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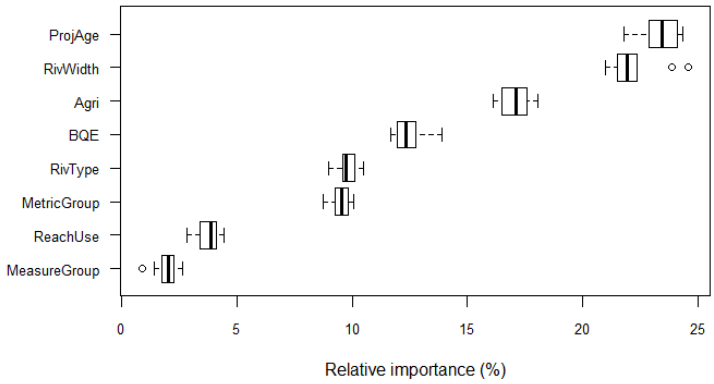
REstoring rivers FOR effective catchment Management



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Summary

An increasing number of rivers have been restored over 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 on biota is limited. Nevertheless, the monitoring results of several projects are available in peer-reviewed scientific literature and have been compiled in recent research projects. Some narrative reviews have already been published, but a comprehensive quantitative meta-analysis which summarizes the findings of these existing studies is lacking.

The objective of the study was to evaluate the effect of hydromorphological restoration on biota based on these existing data. The specific objectives were to quantify restoration success, to identify catchment, river reach, and project characteristics which influence (either constrain or enhance) the effect of restoration, and to derive recommendations for river management.

In the meta-analysis, quantitative research findings from a large number of studies were compiled. A meta-analysis is restricted to one single type of research finding and qualitative information cannot be used, but it is less subjective than narrative reviews and allows to investigate the effect of "moderator variables", i.e. to identify variables which influence restoration success and hence, to identify effective measures and to describe favourable conditions for river restoration. The meta-analysis was complemented by a satellite topic on urban restoration to identify differences between the characteristics of urban and non-urban restoration projects.

Based on the results of the meta-analysis, the satellite topic, and other comprehensive reviews on river restoration already published in literature, the following conclusions were drawn. It is important to note that - as for all statistical analysis - it is not possible to infer causal relationships and hence, results should be interpreted with caution. Furthermore, for most results of this study, restoration success refers to an increase in the number of individuals and taxa simply because these metrics were reported in literature, but other metrics might be better suited to quantify success (e.g. stream-type specific conditions, functional approaches). It is strongly recommended to read the results and discussion sections before applying the results to avoid oversimplified interpretations.

***Overall, the effect of hydromorphological restoration on biota is positive but variability is high**:* Restoration in general has a positive effect on floodplain vegetation, ground beetles, macrophytes, fish, and invertebrates. Since variability is high, adaptive management approaches are recommended.

Restoration effect differs between organism groups: It cannot be expected that all organism groups benefit from restoration to the same extent. Results indicate that, in general, restoration effect on diversity is highest for terrestrial and semi-aquatic groups like floodplain vegetation and ground beetles, intermediate for macrophytes, lower for fish, and lowest for macroinvertebrates.

Restoration has a higher effect on the number of individuals than on the number of taxa: The effect of restoration is more pronounced on the number of fish and invertebrate individuals than on the number of taxa.

Restoration effect does only slightly differ between measures, i.e. there is no single "best" measure: There are no large differences in the overall effect of different measures but there is a tendency that terrestrial and semi-aquatic organism groups like floodplain vegetation and ground beetles as well as macrophytes benefit most from channel-planform measures and aquatic groups like fish and invertebrates from instream measures.

Urban restoration projects do not substantially differ in respect to the pressures occurring and the measures applied: Urban restoration projects were mainly applied in small rivers and length of the restored reaches was shorter compared to non-urban restoration projects. However, approaches towards (sub)urban and non-urban river rehabilitation practices were similar.

Urban restoration projects are rated less successful compared to non-urban projects by scientists and river managers*:* …which is probably due to the low absolute effect, which is rated as a failure, but which often is a high relative effect since urban river start from a low base, and hence, should not be assessed too negative.

Conditions which favour restoration success can be identified but restoration outcome cannot be predicted: Restoration success is especially high for the different organism groups under specific conditions (see section 2.2.5) but the variance explained by the models is too low and low sample size restricted the use of rigorous statistical tests to really predict the restoration outcome.

Overall, restoration success most strongly depends on project age, river width, and is affected by agricultural land use: Success is generally lower but restoration still has a positive effect in catchments dominated by agricultural land use. Since land use is a proxy for e.g. water quality, there is an urgent need to identify the underlying causal relationships. Project age is the most important predictor affecting restoration success, but the direction of the effect of project age on restoration success differs between organism groups (no simple increase of effect with time). There is an urgent need for long-time monitoring to investigate the restoration effect over time, to better understand the trajectories of change induced by restoration measures, and to identify sustainable measures which enhance biota in the long-term.

In summary, it was possible to draw some first conclusion for river management from the evaluation of hydromorphological restoration based on existing monitoring data. However, monitoring data are still scarce and more robust, practical relevant, and quantitative results (e.g. thresholds) could be derived and river management would benefit from (i) original monitoring data, which would allow to use functional metrics to investigate the underlying processes and to infer causal relationships, (ii) full before-after-control-impact monitoring designs, which most probably would substantially decrease scatter in the datasets and analyses, (iii) a larger number of monitored projects, which easily could be accomplished since a large number of hydromorphological restoration measures will be implemented in the upcoming years, (iv) the availability of long-time monitoring data sets to investigate the effect of project age, which was identified as the most important variable affecting restoration success. A more intensive exchange and collaboration between river science and river management in planning monitoring programs is strongly recommended. This would offer a great opportunity to make fundamental advances in our understanding of how river restoration affects river hydromorphology and biota and to identify (cost)-effective restoration measures.

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1. Introduction

Over the last decades, enhancing the hydromorphological and biological state of degraded rivers has become a widely accepted 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, Kail et al. 2007). Nevertheless, monitoring results of several projects have been reported in peer-reviewed scientific literature in the last two and a half decades.

The 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). Besides the studies on specific restoration measures or organism groups, there are few reviews on the effect of river restoration on biota in general, which make use of the growing number of monitoring results available in literature. Most of these narrative reviews qualitatively described restoration success (Roni et al. 2002, 2008) but did not quantify it, except Miller et al. (2010) who investigated the effect of different measures on invertebrate diversity and abundance in a quantitative meta-analysis of peer-reviewed literature using effect sizes. However, a comprehensive meta-analysis on the effect of river restoration on biota is missing.

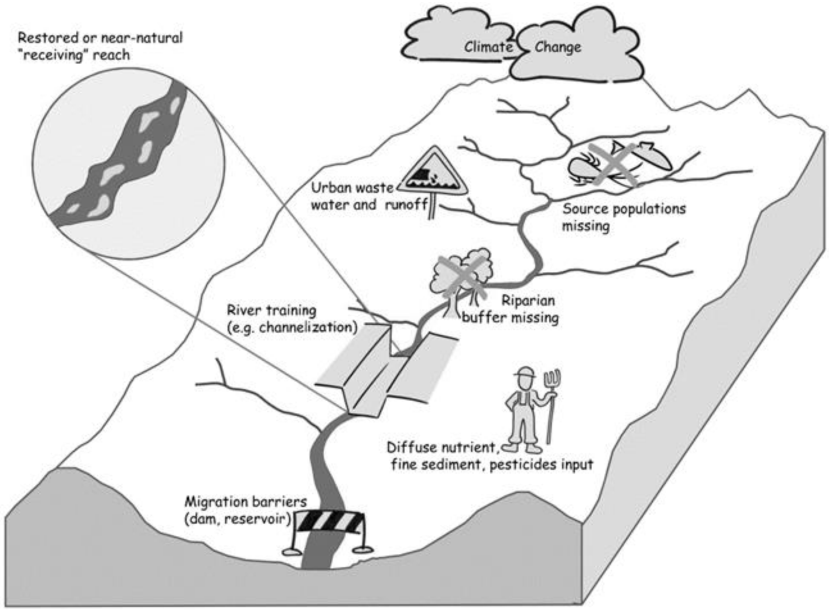


Figure 1-1: Pressures at larger spatial scale affecting reach-scale restoration.

It has been widely stated that the effects of reach-scale river restoration measures are potentially constrained by large-scale pressures (**Figure 1-1**), depend on catchment, river, and project characteristics, and are prone to failure if large scale processes and pressures are not adequately considered (Bond and Lake 2003, Roni 2008, Miller et al. 2010, Palmer et al. 2010). Several empirical studies indicated that large-scale pressures like catchment land-use can be more important in shaping invertebrate and fish communities compared to pressures at smaller spatial scales (Roth et al. 1996, Allan et al. 1997, Black et al. 2004, Hughes et al. 2008, Stephenson and Morin 2009, Sundermann et al. 2013). They might even limit invertebrate and fish assemblages (Wang et al. 2007, Bryce et al. 2010, Kail et al. 2012), and hence, potentially govern river biota and constrain the effect of reach-scale river restoration measures. However, in contrast to the numerous studies on pressure impacts at different spatial scales on the biological state, there is limited empirical evidence for an effect of catchment, river, and project characteristics on restoration success, like the importance of the local fish species pool (Stoll et al. 2014). In a comprehensive meta-analysis, the effect of variables describing catchment, river, and project characteristics could be used as predictor variables to investigate their effect on restoration success.

The first objective of this meta-analysis was to quantify the effect of restoration based on the monitoring results available in peer-reviewed as well as grey literature and unpublished studies. It was hypothesized that the restoration effect differs between organism groups (e.g. fish, macroinvertebrates, macrophytes), biological metrics (e.g. abundance, richness), and restoration measures. The second objective was to identify catchment, river, and project characteristics which influence (either constrain or enhance) the effect of restoration on biota, to investigate interactions between these predictors, and to quantify their relative importance.

Data on the effect of river restoration projects were compiled from two different sources: peer-reviewed literature and original monitoring data obtained from some of the REFORM partners. Originally, we planned to identify and classify restoration effects on hydromorphology, functioning, and ecological status (**Table 1-1**). However, the number of studies investigating the effect of restoration on these three specific response variables was limited. Based on the original monitoring data, biological metrics could be calculated to investigate the effect of restoration on functioning and the ecological status (e.g. ecological quality ratio, functional metrics) but sample size would still have been low. Nevertheless, such a more in depth study of the original monitoring data is planned and will be published in a later stage of the REFORM project. In this meta-analysis, we focused on the biological response variables which were mainly reported in literature (richness, diversity, biomass, and abundance), and for which sample size was large enough to allow for a rigorous statistical analysis.

The meta-analysis was complemented by a satellite topic on urban restoration to identify differences between the characteristics of urban and non-urban restoration projects. Urban river restoration was dealt with in more detail (section 3) particularly because it has been widely stated that restoration in urban rivers fundamentally differs from other settings (primarily because competing land uses and development practises often result in boundary conditions that limit restoration options). The number of urban projects for which monitoring data were available was limited, and hence it was not possible to compare the effect of restoration for urban and non-urban projects in-depth. Instead, the main objective of the urban restoration satellite topic was to identify best practise, effective restoration management by identifying strengths and overcoming limitations in measures identified by building on experiences that were already available and information on restoration projects which have been compiled in WP1. Specific objectives were (i) review available information to evaluate the current state of urban rivers and factors impacting upon them through a review of morphological, hydrological and water quality pressures, (ii) to compare difference in pressures acting on and measures applied in urban and non-urban case studies (iii) to provide an example of an urban cases study where an opportunistic approach was adopted when restoration was carried out in conjunction with flood risk management intervention (see Appendix).

Table 1-1: Description of the task's objectives.

Original task's objectives	**Actual tasks done**
Compile data on restoration cases based on previous work in WP1, WISER and FORECASTER and independently collated by project partners. Data will cover a set of hydrological and morphological variables, the response of BQEs and ecosystem functioning. Data on catchment characteristics (land cover, potential sediment input, water chemistry) will be centrally compiled. Response to restoration will be classified for a set of hydro-morphological, ecological and functional variables.	Data on the effect of river restoration projects were compiled from two different sources: peer-reviewed literature and original monitoring data obtained from REFORM partners. We focused on the biological response variables which were mainly reported in literature (richness, diversity, biomass, and abundance), and for which sample size was large enough to allow for a rigorous statistical analysis. Analysis of the catchment characteristics was limited to land cover since global data on potential sediment input and water chemistry were not available.
Evaluate restoration success. Response to restoration will be… specified for different restoration techniques and river / catchment types. There will be a special focus on large impacted European rivers affected by hydropower, flood prevention, land drainage and navigation (Rhine, Danube).	The effect of different restoration techniques (measures) was investigated, and catchment, river, and project characteristics were identified which influence (either constrain or enhance) the effect of restoration on biota. Large rivers were identified as an important cross-cutting theme in REFORM and this topic was investigated in a new separate task.
Link restoration success to catchment characteristics. Case by case and for groups of similar cases, the magnitude of restoration effects will be linked to catchment characteristics, to pressures potentially constraining restoration success and proxy variables quantifying these pressures. The results will be used to identify thresholds for pressures identified in WP3 that constrain the effect of local and reach-scale restoration measures (e.g. share of urban land-use or impervious cover in the upstream catchment).	Catchment, river, and project characteristics were identified which influence (either constrain or enhance) the effect of restoration on biota. It was further investigated if any of these predictors limits restoration success and if thresholds can be identified.

2. A meta-analysis of restoration effect on biota

2.1 *Materials and Methods*

2.1.1 Methodological background on meta-analysis and effect sizes

In meta-analysis, quantitative research findings from a large number of studies are compiled. In contrast to conventional narrative literature reviews, the research findings are quantified. This approach is less subjective and, besides quantifying the effect, allows to investigate the relationship between study findings (effect sizes) and study characteristics (moderator or predictor variables). In a specific meta-analysis only one type of study findings can be meaningfully compared, for example bivariate correlations or group comparisons.

Studies on the effect of river restoration usually compare the conditions in a restored river reach (treatment) either to the conditions prior to restoration (Before-After monitoring design) or to a nearby, still degraded, unrestored control reach (Control-Impact monitoring design), and hence the findings of river restoration studies belong to the type of "group comparisons". Each single restoration project can be considered a study or experiment where the effect of a treatment (restoration measures) on the state (e.g. abundance, diversity) of the objects of interest (different organism groups) has been investigated.

In restoration studies, different state variables (e.g. biomass, abundance, richness, diversity) were used and measured in different units (e.g. total number of individuals, individuals per square meter). Therefore, in a meta-analysis it is necessary to standardize the different state variables and units using a single effect size.

In the following, more detailed information on effect sizes are given for the interested reader to ease interpretation of the results. However, the main findings can be understood without detailed methodological knowledge.

In traditional meta-analysis, which first was developed in social and medical science, the most widely used effect size for group comparisons (e.g. restored vs. control) is the standardized mean difference d (Lipsey and Wilson 2001, Hunter and Schmidt 2004):

$$d = \frac{\bar{X}_T - \bar{X}_C}{SD_{pooled}} J$$

with $\bar{X}_T$ and $\bar{X}_C$ being the means of the treatment (restored) and control (unrestored or pre-restoration), SD_{pooled} the pooled standard deviation of the treatment and control values, and J a value to weight effect sizes based on the number of samples.

An alternative effect size is the response ratio Δr developed by 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 (unrestored or pre-restoration), values > 0 denoting a positive effect (e.g. increase in diversity or abundance), and negative values a negative effect. The response ratio is dimensionless since $\bar{X}_T$ is divided by $\bar{X}_C$. According to Osenberg et al. (1997), an exponential model is assumed by using a logarithmic function, i.e. a fast increase of the biological metrics in the first years and a smaller increase in the following years until equilibrium is reached.

The response ratio can be made time-invariant by dividing it by the time between implementation of the measures and monitoring t (referred to as project age in the following):

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

For two reasons, the response ratio was used as effect size in this meta-analysis. First, variance based effect sizes like d, where values are standardized using the standard deviation, are considered less suitable for ecological data since variability in the data often are due to factors which have nothing to do with the treatment effect (Osenberg et al. 1997). For example, variability of richness values in one study might simply be higher, and hence effect sizes smaller, due to a higher spatial heterogeneity of the landscape. Second, one main objective of the study was to identify catchment, river, and project characteristics which influence the effect of restoration on biota, called moderator or predictor variables in meta-analysis. Many of these predictor variables can only be quantified if the exact location of the treatment and control reaches are known (e.g. catchment land use, river width) or for single projects (e.g. project age, main measures applied), i.e. mainly for unreplicated studies which investigated one single treated (restored) and control (unrestored) reach. The standardized mean difference d is not applicable for these studies since it needs a standard deviation value to be calculated, i.e. several treatment and control reaches, respectively. In many studies, several samples were taken in a single pair of treatment / control reaches and standard deviation values could be calculated based on these replicate samples. However, this would be a pseudo-replication since all samples were affected by the same conditions (catchment, river, project characteristics).

For interpretation of the results, it has to be noted that a meta-analysis based on variance based effect sizes like d or the response ratio Δr fundamentally differs from studies investigating the effect of restoration on the biological state in respect to two aspects: First, the above mentioned effect sizes are relative values in contrast to the absolute state values, which have been compared between restored and unrestored reaches in most replicate studies (Jähnig et al. 2009, 2010, Schmutz et al. 2013, Lorenz et al. 2012, Pretty et al. 2003, Lepori et al. 2005). As a consequence, a larger increase in absolute values starting from an already high value (e.g. already high taxa number in a moderately disturbed forested river which increased from 50 to 60 by 20%) might result in a similar effect size compared to a smaller absolute increase starting from a low value (e.g. very low taxa number in a heavily degraded urban river which increased from 5 to 6 by 20%). Second, the state value of the restored reach is standardized using the state value of the unrestored reach, which was affected by the same conditions and hence, quantifies the effect of the treatment (restoration) independently from the effect which catchment and river characteristics have on the state of the restored reach. In contrast, studies investigating the effect of catchment and river characteristics (e.g. catchment land use) on the state of the restored reaches (e.g. Lorenz and Feld 2013) are not conceptually different from other studies on the relationship between environmental variables and the biological state of unrestored reaches (e.g. Kail and Wolter 2013, Sundermann et al. 2013).

2.1.2 Data sources and computation of effect sizes

Data on the effect of river restoration projects on biota were compiled from two different sources: peer-reviewed literature and original monitoring data obtained from some of the REFORM partners.

In peer-reviewed literature, studies were identified using the search engines Web of Science and SCOPUS by searching for the following keywords on 10.01.2012:

restor* OR rehabilit* OR revitali* OR renat* OR enhance* OR mitigate*

AND

aquatic habitat* OR reach* OR channel* OR stream* OR river* OR watershed* OR catchment* OR wetla* OR floodpla*

This resulted in 3661 hits, which were then screened using the criteria for inclusion in the analysis listed in **Table 2-1**.

Table 2-1: Criteria for inclusion of peer-reviewed publications in the meta-analysis.

Criteria	Include	Exclude
Ecosystem	River channel, riparian area, floodplain	Lakes, coastal waters
Location	Global (European and non-European countries)	None
Project objectives	Restoration, rehabilitation, mitigation	Conventional engineering or flood protection
Progress	Implemented	Planned (e.g. RBMP)
Measures	Hydromorphological measures	Water quality or river continuity only
Monitoring data	Before/After, Control/Impact, BACI	No monitoring data, only data after restoration
Environmental data	Basic river and project characteristics reported (e.g. location)	Replicate studies with limited information on single projects
Success	Irrespective of success (i.e. success and failures)	None

Out of the 3661 hits, 316 papers met the criteria on ecosystem, location, project objectives, progress, measures, and success, which is a comparable number of papers reviewed by e.g. Roni et al. (2008). The criteria on monitoring data and the basic environmental data reported in the publications further limited the number of suitable studies to n = 74, although first authors of the papers were contacted if necessary and virtually all kindly provided missing monitoring results or environmental data. In some publications, data on more than one restoration project have been reported, and hence the number of peer-reviewed projects was slightly higher (n = 87). The original data obtained from the REFORM partners on 64 restoration projects was compiled in the same database as the peer-reviewed literature. For the original monitoring data, duplicate projects already described in peer-reviewed literature were deleted and species lists were taxonomically adjusted for each pair of restored / unrestored reaches to avoid any artificial differences due to operator bias. Virtually all studies investigated the effect of restoration on fish, macroinvertebrates, or macrophytes, and the only study dealing with algae was excluded since the resulting number of effect sizes was too low for statistical analysis. For the same reason, effect sizes on biomass, abundance, richness, and diversity were included but effect sizes on biological metrics which were reported rarely had to be excluded (e.g. evenness, fish egg survival rates).

The total number of effect sizes for the three organism groups and four biological metrics was considerably higher than the number of projects, since data on more than one organism group and different biological metrics were reported in most of the studies (**Table 2-2**). In the meta-analysis, the effect of restoration on the number of individuals and the number of taxa was investigated separately since density and diversity reflect very different aspects of biological assemblages. The biological metrics were grouped accordingly (biomass/abundance and richness/diversity). Effect sizes which were extracted from the same project, for the same organism group, and the same metric group were combined by calculating mean values to avoid pseudo-replication (e.g. effect sizes of different diversity indices for invertebrates from one project were combined). This resulted in a similar number of unique effect sizes from peer reviewed literature (n = 132) and the original monitoring data (n = 265), and a total number of 397 effect sizes (**Table 2-2**).

Table 2-2: Data sources, number of publications, projects, and effect sizes.

	Peer-reviewed literature	**Original monitoring data**
Publications	74	-
Projects	87	64
Effect sizes	216	299
Unique effect sizes (per project, organism group, metric group)	132	265

The effect sizes were calculated using the time-variant response ratio

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

since time between implementation of the measures and monitoring was used as one of the predictor variables to investigate if the effect size increased with project age.

Furthermore, effect sizes were calculated according to the monitoring design of the project. For projects with a Before/After design (BA), the post-restoration value (after) was divided by pre-restoration value (before), and for projects with a Control/Impact design (CI), the value of the restored reach (impact) was divided by the value of the unrestored reach (control). In case the projects had a full BACI design, the additional information on the state of the control reach prior to restoration was used to adjust the CI and BA values similar to the so-called gain-score for the variance based effect sizes (Lipsey and Wilson 2001). For the CI design, the difference between the restored and unrestored reach prior to restoration was considered by adding the pre-restoration difference ($\bar{X}_{Tb} - \bar{X}_{Cb}$) to the post-restoration value of the unrestored reach $\bar{X}_{Ca}$ (**Figure 2-1**). This is equivalent to correct the post-restoration values by subtracting out any difference prior to restoration. Similarly, the control difference ($\bar{X}_{Ca}$ -$\bar{X}_{Cb}$) was added to the pre-restoration value of the restored reach $\bar{X}_{Tb}$ for the BA design, which is equivalent to correct the BA values of the restored reach by subtracting out any trend over time in the control reach.

Several studies only reported single values for the treatment and control reach respectively, among other reasons because sampling methods only allow calculating one single value, e.g. when an electrofishing sample covered the whole reach. For these studies, the response ratio was calculated based on single values for the treatment and control reach, respectively.

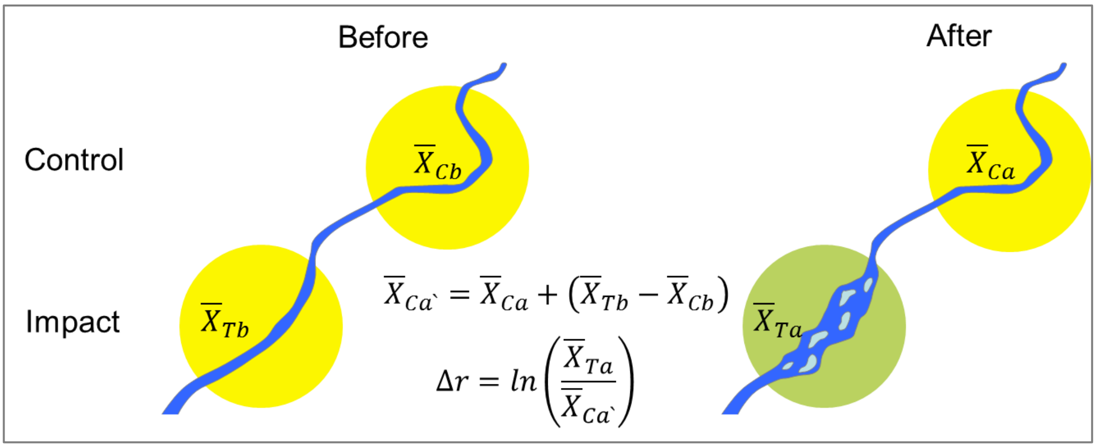


Figure 2-1: Considering the full BACI design for calculating adjusted CI effect sizes.

In meta-analysis, generally, it is not appropriate to combine effect sizes derived from studies with different research designs but it is possible to analyse them separately and compare the results between designs (Lipsey and Wilson 2001). Most of the 397 effect sizes (n = 353, 88.9%) were based on a CI design and only 86 (21.7%) had a BA design (note that percentages sum up to >100% since CI as well as BA effect sizes were calculated for projects with a full BACI design). Since the effect sizes were grouped in sub datasets (grouped by organism group and metric group) which had a sample size too low for statistical analysis for the BA design, the study was restricted to the CI effect sizes. Most restoration measures were included in the CI as well as in the BA dataset, but some restoration measures can, in practice, only be investigated using a BA design since control reaches hardly can be defined, especially flow restoration. Therefore, some restoration measures were not covered by this meta-analysis. In the CI dataset, CI effect sizes which were derived from projects with a full BACI design (n = 39) were adjusted / corrected for the pre-restoration difference as described above to make use of the important information provided by the full BACI designs.

For the following analyses, the CI dataset with n = 353 effect sizes from 120 projects was split in six sub datasets according to the organism group and the biological metric group investigated (**Table 2-3**). Since this resulted in n = 6 multiple comparisons, p-values were adjusted using the adjustment method of Benjamini and Hochberg (1995) if appropriate. It controls the false discovery rate (expected proportion of false discoveries amongst the significant tests) and hence, is less conservative compared to the family-wise error rates (e.g. Bonferroni correction) which correct p-values based on the total number of tests (not only the significant tests).

Table 2-3: Sample sizes of sub datasets used for the meta-analysis.

	Metrics based on individuals	**Metrics based on taxa**
Fish	75	54
Macroinvertebrates	69	75
Macrophytes	40	40

2.1.3 Quantifying catchment, river, and project characteristics potentially influencing restoration success (predictors)

The biological data were complemented by information on catchment, river, and project characteristics which potentially influence (either constrain or enhance) the effect of river restoration. In case these information were not reported in peer-reviewed literature or given by project partners, they were requested from the authors or extracted and cross-checked in Google-Earth (location of restored reach, reach land use, river width).

Catchment land use

The percentage coverage of different land use categories in the catchment upstream of the restoration projects was quantified.

Catchment borders were delineated using the global river network provided in the HydroSheds database (http://hydrosheds.cr.usgs.gov/index.php) and the digital elevation model available on Google Maps (using the hillshade option). For European restoration projects, the more detailed river network and watersheds provided by the ECRINS dataset (http://projects.eionet.europa.eu/ecrins) were used in addition.

Table 2-4: Grouping of the Global Land Cover 2000 (GLC2000) land use classes.

Land use category	GLC 2000 classes
Urban	22 Artificial surfaces and associated areas
Agriculture	16 Cultivated and managed areas 17 Mosaic (cropland, tree cover, other natural vegetation 18 Mosaic (cropland, shrub or grass cover)
Grassland	13 Herbaceous cover, closed-open
Shrub	9 Mosaic (tree cover, other natural vegetation) 10 Tree cover, burnt 11/12 Shrub cover, closed-open, evergreen and deciduous
Forest	1-8 Tree cover ,broadleaved, needle, mixed, regularly flooded
Natural other	14 Sparse herbaceous or sparse shrub cover 15 Regularly flooded shrub and/or herbaceous cover 19 Bare areas 21 Snow and ice
Water	20 Water bodies

Catchment size (km^2) and percentage coverage of the different land use categories were calculated in ArcGIS by grouping the land use classes of the Global Land Cover 2000 data (**Table 2-4**, http://bioval.jrc.ec.europa.eu/products/glc2000/products.php), which is a grid-based global land use dataset with a resolution of 30 arc seconds, corresponding to grid cells with a resolution of about 1000 m in North-South direction and 450-850 m in East-West direction.

River characteristics

Information on the following river characteristics were extracted for each restoration project:

Reach land-use adjacent to the restored reach was categorized as: urban, agriculture, forest, near-natural. Global land use data (e.g. GLC2000) could not be used since the resolution of these datasets was too low to quantify small-scale reach land use.

River types were classified based on the dominant bed substrate (similar to the bed-material calibre types in the river typology developed in REFORM in WP2): step-pool boulder bed streams, gravel-bed rivers, mixed gravel / sand rivers, sand-bed rivers, loess-loam dominated rivers, organic substrate dominated rivers.

In addition, the restoration projects were assigned to one of the four regions based on the location (coordinates): North America, Europe, Australia, North Asia.

Project characteristics

Information on the following project characteristics were extracted for each restoration project:

The restoration measures were classified and single measures were grouped according to the catalogue of measures developed in WP1 (**Table 2-5**): water flow quantity, sediment flow quantity, flow dynamics, longitudinal connectivity. The morphological measures were grouped based on the lateral extent and hence, space needed as instream, riparian, channel-planform, and floodplain measures. With an increase in lateral extent, the space needed to implement these measures, and hence, the conflicts of use increase. These eight groups are referred to as "river compartments" in the following.

As far as possible, the main measure which was implemented in each restoration project was identified, i.e. projects were assigned to one single main measure (**Table 2-5**). This was straightforward for many projects since only 1-2 measure categories and 1-3 single measures have been implemented in about two thirds of the projects. Moreover, the biological monitoring was often directed towards one specific measure (especially in peer-reviewed literature), and hence monitoring results were considered better suited to quantify the effect of these specific measures than of the respective measure combinations. Projects where different measures were implemented without a prominent single measure were classified as having applied multiple-measures. Nevertheless, it has to be noted that more than one single measure has been applied in most projects (78.3%) and restoration projects were not comparable to rigorous scientific experiments, and hence, it was generally not possible to disentangle the effect of multiple measures and to partition restoration effects, which hampers to clearly distinguish the effectiveness of single measures.

The restoration extent was quantified using the restored reach length. In addition project size was calculated following Miller et al. (2010) as ratio of restored reach length to bankfull width. Alternatively, project size could be quantified calculating the ratio of restored reach length to the length of the upstream river network but an appropriate river network covering also the small restored streams was not available. Moreover, the share of the river compartments addressed by the measures was calculated, assuming that projects which implemented measures in different river compartments (e.g. restored in-channel features, channel-channel-planform, and river continuity) can be considered "larger" projects with a larger extent compared to projects which only addressed one single river compartment (e.g. only restored in-channel features).

Table 2-5: Classification and grouping of the restoration measures according to the catalogue of measures developed in WP1, main measures are given in bold.

Measure category (river compartment)	Single measures
Water flow quantity	Reduce abstraction without return, improve water retention, reduce groundwater abstraction, increase minimum flow, water diversion for quantity, recycle used water, improve water storage, reduce water consumption
Sediment flow quantity	Add sediment, reduced undesired sediment input, prevent sedimentation of reservoirs, trap sediments, reduce impact of dredging, improve continuity of sediment transport
Flow dynamics	Establish environmental flows, modify hydropeaking, increase flood frequency in floodplain, shorten impoundments, favour morphogenic flows, reduce anthropogenic peak flows (summarized as **flow restoration**)
Longitudinal connectivity	Facilities for upstream migration, facilities for downstream migration, manage facilities for migration, modify culverts and siphons, remove barriers (summarized as **river continuity**)
Morphology instream (bed and bank)	Remove bed fixation, remove bank fixation, remove sediments (e.g. mud), add sediments (e.g. gravel), remove/modify hydraulic structures, , initiate natural channel dynamics, **weed management, river margins enhancement, large wood placement, riffle creation or enhancement, boulder placement, bank stabilization**
Morphology riparian	Buffer strips to reduce nutrients, buffer strips to reduce sediment, develop natural riparian vegetation (summarized as **riparian buffer development**)
Morphology channel-planform	Shallow/increase bed level, narrow over-widened channel, create low flow channels, initiate lateral channel dynamics, create secondary floodplain, **remeandering**, **widening/rebraiding**
Morphology floodplain	Lower embankments, back-removal of embankments, remove embankments, develop floodplain vegetation, **oxbow creation or re-connection**

2.1.4 Statistical analysis

Bivariate relationships were investigated using standard statistical tests: Welch's t-test and standard t-test to test for differences between two mean values with non-equal and equal variances, respectively. One-way ANOVA to test for differences between means between more than two groups using the Tukey HSD and Dunnett's T3 as post-hoc test for equal variances and non-equal variances, respectively (which was checked with the Levene's test for homogeneity of variance). Pearson's correlation was used to test for significant linear relationships but scatterplots were also visually checked for non-linear relationships and quantile regression.

Regression trees were used to model and predict the effect sizes using all predictors, and to investigate interactions between the predictor variables. Regression trees split the dataset in sub-datasets using one single predictor variable at each split, trying to maximise the difference between the sub-datasets in respect to the response variable, i.e. to maximize the between-groups sum-of-squares in a simply analysis of variance. Regression trees have several advantages: they can capture interactions and non-linear relationships, can handle both, continuous and categorical predictor variables, and are insensitive to outliers. However, they also have some disadvantages: difficulties capturing linear relationships, sensitive to changes in the dataset, and interpretation is difficult if predictors are co-correlated. Regression trees area usually pruned and only splits which significantly increase the performance of the statistical model are included to avoid overfitting. Although the dataset contained all available monitoring results, sample size of the sub-datasets were relatively small and rigorously pruning the trees according to standard procedures would have resulted in single splits only. Therefore, more relaxed pruning criteria were used to identify trends which are not statistically significant but of interest for river management. However, due to this relaxed pruning criteria, regression trees have to be interpreted with care.

Boosted regression trees (BRT) were used to quantify the relative importance of the predictor variables. A boosted regression tree model consists of a sequence of single regression trees, where each successive tree after the first one is built using the residuals of the preceding tree. By using the residuals, the preceding trees focus on the cases which were most difficult to classify. "*The final BRT model is a linear combination of many trees (usually hundreds to thousands) that can be thought of as a regression model where each term is a tree. The relative importance of the predictor variables is calculated based on the number of times a variable is selected for splitting, weighted by the squared improvement to the model as a result of each split, and averaged over all trees. The relative influence (or contribution) of each variable is scaled so that the sum adds to 100, with higher numbers indicating stronger influence on the response.*" (Elith et al. 2008, p. 804, 808). In BRT, only a random sub-dataset is used to develop the regression trees in each model run. This stochasticity improves predictive performance but also results in slight differences between model runs. Therefore, 10 replicate model runs were used to calculate the mean relative importance of the predictor variables and to ensure that differences in predictor importance were larger than this methodological variance.

For some analyses, effect sizes of the six sub datasets were merged to describe specific aspects like the overall restoration effect. This resulted in a pseudo-replication since the merged datasets included more than one effect size for some of the projects (i.e. not all samples were statistically independent), and hence no statistical tests were applied for these few analyses.

As in many meta-analyses, the dataset was rather heterogeneous. Subdividing it in sub-datasets resulted in relatively small sample sizes for some analysis. Therefore, to avoid a Type II error (incorrectly accepting the null hypothesis), and to identify trends in the dataset which are relevant and of interest for river management, a significance level of 0.1 was used in addition to the level of 0.05.

All statistical analysis were performed in “R” (3.0.2) using the following additional packages: car for Levene's test for homogeneity of variance, DTK for Dunnett’s T3 post-hoc test, rpart and rpart.plot for the regression tree analysis and for plotting the trees as well as gbm and dismo for the boosted regression tree analysis.

2.2 Results

2.2.1 Description of restoration projects and measures

The dataset contained 120 restoration projects and covered a wide range of different rivers and projects (Figure 2-2). Different river types were included. While most of the projects were implemented in gravel-bed rivers (55.0%), other river types like sand-bed rivers (13.3%) and mixed gravel-sand river (8.3%) were also covered. The size of the rivers differed, with a 10-90th percentile range of river width and catchment size of 2.0 m - 36.2 m and 6.4 km^2 - 2413.3 km^2, but few large rivers with a catchment size larger than about 10,000 km^2 were included in the dataset (n = 4). Land use pressure in the catchments greatly differed, with a 10-90th percentile range of agricultural land of 0.0% - 85.7% and urban land use of up to about 50%. Moreover, land use adjacent to the restored reach differed. Most projects were bordered by extensive or intensive agricultural areas (69.2%) but forested (15.8%) and urban reaches (13.3%) were also present in the dataset.

In some few projects long river reaches > 10 km in length have been restored but most restored reaches were rather short (10-90th percentile range 0.2 km - 2.6 km). Most of the projects in the database were implemented between 1991 and 2005 (project date) and project age (time between implementation of measures and monitoring) markedly differed with a 10-90th percentile range of 1 to 8 years.

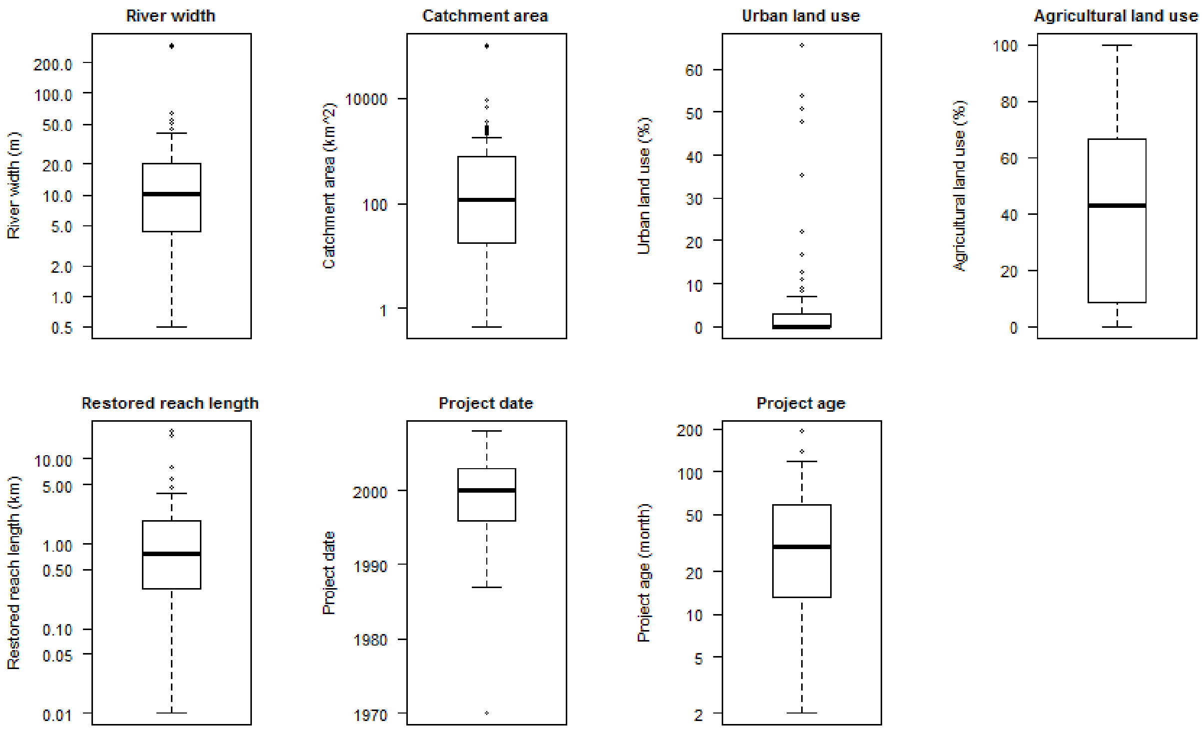


Figure 2-2: River and project characteristics (y-scale log-transformed, except for land use parameters).

A total of 353 effect sizes were extracted from 120 restoration projects. Instream measures have been implemented in most of the projects (83.3%), which either were the only measures applied (25.8%), implemented together with measures to enhance channel-planform (50.8%), the riparian area (20.8%), floodplain area (20.8%) or river continuity (9.2%) (**Figure 2-3**). Planform measures were also frequently applied (55.8%). Moreover, measures to enhance the riparian area (24.2%), floodplain area (23.3%), and river continuity (9.2) were implemented in a substantial part of the projects, whereas sediment, hydrological, and flow restoration measures were virtually missing in the dataset.

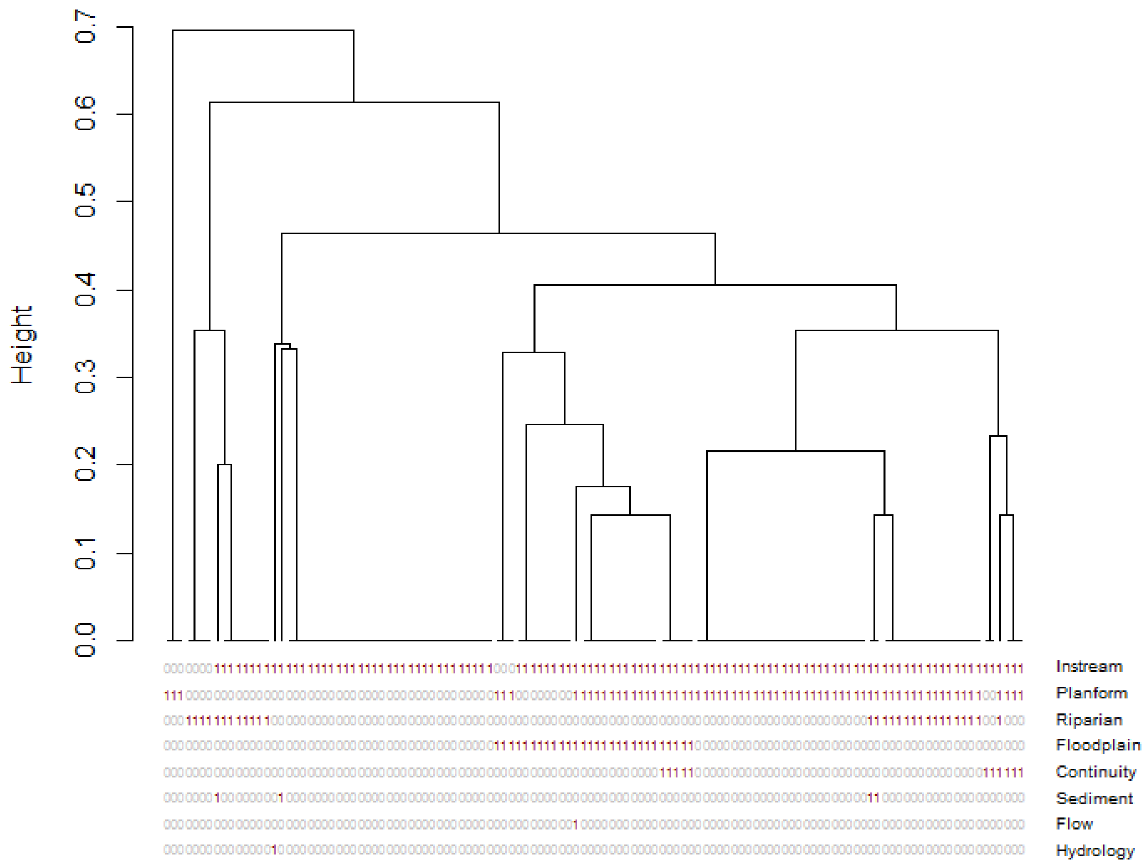


Figure 2-3: Restoration projects clustered by the 8 measure categories (river compartments) given in Table 2-5. Measure categories applied in the projects are indicated by a value of 1.

2.2.2 Methodological aspects

In meta-analyses solely based on peer-reviewed literature, effect sizes may potentially have an upward bias since studies showing an effect or even a high effect, generally have a higher probability of being published, also known as the publication bias. The dataset used in this meta-analysis contained effect sizes from peer-reviewed literature and original monitoring data. The latter not being prone to any selection bias since all effect sizes were included, irrespective of their direction or magnitude, and hence, can be considered to represent unpublished or grey literature. Therefore, the dataset offered the rare opportunity to investigate the publication bias in river restoration literature by comparing the effect sizes of the two data sources for fish and invertebrates (the number of effect sizes for macrophytes from peer-reviewed literature was too low for any significance test).

Effect sizes did not differ between the two data sources, except for the sub dataset on macroinvertebrate taxa (t-test, $t = -3.01$, $df = 73$, $p_{adj} < 0.05$) where the mean effect size extracted from peer-reviewed literature was higher (0.50) compared to the mean value of the original monitoring data (0.08). However, about two thirds (64.3%) of the effect sizes extracted from peer-reviewed literature in this sub dataset originate from projects where instream measures have been applied, whereas this share was significantly lower for the original monitoring data (9.5%, Chi-squared(1, n=78) = 20.91, $p < 0.01$). Since instream measures had a significant higher effect size in this sub dataset (**Figure 2-7**), the higher value for peer-reviewed literature was probably due to this co-correlation. This indicated that there was no or only a small publication bias for invertebrate taxa in river restoration literature.

2.2.3 Quantifying restoration effect

Overall effect of restoration on biota

Considering all three organism groups (fish, macroinvertebrates, macrophytes), the overall effect of restoration on biota was positive, although effect sizes varied greatly and a substantial part of the projects did not enhance or even further degrade biota. The mean effect size for metrics based on taxa and individuals was +0.25 and +0.51, which is equivalent to a $\bar{X}_T$ / $\bar{X}_C$ ratio of 1.28 and 1.67, respectively. Variability was considerably high with a 10-90th percentile range of the effect sizes of -0.25-0.91 (taxa) and -0.76-1.80 (individuals), respectively. While most effect sizes indicated a positive effect (67.5% for taxa, 71.2% for individuals), about one third showed no or a negative effect

Effect of restoration on different biological and metric groups

The effect of restoration differed between biological and metric groups (**Figure 2-4**). Restoration had a higher effect on compared to fish and invertebrates. Moreover, restoration had a higher effect on the number of fish and invertebrate individuals than on the number of taxa but no such differences were detectable for macrophytes, for which both (abundance and number of taxa) were markedly increased by restoration.

The mean effect size based on the number of taxa (richness, diversity) significantly differed between organism groups (One-way ANOVA, $F_{2/166} = 14.50$, $p < 0.01$) and was significantly higher for macrophytes (0.57) compared to the mean effect size for fish (0.18) and invertebrates (0.12) (Dunnett's T3 $p < 0.01$). In contrast, the mean effect size based on the number of individuals (biomass, abundance) did not differ between the three organism groups.

The mean effect size of the individuals based metrics was significantly higher compared to those based on the number of taxa for fish and weakly significantly higher for invertebrates (for fish 0.58 compared to 0.18, Welch's t-test, $t = 3.44$, $df = 108$, $p < 0.01$, for invertebrates 0.35 compared to 0.12, Welch's t-test, $t = 1.65$, $df = 82$, $p < 0.10$). These results indicated that, in general, it is easier to increase the number of individuals than increasing the number of.

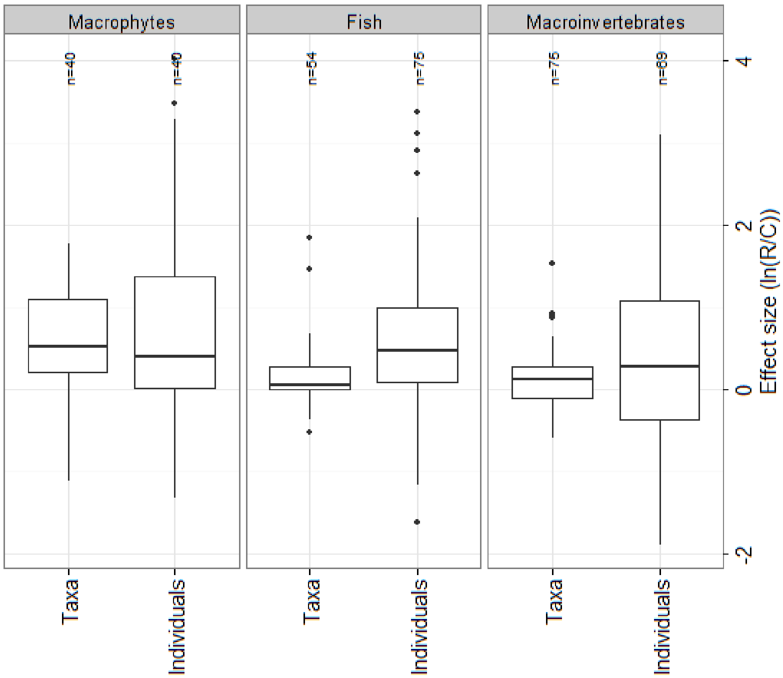


Figure 2-4: Effect sizes of biological and metric groups (six sub datasets).

Effect of different restoration measures

The higher effect of restoration on the number of macrophyte taxa compared to fish and invertebrates (**Figure 2-4**) was mainly due to the fact that macrophytes especially benefited from river widening/rebraiding and remeandering measures.

The effect sizes of taxa based metrics like richness and diversity for macrophytes were significantly higher for the widening/rebraiding (One-way ANOVA, $F_{2/38} = 11.20$, $p < 0.01$, Dunnett's T3 $p<0.05$) and remeandering measures (One-way ANOVA, $F_{2/55} = 8.51$, $p < 0.01$, Dunnett's T3 $p < 0.05$) compared to fish and invertebrates (**Figure 2-5**). These measures were obviously better suited to increase the number of macrophyte taxa and less suited for fish and invertebrates, which is reasonable since these measures usually decrease flow velocity and water depth. Widened and remeandered reaches are often sparsely shaded in the beginning, which favours the establishment of macrophyte species. In addition, remeandering had a significant higher effect on macrophyte abundance compared to invertebrates (One-way ANOVA, $F_{2/58} = 3.75$, Tukey HSD $p < 0.05$) and differences were weakly significant for fish (Tukey HSD $p = 0.10$) (**Figure 2-5**). In contrast, individuals based effect sizes of the different organism groups did not differ significantly for widening/rebraiding. This may be due to the lower nutrient loads in widened mountain rivers compared to the nutrient rich lowland meandering streams which limits the growth and abundance of macrophytes.

Differences between the effects of other restoration measures on the three organism groups are also shown in **Figure 2-5** but sample sizes were too low to test for significant differences.

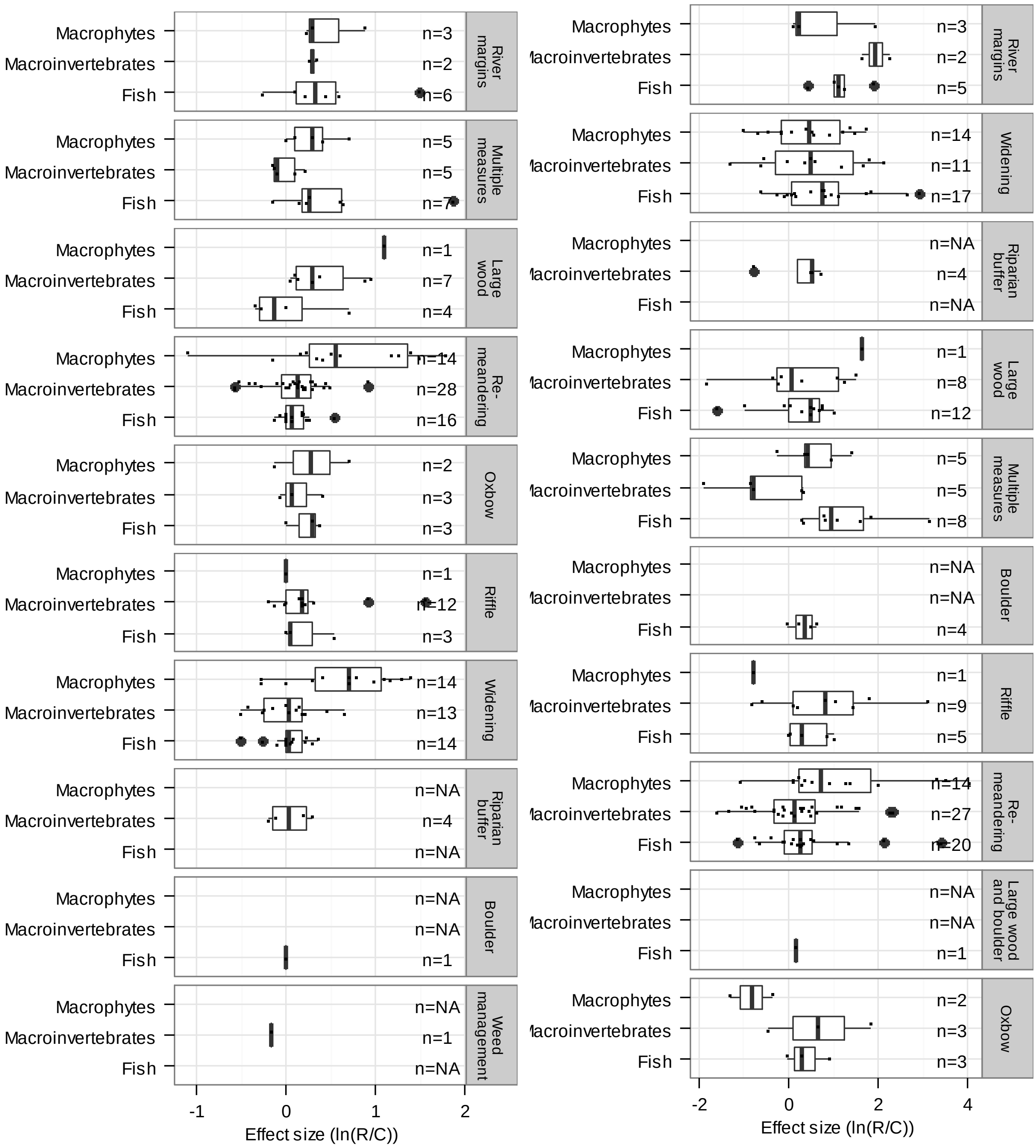


Figure 2-5: Effect sizes based on taxa (richness, diversity, left) and individuals (biomass, abundance, right) of the different organism groups for different main measures. Measures are ordered in the two figures according to the median effect over all organism groups (increasing from bottom to top), and not all measures are given due to data availability.

Restoration measures had a similar effect on biota but instream measures had a higher effect on the number of invertebrate taxa compared to planform and riparian measures (e.g. widening and riparian buffer) and a similar tendency was observed for fish.

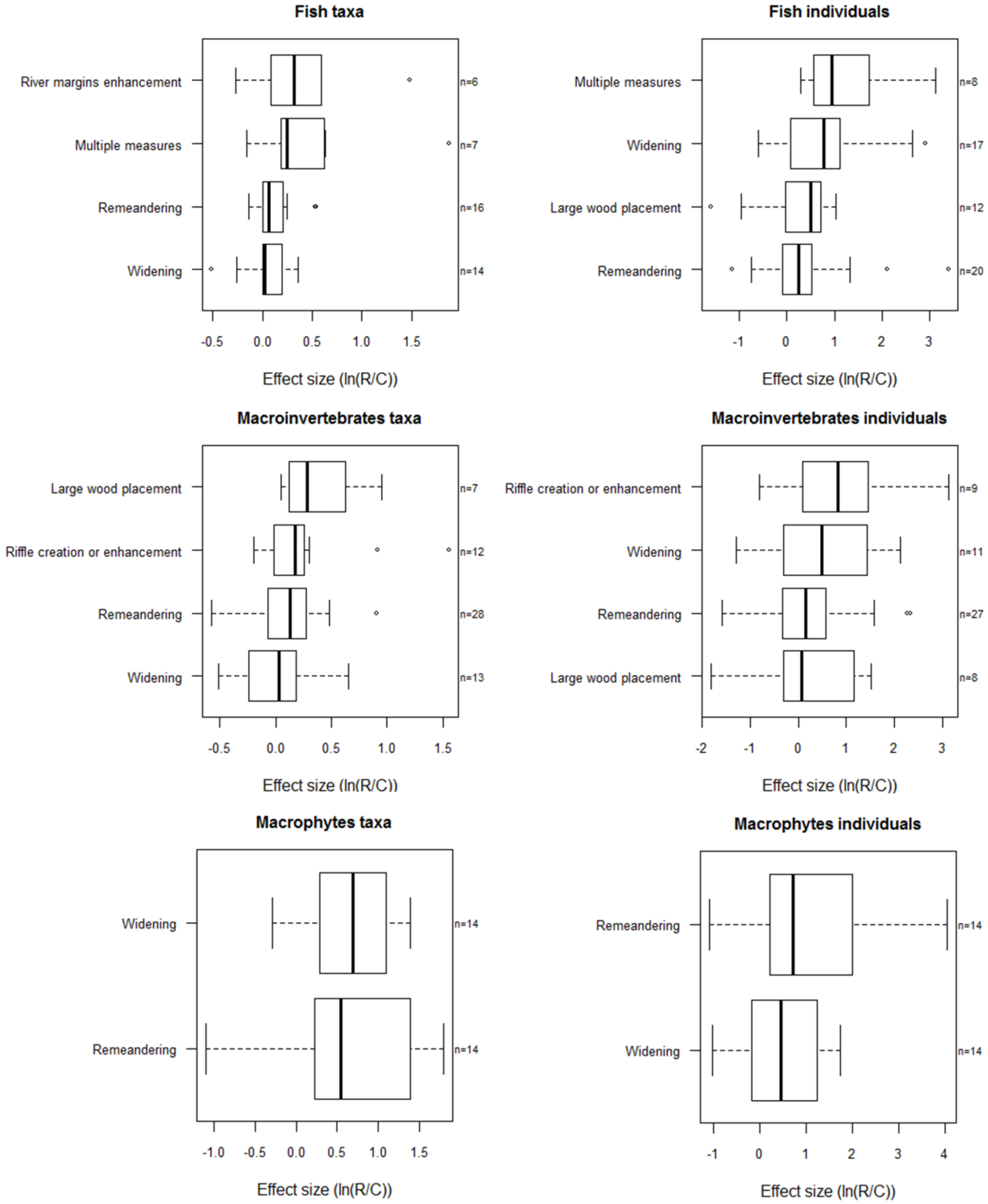
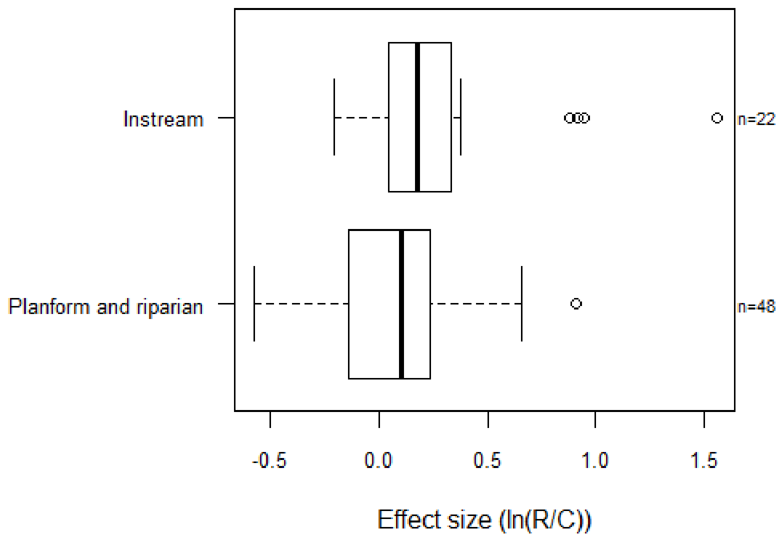


Figure 2-6: Effect of different restoration measures on the different biological and metric groups (six sub datasets), measures were ordered according to the median effect size, measures with n < 6 were excluded.

There were no significant differences between the mean effect sizes of the different measures in the six sub datasets, except for taxa based metrics of fish (One-way ANOVA, $F_{3/39} = 3.39$, $p < 0.05$, measures with $n < 6$ effect sizes excluded, **Figure 2-6**). However, the difference was not significant after p-value adjustment for multiple comparisons ($p_{adj} = 0.15$). For the taxa-based metrics of fish and invertebrates, there was a tendency for instream measures like river margins enhancement and large wood placement to perform better compared to channel-planform and riparian measures like widening/rebraiding and remeandering (Fish taxa and Macroinvertebrate taxa in **Figure 2-6**). Therefore, measures were grouped accordingly as "instream measures" (river margins enhancement, large wood placement, boulder placement, riffle creation or enhancement) and "planform and riparian measures" (widening, remeandering, oxbow creation or re-connection, riparian buffer development). After grouping, instream measures had a significantly higher mean effect size compared to planform and riparian measures for the taxa-based metrics of invertebrates (t-test, $t = -2.58$, $df = 68$, $p < 0.05$) (**Figure 2-7**), which was still significant after p-value adjustment for multiple comparisons ($p_{adj} < 0.05$). These results (i) indicated that the increase in number of invertebrate taxa was higher when instream measures were applied while measures restoring channel-planform and riparian buffers were less effective, and (ii) they might be considered an indication that instream measures were also more effective in increasing fish diversity.

Figure 2-7: Effect of instream compared to planform and riparian measures on taxa based metrics of macroinvertebrates.



2.2.4 Catchment, river, and project characteristics (predictors) affecting restoration success

Catchment land use

Agricultural land use in the upstream catchment adversely affected restoration effect on the number of fish individuals. No other land use category (urban, grassland, shrub, forest, natural other, water) had a detectable effect on restoration success in any of the sub datasets, including non-linear relationships.

The effect sizes for fish metrics based on the number of individuals decreased with an increasing percentage coverage of agricultural land use in the upstream catchment (Pearson, $r = -0.35$, $p_{adj} < 0.05$, $n = 75$, **Figure 2-8**) This relationship potentially could have been solely due to the restoration projects for fish implemented in North-Western USA, which typically were located in forested catchments and had a large effect on fish biomass and abundance. However, effect sizes for fish were also negatively correlated to the percentage coverage of agricultural land use if North-American projects were excluded (Pearson, $r = -0.41$, $p_{adj} < 0.05$, $n = 58$).

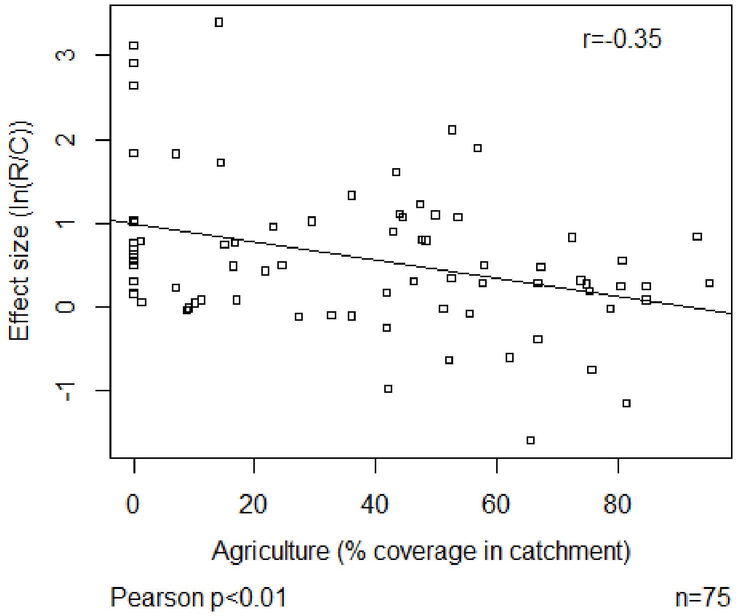


Figure 2-8: Negative effect of agricultural land use on restoration success on the number of fish individuals.

In small catchments, even the most upstream areas can potentially influence the restored reach, whereas the distance to remote areas in large catchments might be too large to affect conditions in the restored reach. Therefore, restoration projects with large catchments >1000 km^2, >500 km^2, and >250 km^2 were progressively excluded from the sub datasets to investigate if catchment land-use effects depend on catchment size. However, the effect of agricultural land use on fish metrics based on the number of individuals remained the only significant relationship.

River reach characteristics

Reach land use adjacent to the restored river reach had no significant effect, but there was a tendency of intensive agriculture to affect restoration success of instream measures.

Effect sizes did not significantly differ between different reach land use categories in the six sub datasets after p-value adjustment for multiple comparisons (reach land use categories with n < 7 were excluded). However, effect sizes of instream measures for fish and invertebrates tended to be higher in forested compared to agricultural reaches (**Figure 2-9**), but these differences were not even weakly significant, most probably due to the high variability and low sample size. This might be considered a first indication that for fish and invertebrates, the success of instream restoration was limited by agricultural land use and was higher in forested reaches that were not affected by direct input of point or diffuse pollution, shaded, and hence provide more favourable conditions for biota in general.

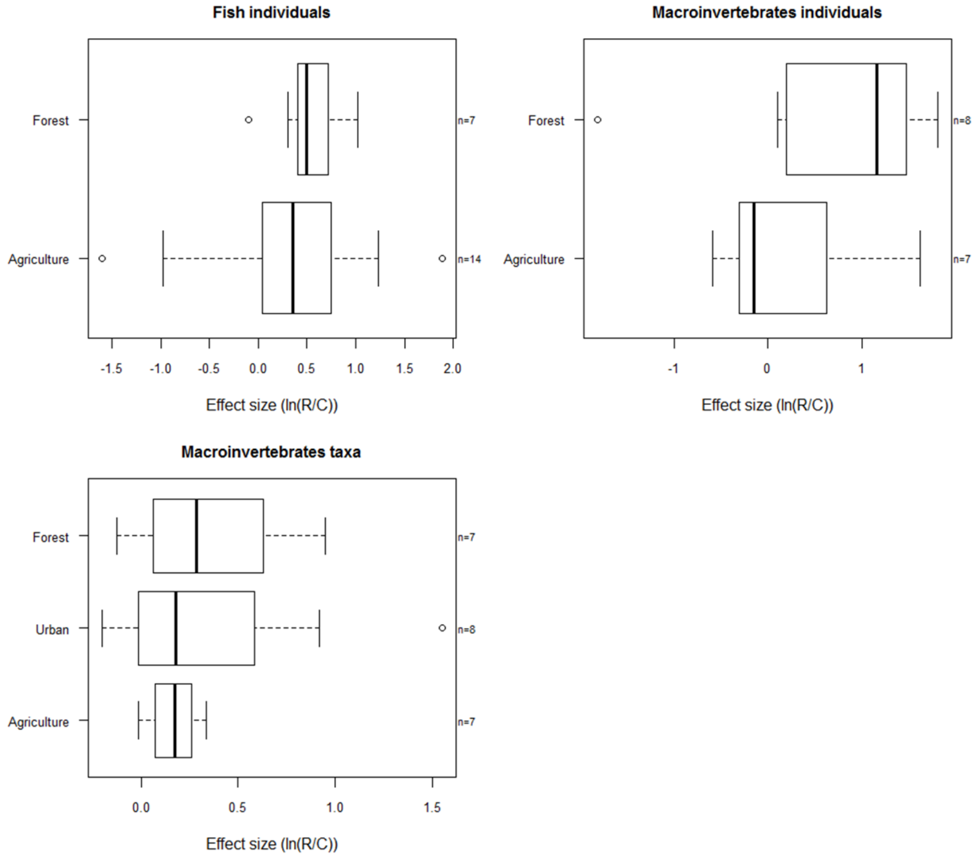


Figure 2-9: Effect of reach land use on the success of instream restoration.

River types influenced restoration success. Restoration projects in gravel-bed rivers had a higher effect on the number of fish and invertebrate taxa as well as fish individuals compared to projects in sand-bed rivers, where restoration had no or even a negative effect.

For fish, restoration effect on the number of individuals was higher in gravel-bed rivers compared to sand-bed rivers (One-way ANOVA, $F_{2/63} = 3.13$, $p < 0.05$, Tukey HSD

$p < 0.05$), with mixed gravel/sand bed rivers having intermediate effect sizes but differences between river types were non-significant after p-value adjustment for multiple comparison ($p_{adj} = 0.17$). However, mean effect sizes of gravel-bed rivers were higher compared to sand-bed rivers for fish metrics based on the number of individuals and taxa, when the analysis was restricted to these two river types but this was only weakly significant after p-value adjustment for multiple comparisons for fish individuals (t-tests with $p_{adj} < 0.1$). Macroinvertebrates showed a similar but non-significant pattern (**Figure 2-10**) and median effect sizes approached zero or were even negative in sand-bed rivers. Differences were more pronounced if only projects were considered which mainly applied instream measures but sample sizes were too low to test for statistical differences.

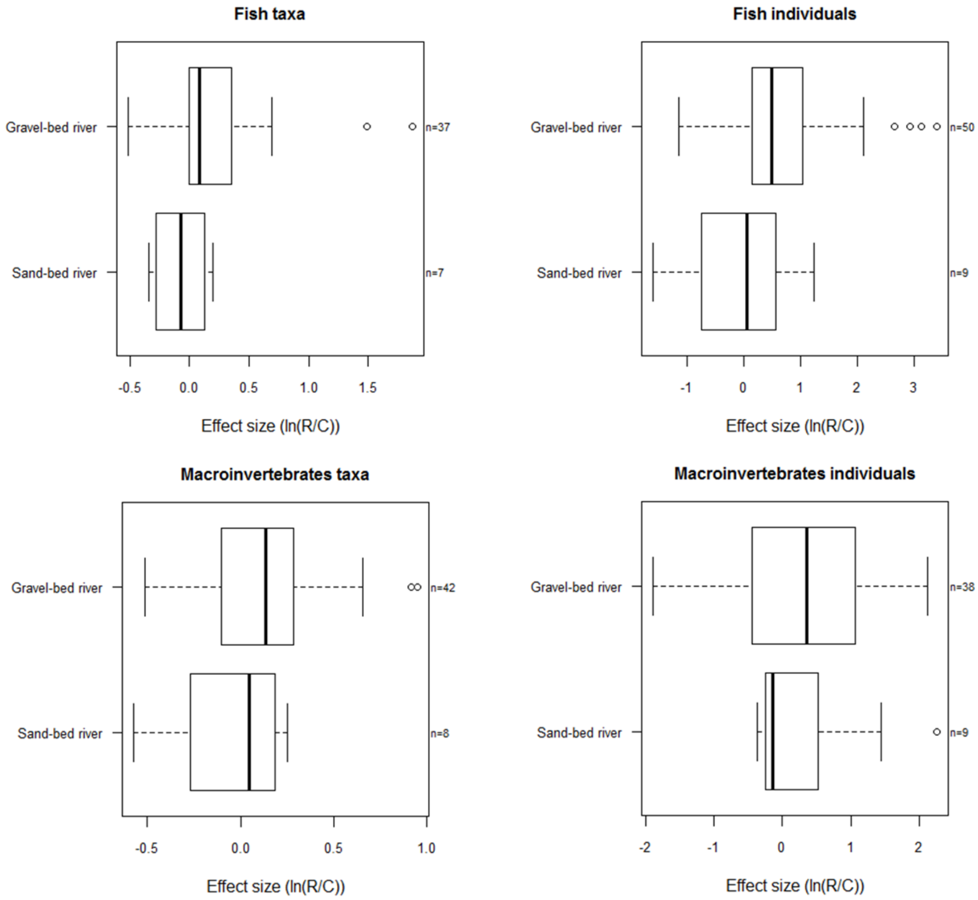


Figure 2-10: Effect of river type on the success of restoration on the number of fish individuals.

River size (width and catchment area) affected restoration success and the effect on the number of fish and invertebrates increased with river width.

For fish, the effect size of metrics based on the number of individuals increased with river width (**Figure 2-11**) and this correlation was significant after p-value adjustment for multiple-comparisons (Pearson, $r = 0.32$, $p_{adj} < 0.05$, $n = 74$)

Catchment size had no significant effect on restoration success in none of the sub datasets, which indicates that river width is better suited as a parameter to describe the influence of river size on the restoration outcome.

Furthermore, restoration success did not differ between *regions* (Europe, North-America) for none of the sub datasets and even not prior to p-value adjustment for multiple-comparisons. This indicated that all other analyses were not affected by differences between regions.

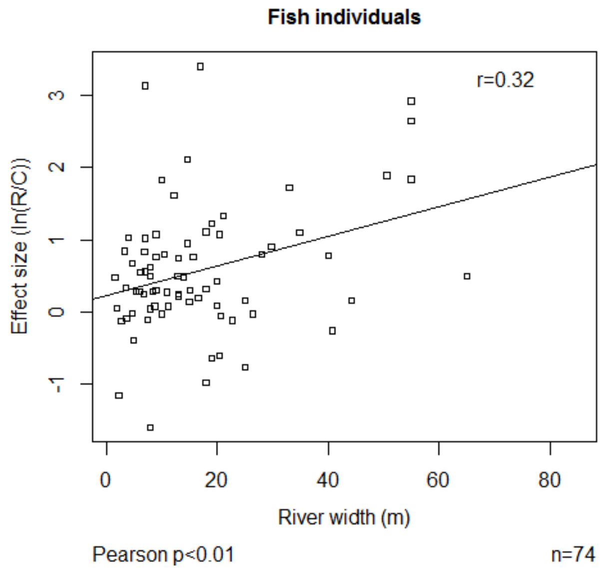


Figure 2-11: Effect of river width on restoration success of fish individuals.

Project characteristics

Restoration success did depend on *project age* (time between implementation of measures and monitoring) and results indicated that restoration success is decreasing over time for macrophyte abundance (**Figure 2-12**).

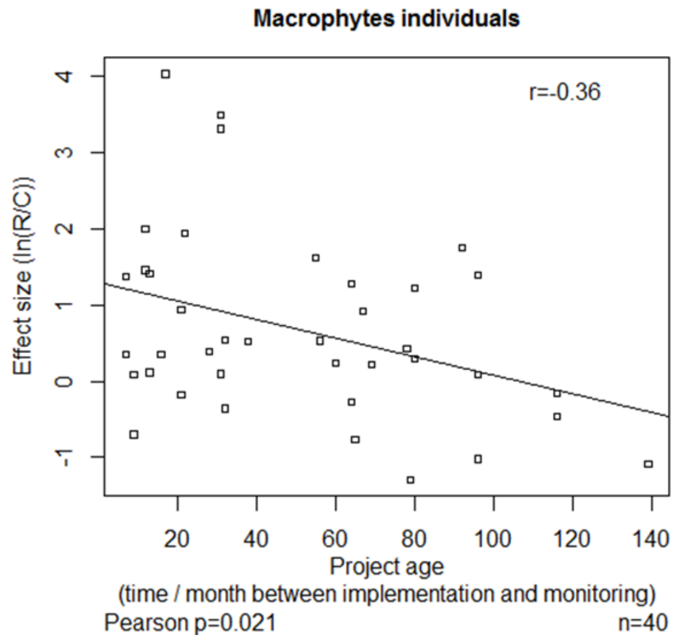


Figure 2-12: Effect of project age on restoration success of macrophytes individuals.

For macrophytes, effect sizes of metrics based on the number of individuals decreased with project age (Pearson, $r = -0.36$, $p < 0.05$, $n = 40$). This relationship was weakly significant ($p_{adj} = 0.09$) after p-value adjustment for multiple comparisons. Moreover, taxa metrics of macrophytes were weakly negatively correlated with project age but this relationship was non-significant after excluding one single outlier (Pearson, $r = -0.20$, $p_{adj} = 0.45$, $n = 39$). In the sub-dataset of macrophyte abundance, project age was highly co-correlated with project date (Pearson, $r = -0.76$, $p < 0.01$, $n = 40$), i.e. time between implementation and monitoring was lower for more recent projects. Therefore, the effect of project age might have been simply due to this co-correlation and the better performance of the most recent projects might have resulted from improved restoration skills over time. However, in the other sub-datasets, effect sizes decreased with project date, which was even weakly significant after p-value adjustment for fish and invertebrate taxa, as well as invertebrate individuals. This would imply that restoration skills worsened for fish and invertebrates. Assuming that restoration skills either increase for all or none of the organism groups, the results rather indicated that restoration effect on macrophytes was higher in the first years after restoration and decreased in the following years, possibly because the most effective measures for macrophytes like widening and remaindering created sparsely shaded, slow flowing shallow areas, but habitats changed to less favourable conditions when channel-features were maturing (e.g. development of riparian vegetation and shading), and hence abundance of the macrophyte species decreased.

Restoration extent (restored reach length, project size, river compartments addressed by measures) had no significant positive effect on restoration success in any of the sub datasets. In general, the length of the restored reaches might differ between different types of measures. For example, widened or remeandered reaches might be generally longer compared to reaches where only instream measures like large wood placement have been applied. However, restoration extent did also have no positive effect in none of the sub datasets if planform/riparian and instream measures were analysed separately. As discussed in more detail in section 2.3, restored reaches were rather short and length might have simply been below a critical threshold to increase restoration success.

2.2.5 Predictor interactions and relative importance

Predictor interactions in sub datasets

Based on the previous results (section 2.2.4), the following variables which significantly affected or at least showed a tendency to influence restoration success in the six sub datasets were selected to investigate interactions between these predictors: Measure group (instream vs. planform and riparian), the percentage coverage of agricultural land use in the upstream catchment (Agriculture), land use adjacent to the restored reach (Reach Use), river type (e.g. gravel or sand-bed rivers), river width (meter), and project age (time between implementation of the measures and monitoring in month).

The regression trees in **Figure 2-13** can be used to identify conditions which favour restoration success. For example, the effect of restoration on the number of invertebrate taxa was highest in projects where instream measures had been applied at least about two years prior to the monitoring (MIV taxa in **Figure 2-13**). In general, the important predictors in the regression trees also significantly affected restoration success in the bivariate analysis (section 2.2.3 and 2.2.4), like the measure group, which was weakly significantly related to the effect sizes of the macroinvertebrate taxa sub dataset (**Figure 2-7**).

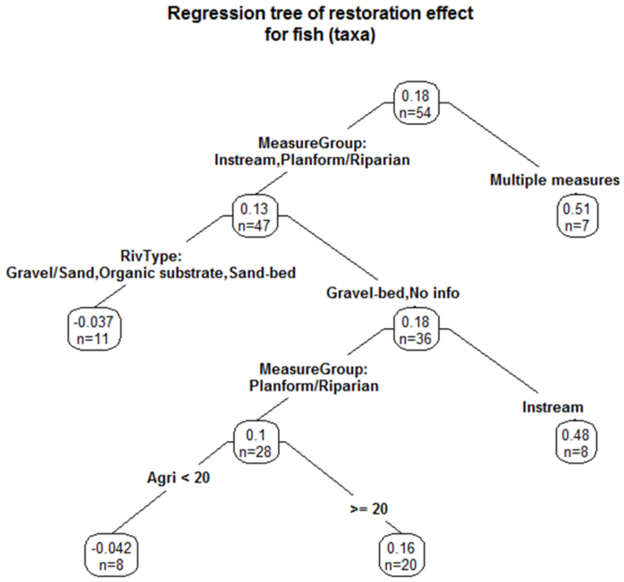
However, in two of the sub datasets (fish taxa, macroinvertebrates individuals), interpretation of the results was difficult since predictors were co-correlated. The second-best predictors at the first split had a similar predictive power compared to the best predictors (**Table 2-6**) but had a low predictive power at the consecutive splits due to the co-correlation with the best predictor, and hence, did not show up in the regression trees.

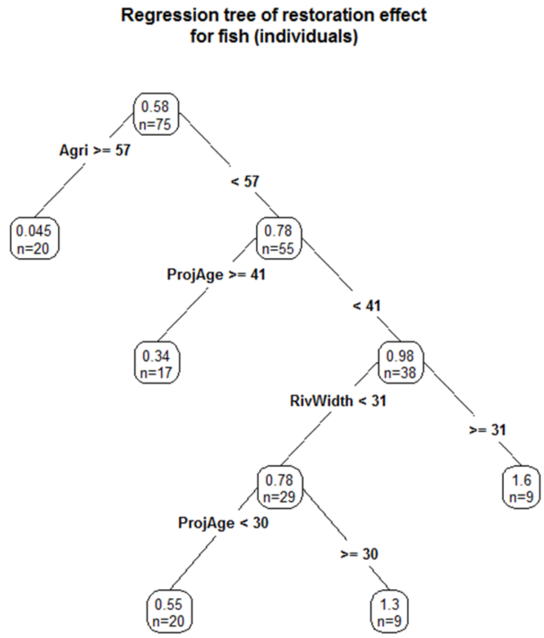
In the fish taxa sub dataset, restoration success was highest (0.48) for instream measures in gravel-bed rivers (after splitting a small group of effect sizes of projects which applied a very diverse set of measures, classified as multiple-measure projects). Reach use was a similar good predictor compared to the measure group and hence, a similar high restoration success (0.46) was observed in a small group of n = 9 effect sizes derived from urban restoration projects. However, since high relative effect sizes in urban restoration projects possibly result from small absolute increases (see section 2.1.1), the regression tree shown in **Figure 2-13** is probably best suited to identify favourable conditions for restoration success in this sub dataset.

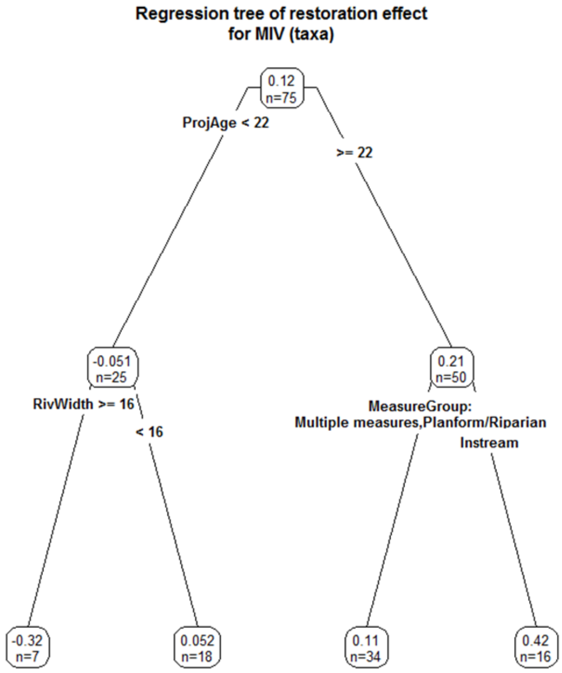
In the macroinvertebrate individuals dataset, restoration success was highest (1.1) in projects of moderate age. River width was an only slightly weaker predictor which did not show up at consecutive splits. There is no obvious reason how river width could be causally linked to restoration success in contrast to project age and hence, the regression tree shown in **Figure 2-13** is probably best suited to identify favourable conditions for restoration success in this sub dataset.

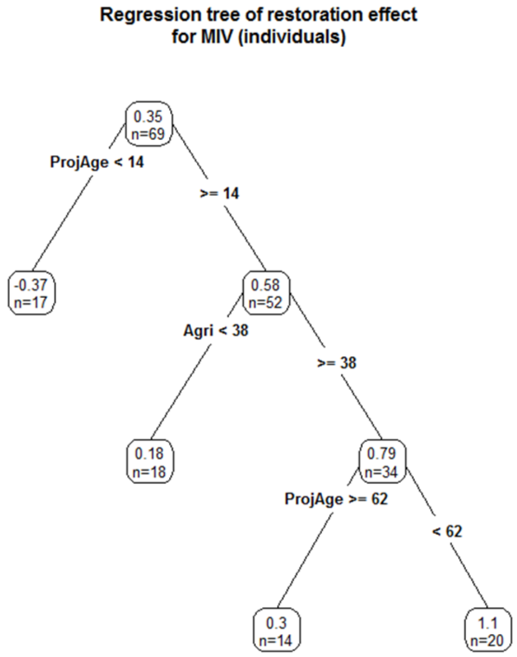
Restoration success in the other sub datasets (fish individuals, macrophytes taxa and individuals) was mainly driven by catchment land use and project age. For fish individuals, it was highest in relatively large rivers and catchments with a low share of agricultural land use. For macrophyte taxa and individuals, it was highest in catchments with a low share of agricultural land use and in addition for macrophyte abundance, in relatively young projects.

In all six sub datasets, the total variance explained by the predictor variables in the regression trees was only about one third (**Table 2-6**). Moreover, it has to be noted that regression trees were not pruned to only show statistically significant splits. Therefore, these results have to be interpreted with care and regression trees show the tendency which condition favour restoration success but cannot be used to predict it. Adaptive management approaches still have to be applied to monitor the effect of the restoration measures and to adapt the specific projects if necessary.


Regression tree of restoration effect
for fish (taxa)
0.18
n=54
MeasureGroup:
Instream,Planform/Riparian
Multiple measures
0.51
n=7
0.13
n=47
RivType:
Gravel/Sand,Organic substrate,Sand-bed
Gravel-bed,No info
-0.037
n=11
0.18
n=36
MeasureGroup:
Planform/Riparian
Instream
0.48
n=8
0.1
n=28
Agri < 20
>= 20
-0.042
n=8
0.16
n=20


Regression tree of restoration effect
for fish (individuals)
0.58
n=75
Agri >= 57
< 57
0.045
n=20
0.78
n=55
ProjAge >= 41
< 41
0.34
n=17
0.98
n=38
RivWidth < 31
>= 31
1.6
n=9
0.78
n=29
ProjAge < 30
>= 30
0.55
n=20
1.3
n=9


Regression tree of restoration effect
for MIV (taxa)
0.12
n=75
ProjAge < 22
>= 22
-0.051
n=25
RivWidth >= 16
< 16
0.21
n=50
MeasureGroup:
Multiple measures,Planform/Riparian
Instream
-0.32
n=7
0.052
n=18
0.11
n=34
0.42
n=16


Regression tree of restoration effect
for MIV (individuals)
0.35
n=69
ProjAge < 14
>= 14
-0.37
n=17
0.58
n=52
Agri < 38
>= 38
0.18
n=18
0.79
n=34
ProjAge >= 62
< 62
0.3
n=14
1.1
n=20

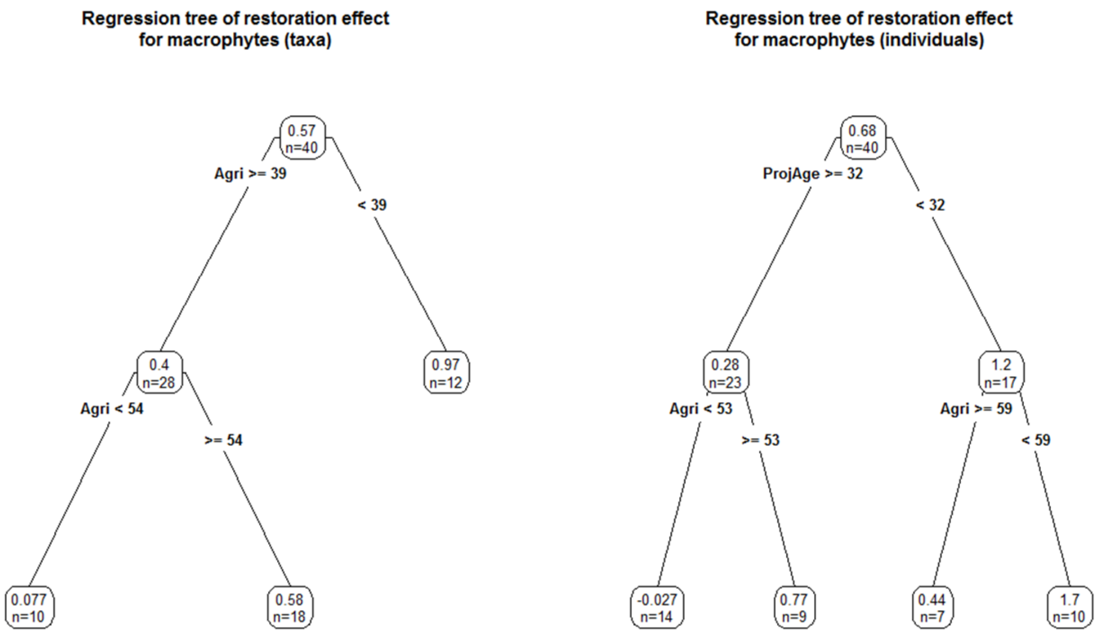


Figure 2-13: Regression trees on the interaction of predictors influencing restoration effect on fish, invertebrates, and macrophytes taxa and individuals based metrics.

Table 2-6: Predictive power of the best, second, and third-best predictor variables at the first split (in parentheses) as well as total variance explained by regression trees.

Sub dataset	Best predictor	Second-best predictor	Third-best predictor	Total variance explained
Fish (taxa)	Measure group (0.106)	Reach use (0.100)	River type (0.081)	0.29
Fish (individuals)	Agriculture (0.132)	River width (0.125)	River type (0.089)	0.35
Inverts (taxa)	Project age (0.111)	Measure group (0.095)	Agriculture (0.050)	0.29
Inverts (individuals)	Project age (0.145)	River width (0.108)	River type (0.073)	0.27
Macrophytes (taxa)	Agriculture (0.191)	Project age (0.076)	River width (0.053)	0.31
Macrophytes (individuals)	Project age (0.141)	River width (0.061)	Project age (0.047)	0.34

Relative importance of the predictors

Overall, restoration success was most strongly influenced by project age, river width, and the percentage coverage of agricultural land use in the upstream catchment, i.e. by project, river reach, and catchment characteristics (**Figure 2-14**). In addition, restoration effect depended and differed between the three organism groups, river types, and biological metrics. The other two predictors were of minor importance (reach land use adjacent to the restored reach, restoration measures). These differences in the relative importance are of relevance since the Boosted Regression Tree model explained a substantial part of the variance in the effect sizes (0.41).

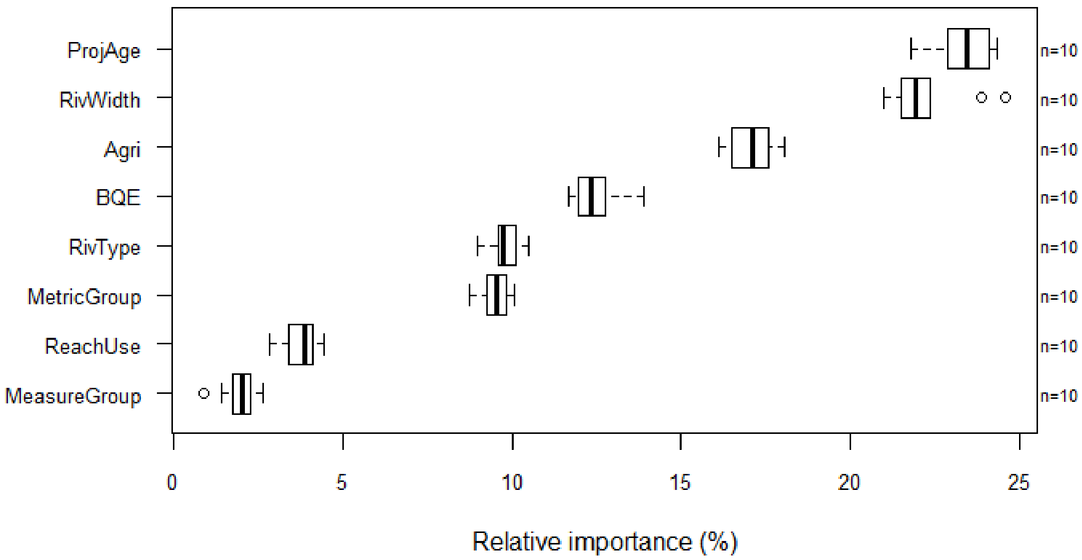


Figure 2-14: Relative importance of predictors in Boosted Regression Tree model of all biological and metric groups (n = 353 effect sizes) with a total variance explained of 0.41. Mean values of 10 replicate BRT model runs are given.

2.3 Discussion

In this meta-analysis, a large dataset on the effect of restoration on biota was compiled. Nevertheless, the dataset had some limitations which have to be considered for interpretation of the results: First, about one third of the effect sizes originated from studies published in peer-reviewed literature, which virtually all used biomass, abundance, richness, and diversity metrics to quantify the effect of restoration on biota, and hence, the analysis had to be restricted to these biological metrics. However, other metrics might be better suited to quantify restoration success if the general objective of restoration is not simply to increase biodiversity or abundance for its own sake, among other reasons because an increase in the number of individuals or taxa is not necessarily tantamount to a better or more natural biological state. For example, an increase in the number of taxa and abundance might be due to the creation of habitats which naturally do not occur in a specific stream type (e.g. addition of gravel and boulders in sand-bed rivers, Kristensen et al. 2011), colonization of invasive species, or the increase of non-stream type specific ubiquitous species. Nevertheless, the meta-analysis was an important first step to assess the effect of restoration on biota based on a comprehensive dataset. Second, the restoration projects investigated in this meta-analysis had mainly been implemented in the second-last decade (10-90th percentile range 1991-2005) and hence, the results reflected the way restoration had been undertaken in the past, limiting transferability to projects with similar catchment, river reach, and project characteristics. The more recent trend to shift from restoring forms (e.g. building channel features) to restoring processes (e.g. natural channel dynamics, sediment transport, natural flow regime) might increase restoration success. However, there is no empirical evidence yet to confirm this hypothesis. Third, most restoration projects applied several measures, monitoring results reflect the response of biota to all these measures, and hence, it was difficult to disentangle the effect of the single measures, which is a fundamental problem and difference to rigorous scientific experimental designs.

The first objective of this meta-analysis was to quantify the effect of restoration and to investigate differences between organism groups, biological metrics, and restoration measures.

The overall effect of restoration on the number of taxa and individuals was positive but a substantial share of the effect sizes showed no or a negative effect. The high variability of the restoration effect sizes might be the reason for the contrasting results reported in literature. Depending on which organism group, biological metric or restoration measure is investigated, results may greatly differ (e.g. Lepori et al. 2005, Palmer et al. 2010 vs. Lorenz et al. 2012, Schmutz et al. 2014).

In this meta-analysis, restoration effect differed between organism groups and the effect sizes for macrophyte taxa were higher compared to fish and invertebrates. This corresponds to the results of replicate studies reporting a positive effect on macrophyte richness and diversity (Lorenz et al. 2012), a low effect on macroinvertebrate diversity and richness (Jähnig et al. 2010, Palmer et al. 2010), and the general finding that restoration effect on taxa based metrics is highest for terrestrial and semi-aquatic groups like floodplain vegetation and ground beetles, intermediate for macrophytes, lower for fish, and lowest for macroinvertebrates (Januschke et al. 2009, Jähnig et al. 2009, Haase et al. 2012). This might be due to the fact that in many of the replicate studies mentioned above and for macrophytes in this meta-analysis, mainly widening and remeandering projects have been investigated, which create pioneer habitats like bare riparian areas and bare gravel bars, reduce flow velocity and water depth, often are sparsely shaded in the beginning, and hence, favour pioneer species in the riparian area and macrophytes in the aquatic zone in the first years. This is supported by the finding that widening and remeandering projects had a higher effect on the number of macrophyte taxa compared to fish and invertebrates.

Furthermore, restoration had a higher effect on the number of fish and invertebrate individuals than on the number of taxa, indicating that, in general, it is easier to increase the number of individuals in the restored reach than establishing new taxa. This is reasonable from an ecological point of view since recent empirical and modelling studies have shown that the species pool and source populations for (re-) colonization are often sparse and limiting the effect of restoration on biodiversity (Radinger and Wolter 2014, Stoll et al. 2014, Tonkin 2014).

The different restoration measures had a similar effect on biota, but instream measures were more effective in increasing the number of invertebrate taxa compared to planform and riparian measures and a similar pattern was observed for fish. Comparing the results of other replicate studies indicated that terrestrial and semi-aquatic organism groups like floodplain vegetation and ground beetles as well as macrophytes benefit most from planform measures and aquatic groups like fish and invertebrates from instream measures (section 2.2.3, Januschke et al. 2009, Jähnig et al. 2009, Miller et al. 2010, Haase et al. 2012, Lorenz et al. 2012). This is reasonable, since many instream measures like the placement of large wood or riffle creation target aquatic habitat conditions and often immediately increase aquatic habitat and substrate diversity. These new aquatic habitats can be quickly colonized by new aquatic species given that respective source populations are located nearby. In contrast, especially planform measures like widening and remeandering also affect instream habitat conditions but first of all elongate the river reach and do not necessarily increase substrate diversity (e.g. re-meandering a pure sand-bed lowland river with limited potential for substrate sorting). However, channel-planform measures often create terrestrial and semi-aquatic pioneer habitats like bare ground and open gravel bars. In this meta-analysis, the restoration projects were assigned to one single main measure, which was surprisingly straightforward for many projects (section 2.1.3). However, in most projects, one or several additional measures were applied which hampered assessing the effect of single measures and probably masked differences in the effect of measures. Therefore, it is assumed that the difference would have been more pronounced if pure instream and planform projects were compared. Subdividing the dataset according to the combinations of measures applied resulted in sample sizes too small for statistical analysis, and hence, larger datasets which focus on specific (common or promising) measure combinations are needed to derive more robust and practically relevant recommendations on the (cost-) effectiveness of measures.

The second objective was to identify whether catchment, river reach, and project characteristics influence the effect of restoration on biota, to investigate interactions between these predictors, and to quantify their relative importance. Overall, the most important predictors affecting restoration success were project age, river width, and the percentage agricultural land use in the upstream catchment.

Several studies showed that catchment land use is strongly related to the biological state, especially to macroinvertebrates, and it is often assumed that it is a proxy for pressures like water pollution, fine sediment loads, hydrological changes or missing source populations which are scarce due to the general low ecological state in the river network (Roth et al. 1996, Allan et al. 1997, Stephenson and Morin 2009, Sundermann et al. 2013, Kail and Wolter 2013). Therefore, it was surprising that agricultural land use only affected the restoration success for fish and not macroinvertebrates. Possibly, this was simply due to a missing gradient in the other sub datasets, i.e. percentage agricultural land was either below any critical threshold in most restored reaches and hence, had no detectable effect or it was above a critical threshold, already limiting biota with any further increase having only a minor impact. The distribution of the percentage coverage values of agricultural land use indeed differed and the 25th percentile value in the fish individuals sub dataset was markedly lower (8.9%) compared to the other sub datasets (20.0%-35.9%). Due to this missing gradient (few restored reaches in forested catchments), it cannot be concluded that agricultural land use did not constrain restoration success in the other sub datasets, and the results rather indicate that it was above a critical threshold in most restoration projects. However, restoration had an overall positive effect (mean effect size of 0.22 for taxa based and 0.43 for individual based metrics) even in catchments dominated by agricultural land use (percentage coverage >50%), and hence, agricultural land use and the associated input of pesticides was not overriding the effect of restoration, as it was recently deduced by the media from a risk assessment of organic chemicals (Malaj et al. 2014), i.e. the results do not question river restoration in agricultural catchments in general.

For the same reason of a short gradient in the dataset, the missing effect of the restored reaches length on restoration success cannot be taken as a proof that it is sufficient to restore short river reaches. In all six sub datasets, virtually all restored reaches were rather short (10-90th percentile range 0.2-2.6 km) and length might have simply been below a critical threshold to increase restoration success. This is supported by the findings of Schmutz et al. (2014) who reported that the effect on the number of rheophilic fish species did depend on restored reach length and was largest for reaches > 3.85 km in length. These two examples illustrate why, although it is tempting to draw conclusions for river management from all results of the meta-analysis, we caution against oversimplified interpretations.

There was no obvious causal explanation why river width was among the most important predictors and restoration success was higher for fish individuals in larger rivers. Among others, this might have been due to co-correlations with other predictors like the significant differences in river width between river types. However, none of the co-correlations pointed in the same direction (e.g. river width was not largest for river types with the highest effect sizes).

Project age was the most important predictor affecting restoration success. It has already been stated in literature that the small effect of restoration measures on biota might be due to time-lags and hysteresis effects in the recovery process (Sundermann et al. 2013). More specifically, the results showed that restoration effect on macrophyte abundance was higher in more recent projects, indicating that restoration effect decreased over time. This is reasonable from an ecological point of view and it is speculated that the number of macrophyte taxa increased in the first years since favourable habitats which were created by the widening and remeandering projects were colonized rapidly from existing seed banks in the restored reaches or by drift of propagules, resulting in the large effect of restoration on the number of macrophyte taxa (**Figure 2-4**). In the following years, channel features possibly matured (e.g. riparian vegetation developed which increased shading, a deeper thalweg developed in the shallow cross-Sections of the widened reaches decreasing the shallow wetted area), and hence abundance of the macrophyte species decreased but not the number of taxa (**Figure 2-12**). Although this is the most probable explanation given the other co-correlations (see text to **Figure 2-12**), there are other possible reasons like an increase in restoration skills in recent years. However, there is limited empirical data to investigate the effect of project age on restoration success in more detail and to test the hypothesis mentioned above; which stresses the need for long-time monitoring to investigate the restoration effect over time, to better understand the trajectories of change induced by restoration measures, and to identify sustainable measures which enhance biota in the long-term.

The interactions of the predictors have been investigated, which can be used to describe conditions which favour restoration success (see section 2.2.5) but the variance explained by the models is too low and low sample size restricted the use of rigorous statistical tests to really predict the restoration outcome. Therefore, there is an urgent need to apply an adaptive management approach, to monitor the effect of restoration measures, and to modify the restoration design if necessary (Downs and Kondolf 2002).

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3. Urban river restoration

3.1 *Introduction to urban river restoration*

River rehabilitation has been practised more frequently in non-urban rivers than urban systems, but the methods used in the former may not be appropriate to be transferred for future actions in the constrained urban environment (Boitsidis & Gurnell 2004). This is because urban rivers are more restricted through channel engineering and artificial structures and may not display the physical habitats that are encountered in less heavily impacted channels, and the objectives of managing rivers in urban areas are often different, providing for public utility whilst protecting against flooding of properties. In addition, small urban rivers do not have the same characteristics as small non-urban rivers such as, channel dimensions reflecting the magnitude and frequency of the fluvial processes (Wharton 1995) and the frequency of geomorphological features, such as pools and riffles (e.g. Leopold & Wolman 1957) (Boitsidis & Gurnell 2004). However, inland waterways are similarly modified in urban and non-urban landscapes, while the principal difference might be riparian buffer zones, paved adjacent river margins and storm water run-off. Understanding the differences in characteristics between urban and non-urban watersheds is becoming more widespread and it is often recommended that approaches should reflect these differences, but in reality this is questionable, and not necessarily feasible.

Many urban areas have developed around rivers and societal demand has caused their degradation through a combination of multiple pressures that act simultaneously, such as domestic housing development and intensification, industry, water supply, flood protection, navigation and transportation, fisheries and recreation (Solimini et al. 2006; Booker & Dunbar 2004; Gurnell et al. 2007; Grimm et al. 2008; Tockner et al. 2009; Everard & Moggridge 2012). Urbanisation in particular, has significantly increased exploitation and degradation of rivers primarily caused by competing land uses that often result in a number of constraining pressures that differ from those in rural areas (Tourbier et al. 2004). These pressures have degraded the majority of urban rivers in Europe to the extent that some cease to provide the ecological requirements of a healthy freshwater ecosystem and further limit options for rehabilitation (Daily 1997; Groffman et al. 2003; Tourbier et al. 2004; Rohde et al. 2006; Grimm et al. 2008; Everard & Moggridge 2012). The most intensive period of river modification occurred during the industrial revolution in the 19th century, in central European lowland river training was done to make rivers navigable between 1830 and 1880. River hydrology was most dramatically changed around 1880 in urban areas, with improved efficiency of water power, the implementation of the water toilet and slightly later with the establishment of large scale sewer fields for waste water treatment. This exploitation has continued over the years with the rise in human population growth and its societal demands. In 2010, more than half the human population lived in urban areas, by 2030 it is expected that 6 out of every 10 people will live in a city, and by 2050, the proportion is expected to increase to 7 out of 10 people (WHO 2013). As a consequence, urban rivers are becoming an important focus for rehabilitation in Europe through the European Union Water Framework Directive's (EU WFD; 2000/60/EEC) objective for artificial and HWMB's to reach 'good ecological potential' (GEP) in the near future. With this in mind, river rehabilitation practise is likely to expand further as urbanisation continues and demands for a sustainable, but enhanced quality of life increases (Clifford 2007).

There have been many problems implementing the WFD due to insufficient understanding of ecological processes and poor scientific knowledge of river rehabilitation (Vaughan et al. 2009; Boon & Raven 2012). To advance river and wetland rehabilitation for both urban and non-urban systems, identifying rehabilitation success is key. However, most studies provide limited understanding when attempting to identify project success, usually because of limitations brought about by existing pressures that restrict the application of rehabilitation measures (Tarzwell 1937; Reeves et al. 1991; Roni et al. 2002; Bernhardt et al. 2005; FAO 2008; Roni et al. 2008). In addition, project success is rarely evaluated (Bernhardt et al. 2007), particularly because expectations have not been clearly defined with measurable objectives (Cowx et al. 2013).

3.1.1 Study scope and objectives

In 2004, the EU URBEM project (Tourbier et al. 2004) produced an overview of 50 existing urban river rehabilitation case studies from across Europe, producing a baseline for knowledge. A decade on, this deliverable intended to investigate urban river rehabilitation schemes across Europe, and to update current knowledge at a larger scale, building on existing experiences: (i) literature on urban river restoration and (ii) the WP1 database on hydromorphological river restoration projects compiled in the REFORM project. In contrast to section 2, this study is not a meta-analysis on the effect of restoration in urban rivers since biological monitoring were not available for a sufficiently large number of urban restoration projects but a description of urban river restoration project characteristics. Prevailing pressures, the application of rehabilitation measures, scale of rehabilitation and the identification of limitations for project success were evaluated from the WP1 database of river and wetland rehabilitation case studies for 161 urban and 878 non-urban case studies (information on European and non-European hydromorphological restoration projects were compiled in this database, irrespective if monitoring data were available, and used in this satellite topic). This not only enabled a large scale overview of urban river cases studies, but also a comparison against non-urban case studies.

Specific Objectives:

- Review current state of urban rivers and factors impacting upon them through a literature review of morphological, hydrological and water quality pressures.
- To compare difference in pressures acting on and measures applied in urban and non-urban case studies.
- To improve understanding of rehabilitation measures applied in urban river restoration

3.1.2 Methods

Literature review

A review was carried out to evaluate the current information on urbanization on rivers, with particular emphasis on morphological, hydrological and water quality pressures and measures used to remediate the ecosystem functioning.

WP 1 database on hydromorphological restoration projects

The WP 1 database was created from web-based and literature research, from past EU projects such as WISER, FORECASTER and RESTORE, and further independently collated by project partners under WP1. Information on hydromorphological restoration projects were compiled in this database, irrespective if biological monitoring data were available. Among others, the analyses of this satellite topic on urban rivers were based on this WP1 database and it was used to identify restoration projects for which monitoring data were available, which were compiled and analysed in the WP4 database for the meta-analysis presented in section 2. The WP 4 database could not be used to apply a meta-analysis to the urban river review because there was very limited data on the biological outcomes of measures for urban rivers.

The WP 1 database used contained 878 rehabilitation case studies mainly from European countries, plus a small number of examples from other parts of the world. The database included detailed information for the categories publication information, rehabilitation site information, pressures and measures, monitoring descriptions and project success/failure. A total of 161 European urban rehabilitation case studies were extracted from the database by selection of the 'reach use' category to create an exclusively 'urban' database to be independently analysed and compared to non-urban case studies for this deliverable. Due to the focus of the REFORM project on hydromorphological alterations, information on hydromorphological restoration projects were compiled in the WP 1 database and *pure* water quality or river continuity projects were excluded. However, if water quality or river continuity measures were implemented *in addition* to hydromorphological measures, this was indicated in the database.

There were ten main groupings for restoration measures, descriptions for 8 of these can be found in Table 2-5, water quality measures (diffuse and point source) were included in the preliminary analysis, but were eliminated from the data set thereafter and are not present in the table. Specific restoration measures make up these main groupings and can be found in the Appendix Table A-2. In most cases, multiple measures were recorded for single case studies.

The measure of restoration success recorded was qualitative and identified by 3 categories - success, failure and unclear. This information was taken from the literature where available by each partner, if information was not available on project success, then it was recorded as 'no information'. The partner inputting the information did not make the decision on project success if it was not clearly stated.

River widths (>5 m, 5-10 m, 10-20 m, 20-50 m and >50 m) and lengths (<0.5 km, 0.5-1 km, 1-3 km, 3-5 km, 5-10 km and >10 km) were recorded as size categories in the database. River widths were used as a 'size category' during the analysis.

Analysis

Several analyses were done for different sub-datasets:

- *Within each size category* – similarity between urban and non-urban case studies, for the four main measure categories applied, within each size category. It was hypothesized that measures with a higher need for space (e.g. re-meandering) might have been applied to a similar extent in small urban and non-urban rivers but less often in larger urban rivers compared to larger non-urban rivers.

- *Combined size categories* – grouping less than (<) and more than (>) for each size category. For example, a size category or 10 m would combine all those case studies with river width <10 m (<5 m and 5-10 m) and then all those >10 m (10-20 m, 20-50 m and >50 m) to produce two values. To investigate similarity between urban and non-urban by comparing frequency of measures for each combined size category.
- *Within each measures* - similarity of urban and non-urban case studies, for each of the four main category measures across all size categories.
- *Urban vs. Sub-urban* – Urban rivers were further categorised into urban and sub-urban to see if the level of pressures influenced the frequency and type of measure applied. Urban rivers were described as those in the centre of cities, where pressures were expected to be magnified in comparison to sub-urban rivers where more green space was expected. Case studies were classified as urban or sub-urban by viewing their location on Google Maps and making the judgment from an aerial view of the local surroundings.

Where possible, data was statistically tested for significance using Chi Square test, however, in some instances analysis was limited by a small sample size (<5).

Correspondence analysis was used to compare measures within the four main restoration measure categories applied to urban and non-urban rivers of different widths. It is a multivariate statistical technique that presents categorical data in a two-dimensional graph.

3.2 *Literature review on urban river pressures, impacts and measures*

Urban areas and human habitation topped amongst the 10 most frequently reported pressures by the EU-25 Member States for lake and river habitat types (EEA 2012). Urban river rehabilitation is becoming increasingly important, especially in collaboration with other projects for city development and urban planning to reach win–win situations: improving flood control and ecological functions (meeting WFD objectives), while offering recreational value and raising the quality of life in urban areas (EEA 2012) (an example of a current case study integrating flood risk management with maintain ecological function can be found in Appendix 2). Although urban rivers tend to make up a small section of a whole river catchment, anthropogenic pressures that impact on freshwater systems can be magnified in contrast to non-urban rivers This is due to the combined effect of multiple pressures such as impoundment and channelization, increased impervious surfaces, water abstraction, pollution, increased sedimentation and alteration of riparian vegetation, (Dynesius & Nilsson 1994; Forman & Alexander 1998; Paul & Meyer 2001; Aarts et al. 2004; Pyrce 2004; Reid 2004; Vinebrooke & Cottingham 2004; Vaughn et al. 2009; Schinegger et al. 2011). All of which have a notable impact on instream habitats and communities and can result in reduced biotic richness such as fish (Wang et al. 2000, 2001; Roy et al. 2006), invertebrates (Beavan et al. 2001; Chadwick et al. 2006) and macrophytes (Suren 2000). In many instances, urban river banks and beds are artificially modified to reduce erosion and substrate movement by the exchange of natural substrate to a more firm, man-made substance and in some cases a lining of the river bed will be completed through a dense urban area (Rocha et al. 2004). As a result, artificial channels have been found to increase overall drainage densities; in addition to an increased slope this contributes to an increase in-stream velocity and conveyance efficiency (Pizzuto et al. 2000; Meyer & Wallace 2001). The construction of culverts to cover streams occurs in many urban areas, for example there is an entire network of rivers culverted under central London (Barton 1992), of which many were once noted for their rich fisheries (Walton 1653; Everard & Moggridge 2012). Some of these culverted streams have also been converted into storm drainage systems (Rocha et al. 2004). In addition, natural land surfaces are replaced by artificial, impervious surfaces such as pavements, roads and roofs meaning vegetation is cleared and soil compacted. Efficient drainage systems in addition to artificial surfaces in urban areas will increase the volume and velocity of runoff that reaches the river and therefore, alters the hydrology of the river system and can lead to peak flow and flood risk downstream (Rocha et al. 2004). This reduces the availability of flow refuge, lowering the diversity and abundance of biota capable of recovering from flooding (Negishi et al. 2002; Lake et al 2007). Urban land run off from impermeable surface, flash flooding and drainage contribute greatly to the poor water quality of urban river systems (Paul & Meyer 2001). Point source pollution results in the introduction of toxic substances (both of organic and inorganic origin) and is generally a consequence of industry, both past and present as well as domestic discharges (Omernik 1976; House et al. 1993; Meybeck 1998; USGS 1999; Winger & Duthie 2000; Wenger et al. 2009). Elevated suspended sediment levels are caused by anthropogenic actions such as mining, road-deposited sediments, industrial point sources and wastewater (Walters et al. 2003; Grimm et al. 2005; Gurnell et al. 2007; Taylor & Owens 2009; Everard & Moggridge 2012) and have effects such as bed sediment changes, nutrient enrichment and turbidity, all of which contribute to reduced diversity of stream macrophytes, degraded riparian buffer zones magnify these effects (Suren 2000).

A European Commission project on Natural Water Retention Measures identified the main urban measures to be buffer strips and swales, permeable surfaces and filter drains, infiltration devices and green roofs. Considerable success in reducing the discharge of pollutants into Europe's waters in recent decades shows that we are on the right track towards reducing pollution from urban and industrial wastewater and agricultural sources (EEA 2012). Continuing improvement in the level of pollutant removal from urban wastewater discharges is anticipated and driven by requirements under the Urban Waste Water Treatment (UWWT) Directive (91/271/EEC) and national legislation (EEA 2012). In addition to improve water run-off and pollution pressures, habitats in urban rivers must be restored with suitable refugia capable of enhancing the resistance and resilience of populations to both natural and anthropogenic disturbances (Sedell et al. 1990; Lancaster & Hildrew 1993; Bond & Lake 2005).

The combined interactions of global climate change and human pressures have a great impact on water bodies. The capacity of an ecosystem to adapt to climate change depends not only on the diversity of species it currently supports, but the number of pressures present. Climate change is predicted to be the cause for the increasing frequency of natural hazards such as floods and droughts (Bernasconia et al. 2005; Moren-Abat et al. 2006). The likely increase in the variability of extreme flood events in urban areas through increased precipitation are now widely recognized as a major challenge and risk for flood management approaches (Douglas et al. 2007). Development of urban centres on floodplains is a common finding and increases the risk of flooding following extreme rainfall events (Wheater 2006). Flood protection may be provided by increasing drainage capacity (e.g. enlarged/straightened channels, raised flood banks) or by reducing flows (e.g. diversions, storage in reservoirs/ enlarged flood plains – and possibly changes in land use/management). As a consequence, pressures from flood protection activities are predicted to intensify in the future because of an increase in extreme flow events (Booth & Jackson 1997; Kemp & Spotila 1997; Schleiger 2000; Wang et al. 2000; Fitzpatrick et al. 2004; Blakely & Harding 2005; Brown et al. 2005; Europa 2006; Schwartz & Herricks 2007; European Commission 2009; Webb & King 2009; Nelson et al. 2009; Wenger et al. 2009).

As a consequence, managers may be required to change the way European waters are conserved, especially as the ecological classifications in the Water Framework Directive are likely to change with climate and therefore cannot be considered as static (Bernasconia et al. 2005). Adaptation and rehabilitation guided through the programme of measures should be more widespread to counteract the collective impacts of human pressures and climate change on European waters. In summary, specific pressures for urban rivers usually mentioned in literature are artificial river banks, beds and surrounding impervious surfaces such as pavements, roads and roofs. Urban land run off from impermeable surface in addition to efficient drainage systems will increase the volume and velocity of runoff that reaches the river altering the hydrology of the river system leading to peak flow, flood risk downstream and water quality problems (Rocha et al. 2004). The main measures applied to urban rivers according to literature are buffer strips and swales, permeable surfaces and filter drains, infiltration devices and green roofs.

3.3 *WP1 database analysis*

3.3.1 Current state of urban river restoration

Previously, river rehabilitation in an urban setting was not a regular occurrence (Hansen 1996; Zöckler 2000). A survey carried out in the early 1990s reported from 66 investigated projects across Europe, only 11% were urban and 21% urban/rural (de Waal et al. 1995). This was still the case 10 years later when Nijland & Cals (2001) established that of 60 papers published in the proceedings of the 2000 conference of the European River Restoration Centre (2001) only 2 referred to urban water courses. In 2004, the most current overview on urban rivers in Europe was from the EU URBEM (Urban River Basin Enhancement Methods) project. The project identified that the majority of river rehabilitation publications up to the year 2004, referred to schemes and processes in rural areas (Tourbier et al. 2004). Nevertheless, over the past 10 years there has been a growing number of completed urban river restoration schemes reported across Europe: a search of Web of Science demonstrates how urban river and wetland restoration has gained more focus over the past 10 years. The number of published papers has more than doubled when comparing the year 2003 (n=34) with 2013 (n=110) (Figure 3-1). Although there is a notable increased focus on urban river restoration, non-urban river rehabilitation remains the key focus, with a higher number of published items (Figure 3-1). Analysis of the WP1 database identified similar findings with only 18% (n=161) of 878 rehabilitation case studies categorised as urban.

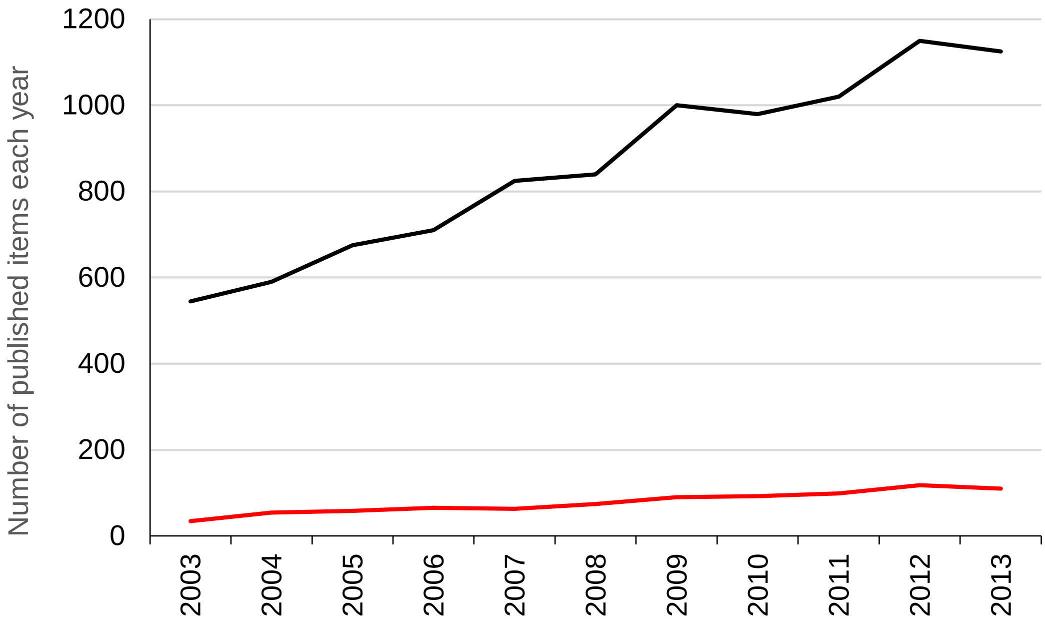

Figure 3-1: Publication trends for evaluating urban — and non-urban — river rehabilitation over the past 10 years. Urban search contained the key words Topic= (river rehabilit* OR river restor* OR wetland rehabilit* OR wetland restor*) AND Topic= (urban*). Non-urban search contained the key words Topic= (river rehabilit* OR river restor* OR wetland rehabilit* OR wetland restor*) (Web of Knowledge completed June 2014).

3.3.2 European and non-European urban case studies

A total of 161 urban and 670 non-urban river rehabilitation schemes, mainly from Europe (urban 73%, non-urban 77%), were identified from the WP1 database (Table 3-1). There was an uneven distribution of urban and non-urban river rehabilitation projects across European countries. For example, Germany reported a high proportion of urban (n = 72, 45% of all urban case studies) than non-urban (n = 240, 36% of all non-urban case studies) case studies (Table 3-1). Whereas France (urban 13% and non-urban 14%), UK (urban 12%, non-urban 8%), Austria (urban 2%, non-urban 6%), and the Netherlands (urban 1%, non-urban 2%) reported a lower proportion of case studies. Low numbers of non-urban case studies were reported for 12 additional European countries (Table 3-1), which may reflect the government policies towards urban development and population density.

Table 3-1: Number of rehabilitation case studies from the REFORM WP1 database for European and non-European countries categorised by urban and non-urban reach use.

Country	Urban	Non-urban
Germany	72	240
France	21	94
UK	20	51
Austria	3	43
Netherlands	2	10
Spain	0	20
Denmark	0	14
Romania	0	15
Belgium	0	2
Bulgaria	0	2
Czech	0	2
Finland	0	8
Switzerland	0	8
Sweden	0	6
Italy	0	3
Norway	0	1
Liechtenstein	0	1
Non-European	43	150
Total	161	670

3.3.3 Pressures identified from the WP1 database

Hydromorphological degradation in urban rivers occurs as a consequence of multiple pressures. The REFORM WP1 database categorised 5 main pressures on rivers, these were morphological, river fragmentation, flow regulation, water abstraction and pollution pressures (Figure 3-2) and there was a significant difference (Chi-square (4, n=2879) = 13.40, $p < 0.01$) when comparing urban and non-urban case studies for each pressure category. Each of these pressure categories has a number of sub-categories for more specific pressures and therefore, multiple pressures were recorded for each of the case studies. The percentages of specific pressures for non-urban and urban case studies were similar on most accounts (Figure 3-2). Overall, a larger number of morphological pressures were recorded compared with the other four pressure categories. Morphological alterations resulting from morphological pressures were recorded as the largest pressures for both non-urban (83%) and urban (89%) water bodies (Figure 3-2). Channelization and instream habitat were also identified as large pressures for both non-urban (55% and 42% respectively) and urban (65% and 42% respectively) water bodies (Figure 3-2). A lower percentage of case studies recorded river fragmentation (non-urban 22%, urban 15%), barriers to upstream (non-urban 16%; urban 10%) and downstream migration (non-urban 13%; urban 11%) (Figure 3-2). Less than 15% of case studies reported flow regulation, water abstraction and pollution pressures for both non-urban and urban reach use (Figure 3-2).

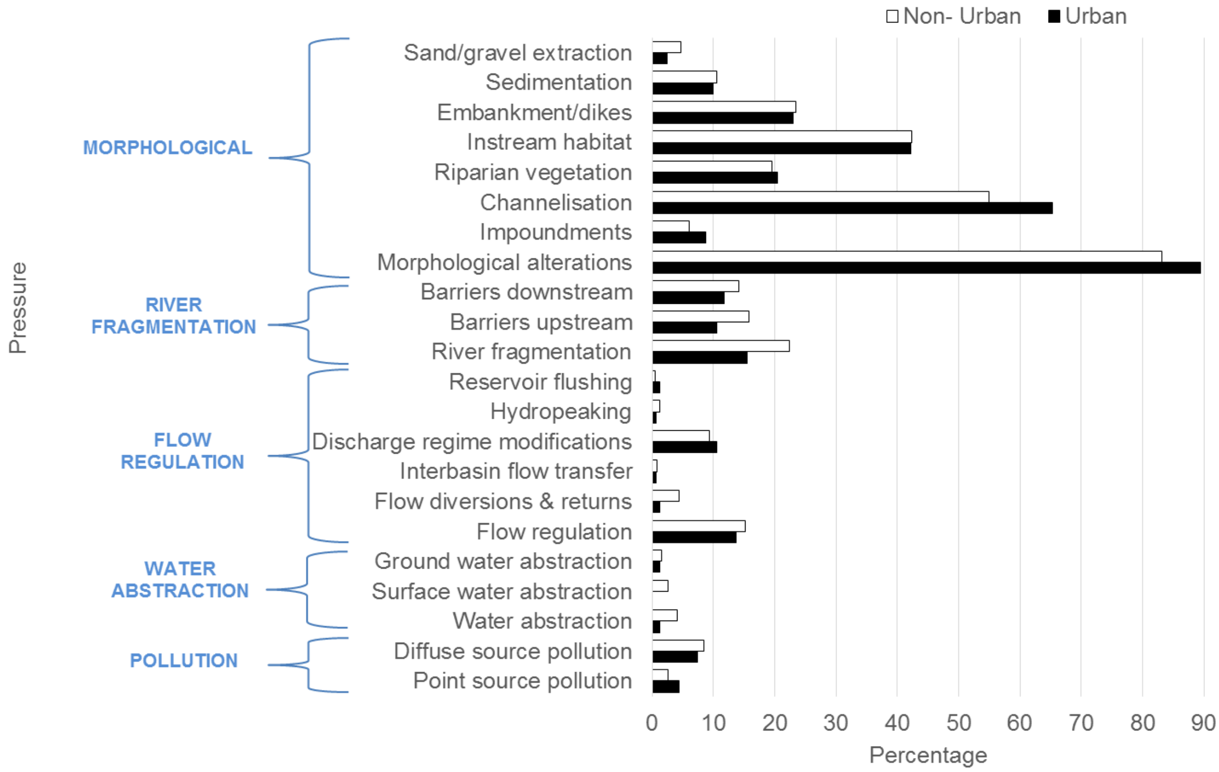

Figure 3-2: Pressures recorded for non-urban (n=670) and urban (n=161) case studies from REFORM WP1 database (multiple pressures present at individual case studies).

When testing specific pressures against the different river widths for urban and non-urban case studies, there were clear similarities between pressures on small to medium urban and non-urban rivers, of the same size category (>5 m, 5-10 m and 10-20 m wide), however, pressures acting on larger rivers (20-50 m and > 50 m wide) were not so similar (Figure 3-3). Morphological alterations (P_Morph) were a frequent pressure occurring in all size categories of urban and non-urban rivers, the similarity between <10 m width case studies and their dissimilarity with all other size categories was the low number of other pressures recorded. Interbasin flow transfer (PFR_Trans), river continuity (P_RiCont) and ground water abstraction (WA_Grund) were most frequently occurring pressures in 10-20 m urban and non-urban case studies (Figure 3-3). Urban river case studies >50 m wide were different to all other case studies because they had a higher frequency of riparian vegetation (PMo_Rip) and impoundment (PMo_Imp) pressures recorded.

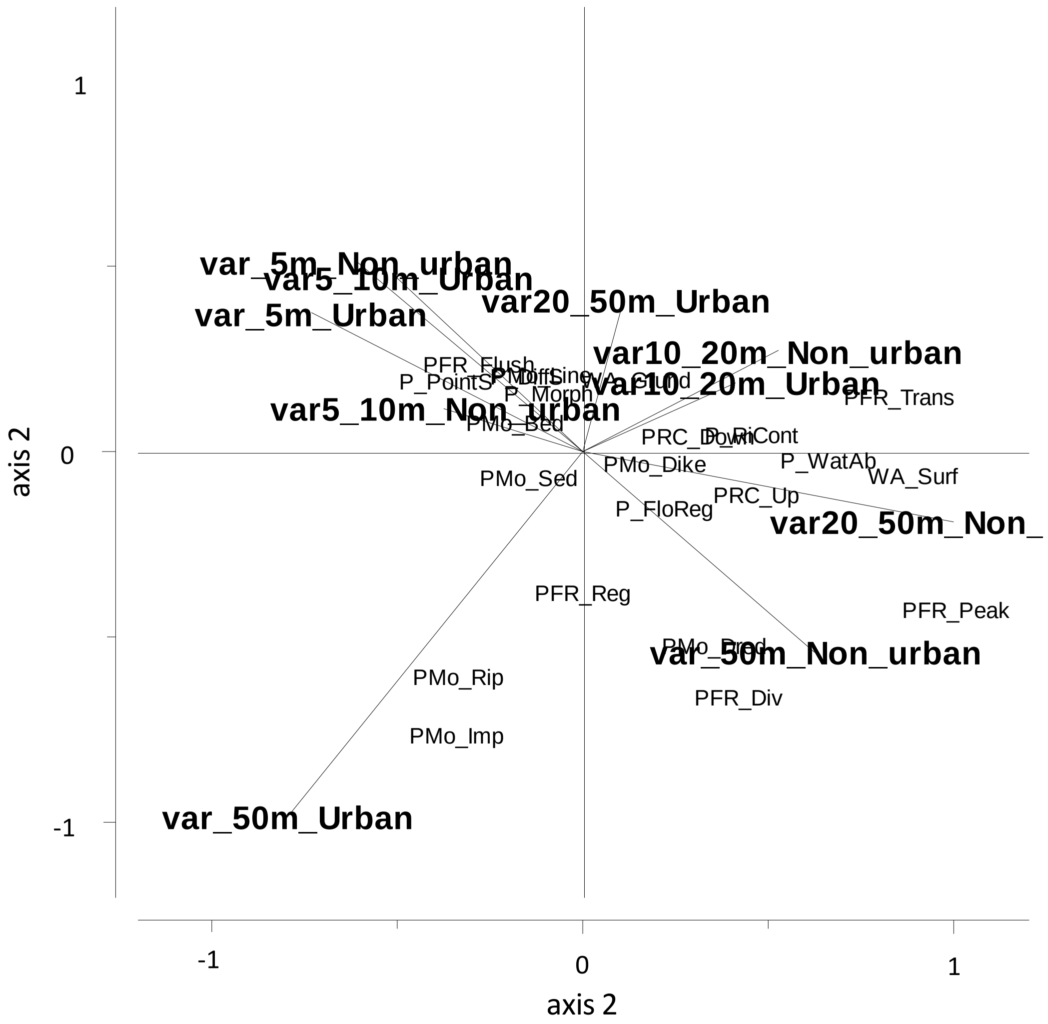

Figure 3-3: Correspondence analysis plot comparing the pressures (Figure 3-2) reported for the five different river widths (Figure 3-6) for urban and non-urban case studies (see Appendix Table A-2 for explanation of abbreviations).

3.3.4 Rehabilitation measures identified from the WP1 database

The WP1 database categorised 10 main rehabilitation measure categories with each measure category having a number of sub-categories of more specific rehabilitation techniques applied for urban and non-urban case studies. Multiple measures were recorded for many case studies (Figure 3-4). The most abundant rehabilitation measure categories applied in both urban and non-urban case studies were instream rehabilitation (68% and 63% respectively), river planform (38% and 37% respectively), riparian zone (36% and 23% respectively) and lateral connectivity (35% and 37% respectively) (Figure 3-4). Less than 5% of rehabilitation measures were recorded to improve sediment quality, flow dynamics, diffuse and point source pollution (Figure 3-4). Case studies where only water quality measures were applied were not included in the database since REFORM focuses on hydromorphological alterations, which was the reason for a low number of case studies recording pollution measures (Figure 3-4).

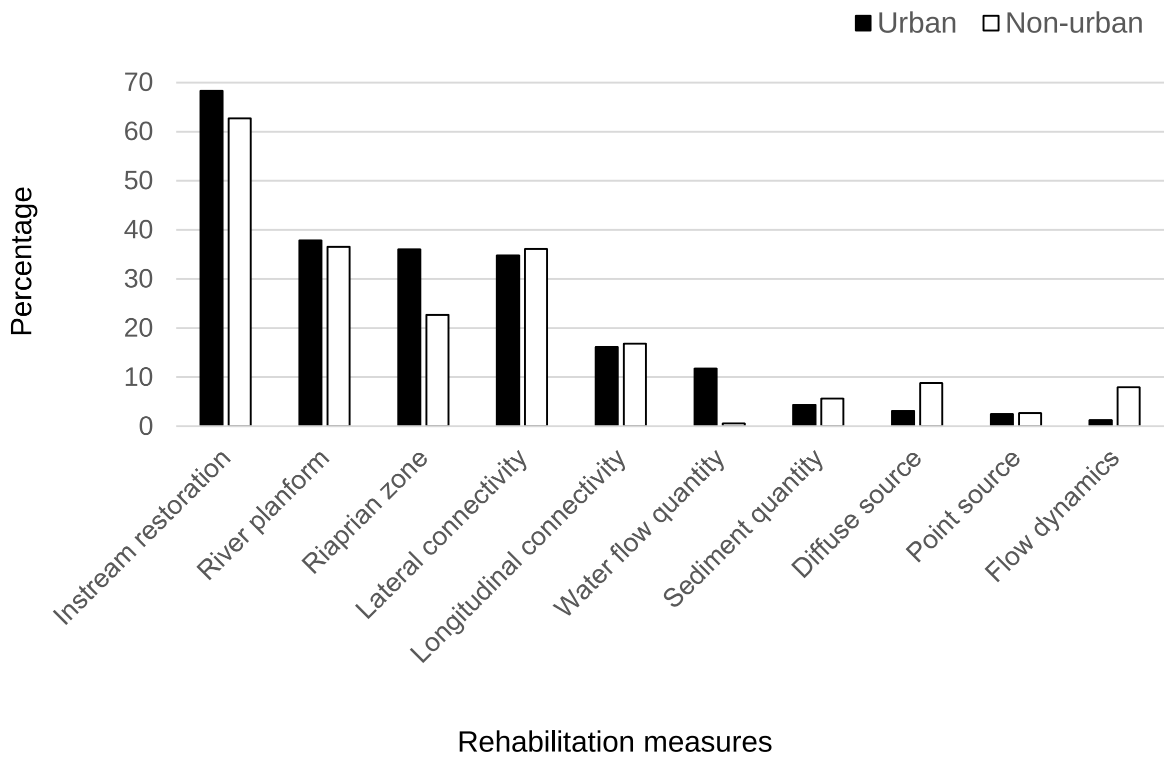

Figure 3-4: River rehabilitation measures recorded for urban (n=161) and non-urban (n=670) case studies from the REFORM WP1 database (multiple measures present at individual case studies).

There were similar trends when comparing the single restoration measures of the four most common measure categories, riparian zone, instream rehabilitation, river planform and lateral connectivity (Figure 3-5). A descriptive analysis was done because there was inadequate sample size (<5) for a number of the measures limiting statistical analysis. Overall, developing natural riparian vegetation was the most abundant measure applied for both urban (36%) and non-urban (28%) case studies, there was infrequent use of buffer strips to reduce nutrient (urban 1%, non-urban 3%) and sediment (urban and non-urban 2%) input in all case studies (Figure 3-5). Eleven options to improve instream rehabilitation were frequently applied for both urban and non-urban case studies (Figure 3-5). Removing bed fixation was the most popular option for instream rehabilitation in urban (26%) case studies but the measure was only used in 5% of case studies in non-urban systems (Figure 3-5). Removing bank fixation (22%) was the second highest option for instream rehabilitation of urban case studies, closely followed by creating artificial bars or riffles (17%) (Figure 3-5). Non-urban case studies followed a similar trend but with lower percentages (15% and 13% respectively) (Figure 3-5). Initiating natural channel dynamics, boulder placement and creating shallow at banks were measures equally applied (12-13%) for urban and non-urban studies (Figure 3-5). Less than 5% number of urban and non-urban case studies managed aquatic vegetation, removed sediments or modified hydraulic structures as instream rehabilitation measures (Figure 3-5). Re-meandering as an option for restoring river planform occurred in 19% of urban and 17% of non-urban case studies, whereas widening and re-braiding were equally applied in urban and non-urban case studies (13%). All other specific river planform rehabilitation measures (<10%) were less frequently applied (Figure 3-5).

Creating semi-natural features was the most abundant urban measure (15%) to improve lateral connectivity (Figure 3-5). Recreate floodplain features was the most abundant rehabilitation measure for improving lateral connectivity for non-urban case studies (20%), where only 10% of urban case studies applied this measure (Figure 3-5). Less than 10% of urban and non-urban rehabilitation measures lowered embankments, furthermore, total removal of embankments was a rare option to improve lateral connectivity for non-urban (4%) case studies and was not applied in urban rivers (Figure 3-5).

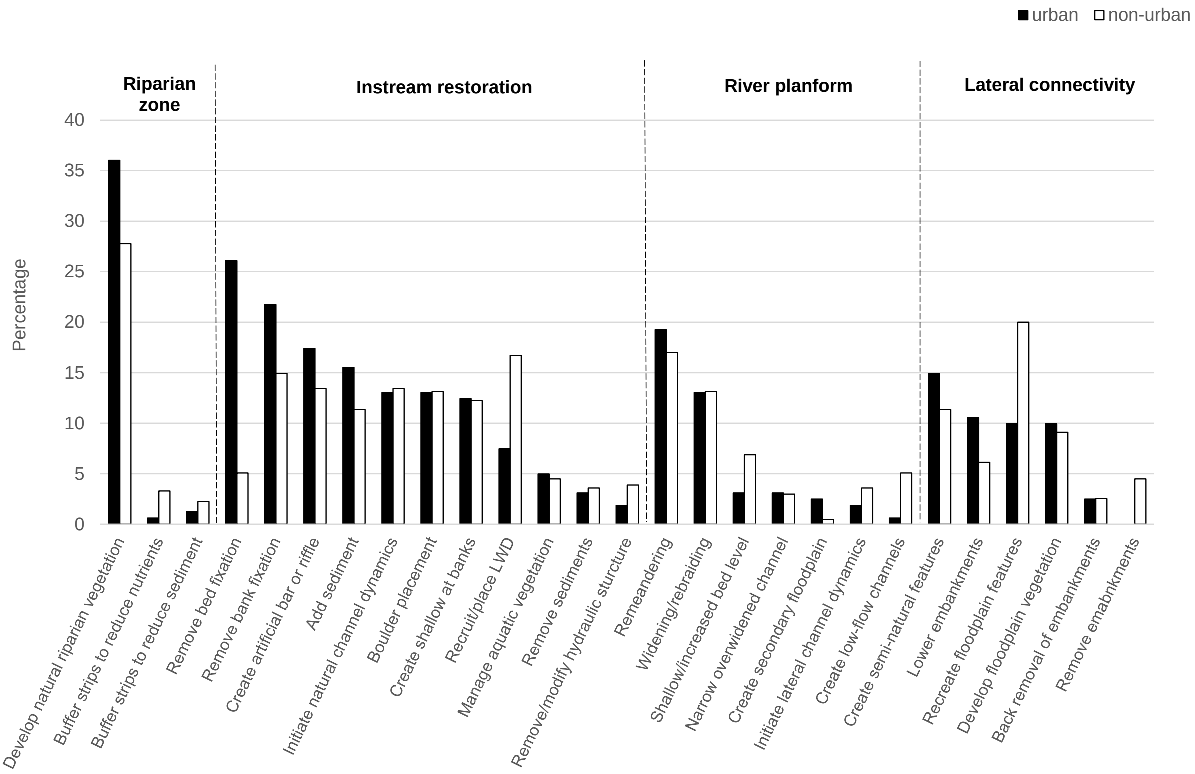

Figure 3-5: Specific rehabilitation measures from four main measure categories taken from Figure 3-4 (lateral connectivity, river planform, riparian zone and instream restoration), recorded for urban and non-urban case studies from the REFORM WP1 database (multiple measures present at individual case studies).

3.3.5 Scale of rehabilitation

River width

Rehabilitation case studies were recorded in urban rivers that ranged from <5 m to >50 m wide and overall, there was a decrease in the frequency of urban rehabilitation cases studies as the river width increased (Figure 3-6). There was a significant difference (Chi-square $(4, n=831) = 9.65$, $p < 0.05$) in the size of rivers restored between urban and non-urban case studies. This was probably due to the highest frequency of urban and non-urban case studies occurred on rivers with widths of <5 m (46% and 26% respectively) (Figure 3-6). Rivers with widths of 5-10 m and 10-20 m had 18% of urban case studies recorded and <10% occurred in rivers with widths of 20-50 m or >50 m (Figure 3-6). All other river widths for non-urban case studies were found in the range of 10-20% (Figure 3-6). There were slightly fewer case studies on non-urban rivers with widths of 5-10 m (16%) and 10-20 m (15%), with a corresponding higher frequency of case studies for rivers with widths of 20-50 m (16%) and >50 m (19%) (Figure 3-6).

Restored river length

The difference in restored river length between urban and non-urban rivers was highly significant (Chi-square (5, n=831) = 24.12, $p < 0.001$). The largest number of urban case studies recorded restored river lengths of <0.5 km (45%), and there were generally fewer case studies with restored river lengths of larger sizes, particularly for restored river lengths of 3-5 km (1%) (Figure 3-6). For example, <20% of urban case studies were recorded for restored river lengths of 1-3 km (1%) and 0.5-1 km (16%) whereas <5% case studies recorded larger restored river lengths of 3-5 km (1%), 5-10 km (5%) and >10 km (4%) (Figure 3-6). A similar trend occurred for non-urban restored river lengths, the largest number of reported non-urban case studies have a restored river length of <0.5 km (26%), followed by river lengths of 0.5-1 km (13%) and 1-3 km (20%), whereas larger restored river lengths had fewer case studies recorded, 3-5 km (5%), 5-10 km (3%) and >10 km (7%) (Figure 3-6).

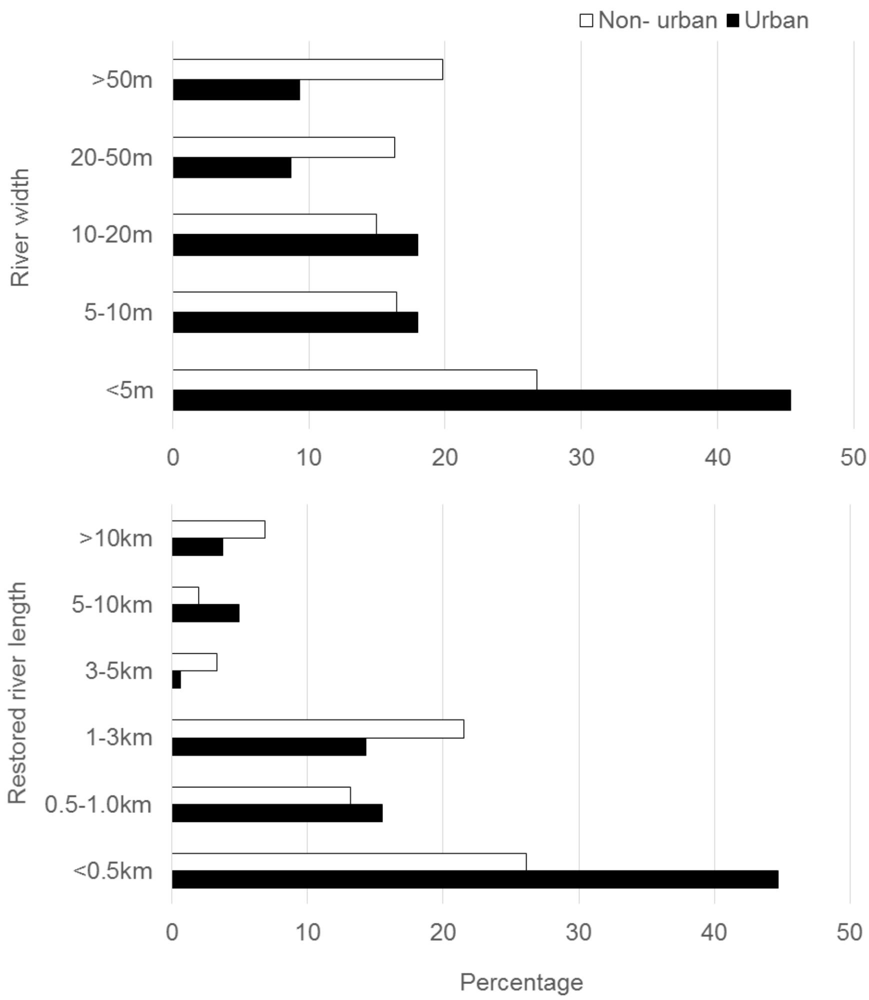

Figure 3-6: River width and restored river length for urban (n= 161) and non-urban (n= 670) case studies from the REFORM WP1 database.

Within each size category

Overall, when grouped into the four main measure categories, measures were similarly applied between urban and non-urban case studies within all width categories (<5 m, 5-10 m, 20-50 m and >50 m widths), but differences were only significant for the <5 m size category (Chi-square (3, n=431) = 9.36, $p < 0.05$). Restoration measures in the riparian zone were applied more frequently in urban (27%) than non-urban (15%) case studies for rivers <5m width possibly causing this difference (Figure 3-7). Instream measure was the most abundant measure for both urban (all >36%) and non-urban (all >37%) case studies within each width category, with the exception of >50 m where lateral connectivity was the main measure applied for urban (40%) and non-urban (35%) (Figure 3-7). Riparian zone was the least applied measure, within each size category, for all non-urban case studies all (<16%), but this only occurred for urban case studies of 5-10 m (19%), 10-20 m (11%) and 20-50 m (8%) widths (Figure 3-7).

Combined size categories

River width categories (<5m and >5m width, <10 m and >10 m width, <20 m and >20 m and <50 m and >50 m width) were used to further investigate how river size influences the four main restoration measures applied, for urban and non-urban rivers. For each of the four measures, case study frequencies were recorded into two groups, these were less than (<) and more than (>). Overall, there was a significant difference when comparing the frequency of the four main measure categories between urban rivers of <5 m and >5 m width (Chi square (3, 284) = 10.92, $p < 0.05$), <10 m and >10 m width (Chi square (3, n=284) = 13.89, $p < 0.05$) and <50 m and >50 m width (Chi square (3, n=284) = 8.34, $p < 0.05$. There was no significant difference between the frequency of different measures applied in urban restored rivers of <20 m and >20 m wide (Chi square (3, n=284) = 7.7, $p > 0.05$). There was a significant difference when comparing the frequency of the four main measure categories between non-urban rivers of <50 m and >50 m width (Chi square (3, n=1029) = 22.132, $p < 0.001$), however, there was no significant difference for <5 m and >5 m wide (Chi square (3, n=1029) = 4.55, $p > 0.05$), <10 m and >10 m wide (Chi square (3, n=1029) = 6.17, $p > 0.05$), and <20 m and >20 m wide (Chi square (3, n=1029) = 8.25, $p > 0.05$).

Within each measure category

Further investigation between urban and non-urban case studies, for individual measures across all river widths found that there was a significant difference for instream restoration (Chi square (4, n=521) = 16.26, $p < 0.05$), but not for lateral connectivity (Chi square (4, n=286) = 6.6, $p > 0.05$). Chi Square was not tested for riparian zone or river planform measures due to a small sample size (<5) for some of the size categories.

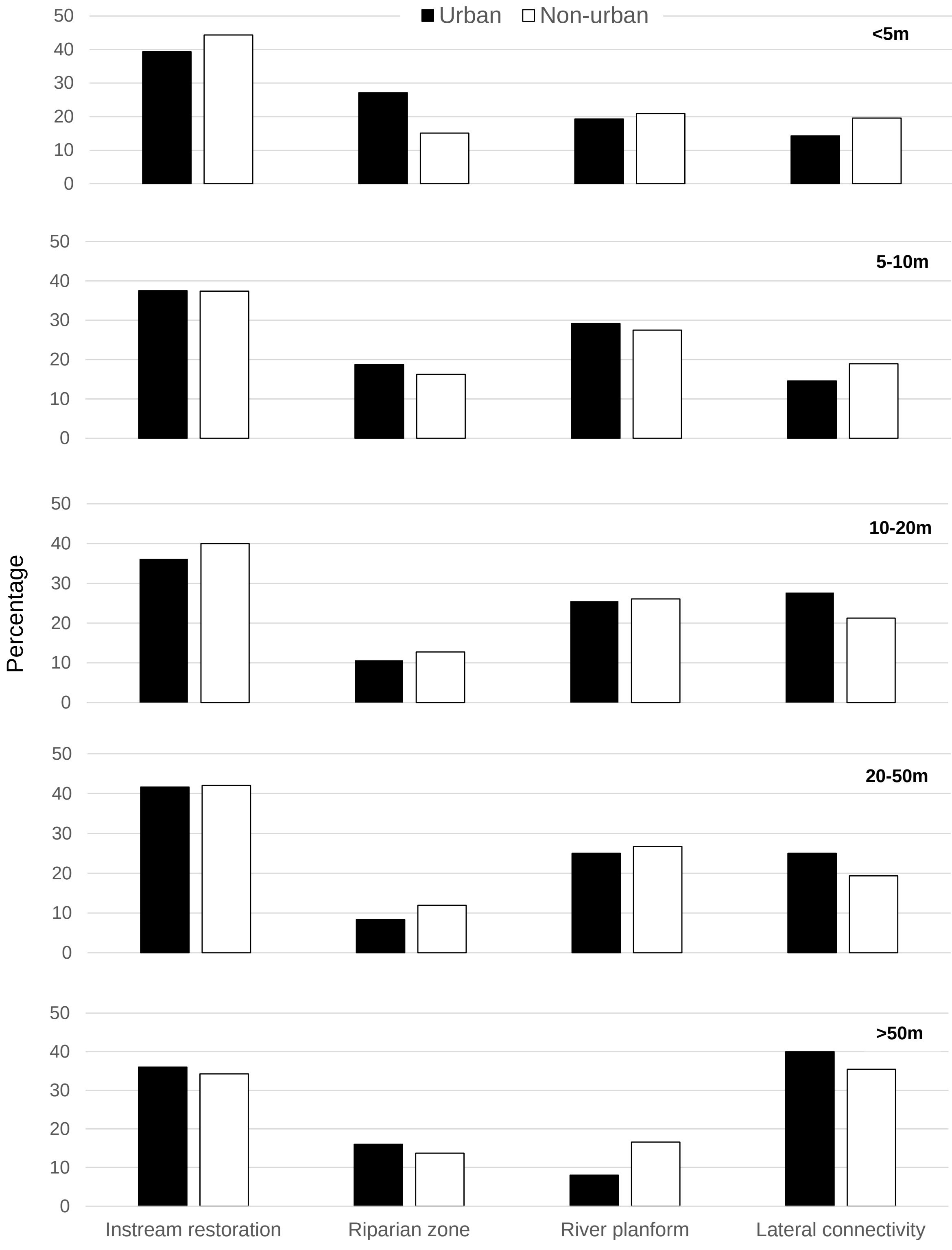

Figure 3-7: River width categories (<5 m, 5-10m, 10-20m, 20-50m and >50m) for the four main measure categories (lateral connectivity, river planform, riparian zone and instream restoration) taken from Figure 3-3, recorded for urban and non-urban case studies from the REFORM WP1 database (multiple measures present at individual case studies).

Comparing between urban and non-urban rivers size

When testing specific measures (Figure 3-5) for the four main measure category groups, against the different river widths for urban and non-urban case studies, there was a clear difference between urban <5 m, 5-10 m and 10-20 m, when compared to the same size categories as non-urban. Removal of bed fixation (MIn_FixBed), creation of secondary flood plain (MP_2Flod) and development of natural vegetation buffer strip (MR_VegBuff) were frequent measures used to restore urban <5 m, 5-10 m and 10-20 m case studies, whereas, instream measures boulder placement (MIn_Boulder) and managing aquatic vegetation (MIn_Veg) were more common for non-urban <5 m and 5-10 m case studies (Figure 3-8). Creating shallows (MIn_Shall), remove bank fixation (MIn_FixBank), create artificial riffle (MIn_Riff), removal of sediment (MIn_RemSed) and addition of sediment (MIn_AddSed) are all instream measures frequently applied in urban 20-50 m case studies, whereas a variety of different measure types are applied to non-urban 20-50 m case studies, such as creating low-flow channels in over-sized channels (MP_LowC) and remove embankments, levees or dikes that impede lateral connectivity (MFP_Remove) (Figure 3-8). Lateral connectivity measures were most frequent for >50 m urban case studies, specifically flood plain measures: create semi natural/artificial back waters (MFP_Create) and those not specifically stated (MFP_Veg) (Figure 3-8). In contrast, >50 m non-urban case studies remove or modify in-channel hydraulic structures (MIn_HyStruc) and allow lateral channel migration (MP_Dynamic) (Figure 3-8).

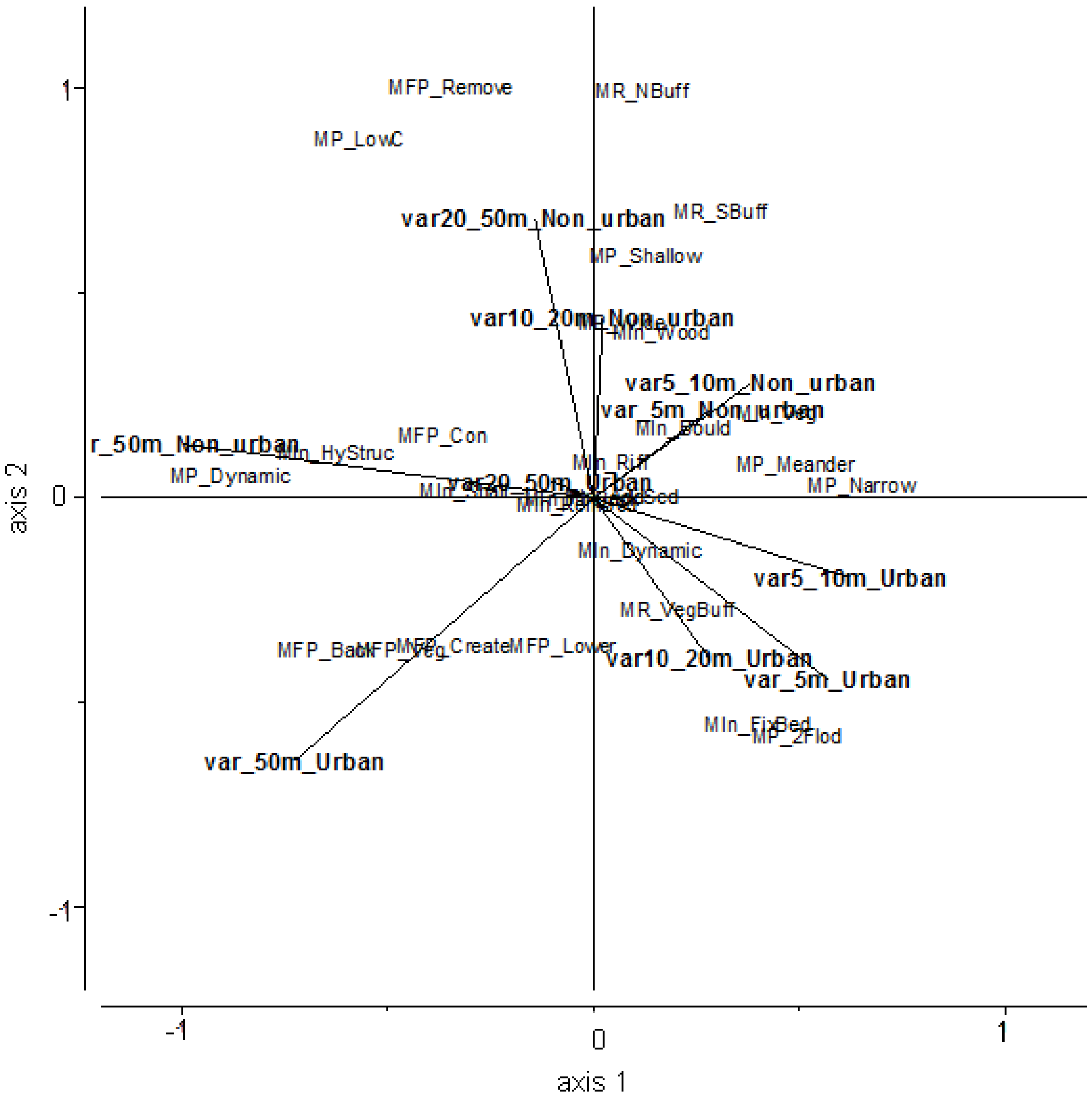

Figure 3-8: Correspondence analysis plot comparing the restoration measures (Figure 3-5) applied to the five different river widths (Figure 3-6) for urban and non-urban case studies (see Appendix Table A-2 for explanation of abbreviations).

Intraurban vs. sub-urban

Urban case studies were then sub-divided into intraurban (case studies that were located in city centres) and sub-urban (case studies located on the suburbs of cities). All four measure categories were applied less frequently in intraurban case studies than sub-urban (Figure 3-9), especially river planform measures (11% in urban compared to 42% in sub-urban). In both sub-datasets (intraurban and sub-urban), instream measures were applied most frequently compared to the other three measure categories (Figure 3-9).

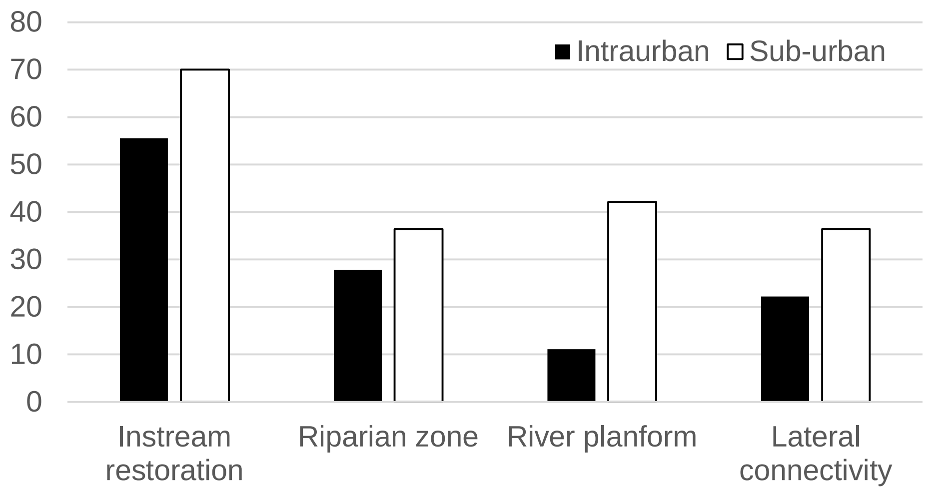

Figure 3-9: Percentage of different measure categories on the number of intraurban (n=18) and sub-urban (n=140) case studies from the REFORM WP1 database.

3.3.6 Identifying success

Ecological improvement was the predominant objective for urban (40%) and non-urban (37%) case studies, whereas, all other main objective categories represented <6% of records (Figure 3-10). When comparing urban to non-urban for each main objective category, there were only small differences between objectives of the rehabilitation (Figure 3-10). Over half of urban (59%) and non-urban (56%) case studies recorded no information about project objectives (Figure 3-10).

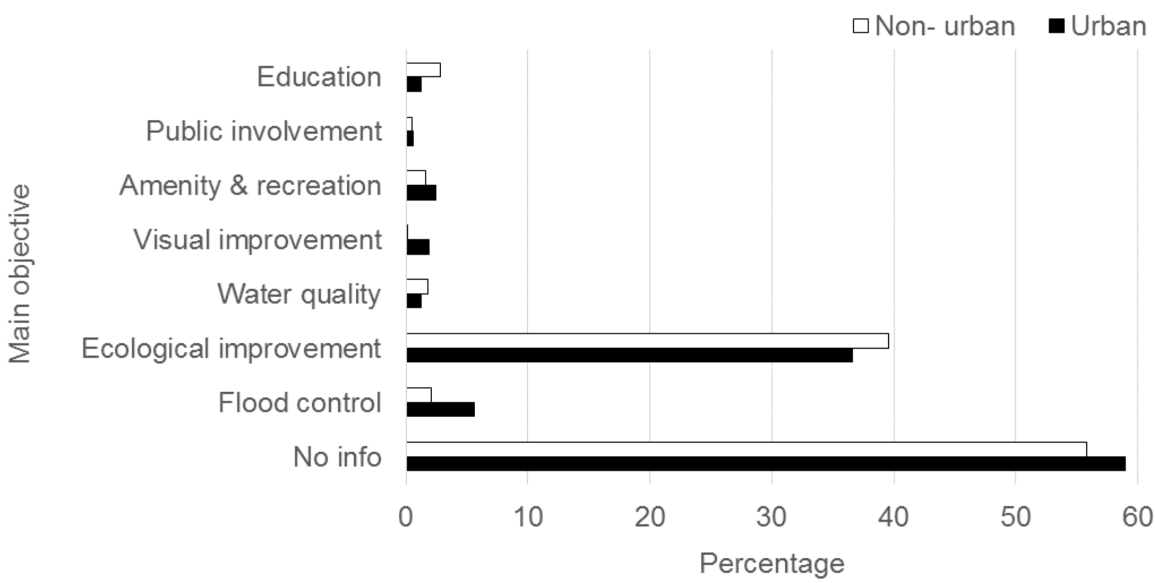

Figure 3-10: Main project objective of each urban (n= 161) and non-urban (n= 670) case study from the REFORM WP1 database.

Monitoring of river rehabilitation case studies occurred in 43% of urban case studies (n=161) and 64% of non-urban case studies (n=670). Of those urban and non-urban case studies monitored, measures of success for biological, morphological and physico-chemical parameters were recorded (Figure 3-11). Successful outcomes were always lower for urban case studies, than non-urban works for biological (14%, 20% respectively), morphological (9%, 14% respectively) and physico-chemical (2%, 5% respectively) parameters (Figure 3-11). Less than 5% of urban and non-urban case studies recorded project failure and < 10% were unclear on project findings, with the exception of 12% of non-urban case studies unclear if biological success had been reached (Figure 3-11).

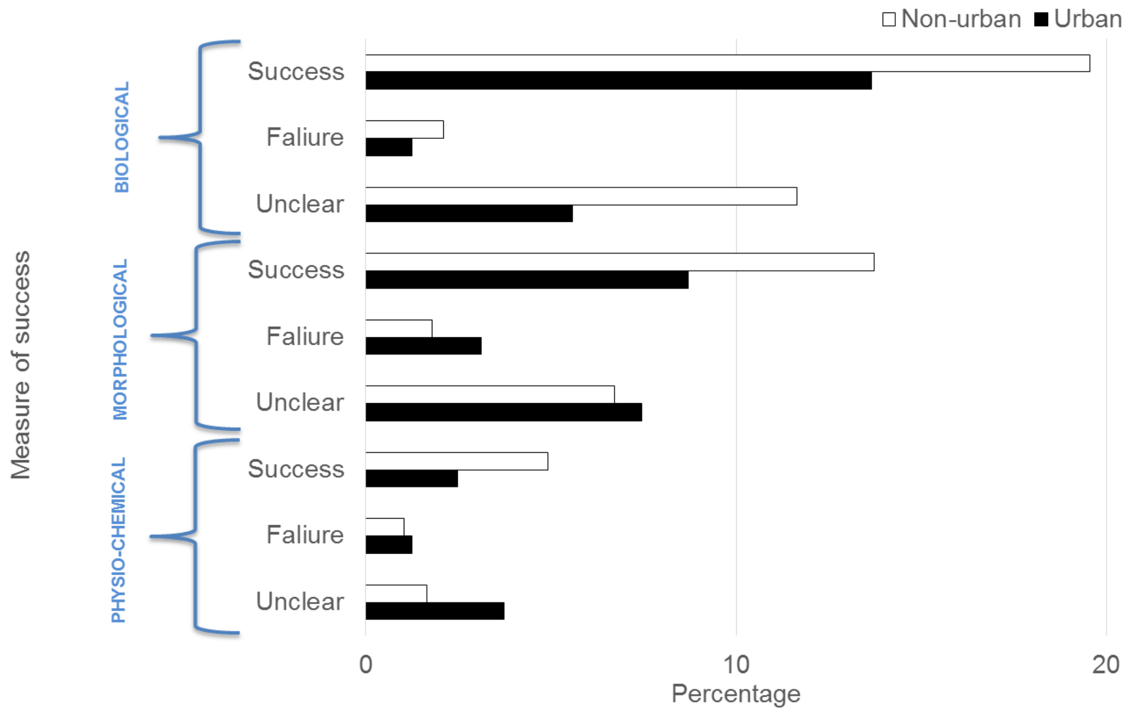

Figure 3-11: Level of success for biological, morphological and physico-chemical components of urban and non-urban case studies from the REFORM WP1 Database (multiple measures of success recorded for individual case studies).

Since monitoring occurred in less than half of urban case studies provided in the WP1 database, it seems prudent to provide an example of a case study on urban restoration. Such an example is provided by Malin Bridge, Sheffield, UK (Appendix A). This is an example where rehabilitation has been carried out in conjunction with FRM intervention. This gives a typical example of the constraints under which urban river rehabilitation has to act and how an opportunistic approach has been adopted to benefit improving the ecological status.

3.4 Discussion

The rapid expansion of urbanisation across Europe and elsewhere as a result of population growth and economic expansion has significantly impacted on rivers, primarily because of competing land uses and development practices that often result in a number of constraining pressures that differ from those in rural areas (Tourbier et al. 2004). The main issues (cause effect) resulting from urbanisation are highlighted in the problem tree (Figure 3-12). Pressures in urban areas impact on morphological, hydrological, physico-chemical and biological components of freshwater systems through impoundment and channelization, increased impervious surfaces, water abstraction, pollution, increased sedimentation, alteration of riparian vegetation and instream habitats (Dynesius & Nilsson 1994; Aarts *et al.* 2004; Pyrce 2004; Reid 2004; Vinebrooke & Cottingham 2004; Vaughn *et al*. 2009; Schinegger et al. 2011; Figure 3-3).

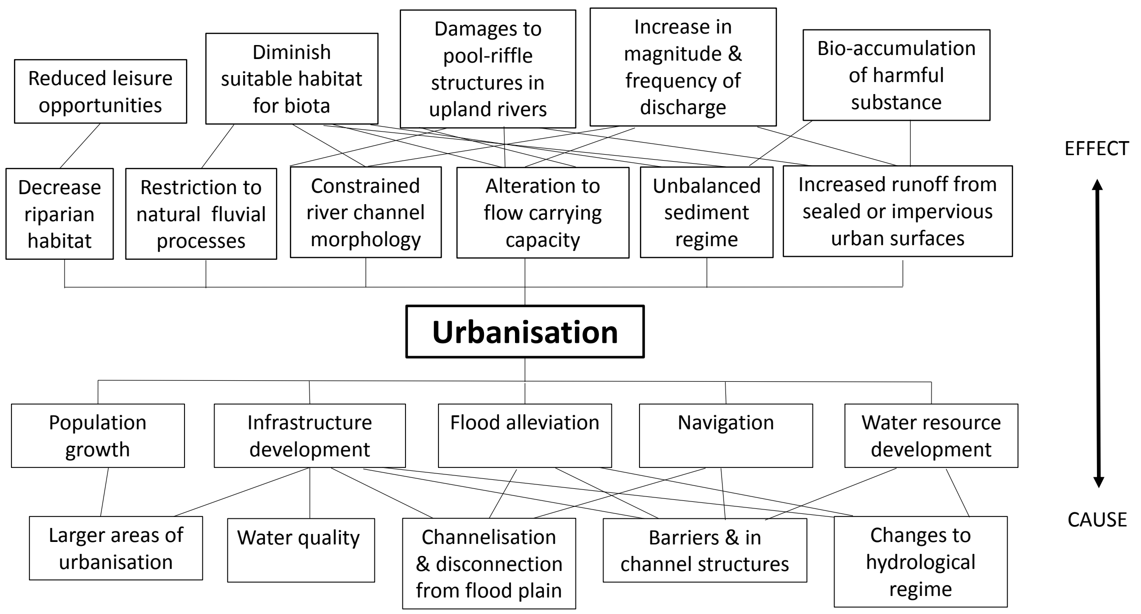

Figure 3-12: Problem tree (cause effect) of urbanisation on river form and functioning.

As a result of these various interventions, urban rivers are now considered heavily modified and under the EU WFD should be the target of rehabilitation activities to reach GEP. Consequently urban rivers are becoming an important focus for rehabilitation in Europe and rehabilitation practise is likely to expand further as urbanisation continues and demands for a sustainable, but enhanced quality of life increases (Clifford 2007). Information collected on a European scale, though the REFORM WP1 database provided a large scale overview of urban river issues and potential rehabilitation practices in Europe. It also highlights potential dissimilarities in the measures applied and their suitability between urban and non-urban river systems.

In 2003, the ICE (2003) stated that "All urban water courses, no matter how small, should be considered for rehabilitation back to nature". Subsequently, urban river and wetland rehabilitation has gained more focus over the past 10 years and the number of peer review journal articles published annually has more than doubled since 2003. Nevertheless this is starting from a low base and progression has been slow compared with non-urban river rehabilitation activities (Figure 3-1). Analysis of the REFORM WP1 database identified only a small percentage of case studies categorised as urban (18%)

within the total number of case studies recorded. This demonstrates that there is a distinct difference in either the approach, or perception, of urban rehabilitation compared with non-urban river rehabilitation. However, further exploration of the WP1 database found considerable overlap in the rehabilitation actions (measures) addressing the main pressures acting on urban and non-urban rivers system (Figure 3-2 and 3-4). Likewise, only small divergences were found between the proportion of specific pressures occurring and specific measures applied in urban and non-urban case studies (Figure 3-4 and 3-5). Overall, the results indicated that approaches towards urban and non-urban river rehabilitation practices were similar in urban and non-urban restoration projects, which is somewhat contradictory of existing knowledge that suggests urban rivers are treated differently as they have a larger number of multiple pressures that limit rehabilitation measures (Findlay & Taylor 2000; Hobbs 2002; Tourbier et al. 2004).

Subdividing urban and non-urban case studies into 5 size categories enabled further investigation in an attempt to disentangle similarities between urban and non-urban pressures and restoration measures between small, medium and large rivers. It was hypothesized that pressures and measures differ between rivers of different size since the space needed, and hence restrictions for restoration increases with river size in an urban setting.

In respect to the pressures reported, the correspondence analysis (Figure 3-3) confirmed there were similarities between pressures on small to medium urban and non-urban rivers, of the same size category (>5 m, 5-10 m and 10-20 m wide), however, pressures acting on larger rivers (20-50 m and > 50 m wide) were not so similar (Figure 3.3). Morphological alterations were actually a frequent pressure occurring in all size categories of urban and non-urban rivers, the similarity between <10 m width case studies and their dissimilarity with all other size categories was because they only recorded a small amount of other pressures. A variety of pressures (interbasin flow transfer, river continuity and ground water abstraction) were most frequently reported for 10-20 m wide urban and non-urban rivers and therefore separated them from others size categories. This is to be expected, especially regarding river continuity as it is a key issue throughout catchments, in both urban areas where weirs were introduced for industry and in non-urban areas where weirs were introduced to hold water back for agricultural purposes and improve reaches for angling. Large urban rivers were the most divergent, from all case studies because a lack of riparian vegetation and a high number of impoundments recorded. This is surprising because small urban rivers also suffer from a lack of riparian vegetation. The similarity between urban and non-urban pressures is still unexpected. Perhaps pressures have been recorded only for the restored reach for each case study and therefore, they do not consider the surrounding land use of the reach or catchment, consequently overlooking complex urban pressures such as large areas of impervious surfaces.

Similar to the pressures, urban and non-urban case studies did only slightly differ in respect to the measures applied even if rivers of different size were distinguished (Figure 3-7). When measures were grouped into the four main measures (instream restoration, riparian zone, river planform and lateral connectivity) they appear to be applied in the same way, for urban and non-urban case studies within all width categories, with the exception of riparian zone that resulted in a significant difference for <5 m urban (27%) and non-urban (15%) case studies. Riparian vegetation as a measure for small urban rivers is to be expected, especially as they tend to be simplistic rehabilitation actions that can be applied in highly constrained channels in heavily urbanised environments that are restricted by space, water availability and political override.

It was only for specific single measures that urban and non-urban rivers differed: Urban case studies of <5 m, 5-10 m and 10-20 m differed when compared to the same size categories as non-urban. Boulder placement and instream vegetation were applied for small to medium non-urban case studies but not for urban of this size. This is to be expected, particularly because urban river restoration has to be integrated with actions that underpin the EU Floods Directive (2007/60/EC), to reduce the risk of flooding to property and infrastructure. Instream measures such as boulder placement and instream vegetation can take up valuable space and reduce the carrying capacity of water needed during a flood, therefore if such measures are to be introduced into an urban area, it has to be done with caution and in collaboration with flood risk teams. However, the removal of bed fixation is an expected measure for urban river channels as they have artificial beds in comparison to non-urban river beds. Creation of a secondary floodplain was a measure that was applied more frequently in small to medium urban rivers compared to non-urban rivers in the same size category. This finding is unusual, especially given that there are significant constraints for space in urban areas. However, most of urban case studies where this secondary floodplains have been build were located in the suburbs (Figure 3-9), where residential areas with green space are available for small lateral modifications of the river channel. An example is the River Brent park project in North West London, England, where a small urban channelized river in the suburbs was modified to create a secondary flood plain.

Within each measures category there was only a significant difference between urban and non-urban case studies applying instream restoration, the difference possibly due to a higher number of <5 m non-urban case studies, in comparison to <5 m urban case studies (Figure 3-7). Even so, the correspondence analysis identified the dissimilarity in measures between urban and non-urban was reversed as river width increased to 20-50 m and instream measures (creating shallows, removing bank fixation, creating artificial riffles and the removal and addition of sediment) were applied more frequently to urban rivers (Figure 3-8). Whereas, river planform (creating low-flow channels in over-sized channels) and lateral connectivity (remove embankments, levees or dikes that impede lateral connectivity) measures were applied more frequently to non-urban 20-50 m case studies (Figure 3-8). Although creating artificial riffles and shallows can reduce the capacity of water in the local area, if managed right, it should not increase flood risk.

Subdividing the urban case studies into intraurban (city centres) and sub-urban enabled to investigate if the degree of urbanization affected which measures have been applied in the restoration projects. The results indicated that measures with a higher need for space (i.e. planform measures) have been applied less frequently in intraurban areas compared to sub-urban case studies (Figure 3-9). This further indicated that land use pressure in the case studies classified as sub-urban was not substantially higher compared to the non-urban restoration projects or at least not high enough to affect the selection of restoration measures.

Urban and non-urban case studies clearly differed in respect to the river size and restored reach length, which was significantly smaller and shorter in urban compared to non-urban restoration projects.

The uneven distribution of European countries represented in the WP1 database and literature for urban and non-urban rivers could influence the results. The information is concentrated in Western Europe lowland rivers where population densities are high and activities restricted by space available. Furthermore Germany contributed to the majority of cases studies in the database, followed by France, UK and Austria. These countries are known to have a large number of modified rivers in both urban and non-urban areas. Land use change for urban or agricultural development is a particularly significant cause of global ecosystem degradation (IUCN 2009). Agricultural intensification in Europe substantially modified the land through land drainage and flood defence schemes (Scrase & Sheate 2005). It would be of interest to gather more information on river restoration case studies from across the whole of Europe, to have a wider overview. More stringent policies in particular countries and insufficient reporting also limit the representation of some countries not only in the WP1 database, but also across other research projects such as FORECASTER, WISER and RESTORE.

Evaluating how successful a rehabilitation project has been, as well as determining reasons for success or failure, are essential if rehabilitation measures are to be carried out in an efficient and cost effective manner. Nevertheless, like in non-urban rivers (Cowx et al. 2013) there is uncertainty with regards to the success of urban river restoration; most projects are either not monitored or are poorly monitored (Bernhardt et al. 2005). Analysis of the WP1 database suggested monitoring was only carried out on 43% of river rehabilitation case studies and 64% of non-urban case studies. From the urban and non-urban case studies monitored, less than 20% identified biological, morphological or physico-chemical success and less than 5% reported project failure. Not only this, but many of the monitored case studies were unclear about the measure of rehabilitation success. The measure of success was always lower for urban case studies than non-urban for biological, morphological and physico-chemical parameters. It is evident that there are constraints that hinder the identification of rehabilitation success, especially for urban cases studies. Scientists and river rehabilitation practitioners have only recently begun to understand the importance of long term monitoring and evaluation of biological condition before and after rehabilitation actions (Lepori et al. 2005; Tullos et al. 2009; Miller et al. 2010). Nevertheless, rehabilitation of urban streams is still not resulting in the recovery of biological, morphological or physico-chemical parameters and therefore, little information has been gathered about successful programmes because evaluation of rehabilitation measures are limited (Roni et al. 2005; Klein et al. 2007; Wolter 2010).

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4. Summary and conclusions

The overall objective of the report was to evaluate hydromorphological restoration projects based on existing data. In a meta-analysis of peer-reviewed literature and original monitoring data, the effect of restoration on biota was quantified, parameters which influence restoration success and favourable conditions were identified, and interactions between these predictors and their relative importance were assessed. The meta-analysis was complemented by a satellite topic on urban restoration to identify differences between the characteristics of urban and non-urban restoration projects.

The following aspects should be considered for interpretation of the results:

- In principle, it is not possible to infer causal relationships from statistical analysis. Therefore, although it is tempting to draw such conclusions for river management, we caution against oversimplified interpretations. However, if the relationships are ecologically meaningful, it is generally accepted to consider the statistical results an indication of a cause-effect relationship.
- Given the relatively high scatter and uncertainty in ecological data, specific (threshold) values should not be considered sharp limits.
- Transferability of the results is limited to similar catchment, river reach, and project characteristics.
- In this report, restoration success refers to an increase in the number of individuals and taxa simply because these metrics were reported in literature. However, other metrics might be better suited to quantify restoration success if the general objective of restoration is not simply to increase biodiversity or abundance for its own sake, among other reasons because an increase in the number of individuals or taxa is not necessarily tantamount to a better or more natural biological state (see discussion in section 2.3).
- Most restoration projects applied several measures, monitoring results reflect the response of biota to all these measures, and hence, it was difficult to disentangle the effect of the single measures, which is a fundamental problem for the monitoring of restoration effects and the assessment of the (cost-) effectiveness of measures.

However, despite all the limitations discussed above, the following conclusions can be drawn based on the report and results from other comprehensive studies (replicate studies on a large number of restoration projects):

Overall, the effect of restoration on biota is positive but variability is high.

Overall, hydromorphological restoration has a positive effect on floodplain vegetation, ground beetles, macrophytes, fish, and invertebrates (section 2.2.3, Januschke et al. 2009, Jähnig et al. 2009, Miller et al. 2010, Haase et al. 2012, Lorenz et al. 2012, Schmutz et al. 2014). Since variability is high (section 2.2.3), adaptive management approaches are recommended which encompass monitoring the effect of restoration in a specific project and including from the beginning alternative restoration strategies in the planning process in case the measures do not have the expected effect (Downs and Kondolf 2002).

Restoration effect differs between organism groups.

Not all organism groups benefit from restoration to the same extent. Results indicate that, in general, restoration effect on diversity is highest for terrestrial and semi-aquatic groups like floodplain vegetation and ground beetles, intermediate for macrophytes, lower for fish, and lowest for macroinvertebrates (section 2.2.3, Januschke et al. 2009, Jähnig et al. 2009, 2010, Palmer et al. 2010, Haase et al. 2012, Lorenz et al. 2012). This is especially true for widening/rebraiding and remeandering measures (section 2.2.4), which have been widely applied and investigated in this study and the references cited above. It is reasonable that these measures have a large effect on terrestrial and semi-aquatic species since they usually result in pioneer habitats like bare riparian areas and bare gravel bars, reduce flow velocity and water depth, restored reaches are often sparsely shaded in the beginning, and hence, favour pioneer species in the riparian area and macrophytes in the aquatic zone in the first years.

Restoration has a higher effect on the number of individuals than on the number of taxa.

The effect of restoration is more pronounced on the number of fish and invertebrate individuals than on the number of taxa (section 2.2.3). This is reasonable from an ecological point of view since the abundance of species which are already present in a restored reach directly benefit from restoration while the (re-)colonization of new habitats by new species strongly depends on the number and location of source populations, the dispersal abilities of the species, and the number and location of migration barriers. Recent studies indicate that the number and location of source populations is even more important for the (re-)colonization of restored habitats than migration barriers (Stoll et al. 2014, Tonkin et al. 2014), especially if source populations and barriers are evenly distributed in the river network (Radinger and Wolter 2014).

Restoration effect does only slightly differ between measures, i.e. there is no single "best" measure.

There are no large differences in the overall effect of different measures but there is a tendency that terrestrial and semi-aquatic organism groups like floodplain vegetation and ground beetles as well as macrophytes benefit most from planform measures and aquatic groups like fish and invertebrates from instream measures (section 2.2.3, Januschke et al. 2009, Jähnig et al. 2009, Miller et al. 2010, Haase et al. 2012, Lorenz et al. 2012). This is reasonable, since many instream measures like the placement of large wood or riffle creation target aquatic habitat conditions and often immediately increase aquatic habitat and substrate diversity. These new aquatic habitats can be quickly colonized by new aquatic species given that respective source populations are located nearby. In contrast, especially planform measures like widening and remeandering also affect instream habitat conditions but first of all elongate the river reach and do not necessarily increase substrate diversity (e.g. re-meandering of a pure sand-bed lowland river with limited potential for substrate sorting). However, channel-planform measures often create terrestrial and semi-aquatic pioneer habitats like bare ground and open gravel bars. In this meta-analysis, the restoration projects were assigned to one single main measure, which was surprisingly straightforward for many projects (section 2.1.3). However, in most projects, one or several additional measures were applied, which hampered assessing the effect of single measures. Subdividing the dataset according to the combinations of measures applied resulted in sample sizes too small for statistical analysis, and hence, larger datasets which focus on specific (common or promising) measure combinations are needed to derive more robust and practically relevant recommendations on the (cost-) effectiveness of measures.

Urban restoration projects do not substantially differ in respect to the pressures occurring and the measures applied.

Urban restoration projects were mainly applied in small rivers (<5m in width) and length of the restored reaches was even shorter compared to non-urban restoration projects. However, the results indicated that approaches towards urban and non-urban river rehabilitation practices were similar in urban and non-urban restoration projects, and it was only for specific single measures that urban and non-urban rivers differed. This is somewhat contradictory of existing knowledge that suggests urban rivers are treated differently as they have a larger number of multiple pressures that limit rehabilitation measures (Findlay & Taylor 2000; Hobbs 2002; Tourbier et al. 2004). Most of the urban case-studies in the database actually were located in sub-urban areas, indicating that land use pressure was not substantially higher compared to the non-urban restoration projects or at least not high enough to affect the selection of restoration measures. The few intraurban projects located in city centres indeed differed in respect to the measures applied.

Urban restoration projects are rated less successful compared to non-urban projects by scientists and river managers.

The subjective rating of the success by the operators compiling the database was always lower for urban case studies than non-urban for biological, morphological and physio-chemical parameters. This is somewhat contradictory to the results of the meta-analysis that urban land use did not affect restoration success. However, in meta-analysis, effect sizes are used which usually quantify the relative effect of the restoration project (section 2.1.1), and the absolute effect might be low and considered a failure by scientists and river rehabilitation practitioners.

Conditions which favour restoration success can be identified but restoration outcome cannot be predicted.

Restoration success is especially high under the following conditions (section 2.2.5):

- Fish number of taxa: Implementation of instream measures in gravel-bed rivers.
- Fish number of individuals: In catchments with a relatively low share of agricultural land use and in medium aged projects (after several years).
- Macroinvertebrate number of taxa: Implementation of instream measures after several years (relatively old projects).
- Macroinvertebrate number of individuals: After at least about one year (relatively old projects).
- Macrophytes number of taxa: In catchments with a relatively low share of agricultural land use.
- Macrophytes abundance: In the first years after implementation of measures in catchments with a relatively low share of agricultural land use.

However, the predictor variables are often co-correlated and hence, these findings should be interpreted with caution. Furthermore, about two thirds of the differences in the restoration success cannot be explained by these variables, and hence, it is not possible to predict the outcome of restoration. This stresses the need to apply adaptive management approaches.

Overall, restoration success most strongly depends on project age, river width, and is affected by agricultural land use.

Overall, the most important predictors affecting restoration success are project age, river width, and the percentage coverage of agricultural land use in the upstream catchment (section 2.2.5).

Many studies show that catchment land use affects the biological state of rivers, especially macroinvertebrates (Roth et al. 1996, Allan et al. 1997, Stephenson and Morin 2009, Sundermann et al. 2013, Kail and Wolter 2013). Therefore, it is not surprising that agricultural land use is also among the most important variables affecting restoration success. It is important to note that, although success is generally lower, restoration still has a positive effect in catchments dominated by agricultural land use, and hence, these findings do not question the implementation of restoration projects in intensively used catchments in general. Catchment land use is not a pressure per se and rather a proxy for water quality aspects like pesticide, nutrient or fine sediment input from agricultural areas or the general conditions in the river network like a low number of source populations. Rivers are also inherently buffered from adjoining land use by their riparian zone, so that condition of the riparian zone will have a strong modulating effect. Therefore, there is an urgent need to identify the underlying causal relationships.

Project age is the most important predictor affecting restoration success, which supports the assumption that the effect of restoration is affected by time-lags and hysteresis effects in the recovery process (Kail and Hering 2009, Sundermann et al. 2013). However, the number of macrophyte taxa was smaller in older projects, indicating that restoration effect rather decreases over time for this organism group. There are also other possible explanations (see section 2.2.4), but it seems most reasonable that the number of macrophyte taxa increases in the first years since favourable habitats which are created by the widening and remeandering projects are colonized rapidly from existing seed banks in the restored reaches or by drift of propagules, resulting in the large effect of restoration on the number of macrophyte taxa. In the following years, channel features probably mature (e.g. riparian vegetation develops which increases shading), and hence abundance but not the number of macrophyte species decreases. However, there is limited empirical data to test these hypotheses, and an urgent need for long-time monitoring to investigate the restoration effect over time, to better understand the trajectories of change induced by restoration measures, and to identify sustainable measures which enhance biota in the long-term.

In summary, it is possible to draw some first conclusion for river management from the evaluation of hydromorphological restoration based on existing monitoring data. However, monitoring data are still scarce and more robust, practical relevant, and quantitative results (e.g. thresholds) could be derived and river management would benefit from (i) original monitoring data, which would allow to use functional metrics to investigate the underlying processes and to infer causal relationships, (ii) full before-after-control-impact monitoring designs, which most probably would substantially decrease scatter in the datasets and analyses, (iii) a larger number of monitored projects, which easily could be accomplished since a large number of hydromorphological restoration measures will be implemented in the upcoming years, (iv) the availability of long-time monitoring data sets to investigate the effect of project age, which was identified as the most important variable affecting restoration success. A more intensive exchange and collaboration between river science and river management in planning monitoring programs is strongly recommended. This would offer a great opportunity to make fundamental advances in our understanding of how river restoration affects river hydromorphology and biota and to identify (cost)-effective restoration measures.

Appendix

A. Malin Bridge case study

An introduction to Malin Bridge case study

Since the year 2000, Europe has suffered more than 175 major floods that had devastating effects on people, property, infrastructure and economy (EEA 2012). Climate change predictions indicate the likely increase in the risk of flooding from rivers caused by excessive rainfall, especially in urban areas where the transport of water is accelerated into rivers that are typically constrained and inadequate to cope with the increased flow volumes (Leopold 1968; Finkenbine et al. 2000; Andjelkovic 2001; Paul & Meyer 2001; Zevenbergen & Gersonius 2007). Flood risk management (FRM) measures are put in place to reduce the impact of flooding and are dependent on political support through legislation, such as the European Floods Directive (EU FD) and in the UK, the Flood and Water Management Act ((FWMA) 2010). Historically, flood protection methodologies in Europe were inclined to support economic and social impacts more than environmental impacts, compromising ecological integrity (Andjelkovic 2001). In recent years FRM approaches have progressed through the integration of river restoration, to lessen its impact and in some cases improve the ecological status of rivers. Therefore, FRM can provide multiple benefits and the opportunity to meet obligations under the European Water Framework Directive (EU WFD) and Habitats Directive (Mainstone & Holmes 2010).

Urban developments are frequently situated on floodplain areas and vertical embankments often restrict lateral flood movement (Everard & Moggridge 2012). The removal of instream features such as trees and islands, by FRM, is a common method to provide a larger volume for water to move through the channel at a faster rate, reducing the effects of flooding on property and infrastructure (McCarthy 1985; Pretty et al. 2003). FRM actions on an already channelized river will increase pressures that restrict ecosystem functioning by reducing river complexity as a direct consequence of deepening and widening the river, removing meanders and eliminating instream riparian habitat features (Hammer 1972; Douglas 1975; Roberts 1989; Booth 1990; Cowx & Welcomme 1998). This can therefore have an undesirable effect on aquatic biota, in particular fish, by diminishing refuges for their feeding and breeding (Brookes 1985; Wilcock & Essery 1991; Hodgson & O'Hara 1994; Cowx & Welcomme 1998; Bernhardt & Palmer 2007). Flood protection activities are predicted to intensify in the future because of an increase in extreme flow events (Booth & Jackson 1997; European Commission 2009; Nelson et al. 2009), but few case studies provide ecological monitoring and evaluation for the integration of FRM and river restoration. The objective of this case study is to provide the outcomes of FRM measures on the fisheries in an urban river.

In England, the Environment Agency is responsible for delivering sustainable FRM whilst reducing potential impacts associated with flood alleviation works through mitigation measures that conserve and enhance the environment (Environment Agency 2010). Following the June 2007 floods in the Sheffield area, the Environment Agency planned and managed FRM works at Malin Bridge, a location on the suburbs of Sheffield where the Rivers Rivelin and Loxley meet. Restoration works were incorporated into the planning to reinstate habitat features and return brown trout (*Salmo trutta* L.)

populations to their base line before FRM works took place. The following case study provides an insight into the effects of FRM and river rehabilitation works on the local brown trout community. Brown trout were selected to monitor the effectiveness of the FRM and rehabilitation schemes because they are categorised as a low tolerance/sensitive species and will be most responsive to rehabilitation works (Karr 1991; Schiemer 2000; Sedgwick 2006). Specific objectives were to compare brown trout 1) habitat quality, 2) population density and 3) population structure pre and post FRM and river rehabilitation works.

Methods

Case study background

Malin Bridge is in the suburbs of Sheffield where the rivers Rivelin and Loxley meet (Figure A-1). The area is representative of the trout zone of temperate rivers in an urban location where property and infrastructure are within 10 m of the river and consequently, there are many urban pressures such as channelisation and river fragmentation. Both rivers are categorised as heavily modified water bodies (HMWB) through the WFD river basin management plans (RBMP) and therefore, need only reach good ecological potential (GEP).

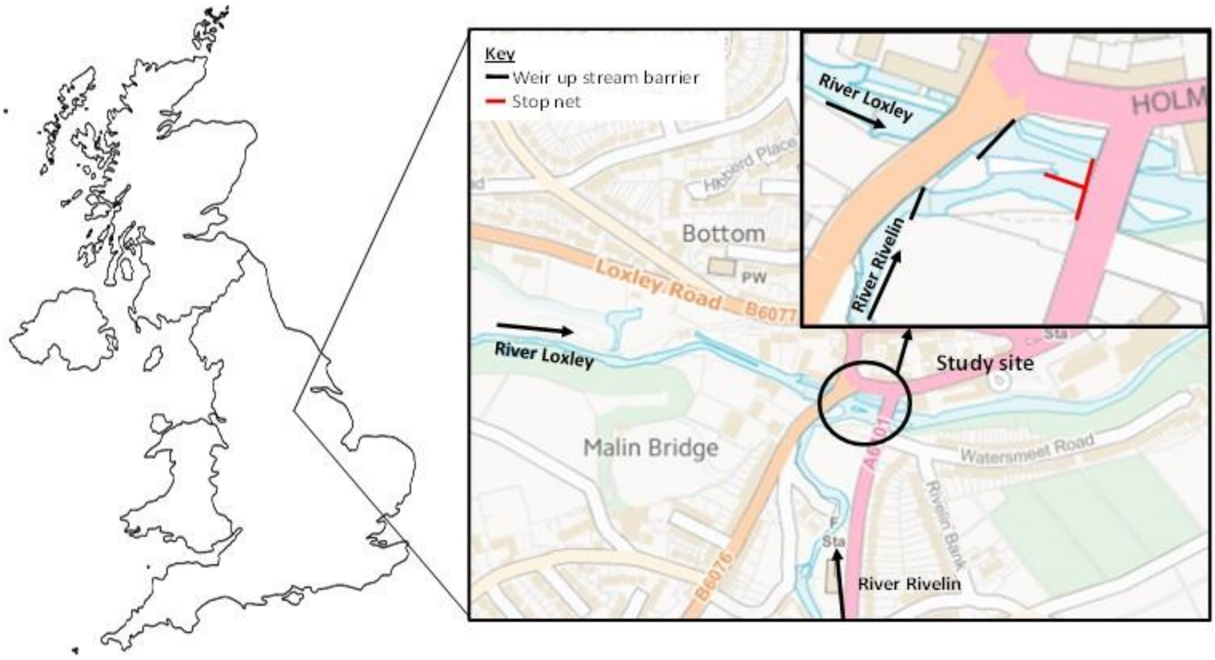

Figure A-1: Malin Bridge case study site where rivers Loxley and Rivelin meet (NGR: SK32578932 between A6101 and B6079 bridges).

Over the years a shoal island (created by the deposition of silts and gravels) had been deposited immediately downstream of the first road bridge, where trees and other vegetation had colonised, reducing the cross Sectional area of the river channel (Figure A-2). The expansion of the island formed serious obstructions, reducing the area available to flood water and therefore, increasing the risk of localised flooding. After the 2007 flooding at Malin Bridge, the Environment Agency planned and managed FRM and restoration works. In 2009, shoal and tree removal was completed to reduce the risk of flooding (Figure A-2). Rehabilitation measures were completed in 2010 and 2011, to recreate the habitat previously found in the river through channel re-profiling, installation of a rock riffle and instream boulders (Figure A-2). Large boulders were used to frame the rock riffle and were of an adequate size not to move under high flows, while smaller material was used to infill between boulders (Figure A-2). Natural re-colonisation of vegetation was the method chosen to re-profile the channel with the intention for it to be maintained in the future (Figure A-2, 2012 and 2013).

Figure A-2: Point photography at Malin Bridge to show the morphological changes made to the River Rivelin (left) and Loxley (right) post-flood works in 2009 and post-restoration works in 2010 and 2011.

Data collection method

Fish survey methodology

Fisheries surveys at the study site were carried out on 3 July 2009 (prior to flood defence works), 21 July 2010 (following flood defence works), 21 July 2011, 19 July 2012 and 20 August 2013 (following rehabilitation works) using quantitative electric fishing (estimates of absolute abundance based on a three-catch removal method (Carle & Strub 1978). The rivers Loxley and Rivelin were sampled separately and were isolated from each other with a stop net running from the island downstream to the A6101 road bridge. A downstream stop net (at the A6101 bridge) ensured there was no escape of fish from, or migration into, the sample areas. Rivelin weir provided an upstream barrier to fish movement on the Rivelin, as did Burgon and Ball weir on the Loxley.

The quantitative electric fishing strategy on the River Loxley involved three operatives (one anode operator and two people netting fish) fishing in an upstream direction, with a fourth operator on the bank supervising safe operation of the electric fishing equipment. A 2-kVA generator powering an Electra catch control box producing a 220 V DC output was employed. During the fishing exercise as many fish as possible were caught in dip nets by operatives positioned either side, and downstream, of the anode; the process was repeated for each run of the three-catch removal method with catches kept separate for data collection. The same methodology was used for surveying the River Rivelin once the catch had been processed from the River Loxley. Following each survey, brown trout were counted and measured to fork length (mm) before being returned to the river.

HABSCORE data collection: HABSCORE is a system for measuring and evaluating stream salmonid habitat features based on empirical statistical models relating the population size of five salmonid species/age (0+ salmon, >0+ salmon, 0+ trout, >0+ trout (<200mm), >0+ trout (>200mm)) combinations (Wyatt et al.1995). Using the information from three HABSCORE questionnaires, the software produces a series of outputs, which includes estimates of the expected populations (the Habitat Quality Score, HQS) and the degree of habitat utilisation (the Habitat Utilisation Index, HUI), for each of five salmonid species/age combinations (Wyatt *et al*. 1995). Salmon are not present in the study river and therefore, HABSCORE data collection and analysis only refer to brown trout. To collect information for HABSCORE analysis a questionnaire on the habitat found at each site was completed following electric fishing surveys. Information on channel width, depth, substrate, flow and sources of cover for >100mm trout were recorded. The methodology of habitat data collection and completion of the relevant form (HABform) are documented by Barnard & Wyatt (1995). To complete the datasets for further HABSCORE analysis two further forms for each site require completion, namely MAPform and FISHform (Barnard & Wyatt 1995). MAPform is completed by collection of relevant information from OS Maps (1:50000) and River Water Quality Maps (1:25000). FISHform is completed by recording of fisheries statistics of three brown trout age classification.

Density estimates and classification of population estimates:

Density estimates of brown trout in the rivers Rivelin and Loxley were derived from estimates of absolute abundance based on the three-catch removal method. Estimates of populations of 0+ and >0+ brown trout were calculated by the Maximum Likelihood Method (Carle & Strub 1978) along with their associated variances. In all cases the population densities were expressed as numbers/100 m^2 and the sampling area was calculated by multiplying the length of the sample site by the overall mean width of the sample site.

Density estimates were used to assess the status of the fish populations according to the Environment Agency Fisheries Classification Scheme (EA-FCS) (Table A-1) and were used in the derivation of HABSCORE outputs. Density estimates derived from surveys in each year were compared. The EA-FCS was developed to allow comparison of juvenile salmonid monitoring data with a juvenile database derived from over 600 survey sites in England and Wales (Mainstone et al. 1994). The classification of salmonid populations is based on a grading scale (A–F) and provides an indication of the status of salmonid populations in study rivers. The EA-FCS grading scheme is translated as follows: Grade A (excellent), Grade B (good), Grade C (fair or average), Grade D (fair/poor), Grade E (poor) and Grade F (fishless) (Table A-1).

Table A-1: Salmonid abundance (N/100m^2) classifications used in the Environment Agency Fisheries Classification Scheme (EA-FCS).

	Abundance classification					
Species group	**A**	**B**	**C**	**D**	**E**	**F**
0+ brown trout	**≥38.0**	**17.0-37.9**	**8.0-16.9**	**3.0-7.9**	**0.1-2.9**	**0**
≥1+ brown trout	**≥21.0**	**12.0-20.9**	**5.0-11.9**	**2.0-4.9**	**0.1-1.9**	**0**

HABSCORE analysis and outputs

HABSCORE outputs were derived using a combination of habitat data and fisheries data collected during the survey period to account for the changes to the habitat over time. In 2009, HABSCORE outputs were derived using habitat and fisheries data collected in 2009 (pre-flood defence works) while in 2010, HABSCORE outputs were derived using habitat data collected in 2010 and only fisheries data collected in 2010 due to the changes in the habitat. The same approach was used to derive HABSCORE outputs from 2011 surveys due to the changes in the habitat because of the rehabilitation works, i.e. habitat data collected in 2011 and only fisheries data collected in 2011. In 2012 to allow for temporal changes in fish populations HABSCORE outputs in 2012 were derived using habitat data collected in 2012 and fisheries data from 2011 and 2012 as no instream habitat modifications were made. In 2013, the HABSCORE outputs were planned to be derived using habitat data from 2013 coupled with annual fisheries data from 2011-2013, but because of the dramatic change in habitat structure due to high flows, particularly in the River Rivelin, it was deemed appropriate to only use fisheries data from 2013.

Data from the three completed forms (HABform, MAPform and FISHform) for the Rivers Rivelin and Loxley were entered into the HABSCORE for Windows program and the following outputs were produced for brown trout populations (definitions from Wyatt et al. (1995)):

HABSCORE population density

Densities were derived from estimates of absolute abundance based on the three-catch removal method and estimates of populations were calculated by the Maximum Likelihood Method (Carle & Strub 1978) along with their associated variances. The HABSCORE programme calculates brown trout density estimates for the three ages classes 0+, >0+ (<200 mm) & >0+ (>200 mm). Population densities for brown trout were expressed as numbers/100 m^2 and the total sampled area was calculated by adding the individual areas for each Section length.

Habitat Quality Score (HQS)

The HQS value is a measure of the habitat quality expressed as the expected long-term average density of fish (in numbers per $100m^2$). The HQS is derived from habitat and catchment features, and assumes that neither water quality nor recruitment are limiting the populations. The HQS is used as an indicator of the potential of the site, against which the observed size of populations may be compared.

Habitat Utilisation Index (HUI)

The HUI is a measure of the extent to which the habitat is utilised by salmonids. It is based on the difference between the 'observed' density and that which would be expected under 'pristine' conditions (i.e. the HQS). When the 'observed' density and the HQS are identical, the HUI takes the value of one; HUI values less than one will occur when the observed densities are less than expected.

HUI lower and upper confidence limits

These are the upper and lower 90% confidence limits for the HUI, expressed as a proportion. An upper HUI confidence interval <1 indicates that the observed population was significantly less than would be expected under pristine conditions. Conversely, a lower HUI confidence interval >1 indicates that the observed population was significantly higher than would normally be expected under pristine conditions.

Results

Density estimates and classification of population estimates

Overall, abundance categories found brown trout populations in 2013 (post flood and restoration works) had returned to their baseline classification of 2009 (pre flood and restoration works) or better, with the exception of >0+ brown trout in the River Loxley (Figure A-3). The presence of 0+ brown trout in the rivers Loxley and Rivelin in all surveys indicated annual recruitment; 2010 (post flood works) was a particularly good year for recruitment for both rivers (Loxley class B; Rivelin class A), but in 2012 (post flood and restoration works) recruitment was poor (class E) in the Loxley and absent (class F) in the Rivelin (Figure A-3). >0+ brown trout densities in the River Loxley decreased after flood works (2010; class C), but increased after restoration works in 2011 and 2012 (class B) (Figure A-3). >0+ brown trout densities in the River Rivelin increased after flood works (2010; class B) and continued to increase through the first stages of river restoration works (2011; class A), before decreasing to the previous state found before flood works (2013; class C) (Figure A-3).

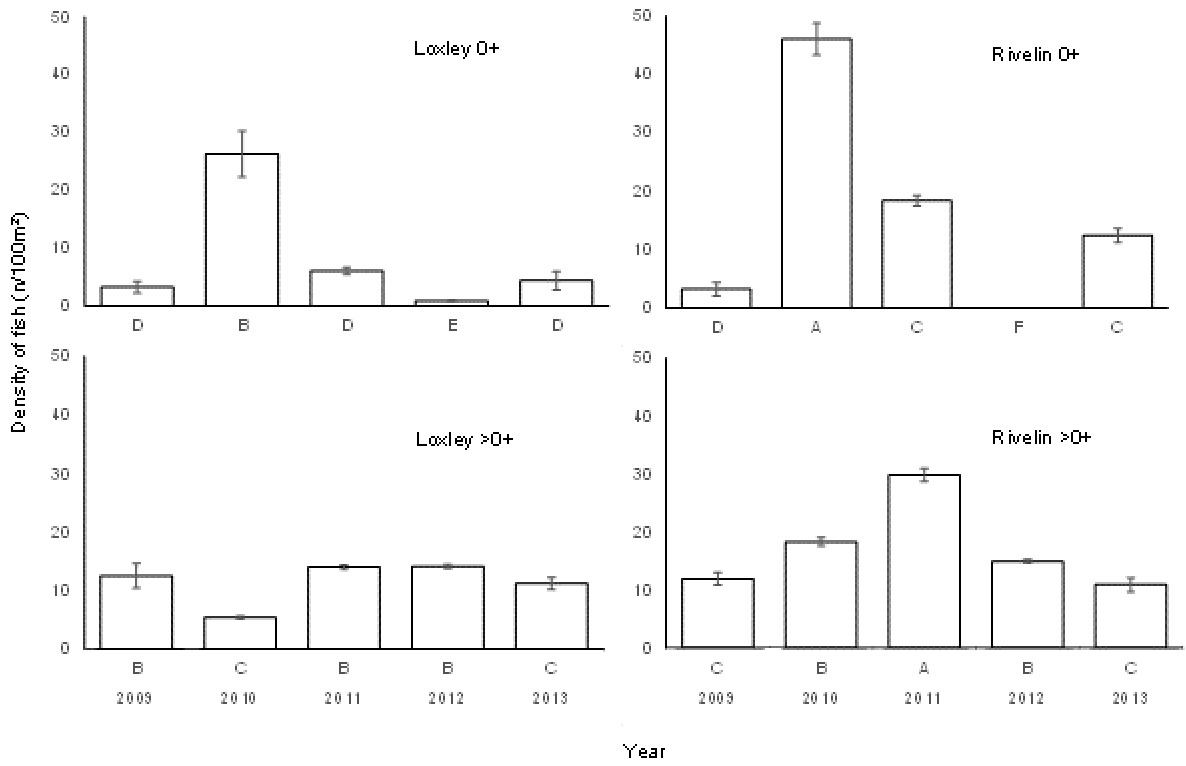

Figure A-3: Density estimates and confidence limits for 0+ and >0+ brown trout in the Rivers Loxley and Rivelin for 2009 (pre flood risk management), 2010 (post flood risk management) and 2011 to 2013 (post river rehabilitation). The Environment Agency Fisheries Classification (EA-FCS) is provided on the x axis for each year and is based on a grading scale (A = excellent, B = good, C = Fair or average, D = Fair/poor, E = poor and F = fishless; Table A-1).

HABSCORE analysis and outputs

HABSCORE outputs for the sites on the rivers Loxley and Rivelin revealed variations in the observed densities, predicted densities (HQS) and habitat utilisation (HUI (Ln)) by trout (Figures A-4 and A-5). HABSCORE data for the River Loxley at Malinbridge suggests that 0+ trout populations were significantly higher (HUI lower CL >1) than the predicted HQS, after flood works in 2010 (Figure A-4). Post-river rehabilitation works, 0+ tout were similar to the predicted HQS in 2011 and 2013, in 2012, 0+ trout densities were lower than the predicted HQS, but not significantly (HUI upper CL >1) (Figure A-4). Observed densities were higher than predicted HQS for >0+ (<200mm) trout across all years for the River Loxley, but only significant in 2009 and 2010 (HUI lower CL >1) (Figure A-4). Observed densities for >0+ (>200mm) trout were similar to predicted HQS in 2010, 2011 and 2012 (Figure A-4). Observed densities in 2009 were lower, but not significantly (HUI upper CL >1) and in 2013 observed densities were significantly higher (HUI lower CL >1) (Figure A-4).

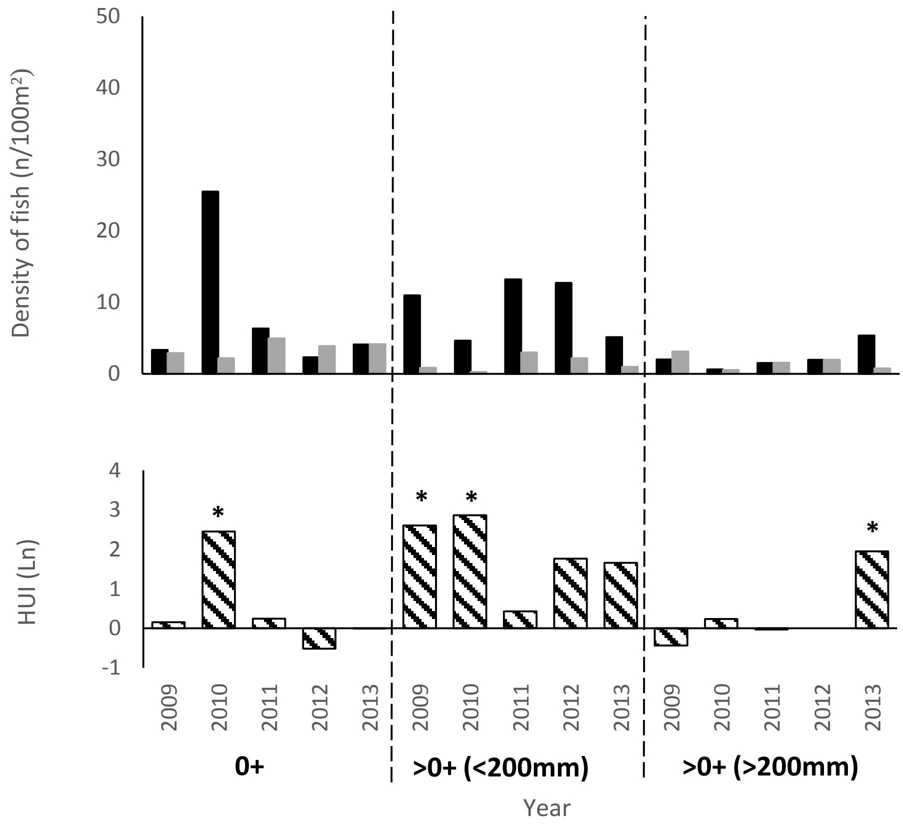

Figure A-4: HABSCORE outputs for the River Loxley at Malinbridge. ■ Observed density (calculated in the HABSCORE programme), ▩ HQS and ▧ HUI (Ln) for each year from 2009 (pre flood risk management), 2010 (post flood risk management) and 2011 to 2013 (post river rehabilitation) for 0+, >0+ (<200mm) and >0+ (>200mm) brown trout. *Represents sites where the observed population was significantly higher than would be expected under pristine conditions. FRM work was carried out between 2009 and 2010 fish and habitat surveys, several stages of river restoration were carried out between 2010 and 2012.

HABSCORE data for the River Rivelin at Malinbridge suggests that 0+ trout populations were significantly higher (HUI lower CL >1) than expected by the HQS after flood works in 2010, but not significantly higher (HUI lower CL <1) after restoration works in 2011 and 2013 (Figure A-5). Observed densities were lower than the predicted HQS in 2012, but not significantly (HUI upper CL >1) (Figure A-5). Observed densities for >0+ (<200mm) trout were higher than the predicted HQS and significantly so (HUI lower CL >1) for 2010 to 2013 catches (Figure A-5). Observed densities for >0+ (>200mm) trout were higher than the predicted HQS, but only significant in 2013 (HUI lower CL >1) (Figure A-5).

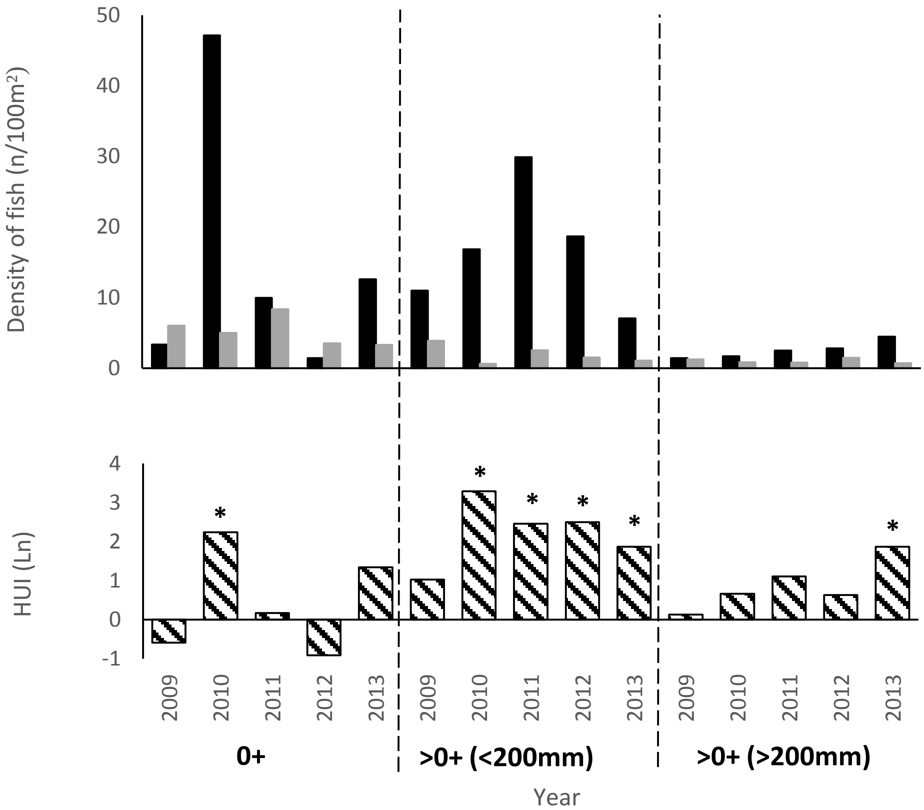

Figure A-5: HABSCORE outputs for the River Rivelin at Malinbridge. ■ Observed density (calculated in the HABSCORE programme), □ HQS and ▧ HUI (Ln) for each year from 2009 (pre flood risk management), 2010 (post flood risk management) and 2011 to 2013 (post river rehabilitation) for 0+, >0+ (<200mm) and >0+ (>200mm) brown trout. *Represents sites where the observed population was significantly higher than would be expected under pristine conditions. FRM work was carried out between 2009 and 2010 fish and habitat surveys, several stages of river restoration were carried out between 2010 and 2012.

Discussion

FRM at Malin Bridge increased the area available for flood water to pass through from the rivers Loxley and Rivelin. This was accomplished by modifying the river bed to a uniform state, reducing the length of the vegetated island and removing the majority of overhanging trees and bankside vegetation. The flood defence works decreased habitat quality (HQS) for all age categories of brown trout in the rivers Loxley and Rivelin and this was to be expected as FRM is known to reduce river complexity and ecosystem functioning (Hammer 1972; Douglas 1975; Roberts 1989; Booth 1990; Cowx & Welcomme 1998). Sequentially, poor habitat quality resulting from FRM would be expected to have detrimental effects on fish, by degrading habitat quality for feeding and breeding (Brookes 1985; Cowx & Welcomme 1998; Bernhardt & Palmer 2007). However, this was not always mirrored in brown trout abundance and the HUI. The presence of good numbers of 0+ brown trout present in the rivers Loxley and Rivelin post flood works indicated successful recruitment, 2010 being a particularly good year as 0+ trout populations were significantly higher than the predicted HQS for both rivers. The flood works created shallow (<30 cm deep), riffle habitat with moderately fast flowing water that is more favourable to juvenile trout (Crisp 2000) and possibly resulted in these good numbers of juvenile trout. By contrast, >0+ brown trout densities in the River Loxley decreased after flood works in 2010, but increased in the River Rivelin, suggesting the flood defence works caused deterioration in the quality of habitat for larger trout in the Loxley, but not the Rivelin. This could be because the flood works levelled the river bed limiting deep pools, with the exception of a weir pool that remained in the Rivelin. It is likely that larger trout from both rivers either moved out of the study reach to find more suitable habitat (Nordwall et al. 2001), or moved into the Rivelin weir pool, which would explain the increase of >0+ trout in the Rivelin.

Following the flood defence works the EA initiated a series of rehabilitation works in 2010/2011 to improve habitat diversity by channel re-profiling and installation of instream boulders (see Figure A-2). Overall, river restoration marginally improved the HQS for all age categories of brown trout in the River Loxley, but only improved the HQS for 0+ trout in the Rivelin. 0+ brown trout populations decreased in both rivers after rehabilitation (2011) in comparison to 2010's particularly good year for 0+ trout populations, but were still similar to the predicted HQS found prior to flood defence works (2009). In 2012, 0+ brown trout populations were very low for both rivers (not significantly lower than the predicted HQS), a feature found throughout the Don catchment suggesting a generally poor year for recruitment in the region. Low numbers of juvenile trout could be associated with the 2012 high late spring/early summer flows in Yorkshire that might have displaced trout alevins. An alternative suggestion for the low number of juvenile trout maybe that the restoration of the river to a more diverse channel has decreased the habitat suitable for 0+ trout and therefore intensified competition, especially as juvenile trout are known to be aggressive, defend territories and compete intensively for resources (Kalleberg 1958; Lahti et al. 2001). Nevertheless, 0+ brown trout populations in both rivers indicated that river habitat had successfully returned to a state to that before the flood and restoration works took place c.f. 2013 and 2009 fish data. >0+ brown trout densities in the River Loxley decreased after flood works in 2010, but increased after restoration works in 2011 and 2012. Whereas, >0+ brown trout densities in the River Rivelin increased after flood works (2010) and continued to increase through the first stages of river restoration works (2011), before decreasing to the state found before the flood works (2013). In addition, the higher densities of >0+ brown trout may also be a result of good recruitment and survival of 0+ trout found in 2010. The data suggest that rehabilitation after the flood defence works may have improved the quality of habitat for larger trout, perhaps a result of the introduced cover from boulders and the deepening of the river bed in places; these being known habitat preferences of large trout (Heggenes 2002; Cowx et al. 2004).

Climate change has a major impact on urban water resources in Europe and flood events are predicted to increase. As a consequence, pressures from flood protection activities will likely intensify. On the whole, abundance categories at Malin Bridge indicated that brown trout populations returned to their baseline classification, when comparing catch data pre flood and restoration works (2009) to post flood and restoration works (2013). This demonstrates that the potential impacts associated with flood alleviation works can be reduced when incorporating river restoration in to FRM, leading to positive outcomes. With the right project planning and collaborations, tradeoffs among conflicting goals, such as EU FWMA and the EU WFD, can generate win-win scenarios.

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Table A-2: Description of pressures

Pollution	
P_PointS	Pressure: Point Source
P_DiffS	Pressure: Diffuse Source
Water abstraction	
P_WatAb	Pressure: Water abstraction
WA_Surf	Surface water abstraction
WA_Ground	Groundwater abstraction
Flow regulation	
P_FloReg	Pressure: Flow regulation
PFR_Div	Discharge diversions and returns
PFR_Trans	Interbasin flow transfer
PFR_Reg	Hydrological regime modification including erosion due to increase in peak discharges
PFR_Peak	Hydropeaking
River fragmentation	
PFR_Flush	Pressure: River continuity
PRC_Up	Artificial barriers upstream from the site
PRC_Down	Artificial barriers downstream from the site
Morphological	
P_Morph	Pressure: Morphological alteration
PMo_Imp	Impoundment
PMo_Line	Channelisation / cross section alteration (e.g. deepening) including erosion due to this
PMo_Rip	Alteration of riparian vegetation
PMo_Bed	Alteration of instream habitat
PMo_Dike	Embankments, levees or dikes
PMo_Sed	Sedimentation
PMo_Dred	Sand and gravel extraction, dredging

Table A-3: Description of measures

Measures: In-channel habitat conditions (bed and bank)	
MIn_FixBed	Remove bed fixation
MIn_FixBank	Remove bank fixation
MIn_RemSed	Remove sediment (e.g. mud from groin fields)
MIn_AddSed	Add sediment (e.g. gravel, overlaps with MS_Add)
MIn_Veg	Manage aquatic vegetation (e.g. mowing)
MIn_HyStruc	Remove or modify in-channel hydraulic structures (e.g. groins, bridges)
MIn_Shall	Creating shallows near the bank
MIn_Wood	Recruitment or placement of large wood
MIn_Bould	Boulder placement
MIn_Dynamic	Initiate natural channel dynamics to promote natural regeneration
MIn_Riff	Create artificial gravel bar or riffle
Measures: Riparian zone (including buffer strips!)	
MR_NBuff	Develop buffer strips to reduce nutrient input
MR_SBuff	Develop buffer strips to reduce fine sediment input
MR_VegBuff	Develop natural vegetation on buffer strips (other reasons than nutrient or sediment input, e.g. shading, organic matter input)
Measures: River planform (e.g. cross-section river bed width and depth variation)	
MP_Meander	Remeander water course (actively changing planform)
MP_Wide	Widening or re-braiding of water course (actively changing planform)
MP_Shallow	Shallow water course (actively increasing level of channel-bed)
MP_Narrow	Narrow over-widened water course (actively changing width)
MP_LowC	Create low-flow channels in over-sized channels
MP_Dynamic	Allow/initiate lateral channel migration (e.g. by removing bank fixation and adding large wood)
MP_2Flod	Create secondary floodplain on present low level of channel bed ("Ersatzaue")
Measures: Lateral connectivity	
MFP_Con	Reconnect existing backwaters, oxbow-lakes, wetlands
MFP_Create	Create semi-natural / artificial backwaters, oxbow-lakes, wetlands
MFP_Lower	Lowering embankments, levees or dikes to enlarge inundation and flooding
MFP_Back	Back-removal of embankments, levees or dikes to enlarge the active floodplain area
MFP_Remove	Remove embankments, levees or dikes or other engineering structures that impede lateral connectivity
MFP_Veg	Measures: Other