormant underground structure) decreased with increasing agriculture, which may reflect that geophytes respond negatively to grazing and in particular phosphorous availability - factors which are both highly associated with agricultural intensity. However, we found no effect of agricultural land use on the effect of restoration on diversity metrics. We speculate that restoration effect on diversity is more strongly affected by larger spatial scales (e.g. bioregion) or historical disturbances, i.e. masked by factors operating at much larger spatial scales and/or over longer time periods.
- Project age (time between implementation of the measures and monitoring): While positive relationships between riparian plant richness and time after restoration have been observed in some restored sections, the time scale investigated (1-20 years) may be far below what is needed for full community recovery. Moreover, effects on diversity may be delayed by time-lags in the recovery of environmental conditions that restoration was unable to target.

- Conclusions:
  - Restoration had an effect on trait composition, while general effects on diversity were limited to the small restoration projects. The high variability in restoration effects could be attributed to factors related to river typology (discharge and altitude), catchment land use and project age. These strong relationships may partly explain why no general effects of restoration on diversity indices were detected. However, communities may also need considerable more time to establish, and an increase in diversity may not be seen within the time frame investigated here.

Stable isotopes

- Quantify restoration effect:
    - Restoration increased the complexity of the macroinvertebrate food web, and hence trophic length (indicated by the range of $\delta^{15}N$, labelled as $\Delta^{15}N$) in some of the restored sections but differences between restored and degraded sections were not significant. In both restored and degraded sections, the $\Delta^{15}N$ of the macroinvertebrate communities was almost within the limits of a single trophic level.
    - Restoration increased the diversity of food sources for macroinvertebrates as indicated by higher $\Delta^{13}C$ in large restoration projects.
- Conditions favouring restoration success:
    - Restoration extent: The diversity of food sources for macroinvertebrates was higher in large restoration projects compared to the corresponding degraded sections but no such differences were found for small restoration projects.
- Conclusions:
    - Results indicated that the effect of restoration on food sources and food webs depends on restoration extent which is possible due to an increase in habitat diversity and in turn food sources as well as increased river-floodplain interactions. Further analysis will a.o. focus on the effect of different restoration measures. In light of the results on the organism groups, it might be expected that especially widening projects will increase species richness and river-floodplain interactions, and hence influence food webs.

11.3 General discussion and conclusions

Transferability of results

The 20 restoration projects investigated in this study were representing good-practice examples in Northern Eastern and Central Europe either targeting medium-sized lowland rivers or medium-sized mountain rivers (Figure 1-2), already covering many river types in Europe, and partly reflecting the relatively long tradition in river restoration in these regions, and hence, availability of good-practice examples. Without substantially increasing the number of restoration projects investigated, which was beyond the capability of a single study, including large rivers or projects from Western or Southern Europe would have resulted in regional differences too large to allow for grouping and a meaningful comparison for statistical analysis. Therefore, transferability of the results is mainly limited to mid-sized rivers in the regions investigated and regional differences have to be considered when applying the results in other river types or regions.

The general effect of restoration on biota - a comparison of different response variables

As hypothesized, the effect of restoration on richness and diversity differed between the response variables investigated. Almost all organism groups showed the expected higher effect on floodplain-related compared to aquatic variables. Restoration had no or only a small effect on species richness or diversity of macroinvertebrates and fish, while restoration had a clear positive effect on richness or diversity of organism groups inhabiting river banks or adjacent shallow shoreline habitats (ground beetles, macrophytes). This is consistent with the findings of several other studies which found that restoration has a high effect on ground beetles (Januschke et al. 2011) and macrophytes (Lorenz et al. 2012) and a smaller or missing effect on fish and macroinvertebrates (Schmutz et al. 2014, Jähnig et al. 2009, Januschke et al. 2014, REFORM deliverable D 4.2 Kail and Angelopoulos 2014). However, the most floodplain-related organism group (floodplain vegetation) showed no increase in richness or diversity, in contrast to other studies reporting a significant higher richness in restored compared to degraded sections (Jähnig et al. 2009, Januschke et al. 2011). These contrasting results were possibly due to the limiting effect of land use, which was much more intense in some of the catchments investigated in this study.

- In general, it can be expected that the effect of restoration on species number and diversity is high for ground beetles, macrophytes, and floodplain vegetation (given a low land use pressure), moderate for fish, and low for macroinvertebrates.
- The effect of river restoration projects should be assessed in a holistic way, including semi-terrestrial and terrestrial organism groups since terrestrial (floodplain) and aquatic ecosystems are closely linked and cannot be considered separately.

In general, the effect of restoration on community structure, traits, and functional indicators was more pronounced compared to its effect on the pure number of species or diversity mentioned above. First, for three organism groups, the significant effect of restoration on species richness or diversity was most pronounced for specific traits (ground beetle species inhabiting sparsely vegetated river, only helophytes but not hydrophytes, small rheophilic fish). Second, organism groups for which richness or diversity was not significantly increased showed effects on community structure (increase of therophytes and annual floodplain vegetation species, increase of food source diversity for invertebrates as indicated by the stable isotopes). Third, the restored and degraded sections were highly dissimilar, also in respect to organism groups for which species number and diversity did not or only slightly change (macroinvertebrates, fish, and floodplain vegetation besides macrophytes and ground beetles, Figure 3-1). These changes in community structure indicate specific functional changes caused by river restoration and can be used to increase our understanding how restoration measures affect aquatic ecosystems and investigate causal relationships.

- Future monitoring and studies should focus more on functional aspects (e.g. species traits, community structure) to investigate how river restoration affects river hydromorphology and biota, which would offer a great opportunity to make fundamental advances in restoration ecology and to identify (cost)-effective restoration measures.
- Restoration projects should also aim at restoring ecosystem functions and focus more on traits besides assessing restoration success based on the effect on species richness and diversity.

Conditions favouring restoration success

The factors potentially constraining or enhancing the effect of restoration were partly correlated, which made it difficult to infer causal relationships (e.g. most old projects were located in gravel-bed rivers where widening was the main restoration measure applied, and catchment land use was less intensive). Nevertheless, it was possible to identify conditions which most probably favour restoration success:

Catchment land use: It has been widely stated that large-scale pressures like water quality and fine sediment loads might constrain the effect of restoration (Palmer et al. 2010, Lorenz and Feld 2013; Sundermann et al. 2013), which in principle should affect aquatic organism groups like macroinvertebrates and fish more strongly compared to riparian and floodplain inhabiting biota or macrophytes, which even might benefit from slightly increased nutrient loads. However, in this study, catchment land use did only affect restoration effect on trait composition of floodplain vegetation. The missing effect of restoration on species richness and diversity of floodplain vegetation was not related to present agricultural land use as a large-scale pressure, and floodplain vegetation possibly has rather been affected by regional differences and historical disturbances (Harding et al. 1998), i.e. factors operating at much larger spatial and temporal scales. It has been widely stated that the effect of restoration on macroinvertebrates is constrained by large-scale pressures like a high land-use pressure in the upstream catchment (Kail and Hering 2009, Lorenz & Feld 2013, Marzin et al. 2013, Verdonschot et al. 2013). However, in this study, catchment land use had no effect and results rather indicated that the low effect on invertebrates was mainly due to a low effect of restoration on the microhabitat conditions relevant for macroinvertebrates (see discussion on habitat conditions below).

Similar non-significant linear relationships between catchment land use and richness as well as abundance were found in a recent meta-analysis of peer-reviewed literature on fish, macroinvertebrates, and macrophytes (REFORM deliverable D 4.2 by Kail and Angelopoulos, 2014). Only the effect of restoration on fish abundanc was negatively affected by agricultural land use. In contrast in the same meta-analysis, agricultural land use was identified as an important predictor for restoration success using other statistical methods which are more appropriate to detect non-linear relationships, upper limits, and threshold effects (Kail and Angelopoulos 2014) but require a much larger sample size compared to the 20 restoration projects investigated in this study.

Species pool and dispersal: The organism groups which did benefit most from restoration also have relatively high dispersal abilities (ground beetles, macrophytes), indicating that lower dispersal abilities (e.g. macroinvertebrates) or migration barriers (e.g. affecting upstream migration of fish) might have limited the effect of restoration on other organism groups. The ecological status of upstream reaches used as a proxy for macroinvertebrate source populations was not related to restoration success, indicating that - at least for macroinvertebrates in the projects investigated - the limited species pool available for re-colonization was not a main factor affecting restoration success. Moreover, the effect of restoration on floodplain vegetation increased with river size (discharge), most probably because dispersal distances of vegetation propagules generally increase with an increasing discharge. A detailed analysis of source populations and dispersal modelling was beyond the scope of this study but there is an increasing number of publications on this topic (Stoll et al. 2013, Tonkin et al. 2014, Radinger et al. 2014) and it clearly merits further investigation since a limited re-colonization potential would need a completely different restoration strategy compared to habitat improvements.

Project age (time between implementation of the measures and monitoring) only had a positive effect on the aquatic habitat conditions but not on any of the organism groups investigated, possibly due to the young age of most projects investigated. Most projects were just implemented 1 to 16 years prior to monitoring, which was probably less than what is needed for full community recovery. In contrast, project age was identified as one of the most important variables affecting restoration success in the REFORM deliverable D 4.2. (Kail and Angelopoulos 2014), which is surprising since the gradient in this dataset was even shorter (10-90th percentile range of 1 to 8 years). These contrasting results stress the need to further investigate the effect of restoration over time in future studies to better understand the trajectories of change induced by restoration measures, and to identify sustainable measures which enhance biota in the long-term.

- The effect of restoration depends on different factors including present large-scale pressures (e.g. water quality, fine sediment input), historical disturbances, a limited species pool and migration barriers hindering re-colonization of the restored section as well as project age. The knowledge on the effect of these factors on restoration success is still limited due to methodological problems or limited data availability and clearly merits further investigation.

Restoration extent: The effect of restoration did not differ significantly between the large and small restoration projects for none of the response variables. However, results indicated that there was a tendency for large restoration projects being more successful. For three organism groups, only large projects had a significant positive effect on some biological metrics which indicate a higher interaction between the river and its floodplain while small projects did not show such a significant effect (ground beetle richness, floodplain vegetation traits indicating higher disturbance by flooding, food source diversity for invertebrates as indicated by stable isotopes). Moreover, the share of small rheophilic fish was larger in longer restored sections and large projects tended to have a higher effect on mesohabitat diversity. Similarly, Kail and Angelopoulos (2014) did not find an effect of restored reach length in a meta-analysis on restoration success (see REFORM deliverable 4.2).

Based on these results, one should not conclude that it is sufficient to restore short river sections and implement small restoration projects. The majority of the large restoration projects were still too small to cause significant differences compared to the smaller projects. For example, restored section length was less than or equalling 2 km, except for two restoration projects (see Annex B). This is consistent with the results of Kail and Angelopoulos (2014) who also concluded that the missing effect of restored section length on restoration success was most probably due to the short length of most restored sections investigated (< 2.6 km). Moreover, it is in line with the results of Schmutz et al. (2014), who observed a higher effect of restoration on the number of rheophilic fish species in long restored as compared to short restored sections but only at length greater than 3.8 km.

- Restoration extent (length of restored section, restoration intensity) is not the main factor determining restoration effects in projects comparable to the once investigated. Most probably, restoration projects implemented in the past were simply too small to benefit from possible positive mitigating effects of restoration extent.

Restoration measures: Widening was applied in many of the projects investigated (11 out of 20) and had a significantly larger effect on ground beetle richness and diversity as well as richness of helophytes compared to other, less intensive measures (e.g. remeandering, flow restoration, instream measures). The results for macrophytes (helophytes) were consistent with the findings of Kail and Angelopulos (2014) who reported that widening had a positive and much higher effect on the richness/diversity of macrophytes compared to fish and macroinvertebrates. Moreover, widening projects also tended to have a higher effect on macroinvertebrate richness as well as macro- and mesohabitat diversity. This is consistent with the widely endorsed assumption that restoring geomorphological processes in longer reaches by e.g. removing bed and bank fixation and widening has a higher effect on hydromorphology and biota compared to other non-process based measures like gravel addition. The projects classified as widening usually comprised a set of restoration measures, including the removal of bed and bank fixation, flattening the river banks, and considerably widening the cross-section in some cases. Therefore, it is difficult to disentangle the effect of single measures and their contribution to the overall effect. Since the positive effect on ground beetles was mainly due to the creation of open pioneer habitats covered by sparse woody vegetation, even single measures like flattening river banks might already suffice but this has to be further investigated.

The higher effect of widening projects do not question the use of pure instream measures restricted to the river bed since transferability in respect to instream measures is limited due to the relatively low number of projects which mainly applied such kind of measures (n=4). In a recent meta-analysis based on a larger number of restoration projects, instream measures in the wetted channel had a significant positive effect on either richness/diversity or abundance/biomass of fish and macroinvertebrates (Kail and Angelopoulos 2014). Moreover, Miller et al. (2010) reported a significant positive effect of typical instream measures (large wood and boulder placement) on macroinvertebrate richness in a meta-analysis.

- Widening (removing bed and bank fixation, flattening river banks, and in some projects considerably widening the cross-section) is one of the most effective restoration measure, especially for ground beetles and macrophytes but instream measures in the wetted channel also can have a significant positive effect.

Habitat conditions: The results indicated that it is crucial to ensure the restoration projects enhance habitat conditions at spatial scales relevant for biota. For ground beetles, the positive effect of widening was mainly due to the strong relationship between ground beetle richness and a specific habitat type: the open pioneer stage covered by sparse woody vegetation, but not to the mere number of habitat types. The effect of restoration on macroinvertebrates (quantifed by the dissimilarty of restored and degraded sections) was higher in restored sections where subsrates differed more strongly from the degraded sections (Chapter 3). Moreover, macroinvertebrate richness and diversity was correlated with microhabitat diversity (Chapter 5). Since restoration had a very low effect on substrate diversity at the microhabitat scale, this possibly was one of the main reasons for the low effect of restoration on macroinvertebrates. Surprisingly, a high effect of restoration on macro- and mesohabitat diversity was not associated with a high effect on microhabitats. Therefore, although a restoration project has enhanced macro- and mesohabitats which often is visually appealing, it still may have failed at increasing microhabitat diversity relevant for macroinvertebrates. Therefore, future projects should aim at increasing and monitoring habitat diversity at spatial scales which are ecologically relevant for the targeted organism groups. Even if microhabitat diversity is improved, other factors seem to constrain the effect of restoration (similar to Liebig’s law of the minimum), and hence, it is essential to identify the main reasons for the low effect of restoration on some organism groups since the different reasons involve completely different restoration strategies.

- It is not necessarily most important to increase the mere number of habitat types (e.g. habitat diversity) but to restore specific habitats which are of special importance.
- It is crucial to ensure that restoration measures create habitats at spatial scales relevant for biota (e.g. substrate diversity at the microhabitat scale for invertebrates).

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12. Annex

12.1 Annex A: List of variables compiled in the database

Table 12-1 Variables and descriptions of the database sheet "Site information"

Parameter	Description
StationCode	unique internal or national code
StationName	Station name [national name of case study reach]
CasestudyType	Type of case study: R1 = large restoration, R2 = small restoration, D1 = degraded reach corresponding to large restoration reach, D2 = degraded reach corresponding to small restoration reach [R1, R2, D1, D2]
BeginLongitude	Begin of reach upstream - Longitude [degrees W (-) or E (+), decimal], WGS84
BeginLatitude	Begin of reach upstream - Latitude [degrees N, decimal], WGS84
EndLongitude	End of reach downstream - Longitude [degrees W (-) or E (+), decimal], WGS84
EndLatitude	End of reach downstream - Latitude [degrees N, decimal], WGS84
ReachLength	Length of reach [km]
ReachArea_cat	Floodplain area (the river and valley bottom, that is flooded and shaped by dynamic processes - under natural conditions!); categorised (<1, 1-10, 10-50, 50-100, 100-500, >500) [ha]
ReachArea_exc	Floodplain area (the river and valley bottom, that is flooded and shaped by dynamic processes - under natural conditions!); exact value (if known) [ha]
FloodplainWidth	Average Floodplain width (average of floodplain transects every 100m of the reach)[m]
AltitudeBegin	Altitude at begin of reach (Meters Above Sea Level) [m]
AltitudeEnd	Altitude at end of reach (Meters Above Sea Level) [m]
Elevation	Elevation, according to the WFD categories [lowland = <200 m, mid-altitude = 200-800 m, high = >800 m]
RiverName	National river name
StrOrder	Stream order, according to Strahler
WaterbodyID	Water body code, according to River Basin Management Plan (RBMP)
CountryID	Country code (see country code list)
BQE_Type	Biological Quality Elements (BQE) of original station [1 = fish, 2 = invertebrates, 3 = phytobenthos, 4 = macrophytes (combinations possible)]
MultipleBQE	Are there more than one BQE samples for this site with the specified timeframe? [yes / no / nodata]
EcoregionID	Ecoregion number, according to WFD (see ecoregions list)
GeologicalType	Geological type according to WFD [calcareous / silicious / organic]
Geol_cat	Geology of catchment upstream according to WFD [calcareous / silicious / organic]
CatchmentArea	Catchment size [km²]
CatchmentCat	Catchment categories according to the WFD categories [very small = <10km² / small = 10-100km² / medium = 100-1000km² / large = 1000-10000km² / very large = >10000km²]
CatchmentName	National name of catchment
MainRiverRegion	Name of main river region
RestDate	Year(s) of restoration [yyyy; yyyy-yyyy]
Rest_TimeAfter_exc	Time after restoration [in years]
Rest_TimeAfter_cat	Time after restoration, categorised [1 = 0-1y / 2 = 2-4y / 3 = 5-12y]

Table continued

Parameter	Description
Mon_Rest	Repeated monitoring, present or absent [0 = no / 1 = yes]
Mon_Freq	Frequency of monitoring [monthly / semi-annually / annually / only once / other { e.g. biennial,…} / nodata]
Mon_Time	Time between implementation of project(s) and monitoring [month]
UpstreamLakes	(if the lake upstream affects a site) [yes / no / nodata]
ProjSum	Brief summary of the project location, pressure situation, objectives and implemented measures
Pictures	pictures (with description) before and after restoration of this site or pictures of degraded reach
ReporterID_site	name and organisation of the person who obtained the data
DataSourceID_site	database name, report, etc. of the obtained data
Comment_site	Any other comment
CatchCode	Code of the catchment polygon
CLC1Perc	CLC2006 class 1 in catchment polygon (% coverage) - Artificial surfaces
CLC2Perc	CLC2006 class 2 in catchment polygon (% coverage) - Agricultural areas
CLC3Perc	CLC2006 class 3 in catchment polygon (% coverage) - Forests and semi-natural areas
CLC4Perc	CLC2006 class 4 in catchment polygon (% coverage) - Wetlands
CLC5Perc	CLC2006 class 5 in catchment polygon (% coverage) - Water bodies
CatchArea(km2)	Area of catchment polygon [km^2]

Table 12-2 Variables and descriptions of the database sheet "Hydromorphology"

Parameter	Description
ID_SC	code of referring site (field "StationCode" of table Site information)
ID_HM	consecutive number, beginning with 1
SampelDate	[dd.mm.yyyy]
ChanPatt	Natural channel pattern [meandering, braiding, wandering, anastomosing, constrained] classification of channel pattern should be consistent with other WPs!
ChanBankW_cat	Bankfull width of river channel (m), categorised [<5, <10, <20, <50, >50]
Slope_exc	Channel slope [in %]
Slope_cat	Channel slope, categorised [<0,1% / 0,1-0,5% / 0,5-1% / 1-3% / >3%]
FlowVel_hm	Mean flow velocity [m/s]
Discharge	Mean discharge [m^3/s]
Mean_river_width	Mean width of water body [m]
Mean_river_depth	Mean depth of water body [m]
River_width_min	Minimum width of water body [m]
River_width_max	Maximum width of water body [m]
River_depth_min	Minimum depth of water body [m]
River_depth_max	Maximum depth of water body [m]
FlowDiversity	Type-specific flow diversity [present, slightly reduced, reduced, absent]
DepthVariability	Type-specific depth variability [present, slightly reduced, reduced, absent]
Substrate_dom_ID	Dominating substrate, categorised; according to codelist
Substrate_divers	Type specific substrate diversity [present, slightly reduced, reduced, absent]
BedFixation	Bed-fixation [yes / no / nodata]
InstrHabit	Type specific instream habitats [present, slightly reduced, reduced, absent] (e.g. sediment bars, pools, rapids, cascades)
RiverDynamics	Features indicating type specific river dynamics [present, slightly reduced, reduced, absent] (e.g. woody debris, undercut banks, islands,..)
Barriers_art	Artificial barriers [present / absent] (e.g. dams, weirs)
ChanForm_modified	Channel form modified [no / intermediate / straightened]
CrossSect_modified	Cross section modified [no / intermediate / technical profile]
Artific_Embank	Artificial embankment [no / slight / intermediate / high]
RiparianVeg_modified	Riparian vegetation modified [no / slight / intermediate / high]
FloodplHabitat	Type specific floodplain habitats [present, slightly reduced, reduced, absent]
BufferZone	Nature-like or extensive land use in the adjacent area along the river - riparian buffer strip [present / absent]
FloodplLanduse_cat	Landuse of floodplain; categorised [(near-)natural / extensive agriculture / intensive agriculture / urban / forestry]
HymoStatus	Mean hydromorphological status [1 = very good / 2 = good / 3 = moderate / 4 = poor / 5 = bad]
HymoStat_Method	Name of hydromorphological survey method
Detail_Hymo	Detailed hydromorphological datasets available [yes / no; if yes]
Detail_Hydrol	Detailed hydrological datasets available [yes, no; if yes, please specify]
HydrolModel	Hydrological model available [yes / no / nodata; if yes, please specify]
ReporterID_hyd	name and organisation of the person who obtained the data
DataSourceID_hyd	database name, report, etc. of the obtained data

Table 12-3 Variables and descriptions of the database sheet "Pressure types"

Parameter	Description
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_PR	Consecutive number, beginning with 1
Impoundment	Impoundments or stagnation [yes / no / nodata]
Hydropeaking	Height of hydropeaking or puls releases [cm]
WaterAbstraction	Water abstraction [yes / no / nodata]
SurfWaterAbstr	Surface water abstraction [yes / no / nodata]
GroundwAbstr	Groundwater abstraction [yes / no / nodata]
FlowRegulation	Change of hydrological regime [yes / no / nodata]
FlowVelIncrease	Flow velocity increase [yes / no / nodata]
SedimentStor	Sediment storage upstream [yes / no / nodata]
NutrPollution	Nutrient pollution [yes / no / nodata]; if yes, specify: [point / diffuse]
Morph_alter	Alteration of morphology: Channelization [yes / no / nodata]
RipVeg_alter	Alteration of riparian vegetation [yes / no / nodata]
InstrHabit_alter	Alteration of instream habitats [yes / no / nodata]
MorphDike	Presence of embankments, levees or dikes [yes / no / nodata]
Sedim_artif	Artificially induced (increased) sedimentation (deposition) [yes / no / nodata]
Sedim_extrac	Sand and gravel extraction, dredging [yes / no / nodata]
BarriersCatchmUp	Presence of barriers in catchment upstream [yes / no / nodata]
BarriersCatchmDown	Presence of barriers in catchment downstream [yes / no / nodata]
NumberBarrierUp	Number of barriers in catchment upstream
NumberBarrierDown	Number of barriers in catchment downstream
DistNextBarrUp	Distance to next barrier upstream [km]
DistNextBarrDown	Distance to next barrier downstream [km]
WaterUse	Water use [HP= Hydropower / I = Irrigation / DW = Drinking Water / SP = Snow Production /FP = Fishponds / CW = Cooling Water/ IW = Industrial Water]; if there are others, please specify; multiple answers possible
PressCatchmUp	Pressure types catchment upstream [CH = channelization / IP = impoundment /WA = water abstraction / HP = hydropeaking / PO = pollution /FA = flow alteration / SA = sediment alteration]; if there are others, please specify; multiple answers possible
ReporterID_pres	name and organisation of the person who obtained the data
DataSourceID_pres	database name, report, etc. of the obtained data
Comment_pres	Any other comment

Table 12-4 Variables and descriptions of the database sheet "Restoration measures"

Parameter	Description
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_RM	Consecutive number, beginning with 1
CompLatShare	CompLat/4
CompAllShare	CompAll/8
CompLat	Sum of M_InC; M_Rip; M_Plan; M_Flood
CompAll	Sum of M_Hydrol; M_Sed; M_Flow; M_Conect; M_InC; M_Rip; M_Plan; M_Flood
M_Hydrol	Sum of MH_Abstr; MH_Ret; MH_GW; MH_Stor; MH_Min; MH_Div; MH_Cycle; MH_Cons
M_Sed	Sum of MS_Add; MS_Input; MS_Reser; MS_Trans; MS_Trap; MS_Dredg
M_Flow	Sum of MF_EFlow; MF_HPeak; MF_FPlain; MF_APeak; MF_Imp; MF_MorphFlow
M_Conect	Sum of MC_Up; MC_Down; MC_Manag; MC_Remov; MC_Culv
M_InC	Part of CompLat (Sum of MIn_FixBed; MIn_FixBank; MIn_RemSed; MIn_AddSed; MIn_Veg; MIn_HyStruc; MIn_Shall; MIn_Wood; MIn_Bould; MIn_Dynamic; MIn_Riff)
M_Rip	Part of CompLat (Sum of MR_NBuff; MR_SBuff; MR_VegBuff)
M_Plan	Part of CompLat (Sum of MP_Meander; MP_Wide; MP_Shallow; MP_Narrow; MP_LowC; MP_Dynamic; MP_2Flod)
M_Flood	Part of CompLat (Sum of MFP_Con; MFP_Create; MFP_Lower; MFP_Back; MFP_Remove; MFP_other)
MP_PointS	Decrease of point source pollution [yes / no / nodata]
MP_DiffS	Decrease of diffuse nutrient or pollution input (other than buffer strips!) [yes / no / nodata]
MH_Abstr	Reduction of surface water abstraction without return [yes / no / nodata]
MH_Ret	Improvement of water retention (e.g. on floodplain, urban areas, overlaps with MFlow_APeak) [yes / no / nodata]
MH_GW	Reduction of groundwater abstraction [yes / no / nodata]
MH_Stor	Improvement/creation of water storage (e.g. polders) [yes / no / nodata]
MH_Min	Increase of minimum flow (to generally increase discharge in a reach or to improve flow dynamics) [yes / no / nodata]
MH_Div	Improving water quantity by water diversion and transfer [yes / no / nodata]
MH_Cycle	Recycling of used water (off-site measure to reduce water consumption) [yes / no / nodata]
MH_Cons	Reduction of water consumption (other measures than recycling used water) [yes / no / nodata]
MS_Add	Adding/feeding of sediment (e.g. downstream from dam) [yes / no / nodata]
MS_Input	Reduction of undesired sediment input (e.g. from agricultural areas or from bank erosion other than riparian buffer strips!) [yes / no / nodata]
MS_Reser	Prevention of sediment accumulation in reservoirs [yes / no / nodata]
MS_Trans	Improvement of continuity of sediment transport (e.g. manage dams for sediment flow) [yes / no / nodata]
MS_Trap	Trapping of sediments (e.g. building sediment traps to reduce washload) [yes / no / nodata]
MS_Dredg	Reduction of impact of dredging [yes / no / nodata]
MF_EFlow	Establishment of environmental flows / naturalise flow regimes (does focus on discharge variability compared to water quantity of MH_Min) [yes / no / nodata]
MF_HPeak	Modification of hydropeaking [yes / no / nodata]
MF_FPlain	Increase of flood frequency and duration in riparian zones or floodplains [yes / no / nodata]

Table continued

Parameter	Description
MF_APeak	Reduction of anthropogenic flow peaks [yes / no / nodata]
MF_Imp	Shortening the length of impounded reaches [yes / no / nodata]
MF_MorphFlow	Favouring morphogenic flows (can also be considered a measure to improve planform or in-channel habitat conditions) [yes / no / nodata]
MC_Up	Installing fish pass, bypass, side channel for upstream migration [yes / no / nodata]
MC_Down	Installing facilities for downstream migration (including fish friendly turbines) [yes / no / nodata]
MC_Manag	Management sluice, weir, and turbine operation for fish migration [yes / no / nodata]
MC_Remov	Removal of barrier (e.g. dam or weir) [yes / no / nodata]
MC_Culv	Modification or removal of culverts, syphons, piped streams [yes / no / nodata]
MIn_FixBed	Removal of bed fixation [yes / no / nodata]
MIn_FixBank	Removal of bank fixation [yes / no / nodata]
MIn_RemSed	Removal of sediment (e.g. mud from groin fields) [yes / no / nodata]
MIn_AddSed	Adding of sediment (e.g. gravel, overlaps with MS_Add) [yes / no / nodata]
MIn_Veg	Management of aquatic vegetation (e.g. mowing) [yes / no / nodata]
MIn_HyStruc	Removal or modification of in-channel hydraulic structures (e.g. groins, bridges) [yes / no / nodata]
MIn_Shall	Creation of shallows near the bank [yes / no / nodata]
MIn_Wood	Recruitment or placement of large wood [yes / no / nodata]
MIn_Bould	Placement of boulders [yes / no / nodata]
MIn_Dynamic	Initiation of natural channel dynamics to promote natural regeneration [yes / no / nodata]
MIn_Riff	Placement of artificial gravel bar or riffle [yes / no / nodata]
MR_NBuff	Development of buffer strips to reduce nutrient input [yes / no / nodata]
MR_SBuff	Development of buffer strips to reduce fine sediment input [yes / no / nodata]
MR_VegBuff	Development of natural vegetation on buffer strips (other reasons than nutrient or sediment input, e.g. shading, organic matter input) [yes / no / nodata]
MP_Meander	Remeandering of water course (actively changing planform) [yes / no / nodata]
MP_Wide	Widening or re-braiding of water course (actively changing planform) [yes / no / nodata]
MP_Shallow	Creation of shallow water course (actively increasing level of channel-bed) [yes / no / nodata]
MP_Narrow	Creation of narrow over-widened water course (actively changing width) [yes / no / nodata]
MP_LowC	Creation of low-flow channels in over-sized channels [yes / no / nodata]
MP_Dynamic	Allowing/initiation of lateral channel migration (e.g. by removing bank fixation and adding large wood) [yes / no / nodata]
MP_2Flod	Creation of secondary floodplain on present low level of channel bed (floodplain compensation) [yes / no / nodata]
MFP_Con	Reconnection of existing backwaters, oxbow-lakes, wetlands [yes / no / nodata]
MFP_Create	Creation of semi-natural / artificial backwaters, oxbow-lakes, wetlands [yes / no / nodata]
MFP_Lower	Lowering embankments, levees or dikes to enlarge inundation and flooding [yes / no / nodata]

Table continued

Parameter	Description
MFP_Back	Back-removal of embankments, levees or dikes to enlarge the active floodplain area [yes / no / nodata]
MFP_Remove	Removal of embankments, levees or dikes or other engineering structures that impede lateral connectivity [yes / no / nodata]
MFP_other	Other measures concerning Floodplain and Vegetation [yes / no / nodata]
MFP_other_name	If you anwered the field "MFP_other" with yes, please specify this measure here
RestLimits	Constraints or limiting factors which might have impeded restoration effects (e.g. multiple stressors, key habitats still missing) [yes / no / nodata]
RestCosts_plann	Planning and project design costs before project implementation [EUR]
RestCosts_dike	Construction costs for dike relocation or extension, if relevant [EUR]
RestCosts_transc	Transaction costs such as administrative and legislative costs [EUR]
RestCosts_acqu	Land acquisition costs, if relevant [EUR]
RestCosts_oth	Other construction and investment costs [EUR]
RestCosts_maint	Annual maintenance costs after project implementation [EUR]
RestCosts_monit	Annual monitoring costs after project implementation [EUR]
RestCosts_total	Total consts of restoration [EUR]
ReporterID_res	name and organisation of the person who obtained the data
DataSourceID_res	database name, report, etc. of the obtained data
Comment_res	Any other comment

Table 12-5 Variables and descriptions of the database sheet "Physico-chemical data"

Parameter	Description
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_PC	Enter a consecutive number, beginning with 1
SampleMean	Does this data set contain means of more than one sample [yes / no / nodata]
SampleDateStart	If you answered the field "SampleMean" with yes, enter the date of your first sample here; if you answered the field "SampleMean" with no, enter the date of your individual sample here [dd.mm.yyyy; hh:mm]
SampleDateEnd	If you answered the field "SampleMean" with yes, enter the date of your last sample here; if you answered the field "SampleMean" with no, leave this field empty [dd.mm.yyyy; hh:mm]
pH	PH 0-14; value at sampling time
WaterTemp	Water temperature, value at sampling time [°C]
Conductivity	Electrical conductivity, value at sampling time [microS/cm];
Oxygen	Oxygen content, value at sampling time [mg/l]
OxygenSaturation	Oxygen saturation [%], if applicable
BOD5	Biological oxygen demand [mg/l]
Nitrite	Nitrite [mg/l] (NOT Nitrit-N!)
Nitrate	Nitrate [mg/l]
Ammonia	Ammonia [mg/l]
Chloride	Chloride [mg/l]
OrthoPhosphate	Ortho-phosphate [microg/l] (NOT PO4-P!)
TotalPhosphate	Total-phosphate [microg/l]
Alkalinity	Alkalinity [mval/l], if applicable
ReporterID_phych	name and organisation of the person who obtained the data
DataSourceID_phych	database name, report, etc. of the obtained data
Comment_phych	Any other comment

Table 12-6 Variables and descriptions of the database sheets "Fish"

Parameter	Description
FishSite	
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_site_fish	Consecutive number, beginning with 1
Site_altitude_fish	Altitude [m]
Site_name_fish	Local or internal name of site
Site_SectBegLong_fish	Begin of sample section upstream - Longitude [degrees W (-) or E (+), decimal], WGS84
Site_SectBegLati_fish	Begin of sample section upstream - Latitude [degrees N, decimal], WGS84
Site_SectEndLong_fish	End of sample section downstream - Longitude [degrees W (-) or E (+), decimal], WGS84
Site_SectEndLati_fish	End of sample section downstream - Latitude [degrees N, decimal], WGS84
FishSample	
ID_site_fish	Number of referring fish site (field "ID_site_fish" of table FishSite)
ID_sample_fish	Consecutive number, beginning with 1
FishSampleDate	Date of sample [dd.mm.yyyy]
SamplingMethod	e.g. electrofishing, demersal line, beach seine, gill net
BoatWade	[boat / wading]
Anode	[fixed / handheld]
AnodeNo	Number of handheld anodes
GenPower	Power of generator [kW]
Voltage	Voltage, e.g. 300 or 600 [V]
Amperage	Amperage during sampling [A]
Barrier	Barrier at upstream end of sample [yes / no / nodata]
SampStrat_el	Sampling strategy (if electrofishing) [partial habitat / partial strip / serial removal]
SampStrat_el_No	Sampling strategy (if electrofishing) [number of runs]
SamplingDuration	Sampling duration / exposure time [hh:mm]
SampleLength	Length of sample [m]
SampleWidth	Width of sample [m]
SampleRiverWidth	River width at sampling site [m]
SampleRiverDepth_av	Average depth at sampling site [cm]
SampleRiverDepth_max	Maximum depth at sampling site [cm]
FlowVel_fish	Flow velocity at sampling site [m/s]
SampleType	[midstream / riparian zone / whole width]
SampleHabitat	[main channel / side channel connected / backwater / oxbow]
SampleHabitatStruct	[rock / boulders / gravel / sand / mud / litter / woody debris / reeds / submersal macrophytes / riparian vegetation]; multiple answers are permitted
CaptureEfficiency	Capture efficiency; estimated for each species and different age/size classes: 100% is the total of visually detected fish [%]
ReporterID_fish	name and organisation of the person who obtained the data
DataSourceID_fish	database name, report, etc. of the obtained
Comment_fish	Any other comment

Table continued

Parameter	Description
FishCatch	
ID_sample_fish	Number of referring sample (field "ID_sample_fish" of table FishSample)
ID_catch_fish	Consecutive number, beginning with 1
FishSpeciesID	ID of scientific name of species (see taxa list)
FishLength	Length of fish [mm]
FishWeight	Weight of fish [g]
Sex	[m/f]
FishAbundJuv	Abundance data of whole community (juveniles), area-related [absolute no. of individuals in sample]
FishAbundAdu	Abundance data of whole community (adults), area-related [absolute no. of individuals in sample]
AddInfo	

Table 12-7 Variables and descriptions of the database sheets "Invertebrates"

Parameter	Description
InvSite	
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_site_inv	Consecutive number, beginning with 1
Site_Long_inv	Longitude at midpoint of sample reach [degrees W (-) or E (+), decimal], WGS84
Site_Lati_inv	Latitude at midpoint of sample reach [degrees N, decimal], WGS84
Site_altitude_inv	Altitude [m]
Site_name_inv	Local or internal name of site
InvSample	
ID_site_inv	Number of referring site (field "ID_site_inv" of table InvSite)
ID_sample_inv	Consecutive number, beginning with 1
InvSampleDate	Date of sample [dd.mm.yyyy]
InvSampleMeth	Sampling methode
Sample_area_inv	Sampling area [m²]
ReporterID_inv	name and organisation of the person who obtained the data
DataSourceID_inv	database name, report, etc. of the obtained data
Comment_inv	Any other comment
InvCatch	
ID_sample_inv	Number of referring sample (field "ID_sample_inv" of table InvSample)
ID_catch_inv	Consecutive number, beginning with 1
Inv_spec_ID	Species ID of invertebrates (according to codelist)
InvSpecAbund	Abundance of species

Table 12-8 Variables and descriptions of the database sheets "Macrophytes"

Parameter	Description
MacrophSite	
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_site_mph	Consecutive number, beginning with 1
MPhSiteLong	Longitude of sample site [degrees W (-) or E (+), decimal], WGS84; midpoint of sample reach
MPhSiteLati	Latitude of sample site [degrees N, decimal], WGS84; midpoint of sample reach
MPhSite_name	Local or internal name of site
MacrophSample	
ID_site_mph	Number of referring sample (field "ID_site_mph" of table MacrophSite)
ID_sample_mph	Consecutive number, beginning with 1
MPhSampleDate	Date of sample [dd.mm.yyyy]
MPhSampleMeth	Sampling methode
ReporterID_mph	Name and organisation of the person who obtained the data
DataSourceID_mph	database name, report, etc. of the obtained data
Comment_mph	Any other comment
MacrophCatch	
ID_sample_mph	Number of referring sample (field "ID_sample_mph" of table MacrophSample)
ID_catch_mph	Consecutive number, beginning with 1
MPhTaxonID	Macrophytes Taxon ID (according to codelist)
MPhEmSub	[emergent / submerged]
MPhGrowthForm	ID of growth form, according to Den Hartog & Van der Velde 1988 and Wiegleb 1991 (see codelist)
MPhAbundance	Abundance of Species; 5-point scale, according to Kohler (1978) [1 = very rare, 2 = rare, 3 = common, 4 = frequent, 5 = abundant, predominant]

Table 12-9 Variables and descriptions of the database sheets "Riparian beetles"

Parameter	Description
BeetSite	
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_site_beet	Consecutive number, beginning with 1
BeetSiteLong	Longitude of sample site [degrees W (-) or E (+), decimal], WGS84; midpoint of sample reach
BeetSiteLati	Latitude of sample site [degrees N, decimal], WGS84; midpoint of sample reach
BeetSite_name	Local or internal name of site
BeetSample	
ID_site_beet	Number of referring sample (field "ID_site_beet" of table BeetSite)
ID_sample_beet	Consecutive number, beginning with 1
BeetSampleDate	Date of sample; for pitfall traps date of installing traps [dd.mm.yyyy]
BeetMH_RipFor	Coverage of mesohabitat 'Riparian forest' [%]: > 25% coverage of woody riparian vegetation; trees cover the area
BeetMH_Past	Coverage of mesohabitat 'Pasture' [%]: Gras land (no tree cover)
BeetMH_Ohv	Coverage of mesohabitat 'Other herbaceous vegetation' [%]: Riparian herbaceous vegetation (no tree cover)
BeetMH_VegS	Coverage of mesohabitat 'Vegetated swamp' [%]: very moist (muddy) vegetated patches
BeetMH_Ogbr	Coverage of mesohabitat 'Open gravel bank/bar' [%]: < 25% vegetation coverage
BeetMH_Osbr	Coverage of mesohabitat 'Open sand bank/bar' [%]: < 25% vegetation coverage
BeetMH_Ombr	Coverage of mesohabitat 'Open mud bank/bar' [%]: < 25% vegetation coverage
BeetMH_Sue	Coverage of mesohabitat 'Steep unvegetated embankment' [%]: < 25% vegetation coverage
BeetMH_othName	If mesohabitat is present, that does not fit to the classification of mesohabitats above, please specify
BeetMH_Oth	Coverage of other mesohabitat, if it doesn't fit to defined mesohabitats [%]
ReporterID_beet	name and organisation of the person who obtained the data
DataSourceID_beet	database name, report, etc. of the obtained data
Comment_beet	Any other comment
BeetCatch	
ID_sample_beet	Number of referring sample (field "ID_sample_beet" of table BeetSample)
ID_catch_beet	Consecutive number, beginning with 1
BeetSampleMeth	Sampling methode [hand collection / pitfall trap]
BeetSampleNo	Enter consecutively from 1 within a sample; each subsample (each trap and handcollection get a unique number)
BeetMesohab	Name of mesohabitat that was sampled
BeetTaxonID	Beetles Taxon ID (according to codelist)
BeetAbundance	Abundance of species absolute

Table 12-10 Variables and descriptions of the database sheets "Floodplain Vegetation"

Parameter	Description
VegSite	
ID_SC	Code of referring site (field "StationCode" of table Site information)
ID_site_veg	Consecutive number, beginning with 1
VegSiteLong	Longitude of sample site [degrees W (-) or E (+), decimal], WGS84; midpoint of sample reach
VegSiteLati	Latitude of sample site [degrees N, decimal], WGS84; midpoint of sample reach
VegSite_name	Local or internal name of site
VegSample	
ID_site_veg	Number of referring sample (field "ID_site_veg" of table VegSite)
ID_sample_veg	Consecutive number, beginning with 1
VegSampleDate	Date of sample [dd.mm.yyyy]
ReporterID_veg	name and organisation of the person who obtained the data
DataSourceID_veg	database name, report, etc. of the obtained data
Comment_veg	Any other comment
VegTransUnit	
ID_sample_veg	Number of referring sample (field "ID_sample_veg" of table VegSample)
ID_transect_veg	Transect number (1, 2 or 3) at which the (length of) vegetation order/unit was mapped
ID_VegCode	Each order or unit can appear more than once per transect or site, so each one is counted separately
ID_vegorder	ID of vegetation order of community (according to codelist)
ID_vegunit	ID of vegetation unit of community (according to codelist)
Veg_othName	If there is a vegetation order or unit that you don't find in the list, please specify here
VegUnitLength	Length of vegetation orders/units at transect number x for all vegetation orders/units present in a sample site [m]
VegTaxa	
ID_sample_veg	Number of referring sample (field "ID_sample_veg" of table VegSample)
ID_transect_veg	Number of referring transect (field "ID_transect_veg" of table VegTransUnit)
ID_VegCode	Number of referring field "ID_VegCode" of table VegTransUnit
VegTaxonID	Taxon ID of plants (according to codelist)
VegTaxonCoverage	Coverage of the taxon within a vegetation unit at an area of 2x3 meters [abundance classes: 1%, 5%, 10%, 15%, 20% and continuing in 10%-steps up to 100%]; only for 3 mapped areas per vegetation unit within a sample site

Table 12-11 Variables and descriptions of the database sheet "BQE Status"

Parameter	Description
StationCode	Station code; unique internal or national code (land code_river name_casestudy type -> eg.: FI_Kuiv_R1, AT_Drau_D1, etc.)
EQC_all_site	Ecological quality class - at case study site [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the site includes more than one waterbody - calculate mean value weighted by length
EQC_all_0 to 1 km_up	Ecological quality class - from upstream end of the site to 1 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length
EQC_all_1 to 5 km_up	Ecological quality class - from 1 to 5 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
EQC_all_5 to 30 km_up	Ecological quality class - from 5 to 30 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Benin_site	Biological quality class - benthic invertebrates at case study site [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the site includes more than one waterbody - calculate mean value weighted by length
BQC_Benin_0 to 1 km_up	Biological quality class - benthic invertebrates - from upstream end of the site to 1 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Benin_1 to 5 km_up	Biological quality class - benthic invertebrates - from 1 to 5 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Benin_5 to 30 km_up	Biological quality class - benthic invertebrates - from 5 to 30 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Macphy_site	Biological quality class - macrophytes at case study site [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the site includes more than one waterbody - calculate mean value weighted by length
BQC_Macphy_0 to 1 km_up	Biological quality class - macrophytes - from upstream end of the site to 1 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Macphy_1 to 5 km_up	Biological quality class - macrophytes - from 1 to 5 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Macphy_5 to 30 km_up	Biological quality class - macrophytes - from 5 to 30 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Phyben_site	Biological quality class - Phytobenthos at case study site [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the site includes more than one waterbody - calculate mean value weighted by length
BQC_Phyben_0 to 1 km_up	Biological quality class - Phytobenthos - from upstream end of the site to 1 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length

Table continued

Parameter	Description
BQC_Phyben_1 to 5 km_up	Biological quality class - Phytobenthos - from 1 to 5 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Phyben_5 to 30 km_up	Biological quality class - Phytobenthos - from 5 to 30 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Diatoms_site	Biological quality class - Diatoms at case study site [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the site includes more than one waterbody - calculate mean value weighted by length
BQC_Diatoms_0 to 1 km_up	Biological quality class - Diatoms - from upstream end of the site to 1 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Diatoms_1 to 5 km_up	Biological quality class - Diatoms - from 1 to 5 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Diatoms_5 to 30 km_up	Biological quality class - Diatoms - from 5 to 30 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Fish_site	Biological quality class - Fish at case study site [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the site includes more than one waterbody - calculate mean value weighted by length
BQC_Fish_0 to 1 km_up	Biological quality class - Fish - from upstream end of the site to 1 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Fish_1 to 5 km_up	Biological quality class - Fish - from 1 to 5 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Fish_5 to 30 km_up	Biological quality class - Fish - from 5 to 30 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Fish_0 to 1 km_down	Biological quality class - Fish - from downstream end of the site to 1 km downstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Fish_1 to 5 km_down	Biological quality class - Fish - from 1 to 5 km downstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Fish_5 to 30 km_down	Biological quality class - Fish - from 5 to 30 km downstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Phypla_site	Biological quality class - Phytoplankton at case study site [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the site includes more than one waterbody - calculate mean value weighted by length

Table continued

Parameter	Description
BQC_Phypla_0 to 1 km_up	Biological quality class - Phytoplankton - from upstream end of the site to 1 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] - if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Phypla_1 to 5 km_up	Biological quality class - Phytoplankton - from 1 to 5 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
BQC_Phypla_5 to 30 km_up	Biological quality class - Phytoplankton - from 5 to 30 km upstream [1 = high, 2 = good, 3 = moderate, 4 = poor, 5 = bad, 0 = nodata] if the buffer includes more than one waterbody - calculate mean value weighted by length
ReporterID_BQE	name and organisation of the person who obtained the data;
DataSourceID_BQE	database name, report, etc. of the obtained data
Comment_BQE	Any other comment

Table 12-12 Variables and descriptions of the database sheet "Colonization Sources"

Parameter	Description
StationCode	Station code; unique internal or national code (land code_river name_casestudy type -> eg.: FI_Kuiv_R1, AT_Drau_D1, etc.)
NHES_Benin_0 to 1 km_up	Number of water bodys with high or good ecological status - BQE benthic invertebrates - from upstream end of the site to 1 km upstream
NHES_Benin_1 to 5 km_up	Number of water bodys with high or good ecological status - BQE benthic invertebrates - from 1 to 5 km upstream
NHES_Benin_5 to 30 km_up	Number of water bodys with high or good ecological status - BQE benthic invertebrates - from 5 to 30 km upstream
TLHES_Benin_0 to 1 km_up	Total length of water bodies with high or good ecological status - BQE benthic invertebrates- from upstream end of the site to 1 km upstream [km]
TLHES_Benin_1 to 5 km_up	Total length of water bodies with high or good ecological status - BQE benthic invertebrates- from 1 to 5 km upstream [km]
TLHES_Benin_5 to 30 km_up	Total length of water bodies with high or good ecological status - BQE benthic invertebrates- from 5 to 30 km upstream [km]
DataSource__CCS_Benin	database name, report, etc. of the obtained data
NHES_Macphy_0 to 1 km_up	Number of water bodys with high or good ecological status - BQE macrophytes - from upstream end of the site to 1 km upstream
NHES_Macphy_1 to 5 km_up	Number of water bodys with high or good ecological status - BQE macrophytes - from 1 to 5 km upstream
NHES_Macphy_5 to 30 km_up	Number of water bodys with high or good ecological status - BQE macrophytes - from 5 to 30 km upstream
TLHES_Macphy_0 to 1 km_up	Total length of water bodies with high or good ecological status - BQE macrophytes - from upstream end of the site to 1 km upstream [km]
TLHES_Macphy_1 to 5 km_up	Total length of water bodies with high or good ecological status - BQE macrophytes - from 1 to 5 km upstream [km]
TLHES_Macphy_5 to 30 km_up	Total length of water bodies with high or good ecological status - BQE macrophytes- from 5 to 30 km upstream [km]
DataSource__CCS_Macphy	database name, report, etc. of the obtained data
NHES_Phyben_0 to 1 km_up	Number of water bodys with high or good ecological status - BQE phytobenthos - from upstream end of the site to 1 km upstream
NHES_Phyben_1 to 5 km_up	Number of water bodys with high or good ecological status - BQE phytobenthos - from 1 to 5 km upstream

Table continued

Parameter	Description
NHES_Phyben_5 to 30 km	Number of water bodys with high or good ecological status - BQE phytobenthos - from 5 to 30 km upstream
TLHES_Phyben_0 to 1 km_up	Total length of water bodies with high or good ecological status - BQE phytobenthos - from upstream end of the site to 1 km upstream [km]
TLHES_Phyben_1 to 5 km_up	Total length of water bodies with high or good ecological status - BQE phytobenthos - from 1 to 5 km upstream [km]
TLHES_Phyben_5 to 30 km	Total length of water bodies with high or good ecological status - BQE phytobenthos - from 5 to 30 km upstream [km]
DataSource__CCS_Phyben	database name, report, etc. of the obtained data
NHES_Diatoms_0 to 1 km_up	Number of water bodys with high or good ecological status - BQE diatoms - from upstream end of the site to 1 km upstream
NHES_Diatoms_1 to 5 km_up	Number of water bodys with high or good ecological status - BQE diatoms - from 1 to 5 km upstream
NHES_Diatoms_5 to 30 km_up	Number of water bodys with high or good ecological status - BQE diatoms - from 5 to 30 km upstream
TLHES_Diatoms_0 to 1 km_up	Total length of water bodies with high or good ecological status - BQE diatoms - from upstream end of the site to 1 km upstream [km]
TLHES_Diatoms_1 to 5 km_up	Total length of water bodies with high or good ecological status - BQE diatoms - from 1 to 5 km upstream [km]
TLHES_Diatoms_5 to 30 km_up	Total length of water bodies with high or good ecological status - BQE diatoms - from 5 to 30 km upstream [km]
DataSource_CCS_Diatoms	database name, report, etc. of the obtained data
NHES_Fish_0 to 1 km_up	Number of water bodys with high or good ecological status - BQE fish - from upstream end of the site to 1 km upstream
NHES_Fish_1 to 5 km_up	Number of water bodys with high or good ecological status - BQE fish - from 1 to 5 km upstream
NHES_Fish_5 to 30 km_up	Number of water bodys with high or good ecological status - BQE fish - from 5 to 30 km upstream
TLHES_Fish_0 to 1 km_up	Total length of water bodies with high or good ecological status - BQE fish - from upstream end of the site to 1 km upstream [km]
TLHES_Fish_1 to 5 km_up	Total length of water bodies with high or good ecological status - BQE fish - from 1 to 5 km upstream [km]
TLHES_Fish_5 to 30 km_up	Total length of water bodies with high or good ecological status - BQE fish - from 5 to 30 km upstream [km]
DataSource__CCS_FishUp	database name, report, etc. of the obtained data
NHES_Fish_0 to 1 km_down	Number of water bodys with high or good ecological status - BQE fish - from downstream end of the site to 1 km downstream
NHES_Fish_1 to 5 km_down	Number of water bodys with high or good ecological status - BQE fish - from 1 to 5 km downstream
NHES_Fish_5 to 30 km_down	Number of water bodys with high or good ecological status - BQE fish - from 5 to 30 km downstream
TLHES_Fish_0 to 1 km_down	Total length of water bodies with high or good ecological status - BQE fish - from upstream end of the site to 1 km downstream [km]
TLHES_Fish_1 to 5 km_down	Total length of water bodies with high or good ecological status - BQE fish - from 1 to 5 km downstream [km]
TLHES_Fish_5 to 30 km_down	Total length of water bodies with high or good ecological status - BQE fish - from 5 to 30 km downstream [km]
DataSource__CCS_FishDown	database name, report, etc. of the obtained data
NHES_Phypla_0 to 1 km_up	Number of water bodys with high or good ecological status - BQE phytoplankton - from upstream end of the site to 1 km upstream
NHES_Phypla_1 to 5 km_up	Number of water bodys with high or good ecological status - BQE phytoplankton - from 1 to 5 km upstream
NHES_Phypla_5 to 30 km_up	Number of water bodys with high or good ecological status - BQE phytoplankton - from 5 to 30 km upstream

Table continued

Parameter	Description
TLHES_Phypla_0 to 1 km_up	Total length of water bodies with high or good ecological status - BQE phytoplankton - from upstream end of the site to 1 km upstream [km]
TLHES_Phypla_1 to 5 km_up	Total length of water bodies with high or good ecological status - BQE phytoplankton - from 1 to 5 km upstream [km]
TLHES_Phypla_5 to 30 km_up	Total length of water bodies with high or good ecological status - BQE phytoplankton - from 5 to 30 km upstream [km]
DataSource__CCS_Phypla	database name, report, etc. of the obtained
ReporterID_CCS	name and organisation of the person who obtained the data
Comment_CCS	Any other comment

Table 12-13 Variables and descriptions of the database sheet "Hydromorphology" of the catchment

Parameter	Description
StationCode	Station code; unique internal or national code (land code_river name_casestudy type -> eg. FI_Kuiv_R1, AT_Drau_D1, etc.)
Name_next_gauge	Name of nearest gauging station to case study site
gauge_up_down	Location of nearest gauging station up- or downstream to case study site [up/down]
Dist_Gauge	Distance from case study site to nearest gauging station [km]
Discharge_NQ	low water level discharge [m³/s] [-999 = nodata]
Discharge_MNQ	mean low water level discharge [m³/s] [-999 = nodata]
Discharge_MQ	mean level discharge [m³/s] [-999 = nodata]
Discharge_MHQ	mean high water level discharge [m³/s] [-999 = nodata]
Discharge_HQ	high water level discharge [m³/s] [-999 = nodata]
DataSourceID_disc	database name, report, etc. of the obtained data
Discharge_data_year	discharge data recorded in year or mean values from time series from -to [yyyy/yyyy-yyyy]
HymoStatus	Mean hydromorphological status for case study site [1 = very good, 2 = good, 3 = moderate, 4 = poor, 5 = bad,0 = nodata] if the site includes more than one waterbody - calculate mean value weighted by length [-999 = nodata]
ReporterID_hyd	name and organisation of the person who obtained the data
DataSourceID_hyd	database name, report, etc. of the obtained data
Comment_hyd	Any other comment

Table 12-14 Variables and descriptions of the database sheet "Hydromorphological pressures"

Parameter	Description
StationCode	Station code; unique internal or national code (land code_river name_casestudy type -> eg.: FI_Kuiv_R1, AT_Drau_D1, etc.)
Impoundment_YN	Impoundments in catchment upstream [1 = yes, 2 = no, 0 = nodata]
Impoundment_perc	Percentage of impounded water courses in catchment upstream [%] - Total Length of water courses in catchment upstream = 100% [-999 = nodata]
Hydropeaking_YN	Waterbodies affected by Hydropeaking in catchment upstream [1 = yes, 2 = no, 0 = nodata]
Hydropeaking_perc	Percentage of water courses with hydropeaking in catchment upstream [%] - Total Length of water courses in catchment upstream = 100% [-999 = nodata]
RiverChan_YN	River channelization in catchment upstream [1 = yes, 2 = no, 0 = nodata]
RiverChan_perc	Percentage of channelized water courses in catchment upstream [%] - Total Length of water courses in catchment upstream = 100% [-999 = nodata]
WaterAbstraction_YN	Water abstraction in catchment upstream [1 = yes, 2 = no, 0 = nodata]
WaterAbstraction_perc	Percentage of water courses with residual flow in catchment upstream [%] - Total Length of water courses in catchment upstream = 100% [-999 = nodata]
tot_pres_hymo	Percentage of water courses affected by at least one HyMo pressure - each water course ist only counted once - Total Length of water courses in catchment upstream = 100% (value must not exceed 100%) [-999 = nodata]
SurfWaterAbstr_YN	Surface water abstraction [1 = yes, 2 = no, 0 = nodata]
SurfWaterAbstr_perc	Percentage of water courses affected by surface water abstraction in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
GroundwAbstr_YN	Groundwater abstraction in catchment upstream [1 = yes, 2 = no, 0 = nodata]
GroundwAbstr_perc	Percentage of water courses affected by groundwater abstraction upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
FlowRegulation_YN	Change of hydrological regime in catchment upstream [1 = yes, 2 = no, 0 = nodata]
FlowRegulation_perc	Percentage of water courses affected by change of hydrological regime upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
SedimentStor_YN	Sediment storage in catchment upstream [1 = yes, 2 = no, 0 = nodata]
SedimentStor_perc	Percentage of water courses affected by sediment storage upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
MorphDike_YN	Presence of embankments, levees or dikes in catchment upstream [1 = yes, 2 = no, 0 = nodata]
MorphDike_perc	Percentage of water courses affected by presence of embankments, levees or dikes upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
Sedim_artif_YN	Artificially induced (increased) sedimentation (deposition) in catchment upstream [1 = yes, 2 = no, 0 = nodata]
Sedim_artif_perc	Percentage of water courses affected by artificially induced (increased) sedimentation (deposition) upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
Sedim_extrac_YN	Sand and gravel extraction, dredging in catchment upstream [1 = yes, 2 = no, 0 = nodata]

Table continued

Parameter	Description
Sedim_extrac_perc	Percentage of water courses affected by sand and gravel extraction, dredging upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
BarriersCatchmUp	Presence of all barriers in catchment upstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrierUp	Number of all barriers in catchment upstream [-999 = nodata]
NumberArtBarrierUp	Number of artificial barriers upstream [-999 = nodata]
NumberBarrFishPassUp	Number of fish passable barriers upstream [-999 = nodata]
Barriers_0 to 1 km_Up_YN	Presence of all barriers from upstream end of the site to 1 km upstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrier_0 to 1 km_up	Number of all barriers from upstream end of the site to 1 km upstream [-999 = nodata]
NumberArtBarrier_0 to 1 km_up	Number of artificial barriers from upstream end of the site to 1 km upstream [-999 = nodata]
NumberBarrFishPass_0 to 1 km_up	Number of fish passable barriers from upstream end of the site to 1 km upstream [-999 = nodata]
Barriers_1 to 5 km_Up_YN	Presence of all barriers from 1 to 5 km upstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrier_1 to 5 km_up	Number of all barriers from 1 to 5 km upstream [-999 = nodata]
NumberArtBarrier_1 to 5 km_up	Number of artificial barriers from 1 to 5 km upstream [-999 = nodata]
NumberBarrFishPass_1 to 5 km_up	Number of fish passable barriers from 1 to 5 km upstream [-999 = nodata]
Barriers_5 to 30 km_Up_YN	Presence of all barriers from 5 to 30 km upstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrier_5 to 30 km_up	Number of all barriers from 5 to 30 km upstream [-999 = nodata]
NumberArtBarrier_5 to 30 km_up	Number of artificial barriers from 5 to 30 km upstream [-999 = nodata]
NumberBarrFishPass_5 to 30 km_up	Number of fish passable barriers from 5 to 30 km upstream [-999 = nodata]
BarriersCatchmDown	Presence of barriers in catchment downstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrierDown	Number of barriers in catchment downstream [-999 = nodata]
NumberArtBarrierDown	Number of artificial barriers downstream [-999 = nodata]
NumberBarrFishPassDown	Number of fish passable barriers downstream [-999 = nodata]
Barriers_0 to 1 km_down_YN	Presence of all barriers from downstream end of the site to 1 km downstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrier_0 to 1 km_down	Number of all barriers from downstream end of the site to 1 km downstream [-999 = nodata]
NumberArtBarrier_0 to 1 km_down	Number of artificial barriers from downstream end of the site to 1 km downstream [-999 = nodata]
NumberBarrFishPass_0 to 1 km_down	Number of fish passable barriers from downstream end of the site to 1 km downstream [-999 = nodata]
Barriers_1 to 5 km_down_YN	Presence of all barriers from 1 to 5 km downstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrier_1 to 5 km_down	Number of all barriers from 1 to 5 km downstream [-999 = nodata]
NumberArtBarrier_1 to 5 km_down	Number of artificial barriers from 1 to 5 km downstream [-999 = nodata]
NumberBarrFishPass_1 to 5 km_down	Number of fish passable barriers from 1 to 5 km downstream [-999 = nodata]
Barriers_5 to 30 km_down_YN	Presence of all barriers from 5 to 30 km downstream [1 = yes, 2 = no, 0 = nodata]
NumberBarrier_5 to 30 km_down	Number of all barriers from 5 to 30 km downstream [-999 = nodata]

Table continued

Parameter	Description
NumberArtBarrier_5 to 30 km_down	Number of artificial barriers from 5 to 30 km downstream [-999 = nodata]
NumberBarrFishPass_5 to 30 km_down	Number of fish passable barriers from 5 to 30 km downstream [-999 = nodata]
DistNextBarrUp	Distance to next impassable barrier upstream -> Startpoint = upstream end of case study site [km] [-999 = nodata]
DistNextBarrDown	Distance to next impassable barrier downstream -> Startpoint = downstream end of case study site [km] [-999 = nodata]
ReporterID_preshymo	name and organisation of the person who obtained the data
DataSourceID_preshymo	database name, report, etc. of the obtained data
Comment_preshymo	Any other comment
other pressures_YN	If there are other pressures, which are not named
other pressures_perc	If there are other pressures, which are not named

Table 12-15 Variables and descriptions of the database sheet "Pressure point / diffuse sources"

Parameter	Description
StationCode	Station code; unique internal or national code (land code_river name_casestudy type -> eg.: FI_Kuiv_R1, AT_Drau_D1, etc.)
sewage_plants_YN	water bodies in the catchment upstream affected by municipal sewage treatment plants [1 = yes, 2 = no, 0 = nodata]
sewage plants_perc	Percentage of water courses affected by municipal sewage treatment plants in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
comb_sewers_YN	Water bodies in the catchment upstream affected by combined sewers (Sewers carrying both sewage and stormwater together) [1 = yes, 2 = no, 0 = nodata]
comb_sewers_perc	Percentage of water courses affected by combined sewers (Sewers carrying both sewage and stormwater together) in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
industrial_YN	water bodies in the catchment upstream affected by industrial facilities (including manufacturing, oil and gas extraction, and service industries) [1 = yes, 2 = no, 0 = nodata]
industrial_perc	Percentage of water courses affected by industrial facilities (including manufacturing, oil and gas extraction, and service industries) in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
mining_YN	water bodies in the catchment upstream affected by priority substances [1 = yes, 2 = no, 0 = nodata]
mining_perc	Percentage of water courses affected by priority substances in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
thermal_poll_YN	water bodies in the catchment upstream affected by priority substances [1 = yes, 2 = no, 0 = nodata]
thermal_poll_perc	Percentage of water courses affected by priority substances in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
Prior_subst_YN	water bodies in the catchment upstream affected by priority substances [1 = yes, 2 = no, 0 = nodata]
Prior_subst_per	Percentage of water courses affected by priority substances in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]

Table continued

Parameter	Description
agricult_YN	water bodies in the catchment upstream affected by nutrients, pesticides, herbicides, fertilizers, animal wastes [1 = yes, 2 = no, 0 = nodata]
agricult_perc	Percentage of water courses affected by nutrients, pesticides, herbicides, fertilizers, animal wastes in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
tot_pres_point_YN	Point sources present in the catchment upstream (like industrial facilities, mining,sewage plants) [1 = yes, 2 = no, 0 = nodata]
tot_pres_point_perc	Percentage of water courses affected by point sources in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
tot_pres_difus_YN	Diffuse sources present in the catchment upstream (like sediments, nutrients, pesticides, herbicides, fertilizers,animal wastes) [1 = yes, 2 = no, 0 = nodata]
tot_pres_difus_perc	Percentage of water courses affected by diffuse sources in catchment upstream [%] Total Length of water courses in catchment upstream = 100% [-999 = nodata]
tot_pres_sourc_YN	Point and/or diffuse sources present in the catchment upstream [1 = yes, 2 = no, 0 = nodata]
tot_pres_sourc_perc	Percentage of water courses affected by point and /or diffuse sources in catchment upstream [%] - each water course ist only counted once - Total Length of water courses in catchment upstream = 100% (value must not exceed 100%) [-999 = nodata]
ReporterID_pres_sourc	name and organisation of the person who obtained the data
DataSourceID_pres_sourc	database name, report, etc. of the obtained data
Comment_pres_sourc	Any other comment
other pressures_YN	If there are other pressures in the catchment upstream, which are not named
other pressures_perc	If there are other pressures in the catchment upstream, which are not named

Table 12-16 Variables and descriptions of the database sheet "Physico-chemical data"

Parameter	Description
StationCode	Station code; unique internal or national code (land code_river name_casestudy type -> eg.: FI_Kuiv_R1, AT_Drau_D1, etc.)
pH_data_year	Year of pH data [yyyy; -999 = nodata]
pH_site_mean	PH 0-14; mean value at case study site [-999 = nodata]
pH_site_min	PH 0-14; minimum value at case study site [-999 = nodata]
pH_site_max	PH 0-14; maximum value at case study site [-999 = nodata]
pH_mean_5_up	PH 0-14; mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_pH	database name, report, etc. of the obtained data
Conduc_data_year	Year of Conductivity data [yyyy; -999 = nodata]
Conduc_site_mean	Electrical conductivity, mean value [microS/cm]; [-999 = nodata]
Conduc_site_min	Electrical conductivity, [microS/cm] minimum value at case study site [-999 = nodata]
Conduc_site_max	Electrical conductivity, [microS/cm] maximum value at case study site [-999 = nodata]

Table continued

Parameter	Description
Conduc_mean_5_up	Electrical conductivity, [microS/cm]; mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_conductivity	database name, report, etc. of the obtained data
TN_data_year	Year of total Nitrogen data [yyyy; -999 = nodata]
TN_site_mean	Total Nitrogen [mg/l] mean value at case study site [-999 = nodata]
TN_site_min	Total Nitrogen [mg/l] minimum value at case study site [-999 = nodata]
TN_site_max	Total Nitrogen [mg/l] maximum value at case study site [-999 = nodata]
TN_mean_5_up	Total Nitrogen [mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_TN	database name, report, etc. of the obtained data
TOC_data_year	Year of total organic carbon data [yyyy; -999 = nodata]
TOC_site_mean	Total organic carbon [mg/l] mean value at case study site [-999 = nodata]
TOC_site_min	Total organic carbon [mg/l] minimum value at case study site [-999 = nodata]
TOC_site_max	Total organic carbon [mg/l] maximum value at case study site [-999 = nodata]
TOC_mean_5_up	Total organic carbon [mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_TOC	database name, report, etc. of the obtained data
BOD5_data_year	Year of biological oxygen demand data [yyyy; -999 = nodata]
BOD5_site_mean	Biological oxygen demand [mg/l] mean value at case study site [-999 = nodata]
BOD5_site_min	Biological oxygen demand [mg/l] minimum value at case study site [-999 = nodata]
BOD5_site_max	Biological oxygen demand [mg/l] maximum value at case study site [-999 = nodata]
BOD5_mean_5_up	Biological oxygen demand [mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_BOD5	database name, report, etc. of the obtained data
NO2_data_year	Year of Nitrite data [yyyy; -999 = nodata]
NO2_site_mean	Nitrite [mg/l] mean value at case study site [-999 = nodata]
NO2_site_min	Nitrite [mg/l] minimum value at case study site [-999 = nodata]
NO2_site_max	Nitrite [mg/l] maximum value at case study site [-999 = nodata]
NO2_mean_5_up	Nitrite [mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_NO2	database name, report, etc. of the obtained data
NO3_data_year	Year of Nitrate data [yyyy; -999 = nodata]
NO3_site_mean	Nitrate [mg/l] mean value at case study site [-999 = nodata]
NO3_site_min	Nitrate [mg/l] minimum value at case study site [-999 = nodata]
NO3_site_max	Nitrate [mg/l] maximum value at case study site [-999 = nodata]
NO3_mean_5_up	Nitrate [mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_NO3	database name, report, etc. of the obtained data
NH3_data_year	Year of Ammonium data [yyyy; -999 = nodata]
NH3_site_mean	Ammonia [mg/l] mean value at case study site [-999 = nodata]

Table continued

Parameter	Description
NH3_site_min	Ammonia [mg/l] minimum value at case study site [-999 = nodata]
NH3_site_max	Ammonia [mg/l] maximum value at case study site [-999 = nodata]
NH3_mean_5_up	Ammonia [mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_NH3	database name, report, etc. of the obtained data
NH4_data_year	Year of Ammonium (NH4+) data [yyyy; -999 = nodata]
NH4_site_mean	Ammonium [mg/l] mean value at case study site [-999 = nodata]
NH4_site_min	Ammonium [mg/l] minimum value at case study site [-999 = nodata]
NH4_site_max	Ammonium [mg/l] maximum value at case study site [-999 = nodata]
NH4_mean_5_up	Ammonium[mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_NH4	database name, report, etc. of the obtained data
Cl_data_year	Year of chloride data [yyyy; -999 = nodata]
Cl_site_mean	Chloride [mg/l] mean value at case study site [-999 = nodata]
Cl_site_min	Chloride [mg/l] minimum value at case study site [-999 = nodata]
Cl_site_max	Chloride [mg/l] maximum value at case study site [-999 = nodata]
Cl_mean_5_up	Chloride [mg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_Cl	database name, report, etc. of the obtained data
OrPh_data_year	Year of Ortho-phosphate data [yyyy; -999 = nodata]
OrPh_site_mean	Ortho-phosphate [microg/l] mean value at case study site [-999 = nodata]
OrPh_site_min	Ortho-phosphate [microg/l] minimum value at case study site [-999 = nodata]
OrPh_site_max	Ortho-phosphate [microg/l] maximum value at case study site [-999 = nodata]
OrPh_mean_5_up	Ortho-phosphate [microg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site [-999 = nodata]
DataSource__PCD_OrPh	database name, report, etc. of the obtained data
PO4_data_year	Year of total-phosphate data [yyyy; -999 = nodata]
PO4_site_mean	Total-phosphate [microg/l] mean value at case study site [-999 = nodata]
PO4_site_min	Total-phosphate [microg/l] minimum value at case study site [-999 = nodata]
PO4_site_max	Total-phosphate [microg/l] maximum value at case study site [-999 = nodata]
PO4_mean_5_up	Total-phosphate [microg/l] mean value for 5km catchment upstream -> Startpoint = upstream end of case study site
DataSource__PCD_PO4	database name, report, etc. of the obtained data
ReporterID_PCD	name and organisation of the person who obtained the data
Comment_PCD	Any other comment

Table 12-17 Variables and descriptions of the database sheet "Additional Parameters"

Parameter	Description
StationCode	Station code; unique internal or national code (land code_river name_casestudy type -> eg.: FI_Kuiv_R1, AT_Drau_D1, etc.)
GDP_data_year	Year of GDP data [yyyy; -999 = nodata]
GDP	Gross domestic product - mean value of all municipalities in catchment upstream [-999 = nodata]
DataSource__GDP	database name, report, etc. of the obtained data
pop_den_data_year	Year of population density data [yyyy; -999 = nodata]
popul_density	inhabitants per km^2 - in catchment upstream [yyyy; -999 = nodata]
DataSource__pop_den	database name, report, etc. of the obtained data
ReporterID_addIn	name and organisation of the person who obtained the data
Comment_addIn	Any other comment

12.2 *Annex B: Description of restoration projects and study sections*

Large restoration project Drava (AT_R1)


Figure 12-1 Large restoration project AT_R1 (left: Amt der Kärntner LR, Abt.16L; S. Tichy, right: BOKU, IHG)

The large restoration project in Austria AT_R1 is situated at the river Drava in the western part of the federal province Carinthia near the village Kleblach. At this point the upper catchment measures about 2433 km². The mean discharge near the site is 62.6 m³/s. In this section the Drava is a 7th order stream and is assigned to the fish region Hyporhithral.

In the years from 2002 to 2003 several restoration measures were implemented over a total length of 1.9 km. On a length of 1.3 km bank fixation was removed and the river bed was widened up to 45 m in several sections. A secondary channel was created with a length of 500 m and a width of 30 m. These measures aimed at stabilizing the river bed and the groundwater level, the creation of gravel banks and the increase of in-stream and bank structures. One of the former side arms was reconnected to the river for annual flooding. This reconnection of floodplain water bodies with the main channel was intended to prevent aggradation processes and provide habitats for juvenile fish and stagnophil fish species. Additionally agricultural land was purchased for the establishment of new floodplain forests (IHG, 2008; Mandler, 2004).

Small restoration project Enns (AT_R2)

Figure 12-2 Small restoration project AT_R2 (BOKU, IHG)

The small restoration project in Austria AT_R2 is situated at the river Enns in the federal province Styria near the village Aich. At this point the upper catchment measures about 809 km². The mean discharge near the site is 21.5 m³/s. In this section the Enns is a 5th order stream and is assigned to the fish region Metarhithral.

In the years from 2003 to 2004 due to efforts to reduce the flood risk for the village Aich protection and restoration measures were implemented. For the protection of the village Aich on the left side of the river an 800 m long flood protection dam was built. In the upper section the bank fixation was removed on the left side and the river bed was widened on a length of 80 m. A new 170 m long secondary channel was created and the new cut bank was shaped with biological engineering measures. The estuary of a former side channel of the Enns was transferred upstream into the new side arm. The barriers for fish migration in this former side channel were removed. Downstream of the bridge the river bed was widened on the right bank. Gravel bars were shaped roughly and dead wood structures were initiated (IHG, 2008; Mandler, 2004).

Large restoration project Thur (CH_R1)

Figure 12-3 Large restoration project CH_R1. Upper left: Gravel bar on the left side of the river, and presence of wood within the main river channel (A. Paillex, Eawag), upper right: part of the restored site with the main river channel on the left of the photo, stagnant water body and alluvial forest on the right (A. Paillex Eawag), lower: view from the middle part of the restored site toward downstream (H. Mottaz, Eawag).

The large restoration project in Switzerland CH_R1 is situated at the river Thur in the north east of Switzerland near the villages Niederneunforn and Altikon. At this point the upper catchment measures about 1605 km². The mean discharge near the site is 52.9 m³/s. In this section the Thur is a 7th order stream and is assigned to the fish region Epipotamal.

The restored reach is 1.55 km in length and was restored in 2002. The river was widened on one side of the main river channel. Embankments along the right side of the river were removed to provide a larger space to the river. Additional artificial structures were added to enhance the ability of the river to braid. Both are expected to increase diversity of instream habitats and corresponding biota. Restoration efforts recreated patterns of erosion and deposition, as well as large gravel bars along the main river channel. Restoration is expected to enhance the terrestrial biodiversity living along and on the re-created gravel bars, and a higher frequency of interaction between the river and the old-disconnected floodplain is expected to happen.

Small restoration project Töss (CH_R2)

Figure 12-4 Small restoration project CH_R2. Upper right: view from downstream toward the upper part of the restored site (Eawag, P. Reichert), upper left: view from the upper part toward the lower part of the restored site (Eawag, P. Reichert), lower right: detail of artificial structure to force the river to create islands (Eawag, P. Reichert)

The small restoration project in Switzerland CH_R2 is situated at the river Töss in the north east of Switzerland. At this point the upper catchment measures about 188 km². The mean discharge near the site is 9.9 m³/s. In this section the Töss is a 6^{th} order stream and is assigned to the fish region Metarhithral.

The 200 m long site was restored in 1999. Before restoration, the river was fully embanked and was a straight canal. During restoration, the river was widened on both sides of the main river channel. Along the course of the river, embankments were removed to provide a large space to the river (Figure 12-4). Additional wood structures and blocks of stone were added to enhance the ability of the river to recreate islands (Figure 12-4). Restoration efforts are expected to increase diversity of instream habitats and corresponding biota. In parallel, recreating gravel bars and islands is expected to enhance the terrestrial biodiversity living along fluvial corridors.

Large restoration project Becva (CZ_R1)

Figure 12-5 Large restoration project CZ_R1 (Karel Brabec)

The large restoration project in the Czech Republic CZ_R1 is situated at the Becva River (Danube water basin) near the village Osek nad Becvou. At this point the upper catchment measures about 1532 km^2. The mean discharge near the site is 16.6 m^3/s. In this section the Becva River is a 7^{th} order stream and is assigned to the fish region Epipotamal.

The large restoration project is one of five river stretches which were passively renaturalized by floods in 1997. In comparison with still regulated channels the 450 m long restored section is characterized by intensive erosional and depositional processes. Wider channel and intermittent occurrence of large woody debris in channel contribute to the development of hydromorphological features characteristic of braiding channel. Higher heterogeneity of river habitats was documented in terms of water chemistry, water temperature, substrate and aquatic biota.

Small restoration project Morava (CZ_R2)

Figure 12-6 Small restoration project CZ_R2 (Karel Brabec)

The small restoration project in the Czech Republic CZ_R2 is situated at the river Morava (Danube water basin). At this point the upper catchment measures about 2305 km². The mean discharge near the site is 17.7 m³/s. In this section the Morava is a 7th order stream and is assigned to the fish region Epipotamal.

The small restoration project is a relatively short river stretch (220 m) where the bank protection was removed by floods in 1997. In comparison with the still regulated channel, the restored section is characterized by higher diversity of bank habitats, by ocurrence of gravel bars and side pools. The upstream river segment is characterized by hydromorphologicaly valuable structures (meandering channel connected with floodplain and containing large woody debris).

Large restoration project Ruhr (DM_R1)

Figure 12-7 Large restoration project DM_R1 (UDE)

The large restoration project in the German mountain area DM_R1 is situated at the river Ruhr in the Federal State of Northrhine-Westfalia in the urban area of the city Arnsberg. At this point the upper catchment measures about 1054 km². The mean discharge near the site is 15.2 m³/s. In this section the Ruhr is a 3rd order stream and is assigned to the fish region lower grayling.

In 2008 a reach 750 m in length was restored. The main aims of the restoration measures were to restore more natural hydromorphological conditions and to re-establish longitudinal connectivity. Moreover, it aimed to increase the aesthetic value of the river section and to raise people's awareness of the importance of biodiversity by making nature tangible. The river bed was widened and the bank fixations were removed to initiate lateral erosion. Two secondary channels were created and sediment and large wood were added to enhance the instream structures of the site.

Small restoration project Lahn (DM_R2)

Figure 12-8 Small restoration project DM_R2 (UDE)

The small restoration project in the German mountain area DM_R2 is situated at the river Lahn in the Federal State of Hesse. At this point the upper catchment comprises about 652 km². The mean discharge near the site is 12 m³/s. In this section the Lahn is a 3^{rd} order stream and is assigned to the fish region lower grayling.

In 2000 measures were implemented on a river length of 240 m. The main aim was a morphological improvement as the river course was straightened and natural instream habitats were largely missing. To initiate bankside erosion bank fixations were removed and a side arm was created. The river bed and banks were restructured to enhance habitat and biotic diversity. Placement of large wood was carried out to improve instream structures at the site.

Large restoration project Vääräjoki (FI_R1)

Figure 12-9 Large restoration project FI_R1 (left: Jukka Aroviita, right: Jaana Rääpysjärvi)

The Finnish large restoration project FI_R1 is situated at the river Vääräjoki. At this point the upper catchment measures about 835 km². The mean discharge at the site is 9.9 m³/s. In this section the Vääräjoki is a 4^{th} order stream and is assigned to the fish region brown trout-European bullhead.

In the timeframe from 1997 to 2006 all the rapids in section from 13 km to 29 km of the river mouth have been restored. One of the rapids is situated within the 1.4 km long large restored section FI_R1. The stream bottom was rearranged using boulders that had originally been removed from the channel during channelization and placed along stream margins Also gravel beds were created to provide nursery habitat for salmonids. The aim of the restoration was to return the heavily modified river closer to natural hydrological and morphological state and especially enable the breeding and migration of fish (Aronen, 1996; HERTTA).

Small restoration project Kuivajoki (FI_R2)

Figure 12-10 Small restoration project FI_R2 (Jaana Rääpysjärvi)

The Finnish small restoration project FI_R2 is situated at the river Kuivajoki. At this point the upper catchment measures about 976 km². The mean discharge at the site is 12.8 m³/s. In this section the Kuvajoki is a 4th order stream and is assigned to the fish region salmon-European bullhead.

In Kuivajoki, altogether about 5 km of the river (consisting of multiple riffle sections in the river) were restored in early 2000s. The stream bottom was rearranged using boulders were removed from the channel and placed along stream margins during channelization. Also gravel beds were created to provide nursery habitat for salmonids. The small restoration section FI_R2, called Hirvaskoski, at River Kuivajoki is 400 m long. Most of the boulders that were removed from the river during channelization were placed back in early 2000s to create more heterogeneous habitat for the stream biota (Aronen, 1996; HERTTA).

Large restoration project Emån (SE_R1)

Figure 12-11 Large restoration project SE_R1. Aerial overview of the site of dam removal (left) and on-ground view of the restoration project upstream of dam removal (right) (SLU 2013)

The Swedish large restoration project SE_R1 is situated near the old mill town of Emsfors in the River Emån in the south-east of Sweden. At this location the upper catchment measures about 4440 km². The mean discharge near the site is 29.3 m³/s. In this section the River Emån is a stream of 6^{th} order and is assigned to the fish region hyporhithral.

Restoration started in 2006 by permanently opening the dam lids of a former hydropower dam with the aim to restore longitudinal connectivity. In the same year, riffles damaged by timber floating located upstream the dam were restored. This was done to compensate for the drop in waterlevel upstream of the dam after dam removal, and thus to protect important floodplain habitats in this area, but also to improve the habitat for salmonid fish. The riffles were restored by boulder and salmonid spawning gravel additions. In total, a 900 m river stretch was restored. In 2010-2011 the hydropower dam and the hydropower station were completely removed and a fishway with low inclination was constructed to further improve conditions for fish migration at the site.

Small restoration project Mörrumsån (SE_R2)

Figure 12-12 Small restoration project SE_R2. Overview (left) and view of the restored section (right) (SLU 2013)

The Swedish small restoration project SE_R2 is situated near the village of Hemsjö in the River Mörrumsån in the south of Sweden. At this site, the upper catchment is ca. 3264 km². At the site of restoration, i.e. the old dry channel, the mean discharge is 12 m³/s, whereas the mean natural discharge in this area of River Mörrumsån is ca 26 m³/s. In this section the Mörrumsån is a stream of 6^{th} order and is assigned to the fish region hyporhithral.

Between 2003 and 2012 several restoration measures have been implemented on a length of 3.3 km to restore longitudinal connectivity and to improve habitat conditions for salmonid fish. To restore longitudinal connectivity, fishways were constructed at the hydropower plants Hemsjö övre and Hemsjö nedre in 2003-2004 and water flow was increased in the dry channel between the hydropower stations. After the initial restoration of longitudal connectivity several habitat improvement projects were carried out in 2004-2006, 2010 and 2012. Spawning gravel was added along the site to improve and create new spawning grounds for salmonid fish.

Large restoration project Skjern (DK_R1)

Figure 12-13 Large restoration project DK_R1. Overview of the restored 40 km reach in River Skjern (left) and closer view on part of the restored reach today 11 years after restoration (right) (Niels Bering Ovesen)

The large restoration project in Denmark DK_R1 is situated at the river Skjern. At this point the upper catchment measures about 1553 km². The mean discharge near the site is 36.6 m³/s. In this section the Skjern is a 5th order stream.

River Skjern is the second largest river in Denmark and drains the western part of the peninsula Jutland. The river was channelized in 1960 and wetlands in the floodplain were drained to improve conditions for agriculture. The river was restored from 1999-2002. This project is the largest single restoration project in Northern Europe. The main aim of the project was to enhance the nutrient retention and biodiversity by restoring the physical and hydrological dynamics of the river and floodplain. The restoration project included re-meandering of the river and re-establishment of the natural water levels and water level fluctuation in the river and its valley with the purpose of enhancing living conditions for plants and animals and safeguarding a high water quality in the river and in the downstream estuary, Ringkøbing Fjord. Specific biological targets included improved habitat conditions for migratory birds, improvements of floodplain and wetland vegetation and increased survival of salmonoid fish.

The restoration work was initiated in June 1999 and was more or less finalized by autumn 2002. The main activities were excavation of about 40 km of new river course, removal of existing dikes from the land reclamation in the 1960s and the filling of the old channelised river reaches. Two pumping stations and a weir established in connection with the river channelisation were also removed. The activities also comprised construction of bridges and paths. Whenever possible, one of the original river banks from before the 1960s formed one of the banks of the restored river.

Small restoration project Storå (DK_R2)

Figure 12-14 Small restoration project DK_R2. Overview of the site restored in River Storå left) and closer view on the gravel bar created in 2011 by addition of coarse material (right) (Niels Bering Ovesen)

The Danish small restoration project DK_R2 is situated in the river Storå At this point the upper catchment measures about 878 km². The mean discharge near the site is 16.1 m³/s. In this section the Skjern is 5th order stream.

River Storå is the 3rd largest river in Denmark and drains the western part of the peninsula Jutland. Some parts of the river were channelized during 1950's, however large reaches of the river have also been left untouched. Smaller in-stream habitat improvements have been conducted over the years, including addition of coarse material for improving salmonid spawning. The main aim of the project was to improve conditions for salmon in the Storeå by creating new spawning areas and additionally for lampreys being embraced by the EU Habitats Directive. Furthermore the project aimed at improving conditions for grayling that is currently declining in Denmark. In autumn in 2011 a total amount of 700 m^3 coarse substrates in the form of gravel, boulders and a few larger stones were added to a 50-60 m long reach in the Storå thereby increasing the area available for spawning for salmonoid fish. The addition of substrates is expected to increase the water level app. 20 cm just upstream of the gravel bar.

Large restoration project Lippe (DL_R1)

Figure 12-15 Large restoration project DL_R1. Upstream view of the widened and shallow cross-section (left) and large wood placement (right) (UDE)

The large restoration project in the German lowlands DL_R1 is situated at the river Lippe. At this point the upper catchment measures about 1896 km². The mean discharge near the site is 17.7 m³/s. In this section the Lippe is a stream of 3rd order and assigned to the fish region barbel.

The 2 km long section was restored in 1996-1997. The bank fixation was removed, sediment was added to the channel bed to elevate it by 2 m to reconnect the river with its former floodplain. Furthermore the channel was widened from 13 to 45 m. Several large trees were placed in the reach to initiate natural channel dynamics and to increase local depth variability. Floodplain land-use was restricted to extensive grazing by primitive Konik ponies and Taurus cattle to allow for natural succession of the floodplain vegetation. The agricultural drainage system was stuffed, except some local ponds. A ramp was built at the downstream end to prevent channel incision (ABU, 2014).

Small restoration project Spree (DL_R2)

Figure 12-16 Small restoration project DL_R2. Overview (upper left, Landesumweltamt Brandenburg, aerialimagery 030.05.2009)

The small restoration project in the German lowlands DL_R2 is situated at the river Spree. At this point the upper catchment measures about 6275 km². The mean discharge near the site is 14 m³/s. In this section the Spree is a 6th order stream and assigned to the fish region Metapotamal.

The 950 m long restoration site was a former oxbow which was reconnected on both sides of the main channel. The former main channel was blocked by a gravel dam to redirect all flow through the new meander. The remaining old main stem stretch serves as new flow protected habitats and their depth and width variability slightly improved by alternating sand bars. The main aims of the restoration were to restore the natural hydrology, morphology and oxygen balance of the river; to improve water retention in the landscape and the development of habitats to improve benthic and rheophil species (Köhler et al., 2002).

Large restoration project Regge (NL_R1)

Figure 12-17 Large restoration project NL_R1. Overview of the section (left, Waterschap Vechstromen) and on-ground view (right, Piet Verdonschot)

The Dutch large restoration project NL_R1 is situated at the river Regge. At this point the upper catchment measures about 339 km². The mean discharge near the site is 4.2 m³/s. In this section the Regge is a 4th order stream and assigned to the fish region of the bream zone.

The section was restored in 2005-2006 over a length of 1.4 km. Two old meanders were excavated (based on topographical maps from 1900) and connected to the channelized riverbed. Subsequently, the latter was dammed, only acting as a bypass during peak discharges. The new meandering channel is less wide and shallower in comparison to the former main channel, improving instream conditions for biota through an increased current velocity. Furthermore, land use of the floodplain was changed from agriculture to nature, embankments were lowered and an underwater weir has been built to prevent bed erosion (Waterschap Regge en Dinkel, 2005).

Small restoration project Dommel (NL_R2)

Figure 12-18 Small restoration project NL_R2 Dommel (Piet Verdonschot)

The Dutch small restoration project NL_R2 is situated at the river Dommel. At this point the upper catchment measures about 399 km². The mean discharge near the site is 2.6 m³/s. In this section the Dommel is a stream of 4th order and assigned to the fish region of the bream zone.

The section was restored in 2007 over a length of 0.9 km. To create more habitat heterogeneity, two secondary channels were dug and the streambed was modified, resulting in more gently sloping banks. Other measures were the excavation of pools in the floodplain and the construction of a fishway (Waterschap de Dommel, 2007).

Large restoration project Narew (PL_R1)

Figure 12-19 Large restoration project PL_R1. Overview (upper left, A. Bielenko, branches of the anastomosing river near Panki Village (upper right, WULS-SGGW) and near Rzedziany Village (lower left, WULS-SGGW)

The Polish large restoration project PL_R1 is situated at the river Narew downstream the Narew national Park. At this point the upper catchment measures about 3680 km². The mean discharge near the site is 16.9 m³/s. In this section the Narew is a 2nd order stream and assigned to the fish region bream.

In 1995 it was decided to restore the degraded section adjacent to the National Park. On a length of 9 km several restoration measures were implemented with the objectives to bring back a natural value of the river valley and to restore the naturally anastomosing river network. Underwater weir structures, functioning as thresholds, were built to raise water level and as consequence flooding old side arms and slowing down the water outflow form the area. Additionally old side channels were re-connected by removing excess sediment and vegetation (PTOP, 2012; Winiecki and Krupa, 2006; Winiecki et al. 2009).

Small restoration project Warta (PL_R2)

Figure 12-20 Small restoration project PL_R2 (WULS-SGGW)

The Polish small restoration project PL_R2 is situated at the river Warta between Zagórów and the village Lad. At this point the upper catchment measures about 14519 km². The mean discharge near the site is 45.3 m³/s. In this section the Warta is a 2nd order stream and assigned to the fish region bream.

The restoration actions were undertaken as a compensation project for losses caused by constructing a highway that damaged the other stretch of the river belonging to the Natura 2000 area. The restoration actions were performed in the years 2006-2008 on a length of 3 km. The aim of the restoration was the improvement of lateral connectivity between the main river channel and floodplain. The main implemented measures are related to re-connecting the river with oxbows and floodplain by building culverts, lowering the embankments at some points and clearing the old connections (PTOP, 2012; Winiecki and Krupa, 2006; Winiecki et al. 2009).

Table 12-18 Characteristics of the large restoration projects (R1)

Site name	AT_R1	CH_R1	CZ_R1	DM_R1	FI_R1	SE_R1	DK_R1	DL_R1	NL_R1	PL_R1_1
Country	Austria	Switzerland	Czech Republic	Germany	Finland	Sweden	Denmark	Germany	Netherlands	Poland
River name	Drau	Thur	Becva	Ruhr	Vääräjoki	Emån	Skjern	Lippe	Regge	Narew
River type	Gravel-bed	Gravel-bed	Gravel-bed	Gravel-bed	Gravel-bed	Gravel-bed	Sand-bed	Sand-bed	Sand-bed	Sand-bed
Latitude (N)	46.75454	47.5918	49.4968975	51.44093	64.054433	57.149095	55.9380926	51.663675	52.4384	53,1500527
Longitude (E)	13.309393	8.77114	17.5211533	7.96223	24.2206639	16.441897	8.6279814	8.23248	6.4417	22,8716193
Altitude (m a.s.l.)	570	371	232	153	60	10	10	72	6	139
Catchment geology	siliceous	calcareous	siliceous	siliceous	organic	siliceous	siliceous	siliceous	siliceous	organic
Corine Land Cover (%)										
artificial surfaces	2	8	6	7	1	2	3	9	13	2
agricultural areas	8	59	43	36	15	13	75	65	70	62
forest and seminatural areas	90	33	51	57	75	80	20	26	17	34
wetlands	0	0	0	0	8	1	2	0	0	2
water bodies	0	0	0	0	1	5	0	0	0	0
Catchment size (km²)	2433	1605	1532	1054	835	4440	1553	1896	339	3680
Mean discharge (m³/s)	62.6	52.9	16.6	15.2	9.9	29.3	36.6	17.7	4.2	16,9
Stream order	7	7	7	3	4	6	5	3	4	2
Ecoregion	Alps	Alps	Hungarian lowlands	Central Highlands	Fenno-scandian shield	Fenno-scandian shield	Central plains	Central plains	Western plains	Eastern plains
Restoration Length (km)	1.9	1.55	0.45	0.75	1.4	0.9	26	2	1.4	9
Local channel slope (%)	0.34	0.09	0.2	0.08	0.13	0.24	0.2	0.03	0.005	0,06
Restoration date	2002-2003	2002	1997	2008	1997-2006	2006-2011	2003	1997	2005-2006	1995-cont.
Main measures	riverbed widening; (partial removal of bank fixation; initiation of secondary channel; reconnection of one sidearm)	riverbed widening; (enhancement of flood protection and biota diversity, removal of embankments)	riverbed widening	riverbed widening	instream measures	Hydro RivCon (dam removal, naturalise flow regime, fishway constr, salmonid spawning gravel and boulder additions)	re-meandering and reconnection of wetlands	re-meandering	re-meandering and reconnection	reconnection side channels (rise water level by thresholds)

Table 12-19 Characteristics of the small restoration projects (R2)

Site name	AT_R2	CH_R2	CZ_R2	DM_R2	FI_R2	SE_R2	DK_R2	DL_R2	NL_R2_2	PL_R2
Country	Austria	Switzerland	Czech Republic	Germany	Finland	Sweden	Denmark	Germany	Netherlands	Poland
River name	Enns	Töss	Morava	Lahn	Kuivajoki	Mörrumsån	Stora	Spree	Dommel	Warta
River type	Gravel-bed	Gravel-bed	Gravel-bed	Gravel-bed	Gravel-bed	Gravel-bed	Sand-bed	Sand-bed	Sand-bed	Sand-bed
Latitude (N)	47.42112	47.46338	49.6570728	50.86588	65.6860429	56.336005	56.3614934	52.377747	51.4103	52,1930314
Longitude (E)	13.816094	8.72825	17.2179975	8.79088	25.6349874	14.700237	8.4982852	13.878897	5.4375	17,8974616
Altitude (m a.s.l.)	692	453	218	191	74	87	10	35	18	75
Catchment geology	calcareous/ siliceous	calcareous	siliceous	siliceous	organic	siliceous	siliceous	siliceous	siliceous	calcareous
Corine Land Cover (%)										
artificial surfaces	4	4	5	6	0	2	7	7	17	6
agricultural areas	12	36	50	40	1	12	80	49	57	69
forest and seminatural areas	84	59	45	54	67	73	11	41	24	25
wetlands	0	0	0	0	29	1	1	0	1	0
water bodies	0	0	0	0	3	13	0	3	0	1
Catchment size (km²)	809	188	2305	652	976	3264	878	6275	399	14519
Mean discharge (m³/s)	21.5	9.9	17.7	12	12.8	12	16.1	14	2.6	45,3
Stream order	5	6	7	3	4	6	5	6	4	2
Ecoregion	Alps	Alps	Hungarian lowlands	Central Highlands	Fenno-scandian shield	Fenno-scandian shield	Central plains	Central plains	Central plains	Central plains
Restoration Length (km)	0.6	0.21	0.22	0.24	0.4	3.3	0.3	0.95	0.9	3
Local channel slope (%)	0.46	0.52	0.15	0.02	0.26	0.8	0.2	0.015	0.015	0,08
Restoration date	2003-2004	a) 1999 b) 2010	1997	2000	2002-2006	2003-2012	2012	2005	2007	2008
Main measures	riverbed widening (partial removal of bank fixation; initiation of one secondary channel)	riverbed widening (enhance biota diversity, remove embankments)	riverbed widening	riverbed widening	instream measures	Hydro RivCon (increased flow, fishway construction and salmonid spawning gravel additions)	instream measures (habitat restoration: salmonid spawning gravel)	remeandering	excavation of secondary channels, streambed modifications	reconnection floodplain

12.3 Annex C: Hydromorphological effects - detailed results

Results of the hydromorphological survey

The comparison of the effect sizes of the survey parameter "mean hymo survey evaluation" (Mean_hymo), which represents the mean of 14 single parameters, reveals first differences between the restoration projects. 19 of the 20 restored sections showed a positive restoration effect; the one exception was a small restored section at the river Dommel (NL_R2). All large restored sections (R1) showed significant differences to the degraded ones; at small restoration extent (R2) only half of the sections differed significantly.

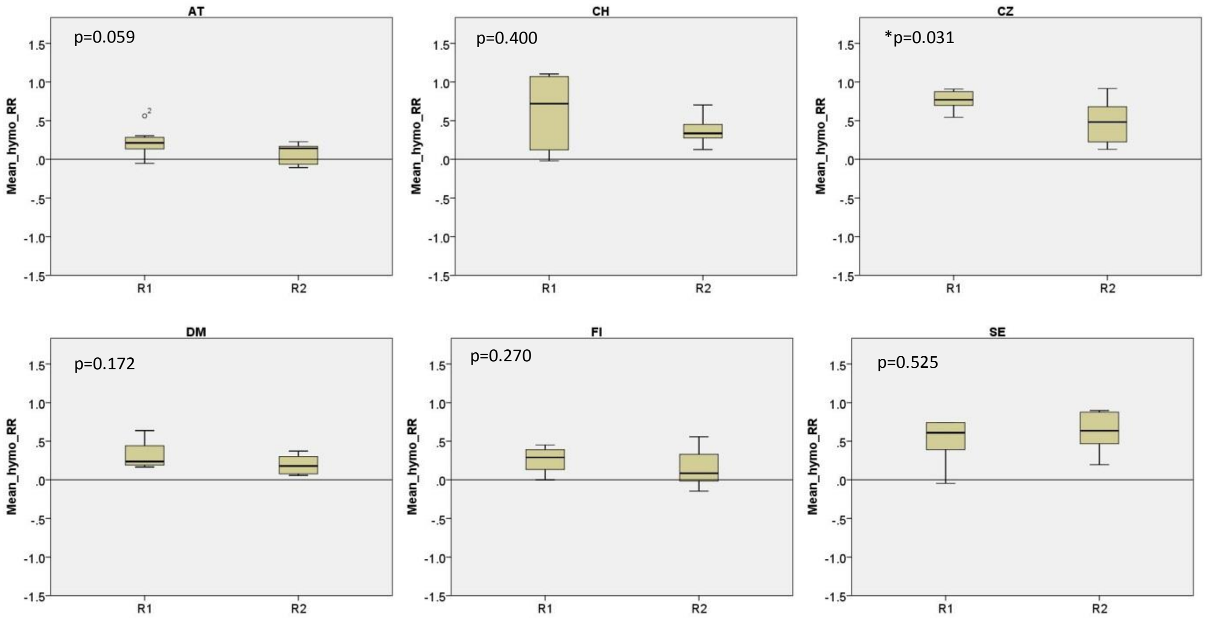


Figure 12-21 Restoration effect of "mean hymo survey evaluation" (Mean_hymo_RR) for gravel-bed rivers (6) differentiated by restoration extent (R1/R2) per country. Mann-Whitney U-Tests R1 vs. R2, p-values were added to the plots (*p < 0.05 significant).

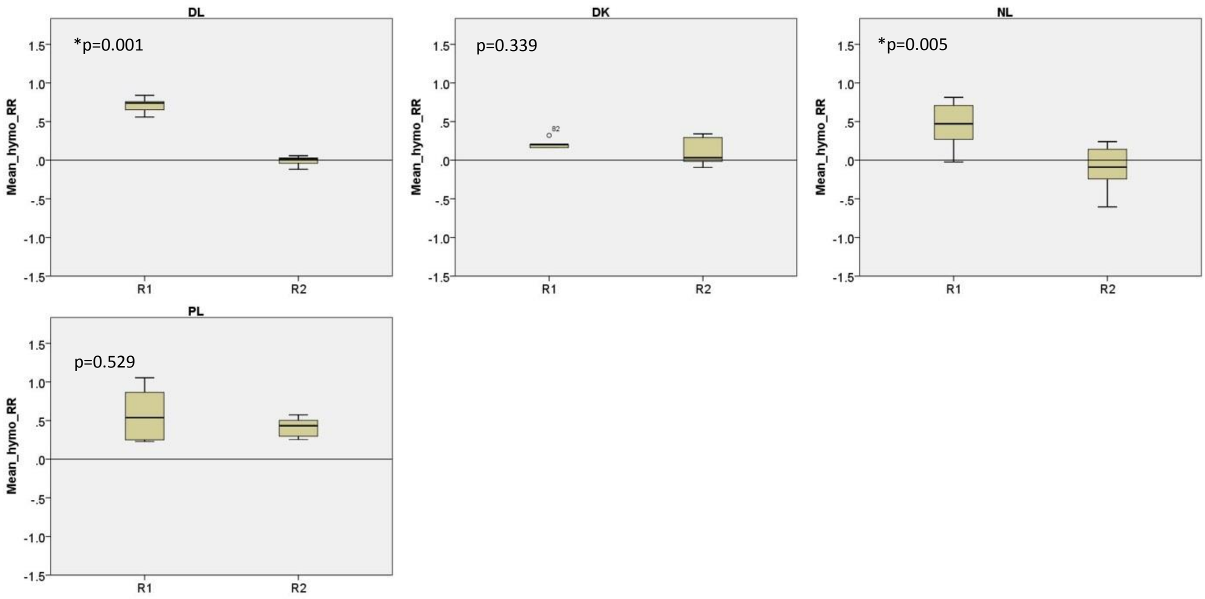


Figure 12-22 Restoration effect of "mean hymo survey evaluation" (Mean_hymo_RR) for sand-bed rivers (4) differentiated by restoration extent (R1/R2) per country. Mann-Whitney U-Tests R1-R2, p-values were added to the plots (*p < 0.05 significant).

Mesohabitat results of case study sites

The change in the "number of natural channel features" along transects (NMchanfeat_nat) within the 20 paired restoration projects is illustrated in Figure 12-23. In half of the restoration projects (AT, CH, CZ, DM, SE) positive restoration effects were evident for both sections with large (R1) and small (R2) restoration. No difference or even lower values were detected in DK, FI, PL, NL for either large or small restoration, in DL at small restorated sections. Figure 12-24 visualizes the variation of "number of natural substrate types" along transects (NMsubstr_nat). In many cases an increase of natural substrate types could be proven, except for DK, NL, FI, PL at large restored sections and for DL, NL, PL for small restoration.

Overall, many restored sections show significantly positive restoration effects for NMchanfeat_nat "number of channel features" (11 out of 20 cases) and NMsubstr_nat "number of natural dominant substrate" (9) (pairwise comparisons R1/D1 – R2/D2 Mann-Whitney U-test, $p<0.05$ – see Table 12-20 This indicates an increase in habitat diversity at the mesohabitat level within the river channel and investigated floodplain area. Within both parameters, no significantly negative effect was found.

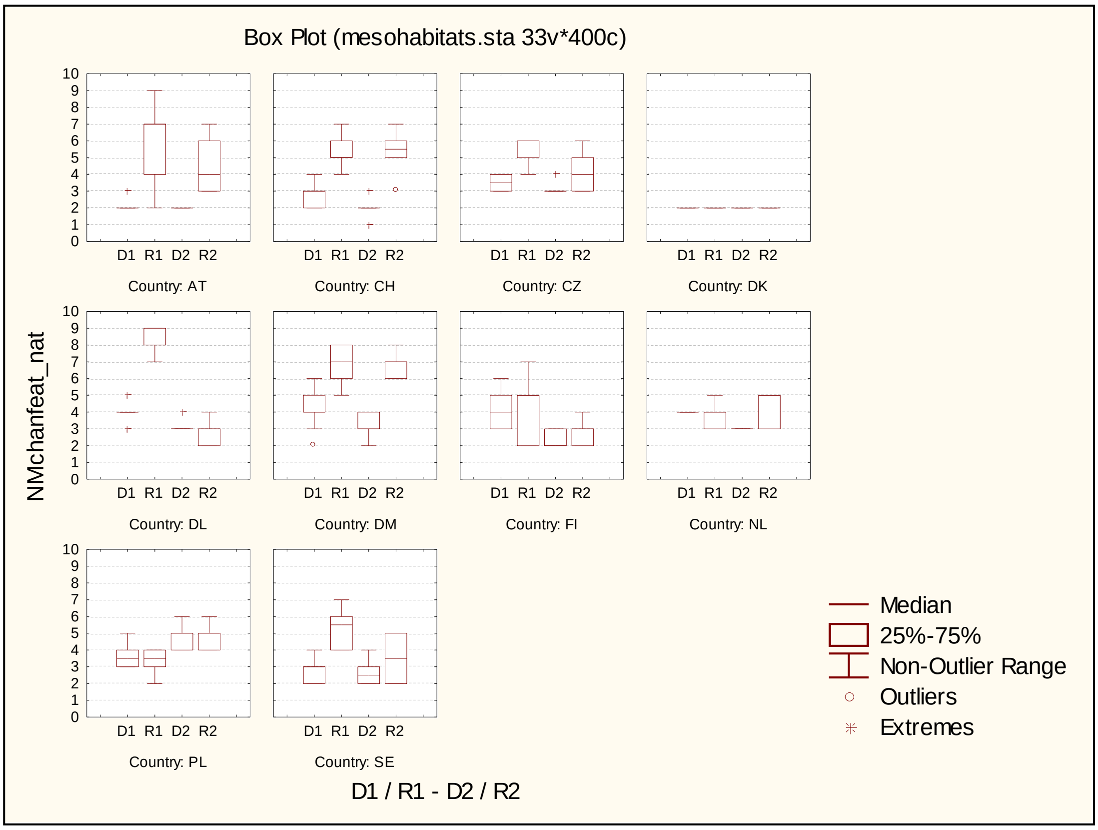


Figure 12-23 Variation of "number of natural channel features" (NMchanfeat_nat) of D1/R1/D2/R2 per country (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration)

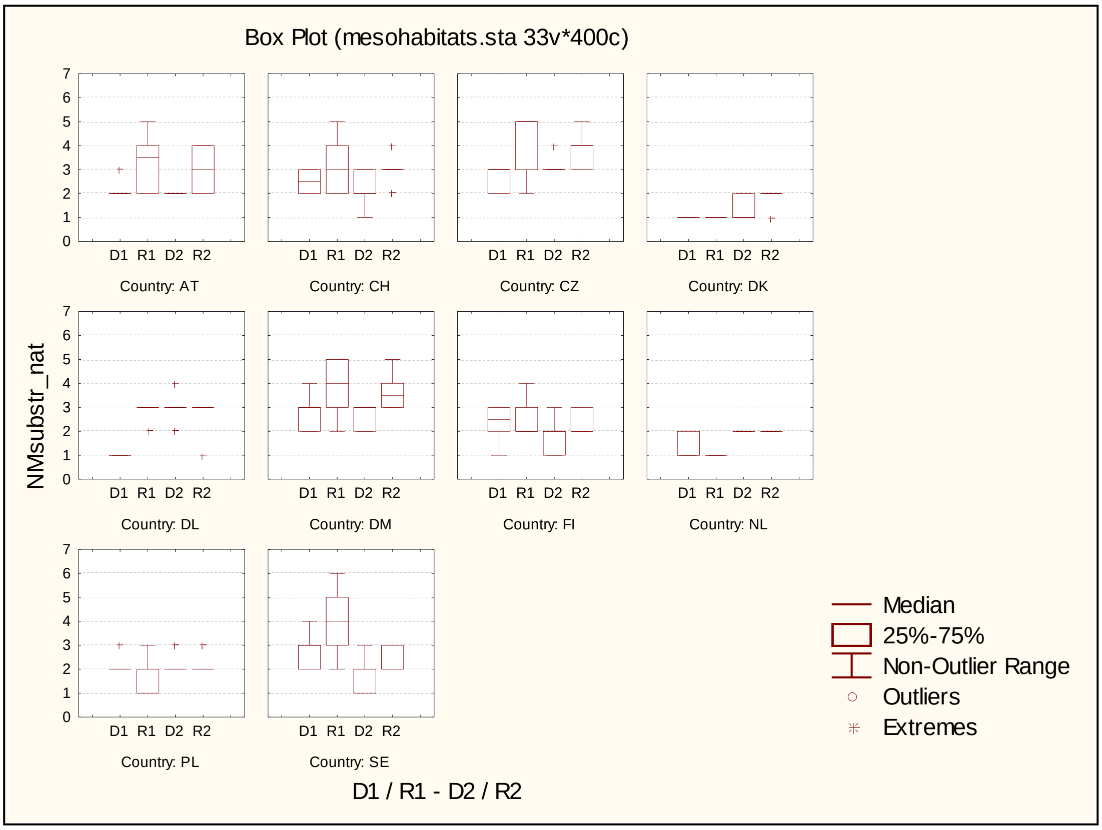


Figure 12-24 Variation of "number of natural dominant substrates" (NMsubstr_nat) of D1/R1/D2/R2 per country (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration)

The parameter "share of main channel width" of total length of transects (Mainchan_share) should reflect the dominance of the river channel in relation to other channel features along a river cross section. We assumed that in restored sections the "share of main channel width" is significantly lower than in degraded ones due to a more diversified morphology. Figure 12-25 proves this assumption in general; in five case study sites (AT, CH, DK, DM, PL) it is demonstrated for both restoration extents (R1 and R2). In FI, only the small restoration (R2) shows a lower share of main channel compared to the degraded sections, whereas in CZ, DL, NL, SE large restored sections correspond to our assumption.

Accordingly, in 12 out of 20 cases the positive difference between restored and degraded sections was statistically significant (pairwise comparisons R1/D1 – R2/D2 Mann-Whitney U-test, $p<0.05$ – see Table 12-20).

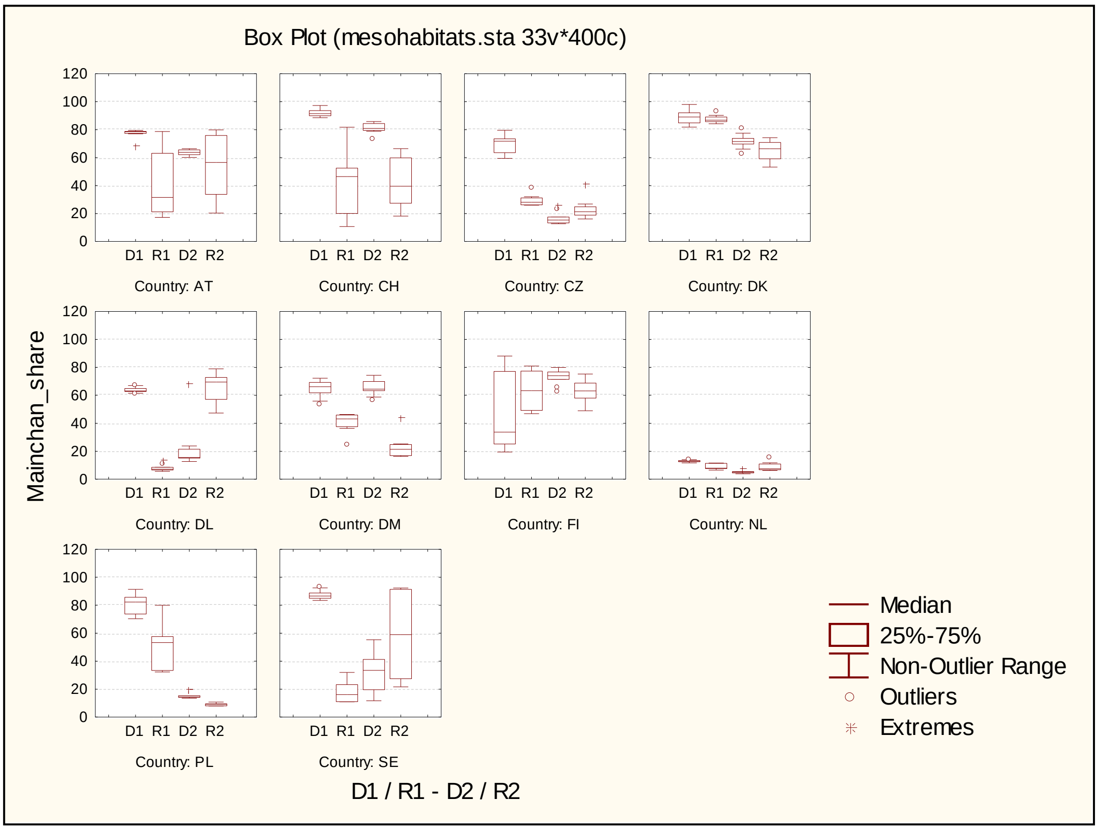


Figure 12-25 Variation of "share of main channel width of total transect length in %" (Mainchan_share) of D1/R1/D2/R2 per country (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration)

Referring to several studies about restoration effects (Jähnig et al. 2008, Feld et al. 2014), we calculated diversity indices (SWI, SDI) for channel features and dominant substrate. The results shown in Figure 12-26 (SWI) and Figure 12-28 (SDI) for channel features correspond strongly to Figure 12-23 (NMchanfeat_nat "number of channel features"). This is consistent because the SWI considers the number of channel features and the proportion of each feature in a transect. Taking into account the spatial composition of channel features along the transect (SDI, Figure 12-28), we were unable to identify a further differentiation between restored and degraded sections. The trend of changes remained the same as in Figure 12-23. Significantly positive differences of SDI_chanfeat "SDI channel features" between restored and degraded sections were slightly reduced (10 out of 20 cases) compared to NMchanfeat_nat, whereas one section now showed a significantly negative change (DK_R1; Table 12-20).

The SWI index of substrate diversity along transects (Figure 12-27) reflects the results in Figure 12-24 (NMsubstr_nat "number of substrate classes"). However, significant values are strongly reduced (Table 12-20). Only four case study sites out of 20 show significantly positive differences between restored and degraded sections; NL_R2 shows a significantly negative effect. This result is consistent with the SDI parameter of substrate diversity (SDI_substrate "Spatial Diversity Index of substrate classes" Figure 12-29). No finer distinction between restored and degraded sites was achieved by considering the spatial composition of substrate types along transects. Only in one case (PL_R2) there was a significantly positive change of SDI_substrate (Table 12-20) whereas both restored sections in NL showed a significantly negative change.

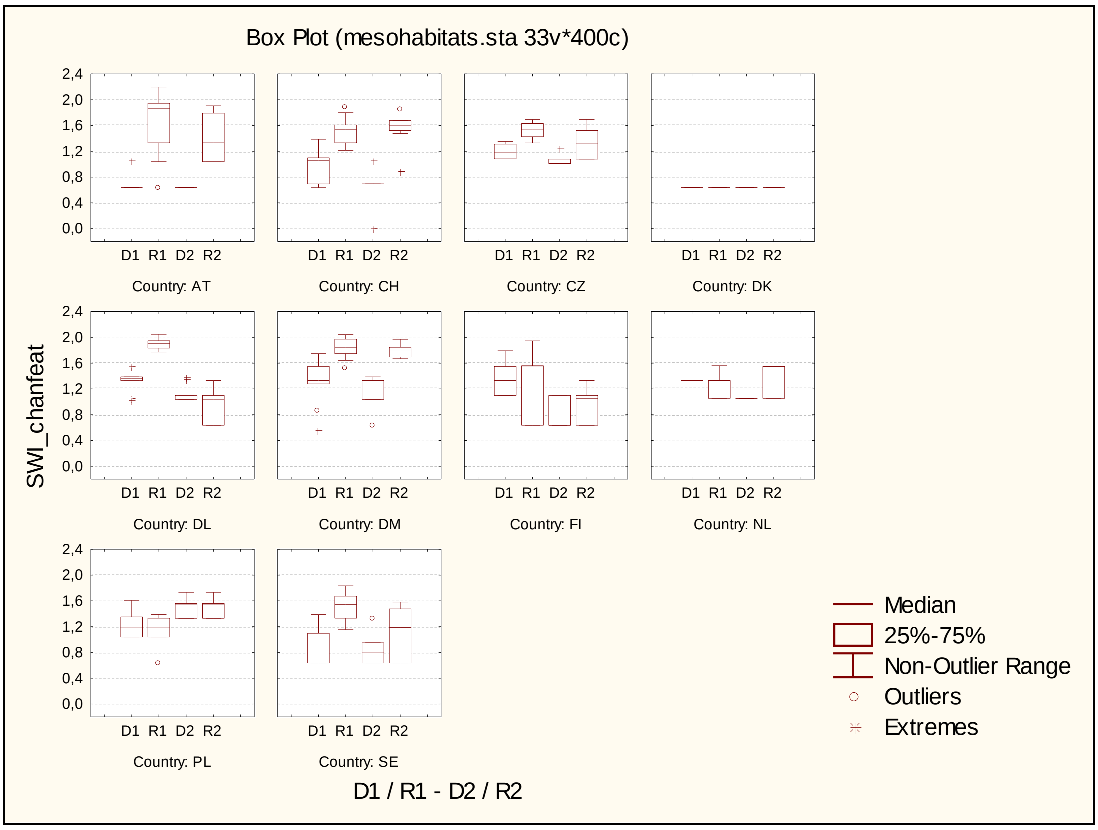


Figure 12-26 Variation of "Shannon–Wiener diversity index of natural channel features" (SWI_chanfeat) of D1/R1/D2/R2 per country (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration)

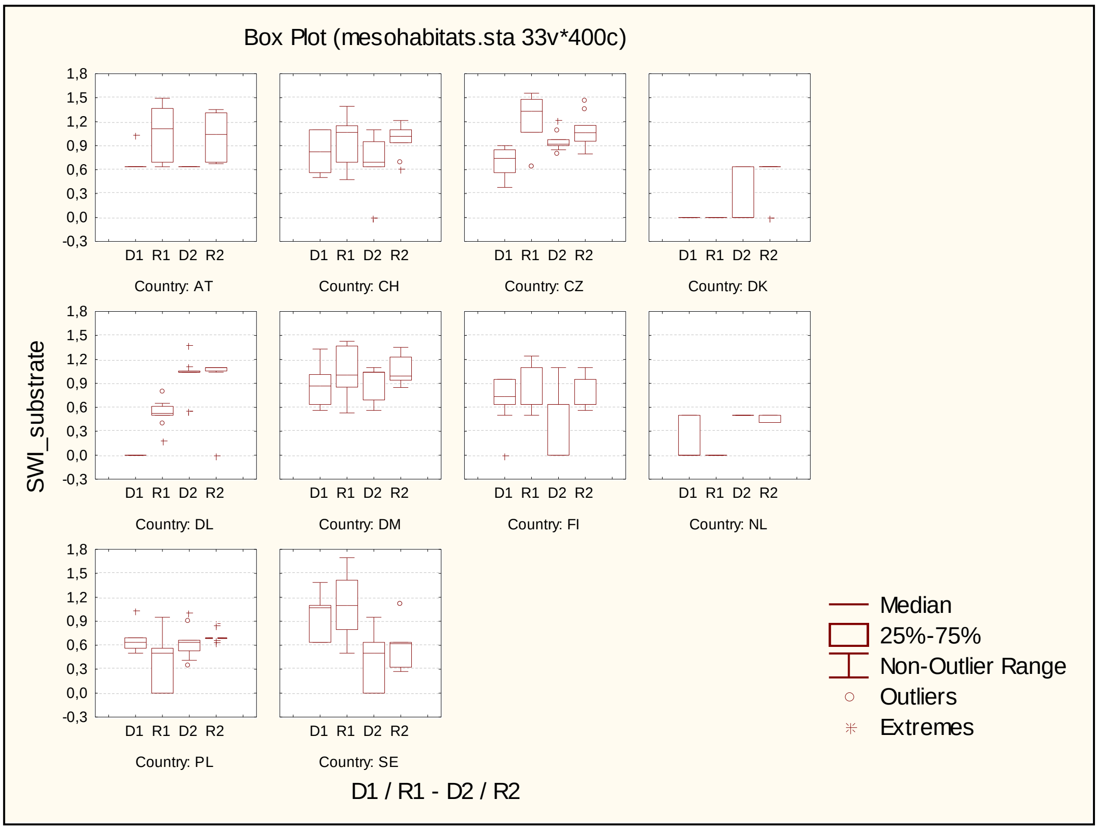


Figure 12-27 Variation of "Shannon–Wiener diversity index of natural substrate" (SWI_substrate) of D1/R1/D2/R2 per country (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration)

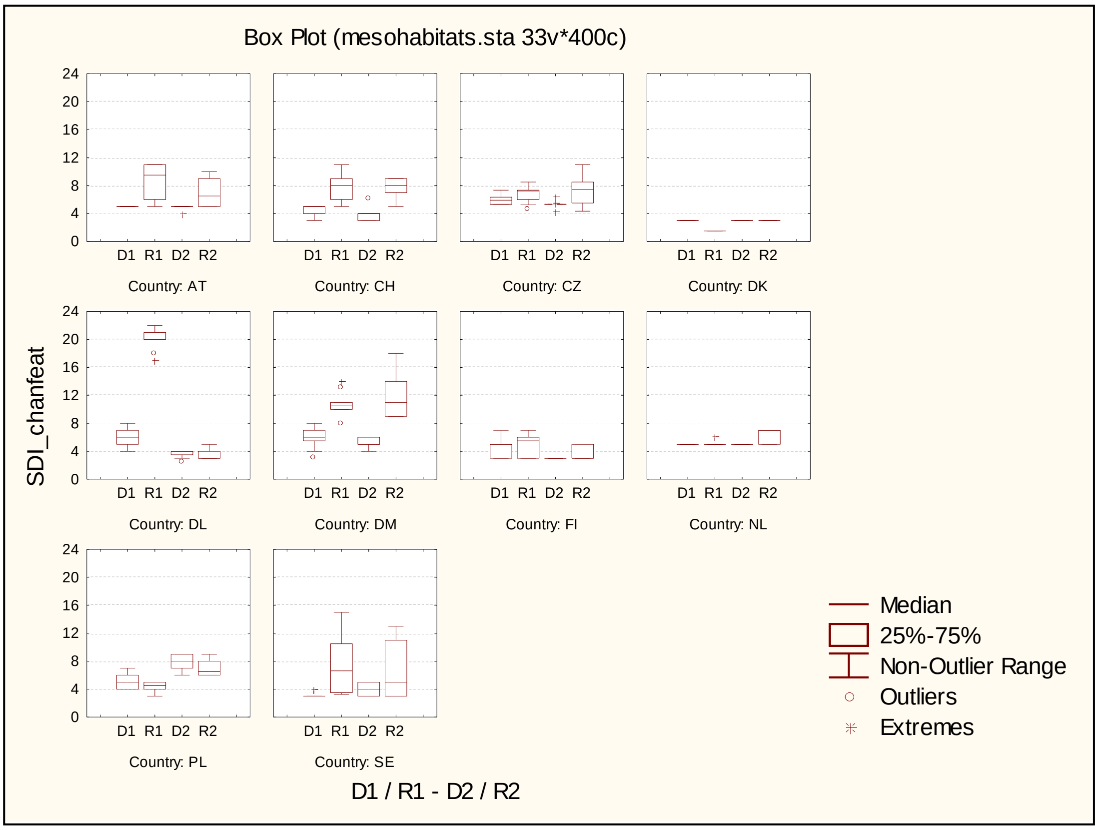


Figure 12-28 Variation of "Spatial Diversity Index of channel features" (SDI_chanfeat) of D1/R1/D2/R2 per country (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration)

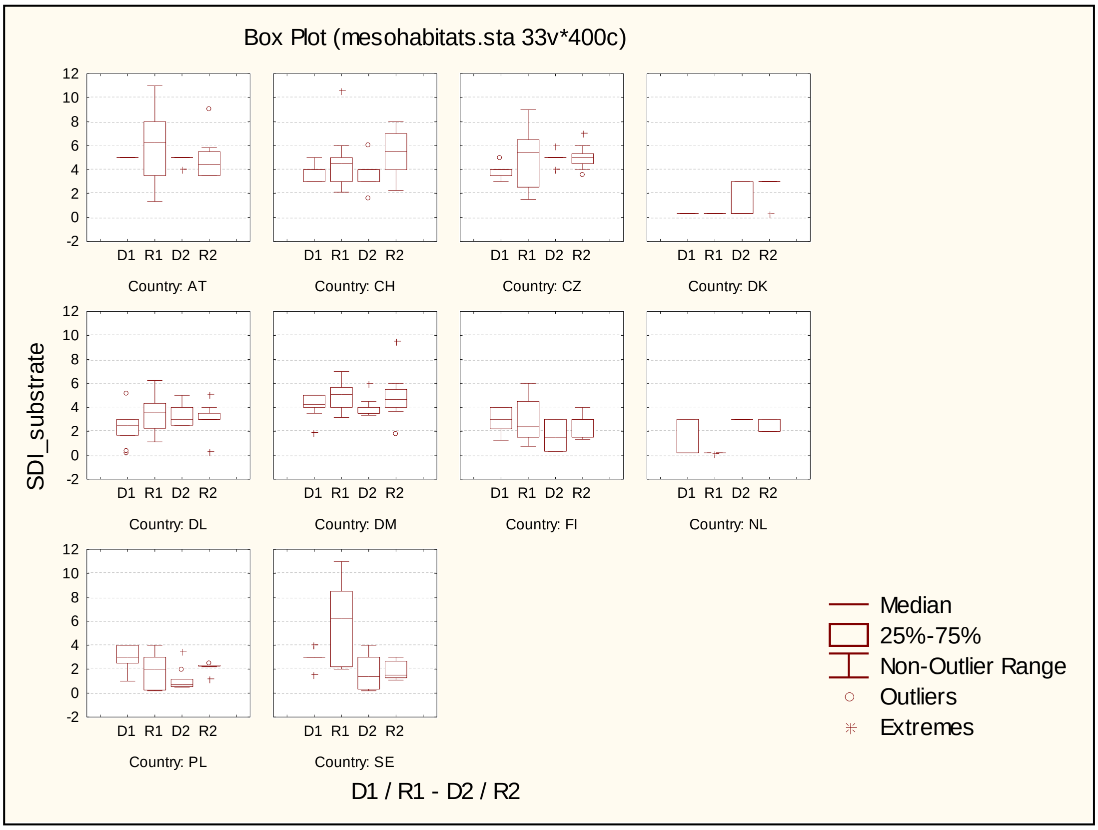


Figure 12-29 Variation of "Spatial Diversity Index of substrate" (SDI_substrate) of D1/R1/D2/R2 per country (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration)

Table 12-20 p-values of Mann-Whitney U test for 7 mesohabitat parameters shown from Figure 12-23 to Figure 12-29 for D1/R1/D2/R2 (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration).

	AT_D1 / AT_R1	AT_D2 / AT_R2	CH_D1 / CH_R1	CH_D2 / CH_R2	CZ_D1 / CZ_R1	CZ_D2 / CZ_R2	DK_D1 / DK_R1	DK_D2 / DK_R2	DL_D1 / DL_R1	DL_D2 / DL_R2
NMchanfeat_nat	**0.001***	**0.000***	**0.000***	**0.000***	**0.001***	**0.041***	1.000	1.000	**0.000***	*0.226*
NMsubstr_nat	**0.014***	**0.023***	0.131	**0.031***	**0.026***	**0.041***	1.000	0.059	**0.000***	*0.970*
Mainchan_share	**0.001***	0.450	**0.000***	**0.000***	**0.000***	**0.010***	0.597	0.070	***0.000****	**0.001***
SWI_chanfeat	**0.001***	**0.000***	**0.001***	**0.000***	**0.000***	**0.001***	1.000	1.000	**0.000***	*0.406*
SWI_substrate	**0.002***	**0.000***	0.290	0.104	**0.007***	0.112	1.000	0.059	**0.000***	0.121
SDI_chanfeat	**0.003***	**0.010***	**0.001***	**0.000***	0.112	**0.034***	***0.000****	1.000	**0.000***	*0.545*
SDI_substrate	0.705	*0.450*	0.545	0.064	0.151	0.791	1.000	0.059	0.096	*1.000*

	DM_D1 / DM_R1	DM_D2 / DM_R2	FI_D1 / FI_R1	FI_D2 / FI_R2	NL_D1 / NL_R1	NL_D2 / NL_R2	PL_D1 / PL_R1	PL_D2 / PL_R2	SE_D1 / SE_R1	SE_D2 / SE_R2
NMchanfeat_nat	**0.000***	**0.000***	0.910	0.104	*0.450*	**0.023***	*0.571*	*0.571*	**0.000***	0.112
NMsubstr_nat	**0.034***	**0.014***	0.821	0.121	*0.131*	1.000	*0.326*	*1.000*	**0.023***	0.070
Mainchan_share	**0.000***	***0.000****	*0.199*	**0.007***	**0.000***	***0.001****	**0.001***	**0.000***	**0.000***	*0.199*
SWI_chanfeat	**0.001***	**0.000***	0.910	0.199	*0.450*	**0.023***	*0.678*	*0.762*	**0.001***	0.064
SWI_substrate	0.174	0.450	0.623	0.140	*0.131*	***0.023****	*0.070*	0.054	0.257	0.326
SDI_chanfeat	**0.000***	**0.000***	0.650	0.131	0.450	**0.023***	*0.273*	*0.082*	**0.002***	0.345
SDI_substrate	0.162	0.054	*0.683*	0.131	***0.049****	***0.023****	*0.151*	**0.003***	0.290	0.450

*significant differences between degraded and restored sections (bold font, $p<0.05$); *negative effect (italic font);* positive effect (normal font)

Microhabitat results of restored sections

Figure 12-30 visualizes the variation of five microhabitat parameters within the ten paired restoration projects. Pairwise comparisons between restored and degraded sections were tested with Mann-Whitney U-tests, $p<0.05$. Restoration effect differed considerably among restoration projects: In some projects positive restoration effects were evident in most parameters, whereas in other restored sections none of the parameters indicated such effects.

The "number of natural microhabitats" along a transect was higher in both large (R1) and small (R2) restoration in only one case study (AT). In two case studies, the number of microhabitats was larger only in the large restoration (DL, NL), and in two case studies larger only in the small-scale restoration (DM, SE). There was no difference in number of microhabitats in cases (CH, FI, DK, PL) for either large or small restoration. In CZ there was a significant decrease in the number of natural microhabitats identified from the degraded compared to the restored section at large restoration. The "Shannon Wiener Diversity" index of natural microhabitats showed a similar pattern. The spatial composition of microhabitats (SDI_micro "Spatial diversity index") along a transect showed only in one case study (NL) higher values for restored sections at both restoration extents (R1 and R2). At small restoration in DL the "Spatial diversity index" was significantly smaller than in the degraded section.

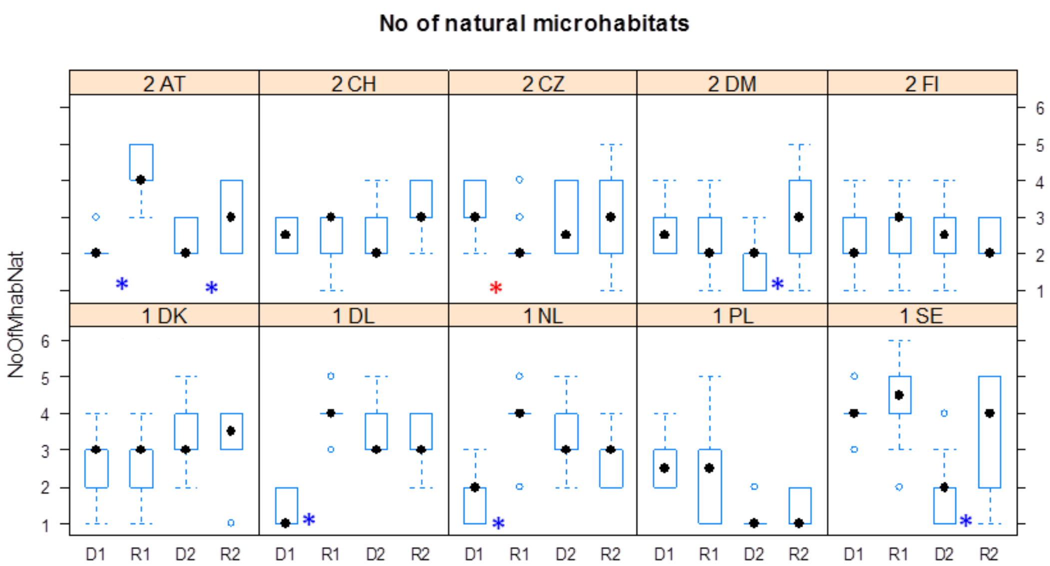
The "variance of depth" differed in three case studies (AT, DL, NL) at both restoration extents. All R1–sites at sand-bed rivers showed significantly higher depth variability than the degraded sites, with the exception of PL_R1.

The "variance of flow" was in many cases also increased in restored sections, in three case study sites (AT, NL, PL) for large and small restoration. In sand-bed rivers at large restoration extent, the flow was significantly more variable in each case study except in SE.

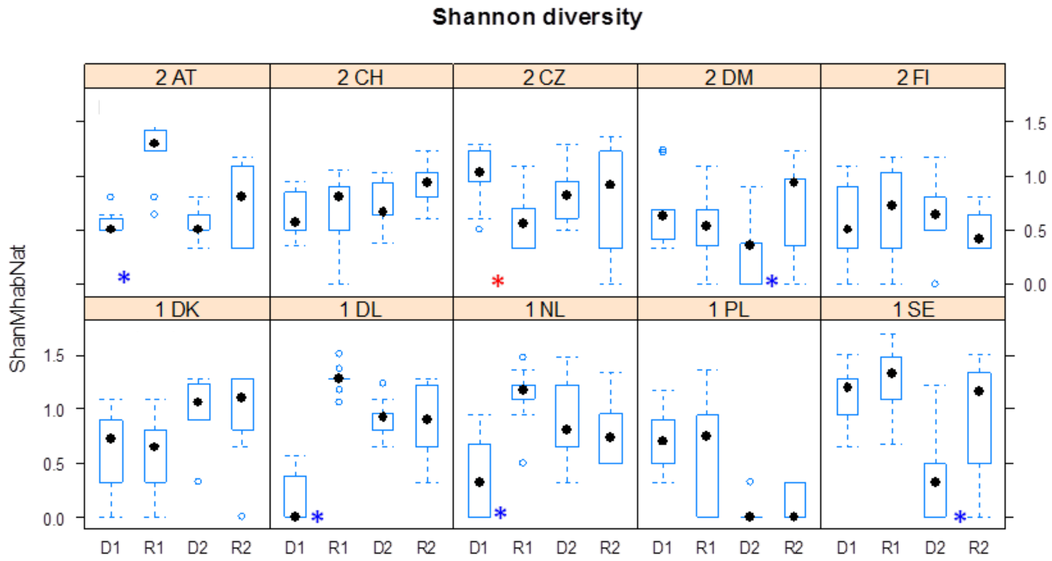
Only in large restoration did at least four (DL) or all five (AT, NL) parameters have significantly higher values in the restored sections than in the degraded ones. Overall, however, significant differences were equally frequent in large and small restoration.

In nine out of the ten case studies (all but CZ) at least one parameter showed a positive restoration effect either in large or small restoration. In three case studies (CH, DM, FI), significantly positive effects were observed only in small restoration. In two case studies, significantly negative effects were observed (CZ_R1, DL_R2), indicating that restoration sites show a decreased diversity in one of the five morphological parameters compared to the degraded sections.

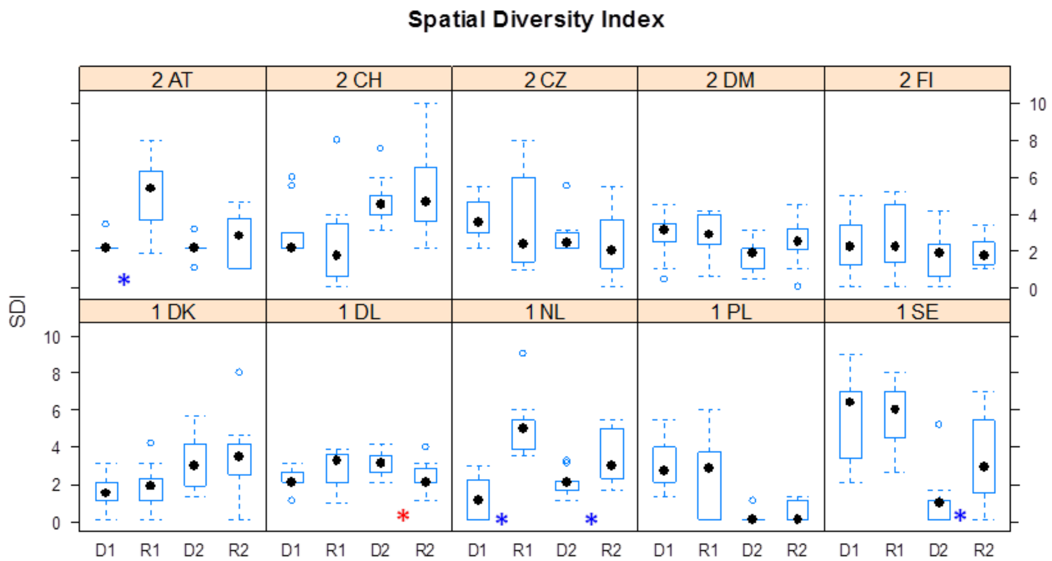
a)



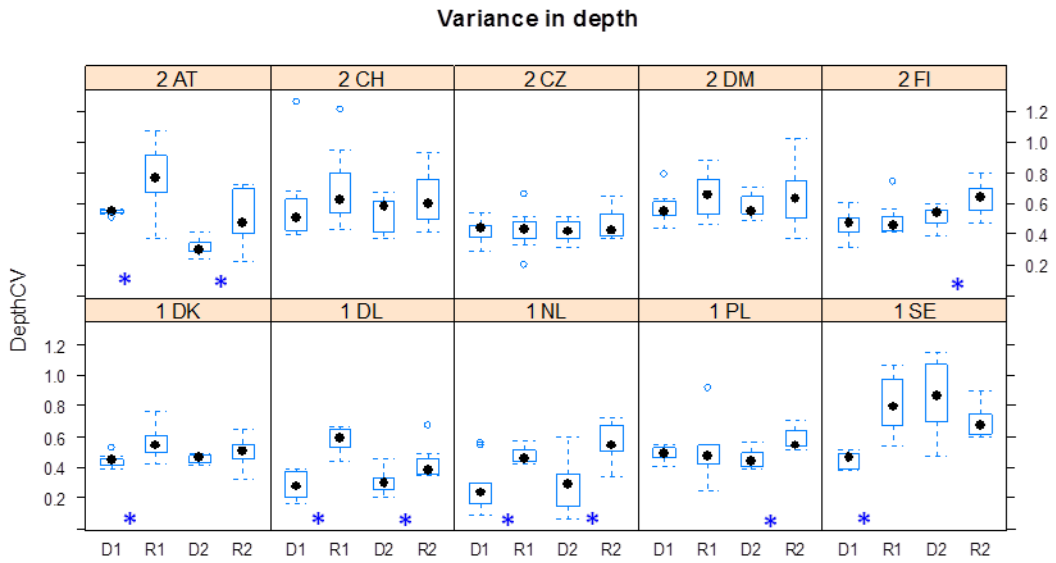
b)


Shannon diversity
2 AT
2 CH
2 CZ
2 DM
2 FI
1 DK
1 DL
1 NL
1 PL
1 SE
ShanMhabNat
0.0
0.5
1.0
1.5
D1
R1
D2
R2

c)


Spatial Diversity Index
2 AT
2 CH
2 CZ
2 DM
2 FI
1 DK
1 DL
1 NL
1 PL
1 SE
SDI
0
2
4
6
8
10
D1
R1
D2
R2

d)



e)

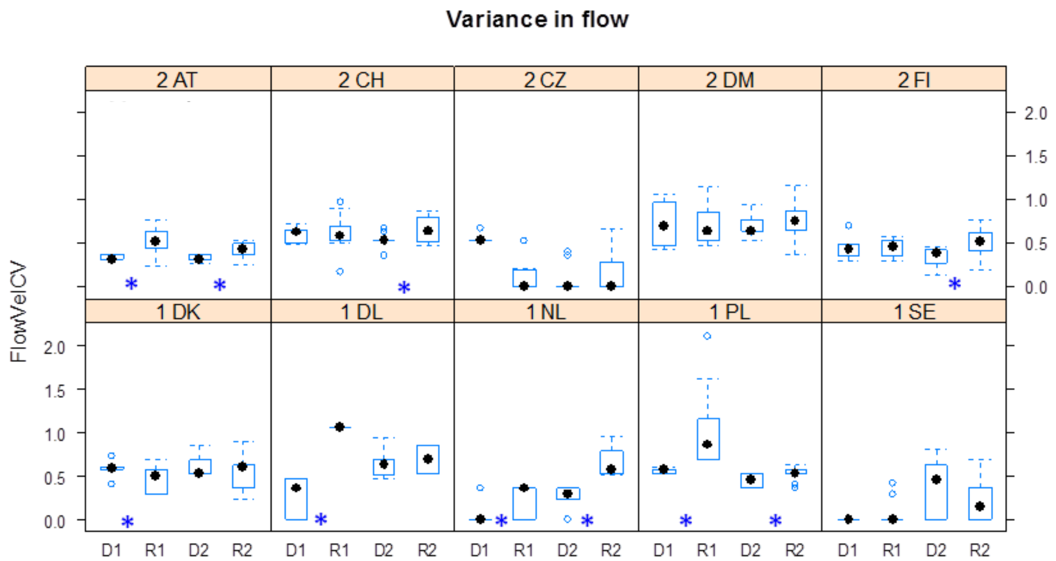
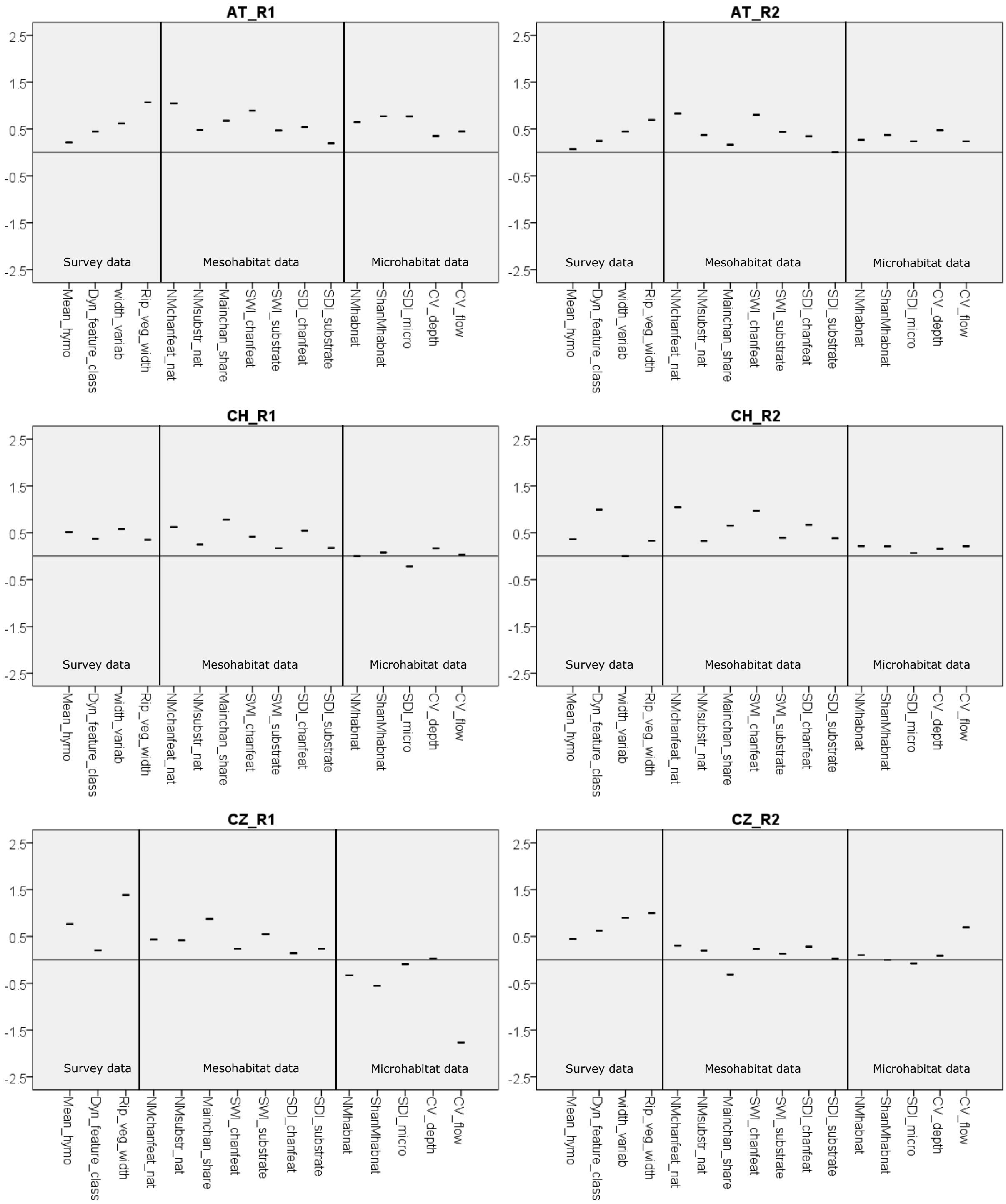
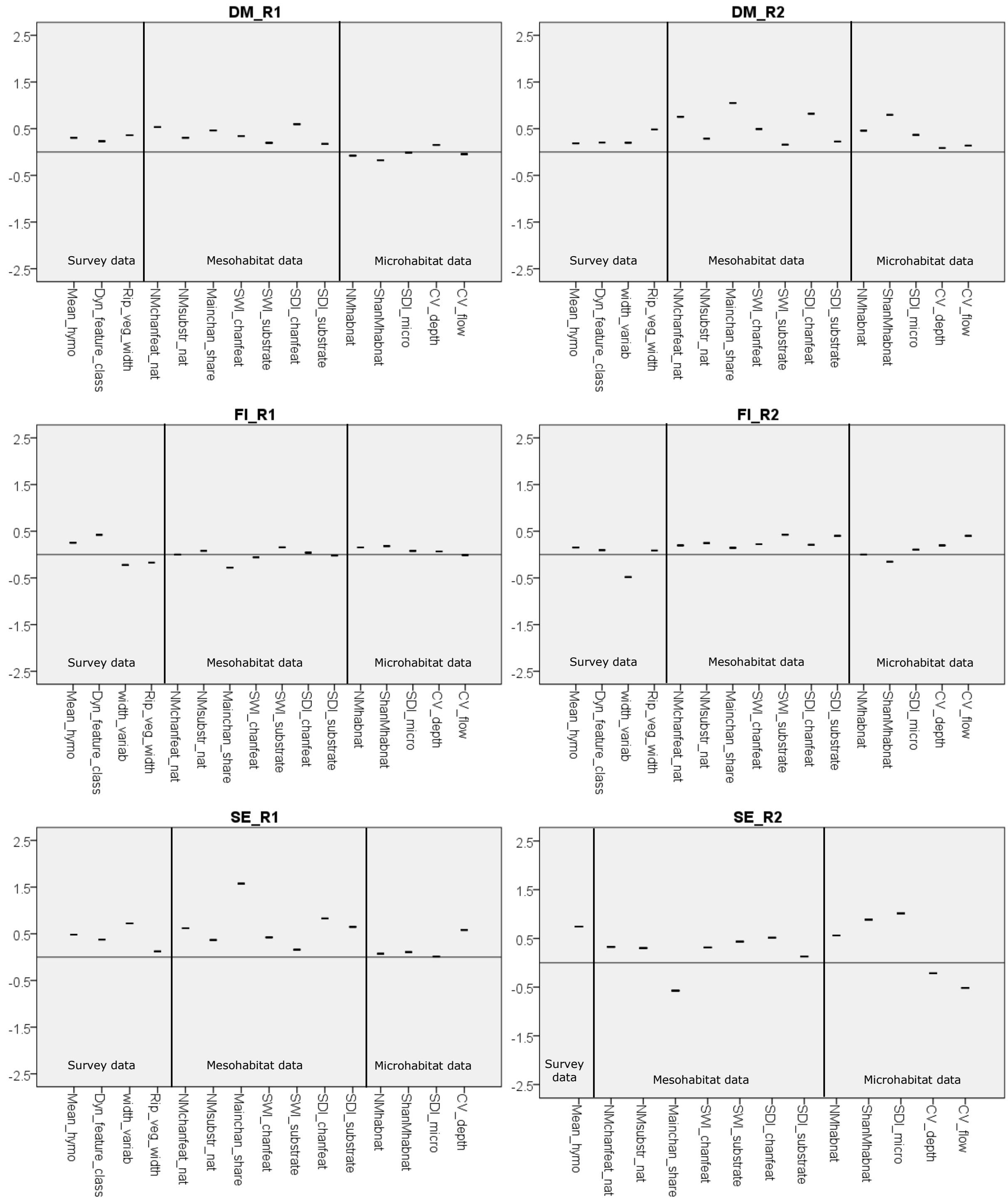
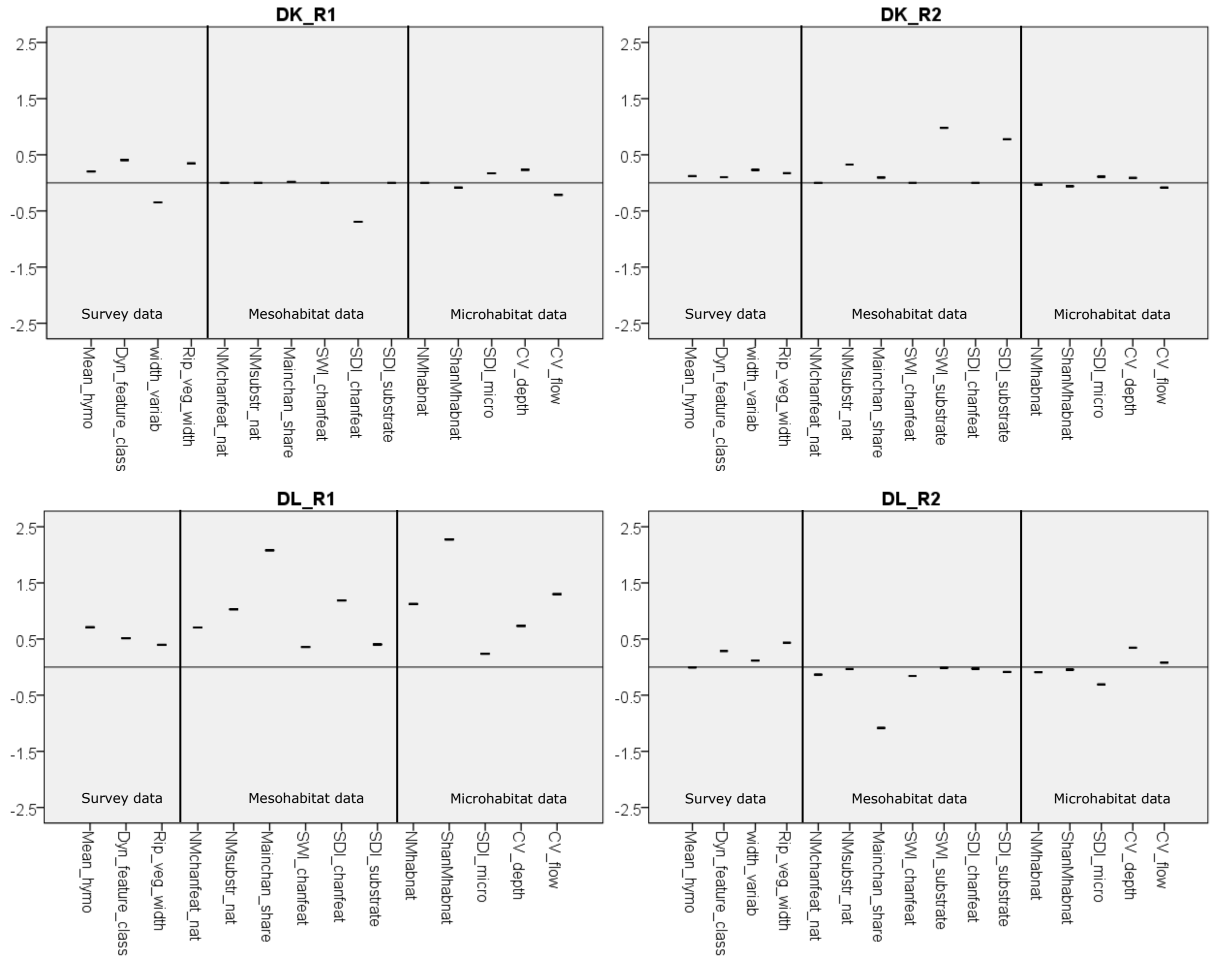


Figure 12-30 a)-e) Variation of the five aquatic microhabitat parameter values a) number of natural microhabitats, b) Shannon diversity, c) Spatial Diversity Index, d) variance of depth, e) variance of flow; of D1/R1/D2/R2 (R – restored; D – degraded; 1 – sections located at river with large restoration; 2 – sections located at river with small restoration); Significant differences (Mann-Whitney U test, p<0.05) between degraded and restored sections are indicated with an asterisk (red: negative restoration effect /blue: positive restoration effect)


AT_R1
AT_R2
CH_R1
CH_R2
CZ_R1
CZ_R2
2.5
1.5
0.5
-0.5
-1.5
-2.5
Survey data
Mesohabitat data
Microhabitat data
Mean_hymo
Dyn_feature_class
width_variab
Rip_veg_width
NMchanfeat_nat
NMsubstr_nat
Mainchan_share
SWI_chanfeat
SWI_substrate
SDI_chanfeat
SDI_substrate
NMhabnat
ShanMhabnat
SDI_micro
CV_depth
CV_flow


DM_R1
DM_R2
FI_R1
FI_R2
SE_R1
SE_R2
Survey data
Mesohabitat data
Microhabitat data
2.5
1.5
0.5
-0.5
-1.5
-2.5
Mean_hymo
Dyn_feature_class
width_variab
Rip_veg_width
NMchanfeat_nat
NMsubstr_nat
Mainchan_share
SWI_chanfeat
SWI_substrate
SDI_chanfeat
SDI_substrate
NMhabnat
ShanMhabnat
SDI_micro
CV_depth
CV_flow


DK_R1
DK_R2
DL_R1
DL_R2
2.5
1.5
0.5
-0.5
-1.5
-2.5
Survey data
Mesohabitat data
Microhabitat data
Mean_hymo
Dyn_feature_class
width_variab
Rip_veg_width
NMchanfeat_nat
NMsubstr_nat
Mainchan_share
SWI_chanfeat
SWI_substrate
SDI_chanfeat
SDI_substrate
NMhabnat
ShanMhabnat
SDI_micro
CV_depth
CV_flow

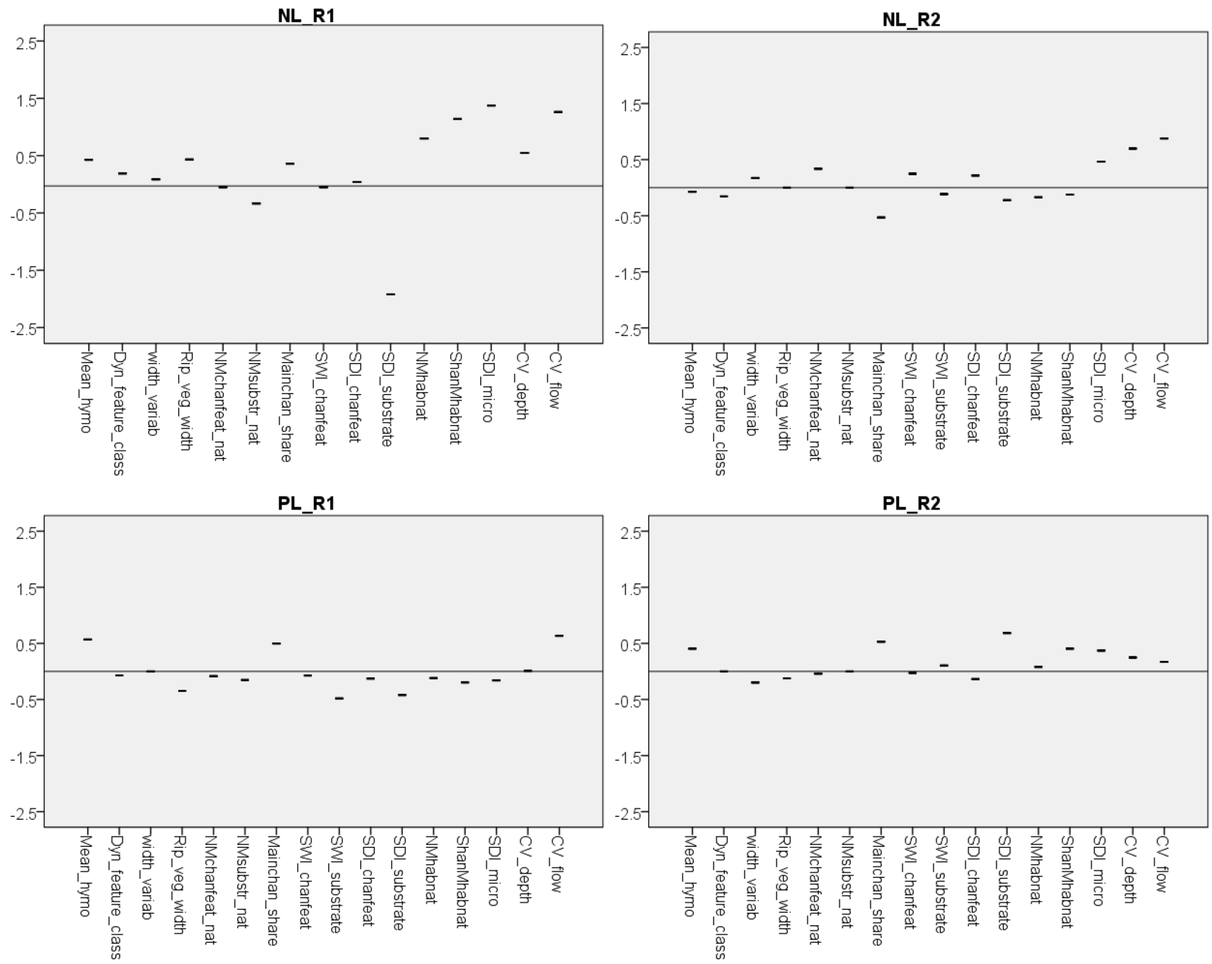


Figure 12-31 Restoration effect (ln(R/D)) of all hydromoprohological parameters differentiated by case study sections.

12.4 *Annex D: Fish data*

Table 12-21 Habitat guild classification and number of fish caught per section and species (coding of sections: country code, river name, degraded D or restored R, small 2 or large 1)

Nr	Species name	Guild	AT_Drau_D1	AT_Drau_R1	AT_Enns_D2	AT_Enns_R2	CH_Thur_D1	CH_Thur_R1	CH_Toss_D2	CH_Toss_R2	CZ_Becva_D1	CZ_Becva_R1	DE_Lahn_D2	DE_Lahn_R2	DE_Lippe_D1	DE_Lippe_R1
1	Abramis brama	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	1
2	Alburnoides bipunctatus	RHEO	0	0	0	0	214	315	0	0	4	155	0	0	0	0
3	Alburnus albidus	EURY	0	0	0	0	0	0	0	0	10	163	0	0	0	0
4	Alburnus alburnus	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	9
5	Anguilla anguilla	EURY	0	0	0	0	7	13	0	0	0	0	0	7	29	4
6	Aspius aspius	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	0
7	Barbatula barbatula	RHEO	0	0	0	0	52	173	45	197	0	4	52	57	16	398
8	Barbus barbus	RHEO	0	0	0	0	58	377	0	0	0	303	13	2	3	20
9	Blicca bjoerkna	EURY	0	0	0	0	0	0	0	0	1	0	0	0	0	0
10	Carassius auratus	LIMNO	0	0	0	0	0	0	0	0	2	0	0	0	0	0
11	Carassius gibelio	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	0
12	Chondrostoma nasus	RHEO	0	0	0	0	0	3	0	0	0	37	1	0	0	33
13	Cobitis taenia	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	35	96
14	Cottus gobio	RHEO	0	0	7	2	1	7	208	652	0	0	1	1	63	74
15	Cottus poecilopus	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0
16	Cyprinus carpio	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	2
17	Esox lucius	EURY	0	0	0	0	0	0	0	0	0	0	0	0	4	9
18	Gasterosteus aculeatus	EURY	0	0	0	0	0	2	0	0	0	0	185	13	5	188
19	Gobio gobio	RHEO	0	0	0	0	0	0	0	0	3	53	47	17	16	123
20	Gymnocephalus cernuua	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0
21	Lampetra planeri	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	1
22	Leuciscus leuciscus	RHEO	0	0	0	0	0	0	0	0	0	15	0	8	2	78
23	Lota lota	EURY	0	0	0	0	0	0	0	0	0	0	0	0	102	19
24	Misgurnus fossilis	LIMNO	0	0	0	0	0	0	0	0	0	0	0	0	0	0
25	Oncorhynchus mykiss	RHEO	4	11	10	12	0	0	0	0	0	0	0	0	0	0
26	Perca fluviatilis	EURY	0	0	0	0	0	0	0	0	6	2	0	0	15	42
27	Phoxinus phoxinus	RHEO	0	0	0	0	6	36	29	385	0	0	836	307	1	7
28	Platichthys flesus	LIMNO	0	0	0	0	0	0	0	0	0	0	0	0	0	0
29	Pseudorasbora parva	LIMNO	0	0	0	0	0	0	0	0	9	5	0	0	0	0
30	Pungitius pungitius	LIMNO	0	0	0	0	0	0	0	0	0	0	0	0	0	37
31	Rhodeus amarus	LIMNO	0	0	0	0	0	0	0	0	3	1	0	0	0	0
32	Romanogobio kesslerii	RHEO	0	0	0	0	0	0	0	0	4	17	0	0	0	0
33	Rutilus rutilus	EURY	0	0	0	0	0	0	0	0	0	0	0	0	1	33
34	Salmo salar	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0
35	Salmo trutta fario	RHEO	13	8	21	34	0	1	263	452	0	0	8	5	2	0
36	Salmo trutta trutta	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0
37	Scardinius erythrophthalmus	LIMNO	0	0	0	0	0	0	0	0	0	0	0	0	0	0
38	Silurus glanis	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	0
39	Squalius cephalus	EURY	0	0	0	0	64	222	0	0	188	136	3	4	4	32
40	Telestes soufia	RHEO	0	0	0	0	0	3	0	0	0	0	0	0	0	0
41	Thymallus thymallus	RHEO	2	32	48	18	0	0	0	0	0	0	8	6	1	5
42	Tinca tinca	LIMNO	0	0	0	0	0	0	0	0	1	0	0	0	0	0
43	Vimba vimba	RHEO	0	0	0	0	0	0	0	0	5	0	0	0	0	0
	Total		19	51	86	66	402	1152	545	1686	236	891	1154	427	299	1211

Table continued

Nr	Species name	Guild	DE_Ruhr_D1	DE_Ruhr_R1	DE_Spree_D2	DE_Spree_R2	DK_Skjern_D1	DK_Skjern_R1	DK_Storaa_D2	DK_Storaa_R2	FI_Kuiva_H_R2	FI_Kuiva_K_D2	FI_Vaara_N_R1	FI_Vaara_P_D1	SE_Eman_D1	SE_Eman_R1	SE_Morrum_D2	SE_Morrum_R2
1	Abramis brama	EURY	0	0	2	2	0	0	1	0	0	0	0	0	0	0	0	0
2	Alburnoides bipunctatus	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3	Alburnus albidus	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
4	Alburnus alburnus	EURY	0	0	41	117	0	0	0	0	0	1	0	0	1	7	11	18
5	Anguilla anguilla	EURY	0	0	1	1	1	0	4	1	0	0	0	0	0	0	0	0
6	Aspius aspius	EURY	0	0	3	8	0	0	0	0	0	0	0	0	0	0	0	0
7	Barbatula barbatula	RHEO	1026	1553	0	0	0	0	0	0	5	8	54	5	0	0	0	0
8	Barbus barbus	RHEO	74	169	0	0	0	0	0	0	0	0	0	0	0	0	0	0
9	Blicca bjoerkna	EURY	0	0	28	51	0	0	0	0	0	0	0	0	0	0	0	0
10	Carassius auratus	LIMNO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	Carassius gibelio	EURY	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0
12	Chondrostoma nasus	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
13	Cobitis taenia	RHEO	0	0	16	37	0	0	0	0	0	0	0	0	0	0	0	0
14	Cottus gobio	RHEO	1378	325	0	0	0	0	0	0	45	72	75	62	0	2	0	0
15	Cottus poecilopus	RHEO	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
16	Cyprinus carpio	EURY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
17	Esox lucius	EURY	0	0	8	2	0	3	1	1	0	0	1	1	7	7	1	3
18	Gasterosteus aculeatus	EURY	4	75	0	0	1	7	8	6	0	0	0	0	0	0	0	0
19	Gobio gobio	RHEO	25	10	0	0	0	0	4	1	0	0	0	0	0	0	0	0
20	Gymnocephalus cernuua	RHEO	0	0	0	2	1	0	1	2	0	0	0	0	0	0	0	0
21	Lampetra planeri	RHEO	26	14	0	0	0	0	0	0	0	0	0	0	0	0	0	0
22	Leuciscus leuciscus	RHEO	1	1	0	0	2	3	9	14	1	0	0	0	0	0	0	0
23	Lota lota	EURY	0	0	1	1	0	1	0	0	0	1	1	1	1	2	0	0
24	Misgurnus fossilis	LIMNO	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
25	Oncorhynchus mykiss	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
26	Perca fluviatilis	EURY	0	0	81	271	5	1	4	0	8	13	1	11	1	0	175	4
27	Phoxinus phoxinus	RHEO	4149	5961	0	0	0	0	0	0	24	0	0	0	0	0	0	35
28	Platichthys flesus	LIMNO	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
29	Pseudorasbora parva	LIMNO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
30	Pungitius pungitius	LIMNO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
31	Rhodeus amarus	LIMNO	0	0	4	3	0	0	0	0	0	0	0	0	0	0	0	0
32	Romanogobio kesslerii	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
33	Rutilus rutilus	EURY	0	0	73	287	2	1	4	0	3	9	5	0	1	8	4	4
34	Salmo salar	RHEO	0	0	0	0	1	10	0	81	0	0	0	0	0	30	0	41
35	Salmo trutta fario	RHEO	46	169	0	0	0	0	0	0	0	0	1	0	0	0	0	0
36	Salmo trutta trutta	RHEO	0	0	0	0	0	0	4	16	0	0	0	0	0	6	0	2
37	Scardinius erythrophthalmus	LIMNO	0	0	50	84	0	0	0	0	0	0	0	0	0	0	0	0
38	Silurus glanis	EURY	0	0	17	24	0	0	0	0	0	0	0	0	0	0	0	0
39	Squalius cephalus	EURY	62	19	38	57	0	0	0	0	0	0	0	0	0	0	0	0
40	Telestes soufia	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
41	Thymallus thymallus	RHEO	60	55	0	0	0	0	0	0	1	1	0	0	0	0	0	0
42	Tinca tinca	LIMNO	0	0	8	9	0	0	0	0	0	0	0	0	0	0	0	0
43	Vimba vimba	RHEO	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Total		6851	8351	373	960	15	28	40	122	87	105	138	80	11	62	191	107