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Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery

DELIVERABLE

Deliverable D5.1-2: Driver-Pressure-Impact and Response-Recovery chains in European rivers: observed and predicted effects on BQEs

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PU	Public	X
PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	
CO	Confidential, only for members of the consortium (including the Commission Services)	

Content

Non-technical summary

This report summarises the results of empirical analysis and literature surveys on both the response of biological assemblages to environmental pressures (stressors) and to pressure reduction (restoration/management) in river ecosystems. Part one of the report introduces the reader to the WISER WP5.1 database underlying all empirical analysis presented in the following. Part II continues with the analysis of pressure-impact relationships for four major groups of environmental pressures (physico-chemical/water quality, hydrological, morphological and land use-related degradation) and four biological assemblages (BQEs: benthic diatoms, macrophytes, benthic invertebrates and fishes). This part highlights the impacts of different stressors and its hierarchy with regard to the effects on different organism groups. The strengths (intensity) of relationships between pressures and organisms as well as the detectable stress (sensitivity) levels are referred to. Part III of the report is dedicated to the analysis of the effects of restoration on riverine organisms. This Chapter is building a on previous report (Deliverable 5.1-1) that outlined the conceptualised effects of river restoration and management on river biota based on a literature survey of restoration studies worldwide. Finally, part IV attempts to summarise the observed and predicted (predictable) effects and draws appropriate implications for River Basin Management to inform the restoration and management practitioners.

Summary

The report presented in the following is part of the outcome of WISER's river Workpackage WP5.1 and as such part of the module on aquatic ecosystem management and restoration. The module does also comprise potential effects of global and climate change, yet these will be subject to a separate Deliverable (D5.1-3) lead by CEMAGREF. The ultimate goal of WP5.1 is to provide guidance on best practice restoration and management to the practitioners in River Basin Management. Therefore, a series of analyses was undertaken, each of which used a part of the WP5.1 database in order to track two major pathways of biological response: 1) the response of riverine biota to environmental pressures (degradation) and 2) the response of biota to the reduction of these impacts (restoration).

This report attempts to provide empirical evidence on the environment-biota relationships for both pathways. Where the analysed data was not sufficient (e.g. limited spatial coverage, limited number of stations), literature surveys complement the results (or replace empirical outcome). Therefore, it is recommended to consult the previous Deliverable 5.1-1 with regard to the results presented in Chapter III.

Altogether, data on 4349 stations in ten countries was available for empirical analysis. Most stations are located in Germany, but rarely cover biological data on more than two organism groups in parallel. If all organism groups are considered in parallel, this data was available for <250 stations. The situation is similar for environmental (pressure) variables, which comprise nearly 70 single parameters, however, which are fully available for <700 stations only. Therefore, the individual studies presented in the following consider tailor-made sub-sets of the entire data base.

The impact of environmental degradation was analysed in the studies of Marzin et al. (French data), Dahm et al. (Austrian data) and Feld (French and German data). Except for the latter only, the studies considered physico-chemical, hydrological, morphological and land-use-derived pressures (also referred to as 'stressors'). Feld considered land use data only and compared the influence of different spatial scales (riparian buffer to whole catchment). In general, biological and ecological traits (metrics) were stronger related to degradation than the taxonomic structure of assemblages. All assemblages were strongly responding to water quality degradation, while the correlations were notably higher for the fauna (fish and invertebrates), when land use was considered. As for fish, macroinvertebrate metrics were very sensitive to morphological degradations such as the presence of an impoundment while diatoms and macrophytes metrics did not show strong responses to these changes. Fish metrics responded the strongest to hydrological perturbations. Overall, macroinvertebrate metrics seem to respond better to local than to catchment-scale pressures.

The correlations of fish, macroinvertebrate and macrophyte metrics with agriculture and forest land cover revealed strong scale-dependent patterns. This analysis identified the near-stream buffer land use to be strongly related to the ecological conditions, while correlations increased with buffer length. Thereby, overall correlations were much higher in mountainous than in

lowland ecoregions. The results imply that near-stream buffer land use is a strong predictor of ecological status in particular in mountainous regions, while the role of catchment-scale land use becomes more important in lowland regions. Along a gradient of percent land use as agriculture, many metrics significantly changed their values at 0–20% agriculture in mountain ecoregions and 30–50% in lowland ecoregions, irrespective of the buffer size.

The response of riverine biota to ecosystem management and restoration was studied by Lorenz, Melcher et al. and Keizer-Vlek et al. While Lorenz used empirical restoration data of about 50 restoration measures located in mountainous and lowland regions of western Germany, Melcher et al. based their meta analysis on a literature survey. Kaizer-Vlek in contrast used time-series data of a single (Vecht) catchment in the Netherlands to investigate biological recovery over time. A central finding of both empirical restoration and literature surveys is that restoration is likely to show effects, even at the local scale, if the important pressures are being tackled by restoration. For instance, local habitat enhancement (e.g. the introduction of spawning substrates for Salmonid fish) can immediately enhance the density of the targeted fish species. This is, however, usually not the case in intensively modified and heavily used agricultural catchments, where large-scale hydromorphological degradation and land use impacts above a restoration site typically superimpose the effects of restoration. Consequently, successful restoration in these catchments also requires large-scale hydromorphological improvements and land use management. Lorenz found in particular the land use conditions and hydromorphological structure (physical habitat) up to 10 km upstream of a restoration to control its biological effect. Interestingly, there was no significant effect detectable for the kind of restoration measure, although the trend was found towards a better improvement following large-scale integrated restoration tackling the bed, bank and riparian area in parallel. Finally, the time-series and other analyses in the Vecht(e) case study did not show any significant improvement over the past 15 years, probably because there were hardly any changes in management reported.

Finally, Verdonschot compared River Basin Management Plans of three countries and found two notable commonalities: 1) water-quality-related measures dominate until 2015 and physical restoration measures are mainly planned for the period after 2015 and 2) there is hardly any attempt detectable to improve restoration monitoring. This deficit in appropriate monitoring of biological and abiotic effects of restoration is the main reason for the lack of data that also largely limited the extent of empirical analysis in this report. This shortcoming is likely to continue to hinder restoration science and adaptive river management. This in particular refers to the sampling before restoration, which is hardly ever done.

The results presented in the following, together with the previous meta analysis of restoration studies conducted in Deliverable 5.1-1 imply that river restoration requires thorough planning and implementation. In order to be ecologically effective (and successful) appropriate restoration measures are those, and only those, that tackle *all* pressures impacting a site or stretch. As long as this prerequisite of ‘scaling appropriateness’ is not fulfilled, will (local) restoration hardly show the desired effects. Consequently, if large-scale land use is impacting a site or stretch, appropriate land use management will be required in parallel to habitat improvement.

CHAPTER I

The WISER WP5.1 river database

Database structure and component biological and environmental data

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Data origin

Representing the only river workpackage in WISER, WP5.1 agreed on collating available biological and environmental data from partner institutions and establishing an own database. This database hereafter is named "*the WP 5.1 large-scale database*" and contains monitoring data from 4349 stations in ten European countries (Figure 1, see also Figure 4 for a map): Austria (AT), Czech Republic (CZ), Denmark (DK), France (FR), Germany (DE), Netherlands (NL), Poland (PL), Sweden (SE), Slovakia (SK) and the United Kingdom (UK). While the majority of sampling stations were located in AT, DE, FR, NL and SE, comparatively few data was included from CZ, DK, PL, SK and UK. Data from the latter countries exclusively originated from the EU-funded Integrated Project 'STAR' (**Sta**ndardisation of **R**iver Classifications).

Table 1: Data included in the WP 5.1 large-scale database. Each row represents a single national or international (project) database. Database IDs correspond with the IDS used in the WISER Meta Database at http://www.wiser.eu/programme-and-results/data-and-guidelines/meta-database/. Biological Quality Elements are abbreviated as FI (fish), BI (benthic invertebrates), MP (macrophytes) and PB (phytobenthos)

DatabaseID	Database name (county of origin)	Biological Quality Elements included
01-LR-NA	LIMNODATA (NL)	FI, BI, MP, PB
07-R-7	EFI+ (EU)	FI
10-R-CAI	National Monitoring Austria GZUEV (AT)	BI, PB
11-R-5	STAR database (EU)	FI, BI, MP
228-R-AT	IHG-DB (AT)	FI, BI
67-LR-NCB	Swedish National trend lakes + streams (SE)	FI, BI, MP, PB
74-R-4	AQEM database (EU)	BI
76-R-CAI	UBA project database (DE)	FI, BI, MP, PB
90-R-4	French rivers - fish database (FR)	FI

DatabaseID	Database name (county of origin)	Biological Quality Elements included
99-R-N	National Monitoring Austria GZUEV (AT)	FI
227-R-AT	SAPRO-NOE (AT)	BI
91-R-4	French rivers - invertebrates database (FR)	BI
213-R-FR-MP	French rivers - macrophyte database (FR)	MP
03-R-7	STAR diatoms (EU)	PB

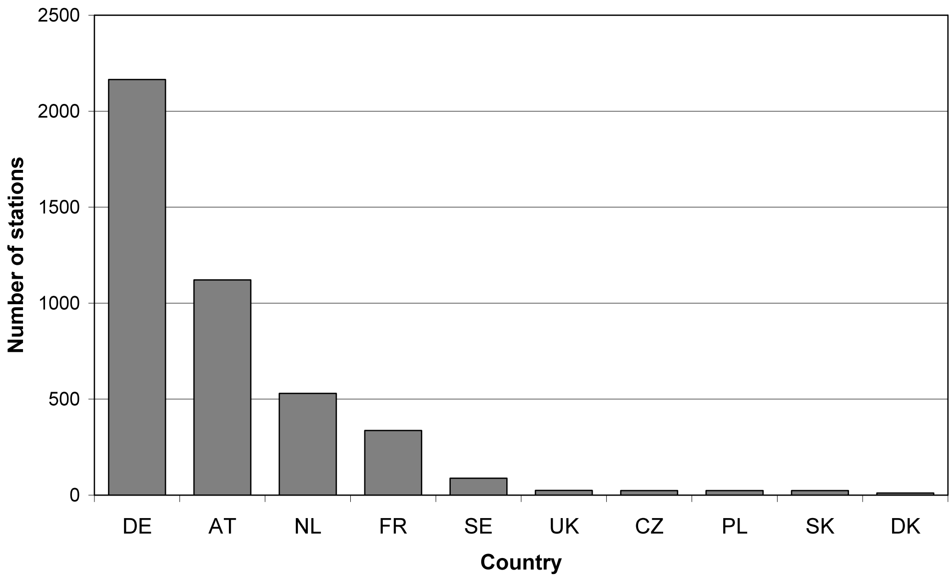

Figure 1: Number of stations per country.

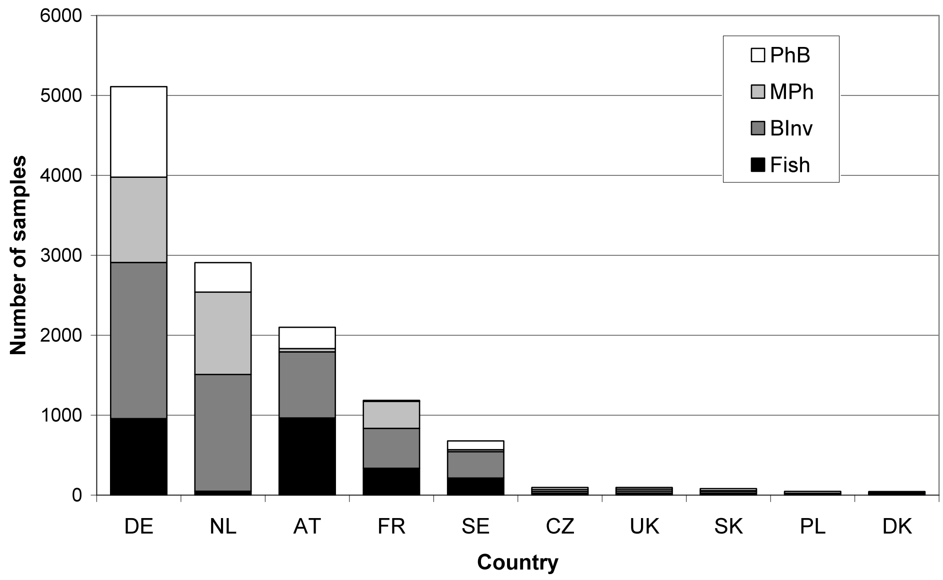

Figure 2: Number of samples per BQE and country.

Table 2: Number of stations covering 1, 2, 3 and 4 BQEs per country.

Country	Number of BQE				
	1	2	3	4	Total
FR			325	12	337
AT	641	332	141	7	1121
CZ			1	23	24
DE	754	590	654	167	2165
DK				11	11
NL	6	406	109	9	530
PL	1	23			24
SE	9	20	49	10	88
SK			15	9	24
UK			5	20	25
Total	1411	1371	1299	268	4349

Biological Quality Components (BQEs)

The WP5.1 large-scale database contains biological samples of altogether four BQEs: fish (FI), benthic invertebrates (BI), aquatic macrophytes (MP) and phytobenthos (PB; mostly benthic diatoms). Raw taxalists from component databases were therefore taxonomically harmonised and stored within four respective database tables. The taxonomic harmonisation aimed at identifying and excluding synonyms, so that for each BQE as far as possible a unique standard taxalist was followed. Due to the country-specific heterogeneity, however, it was not feasible (and also not targeted) to reduce all redundant taxa from the master database, such as different life stages of the same coleopteran species or semi-terrestrial macrophytes, the latter of which are usually not sampled as aquatic macrophytes in standard monitoring programmes of many countries. Individual sub-sets of data generated by individual queries, therefore, must be checked individually for redundancy prior to any analysis of the data! As the number of stations and samples per station depends on individual priorities of the data user, we did not generate standard queries and harmonised taxalists for general use.

The taxonomy applied follows the EFI+ taxalist for fish EFI+ Consortium 2009), http://www.freshwaterecology.info for benthic invertebrates, the intercalibration taxalist for macrophytes (S. Birk, unpublished data) and the OMNIDIA software tool (http://omnidia.free.fr/omnidia_review.htm) for benthic diatoms.

BQE sampling methods and specifications

Fish

Fish samples were primarily taken using electroshocking while sampling effort varied according to the abundance observed (Oberdorff 2001). All fish were counted and measured for length alive and released afterwards. If not indicated separately, only data from the first run are included in the database. In some cases, a sampling stretch (station) was repeatedly sampled in a second and, rarely also in a third run to better achieve whole-stretch representativeness of a sample. For comparability reasons, additional runs must not be confounded with single-run samples. All results are indicated as number of specimens, which can be recalculated to number of specimens per hectare area, if area data is provided (mainly AT and FR). For DE, area data is limited to several hundreds of stations only. The targeted taxonomic level was the species level.

Benthic invertebrates

Benthic invertebrates were predominantly sampled using semi-quantitative or quantitative methods that enable the indication of an area sampled. Where standard multi-habitat sampling was applied (Hering et al. 2004), this area spanned 1.25 m^2 and was conducted using a handnet (ca. 25 x 25 cm frame, 500 µm mesh size) for semi-quantitative sampling or a surber sampler (similar size and mesh) for quantitative sampling. Samples were mostly pooled and either field-sorted (density of highly abundant species was then estimated as abundance class) or lab sorted (density mostly based on counts). All abundances are indicated as number of specimens per m^2, which equals a minimum abundance of 0.8 specimen/m^2 (1 specimen/1.25 m^2). Abundance values being a multiple of 0.8 indicate their origin from standard multi-habitat sampling as described in Hering et al. (2004). The targeted taxonomic level was the species level in all countries except for France, which aimed at the genus level.

Aquatic Macrophytes

Macrophyte surveys were in line with the European Standard EN 14184 and followed the protocols of AFNOR (2003) in France and Schaumburg (2005a,b) in Germany. At each station, the species' coverages were estimated along a 100 m stretch, either while wading across the stretch or using a boat and a rake at non-wadeable stations and converted to the semi-quantitative Kohler scale. Users should note that the sampling effort differed significantly among countries and included semi-terrestrial and terrestrial species (trees) in France, which necessitates a thorough taxonomic adjustment prior to inter-country comparisons of the taxonomic structure.

Benthic diatoms (Phytobenthos)

Sampling techniques differed among countries and habitats and required a standardisation of densities per unit effort. Where total counts of specimens approach the targeted value of 400 specimens, taxa lists are directly comparable. If total taxa counts deviate by more than 5% (i.e. <380 and >420 specimens), taxa richness must not be directly compared. Any user of phytobenthos taxa lists, thus must be aware of this limitation, which can be overcome by rarefaction techniques, i.e. the electronic random sampling of 400 specimens from larger communities. The targeted taxonomic level for benthic diatoms is the species level.

Metric data

For each BQE there is at least one metric table available. The metrics were calculated using the same standard software tool for all samples. This was EFI+ for fish, ASTERICS for benthic invertebrates, the IC Excel macro for aquatic macrophytes and Omnidia for benthic diatoms. After calculation, all metric results were stored in the database and linked to the respective BQE sample tables. Thus, it is easily feasible to link the BQE metrics to the stations.

Environmental variables

Altogether 83 environmental variables were collated in addition to the biological data, representing three different abiotic impact groups and one group of general environmental meta data (= natural descriptors): i) station meta data, ii) physico-chemical variables, iii) hydromorphological variables and iv) land use/cover variables (Table 3). Variable short codes and units of measurement are listed in the original tables further below.

Station meta data contain geographical information, altitude, unique station codes, recent temperature data and some additional information on data availability and the data provider. For 4,284 stations current (Hijmans et al. 2007) and predicted temperatures (Nakicenovic and Swart 2000; International Centre for Tropical Agriculture 2011) are available with the time horizon 2050.

Table 3: 83 Environmental variables divided into four groups and their units of measurement.

Variable group	No. of variables	Name of table
Station meta data	35 plus descriptions	t1_Station
Hydromorphological	16	t2_P_Hydromorphology
Physico-chemical	13	t3_P_PhysChemParam
Land cover/use	19	t4_P_Landuse

Database structure

The database comprises 26 data and help tables. The structure and relations between the main tables and several important help and code tables are shown in Figure 1. Please note that the metric tables are hidden for clarity reasons.

The central station table *t1_Station* defines the main sampling stations (sites) including supporting descriptive information, for example, geographical coordinates, altitude, ecoregion and the biological quality elements (BQE) available at a respective station. The term station is preferred over the term site, since a station might be equivalent to a stretch of several hundreds of metres length, at which different samples of different BQEs have been taken, or can be allocated to, respectively. If different BQE sampling sites do not fall within a stretch of several hundreds of metres, but nevertheless belong to the same station (water body!), the distance between the individual BQE sampling *site* and the common sampling *station* is indicated in the BQE table (not in the station table!).

The station code always starts with a country's two-letter code, followed by alphanumeric characters. The site codes were handled similarly, so that it should be easy to identify the coutry of origin of the data. Because partner UDE (Germany) was responsible for the incorporation of the STAR project data and the Swedish monitoring data, all STAR data has station codes beginning with the country abbreviation 'DE' for Germany. The field "CountryID" provides the information about the BQE data origin; this field was included in all tables to assist the query of country-specific data.

Even if multiple samples were available for a station originating from different sampling seasons or years, each station is considered unique and, thus, represents a spatially homogeneous unit. In contrast, the samples available for a station might reveal notable temporal heterogeneity and may differ up to eight years. The concept of stations as main reference points instead of sites for the BQEs was created because not every partner has more than one BQE at a sampling site.

Hydromorphological variables are stored in the table *t2_P_Hydromorphology* and comprise data on the status of bed, bank and riparian conditions and of the connectivity (weirs/obstacles upstream/downstream of a station). Due to the comparatively heterogeneous (country-specific) sampling protocols used for hydromorphological surveys, the variables were re-scaled to ordinal or categorical units

The main physico-chemical parameters are stored in the table *t3_P_PhysChemParam*. The list covers only basic physical (e.g. pH, conductivity and oxygen content) and chemical parameters (e.g. N and P components, alkalinity), so that the gradients that can be derived from this data address mainly eutrophication and oxygen depletion. This table contains the water temperature measured in parallel to sampling, which is different from the mean recent and predicted air temperatures stored in the station table.

The land use/cover information is stored in the table *t4_P_Landuse*. Land use data is available at different spatial scales spanning from near-stream ares of 0.1 km^2 to the entire catchment. However, near-stream buffer land use/cover is only available for ca. 500 stations while catchment-scale percent cover is available for a larger portion of the data. For a description of small-scale buffer land use and buffer areas, see Figure 2 in Feld (this report) on the impact of land use on riverine assemblages at different spatial scales. All land use information is based on CORINE 2000 (CLC 2000).

The BQE data is organised in 12 tables. For each BQE there is a site table (e.g. *tbl_Fish1Site*) containing site-specific information like coordinates, site name and distance to the station (if there is any). Next follows the sample table (e.g. *tbl_Fish2Sample*), which includes the sampling date and various sample related data like method or sampled area. There can be more samples at one site. The catch table (e.g. *tbl_Fish3Catch*) contains the information about the species and abundances at a sampling site. There are also various help and code tables, providing information about ecoregions, countries, abundance scales (for some Austrian invertebrate data), reporters and finally the BQE taxa lists.

Each of the main tables has at least one primary key, which is normally the station or site code. If there are no unique datasets with only one primary key, the date was chosen as second primary key. This was the case in the sample tables, *t2_P_Hydromorphology* and *t3_P_PhysChemParam*. In case of the catch tables, the taxon ID was the third primary key to define unique datasets. The table *tbl_PhB3Catch* required the use of four primary keys, because of the need to allocate all species to the official reference taxa list; this was not trivial in case of phytobenthic algae due to their complex taxonomy. The table *t4_P_Landuse* also required the use of an auto value as primary key, because of the lack of uniqueness in land use at different spatial scales

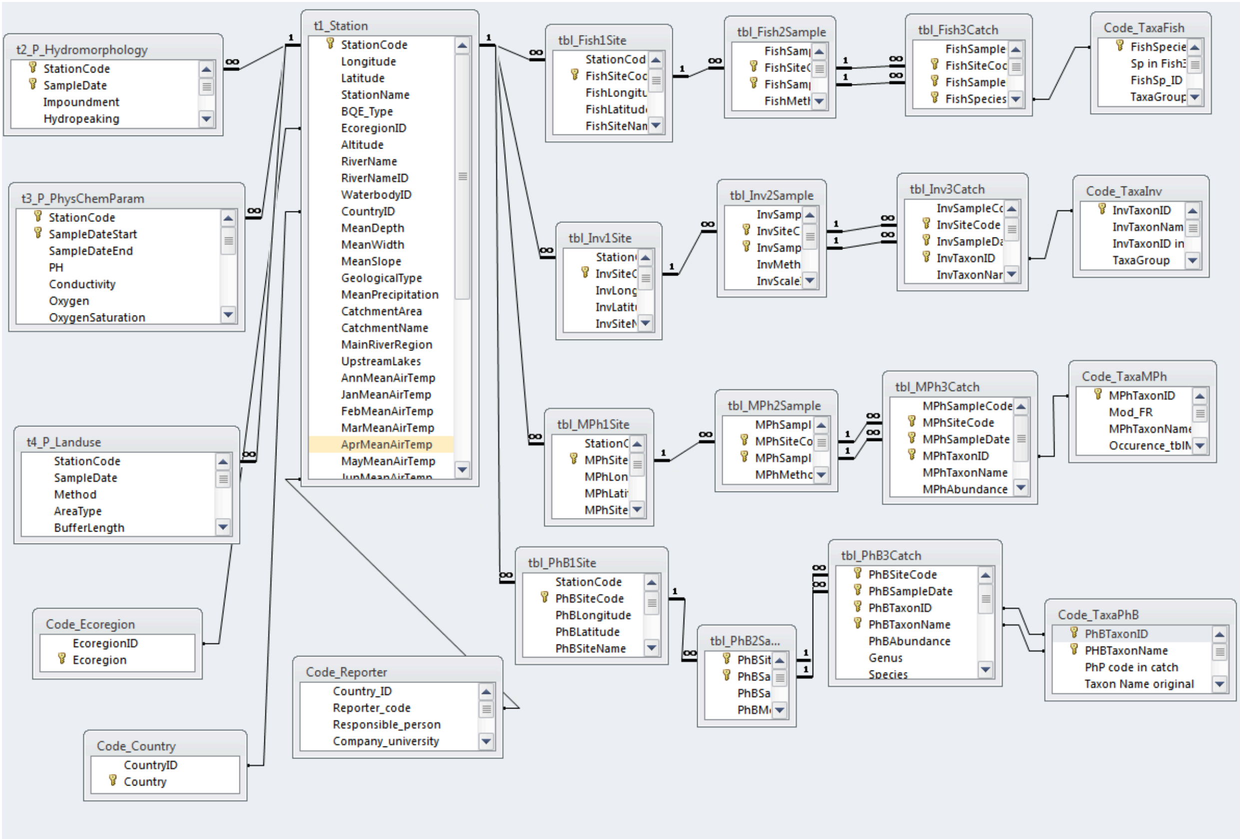

Figure 3: Database relations (main tables and important help/code tables, no metrics tables).

Table 4: Number of data sets in main tables (part I).

country	t1_Station	t2_P_Hydromorphology	t3_P_PhysChemParam	t4_P_Landuse	tbl_Fish1Site	tbl_Fish2Sample	tbl_Fish3Catch	tbl_Inv1Site
AT	1121	1121	1121	17565	842	965	6242	690
CZ	24	24	24	24	24	24	50	24
DE	2165	1331	1511	33486	878	957	6855	1631
DK	11	11	11	11	11	11	56	11
FR	337	337	329	12	337	337	2585	337
NL	530	530	1299	644	43	48	545	500
PL	24	24	24	24	23	23	109	
SE	88	213	1780	27	66	213	717	87
SK	24	24	24	24	24	24	81	24
UK	25	25	25	25	25	25	136	25
	4349	**3640**	**6148**	**51842**	**2273**	**2627**	**17376**	**3329**

Table 4 continued: Number of data sets in main tables (part II).

country	tbl_Inv2Sample	tbl_Inv3Catch	tbl_MPh1Site	tbl_MPh2Sample	tbl_MPh3Catch	tbl_PhB1Site	tbl_PhB2Sample	tbl_PhB3Catch
AT	827	51313	42	42	306	227	265	12541
CZ	24	1384	23	23	88	24	24	365
DE	1954	53060	1048	1067	5272	1007	1132	33373
DK	11	466	11	11	121	11	11	419
FR	498	20440	337	338	5407	12	12	389
NL	1461	83670	549	1030	13773	200	370	10571
PL			24	24	514			
SE	330	4612	26	26	235	57	110	2912
SK	24	1455	9	9	5	24	24	532
UK	25	1101	25	25	206	20	20	496
	5154	**217501**	**2094**	**2595**	**25927**	**1582**	**1968**	**61598**

The table *t1_Station* consists of 4349 stations in 10 countries and covers numerous ecoregions. Most stations are located in Germany, Austria, France and the Netherlands.

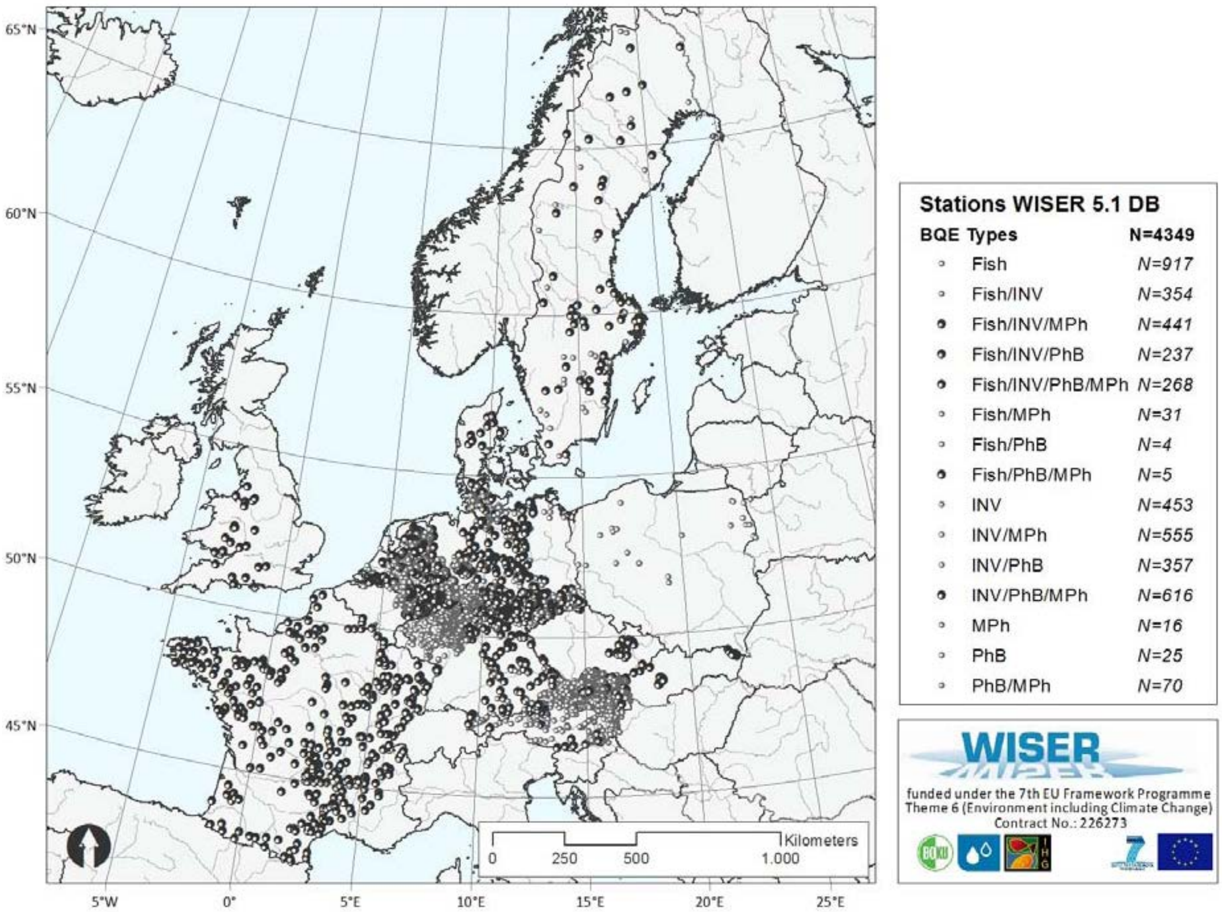

Figure 4: Map of all stations and BQE types/combinations available.

Description of tables

Below follows a short introduction to every table in the WP 5.1 database (field name, data type, table description).

t1_Station

Field Name	Data Type	Description
StationCode	Text	Station Code (created for WP5.1), primary key by partner; starting with country abbreviation: AT, DE, FR, NL (has to be unique)
Longitude	Number	longitude (degrees W (-) or E (+), decimal), WGS84
Latitude	Number	latitude (degrees N, decimal), WGS84
StationName	Text	Station Name, national use
BQE_Type	Number	BQE of original station: 1=fish, 2=invertebrates, 3=phytobenthos, 4=macrophytes (combinations possible)
EcoregionID	Number	Ecoregion number (see codelist)
Altitude	Number	Altitude (Meters Above Sea Level) [m]
RiverName	Text	River name (national)
WaterbodyID	Text	Key field in t_Waterbody (WP 2.2 database)
CountryID	Text	Country Code (see codelist Code_Country)
MeanDepth	Number	Depth of bottom below sampling station (if applicable) [m]
MeanWidth	Number	Width of waterbody
MeanSlope	Number	Slope [‰]
GeologicalType	Text	Geology type: siliceous, calcareous or organic
MeanPrecipitation	Number	Mean annual precipitation [mm]
CatchmentArea	Number	Catchment size [km²]
CatchmentName	Text	Catchment name
MainRiverRegion	Text	Name of main river region
UpstreamLakes	Text	Yes/no/NoData (if the lake upstream affects a site)
AnnMeanAirTemp	Number	Mean annual air temperature [°C]
JanMeanAirTemp	Number	Mean air temperature for January [°C]
FebMeanAirTemp	Number	Mean air temperature for February [°C]
MarMeanAirTemp	Number	Mean air temperature for March [°C]
AprMeanAirTemp	Number	Mean air temperature for April [°C]
MayMeanAirTemp	Number	Mean air temperature for May [°C]
JunMeanAirTemp	Number	Mean air temperature for June [°C]
JulMeanAirTemp	Number	Mean air temperature for July [°C]
AugMeanAirTemp	Number	Mean air temperature for August [°C]
SepMeanAirTemp	Number	Mean air temperature for September [°C]
OctMeanAirTemp	Number	Mean air temperature for October [°C]
NovMeanAirTemp	Number	Mean air temperature for November [°C]
DecMeanAirTemp	Number	Mean air temperature for December [°C]
DataSource	Text	Source of data (database)
ReporterCode	Text	Corresponds to the reporter code in "code_reporter"
MultipleBQE	Yes/No	Are there more than one BQE samples for this site with the specified timeframe? (Yes/No)
BQETypesWord	Text	BQE types in words: fish, inv, PhB, MPh; exampel: "fish, MPh"
BQETypesNum	Text	BQE types in numbers: 1=fish, 2=inv, 3=PhB, 4=MPh; exampel: "14" ->fish and MPh
Comment	Memo	Any other comment
WaterBodyType	Text	only NL: Additional information about waterbody
Select_all	Number	only AT: use in WISER = 1
SelectWiser	Yes/No	only AT: will be used for WISER Level 1 (yes/no)
SelectWiser2	Yes/No	only AT: will be used for WISER Level 2 (yes/no)
owk_check	Number	only AT: check for waterbody - 0=same waterbody, 1=barrier but soame waterbody, 2=not the same waterbody

t2_P_Hydromorphology

Field Name	Data Type	Description
StationCode	Text	Station Code from "t1_Station"
SampleDate	Date/Time	Date (year)
Impoundment	Text	Impoundments or stagnation (yes/no/noData)
Hydropeaking	Text	Hydropeaking or puls releases (yes/no/noData)
WaterAbstraction	Text	Water abstraction (yes/no/noData)
VelocityIncrease	Text	Flow velocity increase (yes/no/noData)
RiparianVegMod	Text	Riparian vegetation modified (no/slight/intermediate/high)
ArtificialEmbankment	Text	Artificial embankment (no/slight/intermediate/high)
NumberBarrierUp	Number	Number of fish migration barrier upstream (segment scale)
BarrierUp	Text	barrier upstream (yes/partial/no) - alternativ to NumberBarrierUp (FR)
NumberBarrierDown	Number	Number of fish migration barrier downstream (segment scale)
BarrierDown	Text	Barrier downstream (yes/partial/no) - alternativ to NumberBarrierDown (FR)
BarriersCatchmentDown	Text	Barriers catchment down (yes/partial/no)
WaterUse	Text	Water use: HP(Hydropower) I(Irrigation) DW(Drinking Water) SP(Snowproduction) FP(Fishponds) CW(Cooling Water) IW(Industrial Water) OT(Others)
InstreamHabitatModified	Text	Instream habitat modified (no/intermediate/high)
DominatingSubstrate	Text	Dominating substrate
ChannelFormModified	Text	Channel form modified (no/intermediate/straightened)
CrossSectionModified	Text	Cross section modified (no/intermediate/technical profile)
DataSource	Text	Source of data (database)
Comment	Memo	Any other comment
CountryID	Text	Country Code (see codelist Code_Country)

t3_P_PhysChemParam

Field Name	Data Type	Description
StationCode	Text	Station Code from "t1_Station"
SampleDateStart	Date/Time	Date of first sample
SampleDateEnd	Date/Time	Date of last sample (if more than one sample)
PH	Number	PH 0-14; value at sampling time
Conductivity	Number	Electrical conductivity [microS/cm]; value at sampling time
Oxygen	Number	Oxygen content [mg/l]; value at sampling time
OxygenSaturation	Number	Oxygen saturation [%], if applicable
BOD5	Text	Biological oxygen demand [mg/l]
Nitrite	Number	Nitrite [mg/l] (NOT Nitrit-N!)
Nitrate	Number	Nitrate [mg/l]
Ammonia	Text	Ammonia [mg/l]
Chloride	Number	Chloride [mg/l]
OrthoPhosphate	Number	Ortho-phosphate [microg/l] (not PO4-P)
TotalPhosphate	Number	Total-phosphate [microg/l]
Alkalinity	Number	Alkalinity [mval/l], if applicable
WaterTemperature	Number	Water temperature [°C], if applicable
DataSource	Text	Source of data (database)
Comment	Memo	Any other comment
CountryID	Text	Country Code (see codelist Code_Country)

t4_P_Landuse

Field Name	Data Type	Description
SampleDate	Date/Time	Date (year)
Method	Text	Method of calculations
AreaType	Text	Type of area used for calculations (e.g. catchment, buffer)
BufferLength	Number	Total calculated length of buffer in meters [m]
CatchmentLandUse	Text	Catchment land use/cover (Corine)
CorineArea	Number	Area in [km²], base for corine percentages
11_UrbanFabric	Number	Percentage; Urban fabric
12_IndustrialCommercialTrans	Number	Percentage; Industrial, commercial and transport units
13_MineDumpConstruction	Number	Percentage; Mine, dump and construction sites
14_ArtificialNon-agriculturalVe	Number	Percentage; Artificial non-agricultural vegetated areas
21_ArableLand	Number	Percentage; Arable land
22_PermanentCrops	Number	Percentage; Permanent crops
23_Pastures	Number	Percentage; Pastures
24_HeterogeneousAgricultural	Number	percentage; Heterogeneous agricultural areas
31_Forests	Number	Percentage; Forests
32_ShrubHerbaceousVegetatic	Number	Percentage; Shrub and/or herbaceous vegetation association
33_OpenSpacesLittleOrNoVeg	Number	Percentage; Open spaces with little or no vegetation
41_InlandWetlands	Number	Percentage; Inland wetlands
42_CoastalWetlands	Number	Percentage; Coastal wetlands
51_InlandWaters	Number	Percentage; Inland waters
52_MarineWaters	Number	Percentage; Marine waters
99_UnclassifiedSurface	Number	Percentage; Unclassified surface
DataSource	Text	Source of data (database)
Comment	Memo	Any other comment
CountryID	Text	Country Code (see codelist Code_Country)
LanduseID	AutoNumber	Landuse ID (Auto)

tbl_Fish1Site:

Field Name	Data Type	Description
StationCode	Text	Station Code from "t1_Station"
FishSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
FishLongitude	Number	longitude (degrees W (-) or E (+), decimal), WGS84
FishLatitude	Number	latitude (degrees N, decimal), WGS84
FishSiteName	Text	Site name, national use
FishRiverName	Text	River name
FishDistToStation	Number	Distance from Sampling Site to Station [m] (optional)
Fish_up_down	Number	Code "1" for upstream distance, Code "2" for downstream distance, Code "0" for Station and Site are identical (distance = 0)
CountryID	Text	Country Code (see codelist Code_Country)
DatabaseID	Text	Dataset IDs for the WP5.1 metadatabase

tbl_Fish2Sample:

Field Name	Data Type	Description
FishSampleCode	Text	Sample Code (optional)
FishSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
FishSampleDate	Date/Time	Sampling date
FishMethod	Text	Sampling method
FishStrategy	Text	Define how the section was sampled, Whole, Partial, NoData
FishArea	Number	Sample area [m²]
FishReporter_code	Text	Corresponds to the reporter code in "Code_Reporter"
CountryID	Text	Country Code (see codelist Code_Country)

tbl_Fish3Catch:

Field Name	Data Type	Description
FishSampleCode	Text	Sample Code (optional)
FishSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
FishSampleDate	Date/Time	Sampling date
FishSpecies	Text	Scientific name of species (update by melcher)
FishTaxonID	Number	Fish taxon ID (see table Code_TaxaFish)
Run1_number_all	Number	All caught individuals of the species in run 1
Run2_number_all	Text	All caught individuals of the species in run 2
Run3_number_all	Text	All caught individuals of the species in run 3
FishBiomass	Text	Estimated biomass (kg) of the species per hectare
FishAbundance	Text	Estimated abundance (no. of individuals) of the species per hectare
CountryID	Text	Country Code (see codelist Code_Country)
AbundanceUnit	Text	DE: Unit in which the abundance was given (UDE usually indidviduals, SLU usually individuals/100 sqm).

tbl_Inv1Site:

Field Name	Data Type	Description
StationCode	Text	Station Code from "t1_Station"
InvSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
InvLongitude	Number	longitude (degrees W (-) or E (+), decimal), WGS84
InvLatitude	Number	latitude (degrees N, decimal), WGS84
InvSiteName	Text	Site name, national use
InvRiverName	Text	River name
InvDistToStation	Number	Distance from Sampling Site to Station [m] (optional)
Inv_up_down	Number	Code "1" for upstream distance, Code "2" for downstream distance, Code "0" for Station and Site are identical (distance = 0)
CountryID	Text	Country Code (see codelist Code_Country)
DatabaseID	Text	Dataset IDs for the WP5.1 metadatabase

tbl_Inv2Sample:

Field Name	Data Type	Description
InvSampleCode	Text	Sample Code (optional)
InvSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
InvSampleDate	Date/Time	Sampling date
InvMethode	Text	Sampling method
InvScaleID	Number	Only AT: >0 means qualitative sampling; see Code_InvScaleAT
InvArea	Number	Sample area [m²]
InvUnit	Text	Only NL: Unit
InvReporter_code	Text	Corresponds to the reporter code in "code_reporter"
CountryID	Text	Country Code (see codelist Code_Country)

tbl_Inv3Catch:

Field Name	Data Type	Description
InvSampleCode	Text	Sample Code (optional)
InvSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
InvSampleDate	Date/Time	Sampling date
InvTaxonID	Number	Inv taxon ID (see table Code_TaxaInv)
InvTaxonName	Text	Inv taxon name (genus species)
InvGenus	Text	Inv genus
InvSpecies	Text	Inv species
InvAbundance	Number	Abundance of species
CountryID	Text	Country Code (see codelist Code_Country)
comment	Text	Comment

tbl_MPh1Site:

Field Name	Data Type	Description
StationCode	Text	Station Code (from table tbl_Station)
MPhSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
MPhLongitude	Number	longitude (degrees W (-) or E (+), decimal), WGS84
MPhLatitude	Number	latitude (degrees N, decimal), WGS84
MPhSiteName	Text	Site name, national use
MPhRivername	Text	River name
MPhDistToStation	Number	Distance from Sampling Site to Station [m] (optional)
MPh_up_down	Number	Code "1" for upstream distance, Code "2" for downstream distance, Code "0" for Station and Site are identical (distance = 0)
CountryID	Text	Country Code (see codelist Code_Country)
DatabaseID	Text	Dataset IDs for the WP5.1 metadatabase

tbl_MPh2Sample:

Field Name	Data Type	Description
MPhSampleCode	Text	Sample Code
MPhSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
MPhSampleDate	Date/Time	Sampling date
MPhMethod	Text	Sampling method
MPhUnit	Text	Only NL: Unit
MPhReporter_code	Text	Corresponds to the reporter code in "code_reporter"
CountryID	Text	Country Code (see codelist Code_Country)

tbl_MPh3Catch:

Field Name	Data Type	Description
MPhSampleCode	Text	Sample Code
MPhSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
MPhSampleDate	Date/Time	Sampling Date
MPhTaxonID	Text	MPh taxon ID (see table Code_TaxaMPh)
MPhTaxonName	Text	MPh taxon name - updated version by Melcher
MPhAbundance	Number	abundance of species
MPhScaleValue	Number	AT: only scale values: 1 (rare) to 5 (mass)
CountryID	Text	Country Code (see codelist Code_Country)

tbl_PhB1Site:

Field Name	Data Type	Description
StationCode	Text	Station Code (from table tbl_Station)
PhBSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
PhBLongitude	Number	longitude (degrees W (-) or E (+), decimal), WGS84
PhBLatitude	Number	latitude (degrees N, decimal), WGS84
PhBSiteName	Text	Site name, national use
PhBRivername	Text	River name
PhBDistToStation	Number	Distance from Sampling Site to Station [m] (optional)
PhB_up_down	Number	Code "1" for upstream distance, Code "2" for downstream distance, Code "0" for Station and Site are identical (distance = 0)
CountryID	Text	Country code (see codelist Code_Country)
DatabaseID	Text	Dataset IDs for the WP5.1 metadatabase

tbl_PhB2Sample:

Field Name	Data Type	Description
PhBSampleCode	Text	PhB Sample ID (optional)
PhBSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
PhBSampleDate	Date/Time	Sampling date
PhBMethod	Text	PhB method
PhBUnit	Text	PhB unit
PhBReporter_Code	Text	Reporter Code (see table Code_Reporter)
CountryID	Text	Country code (see table Code_Country)
Cover_all	Text	Only AT: coverage percentage total
Cover_MA	Text	Only AT: coverage percentage of macro algae
Cover_all_WFD	Text	Only AT: coverage percentage total according to new WFD method
Cover_MA_WFD	Text	Only AT: coverage percentage of macro algae according to new WFD method
Cover_Algae_WFD	Text	Only AT: coverage percentage algae according to new WFD methos

tbl_PhB3Catch:

Field Name	Data Type	Description
PhBSampleCode	Text	PhB Sample ID (optional)
PhBSiteCode	Text	Sampling Site Code; Unique reference number/name/code per sampling site (national)
PhBSampleDate	Date/Time	Sampling date
PhBTaxonID	Text	PhB taxon ID (see table Code_TaxaPhB)
PhBTaxonName	Text	PhB taxon Name (see table Code_TaxaPhB)
PhBAbundance	Number	PhB abundance
Genus	Text	PhB genus
Species	Text	PhB species
AT_Cover_MA	Number	AT calculation of abundance - percentage of macro-algae coverage
AT_Peelings_D	Number	AT calculation of abundance - peelings
AT_ID_PHBTaxon	Number	AT Ecoprof ID
AT_Name_MI	Text	AT name of micro-algae coverage
AT_Cover_MI	Number	AT percentage of micro-algae coverage
AT_Cover_MI_WFD	Number	AT percentage of micro-algae coverage according to WFD (if applicable)
AT_Dominance_MI	Number	AT dominace of micro-algae
AT_PHBSample_ID	Number	AT Ecoprof code
CountryID	Text	Country Code (see codelist Code_Country)

Code_TaxaFish:

Field Name	Data Type	Description
FishSpecies	Text	Wiser species name genus and species separeted by "_" updated by melcher
Sp in Fish3Catch	Text	Species name as in table tbl_Fish3Catch
FishSp_ID	Number	Species ID
TaxaGroup	Text	Group
Family	Text	Family
Genus	Text	Genus
Species	Text	Species
Author	Text	Author
Synonym1	Text	Synonyms (if existent)
Synonym2	Text	Synonyms (if existent)
Shortcode_WISER	Text	Shortcode WISER
Shortcode_FR	Text	Shortcode France
Shortcode_AT	Text	Shortcode Austria
higher taxonomic unit	Text	higher taxonomic unit (yes/...)

Code_TaxaInv:

Field Name	Data Type	Description
InvTaxonID	Number	WISER invertebrate taxon ID updated by melcher
InvTaxonName in InvCatch	Text	Quality check
InvTaxonID in InvCatch	Number	Quality check
TaxaGroup	Text	Taxa group
Family	Text	Family
Subfamily	Text	Subfamily
Genus	Text	Genus
Species	Text	Species
Author	Text	Author
Shortcode	Text	Shortcode
DINNo	Number	DIN Number
TCM_Code	Number	TCXM Code
Furse_Code	Text	Furse Code
Perla_Code	Text	Perla Code
higher taxonomic unit	Text	higher taxonomic unit (yes/...)

Code_TaxaMPh:

Field Name	Data Type	Description
MPhTaxonID	Text	PK; Updated from GIG table by melcher (new code for not listed yet NA ... NA126)
Mod_FR	Text	New TaxonID according to Christian Chauvin: NA### are old codes
MPhTaxonName	Text	Updated from GIG table by melcher (126 which were not listed yet are added, 3 by MS after NL update September 2010)
Occurence_tblMPhCatch	Text	Taxon which are occuring in tblMPhCatch: updated after every change of MPh (current: FR, NL)
Authority	Text	Authority
SYNONYM 1	Text	Synonym
SYNONYM 2	Text	Synonym
SYNONYM 3	Text	Synonym
SYNONYM 4	Text	Synonym
SYNONYM 5	Text	Synonym
SYNONYM 6	Text	Synonym
SYNONYM 7	Text	Synonym
GROUP	Text	Group
TAX LEVEL	Text	Taxonomic level
Aquaticity	Text	Aquaticity of species
Alien	Number	Alien - is the species native or alien (0/1)
Comment	Text	Comment

Code_TaxaPhB:

Field Name	Data Type	Description
PhBTaxonID	Text	During update the "valid code"
PHBTaxonName	Text	ValidTaxon_name includes taxa and var. and subsp.
PhP code in catch	Text	Occurence in tblPhB3catch
Taxon Name original	Text	Original Taxon name
Taxagroup	Text	Group
Taxon Name	Text	Taxon name
ssp_var_mor_fo	Text	PhB species forms and variations
ssp_var_mor_fo_aff_name	Text	PhB species forms and variations
ID_Ecoprof	Number	ID Ecoprof
Genus	Text	Genus
Species	Text	Species
Authority	Text	Authority
ssp_var_mor_fo_aff_authority	Text	Authority PhB species forms and variations
2nd_var_mor_fo	Text	PhB species forms and variations
2nd_var_mor_fo_name	Text	PhB species forms and variations
2nd_var_mor_fo_authority	Text	Authority PhB species forms and variations
higher taxonomic unit	Text	Higher taxonomic unit
origin	Text	Origin of data
origin2	Text	Origin of data

Code_Country:

Field Name	Data Type	Description
CountryID	Text	2-letter countrycode (ISO standard) (see codelist)
Country	Text	Country name

Code_Ecoregion:

Field Name	Data Type	Description
EcoregionID	Number	Ecoregion no.
Ecoregion	Text	Name of ecoregion (see http://dataservice.eea.europa.eu/atlas/viewdata/viewpub.asp?id=499)

Code_Reporter:

Field Name	Data Type	Description
Country_ID	Text	Country ID used in the DB: AT, DE, NL, SE
Reporter_code	Text	Your own personal code (e.g. ALTERRA)
Responsible_person	Text	Name of person responsible for data supply
Company_university	Text	Name of company/Univeristy if applicable
Institute_agency	Text	Name of institute or governmental agency
Street_pob	Text	Street address or post box number
City	Text	Name of city
State_province	Text	State name
Postal_code	Text	Country code + postal code (e.g. SE 116 62)
Country_name	Text	Country in English
Email	Text	Email address of the responsible person

Selected descriptive statistics

BQE representativeness.

Benthic invertebrates were available for roughly a third (35% of all stations) of the stations, followed by fish (25%) (Figure 5).

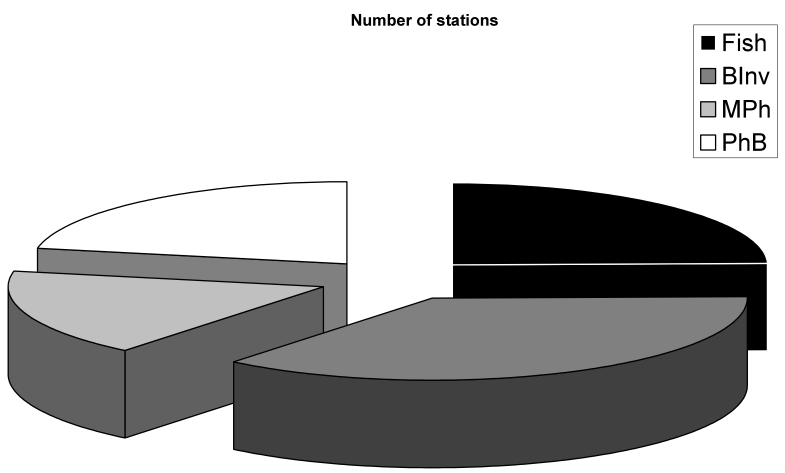

Figure 5: Number of stations per BQE

macrophytes were less frequent, and together with benthic diatoms comprised nearly a third of all stations.

Taxon richness

Fish richness was highest in the Netherlands, followed by France, Germany and Austria. The differences are not significant among the countries. This is different for benthic invertebrates (BInv); the highest richness was found in Austria and the Netherlands, while this was significantly lower in France and lowest in Germany. These differences, however, to some degree represent country-specific 'traditions' in determination: while water mites, oligochaetes and chironomids are determined to species level in the Netherlands (the latter group also in Austria), but not in the other countries, the comparison of taxa lists requires thorough harmonisation prior to analysis in order to account for these artificial richness patterns.

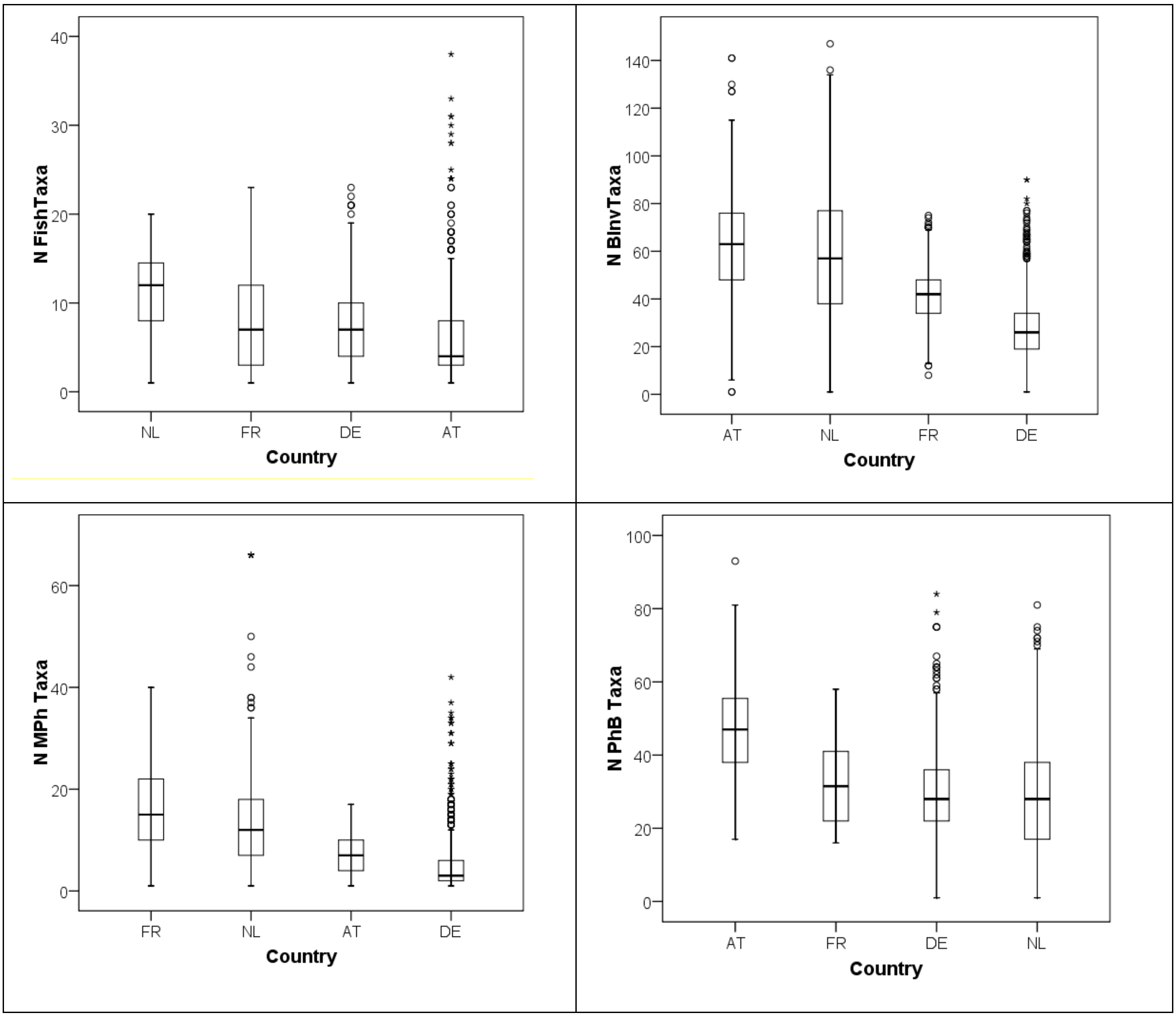

Figure 6: BQE richness per country (AT = Austria, DE = Germany, FR = France, NL = the Netherlands.

Interestingly, invertebrate richness, on average, was higher in France than in Germany, although the French target only genus level identification. The pattern was different for macrophytes (MPh), which were richest France, followed by the Netherlands, Austria and Germany (Figure 6). Again, this is owed to different traditions in the countries; the macrophyte field protocols are different and consider semi-terrestrial and riparian plants in one, but aquatic macrophytes only (submerged and emergent) in another protocol. And again, this requires thorough harmonisation of taxa lists prior to any comparative analysis. Finally, the country-specific differences were less pronounced for benthic diatoms (phytobenthos; PhB). Their richness was highest in the Netherlands, while taxon richness was comparable among the other countries (Figure 6). Nevertheless, it is recommended to use rarefaction techniques in this organism groups, since the number of counted cells differed considerably among samples (but not countries), which is very likely to affect also the richness detected.

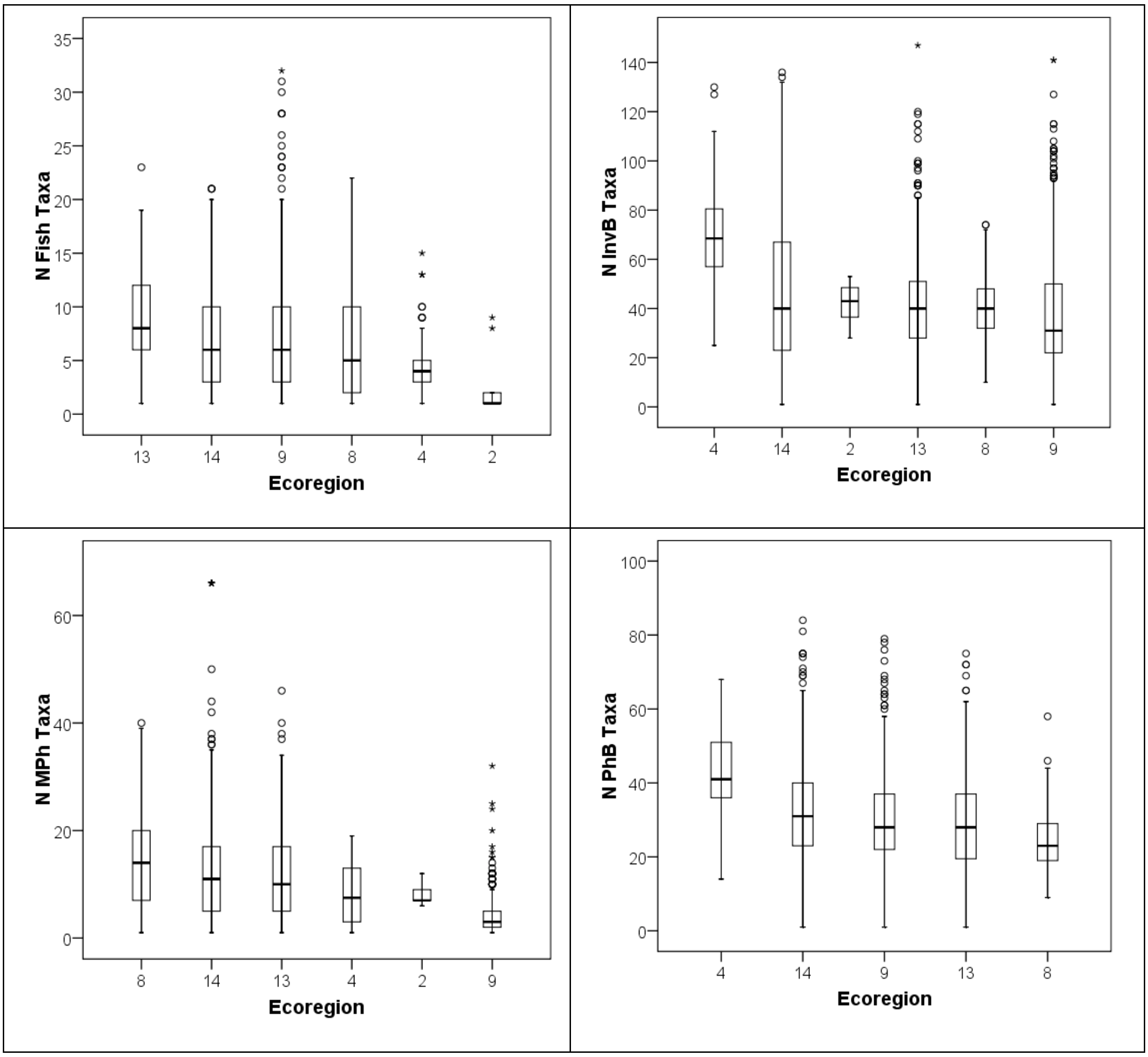

Figure 7: BQE richness per ecoregion (ecoregion numbers after Illies 1978). Ecoregions are: Western and Central Plains (13 and 14, respectively), Western and Central Mountains (8 and 9, respectively) and Pyrenees and Alps (2 and 4, respectively).

Overall, ecoregional differences were less pronounced than richness patterns among countries (Figure 7). The strongest pattern was observed for ecoregions 2 and 4 (Pyrenees and Alps, respectively), where fish richness was lowest, but invertebrate richness peaked highest. The low fish richness is probably due to the small size of streams and rivers in these (alpine and pre-alpine) ecoregions, which naturally limits fish richness in these ecoregions. While benthic invertebrate and phytobenthos richness peaked in the Alps (ecoregion 4), there was no clear preference detectable for macrophytes (Figure 7).

Physico-chemical parameters

Country-specific differences were detectable for total phosphate, which revealed remarkably higher values in Germany (Figure 8).

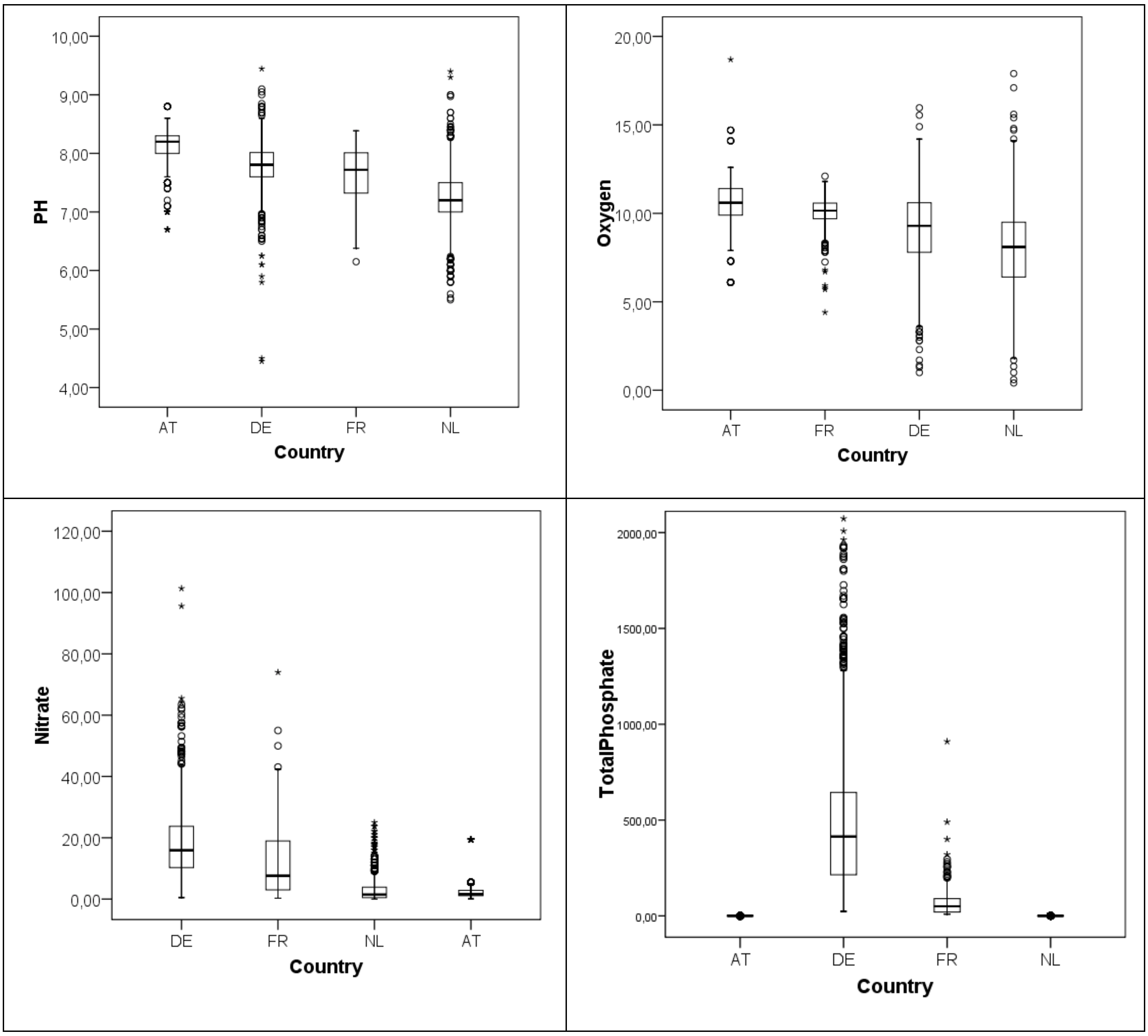

Figure 8: Distribution of selected physico-chemical parameters among the four countries to show differences in physico-chemical gradient lengths (AT = Austria, DE = Germany, FR = France, NL = the Netherlands.

However, owed to the limited availability of physico-chemical data from the Netherlands, one should be cautious if comparing total phosphate among the countries; the low phosphate levels detected for the Netherlands may not be representative for the entire country. The other parameters revealed a common pattern in that their variability was high in lowland regions and lower in mountain regions (Figure 8).

Immutable parameters

The country-specific temperature patterns illustrated in Figure 9 are notable, since the temperature variability was considerably higher in Austria and France, as opposed to Germany and the Netherlands. In particular, the higher peak summer temperatures in the former two countries deserve consideration, since such temperatures are frequently found also in rivers within the latter two countries. This difference is probably owed to two peculiarities: 1. summer temperatures are higher in southern regions (e.g. ecoregion 13 in France) and 2. large rivers are included for Austria and France, but less represented for Germany and the Netherlands. Hence, comparative analysis of temperature data among the countries requires a thorough data mining in order to detect and exclude non-targeted (immutable) temperature effects.

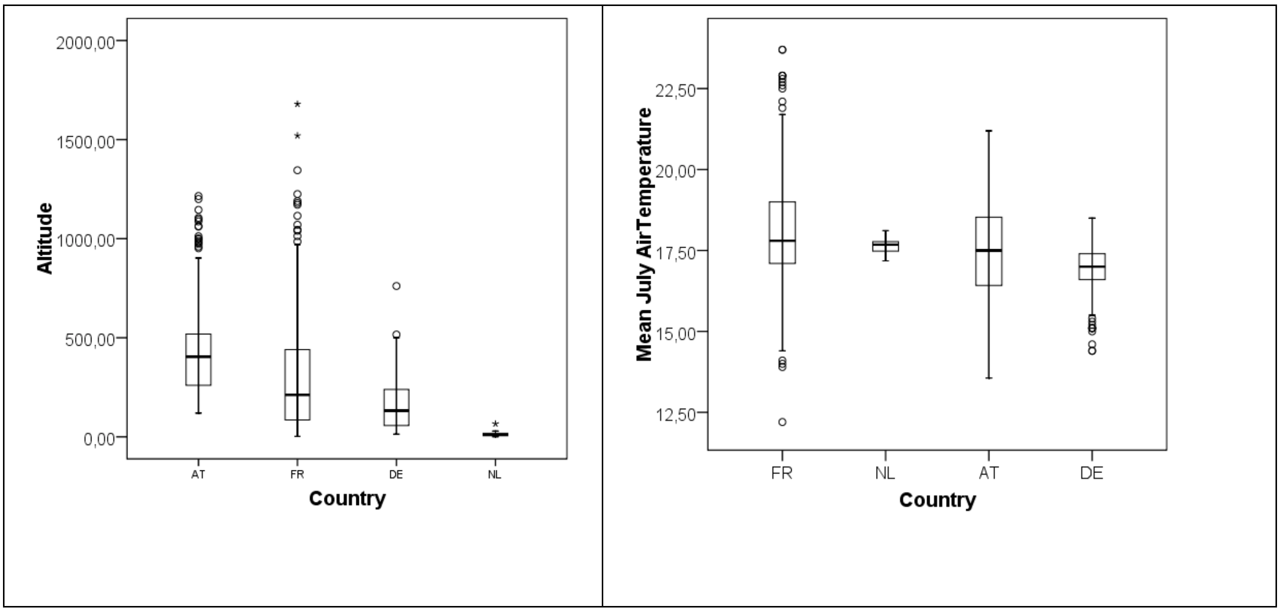

Figure 9: Distribution of selected immutable parameters explaining natural gradients (AT = Austria, DE = Germany, FR = France, NL = the Netherlands.

Table 5: Number of stations for six selected morphological pressure types and stress modality (intensity).

	PressureIntensity				
PressureType	No			High	Total number classified
	0	1	2	3	
Morphology (No, Yes)	198	1515			1713
ArtificialEmbankment	408	388	408	513	1717
RiparianVegMod	425	339	302	680	1746
InstreamHabitatModified	1113		266	443	1822
ChannelFormModified	917		288	373	1578
CrossSectionModified	627		370	365	1362
VelocityIncrease	898	506			1404

CHAPTER II

Driver-Pressure-Impact chains: assessment and indication

Ecological assessment of running waters: do macrophytes, macroinvertebrates, diatoms and fish show similar responses to human pressures?

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Abstract

This study aimed at comparing the intensity and the sensitivity of the responses of four river biological quality elements (BQEs): macrophytes, fish, diatoms and macro-invertebrates, to human pressures excluding natural variations of stream ecosystem functioning. Biological, water quality and hydro-morphological data were compiled for 290 French river sites. National and European indexes (e.g. EFI) and metrics based on the taxonomic composition (e.g. species richness) and ecological and biological traits (e.g. number of lithophilic fish species) were first transformed to acquire independence of natural environmental variability (80 undisturbed sites). In a second step, their responses to human pressures linked to global, water quality, hydrological and morphological alterations were tested and compared using an independent data set (188 impacted and 22 undisturbed sites).

Out of the 93 metrics tested, 51 covering the four BQEs responded significantly to global degradations. The responses to specific pressures were consistent with the BQEs ecological and biological characteristics. The four BQE's metrics responded strongly to water quality degradations. As for fish, macro-invertebrates metrics were very sensitive to morphological degradations such as the presence of an impoundment while diatoms and macrophytes metrics did not show strong responses to these changes. Fish metrics responded the strongest to hydrological perturbations. Although a high proportion of the metrics responded only to high level of human-induced degradations, traits-based metrics seemed the most sensitive and responded to lower level of pressure. Global and water quality degradations of the river appear to be better detected by BQEs' metrics than morphological and hydrological degradations. Finally, studying the effects of single pressures on biota brought new questions about the effects of pressure accumulation, links among pressures and between environment and pressure effects on biological communities.

Highlights

- Macrophytes, fish, diatoms and macro-invertebrates responses to human-induced pressures were compared
- Natural environmental variability was discarded from the analysis
- Functional traits-based metrics were the most sensitive to changes
- Depending on their ecological and biological characteristics, biological groups showed different responses
- Multi-impacted sites have to be removed from the analysis to detect specific pressures

Abbreviations

WFD, Water Framework Directive; BQE, biological quality elements; EQR, ecological quality ratio; MetIND, indexes; MetFUNC, functional trait-based metrics; MetTAX, metrics based on the taxonomic composition; MDC, minimally disturbed conditions; FI, fish; DI, diatoms; MI, macroinvertebrates; MA, macrophytes; EG, environmental gradient; DE, discriminatory efficiency; WS, weighted mean of sensitivity.

Introduction

Throughout Europe, streams have experienced a long history of modification by Man (Petts et al. 1989) and have become one of the most threatened ecosystems (Loh et al. 2005). Since 2000, the Water Framework Directive (WFD; 2000/60/EC) has made a crucial issue of the assessment and the reestablishment of the ecological quality of European rivers. Basically, the WFD recommends the use of multiple biological quality elements (BQEs: macrophytes, phytobenthos, invertebrates, fish and phytoplankton for large rivers) to assess river "ecological status". Facing these institutional needs, freshwater scientists have developed a large number of tools based on various conceptual approaches and biological indicators. It is therefore of primary importance to have a comparative idea of their sensitivity and efficiency to detect human-induced degradation in rivers.

A common approach to evaluate river ecological quality is to measure the deviation of ecosystems from reference status. Such indicators are referred as Ecological Quality Ratio (EQR). As pointed out by Stoddard et al. (2006), "natural variability in indicators always occurs" and has to be taken into account when describing reference conditions. To be able to express the ecological status as EQR, most of the authors define the reference values per stream types standing for homogeneous environmental characteristics (e.g. Verdonschot and Nijboer, 2004). In 2001, Oberdorff et al. have developed a method allowing the consideration of streams and rivers as continuums (River Continuum Concept; Vannote et al. 1980), standardizing their responses regarding the main natural environmental gradients acting at large scale (see also Pont et al. 2006, 2007).

Historically, biological responses were examined through metrics focusing on the most sensitive taxa (e.g. Saprobic index: Pantle and Buck, 1955). More recently, Southwood (1977) and Townsend et al. (1994) hold the idea that combinations of functional traits (ecological and biological) are selected by habitat conditions through the survival ability of individual organisms relative to others (i.e. the fitness). Such integrative approaches were based on functional structures of fish (e.g. Fausch et al. 1990; Index of Biotic Integrity, IBI: Karr, 1981) and macroinvertebrates communities (e.g. Statzner et al. 2001; Usseglio-Polatera et al. 2000).

Responses to human-induced disturbances in rivers have been frequently analyzed separately for the four BQEs: macro-invertebrates (e.g. Archaimbault et al. 2010; Cardoso et al. 2008; Lorenz et al. 2004; Statzner et al. 2001, 2010), diatoms (e.g. Besse-Lototskaya et al. 2011; Carpenter et al. 2000; Fore et al. 2002), macrophytes (e.g. Lacoul et al. 2006; Riis et al. 2000) and fish (e.g. D'Ambrosio et al. 2009; Pont et al. 2006, 2007; Yates et al. 2010). Nonetheless, only few authors have compared the responses to anthropogenic pressures among BQEs. These studies have shown that trajectories (Johnson et al. 2009) and robustness (Johnson et al. 2006b) of the metric's response differ considerably between BQEs, stressors and with stream types (e.g. Heino, 2010; Hering et al. 2006). They observed that nutrient gradient is more correlated to diatom and invertebrate's metrics than fish and macrophyte's (e.g. Hering et al. 2006; Johnson et al. 2006a; Justus et al. 2010) and that diatom and macrophyte's respond earlier than invertebrate and fish's to this gradient (e.g. Johnson et al. 2009). It appears likely that hydro-morphological degradations affect more fish and macrophyte communities than diatoms and macro-invertebrates (e.g. Hughes et al. 2009; Johnson et al. 2006a). More globally, responses of the four BQEs seem stronger for water quality than for hydro-morphological degradations (Hering et al. 2006).

Some authors have demonstrated that traits-based metrics such as the number of tolerant species show the highest sensitivity to human disturbance (e.g. Dolédec et al. 2006; Usseglio-Polatera et al. 2000). However, previous works often relied on metrics based on the taxonomic composition such as the total number of species (e.g. Heino et al. 2005; Johnson et al. 2009) rather than traits-based metrics (Hering et al. 2006; Hughes et al. 2009; Johnson et al. 2006a, 2006b; Justus et al. 2010). In addition, as streams are frequently impacted by multiple linked stressors, single effects of stressors on BQEs have rarely been assessed (Hughes et al. 2009). Based on this literature review, we expected that biological communities would present different responses to anthropogenic disturbances in term of intensity (i.e. discriminatory efficiency, Ofenböck et al. 2004) and sensitivity (i.e. impact of low level of pressure). Also, it was assumed that responses to pressures would be stronger for indexes (MetIND) and functional trait-based metrics (MetFUNC) than for metrics based on the taxonomic composition (MetTAX) and that the standardization method would allow analysing BQEs responses along the whole environmental gradient.

Comparing the responses of the four BQEs (macrophytes, fish, diatoms and macro-invertebrates) to different human pressures, this paper aims at answering the following questions:

(1) Which kind of metrics (MetIND, MetTAX and MetFUNC) is more suitable to detect human pressure impacts?
(2) Are intensity and sensitivity of the responses to general degradation gradient comparable among BQEs?

(3) Do all BQEs detect in a similar way hydrological, morphological, and water quality degradations?
(4) Do these responses change when sites are not multi-impacted?

We focused on a French dataset covering a large range of environmental conditions and human-induced pressures. In this paper, reference status was recognized as the minimally disturbed conditions (MDC) as defined by Stoddart et al. (2006). In comparison with previous studies, natural environmental variation effects were differentiated from human pressure effects transforming beforehand the metrics in standardized metrics independent from the environment for MDC.

Material & Methods

Data compilation

BQEs, natural environmental conditions and reach scale human-induced pressures data (see Table 1) were compiled for 290 French streams. Pressures data included information concerning hydro-morphological degradations (e.g. modification of the channel form) and water quality variables (e.g. oxygen saturation). Sites were samples from 2005 to 2008 during French national monitoring programs. Sites Water quality parameters were used as median values for the period 2005-2007. The four BQEs were sampled at each sites using standardized protocols. When several biological samples were available the most recent was chosen. Fish (FI) data were obtained from national fisheries surveys using electric fishing and expressed as density of each species at a site (number of individuals / sampled area). Macro-invertebrates (MI) were collected using the IBGN (AFNOR, 2004a) method and mostly identified at the genus level. Macrophyte (MA, mainly aquatic phanerogams, bryophytes and colonial algae) were collected using the IBMR method (AFNOR, 2003). They were identified mostly at the species level and density was expressed as the percentage of cover for each taxon. Finally, diatoms (DI) were sampled following the IBD protocol (AFNOR, 2007) and were identified mostly at the species level.

Table 1: Environmental and pressure variables and synthetic gradients

Variables	Transformation	Variables modalities/ranges
Environmental gradients		
Altitude (m)	Log(x)	217 (2 - 1520)
Mean width (m)	Log(x)	7 (0.5 - 93)
Mean slope (‰)	Log(x)	4 (0.1 - 82)
Catchment area (km²)	Log(x)	99 (1 - 13312)
Annual mean air temperature (°C)	Log(x)	10.5 (5 - 15.5)
Distance to the source (m)	Log(x)	17 (0.6 - 372)
Geological type		Siliceous (131) / Calcareous (159)
Upstream lakes		No (288) / Yes (2)
Ecoregions		Alps (6) / Central highlands (2) / Mediterranean (2) / Pyrenees (3) / Western highlands (123) / Western plains (154)
Global degradation gradient		
Presence of an impoundment at		No (263) / Yes (27)

the station		
Hydrological degradation gradient		
Hydrological Regime Modified		No (183) / Slight (59) / Intermediate (26) / High (22)
Hydropeaking		No (264) / Yes (26)
Water Abstraction		No (182) / Slight (77) / Intermediate (10) / High (21)
Morphological (habitat) degradation gradient		
Riparian Vegetation Modified		No (178) / Slight (81) / Intermediate (19) / High (12)
Artificial Embankment		No (253) / Partial (22) / Yes (15)
Instream Habitat Modified		No (233) / Intermediate (34) / High (23)
Channel Form Modified		No (238) / Intermediate (31) / High (21)
Cross Section Modified		No (241) / Intermediate (23) / High (26)
Diked		No (260) / Intermediate (20) / High (10)
Sedimentation		No (170) / Slight (64) / Intermediate (40) / High (16)
Water quality degradation gradient		
Oxygen saturation (%)	Log(x)	94.3 (48.5 - 112.3)
BOD5 (mg O2/l)	Log(x)	1.4 (0.5 - 4.5)
Nitrite (mg NO2/l)	Log(x)	0.03 (0.01 - 0.34)
Nitrate (mg NO3/l)	Log(x)	7 (0.3 - 43.1)
Ammonia (mg NH4/l)	Log(x)	0.05 (0.01 - 0.6)
OrthoPhosphate (µg PO4/l)	Log(x)	60 (10 - 880)
TotalPhosphate (µg P/l)	Log(x)	50 (10 - 490)

Note: Ranges for quantitative variables: median (min-max) or modalities for qualitative variables: number of sites per modality.

Rivers ranged from small to large order (from 1 to 7), were part of small to large catchments (from 1 to 13312 km^2; median = 99 km^2) and were at altitude from 2 to 1520 m above the sea level (median = 217 m). Out of the 290 sites, 102 were slightly perturbed, with no or slight hydro-morphological disturbances and water parameters corresponding to "very good" or "good" status (French stream water quality evaluation system; French Water Agency, 2000). The others 188 sites were considered as impacted sites. For this work, the 290 sites were divided in two datasets: the calibration dataset CAL-80 containing 80 sites randomly selected from the 102 quasi-undisturbed sites (MDC) and the analysis dataset AN-210 including the 188 impacted sites and the 22 other MDC sites. Homogeneous distribution of sites on the French territory (see Fig. 1) and along environmental and pressure gradients were checked for representativeness.

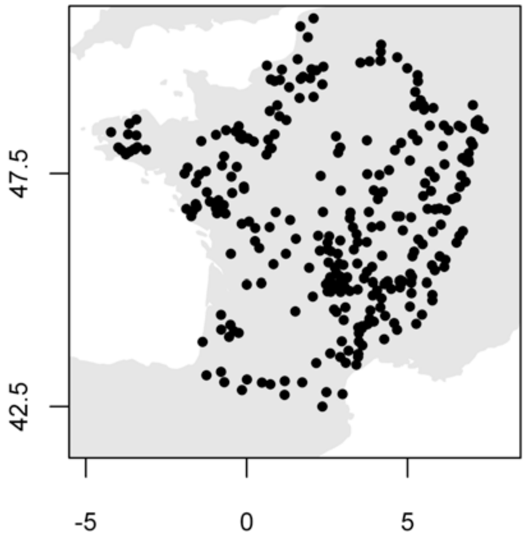

Figure 1: Location of the 290 sites.

Statistical analysis

Definition of environmental and human-induced pressure gradients

Hill and Smith analysis (Hill and Smith, 1976) was used to summarized the nine environmental variables (quantitative and qualitative; see Table 1) into three independent environmental gradients (*EG1*, *EG2* and *EG3*) respectively the first three axis of the analysis.

Four synthetic anthropogenic pressure gradients were developed. The global synthetic degradation gradient resumed all the human-induced pressure variables while the three other synthetic gradients were respectively related to hydrological, morphological and water quality human-induced degradations (see Table 1). PCA (principal composant analysis) and MCA (multiple correspondence analysis) were used to analyse respectively quantitative (water quality gradient) and qualitative (hydrological and morphological degradation gradients) variables and Hill and Smith analysis was used to examine jointly quantitative and qualitative variables (global degradation gradient). Consequently, quantitative variables were log-transformed when necessary to better fulfil the PCA normality assumption and the number of meaningful axis was determined by examining the cumulative inertia of the first few axes. For each pressure gradient, the first axis of the analysis was kept and parted in four classes corresponding in four levels of pressure (gp1 = slight pressure to gp4 = strong pressure) using K-means algorithm (Hartigan & Wong, 1979). It was then verified for each variable that the level of pressure increase along the synthetic gradients.

Metrics calculation and standardization

Ninety-three candidate metrics described in the scientific literature and expected to be impacted by different human-induced degradations were calculated (see Table 2). Metrics were based on biological and ecological functional traits (MetFUNC) (16 for FI, 17 for DI, 14 for MA and 21 for MI), on species composition of the samples (MetTAX) (4 for FI and DI, 3 for MA and 9 for MI) or corresponded to previously published indexes (MetIND) (2 for FI and 1 for DI, MI and MA). Following Pont et al. (2006, 2007), metrics were transformed to discard the effect of the natural environmental gradients. Metrics were expressed by the models E:

(1) Biological metric ~ *EG1* + *EG2* + *EG3* + Residuals

Parameters of these models were estimated using CAL-80 MDC sites. Metrics were then predicted for the AN-210 sites and residuals of the models standardized by the mean and the standard deviation of the CAL-80 residuals. For each site, values of the metrics were replaced by the standardized residuals of the models (i.e. the deviations between observed metrics values and predicted values for MDC). Metrics based on number of species (i.e. count data) were modelled using log-linear models (McCullagh and Nelder, 1989). Metrics relying on continuous positive data and proportional data were respectively log-transformed or square-root arcsin transformed when necessary and modelled using multiple linear models. Models' predictive reliability was assessed by a split-sampling cross-validation method (Harrel et al. 2001; Logez, 2010). Independence of the metrics to the environmental variables has been checked for the MDC sites of CAL-80 before and after transformation using analysis of variance procedures.

Biological responses to human pressures

Biological responses to human pressure were assessed using the transformed metrics (i.e. independent of the natural environmental gradient). In order to consider the possible combined effect of the different types of pressure, data were analyzed in three steps. First, responses of metrics to global pressure gradient were tested for the 210 sites of the AN-210 dataset (step 1). In a second step, responses of these metrics were tested for the three types of pressure (i.e. water quality, hydrological and morphological degradations) and a particular pressure (i.e. the presence of an impoundment at the site) for all the sites of the AN-210 dataset (step 2). Finally, the same analysis was made for each single pressure removing all the sites strongly impaired by other types of pressure (step 3).

The number of sites impacted by one single pressure in the case of morphological degradations (gp1 = 77; gp4 = 4) and the presence of an impoundment (no = 102; yes = 2) was too low to support statistical tests. Thus, the step 3 was not applied for these two pressures.

Kruskal-Wallis non-parametric post hoc tests were used to determine the effect of single and combined human pressures on BQE's metrics. Three criteria were used to describe the metric's responses to pressures: (i) the significance of the metric response between MDC (gp1) and high levels of pressure (gp4), (ii) the discriminatory efficiency (DE; Ofenböck et al. 2004) of the metric, i.e. the percentage of highly impacted sites (gp4) with metric values inferior (superior) to the 5^{th} (95^{th}) percentile of the MDC sites (gp1) for increasing (decreasing) metrics, (iii) and the sensitivity of the response, i.e. the lowest level of pressure detected by the metric. Significant difference between MDC reference sites (gp1) and slightly impacted sites from gp2 (respectively gp3 or gp4) corresponded to high (H) (respectively middle (M) or low (L)) sensitivity of the metric to change. In order to compare BQEs among them, weighted mean of sensitivity (WS) were calculated for each BQE, allocating weigh for the three situations: respectively 1, 2 and 3 for late, middle and early response. WS close to 3 indicates high sensitivity of the metrics to the pressure, i.e. response to low level of pressure, while WS close to 1 indicates low sensitivity, i.e. response only to high level of pressure. Sensitivity was not relevant for presence/absence data and was not determined for the presence of impoundment. All the statistical analyses were implemented using the R software (version 2.10.1)

Results

Environmental gradients and anthropogenic pressure indices

The three first axis of the environmental variables analysis represented 53.5% of the total inertia with 31.6% explained by the first axis (*EG1*), 12.5% explained by the second axis (*EG2*) and 9.3% explained by the third axis (*EG3*). *EG1* was related to rivers longitudinal gradient, increased with altitude and mean slope and decreased with mean width and catchment area. *EG2* was related to the same variables but did not suggest evident interpretation. *EG3* was related to geological types from siliceous to calcareous.

The global synthetic pressure index (first axis of the analysis including all pressure variables) explained 20.1% of the total inertia, decreased when the oxygen saturation increased and increased with all the other chemical variables (e.g. total phosphorus, nitrate) and all the hydro-morphological degradations variables (from modality "no" to "high"; see Table 1). The synthetic indexes corresponding to water quality, hydrological and morphological degradations (first axes of the analyses including respectively only water quality, hydrological and morphological variables) accounted respectively for 55.9%, 23.6% and 22.5% of the total inertia and were related to an increase in modifications and degradations of the associated variables.

Natural environment and standardized biological metrics

For the CAL-80 undisturbed sites, sixty out the 93 untransformed metrics tested varied significantly (ANOVA p-value > 0.05) along the main natural environmental gradient (*EG1*). Not surprisingly, the 33 unvarying metrics included the four EQR tested (IBD: *M22*, IPR: *M43*, EFI: *M44*, IBGN: *M91*). However, p-value of *M91* and *M43* were close to 0.05 (respectively equal to 0.052 and 0.068). Sixteen metrics out of these 33 metrics were related to DI communities, eight to MI and to MA and only two were related to FI communities. As expected, ANOVA procedures did not reveal any residual effect of environment for the 93 transformed metrics (e.g. % of the Plecoptera taxa: *M89*; see Figure 2). In addition, the cross-validation revealed the good stability of the models. The root mean squared error (RMSE) and

Spearman rank correlation between predictions and observations were coherent between CAL-80 and the 200 resamples and deviations rarely exceeding 15% of the initial statistics.

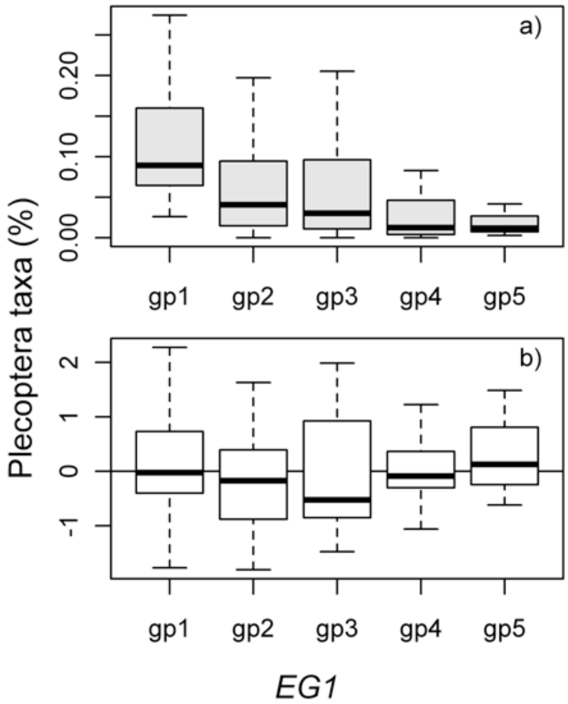

Figure 2: Boxplots of the percentage of Plecoptera taxa along the first environmental gradient (EG1) for undisturbed sites (CAL-80) before a) and after b) transformation.

Biological responses to anthropogenic pressure gradients

Four types of metric were identified regarding its responses to pressures (see examples in Figure 3 and exhaustive results in Table 2): metrics not impacted by pressures (i.e. no response, 27 metrics), decreasing metrics (i.e. negative response, 34 metrics), increasing metrics (i.e. positive response, 26 metrics), and decreasing/increasing metrics depending on the nature of the pressure (six MA metrics).

Metric types' comparison

All the indexes (MetIND), two thirds of the chosen functional metrics (MetFUNC) and a third of the taxonomy-based metrics (MetTAX) responded to the global degradation gradient. MetIND and MetFUNC showed the strongest responses with median discriminatory efficiency (DE) respectively equal to 60% (range: 20-80%) and 49% (range: 6-91%). Half of the MetFUNC were highly sensitive (H: 52%, M: 10%, L: 38%, WS=2.1) while MetIND were globally less sensitive (H: 43%, M: 14%, L: 43%, WS=2). Most of the MetTAX showed weak responses and low sensitivity (DE 23% (9-34%) and H: 40%, L: 60%; WS=1.8). Similar results were observed for the specific pressure gradients, i.e. MetIND and MetFUNC responding more frequently and stronger than MetTAX.

Table 2: Responses of the 93 biological metrics to human pressures: Discriminatory efficiency (positive or negative) Sensitivity

Metric	*BQE*	*Type*	Description	Global	Water quality		Hydrological		Morphological	Impoudment
				Step 1	Step 2	Step 3	Step 2	Step 3	Step 2	Step 2
M1 (12)	*DI*	*MetFUNC*	O2 Intolerant (RA)	57% (-) M	54% (-) M	33% (-) L	19% (-) M		11% (-) E	11% (-)
M2 (12)	*DI*	*MetFUNC*	O2 Tolerant (RA)	43% (+) E	53% (+) E	44% (+) E			11% (+) L	11% (+)
M3 (12)	*DI*	*MetFUNC*	Aquatic strict (RA)							
M4 (12)	*DI*	*MetFUNC*	Terrestrial (RA)				21% (+) L		4% (+) L	
M5 (12)	*DI*	*MetFUNC*	Oligosaprobic (RA)							
M6 (12)	*DI*	*MetFUNC*	Mesosaprobic (RA)							15% (-)
M7 (12)	*DI*	*MetFUNC*	Alphamesosaprobic (RA)		14% (+) L	19% (+) L	11% (+) L			
M8 (12)	*DI*	*MetFUNC*	Alphamesotopolysabrobic (RA)	49% (+) E	61% (+) E	48% (+) E			15% (+) L	11% (+)
M9 (12)	*DI*	*MetFUNC*	Polysaprobic (RA)							
M10 (12)	*DI*	*MetFUNC*	Oligotrophic (RA)							
M11 (12)	*DI*	*MetFUNC*	Mesotrophic (RA)	9% (-) E	14% (-) M	7% (-) M				
M12 (12)	*DI*	*MetFUNC*	Mesoeutrophic (RA)							33% (-)
M13 (12)	*DI*	*MetFUNC*	Trophe indifferents (RA)	43% (-) M	37% (-) M	30% (-) L	28% (-) M		30% (-) L	22% (-)
M14 (12)	*DI*	*MetFUNC*	Sensitive N-Autotrophic (RA)		10% (-) L					
M15 (12)	*DI*	*MetFUNC*	Tolerant N-Autotrophic (RA)							
M16 (12)	*DI*	*MetFUNC*	Facultative N-Heterotrophic (RA)	43% (+) E	37% (+) E	26% (+) E				
M17 (12)	*DI*	*MetFUNC*	Obligatory N-Heterotrophic (RA)	20% (+) L			18% (+) L		15% (+) L	15% (+)
M18	*DI*	*MetTAX*	S	34% (+) E	20% (+) M	22% (+) L				
M19 (7)	*DI*	*MetTAX*	Evenness ($E = e^H / S$)							
M20	*DI*	*MetTAX*	Ni							
M21 (10)	*DI*	*MetTAX*	Shannon diversity (H)	9% (+) L	10% (+) M					
M22 (4)	*DI*	*MetIND*	IBD (French Diatoms Index)	80% (-) E	61% (-) M	52% (-) M	21% (-) M	7% (-) L	7% (-) E	19% (-)
M23 (5)	*FI*	*MetFUNC*	Insectivorous (S)							63% (-)
M24 (5)	*FI*	*MetFUNC*	Insectivorous (RA)	37% (-) L			35% (-) L		30% (-) L	48% (-)

M25 [5]	*FI*	*MetFUNC*	Omnivorous (S)							
M26 [5]	*FI*	*MetFUNC*	Omnivorous (RA)	57% (+) L	25% (+) L		14% (+) M		19% (+) E	11% (+)
M27 [5]	*FI*	*MetFUNC*	O2 Intolerant (S)	71% (-) L			46% (-) M	18% (-) L	48% (-) M	85% (-)
M28 [5]	*FI*	*MetFUNC*	O2 Intolerant (RA)	91% (-) E	61% (-) M	37% (-) M	32% (-) M		37% (-) E	41% (-)
M29 [5]	*FI*	*MetFUNC*	O2 Tolerant (S)	74% (+) E	49% (+) L	52% (+) L			26% (+) L	33% (+)
M30 [5]	*FI*	*MetFUNC*	O2 Tolerant (RA)	89% (+) E	71% (+) M	52% (+) M	32% (+) L		48% (+) E	44% (+)
M31 [5]	*FI*	*MetFUNC*	Habitat Intolerant (S)	66% (-) L	41% (-) L		35% (-) L		41% (-) L	52% (-)
M32 [5]	*FI*	*MetFUNC*	Habitat Intolerant (RA)	14% (-) L	3% (-) L					
M33 [5]	*FI*	*MetFUNC*	Habitat Tolerant (S)	69% (+) E	53% (+) E	52% (+) M				33% (+)
M34 [5]	*FI*	*MetFUNC*	Habitat Tolerant (RA)	91% (+) E	56% (+) M	41% (+) M	39% (+) M		44% (+) E	41% (+)
M35 [5]	*FI*	*MetFUNC*	Rheopar (S)	60% (-) L			28% (-) L		44% (-) L	70% (-)
M36 [5]	*FI*	*MetFUNC*	Rheopar (RA)	91% (-) E	56% (-) M	52% (-) M	39% (-) L		52% (-) E	63% (-)
M37 [5]	*FI*	*MetFUNC*	Lithophilic (S)	63% (-) L			32% (-) L		48% (-) L	74% (-)
M38 [5]	*FI*	*MetFUNC*	Lithophilic (RA)	91% (-) E	54% (-) E	52% (-) E	33% (-) L		41% (-) L	30% (-)
M39	*FI*	*MetTAX*	S							
M40 [7]	*FI*	*MetTAX*	Evenness ($E = e^H / S$)							
M41	*FI*	*MetTAX*	Ni							
M42 [10]	*FI*	*MetTAX*	Shannon diversity (H)							15% (-)
M43 [1]	*FI*	*MetIND*	IPR (French Fish Index)	49% (-) L			32% (-) M	21% (-) M	22% (-) L	22% (-)
M44 [9]	*FI*	*MetIND*	EFI (European Fish Index)	74% (-) L	47% (-) L		21% (-) M		15% (-) L	15% (-)
M45 [3]	*MA*	*MetFUNC*	Water quality Intolerant (S)	9% (-) M			2% (-) M		7% (-) L	
M46 [3]	*MA*	*MetFUNC*	Water quality Intolerant (C)				2% (-) E			
M47 [3]	*MA*	*MetFUNC*	Water quality Tolerant (S)		41% (+) M	56% (+) M				19% (-)
M48 [3]	*MA*	*MetFUNC*	Water quality Tolerant (C)		31% (+) L	26% (+) L				
M49 [3]	*MA*	*MetFUNC*	Amphibious (S)			37% (+) M	25% (-) M			22% (-)
M50 [3]	*MA*	*MetFUNC*	Amphibious (C)				30% (-) M	29% (-) L		37% (-)
M51 [3]	*MA*	*MetFUNC*	Aquatic strict (S)			15% (+) L				33% (-)

M52	*(3)*	*MA*	*MetFUNC*	Aquatic strict (C)		39% (+) M	56% (+) M				
M53	*(3)*	*MA*	*MetFUNC*	Helophytes (S)	43% (+) L	41% (+) M					
M54	*(3)*	*MA*	*MetFUNC*	Helophytes (C)							
M55	*(3)*	*MA*	*MetFUNC*	Intolerant species (S)							26% (+)
M56	*(3)*	*MA*	*MetFUNC*	Intolerant species (C)	6% (+) L						7% (+)
M57	*(3)*	*MA*	*MetFUNC*	Tolerant species (S)		37% (+) E	52% (+) E				26% (-)
M58	*(3)*	*MA*	*MetFUNC*	Tolerant species (C)			30% (+) L				
M59		*MA*	*MetTAX*	S		27% (+) E	33% (+) E				22% (-)
M60	*(7)*	*MA*	*MetTAX*	Evenness ($E = e^H / S$)							
M61	*(10)*	*MA*	*MetTAX*	Shannon diversity (H)			41% (+) L				22% (-)
M62	*(6)*	*MA*	*MetIND*	IBMR (French Macrophytes Index)	20% (-) E	15% (-) M	15% (-) M			11% (-) L	7% (-)
M63	*(11)*	*MI*	*MetFUNC*	Deposit Feeder (RF)			41% (-) M				
M64	*(11)*	*MI*	*MetFUNC*	Shredder (RF)							
M65	*(11)*	*MI*	*MetFUNC*	Maximum size < 2.5 cm (RF)					11% (-) L		
M66	*(11)*	*MI*	*MetFUNC*	Maximum size > 8 cm (RF)							
M67	*(11)*	*MI*	*MetFUNC*	Life cycle < 1 year (RF)	49% (-) L	47% (-) M	33% (-) M			19% (-) L	11% (-)
M68	*(11)*	*MI*	*MetFUNC*	Lyfe cycle > 1 year (RF)	29% (+) L	36% (+) M	33% (+) L				
M69	*(11)*	*MI*	*MetFUNC*	Number of cycle per year < 1 (RF)	40% (-) E	25% (-) M	22% (-) M			30% (-) L	19% (-)
M70	*(11)*	*MI*	*MetFUNC*	Number of cycle per year > 1 (RF)	29% (+) L	22% (+) L				30% (+) L	19% (+)
M71	*(11)*	*MI*	*MetFUNC*	Ovoviviparity (RF)	86% (+) E	68% (+) M	52% (+) M	23% (+) M		37% (+) L	26% (+)
M72	*(11)*	*MI*	*MetFUNC*	Aquatic Passive Dispersal (RF)	57% (+) E	51% (+) M	48% (+) L			22% (+) L	
M73	*(11)*	*MI*	*MetFUNC*	Aerial Active Dispersal (RF)	77% (-) E	75% (-) M	81% (-) M	21% (-) L		48% (-) L	30% (-)
M74	*(11)*	*MI*	*MetFUNC*	Gravel (RF)	51% (-) L					48% (-) L	59% (-)
M75	*(11)*	*MI*	*MetFUNC*	Sand (RF)	46% (-) M	36% (-) L	56% (-) L	19% (-) L		33% (-) L	30% (-)
M76	*(11)*	*MI*	*MetFUNC*	Microphytes (RF)	40% (+) E	29% (+) M	30% (+) M			11% (+) L	
M77	*(11)*	*MI*	*MetFUNC*	Mud (RF)							
M78	*(11)*	*MI*	*MetFUNC*	Xenosaprobic (RF)	26% (-) E	25% (-) L				19% (-) L	11% (-)
M79	*(11)*	*MI*	*MetFUNC*	Oligosaprobic (RF)	49% (-) E	37% (-) L		12% (-) M		22% (-) L	19% (-)

M80 [(11)]	*MI*	*MetFUNC*	Polysaprobic (RF)	26% (+) L	17% (+) L				26% (+) L	26% (+)
M81 [(11)]	*MI*	*MetFUNC*	Oligotrophic (RF)	54% (-) E	39% (-) M		11% (-) M		26% (-) L	15% (-)
M82 [(11)]	*MI*	*MetFUNC*	Mesotrophic (RF)							19% (-)
M83 [(11)]	*MI*	*MetFUNC*	Eutrophic (RF)	43% (+) E	29% (+) L		12% (+) L		22% (+) L	15% (+)
M84	*MI*	*MetTAX*	S							
M85 [(7)]	*MI*	*MetTAX*	Evenness ($E = e^H / S$)							
M86	*MI*	*MetTAX*	Ni							7% (-)
M87 [(10)]	*MI*	*MetTAX*	Shannon diversity (H)	34% (-) L						15% (-)
M88	*MI*	*MetTAX*	Ephemeroptera, plecoptera, trichoptera taxa (%)	23% (-) L	12% (-) L				7% (-) L	7% (-)
M89	*MI*	*MetTAX*	Plecoptera taxa (%)	20% (-) E	19% (-) M	11% (-) M	16% (-) E		22% (-) L	
M90 [(8)]	*MI*	*MetTAX*	GoldInv							33% (+)
M91 [(13)]	*MI*	*MetIND*	IBGN (French Macroinvertebrate Index)	60% (-) M	41% (-) L	33% (-) L	16% (-) L		37% (-) L	30% (-)
M92 [(2)]	*MI*	*MetIND*	BMWP (Biological Monitoring Working Party)	26% (-) L	32% (-) L		16% (-) M		37% (-) E	22% (-)
M93 [(2)]	*MI*	*MetIND*	ASPT (Average Score per Taxa)	80% (-) E	71% (-) M	44% (-) L	37% (-) M		52% (-) E	22% (-)

Note: BQEs: DI = diatoms, FI = fish, MA = macrophytes, MI = macro-invertebrates; Metrics type: MetFUNC = functional traits-based metrics, MetTAX = taxonomy-based metrics, MetIND = Index. Units: S = richness, Ni = number of individuals, RF = relative frequency, C = coverage of a taxon/traits, i.e. indices of cumulated density, RA = relative abundance. Discriminatory efficiency: the percentage of sites highly impacted (gp4) with metric values inferior to the extreme percentile (5% for increasing metrics and 95% for decreasing metrics) of the MDC reference sites (gp1). (-) for negative and (+) for positive responses. L = late response, M = middle term response, E = early response. Sources: (1) AFNOR, 2004b; (2) Armitage et al. 1983; (3) Christian Chauvin (pers. comm.); (4) Coste et al. 2009; (5) Haury et al. 2006; (6) Pont et al. 2006; (7) Pielou, 1966; (8) Pinto et al. 2004; (9) Pont et al. 2007; (10) Shannon et al. 1949; (11) Tachet et al. 2006; (12) Van Dam, 1994; (13) Genin, 2003.

Global degradation

From the 51 metrics responding significantly to the global pressure gradient (significance rates FI: 73, MI: 68, DI: 45, MA: 22 % of the total number of metrics for each BQE), 46 had a discriminatory efficiency over 20% (Table 2). Most of the 16 FI metrics showed a strong response (DE median value 70%, range: 14-91%) and half of them to low pressure level (H: 44%, L: 56%; WS=1.9). The ten DI metrics and the 21 MI metrics showed in average comparable response intensities (DE medians: 43% and ranges DI: 9-80%, MI: 20-86%) and mostly high sensitivity for DI (H: 60%, M: 20%, L: 20%; WS=2.4) and MI (H: 52%, M: 10%, L: 38%; WS=2.1). The four MA metrics responded mainly weakly (DE range: 6-43%) with low sensitivity (H: one, M: one, L: two metrics; WS=1.8).

Water quality degradation

Forty-eight metrics responded significantly to water quality degradation gradients (significance rates MI: 61, DI: 50, FI: 50, MA: 39%) and 39 had a DE over 20% (Table 2). Most of the FI metrics presented rather low sensitivity (H: 18%, M: 36%, L: 45%; WS=1.7) but the strongest responses (DE median 53%, range: 3-71%). Whereas MI metrics showed middle or low sensitivity (M: 53%, L: 47%; WS=1.5), most of the DI (H: 27%, M: 55%, L: 18%; WS=2.1) and MA (H: 29%, M: 57%, L: 14%; WS=2.2) metrics highly sensitive. However, their responses presented comparable median intensities with wider ranges for DI and MI than for MA (DE medians (ranges) respectively MA: 37 (15-41%), DI: 37 (10-61%) and MI: 36% (12-75%)). Thirty-eight metrics still responded to water quality diminution when considered as single pressure (significance rates MA: 56, DI: 41, MI: 39, FI: 32%). DE median values (MA: 30, DI: 35, MI: 37, FI: 52%) and WS (MA: 2, DI: 2, FI: 2, MI: 1.6) remained quasi-unchanged for the four BQEs.

Hydrological degradation

Thirty-three metrics responded to the hydrological degradation gradient (significance rates FI: 59, MI: 32, DI: 27, MA: 22%) and 20 had a DE over 20% (Table 2). Metrics responses were generally weak (BQEs DE medians < 32% and BQEs DE range 2-46%) with low sensitivity (FI WS=1.5, MI WS=1.7, DI WS=1.5) except for the macrophytes (WS=2.3; only two highly sensitive metrics: M46 (MA) and M89 (MI), see Table 2). Only five metrics still showed significant responses when considering hydrological degradation as single pressure (M50 (MA), M22 (DI), M65 (MI), M27 and M43 (FI); see Table 2) and with rather low sensitivity (FI WS=1.5, MI WS=1, DI WS=1, MA WS=1). Morphological degradation

Forty-two metrics presented responses to the morphological degradation gradient (significance rates FI: 64, MI: 61, DI: 32, MA: 11%) and 28 had a DE over 20% (Table 2). Most of the FI metrics showed middle intensity responses (DE median 41% and range: 15-52%) and a third were highly sensitive (H: 36%, M: 7%, L: 57%; WS=1.8). Most of the MI metrics showed weak responses (DE median 26%, range: 7-52%) to high level of pressure (H: 11%, L: 89%; WS=1.2). DI and MA metrics showed mainly very weak responses (DE medians < 11% and range 4-30%). While all the MA metrics showed low sensitivity (WS=1), a third of DI metrics were highly sensitive (H: 29%, L: 71%; WS=1.6).

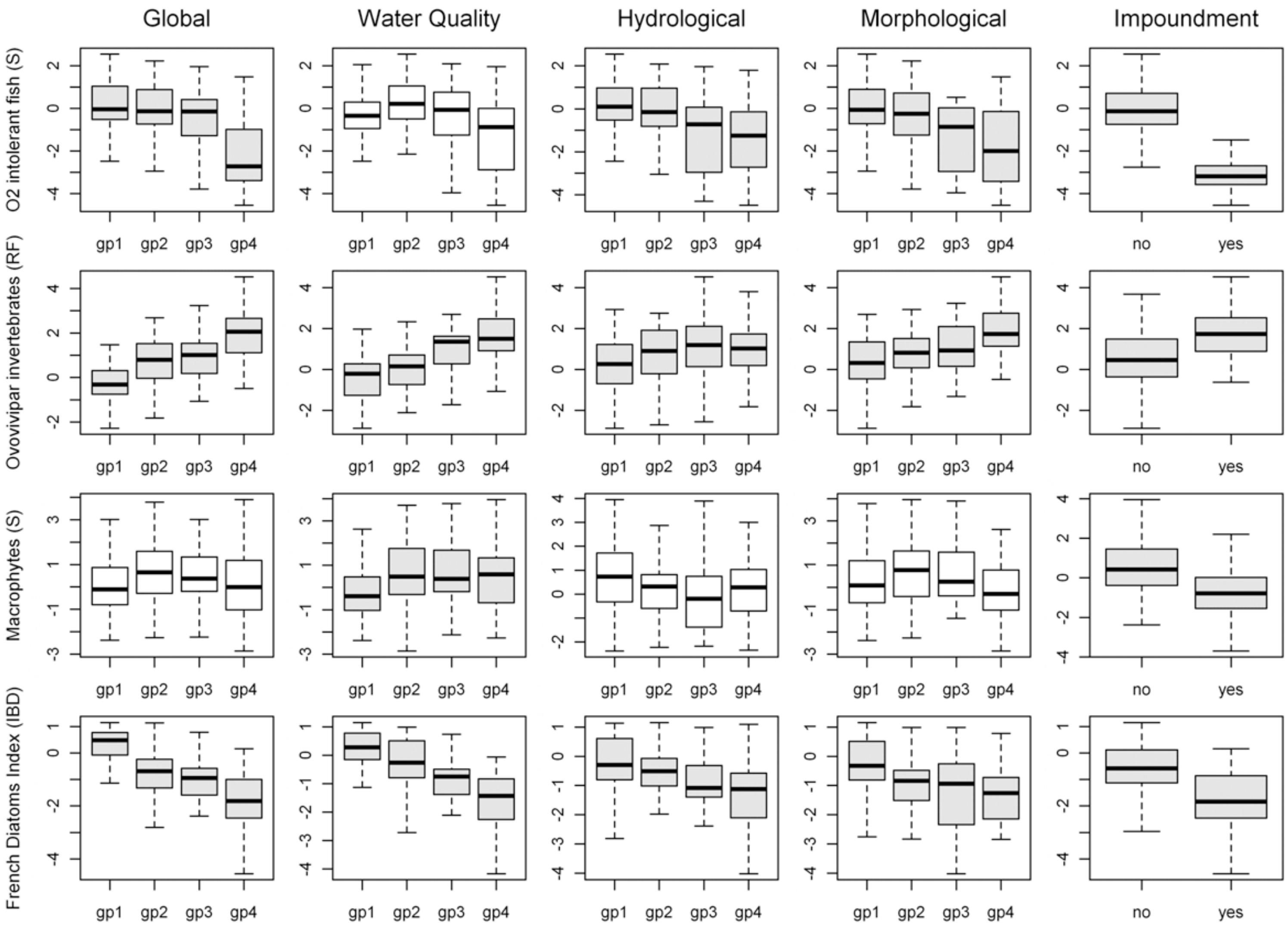

Figure 3: Boxplots of the responses of four metrics to human-induced perturbations, i.e. global, water quality, hydrological, morphological degradation gradients and the presence of an impoundment. White boxes (respectively grey boxes) for non significative (significative) response.

Impoundment

Fifty-five metrics were able to detect the presence of an impoundment (significance rates FI: 77, MI: 65, MA: 56, DI: 36%) and 32 had a DE over 20% (Table 2). Most of the FI metrics responded with medium intensities (DE median 41%, range: 11-85%) while most of the MI, MA and DI metrics responded weakly (DE median (range) respectively 19% (7-59%), 22% (7-37%, and 15% (11-33%)).

Discussion

The main purpose of this study was to test whether the responses to human-induced changes were comparable among four BQEs: macrophytes, diatoms, fish and macro-invertebrates. More particularly, potentials to detect human-induced changes were compared in term of intensity of the response (i.e. discriminatory efficiency) and sensitivity to changes (i.e. precocity of the response along pressure gradients). Metrics were transformed beforehand to keep only the part of the signal linked to human-induced changes. Sixty-six metrics out of the 93 transformed metrics were able to detect at least one of the five anthropogenic pressures considered. Among them, the strongest efficiency and sensitivity were observed for MetFUNC and MetIND. Also, BQEs responded differently, depending of the type of human pressure. As pointed out by Johnson et al. (2006a), discriminatory efficiency and sensitivity varied noticeably among individual metrics, for a given BQE. In addition, the metric responses to single pressures were comparable to responses to combined pressures in the case of water quality degradation, but typically weaker for hydrological pressures.

Taking into account natural environmental diversity in the analysis

As advised by Pont et al. (2007), the calibration dataset cover mainly the range of natural environmental diversity of the French territory what allows us to provide robust predictions of the metrics under MDC. Before transformation, two third of the tested metrics varied significantly with the main environmental variables when only considering weakly disturbed sites. These results are in agreement with previous studies showing that local and regional natural environmental factors are major drivers of change in BQEs structure and function (e.g. Hughes et al. 2009; Johnson et al. 2006a; Logez et al. 2010). This implies that distinction of these two sources of variability is not trivial and should be taken into account before considering biological responses strictly due to human-induced stressors.

Moreover, in this study, biological communities show different responses to the environmental gradients. While most of the FI and MI metrics were correlated with the natural environmental variations, DI metrics were more frequently uncorrelated. This result suggests that for the selected metrics, DI communities are less sensitive to natural environment variability than the three other groups. Previous studies (e.g. Potapova and Charles, 2002) have shown that environment and especially water mineral content plays the most important role in structuring diatom assemblages in rivers. Natural water physico-chemistry variability was not directly considered in the environmental gradients, but approximated through parameters describing the natural longitudinal gradient (e.g. geological type, altitude, mean width). Then, our models were probably not able to catch all the variability of biological communities associated to these environmental parameters (in particular for diatoms). More generally, because BQEs are known to be related differently to their environment, common important environmental parameters and pressure descriptors were selected in this study. Although introducing uncertainty in our model, such compromise is indispensable for comparison.

All the tested metrics were independent of the natural environment for undisturbed sites (CAL-80) after the transformation. These results lead us to believe that the method used is appropriate to discard the part of the metrics variability linked to the direct effect of natural phenomena from the analysis while still considering river as a continuum (Vannote et al. 1980), i.e. not splitting our dataset into different river types.

Choosing metric's types

Indexes (MetIND) and functional traits based metrics (MetFUNC) were generally more sensitive and showed stronger responses to pressures than taxonomy-based metrics (MetTAX). Indeed, several authors have demonstrated that ecological and biological functional traits are adapted for large-scale approaches (e.g. Statzner et al. 2001) and are able to integrate more general phenomena than taxonomy-based metrics (e.g. Dolédec et al. 2006). Also, the five multi-metric indexes tested in this study (see Table 2: *M22*, *M43*, *M44*, *M62*, *M93*) showed strong responses to global degradation. Apart from *M43* (IPR: French fish index) and *M62* (IBMR: French Macrophytes Index) which did not detect respectively water quality and morphological perturbations, multi-metric indexes were significantly affected by the three specific pressure gradients and the presence of an impoundment. Their responses were always negative. These results clearly support the use of ecological and biological functional traits metrics to built multi-metric indexes in order to assess river biotic integrity. We advocate that concerns should not only focus on the BQE but also on the nature of the metric selected to monitor the effects of stressors of interest (i.e. underlying processes, types, and units).

Intensity and sensitivity of BQEs responses to general degradation

The sensitivity and the intensity of metrics responses to human pressure fluctuated considerably among biological groups. DI and MI metrics appear to be more sensitive to the degradation of the global river condition (general gradient) than FI metrics and reacted to lower levels of pressure. However, FI metrics presented the strongest responses. These differences can be partly linked to the migratory capacities of fish and their longer life cycles. FI are generally able to move further in the stream to find favourable conditions when degradations occur. Consequently, as long as favourable habitats and conditions are accessible, changes will remain undetected by metrics. However, when favourable habitats are no longer accessible, FI will show dramatic responses resulting in strong responses to high degradations. Conversely, short life cycle and sedentary organisms such as DI will be impacted by lower level of pressure as soon as local favourable conditions are degraded. Less sensitive but more intense responses of FI metrics would be more adapted to detect high modifications of the stream or first results of restoration measures while DI and MI metrics would be more interesting to detect first impacts of degradation and more advanced stages of restoration.

Compared to the three other groups, a lower number of MA metrics responded significantly and these responses were weaker and less sensitive. These differences can be due to the positive/negative responses of the MA metrics. For example, *M47*, i.e. the number of water quality tolerant macrophyte species, were positively impacted by a diminution of the water quality (DE = 41%, middle term response) but negatively impacted by the presence of an impoundment (DE = 19%). More generally, all MI metrics showing contrasted responses to specific pressures are insensitive to global degradation. Therefore, in the case of multi-impacted sites, losses caused by a pressure could be compensated by benefices linked to other pressures revealing possible antagonist effects. Such metrics are particularly interesting to detect impacts linked to different types of pressure but could be confounding when assessing general degradation of multi-impacted sites.

BQEs responses to hydrological, morphological, and water quality degradations

As in previous studies, metric responses were globally stronger for global degradation than for specific pressures (e.g. Hering et al. 2006). In addition, our study demonstrates that metrics are more sensitive to global pressure. Also, among specific pressures, water quality degradations resulted in the best metric's responses in term of intensity and sensitivity.

In agreement with previous studies, the four BQEs showed significant responses to water quality degradation (e.g. Hering 2006; Johnson et al. 2006a, 2009; Justus et al. 2010). In our study, DI and MA metrics were more sensitive to water quality (response to low to moderate levels of water quality degradation) than FI and MI metrics (response to moderate to high levels of water quality degradation). However, as in Johnson et al. (2009) and as for global pressure, the strongest responses were observed for FI metrics.

Contrary to Hughes et al. (2009) showing that MI and MA were more impacted by water velocity changes and FI by physical disturbance, our results suggest that FI is the most impacted biological group by hydrological perturbations followed by MA and DI. Nonetheless, responses were rather weak (DE median < 32%) and significant for high pressure level for all the groups (WS<1.7) except MA (WS=2.3). Besides, FI and MI's metrics showed the strongest responses to morphological degradations. Although FI, DI and MI metrics seem to be the most sensitive metrics to this pressure, response occurs generally for high degradation.

In previous works, hydrological and morphological perturbations have generally been combined into a single pressure gradient identified as habitat degradation. Our results tend to be in adequacy to those of Hering et al. (2006) showing that MI and FI responded to reach scale hydro-morphological gradients and contrast with those of Johnson et al. (2006a) showing that FI and MA metrics showed higher response than either DI or MI metrics to alteration in such general habitat alteration gradient. These differences may be explained by author's choices concerning the variables describing pressures and the analysis settings.

The four BQE's metrics were affected by the presence of an impoundment and the highest responses were observed for FI metrics. Our results confirmed the particular ecological impact of the presence of an impoundment with relatively strong responses of the four BQEs to this pressure. Indeed, this type of river modification is known to strongly alter both water quality and hydro-morphological conditions upstream of a weir or a dam (e.g. Baxter, 1977; Feld et al. 2011).

In more general terms, fish metrics showed strong intensity but rather late responses, i.e. first answer to high level of pressure. Diatom metrics present mainly medium (global and water quality degradations) to weak (morphological and hydrological degradations) intensity responses and apart from hydrological pressure, the earliest responses, i.e. answer to low level of pressure. Except for global degradation, macro-invertebrate metrics show mainly late answer to pressure and intensities of the responses were medium (global, water quality, and morphological degradations) to weak (hydrological degradation). Macrophyte metrics showed the most sensitive responses to water quality and hydrological degradation.

Detecting combined and single pressure effects

For both water quality and hydrological degradations, the number of significant responses decreased sharply from pressure types (step 2) to single pressure analysis (step 3). For instance, for the FI metrics, the significance rate fell from 59 to 9% for hydrological pressure and from 50 to 32% for water quality degradation. In addition, the same pattern as for the step 2 was observed, i.e. better responses in term of intensity and sensitivity for water quality than for hydrological perturbations.

Intensity and sensitivity of the four BQEs responses to single water quality degradation were quasi-unchanged when removing sites strongly impacted by hydrological or morphological degradations (from step 2 to step 3). This result suggests that the effects on BQE's metrics observed in the step 2 were mainly due to water quality degradation and not to a combined effect with hydro-morphological degradations.

By contrast, for single hydrological pressure gradient, whereas responses of the FI metrics were quasi-unchanged, the MA, MI and DI metrics were less sensitive and show weaker responses when sites strongly impacted by water quality or morphological degradations were not considered (from step 2 to step 3). Therefore, the effects observed on FI metrics in the step 2 appear to be mainly due to hydrological changes and not to a combined effect with water quality and/or morphological degradations. At the opposite, MA, MI and DI metric's responses were probably largely due to the impact of water quality and/or morphological associated degradation. These results bring about new issues about the link between pressures. Unfortunately, understanding of the combined/accumulation effect of several types of pressure on river aquatic communities is typically poor (e.g. Pont et al. 2007) and questions such as the existence of cumulative or multiplicative (i.e. interaction) effects are still open.

Conclusions

Our study clearly demonstrated that the two main sources of variability of biological communities (natural environment diversity and anthropogenic disturbances) should be distinguished a priori when looking at human-induced stressors impacts. Also, when selecting "best" BQEs or metrics to detect stressor impacts, particular care should be taken concerning the nature of the metric. Indeed, indexes and functional traits-based metrics tend to better detect human-induced changes (stronger responses to lower level of pressure) than taxonomy-based metrics. Thus, to deepen such analysis, knowledge on BQEs' biological and ecological traits needs to be improved in particular for macrophytes. This work shows that the four BQEs' metrics are impacted differently by pressures, even if the responses are variable from a metric to another within a given group. Metric selection within a group might have as much importance to detect precisely human-induced change impacts as selecting a BQE. More generally, global and water quality degradations of the river appear to be better detected by BQEs' metrics than (in decreasing order) impoundments, morphological and hydrological degradations. Finally, given our results, there is all the reason to believe that hydrological degradation effects will be confounded with water quality and morphological degradation effects on biota if multi-impacted sites are not removed from the analysis. Nevertheless, the dataset used in this study was limited. Using a dataset covering a larger range of environmental and anthropogenic disturbances, our findings could be confirmed and generalized.

As river assessment research is turning towards multi-metric tools, it is of prime importance to be able to answer the following question before including metrics to indexes: Do the different types of pressure have additive effect, multiplicative or opposite effects? Furthermore, this study have analysed the influence of natural environment on metrics for undisturbed conditions but we believe that complex interactions exist between human pressure effects and the natural environment diversity, i.e. aquatic communities' responses to human pressure will be different depending on the natural environment. Such interaction effects on BQEs responses have not been enough investigated yet and are needed to assess ecological impacts towards restoration of the water bodies.

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Detecting the impact hierarchy of stressors from different spatial scales on aquatic communities

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Introduction

River systems in Europe have been altered all the way throughout history. Usages from log-floating to hydropower and industrialization were often associated with channel modification, water abstraction and impairment of water quality. With most riverine systems impacted by those adverse effects, only few river stretches remained pristine over time. Most alterations took place on a big scale affecting the flora and fauna of the rivers and the floodplains (Tockner et al. 2010). In contrast, river restoration is mostly applied on a rather small spatial scale due to financial and logistical restrictions. The successful implementation of these restoration measures by achieving the good ecological status has not been satisfactory so far (Jähnig et al. 2011). Fish and macrophyte communities have shown moderate positive reactions to restoration measures, while macroinvertebrates in many cases did not respond at all to the newly created habitats (Lorenz and Januschke 2011). Various studies have identified the strongest impact on aquatic communities from the catchment and its urban or agricultural utilization (Vondracek et al. 2005; Walsh et al. 2007). Land use can serve as a proxy to estimate the catchments' nativeness since intensive utilization is responsible for diffuse impacts on water quality and the structural conditions (Blann et al. 2009).

This study focuses on three aquatic organism groups: macroinvertebrates, fish and diatoms (as part of the phytobenthos). Due to their different size, dispersal capacity and habitat requirements for reproduction and feeding, they are known to respond to varying extends to different types of stressors. Phytobenthos indicates short-term effects on water quality like eutrophication or acidification. Community changes are also expected to occur due to water velocity or light level changes. By monitoring macrophyte communities the long-term trophic state of a river can be assessed and structural degradation identified. The quality of macroinvertebrate communities can give information about the saprobic status of the assessed river stretch, acidification and general degradation, while fish indicate a wide range of structural modifications and deficient connectivity within the river system. The environmental data in this study comprises hydromorphological and physico-chemical parameters on the sampling site, and land use proportions in the catchment and in a 100 x 1,000 m strip along the river. Though the physico-chemical data was sampled directly at the sampling site it was classified as large-scale impact. It was assumed that in most cases possible impacts already exist within the water body upstream.

A great amount of information has already been compiled about ecological requirements of specific species. Due to the similarity displayed by different species in their requirements, metrics have been developed which are independent from the presence or absence of specific taxa. These metrics are used to grasp habitat characteristics based on sensitivity or functional traits of taxonomic groups and to make river systems comparable.

The goals of the study are:

- to detect the hierarchy between stressors from different spatial scales in their impact on fish, invertebrates and diatoms. The identification of the impact hierarchy may help water managers to prioritize the measures that are applied in water restoration and to spend resources in a more effective way.
- to examine the response of functional and sensitivity metrics to specific stressors. This may help:
- to identify metrics for indication and
- to identify the dominant impact that is represented by the land use in our datasets.

Material and Methods

Dataset

The dataset used here comprises samples of three Biological Quality Elements (BQEs) taken in Austria (Table 1). For detailed information about the data origin see Melcher et al. (this report). Some of the stations overlap but the datasets were analysed separately due to the different amount of information that was available for each BQE. The stations are located in the lowland and mountainous regions of Austria and within the ecoregions 4, 9 and 11 (Illies 1978).

Table 1: Number of stations of the datasets for fish, macroinvertebrates and diatoms. All stations are located in Austria. Lowland <200 m, Mountain 200–800 m

BQE	Total Number of Stations	Lowland	Mountain
Fish	286	44	242
Macroinvertebrates	227	43	184
Diatoms	85	22	63

The datasets were extracted according to the following procedure:

1. Identification of stations for which data for fish, macroinvertebrates and diatoms were available at the species level. If various biotic sampling dates were present for the same station within one year, the optimal sampling date was chosen according to the following recommended time periods:

- Fish: August/September - recommendation according to FiBS (Dußling 2007)
- Invertebrates: March/April in catchments <1,000 km^2, June/July in catchments >1000 km^2 - recommendation according to AQEM (www.fliessgewaesserbewertung.de)

- Diatoms: August/September - recommendation according to Phylib (Bayerisches Landesamt für Umwelt 2006)

If biotic data was available for the same station for various years, the most recent year was chosen. Stations with catchments >5 000 km^2 and from alpine rivers (Altitude >800 m) were removed to grant improved comparability between the biotic and environmental parameters.

2. Identification of stations for which environmental data was available without any gaps. The parameters were checked for a long gradient within the dataset. Ordinal data was converted to dummy variables. Conductivity values were log10-transformed.

Hydromorphological parameters

- Impoundment (yes/no)
- Riparian vegetation modified (no/slight/intermediate/high)
- Artificial embankment (no/slight/intermediate/high)
- Barrier upstream (yes/partial/no)
- Barrier downstream (yes/partial/no)
- Instream habitat modified (no/intermediate/high)
- Channel form modified (no/intermediate/straightened)
- Cross section modified (no/intermediate/technical profile)

Physicochemical parameters

- pH
- Electrical conductivity [µS/cm]
- Oxygen [mg/l]
- Nitrate [mg/l]
- Total Phosphate [µg/l]

Land use parameters for catchment and buffer (size: 100 x 1,000 m)

- Urban fabric [%]
- Arable land [%]
- Heterogeneous agricultural areas [%]
- Forests [%]

Data adjustment and metric calculation

The macroinvertebrate taxalist was checked for redundancy between species and genera. The taxonomic scale was lifted from species to genus level if the occurrence of individuals identified to genus level was high. Occurrences on the genus level were deleted if only few individuals were recorded, but many individuals were identified to species level. Rarefaction (Program: R!, package: vegan) was applied on the diatom taxalist with 100 valves per subsample.

Fish metrics were calculated with EFI+ (2009), macroinvertebrate metrics with Asterics 3.2 (Meier et al. 2006) and diatom metrics with OMNIDIA (Diatom Software, http://omnidia.free.fr/omnidia_review.htm).

A selection of metrics was used for the further analysis. Sensitivity and functional metrics, that covered habitat, feeding and reproduction preferences were chosen for fish and macroinvertebrates. For diatoms only sensitivity metrics were available.

Data analysis

Multivariate statistics was conducted with Canoco 4.5 (ter Braak and Smilauer, 2002). Stressor gradients in the datasets were detected via principal component analysis (PCA) for each parameter group (hydromorphology, physico-chemistry, land use catchment, land use along the river). The gradients were defined and named according to the dominant parameters. The corresponding stressor values were then assigned to each station. Revision of the gradient lengths of the metrics (standard deviations <3) with Detrended Correspondence Analysis (DCA) identified the redundancy analysis (RDA) as the adequate method to analyse the metric response to the stressor gradient impact. The altitude (lowland: <200m, mountain: 200-799m) and the catchment size were included as covariables into each analysis. Metric values were centred and standardized. Automatic forward selection was performed to rank the stressor variables. Forward selection provides information about the marginal (expressed by the value of Lambda1) and conditional effects (LambdaA) of each stressor variable. Lambda1 specifies the effect that an independent stressor variable adds to the explained variance, while LambdaA specifies the remaining effect a stressor variable adds to the model when other variables have already been loaded. The hierarchy was derived by ranking the values of Lambda1. Information about the inter-correlation of stressor variables were obtained by comparing Lambda1 to LambdaA. Monte Carlo permutation test was performed to test if the conditional effects are significant. Stressor gradients whose marginal effects were zero were removed from the dataset and the analysis was repeated.

It needs to be emphasized, that the presence of gradients does not mean that there is an actual impact at the sampling sites. Metrics already respond to values below critical thresholds and the aim was to derive the intensity of that response.

Results

Functional and sensitivity metrics

For each BQE eleven metrics were selected (Table 2). All metrics had a long gradient within the datasets (box plots not shown).

Table 2: Metrics selected for the analysis. F = Functional, S = Sensitivity

Metrics Fish		Metrics Macroinvertebrates		Metrics Diatoms	
Density intolerant to low O2	S	Average score per Taxon	S	TDI = Trophic Diatom Index (Kelly & Whitton 1995)	S
Density intolerant to acidification	S	BMWP Score	S	IPS = Specific Sensitivity Index	S
Density general water quality intolerant	S	EPT-Taxa [%]	S	GENRE = Indice diatomique generique	S
Density intolerant to habitat degradation	S	[%] Zonal preference littoral	F	DI-CH= Swiss diatom index	S
Density lithophilic reproduction	F	Life Index	F	IDP = Pampean diatom index	S
Density rheophilic flow conditions	F	[%] Zonal preference metarhithral	F	IDAP = Indice diatomique Artois Picardie	S
Density benthic feeding habitats	F	[%] Rheophilic flow conditions	F	EPI-D = Pollution index Dell'Uomo A	S
Density pelagic feeding habitats	F	Rhithron Feeding Type Index	F	Wat = Watanabe	S
Density insectivorous adults	F	[%] Microhabitat preference lithal	F	LOBO = Lobo et al.	S
Density rheopar spawning prefernces adults	F	[%] Comb. Feeding Types: Xyloph. + Shred. + ActFiltFee. + PasFiltFee	F	SLA = Sladecek's Index	S
Density fractional reproduction	F	[%] Microhabitat preference pelal	F	TID = Trophic Index (Rott et al. 1999)	S

Stressor gradients

Table 3 gives a summary of the results of the principal component analysis with the fish, macroinvertebrate and diatom datasets. Axes with Eigenvalues > 10 % were considered in the further analysis if the parameters that accounted for the variation were not yet identified as a dominant parameter in a superior axis. Considering the dominant stressor parameters that account for the variation, five local and seven large-scale stressor gradients were identified. (Table 4).

Table 3: Results of the PCA with environmental parameters. Listed are the dominant parameters with species scores for each axis and the eigenvalue (EIG [%])

Gradients	Fish		Invertebrates		Diatoms	
Hydromorphology	**Parameter (Species scores)**	**EIG [%]**	**Parameter (Species scores)**	**EIG [%]**	**Parameter (Species scores)**	**EIG [%]**
Ax1	Artificial embankment (-0.83), instream habitat mod. (-0.85), cross section mod. (-0.73), channel form mod. (-0.53)	30.5	Artificial embankment (-0.81), instream habitat mod. (-0.79), cross section mod. (-0.71), channel form mod.(-0.58)	28.5	Artificial embankment (-0.73), instream habitat mod. (-0.69), cross section mod.(-0.68), barrier down (-0.57), channel form mod. (-0.51)	28.6
Ax2	Barrier down (0.71), impoundment (0.64)	17.4	Barrier down (0.70), impoundment (0.64)	17.5	Impoundment (0.60), riparian veg. mod. (0.65)	17.2
Ax3	Barrier up (-0.80)	13.8	Riparian veg. (0.65), barrier up (-0.62)	14.3	Barrier up (-0.85)	13.4
Ax4	Riparian veg. (-0.90)	11.7	Riparian veg. (-0.74)	11.7	Barrier down (-0.52)	11.0
Physicochemistry						
Ax1	Nitrate (0.76), total phosphate (0.72), oxygen (-0.44)	28.3	Nitrate (0.79), total phosphate (0.73), oxygen (-0.58)	33.5	Nitrate (0.88), total phosphate (0.88), oxygen (-0.71)	42.3
Ax2	pH (-0.75), conductivity (-0.64), oxygen (-0.46)	25.5	pH (-0.77), conductivity (-0.80)	27.3	pH (-0.80), conductivity (-0.89)	32.1
Ax3	Conductivity (-0.67), oxygen (-0.42)	16.3	Oxygen (-0.53)	10.6		< 9
Ax4	pH (-0.47), oxygen (-0.47)	12.3	pH (0.4)	10.3		< 9
Land use Catchment						
Ax1	Forest (-0.89), arable land (0.69)	41.1	Forest (-0.91), arable land (0.67)	42.2	Forest (-0.96), arable land (0.74)	42.5
Ax2	Heterogenous agriculture (0.8)	24.1	Heterogenous agriculture (0.76)	22.2	Heterogenous agriculture (0.83)	26.2
Ax3	Urban fabric (0.66)	16.6	Urban fabric (0.74)	18.3	Urban fabric (0.79)	18.6
Ax4		< 5		< 5		< 3
Land use along the river (100m x 1000m)						
Ax1	Forest (0.92), urban fabric (-0.6)	34.9	Forest (0.91), urban fabric (-0.56)	34.4	Forest (0.81), heterogenous agriculture (-0.72)	33.6

Ax2	Urban fabric (0.77), heterogenous agriculture (0.56)	29.6	Heterogenous agriculture (0.85), urban fabric (-0.56)	29.8	Arable land (0.83)	31.7
Ax3	Arable land (-0.76), heterogenous agriculture (0.7)	27.6	Arable land (0.83), urban fabric (-0.59)	28.3	Urban fabric (0.89)	29.1
Ax4		< 4		< 4		< 3

Table 4: Summary of the derived stressor gradients with the corresponding PCA axes.

LOCAL	Fish	Invertebrates	Diatoms
Hydromorphology			
Structure	Ax1	Ax1	Ax1
Riparian vegetation modified	Ax4	Ax4	Ax2
Land use along the river (B = Buffer)			
Urban fabric B	Ax1	Ax1	Ax3
Heterogenous agriculture B / Urban fabric* B	Ax2*	Ax2	Ax1
Arable land B	Ax3	Ax3	Ax2
LARGE SCALE			
Hydromorphology			
Impoundment* / Barrier downstream	Ax2*	Ax2*	Ax4
Barrier upstream	Ax3	Ax3	Ax3
Physicochemistry			
Eutrophication	Ax1	Ax1	Ax1
Alkalization	Ax2	Ax2	Ax2
Land use Catchment (C)			
Arable land C	Ax1	Ax1	Ax1
Heterogenous agriculture C	Ax2	Ax2	Ax2
Urban fabric C	Ax3	Ax3	Ax3

Stressor hierarchy

The stressor hierarchy (decreasing impact on the biota) was derived by ranking the values of Lambda1 (marginal effect). The strongest impact on the metrics of all BQEs is exerted by a high percentage of arable land in the catchment (Table 5). Eutrophication is the second strongest stressor for fish and invertebrates. Diatoms are influenced stronger by alkalisation than eutrophication but the marginal effects show only a little difference. Alkalisation is also a stressor of high importance for invertebrates but it was detected only little impact on fish. The local stressor with the highest impact on fish and invertebrates is arable land along the river, while the strongest local stressor for diatoms is structural degradation. Fish and diatoms show a relatively strong response to barriers within the river system. Structural degradation is on sixth and eighth position in the diatom and fish rankings. Inter-correlation was detected for the variable eutrophication. Between 66% and 85% of its effect is explained by other variables. Rather independent variables are arable land in the catchment and alkalization. For the remaining variables the marginal and conditional effects are rather low, so no comparison was deducted.

Table 5: Hierarchy of the stressor impact on fish, invertebrates and diatoms. Lambda1 expresses the marginal effect (Effect of independent variable), LambdaA the conditional effect (additional effect of independent variable). The p-value refers to the significance of the conditional effect. Green = large scale impact, Orange = local impact.

Fish Variable	Lambda1 (marg.)	LambdaA (cond.)	p	F
Arable land Catch	0.07	0.07	0.002	22.13
Eutrophication	0.06	0.02	0.002	7.21
Urban fabric Catch	0.02	0.02	0.002	7.53
Arable land Buffer	0.02	0.01	0.006	3.8
Barrier downstream / Impoundment	0.02	0.02	0.002	6
Structure	0.02	0.02	0.002	5.36
Heterog, agriculture Catch	0.02	0.01	0.002	5.9
Urban fabric / Het. Agric. Buffer	0.01	0	0.098	1.8
Alkalization	0.01	0.02	0.002	5.98
Urban fabric Buffer	0			
Riparian Vegetation mod.	0			
Barrier upstream	0			

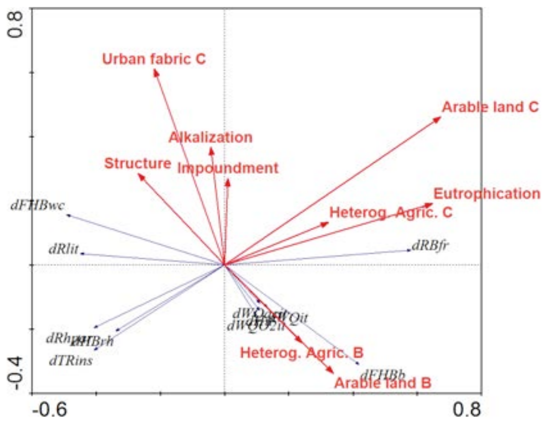
Invertebrates Variable	Lambda1 (marg.)	LambdaA (cond.)	p	F
Arable land Catch	0.09	0.09	0.002	24.06
Eutrophication	0.05	0.01	0.004	4.54
Arable land Buffer	0.04	0.01	0.022	2.63
Alkalization	0.03	0.03	0.002	9.5
Heterog. Agriculture Catch	0.02	0.01	0.066	2.05
Urban fabric Buffer	0.01	0.01	0.058	2.04
Heterog. agriculture Buffer	0.01	0.01	0.928	0.38
Structure	0.01	0.01	0.342	1.1
Urban fabric Catch	0.01	0.01	0.544	0.81
Riparian vegetation mod.	0.01	0	0.584	0.77
Barrier downstream / Impoundment	0			
Barrier upstream	0			

Diatoms Variable	Lambda1 (marg.)	LambdaA (cond.)	p	F
Arable land Catch	0.23	0.23	0.002	27.14
Alkalization	0.1	0.09	0.002	10.75
Eutrophication	0.06	0.01	0.28	1.19
Structure	0.02	0.01	0.15	1.64
Barrier upstream	0.01	0	0.866	0.41
Barrier downstream	0.01	0.01	0.366	1.11
Heterog. Agriculture Catch	0.01	0.02	0.01	3.02
Riparian vegetation mod.	0.01	0.01	0.37	1.04
Heterog. agriculture Buffer	0.01	0.01	0.672	0.68
Urban fabric Buffer	0.01	0	0.866	0.41
Urban fabric Catch	0.01	0.01	0.212	1.44
Arable land Buffer	0.01	0.01	0.632	0.76

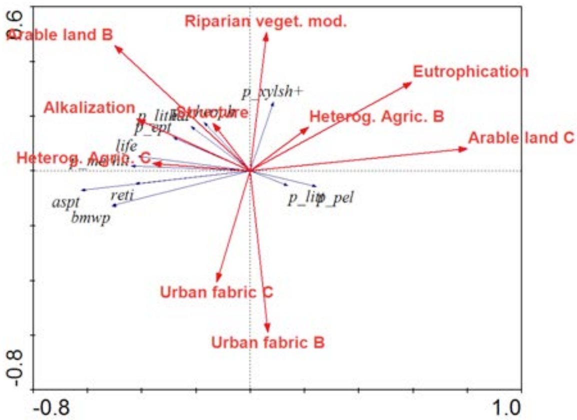
Metric response

The RDA biplots (Redundancy Analysis) illustrate the relationships between the selected metrics and to stressors gradients. A positive correlation is expressed by long vectors pointing into the same direction. Negative correlation is indicated by arrows pointing into opposite directions. Perpendicular arrows indicate that there is no correlation (Figure 1).

Invertebrates:



Fish:



Diatoms:

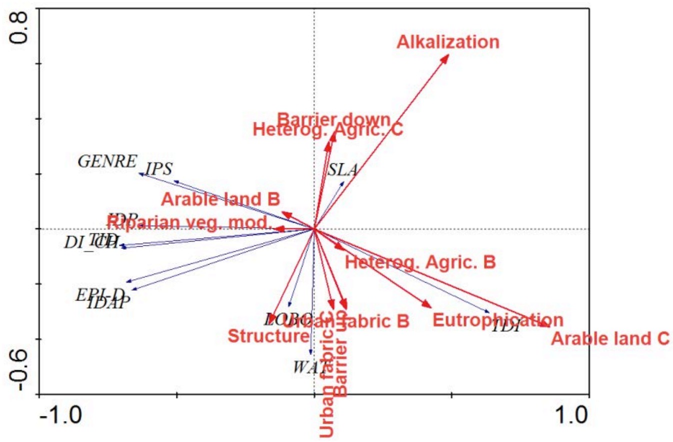


Figure 1: RDA biplots with selected metrics (black vectors) and to stressors gradients (red vectors).Metrics <u>Fish:</u> dWQO2it = density intolerant to low O2, dWQacit = density intolerant to acidification, dWQit = density general water quality intolerant, dHit = density intolerant to habiat degradation, dRlit = density lithophilic reproduction, dHBrh = density rheophilic flow conditions, dFHBb = density benthic feeding habitats, dFHBwc = density pelagic feeding habitats, dTRins = density insectivorous adults, dRHpar = density rheopar spawning preferences adults, dRBfr = density fractional reproduction; <u>Macroinvertebrates</u>: aspt = Average score per taxon, bmwp = BMWP score, p_ept = EPT-Taxa [%], p_litt = [%] Zonal preference littoral, life = LIFE Index, p_merhit = [%] Zonal preference metarhithral, p_rheoph = [%] Rheophilic taxa, p_lithal = [%] Microhabitat preference Lithal, p_xylsh+ = Combination of feeding types, p_pel =[%]Microhabitat preference pelal;<u>Diatoms</u>:see table 2 for abbreviations.

Fish

The sensitivity metrics respond weakly to the stressor variables (Table 6). The highest response was detected for "density intolerant to low oxygen" and "density intolerant to acidification" to urban fabric (-0.165 and -0.232). Almost all functional metrics respond to the strongest stressors arable land in the catchment and eutrophication. "Density benthic feeding habitats" and "Density pelagic feeding habitats" respond to a high number of stressors but some relations are paradox, for example, to Arable land along the river (benthic feeding positively, pelagic feeding negatively correlated).

Table 6: Correlations between fish metrics and stressor variables. dWQO2it = density intolerant to low O2, dWQacit = density intolerant to acidification, dWQit = density general water quality intolerant, dHit = density intolerant to habiat degradation, dRlit = density lithophilic reproduction, dHBrh = density rheophilic flow conditions, dFHBb = density benthic feeding habitats, dFHBwc = density pelagic feeding habitats, dTRins = density insectivorous adults, dRHpar = density rheopar spawning preferences adults, dRBfr = density fractional reproduction.

Stressor	Lambda1	dWQO2it	dWQacit	dWQit	dHit	dRlit	dHBrh	dFHBb	dFHBwc	dTRins	dRhpar	dRBfr
Arable land Catch	0.07					-0.3144	-0.3396		-0.2387	-0.3945	-0.3140	0.4385
Eutrophication	0.06					-0.2925	-0.2612	0.2070	-0.2794	-0.2885	-0.2969	0.4184
Urban fabric Catch	0.02	-0.1652	-0.2320			0.1806		-0.2504				
Arable land Buffer	0.02	0.1567				-0.1483		0.2322	-0.2484			
Barrier downstream / Impoundment	0.02		0.1781					-0.1998	0.1567	-0.1497		
Structure	0.02			-0.1701				-0.1842	0.1848			
Heterog, agriculture Catch	0.02								-0.1662	-0.1937	-0.1677	0.1854
Urban fabric / Het. Agric. Buffer	0.01		0.1928					0.1366				0.1952
Alkalization	0.01							-0.1678	0.1232	-0.1372		

Invertebrates

The BMWP Score and the related Average Score per Taxon respond strongly to intensive land use and water quality impacts (Table 7). Functional metrics that show high correlations are "[%] Zonal preference metarhithral", the LIFE index, "[%] Microhabitat preference pelal" and the Rhithron Feeding Type Index. Response to specific stressors can be observed for "[%] Microhabitat preference lithal" (Arable land along the river), the combination metric of different feeding types "Xyloph. + Shred. + ActFiltFee. + PasFiltFee" (Urban fabric along the river and Riparian vegetation modified) and "[%] Rheophilic flow conditions" (Urban fabric buffer and structure).

Table 7: Correlations between macroinvertebrate metrics and stressor variables. aspt = Average score per taxon, bmwp = BMWP score, p_ept = EPT-Taxa [%], p_litt = [%] Zonal preference littoral, life = LIFE Index, p_merhit = [%] Zonal preference metarhithral, p_rheoph = [%] Rheophilic taxa, p_lithal = [%] Microhabitat preference Lithal, p_xylsh+ = Combination of feeding types, p_pel =[%] Microhabitat preference pelal

Stressor	Lambda1	p_litt	p_merhit	p_lithal	p_pel	p_rheoph	aspt	bmwp	p_ept	life	reti	p_xylsh+
Arable land Catch	0.09		-0.3149		0.2720		-0.4582	-0.3746	-0.2773	-0.3525	-0.3968	
Eutrophication	0.05		-0.2106		0.2163		-0.4302	-0.3721		-0.2029	-0.2490	
Arable land Buffer	0.04		0.2443	0.2097	-0.2411		0.2416	0.2217		0.2016		
Alkalization	0.03	-0.1602	0.2525				0.3028	0.1913		0.2206		
Heterog. Agriculture Catch	0.02		0.2352				0.2195	0.2296		0.1469		
Urban fabric Buffer	0.01	0.1440				-0.1334						-0.1576
Heterog. agriculture Buffer	0.01						-0.1521	-0.1690			-0.1145	
Structure	0.01					0.1123	0.1309	0.1215				
Urban fabric Catch	0.01						0.1251	0.1336				
Riparian vegetation mod.	0.01		-0.1273									0.1230

<u>Diatoms:</u>

The correlations between various sensitivity indices are generally high (Kelly et al. 1995) therefore almost all diatom metrics show relatively high correlations with the stressors. The strongest response to arable land in the catchment connected to eutrophication can be observed for the TDI (Trophic Diatom Index), GENRE (Indice Diatomique Generique) and IPS (Specific Sensitivity Index), while IDAP (Indice diatomique Artois Picardie), EPI-D (Pollution index Dell'Uomo A), DI_CH (Swiss diatom index), IDP (Pampean diatom index) and TID (Trophic Index) respond also to alkalisation. LOBO and WAT show a specific response to alkalization and heterogenous agriculture in the catchment, but the reaction to alkalinity is less strong than the response of the later ones. EPI-D and TID respond to structural changes, while IDP and TDI react to barriers up- and downstream of the sampling site. The weakest response is observed for SLA (Sladecek's Index).

Table 8: Correlations between diatom metrics and stressor variables. See Table 2 for abbreviations.

Stressor	Lambda1	SLA	WAT	TDI	GENRE	IPS	IDAP	EPI-D	DI_CH	IDP	LOBO	TID
Arable land Catch	0.23			0.6904	-0.6129	-0.5146	-0.4686	-0.4943	-0.5672	-0.5612		-0.5835
Alkalization	0.1		-0.2935				-0.4642	-0.4686	-0.4244	-0.3309	-0.2579	-0.3453
Eutrophication	0.06			0.2764	-0.2967	-0.3940	-0.2203	-0.2395	-0.2998	-0.2338		-0.2965
Structure	0.02						0.1469	0.2279				0.2225
Barrier up	0.01		0.1288	0.1335		-0.1685				-0.1799		
Barrier downstream	0.01			-0.2753					-0.1646	-0.1147		
Heterog. Agriculture Catch	0.01		-0.2448								-0.2401	
Riparian vegetation mod.	0.01	-0.1688				0.1846	0.1290			0.1322		0.1484
Heterog. agriculture Buffer	0.01	-0.1559		0.1334	-0.1227						-0.1529	-0.1120
Urban fabric Buffer	0.01			0.2259		-0.1454				-0.1448	0.1202	
Urban fabric - Catch	0.01	-0.2239			-0.1056						0.1187	
Arable land Buffer	0.01					0.1524				0.1276		0.1153

Discussion

Stressor hierarchy

Our analysis reveals a hierarchy between stressors from different spatial scales. Stressors from a larger scale have a higher impact on aquatic communities than local stressors. This does not necessarily mean that local degradation (e.g. the instream habitat) is less severe for the biota than the occurrence of a large-scale disturbance (e.g. the implementation of a barrier). In fact it means that stressors impacting a river over long distances affect aquatic communities rather independently from the local conditions. Since the gradients of the specific stressors were analyzed, this overarching effect can be negative or positive depending on the large-scale conditions.

Arable land in the catchment has been identified as the dominant factor exerting the highest large-scale impact on fish, macroinvertebrates and diatoms. It has to be considered though, that the PCA gradient spans from high percentage of forest to a high share of arable land as a dominant factor. In addition urban areas or heterogeneous agriculture also contribute to the degraded areas. Nonetheless, all these forms of intensive land use imply various impacts on river systems: structural degradation and the alteration of the natural stream flow conditions; lack of riparian and instream habitats for stable aquatic populations; missing vegetation over wide areas leads to erosion of sediments and therefore to an increasing input of organic substrate and fertilizers into the river body (Blann et al. 2009). Agriculture along the river (100 m x 1,000

m) is the strongest local stressor for fish and macroinvertebrates. This scale was found to express the direct impact of the foresaid implications of catchment land use. We assume that the impact of land use along the river increases with the length of the regarded strip and that land use at the catchment scale can be considered an image of the conditions along the river (also indicated by Feld, this report). Wasson (2010) and Urban et al. (2006) pointed at the positive effect of forests in the riparian corridor. Thus, based on our results we recommend focusing the effort of river restoration on the creation of riparian habitats over longer distances.

Furthermore, our results indicate that water quality continues to be an issue that needs to be handled in order to implement the goals of the WFD. The metrics of all organism groups respond strongly to eutrophication and diatoms and macroinvertebrates also to acidification. Carpenter et al. (1998) stated that there is a high correlation of arable land in the catchment and eutrophication, which can also be observed in our results. Alkalization shows a low inter-correlation to other stressor variables. Though it is known that pH shifts towards alkaline conditions are correlated with high nutrient loads, the alkalisation gradient in our dataset is probably influenced by the Austrian geology, which consists of granite rocks in the region of the central Alps and the limestone and dolomite rocks in the calcareous Alps. Taking into account that restoration measures mainly focus on improving the structural conditions of a local river stretch, it is alarming that the local structure is ranked rather low in the detected hierarchies. The structural impact is still relatively high for diatoms due to the fact that diatoms are drifted in high population numbers within the water body. Furthermore they respond quickly to the surrounding conditions by reactivation of latent states and asexual reproduction. Fish, as active dispersers, can reach suitable habitats for reproduction or feeding if connectivity is given within the river system. The low response of macroinvertebrate metrics to structural conditions coincides with the observation that creation of new habitats on a river stretch by restoration often does not lead to significant improvements of the community structure. Dispersal strategies of macroinvertebrate species are diverse but are mainly based on passive drift and active or passive aerial distribution. Life cycles are complex and often include habitat changes. So, many species depend on long river stretches with intact instream and riparian habitats. In reality these river stretches are often interspersed by inhabitable stretches. Thus, the conditions on the catchment scale determine if life cycles can be completed and populations established. In terms of restoration efforts, this means that its success is depending on the species pool in the surroundings (Sundermann et al. 2011)

Metric response

The intensity of the metric response varies between the BQEs with diatoms showing the highest intensity. Furthermore, the response of the single metrics varies between different stressor variables. Due to the high inter-correlation between the stressor variables, also the metrics respond to more than one stressor.

The fish metric that indicates best a high amount of intensive land use in the catchment is “Density fractional reproduction”. The reproduction period is adapted to short but frequent disturbances and can be varied in length depending on temperature, food availability and other parameters. Rather pristine (often mountainous catchments) are best represented by “Density rheophilic flow conditions”, “Density rheopar spawning preferences adults” and “Density lithophilic reproduction”. The response of fish metrics to land use parameters is always correlated to eutrophication. The sensitivity metrics that were chosen (Intolerance to low oxygen, acidification, habitat degradation and general water quality) responded weakly. A possible reason for this is that certain sensitive species like *Salmo trutta f. fario* or *Salvelinus fontinalis* are favoured targets to fish stocking in river stretches all over Austria (Spindler 1997). The occurrence of the species thus may not reflect the habitat conditions that are naturally preferred. Macroinvertebrate metrics with the highest response intensity are the sensitivity metrics BMWP and ASPT. They respond to many different stressor types. This indicates again the high correlation between structural degradations and the nutrient load. Other metrics that indicate natural catchments are: “[%] Zonal preference metarhithral”, “EPT-Taxa [%]”, “Rhithron Feeding Type Index” and the LIFE Index, while “[%] Microhabitat preference pelal” indicates the opposite. The metrics that respond exclusively to local stressors are “Rheophilic flow conditions” (Urban fabric buffer, Structure) and “[%] Microhabitat preference lithal” (Arable land buffer). The sensitivity metrics that were developed to capture trophic or pollution impacts respond well to various stressor types with the same indication as the macroinvertebrate metrics BMWP and ASPT. The metrics that respond most are GENRE, IPS and TID. It is obvious that aquatic communities are altered by multiple stressors and that land use is a broad proxy parameter representing manifold impact types. Thus we could not detect a dominant impact on aquatic organism groups.

The results reveal that the larger scale needs to be taken in account when implementing successful restoration measures. The catchment should be analysed in detail during restoration planning to detect and possibly decrease large scale impacts and to choose restoration sites wisely.

Conclusions

We conclude that:

- there is a hierarchy between stressors from different spatial scales in their impact on aquatic communities.
- agriculture in the catchment and eutrophication exert the strongest large scale impact.
- agriculture along the river and structural degradation (for fish and diatoms) are the strongest local factors.
- the specific impact of land use from the catchment scale is manifold and aquatic communities respond to multiple stressors.

Therefore river restoration needs to take into account the following:

- During restoration planning the impact sources from the catchment need to be detected.
- If rivers are impacted by eutrophication, sources of pollution need to be reduced.
- Riparian buffers should be accomplished over longer distances to hold back the input from the catchment and to create suitable habitats.
- Restoration should start at sites where the catchment still houses populations for colonization and spread then into more degraded areas.

The response of macrophytes, macroinvertebrates and fish assemblages to land use in lotic environments varies across ecoregions and spatial scales

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Abstract

The role of catchment land use and its manifold impacts on the lotic environment down to the local habitat scale has been frequently investigated by research and monitoring studies, yet the results do not provide definite guidance as to whether land use impacts lotic assemblages at the broad (catchment) or rather at the fine scale (stretch/reach). Here, I investigated the response of assemblage metrics of macrophyte, benthic invertebrate and fish communities to agriculture and forest cover at different spatial scales. The data comprised biological samples from 479 stations in four ecoregions in France and Germany and land use/cover generated at the catchment scale and at the scale of 16 buffer sizes ranging 0.1–7.2 km^2 (four lengths upstream of by four widths adjacent to each station). Correlation analysis and multivariate regression were used to identify response patterns and correlation strengths of metrics and land use. In general, land use was correlated much stronger with fish and macroinvertebrate metrics (mean maxima across all spatial scales: Spearman $r = 0.615$ and $r = 0.512$, respectively) than with macrophyte metrics ($r = 0.340$). Except for two out of 30 metrics tested, all correlations were higher for mountain data than for lowland data. Graphical analysis revealed different trends (e.g. unimodal, sigmoidal, linear) of correlations along the buffer size/catchment gradients, however, the majority of metrics tested suggests an important role of land use patterns in the near-stream (100 m wide) buffer in mountain ecoregions, with increasing correlations at increasing buffer lengths. This was supported by multivariate regression, but neither correlation analyses nor regressions revealed comparatively clear and strong trends for the lowland data. Along a gradient of percent land use as agriculture, many metrics significantly changed their values at 0–20% agriculture in mountain ecoregions and 30–50% in lowland ecoregions, irrespective of the buffer size.

Synthesis and applications: The results presented here suggest a better (mountains) of equally good (lowlands) responsiveness of lotic fish and invertebrate assemblages to near-stream land use patterns, which in turn confirms the near-stream buffer strip to be a suitable spatial scale of land use management and restoration.

Introduction

The Millennium Ecosystem Assessment (MA 2005) has identified the continuing increase of agricultural land use on earth to be a major driver of ecosystem change, in particular the conversion of land into row crops during the past 35 years that was unprecedented in human history before. Thereby, agricultural land use can impact the landscape and ecological conditions in entire watersheds through a wide array of stressors connected to it. Allan (2004) reviewed the manifold stressors induced by different forms of land use (e.g. agriculture, urbanisation) and pointed at the paramount role of spatial scaling in this context. Accordingly, agricultural land use degrades entire river ecosystems by an increase of non-point inputs of pollutants, pesticides, fine sediments, nutrient levels and by altering hydrology. At the finer riparian and reach scales, agricultural land use reduces riparian and stream channel habitat, while the loss of riparian vegetation (shading) enhances solar radiation and thus higher levels of algal (Quinn 2000) and macrophyte biomass. This has (often adverse) implications on the aquatic food web (e.g. increase of macroinvertebrate grazers, decrease of macroinvertebrate shredders) (Delong & Brusven 1998) and its component organisms (e.g. loss of cold-stenothermic macroinvertebrate taxa).

Among the multiple stressors induced by agriculture, it is in particular the excessive release of sediments and nutrients from crop fields that are frequently correlating with agricultural area in the catchment above a test site (see Allan 2004 for a review). Jones et al. (2001) found catchment agricultural land use and riparian forest to explain the majority of variance in nutrient and sediment yields at 78 catchments in the mid-Atlantic Highlands, U.S.A. Thereby, a greater in-stream deposition of sediments in agricultural catchments can significantly reduce biological integrity, for instance, through the loss of fish and benthic invertebrate richness, in particular of those species associated with coarse substrate (Wood & Armitage 1997; Walser & Bart 1999). Following the review study of Allan (2004), there is sufficient evidence in the literature that the mere area of catchment agricultural land use is a good predictor (proxy) of the multiple stressor effects of land use on in-stream habitats and the organisms that inhabit them.

For roughly a decade already and primarily in the course of the implementation of the European Union Water Framework Directive (2000/60/EC), an increasing number of European studies built on this knowledge and analysed catchment land use data to seek for biological indicators, suitable to assess the ecological status of in-stream communities along gradients of multiple-stressor impact. For example, Hering et al. (2006) and Johnson & Hering (2009) compared the response of different organism groups (fish, benthic invertebrates, macrophytes and benthic diatoms) to catchment land use based on data from lowland and mountain ecoregions in seven European countries. The comparison of community-level metrics in these studies revealed diatoms and invertebrates to be more responsive than fish and macrophytes. Another group of studies (e.g. Dolédec et al. 2006) investigated the usability of benthic invertebrate species traits and tested their response to varying intensities and forms of catchment land use. These studies often found species traits to respond predictably to *a priori*-hypothesised land use effects, for instance an increase of % sediment feeders/gathering collectors following increased amounts of catchment agricultural land use (Dolédec et al. 2006; Vandewalle et al. 2010). Dolédec et al. (2006) concluded that functional (trait-based) measures successfully complement traditional structural community measures (e.g. total and sensitive species richness). In summary, all these studies imply a general impact of broad-scale land use of in-stream assemblages (fauna and flora) and their attributes (compositional, structural, functional measures).

A common attribute of many studies mentioned above is their focus on the catchment scale. Although there is also much evidence that riparian land use strongly influences in-stream habitat and biota (e.g. Delong & Brusven 1998; Lammert & Allan 1999; Quinn 2000; Stauffer et al. 2000; Jones et al. 2001), such finer scales remain often unconsidered in many indicator studies. Consequently, one cannot answer from these studies as to whether the analysed communities and their community attributes are actually most indicative to catchment-scale land use. The answer, in part, might be derived from Allan's (2004) review study, but the number of considered studies therein that directly compare scales and indicators is low. Moreover, the transferability of results to the European continent, primarily gained in regions outside Europe, is doubtful.

In recent years, an increasing number of ecological river restoration studies referred to catchment land use (see Feld et al. 2011 for a review). Apparently, there is an overriding adverse effects on the ecological success of physical restoration at the finer riparian and reach scales. In brief, it is often assumed (but rarely statistically proven) that intensive catchment land use is likely to "spoil" restoration success at the finer (reach) scale, i.e. the actual scale of the majority of measures, be it biological recovery (e.g. Entrekin et al. 2008) or mere long-term sustainable improvements of physical habitat conditions (e.g. Pretty et al. 2003; Levell & Chang 2008). The obvious conclusion is that physical restoration measures in agricultural catchments cannot be successful without fixing in parallel the (superior) adverse land use effects in the catchment above a restoration. Of paramount importance in this context is again the question as to whether the catchment scale is the most influential one. Or more explicitly: What is the biological effect scale of land use? The answer to this question is not only relevant for the appropriate scaling of indicators to assess the lotic environment, but furthermore can help identify the appropriate scaling of land use management. Doubtlessly, land use management in agricultural landscapes is a likely prerequisite for ecologically successful restoration, yet management would become somehow more realistic (i.e less prone to conflicting uses and interests) if applied to the riparian or reach scale instead of the entire catchment. Catchment-scale 'natural' conditions might not be the appropriate goal in the 21st century (Bishop et al. 2009).

In this study, I use percent arable land use and percent forest cover, generated at three different spatial scales (near-stream buffer, different riparian buffer areas, entire catchment), as proxy measures to investigate possible adverse, but also favourable effects on selected community attributes (metrics) of three organism groups: fish, benthic invertebrates and aquatic macrophytes. I systematically compare these effects among assemblages and spatial scales and test the metrics' response to land use intensity and scaling. The data also allows a comparison of catchment and near-stream land use to test whether catchment land use is a reliable proxy for near-stream land use conditions. The outcome of my study not only helps identify the ecologically relevant land use scaling for lotic ecosystem indication, but also suggests the appropriate scaling of land use management aiming at ecological recovery of in-stream communities.

Materials and methods

Study reach

This study is based on a subset of the WISER river database that contains biological and abiotic monitoring data from 5 European countries spanning 8 major ecoregions. The subset comprises 500 sampling stations in France and Germany (Figure 1) and spans over four ecoregions (Western and Central Mountains: 422 ± 277 m a.s.l.; Western and Central Lowlands: 110 ± 111 m a.s.l., i.e. ecoregion No. 8, 9, 13 and 14, respectively, according to Illies 1978). For each station, it contains biological (fish, benthic invertebrates and aquatic macrophytes; hereafter referred to as 'Biological Quality Elements' [BQEs]) and environmental data (land use at the scale of 16 different buffer sizes and the entire catchment above a station). Thereby, a station is defined as a spatially, but not temporally homogeneous river stretch of up to several hundreds of metres in length that is attributable to samples of all BQEs, whereas the sampling occasions of individual BQEs may differ by more than one year. The temporal heterogeneity is, inter alia, due to the deviation of optimum sampling seasons for the different BQEs: while aquatic macrophytes are usually sampled in late summer and early autumn, the preferred period for benthic invertebrates is spring (small to medium streams) or early summer (medium to large streams and rivers).

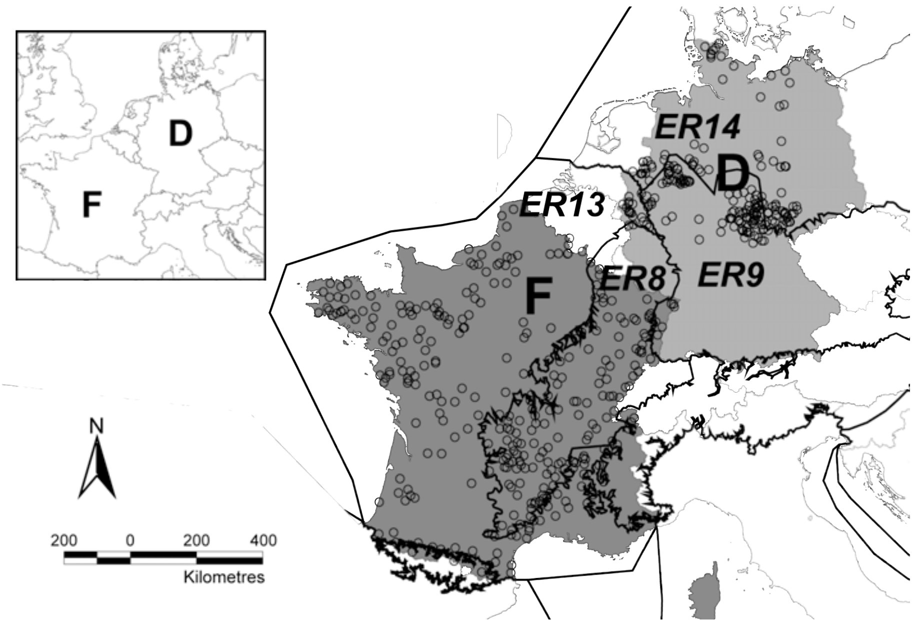

Figure 1: Location of 500 sampling stations in France (F) and Germany (D), divided into four ecoregions (ER): 8 and 9 = Western and Central Mountains, 13 and 14 = Western and Central Lowlands (Illies 1978). Bold lines: ecoregion borders; fine lines = country borders.

Field sampling

Macrophyte surveys were in line with the European Standard EN 14184 and followed the protocols of AFNOR (2003) in France and Schaumburg (2005a,b) in Germany. At each station, the species' coverages were estimated along a 100 m stretch, either while wading across the stretch or using a boat and a rake at non-wadeable stations and converted to the semi-quantitative Kohler scale. Benthic invertebrate samples were taken using a multi-habitat sampling technique (Hering et al. 2004) and a hand-net of ca. 25 x 25 cm (mesh: 500 µm). The samples were either sorted in the field or in the lab and subsequently determined to the lowest feasible level in Germany (Meier et al. 2006), while the genus level was aimed at in France (AFNOR 2010). Finally, fish assemblages were sampled using an electroshocking device either while wading across a section or using a boat. A stretch of one to several hundreds of metres in length was sampled at each station (1st run), while sampling effort varied according to the abundance observed (Oberdorff 2001). All fish were counted and measured for length alive and released afterwards.

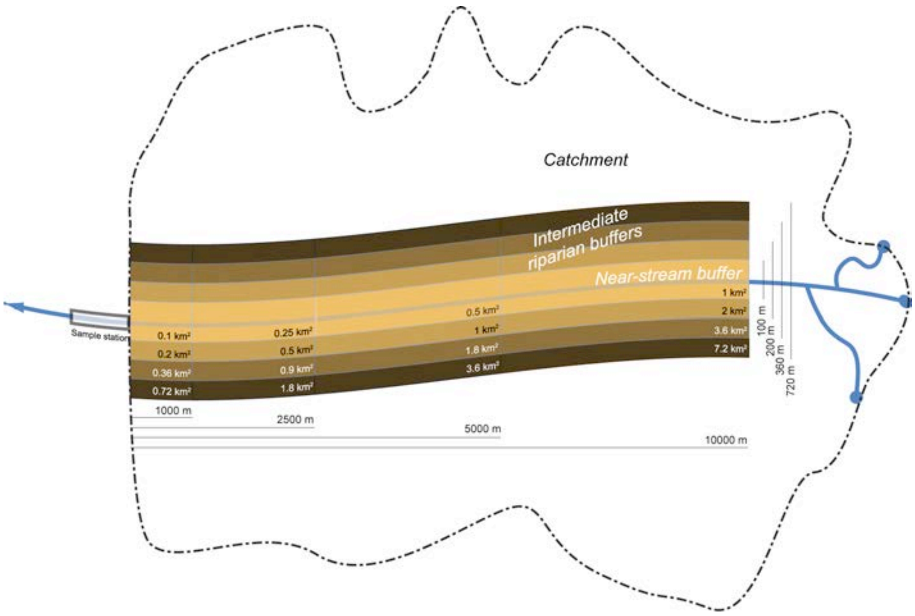

Figure 2: Schematic overview of 16 buffer areas (4 widths by 4 lengths) used to generate land use data. For clarification, four buffer areas are exemplarily delineated by a dotted line and indicated by larger fonts. Buffer generation included the main river course above a sampling station and its tributaries.

Land use and ecoregional descriptor variables

Buffer and catchment land use data were generated for 500 individual stations using ArcGIS 9.3. A script delineated buffer polygons along the river network (i.e. the main river course upstream of each station and its tributaries), so that the river course (line) divided all buffers in equal areas on the left and right hand side (Figure 2). The combination of four buffer lengths (1.00, 2.50, 5.00 and 10.00 km) and four buffer widths (0.10, 0.20, 0.36 and 0.72 km) resulted in 16 different buffers ranging from 0.1–7.2 km^2 area upstream each station. Catchment (watershed) delineation was generated using digital elevation models to automatically identify the (margins of the) entire river network upstream of the upper end of each station. Both buffer and catchment delineations were spot-checked for correctness and then combined with CORINE land cover data (http://www.corine.dfd.dlr.de/ papers_de.html). CORINE data is based on satellite imagery (Landsat 7) and is available as vector or raster data for all over Europe at comparable resolutions. While the raster data is generalised and available at a minimum resolution of 100 x 100 m, the vector maps are based on data at a resolution of 25 x 25 m or better; vector data was used in this study.

For each buffer area, the relative cover (percentage) of 13 land use/cover types (CORINE level 2; hereafter referred to as land use and land use types, respectively) was calculated and visually inspected for variation using box plots. Finally, five land use types showed sufficient variation and were selected for land use gradient analysis: urban, arable (row crop), heterogeneous agriculture, pasture and forest.

Catchment areas ranged 8–9,350 km^2 (median: 111 km^2, $10^{th}/90^{th}$ percentile: 16/1,070 km^2) across all ecoregions and were comparable between ecoregions (Figure 3). Stations <8 km^2 catchment area were omitted to allow for a proper calculation of land use at all buffer scales. In order to account for ecoregional and geographical (i.e. natural) variability in the data, country (France, Germany), ecoregion (8, 9, 13 and 14), longitude/latitude (decimal degrees) and altitude (m a.s.l.) were added for each station and used in direct gradient analysis (see below) to account for co-variation of natural descriptors.

Principal Components Analysis (PCA) was applied to the matrix of percent cover values (arc sin square root-transformed data) of five land use types and 500 stations and revealed arable and forest land use to describe the first principal component (gradient) along axis 1 (results not shown here). This gradient described already 57% of the total variation in the land use data and reflected both an 'impact type' (arable) and a 'buffer type' land use (forest). Hence, both principal land use types fit the aims of this study and were selected for further analysis.

Metric calculation and selection

For all BQEs, taxonomic data was available at the species (fish, aquatic macrophytes) or genus level (benthic invertebrates). After taxonomic harmonisation (to reduce inconsistency and redundancy) and the exclusion of species-poor stations (<6 taxa), biological and ecological metrics were calculated using the EFI+ tool for fish (EFI+ Consortium 2009), ASTERICS 3.1.1 for benthic invertebrates (Meier et al. 2006) and an intercalibration metrics protocol for aquatic macrophytes (S. Birk, unpublished data). Strongly interrelated metrics were indentified using Spearman rank correlation and excluded, while the number of macroinvertebrate metrics was limited *per se* due to the use of genus-level taxa lists, which rendered the calculation of the majority of metrics impracticable (and not reliable, as they require species-level data). The final metric lists comprised: 31 fish metrics at 478 stations, 35 invertebrate metrics at 491 stations (metric calculation limited due to genus-level taxonomy) and 13 macrophyte metrics at 498 stations.

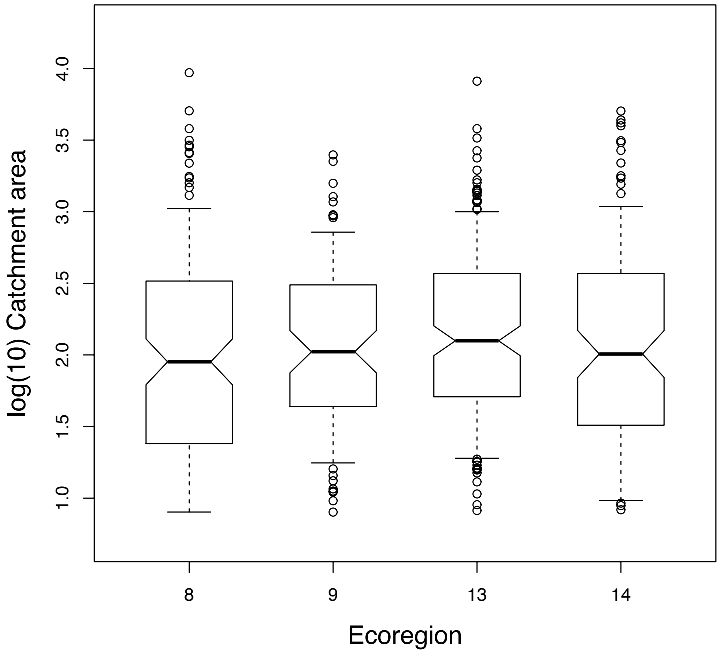

Figure 3: Distribution of catchment areas among the four ecoregions.

Data analysis

A pre-selection of metrics generally responsive to land use was made using partial Redundancy Analysis (pRDA). pRDA was run separately for each combination of three BQEs (log-transformed taxa counts, arc sin square root-transformed percent values), two land use types (arc sin square root-transformed percent values) and four ecoregions, resulting in 24 runs. As a direct form of multivariate gradient analysis, pRDA is seeking for concordance of biological and environmental gradients, the latter of which are linear combinations of environmental variables (ter Braak & Smilauer 2002). 'Partial' here refers to the extraction of 'natural' co-variance using country, latitude, longitude and altitude as co-variables in all pRDA runs. By visual inspection of pRDA biplots, metrics that ordered along the major land use gradient were selected and subjected to correlation analysis. Spearman rank correlation was used to identify the strength of relationships between individual metrics and percent land use and the trend of correlations along a gradient of buffer areas. Due to the important role of ecoregional descriptors, all correlation analysis was separated by ecoregion and land use type in order to reduce the potentially confounding effects of these natural variables.

Finally, Boosted Regression Trees (BRTs) were applied to identify metric change points (or ranges thereof) along land use gradient. BRTs constitute a relatively novel statistical approach successfully applied in species distribution modelling (Leathwick et al. 2006; De'ath 2007; Elith et al. 2008). As opposed to conventional regression modelling (e.g. linear regression, GLM, GAM), BRT raise only modest preconditions on data quality and distribution; in particular they can handle continuous and categorical data in parallel, cope with missing values, do not require normally-distributed variables and allow of the use of correlated predictor variables (Elith et al. 2008). The latter criterion renders BRT particularly useful for the analysis of inter-related land uses at different spatial scales, as in this study. Eight BRTs were run separately for each combination of metric, ecoregion and land use type and were applied to a total of seven metrics (2 macrophyte, 2 fish and 3 invertebrate metrics). This resulted in 56 BRTs to indentify and compare metric change points subject to BQE, ecoregion and land use type.

The procedure of boosted regression in general followed the manual provided by Elith et al. 2008 (supplementary material S3) and Elith & Leathwick (2011), using the standard setting for model training and cross-validation (tree complexity: 5, learning rate: 0.01; bag fraction: 0.5). The variance explained by individual spatial scales (land use buffer areas) was determined using the BRTs summary statistics, while the analysis of metric change points was based on the BRT function's graphical output. All correlation and BRT analyses were run in R (R Development Core Team 2011) using R's standard libraries and in addition for BRT, using the libraries 'gbm' (v.0.7-2, Ridgeway 2010) and 'dismo' (v.0.7-2, Hijmans et al. 2011). PCA and pRDA were run using CANOCO 4.5 (ter Braak & Smilauer 2003).

Results

Scaling and size effects of land use

The comparison of land use at different spatial scales showed a notably consistent pattern: in all ecoregions and for both land use types, near-stream arable and forest land cover were highly correlated with the respective catchment-scale values, but to a lesser degree with the values generated at intermediate buffer scales (Table 1). This catchment-buffer scale relationship may be trivial in small catchments, i.e. when catchment area resembles buffer sizes. Indeed, smaller catchments (<50 km^2) constitute approximately a third of the stations considered here. However, the observed pattern held true for arable land use also after exclusion of 159 sampling stations <50 km^2 catchment size; the correlations even increased for three out of four ecoregions (range: r = 0.699–0.847, Table 2). Contrastingly, the respective correlation of forest cover decreased for all ecoregions after exclusion of small catchments (range: r = 0.347–0.666), i.e. a size effect was clearly obvious for forest, but not for arable land use.

Table 1: Spearman's r (mean r ± SD) for the correlation between near-stream (100 m strip) and intermediate scale buffer land uses with catchment land use.

Ecoregion	arable		forest	
	100 m strip – catchment	200–720 m strip – catchment	100 m strip – catchment	200–720 m strip – catchment
8	0.780 (0.045)	0.737 (0.087)	0.680 (0.061)	0.629 (0.076)
9	0.903 (0.010)	0.751 (0.084)	0.715 (0.026)	0.541 (0.043)
13	0.683 (0.095)	0.548 (0.115)	0.730 (0.065)	0.626 (0.089)
14	0.768 (0.031)	0.605 (0.073)	0.638 (0.024)	0.446 (0.079)

Table 2: Spearman's r (mean r ± SD) for the correlation between near-stream (100 m strip) and catchment land use in all ecoregions (ER), divided by catchment sizes <50 and >50 km^2.

	<50 km^2		>50 km^2	
	arable	forest	arable	forest
ER 8	0.688 (0.100)	0.777 (0.103)	0.816 (0.025)	0.384 (0.074)
ER 9	0.964 (0.003)	0.764 (0.034)	0.847 (0.011)	0.666 (0.032)
ER 13	0.914 (0.021)	0.897 (0.046)	0.699 (0.069)	0.407 (0.080)
ER 14	0.801 (0.036)	0.858 (0.038)	0.699 (0.019)	0.347 (0.022)

Response of BQEs along the buffer size gradient

The comparison of correlations of individual metrics with land use along the size gradient (0.1–7.2 km^2 and catchment area) revealed notable differences that are exemplarily illustrated in Figure 4. In general, four different trends were evident: i) no trend, i.e. similar strength of correlations at all scales; ii) unimodal trend, i.e. a strong relationship at intermediate buffer sizes and less strong the relationship at the fine and broad scales, but; iii) sigmoid trend, i.e. metrics correlate highest either at the fine (buffer) or broad (catchment) scale; iv) linear and monotonic trend, i.e. the correlation strength increased/decreased with buffer/catchment area. The latter linear trend can also cross the x-axis and change the sign as illustrated in Figure 4d for the proportion of xylophilic (wood-preferring/-dwelling) invertebrates. Such trends from positive to negative correlations require a thorough analysis of land use at different spatial scales.

When averaged across ecoregion types and land use types, all assemblages and the majority of metrics revealed the 'unimodal' trend (Figure 4b) to dominate in mountain systems: the strength of the relationship increased with buffer length in the near-stream (100 m) buffer, then dropped down slightly, increased again at intermediate buffer sizes, before it eventually decreased at the catchment scale (Figure 5). Although not further tested here, this finding also implies that the metrics are more responsive to increasing buffer length, in particular in the near-stream buffer strip. No clear dominating trend, however, was evident for lowland systems, where all metrics also revealed notably weaker correlations with land use, regardless of the spatial scale of land use.

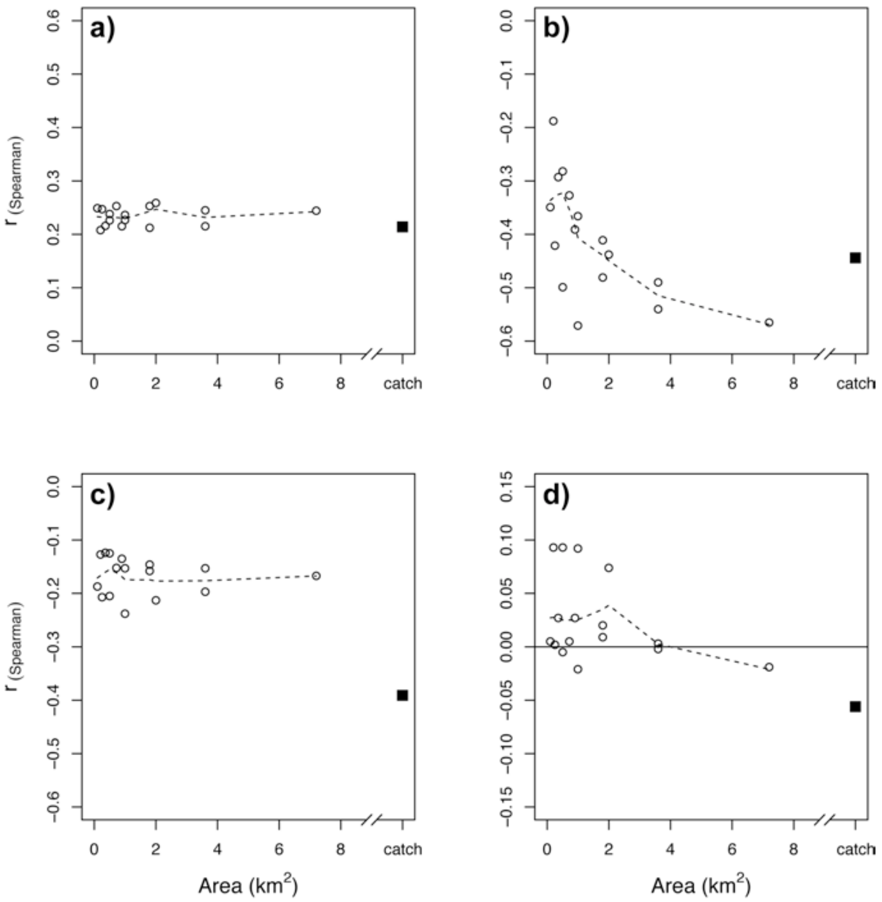

Figure 4: Correlation trends of four metrics along a gradient of buffer (open circles) and catchment area (catch, filled squares). a) # EPT invertebrate taxa and forest in ER 13; b) % water-quality intolerant fish in ER 8; c) # aquatic macrophyte taxa in ER 13; d) % xylophilic invertebrates and forest in ER 13. Buffer values are fitted using a lowess smoother (dashed line).

Table 3: Maximum Spearman's r (absolute values) for the correlation of ten fish metrics with arable and forest land use in four ecoregions (ER), representing two ecoregion types: mountain (ER 8 + 9) and lowland (ER 13 + 14).

$r_{Spearman}$	ER 8		ER 9		ER 13		ER 14	
metric	arable	forest	arable	forest	arable	forest	arable	forest
sWQO2it	0.571	0.625	0.638	0.669	0.189	0.409	0.333	0.238
sHit	0.512	0.592	0.686	0.741	0.222	0.415	0.259	0.229
sCL53it	0.528	0.569	0.676	0.740	0.230	0.416	0.272	0.218
sRhpar	0.449	0.584	0.650	0.698	0.241	0.431	0.311	0.196
sWQit	0.462	0.576	0.676	0.763	0.214	0.399	0.193	0.215
sWQtol	0.527	0.564	0.655	0.646	0.325	0.374	0.237	0.151
sHtol	0.486	0.588	0.721	0.718	0.258	0.305	0.110	0.265
sHBrh	0.552	0.573	0.617	0.642	0.224	0.364	0.270	0.185
sCL53bi1	0.474	0.603	0.644	0.736	0.186	0.402	0.176	0.180
sWQtxit	0.489	0.591	0.633	0.716	0.186	0.403	0.163	0.196

$r_{Spearman}$	total		mountain		lowland	
metric	mean	SD	mean	SD	mean	SD
sWQO2it	0.459	0.191	0.626	0.041	0.292	0.098
sHit	0.457	0.208	0.633	0.101	0.281	0.091

sCL53it	0.456	0.204	0.628	0.097	0.284	0.091
sRhpar	0.445	0.188	0.595	0.108	0.295	0.102
sWQit	0.437	0.221	0.619	0.130	0.255	0.096
sWQtol	0.435	0.190	0.598	0.063	0.272	0.098
sHtol	0.431	0.230	0.628	0.113	0.235	0.086
sHBrh	0.428	0.188	0.596	0.041	0.261	0.077
sCL53bi1	0.425	0.226	0.614	0.109	0.236	0.111
sWQtxit	0.422	0.220	0.607	0.094	0.237	0.112

Table 4: Maximum Spearman's r (absolute values) for the correlation of ten benthic macroinvertebrate metrics with arable and forest land use in four ecoregions (ER), representing two ecoregion types: mountain (ER 8 + 9) and lowland (ER 13 + 14).

$r_{Spearman}$	ER 8		ER 9		ER 13		ER 14	
metric	arable	forest	arable	forest	arable	forest	arable	forest
ASPT	0.601	0.540	0.747	0.660	0.494	0.393	0.320	0.341
No_EPT	0.540	0.436	0.513	0.652	0.339	0.259	0.258	0.341
Life	0.530	0.546	0.575	0.522	0.258	0.296	0.302	0.227
r_K	0.488	0.558	0.616	0.632	0.408	0.207	0.152	0.114
p_EPT	0.417	0.355	0.400	0.416	0.347	0.229	0.191	0.384
p_Lithal	0.385	0.299	0.467	0.477	0.208	0.116	0.310	0.382
p_Psam	0.218	0.095	0.290	0.266	0.366	0.185	0.429	0.412
p_actFF	0.414	0.284	0.365	0.336	0.152	0.154	0.221	0.155
R_Dom	0.375	0.298	0.465	0.530	0.071	0.048	0.078	0.065
Margalef	0.219	0.057	0.447	0.541	0.285	0.124	0.145	0.086

$r_{Spearman}$	total		mountain		lowland	
metric	mean	SD	mean	SD	mean	SD
ASPT	0.512	0.154	0.637	0.088	0.387	0.078
No_EPT	0.417	0.143	0.535	0.089	0.299	0.047
Life	0.407	0.148	0.543	0.023	0.271	0.035
r_K	0.397	0.212	0.574	0.065	0.220	0.131
p_EPT	0.342	0.086	0.397	0.029	0.288	0.092
p_Lithal	0.331	0.124	0.407	0.083	0.254	0.116
p_Psam	0.283	0.116	0.217	0.087	0.348	0.112
p_actFF	0.260	0.105	0.350	0.054	0.171	0.034
R_Dom	0.241	0.199	0.417	0.102	0.066	0.013
Margalef	0.238	0.175	0.316	0.219	0.160	0.087

Table 5: Maximum Spearman's r (absolute values) for the correlation of ten aquatic macrophyte metrics with arable and forest land use in four ecoregions (ER), representing two ecoregion types: mountain (ER 8 + 9) and lowland (ER 13 + 14).

$r_{Spearman}$	ER8		ER9		ER13		ER14	
metric	arable	forest	arable	forest	arable	forest	arable	forest
NMOSS	0.348	0.320	0.542	0.580	0.283	0.191	0.234	0.218
NMACRx	0.331	0.333	0.102	0.282	0.230	0.436	0.144	0.151
NTAXA	0.220	0.303	0.201	0.397	0.222	0.401	0.086	0.101
SWTAXA	0.199	0.264	0.203	0.451	0.335	0.263	0.100	0.096

ABDMOSS	0.171	0.259	0.265	0.313	0.182	0.212	0.159	0.191
NTRUE	0.152	0.300	0.148	0.337	0.274	0.258	0.089	0.161
SWTRUE	0.171	0.243	0.150	0.376	0.362	0.163	0.082	0.096
EVTAXAx	0.105	0.144	0.210	0.364	0.271	0.085	0.095	0.125
EVTAXA	0.116	0.124	0.185	0.348	0.307	0.087	0.098	0.118
EVTRUE	0.148	0.150	0.024	0.244	0.385	0.086	0.056	0.121

$r_{Spearman}$	total		mountain		lowland	
metric	mean	SD	mean	SD	mean	SD
NMOSS	0.340	0.147	0.448	0.132	0.232	0.039
NMACRx	0.251	0.115	0.262	0.109	0.240	0.136
NTAXA	0.241	0.119	0.280	0.090	0.203	0.146
SWTAXA	0.239	0.118	0.279	0.118	0.199	0.120
ABDMOSS	0.219	0.054	0.252	0.059	0.186	0.022
NTRUE	0.215	0.088	0.234	0.098	0.196	0.087
SWTRUE	0.205	0.112	0.235	0.102	0.176	0.129
EVTAXAx	0.175	0.099	0.206	0.114	0.144	0.086
EVTAXA	0.173	0.100	0.193	0.108	0.153	0.104
EVTRUE	0.152	0.116	0.142	0.090	0.162	0.151

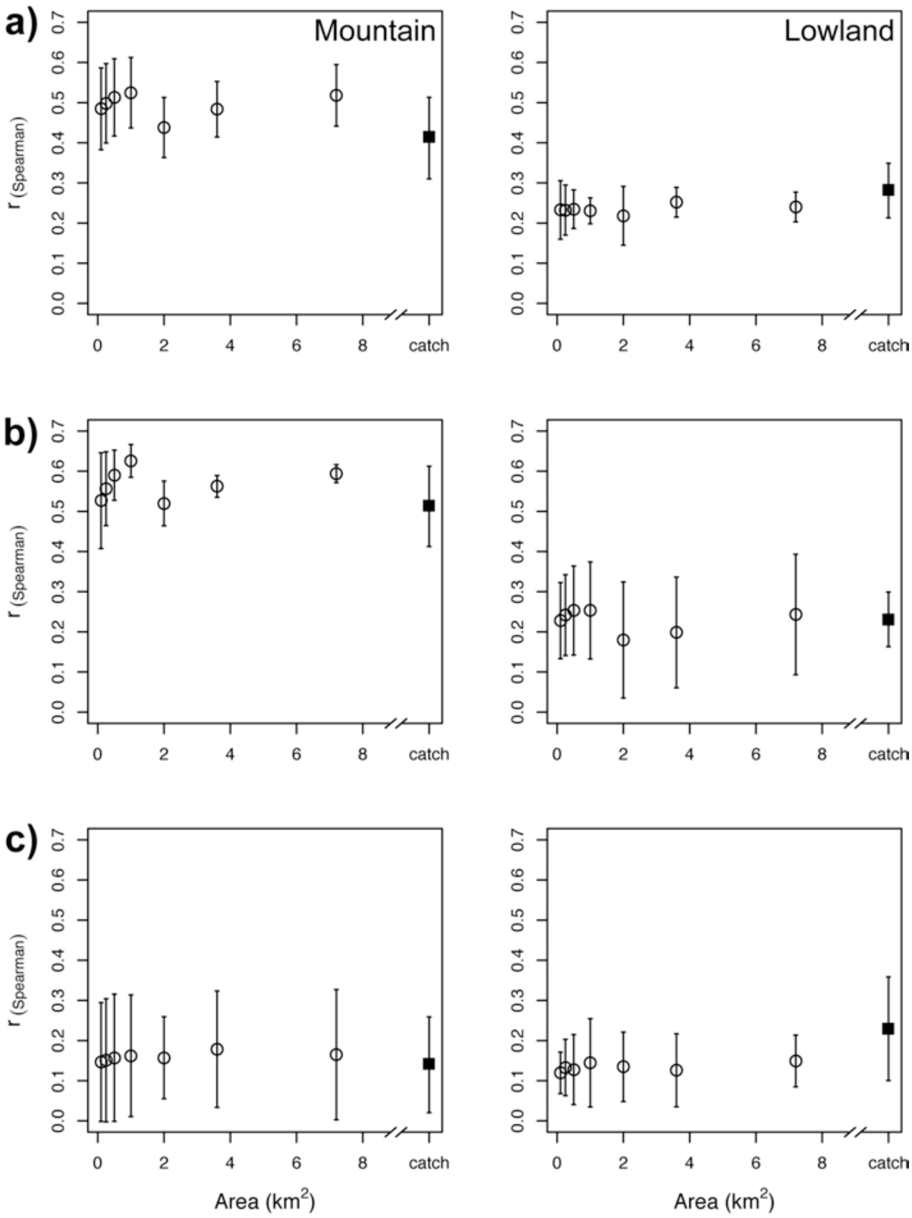

Figure 5: Mean Spearman correlation (absolute values) ± SD of a) benthic invertebrate (# EPT taxa), b) fish (sWQO2it) and c) macrophyte (NMACRx) metrics with arable and forest land use divided by ecoregion type. For clarity reasons, only seven buffers (open circles: all lengths in 100 m strip plus 10 km length in 200, 360 and 720 m strip) and catchment area (filled squares) are displayed.

Response of BQEs to land use type and ecoregion

Irrespective of land use scale, but separated by ecoregion and land use type, the ten best performing metrics are listed for each assemblage in Table 3–5. The direct comparison of correlation maxima ranked fish metrics highest (mean maxima across all spatial scales, land use types and ecoregions: $r = 0.615 \pm 0.090$), followed by benthic invertebrates ($r = 0.512 \pm 0.154$) and, with considerably lower values by aquatic macrophytes ($r = 0.340 \pm 0.147$).

In particular, those metrics performed well that respond to changing water and microhabitat quality, for instance ASPT, # EPT taxa, and % lithal and psammal preferences (p_Lithal, p_Psam) among benthic invertebrates or % oxygen- and habitat-demanding fish (sWQO2it, sHit). Except for ecoregion 14, benthic invertebrates revealed stronger correlations with arable than with forest land use (Table 4), while this finding was reversed for fish (Table 3). Aquatic macrophyte richness and diversity showed less clear differences and correlated consistently higher with forest land use in ecoregion 8 and 9, but not in ecoregion 13 and 14 where no clear difference was evident (Table 5). All assemblages revealed consistently and significantly higher correlations to land use in mountain systems (Table 3–5, also exemplarily illustrated in Figure 5).

A notable pattern was obvious for macrophyte richness metrics that were found to be negatively correlated with forest cover at the catchment scale in all ecoregions and, except for ecoregion 9, also with forest at the near-stream and intermediate buffer scales (Figure 6). Accordingly, true aquatic macrophyte richness (NMACRx) almost consistently decreased with increasing 'natural' land cover, which may affect the general suitability of this assemblage as indicator group for natural land use impact—at least with respect the macrophyte richness measures applied here.

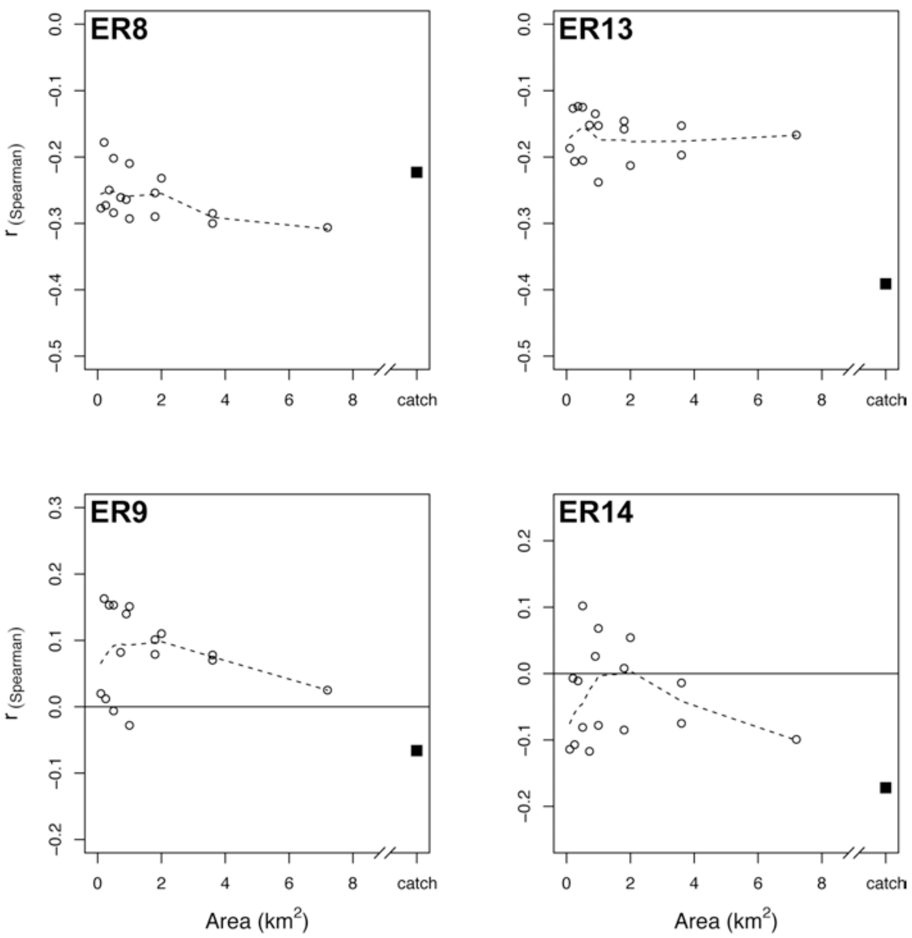

Figure 6: Primarily negative correlation between aquatic macrophyte richness and forest land use along the buffer size gradient (open circles) and at the catchment scale (catch, filled squares) in mountain (ER8, 9) and lowland ecoregions (ER13, 14). Buffer values are fitted using a lowess smoother (dashed line).

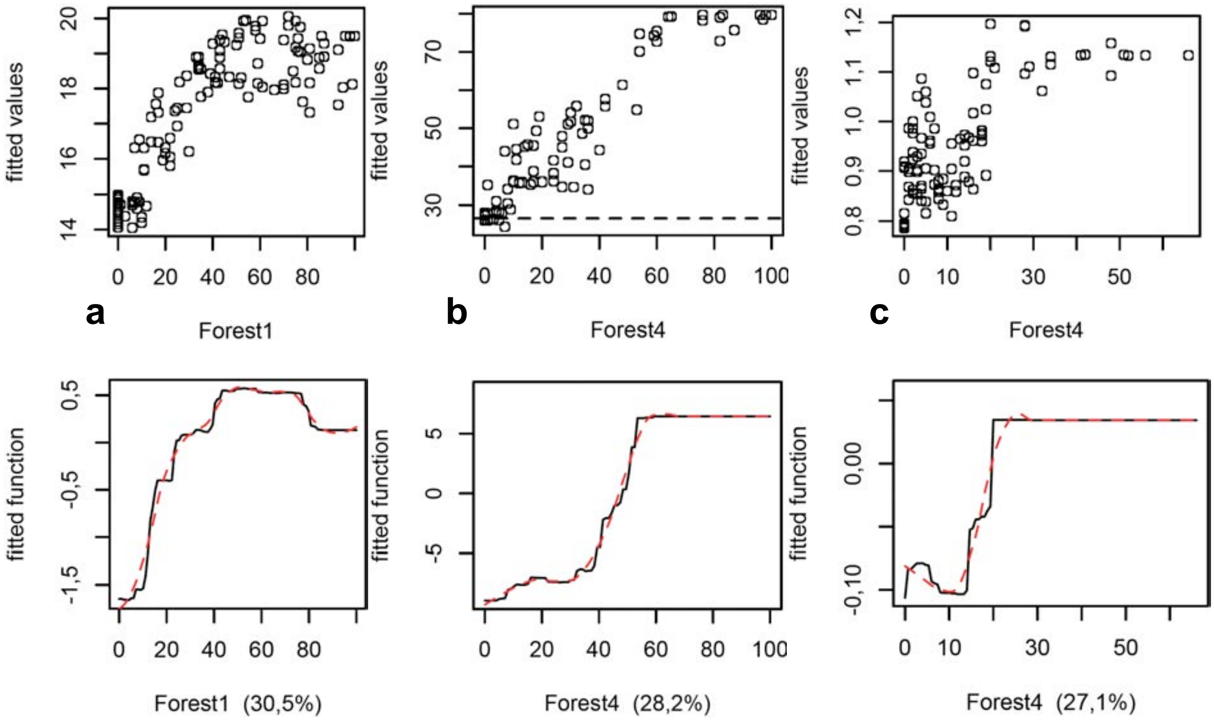

Figure 7: BRT fitted values (top row: original metric values along the land use gradient) and fitted functions (bottom row: solid line = modelled fit, dashed line = smoothed fitted function) for the relationship of near-stream forest cover (100 m buffer strip) and three selected metrics: a) number of EPT taxa in ER8; b) % habitat intolerant fish taxa along in ER 9; c) number of moss species in ER 14. Values in brackets indicate the variance in the metric described by the respective spatial scale (Forest 1 and 4 correspond to forest cover in 0.1 x 1 m and 0.1 x 10 km buffer size, respectively). See text for explanation.

Identification of effective spatial scales and land cover thresholds

I used the outcome of the RDA to rank the order of importance of environmental variables for the overall variance in the biological metrics. Expressed as conditional effect, i.e. is the unique contribution of each of the 17 spatial scales in the analysis, the results revealed a dominating role of near-stream and intermediate buffer sizes (Table 6). In 14 out of 24 (= 60%) combinations of BQE and ecoregion, the near-stream buffer was most descriptive. Notably, the catchment scale was identified only five times (four times in the lowlands and only once in the mountains), while in particular the macrophyte metrics revealed a consistently high responsiveness to buffer scale land use.

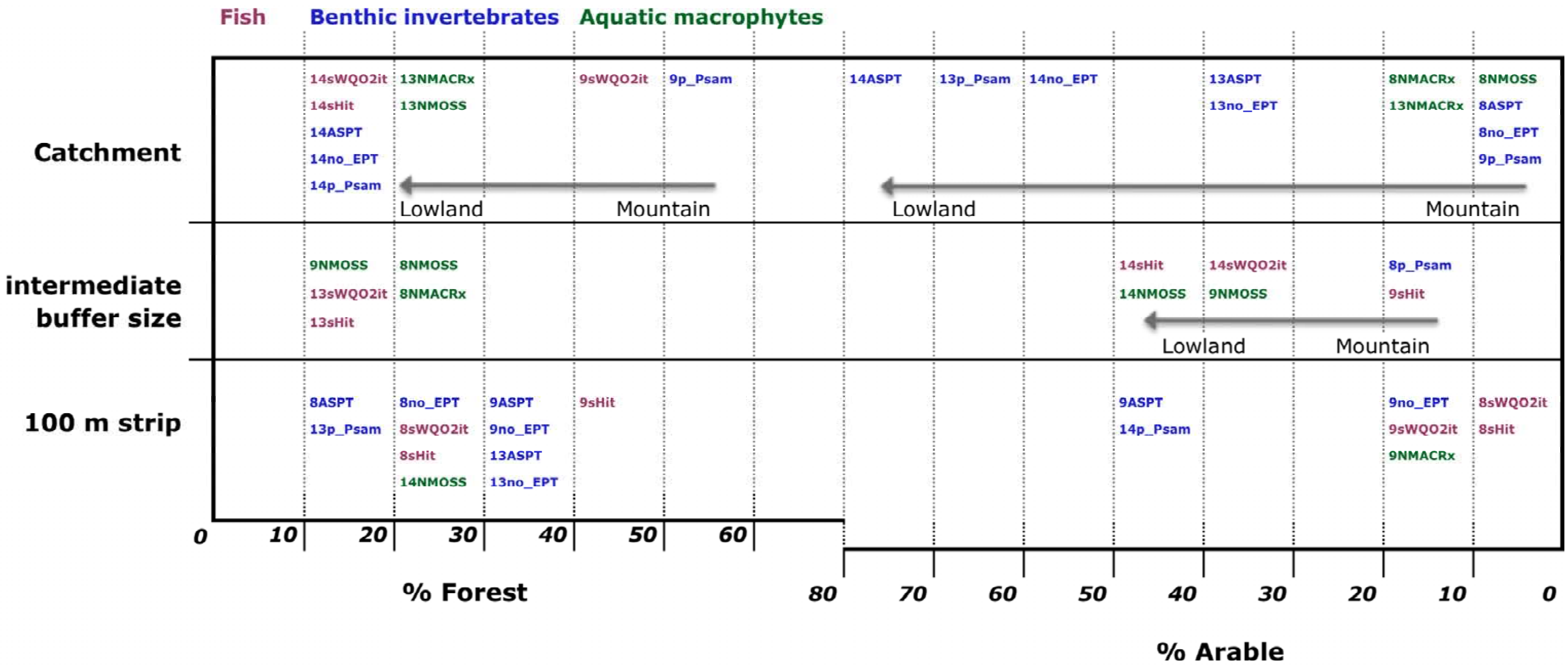

Figure 8: Ranges of change points identified for seven metrics, four ecoregions and three spatial scales and two land use types. Initial numbers in the metric names correspond to the ecoregion for which the relationship was found, while the location in the rectangular matrix indicates the most descriptive spatial scale and land use type for this relationship. For metric descriptions, see Annex 1. Arrows indicate the shift of change points from mountain to lowland systems.

The application of BRT to the same data largely supports the dominating role of near-stream buffer land use (results not shown in detail here). For each environmental descriptor variable, BRT provides the percent variability in the model explained by the variable, all of which sum up to 100%. This outcome revealed the buffer scale to be particularly descriptive in mountain systems (ecoregion 8 and 9). In addition, BRT allows for the identification of change points along the environmental (land cover) gradient, at which biological metrics most strongly rise or drop down, respectively. Figure 7 exemplarily illustrates this for three metrics, one each of which representing the fish (sHit: percent habitat-intolerant fish taxa), benthic invertebrates (no_EPT: number of Ephemeroptera-Plecoptera-Trichoptera taxa) and aquatic macrophytes (NMOSS: number of moss taxa) in relation to near-stream buffer forest cover. While the upper part of Figure 7 displays the modelled metric values (y-axis, original metric scale) along the land use gradient (x-axis), the lower row represents a smoothed version of the fitted values in the models. Accordingly, change points were detectable at: 10–20% forest cover for the number of EPT taxa in ER 8; 40–50% forest cover for habitat-intolerant fish in ER 9; 10–20% forest cover for the number of moss species in ER 14 (Figure 8a, b and c, respectively). The summary of 56 BRTs (seven metrics by four ecoregions by two land use types) is illustrated in Figure 8 and suggests a higher descriptiveness of catchment-scale land use in lowland systems (ecoregion 13 and 14), as opposed to the mountain ecoregions 8 and 9.

*Table 6: Conditional effects of responsive land use scales identified by RDA (multivariate regression) analysis. Only the top-ranking spatial scale (buffer width x length or catchment) is presented for each combination of BQE, ecoregion and land use type. Near-stream (100 m width) buffers indicated in bold. Significance level: ^{ns} (not significant), * (p <0.05) or ** (p <0.01).*

		Fish		Benthic invertebrates		Aquatic macrophytes	
		scale	F	scale	F	scale	F
Mountain	ER 8 arable	**0.1 x 10 km**	10.0**	catchment	9.9**	**0.1 x 10 km**	1.8^{ns}
Mountain	ER 8 forest	**0.1 x 5 km**	24.8**	**0.1 x 1 km**	7.3**	**0.1 x 10 km**	6.6**
Mountain	ER 9 arable	0.72 x 10 km	10.7**	0.72 x 5 km	6.3**	0.36 x 10 km	3.1*
Mountain	ER 9 forest	**0.1 x 1 km**	8.2**	**0.1 x 5 km**	7.5**	0.36 x 10 km	3.7**
Lowland	ER 13 arable	**0.1 x 2.5 km**	7.8	**0.1 x 1 km**	4.6**	**0.1 x 1 km**	3.3*
Lowland	ER 13 forest	**0.1 x 10 km**	14.3**	catchment	2.8*	**0.1 x 10 km**	1.8^{ns}
Lowland	ER 14 arable	catchment	2.6*	0.36 x 5 km	3.4**	**0.1 x 2.5 km**	3.4*
Lowland	ER 14 forest	catchment	1.5^{ns}	catchment	2.0^{ns}	**0.1 x 2.5 km**	2.8*

Of particular importance, however, is the difference of change points in mountain and lowland systems that is clearly evident from the BRT results (Figure 8). The tested metrics respond (change) at 0–20% arable land use in mountain catchments as opposed to remarkably higher values of 30–80% in lowland systems. The values also differ strongly for catchment forest cover (40–60% in mountain *vs.* 10–30% in lowland ecoregions). No clear ecoregional and land use type-specific patterns were detectable for the intermediate and near-stream buffer scales. Hence, buffer forest cover turned out to be influential at 10–40%, while corresponding values for buffer arable land use ranged 0–50% (Figure 8).

Discussion

The central aim of this study was to investigate whether the spatial scaling of land use impacts fish, benthic invertebrates and aquatic macrophytes. Yet, before the attempt can be made here to discuss the findings in light of spatial scaling effects, there is a need to account for the interrelationships between scales and biological assemblages (BQEs) as well as for ecoregional differences, all of which potentially may influence and even flaw the conclusions presented in the following, drawn about the role of land use scaling for the aquatic flora and fauna.

Land use as a proxy for environmental change

The aim of this study was to identify community-based metrics of different lotic assemblages that correlate with arable and forest land use at different spatial scales. To be useful, such metrics ideally respond in a predictable way to the changing environmental states induced by land use, so that they reflect ecologically meaningful relationships. Allan's (2004) review on the influence of land use on stream ecosystems, for instance, summarised nutrients and suspended fine sediments to be major stressor variables in agricultural landscapes. Corresponding examples of ecologically meaningful effects on aquatic macroinvertebrates are, for instance, the increase of fine sediment feeders (gathering collectors) and dwellers (sand-inhabiting taxa), the latter of which was also found significantly correlated to percent agriculture in lowland systems here (Table 4). Furthermore, excessive loads of suspended sediments can cause a loss of benthic invertebrates and fishes, in particular of the species that inhabit coarser substrates (e.g. gravel, stones) or spawn on them (e.g. salmonid fish). Nutrient enrichment, in contrast, is known to directly affect habitat quality, for instance, through algal growth (Quinn 2000), which in turn can impact pollution sensitive fish or benthic invertebrate taxa, for instance, through oxygen depletion (Furse et al. 2006).

Based on the analysis of nearly 500 sample stations, my findings support the existence of such major cause-effect chains from agriculture to macroinvertebrate and fish indicators. This is evident, for instance, by macroinvertebrate habitat preferences (for psammal and lithal) or fish and invertebrate water quality metrics (O_2-intolerant fish, ASPT) that proved highly responsive to catchment agriculture. Although not considered further in this study, these findings support the existence major pathways of land use impact, i.e. via suspended sediments and nutrients as reported by Allan (2004) and the numerous reviewed studies therein.

Catchment land use as a proxy for near-stream conditions

Many bio-indication studies have investigated the effects of whole catchment agriculture to identify suitable general indicators of the impacts of large-scale land use on fish (Roth et al. 1996; Wang et al. 1997), benthic invertebrates (Hering et al. 2004; Feld & Hering 2007) and additional organism groups (e.g. Hering et al. 2006; Johnson et al. 2007). These studies have in common that they have found large-scale land use patterns to explain a significant portion of the sampled biological variability, often comparable to that portion explained by water chemistry and much higher than the variability explained by reach-scale and local hydromorphology. Contrastingly, equally numerous studies have reported land use patterns in the finer stretch or reach (buffer) scale to be stronger related to lotic assemblages than land uses in entire watersheds (e.g. Lammert & Allan 1999; Meador & Goldstein 2003). This contradiction continues to raise the question of what scale is eventually more important for the integrity of the lotic environment and, thus, continues to attract limnologist's interest. For more than a decade, limnologists and water managers seek to answer this question in the context of stream and river monitoring and management at the pan-European scale (Hering et al. 2010). The answer is crucial for river basin managers, who have to implement the European Union Water Framework Directive, since ecologically successful management is depending on the scale of management that should fit the scale of (anthropogenic) impact to be mitigated (Feld et al. 2011).

In this study, I found the catchment and reach scale buffer land use to be strongly interrelated in all ecoregions investigated (Table 2), which implies that catchment-scale land use is a useful predictor for land use patterns at finer scales. This, in particular, applies to percent agriculture, which was found to be a good predictor regardless of catchment size and ecoregion type. Hence, whatever biological metric is strongly related to the percent of catchment as agriculture in the regions subjected here, the same metric is likely to be equally correlated with percent as agriculture at the reach-scale riparian buffer (1–10 km length x 100 m width). Yet, obviously it is not this simple. I also found percent forest cover in the 100 m buffer to largely vary and achieve up to 60%, even under high levels of catchment agriculture, in three out of four ecoregions. This variability in near-stream land cover conditions, however, adds to the variability of catchment land cover (Stauffer et al. 2000). For example, riparian forest can effectively buffer nutrient and sediment entries to the in-stream environment (Osborne & Kovacic 1993; Dosskey 2001), provides organic matter (Lester & Boulton 2008), shade (Broadmeadow & Nisbet 2004), shelter (Brooks et al. 2004), regulates water temperature (Barton et al. 1985), induces bank stability (Lester & Boulton 2008) and can inhibit algal growth (Sponseller et al. 2001). Thus, the use of catchment-scale conditions alone is unlikely to sufficiently address these important ecological mechanisms that control lotic assemblages and ultimately determine the ecological integrity of lotic ecosystems. Consequently, the use of riparian land cover instead of (or at least in addition to) catchment land cover in bio-indication studies is of paramount importance.

Comparison of the response of different BQEs

In light of the major environmental implications of agriculture—increasing suspended sediments and nutrients—the faunal assemblages were found here to be much more responsive to both effects than the macrophytes. The direct comparison of faunal assemblages revealed fish metrics to be significantly better correlated with land use than benthic invertebrate metrics (U-test, $p < 0.0025$), alike the finding of Fitzpatrick et al. (2001). It seems as if the majority of previous land use research preferred fish and/or benthic invertebrates to the aquatic flora. Macrophytes and diatoms have been addressed considerably less often (Hering et al. 2006; Johnson et al. 2007), probably because of their strong linkage to riparian shade (Sponseller et al. 2001), which is also implied by my findings: macrophyte richness responded best to near-stream land cover in six out of eight combinations of ecoregion and land use type (Table 6).

Furthermore, macrophyte richness was significantly and negatively correlated with percent forest cover in near-stream buffers in one mountain and both lowland ecoregions (ER8, 13 and 14). One has to be cautious here, as 'near-stream' in this study is defined as up to 50 m off the stream banks into the floodplain on either side (= 100 m buffer). Due to the spatial resolution of CORINE land cover data, the generation and use of smaller buffer widths was considered unrealistic for this study. Hence, the near-stream buffer strip is not necessarily reflecting well the actual occurrence of wooded vegetation close to a stream, but rather the presence of larger patches of floodplain forest. Nevertheless, it is evident also from previous studies that it is rather the forest land cover close to a stream that controls in-stream plant growth through the control of light (or shade, respectively) and only to a lesser degree (if at all in agricultural landscapes), by nutrient retention (Sponseller et al. 2001).

Comparison of mountain and lowland ecoregions

When comparing mountain and lowland ecoregions, the mountain fauna and flora was found to correlate consistently and significantly better (and less variable the correlations) with land use for all tested metrics, regardless of the land use type (Table 3–5). In other words, the correlation analysis revealed the mountainous assemblages being more responsive to land use. This finding is not trivial to interpret and also the consulted body of peer-reviewed literature does not provide sufficient evidence for a proper reasoning. I assume that this finding might (in part) be induced by the fundamentally different land use legacies in both ecoregion types.

A satellite image impressively and easily shows that mountainous environments in Central and Western Europe are primarily covered by mixtures of forest and pasture; intensive forms of agriculture (e.g. row crops) are largely missing at higher elevations (http://www.eea.europa.eu/data-and-maps/figures/land-cover-2006-and-changes). This is presumably due to geographical constraints (e.g. high slope, low soil quality) that render mountainous regions mostly unsuitable for intensive forms of cultivation. In contrast, the dominating land use form in both lowland ecoregions is intensive agriculture. Agriculture also has a historic dimension in Central Europe—as in many other developed regions around the world. Intensive agriculture, in parallel with deforestation, appeared in the mid-19th century (Foster et al. 2003). Harding et al. (1998) concluded from their comparison of stream biodiversity patterns with land use in the 1950s and 1990s that former effects of land use can continue to impact in-stream ecology even long after land use changed and named it the 'ghost of land use past'.

Besides legacy land use, the *current* length of land use gradients may also have influenced the differences in the response of assemblages between ecoregion types. For instance, the catchments in the Western Mountains (ecoregion 8) revealed a comparatively short agricultural gradient ranging only 0–30% arable land use, while this was 0–80% in the Central Mountains (ecoregion 9) and Western Lowlands (ecoregion 13), and 20–100% in the Central Lowlands (ecoregion 14). Correspondingly, the values for forest cover showed a reversed pattern and ranged 0–100% in the mountainous and 0–50(60)% in the lowland catchments. Yet, these relatively *current* land use patterns and do not seem to have systematically influenced the results presented here; the correlations of all assemblages were consistently better in mountain ecoregions, regardless of land use type.

Hence, although neither proven nor provable in the frame of this study, legacy land use may continue to constitute a severe, permanent and long-term impact, in particular on lowland ecosystems in densely populated areas of Central and Western Europe. This "legacy baseline impact" continues to degrade many lotic environments and thereby shortens the lengths of biological assemblage's gradients. The currently detectable biological gradients, thus, do not (fully) reflect current land use patterns, but to a certain degree are still 'cut off' by legacy land use impacts including, for instance urbanisation and deforestation. That is, the *current* gradients lack an 'unstressed' endpoint, presumably even under comparatively natural riparian land use conditions. Without this endpoint, however, ecosystem indication inevitably runs the risk of failing to indicate reliably the effects of actual land use and other stressors in agricultural landscapes—in particular where monitoring systems neglect regional or water type-specific reference conditions that may provide alternative endpoints.

Comparison of BQEs response to buffer and catchment scale land use

Apart from the discussion about interrelationships of land use at different scales, the central aim of this study was to investigate the role of spatial scaling in the BQEs response. The initial hypothesis was that an 'impact type' of land use (e.g. agriculture) primarily acts at the scale of entire catchments, while a 'natural type' of land use (e.g. forest) acts at finer scales, for instance, at the stretch or reach scale, and potentially buffers agricultural impacts, for example, by retaining nutrients and suspended sediments. Inter alia, this hypothesis was driven by previous research, such as the field study by Jones et al. (2001); the authors investigated 78 streams in the Mid-Atlantic Highland regions of the U.S. and found in particular the proportion of agriculture in the catchment and forest in the riparian zone to explain most of the variation in nutrient and suspended sediment yields.

However, my findings do not support this hypothesis. From the correlation analysis of metrics along the buffer size gradient (0.1–7.2 km^2, Figure 5) and from multivariate analysis of metrics along gradients of percent land use (Table 6), it is evident that the assemblages in general were more responsive (mountain systems) or equally responsive (lowland systems) to the near-stream land use conditions. Furthermore, the BQEs were increasingly responsive to near-stream buffer conditions with increasing buffer lengths, which implies that both buffer widths and lengths require consideration here.

As already stated further above, there are numerous examples in the peer-reviewed literature that support the strong relationship of aquatic assemblages to land use in the riparian buffer (e.g. Lammert & Allan 1999; Stewart et al. 2001; Meador & Goldstein 2003), as there are equally numerous contrasting examples that have found the opposite and highlighted the better correlation of assemblages with catchment-scale land cover patterns (e.g. Frissel et al. 1986; Roth et al. 1996; Feld & Hering 2007). The latter is also supported by fundamental ecological concepts, such as the paradigm that the "valley rules the stream" by Hynes (1975), which later has been put into a conceptual framework of spatially nested environmental factors by Frissell et al. (1986). In this framework, large-scale geomorphic processes shape the environment at smaller scales and, thus, the habitat for instream assemblages at the fine scale. So, one may conclude that, following ecological theory, the catchment scale is superior to subsequent smaller scales nested within the catchment, yet that there is no clear advantage of using one scale over the other in indicator studies.

Lammert & Allan (1999) revisited the same sub-catchment as Roth et al. (1996) and, similar to Roth and co-authors, investigated the response of fish to land use at different spatial scales. Interestingly, while the former have reported land use within a 100 m buffer to be the best predictor of the integrity of the fish assemblage, the latter study has found the fish IBI scores to be correlated best with land use in the catchment above the investigated sites. Lammert & Allan (1999) concluded that the contradictory findings were owed to the different spatial resolution of the sampled data that was reflecting the local conditions better in their study, while Roth et al. (1996) also considered the regional environmental conditions.

Hence, there is evidence that the spatial extent of a whole study can influence the outcome with regard to the importance of particular spatial scales. Furthermore, we have to be cautious when comparing studies that considered catchments <100 km^2 (e.g. Roth et al. 1996; Stewart et al. 2001) with those that include catchments >1,000km^2 (this study, see Figure 3), as the catchment size in small-scale studies may correspond to buffer areas in studies that cover large geographic scales. Therefore, I suggest referring to the area of scales instead of (or in addition to) a mere use of terms such as 'catchment', 'watershed' or 'buffer', the spatial context and understanding of which is highly variable in the literature.

Being put in this spatial context, this study revealed a strong relationship of land use patterns in the 100 m buffer, while the correlations increased with buffer lengths (10 > 5 > 2.5 > 1 km length). At large geographic scales, near-stream buffer land use is presumably a good predictor for in-stream assemblage conditions (see also Meador & Goldstein 2003) and may affect the instream assemblages more directly, as compared to the catchment-wide land use conditions.

Comparison of change points of BQEs response to land use

Change points (or ranges thereof) mark thresholds along the land use gradients at which a dramatic change in the tested assemblage metrics was detectable. Again, this study points at notable differences between mountain and lowland ecoregions, but not between BQEs. In particular, my findings imply the existence of threshold ranges for catchment agriculture (0–20% arable in mountain *vs.* [10–] 30–80% in lowland systems) and catchment forest cover (40–60% in mountain *vs.* 10–30% in lowland systems). Similar trends were detectable for near-stream agriculture (0–20% in mountain *vs.* 30–50% in lowland systems), but not for forest cover in the 100 m buffer (30–50% in both ecosystem types). Thus, it is evident that the mountain assemblages responded much more sensitive (or earlier) to changing degrees of land use impact and to the loss of forest cover, respectively. At first glance, one may hypothesise that mountain catchments respond earlier to land use impact, which is in line with Snyder et al. (2003); these authors found catchments in steeper terrain to be more affected by (urban) land use.

This finding, however, may also support the assumptions made above about the role of land use legacy. Harding et al. (1998) have found that ‘past land use activity, particularly, agriculture, may result in long-term modifications to and reductions in aquatic diversity, regardless of reforestation of riparian zones’. Although the results presented here revealed detectable response of instream biota to riparian land cover even in intensively-used agricultural catchments, the putative tolerance of lowland communities towards the coverage of agricultural land use, as found in parallel, may already mirror a fundamental and long-term degradation of entire catchments that no longer allows the degradation-sensitive community members to reproduce and sustain. Ehlert et al. (2002) have reviewed the historic macroinvertebrate colonisation in 18 medium-sized to large rivers in Germany and reported numerous unionid (mussels), stonefly and mayfly taxa to have gone extinct after 1900, probably because of the manifold complex of stressors that continue to impact lowland rivers since then. These taxa once were common within the German part of the Central European lowlands (e.g. Borcherding 1883) and probably also beyond. The remaining, fragmented benthic invertebrate community currently comprises mostly insensitive taxa (Ehlert et al. 2002), which are presumably incapable of a reliable indication of changes at (or close to) the “unstressed” endpoint of present land use gradients.

The change points identified along the agricultural gradient in the present study resemble those reviewed by Allan (2004): “Streams in agricultural catchments usually remain in good condition until the extent of agriculture is relatively high, more than 30–50%. Slightly more rigorous thresholds, however, have been reported by Fitzpatrick et al (2001), who investigated the response of different river assemblages to land use in a lowland region west of Lake Michigan, U.S.A. These authors found fish IBI scores to drop below ‘good’ when watershed agriculture increased above 30%, whereas the threshold was at as low as 10% for near-stream buffer agriculture. Yet, their sampling design explicitly included high-quality benchmark sites, which may have supported a fish assemblages diverse enough to respond more close to the “unstressed” endpoint of the impact gradient and eventually, which may explain why these authors were able to identify more rigorous change points.

Synthesis and implications for River Basin Management

River Basin Management attempts to assess, monitor and improve the integrity, or ‘ecological quality in terms of the European Union Water Framework Directive (WFD). By definition, the spatial scale of interest, thus, is the entire catchment (= river basin). Consequently, water managers, who are obliged to implement the EU WFD and corresponding water legislations in other regions outside Europe, urgently seek for guidance on the appropriate spatial scaling for RBM. Although numerous research studies have suggested a predominance of catchment scale stressors, such as land use and its related effects on the environment (Harding et al. 1998, Allan 2004), this scale is unlikely to be a realistic scale with respect to practical restoration (Bishop et al. 2009). More likely, restoration measures will continue to be implemented at smaller (reach or stretch) scales as summarised in the reviews of Palmer et al. (2009) and Feld et al. (2011). The ultimate question to practical water managers (and scientists: see Bernhardt & Palmer 2011) hence is whether this rather "bottom-up" management approach will be successful in mitigating the "top-down" and nested effects of land use as described and illustrated by Frissel et al. (1986).

The results presented in this study suggest that catchment-scale land use controls the land use conditions at finer scales and, thus, may provide a reliable and easy-to-measure proxy for environmental assessment also at smaller scales. Yet, regarding the relationship of land use to lotic assemblages of aquatic macrophytes, benthic invertebrates and fish, I was able to show that community metrics responded stronger to percent agriculture and forest in near-stream buffer strips. Hence, catchment land use should be considered a proximate rather than the ultimate factor shaping the in-stream communities and their integrity.

Ultimately, it is more likely the manifold effects of near-stream land use that control in-stream physico-chemistry (Quinn 2000; Jones et al. 2001; Sponseller et al. 2001), hydromorphology (Johnson et al. 2003) and local habitat availability (Roth et al. 1996; Wang et al. 1997). Moreover, riparian buffer restoration can mitigate adverse effects of agriculture through mechanisms of retention (Osborne & Kovacic 1993; Dosskey 2001). Riparian wooded vegetation can control the function of instream food webs (Quinn 2000; Beechie et al. 2010) through the provision of allochthonous organic material (e.g. leaves, large wood). In summary, there is sufficient evidence that restoring the functionality of riparian forests can effectively restore the in-stream habitat and, thereby, the integrity of aquatic assemblages. The results presented in this study underpin the important role of the near-steam buffer scale and imply that riparian land use management is likely to be biologically effective and probably also successful with respect to 'ecological integrity' or 'ecological quality' targets. A smart design and planning of multiple "bottom-up" restoration measures at the riparian scale, then in combination may eventually lead to substantial improvements also at the river basin scale.

Unlike land management at the scale of entire catchments, riparian (buffer strip) restoration is more realistic and feasible also in the short term. It is, however, crucial that riparian buffers are sufficiently wide and long in order to provide an effective buffer against agricultural impacts. A minimum width cannot be derived from this study, since the smallest buffer was 100 m wide (50 m on either side); previous studies suggest a minimum width of 30 m on either side in order to provide a proper functionality (e.g. Castelle et al. 1994; Wenger 1999). The findings presented in this study, however, revealed an increasing strength of the correlation of assemblage metrics with near-stream land use at increasing buffer lengths. This in turn suggests that longer buffers (several to many km in length) are more influential than shorter ones, probably because, for instance, temperature and sediment controls are a function of buffer width and length.

With regard to thresholds of percent land use at which notable changes in assemblage structure and function can be detected, the results presented here suggest a fundamental difference between mountain and lowland systems. Probably owed to the different landscape gradients (slopes) in both ecoregion types (see Snyder et al. 2003), mountain systems seem to respond much more sensitive to agriculture (0–20%) than lowland systems (30–50%), which is in line with previous research (Allan 2004). Accordingly, land use management in mountain systems should aim at keeping percent agriculture below these levels in order to maintain intact assemblages, in particular their water quality- and habitat-sensitive members.

Finally, my findings suggest that the instream fauna is stronger related to land use than the flora. In particular, the fish metrics tested here may constitute promising aggregate indicators of the multiple impacts of land use at both the broad and the fine spatial scale. Before implementing such indicators, however, a thorough testing of response trends along the land use gradients will be required, as these trends may vary fundamentally.

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Spatially based methods to describe the hydro-morphological status of European River Groups

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Abstract

Within the EU funded project WISER (Water Bodies in Europe: Integrative systems to assess Ecological status and Recovery. A Contribution to the Water Framework Directive), a spatially based, river-type specific approach was used to develop an ecological assessment method for European rivers based on existing sampling data. The methodology comprised two main steps: (1) the definition and environmental description of European river groups (2) analyses of hydro-morphological impacted conditions for each group. Clustering abiotic characteristics identified five homogeneous groups in 11 European ecoregions. They encompassed both regional (geographic position in Europe) and local factors (longitudinal zonations) influencing the distributions of riverine fish and benthic invertebrates. To assess the ecological status, the response of more than 300 fish metrics and benthic invertebrate metrics (species diversity and abundance) to human pressures was tested for each river group individually. A maximum of 17 potential fish and 26 benthic invertebrate metrics was selected using logistic regression. The density of intolerant species and migrating guilds regarding fishes and composition as well as functional traits regarding invertebrates respectively had the highest capacity for predicting the intensity of perturbation.

Keywords: Europe, fish, benthic invertebrates, rivers, Water Framework Directive, hydro-morphological pressures

Introduction

The spatially-based approach is one of the methodological options for assessing the ecological status of running waters within the Water Framework Directive (2000/60/EC). The principle is that rivers can be classified into units with homogenous abiotic and biotic characteristics. Multimetric approaches such as the Index of Biotic Integrity (IBI, Karr 1981) generally use an eco-region or bioregion approach and by that are limited to specific regions. Melcher et al. and Schmutz et al. demonstrated in 2007 that “spatially-based” assessment methods can be developed in European ecoregions for specific river segments with homogeneous fish assemblages. Ecoregions are supposed to provide a spatial framework for ecosystem at the large scale (Omernik 1995). Illies (1978) introduced a European classification system dividing the continent into 25 ecoregions. This system has never been evaluated for its ability to discriminate between fish and benthic invertebrate assemblages at a continental scale and no attempt has been made to analyse longitudinal patterns across ecoregions at that scale. Due to the biogeographic situation in Europe some assemblage types, e.g. brown trout dominated communities, can be found throughout Europe. This evidence is supported by the fact that the concept of fish zones and the corresponding macroinvertebrate patterns (Thienemann 1925, Huet 1949, Illies & Botosaneanu 1963) has long tradition all over Europe to explain natural variability of fish communities along the longitudinal gradient of running waters.

In the WFD the spatially-based classification of surface water bodies is based on the assumption that an abiotic river typology is adequate to stratify fish and macroinvertebrate communities sufficiently in order to distinguish between natural and anthropogenic (hydro-morphological pressures) variability. However, so far no efforts have been made to validate this assumption at the European scale for both fish and invertebrates.

The potential similarity between the assemblage types identified across Europe (Schmutz et al. 2007, Melcher et al. 2007) suggested that eco-regional fish types could be merged, thereby reducing the number of assessment methodologies necessary to monitor European rivers.

The objectives of this study are to (1) classify homogenous river-groups at the European level, (2) to compare identified fish and benthic invertebrates assemblage types with environmental and characteristics and (3) to describe hydro-morphological impacted and not impacted river sections along identified river-groups by using a set of defined potential metrics.

Results and Discussion

The data used in this study have been taken from the WISER Workpackage 5.1 database (Melcher et al., this volume). Selected data, with hydro-morphological pressures include in general sampling stations from 11 European eco-regions and nine countries (Table 1). More than 90 percent of the data was provided by Austria, Germany and France.

Hydro-morphological pressures

More sites are morphologically impacted than due to hydrological alterations (Table 2 and 3). For three hydrological pressure types (impoundment, water abstraction and hydro-peaking) only presence / absence information was available for specific river sections. In total only 180 stations out of ca. 1900 are affected directly by impoundment (Table 2). On the other side (Table 3) morphological changes (presence / absence) were stated in 90 percent of all cases.

Table 1: Number of stations with hydro-morphological pressures and their eco-regional and national distribution (source: WISER DB 5.1).

EcoregionID	CountryID									Total	Percent
	AT	DE	FR	SE	CZ	SK	UK	NL	DK		
9	428	198	6		9					641	33.8%
4	336		11							347	18.3%
11	225				14	2				241	12.7%
8		71	144							215	11.3%
13		27	160					8		195	10.3%
14		141		21				7	11	180	9.5%
18							25			25	1.3%
10					1	22				23	1.2%
5	18		0							18	0.9%
2			8							8	0.4%
22				5						5	0.3%
Total	1007	437	329	26	24	24	25	15	11	1898	100.0%
Percent	53.1%	23.0%	17.3%	1.4%	1.3%	1.3%	1.3%	0.8%	0.6%	100.0%	

Table 2: Number of stations out of three hydrological pressures (binary; 0,1) and their combinations.

PressureType	Number of pressures			
	1	2	3	Total
Impoundment	166	8	6	180
WaterAbstraction	31	49	6	86
Hydropeaking	8	45	6	59

Table 3: Number of stations out of six morphological pressure types and their pressure intensity.

	PressureIntensity				
PressureType	No			High	Total number classified
	0	1	2	3	
Morphology (No, Yes)	198	1515			1713
ArtificialEmbankment	408	388	408	513	1717
RiparianVegMod	425	339	302	680	1746
InstreamHabitatModified	1113		266	443	1822
ChannelFormModified	917		288	373	1578
CrossSectionModified	627		370	365	1362
VelocityIncrease	898	506			1404

A principal component analyses (Figure 1) shows the high interaction between six morphological parameters (Degerman et al. 2007). As a consequence, 'artificial embankment' and 'riparian vegetation modified' (RiperianVegetationMod) were selected representative for morphological alterations.

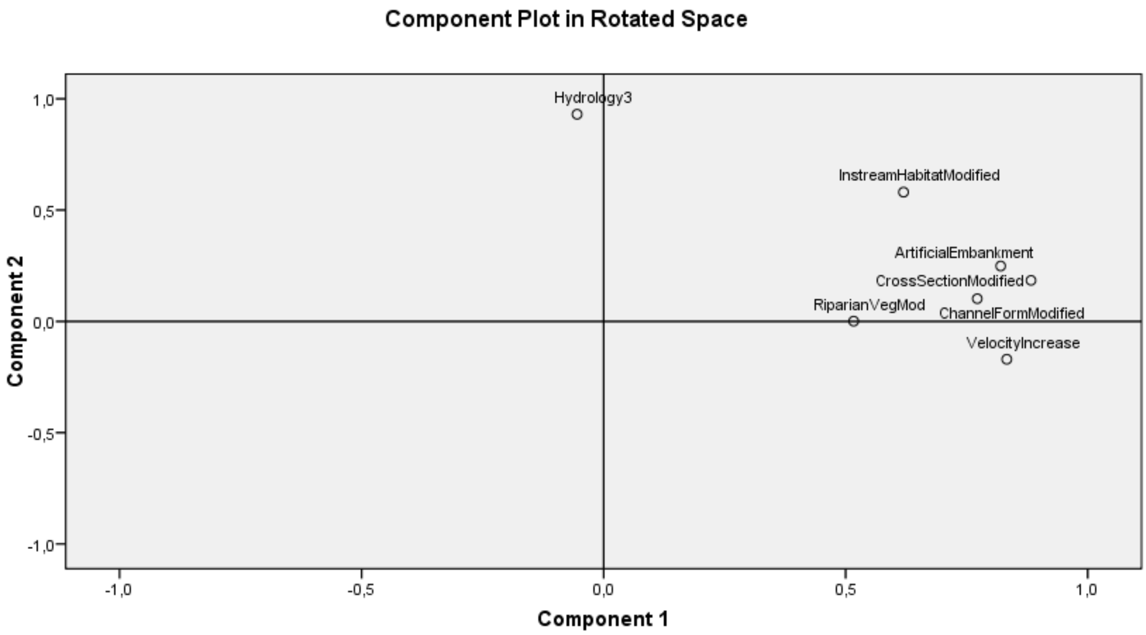


Figure 1: Component plot after factor analyses (SPSS 2003, rotated space; varimax) for hydrology and morphology.

Calculation of different pressure types into specific indices

The following description gives an overview of available hydro-morphological pressure types and their coding:

- Hydrology3 (0, 1): presence/absence for all three types: Impoundment (0, 1), Water Abstraction (0, 1) and Hydropeaking (0, 1)
- A hydrology 'Hy' index (0…6): could be calculated as follows: Hy = Impoundment *2 + WaterAbstraction *2 + Hydropeaking *2
- Morphology2 (0, 1):presence/absence for both types: Artificial embankment (0, 1, 2, 3) and Riparian vegetation modified (0, 1, 2, 3)
- A morphology 'Mo' index (0…6): could be calculated as follows: Mo = ArtificialEmbankment + RiparianVegMod
- A hydro-morphology HyMo index (0 … 12) could be calculated as following: HyMo = Hy + Mo

Figure 2 shows no significant differences between the intensity of morphological pressures and the altitude. At least the highest morphological intensity (index 6) could be allocated in larger catchment sizes.

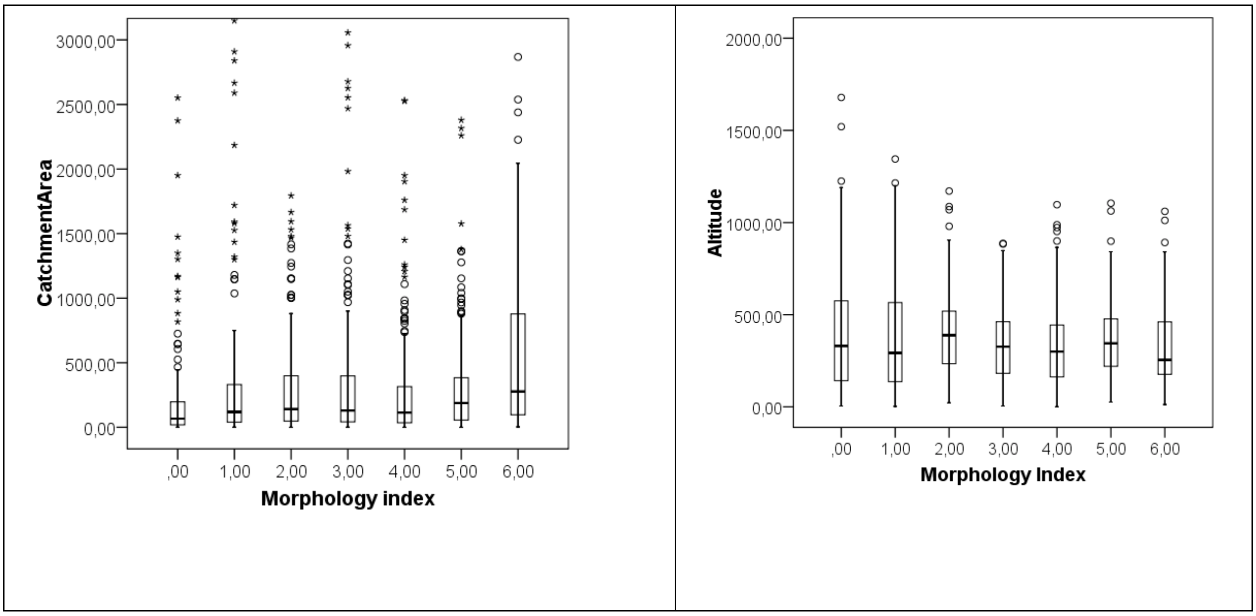


Figure 2: Frequency distribution of the intensity for morphological pressure intensity (Morphology Index) in comparison with size of catchment and altitude.

Defining European River-groups

In a next step so called European River-groups (river types) were identified by using three independent environmental descriptors, i.e. altitude (alt), mean air temperature and the size of catchment. These variables were chosen because they describe both the regional position in the hydro graphic network of European rivers and the organization of sites along the longitudinal continuum of rivers. All stations with invertebrates and fish occurrence (n ~ 1900) were used for clustering (Ward's method, Euclidean distance; Jobson 1992, SPSS 2003) these stations into five different European River-groups (A, B, C, D, E). Table 4 explains the assignment of five distinct European River-groups to European countries and eco-regions.

The cluster analysis splits the data into 5 clusters; cluster (river-groups) A and B are representing headwater assemblages (highland) with low species richness whereas cluster C, D and E are characterized by more divers fish and benthic invertebrate fauna (lowland). The river-groups are described by their species composition, environmental descriptors and the intensity of pressures (HyMo index) in Figure 3.

Table 4: Assignment of five distinct European River-groups to countries and eco-regions.

CountryID * QCL_2 Rivergroups (alt, temp, catch) Crosstabulation

Count

		QCL_2 Rivergroups (alt, temp, catch)					Total
		1 A	2 B	3 C	4 D	5 E	
CountryID	AT	155	298	188	364	2	1007
	CZ	0	15	9	0	0	24
	DE	11	77	280	61	0	429
	DK	0	11	0	0	0	11
	FR	14	14	83	117	109	337
	NL	0	0	10	15	0	25
	SE	14	12	0	0	0	26
	SK	1	16	4	3	0	24
	UK	0	2	22	0	0	24
Total		195	445	596	560	111	1907

EcoregionID * QCL_2 Rivergroups (alt, temp, catch) Crosstabulation

Count

		QCL_2 Rivergroups (alt, temp, catch)					Total
		1 A	2 B	3 C	4 D	5 E	
EcoregionID	2	2	0	4	1	2	9
	4	96	154	89	7	1	347
	5	0	0	1	17	0	18
	8	13	13	54	47	26	153
	9	69	222	238	164	0	693
	10	1	15	5	2	0	23
	11	0	7	16	216	2	241
	13	0	0	31	91	80	202
	14	9	32	136	15	0	192
	18	0	2	22	0	0	24
	22	5	0	0	0	0	5
Total		195	445	596	560	111	1907

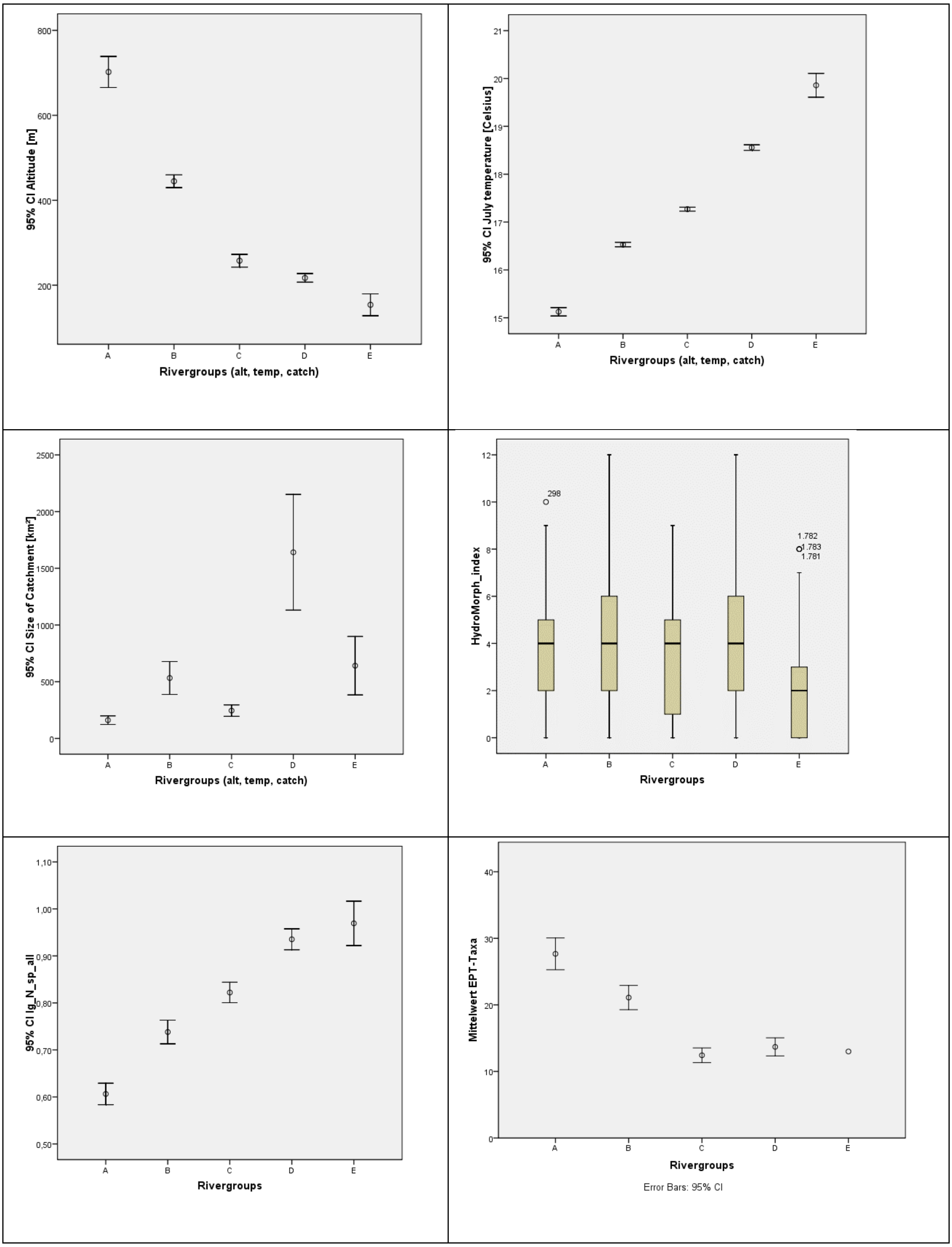


Figure 3: Mean value (log transformed) and 95% confidence interval (CI) of European River-groups: (1) abiotic description - altitude (a), mean July air temperature (b), size of catchment (c) intensity of pressures (d); and (2) biotic description – total number of fish species (e) and number of benthic invertebrates EPT taxa (f).

Benthic invertebrates analyses and metric selection

For the invertebrate assessment 2565 sites of 3 countries (Austria, Germany and the Netherlands) were selected for faunal taxonomical adjustment to an integrative determination level. First faunal analyses provided doubtful results due to a high significant separation of the sites per country. A further analysis of the taxa list showed that some taxa-groups still needed to be adjusted (e.g. Hydrachnidia and Chironomidae) to avoid a country specific separation in respect of taxonomic composition. As a further step of the invertebrate assessment similarities and dissimilarities of the taxa compositions were calculated with taxa-groups at order or suborder level for which no adjustment is needed.

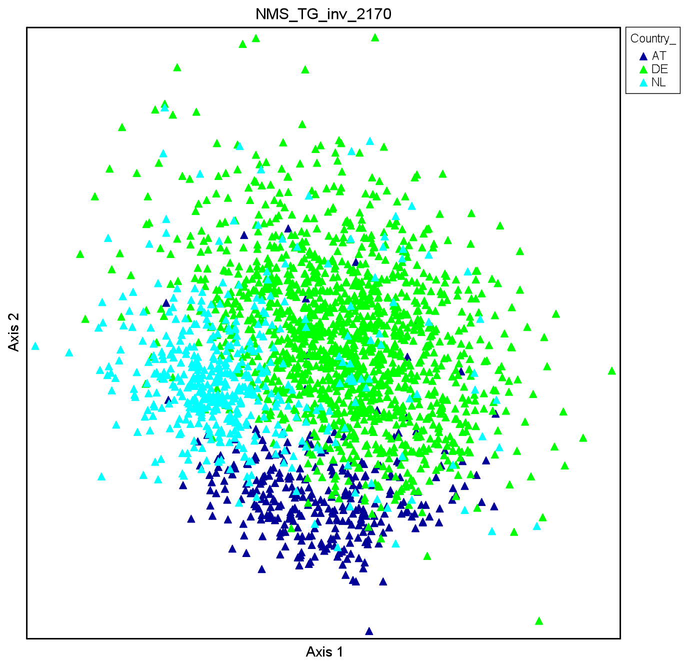
After excluding sites with abundances less than 100 specimens per m² and all qualitative Austrian sites a dataset of 2170 sites remained for further assessment. A detailed schedule is given in Table 5.

Table 5: Number of sites per country and eco-region including minima, maxima, mean and standard deviation of abundance [individuals/m²] and total number of taxa.

Descriptive Statistics

Country	EcoregionID		N	Minimum	Maximum	Mean	Std. Deviation
AT	4	Abundance [ind/m²]	88	413,6000	124152,0000	9505,106818	1,7213288E4
		Number of Taxa	88	25	127	69,74	17,376
		Valid N (listwise)	88				
	5	Abundance [ind/m²]	15	1124,8000	15751,2000	6417,120000	4,2917607E3
		Number of Taxa	15	51	94	73,47	12,626
		Valid N (listwise)	15				
	9	Abundance [ind/m²]	145	174,4000	83623,0000	6529,652414	8,3531629E3
		Number of Taxa	145	20	141	73,15	19,681
		Valid N (listwise)	145				
	11	Abundance [ind/m²]	65	200,0000	54116,0000	8283,964154	7,9338421E3
		Number of Taxa	65	17	106	60,51	21,176
		Valid N (listwise)	65				
DE	4	Abundance [ind/m²]	4	284,0000	1599,2000	1182,962500	606,8584340
		Number of Taxa	4	28	44	38,50	7,550
		Valid N (listwise)	4				
	8	Abundance [ind/m²]	104	103,2000	4641,6000	1277,058654	907,4082606
		Number of Taxa	104	10	74	36,58	13,985
		Valid N (listwise)	104				
	9	Abundance [ind/m²]	685	100,8000	18340,0000	1488,446307	1,7047587E3
		Number of Taxa	685	4	90	27,71	12,414
		Valid N (listwise)	685				
	13	Abundance [ind/m²]	113	100,0000	3371,2000	675,212389	704,4717307
		Number of Taxa	113	7	90	29,45	12,210
		Valid N (listwise)	113				
	14	Abundance [ind/m²]	476	109,6000	27136,0000	1021,925819	1,8472320E3
		Number of Taxa	476	2	82	26,64	11,064
		Valid N (listwise)	476				
NL	13	Abundance [ind/m²]	88	120,8000	12681,6000	1021,218182	1,6582518E3
		Number of Taxa	88	15	118	54,49	24,561
		Valid N (listwise)	88				
	14	Abundance [ind/m²]	383	100,8000	114804,8000	1919,542559	6,5738873E3
		Number of Taxa	383	9	132	62,16	24,112
		Valid N (listwise)	383				

The following NMS-Analysis is based on 27 taxa-groups and comprises data of three countries (Austria=AT, Germany=DE and the Netherlands=NL). Abundances were (log+1) transformed. The scatter plot (Figure) shows a significant separation of the three countries with a marginal overlap. Also in respect of different eco-regions a separation due to physiographic aspects like altitude or catchment is reflected by the countries. Regarding aspects like hydrological or morphological stressors no explicit clustering of impacted or reference sites could be found.



The following options were selected:
ANALYSIS OPTIONS
1. SORENSEN = Distance measure
2. 3 = Number of axes (max. = 6)
3. 250 = Maximum number of iterations
4. RANDOM = Starting coordinates (random or from file)
5. 1 = Reduction in dimensionality at each cycle
6. 0.20 = Step length (rate of movement toward minimum stress)
7. USE TIME = Random number seeds (use time vs. user-supplied)
8. 50 = Number of runs with real data
9. 0 = Number of runs with randomized data
10. NO = Autopilot
11. 0.000010 = Stability criterion, standard deviations in stress over last 15 iterations.

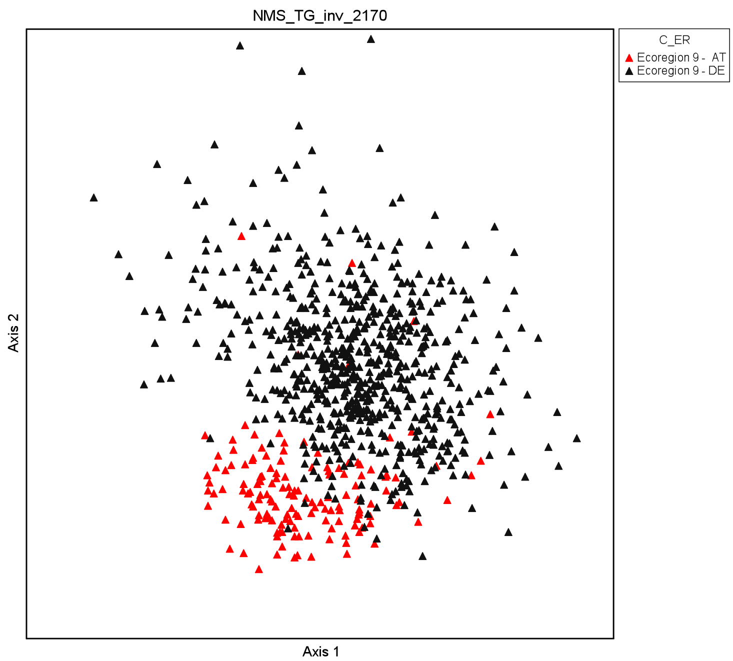
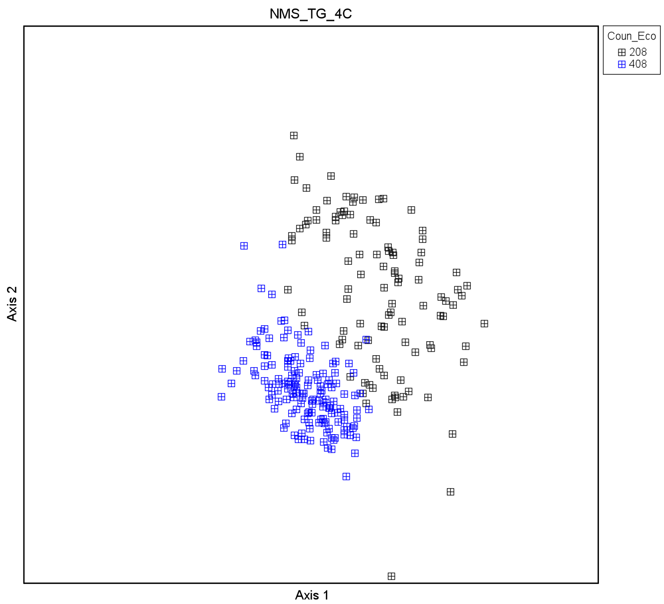


Figure 4: NMS-Scatter plot (axes 1 & 2); 2170 sites, 27 taxa-groups (order level or higher); abundances (log+1) transformed (16.39914 = final stress for 3-dimensional solution; 0.00000 = final instability; 162 = number of iterations); overlay=countries (above); overlay=eco-region 9 - Austria (AT) and Germany (DE) (below).

An obvious separation of countries within the same eco-region is also reflected in Figure 5 where an NMS-Analysis based on 30 taxa-groups comprising data of four countries (Austria, Germany, the Netherlands and France) was done. The fauna of French (western part) and the eastern German part of eco-region 8 (western highlands) shows a considerable split with a narrow overlap.



The following options were selected:

ANALYSIS OPTIONS

1. SORENSEN = Distance measure
2. 3 = Number of axes (max. = 6)
3. 250 = Maximum number of iterations
4. RANDOM = Starting coordinates (random or from file)
5. 1 = Reduction in dimensionality at each cycle
6. 0.20 = Step length (rate of movement toward minimum stress)
7. USE TIME = Random number seeds (use time vs. user-supplied)
8. 50 = Number of runs with real data
9. 0 = Number of runs with randomized data
10. NO = Autopilot
11. 0.000010 = Stability criterion, standard deviations in stress over last 15 iterations.

Figure 5: NMS-Scatter plot (axes 1 & 2); 2507 sites, 30 taxa-groups (order level or higher); abundances (log+1) transformed (15.58275 = final stress for 3-dimensional solution; 0.00006 = final instability; 250 = number of iterations); overlay=eco-region 8 - France (blue) and Germany (black).

These results indicate that the differences of the taxonomical compositions even on a high identification level are enormous between countries and even within the same eco-region so that only detailed analyses of smaller units like river-groups may reflect hydrological and morphological stressors.

An indicator species analysis (ISA) underlines the classification into countries (Table 6).

The analysis based on taxa-groups at family level shows a characteristic dominance of certain taxa-groups per country with high indicator values (IV) for Austria and the Netherlands but not for Germany. The fauna of Austria and the Netherlands is dominated by exclusive family groups. While in Austria mountainous groups of lotic habitats prevail, the benthic invertebrates of the Netherlands are governed by typical lentic organisms of lowlands. Germany shares all surveyed eco-regions and benthic organisms that also occur in the neighbouring countries (and eco-regions) which results in a comparable low IV of reported taxa.

Table 6: Indicator species analysis (ISA); family level; taxa representing indicator values ≥ 50 or top 5 respectively (for DE) and significance p < 0.05.

	Country	Indicator Value (IV)	Mean	S.Dev	p
ELMIDAE	AT	75,2	7,6	0,73	0,0002
EMPIDIDAE	AT	65,7	7,9	0,74	0,0002
LIMONIIDAE	AT	62,5	12,8	0,87	0,0002
HEPTAGENIIDAE	AT	55,2	14	0,89	0,0002
PEDICIIDAE	AT	53,1	15,8	0,9	0,0002
LEUCTRIDAE	AT	52,5	10,8	0,83	0,0002
GAMMARIDAE	DE	33,2	27,2	0,81	0,0002
LIMNEPHILIDAE	DE	26,6	23,7	0,9	0,0058
ERPOBDELLIDAE	DE	24,7	20,7	0,93	0,0012
SERICOSTOMATIDAE	DE	16,1	10,3	0,84	0,0002
EPHEMERIDAE	DE	15	9,5	0,8	0,0002
DYTISCIDAE	NL	57,8	8	0,74	0,0002
HALIPLIDAE	NL	57,4	6	0,66	0,0002
COENAGRIONIDAE	NL	53,1	6,3	0,67	0,0002
ASELLIDAE	NL	52,4	17,4	0,92	0,0002
HYDROPHILIDAE	NL	50	4,9	0,61	0,0002

On the basis of taxonomical composition a significant response to stressors cannot be defined due to a country specific (in case of AT, DE, NL a zoogeographical) and physiographic separation (lowland vs. highland, catchment area etc.) and leads to an assessment of biological traits.

Metrics selection - biased by determination level

A first selection of characteristic metrics was calculated with taxa at the lowest possible determination level (e.g. species level) for each country where samples with abundance less than 100 ind./m² were excluded.

The total number of total taxa is significant lower at German sites because of the poor determination level of groups like Chironomidae and Oligochaeta primarily which is also reflected for eco-regions (8 and 9) dominated by German sites (Figure 6).

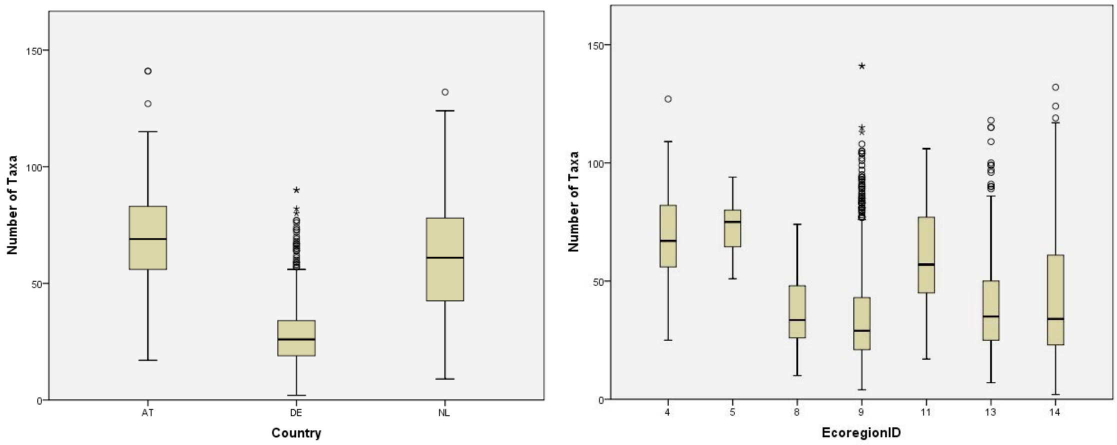


Figure 6: Number of taxa per country (left) and eco-region (according to Illies 1978, right).

Metrics governed by physiographic characteristics (eco-regions)

There is a clear observable gradient along geographic latitude in Europe regarding altitude, slope and water temperature which has led to the implementation of smaller biological units like eco- and bioregions across Europe within the Water framework Directive (WFD). The benthic fauna and therefore many metrics reflect these significant physiographic differences even on a rough taxonomic level and underlines the importance of these parameters for the distribution of freshwater invertebrates.

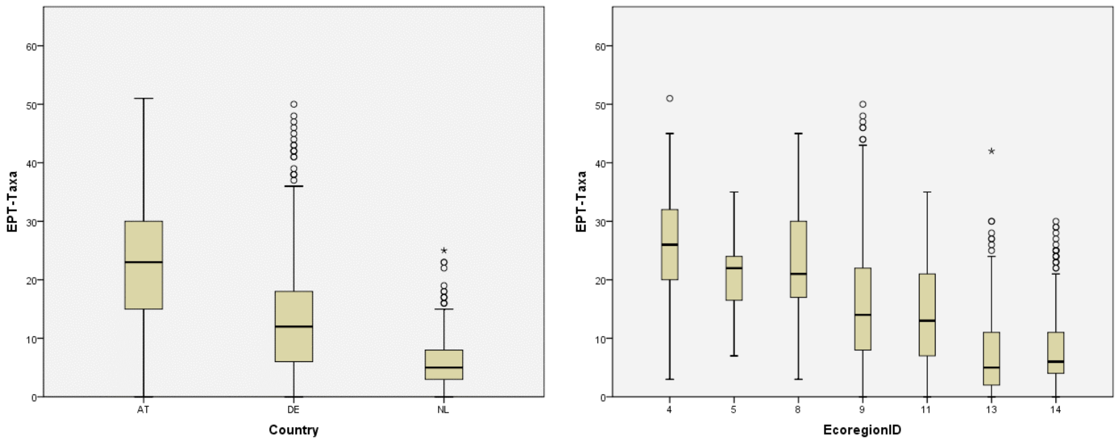


Figure 7: Number of EPT- taxa per country (left) and eco-region (right).

To illustrate this fact the sensitive group of Ephemeroptera-, Plecoptera- and Trichoptera (EPT)-taxa which is determined at a comparable level for all countries shows a numerical decrease from Austria to the Netherlands as well as from alpine to lowland eco-regions (Figure 7).

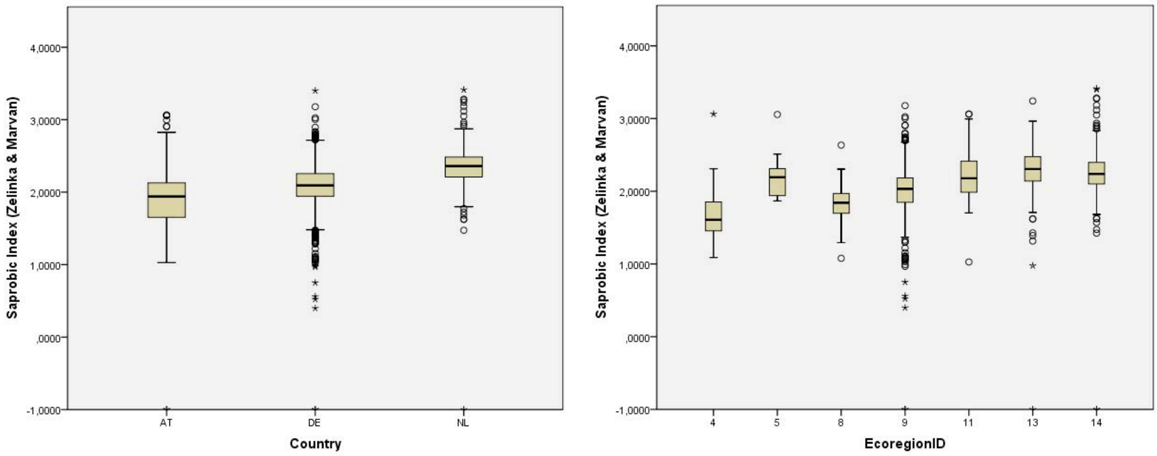


Fig. 8: Saprobic index per country (left) and eco-region (right).

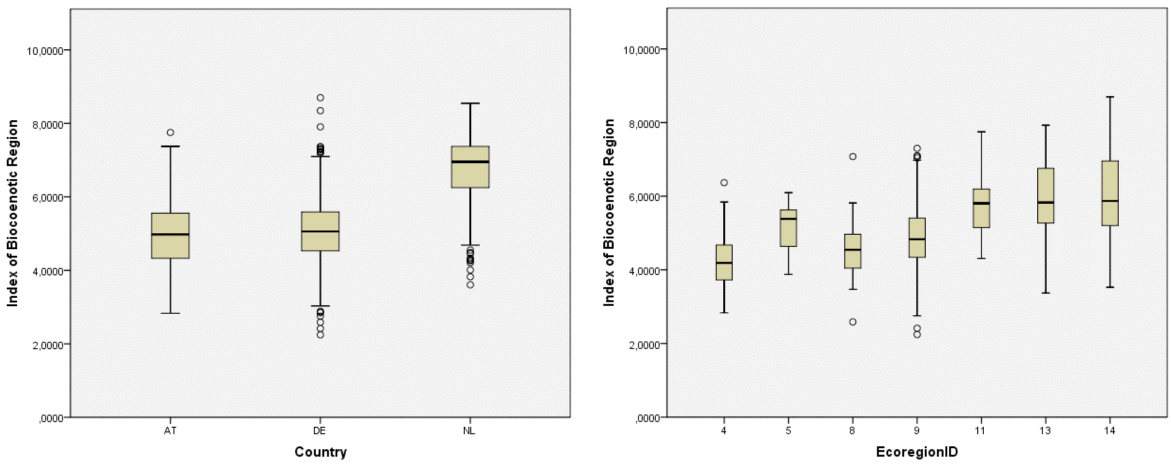


Figure 9: Index of biocoenotic region per country (left) and eco-region (right).

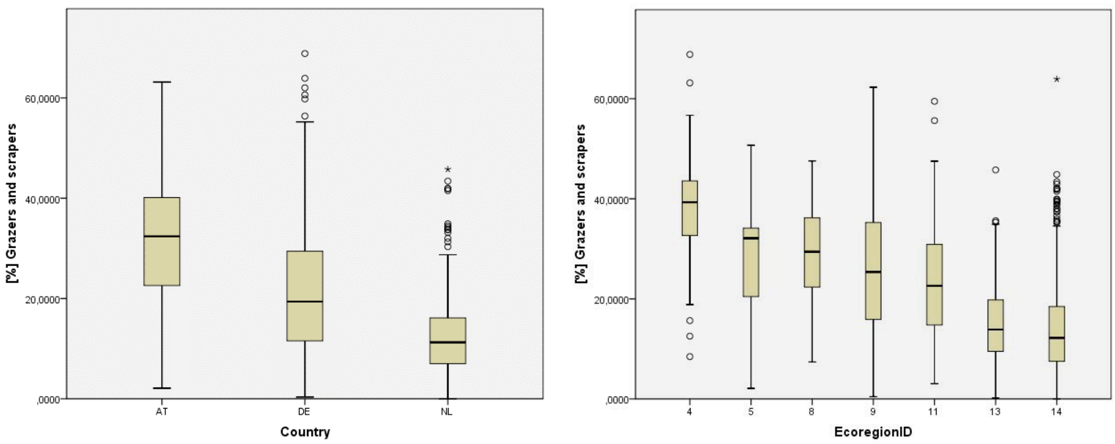


Fig. 10: Proportion of grazers & scrapers per country (left) and ecoregion (right).

Other general trends visible across the overall European dataset of WISER show e.g. an increase by trend of the saprobic index (Zelinka & Marvan) and as well as an increase of the index of biocoenotic region from South (Austria) to the North (the Netherlands) which can also be recognised with the increase of the Eco-region ID (Figure 8).

As an example for species traits the functional feeding guilds of grazers and scrapers show a significant metric reaction. Its share within the benthic assemblages decreases from Austria to the Netherlands (from alpine to lowland eco-regions) (Figure 9 and 10).

<u>Conclusion of large scale stressor-sensitive metrics</u>

Despite the evidence of the above mentioned inhomogenities, a large scale stressor specific metric analysis was done for sites shared by fish data to compare the metric response of the two biological quality elements.

On the basis of logistic regression and Pearson correlation a definition of candidate metrics for each hydrological and morphological stressor predicated on a country specific determination level was done.

A separation into river-groups or eco-region was not possible due to the low number of sites with a sufficient declaration of stressors. Especially hydrological stressors are rarely documented and only comprise 36 impounded sites and 16 sites impacted by hydro-peaking and water abstraction respectively. The random number is too low after a further splitting.

In general meaningful metrics could be listed and the response (increase or decrease) for each metric is given. For all hydrological stressors and for the morphological stressors “artificial embankment” and “riparian vegetation modified” significant metrics could be detected. The other morphological stressors show a large number of metrics, which respond only by trend or are not meaningful due to marginal values.

The appliance indication (good, restricted, vague) in Table 7 reflects the accordance of the classification into impacted and not impacted sites. When the predicted classification in the process of the logistic regression confirms the observed classification, 100% of the sites are correctly classified so that the stressor is indicated by certain metrics. If the accordance is less than 100% some of the impacted sites are not recognized as such in respect of the applied metrics. Stressors obtaining values between 90 and 100% display at least three significant metrics (good). At values between 70 and 90% a metric response is given only by trend (restricted). Stressors obtaining values less than 70% should not be used for metric selection because the response is too low and the effect of the stressor is not observable (vague).

In case of the stressors “cross section modified” and “velocity increase” no measurable responses between reference and impacted site could be computed so that these stressors should not be applied for an overall assessment.

Table 7: Stressor specific candidate metrics of invertebrates: metric response: ↑=increase; ↓=decrease; blue arrows: results from the logistic regression, black arrows: additional metrics reflecting Pearson correlation > 0.5 and significance at the 0.01 level (2-tailed).

Number of sites	36	16	16	292	290	369	204	164	194
Percentage correctly classified	100	90,9	90,9	100	100	85,4	73	65,1	54,4
Viability	**good**	**good**	**good**	**good**	**good**	**restricted**	**restricted**	**vague**	**vague**
	Impound-ment	Hydro-peaking	Water Abstraction	Artificial Embankment	Riparian Vegetation Modified	Instream Habitat Modified	Channel Form Modified	Cross Section Modified	Velocity Increase
EPT – taxa, composition	↓								
Plecoptera taxa, composition	↓			↓					
Oligochaeta & Diptera /Total-Taxa, composition		↑	↑			↑			
Leuctra & Calopteryx [ind/m²], composition	↓								
Number of sensitive taxa [Austria], composition /function	↓				↓				
Number of total taxa, composition						↑			
Xenosaprobic ratio [scored taxa = 100], function	↓			↓	↓				
Epipotamal ratio [scored taxa = 100], function	↑								
[%] littoral + profundal, function	↑								
Xylophagous, shredders, active & passive filterfeeders, function		↓	↓						
Passive filter feeders [%], function	↓	↓	↓	↓	↓				
Gatherers/Collectors [%], function		↑	↑						
Stone dwelling taxa, function	↓								
Rhithron Typie Index, function	↓								
SPEAR organic, function		↑	↑						
Number of (additional) metrics with response given by trend only or insufficient data, various				**4**	**3**	**15**	**8**	**27**	**31**
Number of candidate metrics	**10**	**5**	**5**	**3**	**3**	**2**	**-**	**-**	**-**

Impounded sites with a number of 10 candidate metrics show a clear response and can be assessed precisely. These results can be confirmed by previous studies from Austrian rivers of different sizes where the impact of impoundments in dependency of the distance to the weir was analysed (Ofenböck et al. 2011). For example river Traun indicates a good status at the free flowing section (headrace) and poor or bad status at all following transects in the impounded section. The response of the Austrian Multimetric Index for the river Traun is based on the calculation of metrics (e.g. EPT-taxa, filter feeders, littoral and profundal ratio) used in the Austrian National Method and are also defined as candidate metrics in Table 8. This example illustrates the small scale reaction to stressors of macro invertebrates, and underlines the importance of precise abiotic conditions at the site as prerequisite for further analyses.

Table 8: Results of the detailed method of the impoundment Pucking at river Traun including selected environmental factors and metrics.

Saprobic index (Zelinka & Marvan)*	2,18	2,21	2,14	2,26	2,9	2,72	
ECOLOGICAL STATUS (detailed MZB-method)	**good**	**poor**	**poor**	**bad**	**bad**	**bad**	
Environmental characteristics	*flow direction*						
	Headrace	Impoundment				Before weir	
Bed sediment	**Gravel/Pebbles**				**Mud/Sand**	**Mud**	
Current velocity [m/s]	**0,4**	**0,3**	**0,2**	**0,16**	**0,11**	**0,05**	
Depths [m]	**3,6**	**5,6**	**8,8**	**10,2**	**10,1**	**12,8**	**WEIR**
Distance to weir [m]	**10.000**	**8.000**	**6.000**	**4.000**	**2.000**	**200**	
Biological metrics							
Species richness (total # taxa)	**105,00**	**79,00**	**63,00**	**71,00**	**41,00**	**30,00**	
EPT-taxa*	**23,00**	**11,00**	**10,00**	**4,00**	**0,00**	**1,00**	
% EPT-taxa*	**29,87**	**18,97**	**21,28**	**7,69**	**0,00**	**5,00**	
Litoral & profundal*	**1,49**	**2,39**	**2,87**	**3,82**	**3,81**	**4,14**	
Grazers & shredders quotient (RETI)	**0,58**	**0,22**	**0,14**	**0,09**	**0,00**	**0,01**	
Filtering & gathering collectors quotient (PETI)	**0,42**	**0,78**	**0,86**	**0,91**	**1,00**	**0,99**	
Degradation index*	**44,00**	**0,00**	**0,00**	**0,00**	**0,00**	**0,00**	

*Applied metrics for the calculation of the Multimetric Index for the mentioned rivertype (according to the Austrian method – Bioregion: large alpine rivers; saprobic basic condition=1,75; inner differentiation=Traun)

Hydropeaking and water abstraction are stressors always occurring in combination at the selected sites and therefore are reflected by the same candidate metrics. Functional feeding guilds and the ratio of Oligochaeta and Diptera taxa show significant responses (Table 8). The morphological stressors are indicated by only a small number of metrics, which only give a trend of response in many cases. The combination of these stressors within the dataset may seriously bios the assessment as these impacts may drive the macro-invertebrate reaction in opposite directions.

Modified instream habitats for example can also have a positive effect on the fauna due to different physiographic regions of not impacted and impacted site or due to impacts of other stressors like modified channel forms or cross sections where an instream habitat can provide an attractive alternative for some taxa. This may define the increase of the total number of taxa in Table 7.

In respect to the most frequently responding metric “percentage of passive filter feeders” a considerable reaction along the hydrological, morphological and hydro-morphological index (compare chapter fish assessment) can be noticed. Anyhow, as shown in Table 7 and Figures 11 and 12 the type of stressor is crucial not the total number of stressors. Not impacted sites (HyMo index =0) show maximum values at least by trend.

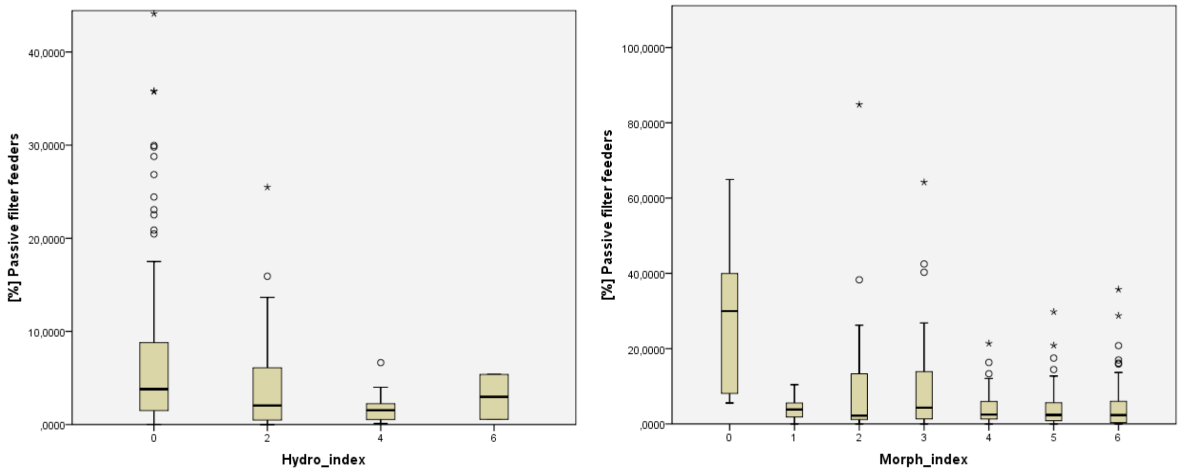


Figure 11: Metric response along hydrological (left) & morphological (right) index; percentage of passive filter feeders.

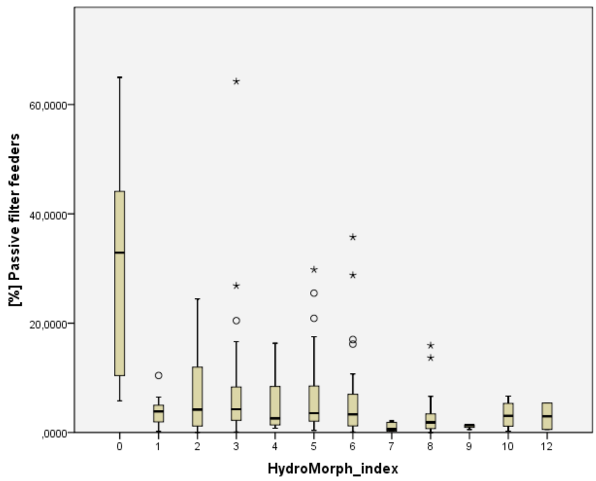


Figure 12: Metric response along hydro-morphological gradient (index); percentage of passive filter feeders.

In general a metric response at the large-scale should not be overestimated. Reference and impacted sites are frequently located in different and non-comparable catchments, altitudes or eco-regions. An analysis including all invertebrate data separated into river-groups, eco-regions or even smaller scale is prerequisite for a sound assessment system.

Fish species analyses and metric selection

The spectrum of the 300 tested metrics comprised overall metrics and specific guild-based metrics (Melcher et al. in prep., Holzer 2008). Five functional metrics groups were considered to water quality, feeding, reproduction, habitat and migration (Oberdorff et al. 2001, Melcher 2007). Tolerant and intolerant groups reflect species sensitivity to any common impact related to altered flow regime, nutrient regime, habitat structure and water chemistry (Karr et al. 1986). Loss of intolerant species is a response to degradation, whereas the number of tolerant species will tend to increase with disturbance (Pont et al. 2006) (Table 9).

Tab. 9: Overview of used fish metrics and their abbreviations (Holzer 2009).

Guild	Functional metric group	Number of species (n.sp)	Relative number of species (perc.sp)	Total density (ind/ha) (n.ha)	Relative density (%) (perc.nha)
Overall Composition	All (all)	n.sp.all		n.ha.all	
Tolerance (Tol)	Intolerant (intol)	n.sp.intol	perc.sp.intol	n.ha.intol	perc.nha.intol
	Tolerant (tol)	n.sp.tol	perc.sp.tol	n.ha.tol	perc.nha.tol
Habitat (Hab)	Water Column (wc)	n.sp.Hab.wc	perc.sp.Hab.wc	n.ha.Hab.wc	perc.nha.Hab.wc
	Benthic (b)	n.sp.Hab.b	perc.sp.Hab.b	n.ha.Hab.b	perc.nha.Hab.b
	Rheophilic (rh)	n.sp.Hab.rh	perc.sp.Hab.rh	n.ha.Hab.rh	perc.nha.Hab.rh
	Limnophilic (li)	n.sp.Hab.li	perc.sp.Hab.li	n.ha.Hab.li	perc.nha.Hab.li
	Eurytopic (eury)	n.sp.Hab.eury	perc.sp.Hab.eury	n.ha.Hab.eury	perc.nha.Hab.eury
Reproduction (Re)	Lithophilic (lith)	n.sp.Re.lith	perc.sp.Re.lith	n.ha.Re.lith	perc.nha.Re.lith
	Phytophilic (phyt)	n.sp.Re.phyt	perc.sp.Re.phyt	n.ha.Re.phyt	perc.nha.Re.phyt
Longevity (Lon)	Long lived (ll)	n.sp.Lon.ll	perc.sp.Lon.ll	n.ha.Lon.ll	perc.nha.Lon.ll
	Short lived (sl)	n.sp.Lon.sl	perc.sp.Lon.sl	n.ha.Lon.sl	perc.nha.Lon.sl
Feeding (Fe)	Piscivorous (pisc)	n.sp.Fe.pisc	perc.sp.Fe.pisc	n.ha.Fe.pisc	perc.nha.Fe.pisc
	Insectivorous/ Invertivorous (insev)	n.sp.Fe.insev	perc.sp.Fe.insev	n.ha.Fe.insev	perc.nha.Fe.insev
	Omnivorous (omni)	n.sp.Fe.omni	perc.sp.Fe.omni	n.ha.Fe.omni	perc.nha.Fe.omni
Migration (Mi)	Long distance (long)	n.sp.Mi.long	perc.sp.Mi.long	n.ha.Mi.long	perc.nha.Mi.long
	Potamodrom (potad)	n.sp.Mi.potad	perc.sp.Mi.potad	n.ha.Mi.potad	perc.nha.Mi.potad

Values for each metric per site and date were computed systematically by a software routine. All metrics were log - transformed [log (x+1)]. We pre-selected metrics by eliminating metrics with insignificant pressure response using Spearman‘s rank correlation ($p < 0.05$).

An important task in discriminating groups is finding a set of metrics that leads to an optimal differentiation and keeps the number of variables low to avoid over fitting. Stepwise logistic regression analyses were performed for each of the river-groups to select metrics best discriminating between impacted and less impacted stations. Thereby, the selection of variables is based on the significant F-test from an analysis of covariance. Variables already chosen, act as covariates and the new variable is used as the dependent one (SPSS 2003).

In total 100 metrics out of more than 300 calculated fish metrics showed significant ($p < 0.05$, Spearman's rank correlation) responses to hydro-morphological pressures. Finally, 17 different metrics were selected from 25 logistic regression analysis models (one model for each river-group and each pressure type separately). The number of metrics per European River-group varied from one to five (Table 10). "Number of lithophilic and potamodromous species" were the most common metric across all models and were used in all 5 European River-groups. Four other metrics, i.e. "density of rheophilic species", "density of habitat degradation intolerant and intermediate species", "number of species tolerant to the loss of oxygen" and "intolerant to water toxicity" were used in two river-groups. Four of the 17 selected metrics were specific to individual river-groups and pressures. All water quality related metrics occurred in lower or warmer parts of the waters with a higher number of species (mainly river-groups D and E)

Table 10: Frequency of significant fish metrics used per guild category and functional metric group used for response to hydro-morphological pressures like impoundment (imp), hydro-peaking (peak), water abstraction(abst), riparian vegetation modified (modi) and artificial embankment (arti). European river-groups are given in capital letters.

			Pressure					
Trait	**Functional metric group**	**metric**	**imp**	**peak**	**abst**	**modi**	**arti**	sum
Habitat guild	rheophilic	density		A,B	A,B			4
Reproductive guild	lithophilic	n species	E	A,D	A	A,D		6
Habitat spawning preferences	rheopar	n species		A	A			2
	intermediate	n species				B,C	A,C	4
Habitat degradation tolerance	intolerant	density	C,D	D				3
	intolerant	n species				D	C,D	3
	potamodromous	n species	C,D	A	A	B		5
Migration guild	long catadromous	density	E				D	2
	long anadromous	density					C	1
Water quality tolerance oxygen	tolerant	n species		D			C,D	3
Water quality tolerance toxicity	intolerant	density			D,E		D	3
	tolerant	density	D		D			2
Water quality tolerance general	tolerant	density				E		1

Table 10 shows the most frequently selected functional groups, i.e. rheophilic and potamodromous species (11 times used) followed by rheophilic species (4 times). Reproductive and habitat guilds were used most often for upper river sections and water quality traits were used only in the lower sections. Metrics responded in most cases in one direction only, i.e. consistently either increased (water quality) or decreased (habitat and migration) in all pressure types. However, metrics referring to migration, and loss of habitat decreased with the intensity of disturbance also (Fig. 13 and 14).

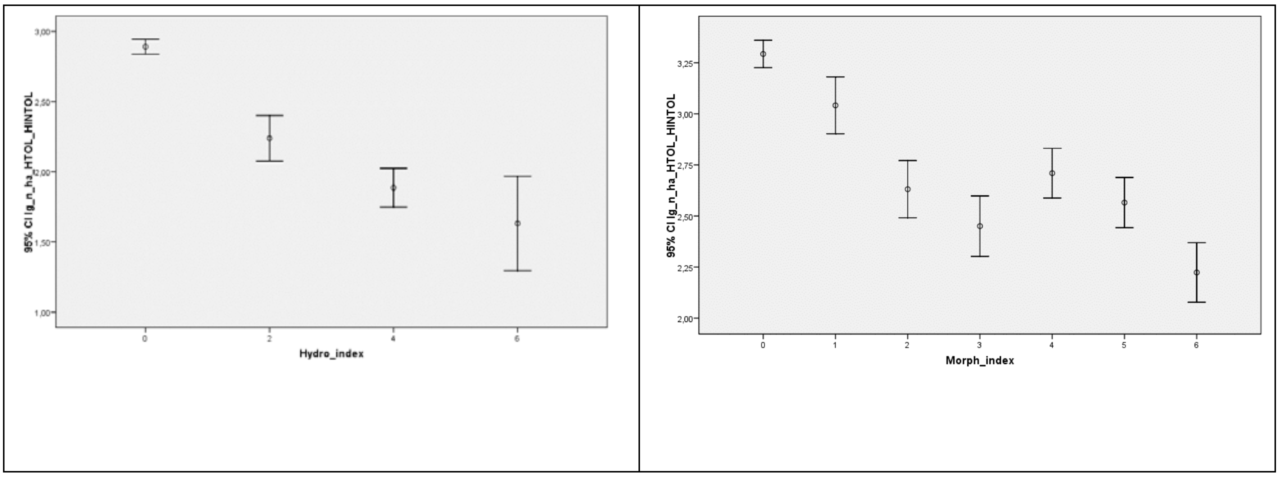


Fig. 13: Response of the frequent fish metric habitat degradation tolerance (density of intolerant species) to hydrological and morphological pressure intensity.

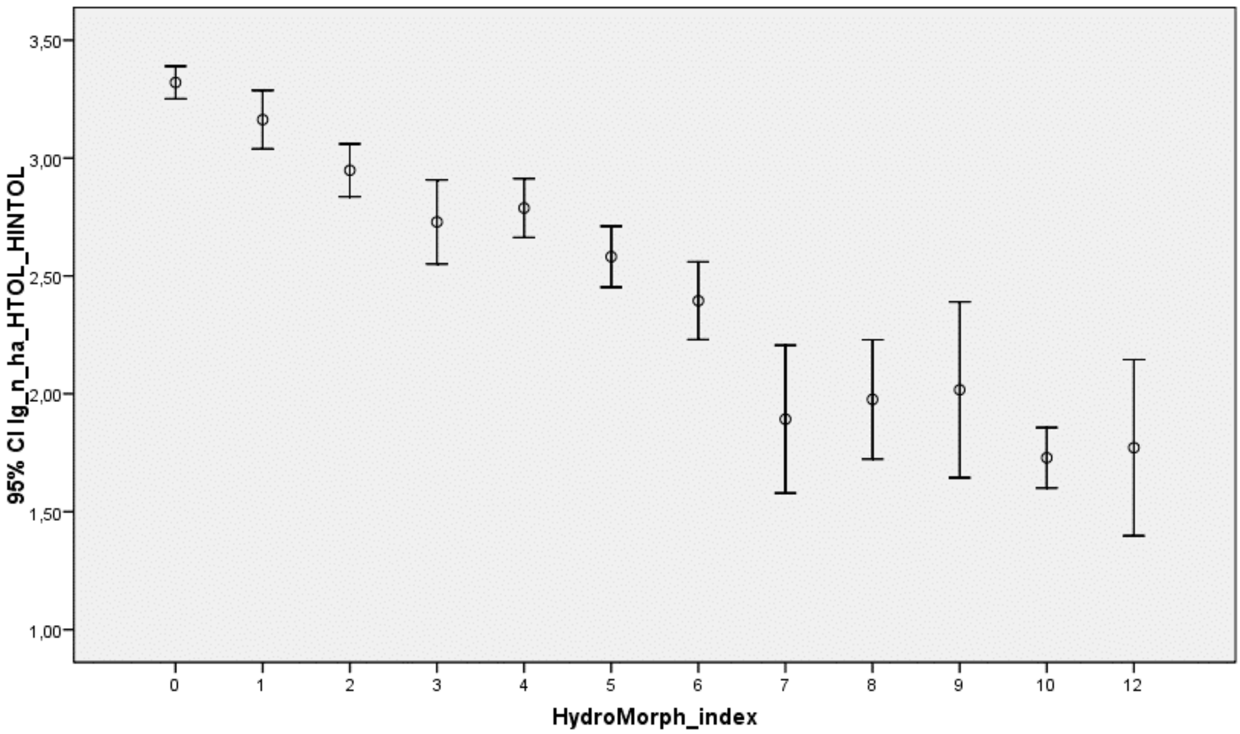


Fig. 14: Response of the frequent fish metric habitat degradation tolerance (density of intolerant species) to hydro-morphological pressure intensity.

Conclusion

In this study a river typology based on environmental descriptors and standardised metrics of biotic integrity at the European scale have been developed to be in accordance with the Water Framework Directive. At present, the approach applies only to rivers belonging to five European river-groups defined in the present work. To extend the geographic range of the method and to include new benthic invertebrates and fish assemblage types additional data have to be collected and the results validated.

The results found in the WISER dataset regarding the invertebrates are largely congruent with findings of Verdonshot (2006), a study dealing with comparable large-scale data. Verdonshot (2006) found that climate (temperature), slope (current velocity) and stream size are the main typological descriptor of benthic macro-invertebrate assemblages and that human stress diminished the natural differences between stream communities and therefore he concluded that reference sites should be used only to generate a sound typological approach. Our analyses of macro-invertebrates of rivers and streams sections covered the eco-regions 4, 5, 8, 9, 11, 13 and 14 (according to Illies, 1978) and showed a clear separation regarding taxonomic structure along a latitudinal gradient as well as along eco-regional units. On basis of a quite rough taxonomical level (order or higher taxonomic units) even a clear divergence between eco-regions was stated indicating delicate differences within eco-regions. The fauna of the French (western part) and the eastern German part of eco-region 8 (western highlands) shows a considerable split and the southern (Austrian) part of eco-region 9 is clearly different to the northern (German) part. In conclusion the eco-regional approach works as rough typological descriptor, but might be too imprecise to reflect faunal inhomogeneities at a small scale. Verdonshot (2006) found sound sub-divisions of eco-regions in his macro-invertebrate analyses, which justify the bioregion-concept as entities realised in many European countries. Despite these results we found a small number of metrics/species traits, which responded to various stressors at a large scale. Nevertheless metric respond was strongest in smaller geographical scales (e.g. Austria) and weakest on the combined, total dataset including sites in Austria, Germany and the Netherlands. Therefore a further separation in smaller units using approaches like river-groupings or those of Hering et al. (2006) and Verdonshot (2006) or national typologies is therefore strongly recommended to make the results more reliable.

In contradiction to Verdonshot's results we found a considerable decrease in diversity along the latitudinal and longitudinal gradient. Especially the distribution of EPT-taxa reflects the physiographic change from alpine to lowland areas. On the other hand stagnophilic groups like odonats increase in diversity towards the lowlands. However, analyses of overall diversity are severely hampered and biased by the lack of taxonomic information on larval stages of macro-invertebrates and the differing level of the national assessment system and its level of taxonomical resolution. Although traits and functional indices may be rather used than faunal composition the taxonomical resolution between sites has to be comparable.

The use of different biological quality elements for different stressors and on different scales is described in several studies. Hering et al. (2006) stated that with the exception of diatoms all organism groups respond to hydro-morphological degradation and the selection of the most appropriate organism group depends on the stream type. Fish should be considered at the reach scale and at the meso-scale for monitoring the effects of hydro-morphological degradation (Johnson et al. 2006, Bain et al. 1988). Effects at the habitat scale are best reflected by macro-invertebrates (Hering et al. 2006). In multiple stressed streams and rivers the use of macro-invertebrates is recommended by Doledéc et al. (1999) and Statzner et al. (2001) because of well known species traits of benthic invertebrates. As fish are sensitive at different scales and also to different stressors than macro-invertebrates a parallel reaction of these biological quality elements is not likely to be detected. Although macro-invertebrates are able to describe global stressors at a large catchment scale (Feld & Hering 2007) they perform the best on a micro-scale (habitat) when broad scale stressors (habitat deterioration at catchment scale) have already been affecting the community since long time. In Central Europe this is likely to have happened in all regions below a certain altitude where agriculture is cost-effective. To evaluate the macro-invertebrate stressor-specific reaction it is therefore crucial to have detailed information on the intensity of a stressor and to include the river-type specific faunal composition of a reference site in the analyses. Within the huge WISER-dataset there is a serious lack of data regarding the above-mentioned parameters and the restricted number of abiotic site-specific information affected the interpretation considerably. Hence, the data did not fulfil the prerequisites to describe different gradients of hydro-morphological pressures from natural reference sites to strongly impacted sites.

On a large scale the effect of stressors is bound to a small group of metrics, which show a significant response. The allocation of candidate metrics can be defined more reliable for hydrological stressors than for morphological stressors.

The concept of spatially-based assessment methods is to test fish metrics response for each identified river group (Melcher et al 2007). Fish metrics were calculated systematically for defined ecological traits and their functional groups. This study showed that number and type of in total 13 selected metrics, using logistic regression, differed among river-groups and pressure types. The number of metrics ranged from two to five metrics per model. Most important and frequent metrics traits were those, who discriminate between impacted and not impacted stations best for reproduction, loss of habitat, migration and water quality (especially for lowland river reaches). The proposed standardised procedure provided the opportunity to examine the relative importance of metric types.

Ongoing studies on the Wiser-dataset will include the following steps: testing the ecological meaningful geographical entities which can be combined for large-scale analyses, filtering out sites with reliable hydro-morphological data, a rigorous check of candidate metrics as well as metrics used in the national assessment systems. As the number of sites with data on fish and macro-invertebrates is in many cases too low for statistical sound analysis, we will focus on sites with macro-invertebrate information only for which a definition of river-groups basing on physiographic parameters will be done.

Finally we can conclude that:

- both biotic data and physiographic characterisation point up a north-south differentiation.
- a type-specific (ecoregion-, and river-group) metric reaction is given by trend
- in respect to the most frequently responding metric “percentage of passive filter feeders” a considerable reaction along the hydrological, morphological and hydro-morphological index can be noticed.
- The reaction of macroinvertebrates is more pronounced regarding hydrological stressors than morphological ones.
- Fish metrics show significant response to specific pressures.
- Loss of habitat, migration and reproduction related metric performed best for fish.
- Fish metrics react due to fish zonations (river-groups).
- In terms of scale-specific response prioritisation of measures with large scale effects should be favoured. Focus should be given to less impacted catchments at first. Largest restoration effects are expected to occur there.
- Development of programmes of measures from (management plans) “large to the small” is necessary, i.e. from catchment (sub-catchment) to the local scale.

CHAPTER III

Response-Recovery chains: restoration and management

Analysing biota's response to river restoration in light of broad-scale land use and physical habitat modification

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Abstract

River restoration is a central issue of present-day improvement of the environment. Local municipalities or regional councils restore stretches of rivers, which flow through their district. Nonetheless, even on the level of municipalities many ownership and administrative constraints hamper the restoration of long stretches of rivers. Thus, the average length of restored river reaches is normally much shorter than 1 km but the total costs for purchase of land and restructuring of the river channel are very high. Similarly, the expectations on the ecological effects are commonly very high (Jähnig et al. 2011). But, investigations of restoration measures showed that the ecological effects are often far smaller if present at all (Palmer et al. 2010). Causes are hypothesized in missing habitat heterogeneity and potential catchment constraints. Nonetheless, a general picture is rarely drawn because the great majority of investigations remain on the case study level. But single case studies might neither understand the overall influences nor be applicable to define general pattern. Therefore, the actual analysis tries to fill this gap by an analysis of biological and catchment information on 47 restoration measures. A database was compiled comprising biological data of three different organism groups (fish, macroinvertebrates and macrophytes) sampled with standardized methods. Furthermore, an evaluation of the river habitat quality and land use in the catchment was conducted. Additionally, close to the restored river reaches an unrestored reaches was sampled for comparison. Thus, with these data insights can be detected on the biological response and the catchment constraints to restoration actions. The results show that isolated restored river reaches are not really isolated in the common sense. Catchment constraints and positive influences either foster or hamper the reshaped local reach and alter ecological effects in many ways:

- Local restoration measures do not always lead to positive responses of the biota:
  - there are no significant differences in the response of the stream biota to the type of restoration measure conducted, i.e. the money spend on a reach
  - the panacea for restoration on a local reach with the intension to improve all biological organism groups is not yet found
  - but: 'larger', more extensive measures, which improve the in-stream habitat heterogeneity and restore the overall river channel patterns have a higher chance in improving fish, macroinvertebrates and macrophytes at the same time

- Biological responses are dependent on the ecological status before the restoration:
    - restoration of reaches which are ecologically in a poor status results in the highest biological response in macroinvertebrates and macrophytes
    - restoration of good and very good reaches is followed by a deterioration in macroinvertebrates and macrophytes
    - fishes respond positive to local restoration measure irrespective of the ecological quality before the restoration
- The more natural the land use and river habitat upstream – the higher the chance of a good ecological quality in restored reaches:
    - fishes and macroinvertebrates are fostered by deciduous forest along the river which provides habitat and food while macrophytes are fostered by arable land
    - restoration measures should at least target an average river habitat assessment score of 4.5 (German river habitat survey assessment system) on long upstream sections to affect positively the biota
    - fishes are strongly influenced by the upstream river habitat quality, the land use in buffers as well as the land use in the catchment
    - macroinvertebrates are mainly influenced by a the river habitat quality in a comparatively short stretch (1,000 m) upstream
    - macrophytes are fostered by the river habitat quality upstream in long stretches (up to 7,500 m) by e.g. providing source populations which might establish in the restored reaches
- Money should be spend wisely: more on buffer improvements upstream than on reach brilliance

Introduction

Since a decade restoration ecology is one of the hot topics in aquatic ecology. A large body of literature addresses case studies all over the world on the reaction of aquatic organism groups to morphological improvements. The reported effects range from very positive effects to no or even negative reaction of individual organism groups (see Bernhardt et al. 2005 for a review). Most studies address only a certain reach of a river and only one organism group. Comparative studies of different organism groups are rare and studies concerning many restoration measures and several organism groups are nearly non-existent. Nonetheless, these studies are becoming more and more necessary to obtain a general picture on the biotic response to morphological restorations. Based on the EU Water Framework Directive (WFD, 2000/60/EC) there is an urgent need for general recommendations on the potential chances and possibilities of restorations measures to improve a water body. Furthermore, since many references reported less improvement than expected the search for causes has started. Hypotheses rank around missing source populations of target species to migration barriers and an overarching effect of the catchment.

The latter parameter will be in the focus of this reach. Based on an evaluation of the land use and the stream morphology in different spatial scales we will answer the following key questions:

- Does the type of restoration measure determine the level of response of the biota?
- Do restoration measures improve fish, macroinvertebrates and macrophytes at the same time?
- Is the ecological quality of restored reaches dependent from the land use and river habitat quality upstream?
- Does the biological response depend on the ecological quality class of the unrestored reaches?
- Are break points in land use and river habitat quality detectable which separate ecologically good reaches from poor or bad ones?

Material and methods

Database

We analysed data from 47 river restoration measures (Annex 4**Fehler! Verweisquelle konnte nicht gefunden werden.**). The measures were reach-scale restorations conducted in the German Federal States of North Rhine-Westphalia, Hesse, Rhineland-Palatinate, Lower Saxony and Thuringia. The rivers had overall catchment areas between 9 and 52,880 km^2 and were located either in the German lowlands (ecoregion 14 following Illies 1978) or in the lower mountainous areas (ecoregion 9).

The physical restoration at the sites comprised the removal of weirs, the removal of bank fixations as well as the creation of backwaters or the creation of multiple-channel or meandering patterns. At some sites only the maintenance of the bank and bed fixations was stopped, which is called passive restoration.

At 41 sites the fish fauna was sampled according to the German standard method (Dussling et al. 2004), which is an EU-WFD compliant method. The sampling was conducted in late summer by wading using a back-pack electroshocking device or by boot using a generator-driven electroshocking gear. All fish were recorded and their length measured.

At 43 sites the macroinvertebrate fauna was sampled according the German standard method (Meier et al. 2006), which is also an EU-WFD compliant method mainly developed in the EU-funded projects AQEM and STAR. Streams with catchment sizes below 100 km^2 were sampled in spring (March and April) and all other rivers in summer (June and July). Sorting and identification was done in the lab and species or genus level was envisaged for all groups except for Chironomidae and Oligochaeta. The numbers of all species were recorded.

Macrophytes were sampled at 43 sites according to the EU WFD compliant method developed by Schaumburg et al. (2004, 2005) for Germany. Sampling took place in the summer months and all species were identified on site on species level except for some mosses, which were identified in the lab. The abundance of the species was recorded according to the Kohler scale (Kohler 1978).

In addition to each restored reach an unrestored reach a few hundred meters upstream of the restored reach was sampled with the same methods. This unrestored control reach resembles the restored reach before the restoration.

The samplings were mainly conducted in a project funded by the DBU and the state of Hesse in the years 2007 and 2008. Additional datasets derive from selected sampling by the staff of the University of Duisburg-Essen and the Planungsbüro Koenzen (planning enterprise).

Metrics

The fish data were imported to the fiBS program (Dussling et al. 2007), which calculates the Ecological quality class for each taxa list based on river specific reference lists. Furthermore, about 50 additional metrics were calculated by the program.

The macroinvertebrate taxalists were imported to the ASTERICS program (www.fliessgewaesserbewertung.de), which calculates the Ecological quality class and about 200 additional metrics.

The ecological quality class for the macrophytes was calculated with the Phylib program (Bayerisches Landesamt für Umwelt 2006) and some additional metrics were calculated by hand.

There were no significant correlations between the years since the restoration measures have been conducted and the response of any metric; i.e. there is no time effect discernable in the data set.

Land use and river habitat survey

The land use was evaluated on the meso-scale in 10 different buffer reaches upstream of the restored sites. Two different buffers width were chosen: 50 m and 100 m (each side of the stream) and 5 different lengths: 0.5 km, 1 km, 2.5 km, 5 km and 10 km.

Furthermore, the land use was evaluated on the whole catchment upstream of the restored sites.

For each buffer of each restored site the percentages of the land use were derived from the CORINE land cover data (CORNE 2000) via a GIS system. For the analysis the land use variables were summarized under the following categories: urban areas, pasture, cropland, deciduous forest and coniferous forest. For some analyses the latter two were additionally summarized to deciduous ('dec') and coniferous ('coni') forest.

The hydromorphological quality was evaluated by the German standard river habitat survey (LAWA 2000, LUA 1998, 2001). For many German rivers this river habitat survey has been conducted by governmental environment agencies in recent years. The rivers have been divided into reaches of 100 m and about 100 hydromorphological parameter had been recorded for each 100 m reach and used for an assessment of the habitat quality. The assessment system is based on stream-type specific reference conditions and gives a final value between 1 (reference) and 7 (totally impaired).

The river habitat quality was evaluated on the same length categories upstream of the restored sites as the riparian land use: 0.5 km, 1 km, 2.5 km, 5 km and 10 km.

Types of restoration measures and their effects on aquatic organism groups

The first analysis focuses on the comparison between different types of restoration measures and their effects on aquatic biota. Hence, the measures have been related to the response of the biota to find out if certain restoration measures have a stronger effect than others. Furthermore, the responses of the individual restoration types have been related to catchment constraints represented by the land use and river habitat survey results in different spatial scales.

Data evaluation

The analysis was sequenced in the following steps:

1. Cluster analysis of the conducted physical improvements to identify groups of restored sites with similar restoration actions.

2. Calculation of the biotic response.

3. Averaging the biotic response for each cluster group.

4. Correlation of the average responses to the land use and habitat survey parameter.

Results

Step 1

A two-way-cluster analysis was performed (Euclidean distance; Ward's linkage method) on a matrix of the conducted measures (Figure 1). The resulting figure shows six different cluster groups, i.e. six different types of restoration measures (from green on the left to blue on the right, Figure 1). In each cluster group the measures, which have been conducted are relatively equal. Furthermore, the number of different measures conducted on a single reach increase from left to right. Each cluster group can be characterized by certain individual measures (from left to right):

- passive restoration with extensification of land use (green cluster group)
- reconnection of backwaters, mainly short measures (red)
- elongation of the river length by remeandering (black)
- weir removal, remeandering (yellow)
- creation of multiple channel pattern, creating a new water course (turquoise)

- placement of flow deflectors, wood placement, creation of multiple channel pattern (blue)

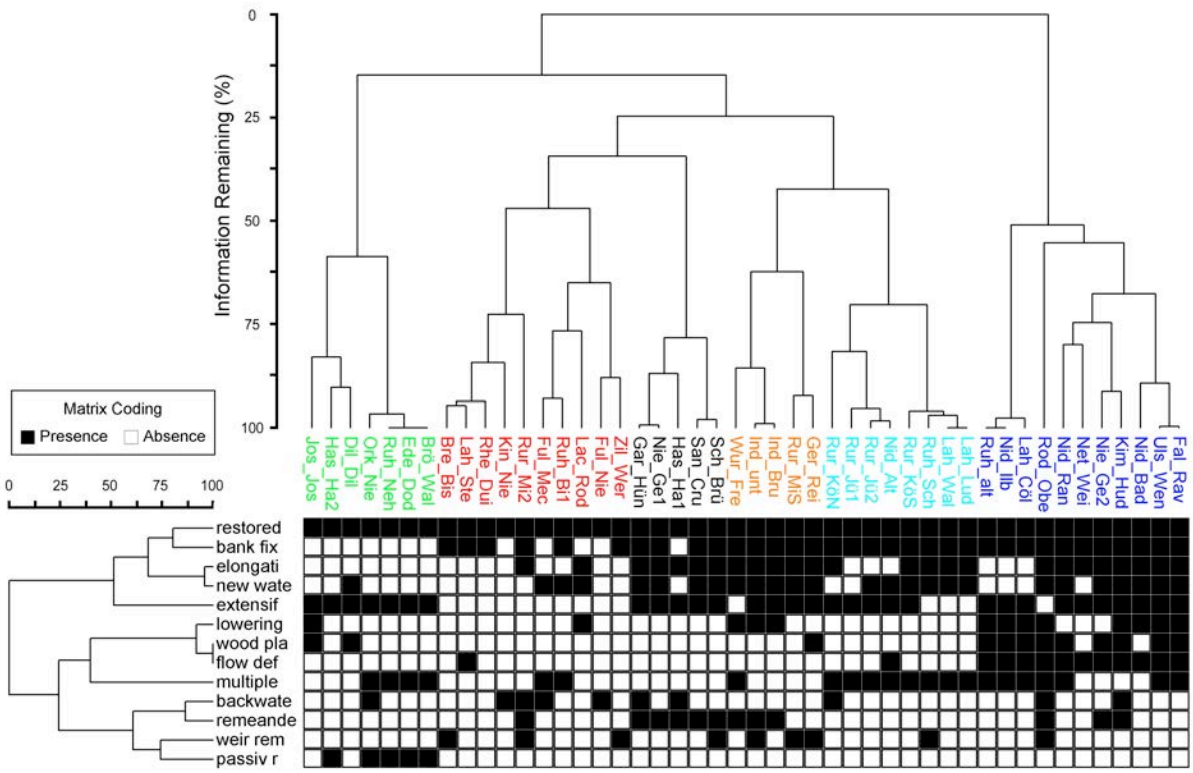

Figure 1: Two-way cluster dendrogram of the conducted measures (horizontal) and the restored sites (vertical).

Step 2

The calculation of the biotic response to the restoration measures was performed by subtracting the metric results of the unrestored reaches from the results of the restored reaches. Finally, the respective results indicate the real change due to the conducted restoration measures. These responses form the basis for further analysis between the cluster groups.

Step 3

The biotic responses were averaged for the 6 types (cluster groups) of restoration measures. Figure 2 shows selected results for the metrics abundance, richness, habitat specifity and assessment result (EQR) for all three biological groups.

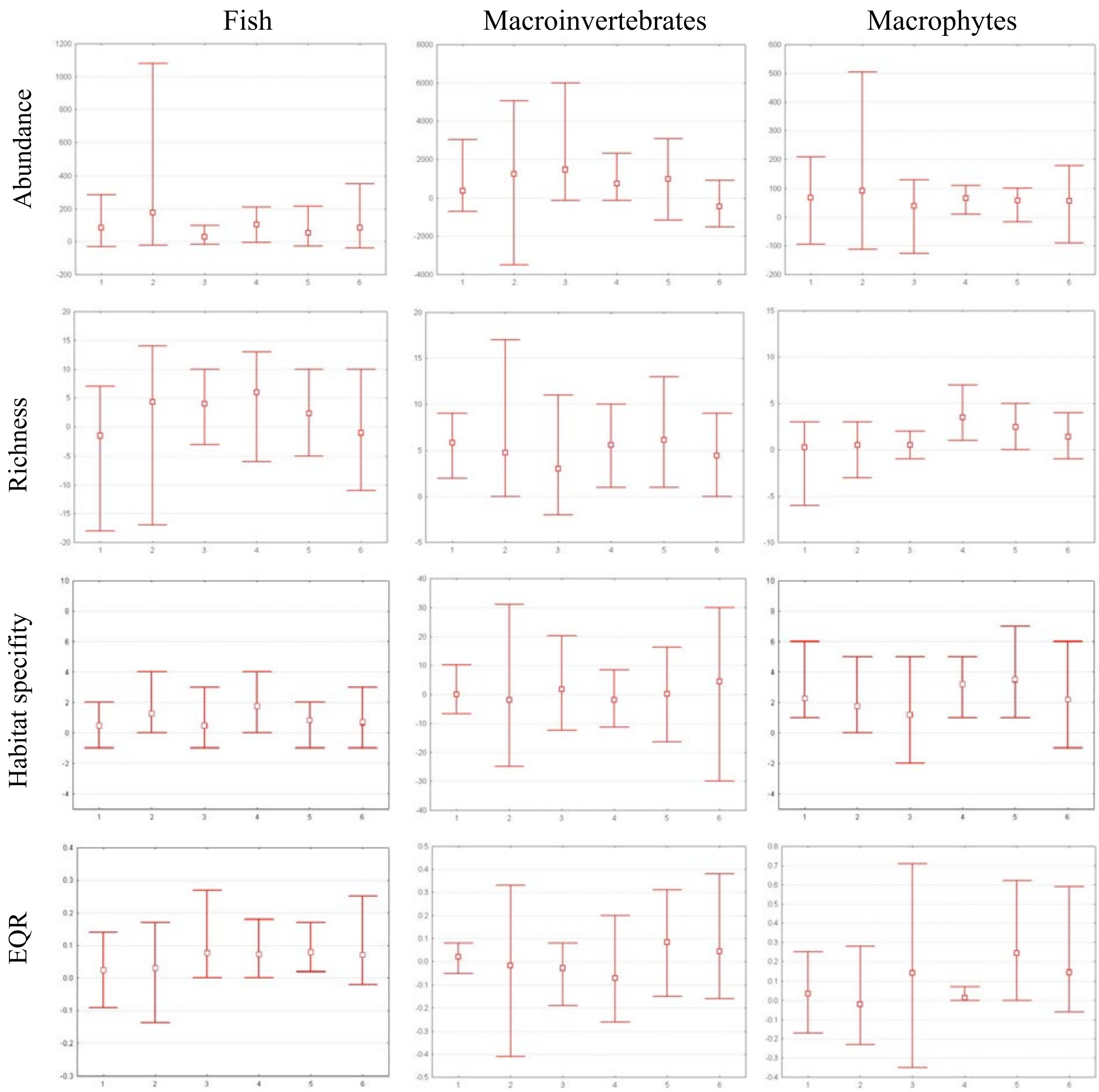

Figure 2: Whisker-plots of selected metrics for the 6 cluster groups (x-axis; Figure 1) in the three organism groups. Abundance: individuals per 100 m (fish), individuals per m^2 (macroinvertebrates) quantity according to the ‚Kohler' scale (macrophytes). Richness: number of taxa. Habitat specifity: number of type-specific species (fish), % of EPT taxa (macroinvertebrates), number of growth forms (macrophytes). EQR: ecological quality ratio derived from the assessment systems. Kruskal-Wallis-test found no significant differences between groups in any metric.

The responses to the restoration differ between the organism groups. In fish and macrophytes the responses are on average positive while macroinvertebrates show nearly no response. Furthermore, there is a wide range of responses within each cluster group indicating that in some cases the restoration measure seems to be effective and in other cases not. Nonetheless, the comparison between the cluster groups showed only subtle and no significant differences irrespective of the metric or organism group.

Step 4

Correlation of the responses of each cluster group to the land use and habitat survey parameters.

This step was omitted, because there are no significant differences in biotic response between the different types of restoration measures.

Congruency in the biological response of three organism groups

The second analysis combines the results of the organism groups by evaluating if there is a congruency in the response. Thus, the question is if there are certain restoration measures, which improve all organism groups or if certain measures favour single organism groups. The results could give water managers a hint which measure they should undertake for their individual goal and how they might achieve the highest ecological benefit.

The analysis is based on the cluster groups of analysis 1. Here, the mean EQR response is calculated for all three organism groups in each cluster group and compared to the responses in the other cluster groups. Additionally, the number of reaches with a positive response in all three organism groups is recorded.

Results

Cluster group 5 has the highest average response of all cluster groups (Table 1). The creation of new water courses with multiple channel patterns seems to foster positive reactions by all three organism groups. The large measures where wood is placed in the river and flow deflectors are constructed in a multiple-channel pattern (cluster group 6) are also followed by comparatively high positive reactions of the three organism groups. Furthermore, in both cluster groups three reaches showed positive responses in all three organism groups. On the other hand, the mean investment was comparatively high in both groups. Short measures and measures in which a reconnection of backwaters was the sole conducted fieldwork had nearly no effect on the biology.

The 9 measures which showed positive responses in all organism groups had no general restoration pattern in common, beside removal of bank fixation and extensification of land use in the floodplain. There is an equal distribution between mountain and lowland reaches and the average restored length was 1,056 m and the average costs covered 800 thousand EUR. In most cases the river channel was changed either by remeandering or by creating multiple channel patterns. Flow deflectors and wood was placed in three reaches and weirs were removed in two of the 9 cases.

Table 1: Mean EQR response, mean investment and number of restoration measures with positive response in all three organism groups for the six cluster groups.

Cluster group	Main measures	Mean EQR response (Fish, MIV and MP)	Mean investment (thousand €)	No. of reaches with positive response in all BQE
1 (n=7)	passive restoration with extensification of land use	0.03 ± 0.10	193	1
2 (n=10)	reconnection of backwaters, mainly short measures,	0.00 ± 0.16	376	0
3 (n=6)	elongation of the river length by remeandering	0.06 ± 0.24	2950	1
4 (n=5)	weir removal, remeandering	0.02 ± 0.18	632	1
5 (n=8)	creation of multiple channel pattern, creating a new water course	0.14 ± 0.17	740	3
6 (n=11)	Placement of flow deflectors, wood placement, creation of multiple channel measures	0.09 ± 0.18	573	3

Conclusions

The mean ecological response of all organism groups combined is positive (0.06 ± 0.18). Nonetheless, the overall response is lower than generally expected. But, there is a wide range in the responses (the standard deviation), which indicates that the biota can respond if certain parameters are improved. Unfortunately, there is no general restoration pattern, which always leads to an improvement of the fauna.

Nearly 20% of the restoration measures (9 of 47) lead to a positive response by fish, macroinvertebrates and macrophytes. That means that restoration measures are possible, which improve the conditions of the three organism groups. But, these measures are expensive and the river channel needs to be widely restructured. Increased in-stream habitat heterogeneity seems to be the main factor improving the ecological quality. Nonetheless, the gross of restoration measures failed to improve all if any organism group. Furthermore, the panacea for restoration on a local reach with the intension to improve all biological organism groups is not yet found. But, local restored reaches are not self-contained ecosystems; they are integral parts of larger catchments and depend on larger scale influences and potential constraints. Thus, further analysis need to consider larger scale influences on the biological response.

The ecological quality and its dependency on the river habitat quality and land use in upstream buffers

This third analysis evaluates the impact of the river habitat quality and the land use variables in different upstream buffers on the ecological quality of the restored and the respective unrestored reaches. With this analysis the dependency of the biota from constraints on the meso-scale will be highlighted. The ecological quality ratio (EQR) of each organism group is correlated to the river habitat survey assessment score of the different buffer lengths (Table 2) and to the land use variables in the different buffer lengths (Table 3). Significant results indicate a dependency of the EQR to the respective variable, while these can be negative or positive relationships.

As a prerequisite a T-test showed no significant differences in the RHS and land use buffers upstream between the unrestored and the restored reaches.

Results

River habitat quality

The organism group fish shows a strong, significant, negative correlation of the EQR to the RHS assessment in the unrestored reaches on all buffers from 500 m to 10 km; this relationship is even stronger for the EQR of the restored reaches from the 500 m to the 5,000 m buffers. In general, this means that fish communities of an individual reach strongly depend on the upstream river habitat quality. In the specific case of the stronger relationship of the restored reaches, this indicates that the river habitat quality upstream fosters the local fish quality (EQR). If in-stream habitats are improved by restoration measures then the fish fauna responds positively under the precondition that the upstream reaches are in a good morphological status. Furthermore, this response is limited to the degree of the potential source populations upstream.

Table 2: Correlation coefficients (r) and Spearman-Test results for the RHS assessments and the EQR in the different buffer lengths of the unrestored and the restored reaches for the three investigated organism groups. Significant correlations in bold.

	Fish		MIV		MP	
Buffer length	Unrestored	Restored	Unrestored	Restored	Unrestored	Restored
500 m	**-0.37** N=32 p=0.035	**-0.44** N=34 p=0.010	**-0.36** N=33 p=0.038	**-0.50** N=35 P=0.002	-0.27 N=34 P=0.128	**-0.49** N=35 P=0.003
1,000 m	**-0.35** N=32 p=0.048	**-0.41** N=34 p=0.002	**-0.38** N=33 p=0.027	**-0.42** N=35 P=0.013	-0.25 N=34 P=0.150	**-0.46** N=35 P=0.005
2,500 m	**-0.51** N=32 p=0.003	**-0.52** N=34 P=0.002	**-0.45** N=33 p=0.008	**-0.40** N=35 p=0.017	-0.32 N=34 P=0.068	**-0.54** N=35 P=0.001
5,000 m	**-0.47** N=32 0.007	**-0.51** N=34 P=0.002	-0.32 N=33 P=0.071	-0.31 N=35 P=0.066	**-0.37** N=34 P=0.034	**-0.45** N=35 P=0.006
7,500 m	**-0.47** N=32 p=0.007	**-0.42** N=34 P=0.014	-0.23 N=33 P=0.208	-0.24 N=35 P=0.165	**-0.36** N=34 P=0.036	**-0.38** N=35 0.023
10,000 m	**-0.50** N=32 0.004	**-0.35** N=34 p=0.043	-0.22 N=33 p=0.229	-0.25 N=35 p=0.147	-0.33 N=34 p=0.060	-0.29 N=35 p=0.089

The influence of the river habitat quality on the EQR of the macroinvertebrates is also significantly negative for both reach groups in the first 3 buffers upstream (500 m, 1,000 m, 2,500 m). Similar to the fish assessment the relationship between the river habitat quality and the local EQR is stronger in the restored reaches compared to the unrestored reaches. Again, this shows that the macroinvertebrate fauna in the restored reaches resembles the upstream habitat quality. The latter can be good as well as bad and determines therefore the EQR of the restored reaches. The strongest relationship between macroinvertebrates (of restored reaches) and the river habitat quality is present on the first 500 m upstream and decreases further upstream. The river habitat quality of buffers equal or longer than 5,000 m does not have a significant influence on the local EQRs. This indicates that the reaches close to the restored reach are the most important for colonization effects in general and the re-colonisation after the restoration measure.

In the macrophytes the effect of the river habitat quality on the EQR in the restored reaches is significantly negative for all buffers up to 7,500 m upstream, while there is only in the 5,000 m and 7,500 m buffer a low but significant effect of the RHS on the unrestored reaches. A good river habitat quality upstream does therefore foster the improvement of the EQR of the flora in the restored reaches and a bad river habitat quality upstream hinders an improvement of the flora. In the unrestored reaches there is nearly no effect of the upstream river habitat quality, which is mainly due to the fact that the flora of unrestored reaches is totally impoverished or not existent. Hence, ecological effects are not discernable.

As a side aspect the results of all three organism groups show that the restoration measures seem to have no adverse effect on the influence of the upstream reaches. The correlations between the EQRs and the river habitat quality are generally stronger in the restored reaches compared to the unrestored reaches. This indicates that local restoration measures do not uncouple the relationship between local EQR and the buffer influence but strengthen it.

Land use quality

Significant correlations of land use variables in any buffer on the EQR are only discernable in the BQE fish and macroinvertebrates. Significant influences of land use variables in the catchment are only found in fish (Table 3). Furthermore, the land use variables: “deciduous forest” and “deciduous and coniferous forest” are the sole variables which have a direct significant influence on the biota.

The results of the fish analysis indicate significance for restored reaches between EQR and land use variables starting not until 5,000 m buffer length, while in the unrestored reaches already the 2,500 m buffer length shows significant positive correlations of deciduous forest and deciduous and coniferous forest to the EQR. In the longer buffers correlations are higher to the EQR in the restored reaches compared to the influence in the unrestored reaches and are furthermore present in two land use variables.

On the catchment scale level a significant negative correlation of arable land with the EQR is seen in the unrestored reaches compared to no significant correlation in restored reaches. The same is true for the percentage of deciduous and coniferous forest. Thus, the effect of the land use practices in the catchment on the fish is mitigated if local restoration measures are conducted.

Table 3: Significant correlations (r; p<0.05 Spearman-test) of land use variables in buffers to the EQR of the fish and macroinvertebrate assessment in the unrestored and the restored reaches.

Organism group	Land use variable	Buffer Length	Buffer width	Fish Unrestored	Fish Restored
Fish	deciduous forest	2500m	50m	0.33	
	dec. and con. forest	2500m	50m	0.34	
	deciduous forest	2500m	100m	0.34	
	dec. and con. forest	2500m	100m	0.33	
	deciduous forest	5000m	50m		0.40
	dec. and con. forest	5000m	50m	0.36	0.39
	deciduous forest	5000m	100m		0.42
	dec. and con. forest	5000m	100m	0.37	0.40
	deciduous forest	10000m	50m		0.47
	dec. and con. forest	10000m	50m	0.42	0.47
	deciduous forest	10000m	100m		0.46
	dec. and con. forest	10000m	100m	0.42	0.41
	arable land	whole catchment		-0.45	
	coniferous forest	whole catchment		0.53	0.37
	dec. and con. forest	whole catchment		0.36	
MIV	deciduous forest	500m	100m		0.33
	deciduous forest	1000m	100m		0.34

For the macroinvertebrates the percentage of deciduous forest in the first two buffers shows a significant positive correlation to the EQR but exclusively in the restored reaches.

Conclusions

All three organism groups are differently influenced by the upstream river habitat quality and the land use practices in buffers and the whole catchment. In fish, there is a strong influence from the upstream river habitat quality, the meso-scale land use in buffers as well as of the land use in the catchment on the EQR, which is in part mitigated by the local restoration measures; i.e. general effects of local upstream in-stream habitat, and land use in buffers as well as of catchment constraints are levelled out by morphological improvements. That means that local restoration measures have strong effects on local fish communities.

Macroinvertebrates are only slightly influenced by meso-scale variables or catchment land use but show effects from the upstream in-stream river habitat quality. There is no effect through the restoration measures on the local fauna. Therefore, the restoration measures do not mitigate the influence of the upstream habitat. That means if the in-stream habitats are in a bad condition upstream than the fauna is poor. But on the other hand if the upstream habitat quality is good and there is a high percentage of deciduous forest in the 500 m and 1000 m buffers then the perspective for a good EQR is also high.

Things are different in the macrophytes. Meso-scale variables and catchment land use do not show any significant influence on the quality of the flora, but the local restoration measures foster effects of the river habitat quality by e.g. providing source populations, which might establish in the restored reaches.

Break points in the relationship between ecological quality, land use and river habitat quality

This fourth analysis is based on the results of analysis 3. The river habitat quality and the land use along the river upstream have a large influence on the EQR of the restored reaches. Therefore, several questions arise regarding the absolute values or reference values for good quality reaches as well as potential break points in the river habitat quality and the land use variables:

- How is the river habitat quality upstream of ecologically good reaches compared to the one of moderate, poor or bad reaches?
- Do the good reaches have a more natural land use upstream than the moderate, poor or bad reaches?
- Which average values in upstream river habitat quality and land use separate good or very good reaches from moderate, poor or bad reaches?

Data evaluation

For the analysis the EQR results of the restored reaches were transformed into the ecological quality classes. According to the EU-WFD, the important break point in terms of the necessity to restore a river reach lies between the quality classes "good" and "moderate", i.e. if the EQR is moderate or worse, restoration actions has to be taken. Therefore, the results were grouped into reference and good reaches on the one hand and moderate, poor and bad reaches on the other hand. Group sizes varied between organism groups because the quality classes "good" and "reference" were achieved only in few cases:

- In the fish seven reaches were in a good status and none in reference status. 32 reaches were in the second group of moderate to bad quality.
- In the macroinvertebrates 12 reaches had the quality class good and one was in reference conditions, while 30 reaches were classified moderate, poor or bad.
- In the macrophytes seven reaches belonged to the first group of which only the Gartroper MB had reference quality. 36 reaches were in the second group.

Then, the river habitat survey assessment and the land use in the quality groups were evaluated and tested (Mann-Whitney-U-Test).

Results

River habitat quality

The river habitat quality is significantly better in the good and reference reaches of the fish and macroinvertebrates compare to the moderate, poor and bad reaches (Figure 3 and Figure 4). For the macrophytes the differences are not significant but obvious (Figure 5). Good reaches have on average a river habitat quality score of 4.5 while moderate, poor and bad reaches have a score of higher than 5 in the first 500 m upstream of the restored reaches.

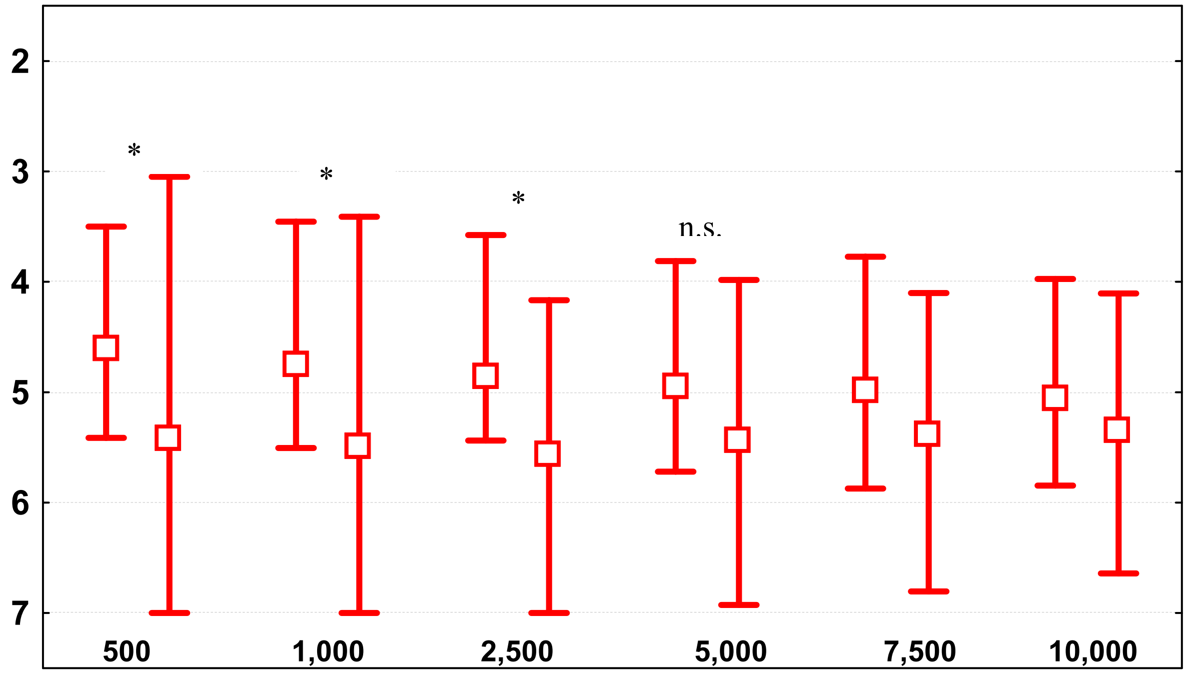

Figure 3: Organism group fish; Whisker-plots of differences between group 1 (reaches with good ecological quality) and group 2 (reaches with moderate, poor or bad ecological quality) in the river habitat survey assessment. Mean, whiskers: Min-Max, asterisks indicate significance p<0.05 (Mann-Whitney-U-Test).

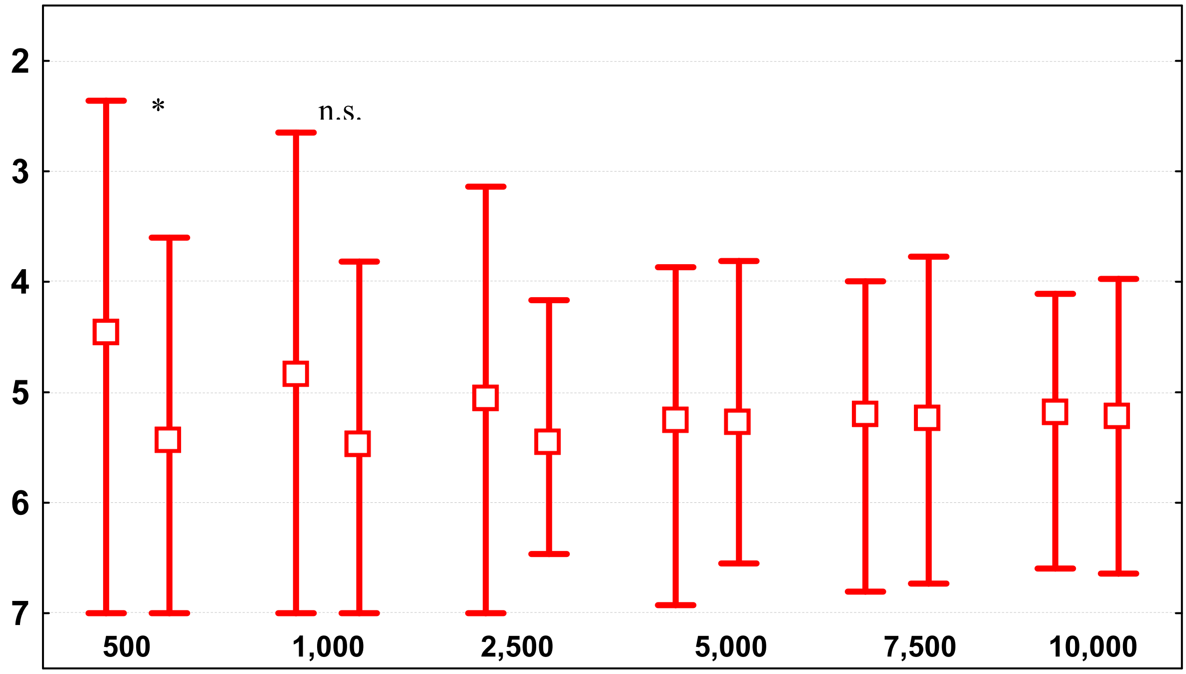

Figure 4: Organism group macroinvertebrates; Whisker-plots of differences between group 1 (reaches with reference and good ecological quality) and group 2 (reaches with moderate, poor or bad ecological quality) in the river habitat survey assessment. Mean, whiskers: Min-Max, asterisks indicate significance p<0.05 (Mann-Whitney-U-Test).

Within the next 2,000 m upstream the average of the first group is in all three organism groups better than 5 but aligns more and more to the score of the second group which is around 5.4. From 5,000 m to 10,000 m the scores of the two groups become similar and no significant differences are detectable in any organism group.

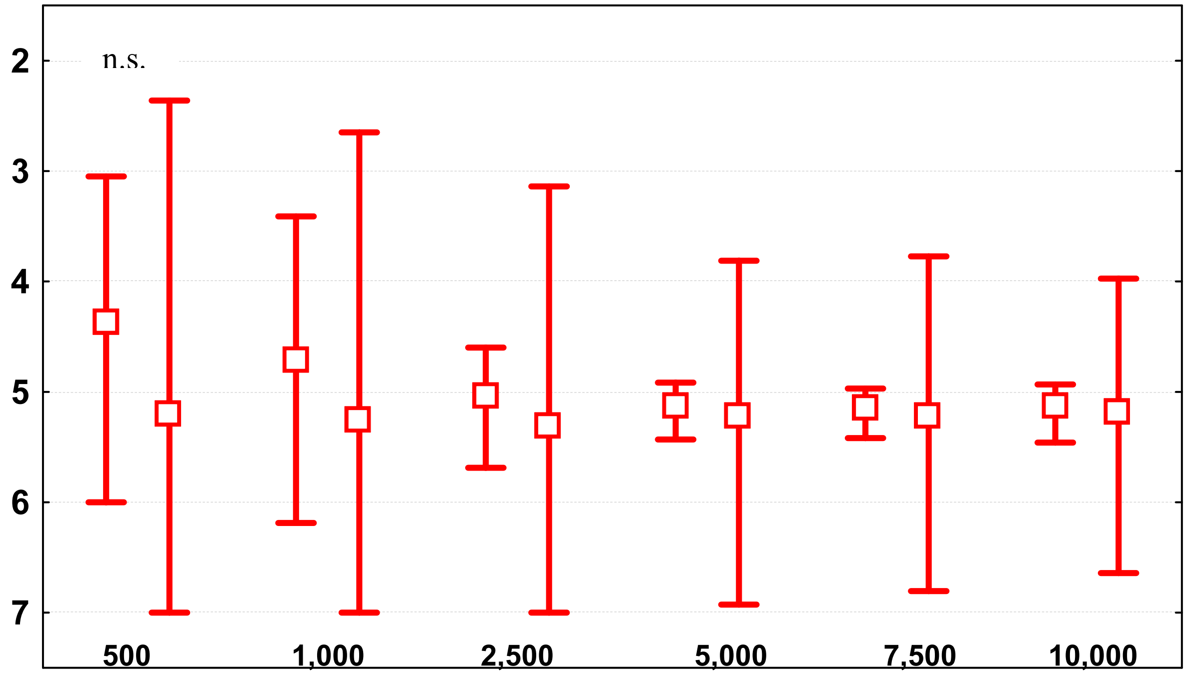

Figure 5: Organism group macrophytes; Whisker-plots of differences between group 1 (reaches with good ecological quality) and group 2 (reaches with moderate, poor or bad ecological quality) in the river habitat survey assessment. Mean, whiskers: Min-Max.

Land use quality

Significant differences between the two ecological quality groups concerning land use variables are mainly found in the short buffers up to 1,000 m upstream (Table 4). The good reaches in the fish fauna are characterized by a higher percentage of deciduous forest and a much lower percentage of arable land compared to the second group. Likewise in the macroinvertebrate, the percentage of deciduous forest is in the first buffers also much higher in the good quality reaches, though it is only significant for one buffer. On the other hand, in the macrophytes the good reaches are characterized by a high percentage of arable land in the first buffers.

Table 4: Mean values of land use variables in buffers with significant differences (p<0.05 U-test) between reference and good reaches (group 1) and moderate, poor or bad reaches (group 2).

		Buffer size		Mean values	
Organism group	Land use variable	Length	Width	Group 1	Group 2
Fish	deciduous forest	500m	50m	18.7	4.1
	deciduous forest	500m	100m	19.4	4.2
	deciduous forest	1000m	50m	20.1	7.1
	deciduous forest	1000m	100m	21.1	6.9
	arable land	500m	50m	14.9	44.2
	arable land	500m	100m	15.2	45.0
MIV	deciduous forest	1000m	100m	22.1	5.3
Macrophytes	arable land	500m	50m	70.2	28.4
	arable land	500m	100m	69.7	29.4
	arable land	1000m	50m	65.2	29.9

Conclusions

The river habitat quality assessment in Germany has a scale from 1 (reference) to 7 (totally degraded). The average of the 500 m upstream sections which were classified ecological as good or reference in all three organism groups was 4.5. This score is worse than the mean on the total scale of the river habitat quality assessment. But it still has positive effects on the biota, when compared to the second group. Thus, restoration measures should at least target an average river habitat assessment score of 4.5 on long upstream sections to affect positively the biota.

In terms of the land use variables in upstream buffers the natural vegetation which is deciduous forest is significantly more present in the good reaches. Thus, fish and macroinvertebrates are fostered by deciduous forest along the river which provides habitat and food while macrophytes are fostered by arable land. The latter is due to the fact that arable land use at the river banks does not shade the river and thus more sunlight reaches the river bottom enhancing macrophyte growth. Even in the good reaches the overall percentage of deciduous forest in the first buffers upstream is relatively low (< 25%). Therefore, potential for further improvement exists.

The biological response and its dependency on the ecological quality of unrestored reaches

This fifth analysis evaluates if there is a relationship between the biological response, i.e. EQR of a restored reach minus EQR of the respective unrestored reach, and the ecological quality class (EQC) of the unrestored reaches. The question behind is, that a restoration measure might have a higher chance of a biological improvement if a reach is strongly degraded than if a reach is only moderately or slightly affected. In a first step the EQR of the unrestored reaches was transformed into the EQC according to the WFD. Then, the responses of each organism group were summarized separately for each quality class. Finally, the values of the different quality classes were tested with a Kruskal-Wallis-Test.

Results

The fish evaluation showed that there is no significant difference in response related to the EQC of the unrestored reaches (Figure 6); i.e. the response is equally positive irrespective of the ecological status in the unrestored reaches.

In the macroinvertebrate evaluation the response was on average negative if the unrestored reaches were already in good and very good status. On the other hand, the reaches, which were classified as moderate, poor or bad showed a significant, positive response in the restored reaches. Particularly, if the unrestored reaches were poor then the restored reaches improved on average by 0.15.

In the macrophytes the restoration measures yielded the highest improvements if the unrestored reaches were in a poor status. This result was significant compared to the generally negative response of the moderate and good reaches.

No very good reaches were found in fish and macrophytes and likewise no bad reaches in the macrophytes.

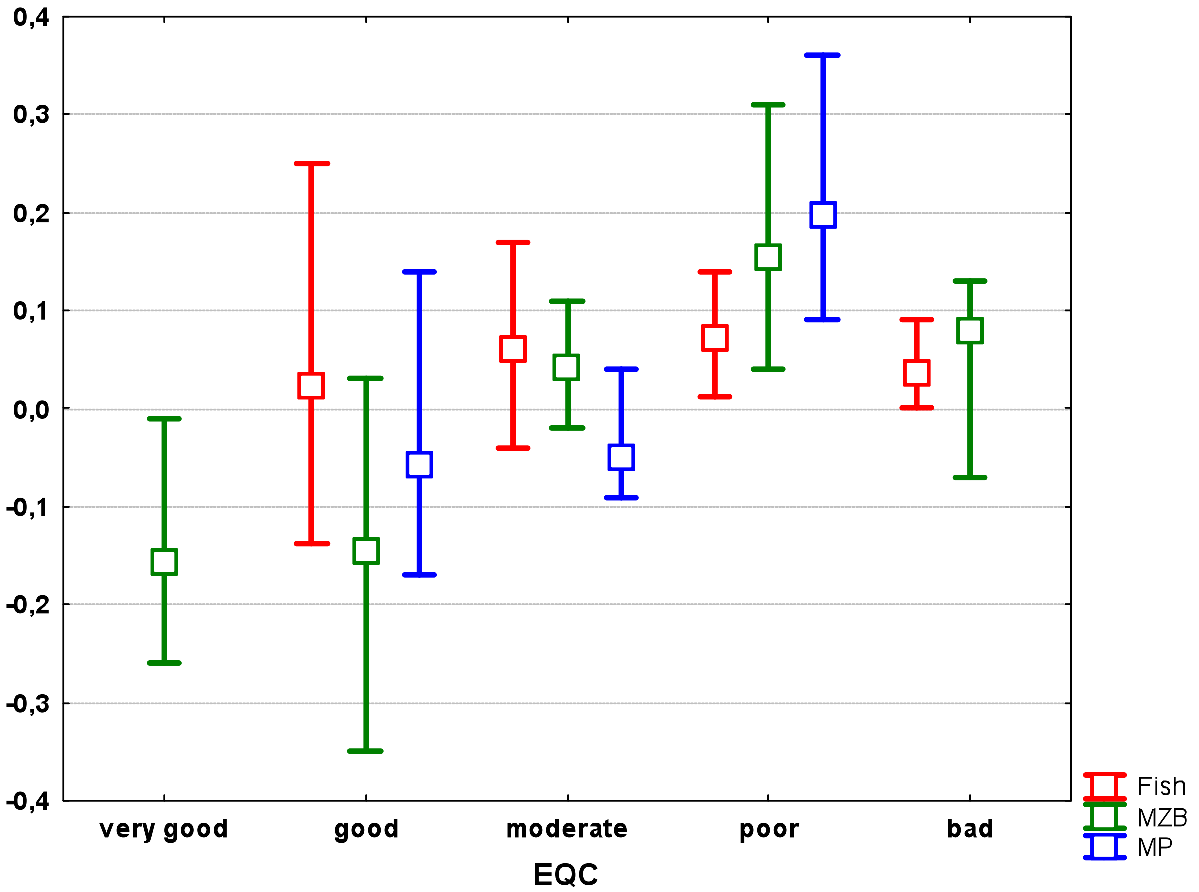

Figure 6: Mean change in EQR (and non-outlier whiskers) for the three organism groups in dependency of the Ecological Quality Class of the unrestored reaches. Kruskal-Wallis-Test found significant differences in macroinvertebrates and macrophytes.

Conclusions

Fish respond positive to restoration measures and the response can raise the EQR up to 0.25 if the unrestored reaches are already in a good or moderate ecological status. The restoration of poor and bad reaches has positive effects as well but to a lower rate. The macroinvertebrates show sensitive reaction to the restoration measures. In rivers where the unrestored reaches were already in a good or very good ecological status the restoration measures lead on average to deterioration probably by the disturbance caused by the restoration works. Reaches, which are in a poor ecological status have the best potential for a high improvement. On the other hand, improvements are minor (generally below 0.1) compared to fish and macrophytes. Macrophytes are also negatively affected by restoration works if the flora is already in a good or moderate status. In poor and bad reaches the flora is normally non-existent and restoration on average enhances growths and abundance of the flora.

Thus, restoration measures targeting an improvement of the fish fauna should focus on moderate and poor reaches but might even improve good reaches, while for macroinvertebrates mainly poor and bad reaches will show a positive response. In the macrophytes the poor reaches should be the focal point of restoration works.

Acknowledgement

The restoration data were gathered in two national projects funded on the one hand by the Bundesamt für Naturschutz (German Federal Agency for Nature Protection) and on the other hand by the Deutsche Bundesstiftung Umwelt (German Environment Foundation) and the Hessisches Ministerium für Umwelt, ländlichen Raum und Verbraucherschutz (Ministry for the Environment Hesse). Furthermore, we're grateful to Planungsbüro Koenzen for providing additional data.

Case study Vecht catchment – Changes in ecological condition of streams in relation to land use change and hydromorphological restoration measures

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Introduction

The European Water Framework Directive's objective is to achieve at least 'good ecological quality status' for all water bodies by 2015. In order to achieve this goal numerous very costly river restoration measures have been taken or will be taken. Given the costs of river restoration measures insight in the effectiveness of these measures is crucial.

River restoration is gaining increased attention in many parts of the world, as reflected in growing numbers of projects and increased financial support (Palmer et al, 2007). To assess the effectiveness of restoration measures collecting appropriate monitoring data is essential (Reitberger et al. 2010). This means monitoring should take place prior to restoration and after restoration, at both unrestored (control) and restored site(s). The effects of river restoration on the ecological condition of streams are often not clear. The main reason for this is the lack of studies using monitoring data from both control and impacted sites, and using pre- and post-restoration data (Jähnig et al. 2010, Reitberger et al. 2010). Another reason is that the impact of local restoration projects on the ecological status of the water body interacts with global and regional pressures such as climate change and land use change. Better understanding of how these restoration projects are impacted by more global/regional pressures is crucial for effective catchment management planning.

The objective of this study was to determine: 1) the relationship between land use cover and the ecological condition of streams and 2) influence of changes in land use cover on the effects of hydromorphological restoration measures.

Methods

Study area

The management district of the regional water authority Regge and Dinkel encompasses 135,000 ha and is located in the Province Overijssel, in the east of the Netherlands, adjacent to the German border. Its borders are almost the same as those of Twente. The main rivers within the management district are the Regge, the Dinkel and the Linderbeek, which flow into the

Overijsselse Vecht. In Twente the most important land use type is agriculture (70%), which is mainly dairy farms and crop farms. Fifteen % of the area is forest or nature reserve and 12% of the area is developed (Catchment Management Plan 2010–2015 Waterschap Regge and Dinkel).

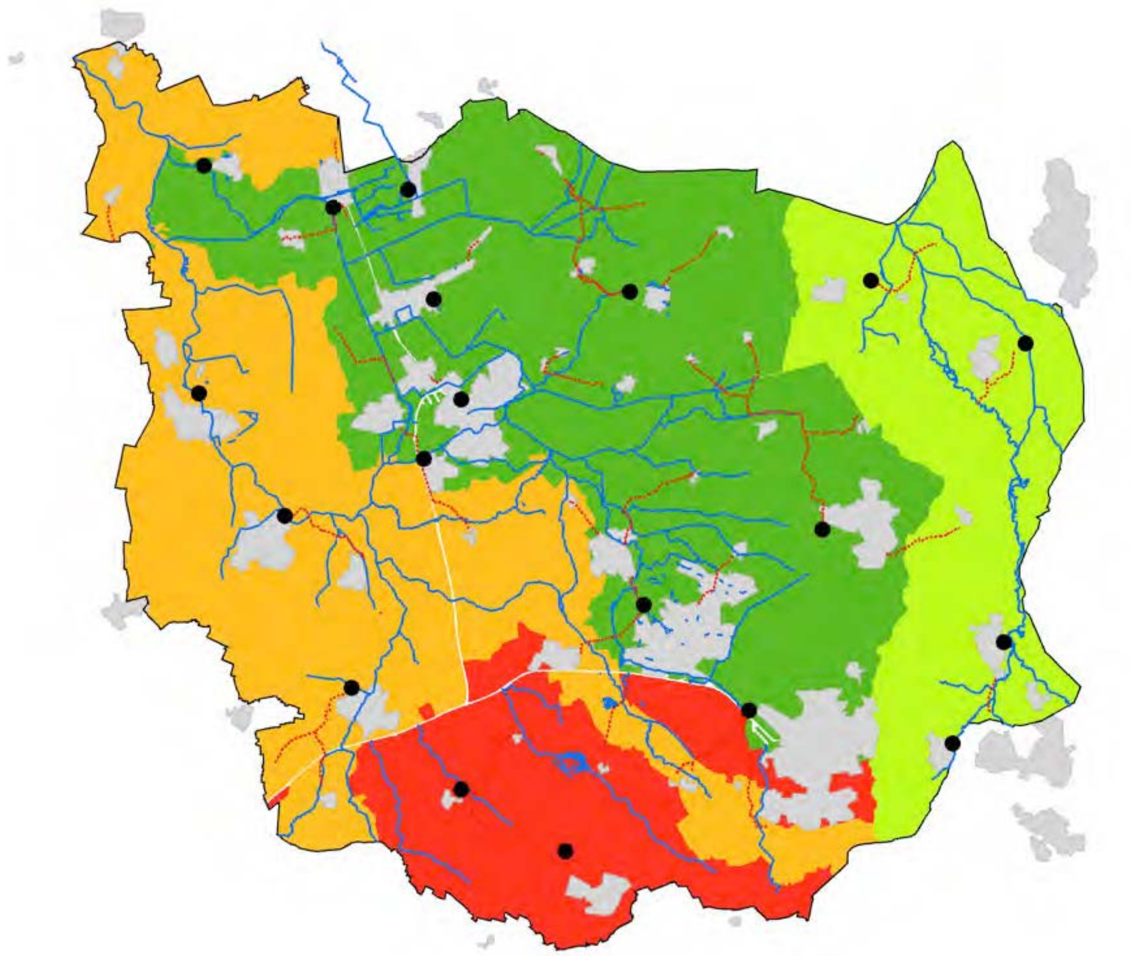

Figure 1: Overview of the main catchments managed by regional water authority Regge and Dinkel (Waterschap Regge en Dinkel, 2010). Orange = Regge, dark green = Linderbeek, light green = Dinkel, and red = Twente canal.

Data collection and preparation data

Since 1979 the regional water authority Regge and Dinkel has monitored the community composition of several aquatic BQE's on several locations in the Regge and Dinkel catchment on a regular basis. This has resulted in a large data set covering a thirty-year period (1979–2008). Depending on the BQE between 16 and 2249 samples were collected within this time period (Table 1).

Table 1: Overview of number of locations and number of samples per BQE.

BQE	number of locations	number of samples
diatoms	157	295
phytoplankton	10	16
desmids	70	99
macrophytes	471	674
fish	512	1085
macroinvertebrates	789	2249

Data analyses

Since the dataset was very large we decided to explore the dataset in a first step to see if biological changes could be detected over time within the management district of regional water authority Regge en Dinkel (step 1). The analyses were performed with the macroinvertebrate data, because this BQE was sampled the most (Table 1 and 2). No major biological changes could be detected over time at catchment scale (paragraph x). The results of the analyses at catchment scale showed high (spatial) variation, which might have confounded the results (paragraph 3.1). To reduce spatial variation we analyzed data at reach scale (within catchments) to see if we could detect change in ecological condition (step 2). At the level of individual locations data availability was marginal (Table 2). Macroinvertebrate samples collected regularly over longer time periods were sparse (Table 2). Therefore, data from different locations within the same reach were combined. Analyses at reach scale were not performed for BQE's other than macroinvertebrates, because data availability for these other BQE's was even less than for macroinvertebrates. In a third step the biological changes at reach scale were related to changes in land use and restoration measures. Since, changes in land use appeared to be non-existing (paragraph 3.3) we performed an final analysis (step 4) were space for time substitution was used to look for relationships between spatial differences in ecological quality status within the management district and cover by different land use types. Step 4 was performed both the catchment/drainage basin and site scale.

Table 2: Overview of number of locations with at least 10 and at least 6 samples per BQE.

BQE	≥10 samples	≥ 6 samples
diatoms	0	3
fytoplankton	0	1
desmids	-	-
macrophytes	0	6
fish	8	31
macrofauna	36	70

Step 1: Biological change in time at catchment scale

To reduce spatial variation in the data set only data from streams were used for analysis. The remaining data set was divided into three major catchments; (1) Regge and Twente canal, (2) Dinkel, and (3) Linderbeek. The analysis was performed separately for these three catchments. The macroinvertebrate data were pre-processed. For each sample the number of individuals per taxon was standardized to a total sampled area of 1.25 m2. A taxonomic adjustment was performed on the data: the lowest taxonomic level was used, unless the frequency of occurrence of the higher level made up more than 20% of the sum of the lower level taxa. In the latter case the lower taxonomic level was changed into the higher taxonomic level. After taxonomic adjustment the macroinvertebrate abundances of each sample were logarithmically transformed (2 log(x+1)). The samples were ordinated by detrended (canonical) correspondence analysis using the program CANOCO (Ter Braak, 1987). DCA was used to determine the variation in the dataset. Based on the results of the DCA it was decided to use a unimodal technique for further analysis. DCA was used because Correspondence Analysis (CA) lead to an arch-effect. The program CANOCO offers different options on how to present and analyze data. The choices made in CANOCO will influence the result of the ordination. All techniques are fully explained by Ter Braak and Šmilauer (1998). In this study the following options were selected:

- downweighting of rare species: reduces the influence of rare species on the analysis;
- detrending by segments;

In the resulting ordination diagrams samples were grouped per 5 years, to make it easier to detect changes in community composition over time.

Step 2: Biological change in time at reach scale

These analyses were performed for the Linderbeek catchment only. We chose the Linderbeek catchment because this was the catchment with the highest diversity of land use types and with a relative high density of restoration projects. We pooled data from different sampling locations in a river stretch as one. We choose those river stretches with the highest density of sampling locations. Within the Linderbeek 129 sampling locations were grouped into 24 stretches of stream. Within one stretch of stream we considered all the sampling location as one. 63 sampling locations weren't allocated to a stretch of river (because they were located in the stream, but in water near the stream), and were left out of the analysis. The Linderbeek data were thus pooled into 24 stretches of river, later referred to as 'catchments'.

To determine whether ecological quality has changed over time within the 24 selected river stretches, multimetric index (MMI) values were calculated based on the macroinvertebrate community composition of the samples taken from the 129 sampling locations. The applied multimetric index has been developed by Verdonschot and Verdonschot (2010) to assess the ecological condition of Dutch streams. For a detailed description of the calculation of the MMI see the same publication.

The multimetric index value is the mean of the following standardized metrics:

- % oligosaprobic taxa
- % Diptera (excl. Chironomidae) taxa
- % Trichoptera taxa
- % Hirundinea taxa
- % Heteroptera taxa
- % taxa with plants as habitat preference (epiphytic, phytophilous)
- % detrivore + detri-herbivore + herbivore individuals

MMI values per catchment were plotted against time. To test whether MMI values changed over time, we conducted a linear regression with the MMI value as dependent variable and time as the independent variable. Linear regression was performed separately for the 24 catchments in R 2.11.1. Based on these results we selected 4 catchments, 3 of which MMI changed significantly and one that didn't change. Any further analysis in step 3 was only done on these four catchments.

Step 3: Biological change in time at reach scale in relation to restoration measures and land use

Based on the analyses in step 2 the catchments 4, 8, 12 and 13 were selected for analyses in step 3. Catchment 13 is located in the northwest of the water authority Regge and Dinkel (Figure 2). The main river stretch on which samples were taken was the Linderbeek. Catchment 4 is located in the north of the water authority Regge and Dinkel (Figure 2). The main river stretch sampled in this catchment was the Hazelbeek. Catchment 12 is located southwest in the water authority (Figure 2). The rivers that have been sampled in this catchment are the Drienerbeek and the Eschbeek. Finally, catchment 8 largely corresponds with the catchment of the Azelerbeek (Figure 2).

GIS information was available of the years 1980, 1988, 1994, 1997, 2000, 2004, and 2008 for the management district of regional water authority Regge and Dinkel Water (Centrum Geo-information, Alterra). The information for the 1980 land use data are based on the topographic maps of that year. The maps of the other years are based on both topographic maps and aerial imagery. The number of original land use types varied between 18 and 45, depending on the year. Because we weren't interested in that level of detail we grouped the different land use types in the following land use categories: 'development', 'forest and nature', 'grass', 'heath', 'agriculture', 'natural grassland', 'reed' and 'water' (Table 3). For catchment 13, 12, 8, and 4 we calculated the percentage coverage of each land use category. Analyses of the GIS data were done in Arcmap (9.3.1). The Water authority Regge and Dinkel provided us with a list of the river restoration projects that took place in the catchments 13, 12, 4 and 8 (see above) during the last thirty years.

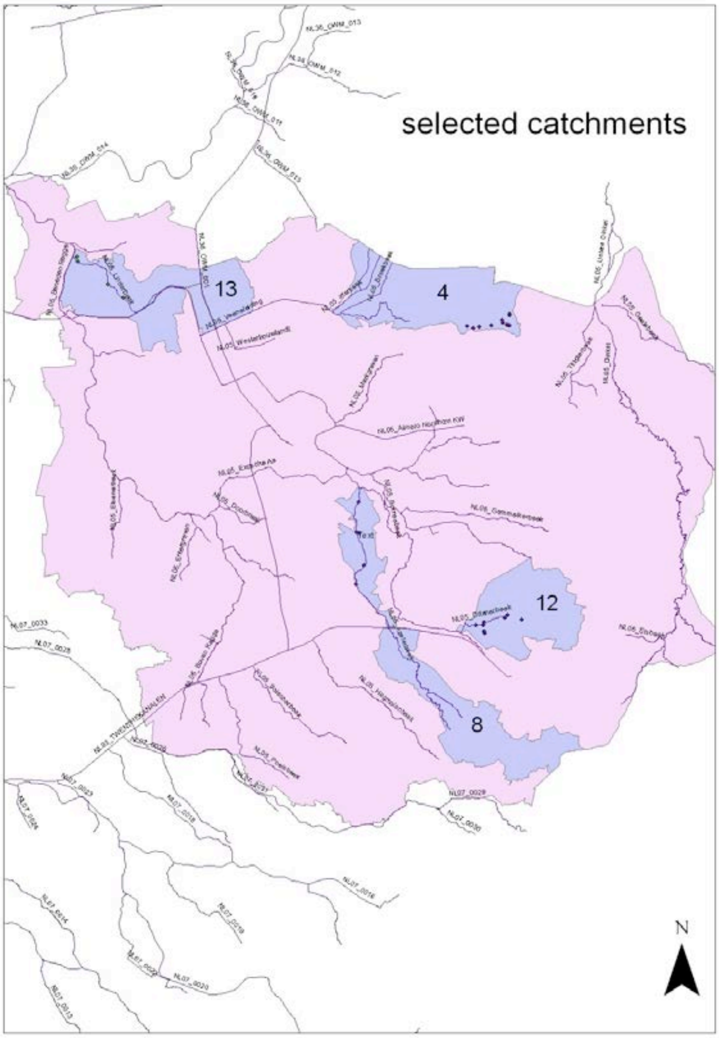

Figure 2. Map showing sampling locations within the selected catchments 13, 4, 8, and 12 in the management district of regional water authority Regge and Dinkel.

Table 3. Overview into which of the six land use categories the original LGN6 land use types were combined for the purpose of analysis.

code	LNG6 land use types	land use category
1	agricultural grassland	grass
2	corn	agriculture
3	potatoes	agriculture
4	beets	agriculture
5	grain	agriculture
6	other agriculture	agriculture
8	cultivation under glass	development
9	orchards	agriculture
10	bulb-cultivation	agriculture
11	deciduous wood	forest and nature
12	coniferous wood	forest and nature
16	freshwater	water
17	salt water	water
18	buildings within the built-up area	development
19	buildings outside the built-up area	development
20	forest within the built-up area	forest and nature
22	forest outside the built-up area	forest and nature
23	grass in the built-up area	grass

code	LNG6 land use types	land use category
24	bare soil in the built-up area	development
25	main roads and railways	development
26	buildings outside the built-up area	development
28	grass outside the built-up area	grass
30	salt marsh	forest and nature
31	open sand in coastal areas	forest and nature
32	dunes with short vegetation (<1m)	forest and nature
33	dunes with tall vegetation (>1m)	forest and nature
34	dunes with heath	heath
35	open drift-sand or river sand	forest and nature
36	heath	heath
37	heath moderately grassy	heath
38	heath strongly grassy	heath
39	bog	forest and nature
40	forest in bog area	forest and nature
41	other swamp vegetation	forest and nature
42	reed	reed
43	forest in swamp area	forest and nature
45	natural grasslands	natural grassland
61	three cultivation	agriculture
62	fruit cultivation	agriculture

For the selected catchments we plotted the percentage of cover for the main land use categories (development, forest and nature, grass, and agriculture) against time and conducted a linear regression between land use and 'time'. Pearson product-moment correlations were performed to test the relationship between the percentage of land use cover and MMI value of the corresponding time. Because the times of land use measurement and measurements of the ecological condition (i.e. macroinvertebrate sampling) don't necessarily coincide, comparisons were made between land use data and estimated values of the ecological condition (MMI). We used predicted MMI values based on the linear regression model. To test the impact of river restoration programs on the ecological status of the 'catchment', MMI values before and after a restoration event were compared using one-way ANOVA's (α=0.05). For this comparison we only used the data from locations that were both sampled before and after the implementation of the restoration measures. Locations that were only sampled before or after the implementation of the restoration measures were excluded from the analysis. Only catchment 13 and 4 met these conditions. All the statistical analyses were done in R 2.11.1

Step 4: Biological change in space in relation to land use

We performed two separate analyses one at the catchment scale and one at site scale. To determine the percentage of cover for different land use types we used the Dutch land use database (LGN). The LGN database is a raster database with 25*25 m resolution, covering the entire Dutch territory and presents the land use in 39 classes. From 1986 the database is frequently updated with a 3–5 years interval. It is based on a combination of geodata and satellite images. The LGN database is a product of the Centre for Geo-Information, which is part of the Wageningen University and Research Centre. For the purpose of this analysis we used

LGN6, the latest version of the database, which is based on data from 2006, 2007 and 2008. We performed two separate analyses one at the catchment scale and one at site scale.

Catchment scale - All samples collected in streams were selected from the database (n=1,521). Sample without an indication of water body type were not used for analysis. All samples were plotted on a topographic map and assigned to a catchment based on a map with drainage basins. In the management district of regional water authority Regge and Dinkel 69 catchments could be discerned based on this map (Table 4). Of these 69 catchments, 38 catchments contained macroinvertebrate samples from streams. Within each catchment the percentage of cover per land use type was calculated in ArcMap 9.3.1. The 39 different land use types in the LGN6 were combined into six different categories: urban (roads and houses), forest, agricultural grassland, agriculture, nature (excluding forest) and freshwater (Table 5). For more details, see http://www.lgn.nl. The software QBWAT version 4.3.1 was used to determine the ecological quality ratios (EQRs) for each macroinvertebrate sample. EQR scores can range between zero and 1 and determine the ecological status (Table 6). Per catchment EQRs for all samples (across years and locations) were averaged. In Table 5 an overview is given of the number of samples collected per catchment between 1978 and 2008. Spearman rank correlations between average EQRs and percentage of land use cover were performed for each of six land use categories. Spearman rank correlations were performed in SPSS Statistics version 19. Also, scatter plots were made.

Table 4. Overview per catchment of the number of samples available for analysis.

catchment	number of samples
AZELERBEEK	84
BENEDEN- EN MIDDEN-REGGE	111
BOLSCHERBEEK	16
BORNSEBEEK	41
BOVEN REGGE - DELDEN A	6
BOVEN REGGE - DIEPENHEIM	30
BOVEN REGGE - GOOR RNOO_L05	54
DINKEL	203
DINKELKANAAL	55
DOORBRAAK/BORNERBROEKSE WATERLEIDING	31
ELSBEEK	23
ELSENERBEEK	11
ENTERGRAVEN	20
EXOSCHE AA	31
GAMMELKERBEEK	28
GEELEBEEK	26
GEESTERSE MOLENBEEK/ BROEKBEEK/ ITTERBEEK	163
GLANERBEEK	45
HAGMOLENBEEK	58
HAMMERWETERING	12
HOLTDIJKSBEEK	6
HOOGELAARSLEIDING	18
KANAAL ALMELO-NORDHORN WEST	23
KOPPELLEIDING	30
LATERAALKANAAL/ VEENELEIDING	35
LINDERBEEK	23
LOOLEE/ OUDE BORNSEBEEK	143

catchment	number of samples
MARKGRAVEN	49
NIEUWE STROOMKANAAL/ GEESTERSCH STROOMKANAAL	10
OMLEIDINGSKANAAL	27
OVERIJSSELS KANAAL	2
POELSBEEK	22
PUNTBEEK	27
RUENBERGERBEEK	48
TILLIGTERBEEK	212
TWENTEKANAAL A	22
TWENTEKANAAL B	1
WESTERBOUWLANDLEIDING	5

Table 5. Overview into which of the six land use categories the original LGN6 land use types were combined for the purpose of analysis.

code	LNG6 land use types	land use category
1	agricultural grassland	agricultural grassland
2	corn	agriculture
3	potatoes	agriculture
4	beets	agriculture
5	grain	agriculture
6	other agriculture	agriculture
8	cultivation under glass	agriculture
9	orchards	agriculture
10	bulb-cultivation	agriculture
11	deciduous wood	forest
12	coniferous wood	forest
16	freshwater	freshwater
17	salt water	freshwater
18	buildings within the built-up area	urban
19	buildings outside the built-up area	urban
20	forest within the built-up area	urban
22	forest outside the built-up area	forest
23	grass in the built-up area	urban
24	bare soil in the built-up area	urban
25	main roads and railways	urban
26	buildings outside the built-up area	urban
28	grass outside the built-up area	agricultural grassland
30	salt marsh	nature
31	open sand in coastal areas	nature
32	dunes with short vegetation (<1m)	nature
33	dunes with tall vegetation (>1m)	forest
34	dunes with heath	nature
35	open drift-sand or river sand	nature
36	heath	nature
37	heath moderately grassy	nature
38	heath strongly grassy	nature
39	bog	nature
40	forest in bog area	forest
41	other swamp vegetation	nature
42	reed	nature
43	forest in swamp area	forest
45	Natural grasslands	nature
61	three cultivation	agriculture
62	fruit cultivation	agriculture

Table 6. Relation between EQR and ecological status.

EQR	ecological status
0.0 - ≤0.2	bad
>0.2 - ≤0.4	poor
>0.4 - ≤0.6	moderate
>0.6 - ≤0.8	good
>0.8 - ≤1	high

Site scale - All samples collected in streams were selected from the database (n=1,521). Sample without an indication of water body type were not used for analysis. All samples were plotted on a topographic map and a circular buffer of 50 m in diameter was created around each site. The percentage of cover per land use category was calculated in ArcMap 9.3.1 for the buffer area (=7,833 m^2) was calculated for each site. Land use data were generated similar to the procedure for catchment-scale analysis. Per site EQR's for all samples (from different dates/years) were averaged (n=520). Spearman rank correlations between average EQR's and percentage of land use cover were performed for each of six land use categories, using the methods already described for catchment-scale analysis.

Results

Biological change in time at catchment scale

Multivariate analyses didn't show a gradual change (improvement in ecological condition) in macroinvertebrate community composition between 1980 and 2008 in the three catchments (Dinkel, Linderbeek and Regge) (Figure 3–5). Only in some cases, a certain time span did form a separate cluster. For example the macroinvertebrate samples collected during '86–'89 in the Regge en Twente canal catchment (Figure 5).

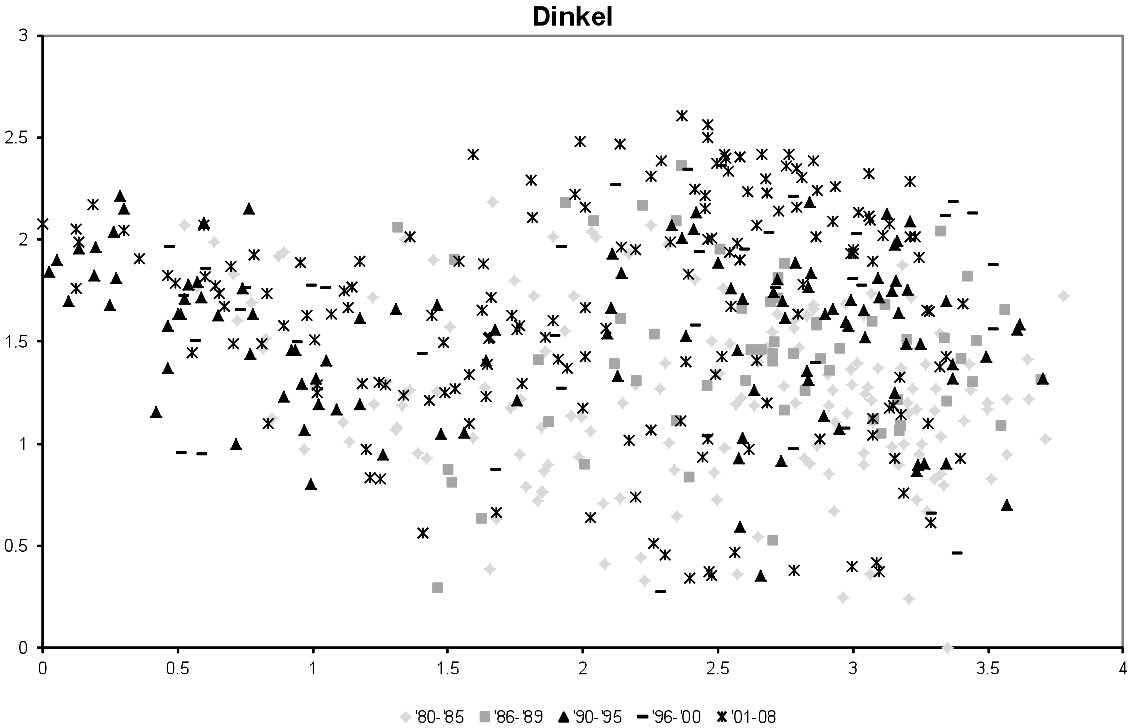

Figure 3. DCA ordination diagram of axis 1 and axis 2 showing macroinvertebrate samples from streams within the Dinkel catchment.

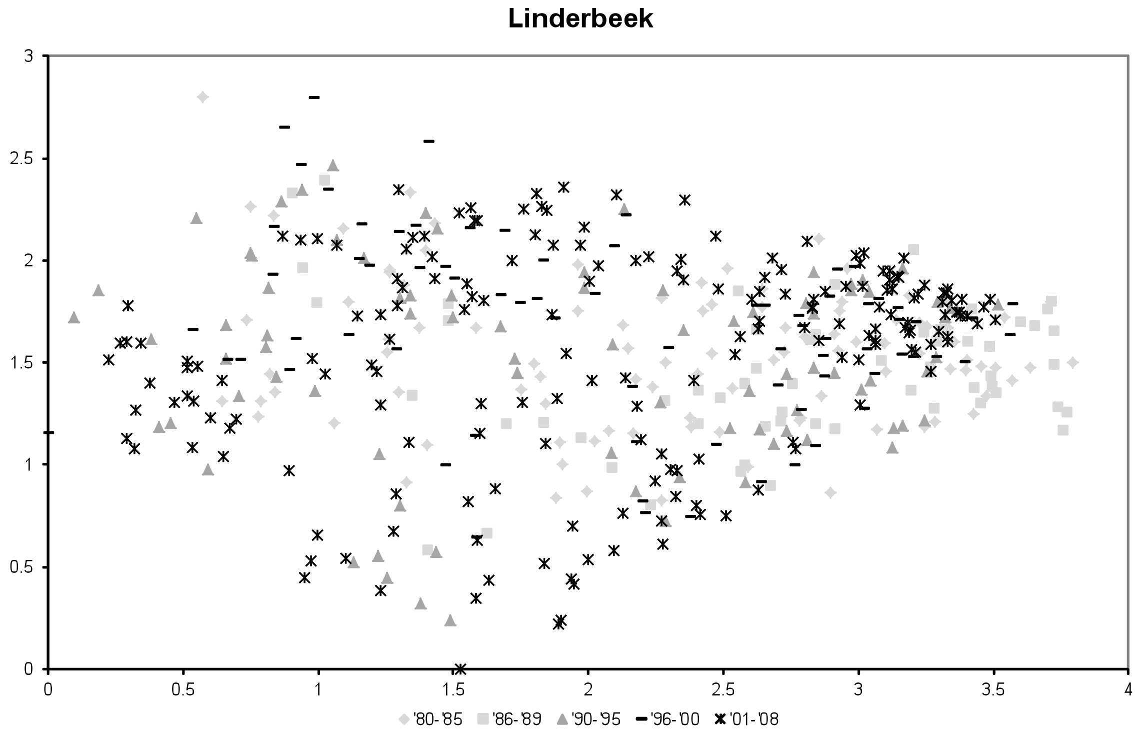

Figure 4. DCA ordination diagram of axis 1 and axis 2 showing macroinvertebrate samples from streams within the Linderbeek catchment.

Regge en Twente canal

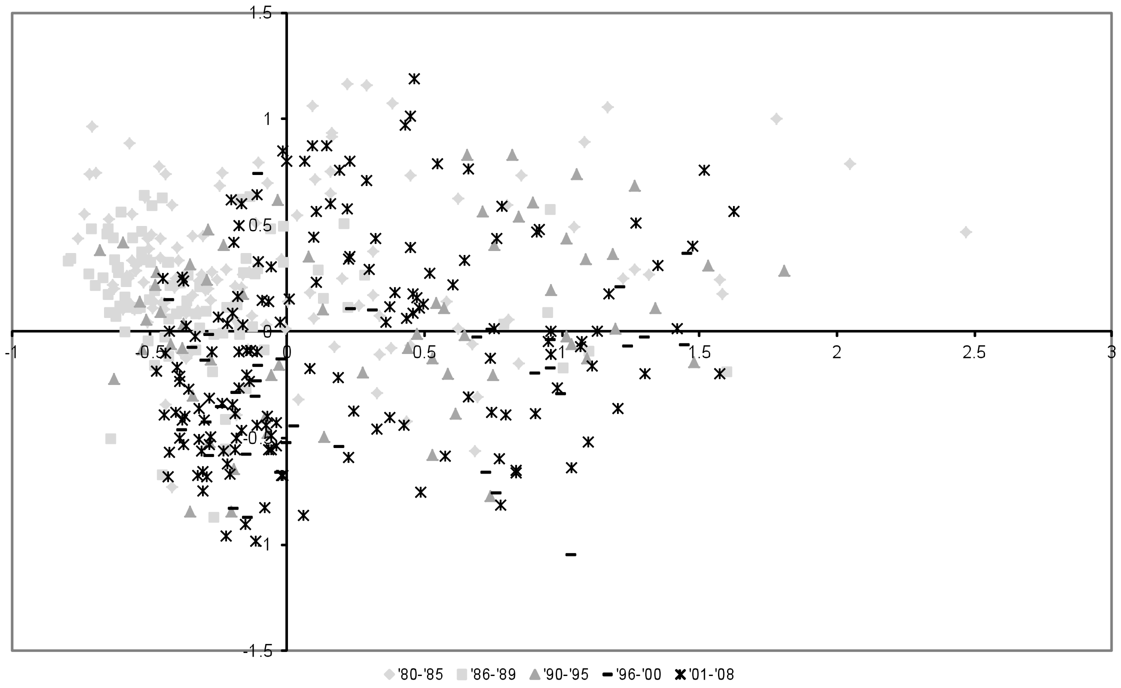

Figure 5. D?CA ordination diagram of axis 1 and axis 2 showing macroinvertebrate samples from streams within the Regge and Twente canal catchment.

Discussion – The fact that no changes in macroinvertebrate community composition were detected within the management district of regional water authority Regge en Dinkel between 1980 and 2008 might have been caused due to high spatial variation in macroinvertebrate community composition. However, analyses in step 2 only showed significant change in MMI values for three out of 24 catchments within the Linderbeek system. This confirms our believe that possible changes in macroinvertebrate community composition between 1980 and 2008 at the scale of the entire management district were not masked by spatial variation. The fact remains that other sources of variation, e.g. sampling by different persons, sampling using different methods and sampling during different seasons might have masked changes in macroinvertebrate community composition.

Biological change in time at reach scale

In figures 6 to 10 we plotted per catchment the MMI values against time. Catchment 2 and 4 had the longest time series, respectively 27 and 29 years. They also have the most samples (62 and 34 respectively). We conducted a linear regression for each of the catchments with MMI values as the dependent variable and time as the independent variable. MMI values in catchment 4 increased significantly with time ($R^2 = 0.16$, $p = 0.01$), so did MMI values in catchment 13 ($R^2 = 0.46$, $p = 0.001$). In catchment 12 MMI significantly decreased over time ($R^2 = 0.26$, $p = 0.02$). None of the other catchments showed significant changes in MMI values over time. Although there was significant change over time, the change is very small (Figure 5 and 7). The change in metric values is in all three cases less than 0.2 (0.2 equals the range of an ecological quality class).

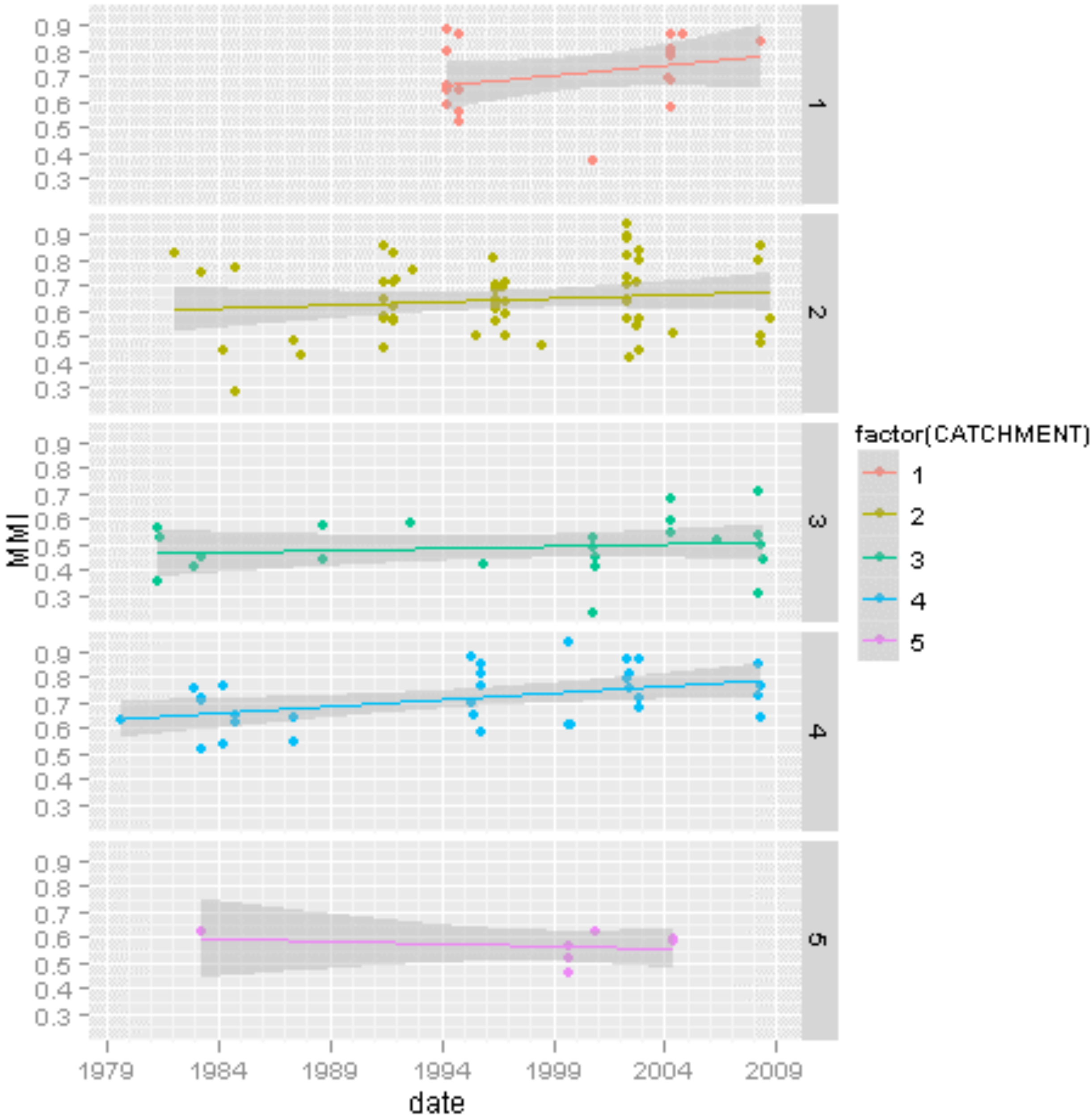

Figure 6. MMI values (ecological condition) over time for catchments 1–5.

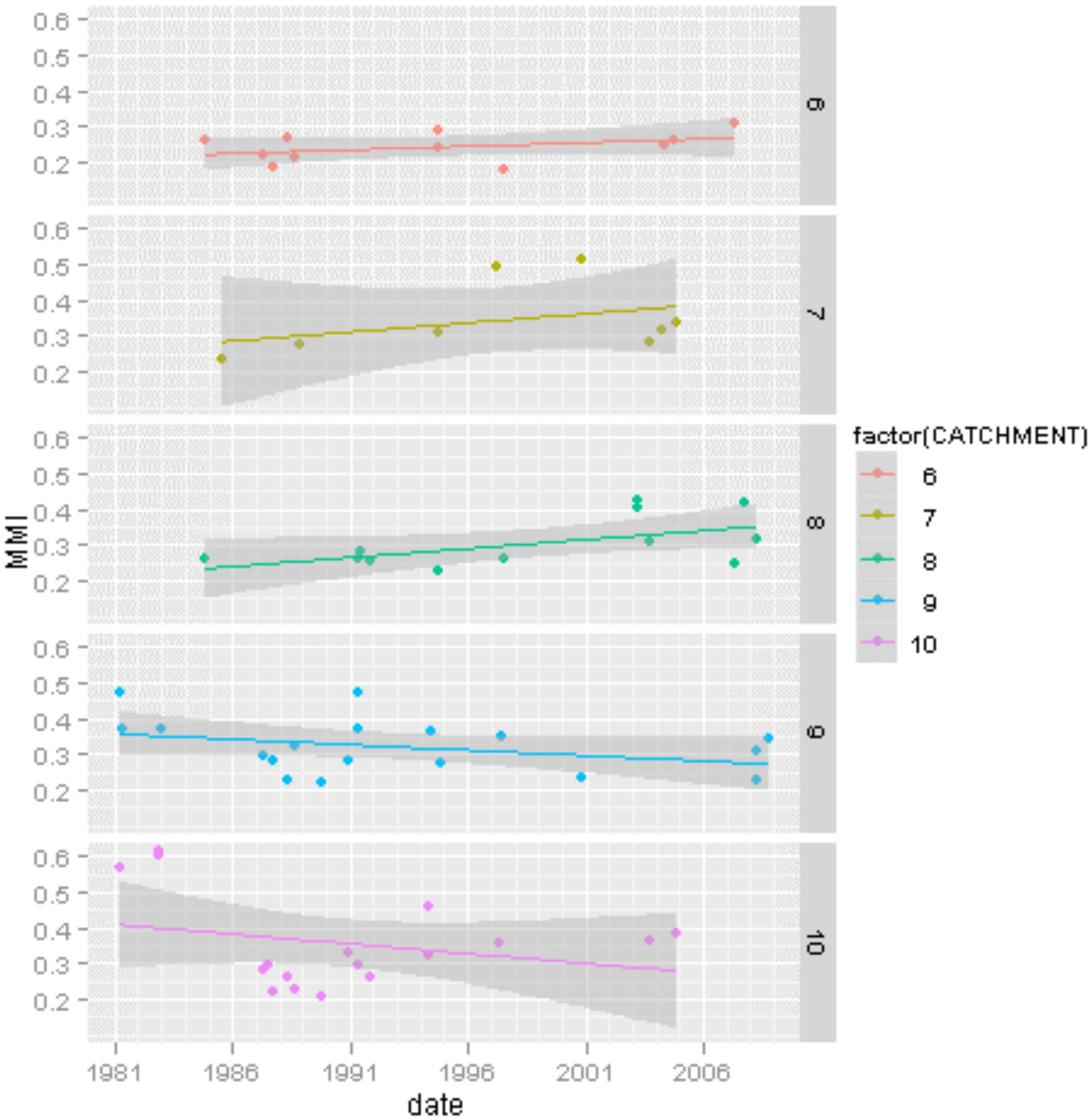

Figure 7. MMI values (ecological condition) over time for catchments 6–10.

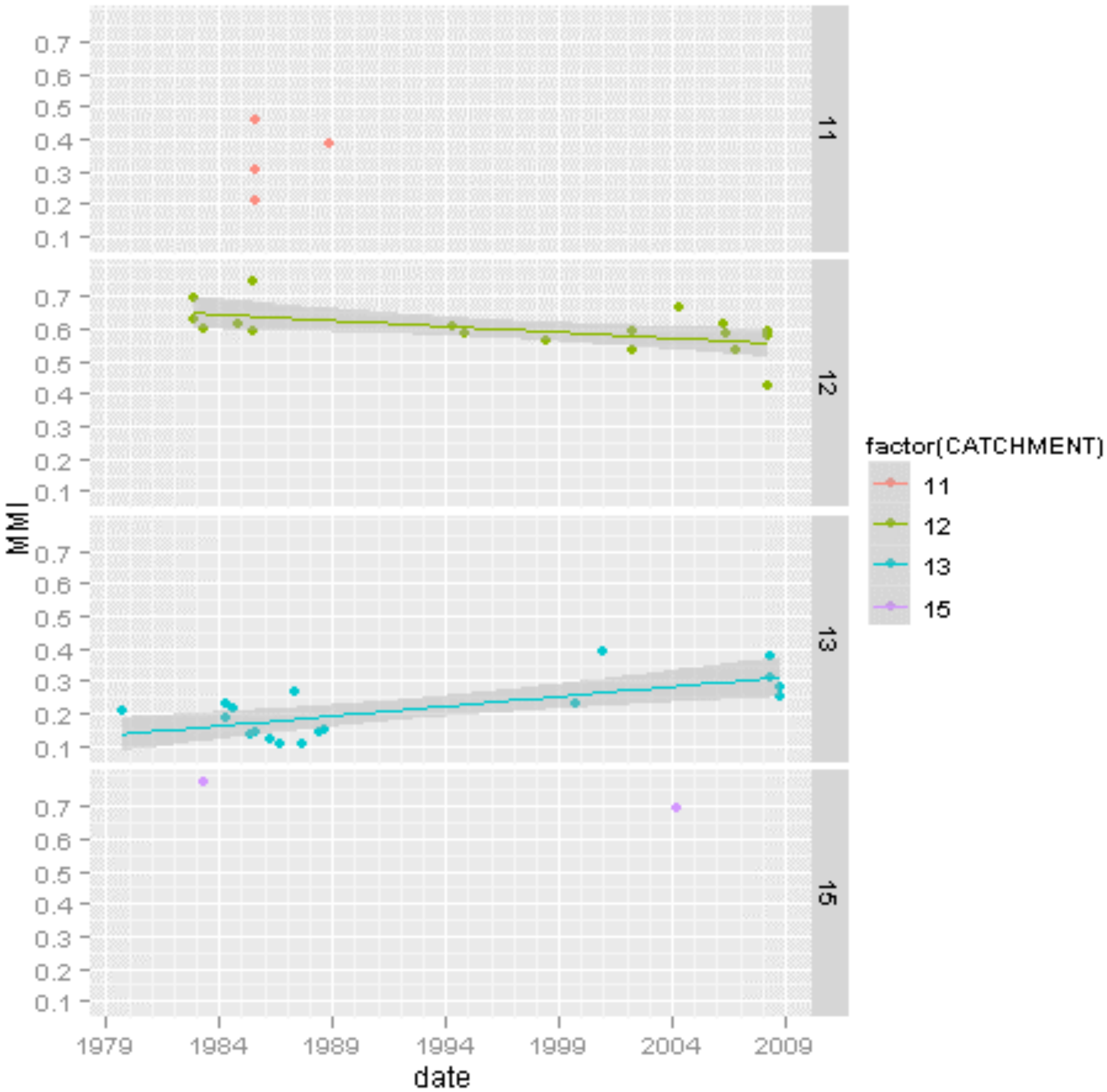

Figure 8. MMI values (ecological condition) over time for catchments 11–15.

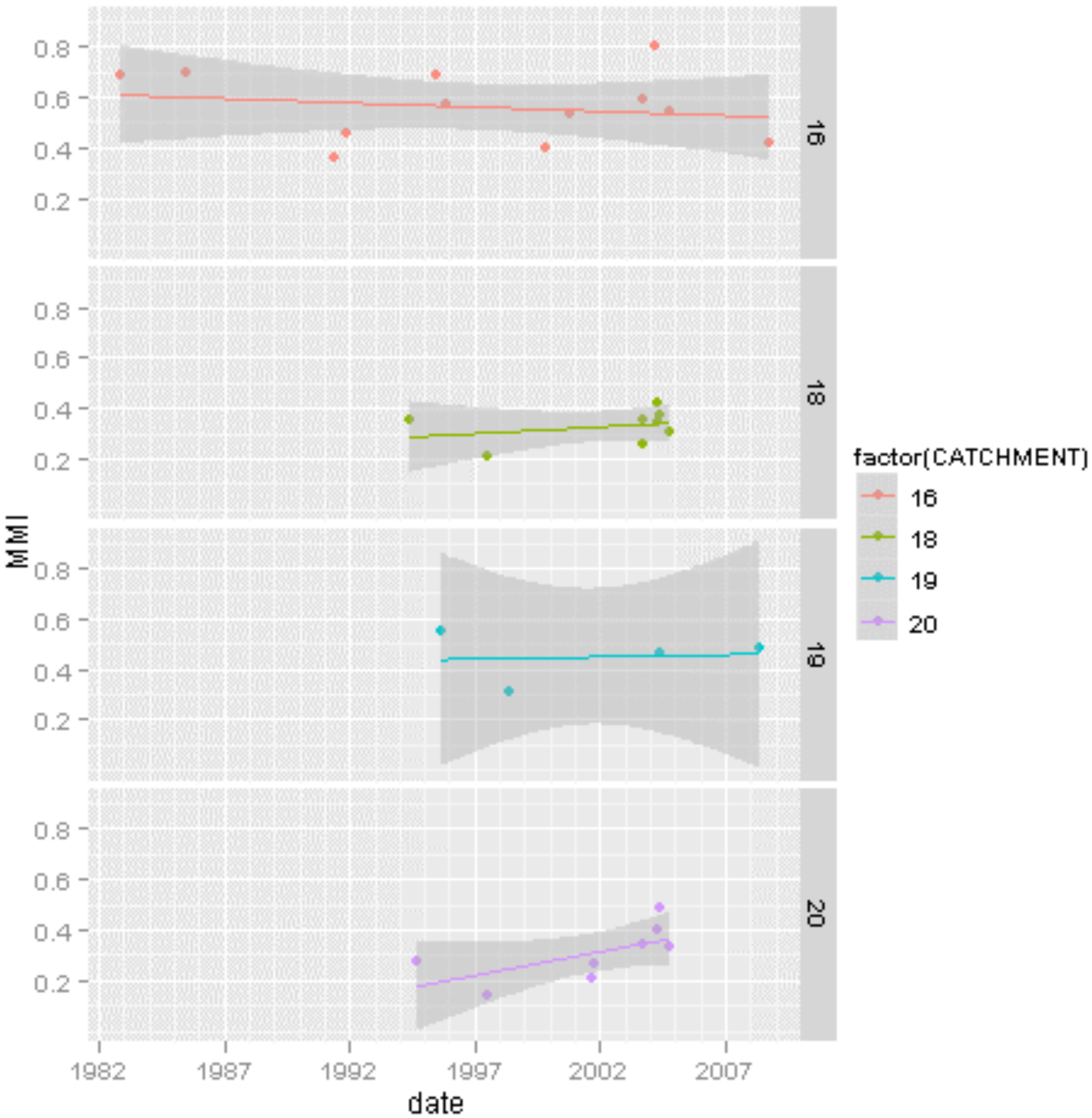

Figure 9. MMI values (ecological condition) over time for catchments 16–20.

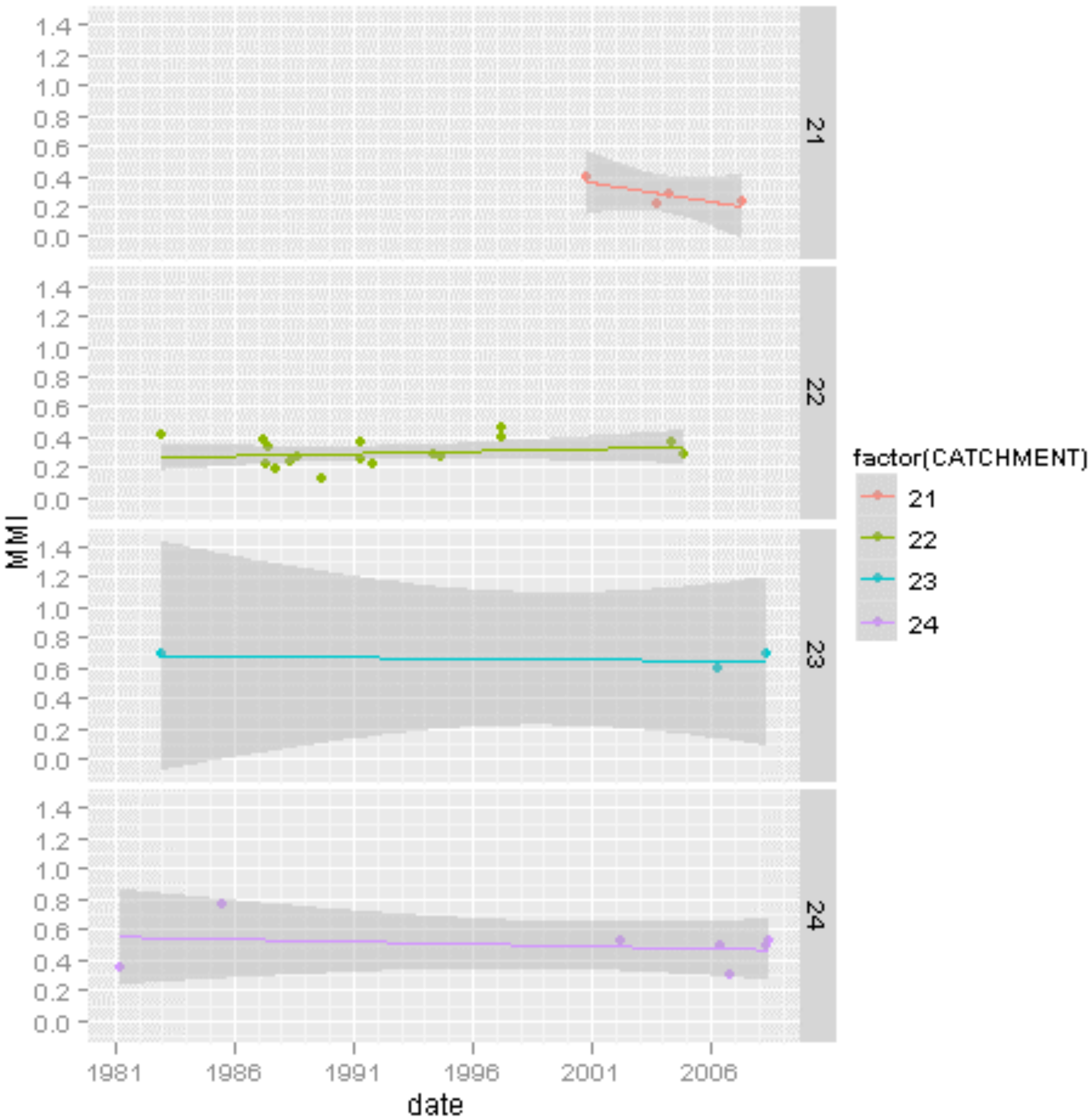

Figure 10. MMI (ecological condition of ecosystems) over time for catchments 21–24.

Discussion - In the majority of the catchments no significant change in MMI values through time could be detected. The lack of change in the ecological status of the different catchments over time was mainly due to the large variation in ecological condition within each catchment. Apparently, sampling sites within a catchment that are located close together, can still differ in ecological condition. This was especially the case in catchments 1 and 2. In three out of 24 catchments we found a significant change in the ecological condition over time. This change, however, was very subtle.

Biological change in time at reach scale in relation to restoration measures and land use

Land use - In figure 11 to 15 the four different land use types for the four different catchments were plotted against time. Land use data were available for seven moments in time (1980, 1988, 1994, 1997, 2000, 2004, and 2008). Figure 11 shows that the percentage of cover for the land use type ‘development’ followed a similar pattern in time for the four different catchments. In 1994 there is a dip in the % coverage ‘development’ and from then onwards there is a slight increase. In 2008 the % coverage ‘development’ levelled off again. For none of the catchments there is a significant correlation with time (Table 6). Figure 12 illustrates the change in % coverage of forested areas and nature areas. Again a similar pattern in time in the percentage of coverage was visible for all four catchments, the % coverage was similar for all years, except in 1988 where the percentage was higher. The % coverage for the land use type ‘forestnat’ was much lower in catchment 13 compared to the other catchments.

There was no clear pattern visible in the % coverage of agriculture for any of the catchments (Figure 13). Percent coverage agriculture was lower in the more southern catchments (8 and 12) than in the northern catchments, 13 and 4 (Figure 13).

The change in % coverage grass showed a similar pattern for the four different catchments (Figure 14). In all the catchments there was a drop in % coverage grass in 1988, followed by an increase in 1994 and then it slowly decreased in the subsequent years (Figure 14).

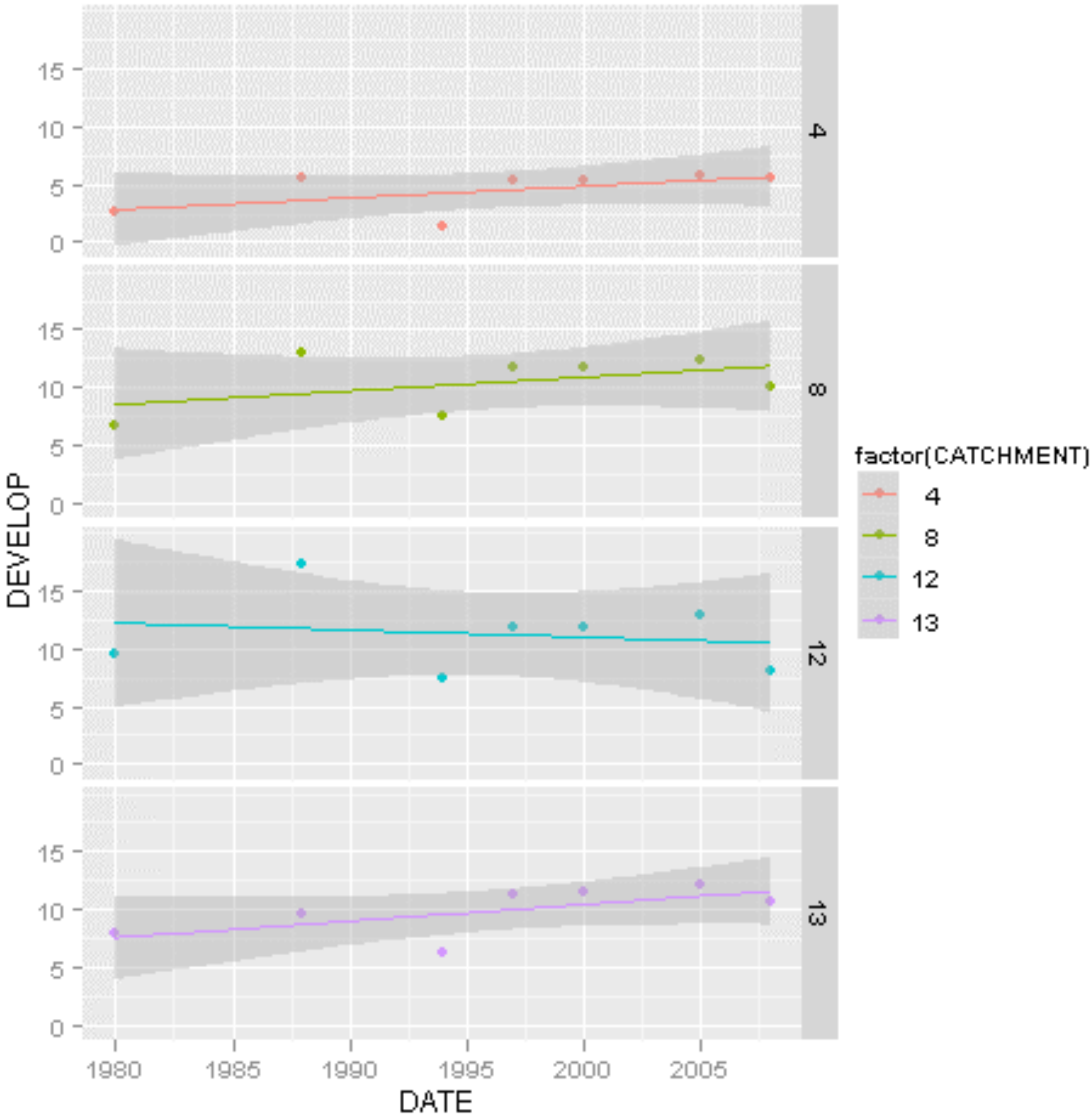

Figure 11. Change in the % coverage of land use type 'development' over time for catchment 4, 8, 12, and 13.

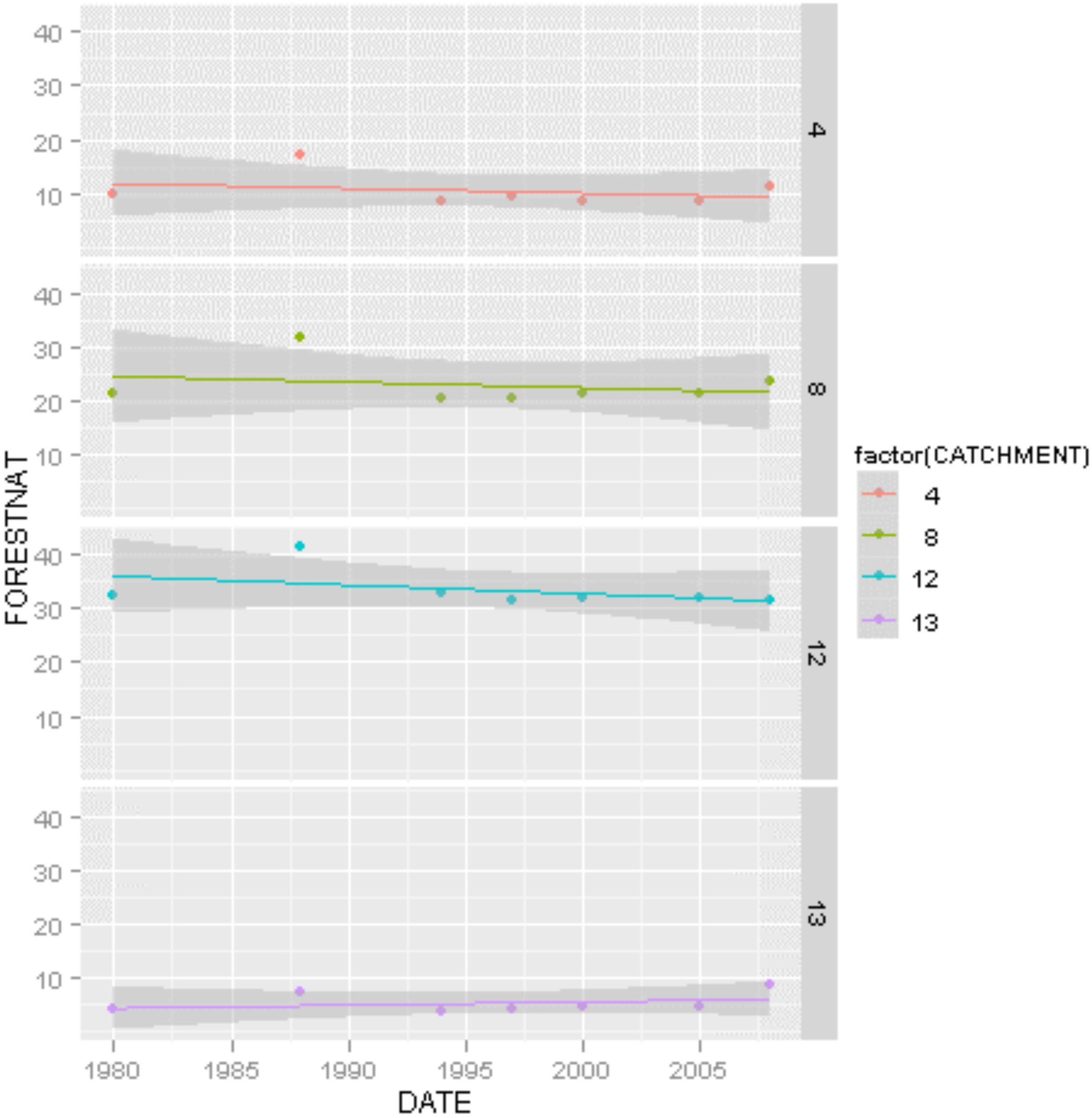

Figure 12. Change in the % coverage of land use type 'forest and nature' over time for catchment 4, 8, 12, and 13.

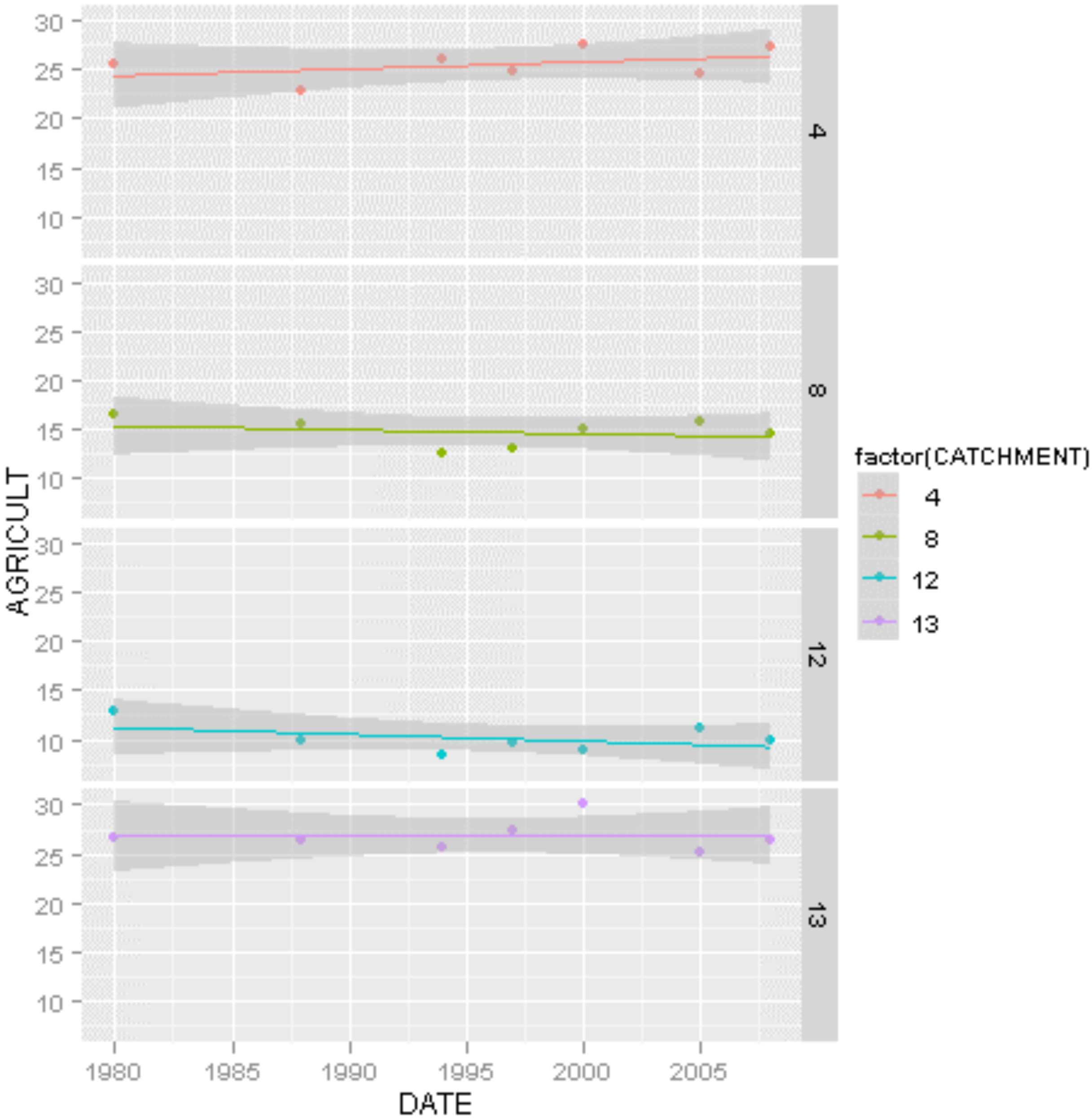

Figure 13. Change in the % coverage of land use type 'agriculture' over time for catchment 4, 8, 12, and 13.

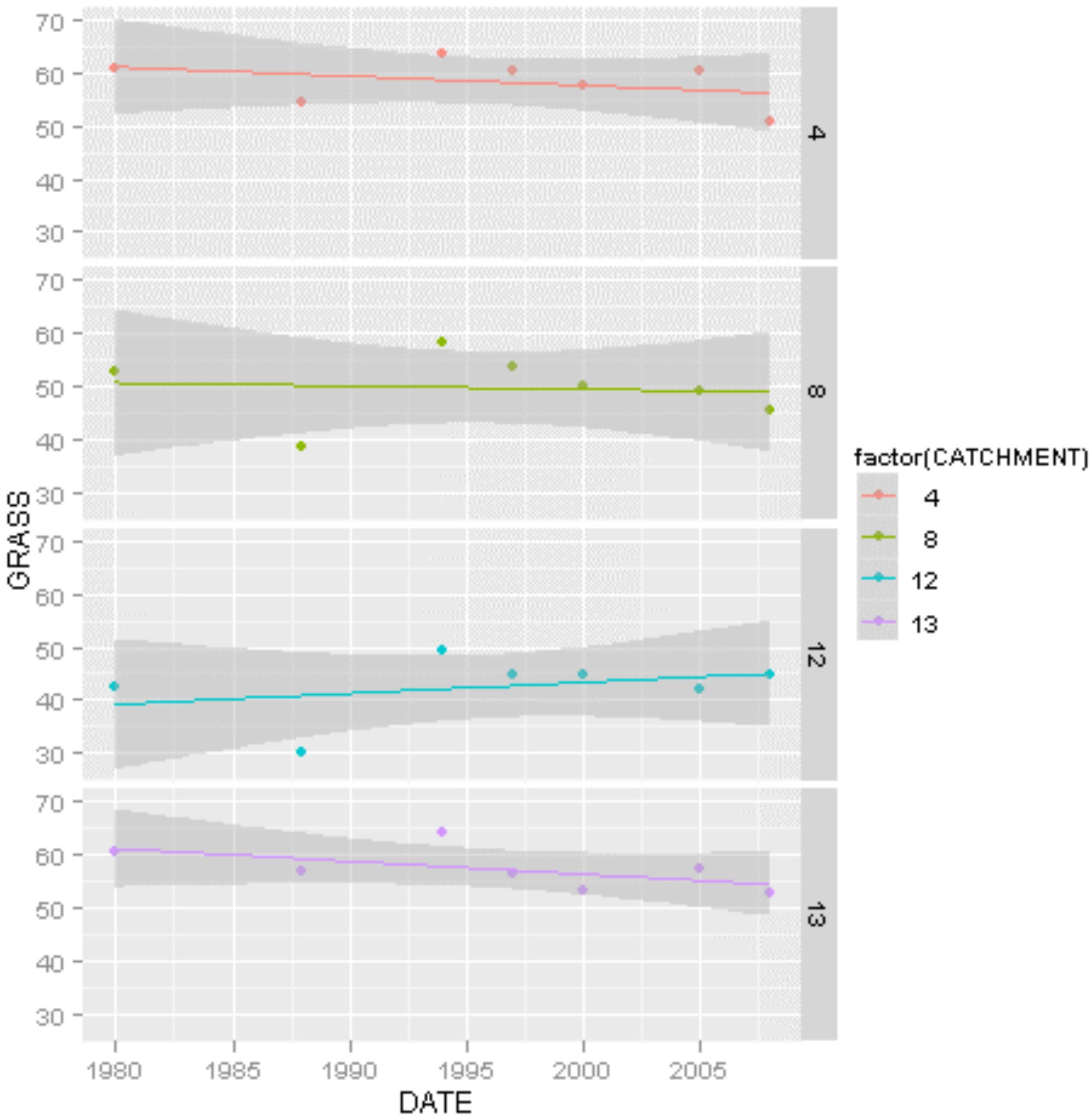

Figure 14. Change in the % coverage of land use type 'grass' over time for catchment 4, 8, 12, and 13.

The percentage of cover by any of the land use categories was not related to time (Table7). Also, MMI values were not related to the percentage of cover by any of the land use categories (Table 8).

Table 7. Pearson's product-moment correlation r for the relationship between the percentage of cover by the four main land use types (development, forest and nature, grass, and agriculture) and time for catchments 13, 4, 8, and 12.

catch.	development		forest and nature		grass		agriculture	
	r	p	R	p	r	p	r	p
13	0.64	0.12	0.30	0.49	-0.57	0.18	0.02	0.97
4	-0.28	0.53	-0.29	0.53	-0.37	0.40	0.40	0.36
8	0.46	0.30	0.25	0.59	-0.09	0.84	-0.27	0.55
12	-0.17	0.71	-0.44	0.32	0.33	0.46	-0.46	029

Table 8. Pearson's product-moment correlation r for the relationship between MMI values and the percentage of cover by the four main land use types (development, forest and nature, grass, and agriculture) for catchments 13, 4, 8, and 12.

catch.	development		forest and nature		grass		agriculture	
	r	p	r	p	r	p	r	p
13	0.65	0.12	0.31	0.50	-0.57	0.18	0.02	0.97
4	0.56	0.19	-0.29	0.53	-0.38	0.40	0.41	0.37
8	0.46	0.30	-0.25	0.59	-0.09	0.84	-0.27	0.55
12	0.17	0.71	0.44	0.32	-0.34	0.46	0.47	0.29

Restoration measures - In 2005 several hydromorphological measures were implemented in the Linderbeek (catchment 13). The driving factors behind the measures were maintenance and management reasons (safety and limited discharge capacity). To repair the failing slopes a constant water level was introduced as well as natural banks. To deal with the limited discharge capacity the river was widened. When we compared the MMI values from before the hydromorphological measures with the MMI values after the hydromorphological measures we found a significant difference (Table 9). The average MMI value after the restoration measures was higher than the average MMI value before the restoration measures (Table 9).

Between 2006 and 2007 in the Hazelbeek a range of ecohydrological measures were implemented, e.g. reduction of manure input, construction of retention basins, and reprofiling from related streams (Table 9). Although there was a significant increase in MMI values over time in catchment 4 (Hazelbeek) (Figure 6), there was no significant difference between MMI values before and after the ecohydrological measures.

In the river basin of the 'Azelerbeek' (catchment 8) several restoration measures were taken during the last decennia. They were all taken in the 'Oelerbeek', which is the upper reach of the 'Azelerbeek'. In 1996 the discharge of the STP 'Enschede-zuid' in the Oelerbeek, a stream that changes into the Azelerbeek, stoped. In 1997 another discharge point in the Oelerbeek was closed. In 2002 changes in water allocation in the Oelerbeek were introduced. In 2005 and 2006 three other STP closed (STP, 'Texoprint', STP 'Delden', STP 'Boekelo'). Because there was only one sample location that was sampled before and after the restoration measures we couldn't conduct an ANOVA on these data.

Table 9. ANOVA results for differences in MMI values before and after restoration measures.

catchment	stream	end-date project	Type measure	df	F	p
12	Drienerbeek	-				
13	Linderbeek	2005	Hydromorphological measures	1, 15	7.63	**0.01**
4	Hazelbeek	2006-2007	Ecohydrological measures	1, 19	0.17	0.68

Discussion- No significant change in the % coverage of the different land use types was found for the four selected catchments. Although, we could not detect linear change in % coverage over time, there seemed to be differences in % coverage between years. However, we suspect these changes are an artefact. During analysis we saw buildings disappearing from the map between 1987 and 1994 and then reappearing in 1997, which seemed strange. The apparent differences in land use coverage between years might be explained by the fact that data from 1980 were based only on topographic maps, and the quality of areal imagery has improved during the years. The fact that no changes in land use over time could be detected in this study is in accordance with the fact that major land use shifts in the Netherlands occurred before 1980.

Since land use showed no change over time in the four selected catchments, logically, no relationship was detected between the % coverage of land use types and MMI values. Furthermore, working with predicted values rather than measured values, lowers the strength of this test and would only detect very strong patterns.

Table 10 gives an overview for catchments 13, 4, 8, and 12 of significant change in MMI values over time, significant differences in MMI values before and restoration, and significant change in the percentage of coverage by the four land use types. Since no major changes in land-use cover took place in the different catchments we can assume the effectiveness of restoration measures has not been affected by changes in land use. Based on ANOVA results it seems like restoration measures taken within catchment 13 in 2005 have improved ecological condition of the catchment (Table 10), however, this is an artefact. During the period prior to restoration four samples were collected in the 1980's, six in 1995, one in 1999, and four in 2002 (Annex 1). When we studied the graphs with MMI values by eye, it appeared MMI values already changed in 2002 (Annex 1), but because so many samples were taken before 2005 this didn't affect analysis results. Also, only four samples were taken after restoration took place. It is the question with these kind of analyses how many years pre- and post- restoration should be or can be included. In our view it is better to visually study the graphs with MMI values. Finally, it should be kept in mind that when significant differences do occur these might not have been caused by the studied restoration measures. To rule out other causes of the improvement in ecological condition data on as many environmental variables as possible from the same time period should be studied.

Biological change in space in relation to land use

Catchment scale – Land-cover composition differed between catchments. Urban land-cover varied between 1 and 71% (Figure 15), land-cover by forest between 3 and 41% (Figure 16), by agricultural grassland between 18 and 61 (Figure 17), by agriculture between 5 and 31% (Figure 18), by nature between 0 and 22% (Figure 19) , and by freshwater between 0 and 6% (Figure 20). EQR values were not related to the percentage of cover by any of the land use categories (Table 11).

Table 10. Overview for catchments 13, 4, 8, and 12 of significant change in MMI values over time, significant differences in MMI values before and restoration, and significant change in the percentage of coverage by the four land use types. Dev. = development and f, n = forest and nature.

catchment	significant change in MMI over time	significant change in land use				restoration	significant difference before and after restoration
		dev	f,n	grass	agriculture		
13	yes (+)	no	no	no	no	yes	yes(+)
12	yes (-)	no	no	no	no	no	-
8	no	no	no	no	no	yes	-
4	yes (+)	no	no	no	no	yes	no

Table 11. Spearman rank correlation coefficients for the relationship between the average ecological condition ratio value and the percentage cover of six different land use categories in the catchments within the management district of regional water authority Regge en Dinkel (N=38).

land use category	Spearman's correlation coefficient	p-value
urban	-0.048	0.773
forest	0.022	0.898
agricultural grassland	0.124	0.459
agriculture	-0.092	0.584
nature	0.114	0.494
freshwater	-0.263	0.111

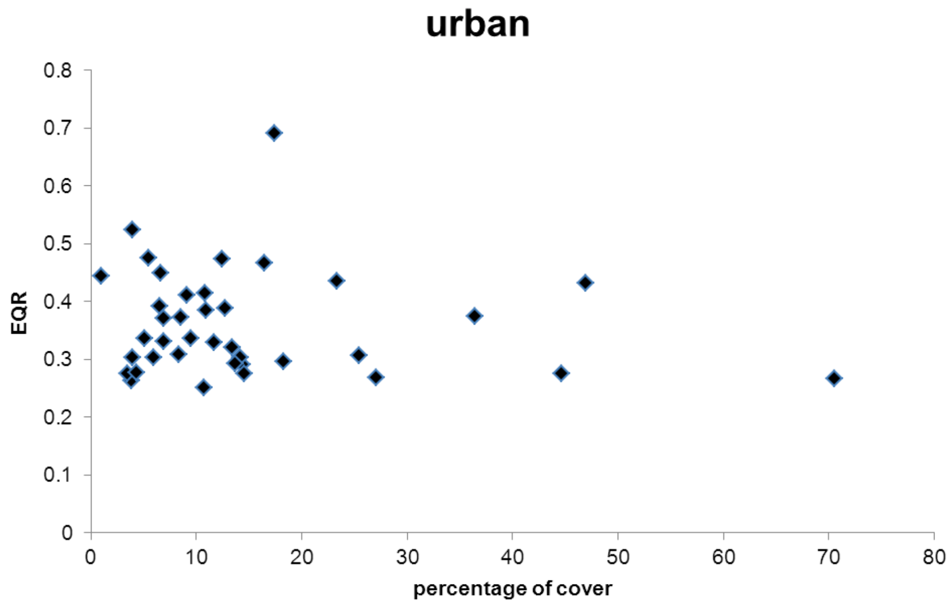

Figure 15. Relationship between the average EQR value and the percentage of urban land cover for catchments within the district of regional water authority Regge en Dinkel (N=38).

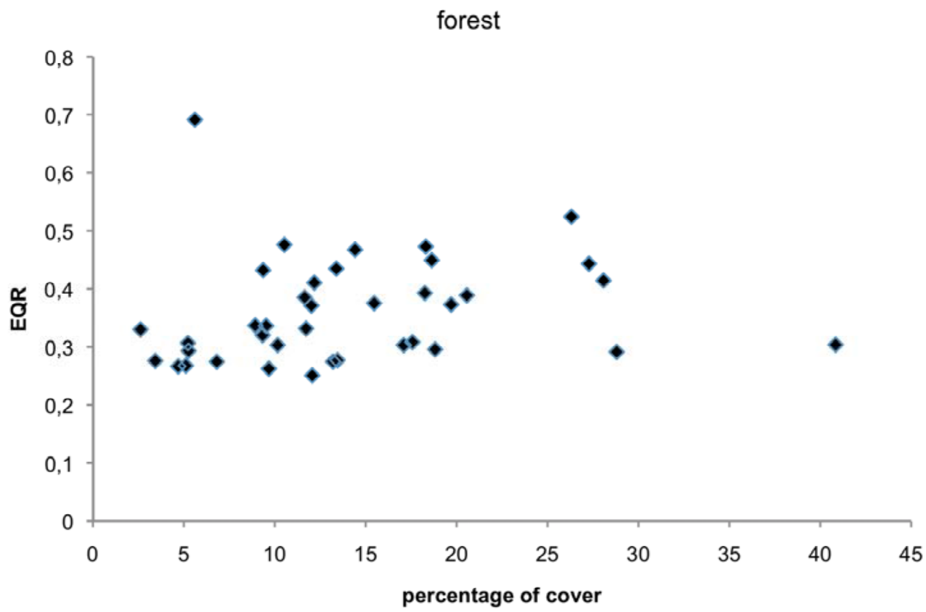

Figure 16. Relationship between the average EQR value and the percentage of forest land cover for catchments within the district of regional water authority Regge en Dinkel (N=38).

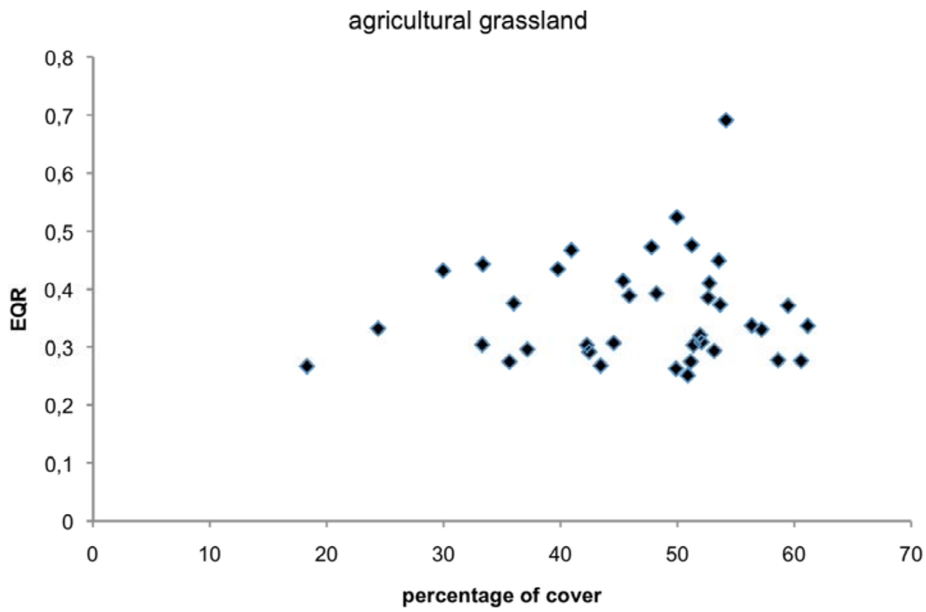

Figure 17. Relationship between the average EQR value and the percentage of agricultural grassland land cover for catchments within the district of regional water authority Regge en Dinkel (N=38).

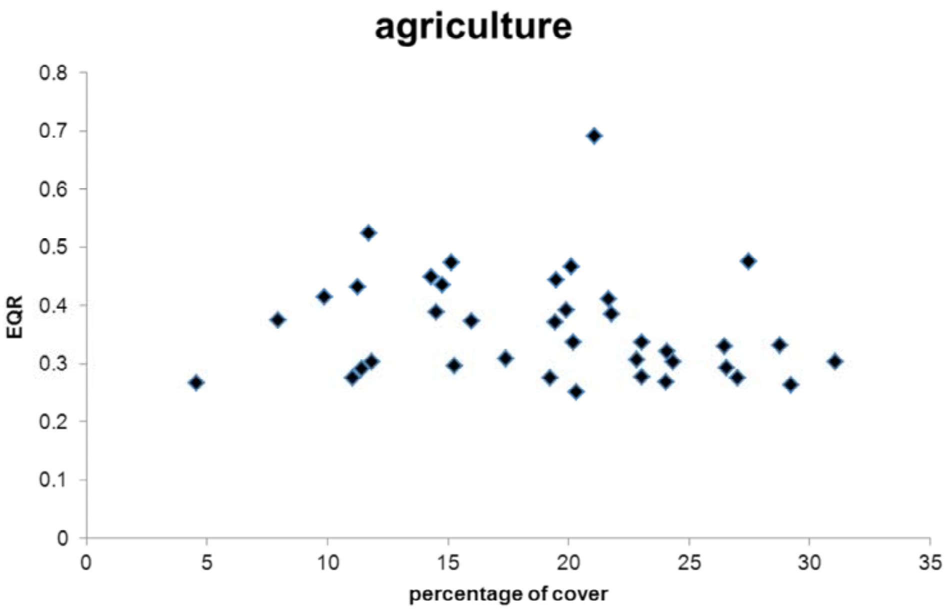

Figure 18. Relationship between the average EQR value and the percentage of agricultural land-use for catchments within the district of regional water authority Regge en Dinkel (N=38).

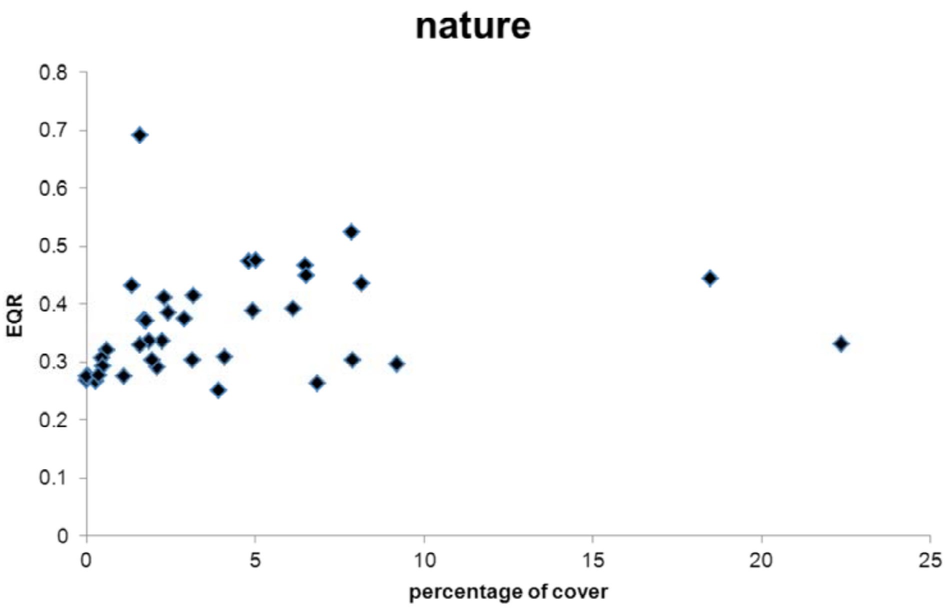

Figure 19. Relationship between the average EQR value and the percentage of natural land cover for catchments within the district of regional water authority Regge en Dinkel (N=38).

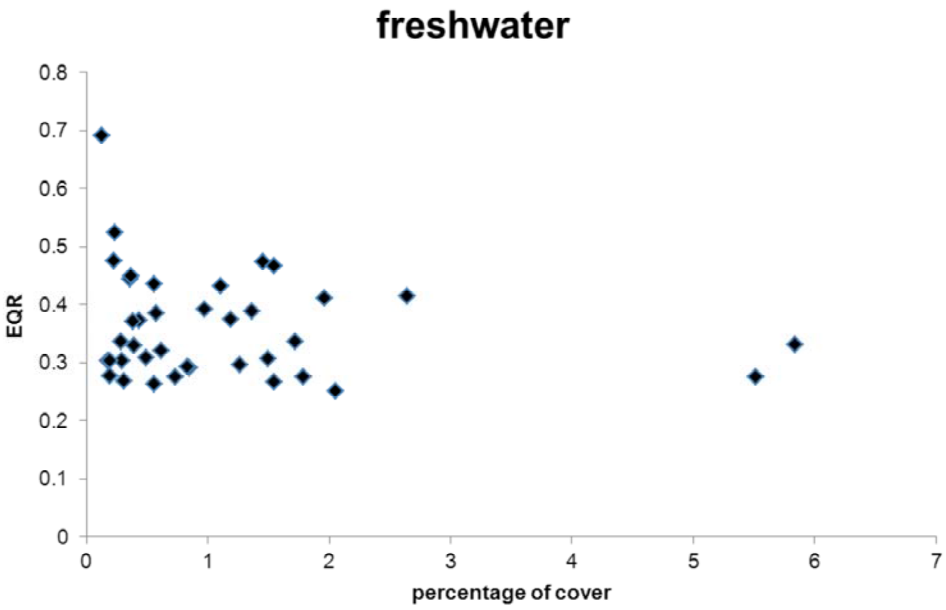

Figure 20. Relationship between the average EQR value and the percentage of freshwater cover for catchments within the district of regional water authority Regge en Dinkel (N=38).

Site scale – Land-cover composition differed between sites. The percentage of land-cover varied between 0 and 100% for all six categories of land use except for freshwater (Figure 21–26). EQR values were significantly related to the percentage of cover for all six land use categories (Table 12). Relationships, however, were weak. The strongest relationships between the percentage of cover and EQR values existed for the land use categories forest (r=0.369) and nature (r=0.305) (Table 12). When we combined the categories forest and nature the relationship with EQR values improved (r=0.415).

Table 12. Spearman rank correlation coefficients for the relationship between the average ecological condition ratio value and the percentage cover of six different land use categories for sites (macroinvertebrate sample locations) within the district of regional water authority Regge en Dinkel (N=520).

land use category	Spearman's correlation coefficient	p-value
urban	-0.156	0.000
forest	0.369	0.000
agricultural grassland	-0.163	0.000
agriculture	-0.163	0.000
nature	0.305	0.000
freshwater	-0.181	0.000
forest and nature	0.415	0.000

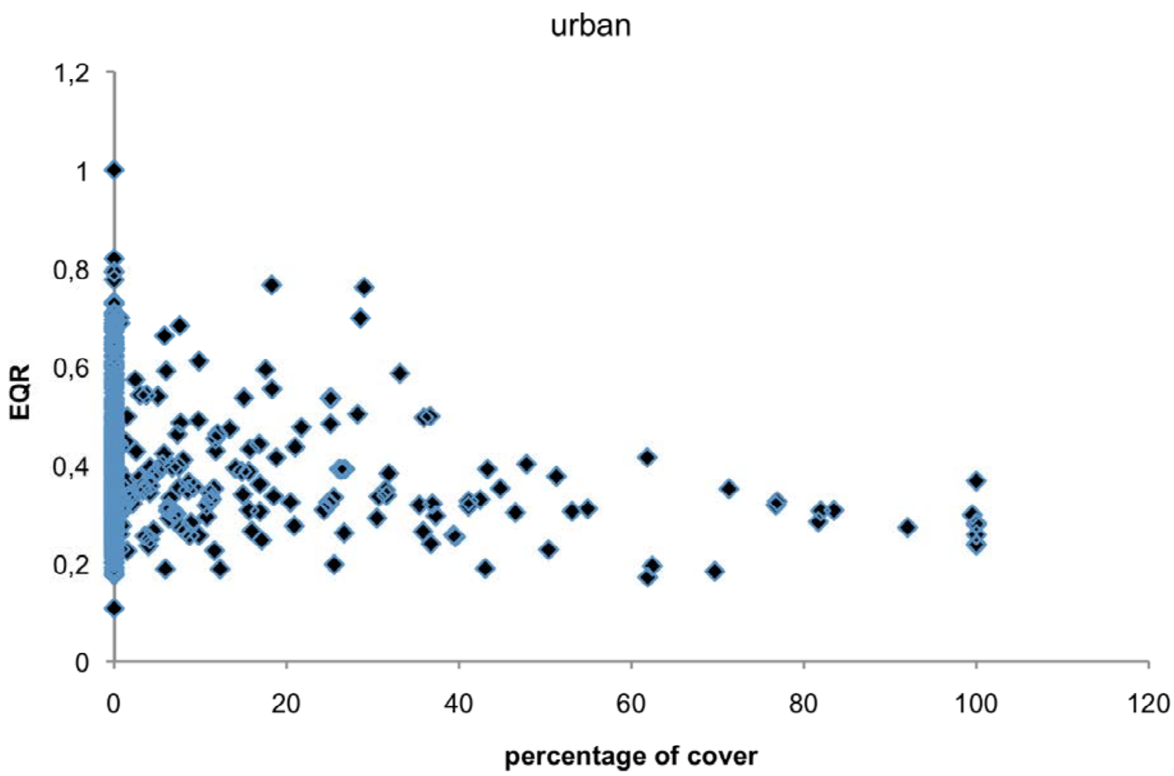

Figure 21. Relationship between the average EQR value and the percentage of urban land-use cover for sites (circular buffer with a diameter of 25 m) within the district of regional water authority Regge en Dinkel (N=520).

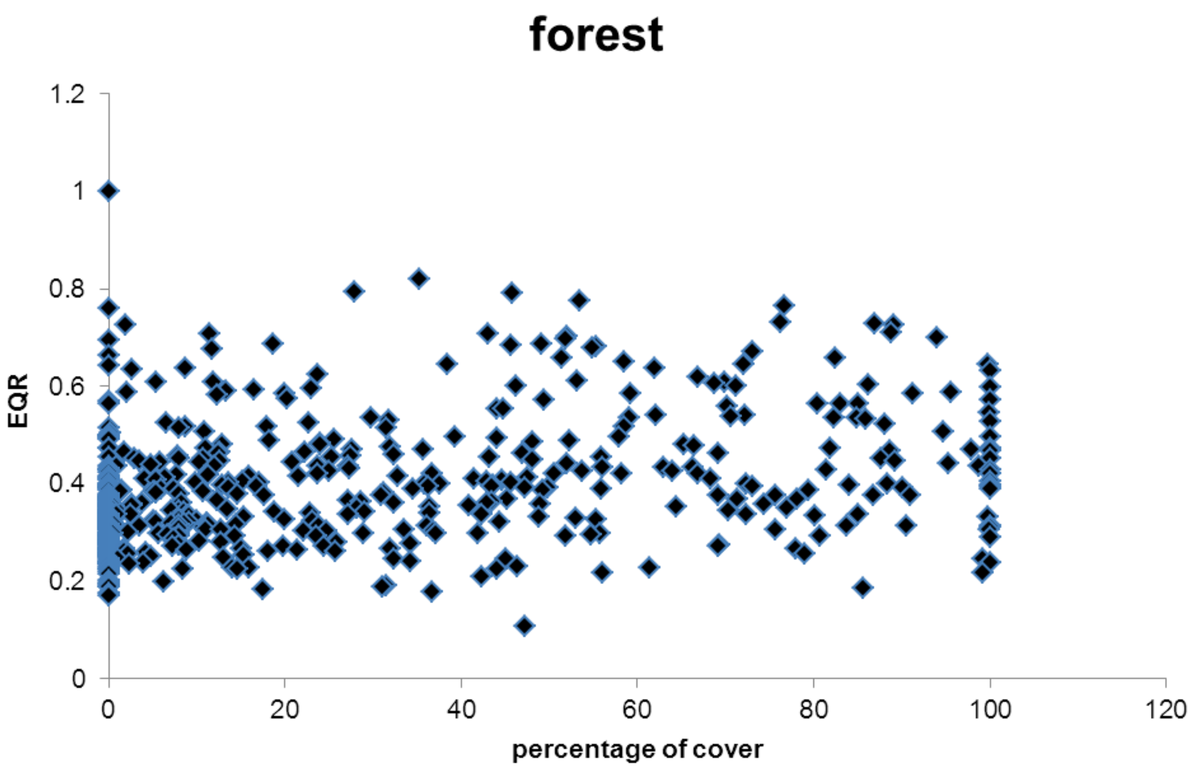

Figure 22. Relationship between the average EQR value and the percentage of forest land-use cover for sites (circular buffer with a diameter of 25 m) within the district of regional water authority Regge en Dinkel (N=520).

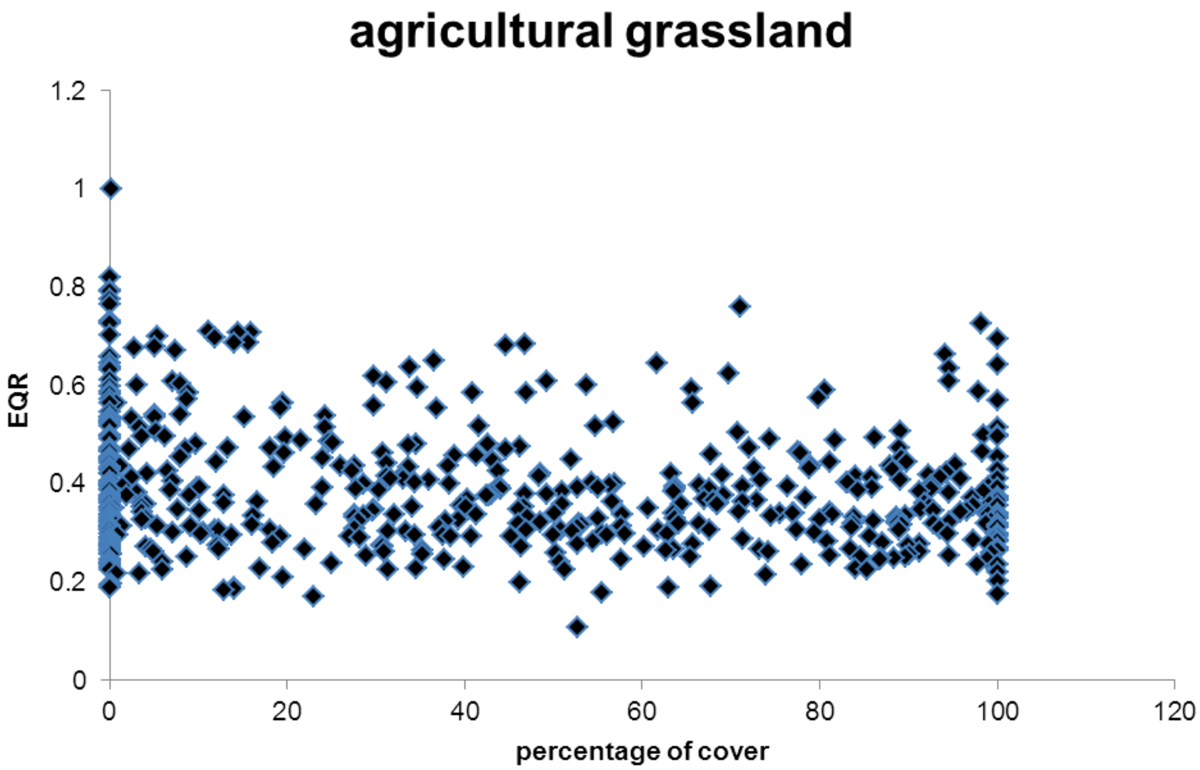

Figure 23. Relationship between the average EQR value and the percentage of agricultural grassland land-use cover for sites (circular buffer with a diameter of 25 m) within the district of regional water authority Regge en Dinkel (N=520).

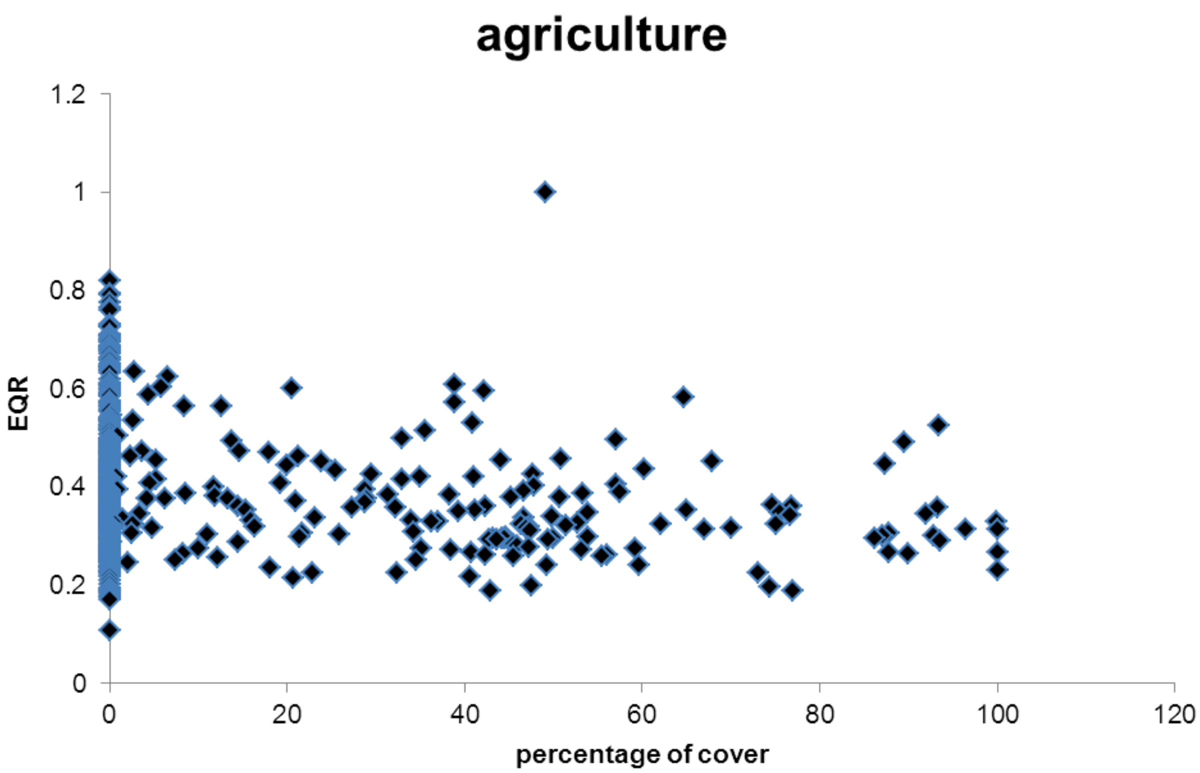

Figure 24. Relationship between the average EQR value and the percentage of agriculture grassland land-use cover for sites (circular buffer with a diameter of 25 m) within the district of regional water authority Regge en Dinkel (N=520).

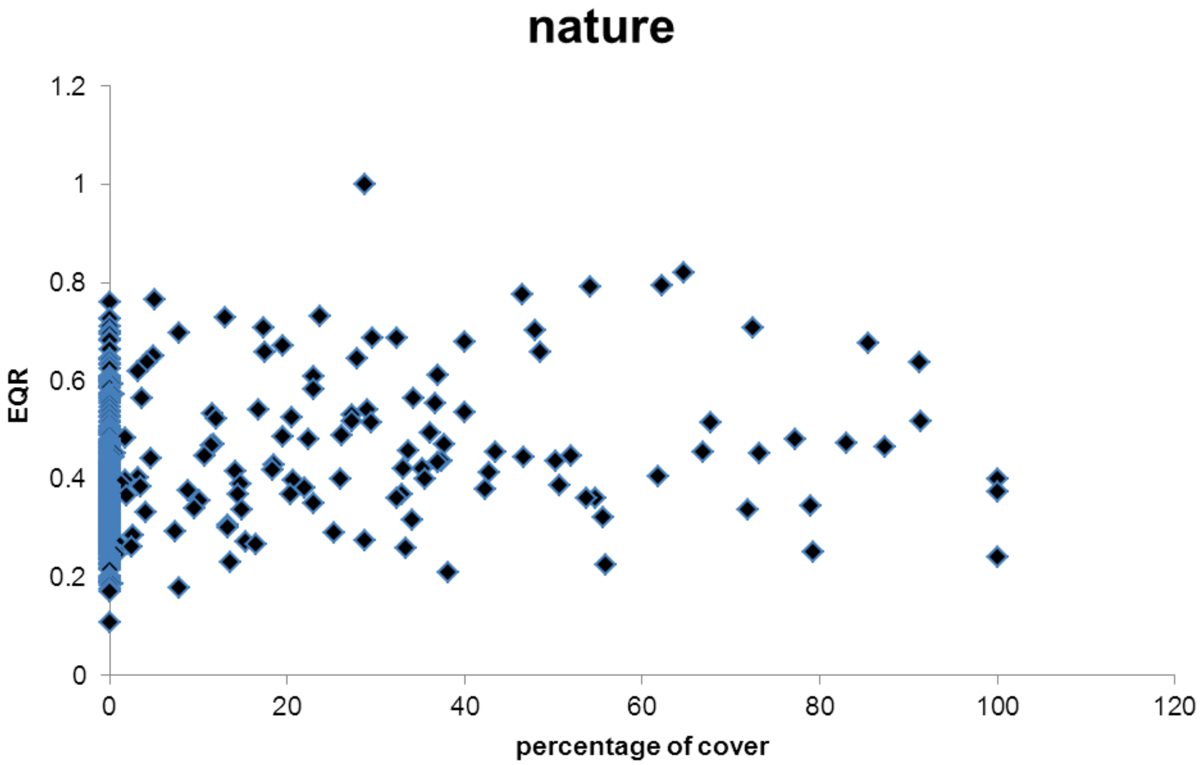

Figure 25. Relationship between the average EQR value and the percentage of nature land-use cover for sites (circular buffer with a diameter of 25 m) within the district of regional water authority Regge en Dinkel (N=520).

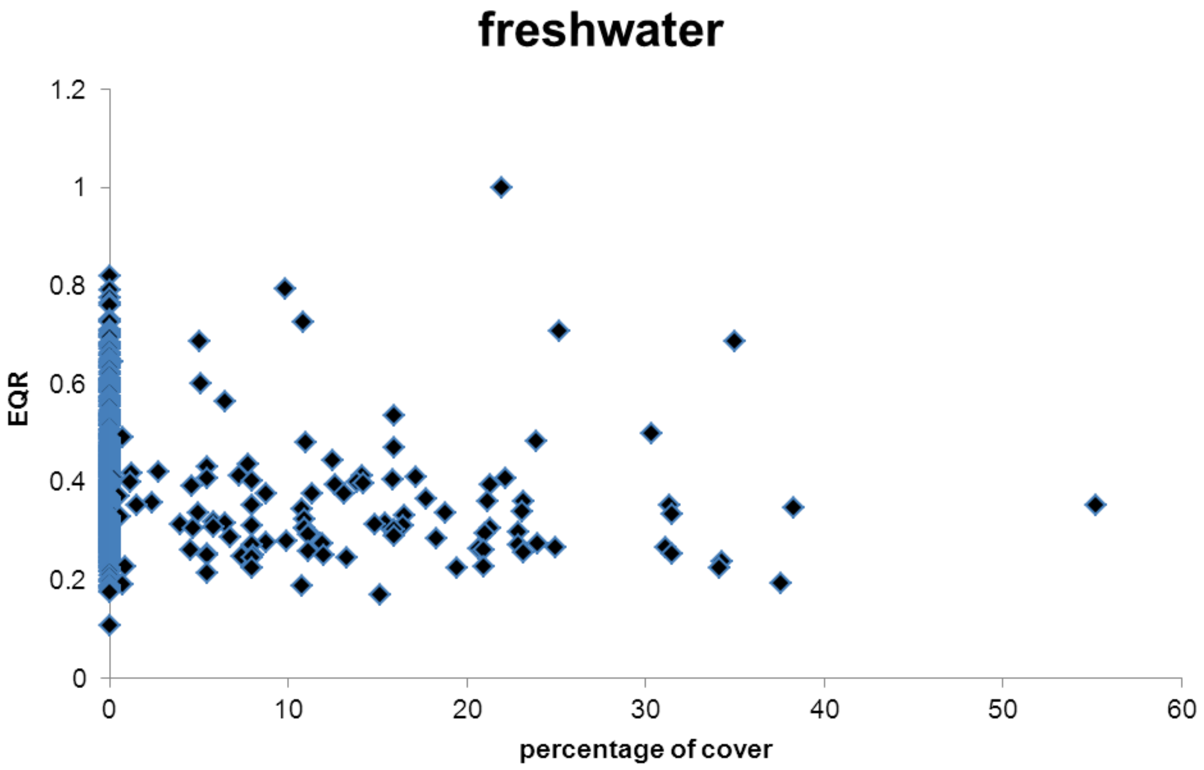

Figure 26. Relationship between the average EQR value and the percentage of freshwater land-use cover for sites (circular buffer with a diameter of 25 m) within the district of regional water authority Regge en Dinkel (N=520).

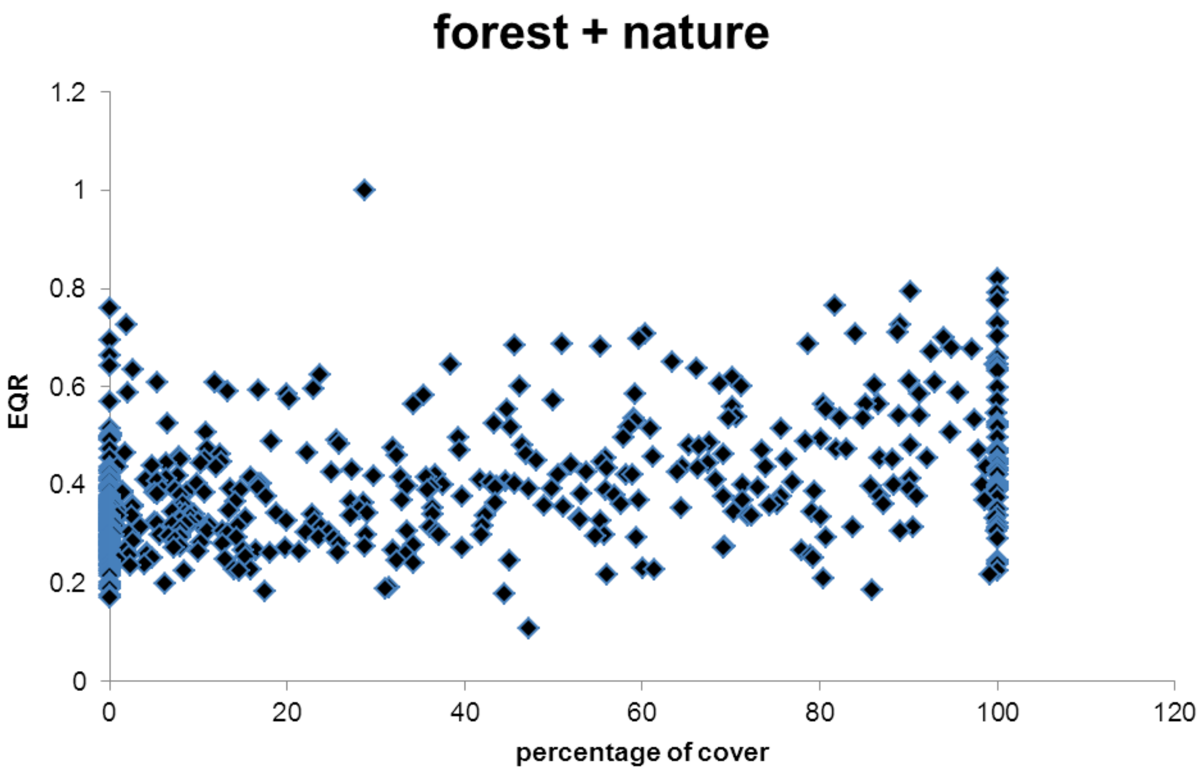

Figure 27. Relationship between the average EQR value and the percentage of forest and nature cover for sites (circular buffer with a diameter of 25 m) within the district of regional water authority Regge en Dinkel (N=520).

Discussion – We detected a weak positive relationship between the percentage of cover by forest and nature within a radius of 25 m around a site and ecological water quality at that site. However, we found no relationship between any of the land use categories and ecological water quality at catchment level.

We didn't necessarily expect a linear relationship between land-use cover and ecological condition. However, we did expect to find threshold values, as did others (e.g., Wang et al. 1997, Quinn 2000, Fitzpatrick et al. 2001, Russell and Collier. 2010). In our study we could detect no threshold values for any of the land use categories, both at site and catchment scale. It is possible that the percentage of cover by urban and/or agricultural land use in our study already exceed some threshold, but not likely given the range in cover studied. More likely, the relationship between ecological condition and land-use cover is too complex for a single threshold to apply (Allan 2004)

Both the lack of thresholds and relationships between land-use cover and ecological condition can have several causes: 1. Our dataset included fast flowing streams (>30 cm/sec), slow flowing streams (<30 cm/sec), headwater streams, and up to 25 m wide streams. These differences in natural environmental variables might have confounded results. 2. For the purpose of this study we used an existing dataset of a regional water authority. As a result number of samples and locations varied per catchment, and sample locations were not selected randomly (as a result sample locations might not reflect ecological condition of the catchment). Both factors might have confounded results. 3. Other environmental variables that were not included in the study might have confounded results. For example, the effect of upstream hydromorphological conditions on local ecological status (Kail & Hering 2009).

We suspect that the correlation between the percentage of cover by forest and ecological condition at site scale detected in this study is directly related to the amount of shading and/or morphological degradation. In general forested sections in the Netherlands are less likely to be channelized/normalized than un-forested sections. Other studies have indicated that forested stream sections are characterized by cooler temperatures, wider channels, fewer sediments, and greater diversity of invertebrates (e.g., Sweeney 1993, Quinn et al. 1997, Abell & Allan (2002).

To answer the question whether land-use cover can be used as indicator of ecological condition in streams more extensive analyses are required. For example, multivariate analyses should be performed to see how much variation can be attributed to land use at different spatial scales, also in comparison to other environmental variables. In this study we used a multimetric index developed by Verdonschot and Verdonschot (2010) as a composite measure to study overall change in ecological condition, however it might be easier to interpret the behaviour of individual/less aggregated response variables (Watzin & McIntosh 1999) and they might prove to be more useful in evaluating mechanisms and pathways (Poff 1997, Usseglio-Polaterra et al. 2000). This issue should also be addressed in future analyses.

Discussion

Multivariate analyses showed there were no major changes in macroinvertebrate community composition between 1980 and 2008 within the three major catchments of the water management district of the regional water authority Regge en Dinkel. To reduce spatial variation we analyzed data at reach scale (within catchments) to see if we could detect change in ecological condition.

Instead of working time series of single sampling locations, we worked with time series of catchments, i.e. a collection of sample sites that are located close together. Through this upscaling we could work with longer time series. Although this study set out to look at the effects of land use and river restoration for all four BQE’s it appeared the dataset was not suited for this goal. Especially at the reach and site scale long-term datasets were not available for fish, phytobenthos and macrophytes. We expect that in the near future more data will become available from BQE’s other than macroinvertebrates, because the WFD forces regional water authorities to sample all BQE’s for the purpose of surveillance monitoring. However as long as regional water authorities do not use a monitoring frequency of at least once a year (surveillance monitoring has to take place with a minimum frequency of only once every six years), it will remain difficult to detect change within a time span of several years, especially given the high variability of ecological data.

In the majority of the catchments no significant change in ecological condition through time could be detected. The lack of change in the ecological status of the different catchments over time was mainly due to the large variation in ecological condition within each catchment. Apparently, sampling sites within a catchment that are located close together, can still differ in ecological condition. In three out of 24 catchments we did find a significant change in the ecological condition over time. This change, however, was very subtle. The change in metric values was in all three cases less than 0.2 (0.2 equals the range of an ecological condition class).

Further analyses of macroinvertebrate data showed that even these data were not suited to study the effects of restoration measures. This is mainly due to the fact that in the Netherlands regional water authorities hardly ever use an experimental design (with samples from restored and unrestored locations before and after restoration) to monitor the ecological impact of restoration measures. Given the fact that millions of Euros are being spent on restoration of streams the lack of (proper) monitoring data necessary to evaluate the ecological effectiveness of restoration measures is astonishing. Although, river restoration is getting increased attention in many parts of the world (Palmer et al. 2007), the lack of proper monitoring data seems to be a general problem. Bernard et al. (2005) stated that of 37.000 river restoration projects in the United States, only 10% included some form of monitoring, and of this information little was either appropriate or available. Feld et al. (2006) conducted a study on the effectiveness of several hundreds of ecological river restoration projects in Germany. They found that for less than a quarter of the studies (23%), post-project evaluation of measures had been conducted.

Our analyses showed no major changes in land use took place between 1980 and 2008 within the water management district of regional water authority Regge en Dinkel. Therefore, the only way to study the relationship between land use/cover and ecological condition of streams at different spatial scales, was to use space for time substitution. Results showed a weak positive relationship between the percentage of cover by forest and nature and ecological condition site scale. These findings are completely opposite to those of Wang et al. (1997), who found stronger correlations at watershed level compared to a more local level (100-m buffer, upstream entire watershed). Also, Russell and Collier (2010) showed catchment-level measures of indigenous forest were more strongly linked with ecological condition indices than the segment- or habitat-level variables. Stewart et al (2001) detected a positive relationship between ecological condition (Hilsenhoff biotic index and the number of EPT species) and the percentage of cover by forest at both the watershed and riparian scale. Stewart al. (2001) mentioned several possible reasons for these apparent differences in results between studies: differences in resolution and age of land cover data, the scale and extent of the stream network used for riparian-corridor analysis, and whether riparian land cover is summarized as part of watershed cover, or if watershed and riparian land cover are summarized separately. Weigel et al. (2003) also noted that the results of studies comparing multiple spatial scales will vary depending on the area studied, sites included in the study, and the selected predictor and response variables.

Lammert and Allan (1999) found that land use within 100-m of the stream was significantly related to biotic integrity. However, they didn't provide a conclusive answer to the question whether local or catchment wide factors have more impact on biotic integrity than local factors. They indicated this will largely depend on the study design. In the study by Lammert and Allan (1999) differences in land use cover at catchment scale were relatively small: 17 to 23% for forest, and 42 to 66% for agriculture. In our study differences in land use cover were also smaller at the catchment scale compared to the site scale. Therefore, we cannot rule out that we did find significant relationships at site scale opposed to catchment scale simply because variation in land cover percentages was higher at the site scale than at the catchment scale. Allan (2004) has suggested that the greater influence attributed to riparian land use (as opposed to catchment land use) in many studies, might be (partly) resulting from the fact that variation in land use is often greater at riparian and reach scales opposed to catchment scale.

We didn't necessarily expect a linear relationship between land-use cover and ecological condition. However, we did expect to find threshold values, as did others (e.g., Wang et al. 1997, Quinn 2000, Fitzpatrick et al. 2001, Russell and Collier 2010). In our study we could detect no threshold values for any of the land use categories, both at site and catchment scale. It is possible that the percentage of cover by urban and/or agricultural land use in our study already exceed some threshold, but not likely given the range in cover studied. More likely, the relationship between ecological condition and land-use cover is too complex for a single threshold to apply (Allan 2004)

Both the lack of thresholds and relationships between land-use cover and ecological condition can have several causes: 1. Our dataset included fast flowing streams (>30 cm/sec), slow flowing streams (<30 cm/sec), headwater streams, and streams up 25 m wide. These differences in natural environmental variables might have confounded results. 2. For the purpose of this study we used an existing dataset of a regional water authority. As a result number of samples and locations varied per catchment, and sample locations were not selected randomly (as a result sample locations might not reflect ecological condition of the catchment). Both factors might have confounded results. 3. Other environmental variables that were not included in the study might have confounded results. For example, the effect of upstream hydromorphological conditions on local ecological status (Kail and Hering 2009). 4. The percentage of cover by the different land use type was determined for the entire catchment and sites were linked to a catchment. However, when a site is located upstream the influence of land-use downstream is probably negligible.

We suspect that the correlation between the percentage of cover by forest and ecological condition at site scale detected in this study, is directly related to the amount of shading and/or morphological degradation. In general forested section in the Netherlands are less likely to be channelized/normalized than un-forested sections. Other studies have indicated that forested stream sections are characterized by cooler temperatures, wider channels, fewer sediments, and greater diversity of invertebrates (e.g., Sweeney 1993, Quinn et al. 1997, Abell and Allan 2002).

To answer the question whether land-use cover can be used as indicator of ecological condition in streams more extensive analyses are required. For example, multivariate analyses should be performed to see how much variation can be contributed to land use at different spatial scales, also in comparison to other environmental variables. In this study we used a multimetric index developed by Verdonschot and Verdonschot (2010) as a composite measure to study overall change in ecological condition, however it might be easier to interpret the behaviour of individual/less aggregated response variables (Watzin & McIntosh 1999) and they might prove to be more useful in evaluating mechanisms and pathways (Poff 1997, Usseglio-Polaterra et al. 2000). This issue should also be addressed in future analyses.

Conclusions

No major change in macroinvertebrate community composition took place between 1980 and 2008 within the three major catchments of the water management district of regional water authority Regge en Dinkel.

No major changes in land-use cover took place between 1980 and 2008 within the water management district of regional water authority Regge en Dinkel.

In most reaches no changes in ecological condition at reach level took place between 1980 and 2008 within the water management district of regional water authority Regge en Dinkel. When change did occur it was very subtle, less than one ecological condition class.

The available monitoring data were not suited to determine the effectiveness of river restoration measures performed within the water management district of the regional water authority Regge en Dinkel.

Results from our study suggest that the introduction of riparian cover in the form of trees in a buffer of 25 m next to the stream might improve the ecological condition of the stream. However, more research is necessary to determine the relative contribution of cover by forest and nature to the ecological condition of a stream opposed to other environmental variables.

Meta-analyses of restoration projects to improve riverine fish populations

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Introduction

Habitat destruction and habitat loss have been identified as greatest single threat to biodiversity in the Anthropocene (e.g. Brooks et al. 2002). The loss of species and pristine ecosystems continues at an alarming rate in Europe and globally (Feld et al. 2011). In this respect running waters play a crucial role, because these ecosystems belong to the most severely human-impacted habitats on earth (Malmqvist et al. 2002). In freshwaters the projected decline in species diversity is about five times higher than it was estimated for terrestrial ecosystems (Pimm et al. 1995), at a rate similar to the historical great extinctions (Malmqvist and Rundle 2002; Barnosky et al. 2011). Today river restoration became widely accepted as an essential complement to conservation and natural resource management (Palmer et al. 2005). For that billions of dollars have been spent every year, highly sophisticated restoration strategies developed (e.g. Bradshaw 1996; Buijse et al. 2002; Palmer et al. 2005; Stanford et al. 1996, 2002; Wohl et al. 2005), thus numerous textbooks and papers on rehabilitation facilities written (e.g. Cowx and Welcomme 1998). For all that very little as well as contradicting information was gathered about restoration success respectively effects (e.g. Bernhardt et al. 2005; Roni et al. 2005; 2008; Palmer et al. 2010). “The work has begun, but we have yet to determine what works best”, as it has been subtitled by Palmer and Allan (2006). In this context, this paper aims to elucidate the most efficient morphological restoration measures in relation to fish ecological improvements from reported assessments of restoration projects.

This investigation was forced by the implementation of the new Water Framework Directive (WFD) requiring to reach the good ecological status of all larger surface water bodies in the EU member states until 2015, measured in terms of macrophytes, phytoplankton, macroinvertebrates and fish communities (http://www.wiser.eu). At the end of 2004, the first status reports of the member states revealed, that the majority of surface water bodies will fail to reach the good ecological status until 2015 (BMLFUW 2005), as e.g. in Germany 60% of all surface water bodies (62% of rivers) were considered “at risk“ of failing the WFD objectives and further 26% “possibly at risk” (Borchardt et al. 2005). Accordingly, there is no doubt, that achieving the good ecological quality goals of the WFD requires significant restoration efforts and highly efficient mitigation measures. In contrast to the effects of degradation, the biotic response to restoration is less well-known and poorly predictable. The timescale of the WFD (obtaining good ecological status in all surface waters by 2027) is over-ambitious (Hering et al. 2010).So far many projects have been performed to revitalise and restore river sections; however, peer reviewed publications on monitored river rehabilitation projects describing biological parameters are rare (Roni et al. 2005). This poses a major problem for river managers responsible to decide on the most environmentally sensitive and cost-effective management schemes and restoration/rehabilitation measures: the challenge is to identify and prioritise the main drivers and responses at appropriate scales for implementing effective management of our natural resources (Feld et al. 2011).

In the present review, available information on river restoration projects performed in countries of the European Union was analysed. Data were gathered from peer reviewed studies available and supplemented by “grey literature”, unpublished reports, mostly from Austria, Germany and Switzerland. The aim of the review is to provide an overview on restoration measures commonly applied to improve fish diversity and on their effect on fish community in different types of rivers. The following hypotheses will be tested: Firstly, restoration measures have a significant effect on the biological parameters n species, density, and biomass. Secondly, the effect size depends on the time lag after the construction, as well as on the kind of restoration measure. After an initial improvement fish diversity might decrease due to natural succession. This influence of succession differs between types of restoration measures. Restoration measures providing active channels are expected to be most sustainable. Thirdly, restoration effect size differs between river types, salmonid or cyprinid, and the catchment size. Large rivers require large restoration efforts to be as same effective as in small rivers.

Materials and methods

Search for references

To determine the state of the art, literature was searched in several bibliographies (ISI web of knowledge, CSA data bases, Google Scholar). Furthermore, technical bureaus, universities, research institutes, and governmental agencies were contacted and asked to provide reports and studies on monitored restoration measures. Only studies with a spatial or temporal control were included in the present study. In total, 28 references from five countries were collected (Austria 20, Switzerland 3, Germany 3, Ireland 1, Great Britain 1) describing 68 different sites with restoration measures there from 21 control sites (see Appendix for references, site codes and classifications). In total 30 were analysed using a spatial control, 15 a temporal control, and two sites were analysed using both, a temporal and spatial control. 132 data sets (data set= combined value of biological parameter before and after restoration measure) were extracted for the three investigated biological parameters (110 without repeated monitoring): n species (47), density (36), and biomass (27). Habitat guild occurrence of fish species before and after restoration measures was revealed for 44 distinct manipulated sites.

Extracted parameters and classifications

Since the set of reported parameters was highly heterogeneous, sites were grouped according the catchment size in three size classes (<500 km²: small, 500-5000 km²: medium, >5000 km²: large).

Restoration measures were grouped in three classes applying a hierarchical approach. This was done according to three types of restoration measures, which could occur non-exclusively at each restored site. The restoration type "instream habitat enhancement" (IHE) was stated when structures to increase the variability of habitat parameters (e.g. width, depth, velocity) were installed within the river channel (e.g. gravel banks, large woody debris, boulder cluster etc.) and no other restoration measure was implemented at this site. The restoration type "dynamic processes" (Dyn) was stated when the restoration measure provided the possibility for side erosion and the dynamic forming and altering of gravel banks. In this group IHE measures may occur concurrently. The restoration measure "active channel" (ACh) was stated when the constructed measure provided side channels or side arms (depending on the type of river). IHE and Dyn may also occur concurrently. According to the information on fish assemblages and species caught in the studies, the rivers and sites were further grouped in two classes of fish regions (salmonid, cyprinid) and fish species were grouped into three habitat guilds: rheophilic, eurytopic, limnophilic (EFI+ 2007).

An unweighted approach was used for harmonisation of data within the meta-analysis by calculating the "log response ratio", ln(effect size), was calculated, which is the natural logarithm of the ratio between the values of the experimental and control group (Gurevitch & Hedges 2001). From studies with repeated monitoring, data from multiple monitoring events were only used for the analysis of the time effect. For all other analyses, only the most recent monitoring data set of the same restoration measure was used to prevent pseudo-replication (Gurevitch & Hedges 2001). From studies comparing one control with several manipulated sites all data were included. In Table 1 the number of sites providing data for the biological parameters n species, density, and biomass and their combinations with the different river types (small, medium, large, salmonid, cyprinid) is listed. Numbers in brackets indicate numbers of sites with repeated monitoring.

Table 1: Number of sites (N=47) providing data for the biological parameters n species, density, and biomass and their combinations with the different river types (small, medium, large, salmonid, cyprinid. Numbers in brackets indicate numbers of sites with repeated monitoring.

river size	fish region	n species	density	biomass
small	salmonid	15 (0)	14 (0)	11 (0)
	cyprinid	4 (3)	4 (3)	4 (3)
medium	salmonid	12 (0)	9 (0)	8 (0)
	cyprinid	9 (1)	6 (1)	4 (0)
large	cyprinid	7 (0)	4 (0)	1 (0)

Note, that in case of an increase in one of the biological parameters, the plain effect size expresses a value above one, and the ln(effect size) a value above zero.

To evaluate the general success of restoration measures, the calculated effect sizes were grouped and the cumulative percentage calculated from groups of high effect sizes to low effect sizes (Fig. 1). The last group in the analysis was the group of studies reporting a drop of the investigated parameter at the manipulated site below the value at the control site.

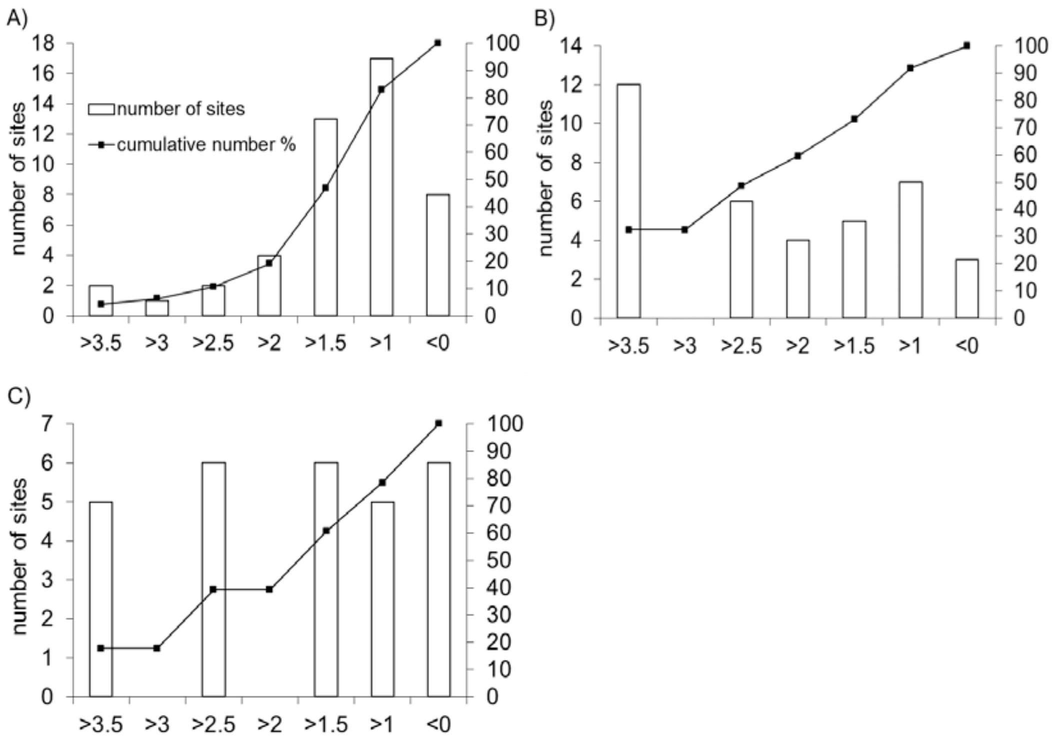

Figure 1: Cumulative number and number of sites with reported effect size of measures addressing A) fish species numbers, B) fish densities, and C) fish biomass.

Non-parametric tests were computed between three classes of restoration measures (IHE, ACh, Dyn), three classes of river sizes (small, medium, large), and three classes of time after restoration measure (0-1y, 2-4y, 5-12y) to detect differences in effect sizes for all three biological parameters. A linear curve fitting was computed for the maximum length of restoration measures and the biological parameters. Further box plots were computed for the occurrence of fish species before and after restoration measure between three habitat guilds (rheophilic, eurytopic, limnophilic) and three types of restoration measure (IHE, Dyn, ACh) for two fish regions. As for large size rivers exclusively sites at the Austrian Danube feature this study we further compared the occurrence of fish species to their specific reference called Leitbild (BAW 2011).

Data were tested for normality. The level of significance was $P < 0.05$. All statistics were calculated using SPSS PASW.

Results

General success of restoration measures

In 64% of the restoration sites all of the investigated biological parameters increased.

The percentage of sites with a positive effect (effect size >1), was highest for density (92%) and n species (83%) and about 79% for biomass (Fig. 1). The percentage with no or even a negative effect (effect size <0) was highest for biomass (21%) followed by n species (17%) and density (8%). Density concluded also high percentage (32%) of sites with an effect size above 3.5.

Differences in success for the three varying restoration measures

The number of restoration sites revealing positive effect sizes for all of the investigated parameters was highest for ACh (9 sites out of 13), followed by IHE (7 sites out of 11) and Dyn (14 sites out of 23).

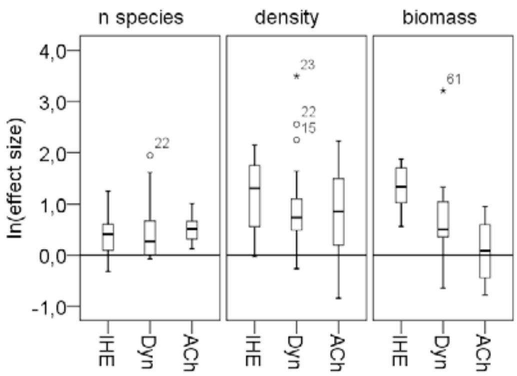

Figure 2: Median (black bar), 25 and 75% percentile (box), and minimum/maximum (whisker), and outliers (open circle; numbers indicate site ID, see text and Appendix S1) of ln(effect size) on three biological parameters (n species, density and biomass) for three classes of restoration measures (instream habitat enhancement, IHE; dynamic processes, Dyn and active channel, ACh).

Highest median ln(effect size) for n species was revealed for ACh measure (0.51, Fig. 2). IHE measure revealed highest median ln(effect sizes) for density (1.31) and biomass (1.33). Maximum ln(effect sizes) were revealed for Dyn measure in all of the parameters (n species= 1.95, density= 3.49, and biomass= 3.21). Minimum ln(effect size) for n species (-0.32) was revealed for IHE measure. In ACh measure minimum ln(effect sizes) were revealed for density (-0.84) and biomass (-0.78). A significant difference between restoration measures was exclusively found for biomass (Kruskal-Wallis, P = 0.007) with high effect sizes for IHE and low effect sizes for ACh measures (Fig. 2).

Differences in river size

No significant differences were revealed for river size classes and biological parameters (Fig. 3). Larger rivers revealed highest median effect size for n species (0.66). Streams revealed highest median effect sizes for density (0.99), and biomass (0.98), and maximum effect sizes for all of the biological parameters (n species= 1.95, density= 3.49, and biomass= 3.21). Medium size rivers revealed minimum effect sizes for all of the biological parameters (n species= -0.32, density= -0.84, and biomass= -0.78).

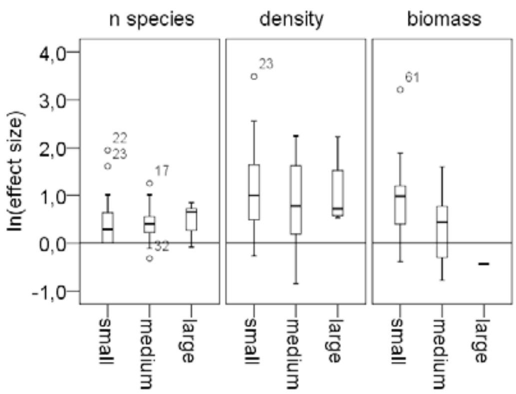

Figure 3: Median (black bar), 25 and 75% percentile (box), and minimum/maximum (whisker), and outliers (open circle; numbers indicate site ID, see text and Appendix S1) of ln(effect size) on three biological parameters (n species, density and biomass) for three river size classes (small, medium, large).

Time effect

All of the 47 manipulated sites were included in the time effect analyses. Of the sites analysed 17% described long-term monitoring (five to twelve years after restoration), 43% medium-term monitoring (two to four years after restoration) and 40% short-term monitoring (until one year after restoration). The longest monitoring of a rehabilitation measure lasted for twelve years after construction. Time had no significant influence on effect size (Fig. 4).

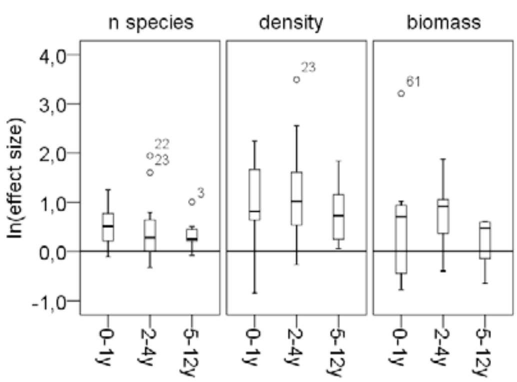

Figure 4: Median (black bar), 25 and 75% percentile (box), and minimum/maximum (whisker), and outliers (open circle; numbers indicate site ID, see text and Appendix S1) of ln(effect size) on three biological parameters (n species, density and biomass) for three classes of time after restoration (0-1 year after restoration, 2-4 years after restoration, 5-12 years after restoration).

Effect of length of restoration measure

The maximum length of restoration measure was highly variable and the highest variability and the longest restoration measure (21 km) are reported for large size rivers (Fig. 5).

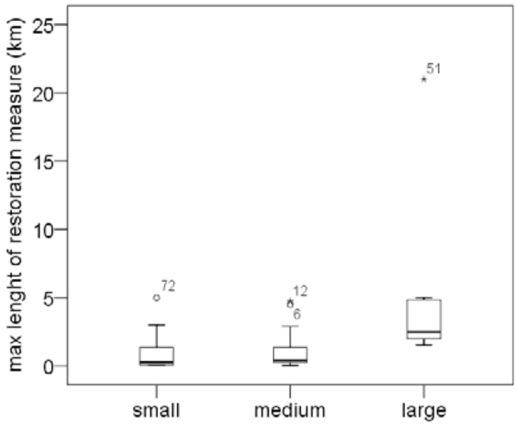

Fig. 5: Median (black bar), 25 and 75% percentile (box), and minimum/maximum (whisker), and outliers (open circle; numbers indicate site ID, see text and Appendix S1) of maximum length of restoration measure on three river size classes (small, medium, large).

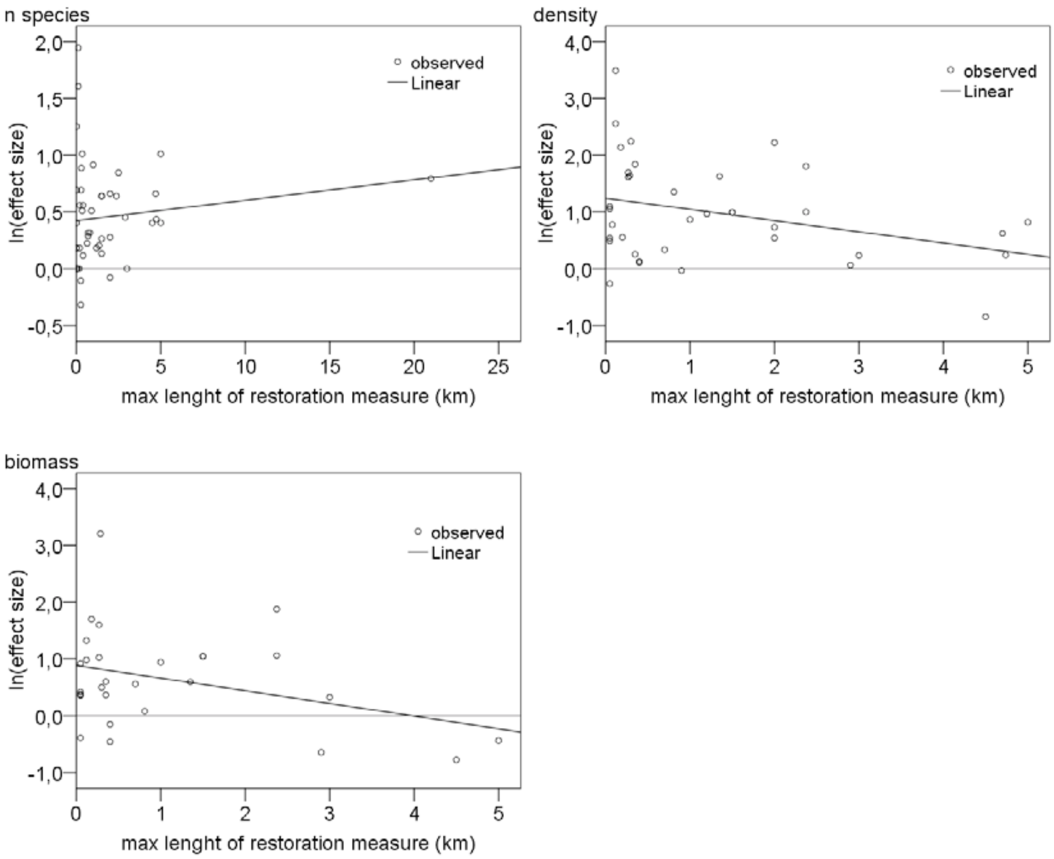

Fig. 6: Linear curve fitting of ln(effect size) on three biological parameters (n species, density and biomass) and maximum length of restoration measure.

There was a significant relation for the parameter density (linear curve fitting, $P = 0.043$, $r^2 = 0.111$, $y = 0.886x - 0.223$). Effect sizes for density decreased with increasing maximum length of a restoration measure. There was further a tendency in effect sizes of n species to increase and of biomass to decrease with length of restoration measure (Fig. 6).

Occurrence of riverine fish species before and after restoration measures

For 44 manipulated sites fish species data was provided before and after restoration measure and therefore we compared the occurrence within habitat guilds for fish region, type of restoration measure, and large size rivers. All comparisons revealed a substantial improvement following restoration (Fig. 7). Salmonid rivers showed a high improvement for rheophilic fish species, cyprinid rivers showed an improvement for rheophilic and eurytopic fish species and limnophilic fish species occurred marginal and were grouped with eurytopic fish species.

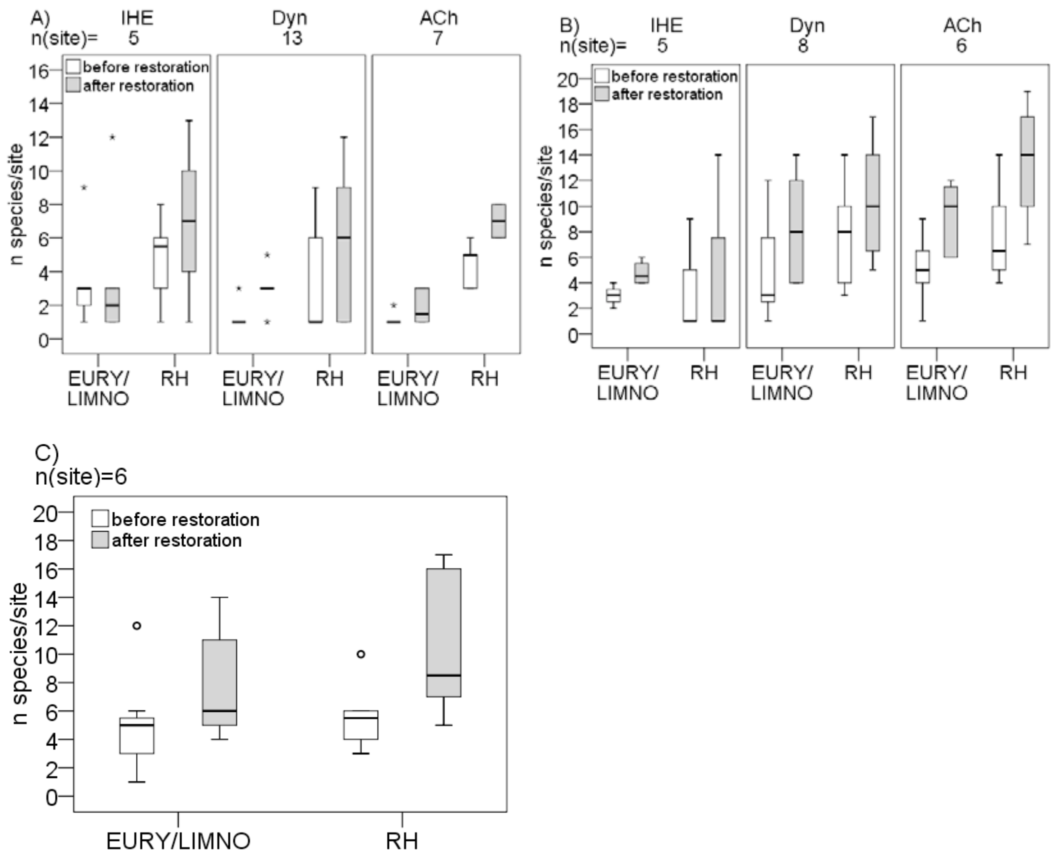

Fig. 7: Median (black bar), 25 and 75% percentile (box), and minimum/maximum (whisker), and outliers (open circle) of n species/site before and after restoration measures grouped in habitat guilds (rheophilic and eurytopic/limnophilic) addressing A) type of restoration measure in salmonid fish region, B) type of restoration measure in cyprinid fish region, and C) large size rivers

In large size rivers (6 sites) median values for rheophilic (six to nine from 36 of 'Leitbild') and eurytopic (five to eight from 14 of 'Leitbild') fish species increased. Median values for limnophilic fish species stayed the same at three (from 7 of 'Leitbild'). In Fig. 7 eurytopic and limnophilic fish species were grouped thus only a marginal number of sites with limnophilic species occur in large size rivers. In comparison to the 'Leitbild' occurrence of fish species improved after restoration measures for each habitat guild.

Discussion

Success of restoration measures to improve riverine fish populations

To evaluate the effects of restoration measures on stream fish populations in an overall view we computed the number of sites that showed positive effects sizes for all investigated biological parameters. Well knowing that not all of the biological parameters were assessed at all sites, 64% of 47 single sites analysed showed positive results for selected parameters after restoration measures. The lack of a consistent data basis possibly influences the overall result. Nevertheless the single calculated ecological metrics in this study indicate that restoration measures improve stream fish populations and can be considered as representative (Fig. 1).

ACh measures were most effective with 69% of 13 single sites revealing an increase in all of the parameters. Further ACh measures revealed highest median effect size for n species. This result confirms the hypotheses that restoration measures providing active channels are expected to be most sustainable. However also IHE measures revealed a striking improvement and highest median effect sizes for density and biomass. 64% of 11 single sites revealed an increase in effect sizes for all of the investigated biological parameters. In contrast Palmer et al. 2010 examined the reach-scale response of invertebrate species richness to restoration actions that increased habitat heterogeneity and only found 33% of the studies attempting to correlate a positive relationship between biodiversity and in-stream heterogeneity. Based on these results they postulated that physical heterogeneity should not be the driving force in selecting restoration approaches concerning benthic invertebrate populations, which have a smaller habitat scale than fish populations. Our results show that if the aim of restoration measure is to improve riverine fish populations, local measures to improve physical heterogeneity can be the adequate approach. Riverine fish as highly mobile organisms are known to be very effective in exploring newly created habitat structures at various life stages. It is therefore the creation and maintenance of a high habitat diversity mainly created by dynamic measures that consistently was followed by an increase in all of the parameters for 61% of 23 single sites and maximum effect sizes for all of the parameters.

Bias in evaluating restoration measures

At a success rate of 64% still the question remains, what the driving key factors leading to small effect sizes in some studies are. Especially the failures are important since they are likely to be continued or repeated elsewhere without monitoring and communicating to a larger audience (Wissmar & Beschta 1998). However, there is an additional bias in assessing effect sizes, which is only weakly related to the measure itself. First of all, there might be an insufficient assessment of the ecological status before the rehabilitation. For example in the Danube River at Orth (Appendix 2, site 70) in the oxbow system, the habitat suitability for spawning has been underestimated due to low water levels at the time of pre-investigation. Secondly, the control sites might be spatially related to the measure site, and thus, influence the assessment result in particular for such a mobile indicator as fish. This holds especially true because of thirdly, the typically high sampling variability at a site is and that only parts of the whole fish community will be represented in single fishing occasions (Zauner *et al*. 2008). Both hamper the detection of rehabilitation effects. Fourthly, more general population metrics might fail in sufficiently detecting changes. For example in the Danube River at the Wachau (Appendix 2, site 65) the total biomass at the control site overtopped those at the restoration site; however, both significantly differed in quality. At the restored site, where new gravel bars were provided, a higher biomass of the riverine target species *C. nasus* and *B. barbus* was observed, whilst the biomass increase at the control site resulted from the catch of a big *C. carpio* and numerous adult eurytopic *A. alburnus*. There is an urgent need for deriving more specific indicators for rehabilitation success.

Finally, restoration measures are always compromises between various stakeholder interests. Therefore, its realisation might not represent an optimum for the environmental objectives but the most agreeable common denominator (measure) at a certain site with its designated uses. Compromises in measure application impact their potential ecological efficiency, which yields biased results in analysing the potential of rehabilitation measures in general. For example, re-meandering may substantially improve the habitat heterogeneity at the reach scale, but fail in ecological improvements if banks and cross sections were fixed to ensure an unchanged discharge capacity for flood protection (Wolter 2010).

As those examples show, failures often are not directly linked to restoration measures but to distinct monitoring practices as well as further degradations and limitations due to additional impacts. Therefore the time for post-investigation must be selected advisedly. Accordingly, effect sizes will be detected despite of seasonal or interannual variations. It has to be concluded, that effect size might be influenced but not fully masked by natural variability. Thus, lack of effect size probably almost results from failing restoration measures (although the reasons to fail broadly vary).

Differences between restoration measures

Differences between restoration measures were significant for biomass with high effect sizes for IHE and low effect sizes for ACh measurement. In the small River Melk (Appendix 2, site 41) the biomass of nearly all fish species increased with a trend to the population value of the former, natural river after the installation of boulder cluster, groynes, bays and pools (Jungwirth *et al*. 1991). In contrast in the medium size River Drau (Appendix S1, site 6) biomass and density decreased after riverbed widening, the connection of oxbow lakes, and the introduction of gravel banks. At the small River Mank (Appendix 2, site 61) riverbed widening and the placement of habitat enhancement structures improved the availability of limiting spawning habitats. Accordingly, spawning of the target species has been observed immediately, one year after restoration (Zitek et al. 2004a).

The suggestion is that distinct restoration measures are extraordinary efficient for distinct stages of life or reproduction and for others on the other hand affect neutral or even negative. Restoration practitioners should aim to take into account the meta population aspects of recolonisation, when planning a restoration scheme (Feld et al. 2011).

Conclusion

It was illustrated the significant success of a restoration measure depends on the type of restoration measure respectively the combination of measures and the type of river. Even small-scale instream habitat enhancements may be extremely successful, if the restoration measure specifically addresses the limiting factors for a species or fish population. For the distribution of restoration type within river size please see Table 2. The review of 345 papers by Roni et al. (2008) on effectiveness of stream rehabilitation techniques as well indicated that some techniques, such as reconnection of isolated habitats, rehabilitation of floodplains, and placement of instream structures, have proven to be effective for improving habitat and increasing local fish abundance under many circumstances.

Tab. 2: Number of sites (N=47) for combinations of river size (small, medium, large) and restoration type (IHE, Dyn, ACh). Numbers in brackets indicate number of sites with repeated monitoring.

river size	IHE	Dyn	ACh
small	4 (1)	14(2)	1 (0)
medium	7 (0)	6 (1)	8 (0)
large	0 (0)	3 (0)	4 (0)

Local habitat enhancement measures are often ‘swamped’ by reach- or watershed-scale pressures upstream that continue to affect the treated sites. These limitations imply that the spatial scaling of restoration schemes must fit the scaling of degradation, that is, the scale of the pressures impacting the system, a point that has been largely ignored to date (Feld et al. 2011). Lepori et al. (2005) suggests that in-stream restoration schemes and the anthropogenic habitat alterations that motivate them have no substantial consequences for the diversity of fish and invertebrates if they affect habitat characteristics at scales different from those structuring biotic assemblages. Equally it has been rarely, thoroughly analysed, which of the essential habitats within the life cycle of a fish is lacking or functionally extinct and restoration decisions are often based on intuition rather than rigorous science (Muotka & Laasonen 2002). We suggest that local (site scale) measures need to be accompanied by reach-scale measures further upstream to control potential confounding effects of watershed-scale pressures.

Further, restoration measures need to be monitored beyond the timescale of typical experimental studies (i.e. the 3–4 years of many research grant funding schemes), in order to detect long-term intergenerational recovery, but also potential adverse effects and potential longer-term reversion to a degraded condition (Feld et al. 2011). The value of monitoring per se is in analysing trends over time. Presently, the spatial resolution of WFD monitoring data is high, though somewhat different between European countries. As the first phase of monitoring has just ended, there is yet no assessment of trends; the monitoring data will be important both for judging short-term effects of individual restoration measures and for analysing long-term trends. Finally we agree to Hering et al. (2010) that the particular value of the WFD monitoring data lies in the combination of a high spatial and a moderate temporal resolution.

Commonalities and differences in WFD River Basin Management Plans

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Introduction

One of the key elements in the Water Framework Directive are river basin management plans. According to the EU the best model for a single system of water management is management by river basin. The river basin is the natural geographical and hydrological unit that better ensures water management instead of areas based on administrative or political boundaries. For each river basin district - some of which will traverse national frontiers - a "River Basin Management Plan (RBMP)" has to be established and updated every six years.

The RBMP is a detailed account of how the objectives set for the river basin (ecological status, quantitative status, chemical status and protected area objectives) are to be reached within the timescale required. The RBMP includes amongst others the river basin's characteristics, a review of the impact of human activity on the status of waters in the basin, estimation of the effect of existing legislation and the remaining "gap" to meeting these objectives; and a set of measures designed to fill the gap.

One additional component is that an economic analysis of water use within the river basin must be carried out to enable there to be a rational discussion on the cost-effectiveness of the various possible measures.

Objectives

Current restoration of surface waters is mostly based on the WFD demands. The WFD river basin management plans must list the measures to reach the WFD objectives. A selection of five RBMPs is chosen to obtain an overview of the measures proposed to reach the WFD objectives. The objective of this study is to describe commonalities and differences in RBMPs in Europe.

Methods

From the WISER perspective four questions were posed:

- Do all waters in your country/catchment belong to a water body?
- To which water category do how many water bodies (or surface area/length) belong?
- How many water bodies belong to which water type?
- Which measures are listed in the RBMPs and to which major category does each measure belong?

The questions were answered for the Netherlands and Austria as a whole and for the catchment of the Seine (France) and the Lippe and Vechte (Germany).

Results

Water bodies

A water body is a discrete and significant element of surface water such as a river, lake or reservoir, or a distinct volume of groundwater within an aquifer. As such, a water body represents the smallest discrete unit subject to river basin management. Biological monitoring within a water body aims at assessing the ecological status of the whole water body, while management and restoration aim at improving the status of a water body as a whole. But surface waters include a large number of very small waters for which the administrative burden for the management of these waters may be enormous. The WFD does not include a threshold for very small "water bodies". However, the according to the CIS Guidance Document 2 the WFD sets out two systems for differentiating water bodies into types, System A and System B. Only the System A typology specifies values for size descriptors for rivers and lakes. The smallest size range for a System A river type is 10–100 km^2 catchment area. The smallest size range for a System A lake type is 0.5–1 km^2 surface area. No sizes for small transitional and coastal waters are given.

The application of system B must achieve, at least, the same level of differentiation as system A. It is therefore recommended to use the size of small rivers and lakes according to system A. However, it is recognised that in some regions where there are many small water bodies, this general approach will need to be adapted. Having said that, it may be appropriate to aggregate water bodies into groups for certain purposes in order to avoid unnecessary administrative burden. However, there are still large numbers of discrete rivers and lakes that are smaller than these thresholds. Member States have flexibility to decide whether the purposes of the WFD, which apply to all surface waters, can be achieved without the identification of every minor but discrete and significant element of surface water as a water body.

In The Netherlands 74% of all lakes and in Austria 100% of the lakes >50 ha are listed as WFD water body (Table 1). The total length of rivers is unknown for the Netherlands. In Austria 100% of river lengths are assigned a water body. Within the Seine catchment, only 38% of the streams and rivers are assigned a water body and thus part of the WFD. For the German basins of river Vechte and Lippe, these figures are not availbale.

Table 1. Overview of all and WFD water bodies in 2 countries and 3 selected catchments.

	Netherlands	Austria*	Seine	Lippe*	Vechte*
	NL	AT	FR	DE	DE
Rivers					
total length of all waters (km)		31,466	50,000	1,897	126
those part of a WFD water body		100%	38%	n.a.	n.a.
those out of a WFD water body		0%	62%	n.a.	n.a.
Lakes					
total surface area (ha) all waters	255,359	103,440	-	250	-
those part of a WFD water body	73.6%	n.a.	-	1	-
those out of a WFD water body	26.4%	n.a.	-	0	-

* Only river stretches >10 km^2 catchment size and lakes > 0.5 km^2 surface area are considered (System A typology)

Water categories

The WFD differentiates between natural, artificial, and heavily-modified and water bodies. Natural water bodies are considered all significant, natural accumulations of water that occur at the earth's surface in contrast with anthropogenic water bodies. The latter refers to water accumulations that are 'strongly' changed in their hydromorphology or that are created by men. For surface waters the overall aim of the WFD is that Member States should achieve "good ecological and chemical status" in all bodies of surface water by 2015. Some water bodies may not achieve this objective. Under certain conditions the WFD permits Member States to identify and designate Artificial Water Bodies (AWB) and Heavily Modified Water Bodies (HMWB). The assignment of less stringent objectives to water bodies and an extension of the timing for achieving the objectives is also possible (derogation).

HMWB are bodies of water, which as a result of physical alterations by human activity are substantially changed in character and cannot, therefore, meet the "good ecological status" (GES). In this context:

- Physical alterations mean changes to the hydromorphological characteristics of a water body, and
- A water body that is substantially changed in character is one that has been subject to major long-term changes in its hydromorphology as a consequence of maintaining specified uses. In general these hydromorphological changes alter morphological and hydrological characteristics.

AWB are surface water bodies, which have been created in a location where no water body existed before and which have not been created by the direct physical alteration, movement or realignment of an existing water body.

Member States may optionally designate surface water bodies as HMWB or AWB where they have been physically altered so that they are “substantially changed in character” or "created by human activity" respectively, and subject to some specific criteria. The first criterion requires that the specified uses of the water body (i.e. navigation, hydropower, water supply or flood defence) or the “wider environment” would be significantly adversely affected by the restoration measures required to achieve good ecological status. The second criterion requires that there are no significantly better environmental options for delivering the specified use that are technically feasible and cost effective. The designation may, in some instances, help to protect wider environmental interests, e.g. when the removal of a modification would lead to the destruction of valuable environmental features.

Instead of "good ecological status", the environmental objective for HMWB and for AWB is good ecological potential (GEP), which has to be achieved by 2015. The designation is not an opportunity to avoid achieving demanding ecological and chemical objectives, since GEP is an ecological objective, which may often, in itself, be challenging to achieve.

The Netherlands has far most artificial water bodies (57%), the other areas comprised less than 1–8% (Table 2). Heavily modified water bodies comprise more than 40% of all water bodies in Netherlands and the Lippe catchment, and even 79% in the Vechte catchment. On the contrary, the Seine catchment and Austria only assigned 1–4% as heavily modified. In the Seine catchment and Austria most water bodies (>90%) are assigned natural. The Netherlands has <1% natural water bodies.

Table 2. Assignment of water bodies to the three major water categories in 2 countries and 3 selected catchments.

	Netherlands	Austria	Seine	Lippe	Vechte
natural (%)	0.9	90.7	92.4	52.7	19.0
heavily modified (%)	42.1	7.8	3.9	46.3	78.6
artificial (%)	57.0	1.3	3.6	1.1	2.4
total number (n)	703	7,661	1,296	283	42

Water types

The WFD requires Member States to identify the location and boundaries of bodies of surface water and to carry out an initial characterisation of all such bodies. The surface water bodies identified must be differentiated according to hydromorphological type. The types are defined under ‘System A’ or ‘System B’. Under System A the typing categories are based on fixed typology descriptors: altitude, catchment area, and geology. Under System B Member States must achieve at least the same degree of differentiation as would be achieved using System A. Accordingly, the surface water bodies must be differentiated into types using the values for the obligatory descriptors and such optional descriptors, or combinations of descriptors, as are required to ensure that type specific biological reference conditions can be reliably derived.

Except for Austria, the number of river types is comparable between Netherlands, France and Germany (Table 3). The high number of river types used in Austria is based on abiotic (ecoregion, altitude, size and geology) and biotic (macro invertebrates, fish, algae, macrophytes) data. The Austrian rivers were divided into 15 bioregions. Furthermore, a longitudinal zonation (8 fish zones, saprobic and trophic conditions) was performed. In addition, some river segments were classified as special types (e.g. large rivers, lake discharge, glacial streams, thermal rivers, water falls, cascades, et cetera). By combining the bioregions and fish zones approximately 100 river water types were defined.

The number of water bodies is low in all areas, except for the Seine catchment.

Table 3. Number of water types and water bodies for rivers and lakes, respectively in 2 countries and 3 selected catchments.

	Netherlands	Austria		Seine	Lippe	Vechte
Area (km2)	41,530	83,855		75,000	5000	745.54
Rivers		natural	all			
Number of water types	12	>100		14	11	5
Number of water bodies	251	6,674	7,335	1,235	273	41
Lakes						
Number of water types	18	11	13	5	1	0
Number of water bodies	368	37	62	41	1	0

Measures and measure categories

The main objective of the Water Framework Directive is to maintain the ‘high and good status’ of waters where it exists, to prevent any deterioration in the existing status of waters and to restore at least ‘good status’ in relation to all waters by 2015. The mechanism, by which this objective will be achieved, is through the adoption and implementation of River Basin Management Plans (RBMPs) and programmes of measures (POMs). A RBMP distinguishes basic measures and, where necessary, supplementary measures.

The basic measures have been implemented by way of national regulation under various statutory instruments, must be complied with in full and are legally binding across the Member State. In some cases additional measures must be identified and considered at local level, i.e. at the river basin or water body.

The following procedure to compile a list of measure is most feasible in Europe. A water body is heavily degraded due to nutrient enrichment from urban wastewater and intensive agriculture practices. The minimum WFD obligation is that (basic) measures are implemented within the river basin as set out in national regulations. Furthermore, if these basic measures alone will not be enough to restore the water body to ‘good status’ by 2015, additional measures must be identified. These include, for example, setting more stringent emission controls than those required by national legislation and requiring stricter regulations on agricultural activities. Other additional measures include the recreation and restoration of the hydromorphology of the water body or even of surrounding wetland areas. More often a combination of additional measures is needed.

Measures are water type specific. Measures taken to improve a lake differ from those for a river. Measures taken in the alpine areas differ from those in the lowlands. To make measures comparable between countries and ecoregions we listed to types of summary categories:

I: key factors (factors that represent the main drivers of the ecosystems).

- hydrology
- hydromorphology
- morphology
- physico-chemistry
- biology
- spatial

II: measure categories (groups of comparable measures)

- source
- immission
- restoration
- maintenance
- connectivity
- policy & guidance
- others
- research & monitoring

In The Netherlands, until 2016, 38% of all measures focus on physico-chemistry and 24% on morphology (Table 4). Until 2027 42% focus on morphology and 24% on physico-chemistry. Biology and hydromorphological measures are limited in both time periods (only 1 and 3%, respectively).

Until 2016, 31% of all measures are restoration measures and 22% immission reduction measures. Until 2027, 54% are restoration measures and in both periods only 1% are source-oriented measures.

Table 4. Number of feasible and payable measure category - key factor combinations listed in the RBMP's in the Netherlands.

measure category	key factor	until 2016	2016-2027
policy & guidance	all	163	8
research & monitoring	all	613	16
restoration	all	110	83
source	all	3	0
policy & guidance	hydrology	59	45
restoration	hydrology	125	77
policy & guidance	hydromorphology	60	21
restoration	hydromorphology	53	26
maintenance	morphology	29	5
policy & guidance	morphology	1	1
restoration	morphology	1037	752
immission	physico-chemistry	692	111
maintenance	physico-chemistry	620	262
restoration	physico-chemistry	57	13
source	physico-chemistry	42	12
immission	physico-chemistry/hydrology	262	30
maintenance	biology	44	21
connectivity	spatial	353	256
policy & guidance	spatial	30	6
restoration	spatial	77	59

In Austria until 2016, 39% are immission measures (Table 5), while in the period thereafter 40% of the measures deal with connectivity. Until 2016, 19% of the measures focus on connectivity, while in the period thereafter 17% are physico-chemical and 24% related to hydrology.

Table 5. Number of water bodies within the feasible and payable measure categories - key factor combinations listed in the RBMP's in Austria.

measure category	key factor	until 2016	2016-2026
restoration	hydrology	139	1,312
restoration	impoundments	125	268
restoration	morphology	220	187
immission	physico-chemistry	447	916
connectivity	spatial	223	3,010

In the Seine catchment, 53% of the measures are physico-chemistry-oriented (Table 6). Here, the measure are related to source solutions (22%), restoration (19%), research & monitoring (17%), immission (16%), and policy & guidance (14%). No different time periods are known.

Table 6. Number of feasible and payable measure category - key factor combinations listed in the RBMP's in the Seine catchment.

key factor	measure category	number of times listed
all	policy & guidance	26
all	research & monitoring	42
all	restoration	37
all	connectivity	56
hydrology	policy & guidance	12
hydrology	restoration	57
hydrology	source	51
hydromorphology	research & monitoring	51
hydromorphology	restoration	16
hydromorphology	restoration/maintenance	23
morphology	restoration	30
physico-chemistry	immission	145
physico-chemistry	source	175
physico-chemistry	maintenance	76
physico-chemistry	policy & guidance	51
physico-chemistry	research & monitoring	89
physico-chemistry/hydrology	immission	18
spatial	policy & guidance	54
spatial	restoration	40

Table 7. Number of feasible and payable measure category - key factor combinations listed in the RBMP's in the Lippe catchment.

key factor	measure category	until 2016	2016-2026
all	policy & guidance	90	
all	research & monitoring	3	
hydrology	policy & guidance	1	
hydrology	restoration	28	24
hydrology	source	1	
hydromorphology	maintenance	48	59
hydromorphology	policy & guidance	13	
hydromorphology	restoration	14	109
morphology	maintenance	4	7
morphology	policy & guidance	24	
morphology	restoration	27	177
physico-chemistry	immission	12	18
physico-chemistry	policy & guidance	280	
physico-chemistry	source	1	63
physico-chemistry/hydrology	immission	7	35
spatial	policy & guidance	7	
spatial	restoration	11	52
temperature	source	2	

In the Lippe catchment, until 2016, 52% of all measures focus on physico-chemistry (Table 7). After 2015 the measures focus on morphology (34%), hydromorphology (31%) and physico-chemistry (21%). Surprisingly, until 2016, the measures in the Lippe catchment are mainly policy & guidance measures (72%). After 2015, 62% of the measures will be on restoration.

In the Vechte catchment, until 2016, 76% of all measures focus on physico-chemistry (Table 8). After 2015 the measures focus on morphology (32%), hydromorphology (32%) and physico-chemistry (27%). As in the Lippe catchment, until 2016, the measures in the Vechte catchment are mainly policy & guidance measures (97%). After 2015, 63% of the measures will be on restoration.

Table 8. Number of feasible and payable measure category - key factor combinations listed in the RBMP's in the Vechte catchment.

key factor	measure category	until 2016	2016-2026
all	policy & guidance	8	0
hydrology	restoration	0	1
hydromorphology	maintenance	0	8
hydromorphology	restoration	0	16
morphology	restoration	0	24
physico-chemistry	immission	1	4
physico-chemistry	policy & guidance	25	0
physico-chemistry	source	0	12
physico-chemistry/hydrology	immission	0	4
spatial	restoration	0	6

Conclusions

Water bodies

It is surprising that the percentages of water bodies being part of the WFD differ strongly between countries and catchments. For example, The Netherlands included about 74% of all water bodies, while France, within the Seine catchment, considered only 38% of the streams and rivers as part of the WFD. The other countries/catchments did not list the smaller water bodies.

Leaving out small, linear water bodies comprising <10 km^2 catchment area standing waters <0.5 km^2 surface area implies a reduction in management effort (monitoring, evaluation, protection and restoration). But it does not mean that water managers need not to manage these small water bodies. Several argument plea for local water management even to reach the WFD demands in larger catchments or water bodies because;

- As stated in the WFD, all surface water bodies fall under the legislation of the Directive.
- In intensively land-used regions, most physico-chemical pollution enters the major water systems through small water bodies,
- Many indicative, rare and vulnerable species inhabit small water bodies and host many positive quality elements and high biodiversity; thus the headwater sections below 10

km^2 catchment area represent important sources for recolonisation of restorations further downstream.

Water categories

The Netherlands has far most artificial water bodies, a finding that is not surprising, as half of the country is below sea level and many water bodies were dug to ensure water safety (drainage ditches). Heavily modified water bodies comprise almost half of the water bodies in Netherlands and Germany. These areas are urbanised and/or have intensive agricultural practices. In the Seine catchment and in Austria, most water bodies are assigned natural which is especially for the Seine surprising as the river runs through densely urbanised areas (e.g. the metropolitan area of Paris).

The major problem with the use of heavily modified status is the argumentation to reduce the water quality status to be reached. Despite criteria listed to sustain the heavily modified status, separation between physico-chemical and hydromorphological degradation is hard to make as the one often is directly linked to the other.

Water types

Despite the differences in surface area of the countries and catchments, respectively, the number of water types is comparable, except for Austria. The high number of river types used in Austria is due to the typology criteria used and most probably approaches the natural or original ecological differences present between stream and river ecosystems. In all other areas, despite the use of System B criteria, the number of both lake and river water types are low. The geomorphological, hydrological and physico-chemical circumstances most probably would result under reference conditions in a much higher number of ecosystem types. The approach chosen in System A, and somewhat less in System B, is not fully ecosystem boundaries based but more pragmatic related to major environmental parameters. Therefore, a difference exists between the number of water types versus the number of naturally occurring ecological types. This sets problems in defining reference conditions.

The high number of water bodies assigned in the Seine catchment is a good example of the reality as stream sites and stretches can differ strongly in pressures and ecological quality. Most measures need to be taken at the level of water bodies that are directly related to local circumstances. The link between small water body size and measures to be taken will be much more effective when the scales of both are comparable.

Measures and measure categories

Connectivity is most important in Austrian rivers and somewhat less in The Netherlands (Table 9). Connectivity measures lack in Germany. The latter is surprising as the catchments are in urbanised and agricultural areas where weirs in rivers are common. Immission reduction remains important in all areas, especially in Austria. Maintenance measures are most important in The Netherlands. The intensive water maintenance puts a high stress upon the water ecosystems. Reduction and changes in maintenance will contribute strongly to ecological status improvement. In Germany policy and guidance measures take an important role until 2016. Here, a lot of these policy and guidance measures focus on physico-chemical circumstances. If these measures accomplish a nutrient load reduction in the catchment the ecological profit will be huge. Only The Netherlands and France put some extra effort in research and monitoring. Overall, the attention to research and monitoring is very disappointing. The effects of many measures are not well or hardly studied. Stream restoration more often did not result in ecological improvement. Therefore, the importance of research and monitoring should be stressed by the European Commission.

Restoration is the most important measure in the studied areas.

That so little source-related measures are taken is disappointing as from an ecological point of view source-related measures are the only sustainable ones. It illustrates that a real integrated approach to deal with land use problems in Europe is still in its early development.

The majority of measures relate to physico-chemistry and morphology. Surprisingly, hydrology is less often tackled while it is the main driver of the ecological status. Probably the limitations of water management to influence or improve run off, groundwater and water use reduces the effort needed in this component.

Table 9. Percentages of feasible and payable measure categories and key factors in the studied areas.

	Netherlands		**Austria**		**Seine**	**Lippe**		**Vechte**	
key factor	**until 2016**	**2016-2027**	**until 2016**	**2016-2027**	**un-defined**	**until 2016**	**2016-2027**	**until 2016**	**2016-2027**
all	20	6			15	16	0	24	0
hydrology	10	8	12	23	11	5	4	0	7
hydromorphology	3	3	11	5	9	13	31	0	32
morphology	24	42	19	33	3	10	34	0	32
physico-chemistry	38	24	39	16	53	52	21	76	27
biology	1	1							
spatial	10	18	19	53	9	3	10	0	8
temperature						0	0		
measure category									
connectivity	8	15	42	31	5				
immission	22	8	39	16	16	3	10	3	11
maintenance	16	17			9	9	12	0	11
policy & guidance	7	5			14	72	0	97	0
research & monitoring	14	1			17	1	0		
restoration	31	54	19	53	19	14	67	0	63
source	1	1			22	1	12	0	16

CHAPTER IV

Synthesis

Observed effects along pressure-impact-response-recovery chains

Environmental stressors act hierarchically (low uncertainty)

There is evidence that broad-scale stressors (catchment water quality deterioration, intensive agriculture above a river site) can overrule the impact of rather fine-scale stressors (local habitat characteristics). The same is true for stretch/reach-scale environmental deterioration (e.g. riparian land use along several km of river length, physical habitat quality upstream of a station), which may superimpose local habitat quality. Empirical analysis in this report (see this report: Marzin et al., Dahm et al., Feld) imply that all BQEs significantly respond to broad-scale water quality deterioration, in parallel to other stressors at finer spatial scales. Conversely, management and restoration at finer scales is unlikely to initiate ecological recovery unless the broad scale impacts are being addressed and managed. This finding is in line with the scientific body of literature (see Paul and Meyer 2001, Allan 2004 and Feld et al. 2011 for an overview).

Broad-scale (catchment) and riparian land use control fine-scale habitat conditions for all river organism groups. At the broad scale, nutrient pollution and fine sediment entry can deteriorate local water quality (e.g. for macrophytes and diatom) and habitat conditions (e.g. for fish and benthic macroinvertebrates) (see this report: Marzin et al. and Feld). This finding is in line with the scientific body of literature (e.g. Allan 2004, Feld et al. 2011).

Biological indication of the impact of environmental stressors is complex (low uncertainty)

BQEs respond differently to (individual) stressors

There is strong empirical evidence that metrics respond differently to individual stressors (see this report: Marzin et al., Feld). While the response (correlation) of a specific metric can be positive to one stressor, this relationship may be negative with another and neutral with a third stressor. Moreover, non-linear response patterns can occur and require appropriate analytical methods. There is also evidence that the response of assemblages to land use is spatial scale-dependent (see this report: Feld); individual metric's response patterns can change across spatial scales and even change their sign (positive => negative or vice versa).

There is empirical evidence that river organisms respond better (i.e. a stronger relationship already at low stress levels) to global (general degradation) and water quality degradation, as opposed to hydrological and morphological degradation. All organism groups revealed a significant response to water quality deterioration in this report (see this report: Dahm et al., Marzin et al.).

Fish and macroinvertebrates show the strongest response to morphological (physical structure) degradation, while fish were found to be the strongest indicator group addressing hydrological degradation. Diatoms and macroinvertebrates showed a strong response to broad-scale and general degradation and responded to low stress levels. This finding implies that both latter organism groups are likely to be weak indicators of local improvement unless broad-scale degradation is reduced (see this report: Marzin et al.). Fish and macroinvertebrates show also a strong relationship to catchment and reach-scale land use (agriculture and forest) (see this report: Feld).

BQEs respond sensitive to different levels of stress

The strength of response (intensity) and the minimum stress level to show response (sensitivity) were considered separately by Marzin et al (this report). There is evidence that fish metrics reveal a strong intensity, but weak sensitivity; they can cope with relatively large stress levels, probably due to their mobility and hence specific ability to evade stress. Except for global degradation, macro-invertebrate metrics reveal a low sensitivity, while the intensity of response was intermediate (global, water quality, and morphological degradations) or weak (hydrological degradation). Macrophyte metrics showed the most sensitive responses to water quality degradation. Diatom metrics show mainly medium (global and water quality degradations) to weak (morphological and hydrological degradations) intensities, but reveal a high sensitivity except for hydrological degradation.

Traits and metrics/indices respond stronger and more sensitive than taxonomic/structural measures

There is evidence that trait-based metrics and multimetric indices reveal a stronger response to environmental stressors and respond more sensitive, if compared to the taxonomic structure of assemblages (see this report: Marzin et al.). This finding is supported by the scientific body of literature and accordingly, due to the broader applicability of functional characteristics of organisms, which are assumed to be more similar in water bodies across different regions as opposed to the taxonomic structure. Furthermore, traits and metrics address already biological or ecological attributes at community level, which can be related to specific habitat characteristics or feeding habits. Hence, contrary to the taxonomic structure of an assemblage, the relationship of its ecological and biological traits to degradation can be causal, for instance, the increase of fine-sediment-dwelling macroinvertebrates and the decrease of gravel-spawning fish due to fine sediment entry from crop agriculture close to a river course.

Measures of management and restoration are linked to and confounded by broad-scale stressors (medium uncertainty)

Local restoration is often unsuccessful

Local restoration refers to restoration at the scale of several tens of metres up to several hundreds of metres of river length. There is evidence that local restoration measures (e.g. introduction of wood or channel improvement) are unlikely to be ecologically successful and sustainable (Feld et al. 2011, http://www.wiser.eu/programme-and-results/management-and-restoration/conceptual-models/). There is little empirical evidence for the underlying reason(s) of failure, but it is a trivial conclusion that local habitat improvements do not (and cannot) address the broad-scale stressors. Nevertheless, there is evidence that specific local habitat enhancement can have positive effects on specific community members, for instance, the introduction of gravel for fish spawning and the subsequent increase of gravel spawning species (see this report: Melcher et al.).

There is empirical evidence that 'larger', more extensive restoration measures, which in parallel improve the in-stream habitat heterogeneity and overall channel patterns are more likely to improve fish, macroinvertebrate and macrophyte assemblages at the same time (see this report: Lorenz).

River restoration can be ecologically successful, if the relevant stressors are being addressed

There is empirical evidence in the body of restoration literature that river restoration is successful, if the relevant stressors are being addressed and their impact is being mitigated (Feld et al. 2011, http://www.wiser.eu/programme-and-results/management-and-restoration/conceptual-models/). Riparian buffer instalment is a suitable and successful measure to reduce nutrient and fine sediment entry from riparian agriculture. A riparian mixed buffer (trees, shrubs, grass strip) is capable of retaining 100% of fine sediments (incl. adhered phosphorus) and up to 90% of nitrogen from the upper groundwater layer. Riparian trees restore temperature regimes, control the riverine food web (allochthonous carbon supply) and shape the riverbed by the provision of woody debris.

There is empirical evidence that active channel restoration (e.g. by construction of side channels/arms) can lead to an improvement (e.g. increased richness, density, biomass) of fish assemblages in mountainous rivers (Melcher et al., Deliverable 6.4-2).

There are ambiguous empirical findings about the effects of local habitat enhancement. In general, local habitat enhancement can foster positive biological response if no other stressor is superimposing habitat enhancement (see this report: Melcher et al.; Zitek et al. 2004a). This is rarely the case, however, in intensively used and modified catchments, typically encountered in the lowlands of Eastern, Central and Western Europe and in the Mediterranean lowlands, where multiple stressors are reported to impact rivers in concert. Local habitat enhancements are also often 'spoiled' by erosion and deposition following major floods. Thus, local habitat enhancement (and structural restoration) must account for regional hydromorphological and geomorphologic settings that altogether control spatial and temporal local habitat conditions (Feld et al. 2011).

Biological response is controlled by ecological and environmental conditions upstream

There is empirical evidence that restoration measures show ecological improvement, if the ecological conditions upstream are poor; in particular macroinvertebrates and macrophytes revealed a strong response to restoration as described in this report (see this report: Lorenz). Notably, fish always responded positive (taxa richness, EQR) to local restoration measure, irrespective of the ecological quality above the restoration measure, probably due to their mobility and, thus, high capability to rapidly recolonise a restored stretch.

There is empirical evidence that many fish and macroinvertebrate metrics significantly change values at 0–20% agriculture in mountain ecoregions and 30–50% in lowland ecoregions, irrespective of the area considered (buffer, reach, catchment) (see this report: Feld). This finding is in line with the body of scientific restoration literature (e.g. Allan 2004).

Restoration is more likely to be successful, if upstream physical habitat degradation and land use impacts are low

There is evidence that fish and macroinvertebrate assemblages respond positively to the % coverage of deciduous forest in the riparian area upstream, presumably because the leaves and woody debris provide habitat and food. In contrast, macrophytes respond to the % coverage of arable land, which is likely due to low percentage of shading (i.e. the lack of riparian trees) upstream.

Empirical analysis implies that restoration measures can initiate biological recovery, if the physical habitat several kilometres upstream of the restoration are of moderate quality or better (e.g. German physical habitat score <4.5). In particular the fish assemblage was found to be strongly influenced by the river habitat quality upstream, but also by riparian and catchment land use upstream. Macroinvertebrates recovery requires shorter stretches upstream to be in a moderate or better quality; empirical data imply that about 1 km length upstream of a moderate or better physical habitat quality might be sufficient (see Lorenz).

In contrast, macrophyte response to restoration was found to be influenced by physical habitat quality up to 7.5 km upstream of the restoration.

Impact thresholds of hydromorphological conditions upstream

There is medium empirical evidence that fish assemblages respond positive to restoration measures, if the hydromorphological quality upstream is moderate or better (see this report: Lorenz). This evidence is based on nearly 50 restoration measures, yet restricted to the German mountain ranges and lowlands of ecoregion 9 and 14, respectively.

Restoration Monitoring is insufficient (low uncertainty)

Most restoration measures lack appropriate monitoring schemes

There is strong evidence that the vast majority of restoration measures reported in the scientific literature lack a monitoring program. Apart from those restoration studies that are part of a scientific research study, before-after-control-impact design monitoring is rarely reported and presumably completely neglected in regular monitoring of restoration and management measures (Feld et al. 2011). In most cases, restoration sites are being compared with non-restored sites close to the restoration (control-impact design, often also referred to as 'space-for-time substitution'). Thereby, control-impact design studies are capable of detecting temporal changes at both sampling sites, but they cannot account for small-scale spatial differences between the unrestored (control) and restored (impact) site. In order to overcome this shortcoming, before-after design monitoring is required in addition in order to be able to compare the conditions before and after restoration at the *same* site. The lack of this data is obvious also from the analyses presented here by Lorenz, Melcher et al. and Vlek et al. (this report). Although river restoration has been conducted for nearly 20 years in many countries (Feld et al. 2011), it is still challenging to compile >50 datasets of before-after restoration monitoring for an individual country.

Restoration monitoring, including both the situation before and changes after restoration, and in addition the changes at an unrestored control site nearby is an inevitable prerequisite for the sound evaluation of biological response and abiotic changes over time. As such, it would provide a sound basis for adaptive restoration/management taking into account, for instance, also other effects such as global/climate change. Without this data, adaptive river basin management, taking into account the manifold and complex interactions of stressors and organisms and their temporal variability and changes, and eventually the effects of external pressures, such as large-scale land use changes or climate change, is impossible! How do we know restoration effects, if we don not look at them?

River Basin Management Plans insufficiently account for research and monitoring demands

The lack of restoration monitoring is likely to continue within the first management period of the WFD (until 2015). This is evident from the River Basin Management Plans (RBMPs) analysed for this report (see this report: Verdonschot et al.). Although they represent only a small part of Europe, the RBMPs concordantly prove the little attention assigned to additional research and monitoring. Thus, it seems as if the implementation of restoration measures primarily planned for the period as of 2016 until 2021/2027 won't be accompanied by appropriate monitoring before and after implementation.

This obvious deficit requires a timely correction in order not to render future river restoration a potential waste of resources.

Predicted effects along pressure-impact-response-recovery chains

The limited availability of data, as outlined before, largely restricted the extent of empirical modelling (statistical modelling) conducted for this report. Consequently, statistical models (e.g. regression, calibration) were rarely applied.

The development of mechanistic models, which are, for instance, used for estimations of nutrient enrichment/reduction on phytoplankton growth in other ecosystem types, is impractical due to the course temporal resolution of environmental parameters; sampling takes place usually once a year or less often, which turned out to be too infrequent for the development of mechanistic approaches.

Riparian and catchment land use and hydromorphological quality control local habitat conditions (medium uncertainty)

Impact thresholds of riparian and catchment land use upstream

There is empirical evidence that widely used assessment metrics (fish and macroinvertebrates) significantly change values, when % cover agriculture upstream exceeds 20% in mountain ecoregions. Lowland assemblages seem to respond less sensitive to agriculture and significantly change values at 30–50%, irrespective of the area considered upstream (see this report: Feld). These findings are fully in line with the thresholds reported by Allan (2004) and the manifold original references therein.

There is medium empirical evidence that near-stream buffer areas should consist of a minimum of 40–50% forest along several kilometres upstream in order to maintain a good ecological quality (see this report: Feld). This finding refers to ecoregions only where forest (wooded vegetation) constitutes the climax vegetation and refers to mountainous as well as to lowland ecoregions. In general, forest cover in the riparian buffer strip should increase with increasing intensification of agriculture on the floodplain upstream.

There is medium empirical evidence that as little as 25% of the length of a river course upstream covered with deciduous forest is sufficient for ecologically successful restoration, if other adverse impacts (e.g. intensive agriculture) are absent from the riparian area (see Lorenz).

Implications for River Basin Management (RBM)

The full list of implications of the findings presented in this report would be comprehensive. This is primarily owed to the large number of individual contributions, each of which addresses a different key topic and uses data of a different spatial extent (e.g. sub-regions within a country, entire country, several countries/ecoregions). Therefore, focus is given here to those implications that are not restricted to specific individual fine-scale case studies, but apply to larger units (e.g. sub-basins or entire river basins). These implications are not necessarily fully supported by this report alone, but are in accordance with and supported by the findings of previous research within the WISER project (Feld et al. 2011) and outside (e.g. Allan 2004 and the references therein; Palmer et al. 2007; Palmer 2009).

Recommendations for the use of biological indicators in river monitoring

The stressors impacting the riverine fauna and flora and, hence the ecological quality, was divided into four major groups for this report: physico-chemical (e.g., nutrients, temperature), hydrological (e.g., stagnation), morphological (e.g. habitat degradation) and land use (e.g. agriculture). It is evident, that the different biological quality elements (BQEs) as well as the community characteristics within a BQE (habitat metrics, biological traits) respond differently to these stressor groups. The following table provides an overview of the BQE's intensity (correlation strength with a stressor) and sensitivity (minimum detectable stress level). The higher both measures are, the more indicative is a BQE/metric against a specific stressor.

BQE		general degradation	physico-chemical	hydrological	morphological	land use
Diatoms	Intensity	high	medium	low	low	medium
	Sensitivity	high	high	low	medium	high
Macrophytes	Intensity	low	medium	medium	low	low
	Sensitivity	medium	high	low	low	low
Macroinverte-brates	Intensity	high	medium	low	medium	medium
	Sensitivity	high	medium	low	low	medium
Fish	Intensity	high	high	medium	high	high
	Sensitivity	medium	medium	medium	medium	low

Despite the detected difference of BQE's response to different stressors, it is common to all BQEs that trait-based measures respond stronger (intensity) to stressors as opposed to taxonomic measures. Trait-based measures also revealed a significantly higher sensitivity to stressors.

River Basin Management must address and reduce all stressors

A typical hydromorphological river restoration targets channel and habitat improvements, for instance, by re-meandering and the introduction of natural, but underrepresented substrates (e.g. wood in lowland rivers, boulders in mountainous rivers). In its typical context, this restoration is implemented along a river stretch of up to several hundreds of metres. In agricultural river basins or smaller catchments therein, this typical approach is *very unlikely* to show significant improvement after restoration.

In agricultural river basins, the impact of agricultural activities (fertilizer and pesticide application, soil degradation and runoff) is omnipresent and can superimpose the impact of other (rather local) stressors. Consequently, *any* local restoration in agricultural catchments must account for these large-scale impacts upstream of a restoration in order to evaluate the response of the biota and to initiate recovery. The recommended best-practice measure to retain nutrients and fine sediments before they can enter and degrade the river system is to allow the establishment of riparian buffer strips. These strips can effectively mitigate agricultural impact. The recommended dimensions are (depending on river size): width: 10–30 m on either side. This also requires land users to make underground drainage systems ‘fine sediment-proof’.

Hydrological degradation has not been specifically addressed in this report. Yet, alike the physico-chemical and fine sediment impacts reported for intensively urbanised and agricultural catchments, hydrological degradation too is very likely to occur at the large-scale and may counteract the improvement of local or stretch/reach-scale hydrology (see Paul and Meyer 2001 and Allan 2004 for an overview).

The scale of river restoration must fit the scale of stressors to be restored

The previous paragraph addressed the impact of large-scale land use impact upstream of a restoration that is likely to hinder recovery unless the agricultural impacts are not being mitigated. The same applies to the large-scale physical habitat quality upstream of a restoration. Local hydromorphological restoration and habitat improvement are very unlikely to initiate recovery, if the hydromorphological conditions in the river course upstream (and downstream) are severely degraded (poor or bad status). Consequently, hydromorphological restoration should primarily target river reaches with moderate or better physical habitat conditions along several kilometres upstream (1–5 km depending on the river size). In a second step, further restoration can be implemented below previously restored reaches and successively lead to the restoration of long stretches of several kilometres or even tens of kilometres of length. If addressing also the riparian buffer strip, such a combined approach is more likely to lead to biological recovery than single and isolated restoration measures.

Yet, combined restoration requires a thorough planning prior to implementation, in order to embed individual measure in a larger context. With combined restoration, broad-scale hydromorphological improvements might be achieved by a smart design of multiple fine-scale restoration measures.

Land use management is required

The broad-scale impacts of agriculture have been discussed in previous sections. The mitigation and restoration measures, however, that were discussed so far consider the symptoms of land use degradation. Although it is expected that the large-scale instalment of riparian buffer strips will be of paramount importance in river basin management, it is the land use management, and land use management only that will help reduce the sources of impact (immission management).

It is crystal clear from hundreds of research studies and monitoring activities that good ecological quality is hardly achievable in agricultural and urbanised river basins as long as intensive land use practices are ongoing with the same level of impact. Consequently, river basin management will have to consider land use changes in the future. Such changes may be restricted to particularly sensitive areas, for instance to headwater sections that inhabit important source populations for recolonisation or to regions that are particularly sensitive to water pollution/eutrophication and fine sediment entry (e.g. spawning areas for Salmonid fish). Despite the land use conflicts of river basin management with, for instance, the agricultural industry, land use management is inevitable to achieve the goals of the WFD.

Restoration monitoring is mandatory

The lack of restoration monitoring data in general is alarming! Not only is the amount of available data surprisingly low, the available data is often very limited and rarely allows for the evaluation of improvements and eventually of success. This is basically due to two reasons. First, an overwhelming majority of restoration measures is not being monitored at all, probably because there is no legal requirement that designates restoration monitoring mandatory. And second, if restoration measures are monitored, the methods and time-scales applied rarely fit the state-of-the-art in freshwater monitoring.

The huge investments in river restoration and management require control of the ecological effects of these investments. Therefore, restoration monitoring should be mandatory. Only by frequent monitoring of biological and abiotic changes after restoration will restoration practitioners and scientist be able to evaluate the success of the restoration measure and eventually of the investment done. And such a frequent monitoring only can provide the information required for adaptive management, i.e. management that takes the environmental changes and biological response into account.

Restoration monitoring requires a smart sampling design that allows of sound statistical analysis according to state-of-the-art methods. First, in order to monitor changes, the status *before* restoration must be recorded at least once. Second, the status *after* restoration must be recorded several times in order to account for the development of a restored site after restoration. And third, a *control* (non-restored) site similar to the restored site before restoration must be monitored in order to detect the effect of natural variability (and climate change) and subtract them from pure restoration effects. This *before-after-control-impact (BACI)* design is standard in scientific research and allows the statistical testing of restoration effects and recovery; there is no alternative.

There is evidence in the scientific restoration literature that hydromorphological restoration may require a decade or longer to show biological recovery. Therefore, restoration monitoring must account for long-term recovery processes. It is recommended that restoration monitoring is designed for time scales of up to two WFD monitoring cycles, i.e. up to 12 years.

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Annex

Annex 1: Metric names and abbreviations according to S. Birk (unpublished data) for macrophytes, Meier et al. (2006) for benthic invertebrates and EFI+ Consortium (2009) for fish.

BQE	Abbreviation	Metric name
Fish	sWQO2it	Number of species intolerant to low oxygen content
	sHit	Number of species intolerant to habitat degradation
	sCL53it	Number of species intolerant to general degradation
	sRhpar	Number of species with rheopar spawning preferences
	sWQit	Number of species intolerant to water quality degradation
	sWQtol	Number of species tolerant to water quality degradation
	sHtol	Number of species tolerant to habitat degradation
	sHBrh	Number of species preferring rheophilic flow conditions
	sCL53bi1	Number of species intolerant to general degradation (alternative species grouping)
	sWQtxit	Number of species intolerant to toxic contamination
Benthic invertebrates	ASPT	Average Score Per Taxon
	No_EPT	Number of Ephemeroptera-Plecoptera-Trichoptera taxa
	Life	Lotic Invertebrate index for Flow Evaluation
	r_K	Relation r- to K-strategists
	p_EPT	Relative abundance of Ephemeroptera-Plecoptera-Trichoptera taxa
	p_Lithal	Relative abundance of taxa preferring stones as habitat (lithal)
	p_Psam	Relative abundance of taxa preferring sand as habitat (psammal)
	p_actFF	Relative abundance of active filter feeding taxa
	R_Dom	Dominance of r-strategists
	Margalef	Margalef diversity
Aquatic macrophytes	NMOSS	Number of moss taxa
	NMACRx	Number of true aquatic (submerged and emerged) taxa
	NTAXA	Total number of taxa
	SWTAXA	Shannon-Wiener diversity of all taxa
	ABDMOSS	Relative abundance of mosses
	NTRUE	Number of taxa excluding mosses
	SWTRUE	Shannon-Wiener diversity of taxa excluding mosses
	EVTAXAx	Evenness of true aquatic taxa
	EVTAXA	Evenness of all taxa
	EVTRUE	Evenness of taxa excluding mosses

Annex 2: Description of study site and restoration measure.

ID ref	ID site	country	river	site	river size	fish region	rest. typ	n species	density	biomass	occurrence	repeated monitoring	control type	monitoring term
2	1	A	Drau	Dellach	m	s	Ach	1	1	1	1	0	s	s
2	2	A	Drau	Greifenburg	m	s	Ach	1	1	1	1	0	s	s
2	3	A	Drau	Kleblach	m	s	Ach	1	1	1	1	0	s	l
2	4	A	Drau	control site	m	s	-	-	-	-	-	-	-	-
3	5	A	Drau	Dellach	m	s	Ach	1	1	1	1	0	s	s
3	6	A	Drau	Spittal	m	s	Ach	1	1	1	1	0	s	s
3	7	A	Drau	Kleblach	m	s	Ach	1	1	1	1	0	s	m
3	8	A	Drau	control site	m	s	-	-	-	-	-	-	-	-
4	9	A	Sulm	widening near Heimschuh	m	c	Dyn	1	1	1	1	0	s	l
4	10	A	Sulm	meander near Heimschuh	m	c	Ach	1	1	1	1	0	s	l
4	11	A	Sulm	control site	m	c	-	-	-	-	-	-	-	-
5	12	CH	Thur	Gütighausen-Eschikofen	m	s	Dyn	1	1	0	1	0	s	l
5	13	CH	Thur	control site	m	s	-	-	-	-	-	-	-	-
7	14	A	Leitha	Zurndorf	m	c	Ach	1	1	1	1	0	t	l
8	15	A	Glan	Ebenthal	m	c	Dyn	1	1	1	1	0	t	s
10	16	GB	Huntspill River	A north bank 97	m	c	IHE	1	1	0	1	0	s	s
10	17	GB	Huntspill River	D south bay 97	m	c	IHE	1	0	0	1	0	s	s
10	19	GB	Huntspill River	B south bank 96	m	c	IHE	1	0	0	1	0	s	m
10	20	GB	Huntspill River	C south bay 96	m	c	IHE	1	0	0	1	0	s	m
10	21	GB	Huntspill River	control site	m	c	-	-	-	-	-	-	-	-
13	22	A	Alterbach	section 2	s	s	Dyn	1	1	1	1	0	s	m
13	23	A	Alterbach	section 3	s	s	Dyn	1	1	1	1	0	s	m
13	24	A	Alterbach	control site	s	s	-	-	-	-	-	-	-	-
13	25	A	Oichten	Oichten	s	s	Dyn	1	1	0	1	0	s	m
13	26	A	Oichten	control site	s	s	-	-	-	-	-	-	-	-
14	27	D	Alte Ammer	Alte Ammer	m	s	IHE	1	0	0	1	0	t	s
14	28	D	Rott	Rott	s	s	Dyn	1	0	0	1	0	t	s
15	29	D	Dosse	Baumannsbrücke	m	s	IHE	1	1	1	1	0	s	s
15	30	D	Dosse	control site Baumannsbrücke	m	s	-	-	-	-	-	-	-	-
15	31	D	Dosse	control site Friedrichsgüte	m	s	-	-	-	-	-	-	-	-
15	32	D	Dosse	Friedrichsgüte	m	s	IHE	1	1	1	1	0	s	m
16	35	CH	Bünz	Wohlen	s	s	IHE	1	1	0	1	0	s	m
16	36	CH	Bünz	control site Wohlen	s	s	-	-	-	-	-	-	-	-
17	37	CH	Thur	Güttighausen	m	s	Dyn	1	0	0	1	0	s	l
17	38	CH	Thur	Schäffäuli	m	s	Dyn	1	0	0	1	0	s	s
17	39	CH	Thur	control site Weinfelden	m	s	-	-	-	-	-	-	-	-
17	40	CH	Thur	control site Frauenfeld	m	s	-	-	-	-	-	-	-	-
19	41	A	Melk	Mittellauf	s	c	IHE	1	1	1	1	1	s+t	m
19	42	A	Melk	control site	s	c								
21	45	A	Bregenzerach	Schnepfau-Mellau	s	s	Dyn	1	1	1	0	0	s	m
21	46	A	Bregenzerach	control site	s	s	-	-	-	-	-	-	-	-
22	47	A	Mödlingbach	Mödling	s	s	IHE	1	1	1	1	0	s	m
22	48	A	Mödlingbach	control site	s	s	-	-	-	-	-	-	-	-
23	49	A	Katschbach	Katsch	s	s	Ach	1	1	0	1	0	s	s
23	50	A	Katschbach	control site	s	s	-	-	-	-	-	-	-	-
24	51	A	Danube	National Park Danube	l	c	Ach	1	0	0	1	1	s	m
24	52	A	Danube	control site	l	c	-	-	-	-	-	-	-	-
26	54	IR	Rye Water	S2E1	s	s	Dyn	1	1	1	1	0	t	m
26	55	IR	Rye Water	S2E2	s	s	Dyn	1	1	1	1	0	t	m
26	56	IR	Rye Water	S3E1	s	s	Dyn	1	1	1	1	0	t	m
26	57	IR	Rye Water	S3E2	s	s	Dyn	1	1	1	1	0	t	m
26	58	IR	Rye Water	S3E3	s	s	Dyn	1	1	1	1	0	t	m
28	59	D	Lippe	control site	m	c	-	-	-	-	-	-	-	-
28	60	D	Lippe	Klostermersch	m	c	Dyn	1	1	0	1	1	s+t	l
29	61	A	Mank	downstream section	s	s	Dyn	1	1	1	1	0	t	s
31	62	A	Danube	Diedersdorfer Haufen + Ybbser Scheibe	l	c	Dyn	1	1	0	0	0	s	l
31	63	A	Danube	control site	l	c	-	-	-	-	-	-	-	-
37	64	A	Melk	near Zelking	s	c	Dyn	1	1	1	1	1	t	m
61	65	A	Danube	narrow channel km 2034 - 2009	l	c	Ach	1	1	1	1	0	s	s
61	66	A	Danube	oxbow lake	l	c	Ach	1	1	0	1	0	s	s
61	67	A	Danube	sidearm	l	c	Ach	1	1	0	1	0	s	s
61	68	A	Danube	control site	l	c	-	-	-	-	-	-	-	-
62	69	A	Vöckla	Stadtpark Vöcklabruck	s	s	IHE	1	1	1	0	0	t	m
63	70	A	Danube	Orth	l	c	Dyn	1	0	0	1	0	t	s
67	71	A	Melk	downstream Diemling	s	c	Dyn	1	1	1	1	1	t	m
69	72	A	Liesingbach	Großmarktstraße bis Kledering	s	c	Dyn	1	0	0	1	0	t	s
79	73	A	Danube	gravel bank	l	c	Dyn	1	0	0	1	0	s	s
79	74	A	Danube	control site	l	c	-	-	-	-	-	-	-	-

Key to reference IDs (primary source, secondary source): 2 (Muhar et al. 2000b); 3 (Unfer et al. 2004); 4 (Zitek et al. 2004b); 5 (Hörger & Kaiser 2003); 7 (Muhar et al. 2000a); 8 (Honsig-Erlenburg 2003); 10 (Langer & Smith 2001); 13 (Rüter & Asche 1996); 14 (Bohl et al. 2004); 15 (Völker 2004); 16 (Sreule 2000); 17 (Hörger & Keiser 2003); 19 (Kaufmann et al. 1991); 21 (Grasser et al. 2001); 22 (Eberstaller et al. 1992); 23 (Schmutz & Melcher 2000); 24 (Chovanec et al. 2002); 26 (Kelly & Bracken 1998); 28 (Schütz et al. 2006); 29 (Zitek et al. 2004a); 31 (Jungwirth et al. 1993); 37 (Jungwirth et al. 1993); 61 (Zauner et al. 2008); 62 (Zauner & Ratschan 2008); 63 (Schabus & Reckendorfer 2002); 67 (Jungwirth et al. 1995); 69 (Panek & Siegel 2005); 79 (Keckeis et al. 2007). River size class: s, small; m, medium; l, large. Fish region: s, salmonid; c, cyprinid. Restoration typ: IHE, instream habitat enhancement; Dyn, dynamic processes; ACh, active channel. Biological parameter (n species, density, and biomass) and occurrence of fish species: 1, data available; 0, data not available. Repeated monitoring: 0, no; 1, yes.

Control type: s, spatial control; t, temporal control; s+t, spatial and temporal control. Monitoring time scale: s, short-term monitoring: 0-1year; m, medium-term monitoring: 2-4years; l, long-term monitoring: 5-12years after restoration.

Annex 3: Occurrence of fish species before and after restoration measure per manipulated site.

	site ID	1		2		3		5		6		7		9		10		12		14		15		16		17		19		20		22		23		25		27		28		29		32		35		37		38		41		47		49		51		54		55		56		57		58		60		61		64		65		66		67		69		70		71		72		73		74	
species	habitat guild (EFI+)	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after	before	after
Abramis brama	EURY															+	+			+	+			+	+		+		+									+	+																			+	+										+				+									+	+					+		
Alburnoides bipunctatus	RH												+	+	+	+	+	+	+	+	+	+										+				+		+								+	+	+		+																						+	+	+	+	+	+	+						+						
Alburnus alburnus	EURY												+	+	+	+			+	+		+															+	+	+	+																	+	+										+	+					+	+	+	+	+	+			+	+					+	+		
Ameiurus spp.																+																																																																																	
Anguilla anguilla	EURY													+			+	+				+															+	+	+	+							+	+																							+	+																					
Aspius aspius	RH																				+																	+	+																			+	+																			+	+	+	+	+	+			+	+								
Barbatula barbatula	RH		+		+				+		+		+	+	+	+	+	+	+	+	+												+		+	+	+					+	+	+	+					+	+	+	+						+										+	+	+	+	+	+		+				+	+	+			+	+	+	+				
Barbus barbus	RH												+	+	+	+	+	+	+	+	+	+																+		+							+	+	+	+	+	+					+	+										+	+	+	+	+	+		+					+	+		+	+	+		+	+	+		
Barbus peloponnesius	RH												|+| |+|																																																																																
Blicca bjoerkna	EURY													+		+			+	+																		+																	+										+															+	+					+		
Carassius auratus	LIMNO																																																																																					+				
Carassius carassius	LIMNO					+				+		+								+																		+													+																						+												+						
Carassius gibelio	EURY												+	+	+	+			+	+		+															+	+																	+										+															+	+		+				
Chondrostoma nasus	RH				+		+						+	+	+	+	+	+	+	+	+	+																+								+		+		+					+	+										+	+		+		+	+	+	+	+	+	+				+		+						
Cobitis elongatoides	RH																				+																																																																+						
Cobitis taenia	RH																																																				+																		+		+		+																			
Cottus gobio	RH		+					+	+	+	+				+				+												+		+	+	+				+					+	+		+			+	+			+	+	+	+												+	+	+	+	+	+		+				+	+	+			+	+			+			
Cypriniden																																																|+|+|+|+|																																																
Cyprinus carpio	EURY													+		+			+	+		+						+										+	+													+					+	+										+	+				+		+							+	+		+						
Esox lucius	EURY					+				+		+	+	+	+	+			+	+		+							+							+	+	+	+	+	+	+			+												+										+	+						+							+	+								
Eudontomyzon spp.														+																																																														+															
Gasterosteus aculeatus	EURY																		+			+					+		+															+	+	+	+					+	+										+												+	+								+		+			+								
Gobio obtusirostris	RH				+		+		+					+	+	+	+	+	+	+	+		+		+		+	+	+	+	+			+					+	+	+		+	+	+	+	+	+	+	+	+	+	+	+																+	+	+	+	+	+		+							+	+		+				
Gymnocephalus baloni	RH																																																										+																																			
Gymnocephalus cernuus	EURY													+		+				+			+	+	+	+	+	+	+	+							+	+												+																							+	+						+								+	+								
Gymnocephalus schraetser	RH																																																										+																						+		+		+			+									
Hucho hucho	RH				+						+	+	+	+	+	+	+	+																																							+					+	+															+		+		+						+										
Lampetra planeri	RH																																											+		+													+														+	+																								
Lepomis gibbosus	LIMNO													+		+							+																																																+																	+	+				+				
Leucaspius delineatus	LIMNO																		+		+			+			+		+																																															+						+		+														
Leuciscus cephalus	RH																																																		|+|+|																																														
Leuciscus idus	RH													+		+																						+	+																		+																			+	+	+	+	+	+			+	+					+		
Leuciscus leuciscus	RH												+	+	+	+	+	+	+	+	+																+	+	+	+	+	+	+	+			+	+		+		+					+	+												+	+		+	+	+	+	+								+	+	+	+		+				
Lota lota	EURY	+	+	+	+	+	+	+	+	+	+	+	+								+	+	+																																																+	+														+	+								
Misgurnus fossilis	LIMNO																				+																																																																											
Neogobio kessleri	EURY																																																																											+	+								+									
Neogobius gymnotrachelus	EURY																																																																																															+
Neogobius melanostomus	EURY																																																																											+	+	+	+	+	+											+	
Onchorhynchus mykiss	RH												+	+	+	+																+						+												+	+			+	+																			+	+							+	+			+	+						
Perca fluviatilis	EURY												+	+	+	+		+	+	+		+	+	+		+	+	+	+	+				+			+	+		+							+		+								+										+	+					+	+							+	+								
Phoxinus phoxinus	RH		+					+	+	+	+	+	+					+	+		+												+		+							+	+	+	+			+	+	+	+	+	+																			+	+	+	+	+	+		+					+	+			+	+	+	+				
Proterorhinus semilunaris	EURY																			+	+																																					+																						+	+	+	+				+	+								
Pseudorasbora parva	EURY												+	+	+	+																																																																									+				
Pungitius pungitius	LIMNO																																										+		+																													+	+																							
R. rutilus x A. brama hybrid																								|+|				|+|																																																																	
Rhodeus amarus	LIMNO													+		+				+																								+									+					+																+										+	+		+						
Romangobio vladykovi	RH																																																									+	+																				+		+												+		
Romanogobio kesslerii	RH												+	+	+	+																																																																											+	+		
Romanogobio vladykovi	RH												+	+	+	+																																																																													
Rutilus meidingeri																																																																																		+										
Rutilus rutilus	EURY				+		+			+	+			+	+	+	+			+	+		+	+	+	+	+	+	+	+	+							+	+		+	+		+						+		+						+												+	+			+	+	+	+	+	+	+	+	+	+	+	+		+						
Rutilus virgo	RH																+				+																																																																									
Sabanejewia balcania	RH																+																																																																												
Salmo salar	RH																																																											+	+	+	+	+	+	+	+	+	+																								
Salmo trutta	RH	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+		+		+									+	+	+	+	+	+		+	+	+	+	+	+	+	+	+	+	+			+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+		+	+	+		+					+	+			+	+				+		
Salvelinus fontinalis	RH																+		+																						+																+	+																											+				+						
Sander lucioperca	EURY													+		+				+																		+																		+												+					+	+	+	+	+	+			+	+						+		
Scardinius erythrophthalmus	LIMNO					+						+		+		+			+	+																		+	+													+						+										+				+		+		+		+				+	+				+				
Silurus glanis	EURY																																					+																					+																																							
Squalius cephalus	RH	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+										+			+	+	+	+	+	+						+	+	+	+									+	+												+	+	+	+	+	+	+	+					+	+	+	+	+	+	+	+				
Telestes souffia	RH				+		+		+		+		+																																			+	+	+	+																																													
Thymallus thymallus	RH	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+		+			+	+														+	+		+		+	+	+	+	+							+	+				+	+																				+	+		+	+	+							+	+			+	+						
Tinca tinca	LIMNO											+		+		+				+																		+	+	+	+																	+										+	+									+					+								
Vimba vimba	RH																						+																																			+																	+		+		+			+	+						+		
Zingel streber	RH												+	+	+	+																																										+																				+				+									+	+		
Zingel zingel	RH																																																									+	+																								+	+												

Annex 4: General characteristics of the restoration measures

River	Nearest town	shortcode	Year of restoration	Restored length (m)	Lowland	Mountain	Catchment size (km^2)	Lowering of entrenchment depth	Removal of bank fixation	Wood placement	Installation of flow deflectors	Elongation of river length	Creating a new water course	Creation of multiple channels	Extensifi-cation of landuse	Re-connection of backwaters	Removal of weirs	passiv restoration	re-meandering
Brend	Bischofsheim	Bre_Bis	2005	100	0	1	110	-	x	-	-	-	-	-	-	-	x	-	-
Bröl	Waldbröl	Brö_Wal	1995	400	0	1	181	-	-	-	-	-	-	x	x	-	-	x	-
Dill	Dillenburg	Dil_Dil	2005	800	0	1	314	-	-	x	-	-	x	-	x	-	-	-	-
Eder	Dodenau_Rössmühle	Ede_Dod	2000	200	0	1	480	-	-	-	-	-	-	x	x	-	-	x	-
Fallbach	Ravolzhausen	Fal_Rav	2002	1000	0	1	29	x	x	x	x	x	x	x	x	-	-	-	-
Fulda	Mecklar	Ful_Mec	2004	1000	0	1	2375	-	-	-	-	-	x	x	-	x	-	-	-
Fulda	Niederaula	Ful_Nie	2005	2000	0	1	1290	-	-	-	-	-	-	-	-	x	-	-	-
Gartroper Mühlenbach	Hünxe	Gar_Hün	2003	1400	1	0	9	-	x	-	-	x	x	-	x	x	-	-	x
Gersprenz	Reinheim	Ger_Rei	2007	1200	1	0	154	-	x	x	-	x	x	-	x	-	x	-	-
Hase	Haselünne_r1	Has_Ha1	2001	1000	1	0	2530	-	-	-	-	x	-	-	x	x	-	-	x
Hase	Haselünne_r2	Has_Ha2	2006	1000	1	0	2520	-	-	-	-	-	-	-	x	-	-	x	-
Inde	Kirchberg - Brücke	Ind_Bru	2002	300	1	0	359	x	x	-	-	x	x	-	x	-	-	-	x
Inde	Kirchberg - unterer Abschnitt	Ind_unt	2002	350	1	0	359	x	x	-	-	x	x	-	x	-	x	-	x
Josbach	Josbach	Jos_Jos	2002	400	0	1	29	x	-	x	-	-	-	-	x	-	-	-	-
Kimmer-Brookbäke	Hude	Kim_Hud	2006	1400	1	0	10	x	x	x	x	x	x	-	x	x	-	-	x
Kinzig	Niederrodenbach	Kin_Nie	2001	200	0	1	885	-	-	-	-	-	-	-	-	x	-	-	-
Lache	Rodenbach	Lac_Rod	2007	800	1	0	11	x	-	-	-	x	x	-	-	-	-	-	-
Lahn	Cölbe	Lah_Cöl	2002	200	0	1	650	x	x	x	x	-	-	x	x	-	-	-	-
Lahn	Ludwigshütte	Lah_Lud	2001	300	0	1	288	-	x	-	-	x	x	x	-	-	-	-	-
Lahn	Sterzhausen	Lah_Ste	2006	200	0	1	350	-	x	-	x	-	-	-	-	-	-	-	-
Lahn	Wallau	Lah_Wal	2000	300	0	1	278	-	x	-	-	x	x	x	-	-	-	-	-
Nette	Weißenthurm - Mündung	Net_Wei	2007	700	0	1	370	-	x	-	x	x	-	-	x	-	-	-	-
Nidda	Bad Vilbel	Nid_Bad	2001	450	0	1	1200	x	x	-	x	x	x	-	x	-	-	-	-
Nidda	Ilbenstadt	Nid_Ilb	2006	1500	0	1	1168	x	x	x	x	-	-	x	x	-	-	-	-
Nidda	Ranstadt	Nid_Ran	2004	2500	0	1	226	-	x	x	x	x	x	x	x	-	-	-	-
Nidder	Altenstadt	Nid_Alt	2002	300	0	1	153	-	x	-	x	-	x	x	x	-	-	-	-
Niers	Geldern-Pont R1	Nie_Ge1	2000	800	1	0	386	-	x	-	-	x	x	-	x	-	-	-	X
Niers	Geldern-Pont R2	Nie_Ge2	2007	800	1	0	386	-	x	x	x	x	x	-	x	-	-	-	X
Orke	Niederorke	Ork_Nie	1998	300	0	1	289	-	-	-	-	-	-	x	x	x	-	x	-
Rhein	Duisburg	Rhe_Dui	2005	700	1	0	52880	-	x	-	-	-	-	-	-	-	-	-	-
Rodau	Obertshausen	Rod_Obe	2002	2000	1	0	71	x	x	x	x	x	x	x	-	x	x	-	X
Ruhr	Arnsberg - Altes Feld	Ruh_alt	2004	800	0	1	844	x	x	x	x	-	-	x	x	-	-	-	-

Ruhr	Arnsberg - Binnerfeld R1	Ruh_Bi1	2008	750	0	1	1050	-	x	-	-	-	x	x	-	-	-	-	-
Ruhr	Arnsberg - Binnerfeld R2	Ruh_Bi2	2009	820	0	1	1050	-	x	x	-	-	x	x	-	-	-	-	-
Ruhr	Arnsberg - Binnerfeld R3	Ruh_Bi3	2010	960	0	1	1050	-	x	x	-	-	x	x	-	-	-	-	-
Ruhr	Neheim	Ruh_Neh	1980	320	0	1	1531	-	-	-	-	-	-	x	x	-	-	x	-
Ruhr	Schellenstein	Ruh_Sch	2006	500	0	1	90	-	x	-	-	x	x	x	-	-	x	-	-
Rur	Jülich 2 - Brücke (R1)	Rur_Jü1	1996	400	1	0	1340	-	x	-	-	-	-	x	x	-	-	-	-
Rur	Jülich 3 - Klärwerk (R2)	Rur_Jü2	1996	200	1	0	1335	-	x	-	-	-	x	x	x	-	-	-	-
Rur	Körrenzig 2 - Insel Süd	Rur_KöS	2001	300	1	0	1472	-	x	-	-	x	x	x	x	-	-	-	-
Rur	Körrenzig 3 - Neutrassierung Nord	Rur_KöN	2001	500	1	0	1470	-	x	-	-	x	-	x	x	x	-	-	-
Rur	Millich 3 - Schanz	Rur_MiS	2002	400	1	0	1730	-	x	-	-	x	x	-	x	-	x	-	-
Rur	Millich 2	Rur_Mi2	2002	300	1	0	1715	-	x	-	-	x	-	-	-	x	x	-	X
Sandbach	Crumstadt	San_Cru	1995	2600	1	0	116	-	x	-	-	x	x	-	x	-	-	-	X
Schwalm	Brüggen	Sch_Brü	1997	4300	1	0	250	-	x	-	-	x	x	-	x	-	x	-	X
Ulster	Wenigentaft	Uls_Wen	2006	400	0	1	384	x	x	x	x	x	x	x	x	-	-	-	-
Wurm	Frelenberg - FB4	Wur_Fre	2007	500	1	0	251	x	x	-	-	x	x	x	-	-	-	-	X
Zillierbach	Wernigerode	Zil_Wer	2002	8000	0	1	23	-	x	-	-	-	-	-	-	-	x	-	-

Annex 5: MMI values (ecological condition) based on macroinvertebrate samples collected before and after stream restoration in catchment 13.