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REFORM

REstoring rivers FOR effective catchment Management

Deliverable: D3.2

Title: **Understanding biological responses to degraded hydromorphology and multiple stresses**

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Dissemination Level

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Summary

- The aim of this deliverable is to conceptually model and empirically test the response of biota to the effects of both hydromorphological pressures acting in concert with one another or with other types of pressures.

- Best use is made of existing large national monitoring datasets (Denmark, UK, Finland, France, Germany, Austria & WISER datasets), case studies and modeling to provide evidence of multiple stressors interacting to alter river biota (Biological Quality Elements: BQE).

- In-stream river plants. The evidence from analyzing plant traits, from UK, German and Danish data is that macrophytes can indicate hydromorphological degradation. One can also gain insights into how different hydromorphology and other stressors are interacting. Channelised sites are not only physically altered but require ongoing maintenance in the form of vegetation management (riparian and in-stream) and dredging which impact on macrophyte traits. The observed interactions between eutrophication and different hydromorphological pressures are explored. There was also evidence that macrophytes may have a role in accumulating/retaining heavy metals in polluted rivers; a fact which requires consideration during the physical restoration of such systems.

- Fish. The sensitivity of species with different physical habitat affinities is considered. The response of fish to over 100 years of hydromorphological degradation at three Austrian case study systems is described. Here the complex nature of hydromorphological impacts on hydromorphological processes is emphasised. In a second study the possibility of creating models which link pressure to processes to fish occurrence is explored. As a pressure may affect more than one hydromorphological process it is important to understand and clear define how pressures interact on physical processes at scales that fish respond to.

- Invertebrates. Existing invertebrate metrics are problematic. To explore alternatives we aimed to elucidate general patterns between the assessment of habitats/biotopes and the diversity of macroinvertebrates using a standardised biomonitoring sampling methodology. We use a high quality dataset that includes a pre-defined gradient in hydromorphological degradation. In addition to identity-based diversity metrics and traits were used. There was some evidence that traits held some potential but sampling methods also need to be revised.

- Joint (BQE) Biota Analysis. Analysis of a very large European data set covering Finland, Denmark and France examined the relative sensitivity of macrophytes, diatoms and invertebrates to nutrient and hydromorphological stresses. Analysis at the community level suggested that nutrient impacts were more detectable than hydromorphological stressors. This finding is not surprising for Finland where hydromorphological degradation is rather low. Sites subject to multiple-stresses tended to have the most common species. There are differences in sensitivity between BQEs at community level; primary producers appear more sensitive to diffuse pollution than invertebrates. The challenges in using monitoring data to identify known causal interactions between biota and multiple stressors are discussed.

- Weir removal. This intervention is now widely advocated across Europe as a means of improving ecological status. The benefits of weir removal are obvious, for example its

removal facilitates upstream migration of anadromous fish; but in multi-stressor environments there may also be unforeseen disadvantages. The removal of a weir from a British weir is modeled and checked against empirical data. The results indicate that the weir creates conditions for denitrification and its removal increases the load of N exported downstream. The cumulative impact of widespread weir removal should be considered carefully in terms of the gross export of rivers to coastal areas where catchments suffer from eutrophication.

- Stressor interactions. A clear conclusion from the evidence available across all biotic groups is that irrespective of whether or not hydromorphological stressors and other forms of stress interact synergistically or antagonistically to alter natural river biota assemblages, each stressor on its own can and do have detrimental impacts. Therefore when remediating damaged sites it is important to understand that while it may be possible to improve system status by tackling one stressor good ecological status is unlikely to be achieved without tackling all significant stressors in systems subject to multiple stress.

- Future Directions. Monitoring data are designed to detect change at individual sites through time and can miss some crucial hydromorphological impacts. Revision of some of the monitoring metholodologies can help, however adherence to monitoring data alone will not supply knowledge and system understanding. Common sense indicates an alternative approach for some WFD operational and investigative monitoring are well designed, replicated field ‘experiments’ elucidate cause-effect relationships on case study systems and are used to complement monitoring data, see WP4 for examples.

Acknowledgements

This document has been internally reviewed by Judy England (EA), María Isabel Berga Cano at CEDEX and Tom Buijse (Deltares).

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Table of Contents

1 Introduction

In this introductory chapter, Deliverable D3.2 is placed within the current context of the REFORM project and river management in Europe. The deliverable is the outcome of work carried out under Tasks 3.1 and 3.2 in WP3, which terminated in project month 36 (October 2014).

The main focus of the deliverable is to get a better understanding of the biological responses to degraded hydromorphology, sediment dynamics and multiple stresses. Within the constraints placed by available data and knowledge, it reports on the mechanistic understanding of biological response to hydromorphological degradation, and how hydromorphological stress can be singled out, or otherwise, in a multi-pressure environment.

The over arching aim of the deliverable is to:

- Conceptually model and empirically test the response of biota to the effects of both hydromorphological pressures acting in concert with one another or with other types of pressures.

This deliverable follows on from D3.1 which used national and pan-European monitoring datasets to investigate the impact of hydromorphological degradation alone on ecological status. D3.1 provides the context for this report. It provided evidence that metrics indicating HYMO impact could be developed from monitoring data on fish and macrophytes; that there was the potential to derive metrics sensitive to fine sediment and evidence that phytobenthos (diatoms), invertebrates and macrophytes have the potential to be used in combined metrics. However it was also found that existing macroinvertebrate metrics lack specificity and can provide false positive responses to HYMO pressure, suggesting that clear conceptually modelling and empirical testing of multi-stressor responses is critical to good diagnosis.

A second sister Deliverable, D3.3, also based on work in Tasks 3.2 and 3.3, is reported in project month 36. It is recommended that the two deliverables be considered together. D3.3 evaluates candidate indicators for case studies including uncertainty. It reports on best indicators of hydromorphological stress, directly tested in case studies, and elucidating their relative precision in separating ecological class boundaries through analysis of uncertainty. There is a strong focus on experimental testing. In addition deliverable D3.4 addresses riparian processes at degraded sites and should also be read in conjunction with this deliverable. It highlights the complex but well described interactions between physical and ecological processes over time. The dramatic response of riparian zones, questions the restricted focus on in-stream processes inherent in the Water Framework Directive. D3.4 is due out in March 2015.

Why do we need to understand the biotic response in rivers to multiple pressures?

The first reports to the EU by member states on Water Framework Directive status indicate hydromorphological degradation is as widespread and important an issue as nutrient pollution (European Environment Agency 2012). The WFD demands remedial action and this must encompass both hydromorphological and water quality degradation if good status is to be achieved.

In D3.1 we previously reported the timeliness of this work. At an end-user conference held in Brussels in February 2013, held as part the REFORM project dissemination activities, 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. In the first phase of implementing the WFD, monitoring systems were put in place to determine the quality of rivers based on their biota. Now in the second phase, sub-standard rivers are subject to remediation.

Previous research by project members has indicated that many rivers across Europe are subject to multiple pressures (Figure 1.1). To create remedies for these rivers requires diagnostic tools. We have previously used the following analogy; river managers are like doctors, they need tools for diagnosing how ill their patient (river) is and how best to treat them cost effectively.

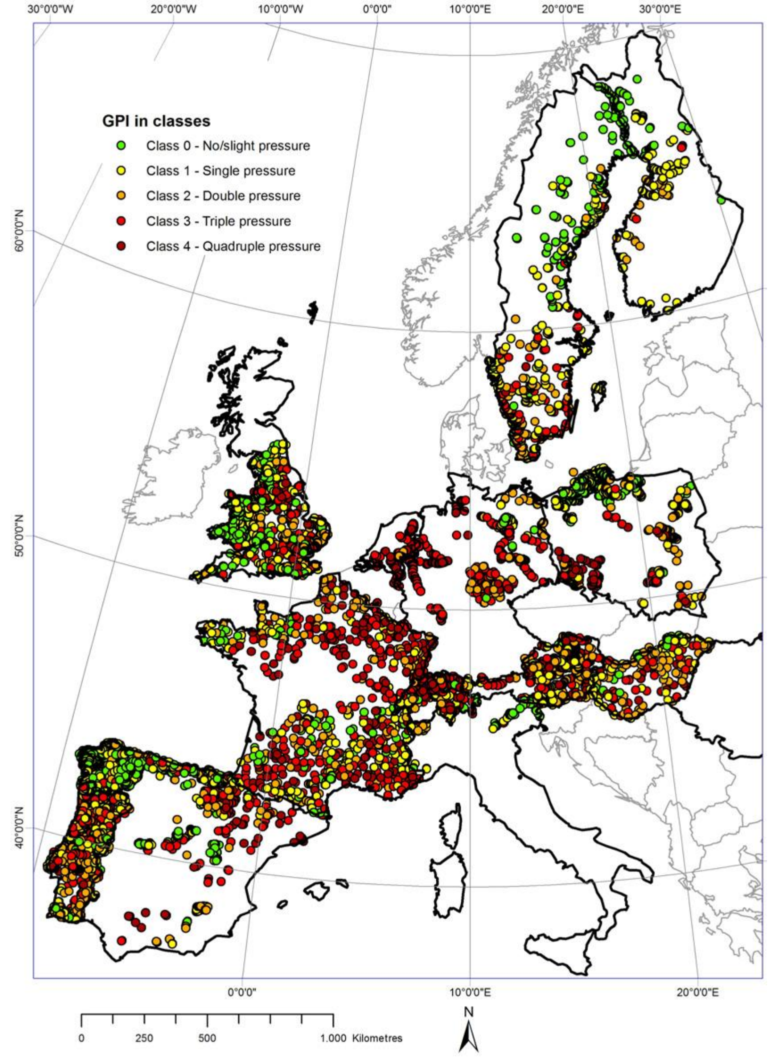

Figure 1.1 The occurrence of multiple pressures/ stressors across European waters, after Schinegger et al 2012.

How do we go about understanding the biotic response in rivers to multiple pressures? The REFORM context

The data available to the project allowed for analysis of hydromorphological issues related to channelisation in various forms and the associated maintenance of channels, weed cutting etc. Water quality was also well represented in these datasets and it was possible to examine its interaction with hydromorphological degradation.

Within this deliverable we build on the findings of the Deliverables from WP1 which review the pre-existing understanding of hydromorphological pressure - impact - responses from the literature. The most direct advance is in the chapter on fish responses where the conceptual models developed in WP1 are refined for individual species. This work is supplemented by a chapter on fish responses in a single system over the sequential and cumulative impact of hydromorphological impact.

The main analysis techniques used here were the ordination of large multivariate datasets and close examination and modeling of key case studies.

1.1 Over arching findings

A key cross cutting finding is that it is important to understand how pressures affect fluvial geomorphological and hydrological processes. To use a well understood process as an analogy; nutrient pollution can alter biota but has little direct effect on other aspects of water chemistry, in contrast pressures on hydromorphology alter hydromorphological processes which then in turn act as stressors on biota (Garcia de Jálon et al. 2013). The effect of these pressures is mediated by process.

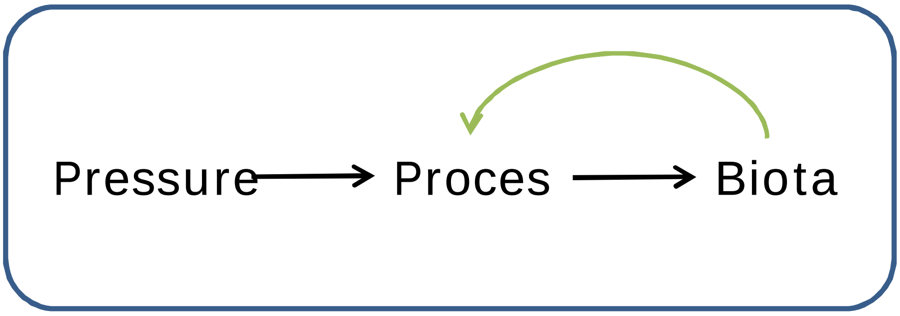

Figure 1.2, The fundamental links in biota sensitivity to hydromorphological pressures. Note biota (vegetation) can cause feedbacks by influencing processes.

For some taxa there were encouraging signs that multi-stressor processes could be understood. Vegetation showed some clear responses and reflects the strong background literature and research on the topic. Subtle shifts in trait responses of taxa can be used to understand how stressors are interacting at site and may help in understanding why the assemblages of some sites may be improved. The linkages between stressors and biotic response described here should act as useful examples for managers, to illustrate the potential impact of any new developments to their systems.

Repeatedly, across different biological groups, we discovered that either the biotic data or the supporting environmental data were not collected in a manner which made it straight forward to elucidate the linkages in Figure 2.2. This raises two distinct issues; how should monitoring be altered to increase sensitivity to hydromorphological degradation and secondly what data might be required to provide a clear understanding of fundamental linkages described in figure 2.2. A cost effective solution would be to carry out focused studies on specific key questions. It should be noted that hydromorphology is not a single pressure, as eutrophication is, it is a number of different pressures each requiring understanding. An undertaking comparable in scope to Vollenweider's classic work on lake eutrophication would be required for each major hydromorphological pressure (Vollenweider 1968).

Simple solutions to hydromorphological degradation which are currently in vogue need to be considered in the context of hydromorphological processes and multiple stresses. The chapter on weir removal cautions about the increased export of nitrogen to estuaries if weirs are removed while work on macrophytes indicates simply applying riparian tree cover can have negative impacts on desirable macrophytes.

1.2 References

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2 Macrophytes - synopsis of stress impacts from multiple hydromorphological stressors and eutrophication

Author: Matthew T O'Hare

This synopsis gives a brief impression of the three detailed chapters which analyse multiple stresses in Denmark, Germany and the UK. Each chapter contains a detailed analysis of monitoring data collected at large numbers of sites. As each country differs in the detail of the multi-stressor interactions, quality of data etc, it made most sense to analyse each national dataset individually using national experts.

A common approach was used whereby, for all countries, macrophyte traits were used to test for differences between sites subject to different stressor interactions.

In all cases the focus was on rivers which are altered in terms of hydromorphology and in many cases are subject to ongoing management to maintain their ability to convey flood waters. They suffer from eutrophication and often from modifications to riparian habitat associated with anthropogenic changes, intensive land use etc. Some of these systems have altered hydrology too although this was not explicitly analysed here. The combination of altered hydromorphology and eutrophication is a common multi-stressor scenario across Europe.

Below the main findings from the three detailed analysis chapters are reported and placed in the context of our current understanding of system function - in particular the implications for eutrophication processes. Although the three countries considered can differ in river types, thee is sufficient overlap to provide comparable results between countries.

It should also be noted that the results presented ehe have limited application in the more arid regions of Europe, which are subject to much higher variation in hydrological regime, which can have in turn a more extreme influence on aquatic plant communities.

Denmark

There was clear evidence that habitat degradation in lowland streams (hydromorphological alterations and eutrophication) mediate selective changes in the functional trait characteristics of the macrophyte community. Could traits be used to distinguish between stressors? Yes for eutrophication: eutrophication included a shift toward free-floating submerged; anchored submerged; multi- and single-apical growth meristem traits. Yes for hydromorphology: the life form trait heterophylly was unique for hydromorphological degradation (channelised planform; a channelised cross-section; weed cutting intensity). This result probably reflect that channelisation and weed cutting, that is performed to preserve the channel profile, both contribute to homogenise the channel leaving restricted space for deposition zones suitable for heterophyllous species. Additionally, the overwintering capacity of the community increased upon degradation (channelised planform; a channelised cross-section; weed cutting intensity) and the abundance of species with basal growth meristem increased with increasing weed cutting intensity. This latter finding likely reflects that species with basal meristems may start re-growth immediately after cutting whereas species with apical growth meristems likely exhibit delayed re-growth. Did eutrophication and hydromrophology interact in a detectable manner? Yes: we did not find an increase in the abundance of species growing from apical meristems, as predicted from theory, with increasing levels of inorganic nutrients but instead the opposite. The sites involved were also subject to weed cutting and we suggest that weed cutting can set aside light as a factor controlling species composition under nutrient rich conditions. Thus, plenty of light may reach the stream bottom following biomass removal and at the same time shading from epiphytic algae is likely to be reduced.

Germany

Riparian shading was a crucial factor in determining the diversity of the macrophyte community. Sites which were not shaded had diverse species assemblages and traits. Riparian tree cover is considered part of a naturally functioning riparian zone, both in terms of physical and biological processes. As riparian tree cover is considered a prerequisite for sites of good status; good sites have poor macrophyte communities. The implication is that a more robustly defined reference condition for should be more robustly defined in terms of riparian tree cover should be included, which takes into account of how those trees would have contributed to creating macrophyte habitat, their natural patchiness, their contribution to localized dammed areas as part of debris dams and their use in dam construction by beavers. Results from the whole study also show that traits can indicate morphological degradation and that in multi-stressor systems eutrophication acts synergistically to hydromorphological degradation. However, these stressors can be split out and be indicated by different macrophyte traits.

UK

The UK data were strongly spatially structured with multiple stressors most obviously manifested in lowland, eastern England, in areas which had been drained for agriculture in historic times. As macrophyte communities differ along similarly spatially structured lines the question must be asked as to whether or not the macrophyte communities in these rivers can all be expected to respond to eutrophication in a similar manner. This is discussed in detail below.

2.1 Implications for the conceptual understanding of eutrophication processes

Eutrophication in rivers is a complex process which can be influenced by hydromorphological processes at various stages. To date the most explicit conceptual model of eutrophication impacts on river vegetation is that by Hilton et al (2006). They present a model where nutrient enrichment fuels the competition for light between aquatic vegetation. This leads to dominance by species which are competitive, the findings from Denmark, where species which exhibit apical growth dominate, is in broad agreement with this concept but differs in the detail.

The change in dominance by macrophytes of different growth forms, proposed by Hilton et al. (2006), where species which can produce floating or emergent leaves out shade submerged species, takes no account of the suitability of the physical habitat for those plants (Figure 2.1). From the studies presented here we know the ability of different growth forms is determined in the first instance by the suitability of the physical habitat and river style (Figure 5.6) and Deliverable 2.2. The idea of a hierarchy of factors influencing the suitability of a site for macrophytes is not new and has previously been proposed by Fox (1992), (Figure 2.2).

It therefore follows that the processes envisaged by Hilton et al. (2006) can only occur in systems which can support a wide range of macrophyte growth forms. Many of the lowland sites represented within the macrophyte chapters in this deliverable fall into this category and hence we see substantial shifts in traits which are however more subtle than those envisaged in the existing conceptual model. This is not surprising if one considers that broad scale shifts in dominance of macrophyte morphotypes is more likely to be determined by the suitability of hydraulic habitat. It is within the hydraulic setting, and the suite of species it supports, that one then sees shifts in plant traits associated with competition for light and other characterisitics.

The chapter on the Danish dataset goes into detail on the response to vegetation to eutrophication and how it interacts with changes to hydromorphology. These interactions are summarised in Figure 3.3 and provide a revised conceptual model for eutrophication in lowland systems.

The UK dataset included high energy upland rivers as well as lowland systems. It is worth noting, however, that in high energy upland river systems eutrophication has fewer potential outcomes. Following the hierarchy proposed by Fox (1992) in such systems in-stream vegetation is either entirely absent or only bryophytes are found, growing on stable boulders or bed rock. The physical habitat is unsuitable for other forms of macrophytes, although occasionally small and or temporary pockets of suitable habitat for higher macrophytes may be found. In such systems there is no possibility for higher macrophytes to outcompete bryophytes as a result of eutrophication. The extent to which they can colonise is restricted by physical habitat suitability. The only possible outcome of eutrophication is to increase the standing crop of bryophytes.

As there is only a handful of obligate submerged bryophyte species found across Europe the chances of significant shifts in community structure are unlikely. The most common submerged bryophyte, *Fontinalis antipyretica,* for example, has a broad ecological tolerance for P and is unlikely to be outcompeted. Algae outcompeting bryophytes in high energy systems seems unlikely too. Bryophytes are designed to grow slowly and can tolerate low light levels and low productivity conditions for sustained periods of time. Algae would therefore need to be present for significant portions of the year and this is again unlikely in high energy systems subject to significant shear stress. An alternative mechanism, where algae grow intertwined amongst bryophytes and thereby increase form drag, is again unlikely to cause bryophytes wash out as submerged bryophytes have exceptional stem /root strength.

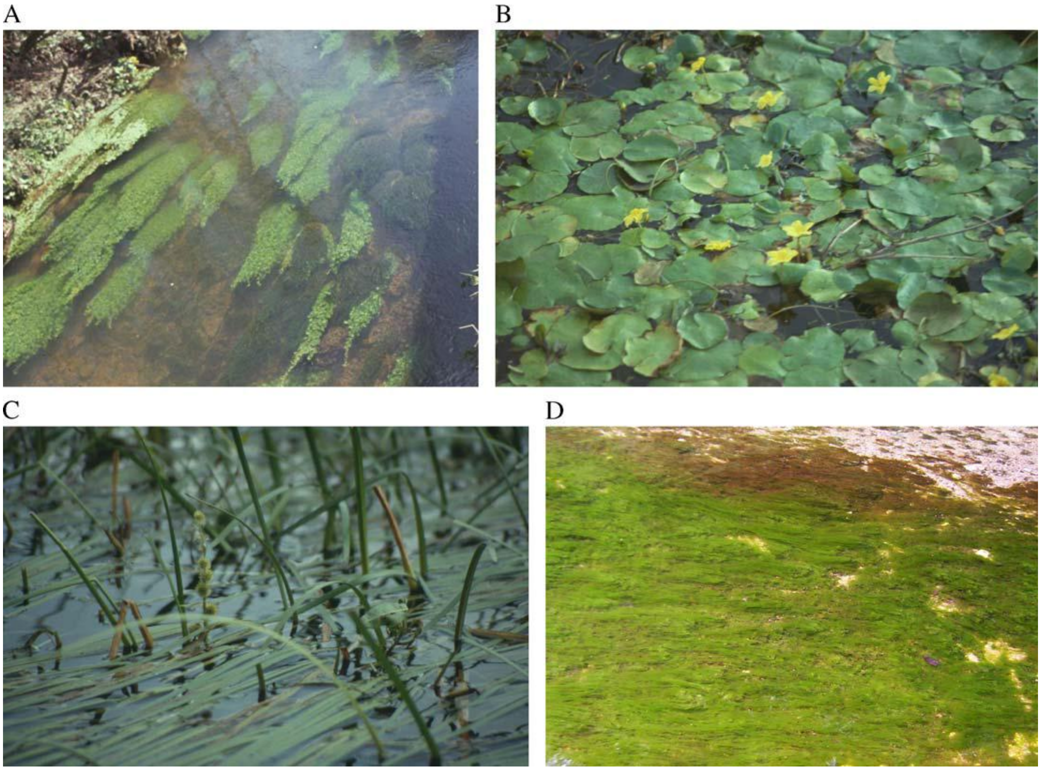

Figure 2.1 A diagram showing the possible changes in macrophyte dominance as eutrophication progreses from A to D; submerged leaved, floating leaved, emergent and finally filamentous algae, after Hilton et al 2006. This type of scenario could only develop in relatively sluggish flowing conditions in low energy systems and is likely to be diriven primarily by hydraulic habitat suitability.

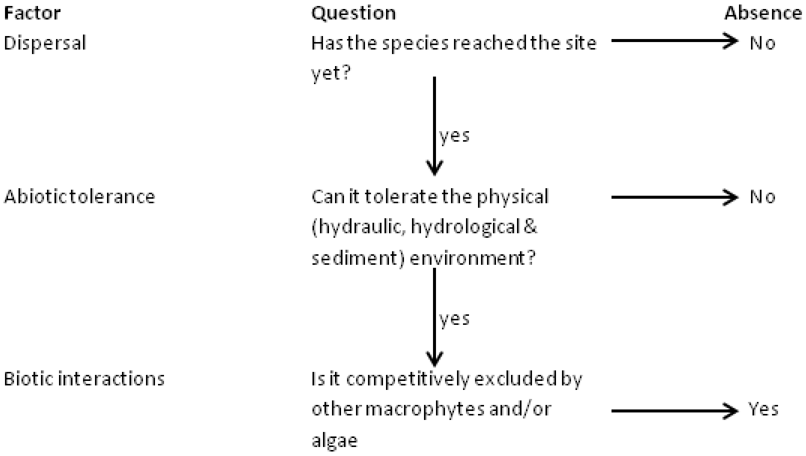

Figure 2.2 A diagram illustrating the hierarchy of factors which determine the occupancy of a site by macrophytes. Abiotic factors equate to hydromorphology in terms of the habitat available and natural disturbance. Biotic competitive interactions are mediated by eutrophication, modified from Fox (1992).

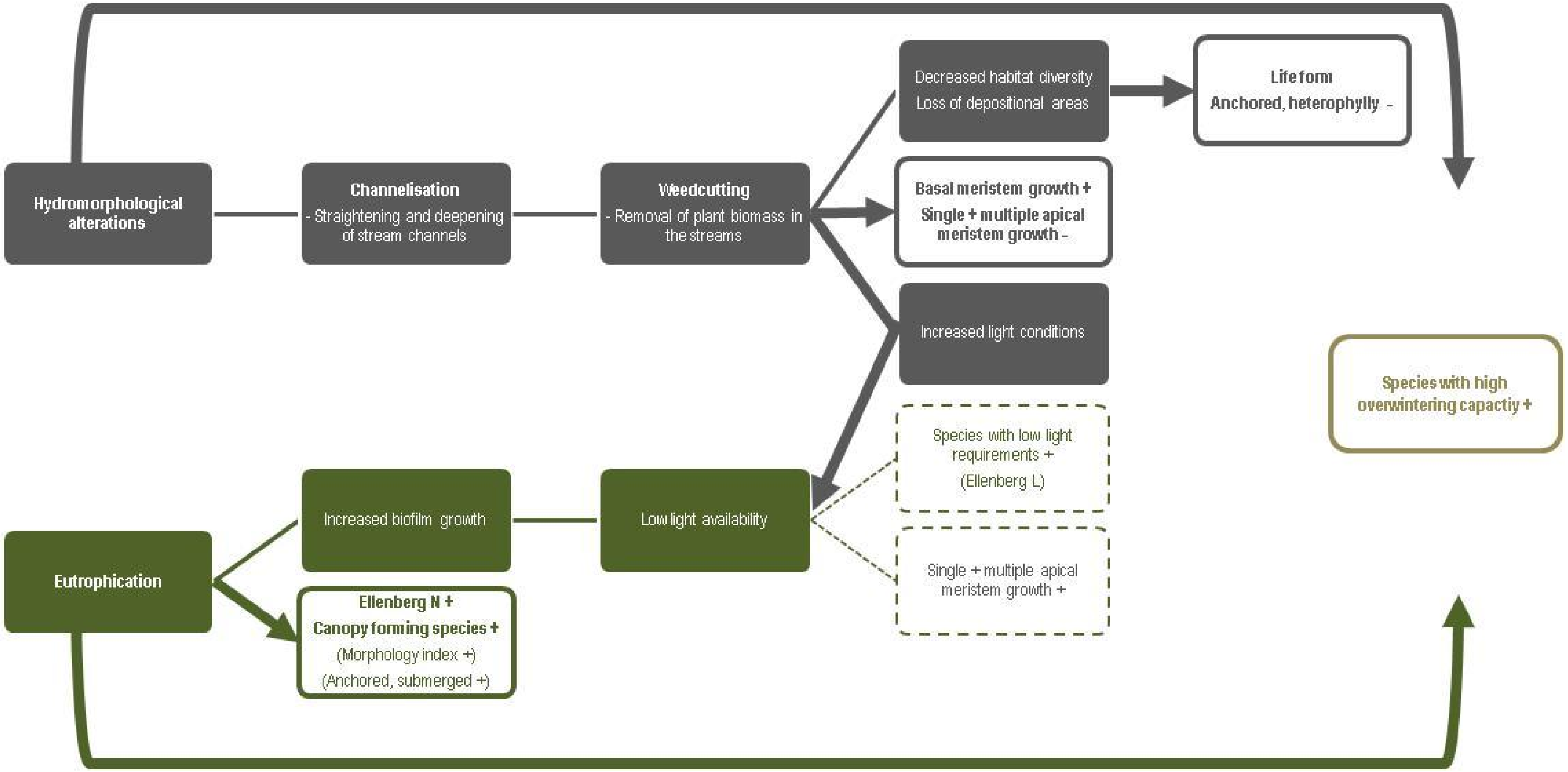

Figure 2.3 Decision tree showing the effect of hydromorphological alterations (filled grey text boxes) and eutrophication (filled green text boxes) on macrophyte trait composition (white text boxes) in lowland streams. White text boxes with grey and green text indicate that a change in trait composition is specific for hydromorphological alterations and eutrophication, respectively. White text boxes with brown text indicates that it is not possible to distinguish which pressure causes the change in trait composition. Plus (+) and minus (-) indicates an increased and decreased abundance of species with a specific trait, respectively. Text boxes with dotted lines indicate that an expected effect may be absent in the presence of certain pressures (here weed cutting).

2.2 Implications for the conceptual understanding of hydromorphological processes

The description of work indicates that ‘*macrophytes will be considered both as a habitat structuring element and a BQE*’. Here we consider the implications of the findings for the role of macrophytes in physical processes.

A channel’s resistance to water flow is varied by plants growing within its margins (Pitlo and Dawson 1990).

The absence of macrophytes from systems with ’reference’ levels of riparian tree cover, as found in Germany, could profoundly alter the hydrology of these systems. Equally, the shift in assemblage structure associated with eutrophication is likely to alter the resistance to flow of the channel as plants of different growth forms (which differ in traits in the Danish example) interact differently with flow. Previous research has demonstrated the amount of vegetation in a channel can increase with phosphorus and is associated with an increase in channel blockage (O’Hare et al 2010a, 2010b). In effect eutrophication exacerbates flood risk by increasing channel blockage by vegetation.

The difficulty in detecting impacts of re-sectioning in the UK was attributed in part to ecosystem engineering processes, by plants, in channels which had not been recently cut or dredged. This process could lead to a degree of ‘re-naturalisation’ but must be considered carefully and in a controlled manner as there are significant implications for flooding as the channels may become blocked.

The accumulation of heavy metals in and around macrophytes, observed in the Czech case study, is consistent with our knowledge that plants accumulate sediment. Considering the wider implications of these findings this type of process will be prevalent in systems where plants can grow and hydraulic conditions are suitable for the deposition of heavy metals in a manner analogous to the deposition of fine sediment (Figure 3.4). Deliverable 2.2 of this project uses a trait based typology to classify riverine plants by their ability to accumulate sediment and by extension they may well accumulate heavy metals in the substrate deposited within their canopies. In some systems such processes could lead to the longer term storage of heavy metals, where plants engineer channels and sediment builds up to the extent that berms and other structures are formed, burying any heavy metals.

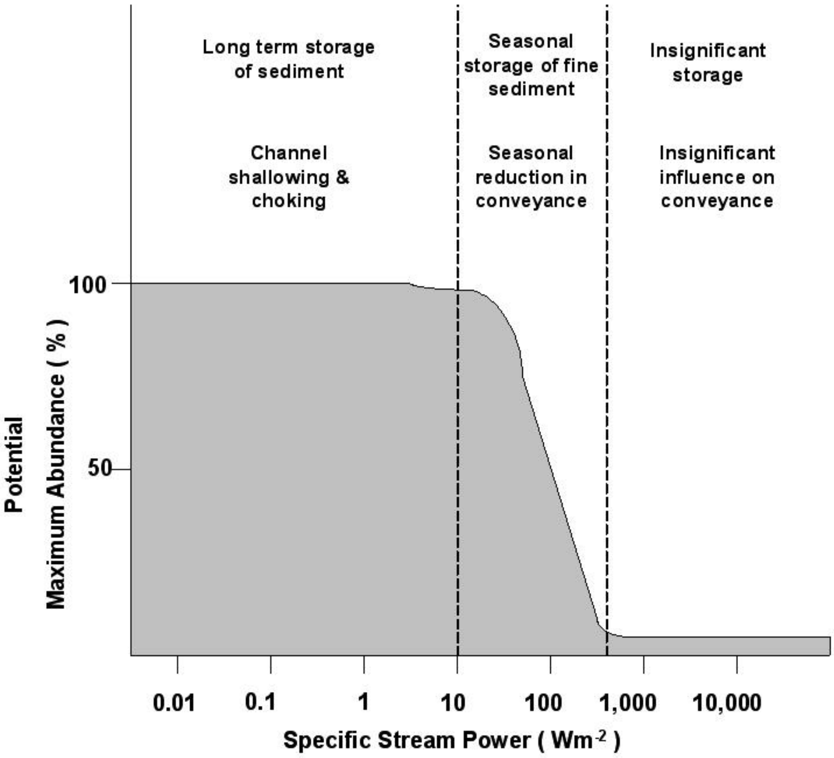

Figure 2.4 A diagram indicating the potential growth of macrophytes in channels of different stream power and their associated ability to retains sediment. After O'Hare et al 2011. Note current evidence indicates the actual boundaries (dotted lines)should be at somewhat higher stream power.

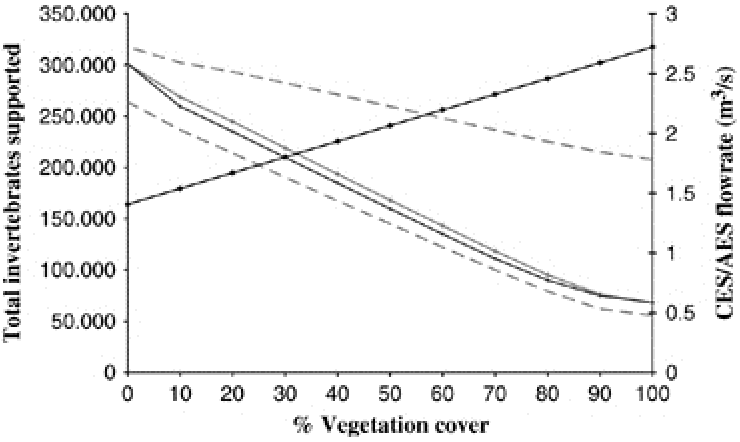

Figure 2.5 Change in number of invertebrates supported and total flow rate with percentage vegetation cover. This diagram illustrates the trade off in cutting macrophytes which block a channel's flow but provide habitat for invertebrates, River Frome, Dorset UK. After McGahey et al 2008.

2.3 Implications for reference conditions

The findings from Germany, where macrophytes were most diverse at sites which were considered to be of non-reference status because of the absence of riparian tree cover, highlights the pitfalls in setting reference conditions.

Unlike the situation for lakes where an impression of the historic macrophyte community can be established from plant fragments buried in chronological order the same cannot be done for macrophyte communities of rivers. Direct evidence is therefore absent but it is worth considering the information available on riparian tree cover. The contention that Europe was covered by forest after the last ice age has been seriously challenged (Vera 2000). The role of herbivores in creating openings in forest have been emphased and there is some field experimental evidence from America that indicates herbivory by large animals, in concert with other drivers, can alter riparian vegetation (Marshall 2014). Before widespread drainage, many areas of lowland Europe would have been wetlands, and bogs. These areas are unsuitable for large riparian trees. Pollen records and coleopteran fossils suggest that during the last interglacial period the vegetation of European floodplains contained about 40% non - arboreal plants and would therefore have been open, if patchy (Svenning 2002). Piecing the evidence together suggests that while there may well have been sections of rivers which had little macrophyte cover due to riparian shading, under natural conditions it is unlikely too that macrophytes were completely excluded.

Within WP2 of REFORM the focus has been on re-focusing on process rather than state. Such an approach may have practical applications in the context of setting reference processes for rivers. The processes relevant to the establishment of macrophyte communities in rivers with dynamic riparian zones are highlighted. Riparian trees can contribute to debris dams which flood small localised areas and would create habitat suitable of macrophytes. Such processes are forestalled in many European rivers as they are considered to cause flooding and reduce channel conveyance capacity. A wider question is therefore raised regarding the trade-off between flood conveyance and allowing rivers to function naturally and is closely related to issues of natural flood management. As modeling of eco-hydraulic processes improves it may be possible to set acceptable levels of ‘natural’ processes for a given channel conveyance capacity, or other desired service such as abstraction etc and relate this to BQEs (see Figure 2.5 for a practical example).

2.4 Recommendations

An overriding issue for all management tasks is setting reference conditions that are reasonable. It is clear from the findings presented here that a careful understanding of the natural bio-physical processes which determine a river’s style are critical to understanding what might be natural or desirable for a particular system. Setting reference conditions is subjective, often because returning a system to its truly natural state is impractical, especially in fertile lowland areas. Due attention should be paid to the descriptions of natural fluvial geomorphic processes described in the WP2 reports, which can act as guidance in this matter.

For managers concerned with the current condition of aquatic vegetation in their river systems attention should be given to the traits of the plants in question which may diagnose the relative importance of either channelisation and maintenance or eutrophication, or alternative synergistic interactions of both pressures. There is currently no simple indicator tool available based on the findings presented here but there is clear potential for one to be developed. To understand the condition of the macrophyte communities information on channel management should be routinely recorded.

For managers concerned with reviewing planning applications that may add to channelisation and eutrophication pressures on rivers, there is clear evidence that both types of pressure can have an individual and detrimental effect on aquatic vegetation. The relationships presented here are qualitative, not quantitative, and as suggested above would require further development to become quantitative. Any tool which is developed should consider the cumulative impact on the entire river of both increases in nutrient load and alterations to hydromorphology.

For managers considering restoring a river and its vegetation and wondering which pressure is having the most negative influence, the diagram in Figure 2.2 can help one decide. A clear message however is that removing one pressure may not achieve an improvement in status if other pressures are not tackled.

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3 Macrophytes - responses in Denmark

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3.1 Introduction

Within the REFORM project D3.2 we address the effects of multiple-stressor interactions, focusing on hydromorphological stresses. Here we investigate the empirical evidence for stressor interactions between eutrophication, channelisation, weed cutting and channel maintenance. Although this study focuses on lowland streams in Denmark, this combination on interacting stressors is common in intensive lowland agricultural and urban areas across central and northern Europe. In Deliverable D3.1 we investigated hydromorphological stressor impacts on their own. Here eutrophication is also considered. The streams types analysed equate to Type R-C1 and R-C4 described in Deliverable 2.1. This chapter should be read in combination with the analysis of macrophytes to a similar set of stressors in central Germany.

Lowland stream ecosystems worldwide are affected by human pressures (Vörösmarty et al. 2010) that dramatically alter habitat conditions and causes loss of biological diversity (Pimm et al. 1995). Historically close attention has been paid to negative impacts associated with nutrient loads to streams and rivers, but today there is a gradual change in this perception as stress related to hydromorphological alterations are considered increasingly important for failure to achieve good ecological status in lowland streams (EEA 2012). Even though the importance of hydromorphological degradation is now accepted as a major stressor, the ability to assess the level of hydromorphological impact on the biological communities remains restricted (e.g. Vaughan et al. 2009; Feld et al. 2014). One reason could be that the high level of spatial and temporal variability characterizing stream and river habitats makes it difficult to assess the hydromorphological impact at a scale relevant for the biological communities. At the reach scale, the biota respond to local hydromorphological features (i.e. the interaction between the flow of water and the channel form) but additionally, the communities reflect larger-scale environmental filters that can mask the effect of local factors on species composition (Poff 1997; Kail and Wolter 2013).

A large majority of studies conducted on the effects of hydromorphological degradation on biological communities have focused on species richness and/or multivariate descriptors of species composition (e.g. Hering et al. 2006; Dahm et al. 2013 but also see Feld et al. 2014 and, Elosegi and Sergi 2013 and references therein). However, taxonomic composition may differ between regions due to spatial constraints on community assembly making compositional approaches vulnerable to scale-dependent processes. That is, processes operating at larger (e.g. regional) spatial scales have already filtered the species pool before local environmental factors ultimately determine local species composition (Poff 1997) and consequently, regional processes often mask the effect of local factors on species composition (Poff 1997). Functional community characteristics have been suggested as an alternative or complement to compositional characteristics. Because similar traits (responding to similar environmental conditions) can be applied to most species in the world, functional composition is thought to be less vulnerable to scale-dependent processes (e.g. Dolédec et al. 2006; Friberg et al. 2011). Therefore, trait composition can be expected to show high and limited correspondence with local environmental and regional factors, respectively. Additionally, traits provide a means to gain insight into the mechanisms behind the in response to natural and anthropogenic drivers of change (Diaz et al. 2007; Moretti and Legg 2009). This insight may help identifying trait characteristics of species with specific sensitivity towards major anthropogenic pressures in different types of ecosystems, which, in turn, may enable society to take adequate measures to combat species loss.

In this study we explore whether functional characteristics of the macrophyte community are suitable for distinguishing between different forms of hydromorphological degradation and eutrophication in lowland streams. Generally, macrophytes are abundant in this stream type, quite stable over summer and additionally, they are likely to respond to hydromorphological degradation at a scale normally applied in hydromorphological assessments (i.e. the reach scale). We focus on hydromorphological alterations typical in lowland areas with intensive agriculture, where a large majority of streams have been channelised (e.g. deepened, widened and straightened; Figure 3.1 - 4.3) to optimise water runoff and reduce flooding and sedimentation of adjacent areas. These changes inevitably mediate a homogenisation of in-stream channel habitats with the development of more uniform flow patterns, substrate conditions and depth characteristics (Baattrup-Pedersen and Riis 1999; Rambaud et al. 2009). Furthermore, these characteristics are actively maintained in many places by regular dredging and mechanical removal of the vegetation (Fox and Murphy 1990; Baattrup-Pedersen et al. 2009; Wiegleb et al. 2014).

We use an approach based on the mean functional trait characteristics of the macrophyte community. A wide range of traits can be used to describe the response of species to their environment and different traits may capture different aspects of species resource use and habitat requirements. For plants, traits related to life form characteristics, growth forms, growth rates, photosynthetic pathways, leaf morphology and chemistry have been used to describe responses to environmental conditions as they may affect species growth, survival and reproductive output (e.g. Violle et al. 2007). For aquatic species, however, the numbers of traits that can be obtained remain few (Willby et al. 2000; Klotz et al. 2002; Kühn et al. 2004; Cavalli et al. 2014). Therefore we were restricted to traits related to the ecological preferences, life form, morphology, dispersal and survival.

Specifically, we aimed to investigate the link between community trait characteristics and specific types of hydromorphological stress in systems subject to eutrophication. Furthermore we addressed the question, can trait characteristics provide insight into the mechanisms that mediate changes in community structure taking into account the level of eutrophication? (Baattrup-Pedersen et al. 2014) We hypothesized that (i) macrophyte traits describing life form and growth characteristics respond to hydromorphological degradation such as stream channelisation and maintenance practice and (ii) specific traits can distinguish hydromorphological degradation from eutrophication of the streams.

Figure 3.1 Part of Skals å situated in Jutland, Denmark. This section has been straightened and deepened to improve drainage from adjacent areas.

Figure 3.2 Part of Vorgod å situated in Jutland, Denmark. This section has a natural planform and a well-developed in-stream vegetation. Variation in depth is large and increases over summer.

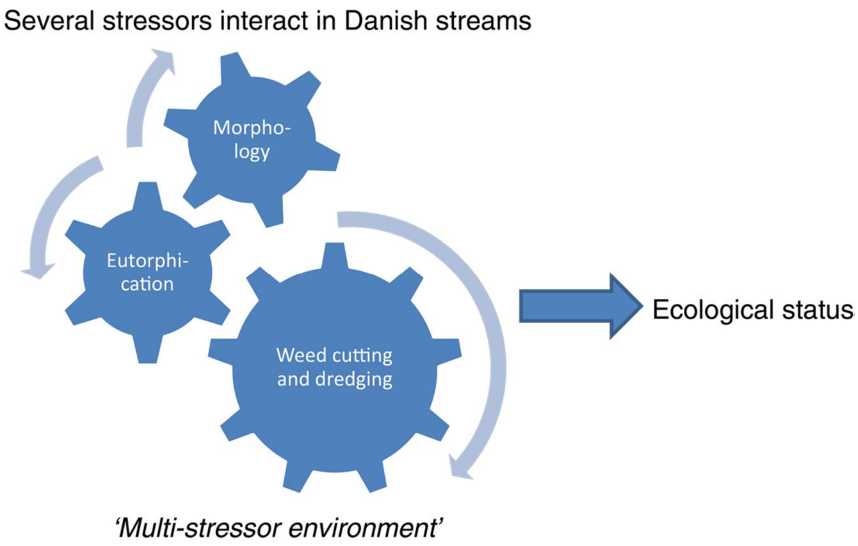


Figure 3.3 Diagram showing the most important stressors in Danish streams with detrimental impacts on the ecological status.

3.2 Methods

3.2.1 Macrophyte and environmental data

We used monitoring data from a total of 147 stream reaches where we had combined data on macrophyte species abundance, catchment and buffer strip land use, hydromorphological alterations (i.e. cross section, planform, weed cutting) and water chemistry parameters (NH_4, PO_4, NO_3). These 147 sites were all categorised as middle-sized (type 2 and 3; Baattrup-Pedersen et al. 2004) with a catchment area larger than 10 km^2; they were distributed throughout Denmark and covered existing gradients in alkalinity and type of catchment. Macrophyte data were collected following the protocol described in Pedersen et al. (2007). In each stream reach, macrophyte recordings were made in approximately 250 plots (25 x 25 cm) placed adjacently along 10 cross-sectional transectsof a 100m long stream reach (Figure 3.4). A cover score was allocated to each species present in the plots using the following abundance scale: 1=1-5%, 2=6-25%, 3=26-50%, 4=51-75%, 5=76-100%. Local species abundance in each stream reach was then calculated as the sum of cover scores relative to the maximum score sum (i.e. the number of plots multiplied by the maximum score of five).

The catchment upstream of the stream reaches and in a 50 m wide buffer along upstream reaches in the catchment were delineated using the Analysis Tools in ESRI ArcGIS 9.2. Agricultural land use was then determined from a national land cover raster map (25 m grid) with 12 land cover classes (Nielsen 2000) and information on land use mandatory reported by all farmers annually to the Danish Ministry of Agriculture as a requirement for obtaining EU subsidies (DFFE 2008). This latter source contains information on field location and crop type. Only land cover polygons classified as arable land were allocated to agriculture.

Figure 3.4 Survey of macrophyte vegetation in a Danish stream. According to the Danish Monitoring Programme, macrophyte recordings are made in approximately 250 plots (25 x 25 cm) in each stream reach placed adjacently along 10 cross-sectional transects.

Table 3.1 Overview of number of stream reaches categorized into the hydromorphological categories. More information on how the streams were categorized is given in the method section.

Variable	Category	N	%
Channel planform	PL_Channelised	27	18.4
	PL_Meandering	42	28.6
	PL_Straight, natural	23	15.6
	PL_Sinuous	55	37.4
Cross-section	CS_Channelised	52	35.4
	CS_Natural	95	64.6
Weed-cutting intensity	Weed>halfwidth	68	46.3
	Weed<halfwidth	22	15.0
	Weed_none	57	38.8

Hydromorphological features of the stream sites (i.e. cross-section and planform) were recorded at the time of the macrophyte sampling whereas information on weed cutting practice was obtained from the water authorities. Cross-section was categorized into natural and channelised from the depth characteristics. Channelised cross sections had a trapezoid form with similar depths across the channel profile, whereas natural cross sections had variability in depth characteristics. Furthermore, channelised cross sections were deeply positioned compared to the surroundings whereas natural cross sections were in level with the surroundings. Information on stream shape was based on a recording in the field and from photos to assess the sinuosity of the channel. The following categories were used: channelised planform; straight (natural) planform, sinuous planform, and meandering planform. Weed cutting was categorized into three categories depending on the extent of the cutting in the channel in either full-half width including stream reaches being cut full to the half width of the cross section, net-half width including stream reaches being cut from half width to less than half width of the cross section and reaches that were left uncut. The numbers of stream sites within each of the different categories are given in Table 4.1 and Figures 4.5 - 4.7.

Water chemistry data were based on samples taken five times yearly, that were analysed in the laboratory according to European standards for ortho-phosphate (EN ISO 6878), nitrate (EN ISO 13395), alkalinity (DS/EN ISO9963-2 1996), total nitrogen (EN ISO 11905-1) and total phosphorous (EN ISO 6878).

Figure 3.5 Weed cutting in a middle-sized Danish stream. The vegetation is cut close to the stream bottom from razors mounted on the boat.

Figure 3.6 Weed cutting in a small Danish stream. Cutting is performed manually using a scythe.

Figure 3.7 In some streams dredging is performed to maintain the channel dimensions.

3.2.2 Description of traits

A total of 52 submerged and amphibious taxa were observed of which we were able to allocate traits to 41 species representing 79% of the total submerged and amphibious species pool. The traits covered traits related to ecological preference values (Ellenberg N and L), life form and traits related to dispersal, reproduction and survival (Table 3.2). The Ellenberg indicator values (Ellenberg et al. 1991) offer autecological information on the response of approximately 2000 species to a range of climatic and edaphic factors in central Europe. They have been applied to aquatic vegetation in a few recent studies and we therefore decided to integrate them also here to analyse their variability with community trait characteristics.

Trait data were extracted from the literature and online databases (Willby et al. 2000; Klotz et al. 2002; Kühn et al. 2004; Table 3.2). The trait life forms (LF) were divided into six: free floating (surface and submerged), anchored with both floating and submerged leaves, and amphibious species with homophyllus emergent leaves and heterophyllus emergent leaves. Growth morphology was divided into four forms: single basal, multi apical, single apical or multi+single apical (Table 3.2). Plant morphological traits also included a morphology index building on the height and lateral extension of the canopy and the leaf area of the species. Dispersal was characterised by four traits. Dispersal was inferred from the ability to disperse by forming extensive root-rhizome systems, the ability to reproduce by fragmentation, the number of seeds and number of reproductive organs produced by the species. We also integrated traits related to survival in terms of overwintering organs such as tubers, turions and rhizomes.

The life form traits, and traits covering fragmentation, seeds, overwintering organs and rhizomes, were based on presence/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). The morphology index was also classified into categories (2, 3-5, 6-7, 8-9 and 10) with values ranging from 1 to 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 to the particular trait (i.e. 1.5, 2.5, 3.5 and 4.5).

Table 3.2 The 19 functional traits used in the present study to characterize the plant species. The selected traits give information on ecological preference (Ellenberg Light and Ellenberg Nitrogen), life form, morphology (meristem characteristics; leaf area; canopy characteristics), dispersal (root-rhizome growth; fragmentation; seed production) and survival (overwintering organs). See text for further explanations.

Short trait name	Explanation	Category
LE	Ellenberg Light	Ecological preference
NE	Ellenberg Nitrogen	Ecological preference
Frflsr	Free floating, surface	Life form
Frflsb	Free floating, submerged	Life form
Anflle	Anchored, floating leaves	Life form
Ansule	Anchored, submerged leaves	Life form
Anemle	Anchored, emergent leaves	Life form
Anhete	Anchored, heterophylly	Life form
meris.ma	Meristem multiple apical growth point	Morphology
meris.sb	Meristem single basal growth point	Morphology
meris.sa	Meristem single apical growth point	Morphology
meris.sa.ma	Meristem single+multiple apical growth point	Morphology
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

3.2.3 Data analysis

As a first step, we wanted to assess the relative importance of different groups of environmental variables (describing the most important types of anthropogenic impacts) for macrophyte trait composition in the streams. For this purpose, we performed variation partitioning analysis (function varpart in R package vegan; Oksanen et al. 2013) which uses partial redundancy analysis (pRDA) to calculate how much of the variance in trait composition that is explained uniquely by two or more explanatory matrices, as well as the shared fraction explained between them. Trait composition was assessed using community weighted means (CWMs) of the traits per plot. CWMs were then calculated using R package FD (Laliberté and Shipley 2011). The CWMs were calculated with the function functcomp as:

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

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

Environmental variables were grouped as agricultural activity, local hydromorphological alterations, nutrients and typology. The environmental variables included in the three first groups (which were of main interest for this study) are described above. We also included typology (a dummy variable indicating typology 2 or 3) to ensure that not too much variance in trait composition was uniquely explained by this factor. In that case, significant patterns in RLQ and fourth-corner analyses (described below) may be masked by the influence of typology (which was not of prime interest here) and separate analyses may have to be performed for each stream type. We also wanted to ensure that not too much variance in trait composition was jointly explained by the different environmental groups as this would make it difficult to distinguish effects of one variable type from another (e.g. nutrient vs. hydromorphological effects) in the RLQ and fourth-corner analyses. Before conducting the actual variation partitioning analysis, we ran Redundancy Analyses (RDAs) (function rda) on the environmental variable groups vs. trait composition separately (hereafter called global models). If the global models were not significant, no further analyses were conducted and we concluded that these environmental variables were not significantly explaining variation in trait composition. In case of significant global models, we proceeded with a forward selection procedure (function forward.sel in R package packfor; Dray 2009) and significant variables were retained for further analyses. In the forward selection procedure, the selected variables had to be significant at alpha 0.05 and the adjusted R2 of the final model had to be lower that the adjusted R2 of the global model (Blanchet et al. 2008). The retained variables were then used in the variation partitioning analysis, in which the unique fraction of total variation in trait composition significantly explained by each set of environmental variables was estimated (using adjusted R2 values, Peres-Neto et al. 2006).

We used RLQ and fourth-corner analyses to assess the covariation between environmental variables (hydromorphological and nutrient variables) and traits. RLQ analysis is an extension of co-inertia analysis that provides an overview of the multivariate associations by searching for a combination of traits (Table Q) and environmental variables (Table R) with maximal covariance, which is weighted by the abundance of the species in the plots (Table L) (Dolédec et al. 1996). First, correspondence analyses (CA) was applied to Table L, principle component analysis was applied to Table Q and Hill and Smith analysis (Hill and Smith 1976) was applied to Table R as it contained a mix of qualitative (hydromorphological variables) and quantitative (water chemistry variables) variables. RLQ analysis was then applied which combines the three separate ordinations and identifies the main associations between Table R and Q, linked by Table L. In the RLQ analysis, the site scores in Table R constrain the site scores in Table L, and the species scores in Table Q constrain the species scores in Table L. The axis that maximises the covariance in Table L is then selected which result in a compromise between the best joint combination of site scores by their environmental characteristic, the best combination of species scores by their trait attributes and the simultaneous ordination of sites and scores. The overall significance of the relationship between the environmental variables (R) and species traits (Q) was assessed with a Monte Carlo test with 999 permutations on total inertia of the RLQ analyses (Dolédec et al. 1996).

We also performed a fourth-corner analysis on our data which, similarly to RLQ, computes a new matrix that relates the environmental variables to biological traits (Legendre et al. 1997).

However, the fourth-corner method provides an additional significance test of all possible bivariate associations between single traits and environmental variables which allows for a more detailed and specific interpretation of trait-environment associations. We first performed an analysis of variance statistic (i.e. the global F-statistic) for the categorical environmental variables (cross-section, river planform and weed cutting) to test whether an overall trait-environment association existed. In case of a significant F, at least one environmental category differed from the others in terms of species traits. We then explored the bivariate relationships further. The statistics of the fourth-corner method depend on the type of variables. In case of two quantitative variables (i.e. when both the trait and the environmental variable are quantitative) the Pearson product-moment correlation (r) coefficient is used. However, in case of quantitative traits and qualitative environmental variables, Legendre et al. (1997) suggested the use of either homogeneity statistic (d) or Pearson product-moment correlation coefficient (r). We used the latter option (r) to obtain the both the strength and direction (positive or negative) of association between the environmental variables and traits. The significance of r and F was obtained by permuting simultaneously the rows of Tables R and Q (999 runs) following the model proposed by Doledec et al. (1996). RLQ and fourth-corner analyses were all performed in R package ade4 (Dray and Dufour 2007).

3.3 Results

The stream reaches were quite different regarding total macrophyte coverage, percentage of agriculture close to the stream (50 m wide zone) and in the catchment and also regarding the concentration of major nutrients for plant growth in the stream water (Table 3.3).

Table 3.3 Key characteristics of the study reaches. Nutrient concentrations are based on five yearly samples. Percentage of agriculture were derived from GIS in the whole catchment and in a 50 m wide zone from the stream channel.

	Mean	SE	Min	Max
Macrophyte coverage (%)	32.7	1.7	0.7	84.1
NH_4N (mg l^{-1})	0.10	0.01	0.02	0.39
NO_3N (mg l^{-1})	3.42	0.17	0.04	8.48
PO_4P (mg l^{-1})	0.05	0.00	0.01	0.18
Agriculture buffer (%)	44.3	1.6	0.9	84.1
Agriculture catchment (%)	62.6	1.4	0.6	86.8

Typology did not significantly explain variation in trait composition (RDA ANOVA; $p>0.05$) suggesting that we could treat the two typologies together in the following analyses. Hydromorphological degradation, eutrophication and agricultural land use all explained unique and significant ($p<0.01$) fractions of the variation in macrophyte trait composition (4%, 4% and 3%, respectively) with only a limited amount of variation explained jointly by the three environmental groups (1%) (Figure 3.8). We, therefore, infer that trait characteristics can distinguish stressors related to hydromorphological degradation, eutrophication and agricultural land use. For the remaining analysis we only retained variables related to hydromorphological degradation and nutrients in the stream water being those most directly related to habitat conditions in the streams.

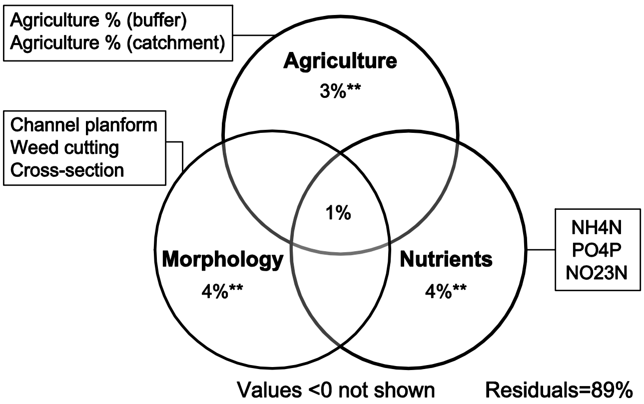


Figure 3.8 Venn diagram showing the results of the variation partitioning analysis. Unique fractions (%) of macrophyte trait composition explained by agricultural activities, morphological alterations and nutrients are shown in resp. circle. The significance of each fraction explained is indicated in the figure (**p<0.01). Shown is also fractions explained jointly (overlapping parts of the circles) by all three explanatory matrices, agricultural activities and morphological alterations, morphological alterations and nutrients, and nutrients and agricultural activities. The residuals (i.e. unexplained variance) in the analysis are shown in the lower right hand corner. The sum of all variances shown in the figure can exceed 100%. This is because variances can be negative (Legendre and Legendre 1998), but these values are not shown (values <0 not shown). The text boxes next to each circle indicate which variables were included in the variation partitioning analysis (i.e. variables selected in the forward selection). The variables are displayed in the order that they were selected.

We found that the distribution of macrophytes in the streams could be explained by linking the trait characteristics of the species to the environmental conditions (Monte-Carlo test; P<0.001). The first three axes of the RLQ explained 51%, 22% and 16% of the total variance that links the environmental characterises in table R with species traits in table Q (Figure 3.9; Table 3.4). The first RLQ axis differentiated channelized reaches from more meandering reaches (straight natural; meandering) without cuttings (Figure 3.9) whereas the second axis differentiated reaches with a channelised cross section and less cutting intensity from reaches with a natural cross section but more intensive cutting. The amounts of nutrients in the stream water were related to both axis one and two. Interestingly though, reaches with high levels of PO_4P were differentiated from those with high levels of NO_3N (Figure 3.9b). For the traits, the first RLQ axis differentiated survival (overwintering), productivity (EN) and dispersal by rhizome growth from some of the specific life form characteristics (heterophylly; anchored floating leaved) and dispersal by fragmentation, whereas the second axis differentiated meristem characteristics i.e. apical and multi-apical meristem growth from basal meristem growth (Figure 3.9b). The differentiation of species according to RLQ axes are shown in Figure 3.9c.

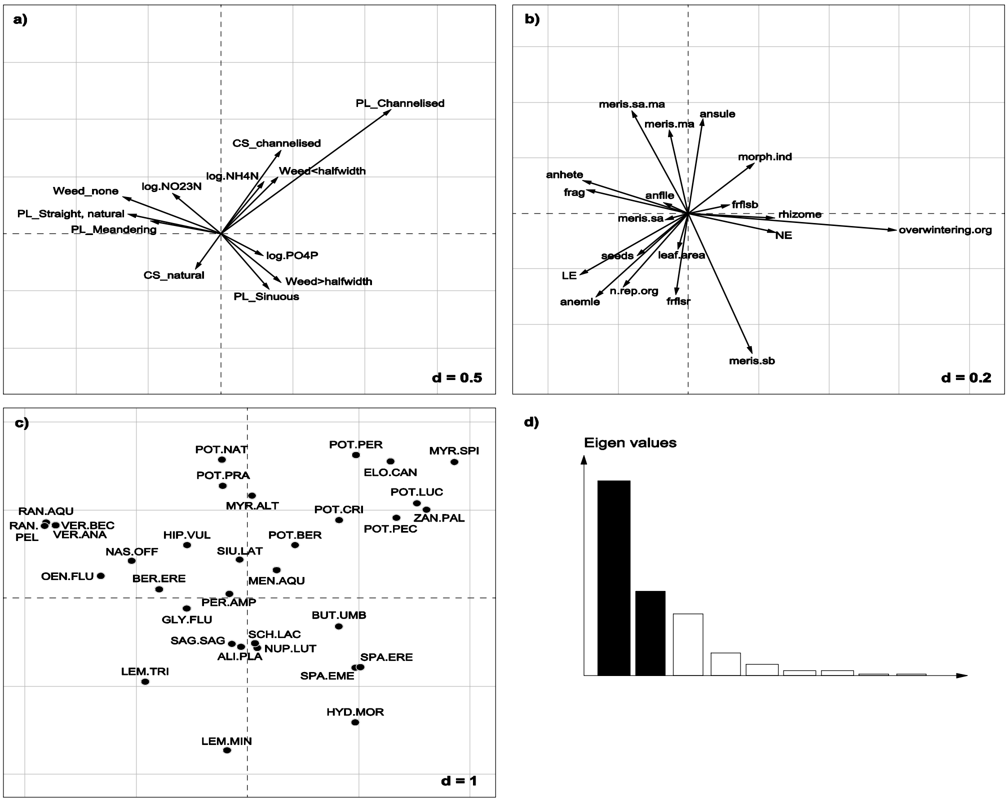


Figure 3.9 Results of the first two axes of RLQ analysis: (a) coefficients for the environmental variables, (b) coefficients for the trait, (c) eigenvalues and scores of species and (d) eigenvalues, with the first two axes in black. The values of d give the grid size. Codes for variables are given in table 1 and 2 and for species in Appendix A.

Table 3.4 Summary of the RLQ analysis: eigenvalues and percentage of total co-inertia accounted for by first three RLQ axes, covariance refers to the covariance between the two new sets of factorial scores projected onto the first three RLQ axes (square root of eigenvalue); correlation refers to the correlation between the two new set s of factorial scores projected onto the first three RLQ axes; cumulative inertia refers to the variance of each set of factorial scores computed in the RLQ analysis, both for the environment and for the traits.

	Axis 1	Axis 2	Axis 3
Eigenvalues	0.37	0.16	0.12
% of total co-inertia	51%	22%	16%
Covariance	0.61	0.40	0.34
Correlation	0.33	0.21	0.15
Cumulative inertia (environment)	1.62	2.88	4.35
Cumulative inertia (traits)	2.15	5.06	8.71

We found several significant associations between the environmental variables and the trait characteristics (Table 3.5). Generally, we found that traits within each of the five categories (ecological preference; life form; morphology; dispersal; survival) were significantly related to one or more of the hydromorphological variables (planform; cross-section; weed-cutting; Table 3.5). One of the life form traits (heterophylly) and one trait related to survival (overwintering) were responded to all types of hydromorphological degradation (i.e. planform, cross section and weed cutting) (Table 3.5).

Looking into the specific hydromorphological stressors, we found significant bivariate associations to several trait characteristics (Figure 3.12). In many instances, the relationships were similar regarding planform and cross-section. LE, anchored emergent leaves and anchored heterophylly were all negatively associated with channelisation whereas overwintering organs was positively associated with channelisation (Figure 3.12). Additionally, the morphology index and reproduction by rhizome growth were positively associated with a channelised planform but not with a channelised cross-section. Anchored heterophylly was negatively associated with weed-cutting together with single+multi apical growth meristem, whereas both traits were positively associated with no cutting. In contrast, single basal growth meristem was positively associated with weed-cutting but negatively with no cutting. NE and overwintering were also traits negatively associated with no cutting.

Figure 3.10 *Oenanthe fluviatilis* (Bab.) Coleman is a rare heterophyllous species in Danish streams. The submerged leaves are finely sectioned.

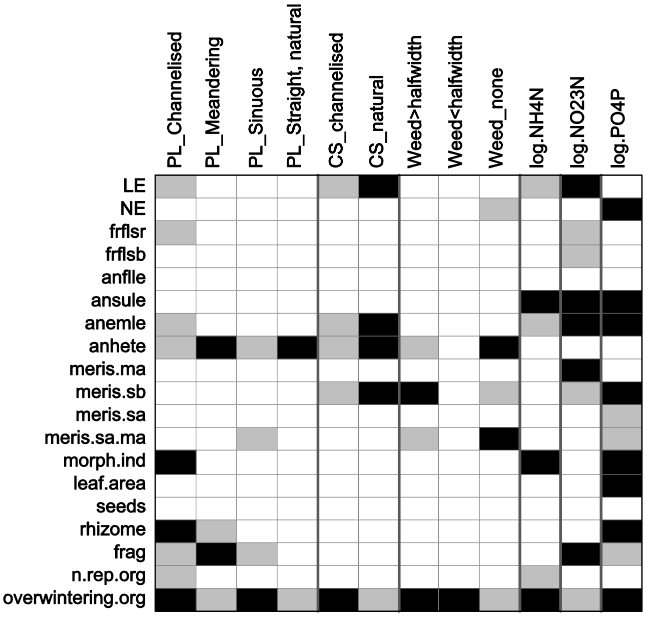
Figure 3.11 A mixture of several species in a hydromorphologically unmodified stream reach. Note the submerged and aerial leaves of *Ranunculus*. The water current is fast under the biomass of the plants and between the plants.

Several traits were also significantly related to the level of eutrophication (Figure 3.12), but the response varied for the different types of nutrients. The ecological preference for light (LE) decreased with increasing levels of NH_4N but increased with increasing levels of NO_3N, whereas no significant relationships was found between LE and PO4P (Figure 3.12). Instead a significant relationship was found between the ecological preference for nutrients (NE) and PO_4P (Figure 3.12). Several life form characteristics were also related to the concentration of nutrients in the stream water. For example, anchored species with submerged leaves increased with increasing concentrations of all nutrients, whereas anchored species with emergent leaves increased with increasing concentrations of NO_3N and PO_4P, but not with increasing concentrations of NH_4N (Figure 3.12). We also found that several morphological traits (i.e. position of growth meristems and canopy characteristics) were significantly related to the concentration of nutrients (Figure 3.12). In particular, apical and single+multi apical growth meristems were negatively associated with increasing levels of PO_4P whereas single basal growth meristems were positively associated with increasing levels of PO_4P (Figure 3.12). The overwintering capacity increased significantly with increasing concentrations of all nutrients (Figure 3.12).

Table 3.5 Pseudo F-statistics of the fourth-corner analysis for the categorical hydromorphological variables (cross-section, river shape and weed cutting).

		River shape			Cross section			Weed cutting		
	Stat.	Value	Prob.		Value	Prob.		Value	Prob.	
LE	F	2.75	0.012	*	4.06	0.013	*	0.59	0.422	
NE	F	1.11	0.133		0.85	0.229		2.42	0.027	*
frflsr	F	0.94	0.209		0.66	0.303		0.52	0.425	
frflsb	F	0.45	0.529		1.29	0.152		0.38	0.515	
anflle	F	0.10	0.907		2.55	0.046	*	1.52	0.098	.
ansule	F	1.08	0.173		0.54	0.345		0.62	0.379	
anemle	F	1.96	0.036	*	9.10	0.001	***	0.89	0.271	
anhete	F	3.27	0.004	**	2.52	0.042	*	6.23	0.001	***
meris.ma	F	0.53	0.459		2.03	0.078	.	0.63	0.375	
meris.sb	F	1.17	0.150		3.21	0.025	*	3.32	0.006	**
meris.sa	F	0.77	0.326		0.34	0.474		0.24	0.692	
meris.sa.ma	F	1.85	0.047	*	1.22	0.176		3.56	0.004	**
morph.ind	F	1.11	0.155		0.41	0.419		1.45	0.109	
leaf.area	F	0.17	0.860		1.27	0.147		0.13	0.799	
seeds	F	0.53	0.462		1.26	0.154		1.01	0.209	
rhizome	F	2.21	0.021	*	0.29	0.479		1.17	0.163	
frag	F	2.91	0.005	**	0.13	0.648		1.15	0.175	
n.rep.org	F	1.16	0.157		2.14	0.065	.	1.32	0.141	
overwintering.org	F	6.12	0.001	***	4.80	0.009	**	9.64	0.001	***

. p <0.1, * = p<0.05, ** = p<0.01, *** p < 0.001



	PL_Channelised	PL_Meandering	PL_Sinuous	PL_Straight, natural	CS_channelised	CS_natural	Weed>halfwidth	Weed<halfwidth	Weed_none	log.NH4N	log.NO23N	log.PO4P
LE	▨	□	□	□	▨	■	□	□	□	▨	■	□
NE	□	□	□	□	□	□	□	□	▨	□	□	■
frflsr	▨	□	□	□	□	□	□	□	□	□	▨	□
frflsb	□	□	□	□	□	□	□	□	□	□	▨	□
anflle	□	□	□	□	□	□	□	□	□	□	□	□
ansule	□	□	□	□	□	□	□	□	□	■	■	■
anemle	▨	□	□	□	▨	■	□	□	□	▨	■	■
anhete	▨	■	▨	■	▨	■	▨	□	■	□	□	□
meris.ma	□	□	□	□	□	□	□	□	□	□	■	□
meris.sb	□	□	□	□	▨	■	■	□	▨	□	▨	■
meris.sa	□	□	□	□	□	□	□	□	□	□	□	▨
meris.sa.ma	□	□	▨	□	□	□	▨	□	■	□	□	▨
morph.ind	■	□	□	□	□	□	□	□	□	■	□	■
leaf.area	□	□	□	□	□	□	□	□	□	□	□	■
seeds	□	□	□	□	□	□	□	□	□	□	□	□
rhizome	■	▨	□	□	□	□	□	□	□	□	□	■
frag	▨	■	▨	□	□	□	□	□	□	□	■	▨
n.rep.org	▨	□	□	□	□	□	□	□	□	▨	□	□
overwintering.org	■	▨	■	▨	■	▨	■	■	▨	■	▨	■

Figure 3.12 Results of the fourth-corner analysis. The table shows all possible bivariate associations between the environmental variables (hydromorphological categories and nutrient variables) and macrophyte traits. Significant ($P<0.05$) positive associations are represented by black cells, and significant negative associations are represented by grey cells. Non-significant associations are white. Codes for traits and environmental variables are explained in Table 3.1 and Table 3.2.

3.4 Discussion

We found a highly significant relationship between macrophyte trait characteristics as measured by traits related to ecological preferences, life form, morphology, dispersal and survival and the environmental characteristics of the stream habitats. This finding provides evidence that environmental conditions associated with habitat degradation (hydromorphological alterations and eutrophication) mediate selective changes in the mean functional trait characteristics of the community. We found that traits related to both ecological preference, life form, morphology, dispersal and survival were associated with habitat degradation, but only a few of these were unique, however, for either hydromorphological degradation (heterophylly) or eutrophication (free-floating submerged; anchored submerged; multi- and single-apical growth meristem; leaf area). In particular it seems that a high fraction of heterophyllous species was associated with an unmodified hydromorphological environment reflected in positive associations between heterophylly and a natural planform (meandering or naturally straight), a natural cross-section and absence of weed cutting. The consistency in the response of this trait to hydromorphological alterations was also reflected in negative associations between heterophylly and a modified hydromorphological stream environment (channelised planform; channelised cross-section; weed-cutting).

Several lines of evidence exist that populations of aquatic plants that routinely experience heterogeneous environmental conditions also exhibit the greatest degree of heterophylly (Cook and Johnson 1968; Levins 1963; Bradshaw 1965). In line with this, we suggest that the higher fraction of heterophyllous species in hydromorphologically unmodified stream reaches is related to more diverse habitat conditions compared to what is found in the modified reaches. In natural sinuous and meandering streams a large variability in depth and substrate conditions exist (Pedersen et al. 2006) and this variability may intensify during summer as the flow velocity declines. Along the outside of the meander bends sediment is eroded and deposition occurs further downstream on the inside of meanders where the current velocity is lower. Depositional areas that build up during summer on the inside of meanders are suitable habitats for heterophyllous species. These species maximise resource uptake by producing submerged leaves under high flow velocities in winter and early spring and then later on during summer as the sediment build up they produce floating or aerial type leaves (Allsopp 1965; Sculthorpe 1967).

The clear negative response of the abundance of heterophyllyous species to weed cutting emphasizes that management add to habitat homogenisation. Management serves to lower the water levels and maintain the channelised cross section by removing deposition zones either indirectly by increasing flow velocity through the removal of biomass or directly by dredging. This interference may prevent deposition zones from developing in the river channel that can limit the habitat availability for heterophyllous species. Under natural conditions, the land-water zone is characterized by a high number of heterophyllous species that may also grow extensively in the stream channel provided that water depths are suitable (Riis et al. 2001). We, therefore, suggest that management may restrict diversity not only directly by leaving only species being robust to cutting (Pedersen et al. 2006), but also indirectly by preventing species from colonizing the streams from the land-water ecotone during summer.

Another very consistent pattern in our results was that the abundance of species with a high overwintering capacity (i.e. species with extensive formation of vegetative propagules such as tubers, turions and rhizomes) was positively associated with hydromorphological degradation both in terms of a degraded channel form (channelised cross section) and high weed cutting intensity. A common characteristic for vegetative propagules are that they remain dormant during the coldest seasons (Sculthorpe 1967). Species with extensive formation of propagules may therefore better survive unfavourable conditions during winter. Species with a high overwintering capacity were a highly diverse group comprising several submerged species (e.g. *Potamogeton* spp., *Myriophyllum* spp., *Ceratophyllum* spp., *Elodea canadensis*), free-floating species (e.g. *Lemna* spp., *Spirodela polyrhiza*, *Utricularia* spp.) and species producing both submerged and emergent leaves (e.g. Sparganium spp., *Sagittaria sagittifolia*, *Myositis palustris*). A majority of the species were, however, submerged reflecting that propagules also serve for propagation and dispersal in streams with limited occurrence of species being dispersed by seeds (Sculthorpe 1967).

The coupling between hydromorphological degradation and the abundance of species with a high overwintering capacity can be associated with a generally higher resilience of the macrophyte community in anthropogenic disturbed habitats. Additionally, we suggest that weed cutting may provide a competitive advantage for species with a high overwintering capacity in temperate regions. Management in terms of weed cutting and dredging may extend the growing season by improving the amount of light that reach the stream bottom (Dawson 1976; Ham et al. 1981). Dawson (1976) observed that cutting during summer increased the summer biomass next year by giving rise to a higher overwintering biomass from which re-growth could take place. In line with this, we suggest that managed streams generally can have a higher late summer biomass compared to unmanaged streams that experience a high degree of natural dieback towards the end of the growing season. The higher biomass may result in a higher amount of propagules formed which can be an advantage for propagule forming species in particular following winters with extensive dieback of the standing biomass.

As expected, we found that growth meristem characteristics of the macrophyte community were influenced by the management regime. A clear and consistent pattern was found with a higher abundance of species growing from basal meristems in stream reaches with a high weed cutting intensity and a lower abundance of these species in stream reaches without weed cutting. At the same time we found the opposite pattern for species growing from apical meristems (single+multiple). Previous studies have shown that weed cutting can have a severe influence on community structure (Baattrup-Pedersen and Riis 1999; Baattrup-Pedersen et al. 2003, 2004; Pedersen et al. 2006), and here we provide evidence that these changes are likely to be controlled by growth meristem characteristics. This finding seems intuitively logical since the position of the growth meristem is likely to determine the potential for re-growth following cutting. That is, species with basal meristems may start re-growth immediately after the intervention whereas species with apical growth meristems likely exhibit delayed re-growth. One taxa previously identified as being highly tolerant to high intensity weed cutting is *Sparganium* spp. (Baattrup-Pedersen et al. 2003). A likely reason for this could be that the leaf-producing meristems are located just above the stream bottom and therefore remains intact following cutting. Additionally, this species also has extensive rhizomes that may also provide a competitive advantage in disturbed environments (Baattrup-Pedersen et al. 2003; Wiegleb et al. 2014). Using continuous multi-year data, Wiegleb et al. (2014) noticed an increase in *Sparganium emersum* along with other rhizomatic species in response to increasing anthropogenic disturbance in German rivers when combining both hydromorphological stressors, e.g. cutting, dredging and construction work in the stream bed, and stressors associated with water quality, e.g. malfunction of sewage plants and intensification of agricultural land use. Interestingly, Sparganium emersum is also widely distributed and abundant in the least-disturbed lowland streams in Europe (Baattrup-Pedersen et al. 2008), but still subordinate in comparison to what has been noticed in anthropogenic disturbed streams (Riis et al. 2000; Baattrup-Pedersen et al. 2002; Pedersen et al. 2006; Birk and Wilby 2010; Steffen et al. 2013).

As opposed to our expectations, we did not find an increase in the abundance of species growing from apical meristems with increasing levels of inorganic nutrients. This finding was puzzling. Recently, it was shown that the abundance of species with this trait characteristic increased in response to increased agricultural intensity in the catchment of European lowland streams and it was argued that the driver for changes in community structure was light limitation giving species with an ability to concentrate the photosynthetic active biomass in the uppermost waters a competitive advantage under eutrophication (Baattrup-Pedersen et al. submitted). The apparent opposite response found here may link to differences in co-operating stressors in the streams. We found that phosphate levels were positively associated with weed cutting intensity and, consequently, streams with high phosphate levels also experienced regular cuttings. These cuttings probably set aside light as a factor controlling species composition under nutrient rich conditions (e.g. Hilton et al. 2006). Thus, plenty of light may reach the stream bottom following biomass removal and at the same time shading from periphytic algae becomes less important since it will take some time before mats develop on the new leaves being formed following cutting. This interpretation of our results also implies that the positive response observed between nutrient levels in the stream water and the abundance of species with basal growth meristems is without causality but merely reflect that species growing from basal meristems dominate in regularly cut reaches.

Interestingly, we also observed a direct and positive response between traits that we associate with species productivity (NE and a high morphology index; Birk et al. 2006; Dudley et al. 2013) and phosphate levels in the stream water indicating that nutrients played a direct role for community composition. Again, this finding is somewhat contradictory to previous results obtained in the large set of European streams mentioned above where no evidence was found that nutrients played any direct role in shaping the macrophyte communities. According to the explanation above, however, this finding can highlight that constraints associated with low light availability in eutrophic streams (Hilton et al. 2006) is relieved if these streams are cut regularly. Biomass removal and consequently, a better light climate may enable productive species that are inefficient in light capture or utilisation to compete successfully provided that they are resilient towards weed cutting.

Can indicators be developed from the trait relationships described here? As has previously been pointed out in REFORM Deliverable 3.1 the question is not whether or not a hydromorphological stressor exists as they are not invisible like nutrient inputs. The relevance of indicators is therefore somewhat different for hydromorphological stressors. Hence, the key questions are related to the kind of impact the stressor has on the ecological status, if is it the dominant impact and if it interacts with other stressors to have a detrimental impact on ecological status. In terms of the hydromorphological impacts considered here it is clear that they have a significant impact on macrophyte assemblage structure mediated through the traits of individual species. It is also clear that weed cutting for example has a dominant influence over eutrophication. The implications for a river manager hoping to restore a site are stark however, as each of the stressors considered can substantially alter the natural assemblage structure on their own or in concert. Therefore, the message is simple: to achieve good status all forms of stress must be addressed.

3.5 Conclusions

We found clear evidence that habitat degradation in lowland streams (hydromorphological alterations and eutrophication) mediate selective changes in the functional trait characteristics of the macrophyte community.

Could traits be used to distinguish between stressors?

Yes for eutrophication: eutrophication included a shift toward free-floating submerged; anchored submerged; multi- and single-apical growth meristem traits.

Yes for hydromorphology: the life form trait heterophylly was unique for hydromorphological degradation (channelised planform; a channelised cross-section; weed cutting intensity). This result probably reflect that channelisation and weed cutting, that is performed to preserve the channel profile, both contribute to homogenise the channel leaving restricted space for deposition zones suitable for heterophyllous species.

Additionally the overwintering capacity of the community increased upon degradation (channelised planform; a channelised cross-section; weed cutting intensity) and the abundance of species with basal growth meristem increased with increasing weed cutting intensity. This latter finding likely reflects that species with basal meristems may start re-growth immediately after cutting whereas species with apical growth meristems likely exhibit delayed re-growth.

Did eutrophication and hydromrophology interact in a detectable manner? Yes: we did not find an increase in the abundance of species growing from apical meristems, as predicted from theory, with increasing levels of inorganic nutrients but instead the opposite. The sites involved were also subject to weed cutting and we suggest that weed cutting can set aside light as a factor controlling species composition under nutrient rich conditions. Thus, plenty of light may reach the stream bottom following biomass removal and at the same time shading from epiphytic algae is likely to be reduced.

Each of the stressors examined had the potential to individually damage macrophyte communities, any remedial action on river systems would therefore have to address all of the stressors.

3.6 References

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4 Macrophytes - responses in Germany (North Rhine-Westfalia)

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4.1 Introduction

The Central European Lowlands (Illies 1978) cover an area from the Netherlands and parts of Belgium in the West to the Poland in the East. The southern boundaries are the Central European Highlands while the North and the Baltic Sea form natural boundaries in the north. Highly productive soils cover large parts of this area. Five to eight thousand years B.C., agricultural land use started in river valleys by small groups of people. Over time the area of farmed land increased. In the last century economic pressures have led to an intensification of agriculture. The natural woodlands, formerly present have largely been cleared.

The clearances also resulted in adjustments and transformations of the morphology of streams and rivers and their floodplains. Drainage systems have been optimised for agricultural land use in the floodplains, and straightened and deepened river courses ensure a fast effluent. The last beavers, and thus their transverse structures and habitat formations, were eradicated in the middle of the 19th century. Woody riparian vegetation has often been removed and replaced by riprap as bank fixation. Natural morphological features and flow characteristics have thus been completely changed (Baattrup-Pedersen and Riis 1999). In addition, fertilizers and fine sediments from commercial farming polluted run-off. These systems can therefore be considered to be subject to multiple stresses with channel modification and eutrophication as common interacting stresses.

Macrophytes are an integral part of life in streams and rivers. Depending on gradient, alkalinity and discharge different macrophyte communities evolve (Schaumburg et al. 2004). Furthermore, their existence depends on multiple morphological features (REFORM Deliverable 3.1; Alonso et al. 2013) while on the other hand they form and structure their own aquatic environment (REFORM Deliverable 2.1; Gurnell et al. 2014). The most diverse and abundant macrophyte communities are found in mid-sized lowland rivers (Baattrup-Pedersen et al. 2006). Those naturally meandering systems are characterized by multiple patches of diverse flow, depth and substrate conditions and with different levels of shading by woody riparian vegetation. These conditions lead to a co-existence of many species and growth forms (LUA 2001). In small streams the shading influence of woody riparian vegetation is more pronounced and a particularly important factor shaping and in parts limiting macrophyte growth and diversity (Baatrup-Pedersen et al. 2006). Furthermore, woody debris and beaver ponds create a mosaic of patches of light and stagnant and flowing conditions (Gurnell and Linstead 1998; Gurnell and Sweet 1998, Smock et al. 1989).

The study of the ecology of plants has a long tradition. In recent times several authors have tried to summarize this information into ecological profiles of plants (Kleyer et al. 2008; Kühn et al. 2004; Klotz et al. 2002; Ellenberg et al. 2001). Traits are more versatile and can integrate at the community level better than simple species lists or richness measures. Thus, traits have attracted the attention of many researchers to use as indicators of impact (e.g.

Endels et al. 2007; Verheyen et al. 2003).

In a case study the small sandy lowland streams of western Germany were selected for an in-depth analysis of the influence of anthropogenic stress from eutrophication and hydromorphological degradation on the ecological traits of macrophyte communities. Small sandy lowland streams form the majority of waterways in the German lowlands, e.g. 53 % of all river kilometer in the federal state Northrhine-Westphalia belong to this stream type (LUA 2014). Unfortunately, the vast majority has been straightened and fixed with hard materials. The floodplains were drained for intense agricultural production. Few naturally meandering streams still exist and the form of morphological impairment they are subject to varies in terms of bank and bed structure, sinuosity and riparian vegetation. In Denmark, Baatrup-Pedersen et al. (in this REFORM Deliverable 3.2) also studied lowland streams under multiple stress using plant traits as indicators. Those streams varied in channelisation and particularly weed cutting intensity. Thus, the text at hand has to be considered in conjunction with Baatrup-Pedersen et al. In comparison to the study in Denmark a larger number of abiotic variables describing the in-stream and bank structures were used. Furthermore, German sites were characterized by larger gradient in such hydromorphological features and particularly in the configuration of the woody riparian vegetation. On the other hand, data on weed cutting were not available for the German lowland streams.

However, multiple hydromorphological stresses act on the streams together with different levels of nutrient run-offs from adjacent agricultural areas. The goal of this study was to detect and describe these multi-stressor interactions on the macrophyte communities.

4.2 Material and Methods

4.2.1 Locations

The basis for the analysis was a data set of 1136 sampling sites stemming from the national monitoring system in the federal state of Northrhine-Westphalia (western Germany). The sites span over the whole federal state (Figure 4.1) comprising different stream types from small mountain streams to large lowland rivers.

The state agency classified the sampling sites into 10 different stream types (Table 4.1). The number of sampling sites per stream type varied between 3 and 382.

For an in-depth analysis of the relationship between species traits and a multi-stressor environment the stream type TRk (small, high gradient lowland streams) was chosen due to the high number of sites and its representativeness in the federal state. This stream type comprises small lowland streams with catchment areas between 1.2 and 219 km^2 (mean 31.9 km^2). A GIS shape file provided by the state agency for environment included the bed slope of the sampling sites, which was on average 3.1 ‰ (range: 0.01 to 11.4 ‰). The GIS shape is based on the situation with regard to the current longitudinal planform, which is often artificially straightened. Straightening normally increases the bed slope. Thus, bed slope has a natural as well as anthropogenic aspect.

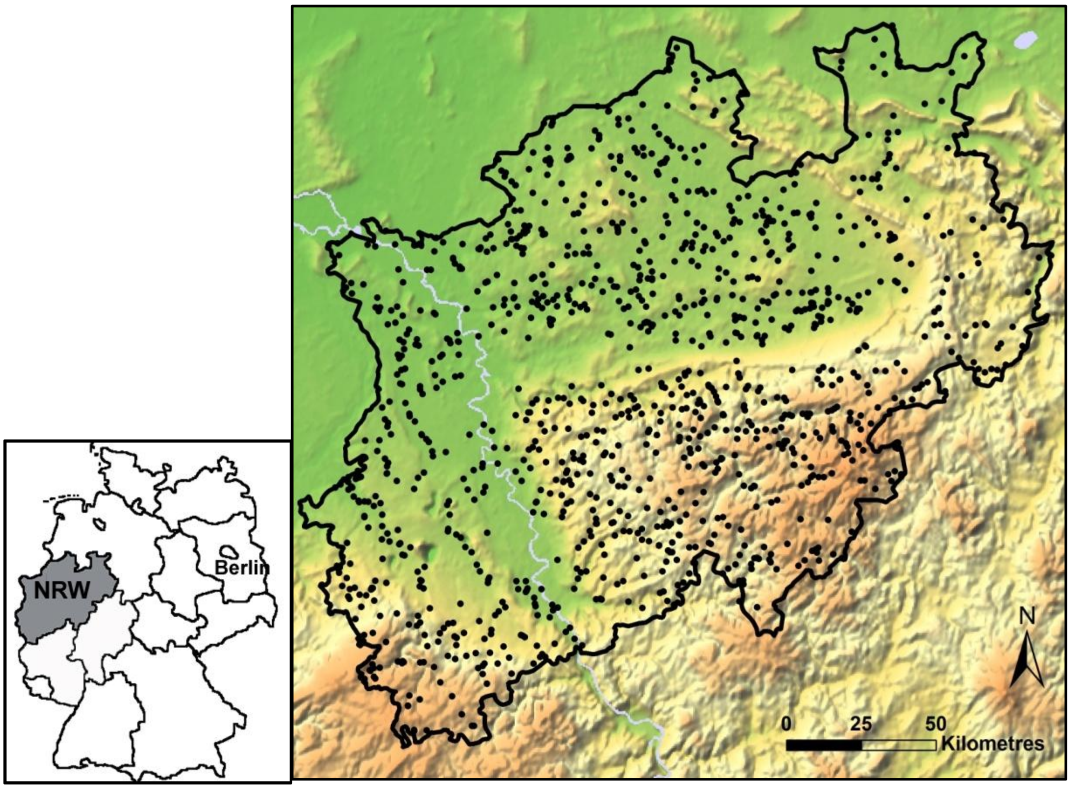

Figure 4.1 Location of the federal state North Rhine-Westphalia (NRW) in Germany and the position of the monitoring sampling sites within NRW.

The morphology of the stream type in a natural state is describes as follows (http://www.wasserblick.net):

"Strongly meandering channel in a shallow trough or wide u-shaped valley. The river is more stretched when it is groundwater influenced. Besides the dominating sandy substrates, gravel can form small but well-established patches (gravel bars); locally marl and clay occur. Important secondary habitat structures include coarse woody debris, alder roots, macrophytes and fallen leaves. These organic substrates are however subordinate. The generally shallow profile exhibits deep furrows and scoured pools behind debris dams. Slip-off slopes and stable bank cliffs are well established. Eroding bank cliffs and slightly undercut banks occur. Valley bogs and fens can occur in the floodplain." This description explicitly concerns the potentially natural status of the stream type and is hence the reference status for a hydromorphological assessment system for all reaches classified as this stream type. In contrast, the typology of Gurnell et al. (REFORM Deliverable 2.1) focused on natural processes without referring to a reference status. Thus, e.g. current planform features like anthropogenically straightened river courses are assessed as natural situation irrespective of the anthropogenic origin. Therefore, in this study the German stream typology is the means of choice, as the individual sites' deviation from the reference conditions is one of the main predictors for the analysis. The stream type TRk comprised a set of 228 sampling sites. In a first step, sites with less than two species were eliminated. The data set was further reduced to sites with a total phosphorous concentration of less than 0.6 P mg/l for a higher emphasis on hydromorphological constraints rather than eutrophication problems. The total P limit of 0.6 mg/l was chosen first to not further reduce the data set and second to analyze the influence of phosphate for this stream type and third with the idea to use in a second step of the analysis this variable as a co-variable. This lead to final data set of 181 sites for the analysis.

4.2.2 Macrophytes sampling

The macrophyte samples were taken in the summer months in the years 2006 to 2013. Following a standardized method (Schaumburg et al. 2004) a 100m-sample reach was survey for macrophytes. All species were identified to the species level and their abundance recorded according to a five-class-scale (Kohler 1978).

4.2.3 Hydromorphological recording

The morphology was surveyed according to the German standard (LUA NRW 1998; Gellert et al. 2014). 24 single hydromorphological parameters were recorded and assessed on a 7-class-scale. These were reduced to 13 parameters, which evaluate the main hydromorphological features (Table 4.2). For each parameter a score between 1 (reference) and 7 (totally anthropogenically altered) was used in the analysis. Additionally, slope, catchment size and total P were used as explanatory variables.

4.2.4 Macrophyte traits

Several trait databases were reviewed for ecological and biological information on the species (Table 5.3). Cristiana Viera, a bryophyte expert, assisted in classifying the mosses to individual traits. In total 10 different trait groups, comprising 62 different single traits were gathered for the analysis.

4.2.5 Analysis

For each trait the abundances of species classified in this trait were summed up individually for each site. Then, the trait results were divided by the number of present species of the individual site for standardization. The standardization was necessary to account for the gradient in richness, which would cause an overestimation of traits in species-rich sites. Traits, which were present in less than four sites, were excluded from further analysis. Canonical correspondence analysis was performed with the program Canoco (Ter Braak and Smilauer 2002) to identify traits, which are related to certain morphological and eutrophication parameters. Downweighting of rare traits was selected in the analysis.

Table 4.1 Stream types and number of sampling sites (N).

stream type	abbreviationnnn	N
small, low gradient lowland streams	TNk	250
mid-sized, low gradient lowland streams	TNm	92
large, low gradient lowland rivers	TNg	30
small, high gradient lowland streams	TRk	228
mid-sized, high gradient lowland streams	TRm	22
large, high gradient lowland rivers	TRg	3
siliceous, high gradient mountain streams	MRS	382
carbonate, high gradient mountain streams	MRK	115
low gradient mountain rivers	MP	11
large mountain rivers	Mg	3

Table 4.2 Hydromorphological features (according to Gellert et al. 2014) for the assessment of the hydromorphological situation.

Parameter	Abbreviation
Curvature/bends	Cur_ben
Special channel structures	Sp_cha_str
Current diversity/flow variation	Cur_div
Depth diversity/variation in depth	Dep_var
Substrate diversity	Sub_div
Bed protection	Bed_prot
Special bed features	Sp_bed_fea
Profile type	Pro_typ
Section depth	Sec_dep
Width diversity/variation of width	Wid_var
Riparian vegetation	Rip_veg
Bank protection	Ban_prot
Special bank features	Sp_ban_fea

Table 4.3 Characterization and references for the traits used.

Trait group	Trait	Abbreviation	Reference
Perenation/Plant life span			Kleyer et al. 2008
	annual	ann	
	biennual	bia	
	perennials	per	
Growth form			Den Hartog & van der Velde 1988
	16 different types		
Ellenberg value			Ellenberg et al. 2001
	light	EV_LZ	
	moisture	EV_FZ	
	nitrogen	EV_SZ	
Life form			Kleyer et al. 2008
	geophyte	LF_Geo	
	hydrophyte	LF_Hyd	
	hemi-cryptophyte	LF_HeKr	
	hemi-phanerophyte	LF_HePh	
	therophyte	LF_Th	
	chamaephyte	LF_Cha	
Leaf persistence			Kleyer et al. 2008
	summergreen	LP_sugr	
	evergreen	LP_evgr	
	overwintering green	LP_ovgr	
Floristic status			Kleyer et al. 2008
	indigenous	FS_Ind	
	neophyte	FS_Neo	
Vegetative propagation			Kleyer et al. 2008
	rhizome	VP_rh	
	rhizome-like pleiocorm	VP_rp	
	bulbil	VP_b	
	brood shoot	VP_bs	
	runner-like rhizome	VP_ar	
	root shoot	VP_ws	
	phyllogenous shoot	VP_ps	
	tuft	VP_h	
	shoot tuber	VP_sk	
	turio	VP_t	
	fragmentation	VP_fr	
	runner with tuberous tip	VP_ak	

Trait group	Trait	Abbreviation	Reference
	runner	VP_a	
Type of reproduction			Kleyer et al. 2008
	vegetative	TR_v	
	seeds	TR_s	
	mainly seeds	TR_ssv	
	seeds and vegetative	TR_sv	
	mainly vegetative, rarely seeds	TR_vvs	
Strategy type			Grime et al. 2007
	competitors	ST_C	
	competitors/ruderals	ST_CR	
	competitors/stress-tolerant	ST_CS	
	competitors/stress-tolerators/ruderals	ST_CSR	
	ruderals	ST_R	
	stress-tolerants	ST_S	
	stress-tolerators/ruderals	ST_SR	
Meristem			Baatrup-Petersen et al. (in this deliverable)
	single apical growth point	Meris_sa	
	single basal growth point	Meris_sb	
	multiple apical growth point	Meris_ma	
	single-multiple apical growth point	Meris_sm	

4.3 Results

Between two and 13 different macrophyte species (mean 4.6 ± 2.2) were recorded in the sites. In total 96 different species were identified. The most abundant species were *Phalaris arundinacea* present in 80 sites, *Berula erecta* (65 sites) and *Leptodictyum riparium* (63 sites). On the other end, 30 species occurred in a single site only and 53 species in less than four sites. For most of the species ecological traits information was available which favors a traits-environment analysis rather than species-environment analysis.

In a first step a CCA was performed to identify the influence of general abiotic parameters (slope and catchment size) and anthropogenic stress parameters (nutrients and hydromorphological stress) on the macrophyte communities (Table 4.4). Automatic forward selection was used and the eight best environmental variables were sequentially included in the model on the basis of maximum extra fit. This first CCA revealed a high influence of slope, catchment size and total P on the macrophyte traits (Figure 4.2). Hereby, slope was inversely related to the size of the catchment. This relationship follows the natural pattern of streams starting with higher gradients at the source and decreasing gradient with increasing catchment size. Morphological parameters formed the second axis with substrate and current diversity being the most important parameter. Total P was related to the hydromorphological variables and increased with increased hydromorphological stress. Furthermore, the parameter slope showed a two-sided character. On the one side it expressed the natural aspect explained above and on the other side a correlation to morphological degradation on the y-axis.

Several macrophyte traits are associated to specific environmental characteristics (Figure 4.3). On the right side the growth form elodeids and parvopotamids prevailed with increasing catchment size and relatively high total P. Some of the elodoeids are neophytes and both growth forms comprise species, which mainly reproduce vegetatively and prefer high nitrogen contents. High slope and low catchment size is preferred by the growth form helophytes and the life form hemi-cryptophyte. Those species often occur at straightened stream margins without shading. The later is expressed by the correlation to the variable riparian vegetation, which increased on the y-axis with decreasing riparian shade. In morphologically degraded sites with low substrate and current diversity the growth form myriophyllids prevailed and the life form geophyte. The bottom left part of the plot displays traits related to reference status hydromorphology. This segment showed the trait annual life span and species preferring high moisture content and in parts species which reproduce by seed and vegetatively. To overcome potential typological differences and as we were interested in the anthropogenic influence on the hydromorphology the variables slope, catchment size and additionally total P were parceled out by assigning those as co-variables. Then the CCA analysis was conducted again and the number of environmental variables was constrained by automatic forward selection to the six most important ones (Table 4.4).

The eigenvalues of the first two axes decreased but the variance of the species (in this case: traits)-environment relationship is still considerably high (55.9 for the first two axis).

In the CCA space of these solely hydromorphological variables the main explanatory axis is formed by variables describing in-stream degradation like low substrate and current diversity or artificial bed substrate (Figure 4.4). The second axis is characterized by transversal profile and bank parameters like incision depth and the configuration of the (woody) riparian vegetation.

Table 4.4 Statistics for the CCA with all environmental variables.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.071	0.031	0.030	0.011	1.570
Species-environment correlations	0.568	0.523	0.440	0.311	
Cumulative percentage variance of species data	4.5	6.5	8.4	9.1	
Cumulative percentage variance of species-environment relation	43.7	62.5	81.0	87.5	
Sum of all eigenvalues	1.570				
Sum of all canonical eigenvalues	0.164				

Table 4.5 Statistics for the CCA with all hydromorphological variables; slope, catchment size and total P were assigned as co-variables.

Axes	1	2	3	4	Total inertia
Eigenvalues	0.037	0.018	0.014	0.011	1.570
Species-environment correlations	0.532	0.339	0.363	0.321	
Cumulative percentage variance of species data	2.5	3.7	4.6	5.3	
Cumulative percentage variance of species-environment relation	37.5	55.9	70.1	81.0	
Sum of all eigenvalues	1.477				
Sum of all canonical eigenvalues	0.097				

Although only the hydromorphological variables were considered in this second analysis, the macrophyte traits revealed the same trend as in Figure 4.3. Degraded in-stream morphology was favored by the growth forms elodeids and parvopotamids and neophytes and species, which reproduce mainly vegetatively and are geophytes (Figure 4.5). In sites with low current variability and no woody riparian vegetation as well as a high section depth lemnids dominated and competitor species. More natural sites and hence traits indicating more natural situations are displayed in the left side of the CCA plot. Again, species with an annual life span were depicted and the species reproducing either with seeds or vegetatively. Furthermore, species with a multiple apical growth point preferred these sites and the species were mainly indigenous.

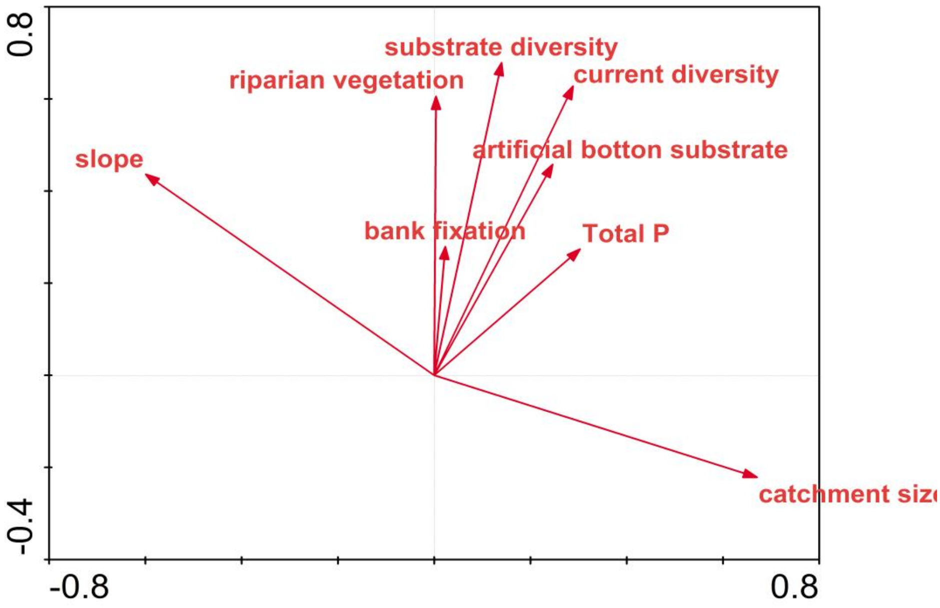

Figure 4.2 The eight most important environmental variables in the CCA.

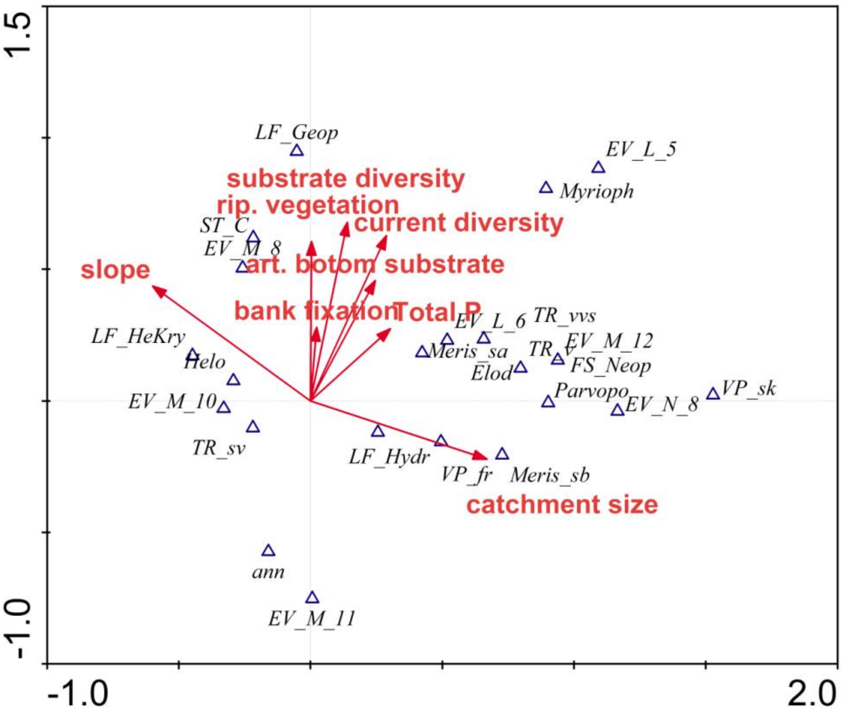

Figure 4.3 CCA plot of environmental variables and related macrophyte traits. See Table 4.3 for trait abbreviations. Only those macrophyte traits are shown which had at least a fit range of 3%.

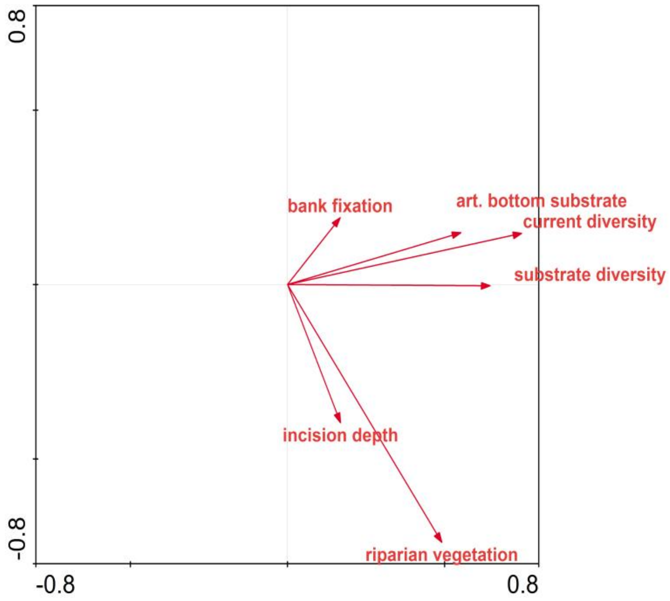

Figure 4.4 CCA plot of the most important environmental variables, which remain when slope, catchment size and total P are set as co-variables. Red arrows point in direction of degradation.

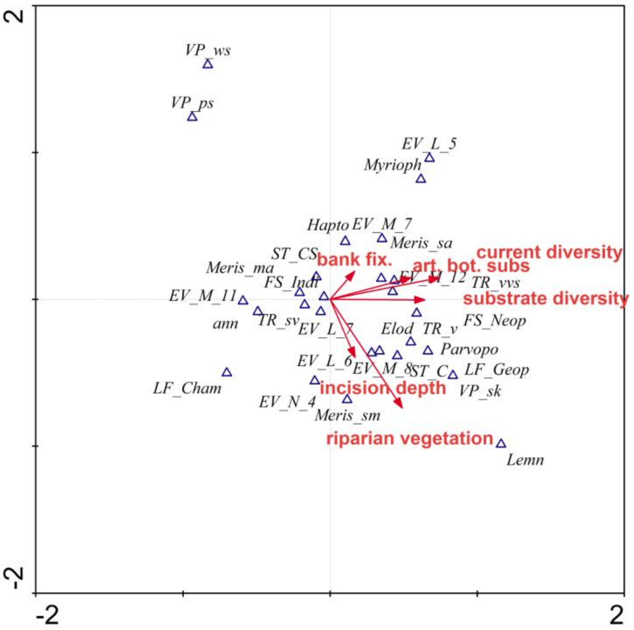

Figure 4.5 CCA plot displaying the most important hydromorphological parameters and the related macrophyte traits for the small (high gradient) lowland streams in Northrhine-Westphalia (Germany). The red arrows point in direction of degradation. Slope, catchment size and total P were set as co-variables. See Table 4.3 for abbreviations of traits. Only those macrophyte traits are shown which had at least a fit range of 3%.

4.4 Discussion

The first analysis showed that the macrophyte traits in this steam type responded to abiotic parameters and anthropogenic pressures. Catchment size and inversely slope had a very high influence while anthropogenic pressures acted contradictory to these general abiotic parameters. The mainly land use-induced variable eutrophication acted synergistically with hydromorphological pressures. Furthermore, we found that morphological degradation was not a single stressor for macrophytes and that different features of the hydromorphological alterations could not be lumped together.

There were two strong gradients: one related to channel incision and degradation of woody riparian vegetation, thus pressures acting on the river banks and the transversal profile. And the other gradient was connected with altered in-stream substrate, depth and flow diversity, thus reflecting in-stream homogeneity. Those factors if seen in a general concept go hand in hand with the impacts of industrial farming along the streams, which on top entails increased nutrient input. Thus, small lowland streams encounter multiple stresses, that are often additive. But unlike other organism groups several specific macrophytes and growth forms responded positively in terms of abundance and growth. The macrophyte communities changed from specific annual indigenous species with single apical meristems to geophytes, often neophytic species, preferring higher nutrient contents and a large amount of light. The nutrient and light availability together with homogenous flow, depth and substrate situations promoted biomass production. This, in turn was followed by enhanced management actions like dredging and mowing, which can on occasion be several times a year. Depending on timing mowing can stimulate production of macrophyte biomass in tolerant species (Dawson 1976).

Eutrophication is a widespread problem in the agricultural landscape of the German lowlands. Fertilizers, which are used in great quantity to increase production, are washed out as run-offs into the streams. In the streams phosphates increases macrophyte grows but additionally changes the macrophyte communities according to the affinity of the individual species. The traits-environment relationships unveiled the anthropogenic pressures. With increased catchment size more nutrients enter the streams and thus typical eutrophication indicators like elodeids and myriophyllids and neophytes dominated (O'Hare et al. 2006). Furthermore, those streams are subject to regular maintenance and thus species reproducing mainly vegetatively occured and those which propagate by fragments. In concordance with Baatrup-Pedersen et al. (in this deliverable) we found geophytes species (overwintering organs in Baatrup-Pedersen et al.) being present in channelized, degraded sites.

When we analytically turned this multi-stressor system of general abiotic variables, eutrophication and hydromorphological alteration into a one-stressor system focusing on hydromorphology, additional conclusions could be drawn. The in-stream diversities of flow and substrate and the riparian vegetation were the main parameter structuring the macrophyte communities. Both variable groups are associated with different responses by macrophyte traits. O'Hare et al. (2006) showed that a decrease in substrate and flow diversity was the main driver of degradation. Furthermore, homogeneity enhanced certain growth forms and traits, which in turn might be used for indication.

But, the results revealed a controversy of near-natural status and the presence of macrophytes. Nearly all macrophyte traits analysed, increased with increasing degradation. This can be explained mainly by two factors, light and substrate. Near-natural streams are considered to be at least 80 percent shaded as flowing through dense deciduous forest (LUA 1999), thus only a very limited amount of light would reach the stream and its banks. Hence, only sparse macrophyte grows would be possible. However real reference conditions for these hydromorphological conditions and their related reference macrophyte communities are not available anymore (Riis and Sand-Jensen 2001) which make these assumptions on reference and their underlying standards and their wide spread application a subject for further reviews. Given the idea that beavers shaped the landscape particularly in small streams creating beaver ponds and that dead wood needs very long time to decay in river systems (Hyatt and Naiman 2001) a patch habitat system with a variety of light and flow settings might be more a realistic situation as reference condition. Under these circumstances, a diverse but less abundant macrophyte community might establish (Ray et al. 2001: Dawson and Haslam 1983).

The diversity of metrics, which are associated with different morphological impairments and the text of Baatrup-Pedersen et al. (in this deliverable) showed the possibilities, which are inherent to macrophytes and their indication potential for multiple stresses. In addition, web-based plant trait databases expand in an increasing rate giving even more options for impact detection and assessment. Thus, this study is a promising starting point for future analysis particularly of other stream types.

Macrophyte traits have the potential to be used for anthropogenic impact assessment. Furthermore, they could be used to indicate multiple stresses. Eutrophication as well as different forms of morphological impairment can be projected by individual traits. Seen from another perspective depending on the traits and the reference situation different traits respond to individual stresses. But, the development of an assessment system for hydromorphological degradation in small lowland streams (in Germany) would probably be contrary to general assessment systems. In invertebrate, fish or macrophyte assessment system for the water framework directive (e.g. RIVPACS, PERLODES, Mean Trophic Rank, Fibs) much weight is given to the presence or absence of specific (indicator) species. Furthermore, in systems, which work on the basis of metrics normally a deviation of the reference condition is shown as a decrease. In contrast, for the small lowland streams in Germany metric values for reference or good status would be minimal or absent, while the values will increases with increasing degradation. The investigation should therefore be focused on macrophyte traits, which decrease with naturalness. Furthermore, due to the high variability and low predictability of species a specific species indicator system is not favored. Hence, this will pose a specific challenge for the development of an assessment system for the small lowland streams in Germany.

However, and in conjunction with the study of Baattrup-Pedersen et al. (in this deliverable), the vicious circle of enhanced macrophyte growth in channelized and eutrophic small lowland streams can only be broken if all stressors are tackled, the eutrophication and the multiple hydromorphological degradations.

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5 Macrophytes - responses in the UK

Authors: Matthew T. O'Hare, Iain D. M. Gunn and Claire McDonald

5.1 Introduction

Here we investigate the response of macrophyte vegetation to channel re-sectioning and eutrophication (elevated phosphorus). The combination of these two stressors is common place in the UK, especially in lowland areas which are intensively farmed. We use multivariate analysis to determine the sensitivity of macrophytes to these stressors and their ability to distinguish between the two stressors. Unlike the findings presented here from Denmark we were unable to detect a strong response at a trait or species composition to re-sectioning. We attribute this in part to the inherent difficulties in assessing impact when there is no information is available on channel maintenance, weed cutting and dredging and the lack of high quality reference sites.

Re-sectioning is where the natural cross section of a channel is remodeled to a standard trapezoid cross section. It is the basic solution applied by engineers attempting to maximise the flood conveyance capacity of a channel and is often accompanied by over widening and straightening of the channel. The channel may be embanked too and possibly re-enforced. The in-stream habitat is homogenised as the bed is flattened and any hydraulic variability is lost. As the banks are steepened, the angle at which the bank meets the channel rarely allows for the development of marginal vegetation, as this requires a more gradual transition.

Most re-sectioning work was done historically in the UK. However the channels rarely maintain conveyance capacity as the new cross section is not in balance with the system's forces and vegetation often colonises. As the vegetation reduces conveyance capacity it is either cut or dredged out (O'Hare et al. 2010a). The frequency with which this maintenance occurs is key, as it resets the community. There is evidence from the Danish study, in this deliverable, that it selects for competitive species.

Eutrophication is also widespread in the UK and is considered to have a detrimental impact on aquatic vegetation. Elevated levels of nutrients, including phosphorus are known to occur in many lowland UK systems and have been implicated with substantial changes in the macrophyte assemblages and overall biomass, although these changes are inferred from comparing sites of different P levels rather than documented cases of assemblages changing over time with increasing P levels, (O'Hare et al 2010b).

Here we ask the following key questions:

Are re-sectioned sites grouped in space and are they subject to multiple pressures?

Do re-sectioned sites support different macrophyte assemblages or traits grouping?

5.2 Methods

Macrophyte abundance information and the majority of the required physical properties were obtained from surveys of 100m lengths of river channel carried out by the Centre for Ecology & Hydrology and the Environment Agency of England and Wales (EA) using the “Mean Trophic Rank” (MTR) macrophyte survey method (Dawson et al. 1999; Holmes 1999). In addition, bankfull channel width (width) was obtained from the Environment Agency’s River Habitat Survey database (RHS, Environment Agency, 2003) of 500m reaches, by identifying sites close to MTR sites. Both the MTR and RHS are rapid surveys whose quantitative indices are based on visual estimates or information from Ordnance Survey maps rather than precise field measurement schemes.

As with other UK-based surveys, the Ordnance Survey 10 x 10 km grid squares were used as a sampling framework. For convenience, however, all coastal squares with less than 50 percent of land area above high water mark were omitted from the baseline reference site network. The original focus for RHS development was England and Wales, and a three-year sampling period was planned.

Traits

Information on the habitat preferences of species and their Ellenberg values, indicating plant preferences for system fertility and soil moisture, were taken from the PLANATT database (Hill et al. 2004).

Data analysis

Principal component analysis (PCA) was carried out to determine which environmental variables could explain the variation in macrophyte abundance across the sites. The influence of the degree of re-sectioning at a site on this relationship was also explored in the PCA plots. PCA was then repeated using the different types of species present at a site to explain the variation in macrophyte abundance across sites. The results of these analyses were used to determine which sites were directly comparable in terms of river style and the stressors that exist at the site. The data from these selected sites were used to determine which plant traits could explain the variation across the sites. All data analysis was carried out using statistical software R. Analysis code can be supplied by the authors on request.

5.3 Results

Multivariate analysis and mapping of community structure and stressor distribution in relations to re-sectioning

PCA analysis indicated that there were strong gradients in the dataset from upland to lowland sites, with strong associations between site altitude, slope, dominant substrate size, and nutrient levels (phos, phosphorus) (Figure 5.1).

Sites which were full resectioned, indicating a trapezoid channel grouped to the right of the PCA plots and were associated with other stressors such as elevated P levels but also natural parameters such as high alkalinity and low altitude.

A map of the PC axis 1 scores for the environmental variable based analysis revealed strong spatial structuring, (Figure 5.3). The sites which are full resectioned are most often found in the east of the country in lowland areas. These areas have seen massive changes in landscape drainage since historic times, with Lincolnshire and East Anglia especially strongly affected.

A species based analysis did not reveal any obvious separation of resectioned sites and non-resectioned sites suggesting other structuring influences are more important across the dataset on assemblage structure, (Figure 5.3). Axis 1 reveals a strong gradient from sites dominated by bryophyte species *Fontinalis antipryetica* on the right of the diagrams, to sites dominated by emergent species, such as *Sparganium erectum* to the left of the diagram. In the UK this shift in community structure is associated with the change from rivers which differ also in physical style, see Deliverable 2.1 for details on rivers of different style.

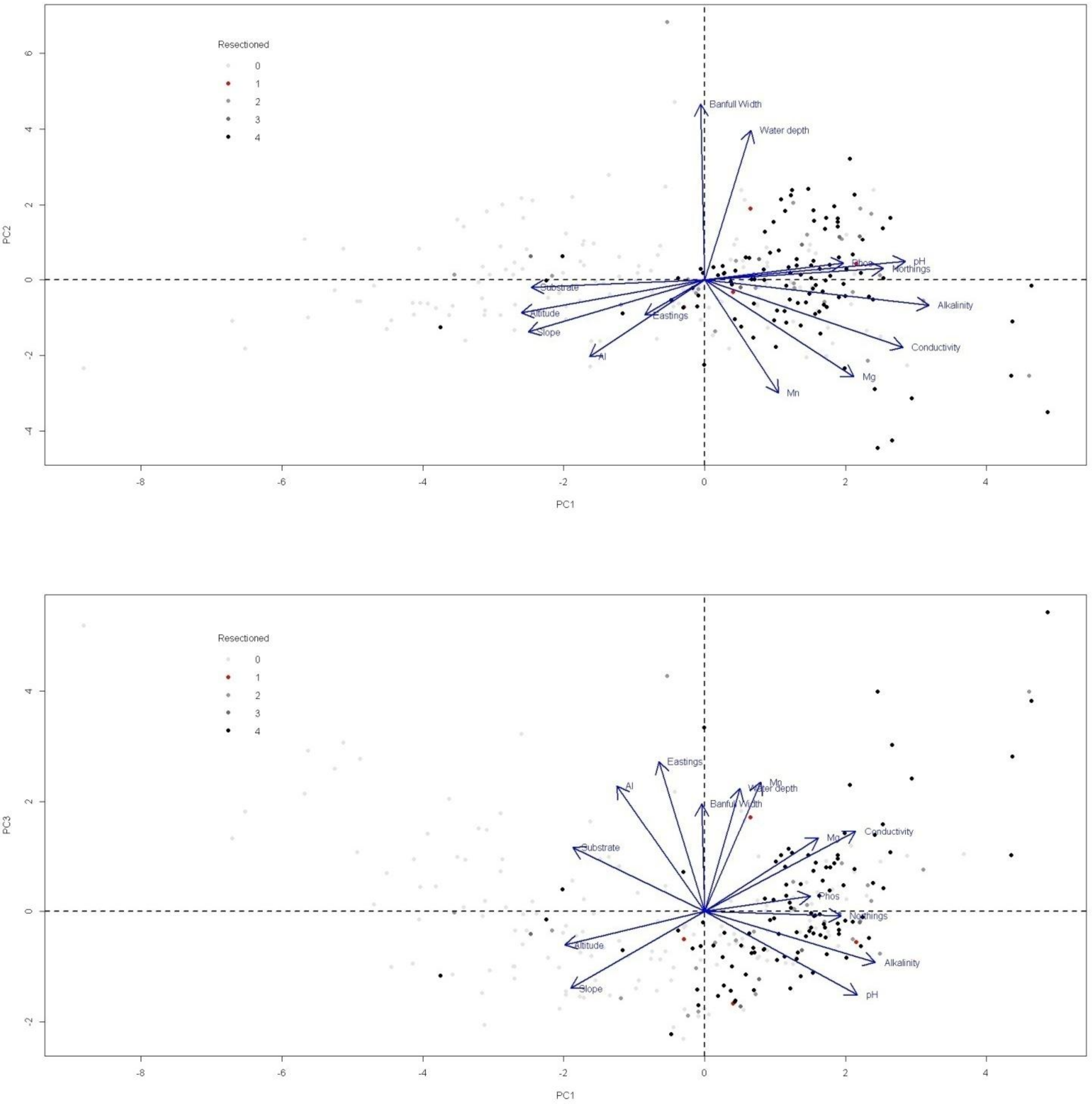

Figure 5.1 Plots of macrophyte sites included in the analysis using PC axis 1 and 2 and axis 1 and 3. The degree of channel resectioning is indicated on a 0 to 4 scale.

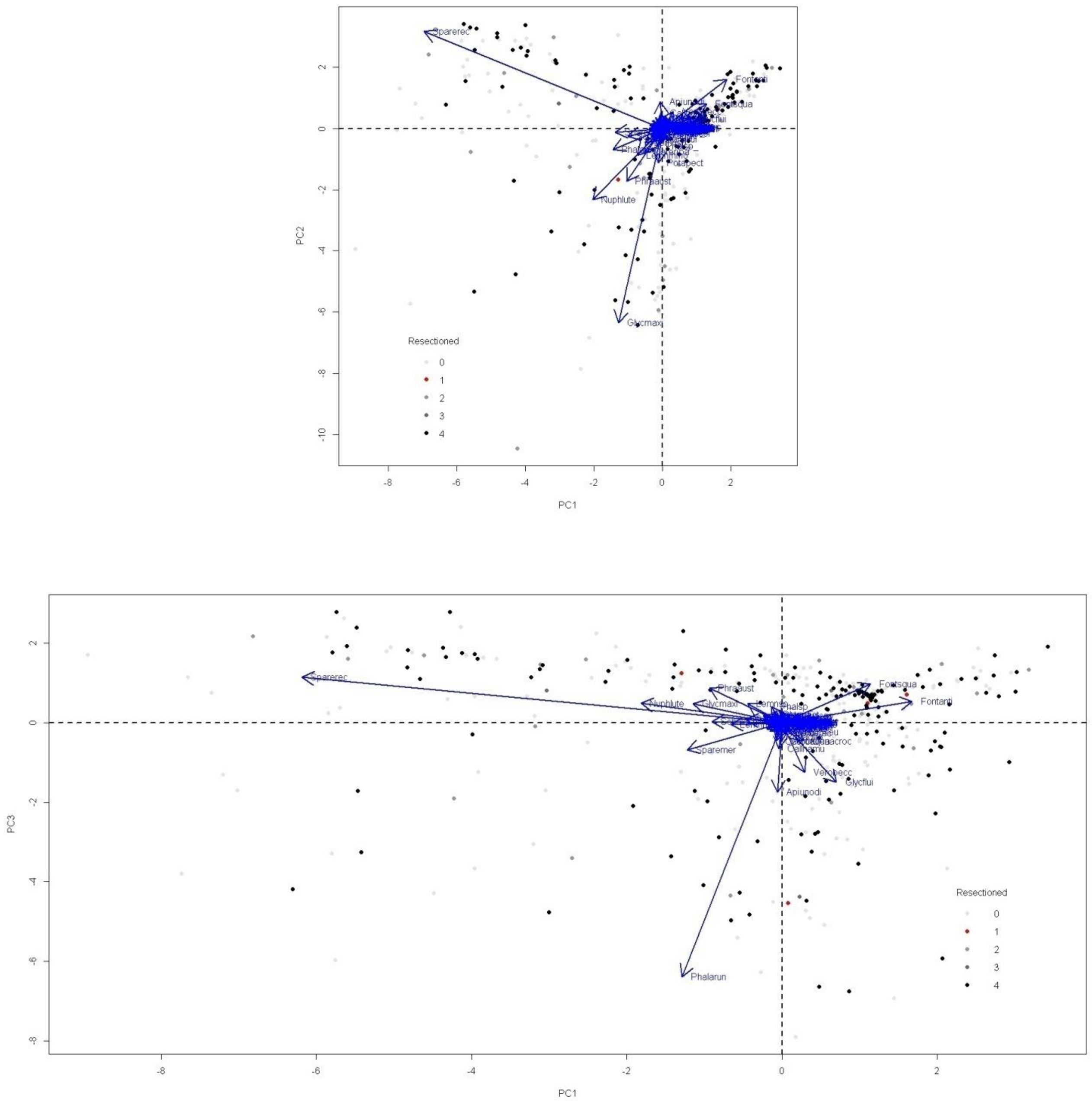

Figure 5.2 Plots of macrophyte species included in the analysis using PC axis 1 and 2 and axis 1 and 3. The degree of channel resectioning is indicated on a 0 to 4 scale.

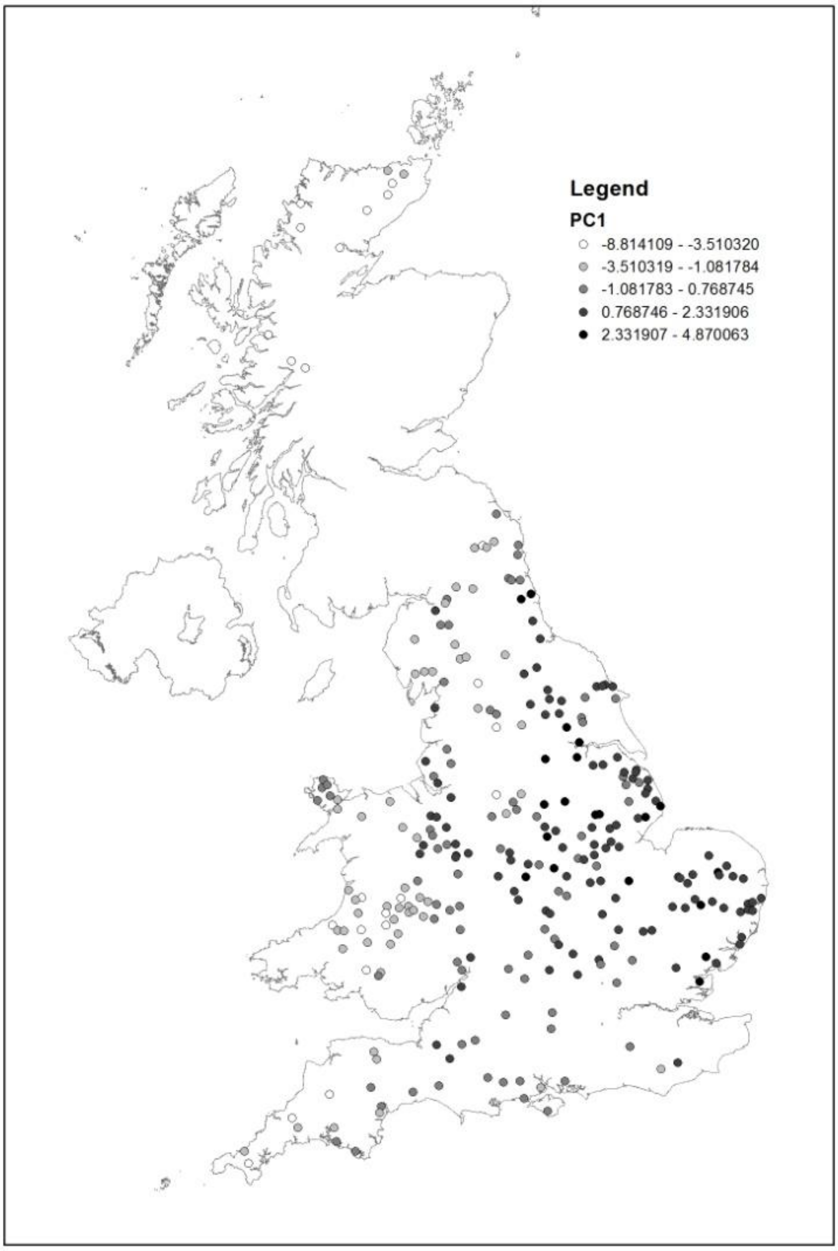

Figure 5.3 a map showing the spatial distribution of sites across the UK colour coded by their Principal components axis 1 scores from the site based analysis of environmental variables.

Traits Analysis

After the initial multivariate analysis and mapping of community structure and stressor distribution in relations to resectioning it was decided to exclude sites to the left of the origin in the PC1 versus PC2 plot of environmental variables, Figure 5.1. Sites to the left of the origin are different in river style and were expected to differ in community response to multiple stressors.

A PCA of traits was carried out, Figure 5.4. Life form and clonality scores were weighted so that instead of just presence/absence scores they were a weighted mean of the abundance of the species which were present in the categories. The first two axes only explain around 36% of the variation in the data the separation of resectioned vs more natural sites doesn't appear at all.

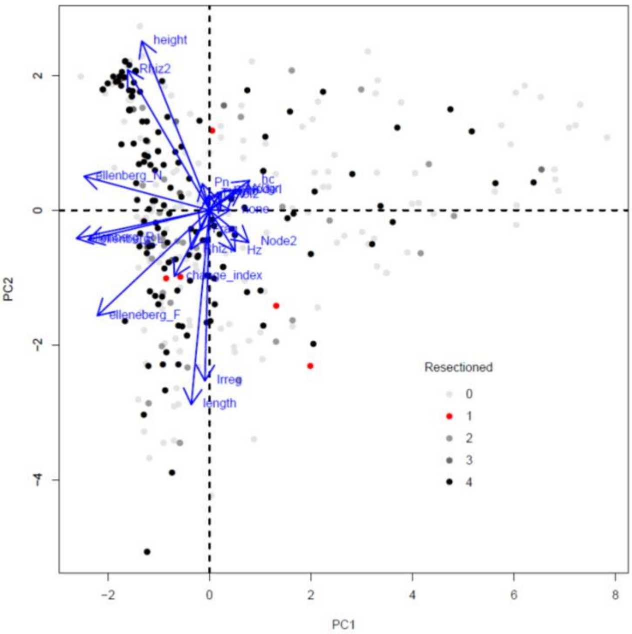

Figure 5.4 A traits based principle components analysis of macrophytes across sites. Traits are weighted by the abundance of species at the sites.

5.4 Discussion

Why was it not possible to detect differences in macrophyte assemblages between resectioned and ‘natural’ channels? There is strong evidence that macrophyte assemblages are seriously altered following major in-stream alteration and it has previously been shown this is associated with homogenisation of in-stream habitat (O’Hare et al. 2006). One likely explanation is that some of these channels have started the process of re-naturalisation where macrophyte vegetation colonises the resectioned channel and engineers the in-stream habitat creating a heterogeneous environment where a variety of macrophytes can thrive. This process leads to sites which are classified as re-sectioned supporting a variety of assemblage types which can therefore overlap in assemblage structure, with sites which have a natural channel cross section (Figure 5.5).

In lowland British rivers the succession process which follows dredging and weed cutting is well described. Over time the macrophyte assemblage changes rapidly in species richness (Wade and Edwards 1980; Wade 1993). A process of ecosystem engineering can take place where sediment is built up in-stream around early colonising rigid emergent macrophytes like *Sparganium erectum* (Figure 5.5; Liffen et al. 2013). Overwidening which is common practice in the UK, is likely to encourage colonisation by *S. erectum* as the channel adjusts and narrows (Gurnell *pers comm*.). This process of bench formation, distinct from simple succession over time, is important because it diversifies in-stream hydraulics and in places recreates marginal habitat which is missing in resectioned channels.

In the UK, the Environment Agency does record information on channel maintenance but not in a form THAT is practical to use for data analysis. Where such information is available it can be used to distinguish between multi-stressor effects on macrophyte traits, see Danish example in this deliverable for details. It is likely that if similar information were available for the UK the Danish findings could be confirmed.

Multi-stressor interactions

Within the dataset analysed, from the UK there was clear associations between eutrophication stress and channel resectioning. There was also a strong spatial pattern revealed with these sites, subject to multiple stress located in the easterly areas of the UK. In centuries past these former wetland areas were drained and are now farmed intensively, in a manner similar to that found in the Netherlands. It is likely these areas are also subject other hydromorphological stresses not represented in this dataset such as high loads of fine sediment from tilled fields.

Upland sites were less subject to high P values and resectioning. There were also clear differences in macrophyte assemblage structure between the more upland and lowland sites. This reflects fundamental differences in the quality and type of in-stream habitat available for these plants and can be related directly to physical processes (Figure 5.6; Gurnell et al. 2010). Fox (1992) has argued that physical habitat is the primary process which determines the potential macrophyte vegetation of a site, with other processes the, such as dispersal and eutrophication acting as selection factors thereafter to determine the actual assemblage found. This underpins the reasons for excluding sites which were found to the left of the PCA from further trait-based analysis. Not only were they mostly not resectioned they were also dominated by different macrophytes. Bryophyte based assemblages in particular are expected to respond very differently to multiple stress from say vascular macrophytes. Bryophytes grow very slowly and competition resulting from nutrient enrichment is unlikely to be an important process.

The dominance of *S erectum* at the sites subject to multiple stresses is consistent with previous findings which suggest it is a highly tolerant species.

Within the dataset it is difficult to identify many lowland sites which can be viewed as being in a truly natural condition. Their current condition is likely to differ dramatically from the historic condition before extensive land drainage, when many of the resectioned sites, known to be located in historic wetlands, would have had aquatic floras and been subject to traditional management which depended on floodplain inundation, (Cook 2010). Typical lowland wetland communities in the UK that are associated with river margins include: swamps and transitional fens, flood and water meadows, wet woodland of alder and willow, and more locally valley mires (Rodwell 1995). All these would have contributed to a very diverse assemblage of macrophytes. Elsewhere in Europe an acceptance of the lack of suitable reference sites has meant reference sites have been sought in other countries in the same ecoregion (Baattrup-Pedersen et al. 2008). It can be argued therefore that sites that truly represent the natural condition and which can be used as a realistic benchmark are under represented making it difficult to truly understand the impact of multiple-stressors on lowland UK systems. The near ubiquitous nature of eutrophication and resectioning across the UK lowlands also suggests that the cumulative impact of these stressors should be determined in future work.

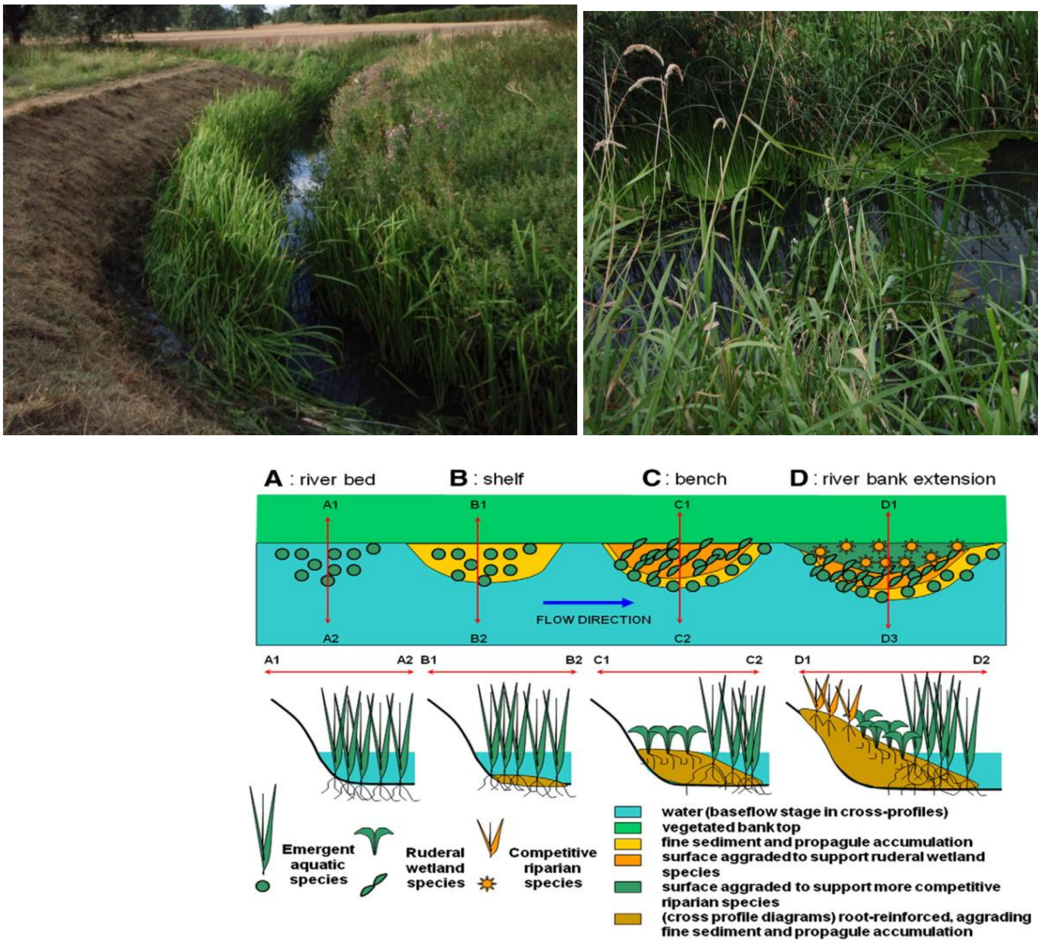

Figure 5.5 Two photographs of channels which have been resectioned. The one on the left has recently been subject to clear cutting on the bank while the in-stream vegetation has been untouched. The example on the right is of a diverse macrophyte community in a resectioned stream which has not been subject to management for many years. A diagram of a process model of bench formation in response to emergent vegetation growth, after Gurnell.

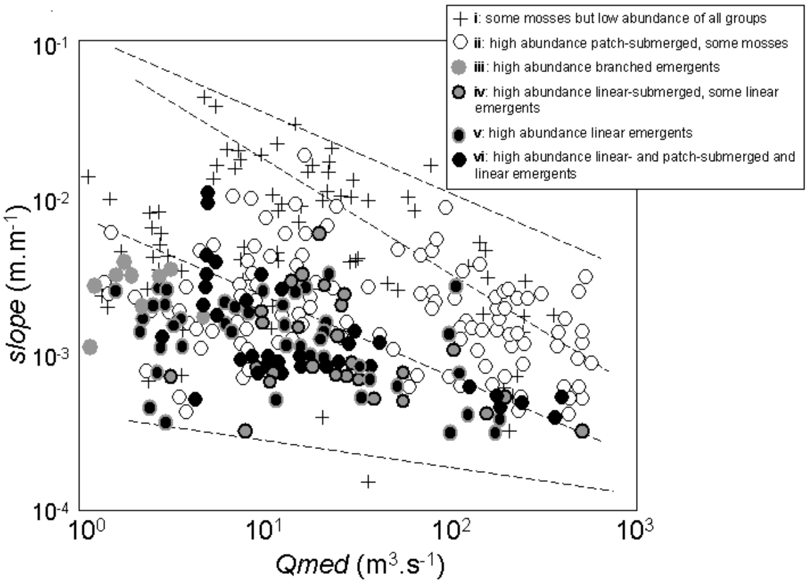

Figure 5.6 a figure depicting the dominance of different groups of in-stream macrophytes in relation to Q med Median Discharge and channel slope, at sites in the UK, after Gurnell et al 2010.

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6 Macrophytes - heavy metals in streams, Czech case study

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6.1 Introduction

Stream restoration projects that are focused on improvement of local hydromorphological conditions consider natural characteristics and anthropogenic pressures in the catchment. When considering a river restoration project where one may wish to improve physical degradation the effects of other stressors should be considered. In systems which have been subject to heavy metal pollution, the metals may be deposited in-stream and focused in particular fluvial geomorphological units, where conditions were suitable for their deposition and retention.

Therefore, if one outcome of a restoration is to alter the in-stream fluvial geomorphological units then one must appreciate how this may affect heavy metal deposition and retention within the system. The quantity and type of sediment load may interact with in-stream structures to determine the sustainability of restoration efforts in the long term. Stream substrates and fluvial geomorphological units may accumulate contaminants at different rates. Driving factors behind this differential rate are physical processes that lead to in-stream hydraulic conditions (reduction of current velocity → sedimentation), physicochemical (sorption) or biochemical (bioconcentration).The distribution of macrophytes within stream channel is influenced by similar hydraulic factors and light conditions. Macrophytes, operating as obstacle clasts, can alter streambed topography and surface-subsurface flux paths and in turn act to focus deposition of contaminants, (O'Hare et al. 2011; Folkard 2011; Gibbs et al. 2014). Stream vegetation alters current velocity conditions and enhances sedimentation in areas covered by their stands or behind them, in their wake.

River restoration projects frequently modify depth complexity, riffle/pool proportions and riparian vegetation (shading). Consequently the extent and species composition of macrophyte communities may be altered (Lorenz et al. 2012). Different channel units differ in accumulation of sediments according to local hydraulic conditions. The bank zone is usually heavily affected by channelisation and subsequently restoration strategies are targeted to change its complexity/heterogeneity to more natural conditions.

In addition, to the physical process of heavy metal deposition, there is also another mechanism by which heavy metals may be either retained in the system or enter the food chain. Bioconcentration of heavy metals in stream vegetation is described from stagnant and running waters (e.g. Martinez and Shu-Nyamboli 2011; Xing et al. 2013). Metal uptake in aquatic plants depends on the type of plant. Direct absorption from the water column to the plant surface is followed by passive or active transport across membranes and, to a lesser extent, root uptake (e.g. Rai et al. 1995). Biofilms, as a stream 'matrix' can also collects heavy metals from the surrounding solution (e.g. Drahota et al. 2014; Flemming et al. 1996; Headley et al. 1998).

Here a study is described which was undertaken as part of the planning process for the restoration of the Svratka River. The river suffers from multiple stresses, in terms of catchment land use, hydromorphological degradation and contamination from heavy metals. As the dynamics of polluted water releases are highly unpredictable, monitoring of concentrations in stream water does not provide useful information. We sampled stream fluvial geomorphological units for dissolved and particulate forms of pollutants. Within Europe, habitat specific sampling strategies are rare in routine monitoring of heavy metal contaminants. These geomorphological units represent features which are often incorporated in restoration projects (riffles, pools and macrophyte beds) and it is therefore important to consider their association with focusing and retention of heavy metals at the design stage of restoration projects. Study results would support a complex approach to stream restoration considering multiple spatial scales and interactions among environmental risks and hydromorphology.

Study aims:

- To study distribution of heavy metals in stream macrophytes and sediments associated with them
- To compare heavy metals concentrations in different stream fluvial geomorphological units

6.2 Methods

6.2.1 Study area

Sampling sites covered a 40 km long segment of the Svratka River (Table 6.1;Figure 6.1) The upper part of the river is subject to point pollution sources and channelisation, which is followed by relatively lower loads of pollutants and more nature hydromorphology downstream (Figure 6.2 Figure 6.4). Major potential industrial sources of heavy metals are located in the towns of Svratka (Cu, Zn) and Svratouch (between sites SV1 and SV2). However potential sources of heavy metals also include domestic effluents (e.g. Arsenic associated with phosphates in washing powder) and diffuse agriculture sources. The Vir Reservoir near site SV7 supplies drinking water. Anthropogenic pressures are limited in the reservoir drainage area.

A spatial and temporal study of the distribution of heavy metal across sites was undertaken at the same time as the study presented here. Full details of that study can be supplied by the authors. In summary, it found all metals exhibited a longitudinal pattern in their concentrations in sediments associated with plants. Industrial effluent entering the river between sites SV1 and SV2 is a major source of heavy metal contamination for some metals. Coniferous forest is the predominating land cover category in sub-catchments gradually decreasing downstream from 73 % at site SV1 to 43 % at site SV7. The proportion of pastures is relatively stable across all sub-catchments (14-19 %). Arable land exhibited a sudden increase between SV5 and SV6. While values at SV1-SV5 ranged between 5.1 and 9.1 %, SV6 and SV7 was 22.1 and 22.5 %, respectively. Agriculture combined with natural vegetation increased gradually along the river from 0.01 % at SV1 to 12.3 % at SV7. Artificial areas representing urban, industrial and sport areas divided sites into two groups: lower level sites (SV1, SV4 and SV5; 1.8-2.2 %) and higher level sites (SV2, SV3, SV6 and SV7; 3.3-4.4 %). Metals in aquatic ecosystems may originate from natural weathering of soil and rocks (Bradl 2005), industry and mining activities (Bradl 2005; Douglas and Beveridge 1998) and urbanisation of catchment (Ancion et al. 2013; Chalmers et al. 2007). The urbanisation of a catchment entering the Svratka River between SV5 and SV6 in combination with slow flow velocity conditions at site SV6 could explain a rise in metal concentrations at SV6. Ancion et al. (2013) reported increasing concentrations of zinc, copper and lead in both sediments and biofilm with the degree of urbanisation in the catchment area.

6.2.2 Hydromorphology

The Svratka River within the studied segment reaches a Strahler stream order of 4 and 5 (Table 6.1). The slope profile of Svratka River with location of study sites is in (Figure 6.3). The studied river stretch has relatively natural hydromorphological conditions, with river regulation occurring more upstream (Figure 6.4). Long term mean annual discharges at relevant gauging stations are 1.45 $m^3.s^{-1}$ (station Borovnice between SV5 and SV6) and 3.44 $m^3.s^{-1}$ (station Dalecin near to SV7). The hydrograph of daily mean discharge over a three year period at the Borovnice gauging station is given in Figure 6.5.

Table 6.1 Site characteristics

sites	SV1	SV2	SV3	SV4	SV5	SV6	SV7
stream km	157.9	155.6	147.7	139.4	132.9	127.2	116.7
catchment area (km^2)	32	45	64	95	134	247	369
altitude (m a.s.l.)	634	626	590	554	517	509	468
stream order	4	4	4	4	4	5	5
% arable land (211)	6.9	9.1	6.4	5.1	6.8	22.2	22.5
% artificial areas (112+121+142)	1.9	4.4	3.3	2.2	1.8	4	3.7
% forest (311+312+313)	76.3	66.7	69.7	67.8	63.8	46.6	44
pH	6.2	6.1	6.6	7	6.8	6.3	6.7
Ca+Mg ($mmol.l^{-1}$)	0.32	0.39	0.36	0.38	0.37	0.61	0.53
Ca ($mg.l^{-1}$)	9.2	12	11.3	11.5	11	18.6	15.4
Mg ($mg.l^{-1}$)	2.07	2.12	2	2.15	2.21	3.45	3.46
BOD5 ($mg.l^{-1}$)	0.6	0.8	0.8	1.8	<0.5	0.9	0.9
Total organic carbon ($mg\text{-}l^{-1}$)	7.8	8.1	8.2	7.5	6.9	7.3	7.2
Chloride ($mg\text{-}l^{-1}$)	3.1	8.7	6.3	6.7	5.8	10.3	8.3
Amonium NH4- ($mg\text{-}l^{-1}$)	0.123	0.283	0.066	0.021	0.006	0.012	0.011
Nitrite ($mg\text{-}l^{-1}$)	0.03	0.06	0.03	0.01	0.01	0.01	0.02
Nitrate ($mg\text{-}l^{-1}$)	5.36	6.26	5.35	5.34	6.03	4.07	7.02
Phosphate PO4- ($mg\text{-}l^{-1}$)	0.05	0.05	0.05	0.03	0.03	0.03	0.03
Total phosphorus ($mg\text{-}l^{-1}$)	0.035	0.048	0.037	0.029	0.023	0.031	0.034
Conductivity ($\mu S.cm^{-1}$)	94.8	132.9	114.5	118.4	118.6	186.1	163.9
Dissolved oxygen ($mg.l^{-1}$)	9.89	9.54	9.65	10.42	10.22	6.35	10.8
Oxygen saturation (%)	87.9	89.4	88.7	94.9	94.9	59.6	100.6
Water temperature (°C)	7.4	9.6	9.0	8.5	9.2	10.0	9.7

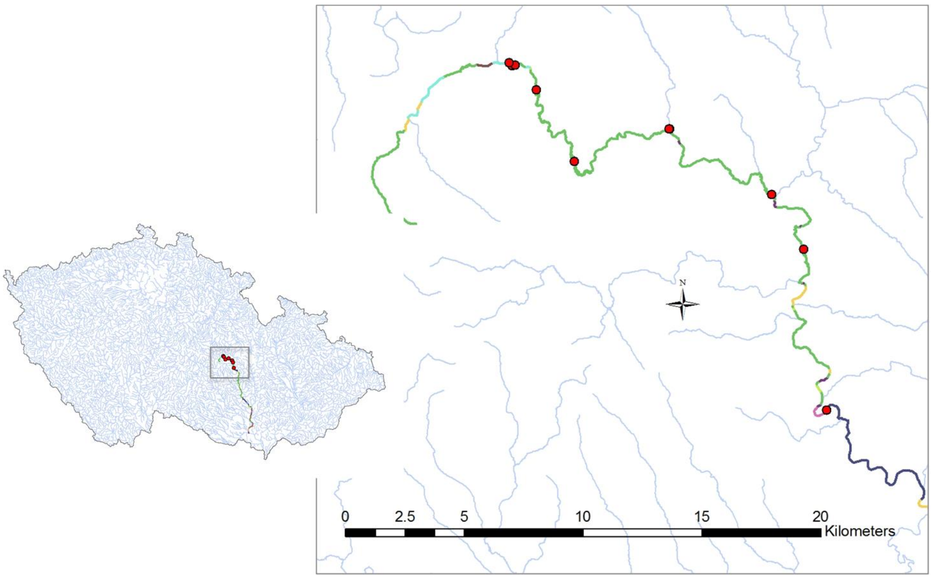


Figure 6.1 Location of study area within Czech Republic, study sites and hydromorphological types at Svratka River.

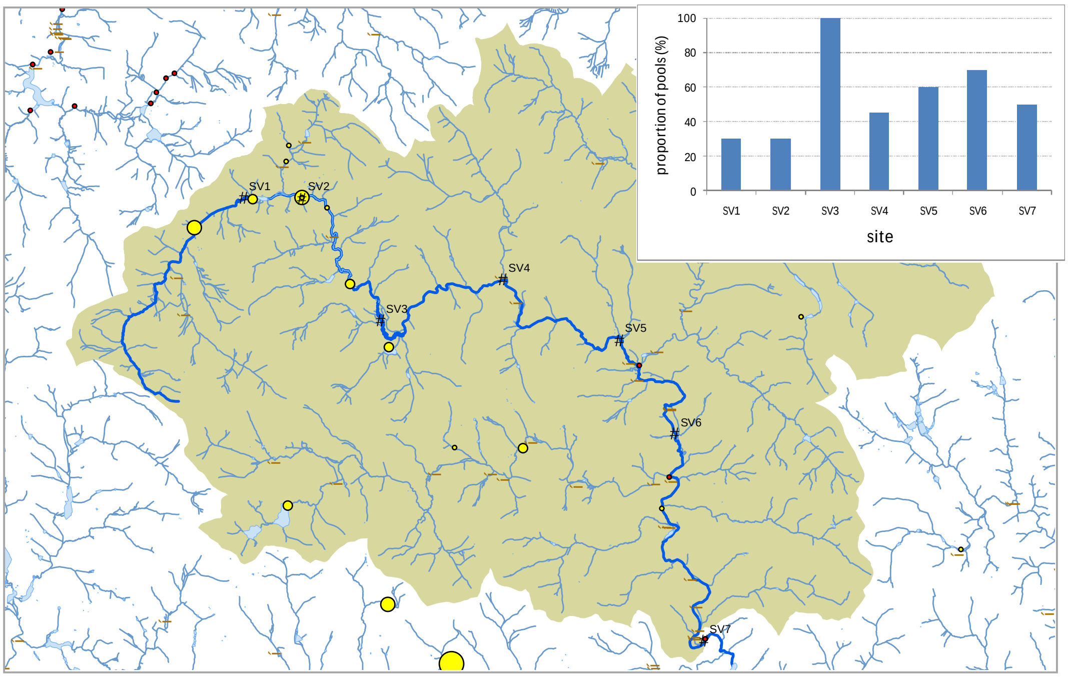


Figure 6.2 Location and magnitude of point sources of pollution in studied catchment (size of circles is related to year sum of released polluted water weighted by chemical oxygen consumption (COD); brown columns indicate position of weirs.

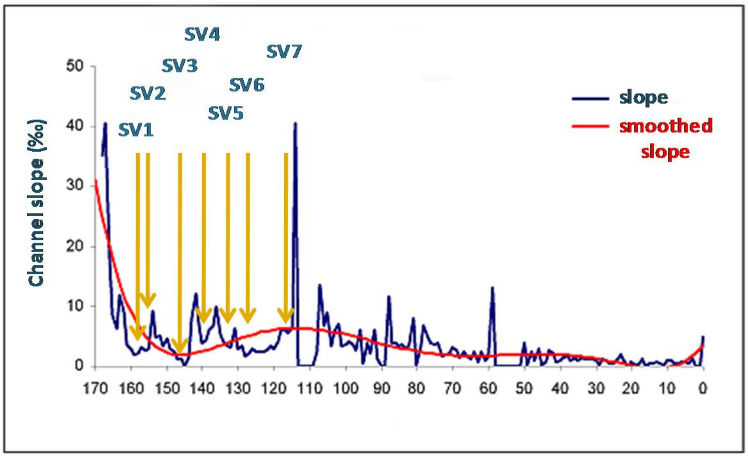


Figure 6.3 Longitudinal profile of Svratka River channel slope.

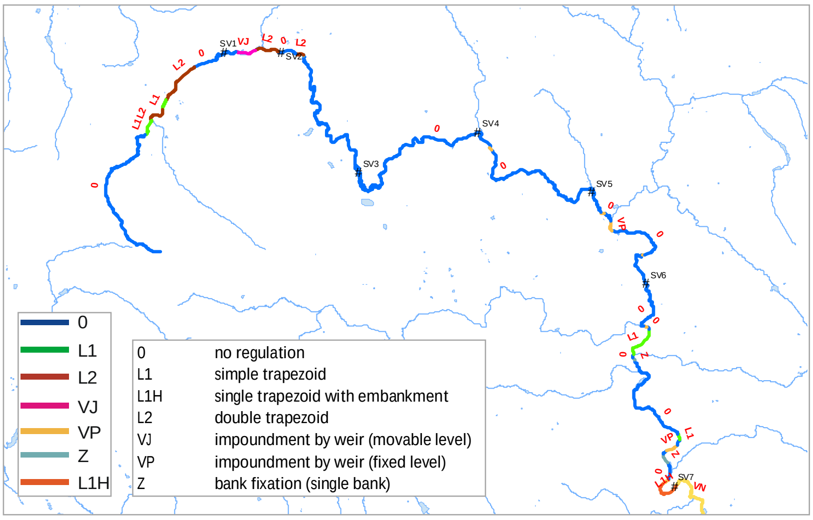


Figure 6.4 Channel modifications along studied Svratka River segment.

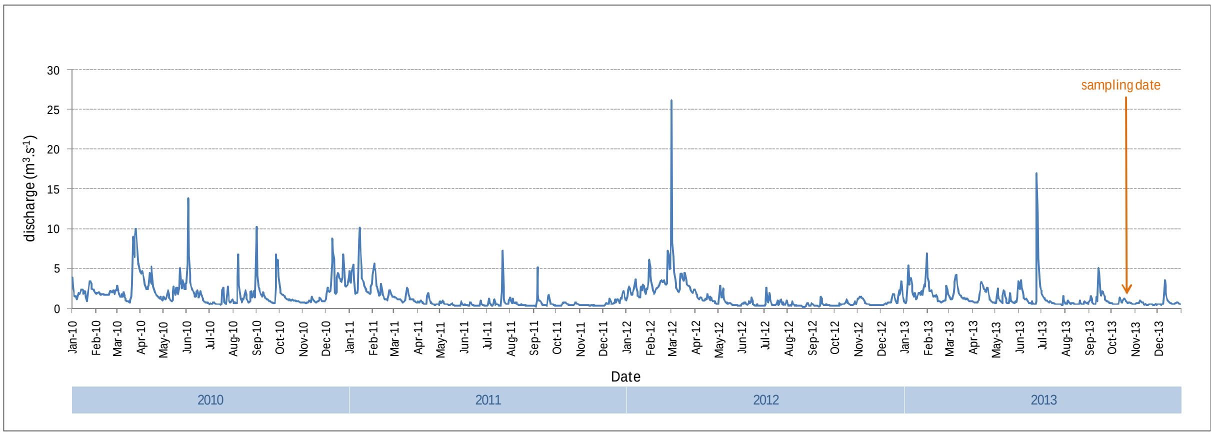


Figure 6.5 Hydrograph from gauging station at Borovnice

6.2.3 Sampling

At each of the seven sites, in the following fluvial geomorphological units, sediment and/or plant material were sampled for heavy metals, marginal vegetation, submerged macrophyte (*Callitriche* spp.), pools, and biofilm in riffles. Surface sediments were collected using a piston sampler (100 ml) from pools and from sediment deposited within or in the immediate vicinity of *Callitriche* and marginal vegetation stands, that is vegetation deposits resulting from the plants acting as an obstacle clast, see Deliverable 2.2 for details on plants as depositional units. In addition, the plants were uprooted and the roots were washed in a container with stream water to collect root attached sediments.

A total of 136 samples were collected but one sample was damaged during laboratory processing. Samples were categorized according geomorphological units and sub-matrices:

CLB – *Callitriche* body, CLR – *Callitriche* root, CLS – *Callitriche* related surface sediment, CLC – subsurface sediment washed from *Callitriche* roots.

MVB – body of marginal vegetation, MVR - root of marginal vegetation, MVS – surface sediments related to marginal vegetation, MVC - subsurface sediment washed from marginal vegetation.

PS – surface sediment in channel pools, R – biofilm in riffles, W – water.

In the laboratory, the roots were separated from the remainder of the plant body (*Callitriche* and marginal grass vegetation) and were analysed separately. Marginal vegetation was represented by *Phalaris arundinacea*. In addition, biofilm was brushed from the upper surfaces of four stones (each was approx. 4x4 cm in area).

At each sampling point, water depth, current velocity (at 40 % of depth above substrate; and also at other vertical points if possible), and substrate composition (estimation of surface coverage) were measured.

6.2.4 Sample processing and HMs analyses

Water samples were filtered through a 0.45 µm membrane syringe filter and then acidified with 2 ml of ultra-pure nitric acid per 100 ml of water sample immediately after collection.

Sediment samples were freeze dried and subsequently sieved using 1 mm mesh sieve. The sub-1mm size fraction was used for further processing. The content of heavy metals was determined in the sediment samples after aqua-regia leaching of samples according to ISO 11466 adapted to our analytical instrumentation.

Biota samples (vegetation biomass) were freeze dried and cut into small pieces using zirconium dioxide scissors. Samples were further digested in a microwave digestion system (MWS3+, Berghof) with concentrated nitric acid and hydrogen peroxide.

Selected elements were quantified in all types of samples by inductively coupled plasma mass spectrometry (7700x Agilent ICP-MS) equipped with collision/reaction technology for suppression of spectral interferences.

6.2.5 Statistical analyses

Statistical analyses were done in Statistica 12 software. Principle component analysis (PCA) was based on correlations of log-transformed data. Outlying sample SV4_R1 (biofilm at site SV4) were omitted from this analysis. Based on correlations (Pearson), we selected some heavy metals for the PCA and others were displayed in chart as supplementary variables (Figure 6.8).

The Mann–Whitney test (non-parametric) was used to identify significant differences in metal concentrations among fluvial geomorphological units. Spearman rank correlation coefficients (r_S) and a Mann-Whitney and Kruskal-Wallis test were performed on non-transformed data. A Statistica macro was used for non-parametric post hoc testing following Kruskal-Wallis analysis.

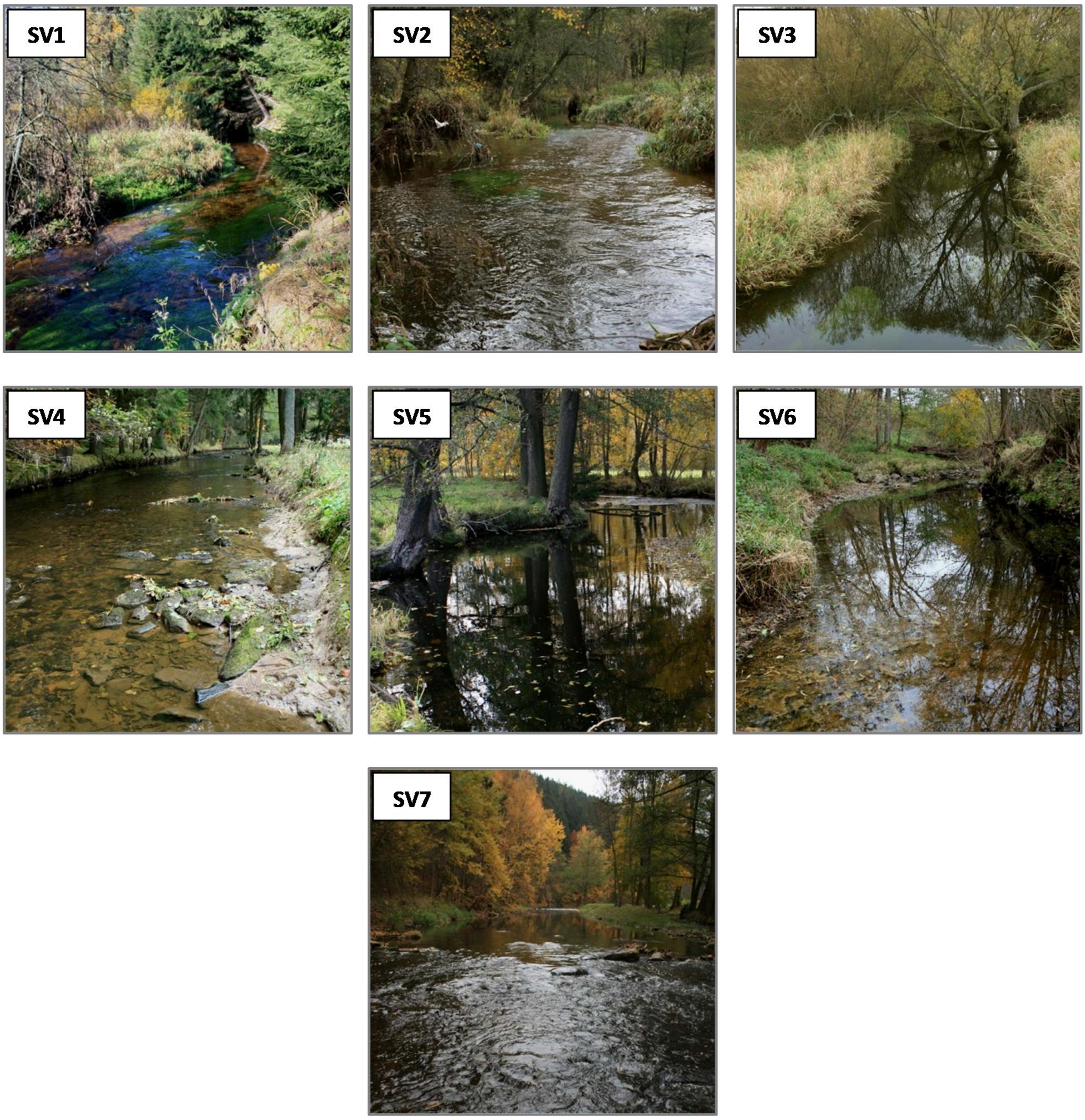


Figure 6.6 Photographs of studied sites SV1-SV7.

6.3 Results

Are heavy metals accumulates associated with fluvial geomorphological units?

We compared metal concentrations in surface layer vs. deeper sediments surrounding plant roots, surface sediments in channel pools vs. sediments accumulated behind plants (*Callitriche* and marginal vegetation).

The highest concentrations of many studied metals were found in the roots of the submerged macrophyte taxa *Callitriche*, and/or in sediments washed from their roots (Figure 6.7).

There were significantly higher concentrations of the majority of the studied metals in sediments washed from roots of both *Callitriche* and marginal vegetation than in surface sediments accumulated behind the plants; all differences were significant (Mann-Whitney test, $p<0.001$ $N=12$), except Cu in *Callitriche*-related sediments and As and Zn in sediments related to marginal vegetation (both $p<0.01$). No significant differences were found for Pb.

The metal concentrations in surface sediments taken from pools differed slightly from those in surface sediments which accumulated behind plants we found weak difference only (Cr, Ni and V was $p<0.05$). All others comparisons were not statistically significant.

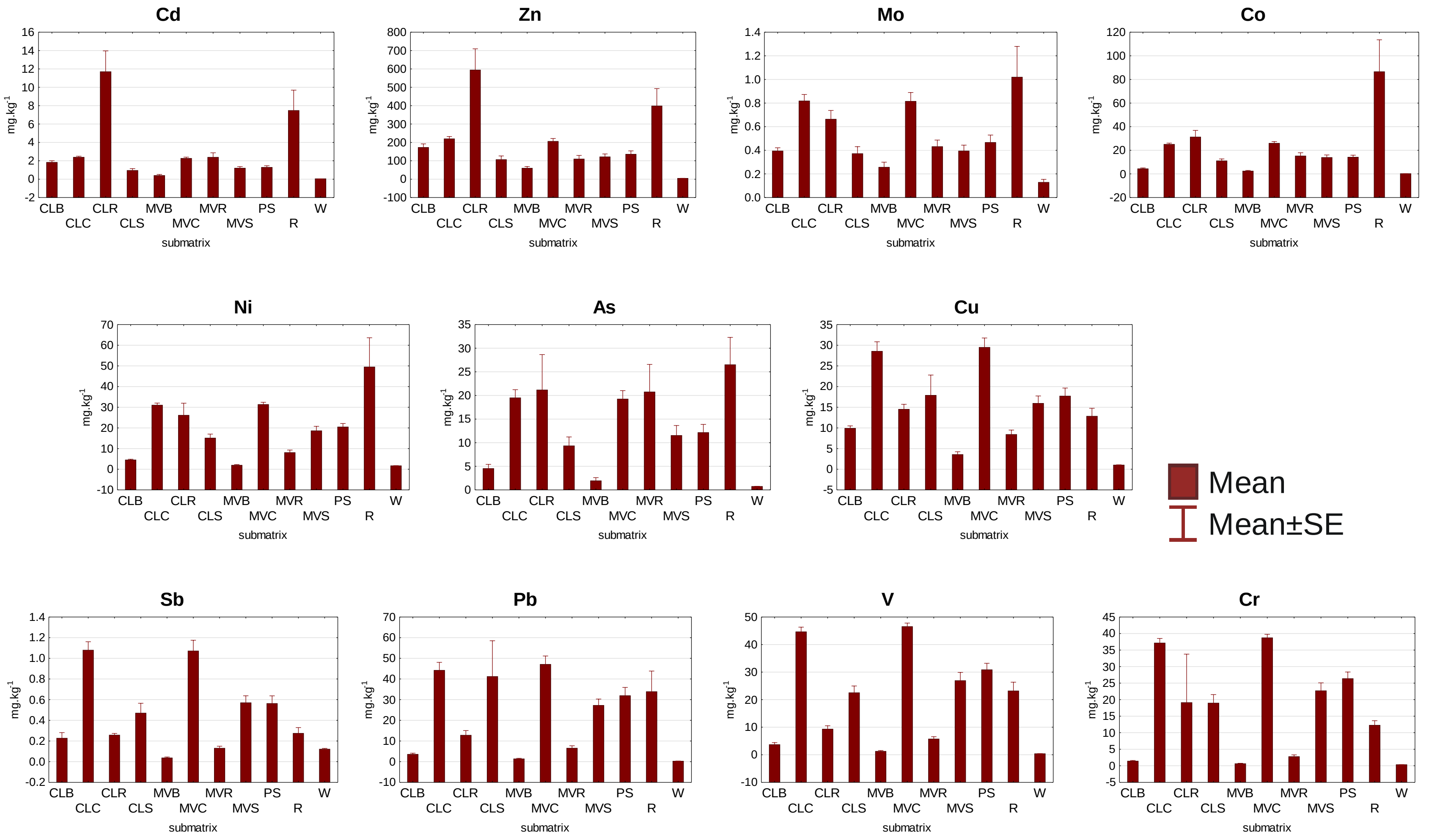


Figure 6.7 Mean concentration of heavy metals in fluvial geomorphological units (all sites combined). CL = the submerged macrophyte taxa *Callitriche* spp.; MV = marginal vegetation; PS = pool sediment; R = riffle biofilms and W = water. For the vegetation categories (B = shoots, R = roots, S = plant related sediment, C = sediment washed from plant roots).

Are there consistent patterns across sites?

Principle component analysis (PCA), based on correlations among metal concentrations in all of the samples, identified some main components that explain a large portion of the variance. The first two PCA factors explained 94.3 % of model variance (Figure 6.8). The first component (83.2 %) was mostly related to concentrations of Cu, Pb and Co. Two other metal groups diverge from this first component: a group represented by Cr and also including Ni and As; another group of interrelated metals is Zn, Cd, Co and Mo (also related to Cu).

When the samples scores on factors 1 and 2 were plotted (Figure 6.8) groups of samples related to different fluvial geomorphological units were differentiated in a diagonal direction across the plot that was related to zinc concentration (and related metals). Separation of sediment samples from plant biomass samples can be explained by difference in Cr-related metals (Figure 6.8).

Do plants bio-accumulate metals?

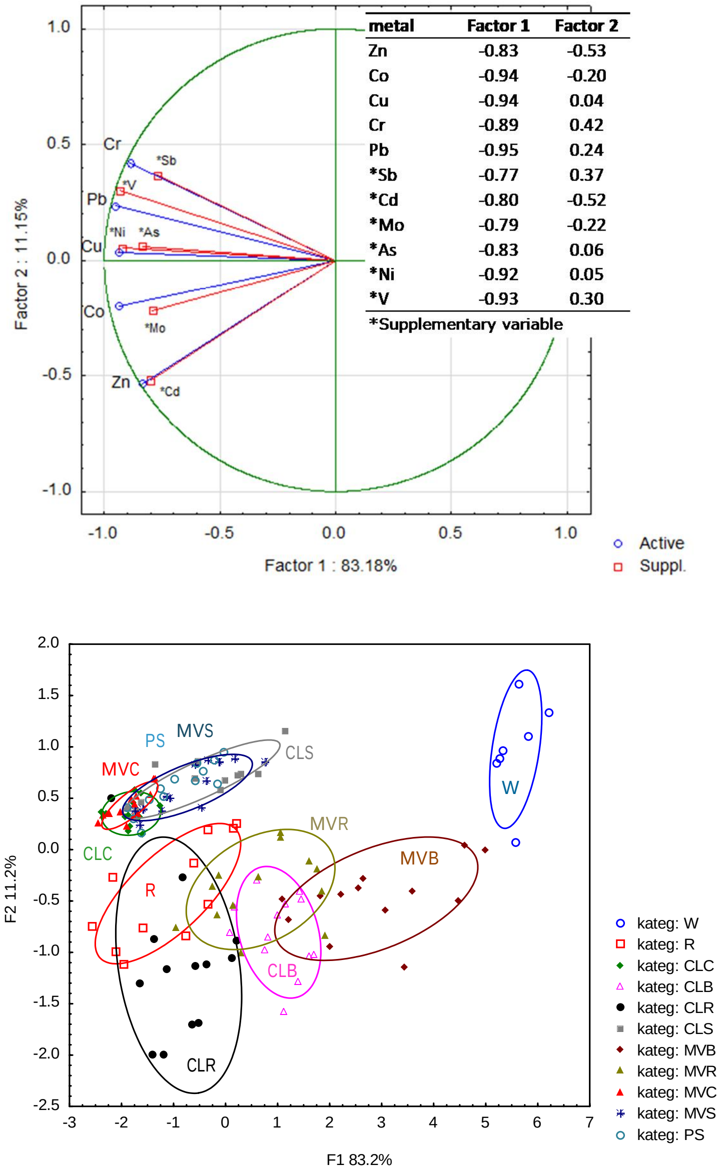
Concentrations in roots were significantly higher than in plant shoots (As, Cu, Mo $p<0.01$; all others $p<0.001$;Figure 6.9, Figure 6.10), except Sb which was non significant; r_S, N=13).

Both roots and shoots of *Callitriche* contained more heavy metals than marginal vegetation. In roots very significant differences were found for Cd, Ni, Sb, Zn (M-W test $p<0.001$; N=13) and no difference exhibited As and V. In aboveground parts of plants the concentrations of Cd, Cu, Ni, Sb, Zn (M-W test $p<0.001$; N=13) in *Callitriche* greatly exceeded those in marginal vegetation.

For example, the highest zinc concentrations were found in Callitriche roots, dominating not only *Callitriche*-related fluvial geomorphological units but also showing the maximum within all studied fluvial geomorphological units (Figure 6.9). The translocation factor for *Callitriche* (concentration in shoots / concentration in roots; TF), ranged between 0.11 and 0.63 (mean 0.35), whereas the mean bioconcentration factor (concentration in plant / concentration in water; BCF), was 4.9×10^4 for shoots and 18.2×10^4 for roots.

All heavy metals exhibited significantly higher concentrations in roots of marginal vegetation than in shoots (Mo, Zn $p<0.05$; all others $p<0.001$;Figure 6.8). The above-ground part of the marginal vegetation contained a significantly lower amount of heavy metals than was observed for *Callitriche* (Cd, Cu, Ni, Sb, Zn $p<0.001$; As, Cr, Mo, Pb, V $p<0.01$; Co $p<0.05$). Lastly, the mean BCF of arsenic was 5.8×10^3 for shoots and 2.7×10^4 for roots.

Statistically significant relationships among metal concentrations in *Callitriche*-related fluvial geomorphological units were more frequent than in marginal vegetation (Table 6.2). Concentrations in *Callitriche* shoots were related to those in the roots (Co, Cr, Pb, Zn), and to sediments washed from roots (As, Cr, Cu, Pb, Sb). Metal content in surface sediments (CLS) was related to all other *Callitriche*-related fluvial geomorphological units for As and Cu, to the roots for Mo, Sb and V, and there was no significant relationships among *Callitriche*-related fluvial geomorphological units for Cd and Ni.



metal	Factor 1	Factor 2
Zn	-0.83	-0.53
Co	-0.94	-0.20
Cu	-0.94	0.04
Cr	-0.89	0.42
Pb	-0.95	0.24
*Sb	-0.77	0.37
*Cd	-0.80	-0.52
*Mo	-0.79	-0.22
*As	-0.83	0.06
*Ni	-0.92	0.05
*V	-0.93	0.30

*Supplementary variable

Figure 6.8 PCA plot of samples marked by fluvial geomorphological units (elipses included 95 % of points). CL = the submerged macrophyte taxa *Callitriche* spp.; MV = marginal vegetation; PS = pool sediment; R =riffle biofilms and W = water. For the vegetation categories (B= shoots, R= roots, S=plant related sediment, C=sediment washed from plant roots).

Table 6.2 Significant correlations (Pearson r, N=13; p<0.05) among metal concentrations in different fluvial geomorphological units related to stream vegetation (B=plant shoots, R=plant roots, S=plant related sediment, C=sediment washed from plant roots).

Metal	Marginal vegetation	Callitriche
As	B x R	C x B; S x all
Cd	S x R	-
Co	C x R	B x R
Cr	-	B x C; B x R
Cu	B x R	C x B; C x R; S x all
Mo	-	R x S; C x R
Ni	-	-
Pb	-	C x B; B x R; C x R
Sb	C x R	B x C; R x S
V	-	R x S
Zn	B x S	B x R; C x S

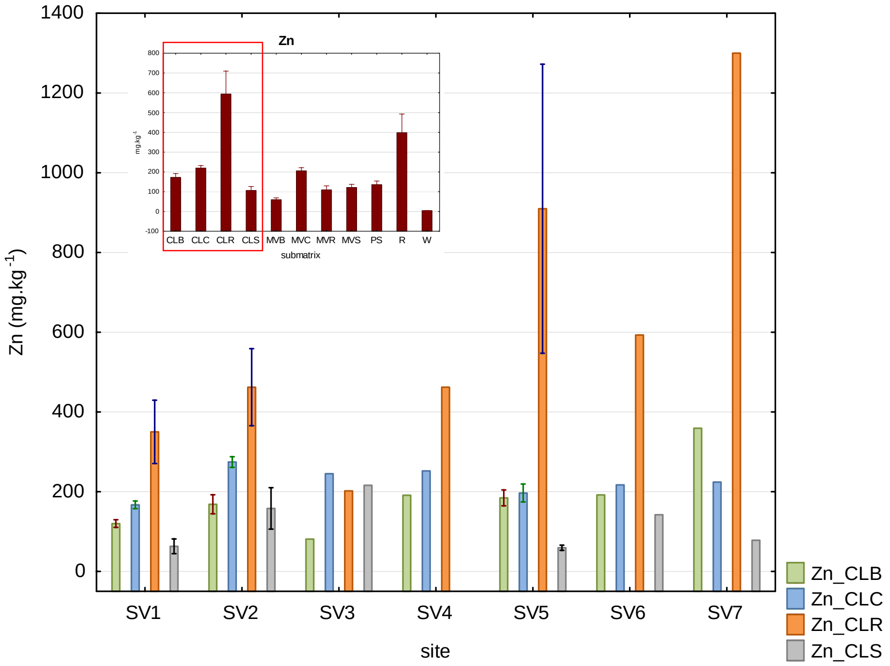


Figure 6.9 Zinc concentration in fluvial geomorphological units related to *Callitriche* at studied sites (small chart shows summarization across all sites); means±SE.

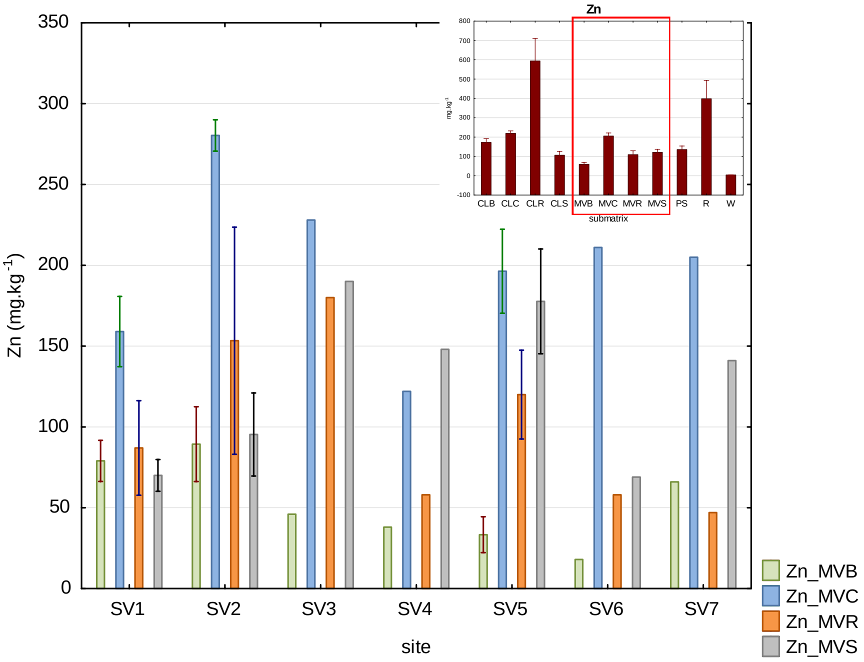


Figure 6.10 Zinc concentration in fluvial geomorphological units related to marginal vegetation of all studied sites (small chart shows summarization across all sites); means±SE.

6.4 Discussion

The majority of studies dealing with heavy metal contamination are based on sediment samples or biofilms covering larger substrates that dominate in glide and riffle habitats of streams (Mages et al. 2004; Ramelow et al. 1987). In recent years, many studies have been conducted on the heavy metal accumulation by aquatic plants, (e.g. Brooks and Robinson 1998; Rai 2009; Robinson et al. 1995; Robinson et al. 2006). Our results confirm the importance of looking at all of these potential sources to gain a more comprehensive picture of heavy metal retention within a stream system. The strong links between fluvial geomorphic units, vegetation and heavy metal deposition suggest a clear understanding is required before planning a restoration project. In Deliverables D2.1 and D2.2 of REFORM the processes by which fine sediment and, by inference, heavy metals may deposit around vegetation are described and placed in the context on individual river styles. Before any physical restoration of the study river is under taken a prudent course of action would be to evaluate the changing distribution and frequency of the different geomorphic units. The findings from our case study are not unique. Quantifying the accumulation and retention of sediments by in-channel vegetation in urban rivers has been demonstrated to be of importance in understanding metal contamination and in the planning and management of urban river restoration schemes (Gibbs et al. 2014).

A more thorough understanding of the interactions between processes is clearly required for understanding the broader application of the findings to other rivers that also suffer from the same multiple stresses. For example, further research is needed to understand why sub-surface sediments contained more heavy metals than surface ones. This finding is most probably related to sub-surface conditions than to any association with plant roots. Further sampling of sub-surface sediments in un-vegetated areas in channel pools could clarify this relationship.

How the key process of bioaccumulation interacts with biofilm or plant preferences for fluvial geomorphic zones also requires further attention. The results presented here are in line with previous studies where aquatic plants (algae, bryophytes and angiosperms) are considered indicators of trace element pollution (e.g., Cenci 2000; Markert, 1993). Some species of submerged macrophytes such as *Callitriche stagnalis* and *Myriophyllum propinquum* have revealed a high potential to accumulate As (Robinson et al. 2006). Bioconcentration factors reported by Favas et al. (2012) for arsenic in *Callitriche hamulata* and water (mean BCF 5.9 x 10^3) corresponded with our results in Callitriche shoots (mean BCF 5.8 x 10^3). However, high variability of BCF values in both studies indicated the potential influence of other factors; possibilities include pH, oxygen saturation and hydraulic conditions. A possible source of variation could also be the combination of adsorption and absorption processes that were not differentiated in this study, nor in Favas et al. (2012).

Ladislas et al. (2012) identified correlation between cadmium and zinc concentrations in the roots of the rooted plants. We found such relationships most significant in surface sediments (CLS, MVS and PS; R^2 ranged 0.87-0.95), biofilm (R; R^2=0.9), *Callitriche* roots (CLR; R^2=0.77).

There is a strong body of evidence that microbial biofilms on mineral substrates and the growth of periphyton influence heavy metal retention (e.g. Bradac et al. 2010). Zinc and copper are frequently enriched in biofilm compared to sediments (Ancion et al., 2013). Kohusova et al. (2011) identified correlations among contents of heavy metals in river sediments and biofilm. Biofilm predominated among studied fluvial geomorphological units in content of Cd, Zn, Mo, Co, Ni and As. Our results broadly support the hypotheses that metals may accumulate in biofilms, for some metals; comparing values observed by Meylan et al. (2003) in biofilms of streams historically contaminated by metal industry and our results we found similar ranges of zinc (183- 1504 mg kg^{-1} dry wt and 77-1200 mg kg^{-1} dry wt respectively. In terms of copper concentrations we found lower values.

In conclusion, the case study at the Svratka River demonstrates the importance of comprehensively assessing multi-stressor interactions during the planning phase of a restoration project. In addition, to the study presented here, a spatial and temporal assessment of heavy metal deposits was carried out across sites and an assessment of catchment land-use pressures. All these studies have provided useful data for understanding both the drivers of multiple stresses and how to stresses are interacting in-stream.

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7 Linking fish distribution to hydromorphological stress - method development

Authors: Alonso C. and R. A. Noble

7.1 Introduction

This chapter describes the development of a method which produces conceptual models that link pressures —> processes —> responses of fish species. It combines output from D1.2, that used an analysis of peer reviewed journal papers to link hydromorphological (HYMO) pressures to hydromorphological processes, with output from D3.1, that used a large European set of field data to statistically link pressures to the occurrence of fish species. The process is an efficient means of generating standard conceptual models for many species. We discuss in outline the steps required to take the process from its novel form, as presented here, to a robust and reliable tool.

According to the DPSIR (Driver, Pressure, Stress, Impact, Response) scheme, stresses are caused by pressures that change processes. Currently these alterations do not come in isolation, and the usual altered conditions include several pressures acting simultaneously on a set of processes. Pressures can be of many types, but here only hydromorphological pressures will be considered. Which HYMO pressures induce a response in a given fish species was addressed in D3.1 How HYMO processes are altered by this HYMO pressures was discussed in D1.2. However, as Deliverable 1.2 concluded, there is a great number of studies that address the effects of HYMO pressures on biota, but there is a lack of observational reports on the mechanisms through which pressures influence hydromorphological processes, and how process alterations affect HYMO variables, and, finally, how these disturbed HYMO variables modify biological metrics (García de Jalón et al. 2013). Therefore, the links between processes and fish community response has to be deduced from what it is already known.

A practical accounting of the multiple-stressor condition is to link all the HYMO pressures that are found to significantly influence the probability of fish occurrence (D3.1) and all the changes that these pressures induce on HYMO processes (D1.2). Therefore, this study focuses on HYMO stresses interacting with one another. If the path that links HYMO pressures->HYMO processes->fish responses is known for every HYMO process and species, then the HYMO stressor that is inducing an observed response can be estimated in a multiple stress environment

The relationship between HYMO stressors and fish is twofold: it depends on the nature of the stressor but also on the use that fish make of their habitat. The mechanistic understanding of the responses of the fish community to HYMO stressors needs to consider how fish perceive their habitat. This perception is scale dependent. At the micro-scale (100 m^2 - 10^4 m^2), physical habitat is perceived by fish as a combined array of depth, velocity, temperature, substrate and turbidity. The spatial distribution of these factors along the river reach and its dynamics through the year define how habitat is perceived at the meso-scale (10^4 m^2 - 10^9 m^2). The size, shape and structure of the river network define the habitat at the macro-scale (10^9 m^2 - 10^{14} m^2). At the micro-scale, the trade-off between the energetic cost of maintaining a position and the incomes (in terms of food and shelter) that provides that position food and shelter during a short time, is what determines whether a given individual of a given species occupies a particular position. At the meso-scale, this trade off along a longer period of time determines the species composition of the fish community. And at the larger scale, climate and biogeography is what determines species distribution ranges. Fish can therefore respond to processes at a variety of scales. It is appropriate to ask the key question of this deliverable at the meso-scale; 'How do species respond to environmental stressors?' (McCluney et al. 2014). It is the scale at which the hydromorphology of riverine ecosystems has been typically studied (Poole 2010). It is, therefore, expected that the fish community species composition will be sensitive to stressors acting at the reach scale.

In this chapter, the responses of the fish community to multiple HYMO stressors are explored and expressed by means of conceptual schemes. These schemes include the processes that are altered by HYMO pressures, which, in turn, affect the physical habitat as it is perceived by fish at the community (meso) scale. The results of these conceptual models are synthesised according to the gradient in fluvial conditions characterized by Huet's river zones. Models for two species whose ecology is very well understood are explored in some detail to test the veracity of the individual models and highlight any limitations. They are: brown trout (*Salmo trutta*), and tench (*Tinca tinca*). They represent species with preferences for very different river zones, according to Huet's zonation.

The main hypothesis to be tested is: fish community composition responses to hydromorphological pressures through hydromorphological processes can be tracked in sequential schemes. Therefore, fish communities can be used to assess and predict impacts of hydromorphological alterations on the ecological status of rivers. We also ask if species with particularly narrow tolerances to physical habitat show the clearest responses HYMO pressures?

7.2 Methods

The procedure of this first approach consists in synthesising the schemes that represent the effects of HYMO pressures on HYMO processes which were obtained by means of a literature review in D1.2; and the observed effects of HYMO pressures on the probability of occurrence of fish species, detected by means of generalized linear analysis (logistic regression) in D3.1. The literature review that was used to build the pressures->processes links included more than 200 specific articles. The data-driven relationships between pressures and probability of occurrence of fish species were found by analyzing datasets from Denmark, Finland, Netherlands, Spain, Sweden and WISER (Table 4.18 in D3.1).

The conceptual schemes linking the response of several hydromorphological processes to hydromorphological pressures were drawn in the Deliverable 1.2 of this project. In these *pressure->process schemes* the pressure effects were expressed separately for each

HYMO pressure by means of a diagram showing its direct effects on the processes. The *pressure->process schemes* were made on the basis of an extensive literature review. The outcomes of this literature analysis in D1.2 were incorporated into one single effect matrix, (Figure 7.1).

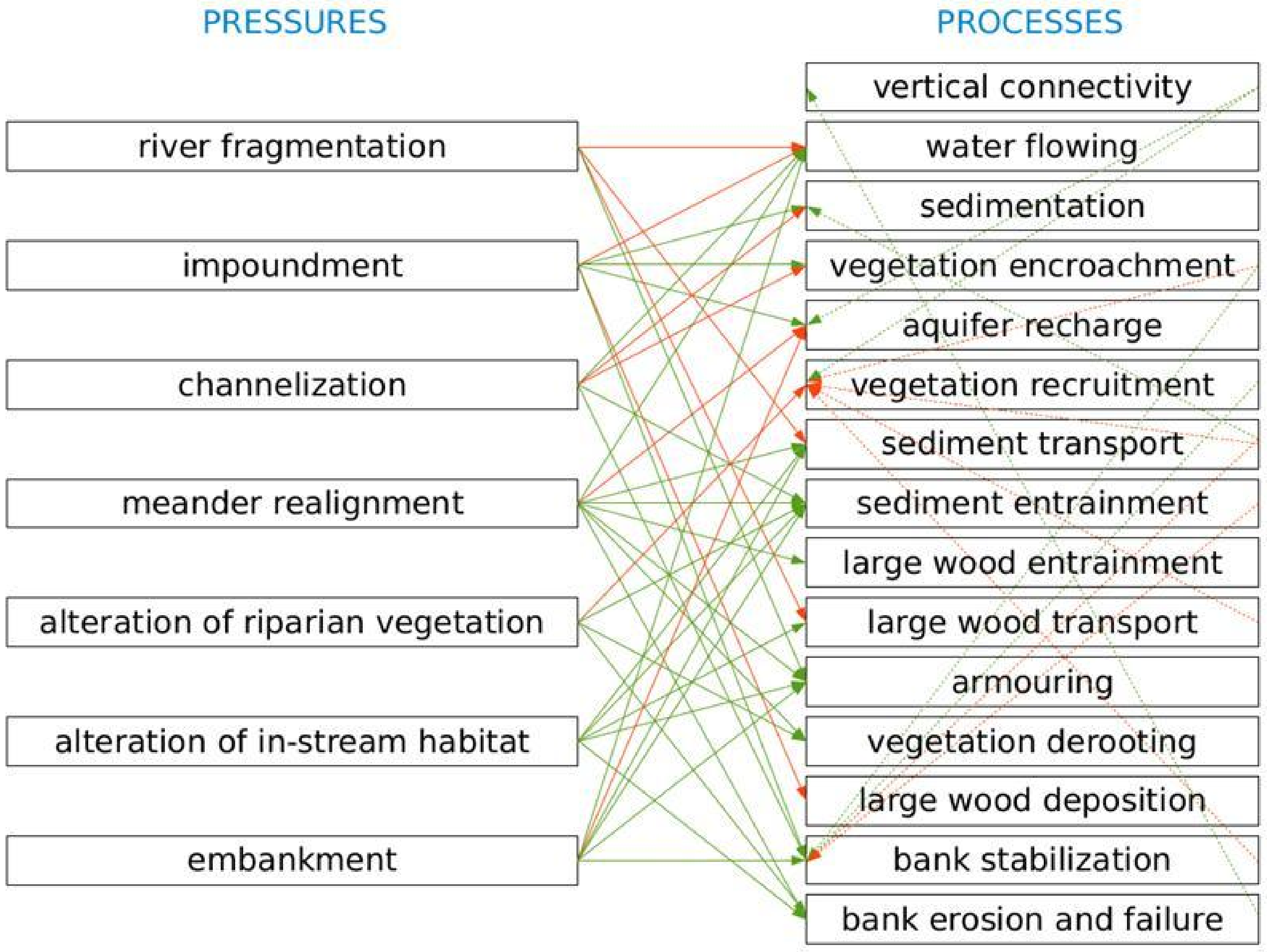

Figure 7.1 Representation of the adjacency matrix resulting from the synthesis of the conceptual schemes linking the effects of hydromorphlogical pressures on hydromorphological processes (pressures->processes schemes). Solid green arrows indicate positive (direct) relationship and solid red arrows indicate negative (inverse) relationship among pressures and processes. Dashed arrows indicate the links among pairs of processes

In a parallel study, the effects of HYMO pressures on every fish species were determined in Deliverable 3.1 from a common database of 2,796 sampling sites and 3,252 fishing occasions. The probability of occurrence of a number of European freshwater fish species was found to be significantly ($p<0.05$) related to the presence of several hydromorphological pressures (Table 7.1 (copied from Table 4.20, Deliverable 3.1) summarises the logistic models that were found to be significant between the presence/absence of every fish species a set of HYMO pressures. Estimated values >0 indicate a positive (direct) relationship and values <0 indicate a negative (inverse) response to the probability of a fish species occurring in the presence of a given pressure (*pressure->fish occurrence schemes*).

It should be noted that the D1.2 output is dependent on the quality of the underlying literature and the statistical analysis of D3.1 did not address the influence of other parameters or attempt to remove their influence. So although the models presented here may infer causality that has not been proven and would require further analysis and development.

In the current study, both types of schemes (namely, *pressures->processes schemes*, and *pressures->fish occurrence schemes*) were synthesised to deduce the links between HYMO processes and fish occurrence (*process->fish occurrence schemes*). For every HYMO pressure, a conceptual scheme representing its links to HYMO processes is considered (Figure 7.2). For a given fish species, the HYMO pressures that induced a significant response on its probability of occurrence (outcomes of D3.1) were selected:

(1) The effects of these selected HYMO pressures on the HYMO processes (outcomes of D1.2) were drawn and coded: a green arrow if the response was positive; a red arrow if negative.

(2) The effect of the alteration of the HYMO process on the probability of occurrence of the given fish species was deduced as follows: (a) if pressure induces a positive response on the fish occurrence and a positive response on the process, then a positive response of the fish occurrence to the process can be deduced (coded green rectangle); (b) if pressure induces a positive response on the fish occurrence and a negative response on the process, then a negative response of the fish occurrence to the process can be deduced (coded red rectangle); (c) if pressure induces a negative response on the fish occurrence and a positive response on the process, then a negative response of the fish occurrence to the process can be deduced (coded red rectangle); (d) if pressure induces a negative response on the fish occurrence and a negative response on the process, then a positive response of the fish occurrence to the process can be deduced (coded green rectangle).

(3) The global response of the fish species to the alteration of the HYMO process produced by all the HYMO pressures selected in step 1 was then summarised. If more than 50% of all deduced effects (coded rectangles) were positive effects (coded green) a positive global (coded green) effect was deduced. Analogously, negative global effects were deduced when more than 50% were negative relations. Unclear cases (~50% positive ~50% negative effects) were coded with a broken red and green rectangle.

Table 7.1 (Copied form Table 4.20, Deliverable 3.1) Summary of the fitting of HYMO pressures (explanatory variable) to presence/absence of every fish species (response variable) by means of a logistic regression (pressures->fish occurrence schemes): estimates and p-values. Relationships 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							247.5	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***			516.5	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			-0.58	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
Gobiidae	*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				

Figure 7.2 shows the graphical schemes followed in this procedure as an example: it is known from Deliverable 1.2 that pressure 1 induces, among others, a negative response on process 2(red solid arrow) and positive on process 11 (green solid arrow); and indirectly (dashed arrows) on process 3 (negative*positive=negative) and 6 (negative*negative=positive). From Deliverable 3.1 it has been found a positive response of the probability of occurrence of fish species a to the presence of pressure 1 (green coloured pressure 1). It is therefore deduced the probability of occurrence of species a is negatively related to process 2 (red rectangles); and positively related to process 11 (green rectangle). The opposite relationships are deduced for fish species b, since the effect of pressure 1 on its probability of occurrence is negative (red coloured pressure 1). To account for the effect of multi-stressor scenarios, every statistically significant link was merged into a single graph.

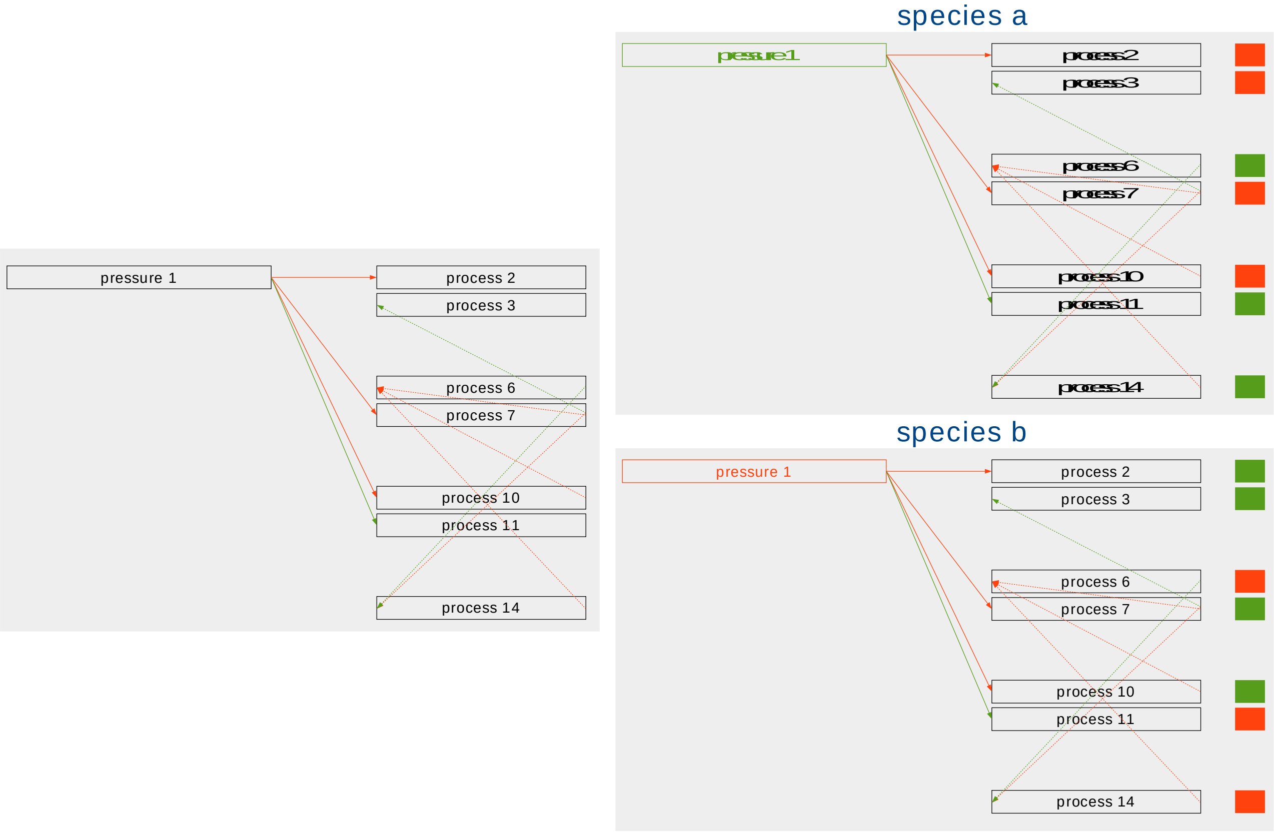

Figure 7.2 Generic procedure to deduce the link between HYMO processes and fish occurrence, from the known links between a HYMO pressure and several HYMO processes.

A species whose probability of occurrence shows the same response to the changes in a HYMO process that are induced by all the HYMO pressures (all green or red rectangles) will produce a more consistent scheme than a species showing opposed responses to the same HYMO process depending on the HYMO pressure that is altering the process.

The response of the species probability of occurrence is expected to be uniform for every HYMO process. Therefore, a summary of the process->species links has been built to establish the effects of every HYMO pressure on the probability of occurrence of every fish species. Only the consistent process->species links have been considered in this summarized conceptual model, built with the global responses of fish species probability of occurrence to changes in HYMO processes. Results of this summary can be expressed in a matrix form.

With this approach, a data-driven conceptual scheme for every fish species whose probability of occurrence was significantly related to the presence of a HYMO pressure was built. In order to make some generalizations, all fish species were sorted according to an index of rheophylly. The Fish Region Index (FRI), (Wolter et al. 2013) was chosen for this purpose. In this way, it could be checked as to whether or not the links between HYMO processes and fish species probability of occurrence followed a gradient along the river continuum.

7.3 Results

A single conceptual model was built for every fish species that showed a significant relationship between a HYMO pressure and its probability of occurrence. We discuss below, in some detail, the models of two contrasting species, brown trout (*Salmo trutta*) and tench (*Tinca tinca*), which have well described ecology, to help evaluate the models and we also describe some general patterns from an initial analyses

Brown trout is a commonly monitored species in fisheries management programs, so knowledge on its ecology and habitat requirements is deep. Brown trout mainly inhabits clean, fresh and well oxygenated reaches. The habitat use and preferences exhibit ontogenetic changes, and, therefore, populations are structured along the river continuum. Fry hatch in the spawning gravel areas located in well oxygenated running waters in the upper reaches, and keep feeding on the yolk sac until they absorb 75% of its volume. Then they emerge and start feeding in the water column near the redd. At this stage fry are very territorial (Kallenberg 1958; Héland 1999; Lahti et al. 2001) and defend an individual feeding area in shallow waters (<30 cm) and near the shores with moderate current (0.2-0.5 m^3s^{-1}) (Roussel and Bardonnet 1999). Individuals that are not able to defend their territory drift with the current to downstream reaches and many of them die (Elliot 1994); although this is highly dependent on the density (Crisp 2000). The habitat use changes allometrically with the size of the individuals. In the model meander realignment is considered a negative impact, which is in line with the homogenisation of instream habitat, whereas direct alteration to instream habitat is considered positive (Figure 7.3). This suggests a bias in the underlying literature toward papers focusing on diversification of instream habitat rather than the homogenisation and disturbances caused by dredging etc. In resident populations, trout can stay close to their birth reaches or move, changing with some frequency, the feeding areas. Young (1+) trout seek shelter in the upper reaches, in the interstitial spaces of boulders. Adult trout usually inhabit pools and tables where they find shelter behind submerged roots and macrophytes (Heggenes et al. 2002).

The main effects of HYMO processes on its probability of occurrence may differ between sites. This species shows low tolerance to hypoxic conditions and high temperatures. Taking Iberian rivers, as an example, they typically suffer from highly variable flow regimes, both inter- and intra-annually. Irrigation programs usually store water during the rainy season (winter in Mediterranean climate) and release it during the dry-irrigation season (in summer). Therefore, irrigation programs cause a smoothening of both hydrographs and thermal regimes downstream from dams. Consequently, in the southern range of its distribution trout may react positively to flow regulation whereas it shows a negative response to hydro-peaking that enhances the naturally variable flow regimes. Therefore, its response to HYMO processes might be obscured by the simultaneous concurrence of multiple pressures. Large woody debris provides shelter for fry, and fine sedimentation saturates the interstitial habitat for eggs and larvae. The model as it stands does not pick up on the negative impacts of sedimentation associated with channelisation and alterations to in-stream habitat and suggests large wood debris would be a negative influence.

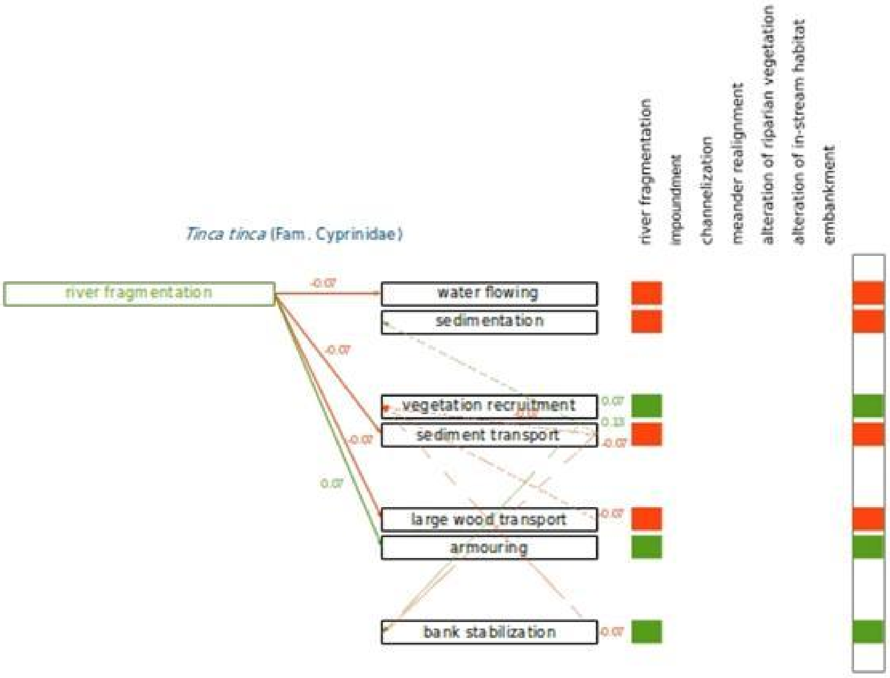

Figure 7.3 Data driver schemes showing the links between HYMO processes and the probability of occurrence of a fish species, from the links deduced from known relations among pressures and the probability of occurrence of that fish species. HYMO pressures in red (green) indicate that are negatively (positively) related to the probability of occurrence of the species. Solid red (green) arrows indicate negative (positive) relations among processes and species probability of occurrence. Dashed red (green) arrows indicate negative (positive) relations among processes. Red (green) rectangles indicate negative (positive) effect of the HYMO process on the species probability of occurrence. The global response of the species to every process is highlighted within a framework.

In river systems tench typically lives in shallow, densely vegetated backwaters. Often it overwinters buried in mud. It spawns among dense vegetation in still water. Tench feed on detritus, benthic animals and plant material. It has been observed that when temperature fluctuates strongly there is a high embryonic mortality (Kottelat and Freyhof 2007). It is reported to be locally threatened by river engineering (Doadrio, 2002; Kottelat and Freyhof 2007). The model did pick up on tench's strong association with vegetation but like many of the species there was very limited information available in journal papers regarding the impact of specific hydromorphological pressures (Figure 7.4).

We looked for patterns in response across the fish species. Their responses were species specific. Only species with narrow tolerances to physical habitat have been found to show a clear, consistent response to HYMO processes alteration. For instance, benthic fish like Cobitidae, Nemacheilidae, Cottidae or Gobiidae show the most consistent response to HYMO processes alteration. This is to some extent expected, since these species are highly dependent on the sediment dynamics of the substrate and besides they have limited swimming skills that at make them less prone to abandon a given reach when a disturbance is acting.

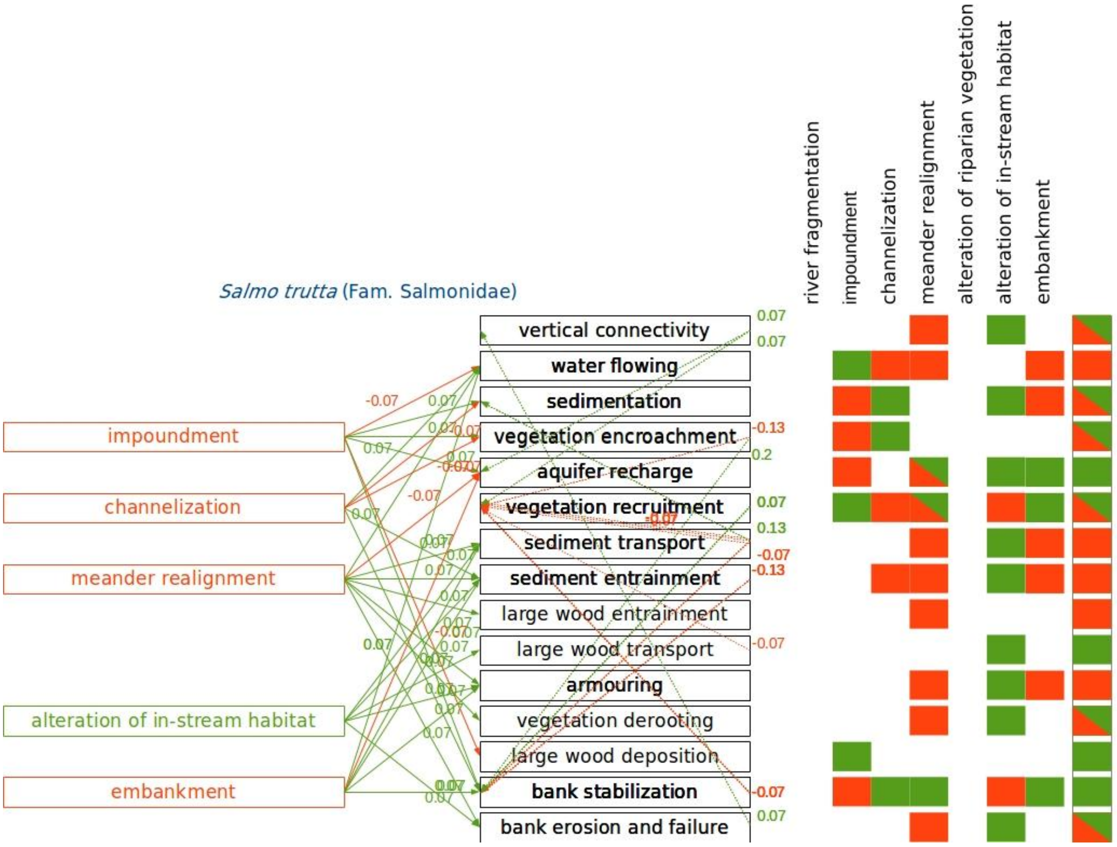

Figure 7.4 Data driver schemes showing the links between HYMO processes and the probability of occurrence of a fish species, from the links deduced from known relations among pressures and the probability of occurrence of that fish species. HYMO pressures in red (green) indicate that are negatively (positively) related to the probability of occurrence of the species. Solid red (green) arrows indicate negative (positive) relations

among processes and species probability of occurrence. Dashed red (green) arrows indicate negative (positive) relations among processes. Red (green) rectangles indicate negative (positive) effect of the HYMO process on the species probability of occurrence. The global response of the species to every process is highlighted within a framework.

7.4 Discussion

The models provided a good means of expressing a significant amount of information in a single diagram. We believe the crucial link is to place process between pressure and biota. In doing so it demands that those interpreting the diagram stop and consider process, which can differ between systems.

The diagrams bridge a number of disciplines and can therefore be seen as a tool for translation between those disciplines. A real world application would be during the early stages of a restoration scheme to use such diagrams to aid discussion between catchment mangers, ecologists, fluvial geomorphologists and engineers. They should be considered as a starting point for discuss rather than a tool which provides a complete answer to our understanding of how physical degradation links to biotic response. We believe such a tool has significant potential but that it would require significant development .

Recommendations for future development:

- The underlying literature analysis could be replaced by a more comprehensive review of fish ecology. Much fish ecology is in book form rather than in peer review papers. It is therefore less accessible to automatic electronic searches.
- Our understanding of the ecology of fish often provides strong evidence for in-stream habitat preferences. In future model iterations linking habitat preferences to known hydromorphological processes would be a useful additional step.
- The statistical analyses linking fish occurrence to pressures should be revised using an analysis system that can detect the residual effects of the pressures once other drivers of occurrence have been removed.
- Throughout this deliverable a recurring issue is that particular stresses are associated with certain river styles. Sub-setting analyses to reflect river style and bioregion, as appropriate, for individual fish species should refine the models. Given the strong relationship between the distribution of fish species and fluvial habitat type (FRI) this process should help focus on the most impacted species.
- There are clear gaps in our knowledge of how hydromorphological degradation may affect some fish species. These gaps must be tackled by primary research.

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8 Fish - stress from multiple interacting hydromorphological stressors over time - three Austrian Case studies

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8.1 Introduction

Most of the studies presented in this deliverable compare biological responses to multiple stressors across river systems that differ in their level of stress. This chapter is different, it tracks a river through time starting with its near-natural river form in the 1880s and ending with the current heavily modified river flowing through a monotone canal-like channel. It is compared to two other case studies. What is apparent from all the case studies is that hydromorphological degradation is not one form of stress but multiple stresses. Human alterations to these channels are not static impacts but alter dynamic fluvial processes. In this regard hydromorphological stresses are fundamentally different from water quality pollutants.

The focus is on mechanistic understanding of the biological response to hydromorphological degradation, including altered sediment dynamics. As rivers are subject to self-steering dynamic processes over time, single historic and actual inspections show just a moment within the river history. The main emphasis of this chapter lies on the detailed analysis of the processes of degradation. These process data are then related to changes in the fish community.

While it was possible to model the hydromorphology in great detail, the data available on fish was limited and it is not possible to determine direct causal links over such a long time periods when a myriad of influences has effected fish populations. Despite that the substantial loss of some habitat forms is clearly illustrative of change in habitat availability. A broad list of hydromorphological parameters that influence biota was produced in Deliverable 3.1. The influence of hydromorphological parameters and pressures on several hydromorphological processes was also dealt with within Deliverable 1.2. Evaluations of parameters in this chapter are based on the results of these deliverables. The first part of the chapter proceeds from general information of the study sites to specific descriptions of hydrological, morphological and biological conditions. In the second part hydrological and hydraulic model results were analysed and input data files for the conceptual fish model were prepared.

Here we compare three case studies; the River Traun where it is possible to compare historic and recent habitat extent and quality which can be implicated, qualitatively as a contributory in the loss of some fish species; the River Mur where in-stream conditions, although not ideal allow key fish species to survive and the River Drau where fine sediment flushing from reservoirs has clogged spawning gravels and is implicated in the loss of fish species.

8.2 Methods

8.2.1 Study Sites

The most detailed analyses were performed for a river section of the River Traun. With a total length of 73 km, the River Traun drains a catchment of 2770 km². The study site of the 6th order alpine stream has a total length of approx. 5.5 km and a total project area of approx. 730 ha. According to the extended river typology (D2.1, Part 1, chapter 7) the historical stream course is classified as type 9, island braided gravel-bed river. The degraded river situation is classified as type 13, sinuous – straight gravel-bed river in the extended classification (Gurnell et al. 2014a). The natural River Traun is described by analysing site descriptions and historic maps. The history of the Traun stream course and the river bed morphology has been documented since the 17th century. The natural situation was described using detailed data from 1885. The actual degradation was described from two datasets on the channelized river. One dataset of the degraded situation from a biological quality assessment of the project site fulfilled in the course of a feasibility study of a water power plant in the 1980s and a second dataset of the fundamental data analysis of a restoration project starting in the year 2000. Hydrological classification corresponds with a moderately Nival Flow Regime with a distinct annual character. The current Heavily Modified River flows through a monotone canal-like channel with a corresponding degraded aquatic community, remnants of running dry riparian forests remain. Further information of this case study site is available on the project website (www.reformrivers.eu). The River Traun is a large alpine river. The study site is located in an epipotamal fish region within the Central highlands ecoregion and the bioregion of the Bavarian-Austrian Alp Foothills.

To provide evaluation data from different river regions, five sites at the upper River Mur were analysed too. According to the extended river typology (D2.1, Part 1, Chapter 7) the historical stream course is classified as type 13, sinuous – straight gravel-bed river. The degraded river situation is classified as type 0, straight-sinuous with some superficial bars in the extended classification (Gurnell et al. 2014a). Field survey data of the current situation was used for modelling and evaluating of hydromorphological data. Fish data was collected for the potential historical situation from established sources as well as for the current situation based on recent surveys at all five sites. The two upper study sites of the River Mur are located in a metarhithral fish region. The other three study sites of the River Mur are located in a hyporhithral (large) fish region. All 5 sites are part of the ecoregion 4 (Alps).

An additional dataset from the River Drau was included to provide a broader variation of different validation data. Only the degraded hydromorphological situation of the river section was analysed for this chapter. According to the extended river typology (D2.1, Part 1, Chapter 7) the degraded river situation is classified as type 0, straight-sinuous with some superficial bars. The historical stream course is classified as type 13, sinuous – straight gravel-bed river in the extended classification (Gurnell et al. 2014a). Information about fish was provided for the historical and current situation. Further information on this study site, which is also part of WP2 and WP4, can be viewed on the project website (www.reformrivers.eu). The study site of the River Drau is located in the hyporhithral fish region in ecoregion 4 (Alps).

8.2.2 Data

8.2.2.1 Historical hydromorphological data

Data analyses of the study site at the River Traun were conducted for several datasets beginning in the 17th century. A detailed technical survey, local maps, cross section data and a longitudinal profile of the natural situation of 1885 were available which were used to generate a digital terrain model of the natural river.

The several hundred years flood in 1797 and the upcoming change of the hydraulic gradient caused a significant change of river type of the River Traun from meandering to braiding. Records about a long time series of great damages caused by were found in several chronicles of nearby towns and a salt shipping company. While major flood events of 1542, 1546, 1548, 1563, 1572, 1787, 1797, 1821, 1862 resulted in major natural hydromorphological changes caused by natural processes in the main river and the side arms, major flood events after the river channelisation (1920, 1928, 1949, 1954, 1956, 1959 and 1985) shows damages in the more intensively used agricultural landscape in the floodplain area. (Mader et al. 2001).

8.2.2.2 Hydrology data

For the hydrological analysis of the rivers Traun, Mur and Drau mean daily flow values were used (Hydrographic Service of Austria (1895-2011)). Based on this data set mean monthly flows were calculated according to the current method of Mader et al. (1996) with some minor adjustments. The analysis of the data series was carried out in two different ways. Firstly in periods of decades (e.g. 2001-2010, 1991-2000 etc.) and secondly considering the whole observation period time frame (Lebiedzinski et al. 2014). The results are given as mean monthly hydrographs. In addition, low flow and mean flow statistics were carried out for supplementary specification, as especially low flow is a good indicator to ensure undisturbed biocenosis (Heger and Moog 1986; Mader 1992). In order to investigate further characteristics, flow regimes were analysed according to the described method in Annex C of Deliverable 2.1 (Gurnell et al. 2014b).

The hydrological analysis of the River Traun was carried out at the gauging station Wels-Lichtenegg. The observation period reaches from 1895 up to 2010. From 1938 to 2010 the data was given directly as flow data. The historical data records from 1895 to 1937 were available as water levels and had to be converted into flow data with revised rating curves of 1904 (after river channelisation), 1947, 1948 and 1949 after morphological changes of the transect at the gauging station. For the hydrological analysis of the River Mur the mean daily flow values at the gauging station Mörtelsdorf were investigated. Since in this case the available data series is a relatively short observed period from 1971 to 2010, only the current situation could be assessed.

The observation period of the River Drau at the gauging station Sachsenburg (Brücke) reaches from 1951 to 2010.

Fish - Data

Archive data is the basis of the historical condition of fish abundance. These datasets are applied to model results of the hydromorphological dataset 1885 of the River Traun and also to the historical hydromorphological condition for the study sites Mur and Drau

(BAW-IGF 2012).

Fish sampling data was collected in the course of water power plant planning in 1987 for the study site at the River Traun. Quantitative samples were taken in 1985 by electrofishing at 5 sites in and around the study site area (Jungwirth et al. 1987). Renewed assessment for the River Traun was done for a step wise restoration project starting in 2000 (Essl and Schuster 2001).

For analysing the hydromorphological condition of the River Traun in detail, a project site related classification for water depth and flow velocities was required. To illustrate the loss of habitats relevant for different life stages of fish, a classification of water depths and velocities was established based on a literature review (Table 8.1). The evaluation is focusing only on the flagship fish species of the River Traun specified by the Austrian Federal Agency for Water Management (BAW-IGF 2012). Parameters were analysed for *Thymallus thymallus* (Sempeski and Gaudin 1995; Nykänen and Huusko 2002; Nykänen 2004; Mouton et al. 2008; Fukuda et al. 2013), *Barbus barbus* (Giesecke et al. 2009) and *Chondrostoma nasus* (Altzinger 2011). Additionally, results were harmonized using defined classes for habitat evaluation (Gorman and Karr 1978). Three classes are differentiated for the life stages 0+ individuals, juveniles and spawners as well as adult individuals. Additional definitions for dry areas and open water complete the classification.

Table 8.1 Classification of specific fish habitats at the study site Traun

	Dry area	0+	Juveniles/ Spawners	Adults	Open Water
water depth [m]	< 0,0	0,0 – 0,2	0,2 – 0,6	0,6 – 1,6	> 1,6
Velocity [m/s]	< 0,0	0,0 – 0,2	0,2 – 0,4	0,4 – 1,0	> 1,0

Information about fish population as well as lengths frequency distribution datasets for the five study sites at the River Mur is based on surveys done by means of electrofishing. All samples were taken between 29th of September and 8th of November 2011 (Salzburg_AG 2011).

Fish data, which is also part of analysis within WP4, was used for the study site at the River Drau. Sampling was done in mid of November 2011 by electrofishing (Schmutz et al. 2013).

Electrofishing is a selective fishing method, as large fish show a better response to electricity as smaller fish and juveniles. For this reason, juveniles and small fish species are underrepresented in the results. Electrofishing in large rivers like the Mur, the Traun and the Drau is a cause of the high variability of the abiotic HYMO-components and its spatial and temporal variation, time consuming, and a scientifically and methodically complicated procedure. Because of this, the available single electrofishing results at a specific time may result in technical inaccuracies and errors.

8.2.3 Hydraulic Modelling

The software SMS (Surface-Water Modelling System) was used for setting up the hydraulic model of the River Traun. The hydraulic modelling of this site was carried out using the software Hydro_AS-2D. The model was adjusted using observations and records from the flood in 2002 (1568 m^3s^{-1}). Numerous simulations were performed at flows from low flow (41.4 m^3s^{-1}) up to three-hundred-year flood events (1750 m^3s^{-1}).

The hydraulic modelling of the five sites of the River Mur was performed using the software River2D. The model was calibrated using observations and records of a calibration flow (150 % $MJNQ_t$) from the year 2009.

The hydraulic modelling of the Drau site was performed using the Model CCHE2D. Data for variables and maps were allocated from the hydromorphological dataset applying to WP2 (Klösch and Blamauer 2014).

A list of all described measures and pressures based on human impacts was compiled for all seven study sites (Alonso et al. 2013). Based on the classification of pressures established in D1.2 (Jalón et al. 2012), a list of all described measures and pressures was selected for the study sites. Additional information was provided by the site description on the project website (2014). Selected processes as well as significant and available hydromorphological variables were chosen for the further evaluation of parameters to validate the fish model established within the REFORM project. A qualitative evaluation of several parameters was fulfilled to validate the conceptual fuzzy logic model within chapter 6. A more detailed quantitative evaluation of selected relevant variables was carried out to provide validation data used in D3.3 (in prep.).

8.2.4 GIS - Processing

The software ArcGIS 10.2 was used for further data processing of the model results of the rivers Traun, Mur and Drau. Data of the hydraulic models of the River Traun and River Mur was transformed to a standardized grid of 1 m to generate datasets for different parameters and flows.

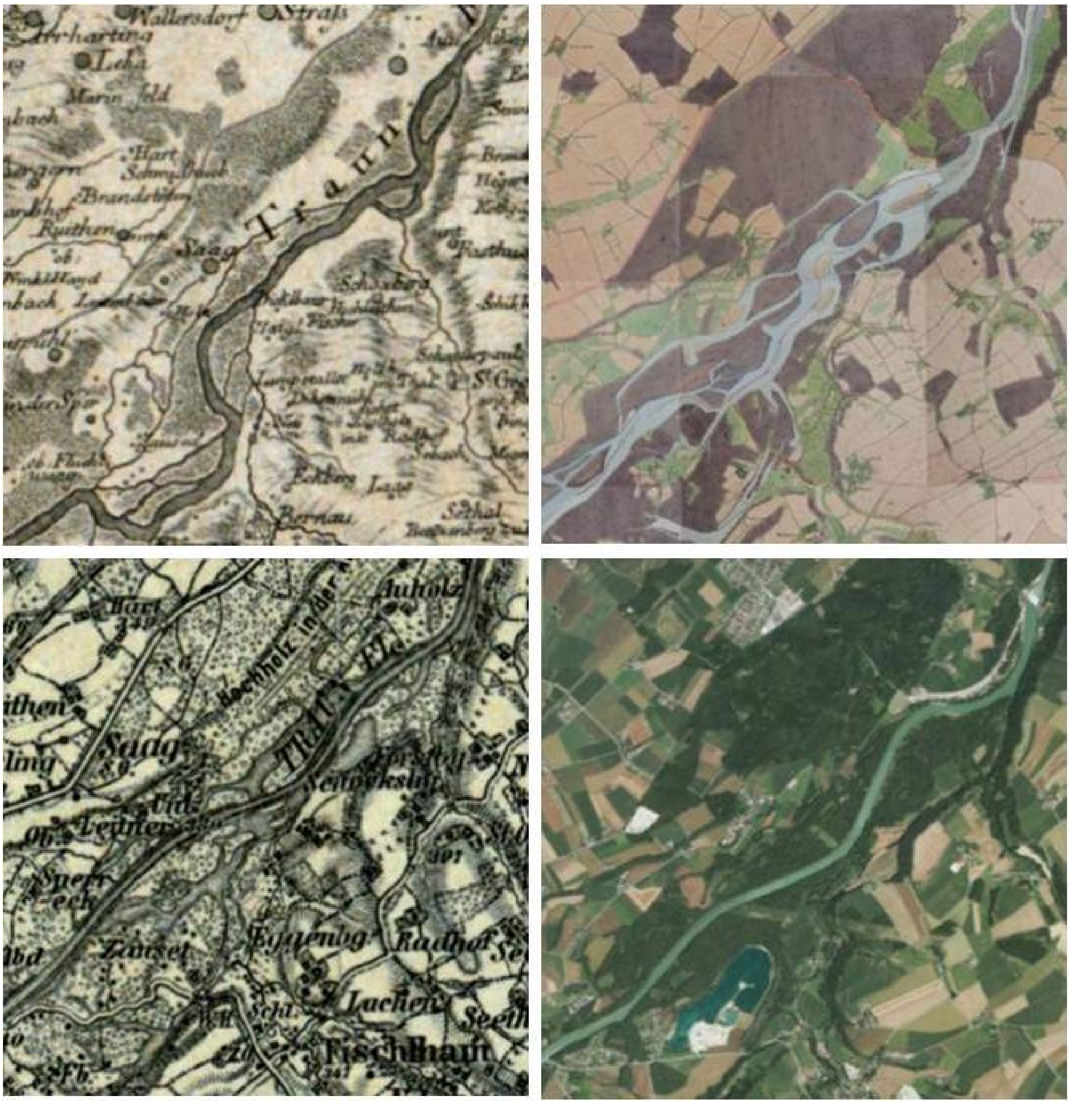


Figure 8.1 Historical development of the study site River Traun 1787 - 1830s - 1904 – 2006

Table 8.2 shows the quantitative analysis of hydromorphological parameters over time. These parameters are also used to quantify variables to describe processes caused by various pressures as defined in deliverable 1.2. Additional analyses for previous situations (before 1885) are provided to give an impression of the self steering dynamic processes of the natural condition of the River Traun. The table illustrates the strong changes in between the natural condition in 1885 and the current degraded situation. The variability of hydromorphological parameters like the channel width was substantially reduced (reduction of variability of channel width ranging from 65 m up to 195 m in the natural situation in 1885 down to 62 m up to 77 m in the degraded situation).

Table 8.2 Quantification of deficits - the study site River Traun from the 17th century

Parameter \ Map	17th Century	1763 – 1787 (Josephini land survey	1805 – 1869 (Franciscan survey map)	1869 – 1887 (Franciscan - Josephini land survey)	1885 (before river regulation)	2004 (current state)
Rearrangement width max. [m]	510	793	1311	1316	718	77
Width max. [m]	145	117	160	356	195	77
Width min. [m]	48	55	100	100	65	62
Width mean [m]	93	87	133	237	117	68
Variance of rearrangement width [cm²]	170 Mio.	528 Mio.	1870 Mio.	1770 Mio.	275 Mio.	0,3 Mio.
Variance of river width [cm²]	8,4 Mio.	2,2 Mio.	4,1 Mio.	78,6 Mio	11,8 Mio.	0,3 Mio.
Fluctuation coefficient w_{FC} [-]	3,0	2,1	1,6	3,6	3,0	1,2
River length [km]	7,10	6,68	5,76	5,67	5,77	5,66
Air-line distance [km]	5,28	5,24	5,28	5,26	5,26	5,26
River length / air-line distance [-]	0,34	0,27	0,09	0,08	0,10	0,08
Number of sidearms max.	4	4	4	5	5	1
Number of sidearms mean	2,1	2,3	2,2	3,1	2,4	1,0
Radius of meanders max. [m]	1800	1120	520	500	660	1300
Radius of meanders min. [m]	60	140	160	110	90	300
Mean radius of meanders [m]	370	390	330	300	380	650
Number of meanders	8	8	8	9	8	6
Islands & gravel banks	2	0	4	7	6	0

8.3 Results

8.3.1 River Traun Case study

River regulation measures at the River Traun were introduced in 1885 (Figure 8.1, Figure 8.2). Dams were raised in order to prevent the self-steering widening of the water caused by flood events. Several bayous were cut off from the main river bed. A flood protection embankment (Saag) was constructed in the 1970s. In the course of these measures the downward gradient was increased by straightening the River Traun over the whole reach of the study site. Water depths and flow velocities increased significantly due to the transformation into a steep and straightened single bed river course. The transport capacity of the new river bed was optimised hydraulically. The river banks of the study site were stabilized. The river width was reduced dramatically. As a consequence of the reduced river width, increased flow velocity and the increased downward gradient the river bed was lowered several meters over the period of 100 years. This lowering was caused by the increased transport capacity combined with a lack of sediment input from the River Traun and its tributaries. A permanent weir and a hydropower plant were established in 1927 (subsequent increase of the weir height in 1950). Within the impoundment upstream of the weir flow velocities were reduced and almost stagnated in some areas while the water depth is increasing and sediments are deposited in the reservoir upstream the weir.

In the area around the study site small scale changes in land use are recorded in recent decades. In particular these changes were planting of spruce mono-cultures and occasional riparian forest clearance for intensive agriculture. Most of the study site is classified as a Natural River except a length of approx. 800 m connected to the weir, which is classified as Heavy Modified Water Body. Between the year 2000 and 2010 a restoration project was established in several stages. The first constructional measures started in 2009.

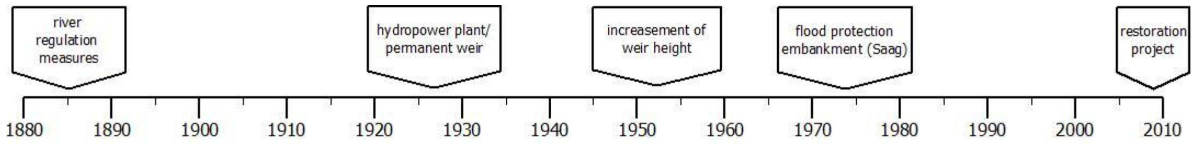


Figure 8.2 Timeline of significant changes of the river Traun.

With regards to the lists of hydromorphological processes and pressures from Deliverable 1.2 the following multiple-pressures affect this river section at the River Traun:

- River fragmentation
- Channelisation – cross section alteration
- Alteration of riparian vegetation – logging and tree removal – transformation into farming lands
- Water abstraction – groundwater abstraction
- Embankments, levees or lateral dykes
- Sand and gravel extraction – gravel pits
- Alteration of in-stream habitat – bottom rigidifying

- Hydromorphology changes

The current as well as the historical condition of the River Traun were modelled beginning from low flow up to a 300-year flood event. Results of water depth distribution and flow velocity distribution are shown schematically for the natural conditions 1885 and the degraded conditions 2006 for different flows (Figure 8.3).

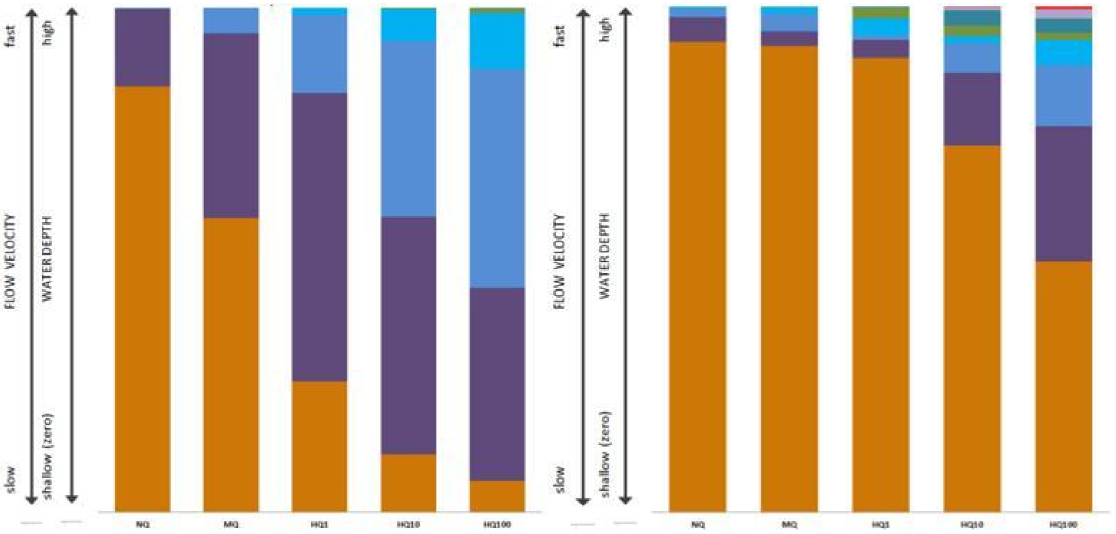


Figure 8.3 River Traun Modelled water depths and flow velocity distribution 1885 (left) and 2006 (right) for different flows. The classification shown in these graphs is based on results of D2.2 (Gurnell et al. 2014c).

In 1885 there was a mean active channel width of approx. 650 m, several side arms and a broad variability of water depths, the river Traun shows a high diversity in habitat structures (Figure 8.4). The impact of the weir on the downstream end of the study site can be detected by significant increase of water depth up to over 4 meters and significant decrease of flow velocities in the backwater area of the degraded situation.

The degraded situation is presented by a monotonous single bed river with a reduced mean active channel width of one sixth of the natural condition. The figure of water velocity illustrates the significant loss of wetted area, especially in areas showing low flow velocities in 1885. The reduction of the mean active channel width and the dramatic loss of inundated areas with low water depth and low flow velocities become particularly obvious when comparing the natural historical situation to the degraded situation at mean flow.

Additional to the higher water depths, Figure 8.4 shows an increase in the flow velocities in the degraded situation at mean annual daily low flow.

The decrease of low water depth in the side channels and increase of great depth in the impoundment area above the weir is also clearly shown in the comparison of historical (1885) and degraded (2006) situation at mean annual daily low flow. Results of 2D hydraulic modelling show the dramatic loss of wetted area and available aquatic habitat at low flow conditions as well as at mean flow. Moreover, the considerable reduction of water flowing branches at mean and low flow conditions underscores the decline of the function of the riparian area as an aquatic, semi-terrestrial and terrestrial living space.

The hydrology of the River Traun and consequently the habitat conditions were affected by early channelisation measures, for example, the control “Seeklause” of Traun lake for flow regulation, and hydro power. In Figure 8.5 the mean monthly hydrographs at the gauging station Wels-Lichtenegg are shown in 10-year periods from 1895 to 2010.

A considerable alteration of the hydrographs in the course of time is visible in the observed period. As Figure 8.5 also illustrates, these recurring fluctuations occur in combination with a decrease of the peak value in May and an increase of the low flows in the winter months. This phenomenon is most apparent in recent decades resulting in modified hydrographs.

On closer examination, the low flow shows a significant jump around 1960 and a constantly increasing trend in the following years (Figure 8.6). Correspondingly, at the same time the trend of the mean annual daily low flow progression is reversed from decreasing into increasing. Due to the fact that at this time no additional river intervention occurred and the relation of hydropower impact through water storage and mean flow is about one percent, the cause of this process is still unclear. Even an additional effect caused by the operation of the upstream located “Seeklause” can be excluded, since it was built between 1629 and 1630 and is nowadays protected as a historic monument. As it cannot be regulated even at flood events, this influence can be considered a permanent impact.

The result of the flow regime analysis according to the method described in Annex C D2.1 (Gurnell et al. 2014b) is shown below in Figure 8.7. For the time series from 1980 to 2010 a perennial stable flow regime was determined.

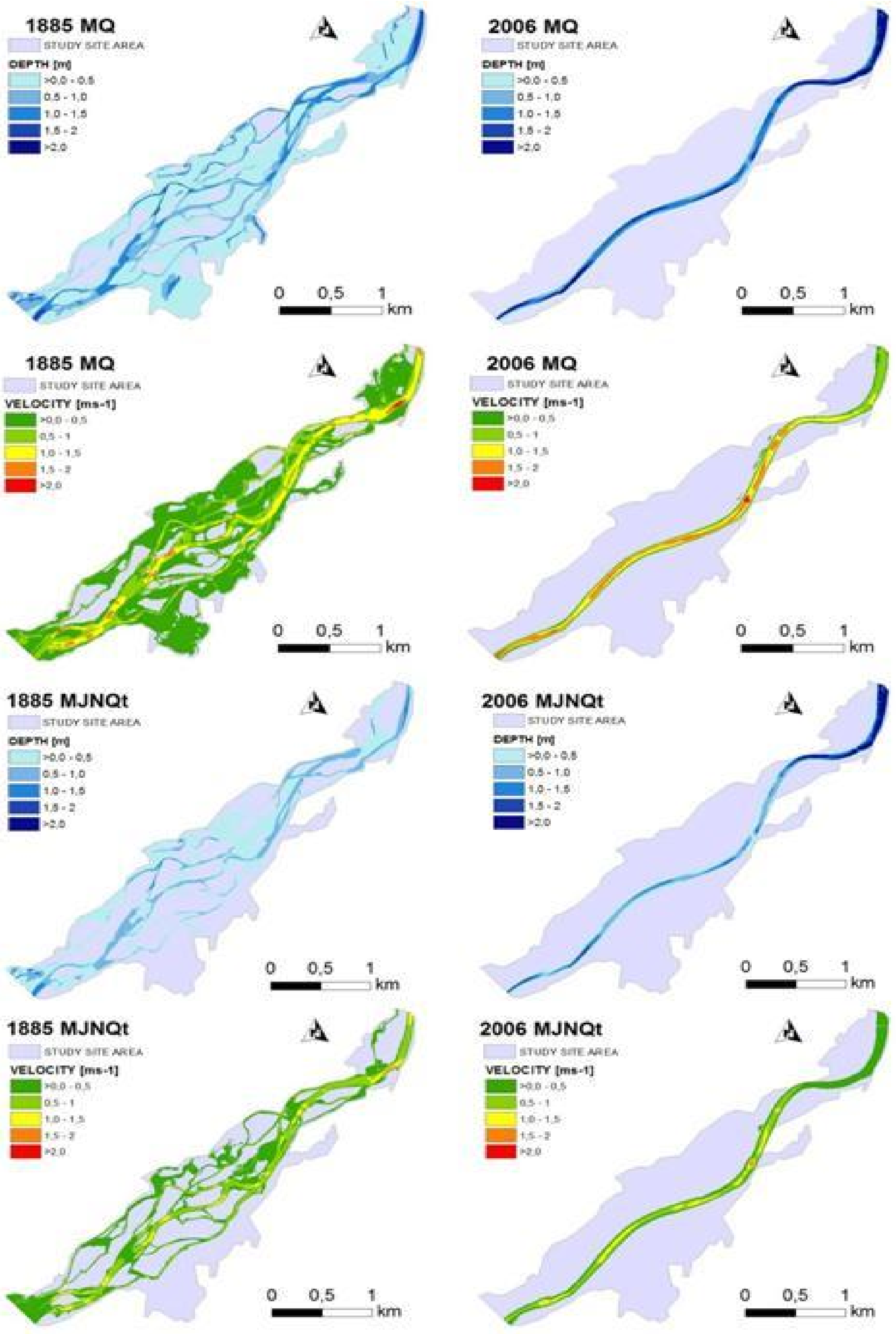


Figure 8.4 Water depth and velocity in 1885 and 2006 for the River Traun for MQ and MJNQ$_t$ (mean annual low flow on an annual series t).

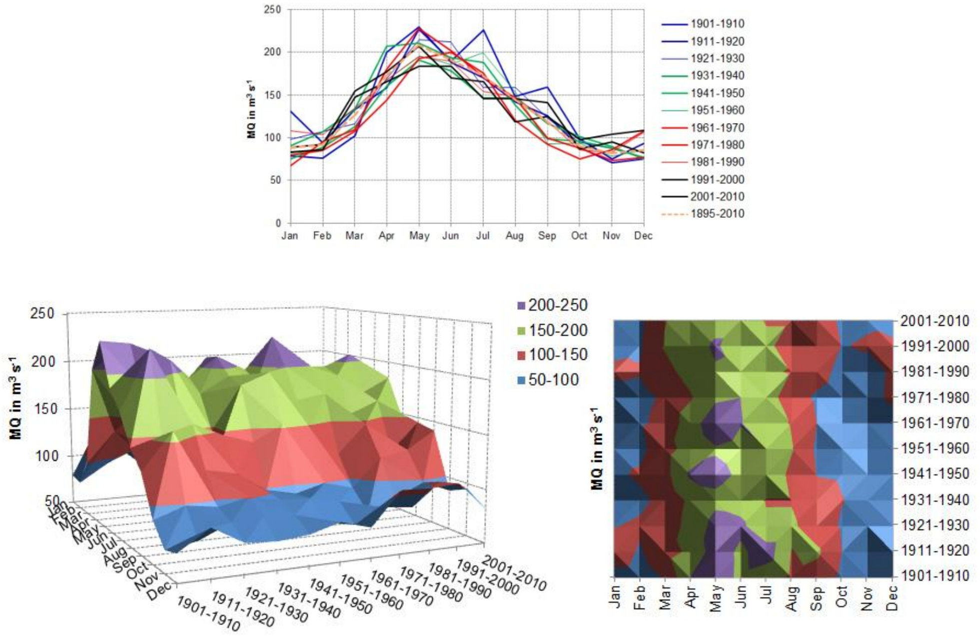


Figure 8.5 Mean monthly hydrographs of river Traun at Wels-Lichtenegg (dashed line - whole observation period, continuous lines – decades) (Lebiedzinski et al. 2014). 3D colour surface chart of mean monthly hydrographs of River Traun at Wels Lichtenegg.

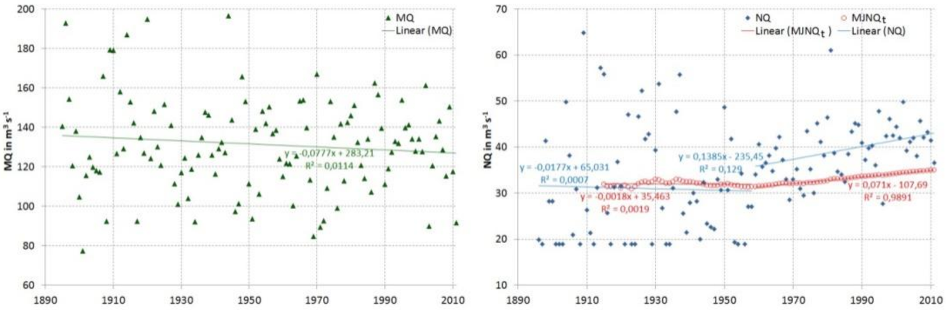


Figure 8.6 Mean flow (MQ), low flow (NQ) and mean annual daily low flow ($MJNQ_t$) progression of river Traun at Wels-Lichtenegg (1895-2011).

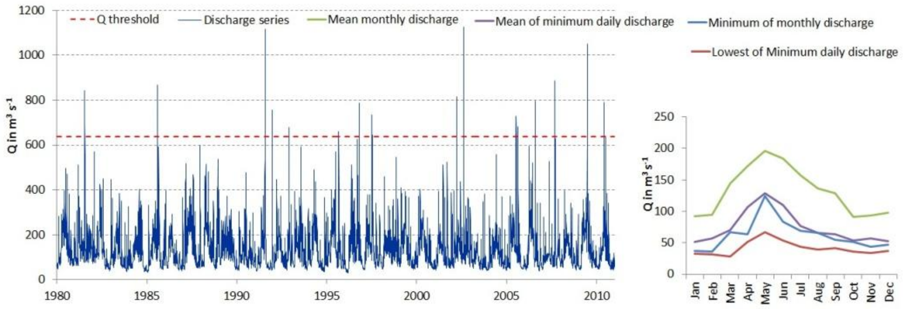


Figure 8.7 Flow regime analysis of River Traun at Wels-Lichtenegg - maximal flood events over threshold and characteristic monthly flows (1980-2010) based on the method in Annex C D2.1 (Gurnell et al. 2014b).

8.3.1.1 Implications of changes for fish

Quantitative parameters were calculated for low flow, mean flow and the annual flood event, see appendix for details. Distinctive profiles as well as an area analysis of the 2-dimensional modelling results built up the basis for these calculations. Results of analysed water depth and flow velocity distributions of the study site of the River Traun are shown and discussed based on the total wetted area at mean flow condition of approx. 263 ha in the historical condition of 1885 including riverbanks and islands. At mean flow 22 % of the total area of 263 ha is classified as dry area (riverbanks and islands) and 78 % as wetted area. The wetted area is subdivided in 4 classes for water depth analysis and respectively 5 classes for flow velocity analysis. This classification was established considering the different requirements of habitat for all life stages of key fish species in the specific fish region of the River Traun (epipotamal). The significant alteration caused by human measures is clearly illustrated by the graphs of modelled water depths and flow velocities of the natural and the degraded River Traun. Comparison of the historical and the degraded situation at mean flow is presented for water depths and flow velocities. In 2006 only 15 % of the total area of 263 ha remains as wetted area.

The detailed analysis of the habitat relevant area shows a drastic reduction of available shallow habitat, especially for juvenile fish and fry. Very shallow and shallow areas are cut down from nearly 60 % of the total area in 1885 to 5 % in 2006 while the percentage of great depth area is increasing (Figure 8.8). The loss of more than 60 % of wetted area over the last 120 years has serious consequences on the proportionate areas of flow velocity classes. Similar to the distribution of depths at mean flow, very slow and slow flow velocity areas are reduced dramatically. The results for the mean annual daily low flow conditions also refer to the benchmark of the defined area of 263 ha at mean flow. 40 % of these defined areas are rated as wetted area at mean annual daily low flow in 1885. Due to anthropogenic influences over 120 years, the wetted area at mean annual daily low flow is cut down to 14 % in 2006 (Figure 8.8).

As shown in Figure 8.8 very shallow and shallow areas of the River Traun were decreasing from over 30 % of the total area to 7 %. Even the percentage change of several habitats is not as prominent at mean annual daily low flow as it is at mean flow; the negative effects on important habitats for aquatic organisms are significant. Further on, the change in the composition of the depth distribution also has a negative impact on aquatic biocoenoses in general.

A closer look into the analysis of flow velocity alteration suggests that there is a considerable decrease of low flow areas also at mean annual daily low flow. Almost no areas of stagnating velocity could be detected when analysing the degraded situation in 2006. The comparison of the modelling results at the natural and the degraded river shows the dramatic loss of habitat caused by a 20 % reduction of wetted area at mean annual daily low flow conditions and an 80 % reduction of wetted area at mean flow conditions. Details of the percentage changes for the considered flow conditions are displayed for water depths (Table 8.3) and flow velocities (Table 8.4) for the historical natural and the current degraded situation.

As an overall result of the multi pressure degradation at the River Traun study site, including both local and regional pressures, only between 8 (1985) and 15 (2001) out of the 39 fish species present in 1885 were observed. Table 8.5 illustrates the presence and absence of fish species at the River Traun study site in 1885, 1985 and 2001. The increase between 1985 and 2001 is mainly attributable to water quality improvement. The historical fish population was evaluated considering historical records and expert knowledge (Ratschan et al. 2008) *Thymallus thymallus* and *Barbus barbus* are listed as key fish species (epipotamal fish region) and were also found in current sample results. The third key fish species *Chondrostoma nasus* was detected in none of the recent samplings at the study site. Historical sources report an enormous population of *Chondrostoma nasus* (Scheiber 1930). The absence of this species is most likely caused by the morphological situation downstream (insurmountable artificial barriers, fixed river bed, reduction of flow velocity) and not caused by local pressures. In general, this species is a reasonable indicator for good ecological condition because of special food and complex ecological requirements. The population declines due to morphological alteration (Melcher 1999) as impoundments downstream make spawning migration difficult. Some of the hydropower plants still have no fish passes installed making fish migration impossible. Due to this also the downstream migration of juvenile nase is not possible (Berg and Gumpinger 2009).

Hucho hucho is a typical side species for this region. The above mentioned interruptions of the aquatic continuum make the essential spawning migration and therefore a self-reproducing population impossible. The study site is rated as good potential habitat including good spawning areas for *Hucho hucho*.

Homogenisation of flow dynamics of the River Traun cause loss of aquatic habitats, changes of temporal and spatial habitat availability for aquatic flora and fauna and reduced lateral connectivity. There is a low biomass determined over the whole river segment mainly due to artificial barriers, hydromorphological degradation and loss of structures. There is an extensive loss of habitats for rheophilic species in the impoundment area upstream the weir (Höfler et al. 2013)

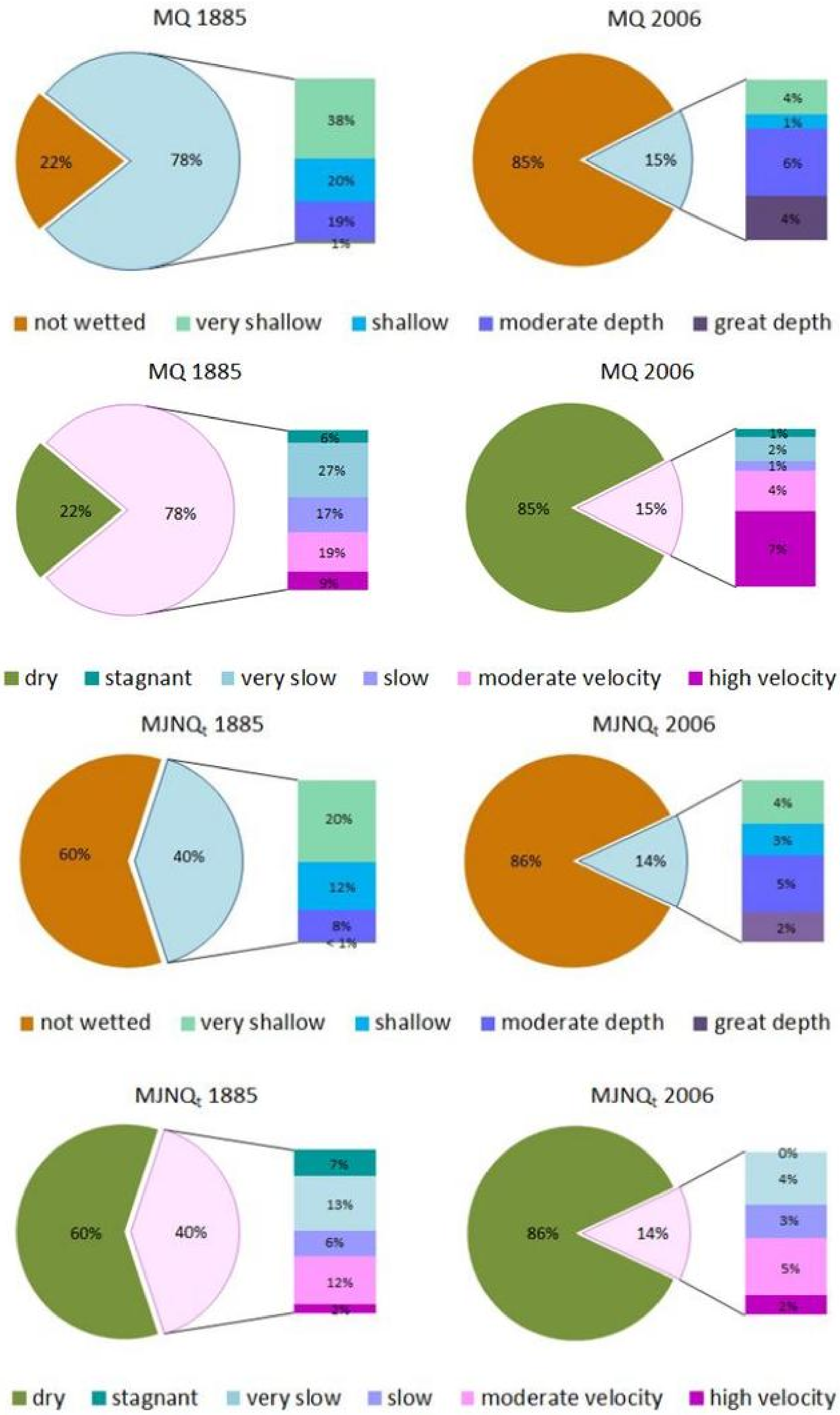
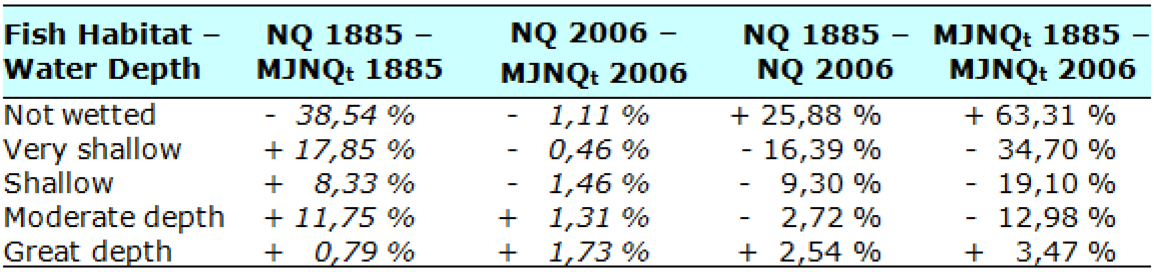


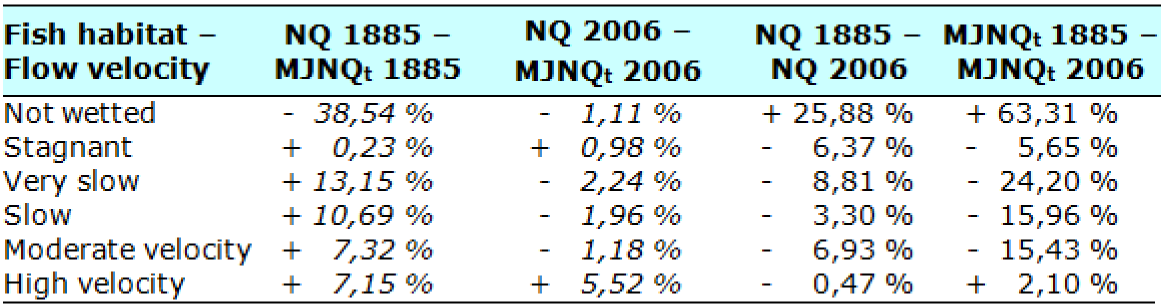
Figure 8.8 A comparison of water depths and velocities in 1885 and 2006 at mean flow (MQ) and at mean annual daily low flow (MJNQ_t).

Table 8.3 Comparison of water depths in 1885 and 2006 at mean annual daily low flow situation and mean flow as a percentage



Fish Habitat – Water Depth	NQ 1885 – MJNQ$_t$ 1885	NQ 2006 – MJNQ$_t$ 2006	NQ 1885 – NQ 2006	MJNQ$_t$ 1885 – MJNQ$_t$ 2006
Not wetted	*- 38,54 %*	*- 1,11 %*	+ 25,88 %	+ 63,31 %
Very shallow	*+ 17,85 %*	*- 0,46 %*	- 16,39 %	- 34,70 %
Shallow	*+ 8,33 %*	*- 1,46 %*	- 9,30 %	- 19,10 %
Moderate depth	*+ 11,75 %*	*+ 1,31 %*	- 2,72 %	- 12,98 %
Great depth	*+ 0,79 %*	*+ 1,73 %*	+ 2,54 %	+ 3,47 %

Table 8.4 Comparison of flow velocities in 1885 and 2006 at mean annual daily low flow situation and mean flow as a percentage



Fish habitat – Flow velocity	NQ 1885 – MJNQ$_t$ 1885	NQ 2006 – MJNQ$_t$ 2006	NQ 1885 – NQ 2006	MJNQ$_t$ 1885 – MJNQ$_t$ 2006
Not wetted	*- 38,54 %*	*- 1,11 %*	+ 25,88 %	+ 63,31 %
Stagnant	*+ 0,23 %*	*+ 0,98 %*	- 6,37 %	- 5,65 %
Very slow	*+ 13,15 %*	*- 2,24 %*	- 8,81 %	- 24,20 %
Slow	*+ 10,69 %*	*- 1,96 %*	- 3,30 %	- 15,96 %
Moderate velocity	*+ 7,32 %*	*- 1,18 %*	- 6,93 %	- 15,43 %
High velocity	*+ 7,15 %*	*+ 5,52 %*	- 0,47 %	+ 2,10 %

.

Table 8.5 Comparison of presence and absence of fish species at the River Traun study site

scientific name	species	Case Study Site TRAUN		
		1885	1985	2006
Thymallus thymallus	european grayling	✓	✓	✓
Barbus barbus	barbel	✓	✓	✓
Chondrostoma nasus	nase	✓	✗	✗
Lota lota	burbot	✓	✓ tributary	✓
Squalius cephalus	european chub	✓	✗	✓
Salmo trutta fario	brown trout	✓	✓	✓
Barbatula barbatula	stone loach	✓	✓	✓
Abramis brama	common bream	✓	✗	✓ tributary
Phoxinus phoxinus	eurasian minnow	✓	✗	✓
Perca fluviatilis	eurasian perch	✓	✓	✓
Gobio gobio	gudgeon	✓	✓	✓
Leuciscus leuciscus	eurasian dace	✓	✗	✗
Esox lucius	northern pike	✓	✓	✓ tributary
Hucho hucho	danube salmon	✓	✗	✗
Cottus gobio	bullhead	✓	✗	✓
Alburnus alburnus	bleak	✓	✗	✗
Leuciscus idus	danube ide	✓	✗	✗
Rutilus rutilus	roach	✓	✗	✓ tributary
Rhodeus amarus	bitterling	✓	✗	✗
Rutilus pigus	danube roach	✓	✗	✗
Carassius carassius	crucian carp	✓	✗	✗
Gymnocephalus cernuus	ruffe	✓	✗	✗
Eudontomyzon mariae	ukrainian brook lamprey	✓	✗	✗
Rutilus meidingeri	perlfish	✓	✗	✗
Scardinius erythrophthalmus	rudd	✓	✗	✗
Vimba vimba	vimba	✓	✗	✗
Aspius aspius	asp	✓	✗	✗
Misgurnus fossilis	weatherfish	✓	✗	✗
Tinca tinca	tench	✓	✗	✓
Alburnoides bipunctatus	spirling	✓	✗	✓
Salmo trutta lacustris	lake trout	✓	✗	✗
Alburnus mento		✓	✗	✗
Cobitis elongatoides	spined loach	✓	✗	✗
Acipenser ruthenus	sterlet	✓	✗	✗
Zingel streber	danube streber	✓	✗	✗
Telestes souffia	sourie, vairone	✓	✗	✗
Silurus glanis	wels	✓	✗	✗
Sander lucioperca	zander	✓	✗	✗
Zingel zingel	zingel	✓	✗	✗

8.3.2 River Mur Case study

The sites 1 to 4 of the River Mur are described as altered in its character, but not classified as Heavily Modified Water Bodies. The assessment of the hydromorphological condition is moderate within the first four sites. Only site 5 is classified as a Natural River with a good ecological condition. Sites 1 and 2 are part of the river section which is influenced by a ramp located between study site 2 and 3. The ramp leads to a reduced downward gradient in the surrounding area. Most of the tributaries in the river section, of which sites 3 and 4 are a part of, are disconnected from the River Mur. Channelisation measures started in the 1870s (Figure 8.9). Large parts of the analysed sites are affected by embankments established in the course of the nearby train track construction in the 1890s. The stabilisation of the river banks and the straightening of the river caused a shortening of the river course, a loss of depths and velocity variability, higher water depths and flow velocities in the single bed river course and therefore a higher downward gradient. In the area around the study site small scale changes in land use are recorded over the last 140 years. This led to the loss of riparian vegetation around the study sites. A hydropower plant was built and operated within study site 3 between 1900 and the early 1990s. The plant mainly affected in-stream habitat and sediment transport. Two large storage power plants were established between 1984 and 2008 upstream of the study sites causing slight short term flow fluctuations in the River Mur. Due to the operational mode of the storage power plants a hydrological regime modification can be determined. The whole river segment with all five study sites is evaluated as natural fish habitat. Initial measures to enhance the self-steering processes and relocating capacity are recommended in this area.

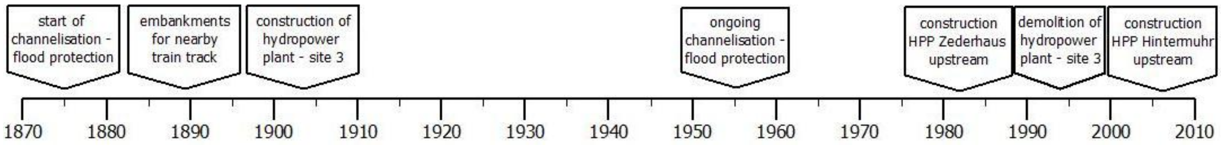


Figure 8.9 Timeline of significant changes of the River Mur.

With regards to the lists of hydromorphological processes and pressures from Deliverable 1.2 the following multiple-pressures affect the river sections at the River Mur:

- Hydrological regime modification – hydrological alteration by reservoirs
- Channelisation – cross section alteration
- Alteration of riparian vegetation – logging and tree removal – transformation into farming lands
- Hydropeaking – short term flow fluctuation by hydropower plants
- Alteration of in-stream habitat – bottom rigidifying
- Hydromorphology changes

Terrain models and hydraulic parameters of five study sites served as basis for the hydromorphological analysis at the River Mur. The sites are distributed along a river stretch of approx. 15 km. There are fewer anthropogenic influences and impacts compared to the study site Traun. The hydromorphological condition was analysed for

2009, there was no comparable historical dataset. Modelling results for water depth and flow velocity distributions at a habitat relevant flow ($MJNQ_t$) for the five sites are shown in (Figure 8.10). All sites are characterised by homogenous and relatively monotone river geometry in lateral as well as longitudinal direction.

The first study site (site 1) shows very homogeneous hydromorphological conditions due to channelisation of the River Mur. No notable structures occur with the exception of a small gravel bank in the middle of the section. A slightly greater variability of water depths and flow velocities could be detected at site 2 caused by a change of direction of the stream course (Figure 8.10).

Due to anthropogenic influences at the River Mur, study site 4 also shows very homogeneous hydromorphological conditions. No notable structures occur in this site except a hardly notable gravel bank on the orographic left side which can be seen at mean annual daily low flow conditions. The monotonous conditions of the regulated river can be also seen in Figure 8.10 where the homogenous structure of the study site is interrupted only by a small gravel bar on the orographic left side of the river that becomes apparent at mean annual daily flow in site 5.

One of the few remaining structures leading to greater variety of aquatic habitats appears in site 3 of the upper Mur. The highest variability of water depths and flow velocities of the five datasets of this river are found in this area.

Based on the hydromorphological parameters determined in Deliverable 3.1 and the linkage of pressures, processes and variables explained in Deliverable 1.2, the qualitative and quantitative availability is assessed in a table of selected pressures which can be found in Table 13.3 (appendix). Within the same table, the qualitative evaluation was executed using a simple rating system ranging from increase to decrease of the hydromorphological variable. Processes and variables were analysed separately for each hydromorphological pressure occurring at the study sites.

These indicators were calculated for all five sites at mean annual daily low flow, see Table 13.4 (appendix). Distinctive profiles as well as an area analysis of the 2-dimensional modelling results built up the basis for these calculations.

The hydrology of the River Mur is mainly influenced by moderate nival flow characteristics. At the gauging station Mörtelsdorf the river is affected by a storage power plant which was built in 1991 and extended to a pump storage power plant in 2008. The relation of hydropower impact due to water storage and mean flow is about six percent. As a result of this intervention a higher flow in autumn and winter as well as a moved peak towards spring can be detected as shown in Figure 8.11. Additionally a decrease of the peak value can be seen. Concerning this development, a steady positive trend towards a higher mean low flow has appeared as a secondary effect of the storage activity (Figure 8.12). However, the mean flow progression shows no significant changes. Figure 8.13 illustrates the maximal flood events over threshold and characteristic monthly flows. On the basis of the time series from 1971 to 2010 a perennial stable flow regime was determined.

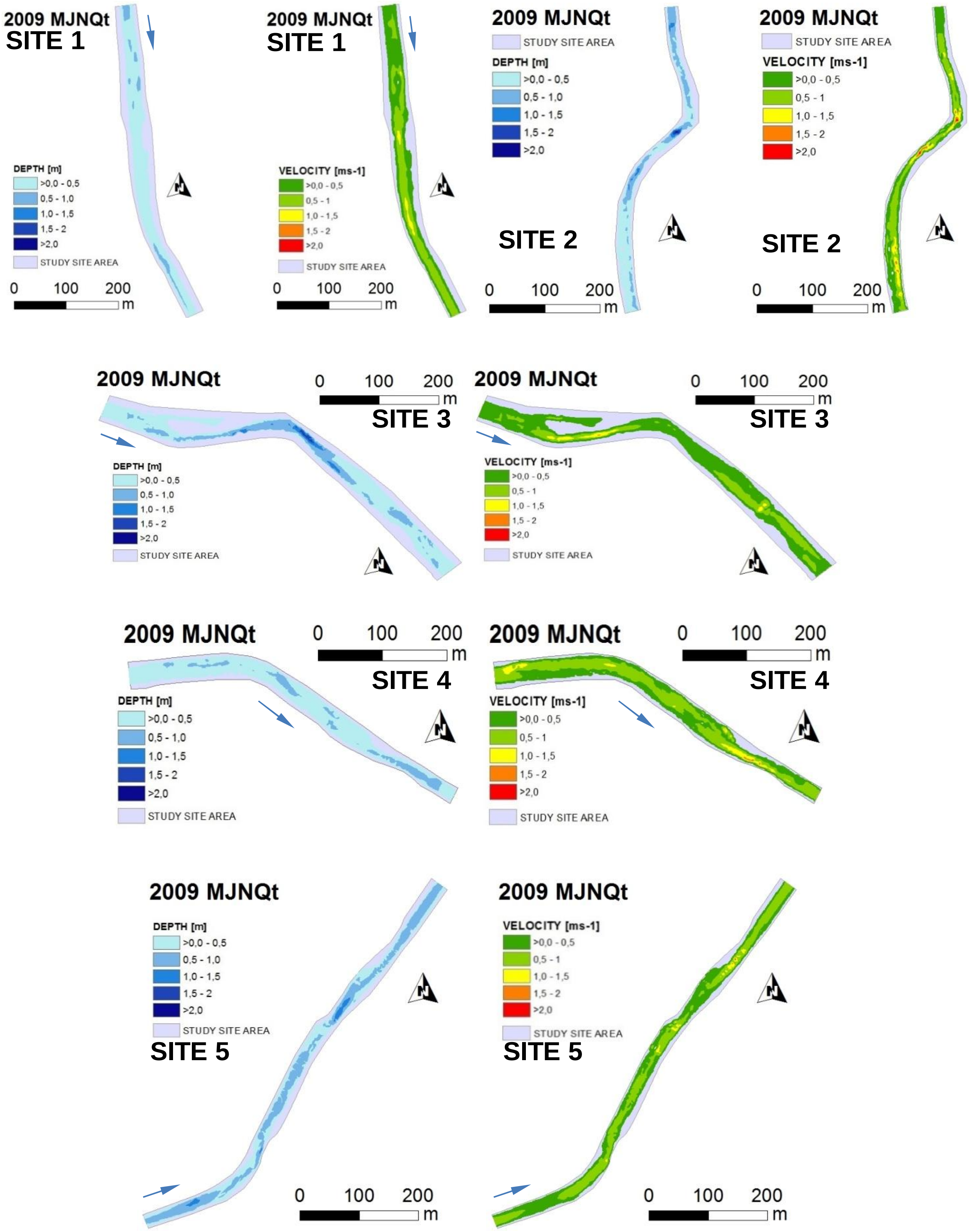


Figure 8.10 Water depth and velocity for MJNQ$_t$ at the four study sites on the River Mur.

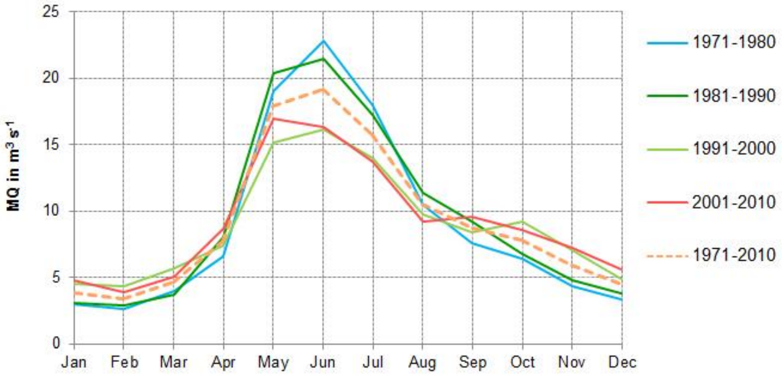


Figure 8.11 Mean monthly hydrographs of River Mur at Mörtelsdorf (dashed line - whole observation period, continuous lines – decades) (Lebiedzinski et al. 2014).

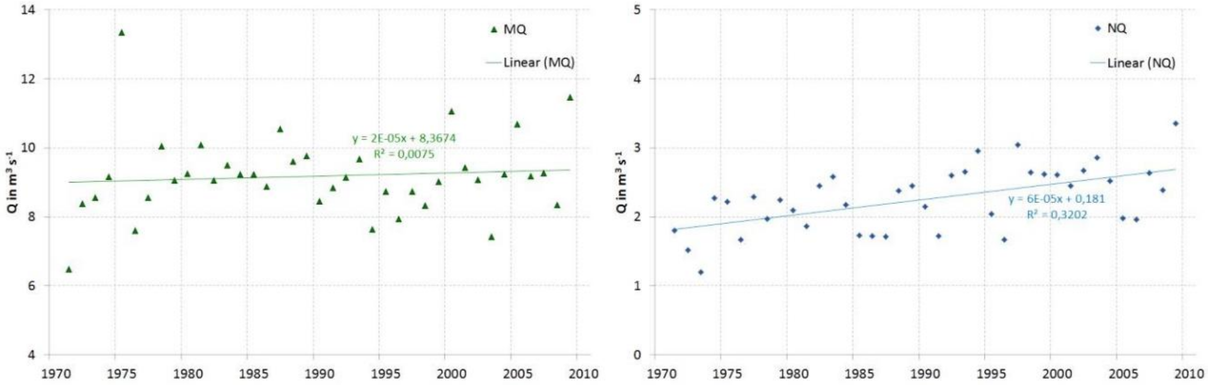


Figure 8.12 Mean flow (MQ) and low flow (NQ) progression of River Mur at Mörtelsdorf (1971-2010).

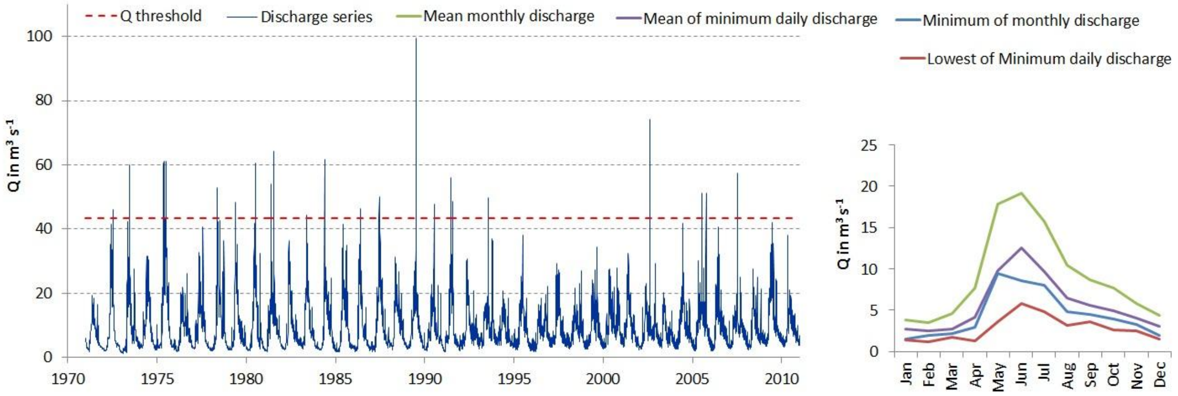


Figure 8.13 Flow regime analysis of River Mur at Mörtelsdorf - maximal flood events over threshold and characteristic monthly flows (1971-2010) based on the method in Annex C D2.1 (Gurnell et al. 2014b).

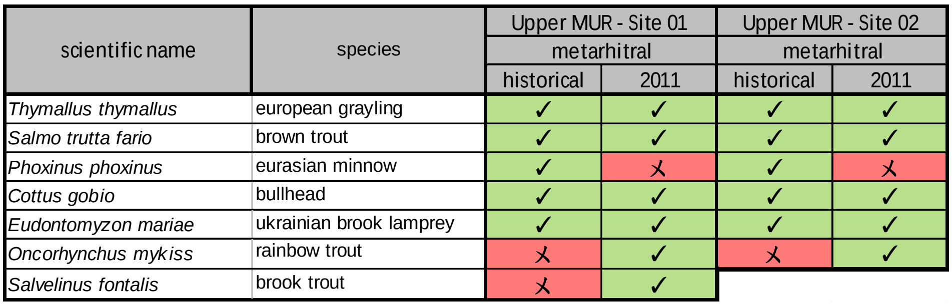
8.3.2.1 Fish community changes

Sampling results at the River Mur show that the key species are occurring in all of the five study sites at the latest sampling. A comparison of the presence and absence of fish species at the study sites 1 and 2 is shown in Table 8.6 and in Table 8.7 for the sites 3,4 and 5. Site 1 and 2 are located in the metarhithral fish region. The historical fish population was evaluated counting five fish species based on historical records and expert knowledge (Wiesner et al. 2005).

The key fish species for this area (*Salmo trutta fario*, *Thymallus thymallus* and *Cottus gobio*) show naturally functioning reproduction behaviour. However few sites provide appropriate substrate for spawning activities. Within the river reach only small scale spawning areas are available. Due to the river size not all spawning sites were detected within the field investigations. Very limited self-steering dynamic processes and clogging were determined.

Oncorhynchus mykiss and a single individual of *Salvelinus fontinalis* were detected as alien species for this area. Site 3, 4 and 5 of the River Mur are located in the hyporhithral fish region. Based on historical records as well as expert knowledge only six fish species were evaluated as present in the historical condition (Woschitz et al. 2007). *Oncorhynchus mykiss* and *Salvelinus fontinalis* were detected as alien species for this area. Site 4 received the lowest evaluation attributed to continuous bank stabilisation (high percentage of technical installations) and absence of structures. Beside this, all sites show typical key species and good populations. Only *Thymallus thymallus* is under represented. Small scale habitats in terms of sand- and mud banks are available and used by Ukrainian brook lamprey throughout the study sites despite the river regulation.

Table 8.6 Comparison of presence and absence of fish species at the study sites 1 and 2 at the River Mur. Note data are limited to snap shot surveys.



scientific name	species	Upper MUR - Site 01		Upper MUR - Site 02	
		metarhitral		metarhitral	
		historical	2011	historical	2011
Thymallus thymallus	european grayling	✓	✓	✓	✓
Salmo trutta fario	brown trout	✓	✓	✓	✓
Phoxinus phoxinus	eurasian minnow	✓	✗	✓	✗
Cottus gobio	bullhead	✓	✓	✓	✓
Eudontomyzon mariae	ukrainian brook lamprey	✓	✓	✓	✓
Oncorhynchus mykiss	rainbow trout	✗	✓	✗	✓
Salvelinus fontalis	brook trout	✗	✓		

Table 8.7 Comparison of presence and absence of fish species at the study sites 3,4 and 5 at the River Mur. . Note data are limited to snap shot surveys.

scientific name	species	Upper MUR - Site 03		Upper MUR - Site 04		Upper MUR - Site 05	
		hyporhitral		hyporhitral		hyporhitral	
		historical	2011	historical	2011	historical	2011
Thymallus thymallus	european grayling	✓	✓	✓	✓	✓	✓
Salmo trutta fario	brown trout	✓	✓	✓	✓	✓	✓
Phoxinus phoxinus	eurasian minnow	✓	✓	✓	✗	✓	✓
Barbatula barbatula	stone loach	✓	✗	✓	✗	✓	✗
Cottus gobio	bullhead	✓	✓	✓	✓	✓	✓
Eudontomyzon mariae	ukrainian brook lamprey	✓	✓	✓	✓	✓	✓
Oncorhynchus mykiss	rainbow trout	✗	✓	✗	✓	✗	✓
Salvelinus fontalis	brook trout	✗	✓	✗	✓	✗	✓

8.3.3 River Drau Case study

8.3.3.1 Timeline

The first channelisation project for flood protection reasons was introduced in the 1880s at the River Drau. Embankments were established shortly before these measures in the course of train track constructions upstream of the study site, Figure 8.14. These alterations caused higher flow velocities, water depths and a higher downward gradient due to the shortened river course, even though the alteration is not as significant as at the study sites Traun and Mur. The River Drau is classified as a natural river and evaluated as natural fish habitat with a good chemical and ecological condition.

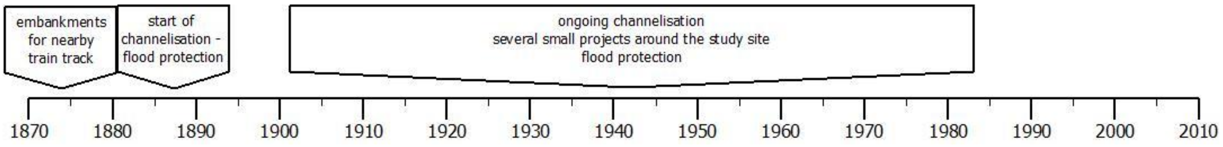


Figure 8.14 Timeline of significant changes of the River Drau

With regards to the lists of hydromorphological processes and pressures from Deliverable 1.2 the following multiple-pressures affect this river section at the River Drau:

- Channelisation – cross section alteration
- Embankments, levees or lateral dikes

8.3.3.2 Hydromorphology changes

River Drau terrain models and hydraulic parameters of a degraded river section served as basis for the hydromorphological analysis of the River Drau. The overall site length is approximately 2 km. For the purpose of this analysis, the hydromorphological condition was investigated only for a degraded condition at three different flow situations. No historical hydromorphological dataset was available. Modelling results for water depth and flow velocity distributions at mean flow are shown in Figure 8.15. In the study site, areas of shallow water depth are almost completely missing in the distribution map. Due to the river regulation measures, the monotonous channel is dominated by high water depths in combination with high flow velocities. This combination occurs mainly in the main flow direction, especially in the outside of the river bends.

The lack of opportunity for self-dynamic river development leads to a limited variety of aquatic habitats. Thus, the canal-like character of this river reach causes altered habitat conditions for aquatic organisms.

The hydrology of the River Drau at Sachsenburg (Brücke) is mainly influenced by nival and glacial flow characteristics of the Alps. In Figure 8.16, the mean monthly hydrographs show a change of flow in the course of time. Since the installation of water abstraction in 1959, 1969 and 1984 the course of the mean monthly hydrographs changed rapidly. The mean monthly flows are clearly reduced in summer and autumn, while a significant decrease of the peak value in June has appeared. In the winter months, there were hardly any changes in the flow progress over the observed period.

The flow regime was determined as perennial snowmelt based on the time series from 1951 to 2010. The maximal flood events over threshold and characteristic monthly flows are illustrated in Figure 8.17.

8.3.3.3 Fish community Changes

At the study site on the river Drau only 2 (*Thymallus thymallus* and *Salmo trutta fario*) out of five key species for this fish region (hyporhithral) were detected. *Oncorhynchus mykiss* were found and classified as alien species for this area. Only two out of the 22 recorded fish species in the historical condition (Unfer et al 2004) were found when sampling the site in 2011. 20 of the fish species recorded in historically disappeared in the study site area.

Investigations have shown that the number of insect larvae and small crustaceans, the food supply for certain fish species, decreased dramatically in recent decades. In upstream tributaries fine glacial striation is deposited in the storage reservoirs of power plants and transported into the river Drau when the reservoirs are flushed. These fine sediments clog the gravel banks and this reduces the availability of spawning habitats. These factors influence mainly spawning by *Thymallus thymallus* but also *Eudontomyzon mariae, Hucho hucho, Cottus gobio, Telestes souffia*, spined loach and others. Only the population of *Salmo trutta fario* increased within the last decades. Juvenile stages of *Hucho hucho, Chondrostoma nasus, Lota lota* and *Thymallus thymallus* are endangered by desiccation of tributaries. Reasons for this may be water extractions in upper regions of tributaries for agricultural reasons as well as hydrological reasons (lower precipitations). The loss of habitats and structures caused by channelisation, the alteration of tributaries and clogging caused by glacial striation were detected as the main reasons for the reduction and negative impacts on fish communities. Beside this impacts stocking with alien fish species and populations (*Oncorhynchus mykiss, Salvelinus fontinalis*), predation pressure by *Phalacrocorax carbo*, extensive fishing, loss of floodplains and others cause negative effects on the fish communities in the upper Drau (Uiblein et al. 2002).

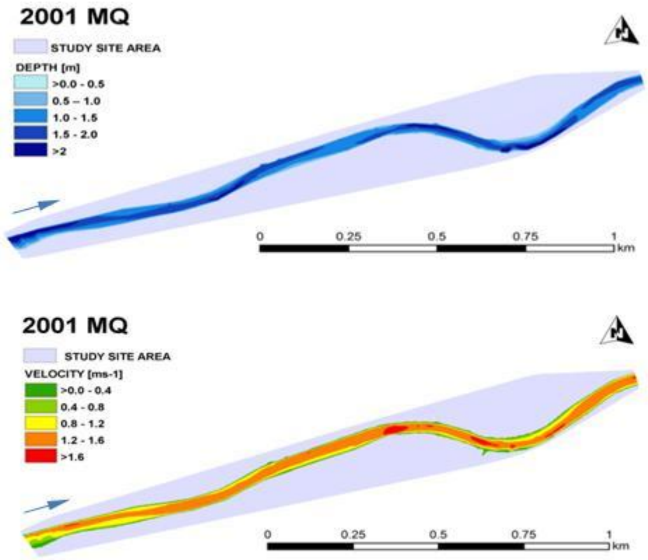


Figure 8.15 Depth and velocity for MQ in 2001 in the River Drau.

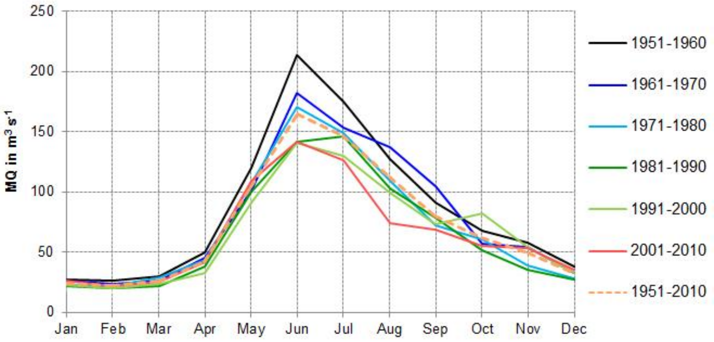


Figure 8.16 Mean monthly hydrographs of River Drau at Sachsenburg (Brücke) (dashed line - whole observation period, continuous lines – decades) (Lebiedzinski et al. 2014).

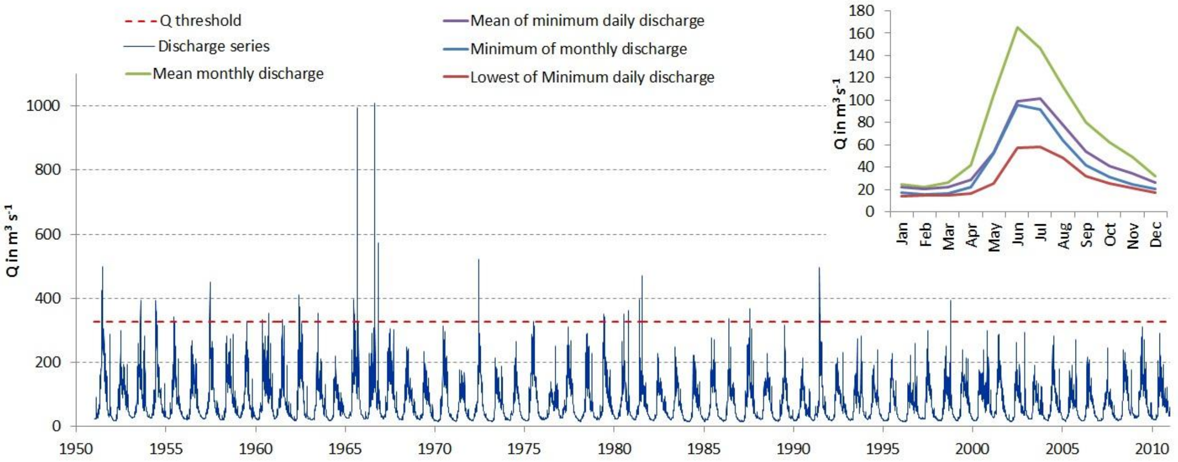


Figure 8.17 Flow regime analysis of river Drau at Sachsenburg (Brücke) - maximal flood events over threshold and characteristic monthly flows (1951-2010) based on the method in Annex C D2.1 (Gurnell et al. 2014b).

Table 8.8 Observed effects of the alteration of HYMO processes on the probability of occurrence of fish species in the validation dataset in rivers Drau (see chapter 6.2 for method). Names in brackets indicate fish species that have been extirpated from the community.

River	Species	vertical connectivity	water flowing	sedimentation	vegetation encroachment	aquifer recharge	vegetation recruitment	sediment transport	sediment entrainment	large wood entrainment	large wood transport	armouring	vegetation derooting	large wood deposition	bank stabilization	bank erosion and failure
Drau	*[Alburnus alburnus]*		-1	1	-1			-1	1			1			0	
	[Barbatula barbatula]		-1	1	-1			-1	1			1			0	
	[Carassius carassius]		-1	1	-1			-1	1			1			0	
	[Chondrostoma nasus]		-1	1	-1			-1	1			1			0	
	[Cobitis elongatoides]		-1	1	-1			-1	1			1			0	
	[Cottus gobio]		-1	1	-1			-1	1			1			0	
	[Eudontomyzon mariae]		-1	1	-1			-1	1			1			0	
	[Gobio gobio]		-1	1	-1			-1	1			1			0	
	[Hucho hucho]		-1	1	-1			-1	1			1			0	
	[Lota lota]		-1	1	-1			-1	1			1			0	
	Oncorhynchus mykiss		1	-1	1			1	-1			-1			0	
	[Perca fluviatilis]		-1	1	-1			-1	1			1			0	
	[Phoxinus phoxinus]		-1	1	-1			-1	1			1			0	
	[Rutilus rutilus]		-1	1	-1			-1	1			1			0	
	Salmo trutta fario		0	0	0			0	0			0			0	
	[Squalius cephalus]		-1	1	-1			-1	1			1			0	
	[Telestes souffia]		-1	1	-1			-1	1			1			0	
	Thymallus thymallus		0	0	0			0	0			0			0	
	[Tinca tinca]		-1	1	-1			-1	1			1			0	

Table 8..9 Comparison of presence and absence of fish species at the River Drau study site.

scientific name	species	Case Study Site DRAU	
		historical	2011
Lota lota	burbot	✓	✗
Squalius cephalus	european chub	✓	✗
Thymallus thymallus	european grayling	✓	✓
Salmo trutta fario	brown trout	✓	✓
Barbatula barbatula	stone loach	✓	✗
Barbus barbus	barbel	✓	✗
Abramis brama	common bream	✓	✗
Phoxinus phoxinus	eurasian minnow	✓	✗
Perca fluviatilis	eurasian perch	✓	✗
Gobio gobio	gudgeon	✓	✗
Esox lucius	northern pike	✓	✗
Hucho hucho	danube salmon	✓	✗
Carassius carassius	crucian carp	✓	✗
Cottus gobio	bullhead	✓	✗
Alburnus alburnus	bleak	✓	✗
Chondrostoma nasus	nase	✓	✗
Eudontomyzon mariae	ukrainian brook lamprey	✓	✗
Rutilus rutilus	roach	✓	✗
Scardinius erythrophthalmus	rudd	✓	✗
Tinca tinca	tench	✓	✗
Cobitis elongatoides	spined loach	✓	✗
Telestes souffia	sourie, vairone	✓	✗
Oncorhynchus mykiss	rainbow trout	✗	✓

8.4 Discussion

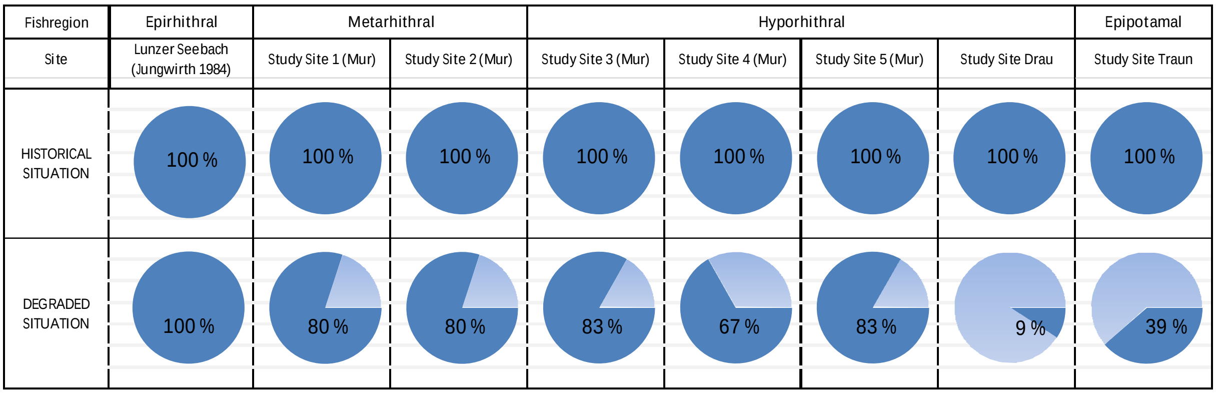
One of the overall goals of the Water Framework Directive is to maintain and enhance the biological status of rivers in all member states of the EU by 2027. The negative impacts of river channelisation and multi pressure caused HYMO alterations on fish populations are well know described for several decades and documented by various authors (Gorman and Karr 1978; Bless 1981; Karr and Schlosser 1978; Larimore and Smith 1963; Lelek and Lusk 1965; Sheldon 1973). In certain cases, alterations show a dramatic reduction of fish stocks of up to 95% regarding stock density and biomass (Jungwirth and Winkler 1983).

One of the most common causes of fish species decline is habitat alteration caused by river channelisation. Although one or two principal causes of decline can be identified for each species, the decline is typically the result of multiple, cumulative, long-term effects (Moyle and Leidy 1992). Literature research on river bed incision shows the long term impacts interferes with other environmental factors and ecological processes. Caused by multiple stresses the ecosystem responses to river incision, especially over long time scales, are not deterministic. Effects may be compensated by other processes or might be amplified (Bravard et al. 1997)

Since the 18th century it has been documented, that the construction of embankments leads to a clear reduction of dynamic habitats in braided rivers, (Stummer 1982) like the

Traun study site. With increasing degrees of channelisation, the variability of characteristic parameters like water depth and variance of river width are decreasing drastically (Jungwirth 1986). Reduced spatial heterogeneity due to river straightening resulted in decreasing species number, diversity, stock density and biomass (Jungwirth et al. 1995). Smaller habitat complexities in channelised river reaches lead to lower biomass values (Jungwirth et al. 1995; Oscoz et al. 2005; Penczak 1995). Further factors suggested to be responsible for reducing fish biomass in channelized streams are: reduced space, loss of riffles, decreased habitat diversity, habitat instability in terms of substrate, water levels and water temperature (Portt et al., 1986). Results from the River Mur sites 1 and 2 show clearly, that increasingly monotonous river habitats due to regulation measures such as channelisation result in unbalanced population structures in epirhithral or metarhithral rivers while it results in loss of species diversity in hyporhithral or epopotamal rivers (Jungwirth et al. 2003) such as seen out of results of study sites at River Traun, River Drau and River Mur, sites 3, 4 and 5).

Figure 8.18 illustrates the reduction of the number of fish species comparing the natural historical situation with 100 % presence of the number of fish species with the number of fish species in the degraded situation of all surveyed study sites fish regions from metarhithral to epipotamal. The case study of the Lunzer Seebach, a small river in the epirhithral fish region (Jungwirth 1984) is added.



Fishregion	Epirhithral	Metarhithral		Hyporhithral				Epipotamal
Site	Lunzer Seebach (Jungwirth 1984)	Study Site 1 (Mur)	Study Site 2 (Mur)	Study Site 3 (Mur)	Study Site 4 (Mur)	Study Site 5 (Mur)	Study Site Drau	Study Site Traun
HISTORICAL SITUATION	100 %	100 %	100 %	100 %	100 %	100 %	100 %	100 %
DEGRADED SITUATION	100 %	80 %	80 %	83 %	67 %	83 %	9 %	39 %

Figure 8.18 Single results from alterations on percentage reduction of the number of fish species caused by hymo pressures

As shown in the case study of natural conditions of River Traun, typically, natural sections have a high variety of widths, depths, substrates and velocities and well developed riffle and pool sequences in the main river and in side-arms caused by natural HYMO processes. On the other hand, channelisation causes monotonous stream width, depth and substrate with much less opportunities for fish species diversity. The disappearance of very high percentages of fish species at the Traun study site in the epipotamal fish region and on the River Drau in the hyporhithral fish regions caused on the one hand side by the loss of wetted area as well as the determined change of depth and velocity distributions at habitat relevant low flows (MJNQ$_t$) and at the other hand side by the alteration of the hydrograph had a cumulative effect of changes in land use, reduction of riparian forests and retention areas, systematic regulation and channelisation of tributaries and the effects of climate change. In opposition to that, a clear alteration of HYMO processes in epirhithral and metarhithral fish regions mainly results in an alteration of the length frequency distributions. The reduction of morphological variability, the loss of riverine habitats and structures and the long term effects of missing natural HYMO processes of substrate erosion and deposition caused by river channelisation and stabilisation of the two metarhithral study sites at the River Mur and the Lunzer Seebach in the epirhithral fish region reveals mainly significant reduction of stock density and biomass of fish species whereby river regulations primarily cause loss of juveniles and adults remaining the middle age category. Juvenile fish are more sensitive to habitat changes than adults (Blaxter 1974; Jurajdy et al. 2001). The correlation of variance of maximum depth and fish length was already detected by Jungwirth (Jungwirth 1984, 1981). The losses of fish species ranged between close to or lower than approximately 25% in the metarhithral region to zero in epirhithral fish region. These findings out of detailed case study analyses appear to be consistent with other studies (Wyzga et al. 2014). Reduced cross-sectional variation in (near-bed) flow velocity and bed material size in channelized cross-sections of a gravel-bed mountain river was reflected in no difference in the species richness of fish assemblages, although the channelised cross-sections supported significantly lower number of both juvenile and older fish individuals(Wyzga et al. 2014).

In conclusion, the available limited number of highly detailed survey results show that fish presence or absence is a potential useful indicator in hyporhithral and potamal river systems but that it must be understood in terms of the impact human alteration to a channel are having on dynamic fluvial processes. To evaluate the situation in epirhithral systems with just one to two fish species and metarhithral systems with less than five to six fish species is not recommended and more detailed investigations on length frequency distributions should be undertaken. Data of the species composition of the fish community and the presence of HYMO process alteration (qualitative data: increase, decrease or absence) for these three rivers between historical (1885) conditions and current (2006) conditions are available from records from electrofishing and detailed analyses of HYMO parameters at 7 case study sites. This makes them useful for validation purposes. The results of this validation exercise shows that salmonids like *Oncorhynchus mykiss* or *Hucho hucho*, did perform well when predicting the effects observed in the case studies. On the contrary, species that showed low consistency in their response to HYMO pressures in multiple stressor scenarios, for example, *Eudontomyzon mariae*, *Telestes souffia*, *Alburnoides bipunctatus*, *Salmo trutta* and *Thymallus thymallus*, are the least scored when contrasted to the case study (the last three species even failed to respond to the observed changes in the processes). Therefore, the consistency of its responses in the conceptual schemes (see chapter 6) can be used as a measure of the expected performance of a given species as indicator species of HYMO processes alteration in multi-stressor scenarios.

8.5 Glossary

Abbreviations for different flows referring to the Austrian standard document ÖNorm 2400:

NQ	low flow; lowest flow value in the observed period
$MJNQ_t$	mean annual daily low flow on an annual series t

MQ	mean flow; mean flow value in the observed period
HQ	flood; highest flow value in the observed period
HQ1	annual flood, 1-year flood event; flood event with 1 year return period
HQ10	10-year flood event; flood event with 10 years return period
HQ100	100-year flood event; flood event with 100 years return period

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APPENDICES including 6 FIGURES can be found at the end of the document.

9 Invertebrate diversity along a stream morphological gradient

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9.1 Introduction

Streams and river ecosystems are among the most impacted on earth, with human-induced changes probably resulting in irreparable changes within the next twenty years (e.g. Malmqvist and Rundle 2002). Rivers and streams are mainly affected by habitat modification (hydrological changes, dams and weirs), water chemistry alteration (nutrient additions, acidification, metals, and organic compounds), and introduction or removal of species. In the EU Water Framework Directive [WFD] (2000/60/EC Establishing a Framework for Community Action in the Field of Water Policy) the classification of ecological status is based on the assessment of organism community composition and diversity. For running waters, the focus of the WFD is on fish, macrophytes, benthic macroinvertebrates, and phytobenthos. These organisms are generally believed to respond to pressures on different spatial scales, with fish (e.g. Snyder et al. 2003; Hoeinghaus et al. 2007) responding to large-scale catchment pressures, macrophytes to a combination of regional factors and in-stream physical variables (e.g. Baattrup-Pedersen et al. 2003) and alterations, whereas benthic macroinvertebrates (e.g. Vinson and Hawkins 1998; Sandin and Johnson 2004; Sandin 2009) and phytobenthos (e.g. Coring 1999) are thought to respond more directly to local scale variability such as water chemistry and habitat quality (but see Johnson et al. 2007).

The internationally developed biodiversity indicators for freshwaters usually do not measure biodiversity in terms of species richness, but instead measure threats, including connectivity/fragmentation of rivers, and water quality in aquatic ecosystems such as the Streamlining European 2010 Biodiversity Indicators (European Environment Agency unpublished). The European Environmental Agency core set of indicators does not contain any freshwater indicators for biodiversity (European Environment Agency 2005), but the European Environment Agency suggests the use of salmon (*Salmo salar* L.) as an indicator of river health (European Environment Agency, 2006). The rationale for not including direct measurements of change in freshwater biodiversity, but instead measuring threats (such as fragmentation and deterioration in water quality) are: i) "…weaknesses in direct measurements of biodiversity", and ii) "the subject of priorities for direct indicators of biodiversity versus indirect indicators (of threats) has not been systematically addressed for inland waters" (Convention on Biological Diversity 2006). Setting priorities for such indicators requires a mechanistic understanding between threats (stressors) and biodiversity. While the implementation of the Water Framework Directive has gone a considerable way in addressing such issues there are still outstanding questions. The topic of this deliverable is to understand the biological response of river organisms to the interactions between stresses where at least one of those stressors is a form of hydromorphological degradation. As many European waters are subject to multiple stress this is a pragmatic approach. The multiple stresses we address here are nutrient stress and alterations to instream habitat complexity as a means of assessing hydromorphological alterations to the in-stream environment. Nutrient stress is thought to effect macroinvertebrate communities by homogenizing the benthic macro invertebrate communities, whereas altering instream habitat complexity and variability is also influencing both the biotic patterns as well as processes in stream ecosystems.

The aim of this deliverable is to describe how multiple stressors impact benthic macroinvertebrate traits and thus identify potential trait based indicators to assess mainly stream morphological stress effects on stream biota. We take a trait based approach to be able to develop indicators which will potentially be usable across large spatial scales (such as the countries within EU) as opposed to existing benthic macro invertebrate based indicators which much generally be adapted to be usable across larger spatial scales.

Using benthic macroinvertebrate, data from 226 streams sites across central and northern Europe we address the following questions:

i) Is there a link between habitat/biotope diversity and macroinvertebrate diversity?

ii) Will the use of traits and trait diversity alter this relationship?

Our aim is to elucidate general patterns relating to the relationship between the assessment of habitats/biotopes and the diversity of macroinvertebrates using a standardised biomonitoring sampling methodology. We use a high quality dataset that includes a pre-defined gradient in hydromorphological degradation. In addition to identity-based diversity metrics we will use traits, aiming to identify candidate traits, or combinations of traits, that are particularly sensitive to hydromorphological stress.

9.2 Methods

9.2.1 Study area

Two-hundred and twenty six stream sites (in different streams and catchments) were sampled for benthic macroinvertebrates in the spring of 2002 as part of the EU-funded STAR project (Furse et al. 2006). The stream sites were situated in ten countries (Austria, Czech Republic, Denmark, France, Germany, Latvia, Poland, Slovakia, Sweden and the UK) with 124 small-sized and 102 medium sized streams, with 126 one streams sampled at below 200 m.a.s.l. and 100 stream sites sampled at an altitude between 200 and 800 m.a.s.l. The stream sites were chosen *a priori* to constitute a hydromorphological or nutrient enrichment gradient using existing data of stream water chemistry as well as expert opinion from Local County Board managers. This also included possible reference sites (high status or minimally disturbed), with low land use practice in the catchment, natural river channels and habitats, as well as riparian vegetation and each site was classified *a priori* into one of the ecological status classes from ´high´ to ´bad´ (see Furse et al. 2006 for a full description of reference criteria). Generally, only environmental and not biological (organismal) variables were considered in stream selection. A total of 68 sites out of the 226 used in the study were pre-classified from ´high´ to ´bad´ with degradation in stream morphology as the dominant stressor (as opposed to organic pollution). These 68 sites were also used in further steps in the present study to assess the relationship between morphological effects on stream ecological classification and substratum as well as biological diversity.

We used the assessment of habitats that are part of the AQEM-STAR sampling method for macroinvertebrates and which decides where kick-sampling is undertaken (details below). The rationale for using this assessment rather than the River Habitat Survey (RHS), which was employed on all reaches in the STAR-project, was two-fold: i) a tighter coupling between habitats assessed and macroinvertebrate composition in the samples as they are taken proportional to presence of habitats ii) the linkage between RHS scores and macroinvertebrates, both using various metrics and traits, are shown to be weak (Friberg et al. 2009; Feld et al. 2014) and at least partly due to differences in sampling scales.

9.2.2 Field sampling

Benthic macroinvertebrates were sampled once in each stream in the spring of 2002. In each stream, a sampling section of 500 m was chosen and all benthic macroinvertebrate samples and water chemistry measurements were taken within this area. The reach was chosen to be as homogeneous as possible regarding riparian vegetation and surroundings (i.e. c. 50 meter perpendicular to the stream). The length of the sampling reach for macroinvertebrates were dependent on stream size and was approximately 20-50 m in small (1-100 km2 catchments) streams and 50-100 m in length in medium (100-1000 km2 catchments) streams. The sampling always covered the entire stream width and was representative in terms of habitat composition of the larger 500 m reach in which the macroinvertebrate sampling was nested.

Benthic macroinvertebrates were sampled at each reach using the standardised STAR method in which a sample consists of 20 sub-samples taken from all microhabitat types with a share of at least 5% coverage. Each sub-sample was obtained by positioning a standard hand-net (0.25 x 0.25 m frame; 0.5 mm mesh size) randomly within a given micro-habitat type and then disturbing the substrate for a distance that equals the square of the frame width upstream of the net. This procedure resulted in approximately 1.25 m^2 of the stream bed being sampled. The 20 sub-samples were distributed in accordance with the proportional coverage of microhabitats that was assessed prior to sampling of macroinvertebrates. All microhabitats with at least 5% cover were recorded to the nearest 5% interval. Mineral substrates would always add up to 100% while the biotic microhabitats overlaying these would range between 0 and 100 %. The mineral substrates were divided into the following classes: boulders (>40 cm); stones (>20-40 cm); cobbles (>6-20 cm); pebbles (>2-6 cm); gravel (>0.2-2 cm); sand (>6 µm-2 mm) and silt (<6 µm). The biotic microhabitats consist of the following: algae (filamentous; turfs); submerged macrophytes; emergent macrophytes; living terrestrial plant parts (e.g. tree roots); dead wood; CPOM; FPOM and mud. Samples were preserved in ethanol and returned to the laboratory for sorting and identification. Individuals were mostly identified to species, but some to genus, family or higher taxonomic levels, depending on the availability of published keys and the condition of the animals. Analyses of water chemistry (total nitrogen, total phosphorous, NO2+NO3, NH4-N, and PO4-P) followed international (ISO) or European (CEN) standards where applicable.

9.2.3 Traits information

We used 20 traits with a total of 107 trait modalities in the study based on information on genera of benthic macroinvertebrates (Usseglio-Polatera et al. 2000).

9.2.4 Statistical tests

The mineral substratum classification was used as a basis for measurements of substratum diversity, which was calculated into five different mineral and five organic substratum diversity indicators. These were: i) number of substratum types found at the site, ii) Simpsons diversity, iii) Shannon-Wiener diversity, iv) Evenness, and v) a Mineral and Organic index based on summing the presence of each coarse substratum type (hygopteric, megalithal, macrolithal, mesolithal, microlithal) present at the site scoring +1 with akal (sand) scoring 0 and with the negative mineral substratum types (psammal/psammalopelal and argyllal) present scoring -1. In the same way the organic indicator is based on the presence of coarse organic material such as submerged and emergent macrophytes, living parts of planrs, xytal and Coarse Particulate Organic Matter present scoring +1, Fine Particulate Organic Matter present scoring 0 and organic mud scoring -1.

The biological diversity of the benthic macroinvertebrates were in the same way used to calculate four biodiversity indicators: i) number of taxa, ii) Simpsons diversity, Shannon-Wiener diversity, and iv) Evenness.

All statistical testing was done using Single Linear Regression models with water chemistry or substratum diversity as the independent and in most cases the benthic macroinvertebrate diversity or trait as the dependent variable. We also tested for the relationships between the ten mineral and organic substratum diversity indicators and the 21 trait variables with a total of 216 trait modalities using Type III ANOVAs where each of the combinations were tested individually.

We tested in total 11 biological traits and 9 ecological traits. As we were mainly interested in whether specific trait modalities were related to the substratum diversity variables, we did not compensate for the large number of tests by adjusting the p-values, but rather used $p<0.05$ as a way to get a general idea of which traits modalities were related to substratum diversity and thus potentially interesting when developing a macroinvertebrate trait indicator of stream substratum morphological stress. We tested in total 11 biological traits and 9 ecological traits.

9.3 Results

9.3.1 Stressor interactions: Water chemistry – substratum diversity relationships

Across the dataset as a whole there were no statistical differences in water chemistry between small and medium sized streams in the dataset for nitrate (189 obs) or total phosphate (190 observations). The same was true for the two altitudinal typology classes. Number of mineral substratum types, mineral Simpson and Shannon substratum diversity, mineral index was statistically correlated with nitrate, and total P ($p<0.01$) but the adjusted r^2 value was low in all cases ($r^2<0.016$).

However , a test of a subset of the data (Swedish and Danish sites) showed signs of a synergistic effect of substratum complexity and phosphorous level on total number of macroinvertebrate taxa, (Figure 9.1). The sampled sites were divided into those with a low total P concentration of (<25 µg l^{-1}) and those defined as having a high total P concentration (>85 µg l^{-1}). When comparing taxon richness at sites with a simple versus complex substratum composition and at the same time a low total P concentration, there were no difference between number of taxa (Figure 9.1 left panel). When on the other hand the same comparison was made for sites with a high total P concentration there was a clear statistical significant difference in number of macro invertebrate taxa found, i.e., 7.9 ±0.33 (Std Error) at the complex sites, whereas the sites with simple substratum types has a mean of 3.08±0.32 taxa ($p<0.0001$), (Figure 9.1, right panel).

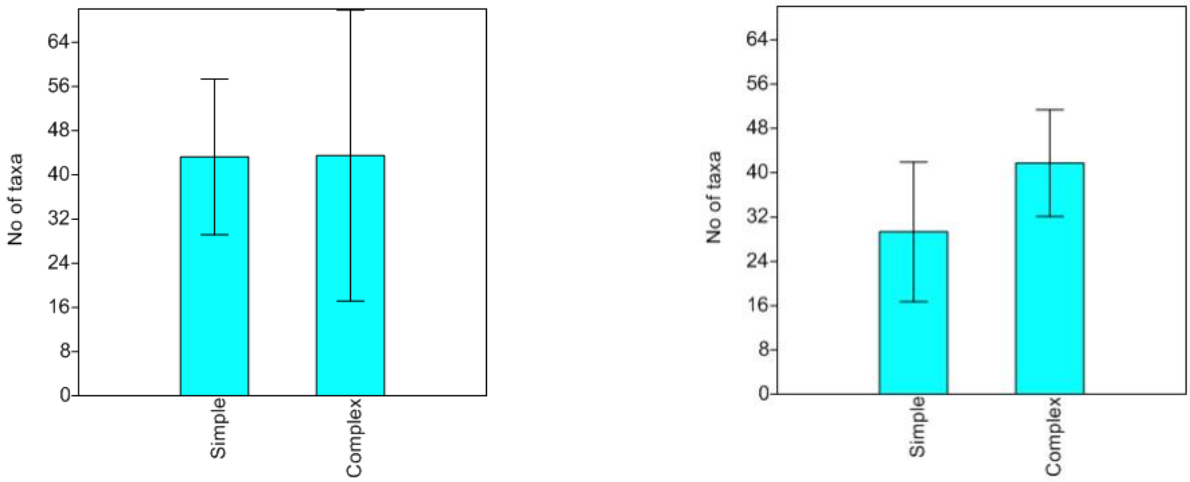

Figure 9.1 Left panel represent sites with low total phosphorous concentrations (<25 µg l^{-1}) and the right panel high phosphorous concentrations (>85 µg l^{-1}) in stream water samples. Stream sites were grouped as either simple, i.e. with few substrate types (inorganic only) and complex with a high number of different substrate types.

9.3.2 Substratum diversity

Each sampled site had between one and seven substratum types recorded. The substratum diversity differed in some cases between the small and the medium sized streams/rivers with a higher mineral Simpson, Shannon-Wiener diversity and Evenness in the medium sized rivers ($p<0.05$ in all cases), whereas no such differences were found for the organic substratum diversity. The mineral substratum diversity also differed among the two altitudinal stream types with a higher mineral Simpson diversity and Evenness ($p<0.05$ in both cases) at the sites < 200 m.a.s.l. compared to the sites at 200-800 m.a.s.l. No such differences were found for the organic substratum diversity ($p>0.05$ in all cases).

9.3.3 Biological diversity

Total number of taxa were statistically significantly higher in small-sized compared to medium-sized streams ($p<0.001$), no other differences in biological diversity was found in relation to stream size. The same was true for the altitudinal typology where sites at

200-800 m.a.s.l. had a higher taxonomic richness ($p<0.0005$) compared to sites at <200 m.a.s.l, none of the other measurements of biological diversity differed between the two altitudinal typology groups.

9.3.4 Hydromorphological pre-classification

There were 68 sites that were included in the STAR project because the dominant stressor was degradation in stream morphology. For these sites there were no statistical difference in any of the substratum diversity measurements included in the study either for mineral or organic substratum ($p>0.05$ in all cases). For the biological diversity on the other hand, it was clear that the ´bad´ status based on degradation in stream morphology had lower number of taxa (Figure 9.2), Simpsons and Shannon-Wiener diversity as well as Evenness ($p<0.005$ in all cases). Using Tukey-Kramer HSD tests it was clear that it was the ´bad´ category that differed from the four other categories.

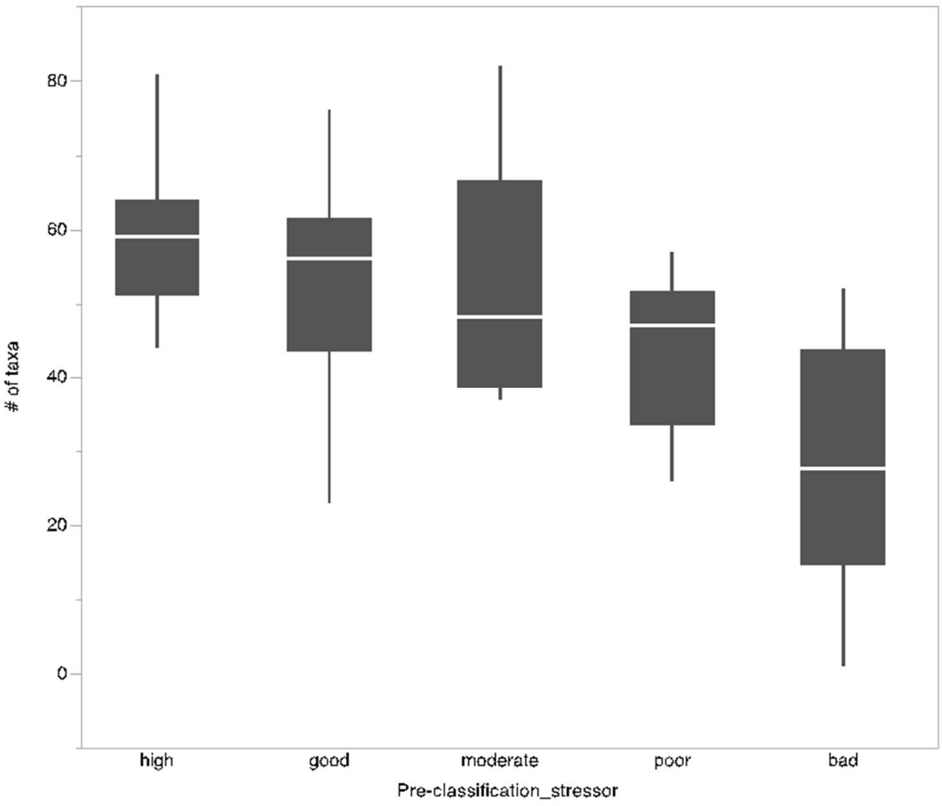

Figure 9.2 Relationships between pre-classified hydromorphological status of the streams and number of taxa of benthic invertebrates.

9.3.5 Relationships between substratum and biological diversity

We tested the relationship between substratum diversity, both mineral and organic and biological diversity using simple linear regressions. There were statistically significant relationships in some instances, but these were always statistically non-significant with very low r^2 values. When analysing the 68 sites that had been sampled with a focus on degradation in stream morphology there were one statistically significant relationship between substratum evenness and biological diversity and no ecologically relevant relationships.

9.3.6 Relationships between substratum diversity and traits

We tested the relationship between the modality traits and the substratum diversity categories using ANOVA models, where the substratum categories (combined) could explain between 16.7 and 5.6% of the variation in the modalities assessed have a statistical relationship. This was based on the inclusion of all statistically significant substratum diversity variables in the models (Table 9.1).

The relationship between the biological trait maximum adult size was clearly related to the organic substratum indicators, especially number of organic substratum types and less so with the mineral substratum indicators. There was also a clear pattern that the smallest organisms responded most strongly to substrate diversity, both mineral and organic.

Testing the traits life cycle duration and potential number of reproductive cycles per year showed generation time longer than one year as well as organisms with more than one generation time per year were both related to most organic substratum diversity types and with some of the mineral substratum diversity types as well as in most cases a higher substratum diversity was negatively related to abundances of taxa with a certain life cycle duration as well as number of reproductive cycles per year.

With regard to the trait aquatic stages the modality adult, i.e. where the organism has its adult life stages in the water (in contrast to most insects), was related significantly to all but three of the mineral substratum diversity indicators, whereas larva and egg was related to most organic substratum diversity indicators.

The relationship between the trait reproduction and substratum indicators showed that the trait free isolated eggs were correlated with most substratum indicators as was the indicator asexual reproduction (Appendix Table 4). In most cases there were a negative correlation between substratum diversity and abundances of taxa with isolated eggs. With regard to the trait dispersal the modalities passive and active dispersal were both strongly related to most organic substratum diversity types. The trait resistance form showed that the modality of forming cocoons as a protection strategy against drought was strongly correlated with most substratum diversity indicators, while the building of housing was related to a few of the organic substratum indicators. When testing the trait respiration only the modality relating species with tegument respiration, i.e. by diffusion over the body cells, was correlated with mainly organic substratum indicators, whereas none of the other breathing apparatus traits were related to substratum diversity.

Several of the modalities related to the trait locomotion and substrate relation, i.e., full water swimmer, burrower, interstitial, and permanently attached was correlated with most of the substratum diversity indicators. There were nine trait modalities related to trait *food* of the benthic macroinvertebrates, only three of them were statistically significantly related to the substratum diversity traits. Several of the modalities that belong to the trait feeding habits were related to the substratum diversity types with absorbers, deposit feeders, and scrapers were the main ones. Several of the modalities related to the ecological trait transversal distribution were related to mainly the organic substratum diversity indicators, whereas the trait modality groundwaters were related to most of the substratum indicators.

Similarly, most of the modalities related to the trait longitudinal distribution were correlated with the organic substratum diversity indicators. With regard to the ecological trait altitude the lowlands modality was related to most organic substratum indicators with piedmont and alpine modalities being related to one or two of the organic indicator(s). Many of the modalities related to the trait preference substratum were related to mainly the organic substratum diversity indicators but also a few mineral indicators. The modality slow velocity preference belonging to the traits preference current velocity was related to most organic substratum diversity types, whereas the modalities medium and fast stream flow was related only to number of organic substratum types.

Modalities mesotrophic and eutrophic nutrient status (trait: trophic status) were related to organic substratum indicators. Whether the organisms had a preference for brackish water or not (trait: *salinity*) was related to the organic substratum diversity indicators. The modality thermohilic (i.e. organisms that thrive at high temperatures), belonging to the trait preference temperature was correlated with most substratum diversity indicators. Beta and alpha mesosaprobic modalities (trait: saprobity) were correlated with substratum diversity indicators, mainly for the organic substratum types.

Table 9.1 List of traits and modalities that showed a statistical significant relationships with the substratum diversity. R2 values are calculated using a ANOVA model including all statistically significant substratum diversity variables.

Traits	Modalities = Categories	r^2 - values
Body size	≤ .25 cm	0.16
	> .25-.5 cm	0.12
	> .5-1 cm	0.056
	> 1-2 cm	0.07
Life cycle duration and generation time	> 1 year	0.15
	> 1	0.12
Aquatic stages	adult aquatic life stage	0.16

Traits	Modalities = Categories	r^2 - values
Reproduction	isolated eggs, free	0.16
	isolated eggs, cemented	0.08
Dispersal	aquatic passive	0.11
	aquatic active	0.11
Resistance form	cocoons	0.15
Respiration	tegument	0.09
Locomotion and substrate relations	full water swimmer	0.14
	burrower	0.15
	interstitial	0.14
	permanently attached	0.16
Food	microorganisms	0.16
	detritus < 1mm	0.11
Feeding habits	deposit feeder	0.15
Transversal distribution	groundwaters	0.16
Longitudinal distribution	crenon	0.12
	estuary	0.16
Altitude	lowlands	0.09
Preference substratum	gravel	0.12
	sand	0.13
	silt	0.16
	microphytes	0.14
	mud	0.15
Preference current	slow	0.09
Trophic status	mesotrophic	0.12
	eutrophic	0.10

Traits	Modalities = Categories	r^2 - values
Salinity	brackish water	0.14
Preference temperature	thermophilic	0.14
Saprobity	a-mesosaprobic	0.14

9.4 Discussion

We found a clear trend in a small data set that nutrient status (phosphorous (P) concentration) and habitat complexity interacted, where sites with a combination of high P levels and low complexity in average had lower macroinvertebrate diversity. In the analysis of the larger data set we could not find any clear trends between chemistry measured and hydromorphological variables with all correlations being very weak. We could therefore not pursue questions relating to how the two stressors interact and focused on testing the relationships between our measures substrate diversity with macroinvertebrate diversity, both identity and trait based. This will naturally introduce additional variability related to e.g. nutrients that will make relationships weaker. Therefore, we will also evaluate consistent patterns in the direction of relationships (e.g. consistently negative relationships between all hydromorphological variables and a given trait), although the individual relationships might relatively weak as the level of statistical significance.

We found a negative relationship between number of taxa and the pre-classification of stream sites along a hydromorphological degradation gradient. It was, however, also clear that this relationship does not appear linear but may reach an asymptote with the range of poor-high hydromophological quality only differing in average by 10 taxa, while the difference between poor and bad was almost 20 taxa in average. However, strikingly, there was no relationship between substrate diversity and the hydromorphological pre-classification. We attribute this to the fact that the pre-classification, which was subjective in nature, encompassed degradation across scales and hence included information on impacts beyond substrate simplification such as channelisation and change of channel planform.

It is evident from our results that micro-habitat assessment at the reach where macroinvertebrates are sampled had very little relationship with their diversity. So despite using hydromorphological data at this scale, our findings support previous studies showing weak relationships between macroinvertebrates and standard measures of hydromorphology (Friberg et al. 2009; Feld et al. 2014). This clearly lends support to the contention that macroinvertebrate community structure at the reach scale are governed by a range of physical and ecological processes acting on larger scales, as well as a range of local factors that are not quantified in standard assessments (Friberg 2014). We furthermore found indications that the way macroinvertebrates are sampled has a bias towards upland streams with coarse substrates. In contrast to theory and actual findings (e.g. Göthe et al. 2014), diversity of macroinvertebrates were highest for the smallest streams in the altitude range between 200-800 m.asl. At the same time mineral substrate diversity was highest in lowland streams (< 200 m). These findings could indicate that there is a sampling artifact, which seriously hamper our ability to detect relationships between hydromorphological degradation and macroinvertebrate community composition. The relationships between identity based diversity and indicators of substrate diversity were either not significant or very weak. This suggests an inherent weakness of the data set and will also influence our ability to draw conclusions from the trait-based diversity metrics. However, as species traits will be more closely related to the environmental filters, they should be more sensitive even when using a data set with the issues mentioned above (e.g. Menezes et al. 2010). It is important to stress that the sampling methodology in the present study is commonly used and fully in line with methods used in the monitoring of stream and rivers across Europe (Friberg et al. 2005).

In summary, the analysis of a wide range traits gave a short list of associations between specific modalities and the substrate diversity metrics used. It was evident from the analysis that the organic substrates had more significant relationships with traits, and their modalities, than was the case with inorganic substrates. This might highlight the importance of features such as large wood or aquatic vegetation for the macroinvertebrates and is in accordance with findings of e.g. Demars et al. (2012) and the findings in WP 1 and WP 2 in the REFORM project. However, it was also evident that the organic substrate index we developed for this study was not usable as it most often showed no significant relationships to traits, and their modalities, that otherwise was significantly related to one or more of the other organic substrate diversity indicators that we used. The reason for this could be that the subjective scores used in the organic substrate index was not appropriate and obscured the direct relationship between organic substrate diversity and the expression of traits. We will therefore disregard this index in the following discussion. The inorganic substrate index did appear, however, to follow the other inorganic substrate diversity metrics. A number of traits and their modalities showed significant differences and consistent patterns, which might indicate that they could be candidate components in a trait-based metric sensitive towards hydromorphological degradation. However, several of the responses are not easily explained from an ecological point of view and some are contradictory. An example is body size and longevity, which both show a negative response to substrate diversity. When substrate gets less divers macroinvertebrates get smaller, which makes perfect sense, but they also live longer and this combination contradicts fundamental biological theory and empirical knowledge (e.g Woodward et al. 2005). A number of the relationships furthermore suggest that diversity of organic substrates might be related to a certain group of streams within the data set that are in some way degraded and hence confounding the results. The positive relationship to trophic status and the modalities mesotrophic/eutrophic supports this in combination with e.g. the negative relationship to macroinvertebrates with tegument respiration. Macroinvertebrates with this respiratory strategy are generally sensitive to low levels of oxygen (e.g. Friberg et al. 2010). This specific trait is, as a consequence, a poor indicator of degraded hydromorphology as it will respond more strongly to e.g. organic pollution. Our results are overall in agreement with the findings of Feld et al. (2014) that found weak responses in trait diversity of macroinvertebrates to hydromorphological alteration in lowland streams. However, as shown by Pollard and Yuan (2010) relationships to certain traits can be established if data sets are large enough to overcome the “noise” that is an inherent property of this type of data with multiple stressors, a large amount of unmeasured variables and biased sampling methodology. Therefore, the next step is to test the traits that are ecology meaningful and listed in table 21, on a large data set. From table 21 body size (small), aquatic stages (adult aquatic stages), food (fine sediment and microorganism), feeding habitats (deposit), current preferences (slow) show promise. Traits provide a mechanistic understanding of how environmental change influences biotic communities and are therefore a step towards being able to diagnose cause and effects drivers (Doledec et al. 1999; Rabeni et al. 2005). The trait modalities found in this analyses that could be linked to hydromorphological degradation are all functionally meaningful, although e.g. fine sediments as food and habitat can also indicate eutrophication/organic pollution. However, habitat features are the major driver for the expression of these trait modalities and we assume that they for this reason would be able to detect hydromorphological degradation in a multi-stress environment.

In conclusion, however, this analysis undertaken here supports previous findings, both within the REFORM project and published elsewhere, that the link between macroinvertebrates and hydromorphological degradation is weak when using the data currently available. This will ultimately influence our ability to detect impact even with a trait based metric that is specifically sensitive to hydromorphological stress, as there are limits to how far we can progress when developing metrics on currently available data. Underlying methods (both hydromorphological and biological) should be radically revised to address the challenges that managers are facing with regard to linking the massive loss of habitat complexity to the many riverine water bodies that cannot obtain good ecological status.

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10 Joint biota (Biological Quality Elements) - stress from altered channel morphology and land use in multiple-countries

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10.1 Introduction

In this section we assess the importance of two pervasive anthropogenic pressures, alterations to channel morphology and land use induced diffuse pollution, on the community response of multiple biological quality elements (BQEs) in European streams. We use existing biomonitoring datasets from mainly co-located BQE sampling sites compiled earlier in REFORM (see Deliverable 3.1; Friberg et al. 2013) and have two primary study aims. First, we aim to disentangle whether the sensitivity of the BQEs differ between the two pressures. We specifically investigate whether morphological pressures can be singled out in multi-pressure biomonitoring data. This information is required to plan and prioritise management actions to mitigate the impacts of anthropogenic pressures most harmful to the ecological quality. Second, we investigate whether the different BQEs respond differently to the two pressures that alter the local habitat conditions. This information will help plan monitoring schemes evaluating the effectiveness of river restoration projects in selecting the most relevant BQEs to detect change in ecosystems.

Degradation of morphological quality is the most visually obvious anthropogenic alteration of running water ecosystems. Channelisation and modifications of river bed and riparian habitat alter natural physical habitat variability (e.g. natural supply of wood and sediment, depth and substrate variability), flow regime, longitudinal connectivity and lateral connections to floodplains (Poff et al. 1997; Nilsson et al. 2005, Tockner et al. 2010). These disruptions to the geomorphic processes that form habitats have caused large scale loss in natural habitat heterogeneity (see REFORM deliverables D2.1 and D2.2) which is often regarded an essential requirement for high biodiversity (Benton et al. 2003; Tews et al. 2004; Beisel et al. 2000; Wyżga et al. 2012). Further, morphologically altered rivers often drain catchments with intensive agricultural, forestry and urban use (Schinegger et al. 2012). Agricultural rivers are often channelized for land drainage (Allan 2004), but drainage for forestry and associated alterations of riparian habitat can also be locally significant (Campbell and Doeng 1989; Turunen 2008). Urban streams are often heavily altered in terms of both hydrology and morphology (Paul and Meyer 2001). In addition, these land use activities typically alter hydrological patterns and cause diffuse pollution which results in increased nutrient levels and altered temperature (e.g. Genito et al. 2002; Buck et al. 2003; Roy et al. 2003; Matthaei et al. 2006), but may also alter channel hydromorphology by increasing fine sediment loads, which decreases average sediment size. (Walser and Bart 1999; Paul and Meyer 2001; Roy et al 2003).

Land use induced diffuse pollution is known to have clear negative impacts on the quality of BQEs (e.g. Walser and Bart 1999; Genito et al. 2002; Matthaei et al. 2006), whereas observed responses to morphological degradation have been more variable for fish (Lepori et al. 2005) and invertebrates (Negishi et al. 2002; Wyẑga et al. 2012) and can be difficult to characterize using monitoring data (e.g. Villeneuve et al. 2015). Restoration projects that have focused on re-establishment of natural hydrological and morphological variability have resulted in modest or even non-existent biotic responses from fish and macroinvertebrates (Lepori et al. 2005, Jähnig et al. 2010; Louhi et al. 2011; Nilsson et al. 2014). The commonly argued reason for this ineffectiveness has been that poor land use management constrains the positive response to the increased hydromorphological quality (Pretty et al. 2003; Palmer et al. 2010; Stranko et al. 2011, Sundermann et al. 2011). On the contrary, aquatic macrophytes are found to respond to physical alterations and restoration activities (O'Hare et al. 2006; Lorenz et al. 2012).

Many river ecosystems suffer from multiple pressures concurrently, and planning their restoration in a cost-effective way requires that the primary causes for the degraded ecological status are known to allow prioritisation of measures. Hence, it is of fundamental importance to know the unique effect that individual pressures have on BQEs. Recent experimental research has increased understanding about multiple stressor effects (Townsend et al. 2008; Matthaei et al. 2010; Wagenhoff et al. 2011, Wagenhoff et al. 2012; Magbanua et al. 2013). However, in natural river systems the relative sensitivity of BQEs to morphological degradation and land use induced diffuse pollution is insufficiently understood.

Recent studies have indicated that land use in European rivers may mask potential biotic metric response to hydromorphological degradation (Marzin et al. 2012; Dahm et al. 2013). Thus, a significant challenge is how to single out potential effect of hydrological and/or morphological alterations and at same time control for land use impacts. Previous analysis of biotic metrics in Central European rivers have also indicated that the shape of response and sensitivity to different pressures can vary among BQEs (Hering et al. 2006; Johnson et al. 2006; Marzin et al. 2012). For land use pressure, it seems that the BQEs that are primary producers may respond at very low levels of pressure (e.g. Hilton et al. 2006). For example, Johnson et al. (2013) found a marked change in diatom community composition at very low nutrient levels (total phosphorus concentrations <50 µg/L). By contrast, consumers, such as benthic invertebrates, seem to be relatively insensitive to low level increases of nutrients (e.g. Johnson et al. 2013).

In this chapter, we investigate the effects of morphological and land use pressure on community structure of multiple biological groups. We use two separate large-scale monitoring data sets: I) diatom, macrophyte and macroinvertebrate community data from Finland and II) macrophyte and macroinvertebrate community data from Finland, Denmark, France, Germany and Austria. We use multivariate analysis on the pressure data to explore whether the two pressure types can be distinguished in the data sets and, to divide the sampling sites into treatment groups, with an aim of exploring whether the BQE communities respond more strongly to only one or both of the pressures. Instead of pre-selected metrics that can be stressor-specific, we use community composition (similarity) to quantify the BQE response to the pressures. We analyse three different alterations to morphology (modifications of channel form, cross section and habitat) as a single pressure gradient because all can cause to some degree similar modifications of local habitat conditions at site-level. Moreover, our primary aim is to quantify the relative importance of land use pressure and any type of morphological alteration to the BQEs.

Specifically, we address the following three hypotheses:

Land use pressure will have a stronger effect on BQEs than degradation of channel morphology even at low pressure levels, because low increases in nutrients and associated stress are likely to alter the community composition. In contrast, low level of alterations to channel morphology might not alter community composition if suitable microhabitats for the biota remain present.

Macroinvertebrates will show a weaker response to land use pressure than diatoms or macrophytes, as primary producers are more directly linked to changes in nutrient concentrations that limit growth (Johnson and Hering 2009; Johnson et al. 2013).

Given the strong responses of the primary producer BQEs to land use already at low pressure levels, responses to alterations of channel morphology can be detected only when land use pressure is minimal. In contrast, response of macroinvertebrates to morphology should be less dependent on the degree of land use.

10.2 Materials & Methods

10.2.1Study sites of Dataset I from Finland

The Dataset I from Finland included in total 123 river reaches from southern and central Finland (60-65°N and 21- 31°E). The sites represent small to mid-sized rivers (mean catchment area 414 km^2, range 24-3550 km^2) that drain catchments with peatland or mineral soils. All the sampling sites are swiftly flowing hard-bottom riffle reaches. The sites represent both national reference sites minimally impacted by land use activity, morphological alteration or other anthropogenic pressures, and sites impacted by agriculture, forestry and morphological alteration of channel and riparian habitat (Aroviita et al. 2014). The main morphological alteration is channelisation of riffles by removal of boulders (dredging) to facilitate transport of timber. The dredging has reduced land-water interaction, variability of flow and depth, as well as increased flow velocities and erosion of fine sediment, mainly locally at the riffle sections (Nilsson et al. 2005). These alterations took place in Finnish rivers mainly between 1850s and 1970s. Even though a few of the study sites are affected by bank reinforcement (bridges) or flow regulation, dredging is the dominant morphological alteration. Alterations of the riparian habitat include agricultural land use, pasture, forest harvest, and drainage ditches for agriculture and forestry.

Benthic macroinvertebrates, diatoms and macrophytes were sampled in 2006–2012. All the three biotic groups were sampled at 94 sites. The sites were sampled during late summer or autumn using the national sampling methods of the Finnish environmental administration (Meissner et al. 2013). Environmental data consist of measurements of water chemistry, Corine land use of catchments, measurements of morphological alterations (River Habitat Survey [RHS]), and an accompanying evaluation of dredging and other in-stream alterations. RHS was done along a 500 m reach at each site to evaluate the morphological condition (see Raven et al. 1998). From RHS data we calculated the Habitat Modification Score (HMS), a value that describes the intensity of the anthropogenic modification of the stream habitat. The intensity of dredging (channelisationand removal of substrate) was additionally estimated using a 3 point scale (0, 1, 2) at ten evenly distributed spots within the reach. Zero indicated no alteration, 1 moderate alteration and 2 severe alteration. The sum of these values was used as an estimate of overall dredging intensity of the reach. As the biota were sampled from a riffle in each reach, we also calculated an average dredging intensity of the riffle sections (values between 0-2).

10.2.2Study sites of Dataset II

For Dataset II, ecological and environmental data from river sites from Denmark, Finland (mainly the Dataset I), and WISER EU -project were combined (Friberg et al. 2013; Tables 1 and 2). For the purpose of this work, we specifically screened the REFORM data base for sites with co-located BQE samples, measurements of hydrological and morphological alterations, and land use pressures. We initially also screened UK Countryside Survey data (http://www.countrysidesurvey.org.uk/) of hydomorphology, macrophytes and macroinvertebrates, but due to lack of land use data it was excluded from the analysis.

We aimed at consistency in environmental measures across the dataset. The original WISER project data (macrophyte sites N = 2051, macroinvertebrate sites N = 3329) consisted of data from Austria, Germany, Czech Republic, United Kingdom, Denmark, Sweden, France, Slovak Republic and Netherlands. The majority of the WISER-project sites were excluded due to lack of information of either water quality or hydromorphological degradation. Sets of data from Austria, France and Germany had consistent environmental variables. Even then, to harmonise different methods used among the datasets, we ranked morphological alterations of instream habitat modified, channel form modified and cross section modified to bivariate presence/absence (1/0) scale. Hydrological alterations were not measured consistently for any constituent of this large-scale data set. We further excluded some of the sites to balance the statistical design (see below), e.g. sites with a very large catchment (>5000 km^2) or a very high phosphate concentration (>1000 µg/L). The final data consisted of macrophyte (536 sites) and macroinvertebrate (772 sites) samples and associated environmental data (Table 10.1). These sites represent mainly small or medium-sized rivers from Austria, Finland, France and Germany, and small rivers from Denmark. Altogether 454 sites had both macrophytes and macroinvertebrates sampled.

Table 10.1 The ecoregions and number of sampling sites used in the analysis from each country.

	Austria	Denmark	Finland	France	Germany	Total
Ecoregions covered (EEA*)	4, 5, 9, 11	14	22	2, 4, 8, 9, 13	8, 9, 13, 14	
N, environmental variables	156	161	146	315	169	947
N, macrophyte sampling sites	1	82	118	310	25	536
N, macroinvertebrate sampling sites	149	152	132	170	169	772

*http://www.eea.europa.eu/data-and-maps/data/ecoregions-for-rivers-and-lakes

Table 10.2 The environmental variables combined from WISER, Finnish and Danish data sets. The pressure variables marked with x were log-transformed and used in the Hill & Smith multivariate analysis.

Environmental variable	Unit	Pressure
Ecoregions for rivers and lakes (EEA)		
Catchment area	km^2	
Hydro-morphological alterations		
Instream habitat modified	yes / no	x
Channel form modified	yes / no	x
Cross section modified	yes / no	x
Water chemistry		
pH		
Phosphate-P	µg/l	x
Nitrite-N + Nitrate-N	µg/l	x
Ammonia-N	µg/l	x
Catchment variables		x
Urban land use	Corine land cover (%)	
Arable land	Corine land cover (%)	x
Heterogeneous agricultural areas and permanent crops	Corine land cover (%)	x
Pastures	Corine land cover (%)	x
Forests	Corine land cover (%)	x
Shrub and/or herbaceous vegetation	Corine land cover (%)	
Open spaces with little or no vegetation	Corine land cover (%)	
Inland waters	Corine land cover (%)	
Inland wetlands	Corine land cover (%)	
Coastal wetlands	Corine land cover (%)	
Marine waters	Corine land cover (%)	

10.2.3 Biological data

Diatoms

The diatoms in Dataset I from Finland were sampled from five stones from each site (Eloranta et al. 2007). The stones were placed in a bucket, where the upper surface of the stones was washed with water using a tooth brush. The dislodged material was put in a sample bottle and preserved in ethanol. From each sample 500 diatom valves were counted and identified to species level. The relative abundance of species (proportion of valves in the sample) was used in all analyses.

Macrophytes

In Dataset I from Finland, macrophytes (both vascular plants and bryophytes) were surveyed along a 100 m reach at each site. Each reach was divided into five 20 m sections where both abundance and frequency of vascular plant species were estimated (Rääpysjärvi et al. 2013). For vascular plants, only those with underwater roots were included. Abundance of each species was estimated by the percent coverage in typical stands in a square meter of the 20 m section. For frequency, first the surface area of one percent of the 20 m section was estimated. Then the frequency of each species was counted by the number of one percent squares in which the species was observed. Abundance of bryophyte species was estimated as percentage coverage at two 1x2 m plots within each 20 m section. The plots were placed at 10 m intervals starting at 5 m distance from the beginning of the section. The frequency of bryophyte species was counted by the number of plots where the species occurred.

We used the vegetation index (VI) of Ilmavirta and Toivonen (1986) in all analysis. VI = 2^(abundance+frequency-1), where abundance and frequency of vascular plants and abundance of bryophytes are ranked to values 1 to 7 by 0 %=0, <0.5 %=1, 0.5-1 %=2, 1-5 %=3, 5-25 %=4, 25-50 %=5, 50-75 %=6, 75-100 %=7; and frequency of bryophytes: 0=0 %, 1=10 %, 2=20 %, 3=30%, 4=40 %, 5=60-70%, 6=80-90%, 7=100%.

The macrophyte sampling methods of Dataset II are described for Denmark in Riis and Sand-Jensen (2002), for Finland see above, and for the WISER-project in Feld et al. (2012). The basic unit of all sampling methods was 100 meter long stream reach, where macrophyte species and their abundances were recorded. The abundance unit varied between the methods and therefore the abundance unit was changed into the semi-quantitative five-point scale used in WISER: 1 = very rare, 2 = rare, 3 = common, 4 = frequent, 5 = abundant, predominant (Kohler 1978). The abundances of the Finnish and Danish data were transformed to a Kohler scale with the following ranges: <0.1% = 1, 0.1–1% = 2, 1–5% = 3, 5–10% = 4, >10% = 5 following the range presented by Holmes and Whitton (1977). The taxonomy used in the analysis was according to the species list developed as part of the WISER and REBECCA EU-projects (available at http://www.aqplants.ceh.ac.uk/). Rare taxa were omitted from all analyses.

Macroinvertebrates

In Dataset I from Finland, benthic macroinvertebrates were sampled by taking four 30 second kick-net samples on substrates that were dominated by large and small stones. In the field invertebrates were preserved with 95 % ethanol (c. 70 % final concentration) and later sorted in full in a laboratory. Identification was mainly done to species or genus level (with the exception of dipteran families and oligochaete worms). In Dataset I, log-transformed abundance data were used in all analyses.

The macroinvertebrate sampling procedure in Dataset II followed national standards in Denmark and Finland (see above). WISER-project data is described in Feld et al. (2012). One sample was chosen to represent each site. To take into account seasonal variation across Europe, spring and summer samples were predominantly chosen from the WISER and Danish data whereas autumn samples from the Finnish sites were preferred. The taxonomy was harmonized to the lowest feasible level consistent across the data (mainly genus level, in some cases family-level). Samples with poor taxonomic resolution were discarded and only those with identification to the operative taxonomic level were accepted. Oligochaeta, mites, and chironomids were not consistently identified across the data, so they were retained at order or family level. To avoid “ghost taxa”, specimens identified a lower level (e.g. to family) than in the operative taxomony (e.g. genus) were assigned to the operative taxa (genus) in proportion to the corresponding family identified in the sample. To equalize differences in sampling effort among the datasets, the macroinvertebrate abundances per unit sampling effort were transformed to relative abundances in each sample. In addition, rare taxa were omitted from all analyses.

10.2.4Statistical analyses

General approach

First, the Finnish dataset was analyzed separately to investigate the importance of alterations to channel morphology in a region where land use and nutrient enrichment is low compared to most other parts of Europe. Then, a pan-European analysis was conducted forming common pressure gradients using the large scale dataset II.

To investigate the effect of individual pressures, we divided the study reaches into “treatment” groups based on their pressure levels. Dataset I and particularly II cover a wide range of natural gradients (river types) in European landscape, with associated natural variation of the BQEs. We excluded some sites from the analyses (see above) with the aim of forming roughly balanced treatment groups but including as many sites as possible. Each comparison (e.g. between low to medium level of morphology at low level of land use) will thus represent a wide but similar range of natural variation. We expected that any major change in the global pressure level (e.g. from low to medium morphological alteration) affecting the BQE communities would be detected as a difference in the community similarity among the pressure treatments.

Pressure gradients

In Dataset I from Finland, Principal Component Analysis (PCA) was used to reduce the multidimensionality of the environmental variables by combining redundant environmental variables to a few principal components. In PCA the environmental variables were centered and standardized.

In the Dataset II, the pressure variables were summarized using Hill and Smith analysis, which is a multivariate analysis for mixed and multivariate data (Dray & Dufour 2007, Hill & Smith 1976) and, thus, served well for Dataset II (see Table 10.2). In Hill and Smith, multiple correspondence analysis is used for qualitative variables. Since Hill and Smith analysis uses principal component analysis (PCA) for the multivariate data, combining variables and logarithmic transformation were done for the multivariate water quality and catchment variables to better fulfill the assumption of normality in PCA.

Site groupings for pressure treatments

Dataset I from Finland had more detailed information about the degree of anthropogenic alterations, so we grouped the sites based on their deviation from reference conditions of morphology and nutrient status. The reach was considered morphologically degraded if the dredging intensity index within the riffle habitat was >0.2. As the estimate of dredging in each spot was evaluated either 0, 1 or 2, a dredging intensity value of 0.2 indicates that 20 % of riffle section was dredged moderately or 10 % severely. The site was considered impacted by diffuse pollution due to land use if the total phosphorus concentration was higher than the national boundary between good and moderate nutrient status class for the respective stream type (Aroviita et al. 2012). The approach thus takes into account natural variation in nutrient the levels across different river types. The grouping resulted in four pressure treatments: (i) Neither of the pressures present (near-pristine or slightly impacted, hereafter “reference” sites) (N=27), (ii) morphologically degraded sites with no or minor diffuse pollution (N=32), (iii) sites with land use pressure but no or minor morphological degradation (N=35) and (iv) sites with both pressures present (N=29).

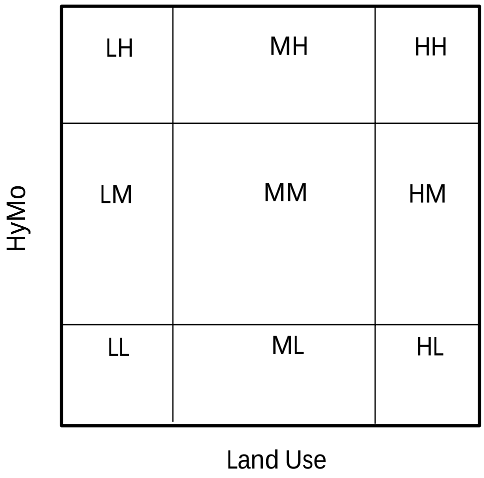


Figure 10.1 A conceptual scheme how common land use and (hydro)morphological (HyMo) pressure treatments were formed from the field data in Dataset II. L, M and H indicate low, medium and high common pressure levels derived from the Hill & Smith PCA axis distributions (in each box, first letter is for land use, second for HyMo). See text for details.

For the large-scale dataset II, we aimed to develop pan-European pressure gradients to allow for comparability of the extent of response among regions. The pressure grouping in Dataset II was based on the distribution of PCA-axes values for morphology and land use pressure so that sites with <25th percentile of the axis values ranked as low pressure, sites with 25-75th percentile as medium stress, and sites with >75th percentile of the axis values ranked as high stress (Johnson et al. 2006). This approach allowed a common European pressure gradient to be formed, resulting in nine pressure treatments where both morphological and land use had levels of low, medium and high pressure (Figure 10.1).

Community structure

In the Dataset I, Detrended Correspondence Analysis (DCA) (Hill and Gauch 1980) was used to visualize the community structure of different organism groups. Macroinvertebrate and macrophyte data were log (x+1)-transformed and diatom data arcsine square root transformed. The PCA gradients that described diffuse pollution and morphology were correlated with Spearman rank correlation to DCA-axis scores to explore how the change in community structure was attributable to the measured intensity of human disturbance. We used the DCA instead NMS to be able to examine the main gradient of community turnover among the sites.

In Dataset II, a non-metric multidimensional scaling (NMS) was used to visualize the variation in macrophyte and macroinvertebrate community composition, and its relationship with the variation in pressure components. NMS was first performed across all data to examine the main gradients. Then, NMS (and subsequent analyses, see below) were performed separately on each country specific dataset.

In both Datasets I and II, permutational multivariate analysis of variance for distances ('Adonis' command in R-package vegan 2.0-5, Oksanen et al. 2012) was used to test for significant differences in community structure between the pressure groups, i.e. whether diffuse pollution or morphological degradation could explain these differences. According to Oksanen et al. (2012), Adonis is a robust technique that partitions distance matrices among sources of variation and uses F-tests based on sequential sums of squares from permutations of the raw data to assess statistical significance.

10.3 Results

10.3.1Dataset I: Finland

Pressure gradients

In the Dataset I from Finland, the first four PC-axes of environmental variables had meaningful interpretation, explaining 48.9 % of the variance in the environmental data. The 1st PC-axis can be interpreted as a diffuse pollution gradient as land use and water quality variables contributed strongly to the axis (Table 10.3, Figure 10.2). The 2nd axis describes a natural gradient in water colour, pH, COD that is related to the dominance of peat lands in the catchment. The 3rd axis can be interpreted as a gradient of morphological degradation and 4th axis as a gradient of natural afforestation.

Table 10.3 Varimax rotated principal component loadings for environmental variables. Bold values indicate the highest loading of each variable to the components. HMS is Habitat Modification Score form River Habitat Survey (RHS). HQA is Habitat Quality Assesment score from RHS where higher scores represent more natural sites.

	PC1	PC2	PC3	PC4
Eigenvalue	9.57	4.93	3.25	2.29
% of variance	23.3	12	8	5.9
Total phosphorus	0.888	0.268	0.029	0.023
Turbidity	0.855	-0.153	0.043	-0.01
Field%	0.848	-0.338	0.093	0.003
Total phosphorus max.	0.826	0.274	0.007	0.049
Conductivity	0.814	-0.185	0.047	0.157
SS	0.788	-0.168	-0.047	0.005
Total nitrogen	0.731	0.166	0.078	0.188
Alkalinity	0.703	-0.424	-0.003	-0.113
Forest%	-0.692	0.063	-0.112	0.461
Altitude	-0.674	0.019	-0.009	-0.102
Total nitrogen max.	0.635	0.02	0.055	0.117
Coniferous%	-0.583	-0.09	0.046	0.199
Lake%	-0.534	-0.507	-0.033	-0.142
Urban%	0.517	-0.638	-0.038	0.213
Fine sediments	0.511	0.172	-0.179	0.329
HQA	-0.447	-0.122	-0.023	-0.194
Other arable land %	0.414	-0.123	-0.072	0.296
Coarse wood > 10 Ø	-0.396	-0.228	-0.331	0.371
Canopy cover	-0.383	-0.188	-0.095	-0.018
Pasture%	0.31	-0.363	-0.049	0.051
Cutting	-0.249	-0.069	0.087	-0.09
COD	0.068	0.872	0.023	0.111
Colour	0.337	0.824	0.001	-0.019
Peatland%	-0.241	0.77	0.059	-0.328
pH	0.333	-0.683	-0.12	-0.22
Fe	0.552	0.599	-0.021	-0.195
Dredging	-0.15	-0.026	0.838	0.166
Dredging intensity	-0.131	-0.026	0.819	0.262
Dredging in riffle	-0.08	0.076	0.709	0.142
HMS	-0.125	-0.171	0.664	0.372
Channel straightening	-0.099	-0.218	0.529	0.015
pHmin	-0.04	-0.264	-0.212	-0.039
Fine wood < 10 Ø	-0.388	-0.106	-0.323	0.447
Catchment area	-0.014	-0.087	0.259	-0.416
Mixed forest%	-0.336	0.167	-0.165	0.405
Deciduous%	-0.093	0.233	-0.278	0.354
Number of riffles	-0.087	0.354	-0.025	0.054
Number of pools	0.01	0.179	0.063	-0.212
Culverts	0.151	0.087	0.108	0.362
Mean particle size	-0.139	0.083	0.286	-0.319
Ditches	0.165	0.358	-0.085	0.288

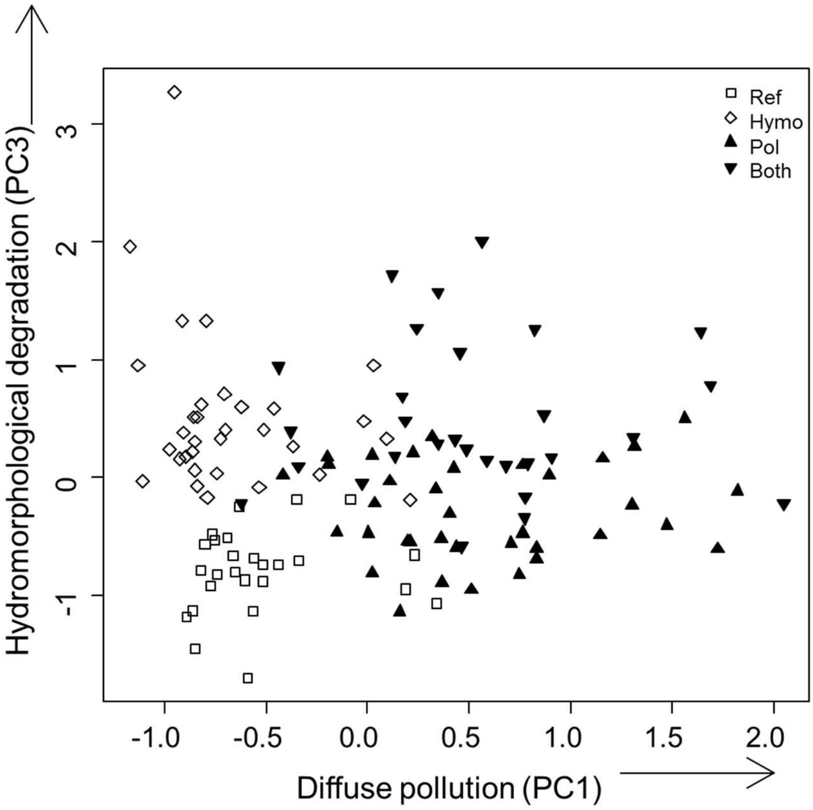


Figure 10.2 PCA-ordination of the 123 study sites from Finland. Ref=Reference sites with low levels of morphological alteration or diffuse pollution, Hymo=Sites with only morphologically alteration, Pol=Sites with only diffuse pollution, Both= Sites with both morphological alteration and diffuse pollution pressure

Community structure and response to pressures

In the Finnish dataset, the main compositional turnover of macroinvertebrate and diatom assemblages (DCA axis 1) correlated strongly with the natural “peatland gradient” (Figure 10.3, Table 10.4, Appendix 1). The result suggests that the macroinvertebrate and diatom community variation was partly attributable to natural variation in the catchment geology. The site scores on DCA axis 2 correlated significantly and relatively strongly with the diffuse pollution gradient (Table 10.4). The main turnover of macrophyte assemblages (DCA axis 1; Figure 10.3 3b) was significantly and strongly related to the gradient of diffuse pollution. However, none of the organism groups showed community turnover along the gradient of morphological degradation (Table 10.4).

Table 10.4 Spearman correlation coefficients between DCA-axes scores of macroinvertebrates, diatoms and macrophytes with the PCA-gradients. Significant coefficients are bolded.

	Macroinvertebrates		Diatoms		Macrophytes	
DCA-axis	DCA1	DCA2	DCA1	DCA2	DCA1	DCA2
PCA1 ("Diffuse pollution")	0.22	0.54	0.52	0.75	-0.82	0.05
PCA2 ("Peatland gradient")	-0.62	0.26	-0.61	0.16	0.24	-0.12
PCA3 ("Morphology")	0.12	-0.02	-0.06	-0.02	-0.12	0.16

Permutational multivariate analysis of variance for distances (Adonis) indicated that diffuse pollution was a significant factor explaining differences in community structure (Table 10.5). In Adonis, morphological degradation did not explain differences in community dissimilarity in any of the BQEs. As expected, diatoms and macrophytes showed the strongest response to land use induced diffuse pollution, followed by macroinvertebrates.

Table 10.5 Permutational multivariate analysis of variance for distances (Adonis) within the Dataset I from Finland. Significant differences ($p < 0.01$) are bolded.

Organism group	Stressor	R^2	p
Macroinvertebrates	Diffuse pollution	0.035	<0.001
	Hymo degradation	0.010	0.299
	Diffuse poll.* Hymo degr.	0.008	0.545
Diatoms	Diffuse pollution	0.055	<0.001
	Hymo degradation	0.010	0.331
	Diffuse poll.* Hymo degr.	0.015	0.048
Macrophytes	Diffuse pollution	0.067	<0.001
	Hymo degradation	0.010	0.257
	Diffuse poll.* Hymo degr.	0.011	0.206

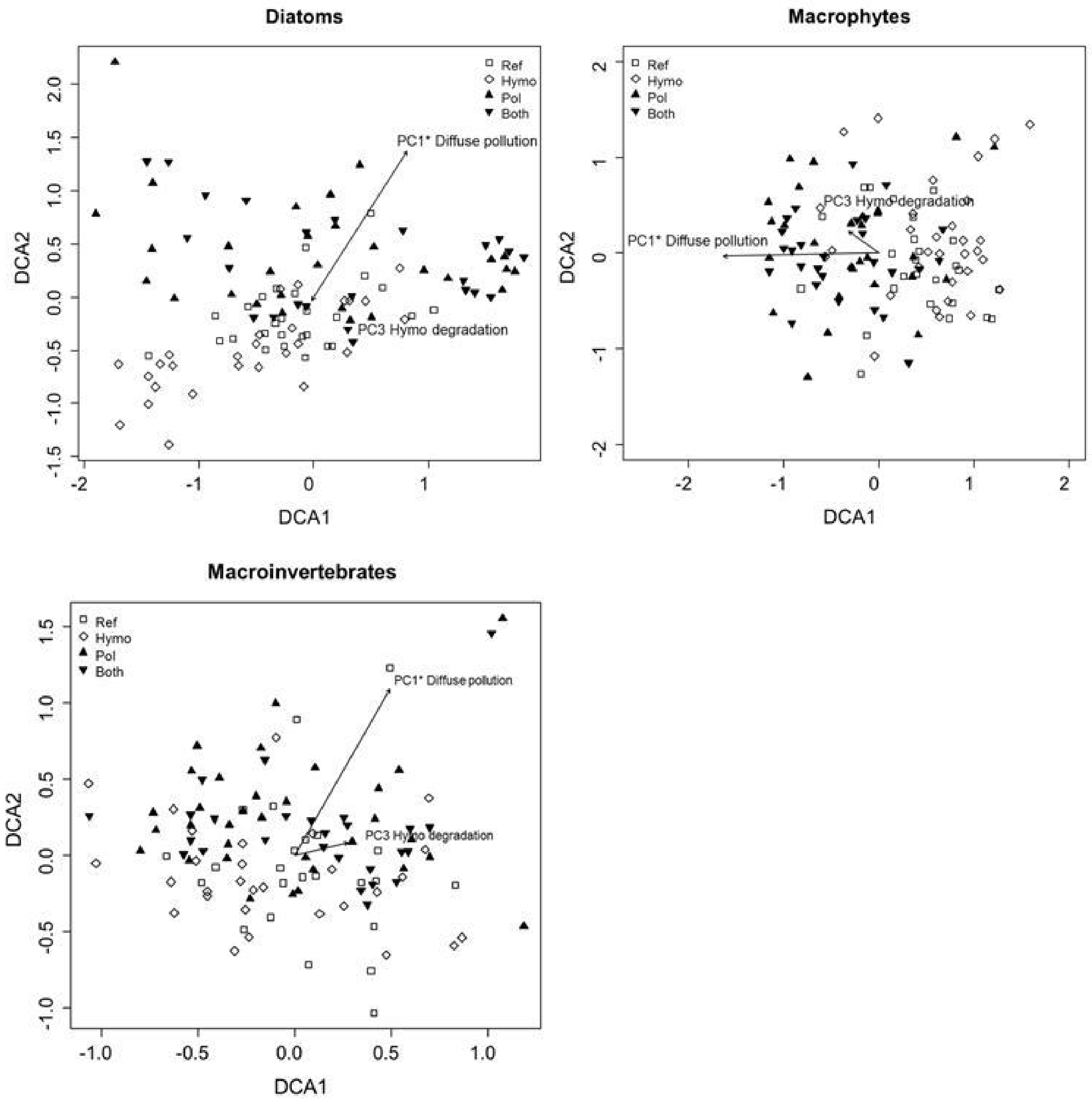


Figure 10.3 DCA-ordination of diatom, macrophyte and macroinvertebrate communities in the Finnish data set. See Appendix I for the species ordination.

10.3.2 Dataset II: Finland, Denmark and Central Europe

Pressure gradients

In the large-scale Dataset II of 947 sites from Finland, Denmark, France and Austria, the first three components of the multivariate analysis explained 60.1 % of the total inertia (Table 10.6). The first component accounted for 28.5 %, the second 18.3 % and the third 13.3 % of the variation. The first pressure component can be interpreted as variation in the water chemistry across Europe due to intensive agriculture (Figure 10.4). Also channel form modification made a strong contribution to the first component, most likely due to channelisation of agricultural streams.

The second component represented a gradient from modified to unmodified local morphological habitat conditions (Table 10.6). Instream habitat modification had the highest contribution to this component. Local morphological changes were linked to urban land use in the catchment, illustrating the multi-pressure nature of European rivers. The second component also incorporated nutrient enrichment, so that natural local hydro-morphological conditions were linked to high nitrate and phosphorus concentrations (Figure 10.4). This counterintuitive gradient was likely to be due to the fact that parts of the data represented rivers with few modifications to channel form, cross section or instream habitat (66 % of the French and 55 % of the Danish sites had no morphological modifications) but high phosphate and nitrate concentrations (see Appendix I and II), and that many Austrian sites had many types of local HyMo-modifications, a high percentage of urban land use and low values of phosphate and nitrate. Therefore, the HyMo pressure component can be interpreted as representing not only the presence or absence of morphological modifications but also more natural HyMo-conditions in the countryside compared to more morphological pressure in the urban areas. The third pressure component represented less intensive agriculture with pastures and heterogeneous agricultural areas. As always with large-scale datasets, the pressure components also incorporate inherent natural variation across Europe.

Community structure and response to stressors

Both macrophytes and macroinvertebrate communities showed a clear geographical pattern across Europe (Figure 10.5 and Figure 10.6). Therefore, further analyses were performed within the countries. In most countries two pressure levels were present. In France, comparisons could be done at two pressure levels where both land use and morphological alteration ranged from low to medium (France1) and medium to high (France2).

Permutational multivariate analysis of variance for distances (Adonis) indicated that land use was a significant factor explaining differences in community structure in all datasets (Table 10.7; Figure 10.7; Figure 10.8). The only exception was between medium and high level of land use pressure, when no significant effect of land use pressure was detected in macrophytes (Denmark, France2). Macrophytes showed a stronger response than macroinvertebrates to land use in France1 and in Finland. In contrast, in Denmark macroinvertebrate communities showed response to land use whereas macrophytes did not.

Morphological alterations were a significant factor explaining differences in community dissimilarity only for macrophytes when land use pressure was at low level (Low-Medium morphological pressure, dataset France1). In the Finnish datasets and in the French and Austrian macroinvertebrate datasets significant community responses to morphological alteration were not detected at low land use pressure.

Table 10.6 Column normalised scores of the pressure variables to the first three components of Hill & Smith PCA analysis in the Finnish, Danish and Central European datasets (Dataset II, N = 947).

	CS1	CS2	CS3
Eigenvalues	3.137	2.008	1.464
Ammonia-N	-0.40	-0.11	-0.17
Nitrite-N + Nitrate-N	-0.44	-0.26	0.02
Phosphate-P	-0.37	-0.37	-0.11
ArableLand	-0.40	0.02	-0.26
Channel form modified	-0.43	0.45	0.09
Cross section modified	-0.27	0.47	-0.04
Urban landuse	-0.27	0.34	0.01
Instream habitat modified	0.17	0.70	0.08
pH	-0.23	0.03	0.43
Heterogeneous agricultural areas and permanent crops	-0.08	0.05	0.61
Pastures	-0.08	-0.23	0.57
No channel form modified	0.29	-0.31	-0.06
No cross section modified	0.27	-0.47	0.04
No instream habitat modified	-0.08	-0.36	-0.04

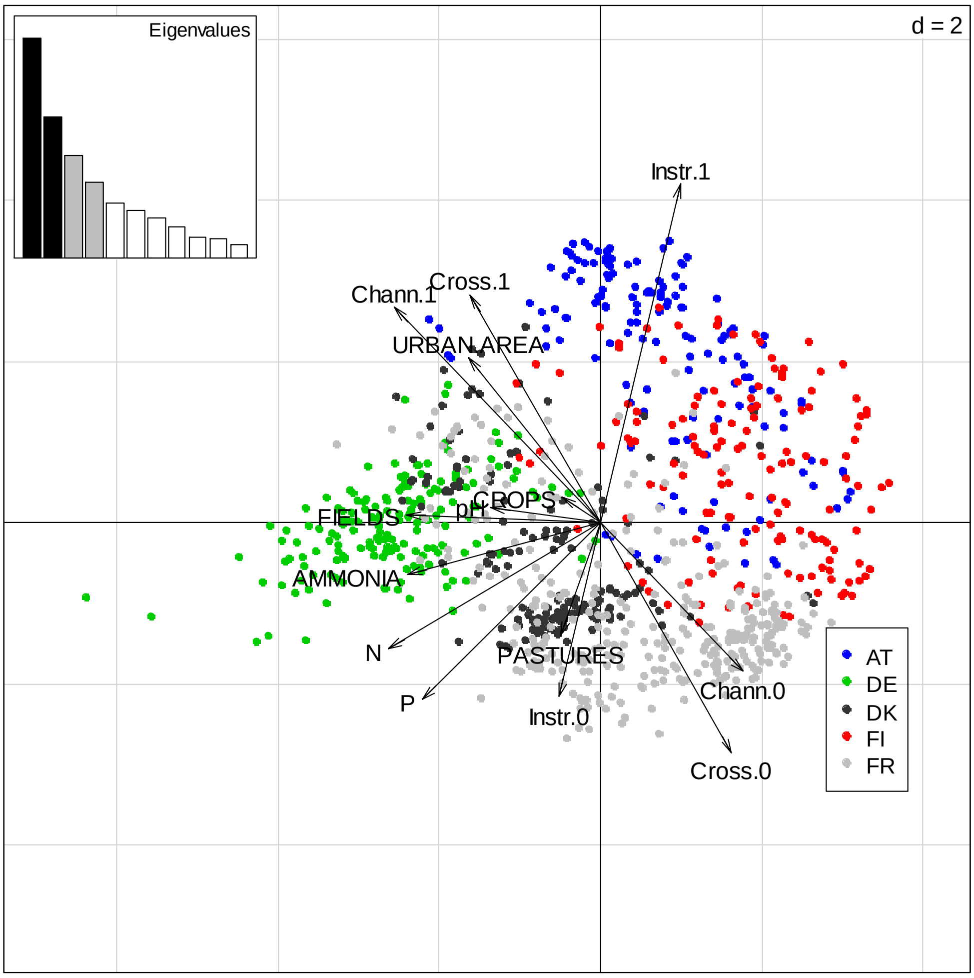


Figure 10.4 A Hill and Smith PCA analysis summarizing the pressure gradient in the Finnish, Danish and Central European datasets (Dataset II, N = 947). Instr. 0 = no instream habitat modified, Instr. 1 = instream habitat modified, Chann 0 = no channel form modified, Chann 1 = channel form modified, Cross 0 = No cross section modified, Cross 1 = cross section modified, P = Phosphate-P , N = Nitrite-N + Nitrate-N, AMMONIA = Ammonia-N, URBAN AREA = urban land use %, FIELDS = arable land, CROPS = heterogeneous agricultural areas and permanent crops (%), PASTURES = pastures in the catchment (%).

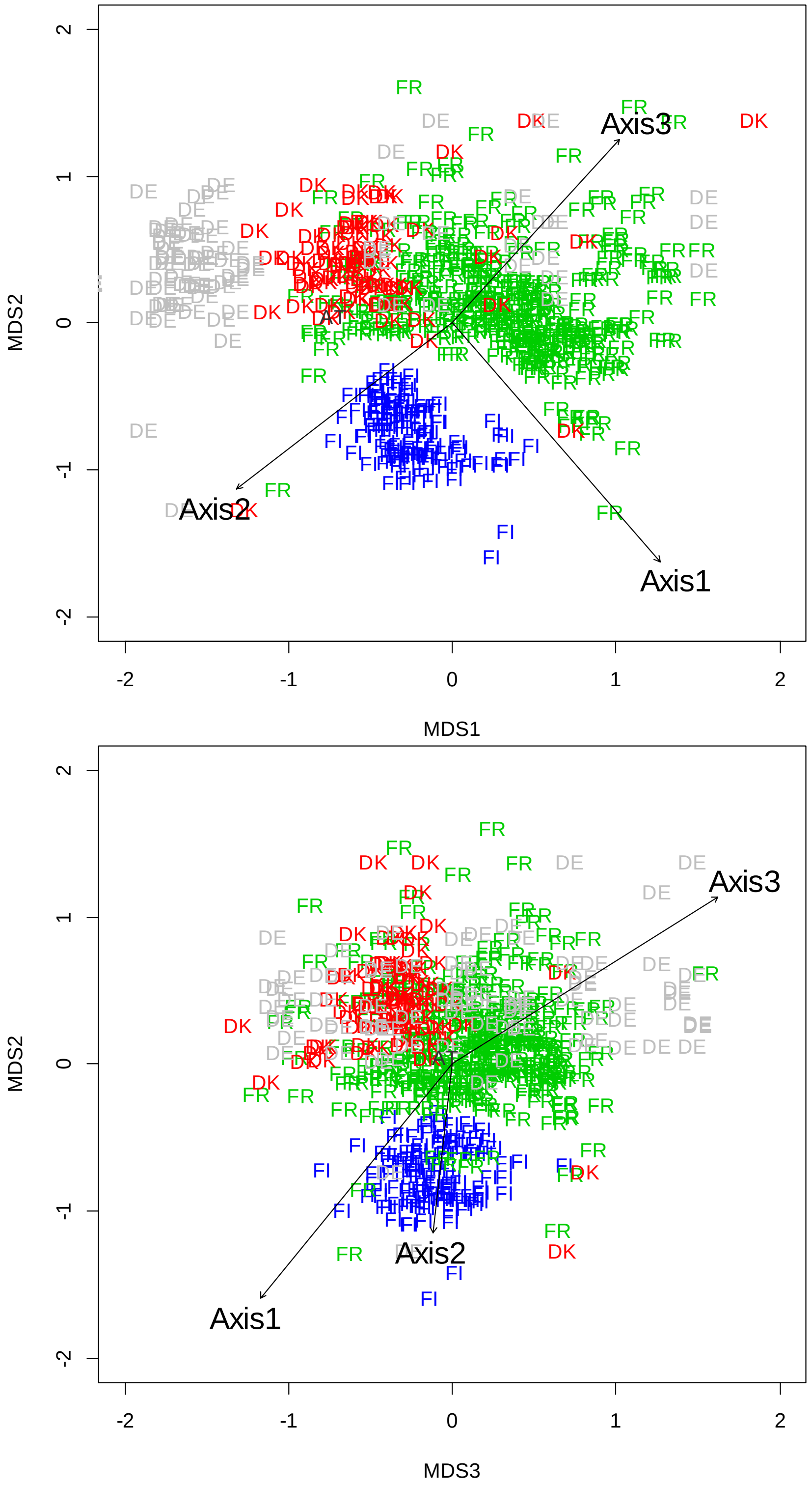
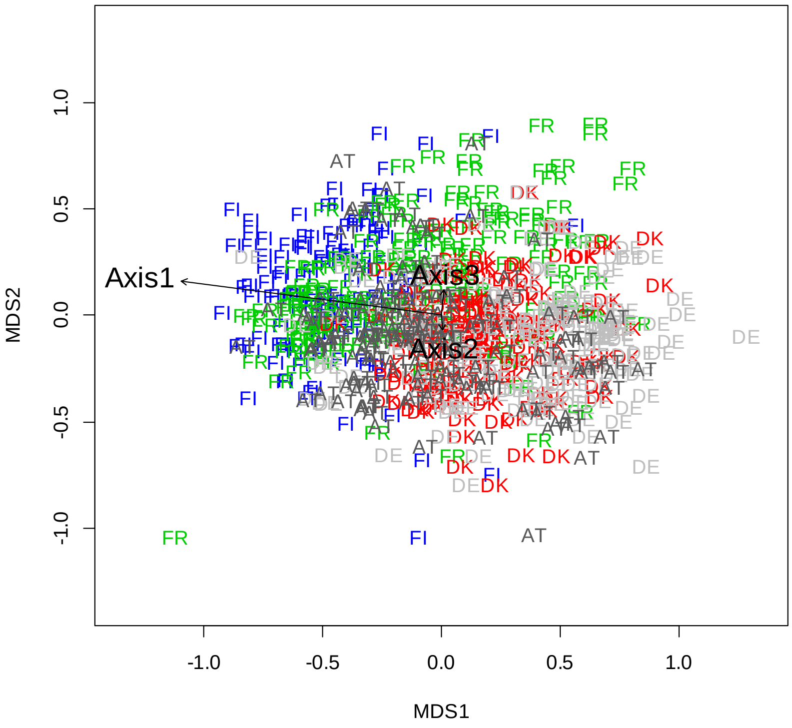


Figure 10.5 A NMS-ordinaton of macrophyte communities across the Finnish, Danish and Central European datasets (Dataset II, N = 536). The arrows show the correlation to the PCA-gradients. Stress: 0.144.



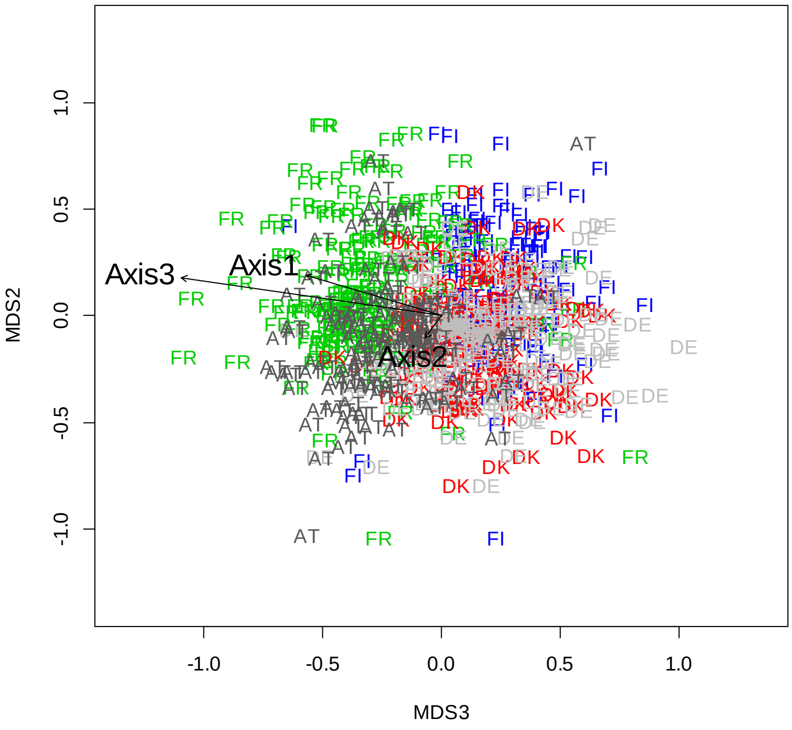


Figure 10.6 A NMS ordination of macroinvertebrate communities across the Finnish, Danish and Central European datasets (Dataset II, N = 772). The arrows show the correlation to the PCA-gradients. Stress: 0.195.

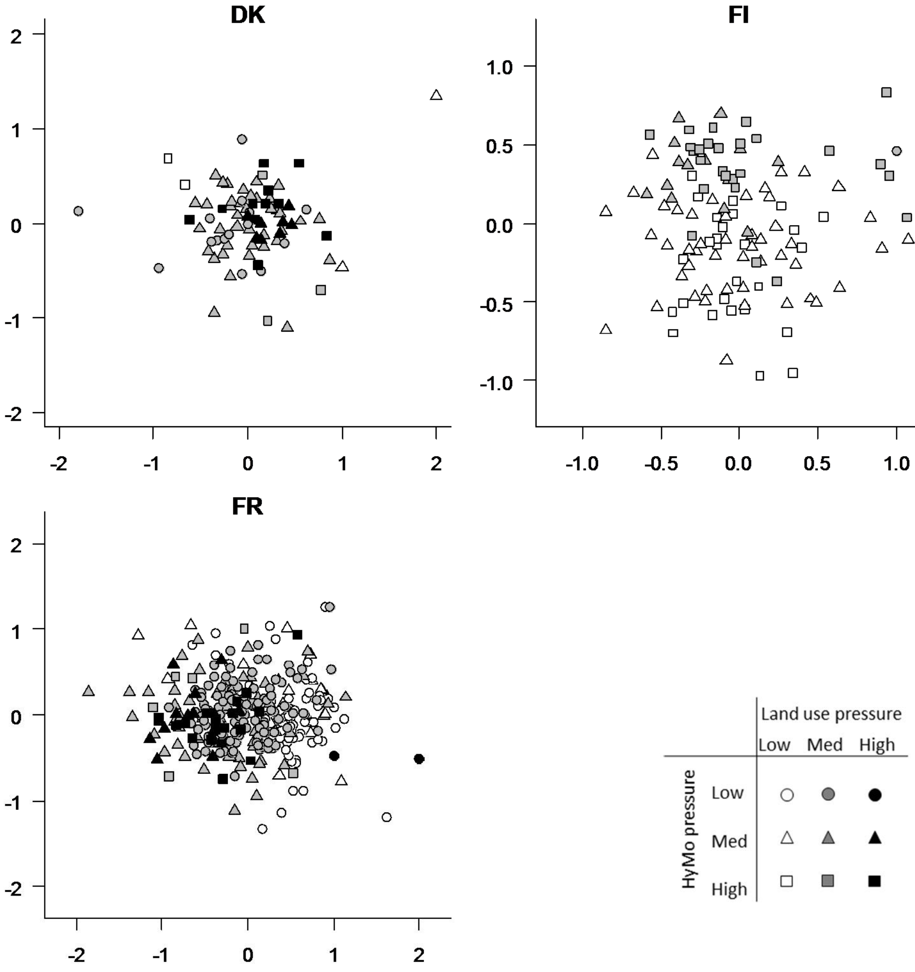


Figure 10.7 Variation of macrophyte community structure within the Danish (Stress: 0.145), Finnish (Stress: 0.228) and French datasets (Stress: 0.215). The symbols refer to common low, medium and high pressure levels derived from the Hill & Smith PCA pressure gradients. See Appendix VIII for the species ordination.

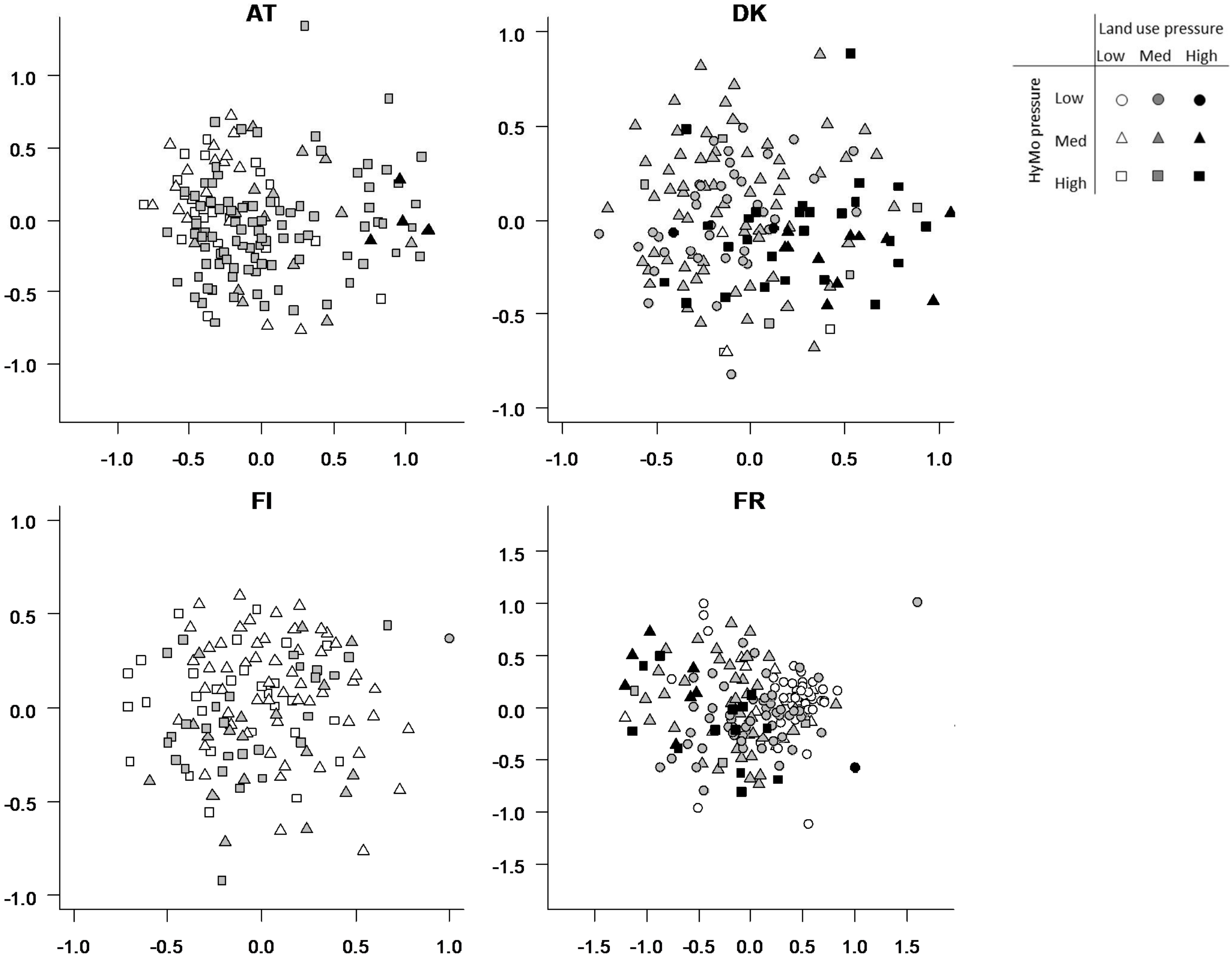


Figure 10.8 Variation of macroinvertebrate community structure within the Austrian (Stress: 0.221), Danish (Stress: 0.237), Finnish (Stress: 0.268) and French datasets (Stress: 0.236). The symbols refer to common low, medium and high pressure levels derived from the Hill & Smith PCA pressure gradients. See Appendix IX for the species ordination.

Table 10.7 Permutational multivariate analysis of variance for distances (Adonis). Significant differences (p < 0.01) are bolded. L, M and H indicate low, medium and high pressure levels derived from the Hill & Smith PCA gradients for both of the pressures (see text for details).

Organism group	Country	Pressure (level class comparison)	R2	p
Macroinvertebrates	Austria	Land use (L – M)	0.024	<0.001
		Hymo degradation (M – H)	0.007	0.335
		Land use*Hymo degradation	0.013	0.017
	Denmark	Land use (M – H)	0.073	<0.001
		Hymo degradation (M – H)	0.018	0.048
		Land use*Hymo degradation	0.007	0.603
	Finland	Land use (L – M)	0.017	0.006
		Hymo degradation (M – H)	0.013	0.024
		Land use*Hymo degradation	0.009	0.223
	France1	Land use (L – M)	0.040	<0.001
		Hymo degradation (L – M)	0.008	0.283
		Land use*Hymo degradation	0.007	0.439
	France2	Land use (M – H)	0.019	0.328
		Hymo degradation (M – H)	0.016	0.447
		Land use*Hymo degradation	0.018	0.335
Macrophytes	Denmark	Land use (M – H)	0.020	0.149
		Hymo degradation (M – H)	0.020	0.097
		Land use*Hymo degradation	0.030	0.028
	Finland	Land use (L – M)	0.090	<0.001
		Hymo degradation (M – H)	0.008	0.422
		Land use*Hymo degradation	0.013	0.057
	France1	Land use (L – M)	0.063	<0.001
		Hymo degradation (L – M)	0.010	<0.001
		Land use*Hymo degradation	0.005	0.075
	France2	Land use (M – H)	0.018	0.012
		Hymo degradation (M – H)	0.012	0.176
		Land use*Hymo degradation	0.008	0.785

10.4 Discussion

In this analysis, large biomonitoring datasets were compiled to address whether alterations to channel morphology could be singled out in European river ecosystems that are often heavily altered by land use. A specific emphasis was on the response patterns of multiple BQE communities to multiple stresses. The results clearly indicated that land use pressure had a strong effect on the taxonomic composition of BQEs (macroinvertebrates, diatoms or macrophytes), a result similar to earlier studies (Johnson et al. 2006; Marzin et al. 2012; Dahm et al. 2013). In contrast, the overall effect of morphological degradation on community structure was much weaker and not consistent across the datasets. Generally, the results indicate that diffuse pollution and associated land use induced stressors affect the taxonomic composition of the BQEs more strongly than alterations of channel morphology alone. However, the results should be interpreted cautiously: The datasets cover large geographical and stressor gradients across a range of longitudinal variation of rivers. The results, thus, do not indicate that alteration of channel morphology does not affect BQEs, but the results do indicate that community level responses of the BQEs to channel morphology are difficult to detect, particularly with current monitoring methods and can be masked by the effects of land use pressure.

The results supported our first hypothesis in that even at low pressure levels land use would have stronger effect on BQEs than morphological degradation. A likely reason for the sensitivity to land use pressure is that communities of primary producers such as diatoms and macrophytes respond directly to nutrient concentrations that limit growth (Buck et al. 2003; Johnson and Hering 2009). Indeed, at low to medium land use gradient macrophytes (and diatoms) showed a stronger response than macroinvertebrates, supporting our second hypothesis. The benthic macroinvertebrates that are consumers seem to be quite tolerant to changes in nutrient concentrations. Recently Johnson et al. (2013) found that stream benthic invertebrate assemblages showed change only at high nutrient levels. Similarly, in the Danish dataset macroinvertebrate communities (but not macrophytes) responded to the land use gradient that ranged from medium to high pressure (see Table 10.7). The results suggest that diatoms and macrophyte communities have a higher sensitivity to diffuse pollution than macroinvertebrates and could thus be more useful biological groups to assess and monitor impacts of low to moderate levels of diffuse pollution in stream ecosystems. The correlation between macrophytes and P is unlikely to be causal across the whole range of river types represented here; for example high energy river systems support macrophyte assemblages dominated by bryophytes, which are relatively insensitive to eutrophication. Such rivers are common in both Finland and France. In Chapter 3 of this deliverable eutrophication and its interaction with hydromorphology is discussed. Detailed analysis of macrophyte data from Denmark (Chapter 4) also indicates responses to eutrophication may be most easily determined by examining traits rather than species data.

Several reasons may exist why a consistent response of BQE community structure to alterations of channel morphology was not detected across the datasets. At low to medium land use pressure, only macrophytes in the France1 dataset showed a significant response to morphological pressure, whereas macrophytes in Finland and all macroinvertebrate datasets indicated an non-significant response. One explanation for these results could be differences in average river sizes between the datasets. The France1 dataset covered mainly small rivers with catchment size < 100 km^2, whereas the Austrian, Finnish and France2 datasets represented mainly medium to large-sized rivers (100-1000 km^2). It might be that the community effects of alterations to channel morphology are, for some yet unidentified reasons, larger in smaller rivers. As we expected, at medium to high land use pressure levels (datasets Denmark, France2), the compositional change in BQEs due to high nutrients and associated stress seem to mask the effects of morphological pressures (Marzin et al. 2012; Dahm et al. 2013).

The small observed effect of morphological degradation at the site scale might also be due to limited comparability of morphological data between countries (see Dahm et al. 2013). We ranked the instream habitat, channel form and cross section modifications to a coarse bivariate presence/absence scale. This allowed construction of a common European pressure gradient across the large data set and the ranking all sites to one of the nine pressure treatment classes. The pressure gradients could be useful in comparing pressure levels between different regions and types of rivers. For example, in Dataset II small rivers were generally present in all pressure treatments, whereas larger rivers were mainly absent at low pressure levels (except for land use in Finland). Nevertheless, the bivariate nature of the morphological pressure gradient does not necessarily represent the intensity of morphological alterations, rather the complexity of the local modification types. Indeed, the sites classified in our analyses as having low morphological pressure according to the PCA (Figure 10.4) could still have been structurally altered in many ways.

In Dataset I from Finland, we did not observe a response in any of the BQEs to morphological alterations of the channels. Moreover, it has been found that more drastic morphological alterations that those commonly found in Finland do not necessarily lead to a biodiversity loss. Recently Feld et al. (2014) found that although macroinvertebrate assemblages changed distinctly along a morphological degradation gradient, overall species diversity remained the same. In Finland the main morphological modification has been channelisation of riffle habitat, mainly for the purpose of timber floating (Nilsson et al. 2005; Muotka and Syrjänen 2007). This practice has resulted in a modified channel configuration, but does not seems to cause such major changes to the community structure of macrophytes, diatoms or macroinvertebrates as diffuse pollution does (despite land use being ranked as low to medium pressure at the European level). Channelisation for timber floating usually requires only the removal of the largest boulders and wood, so that habitat loss is partial (Louhi et al. 2011): it might be that the reduction in morphological variability is not sufficient to restrict community structure and diversity. Indeed, Lepori et al. (2005) and Louhi et al. (2011) found no effect of increased habitat heterogeneity in channelized streams on macroinvertebrates.

Many stream restoration projects have focused on re-establishment of natural morphological variability to degraded stream reaches (Bernhardt and Palmer 2011). As speculated by many authors (e.g. Pretty et al. 2003; Palmer et al. 2010; Stranko et al. 2011; Sundermann et al. 2011), the common reason for the lack of response to increased habitat heterogeneity might be that poor water quality and land use management restrict the positive response to morphological stream restoration. Our results, indicating strong land use effects and weak HyMo-effects, strongly support this idea.

10.5 Conclusions

Land use induced diffuse pollution is an overruling pressure in European river ecosystems. Macrophyte and diatom BQE community structure is sensitive to low levels of land use pressure. The community changes due to land use pressure seem to be more drastic than those caused by alterations of channel morphology alone.

Biological community responses to morphological alterations were observable only when land use pressure was minimal. Even then, BQE community responses were not always detected. The results indicate that community level responses of the BQEs to channel morphology are difficult to detect with current biomonitoring datasets. Morphological alterations are typically monitored in a coarse way which, even if land use is controlled for, hinders the detection of community changes due to altered channel morphology.

Restoration of hydromophological conditions in rivers is not likely to result in improved ecological conditions, unless diffuse pollution and associated stresses due to land use are mitigated.

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11 Modelling hydromorphology - water quality interactions: weir removal

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11.1 Introduction

The overall aim of REFORM is to provide a framework for improving the success of hydromorphological restoration measures to reach, in a cost-effective manner, target ecological status or potential of rivers. Success is defined as being hydromorphologically sustainable, ecologically effective, and exploiting the full potential within the socio-economic setting. To achieve this aim the REFORM consortium in deliverable D3.2 is considering multi-stressor interactions. A key interaction is that between eutrophication and various forms of hydromorphological alteration. This is a widespread form of multi-stressor impact found across Europe and is the subject of various chapters in Deliverable 3.2. Here we consider what would happen following a restoration project which removes a weir and consider the impact on nutrient flux in the channel. In doing so we fulfill the requirement in Task 3.2 to use of a pre-existing, calibrated dynamic process-based catchment water quality model to assess the relative impact of hydromorphological degradation on water quality processes; specifically testing the following interactions: Flow + weirs ->nutrient dilution/retention -> algal production.

In over 65% of river basin districts in the EU the removal of structures such as weirs is proposed as a hydromorphological measures to meet WFD requirements (Kristensen 2013). No other measure is proposed to such a widespread degree. Whilst the benefits described below that are accrued when weirs are removed are widely acknowledged, what is not known is the impact on many aspects of water quality. There are some studies that quantify impacts of weir removal but only in terms of hypothetical scenarios using models, and often as part of scenarios including other restoration practices (e.g. in terms of denitirifcation: Wagenschein and Rode 2008). The self-purification of water in river channels is a potentially important ecosystem service as biotic communities have the potential to remove excess nutrients.

The removal or installation of weirs has a potential impact on ecosystem services (Bryan et al. 2013): notably water purification, fish habitat and hazard regulation (in terms of flood control). Specific emphasis is warranted on evaluating river restoration and its impact on river flows and water quality (Wagenschein and Rode 2008), notably temperature, nutrients, dissolved oxygen, sediment, and algae). The presence or absence (and design) of a weir has implications for (i) fish passage and habitat (river ecosystem and environmental conditions), and (ii) flood risk. Weirs are useful to help gauge river flows and removing them has an adverse effect on quality of monitoring networks. Weirs also provide aeration, the benefits of which can last over many kilometers downstream and are important in very low gradient systems . Estimates using the QUESTOR model (Hutchins et al. 2010) suggest that the Skip Bridge weir on the River Nidd, the subject of the present study, served to elevate the 1997 daily mean dissolved oxygen content 1.3 km downstream by 0.46 mg L^{-1} on average.

The objective of the present study was to evaluate the impact of a weir on the nitrate retention occurring in the channel of a lowland river (the River Nidd) in Yorkshire North East England, UK. As a case study this provided a unique opportunity because a weir was removed in 1999. Two National River Flow Archive (NRFA) gauging stations bound a 15.8 km stretch from which the weir was removed, the downstream station being downstream of the site of the former weir (at Skip Bridge). Furthermore the stretch is only influenced by abstractions and inputs from tributaries and sewage effluents to a minor degree.

To achieve the objective, simulation using mathematical models of the hydraulics and water quality was undertaken for two calendar years, one before and one after 1999 The model was validated in terms of (i) hydraulic parameters using the English Environment Agency Acoustic Doppler Current Profiling data collected as spot measurements before and after the weir removal (ii) nitrate removal using nitrate measurements along the reach. The work described here will use hydraulic models (HEC-RAS) to represent river hydromorphology (hydraulics and banks, floodplain, river bed and basic vegetation characteristics) which when combined with river flow and quality models for catchment networks can be used to estimate water quality criteria. Specifically here, the impact of presence or absence of a weir on the rate of river channel denitrification is estimated. These estimates will be set in context of the assumptions inherent in the chosen modelling approach. From this we illustrate how quantification can be made of the impact weirs may have on the service of water purification (nutrient nitrogen removal) along a river stretch.

11.2 Methods

The 15.8 km river stretch of the River Nidd in Yorkshire UK between Hunsingore (18m amsl) and Skip Bridge (8m amsl) was chosen for the case study. The catchment areas drained at the upstream and downstream ends of this stretch are 484.3 and 516.0 km^2 respectively. Mean annual rainfall of the catchment is 972 mm. Flows are gauged at each end of the stretch (gauging station IDs (27001 and 27062) and long term data reveal mean flow to increase from 7.95 to 8.29 m^{3}s^{-1} along the stretch. The catchment is predominantly agricultural grassland but with substantial areas of tilled land. The upland area (maximum amsl 703 m) is characterised by moorland and numerous reservoirs significantly affect runoff in this part of the catchment.

11.2.1Calculation of hydraulic variables

Applications of the HEC-RAS model (Hydrologic Engineering Centre (HEC); River System Analysis (RAS) – a hydraulic model developed by the US Army Corps of Engineers) were used to explore the hydraulic impact along a river stretch of removing or introducing a weir. HEC-RAS was used to represent the geomorphology of the stretch on the river Nidd (a tributary of the Ouse) between Hunsingore gauging station (27001) and Skip Bridge (27062) and to perform river hydraulics calculations. The output of the model, velocity and hydraulic depth, were then transferred into a denitrification model used for simulating the impact of the weir on downstream nitrogen retention. Two model run types were performed: with weir and without (see Table 11.1 for details). The HEC-RAS software permits one-dimensional (1D) steady and unsteady river flow hydraulics calculations, sediment transport computations and water quality analyses (Brunner 2010a; Brunner 2010b; Warner et al. 2010). The key element is that all those 4 components use a common geometric data representation, and common geometric and hydraulic computation routines. Daily data on river discharge were provided by the English Environment Agency (for details see Booker and Dunbar 2008). River geometry (slopes, elevations, cross section dimensions, and distances between cross sections) and the Manning's roughness coefficients of the river bed and banks were derived from the ISIS model provided by the English Environment Agency. The design of the weir at the downstream end of the river stretch (Skip Bridge weir) was provided by the NRFA (UK). In the case study we conducted steady flow for calculating water surface profiles for steady gradually varied flow. Gradually varied flow is characterized by minor changes in water depth and velocity from cross-section to cross-section. The steady flow component in HEC-RAS is capable of modelling subcritical, supercritical and mixed flow regime.

11.2.2Denitrification calculation

Data spanning a wide range of river environments worldwide has revealed the fraction of nitrate denitrified by micro-bacterial reactions in bed sediments to be closely related to the hydraulic load (Seitzinger et al., 2002). Hydraulic load is represented in the denitrification rate (k) calculated here which was undertaken on a daily basis through 1997 and 2000 using the equation applied by Whitehead and Williams (1982):

$$\boldsymbol{k} = \frac{\boldsymbol{a}}{\boldsymbol{h}} \cdot \mathbf{10}^{\mathbf{0.0293} \cdot \boldsymbol{\theta}} \qquad \text{[Eq. 1]}$$

Where: (i) a = 0.05 (a constant based on a range of UK river basin water quality studies), (ii) h = depth (m), (iii) θ = water temperature (°C). Temperatures were taken from the EA WIMS database of periodic (fortnightly or weekly) monitoring, and daily values interpolated from these.

When reformatted as a rate expression, the travel time (in days) is accounted for to derive the nitrate concentration at the downstream end of the reach (c_t) from the concentration input at the top (c_0):

$$\boldsymbol{c_t} = \boldsymbol{c_0} \cdot \boldsymbol{e}^{-\boldsymbol{kt}} \qquad \text{[Eq. 2]}$$

Where: (i) travel time (t) is derived from velocity as estimated by HEC-RAS, (ii) daily series of nitrate concentrations (mg N L^{-1}) at Hunsingore (c_0) are taken from an existing application of the QUESTOR water quality model (Hutchins et al. 2010)

Conceptually, the value of "a" (0.05) embodies the inclusion in our estimates of denitrification rate of factors that are attributable to the characteristics of the bed sediment. These characteristics are obviously highly localised in nature.

i. Performed runs

In making an assessment of the effect of a weir on the nitrate retention in the river channel the impact of ambient hydro-climatological conditions is likely to be large and needs to be controlled. Therefore four year-long model runs were undertaken (Table 11.1):

Table 11.1 Performed model runs

Run	Inputs (daily flows and nitrate concentrations at the upstream boundary)	Weir present	Data for validation available?
1	1997	Yes	Yes
2	2000	No	Yes
3	1997	No	No. Counterfactual
4	2000	Yes	No. Counterfactual
5	21st June 2013	No	Yes

11.3 Validation

The model was validated in two respects.

11.3.1Testing HEC-RAS model

The performance of HEC-RAS was assessed against Acoustic Doppler Current Profile data which had been collected periodically by the English Environment Agency (Table 11.2).Testing hydraulic components of the model for specific days during 1997 and 2000 EA ACDP data were collected at Skip Bridge and these were used to test the model. In addition the model was run for some other specific days in 1995, 1998 and 1999 when ACDP data were collected. ACDP provides data on hydraulic depth and velocity (variables required as input tot eh water quality model) which. In addition cross sections were measured at the time of ACDP, permitting flow area to be calculated and compared with the HEC-RAS model outputs.

Table 11.2 Performance of HEC-RAS at simulating velocity (v), depth (d) and flow area at the downstream (Skip Bridge) end of the stretch for 10 days of monitoring: A. when weir was still in place, B. after weir removed

A

Date	Model run, measured Q		Measurements		% relative error		Absolute error	
	v (m/s)	flow area (m^2)	v (m/s)	flow area (m^2)	v %	flow area %	v	flow area
15/09/1995	0.32	4.28	0.39	3.883	17.95	-10.22	0.07	0.397
15/09/1995	0.32	4.28	0.389	3.903	17.74	-9.66	0.069	0.377
18/09/1995	0.32	4.28	0.369	3.684	13.28	-16.18	0.049	0.596
10/12/1997	0.67	49.57	0.785	42.556	14.65	-16.48	0.115	7.014
09/01/1998	0.81	90.02	0.763	95.768	-6.16	6	0.047	5.748
15/01/1998	0.6	25.54	0.548	28.192	-9.49	9.41	0.052	2.652
16/06/1998	0.57	19.75	0.493	22.783	-15.62	13.31	0.077	3.033
03/03/1999	0.71	59.99	0.826	51.225	14.04	-17.11	0.116	8.765

B

Date	Model run, measured Q		Measurements		% relative error		Absolute error	
	v (m/s)	flow area (m^2)	v (m/s)	flow area (m^2)	v %	flow area %	v	flow area
10/08/2000	0.24	0.7	0.25	0.71	4	1.41	0.01	0.01
20/03/2000	0.32	0.87	0.32	0.86	0	-1.16	0	0.01

ii. Testing model against nitrate concentrations

In addition the model was run on individual days. The model was set up for 21st June 2013 when nitrate data were collected along the stretch of the Nidd covered by the modelling study (Run 5). Information about change in nitrate concentrations along the stretch was not available, therefore data were collected during a low flow period in summer 2013 (21st June). The stretch of the River Nidd has three influences:

Hunsingore STW

Fleet Beck Tributary which includes the Tockwith STW

Kirk Hammerton

For the validation application of 21st June 2013, it was necessary to make an estimate of the nitrate load entering the river along the stretch. The values for flow and nitrate-N concentration in these influences were set as follows (Table 11.3).

Table 11.3 Flow and nitrate-N concentrations of the small tributaries and point sources joining the River Nidd along the stretch

	Flow (m^3 s^{-1})	N (mgL^{-1})	HEC-RAS Reach ID
Hunsingore STW	0.002	15	22243
Fleet Beck	0.05	5	15474
Tockwith STW	0.02	11.79	15474
Kirk Hammerton WPC	0.01	20.4	10676.2

Flows are based on people equivalents served by the works. For Fleet Beck the catchment area was estimated as being approximately 15 km^2, the Q70 value from a nearby small river in the hydrometric register (0.06 m^3 s^{-1}, at Cundall Beck NGR SE419724) was taken and scaled by catchment area accordingly.

For nitrate measured mean values of total inorganic nitrogen were used, it being assumed that all ammonium in sewage effluents would be oxidised to nitrate in surface waters. For Fleet Beck a mean summer value (5 mg N L^{-1}) for rivers in the nearby locality appearing in the EA WIMS dataset was used. For Hunsingore STW no data were available and a concentration of 15 mg N L^{-1} assumed.

11.4 Results

When running the models for 1997 (Run 1: weir present) and 2000 (Run 2: weir absent) results suggest at first glance the weir to be having a large beneficial effect, enhancing the removal of nitrate by denitrification (Table 11.4). It is apparent that denitrification is markedly more effective in percentage terms in the summer when conditions are warmer and flows lower (in both 1997 and 2000 respectively: Figure 11.1A and B). The absolute flux of nitrate-N from denitrification is less dominated by summer conditions as the input loads are higher in autumn through to spring, as is typical in rural rivers dominated by diffuse sources of nitrate rather than sewage inputs. However, conditions in 1997 and 2000 were contrasting. When re-running the model to remove the confounding influence of year-specific conditions (Runs 3 and 4) it became apparent that the impact of the weir is less marked although apparently still beneficial (Figure 11.1C). Clearly the variability in weather conditions in specific years has an over-riding influence on the amount of denitrification occurring along a stretch of river.

Table 11.4 The amount of denitrification occurring along the stretch of river

	1997: weir present (Run 1)	1997: weir absent (Run 3)	2000: weir present (Run 2)	2000: weir absent (Run 4)
kgN denitrified	31752	30940	22045	21663
Mean % of N denitrified	9.52	9.22	5.93	5.86
Overall fraction of N denitrified (%)	5.72	5.58	3.02	2.97

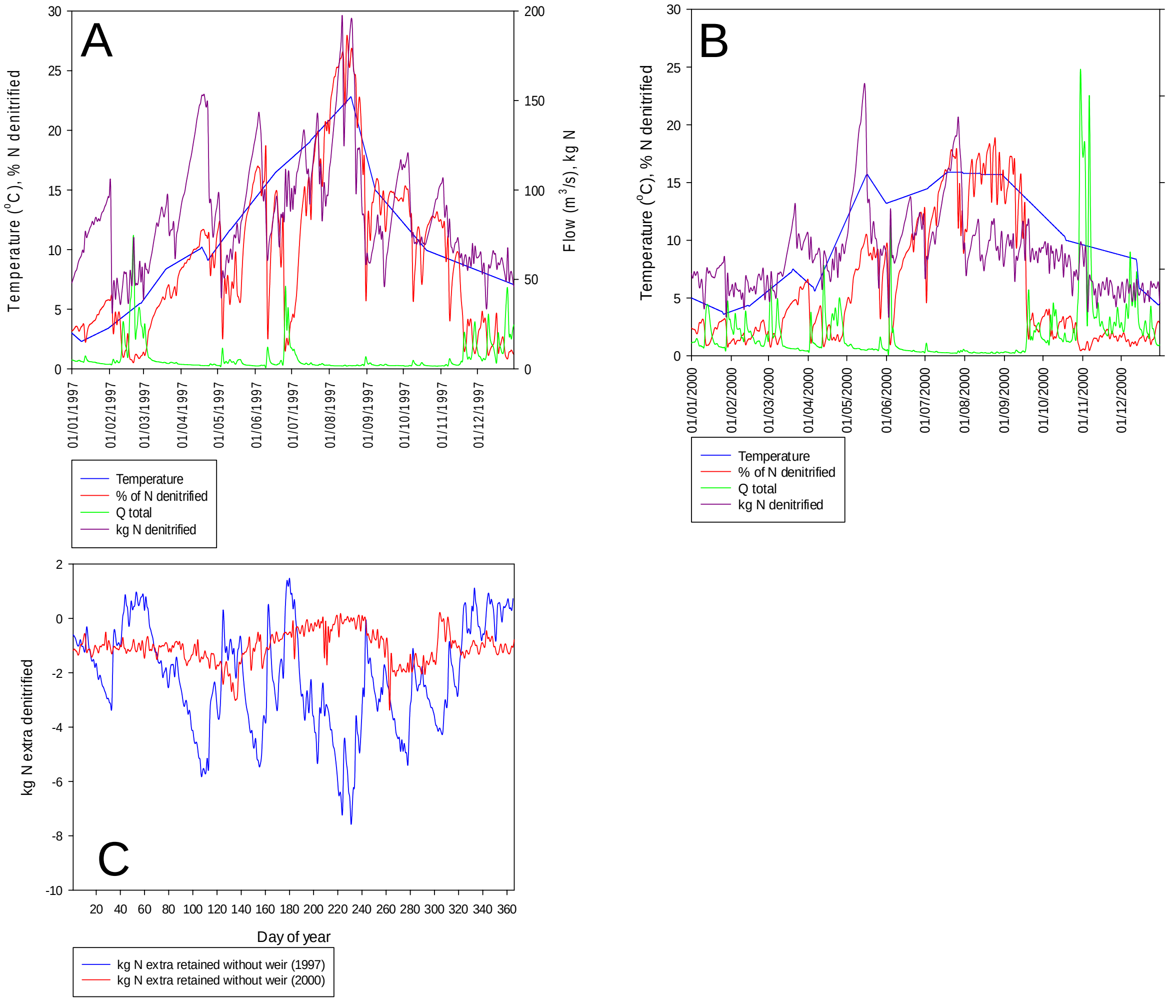

Figure 11.1 Simulations of denitrification using the process-based model (A. Year 1997, B. Year 2000, C. Impact of the weir on denitrification)

The variation in velocity, hydraulic depth and nitrate concentration as simulated by the model is illustrated for two individual days at low flow (2.854 m^3s^{-1} and 2.849 m^3s^{-1}respectively (

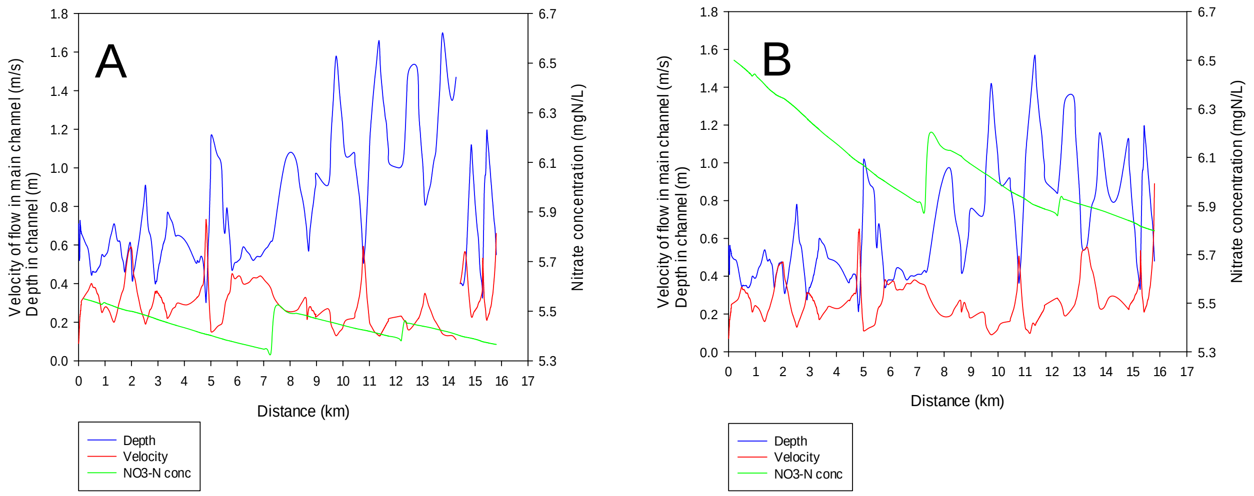

Figure 11.2 Figure 11.2A and B). It can be seen that the profiles of hydraulic depth and velocity are very similar until approximately 13 km along the stretch at which point the influence of the weir (a further 1.3 km downstream) becomes noticeable. Downstream of the weir, the patterns again converge within 0.5 km. The profiles of nitrate concentrations are notably different throughout, a feature which will be discussed later. In terms of hydraulic parameters the relative errors in velocity and flow area are all within 20%, and in the case of the situation in 2000 postdating weir removal, much smaller than this (Table 11.1). Validation of the nitrate concentrations is illustrated for 21st June 2013 (Figure 11.3).

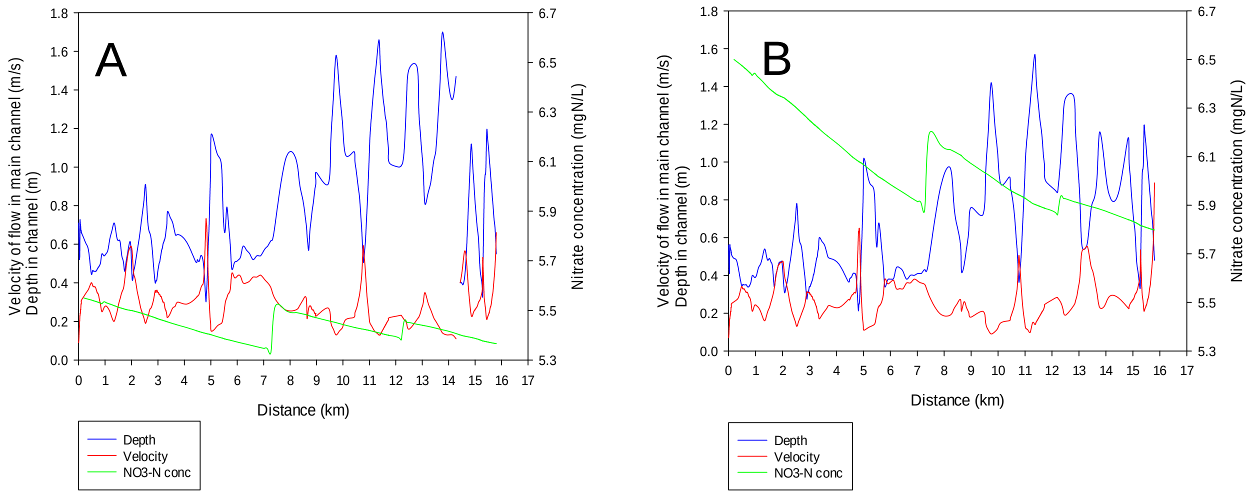

Figure 11.2 Variation of average velocity of flow in main channel, hydraulic depth in channel and nitrate concentration along the stretch for individual days (A: weir present 25/04/1997, B: weir absent 10/08/2000)

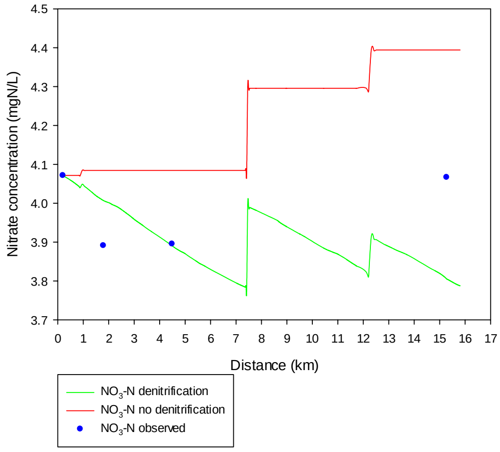

Figure 11.3 Observed and modelled nitrate concentrations on the 21st of June 2013

11.5 Discussion

11.5.1 What is the effect of removing a weir on denitrification?

Under 1997 conditions, removal of nitrate by denitrification would have been 812 kg N less if the weir had already been removed. In 2000 if the weir had still been present, an extra 382 kg N would have been denitrified. The biggest benefits of a weir are seen at low flows in summer (Figure 11.1 C) but can be considerable at low flows during other times. At most times the presence of a weir is beneficial and only during periods of elevated flow during 1997 was this not the case.

The land in the vicinity of the stretch of the River Nidd is predominantly agricultural. Results from the NALTRACES model (Hutchins 2012) suggest approximately 42 kg N ha^{-1} was leached from approximately 24 km^2 of this land in 2000. The detrimental effect of weir absence in 2000 roughly equates to the nitrate-N leached from 9 ha of this land. Projecting back to 1997 the beneficial effect of the weir at that time could have equated to leaching from 19 ha of land.

In terms of denitrification it is clear that the impacts of a weir may be substantial. What is not clear is how the river downstream from the weir will deal with the increased flux of N following weir removal. The capacity of the channel downstream to process more N is crucial. The Nitrate Vulnerable Zones (NVZ) action programme has resulted in a benefit of reduced nitrate leaching loads in protected areas of 5% (Lord et al. 2009). To offset the shortfall brought about by removal of a weir (in a year such as 1997 having conditions where impacts are largest) over 3 km^2 of land would need to be identified as a protected area under the UK Nitrate Vulnerable Zone designation.

The Nidd catchment is a typical lowland intensive agricultural area (mixed cropland and dairy cattle) of low gradient and moderately low annual rainfall. It is fed by more dilute headwaters draining moorland, hence has mean annual nitrate concentrations of approximately 4 mg N/L. Nevertheless catchments of this type can be a target for measures to reduce diffuse nitrate pollution as drinking water guidelines may be at risk of violation. The biogeographic situation of the Nidd is typical of the Humber River Basin District (area of 26000 km^2). If we are to assume 1997 and 2000 span a broad spectrum of conditions under present day climate, our research on weir impacts suggest that in a typical large European river basin district similar to the Humber a 10% increase in annual nitrate export to the marine environment (i.e. equating to mean annual nitrate concentration) will occur if between 70 and 200 similar weirs in locations comparable to that of the Nidd were to be removed.

In terms of considering whether or not weirs and eutrophication work in synergy it is clear they are antagonistic, in this context, and that the removal of a weir could have an indirect effect of increasing eutrophication impact by exporting N downstream. There is therefore a choice for managers do they retain weirs, and accept the local ecological impact and impact to migration or do they remove weirs and accept that will enhance downstream nutrient flux. A key consideration in future studies is therefore to consider the cumulative impact of weir removal and to consider the river lengths effected by enhanced N flux downstream of removed weirs. It is likely that the impact will be related to the retention time of the rivers which in turn is likely to be related to the river’s style. The categorisation of rivers into different styles within the REFORM project uses key parameters such as sinuosity which are directly related to the internal storage of a channel and the extent of dead zones. Taking this work forward it is recommended that any widespread review across river styles uses the REFORM river typology described in D2.1.

11.5.2 Why it is important to remove confounding factors to avoid misleading conclusions?

We emphasize that it is necessary as far as possible to remove confounding factors that are introduced via weather conditions. It is too easy to draw misleading conclusions by for example comparing model outputs between a dry year and a wet year (as illustrated by Table 11.4). As 2000 was in general a wetter and colder year than 1997 (as shown by comparing Figure 11.2 A and B) total denitrification was simulated to be approximately 50% lower. Also when comparing individual days (April 97 and August 2000 – which were specifically selected as being at low flow and in relatively warm periods) the simulated change in nitrate concentration is less in April 97 (0.012 mg L^{-1} km^{-1}) than in August 2000 (0.045 mg L^{-1} km^{-1}) despite our analysis revealing that a weir is beneficial. This is because the water temperature in August 2000 was considerably higher (15.8 ºC) then in April 1997 (9.4 ºC). As denitrification is modelled as a first order process with respect to nitrate concentration, the concentration of nitrate-N at the upstream end of the stretch is also significant in determining the rate of change along the stretch, concentrations being higher in August 2000 (6.5 mg L^{-1}) than in April 1997 (5.55 mg L^{-1}). Run 3 and Run 4 (see Table 11.1 for reference) were done in order to remove these confounding factors.

11.5.3 Assumptions

As illustrated on Figure 11.2 and Figure 11.3, the small tributaries and point source effluents joining the River Nidd along the stretch (listed in Table 3) serve to increase the nitrate concentrations in the main river channel (and thereby promote some additional denitrification). However these influences were not included in the whole-year calculations of nitrate retention because of lack of data on seasonal variation of flows and nitrate concentration in a natural tributary in this area. The input fluxes from Fleet Beck are likely to change seasonally. However when comparing the two 1997 model applications (Runs 1 and 3) and the two 2000 model applications (Runs 2 and 4) undertaken to evaluate the impact of weir removal the relative effects of these neglected influences will be the same.

The ISIS model contained data from 48 measured cross sections along the 15.8 km stretch. In HEC-RAS interpolation of cross sections was carried out between these measurements.

Weir design dimensions were only available in the case of Skip Bridge. For Hunsingore Weir such dimensions were lacking. However, given the evidence of weir backwater length in similar UK rivers (Samuels 1989) it is not thought that the Hunsingore Weir is likely to be influence the hydrological dynamics over 15 km downstream at Skip Bridge.

Using HEC-RAS it is not possible to include temporal variation of the coefficient related to roughness. The growth and die back of aquatic vegetation through the seasons is likely to have a considerable impact on channel roughness (Fathi-Moghadam and Drikvandi 2012; Hamill 1983; McGahey et al. 2008). Due to lack of suitable survey information to include such variation was outside the scope of the paper. It is thought that the impact of this assumption would be that a constant Manning N would represent an overestimate in the winter and an underestimate in the summer whilst preserving realistic annual average value. Underestimation in the summer is likely to have bigger implications for simulation of velocity, depth and nitrate removal than errors at other times of the year. However, the errors are introduced regardless of whether a weir is present and should not affect the relative values of nitrate removal (i.e. differences between model Runs 1-4).

Suitable topographic data describing the floodplain were not available for this stretch of river. Therefore it was assumed that all flow remained within bank. Whilst this may not be a valid assumption under high flow conditions in winter it is the low flow summer conditions that requires study as the vast majority of annual denitrification occurs at this time as shown in this work and by others (e.g. Whitehead and Williams 1982).

The characteristics of the bed sediment affect rates of denitrification. In the absence of observations it was assumed that sediment characteristics are invariant along the stretch. As recommended in other studies (Wagenschein and Rode 2008), further investigations should be carried out to relate sediment characteristics to measured rates of denitrification, for example as determined by Pattinson et al. (1998) in the nearby Ouse and Swale rivers using the acetylene blockage technique. Such research would refine estimates of denitrification and reduce the inherent uncertainties in the calculations. To reduce uncertainties in the calculation of denitrification with respect to weirs further investigations are needed, for example measurements of sediment characteristics upstream and downstream of weirs to refine denitrification rate constant.

11.6 Conclusions

A short stretch along the River Nidd in Yorkshire was studied. Here a weir was removed in 1999. Our model simulations suggest that weirs are beneficial in terms of denitrification, but only to a small extent. The benefits are largely seen during summer low flow periods.

The research has highlighted the importance of taking bed sediment measurement to identify the impact of local bed sediment characteristics on denitrification (to refine the value of “a” in Eqn 1)

It would be important to put these results in terms of nitrate change in the context of other water quality measurements such as phosphorus, phytoplankton and sediment. This can be done using the same modelling tools/protocol.

In the context of the proposed widespread removal of weirs across European river, a thorough evaluation of the trade-off between denitrification versus habitat enhancement should be undertaken on a catchment wide basis for all affected catchments.

11.7 References

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12 Supplementary material

12.1 Supplementary tables for chapter 7

Table 13.1: Selected pressures, processes and variables for the study site Traun

PRESSURE: River Fragmentation – Artificial Barriers Downstream of the Site

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
sediment	width/depth ratio	✓	-1	✓
transport	channel width	✓	-1	✓
LW transport	substrate size	✓	1	
armouring				

PRESSURE: Channelisation – Cross Section Alteration

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing	flood depth	✓	1	✓
	channel width	✓	-1	✓
sedimentation	water depth	✓	1	✓
sediment	thalweg altitutde	✓	1	✓
entrainment	substrate size	✓	1	
bank stabilization	thalweg variability	✓	-1	✓
vegetation	bank stability	✓	1	
encroachment	LWD	(✓)	-1	
	lateral connectivity	✓	-1	✓

PRESSURE: Alteration of Riparian Vegetation – Logging and Tree Removal – Transformation Into Farming Lands

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
bank erosion and	channel width	✓	-1	✓
failure	water depth	✓	1	✓
sediment	LWD	(✓)	-1	
entrainment				
vegetation				
recruitment				

PRESSURE: Water Abstraction – Groundwater Abstraction

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing	channel width	✓	-1	✓
sedimentation	water depth	✓	1	✓
vegetation	water velocity	✓	1	✓
encroachment				

*key: 1= increased, 0= slightly increased, steady or slightly decreased, -1= decreased

PRESSURE: Embankments, Levees or Lateral Dikes

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing sediment entrainment sediment transport armouring upstream sediment-entrainment	flood depth	✓	1	✓
	thalweg altitutde	✓	1	✓
	lateral connectivity	✓	-1	✓
	substrate size	✓	1	
	LWD	(✓)	-1	

PRESSURE: Sand and Gravel Extraction – Gravel Pits

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
bank erosion and failure	lateral connectivity	✓	-1	✓
	thalweg altitude	✓	1	✓
	substrate size	✓	1	
vegetation encroachment				

PRESSURE: Alteration of Instream Habitat – Bottom Rigidifying

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
bank erosion and failure	thalweg variability	✓	1	✓
	bank structure	✓	-1	
	variability	✓	1	
	bank stability	(✓)		
large wood transport	LWD			

*key: 1= increased, 0= slightly increased, steady or slightly decreased, -1= decreased

Table 13.2 Parameter for model evaluation (D3.3) study site Traun as result of 2D-modelling

River Traun Metrics		**Arithmetic Mean**	**Max. Value**	**Median**	**25% Quantil**	**75% Quantil**	**Variance**	**Standard Deviation**	**Empiric Var. Coefficient**
active channel width [m]	1885 -MJNQ$_t$	534,4	963,5	568,5	319,9	644,7	54298	233,02	0,44
	2006 -MJNQ$_t$	106,9	146,6	100,3	84,2	131,9	753	27,44	0,26
	1885 -MQ	664,8	1262,5	678,9	352,7	880,5	111978	334,63	0,50
	2006 -MQ	115,8	159,8	111,5	95,9	141,0	781	27,95	0,24
	1885 -HQ1	1032,0	1430,2	1014,1	845,8	1138,8	57048	238,85	0,23
	2006 -HQ1	124,4	171,9	115,4	102, 7	149,23	809	28,43	0,23
max depth [m]	1885 -MQ	1,52	2,02	1,46	1,36	1,64	0,06	0,24	0,16
	2006 -MQ	1,99	2,80	1,98	1,59	2,24	0,27	0,52	0,26
	1885 -HQ1	2,12	2,57	2,08	1,97	2,16	0,06	0,25	0,12
	2006 -HQ1	3,49	4,44	3,44	3,13	3,70	0,22	0,46	0,13
no of side channels	1885 -MJNQ$_t$	2,1	4	2,0	2,0	2,8	1,2	1,1	0,52
	1885 -MQ	3,0	4	3,0	2,3	4,0	1,1	1,1	0,35
	2006 -MJNQ$_t$	0,1	1	0,0	0,0	0,0	0,1	0,3	3,16
	2006 -MQ	0,1	1	0,0	0,0	0,0	0,1	0,3	3,16
width/ depth ratio	1885	494,4	697,7	488,2	400,0	567,1	16717	129,3	0,26
	2006	36,5	53,8	34,7	28,7	45,7	113	10,6	0,29

Table 13.3 Selected pressures, processes and variables for the study sites upper Mur

PRESSURE: Hydrological Regime Modification – Hydrological Alteration by Reservoirs

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing	Base flow (summer)	✓	-1	
	Phreatic level	✓	0	
vegetation encroachment	Channel width	✓	-1	✓
	Bank stability	✓	1	
	Riparian corridor width	✓	-1	
large wood deposition	LWD	(✓)	-1	

PRESSURE: Channelisation – Cross Section Alteration

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing	flood depth	✓	1	
	channel width	✓	-1	✓
sedimentation	water depth	✓	1	✓
sediment entrainment	thalweg altitutde	✓	0	✓
	substrate size	✓	1	
bank stabilization	thalweg variability	✓	-1	✓
vegetation encroachment	bank stability	✓	1	
	LWD	(✓)	-1	
	lateral connectivity	✓	0	✓

PRESSURE: Alteration of Riparian Vegetation – Logging and Tree Removal – Transformation Into Farming Lands

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
bank erosion and failure	channel width	✓	-1	✓
sediment entrainment	water depth	✓	1	✓
veget. recruitment	LWD	(✓)	-1	

PRESSURE: Hydropeaking – Short Term Flow Fluctuation by Hydropower Plants

Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing	water level rise rate	✓	0	
sediment entrainment	channel width	✓	-1	✓
	bank stability	✓	1	
vegetation encroachment	water velocity	(✓)	0	

PRESSURE: Alteration of Instream Habitat – Bottom Rigidifying

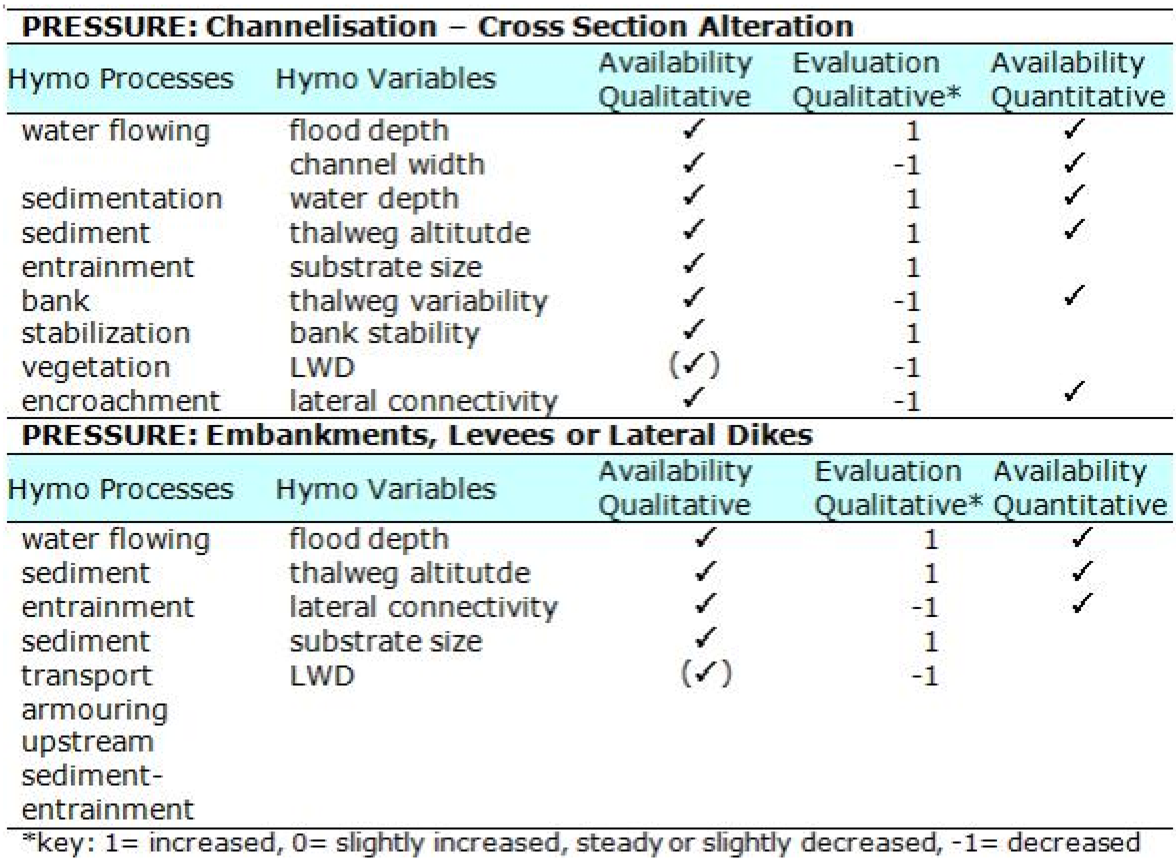
Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
bank erosion and failure	Thalweg variability	✓	1	✓
	Bank structure variability	✓	1	
	Bank stability	✓	-1	
large wood transport	LWD	(✓)	1	

*key: 1= increased, 0= slightly increased, steady or slightly decreased, -1= decreased

Table 13.4 Parameter for model evaluation (D3.3) study site upper Mur as result of 2D-modelling

	River Mur **2009-MJNQt** **Metrics**	**Arithmetic Mean**	**Max. Value**	**Median**	**25% Quantil**	**75% Quantil**	**Variance**	**Standard Deviation**	**Empiric Var. Coefficient**
SITE - 1	active channel width [m]	25,7	30,9	25,7	24,3	27,5	12,0	3,5	0,13
	max. Depth [m]	0,49	0,65	0,49	0,42	0,52	0,01	0,09	0,19
	width/depth ratio	54,7	82,9	54,9	47,0	61,5	194,5	13,9	0,26
	No. of side channels	0	0	-	-	-	-	-	-
SITE - 2	active channel width [m]	16,6	23,4	14,8	13,1	20,8	22,9	4,8	0,29
	max. Depth [m]	0,78	2,19	0,56	0,50	0,79	0,29	0,54	0,69
	width/depth ratio	28,7	66,9	23,8	18,0	40,0	327,9	18,1	0,63
	No. of side channels	0	0	-	-	-	-	-	-
SITE - 3	active channel width [m]	27,6	40,3	28,4	21,0	33,8	77,8	8,8	0,32
	max. Depth [m]	0,92	1,68	0,78	0,60	1,19	0,16	0,39	0,43
	width/depth ratio	38,0	66,2	41,4	19,2	53,5	483,1	22,0	0,58
	No. of side channels	0	1	0	0	0	0	0	2
SITE - 4	active channel width [m]	23,6	29,7	25,6	22,9	26,3	31,5	5,6	0,24
	max. Depth [m]	0,59	0,97	0,57	0,48	0,59	0,03	0,17	0,29
	width/depth ratio	44,4	68,2	47,3	32,7	52,1	322,7	18,0	0,40
	No. of side channels	0	0	-	-	-	-	-	-
SITE - 5	active channel width [m]	19,6	29,9	19,2	16,6	22,3	23,4	4,8	0,25
	max. Depth [m]	0,80	1,22	0,76	0,64	0,95	0,06	0,24	0,30
	width/depth ratio	27,3	45,8	25,8	18,0	35,5	143,0	12,0	0,44
	No. of side channels	0	0	-	-	-	-	-	-

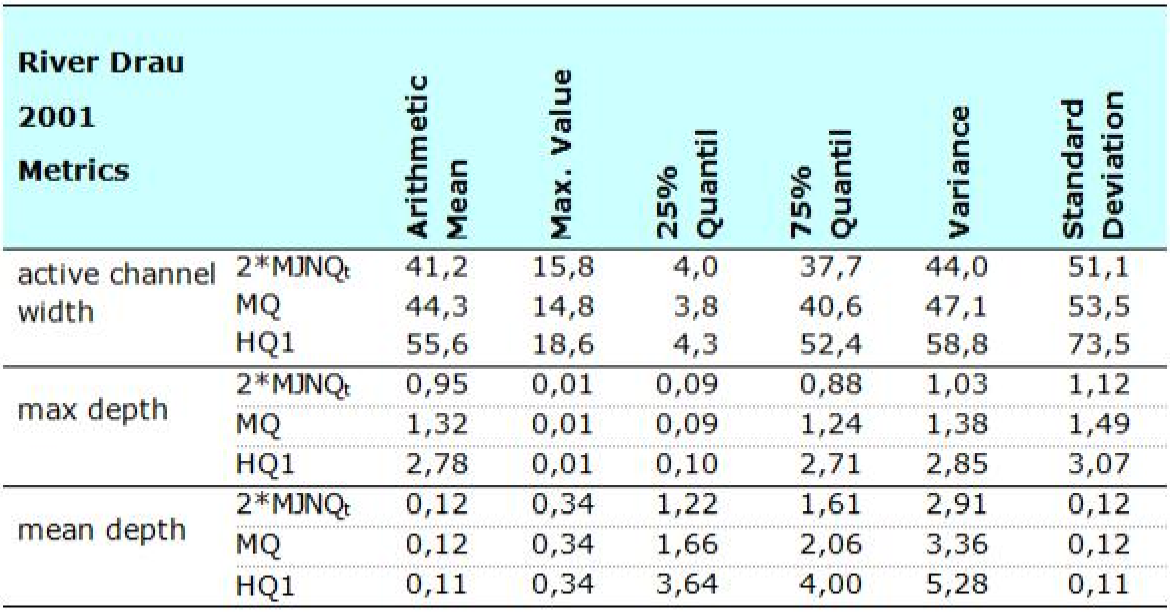
Table 13.5 Selected pressures, processes and variables for the study site Drau



PRESSURE: Channelisation – Cross Section Alteration				
Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing	flood depth	✓	1	✓
	channel width	✓	-1	✓
sedimentation	water depth	✓	1	✓
sediment	thalweg altitutde	✓	1	✓
entrainment	substrate size	✓	1	
bank	thalweg variability	✓	-1	✓
stabilization	bank stability	✓	1	
vegetation	LWD	(✓)	-1	
encroachment	lateral connectivity	✓	-1	✓
PRESSURE: Embankments, Levees or Lateral Dikes				
Hymo Processes	Hymo Variables	Availability Qualitative	Evaluation Qualitative*	Availability Quantitative
water flowing	flood depth	✓	1	✓
sediment	thalweg altitutde	✓	1	✓
entrainment	lateral connectivity	✓	-1	✓
sediment	substrate size	✓	1	
transport	LWD	(✓)	-1	
armouring				
upstream				
sediment-				
entrainment				

*key: 1= increased, 0= slightly increased, steady or slightly decreased, -1= decreased

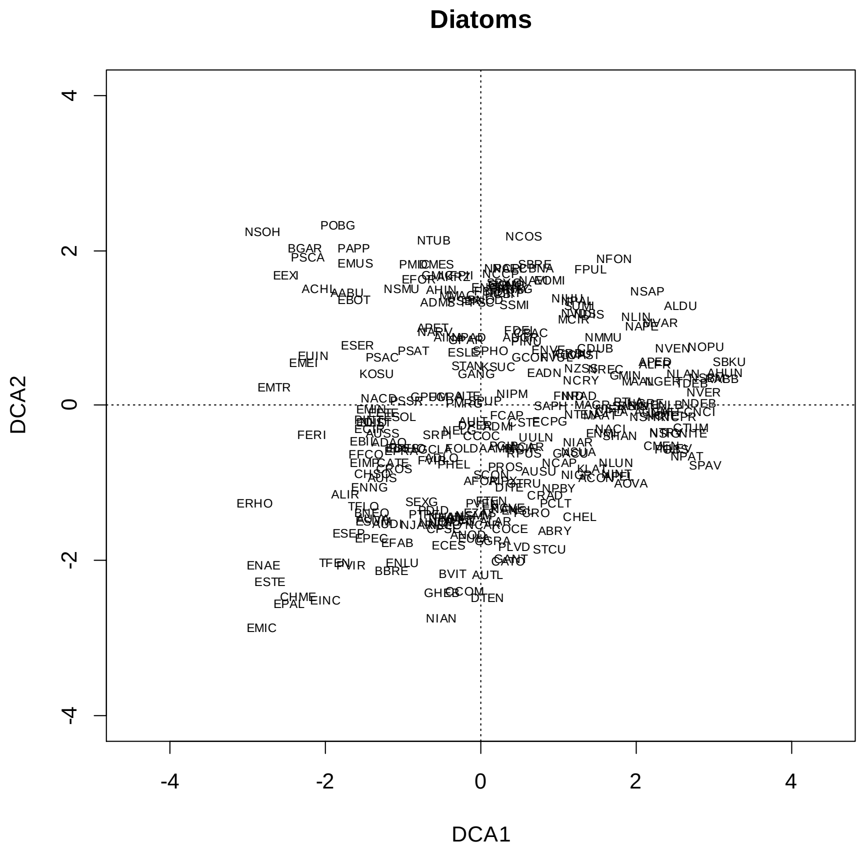
Table 13.6: Parameter for model evaluation (D3.3) study site Drau as result of 2D-modelling

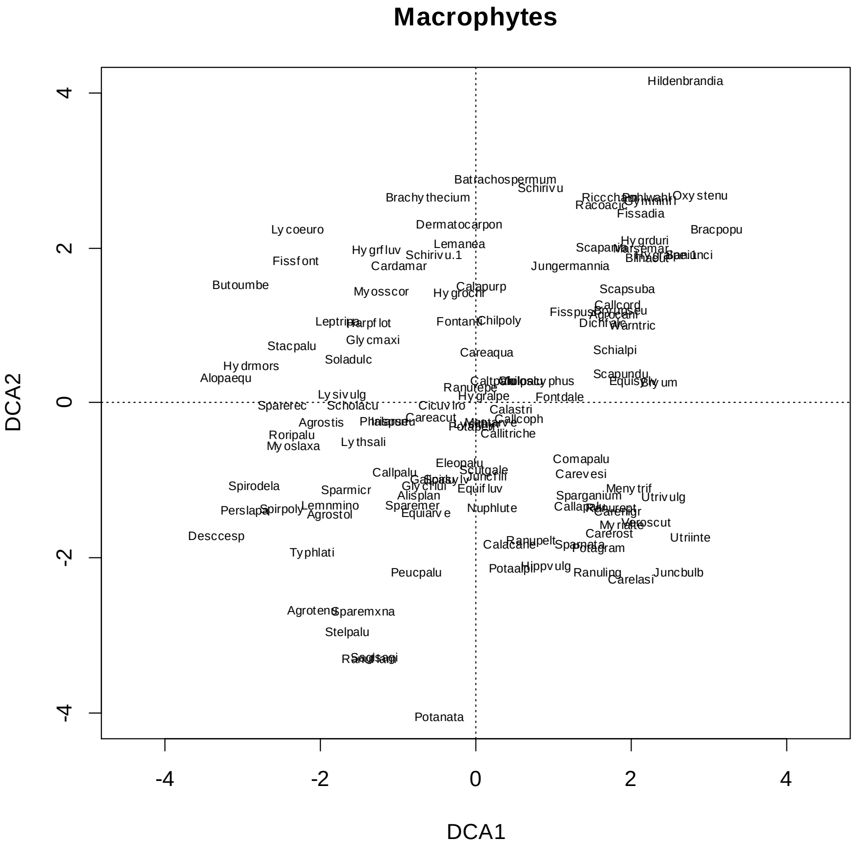


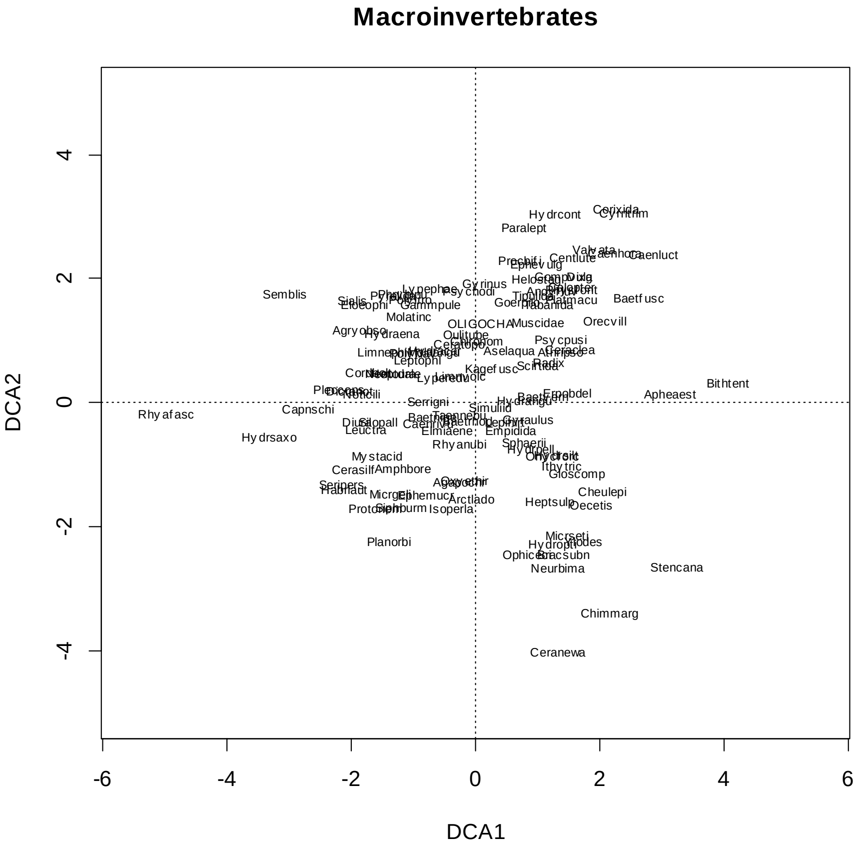
River Drau **2001** **Metrics**		**Arithmetic Mean**	**Max. Value**	**25% Quantil**	**75% Quantil**	**Variance**	**Standard Deviation**
active channel width	2*MJNQ$_t$	41,2	15,8	4,0	37,7	44,0	51,1
	MQ	44,3	14,8	3,8	40,6	47,1	53,5
	HQ1	55,6	18,6	4,3	52,4	58,8	73,5
max depth	2*MJNQ$_t$	0,95	0,01	0,09	0,88	1,03	1,12
	MQ	1,32	0,01	0,09	1,24	1,38	1,49
	HQ1	2,78	0,01	0,10	2,71	2,85	3,07
mean depth	2*MJNQ$_t$	0,12	0,34	1,22	1,61	2,91	0,12
	MQ	0,12	0,34	1,66	2,06	3,36	0,12
	HQ1	0,11	0,34	3,64	4,00	5,28	0,11

12.2 Supplementary material for chapter 8

Appendix I. The PCA-ordination of the diatom, macrophyte and macroinvertebrate species from 123 study sites from Finland (Dataset I). See other Appendix II, III and IV for the codes of the species.




Macrophytes
DCA2
DCA1
4
2
0
-2
-4
Hildenbrandia
Batrachospermum
Schirivu
Brachythecium
Riccchal
Pohlwahl
Oxystenu
Racoacic
Gymninf
Fissadia
Lycoeuro
Dermatocarpon
Bracpopu
Hygrfluv
Lemanea
Schirivu.1
Scapania
Hygrduri
Marsemar
Fissfont
Cardamar
Jungermannia
Blinacut
Hygrapal
Scapunci
Butoumbe
Myosscor
Hygrochr
Calapurp
Scapsuba
Callcord
Fisspusi
Brympseu
Agrocani
Leptripa
Harpflot
Fontanti
Chilpoly
Dichfalc
Warntric
Glycmaxi
Stacpalu
Careaqua
Soladulc
Schialpi
Hydrmors
Scapundu
Alopaequ
Caltpalu
Amblyrhy phus
Equisyl
Bryum
Ranurepe
Lysivulg
Hygralpe
Fontdale
Sparerec
Scholacu
Cicuviro
Calastri
Careacut
Callcoph
Agrostis
Phalarun
Irispseu
Potapect
Lycoeur
Mentarv e
Callitriche
Roripalu
Myoslaxa
Lythsali
Eleopalu
Comapalu
Callpalu
Scutgale
Carevesi
Galipalu
Sparsylv
Juncfili
Spirodela
Sparmicr
Glycflui
Equifluv
Menytrif
Alisplan
Sparganium
Utrivulg
Perslapa
Spirpoly
Lemnmino
Sparemer
Nuphlute
Callapalu
Ranurent
Agrostol
Equiarve
Carenigr
Veroscut
Myrialte
Desccesp
Carerost
Utriinte
Calacane
Ranupelt
Sparnata
Typhlati
Potagram
Hippvulg
Peucpalu
Potaalpi
Ranuling
Juncbulb
Carelasi
Agrotenu
Sparemxna
Stelpalu
Sagisagi
Ranuflam
Potanata
-4
-2
0
2
4


Macroinvertebrates
DCA2
DCA1
4
2
0
-2
-4
-6
-4
-2
0
2
4
6
Hydrcont
Corixida
Cymtrim
Paralept
Valvata
Caenhora
Caenluct
Prechifi
Centlute
Ephevulg
Compdixa
Dixa
Helostag
Gyrinus
Semblis
Sialis
Eloeophi
Lypephae
Psychodi
Gammpule
Andromy
Paralept
Tipulida
Goerpilo
Platmacu
Habanida
Baetfusc
Molatinc
OLIGOCHA
Muscidae
Orecvill
Agryobso
Hydraena
Oulitube
Chironom
Ceratopo
Psycpusi
Aselaqua
Athripso
Ceraclea
Limnephi
Leptophi
Radix
Kagefusc
Scirtida
Cordulia
Nepoidae
Lyperedu
Limnvolc
Bithtent
Plecons
Dicrano
Noticili
Erpobdel
Apheaest
Baetvern
Hydrangu
Serrigni
Rhyafasc
Capnschi
Simuliid
Taennebu
Baethera
Baetrhod
Caenriva
Diura
Silopall
Leuctra
Lepihirt
Gyraulus
Hydrsaxo
Elmiaene
Empidida
Rhyanubi
Sphaerii
Hydrpell
Hydrsilt
Mystacid
Ithytric
Cerasilf
Amphbore
Gloscomp
Oxyethir
Agapochir
Seripers
Habriaut
Micrgei
Ephemucr
Arctlado
Cheulepi
Heptsulp
Oecetis
Protonem
Siphburm
Isoperla
Micrseti
Planorbi
Hydropti
Ylodes
Ophicera
Bracsubn
Neurbima
Stencana
Chimmarg
Ceranewa

Appendix II. The codes of the diatom species used in Appendix I and the explanations.

Code	Name
AABU	Achananthes abundans
AAMB	Aulacoseira ambigua
ABRY	Adlafia bryophila
ACAR	Achnanthes carissima
ACHL	Achnanthes chilidanos
ACON	Achnanthes conspicua
ADAO	Achananthes daonensis
ADLO	Achnanthidium linearioides
ADMI	Achnanthidium minutissimum group II (mean width 2,2-2,8µm)
ADMS	Adlafia minuscula
AFOR	Asterionella formosa
AHIN	Achnanthes hintzii
AHUN	Achnanthes hungarica
AINA	Amphora inariensis
AIPX	Achnanthes impexa
AITE	Aulacoseira italica var. tenuissima
AKRZ	Achnanthes krantzil
ALAR	Achnanthes lanceolata ssp. Rostrata
ALDU	Achnanthes lanceolata ssp. dubia
ALFR	Achnanthes lanceolata ssp. Frequentissima
ALIR	Aulacoseira lirata
ANOD	Achnanthes nodosa
AOVA	Amphora ovalis
APED	Amphora pediculus
APEL	Amphipleura pellucida
APET	Achnananthes petersenii
AUAL	Aulacoseira alpigena
AUDI	Aulacoseira distans
AUGA	Aulacoseira granulata var. angustissima
AUGR	Aulacoseira granulata var. granulata
AUIS	Aulacoseira islandica var. islandica
AUIT	Aulacoseira italica
AUSS	Aulacoseira subarctica f. subborealis
AUSU	Aulacoseira subarctica
AUTL	Aulacoseira tenella
AUVA	Aulacoseira valida
BBRE	Brachysira brebissonii
BGAR	Brachysira garrensis
BNEO	Brachysira neoexilis
BVIT	Brachysira vitrea
CATE	Caloneis tenuis
CATO	Cyclotella atomus
CBAC	Caloneis bacillum
CBNA	Cymbopleura naviculiformis
CCOC	Cavinula cocconeiformis
CCOM	Cyclotella comta var. radiosa
CDUB	Cyclostephanos dubius
CHEL	Cymbella helvetica
CHME	Chamaepinnularia mediocris
CHSO	Chamaepinnularia soehrensis var. soehrensis
CMEN	Cyclotella meneghiniana
CNCI	Cymbella neocistula
COCE	Cyclotella ocellata
CPLA	Cocconeis placentula incl. varieties
CPSE	Cavinula pseudoscutiformis
CPST	Cyclotella pseudostelligera
CRAD	Cyclotella radiosa
CRBU	Craticula buderi
CROS	Cyclotella rossii
CSTE	Cyclotella stelligera
CTUM	Cymbella tumida
DITE	Diatoma tenuis
DMES	Diatoma mesodon
DPER	Diadesmis perpusilla
DTEN	Denticula tenuis
EADN	Epithemia adnata
EBIL	Eunotia biluaris
EBOT	Eunotia botuliformis
ECES	Encyonopsis cesatii
ECIR	Eunotia circumborealis
EDES	Encyonopsis descripta
EDIO	Eunotia diodon
EETE	Eunotia exigua var. tenella
EEXI	Eunotia exigua var. exigua
EFAB	Eunotia faba
EFOR	Eunotia formica
EIMP	Eunotia implicata
EINC	Eunotia incisa var. incisa
EMEI	Eunotia meisteri
EMIC	Eunotia microcephala
EMIN	Eunotia minor
EMTR	Eunotia muscicola var. Tridentula
EMUS	Eunotia muscicola var. muscicola
ENAE	Eunotia naegeli
ENCY	Encyonema
ENLB	Encyonema lange-bertalotii
ENLU	Encyonema lunatum
ENMI	Encyonema minutum
ENNG	Encyonema neogracile var. neogracile
ENSL	Encyonema silesiacum var. lata
ENVE	Encyonema ventricosum var. ventricosum
EOMI	Eolimna minima
EPAL	Entomoneis paludosa
EPEC	Eunotia pectinalis var. pectinalis
EPRA	Eunotia praerupta
ERHO	Eunotia rhomboidea

ESEP	Eunotia septentrionalis
ESER	Eunotia serra var. serra
ESLE	Encyonema silesiacum var. silesiacum
ESOL	Eunotia soleirolii
ESTE	Eunotia serra var. tetraedron
ESUM	Encyonopsis subminuta
EUIN	Eunotia intermedia
EULA	Eucocconeis laevis
EUNO	Eunotia
EZAS	Eunotia zasuminensis
FBID	Fragilaria bidens
FCAP	Fragilaria capucina var. capucina
FCBI	Fragilaria construens
FCPG	Fragilaria capucina group
FCRO	Fragilaria crotonensis
FCRP	Fragilaria capucina var. rumpens
FCVA	Fragilaria capucina var. vaucheriae
FCVE	Fragilaria construens f. venter
FDEL	Fragilaria delicatissima
FERI	Frustulia erifuga
FFCO	Fragilariforma constricta
FGRA	Fragilaria gracilis
FIND	Navicula indifferens
FNAN	Fragilaria nanana
FOLD	Fragilaria oldenburgiana
FPII	Fragilaria pinnata var. intercedens
FPSC	Fragilaria parasitica var. subconstricta
FPUL	Fragilaria pulchella
FRAG	Fragilaria
FRHO	Frustulia rhomboides
FTEN	Fragilaria tenera
FVIR	Fragilaria virescens
FVUL	Frustulia vulgaris
GACU	Gomphonema acuminatum
GANG	Gomphonema angustatum
GANT	Gomphonema angustum
GCLA	Gomphonema clavatum
GCON	Gomphonema constrictum
GGRA	Gomphonema gracile
GHEB	Gomphonema hebridense
GMIC	Gomphonema micropus
GMIN	Gomphonema minutum
GOMP	Gomphonema
GPAR	Gomphonema parvulum
GPUM	Gomphonema pumilum
GTRU	Gomphonema truncatum
KLAT	Achnanthes laterostrata
KOSU	Kobayasiella subtilissima
KSUC	Achnanthes suchlandtii
MAAL	Mayamaea atomus var. alcimonica
MAAT	Mayamaea atomus var. atomus
MAGR	Mayamaea agrestis
MCIR	Meridion circulare var. circulare
MMAC	Microcostatus maceria
MVAR	Melosira varians
NAAN	Navicula angusta
NACD	Nitzschia acidoclinata
NACI	Nitzschia acicularis
NAPE	Navicula atomus var. permitis
NARV	Navicula arvensis var. arvensis
NAVI	Navicula
NCAP	Navicula capitata
NCAR	Navicula cari
NCCP	Navicula cryptocephaloides
NCOS	Navicula costulata
NCPL	Nitzschia capitellata
NCPR	Navicula capitatoradiata
NCRY	Navicula cryptocephala
NCTE	Navicula cryptotenella
NDEB	Nitzschia debilis
NDIF	Navicula difficillima
NDIS	Nitzschia dissipata
NELG	Navicula elginensis
NELO	Naviculadicta elorantana
NFEN	Navicula fennica
NFON	Nitzschia fonticola var. fonticola
NGER	Navicula germainii
NGRE	Navicula gregaria
NHAN	Nitzschia hantzschiana
NIAN	Nitzschia angustata
NIAR	Nitzschia archibaldii
NIFR	Nitzschia frustulum var. frustulum
NIGR	Nitzschia gracilis
NIHU	Nitzschia hungarica
NINT	Nitzschia intermedia
NIPM	Nitzschia perminuta
NITE	Nitzchia tenuis
NJAR	Navicula jaemefeltii
NLAN	Navicula lanceolata
NLEV	Nitzschia levidensis var. levidensis
NLIN	Nitzschia linearis var. linearis
NLST	Navicula leptostriata
NLUN	Navicula lundii
NMLF	Navicula molestiformis
NMMU	Navicula miniscula var. Muralis
NNOT	Navicula notha
NOPU	Navicula oppugnata
NPAD	Nitzschia palea var. debilis
NPAE	Nitzschia paleacea
NPAL	Nitzschia palea var. palea
NPAT	Nitzschia palea var. tenuirostris
NPBY	Navicula pseudobryophila
NRAD	Navicula radiosa
NREC	Nitzschia recta
NRHY	Navicula rhynchocephala
NSAP	Navicula saprophila
NSBM	Navicula subminuscula
NSHR	Navicula schroeteri
NSIG	Nitzschia sigma
NSMM	Navicula schmassmannii

NSMU	Navicula submuralis
NSOH	Navicula soehrensis var. hassica
NSUA	Nitzschia subacicularis
NTEN	Navicula tenelloides
NTRV	Navicula trivialis
NTUB	Nitzschia tubicola
NVDS	Navicula seminulum
NVEN	Navicula veneta
NVER	Nitzschia vermicularis
NZSS	Nitzschia ssp.
PAPP	Pinnularia appendiculata
PCLT	Placoneis clementis
PDID	Psammothidium didymum
PGIB	Pinnularia gibba
PHEL	Psammothidium helveticum
PINT	Pinnularia interrupta
PINU	Pinnularia
PLVD	Psammothidium levanderi
PMIC	Pinnularia microstauron var. microstauron
PMRG	Psammothidium marginulatum
PNOD	Pinnularia nodosa var. nodosa
POBS	Pinnularia obscura
PROS	Psammothidium rossii
PSAC	Psammothidium sacculum
PSAT	Psammothidium subatomoides
PSBR	Pseudostaurosira brevistriata
PSCA	Pinnularia subcapitata var. subcapitata
PSSR	Pinnularia subcapitata var. subrostrata
PTLA	Planothidium lanceolatum
PTPE	Planothidium peragallii
PVEN	Psammothidium ventralis
PVIR	Pinnularia viridis var. viridis morphotype 1
RABB	Rhoicosphenia abbreviata
RPUS	Rossithidium pusillum
RSIN	Reimeria sinuata
SANG	Surirella angusta
SAPH	Surirella amphioxys
SBKU	Surirella brebissonii var. kuetzingii
SBRE	Surirella brebissonii var. brebissonii
SCON	Staurosira construens var. construens
SEXG	Stauroforma exiguiformis
SHAN	Stephanodiscus hantzschii
SPAV	Stephanodiscus parvus
SPHO	Stauroneis phoenicenteron
SPUP	Sellaphora pupula
SPYG	Stauroneis pygmaea
SRPI	Staurosira pinnata var. pinnata
SSMI	Stauroneis smithii var. smithii
STAN	Stauroneis anceps
STCU	Stenopterobia curvula
SUMI	Surirella minuta
TDEB	Tryblionella debilis
TFEN	Tabellaria fenestrata
TFLO	Tabellaria flocculosa
UULN	Ulnaria ulna var. ulna

Appendix III. The codes of the macrophyte taxa used in Appendix I and VIII the explanations.

Code	Name
Achiptar	Achillea ptarmica
Acorcala	Acorus calamus
Agrocani	Agrostis canina
Agrostol	Agrostis stolonifera
Alisplan	Alisma plantago aquatica
Alopaequ	Alopecurus aequalis
Alopgeni	Alopecurus geniculatus
Amblfluv	Amblystegium fluviatile
Ambltena	Amblystegium tenax
Andrpoli	Andromeda polifolia
Aneuping	Aneura pinguis
Apiunodi	Apium nodiflorum
Beruerec	Berula erecta
Bideradi	Bidens radiata
Bidetrip	Bidens tripartita
Blinacut	Blindia acuta
Bracrivu	Brachythecium rivulare
Bryupseu	Bryum pseudotriquetrum
Bryuweig	Bryum weigelii
Butoumbe	Butomus umbellatus
Calacane	Calamagrostis canescens
Calastri	Calamagrostis stricta
Callapalu	Calla palustris
Callbrut	Callitriche brutia
Callcord	Calliergon cordifolium
Callcoph	Callitriche cophocarpa
Callgiga	Calliergon giganteum
Callobtu	Callitriche obtusangula
Callpalu	Callitriche palustris
Callplat	Callitriche platycarpa
Callstag	Callitriche stagnalis
Calltrun	Callitriche truncata ssp. occidentalis
Caltpalu	Caltha palustris
Calysepi	Calystegia sepium
Cardamar	Cardamine amara
Cardprat	Cardamine pratensis
Careacut	Carex acuta
Careacxni	Carex acuta x nigra
Careaqua	Carex aquatilis
Careelat	Carex elata
Carelasi	Carex lasiocarpa
Carelimo	Carex limosa
Carenigr	Carex nigra
Carepani	Carex paniculata
Carepend	Carex pendula
Carerost	Carex rostrata
Carevesi	Carex vesicaria
Cataaqua	Catabrosa aquatica
Cerademe	Ceratophyllum demersum
Chilpall	Chiloscyphus pallescens
Chilpoly	Chiloscyphus polyanthos
Chryoppo	Chrysosplenium oppositifolium
Cicuviro	Cicuta virosa
Cincaqua	Cinclidotus aquaticus
Cincdanu	Cinclidotus danubicus
Cincfont	Cinclidotus fontinaloides
Cincripa	Cinclidotus riparius
Climdend	Climacium dendroides
Comapalu	Comarum palustre

Conoconi	Conocephalum conicum
Cratcomm	Cratoneuron commutatum
Cratfili	Cratoneuron filicinum
Desccaes	Deschampsia caespitosa
Dichfalc	Dichelyma falcatum
Egerdens	Egeria densa
Eleopalu	Eleocharis palustris
Elodcana	Elodea canadensis
Elodnutt	Elodea nuttallii
Epilhirs	Epilobium hirsutum
Epilpalu	Epilobium palustre
Epilpavi	Epilobium paviflorum
Equiarve	Equisetum arvense
Equifluv	Equisetum fluviatile
Equipalu	Equisetum palustre
Equisylv	Equisetum sylvaticus
Erioangu	Eriophorum angustifolium
Eupacann	Eupatorium cannabinum
Falljapo	Fallopia japonica
Festrubr	Festuca rubra
Filaalga	Filamentous algae
Filiulma	Filipendula ulmaria
Fissadia	Fissidens adianthoides
Fisscras	Fissidens crassipes
Fissgran	Fissidens grandifrons
Fissosmu	Fissidens osmundoides
Fisspusi	Fissidens pusillus
Fissrivu	Fissidens rivularis
Fissrufu	Fissidens rufulus
Fisstaxi	Fissidens taxifolius
Fontanti	Fontinalis antipyretica
Fontdale	Fontinalis dalecarlica
Fonthypn	Fontinalis hypnoides
Fontsqua	Fontinalis squamosa
Galimoll	Galium mollugo
Galipalu	Galium palustre
Glechede	Glechoma hederacea
Glycflui	Glyceria fluitans
Glycmaxi	Glyceria maxima
Groedens	Groenlandia densa
Harpflot	Harpanthus flotovianus
Hippvulg	Hippuris vulgaris
Hydrmors	Hydrocharis morsus ranae
Hygralpe	Hygrohypnum alpestre
Hygrduri	Hygrohypnum duriscululm
Hygrluri	Hygrohypnum luridum
Hygrochr	Hygrohypnum ochraceum
Irispseu	Iris pseudacorus
Isoeechi	Isoetes echinospora
Isoelacu	Isoetes lacustris
Juncacut	Juncus acutiflorus
Juncarti	Juncus articulatus
Juncbufo	Juncus bufonius
Juncbulb	Juncus bulbosus
Junccong	Juncus conglomeratus
Junceffu	Juncus effusus
Juncfili	Juncus filiformis
Juncinfl	Juncus inflexus
Jungatro	Jungermannia atrovirens
Jungexse	Jungermannia exsertifolia ssp. cordifolia
Lemngibb	Lemna gibba

Lemnmino	Lemna minor	Nuphx.sp	Nuphar x sp.enneriana
Lemnminu	Lemna minuta	Nympalba	Nymphaea alba
Lemntris	Lemna trisulca	Nymptetr	Nymphaea tetragona
Leptripa	Leptodictyum riparium	Octofont	Octodiceras fontanum
Lobedort	Lobelia dortmanna	Oenacroc	Oenanthe crocata
Lotupedu	Lotus pedunculatus	Osmurega	Osmunda regalis
Ludwpepl	Ludwigia peploides	Pedipalu	Pedicularis palustris
Lunucruc	Lunularia cruciata	Pellsp.	Pellia sp.
Lycoeuro	Lycopus europaeus	Persamph	Persicaria amphibia
Lysinumm	Lysimachia nummularia	Pershydr	Persicaria hydropiper
Lysithyr	Lysimachia thyrsiflora	Petahybr	Petasites hybridus
Lysivulg	Lysimachia vulgaris	Peucpalu	Peucedanum palustre
Lythport	Lythrum portula	Phalarun	Phalaris arundinacea
Lythsali	Lythrum salicaria	Philfont	Philonotis fontana
Marcsp.	Marchantia sp.	Phraaust	Phragmites australis
Marsemar	Marsupella emarginata	Plagdent	Plagiothecium denticulatum
Mentaqua	Mentha aquatica	Platripa	Platyhypnidium riparioides
Mentarve	Mentha arvensis	Poapalu	Poa palustris
Mentlong	Mentha longifolia	Poatriv	Poa trivialis
Menytrif	Menyanthes trifoliata	Pohlwahl	Pohlia wahlenbergii
Molicaer	Molinia caerulea	Polylapa	Polygonum lapathifolium
Myoscesp	Myosotis cespitosa	Polymite	Polygonum mite
Myoslaxa	Myosotis laxa	Porecord	Porella cordeana
Myosscor	Myosotis scorpioides	Porepinn	Porella pinnata
Myrialte	Myriophyllum alterniflorum	Potaalpi	Potamogeton alpinus
Myrisibi	Myriophyllum sibiricum	Potaberc	Potamogeton berchtoldii
Myrispic	Myriophyllum sp. icatum	Potabexob	Potamogeton berchtoldii x obtusifolius
Myrivert	Myriophyllum verticillatum	Potacris	Potamogeton crispus
Nastoffi	Nasturtium officinale	Potagram	Potamogeton gramineus
Nuphlute	Nuphar lutea	Potagrxpe	Potamogeton gramineus x perfoliatus
Nuphpumi	Nuphar pumila	Potanata	Potamogeton natans

Potanodo	Potamogeton nodosus
Potapect	Potamogeton pectinatus
Potaperf	Potamogeton perfoliatus
Racoacic	Racomitrium aciculare
Ranuaqua	Ranunculus aquatilis var aquatilis
Ranucirc	Ranunculus circinatus
Ranudiff	Ranunculus aquatilis var diffusus
Ranuflam	Ranunculus flammula x reptans
Ranuflui	Ranunculus fluitans
Ranuling	Ranunculus lingua
Ranupelt	Ranunculus peltatus
Ranupeni	Ranunculus penicillatus
Ranurepe	Ranunculus repens
Ranurept	Ranunculus reptans
Ranuscel	Ranunculus sceleratus
Rhizsp.	Rhizomnium sp.
Ricccham	Riccardia chamaedryfolia
Roriamph	Rorippa amphibia
Roripalu	Rorippa palustris
Rumeacet	Rumex acetosa
Rumeaqua	Rumex aquaticus
Rumecris	Rumex crispus
Rumehydr	Rumex hydrolaphatum
Rumeobtu	Rumex obtusifolius
Saginata	Sagittaria natans
Saginodo	Sagina nodosa
Sagisagi	Sagittaria sagittifolia
Sagix.lu	Sagittaria x lunata
Scapsuba	Scapania subalpina
Scapundu	Scapania undulata
Schiagas	Schistidium agassizii
Schiapoc	Schistidium apocarpum
Schirivu	Schistidium rivulare
Scholacu	Schoenoplectus lacustris
Scirsylv	Scirpus sylvaticus
Scroauri	Scrophularia auriculata
Scutgale	Scutellaria galericulata
Sparemer	Sparganium emersum
Sparemxna	Sparganium emersum x natans
Sparerec	Sparganium erectum
Sparglom	Sparganium glomeratum
Sparmicr	Sparganium microcarpum
Sparnata	Sparganium natans
Sphaesp.	Sphaerotilus sp.
Sphagsp.	Sphagnum sp.
Spirpoly	Spirodela polyrhiza
Stacpalu	Stachys palustris
Stelalsi	Stellaria alsine
Stelpalu	Stellaria palustris
Sympoffi	Symphytum officinale
Thalflav	Thalictrum flavum
Thamalop	Thamnobryum alopecurum
Tussfarf	Tussilago farfara
Typhlati	Typha latifolia
Urtidioi	Urtica dioica
Utriinte	Utricularia intermedia
Utrimino	Utricularia minor
Utriochr	Utricularia ochroleuca
Utrivulg	Utricularia vulgaris
Warnproc	Warnstorfia procera
Warntric	Warnstorfia trichophylla
Veroanag	Veronica anagallis aquatica

Verobecc	Veronica beccabunga	Violpalu	Viola palustris
Verolong	Veronica longifolia	Zannpalu	Zannichellia palustris
Veroscut	Veronica scutellata		
Vicisp.	Vicia sp.		

Appendix IV. The codes of the macroinvertebrate taxa used in Appendix I and the explanations.

Code	Taksoni
OLIGOCHA	OLIGOCHAETA
Gloscomp	Glossiphonia complanata
Helostag	Helobdella stagnalis
Erpobdel	Erpobdella
Valvata	Valvata
Bithtent	Bithynia tentaculata
Radix	Radix
Planorbi	Planorbidae
Physfont	Physa fontinalis
Gyraulus	Gyraulus
Ancyfluv	Ancylus fluviatilis
Sphaerii	Sphaeriidae
Hydracar	Hydracarina
Aselaqua	Asellus aquaticus
Gammpule	Gammarus pulex
Leptophl	Leptophlebia
Paralept	Paraleptophlebia
Habrlaut	Habrophlebia lauta
Ephevulg	Ephemera vulgata
Ephemucr	Ephemerella mucronata
Serrigni	Serratella ignita
Caenhora	Caenis horaria
Caenluct	Caenis luctuosa
Caenrivu	Caenis rivulorum
Heptdale	Heptagenia dalecarlica
Kagefusc	Kageronia fuscogrisea
Heptsulp	Heptagenia sulphurea
Baetrhod	Baetis rhodani
Baetnige	Baetis niger group
Baetvern	Baetis vernus group
Baetfusc	Baetis fuscatus
Centlute	Centroptilum luteolum
Procbifi	Procloeon bifidum
Calopter	Calopteryx
Gompvulg	Gomphus vulgatissimus
Ophiceci	Ophiogomphus cecilia
Onycforc	Onychogomphus forcipatus
Cordbolt	Cordulegaster boltoni
Taennebu	Taeniopteryx nebulosa
Leuctra	Leuctra
Capnschi	Capnopsis schilleri
Amphbore	Amphinemura borealis
Protonem	Protonemura
Nemoura	Nemoura
Diura	Diura
Isoperla	Isoperla
Siphburm	Siphonoperla burmeisteri
Corixida	Corixidae
Apheaest	Aphelocheirus aestivalis
Sialis	Sialis
Rhyanubi	Rhyacophila nubila
Rhyafasc	Rhyacophila fasciata
Agapochr	Agapetus ochripes
Hydropti	Hydroptila
Ithytric	Ithytrichia
Oxyethir	Oxyethira
Chimmarg	Chimarra marginata

Lypephae	Lype phaeopa
Lyperedu	Lype reducta
Psycpusi	Psychomyia pusilla
Neurbima	Neureclipsis bimaculata
Pleccons	Plectrocnemia conspersa
Polyflav	Polycentropus flavomaculatus
Polyirro	Polycentropus irroratus
Cyrntrim	Cyrnus trimaculatus
Hydrpell	Hydropsyche pellucidula
Hydrsaxo	Hydropsyche saxonica
Hydrsilt	Hydropsyche siltalai
Hydrangu	Hydropsyche angustipennis
Hydrcont	Hydropsyche contubernalis
Ceranewa	Ceratopsyche newae
Cerasilf	Ceratopsyche silfvenii
Cheulepi	Cheumatopsyche lepida
Arctlado	Arctopsyche ladogensis
Agryobso	Agrypnia obsoleta
Phrybipu	Phryganea bipunctata
Semblis	Semblis
Bracsubn	Brachycentrus subnubilus
Micrgeli	Micrasema gelidum
Micrseti	Micrasema setiferum
Lepihirt	Lepidostoma hirtum
Limnephi	Limnephilidae
Goerpilo	Goera pilosa
Silopall	Silo pallipes
Seripers	Sericostoma personatum
Noticili	Notidobia ciliaris
Molaangu	Molanna angustata
Molatinc	Molannodes tinctus
Ceraclea	Ceraclea
Athripso	Athripsodes
Mystacid	Mystacides
Ylodes	Ylodes
Oecetis	Oecetis
Pyralida	Pyralidae
Ceratopo	Ceratopogonidae
Dixa	Dixa
Empidida	Empididae
Dicranot	Dicranota
Eloeophi	Eloeophila
Muscidae	Muscidae
Psychodi	Psychodidae
Simuliid	Simuliidae
Tabanida	Tabanidae
Tipulida	Tipulidae
Chironom	Chironomidae
Platmacu	Platambus maculatus
Stencana	Stenelmis canaliculata
Elmiaene	Elmis aenea
Oulitube	Oulimnius tuberculatus
Limnvolc	Limnius volckmari
Gyrinus	Gyrinus
Orecvill	Orectochilus villosus
Hydraena	Hydraena
Scirtida	Scirtidae

Appendix V. The codes of the macroinvertebrate taxa used in Appendix IX and the explanations.

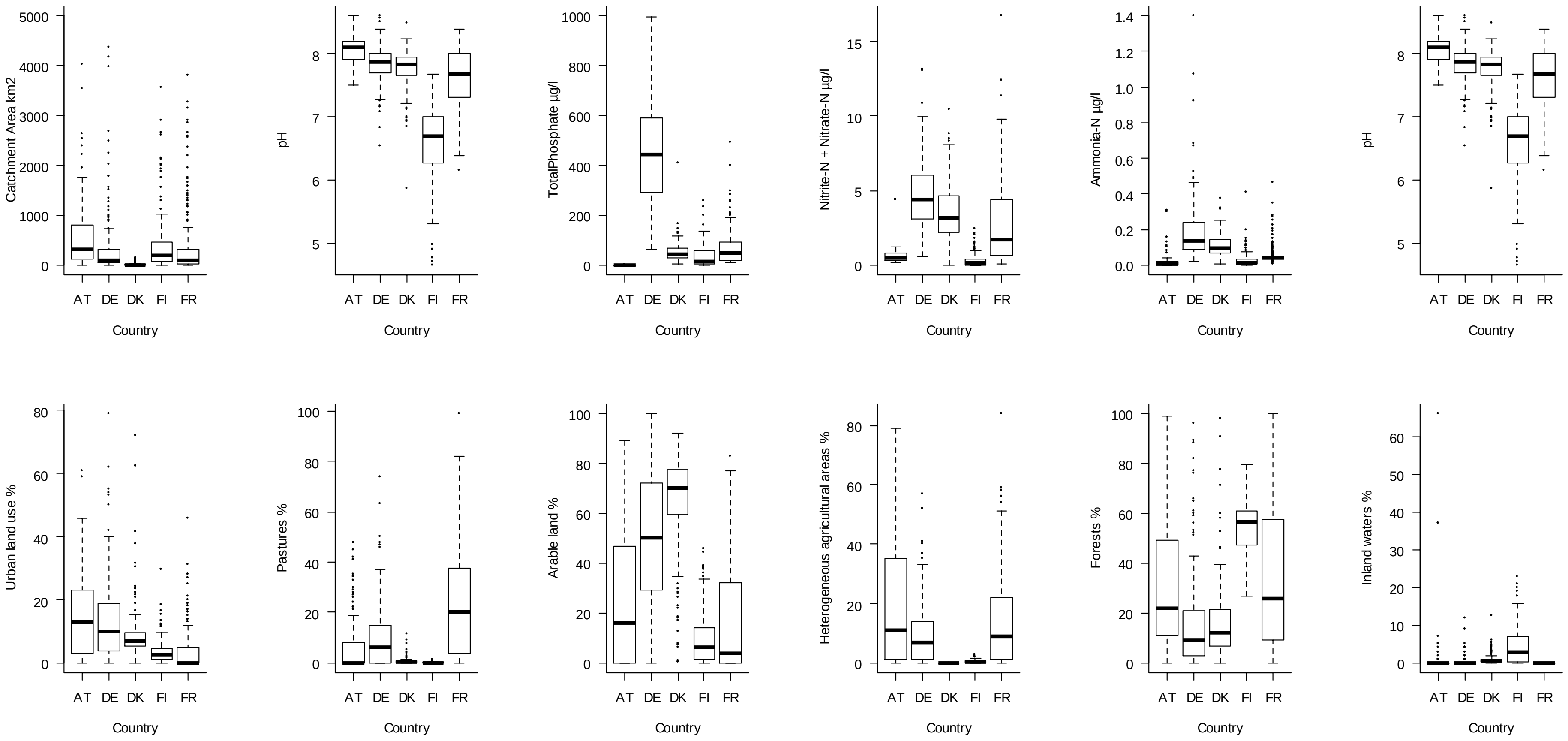
Code	Name
Acentr	Acentrella sp.
Acrolo	Acroloxus sp.
Adicel	Adicella sp.
AeshniG	Aeshnidae Gen. sp.
Agabus	Agabus sp.
Agapet	Agapetus sp.
Agriot	Agriotypus sp.
Agrypn	Agrypnia sp.
Amelet	Ameletus sp.
Amphin	Amphinemura sp.
Anacae	Anacaena sp.
Ancylu	Ancylus sp.
Anisus	Anisus sp.
AnthomG	Anthomyiidae Gen. sp.
Apatan	Apatania sp.
Aphelo	Aphelocheirus sp.
Arctop	Arctopsyche sp.
Asellu	Asellus sp.
AtheriG	Athericidae Gen. sp.
Athrip	Athripsodes sp.
Baetisalp	Baetis alpinus
Baetisbuc	Baetis buceratus
Baetislut	Baetis lutheri
Baetismel	Baetis melanonyx
Baetisnig	Baetis niger Gr.
Baetisrho	Baetis rhodani
Baetisvard	Baetis vardarensis
Baetisver	Baetis vernus Gr.
Bathyo	Bathyomphalus sp.
Beraeo	Beraeodes sp.
Bithyn	Bithynia sp.
BlephaG	Blephariceridae Gen. sp.
Brachyce	Brachycercus sp.
Brachycen	Brachycentrus sp.
Brachypt	Brachyptera sp.
Brychi	Brychius sp.
Bythin	Bythinella sp.
Bythio	Bythiospeum sp.
Caenisbes	Caenis beskidensis/pseudorivulorum
Caenishor	Caenis horaria
Caenisluc	Caenis luctuosa macrura
Caenisriv	Caenis rivulorum
Calopt	Calopteryx sp.
Capnia	Capnia sp.
Capnop	Capnopsis sp.
Centro	Centroptilum sp.
Ceracl	Ceraclea sp.
CeratoG	Ceratopogonidae Gen. sp.
Cheuma	Cheumatopsyche sp.
Chimar	Chimarra sp.
ChironG	Chironomidae Gen. sp.
Chloro	Chloroperla sp.
Chryso	Chrysopilus sp.
Cloeonins	Cloeon.inscriptum
CoenagG	Coenagrionidae Gen. sp.
Corbic	Corbicula sp.
Cordul	Cordulegaster sp.

CorixidG	Corixidae Gen. sp.
Crango	Crangonyx sp.
CurculG	Curculionidae Gen. sp.
Cyrnus	Cyrnus sp.
Dicrano	Dicranota sp.
Dictyo	Dictyogenus sp.
Dina	Dina sp.
Dinocr	Dinocras sp.
Diura	Diura sp.
DixidaG	Dixidae Gen. sp.
Dreiss	Dreissena sp.
Dryops	Dryops sp.
Dupoph	Dupophilus sp.
Ecdyon	Ecdyonurus sp.
Echino	Echinogammarus sp.
Ecnomu	Ecnomus sp.
Electr	Electrogena sp.
Elmis	Elmis sp.
Elodes	Elodes sp.
EmpidiG	Empididae Gen. sp.
Epeoru	Epeorus sp.
Ephemeaur	Ephemerella aurivillii
Ephemedan	Ephemera danica
Ephemeign	Ephemerella ignita
Ephememuc	Ephemerella mucronata
Ephemenot	Ephemerella notata
Ephemevul	Ephemera vulgata
Erpobd	Erpobdella sp.
Esolus	Esolus sp.
Ferris	Ferrissia sp.
Galba	Galba sp.
Gammar	Gammarus sp.
Gerris	Gerris sp.
Glossi	Glossiphonia sp.
Glosso	Glossosoma sp.
Goera	Goera sp.
GomphiG	Gomphidae Gen. sp.
Gyraul	Gyraulus sp.
Gyrinu	Gyrinus sp.
Habrol	Habroleptoides sp.
Habrop	Habrophlebia sp.
Haemop	Haemopis sp.
Halipl	Haliplus sp.
Helobd	Helobdella sp.
Heloph	Helophorus sp.
Heptagdal	Heptagenia dalecarlica
Heptagfla	Heptagenia flava
Heptagsul	Heptagenia sulphurea
Holoce	Holocentropus sp.
HydracG	Hydrachnidia Gen. sp.
Hydrae	Hydraena sp.
Hydrob	Hydrobius sp.
Hydrocyp	Hydrocyphon sp.
Hydrome	Hydrometra sp.
Hydropo	Hydroporus sp.
HydroporiG	Hydroporinae Gen. sp.
Hydrops	Hydropsyche sp.
Hydropt	Hydroptila sp.
Isoper	Isoperla sp.
Ithytr	Ithytrichia sp.
Kagerofus	Kageronia.fuscogrisea
Lepido	Lepidostoma sp.

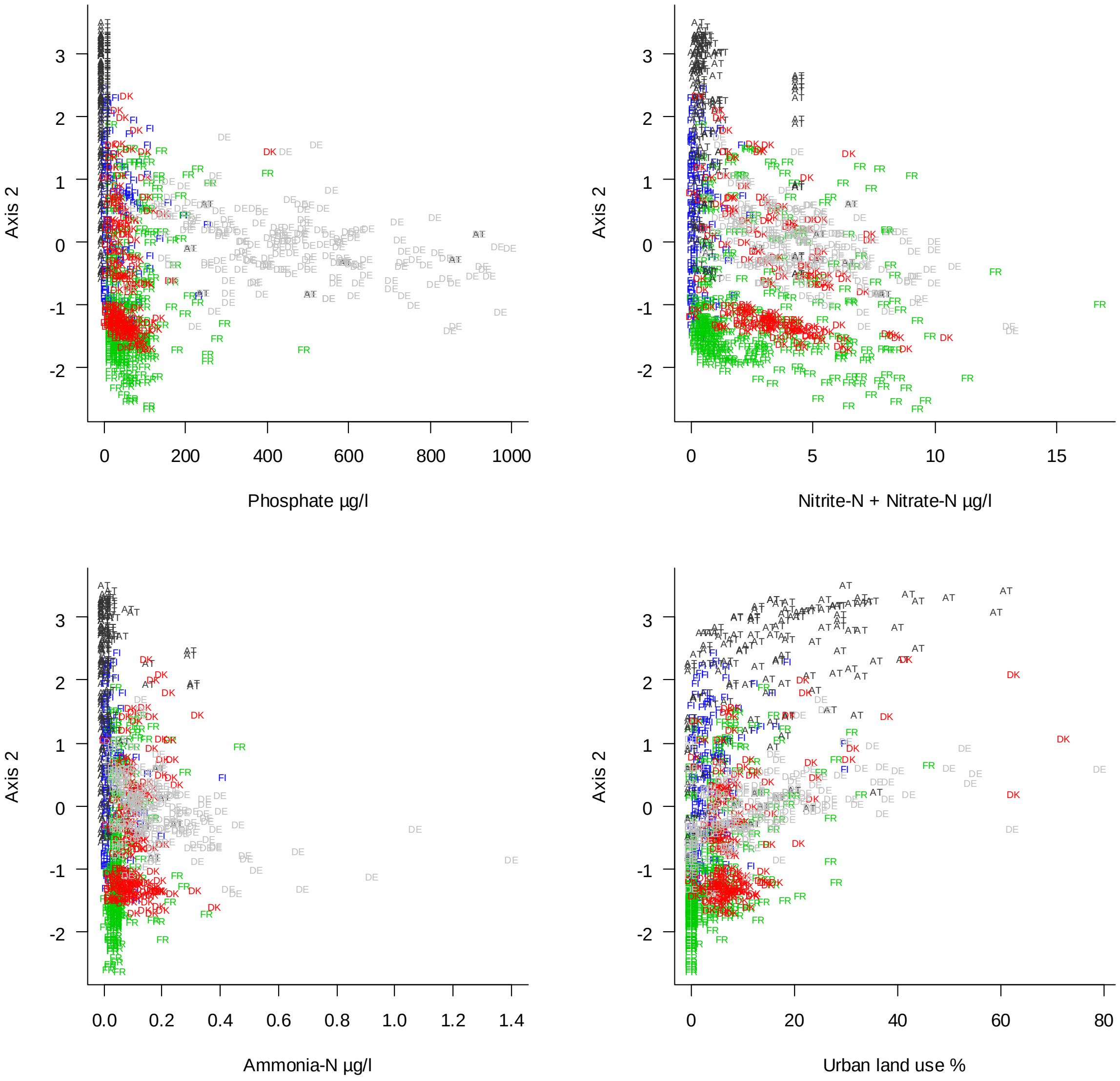
Leptop	Leptophlebia sp.	Oulimn	Oulimnius sp.
Leuctr	Leuctra sp.	Oxyeth	Oxyethira sp.
LimnepdG	Limnephilidae Gen. sp.	Parale	Paraleptophlebia sp.
Limniu	Limnius sp.	Perla	Perla sp.
LimoniG	Limoniidae Gen. sp.	Perlod	Perlodes sp.
Lithax	Lithax sp.	Philop	Philopotamus sp.
LymRad	Lymnaea/Radix sp.	Phryga	Phryganea sp.
Lype	Lype sp.	Physa	Physa sp.
Macron	Macronychus sp.	Physel	Physella sp.
Menetu	Menetus sp.	PiscicG	Piscicolidae Gen. sp.
Micras	Micrasema sp.	Pisidi	Pisidium sp.
Molanna	Molanna sp.	Planorbi	Planorbis sp.
Molanno	Molannodes sp.	Platam	Platambus sp.
MuscidG	Muscidae Gen. sp.	Platyc	Platycnemis sp.
Mystac	Mystacides sp.	Plectr	Plectrocnemia sp.
Nebrio	Nebrioporus sp.	Polycen	Polycentropus sp.
NematodG	Nematoda Gen. sp.	Pomati	Pomatinus sp.
NematomG	Nematomorpha Gen. sp.	PorifeG	Porifera Gen. sp.
Nemour	Nemoura sp.	Potama	Potamanthus sp.
Nemure	Nemurella sp.	Potamopyr	Potamopyrgus sp.
Neurec	Neureclipsis sp.	Proase	Proasellus sp.
Norman	Normandia sp.	Prosto	Prostoma sp.
Notido	Notidobia sp.	Proton	Protonemura sp.
Notone	Notonecta sp.	PsychodidG	Psychodidae Gen. sp.
Odonto	Odontocerum sp.	Psychom	Psychomyia sp.
Oeceti	Oecetis sp.	Ptycho	Ptychoptera sp.
OligocG	Oligochaeta Gen. sp.	Rhabdi	Rhabdiopteryx sp.
Oligon	Oligoneuriella sp.	Rhithr	Rhithrogena sp.
Orecto	Orectochilus sp.	Rhyaco	Rhyacophila sp.
Oreody	Oreodytes sp.	Riolus	Riolus sp.
Orthot	Orthotrichia sp.	Sembli	Semblis sp.

Serico	Sericostoma sp.
Serratign	Serratella.ignita
Setode	Setodes sp.
Sialis	Sialis sp.
Silo	Silo sp.
SimuliG	Simuliidae Gen. sp.
Siphon	Siphonoperla sp.
Sphaer	Sphaerium sp.
Stenel	Stenelmis sp.
StratiG	Stratiomyidae Gen. sp.
TabaniG	Tabanidae Gen. sp.
Taenio	Taeniopteryx sp.
Theodo	Theodoxus sp.
Therom	Theromyzon sp.
Thremm	Thremma sp.
Tinode	Tinodes sp.
TipuliG	Tipulidae Gen. sp.
Torley	Torleya sp.
TurbelG	Turbellaria Gen. sp.
Unio	Unio sp.
Valvat	Valvata sp.
Velia	Velia sp.
Wormal	Wormaldia sp.
Ylodes	Ylodes sp.

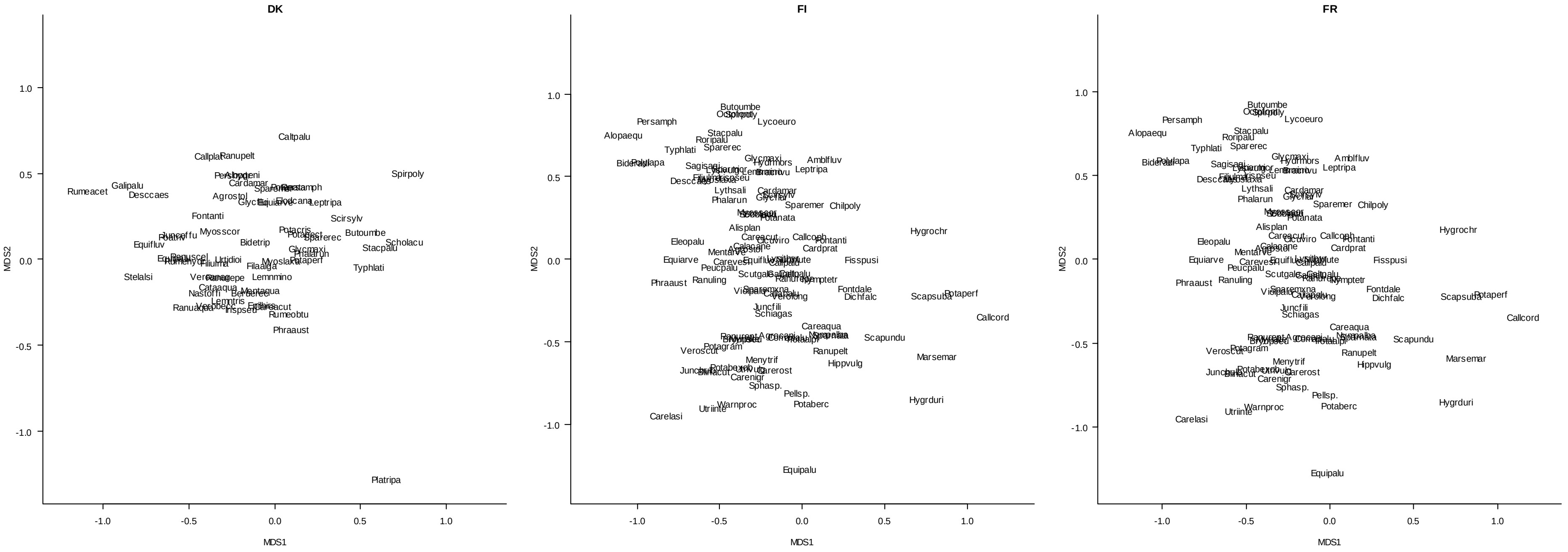
Appendix VI. The variation of the environmental variables in within Austria (AT), Denmark (DK), Finland (FI) and France (FI) in Dataset II.



Appendix VII. The second component of the Hill and Smith PCA analysis plotted with water quality and urban land use variables within Austria (AT), Denmark (DK), Finland (FI) and France (FI) in Dataset II (N = 947). Hydromorphological pressure increases with increasing PCA axis 2-values.



Appendix VIII. The NMS-ordination of the macrophyte taxa from Denmark, Finland and France (Dataset II). See Appendix III for the explanations of the taxa codes.



Appendix IX. The NMS-ordination of the macroinvertebrate taxa from Austria, Denmark, Finland and France (Dataset III). See Appendix V for the explanations of the taxa codes.