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REstoring rivers FOR effective catchment Management

Deliverable	D3.1 Impacts of HyMo degradation on ecology
Title	Impacts of hydromorphological degradation and disturbed sediment dynamics on ecological status
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PU	Public	X
PP	Restricted to other programme participants (including the Commission Services)	
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Partner affiliation

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3	AU-NERI
4	BOKU
5	Cemagref
6	DDNI
7	EAWAG
8	Ecologic
9	FVB.IGB
10	JRC
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17	UHULL
18	UNIFI
19	UPM
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21	WULS
22	CEDEX
23	DLG
24	EA
25	ISPRA

Summary

Background

The purpose of WP3 is to address degraded river systems and under D3.1 to specifically look at impacts of hydromorphological degradation on ecological status using existing data. The aim of the work was to begin the development of metrics which indicate the impact of hydromorphological degradation on biota. The authors were conscious of the need of stakeholders to both evaluate current condition but also evaluate the success of river rehabilitation projects.

HYMO indicators of degradation

A possible approach for developing a method of evaluating the ecological and morphological conditions of a river influenced by human intervention is presented. The method is based on a source pool of detailed physical parameters and indicators (metrics) that are linked to the data and outputs of other work packages (WP1 & WP2) within the REFORM Project. Depending on the focus of an evaluation (to choose from morphology, vegetation, benthos and/or fish), experts can use these approaches to identify a subset of key indicators from this pool. When using the approach an evaluation is performed comparatively between the benchmark condition of the river and the river condition affected by human intervention. The output, an informed choice of key metrics, aims to support the stakeholder decision making processes and their ability to target desired project goals. These indicators of degradation should be viewed as an interim solution while a more comprehensive and tested approach is produced from WP2 and the final system developed will be an integral part of WP6. The impact of hydromorphological degradation on individual biological Quality elements is reviewed in the subsequent chapters.

Phytobenthos

It was not possible to detect any effect of the hydromorphological alterations tested, which included alterations that influence the flow velocity, the rate of sedimentation and the in-stream habitat on metrics based on phytobenthos, although it is reassuring that metrics developed to assess eutrophication stress (e.g. TDI, IPS and related indices) appear robust to hydromorphological alteration. Furthermore, it was not possible to demonstrate an effective response of the proposed index of fine sediment stress based on phytobenthos; % motile taxa appears to be related more to nutrient availability than to fine sediment.

Macrophytes

No macrophyte metrics sensitive to hydromorphological pressures exist despite extensive literature on the well-described and consistent responses of vegetation to damming, weed cutting and dredging. To make the best use of pre-existing monitoring datasets a trait-based approach was used to examine the potential for new metrics. Clear differences in the dominance of plant morphotypes were detected between rivers of different geomorphological style, indicating plant responses may differ between different river styles. While general responses to hydromorphological degradation were difficult to

detect in some of the large noisy datasets, clear responses of some plant traits were detected in others. For stakeholders, the results indicate plants have particularly strong potential as useful metrics, especially given their intimate role in geomorphological processes but significant development is required.

Macroinvertebrates

Metrics developed to detect hydrological impairment and hydromorphological degradation were not more discriminative than a number of metrics sensitive to other pressures. These findings leave water managers with a significant challenge when diagnosing the reason for not obtaining good ecological status in a waterbody. The reasons for the lack of sensitivity can be attributed to a number of different factors; explanatory variables which are not measured as part of routine monitoring programmes and hydromorphological assessment schemes that do not necessarily record variables of importance to the in-stream biota. In addition, the present findings suggest that both metric development and sampling scales need to be scrutinised to improve sensitivity.

Fish

Sensitivity of fish to hydromorphological pressures was detected. Logistic regression analyses revealed 69% of the analysed European freshwater species display a significant (>90% c.l.) response to HYMO pressures. Responses could be both positive and negative depending on whether an alteration had improved or degraded the habitat suitability for a species. The confounding effects of multiple stressors on these potential metrics will be elucidated in Deliverable 3.2.

Joint BQE

Initial tests on data on diatoms, fish and invertebrates collected at the same sites revealed that Ecological Quality Ratios for these groups follow broadly similar patterns. Many of the sites examined were subject to multiple pressures, and the results demonstrated that with these data development of a joint metric sensitive to hydromorphological pressures was confounded by the overwhelming signal from sensitivity to water quality. The data, from Finland, exhibit strong water quality gradients but relatively weak hydromorphological gradients.

Fine sediments

Excess fine sediment input is a diffuse form of hydromorphological pressure which is widespread throughout Europe with known or potential impacts on all BQEs. It is not possible to parameterise it effectively using data collected by standard hydromorphological monitoring techniques such as the River Habitat Survey. The project had access to a specialist dataset where this pressure was examined directly, and the response of invertebrates could be elucidated by using traits. In general, invertebrates shifted to smaller taxa in response to fine sediment input suggesting that there is the potential to develop biotic metrics sensitive to this pressure.

Habitats Directive

The vulnerability of fish protected under the Habitats Directive and aquatic vegetation Habitat 3260 were examined. The potential to assess vulnerability using current monitoring techniques was reviewed for fish, and the extent of hydromorphological pressures at Special Areas of Conservation with Habitat 3260 was quantified. The future vulnerability of Habitat 3260 sites to changes in hydrology, driven by climate change and socio-economic scenarios, is presented graphically and suggests increasing vulnerability.

Sediment quality

The study showed that river water quality is not only dictated by diffuse or source point emissions of contaminants; it is also strongly related to the quality of sediments. High discharge events, which may occur more often in the future as predicted in future climate scenarios, may mobilise the associated contaminants. Increased contaminant loads at high discharge are commonly not signaled or detected by monitoring programmes because of the masking effect of particle size and dilution. An impact on ecosystem health in sedimentation areas cannot be excluded, however.

Groundwater

Both data analysis and spatially distributed groundwater-surface modelling showed that groundwater is an important driver of maintaining good environmental flows during dry periods in sandy catchments. Groundwater conditions and environmental flows have deteriorated due to anthropogenic changes over the past 150 years, and climate change will probably amplify this deterioration. Our study showed that catchment-scale alterations may significantly improve groundwater conditions and stream discharge, for instance via changes in groundwater abstraction regime, drainage systems and re-naturalisation projects.

Effect of stream corridor and catchment characteristics

Land-use in both stream corridors and catchments influences macroinvertebrate community composition although natural features such as altitude have to be considered. Significant relationships of a number of macroinvertebrate metrics were found with agriculture and urban land use in stream corridor and catchment. Additionally metrics correlated with proportion of natural or semi-natural vegetation in stream corridor upstream studied sites. These findings imply that land-use data could be a more robust way of assessing impacts than reach specific data and could be more relevant in the decision making process on land-use and HYMO degradation in a catchment management perspective.

Brief overall conclusions / Executive Summary

- There is an acknowledged need among stakeholders that new hydromorphological metrics are required to facilitate site remediation and for reporting at national and European levels.
- Pressure/ impact data were assembled from across Europe. The task was challenging, but useful information was gathered.
- For each major hydromorphological pressure, the physical response gradients of rivers was summarised as diagnostic diagrams.
- For the first time we provide evidence that metrics indicating HYMO impact could be developed from monitoring data on fish and macrophytes.
- For the first time we demonstrate the potential to derive metrics sensitive to fine sediment.

- We provide evidence that phytobenthos (diatoms), invertebrates and macrophytes have the potential to be used in combined metrics.
- We found that many existing macroinvertebrate metrics lack specificity and can provide false positive responses to HYMO pressure, suggesting that disentanglement of multi-stressor responses is critical to good diagnosis.
- There is evidence that aquatic habitats protected under the Habitats Directive will be increasingly vulnerable to hydrological pressures with the changing climate.
- Frequently, overlooked topics such as sediment quality and groundwater issues ought to supplement or be included in HYMO assessments due to their potential for explaining variance in biological datasets.
- Land-use data on a spatial scale beyond the reach scale (corridor and catchment) relates to site-specific macroinvertebrate metrics and could be a more robust way of assessing impacts.

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Contents

1. Introduction

In this introductory chapter we review and synthesise deliverable D3.1 and place the work within the current context of the REFORM project and river management in Europe. The deliverable is the outcome of Task 3.1. in WP3, which terminated in Month 24, and D3.1 reports the work undertaken as well as a number of findings. The core of Task 3.1. is to use existing data on hydromorphological degradation and Water Framework Directive (WFD) biological quality elements to test existing assessment systems and develop new candidate metrics. These new metrics will be refined and undergo further testing later, in Tasks 3.2. and 3.3. The main elements of the work have focused on standardization of data, statistical analyses and subsequent reporting of results in this deliverable D3.1 *"Impacts of hydromorphological degradation and disturbed sediment dynamics on ecological status"*.

The aims of the deliverable are to:

1. Collate existing knowledge from a number of sources and formats:
    a. Collect and standardise existing pan-European monitoring data, including both biological quality elements, data on hydromorphological degradation and other variables to quantify other stressors/multiple stress scenarios.
    b. Identify single datasets or case studies that can be used to investigate relationships that cannot be tested on the large scale dataset (a) due to lack of appropriate data.
    c. Published (grey and peer-reviewed literature) and web-based information on pressures and indicators of hydromorphology.
2. Quantify impacts of human interventions on hydromorphological processes and forms.
3. Review and investigate relationships between biota, biological metrics and hydromorphological pressures including an assessment of the vulnerability to hydromorphological degradation of specific Habitats Directive freshwater species and habitats and the role of fine sediment.

The timeliness of this work was underlined at the end-user conference held in Brussels in February 2013 as part the REFORM project dissemination strategy. Representatives from a wide range of water management organisations, from across Europe, expressed a clear need for improved diagnostic tools. They need tools that indicate whether or not a hydromorphological alteration to rivers causes biological degradation, how serious the degradation is, and how might it be remedied. Stakeholders strongly emphasised the need for new tools (metrics), and many representatives expressed the view that targeted monitoring could provide the data needed to develop such tools. The first aim of D3.1 addresses this question by looking at the suitability of existing data. In summary, despite intercalibration processes, many technical differences remain between the methods of collecting data, especially physical data, and these limit the scope for pan-European analyses. Furthermore, it has been an extremely time-consuming exercise to standardise these datasets as they are highly variable with regard to both parameters measured and units used. Moreover, typically, documentation is in a national language and required translation. Consequently, the crude data were refined into very comprehensive national datasets and pre-existing pan-European datasets such as the intercalibration dataset and data collected as part of the previous EU Framework Programme projects STAR and WISER. Of these national datasets we have asked similar questions allowing consensus opinions to be developed. Subsequent deliverables from WP3 will suggest modifications to field methods and sampling strategy which will include options for standardization to help avoid the difficulties encountered in assembling the REFORM data.

Why are biological metrics sensitive to hydromorphological pressures required? Since the inception of the EU Water Framework Directive, assessment of the quality of waterbodies has been synonymous with ecological status, i.e. the state of biological communities relative to reference conditions. The short answer is therefore that such metrics are required because they identify the pressure primarily responsible for degradation of the ecological status class. Within the legislative framework there is a need for tools that allow managers to diagnose and prescribe local site remediation, tools that facilitate national strategic solutions, and tools that summarise the impact at various levels of the management hierarchy (river basin, national) for reporting to DG Environment. Reporting tools may also be used to inform and educate the general public. So far, this wide array of requirements has been met with a small suite of biological tools (metrics) that are either based on existing water quality metrics or new metrics developed under conditions of some urgency. Consider the origin of biological metrics for freshwaters, when the first biological metrics were designed for use with bacteria and benthic invertebrates to diagnose the causes of deterioration in biota at a site following a pollution incident. A before and after incident condition could be compared. They were designed to be sensitive to organic pollution, eutrophication or other chemical pollutant, pesticides, acidification, etc. Critically, all of these pollutants are not often visible to the naked eye, they may leave a lasting impact on the biota for long periods after being washed downstream, and their impact is determined by their concentration, which declines with downstream mixing. Typically, metrics are reported as single values, or if used to summarise the quality of a large number of sites, in a small number of categories. It is unusual to see them reported in a manner indicating a response trajectory which would be useful, for example, for assessing site restoration. Typically, they are presented as Ecological Quality Ratios (EQRs) that define an observed state in a river against an ideal reference condition. Hydromorphological changes to systems are fundamentally different in that direct changes to fluvial geomorphology are much more visible to the naked eye. A dam is obvious and a channelised river reach is often obvious too, even when it has not been maintained. Their presence is a constant. Whilst hydrological changes are not always directly visible, it is common for them to be monitored directly through channel gauging and abstraction licensing, allowing remedial environmental flows to be set. The question of applying metrics to hydromorphology is therefore somewhat different in terms of diagnosis from chemical pollutants, as the presence of potential geomorphological pressures is known. Therefore, the usefulness of metrics is to encapsulate the response and change to biological elements, which is attributable to hydromorphological alterations. Basically, the purpose is to answer the question: ***Is a hydromorphological alteration a pressure or not?***

In a manner analogous to the work of medical doctors, river managers need tools for diagnosing how ill their patient (river) is and how best to treat them cost effectively. Such a toolbox of metrics would supplement WFD metrics already in place and enshrined through the intercalibration process. It has often been stated that hydromorphology is a mix of many pressures; the term itself is an uneasy amalgam of hydrology and fluvial geomorphology. Each major pressure will require a diagnostic metric which can be used by managers to improve the condition of a site. Furthermore, a single combined metric may be required for reporting purposes.

Both aspects relevant to the scientific community and aspects relevant to the end-user community are addressed in this deliverable. It is, however, important to stress that none of topics addressed is completely exhausted and that the data collected merit further analyses, which will be undertaken at a later stage of the work package. Deliverable 3.3 will specifically address the issue of BQE responses to multiple-stressors. A recurring finding in this report, across BQE groups, is the need to address the biotic response to hydromorphological pressures in a multi-pressure context.

2. Data compilation and standardization

2.1 Introduction

Work package 3 (WP3) is based primarily on analysis of existing monitoring data. One of the first tasks in WP3 was to collect empirical data from monitoring programmes with strong gradients in hydromorphological degradation. The partners in WP3 were asked to supply monitoring data consisting of/in the form of taxa lists for Water Framework Directive biological quality elements: macroinvertebrates, macrophytes, fish and diatoms, together with hydromorphological measures and other pressures such as water quality measured at the same sites as the biological elements.

The hydromorphological measures were to include parameters such as flow data, base-flow index, ground water level, water abstraction, stream power, velocity, slope, substrate composition given as coverage of main types, stream/river dimensions given as width and depth, a national normalized hydromorphological indicator/degradation index, pressures such as erosion, sedimentation, management of in-stream vegetation, riparian vegetation, land use, barriers, dams and channel modifications.

Data on water quality were to be supplied to level out the influence of chemistry on the relationship between biology and hydromorphology. So water chemistry parameters such as nutrients nitrogen (N) and phosphorus (P), alkalinity and pH were to be included in the delivered datasets.

2.2 Received data

Generally, extensive data were submitted, geographically covering most of Europe. Data were contributed by the following institutions:

Country	**Biological quality element**	**Number of sites**	**Hydrology**
The Czech Republic (MASARYK)	Macroinvertebrates - indices		No
Denmark (AU-NERI)	Macroinvertebrates - species Macrophytes – species Fish - species	~200	Yes, available data
Finland (SYKE)	Macroinvertebrates – species Macrophytes – species Moss - species Fish – species Diatoms - species	~80	Yes, for ~30 of the 80 sites
Great Britain (CEH, QMUL)	Macroinvertebrates – species Macrophytes - species	250 265	Yes
Italy (UNIFI)	Macroinvertebrates – family Macrophytes – family Diatoms – species	~100	Yes, for a subset
The Netherlands (ALTERRA)	Macroinvertebrates – species Macrophytes – species Fish – species Diatoms – species	~100 ~10 ~10 8	Only for a few sites

Country	Biological quality element	Number of sites	Hydrology
Sweden (SLU)	Macroinvertebrates – species Macrophytes – species Fish – species	~800	Yes
Spain (Universidad Politécnica de Madrid, CEDEX)	Macroinvertebrates – family Fish – species Diatoms - species	~70 ~200 ~50	Yes
STAR project	Macroinvertebrates – species Macrophytes – species Fish – species Diatoms - species	~100	No
WISER project	Macroinvertebrates – species Macrophytes – species Fish – species Diatoms - species	~1500	Yes

The Austrian partner BOKU has submitted data from one site in River Traun. The data cover the historical, affected and restored conditions and consist of very high quality hydraulic and morphologic data. This type of data enables comparisons on a temporal scale between different conditions. Unfortunately biological quality elements and chemistry have not been submitted.

All four biological quality elements were included in the submitted data, and most data were accompanied by macroinvertebrate taxa lists. In total, 26 datasets encompassing approximately 4000 sampling sites were submitted, macroinvertebrate data including approximately 3200 sites, macrophyte data 1900 sites, fish data 2100 sites and, lastly, diatom cover data encompassing approximately 400 sites. Below, a summary of the data submitted by each country is given.

The Czech Republic A dataset on macroinvertebrates. The dataset does not contain taxa lists; however, a number of calculated metrics are provided. Furthermore, the dataset includes the following abiotic data: coordinates, altitude, Strahler index, Corine data for a 200-meter buffer zone and Corine data on catchments. Storing format: Excel spreadsheets.

Denmark Biological quality elements: macroinvertebrates, macrophytes and fish. The dataset covers approximately 200 sites.

Abiotic data: coordinates, water chemistry, physical data such as substrates and the physical index. Flow data. Storing format: Dbase files.

Finland Biological quality elements: macroinvertebrates, macrophytes, fish (not yet submitted), diatoms and mosses. Dataset covers approximately 80 sites.

Abiotic data: coordinates, water body information, Corine data, water chemistry, RHS and hydromorphological data. Flow data are supplied for 33 sites. Storing format: Excel spreadsheets.

UK Biological data: a dataset on macroinvertebrates (250 sites) and a dataset on macrophytes (265 sites).

Abiotic data, macrophytes: altitude coordinates, chemistry, substrate, bank-full width, slope, median flow, stream power. Macroinvertebrates: chemistry, coordinates, HMS features, HQA features. Flow data included for the invertebrate data. Storing format: Excel spreadsheets and

Access database.

Italy Biological data: Macroinvertebrates, macrophytes, diatoms. Biological data on approximately 100 sites. Macroinvertebrate and macrophyte data are given at family level, other elements at species level.

Abiotic data: coordinates, RHS (most stations), sinuosity, slope, width, sediment, morphological alteration index, morphological quality index, bed configuration, IQM, confinement. Flow data submitted for a subset of the sites. Storing format: Excel spreadsheets.

The Netherlands Biological data: macroinvertebrates (100 sites), macrophytes (10 sites), fish (10 sites) and diatoms (8 sites).

Abiotic data, H&A dataset: dominant substrate, vegetation cover, chemistry and discharge. R&D dataset: morphology, shadow, drainage, substrate, land use, stream profile, discharge and chemistry. R&O dataset: morphology, shadow, drainage, substrate, land use, stream profile, temperature, chemistry. Storing format: Excel spreadsheets.

Spain Biological data: macroinvertebrates (70 sites, family level), diatoms (50 sites) and fish (200 sites).

Abiotic data: coordinates, flow, index for connectivity. IPA index, substrates, IHF index, for some stations data on channelization, morphology (index). For Jucar, a few chemical variables (conductivity, oxygen, pH, water temperature, saturation), QBR, IHF, mean flow. Storing format: Excel spreadsheets, PDF, Word and Access databases.

Sweden Biological data: macroinvertebrates, macrophytes and fish. Approximately 800 sites with biological data.

Abiotic data: coordinates, stream width, mean depth, bed features (water velocity, substrates, aquatic vegetation), riparian vegetation (buffer land cover 30 m, vegetation type 5 m), chemistry and toxic substances. Flow data for a subset of supplied stations. Storing format: Excel spreadsheets.

STAR data Biological data: macroinvertebrates, macrophytes, fish and diatoms for 100 sites.

Abiotic data: coordinates, chemical variables, land use, hydromorphology and microhabitat parameters. Storing format: Excel spreadsheets.

WISER data Biological data: macroinvertebrates, macrophytes and fish. Data for approximately 1500 sites.

Abiotic data: coordinates and other station-specific information, climate, land use chemical variables. Storing format: Access databases.

And here summarized by biological quality elements:

Macroinvertebrates:
The Czech Republic: metrics, a number of phylums/classes/orders listed with abundance data
Denmark: species (abundance)
Finland: species (abundance)
UK: species (abundance)

Italy: family/genus level (abundance)
The Netherlands: species (abundance)
Sweden: species (abundance)
Spain: family (abundance)
STAR: species (abundance)
WISER: species (abundance)

Macrophytes:
Denmark: species (cover)
Finland: species (cover, abundance)
UK: species (abundance)
Italy: species/genus/family (cover)
The Netherlands: species (cover and presence/absence)
Sweden: species (abundance)
Star: species (cover)
WISER: species (cover)

Fish:
Denmark: species (abundance)
The Netherlands: species (abundance)
Sweden: species (abundance)
Spain: species (abundance)
Star: species (abundance)
WISER: species (abundance)

Diatoms:
Finland: species (4 letter code, abundance)
Italy: species (abundance)
The Netherlands: species (abundance)
Spain: species (abundance)
STAR: species (4 letter code, abundance)

Mosses:
Finland: Coverage in percentage with species and family level.

The following tables present overviews of the received datasets divided into country, including information on pressures and states allocated to the specific datasets.

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
The Netherlands (ALTERRA)	R&D Algae (25) Macrophytes (34) Macroinvertebrates (239) Fish (155)	Weed cutting Alteration of riparian vegetation Embankments, levees or dikes Point sources Land use as proxy for point and diffuse source pollution Groundwater	Bed features Organic pollution Physico-chemistry Bank features Riparian vegetation Floodplain features (very basic data)	13 streams, 8 of which have all biological elements

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
		abstraction		
	H&A Algae Macrophytes Macroinvertebrates (>10) Fish (38)	Weed cutting Alteration of riparian vegetation Embankments, levees or dikes Point sources Land use as proxy for point and diffuse source pollution Groundwater abstraction	Bed features Organic pollution Physico-chemistry Bank features Riparian vegetation Floodplain features (very basic data)	2 streams
	R&O Algae Macrophytes Macroinvertebrates (424) Fish	Weed cutting Alteration of riparian vegetation Embankments, levees or dikes Point sources Land use as proxy for point and diffuse source pollution Groundwater abstraction	Bed features Organic pollution Physico-chemistry Bank features Riparian vegetation Floodplain features	81 streams

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
Sweden (SLU)	Dalarna Macroinvertebrates Macrophytes Fish	Weed cutting Sand and gravel extraction, dredging Alteration of riparian vegetation Embankments, levees or dikes Impoundments Hydropeaking Point sources (Land use as proxy for point and diffuse source pollution) Groundwater abstraction	Bed features Organic pollution Physico-chemistry Riparian vegetation (Floodplain features)	~75 sites
	Em	Same as Dalarna	Same as Dalarna	207 sites
	Em study	Same as Dalarna	Same as Dalarna	30 sites
	Emå	Same as Dalarna	Same as Dalarna	10 sites
	South Sweden	Same as Dalarna	Same as Dalarna	10 sites
	National survey	Same as Dalarna	Same as Dalarna	500-750 sites
	NILS	Same as Dalarna	Same as Dalarna	29 sites

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
			+ bank features	

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
UK (CEH)	A: Dried up Macroinvertebrates	Embankments, levees or dikes Point sources Land use as proxy for point and diffuse source pollution Groundwater abstraction Surface water abstraction Channelization (Width of buffer strips)	Bed features Organic pollution Inorganic pollution Physico-chemistry Bank features Riparian vegetation Floodplain features (basic)	250 sites
	B: Country side survey Macroinvertebrates	Embankments, levees or dikes Point sources Land use as proxy for point and diffuse source pollution Groundwater abstraction Surface water abstraction Channelization (Width of buffer strips)	Bed features Inorganic pollution Physico-chemistry Bank features Riparian vegetation Floodplain features (basic)	220 sites

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
WISER Data from Netherlands, Germany, Austria, France	Fish (1567 sites) Macroinvertebrates (2043 sites) Macrophytes (1066 sites)	In stream habitat modified (2366 sites) Alteration of riparian vegetation (3042 sites) Embankments, levees or dikes (2590 sites) Channel form modified (3084 sites) Cross section modified (2828 sites) Impoundments (2368 sites) Hydropeaking (1403	Bed features (2145 sites) Organic pollution (1487 sites) Physico-chemistry (1000-3000 sites) Nutrients (2500 sites) Land use (3692 sites)	

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
		sites) Artificial barriers upstream (220-1838 sites) Artificial barriers downstream (220-1775 sites) Water use 1769 sites) Velocity increase (1495 sites) Water abstraction (1486 sites)		

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
Spain (Ministry of Agriculture, Food and Environment)	Macroinvertebrates Diatoms	IHF fluvial habit index QBR riparian vegetation quality index	Physico-chemical	50 stations

Country	Biological elements (Samples)	Data on listed pressures	Data on listed states	Total number of sites
Czech Republic (MASARYK)	Morava river basin 2007 Waiting for response concerning biological elements	Point sources Land use as proxy for point and diffuse source pollution	Bed features Organic pollution Physico-chemistry Riparian vegetation Floodplain land use	105 sites
	STAR data	Alteration of in-stream habitat Point sources Land use as proxy for point and diffuse source pollution	Bed features Channel planform Organic pollution Physico-chemistry Bank features Riparian vegetation	23 sites
	Becva river	Channelization / cross sectional alterations Alteration of in-stream habitat Embankments, levees or dikes Land use as proxy for point and diffuse source pollution	Floodplain land use Bed features Organic pollution Hydrology Physico-chemistry Bank features Riparian vegetation Floodplain land use	3 sites (107 samples)

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
Finland (SYKE)	Macrophytes Macroinvertebrates Diatoms Fish	Sand and gravel extraction, dredging Impoundments Hydropeaking Point sources Land use as proxy for point and diffuse source pollution	Bed features Organic pollution Physico-chemistry Bank features Riparian vegetation Floodplain features	80 sites

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
UK (QMUL)	RHS Macrophytes	Point sources Diffuse nutrient and fine sediment input Channel station / Cross section alterations Alteration of riparian vegetation Alteration of in-stream habitat Embankments, levees or dikes Anthropogenic alterations in sediment dynamics Land use as proxy for point and diffuse source pollution	Bed features Physico-chemistry Bank features Riparian vegetation	265 sites

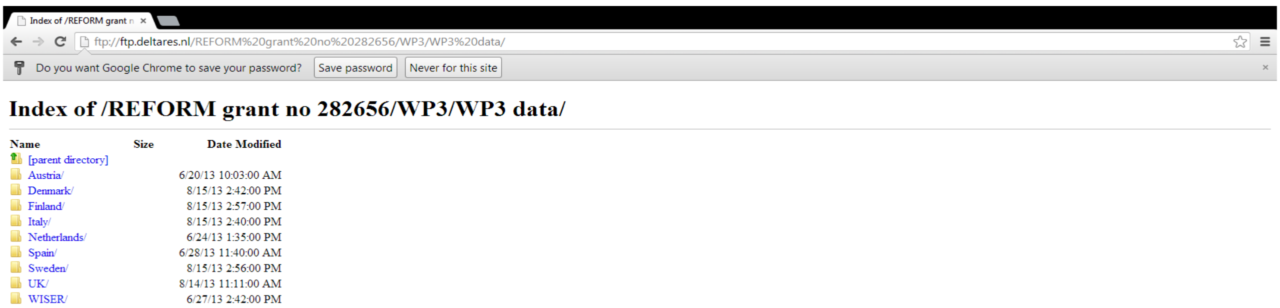
Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
Spain (Universidad Politécnica de Madrid)	Series 1 Duero Macroinvertebrates	Channelization / Cross section alterations Alteration of in-stream habitat Sand and gravel extraction, dredging Alteration of riparian vegetation Embankments, levees or dikes Point sources Surface water abstraction Discharge diversions and returns Inter-basin flow transfer Artificial barriers upstream Artificial barriers downstream Impoundments Diffuse nutrient and fine sediment input Groundwater abstraction	Physico-chemistry Bed features Fine sediment load Low flow and drought Bank features Riparian vegetation Floodplain features	52 sites
	Series 2 Duero Macroinvertebrates Fish	Same as series 1	Same as series 1	17 sites

Country	Biological elements (samples)	Data on pressures	Data on states	Total number of sites
Denmark (AU-NERI)	Macroinvertebrates Fish Macrophytes	Weed cutting Sand and gravel extraction, dredging Channelization / Cross section alterations Alteration of in-stream habitat Alteration of riparian vegetation Artificial barriers upstream Artificial barriers downstream Diffuse source pollution Point sources	Bed features Organic pollution Nutrients Low flow and drought High flow events Physico-chemistry Bank features Riparian vegetation Floodplain Features	~200 sites

Country	Biological elements (samples)	Data on pressures	Data on states	Total number of sites
		Land use as proxy for point and diffuse source pollution Soil type Classification of main pressure Catchment area		

Country	Biological elements (Samples)	Data on pressures	Data on states	Total number of sites
STAR/AQEM (Austria, Czech Republic, Germany, UK, Denmark, Sweden, France, Greece, Italy, Portugal, Poland, Slovakia, Latvia)	Lowlands Diatoms Fish Macrophytes Macroinvertebrates			98 sites
	Mountains Diatoms Fish Macrophytes Macroinvertebrates			86 sites

2.3 Standardising biological data



The abiotic data (hydromorphological measures, chemistry, etc.) have not been standardized but were uploaded as received, and most the datasets have been merged with the standardized taxa lists.

2.4 Conclusion

Substantial monitoring data have been submitted to allow analyses of the relationship between biological quality elements and hydromorphological measures. We had hoped to receive the data as Excel spreadsheets with taxa lists and combined with abiotic measures; however, the data were submitted in a wide variety of ways as to storage media, species names, and hydromorphological and chemical variables. This required extensive standardization of data, and not least the process of standardizing species names was highly time-consuming.

The problems were:

- Data were submitted in different formats. Standardized formats should have been used and agreed upon in the early project state.
- Species names in taxa lists differed widely between the submitted datasets.
- Biological sampling methods differed and were not always described.
- Abiotic variables (values, fractions, names, definitions) and also geo-referenced coordinates were not presented by the same method/submitted in the same format.
- Varying data storage formats were used, e.g. Excel, Access, csv, SAS. In the submitted Excel files the data were differently presented as columns/rows or tables in the middle of the spreadsheets. This presented a challenge and required a huge work effort to make the data readable by the SAS software.

3. Effects of historical and contemporary human interventions on hydromorphological parameters, forms and processes

3.1 Introduction

We often forget that human impacts on hydromorphology began long before the industrialisation in the 18th century. Right from the beginning, human beings had an effect on water, simply because we needed it. For instance, advanced civilizations in Mesopotamia and Egypt built reservoirs (e.g. Wadi Tharthar) and used stream water for irrigation. Thus, water supply and channelization became very important, and a huge network of waterways and channels soon came into existence. In Roman times, there was a tradition of shipping on large rivers in Middle Europe (e.g. the River Donau, the River Inn). In the early Middle Ages people started to use water power and established more waterways. Up to now, industry, water power and flood protection as well as pressures arising from spatial planning and navigation are some of the main factors affecting our river systems (Bayerisches Landesamt für Wasserwirtschaft 2003; Flemming 1967).

This chapter is a review of the effects on hydromorphological parameters, forms and processes caused by human interventions. A key issue in WP3 is the impact of hydromorphological degradation on biota, which calls for an in-depth understanding of abiotic drivers of change. The aim is to give a clear overview of the hydromorphological parameters that are essential drivers for change in the biota and to provide potential insights into the linkages between specific parameters, forms and processes and biological elements. The chapter adds to the knowledge generated as part of other deliverables such as 1.2 and 2.1 within the REFORM project umbrella.

The chapter is subdivided into two main parts: i) a hierarchical setup of abiotic parameters divided into main groups and subgroups and ii) a list of anthropogenic interventions and a proposal for the assignment of main groups, and main subgroups, of abiotic parameters for each impact. The indicators defining the parameters are strongly linked to the indicators used within WP2 (deliverable 2.1; chapter 7).

Furthermore, data reconciliation with deliverable 1.2 was performed. The terminology within this present chapter does not exactly match the terminology of D1.2. The connection between human interventions used within WP3 and the pressures used in WP1 has been made within 3.2.1 Human interventions affecting surface water bodies. The methods produced here should be viewed as complimentary to those reviewed in D1.2 and were originally developed with Austrian requirements in mind.

It should be taken into account that listing of parameters, forms and processes, as well as the allocation to human interventions, are at an early stage, and changes and adaptions will be made as the work progresses during the duration of the project. For this reason, there is no guarantee at this stage that all aspects relating to hydromorphological degradation are fully described and the approach suggested is a proposal to be further developed during the course of the project. Under WP2 detailed work on indicators of hydromorphological processes are under development and it is likely that these will produce more targeted indicators in the latter stages of the REFORM project as part of WP6.

3.2 Abiotic parameters, forms and processes

The following classification of hydromorphological parameters, forms and processes has been prepared in compliance with the list of significant stresses and anthropogenic impacts on the condition of surface waters presented in the Austrian national water management plan (NGP) 2009 (BMLFUW 2009), which is a planning tool for the implementation of the EU Water Framework Directive.

Four main groups are distinguished:

Hydrology

Hydromorphology

Emissions from punctual or diffuse sources (material and physical)

Others

Within several subgroups, a wide range of abiotic parameters are provided based on a broad list of topic-relevant literature.

Selected indicators from WP2 Deliverable 2.1 Table B.4 are shown in **blue letters.**

3.2.1 Hydrology

River-related hydrological indicators

Parameter	Indicator group	Indicator
runoff/discharge	hydrological regime	sh. WP2 D2.1 - 7.2.3 Flow Regime or WP6 discharge regime
		magnitude of average discharge
		mean daily discharge
		daily discharge - coefficient of variation
		base Flow index, %
		extent of intermittency (number of days)
		magnitude of monthly discharge
	discharge /stream flow	annual minima, 1-day mean
		annual minima, 3-day means
		annual minima, 7-day means
		annual minima, 30-day means
		annual minima, 90-day means
		number of zero-flow days
		base flow index: 7day minimum flow/mean flow for year
		annual maxima, 1-day mean
		annual maxima, 3-day means
		annual maxima, 7-day means
		annual maxima, 30-day means
		annual maxima, 90-day means
		channel forming discharge - 2 year return period peak discharge
		channel forming discharge - 10 year return period peak discharge

Parameter	Indicator group	Indicator
		number of low pulses within each water year
		mean or median duration of low pulses (days)
		number of high pulses within each water year
		mean or median duration of high pulses
		rise rates: mean or median of all positive differences between consecutive daily values
		fall rates: mean or median of all negative differences between consecutive daily values
		number of hydrologic reversals
		water level fluctuation
		mean monthly runoff
		alteration of unit stream power from naturalised conditions D2.1 - 7.3.
	low flow	low flow duration
		low flow frequency
		julian date of each annual 1-day minimum
	dynamics of flooding	duration of average floodplain inundations
		flood frequency, 1/yr
		seasonal flood predictability
		timing of floods; day
		julian date of each annual 1-day maximum
		frequency of inundations/floods/peaking
		number of hydropeaking events per year
		julian date of hydropeaking events
		slew rate and rate of descent
		flood peak discharge
	connectivity	lateral connectivity river - floodplain
		proportion of banks with levées / embankments within 0.5 channel width or on bank top D2.1 - 7.3.12
		vertical connectivity (interstitial)
groundwater	groundwater dynamics	interaction from river to groundwater
		relative distances to groundwater surface

Floodplain-related hydrological indicators

Parameter	Indicator group	Indicator
runoff/discharge	flow	surface flow
		base flow
		runoff coefficient
	characteristics of inundations	duration of average floodplain inundations
		date/season of inundation
		frequency of inundations
		development/form of flood wave
	connectivity	lateral connectivity river - floodplain
evapotranspiration	evapotranspiration	actual evapotranspiration
		potential evapotranspiration
interception and snow	interception	storage capacity
		field capacity / duration of interceptional water storage
		relative permeability
	snow	melt water
		depth of snow cover

Parameter	**Indicator group**	**Indicator**
groundwater	groundwater dynamics	duration of snow cover phreatic line connectivity from floodplain to groundwater relative distances to groundwater surface

Catchment area-related hydrological indicators

Parameter	Indicator group	Indicator
precipitation	rainfall	rain magnitude drainage rate
	snow	snow magnitude
runoff/discharge	flow	surface flow base flow runoff coefficient basin drainage ipsometric Curve concentration ratio / time of concentration hydrograph
	connectivity	lateral connectivity river - floodplain
Interception and snow	Interception	storage capacity field capacity / duration of interceptional water storage
	snow	(relative) permeability glacier ratio melt water depth of snow cover duration of snow cover
evapotranspiration	evapotranspiration	actual evapotranspiration potential evapotranspiration
groundwater	groundwater dynamics	phreatic line interaction from river to groundwater relative distances to groundwater surface

3.2.2 Hydromorphology

River-related hydromorphological indicators

Parameter	**Indicator group**	**Indicator**
water body type	fluvial landform	slope /gradient energy grade line length of shoreline ratio between section with and without influence (e.g. of hydropeaking)
longitudinal	water depth	mean water depth variation of water depth (Variation coefficient)
	flow velocity	mean flow velocity surface velocity velocity near river bed variability of flow velocity

Parameter	Indicator group	Indicator
		flow velocity distribution
	wetting	wetted surface / wetted area
	turbulence	reynolds number
	continuum	local longitudinal continuum affected
		regional longitudinal continuum affected
		alteration of Segment longitudinal Continuity D2.1 - 7.2.10
		alteration of reach longitudinal Continuity D2.1 - 7.3.8
lateral	water depth	mean water depth
		variation of water depth (Variation coefficient)
	flow velocity	mean flow velocity
		surface velocity
		velocity near river bed
		variability of flow velocity
		flow velocity distribution
	wetting	wetted surface
	continuum	clogging of hyporheic sediments
		total proportion of the potentially erodible channel margin D2.1 - 7.3.11
		lateral barriers
	turbulence	reynolds number
	roughness	mannings value
	dimension	spatial extension of the aquatic area
		extent of riparian corridor D2.1 - 7.2.7
		structure of riparian corridor D2.1 - 7.2.7
		diversity of channel width and bank forms /processes appropriate to channel type D2.1 - 7.3.3
vertical	flow velocity	mean flow velocity
	turbulence	reynolds number
	connectivity	clogging of hyporheic sediments
		connectivity from river to groundwater
	sediment	soil type
		temporal distribution
		spatial distribution
	sediment relocation	relocation capacity
	erosion	side erosion
		bed erosion
		erodible corridor insufficient for river type D2.1 - 7.3.10
	sedimentation	sedimentation rate
	transport	entrainment
	layer composition	grain size distribution of cover layer substratum (soil texture)
		soil type of cover layer
		hydraulic conductivity of cover layer
		thickness of cover layer / lower layers
		bulk density of cover layer /lower layers
		porosity
		grain size distribution of lower layer substratum (soil texture)
		soit type of lower layer
		hydraulic conductivity of lower layer
		thickness of lower layer to consolidated bedrock /clay

Parameter	Indicator group	Indicator
	turbidity	FTU (Formazin Turbidity Unit)
	roughness	grain roughness
	condition	alteration of bed condition D2.1 - 7.3.9
sediment	sediment composition	grading curve
	sediment input	concentration
		temporal distribution
		spatial distribution
		soit type
sediment	sediment relocation	relocation capacity
	erosion	side erosion
		bed erosion
		erodible corridor insufficient for river type D2.1 - 7.3.10
	sedimentation	sedimentation rate
	transport	entrainment
structures	woody debris	input
		transport
		deposition
		driftwood jam
	geomorphic structures	diversity of geomorphic structures
floating solids		amount of floating matter
ice	surface ice	thickness of ice layer
		duration of cover
		ice accumulation/ice jam
	anchor ice	thickness of ice layer
		duration of cover

Floodplain-related hydromorphological indicators

Parameter	Indicator group	Indicator
sediments	soil composition	height of soil layer
		amount of organic matter
		water retention curve (pF-curve)
		hydraulic conductivity
		porosity
	sediment transport/ movement	erosion (erosivity/erodibility)
		accumulation
		erodibility
structures	geomorphic structures	diversity of geomorphic structures
	cover	riparian margin vegetation structure D2.1 7.3.5
		extent of emergent vegetation (relative to that achievable) D2.1 - 7.3.6
	geometry	spatial extension of the floodplain zone

3.2.3 Material and physical emissions from punctual or diffuse sources

Material and physical emissions from punctual or diffuse sources related to the section **river:**

Parameter	Indicator group	Indicator
punctual	temperature (water)	mean water temperature (emission)
		minimum and maximum values of temperature (emission)
		amplitude of water temperature (emission)
		temporal variation of emission (frequency)
		amplitudes of river and emission
		temperature change within river below emission
		temporal variation
	hydrogen ions activity	pH value
	salinity	salinity
	turbidity	abrasion
		FTU (Formazin Turbidity Unit)
	organic matter	BOD (biological oxygen demand)
punctual		DOC (dissolved organic carbon)
		ammonium
		nitrite
	water characteristics	density
		viscosity
	harmful inorganic matter	COD (chemical oxygen demand)
diffuse (interaction with groundwater, precipitation…)	temperature (water)	spatial variation
		temporal variation
	turbidity	abrasion
		FTU (Formazin Turbidity Unit)
	organic matter	BOD
		DOC (dissolved organic carbon)
		ammonium
		nitrite
	harmful inorganic matter	COD (chemical oxygen demand)
	hydrogen ions activity	pH Value
	salinity	salinity
	water characteristics	density
		viscosity
	oxygen content	oxygen saturation

3.2.4 Others

Parameter	Indicator group	Indicator
solar radiation	shading	degree of shaded surface area
	sunshine duration	sunshine duration
air humidity		absolute humidity
		relative air humidity
wind		velocity of wind
		wind turbulences
		wind direction

Parameter	Indicator group	Indicator
waves		pounding force of waves
physical properties	water characteristics	surface tension
temperature	water temperature	mean temperature
		minimum and maximum values
		amplitude of air temperature
		spatial variation of water temperature
	air temperature	mean temperature
		minimum and maximum values
		amplitude of air temperature

Additional parameters such as climate change (temperature, evapotranspiration, precipitation, snow melt, glacier/ice melt), geology, tectonic, volcanism, acceptance and economic efficiency should be considered. These topics are, however, not taken into account in this chapter.

3.2.5 Elucidation of key literature

Among other sources, references for data were abiotic parameters from the riverine landscape using data from the project Riversmart (Egger et al. 2005), the effect of temperature on rivers according to Wunderlich (1996) and emissions of waste water treatment plants from the Handbook for Evaluating Rehabilitation Projects in Rivers and Streams (Woolsey et al. 2005).

Historical and contemporary human interventions

3.2.6 Human interventions affecting surface water bodies

To ensure that compliance with the other WPs is as comprehensible as possible, a short description is given for all main types of human interventions. This explanation leads to the related pressures within deliverable 1.2 (WP1). For a more detailed description of the related pressures, given in italics, please refer to D1.2.

Several effects can be assigned to a number of 9 main groups of interventions. The groups are strongly linked to the chapter on hydromorphological pressures within D1.2 (connections given in italics). The single hydromorphological pressures defined within D1.2 can often be found in more than one of the nine groups of human interventions.

A distinction of punctual (small scale) and extensive (larger scale) effects on parameters caused by human impacts should be considered. "The forecast-based decision to which extent hydromorphological change is still tolerable with regard to compliance with the quality objective is to be made easier by §5 to the extent that small-scale exceeds of the quality objective in the area of hydromorphological change, which are defined in greater detail do not constitute an obstacle for compliance with the quality objective" (Explanations Austrian Quality Objective Ordinance – Ecological Status of Surface Waters ´Ecology´§5, 2003). It is therefore conceivable that morphological measures such as bank reinforcements yield a much less severe impact than assumed from the range of pressure. It may also be concluded that even if the impacts are punctual, they may have extensive and long-term effects; examples of this are transverse structures, spawning grounds, pollution (material or physical) and hydropeaking.

A – Hydropower [Jungwirth et al. 2003; Gieseck et al., 2009; Kemp et al. 2008; Ugedal et al. 2008]

This effect includes the following pressures used in WP1; D1.2: hydrological regime/water abstraction (discharge diversions and returns), inter-basin flow transfers, hydrological regime modification (flow timing or quantity), hydropeaking, reservoir flushing, sediment discharge, river fragmentation, impoundment, channelization: channel cross section alteration, embankments, levees or dikes, alteration of riparian vegetation and alteration of in-stream habitats.

Possible accompanying measures:

Transverse structures (weirs, dams)/ connectivity interruptions (Nomachi et al. 2013; Magillan & Noslow 2005; Brandt 2000; Poff et al. 2007).

Large dams (transverse)

Longitudinal embankments

Piping, tubing

…

Common effects:

Sediment transport (Walling 2009)

Minimum flow (Mader 1992)

Recess

Woody debris

Water extraction (Egger et al. 2004)

Hydropeaking (Boavida et al. 2013; Young et al. 2011; Bruder et al. 2012; Baumann & Klaus 2003)

Storage

Impoundment (Lozán & Kausch 1996; Vörösmarty et al. 2003)

Ice

…

B – Spatial planning and rural development (Ligon et al. 1995; Mader 2005; Jungwirth 1986; Gregory 2006; Mangelsdorf & Scheuermann 1980; Niehoff 1996; Jürging et al. 2005).

This effect includes the following pressures used in WP1; D1.2: hydrological regime/water abstraction (discharge diversions and returns), river fragmentation, channelization: channel realignment, channelization: channel cross section alteration, sand and gravel extraction, floodplain soil sealing and compaction, alteration of riparian vegetation and alteration of in-stream habitats.

Possible accompanying measures:

Embankments and dams for infrastructure – road and railway network

Surface sealing (settlements)

Soil compaction (e.g. caused by heavy agricultural machinery)

Deforestation / reduction of forests and orchards

Bridges

Cutting off side channels (Wyzga & Zawiejska 2012)

Channelization

Bed stabilisation

Transverse structures

Straightening of the river

Drainage of alluvial area and swamps for land reclamation

Piping / tubing

…

C – Water extraction (Mangelsdorf & Scheuermann 1980, Niehoff 1996, Jürging & Oatt 2005; Mader 2000)

This effect includes the following pressures used in WP1; D1.2: hydrological regime/water abstractions(discharge diversions and returns), inter-basin flow transfers, hydrological regime modification (flow timing or quantity), channelization: channel cross section alteration, river fragmentation, impoundment, alteration of riparian vegetation and alteration of in-stream habitats.

Possible accompanying measures:

Punctual extraction

Diversion hydropower station

Split into several channels (historical mill canals)

Irreversible water withdrawal - transition to other catchment areas

Irreversible water withdrawal - drinking water supply

Irreversible water withdrawal - agriculture and animal husbandry

…

Common effects:

Sediment alteration

Clogging

Silting

D – Flood protection (Vriend 2013; Dorner et al. 2008; Auerswald 2002)

This effect includes the following pressures used in WP1; D1.2: hydrological regime/water abstraction (discharge diversions and returns), hydrological regime modification (flow timing or quantity), reservoir flushing, sediment discharge, impoundment, large dams and reservoirs, embankments, levees or dikes, alteration of riparian vegetation and alteration of in-stream habitats.

Possible accompanying measures:

Artificial levees

Flood retention dams/ basins

Dykes

Bank reinforcement

…

Common effects:

Recess

Loss of structures

…

E – Navigation (waterways) (Arlinghaus et al 2002, Wolter & Arlinghaus 2003)

This effect includes the following pressures used in WP1; D1.2: river fragmentation, embankments, levees or dikes, alteration of riparian vegetation and alteration of in-stream habitats.

Possible accompanying measures:

Locks

Port facilities

Navigation infrastructure

Groynes (Kadota & Asayama 2013)

Channelling

Dredging

…

Common effects:

Recess

Loss of structures

...

F – Pollution (Calow 1994)

This effect includes the following pressures used in WP1; D1.2: sediment discharge, river fragmentation, thermal changes, eutrophication (nutrient changes) and organic discharge, alteration of riparian vegetation and alteration of in-stream habitats.

Common effects:

Emissions from agriculture (Hancock 2002)

Emissions from animal husbandry

Emissions from wastewater treatment plants (Welch & Lindell 1992)

Emissions from infrastructure (roads, railways, …)

Uncontrolled emissions from inhabitants

Pollutant emissions from industry

Emissions from water-driven cooling systems (industry and power stations) (Wunderlich 1996)

Emissions from navigation (and fishery)

…

G – Biological imbalance

This effect includes the following pressures used in WP1; D1.2: impoundments, alteration of riparian vegetation and alteration of in-stream habitats.

Possible accompanying measures:

Fishery management (Hickley et al. 1995; Maitland 1995)

…

Common effects:

Alien Species – Neobiota (Jungwirth et al. 2003)

...

H – Measures for tourism, recreation, leisure and aesthetical reasons (Fillipson et al. 2009)

This effect includes the following pressures used in WP1; D1.2: hydrological regime/water abstraction (discharge diversions and returns), impoundment, alteration of riparian vegetation and alteration of in-stream habitats.

Possible accompanying measures:

Embankments

Water abstraction

Seasonal impact load of wastewater treatment plants

…

Common effects:

Temperature

Pollution

Loss of structures

…

I – Acceleration of climate change (Magillan & Kausch 2005; Nakicenovic, 2000; Gerstengarbe & Werber 1999)

This effect includes the following pressures used in WP1; D1.2: hydrological regime modification (flow timing or quantity), alteration of riparian vegetation and alteration of in-stream habitats.

Common effects:

Change of hydrological parameters – regime (Nester et al. 2005)

Temperature (Lozán & K.H. 1996) …

3.2.7 Description for evaluation of influence

Based on a handful of key indicators this method provides an opportunity to assess he effects of human interventions on rivers. From this knowledge, we can draw important conclusions for the future development of river management. As an additional benefit. the evaluations help us, among other things, to learn from our mistakes, to promote further development of positive effects and influences, and to adapt monitoring methods.

Guidance for evaluation sheets

The evaluation sheets for human interventions are structured as follows:

The title of the evaluation sheet refers to the type of intervention. Therefore, one of the 9 main types of human impacts (3.2.1) has to be chosen:

A – Hydropower
B – Spatial planning and rural development
C – Water extraction
D – Flood Protection
E – Navigation (Waterways)
F – Pollution
G – Biological imbalance
H – Measures for tourism, recreation, leisure and aesthetical reasons
I – Acceleration of Climate Change

As an introduction a general description is provided:

To assure statistical reliability and comparability of the datasets, a number of comparable parameters are required. These include, on the one hand, System A of the Typology of Surface Water Bodies (shown below) and, on the other hand, general information pm hydrological runoff/discharge parameters and geometrical dimensions of the effect of the measure.

Main group of measure: shorthand detailed description of the main measure (e.g. pump storage-power station, waste water outlet, flood retention basin …)

Constructed in [year]: year of completion and, if known, the construction period

Main measure(s): list of additional decisive structures of the measure and/or accompanying measures (e.g. transversal structures, bed stabilisation)

Main impact(s): impacts that reflect the negative environmental effect of pressures (e.g. sediment trapping, recess, loss of structures)

Ecological-oriented measures to reach the state-of-the-art: (vegetation, benthos, fish). The ecological-oriented measures depending on the edited project. A small selection of essential measures is listed below:

Hydropower
- sediment management
- groundwater management
- longitudinal connectivity such as fish pass, minimum flow
- lateral connectivity

Spatial planning and rural development
- stream course morphology
- river bed morphology
- longitudinal connectivity
- lateral connectivity

Water extraction
- longitudinal connectivity such as fish pass, minimum flow

Flood protection
- stream course morphology
- river bed morphology

longitudinal connectivity
lateral connectivity
groundwater management
management of riparian forests and floodplains

Navigation river bed morphology
lateral connectivity (connection of tributaries)

Pollution multiport systems (distribution of input)

Segment length of direct longitudinal influence: approximate value

Spatial distribution: punctual
longitudinal / lateral
spatial

Typology: Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000; System A: (Water Framework Directive):

Ecoregion: shown on map A in Annex XI

1. Iberic-Macaronesian region
2. Pyrenees
3. Italy, Corsica and Malta
4. Alps
5. Dinaric western Balkan
6. Hellenic western Balkan
7. Eastern Balkan
8. Western highlands
9. Central highlands
10. The Carpathians
11. Hungarian lowlands
12. Pontic province
13. Western plains
14. Central plains
15. Baltic province
16. Eastern plains
17. Ireland and Northern Ireland
18. Great Britain
19. Iceland
20. Borealic uplands
21. Tundra
22. Fenno-Scandian shield
23. Taiga
24. The Caucasus
25. Caspic depression

Altitude typology:

	high	(> 800 m)
	mid-altitude	(200 m to 800 m)
	lowland	(< 200 m)

Geology: calcareous
siliceous
organic

Catchment area:	small	(10 to 100 km²)
	medium	(> 100 to 1,000 km²)
	large	(> 1,000 to 10,000 km²)
	very large	(> 10,000 km²)

Datasets / models: provide information on which data the evaluation of the measure is based (e.g. maps, project planning data/documents, field studies).

EEvaluation category: provide information on which topic the evaluation is focused. This focus will mainly match the topic in which the person processing the evaluation is an expert.

focus on morphology
focus on vegetation
focus on benthos
focus on fish

Overview - site plan: provision of pictures, maps and/or project plans to illustrate the affected area

---

Based on the type of human intervention (A-I) and the assigned main measures and impacts for this intervention, the expert handling the evaluation of the measures prepares a list of approximately 10 –15 key indicators for the evaluation scale. These parameters are selected from the extensive list of parameters and indicators provided within topics 3.1.1. – 3.1.4. The accuracy of the indicators should be at the level of indicator groups to provide clear and adequate results. The indicators may vary depending on the eEvaluation category focused upon by the user based on expert knowledge.

There will be no strictly uniform application of a set of indicators for the 9 main types of human interventions as the evaluations and main measures and impacts will differ significantly among different countries and individual interventions. A general description of the measure including the spatial distribution of the measure and general hydrological data is given at the beginning of the characterisation.

The assessment table contains both the evaluation of the chosen key parameters and further general information. Arrangement and order of the indicators follow the structure within 3.1 in terms of hydrological indicators, hydromorphological indicators, material and physical emissions from punctual or diffuse sources and others. It is not mandatory to insert variables of each of the four groups. Information given in this chart should rather be qualitative than quantitative. Four selection options are available for assessing the interference of an indicator related to the measure (no effect / minor interference / moderate interference / high interference). The information provided refers to the condition after completion of the measure and includes the whole impact discussed within the evaluation sheet.
As general information on hydrological indicators, further details regarding the flow regime (WP 6 in prep.) are defined.

The result is demonstrated in the radar chart of rated key indicators where also the values of the benchmark of the river condition are shown. The benchmark of the river condition can refer to the utterly unaffected condition as well as to a modified condition (historical). The graph of the benchmark of the river is assessed following the same scheme as the evaluation of the measure (no effect / minor interference / moderate interference / high interference). The option not evaluated is the only additional possible selection only available for the benchmark conditions. The correct order of the evaluation measures within the graph from the outside to the inside:

- no effect
- minor interference
- moderate interference
- high interference
- not evaluated

The graph illustrates in a simple way which parameters have been seriously affected by human impacts compared to the unaffected river condition, while simultaneously showing which parameters do not exhibit significant changes due to anthropogenic influence.

Based on the graphical visualization, the expert in charge can define a list of recommended measures to improve the current situation. These recommendations may form the basis for further processing and more detailed projects.

Finally, an expert summary statement is provided at the bottom of the evaluation spread sheet. This explanation is to describe special features of the individual human intervention. Alterations of individual parameters should be described relative to the framework conditions of the project. Next, the required measures should be elucidated. The expert providing the recommendation for required measures needs to consider the maintenance of existing rights as well as the functional capability of existing measures.

3.2.8 Application examples

The following section presents examples of application of human interventions affecting surface water bodies relative to most of the nine main groups. The main parameters have been selected with a view to further linkage and development in connection with biotic parameters. As a result some human influences such as pollution and biological imbalance have not been addressed using an example focussing on morphology. Also the acceleration of climate change is not illustrated with the aid of an example within this deliverable.

Two out of 9 of the main groups of human interventions are represented by more than one example.

Two run-of-river power stations were described to illustrate how easy it is to provide a useful decision support for stakeholders by using the described evaluation method.

Two different types of flood protection measures were discussed within the intervention group of flood protection to show that when contemplating the same group of measures, strong variation exists between the most decisive indicators.

Intervention: Hydropower (A)

General description

Main group of measure: run-of-river power station

Constructed in [year]: 2009

Main measure(s): transversal structures, longitudinal embankment

Main impact(s): sediment trapping, recess, hydropeaking

Ecological oriented measures to reach state of the art:

☒ sediment management in year: 2009

☒ groundwater management in year: 2009

☒ longitudinal connectivity (fish pass, minimum flow) in year: 2009

☒ lateral connectivity in year: 2009

Segment length of direct longitudinal influence: 5 to 10 km

Spatial distribution: ☐ punctual ☐ longitudinal / lateral ☒ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Alps Altitude typology: mid-altitude (200 to 800 m)

Geology: calcareous Catchment area: large (>1.000 to 10.000 km²)

Datasets / models: historical maps (Land Register of Francis I.), project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

Figure: Status of the stream morphology a) in the early 19th century and b) in 2010.

<u>Key indicators for evaluation scale</u>

List of key indicators:

- Interaction river – groundwater
- Lateral connectivity river – floodplain
- Longitudinal continuum
- Lateral continuum river – floodplain
- Variation of flow velocities
- Variation of water depths
- Sediment input
- Sediment transport
- Diversity of geomorphic structures
- Effect on water temperature

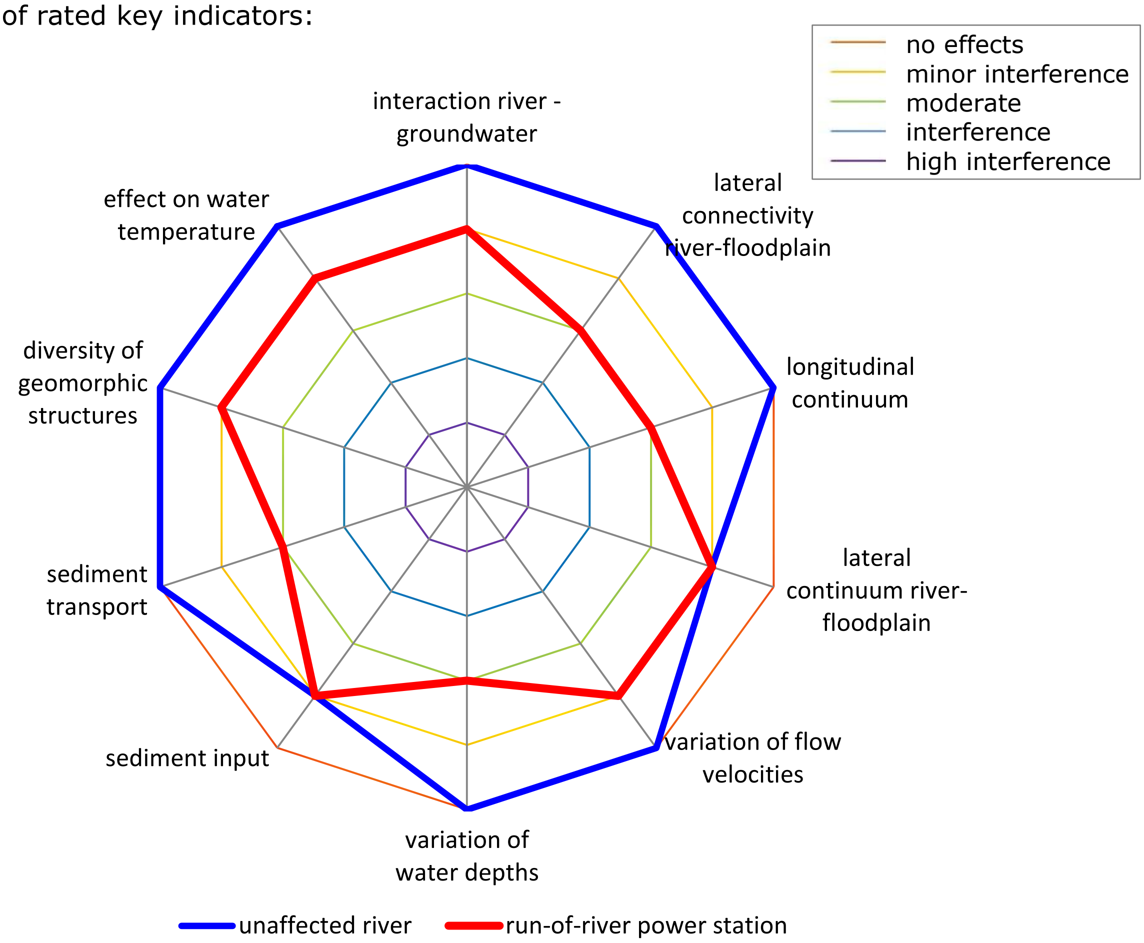
Assessment table:

Hydrological indicators	
Runoff/discharge	Change of flow regime: ☒ no ☐ yes Mean annual discharge [m³/s]: ☐ 1 to 20 ☐ >20 to 100 ☒ >100 to 500 ☐ >500 Lowflow duration [month, period]: winter Occurrence of max. month: June
Groundwater	Interaction from river to groundwater ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference
Runoff/discharge	Lateral connectivity river – floodplain ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Hydromorphological indicators	
Longitudinal	Continuum affected ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Lateral	Interaction from river to groundwater ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference
Runoff/discharge	Lateral continuum river – floodplain ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference Variation of flow velocities ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference Variation of water depths ☐ no affects ☐ minor interference ☒ moderate interference ☐ high interference
Sediment	Sediment input ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference Sediment transport ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Structures	Diversity of geomorphic structures ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference

Others	
Temperature	Effect on water temperature ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference
Turbidity	Effect on water turbidity ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference

Chart of rated key indicators:



Required measures

To enhance the present status, the following measures are recommended to improve the current situation:

- ☐ Interaction river – groundwater
- ☐ Lateral connectivity river – floodplain
- ☐ Longitudinal continuum
- ☐ Lateral continuum river – floodplain
- ☐ Variation of flow velocities
- ☐ Variation of water depths
- ☐ Sediment input
- ☒ Sediment transport
- ☒ Diversity of geomorphic structures
- ☐ Effect on water temperature
- ☐ Others

Expert summary statement

Regarding the analysis of the run-of-river power station project, no atypical effects or measures were determined within the evaluation. The corresponding data fundamentals mainly consist of historical maps (Franziszeische Landesaufnahme) as well as available project documents provided by the operator of the water power station. Because of the initial state of the river concerning stream course and river engineering measures, the interaction of river to groundwater, the lateral connectivity between river and floodplain as well as the sediment input are only slightly affected. A minor impact on water temperature is caused by the impoundment of the power station. Considering the year of construction (2009), it is concluded drawn that the vast majority part of the power plant is planned and constructed according to the state of the art. Taking this fact into account, only very minor improvements are feasible, and it should be noted that all measures to enhance the present (hydromorphological) status would have adverse impacts on existing utilisations.

Intervention: Hydropower (A)

General description

Main group of measure: run-of-river diversion power station

Constructed in [year]: 1871 (conversion 1925)

Main measure(s): run-of-river diversion power station,

transversal structures, longitudinal embankment, water extraction

Main impact(s): sediment trapping, connectivity interruption, minimum flow reach

Ecological oriented measures to reach state of the art:

- ☐ sediment management in year:
- ☐ groundwater management in year:
- ☐ longitudinal connectivity (fish pass, minimum flow) in year:
- ☐ lateral connectivity in year:

Segment length of direct longitudinal influence: 2 km

Spatial distribution: ☐ punctual ☐ longitudinal / lateral ☒ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Alps Altitude typology: mid-altitude (200 to 800 m)

Geology: calcareous Catchment area: large (>1.000 to 10.000 km²)

Datasets / models: historical maps (Land Register of Francis I.), project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

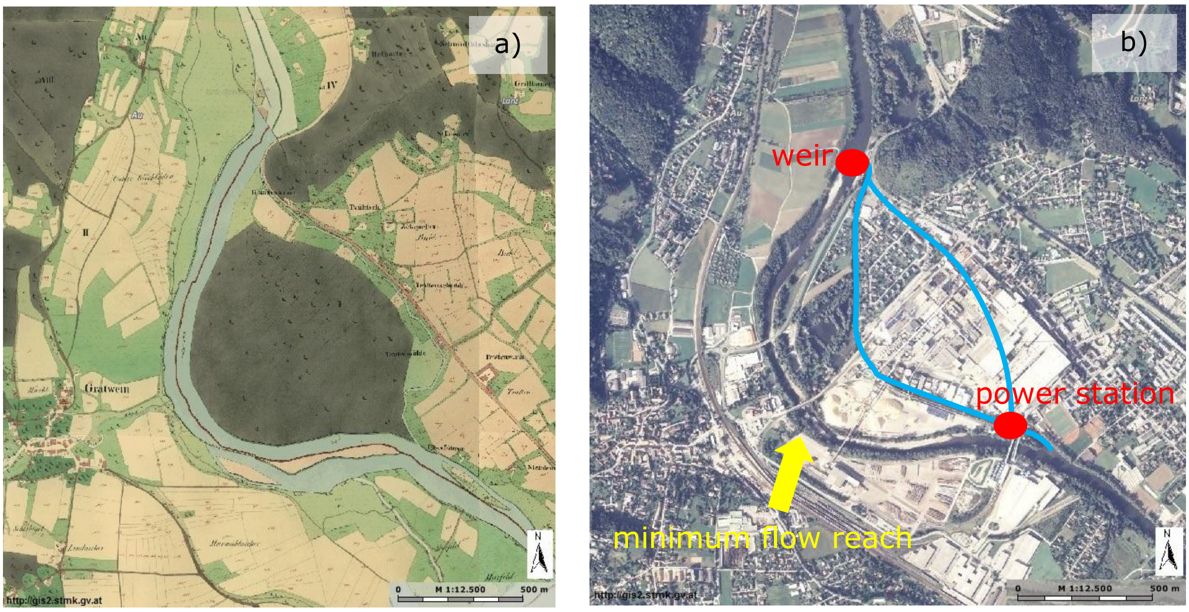


Figure: Status of the stream morphology a) in the early 19th century and b) 2010.

Key parameters for evaluation scale

List of key indicators:

- Interaction river – groundwater
- Lateral connectivity river – floodplain
- Longitudinal continuum
- Lateral continuum river – floodplain
- Variation of flow velocities
- Variation of water depths
- Sediment input
- Sediment transport
- Diversity of geomorphic structures
- Effect on water temperature

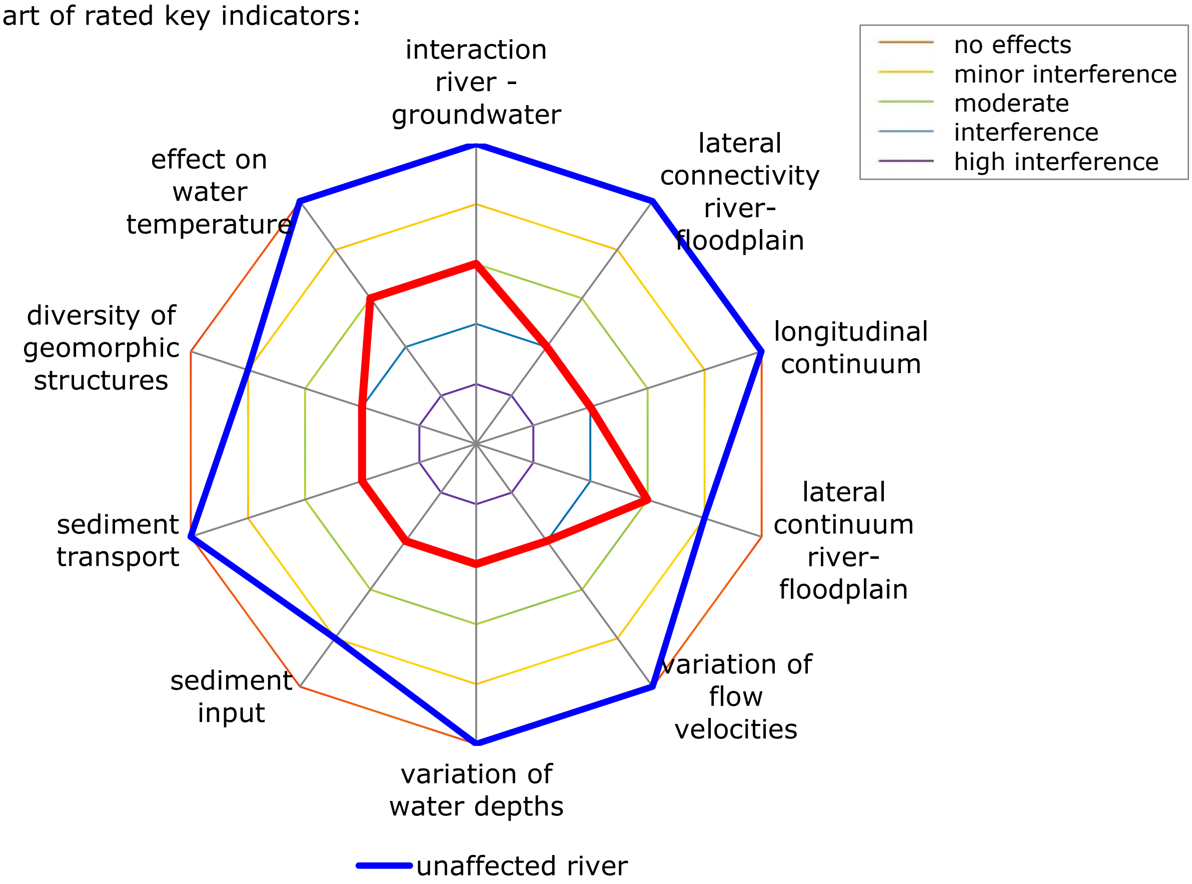
Assessment table:

Hydrological indicators	
Runoff/discharge	Change of flow regime: ☒ no ☐ yes Mean annual discharge [m³/s]: ☐ 1 to 20 ☐ >20 to 100 ☒ >100 to 500 ☐ > 500 Lowflow duration [month, period]: spring Occurrence of max. month: Mai
Groundwater	Interaction from river to groundwater ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Runoff/discharge	Lateral connectivity river – floodplain ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Hydromorphological indicators	
Longitudinal	Continuum affected ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Lateral	Interaction from river to groundwater ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Runoff/discharge	Lateral continuum river – floodplain ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Variation of flow velocities ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Variation of water depths ☐ no affects ☐ minor interference ☐ moderate interference ☒ high interference
Sediment	Sediment input ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Sediment transport ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Structures	Diversity of geomorphic structures ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Material and physical emissions from punctual or diffuse sources	
Temperature	Effect on water temperature ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Turbidity	Effect on water turbidity ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Chart of rated key indicators:



Required measures

To improve the current situation, the following measures are recommended:

- ☐ Interaction river – groundwater
- ☒ Lateral connectivity river – floodplain
- ☒ Longitudinal continuum
- ☐ Lateral continuum river – floodplain
- ☒ Variation of flow velocities
- ☒ Variation of water depths
- ☒ Sediment input
- ☒ Sediment transport
- ☒ Diversity of geomorphic structures
- ☐ Effect on water temperature
- ☐ Other

Expert summary statement

The corresponding data fundamentals mainly consist of historical maps (Franziszeische Landesaufnahme) as well as available project documents provided by the operator of the water power station. Due to the year of construction (1871) as well as the adaption measures performed in the 1920s, it can be confirmed with certainty that the goals of the EC Water Framework Directive have not been taken into account. To sum up measures regarding minimum flow, longitudinal continuum, fish migration and groundwater management should be taken into serious consideration in the near future.

Intervention: Spatial planning and rural development (B)

General description

Main group of measure: straightening of the river

Constructed in [year]: in the 1920s

Main measure(s): cutting off meander, transverse structures

Main impact(s): recess, loss of structures, effects on flooding

Ecological oriented measures to reach state of the art:

☐ stream course morphology in year:

☐ river bed morphology in year:

☒ longitudinal connectivity in year: 1920ies

☒ lateral connectivity in year: 1920ies

Segment length of direct longitudinal influence: approx. 1 km

Spatial distribution: ☐ punctual ☐ longitudinal / lateral ☒ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Alps Altitude typology: mid-altitude (200 to 800 m)

Geology: calcareous Catchment area: large (> 1.000 to 10.000 km²)

Datasets / models: historical maps (Land Register of Francis I.), project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

Figure: Status of the stream morphology a) in the early 19th century and b) in 2010.

Key indicators for evaluation scale

List of key indicators:

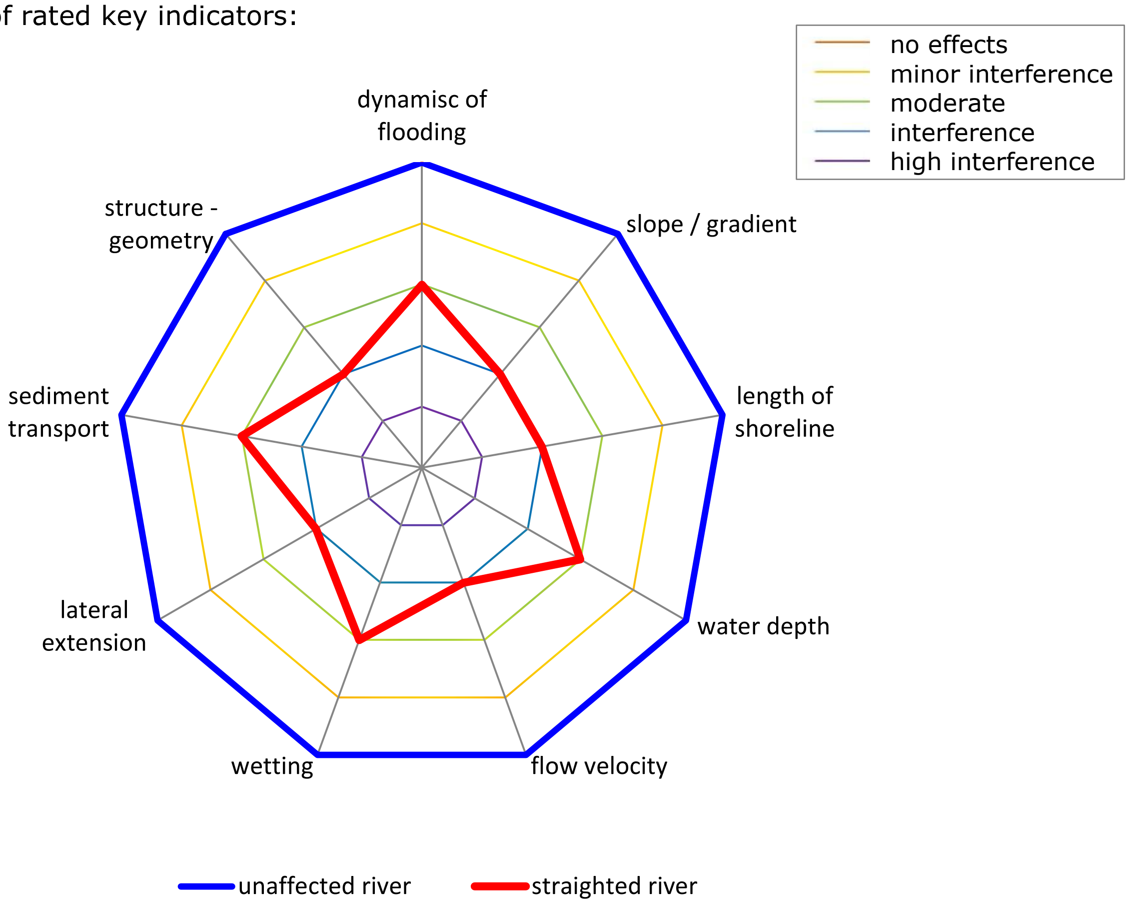
- Dynamics of flooding
- Slope / gradient
- Length of shoreline
- Water depth
- Flow velocity
- Wetting
- Lateral extension
- Sediment transport
- Structure - geometry

Assessment table:

Hydrological indicators	
Runoff/discharge	Change of flow regime: ☒ no ☐ yes Mean annual discharge [m³/s]: ☐ 1 to 20 ☒ >20 to 100 ☐ >100 to 500 ☐ > 500 Lowflow duration [month, period]: winter Occurrence of max. month: June
Dynamics of flooding	Effects on flood peak discharge ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Hydromorphological indicators	
Fluvial landform	Slope / gradient ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Length of shoreline ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Water depth	Variation of water depth ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Flow velocity	Flow velocity distribution ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Lateral	Wetting ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference Spatial extension on the aquatic area ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Sediment	Sediment transport ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Structures	Diversity of geomorphic structures ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Chart of rated key indicators:



Required measures

To enhance the present status, the following measures are recommended to improve the current situation:

- ☐ Dynamics of flooding
- ☒ Slope / gradient
- ☒ Length of shoreline
- ☐ Water depth
- ☒ Flow velocity
- ☐ Wetting
- ☒ Lateral extension
- ☒ Sediment transport
- ☒ Structure - geometry
- ☐ Other

Expert summary statement

At the end of the 19th and beginning of the 20th century large river engineering measures were implemented all over Europe for land reclamation and flood protection purposes. The primary aim of this was to achieve fast flood conveyance including removal of sediments. This was realised by installing hydraulically optimised channels as fixed trapezoidal river cross sections with improved flow characteristics and maximised discharge capacity in combination with bank reinforcements. Unfortunately, this river morphology adjustments, which were carried out by artificial adaption of the stream course related to the valley landscape, led to atypical stream geometries. Furthermore, the resulting modifications such as long plain curves instead of meander swith secondary channels dramatically influenced river properties (length of shoreline, lateral extension, river slope, flow velocity etc.). Hence, the natural structures of the complex system of the unaffected river almost disapperared completely. Of course, for a short period of time these measures improved the flood protection. However, as long-term, adverse consequences recess, monotony, lowering of the groundwater level, sorting of the sediment and also reduced flood protection occurred. It is recommended to consider an extensive restoration project in terms of river widening in the area of the former meander. A possible difficulty might arise taking into account existing utilisation and water rights.

Intervention: Water extraction (C)

General description

Main group of measure: water extraction, diversion hydro power station

Constructed in [year]: 1924

Main measure(s): punctual extraction, transversal structures, piping / tubing

Main impact(s): recess, loss of structures, impoundment, minimum flow reach

Ecological oriented measures to reach state of the art:

☒ longitudinal connectivity (minimum flow) in year: 1924

Segment length of direct longitudinal influence: approx. 11 km

Spatial distribution: ☐ punctual ☒ longitudinal / lateral ☐ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Central highlands Altitude typology: mid-altitude (200 to 800 m)

Geology: siliceous Catchment area: medium (> 100 to 1.000 km²)

Datasets / models: historical maps (Land Register of Francis I.), project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

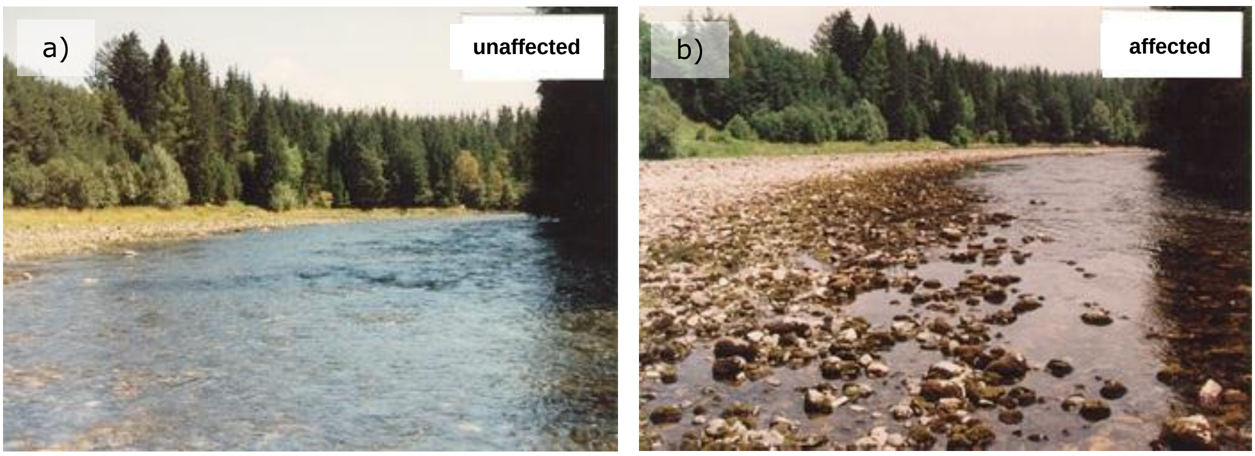


Figure: Status of the stream morphology a) unaffected and b) affected.

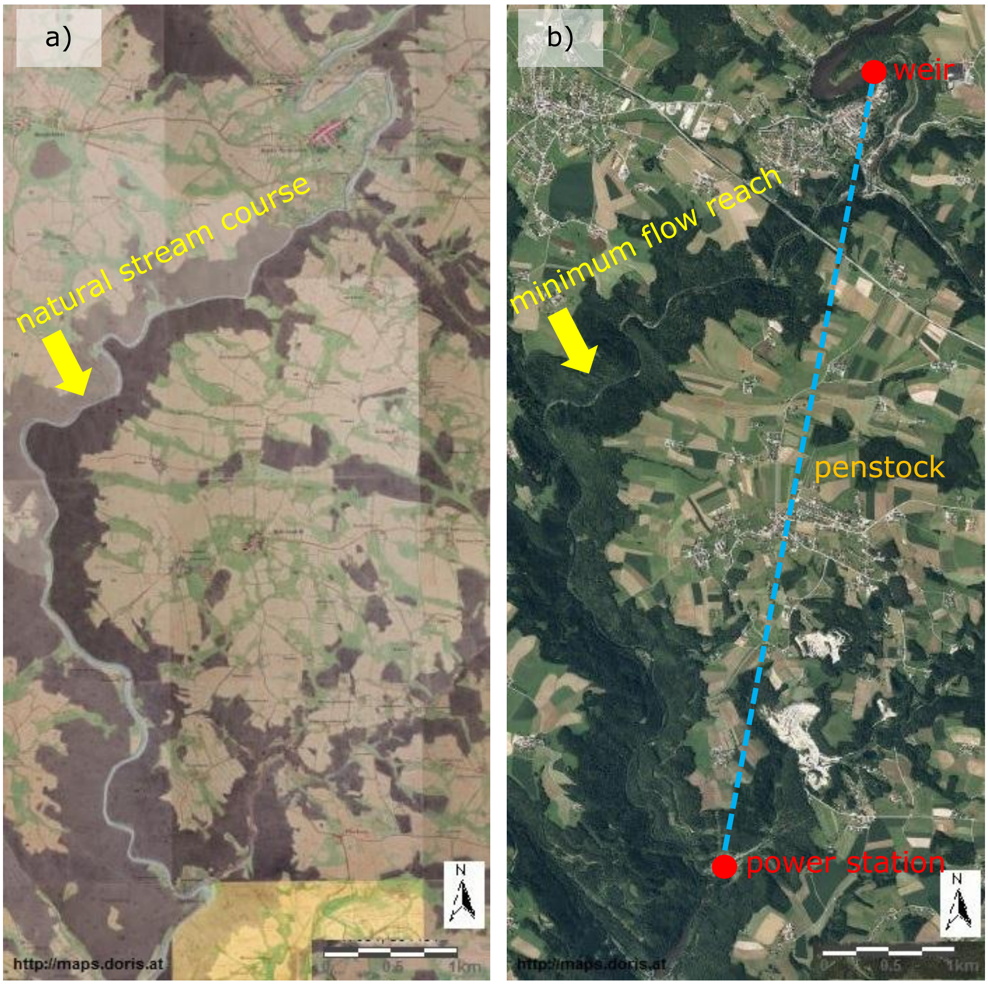


Figure: Status of the stream morphology a) in the early 19th century and b) in 2010.

Key indicators for evaluation scale

List of key indicators:

- Change of flow regime
- Interaction river – groundwater
- Energy gradient line
- Mean water depth
- Variation of water depth
- Mean flow velocity
- Velocity near river bed
- Effect on continuum
- Sediment composition
- Sediment transport
- Effect on water temperature

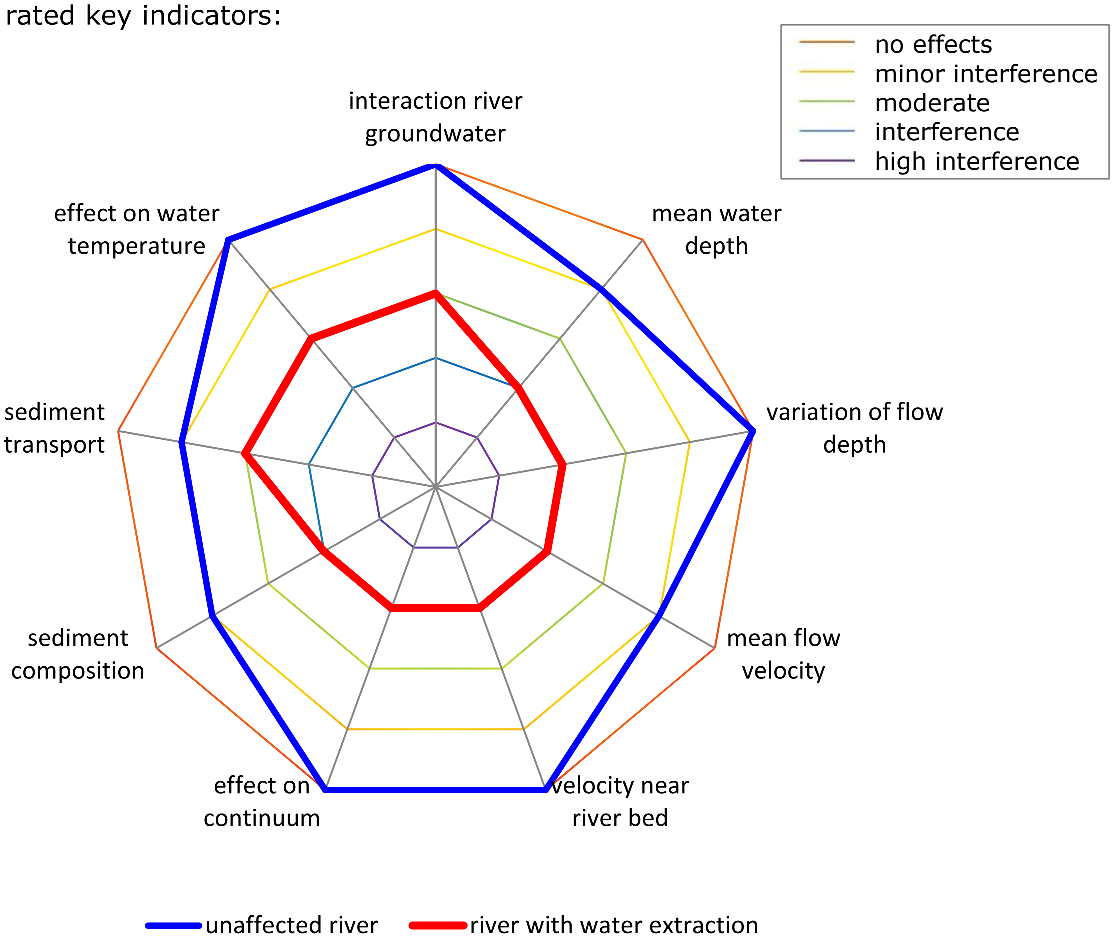
Assessment table:

Hydrological indicators	
Runoff/discharge	Change of flow regime: ☐ no ☒ yes Mean annual discharge [m³/s]: ☒ 1 to 20 ☐ > 20 to 100 ☐ > 100 to 500 ☐ > 500 Lowflow duration [month, period]: winter Occurrence of max. month: March / April
Groundwater dynamics	Interaction from river to groundwater ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Hydromorphological indicators	
Water Depth	Mean water depth ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Variation of water depth ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Flow velocity	Mean flow velocity ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Velocity near river bed ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Continuum	Longitudinal continuum affected ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Sediment	Sediment composition / grading curve ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Sediment transport ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Others	
Temperature	Effect on water temperature ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Chart of rated key indicators:



Required measures

To enhance the present status, the following measures are recommended to improve the current situation:

:

- ☒ Change of flow regime
- ☐ Interaction river - groundwater
- ☒ Mean water depth
- ☒ Variation of water depth
- ☒ Mean flow velocity
- ☒ Velocity near river bed
- ☒ Effect on continuum
- ☒ Sediment composition
- ☐ Sediment transport
- ☐ Effect on water temperature
- ☐ Other

Expert summary statement

As a result of the establishment of the water power station, a relatively long minimum water section was developed. For a long time the river bed was drained completely. In the course of a study, which was one of the main datasets used for this evaluation, it was successfully proved that in the current situation all prescribed hydromorphological parameters are met. Nevertheless, significant degradation of all indicators assessed appears within the chart of rated key indicators. Therefore, further hydrological restoration would be advantageous. It is recommended to prescribe an ecologically oriented minimum water flow management to achieve the goals of the good ecological status according to the EC Water Framework Directive. Methods to determine values for minimum flow management should be based on state of the art such as habitat modelling and in-stream flow tests.

Intervention: Flood protection (D)

General description

Main group of measure: flood control reservoir

Constructed in [year]: 2003

Main measure(s): transverse structures

Main impact(s): effects on peak discharge at flooding, temporal impoundment

Ecological oriented measures to reach state of the art:

- ☐ stream course morphology in year:
- ☐ river bed morphology in year:
- ☒ longitudinal connectivity (passable for fish) in year: 2003
- ☐ lateral connectivity in year:
- ☐ groundwater management in year:
- ☐ management of riparian forests and floodplains in year:

Segment length of direct longitudinal influence: approx. 50 m

Spatial distribution: ☒ punctual ☐ longitudinal / lateral ☐ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Alps Altitude typology: mid-altitude (200 to 800 m)

Geology: calcareous Catchment area: small (10 to 100 km²)

Datasets / models: historical aerial photograph, project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

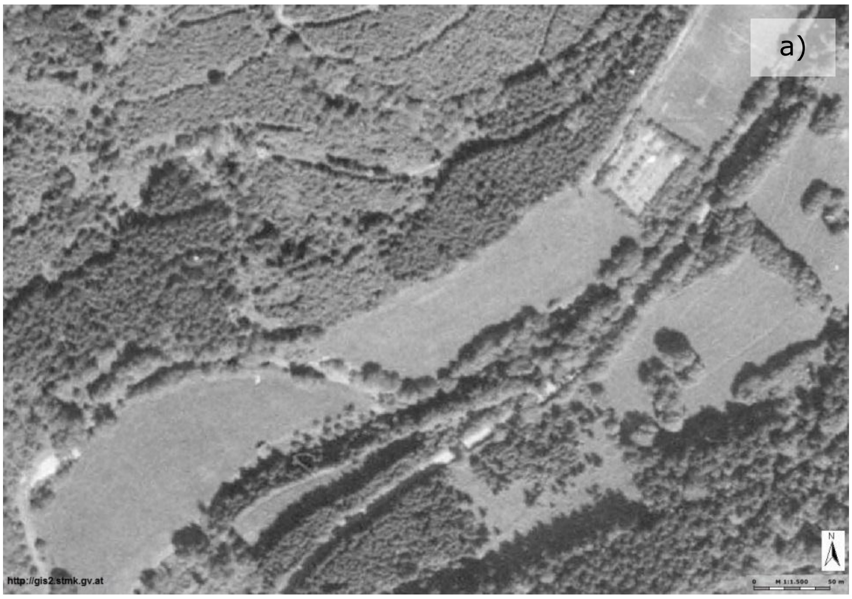


Figure: Status of the stream morphology a) in 1991 and b) in 2011.

Key indicators for evaluation scale

List of key indicators:

- Dynamics of flooding
- Interaction river – groundwater
- Slope / gradient
- Length of shoreline
- Variation of water depths
- Variation of flow velocities
- Wetting
- Lateral extension
- Sediment transport

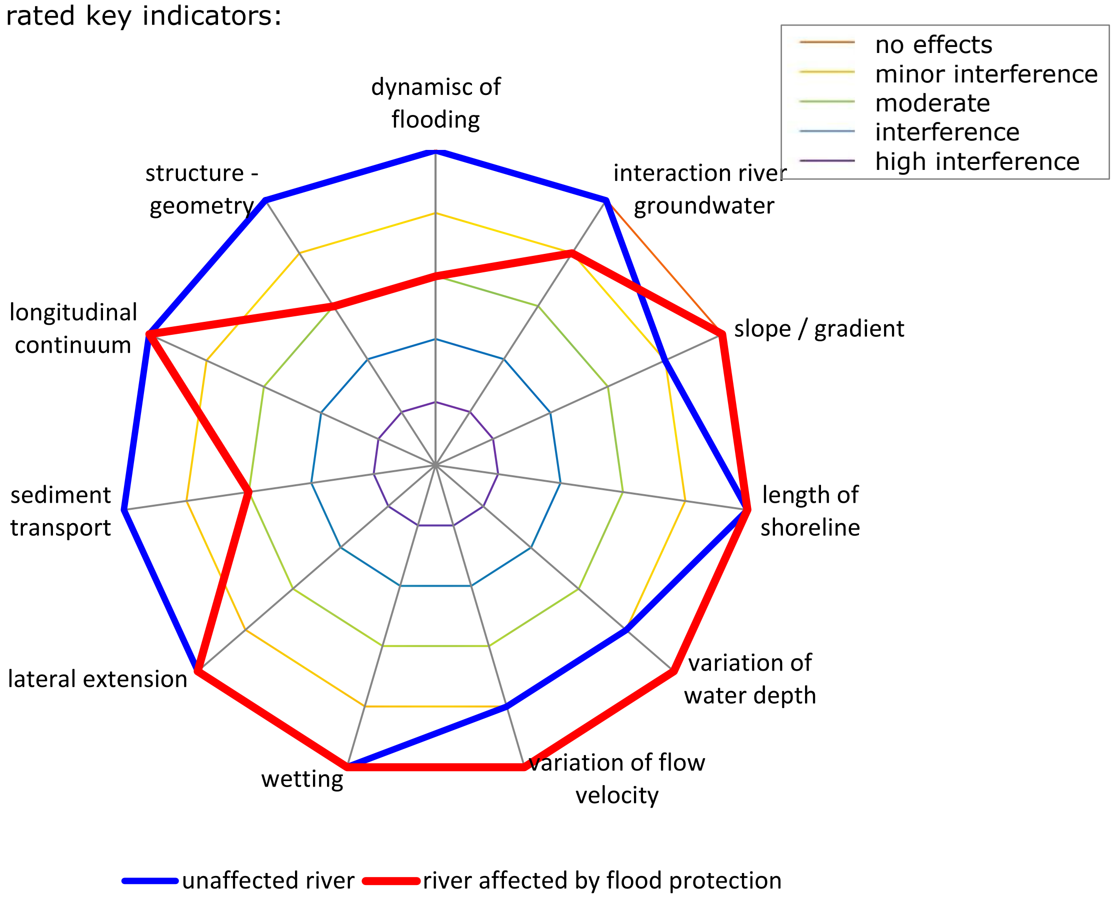
- Longitudinal continuum
- Diversity of geomorphic structures

Assessment table:

Hydrological indicators	
Runoff/discharge	Change of flow regime: ☒ no ☐ yes Mean annual discharge [m³/s]: ☒ 1 to 20 ☐ >20 to 100 ☐ >100 to 500 ☐ > 500
Dynamics of flooding	Effects on flood peak discharge ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Groundwater	Interaction from river to groundwater ☐ no effects ☒ minor interference ☐ moderate interference ☐ high interference

Hydromorphological indicators	
Fluvial landform	Slope / gradient ☒ no effects ☐ minor interference ☐ moderate interference ☐ high interference Length of shoreline ☒ no effects ☐ minor interference ☐ moderate interference ☐ high interference
Water depth	Variation of water depth ☒ no effects ☐ minor interference ☐ moderate interference ☐ high interference
Flow velocity	Variation of flow velocities ☒ no effects ☐ minor interference ☐ moderate interference ☐ high interference
Lateral	Wetting ☒ no effects ☐ minor interference ☐ moderate interference ☐ high interference Spatial extension on the aquatic area ☒ no effects ☐ minor interference ☐ moderate interference ☐ high interference
Sediment	Sediment transport ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Longitudinal	Continuum affected ☒ no effects ☐ minor interference ☐ moderate interference ☐ high interference
Structures	Diversity of geomorphic structures ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Chart of rated key indicators:



Required measures

To enhance the present status, the following measures are recommended to improve the current situation:

- ☐ Dynamics of flooding
- ☐ Interaction river – groundwater
- ☐ Slope / gradient
- ☐ Length of shoreline
- ☐ Variation of water depths
- ☐ Variation of flow velocities
- ☐ Wetting
- ☐ Lateral extension
- ☐ Sediment transport
- ☐ Longitudinal continuum
- ☐ Diversity of geomorphic structures
- ☐ Other

Expert summary statement

Apart from flood events only minor differences could be discerned when comparing the initial condition and the area after the establishment of the flood protection measure. Existing measures show good environmental compatibility. Currently there is no necessity to take further actions to enhance the present status.

Intervention: Flood protection (D)

General description

Main group of measure: flood protection

Constructed in [year]: approx. 1830

Main measure(s): longitudinal structures, embankments

Main impact(s): effects on peak discharge at floodings, recess, loss of structures

Ecological oriented measures to reach state of the art:

- ☐ stream course morphology — in year:
- ☐ river bed morphology — in year:
- ☒ longitudinal connectivity (passable for fish) — in year: 1830
- ☐ lateral connectivity — in year:
- ☐ groundwater management — in year:
- ☐ management of riparian forests and floodplains — in year:

Segment length of direct longitudinal influence: approx. 60 km

Spatial distribution: ☐ punctual ☒ longitudinal / lateral ☐ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Alps — Altitude typology: mid-altitude (200 to 800 m)

Geology: calcareous — Catchment area: medium (100 to 1.000 km²)

Datasets / models: local historical maps, project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

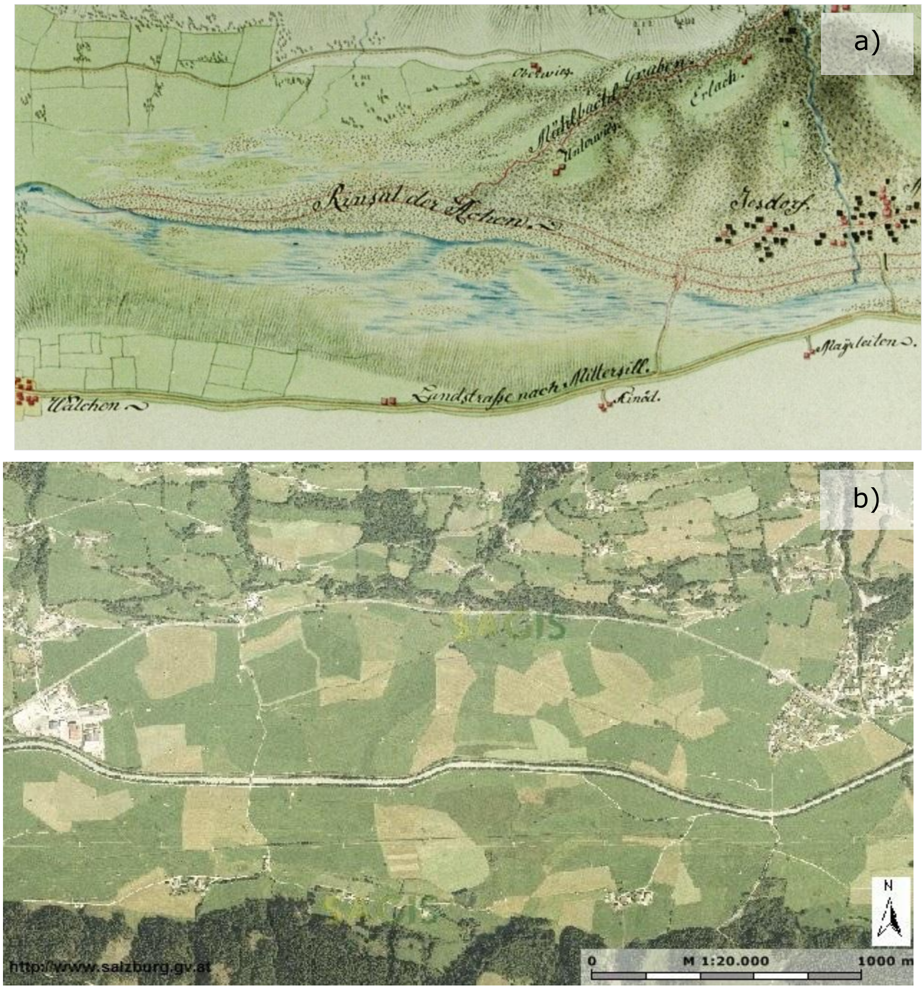


Figure: Status of the stream morphology a) in the 16th century (scheme) and b) in 2004

Key indicators for evaluation scale

List of key indicators:

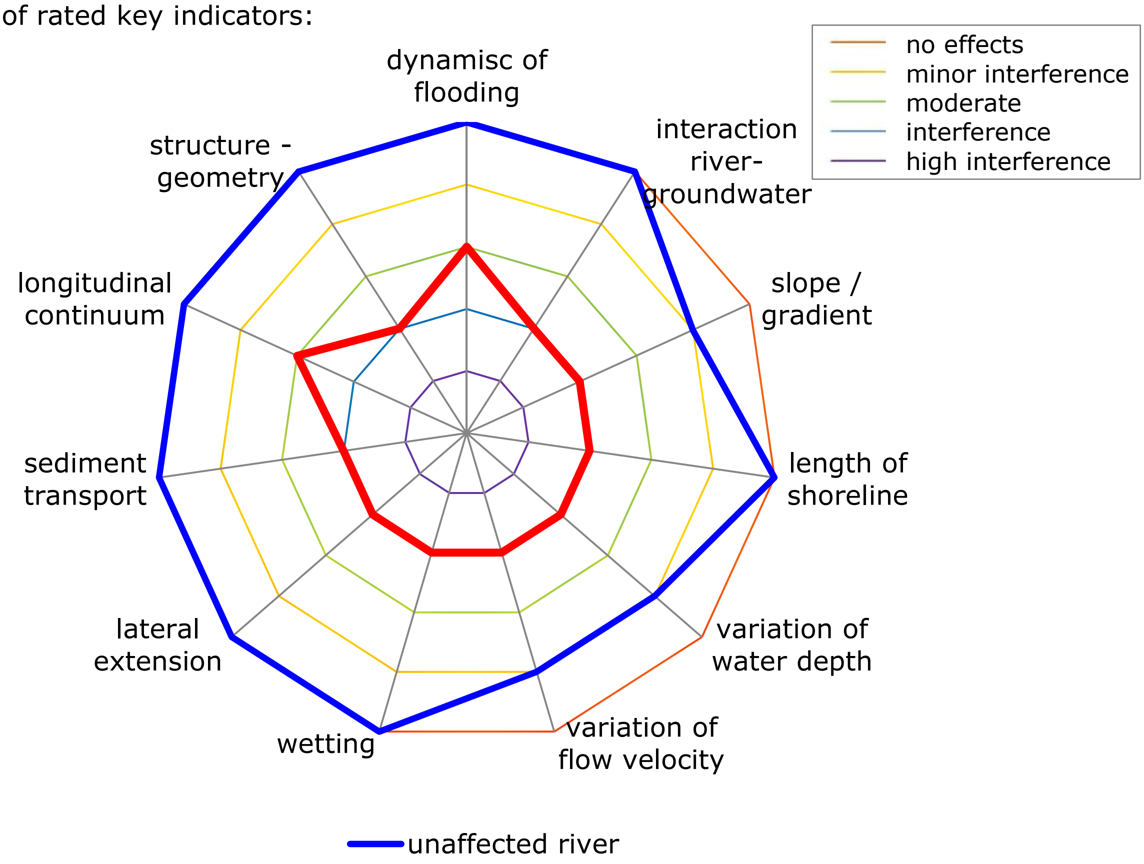
- Dynamics of flooding
- Interaction river – groundwater
- Slope / gradient
- Length of shoreline
- Variation of water depths
- Variation of flow velocities
- Wetting
- Lateral extension
- Sediment transport
- Longitudinal continuum
- Diversity of geomorphic structures

Assessment table:

Hydrological indicators	
Runoff/discharge	Change of flow regime: ☒ no ☐ yes Mean annual discharge [m³/s]: ☐ 1 to 20 ☒ >20 to 100 ☐ >100 to 500 ☐ > 500
Dynamics of flooding	Effects on flood peak discharge ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Groundwater	Interaction from river to groundwater ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Hydromorphological indicators	
Fluvial landform	Slope / gradient ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Length of shoreline ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Water depth	Variation of water depth ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Flow velocity	Variation of flow velocities ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Lateral	Wetting ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Spatial extension on the aquatic area ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Sediment	Sediment transport ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Longitudinal	Continuum affected ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Structures	Diversity of geomorphic structures ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Chart of rated key indicators:



Required measures

To enhance the present status, the following measures are recommended to improve the current situation:

- ☐ Dynamics of flooding
- ☒ Interaction river – groundwater
- ☒ Slope / gradient
- ☒ Length of shoreline
- ☒ Variation of water depths
- ☒ Variation of flow velocities
- ☒ Wetting
- ☒ Lateral extension
- ☒ Sediment transport
- ☐ Longitudinal continuum
- ☒ Diversity of geomorphic structures
- ☐ Other

Expert summary statement

The measures have caused dramatic changes in shoreline length and lateral extension. Natural structures of the complex system of the unaffected river, which covered almost the whole valley floor, nearly disappearedcompletely. The river slope and therefore also flow velocities increased considerably. Almost all measures to enhance the present status will have adverse impacts on the existing utilisation.

Intervention: Navigation (Waterways) (E)

General description

Main group of measure: preservation of a navigable waterway

Constructed in [year]: beginning in 1870

Main measure(s): groynes, dredging, bank reinforcement

Main impact(s): recess, loss of structures

Ecological oriented measures to reach state of the art:

☐ river bed morphology in year:

☐ longitudinal connectivity (connection of tributories) in year:

Segment length of direct longitudinal influence: approx. 50 km

Spatial distribution: ☐ punctual ☒ longitudinal / lateral ☐ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Hungarian lowlands Altitude typology: lowland (< 200 m)

Geology: siliceous Catchment area: very large (> 10.000 km²)

Datasets / models: historical maps (Land Register of Francis Joseph I.), project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

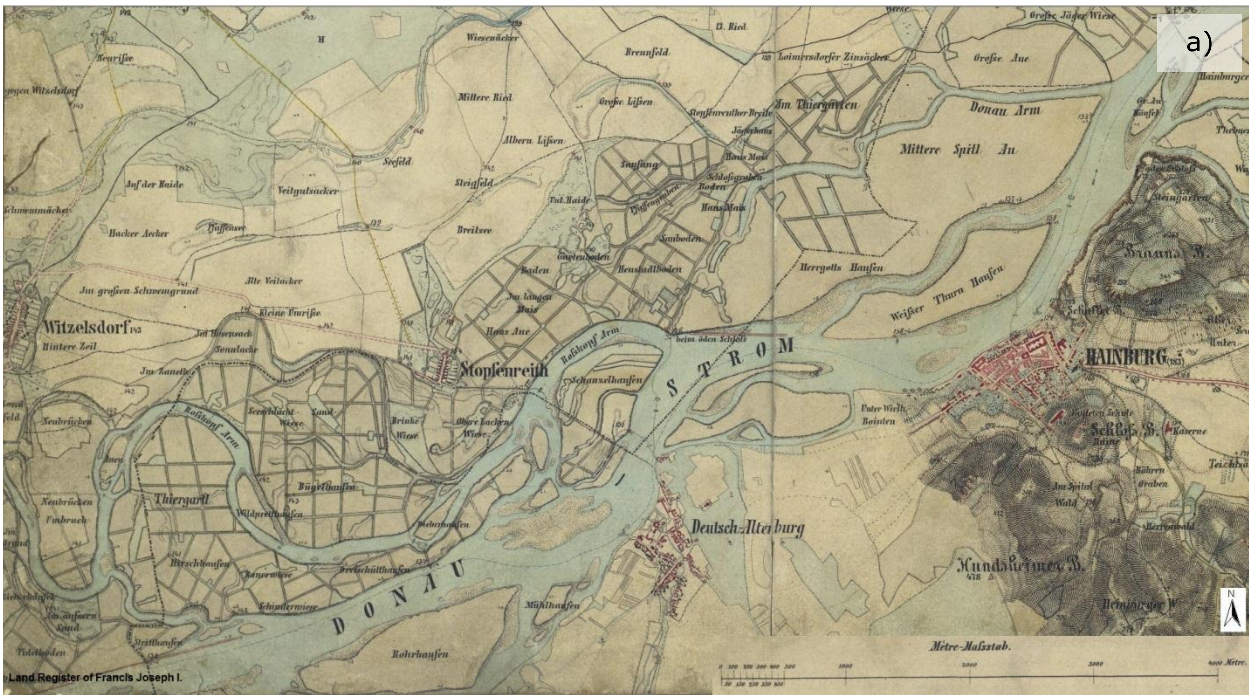


Figure: Status of the stream morphology a) at the end of the 19th century and b) in 2010

Key indicators for evaluation scale

List of key indicators:

- Groundwater dynamics
- Water depth
- Flow velocity
- Turbulence
- Turbidity
- Sediment relocation
- Erosion
- Structure - geometry
- Waves

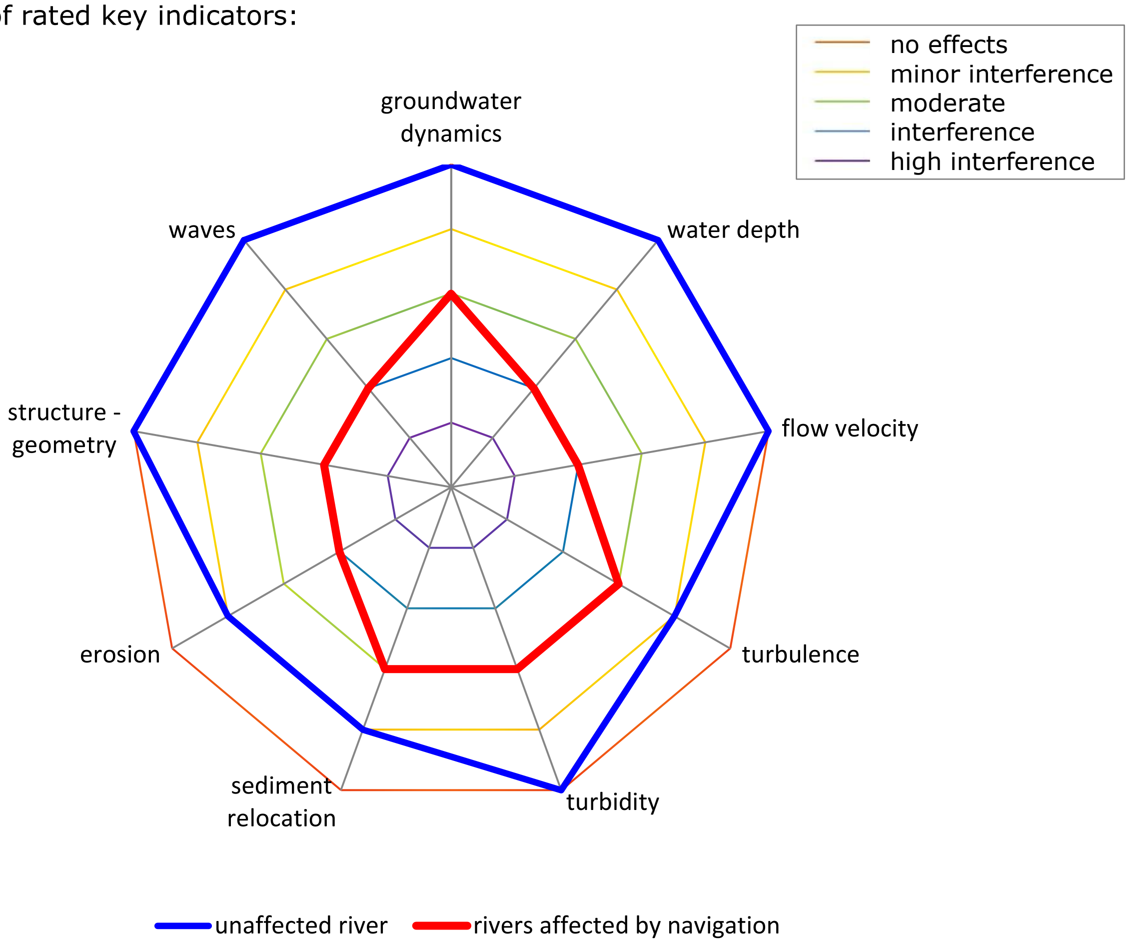
Assessment table:

Hydrological indicators	
Runoff/discharge	Change of flow regime: ☒ no ☐ yes Mean annual discharge [m³/s]: ☐ 1 to 20 ☐ > 20 to 100 ☐ > 100 to 500 ☒ > 500
Groundwater dynamics	Relative distance to groundwater surface ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Hydromorphological indicators	
Water depth	Variation of water depth ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Flow velocity	Flow velocity distribution ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Turbulence	Turbulence ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Turbidity	Turbidity ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Sediment	Sediment relocation ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference Erosion ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Structures	Geomorphic structures ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Others	
Waves	Effect of waves ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Chart of rated key indicators:



Required measures

To enhance the present status, the following measures are recommended to improve the current situation:

- ☐ Groundwater dynamics
- ☒ Water depth
- ☒ Flow velocity
- ☐ Turbulence
- ☐ Turbidity
- ☐ Sediment relocation
- ☒ Erosion
- ☒ Structure - geometry
- ☒ Waves
- ☐ Other

Expert summary statement

In this evaluation only the impacts of construction of a navigable waterway is considered despite that other river engineering measures exist. The main impacts of this particular intervention are the progressive recess, groundwater table drawdown and drying up of riparian forests. Flooding dynamics in connection with the riparian forests as well as the connection to oxbow lakes and cut-off meanders decreased dramatically and most of the transitional structures therefore disappeared. Simultaneously, high flow velocities and significant erosion processes were observed within the navigable waterway. In principle, the question whether rivers should be adjusted to the requirements of ships calls for consideration. Near-natural water management implies matching of river shipping needs with the existing size of a river

Intervention: Measures for tourism, recreation, leisure and aesthetical reasons (H)

General description

Main group of measure: bathing area, wooden lounge jetties

Constructed in [year]: 1900

Main measure(s): recreational structures

Main impact(s): change of structures and geometry, sediment conditions, turbidity

Segment length of direct longitudinal influence: 5 km

Spatial distribution: ☐ punctual ☐ longitudinal / lateral ☒ spatial

Typology: (EC WFD SYSTEM A)

Ecoregion: Hungarian lowlands Altitude typology: lowland (< 200 m)

Geology: siliceous Catchment area: very large (> 10.000 km²)

Datasets / models: historical maps (Land Register of Joseph II.), project data

Evaluation category: ☒ focus on morphology ☐ focus on vegetation

☐ focus on benthos ☐ focus on fish

Overview - site plan:

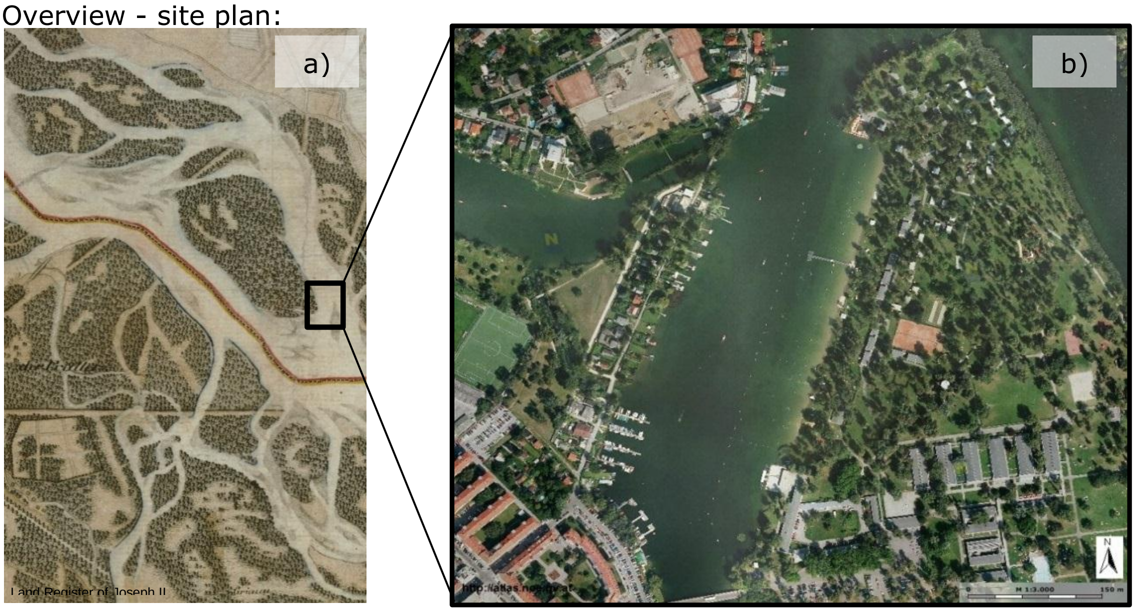


Figure: Status of the stream morphology a) in 1790 (scheme) and b) in 2011.

Key indicators for evaluation scale

List of key indicators:

- Flow regime
- Length of shoreline
- Wetting
- Clogging of hyporheic sediments
- Sediment distribution
- Diversity of structures
- Turbidity
- Effects on shading

Assessment table:

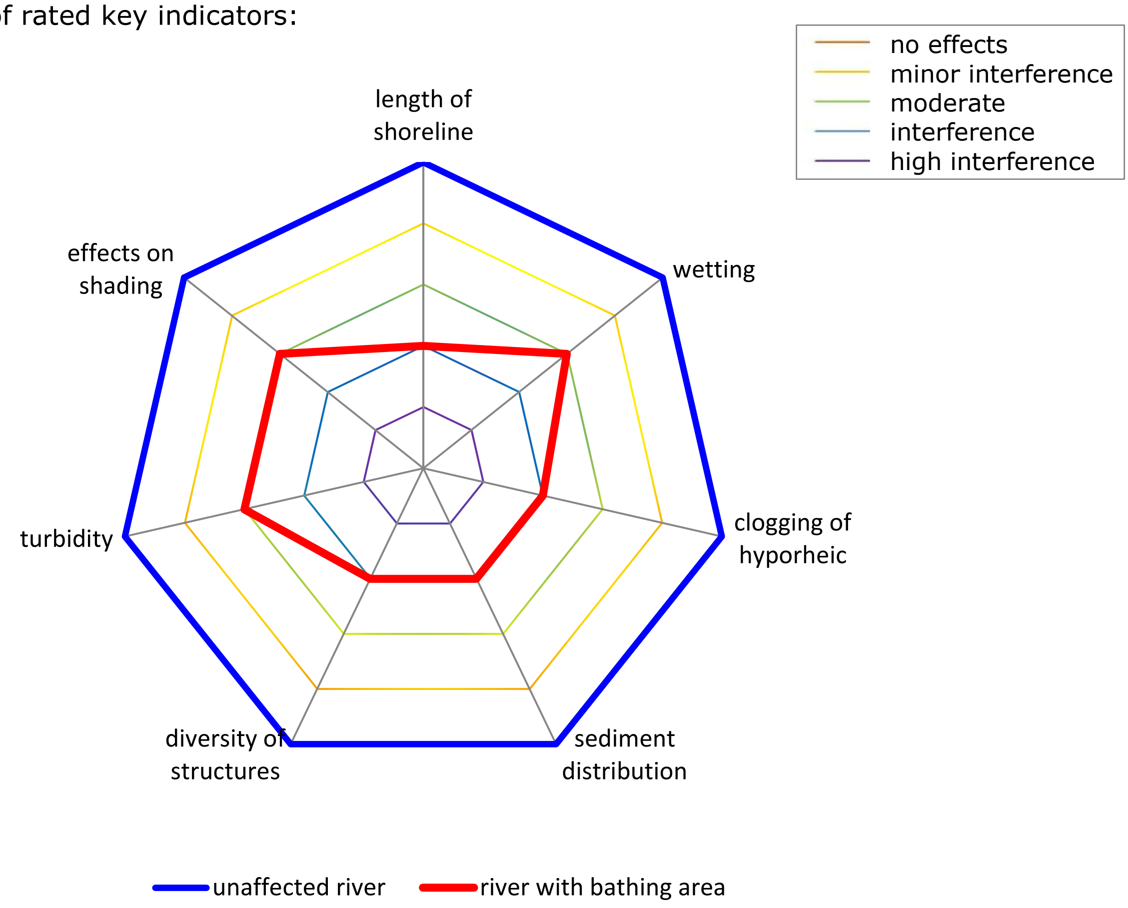
Hydrological indicators	
Runoff/discharge	Change of flow regime: ☐ no ☒ yes Mean annual discharge [m³/s]: ☐ 1 to 20 ☐ > 20 to 100 ☐ > 100 to 500 ☒ > 500

Hydromorphological indicators	
Fluvial landform	Length of shoreline ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Wetting	Wetted surface / wetted area ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference
Sediment	Clogging of hyporheic sediments ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference Spatial distribution ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference
Structures	Diversity of structures ☐ no effects ☐ minor interference ☐ moderate interference ☒ high interference

Material and physical emissions from punctual or diffuse sources	
Turbidity	Effect on water turbidity ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Others	
Solar radiation	Effects on shading ☐ no effects ☐ minor interference ☒ moderate interference ☐ high interference

Chart of rated key indicators:



Required measures

To enhance the present status, the following measures are recommended to improve the current situation:

- ☐ Flow regime
- ☐ Length of shoreline
- ☐ Wetting
- ☒ Clogging of hyporheic sediments
- ☒ Sediment distribution
- ☒ Diversity of structures
- ☐ Turbidity
- ☐ Effects on shading
- ☐ Other

Expert summary statement

In the considered section riparian forests and side channels of braiding systems formerly covered large areas. Today, the use of the area has changed completely. Run-off areas turned into settlements, only small water areas have remained, and a great number of pioneer habitats have disappeared. Realignment of the current channel with its benchmark condition will have adverse impacts on the existing utilisation and is therefore incompatible with the public interest.

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4. Relationships between biological metrics / indices and hydromorphological pressures

4.1 Phytobenthos – benthic algae

It has been suggested that benthic algae are particularly prone to the impact of increased fine sediment loads (Jones et al. n press). As benthic algae are photosynthetic, they are dependent upon light; any increase in the turbidity of the water column caused by suspended fine sediment will reduce light availability and, hence, photosynthesis and biomass of benthic algae. However, the most profound effect of fine sediment is the smothering of substrata to which benthic algae attach by deposited material. The relatively unstable deposited fine sediments (compared with larger particles) are not suitable for the attachment of long-lived sedentary species. Hence, non-motile, and particularly chain-forming taxa, cannot establish easily, further pushing the assemblage towards single-celled and motile taxa. A shift in assemblage composition towards motile taxa can be seen even where larger particles are covered with a layer of fines (Dickman *et al.*, 2005). The lack of stability in patches where easily erodible fine sediments accumulate, either accreting or eroding dependent upon flow conditions, tends to result in reduced taxon richness and biomass compared to more stable patches (Biggs et al. 1998; Biggs & Smith 2002; Matthaei et al. 2003). When comparing across streams, those with stable bed sediments support a higher biomass of diatoms than those that have unstable beds, for example due to high amounts of deposited fine sediment (Iversen et al. 1991; Biggs 1995; Jowett & Biggs 1997; Biggs et al. 1999; Biggs & Smith 2002).

For motile diatoms, shading from deposited fine sediment may not present a substantial problem, as they can move through the deposited sediment to higher light intensities at the river bed surface (Harper 1976; Hay et al. 1993). As a consequence, there is a tendency for the diatom assemblage to become dominated by motile taxa where rates of deposition of fine sediments are high (Yamada & Nakamura 2002; Dickman et al. 2005), and there is the potential for total diatom biomass to compensate for losses due to the shading effects of deposited fine sediment.

As a consequence of the impact of fine sediments on benthic diatom assemblages it is not surprising that indices based on benthic diatom assemblage structure have been proposed. These comprise simply the relative abundance of motile species (e.g. Bahls 1993; Dickman et al. 2005). This measure is based on the fact that many raphid species are capable of migrating through deposited sediment to avoid its negative impacts.

Further negative effects of hydromorphology could be expected through both direct and indirect impacts on the substrate on which benthic algae grow. Reductions in flow velocity, for example caused by impoundments, would tend to reduce flow velocity and increase the deposition of fine sediment altering both bed substrate and the potential for planktonic algae to thrive. Direct modification of in-stream and marginal habitat has the potential to alter the substrate on which benthic algae grow.

Here we have tested the relationship between indices based on the benthic algal community, particularly benthic diatoms, and hydromorphological alteration. In general, these indices have been developed to asses eutrophication stress. We have tested whether any alteration in benthic algal community associated with hydromorphological alteration influences the relationship between these indices and nutrient stress. Furthermore, we have explored the relationship between the relative abundance of deposited fine sediment in the river bed and a key descriptor of benthic diatom communities, the percentage of motile taxa.

4.1.1 Methods

Using data collected during WISER, the impact of hydromorphological pressure on the relationships between indices based on phytobenthos and phosphorus concentration was investigated using ANCOVA. Twelve indices of phytobenthos were investigated, namely Descy, Watanabe, TDI, % planktonic taxa, IPS, IDAP, EPI-D, D-CH, IDP, LOBO, TID and % motile taxa. The influence of six hydromorphological alterations was investigated, namely channel modification, artificial embankment, impoundment, modification of instream habitat, modification of riparian vegetation and velocity increase. Sites were categorized according to the extent of hydromorphological alteration with multiple categories used to describe increasing severity of alteration. Where significant effects of hydromorphological alteration on the relationship between the index and $\log_{10}$ orthophosphate concentration were found, relationships were checked to establish if the results were trivial, i.e. data from modified sites were within the range of scatter of unmodified sites.

Using data collected during STAR, the relationship between deposited fine substrate on the bed (visual estimates of % composition of sand and silt (2 mm – 6 µm), clay (< 6 µm) and all fine sediment (total < 2 mm)) and the % motile taxa was investigated using regression. The relationship between % motile taxa and water chemistry variables was also investigated. Where significant relationships were detected with bed composition analysis was repeated with all sites with zero fine substrate excluded to determine if the results were trivial, i.e. the influence of zero recorded fines was driving the relationship.

All statistical analyses were undertaken in SAS.

4.1.2 Results

There was a significant relationship with $\log_{10}$ orthophosphate for almost all indices. However, no significant effect of any hydromorphological alteration on this relationship was evident for any of the indices of phytobenthos tested. Whilst indices developed to detect the impact of nutrient pollution on phytobenthos should be robust to hydromorphological alteration, the result was somewhat surprising when considering the impacts of alterations such as impoundments (Figure 4.1), channel modification (Figure 4.2) and in-stream habitat modification (Figure 4.3) on the proportion of planktonic and motile taxa in the community.

Only weak relationships were found between the % motile taxa and the % fine sediment in the substrate, and these were driven by sites where zero fines had been recorded (Figre 4.4a-c). However, the relationships between % motile taxa and water chemistry showed a strong response to conductivity and phosphate concentration (Figure 4.4d-f).

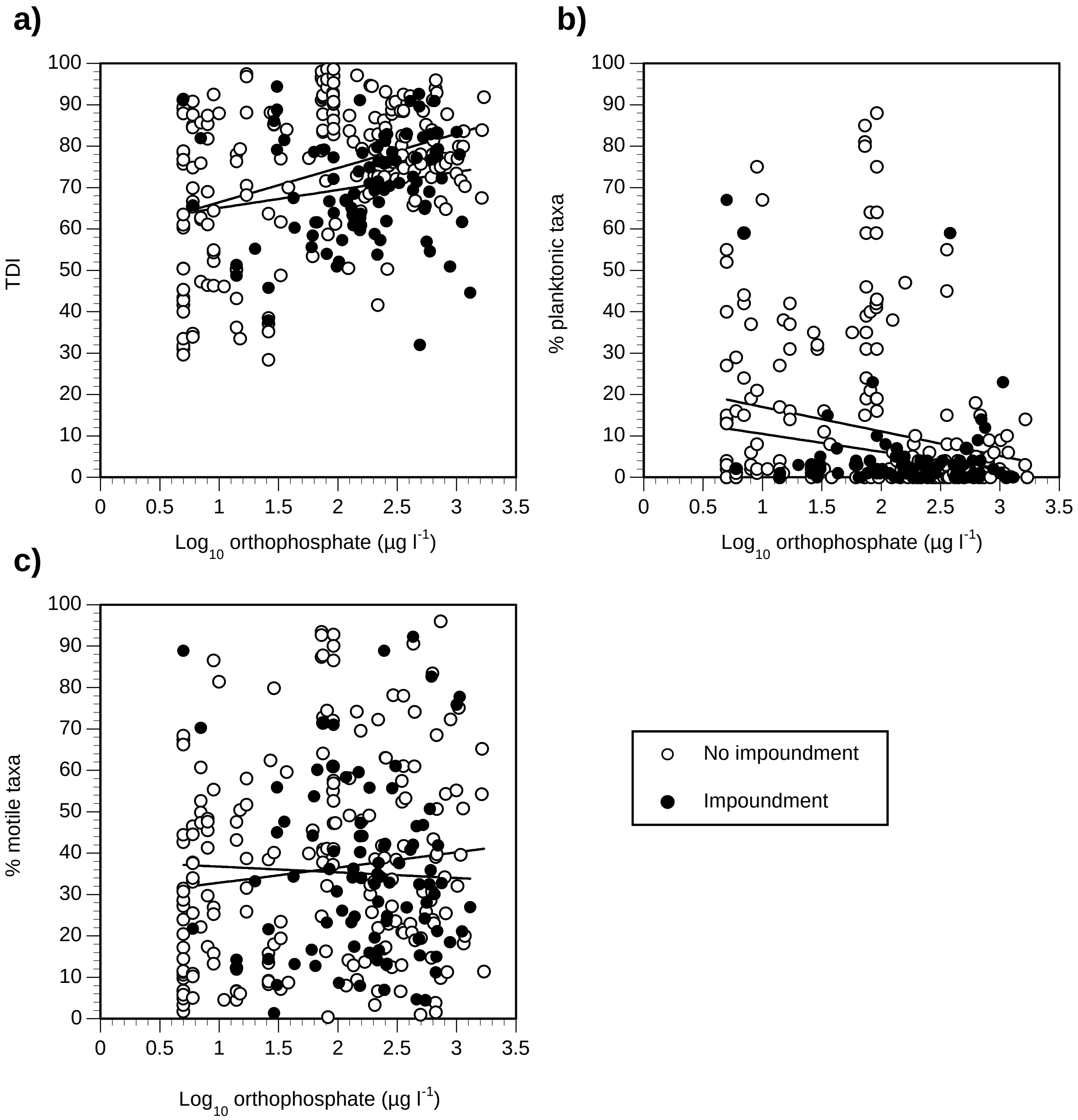

Figure 4.1. Influence of impoundments on the relationship between log_{10} orthophosphate concentration and a) TDI, b) % planktonic taxa and c) % motile taxa.

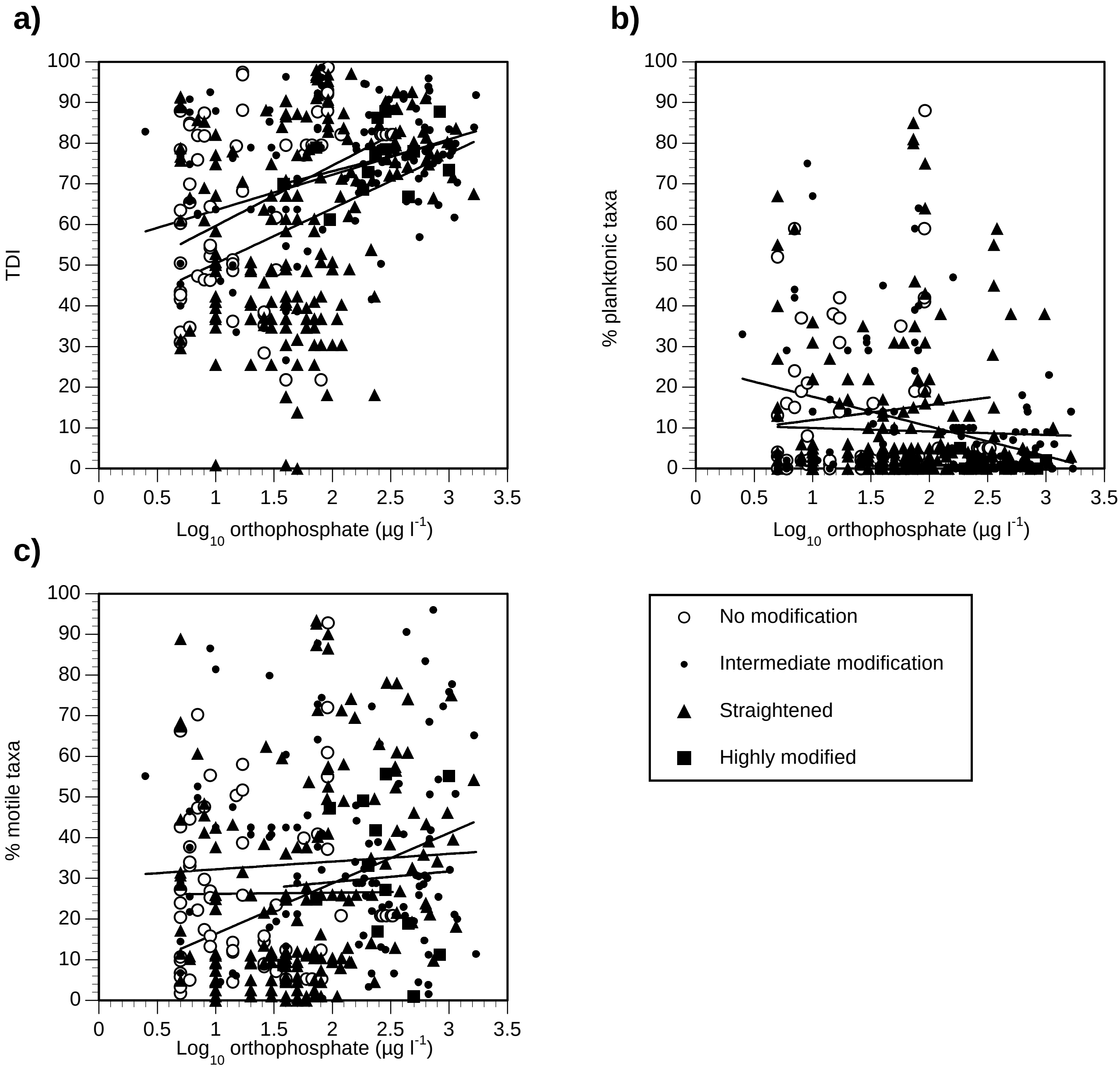

Figure 4.2. Influence of channel modification on the relationship between $\log_{10}$ orthophosphate concentration and a) TDI, b) % planktonic taxa and c) % motile taxa.

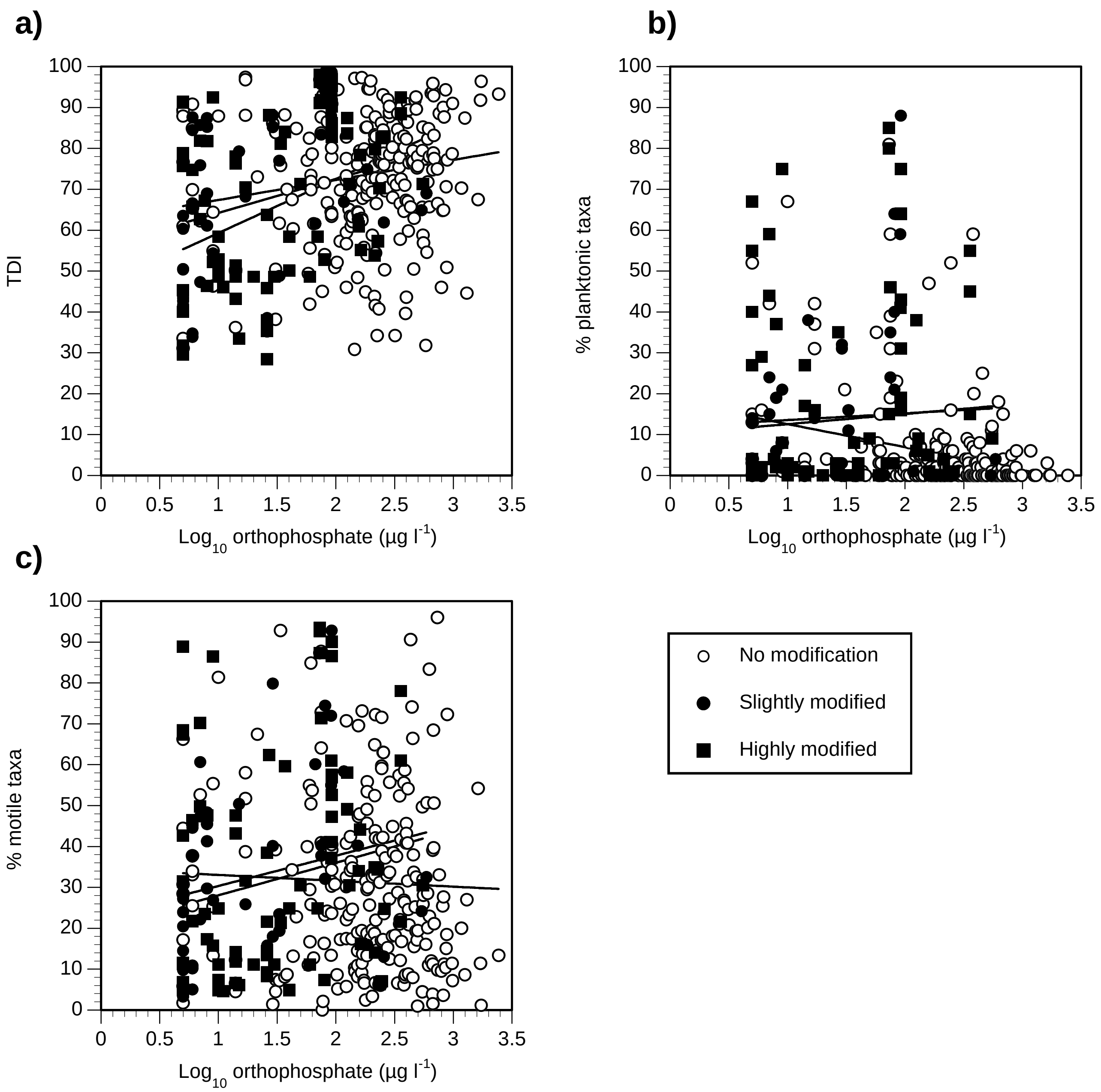

Figure 4.3. Influence of in-stream habitat modification on the relationship between $\log_{10}$ orthophosphate concentration and a) TDI, b) % planktonic taxa and c) % motile taxa.

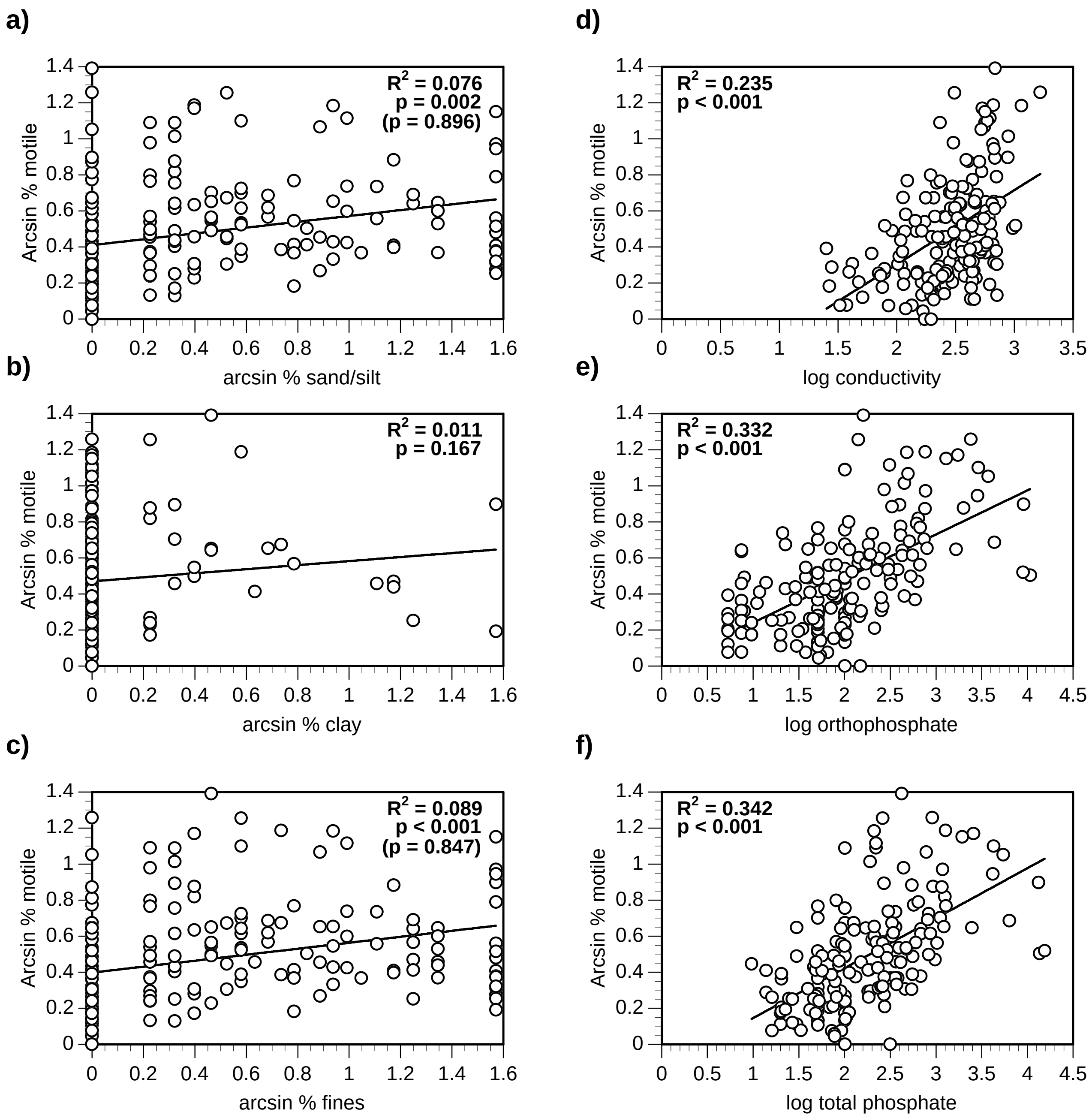

Figure 4.4. Relationships between the relative abundance of motile diatom taxa and measures of deposited fine sediment and water chemistry. a) % sand and silt (6 µm - 2 mm), b) % silt and clay (< 6 µm), c) % fine sediment (sand, silt and clay), d) conductivity, e) orthophosphate (µg l^{-1}), and f) total phosphate (µg l^{-1}). R^2 and p shown, where p is in brackets zero values for bed composition have been excluded.

4.1.3 Conclusion

It was not possible to detect any effect of the hydromorphological alterations tested, which included alterations that influence flow velocity, the rate of sedimentation and in-stream habitat, on indices based on phytobenthos. Although it is reassuring that indices developed to assess eutrophication stress (e.g. TDI, IPS and related indices) appear robust to hydromorphological alteration. This result is somewhat surprising as it was assumed that general descriptors of phytobenthos such as % planktonic taxa and % motile taxa would respond to hydromorphological alterations. Indeed, substrate is thought to have a substantial influence on benthic algal community composition (Stevenson & Pan, 1999, Stevenson *et al.*, 2008), and % motile taxa has been proposed as an index of deposited fine sediment (Bahls, 1993), one of the key aspects associated with hydromorphological alteration. Furthermore, % motile taxa should be used when interpreting indices such as TDI (Kelly et al. 2001).

It is possible that the categorisations of hydromorphological alteration did not adequately describe the extent of alteration, resulting in the negative result. However, the STAR data indicated that % motile taxa is not related to visual estimates of the percentage fine sediment in the bed substrate. Rather, % motile taxa appears to be related to nutrient conditions. This could be a consequence of competition for light between algal species favouring those taxa that can migrate to the top of the layer of benthic algae when nutrients are abundant, or simply that species with these characteristics (small, rapidly growing, motile) are indicative of high nutrient conditions (Kelly et al. 2001). There is a concern that if fine sediment does influence the % motile taxa, as other researchers have described through experimental manipulation, indices of eutrophication stress will be confounded by change in the % motile taxa. We suggest that more complex models of diatom assemblage response, such as classification and regression trees (CART) and RIVPACS (River InVertebrate Prediction And Classification System) models, are likely to be more capable of interpreting the impact of stressors on diatom assemblages than simple indices (Cao et al. 2007, Feio et al. 2007).

To conclude, using these existing data it was not possible to detect any effect of hydromorphological alteration on phytobenthos. Furthermore, it was not possible to demonstrate an effective response of the proposed index of fine sediment stress based on phytobenthos; % motile taxa appears to be related more to nutrient availability.

4.1.4 References

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4.2 *The potential for macrophyte metrics sensitive to hydromorphological pressures*

4.2.1 Introduction

Aquatic macrophytes are sensitive to physical alterations in rivers and streams. Here, the potential of macrophyte metrics to indicate physical alteration is assessed. The European Union (EU) requires member states to categorise the quality of their rivers, primarily using aquatic organisms (European Commission, 2000). Macrophytes are included on the list of organisms with other phytobenthos. Alterations to a river, including physical alteration that degrades the biota and causes a site to be categorised as impacted, must be mitigated against. The need to restore systems is now pressing as the implementation of the Water Framework Directive (WFD) moves into a new phase of restoration after the initial assessment of system quality. This new phase places more emphasis on metrics which are useful for diagnosing interactions at an individual site level and hence facilitating remediation, than previously when the emphasis was on metrics which were useful for providing overviews of impacts at a national level.

The underlying aim of the WFD is to manage aquatic systems using measures of ecosystem health to assess success (Pollard & Huxham 1998). The inclusion of hydromorphology in the assessment of ecological status is significant. In the past, monitoring in running water has focused on chemical parameters and benthic invertebrates. The WFD now widens that focus and implicitly requires that habitats are linked to biota, including macrophytes to physical habitat quality (Logan & Furse 2002). There is therefore a clear management need to appraise the sensitivity of European macrophytes to physical habitat alteration.

Man's alterations to rivers through impoundments, realignment of channels, and in-stream engineering works alter depth, velocity, substrate type, flow types and flow variability (Petts 1984). These variables define the physical niches in rivers and macrophytes have known preferences for these variables (Haslam 1978; Fox 1992), in fact, the assemblage structure of macrophytes has been show to follow the physical style of a river in the UK (Gurnell et al. 2010). Hence, we have a reasonable mechanistic understanding of how physical alterations may influence vegetation. Since historic times macrophytes have been grouped by depth preference as emergent, marginal and submerged (Sculthorpe 1967). In recent times, niche separation and range preferences for other physical variables have been demonstrated for many macrophytes (Westlake 1975; Chambers et al. 1991; French & Chambers 1996; Dawson et al. 1999a). It is therefore not surprising that studies of physically altered rivers show impacts on macrophyte community structure. Following impoundment and channelisation changes include loss of species, altered species dominance and relative abundance (Petts 1984b; Baattrup-Pedersen & Riis 1999).

Despite our knowledge of hydromorphological impacts on vegetation there has been little research carried out on the use of macrophytes as indicators of hydromorphological pressures in the context of the WFD, although there have been a number of studies linking hydromorphological pressures to aquatic vegetation directly (see supplementary material for references). There is therefore a clear gap in the literature.

As a first step in the evaluation of the potential use of macrophytes, it is important to describe the basic relationship between plants and river styles as this relationship is specific and likely to mediate the response of vegetation to different pressures, for example bryophytes in headwaters will respond differently to channelization than emergent vegetation in lowland rivers. In datasets which contain a wide range river styles, it is possible that any differences in plant assemblage between impacted and reference sites may not be transparent in multivariate analysis as the differences are swamped by differences in river style. This would suggest that a reference based approach, originally adopted for invertebrates, would be necessary. In such an approach an expected flora of a site is predicted and compared to the observed flora and presented as an Ecological Quality Ratio. Alternatively an approach could be taken where clearly defined, river types with defined reference communities is developed to which an observed flora is compared.

In addition to using species assemblage-based metrics, as is common for benthic invertebrates, both summary community metrics (evenness, total abundance and species richness) and trait-based metrics are considered. Trait-based approaches have been used successfully with macrophytes and other vegetation groups (Ali et al. 1999). Traits describe an attribute of the vegetation which transcends taxonomic groups and may therefore avoid some difficulties in using assemblage-based metrics. While benthic invertebrate metrics have been based on assemblage structure and summarise the shift in assemblage as sensitive taxa are lost and insensitive taxa increase in dominance. To detect and represent a broad gradient of response therefore requires a relatively broad assemblage of taxa. Typically, however, for in-stream macrophytes recordings by some standard WFD protocols (MTR) produce a rather small number of species, and for a given set of river conditions the species present can vary although their traits may be similar. It has been observed that while species may not occur consistently, morpho-groups (which can be viewed as simple trait groups) are more reliable.

It has often been stated that hydromorphology is a mix of many pressures, the term itself is an uneasy amalgam of hydrology and fluvial geomorphology. Each major pressure will require a diagnostic metric which can be used by managers to improve the condition of a site. Furthermore, a single combined metric may be required which is used for reporting purposes. Here, we focus on investigating the potential for a macrophyte metric generally sensitive to hydromorphological pressure. Within the datasets examined, resectioning is the most likely alteration, although weed cutting and dredging occur too. Where data (UK dataset) was available on specific alterations, resectioning is examined. Resectioning involves changing a channel cross section from its natural form to a standard trapezoid form designed to maximise channel conveyance capacity. Typically, it is associated with straightening and channelization and creates homogenous in-stream conditions where pools become infilled (Gurnell, *pers comm)*.

Here, we take two complementary approaches to develop useful riverine vegetation metrics which indicate hydromorphological pressure. We analyse large monitoring datasets (the intercalibration dataset focusing on Northern Europe, UK data and North Rhine-Westphalia, Germany) using multivariate statistics to associate plant traits with different natural states of hydromorphology and test the ability of trait groups to indicate hydromorphological pressures. The hydromorphological pressures represented in the monitoring datasets are predominately related to channelization. For brevity, only an outline of the methods is presented together with key findings from the results. Full methods and the complete results are provided as supplementary material.

Specific hypotheses

1. There are broad associations between macrophyte assemblage structure and rivers differing in geomorphological type.

2. Sites identified as hydromorphological degraded are physically distinct from sites which are not degraded.
3. Hydromorphological degradation in terms of an altered channel morphology alters macrophyte trait characteristics or species composition.

4.2.2 Methods

Only a brief synopsis of the methods is given here in the first parts of the sections, full details are provided in supplementary material in the end of the deliverable (from p. 233).

Data

Data were available from three sources: a large UK dataset, a German dataset from North Rhine-Westphalia and a dataset covering two river types assembled for the intercalibration exercise.

Intercalibration dataset

A total of 772 stream sites were included in the data analysis, all being part of the IC dataset (Birk and Wilby 2011). The stream sites were all situated in the Central Baltic area with sites in Germany (DE), Denmark (DK), Belgium, Flandern (BE (FL)), France (FR), Great Britain (UK (GB)), Northern Ireland (UK (NI)), Ireland (IE), Italy (IT), Lithuania (LT), Latvia (LV), Netherlands (NL), Poland (PL), and Belgium, Wallonia (BE (WL)). Hydromorphological impact was recorded on a 3-point scale and the type of impacted was not specified.

UK dataset

A total of 467 sites were included in this dataset. The macrophyte abundance and site physical and chemical data were originally collected during surveys carried out by the Centre for Ecology and Hydrology (CEH) and the Environment Agency of England and Wales (EA) using the "Mean Trophic Rank" (MTR) macrophyte survey method (Dawson et al. 1999a).These data and associated physical parameters are described in. (Gurnell et al 2010).

German dataset

A total of 1136 sites were included in the dataset covering the whole topographical gradient of the federal state of North Rhine-Westphalia in Germany. Macrophyte sampling was conducted according to the German standard method (Schaumburg et al. 2005a,b). Here. a 100-m reach was surveyed for macrophytes by zigzagging through the river and walking along the riverbank in the summer months at low flow conditions. All macrophyte species were recorded and identified to the species level. The surveys included all submerged, free-floating, amphibious and emergent angiosperms, liverworts and mosses. The abundance of each species was recorded according to the 5-point scale devised by Kohler (1978): 1 = very rare, 2 = rare, 3 = common, 4 = frequent, 5 = abundant, predominant. Furthermore, the growth form for each species was recorded according to Den Hartog & Van der Velde (1988) and Wiegleb (1991). The growth forms comprise different plant species that realized the same or comparable phenotypical adaptations to the aquatic environment.

Trait data

Data on traits were extracted from the literature and online databases (Willby et al. 2000; Klotz et al. 2002; Kühn et al. 2004) for use with the intercalibration dataset, while the PLANTATT trait dataset was used with UK data.

Analyses

A complementary set of multivariate statistical approaches were used to examine the relationships between macrophytes and hydromorphological degradation. For the UK and German datasets, a PCA was created using physical parameters to test hypothesis 1; further PCAs were used to describe trait distribution and test hypotheses 2 and 3. For the intercalibration dataset, PCAs were used to analyse species by sites and species by traits tables with Co-Inertia Analysis used to link the tables. Trait / Species groupings identified using cluster analysis were further tested for preferences in hydromorphological impact using parameter statistics.

4.2.3 Results

Are there broad associations between macrophyte assemblage structure and rivers differing in geomorphological type?

Yes, our results based on German data confirmed previous findings for the UK data that there are broad scale shifts in the dominance of vegetation across river types (Figure 4.5). Upland rivers on steep slopes tend to be dominated by mosses and liverworts, middle reaches support a range of taxa although submerged species often dominate, while lowland reaches are often characterised by emergent vegetation.

With the intercalibration dataset it was possible to examine detailed differences in macrophyte community characteristics between small (IC type RC1) and medium-sized (IC type RC4) lowland streams. We found that community trait characteristics differed significantly between small (RC1) and middle-sized (RC4) lowland streams (Adonis: Permutational Multivariate Analysis of Variance Using Distance Matrices; $p<0.001$). Specifically it was found that trait based groups (1-3) characterised as floating species or submerged species with single or multiple apical meristems were generally more widely distributed in small as compared to middle-sized streams, while groups (4 & 5), characterised by either submerged species or homophyllus amphibious species, were more widely distributed in middle-sized streams.

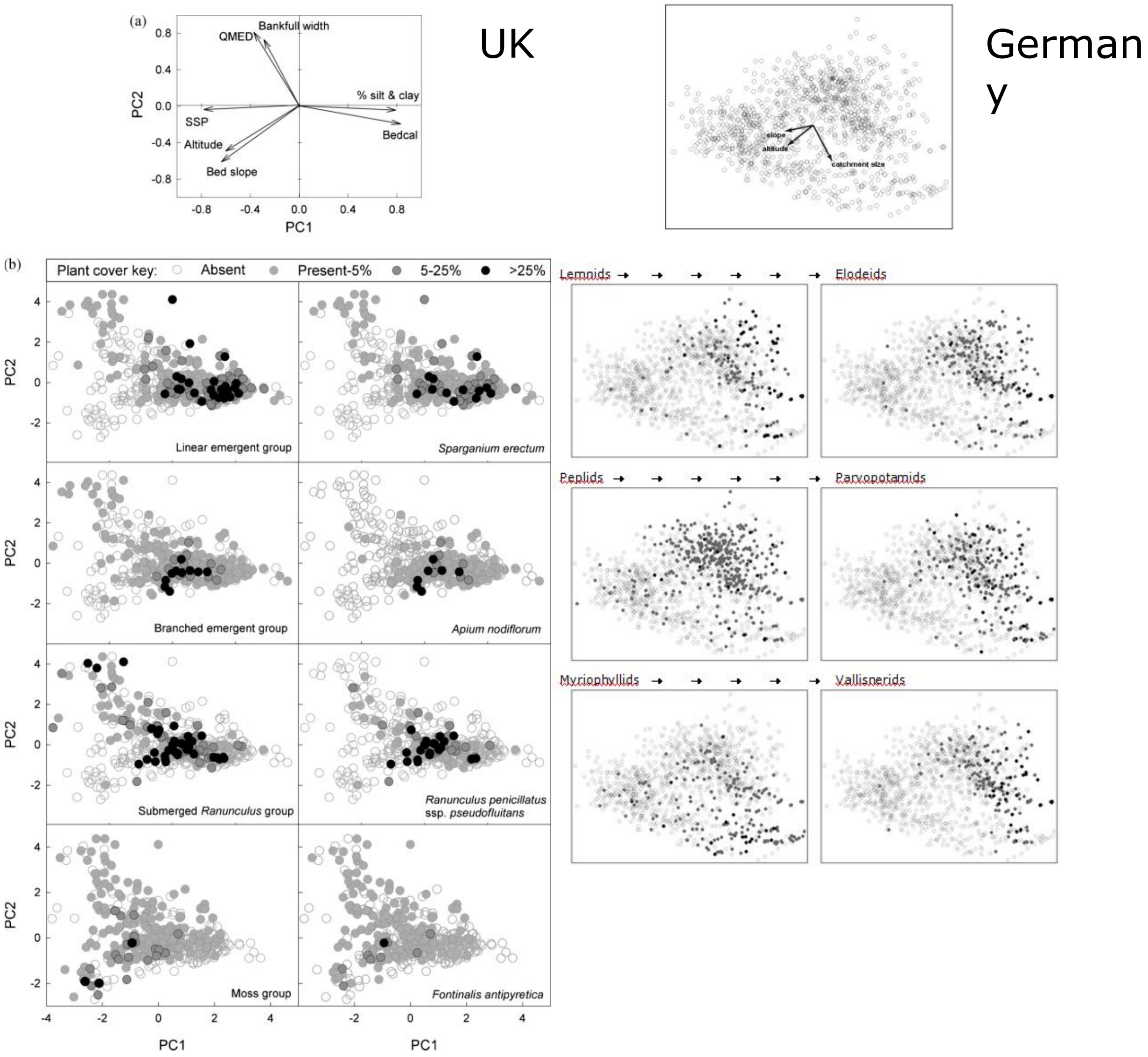

Figure 4.5. PCA plots of sites for the UK dataset (after Gurnell et al 2010) and German dataset demonstrating the distribution of key morpho-types of vegetation with physical parameters.

Are hydromorphological degraded sites physically distinct from sites which are not degraded?

Our results suggest that there is an issue of scale when it comes to detecting differences in the physical character of degraded versus non degraded sites. With the large datasets from the UK and Germany covering a range of river types it was not possible to distinguish between degraded and non-degraded sites using physical variables routinely collected during monitoring exercises.

What was clear from the UK data was that resectioning in particular was associated with lowland sites. Amongst lowland sites, neither substrate type, channel slope, depth nor width allowed resectioned and non-resectioned sites to be distinguished (Figure 4.6).

In the German dataset a morphological gradient is inherent to the dataset, but the morphological quality classes are distributed fairly evenly in the space of the abiotic variables slope, altitude and catchment size (Figure 4.7). This indicates firstly that anthropogenic alteration is widespread and was conducted independent of river size and topographical location; thus, abiotic landscape factors cannot be used for a morphological assessment of human influence. Further on this implies that the dataset has to be divided primarily into river types before a gradient analysis in terms of human influences on the flora can be applied.

It was not possible to do a comparable analysis on the intercalibration dataset as physical habitat variables were not collected.

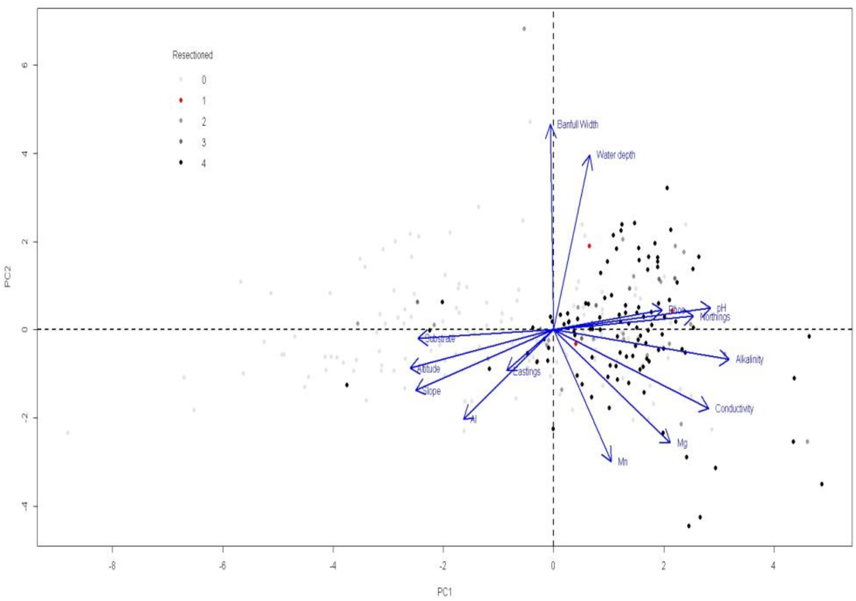

Figure 4.6. A PCA of the UK data demonstrating the distribution of resectioned sites from 0 no resectioning to 4 both banks resectioned. Resectioning is absent to the left of the diagram where sites are found at altitude and have steep slopes, upland sites. However, in the lowland sites, to the right of the diagram both non-resectioned and resectioned sites overlap, indicating similar physical characteristics in the parameters used in the ordination A PCA using only physical variables produces a similar pattern to this one which includes additional information on other stressors such as nutrients.

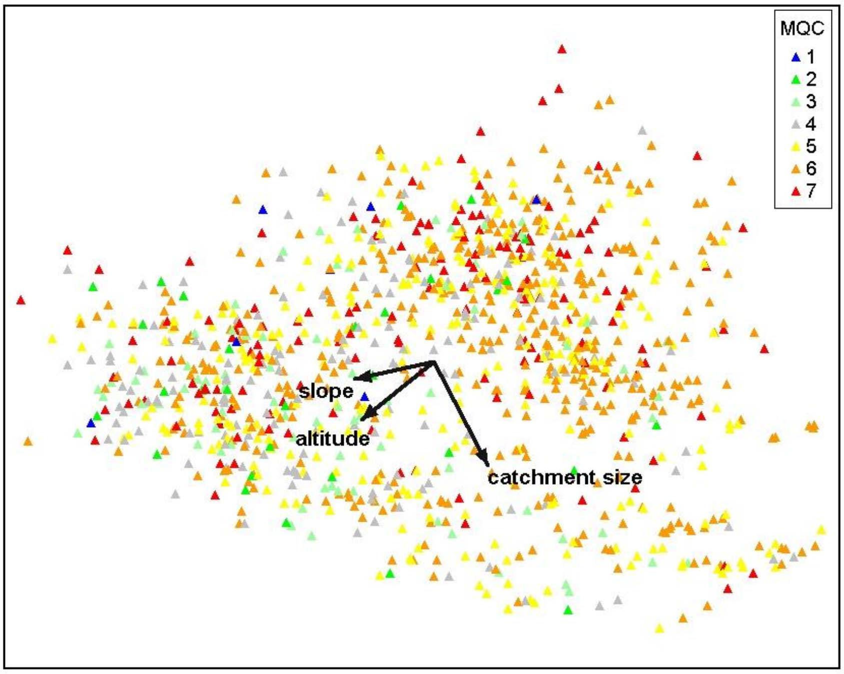

Figure 4.7. PCA of the German data on the three parameters: slope, altitude and catchment size, showing the distribution of morphological quality classes (MQC) between sites. The morphological quality gradient ranges from 1 = near-natural morphology to 7 = totally morphologically altered.

Does hydromorphological degradation in terms of an altered channel morphology alter macrophyte trait characteristics or species composition?

Analysis of the intercalibration dataset did reveal differences in plant traits across hydromorphological degradation gradients (Figure 4.8). We observed that trait characteristics changed significantly in response to hydromorphological degradation in small streams (Adonis: Permutational Multivariate Analysis of Variance Using Distance Matrices; $p<0.05$). The ecological preference of the macrophyte community changed in modified streams with an increase in the abundance of productive species as inferred from increasing weighted averages of Ellenberg N. At the same time we observed an increase in the abundance of free-floating species, whereas the abundance of submerged and amphibious species with heterophyllus leaves declined. We also found that the abundance of species growing from a single basal meristem decline, whereas species with a high overwintering capacity increased in abundance in degraded streams.

For the UK dataset there was no obvious differences in either species preferences or traits between resectioned and non-resectioned sites (Figure 4.9). Sites were ordered along the first PCA axis, of a species by sites plot, characterised as running from sites dominated by the emergent *Sparganium erectum* to sites dominated by the aquatic moss *Fontinalis antipyretica.* The main axis of the traits by sites plot was characterised by the Ellenberg N index which indicates system fertility. The large differences in community structure suggest that these assemblages may vary in their response to hydromorphological pressures.

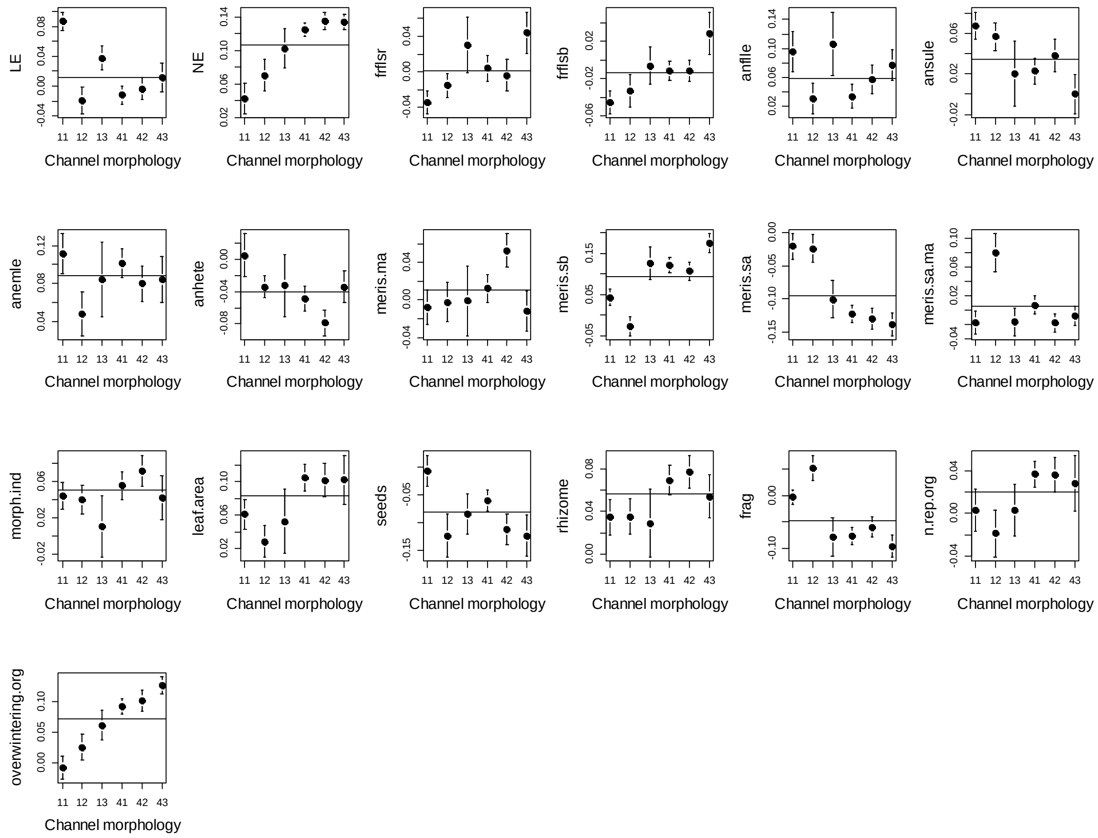

Figure 4.8. Mean trait values (community weighted means obtained through COIA) of unmodified (RC1 streams: 11; RC4 streams: 41), slightly modified (RCI streams: 12; RC4 streams: 42) and highly modified (RC1: 13; RC4: 43) stream sites in terms of channel morphology. Error bars indicate standard error. Horizontal line indicates the grand mean for all sites.

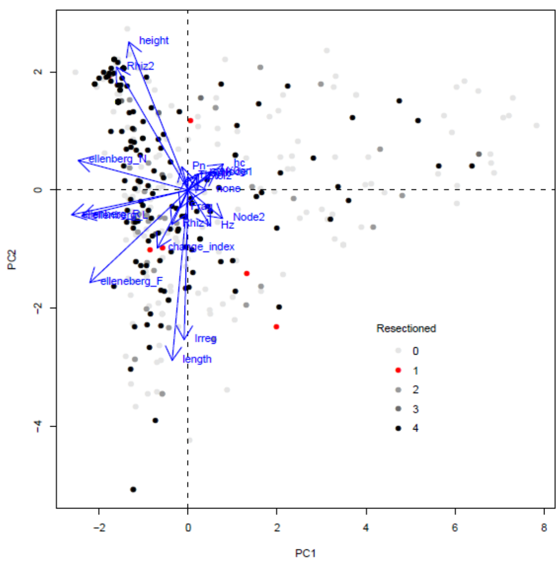

Figure 4.9. A PCA of the UK dataset with site averaged values for traits added as vectors. Traits could not be used to separate resectioned from non-resectioned sites.

4.2.4 Discussion

There are broad associations between macrophyte assemblage structure and rivers differing in geomorphological type.

Our findings confirm previous work associating river vegetation with different sections or physical styles of rivers. These results broadly agree with findings by Holmes et al. (1998) that coarse scale variation in macrophyte community type is associated with contrasting physical characteristics (slope, altitude and substrate size), while more subtle differences are related to geology type. The gradients represent a transition not only from low to high disturbance, but also from low to high ecological stress, as revealed by patterns of generally declining nutrient concentrations with increasing stream power and coarsening substrate. The association of different vegetation types with river style links directly with the approach adopted in WP2 of defining different river styles and linking them to different vegetation types. In WP2 vegetation is viewed as an active component in fluvial geomorphological processes where it can act as an ecosystem engineer to stabilise sediment and impede flow. Key to the successful outcome of REFORM is a need to strengthen the conceptual understanding of natural physical processes from WP2 with those of degraded systems analysed in WP3. Sites identified as hydromorphological degraded are physically distinct from sites which are not degraded.

We did not find differences in in-stream habitat using the variables measured. This suggests that the very broad differences in habitat type between rivers of different style have indeed overwhelmed the signal along gradients in impact. Previous studies have demonstrated clear differences in the physical condition of sites including homogeneity in in-stream conditions in terms of depth, substrate type and flow. Correct representation of in-stream conditions requires further consideration as some differences are obvious to the naked eye, but are not normally recorded using standard monitoring techniques, for example the steep angle of trapezoid banks where they meet the water is unnatural and makes poor habitat for marginal vegetation, but is not normally measured. Another issue is that sites identified as hydromorphologically altered may, in fact, have begun to renaturalise through the formation of side bars, thereby making in-stream conditions more heterogeneous. In the UK in particular such processes have been commonly observed in lowland rivers which are less intensively maintained than previously.

Hydromorphological degradation in terms of an altered channel morphology alters macrophyte trait characteristics or species composition. The most successful analysis of traits was within river styles where some traits showed clear differences between degraded and non-degraded sites. In small lowland streams, with increasing degradation the productivity level of the community increased and there was also a change in dominant life forms with an increased abundance of free-floating and floating leaved species. We also observed in small streams that species growing from single basal meristems declined and that species with a high overwintering capacity increased as observed. These results suggest plant strategy may determine the plants success in degraded systems and the productivity level of the community suggests growing conditions may be better and a possible link to eutrophication. It is possible these sites are subject to multi-stressors.

The geographic coverage of this analysis was limited to central and northern Europe by the available data. Additional data from the Mediterranean which is suitable for analysis will become available to the project and should be incorporated in future analyses for Deliverable D3.2.

The scope of this work was also limited by the lack of information or data on some key hydromorphological pressures. For those pressures, such as regulated rivers and dammed systems we carried out a literature review and tabulated the known vegetation responses, seethe following section in the supplementary material; A review of the impacts of hydromorphological pressures on potential macrophyte metrics. In all cases information on plant traits were not available but information did suggest changes in both vegetation

productivity and species composition were common. Riparian vegetation as well as in-stream vegetation were involved in the system response. The information collected suggested it is possible to develop a clear mechanistic understanding of how some hydromorphological pressures impact upon vegetation and allow diagnostic metrics to be developed. For example irrespective of river size vegetation development typically increases downstream of dams were discharge from the dam stabilises flows.
In summary it will be possible to develop useful indicators of system response to both general hydromorphological degradation and also to specific forms of degradation but if these are to be helpful to managers they must be developed with a clear conceptual understanding of cause-effect processes.

4.2.5 Conclusions

1. There is potential to use macrophyte trait-based metrics to indicate hydromorphological impacts.

2. As macrophyte assemblages change with physical river type, trait responses may be specific to individual river types. The typology under development in WP2 should be considered in future analyses.

3. The association between channelization and lowland sites suggests a strong spatial pattern, as lowland sites are known to be subject to multiple stressors future analysis should look at hydromorphological pressures in combination with other stressor gradients, including eutrophication gradients.

4. Questions have been raised regarding the suitability of current data collection methods, these should be addressed under Deliverable 3.3.

5. The relationship between vegetation and hydromorphological alterations is multi-faceted and there is a need for clear conceptual models describing cause-effect relationships. A priority for future WP3 work should be to fill this gap under Deliverable 3.2.

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4.3 Macroinvertebrates

4.3.1 Introduction

Macroinvertebrate metrics have been profusely developed in the past decades for biomonitoring purposes, including, more recently, metrics designed to respond to hydromorphological conditions (AQEM Consortium 2004), and they are generally used by EU Member States for the evaluation of ecological status of rivers (European Commission, 2012a).

Hydromorphological measures have been proposed for most of the river basin districts in the first planning cycle of the Water Framework Directive. However, there is great uncertainty as to how the measures can contribute to the achievement of the environmental objectives (European Commission 2012b), as knowledge remains limited of how macroinvertebrates and in general riverine biota respond to changes in hydromorphology (Vaughan et al. 2009; Friberg 2010; Lancaster & Downes 2010).

The main objective of this section is to estimate *a priori* the relationships between macroinvertebrate and individual and combined hydromorphological pressure indicators in multi-pressure environments from datasets derived from national monitoring programmes. The specific aims are to quantify the relationships: (1) between hydrological and hydromorphological pressures and conditions and (2) between both of these indicators and available macroinvertebrate metrics. The results obtained for macroinvertebrate metrics designed to respond to hydromorphological conditions and to general degradation or pollution are compared, and the performance of hydromorphological methods established at national level is evaluated to explore their potential function as a baseline for the development of new indicators of hydromorphological degradation in the future.

4.3.2 Strategy

The analyses are done using a 3-tiered approach in which a 1) comprehensive national (Denmark) dataset was analysed on its own; 2) a smaller national (Denmark) dataset with hydrological time-series was analysed to single out the effects of hydrology and 3) three national datasets (Denmark, Spain and UK) were analysed in a similar manner. In the following, the analysis, results and discussion of 1) and 2) is presented first as one entity followed by 3) an entity that uses a slightly different approach. Therefore, certain elements of repetition can occur throughout the text.

4.3.3 Large scale and hydrological dataset (Denmark)

Large scale dataset

The data were compiled in 2004-2005 from 219 Danish streams sites distributed almost evenly within the country. For each site the data only represent one year's sampling. Data were obtained according to the Danish National Monitoring Programme for Nature and the Aquatic Environment (NOVANA). The studied sites represent 1-5 order streams, mainly located in agricultural areas that make up 67% of the total area of the country. The stream sites had a pre-defined typology according to land use and soil characteristics: REF – reference sites (83); PHY – highly physically modified sites (14); AG1 – >50% of catchments with agricultural activities, sandy/loamy sand, hilly topography, intense agricultural activity in the riparian areas (22); AG2 - >50% of catchments with agricultural activities, variable soil type, topography and agricultural activity in the riparian areas (36); AG3 - >50% of catchments dominated by agriculture, variable soil type and topography, and no or minor agricultural activities in the riparian areas (26); PS1 - >50% of catchments dominated by agricultural activities and significant contribution by waste water from scattered dwellings (13); PS2 – significant contribution of waste water from point sources (25). Among these, the REF sites are not true reference sites, but represented the least impacted according to their macroinvertebrate assemblages.

Macroinvertebrates were sampled using a standardised handnet (25x25 cm handnet frame length) with 500 µm mesh size. At each site kicking was carried out at three transects at 10%, 50%, 75% and 100% of the distance between the stream banks (Skriver et al. 2000). The resulting 12 subsamples were pooled into one sample. All samples were preserved in 96% ethanol and sorted and analysed in the laboratory (Skriver et al., 2000). The level of identification was species or genus for most groups; however, for some groups family or higher levels were applied. Sampling and sample processing were carried out by staff at the regional environmental institutions or by private consultants. Rare taxa occurring in less than 10% of the samples were excluded from the analyses.

We calculated more than 200 different metrics and indices using the ASTERICS program version 3.3 (http://www.fliessgewaesser-bewertung.de/en/download/berechnung/). Before importing the data to ASTERICS, we used the taxa and autecology database for freshwater organisms version 5.0 (www.freshwaterecology.info; Schmidt-Kloiber and Hering 2012) to exclude macroinvertebrate ID numbers and international taxa names. In addition to assessing hydromorphological stress, we calculated the Estonian MESH index (Timm et al. 2011), a similar German index (Schmedtje and Colling 1996), reflecting the flow sensitivity of stream macroinvertebrates, and the British LIFE index (Extence et al. 1999). The verbal categories of rheophility of the German index were transformed into six categorical classes. In addition, biological traits such as % crawlers, % interstitial taxa, and % temporarily attached taxa were designated for the occurring taxa using Tachet et al. (2010). Biotic metrics and species traits were assigned to four different groups: composition and abundance metrics (1), richness and diversity metrics (10), sensitive and tolerance metrics (18), and functional metrics (10). We followed the concept of Feld and Hering (2007) on metrics classification.

To exclude redundant macroinvertebrate metrics, we used Spearman Rank correlation analysis. If two metrics had a correlation coefficient (r) higher than 0.8, the one with the highest mean correlation was excluded (Feld and Hering 2007). The procedure was similar for the environmental parameters for which the threshold r value was 0.7, however. Proportional macroinvertebrate metrics and environmental parameters were arcsin(sqrt(x/100))0.5 transformed (Podani 2000). Log-transformation was used for other metrics and environmental parameters that did not meet the requirements of normality. Species abundances were not log transformed prior to analysis since CCA allows log-transformation during analysis.

Data on catchment area (land use), riparian areas (land use), physical habitat quality and hydrochemistry (BOD5, NH4-N, PO4-P, alkalinity, pH, Fe2+) were gathered following technical guidelines prepared especially for the programme. Chemical analyses were all carried out at accredited laboratories, whereas field sampling activities were undertaken and other data were provided by staff at the regional environmental institutions.

Hydrological dataset

The data were collected according to the National Monitoring Programme for Nature and the Aquatic Environment (NOVA) from 34 routine monitoring sites situated nearby flow gauging stations. The studied sites are small, unshaded and are distributed in small catchments (2-48%) affected by agriculture. The sampling was carried out in spring (February-April) and in summer (June-August) 1998, 2000 and 2003. In this study we only used summer data.

Macroinvertebrates were sampled in a similar manner as described under the large scale dataset and metric analyses were undertaken in the same way.

Biotic metrics and species traits were assigned in the following way for this dataset: composition and abundance metrics (14), richness and diversity metrics (20), sensitive and tolerance metrics (23) and functional metrics (33). We followed the conception of Feld and Hering (2007) on how to classify the metrics.

Each study site was located close to the flow gauging station, and the distance between gauge and site was usually 2000 m or less The flow dataset included maximum, minimum, Q10 (flow magnitude exceeded 10% of the time), Q50 (flow magnitude exceeded 50% of the time) and Q90 (flow magnitude exceeded 90% of the time) were calculated. For more information on the calculation of flow statistics see Dunbar et al. (2010).

Other variables were the same as in spatial dataset described under the large scale dataset.

Statistical analyses

We used canonical ordination to identify environmental gradients and to study their relationship with macroinvertebrate species and metrics. Detrended Correspondence Analysis (DCA) was used to test if the response model should be unimodal (Canonical Correspondence Analysis; CCA) or linear (Redundancy Analysis; RDA). The rule of Jongman et al. (1995) says that if the length of the ordination axes is less than 2 s.d., the data have a linear relationship and RDA should be used; if the length of the ordination axes is longer than 4 s.d, the data with strong non-linear relationship and CCA should be used. Accordingly, CCA was used for species data and RDA for metrics data.

A total of 34 environmental variables were tested using forward selection to qualify for inclusion in the model, involving running of a Monte Carlo permutation test with 999 unrestricted permutations, with a significance level α of 0.05.

Large scale dataset

Environmental variables were divided into two separate groups: physical data and chemical data. We used variance partitioning to test whether physical and chemical data were redundant with each other or whether they explained unique aspects of species composition or metrics. Variance partitioning was done by examining the relationship between species (CCA) or metrics data (RDA) with the environmental data including both physical and chemical variables. Next, the same analysis was run using physical data as environmental data and chemical parameters as covariables. In the third step, chemical data were used as environmental data and physical data were included as covariables. Intersection of the two datasets was examined by subtracting the variation explained by physical data (chemical data as covariable) and chemical data (physical as covariable) from the physical and chemical data combined.

During step two, we examined the relationship between species or metrics data and environmental data (physical and chemical data combined), excluding the variation explained by stream typology (REF, FYS, AG1, AG2, AG3, PS1 and PS2 used as covariables). Since the data are collected all over the country, we separated the study sites relative to bottom substrate into sandy bottom (eastern Denmark) and lime (western Denmark), and we included binomial distribution as covariable. We used SIMPER analysis of Bray-Curtis dissimilarity to identify the species responsible for the observed differences between different stream typologies. Prior to analyses, species abundances were transformed using Hellingers transformation.

Ordination was done in CANOCO for Windows version 4.5 (ter Braak and Smilauer 1998), and SIMPER analyses were conducted in PAST version 2.14 (Hammer et al. 2001).

Hydrological dataset

The analytical framework was the same as for the large scale dataset, the only difference being that the effect of year was removed. We used partial constrained ordination to remove the effect of year (1998, 2000 and 2003) from the computations by means of multiple linear regression (ter Braak 1988). In the second part of the analyses we used eastern and western Denmark as a covariable to exclude the impact of soil type from the analyses.

<u>Results</u>

Large scale dataset

The proportion of variance explained by environmental variables was much higher for RDA (metrics) than for CCA (species). Environmental variables explained 20.2% and 4.8% of the variance in the metrics data for the physical and chemical data, respectively, compared to 19% and 4% of the taxa variance for the same dat set, respectively (Table 4.1; Figure 4.10).

Table 4.1. Results of multivariate regression models of taxa (CCA) and RDA.

	Taxa (CCA)	**Metrics (RDA)**
No. taxa/metrics in all analyses	64	41
No. stations	219	219
Total inertia (variance in the species/metric dataset)	2.9	1.00
Sum of all canonical eigenvalues		
Physical data (chemical data as covariable)	0.553	0.202
Chemical data (physical data as covariable)	0.149	0.048
Intersection of physical and chemical data	0.101	0.079
Cumulative % of species/metric-environment relationship of axes 1 and 2		
Physical data (chemical data as covariable)	51.5	92.7
Chemical data (physical data as covariable)	59.0	85.9

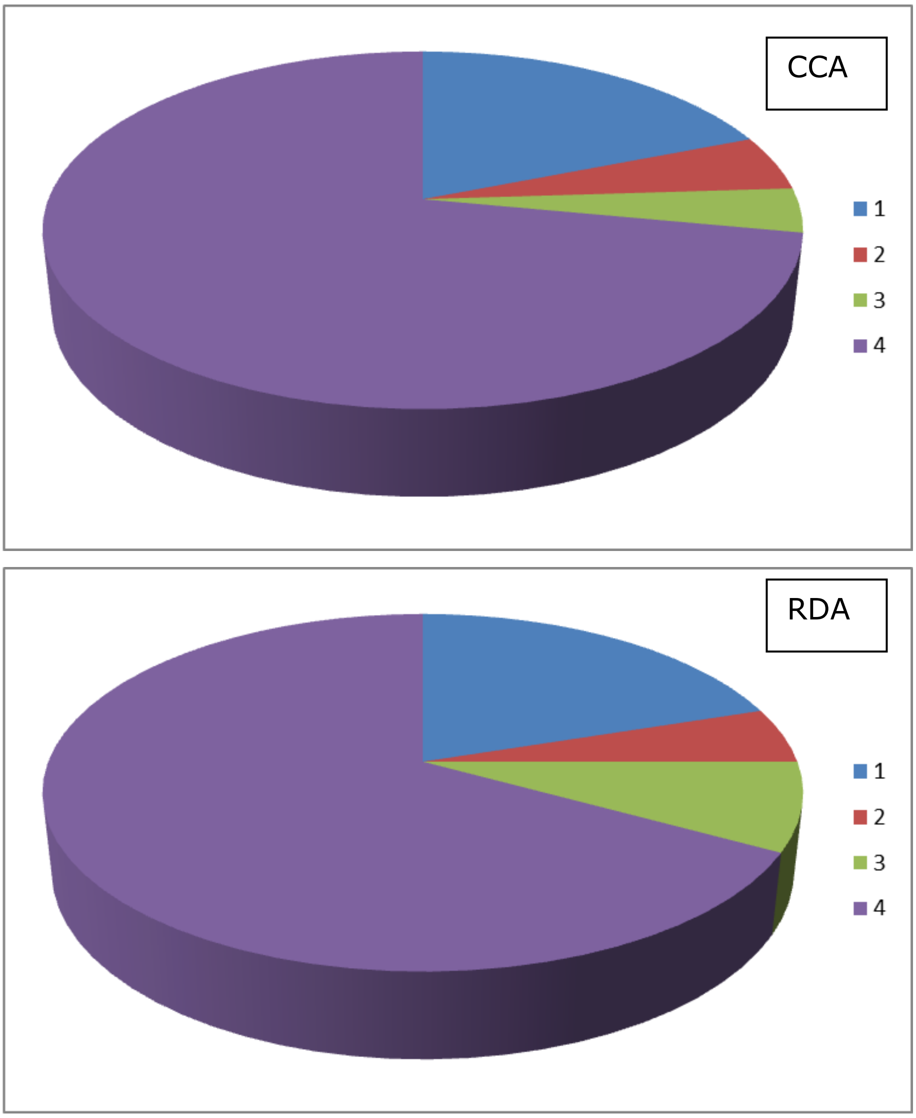


Figure 4.10. Variance partitioning of taxa (CCA) and metrics (RDA) datasets using macroinvertebrates. 1: Physical variables; chemistry as co-variable; 2: Chemical variables; physical variables as co-variable; 3: Intersection of physical and chemical variables; 4: Unexplained variability.

Variance partitioning in species data (CCA)

Twenty-three out of 34 environmental variables were significant in explaining the variation in species data (Figure 4.11). The first axis explained 8.5% of the total variation in the dataset. Combined, the two first axes explained 13.3% of the total variation in the dataset.

The sum of all constrained eigenvalues using 23 environmental parameters (both physical and chemical) and no covariables was 0.83. The total variance (inertia) in the dataset was 2.9, and less than one third (28.6%) of the variance in the species data could be explained by these explanatory variables. Physical components were clearly the most important factor, explaining 19% of total explained variability. Chemical factors explained 4%, and the variation explained by the intersection of both physical and chemical parameters explained 5% of total explained variability. The intersection of physical and chemical data variation was not redundant in explaining species composition, and each dataset (physical and chemical data separately) explained unique aspects of species composition.

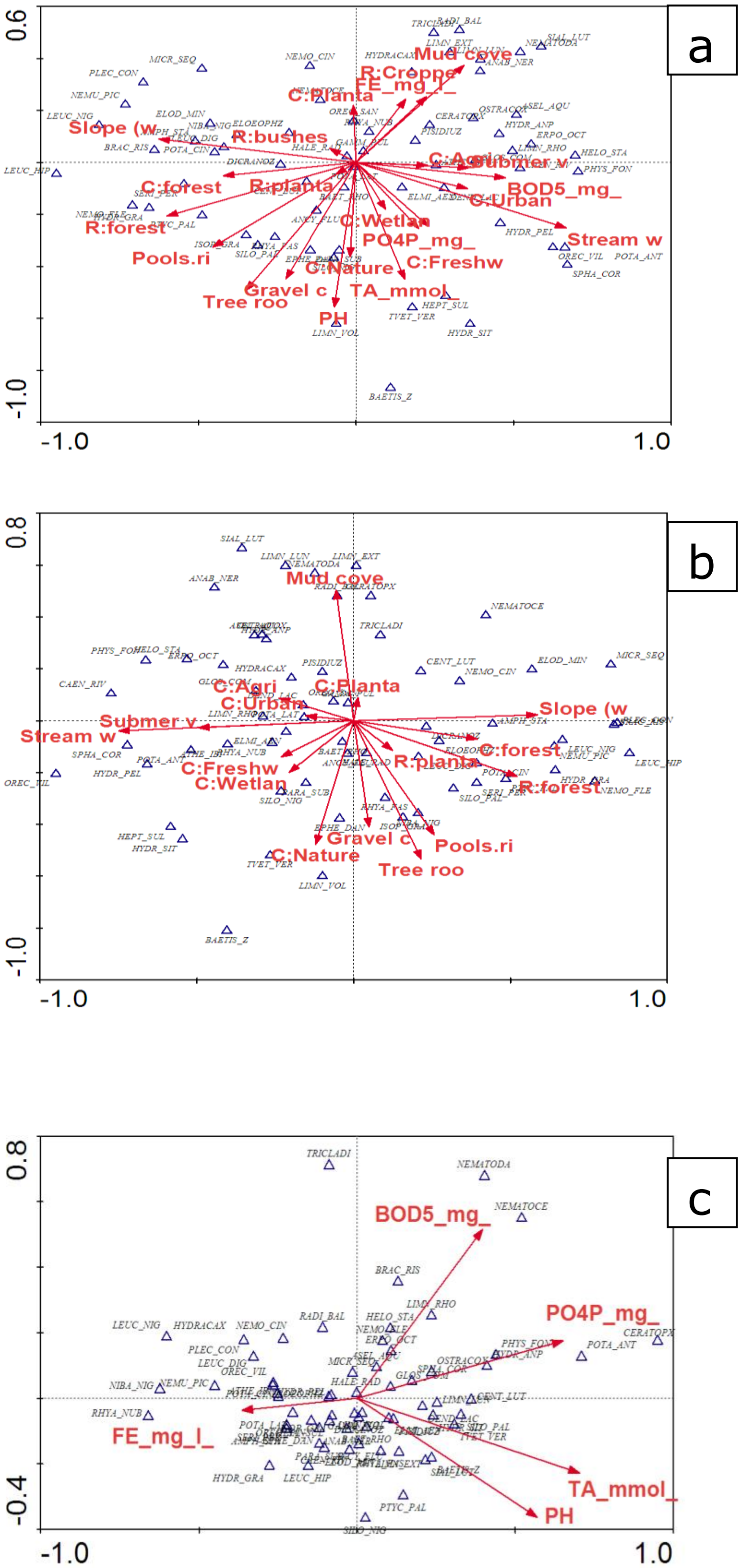


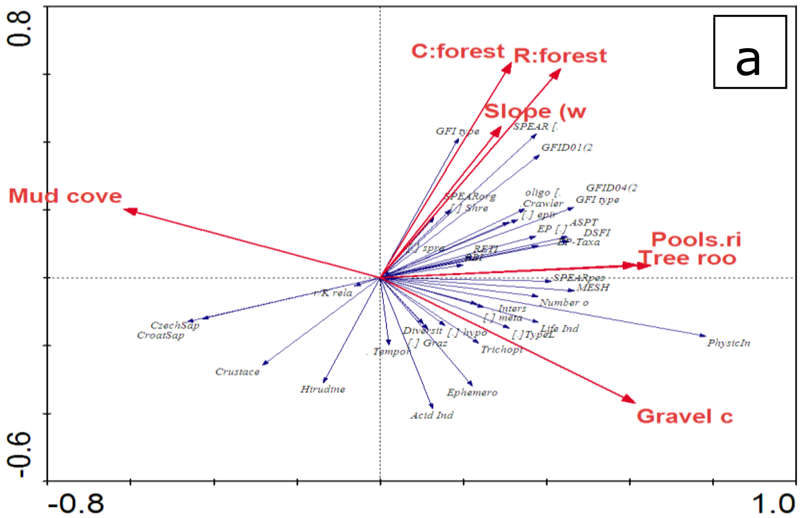
Figure 4.11. CCA of 64 species and 34 environmental parameters. a – both physical and chemical data without covariables; b –physical data as environmental data and chemical data

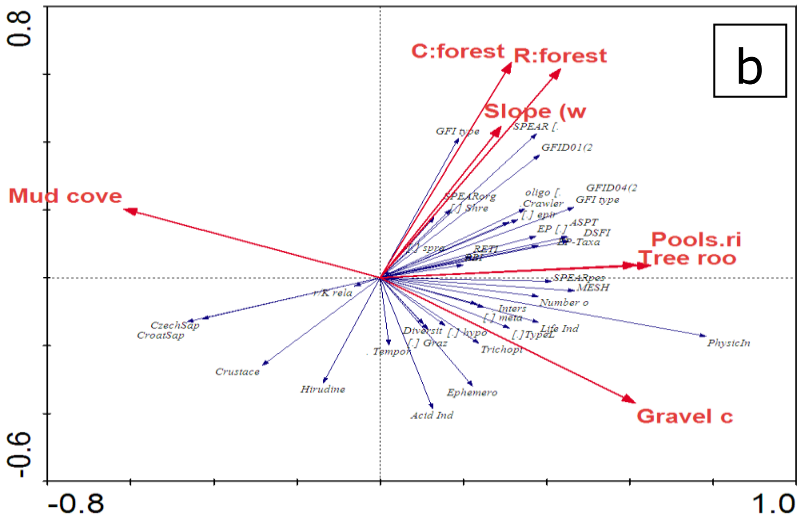
as covariables; c – chemical data as environmental data and physical data as covariables.

Variance partitioning in metrics (RDA)

Ten out of 41 environmental variables were significant in explaining the variation in metrics data (Figure 4.12). The first axis explained 23.5% of total variation in the dataset. Combined, and the two first axes explained 28.9% (1/3) of total variation in the dataset.

The sum of all constrained eigenvalues using 10 environmental parameters (both physical and chemical) and no covariables was 0.329. Total variance (inertia) in the dataset was 1.00, and more than one third (32.9%) of the variance in the species data could be explained by these explanatory variables. Again, physical components were clearly the most important factor, accounting for 20.2% of total explained variability. Chemical factors explained 4.8%, and the variation explained by the intersection of both physical and chemical parameters constituted 7.9% of total explained variability. The intersection of physical and chemical data variation was not redundant in explaining species composition, and each dataset (physical and chemical data separately) explained unique aspects of species composition. The analysis had higher explanatory power for metrics than for species (Figure 4.10).





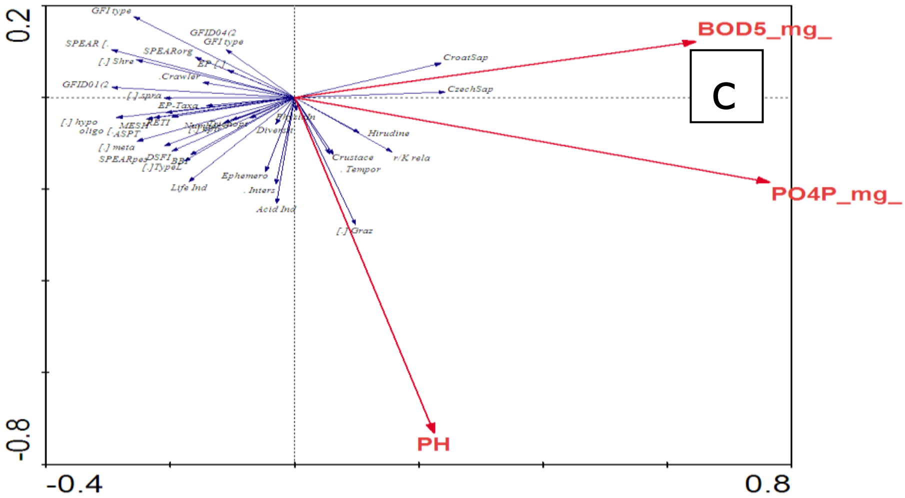


Figure 4.12. RDA of 37 metrics and 41 environmental parameters. a –physical and chemical data together without covariables; b –physical data as environmental data and chemical data as covariable;, c – chemical data as environmental data and physical data as covariables.

The effect of stream typology and soil type (eastern and western Denmark)

Removing the effect of stream typology (including it as covariable), the variance explained by environmental data (both physical and chemical) was smaller than omitting them as covariables (inclusion of stream typology as covariable accounted for 22.8% of the explained variance, while the variance explained amounted to 28.6% when excluding it) (Table 4.1, Table 4.2). For metrics, when omitting the effect of stream typology, the explanatory power was lower than when including it (inclusion of stream typology as covariable accounted for 20.9% of the explained variance compared to 32.9% when omitting it). Similarly, including eastern and western Denmark as covariables did not reveal higher explanatory power than when omitting them as covariables.

Table 4.2. Results of multivariate regression models of taxa (CCA) and metrics (RDA) with environmental data using stream typology and eastern/western Denmark as covariables.

	Taxa (CCA)	Metrics (RDA)
No. taxa/metrics in all analyses	64	41
No. stations	219	219
Total inertia (variance in the species/metrics dataset)	2.9	1.00
Sum of all canonical eigenvalues		
- Stream typology as covariable	0.662	0.209
- Eastern and western Denmark as covariables	0.669	0.330
Cumulative % of species/metric-environment relationship of axes 1 and 2		
- Stream typology as covariable	40	78.3
- Eastern and western Denmark as covariables	50.1	87.9

Species dissimilarity

The SIMPER test identified the species contributing the most to the assemblage differences between different stream typologies and between eastern and western Denmark. Average dissimilarity between different stream typologies was 66.75% (Table 4.3). The highest dissimilarity occurred between reference sites and sites affected by point source pollution (PS2; dissimilarity 72.8%) followed by reference and physically disturbed sites (dissimilarity 70.3%). On average, *Gammarus pulex* contributed significantly to the differences between stream typologies and between eastern and western Denmark, followed by *Baetis rhodani* and *Nemoura cinerea*. The results revealed a similar average dissimilarity, 68.1%, in eastern and western Denmark, and the three main macroinvertebrate species responsible for the difference were the same: *Gammarus pulex* (6.78%), *Baetis rhodani* (4.47%), and *Pisidium* sp. (3.3%).

NMDS showed that there were no well-separated groups (Figure 4.13; Figure 4.14). Figure 4.13 revealed that reference sites were clustered more closely together, and other stream typology groups appeared to be more widespread. Moreover, eastern and western Denmark was not well-separated relative to macroinvertebrate abundance (Figure 4.14).

Table 4.3. Bray-Curtis similarity percentages (SIMPER) test of the contribution (%) of the different macroinvertebrate species to stream typology. Explanation of stream typology abbreviations are given in the chapter on data preparation and analysis. OD – overall dissimilarity between samples.

Stream typology	Macroinvertebrate species	Contribution (%)
PHY	*Gammarus pulex*	6.8
OD 70.3%	*Baetis rhodani*	4.48
	Nemoura cinerea	3.73
AG1	*Gammarus pulex*	6.58
OD 68.47%	*Baetis rhodani*	4.27
	Pisidium sp.	4.09
AG2	*Gammarus pulex*	5.94
OD 68.8%	*Baetis rhodani*	4.47
	Nemoura cinerea	2.87
AG3	*Gammarus pulex*	6.52
OD 69.0%	*Baetis rhodani*	4.44
	Pisidium sp.	2.96
PS1	*Gammarus pulex*	6.7
OD 66.5%	*Nemoura cinerea*	4.2
	Baetis rhodani	3.75
PS2	*Gammarus pulex*	6.75
OD 72.8%	*Baetis rhodani*	2.26
	Asellus aquaticus	4.06

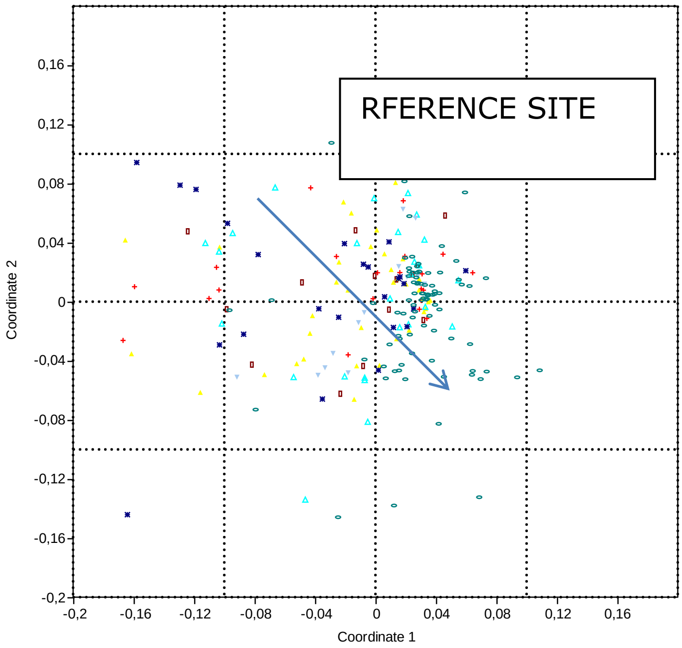


Figure 4.13. NMDS for species in different stream typologies according to the Bray-Curtis dissimilarity measure.

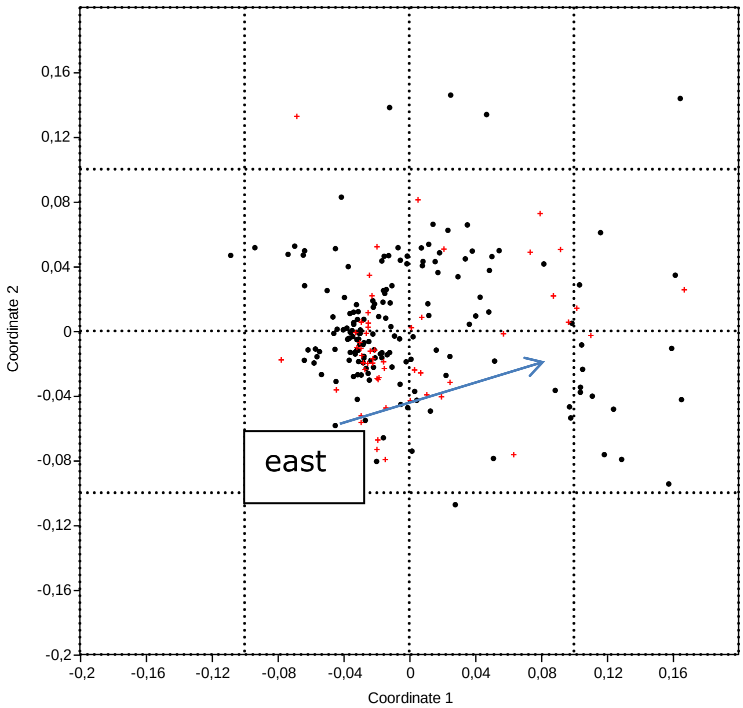


Figure 4.14. NMDS for species in eastern and western Denmark according to the Bray-Curtis dissimilarity measure.

Physically disturbed sites

Running variance partitioning for only physically disturbed sites (n=14) revealed that chemical data did not contribute significantly to the explained variance when tested individually (including physical data as covariables). Thus, three out of 34 environmental variables (all physical) explained 42% of the total variation in species data (Figure 4.15). Four out of 34 environmental variables were significant in explaining the variation in metrics data (Figure 4.16). Three of them were physical and one chemical (pH). Metrics and species data and the % of tree roots in the stream explained most of the variation.

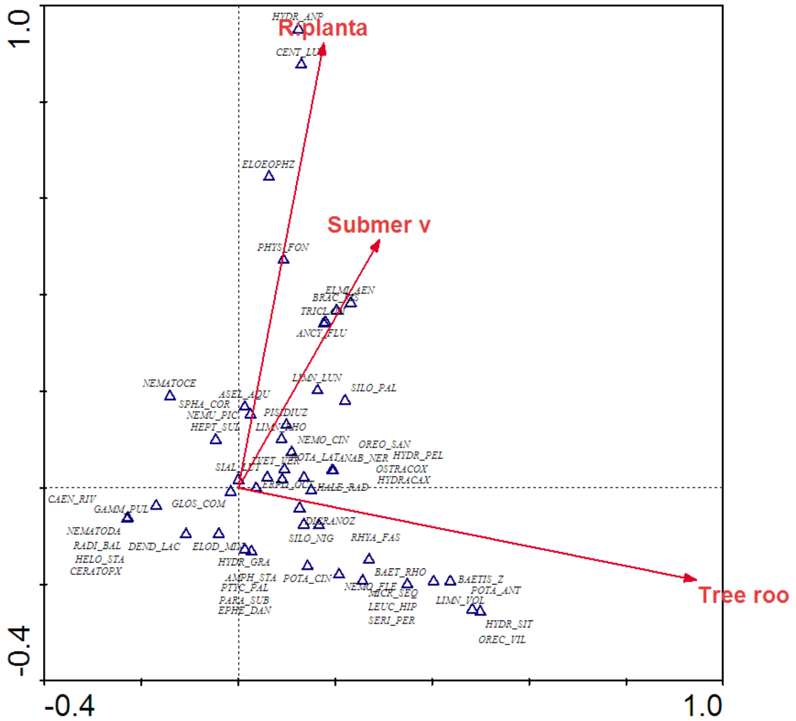


Figure 4.15. CCA analysis including only physically disturbed streams (n=14).

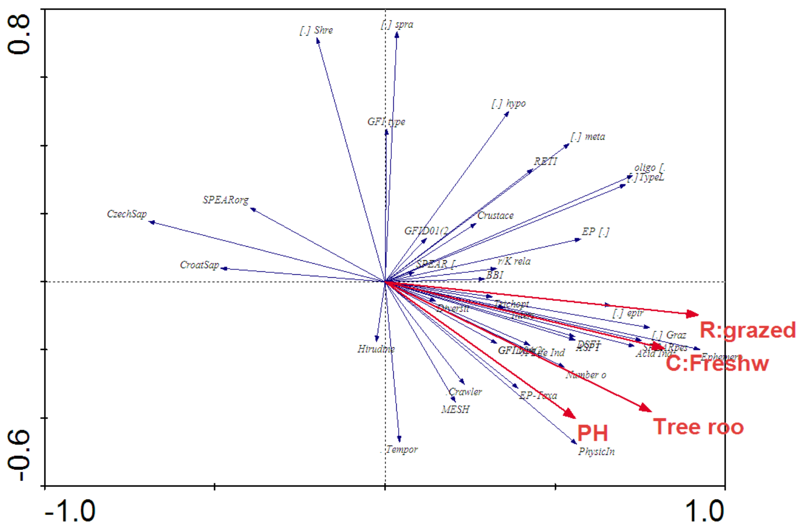


Figure 4.16. RDA analysis including only physically disturbed streams (n=14).

Reference and physically disturbed sites combined

The sum of all constrained eigenvalues using 15 environmental parameters (both physical and chemical) and no covariables was 0.832 (Table 4.4; Figure 4.17). Total variance (inertia) in the dataset was 2.50%, and more than one third (33.2 %) of the variance in the species data could be explained by these explanatory variables. Physical components were clearly the most important factor and explained 20.5% of total explained variability. Chemical factors explained 6.5%, and the variation explained by the intersection of physical and chemical parameters explained 6.1% of total explained variability. The intersection of physical and chemical data variation was not redundant for explaining species composition, and each dataset (physical and chemical data separately) explained unique aspects of species composition.

Table 4.4. Results of multivariate regression models of taxa (CCA) and metrics (RDA) with environmental data using only physically disturbed streams and reference data.

	Taxa (CCA)	Metrics (RDA)
No. taxa/metrics in all analyses	64	41
No. stations	97	97
Total inertia (variance in the species/metrics dataset)	2.505	1.00
Sum of all canonical eigenvalues		
Physical data (chemical data as covariable)	0.832	0.235
Chemical data (physical data as covariable)	0.164	0.019
Intersection of physical and chemical data	0.154	0.04
Cumulative % of species/metric-environment relationship of axes 1 and 2		
Physical data (chemical data as covariable)	50.2	84.4
Chemical data (physical data as covariable)	62.0	52.0

Variance partitioning in metrics (RDA)

The sum of all constrained eigenvalues using six environmental parameters (physical and chemical) and no covariables was 0.294 (Table 4.4; Figure 4.18). Total variance (inertia) in the dataset was 1.00, and less than one third (29.4 %) of the variance in the species data could be explained by these explanatory variables. Again, physical components were clearly the most important factor, accounting for 23.5% of total explained variability. Chemical factors explained 1.9%, and the variation explained by the intersection of physical and chemical parameters accounted for 4.0% of total explained variability. The intersection of physical and chemical data variation was not redundant in explaining species composition, and each dataset (physical and chemical data separately) explained unique aspects of species composition. This analysis showed higher explanatory power for metrics than for species (Table 4.4).

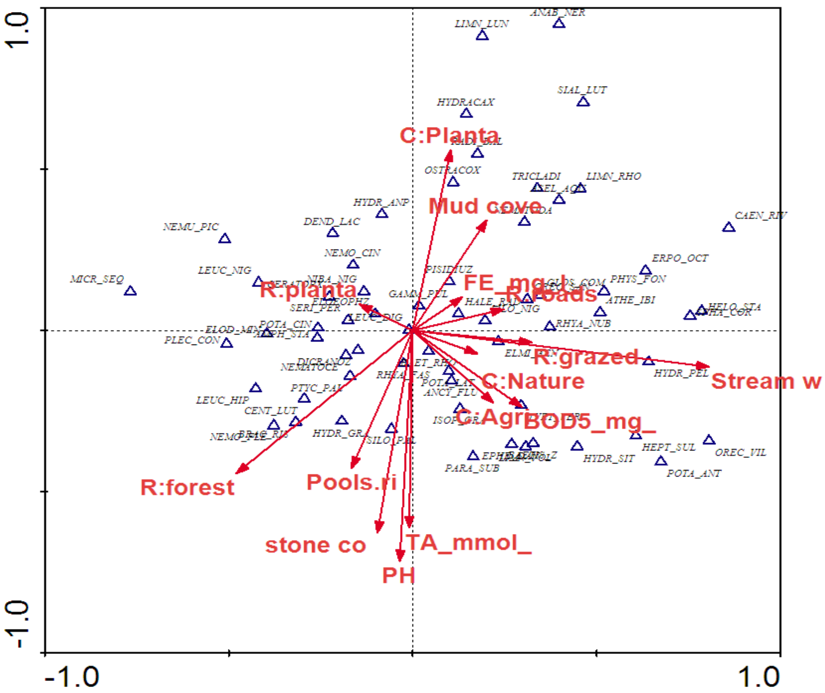


Figure 4.17. CCA analysis with reference sites and physically disturbed streams (n=97).

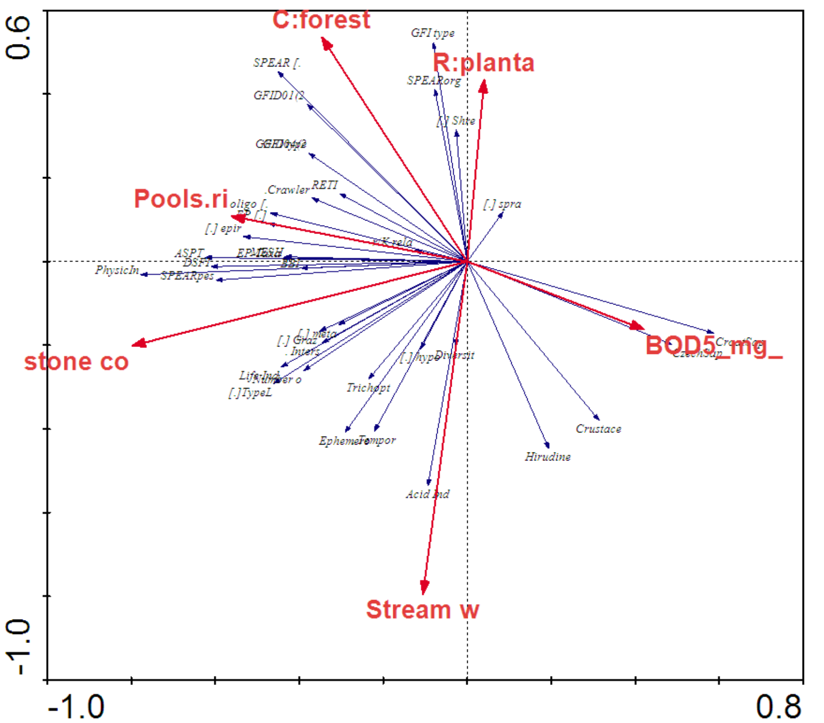


Figure 4.18. RDA analysis with reference sites and physically disturbed streams (n=97).

Hydrological dataset

The proportion of variance explained by environmental variables was much higher for RDA on metrics than CCA on species (Table 4.5). The explanatory power was delivered from the sum of canonical eigenvalues divided by the total inertia. Environmental parameters explain 17.3% of the variance in the species and 20.5% of variance in the metrics data. The first two axes in CCA explain 9.1% of the total variation in species data compared to 19.8% in the metrics data, and is higher for RDA than CCA.

Eight out of 17 environmental data were significant in explaining the variation in species data (Figure 4.19). The first axis expresses Q90 flow magnitude exceeded for 90 % of the time. In the second axis higher values were related to physical parameters as the % of clay and fine gravel and also total alkalinity (Figure 4.19). In RDA, the first axis is highly related to % coarse sand and fine gravel in the sites and BOD5. The second axis expresses % clay (Figure 4.20).

Table 4.5. Results of multivariate regression models of taxa (CCA) and metrics (RDA) with environmental data.

	Taxa (CCA)	**Metrics (RDA)**
No. taxa/metrics in all analyses	92	90
No. stations	34	34
Total inertia (variance in the species/metric dataset)	2.324	1.00
Sum of all canonical eigenvalues		
Without any covariables	0.402	0.205
Year as a covariable	0.443	0.408
Soil type as covariable	0.366	0.395
Cumulative % of species/metric-environment relationship of axes 1 and 2		
Without any covariables	52.7	96.6
Year as a covariable	49.0	87.7
Soil type as covariable	59.7	88.3

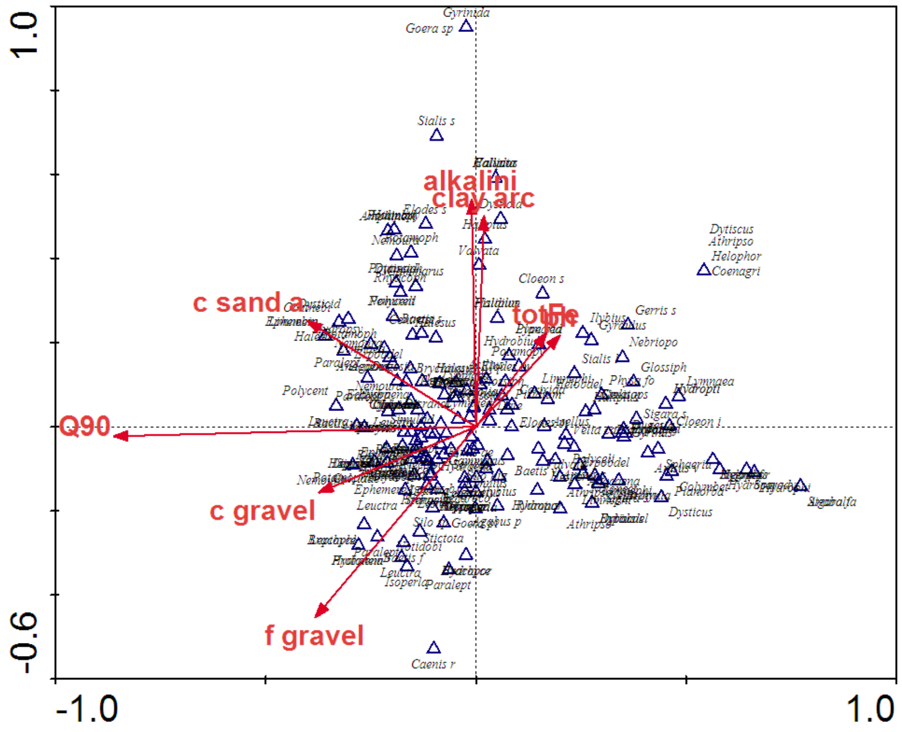


Figure 4.19. CCA analysis with 92 taxa and 26 environmental parameters.

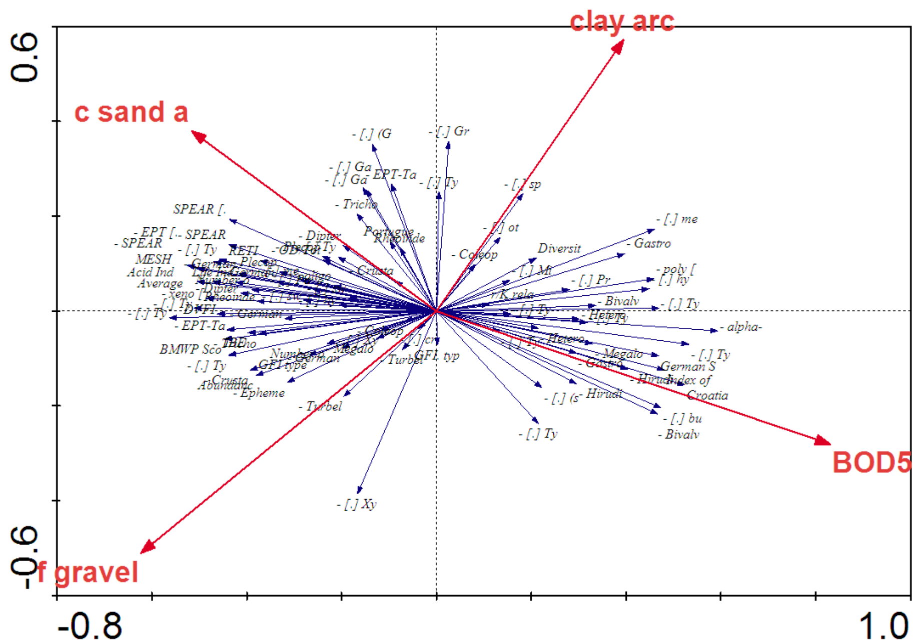


Figure 4.20. RDA analyses with 90 macroinvertebrate metrics and 26 environmental parameters.

Variance partitioning in metrics (RDA) and species data (CCA) with year as covariable

Nine out of 17 environmental variables were significant in explaining the variation in species data. Including year as a covariable, environmental parameters accounted for 18.6% of the explained variance in species data and for 40.8% of the explained variation in metrics data. Including year as a covariable increases the explained variation in both in species and especially in metrics data (Table 4.5).

Soil type as covariable

Seven out of 17 environmental variables were significant in explaining the variation in species data. When including soil type (eastern and western Denmark) as a covariable, environmental parameters accounted for 15.7% of the explained variance in species data and 39.5% of the explained variation in metrics data. Including year as a covariable increases the explained variation in both in species and especially again in metrics data (Table 4.5).

The relationship of selected macroinvertebrate metrics and hydrological data

Most of the selected metrics (except EPT) had strong significant correlation to flow statistics Q90 and Q10 (Table 4.6). The correlation coefficient was mildly lower for Q10 compared to Q90.

Table 4.6. Pearson correlation matrix for maroinvertebrates metrics[a] and flow statistics[a].

	MESH	ASPT	LIFE	EPT	SPEAR (%)
Q90	0.61	0.59	0.52	0.44	0.6
Q10	-0.58	-0.52	-0.47	-0.43	-0.55

[a] MESH – Maroinvertebrates of Estonia: Score of Hydromorphology, ASPT – Average Score Per Taxon, LIFE – Lotic-invertebrate Index for Flow Evaluation, EPT – the total number of taxa belonging to Ephemeroptera, Plecoptera, Trichoptera order, SPEAR (%) – indicated toxicity of pesticides and organic pollution in water, Q10 - flow magnitude exceeded for 10 % of the time, Q90 - flow magnitude exceeded 90% of the time.

All selected macroinvertebrate metrics revealed a significant relationship ($p<0.001$) with flow statistics (Q90 and Q10), although the relationship was not strong ($R^2<0.37$). With increasing discharge (Q90, flow magnitude exceeded 90% of the time), all metrics showed an increase in value. The response of MESH, LIFE and SPEAR (%) to increasing Q90 was logarithmic (Figure 4.21, Figure 4.23, Figure 4.28), ASPT, EPT showed a linear relationship (Figure 4.25, Figure 4.27). Most of the metrics showed significant linear relationship to Q10, except MESH and ASPT (Figure 4.31). MESH index was most strongly related to flow statistics (highest R2 value).

The MESH index increased steeply with increasing discharge until a threshold value (2.7) was reached (Figure 4.21). The highest variation in the MESH index was in very low Q90 values. MESH decreased steadily with increasing Q10 values (Figure 4.22).

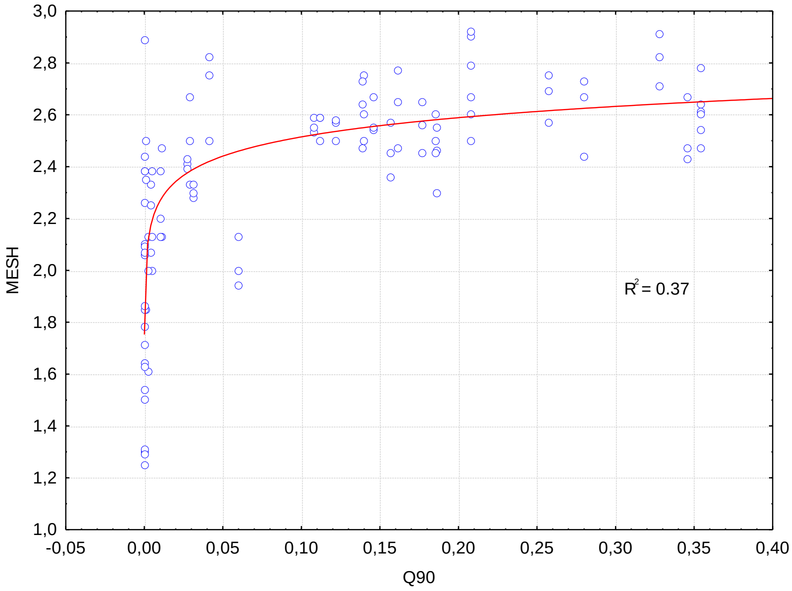


Figure 4.21. Logarithmic regression between MESH index and Q90 (R^2=0.37, p<0.001).

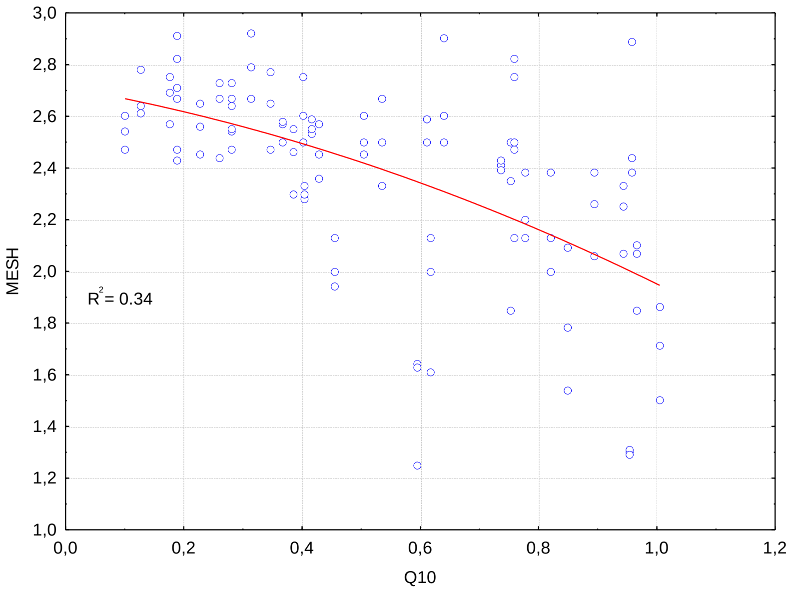


Figure 4.22. Polynomial regression between MESH and Q10 (R^2=0.34, p<0.001).

LIFE index reacted in the same way as MESH by increasing with decreasing discharge (R^2=0.27, p<0.001) and decreasing with increasing Q10 value (R^2=0.22) (Figure 4.23, Figure 4.24).

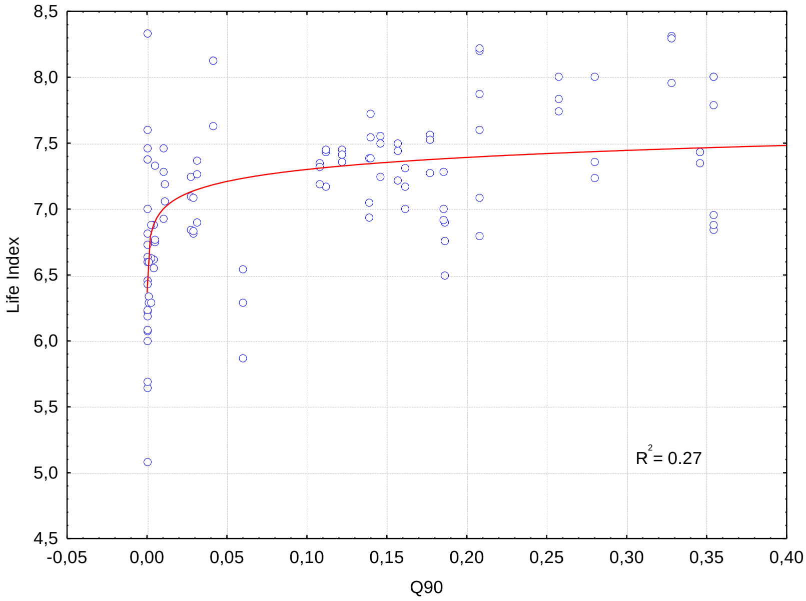


Figure 4.23. Logarithmic regression between LIFE index and Q90 (R^2=0.27, p<0.001).

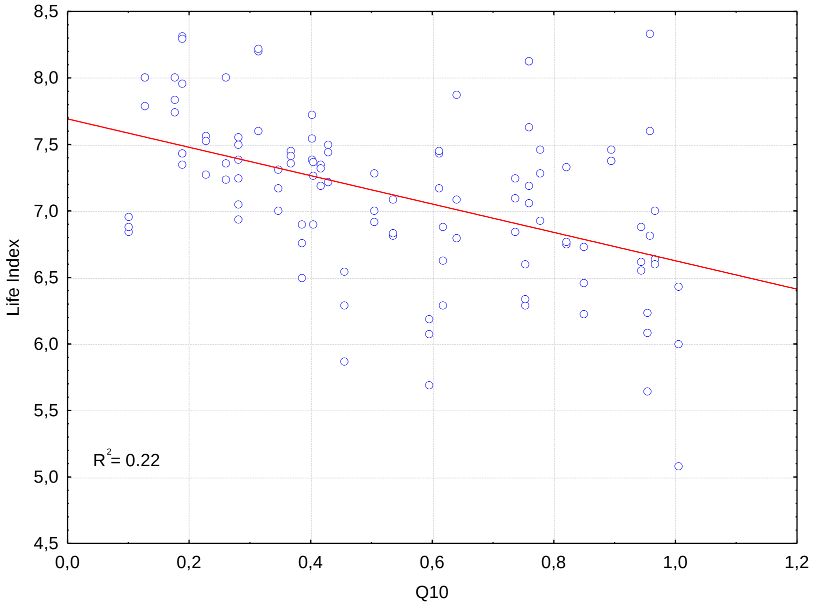


Figure 4.24. Linear regression between LIFE index and Q10.

Figure 4.25 and 4.26 show a significant linear relationship between ASPT and flow statistics (Q90, Q10) (R^2=0.34, p<0.001), similar to the relationship as the MESH and LIFE index.

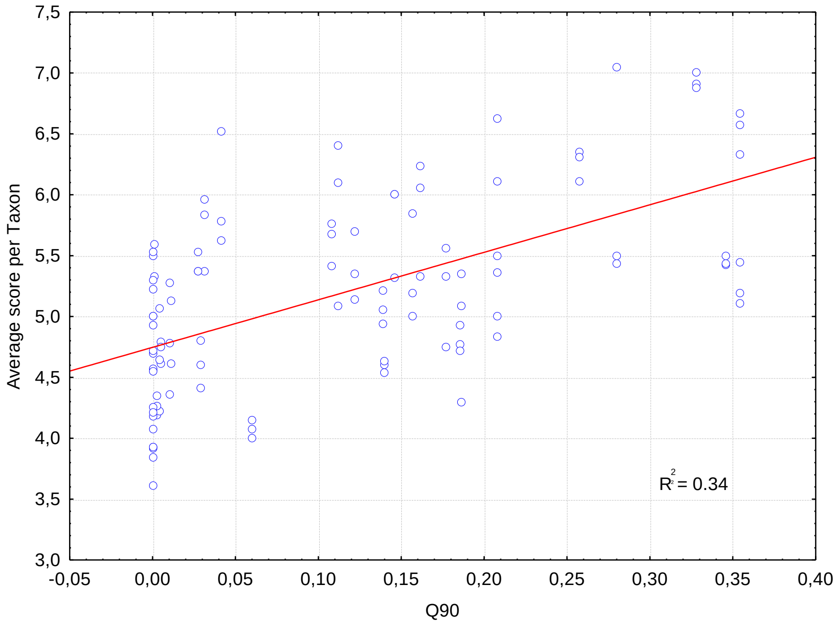


Figure 4.25. Linear regression between ASPT index and Q90.

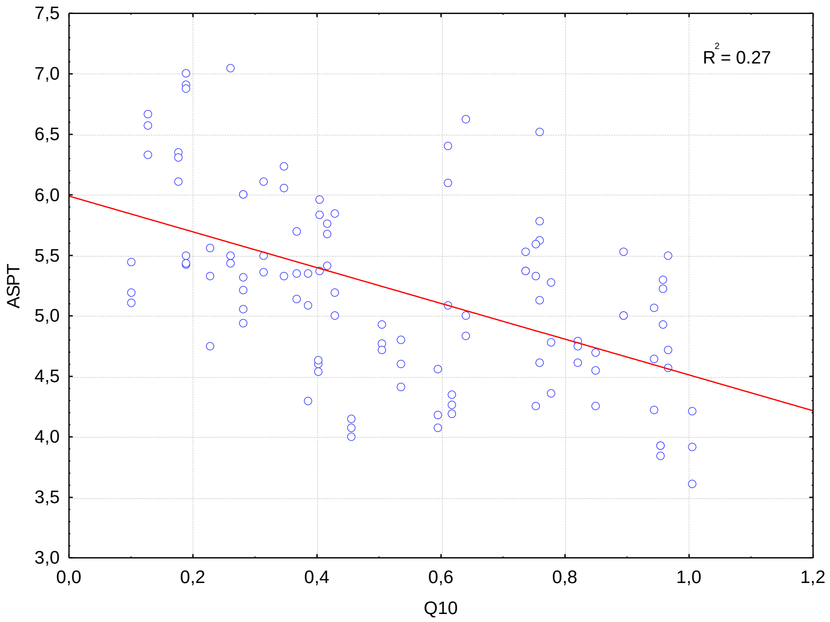


Figure 4.26. Linear regression between ASPT and Q10.

There was also a strong significant relationship between EPT and flow statistics (Figure 4.27, Figure 4.28) (Q90, Q10). The EPT index increased significantly with increasing discharge and decreased with increasing Q10.

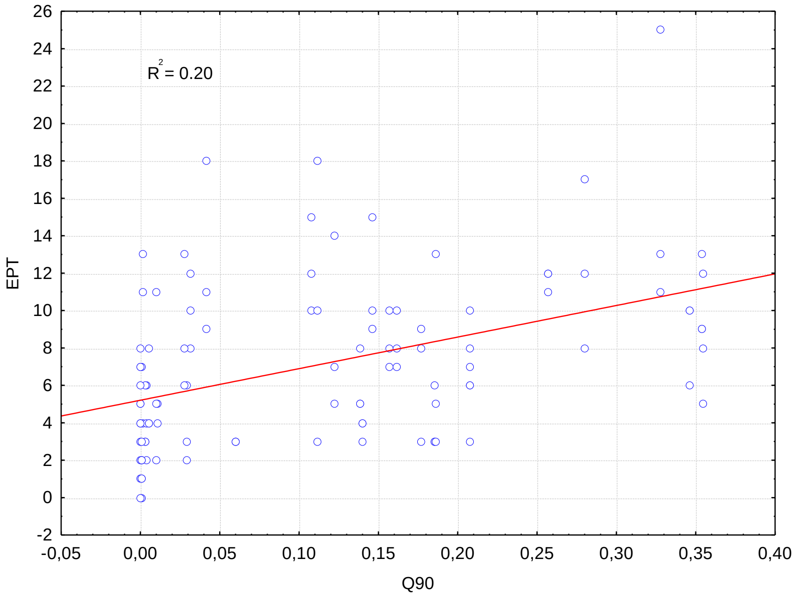


Figure 4.27. Linear relationship between EPT index and Q90.

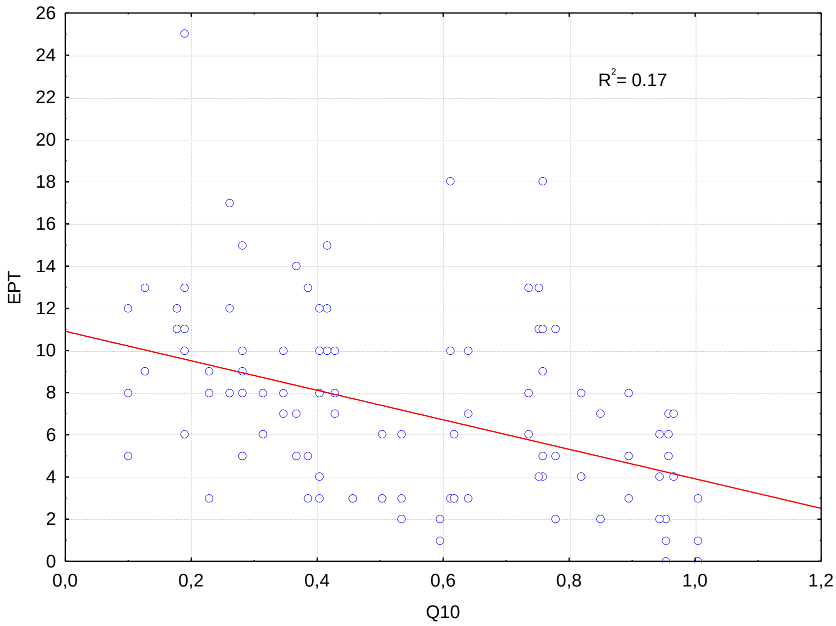


Figure 4.28. Linear relationship between EPT and Q10.

SPEAR (%) index increased sharply with increasing discharge until it reached he threshold value (Figure 4.29). Like most of the other metrics, SPEAR (%) revealed a significant linear relationship with Q10 (R^2=0.30) (Figure 4.30).

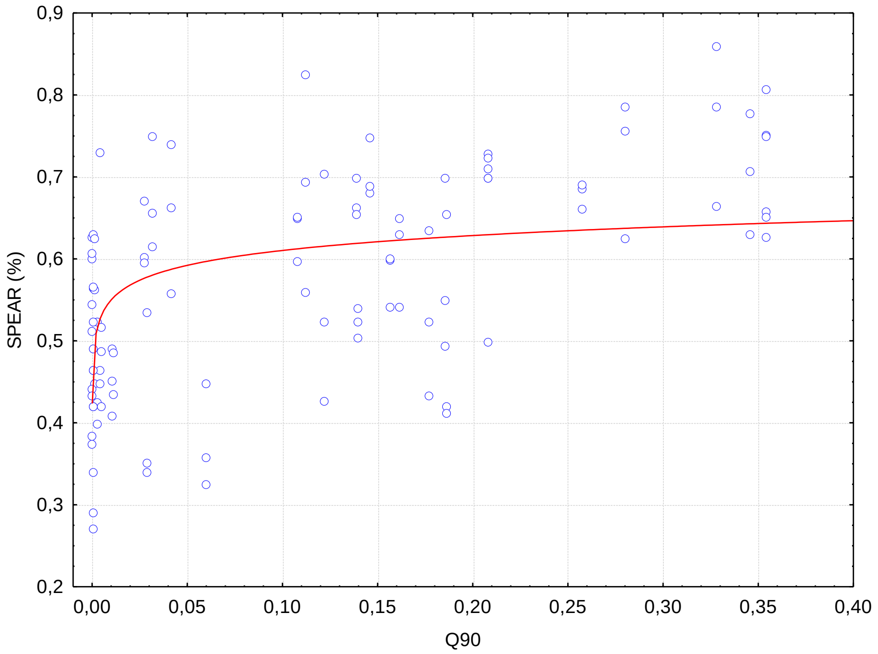


Figure 4.29. Logarithmic relationship of SPEAR (%) and Q90.

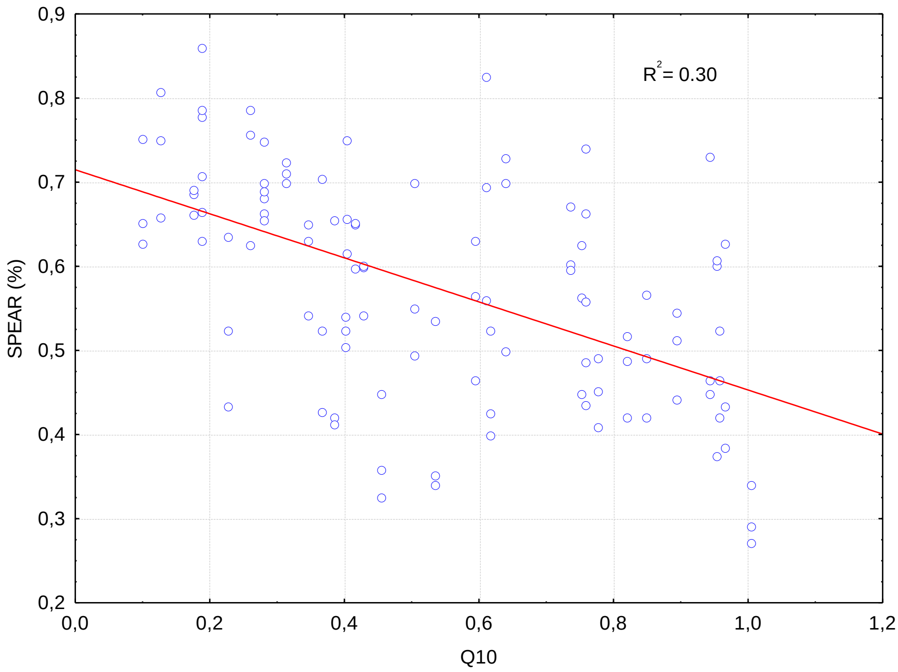


Figure 4.30. Linear relationship of SPEAR (%) and Q10.

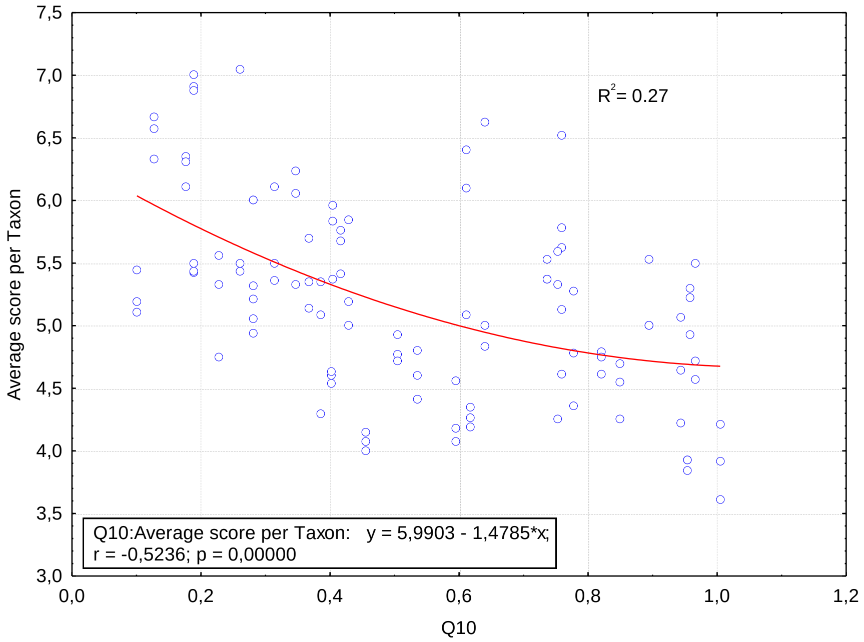


Figure 4.31 Relationship between ASPT and Q10.

Discussion

The results showed that the largest amount of variation in macroinvertebrate species composition was explained by physical variables, strongly indicating the importance of hydromorphological features in shaping riverine biological communities. A slightly higher percentage of variability was explained using metrics rather than species data. However, it should be stressed that the majority of the variability in species composition could not be explained by either the physical or the chemical variables standardly sampled as part of the monitoring of Danish streams. As both dispersal limitations and gradients in the landscape are low in Denmark, it is unlikely that unmeasured natural features contribute substantially to the unexplained variability. This contention is supported by the low effect of stream typology in explaining macroinvertbrate species composition. The explanation is more likely attributed to an insufficient sampling strategy with regard to explanatory variables, both physical and chemical. This could be related to both the actual variables sampled and to the spatio-temporal scale on which they are sampled.

It was evident that a large proportion of the measured variables was significant in explaining variation in macroinvertebrate species composition with plant cover, both riparian and submerged vegetation, tree roots and bed substrates being important. Mud cover on the bed is an important variable that is often perpendicular to variables such as gravel cover, tree roots and pool habitats, indicating degradation in accordance with previous findings (Pedersen et al. 2004).

Analyses of the sub-set of monitoring sites where hydromorphological pressures are perceived to be the main drivers of degradation had a higher explanatory power (42%). Important structural elements were riparian plants, submersed vegetation and tree roots that are important in creating habitats in these otherwise unified streams. There was a tendency to tree roots (axis 1) or vegetation (axis 2) to be important variables, most likely reflecting that both elements rarely occur at the same stream reaches in hydromorphologically degraded streams.

From the smaller dataset with hydrological time-series it is evident that the diagnostic capabilities of metrics designed to be sensitive to hydrological change are limited in comparison with general degradation metrics or the SPEAR index that is sensitive to pesticide pollution. However, these findings have to be viewed in the light of the relatively limited number of sites involved and it should be noted that none of the metrics tested were developed or calibrated to Danish conditions. It is a challenge to find a sufficiently large database to test the link between hydrology and macroinvertebrates, as the number of gauging stations and sampling sites across Europe is very limited.

4.3.4 Comparison of several national datasets (Denmark, Spain and UK)

Dataset description

Individual datasets from Denmark, Spain and the United Kingdom were used for the analyses.

Denmark

Data were derived from the national monitoring programme. One observation from each site was selected randomly from the sites monitored in spring (86% of the total), which resulted in a total of 114 observations. The samples were taken between 2004 and 2011, and the macroinvertebrates were identified to species level for the majority of the taxa.

Land use in the catchments of the monitoring sites is mainly artificial, and specifically mainly urban discontinuous (forested), according to the CORINE classification. There is a minimum of 70% artificial land cover for all the observations, and a minor proportion of land dedicated to agriculture (maximum 3%) and to forest and semi-natural areas (maximum 23%).

Nine per cent of the sites are affected by ochre pollution (50% mild and 50% intense). As for other pollutants, there is a median concentration of 0.09 mg/l ammonium in the available samples (maximum 0.38 mg/l), of 3.2 mg/l nitrate-nitrite nitrogen (maximum of 10.5 mg/l) and of 0.04 mg/l ortophosphate (maximum of 0.13 mg/l).

Spain

Data were derived from a subset of 50 sites monitored between 2007 and 2009 within the Spanish national monitoring programme. One observation from each site was selected randomly from sites monitored in spring and summer (89% of the total), which resulted in a total of 49 observations. Invertebrate data represent family level, and abundances were not used as they were not available for all the sites.

The sites are distributed across Spain, excluding the regions with oceanic climate, and belong to 11 of the total of 32 river types identified at the national level. Data available for chemical parameters indicate that the sites are representative of a wide range of conditions as regards impacts by nitrogen, phosphorous and organic pollution.

United Kingdom

Data were derived from the national monitoring programme. One observation from each site was selected randomly from those that could be matched with hydromorphological data (less than one year in difference between the biological and hydromorphological sampling) and those monitored in spring and summer (85% of the total), which resulted in a total of 89 observations. The samples were taken between 1993 and 2010. The invertebrates are identified to species level for the majority of the taxa.

Macroinvertebrate metrics

Nine macroinvertebrate metrics (Table 4.7) were calculated for all the datasets using the ASTERICS software (AQEM River Assessment Program; Furse et al. 2006; available for download from http://www.fliessgewaesser-bewertung.de/en/download/berechnung). As an exception, the Lotic-invertebrate Index for Flow Evaluation and the $SPEAR_{pesticides}$ index were not calculated if abundance data were not available.

The metrics have been designed to respond to different stressors (AQEM Consortium 2004). LIFE and RHEO measure the presence of taxa with a preference for specific flow and substrate type conditions, while TAXA and EPTT are expected to respond to general degradation, BMWP and ASPT to organic pollution, and SPRP to pesticide pollution.

The value of all the invertebrate metrics is expected to decrease along a gradient of increasing degradation. As an exception, the percentage of taxa with a preference for fine-grained microhabitats (PLAL) is expected to increase.

Table 4.7. Macroinvertebrate metrics.

Code	Metric	Metric description
LIFE	Lotic-invertebrate Index for Flow Evaluation	Average flow score per taxon (species or family). The score for each taxon is obtained from a combination of the assigned flow group and the estimated abundance class and ranges from 1 to 12 (rapid flow preference and highest abundance category) (Extence et al. 1999).
RHEO	Reophilic taxa	Percentage of taxa with a preference for reophilic conditions (moderate to high flow rate) from the total of scored taxa (AQEM Consortium 2004).
PLAL	Preference for Pelal microhabitat	Percentage of taxa with a preference for Pelal microhabitat from the total of scored taxa (AQEM Consortium 2004). Microhabitat: mud (grain size <0.063 mm). Representatives: some tubificids, some aquatic beetles and some chironomids (Sychra & Adámek 2011).
PSAM	Preference for Psammal, Akal or Lithal microhabitat	Percentage of taxa with a preference for Psammal, Akal or Lithal microhabitats from the total of scored taxa (AQEM Consortium 2004). Psammal: sand (grain size from 0.063 to 2 mm); representatives: some tubificids and some chironomids. Akal: fine to medium-sized gravel (0.2-2 cm); representatives: some oligochaetes and some chironomids. Lithal: coarse gravel, stones, boulders (>2 cm); representatives: some leeches and some Diptera (Sychra & Adámek 2011).
TAXA	Number of taxa	Sum of the total number of taxa (AQEM Consortium 2004).
EPTT	EPT Taxa	Number of Ephemeroptera, Plecoptera and Trichoptera taxa (AQEM Consortium 2004).
BMWP	Biological Monitoring Working Party	Sum of the scores assigned to the macroinvertebrate families in the sample. The maximum score for families that are most sensitive to organic pollution is 10 (Armitage et al. 1983).
ASPT	Average Score per Taxon	BMWP divided by the number of families that contributed to this metric (Armitage et al. 1983).
SPRP	$SPEAR_{pesticides}$ (Species At Risk)	Sum of the logarithms of the abundances (plus 1) of the taxa (species or families) sensitive to pesticide pollution divided by the sum of the logarithms of the abundances (plus 1) of all the taxa (AQEM Consortium 2004).

Hydromorphological indicators

Two main groups of indicators were used: indicators of hydromorphological pressures and of hydromorphological conditions. Pressures covered in the datasets comprise flow regulation, surface and groundwater abstraction, channelisation, dredging, bank reinforcement, vegetation removal and general hydromorhological alteration. In turn, hydromorphological conditions quantified in the datasets include substrate composition, conservation of pools and riffles, frequency of high velocity areas, width of riparian buffer, flow diversity, substrate and vegetation types, general habitat diversity and general hydromorhological degradation.

Denmark (hydromorphological conditions and pressures)

The metrics used for the characterisation of hydromorphological conditions and pressures are presented in Table 4.8.

Table 4.8. Hydromorphological indicators (Denmark)

Code	Hydromorphological conditions
NPHI	National Physical Habitat Index (Pedersen et al. 2006)
PLRF	Pools and riffles (percentage of number in natural conditions); 4 possible values: 0 (0%), 13 (1-25%), 50.5 (26-75%) or 88 (> 75%)
HVEL	High velocity (% of stream); 4 values: 0 (0%), 5.5 (1-10%), 18 (11-25%) or 63 (> 25%)
RIPB	Width of riparian buffer; 4 values: 1 (0-2m), 3.5 (2-5m), 7.5 (5-10m) or 10 (> 10m)
GRVL	Gravel (coverage); 4 values: 0 (0%), 5.5 (1-10%), 18 (11-25%) or 63 (> 25%)
MUD	Mud (coverage); 4 values: 2.5 (0-5%), 8 (6-10%), 18 (11-25%) or 63 (> 25%)
Code	**Hydromorphological pressures**
UBRRD	Distance to upstream barriers (km)
DBRRD	Distance to downstream barriers (km)
CHNN	Channelisation (0/1)
DREDG	Dredging (0/1)
REINF	Bank reinforcement (0/1)
RRVEG	River vegetation removal (0/1)
RBVEG	Bank vegetation removal (0/1)
DAMS	Hydrological regime affected by dams (0/1)
DRAIN	Hydrological regime affected by drainage (0/1)
GWABS	Hydrological regime affected by groundwater abstraction (0/1)

Hydromorphological conditions are evaluated using the National Physical Habitat Quality Index methodology (Pedersen et al. 2006). In the study area the index value varies from -1 to 50, indicating a gradient from highly impacted to reference conditions. The NPHI evaluates 16 parameters related to physical conditions plus ochre pollution. The score assigned to each parameter is proportional to the length or surface covered, in the case of elements such as submerged vegetation, or to the intensity, in the case of characteristics such as sinuosity. The sign of the score is given by default for each parameter depending on whether it is considered positive or negative as regards habitat quality.

Apart from the NPHI, five of the individual parameters which it comprises were selected for the analyses. These parameters are the frequency of pools and riffles (PLRF, median 50.5% as related to natural conditions in the observations analysed), the percentage of the stream with high velocity (HVEL, median 18%), the width of the riparian buffer (RIPB, median of more than > 10 m), and the substrate coverage by gravel (GRVL, median of 18%) and mud (MUD, median 2.5%).

High values of PLRF and HVEL are expected to favour taxa with preference for rapid flows and, therefore, higher values of LIFE and RHEO. In turn, high percentages of fine sediments (low GRAVEL and high MUD) are expected to be associated with high values of PLAL and low values of PSAM. The NPHI index is expected to be significantly correlated with the four aforementioned macroinvertebrate metrics. However, the correlation is not expected to be strong, as less than 30% of its total score specifically addresses either flow velocity or substrate composition. Finally, the width of the riparian buffer is not expected to be strongly correlated with these macroinvertebrate metrics, although riparian vegetation can trap sediment and therefore influences substrate composition.

The information available on hydromorphological pressures is categorical (presence/absence) with the exception of the distance to upstream and downstream barriers. The median distance to an upstream barrier is 2.2 km (maximum 45 km), and the median distance to a downstream one is of 1.7 km (maximum 62 km). Many of the categorical pressures are present in less than 10% of the total sites where this information is recorded: 5% or less for channelisation, dredging, bank reinforcement and river vegetation removal, and 9% for bank vegetation removal. The most widespread pressure is modification of the hydrological regime, either by dams (17% of the sites), by draining in catchment (55%) or by groundwater abstraction (35%).

Spain (hydromorphological conditions)

Information was available on the two indices established at the national level to characterise morphological conditions for rivers: the IHF fluvial habitat index for Mediterranean rivers (Pardo et al. 2002) and the QBR quality of riparian habitat index (Munné et al. 2003). The values for the IHF index range from 40 to 83 (median 64), and the values of the QBR index from 0 to 100 (median 70). Both indices are significantly correlated (Spearman correlation coefficient of 0.59, $p<0.05$).

The fluvial habitat assessment method evaluates seven components through a field survey: substrate embeddedness or sediments in pools, substrate composition, rapid frequency, velocity/depth conditions, percentage of shading, presence of heterogeneity elements and aquatic vegetation cover. The IHF index aims to characterise the diversity of in-channel habitats, with a higher score assigned as the heterogeneity of each component increases (maximum 100). Because of this, the IHF index is expected to be a poor predictor for invertebrate metrics related to the preference of a specific flow condition (LIFE and RHEO) or substrate grain size (PLAL and PSAMM) and a better predictor for metrics related to invertebrate diversity.

The QBR riparian habitat index evaluates four components through a field survey, and its value ranges from 0 to 100 (maximum quality). Given that only 25% of the total score of the index depends on the morphological conditions of the river channel (and specifically on river channel naturalness), the QBR index is expected to be a poor predictor for invertebrate metrics related to hydromorphological conditions and a better, though unspecific, predictor for metrics related to general degradation.

United Kingdom (hydromorphological conditions and pressures)

Hydromorphology is evaluated through the Habitat Quality Assessment (HQA) Score and the Habitat Modification Score (HMS) (Raven et al. 1998). The HQA score quantifies the presence and extent of habitat features of wildlife interest, while the HMS score measures the degree of artificial modifications to the channel. HQA and HMS scores increase, respectively, as habitat quality and the degree of alteration increase. The observed values for the HQA score range from 10 to 75, and the HMS values range from 0 to 5020, indicating a gradient from pristine/semi-natural conditions to severely modified sites.

Additional hydromorphological indicators used to characterise site conditions are (Table 4.9) three of the component scores of HQA (related to channel flow types, substrate and vegetation) and the three HMS component scores devised to evaluate the three main types of physical alteration of the channel (reinforcement, resectioning and flow regulation).

Table 4.9. Hydromorphological indicators (United Kingdom).

Code	Hydromorphological conditions
HQA	Habitat Quality Assessment Score and (Raven et al. 1998)
FLOWT	Diversity of flow types
SUBST	Diversity of substrate types
RVEGT	Diversity of in-channel vegetation types
Code	**Hydromorphological pressures**
HMS	Habitat Modification Score (Raven et al. 1998)
REINF	Bank reinforcement with concrete, steel piling, gabion, rip-rap, etc.
DREDG	Reprofiling through dredging of the bed and banks
FLOWR	Flow regulation by impounding structures

Data analysis

Relationships between macroinvertebrate and hydromorphological indicators were evaluated using Spearman correlation (Harre 2010) and Mann-Whitney (R Development Core Team 2010) tests.

Generalised linear models were constructed to predict macroinvertebrate metrics from hydromorphological indicators (R Development Core Team 2010). Binomial error distributions were used for the prediction of percentages, and Poisson distributions were used for the rest of response variables. Quasi-likelihood models were employed when overdispersion was detected, and models were selected through manual backward elimination. The significance of increase in deviance provided by more complex models was evaluated through chi-squared tests, or F tests in the case of quasi-likelihood models. Models were evaluated through analysis of influential points and collinearity between explanatory variables: Cook's distance was less than 1, and tolerance (Fox & Weisberg 2010) was greater than 0.1 for all models unless specified otherwise.

Additionally, the explanatory capacity of hydromorphological indicators in the prediction of macroinvertebrate metrics with random forests was quantified trough the importance (Liaw & Wiener 2002).

Analyses were performed in R version 2.10.1 (R Development Core Team 2010).

Results and discussion (1) Relationships between hydromorphological pressures and conditions

Hydromorphological effects of hydromorphological pressures monitored in the datasets are widely reported in the literature, including the effects of water abstraction (Dewson et al. 2007), flow regulation and impoundment, especially by large structures (Thoms et al. 2005; Poff et al.

2007; Grant 2012), dredging (Kondolf et al. 1997; Rinaldi et al. 2005), and alteration of the natural vegetation (Allan 2004; Gurnell et al. 2012).

Denmark

The majority of relationships that could have been expected from literature findings were not identified as significant in the study area. The distance to upstream and downstream barriers does not show a significant correlation with hydromorphological conditions (Spearman correlation coefficient, p<0.05). Likewise, these conditions do not show a significant relationship with the majority of qualitative pressure indicators (Mann-Whitney test, p<0.05).
Figure 4.32 depicts the difference in mean values of individual hydromorphological condition indicators between sites where pressures are and are not present for the significant relationships. Apart from these relationships, NPHI shows lower values in sites that have been dredged or where river vegetation has been removed.

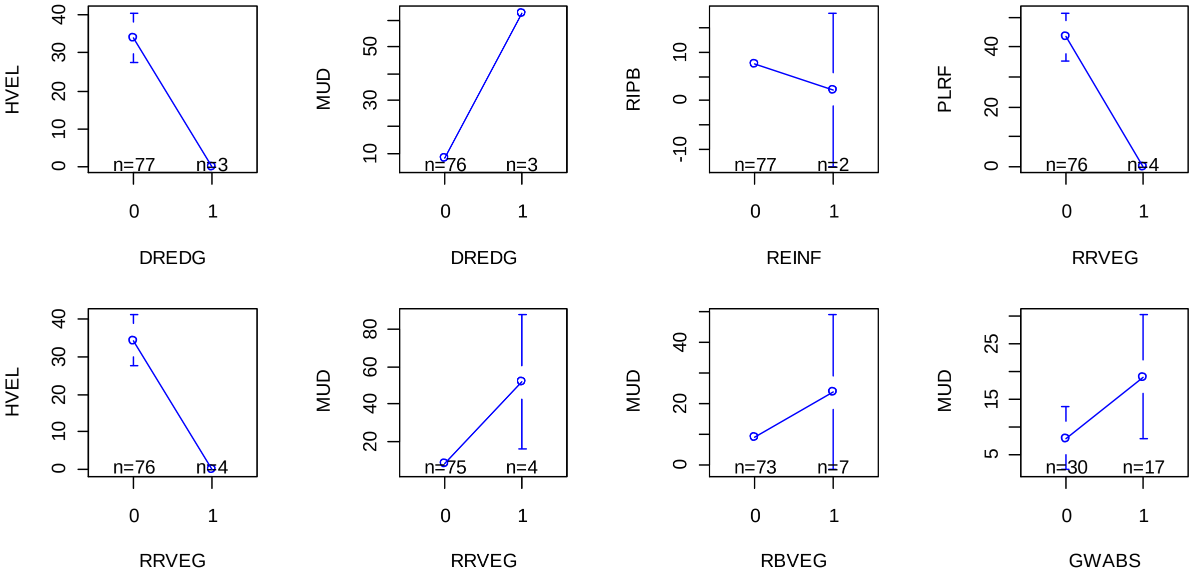


Figure 4.32. Group means and 0.95 confidence intervals for significant relationships (Mann-Whitney test, p<0.05) between qualitative hydromorphological pressures (yes/no) and conditions (Denmark).

The substrate has a greater proportion of fine particles in the sites that have been dredged, which typically leads to coarsening of bed material (Kondolf et al. 1997; Rinaldi et al. 2005), or where river or bank vegetation is removed, which can reduce sediment trapping and increase bank and channel erosion (Allan 2004; Gurnell et al. 2012). Additionally, flow velocity is lower in the first two cases, and the pool-riffle sequence has been more altered in sites where river vegetation has been so too. However, as mentioned before, the percentage of sites where these pressures are present is low, and because of this it is not possible to obtain solid conclusions, especially regarding the size of the effects.

The only pressures with a close to balanced proportion of impacted to non impacted sites are those related to hydrological regime alteration. In this case, the only pressure that shows a significant relationship with hydromorphological conditions is groundwater abstraction, which can be expected to increase the proportion of fines in the substrate through the decrease in flow velocity (Dewson et al. 2007; Parkin et al. 2007).

United Kingdom

The Habitat Quality Assessment Score is negatively correlated with the Habitat Modification

Score, as should be expected (Table 4.10; Figure 4.33). The only hydromorphological condition (HQA component) that is significantly correlated with hydromorphological pressures (HMS components) is the diversity of flow types, which shows higher values in sites with higher values of pressures related to bank reinforcement and flow regulation, which are in turn significantly (and positively) correlated. However, the dataset does not allow evaluation of whether flow diversity is naturally higher in the areas where banks are reinforced and flow is regulated more frequently.

Table 4.10. Significant correlations (Spearman correlation coefficient, $p<0.05$) between hydromorphological conditions and pressures (N = 89) (United Kingdom).

r	HQA	FLOWT	SUBST	RVEGT
HMS	-0.24			
REINF		0.27		
DREDG	-0.31			
FLOWR		0.24		

r	REINF	DREDG
REINF	-	-
DREDG		-
FLOWR	0.41	

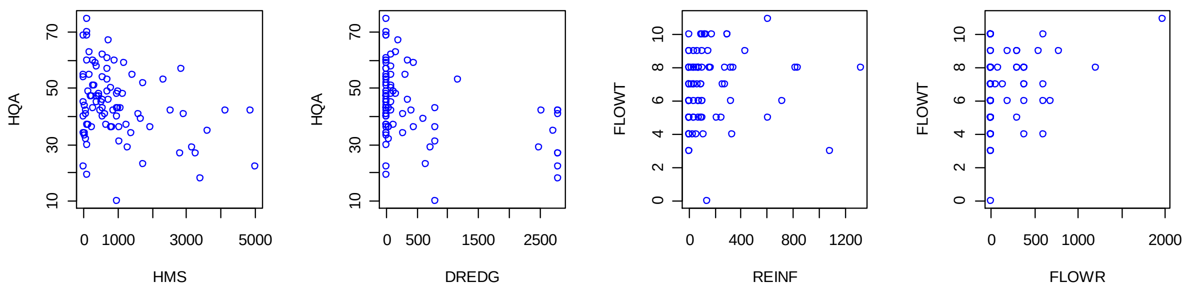


Figure 4.33. Examples of correlation patterns between hydromorphological indicators (United Kingdom).

Results and discussion (2) Relationships between macroinvertebrate metrics and hydromorphological indicators

Hydromorphological conditions are key factors in the structuring of macroinvertebrate communities (Lytle & Poff 2004; Allan & Castillo 2007). Macroinvertebrate indicators were developed originally for the assessment of pollution, but increasing attention is being paid to how they respond to hydromorphological alterations, including modification of natural flow regimes (Poff & Zimmerman 2010) and in-stream habitat conditions (Dunbar et al. 2010). However, relatively few cases of clear impacts of habitat degradation have been documented so far (Friberg 2010), and significant progress is still needed in the analysis of the direct and indirect mechanisms underlying the relationships between flow regime and aquatic biota (Lancaster & Downes 2010).

<u>Denmark</u>

Quantitative hydromorphological indicators

Multiple significant correlations were identified between macroinvertebrate metrics and hydromorphological indicators, with the exception of the total number of taxa and the distance to upstream and downstream barriers (Table 4.11). The absence of relationships with barriers was to be expected given that, as shown above, the distance to upstream and downstream barriers does not show significant correlation with hydromorphological conditions in the dataset analysed.

The key expected significant correlations between in-channel specific conditions and the corresponding macroinvertebrate metrics (AQEM Consortium 2004) are observed in the results: the conservation of the natural pool-riffle sequences and the existence of high velocity areas are correlated with higher values of LIFE and RHEO, and a greater proportion of fines in the substrate is correlated with higher values of PLAL and lower values of PSAM. However, it must be noted that the correlations of these four macroinvertebrate metrics are often stronger with other hydromorphological indicators, and especially with the NPHI index.

Riparian vegetation can have effects on multiple in-stream conditions, ranging from bed granulometry and erosion to shading and water quality improvement (Allan 2004; Gurnell et al. 2012). In the Danish dataset, the width of the riparian buffer is correlated with the macroinvertebrate metrics representative of general degradation; however, it is not correlated with those specifically related to substrate type preferences.

Table 4.11. Significant correlations (Spearman correlation coefficient, p<0.05) between hydromorphological indicators and macroinvertebrate metrics (N between 111 and 114 depending on the case) (Denmark).

r	LIFE	RHEO	PLAL	PSAM	TAXA	EPTT	BMWP	ASPT	SPRP
NPHI	0.54	0.56	-0.47	0.33		0.37	0.34	0.50	0.43
PLRF	0.43	0.45	-0.43			0.30	0.33	0.44	0.32
HVEL	0.48	0.53	-0.42	0.35				0.27	0.31
RIPB						0.19	0.21	0.22	0.21
GRVL	0.40	0.34	-0.27	0.20		0.26	0.21	0.32	0.24
MUD	-0.45	-0.50	0.44	-0.38		-0.30	-0.27	-0.33	-0.27
UBRRD									
DBRRD									

In general, the correlation coefficients are low to moderate, with absolute values ranging from 0.19 to 0.56. This along with the observed "wedge-shaped" relationships (Figure 4.34) is to be expected in multi-pressure environments where data on different environmental gradients are typically necessary to explain the behaviour of indicators of biological quality elements (Friberg 2010).

Finally, the results suggest that the macroinvertebrate metrics designed to respond to hydromorphological pressures do not show a greater sensitivity to hydromorphological indicators than those intended to respond to general degradation and pollution. In fact, the values of the correlation coefficients for the former are only slightly higher in average than for those of the latter.

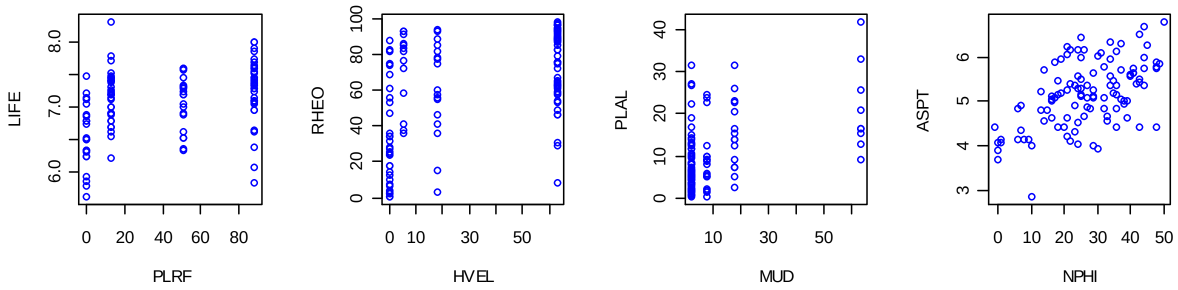


Figure 4.34. Examples of correlation patterns between macroinvertebrate metrics and hydromorphological indicators (Denmark).

Qualitative hydromorphological indicators

Significant relationships are observed for both macroinvertebrate metrics designed to respond to hydromorphological pressures and to general degradation and pollution (Table 4.12), as was the case for quantitative hydromorphological indicators.

Here, however, a great percentage of the expected relationships was not identified. Most notably, no significant relationships are observed for three of the most relevant pressures: channelisation, bank reinforcement and flow regulation by dams. This could be partially explained by the fact that the majority of the pressures are only present in a limited proportion of the sites (Table 4.12), and was to be expected given the few relationships identified between hydromorphological pressures and conditions in the previous subsection.

In the case of significant relationships, and as anticipated, the absence of hydromorphological pressures is associated with lower values of PLAL and higher values of the rest of the invertebrate metrics (Table 4.12). Figure 4.35 represents observed size effects, especially for pressures related to drainage and groundwater abstraction, which are the most extended and therefore allow for a more balanced analysis. Reduced flows derived from surface or groundwater abstractions have the potential to reduce water depth and velocity, dissolved oxygen levels and substrate size (Dewson et al. 2007). They are generally associated with a decline in macroinvertebrate abundance and diversity, although an increase in both parameters can also sometimes be observed (Poff & Zimmerman 2010).

Table 4.12. Significant relationships (Mann-Whitney test, $p<0.05$) between macroinvertebrate metrics and hydromorphological pressures (yes/no) (Denmark). PRESS: percentage of samples with pressure. Effect positive (+)/ negative (-).

Effect	N	PRESS	LIFE	RHEO	PLAL	PSAM	TAXA	EPTT	BMWP	ASPT	SPRP
CHNN	78	4%									
DREDG	80	4%		-	+	-		-	-	-	-
REINF	79	3%									
RRVEG	80	5%	-	-	+	-	-	-	-	-	-
RBVEG	81	9%			+	-					
DAMS	52	17%									
DRAIN	49	55%	-				-	-	-	-	
GWABS	48	35%	-		+	-	-	-	-	-	-

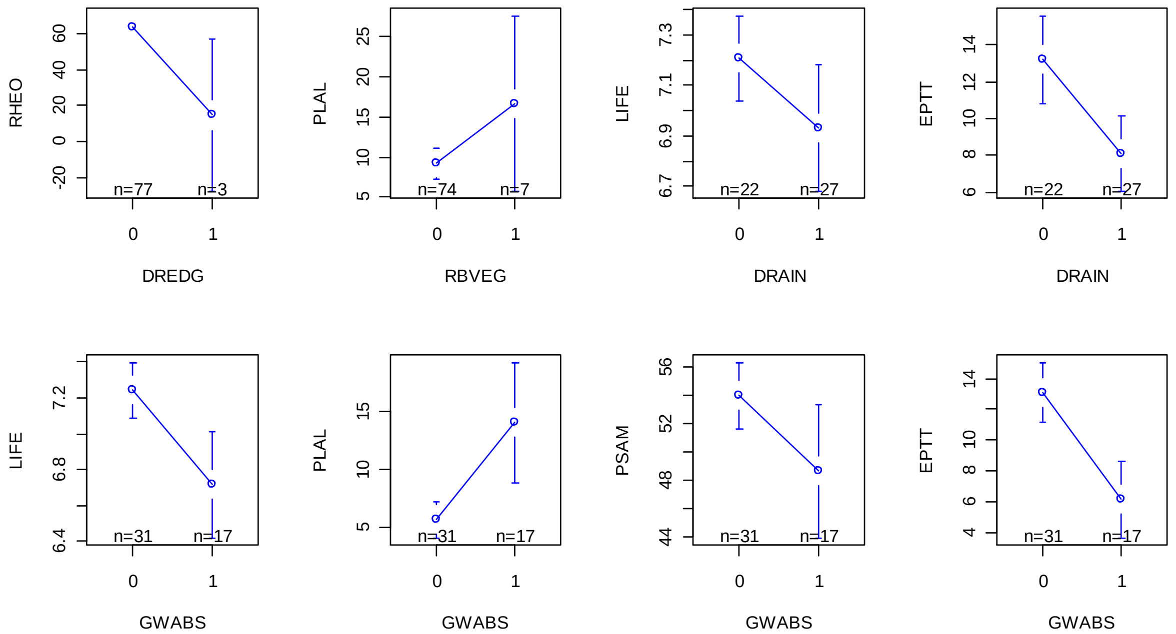


Figure 4.35. Group means and 0.95 confidence intervals for significant relationships (Mann-Whitney test, p<0.05) between macroinvertebrate metrics and hydromorphological pressures (yes/no) (Denmark).

Spain

As expected, the correlations between the IHF and QBR indices and the macroinvertebrate metrics designed to respond to hydromorphological conditions are, with a single exception, not significant (Table 4.13; Figure 4.36). For the rest of the macroinvertebrate metrics, the values of the correlation coefficients are moderate, with absolute values ranging from 0.40 to 0.63. As stated earlier, given that the IHF index (Pardo et al. 2002) measures the diversity of the fluvial habitat but not its naturalness, and given that the morphological conditions of the river channel account for only 25% of the score of the QBR quality of riparian habitat (Munné et al. 2003), these indices are more appropriate for evaluating general degradation than specific hydromorphological alterations.

Table 4.13. Significant correlations (Spearman correlation coefficient, p<0.05) between macroinvertebrate metrics and hydromorphological indicators (Spain).

R	N	RHEO	PLAL	PSAM	TAXA	EPTT	BMWP	ASPT
IHF	49		-0.30		0.40	0.45	0.44	0.49
QBR	48				0.51	0.55	0.54	0.63

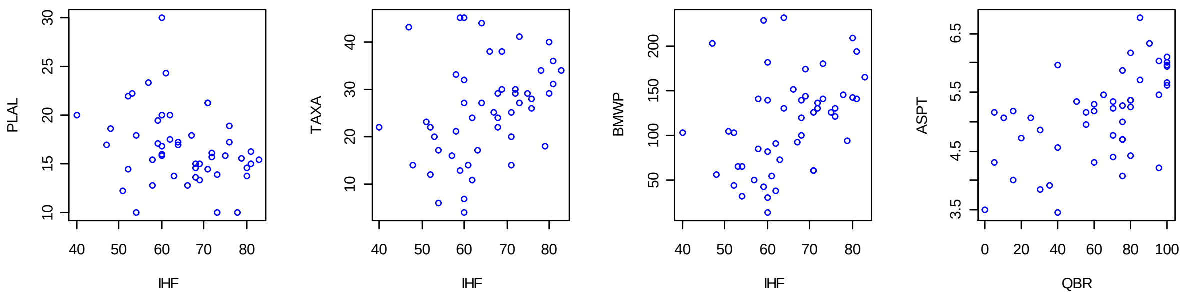


Figure 4.36. Examples of correlation patterns between macroinvertebrate metrics and hydromorphological indicators (Spain).

United Kingdom

The majority of relationships that could have been expected a priori between hydromorphological pressures (Habitat Modification Score and components) and macroinvertebrate metrics were not identified as significant in the study area (Table 4.14). In fact, no significant correlations were identified for the HMS components related to bank reinforcement and flow regulation, which were observed to be correlated with a higher diversity of flow types in the previous analyses. The two observed relationships are lower values of LIFE and higher values for PLAL associated with bank and bed dredging.

In turn, the Habitat Quality Assessment Score shows stronger correlations with macroinvertebrate metrics designed to assess general degradation and pollution than with those related to flow and substrate type preference, which was to be expected given that the HQA score quantifies the habitat heterogeneity (Raven et al. 1998) and not the intensity of hydromorphological impacts.

In any case, the significant relationships identified are weak, with absolute values for the correlation coefficients ranging from 0.22 to 0.36 and "wedge-shaped" patterns frequently occurring (Figure 4.37).

Table 4.14. Significant correlations (Spearman correlation coefficient, p<0.05) between hydromorphological indicators and macroinvertebrate metrics and (N from 87 to 89 depending on the case) (United Kingdom).

r	LIFE	RHEO	PLAL	PSAM	TAXA	EPTT	BMWP	ASPT	SPRP
HQA	0.27				0.24	0.30	0.31	0.26	0.28
FLOWT									0.33
SUBST						0.23			
RVEGT		0.23			0.30		0.25		
HMS	-0.33							-0.25	
REINF									
DREDG	-0.36		0.25					-0.22	
FLOWR									

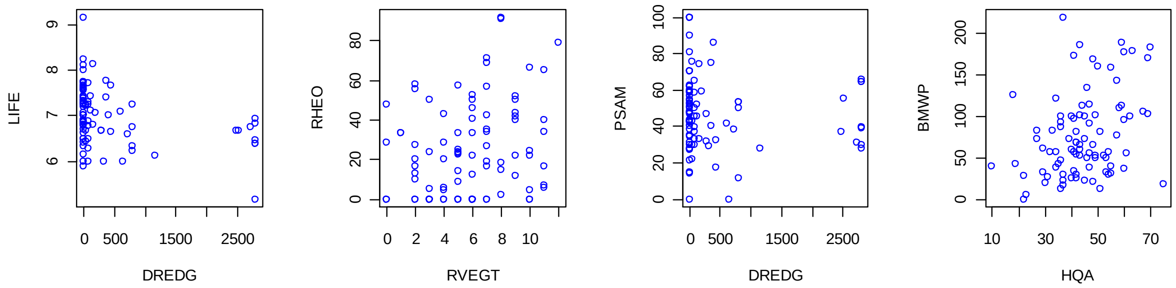


Figure 4.37. Examples of correlation patterns between macroinvertebrate metrics and hydromorphological indicators (United Kingdom).

Results and discussion (3) Prediction of macroinvertebrate metrics from hydromorphological indicators

Denmark

Analysis of importance in random forest models shows a low explanatory capacity of individual predictor variables (NPHI, PLRF, HVEL, RIPB, GRVL, MUD, UBRRD and DBRRD), as was expected from the results obtained previously.

The maximum increase of the mean square error in invertebrate metric predictions after permuting each predictor variable is approximately 8% (examples for LIFE, RHEO and PLAL in Figure 4.38).

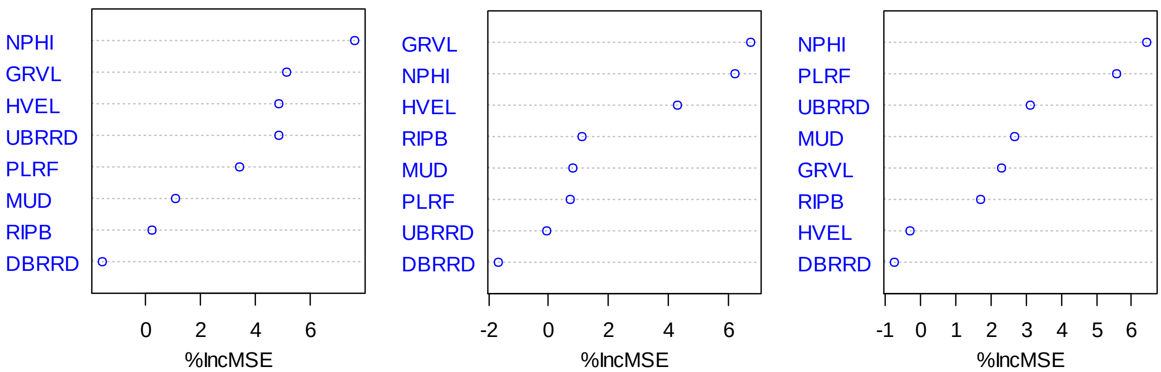


Figure 4.38. Importance of predictor variables in random forest models (left: LIFE; centre: RHEO; right: PLAL) (Denmark).

Generalised linear models were constructed to predict macroinvertebrate metrics designed to respond to hydromorphological pressures (LIFE, RHEO and PLAL) plus ASPT. As PLRF, HVEL and MUD are expected to be related with these biological metrics and have been observed to be correlated with them in the study area, they were selected as predictors. Models were developed firstly for the individual predictors expected to be most strongly correlated with the corresponding invertebrate metric and secondly for a combination of hydromorphological indicators. Additionally, NPHI was included as a predictor in a third round of analysis.

As expected, models explain a low to moderate variance in the data (Table 4.15). The model deviance is very similar for the four response variables and ranges from 24% to 36% when using a combination of hydromorphological indicators, and is significantly lower for single predictors.

Models constructed using NPHI as an additional predictor explain a slightly higher variance, excepting PLAL where NPHI is not significant. However, NPHI is less interesting than individual pressure indicators when analysing pressure-response relationships, as is the case in general with metrics designed to integrate the effects of different impacts, NPHI offering less insight into the mechanisms by which hydromorphological pressures may impact the biota.

Table 4.15. Generalised linear models for the prediction of macroinvertebrate metrics from hydromorphological indicators (Denmark).

Predictors	Inv. metric	Model	N	D^2
Single predictor	LIFE	$= e^{(1.9231\pm0.0093) + (0.0010\pm0.0002)\ HVEL}$	114	17%
	RHEO	$= \frac{100\ e^{-(0.212\pm0.151) + (0.021\pm0.004)\ HVEL}}{1 + e^{-(0.212\pm0.151) + (0.021\pm0.004)\ HVEL}}$	114	21%
	PLAL	$= \frac{100\ e^{-(2.478\pm0.098) + (0.020\pm0.004)\ MUD}}{1 + e^{-(2.478\pm0.098) + (0.020\pm0.004)\ MUD}}$	113	21%
	ASPT	$= e^{(1.475\pm0.028) + (0.006\pm0.001)\ NPHI}$	111	29%
NPHI excluded	LIFE	$= e^{(1.9520\pm0.0114) + (0.0006\pm0.0002)\ HVEL - (0.0017\pm0.0004)\ MUD}$	113	28%
	RHEO	$= \frac{100\ e^{(0.078\pm0.232) + (0.007\pm0.003)\ PLRF + (0.012\pm0.004)\ HVEL - (0.032\pm0.009)\ MUD}}{1 + e^{(0.078\pm0.232) + (0.007\pm0.003)\ PLRF + (0.012\pm0.004)\ HVEL - (0.032\pm0.009)\ MUD}}$	113	36%
	PLAL	$= \frac{100\ e^{-(2.165\pm0.137) - (0.009\pm0.003)\ HVEL + (0.015\pm0.004)\ MUD}}{1 + e^{-(2.165\pm0.137) - (0.009\pm0.003)\ HVEL + (0.015\pm0.004)\ MUD}}$	113	27%
	ASPT	$= e^{(1.6257\pm0.0230) + (0.0011\pm0.0004)\ PLRF - (0.0028\pm0.0008)\ MUD}$	113	24%
NPHI included	LIFE	$= e^{(1.9022\pm0.0209) + (0.0023\pm0.0006)\ NPHI - (0.0010\pm0.0005)\ MUD}$	111	32%
	RHEO	$= \frac{100\ e^{-(0.493\pm0.369) + (0.045\pm0.011)\ NPHI - (0.026\pm0.010)\ MUD}}{1 + e^{-(0.493\pm0.369) + (0.045\pm0.011)\ NPHI - (0.026\pm0.010)\ MUD}}$	111	38%
	PLAL	$= \frac{100\ e^{-(2.165\pm0.137) - (0.009\pm0.003)\ HVEL + (0.015\pm0.004)\ MUD}}{1 + e^{-(2.165\pm0.137) - (0.009\pm0.003)\ HVEL + (0.015\pm0.004)\ MUD}}$	113	27%
	ASPT	$= e^{(1.475\pm0.028) + (0.006\pm0.001)\ NPHI}$	111	29%

<u>Spain</u>

The IHF and QBR indices show a poor capacity for predicting invertebrate metrics designed to respond to hydromorphological conditions and a moderate capacity to predict metrics related to general degradation. In fact, they are not significant for the prediction of RHEO and PSAM. This was to be expected from the characteristics of both indicators, which are more adequate as predictors for general degradation analyses in Spanish Mediterranean catchments (Catalinas & García 2011), and supports the findings of the correlation analyses.

Table 4.16 presents, as an example, the generalised linear models constructed to predict PLAL and ASPT from hydromorphological indicators. IHF and QBR are not significant when used in combination in these two cases.

Table 4.16. Generalised linear models for the prediction of macroinvertebrate metrics from hydromorphological indicators (Spain).

Inv. metric	Model	N	D^2
PLAL	$= \frac{100\ e^{-(1.086\pm0.243) - (0.008\pm0.004)\ IHF}}{1 + e^{-(1.086\pm0.243) - (0.008\pm0.004)\ IHF}}$	49	9%
	$= \frac{100\ e^{-(1.432\pm0.086) - (0.003\pm0.001)\ QBR}}{1 + e^{-(1.432\pm0.086) - (0.003\pm0.001)\ QBR}}$	48	10%
ASPT	$= e^{(1.217\pm0.128) + (0.006\pm0.002)\ IHF}$	49	19%
	$= e^{(1.447\pm0.044) + (0.003\pm0.001)\ QBR}$	48	32%

United Kingdom

Analysis of importance in random forest models shows a low explanatory capacity of individual predictor variables (HQA, HMS and their respective components), as was expected from the results obtained in the correlation analyses.

The maximum increase of the mean square error in macroinvertebrate metric predictions after permuting each predictor variable varies between approximately 4% and 8% for the metrics related to flow and substrate type preference (examples for LIFE, RHEO and PSAM in Figure 4.39.

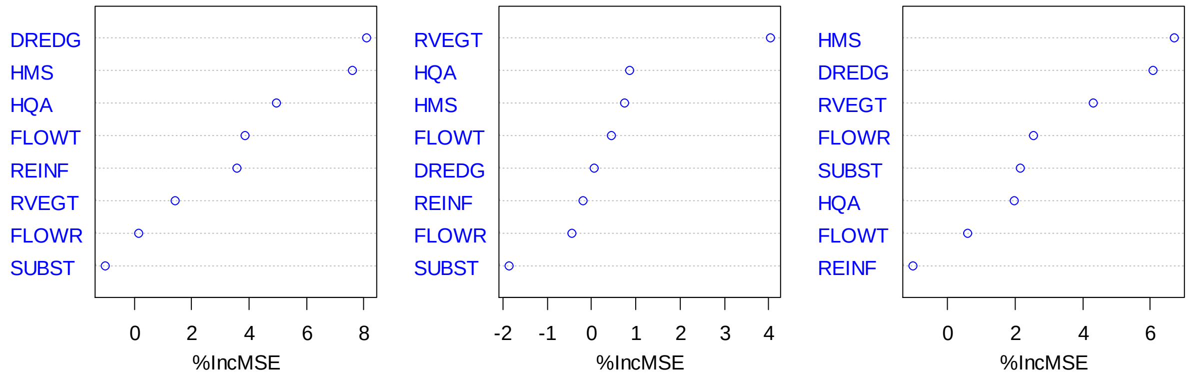


Figure 4.39. Importance of predictor variables in random forest models (left: LIFE; centre: RHEO; right: PSAM) **(United Kingdom).**

Generalised linear models were constructed to predict macroinvertebrate metrics designed to respond to hydromorphological pressures (LIFE, RHEO and PLAL) plus ASPT. HMS and its components were selected to be used as predictors, as they were expected to be related with these biological metrics a priori. HMS and HQA were used as explanatory variables in a second round of modelling.

As expected, models explain a low variance in the data, with a maximum deviance of 13% (Table 4.17). Specifically, no significant models were obtained for RHEO, nor for PLAL when HMS and HQA were used as predictors.

Table 4.17. Generalised linear models to predict macroinvertebrate metrics from hydromorphological indicators (United Kingdom).

Predictors	**Inv. metric**	**Model**	**N**	**D^2**
HMS & HMS components	LIFE	$= e^{(1.959\pm9.782) - (3.880\pm1.117)\ \mathrm{DREDG}}$	88	13%
	RHEO	-	-	-
	PLAL	$= \dfrac{100\ e^{-(1.896\pm9.656) + (1.958\pm8.706)\ \mathrm{DREDG}}}{1+ e^{-(1.896\pm9.656) + (1.958\pm8.706)\ \mathrm{DREDG}}}$	87	5%
	ASPT	$= e^{(1.772\pm1.798) - (4.673\pm2.071)\ \mathrm{DREDG}}$	88	6%
HMS & HQA	LIFE	$= e^{(1.970\pm1.179) - (2.972\pm8.807)\ \mathrm{HMS}}$	88	12%
	RHEO	-	-	-
	PLAL	-	-	-
	ASPT	$= e^{(1.600\pm0.062) + (0.003\pm0.001)\ \mathrm{HQA}}$	88	7%

Summary and conclusions of national inter-comparison

Three datasets from the national monitoring programmes of Denmark, Spain and the United Kingdom were analysed independently to quantify the relationships between macroinvertebrate metrics and between hydromorphological pressures and conditions in multi-pressure environments.

The majority of relationships that could have been expected from the literature between hydromorphological pressure and condition indicators were not identified as significant. In turn, correlations between hydromorphological and macroinvertebrate indicators were in general low to moderate, with maximum Spearman correlation coefficients of approximately 0.55 in absolute value. Generalised linear models constructed to predict macroinvertebrate metrics exclusively from hydromorphological indicators explained a low to moderate variance in the data, with a maximum of approximately 40% of deviance when using a combination of hydromorphological indicators as predictors. Hydromorphological indicators also showed a poor predictive capacity when using random forests.

The results obtained illustrate the fact that indicators currently monitored by Member States generally have severe limitations for valuation of the hydromorphological conditions of waterbodies and the corresponding ecological status, regarding both hydromorphological and macroinvertebrate metrics. THus, they exemplify the need to derive new indicators of hydromorphological degradation to be implemented for the review of the first round of river basin management plans developed to comply with the Water Framework Directive. One of the most important limitations of the analysed datasets is that they lacked a comprehensive quantitative description of the extent and intensity of individual hydromorphological pressures.

Some of the strongest relationships identified were between macroinvertebrate metrics that measure the presence of taxa with a preference for specific flow or substrate type conditions and between indicators of these specific conditions, which reflects the potential of using trait-based metrics to evaluate hydromorphological conditions. However, macroinvertebrate metrics designed to respond to hydromorphological pressures did generally not show greater sensitivity to related hydromorphological indicators than metrics designed to respond to pollution or general degradation. Despite the abundant literature on macroinvertebrates, the knowledge on how specific taxa relate to hydromorphological conditions and respond to changes in these is still limited, especially when factoring in other pressures. Experimental research including the monitoring of river restoration programmes with before-after-control-impact studies is urgently needed to improve the understating of the underlying mechanisms of macroinvertebrate-hydromorphology relationships.

4.3.5 Summary and conclusions

None of the relationships between hydromorphological degradation and macroinvertebrate metrics were very strong. Metrics developed to detect hydrological impairment and hydromorphological degradation were not more discriminative than a number of metrics sensitive to other pressures. These findings leave water managers with a significant challenge when diagnosing the reason for not obtaining good ecological status in a waterbody. The reasons for the lack of sensitivity can most likely be attributed to a number of explanatory variables not being measured as part of routine monitoring programmes or in the currently used hydromorphological assessment schemes, which do not necessarily register elements of importance to the in-stream biota. In addition, the present findings suggest that both metric development and sampling scales need to be scrutinised to improve sensitivity, if possible.

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4.4 Fish

4.4.1 Introduction

In this section the effects of hydromorphological (HYMO) pressures on fish community composition and abundance are assessed. The objective is to identify the HYMO pressures determining at least one of these fish community traits.

4.4.2 Methods

Due to the early state of the study and the heterogeneous nature of the data, an exploratory analysis was planned. In this context, the main methodological criterion is "to keep it simple" and search for results from which to obtain straightforward conclusions; Therefore, the response variable is the presence/absence of a species. The explanatory variables are selected from the set of HYMO pressures considered in WP 1.2.

Previously, the database uploaded in the REFORM ftp was homogenised to the lowest level of information of all the datasets from which the database is based. The database includes datasets from Denmark, Finland, Netherlands, Spain, Sweden and WISER (Table 4.18). The HYMO pressures identified in the datasets have been reclassified into the HYMO pressure types of WP 1.2: (1) Effects of water abstractions; (2) Effects of flow regulation; (2.1) Increased flow; (2.2) Flow regime modification timing; (2.3) Hydropeaking; (3) Effects of river fragmentation; (4) Effects of morphological alterations; (4.1) Impoundment; (4.2) Large dam and reservoir; (4.3) Channelisation/Cross section alteration; (4.4) Channelisation/Meander realignment; (4.5) Alteration of riparian vegetation; (4.6) Alteration of in-stream habitat; (4.7) Embankments. levees or dikes; (4.8) Sedimentation; (4.9) Sand and gravel extraction; (4.10) Loss of vertical connectivity; (5) Effects of other pressures. Since the lowest level of information is presence/absence of the pressure, the HYMO pressures have been coded as (1=presence, 0=absence, no data) for all observations, whereas the WISER dataset classified pressures into four categories according to intensity: none, low, intermediate, high. No data were available on Increased Flow, Flow Regime modification; Timing; Loss of vertical connectivity; and Effects of other pressures, and these variables were consequently removed from the analysis.

Table 4.18. Summary table of the provided data sorted by countries. Data of Austria, France, Germany and Netherlands (partially) were provided by WISER dataset.

	Country	Austria	Denmark	Finland	France	Germany	Netherlands	Spain	Sweden	Total
	No. sites	827	135	75	325	1,068	65	207	94	2,796
	No. occasions	950	237	75	325	1,294	70	207	94	3,252
	No. taxa	75	33	26	53	55	39	26	14	119
No. observations	4.1 Effects of Water abstractions	743	0	0	0	54	0	0	0	797
	4.2 Effects of Flow regulation	0	9	72	0	0	0	0	0	81
	4.2.1 Increased Flow	0	0	0	0	0	0	0	0	0
	4.2.2 Flow Regime modification: timing	0	0	0	0	0	0	0	0	0
	4.2.3 Hydropeaking	743	0	0	0	0	0	0	0	743
	4.3 Effects of River fragmentation	743	8	0	0	469	15	39	0	1,274
	4.4 Effects of Morphological alterations:	-	-	-	-	-	-	-	-	-
	4.4.1 Impoundment	743	0	74	0	433	0	0	0	1,250
	4.4.2 Large Dam and Reservoir.	0	0	0	0	0	0	14	0	14
	4.4.3 Channelisation / Cross section alteration	743	8	74	0	420	15	207	0	1,467
	4.4.4 Channelisation / Meander Realigment	743	9	73	0	475	15	0	0	1,315
	4.4.5 Alteration of riparian vegetation	743	9	72	0	483	15	0	0	1,322
	4.4.6 Alteration of instream habitat	743	0	74	0	368	2	207	0	1,394
	4.4.7 Embankments, levees or dikes	743	9	0	0	272	2	207	0	1,233
	4.4.8 Sedimentation	0	0	0	0	0	0	0	26	26
	4.4.9 Sand and gravel extraction	0	0	72	0	0	0	0	0	72
	4.4.10 Loss of vertical connectivity	0	0	0	0	0	0	0	0	0
	4.5 Effects of other pressures	0	0	0	0	0	0	0	0	0

Two consecutive logistic regression analyses were performed to find the HYMO pressures that significantly explain the probability of occurrence of every fish species. The logistic regression is a statistical procedure that allows prediction of the probability of occurrence of an event as a function of a set of explanatory variables. Therefore, the response variable is the occurrence (value = 1) or non-occurrence (value = 0) of the event. In this case, the event is the presence (1) or absence (0) of a given fish species.

Analysis 1 is a Generalized Linear Model (GLM) where a logistic curve ($Y = [\exp(a + bX)] / [1 + \exp(a + bX)]$) is fitted to the relationship between the explanatory variable and the response variable. This logistic regression is usually made to fit a regression curve when the response variable consists of binary codes (1=presence; 0=absence) data. This first analysis was made univariate. In order to simplify the fitting of the model, this particular analysis was conducted using the HYMO pressures (1, 0, no data) as response variables and species as categorical variables to be selected depending on the statistical significance (>90% c.l.). This is not a problem since the causality of the relation between HYMO pressures and the presence/absence of a fish species is obvious.

Analysis 1 allows pre-selection of a set of independent variables to be used in the second analysis. In Analysis 2, a multivariate logistic regression was fitted between presence/absence (response variable) for every species with a significant (>90% c.l.) relation with HYMO pressures (resulting from Analysis 1) and the HYMO pressures to which it was found to be related (explanatory variables). To undertake this analysis, the absence of a given species out of its geographical area of distribution (Kotellat and Freyhof 2007) was consigned as missing data in the observations dataset. Significant HYMO pressures were selected by means of a stepwise procedure. The set of significant pressures was selected by the Akaike Information Criteria (AIC) using a backward stepwise procedure. To fit the models on the same dataset, all missing data were removed from the dataset of pre-selected explanatory variables and the response variable.

All statistical analyses were conducted in R (R Core Team, 2013).

Once the exploratory results are outlined, a more focused analysis should be undertaken (as response variables, for instance: abundance, diversity, guilds composition, etc.; and explanatory variables: multivariate variables, including mesological, geographical, variables)

4.4.3 Results

The species that are significantly (>90% c.l.) related to HYMO pressures by means of a logistic model (Analysis 1) are shown in Table 4.18. Up to 60% of fish species (65 out of 108) have been found significantly related to the absence/presence of considered HYMO pressures.

Table 4.19. Summary of the fitting of fish species (explanatory variable) significantly related to presence/absence of HYMO pressures (response variable) by means of a logistic regression (Analysis 1): estimates and p-values. (*) *Gymnocephalus cernua* and *G. cernuus* to be merged.

	River fragment.		Impoundment		Cross sect. Alt.		Meander Realig.		Alt. riparian veg.		Alt. instream hab.		Emb., levees, dikes	
Species	**Estimate**	**Pr(>|z|)**	**Estimate**	**Pr(>|z|)**	**Estimate**	**Pr(>|z|)**	**Estimate**	**Pr(>|z|)**	**Estimate**	**Pr(>|z|)**	**Estimate**	**Pr(>|z|)**	**Estimate**	**Pr(>|z|)**
(Intercept)	1.204	2.30E-07	-0.614	0.004	2.091E+00	6.08E-11	1.247	1.37E-07	1.204	2.30E-07	3.868E-01	0.073	4.357	1.50E-05
Achondrostoma arcasii					-4.048E+00	< 2E-16					-2.009E+00	8.28E-09	-4.754	3.72E-06
Alburnoides bipunctatus	-0.499	0.098	-1.925	2.41E-06	-9.147E-01	0.017	-0.945	0.001						
Alburnus alburnus					-1.521E+00	1.51E-05	-0.857	0.003			-4.260E-01	0.098	-2.735	0.008
Anguilla anguilla	1.256	0.000	-0.755	0.005			1.051	0.001	0.636	0.032	-1.499E+00	3.06E-08		
Aspius aspius	-0.916	0.027									8.862E-01	0.064		
Ballerus sapa			1.531	0.077										
Barbatula barbatula			-1.539	1.45E-09	-7.574E-01	0.024					-1.345E+00	1.19E-08	-2.049	0.045
Barbus barbus			-1.151	6.66E-05										
Barbus bocagei					-4.025E+00	< 2E-16					-1.709E+00	2.58E-07	-4.806	2.94E-06
Carassius auratus					-1.803E+00	0.000							-3.440	0.001
Carassius carassius			-1.044	0.075			1.932	0.065			-1.175E+00	0.043		
Carassius gibelio			-0.936	0.010			0.964	0.029						
Cobitis calderoni					-5.086E+00	2.15E-06					-1.834E+00	0.002	-5.050	5.15E-06
Cobitis elongatoides	-1.966	0.000	-2.430	0.020										
Cobitis paludica					-3.295E+00	6.72E-06					-1.591E+00	0.022	-3.153	0.009
Cobitis taenia	1.435	0.011					1.625	0.011	2.803	0.007	-7.095E-01	0.048		
Cobitis vettonica													-4.357	0.012
Cottus gobio			-1.911	3.17E-13	-1.420E+00	1.68E-05	-1.092	1.11E-05			-9.566E-01	3.39E-05	-2.229	0.028
Cyprinus carpio					-1.105E+00	0.004							-2.618	0.013
Esox lucius					-9.974E-01	0.005					-7.716E-01	0.002	-2.362	0.022
Eudontomyzon mariae	-1.609	0.002	-1.583	0.041	-1.472E+00	0.009	-2.094	0.000						
Gambusia holbrooki											-1.437E+00	0.003	-5.050	3.32E-06
Gasterosteus aculeatus	0.618	0.030	-0.838	0.001	1.770E+00	0.001	1.462	1.27E-05	0.469	0.094	-1.645E+00	9.50E-10		
Gobio gobio			-1.105	9.48E-06							-9.800E-01	3.55E-05		
Gobio lozanoi					-4.054E+00	< 2E-16					-2.152E+00	6.18E-11	-4.762	3.18E-06
Gymnocephalus cernua			-0.595	0.090					0.702	0.078	-1.101E+00	0.001		
Gymnocephalus cernuus					-4.037E+00	0.000								
Gymnocephalus schraetser	-3.401	0.002	1.462	0.043										
Hucho hucho	-1.070	0.005	-1.077	0.020			-0.748	0.054			1.693E+00	0.001		
Lampetra fluviatilis			1.308	0.044										
Lampetra planeri	1.281	0.008	-1.349	0.002					0.756	0.065	-3.942E+00	1.41E-07		
Lepomis gibbosus			-1.332	0.090	-3.027E+00	2.80E-13					-1.058E+00	0.002	-4.502	1.36E-05
Leucaspius delineatus	1.322	0.086	-1.177	0.074							-1.773E+00	0.003		
Leuciscus cephalus					-3.189E+00	0.008	-2.345	0.047	-2.303	0.051				
Leuciscus idus	-0.779	0.045					1.005	0.081						
Leuciscus leuciscus			-1.143	3.72E-05							-1.032E+00	4.91E-05		
Lota lota	-1.552	1.15E-05			-1.824E+00	2.72E-06	-0.881	0.007	-0.738	0.023				
Micropterus salmoides											-1.485E+00	0.079	-5.455	2.56E-05
Neogobius kessleri	-2.526	3.36E-05									1.287E+00	0.053		
Neogobius melanostomus			1.084	0.075										
Oncorhynchus mykiss			-1.380	8.39E-08	-1.130E+00	0.001	-0.990	9.44E-05					-2.549	0.012
Perca fluviatilis			-0.593	0.017	-8.924E-01	0.009	-0.561	0.030			-1.141E+00	3.09E-06		
Phoxinus bigerri											-1.913E+00	0.000	-6.149	7.31E-08
Phoxinus phoxinus			-1.251	9.39E-06							-9.464E-01	0.000	-1.954	0.060
Proterorhinus marmoratus	-1.358	0.024												
Proterorhinus semilunaris	-1.291	0.007					1.845	0.079			1.171E+00	0.047		
Pseudochondrostoma duriense					-4.057E+00	3.26E-15					-1.818E+00	5.27E-06	-5.130	9.37E-07
Pseudochondrostoma nasus	-0.777	0.010			-8.715E-01	0.025	-0.782	0.010			7.354E-01	0.016		
Pseudorasbora parva			-2.553	4.10E-06										
Pungitius pungitius	1.302	0.022							1.995	0.009	-2.926E+00	4.43E-06		
Rhodeus amarus			-0.754	0.041					1.209	0.013				
Romanogobio vladykovi	-1.204	0.007	-1.951	0.011							1.733E+00	0.007		
Rutilus pigus	-1.715	0.025												
Rutilus rutilus			-0.591	0.016	-6.461E-01	0.060					-9.990E-01	3.39E-05		
Salmo trutta fario			-1.506	5.18E-10	-9.766E-01	0.003	-0.792	0.001			-7.461E-01	0.001	-2.349	0.020
Salmo trutta trutta			-1.159	0.006	-4.508E+00	< 2E-16	-3.038	1.76E-12	-2.205	9.74E-10	-2.402E+00	7.90E-14	-5.050	8.33E-07
Salvelinus fontinalis			-1.935	0.000	-1.264E+00	0.002	-1.043	0.002					-2.277	0.036
Salvelinus umbla							-2.345	0.047						
Silurus glanis			0.781	0.092										
Squalius alburnoides					-3.700E+00	1.77E-07					-1.996E+00	0.003	-4.580	3.85E-05
Squalius carolitertii					-5.919E+00	2.36E-08					-1.695E+00	4.75E-05	-4.745	6.15E-06
Squalius cephalus			-0.997	5.12E-05							-5.895E-01	0.012		
Telestes souffia	-1.291	0.007			-1.049E+00	0.067	-1.334	0.005						
Thymallus thymallus			-1.317	5.44E-07	-8.900E-01	0.009	-0.857	0.001					-2.236	0.029
Tinca tinca	0.663	0.058									-1.187E+00	0.000	-2.291	0.032
Vimba vimba	-1.715	0.000									1.223E+00	0.038		
Zingel zingel	-3.283	0.002												
No fish													-4.357	0.012
Null deviance	9439.3		7650.7		10490.7		10781		9042.7		11780		5932.7	
degrees of freedom	8561		8335		9248		8737		8804		8725		7636	
Residual deviance	8837.3		7162.5		8313.2		10043		8758.7		10779		4541.8	
degrees of freedom	8467		8254		9150		8651		8718		8627		7541	
AIC	9027.3		7326.5		8511.2		10217		8932.7		10977		4733.8	

Starting from the relations among fish species and HYMO pressures, Analysis 2 show the effect of HYMO pressures on the probability of occurrence of every fish species shown in Table 4.19. The results of Analysis 2 are presented in Table 4.20.

Table 4.20. Summary of the fitting of HYMO pressures (explanatory variable) to presence/absence of every fish species (response variable) by means of a logistic regression (Analysis 2): estimates and p-values. Relationships with significance greater than 99% c.l. are highlighted in bold letters.

Family	Species	(Intercept)		River fragment.		Impoundment		Cross sect. Alt.		Meander realig.		Alt. Ripar. veg.		Alt. Instr. hab.		Emb., lev., dikes		Residual deviance	Degrees of freedom	AIC
		Est.	Pr(>|z|)	Est.	Pr(>|z|)	Est.	Pr(>|z|)	Est.	Pr(>|z|)	Est.	Pr(>|z|)	Est.	Pr(>|z|)	Est.	Pr(>|z|)	Est.	Pr(>|z|)			
Petromyzonidae	*Eudontomyzon mariae*	-4.46	<2E-16	-0.94	0.040					-1.17	0.017							247.8	4646	254
	Lampetra fluviatilis	-6.66	<2E-16			**2.42**	***8.37E-005***											145.3	3685	149
	Lampetra planeri	-6.60	<2E-16	1.22	0.042							1.35	0.024	**-3.63**	***3.36E-004***			516.8	7011	525
Anguillidae	*Anguilla anguilla*	-5.87	<2E-16	**0.98**	***0.001***	**0.54**	***0.006***			**1.45**	***8.15E-007***	0.53	0.037	**-1.10**	***3.23E-008***			1490.5	6972	1503
Cyprinidae	*Alburnoides bipunctatus*	-3.58	<2E-16			**-0.94**	***0.007***			**-0.67**	***2.64E-004***							1243.3	7802	1249
	Alburnus alburnus	-5.27	1.24E-013											0.40	0.028	1.51	0.036	1289.6	5984	1298
	Aspius aspius	-5.83	<2E-16	-0.68	0.060									**1.32**	***0.002***			394.1	7318	400
	Ballerus sapa	-8.09	<2E-16			**2.46**	***0.003***											102.7	7949	107
	Carassius auratus	-4.58	<2E-16													-0.94	0.013	437.9	7250	442
	Carassius carassius	-7.80	6.24E-015							2.01	0.052							221.4	7365	225
	Carassius gibelio	-5.72	<2E-16							**1.35**	***3.14E-004***							882.0	8215	886
	Gobio gobio	-2.65	<2E-16											**-0.34**	***0.001***			3296.8	7479	3301
	Gobio lozanoi	-1.75	<2E-16											-0.51	0.081			722.7	845	729
	Leucaspius delineatus	-7.04	5.11E-012	1.75	0.090									-1.05	0.061			263.9	7079	270
	Leuciscus idus	-6.49	<2E-16	**-0.93**	***0.004***					**2.08**	***0.004***							484.2	8164	490
	Leuciscus leuciscus	-3.32	<2E-16											-0.34	0.015			2041.6	7479	2046
	Phoxinus bigerri	-3.06	<2E-16													-1.28	0.019	238.8	846	243
	Pseudochondrostoma nasus	-4.42	<2E-16					**-0.68**	***0.003***					**1.40**	***4.49E-010***			1117.0	7002	1123
	Pseudorasbora parva	-4.27	<2E-16			**-1.61**	***0.002***											1058.0	8334	1062
	Rhodeus amarus	-5.54	<2E-16									0.94	0.029					785.1	7877	789
	Romanogobio vladykovi	-6.66	<2E-16			-1.53	0.038							**2.37**	***1.29E-004***			332.7	7078	341
	Rutilus rutilus	-2.90	<2E-16			**0.57**	***4.95E-005***							**-0.52**	***1.72E-005***			2783.6	7367	2790
	Squalius cephalus	-2.86	<2E-16									0.21	0.090					3464.0	7347	3468
	Telestes souffia	-4.62	<2E-16	-1.01	0.017					-0.96	0.022							301.9	7805	308
	Tinca tinca	-20.06	0.969	0.70	0.042													675.2	6362	681
	Vimba vimba	-6.01	<2E-16	**-1.29**	***0.003***									**1.53**	***0.006***			298.5	7318	304
Cobitidae	*Cobitis elongatoides*	-4.66	<2E-16	-1.18	0.011													268.4	4646	274
	Cobitis paludica	-5.18	<2E-16													1.77	*0.007*	125.6	846	130
	Cobitis taenia	-5.85	5.22E-009											**1.95**	***4.78E-010***			404.7	2555	411
Nemacheilidae	*Barbatula barbatula*	-2.58	<2E-16			**-0.56**	***0.003***	0.28	0.048					**-0.61**	***2.82E-007***			2717.4	5955	2725
Siluridae	*Silurus glanis*	-6.39	<2E-16			**1.72**	***2.86E-005***											310.1	7949	314
Salmonidae	*Hucho hucho*	-6.23	<2E-16											**2.26**	***2.77E-006***			501.5	6974	509
	Oncorhynchus mykiss	-1.53	<2E-16			**-0.48**	***0.002***			**-0.69**	***1.58E-011***					**-0.69**	***5.64E-006***	3034.7	6338	3043
	Salmo trutta	-1.03	2.31E-016			**-0.71**	***2.80E-007***	-0.27	0.027	**-0.44**	***2.75E-005***			0.21	0.026	**-0.62**	***9.12E-006***	4091.8	5953	4104
	Salvelinus fontinalis	-4.16	<2E-16			-0.88	0.061			-0.56	0.029							697.4	6339	703
	Salvelinus umbla	-6.64	<2E-16							-2.04	0.078							65.2	8181	69.2
Thymallidae	*Thymallus thymallus*	-2.17	<2E-16			-0.36	0.035			**-0.53**	***5.12E-006***					-0.40	0.034	2543.3	6338	2551
Lotidae	*Lota lota*	-4.58	<2E-16	**-1.72**	***2.56E-009***					0.69	0.052							598.2	7975	604
Gasterosteidae	*Gasterosteus aculeatus*	-6.35	<2E-16			0.44	0.020	1.31	0.039	**2.25**	***5.40E-006***			**-1.35**	***1.21E-013***			1637.2	6971	1647
	Pungitius pungitius	-21.82	0.9770											**-2.33**	***0.002***			330.6	7250	339
Cottidae	*Cottus gobio*	-1.57	<2E-16			**-1.11**	***3.40E-009***	-0.24	0.095	**-0.42**	***0.001***					**-0.44**	***0.008***	3220.6	5953	3233
Centrarchidae	*Lepomis gibbosus*	-7.20	<2E-16											1.88	0.015			187.6	5956	194
	Micropterus salmoides	-5.06	<2E-16													**-2.99**	***2.54E-004***	108.9	7204	113
Percidae	*Gymnocephalus cernua*	-4.41	<2E-16			0.72	0.017							**-0.98**	***0.001***			794.8	7351	801
	Gymnocephalus schraetser	-6.46	<2E-16	**-3.55**	***0.001***	**2.75**	***7.49E-005***											118.0	7913	124
	Perca fluviatilis	-2.78	<2E-16			**0.67**	***5.31E-006***			-0.26	0.029			**-0.71**	***5.43E-008***			2613.9	7361	2622
	Zingel zingel	-5.41	<2E-16	-2.55	0.016													120.5	4647	124
Gobidae	*Neogobius kessleri*	-5.93	<2E-16	**-2.10**	***2.24E-004***									1.44	0.024			234.2	7289	240
	Neogobius melanostomus	-7.18	<2E-16			**2.02**	***4.05E-004***											180.4	7949	184
	Proterorhinus marmoratus	-5.67	<2E-16	-1.26	0.023													188.7	8218	193
	Proterorhinus semilunaris	-8.10	1.12E-012	-0.80	0.066					2.21	0.031			1.46	0.011			283.7	7188	292
	Mean estimate	-5.59		-0.84		0.32		0.08		0.25		0.75		0.12		-0.45				
	SD estimate	3.58		1.37		1.36		0.69		1.33		0.43		1.53		1.34				
	% negative estimate	100%		75%		47%		60%		59%		0%		52%		78%				
	% positive estimate	0%		25%		53%		40%		41%		100%		48%		22%				
	No. cases >90% c.l.	49		16		19		5		17		4		23		9				
	No. cases >95% c.l.	49		13		18		4		14		3		21		9				
	No. cases >99% c.l.	49		6		14		1		9		0		15		5				

4.4.4 Discussion

A rough validation test of this approach is to check whether ecologically and taxonomically similar species with separate distribution (which assures that the dataset from which they come are independent) show similar responses to a given HYMO pressure. In Analysis 1 (Table 4.19) *Phoxinus phoxinus* and *P. biguerri*, both show identical response to the alteration of in-stream habitat. *Salmo trutta trutta, Salmo trutta fario* (which in this case are wrongly consigned as different subspecies, since they represent different life cycles of the same species *Salmo trutta*)and *Salvelinus fontinalis* share a common response to several -showing a high sensitiveness to- HYMO pressures.

The results of Analysis 2 (Table 4.20) are sorted by families, so it is easier to notice whether species with similar ecological requirements (Noble et al. 2007) share common responses to a given HYMO pressure. For instance, Salmonidae are quite sensitive to HYMO pressures and four out of five species show the same negative effect to the presence of impoundments and meander realignment. Three out of four Percidae show a positive response to the presence of impoundment, instead. Both noticeable effects remark the rheophilic character of Salmonidae and the limnophilia of Percidae. As it could be expected, *Silurus glanis* shows a positive response to impoundment, as well. Gobiidae is the most sensitive familie to river fragmentation (Figure 4.40). Due to its species diversity, Cyprinidae show a rather heterogeneous response to several HYMO pressures. For instance, genus Carassius (i.e. *C. carassius* and *C. gibelio*) share a positive response to meander realignment, whereas *Alburnoides bipunctatus* and *Telestes souffia* response negatively to the same pressure.

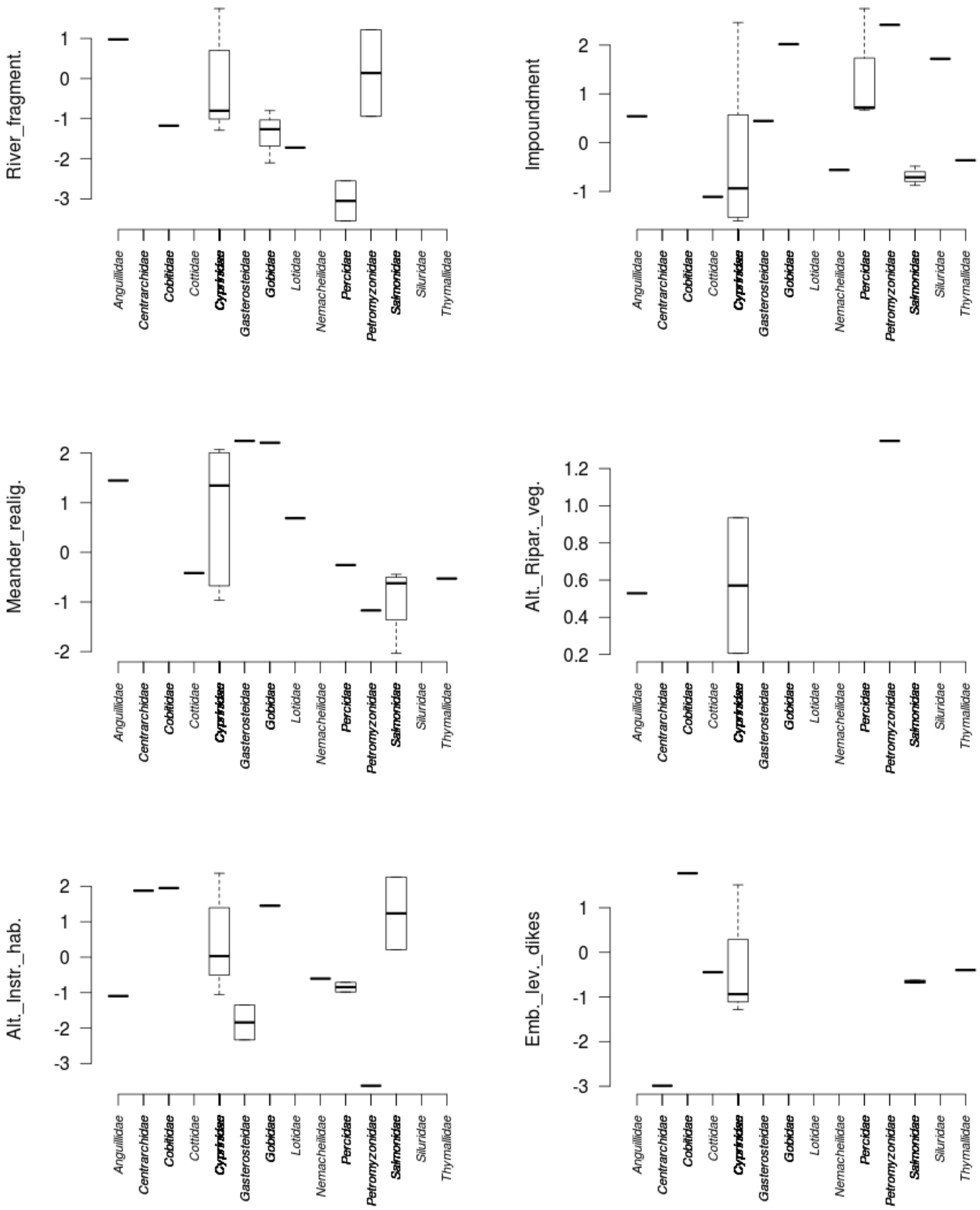

Figure 4.40. Estimates of the significant (>90% c.l.) HYMO pressures on freshwater fish families obtained from logistic regression analyses.

Alteration of in-stream habitat and impoundment are the HYMO pressures that cause the most widespread responses (on 23 and 19 species, respectively). However, the sign of the response is not clearly negative or positive; nearly 50% of species show a positive response to the presence of both HYMO pressures. However, almost 80% of species whose presence responses significantly to embankments, leeves or dikes show a negative response, this pressure is followed by river fragmentation (75%) (Figure 4.41).

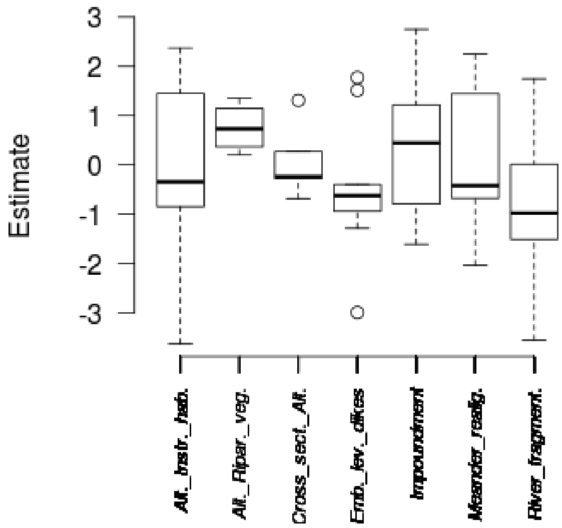

Figure 4.41. Estimates of the significant (>90% c.l.) relations among presence/absence of HYMO variables and presence/absence of fish species.

These results show that 49% of the studied European freshwater species show a significant (>90% c.l.) response to HYMO pressures. However, there are still 52 fish species that show no response to these pressures. Among the latter are eurytopic native taxa like genus Barbus and Squalius, Cyprinus carpio and Esox lucius, and alien taxa like genus Ameiurus and Gambusia, Hypophthalmichthys molitrix and Lepomis gibbosus.

As it has been said, this is an exploratory analysis. It is therefore necessary to undertake a much more detailed test. However, it is a good starting point to set the focus of further studies.

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4.5 Multiple biological quality elements

4.5.1 Aim

In this chapter we investigate; 1) relationships of status estimates of multiple Biological Quality Elements (BQEs) with each other and 2) their response to hydromorphological degradation in a multi-pressure river environment.

Data

In this preliminary exercise, we used biomonitoring data from Finland including multiple biotic groups sampled from same sites. Later work will explore the patterns across a larger geographical extent in Europe using the common REFORM ftp database. We used biological and environmental monitoring data on diatoms, invertebrates and fish sampled mainly in years 2006-2012 from 150 river sites across Finland. Ninety-four sites had all three biotic groups sampled, 40 had two groups and 16 one group sampled. All sites are fast-flowing river riffle reaches in lowland (median 80 m a.s.l.), and the streams are mainly mid-sized (median catchment area 196 km^2) and located in both peatland and mineral land catchments.

The sites include approximately 50 national biomonitoring sites with diffuse loading from agriculture and forestry, the two most widespread anthropogenic pressures in Finland. About 100 sites in the dataset are other national monitoring sites which mostly represent regional least impacted or reference conditions. The data thus covers a wide gradient of least disturbed to altered rivers in Finland.

For each of the three BQEs, we calculated status indices based on bioassessment systems and classification criteria used for the planning of the WFD 2nd RBMPs in Finland (Aroviita et al. 2012). In this national system reference conditions for all metrics are river type-specific and based on a larger pool of least-impacted reference sites across Finland. Status class boundaries for all metrics are based on deviation from the reference values. The status of diatom communities was assessed by mean EQR over EQR values of two indices: occurrence of type-specific taxa (Aroviita et al. 2008) and occurrence and a Percent Model Affinity (Novak & Bode 1992). The status of benthic invertebrate communities was assessed by the mean EQR value over two of the above-mentioned indices and the occurrence of type-specific EPT-families. Fish communities were assessed with a multimetric index named Finnish Fish Index (FiFI, Vehanen et al. 2010).

To allow comparability of BQEs, all metric EQRs were linearly re-scaled so that at the re-scaled EQR scale class boundaries of bad, poor, moderate and good classes equalled EQR-values 0.2, 0.4, 0.6 and 0.8. If the same site was sampled in multiple years, a mean EQR over the multiple sampling occasions was used for the site.

We first correlated the BQE status estimates (i.e. EQRs) with each other to explore their relationships. Then, to explore how the BQE status estimates were attributable to the measured intensity of human disturbance, we correlated the EQRs variables with measures of anthropogenic degradation. Environmental data included measures of land use (Corine), standard water chemistry measurements, River Habitat field Surveys (RHS) and field surveys of hydromorphological degradation. We conducted a varimax-rotated PCA with the aim to extract main, simplified gradients in the environmental data. Last, to explore whether the EQRs were attributable to the measured intensity of human disturbance gradients, we conducted a stepwise multiple linear regression (variable entry if $p < 0.05$, removal if $p > 0.1$) to the PCA gradients.

4.5.2 Results

The EQRs of the BQEs correlated significantly with each other (Figure 4.42). The strength of the correlations was, however, relatively low (r=0.3-0.5). The varimax-rotated PCA extracted four main gradients for the data (Table 4.21). These were PCA1 related to agriculture (nutrient concentrations and agricultural land use), PCA2 related to hydromorphological degradation (HMS from RHS, intensity of dredging), PCA 3 related to urban land use and pH, and PCA4 related to naturalness of the habitat (riparian canopy cover and amount of coarse wood in the streams).

A considerable part of the variation in EQR values of all the BQEs was attributable to the observed intensity of human disturbance. The BQE EQRs correlated strongly with the PCA1 (agriculture gradient) (Table 4.22). However, all other correlations were low and mostly not significant. None of the BQEs were related to the measured morphological alterations (PCA2). In the multiple linear regression, only PCA1 explained variation of the EQRs. Adding of other PCA-axes to the stepwise multiple linear regressions did not increase the explanatory power.

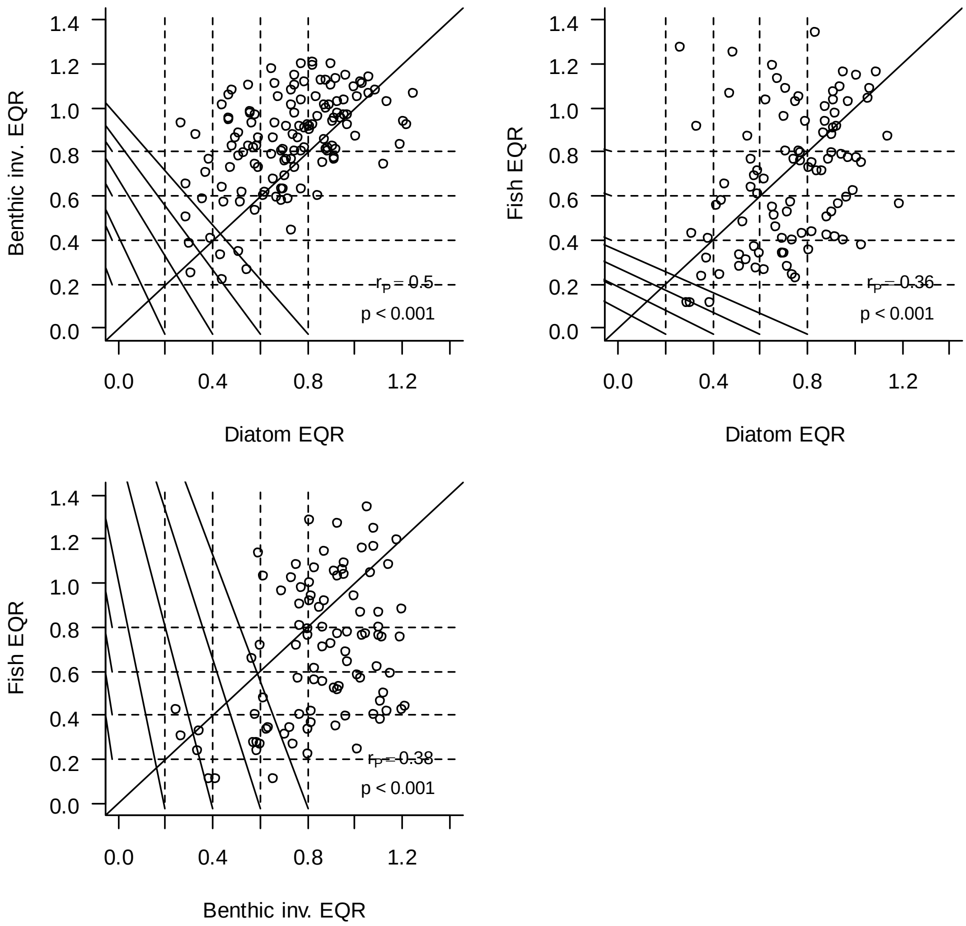

Figure 4.42. Relationship between diatom, benthic invertebrate and fish EQRs in Finnish streams. Pearson correlation coefficient is shown in each panel. The dashed lines show the status class boundaries. 1:1 line is also shown.

Table 4.21. Varimax rotated principal component loadings for a selected set of pressures and stressors in 149 river sites in the Finnish dataset. Bold values indicate the highest loading of each variable to the components. HMS is Habitat Modification Score from River Habitat Survey (RHS), HQA_Adj is Adjusted Habitat Quality Assessment score from RHS where higher scores represent more natural sites.

	PCA1	PCA2	PCA3	PCA4
Eigenvalue	2.99	1.989	1.717	1.24
% of variance	24.9	16.6	14.3	10.3
Total nitrogen	**0.913**	0.046	-0.009	-0.184
Total phosphorus	**0.856**	-0.05	-0.048	-0.134
Fields (%)	**0.713**	0.078	0.547	-0.213
HMS-index	0.081	**0.849**	-0.069	0.034
Dredging intensity	-0.083	**0.794**	-0.134	-0.133
Straightening intensity	-0.021	**0.652**	0.206	0.068
Amount of artificial features	0.369	**0.431**	0.104	0.112
pH	-0.185	-0.057	**0.896**	-0.042
Urban land (%)	0.414	0.125	**0.749**	0.160
Riparian canopy cover (%)	0.049	0.035	-0.042	**0.772**
Amount of coarse wood	-0.132	-0.082	0.065	**0.724**
HQA_Adj-index	-0.242	0.086	0.006	**0.628**

Table 21. Spearman correlation coefficients of EQRs of diatoms, benthic invertebrates and fish, and the mean EQR of the three groups, against the PCA-gradients. Significant coeffcients are given in bold.

	EQR Diatoms	EQR Macroinvert.	EQR Fish	Mean EQR
N	132	139	96	91
PCA1 (”agriculture”)	**-0.550**	**-0.563**	**-0.488**	**-0.670**
PCA2 (”morphological alteration”)	-0.060	0.023	-0.067	-0.025
PCA3 (”urban”)	-0.064	-0.086	0.154	0.151
PCA4 (”naturalness”)	**0.186**	0.097	0.065	0.121

4.5.3 Discussion

The Water Framework Directive requires the use of multiple biotic groups in the assessment and monitoring of the status of European rivers. However, information still remains inconclusive as to the response of different taxonomic groups to different anthropogenic stressors (e.g. Johnson & Hering 2009). We found relatively low, albeit clearly significant, correlations between assessments of diatom, macroinvertebrate and fish communities in the Finnish rivers. We also found some differences in the response signals –benthic invertebrate assemblages were on average in better condition than diatom or fish assemblages. Thus, similar environmental changes may not be similarly detrimental to different taxonomic groups. These results indicate that multiple biotic groups are not redundant but provide complementary information (see also e.g. Heino et al. 2005; Mykrä et al. 2008; Johnson et al. 2013) in bioassessment.

A considerable part of the variation of the EQRs of all BQEs was attributable to the measured intensity of agriculture, whereas, in contrast, correlation with the intensity of morphological alterations was insignificant in all biotic groups. The results indicate that in Finnish rivers biotic degradation due to agricultural activities is more severe than that caused by morphological alterations only. Loss of habitat heterogeneity by historical morphological alterations which would limit, for instance, benthic invertebrate communities may have been only partial (Louhi et al. 2010). The potential biotic impairment due to morphological alterations may be subtle and possibly also ‘overruled’ by the much stronger agricultural gradient in the dataset. These findings are similar to previous recent works that have reported negligible or weak relationships between stream biota and hydromorphology (e.g. Friberg et al. 2009; Vaughan et al., 2009). Morphological alterations may also often remain undetected. For example, increased siltation, which is common in agricultural streams and a severe stressor for, for example, benthic invertebrates, was not quantitatively measured in RHS or in the accompanying HyMo-field surveys. There is a clear need to improve ways to measure morphological degradation, also keeping in mind that many of the morphological ”stressors” exhibit marked natural variation among streams.

Many European WFD-based assessment systems such as the EQRs used in this exercise measure departure of biotic properties from their river type-specific reference conditions, which may, however, be able to indicate only the most obvious taxonomic impairment. Indeed, use of site-specific expectations might be needed to elucidate more subtle biotic impairment, particularly at large geographical scale (Aroviita et al. 2009). Furthermore, as morphological alterations are most likely to be detrimental only for particular organism traits, future work should aim to complement existing taxonomic identity-based assessment with organism trait-based assessment (e.g. Stazner et al. 2001) to develop a comprehensive tool of assessment of the ecological quality of freshwater ecosystems.

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5 Vulnerability to hydromorphological degradation of selected Habitats Directive freshwater species and habitats

5.1 The vulnerability to hydromorphological degradation of plain to montane level water courses with Habitat 3260, Ranunculion fluitantis and Callitricho-Batrachian vegetation

5.1.1 Introduction

Vulnerability refers to the inability to withstand the effects of a hostile environment. Saunders looks at strategies for conservation of freshwater protected areas and identifies altered hydrology as one of three primary threats [Saunders et al. 2002]. In Europe both altered hydrology and geomorphology have been identified as major impacts to European freshwaters, [Agency 2012] and that the vulnerability of Europe's water resources is effected by a wide variety of pressures which are in turn effected by socio-economic drivers [Agency 2012].

Habitat 3260 is represented at 13% of sites designated under the Habitats Directive, it is the 7th most commonly used of all habitat types in site designations and is easily the most commonly applied of the 9 river habitat types, the next mostly commonly used is Habitat 3270 ranked 53rd and represented in only 2.6% of site designations.

In ANNEX I of the Habitats Directive there are 9 river types defined by vegetation type; 1 Fennoscandian, 1 muddy banks, 3 Alpine, 3 Mediterranean and Habitat 3260. It is therefore the only river type available for the designation of rivers in the plain to montane areas of Europe outside the three specific geographic regions specified in the other river types. The major species of submerged riverine vegetation are widespread across the continent with the majority of species belonging to a small number of taxa; Ranunculus, Callitriche, Myriophyllum, Potamogeton and mosses. These key taxa are all acceptable components of the Habitat 3260 definition. It is therefore not surprising that it has been possible to apply this habitat code across the continent (Figure 5.1).

There has been significant concern that submerged aquatic vegetation in general is potentially vulnerable to hydromorphological degradation. It has been hypothesised that the physical habitat, which is what is altered by hydromorphological degradation, is the principle determinand of riverine vegetation after distribution processes, [Fox 1992]. In her seminal work on river plants of western Europe Haslam observed a number of forms of hydromorphological degradation [Haslam 1987]. A review of impacts upon Habitat 3260, which focused on the UK, also implicated hydromorphological degradation in the reduction of habitat quality, although empirical evidence was limited [Hatton-Ellis & Grieve 2003]. However there have been few detailed reviews of the causal effects of hydromorphological pressures and aquatic macrophytes and none which cover all major impacts.

Previous studies on this community have focused on instream vegetation of the middle reaches of river systems. However the physical river types that support this habitat are broader. The inclusion of mosses in the habitat definition has allowed the designation of quite high energy river systems, while the inclusion of all Potamogeton species, and specifically Ranunculus species, of still or sluggish waters has allowed sites to be designated on the basis of vegetation found in secondary channels or floodplain water bodies with intermittent connectance to the main channel. In terms of the river continuum concept, reaches from headwater through middle reaches to lower reaches are all designated under Habitat 3260. As physical systems their response and susceptibility to hydromorphological pressures may vary significantly as does the vegetation they support [Gurnell et al. 2010; O'Hare et al. 20118].

A distinction can be made between total loss of habitat, such as temporary floodplain waterbodies and alterations to habitat such as changes in instream habitat. The plants of Habitat 3260 clearly have no ability to recover from lost habitat but the state of communities and their vulnerability to altered habitat has not been considered in much detail. Riverine vegetation communities can be considered as disturbed communities, their composition determined directly by the influence of physical processes on disturbance and establishment processes. Information has become available on the traits of key species from Habitat 3260 which may indicate vulnerability to altered physical processes.

With a view to the future scenarios are now available on how the hydrology of Europe may alter with changing climate and management [Schneider et al. 2013; Laizé et al. 2013]. These provide a useful indicator of the future vulnerability to hydromorphological degradation.

Here we review the hydromorphological impacts recorded for sites designated with Habitat 3260, we bring together relevant work in the peer review literature, and we consider

- What are the hydromorphology impacts at designated sites?
- We review the traits of key taxa in terms of their role and vulnerability to hydromorphological degradation.
- We review the vulnerability of Habitat 3260 sites to future scenarios of how the hydrology of Europe may alter with climate and management.

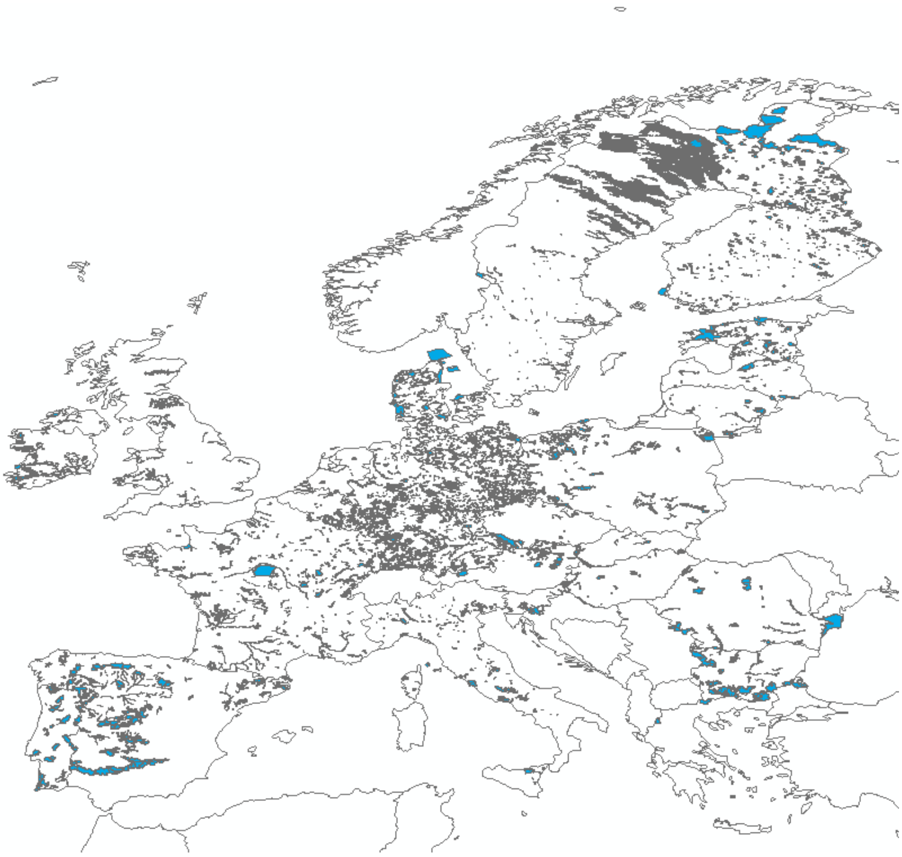

Figure 5.1. A map of sites designated across Europe for Habitat 3260, Water courses of plain to montane levels, with submerged or floating vegetation of the Ranunculion fluitantis and Callitricho-Batrachion (low water level during summer) or aquatic mosses (Habitat 3260).

5.1.2 Methods

Definition of Habitat 3260.

We used the formal definition of Habitat 3260 which is:

The original definition of Habitat 3260 was based on the CORINE classification system (EEC1991).This has now been replaced with the EUNIS system (Council Directive 97/62/EC) as: Water courses of plain to montane levels, with submerged or floating vegetation of the Ranunculion fluitantis and Callitricho-Batrachion (low water level during summer) or aquatic mosses (Habitat 3260). The EUNIS habitat classification was developed to facilitate harmonised description and collation of data across Europe, and builds on the original CORINE and Palaearctic Habitat Classifications. The EUNIS website [http://mrw.wallonie.be] describes how the various habitat codes and definitions relate between classifications. EUNIS guidance lists the plants associated with Habitat 3260 as: Ranunculus saniculifolius, R. trichophyllus, R. fluitans, R. peltatus, R. penicillatus ssp. penicillatus, R. penicillatus spp. pseudofluitans, R. aquatilis, Myriophyllum spp., Callitriche spp., Sium erectum (Berula erecta), Zannichellia palustris, Potamogeton spp. and Fontinalis antipyretica.Ranunculion fluitantis and Callitricho-Batrachian vegetation

Data and analysis for assessing the current hydromorphology impacts at designated sites

Data on sites designated under the Habitats and Species Directive were supplied by the European Environment Agency data portal: http://www.eea.europa.eu/data-and-maps/data/natura-1

These data contain an ACCESS database and GIS layer for all Natura 2000 sites. Sites with Habitat 3260 can be identified and the pressures on the site extracted. Pressures on the sites were identified by the designating body in each country and they were working from a fixed list of potential pressures which contained a limited suite of hydromorphological pressures.

Data on plant traits was taken from the literature and PLANTATT. A database of plant traits assembled by CEH.

Given the nature of the data limited statistical analysis was possible and simple graphs and tables were used.

Data and Analysis to assess the vulnerability of Habitat 3260 sites to future scenarios of how the hydrology of Europe may alter with climate and management

Future hydrological scenarios for Europe suitable for assessing the future vulnerability of habitat 3260 were taken from Laize 2013, which included 4 socio economic scenarios and 2 climate change scenarios. The results are presented graphically as maps showing where each scenario will impact overlain with the location of habitat 3260 sites.

Summary details of the scenarios but full details and the origins of the scenarios can be found in Laize 2013. Projected future climate data for the period 2040–2069 (i.e. '2050s') were taken from two Global Circulation Models (GCMs): (i) IPSL-CM4, Institut Pierre Simon Laplace, France ('IPCM4' thereafter); and (ii) MIROC3.2, Center for Climate System Research, University of Tokyo, Japan ('MIMR' thereafter). For both GCMs, the IPCC SRES A2 emission scenario (IPCC, 2007) was selected; it describes a very heterogeneous world with high population growth, slow economic development and slow technological change (global greenhouse gas emissions projected to grow steadily during the whole 21st century and possibly to double by 2050 compared with that in the year 2000). Under SRES A2 IPCM4 predicts a high temperature increase and a low precipitation increase/decrease ('warm and dry'), whereas MIMR predicts a high temperature increase and a high precipitation increase or a low decrease ('warm and wet'). The four economic scenarios were as follows: economy first (EcF), economy-oriented towards globalization and liberalization with intensified agriculture and slow diffusion of water-efficient technologies; • fortress Europe (FoE), closed-border Europe concentrating on common security issues with food and energy independence as the main focus of the European coalition; • policy rules (PoR), stronger coordination of policies at the European level, driven in part by high energy costs and reduced access to energy supplies, expectation of climate change impacts and increasing water demand; • sustainability eventually (SuE) transition from globalizing, market-oriented Europe to environmental sustainability with quality of life as a central point.

5.1.3 Results

Current hydromorphology impacts at designated sites

There are over 3000 sites designated as Special Areas of Conservation across Europe with Habitat 3260. The majority have good or excellent representation of the habitats (Figure 5.3).

The top impacts are drainage, modifying structures and modification of hydrographic functioning, these effect, respectively, 23%, 12% and 10% of designated sites, Table 5.1. The rank order of the impacts differs little between the top three categories of site representivity.

It is clear that it is possible to have sites in good or excellent condition despite hydromorphological impacts. The size and spatial extent of the physical alterations are likely to determine their actual impact on the vegetation. A typical case study is the River Tweed SAC (Site code UK0012691) which hosts Habitat 3260 in good condition and bankside management is acknowledged as an important impact. In the 18th and 19th centuries the land in the bottom of the valley was drained to create good agricultural land while sections of the river were straightened and the floodplain protected from the river by setback embankments in many places. In the present day the valley bottom is well drained and secondary channels have been artificially cut off (Figure 5.2). Instream conditions are relatively unaffected and many riffle reaches support good stands of *Ranunculus pencillatus*. However the diversity associated with floodplain waterbodies is mostly lost, the cut off meander bend in the photograph was at one time a backwater pool, a relatively rare habitat in this system, but one which would have supported a diverse flora which would have contributed significantly to Habitat 3260.

Figure 5.2. River Tweed SAC Scotland UK with temporary floodplain water body artificially cut off from the river some time before 1856.

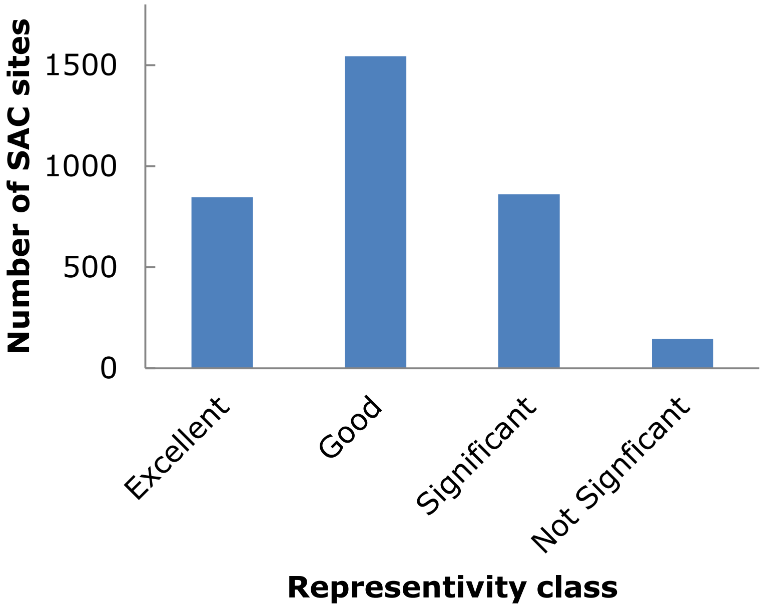

Figure 5.3. The number of SAC sites with Habitat 3260 and how representative they are of that habitat. Data from http://www.eea.europa.eu/data-and-maps/data/natura-2

Table 5.1. Rank ordered counts of SAC sites supporting habitat 3260 subject to hydromorphological related activities. Sites are categorised by how well they represent the Habitat type. Data from EEA and extracted from reports by National representative bodies.

	Representivity of the sites			
	Excellent	**Good**	**Significant**	**Not Significant**
Drainage	139	184	115	29
Modifying structures of inland water courses	110	177	98	16
Modification of hydrographic functioning, general	104	169	69	17
Management of water levels	69	93	62	7
Bridge, viaduct	65	65	46	7
Canalisation	61	80	61	13
Other human induced changes in hydraulic conditions	52	75	68	7
Sand and gravel extraction	44	101	39	13
Management of aquatic and bank vegetation for drainage purposes	43	77	52	1
Dykes, embankments, artificial beaches, general	27	49	33	13
Irrigation	22	47	30	5
Silting up **Activity**	21	53	30	5
Infilling of ditches, dykes, ponds, pools, marshes or pits	20	56	20	5

The vulnerability of Habitat 3260 sites to future hydrological scenarios

The highest density of Habitat 3260 sites can be found in central Europe and Iberia (Figure 5.1). Good representation is also found in other areas such as Finland and Scandinavia while a small number of sites are found in other areas such as the British Isles and France. These geographic differences are most likely to be explained by the stringency of national interpretation of the Habitats Directive guidelines rather than the distribution of the key flora. All the major taxa have a Pan-European distribution.

We compared the distribution of the designated sites against future hydrological change scenarios based on 2 climate change scenarios and 4 socio-economic scenarios (Figure 5.4). Vulnerability was defined as significant changes in the magnitude or timing or either seasonal, low or high flows. Southern and central Europe are most vulnerable to change. Under all IPCM climate scenario combinations Germany and the Czech Republic appear vulnerable while under both MIMR and IPCM climate scenarios Iberia and southeastern Europe appear vulnerable. The differences between the socio-econmic scenarios is subtle and do not mitigate against the climatic scenarios.

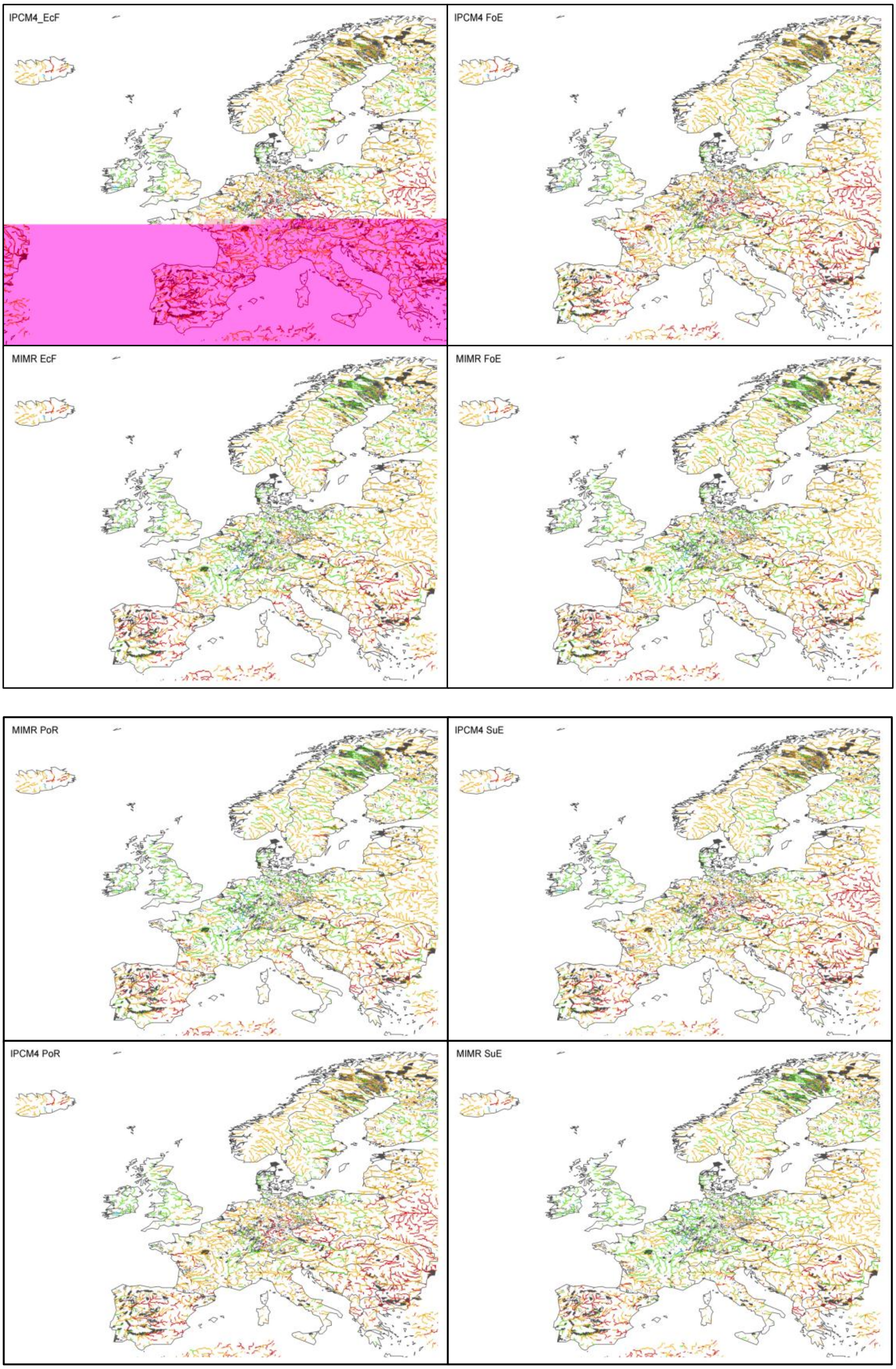

Figure 5.4. ERFA geographical location changes between Natural and socio-economic scenarios After Laize 2013); economy first (EcF),fortress Europe (FoE), policy rules (PoR), sustainability eventually (SuE); climate models A2-IPCM4 (IPCM4) A2-MIMR (MIMR). Model outputs level of

change as; blue, no risk; green, low risk; amber, medium risk; red, high risk. Habitat 3260 sites are overlain in black.

5.1.4 Discussion

From our results it is clear that Habitat 3260 sites across Europe are subject to a wide range of hydromorphological pressures. It is equally clear that at the majority of sites where this habitat is excellently represented have no noted hydromorphological impact and it is possible for this habitat to be present even where the site is impacted. In the information available on the types of impacts there was nothing to describe the intensity of impact it is therefore possible that the actual impact was low on some sites.

The case study presented for the River Tweed SAC is instructive where natural habitat has been lost but the site remains in good condition. As with the River Tweed many fluvial geomorphological alterations have been in place for very long periods and in the case of lowland drainage networks represent the cumulative effort of generations.

It is important therefore to acknowledge the historic damage to our riverine macrophyte communities, especially the loss of connectivity with temporary floodplain waterbodies.

Macklin identified that in the UK that large-scale influx of fine sediment and transformation of channels and floodplains related to agricultural activities occurred during the Middle Ages (c. 10th to early 14th Century) [Foulds & Macklin 2006; Macklin et al. 2006)]. There is a similar story in continental (northern) Europe with small scale impacts in the late prehistoric and early historical periods followed by major changes in the Middle Ages. Later drainage of lowland systems accelerated with the transfer of techniques such as polders developed by the Dutch to other countries.

Whilst there is a documented acceleration in channelization and dam building associated with the industrial revolution of the 1800s the drainage programs associated with the agricultural revolution began in the proceeding centuries. It is arguable therefore that the standard approach to setting a temporal reference state for channelized rivers across Europe of a pre-industrial condition is not useful. Equally it is arguably returning European rivers to a pre-agricultural revolution state is neither practical nor desirable. Dam building is different and is more closely linked to the technical advances in the industrial era.

A key change was the drainage of low energy braided rivers from low lying ground across northern Europe which drain into the North Sea. These valuable systems have been heavily exploited and today are subject to multiple stressors. One of the few remaining examples is the river Narew in Poland where the strategic importance of this natural barrier to the east limited its drainage. Such channels are anabranching and multi thread where vegetation determines channel form. These sluggish channels have good growing conditions for macrophytes. Elsewhere such systems have mostly been replaced by a series of artificially maintained channels and bear limited resemblance to their natural form. In fact they are so entirely absent from our landscapes that there is no living memory in most of northern Europe as to what these landscapes would once have looked like. Their absence from the common consciousness also effects scientists. Hence we base our conservation targets on what we have rather than what we once had under more natural conditions

In terms of future vulnerability the overlap of regions with high densities of Habitat 3260 sites and the climatic scenarios suggest Habitat 3260 are at ongoing risk which is not likely to be alleviated by anything but the most drastic socio-economic changes. It is arguable that some change in climate is now unstoppable and mitigation measures should be considered on the ground to preserve a wide variety of flora and habitats. To do so would require a stronger understanding of the links between hydrology, fluvial geomorphology and Habitat 3260 species.

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6. Empirical biological response to altered sediment dynamics

6.1 Introduction

River managers and the freshwater scientific community have a long-standing awareness of the detrimental impacts of fine sediment (inorganic and organic particles of less than 2 mm diameter) on aquatic ecosystems (Ellis 1936; Jones et al. 2012a). Excessive delivery and retention of fine sediments can impact all components of the biological community of freshwaters (Collins et al. 2011; Kemp et al. 2011; Jones et al. 2012a; Jones et al. 2013). This leads to both direct and indirect impacts on the benthic macroinvertebrate community (Jones et al. 2012b). Different components of the macroinvertebrate community are likely to respond to different aspects of these impacts depending on their intrinsic biological traits, for example certain taxa are likely to be susceptible to the chemical changes associated with the amount of organic matter deposited on the river bed, whereas others may be more susceptible to the physical impacts of inorganic fine sediments (Culp et al. 1986). Community composition may also respond to changes in habitat availability induced both directly or indirectly (e.g. through changes in the availability of macrophyte habitat) by increased fine sediment inputs (Pardo & Armitage 1997).

Recent research in the UK has improved our understanding of how benthic macroinvertebrate communities respond to increasing fine sediment stress (Extence et al. 2013; Murphy et al. in prep.). However, these findings focus on compositional changes. There have been recent developments in European and North American freshwater research into the examination of multiple biological traits of aquatic organisms in the context of various environmental constraints (Statzner et al. 2001; 2008; Zuellig & Schmidt 2012). There is a need to better understand how the prevalence of biological traits in the macroinvertebrate community changes along a gradient of increasing fine sediment stress. The biological trait approach could lead to more widely applicable diagnostic indices of impact as opposed to the composition-based indices that can be limited to the original development biogeographic region.

The current study seeks to quantify the changes in the lotic macroinvertebrate biological trait assemblage across a gradient of increasing agricultural fine sediment delivery and retention. These analyses are carried out on a biological and environmental dataset collected as part of a UK government-funded project with the objective of extending the evidence base on the ecological impacts of fine sediment on freshwaters. The dataset will be interrogated anew, as part of REFORM WP3, from a species trait perspective, with the aim of identify suites of traits that are associated with fine sediment stress, and conversely those associated with low stress conditions.

6.2 Methods

6.2.1 Field survey

In total, 205 stream sites across England and Wales were sampled between spring 2010 and autumn 2011 (Figure 6.1). Each site was on an independent watercourse, was sampled once with a sample of the macroinvertebrate community and deposited fine sediment being collected. These sites were selected from a larger pool of sites each of which was confirmed to be:

- not impacted by sewage inputs
- not to have large urban areas in the catchment
- not to have upstream reservoirs or lakes

The delivery of fine sediment from the catchment to each river sites was modeled using PSYCHIC, a process-based model of suspended sediment mobilisation in land run-off and subsequent delivery to watercourses (Davison *et al,.* 2008). Based on these data we only considered sites with predominantly (>75%) agricultural fine sediment sources.

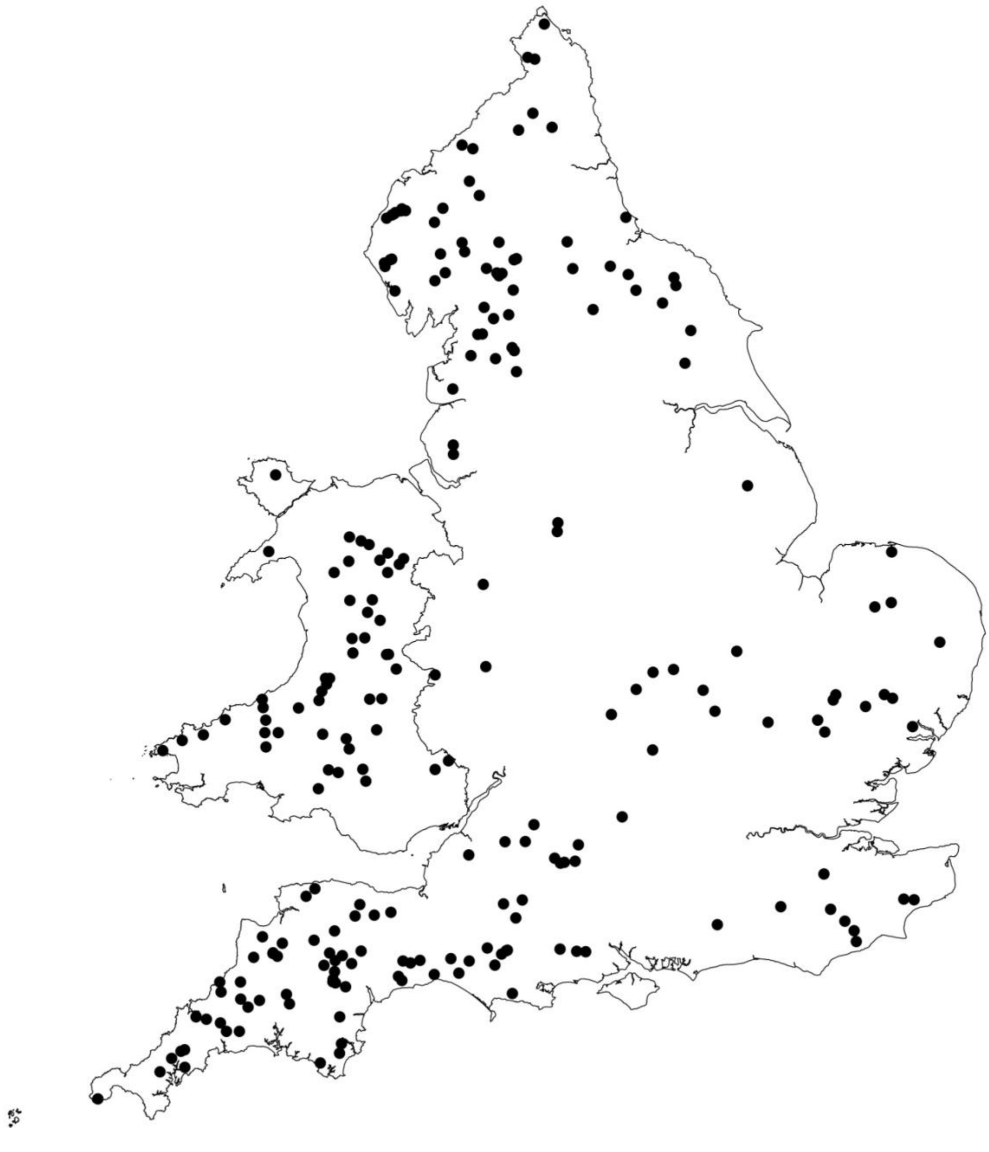

Figure 6.1. Location of sampled stream sites across England and Wales.

To ensure that the sampled macroinvertebrate communities came from as wide a range of natural river types as possible, within the limits set by the other site selection criteria, each site was allocated to one of four approximate stream types based on four map-based physical variables; their catchment geology, distance from source (km), altitude (m asl) and slope (m km^{-1}). The boundary values for this guideline stream typology were loosely based on those associated with the seven RIVPACS IV super end groups (Davy-Bowker et al. 2008). The fundamental aim was to ensure as equal a sampling effort as possible across the fine sediment stressor gradient for each broad stream type, thus ensuring that a representative sample of streams was included in the study where fine sediment pressure was the main driver of differences in species occurrence.

Macroinvertebrates were sampled at each site using the RIVPACS method; a standard three-minute kick/sweep and one minute search sample with a pond net (1 mm mesh-size) (Furse et al. 1981; Murray-Bligh et al. 1997). A field measurement of pH and conductivity was also taken (Hanna Instruments Combo HI98129). Associated RIVPACS environmental variables were recorded either at the site (stream width and depth, velocity, substrate composition), or from map-based data (discharge category, altitude, distance from source and slope). Macroinvertebrate community samples returned to the laboratory for subsequent identification and quantification to the lowest practicable taxonomic level.

Fine sediment deposits on the stream bed were quantified immediately upstream of the macroinvertebrate sampling area using the disturbance technique described in Lambert and Walling (1988) and refined by Collins and Walling (2007). Here a steel cylinder (height 75 cm, diameter 48.5 cm) was inserted into an undisturbed section of the stream bed and the water column vigorously agitated for one minute, without touching the stream bed, to raise any fine sediment deposited on the surface of the stream bed. A pair of water samples was then collected quickly from within the cylinder. The stream bed was then disturbed to a depth of approximately 10 cm, and the water and bed vigorously agitated for one minute to raise any sub-surface fine sediment in addition to the re-suspended surface deposits. A second pair of water samples was then collected from within the cylinder. Four such sets of water samples (surface, and combined surface and subsurface) were collected from each site, two from erosional patches and two from depositional patches. The samples were then refrigerated and returned to the laboratory within 5 days, where they were processed for dry mass and organic content (i.e. volatile solids following combustion at 550°C). The particle size distribution of material <1mm diameter was also measured using a Malvern Mastersizer 2000. Reach-averaged values for surface and total (combined surface and subsurface) deposited fine sediment were subsequently derived.

In summary, for each site, there was an estimate of the quantity of fine sediment being delivered from the catchment (kg ha^{-1} yr^{-1}), derived from the PSYCHIC model, as well as actual measurements of deposited fine sediment mass and composition and the in-stream biological community.

6.2.2 Compilation of biological trait data

It is important at this stage to make the distinction between *biological* and *ecological* traits. Biological traits describe intrinsic characteristics of the species, independent of the environment in which they exist. Ecological traits describe the tolerances/preferences of a species for different aspects of its environment. We were only interested in using biological traits in this study as there is an inherent circularity in using ecological traits to describe how a community varies across a stressor gradient i.e. it is not informative to discover that traits indicating a preference for fine sediment increase in sites stressed by fine sediment.

Two existing freshwater macroinvertebrate species trait resources were used to gather available biological trait information for the 192 taxa identified in the dataset.

- www.freshwaterecology.info

- French Genus Trait Database (Tachet et al. 2000)

The former was originally compiled as part of the EU FP5-funded AQEM project and supported and further developed by subsequent EU-funded projects; STAR (FP5), Euro-limpacs (FP6), BIOFRESH and REFRESH (both FP7). The latter dataset was compiled by French biologists for those taxa found in French waters; many of which are also found in the UK. The French data was the primary source of information and was supplemented with information from freshwaterecology.info for those taxa or traits that were not included in the French database.

Each biological trait was described by several trait-classes. The trait characteristics of each taxon were scored by assigning a value to each trait-class reflecting the affinity of the taxon to the trait-class. Scores ranged from 0 to 5 indicating no to high affinity respectively (Chevenet, Dolédec & Chessel, 1994). The way traits were scored differed between the two databases, as did the range of trait-classes for some traits. For those traits that did match (life cycle duration, resistance form, reproduction, dispersal, respiration) we identified the taxa in freshwaterecology.info that were not already included in the French data. These data were then appended to the French data to create the final trait dataset. As the two datasets use a different scoring method, we also had to convert the freshwaterecology.info 1-10 method to the French 1-5 method by simply converting 5s to 3s and 10s to 5s, leaving 1s as 1s.

The final dataset contained information on 10 biological traits for 192 distinct taxa (Table 6.1).

Table 6.1. Biological traits used in the analysis with their associated trait classes.

TRAIT	TRAIT-CLASS	Abbreviation
Maximal potential size	≤ .25 cm	MaxS_25cm
	> .25-.5 cm	MaxS_5cm
	> .5-1 cm	MaxS_1cm
	> 1-2 cm	MaxS_2cm
	> 2-4 cm	MaxS_4cm
	> 4-8 cm	MaxS_8cm
	> 8 cm	MaxSm8cm
Life cycle duration	≤ 1 year	Lcyc_m1
	> 1 year	Lcyc_l1
Potential number of cycles per year	< 1	Pcyc_lt1
	1	Pcyc_1
	> 1	Pcyc_gt1
Aquatic stages	egg	AqSt_eg
	larva	AqSt_la
	nymph	AqSt_ny
	adult	AqSt_ad
Reproduction	ovoviviparity	Repr_ovo
	isolated eggs, free	Repr_ief
	isolated eggs, cemented	Repr_iec
	clutches, cemented or fixed	Repr_ccf
	clutches, free	Repr_cfr
	clutches, in vegetation	Repr_cvg
	clutches, terrestrial	Repr_ctr
	asexual reproduction	Repr_asr
Dispersal	aquatic passive	Disp_aqp
	aquatic active	Disp_aqa

TRAIT	TRAIT-CLASS	Abbreviation
	aerial passive	Disp_aep
	aerial active	Disp_aea
Resistance forms	eggs, statoblasts	Rest_egg
	cocoons	Rest_coc
	housings against desiccation	Rest_hou
	diapause or dormancy	Rest_dia
	none	Rest_non
Respiration	tegument	Resp_teg
	gill	Resp_gil
	plastron	Resp_pla
	spiracle	Resp_spi
Locomotion and substrate relation	flier	Loco_fli
	surface swimmer	Loco_ssw
	full water swimmer	Loco_swi
	crawler	Loco_crw
	burrower	Loco_bur
	interstitial	Loco_int
	temporarily attached	Loco_tpa
	permanently attached	Loco_pat
Food	microorganisms	Food_mio
	detritus (< 1mm)	Food_det
	dead plant (>= 1mm)	Food_dep
	living microphytes	Food_mip
	living macrophytes	Food_map
	dead animal (>= 1mm)	Food_dea
	living microinvertebrates	Food_mii
	living macroinvertebrates	Food_mai
	vertebrates	Food_vrt

6.2.3 Data analysis

To summarise and illustrate the covariation between the biological trait data and environmental characteristics (both natural and stressor-related) of the sites we used RLQ analysis (Dolédec et al. 1996). RLQ analysis is a three-table ordination method that allows the simultaneous comparison of patterns in matrices of site environmental data (R-table) and species trait data (Q-table) via a connecting matrix of species composition by site data (L-table). It finds linear combinations of the environmental variables and linear combinations of the species traits of maximal covariance weighted by species abundance data.

In preparation for the analysis we constructed three matrices; L-table of the $\log_{10}(x+1)$-transformed abundance of 192 taxa at each of 205 stream sites, Q-table of 54 trait-class scores for each of 192 taxa and an R-table of values for 21 environmental variables at each of the 205 sites (Table 6.2). RLQ analysis was carried out using R 2.15.3 with the additional ade4 package (Chessel *et al*., 2004).

Table 6.2. Environmental variables recorded for each site.

Variable type	Environmental variable	Abbreviation
Natural environmental variable	Discharge Category (long-term historical average discharge; 1-10) (1= <0.31, 2= 0.31-0.62, 3= 0.62-1.25, 4= 1.25-2.5, 5= 2.5-5.0, 6= 5-10, 7= 10-20, 8= 20-40, 9= 40-80, 10= 80-160 m^3s^{-1})	DISCHCAT
	Distance from source (km) (log-transformed)	DISTSOU
	Altitude of site (masl) (log-transformed)	ALTITUDE
	Slope of site ($m.km^{-1}$) (log-transformed)	SLOPE
	Catchment area (km^2) (log-transformed)	CatcArea
	Stream width (m) (log-transformed)	WIDTH
	Stream depth (cm) (log-transformed)	DEPTH
Measured fine sediment variables	Geomean total sediment mass (log-transformed)	SedMass
	Geomean Depositional area sediment mass (log-transformed)	DpSedMas
	Geomean Erosional area sediment mass (log-transformed)	ErSedMas
	Geomean total organic mass (log-transformed)	VsMass
	Geomean Depositional area organic mass (log-transformed)	DpVsMas
	Geomean Erosional area organic mass (log-transformed)	ErVsMass
	Mean % organic (log-transformed)	PctOrg
	Mean Depositional area % organic (log-transformed)	DpPctOrg
	Mean Erosional area % organic (log-transformed)	ErPctOrg
	% by volume of particles in sand size category	PctSa
	% by volume of particles in silt size category	PctSi
	% by volume of particles in clay size category	PctCl
Modelled fine sediment inputs	PSYCHIC estimate of local bank erosion fine sediment load to site from catchment (kg/ha/yr) (log x+1-transformed)	LBESedLd
	PSYCHIC estimate of agricultural fine sediment load to site from catchment (kg/ha/yr) (log x+1-transformed)	AgSedLd

RLQ provides a biplot that graphically illustrates the main patterns of co-variation between the trait and environmental data. It also provides correlations of each environmental variable and trait-class against each RLQ axis and correlations directly between the trait-classes and the environmental variables.

As an alternative analytical approach we undertook a series of more conventional ordinations (Redundancy Analyses) between the environmental data and a matrix of the relative prevalence of each trait-class (within each of the 10 traits) at each of the 205 sites. This latter table was created by log-transformed abundance weighting the taxon scores for each trait-class for a given site. The sums of weighted scores (one per trait-class) were then expressed as the relative abundance distribution (within a trait), giving the site trait profile (Archaimbault et al. 2010).

Initial detrended correspondence analyses (both with and without down-weighting of rare traits) showed that there was relatively little variation in the trait data across the 205 samples. Therefore it was appropriate to use redundancy analysis (RDA) for the direct gradient analysis between the environmental and trait matrices. Using RDA with manual forward selection (999 permutations), excluding variables that don't make a significant additional contribution to the power of the model (P<0.005), we determined which of the measured 'natural' and 'stressor' environmental variables were best associated variation in the trait data is across the 205 samples. We could then use partial RDA (pRDA), with significant natural environmental variables considered as co-variables describing the natural environmental gradient through the dataset. Having factored out this variation we could determine which of the fine sediment stressor variables were significantly associated with the residual variation in the trait data.

It cannot be assumed that the factoring out of variation associated with natural environmental variables will lead to a better subsequent ranking of traits according to their fine sediment tolerances. This needs to be demonstrated to be the case. To do this we compared the percentage of the variation in the biological trait data explained by the 1st ordination axis in a pRDAA (with fine sediment variables as the only constraining variable and other confounding variables as co-variables) with that in a RDA with fine sediment variables as the only constraining variable. If the percentage value was greater in the partial ordination then this would support the case for using pRDA to achieve a better ranking of traits according to their fine sediment tolerances.

The DCA/RDA analyses were carried out using CANOCO 4.5 (ter Braak & Šmilauer, 2002).

6.3 Results

6.3.1 RLQ analysis

The first two RLQ-axes represent 98.7% of the variance explained by correspondence analysis on the L-table (species x site matrix), with axis 1 predominant (96.4%) (Table 6.3).

Table 6.3. RLQ eigenvalue decomposition.

	Eigenvalue	Co-variance	L-correlation
Axis 1	4.047	2.012	0.325
Axis 2	0.097	0.311	0.105

A Monte-Carlo permutation test demonstrated a significant relationship between the environmental characteristics of sites (R) and biological traits of the communities at those sites (Q) (Observed total inertia= 4.197, p < 0.001).

The markedly dominant RLQ axis 1 was most positively associated with an increasing mass of fine sediment and with decreasing levels of modelled agricultural fine sediment input (Table 6.4). The much weaker RLQ axis 2 was related to increasing catchment area and decreasing organic fine sediment (as a proportion of total volume of fines). When considered against the trait data, RLQ axis 1 was most positively associated with a decreased prevalence of active aerial dispersal and crawling locomotion and conversely with an increased prevalence of aquatic adults, and multiple generations within a year (Table 6.5).

Examination of the RLQ trait-environment correlations and RLQ biplot (Figure 6.2) revealed that trait-classes whose prevalence in the community increased as the mass of fine sediment on the stream bed increased included aquatic adults, ovoviviparity, and multiple generations within a year. Conversely the laying of isolated cemented eggs, active aquatic dispersal, active aerial dispersal, eggs or statoblasts as resistant forms & crawling locomotion ecreased in prevalence with increases in the mass of fine sediment on the stream bed. Furthermore, ovoviviparity prevalence decreased and active aerial dispersal and crawling prevalence increased with increasing altitude, stream width, inputs of agricultural fine sediment and natural bank erosion inputs.

Table 6.4. Correlations between the first two RLQ axes and the 21 environmental variables offered to the RLQ analysis. Correlations coefficients greater than 0.65 and less than -0.65 are highlighted in bold. See Table 6.2 for key to environmental variable abbreviations.

Environmental variable	RLQ axis1	RLQ axis 2
SedMass	**0.9493526**	0.29884522
VsMass	**0.9344715**	0.12137952
ErSedMas	**0.9132449**	0.19863008
DpSedMas	**0.8715874**	0.37041623
DpVsMas	**0.8714417**	0.20053986
ErVsMass	**0.8649228**	0.03056473
PctSa	0.39244	0.62941294
PctCl	-0.1352315	-0.39335662
DpPctOrg	-0.2817423	**-0.66179116**
CatcArea	-0.2845193	**0.73420852**
SLOPE	-0.2930095	-0.61639064
DEPTH	-0.2998104	0.57416331
DISTSOU	-0.4306106	0.64733281
PctSi	-0.4307757	-0.62562503
DISCHCAT	-0.4417904	0.28240878
PctOrg	-0.452721	**-0.66642531**
ErPctOrg	-0.5239508	-0.59209316
WIDTH	-0.6014054	0.514598
ALTITUDE	-0.6056211	-0.15364768
LBESedLd	-0.648299	-0.03677813
AgSedLd	**-0.7084073**	-0.09202028

Table 6.5. Correlations between the first two RLQ axes and the 54 biological trait-classes offered to the RLQ analysis. Correlations coefficients greater than 0.6 and less than -0. 6 are highlighted in bold. See Table 6.1 for key to biological trait-classes abbreviations.

Biological trait-class	RLQ axis1	RLQ axis 2
Pcyc_gt1	**0.64273226**	-0.10448861
AqSt_ad	**0.6183783**	0.43769777
MaxS_8cm	0.51179006	0.13796665
Repr_ovo	0.50236184	0.14962053
Repr_asr	0.48630579	0.16701771
Rest_coc	0.45305013	0.16530365
Loco_bur	0.38285959	-0.12781595
MaxSm8cm	0.35090992	0.1692635
Loco_int	0.32622994	-0.0333533
MaxS_4cm	0.27876099	0.1032638
Food_mio	0.2720088	0.0197418
Food_det	0.26491247	-0.10497575
Repr_cfr	0.21389573	-0.4523788
Food_mii	0.17338763	-0.16841745
Resp_spi	0.15227412	-0.32976351
Food_dea	0.13526893	-0.11492149
Loco_tpa	0.11737183	-0.14930223

Biological trait-class	RLQ axis1	RLQ axis 2
MaxS_5cm	0.10676863	0.23381871
Loco_ssw	0.08481306	-0.29778724
Rest_hou	0.06886461	-0.07032037
Lcyc_l1	0.05655208	0.3256456
Food_vrt	0.05094389	0.06737677
Resp_teg	0.05060983	-0.20328733
Disp_aep	0.04097915	-0.24217957
Repr_cvg	0.03783435	0.06605943
Food_dep	0.03712273	0.13094473
AqSt_ny	0.02994072	-0.26853339
Repr_ctr	0.02850071	-0.36696132
Food_map	-0.01336275	0.17505266
Loco_swi	-0.03300338	-0.24307827
Rest_dia	-0.03551891	-0.11950529
Loco_pat	-0.04032823	0.20616014
MaxS_25cm	-0.04167468	-0.12113419
Food_mai	-0.04790544	-0.37986425
Rest_non	-0.06511089	0.12406299
MaxS_1cm	-0.07746364	-0.50441016
Loco_fli	-0.09742682	0.41833021
Resp_pla	-0.14217251	0.50717735
Food_mip	-0.15650683	0.18571939
Disp_aqp	-0.2297797	0.05769737
Repr_ief	-0.23765335	-0.09036062
Repr_ccf	-0.26388431	0.28447154
AqSt_eg	-0.27494063	**0.59548208**
AqSt_la	-0.27506094	0.08245679
MaxS_2cm	-0.27527027	-0.06771145
Resp_gil	-0.29154859	0.46973598
Lcyc_m1	-0.37097008	-0.29218908
Pcyc_lt1	-0.40077215	-0.08094134
Repr_iec	-0.48608594	-0.23560602
Rest_egg	-0.51596851	-0.15294436
Disp_aqa	-0.52344955	-0.17380158
Pcyc_1	-0.58761437	-0.03856079
Disp_aea	**-0.73497196**	0.11003842
Loco_crw	**-0.75766661**	0.10754586

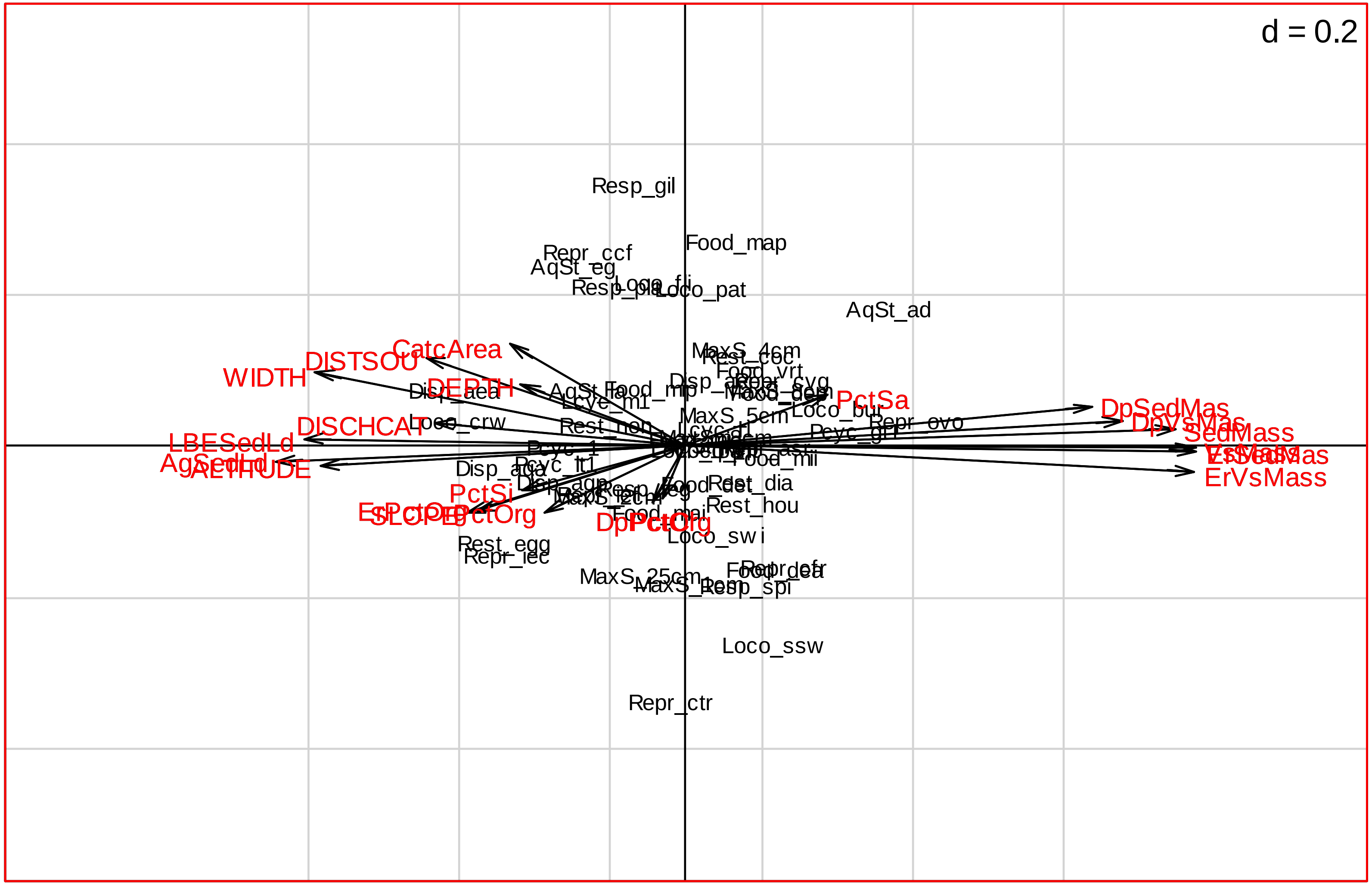

Figure 6.2. RLQ biplot representing biological traits (black labels) and environmental variables (arrows, with red labels). See Table 6.1 and 6.2 for key to biological trait-class and environmental variable abbreviations.

6.3.2 Redundancy Analysis

The initial RDA (Table 6.6) confirmed that the variation in the biological trait data across the 205 sites was best explained by a model incorporating site-averaged mass of organic fine sediment, mass of organic fines in depositional areas, stream width and slope, and modelled inputs of agricultural fine sediment. This model could account for 35.8% of the variation in the biological trait data, with site-averaged mass of organic fine sediment being the dominant explanatory variable (75% of the model explanatory power).

After the variation in the trait data that was associated with natural environmental differences between sites (stream width and slope) was factored out, the three fine sediment stress variables (site-averaged mass of organic fine sediment, mass of organic fines in depositional areas and modelled inputs of agricultural fine sediment) could account for 13.4% of the residual variation in the biological trait data, of which 10.9% associated with axis 1. This was not as great as an RDA constrained only by the three selected fine sediment stress variables, in which the forst axis could account for 30% of the biological trait variation. We therefore proceeded with the latter RDA to quantify the association between the fine sediment stress gradient and the prevalence of biological traits in the macroinvertebrate assemblage.

Table 6.6. Summary of redundancy analysis (RDA).

pRDA Axes	1	2	3	4	Total variance
Eigenvalues	0.299	0.015	0.004	0.187	1
Species-environment correlations	0.8	0.363	0.255	0	
Cumulative percentage variance:					
of species data	29.9	31.4	31.8	50.6	
of species-environment relation	94	98.7	100	0	
Sum of all eigenvalues					1
Sum of all canonical eigenvalues					0.318

Among the traits most strongly associated with an increasing mass of fine organic sediment in the stream bed (and conversely decreasing modelled agricultural fine sediment inputs) were ovoviviparity, burrowing locomotion, prolonged adult aquatic stage, multiple generations in a year, maximum potential size of 4-8cm and passive aerial dispersal Figure 6.3; Figure 6.4). The traits most strongly associated with increasing modelled agricultural fine sediment inputs (and conversely decreasing mass of fine organic sediment) included active aerial dispersal, crawling locomotion, eggs or statoblasts as resistant forms, prolonged egg and larval aquatic stage, and the laying of cemented isolated eggs (Figure 6.3; Figure 6.4).

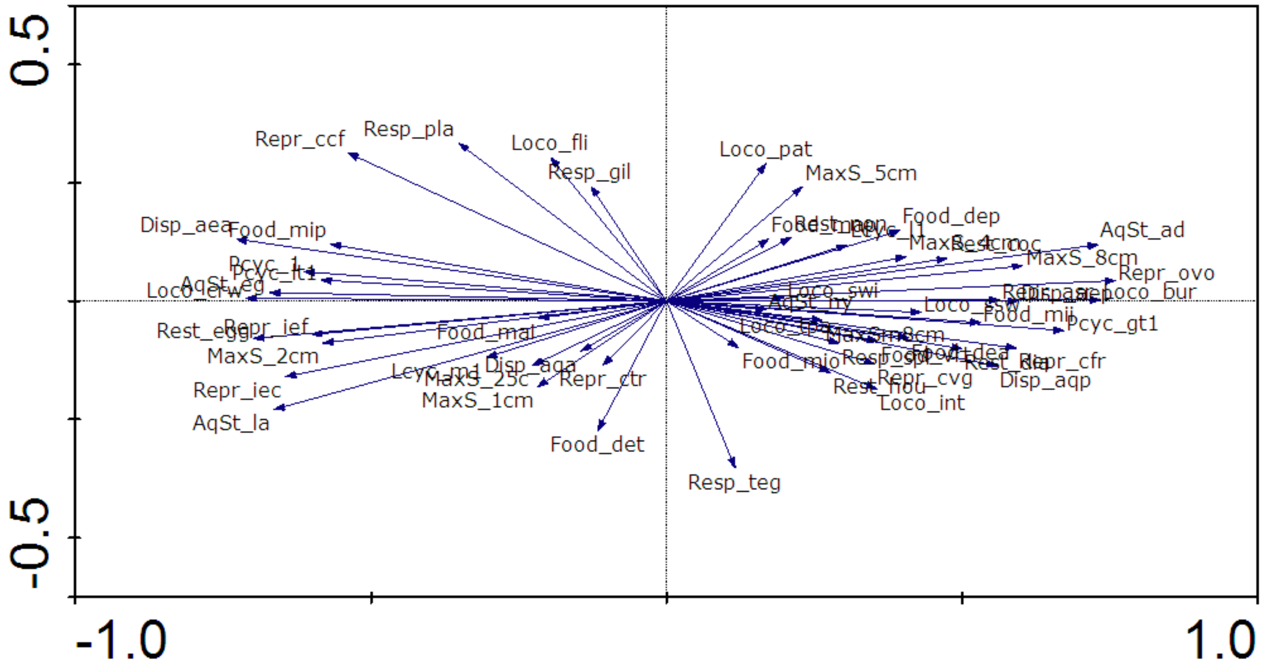

Figure 6.3. RDA ordination plot of biological trait-classes. See Table 6.1 for key to abbreviations.

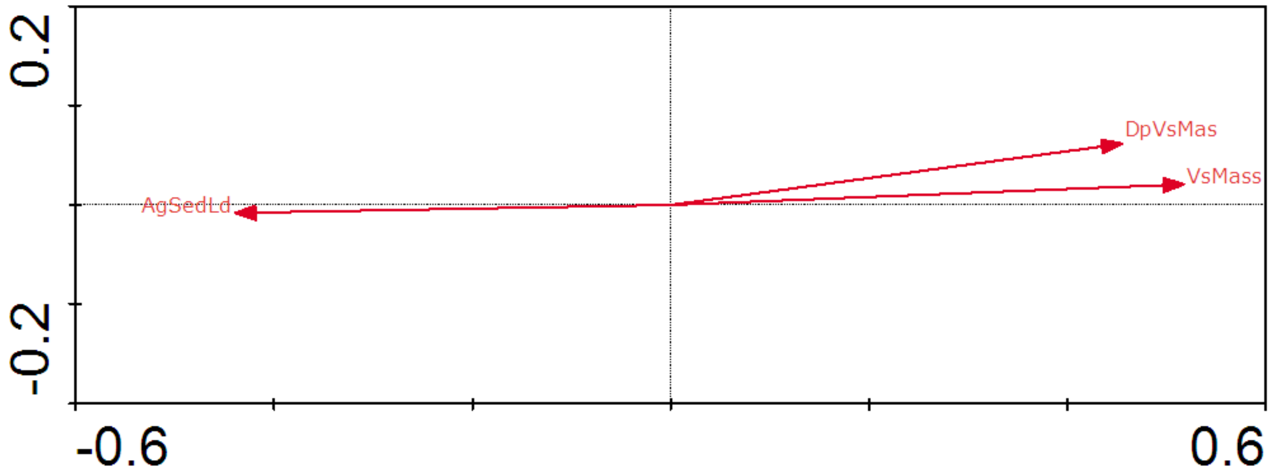

Figure 6.4. RDA ordination plot of environmental variables. See Table 6.2 for key to abbreviations.

6.3.3 Comparison of approaches

Despite the different analytical approaches, there was broad agreement between the results of the RLQ and RDA. Both found that the prevalence of ovoviviparity, prolonged adult aquatic stage, and multiple generations in a year increased with an increasing mass of fine sediment in the stream bed. The association between increasing modelled agricultural fine sediment inputs and active aerial dispersal, crawling locomotion, and eggs or statoblasts as resistant forms was also consistent across the two methods (Table 6.7).

Table 6.7. Summary comparison of RLQ and RDA analyses of relationship between biological traits and fine sediment stress in streams.

Traits	**Trait-classes**	**RLQ**		**RDA**	
		Increasing mass of fine sediment	Increasing inputs of agricultural fine sediment	Increasing mass of fine sediment	Increasing inputs of agricultural fine sediment
Maximal potential size	> 4-8 cm			↑	
Potential number of cycles per year	Multiple generations in a year	↑		↑	
Aquatic Stages	Prolonged egg aquatic stage				↑
	Prolonged larval aquatic stage				↑
	Prolonged adult aquatic stage	↑		↑	
Reproduction	Ovoviviparity	↑	↓	↑	
	Laying of free isolated eggs				↑
	Laying of isolated cemented eggs	↓			↑
	Laying of free clutches of eggs			↑	
Dispersal	Active aquatic dispersal	↓			
	Active aerial dispersal	↓	↑		↑
	Passive aerial dispersal			↑	
Resistance forms	Eggs or statoblasts as resistant forms	↓			↑
Locomotion and substrate relation	Crawler	↓	↑		↑
	Burrower			↑	

6.4 Conclusions

The strength of the trait-based approach is that it overcomes biogeographic differences between regions and allows a more valid comparison of the biological condition of watercourses to be made. A weakness is the problem of consistently describing the traits of macroinvertebrate taxa on the same scale; such information is often lacking for many taxa in many regions (Bonada *et al*. 2006). The trait-based approach is however underpinned more directly by ecological theory which allows investigators to make specific predictions of trait responses to environmental change and to better understand the mechanisms of impact (Zuellig & Schmidt 2012).

This work has confirmed that there is a statistically significant association between the condition of streams, in terms of the quantity of benthic fine sediment, and the biological trait characteristics of the macroinvertebrate community found in the stream bed. Correlative analysis of a spatially extensive dataset, specifically designed and collected to investigate benthic fine sediment impacts, has identified consistent patterns in the trait assemblage that could in the future be applied to more manipulative experimental situations or other broad-scale bioassessment surveys.

Biological traits such as ovoviviparity were strongly associated with increasing fine sediment stress. This trait is commonly found in freshwater Crustacea, Hirudinea, and some Mollusca and may offer an advantage to progeny in terms of boosting their survival chances in the stressed environment. The prevalence of the laying of isolated cemented eggs decreased with increasing fine sediment stress. This trait is more often associated with Plecoptera and Ephemeroptera taxa and it not difficult to understand how excessive deposition of fine sediment on benthic surfaces could reduce the availability of suitable egg-laying substrates for gravid females.

In a recent review of biological monitoring approaches Bonada et al. (2006) considered that the trait-based approach met 10 of their 12 criteria for defining an ideal biomonitoring tool. These exploratory analyses demonstrate the promise that biological trait-based biomonitoring offers.

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7 Remobilization of historically contaminated sediments during high discharges – Pilot River Rhine

7.1 Introduction

During the post-war period industrial activity increased resulting in higher emission rates of organic and inorganic contaminants, causing an overall degradation of biological conditions in the Rhine over the period 1960-1970. Due to action plans and measures to regulate emissions, the concentrations of most heavy metals in the water column slowly decreased over the years. However, it is hypothesized that high discharge events may cause resuspension of these sediment layers and result in mobilization of historically deposited contaminants.

7.2 Experiment and discussion

Large monitoring datasets of the River Rhine covering actual and historical data were used to analyse the relations between discharge, suspended matter, and associated contaminants. The data used are derived from the monitoring programmes of Rijkswaterstaat (1970-2011), the German BfG - Federal Institute of Hydrology, and MWTL monitoring databases. During higher discharges, suspended matter in the water column increases significantly. As a result, contaminant loads increase. See Figures 7.1 and 7.2.

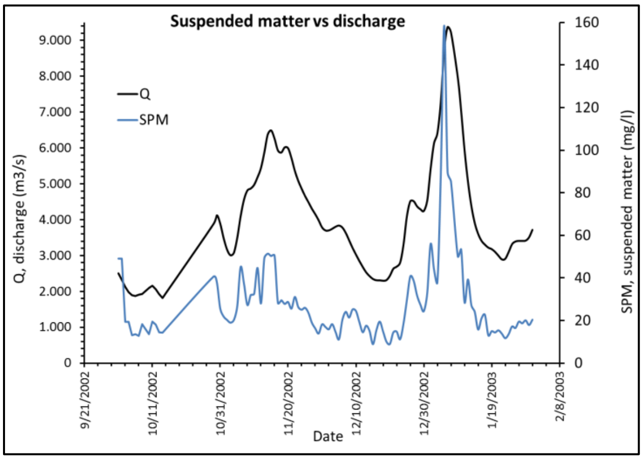

Figure 7.1. Suspended matter and discharge over time

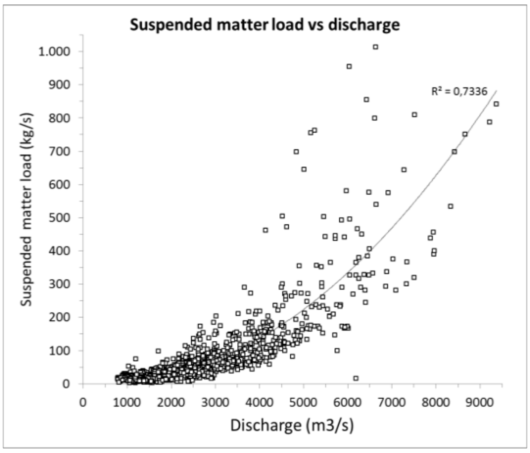

Figure 7.2. Relation between sediment load and discharge.

Measurements during high discharge events almost always show decreasing dissolved concentrations of contaminants compared to low discharge events. This seems to contradict the previous observation of increasing concentrations of suspended particulate matter (SPM). However, relations between discharge and the amount of contaminants released are difficult to establish, mainly due to: 1) data points at high and extreme discharge events are scarce; 2) quantification of the dilution effect at high discharges. The dilution effect masks the actual effects in two ways. Firstly, dissolved concentrations decrease as a result of more water at high discharge; however, the load increases. Secondly, the median size of the particles increases at high discharge, so expressions of concentrations by weight are obsolete.

Therefore, a numerical model, Delft-3D-WAQ, is used to analyse the physics behind high discharge events. Two numerical experiments are performed, one with a constant emission pattern and the other with a variable emission pattern with emission peaks between 1968 and 1975.

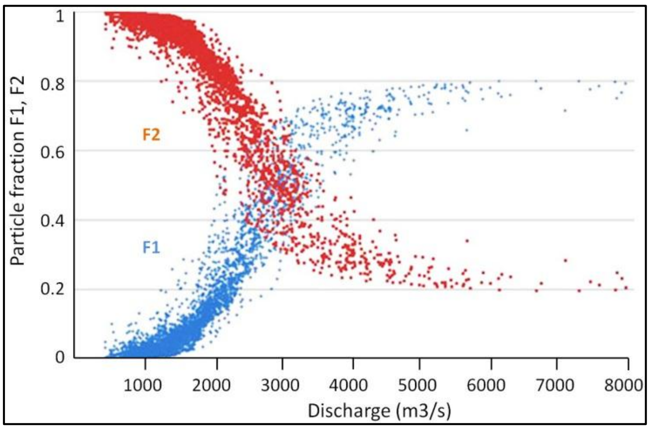

Figure 7.3. Particle fractions of SPM over discharge. F1 are large-, F2 are small-sized particles.

Masking of effects as described above were considered by correcting the SPM composition with particle size fractions (see Figure 7.3). These functions were adopted in the descriptions of sediment shear stress. In order to get a yearly load of contaminants corresponding to the loads reported in the literature, the model was calibrated by applying a constant emission of zinc. The model administrates in- and outfluxes per river segment, which enables monitoring of the total amount of contaminants passing the river system. Comparison of model calculations and monitoring data showed good agreement: a difference of less than 0.4% with the observed load between 2000 and 2004 was found. In any case, model scenarios demonstrate deterioration of suspended particulate matter quality with increasing discharge (Figure 7.4), indicating mobilization of historically contaminated sediment layers.

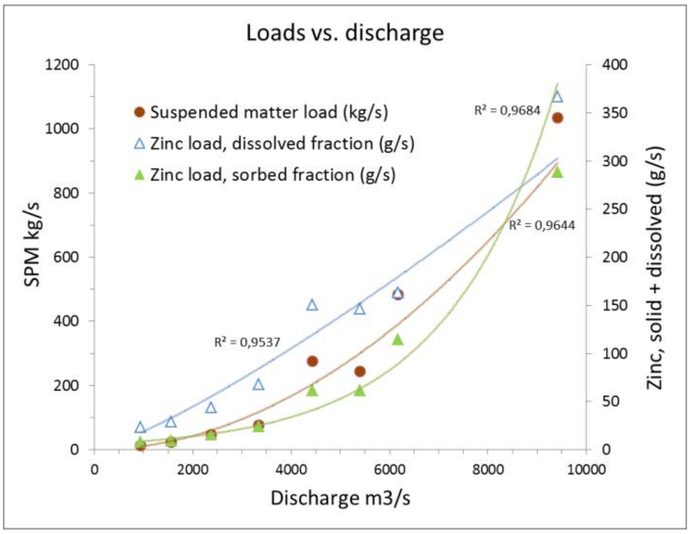

Figure 7.4 Relations between discharge, suspended matter and zinc loads.

7.3 Lessons learnt

The study showed that river water quality is not only dictated by diffuse or source point emissions of contaminants, but is also strongly related to the quality of sediments. High discharge events, which may occur more often in the future as predicted in future climate scenarios, may mobilize the associated contaminants. Increased contaminant loads at high discharge are commonly not signaled or detected by monitoring programmes because of the masking effect of particle size and dilution. An impact on ecosystem health in sedimentation areas cannot be excluded, however.

7.4 References

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8 Groundwater as a stressor of base flow and discharge dynamics in sandy catchments in The Netherlands

8.1 Introduction

In The Netherlands, a country known for its temperate climate with a long standing average net rainfall surplus of approximately 300 mm y^{-1}, streams in catchments with sandy unconfined aquifers can become at risk of failing both the environmental flow standards determined by local experts and the environmental objectives of the Water Framework Directive (WFD). During recent years (2003, 2006, 2008 and 2011) water boards in The Netherlands experienced water shortages and low flow conditions which affect both agriculture and (aquatic) ecology. This trend may persist because more frequent warm, dry summer periods are a possible outcome of future climate change for The Netherlands (Van den Hurk et al. 2006). Small streams are most vulnerable in areas with thin aquifers and limited water inlet possibilities where the water demand for agriculture, drinking water and industrial production is high. Future projections for discharge in large rivers also indicate a decrease in summer periods due to climate change, thereby limiting the possibilities for water inlet and increasing the dependency of Dutch water systems on rainfall and groundwater in the catchment (Klijn et al. 2012). Currently, various water boards are implementing measures, mostly in relation to local stream morphology, to improve the ecological status of surface water bodies in response to the WFD requirements (Verdonschot and Nijboer 2002; PBL 2008). Potentially this can have a positive effect on flow velocity and the water depth of streams. However, effects of these restoration efforts can be seriously hampered if the status of groundwater bodies is too poor to allow sufficient base flow during low flow conditions.

Within the European Framework project REFORM, Deltares investigated possibilities for improvement of groundwater conditions and stream discharge with respect to the aquatic ecology during two research projects.

The first research project consisted of a literature review, geohydrological modelling of two case study areas in the Netherlands, and a critical evaluation of the Dutch implementation of the WFD threshold values for low flow. This research project resulted in a peer-reviewed publication by D.M.D. Hendriks, M.J.M. Kuijper and R. van Ek in the special issue in environmental flow needs of Hydrological Sciences Journal. The paper is titled “Groundwater impact on environmental flow needs of streams in sandy catchments in The Netherlands”. The first paragraph of this chapter of the REFORM deliverable D3.1 contains a summary of this peer-reviewed paper.

The second research project focused on a historic evaluation of discharge and groundwater data series over a long period (1956-2003) from the Regge River in the eastern part of The Netherlands. During the past century large scale anthropogenic alterations have been made in the Regge catchment, affecting the geohydrology of the area. In the research project, an analysis was made of the relations between changes in geohydrology (groundwater levels and stream flow), climate and anthropogenic alterations in the area. This was done by a historical analysis of anthropogenic changes in the area (interviews and literature review) and time series analysis and statistical trend tests of the meteorological data, the discharge data and the groundwater level data.

8.2 Groundwater impact on environmental flow needs of streams in sandy catchments in The Netherlands

D.M.D. Hendriks, M. Kuijper, R. van Ek

This secton contains a summary of the paper published as "Hendriks, D. M. D., Kuijper, M.J.M., Van Ek, R. Groundwater impact on environmental flow needs of streams in sandy catchments in The Netherlands. Hydrological Sciences Journal, accepted. To be published 2013."

8.2.1 Introduction

Direct determination of base flow decrease as a result of anthropogenic alterations to the groundwater system is difficult, due to scale and heterogeneity. First of all, base flow is generally affected by anthropogenic alterations or measures that affect the groundwater conditions on the catchment scale. Therefore, an assessment of changes in base flow should be carried out at the catchment scale rather than at the local scale. Furthermore, availability of long time series of stream discharge is limited and discharge measurements recorded during the undisturbed situation are seldom available. Therefore, the effects of anthropogenic alterations and possibilities for base flow restoration cannot be directly deducted from existing information alone, but require some kind of model projection approach. Additionally, the effects of future changes and the sensitivity for climate change need to be assessed by model projection. Although some work has already been done, no generally accepted method has been established yet. In different countries, e.g. in UK and Denmark, various basic methods have been used in precautionary assessments or nationwide screening purposes (e.g. Henriksen et al. 2007, Ward and Fitzsimons 2008, Acreman and Ferguson 2009). However, there are no catchment specific evaluations of base flow reductions due to various anthropogenic stressors like groundwater abstraction, artificial drainage, and climate change impacts. In addition, no generally accepted threshold value for minimum base flow or maximum base flow reduction exists. In relation to objectives of the Water Framework Directive, it has been suggested that a threshold for significance is where more than 50% of the allowable surface water abstraction within the total upstream catchment can be attributed to groundwater (European Commision 2009, Blum et al. 2009). However, no evidence has been given to support this value and a methodology for quantification of the impact of such stream discharge reductions is mostly lacking.

The goal of our study was to present a methodology that quantifies the effects of past and future anthropogenic alterations and climate change on base flow of streams. Additionally, our aim was to test if past, current and future base flow satisfies EFN threshold values and to determine tolerable base flow loss of streams in sandy catchments. To test the proposed method and to investigate the magnitude of the effects of anthropogenic alterations and climate change, modeling studies were carried out for two well documented sandy catchments in The Netherlands: the Merkske catchment and the Hollandse Graven catchment (Figure 8.1).

Figure 8.1. Location of the case study areas in The Netherlands.

8.2.2 General methodology

To assess the impact of anthropogenic alterations on base flow and to determine the ecological status of the streams with respect to base flow, the following steps were taken: First, information and data on the geohydrology, discharge regime, stream morphology, anthropogenic alterations and climate of the catchment were gathered. Next, EFN threshold values for base flow were established, based on the methodology described below. Then, a geohydrological model was made and simulations were done for the past, undisturbed situation, for the current situation, and for scenarios with various anthropogenic alterations. Additionally, various future scenarios including climate change prognoses and anthropogenic alterations were assessed with the geohydrological model in order to assess the base flow in worst case situations when the conditions of the groundwater body are further degraded. Finally, base flow was calculated from all model runs at various scales (primary, secondary and tertiary streams). A subsequent comparison of the base flow values from various model calculations and a comparison of modeled base flow with the EFN threshold values provides (1) an indication of the level of base flow reduction due to alterations in the catchment, (2) the extent to which the groundwater body currently supports the surface water body in maintaining sufficient water depth and flow rate during dry periods, and (3) the relative effects of anthropogenic alterations on base flow, compared to possible climate change effects (sensitivity analyses).

8.2.3 Description of case study areas

The *Merkske catchment* is situated in the southern part of The Netherlands in the province of Noord-Brabant on the border with Belgium and covers an area of approximately 60 km2 (Table 8.1). Discharge measurements of the main stream at the outlet of the Merkske catchment are available for the period 2004 to 2008. Average discharge in this period was

0.605 m3 s-1. During winter months stream discharge showed peaks of 3.0 to 4.0 m3 s-1, while during dry summer periods stream discharge reduces to 0.10 m3 s-1 and less (internal comm. water board Brabantse Delta, Hendriks and Van Ek 2009). The geohydrology of the area is characterized by a thin phreatic aquifer (2 - 10 m) consisting of sand, on top of a layer with low permeability (thickness approximately 40 m) consisting of fine organic rich river sediments. This in turn overlies a semi-confined aquifer (thickness > 200m) consisting of marine sediments. Base flow is generated mostly by the continuous seepage flow from the semi-confined aquifer (Figure 8.2). Since 1850 large areas of the catchment have been cultivated, resulting in an extensive network of ditches, trenches and tile drainage. In addition, several stream trajectories have been straightened, and dams and weirs were installed. Due to these changes, the surface areas of both infiltration and seepage zones in the catchment have decreased. A few kilometres outside of the catchment two large groundwater abstractions are located that abstract water from the semi-confined aquifer. During the summer period groundwater abstraction for irrigation takes place from the shallow part of the semi-confined aquifer at approximately three wells per km^2. Land use in the catchment consists of grasslands, mixed forests, agriculture, some heathland and some small villages (De Louw and Stuurman 2000, Van der Velde and De Louw 2006, Kuijper et al. 2012).

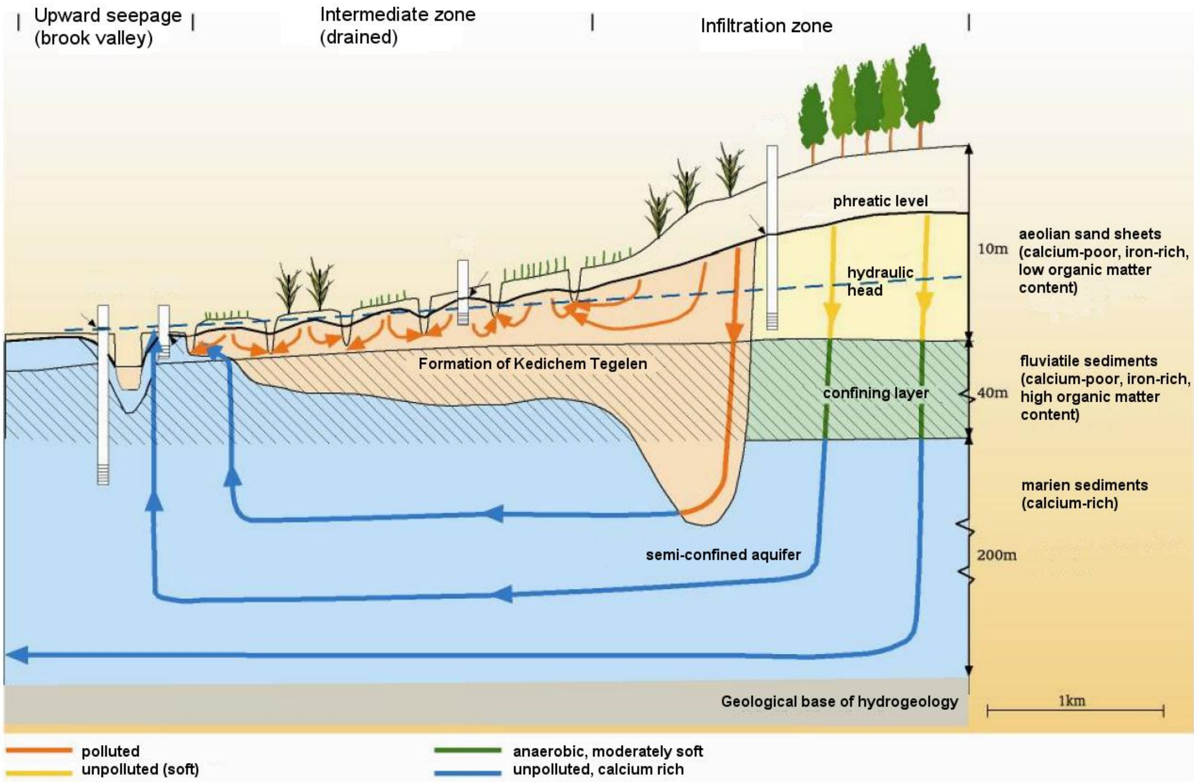

Figure 8.2. Schematic representation of the regional groundwater system of the case study area Merkske catchment.

The *Hollandse Graven catchment* is part of the larger Dinkel catchment and covers a catchment area of approximately 62 km^2 (Table 8.1). Discharge of the main stream was measured at the outlet of the Hollandse Graven catchment over the period 2000 to 2011. Average discharge in this period was 0.370 m3 s-1. During winter months stream discharge showed peaks of 4.5 to 5.5 m3 s-1, while during dry summer periods stream discharge reduces to 0.03 m3 s-1 and less. In the summer of 2003 the stream became practically stagnant. Only the use of weirs prevented parts of the stream from falling dry (Kuijper et al.

2012). The morphology of the area is characterized by the hill slopes of two clay-rich ice-pushed ridges. The main aquifer consists of a thin phreatic system of Pleistocene sands deposited on top of a thick, low permeability layer of moraines and marine sediments. On the hill slopes, the phreatic aquifer is relatively thin or absent. Base flow originates mainly from tributaries discharging seepage from hill slope springs, some of which show continuous discharge, others fall dry in summer months due to thin aquifers and intensive agricultural drainage. Additionally, year round effluent of treated sewerage water contributes to the streams base flow (Figure 8.3). Nowadays the main land use is agriculture: pasture and corn. The groundwater dependent nature areas are small and mainly connected to seepage zones around secondary or tertiary streams. Furthermore, the Hollandse Grave catchment includes the town Ootmarsum and some smaller villages (Kuijper et al. 2012).

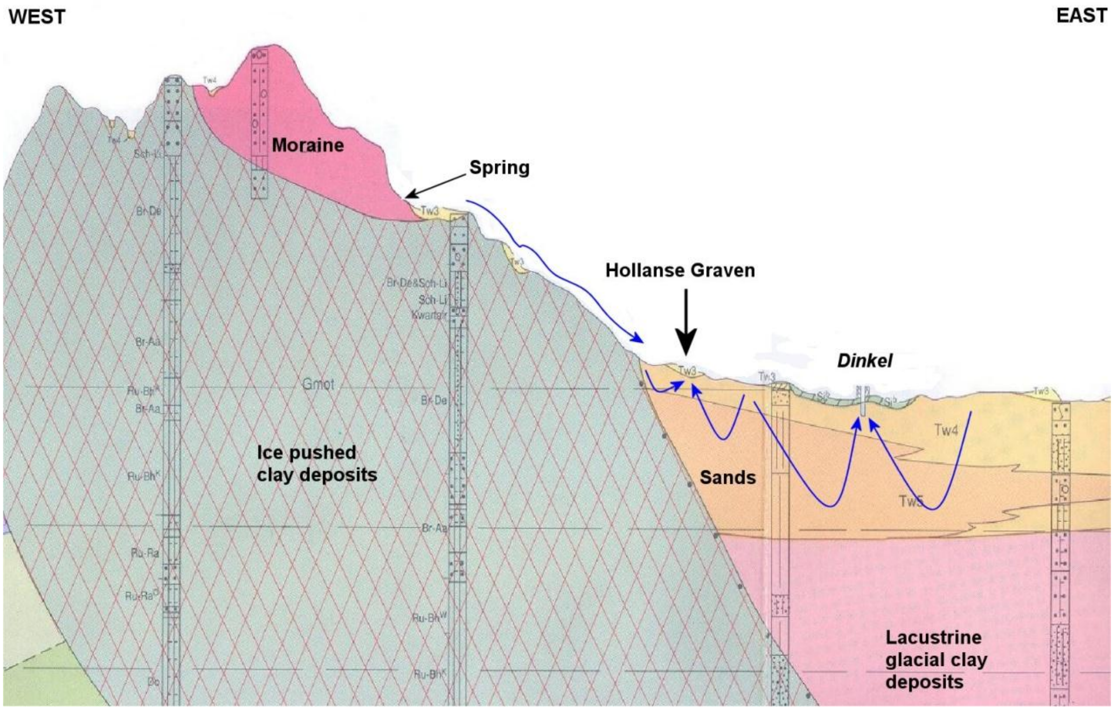

Figure 8.3. Schematic representation of the regional groundwater system of the case study area Hollandse Graven catchment

Table 8.1. Characteristics of the two case study areas (Merkske catchment and Hollandse Graven): geohydrology, hydromorphology, hydrology, and groundwater abstraction.

	Merkske catchment	Hollandse Graven catchment
surface area (km^2)	59.5	62.33
altitude (m + sea level)	11 - 31	19 - 78
slope gradient (m per 100 m)	0.13	0.7 - 4
thickness phreatic aquifer (m)	2 - 10	2 - 15
lithology phreatic aquifer	sand (aeolian)	sand (aeolian; moraines; fluvial)

	Merkske catchment	Hollandse Graven catchment
land use type	agriculture	agriculture, nature, urban
stream type (WFD)	R4	R4
stream depth (m)	0.1 - >2	0.1 - >2
stream width	1 - 4	1 - 5
large abstractions 2003 (mm per year)	--	outside catchment
spray irrigation 2003 (mm per year)	--	35
large abstractions 2005 (mm per year)	outside catchment	outside catchment
spray irrigation 2005 (mm per year)	*6 (0.133 mm d^{-1} during irrigation period)*	30

8.2.4 EFN threshold values for base flow

With respect to EFN, threshold values for minimum flow rate and water depth are relevant hydrological conditions and have been subject of ecological studies (e.g. Verdonschot and Nijboer 2002, Verdonschot and Van den Hoorn 2010, Verdonschot et al. 2012). In our study, we used generic limits of the EFN parameters based on the Dutch implementation of WFD requirements for reference stream types as well as local expert knowledge (Van der Molen and Pot 2007, Samuels and Van Nispen 2008, Altenburg et al. 2012, water board Regge and Dinkel 2010). Values are provided for minimum stream flow velocity and minimum water depth required to meet the EFN or the required aquatic ecological status of stream types (Table 8.2). By combining this information with the information on the stream morphology, the base flow (discharge) that is needed to sustain minimum flow velocity and water depth was determined for primary and secondary streams.

Table 8.2.Hydromorphological requirements of stream type R4 (continuously slow flowing headwater on sand) and base flow threshold values for Merkske and Hollandse Graven catchment.

parameter	EFN threshold values Merkske	EFN threshold values Hollandse Graven	
Requirements R4 stream type			
based on	WFD references	WFD references	local experts (WRD, 2010)
minimum water depth (m)	0.02	0.02	0.05
minimum flow velocity (m s^{-1})	0.03	0.03	0.15
Local stream widths (m)			

parameter	EFN threshold values Merkske	EFN threshold values Hollandse Graven	
1st order streams	3 - 5	5	5
2nd order streams	2 - 4	-	-
Base flow threshold (m^3 s^{-1})			
1st order streams	0.0018 - 0.0030	0.003	0.038
2nd order streams	0.0012 - 0.0024	-	-

8.2.5 Geohydrological modeling

Like in most areas, no data were available from the undisturbed situation in our study catchments. Therefore, the groundwater conditions and discharge regime preceding anthropogenic alterations was estimated using a deterministic spatially distributed hydrogeological model. In The Netherlands, on-line coupled modeling tools are developed and maintained to coherently model the interconnected hydrological system on the local to regional scale (e.g. the MIPWA instrument: Berendrecht et al. 2007). Because of the use of highly detailed spatial information this model type was considered to be suitable for estimating the effect of various measures, under current and future meteorological conditions, on groundwater and surface water regimes. In our case, such MODFLOW models were already available and validated. For further information on the geohydrological modeling we refer to Hendriks et al. (2013, HSJ accepted). An overview of specific model scenarios is given in Table 8.3

Table 8.3. Overview of model scenarios of the Merkske case study and the Hollandse Graven case study including anthropogenic alterations and climate change.

	intensive drainage	large groundwater abstractions	groundwater abstraction for spray irrigation	3 times total groundwater abstraction	Year (climate projection)
Scenarios Merkske case study					
undisturbed situation	--	--	--	--	2005
current situation	X	X	X	--	2005
scenario 1	X	--	--	--	2005
scenario 2	X	X	--	--	2005
scenario 3	X	--	--	X	2005
scenario 4	X	--	--	--	2050 (W+)
scenario 5	X	X	X	--	2050 (W+)
scenario 6	X	--	--	X	2050 (W+)
scenario 7	X	--	--	--	2100 (W+)
scenario 8	X	X	X	--	2100 (W+)

	intensive drainage	large groundwater abstractions	groundwater abstraction for spray irrigation	3 times total groundwater abstraction	Year (climate projection)
scenario 9	X	--	--	X	2100 (W+)
Scenarios Hollandse Graven case study					
current situation (dry)	X	--	X	--	2003
current situation (average)	X	--	X	--	2005
scenario 1 (dry)	--	--	X	--	2003
scenario 1 (average)	--	--	X	--	2005
scenario 2 (dry)	X	--	--	--	2003
scenario 2 (average)	X	--	--	--	2005
scenario 3 (CC, dry)	X	--	X	--	2050 (W+)
scenario 4 (CC, average)	X	--	X	--	2050 (W+)

8.2.6 Model results

Results of all model scenario calculations and the EFN threshold values for base flow (Q95) of the are summarized in Figure 8.4. For the Merkske case study, model results showed that Q95 in the main stream has decreased with 35-45% compared to the undisturbed situation. Base flow is reduced with 25-35% due to artificial drainage with tiles and manmade ditches. In addition, current groundwater abstractions cause a Q95 reduction of approximately 10%, of which half is caused by the spray irrigation during the summer. The effect of the large abstractions near the catchment is small, although the water is abstracted from the same aquifer that is underlying the Merkske catchment. The influence of the abstraction is confined to the area of the abstraction cone and therefore affects only a small part of the Merkske catchment. Additionally, because the aquifer that provides the groundwater is very thick (>200 m) the decrease in pressure head in the confined aquifer is limited. Model scenarios of future developments showed further substantial decrease of base flow. In case of the climate change projections, Q95 is reduced with approximately 33% in 2050 (red line) and with approximately 44% in 2100 (blue line) at catchment scale compared to the current situation (upper graph, Figure 8.4). Future scenarios with increasing groundwater abstraction show a more or less linear Q95 reduction with increasing abstraction discharge. According to the model results, the base flow of the main stream in the Merkske catchment is currently sufficient to maintain the required EFN, but ne threatened in the future.

Model results for the Hollandse Graven case study showed that under the current situation, Q95 in the dry year 2003 is approximately 12 % of the Q95 in the average climatic year 2005 (Figure 8.4). Further, model results showed that Q95 in the main stream has decreased with approximately 30 % due to artificial drainage in 2003 and with approximately 16 % in 2005. The groundwater abstraction for spray irrigation has caused a Q95 reduction of approximately

55 % in 2003 and a reduction of approximately 5 % in 2005. Climate change also has a large potential impact on base flow in the Hollandse Graven catchment. Model results showed that, under the current management, Q95 values will have decreased by approximately 57 % in 2050. Modelled base flow in the Hollandse Graven is equal to or greater than the minimum EFN threshold values based on Dutch WFD standards for the R4 stream type for all scenarios. A comparison with the minimum base flow value based on local expert knowledge, gives a different impression of the ecological status of the stream (Figure 8.4). For all model scenarios based on the meteorological conditions of 2003 and the for the KNMI 'W+' climate change prognoses, the Q95 value drops below this base flow minimum (0.038 m3 s-1). This indicated that the ecological status required by the local water managers is seriously at risk in dry years.

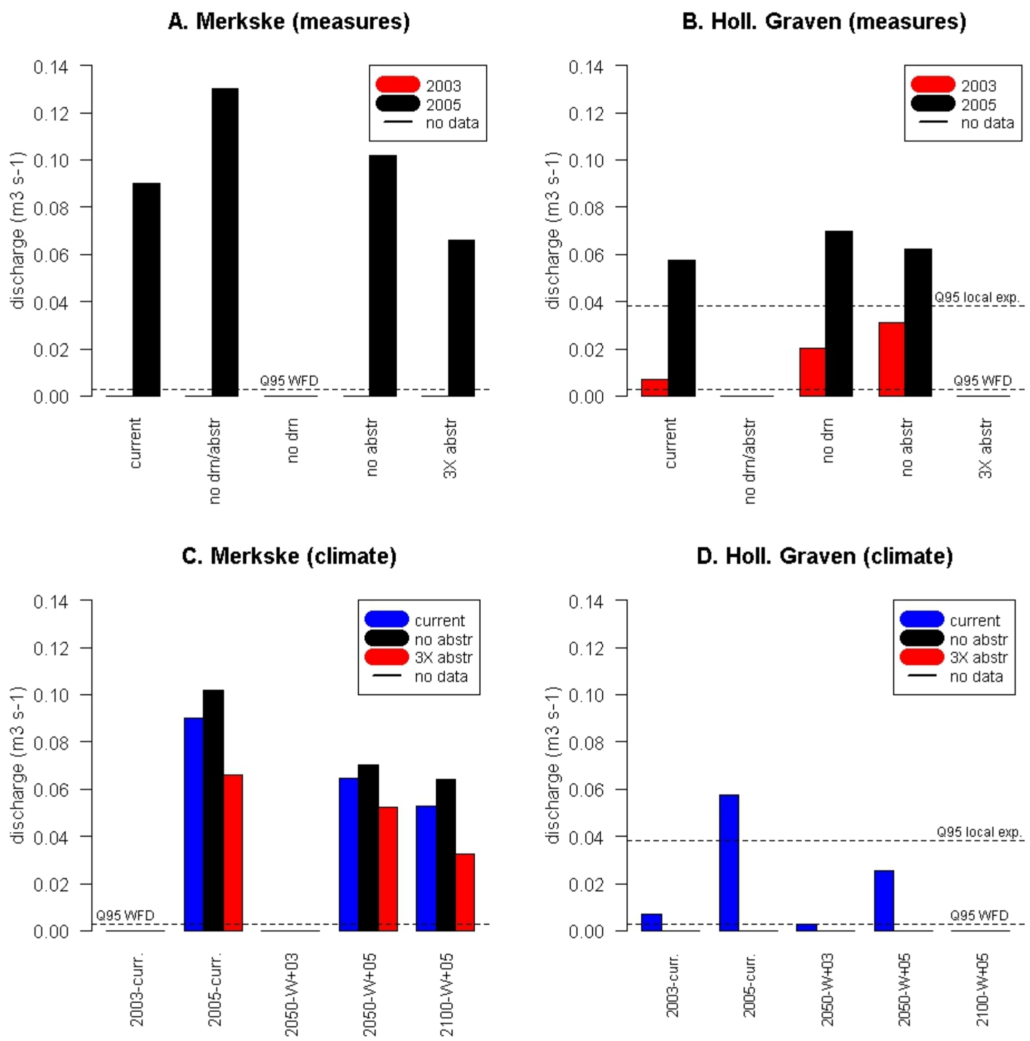

Figure 8.4. Results of simulated base flow (Q95 values) for the total Merkske catchment (graphs: A and C), and for the Hollandse Graven catchment (graphs B and D). In graphs A and B, the results of the model simulations for all scenarios including anthropogenic alterations are shown for 2003 (only Hollandse Graven) and 2005: current drainage and abstraction (current), tile drainage, ditches and abstractions removed (no drn/abstr), only tile drainage and ditches removed (no drn), abstractions removed (no abstr), tripled abstractions (3X abstr). In graphs C and D, the results of the model simulations for all climate scenarios are shown. In the Merkske case study, climate scenarios are combined with anthropogenic impact. The dashed lines indicate EFN threshold values for minimal base flow based on the Dutch implementation of WFD and on local expertise (only Hollandse Graven).

8.2.7 Conclusions

Determination of EFN threshold values for base flow is still under development. In this paper we proposed a method to derive and evaluate EFN threshold values for base flow from minimum required flow velocity and minimum water depth as determined in national implementations of Water Framework Directive (WFD) standards. The method, which comprises scenario analyses using detailed geohydrological models, has proven applicable and useful to determine the ecological status of streams with respect to base flow. However, it was also found that environmental flow standards determined by local experts may be more ambitious than the environmental objectives from WFD standards, as local water managers take into account specific catchment characteristics, local knowledge and a combination of ecological and agricultural purposes.

Scenario analyses in two sandy catchments in The Netherlands according to the proposed method showed that anthropogenic alterations have caused a significant impact on base flow in slow flowing streams in sandy catchments. This study showed that compared to the historic, undisturbed situation, base flow has been reduced significantly. The current artificial drainage (tile drainage and ditches) caused 25-40% base flow reduction, while groundwater abstractions caused a reduction of 5-28%. As a result it could be concluded that the resilience of the studied catchments has decreased as a result of the anthropogenic alterations made during previous centuries.

The reduced resilience of the water systems hampers the ability of the catchments to generate sufficient base flow during dry periods, and as a result the ecological status is likely to be increasingly at risk. In case of climate change for example, maintenance of base flow for EFN might become increasingly difficult. Model results showed that the driest climate change projection (‘W+’ scenario KNMI) potentially causes a base flow reduction of 33–70% in 2050 compared to the current situation.

Most vulnerable are streams in catchments or sub-catchments with thin aquifers, limited natural groundwater-surface water connectivity, low groundwater conductivity, or small surface areas. Our results are considered representative for sandy lowland catchments in different hydrogeological settings. These hydrogeological differences are reflected in base flow characteristics. Catchments with thick aquifers, like the Merkske catchment, have a stronger resilience towards drought and anthropogenic alterations. They respond more slowly and are more robust under drought conditions. The Merkske catchment, with its thick semi-confined aquifer and gentle slopes, shows a relatively high base flow under average meteorological conditions. In the Hollandse Graven catchment, the phreatic aquifer is underlain by a thick confining layer consisting of moraines and hill slopes are relatively steep. These characteristics cause limited groundwater-surface water connectivity, and result in a relatively low natural base flow. Consequently, meteorological droughts and anthropogenic alterations have a larger impact on the water system of the Hollandse Graven catchment, resulting in larger base flow reductions. Even within a catchment hydrogeological differences exist, that cause different base flow responses to anthropogenic alterations and climate change. For example, in the Merkske catchment areas with a relatively thin aquifer (sub-catchment 1) show a stronger decrease in base flow due to groundwater abstractions and climate change than other sub-catchments.

8.2.8 Discussion: EFN threshold for base flow too low

Our results show that the ecological statuses of the main streams in the case studies are not at risk when EFN threshold values based on Dutch WFD references are used. However, during a workshop with water managers, this minimum EFN threshold for base flow was seriously questioned. Water managers found that under current circumstances, base flow in the main streams of the studied areas is too low for the assigned ecological and agricultural functions. For the Hollandse Graven catchment, local experts of water board Regge en Dinkel (WRD) established other minimum values for water depth and flow velocity (WRD 2010). Based on these values, the minimum EFN threshold for base flow is more than 10 times higher than the threshold based on Dutch WFD references, indicating that the ecological status of the main stream is at risk indeed.

The threshold values defined according to the Dutch implementation of the WFD are fairly generic and have been established based on different sources mainly in relatively undisturbed reference catchments (Van der Molen and Pot 2007; Samuels & Van Nispen 2008; Altenburg et al. 2012). All surface water bodies classified as R4 (slow flowing streams) have the same threshold value for minimum flow velocity and water depth. In reality the R4-type surface water bodies differ in size, shape and geohydrological setting. Therefore, it is reasonable to expect that local expert knowledge leads to different threshold values for a specific R4-type surface water body. However, it should be recognized by water managers that local threshold values can differ substantially from national precautionary thresholds. Moreover, it could be argued that WFD threshold values for EFN should be based on more locally based, precautionary assessments and that specifications should be given for a gradient of geohydrological situations, catchment sizes, and level of anthropogenic alteration. For the two catchments in our analysis, local experts regarded the minimum hydrological threshold values based on Dutch WFD standards as large underestimations. Therefore, it may be valuable to reassess the hydrological boundary conditions through a broader inventory of ecological indicators in order to determine more realistic threshold values for environmental flows.

Recent research results have shown that for aquatic ecology in slowly flowing streams, besides thresholds for minimum and maximum flow velocity and water depth, a limited variation in flow velocity is important (Verdonschot et al. 2012). Anthropogenic alterations like intensive drainage systems, straightened streams, artificial meanders, and deepened stream beds may affect both low flow and high flow events. Additionally, climate change affects peak flows and flow variability: the KNMI 'W+' climate projection includes an increase of extreme precipitation events, probably causing more frequent peak flow events in streams. We therefore suggest that the range of flow velocities in a stream should be assessed in coherence, including catchment-wide sensitivity analyses of the impacts of the status of groundwater bodies and the effects of anthropogenic measures and climate change on both low flows.

8.3 Historical data assessment of the impact of groundwater and catchment scale alterations on discharge dynamics of the Regge Catchment

8.3.1 Introduction

During the previous study (section 0), evidence was found for the effects of catchment-wide anthropogenic changes and climate on groundwater level and base flow. However, this was achieved through modeling assessment and uncertainties remain. To strengthen our findings, te next study assessed the impact of catchment-wide anthropogenic changes on groundwater, discharge dynamics and base flow using data series of meteorology, groundwater levels and discharge. In addition, a historical analysis of large scale changes in the catchment was carried out and findings were linked to the data series analysis.

For this purpose, we chose a catchment area in which a REFORM case study is being carried out (WP4 Regge case study, Alterra) and on which long term data series are available (1956 – 2003). In consultation with Alterra (Piet Verdonschot), we analysed not only effects base flow but a larger range of flow parameters and variability characteristics that are relevant for aquatic ecology).

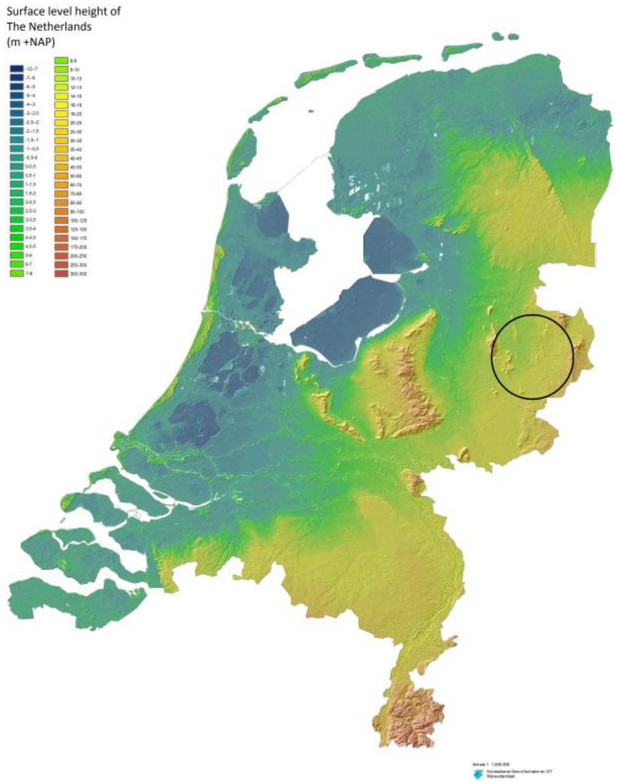

Figure 8.5. Overview of The Netherlands and its surface level height in m +sea level (NAP). The location of the Regge catchment is in the encircled area.

8.3.2 Methodology

Description of the Regge River and its catchment

The Regge catchment is situated in the eastern part of the Netherlands (Figure 8.5) and has a temperate marine climate zone with long standing mean precipitation of 800 - 850 mm per year, long standing mean temperature of 9.3 - 9.9 ^{o}C, and long standing mean evaporation of 560 - 570 mm per year (Royal Netherlands Meteorological Institute KNMI). The Regge catchment covers a area of approximately 87.4 km^2. The basin is characterised by a north western slope of 35 to 65 m, reaching from the eastern head waters (approximately 40-70 m +NAP) to the north western downstream area near the town of Ommen (approximately 5 m +NAP). The Regge River used to originate in Germany, but by anthropogenic impacts the upstream parts were cut off in the early 20th century. Near Ommen the Regge merges into the river Vech (Figure 8.6). With approximately 60% agriculture represents the main land use in the area. Besides these agricultural areas most of parts of the river basin are characterised by rural, small scale forest and nature area. Larger urban areas in the catchment are the towns of Enschede, Almelo and Hengelo (De Louw 2006).

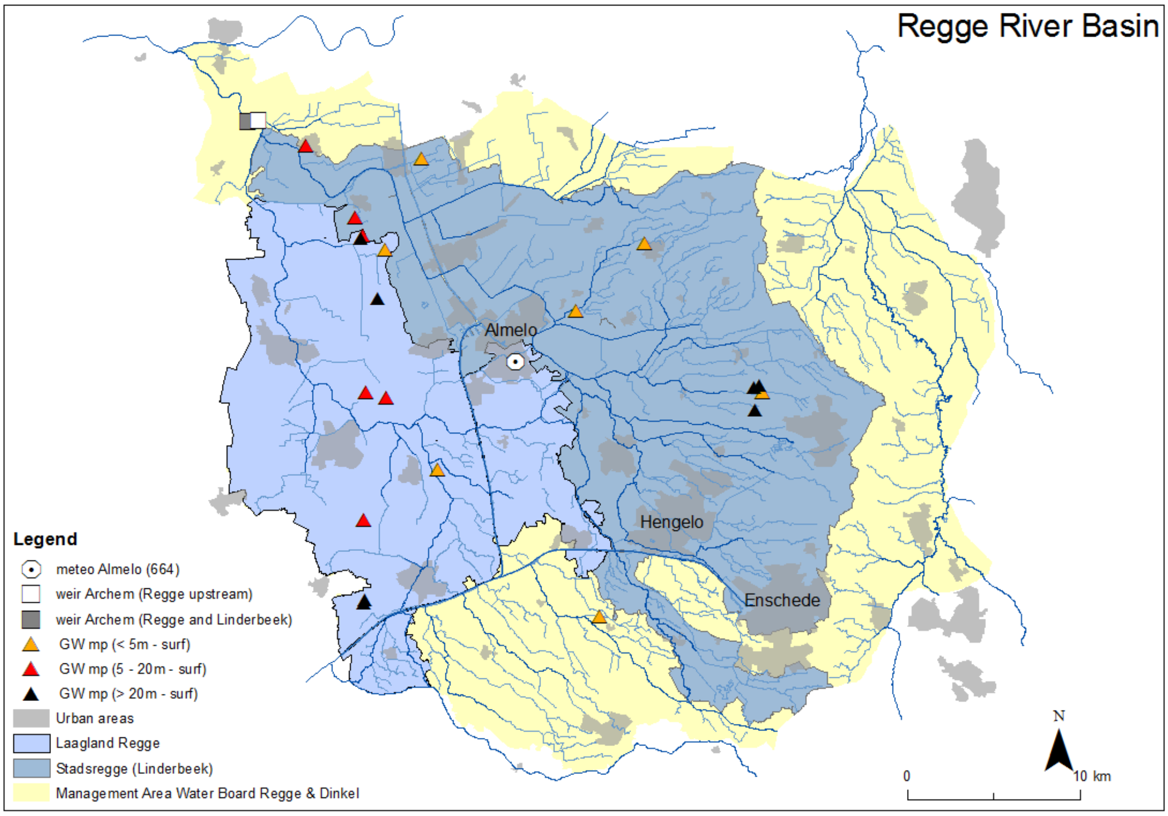

Figure 8.6. Regge catchment with main sub-catchments, streams, urban areas and measurement locations of meteorological station, weirs for discharge measurements, and groundwater level observations.

The geohydrological structure of the river basin is characterised by significant differences from east to west. In the western part the subsurface consists of thick, aquiferous sediments that reach a depth of approximately 150 m and are intervened by impermeable clay deposits, whereas the geohydrological base in the eastern part reaches a depth between 10 – 20 m below the surface and is determined by small scale terminal moraines and a low infiltration capacity (De Louw, 2006). Precipitation represents the main input in the water balance (95%). In addition, water is introduced into the catchment from large rivers. Loss of water from the catchment occurs mainly through evapotranspiration (55%) and discharge of surface water (37%). Also some large groundwater abstractions are located in the area, which cause significant loss of water.

The catchment of the Regge itself can be subdivided in two main sub-catchments with different characteristics (Figure 8.6). The main stream of the Regge River originates from the south-western sub-catchment 'Laagland Regge.' This is the second largest sub-catchment (40.4 km^2) and is characterised by rural landscape and nature areas. The largest sub-catchment (47 km^2) is the 'Stadsregge' and is found in the eastern part. Dominated by mainly urban areas of the catchment, this main tributary of the Regge River is called 'Linderbeek'. Other tributaries of the Regge River are the Hagmolenbeek (De Louw 2006; Reggevisie 1998).

Water management in the Regge catchment is the responsibility of the Water Board Regge and Dinkel, located in Almelo. Established in 1884 the water board is in charge for most aspects of water management, combining various interests like environment, agriculture, nature, etc. During the previous 150 years the landscape of the Regge catchment has been influenced to a large extent by agriculture; hence also the measures of the water board and farmers on water management were driven by agricultural interests. Big changes in these management approaches occurred during the 1980s towards a more nature orientated river basin management (Huitema 2002). Especially during the last 15 years the implementation of measures and re-naturalization became major part of water board's activities. Detailed information on anthropogenic influence on the Regge catchment is found in section 0.

8.3.3 Data acquisition

In this study we combined meteorological data with geohydrological data (groundwater levels and discharge) and historical information on anthropogenic alterations in the catchment. Below, information on data acquisition is summarized, and in Table 8.4 an overview is given of all available data that were used in this study.

Meteorological time series of precipitation and potential evaporation were collected from the database of the Royal Netherlands Meteorological Institute (KNMI). KNMI weather station 'Almelo' was used for the precipitation data because it provided measurements over the period 1951-2013 and was located in the centre of the Regge catchment (see Figure 8.6). Potential evaporation data were taken from KNMI weather station 'De Bilt', because this is the only station with long time series (1950-2010).

Table 8.4. Summary of meta-data of the available meteorological, geohydrological and historical observations from the Regge catchments.

Data type	Number of locations	Location	Measurement frequency	Measurement Period	Source
Precipitation	1	Almelo	daily	1951-2012	KNMI
Evapotranspiration	1	De Bilt	daily	1902-2012	KNMI
Groundwater level < 20 m -surf	13	see map (fig. 4)	daily to two times per month	1951-2012	TNO (DINO)
Groundwater level 20 - 30 m -surf	7	see map (fig. 4)	daily to two times per month	1949-2005	TNO (DINO)
Discharge measurements	2	Archem	daily	1956-2003	Waterboard Regge and Dinkel, Province Overijssel
Changes in water management	--	--	--	~1934-1995	Waterboard Regge and Dinkel
Changes in stream morphology	--	--	--	1924-1995	Waterboard Regge and Dinkel
Changes in landuse	--	--	once per year (31 december)	1951-2012	statistical office Netherlands
Population development	6	Almelo, Enschede, Hellendorn, Borne, Haaksbergen , Hengelo	once per year (31 december)	1951-2012	statistical office Netherlands

Groundwater level time series were collected from the database of the Dutch Applied Research Institute (TNO). Initially, time series from within the Regge catchment were selected based on the following criteria: starting date before 1965, length of time series at least 35 years, and minimum a bimonthly measurement frequency. Then, in order to avoid the influence of local, shallow interferences measurement locations with filter depths of less than 5 m below surface were removed. Additionally, measurement locations with inexplicable disturbances in the time series were removed from our analysis (spikes, large step-changes, data gaps). Finally, we kept a dataset of 21 groundwater level time series spread over the Regge catchment (Figure 8.6).

Discharge time series of the Regge River were provided by the Water Board Regge and Dinkel and the Province of Overijssel. At weirs near Archem, where the main tributary Linderbeek flows into the main Regge stream, discharge has been recorded since 1956. Two datasets are available. The first dataset, which is named 'Regge and Linderbeek' (RL), covers the whole catchment and contains measurements over the period 1956-2003. The other dataset, which is named upstream Regge (RU), is collected by at a weir positioned before the confluence of the main Regge stream and Linderbeek stream and covers the western part of the Regge over the period 1974-2003. Figure 8.6 gives an overview of the sub-catchments RL and RU and the positions of the weirs. Both data series show a large data gap during the years 1984-1989 and measurement techniques differ before and after this period. For the period 1956 to 1983 water level measurements were made once a day at 08:00 and converted to daily discharge records by a Qh-relation:

$$Q = 6.846(h - 2.0)^{1.832} \qquad \text{(eq. 1)}$$

From 1990 to 2003 daily average data were made available by acoustic discharge measurements.

Based on literature studies and interviews, an overview of the anthropogenic alterations in the catchment was made for the period 1950 – 2010. The overview represents two categories: alterations of the main streams (e.g. deepening, canalisation, re-meandering) and large scale alterations in the Regge catchment (e.g. land use change, implementation of artificial drainage, large scale groundwater abstraction, urbanisation, relocation of land). Both groups of anthropogenic alterations have an effect on stream discharge dynamics. These large scale alterations in the Regge catchment affect the discharge mainly through changes in groundwater conditions and changes in the partitioning of precipitation to infiltration and fast runoff.

Data series analysis

In this study we combined various data series analysis techniques in order to assess the effects of climate and anthropogenic alterations on the hydrogeological system: calculation of flow parameters, time series analyses using impulse-response modeling of groundwater data series, and calculation of the standard index (or 'Drought Index') of precipitation (SPI), evaporation (SEI), and discharge (SQI). Finally, we applied statistical trend tests to detect non-stationarity in the time series of precipitation, evapotranspiration, groundwater levels and discharge. Below, the various techniques are described in detail.

With respect to environmental flows, it is required that the discharge through a stream is further specified as *flow parameters* and that the variability of a stream is quantified (WMO 2008:47, Verdonschot 2010:1493). Additionally, in order to study groundwater related flow (base flow) we needed parameters that characterise low flow periods of a stream. For this purpose we calculated a number of flow parameters for each year of the analysis period (1956-2003) that had more than 90% data coverage: mean discharge, median (*Q50*) discharge, high flow parameters (*Q25* and *Q5*), low flow parameters (*Q95* and *Q75*), and flow variability (*Q95/Q5*, *Q75/Q25*, *Q95/Q50*, and *Q75/Q50*). These '*Qx*' flow parameters describe the number of exceedances of a predefined discharge within a specified time period (in our case: one year).

In our study we calculated Standard Indexes (or 'Drought Indexes') based on the available time series of precipitation, evaporation and discharge. The 'Drought Index' is a prime variable for assessment of effects of a drought and defining different drought parameters (intensity, duration, severity, spatial extend). It is a probabilistic index based on long-term data records, which are transformed to a normalised distribution for the full record period. This means that the mean value of the data record obtains a value of zero. Data values that are one standard deviation below the mean obtain a value of -1; values of one standard deviation above the mean obtain a value of +1. Hence, when this is applied to discharge or precipitation, negative values are indications of dry conditions and positive values of wet conditions. With severity of the events also the index gets more extreme. In the literature, transformed precipitation time series are known as the Standard Precipitation Index (SPI) (McKee et al. 1993; Mishra 2010:207). However, it can be adjusted to analyse other hydrological variables; therefore, we call the general version the Standard Index. In this study we used the Standard Index also to assess the time series of evaporation (SEI) and discharge (SQI). The Standard Index can be used at various timescales (Mishra 2010:207); in our analyses a time interval of 10 days ('decade') was used. For the SPI we thus used the 10-day precipitation sum, for SEI the 10-day evaporation sum, and for SQI the 10-day average discharge.

Changes in the condition of the groundwater bodies were analysed with *impulse-response modeling* of groundwater level data series and discharge data series. For this purpose the METRAN technique was applied (Berendrecht 2004). The principle of the impulse-response model is displayed in Figure 8.7. A measured data series is presumed to be compiled of various components. The first component results from meteorological impulses can be estimated from local measurements of P and ET. Other components can be groundwater abstraction, local surface water level fluctuations other changes in water management or land use changes that affect the groundwater level.

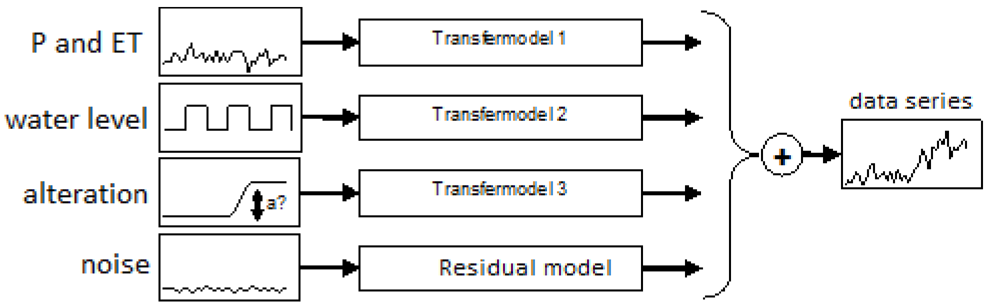

Figure 8.7. Schematic presentation of an impulse-response model.

Both the impulse-response model and the residual or noise model function according to an autoregressive principle: the response on an impulse (e.g. precipitation peak) proceeds over time and the measurement is not only caused by the impulse at that instance, but also by previous impulses. The effect of an impulse attenuates over time, and this process is described by an exponential decrease in the METRAN technique. The attenuation length is dependent on the inertia of the geohydrological system. For example, when concerning groundwater: a catchment with a thick (semi) phreatic aquifer has a slow response compared to a catchment with a thin (semi) phreatic aquifer.

In this study, no other components than P and ET were added to the impulse-response model;

the residual series (the unexplained part of the data series) was analysed for the occurrence of changes and trends. The residual consists partly of noise due to measurement errors. In addition, effects from anthropogenic changes on the land use and stream morphology might be observed in the residual as a step change or a trend.

There are several *statistical trend tests* available for testing stationarity of time series (Hirsch et al. 1993). These tests start from a null hypothesis that the observations are samples from a stationary process. The likelihood of this hypothesis is evaluated based on the value of a test statistic, a property of the dataset. A large deviation of the test statistic from the stationary value is unlikely to be coincidental. The P-value is the probability that the deviation of the test statistic from the homogeneous case is coincidental. If this probability is sufficiently small, then the null hypothesis is rejected and the alternative hypothesis is believed to be true: that the process is non-stationary. An A P-value of 5%, which we used in this study, is a common critical value for accepting statistical significance. Rejection of the null hypothesis at the 5% significance level means that we are 95% confident of non-stationarity. In this study, we were interested in both positive and negative trends. The significance levels were therefore set for a two-sided test. For thorough testing of our data series, we employed three types of statistical trend tests. Below, the tests are briefly described.

The classical Student's t-test evaluates the significance of the correlation between the values of the AM discharges and their years of observation. Pearson's correlation coefficient ρ is calculated from the covariance and standard deviation of both variables. Student's t-test is then used to test the P-value of the test statistic ρ. This is a parametric test, because it assumes that ρ follows a Student's t distribution. If this assumption is not true, the conclusions may be invalidated.

The Spearman's rank correlation test is the non-parametric analog of the Pearson t-test. The test statistic is Spearman's rank correlation coefficient r_s, which is the correlation between the ranks of values and their date of observation. Because of the use of ranks instead of the absolute values, the sampling distribution of r_s for a stationary process can be calculated without the assumption of a distribution function (Best and Roberts 1975).

The Mann-Kendall test is another non-parametric significance test for a monotonic trend in a time series based on the Kendall's τ (Mann 1945; Kendall 1975). This test compares the ranks for all pairs of AM discharges. This amounts to $N*(N-1)/2$ combinations. The test statistic τ is the difference between the number of pairs that supports a positive trend and the number of pairs that supports a negative trend, divided by the standard deviation. The null hypothesis is that the data are independent and thus randomly ordered. For a substantial number of observations the test statistic τ will then be normally distributed.

8.3.4 Results

Presentation of data series

In this section, the data series used in our research are presented and described shortly. In the following sections, the results of further analyses based on these data are described.

In Figure 8.8 the daily measurements of precipitation (P) and potential evapotranspiration (ET) are shown over the period 1950 – 2012. Average P over the measurement period was 2.24 mm/d, while the maximum daily P amounted to 88.2 mm/d (19-06-1966). Average ET over the measurement period was 1.51 mm/d, while the maximum daily ET was 5.8 mm/d (04-07-2001). Also, the annual sums of these meteorological measurements are shown for this period in Figure 8.8. On average, annual total P amounted to 819 mm. In 1959, P was lowest (477 mm/year), while P was highest in 1966 (1240 mm/year). On average, annual total ET amounted to 550 mm. In 1998, ET was lowest (492 mm/year), while ET was highest in 2003 (635 mm/year). From the annual P and ET values, recharge was calculated:

$$\text{Recharge} = P - f_{crop} \times ET \qquad \text{(eq.2)}$$

With f_{crop} estimated on 0.789, based on crop annual factors for various types of land use from literature (Van Walsum et al. 2004; Dik 2004) and an estimation of the land use in the Regge catchment (50% agriculture; 25% forested area; 25% urban area). This resulted in an annual recharge between a minimum of -13 mm/year (net loss of water) in 1959 and a maximum of 843 mm/year in 1966. The average annual recharge amounted to 386 mm/year (Figure 8.8).

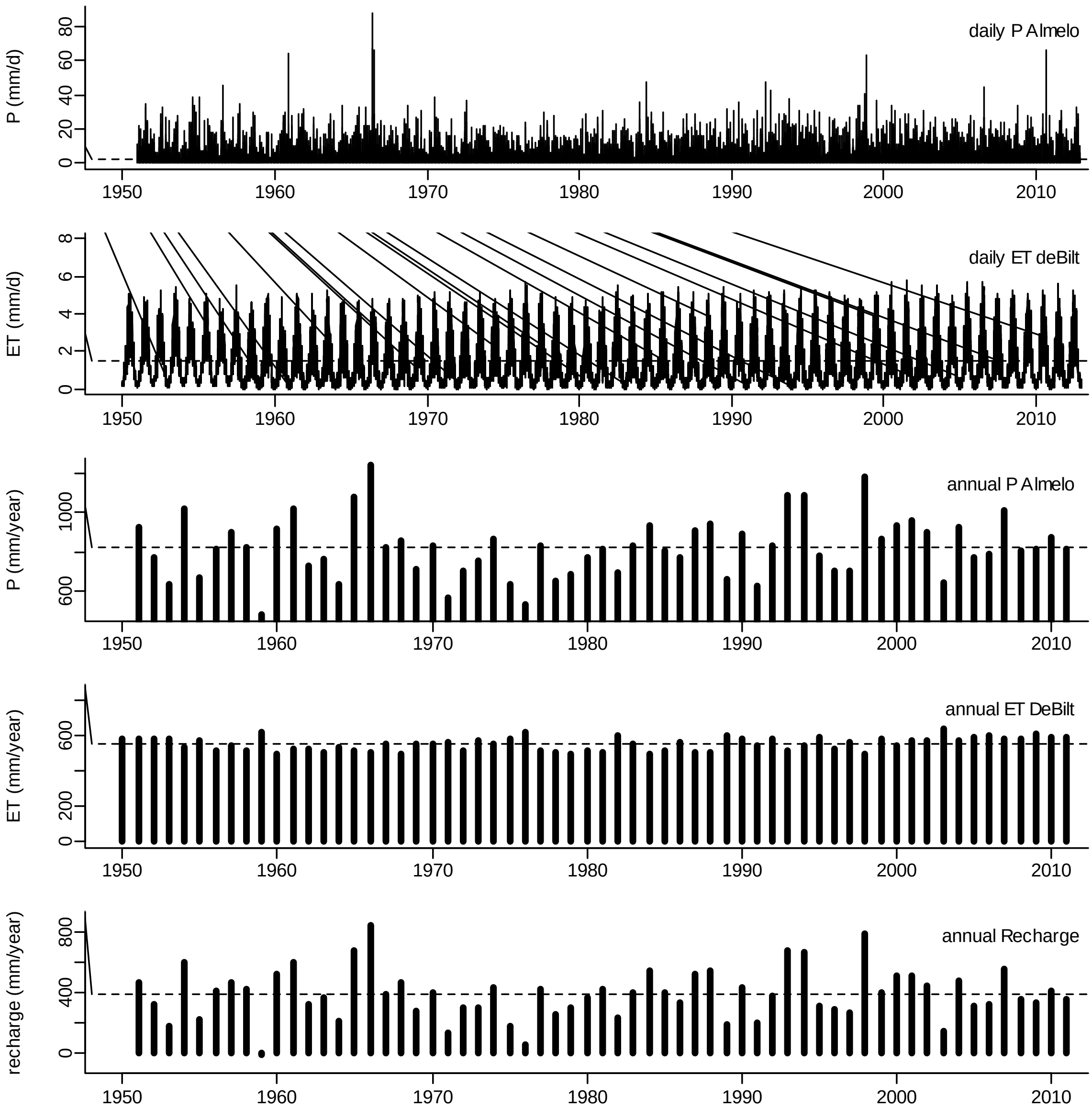

Figure 8.8. Data series of precipitation (P) and evapotranspiration (ET) used in this study. The upper two graphs show the daily recordings, while the lower three graphs show the yearly sum of P, ET, and calculated recharge. The dotted lines indicate the average daily or annual P, ET, and recharge.

Thirteen *groundwater level locations* with measurements starting in the 1960s or earlier and a measurement period of more than 35 years were selected for our analysis. An overview of the locations and their characteristics is given in Table 8.5 and in Figure 8.9. In Figure 8.9 the three longest groundwater level data series are shown as groundwater level below the surface. In general, all the groundwater levels are shallow (0.5 to 3 m below the surface) and showed a decreasing trend over the 35 to 55 years that they were recorded. At some locations this trend is stronger (e.g. location B28B0032) than at others (e.g. location B28B0159). On shorter time scales, all groundwater levels showed significant fluctuations from day to day or month to month. These were probably mainly responses to precipitation and evapotranspiration.

Table 8.5. Meta data of the groundwater level measurement locations used in this study.

grw series TNO ID	start date	end date	surfacelevel (m + NAP)	top of filter (m + NAP)	filter depth (m below surf)
B28B0007	13-06-49	14-10-05	8.95	-0.24	9.19
B28D0066	28-03-50	14-09-00	15.69	1.24	14.45
B28B0159	14-07-51	14-01-01	7.03	-7.79	14.82
B28B0045	28-11-62	25-11-01	10.32	-7.16	17.48
B28D0078	09-10-61	14-04-10	9.35	-8.73	18.08
B28D0079	09-10-61	14-04-10	9.05	-9.50	18.55
B28B0032	07-06-49	14-10-05	10.19	-10.80	20.99
B28B0046	20-10-61	25-11-01	10.05	-13.45	23.50
B34B0160	05-01-67	14-08-05	13.22	-11.78	25.00
B28H0230	27-12-65	14-10-05	20.37	-5.65	26.02
B34B0031	30-09-59	14-08-05	13.78	-15.22	29.00
B28H0229	27-12-65	28-11-05	20.51	-14.49	35.00
B28H0228	17-05-65	28-11-05	23.13	-25.28	48.41

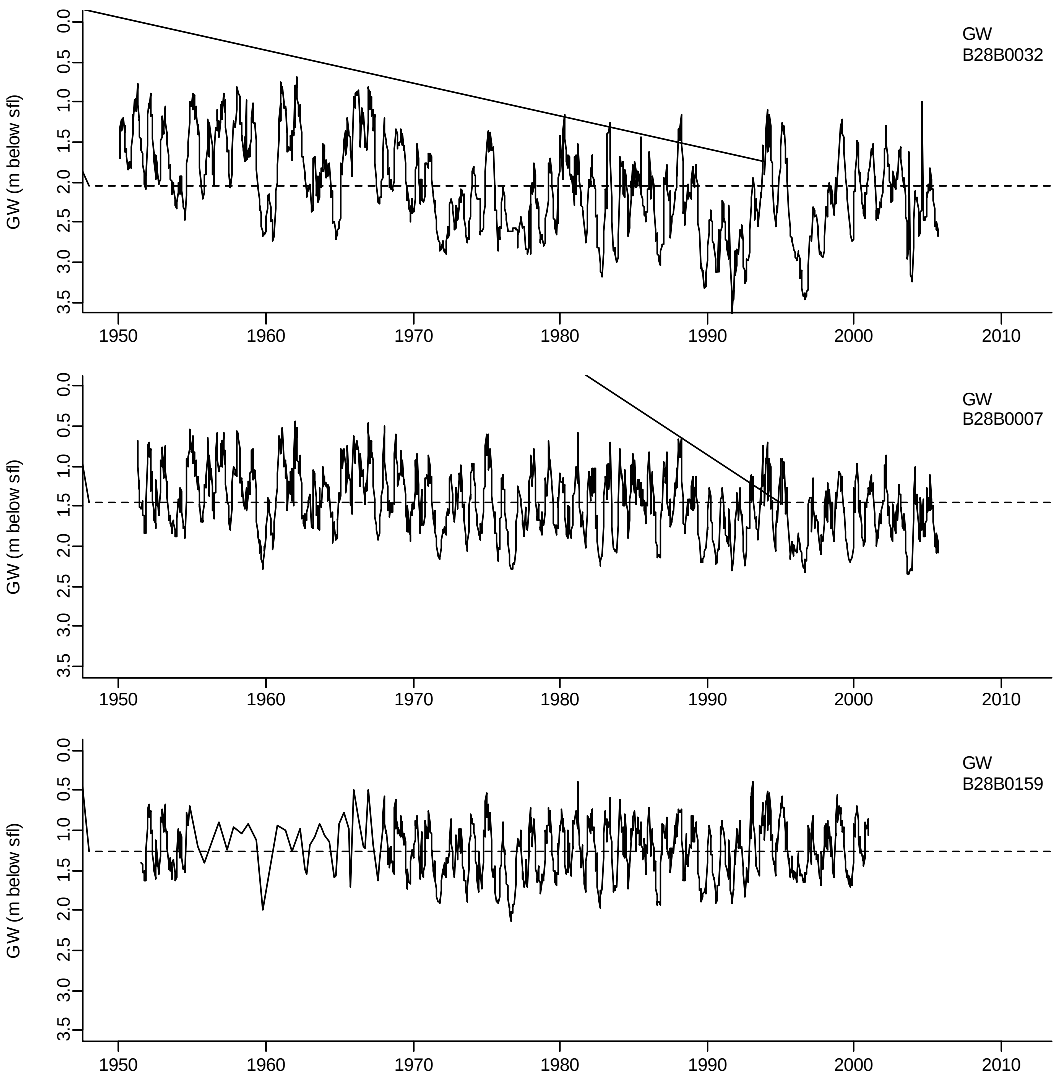

Figure 8.9. Groundwater level measurements at three locations in the Regge catchment with the longest record. The dotted lines indicate the average groundwater level below the surface over the full measurement period at that location.

In Figure 8.10 the discharge measurements of the upstream Regge (RU) and total Regge (RL) are shown. The upper two graphs show the daily recordings in m^3/s of the weirs for RU and RL, while the lower two graphs show the yearly sum in mm per year for both weirs. Average daily Q over the measurement period was 6.94 m3/s for RU and 9.20 m3/s for RL. At the weir of RU, the maximum daily Q amounted to 84.18 m3/s on 29-10-1998 and the minimum daily Q was 0 m3/s on several days in summer periods of 1991, 1992, 1993 and 1997. At the weir of RL, the maximum daily Q amounted to 113.18 m3/s at 29-10-1998 and the minimum daily Q was 0 m3/s at 02-08-1992. Annual averages were calculated for all years with more than 90% data coverage. The average annual sum Q over the measurement period was 229 mm/year for RU and 137 mm/year for RL. At the RU weir, the maximum annual sum of Q amounted to 381 mm/year (1998), and the minimum annual sum of Q amounted to 126 mm/year (1976). At the RL weir, the maximum annual sum of Q amounted to 216 mm/year (1966), and the minimum annual sum of Q amounted to 60 mm/year (1971).

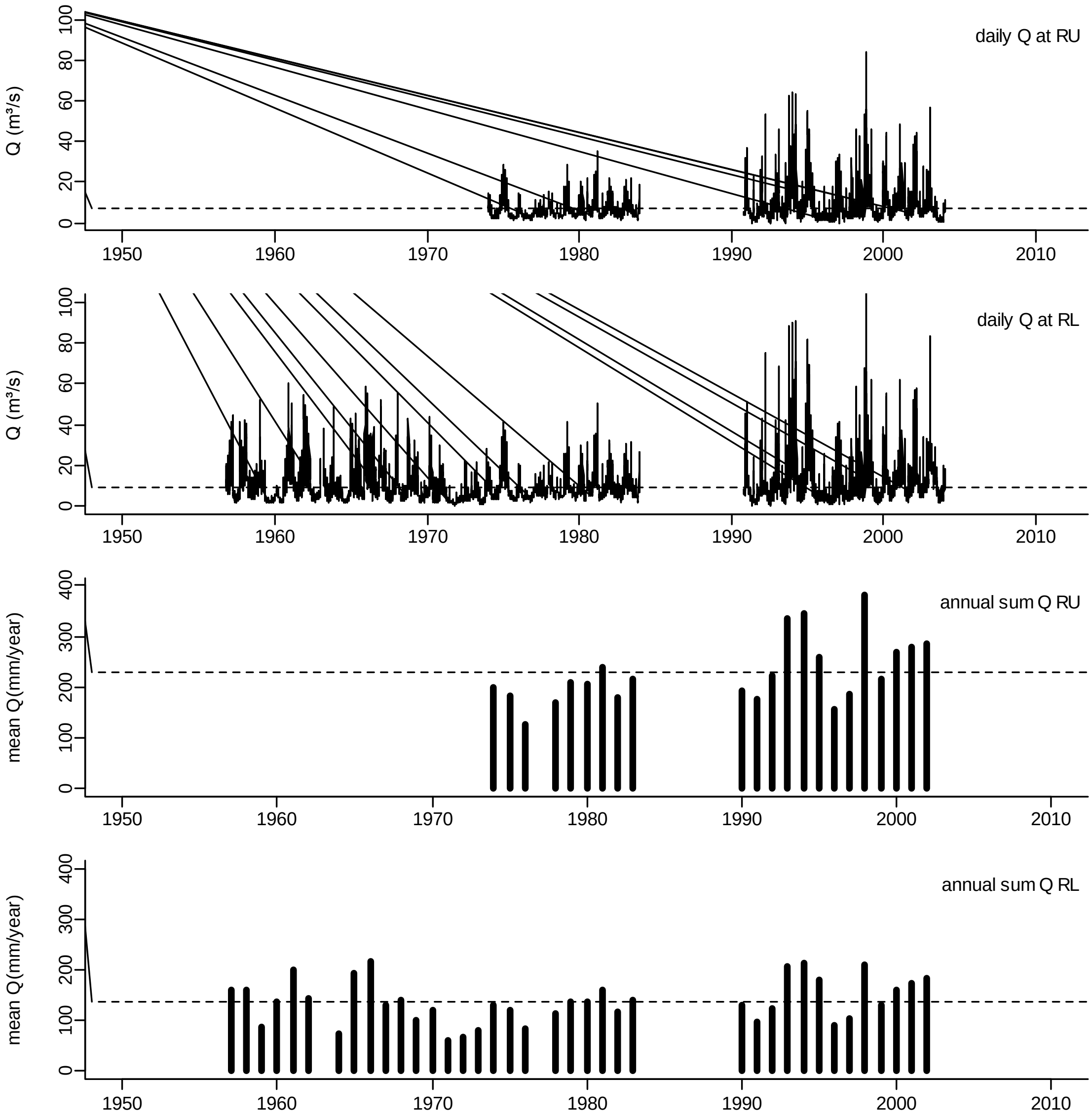

Figure 8.10. Data series of discharge (Q) at the weirs at Archem of the upstream Regge catchment (RU) and the Regge & Linderbeek catchment (RU). The upper two graphs show the daily recordings in m^3/s, while the lower two graphs show the annual sum in mm per year. The dotted lines indicate the average daily or annual Q.

Standard indexes

Figure 8.11 shows the standard indexes of P (Almelo), ET (De Bilt) and Q RL (outflow of whole Regge catchment) calculated on a decadal basis, named, respectively, SPI, SEI and SQI. For all standard indexes, the value "0" is the average for the whole period. For SPI and SQI, negative values indicate a dry period. For SEI, negative values indicate a reduced evapotranspiration and thus less dry conditions. It can be observed that for SQI clear drought periods occur, for instance around 1960 and over the period 1971-1975. For SPI and SEI such multi-year periods are less obvious.

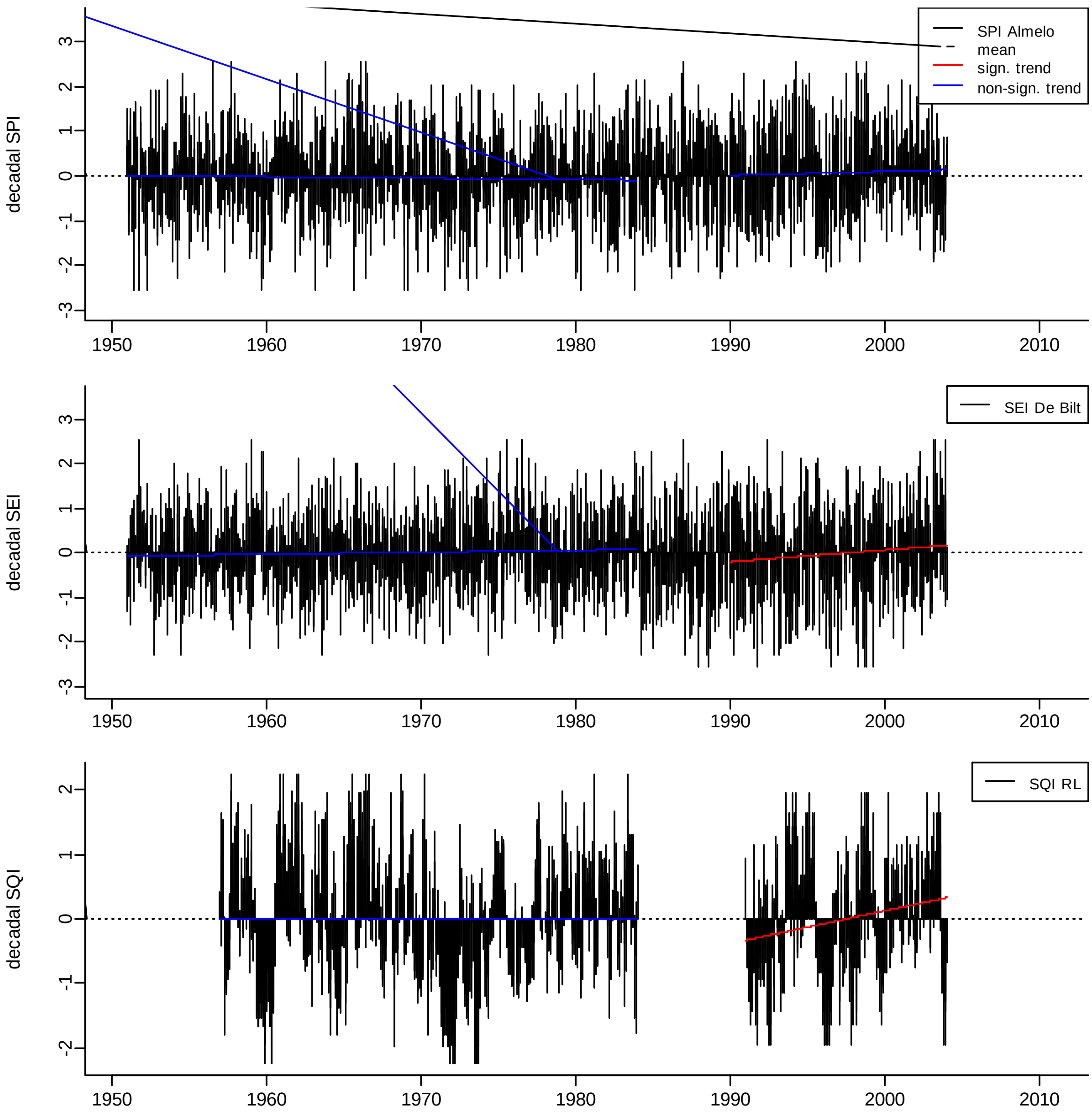

Figure 8.11. SPI (upper graph), SEI (middle graph) and SQI (lower graph) on a decadal basis. Dotted lines indicate average values; red lines indicate significant trend lines; blue lines indicate non-significant trend lines.

As described above, statistical trend tests to detect non-stationarity were applied to the time series of flow parameters. In order to compare any detected trends in SQI with meteorological trends, the time series of SPI and SEI were split up in two periods (1951-1983 and 1990-2003) before statistical trend testing as well. In the graphs in Figure 8.11, the linear trend lines of the two measurement periods are shown. Trend lines that were significantly non-stationary ($p < 0.05$) were considered to have a significant increasing or decreasing trend. These trend lines have a red colour, while non-significant trend lines have a blue colour.

It can be observed that none of the test periods of SPI showed a significant trend. For both SEI and SQI, the first test period (1951-1983) showed no significant trend, while the second test periods did show significant trends. The trend of SEI indicated an increase in evapotranspiration over the second period, while the trend in SQI indicated a decrease in discharge over the second period. As neither SPI nor SEI indicated a wetting of the climate, it may be suggested that some other hydrogeological mechanism occurred that induced an increase in discharge over this period (1990-2003).

Flow parameters

The calculation of hydrological parameters provides better insight into average flow conditions as well as into changes in high and low flow discharge values. The graphs in Figure 8.12 and Figure 8.13Figure 8.13 display the annual values of the flow parameters *mean Q, Q95*, *Q75*, *Q25*, and *Q5* for the upstream Regge catchment (weir at RU; Figure 8.12) and the whole Regge catchment (weir at RL; Figure 8.13). The variability of the discharge was characterised by calculations of the fractions Q95/Q5, Q95/meanQ, Q75/Q25. The lowest three graphs of both figures show these characteristics for flow variability over time. In all the graphs the mean value of the flow parameter or variability characteristic is given.

As described above, statistical trend tests to detect non-stationarity were applied to the time series of flow parameters. For this purpose the time series of the two measurement periods were considered separately. In the graphs in Figure 8.12 and Figure 8.13, the linear trend lines of the two measurement periods are shown. Trend lines that were significantly non-stationary ($p < 0.05$) were considered to have a significant increasing or decreasing trend. These trend lines have a red colour, while non-significant trend lines are blue.

For the second measurement period (1990 -2003) of the upstream Regge catchment (RU), Q95 showed a significant increasing trend as well as Q95/Q5 and Q75/Q25. This indicated an increase of very low flow and decrease of flow variability over this period. Moderate and high flows remained relatively constant over the period.

For the first measurement period (1974 - 1983) of the upstream Regge catchment (RU), only Q75 showed a significant increasing trend, indicating an increase of low flow over this period. Probably, the measurement period was too short for statistical trend testing.

For the first measurement period (1956 - 1983) of the whole Regge catchment (RL), Q25 and Q5 showed a significant decreasing trend, indicating decrease of high flow and peak flow over this period. Low and moderate flow parameters remained relatively constant. All variability characteristics (Q95/Q5, Q95/meanQ (base flow index), and Q75/Q25) showed a significant increasing trend, indicating decrease of flow variability over the period.

For the second measurement period (1990 -2003) of the whole Regge catchment (RL), Q95 and Q75 showed a significant increasing trend as well as all variability characteristics (Q95/Q5, Q95/meanQ, and Q75/Q25). High flow parameter Q25 showed a significant decreasing trend over this period. This indicated an increase of (very) low flow and a decrease of high flow and flow variability over this period.

Overall, flow variability decreased over the analysis period as high flows decreased over the period 1956 - 1983 and low flow increased over the period 1990 - 2003.

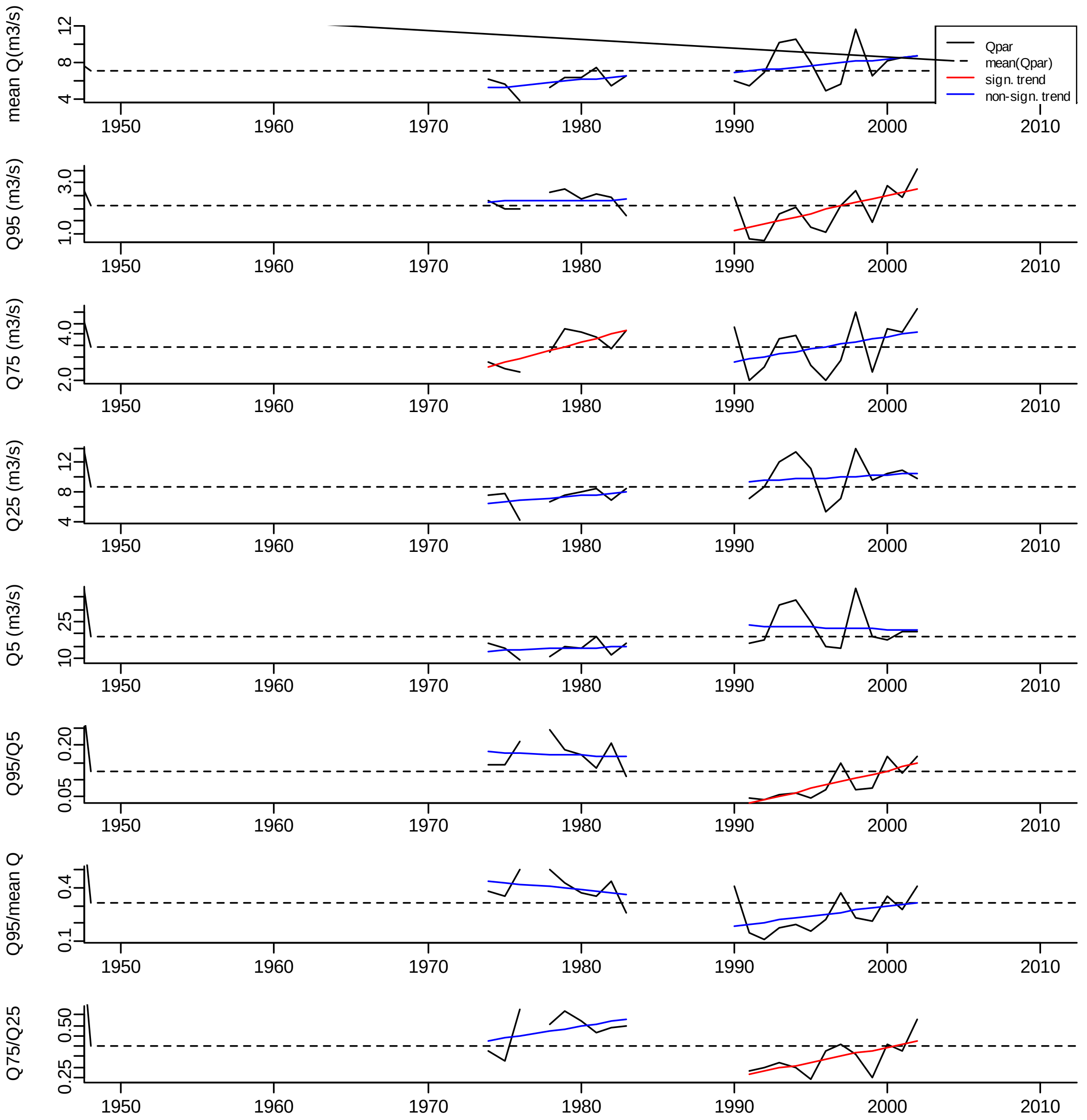

Figure 8.12. Annual values of the flow parameters and variability characteristics for the upstream Regge catchment (RU). Dotted lines indicate average values; red lines indicate significant trend lines; blue lines indicate non-significant trend lines.

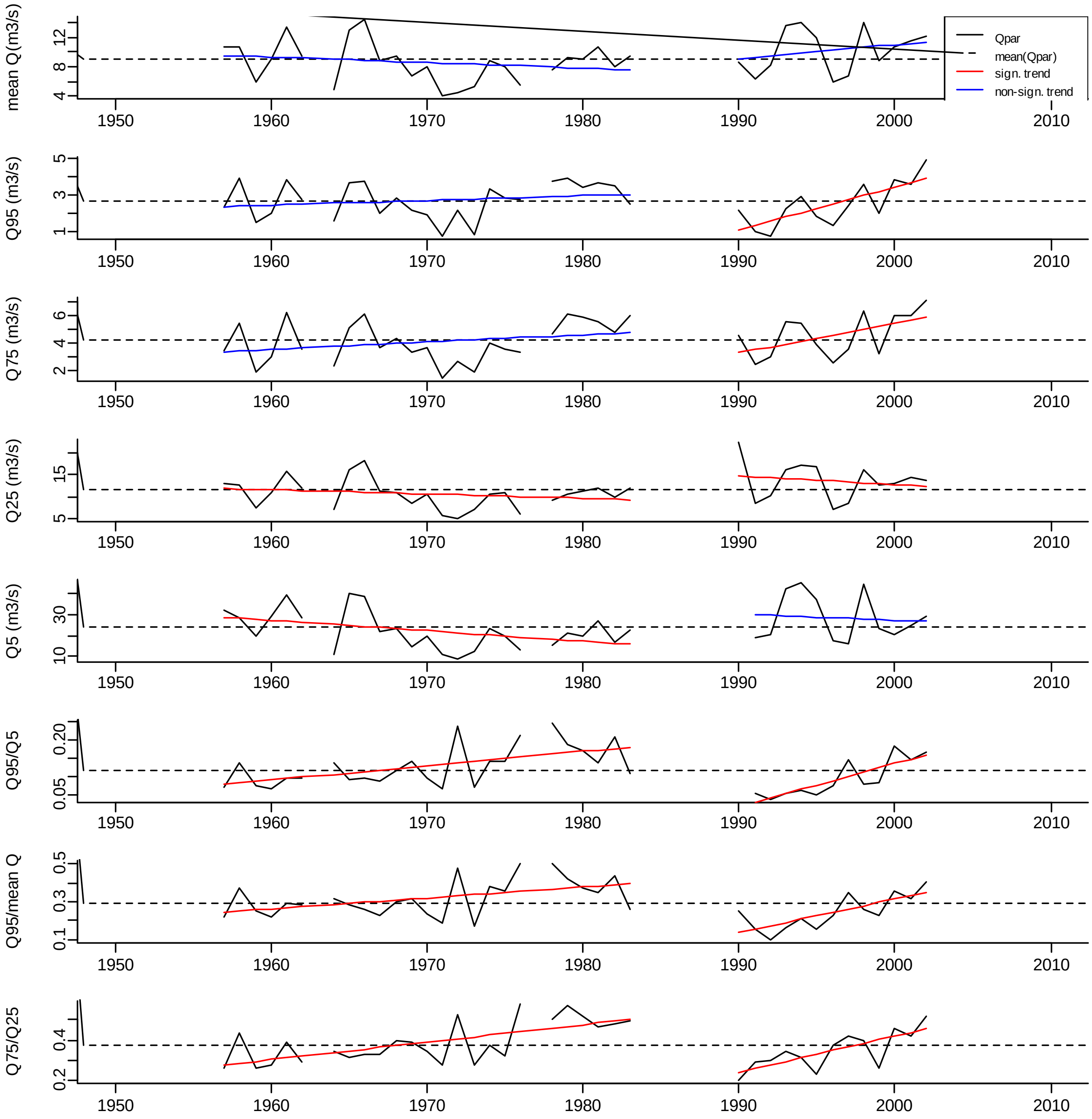

Figure 8.13. Annual values of the flow parameters and variability characteristics for the whole Regge catchment (RL). Dotted lines indicate average values; red lines indicate significant trend lines; blue lines indicate non-significant trend lines.

Time series analyses with impulse-response modelling

The time series analyses of the groundwater level data series showed that all data series for 60-80% were explained by meteorological variability. This indicated that meteorological impulses played an important role in determining the groundwater level. However, also other processes are important. Using the time series analyses with impulse-response modeling technique, the effects of meteorological variability (P and ET) were fit in an impulse-response model. Next, the residual time series were calculated by subtracting the meteorological impulse-response model from the groundwater level data series.

Figure 8.14 shows the result of the time series analyses with impulse-response modeling for the three longest groundwater level data series. The graphs show the time series of the residual of the groundwater level date series. This residual is not affected by meteorological impulses (P and ET), and any trends in the data are due to other impulses or developments in the area or catchment that affect the groundwater. The graphs in Figure 8.14 show that the groundwater level decreased between 1950 and 1990 and increased between 1990 and 2005, independently from meteorological impulses. In the graphs the linear trend lines of these two periods are shown. All trends were significantly non-stationary ($p < 0.05$); hence the residual time series were considered to have a significant increasing or decreasing trend.

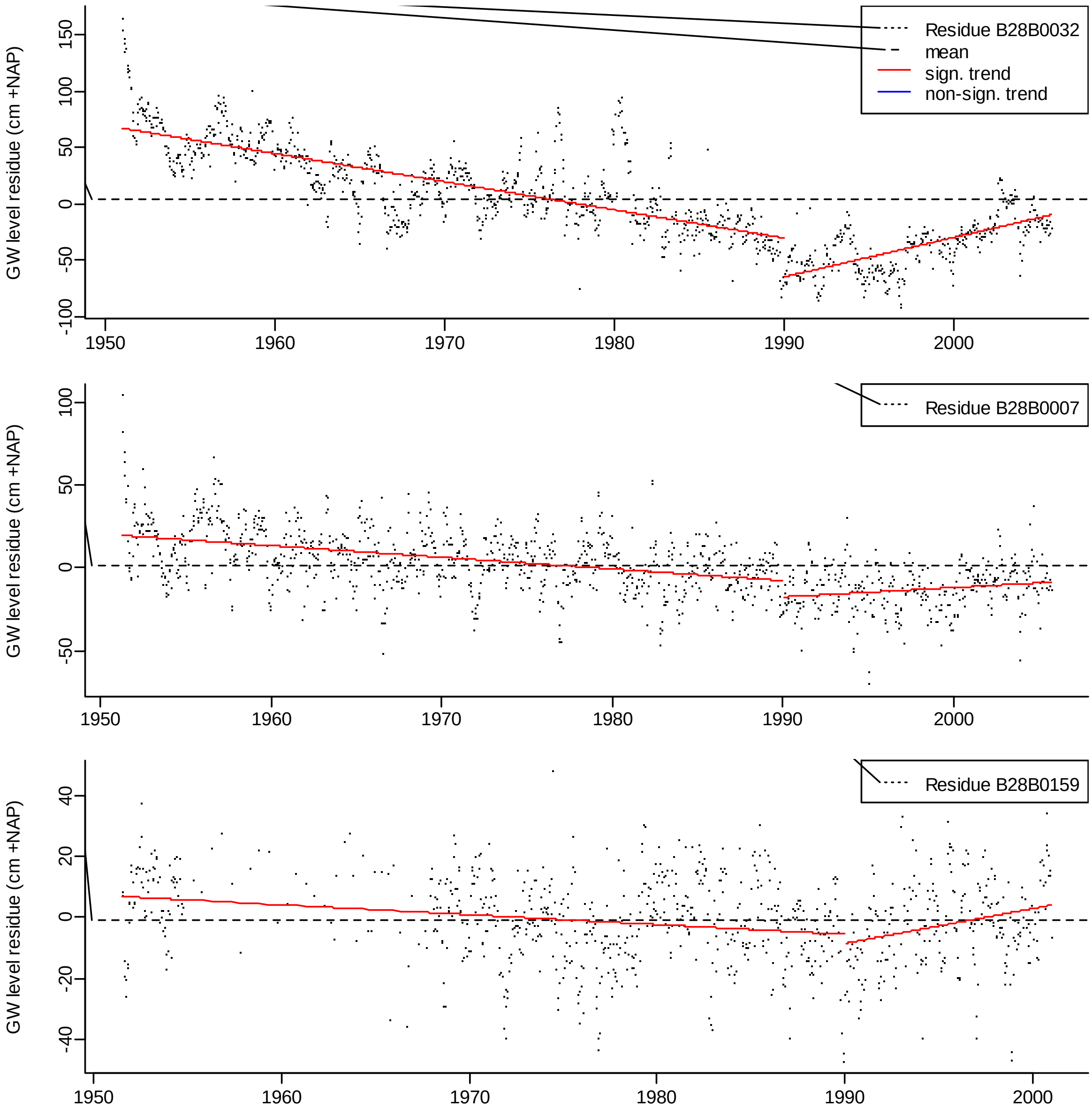

Figure 8.14. Groundwater level residuals for the three groundwater level locations with longest data series (upper three graphs). Dotted lines indicate average values; red lines indicate significant trend lines for two periods: 1950-1990 and 1990-2005.

The time series of QRU showed an explanation by meteorological variability of 64% for the first period and 68% (1956-1983) for the second period (1990-2003). The time series of QRL showed an explanation by meteorological variability of 63% for the first period and 66% (1956-1983) for the second period (1990-2003). This indicated that meteorological impulses played an important role in determining the discharge of the Regge catchment. However, also other processes are important. Using the time series analyses with impulse-response modeling technique, the effects of meteorological variability (P and ET) were fit in an impulse-response model. Next, the residual time series were calculated by subtracting the meteorological impulse-response model from the QRL data series.

Figure 8.15 shows the result of the time series analyses with impulse-response modeling for the two discharge series. These residuals were not affected by meteorological impulses (P and ET), and any trends in the data are due to other impulses or developments in the area or catchment that affect the discharge. The graphs in Figure 8.15 show that QRL slightly decreased between 1956 and 1990 and increased between 1990 and 2005, independently from meteorological impulses. In the graphs the linear trend lines of these two periods are shown. The trends were of the residuals of QRL are significantly non-stationary ($p < 0.05$); hence the residual time series were considered to have a significant increasing trend. For the discharge of the upper Regge sub-catchment no significant increase or decrease was observed.

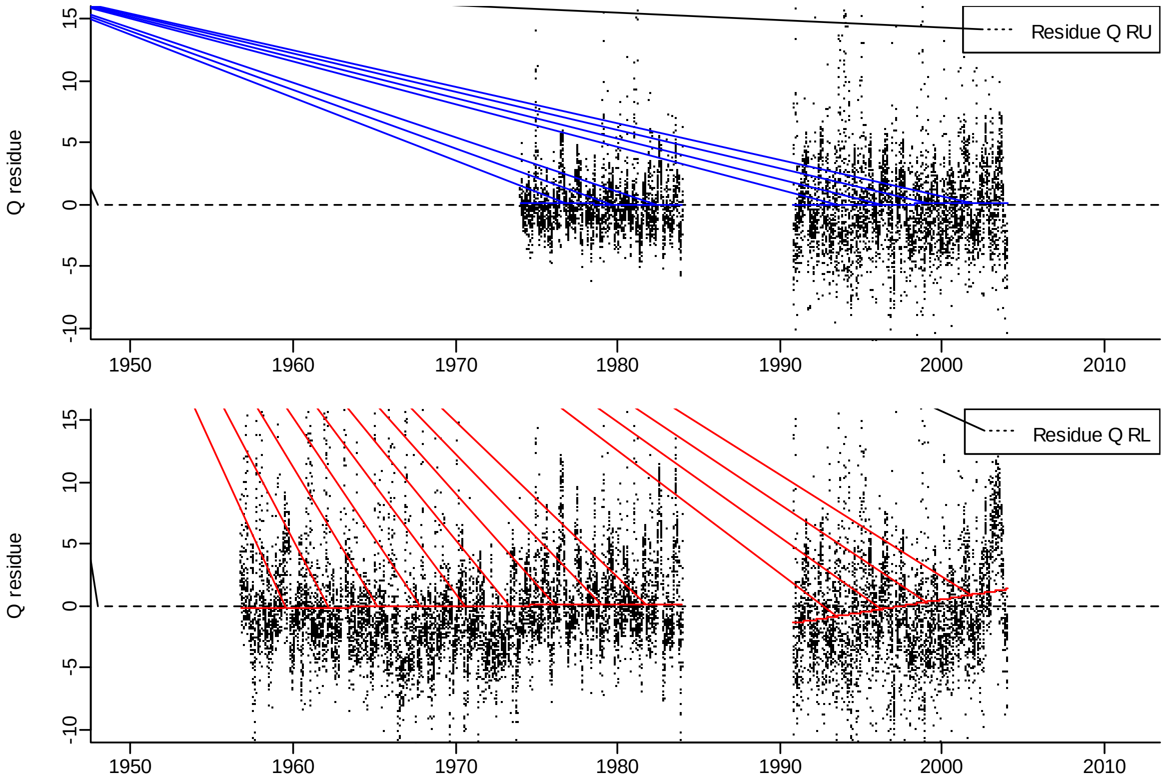

Figure 8.15. Residuals of QRU (discharge upstream Regge catchment) and QRL (discharge whole Regge catchment). Dotted lines indicate average values; red lines indicate significant trend lines for two periods: 1956-1990 and 1990-2005.

8.3.5 Historical overview anthropogenic alterations

Based on Donker (1996), information from the Central Statistical Bureau of the Netherlands (CBS) and an interview at Water Board Regge and Dinkel (B. Ordelmans) a historical overview of the major anthropogenic alterations was complied. In Figure 8.16, the overview is shown for both alterations on the streams and alterations on land use.

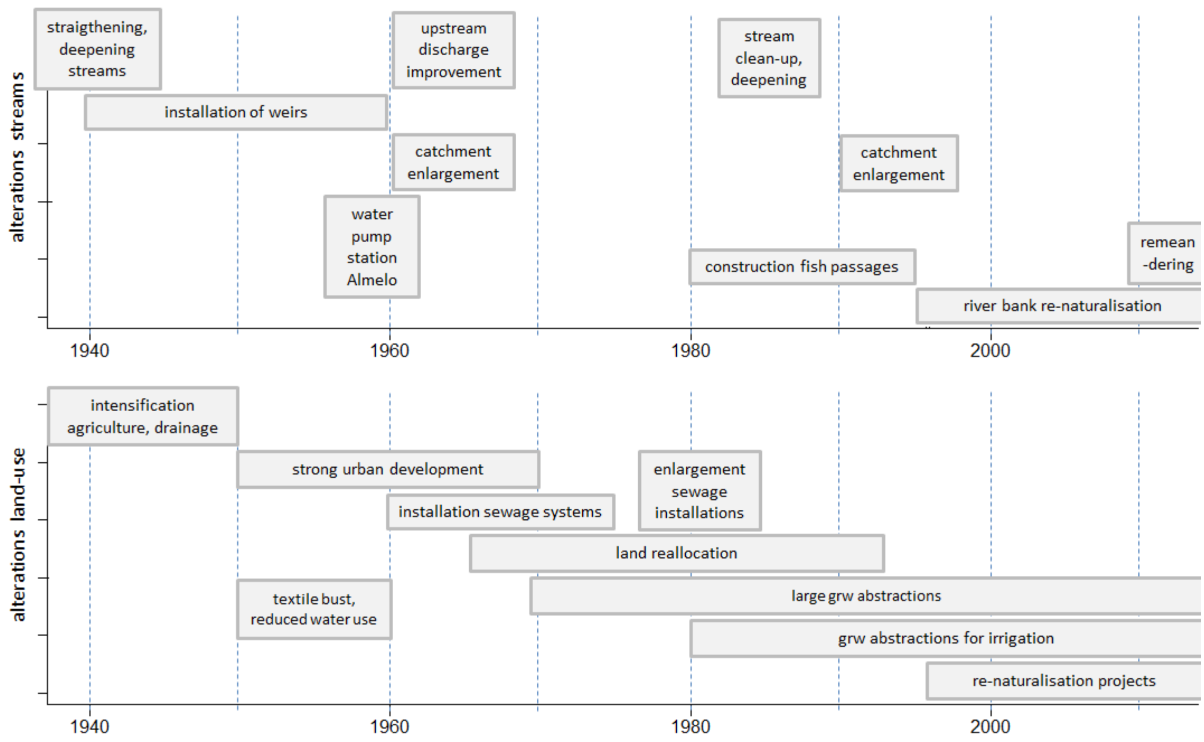

Figure 8.16. Overview of anthropogenic alterations on streams (upper graph) and land use (lower graph) in the Regge catchment for the period 1940-2013.

Anthropogenic alterations of streams started around 1900 with straightening and deepening of streams to prevent floods in agricultural areas. Simultaneously, drainage of agricultural areas was undertaken. These activities were followed by installation of weirs in many streams to prevent the streams from drying up. Activities to improve discharge in upstream areas of the catchment were made in the 1960s. Additionally, between 1983 and 1990 the main streams in the catchment were cleaned, causing a significant deepening of the streams. Both in the 1960s and in the 1990s, the Regge catchment was enlarged with some small upstream sub-catchments. Starting in the 1980s, re-naturalisation activities were carried out. First, fish passages were constructed (190-1995); next, river bank re-naturalisation was done (1995-now);and, finally, re-meandering projects were started around 2010 at the main stream of the Regge.

Extensive anthropogenic alterations of land use started around 1850 when forested areas and marsh lands were turned into agricultural areas. Since the 1930s important intensification of agriculture have taken place accompanied by straightening and deepening of streams and construction of drainage networks. During the period 1950-1970 the area was prone to strong urban development. On the other hand, in the 1970s, textile industries busted, causing a reduction of water use. Following the urban expansion, sewage water systems and installations were constructed over the period 1960-1975. Around 1980, many of the installations were enlarged. Between 1970 and 2000, large land reallocations took place causing up-scaling and further intensification of agricultural activities. Probably, drainage of the agricultural areas was intensified and optimised in this period as well. Around 1975, groundwater abstraction for drinking water and industrial purposes increased, and discharges increased significantly to form an important water source for the area. As in and around the streams, re-naturalisation of upstream areas and areas bordering the main stream started around 1995.

8.3.6 Conclusions and discussion

Our research showed that groundwater levels in the Regge catchment decreased over the period 1956 – 1990 and increased over the period 1990 - 2003. Discharge of the Regge River showed no trend over the first period (1956 – 1983), while an increase in discharge was observed over the second period (1990 – 2003). Concerning the environmental flow parameters of the Regge River, the first measurement period (1956 – 1983) showed a decrease of high flows (Q5 and Q25), while low flows (Q95 and Q75) did not show a significant trend. During the second period, low flows (Q95 and Q75) increased and high flow (Q75) decreased. Flow dynamics decreased for both periods and the Base Flow Index (base flow/mean discharge) increased, indicating that the yearly discharge of the stream consisted increasingly of base flow.

It was found that precipitation affected groundwater and flow dynamics in short term periods (>1-5 year); this is in accordance with previous research on impulse response modelling of the groundwater levels in the area (Hendriks et al., 2010). Additionally, long term changes in groundwater level, discharge and flow parameters were not traced in meteorological data series. Precipitation showed no significant trend for both analyses periods, and evapotranspiration showed no trend between 1956 and 1983, and a positive trend between 1990 and 2003. Hence, the significant long term trends in groundwater levels, discharge, and flow dynamics are probably not caused by meteorological changes.

The historical assessment of anthropogenic changes in the stream morphology and in the catchment showed that over the research period (1956 - 2003) many alterations occurred that might have affected groundwater, discharge and flow dynamics. Unfortunately, the largest changes to the catchment occurred before the measurements of discharge or groundwater levels started. These changes consist of extensive intensification of agriculture, construction of artificial drainage, straightening and deepening of streams. Probably, these have signifantly altered the hydrology of the catchment, not only by changes the groundwater levels and stream density and morphology, but also due to removal of streams beds with reduced transmissivity (e.g. clay, peat layers).

The decrease of groundwater levels during the measurement period was probably partly due to the large scale changes in the catchment occurring during the first half of the 20th century. In addition, a large part of the groundwater level decrease was most probably caused by the combined effect of groundwater abstractions for drinking water, industry and irrigation purposes (1970-2003) and land reallocation (1965-1985). However, we also expected a decrease of low flow (Q95 and Q25) due to the decreased groundwater level as well as an increase of high flow (Q5 and Q25) as a result of urbanisation and increased drainage (more direct runoff). Possibly, a decrease of low flows was reduced by other anthropogenic changes, such as catchment enlargement, the textile bust, and installation of sewage systems that generate a relatively constant discharge. The decrease of high flow that was observed on our measurements could be an effect of sewage water installations and catchment enlargement.

Opposed to the changes during the first analyses period, the significant changes in groundwater level (increase) and low flows (increase) could be linked more directly to re-naturalisation projects that have taken place in the catchment since the 1990's. Although no direct causal relation could be proved, no other large scale changes in the catchment were reported that could cause such a reversal in the geohydrological condition of the catchment.

From our study, several conclusions can be drawn. However, many uncertainties remain which require more detailed research activities. Fully unraveling the specific cause-effect relations was not possible in the study as many anthropogenic changes showed overlap and were implemented gradually rather than instantaneously. Further research, including more local sub-catchment analyses as well as inclusion of water quality data, maycontribute to further pinpointing the specific effects of anthropogenic changes on discharge, base flow and flow dynamics. Unfortunately, a general problem is the length of available data series: most sub-catchments have data series that do not go back further than the 1980s. Another (but time consuming) option is to carry out dedicated field experiments during anthropogenic alterations of land use and stream morphology.

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8.5 Acknowledgements

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9 Macroinvertebrate response to stream corridor and catchment characteristics

9.1. Macroinvertebrate response to land use in stream corridors and upstream catchment

9.1.1 Introduction

According to hierarchical organization of fluvial ecosytems the local stream hydromorphology depends on upstream structures and processes operating at catchment and in stream corridor scales. Considering interactions between anthropogenic impacts and natural processes operating at scale of entire water basin is important for investigation of ecological consequences of degraded hydromorphology. Interactions between fluvial ecosystems and adjacent land reflect in channel dynamics, sediment characteristics, including their contamination by pollutants.

Agriculture activities and point releases of polluted water are main stressors operating in multiple-stressed stream ecosystems of central Europe. Agriculture may affect streams by eutrophication, supply of degradable organic matter and fine particles, contamination by pesticides, melioration and discharge alteration. Chemical substances and sediments are transformed and transported according to physical characteristics of channel, floodplain and catchment. The majority of stream network in the Czech Republic has got altered hydromorphological conditions. Arable land being directly adjacent to stream channel increase probability of contribution of eroded soil into stream sediemnts. Natural or seminatural vegetation in stream corridor may reduce detrimental effects of agriculture land use in the catchment. Vegetation act as buffer zone (nutrients, pesticides, erosion) and it also influence channel morphology by bank stabilization and enriching substrate (habitat) heterogeneity.

For planning measures to be implemented for improving degraded conditions are needed indicators of general degradation and indicators specific to individual aspects of complex impairment as well. In the central Europe the traditional representative of the first type is Saprobic index. As a specific index responding predominantly to contamination was recently developed SPEARpesticides index.

The aim of this study was to compare response of Saprobic index, ASPT, SPEARpesticides and other selected characteristics of macroinvertebrate communities to potential risk for stream ecosystems associated with various land use/cover in stream corridors and catchments. We tested relationships between stress effects and natural factors (altitude, stream size, natural vegetation in corridor).

9.1.2 Data

Data set consisted of 272 macroinvertebrate samples being taken from 272 sites of national monitoring network in spring season of 2007 (see Figure 9.1). Macroinvertebrate data were taxonomicaly adjusted in a way fitting to trait-based metrices and biotic indices. Such adjustment didn’t allow using taxa richness and species diversity metrices. Macroinvertebrate metrices were calculated using ASTERICS software (version 2.6) and SPEARcalculator (http://www.systemecology.eu/spear/spear-calculator/). Basic environmental description of sites (altitude, stream order, coordinates) were supplemented by GIS analyses in upstream catchments and in stream network corridors.

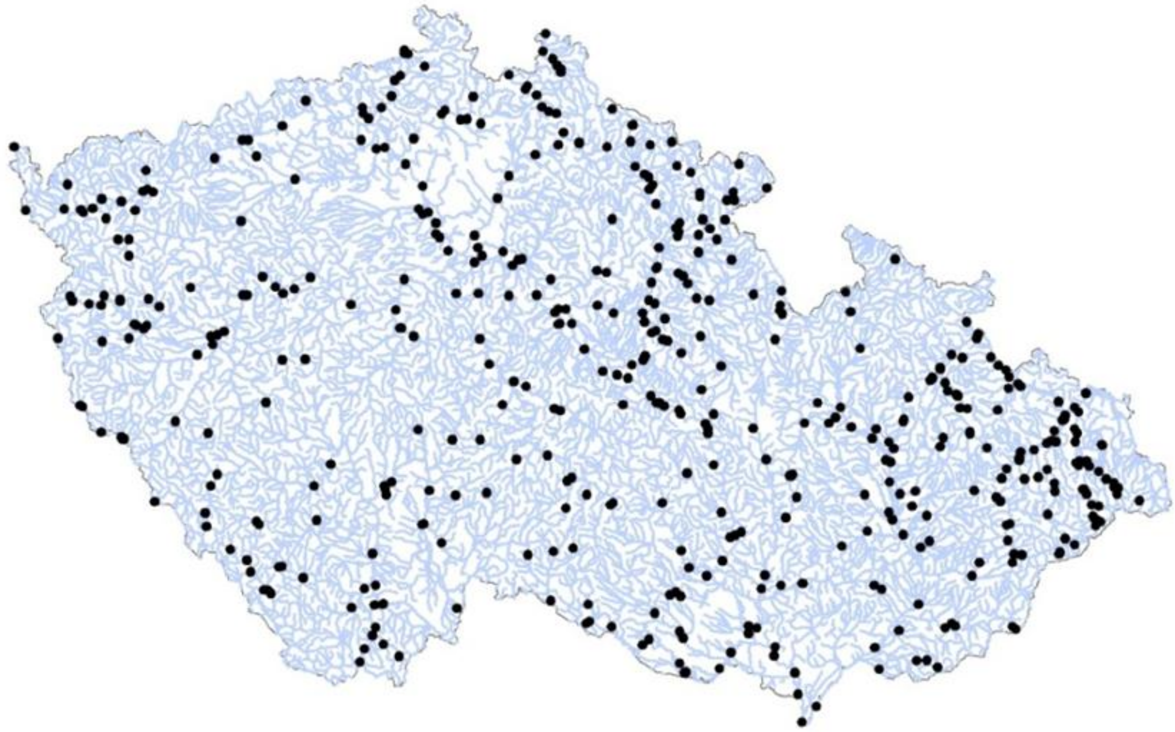


Figure 9.1. Location of sampling sites.

9.1.3 Methods

Stream corridors (Figure 9.2) were defined as 200 m wide stream network buffers (2x100 m). They were delineated in various extent upstream the studied sites (100 m, 200 m, 500 m, 1 km, 2 km, 5 km, 10 km and entire stream network upstream the site).

The CORINE land cover (CORINE 2000) data were analyzed within sites catchments (catch_ CORINE category) and corridors arround stream network upstream sites. With respect to CORINE resolution and considering average channel/floodplain width we used 200m-wide corridor. Three longitudinal extents of corridors were applied: (i) a 2 km (bf200_2_ CORINE category) and (ii) a 5 km extent to the stream network upstream the studied site (bf200_5_ CORINE category) and (iii) a buffer around entire stream network upstream the site (bf200_all_CORINE category).

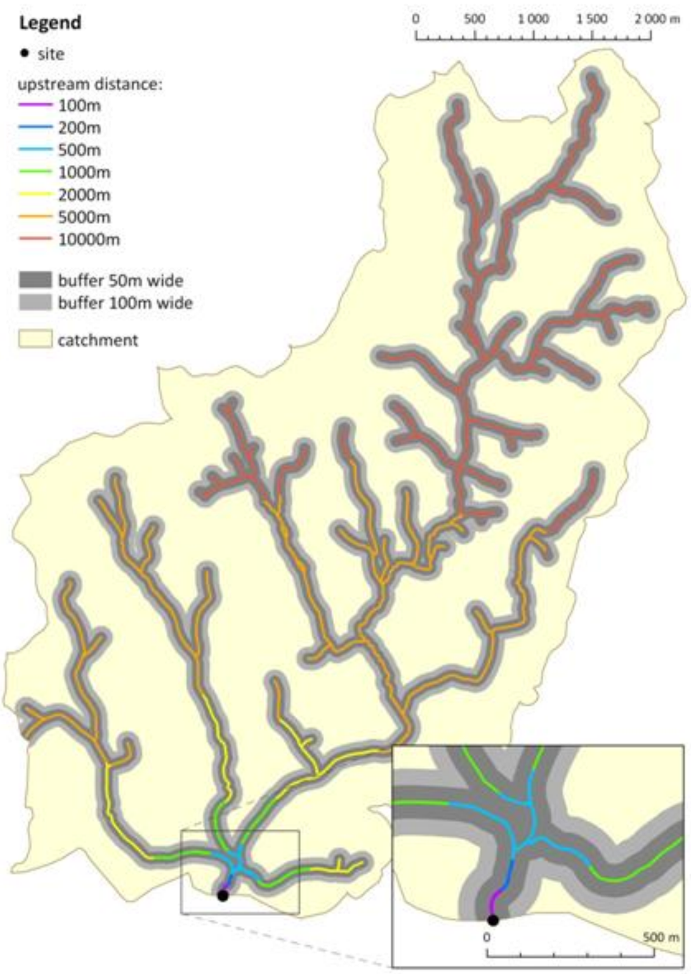


Figure 9.2. Scheme of delineation of stream network buffers (we used for this study the 2x100 m wide buffer of upstream extent 2 km, 5 km and entire network).

Certain CORINE categories were aggregated to express total proportion of land having similar effect on fluvial ecosystems. Agriculture areas (AGR) included the non-irrigated arable land (CORINE code 211), complex cultivation patterns (code 242) and land principally occupied by agriculture, with significant areas of natural vegetation (code 243). Natural and seminatural vegetation in corridors was created by merging of broad-leaved, coniferous and mixed forests (codes 311, 312 and 313 respectively), natural grassland (code 321) and transitional woodland shrub (code 324).

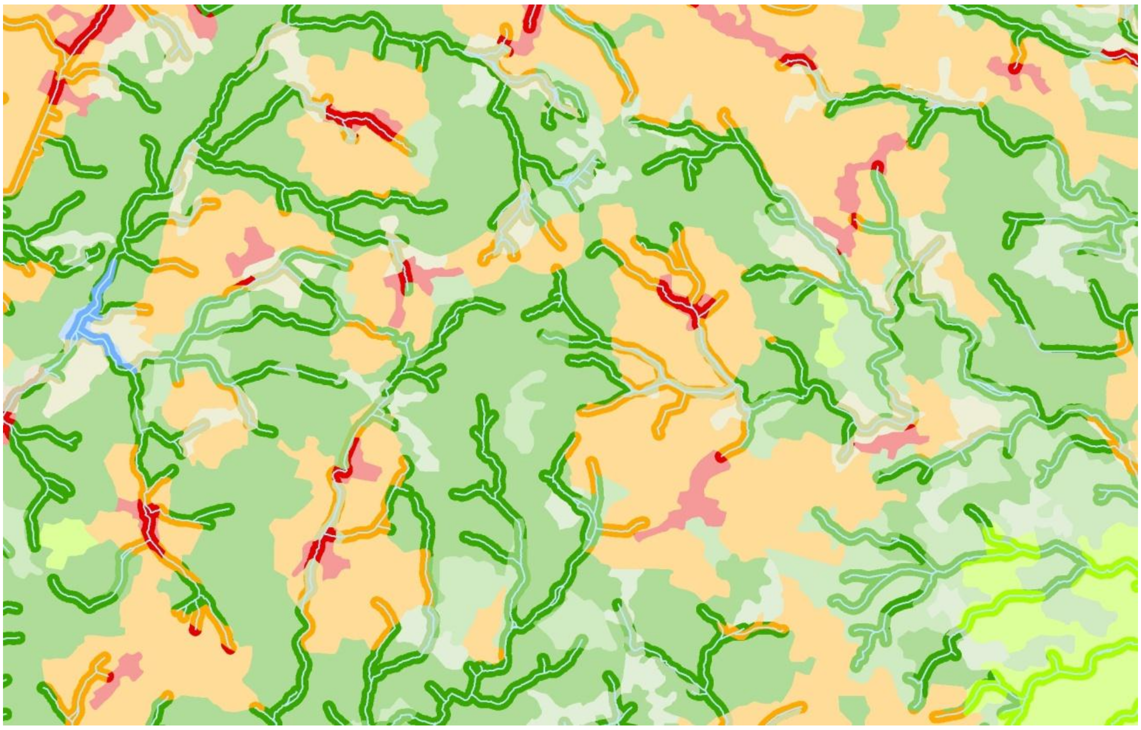


Figure 9.3. CORINE land cover within stream corridors and catchments.

9.1.4 Results

Based on correlation analyses and visualization of data we considered altitude, stream order as main typological parameters (Figure 9.4). The proportion of agriculture in corridors/catchments, natural vegetation in corridors and urban land use in corridors were found as descriptors of anthropogenic risks and buffering capacity.

Agriculture in catchment and upstream corridors of extent 2 km, 5 km and entire stream network correlated with altitude (all Pearson correlation coefficients $p<0.001$). Similar relationship to altitude was found for natural vegetation extent in upstream corridors (2 km, 5 km, total) and catchment (Figure 9.3).

	bf100_all_agr	bf100_all_natveget	catch_all_agr	catch_all_natveget	bf100_5_agr	bf100_2_agr	bf100_2_natveget
r	-0.52	0.43	-0.51	0.48	-0.45	-0.36	0.41

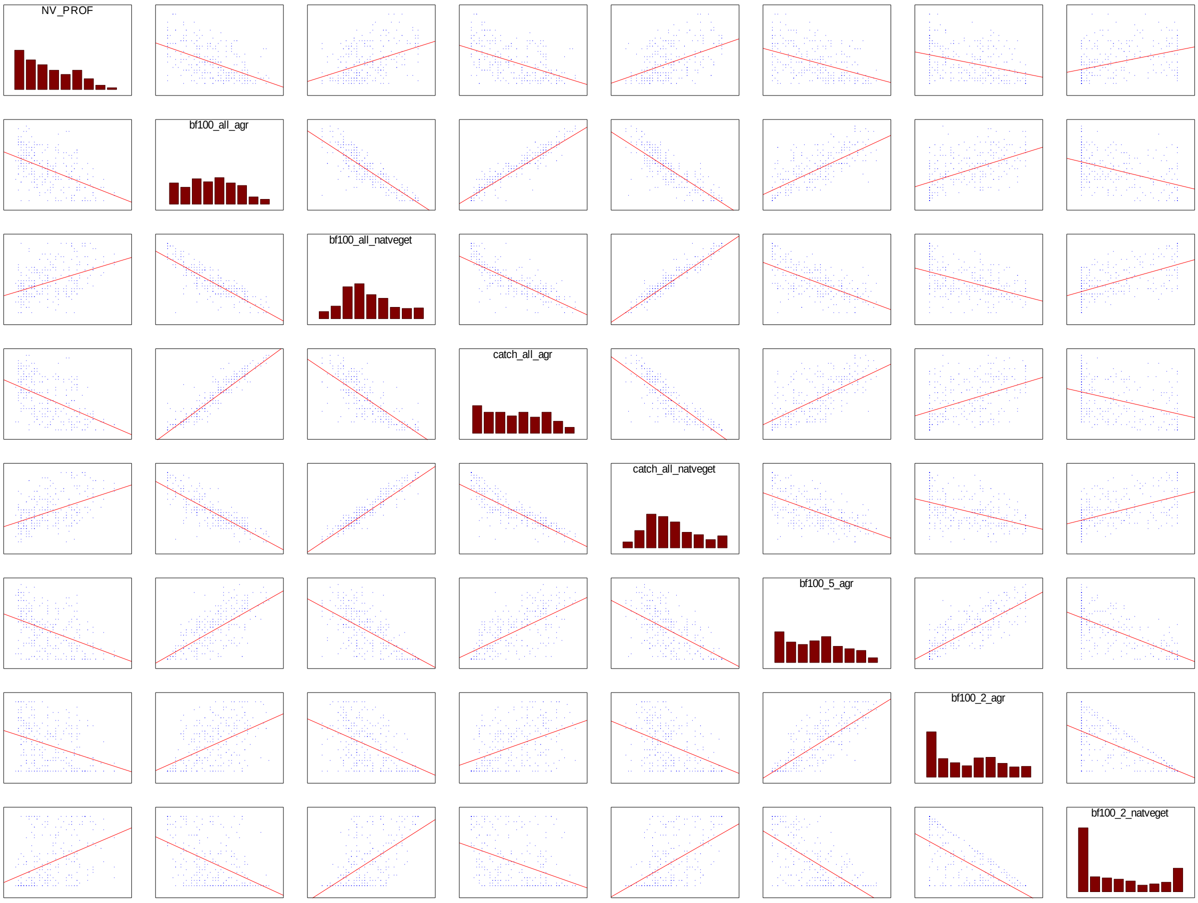


Figure 9.4. Relationships among altitude and land use variables (N=272); values above charts are Pearson correlation coefficient values for altitude.

Relationships among biological metrics

In conditions of selected streams in the Czech Republic were evident relationships among SPEARpesticides and other macroinvertebrate metrics being reported as indicators of general (complex) degradation.

There were publications where specific indication of SPEARpesticides was supported by absence of relationships of this index with other biotic indices. Von der Ohe et al (2007) reported no significant relationship between SPEARpesticides and BMWP indices. We found relatively weak correlation between these indices in our dataset. However ASPT (Average Score Per Taxon) index derived from BMWP correlated with SPEARpesticides significantly (Table 9.1 and Figure 9.5).

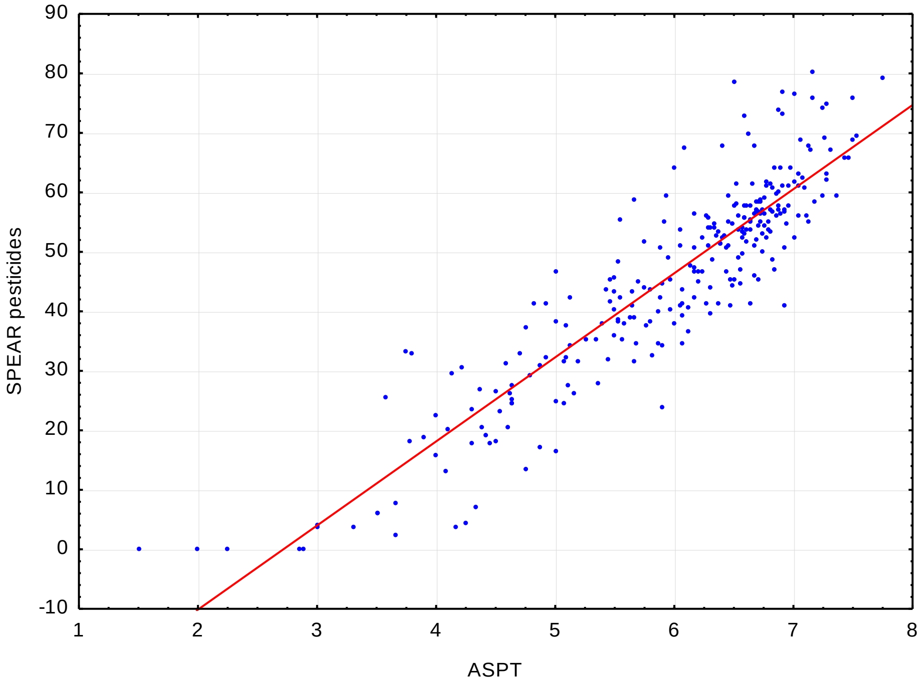


Figure 9.5. Relationship between SPEARpesticide and ASPT indices (r^2=0.82, p<0.001, N=272).

We included in study macroinvertebrate metrics which have been referred as indicators of land use, riparian zone characteristics and hydromorphological conditions (or potentially sensitive to these factors). Our aim was also to include various types of metrics (biotic indices, microhabitat/substrate preferences, feeding strategies, taxonomic richness and community composition. Additionaly to relationship between ASPT and SPEARpesticides indices we found another related metrics (Table 9.1). Number of genera and families reached the highest observed correlation (r=0.95, p<0.001). Relative abundance of EPT taxa correlated with lithal microhabitat preference, and SPEARpesticides. Czech version of Saprobic index was highly related to longitudinal zonation of macroinvertebrate communities (Index of Biocenotic Region), lithal preference and Life index.

Table 9.1. Pearson correlation coefficient among selected macroinvertebrate metrics (p<0.001 marked in red).

Metric	code	b34	b39	SPEAR	b83	b87	b120	b124	b135	b148	b151	b154	b155	b168	b190	b207	b279	b281	b283	b322
Czech Saprobic Index	b34	1.00	-0.68	-0.72	0.17	-0.68	-0.74	-0.82	-0.76	-0.67	-0.22	-0.40	0.64	-0.60	0.55	-0.60	-0.35	-0.26	0.76	-0.68
Average score per Taxon	b39	-0.68	1.00	0.90	0.16	0.73	0.68	0.73	0.54	0.42	0.18	0.13	-0.43	0.33	-0.46	0.60	0.59	0.49	-0.51	0.71
SPEARpesticides	SPEAR	-0.72	0.90	1.00	0.07	0.63	0.73	0.75	0.61	0.46	0.26	0.15	-0.49	0.44	-0.53	0.74	0.38	0.30	-0.58	0.74
Potamon-Typie-Index	b83	0.17	0.16	0.07	1.00	0.10	-0.01	-0.15	-0.07	-0.13	0.00	-0.26	0.06	-0.20	-0.04	0.03	0.27	0.21	0.25	-0.04
Number of sensitive taxa (Austria)	b87	-0.68	0.73	0.63	0.10	1.00	0.57	0.73	0.53	0.56	0.22	0.12	-0.40	0.35	-0.31	0.48	0.76	0.72	-0.49	0.50
Rheoindex (Banning, with abundance)	b120	-0.74	0.68	0.73	-0.01	0.57	1.00	0.74	0.64	0.42	0.28	0.18	-0.53	0.46	-0.59	0.67	0.38	0.27	-0.74	0.76
Rhithron Typie Index	b124	-0.82	0.73	0.75	-0.15	0.73	0.74	1.00	0.73	0.65	0.35	0.18	-0.56	0.52	-0.43	0.67	0.41	0.30	-0.72	0.76
Lithal microhabitat preference	b135	-0.76	0.54	0.61	-0.07	0.53	0.64	0.73	1.00	0.58	0.30	0.17	-0.66	0.44	-0.51	0.72	0.21	0.12	-0.62	0.61
Stone-dwelling taxa (Braukmann)	b148	-0.67	0.42	0.46	-0.13	0.56	0.42	0.65	0.58	1.00	0.36	0.09	-0.45	0.45	-0.29	0.44	0.26	0.19	-0.58	0.50
Grazers and scrapers	b151	-0.22	0.18	0.26	0.00	0.22	0.28	0.35	0.30	0.36	1.00	-0.24	-0.17	0.58	-0.07	0.49	0.07	-0.01	-0.25	0.40
Shredders	b154	-0.40	0.13	0.15	-0.26	0.12	0.18	0.18	0.17	0.09	-0.24	1.00	-0.37	0.58	-0.20	0.03	-0.01	-0.03	-0.28	0.15
Gatherers/Collectors	b155	0.64	-0.43	-0.49	0.06	-0.40	-0.53	-0.56	-0.66	-0.45	-0.17	-0.37	1.00	-0.59	0.49	-0.43	-0.14	-0.04	0.55	-0.48
RETI	b168	-0.60	0.33	0.44	-0.20	0.35	0.46	0.52	0.44	0.45	0.58	0.58	-0.59	1.00	-0.29	0.49	0.07	-0.02	-0.52	0.50
Oligochaeta (%)	b190	0.55	-0.46	-0.53	-0.04	-0.31	-0.59	-0.43	-0.51	-0.29	-0.07	-0.20	0.49	-0.29	1.00	-0.45	-0.11	-0.07	0.39	-0.44
EPT-taxa (%)	b207	-0.60	0.60	0.74	0.03	0.48	0.67	0.67	0.72	0.44	0.49	0.03	-0.43	0.49	-0.45	1.00	0.22	0.12	-0.53	0.63
Number of families	b279	-0.35	0.59	0.38	0.27	0.76	0.38	0.41	0.21	0.26	0.07	-0.01	-0.14	0.07	-0.11	0.22	1.00	0.95	-0.24	0.25
Number of genera	b281	-0.26	0.49	0.30	0.21	0.72	0.27	0.30	0.12	0.19	-0.01	-0.03	-0.04	-0.02	-0.07	0.12	0.95	1.00	-0.14	0.15
Index of Biocoenotic Region	b283	0.76	-0.51	-0.58	0.25	-0.49	-0.74	-0.72	-0.62	-0.58	-0.25	-0.28	0.55	-0.52	0.39	-0.53	-0.24	-0.14	1.00	-0.66
Life Index	b322	-0.68	0.71	0.74	-0.04	0.50	0.76	0.76	0.61	0.50	0.40	0.15	-0.48	0.50	-0.44	0.63	0.25	0.15	-0.66	1.00

Macroinvertebrate response to environmental risks

First analyses of relationships among biological metrics and characteristics of land adjacent to streams were done for entire dataset (272 sites). It was found that proportion of natural vegetation in stream corridor has slightly higher importance for selected biological metrics than proportion of agriculture in catchment and stream corridor (Table 9.2). Generally these correlations were not very high and only in several cases exceeded value 0.6 (e.g. Figure 9.6).

Table 9.2. Correlations between biological metrics and land use parameters (Pearson correlation coefficient, N=272, p<0.001 is marked in red).

		catchment			corridor - entire network			corridor - 5 km		corridor - 2 km		
metric		agr	natveget	LU211	agr	natveget	LU211	agr	LU211	agr	natveget	LU211
Czech Saprobic Index	b34	0.54	-0.64	0.51	0.57	-0.65	0.55	0.54	0.56	0.39	-0.62	0.46
Average score per Taxon	b39	-0.48	0.54	-0.46	-0.46	0.53	-0.48	-0.40	-0.50	-0.25	0.52	-0.40
SPEARpesticides	SPEAR	-0.53	0.59	-0.52	-0.51	0.56	-0.54	-0.43	-0.56	-0.30	0.53	-0.47
Potamon-Typie-Index	b83	0.18	-0.16	0.16	0.23	-0.17	0.18	0.13	0.13	0.15	-0.11	0.11
Number of sensitive taxa (Austria)	b87	-0.42	0.53	-0.39	-0.44	0.55	-0.39	-0.45	-0.44	-0.36	0.58	-0.37
Rheoindex (Banning, with abundance)	b120	-0.43	0.46	-0.45	-0.42	0.45	-0.50	-0.38	-0.54	-0.27	0.49	-0.48
Rhithron Typie Index	b124	-0.59	0.63	-0.59	-0.59	0.63	-0.61	-0.53	-0.61	-0.42	0.58	-0.55
Lithal microhabitat preference	b135	-0.46	0.46	-0.43	-0.47	0.47	-0.46	-0.46	-0.48	-0.32	0.49	-0.42
Stone-dwelling taxa (Braukmann)	b148	-0.50	0.55	-0.46	-0.51	0.56	-0.46	-0.44	-0.39	-0.34	0.43	-0.32
Grazers and scrapers	b151	-0.25	0.20	-0.30	-0.17	0.15	-0.29	-0.11	-0.24	-0.06	0.04	-0.21
Shredders	b154	-0.15	0.24	-0.08	-0.20	0.27	-0.09	-0.23	-0.11	-0.18	0.27	-0.10
Gatherers/Collectors	b155	0.40	-0.37	0.38	0.38	-0.37	0.39	0.43	0.45	0.34	-0.42	0.42
RETI	b168	-0.41	0.44	-0.39	-0.38	0.42	-0.38	-0.36	-0.37	-0.27	0.33	-0.33
Oligochaeta (%)	b190	0.29	-0.27	0.31	0.25	-0.24	0.31	0.26	0.31	0.14	-0.30	0.25
EPT-taxa (%)	b207	-0.49	0.49	-0.50	-0.43	0.44	-0.48	-0.35	-0.49	-0.25	0.40	-0.44
Number of families	b279	-0.19	0.28	-0.18	-0.20	0.29	-0.19	-0.19	-0.25	-0.16	0.33	-0.19
Number of genera	b281	-0.13	0.25	-0.12	-0.15	0.26	-0.12	-0.15	-0.17	-0.16	0.31	-0.13
Index of Biocoenotic Region	b283	0.47	-0.50	0.46	0.49	-0.50	0.52	0.44	0.51	0.37	-0.48	0.50
Life Index	b322	-0.49	0.49	-0.51	-0.47	0.47	-0.55	-0.39	-0.54	-0.30	0.41	-0.47

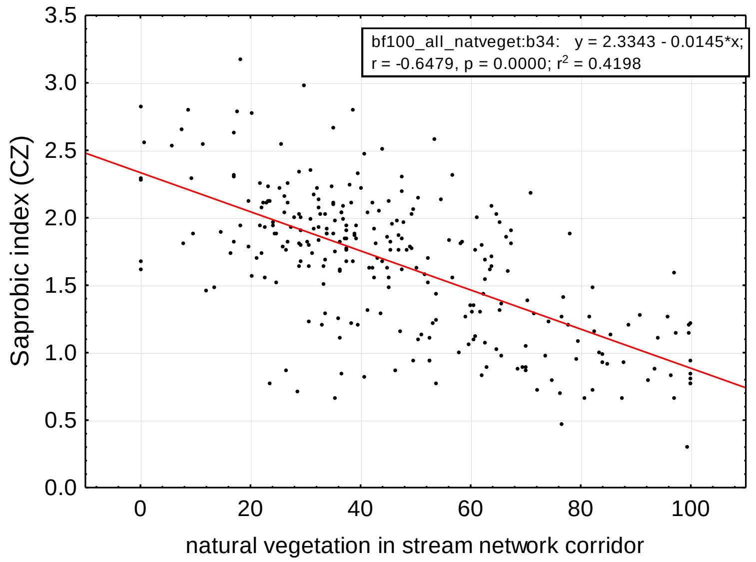


Figure 9.6. Relationship between Saprobic index and proportion of natural vegetation in stream corridor of entire stream network (N=272).

Based on these results we proceeded to analyses within stream types defined by categorization of altitude, stream order and land use.

Modelled pesticide risk in studied catchments

We aimed to analyze specificity of macroinvertebrate metrics to particular stressors. We found some relationships between theoretically specific and general indicators (e.g. SPEARpesticides and ASPT). Then we searched for factors describing potential pesticide risk.

Since we have not available data on pesticide concentration in stream water we decided to analyze DDT risk within studied catchments (Kubosova et al. 2009) (Figure 9.7). Another environmental variable being applied for description of pesticide risk is runoff potential and ecological risk (Schriever & Liess 2007).

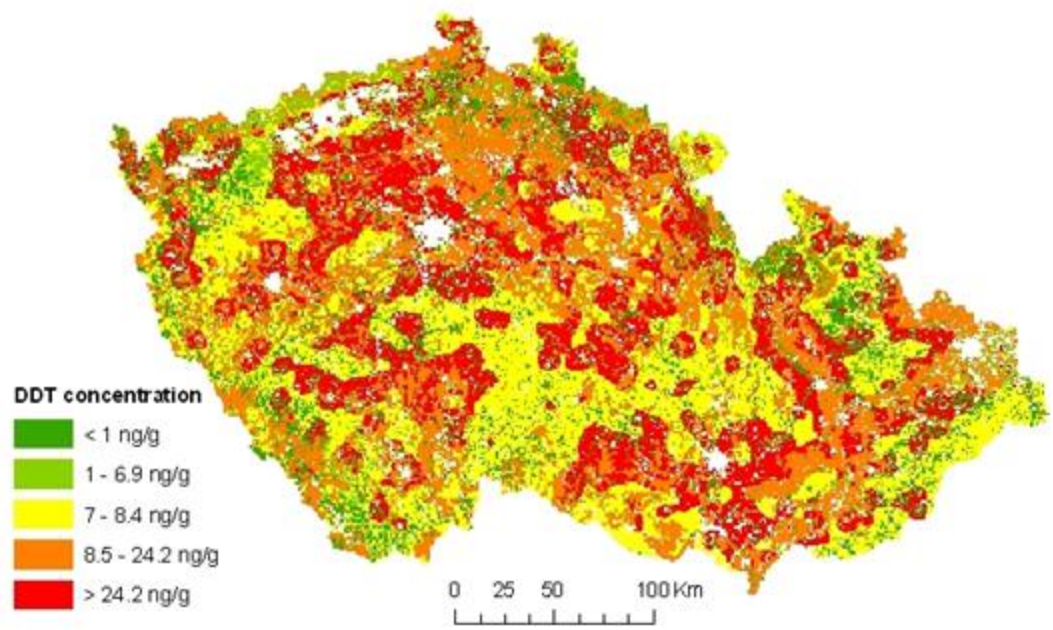


Figure 9.7. Modelled DDT concentration in the Czech Republic (except urban areas) – taken from Kubosova et al, 2009.

Table 9.3. Correlations between biological metrics and toxic compounds risk.

		DDT model					Runoff potential					Ecological Risk			
metric		DDT <1.0	DDT 1.0-6.9	DDT 7.0-8.4	DDT 8.5-24.2	DDT >24.2	RP1	RP2	RP3	RP4	RP5	ER1	ER2	ER3	ER4
Czech Saprobic Index	b34	-0.23	0.03	-0.37	0.18	0.31	-0.26	-0.27	-0.07	0.31	-0.06	-0.36	-0.24	0.16	0.44
Average score per Taxon	b39	0.29	-0.09	0.35	-0.27	-0.16	0.09	0.21	0.11	-0.23	0.07	0.20	0.28	-0.04	-0.48
SPEARpesticides	SPEAR	0.27	-0.05	0.31	-0.22	-0.17	0.15	0.27	0.09	-0.28	0.05	0.29	0.29	-0.10	-0.49
Potamon-Typie-Index	b83	0.01	-0.09	0.05	-0.01	0.03	-0.01	-0.11	-0.02	0.08	-0.06	-0.08	-0.09	0.15	0.03
Number of sensitive taxa (Austria)	b87	0.22	-0.13	0.40	-0.19	-0.29	0.25	0.20	0.03	-0.25	0.13	0.31	0.12	-0.05	-0.36
Rheoindex (Banning, with abundance)	b120	0.25	0.00	0.25	-0.17	-0.18	0.20	0.17	0.09	-0.24	0.04	0.25	0.22	-0.05	-0.43
Rhithron Typie Index	b124	0.28	-0.01	0.37	-0.28	-0.29	0.23	0.35	0.15	-0.39	0.10	0.39	0.26	-0.13	-0.52
Lithal microhabitat preference	b135	0.21	0.00	0.26	-0.16	-0.25	0.24	0.24	0.07	-0.29	0.02	0.32	0.19	-0.12	-0.38
Stone-dwelling taxa (Braukmann)	b148	0.18	0.09	0.18	-0.05	-0.25	0.32	0.21	0.05	-0.30	0.07	0.36	0.24	-0.25	-0.34
Grazers and scrapers	b151	0.24	0.07	0.12	-0.15	-0.18	0.35	0.23	-0.03	-0.25	-0.02	0.39	0.05	-0.15	-0.24
Shredders	b154	-0.03	-0.09	0.18	0.04	-0.22	0.01	0.05	0.06	-0.07	0.11	0.04	0.10	-0.10	-0.06
Gatherers/Collectors	b155	-0.20	-0.04	-0.15	0.04	0.27	-0.16	-0.19	-0.19	0.32	-0.08	-0.24	-0.22	0.12	0.35
RETI	b168	0.16	0.04	0.25	-0.07	-0.36	0.30	0.27	0.07	-0.33	0.09	0.39	0.16	-0.22	-0.30
Oligochaeta (%)	b190	-0.12	-0.07	-0.04	0.10	0.03	-0.14	-0.16	-0.09	0.21	-0.04	-0.20	-0.14	0.09	0.25
EPT-taxa (%)	b207	0.30	-0.07	0.24	-0.15	-0.21	0.27	0.27	0.12	-0.35	-0.03	0.37	0.20	-0.13	-0.42
Number of families	b279	0.16	-0.09	0.31	-0.16	-0.19	0.13	0.04	0.06	-0.13	0.07	0.12	0.11	0.01	-0.24
Number of genera	b281	0.11	-0.09	0.28	-0.14	-0.14	0.11	0.01	0.03	-0.08	0.09	0.08	0.07	0.02	-0.18
Index of Biocoenotic Region	b283	-0.20	-0.14	-0.20	0.09	0.27	-0.24	-0.24	-0.13	0.33	-0.05	-0.33	-0.24	0.17	0.39
Life Index	b322	0.30	0.04	0.26	-0.24	-0.20	0.23	0.23	0.06	-0.27	0.08	0.32	0.23	-0.07	-0.49

Macroinvertebrate response to pesticide risk is lower than to more general land use parameters. The highest correlation was found among Ecological Risk (sensu Schriever & Liess, 2007) and biotic indices (Table 9.3).

Analyses based on categorized dataset

Theoretical ideal indicator should be constant across natural gradients and strongly responding to anthropogenic pressures. In real conditions natural and stress factors are interacting. Therefore categorization of studied sites is needed. Water Framework Directive suggested stream typology based on altitude, stream size and natural characteristics. First we need to explore how fine categorization would provide enough wide stress gradient and enough number of sites within categories (Tabs. 9.4, 9.5 and 9.6).

Table 9.4. Frequency table based on altitude categories and Strahler stream order.

		Strahler stream order						Total
		1	2	3	4	5	6	
Altitude (m a.s.l.)	200-300	0	3	14	36	28	12	**93**
	300-400	0	5	3	28	23	5	**64**
	400-500	0	4	8	20	12	1	**45**
	500-600	1	5	17	18	7	0	**48**
	600-700	0	7	5	6	0	0	**18**
	700-800	1	1	0	2	0	0	**4**
Total		**2**	**25**	**47**	**110**	**70**	**18**	**272**

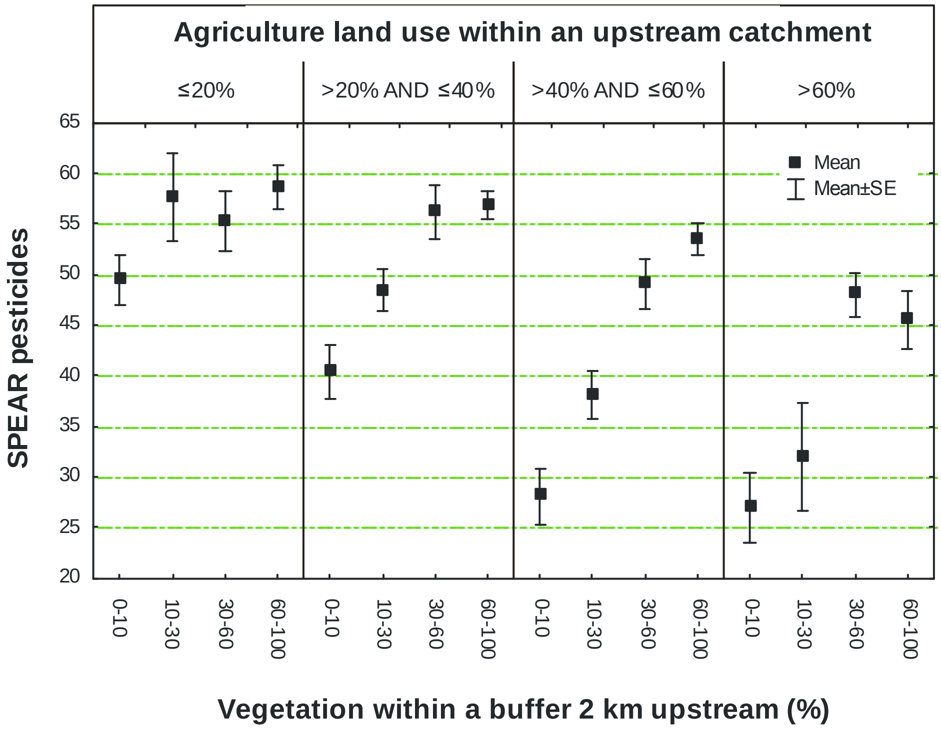
Table 9.5. Frequency table of agriculture landuse in catchment and natural vegetation in corridor.

		Natural vegetation in corridor of 2 km extent (%) (bf100_2_natveget)				Total
		0-10	10-30	30-60	60-100	
Agriculture in catchment (%) (catch_all_agr)	0-20	13	16	16	39	84
	20-40	23	10	18	16	67
	40-60	34	16	14	8	72
	60-100	30	4	6	9	49
Total		100	46	54	72	272

Table 9.6. Frequency table of agriculture landuse in catchment and urban area in corridor.

		Urban area in corridor of 5 km extent (%) (bf100_5_112)				**Total**
		0	0-10	10-20	20-100	
Agriculture in catchment (%) (catch_all_agr)	0-20	45	19	10	10	**84**
	20-40	18	19	15	15	**67**
	40-60	18	21	15	18	**72**
	60-100	5	26	13	5	**49**

Total	**86**	**85**	**53**	**48**	**272**



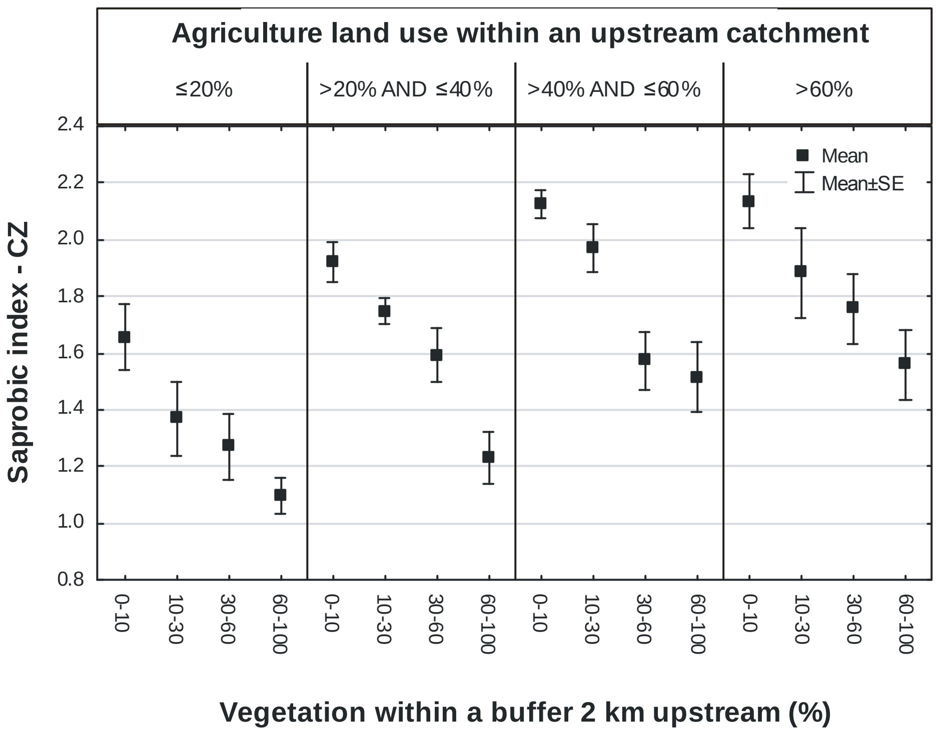


Figure 9.8. Variation of SPEARpesticide index and Saprobic index among categories of agriculture in upstream catchment and vegetation in 200m-wide stream corridor within 2km upstream the site.

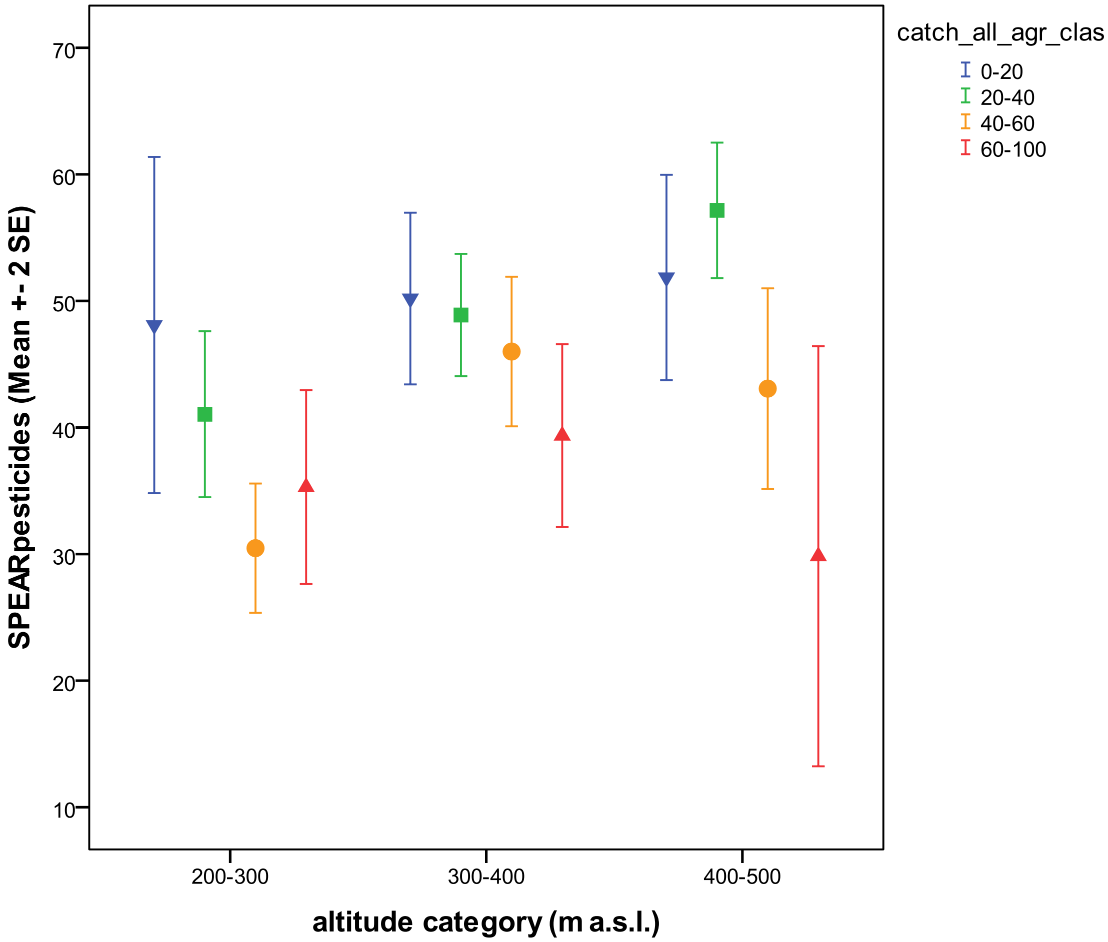


Figure 9.9. SPEARpesticides in relation to agriculture in catchment, different patterns in altitudinal categories (RT2 only = Strahler stream order 4-6).

The most distinct pattern along agriculture proportion gradient is obvious within altitudes 400-500 m a.s.l. With decreasing altitude differences in SPEAR index decrease as well within altitudinal categories. Considering agriculture categories separately SPEAR index showed incerase with increasing altitude for categories with agriculture proportion in catchment 0-40 %. Both higher categories (40-60 % and 60-100%) were associated with decrease of SPEAR mean at altitudes 400-500 m a.s.l. below trend of lower altitudes.

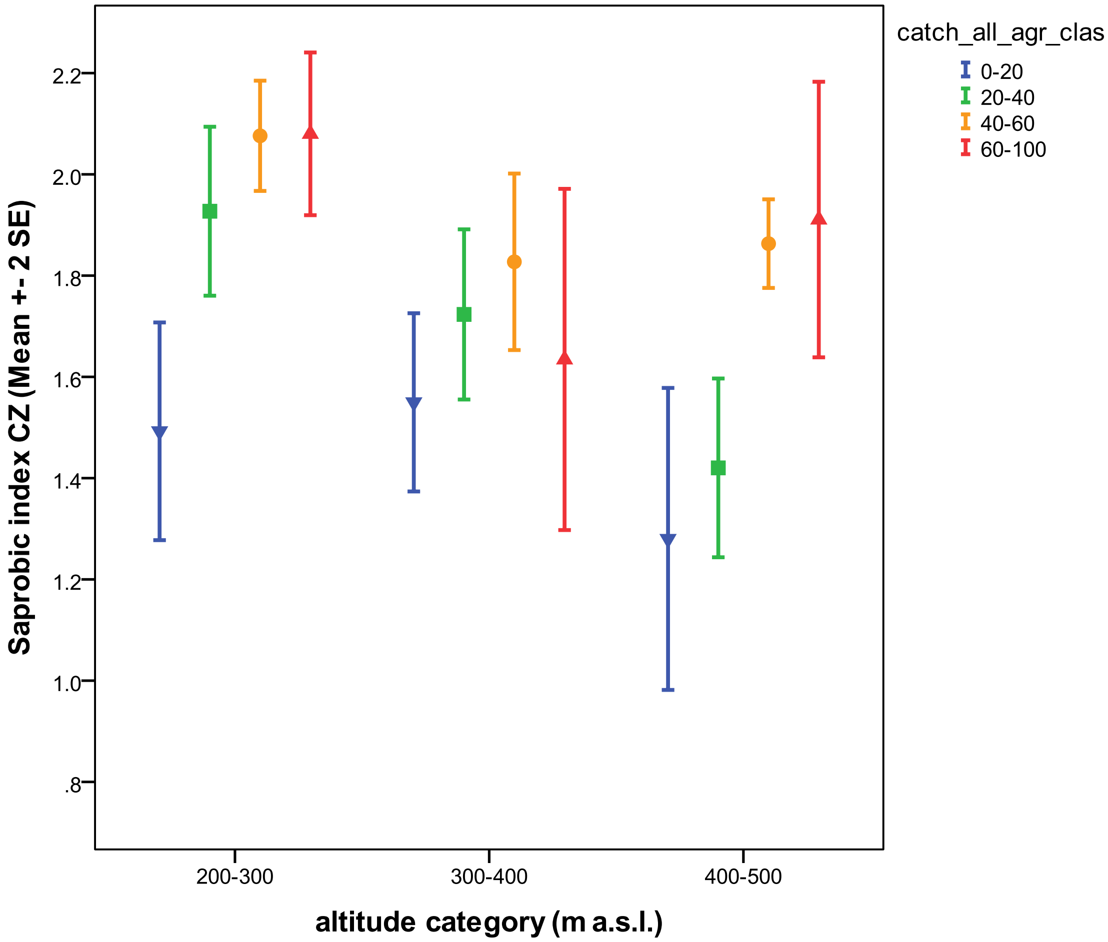


Figure 9.10. Saprobic index (CZ) in relation to agriculture in catchment, different patterns in altitudinal categories (RT2 only = Strahler stream order 4-6).

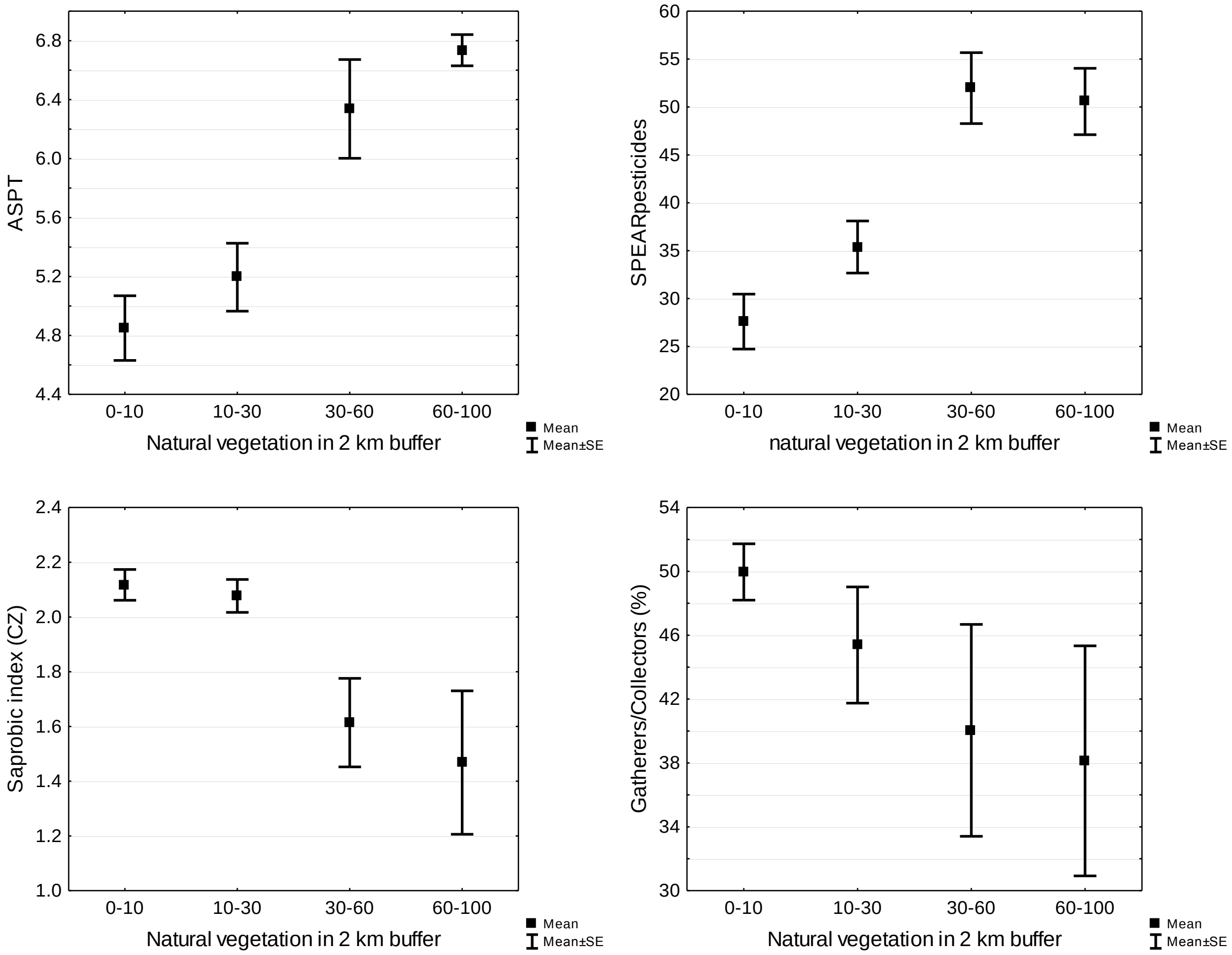


Figure 9.11. Effect of natural vegetation in 2 km corridor on macroinvertebrate metrics (selection of sites of stream order 4-6, altitude 200-350 m a.s.l., percentage of agriculture in catchment 40-60%).

SPEARpesticides responded not only to agriculture risk but also to urbanization of stream corridor with potential point sources of sewage water (Figure 9.12). We found significant relationships among biological metrics and stream corridor/catchment characteristics which were more clear within particular stream types (Figures 9.8, 9.11, 9.13).

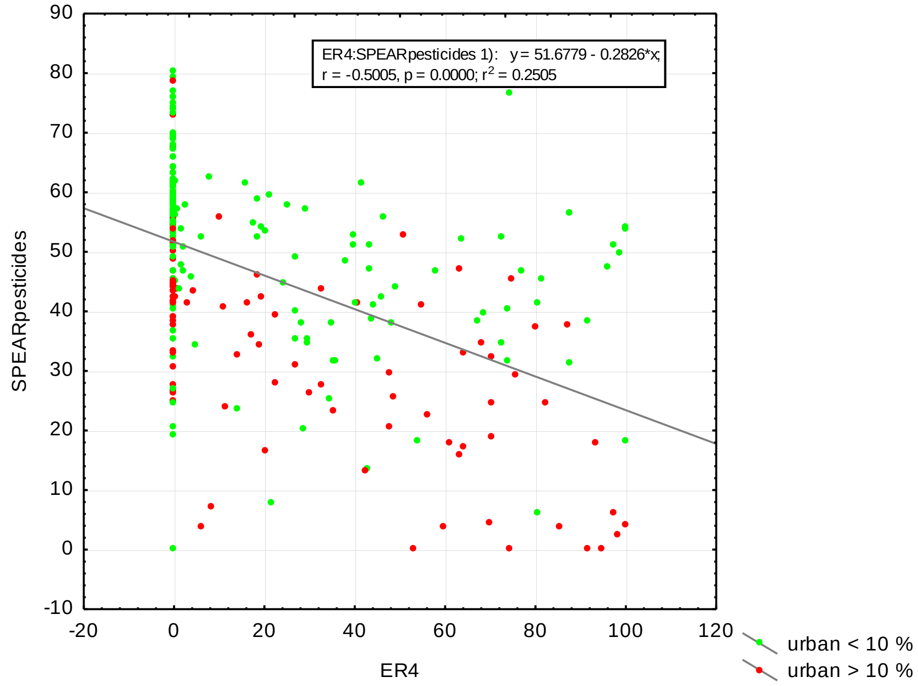


Figure 9.12. SPEARpesticides index response to proportion of the highest Ecological Risk category in catchment (sensu Schriever & Liess, 2007); point colour indicates threshold of urban land use in 5 km corridor (10%).

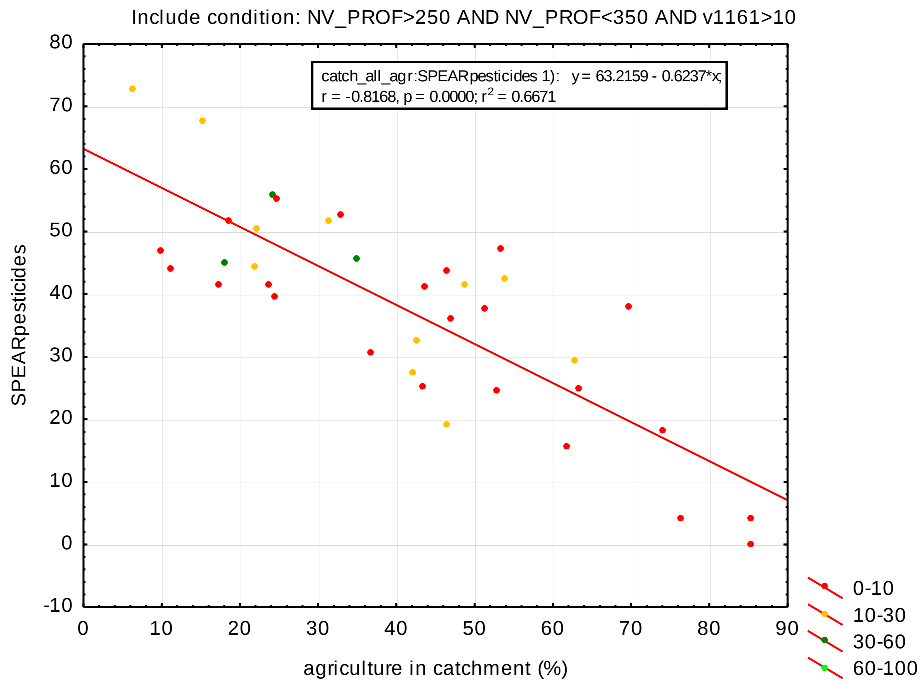


Figure 9.13. SPEARpesticides response to agriculture proportion in catchment within altitude category 250-350 m a.s.l. and proportion of urban land use in 5 km corridor higher than 10% (symbol colour is based on proportion of vegetation in 2 km corridor).

9.1.5 Conclusion

Results showed difficulties with separation of effects of agriculture, urbanization and natural vegetation on macroinvertebrate communities due to dependence all factors on altitude. When we stratified data set using categories of these parameters only minority of combined categories allowed suficient number of samples distributed along studied gradients to be statisticaly relelevant.

Data set we used for this study do not allow to evaluate relationship of SPEARpesticide index to pesticide concentration in streams. However we found strong relationship between SPEARpesticide and other macroinvertebrate metrics. Significant relationships of these interrelated biological indicators were found with agriculture and urban land use in stream corridor and catchment. Additionally metrics correlated with proportion of natural or semi natural vegetation in stream corridor upstream studied sites.

9.1.6 References

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Von der Ohe, P.C., Prüß, A., Schäfer, R.B., Liess, M., de Deckere, E., Brack, W., 2007. Water quality indices across Europe—a comparison of the good ecological status of five river basins. Journal of Environmental Monitoring 9, 970–978.

10 Summary and conclusions

HYMO indicators of degradation

A possible approach for developing a method of evaluating the ecological and morphological conditions of a river influenced by human intervention is presented. The method is based on a source pool of detailed physical parameters and indicators (metrics) that are linked to the data and outputs of other work packages (WP1 & WP2) within the REFORM Project. Depending on the focus of an evaluation (to choose from morphology, vegetation, benthos and/or fish), experts can use these approaches to identify a subset of key indicators from this pool. When using the approach an evaluation is performed comparatively between the benchmark condition of the river and the river condition affected by human intervention. The output, an informed choice of key metrics, aims to support the stakeholder decision making processes and their ability to target desired project goals. These indicators of degradation should be viewed as an interim solution while a more comprehensive and tested approach is produced from WP2 and the final system developed will be an integral part of WP6. The impact of hydromorphological degradation on individual biological Quality elements is reviewed in the subsequent chapters.

Phytobenthos

It was not possible to detect any effect of the hydromorphological alterations tested, which included alterations that influence the flow velocity, the rate of sedimentation and the in-stream habitat on metrics based on phytobenthos, although it is reassuring that metrics developed to assess eutrophication stress (e.g. TDI, IPS and related indices) appear robust to hydromorphological alteration. Furthermore, it was not possible to demonstrate an effective response of the proposed index of fine sediment stress based on phytobenthos; % motile taxa appears to be related more to nutrient availability than to fine sediment.

Macrophytes

No macrophyte metrics sensitive to hydromorphological pressures exist despite extensive literature on the well-described and consistent responses of vegetation to damming, weed cutting and dredging. To make the best use of pre-existing monitoring datasets a trait-based approach was used to examine the potential for new metrics. Clear differences in the dominance of plant morphotypes were detected between rivers of different geomorphological style, indicating plant responses may differ between different river styles. While general responses to hydromorphological degradation were difficult to detect in some of the large noisy datasets, clear responses of some plant traits were detected in others. For stakeholders, the results indicate plants have particularly strong potential as useful metrics, especially given their intimate role in geomorphological processes but significant development is required.

Macroinvertebrates

Metrics developed to detect hydrological impairment and hydromorphological degradation were not more discriminative than a number of metrics sensitive to other pressures. These findings leave water managers with a significant challenge when diagnosing the reason for not obtaining good ecological status in a waterbody. The reasons for the lack of sensitivity can be attributed to a number of different factors; explanatory variables which are not measured as part of routine monitoring programmes and hydromorphological assessment schemes that do not necessarily record variables of importance to the in-stream biota. In addition, the present findings suggest that both metric development and sampling scales need to be scrutinised to improve sensitivity.

Fish

Sensitivity of fish to hydromorphological pressures was detected. Logistic regression analyses revealed 69% of the analysed European freshwater species display a significant (>90% c.l.) response to HYMO pressures. Responses could be both positive and negative depending on whether an alteration had improved or degraded the habitat suitability for a species. The confounding effects of multiple stressors on these potential metrics will be elucidated in Deliverable 3.2.

Joint BQE

Initial tests on data on diatoms, fish and invertebrates collected at the same sites revealed that Ecological Quality Ratios for these groups follow broadly similar patterns. Many of the sites examined were subject to multiple pressures, and the results demonstrated that with these data development of a joint metric sensitive to hydromorphological pressures was confounded by the overwhelming signal from sensitivity to water quality. The data, from Finland, exhibit strong water quality gradients but relatively weak hydromorphological gradients.

Fine sediments

Excess fine sediment input is a diffuse form of hydromorphological pressure which is widespread throughout Europe with known or potential impacts on all BQEs. It is not possible to parameterise it effectively using data collected by standard hydromorphological monitoring techniques such as the River Habitat Survey. The project had access to a specialist dataset where this pressure was examined directly, and the response of invertebrates could be elucidated by using traits. In general, invertebrates shifted to smaller taxa in response to fine sediment input suggesting that there is the potential to develop biotic metrics sensitive to this pressure.

Habitats Directive

The vulnerability of fish protected under the Habitats Directive and aquatic vegetation Habitat 3260 were examined. The potential to assess vulnerability using current monitoring techniques was reviewed for fish, and the extent of hydromorphological pressures at Special Areas of Conservation with Habitat 3260 was quantified. The future vulnerability of Habitat 3260 sites to changes in hydrology, driven by climate change and socio-economic scenarios, is presented graphically and suggests increasing vulnerability.

Sediment quality

The study showed that river water quality is not only dictated by diffuse or source point emissions of contaminants; it is also strongly related to the quality of sediments. High discharge events, which may occur more often in the future as predicted in future climate scenarios, may mobilise the associated contaminants. Increased contaminant loads at high discharge are commonly not signaled or detected by monitoring programmes because of the masking effect of particle size and dilution. An impact on ecosystem health in sedimentation areas cannot be excluded, however.

Groundwater

Both data analysis and spatially distributed groundwater-surface modelling showed that groundwater is an important driver of maintaining good environmental flows during dry periods in sandy catchments. Groundwater conditions and environmental flows have deteriorated due to anthropogenic changes over the past 150 years, and climate change will probably amplify this deterioration. Our study showed that catchment-scale alterations may significantly improve groundwater conditions and stream discharge, for instance via changes in groundwater abstraction regime, drainage systems and re-naturalisation projects.

Effect of stream corridor and catchment characteristics

Land-use in both stream corridors and catchments influences macroinvertebrate community composition although natural features such as altitude have to be considered. Significant relationships of a number of macroinvertebrate metrics were found with agriculture and urban land use in stream corridor and catchment. Additionally metrics correlated with proportion of natural or semi-natural vegetation in stream corridor upstream studied sites. These findings imply that land-use data could be a more robust way of assessing impacts than reach specific data and could be more relevant in the decision making process on land-use and HYMO degradation in a catchment management perspective.

Brief overall conclusions / Executive Summary

- There is an acknowledged need among stakeholders that new hydromorphological metrics are required to facilitate site remediation and for reporting at national and European levels.
- Pressure/ impact data were assembled from across Europe. The task was challenging, but useful information was gathered.
- For each major hydromorphological pressure, the physical response gradients of rivers was summarised as diagnostic diagrams.
- For the first time we provide evidence that metrics indicating HYMO impact could be developed from monitoring data on fish and macrophytes.
- For the first time we demonstrate the potential to derive metrics sensitive to fine sediment.
- We provide evidence that phytobenthos (diatoms), invertebrates and macrophytes have the potential to be used in combined metrics.
- We found that many existing macroinvertebrate metrics lack specificity and can provide false positive responses to HYMO pressure, suggesting that disentanglement of multi-stressor responses is critical to good diagnosis.
- There is evidence that aquatic habitats protected under the Habitats Directive will be increasingly vulnerable to hydrological pressures with the changing climate.
- Frequently, overlooked topics such as sediment quality and groundwater issues ought to supplement or be included in HYMO assessments due to their potential for explaining variance in biological datasets.

- Land-use data on a spatial scale beyond the reach scale (corridor and catchment) relates to site-specific macroinvertebrate metrics and could be a more robust way of assessing impacts.

Supplementary Material on Macrophyte Analyses

An analysis of Intercalibration data

Aim

Our aim was to investigate macrophyte trait characteristics in lowland streams in the Central Baltic region in Europe and analyse variation in community trait characteristics in relation to stream typology and hydromorphological pressures.
Specifically we test the following hypotheses:

1. Macrophyte community characteristics differ between small (IC type RC1) and medium-sized (IC type RC4) lowland streams situated in the Central Baltic region in Europe.
2. Hydromorphological degradation in terms of an altered channel morphology alters macrophyte trait characteristics in both small (RC1) and middle-sized (RC4) streams.

Methods

Data

A total of 772 stream sites were included in the data analysis all being part of the IC dataset (Birk and Wilby 2011). The stream sites were all situated in the Central Baltic area with sites in Germany (DE), Denmark (DK), Belgium, Flandern (BE (FL)), France (FR), Great Britain (UK (GB)), Northern Ireland (UK (NI)), Ireland (IE), Italy (IT), Lithuania (LT), Latvia (LV), Netherlands (NL), Poland (PL), and Belgium, Wallonia (BE (WL). The macrophyte communities were not clearly separated according to country or region as inferred from the result of a detrended correspondence analysis (DCA; Figure. 1). Data from the different countries were therefore treated in the same analyses.

Description of traits

A total of 120 submerged and amphibious species were present in the dataset, but we were only able to allocate traits to 77 species representing 64% of the total species pool. The traits covered ecological indicator values (Ellenberg N and L), life form and traits related to dispersal, reproduction and survival (Table 1). The Ellenberg indicator values (Ellenberg et al. 1991) offer autecological information on the response of some 2000 species to a range of climatic and edaphic factors in central Europe. Although the intuitive nature of the Ellenberg indicator values has been criticised (Thompson et al. 1993), we decided to integrate them in this study as they remain the most wide-ranging source of ecological plant species information available in Europe.

Short trait name	Explanation	Category
LE	Ellenberg Light	Ecological preference
NE	Ellenberg Nitrogen	Ecological preference
frflsr	Free floating, surface	Lifeform
frflsb	Free floating, submerged	Lifeform
anflle	Anchored, floating leaves	Lifeform
ansule	Anchored, submerged leaves	Lifeform
anemle	Anchored, emergent leaves	Lifeform
anhete	Anchored, heterophylly	Lifeform
meris.ma	Meristem single apical growth point	Morphology
meris.sb	Meristem single basal growth point	Morphology
meris.sa	Meristem multiple apical growth point	Morphology
meris.sa.ma	Meristem single+multiple apical growth point	Morphology

Short trait name	Explanation	Category
morph.ind	Morphology index =(height + lateral extension of the canopy)/2	Morphology
leaf.area	Leaf area	Morphology
seeds	Reproduction by seeds	Dispersal
rhizome	Reproduction by rhizomes	Dispersal
frag	Reproduction by fragmentation	Dispersal
n.rep.org	Number of reproductive organs per year and individual	Dispersal
overwintering.org	Overwintering organs	Survival

Data on the 19 traits were extracted from the literature and online databases (Willby et al. 2000; Klotz et al. 2002; Kühn et al. 2004). The trait life form (LF) was divided into 6 forms: free floating (surface or submerged), anchored with either floating or submerged leaves, and amphibious species with either homophyllus emergent leaves or heterophyllus emergent leaves. Growth morphology was divided into 4 forms: single basal, multi apical, single apical or multi apical-single apical (Table 1). Plant morphological traits included also a morphology index building on the height and lateral extension of the canopy and the leaf area of the species. Dispersal was characterised by 4 traits. Local dispersal was inferred from the root-rhizome system and canopy spread, while regional dispersal was extrapolated from the ability to disperse/reproduce by fragmentation as well as by the number of seeds and reproductive organs produced by the species. We integrated traits related to survival in terms of overwintering organs, such as tubers, turions and rhizomes, and in terms of regeneration after disturbance, for instance position and number of meristems.

The life form traits, and traits covering fragmentation, seeds, overwintering organs and rhizomes, were based on presence or absence of the attribute, with a score of 0 for absence, 1 for occasionally but not generally present attributes and 2 for present attributes. The morphology traits describing the meristem growth point type were based on presence (1) or absence (0) of the attribute. The number of reproductive organs was classified into low (<10), medium (10-100), high (100-1000) and very high (>1000), with values ranging from 1 to 4 based on number per individual per year. Leaf area was classified according to the leaf size categories with values ranging from 1-4 representing small (<1 cm^3), medium (1-20 cm^3), large (20-100 cm^3) and very large (>100 cm^3) leaf sizes. The morphology index was also classified into categories (2, 3-5, 6-7, 8-9 and 10) with values ranging from 1-5. In some cases species were classified in-between two categories regarding number of reproductive organs, leaf area and morphology index (Willby et al. 2000). In these cases a classification code in-between was allocated for the particular trait (i.e. 1.5, 2.5, 3.5 and 4.5).

Data analyses

The relationships between species abundance and trait variables were examined using multivariate ordination techniques. To analyse relationships between the spatial distribution of macrophyte species and their traits, we constructed two tables: a table with site and species abundance information and a table with species and trait information. First, we analysed the two tables separately and then used coinertia analysis (COIA) to couple the two tables (Dolédec et al. 1996; Dray et al. 2008). The site by species abundance table was analysed by applying a Principal Component Analyses (PCA) comprising 722 sites and 77 species. The species by trait table was also analysed with a PCA combining the 19 (log transformed) traits with the 77 species. The PCA constructs a distance matrix based on Euclidian distance to detect a linear combination of the original variables that maximises the variance.

The site by abundance table and species by trait table were then linked with COIA to study co-structure by maximising covariance between trait ordination scores and abundance scores in the 2 PCA (Dray et al. 2003). This method permits a combined analysis of ecological and spatial distribution of species and allows quantitative and qualitative data to be mixed. All the analyses were performed in R with the ADE4 package.

Groups of species sharing similar distributional patterns and ecological traits were then identified by performing a hierarchical clustering analysis based on the Euclidian distance results obtained in the COIA. We cut the cluster dendogram at five groups of species. The cluster analysis was run using the package STATS in R.

For a total of 221 stream sites we had information on channel morphology. Stream sites were categorised into either unmodified, slightly modified and highly modified channels based on an evaluation of lateral, longitudinal and transverse profile modifications. We compared trait values (community weighted means obtained through COIA) from unmodified, slightly and highly modified stream sites to analyse if specific traits could be identified that link to hydromorphological degradation in both small (RC1) and middle-sized streams (RC4).

Results

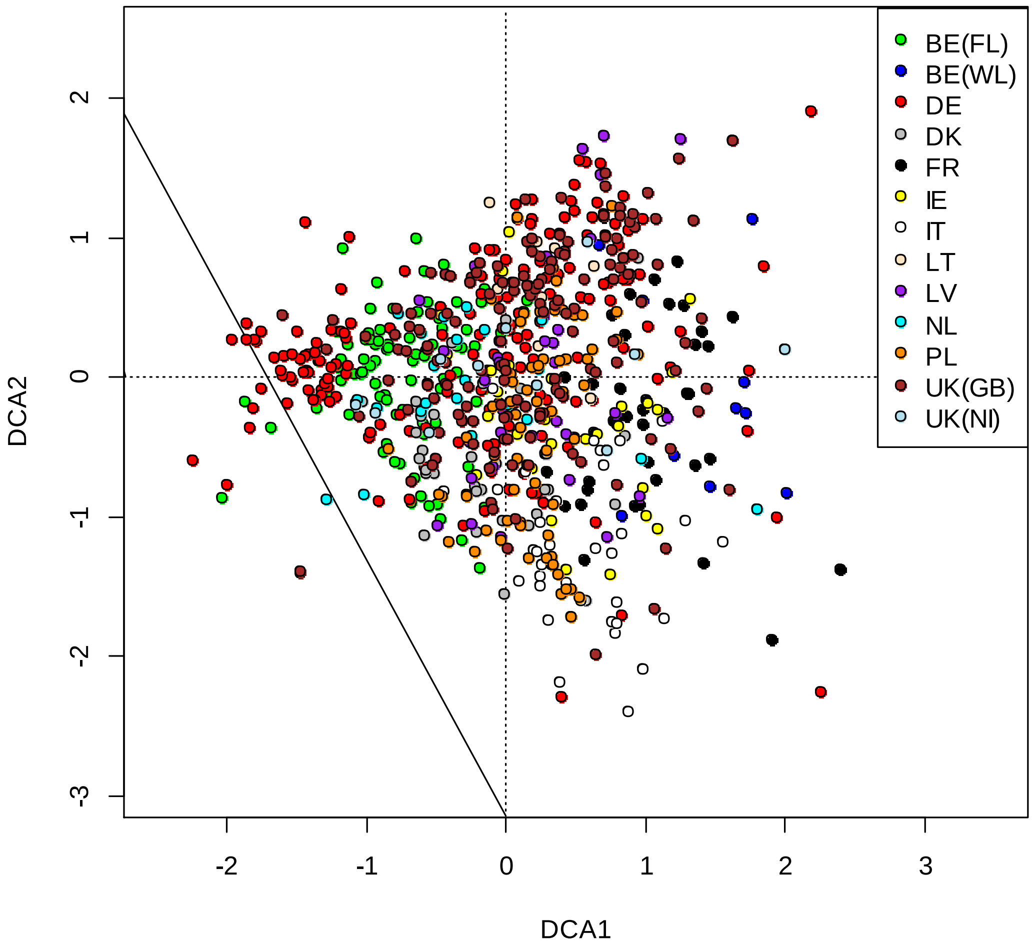
The most abundant species in the studied streams were Sparganium emersum, Sparganium erectum, Potamogeton pectinatus, Potamotegon natans, Nuphar lutea, Lemna minor and Elodea canadensis. We found that species abundance patterns and trait characteristics were significantly related (Monte Carlo test for COIA; p=0.019; Supplementary Figure 2). Both axis 1 and 2 separated species according to vegetative and regenerative characteristics (Supplementary Figure 2). Axis 1 separated species according to morphological traits such as meristem characteristics (single basal; single apical and multiple apical) and leaf area, and to life form traits (emergent and floating leaves) and dispersal, for instancegrowth from rhizomes and shoot fragmentation. Species positively associated with axis 1 (e.g. *Spargaium emersum*, S*parganium erectum*, *Nuphar lutea*) possessed single basal meristems, they had a high leaf area index and they were anchored to the stream bottom, but with an ability to reach the water surface with either floating leaves or leaves reaching out of the water. Species located to the left side on axis 1, on the other hand, possessed apical meristems. The second axis also separated species according to dispersal, morphology and lifef orm characteristics. Species positively associated with axis 2 produced a large number of seeds, and they were either anchored with heterophyllus leaves or free-floating, whereas species negatively associated with axis 2 were submerged species with multiple apical growth meristems and an ability to form extensive canopies.

We identified five species groups using axes 1-3 scores from the CoA in a cluster analysis (Supplementary Figure 3). Group 1 and 2 were large groups consisting of a mixture of submerged and amphibious species. Group 1 species possessed single and multiple apical meristems, while group 2 species possessed single apical meristems (Figure 4). Group 1 species spread more extensively by rhizome growth as compared to group 2 species. Group 3 consisted of primarily free floating species such as several *Lemna* species, *Hydrocharis morsus-ranae* and *Stratiotes aloides*, whereas group 4 consisted of submerged species, i.e. *E. canandensis*, *M. spicatum* and *P. pectinatus*. Group 5 consisted of homophyllus amphibious species, i.e. *S. emersum*, *S. erectum* and *S. sagittifolia* that able to produce both submerged and emerged leaves, and one floating leafed species, *N. lutea*. Additionally, groups 4 and 5 were separated with respect to growth meristems, with group 4 possessing multi-apical meristems and group 5 possessing single basal meristems, also with respect to dispersal with group 4 spreading by fragmentation and group 5 by seeds. Group 5 also had a higher specific leaf area compared to group 4 (Supplementary Figure 4).

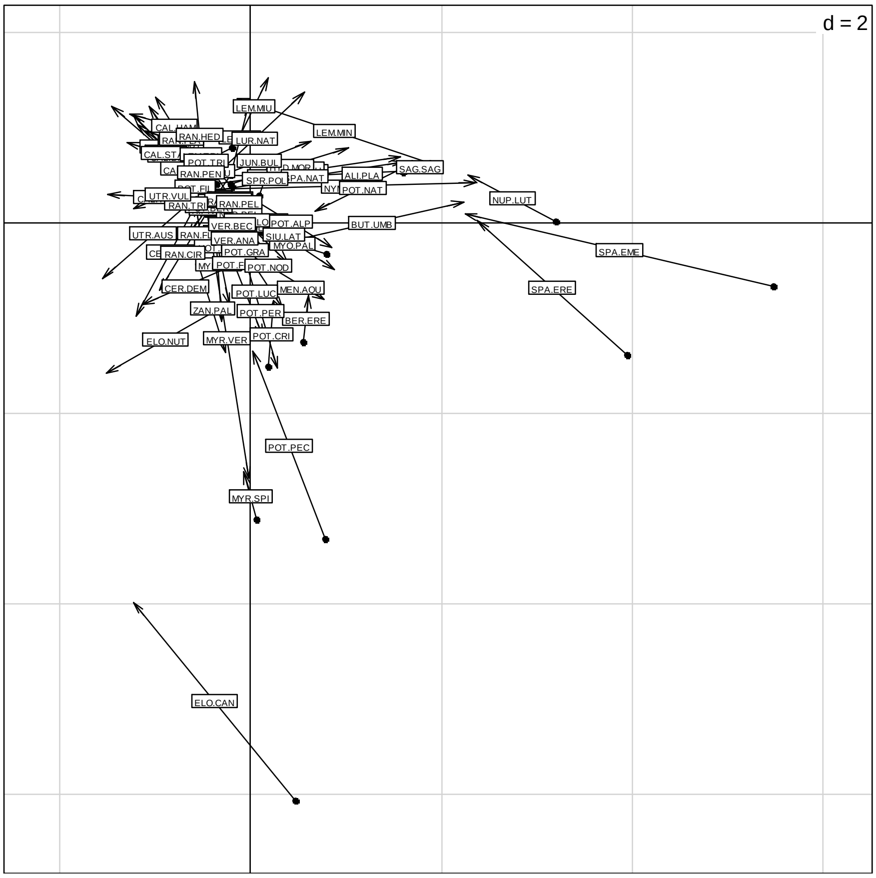
We observed that species groups 1-3 were generally more widely distributed in small than in middle-sized streams, whereas species group 4 and 5 were more widely distributed in middle-sized streams. In contrast, the abundance of all species groups was either higher (group 1, 2 and 4) or similar (group 3 and 5) in small and middle-sized streams (Supplementary Figure. 5).

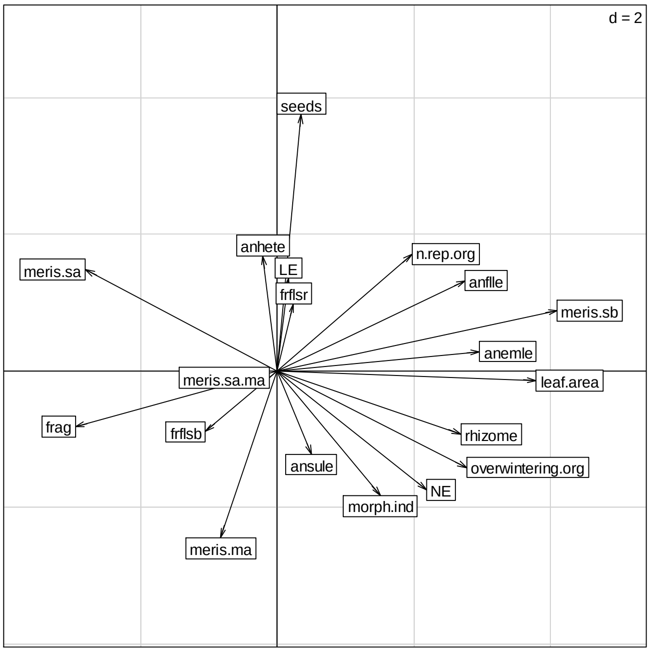
We found that community trait characteristics differed significantly between small (RC1) and middle-sized (RC4) lowland streams (Adonis: Permutational Multivariate Analysis of Variance Using Distance Matrices; $p<0.001$, Figure 4.8 main macrophyte text). Furthermore, we observed that trait characteristics changed significantly in response to hydromorphological degradation in small streams (Adonis: Permutational Multivariate Analysis of Variance Using Distance Matrices; $p<0.05$;). The ecological preference of the macrophyte community changed in modified streams with an increase in the abundance of productive species as inferred from increasing weighted averages of Ellenberg N. At the same time we observed an increase in the abundance of free-floating species, whereas the abundance of submerged and amphibious species with heterophyllus leaves declined. We also found that the abundance of species growing from a single basal meristem declined, whereas species with a high overwintering capacity increased in abundance in degraded streams.

Overall, we did not find that trait characteristics changed upon morphological alterations in middle-sized streams (Adonis: Permutational Multivariate Analysis of Variance Using Distance Matrices; $p=0.493$). It was possible, however, to identify specific traits that responded to hydromorphological degradation, and many of these traits were similar to those found to change in small streams (Figure 4.8 main macrophyte text). Thus, similarly to what we observed in small streams, the productivity level of the community increased, and there was also a change in dominant life forms with an increased abundance of free-floating and floating leaved species. We also observed that species growing from single basal meristems declined and that species with a high overwintering capacity increased, as observed also in small streams.

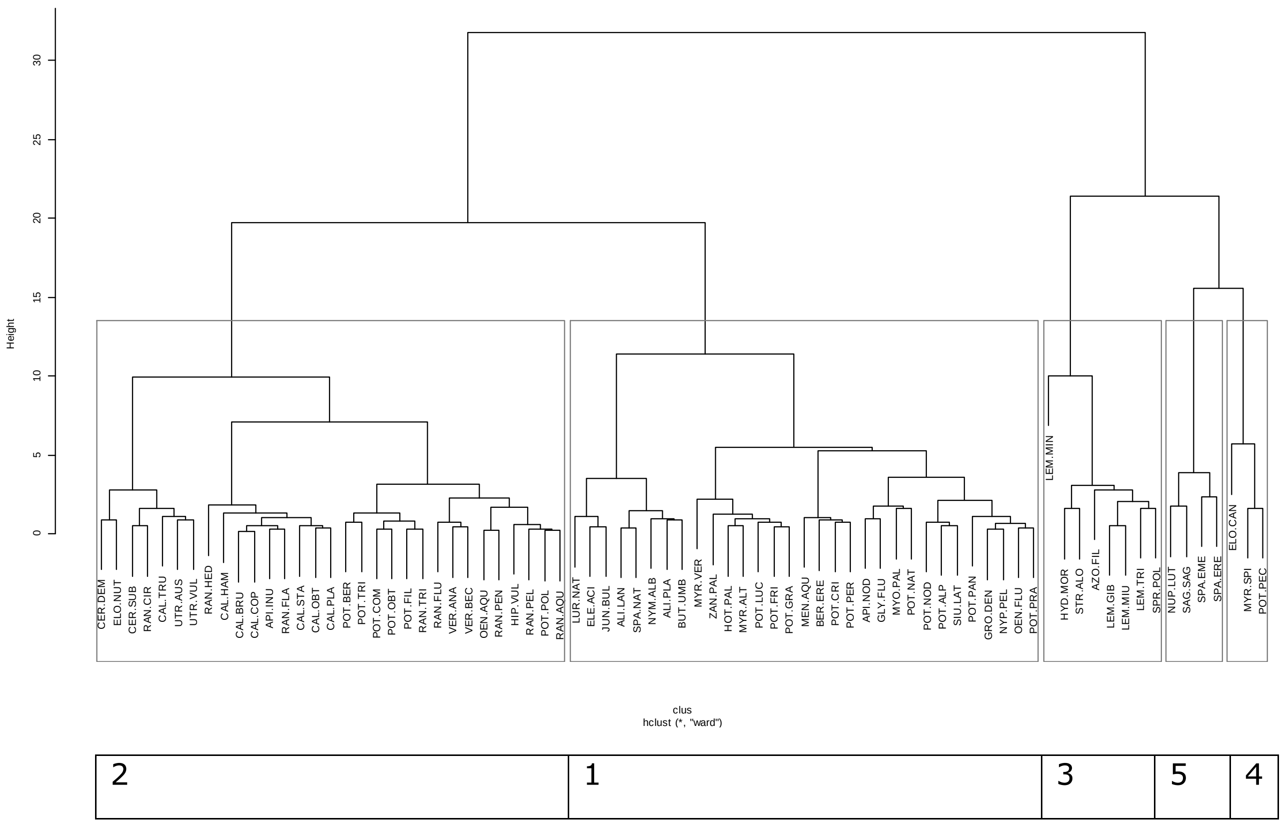


Supplementary Figure 1. Bi-plot of DCA axes 1 and 2 based on an analysis of a total of 772 stream sites and 77 species including submerged and amphibious species. The stream sites were all located in the central Baltic region in Europe and covered small and middle-sized reaches. The small stream sites all belonged to the RC1 stream type and the middle-sized streams all belonged to the RC4 stream type when categorised into stream types according to the stream typology used in the intercalibration process. Country codes are as follows: Germany (DE), Denmark (DK), Belgium, Flandern (FL), France (FR), Great Britain (UK, GB), Northern Ireland (UK, NI), Ireland (IE), Italy (IT), Lithuania (LT), Latvia (LV), Netherlands (NL), Poland (PL), and Belgium, Wallonia (BE, WL).

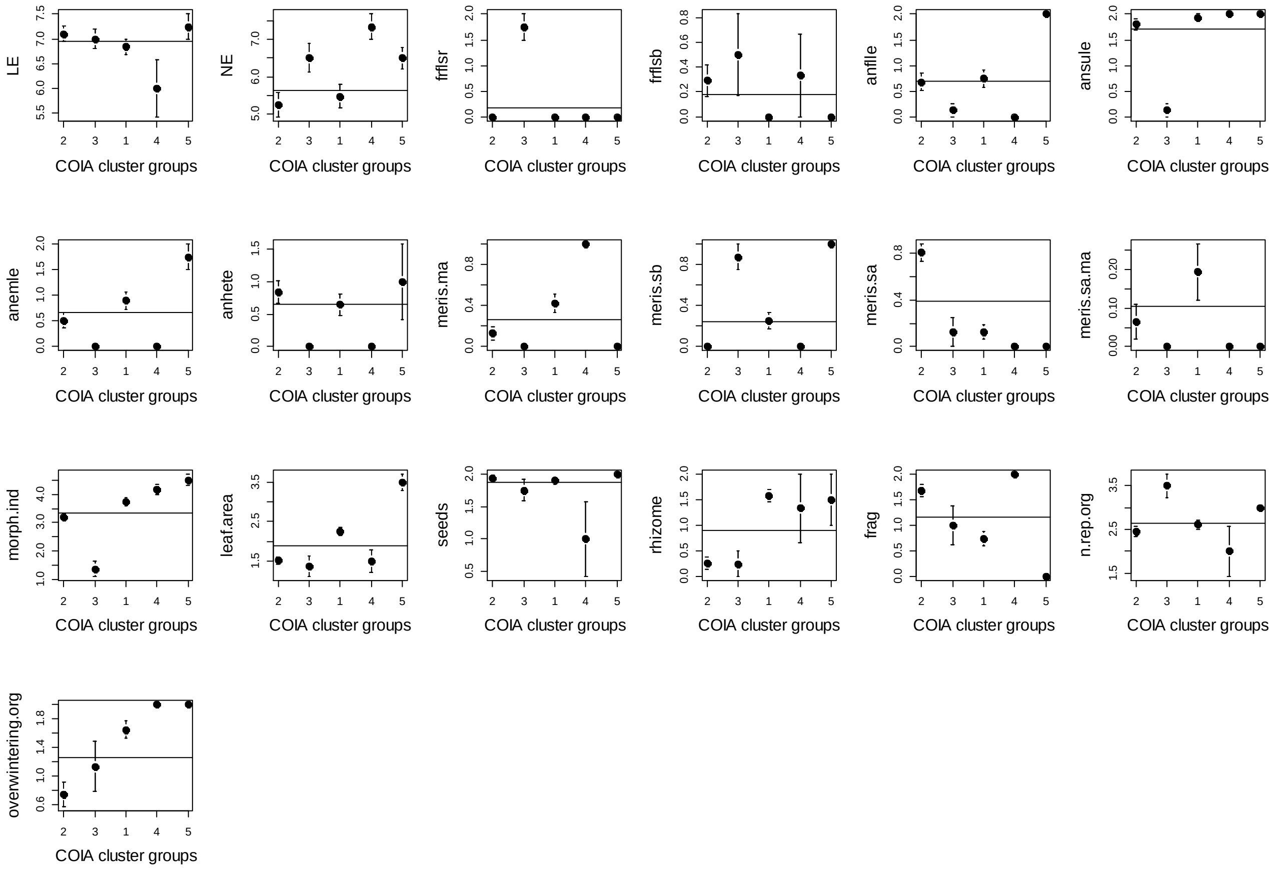




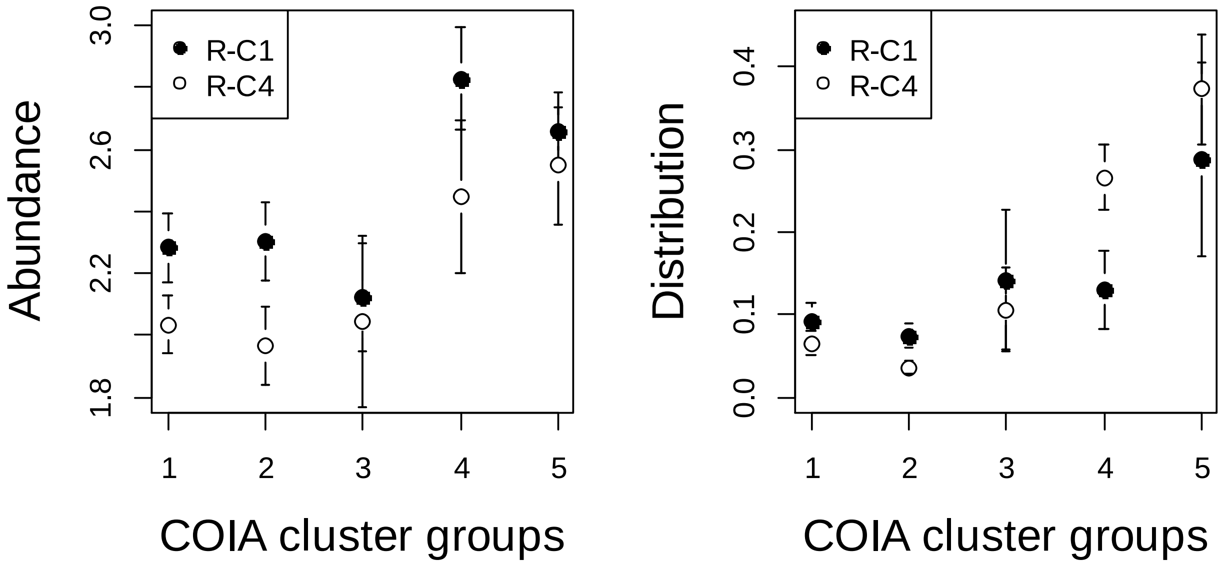
Supplementary Figure 2. Coinertia analysis (COIA) of scores obtained by principal component analyses of species traits (19 traits) and species abundances (77 macrophyte species). In total, 772 stream sites were included in the analysis, all of them located in the central Baltic region of Europe. The first plot shows how species associate with the first 2 axes of the COIA, where the origin of the arrow indicates the ordination of species according to their abundances (normed row scores from the COIA), and the end of the arrow indicates the ordination of species according to their traits (normed row scores from the COIA). The second plot shows how traits associate with the first 2 axes of the COIA (a projection of the canonical weights of species traits).



Supplementary Figure 3. Dendogram showing groups of species sharing similar distributional patterns and ecological traits identified by performing a hierarchical clustering analysis based on the Euclidian distance results obtained in the COIA. We cut the cluster dendogram at 5 groups of species.



Supplementary Figure 4. Plots showing mean trait values for species belonging to COIA cluster group 1-5. Error bars indicate standard errors and the horizontal line indicates the grand mean for all species groups.



Supplementary Figure. 5. Plots showing mean local abundance and distribution (range size) of species belonging to cluster groups 1-5 in R-C1 (black points) and R-C4 (white points). Error bars indicate standard error. Local abundance was calculated as the average abundance for each site where the species was present. Distribution (range size) was calculated as the sum of sites where the species was present divided by the total number of sites.

An analysis of UK national data

Aim

Specific hypotheses

1. There are broad associations between macrophyte assemblage structure and rivers differing in geomorphological type.
2. Sites identified as hydromorphological degraded are physically distinct from sites which are not degraded.
3. Hydromorphological degradation in terms of an altered channel morphology alters macrophyte trait characteristics or species composition.

Methods

UK dataset

A total of 467 sites were included in this dataset. The macrophyte abundance and site physical and chemical data were originally collected during surveys carried out by the Centre for Ecology and Hydrology and the Environment Agency of England and Wales (EA) using the "Mean Trophic Rank" (MTR) macrophyte survey method (Dawson et al. 1999a).These data and associated physical parameters are described in Gurnell et al 2010.

Trait data

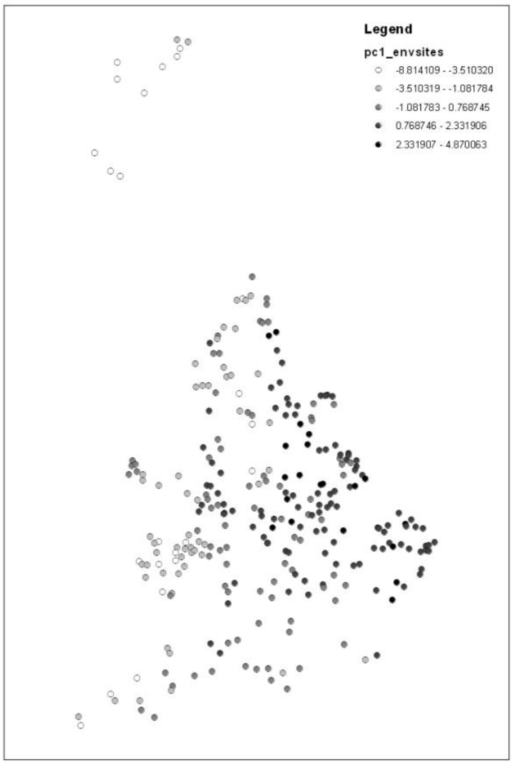
The PLANTATT trait dataset was used with the UK data. Weighted averages were used to create site values. For the numerical categories, we took the median and then multipled it by the abundance For the nominal categories we used fuzzy coding where each level within a trait was turned into a binary form.

Analyses

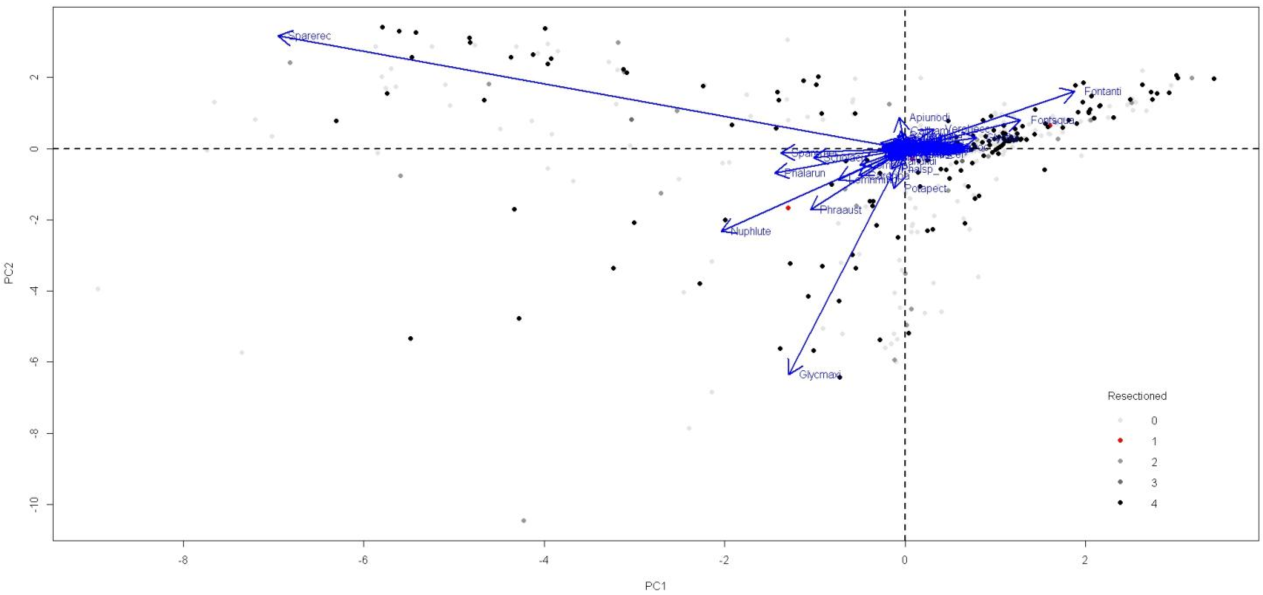
A complementary set of multivariate statistical approaches was used to examine the relationships between macrophytes and hydromorphological degradation. A PCA was created using physical parameters to test hypothesis 1, further PCAs were used to describe trait distribution and test hypotheses 2 and 3. All analyses were carried out in R.

Results

The PCA analyses based on physical parameters alone or in combination with water chemistry variables did not separate sites with different levels of resectioning, ranging from sites with no resectioning to those with both banks resectioned. Resectioning referes to the reprofiling of the river cross section to a standard trapezoid form, Figure 4.. There was however some indication that upland sites, to the left of the PCA diagram, were rarely resectionined, while lowland sites, to the right of the diagram, could have a wide variety of resectioning irrespective of physical character, as measured using the available physical parameters. A map of sites, with sites colour coded by axis 1 PCA score indicates a strong west to east gradient in the data, suggestion strong spatial auto –correlation, (Supplementary Figure 6). Neither species based distribution or plant trait distribution distinguished between sites with differing degrees of resectioning, (Supplementary Figure 7).



Supplementary Figure 6. Spatial pattern of channel resectioning in the UK sites colour coded based on PCA axis one scores in Figure 4.**, main macrophyte text. .**



Supplementary Figure 7. A PCA of sites ordinated by species and a sister PCA plot of plant traits can be found in Figure 4.9 **in the main macrophyte text.**

An analysis of Northrhine-Westfalia (Germany) data

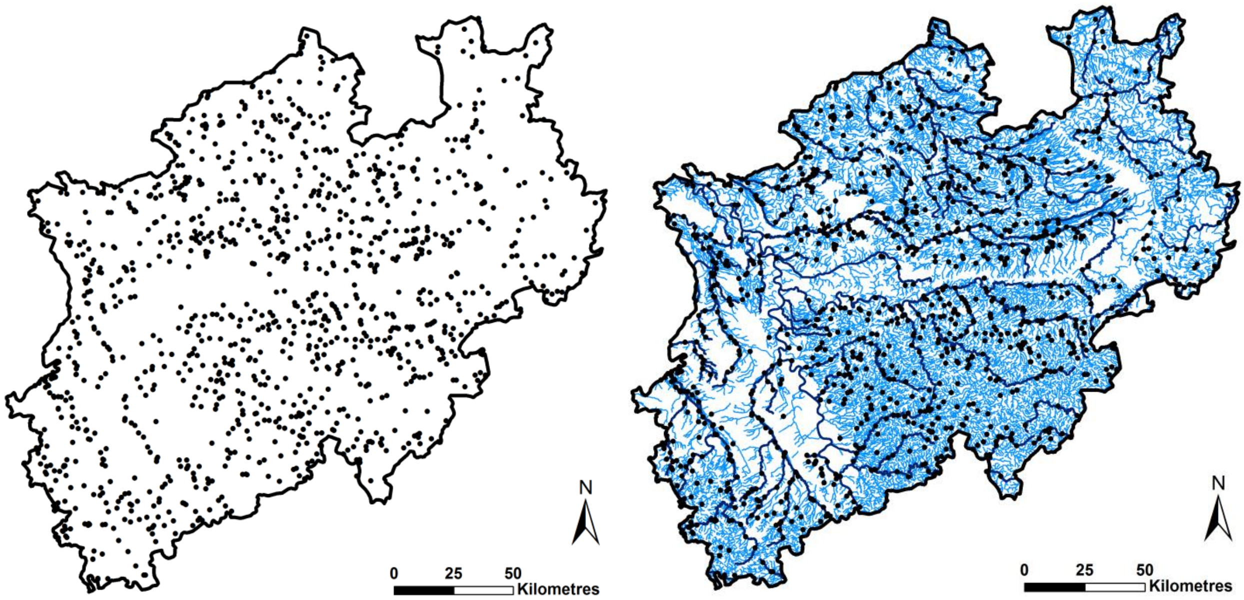
Site selection and descriptions

North Rhine-Westfalia is one of 16 federal states in Germany covering an area of 34,097 km^2. The population is about 17.5 million people. About half of the area is covered by lower mountainous areas (the highest mountain is 834 m a.s.l.), while the other half belongs to the Central European plains. The River Rhine crosses the western part of Northrhine-Westfalia from the south to the north-west. Two large rivers (Ruhr, Lippe) discharge into the River Rhine, while other rivers belong to the catchments of the Ijssel, Weser, Ems and Meuse.

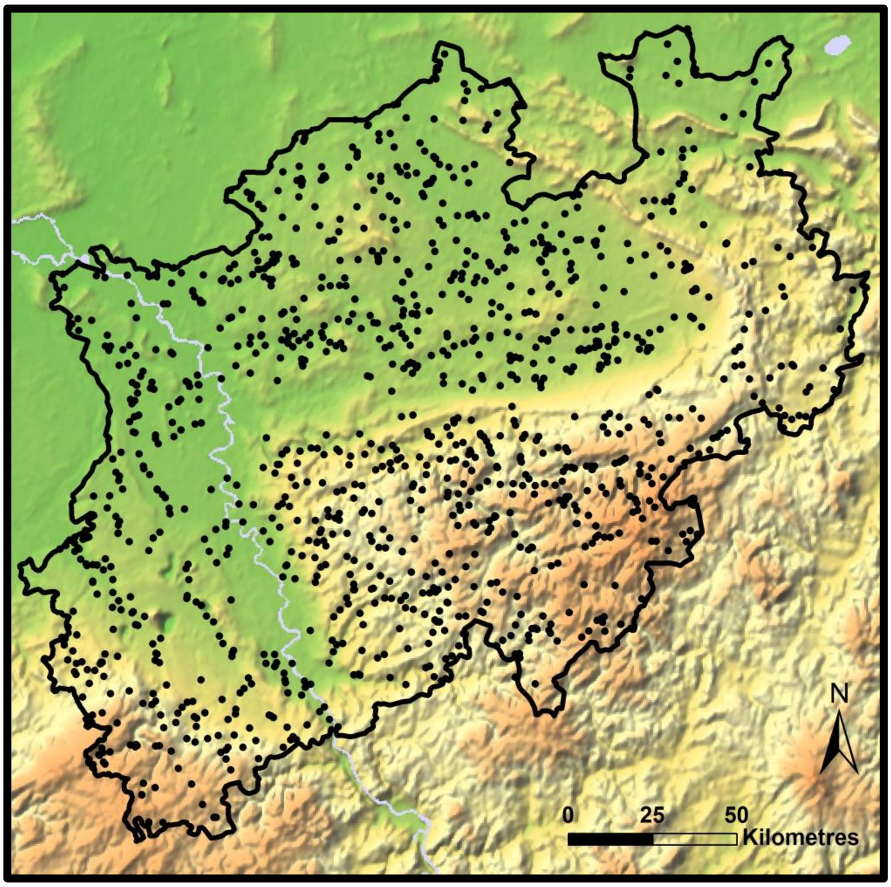
Since the introduction of the EU Water Framework Directive standardised sampling of aquatic organism groups has been undertaken. Besides fish and invertebrates, a large number of standardised macrophyte samplings have been conducted.

The location of sampling sites for monitoring activities according to the legislative requirements was decided by the Landesamt für Natur, Umwelt und Verbraucherschutz (state agency for nature, environment and consumer protection) and the water boards of the individual catchments. In total, 1,206 sampling sites were appointed. These are distributed over the whole federal state (Supplementary **Figure** 8a and b) and cover the whole altitudinal gradient (10), ranging from small gravel-bed mountain brooks to large sand-bottom lowland rivers.

For our analyses, we selected the sites for which the three parameters altitude, catchment size and slope of the river bottom were available as abiotic factors. Thus, in total we selected 1,136 sampling sites.



Supplementary Figure 8. Distribution of sampling sites in Northrhine-Westfalia (a) and within the river network (b).



Supplementary Figure 9. Distribution of the sampling sites with reference to the topography.

Macrophyte sampling and data

Macrophyte sampling was conducted according to the German standard method (Schaumburg et al. 2005a,b). Here, a 100-m reach was surveyed for macrophytes by zigzagging through the river and walking along the riverbank in the summer months at low flow conditions. In non-wadeable areas, the river bottom was raked with a rake (on a long pole or at the end of a rope) to reach the macrophytes. All macrophyte species were recorded and identified to species level. The surveys included all submerged, free-floating, amphibious and emergent angiosperms, liverworts and mosses. The abundance of each species was recorded according to the 5-point scale devised by Kohler (1978): 1 = very rare, 2 = rare, 3 = common, 4 = frequent, 5 = abundant, predominant. Furthermore, the growth form for each species was recorded according to Den Hartog & Van der Velde (1988) and Wiegleb (1991). The growth forms comprise different plant species that realized the same or comparable phenotypical adaptations to the aquatic environment.

In total, we analysed 1,136 samples, i.e. one sample for each of the 1,136 sites. 204 different macrophytes species were detected in the samples and associated to 16 different growth forms.

Data analysis and presentation

We conducted a principle component analysis for all sampling sites with the three abiotic factors: altitude, catchment size and slope of the river bottom. This abiotic space served as a template for the distribution of the growth forms at the sampling sites.

For each sample, the number of species associated with each growth form was calculated, and the abundances of each growth form were summed up. The number of species was classified as: 0 if no species were found, 1 if one species was found and 2 if more than one species was found. The abundances in each growth form and sample were classified into the following four groups: 0 = not present, 1 = rare (abundance sum of 1 and 2), 2 = common (abundance sum of 3 and 4) and 3 = abundant (abundance sum of more than 4).

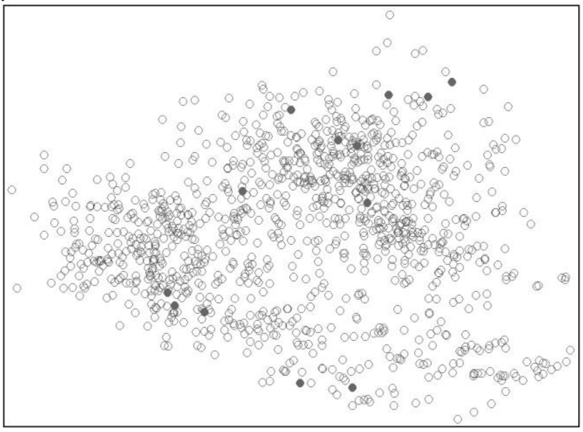
The resulting classes of species numbers and abundances for each growth form for each sample were used as overlays for the sampling sites in the PCA space.

Results

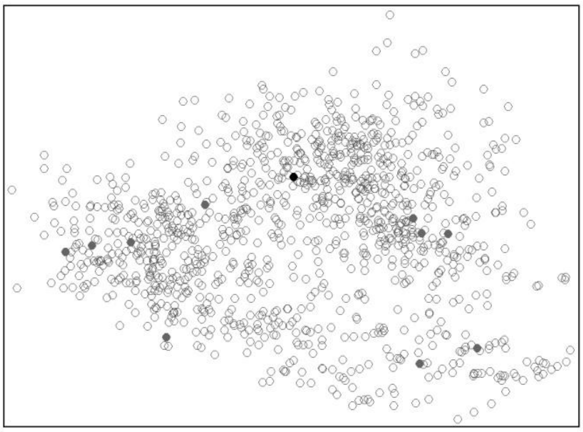
The PCA ordination displays a clear separation of the sites in the space of catchment size, altitude and slope of the river bottom, Figure main macrophyte text . Lowland and mountain sites differentiate as well as small and large river sites.

The number of species within the growth forms show clear patterns comparing the individual growth forms and in the space of the three abiotic factors (Supplementary **Figure**10). While riparian growth forms such as Equisetids, >Juncids and Helophytes are scattered over the whole PCA gradient, the more aquatic growth forms display specific distributions with regard to certain abiotic factors and thus river types. Ceratophyllids, Charids and Lemnids are predominantly found at low altitude, low slope rivers of larger catchment sizes, while the number of moss species (Haptophytes) is particular high in small streams of high altitude and slope. Peplid species (mainly the genus *Callitriche*) are found at nearly every site of low altitude and relatively low slope inrespective of catchment size. Elodeids, Parvopotamids, Valliserids and Nymphaeids are growth forms in which the species numbers are particular high in the larger lowland rivers with low slope, while higher species numbers of Myriophyllids mainly occur in larger mountain rivers of medium slope. Magnopotamids on the other hand are found in lowland rivers of medium size and slope.

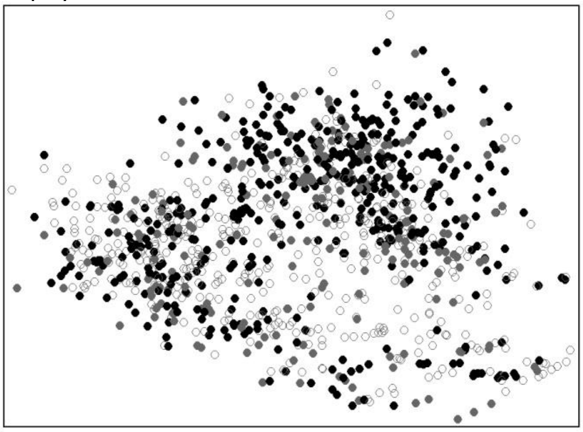
Equisetids



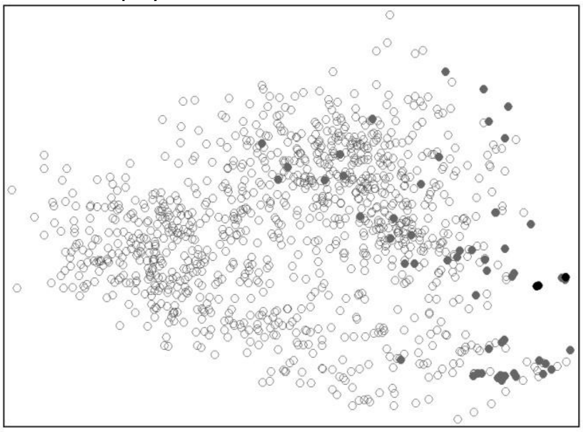
Juncids



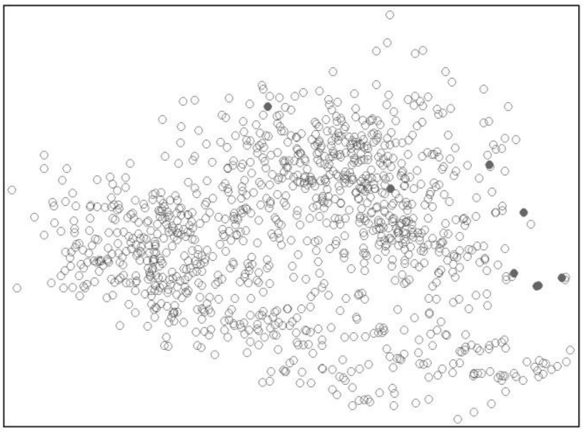
Helophytes



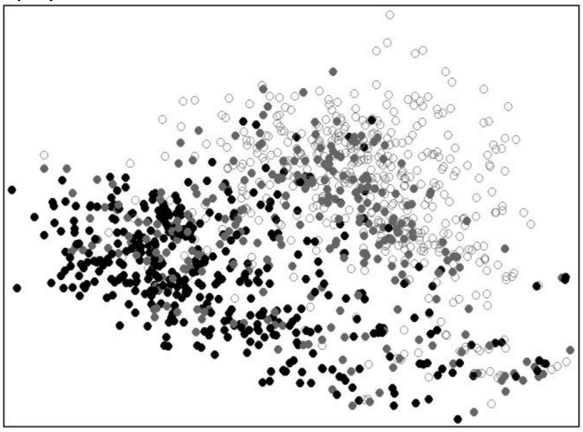
Ceratophyllids



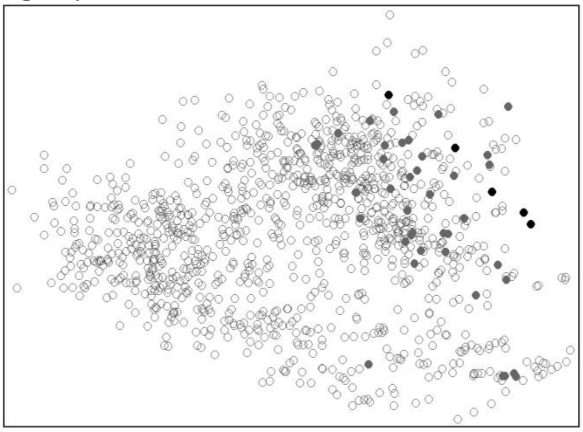
Charids



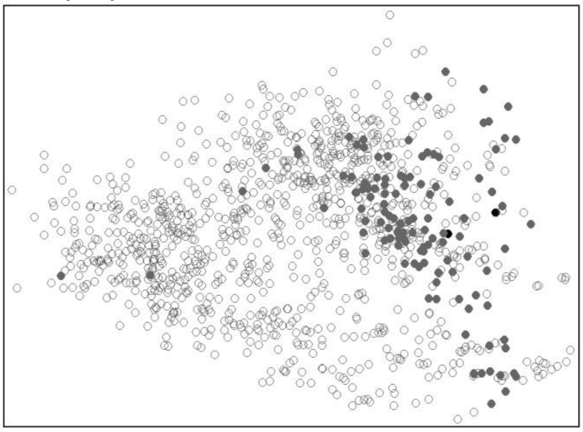
Haptophytes



Magnopotamids



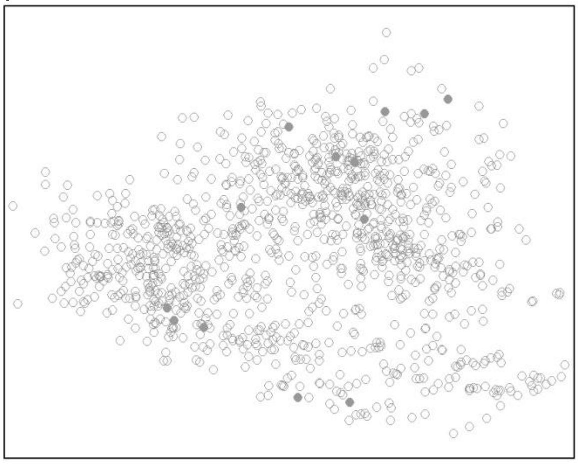
Nymphaeids



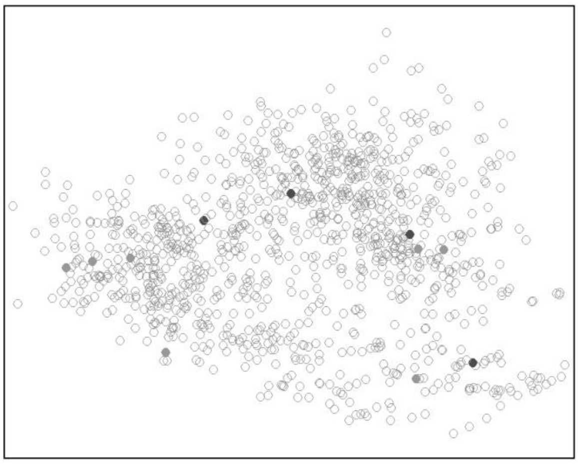
Supplementary Figure 10. Number of species in each growth form and sample. Blank dots = no species found, grey dots = 1 species found, black dots = 2 and more species found. Additional growth form diagrams are presented in Figure **of the main macrophyte text.**

The results for the abundance classes within each growth form and sample resembled mainly the results of the species numbers (Supplementary Figure). Exceptions were the results for Elodeids and Nymphaeids which showed more pronounced patterns with higher abundances in the mid-sized lowland rivers of medium slope. Furthermore, dominant abundances of Myriophyllids were found in mountain rivers of medium slope and size, and Peplids showed their highest abundances in low gradient, medium-sized lowland rivers.

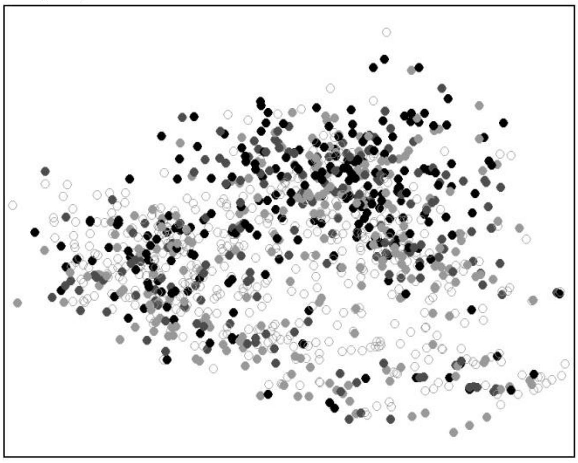
Equisetids



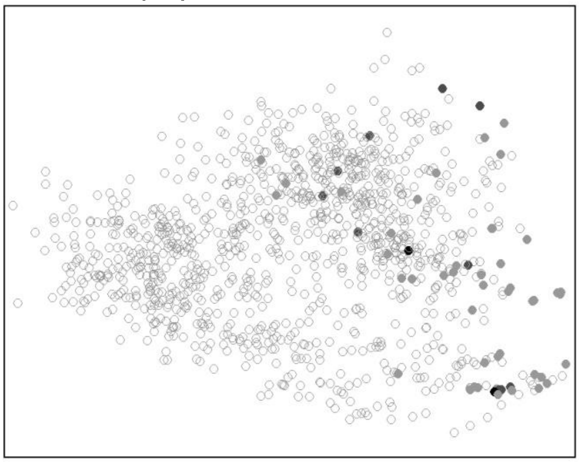
Juncids



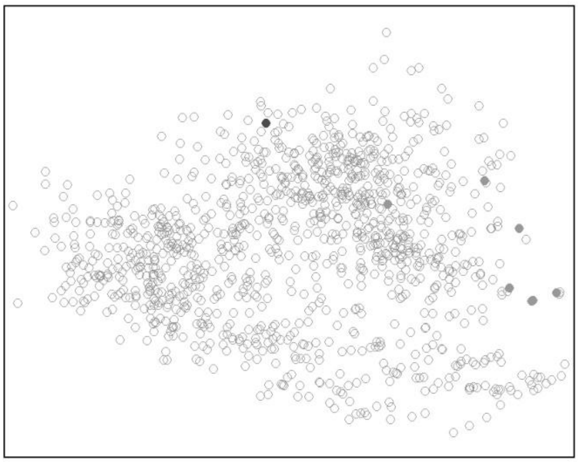
Helophytes



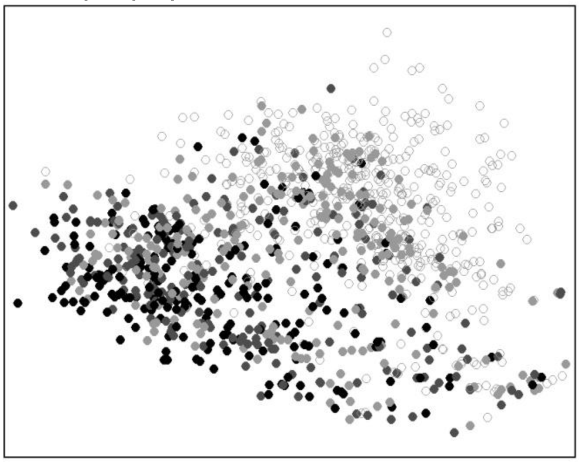
Ceratophyllids



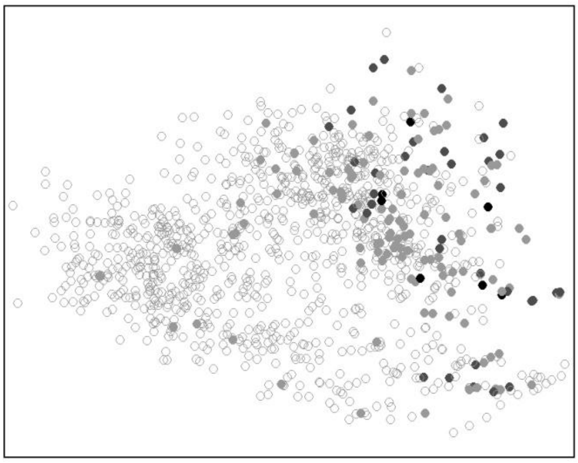
Charids



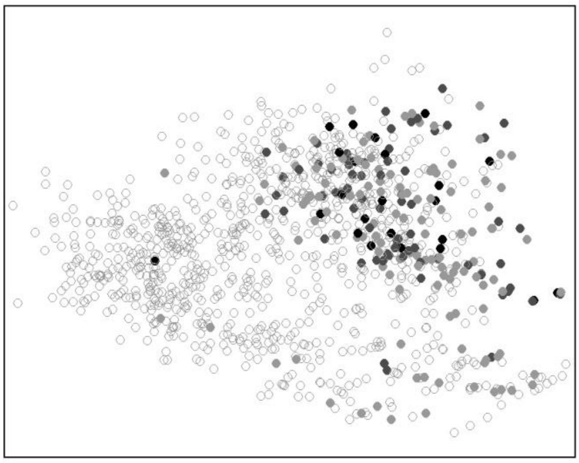
Haptophytes

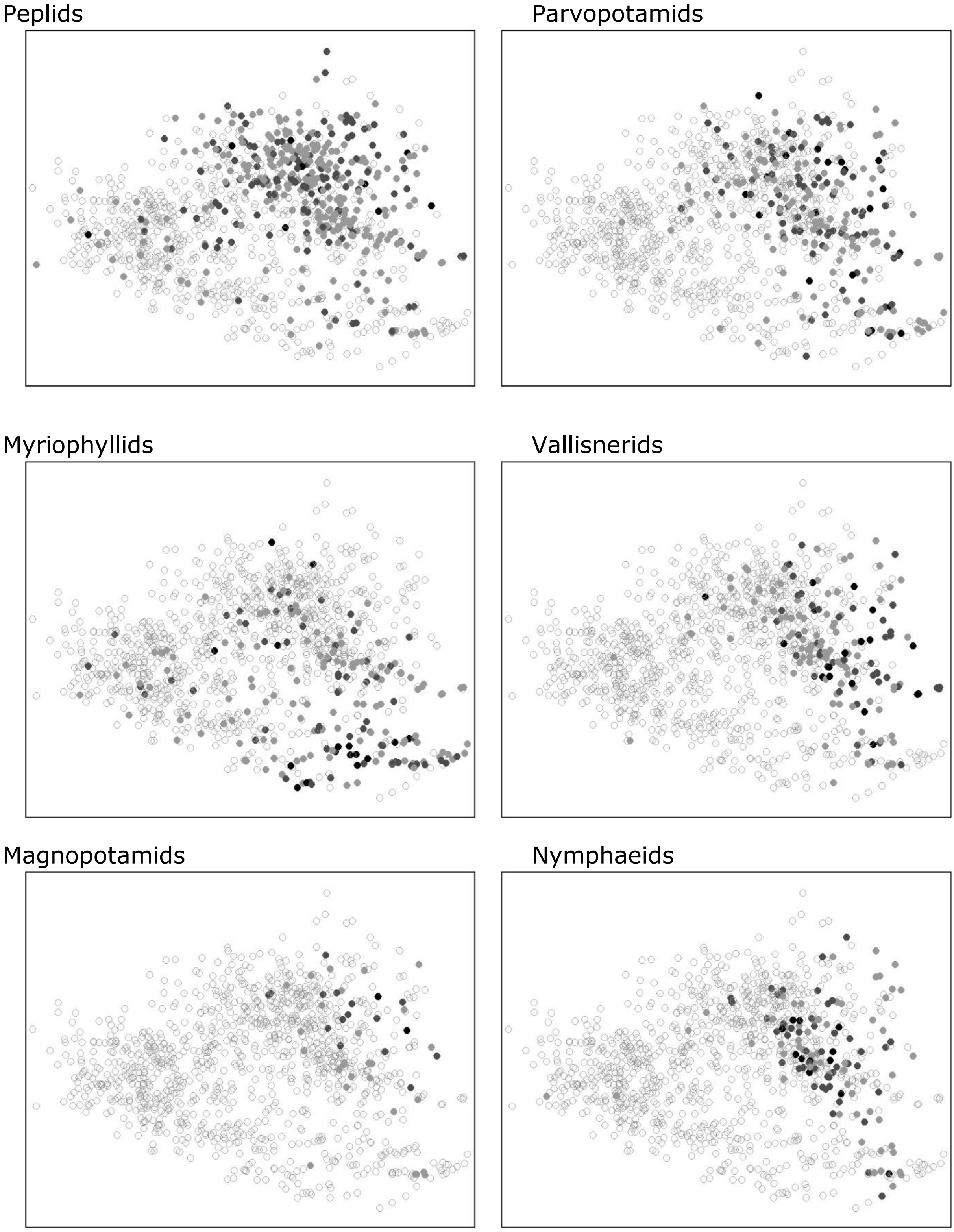


Lemnids



Elodeids





Supplementary Figure 11. Classified abundances for each growth form and sample. Blank dots = not present, light grey dots = rare (abundance sum of 1 and 2), dark grey dots = common (abundance sum of 3 and 4), black dots = abundant (abundance sum of more than 4).

A review of the impacts of hydromorphological pressures on potential macrophyte metrics

Dams/reservoirs/stream regulation

Parameter	Modification	Effect on Plant Quantity	Effect on Plant Quality	Evidence
Discharge	Flows more regulated under some management schemes, e.g. basic compensation flows from water supply reservoirs Typically water depth will be lower.	Increased standing crop	Promotes growth of filamentous algae and in-stream vegetation.	Lowe, 1979
			In semi-arid areas riparian vegetation may encroach towards mid-channel and	Dolores Bejarano et al., 2011; Dolores Bejarano & Ward, 2011
		Riparian communities have a lower density and reduced cover	Riparian plant communities have lower species richness	Nilsson et al., 1991; Nilsson et al., 1997; Jansson et al., 2000
Discharge	Widely fluctuating flows - power and irrigation dams	Decreased standing crop of in-channel vegetation.	Lower species richness.	Jansson et al., 2000;
		Increased cover and density of woody riparian vegetation	Close to dams a flora typical of intermittent rivers,	Bernez & Ferreira, 2007; Garafano-Gomez et al., 2013; Catford et al., 2011; Greet et al., 2013
Turbidity	Decreased by reservoirs, often relatively low in regulated streams improving light penetration	Increased standing crop	Favours submerged vegetation	Lowe 1979
Sediment	Sediment supply reduced leading to channel erosion	Decreased standing crop	Loss or change submerged species	Lowe, 1979
		Riparian herbaceous cover lowered	Riparian herbaceous species richness lowered	Beauchamp & Stromberg, 2008
Water Temperature	Warmer winter, often ice free, cooler summer	Increased standing crop	Favours cool-water stenotherms and green algae	

Channelization

Parameter	Modification	Effect on Plant Quantity	Effect on Plant Quality	Evidence
Sediment	Increased sedimentation during channelization construction	Reduced biomass if sufficient depth of sediment to cause smothering of plants species downstream of channelization works	diversity reduced, particularly submerged plants. Emergent plants may be less affected	Brookes, 1986
Channel geometry	Cross section altered to trapezoidal form – bank angle steepened	Reduced biomass of marginal species due to very limited available habitat	it may permit larger migration of species from banks into streams and increase species diversity in channel	Pedersen et al., 2006
Channel geometry	Enlarged (deepened/widened) channel + embankments – Initial homogenisation of substrate and loss of in-stream habitats. followed by planform adjustment	Total channel macrophyte coverage similar. Reduced biomass due to simplification of in-stream habitat	Reduced species richness and diversity in channel – species associated with finer substrate + shift to more opportunistic species	Baattrup-Pedersen & Riis, 1999; Brookes, 1995; Rambaud et al., 2009
Channel geometry	Straightening – steeper bed slope resulting in increased stream power (but only a little)	Variable Reduced biomass as stream power increases	Variable Shift to species more tolerant of higher stream power Species richness and composition similar to less impacted Iberian Mediterranean-type river sites	Aguiar et al., 2001
Channel geometry	Loss of floodplain connectivity	Riparian species less abundant along channelized stream	riparian species richness lowered along channelized	Baatrup-Pedersen et al., 2005

Weed cutting

Parameter	Modification	Effect on Plant Quantity	Effect on Plant Quality	Evidence
Hydraulic flow resistance	Reduced by regular cutting of submerged plant biomass (above sediment) across whole channel	Submerged plant biomass decreases but plant cover remains similar	Species richness, diversity and patch complexity decreases. Favours plant species with more ruderal traits, i.e. rapid growth and high dispersal capacities	Baatrup-Pedersen et al., 2002; Baatrup-Pedersen et al., 2003; Westlake & Dawson, 1986
Hydraulic flow resistance	Reduced by irregular and variable cutting (e.g. whole channel or part of channel) of submerged plant biomass (above sediment)	Submerged plant biomass decreases but plant cover remains similar	No changes in species diversity with different cutting regimes but directional changes towards dominance by species more tolerant of disturbance, e.g. *Ranunculus,* with increases in cutting frequency of whole channel or focused cutting on one part of channel	Baatrup-Pedersen & Riis, 2004; Sabbatini & Murphy, 1996; Westlake & Dawson, 1982;

Dredging/sediment removal

Parameter	Modification	Effect on Plant Quantity	Effect on Plant Quality	Evidence
Hydraulic flow resistance/silt accumulation	Reduced by removal of both above and below sediment submerged plant biomass	Submerged plant biomass wholly removed	Plant community in channel destroyed. Re-establishment dependent on plant fragments left behind and/or re-colonisation by plant propagules -favours dominance of plants with disturbance tolerant traits, e.g. floating species	Sabbatini & Murphy, 1996; Wade, 1993; Wade & Edwards, 1980; van Zuidam et al., 2012