Assessing hydromorphological degradation of sand-bottom lowland rivers in Central Europe using benthic macroinvertebrates

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Contents

1 Introduction

1.1 The European Water Framework Directive (EU WFD)

In October 2000 the European Water Framework Directive (WFD) passed the European Parliament; it was published in December 2000. The WFD set a milestone in future water management and monitoring within the EU. The ecological quality of streams, lakes, transitional, and coastal waters has to achieve a 'good ecological quality' by the end of 2015. For the first time several 'Biological Quality Elements' (BQE; fish, benthic macroinvertebrates, aquatic macrophytes, and benthic algae and phytoplankton) have been designated instead of abiotic factors to be predominantly used for assessment. Abiotic parameters, such as physical-chemical and hydromorphological variables are to be considered in addition, but are designated to only support the bio-indicator-based assessment, not to replace it anymore. Moreover, the WFD has set several general conditions to be fulfilled by future biotic assessment systems, of which those relevant for stream and river assessment are referred to in the following.

First of all, future assessment has to be stream type-specific, since the composition of the in-stream biota is strongly controlled by natural constant variables setting the large-scale environmental framework of a site. Stream typologies have already come into focus of hydro-biologists and water managers in the last decades and represent the basis for stream assessment systems all over the world (e. g., Clarke, 1993; Omernik, 1995; Verdonschot, 1995; Wimmer et al., 2000). They can be organized either 'top down' by using geomorphological characteristics of river landscapes and the individual streams (Omernik, 1995; Sommerhäuser, 1998; Wimmer & Chovanec, 2000; Schmedtje et al., 2001; Briem, 2003; Pottgiesser & Sommerhäuser, 2004), 'bottom up' based on aquatic communities, or by a synthesis of both (Verdonschot, 1995; Hawkins & Norris, 2000; Moog et al., 2001). In general, a stream typology classifies streams or stream reaches into entities with a limited variability of both community composition and abiotic factors. The WFD defines several 'top-down' type descriptors to be used for the establishment of stream typologies by the member states. System A (EU commission, 2000, Annex II) comprises the obligatory descriptors ecoregion, catchment geology, altitude, and catchment size and represents a minimum demand. The more detailed System B includes the obligatory and additional alternative variables, for example, the mean channel width, depth, and slope, the distance from the source, the substrate composition, and the valley shape. While an official European typology is still missing, Pottgiesser & Sommerhäuser (2004) have recently published the official German stream typology that defines 24 stream types by using System B descriptors. The typology has been checked 'bottom-up' based on benthic invertebrates recently, too by Lorenz et al. (2004), who in general supported the major types, but who were not able to distinguish all 24 stream types. The German typology was used for the delineation of stream types in this thesis.

In order to fulfil the second general condition future assessment has to be based on stream type-specific reference conditions. Therefore, reference communities or community properties have to be defined for each stream type. The ecological status is obtained by Ecological

Quality Ratios (EQR) representing the deviation of a test site's community properties from those defined for the respective stream type's reference community. EQRs are classified into five classes (high = reference, good, moderate, poor, and bad) representing the final ecological status.

The third general condition for future assessment is stressor specifity. As several stressors may act simultaneously, but may differ in terms of the degree of impairment, it is important to identify the main stressor(s) and focus the assessment on the detection of its specific impact(s).

1.2 The need for advanced assessment systems

Until the early nineties water quality monitoring in most EU member states was mainly based on physical-chemical parameters. Since then, a wide variety of biology-based stream assessment methods often using benthic macroinvertebrates have been developed in many European countries. In general, macroinvertebrates are particularly well suited for assessment systems, since a comparatively large amount of data exists, the identification is relatively simple, and they occur in large numbers in all stream types (Hellawell, 1986; Rosenberg & Resh, 1993; Davis & Simon, 1995). The methods applied in European countries prior to 1996 have been summarized by Nixon et al. (1996). Most of these and further methods indicate the anthropogenic impact through organic pollution on the in-stream benthic community (e. g., Armitage et al., 1983; Alba-Tercedor & Sánchez-Ortega, 1988; DEV, 1992; CSN, 1998; Rolauffs et al., 2004). Several other systems aim at the detection of the impact of eutrophication, acidification, and salinization.

However, since 2000 physical habitat evaluation has been brought into focus in Europe (Raven et al., 2002). As a result, hydromorphological degradation has been identified to be an important stressor affecting the in-stream biota in many Central European stream types (Feld et al., 2002a; Raven et al., 2002; Lorenz et al., 2004b; Ofenböck et al., 2004). In most German streams and rivers hydromorphological degradation is supposed to be the main stressor at present (Feld, 2004; Lorenz et al., 2004). In this context, saprobic indices presumably have a restricted applicability in future assessment, since they aim at detecting a single stressor, i.e. organic pollution. Therefore, there is an urgent need for the development of new advanced tools to assess the ecological quality of streams and rivers throughout Europe (Feld, 2004; Hering et al., 2004a). A fundamental shift from a single index system towards a more 'holistic' approach referring to multiple indices and capable of assessing various impacts on the in-stream biota is necessary.

1.3 The application of multi-metric assessment systems

Scientists facing the task to make decisions about complex systems need multiple information about the system. If an economist is asked for the assessment of the economics' health, he falls back on widely used economic indices and indicators to track the information. For example, the consumer price index, the Dow Jones Index, or other stock market indicators provide the information needed, all of which are integrating multiple economic factors

(Karr & Chu, 1999). Thus, referring to ecosystems, which are presumably as complex or even more complex than the national economy, the use of multiple metrics, each of which reflecting an aspect of the system's biological conditions may also provide a suitable overall measure for ecosystem health. The more metrics are combined, the more complexity is presumably accounted for by the metric set, whereas a certain correlation of the metrics is acceptable if they refer to different properties of the community (Karr & Chu, 1999). And *vice versa* the complexity can be divided into single metrics (= measures) and makes multi-metric systems transparent for water managers. Moreover, the use of multi-metric systems also allows of the incorporation of different spatial and temporal scales in the assessment as it focuses on the community rather than on single taxa, the latter presumably more dependent on spatial and temporal scales.

One of the first multi-metric indices was presented by Karr (1981) to assess the biological integrity of fish communities by a selection of twelve metrics. Later work also included macroinvertebrates (Plafkin et al., 1989; Barbour et al., 1996, Fore et al., 1996; Resh et al., 2000), which were considered in the US Rapid Bioassessment Protocol for river assessment (Plafkin et al., 1989; Barbour et al., 1999).

Compared to the more than 20-year-old tradition of multi-metric indices in US American river assessment, its application in Europe is fairly new and was strongly influenced and promoted by the WFD. As future river assessment faces the indication of the impact of multiple stressors (organic pollution, toxic substances, hydrological alteration, morphological degradation, sediment entry, acidification, etc.) throughout Europe, the development of evenly multiple systems is needed. Multi-metric indices are supposed to cope with this new situation. This was recently proved by Lorenz et al. (2004), who presented multi-metric indices for the assessment of hydromorphological degradation in five German stream types. Vlek et al. (2004) developed a multi-metric index for two Dutch stream types incorporating ten metrics and targeting the assessment of the overall ecological quality. Böhmer et al. (2004) presented a multi-metric index based on twelve single metrics and suited to assess the impact of multiple stressors on the benthic invertebrate community throughout Germany and Ofenböck et al. (2004) developed several multi-metric indices for Austria, capable of assessing the impact of multiple stressors and incorporating five to nine single metrics.

1.4 The role of spatial scales

The faunal composition of stream macroinvertebrates is controlled by environmental variables acting at different spatial scales (Frissell et al., 1986; Corkum, 1992; Clarke & Ainsworth, 1993; Allan & Johnson, 1997; Poff, 1997; Townsend et al., 1997; Fitzpatrick et al., 2001; Griffith et al., 2001; Brosse et al., 2003; Snyder et al., 2003; Weigel et al., 2003). Some studies highlight catchment scale variables, such as river basin diameter, relief ratio (Townsend et al., 2003) or catchment geology (Allan & Johnson, 1997) as important determinants, while other studies focus on parameters at the reach scale (Richards et al., 1997; Roy et al., 2003). The role of 'natural' typological descriptors has already been focussed on before. The mean particle size, acting at the habitat scale, may also be a good predictor of stream insect diversity (Brosse et al., 2003). The interaction of environmental variables at

different spatial scales further complicates the relationship of stream biota and environmental variables. For example, catchment geology may control surficial geology and soil at the reach scale, which in turn may determine the particle size at a single site. This internal, predominantly one-directional structure is reflected in the 'hierarchical concept of landscape' (Frissell et al., 1986). However, the degree of hierarchical constraints that large-scale habitat descriptors may impose on small-scale habitat features does not seem to be well understood (Poff, 1997). This is also valid for the interaction of 'non-natural' human-induced environmental variables, such as catchment land use, river bank and bed modification, floodplain devastation, or substrate clogging. It is supposed that a hierarchical organisation applies also to the 'non-natural' descriptors, however studies addressing this topic are scarce.

1.5 Scope of this thesis

This thesis aims at the development of a system to assess the hydromorphological status of sand-bottom lowland rivers. Therefore, the role and interaction of different spatial scales was highlighted for both abiotic and biotic variables.

The delineation of the appropriate stream type in terms of ecoregion and catchment size and further descriptors is presented in Chapter 2 by the comparison of two different benthic invertebrate lowland datasets.

Chapter 3 presents the identification and measure of hydromorphological degradation by environmental variables. The analysis includes environmental features at three different spatial scales (catchment, reach, site) and identifies suited indicators at the scales.

Chapter 4 shows a multivariate approach to combine abiotic and biotic datasets in order to identify biotic indicators (taxa, metrics) to assess the impact of certain environmental variables; the analysis is separated into three spatial scales, too.

The relation of single benthic invertebrate taxa to certain environmental variables is presented in Chapter 5. Therefore, simuliids (Blackflies, Diptera) and environmental variables are analysed with regression analysis. The analysis includes Central Mountain datasets in order to identify ecoregional relations, too.

The major findings of Chapter 2 to 5 set the conceptual framework for the development of the multi-metric index in Chapter 6. The development process is guided by that framework and the index fulfils the demands of the WFD, i. e. the index is stream type- and stressor-specific, and is based on the comparison with reference conditions.

As the central step within the development process was the linkage of environmental and community properties, the compilation of new datasets was favoured, for which a common sampling procedure and field protocol was applied in order to reduce the sample bias (Hering et al., 2003, 2004a).

The samples used in this thesis were compiled within four different scientific projects:

1. „Validation der Fließgewässertypologie Deutschlands, Ergänzung des Datenbestandes und Harmonisierung der Bewertungsansätze der verschiedenen Forschungsprojekte zum Makrozoobenthos zur Umsetzung der Europäischen Wasserrahmenrichtlinie (Modul Makrozoobenthos)“ (LAWA O3.02: Haase et al., 2004).
2. “The development and testing of an integrated assessment system for the ecological quality of streams and rivers throughout Europe using benthic macroinvertebrates” (AQEM: Hering et al., 2004a).
3. “The definition of macroinvertebrate-based reference conditions” (DEMARECO: Feld & Bis, 2003).
4. “Standardisation of River Classifications” (STAR: www.eu-star.at).

Samples obtained by the German LAWA (‘Länderarbeitsgemeinschaft Wasser’) project O3.02 included routine monitoring samples of numerous German stream types which were consistent with Pottgiesser & Sommerhäuser (2004). For the location of the sites see Lorenz et al. (2004).

The AQEM lowland dataset comprised a total of six designated stream types sampled in three countries; the stream types do not always correspond to official national stream typologies but are consistent with System A or System B of the WFD (EU commission, 2000).

Sweden, type S05, ‘Medium-sized Central Lowland streams in south Sweden’.

The Netherlands, type N13 ‘Small streams in the Western Lowlands’ and N14 ‘Small streams in the Central Lowlands’. Samples of both types have been also split up into two alternative types N01 ‘Small Dutch slow running streams’ and N02 ‘Small Dutch fast running streams’.

Germany, type D01 ‘Small sand-bottom streams in the Central Lowlands’, D02 ‘Small organic brooks in the Central Lowlands’, and D03 ‘Medium-sized sand-bottom rivers in the Central Lowlands’.

The coding within AQEM was not consistent with the recently published official German typology (Pottgiesser & Sommerhäuser, 2004); the respective official German type codes are: D01 = type 14, D02 = type 11, and D03 = type 15.

The samples obtained by the German-Polish project DEMARECO and selected samples from STAR were limited to the AQEM type D03 and mainly represent reference or near-natural conditions.

All samples coded D03 have been taken by myself.

2 Delineation of German and Central European lowland stream types

2.1 Scope

The benthic macroinvertebrate community of streams and rivers is, besides biotic interaction, strongly controlled by environmental variables at different temporal and spatial scales. This inevitably leads to an enormous environmental variability over time and space. As a consequence, the in-stream biota reflect the environmental conditions in more or less diverse communities depending on the overall habitat quality. In general, the more diverse the environment is, the more diverse the community responses. The linkage of in-stream benthic macroinvertebrates and environmental variables on different spatial scales offers the basis for the development of assessment systems capable of assessing the hydromorphological status of rivers.

However, the question arises whether one can – in general – put together data of various different streams over wide geographical areas to assess their hydromorphological status with the same system. In other words: Do they all have the same reference conditions? Apparently, the answer must be 'no', since it is, for example, not likely that alpine streams and large lowland rivers have the same geo-hydromorphological reference conditions. This fact necessitates the development of river classification systems before any assessment starts in order to take the possible existence of various types of rivers and thus evenly various reference conditions into consideration. This 'stream typology' already exists for Germany, and, moreover, the EU WFD defines several abiotic stream type descriptors (e. g., ecoregion, altitude, catchment area) either to be used for or covered by national systems of the EU member states.

To come back to the linkage of environmental variables and benthic macroinvertebrates, it is crucial to know (i) whether the macroinvertebrate community reflects the abiotically defined stream types and (ii) which of the abiotic descriptors are 'natural' (e. g., ecoregion, altitude) or 'non-natural' (e. g., substrate diversity, flow conditions, land use), respectively. It is only the latter category, of which a measure of human-induced hydromorphological degradation can be derived. The role of 'non-natural' environmental variables is presented in Chapter 3 and Chapter 4.

This Chapter highlights the role of 'natural' type descriptors by comparison of the abiotically defined German stream typology with benthic macroinvertebrate samples. The detailed analysis is limited to Central Lowland samples. Moreover, the analysis is repeated for five Central and Western Lowland stream types of the AQEM project and an additional lowland dataset from Poland.

2.2 Summary

Based on 390 benthic macroinvertebrate samples from near-natural streams in Germany, Non-metric Multidimensional Scaling (NMS) identified ecoregion as the prevailing discriminator. The analysis was restricted to six dominating taxonomical groups, Mollusca, Ephemeroptera, Odonata, Plecoptera, Trichoptera, and Coleoptera (MEOPTC), due to heterogeneous determination levels available for other taxonomical groups. At genus level, the Central Lowlands, Central/Western Mountains, and Alps were clearly separated. Detailed analysis of the lowland species data revealed stream (catchment) size as an important predictor within the ecoregion. Classification of the fauna data identified five groups of samples representing "bottom up" defined stream types. The analysis was repeated with a Central and Western Lowland dataset of Swedish, Dutch, German, and Polish sites at species level. Similar to the first dataset, stream size discriminated the in-stream macroinvertebrate community best. Ecoregion/sub-ecoregion were good descriptors, too, whereas substrate showed weak descriptive power in both datasets.

2.3 Introduction

As a framework for national 'top-down' typologies the 25 European ecoregions defined by Illies (1978) are frequently used, particularly for applied purposes, like the implementation of the EU Water Framework Directive. In some cases they have been divided into sub-ecoregions (Moog et al., 2004) or 'river landscape units' (Briem, 2003). Germany shares four Illies' ecoregions: Alps (ecoregion 4), Central and Western Mountains (ecoregions 8 and 9), and Central Lowlands (ecoregion 14). However, for many water management purposes, such as assessment and restoration, more differentiated categories of streams are required and useful. In a first attempt to establish a more detailed stream typology for Germany, Schmedtje et al. (2001) suggested a 'top-down' system, which was based on ecoregion, altitude, geology, stream size, and further abiotic variables, such as, for example, the dominant substrate. The system was improved and refined by Pottgiesser & Sommerhäuser (2004), who finally ended up with the delineation of 24 stream types supposed to provide specific environmental conditions to determine the benthic invertebrate, macrophyte, and phytobenthos communities. Although the relevance of such a typology has to be proved 'bottom-up' with community data, a national survey in Germany has not yet been established. However, data availability and quality recently increased, due to several national projects with particular focus on benthic invertebrates (Böhmer et al., 2004; Hering et al., 2004a; Lorenz et al., 2004b; 2004c). A dataset derived from different monitoring programmes and various scientific researches was, although quite heterogeneous, used to test the typology from 'bottom-up'. A focus was laid on the lowland data in order to compare the results with another rather homogeneous lowland dataset originated from the EU project AQEM (Hering et al., 2004a). This chapter addresses the answers to the following questions: 1) Do the WFD System A descriptors ecoregion, catchment size, and geology (substrate) discriminate the benthic invertebrate community in Germany? 2) Does the order of descriptors change at ecoregion scale in the lowlands? 3) Is the 'top-down' typology in the lowlands reflected by benthic macroinvertebrates?

2.4 Material and methods

2.4.1 Data source and preparation

390 macroinvertebrate reference samples from routine monitoring programmes of several German Federal States and various scientific studies were used. The sites were located in four ecoregions according to Illies (1978): Alps (ecoregion 4), Western and Central Mountains (ecoregions 8 and 9), and Central Lowlands (ecoregion 14), and the entire dataset was used on genus level to explore ecoregional patterns of the macroinvertebrate community. See Lorenz et al. (2004c) for a detailed location and description of the sites. For comparison with the AQEM lowland dataset, further analysis of the German monitoring dataset was restricted to lowland samples, too. Non-reference samples were excluded applying the following filter criteria: catchment area < 9 km² (no very small sites); German Saprobic Index > 2.3 (no polluted sites); Gewässerstrukturgüteindex (LAWA 2000) > 3 (no significant structural impairment); catchment land use urban > 9 %/crop land > 19 % (Wang et al., 1997; Roy, 2003); number of taxa < 10 (ecoregions 4, 9) or < 8 (ecoregion 14) (minimum richness). Reference conditions were usually lacking in areas with a high proportion of residential and/or agricultural land use, in particular at medium-sized and large rivers. The macroinvertebrate samples were predominantly taken with handnets (500 µm) using a time-limited method and covering all occurring habitats within the sampling reach (DEV, 1992). For each sampling site selected abiotic parameters were compiled from data providers, maps and GIS and used to explain the observed benthic community patterns. Because the taxa lists were heterogeneous concerning the identification level, sampling season, and sampling and sorting methods, they were harmonised prior to analysis by (i) transformation into qualitative data (presence/absence level) and (ii) selection of only the six most frequently sampled taxonomical groups: Mollusca, Ephemeroptera, Odonata, Plecoptera, Trichoptera, and Coleoptera (MEOPTC). These taxonomically well-known groups were identified mainly to species level. Other taxonomical entities (e. g., Oligochaeta, Chironomidae) revealed considerable heterogeneity in terms of identification levels and have, therefore, been omitted, just as highly frequent taxonomical groups (e. g., Gammaridae), which were not likely to add much explanatory power on the presence/absence level.

The second lowland dataset included 94 macroinvertebrate samples of 53 sites in Central and Western Lowland streams and rivers in Sweden (S05), The Netherlands (N13, N14), and Germany (D03) and additional samples of Western and Central Poland (PL) (Figure 2.1). Samples were taken using a modified Multi-Habitat Sampling (Barbour et al., 1999): Within a sampling reach, 20 ‘sample units’ (each 25 x 25 cm) were taken according to its proportion in the whole reach using a shovel or Surber sampler. Only dominant habitats were sampled with each sampling unit representing 5 % of the stream bottom. The 20 sample units were pooled and preserved with ethanol (96 %) in the field. In addition, 130 environmental variables were recorded in a field protocol, including ‘natural’ and ‘non-natural’ descriptors at different spatial scales. Only ecoregion, altitude, catchment size, and dominating substrate were used as environmental descriptors here, whereas in particular the role of ‘non-natural’ variables is presented and discussed in Chapter 3.

In the lab, samples were rinsed with water over a 1000 µm sieve in order to separate the coarse (> 1000 µm) and fine fraction. The coarse fraction was completely sorted out and specimens were preserved in ethanol (70 %) until identification; the fine fraction was omitted. Identification aimed at species level except for Chironomidae (genus level) and Oligochaeta and most Diptera (family level). For details on sampling and sorting see Hering et al. (2003, 2004a).

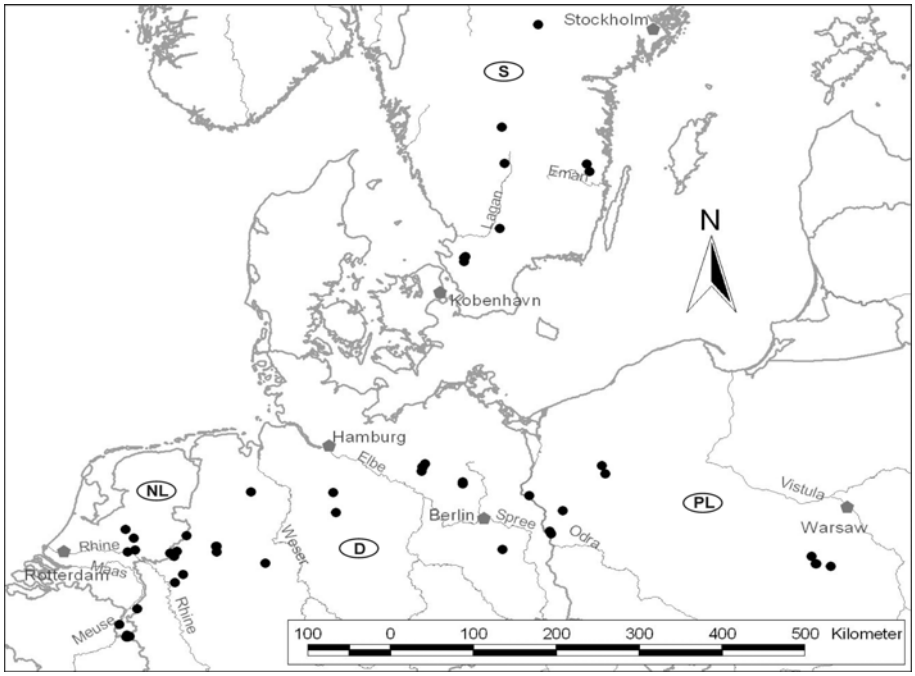

Figure 2.1: Location of 53 sampling sites (●) in Sweden (S), The Netherlands (NL), Germany (D), and Poland (PL).

A taxonomical adjustment was applied to both macroinvertebrate datasets, which is a prerequisite to the comparison of taxalists of different origin and taxonomical resolution (Schmidt-Kloiber & Nijboer, 2004; Feld & Rolauffs, 2005). In general, the artificial heterogeneity (noise) caused by, for example, different researcher-based identification levels (skills) or sampling habits (preference) can be as far harmonized as possible by taxonomical adjustment. The adjustment aimed at keeping the lowest common and safely identified taxonomical level of all samples. This was either species level if species identification was commonly achieved or genus level if > 20 % of the samples only obtained the genus level. After adjustment, taxa with a frequency < 3 were omitted.

The further analysis was exclusively run with the German monitoring dataset that was split up into two seasonal subsets (spring: February–June, 123 samples; summer: May–September, 109 samples; overlap to allow for sufficient sample size for both seasons) in order to take seasonal community patterns into consideration, which could not have been ana-

lytically excluded due to the heterogeneous dataset. In contrast, the homogeneous AQEM dataset was *a priori* analysed with regard to seasonal differences using ordination. Since season did not seem to have a descriptive effect on the data, all AQEM samples were analysed together.

2.4.2 Statistical analysis

Non-metric Multidimensional Scaling (NMS) was used to detect and visualize differences in the benthic invertebrate communities (for background and advantages of NMS see Dufrêne & Legendre, 1997). NMS is an iterative ordination technique, based on a dissimilarity matrix, here the Bray-Curtis distance (Bray & Curtis, 1957; Beals, 1984). NMS tries to as best as possible fit the multi-dimensional dissimilarity matrix into a low (two–three-) dimensional ordination plot. The correspondence between the matrix and the plot is explained by the term 'stress', a measure of goodness of fit. It is zero in case of perfect concordance, whereas values < 0.05 (= 5 %) represent excellent, < 0.10 good, and < 0.20 interpretable results. A higher stress is critical and values > 0.30 represent an arbitrary fit of sample points in the ordination plot (Clarke & Warwick, 2001). The ordination axes are not ordered according to the proportion of the total variance they explain. Therefore, those two axes are shown in the ordination plots that account for the most explained variance.

A cluster analysis was run with macroinvertebrate data to derive biotic cluster groups to be used as an additional layer for the ordination plots and, hence, to aid the interpretation. NMS and Cluster Analysis were run with PC-Ord 4.3 (McCune & Mefford, 1999). NMS was set to autopilot method (Bray-Curtis similarity) with best thoroughness; repeated runs ensured the stability of results. Cluster analysis was run with Bray-Curtis similarity, too and 'flexible *beta*' linkage (*beta* = -0.25). The linkage method is similar to Ward's method (Dufrêne & Legendre, 1997). Sample groups were tested for significant differences with XLSTAT 5.2 (AddinSoft, 2002). Analysis of Similarity (ANOSIM) was used to test the classification of environmental descriptors (season, ecoregion, catchment area, and dominant substrate) and the cluster groups for strength (R) and significance (p). R values usually range between 0 and 1, whereas large values indicate a strong classification, i. e. a high similarity of members within a group compared to their similarity to members of other groups (Clarke & Warwick, 2001). ANOSIM was run with PRIMER E (Clarke & Gorley, 2001).

2.5 Results

2.5.1 German monitoring data

A clear separation of Central Lowland, Central/Western Mountain, and Alpine samples was evident for the whole dataset on genus level and was discussed in detail by Lorenz et al. (2004c). The authors' conclusions referring to the discrimination of ecoregions is supported by ANOSIM with an overall (global) similarity of R = 0.409 ($p < 0.001$) for the four ecoregions in Germany (Table 2.1). However, there was no discrimination evident for Western and Central mountain streams (R = -0.070; p = 0.872). Further analysis focussed on the lowland dataset with 123 spring (143 taxa) and 109 summer samples (136 taxa).

Both seasonal subsets revealed a gradient along the catchment area of the sites (Figure 2.2A and B). While small streams and medium-sized rivers clustered apart in both seasons, the discrimination of large rivers (> 1,000 km²) was solely evident from the summer dataset, a 10 km² borderline (very small/small streams) was not well displayed in both subsets. The proportion of the total variance in the faunal data explained by the two strongest ordination axes was nearly 50 % in spring and 60 % in summer. In Figure 2.3A and B, the same pair of ordination plots was overlaid by the substrate classes 'organic' (bog mosses, POM), 'sand', and 'gravel'. Organic samples formed a rather well separated group in spring; however, this was not obvious from the summer subset due to only two sites belonging to the category. In contrast, sand- and gravel-dominated streams were not separated in any season. As a third overlay, five cluster groups derived from a cluster analysis with the identical fauna data were used: For the spring data (Figure 2.4A), cluster group 1 predominantly comprised medium-sized and large sand-bottom rivers. Large river samples (> 1,000 km²) were clustered at the upper left of the group, so that a moderate size gradient can be observed. Cluster group 2 represented the majority of gravel streams with some overlap to the adjacent sand-bottom (group 1) and organic samples of cluster group 3. The latter consisted of samples from streams with organic substrates and itself showed some overlap at the transition to gravel-bed streams of group 2 and sand-bottom streams of group 4. From cluster groups 1 to 3, a size gradient was obvious ranging from large (at the upper left) to small catchments at the lower right (Figure 2.4A). The gradient corresponded well with ordination axis 2 and accounted for 25 % of the fauna's total variance. The well defined cluster group 4 comprised a mixture of small gravel-bed and sand-bottom streams (< 100 km²). Cluster group 5 contained samples from small gravel-dominated streams, streams in the floodplain of larger rivers, and two samples from medium-sized sand-bottom rivers on the outer left hand side of the cluster. Hence, the cluster groups 4 and 5 reveal a size gradient along the second axis of Figure 2.4A, too.

The clusters derived from the summer data confirmed the results: Again a size gradient was obvious ranging from small sand/gravel streams (group 4) at the upper left to large sand-bottom rivers (group 1) at the lower right hand side of the NMS ordination plot (Figure 2.4B). However, in contrast to the spring data, cluster group 1 comprised a mixture of sand- and gravel-bottom rivers in summer. Organic streams (group 3) were underrepresented in summer (N = 2), so that no further interpretation was possible. Analogous to the spring data, Group 5 comprised another group of mixed sand- and gravel bottom streams that was clearly separated from group 4 in summer.

ANOSIM identified the cluster groups as best discriminating the benthic macroinvertebrate communities of the German lowlands in both seasons (Table 2.1). Catchment area (size) was the second most important descriptor for the summer data, whereas in spring it was less important. The dominant substrate category as allocated by expert judgement *a posteriori* showed only a weak discrimination strength.

Table 2.1: Classification strength of predictor variables used as overlays for NMS ordination plots of the German monitoring dataset. Classification strength is expressed as global ANOSIM R with values > 0.500 indicated in bold. Descriptors and groups are explained in the text. p = level of significance.

	Germany		Central Lowlands	
Descriptor	R	p	R	p
Ecoregion	0.409	< 0.001		
Spring				
Catchment area (WFD classes)			0.330	< 0.001
Dominant substrate			0.177	< 0.001
'Bottom-up' types (cluster groups)[a]			**0.600**	< 0.001
Summer				
Catchment area (WFD classes)			**0.514**	< 0.001
Dominant substrate			0.200	0.001
'Bottom-up' types (cluster groups)[a]			**0.578**	< 0.001

[a]Cluster groups according to Figure 2.4.

2.5.2 AQEM lowland data

The analysis of the total dataset with 94 samples and 225 taxa of three seasons did not reveal a seasonal pattern (Figure 2.5A) and, hence, all analysis was run without further separation of seasonal subsets. This was underlined by low R values for ANOSIM (Table 2.2). As samples from Western Lowlands (N13) were available only for the Dutch data, NMS did not discriminate ecoregions if the whole data were considered (Figure 2.5B and C; Table 2.2); samples of both Dutch stream types formed a well mixed cluster at the lower left hand side of the ordination plot. But nevertheless, if the Swedish samples were excluded, ecoregion discriminated the remaining samples pretty good (ANOSIM: R = 0.454; $p < 0.001$) and even slightly better if ANOSIM was restricted to the Dutch samples only (R = 0.504; $p < 0.001$).

Within ecoregion 14 two other groups were separated by ordination: the Swedish stream type S05 clustered well apart from the remainder (Figure 2.5C). If the data were compared for sub-ecoregional differences within the Central Lowlands, the latitude of the Swedish sites ranged from 56 to 59° N, whereas Dutch, German, and Polish sites ranged from 51 to 53° N. Hence, latitude showed the highest correlation with NMS axis 2 ($r = 0.775$, $p < 0.001$). Besides, Swedish stream reaches were characterized by a higher mean proportion of cobbles on the stream bottom (19.5 % for S05 *vs.* 1.5 % for the rest) and a higher mean stream width (10.7 *vs.* 5.7 m, respectively), whereas their mean number of organic substrates (1.6 *vs.* 3.7, respectively) and mean number of logs on the stream bed (1.4 *vs.* 29.2, respectively) were much lower than for the other sample reaches. All differences were significant (Mann-Whitney-U-Test, $p \leq 0.01$).

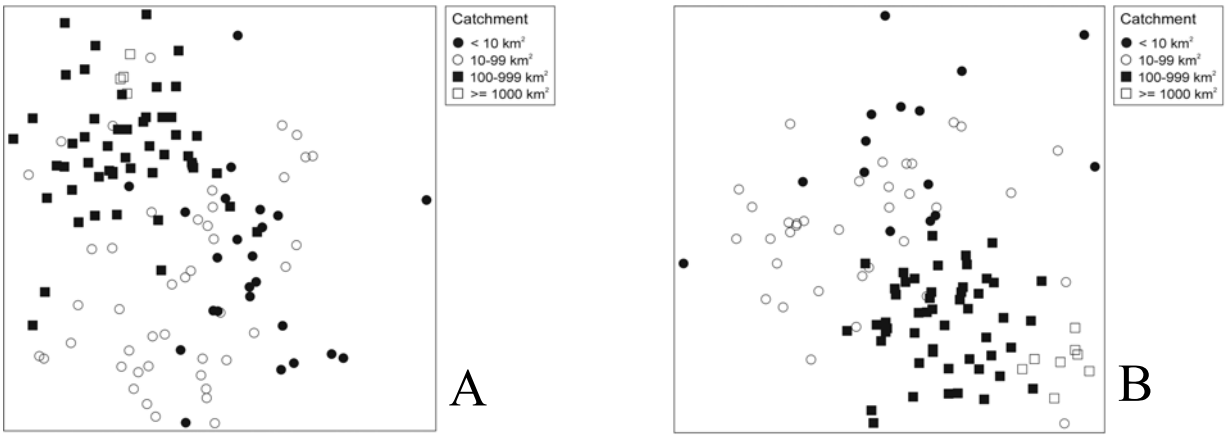

Figure 2.2: NMS ordination of lowland samples at species level. Catchment area was used as overlay after ordination. A) Spring data with 123 samples and 143 taxa. Final stress: 0.198. Variance explained: Axis 1 = 25.3 %; axis 2 = 24.1 %. B) Summer data with 109 samples and 136 taxa. Final stress: 0.207. Variance explained: Axis 1 = 18.3 %; axis 2 = 42.0 %.

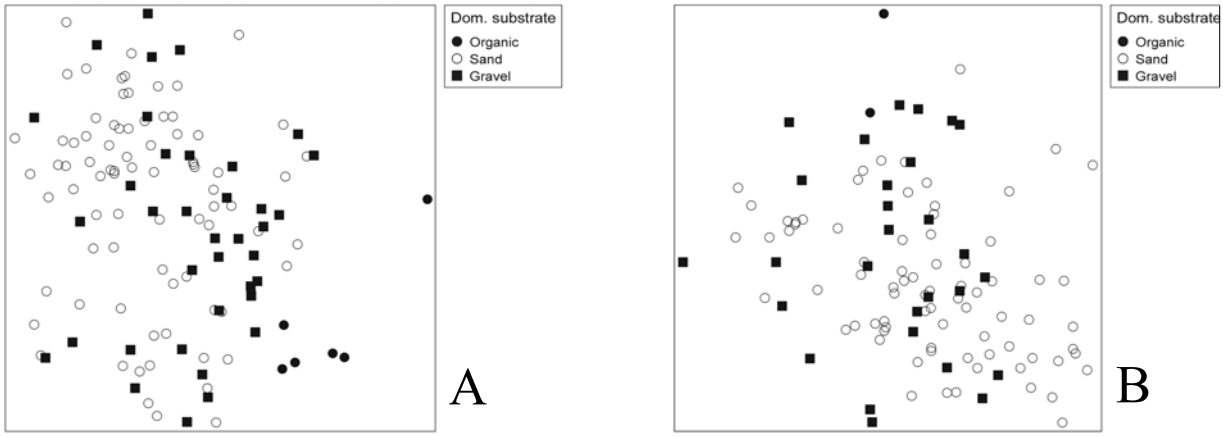

Figure 2.3: NMS ordination of lowland samples at species level. Dominant substrate was used as overlay after ordination. A) Spring data, B) Summer data. Number of samples and taxa, final stress and explained variance as in Figure 2.2.

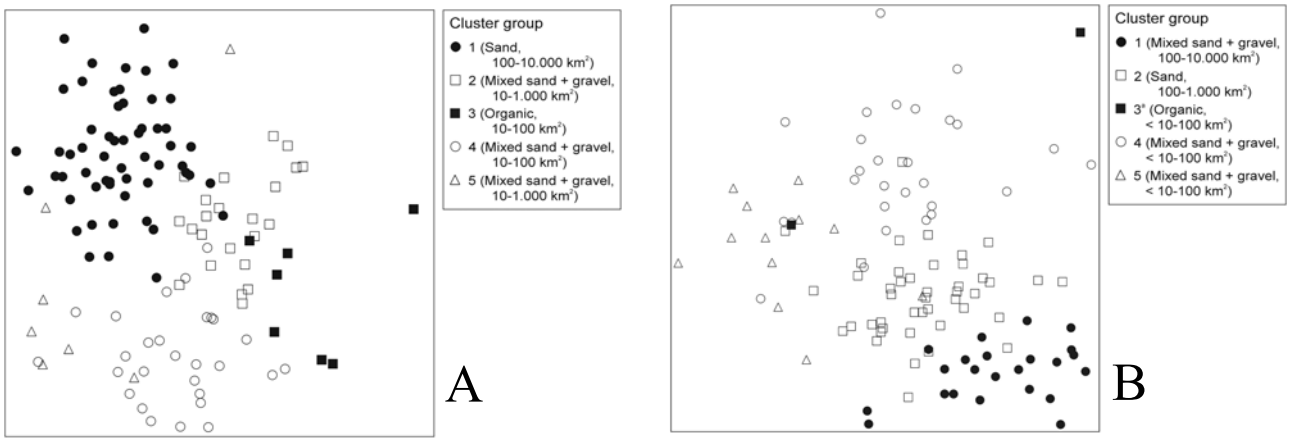

Figure 2.4: NMS ordination of lowland samples at species level. Cluster groups derived from the same fauna dataset were used as overlay after ordination for A) spring and B) summer data. Number of samples and taxa, final stress and explained variance as in Figure 2.2.

[a]Cluster group 3 is not representative due to N = 2 for organic streams in summer.

A size gradient was obvious for all samples except the Swedish from the lower left to the upper right hand side of Figure 2.5D. The existence of a size gradient was explicitly supported by ANOSIM (Table 2.2) for the whole data as well as for the non-Swedish data. If the classification of a cluster analysis was used as an overlay (Figure 2.7), two size classes were discriminated along the main size gradient: The first (group 1) almost exclusively comprised very small and small catchments (range: 0.5–90.5 km²; mean ± SD: 13.3 ± 24.6 km²), the second (group 2) mainly medium-sized catchments (3.4–760 km²; 240.1 ± 196.3 km²). The fauna-based size classification promotes a class boundary at about 40–50 km² of catchment area and was strongly supported by the analysis of similarity (Table 2.2, 'Global excl. S05'). The gradient was also visible for the Swedish samples, however, it was confounded by two samples originating from large rivers and in general was too small for analysis of similarity and further interpretation.

Substrate did not seem to discriminate well between samples as far as the non-Swedish samples are regarded (Figure 2.6; Table 2.2). But although substrate seems to separate the Swedish streams from the rest, it was not clear whether this was due to substrate differences or rather sub-ecoregional properties of type S05 as already mentioned before.

Table 2.2: Classification strength of predictor variables used as overlays for the NMS ordination plots of the AQEM lowland dataset. Classification strength is expressed as global ANOSIM R with values > 0.500 indicated in bold. Descriptors and groups are explained in the text.

	Global		Global excl. S05	
Descriptor	R	p	R	p
Season	0.083	0.060	0.115	< 0.001
Ecoregion	0.277	0.010	0.454	< 0.001
Catchment area (WFD classes)	0.492	< 0.001	**0.608**	< 0.001
Dominant substrate	0.354	< 0.001	0.177	0.020
'Bottom-up' types (cluster groups)[a]	**0.736**	< 0.001	**0.634**	< 0.001

[a]Cluster groups according to Figure 2.7.

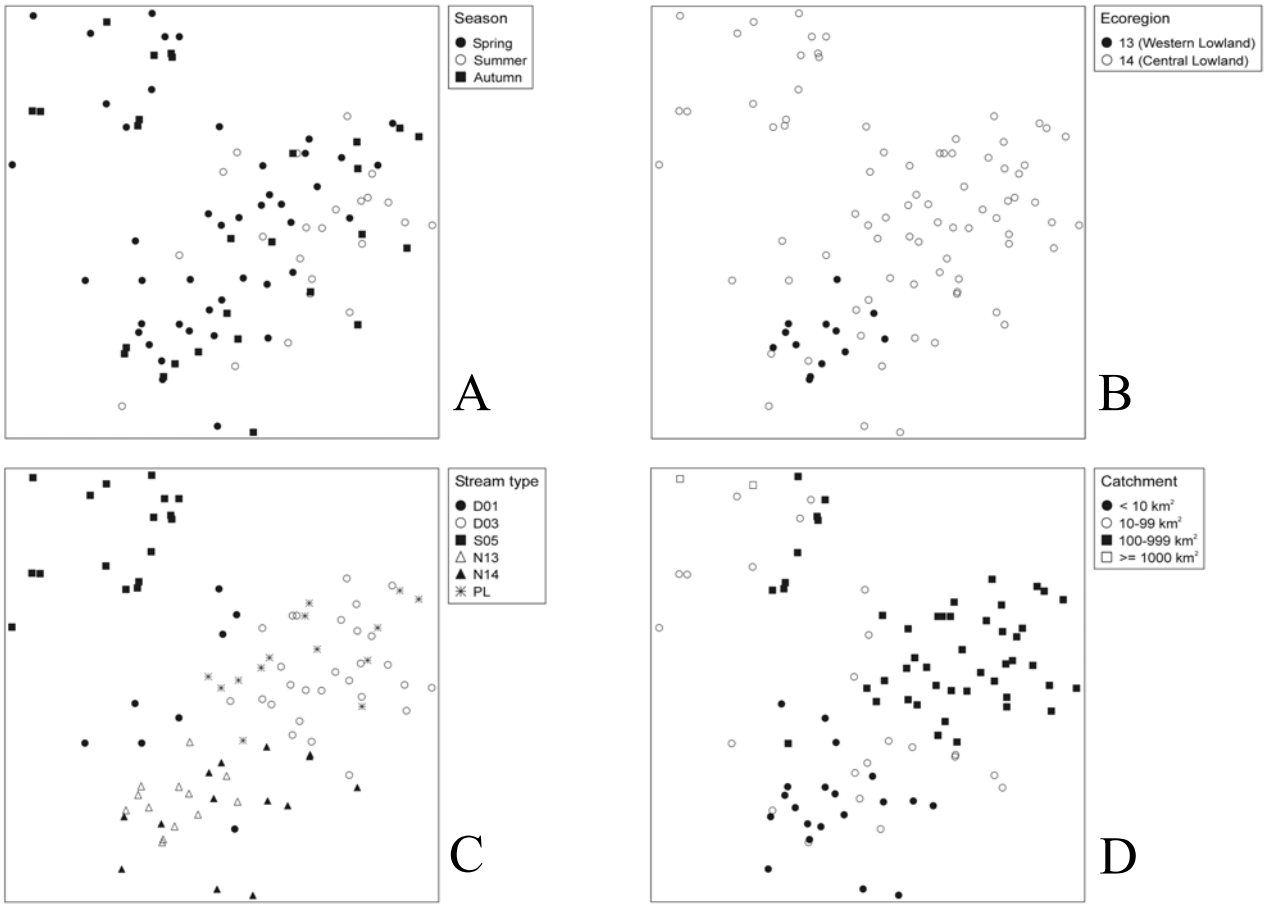

Figure 2.5: NMS ordination of 94 AQEM lowland samples with 225 taxa at species level. Season (A), ecoregion (B), stream type (C), and catchment area (D) were used as overlays after ordination. Final stress: 0.170. Variance explained: Axis 1 = 29.1 %; axis 2 = 32.9 %.

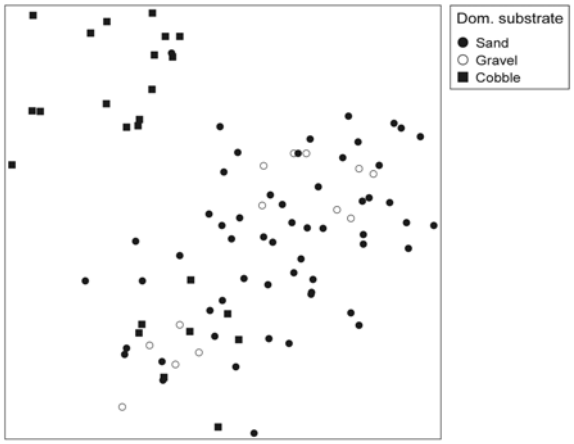

Figure 2.6: NMS ordination of AQEM lowland samples with the dominant (prevailing) substrate category used as overlay after ordination. Number of samples and taxa, final stress, and explained variance as in Figure 2.5.

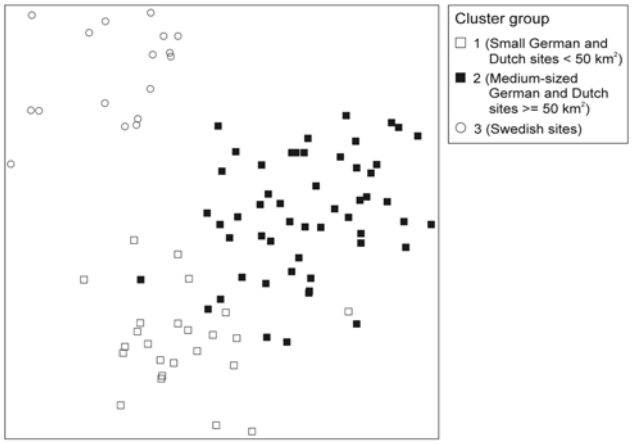

Figure 2.7: NMS ordination of AQEM lowland samples with cluster groups derived from the same fauna dataset as overlay after ordination. Number of samples and taxa, final stress, and explained variance as in Figure 2.5.

2.6 Discussion

2.6.1 German monitoring data

Like in other countries (e. g., Feminella, 2000; Gerritsen et al., 2000; Sandin & Johnson, 2000) ecoregion was the prevailing descriptor affecting the benthic invertebrate community of German streams and rivers. Accordingly, the faunal composition was mainly controlled by ecoregional properties, such as altitude, slope, hydrology, temperature, and geology (substrate). The community-based separation of the Central Lowlands, Central/Western Mountains, and Alps was recently discussed in detail by Lorenz et al. (2004c). The analysis of similarity (ANOSIM) applied here to the same data supported the discrimination strength of ecoregion very well. The importance of ecoregional properties was also reported by Moog et al. (2004) for Austria, where ecoregions and sub-ecoregions accounted for the most benthic macroinvertebrate variability. Verdonschot & Nijboer (2004) found ecoregion and other System A descriptors of the EU WFD discriminating the benthic invertebrate communities throughout Europe. However, Waite et al. (2000) found ecoregion not being satisfying if it was exclusively considered for the Mid-Atlantic Highlands, USA. Other factors had to be regarded in addition to explain the in-stream fauna composition. Also Rundle et al. (1993) and Brewin et al. (1995) found catchment area to be an important factor in separating stream assemblages, besides ecoregions. The size gradient shown for the Central Lowland data of both seasons in Figure 2.2 confirms the role of stream size (catchment area) in structuring the in-stream fauna. Whether directly by using the WFD size classification or by cluster groups derived from the fauna data, stream size was the strongest descriptor for community patterns in the current study. This is consistent with basic ecological concepts for the longitudinal zonation of streams (e. g., Illies, 1961; Vannote et al., 1980). Hence, stream size is a prevailing ‘typologically relevant’ parameter and should be generally considered for and included in stream typologies. Size may be expressed either as catchment area, stream width, discharge, or distance to source.

Several samples did not fit perfectly into the size gradient as shown in Figure 2.2, which is supposed to be due to the WFD size classification applied here: < 10, 10–99, 100–999, and ≥ 1,000 km² (EU commission, 2000, Annex 5, System A). The classification rather represents an artificial system not reflecting the natural features that discriminate streams, medium-sized, and large rivers very well. In particular this applies to small streams with catchments << 100 km², for which another boundary was reflected by the macroinvertebrate community (Figure 2.2). As discussed later for the AQEM dataset, alternative class boundaries might be derived from Figure 2.5D and 2.7.

Another ‘potentially relevant’ parameter for lowland streams was summarized by the term ‘substrate’. This term combines the substrate type (organic *vs*. mineral) and its grain size (here: sand, gravel, cobble), the latter applicable only for mineral substrates. Previous papers dealing with stream typology in the German lowlands usually apply the substrate classes organic material, sand, and gravel (LUA NRW, 1999; 2001; Sommerhäuser & Schuhmacher, 2003) and, thus, they were also used in the current study. NMS separated organic and mineral streams, but surprisingly a further separation of mineral streams was not

revealed. Both spring and summer data showed a considerable overlap of gravel- and sand-bottom samples (Figure 2.6) that lead to very weak R values of ANOSIM. Thus, the dominant substrate does not seem to be a strong descriptor of the macroinvertebrate community. This is not consistent with the well reported role of substrate (grain size) for structuring macroinvertebrate communities (Brusven & Prather, 1974; Minshall, 1984; Allan, 1995). This contradiction to the literature may be due to three reasons: 1) The relation of the fauna and its environment is scale-dependent. While substrate becomes more important at the site-scale (habitat), it may be a comparatively weak descriptor on the large ecoregion-scale. 2) Community patterns were analysed by the overall Bray-Curtis similarity and, moreover, were based on presence/absence level in case of the German monitoring data. Therefore, changes in the total community composition were analysed rather than specific differences of certain taxa. 3) The weak discrimination between sand- and gravel-bottom streams does not necessarily reject the existence of a substrate gradient. Rather, it is connected with the data quality in this case, in particular with the subjective judgement on the prevailing grain size at the sampled stream reaches. This was often assigned *a posteriori* by researchers based on their expert judgement and may not reflect the substrate conditions truly present while macroinvertebrate samples have been taken. On the other hand, the discrimination was neither obvious for the AQEM lowland sites that were classified according to directly recorded substrate estimations in parallel to sampling in the field (see below).

In order to overcome the lack of an objective substrate classification here, benthic macroinvertebrates were used twice: i) for the NMS and ii) for a cluster analysis that provided the overlay (cluster groups) for the NMS ordination plot (Figure 2.4). The first cluster group represents typical medium-sized and large, mainly sand-bottom rivers and forms a well separated unit regarding the benthic invertebrate community in both seasons. *Sphaerium* sp. (Bivalvia), *Baetis vernus*, *Heptagenia flava* (Ephemeroptera), *Gomphus vulgatissimus* (Odonata), *Isoptena serricornis* (Plecoptera), and *Hydropsyche pellucidula* (Trichoptera) were abundant in samples of this cluster. The faunal composition also corresponds well with cluster group 2 of the summer samples. Another well defined unit in spring was formed by small organic streams (group 3). They represent a stream type with, amongst other aspects, a specific benthic macroinvertebrate community. The stream type and its specific community was documented by Sommerhäuser & Schuhmacher (2003). *Cordulegaster boltoni* (Odonata) or *Glyphotaelius pellucidus* (Trichoptera) are typical species and they were frequently found in samples of the cluster group. Samples of organic streams were almost lacking for the summer data and, therefore, are not further discussed here. The samples of cluster group 2 originated from sand- or sand- and gravel-dominated streams and medium-sized rivers. However, those samples reported from sand-bottom streams and rivers apparently had a 'gravel-fauna' in spring and may, thus, be examples of streams with a comparatively small proportion of gravel, but nevertheless determining its faunal composition. This hypothesis is confirmed by the benthic community: *Capnia bifrons* (Plecoptera), *Agapetus fuscipes*, or *Odontocerum albicorne* (Trichoptera) frequently occurred in spring samples of this cluster group, all of which are gravel- (cobble-) preferring species (Schmedtje & Colling 1996). This corresponds to the fauna found in samples of cluster group 4 of the summer samples: *Electrogena* sp. (Ephemeroptera), *Nemurella pictetii* (Plecoptera),

Agapetus fuscipes, *Beraeodes minuta*, *Plectrocnemia conspersa*, and *Sericostoma* sp. (Trichoptera) frequently occurred. The question arises of what "dominant substrate" means. Regarding the data presented here, gravel and cobbles may likely control the benthic community with a proportion far below 50 % within a stream reach.

The second mixed cluster group 4 comprises a set of sand- and gravel-dominated small streams in spring and was characterized by the presence of *Lymnaea stagnalis* (Gastropoda), *Limnephilus flavicornis*, or *Phryganea grandis* (Trichoptera). According to Moog (1995) and (Schmedtje & Colling 1996), these species rather prefer fine sediments, lentic flow conditions, and the presence of macrophytes. In summer, cluster group 5 was characterized by the same species and, hence, may represent comparable conditions. As a consequence, the five cluster groups identified for the summer data corresponded pretty good to the first four groups of the spring data. The community summarized by spring samples of cluster group 5 comprised several Potamon-specific species: *Acroloxus lacustris*, *Anisus vortex*, *Theodoxus fluviatilis,* or *Viviparus viviparus* (Gastropoda). Therefore, it presumably represented samples from medium-sized to large rivers, characterized by a domination of sand and mud on the river bottom and occasionally covered with large stands of macrophytes. Corresponding samples were presumably lacking in summer due to the lack of those Potamon-specific species in the summer dataset.

2.6.2 AQEM lowland data

The main descriptor to explain the benthic community structure of the AQEM lowland dataset was stream size, here expressed as catchment area, and visible for both groups of samples, S05 and the remainder (Figure 2.5D). Moreover, ANOSIM proved the size gradient identified by cluster groups for the non-Swedish data (Figure 2.7) to be the strongest descriptor for the AQEM lowland dataset. Yet, similar to the German monitoring data, the size classification given by the EU WFD was only partly reflected by the benthic macroinvertebrates presumably due to its rather artificial character. In particular the 10 km² boundary (very small *vs.* small streams) was not shown at all by the fauna. In contrast, the separation of small streams and medium-sized rivers was partly reflected, even if there was considerable overlap at the 100 km² boundary (Figure 2.5D). The results presented support the assumption that a 'faunistically relevant' size class boundary might be expected between 40 and 50 km² catchment area. If applied to the AQEM lowland data, smaller catchments were characterized by the almost exclusive and frequent presence of *Gammarus fossarum* (Crustacea), *Elodes* sp. (Coleoptera), *Sericostoma* sp., *Silo* sp. (Trichoptera), *Eloeophila* sp., and *Polypedilum* sp. (Diptera), whereas larger catchments promoted the presence of taxa preferring the lower Rhithral or upper Potamal zone of rivers: *Pisidium amnicum* (Bivalvia), *Gammarus roeselii* (Crustacea), *Aphelocheirus aestivalis* (Heteroptera), *Orectochilus villosus* (Coleoptera), *Hydropsyche pellucidula* (Trichoptera), and *Prodiamesa olivacea* (Diptera). However, more data are needed to statistically verify the alternative size classification.

Ecoregion seemed to be the second important descriptor, although ecoregion 13 (Western Lowlands) was represented by only few Dutch samples and the discrimination was not obvious from the NMS (Figure 2.5B). But regarding the ANOSIM results, the mean similarity of both ecoregions was comparable to that calculated for the whole German monitoring dataset. This discrepancy shown for NMS and ANOSIM results leads to the assumption that the number of samples of ecoregion 13 and the geographical area covered by the samples may be too small for sound interpretation of the results. Environmental variables other than ecoregion may be responsible for the discrimination of the respective Dutch samples, for example, slope or discharge. The same applies to the discrimination of Swedish samples: Although located in the Central Lowlands, they clustered apart from all the others and formed a distinct group (type S05, Figure 2.5C). The differences shown for the latitudes of both groups seem to support this finding. However, the environmental properties of type S05 rather prove its hydromorphological peculiarity as shown, for example, by the significantly higher mean proportion of cobbles and mean stream width and the significantly lower mean number of logs and organic substrates on the river bottom. The characteristic of type S05 was underlined by several taxa that frequently and almost exclusively occurred in the respective samples: *Leptophlebia marginata*, *L. vespertina*, *Nigrobaetis niger* (Ephemeroptera), *Isoperla difformis*, *Leuctra hippopus* (Plecoptera), and *Agapetus ochripes* (Trichoptera).

Similar to ecoregion, the prevailing substrate classification as applied here was shown to be a weak descriptor of the benthic macroinvertebrate composition, too (Figure 2.6, Table 2.2). Samples except those of type S05 were scattered all over the ordination plot independent of whether they were originating from sand-, gravel-, or cobble-bottom streams and rivers. In particular the sand-bottom samples were evenly spread along the size gradient. The Swedish samples form a distinct cluster characterized by the domination of cobbles. This implies that substrate rather than sub-ecoregion was the main descriptor for the separation of S05 as already discussed above. Up to 50 % of macrolithal, i. e. head-sized cobbles (20–40 cm), were recorded for the Swedish sites (mean: 19.5 %). Since comparable proportions of macrolithal have not been recorded for either natural Dutch, German, or Polish Central Lowland rivers, its dominance in Swedish rivers presumably means a sub-ecoregional peculiarity, too. Except for stream reaches that run through end moraines, the proportions of cobbles recorded for the Swedish sites are not likely to naturally occur over a comparatively large geographical area in one of the other countries that share the same ecoregion. Natural rivers in the Central Lowlands of The Netherlands, Germany, or Poland usually run through glacial deposits of the last two glacial periods (ground moraines) and, therefore, most often are dominated by either gravel or sand. Cobbles and even boulders may occur, but are usually restricted to short sections running through end moraines.

2.7 Conclusions

Despite the constraints discussed in context with data heterogeneity and representativeness, the results presented here support the assumption that ecoregion and stream size are major descriptors structuring the in-stream benthic invertebrate community. Within an ecoregion, as shown for the Central Lowlands, stream size becomes an important descriptor. However, the boundaries reflected by the macroinvertebrates seem not to fit well into the size classification given by the EU WFD. At least for the separation of small streams and medium-sized rivers the present data support an alternative boundary at approximately 40–50 km² catchment area. In contrast, the applied substrate classification 'sand-gravel-cobbles' proved weak to explain community patterns of benthic macroinvertebrates at larger (ecoregion) scales. It may become a stronger descriptor at smaller spatial scales (e. g., site, reach), however this was not subject of the present study. The classification of the German lowland fauna data implies four to five entities that may represent distinct stream types:

1) small organic streams;
2) small lotic sand-/gravel-bottom streams;
3) small lentic sand-/gravel-bottom streams;
4) medium-sized and large lotic sand-/gravel-bottom rivers
5) medium-sized (and large) lentic sand-/gravel-bottom rivers

Taking the basic results into consideration, further analysis is focussed on streams and rivers in the Central European Lowlands (ecoregion 14 according to Illies, 1978). Besides faunistic aspects the hydromorphological differences between stream types and – within a certain type between sites of different 'quality' is focussed. First of all the 'hydromorphological degradation' is analysed addressing only physical habitat parameters at different spatial scales.

3 Identification and measure of hydromorphological degradation in Central European lowland streams

3.1 Scope

With the previous Chapter the role of different 'natural' typological descriptors in structuring in-stream benthic invertebrates was highlighted. In brief, ecoregion and catchment size were found to strongly influence the community. Hence, the analysis of the response of the community to hydromorphological impacts has to take those 'natural' typological peculiarities into consideration. Otherwise, the clear discrimination of natural stream type-specific and artificial (man-made) properties will be confounded. If the task is to assess the impact of hydromorphological degradation of the in-stream community, the questions arise, which additional hydromorphological variables may act as typological descriptors and, therefore, represent natural differences and which of the variables reflect hydromorphological degradation.

This Chapter aims at the definition of hydromorphological degradation and the identification of hydromorphological variables suited to define the degradation. Therefore, the site protocols of the AQEM lowland dataset of ecoregions 13 and 14 were analysed.

3.2 Summary

Stream type-specific and spatial scale-dependent multivariate analysis (Non-metric Multidimensional Scaling, NMS) of 106 hydromorphological variables derived from 275 samples at 147 sites and Indicator Value Analysis (IndVal) resulted in the identification of 'key factors' describing hydromorphological differences in Central European lowland streams. Sample sites represented six European stream types from Sweden (1 stream type), The Netherlands (2 stream types), and Germany (3 stream types). The four large-scale hydro(geo)morphological variables: catchment size, geology ('% moraines', '% alluvial deposits'), and natural land use ('% natural forest') explained inter-stream type differences best. On the smaller site scale, riparian vegetation described inter-stream type differences best.

At catchment scale, '% natural forest' and '% agricultural land use' illustrated inter-stream type hydromorphological degradation of all six stream types very well. Four site-related variables ('% wooded riparian vegetation', '% shading', 'average stream width', and '% macrolithal' (cobbles) accounted for hydromorphological degradation at the smaller reach-scale. An analysis of indicator variables restricted to German stream types resulted in four factors, namely '% xylal' (tree trunks, branches, roots), 'no. of debris dams > 0.3 m³', 'no of logs > 10 cm Ø', and '% fixed banks' as important descriptors of hydromorphological degradation.

Intra-stream type hydromorphological degradation is illustrated for medium-sized sand-bottom rivers in the German lowlands. For this stream type, a clear gradient of degradation was revealed and 25 variables were identified to entirely characterize reference conditions and degradation. The variables that described the degradation gradient best were combined

to the new German Structure Index (GSI), which can be used to continuously measure hydromorphological degradation.

3.3 Introduction

Running water ecosystems are controlled mainly by geological, hydrological, morphological, and water chemistry attributes that form the physical habitat (Franquet et al., 1995; Hildrew, 1996; Richards et al., 1996). The physical habitat controls the in-stream biota at both temporal and spatial scales (Allan et al., 1997; Beisel et al., 1998a; 1998b; Davies et al., 2000; Sponseller et al., 2001). In particular, the scale-dependent relation between hydromorphology and the macroinvertebrate community in streams and rivers has been widely discussed (e. g., Rabeni, 2000; Sponseller et al., 2001; Statzner et al., 2001). Some authors emphasize the role of large-scale variables, such as catchment geology, while others state sub-catchment, such as land use and reach-scale habitat attributes, such as riparian buffer width, to mainly influence the community. Moreover, at a finer spatial scale the influence of single hydromorphological features, for example, large wood or riparian vegetation on benthic invertebrates is well-known and was broadly discussed (Dudley & Anderson, 1982; Benke et al., 1985; Richards et al., 1996; Hoffmann & Hering, 2000).

Several methods to measure habitat quality and habitat degradation exist (e. g., Agence de l'Eau Rhin-Meuse, 1996 for France; Barbour et al., 1999 for the USA; Raven et al., 1998, 2002 for the UK; LAWA, 2000 for Germany). But Raven et al. (2002) have also shown that the cited methods lead to different results due to the different definition of 'near-natural land use' in the French and German protocol. Moreover, the lack of stream type specifity, as applies to, for example, the German 'Strukturgütekartierung', requires a revision of existing methods to fulfil the demands of the WFD. Due to the complex relationship between hydromorphological attributes and the in-stream community it still remains controversial how to define habitat degradation and at which spatial scale(s). Hydromorphological assessment within the EU-funded research project AQEM generally followed the approach to compare test site characteristics with specific reference characteristics per stream type (Barbour et al., 1999; Raven et al., 2002). Therefore, stream type-specific hydromorphological reference conditions had to be defined prior to assessment. This step demands knowledge of the hydromorphological conditions occurring under undisturbed conditions (high status) as a basis for the definition of four hydromorphological degradation classes (good, moderate, poor, bad status) as demanded by the five-class classification of the WFD. Three major questions were addressed in the following: 1) What is hydromorphological degradation? 2) Which spatial scale is appropriate to describe the hydromorphological status? 3) Which groups of hydromorphological variables (e. g., land use, hydrograph, reach, riparian area) are suited and minimally necessary to measure hydromorphological degradation?

This Chapter presents stream type-specific results based on spatial scale-dependent statistical analysis of hydromorphological characteristics of six stream types in ecoregions 13 and 14 of Europe (according to Illies, 1978). The aim was to analyse spatial scale-dependent hydromorphological differences and to identify hydromorphological variables suited to describe reference conditions and different states of degradation within a single stream type.

3.4 Material and methods

3.4.1 Data collection

In total, 275 samples collected at 147 sites belonging to six different stream types and distributed over three different countries (Sweden, The Netherlands and Germany) were analysed (Table 3.1, Figure 3.1). German and Swedish sites were sampled twice in March/April/May 2000 and June/July 2000, with the exception of sites of stream type D03, which were sampled three times in June and September 2000, and March 2001. Dutch sites were sampled once or twice in April/May/June and/or August/September/October 2000. All sites belong to the Central European Lowlands (ecoregion 14), except Dutch sites south of River Rhine, which belong to the Western European Lowlands (ecoregion 13).

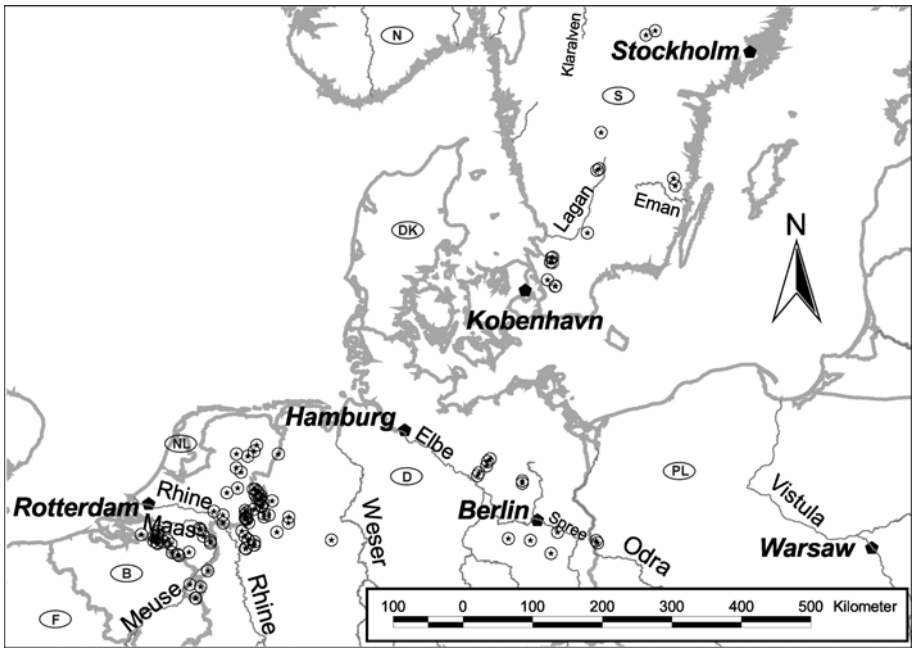

Figure 3.1: Location of the 147 sites in Sweden, Germany, and The Netherlands.

The hydromorphological status of each site was derived from a set of variables compiled using the AQEM site protocol. A detailed description and a downloadable site protocol is available at www.aqem.de (see also AQEM consortium, 2002; Hering et al., 2003). In total, 130 hydromorphological and geological variables were recorded at three different spatial scales:

(1) Catchment-related variables consider the whole catchment from the stream source to the sample site, for example, distance to source, stream order, catchment geology, and catchment land use. They were derived from topographical and geological maps (scale: 1 : 50,000 to 1 : 300,000). When available, land use was measured using ArcView GIS and data from Corine Landcover (e. g., Statistisches Bundesamt, 1997

for Germany). Since catchment variables are generally constant over a long period of time, they were recorded only once for each sample site.

(2) The longitudinal extent of reach-related (up-/downstream) variables depends on the size class of a stream type. For small streams (10–100 km² catchment area), a stretch of 5 km up- and downstream of the sample site was taken into consideration (= 10 km), whereas in case of medium-sized rivers (100–1000 km²) a stretch of 10 km up- and downstream was analysed (= 20 km). Percent (%) length of impoundments, lack of natural vegetation, or water abstraction represent typical up-/downstream variables, which were usually derived from topographical maps (scale: 1 : 50,000) and recorded once per sampling site.

(3) Site-related variables were recorded for each sampling occasion separately. They refer to a stretch of 250 m up- and downstream (= 500 m) of the sample site for small streams and 500 m up- and downstream (= 1,000 m) in case of medium-sized rivers. Habitat composition and physical-chemical variables are typical site related variables.

3.4.2 Stream characteristics

General stream characteristics are summarized in Table 3.1.

Sites of ‘medium-sized lowland rivers in south Sweden’ (type S05) are usually slow-flowing permanent streams without a distinct valley. The natural low-gradient stream course is usually meandering. Benthic diatoms represent dominating primary producers in lotic reaches, whereas deep and slowly flowing reaches are dominated by macrophytes and epiphytic algae as primary producers. The prevailing degradation factor is a mixture of organic and nutrient pollution (eutrophication), and locally acidification is very important. Degraded sites of this stream type are also hydromorphologically impaired (e. g., through straightening) and situated in agricultural areas (see also Dahl et al., 2004).

The Dutch streams belong to two stream types: ‘Small Dutch slow running streams’ (type N01) and ‘small Dutch fast running streams’ (type N02). The latter are characterized by higher gradients (mean slope of the thalweg), situated in U-shaped valleys with higher proportions of gravel on the stream bottom. ‘Small Dutch fast running streams’ show a permanent and relatively constant discharge pattern. Stream morphology is always altered by channel regulation and agricultural land use. Thus, high quality reference sites are almost completely lacking.

‘Small Dutch slow running streams’ (type N01) naturally have a plain floodplain with a meandering channel, and in-stream habitat comprises a higher proportion of sand and particulate organic material, when compared to hill streams. Due to extensive alteration of the stream morphology (straightening, scouring, and removal of floodplain vegetation) and eutrophication, this stream type is almost entirely affected by severe degradation (see also Vlek et al., 2004).

Pristine (reference) sites of ‘small sand-bottom streams in the German lowlands’ (type D01) are characterized by sand of fine to medium grain size and a meandering channel flowing in varying valley forms (trough valley, meander valley, plain floodplain). Organic substrates range from 10 to 50 % with a considerable amount of large wood (logs, debris dams).

‘Small organic type brooks in the German lowlands’ (type D02) are naturally characterized by a U-shaped valley and a braided channel. Organic microhabitats cover most of the stream bottom, for example phytal (floating stands of *Potamogeton polygonifolius* Pourr. and water mosses, such as *Sphagnum* spp. and *Scapania undulate* [L.]), xylal (wood, root mats) and CPOM (coarse particulate organic matter: fallen leaves, twigs). The brownish water is often acidic. Both small stream types have been nearly completely degraded by scouring, straightening, impoundments, stagnation, removal of large wood, and devastation of floodplain vegetation in the past.

References of ‘medium-sized sand bottom rivers in the German lowlands’ (type D03) are characterized by sand of fine to coarse grain size and a sinuate to meandering channel flowing in a meander valley or a plain floodplain. Organic substrates cover between 10 and 50 % of the bottom, of which large wood (logs, debris dams) causes a high substrate and current diversity. The wide floodplain is dominated by deciduous wooded vegetation and standing water bodies (side arms, backwaters) occur regularly except during summer when they dry out. Almost all streams of this stream type have been extensively degraded by scouring, straightening, impoundments, stagnation, removal of large wood, and devastation of floodplain vegetation due to agricultural land use. Small near-natural fragments occur in north-eastern Germany and Poland (Pauls et al., 2002).

Table 3.1: General characteristics of investigated stream types (stream type codes according to Hering et al., 2003).

Stream type	Code	River system(s)	Ecoregion (Illies, 1978)	Catchment size (km²)	Altitude (m a.s.l.)	pH	Conductivity (µS cm^{-1})	No. of reference sites	No. of reference samples	Total no. of sites	Total no. of samples
Small sand-bottom streams in the German lowlands	D01	River Rhine, Ijssel, Ems	14	9–151	33–136	6.7–8.3	295–1,750	1	2	12	23
Small organic type brooks in the German lowlands	D02	River Rhine	14	0.1–11.3	30–50	4.2–7.4	200–640	4	4	13	13
Medium-sized sand-bottom rivers in the German lowlands	D03	Ijssel, Ems, Elbe, Odra	14	120–760 (–6,400)[a]	25–60	7.2–8.5	330–815	5	15	18	54
Medium-sized rivers in South Swedish lowlands	S05	Norrström, Motala ström, Virån, Helge å, Kävlingeån, Saxån, Rönne å, Lagan	14	32–1,005	15–200	5.2–8.2	60–1,550	5	10	15	30
Small Dutch slow-running streams	N01	River Rhine, Meuse (Maas), Drentse A	13, 14	0.5–190	1–180	4.4–8.6	100–895	32	58	78	141
Small Dutch fast-running streams	N02	River Rhine, Meuse (Maas), Drentse A	13, 14	0.5–137	5–180	6.5–8.4	120–950	6	8	11	14
	Sum							53	97	147	275

[a]Single site at River Spree (Brandenburg, Germany).

3.4.3 Selection of sampling sites

Due to an extensive sampling programme, the number of samples taken for a single stream type was restricted. Therefore, sample sites were pre-selected according to a subjective estimation of their degradation status. The aim of the pre-selection was a set of sites that covered a degradation gradient from reference (high status) to heavily degraded sites (bad status). Degradation was related to the (main) stressor affecting a single stream type, which was organic/nutrient pollution (type S05), hydromorphological degradation (types D01, D02, and D03), or general degradation (types N01 and N02). The pre-selection was supported by information derived from maps, for example, channel form, stream size, stream order, or accessibility. Additional information on stream status and stream reaches was compiled using data from earlier studies, monitoring reports, and data on habitat quality, such as the German river habitat survey 'Strukturgütekartierung' (LAWA, 2000). The pre-selection was then evaluated during field trips yielding the final set of sample sites (Figure 3.1).

As a general frame, a set of sites for a single stream type comprised at least three sites each of a supposed high (reference conditions), good, and moderate quality, respectively. Poor and bad states were each represented by at least one site, so that at least eleven sites were sampled per stream type (see also Hering et al., 2004a). Definition of reference sites followed the basic statements of Hughes (1995) and Wiederholm & Johnson (1996) and aspects defined by Nijboer et al. (2004). When reference sites were not available due to degradation of an entire stream type the best available sites served as 'assessment references', which was the case for the Dutch stream types N01 and N02. The 'assessment references' represented a 'good ecological quality' instead of a 'high ecological quality' according to the WFD.

3.4.4 Evaluation of stream type assignment and hydromorphological degradation

Stream type definition and assignment followed System B of the WFD (for detailed description see Hering et al., 2004a). When available, stream type tables were used to support proper stream type assignment (e. g., LUA NRW, 2001 for German stream types). The analysis of typologically relevant hydromorphological variables was exclusively related to 97 samples of a supposed good or high quality, since any kind of degradation may affect or superimpose the results. Six samples were excluded from the analysis due to missing data.

In order to visualize the general structure of the environmental dataset, the whole set comprising 275 sampling occasions including 106 out of 130 recorded hydromorphological and geological variables was used. Twenty-four site protocol variables were excluded from the analysis due to the casewise deletion of missing data. For the analysis of inter-stream type hydromorphological degradation, a two-class classification was introduced, since a reduced classification was supposed to facilitate the recognition of a general hydromorphological gradient. Therefore, samples pre-classified as being of high or good hydromorphological quality were summarized to the category 'unstressed', whereas lower quality sites (moderate, poor, or bad) were defined as 'stressed'.

The hydromorphological degradation of the German stream types D01, D02, and D03 was analysed using 90 samples with 104 site protocol variables. Hydromorphological degradation was supposed to be the main stressor in German stream types.

3.4.5 Development of a Structure Index for German lowland streams

The German Structure Index (GSI) combines several stream type-specific hydromorphological features on different spatial scales, such as land use, channel morphology, or riparian vegetation, to a single index value. Because the GSI was based on objective variables recorded from either field surveys or maps, it provides a more objective measure of hydromorphological degradation compared to the rather subjective judgment of the pre-selection. NMS and subsequently Indicator Value Analysis were used to identify hydromorphological variables suited to describe a hydromorphological gradient. The variables were divided into 'positive' or 'negative', representing either high/good or moderate/poor/bad hydromorphological conditions. Selected variables were tested for significant differences between the two groups (Mann-Whitney-U-Test). Redundant variables were identified using correlation analysis (Pearson). However, similar variables may provide different information when recorded at different spatial scales and, hence, the information on the hydromorphological status of a site is also different, even if strong inter-correlation between those variables occur. For example a high proportion of native forest in the catchment indicates the morphological integrity of a site, whereas '% shading at zenith (foliage cover)' of a site provides information about the riparian vegetation and in-stream habitat quality itself, without being necessarily linked to a high proportion of native forests in the catchment. Hence, variables were not automatically rejected, if interdependence was high (Pearson correlation $r > 0.700$).

Table 3.2: Hydromorphological variables used to calculate group indices for medium-sized sand-bottom rivers in the German lowlands (D03), with respective spatial scale and calculation formula.

	Group index	Hydromorphological variable	Spatial scale	Calculation formula
'Positive'	*Debris Index*	No. of debris dams, No. of logs	Site	3 * Debris dams + Logs
	Organic substrate Index	% Xylal (twigs, branches, roots), % Organic substrates	Site	% Xylal / % Organic substrates
	Shading Index	% Shading at zenith, Average stream width	Site	% Shading * Average stream width
	Shoreline Index	% Shoreline covered with wooded vegetation, Average width of wooded riparian vegetation	Reach/ site	% Shoreline * Average width

Table 3.2, continued.

	Group index	Hydromorphological variable	Spatial scale	Calculation formula
'Negative'	*'Positive'/ 'Negative' Index*	Presence/absence: - Backwaters - Stagnation - Straightening - Impoundments - Removal of CWD	Reach/ site	Backwaters (0/1) – Stagnation (0/1) – Straightening (0/1) – Impoundments (0/1) – Removal of CWD (0/1)
	Land Use Index	% Pasture/grass-/bushland % Crop % Urban settlement/ industry	Catchment/ reach	% Urban * 5 + % Crop * 3 + % Pasture/grass-/bushland
	Scouring Index	Scouring below floodplain level	Reach/ site	Original measure from site protocol [cm]
	Bank Fixation Index	% Concrete % Stones % Wood/trees	Site	% Concrete * 5 + % Stones * 3 + % Wood/trees

A group index was calculated for each variable group, representing a certain habitat quality feature (Table 3.2). Three group indices ('Debris Index', 'Land Use Index', 'Bank Fixation Index') were calculated using weighing factors in order to consider the different quality of categories present for a single variable. For example, in case of the 'Bank Fixation Index', concrete-fixed banks are weighed higher than stones (rip-rap) and stones more than wood-fixed banks (Table 3.2). 'Positive' and 'negative' group indices were finally summed up to form the GSI. A list of site protocol variables used for the analysis with information on the spatial scale is given in Appendix 1. The GSI was used to correlate biota (represented by biocoenotic metrics) with hydromorphological quality of a site (see also Feld et al., 2002a; Pauls et al., 2002; Lorenz et al., 2004a; 2004b).

3.4.6 Statistical analysis

Correlation analysis and Mann-Whitney-U-Tests were performed with the XLStat 5.2 statistical software package (Addinsoft SARL, 2002). The Mann-Whitney-U-Test for non-parametric data was chosen, since frequency plots revealed a lack of normal distribution for all variables. As variables differed in numerical scaling and units of measurement (nominal (binary), ordinal, and interval scales), Non-metric Multidimensional Scaling (NMS) was used for multivariate analysis, as it provides an appropriate tool for non-parametric data of different numerical scales (McCune & Mefford, 1999).

To provide comparability between hydromorphological variables of different measurement units, all variables were standardized by dividing each value by the square root of the respective variables sum of all squared values (Formula 3.1). Thus, the sum of squares will become 1 for each variable, which equalizes the contribution of variables to the analysis (Podani, 2000).

$$b = \frac{x_{ij}}{\sqrt[2]{\sum_{j=1}^{n} x_{ij}^2}}$$

Formula 3.1

b = standardized value
x_{ij} = raw value of the i-th variable in the j-th sample

All NMS analysis was performed using PC-Ord's (McCune & Mefford, 1999) 'autopilot' settings: a four-dimensional solution as a starting point based on Bray-Curtis distance measures (Bray & Curtis, 1957) with medium speed and thoroughness; 15 runs with real data and 30 runs with randomized data, and a stability criterion of 0.0001. The variance explained by each multivariate axis and Pearson's Correlation Coefficient for the correlation of hydromorphological variables with each multivariate axis were calculated using PC-Ord. Presented two-dimensional ordination plots always show axes pairs, which explain the maximum variance of the hydromorphological variables used for the respective analysis. The 'final stress', a measure that explains the discrepancy between the multidimensionality of the data and the final (low-dimensional) ordination is given. According to Clarke (1993), Clarke & Warwick (2001), and Podani (2000), stress values below 0.2 represent acceptable and interpretable results.

Joint plots show the relationship between sample units and hydromorphological variables, the latter drawn as lines radiating from the centroid of the ordination scores. The angle and length of the line tell the direction and strength of the relationship (McCune & Mefford, 1999). For a given variable, the line forms the hypotenuse of a right triangle with the two other sides being correlation coefficients (r values) between the variable and the two axes. Only variables (lines) are shown, whose r value exceeded 0.500 (cut-off level = 0.5).

'IndVal' provides a tool to analyse species assemblages and uncover indicator species (Dufrêne & Legendre, 1997). 'IndVal' was used in a different way to identify hydromorphological variables that are suited to indicate high or low quality sites. Therefore, similar to Discriminant Analysis, a site-grouping variable had to be defined prior to analysis. Consequently, results are strongly affected by subjective judgment on group membership of sites, which was performed during pre-selection of sampling sites. In order to minimize the influence of a subjective judgment on statistical analysis and to make group allocation as transparent as possible, NMS analysis was used *a posteriori* to determine the number of groups and the sites belonging to a single group (Figure 3.2). Accordingly, the samples were divided into two groups: reference (high status) and heavily degraded (poor or bad status) (Table 3.3). The two groups represented extremes of the hydromorphological gradient without any overlap to adjacent quality classes (Figure 3.2) and comprised 15 samples each. Samples of a pre-classified 'good' or 'moderate' status were omitted.

The better a (hydromorphological) variable explains a group, the higher is the resulting 'IndVal' index. The highest explanation is reached (i.e. the index reaches its maximum value of 100 %), if all records of a single variable are found in a single group of samples and if the variable occurs in all samples of that group.

The statistical significance of the 'IndVal' Index values is evaluated using a randomization procedure (Dufrêne & Legendre, 1997).

3.5 Results

3.5.1 Stream type assignment

The first two axes of the NMS of the hydromorphological variables accounted for 83 % of its total variance (Figure 3.3). The first axis was mainly correlated with large-scale catchment characteristics, such as catchment size, geology, and natural land use practices, whereas the second axis was correlated with agricultural land use at the catchment scale and the natural shoreline vegetation and the degree of shading at the reach and site scale (Table 3.4). Reach- or site-related variables are also typologically important, if the substrate composition at a site is taken into consideration.

Out of the stream types pre-defined using the WFD, five types can be identified from Figure 3.3: Small organic type brooks in the German lowlands (type D02), small and medium-sized sand-bottom streams and rivers in the German lowlands (D01 and D03), and medium-sized rivers in the South Swedish lowlands (S05). However, sites of type D01 comprised only two samples and, thus, lack a sufficient sample size for a valid separation. Taking this into consideration, Figure 3.3 reveals only four stream types. Dutch samples formed a distinct cluster separated from other stream types, but with considerable overlap of Dutch slow running streams (N01) and Dutch fast running streams (N02).

3.5.2 Evaluation of hydromorphological degradation: All stream types

A gradient of hydromorphological degradation was evident along axis 1 of Figure 3.4. Both axes of the NMS plot accounted for nearly 85 % of the total variance of the environmental dataset. The first axis (60 % variance) represented the degradation and was, for example, negatively correlated with '% native forest', '% shoreline covered with wooded vegetation', and '% shading at zenith (foliage cover)' (Table 3.5). These variables indicate high hydromorphological quality ('unstressed') and were represented by sites located on the left hand side of the NMS plot (empty symbols in Figure 3.4). In contrast, 'stressed' sites were best explained by, for example, '% agriculture', which was positively correlated with the first axis of the NMS plot.

Table 3.3: Median value and range of hydromorphological variables of stream type D03, significantly differing between reference and heavily degraded sites (poor or bad hydromorphological status, see Figure 3.2) ($p < 0.001$, Mann-Whitney-U-Test).

Hydromorphological variable	Reference	Heavily degraded
	Median (range)	Median (range)
Catchment: % Native forest	20 (0–40)	0 (0)
Site: % Native forest	90 (80–100)	0 (0)
Site: % Total agriculture	0 (0–10)	85 (10–100)
Reach: % Impoundments/dams up-/downstream	0 (0)	85 (40–100)
Site: % Shading at zenith (foliage cover)	80 (60–80)	0 (0)
Site: Average width of wooded riparian vegetation [m]	150 (110–200)	6 (0–16)
Site: No. of debris dams (> 0.3 m³)	4 (3–22)	0 (0)
Site: No. of logs (> 10 cm diameter)	63 (35–100)	0 (0)
Site: % Shoreline covered with wooded riparian vegetation	100 (90–100)	20 (0–75)
Site: % Bank fixation stones (rip-rap)	0 (0)	100 (20–100)
Site: No. of organic substrates	3 (2–5)	1 (0–2)
Site: Max. current velocity [cm s^{-1}]	43 (31–63)	26 (7–53)

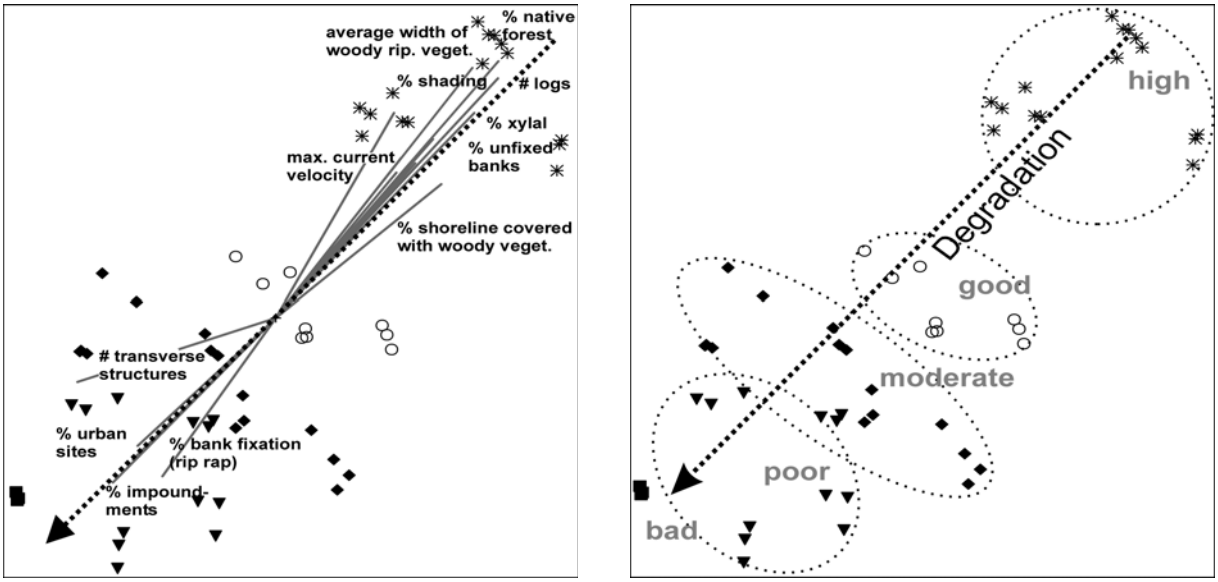

Figure 3.2: NMS joint plot of 95 hydromorphological variables of 54 samples of 'medium-sized sand-bottom rivers in the German lowlands'. Lines indicate strongest variables to describe the hydromorphological status (cut-off level: 0.500) and arrow indicates hydromorphological degradation. Final Stress: 0.114. Variance explained: Axis 1: 58.8 %; axis 2: 28.9 %. 'High' represents reference, 'poor' and 'bad' represent heavily degraded.

The second axis of the NMS ordination plot (Figure 3.4) was strongly correlated with catchment geology. Sites dominated by alluvial deposits are situated in the upper part of the NMS plot, whereas moraine-dominated sites were located at the bottom. '(%) Native forest' was negatively correlated with NMS axis 2 (Table 3.5). Sites with a high proportion of native forest in their catchment, a rather strong descriptor of hydromorphological reference conditions, were clustered in the lower left corner of the NMS plot (in particular stream type S05). Figure 3.4 reveals a clear gradient of hydromorphological degradation for the German stream types (D01–D03) (see also Figure 3.5), coinciding with the presumed main stressor 'hydromorphological degradation' for these stream types. In contrast, stream types S05, N01, and N02 show a considerable overlap of 'unstressed' and 'stressed' sites.

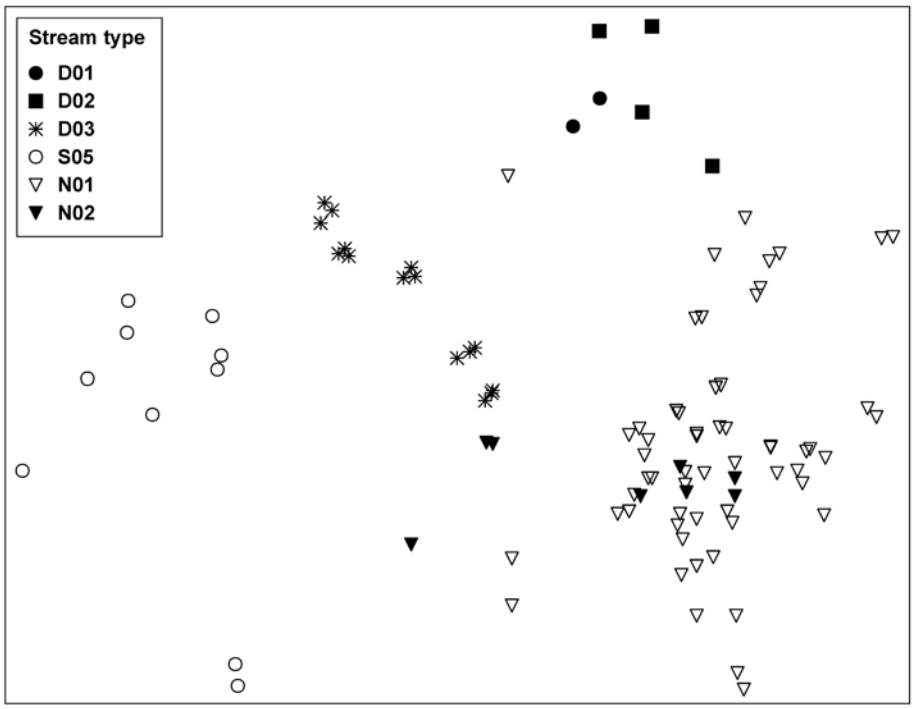

Figure 3.3: NMS ordination plot of 97 reference samples of six European stream types. Final stress: 0.155. Variance explained: Axis 1: 56.7 %; axis 2: 26.4 %.

Table 3.4: Pearson's correlation coefficient (r) for hydromorphological variables with the first two NMS axes of the ordination of typological aspects (Figure 3.3). Only correlations > 0.500 are listed.

Axis 1	r	Axis 2	r
Catchment: % Moraines	-0.884	Site: % CPOM	0.608
Catchment: % Native forest	-0.851	Catchment: % Pasture	-0.585
Catchment: % Alluvial deposits	0.823	Catchment: % Agriculture	-0.559
Catchment: % Wetland	-0.678	Site: % Shading at zenith (foliage cover)	0.524
Catchment: % Non-native forest	0.608	Site: % Shoreline covered with wooded vegetation	0.507
Site: % Psammal/psammopelal (sand/sand and mud)	0.596		
Site: Average stream width	-0.596		
Site: % Macrolithal (cobbles)	-0.585		
Catchment: Distance to source	-0.567		
Catchment: Catchment area	-0.566		
Site: % Megalithal (large cobbles and boulders)	-0.555		
Site: % Shoreline covered with wooded vegetation	-0.531		
Catchment: % Acid silicate rocks	-0.526		
Reach: Altitude	-0.523		
Catchment: % Organic formations	-0.519		

Table 3.5: Pearson's correlation coefficient (r) of hydromorphological variables with the two NMS axes of the ordination of habitat degradation (Figure 3.4). Only correlations > 0.500 are listed

Axis 1	r	Axis 2	r
Catchment: % Native forest	-0.713	Catchment: % Moraines	-0.763
Site: % Shading at zenith (foliage cover)	-0.630	Catchment: % Native forest	-0.727
Site: % Shoreline covered with wooded vegetation	-0.595	Catchment: % Alluvial deposits	0.662
Catchment: % Wetland	-0.506	Catchment: % Non-native forest	0.573
Catchment: % Agriculture	0.509	Site: Average stream width	-0.559
		Site: % Macrolithal (cobbles)	-0.525

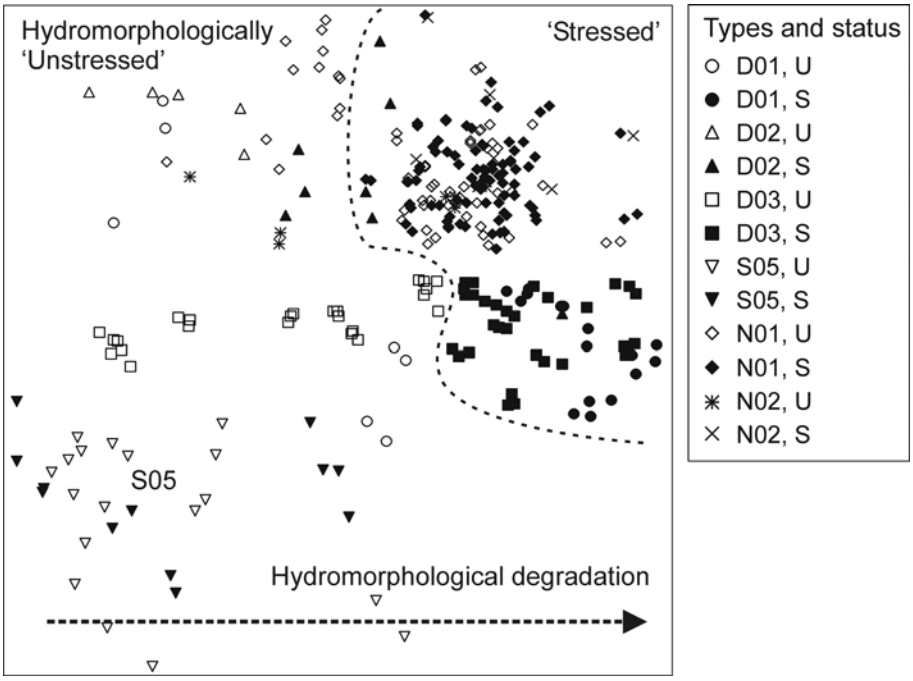

Figure 3.4: NMS ordination plot of 275 samples of six investigated stream types (explanation of stream types in Table 3.1). Symbols indicate stream type and status of degradation pre-classified as 'U' = unstressed (empty symbols, pre-classified 'high' or 'good status') and 'S' = stressed (filled symbols, pre-classified moderate, poor, or bad status). Final stress: 0.172. Variance explained: Axis 1: 60.2 %; axis 2: 24.2 %.

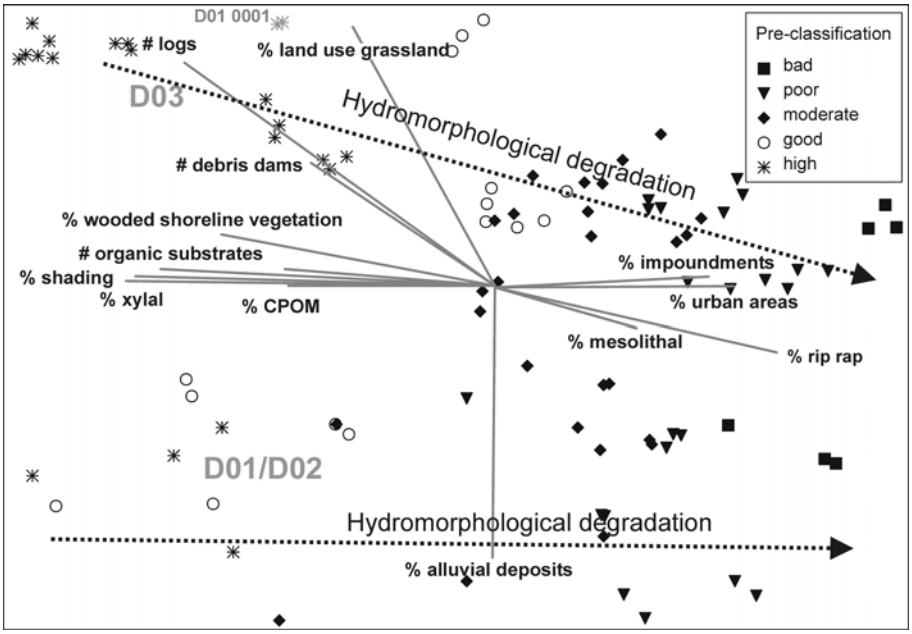

Figure 3.5: NMS joint plot of hydromorphological degradation of 90 samples of three German stream types (D01, D02, and D03). Lines indicate variables that describe the gradient best (cut-off level: 0.500). Arrows indicate gradients of hydromorphological degradation. Final Stress: 0.108. Variance explained: Axis 1: 53.3 %; axis 2: 18.5 %.

3.5.3 Evaluation of hydromorphological degradation: German stream types

A gradient of hydromorphological degradation was evident along axis 1 (Figure 3.5) for both small and medium-sized streams and rivers. This gradient was best explained by site-related variables (Table 3.6), such as the proportion and number of organic substrates on the stream bed, the proportions of wooded shoreline, and bank fixation. At the catchment scale, it was the proportion of urban areas that indicateed hydromorphological degradation for the three German stream types. The separation of small and medium-sized samples along axis 2 was predominantly based on catchment geology ('% alluvial deposits' *vs.* '% moraines'), '% grass-/bushland', and 'no. of logs > 10 cm Ø' on the stream bed (Table 3.6), the latter being more frequent in medium-sized rivers. However, the pre-classified hydromorphological reference site of type D01 (D01 0001 in Figure 3.5) was clustered together with the reference sites type D03 due to comparatively high proportions of organic substrates on the stream bottom. In particular, the number of logs > 10 cm Ø' on the streambed resembled those recorded for D03 reference sites. Moreover, regarding the stream width, the 'small' D01 site was similar to the medium-sized sites of type D03.

Hydromorphological degradation of type D03 can be derived almost entirely from the site protocol variables, as reflected by a clear gradient for this stream type. The overlap at the transition from good to moderate and from moderate to poor status (Figure 3.5) disappeared, when stream type D03 was analysed separately (Figure 3.2). Here, the pre-classification was well reflected by the NMS ordination, which accounted for almost 88 % of the total variance in the environmental dataset. A similar result was evident for small sand-bottom streams (D01) and organic type brooks (D02), when analysed separately (not shown here). Hence, the three German stream types, as well as their hydromorphological status can be identified solely by environmental variables.

3.5.4 Development of a Structure Index for German lowland streams

In total 'IndVal' analysis revealed 25 variables, which significantly described the end points of the hydromorphological gradient (Table 3.7). The variables can be separated into those, which predominantly indicated reference conditions ('positive') and those which were connected with a heavily degraded hydromorphology ('negative'). Some variables revealed a considerable correlation, as it was, for example evident for the proportion of native forests on catchment and reach scale and the number of logs in the stream channel (Figure 3.6).

Measures of several hydromorphological variables were significantly different between reference and heavily degraded sites (Table 3.3). Consequently, heavily degraded sites were mainly characterized by extensive agricultural land use in the floodplain, extensive bank modification, the lack of a dense riparian wooded vegetation, and thus, the lack of shading and large wood on the stream bottom. In addition, only a small amount of organic substrate occurred at sites of a poor or bad hydromorphological status, and hydrology was strongly affected by stagnation due to weirs, which significantly reduced the maximum current velocity.

In a next step, variables representing a certain habitat quality feature (e. g., large wood, channel modification, or land use), were combined to group indices related to different spatial scales. Altogether, eight group indices were defined and calculated (Table 3.2).

Table 3.6: Pearson's correlation coefficient (r) of hydromorphological variables with NMS axes of the ordination of habitat degradation in German stream types (Figure 3.5). Only correlations > 0.500 listed.

Axis 1	r	Axis 2	r
Site: % Xylal (e. g., dead wood, branches, roots)	-0.761	Catchment: % Alluvial deposits	-0.651
Site: % Shading at zenith (foliage cover)	-0.750	Catchment: % Open grass-/bush land	0.637
Site: % Unfixed banks	-0.725	Site: No. of logs (> 10 cm diameter)	0.594
Site: No. of logs	-0.700	Catchment: % Sander	0.505
Site: % Bank fixation stones (rip-rap)	0.666	Catchment: % Moraines	0.502
Site: % Shoreline covered with wooded vegetation	-0.657		
Catchment: Land use: % Urban sites	0.612		
Reach: % Impoundments	0.600		
Site: No. of organic substrates	-0.576		
Site: % CPOM	-0.569		
Site: No of debris dams (> 0.3 m³)	-0.537		
Catchment: % Native forest	-0.536		

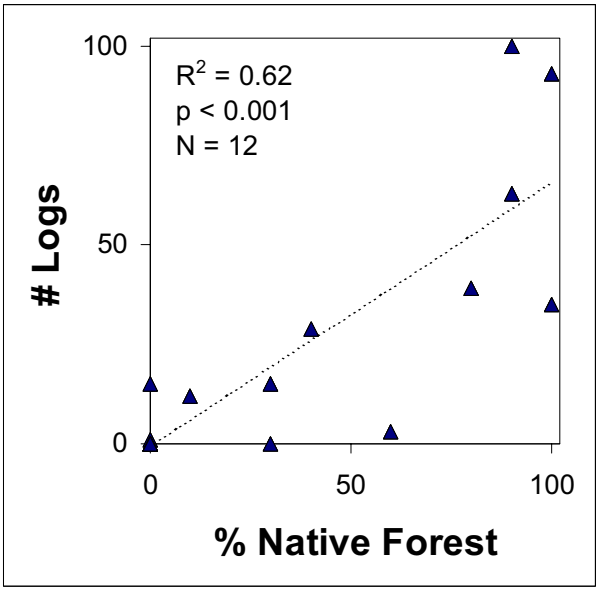

Figure 3.6: Correlation of % native forests in the floodplain and in-stream number of logs for 12 sites in medium-sized sand-bottom lowland rivers (D03).

Table 3.7: 'IndVal' results of suitable core variables to describe the hydromorphological gradient detected for stream type D03 (significance level: < 0.05, 499 iterations). 'Positive' variables indicate reference conditions (high quality), 'negative' variables heavily degraded conditions (poor or bad quality). (IV = 'IndVal' index)

'Positive' variable	IV	'Negative' variable	IV
Site: Max. current velocity [cm s^{-1}]	95.54	Reach: % Urban sites	100.00
Site: No. of logs (> 10 cm diameter)	75.63	Reach: Culverting up-/downstream	100.00
Reach: % Native forest	63.52	Reach: No of dams obstructing migration up-/downstream	100.00
Site: Average width of wooded riparian vegetation	61.62	Site: % Bank fixation stones (rip-rap)	56.23
Catchment: % Native forest	60.31	Site: % Bed fixation stones	50.00
Site: % Xylal (e. g., dead wood, branches, roots)	55.56	Reach: % Impoundments/dams	45.07
Site: No. of debris dams (> 0.3 m³)	50.65	Reach: No of transverse structures (e. g., weirs, dams, bridges)	44.69
Site: % Unfixed banks	43.10	Reach: Stagnation	43.80
Site: % CPOM	35.46	Reach: Straightening	38.46
Site: % Shoreline covered with wooded riparian vegetation	33.01	Site: Removal of large wood	30.30
Site: CV depth	27.50	Reach: Channel form	27.95
Site: % Shading at zenith (foliage cover)	43.48	Site: Scouring	25.00
Site: No. of organic substrates	29.90		

The ´Debris Index´ weighs debris dams more (factor 3) than logs, for debris dams provide a higher habitat complexity and diversity. The relative '% xylal' in relation to the total '% organic substrates' was defined the ´Organic Substrate Index´. As the maximum degree of shading usually decreases with increasing stream channel width, the ´Shading Index´ considers both by the relation to the width-dependent maximum value. However, if a sample site is nearly complete shaded, 100 % is taken as the resulting shading index independent of the respective stream width. The ´Shoreline Index´ refers to the two dimensional extension of the wooded riparian vegetation (along the stream course as well as in the floodplain), and thus assesses the buffer strip functionality. Certain 'positive' and 'negative' hydromorphological features are – on a presence/absence level – combined to the ´Positive/Negative Index´. The extent of land use in the floodplain is summarized with the ´Land Use Index´, and a weighing factor allows for the severity (urban areas > crop land > pasture, meadow, or open grassland). The ´Scouring Index´ directly represents the measured incision depth of the stream channel. The ´Bank Fixation Index´ is related to the total share of fixed banks and different qualities of fixation are allowed for by weighing (concrete > stones > wood or trees). For each sample, group indices were calculated and related to the respective stream type-specific maximum value of a single index plus 10 %. Thus, each index value was related to a 110 % basis, following the assumption that the samples did not necessarily reflect the whole range of best to worst conditions for a certain stream type. Finally, the re-scaled proportional values of 'negative' group indices were simply added up and subtracted from the sum of 'positive' group indices, representing the GSI.

3.6 Discussion

The objective of this Chapter was to identify suitable variables to describe hydromorphological degradation of stream types in ecoregions 13 and 14 of Central Europe. If data analysis was shifted from several stream types to a single stream type, the respective scale of hydromorphological variables also changed from catchment-scale to reach- or site-scale. Thus, the set of hydromorphological variables to identify hydromorphological degradation strongly depends on the spatial scale. Earlier studies have also stressed the role of spatial scale in physical habitat assessment (Richards et al., 1996; Allan et al., 1997; Davies et al., 2000; Sponseller et al., 2001), and some have argued a distinct spatial hierarchy existed that influenced environmental variables of riverine habitats (Frissell et al., 1986; Rabeni, 2000). The results presented here support this hierarchical organisation of hydromorphological variables.

3.6.1 Stream type assignment

The results presented here support the typological relevance of the hydrological and geological descriptors according to the EU WFD (EU commission, 2000, System A and B; see also Introduction, Section 1.1). Catchment geology, altitude, substrate composition, and stream and catchment size discriminated between the investigated stream types of ecoregions 13 and 14 in Central Europe (Table 3.4). In addition, the current study revealed land use characteristics as important typological variables at the catchment scale. For example, the proportion of native forest correlated very well with axis 1 of the typological NMS (Table 3.4). Yet, catchment land use characteristics rather reflect the degree of human activities in the catchment and thus already represent hydromorphological degradation. In case of type S05, both outlier samples (Figure 3.3) were influenced by intensive agricultural land use and, therefore, likely did not represent real hydromorphological reference sites. Allan et al. (1997) and Richards et al. (1996) found catchment geology and land use attributes, in particular the proportion of (intensive) row-crop agriculture, to be strong descriptors of stream habitat conditions and macroinvertebrate communities. The land use-controlled discrimination between lowland stream types of Central and Western Europe does not correspond very well with the potential natural vegetation expected for this region: deciduous forest (Ellenberg, 1996). Land use appears to reflect degradation rather than typological aspects. The consideration of additional site-scale hydromorphological features, such as the proportions of the shoreline covered with wooded riparian vegetation and shading of the stream bottom at zenith supports this assumption. Both variables were closely related to degradation and a dense riparian vegetation, usually dominated by *Alnus glutinosa* (Black Alder) and *Salix* spp. (Willow), which can be expected along streams and rivers in both ecoregions (Ellenberg, 1996). With regard to catchment land use properties, the reference dataset considered here does not appear to fulfil the essential requirements on reference conditions (Hughes, 1995; Wiederholm & Johnson, 1996; Hering et al., 2003; Nijboer et al., 2004).

The Dutch stream types were not separated when using hydromorphological variables at a large spatial scale (Figure 3.3). Thus, it seems that they are similar from a hydromorphological point of view, even if they represent two ecoregions. This is in contrast with the faunistic differences of the Dutch stream types that were previously identified in Chapter 2 and may be an example for the deviation of stream type-specific abiotic and biotic properties. Consequently, hydromorphological degradation could be defined commonly for both stream types, whereas the assessment of its impact on the in-stream community must refer to stream type-specific reference communities.

With regard to the small German stream types D01 and D02, the results imply a similar conclusion. Both German types are well defined (Pottgiesser & Sommerhäuser, 2004) and characterized by different benthic invertebrate reference communities. Moreover, the bottom of small sand-dominated lowland rivers (D01) is usually dominated by sand, whereas a domination of sand in small organic substrate-dominated rivers (D02) already indicates a certain level of degradation. The bottom of the latter type is naturally dominated by mosses (e. g., *Sphagnum* spp.) and/or CPOM (e. g., leaves, twigs) (Sommerhäuser & Schuhmacher, 2003). Yet, despite of the (micro-scale) habitat difference, both types revealed similar hydromorphological conditions at larger (reach, catchment) spatial scales as shown in Figure 3.3. Thus, if indicated by reach-scaled variables, hydromorphological degradation may be defined commonly for both stream types.

3.6.2 Evaluation of hydromorphological degradation: All stream types

The analysis of the hydromorphological degradation revealed two groups: The first group comprised the Dutch and Swedish types, the second group included the German types. For the latter hydromorphological impact was clearly and continuously detectable along a hydromorphological gradient, whereas Dutch and Swedish samples of various pre-classified status clustered together (Figure 3.4). This finding was not surprising, since the identification of a hydromorphological gradient was only aimed at in case of the German types. The pre-selection in Germany was focussed on covering the different hydromorphological conditions from reference to bad status as good as possible. However, while the identification of a general (pollution, hydromorphology, nutrients) degradation was focussed on in The Netherlands, the presumed main stressor was organic pollution in Sweden. The pre-selection of Dutch and Swedish sites presumably did not represent a hydromorphological gradient, too. Consequently, Swedish and Dutch samples clustered opposite one another in the ordination space (Figure 3.4). Thus, Swedish sites were only weakly affected by hydromorphological degradation, whereas Dutch sites were predominantly in a moderate to bad hydromorphological condition. This was supported by a comparison of, for example, the land use at Dutch and Swedish sites: The proportion of natural forest was zero in case of all Dutch sites and ranged from 20–90 % (mean: 63 %) for the Swedish sites. Hence, regarding the catchment scale, a severe land use impact was evident for all Dutch sites. The analysis of hydromorphological variables, comparing different stream types, was mainly governed by catchment properties. However, at the reach- and site-scale, several variables, such as the proportions of shoreline covered with wooded vegetation and shading at zenith were shown to be suitable descriptors of hydromorphological impact. Therefore, the evaluation of the

hydromorphological status should include variables measured for stretches of 10 m up to at least several km. The AQEM site protocol considers different spatial scales, of which only catchment properties and some up-/downstream (stretch of 500–1000 m) variables are available through topographical and geological maps. Thus, physical habitat assessment necessitates field work to obtain several important variables at smaller spatial scales.

3.6.3 Evaluation of hydromorphological degradation: German stream types

The significance of 'small-scale' (reach, site) hydromorphological variables was evident if the analysis was limited to the German stream types, representing a smaller geographical extent. Yet the discrimination of German lowland stream types was, amongst other variables, controlled by catchment geology (Bundesanstalt für Geowissenschaften und Rohstoffe, 1993). The majority of medium-sized sites were located in East Germany (dominated by moraine and sander deposits of the Weichselian glaciers) whereas small sites were exclusively located in the part of West Germany that was unaffected by the last ice age (characterized by alluvial (fluviatile) deposits). Thus, the discrimination of small and medium-sized types in Figure 3.5 reflects the pre-selection of sites rather than typological differences.

Reach- and site-related variables became major descriptors of hydromorphological degradation, in particular the amount and quality of organic substrates (large wood, CPOM) and variables describing the riparian vegetation and channel modification. In particular, large wood appeared to be an important factor influencing the hydromorphological conditions of these stream types which is consistent with the findings of Harmon et al. (1986), Gurnell et al. (1995), Hering & Reich (1997), Mutz (2000), and Kail (2003). Riparian buffer strips are important to control the influence of sediment input from row-crop agricultural areas on the riverine benthic community (Newbold et al., 1980; Allan et al., 1997; Tabacchi et al., 1998). Newbold et al. (1980) defined a minimum width of 30 m for riparian buffer strips as sufficient to provide optimal habitat conditions for macroinvertebrates. Allan et al. (1997) stressed the role of riparian buffer strips as a barrier for nutrient supply and sediment delivery. The importance of both a dense and wide riparian buffer was also supported by the current study. The 'IndVal' analysis of hydromorphological variables for type D03 revealed the proportion of shoreline covered with wooded vegetation and the average width of wooded riparian vegetation to significantly differ between reference sites and sites of a poor or bad hydromorphological status in medium-sized sand-bottom rivers. Reference sites were characterized by riparian trees, which covered 90–100 % of the shoreline and extended between 110 and 200 m into the floodplain. It appears that the extent of riparian vegetation in the floodplain plays a major role, which was accounted for by the ´Shoreline Index´ being part of the GSI. At the level of a single stream type, hydromorphological degradation appeared to be particularly related to site-scale variables. Hence, the smaller the spatial extent of the sites was, the smaller was the spatial scale of well-discriminating environmental variables. As a consequence, site-related physical habitat evaluation becomes important, if applied at a small spatial scale. Several methods integrate those site-related features in Europe, such as the British River Habitat Survey (RHS, Raven et al., 1997, 1998, 2002), the German 'Strukturgütekartierung' (LAWA, 2000) or the French SEQ-MP (Agence de l'Eau Rhin-Meuse,

1996). However, a common lack of all methods is that they do not cover the indicator variables listed in Table 3.3 and 3.7. The habitat evaluation protocols may be improved simply by adding specific items addressing meso- (reach) and micro-scale (site, habitat) variables within a reach of 10 m to several km, such as the proportion of shading on the stream bottom, the number of in-stream debris dams and logs, the proportion of wooded riparian vegetation and bank fixation, and the proportion and number of organic substrates. The importance of the meso-scale was also stressed by Beisel et al. (1998a, b). Physical habitat evaluation applying the AQEM site protocol includes these specific and detailed information.

3.6.4 Development of a Structure Index for medium-sized sand bottom rivers in the German lowlands

The results underline the importance of environmental variables for the development of tools to assess the river health in European (lowland) rivers. Future assessment systems for European rivers have to be predominantly based on the riverine communities (fauna and flora) as demanded by the EU WFD (EU commission, 2000, see Chapter 1 Introduction). However, there is an urgent need to identify and define the major impacts on the aquatic fauna and flora first. The identification and measure of environmental (impact) gradients provides a tool to calibrate community-based assessment systems during the process of development and to validate the final systems. By combining eight groups of hydromorphological variables (large wood, organic substrates, shading, shoreline, positive and negative structure elements, land use, scouring, and bank fixation) the German Structure Index (GSI) provides such a measure and continuously describes the hydromorphological status of a site. Finally, by relation to the GSI, single community measures (ecological traits or metrics, e. g., feeding types, current preferences, substrate preferences) and indicator taxa indices can be identified to be suited candidates of a multi-metric index to assess the impact of hydromorphological degradation on the aquatic macroinvertebrates at a site (Hering et al., 2004a, b, see also Chapter 6). Lorenz et al. (2004a, b) documented the correlation of the hydromorphological status of a site and numerous metrics derived from the macroinvertebrate community sampled at that site. Feld et al. (2002a) found the number of Simuliid taxa to be significantly higher at hydromorphologically ‘unstressed’ sites.

In comparison with the existing methods of physical habitat evaluation (e. g., the German ‘Strukturgütekartierung’; LAWA, 2000), the GSI provides two advantages: Firstly, the GSI is a continuous measure of hydromorphological quality, allowing of simple correlation with biocoenotic metrics. Secondly, the GSI refers to a hydromorphological gradient derived from numerous environmental variables and likely covers the present range of hydromorphological conditions from reference to bad status within the examined stream types. Feld et al. (2002a), Pauls et al. (2002), and Lorenz et al. (2004a, b) reported the GSI being a suitable measure for the identification of metrics to assess the impact of hydromorphological degradation on benthic macroinvertebrates.

A potential deficit of the approach that was followed here was the subjective pre-selection of candidate sites according to the researcher’s subjective judgment on the stressor-specific ecological status of the sites. Since the approach aimed at covering the whole gradient of

the presumed main stressor, this was an inevitable prerequisite for the detection of a gradual impact of the stressor. An alternative procedure would have been to randomize the selection of sites, however this would have lead to a multiplied effort in order to cover a comparable gradient length, whereas it is unlikely that a different set of environmental descriptors would have been identified to define and measure hydromorphological degradation. The pre-selection thus offers a practical and successful method to gain a set of sites suited for the development of stream type-specific assessment system.

4 Linking macroinvertebrate taxa and derived ecological metrics to hydromorphology and land use at different spatial scales in Central European lowland rivers: a multivariate approach

4.1 Scope

The previous Chapters in general addressed the analysis of biotic and abiotic properties at different spatial scales. However, the analysis of both Chapters was limited to either faunistic data (Chapter 2) or hydromorphological variables (Chapter 3). The statistical approaches that have been applied are members of the 'family' of indirect gradient analysis. The term 'indirect' means that the ordination axes (gradients) derived from the community composition are not linked to other variables. However, environmental variables may be used to interpret the community gradients. One advantage of indirect gradient analysis is that the results are not biased by any other kind of information; they 'speak for themselves'. Thus, indirect ordination provides a useful tool for experienced scientists to analyse and recognize the inherent structure of a abiotic or biotic dataset and helps identifying the potential underlying mechanisms responsible for the structure. This was aimed at in the previous sections, so that indirect methods, such as 'Non-metric Multidimensional Scaling' have been chosen.

Another 'family' of ordination can be summarized by the term 'direct gradient analysis'. In direct ordination both the community and environmental datasets are analysed simultaneously. Canonical ordination, for example, aims at ordering the main structure in the fauna data along the main environmental gradient. Since the first ordination axis represents a linear combination of the environmental variables canonical ordination is also known as 'constrained ordination'. This specific characteristic was the reason to use direct gradient analysis in this study to link the fauna and environmental data. The analysis aims at identifying the variation of the fauna that is explained explicitly by the environmental variables used for the analysis. Furthermore, the individual suitability of certain taxa, metrics, and environmental variables to identify and describe hydromorphological degradation is explored. The analysis again considers the whole AQEM lowland dataset of ecoregions 13 and 14.

4.2 Summary

The correlation of the faunal composition of benthic invertebrates and derived ecological metrics with hydromorphological and land use gradients in European lowland rivers at four different spatial scales is presented: supra-catchment ('mega'-scale), catchment (macro-), reach (meso-), and habitat (micro-). Field surveys and maps yielded 130 parameters characterizing hydromorphology and land use gradients covering 75 river sections in Sweden, The Netherlands, Germany and Poland. In total, 244 macroinvertebrate taxa and 84 derived ecological metrics, such as functional guilds, assemblage diversity and composition measures, and different biotic indices were included in the analysis of the faunal characteristics. Direct multivariate analysis, Canonical Correspondence Analysis (CCA) and Redundancy

Analysis (RDA) identified hydromorphological and land use variables that significantly contributed to the multiple regression of taxa and metrics at the four spatial scales. Environmental variables explained 17.5, 8.0, 19.3, and 14.0 % of the species' variance at the 'mega-', macro-, meso-, and micro-scales, respectively. These variables explained 21.3, 8.1, 20.3, and 14.1 % of the metrics' variance at the respective spatial scales. The linkage of taxa, metrics and 34 non-correlative environmental variables was analysed with Indicator Species Analysis (ISA). All taxa/metrics having an Indicator Value (*IV*) > 0.30 were defined as good indicators. For the 244 taxa tested with ISA, 76 were indicative at the macro-scale, 112 at the meso-scale, and 62 at the micro-scale. Trichoptera were most indicative at the macro and meso-scales, Diptera at the micro-scale. For the 84 metrics tested, most indications were observed at the meso-scale (77), followed by the macro (64) and micro-scales (56). The mean proportion of significant correlations for metrics was much higher than for taxa (78.2 *vs*. 34.2 %) indicating that metrics are more closely related to environmental gradients. Richness/diversity measures showed highest scale-dependent differences and were most indicative at the micro-scale. Functional measures were indicative at the macro, meso, and micro-scale.

4.3 Introduction

The major environmental variables that control the composition of aquatic macroinvertebrates act at different spatial scales (Frissell et al., 1986; Corkum, 1992; Poff, 1997; Townsend et al., 1997; Fitzpatrick et al., 2001; Brosse et al., 2003; Weigel et al., 2003). An overview of the main factors has already been given in Section 1.4 (Introduction). However most of the studies (e. g., Allan & Johnson, 1997; Townsend et al., 2003) focussed on 'natural' environmental variables, such as surface geology, basin diameter, or relief ratio. Those variables are constant regarding the time scale that is relevant here. Investigations analysing the impact of hydromorphological parameters as a human induced stressor, however, are related to 'non-natural' environmental variables that are usually more or less variable. Those studies are fairly rare. In large areas of Europe hydromorphological degradation is thought to be the most important stressor affecting in-stream macroinvertebrates as previously discussed and also stated by Raven et al. (2002). This may apply to other regions, too, as implied by studies focussing on the impact of hydromorphological gradients throughout the USA (Barbour et al., 1999; Griffith et al., 2001; Snyder et al., 2003) or in New Zealand (Townsend et al., 2003).

Yet, the spatial hierarchy of the variable (human-induced) environmental descriptors has rarely been linked to the biota of rivers. This link is needed to identify, separate, and assess the effects of perturbations on the community and to derive appropriate management options. For future biomonitoring and assessment approaches it is crucial to understand which type of hydromorphological changes affect which part of the biota. Stream macroinvertebrates are principally good indicators of environmental stress at the catchment, reach, and habitat scale due to their sensitivity for differently scaled environmental parameters (Hellawell, 1986; Rosenberg & Resh, 1993), and several assessment systems are based on them (Wright, Furse & Armitage, 1993; Barbour et al. 1999; Smith et al., 1999; Hering et al.,

2004a). However, most assessment systems aim at the indication of the impact of organic pollution or an overall 'ecological quality' of a river.

Using an extensive hydromorphological and macroinvertebrate dataset of European lowland rivers this chapter addresses the identification of: i) hydromorphological and land use variables describing hydromorphological gradients at different spatial scales, ii) macroinvertebrate taxa and metrics related to the environmental gradients, iii) taxa and metrics, such as functional guilds, diversity and composition measures, and different biotic indices best suited to assess the impact of hydromorphological degradation, and iv) hydromorphological and land use variables having the strongest impact on macroinvertebrate assemblage composition at different spatial scales.

4.4 Material and methods

4.4.1 Study site

75 stream and river sections in Southern Sweden, The Netherlands, Northern Germany, and Western and Central Poland were selected (Figure 4.1). The sites were pre-selected to cover a hydromorphological gradient characterized by different degrees of catchment and floodplain land use, riparian degradation, and flow regulation. This selection was restricted to unpolluted or slightly polluted sites (Feld, 2004; Hering et al., 2004a; Lorenz et al., 2004b). All sites were located in ecoregion 14 ('Central Lowlands' according to Illies, 1978), except nine Dutch sites in ecoregion 13 ('Western Lowlands'). Altitude ranged between 6.5 and 200 m a.s.l. and catchment size between 0.50 and 6,400 km² with 90 % of sites between 1 and 740 km². Except for the Polish sites a detailed description of the sampled stream types is given in Chapter 3. Sites in Poland represented the least disturbed conditions in the dataset with only very slight hydromorphological alteration. Land use in the catchment was dominated by forestry and, therefore, sites were characterized by the frequent presence of large wood on the stream bed. Catchment geology of all sites was dominated by moraines or sanders originated from the Weichselian ice age that covered eastern Germany, Poland, and Sweden and the 'Saale' ice age that covered Western Germany and The Netherlands. Consequently, channel substrates were dominated by sand with a small proportion of gravel. Cobbles and boulders occurred where channels cut into end moraines.

4.4.2 Sampling and sample processing

Taxa and environmental data (geographical, geological, hydrological, morphological, and land use parameters) were recorded from three research projects: AQEM (Hering et al., 2004a), DEMARECO (Feld & Bis, 2003), and STAR (www.eu-star.at). In all projects a standardized multi-habitat sampling procedure (Hering et al., 2003, 2004) was applied to sample macroinvertebrates with a 25 x 25 cm frame shovel sampler (500 µm mesh). 20 microhabitat patches were sampled within a section of 50 to 100 m of length. The 20 sample units were pooled and preserved in the field (ethanol, 96 %). The macroinvertebrates in these pooled samples were identified to species level except for Oligochaeta and most Diptera (both family level), and Chironomidae (genus level where possible). The sam-

ples were taken in spring (March to May; all countries), summer (June to July; all countries except Sweden), or autumn (August to October; all countries).

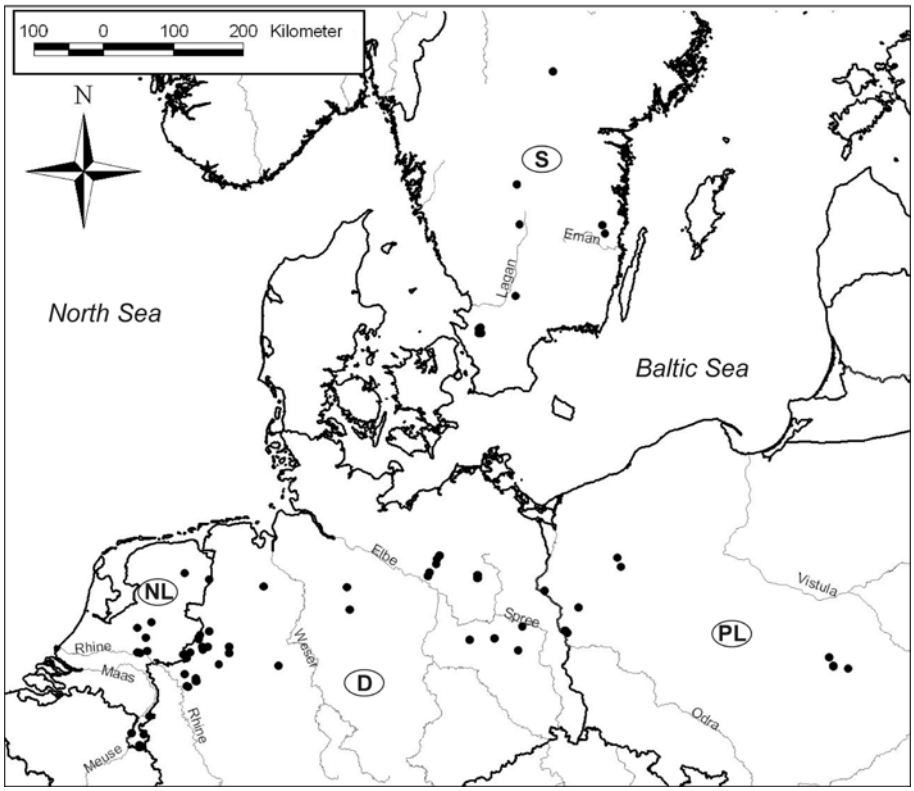

Figure 4.1: Location of the 75 study sites (●) in Central and Western Europe (S = Sweden, NL = The Netherlands, D = Germany, and PL = Poland).

Field data or data derived from maps and GIS yielded 130 environmental parameters. These parameters were assigned to four different spatial scales (Appendix 2) and followed the hierarchical concept of landscape (Frissell et al., 1986): The 'mega'-scale comprised 'supra-catchment' variables (sampling season, latitude, longitude, altitude, ecoregion) and the catchment area. These were used as covariables to determine their influence on the 'natural' variability introduced by them to the macroinvertebrate community. Catchment land use (e. g., % pasture, % crop land, and % urban settlement/industry) was assigned to the macro-scale. Meso-scale environmental variables referred to a reach of 250–500 m up- and downstream of a site, for example, floodplain land use, density and width of riparian vegetation, bank fixation, channel alteration, and flow regulation. Habitat variables were assigned to the micro-scale and refer directly to the sampling site, a stretch of 50–100 m of stream length.

4.4.3 Data analysis

Prior to data analysis two procedures were applied to improve the comparability of the data: 1) The selection was restricted to sites exclusively affected by hydromorphological degradation without any significant sign of organic pollution. The Saprobic Index (SI; Zelinka &

Marvan, 1961) and the ASPT (Armitage et al., 1983) were applied as filter criteria. While saprobic indices are widely used in Central Europe (e. g., in The Netherlands, Germany, Austria, Czech Republic) to detect organic pollution the suitability of the ASPT for detecting organic pollution in Central European rivers was recently stressed by Sandin & Hering (2004). Polluted sites with a Saprobic Index > 2.21 and an ASPT < 5.00 were excluded from further analysis. Following this filter procedure the database comprised 144 samples, of which 19 were taken in Sweden (10 spring and 9 autumn), 32 in The Netherlands (18 spring, 3 summer, and 11 autumn), 79 in Germany (28 spring, 35 summer, and 16 autumn), and 14 in Poland (10 spring, 2 summer, and 2 autumn). 2) A taxonomical adjustment was applied to improve commensurability of the biotic data. Taxa lists originating from different research studies and researchers are usually biased in terms of the taxonomical resolution that was reached by the identifiers (Feld & Rolauffs, 2005). In addition, a lot of higher-level taxonomical units (tribe, sub-family, family, order, class) often occur in parallel to genera and species, representing mainly redundant taxa. The taxonomical adjustment aims at reducing those redundant information and thus the 'noise' in the biotic dataset. In particular, different levels of determination were found for Mollusca, Oligochaeta, Ephemeroptera, Plecoptera, Coleoptera, Trichoptera, and Diptera. Rare taxa, occurring in < 5 samples (= 3.5 % of all samples) were deleted, leading to a total of 244 taxa mainly on species and generic level.

The adjusted taxa lists were used to calculate 109 metrics that were assigned to four metric groups (Hering et al., 2004a): Composition/abundance (19 metrics), richness/diversity (28), sensitivity/tolerance (23), and functional (39) (Appendix 3). This assignment followed the conceptual framework of metrics (Barbour et al., 1995), however, the author's first metric group 'community structure' was divided here into 'richness/diversity measures' that were mainly related to the number and dominance structure of taxa and 'composition/abundance' metrics, the latter being rather related to the abundance of taxa. 'Individual condition' metrics proposed by Barbour et al., (1995) have not been included, as this information is rarely available for species- and genus-level European taxa (Moog, 1995; Schmedtje & Colling, 1996).

Metrics were analysed for redundancy using Spearman's rank correlation (calculated with STATISTICA 6.1): If metric results correlated with r > 0.800, those metrics that showed the higher overall mean correlation coefficient in the triangular correlation matrix were excluded from further analysis. A similar procedure was applied to the environmental variables; however, the threshold for excluding parameters was set to r > 0.700. These procedures led to a total of 51 environmental variables (Table 4.1, Appendix 2) and 84 metrics (Appendix 3).

Taxa abundances were log-transformed for multivariate analysis (log x + 1) to approximate normal distribution. Proportional metrics (e. g., % individuals) and environmental variables (e. g., % land use) were arc sin $(x/100)^{0.5}$-transformed (Podani, 2000). All other environmental variables except pH and binary variables were log-transformed (Appendix 2).

Table 4.1: Main statistics of multivariate analysis with environmental variables at three spatial scales, taxa (CCA), and metrics (RDA). Significance levels indicated by '**' ($p < 0.01$) or '*' ($p < 0.05$). n. s. = not significant.

		Taxa (CCA)	Metrics (RDA)
Gradient length (DCA)		> 3.81	< 0.76
No. taxa/metrics in all analyses		244	84
No. of samples		144	144
No. of sites		75	75
No. environmental variables			
'Mega'-scale		9	9
Macro-scale		8	8
Meso-scale		21	21
Micro-scale		13	13
Total inertia (variance in the species/metric dataset)		6.800	1.000
'Mega'-scale	sum of all canonical eigenvalues:	1.190	0.213
Axis 1		F = 7.404**	F = 14.706**
Axis 2		F = 7.062**	F = 8.788**
Axis 3		F = 4.419**	F = 7.926**
Axis 4		F = 3.283**	F = 3.898**
Macro-scale	sum of all canonical eigenvalues:	0.543	0.081
Axis 1		F = 3.134**	F = 4.243**
Axis 2		F = 2.415**	F = 4.037**
Axis 3		F = 2.336**	F = 3.213**
Axis 4		F = 2.020**	F = 3.096*
Meso-scale	sum of all canonical eigenvalues:	1.309	0.203
Axis 1		F = 3.892**	F = 11.375**
Axis 2		F = 2.910**	F = 4.990**
Axis 3		F = 2.933**	F = 3.978**
Axis 4		F = 2.211**	F = 4.737**
Micro-scale	sum of all canonical eigenvalues:	0.955	0.141
Axis 1		F = 4.214**	F = 7.044**
Axis 2		F = 2.791**	F = 5.731**
Axis 3		F = 2.269**	F = 3.557*
Axis 4		F = 2.106**	F = 3.077$^{n.\ s.}$
Cumulative % of species/metric-environment relationship of axes 1–4			
'Mega'-scale		81.1	91.1
Macro-scale		70.8	82.8
Meso-scale		40.9	63.3
Micro-scale		51.6	70.9

4.4.4 Statistical analysis

Canonical ordination was used to (i) identify environmental variables that significantly contributed to the multiple regression on taxa/metrics and (ii) to identify hydromorphological gradients and their relation to taxa/metrics. In canonical ordination biotic data are ordered along environmental gradients, which are constrained linear combinations of environmental variables (ter Braak & Smilauer, 2002). For the selection of the appropriate response model, both the taxa and metric datasets were first analysed for gradient length with Detrended Correspondence Analysis (DCA). A gradient length << 3 implies a linear model, whereas values > 4 require the assumption and use of unimodal models (ter Braak & Smilauer, 2002). Based on the DCA results (Table 4.1), Canonical Correspondence Analysis (CCA) was selected for taxa and Redundancy Analysis (RDA) for metrics. All ordinations were run with CANOCO 4.5 (ter Braak & Smilauer, 2002, 2003). In CCA focus was given to 'inter-species distance' with 'biplot scaling'. In RDA focus was set to 'inter-species (metrics) correlations' with 'species scores divided by standard deviation' to display standardized metrics and correlations in the ordination plot instead of co-variances (ter Braak & Smilauer, 2002). Environmental variables at each spatial scale ('mega'-, macro-, meso-, and micro-) were processed separately leading to eight canonical ordinations, four CCA on the taxa and four RDA on the metrics. For both CCA and RDA variables were first checked for collinearity: Variables with a variance inflation factor (VIF) > 8 were omitted, since in several cases a VIF > 8 was related to a Spearman correlation coefficient r > 0.700. The analysis was then re-run with the reduced set of non-collinear environmental variables using CANOCO's forward selection procedure (499 permutations) in order to analyse the contribution of each environmental variable in the multiple regression model. The variable's contribution is expressed by '*Lambda*-A', a measure that shows the additional contribution of a variable to the regression at the time it is incorporated in the model to the variables that are already in the model (ter Braak & Smilauer, 2002).

The first CCA and RDA identified five uncorrelated 'mega'-scaled variables with significant 'conditional effects' to the multiple regression (*Lambda*-A): Catchment size, latitude, longitude, ecoregion, and season. In the following analyses on the macro-, meso- and micro-scales, the five 'mega'-scaled variables were included as covariables in order to partial out their 'natural' influence. Ordination axes' significance was analysed according to the procedure described by ter Braak & Verdonschot (1995).

The linkage of taxa, metrics, and environmental variables was analysed with Indicator Species Analysis (ISA), which is part of the software package PC-Ord 4.3 (McCune & Mefford, 1999). ISA is a procedure to identify the frequency and abundance of species (resp. taxa or metrics) in relation to *a priori*-defined groups of samples (Dufrêne & Legendre, 1997). The indication strength of a taxon/metric is expressed by its Indicator Value (IV) that ranges between 0 and 100. As shown with Formula 4.1 the *IV* increases with (i) the mean relative abundance (*RA*) of a taxon/metric j in the samples i of a group k compared to the overall mean ('specifity' according to Dufrêne & Legendre, 1997) and (ii) the relative frequency (*RF*) of a taxon/metric j of a group k compared to that of the other groups ('fidelity').

$$IV = RA_{jik} \bullet RF_{jk} \bullet 100$$ Formula (4.1)

The significance of *IV* is calculated using the related Monte Carlo test in PC-Ord with 1,000 permutations (McCune & Mefford, 1999). Since ISA does not underlie any constraint referring to the numerical distribution of values to be analysed, it can be processed with original taxa lists containing abundances, which is an advantage over many other analytical methods.

ISA was applied to a total of 34 environmental variables identified by CCA/RDA to be non-collinear and to have significant conditional effects in multiple regression models of CCA/RDA at the level of $p < 0.05$ (Table 4.2). *A priori* groups were individually created according to the range and distribution of the respective environmental variable with five groups at most (Appendix 4). Per definition, binary variables comprised only two groups: item present/yes and item absent/no. For other variables, except for pH, electric conductivity, number of logs within 500 m up- and downstream of sampling site, stream width, and % psammal/psammopelal microhabitat the first group always represented '0', which referred to zero impact of the respective variable (Appendix 4).

The total sum of ISA indications fulfilling the following criteria was counted for taxa and metrics: (i) the respective *IV* was ≥ 30 and (ii) the result was significant at the level of $p < 0.05$. The threshold was set to $IV \geq 30$, since either *RA* or *RF* must reach at least $\sqrt{0.30}$ (= 0.54), which is equivalent to either half of the mean relative abundance or half of the relative frequency of a species to be covered by a certain group (Formula 4.1). In order to evaluate the indicative potential of taxa/metrics significant indications by ISA (SI-ISA) were entered into a matrix of taxa/metrics by environmental variables at the different spatial scales; finally, the total number of SI-ISA per taxon/metric and environmental variable was counted per spatial scale. Ordination plots were created with CanoDraw 4.1 (ter Braak & Smilauer, 2003). For clarity, species (bi)plots show only those taxa with a weight of ≥ 30 % of the maximum weight that occurred in the respective unimodal model (CCA). The species weight represents the impact of a species/taxon on the analysis results (ter Braak & Smilauer, 2002). In linear models (RDA) weights are not calculated. For clarity in RDA (bi)plots the species fit was used. It is a measure for the proportion of variance of the metric values explained by a given ordination subspace, which was set to ≥ 10 %. Other diagrams were produced with Statistica 6.1 (StatSoft 2003).

Table 4.2: Hydromorphological variables with significant conditional effects in forward selection of canonical ordination of taxa (CCA) and metrics (RDA). The environmental variables were also used for ISA (see text).

	CCA (taxa)			RDA (metrics)		
Variable	*Lambda*-A	p	F	*Lambda*-A	p	F
Macro-scaled						
% Pasture (catchment)	0.11	0.002	2.77	0.02	0.004	2.33
% Artificial standing water bodies (catchment)	0.10	0.002	2.33	0.01	0.020	1.86
% Grass-/bushland (catchment)	0.08	0.002	1.84			
% Wetland (catchment)	0.08	0.002	2.07	0.01	0.002	2.62
Lakes in the stream continuum upstream of the sampling site (yes/no)	0.07	0.004	1.58	0.02	0.002	3.07
% Urban settlement/industry (catchment)	0.06	0.002	1.66	0.01	0.008	2.17
Meso-scaled						
pH	0.14	0.002	3.33	0.02	0.002	2.47
Proportion of bank fixation stones (1,000 m reach)	0.12	0.002	2.99	0.05	0.002	8.47
No. of logs (1,000 m reach)	0.09	0.002	2.24	0.01	0.010	1.98
% Grass-/bushland, reeds (floodplain)	0.07	0.002	1.71	0.01	0.008	1.99
% Pasture (floodplain)	0.07	0.002	1.75	0.01	0.020	1.73
Average stream width [m]	0.07	0.002	1.59			
Proportion of shoreline covered with wooded riparian vegetation	0.06	0.016	1.38	0.01	0.012	1.96
% Urban settlement/industry (floodplain)	0.06	0.004	1.66			
Straightening (yes/no)	0.06	0.006	1.43	0.02	0.002	4.30
Electric conductivity [µS/cm]	0.05	0.004	1.52			
Shading at zenith [%]	0.05	0.044	1.24			
Meandering channel (yes/no)	0.05	0.006	1.41			
Sinuate channel (yes/no)	0.05	0.024	1.32			
Stagnation (yes/no)	0.05	0.030	1.31	0.01	0.018	1.83
Presence of standing water bodies in the floodplain (yes/no)	0.05	0.040	1.28	0.01	0.002	2.70
% Crop land (floodplain)	0.04	0.016	1.31	0.01	0.008	1.86
Micro-scaled						
% CPOM (coarse particulate organic matter)	0.12	0.002	2.93	0.01	0.008	1.90
% Submerged macrophytes	0.09	0.002	2.12			
% FPOM (fine particulate organic matter)	0.08	0.002	1.96	0.01	0.004	2.02
% Akal (> 0.2–2 cm)	0.07	0.002	1.85			
% Macrolithal (> 20–40 cm)	0.07	0.002	1.64			
% Mesolithal (> 6–20 cm)	0.07	0.002	1.70	0.02	0.002	2.48
No. of organic substrates	0.07	0.002	1.93	0.01	0.012	1.82
% Xylal (wood)	0.07	0.002	1.74	0.02	0.002	4.24
% Emergent macrophytes	0.06	0.016	1.43	0.01	0.026	1.71
% Psammal/psammopelal (sand and/or mineral mud)	0.06	0.006	1.41	0.02	0.002	3.75
% Microlithal (> 2–6 cm)	0.05	0.036	1.29			
% Living parts of terrestrial plants	0.05	0.006	1.45			

4.5 Results

Generally, the sums of canonical eigenvalues (SCE) were much higher for taxa (from 0.543 at the macro to 1.309 at the meso-scale) than for metrics (from 0.081 at the macro to 0.213 at the 'mega' scale) (Table 4.1). Compared to the total inertia, environmental variables explained 17.5, 8.0, 19.3, and 14.0 % of the species' variance and 21.3, 8.1, 20.3, and 14.1 % of the metric's variance at the 'mega'-, macro-, meso-, and micro-scale, respectively. Hence, 'mega'- and meso-scaled variables accounted for the highest proportion of variance. However, the CCA axes 1–4 barely cover more than 50 % or even less of the cumulative species-environment relationship in case of meso- and micro-scaled environmental variables (Table 4.1). All axes were significant at the level of $p < 0.05$, except for the non-significant axis 4 of the RDA on micro-scales variables ($p = 0.210$).

The canonical ordination of nine 'mega'-scaled variables (Appendix 2) identified five variables with a significant conditional effect (*Lambda*-A): Catchment size, latitude, longitude, ecoregion, and sample season summer. These 'natural' environmental variables were included as covariables for all analyses at the other spatial scales and, hence, were not considered separately in the following.

4.5.1 Taxa analysis (CCA)

4.5.1.1 Macro-scale

Using seven catchment land use variables, the first CCA axis (eigenvalue: 0.134) described a gradient with the end points % grass-/bushland and % pasture, and % crop land (Figure 4.2). Although not included in the analysis the gradient was also explained by % forest, which was negatively correlated with the rather intensive land use categories % crop land (Spearman, $r = -0.822$, $p < 0.001$) and % pasture (Spearman, $r = -0.731$, $p < 0.001$). The second axis (eigenvalue: 0.101) was related to the proportion of artificial water bodies in the catchment (e. g., fish ponds) and the presence of lakes in the stream continuum upstream of the sampled site. The third and fourth CCA axes (eigenvalues: 0.096 and 0,081, not shown in Figure 4.2) were related to % wetland and % urban settlement/industry, respectively, whereas both show also relation to lakes in the stream continuum upstream.

Gammarus pulex, *G. fossarum*, (Crustacea), *Nemoura* sp. (Plecoptera), *Dicranota* sp., *Polypedilum* sp., and *Simulium* sp. (Diptera) revealed a positive relation to % grass-/bushland. The other end of the land use gradient along axis 1 was characterized by *Baetis rhodani*, *Ephemera danica* (Ephemeroptera), and *Limnius volckmari* (Coleoptera). However, *Baetis rhodani* and *Limnius volckmari* also showed a relationship to % wetland in the catchment.

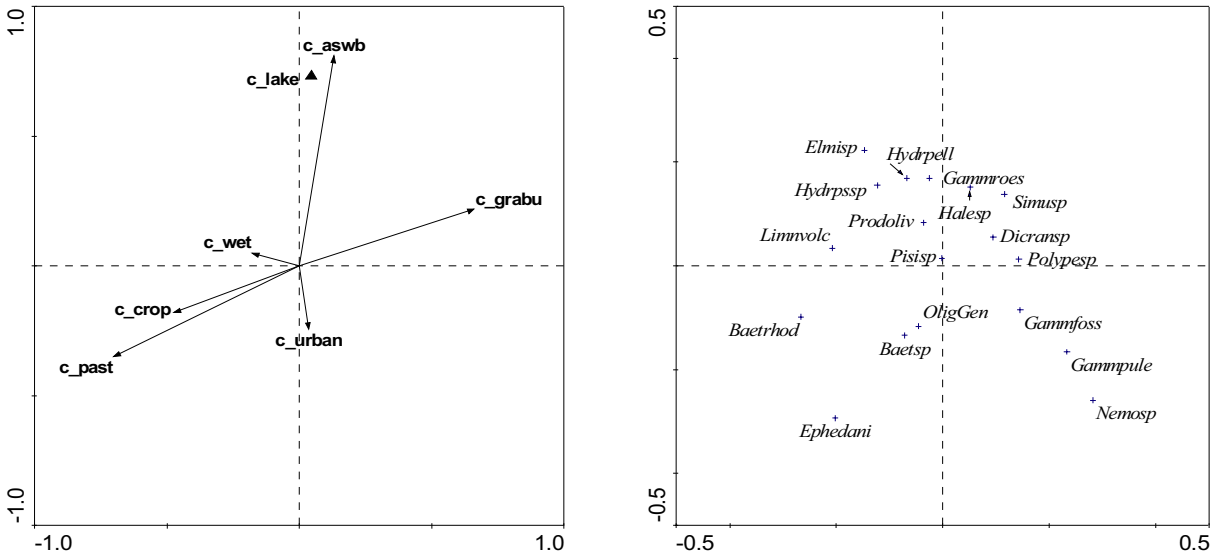

Figure 4.2: Partial CCA (axis 1 *vs.* axis 2) of 244 taxa and seven non-collinear macro-scale catchment land use categories [%]: c_aswb = artificial standing water bodies; c_lake = lakes; c_grabu = grass-/bushland; c_wet = wetland; c_crop = crop (tilled) land; c_urban = urban settlement/industry; c_past = pasture. Taxon codes: Pisisp = *Pisidium* sp. ; OligGen = Oligochaeta Gen. sp.; Gammfoss = *Gammarus fossarum*; Gammpule = *Gammarus pulex*; Gammroes = *Gammarus roeselii*; Baetrhod = *Baetis rhodani*; Baetsp = *Baetis* sp.; Ephedani = *Ephemera danica*; Nemosp = *Nemoura* sp.; Hydrpell = *Hydropsyche pellucidula*; Hydrpssp = *Hydropsyche* sp.; Halesp = *Halesus* sp.; Elmisp = *Elmis* sp. ; Limnvolc = *Limnius volckmari*; Polypesp = *Polypedilum* sp.; Prodoliv = *Prodiamesa olivacea*; Dicransp = *Dicranota* sp.; Simusp = *Simulium* sp.

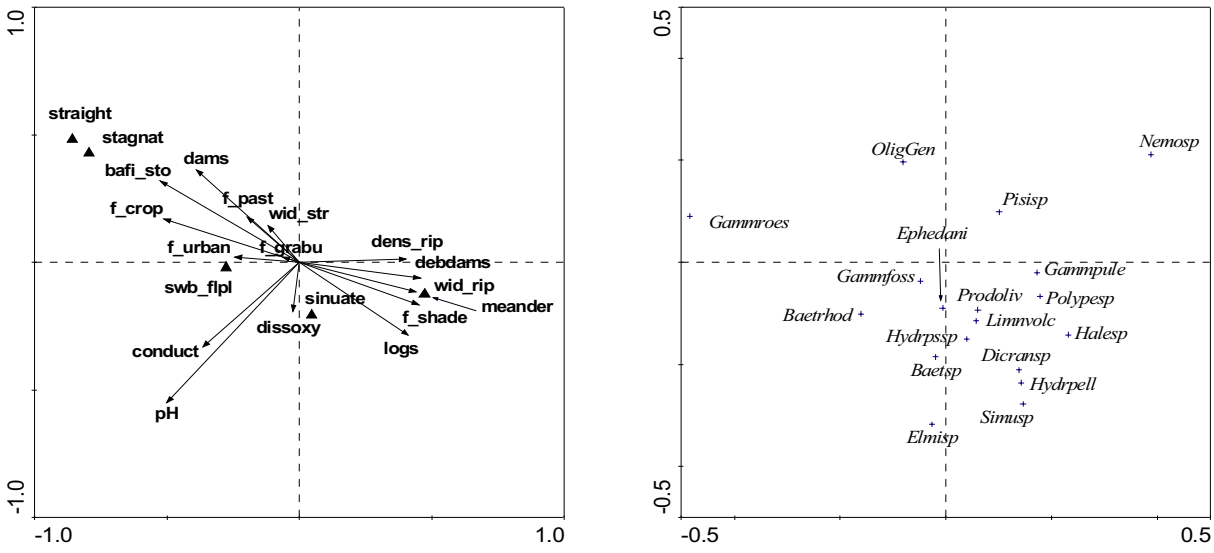

Figure 4.3: Partial CCA (axis 1 *vs.* axis 2) of 244 taxa and 20 non-collinear meso-scale environmental variables. Floodplain land use categories [%]: f_grabu = grass-/bushland; f_wet = wetland; f_crop = crop (tilled) land; f_urban = urban settlement/industry; f_past = pasture. Hydromorphological variables: straight = straightening; stagnat = stagnation; bafi_sto = bank fixation stones (rip-rap); wid_str = average stream width; dens_rip = density of riparian vegetation; debdams = debris dams; wid_rip = width of riparian vegetation; f_shade = shading; dissoxy = dissolved oxygen; conduct = electric conductivity; swb_flpl = standing water bodies in the floodplain. For taxon codes, see Figure 4.2.

4.5.1.2 Meso-scale

The 20 non-collinear meso-scaled environmental variables were ordered along an impact gradient that was predominantly given by the first ordination axis (Figure 4.3; eigenvalue: 0.184). Variables on the left indicated human impact through flow regulation, streambank modification, or agricultural land use. The variables on the opposite represented natural habitat conditions: meandering stream course, riparian vegetation, and the proportion of large wood on the stream bottom. Axis 2 (eigenvalue: 0.134) also explained part of the hydromorphological gradient, but it was also related to pH and electric conductivity.

Several species were related to the hydromorphological gradient along axis 1, of which most were located at the 'natural end': *Pisidium* sp. (Bivalvia), *Gammarus pulex*, *Nemoura* sp., *Elmis* sp. (Coleoptera), *Halesus* sp., *Hydropsyche pellucidula* (Trichoptera), *Dicranota* sp., *Polypedilum* sp., *Prodiamesa olivacea*, and *Simulium* sp. (Diptera) and few at the 'impacted end': Oligochaeta, *Gammarus roeselii* (Crustacea), and *Baetis rhodani*.

4.5.1.3 Micro-scale

The 14 micro-scaled substrate (habitat) categories ordered along a gradient mainly given by the first ordination axis (eigenvalue: 0.189). The gradient was characterized by (mineral) hard substrates (boulder, cobble, pebble and gravel) on the left hand and soft organic substrates (submerged and emerged macrophytes), and coarse particulate organic matter on the right hand side of Figure 4.4. Soft substrates also include (mineral) sand/silt. Axes 2–4 (eigenvalues: 0.122, 0.097, and 0.088, respectively) were related to the proportion of macrophytes on the stream bottom, the proportion of large wood (twigs and branches > 10 cm diameter), and fine particulate organic matter. Thus, while axis 1 separated mineral hard substrates from soft and organic substrates, axes 2–4 (only the two main axes shown in Figure 4.4) differentiated between organic substrates that represented rather 'natural' conditions (e. g., xylal, FPOM) and those that indicated a certain degree of human impact, in particular the proportion of submerged and emerged macrophytes and organic mud on the stream bottom.. The latter, if encountered in large proportions at a site, was representative for lentic (stagnated) flow conditions and a lack of shading promoting the development of large stands of, for example, arrowhead *Sagittaria sagittifolia* and yellow water-lily *Nuphar lutea*. The human impact can also be derived from axis 1 if we presume that significant proportions of mineral hard substrates may indicate the presence of rip-rap and, thus, a certain degree of human impact in sand-bottom lowland streams.

Oligochaeta, *Gammarus roeseli*, *G. fossarum*, *Baetis rhodani*, *Elmis* sp., and *Limnius volckmari* were related to sites with relatively large proportions of mineral hard substrates. Oligochaeta, *Gammarus roeseli*, and *Baetis rhodani* also showed a relationship to bank fixation with stones at the reach scale (Figure 4.3, axis 1). By contrast, the organic (soft) substrates were correlated with *Gammarus pulex*, *Nemoura* sp., *Halesus* sp., *Hydropsyche pellucidula*, *Polypedilum* sp., and *Simulium* sp.

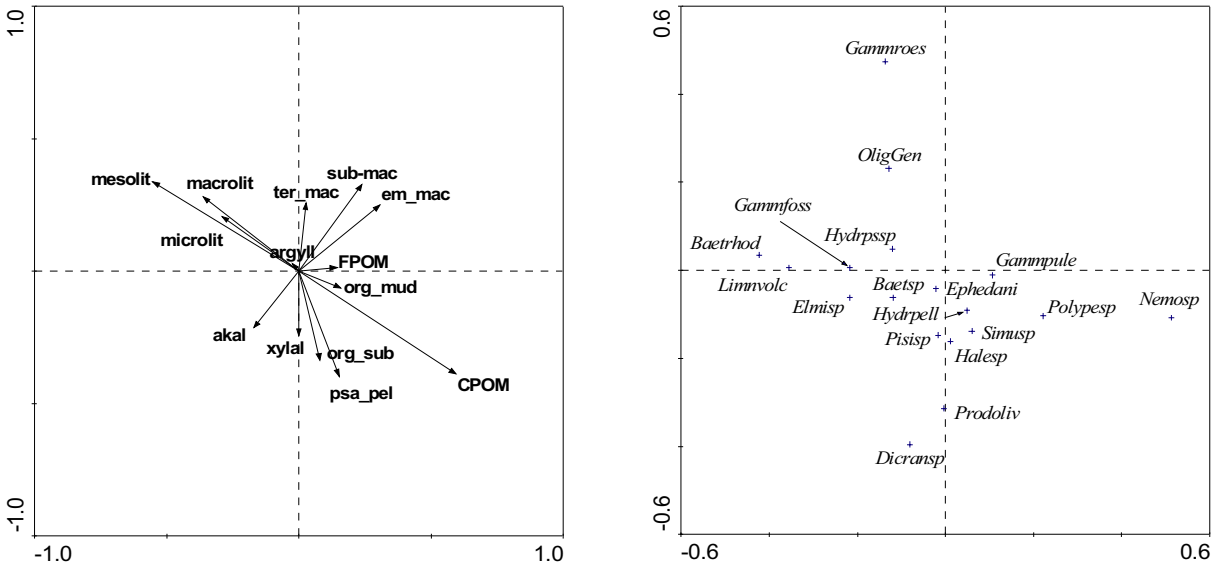

Figure 4.4: Partial CCA (axis 1 *vs.* axis 2) of 244 taxa and 14 non-collinear substrate (habitat) categories [%]: macrolit = macrolithal; mesolit = mesolithal; microlit = microlithal; argyll = argyllal; psa_pel = psammal/psammopelal; ter_mac = living parts of terrestrial plants; sub_mac = submerged macrophytes; em_mac = emergent macrophytes; FPOM = fine particulate organic matter; CPOM = coarse particulate organic matter; org_mud = organic mud; org_sub = no. of organic substrates. For taxon codes, see Figure 4.2.

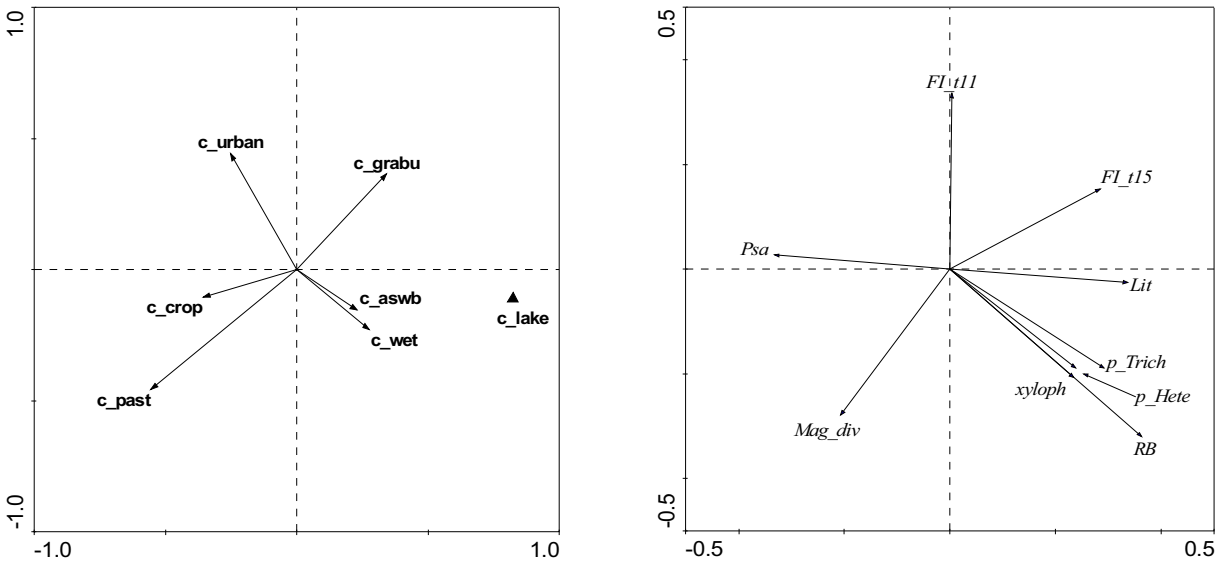

Figure 4.5: Partial RDA (axis 1 *vs.* axis 2) of 84 metrics and seven non-collinear macro-scale catchment land use categories (for land use codes, see Figure 4.2). Metric codes: FI_t11 = German Fauna Index type 11; FI_t15 = German Fauna Index type 15; Psa = psammal preferences; Lit = lithal preferences; xyloph = wood preferences; metrhit = metarhithral preferences; actfilt = active filterer; pasfilt = passive filterer; Mag_div = Margalef diversity; RB = rheobiont; p_Trich = % individuals Trichoptera; p_Hete = % individuals Heteroptera.

4.5.2 Metric analysis (RDA)

4.5.2.1 Macro-scale

Although the RDA revealed two possible land use gradients (Figure 4.5) neither was dominant and related to a single ordination axis. This was indicated by almost equal eigenvalues for the ordination axes. Moreover, the sum of canonical eigenvalues was the lowest observed (Table 4.1). Along axis 1 the intensive land use categories (left) were separated from the 'natural' proportion of artificial water bodies in the catchment (right) (Figure 4.5). Axis 1 *vs.* 2 showed two potential gradients: one represented by % crop land and % pasture at the 'impacted end' and % grass-/bushland at the other, and another gradient located almost perpendicular to the first, characterized by % urban settlement/industry at the 'impacted' and the proportion of artificial water bodies in the catchment and % wetland at the 'natural' end.

Regarding the metrics % urban settlement/industry was shown to be negatively related to rheobiont and xylophagous taxa, but also the relative abundance of Heteroptera and Trichoptera (Figure 4.5). The impact of % crop land and % pasture was positively related to psammal preferences, active filterers, and Margalef's diversity, whereas a negative relation was found for lithal and metarhithral preferences, rheobiont taxa, passive filterers, the relative abundance of Heteroptera and Trichoptera, and the German Fauna Index.

4.5.2.2 Meso-scale

The 17 reach-related variables were predominantly ordered along axis 1 (eigenvalue: 0.069) and showed a clear gradient of hydromorphological impact (Figure 4.6). Variables indicating 'natural' conditions were located on the left as opposed to those variables connected with degradation, which could be found on the right. Axis 1 accounted for 34 % of the total variance.

The relation of metrics to the gradient revealed in particular richness/diversity and sensitivity/tolerance measures as indicative for natural conditions (left hand side in Figure 4.6). On the opposite, several saprobic indices and functional metrics were located. In summary, the 'impacted' end was correlated with slow flow conditions, supporting the growth of large macrophyte stands (Phytal) and, as a result, the accumulation of organic mud (Pelal), the latter directly promoting gathering collectors.

4.5.2.3 Micro-scale

The ordination of 14 site-related habitat variables again revealed a gradient along axis 1 (eigenvalue: 0.043; Figure 4.7). Xylal (wood), CPOM, and FPOM represented the left 'natural' end, whereas macro- and mesolithal, and aquatic macrophytes marked the 'impacted' end. Axis 2 (eigenvalue: 0.031) was related to the proportion of sand (psammal/psammopelal) and the number of organic substrates.

The 'natural' end of the substrate gradient along axis 1 was related to metrics representing the number of sensitive taxa (German Fauna Index type 15 and the respective number of indicator taxa, total number taxa, Trichoptera taxa and abundance, BMWP, and proportion of individuals with rheophilic preferences). By contrast, human impact resulting in higher pro-

portions of lithal and macrophytes on the stream bottom was positively related to mainly functional metrics (Figure 4.7), such as the proportion of individuals with preferences for epirhithral, hypocrenal, hypopotamal, and littoral zones, the feeding types grazer/scraper and gathering collector, and the proportion of individuals living on macrophytes. The number of Ephemeroptera-Plecoptera-Trichoptera taxa and Diptera taxa, and Margalef's diversity were negatively related to macro-, mesolithal, and submerged macrophytes (not shown in Figure 4.7).

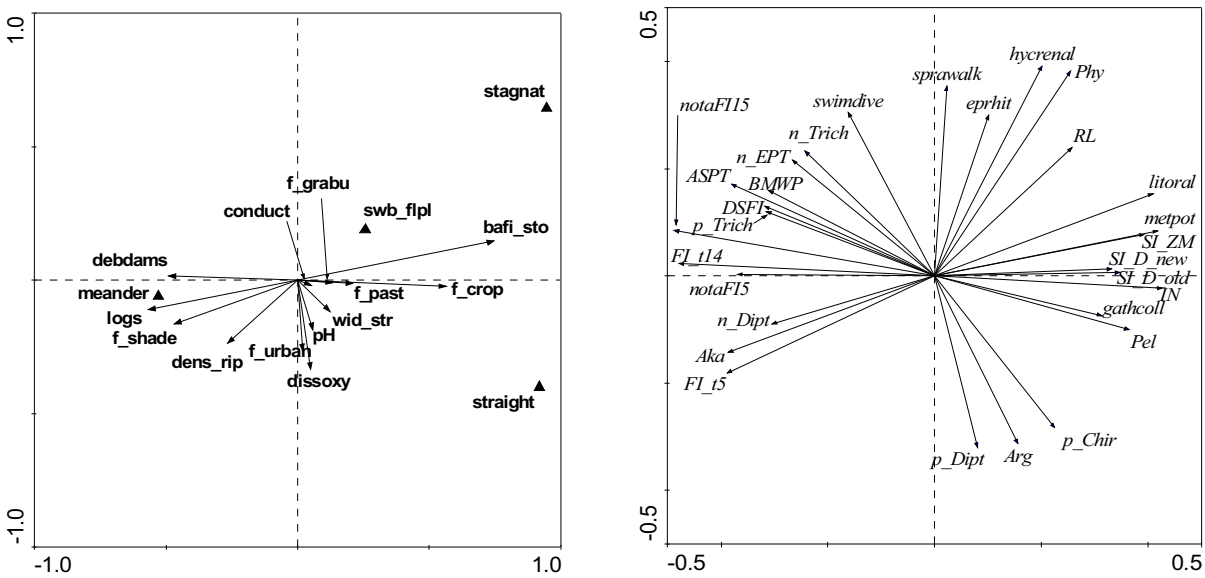

Figure 4.6: Partial RDA (axis 1 *vs.* axis 2) of 84 metrics and 17 non-collinear meso-scale environmental variables (for variable codes, see Figure 4.3). Metric codes: FI_t5, t11, and t14 = German Fauna Indices types 5, 11 and 14; notaFI5, 11, and 15 = number of indicator taxa German Fauna Indices types 5, 11, and 15; BMWP = British Monitoring Working Party (index); ASPT = Average Score per Taxon; DSFI = Danish Stream Fauna Index; SI_ZM = Saprobic index after Zelinka & Marvan; SI_D_old = German Saprobic Index; SI_D_new = German Saprobic Index revised; SI_NL = Dutch Saprobic Index; n_Trich = number of taxa Trichoptera; n_EPT = number of taxa Ephemeroptera-Plecoptera-Trichoptera; n_Dipt = number of taxa Diptera; p_Plec = % individuals Plecoptera; p_Cole = % individuals Coleoptera; p_Trich = % individuals Trichoptera; p_EPT = % individuals Ephemeroptera-Plecoptera-Trichoptera; p_Dipt = % individuals Diptera; p_Chir = % individuals Chironomidae; Aka = akal preferences; Arg = argyllal preferences; Phy = phytal preferences; Pel = pelal preferences; hycrenal = hypocrenal preferences; eprhit = epirhithral preferences; metpot = metapotamal preferences; RL = rheo- to limnophilic current preferences; litoral = littoral preferences; IN = indifferent current preferences; swimdive = swimmer/diver; sprawalk = sprawler/walker; gathcoll = gatherer/collector.

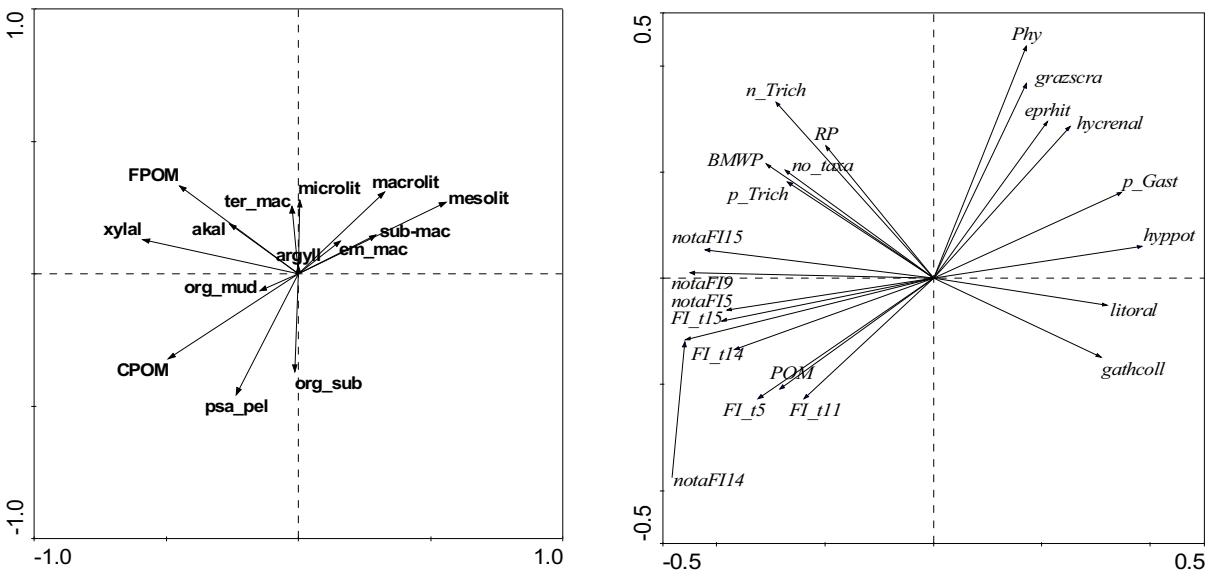

Figure 4.7: Partial RDA (axis 1 *vs.* axis 2) of 84 metrics and 14 non-collinear substrate (habitat) categories (for habitat codes, see Figure 4.4; for metric codes, see Figure 4.6): FI_t15 = German Fauna Index type 15; notaFI9 and 14 = number of indicator taxa German Fauna Indices types 9 and 14; Mag_div = Margalef diversity; no_taxa = total number of taxa; p_Gast = % individuals Gastropoda; POM = preferences for particulate organic material; hyppot = hypopotamal preferences; RP = rheophilic current preferences; grazscra = grazer/scraper.

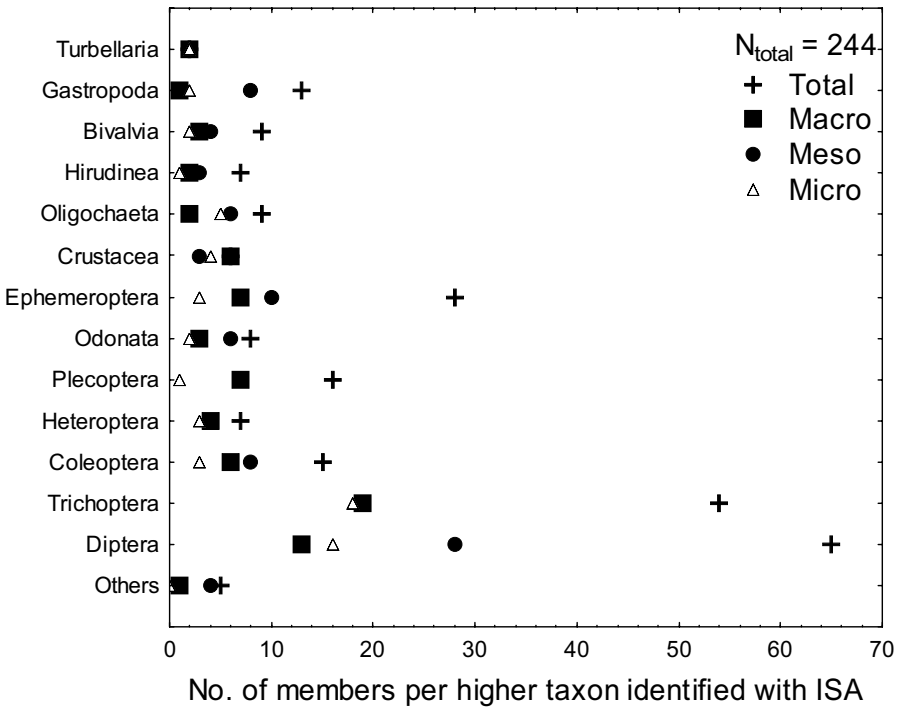

Figure 4.8: Number of taxa identified with Indicator Species Analysis (ISA) in relation to the total number of taxa per taxonomical unit (order/class) and spatial scale.

4.5.3 Indicator Species Analysis (ISA)

In total, 34 non-collinear and non-correlating environmental variables remained for the ISA (Table 4.2).

4.5.3.1 Indicative potential of taxa at different spatial scales

Among the total of 244 taxa tested with ISA 76 were indicative with a significant *IV* > 30 (SI-ISA) at the macro-, 112 at the meso-, and 62 at the micro-scale. When species/genera were grouped into higher taxonomical entities Diptera and Trichoptera were dominant at all spatial scales (Figure 4.8) followed by Ephemeroptera, Plecoptera and Coleoptera (EPCTD taxa). Trichoptera dominated at the macro and micro-scales (25.0 and 25.8 %), Diptera at the meso-scale (25.0 %). The EPCTD taxa revealed most SI-ISA at the macro-scale, whereas Gastropoda entered the top-five taxa at the meso-scale, and Oligochaeta and Crustacea at the micro-scale (Figure 4.9, pies). However, the indicative potential of a taxonomical unit was related to the total number of the respective species/genera entering the analysis. Therefore, the deviation of SI-ISA of a taxonomical unit in relation to its proportion in the total taxa list was also calculated (Figure 4.9, bar plots).

Table 4.3: Top taxa with ≥ 5 significant indications in Indicator Species Analysis (ISA).

	No of SI-ISA		
Taxon	Macro	Meso	Micro
Bithynia tentaculata (Linnaeus, 1758)	1	5	1
Pisidium sp.	1	2	2
Oligochaeta Gen. sp.	2	6	4
Asellus aquaticus (Linnaeus, 1758)	3	2	
Gammarus fossarum Koch in Panzer, 1836	1	4	3
Gammarus roeselii (Gervais, 1835)	1	6	1
Baetis rhodani Pictet, 1843–1845	2	5	
Leuctra sp.	3	3	
Aphelocheirus aestivalis (Fabricius, 1794)	3	2	1
Velia sp.	1	3	2
Hydropsyche pellucidula (Curtis, 1834)	1	4	4
Halesus sp.		3	3
Elmis aenea (Müller, 1806)	1	1	3
Limnius volckmari (Panzer, 1793)		5	2
Atherix sp.	2	4	1
Ceratopogonidae Gen. sp.	1	5	1
Microtendipes pedellus (de Geer, 1776)	1	5	2
Polypedilum sp.	2	6	2
Empididae Gen. sp.	2	4	2
Simulium sp.	1	5	5

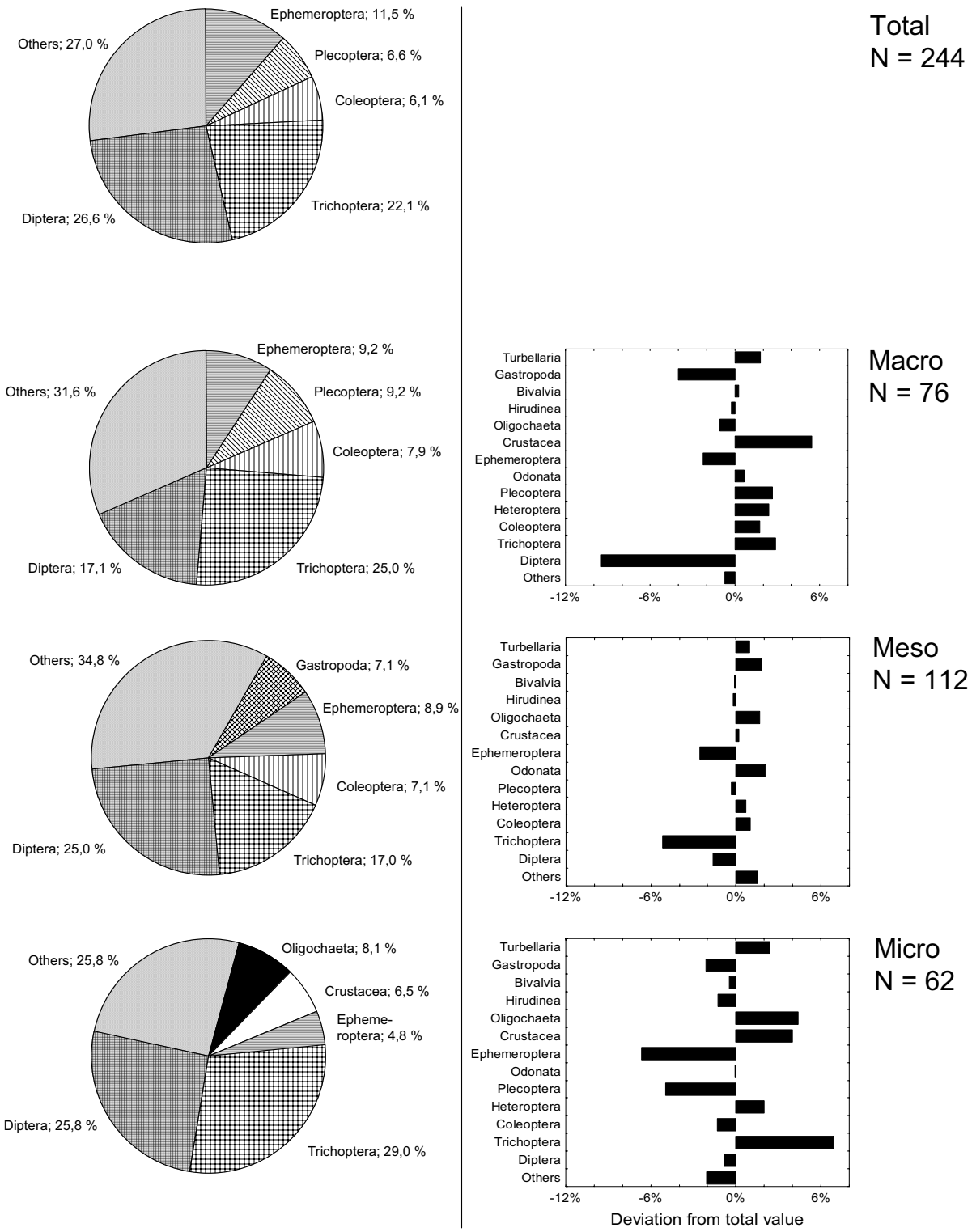

Figure 4.9: Proportion of the five dominant taxonomical units for the total taxa dataset and identified with Indicator Species Analysis (ISA) per spatial scale (pie plots). Bar plots show the deviation of the taxa number identified with ISA to the total number per taxonomical unit that was entered the analysis.

SI-ISA of Crustacea at the macro-scale were 5.4 % higher than their proportional representation in the taxa list, those of Plecoptera, Heteroptera, and Trichoptera around 3 % higher, and those of Diptera much lower (-9.5 %). At the meso-scale, Trichoptera and Ephemeroptera were less indicative than expected and at the micro-scale, Trichoptera, Oligochaeta, and Crustacea had more, Ephemeroptera and Plecoptera less indications compared to their relative proportions in the taxa list. At the species/genus level several indicator taxa were identified by multiple SI-ISA (Table 4.3). Most indications were observed at the meso-scale, which is in accordance with the results of the CCA (Table 4.1). Oligochaeta Gen. sp. was the most indicative taxon with 12 indications followed by the Diptera taxa *Simulium* sp. (11) and *Polypedilum* sp. (10).

4.5.3.2 Indicative potential of metrics at different spatial scales

Functional metrics (e. g., feeding types, habitat preferences) covered about half of the total number of the 84 metrics (Figure 4.10), whereas only 15.5 % belonged to the sensitivity/tolerance metric group. Most indications were observed at the meso-scale (77), followed by the macro (64) and micro-scale (56). Considering the total number of metrics in the analysis, the overall mean SI-ISA for metrics was much higher than for taxa (78.2 *vs.* 34.2 %); thus, metrics were in general much more related to the environmental variables targeted in this Chapter. Richness/diversity measures showed the greatest differences between spatial scales. Their indicative potential was higher at the micro-scale and lower at the macro and meso-scales. In contrast, composition/abundance and sensitivity/tolerance metrics were slightly higher indicative at the macro and meso-scales but worse at the micro-scale. Functional measures had a high indicative potential at all spatial scales and the highest at the meso-scale. The higher overall mean SI-ISA for metrics is reflected by Table 4.4 that comprises 47 metrics with multiple (≥ 5) SI-ISA compared to only 22 taxa listed in Table 4.3. The most indications were observed for % individuals with argyllal (12) and crenal (11) preferences, a semi-sessil locomotion type (10), xylophagous feeding (10), and the number of indicator taxa of the German Fauna Index type 15 (11). Taxa number of Chironomidae (8), Plecoptera (8), EPT (7), and Oligochaeta (7) represent high ranking composition/abundance measures, the German Fauna Indices type 14 (8) and type 15 (8) the highest scoring sensitivity/tolerance metrics.

Table 4.4: Top metrics with ≥ 5 significant indications in Indicator Species Analysis (ISA). Metric group abbreviations: C/A = composition/abundance; F = function; R/D = richness/diversity; S/T = sensitive/tolerant taxa.

		No. of SI-ISA		
Metric name	Metric group	Macro	Meso	Micro
Chironomidae [%]	C/A	2	4	2
Plecoptera [%]	C/A	2	5	1
EPT (Ephemeroptera, Plecoptera, Trichoptera) [%]	C/A	3	4	0
Oligochaeta [%]	C/A	0	4	3
Hirudinea [%]	C/A	1	2	3

Table 4.4, continued.

Metric name	Metric group	No. of SI-ISA Macro	Meso	Micro
Bivalvia [%]	C/A	1	3	1
Coleoptera [%]	C/A	1	2	2
Ephemeroptera [%]	C/A	0	2	3
Gastropoda [%]	C/A	2	3	0
Odonata [%]	C/A	1	2	2
Trichoptera [%]	C/A	2	2	1
Turbellaria [%]	C/A	1	1	3
Argyllal preferences (silt, loam, clay) [%]	F	3	5	4
Crenal preferences [%]	F	1	6	4
(Semi-)sessil [%]	F	2	4	4
Xylophagous taxa [%]	F	1	6	3
Active filterers [%]	F	4	2	3
Burrowing/boring [%]	F	4	3	2
Hypopotamal preferences [%]	F	3	3	3
Indifferent current preferences [%]	F	2	6	1
Littoral preferences [%]	F	2	5	2
Pelal preferences (mud) [%]	F	3	4	2
Hypocrenal preferences [%]	F	2	3	3
Passive filterers [%]	F	1	5	2
Epirhithral preferences [%]	F	1	2	4
Particulate Organic Matter preferences (CPOM, FPOM) [%]	F	0	4	3
Akal preferences (fine to medium gravel) [%]	F	1	4	1
Metapotamal preferences [%]	F	1	4	1
Metarhithral preferences [%]	F	1	3	2
RETI (Rhithron Feeding Type Index) (Schweder, 1992; Podraza et al., 2000)	F	2	3	1
Shredders [%]	F	3	2	1
Gatherers/Collectors [%]	F	0	2	3
Grazers/scrapers [%]	F	1	1	3
Lithal preferences (coarse gravel, stones, boulders) [%]	F	3	2	0
Phytal preferences (mosses, macrophytes, parts of terrestrial plants) [%]	F	1	3	1
Rheobiont individuals [%]	F	2	2	1
Sprawling/walking [%]	F	3	1	1
Swimming/diving [%]	F	2	2	1
German Fauna Index type 15: No. of indicator taxa (Lorenz et al., 2004b)	R/D	2	4	5
German Fauna Index type 14: No. of indicator taxa (Lorenz et al., 2004b)	R/D	1	5	2
German Fauna Index type 9: No. of indicator taxa (Lorenz et al., 2004b)	R/D	1	3	4
No. taxa Oligochaeta	R/D	2	1	2
German Fauna Index type 14 (Lorenz et al., 2004b)	S/T	2	4	2
German Fauna Index type 15 (Lorenz et al., 2004b)	S/T	2	4	2
ASPT (Average Score per Taxon) (Armitage et al., 1983)	S/T	3	2	1
Dutch Saprobic Index (Lorenz et al., 2004b)	S/T	2	3	1
German Fauna Index type 5 (Lorenz et al., 2004b)	S/T	1	3	2

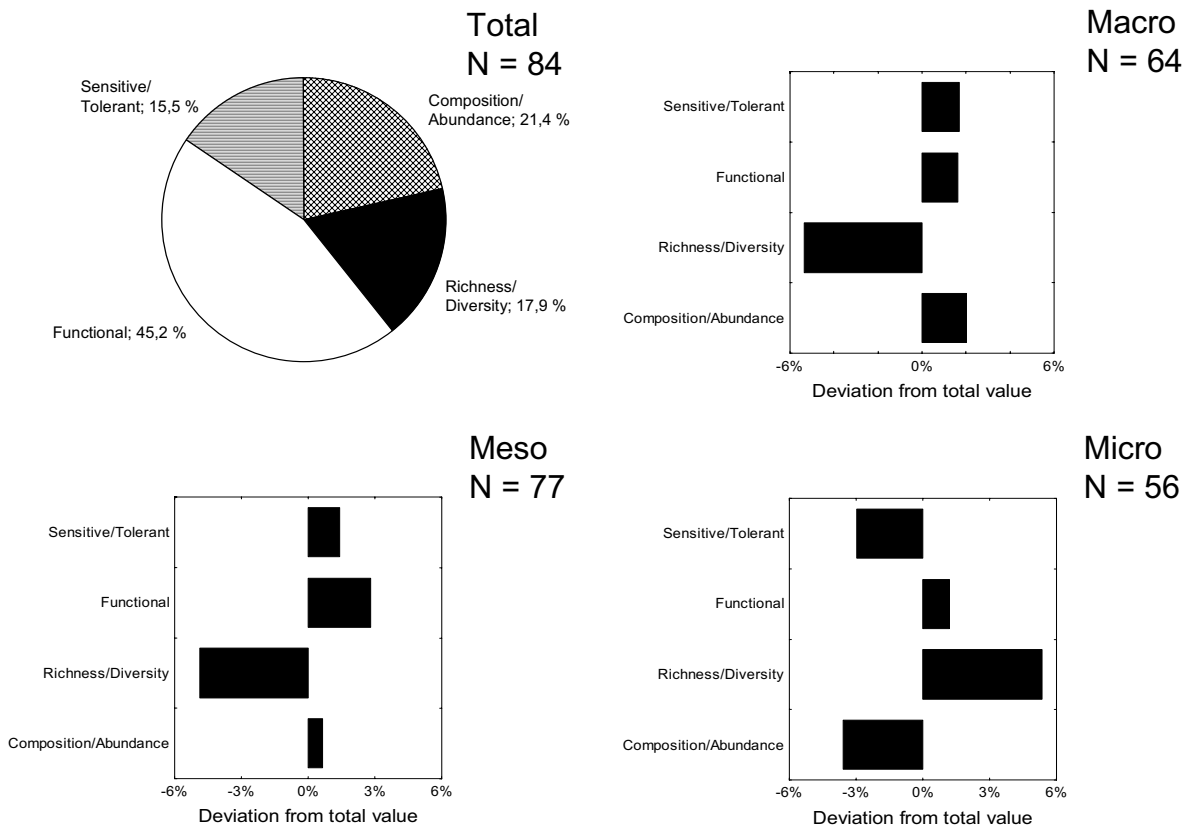

Figure 4.10: Proportion of total metrics per metric group (pie plot) and deviation of the number of metrics identified with Indicator Species Analysis (ISA) for metric groups and spatial scales to the total number and proportion (bar plots).

4.5.3.3 Indication of hydromorphological variables by taxa and metrics at different spatial scales

In general, taxa and metrics with most SI-ISA were related to different environmental variables showing that both biotic categories may provide a different internal relation to the environment. At the macro-scale, the proportion of wetland (22) and grass-/bushland (19), the presence of lakes in the river continuum upstream of the sampling site (37), and the proportion of urban settlement/industry (33) were most often related to taxa and metrics, respectively. Regarding the meso-scaled reach variables, pH (35), mean stream width (29), and the proportion of crop land in the floodplain (25) had most SI-ISA by taxa. At the same scale metrics showed the highest relation to straightening (49), bank fixation with rip-rap (32), and stagnation (31). Among the microhabitats the proportion of emergent macrophytes (16) and living parts of terrestrial plants (15) were most often related to taxa, the proportion of xylal (29) and CPOM (19) to metrics.

Table 4.5: Top hydromorphological variables with ≥ 10 significant indications for taxa or metrics in Indicator Species Analysis (ISA). n. s. = conditional effect of variable in CCA/RDA not significant at $p < 0.05$.

	No. of SI-ISA	
Environmental variable	Taxa (CCA)	Metrics (RDA)
Macro-scaled		
Lakes in the river continuum upstream (y/n)	6	37
Land use catchment: Artificial standing water body [%]	18	20
Land use catchment: Grass-/bush land [%]	19	n. s.
Land use catchment: Pasture [%]	18	10
Land use catchment: Urban settlement/industry [%]	10	33
Land use catchment: Wetland [%]	22	11
Meso-scaled		
Bank fixation: Stones (rip-rap) [%]	12	32
Land use reach: Crop land [%]	25	18
Land use reach: Grass-/bush land [%]	14	7
Land use reach: Pasture [%]	10	8
Mean stream width [m]	29	n. s.
Meandering stream course (y/n)	11	n. s.
No. of logs (reach)	9	14
pH	35	14
Presence of standing water bodies in the floodplain (y/n)	11	24
Proportion of wooded riparian vegetation [%]	13	13
Stagnation (y/n)	17	31
Straightening (reach) (y/n)	14	49
Micro-scaled		
CPOM [%]	0	19
Emergent macrophytes [%]	16	8
FPOM [%]	10	13
Living parts of terrestrial plants [%]	15	n. s.
Mesolithal [%]	13	18
No. of organic substrates	11	16
Psammal/psammopelal [%]	5	11
Xylal [%]	7	29

4.6 Discussion

Based on canonical ordination, the catchment or even 'supra-catchment' environmental variables explained a large proportion of variance in both taxa and metrics data. Catchment size, latitude, longitude, altitude, ecoregion, and sampling season accounted for 17.5 % of the total taxa inertia and 21.3 % of the total metric inertia. This high descriptive potential of 'mega'-scaled variables is not surprising, since ecoregion (e. g., Corkum, 1992; Johnson & Goedkoop, 2002), catchment size (e. g., Vannote et al., 1980), and sampling season (e. g., Furse, Wright & Armitage, 1984) are known to control macroinvertebrate communities of streams and rivers. However, if a large geographical range is considered, as it was the case for the current analysis, in particular catchment size and ecoregion were proved good and important descriptors. Both had high conditional effects in forward selection of CCA and

RDA and both were also identified in Chapter 2 to influence the benthic community at large scales. The large proportion of variability explained by catchment size and ecoregion, however, may also imply a disadvantage, since the natural variability may have confounded the results. This cannot be completely ruled out, but the previous analysis addressing the identification of hydromorphological degradation (Chapter 3) identified similar gradients at different spatial scales from ecoregion to a single stream type. Thus, the gradients derived here from direct gradient analysis seem in general to be valid. The low conditional effects observed for season and, hence, its weak descriptive power is consistent with the results of Chapter 2, where season was identified by analysis of similarity (ANOSIM) to be a rather weak descriptor for the benthic community's variability at large scales. Yet, it is opposed to the findings of Townsend et al. (1997), who in general stated the temporal variability to be as important or even more important than the spatial variability in streams. The role of the seasonal variability did not change if the data covered a smaller area, for example, a certain stream type as shown in Chapter 6. Because of the contradictory findings, season and the other 'mega' scaled variables with a *significant* conditional effect were used here as covariables to detect and partial out their influence on subsequent ordinations targeting lower spatial scales. The focus was laid on the impact of human-induced hydromorphological alteration rather than the role of 'natural' environmental descriptors.

Environmental variables explained more variance in the metric data than in the taxa data at all spatial scales (Table 4.1). This may partly be related to the number of parameters that were entered in the analysis, which were about three times as high for taxa as for metrics and which are thought to be positively related to the biological variation. Moreover, as shown by the gradient length of the first DCA axis, the species turnover (> 3.81) was five times the metric turnover (< 0.76). Metrics derived from benthic invertebrates appeared to be better suited to account for the impact of environmental variables, because their applicability over wide geographical areas (regions) was more constantly expressed by a lower turnover. As metrics, such as functional guilds or diversity measures summarize the functional aspect of a community, this usually leads to a lower number of metrics compared to the number of taxa needed to describe a community or its relation to the environment. Another advantage of metrics may be the lower turnover of the metric structure between seasons. While the occurrence of larvae of several aquatic insect orders (e. g., Plecoptera, Ephemeroptera, Trichoptera) may show a strong seasonal patterns, Haybach et al. (2004) recently reported the trait structure of the communities within several large rivers to be independent of the sampled season. They found, for example, samples of the northern Upper-Rhine to be almost constant between seasons in terms of their structure of 14 biological traits derived from the benthic invertebrates.

4.6.1 Ordination of environmental variables

Canonical ordination is a 'direct' gradient analysis that aims at the detection of the main pattern in the relationship between the species and the observed environmental variables (Jongman et al., 1995). The gradients are usually ordered along the ordination axes. Several gradients were revealed in the ordination plots (Figure 4.2–4.7) that can mainly be described as hydromorphological and/or land use gradients, since almost exclusively variables

related to human impact were used. The land use gradient at the macro-scale predominantly separated the intensive agricultural utilization (pasture, crop land) from grass-/bushland. While the proportion of pasture and (tilled) crop land marked the degree of human impact on the aquatic macroinvertebrates, the proportion of natural grass-/bushland (and forest) indicated the naturalness of a site's catchment in the Central European Lowlands. Although forest area was not used in canonical ordinations due to its high collinearity, its indicative value was obvious from a high negative correlation with the proportion of pasture and crop land in the catchment. The results support the relation of catchment land use categories presented in Chapter 3 and also stated by Fitzpatrick et al. (2001) for row-crop agriculture-dominated lowland rivers in Wisconsin, USA. The proportion of wetland and urban settlement/industry described an additional gradient and both variables were negatively related (Spearman rank correlation; $r = -0.317$; $p < 0.001$). It seems that at least two land use gradients control the macroinvertebrates at the macro-scale: an agricultural gradient and an urban gradient. Comparing CCA and RDA the urban gradient was marginally better related to metrics than to taxa, however both identified the same agricultural gradient.

The reach scale variables described mainly a hydromorphological gradient along the first axis, which was underlined by the highest eigenvalues observed for meso-scaled environmental variables in both CCA and RDA. Consequently, bank and flow modification, and the degradation of the riparian area had the strongest impact on the aquatic macroinvertebrates and the derived metrics. This is consistent with the findings presented in Chapter 3, too. Accordingly, floodplain land use, bank and flow modification, riparian buffer vegetation, shading, and large wood were found to be related to the predominant environmental gradient in a lowland dataset. It implies that the reach scale is of major importance in the management of the investigated stream types and it is consistent with Griffith et al. (2001), who reported the reach scale as most appropriate for stream management in the mineral belt of the Southern Rockies Ecoregion in Colorado. The riparian disturbance as a result of intensive (reach) agricultural land use was best associated with the hydromorphological gradient in their study.

The two substrate gradients at the micro-scale may be related to hydromorphological degradation. The first 'hard-soft substrate' gradient separates sites dominated by mineral hard substrates, such as cobbles and pebbles, from sites dominated by soft substrates, such as CPOM or sand. The inherent degradation of this gradient becomes evident if referred to the investigated stream types: The study predominantly covered sand-bottom streams and rivers (mean proportion sand : 66 %; median: 80 %), where the presence of cobbles or even boulders was usually related to bank fixation with rip-rap. Hence, the reach-related variable 'bank fixation' controlled the substrate composition at a site, which may be an example for the internal hierarchical structure of the environment, as described by Frissell et al. (1986), although artificially induced in this case. This also applies to the decrease of CPOM on the other side of the gradient. Bank fixation is usually related to straightening and as a consequence increased flow velocities and reduced retention of CPOM. The retention of POM is further decreased due to the lack of respective morphological features functioning as POM sinks (e. g., large wood, pools). The second 'aquatic macrophyte' gradient was marked by

the proportion of emerged and submerged macrophytes on the stream bottom, which grow most abundant under stagnant flow conditions and in unshaded stream sections lacking riparian wooded vegetation. The gradient was fairly independent from the first ('hard-soft substrate') gradient as shown by the almost perpendicular orientation in Figure 4.4 for the CCA and to a lesser extent in Figure 4.7 for the RDA. In conclusion, hydromorphological degradation may be indicated solely by substrate characteristics, since the micro-scale seems to be largely controlled by reach-related variables.

4.6.2 Ordination of macroinvertebrate taxa

Oligochaeta Gen sp., *Baetis rhodani*, and *Gammarus roeseli* occurred at the 'impacted' gradient ends at the meso and micro-scale, whereas at the macro-scale only *Baetis rhodani* marked the 'impacted' end of the agricultural land use gradient. In contrast, *Nemoura* sp., *Halesus* sp., *Prodiamesa olivacea*, *Polypedilum* sp., and *Simulium* sp. showed negative relationships to the degree of human impact et all spatial scales. As Oligochaeta and *Gammarus roeseli* are known to have a positive relation to impacted sites (Lorenz et al., 2004b), the indicative potential of *Baetis rhodani* may be due to its rheophilic character and preference for stones (Schmedtje & Colling, 1996), indicating the presence of stones on the stream bottom. This was strongly related to hydromorphological alteration (rip-rap) in the dataset. The ordination also revealed a high general indicative potential of dipteran taxa, namely of the family Chironomidae, which stresses the importance of a respective taxonomical resolution for indicating the effects of hydromorphological degradation. At present, bioassessment and monitoring protocols often omit lower taxonomical units of Chironomidae and other dipterans (Armitage et al., 1983; Lorenz et al., 2004b).

The decrease of taxa richness and diversity due to habitat degradation was common at the meso-scale (Figure 4.3), where most taxa were located on the right ('natural') hand side of the plots. Thus, habitat degradation may rather be indicated by the lack of taxa typical of natural conditions than by the presence of taxa typical of impacted sites. Several biotic indices were based on sensitive (intolerant) taxa and, moreover, consider only high level taxonomic units (family level): BMWP/ASPT (Armitage et al., 1983); BBI (De Pauw & Vanhooren, 1983); Hilsenhoff's FBI (Hilsenhoff, 1988); DSFI (Skriver et al., 2001). As a consequence, these indices inevitably lack the capability of discriminating the impacts of different stressors. Any decrease in taxon (family) richness/diversity is interpreted as impairment and, therefore, its use is restricted to those rivers known to have a linear relation of richness/diversity and impairment. Consequently, the use of genus-/species-based richness/diversity measures is advantageous, since those measures rather provide the potential to distinguish between different sources of impact (stressors) and moreover comprise more tolerant taxa. The German Fauna Index, for example, specifically aims to assess the impact of hydromorphological degradation and it is based on macroinvertebrate species and genera previously identified to be indicative for certain hydromorphological variables of both positive and negative quality (Lorenz et al., 2004b; see also Chapter 5). The relation of 'positive' to 'negative' taxa varies from 1 : 1 to 5 : 1, depending on the stream type, whereas the number of indicator taxa varies from 86 to 189. This keeps the index applicable even if sensitive taxa are naturally reduced, for example, in naturally acidified streams.

4.6.3 Ordination of macroinvertebrate metrics

The metrics revealed a different pattern with an almost equal number of metrics typical of natural conditions and metrics typical of impacted sites at all spatial scales. The strongest relation was observed at the meso-scale. Therefore, in contrast to taxa metrics provide a basis to indicate hydromorphologically impacted sites, too. Referred to the metric groups, RDA identified functional and composition/abundance metrics to be indicative at the macro-scale: psammal-preferring taxa indicated a high agricultural and urban impact, probably due to the lack of POM, large wood, and pebble at degraded sites. Instead, the stream bottom was likely completely covered by sand. On the other hand, the increase of xylophagous and rheobiont taxa with decreasing agricultural and urban land use confirms the assumption. Nevertheless, the relation of land use practices and metrics is always indirect and remains rather speculative.

This did not apply to the reach-scale hydromorphological gradient that was related to several metrics with sensitivity/tolerance and richness metrics being most indicative at the 'natural' end of the gradient. It confirms the above discussed decrease of taxa richness and diversity with increasing human impact. Indices that consider the sensitivity and tolerance of certain taxa against pollution or other stressors, such as the British Monitoring Working Party (index) (BMWP) and the related Average Score Per Taxon (ASPT; Armitage et al., 1983), the Danish Stream Fauna Index (Skriver et al., 2001), and the German Fauna Index (GFI; Lorenz et al., 2004b) are *per se* positively related to the taxa richness. Consequently, richness measures were located close to these indices in Figure 4.6, for example, the number of Trichoptera, Ephemeroptera-Plecoptera-Trichoptera, and Diptera taxa. The 'impacted' end of the hydromorphological gradient was marked by functional measures and several types of saprobic indices. Metrics that were derived from the relative abundance of functional guilds, therefore, may have potential to characterize habitat degradation at the reach scale. Taxa preferring littoral and metapotamal sections characterized lentic to stagnant flow conditions, presumably leading to the accumulation of fine sediments. Consequently, pelal (mud) dwellers and gathering collectors, which feed on organic mud and FPOM occurred in high abundances. Since species level determination is not imperative to calculate functional metrics, this metric group may be valuable in evaluating meso-scale habitat degradation. The indicative potential of saprobic indices rather implies their stressor-insensitive indication, since only unpolluted sites were included in the analysis (mean revised German Saprobic Index: 1.91 ± 0.20). Saprobic systems are mainly based on sensitive/tolerant taxa that are not exclusively related to pollution and the absence of those taxa automatically leads to higher index values. However, the overall decrease in taxon richness due to hydromorphological degradation has a similar effect.

Several sensitivity/tolerance and richness/diversity metrics were also positively related to CPOM, FPOM, and large wood, which mark the 'natural' end of the substrate gradient (Figure 4.7). These habitats support a rich and diverse macroinvertebrate community besides those taxa that are sensitive to hydromorphological impacts in the investigated stream types. In contrast, the increase of aquatic macrophytes, cobbles, and pebbles on the stream bottom and the abundance of the respective taxa was related to the decrease of taxa richness and

sensitive taxa. Thus, a shift from richness/diversity and sensitivity/tolerance to functional metrics was obvious along the main gradient, as already revealed at the meso-scale. Similar to the meso-scale, the 'impacted' end of the habitat gradient was related to lentic or even stagnant conditions (hypopotamal and littoral preferences) and the proportion of gathering/collecting macroinvertebrates. This also applies to the proportion of phytal-dwelling individuals, grazer/scrapers, and those with preferences for the hypocrenal and epirhithral longitudinal zonation, too. But the latter four metrics rather represent straightened sections with artificial lotic flow conditions and rip-rap, promoting grazing/scraping taxa and those preferring hypocrenal and epirhithral flow conditions. Hence, both slow and fast flow conditions may be related to hydromorphological degradation.

4.6.4 Indicator species/metrics analysis (ISA)

ISA was used in order to quantify the relation of macroinvertebrate taxa and metrics indicated by the canonical ordinations. Although partial constrained analysis (pCCA, pRDA) provides a powerful tool for similar purposes (Borcard et al., 1992; Johnson & Goedkoop, 2002), ISA was used here instead for two reasons: 1) It provides a simple and 'easy-to-understand' tool to detect and describe the value of different species/metrics for indicating environmental conditions (Dufrêne & Legendre, 1997). Therefore, plain data can be used without any further standardization and transformation. 2) With ISA the relation of a single environmental variable to the biotic data is possible rather than the common analysis of groups of variables with pCCA or pRDA. Hence, the combination of both ordination and ISA in this case provides two approaches that may even seem somewhat redundant. Yet, as both statistical approaches led to similar overall results, while providing differently-detailed results, the combination supports the validity of the results. So, in concordance with the ordination results, the most significant relations of both taxa and metrics were identified at the meso-scale. Reach variables, such as floodplain land use, bank modification, flow regulation, straightening, large wood, and the riparian vegetation had the strongest relation to the macroinvertebrate community (Table 4.4). ISA also identified more metrics indicative as opposed to the taxa. This is supported by Griffith et al. (2001) who found similar relations when analysing macroinvertebrate taxa, metrics, and environmental variables. In their study, 96 % of the metric-environment relationship and only 45 % of the genera-environment relationship were explained. The value of metrics ('ecological traits') as functional units in stream ecology is also stressed by Poff (1997). The aggregation of species into functional groups (= guilds) makes multi-species analysis more tractable. Another advantage of metrics is their lower turnover in large datasets, in particular at large spatial scales, which allows of the use of linear regression models instead of unimodal approaches for taxa analysis. Finally, metric calculation does not depend solely on species determination, as many functional metrics can be calculated on the basis of genera or even families.

The ordination revealed the richness/diversity and sensitivity/tolerance metrics to be strongly related to the 'natural' ends of the land use and hydromorphological gradients at the meso- and micro-scale. Concerning the input/output ratio metrics per metric group, richness/diversity measures have the best indicative potential at the microhabitat scale (Figure 4.10). Taxa richness and diversity was directly related to the habitat characteristics. The

indicative potential of functional measures was above average at all spatial scales showing the general suitability of functional measures to indicate the impact of land use and hydromorphological degradation on the benthic community. From Figure 4.8 and 4.9 the dominant role of the taxonomic groups Ephemeroptera, Plecoptera, Coleoptera, Trichoptera, and Diptera among the total 244 taxa was evident. As Trichoptera and Diptera represented 40 to 55 % of indicative taxa at the three spatial scales (Figure 4.9) they are supposed to be the 'key groups' to detect the impact of hydromorphological degradation. This may even apply to trichopterans alone at the habitat scale. Moreover, autecological studies have addressed trichopterans for about 100 years and thus a good knowledge of the taxa's ecological background is available, summarized, for example, by Moog (1995) and Schmedtje & Colling (1996). Less effort has been spent on oligochetes and dipterans, so that their potential role in bio-indication might be improved within the next years. Chapter 5 is a small step forward as it focuses on the relation of the dipteran family Simuliidae to hydromorphological degradation.

5 The impact of hydromorphological degradation on Simuliidae (Diptera)

5.1 Scope

In the previous Chapter the insect order Diptera was identified to be of major importance regarding the assessment of hydromorphological degradation with benthic invertebrates. Various relations of dipteran taxa to hydromorphological variables were identified at different spatial scales. The findings underline the suitability of dipterans to detect hydromorphological impact. However, being only part of a long taxa list the specific role of certain dipteran species was not focussed on in the previous analysis. This Chapter highlights the specific role of the dipteran family Simuliidae. By restriction to only few taxa the identification of those taxa suitable for the indication of hydromorphological degradation was targeted. The analysis comprises the German AQEM sites and includes the Central Mountain data. If a taxon, for example, exclusively or almost exclusively occurred in the Central Mountains it is not likely to be suited to detect hydromorphological impact in lowland rivers and vice versa. Hence, the inclusion of both ecoregions in the analysis enabled the identification of such restrictions.

5.2 Summary

Blackfly communities from five German stream types covering two ecoregions were compared (small and medium-sized siliceous gravel-bed mountain streams and rivers of the Central European mountains, and organic type brooks, small and medium-sized sand-bottom Central European lowland streams and rivers). Ecoregional and stream type-specific differences of Simuliidae (Diptera) were revealed. The presence of *Prosimulium* sp. was restricted to mountain streams, whereas *Simulium lineatum* seemed to prefer medium-sized sand-bottom lowland rivers, and *S. vernum* showed a clear preference for lowland streams. The German Structure Index (GSI) was used to divide sites into morphologically 'unstressed' (high or good hydromorphological status) or 'stressed' (moderate, poor, or bad hydromorphological status), and biocoenotic differences of the two classes were discussed. Two stream types and the entire dataset showed significantly higher numbers of taxa at 'unstressed' sites. Linear Multiple Regression (LMR) was used to identify geo-hydromorphological parameters that significantly explained the variance of the three most frequent taxa, *Prosimulium* sp., *P. hirtipes*, and *Simulium* sp. in a LMR model.

5.3 Introduction

Several EU-funded projects have recently developed tools to assess the ecological status of rivers throughout Europe (e. g., AQEM consortium, 2002; Schmutz & Haidvogl, 2002; Hering et al., 2004a; Lorenz et al., 2004b). Including also national systems many projects focus on macroinvertebrates that are in general well-suited for assessment and quality indication systems, since a comparatively large amount of data exists, the identification is relatively simple, and they occur in large numbers in all stream types (Hellawell, 1986; Rosenberg &

Resh, 1993; Davis & Simon, 1995). Nevertheless, the insufficient ecological knowledge of several taxonomical groups (e. g., Simuliidae, Chironomidae) requires further extensive research efforts to allow these groups to be included in a sound explanation of macroinvertebrate reference conditions.

Blackfly larvae are widespread and regular members of the lotic community, and they inhabit most types of running waters. Several species are restricted to specific ranges of biotic and abiotic parameters, thus, meeting a minimum demand of indicator species(groups) for assessment. Moreover, certain species are known to be sensitive, for example, to acidification or organic pollution (Seitz, 1992), however, the impact of structural degradation on blackfly communities has not yet been well studied.

Two main questions were addressed in this Chapter: 1) Do blackfly communities show the ecoregional differences in presence/absence and species composition that have been identified on the basis of the whole benthic invertebrate community in Chapter 2? Therefore, taxa lists originating from Central Mountain streams and rivers were included in the analysis. 2) Is the presence/absence of blackfly communities related to specific hydromorphological features that provide an opportunity to indicate hydromorphological impact (stress) by certain indicator species or species groups?

5.4 Material and methods

5.4.1 Site selection and study area

The five German AQEM stream types were defined according to the German typology (Pottgiesser & Sommerhäuser, 2004) of which the lowland types have already been characterized in Chapter 3, Table 3.1. Where real reference sites have not been found (e. g., for medium-sized rivers in ecoregion 9 (type D05)), the ‘best available’ sites were taken as ‘assessment references’. However, assessment references were regarded to represent a ‘good ecological quality’ instead of a ‘high ecological quality’ and, thus, did not replace the (potential) reference conditions.

Macroinvertebrate samples were taken from small and medium-sized gravel to cobble-bed rivers of ecoregion 9, and small organic brooks, and small and medium-sized sand-bottom rivers of ecoregion 14. Stream type codes and general type description are given in Table 5.1. A total of 92 sites were sampled: 49 sites (= 53 %) at mountain streams and 43 sites (= 47 %) at lowland streams.

Table 5.1: Stream type properties (NRW = North-Rhine/Westphalia; RP = Rhineland-Palatinate; HE = Hesse; BB = Brandenburg; PL = W. Poland). Size classification according to EU commission (2000), Annex II, ecoregions according to Illies (1978).

Stream type	Name (catchment area)	Eco-region	Federal state	Sampled seasons	No. of sites	No. of samples
D01	Small sand-bottom lowland streams, (10–100 km²)	14	NRW	Spring, summer	12	24
D02	Small organic brooks, lowland (10–100 km²)	14	NRW	Spring	13	13
D03	Medium-sized sand-bottom low-land rivers, (100–1,000 km²)	14	NRW BB PL	Spring, summer, autumn	5 11 2	15 33 6
D04	Small siliceous gravel-bed moun-tain streams, (10–100 km²)	9	NRW RP	Spring, summer	19 10	38 20
D05	Medium-sized siliceous gravel-bed mountain rivers (100–1,000 km²)	9	NRW RP HE	Spring, summer	9 7 4	18 14 8
				Sum	92	189

Small sand-bottom streams and organic brooks (D01, D02) were limited to the western part of ecoregion 14 (North-Rhine/Westphalia), whereas medium-sized sand-bottom rivers (type D03) were distributed throughout the Central European Lowland with reference sites in East Germany (Brandenburg) and western Poland (Figure 5.1). Sand-bottom streams and rivers were naturally dominated by fine to coarse sand; bands of small gravel occur and may occasionally cover up to 50 % of the bottom. Degraded stretches were regulated and characterized by severe bed and bank modifications reaching also the riparian area. Organic brooks (type D02) were naturally dominated by bog mosses (e. g., *Sphagnum* sp.) and particulate organic matter (POM), which almost entirely covered the streambed. The proportion of mineral substrates (fine to coarse sand) increased with increasing degradation. Catchment size of the sites in the Central European lowland varied between 10 km² in small streams and 750 km² in medium-sized rivers. For a detailed description of the lowland types, see Section 3.4.2.

The investigated mountain streams were located in the Low Mountain Ranges of three German Federal states: North-Rhine/Westphalia (‘Sauerland’, ‘Rothaargebirge’), Hesse (‘Rothaargebirge’) and Rhineland-Palatinate (‘Eifel’) (Figure 5.1). The predominant geological formation of all catchments was silicate rock (palaeozoic clayey slate). The substrate grain size ranged from fine to coarse gravel and cobble and the size of the catchments ranged from 8 km² in small streams (stream type D04) up to 1,020 km² in medium-sized rivers (stream type D05).

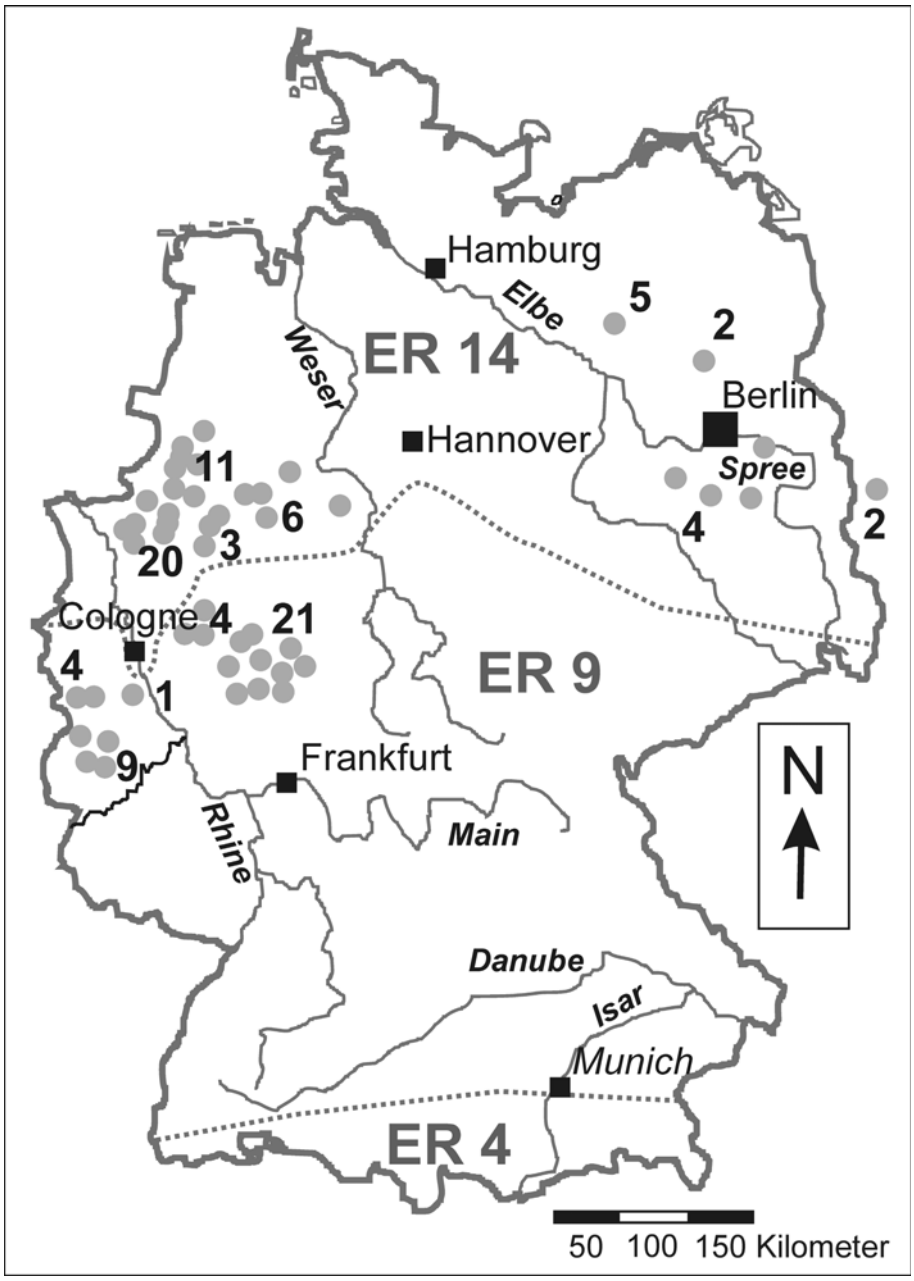

Figure 5.1: Study area and location of 92 sample sites in ecoregions 9 and 14.

The pre-selection of sites was based on hydromorphological maps and field judgement, assuming that hydromorphological degradation was the main stressor for the investigated stream types. This assumption was – concerning organic pollution – *a posteriori* confirmed by calculation of the German Saprobic Index (SI) for all macroinvertebrate samples. The SI ranged almost constantly within class boundaries of saprobic class II (*beta*-mesosaprobic, moderately loaded) and showed only slight differences between reference sites and sites of poor or even bad hydromorphological status. Hence, all sites were comparable in terms of saprobic load. In order to cover a hydromorphological gradient the pre-selection aimed also at comprising sites of different hydromorphological status, as far as possible ranging from high to bad quality.

5.4.2 Sampling, and sample processing

All macroinvertebrate samples were taken with a shovel-sampler (frame: 25 x 25 cm; mesh-size: 500 µm) using a modified Multi-habitat Sampling (MHS) technique (Barbour et al., 1999; Hering et al., 2004). A total of 20 sample units, each 25 x 25 cm, were taken from each habitat covering more than 5 % of the bottom representative to its total proportion. Hence, for example, a habitat that covered 20 % of the bottom was sampled with four out of 20 units; if a habitat covered less than 5 % of the sampled reach it was omitted but recorded as present in the site protocol. The restriction to a minimum coverage of 5 % inevitably affected the effectiveness of sampling in terms of blackfly larvae. In parallel to sampling, several hydromorphological parameters were recorded at the site, reach, and catchment scale in order to provide the basis for analysis of relations between species/species groups and environmental features. All sites were sampled twice in spring and autumn 2000, except for stream type D03 that was sampled three times in summer and autumn 2000, and in spring 2001. The identification of blackfly larvae was carried out to genus level except for mature larvae with well-developed gills; those specimens were determined to species level if possible as were also simuliid pupae. If individuals were not identifiable to species level (immature larvae) the respective species group or genus was recorded.

5.4.3 Statistical analysis

Sites were divided first into hydromorphologically 'stressed" and 'unstressed'. Therefore, selected site protocol parameters were used to calculate the German Structure Index (GSI). The selection of environmental variables and the calculation formula for the lowland types was described in Chapter 3 (Section 3.4.4). A different set of variables was used to calculate the GSI for the mountain stream types (Lorenz et al., 2004b; A. Lorenz, P. Rolauffs, pers. comm.). Ecoregional and stream type-specific differences of the blackfly communities were explored with Non-metric Multidimensional Scaling (NMS) using Jaccard dissimilarities. Qualitative (presence/absence) data were used, since the sampling method (MHS) was presumably not suited to get quantitative and representative data. The restriction to 'unstressed' aimed at minimising the overlap of degradation-dependent variation in the community. Only those taxa with a frequency > 5 % were included in the analysis: *P. tomosvaryi, S. angustipes-aureum-velutinum*-gr., *S. costatum, S. lundstromi, S. morsitans, S. naturale, S. paramorsitans, S. reptans, S. rostratum*, and *S. variegatum* were excluded from the analysis, as were all sites completely lacking blackflies. NMS was run with PC-Ord 4.3 (McCune & Mefford, 1999).

A Linear Multiple Regression (LMR) was used to reveal the relationship between the distribution of Simuliidae and the morphological quality of sites. However, the frequency of many species was too low for sound statistical analysis and, therefore, multiple regressions were exemplarily calculated for three taxa: *Simulium* spp., *Prosimulium* spp., and *P. hirtipes*. Stream type D02 was entirely excluded from regression analysis, since data on hydromorphological variables were very heterogeneous and incomplete regarding the information targeted here. Regression analysis and significance tests were run with STATISTICA 5.5 (StatSoft, Inc. 2000).

5.5 Results

5.5.1 Taxa richness and species composition

Blackfly larvae occurred at 86 sites (= 93 % of all sites), 47 sites (= 96 %) in ecoregion 9 and 39 sites (= 91 %) in ecoregion 14. At six sites simuliids were completely lacking; the sites represented both the worst hydromorphological and physico-chemical conditions in the total dataset (e. g., current velocity below 6 cm/s, up to 140 mg/l NO_3, 12 mg/l BOD_5, and 1740 µS/cm). A total of 189 samples was analysed, of which 98 (= 52 %) were located in the Central Mountains and 91 (= 48 %) in the Central Lowland. Simuliid larvae were found in 80 % of all samples. In all, 17 species were identified (Table 5.2): three of the genus *Prosimulium* and 14 of the genus *Simulium*.

5.5.2 Ecoregional differences

Ten species and species groups were recorded for lowland streams and 16 for mountain streams (Table 5.2). Nine species(groups) were restricted to mountain streams (*P. hirtipes, P. rufipes, P. tomosvaryi, S. costatum, S. naturale, S. paramorsitans, S. reptans, S. rostratum*, and *S. variegatum*), whereas only three were exclusively encountered in lowland streams (*S. angustipes-aureum-velutinum-gr.*, *S. lundstromi*, and *S. morsitans*). *S. equinum, S. ornatum, S. urbanum*, and *S. vernum* were predominantly found in lowland streams. In case of *S. vernum* this preference was significant, whereas *S. erythrocephalum* revealed a significant preference for mountain streams. Ecoregional differences were mainly due to the frequent occurrence of *Prosimulium* spp. that was restricted to mountain streams in ecoregion 9. *Prosimulium* spp. occurred at 42 sites (86 %).

5.5.3 Comparison of 'unstressed' and 'stressed' sites

For the entire dataset, the mean number of blackfly species was significantly higher at 'unstressed' sites (Table 5.3). This was also evident for sites of ecoregion 9 and on the smaller stream type scale for stream types D03 and D04. 'Unstressed' sites were colonised by 3.9–4.1 taxa, whereas only 1.7–2.9 taxa were found at 'stressed' sites. In contrast, small lowland streams (types D01 and D02) contained a considerably higher number of taxa at 'stressed' sites, yet the differences were not significant. While several species and higher-level taxa occurred with almost the same frequency at 'stressed' and 'unstressed' sites (*Prosimulium* sp., *S. argyreatum*, *S. costatum*, *S. equinum*, *S. erythrocephalum*, *S. ornatum*-gr., *S. urbanum*, and *S. vernum*) others seemed to be more sensitive to hydromorphological degradation (Table 5.2). *S. naturale* and *S. paramorsitans*, for example, were found only at 'unstressed' sites. *S. lineatum* was the only species that showed a clear and stream type-specific preference for hydromorphologically undisturbed sites. *S. lineatum* most frequently (50 %) colonized medium-sized sand-bottom lowland rivers (type D03), where its preference for 'unstressed' sites was significant (Mann-Whitney-*U*-test, $p < 0.05$). Regarding both ecoregions, its frequency at 'unstressed' sites was about three times the frequency at 'stressed' sites. In contrast, five species(groups) occurred exclusively, but with low frequencies (1–3), at 'stressed' sites: *S. angustipes-aureum-velutinum*-gr., *S. lundstromi*, *S. morsitans*, *S. rostratum*, and *S. variegatum*.

Table 5.2: Taxa list with frequency of occurrence in ecoregions and 'unstressed' and 'stressed' sites (bold = preference for ecoregion or morphological state).

	No. of sites		Morph. status of sites	
Taxon	ER 14	ER 9	'Unstressed'	'Stressed'
Prosimulium hirtipes (Fries, 1824)	–	**25**	13	13
Prosimulium rufipes (Meigen, 1830)	–	**7**	4	3
Prosimulium sp.	–	**42**	21	21
Prosimulium tomosvaryi (Enderlein, 1921)	–	**3**	1	2
Simulium angustipes-aureum-velutinum-gr.	1	–	–	1
Simulium argyreatum Meigen, 1838	6	10	8	8
Simulium costatum Friedrichs, 1920	–	3	1	2
Simulium equinum (Linnaeus, 1758)	**5**	1	3	3
Simulium erythrocephalum (de Geer, 1776)	2	**15**	9	8
Simulium lineatum (Meigen, 1804)	7	4	**8**	3
Simulium lundstromi (Enderlein, 1921)	1	–	–	1
Simulium morsitans (Edwards, 1915)	1	–	–	1
Simulium naturale (Davies, 1966)	–	1	1	–
Simulium ornatum-gr.	10	14	13	11
Simulium paramorsitans (Rubzov, 1956)	–	3	3	–
Simulium reptans (Linnaeus, 1758)	–	2	1	1
Simulium rostratum (Lundström, 1911)	–	3	–	3
Simulium sp.	39	43	38	44
Simulium urbanum (Davies, 1966)	**7**	2	5	4
Simulium variegatum Meigen, 1818	–	1	–	1
Simulium vernum Macquart, 1826	**10**	1	5	6

Table 5.3: Mean number of taxa ± SD at 'unstressed' and 'stressed' sites (min. frequency of taxa: 5 %). p = significance level (Mann-Whitney-U-test; n. s. = not significant at a level of $p < 0.05$). N = number of sites.

	'Unstressed'			'Stressed'	
Stream type	Mean ± SD	N	p	Mean ± SD	N
D01, small	0.8 ± 0.5	4	n. s.	2.0 ± 1.8	8
D02, small	1.6 ± 0.6	5	n. s.	2.0 ± 1.1	8
D03, medium-sized	3.9 ± 0.6	9	< 0.01	1.7 ± 1.4	9
D04, small	4.1 ± 1.1	13	< 0.01	2.9 ± 1.1	16
D05, medium-sized	4.8 ± 1.6	8	n. s.	3.6 ± 2.5	12
Ecoregion 14	2.6 ± 1.5	18	n. s.	1.9 ± 1.4	25
Ecoregion 9	4.3 ± 1.3	21	< 0.01	3.2 ± 1.8	28
Entire dataset	3.5 ± 1.7	39	< 0.01	2.6 ± 1.7	53

Besides hydromorphological degradation the analyses also revealed a significant difference between small and medium sized streams. 'Unstressed' medium-sized lowland rivers (type D03) were colonised by significantly more taxa than small streams of the same ecoregion (types D01 and D02) (Mann-Whitney-U-test, $p < 0.001$). A reversed trend was evident

for ‘stressed’ lowland sites, however, without a clear relation of stream size and number of taxa. Contrasting to the lowland streams ‘stressed’ sites of mountain streams were colonized by a mean of 2.9–3.6 taxa and stream types of ecoregion 9 showed increasing taxa numbers with increasing stream size for both ‘stressed’ and ‘unstressed’ sites, even if the type-specific differences in taxa numbers were not significant (Mann-Whitney-*U*-test, $p > 0.250$).

5.5.4 Multivariate comparison of ecoregions and stream types

The MDS ordination plot clearly reveals two major groups (Figure 5.2): sites of ecoregion 9 are located in the lower and right hand part, those of ecoregion 14 in the upper and left hand part of the plot. Ecoregion proved to be the predominant factor explaining blackfly community variance. Besides *Prosimulium* spp., *S. erythrocephalum* discriminated between ecoregions. Moreover, the small and medium-sized lowland streams and rivers cluster separately, which reveals stream size-dependent differences in the blackfly community. The discrimination of small and medium-sized streams and rivers in ecoregion 14 was due to *S. lineatum* and *S. ornatum*-gr.: both taxa were very rarely found in small streams.

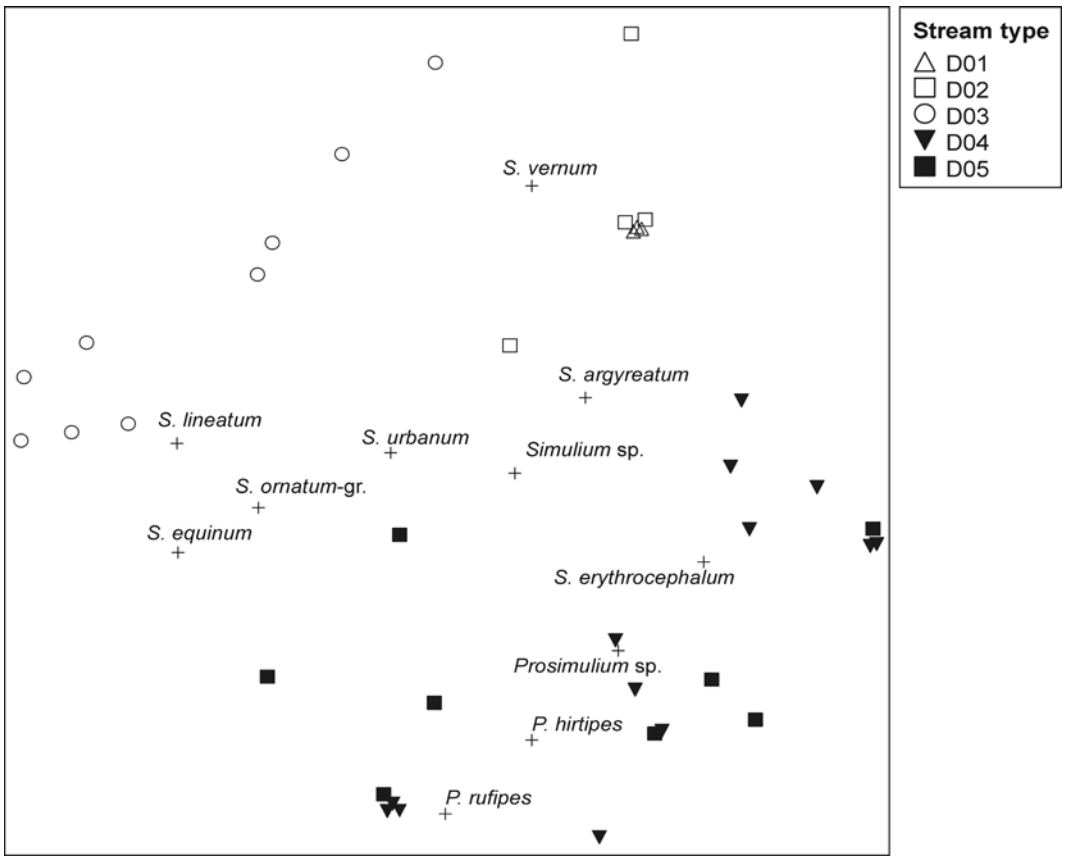

Figure 5.2: NMS ordination plot of eleven Simuliid taxa and 38 ‘unstressed’ sites of five stream types. For stream type codes see Table 5.1. Species’ location indicated by “+”. Distance measure: Jaccard. Final stress: 0.141. Variance explained: Axis 1: 19.1 %; axis 2: 33.6 %.

5.5.5 Multiple regressions

Simulium spp.

The first LMR model was calculated with *Simulium* spp. and a set of 44 hydromorphological variables (predictors), of which nine were finally included in the model (Table 5.4). Three site-specific variables were identified as significant discriminators: '% bank fixation with wood/trees', '% macrophyte coverage', and 'mean current velocity'. Because the first model explained only 38 % of the total variance (R^2) a second model was calculated with six out of the nine predictors from the first model. Supplementary, three outliers identified through the first model were *a priori* excluded. The second (final) model explained 47 % of the total variance of *Simulium* spp. with three significantly discriminating environmental variables at the site scale: '% shading', '% coverage of macrophytes', and 'mean current velocity'. Other (non-significant) parameters in the model were 'number of debris dams', 'number of organic substrates', and 'CV depth'.

Table 5.4: Site protocol variables and statistical properties of two linear multiple regression models on *Simulium* spp. Variables included in a model indicated by "+". Significant values for *beta* indicated in bold.

Site protocol variable	Model 1		Model 2	
	$F = 4.74$; $p < 0.001$; $R^2 = 0.380$	*beta*	$F = 10.10$; $p < 0.001$; $R^2 = 0.470$	*beta*
% Shading at zenith	+	0.131	+	**0.255**
No of debris dams	+	0.172	+	0.139
% Shoreline covered with wooded vegetation	+	-0.020		
% Bank fixation stones	+	-0.070		
% Bank fixation other materials	+	**0.267**		
No. of organic substrates	+	0.030	+	0.232
% Macrophytes	+	**0.273**	+	**0.259**
CV depth	+	0.233	+	0.151
Mean current velocity	+	**0.429**	+	**0.552**

Prosimulium spp.

The same procedure of LMR as described above for *Simulium* spp. was applied to *Prosimulium* spp. The second model ($F = 5.43$, $p < 0.001$) finally included nine hydromorphological descriptors and explained 57 % of the distribution of *Prosimulium* spp. in Central Mountain streams and rivers. Environmental variables with a significant *beta* were '% urban land use' (catchment and floodplain/site), 'number of organic substrates', and 'number of debris dams'. Other variables included in the model were: '% crop' (catchment and floodplain/site), 'average width of wooded riparian vegetation', '% lithal', and 'mean channel depth'.

Prosimulium hirtipes

The final LMR model included seven hydromorphological variables and explained almost 70 % of the total species' variance (Table 5.5). Besides the predictors identified as influential for *Prosimulium* spp. the variance in *P. hirtipes* was additionally explained by '% native forest catchment', 'width of the floodplain', and 'number of transverse structures upstream of the site' seemed to have a strong influence on *P. hirtipes* as shown by high values for *beta*. However, the model itself lacked robustness and the analysis of residuals did not show all parameters to follow a normal distribution, which is a prerequisite for statistically sound results.

Table 5.5: Site protocol variables and statistical properties for the linear multiple regression model on *P. hirtipes*. Variables included in a model indicated by "+". Significant values for *beta* indicated in bold.

Site protocol variable	Model F = 11.38; p < 0.001; R^2 = 0.690	*beta*
% Native forest (catchment)	+	**0.324**
% Urban settlement/industry (catchment)	+	**0.288**
Width of the floodplain	+	**0.624**
% Crop land (folldplain)	+	**0.581**
No. of transverse structures (upstream)	+	**0.566**
% Lithal	+	**0.250**
CV current velocity	+	**0.392**

5.6 Discussion

5.6.1 Methodological constraints

Multi-habitat Sampling (MHS) is known to be well-suited to record benthic macroinvertebrates, since shovel-sample units are taken from each representative habitat, i. e. substratum (Barbour et al., 1999, Hering et al, 2004a). However, for two reasons MHS was presumably insufficient to sample blackflies quantitatively: First, substrates frequently covering considerably less than 5 % of the site (e. g., few stands of macrophytes, single boulders, or few pieces of large wood) were omitted with MHS. Therefore, substrates likely to be preferred by larval and pupal simuliids were likely underrepresented in the samples. Second, the precise determination of pre-imaginal blackflies depends on pupal features, yet shovel sampling as applied within MHS often leads to small and, hence, unidentifiable larvae. As a consequence, the number of reliably identified species was comparatively small. A mean number of 4.3 species was, for example, recorded for 'unstressed' mountain streams, whereas Reidelbach (1994) recorded 13 species exclusivle in a small stream, the Breitenbach (Hesse). Seitz (1992) found between seven and eleven species in different 'unstressed'

streams in Bavaria. A total of 18 species were recorded, which was merely half the number of species expected for the investigated area (Schmedtje & Colling, 1996; Timm & Juhl, 1992; Seitz, 1992). But since the main objective of the AQEM project was to assess the hydromorphological status by comparison with reference conditions, the methodological disadvantages mentioned above were inevitable and acceptable. If all available habitats had been extensively sampled at each site regardless of its proportion the resulting community probably would not have reflected a hydromorphological condition representative for the sampled reach.

Zwick (1974) described methodological problems in case of quantitative samples, too, when a site comprises a diverse habitat composition. The author found time-restricted samples with 10 minutes effective sampling time (5 minutes for large stands of floating macrophytes) as best-suited and comparable for sampling representative blackfly communities. On the other hand time-restricted sampling is likely related to the researcher's experience and skills and is, thus, probably not suited to get standardized macroinvertebrate samples for assessment purposes, as it was targeted in this thesis.

5.6.2 Taxa richness and species composition

Simuliidae were frequently encountered in all investigated stream types of both ecoregions: blackflies occurred at 93 % of the sampled sites and in 80 % of the samples. The frequent and wide distribution fulfils a major demand on macroinvertebrate indicator organisms for river assessment (Seitz, 1992; Hering et al., 2004a). The present study, which was based on either presence/absence data or on the most constant (determinable) taxa (*Simulium* spp., *Prosimulium* spp., *P. hirtipes, S. lineatum*, *S. ornatum*-gr., and *S. vernum)* revealed remarkable results concerning blackfly relationships to morphological habitat features. Ecoregions and two lowland stream types were clearly distinguished by the blackfly community.

Species' preferences for certain stream types have already been described earlier. Seitz (1992), for example, found *S. costatum, S. naturale, S. variegatum*, and *S. reptans* in mountain streams and rivers. At least for some species the underlying ecological background probably responsible for a species' restriction to a region or stream type, is known quite well. Species of the genus *Prosimulium* are known to colonize mosses, which are typically found on large stones and other stable and solid substrates at the shoreline of mountain streams. Females of *Prosimulium* sp. depend on the presence of these mosses for oviposition (Zwick & Zwick, 1990; Timm, 1993; Timm & Klopp, 1993). The restriction of *Prosimulium* sp. to mountain streams (Table 5.2) corresponds to the results of earlier studies. Wirtz et al. (1990) stated a sharp borderline with *Prosimulium* sp. restricted to altitudes > 200 m where ecoregion 9 (Central Mountains) borders on ecoregion 14 (Central Lowlands). *S. variegatum* is likely dependent on high current velocities and oxygen levels (Kiel & Frutiger, 1997) and, thus, frequently occurs at higher altitudes. *S. costatum* and *S. naturale* seem to prefer springs or spring fed streams and are, therefore, found more frequently in mountain streams.

On the other hand, high frequencies were revealed for *S. equinum, S. ornatum*-gr., *S. urbanum*, and *S. vernum* in the Central Lowlands, which corresponds to the ecological knowledge on the species (e. g., Seitz, 1992; Schmedtje & Colling, 1996). Although all species are usually encountered in ecoregion 9 as well they prefer lowland streams and often occur with large populations. In particular, *S. equinum* and *S. ornatum*-gr. are frequently found in greater numbers on densely growing submerged macrophyte leaves and at sites with little or lacking riparian wooded vegetation (Timm & Klopp, 1993). *S. vernum* prefers shaded lowland sites with a well developed wooded riparian vegetation. Although *S. ornatum*-gr. may occur also there, *S. vernum* usually replaces *S. ornatum*-gr. at shaded sites due to different requirements for oviposition sites (Timm, 1994).

Regarding *S. erythrocephalum*, *S. rostratum*, and *S. morsitans* the results differ from other published data. For example, *S. erythrocephalum* was most frequently found in mountain streams, although the species is usually reported as preferring medium-sized to large lowland rivers (Seitz, 1992; Timm, 1995). *S. lundstromi* was missing in mountain streams but should have been present there according to Seitz (1992), and *S. rostratum*, a species that usually colonizes lake outflows in both ecoregions was restricted here to mountain streams. At least in terms of *S. lundstromi* and *S. rostratum* the discrepancies are probably due to methodological constraints (MHS, large number of unidentifiable larvae).

5.6.3 Factors to assess 'unstressed' and 'stressed'

As most Simuliidae are passive filter feeders, current velocity and flow pattern are very important for their development. Among other factors, the confinement of several species to a particular flow pattern seems to be based on morphometric features. According to the result of Malmqvist et al. (1999) there was a strong (negative) correlation between current velocity and stream size, and the size of larval headfans: species colonizing streams with low current velocities had large headfans, while species preferring larger rivers and high current velocities had small head fans. The authors concluded that flow pattern thus should govern species richness in a stream, because fans which are too large in relation to maximum flow might collapse while fans which are too small will be ineffective at slow flow velocity. The results presented here support this assumption, at least for Central Lowland streams and rivers. At 'unstressed' lowland sites species richness and stream size were correlated, and a significant difference between small streams and medium-sized rivers was evident. The mean number of species in medium-sized 'unstressed' lowland rivers was, for example, four times the number of small lowland streams. Malmqvist et al. (1999) found that sites in small streams and large rivers formed distant groups in a ordination plot. Hence, their data also indicated size-dependent differences in species composition.

Linear multiple regression revealed current velocity to be one of the parameter suited to explain the variance in the distribution of species which occurred frequently (> 50% of the sites), for example, *Simulium* sp., *Prosimulium* sp., and *P. hirtipes*. Despite the obvious influence of current velocity further factors also affect the blackfly community structure. According to the multivariate analyses of Malmqvist et al. (1999), variables that correlate with stream size – especially discharge, depth, channel width, and substratum – are of paramount importance. The authors further showed that small streams changed comparably more with

increasing width than did large rivers. Hence, even if channel size and corresponding parameters change only slightly, a focus on these features may still be important. The high number of abiotic variables recorded during the fieldwork helps revealing the relation of structural parameters and species' occurrence. Shading, for example, indicates the presence of riparian wooded vegetation, which also forms a source of large wood and other particulate organic matter (e. g., FPOM and CPOM). As CPOM together with large wood often causes an increased variability of current velocity and depth these features indicate more natural flow conditions and a more natural variability of morphometry. Thus, shading is presumably a useful (indirect) parameter to describe both the hydrological and structural condition.

To a certain extent, this is true for the macrophytes, too. Naturally, macrophytes would cover hardly more than 15 % of the stream bottom in small streams as was evident from the present study. Therefore, a negative correlation exists between the degree of shading at site and the % coverage of macrophytes on the stream bottom; both directly and indirectly affect the benthic community. The differences in blackfly species composition at small streams and medium-sized rivers may be due to a decrease in shading by riparian vegetation, which causes an increase in aquatic macrophyte density. Results from Wright et al. (1993) revealed the importance of plant leaves for blackfly densities. The authors showed that abundances of larval Simuliidae were more than ten times as high on *Berula* sp. and *Ranunculus* sp. stands than they were on bare sand or gravel. Larvae and pupae need firm contact to the substratum, because blackfly larvae attach themselves to the substrate for filter feeding and to fix pupal cocoons (Barr, 1982; Eymann & Friend; 1988, Reidelbach & Kiel, 1990). Bare sand or gravel bars usually provide suboptimal conditions for larval or pupal attachment. With increasing current velocities sand and gravel are mobilized, thus forcing blackflies to detach and move. Therefore, in unstressed sandy streams highest blackfly population usually grow up at sites with floating macrophyte leaves. The lower degree of shading, which enables a higher percentage of macrophyte coverage in lowland streams, may also be responsible for the colonization through a considerably higher number of taxa at 'stressed' sites than at 'unstressed' sites (2.0 *vs.* 0.8 taxa, respectively) in small sand-bottom lowland streams. Although current velocity and food supply may be optimal at 'unstressed' lowland sites, blackfly larvae likely do not find optimal conditions there. Stable substrate surfaces might be the limiting factor and are usually quite rare in most streams of this type. They are frequently colonised by other macroinvertebrates like, for example, larvae of the Trichopteran genus *Hydropsyche*, which are known to be successful competitors (Hemphill & Cooper, 1983; Hemphill 1988).

Current velocity proved to be one of the main hydromorphological features determining the presence of *Prosimulium* sp. and *Simulium* sp. in the present study. It might be the reason, why the organic brooks (type D02) were colonised by slightly more taxa at 'stressed' sites than at 'unstressed' (2.0 *vs.* 1.6 taxa, respectively). As the bed of organic brooks is usually almost completely covered by mosses and those streams usually have low current velocities (Rasper, 2001), blackflies are rare and characterized by a clumped distribution. In contrast, degraded sites are characterized by increasing current velocities. Thus, organic brooks may

be colonized by a more diverse blackfly community when slightly degraded. Moreover, hydromorphological degradation apparently affected the pH in this stream type, which was 5.6 ± 1.3 at 'unstressed' and 6.4 ± 0.5 (mean ± SD) at 'stressed' sites. Although not proved here, increasing pH may be a reason for higher taxa numbers at 'stressed' sites of organic brooks.

5.6.4 The impact of hydromorphological degradation on Simuliidae

The relation of Simuliidae and particular hydromorphological variables was evident from the results. Besides hydromorphological features, the relations strongly depend on typological aspects, mainly ecoregion and catchment size. Hence, simuliids provide important characteristics to assess the impact of hydromorphological degradation. The major findings have already been implemented with the AQEM Assessment System (Hering et al., 2004a; Lorenz et al., 2004b). This multi-metric system aims at assessing the ecological quality of German streams and rivers and uses (amongst other taxonomical groups) blackflies at two distinct levels: First, at the community level blackflies strongly influence the proportion of several functional metrics, such as '% rheophilic preferences', '% filter-feeders', and '% lithal preferences'. Moreover, selected black fly taxa add to the assessment system at the species level. On the basis of both presence/absence and abundance eight species were used to calculate the German Faunaindex (Feld et al., 2002a; 2002b; Lorenz et al., 2004a; 2004b). The Faunaindex corresponds to the concept of sensitive and tolerant taxa in terms of hydromorphological (habitat) degradation. The Faunaindex currently incorporates: *Prosimulium hirtipes*, *P. tomosvaryi*, *Simulium equinum*, *S. erythrocephalum*, *S. lineatum*, *S. paramorsitans*, *S. urbanum*, and *S. vernum*. *Simulium lineatum*, for example, proved to be suitable – on the basis of the present study – to indicate 'unstressed' hydromorphological conditions in medium-sized lowland rivers (type D03). But, as far as small sand-bottom lowland streams (type D01) are concerned this species indicates rather a morphologically 'stressed' situation. The contrasting indicator characteristics of *S. lineatum* within two similar stream types are presumed to be due to the preferred flow conditions in Central Lowland streams. Its significant preference for 'unstressed' sites corresponds to its (epi)potamal preference (Moog, 1995; Schmedtje & Colling, 1996). In small lowland streams, however, *S. lineatum* rather indicates a shift from natural rhithral to degraded potamal conditions caused, for example, by stagnation (weirs) or an non-natural expansion of submerged macrophyte stands.

6 Development of a multi-metric system to assess the impact of hydromorphological degradation on benthic macroinvertebrates

6.1 Scope

The last Chapter of this thesis presents an assessment system to assess the impact of hydromorphological degradation on benthic invertebrates. The development process is presented step by step from the selection of appropriate sample sites to the validation of the results based on expert judgement on the hydromorphological condition of a site. This inevitably includes the main findings presented in the previous Chapters which, therefore, already represent a certain part of the development process. For example, as a consequence of the results presented in Chapter 2, a stream type-specific approach was chosen and the analysis was limited to the comparatively homogeneous German AQEM lowland rivers with a catchment area > 50 km². Chapter 3 revealed that the identification of stressor gradients is a crucial step on the way towards a multi-metric assessment system. Therefore, numerous environmental variables are needed in order to properly identify the gradients and to calibrate the biotic indicators along the gradients. The calibration was implemented in this thesis by using direct gradient analysis. The procedure was presented and discussed in Chapter 4 based on a larger lowland dataset. Furthermore, the role of reach-related (meso-scale) hydromorphological variables was stressed before and is focussed on in the following. And finally, the comparison of the relation of taxa and metrics to environmental variables suggests a multi-metric approach for the assessment of the hydromorphological status, which was consequently pursued in the following Chapter.

6.2 Summary

Based on 82 macroinvertebrate samples out of 40 Central European medium-sized sand-bottom lowland rivers, the relation of environmental variables and metrics was examined at three different spatial scales using Redundancy Analysis (RDA). The main hydromorphological gradients revealed by the RDA were correlated with numerous metrics, of which five core metrics were selected to develop the multi-metric index (MMI): The German Fauna Index type 15, the number of indicator taxa of the respective Fauna Index, the relative abundance of rheophilic and littoral-preferring taxa, and the relative abundance of pelal-dwelling taxa. Two different scoring systems and the calculation of ecological quality ratios (EQR) were compared for the combination of core metrics to the MMI. The EQR method revealed the highest correlation with RDA gradients ($|r| > 0.910$ at all spatial scales). The performance of the MMI was tested by the comparison with scores based on expert judgement, leading to 85 % 'correct' identifications of stress (88 % for spring data only) and 51 % 'correct' classifications if referred to a five-class classification (61 % for spring data only.)

6.3 Introduction

The role of benthic invertebrates in river assessment was restricted to the indication of organic pollution for many decades. More than 40 years ago Zelinka & Marvan (1961) presented a saprobic index, followed by the British BMWP/ASPT (Armitage et al., 1983) and the German saprobic index (DEV, 1992). Other approaches aim at assessing the general impairment of the macroinvertebrate fauna caused by multiple impacts, such as land use in the catchment and floodplain or a severe bed and bank modification. The assessment can either be based on single indices, such as the Danish Stream Fauna Index (Skriver et al., 2001), multi-metric indices (Barbour et al., 1999; Karr & Chu, 1999), or by using predictive models that measure the distance of a test site's community to the expected community under reference conditions (Wright et al., 1993; Smith et al., 1999).

In Germany, river assessment with macroinvertebrates was solely based on the calculation of the saprobic index in the past (DEV, 1992). It was focussed on the detection of organic pollution, as it was supposed to be the main stressor affecting the in-stream fauna for decades. However, the general water quality in Germany improved during the last decade due to enormous efforts addressing waste water treatment. As a consequence, organic pollution is no longer dominant in German rivers: for example, 80 % of Hessian stream and river sections are either unpolluted or slightly polluted (oligosaprobic to *beta*-mesosaprobic; HMULF, 1999). In contrast, the German 'Strukturgütekartierung' (river habitat survey; LAWA, 2000) which is coherent to the respective CEN standard, resulted in only 20 % of the same sections being in an acceptable hydromorphological status (class 1–3). Yet, 65 % of the sections in Hesse were in a poor or bad hydromorphological status (HMULF, 1999). The situation is even worse in the Central Lowland of North Rhine-Westphalia, where only 2 % of the river sections were assessed to have a good status (highest two out of seven classes) opposed to 54 % that were in a poor hydromorphological condition (lowest two out of seven classes; StUA Münster, unpubl.). In the Netherlands, only approximately 4 % of the river sections provide near-natural hydromorphological conditions (Verdonschot & Nijboer, 2002) and in Denmark only 2 % are more or less natural (Brookes, 1987). It is apparent that physical habitat degradation has the most important impact at present and dramatically threats the aquatic and riparian biodiversity. Straightening, damming, severe bed and bank modification, or the disconnection of a river from its floodplain may cause a loss of several habitat types and associated species (Zwick, 1992).

Future river assessment has to focus on river type-specific assessment by comparison of a test site's community with the type-specific reference conditions (EU commission, 2000). The question of whether benthic invertebrates reflect the hydromorphological conditions of a site or reach has been answered in the previous Chapters. Besides, numerous aquatic species are reported to prefer a certain habitat, flow condition, or diet (Moog, 1995; Schmedtje & Colling, 1996) and are known to need certain morphological structures for oviposition, pupation, and also the habitat for the terrestrial adult stages to successfully complete their life cycles (e. g., Resh & Rosenberg, 1993; Merritt & Cummins, 1996; Hoffmann & Hering, 2000). Thus, the in-stream macroinvertebrate community is supposed to well reflect the structural integrity of a river.

This Chapter presents a method to assess the hydromorphological status of a river. Therefore, the relation of hydromorphological variables and ecological traits (metrics) of the benthic invertebrate community was examined following a river type-specific approach as demanded by the EU WFD.

6.4 Material and methods

6.4.1 Study site

The dataset originated from 40 sand-dominated lowland rivers in Germany and Poland and comprised a total of 82 invertebrate samples and associated site protocol data taken in three seasons: spring, summer, and autumn. For a physical-chemical characterization, site description, and site location, see Feld (2004) and Chapter 3. The pre-selection of sites aimed at i) covering a hydromorphological gradient as good as possible and additionally ii) excluding polluted sites in order to reduce the overlapping impact of different stressors at a single site (Feld, 2004; Lorenz et al., 2004b). Even if the German sites were originally divided into small and medium-sized sand-bottom rivers following the size classification of the WFD the current selection follows the alternative size classification presented in Chapter 2 (see Figure 2.7); the catchment area of the sites ranged from 50–760 km².

6.4.2 Sampling and sample processing

For a detailed description of sampling and sample processing see Chapter 2 and Hering et al. (2004a), for the selection, preparation, and spatial scaling of 130 site protocol variables used to analyse the fauna-environment relationship see Chapter 4 and Feld (2004).

A taxonomical adjustment was applied to the taxa list according to the procedure described in Chapter 2 and by Feld & Rolauffs, 2005). Taxa encountered in less than three samples (3.7 %) were excluded from the list. The adjusted taxa list was fed into the AQEM Software (Hering et al., 2004a) to calculate approximately 200 metrics and biotic indices. The metrics were assigned to four metric groups, each of which representing a different ecological aspect of the benthic invertebrate community (Hering et al., 2004a; 2004b, Lorenz et al., 2004b): composition abundance, richness/diversity, sensitive/tolerant, and function. In order to reduce the extensive metric list two filter procedures were applied: First of all, each metric was explored by box/whisker plots to identify those metrics with an insufficient range of the metric values within the 82 samples. For example, if the feeding type ‘miner’ was represented by only 0.05 to 0.56 % of the community the metric was defined unsuited for assessment purposes and omitted from further analysis. Secondly, a cross-correlation matrix was calculated for the remaining metrics (Spearman rank correlation) and metric pairs with a correlation coefficient $r > 0.800$ were defined as redundant. In case of redundancy, the correlation of each pair’s members with the other metrics was calculated and, finally, that metric was omitted that showed the higher overall mean correlation. All correlations and descriptive analysis were calculated with XLStat 5.2 (AddinSoft SARL, 2002). A total of 84 metrics was kept after application of the filter procedures (Appendix 3), of which the four Saprobic Indices have finally been excluded. A similar procedure was applied to the environmental variables, yet with redundancy at $r > 0.700$, leading to 49 variables at four spatial

scales remaining for statistical analysis. The variables and respective spatial scales are listed in Appendix 2, of which ecoregion, average stream width, and electric conductivity have been excluded due to the filter procedures. Proportional metrics (e. g., % feeding types, longitudinal zonation) and environmental variables (e. g., % land use) were arc sin $(x/100)^{0.5}$-transformed according to Podani (2000). Except for pH and binary variables all other environmental variables were log-transformed (Appendix 2).

6.4.3 Statistical analysis

Since seasonal differences of the benthic community may have a strong influence on certain metrics, the metric dataset was analysed first with analysis of similarity (ANOSIM) for seasonal patterns (see Chapter 2 for a detailed description of the method). However, in contrast to several other studies (e. g., Furse et al., 1984; Ward, 1989, Townsend et al., 1997), the metrics did not show any seasonal pattern within the 82 samples of medium-sized sand-bottom lowland rivers that were analysed here. The global ANOSIM R (0.039; $p = 0.077$) revealed the mean similarity of samples within a season and between seasons to be almost identical. Moreover, metrics representing the richness and abundance of certain taxa with a season-dependent occurrence of aquatic (larval) stages did not reveal a seasonal pattern in the dataset. The seasonal differences of, for example, the number of Plecoptera individuals (ANOVA: $F = 1.572$, $p = 0.214$, $df = 2$) and Ephemeroptera-Plecoptera-Trichoptera (EPT) taxa (ANOVA: $F = 2.836$, $p = 0.065$, $df = 2$) were not significant between any pair of seasons (Tukey's HSD test). Therefore, data of all seasons were analysed together in subsequent analysis.

Canonical ordination was selected to explore the relation of metrics and environmental variables and to identify appropriate metrics for a multi-metric assessment system. Representing a direct gradient analysis the canonical ordination axes of the fauna data are (constrained) linear combinations of the environmental variables. Canonical ordination detects the pattern of variation in the fauna data that is 'best' explained by the environmental variables (Jongman et al., 1995) and was applied in many previous studies with similar purposes (e. g., Richards et al., 1993; ter Braak & Verdonshot, 1995; Ruse, 1996; Weigel et al., 2003, or Johnson et al., 2004). Depending on the gradient length of the fauna data, which is a measure of the species/metric turnover, either a unimodal (long gradients > 3) or a linear relationship (gradient length < 2) can be assumed (ter Braak & Smilauer, 2002). Detrended Correspondence Analysis (DCA, detrending by segments) was used to calculate the gradient length of the metric data (= 0.767) and accordingly a linear approach was selected. The respective canonical ordination method, therefore, was Redundancy Analysis (RDA). For each spatial scale a separate RDA was calculated with CANOCO 4.5 (ter Braak & Smilauer, 2002, 2003) and a metric-environment biplot was drawn with CanoDraw 4.1 (ter Braak & Smilauer, 2003). By forward selection with 499 permutations each variable's contribution to the multiple regression was calculated (*Lambda*) representing a measure for the relation to the metric's variability. '*Lambda*-1' is a measure for the single contribution of a variable if it was the only one in the multiple regression (marginal effect), whereas *Lambda*-A represents the power of a variable in comparison with and additional to the others in the model (conditional effect). Season and catchment area were used as covariables in all RDA in or-

der to partial out the proportion of variance explained by the two 'mega'-scaled variables. Although seasonal differences were presumably negligible as shown before the selection of season aimed at excluding as much as possible the sources of 'natural' variation provided by the environmental variables.

6.4.4 Selection of candidate metrics

In river assessment, a candidate metric is a metric that is on principal suited to assess the impact of a stressor on the community from which the metric is derived. The identification of whether a metric is suited or not may be based on direct correlation with the impact. Yet, this approach is often difficult, since single hydromorphological impact measures need to simplify the information of various detailed hydromorphological variables, leading to an inevitable loss of information. Therefore, the individual metric's variability ('metrics fit') that was explained by the first canonical axis was used here to identify suitable indicator metrics at each spatial scale. As the first canonical axis is constrained in direct gradient analysis, it represents the main hydromorphological gradient and arranges the main community variability along this gradient. Metrics with a metrics fit value > 75th quantile (i. e. 25 % highest values) were defined as suited candidates at each spatial scale separately.

6.4.5 Selection of core metrics

A core metric is a metric of the candidate list that is used for the final multi-metric assessment system. To be selected as a core metric, three prerequisites must have been fulfilled here: 1) The metric had to perform well at each of the three spatial scales. 2) The relations to the hydromorphological main gradients had to be strong. This was indicated by the rank of performance: the best candidate (= highest metrics fit value) at each spatial scale was ranked 1, the second best 2, and so on. Thus, a metric was strong if its mean rank over all three spatial scales was low. 3) Finally, the core metrics had to cover the four metric groups composition/abundance, richness/diversity, sensitive/tolerant, and function with at least one representative.

6.4.6 Development of a multi-metric index

Two alternative methods have been applied for the combination of core metrics to a multi-metric index: a scoring system and the calculation of ecological quality ratios (EQR). The scoring system in general followed the approach of Barbour et al. (1999) and Karr & Chu (1999), but was modified with regard to the concept that least impacted conditions get the lowest scores. Originally, the authors scored the least impacted conditions highest (5), but this was reversed here so that the correlation of score and impact was consistent (positive) with other index-based assessment systems in Germany (e. g., DEV, 1992; LAWA, 2000; Rolauffs et al., 2004) and assigns the lowest value (1) for the least impact. The metric values were transformed into scores ranging from 1 to 5 following two alternative classifications (Figure 6.1). For the three-point system class boundaries were set to the 25th and 75th quantile. If a metric was positively correlated with the impact gradient metric values > 75th quantile were scored five points, values between the 75th and the 25th quantile three points, and values < 25th quantile one point. If the correlation with the impact was negative, the

scores were reversed. The class boundaries of the five-point system were set to the 80th, 60th, 40th, and 20th quantile and the scores were assigned accordingly as shown in Figure 6.1. Again, reversed scores were assigned if metrics correlated negatively with the impact gradient.

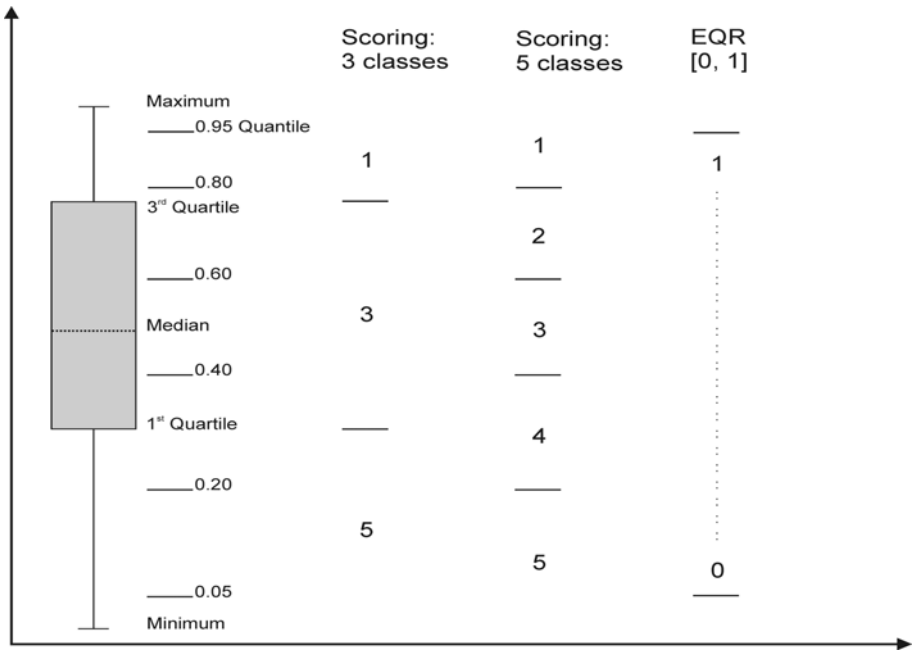

Figure 6.1: Three alternative schemes for the conversion of metric values into scores and ecological quality ratios (EQR), respectively.

The calculation of EQR was explicitly favoured by the EU WFD (EU commission, 2000) and followed the description of Böhmer et al. (2004). Therefore, each metric value was related to the respective metric's range, however the range excluded the lower and upper five percent of metric values, assuming that they represent lower and upper outliers (Formula 6.1). Since the conversion allowed for values outside the interval [0, 1], values < 0 were set to 0 and values > 1 were set to 1. For metrics negatively correlated with the impact gradient, the 5th and 95th quantiles were replaced by each other in Formula 6.1. With ecological quality ratios severe impact was indicated by values near zero.

Finally, the arithmetic mean score and EQR was calculated for each sample representing the multi-metric index (MMI).

$$EQR = \frac{metric\ value - 95^{th}\ quantile}{5^{th} - 95^{th}\ quantile} \qquad \text{Formula 6.1}$$

6.5 Results

6.5.1 Relation of metrics and environmental variables with RDA

The canonical ordination revealed a clear main hydromorphological gradient at each spatial scale that was ordered along the first ordination axis. At the macro-scale the gradient was characterized by the percentage of pasture and urban settlement/industry in the catchment (Figure 6.2A). The additional forward selection during the RDA calculated the highest conditional effects (*Lambda*-A) for pasture (0.06, p = 0.002) and the second-highest for urban settlement/industry (0.02, p = 0.006), which were the only significant results among all catchment land use variables.

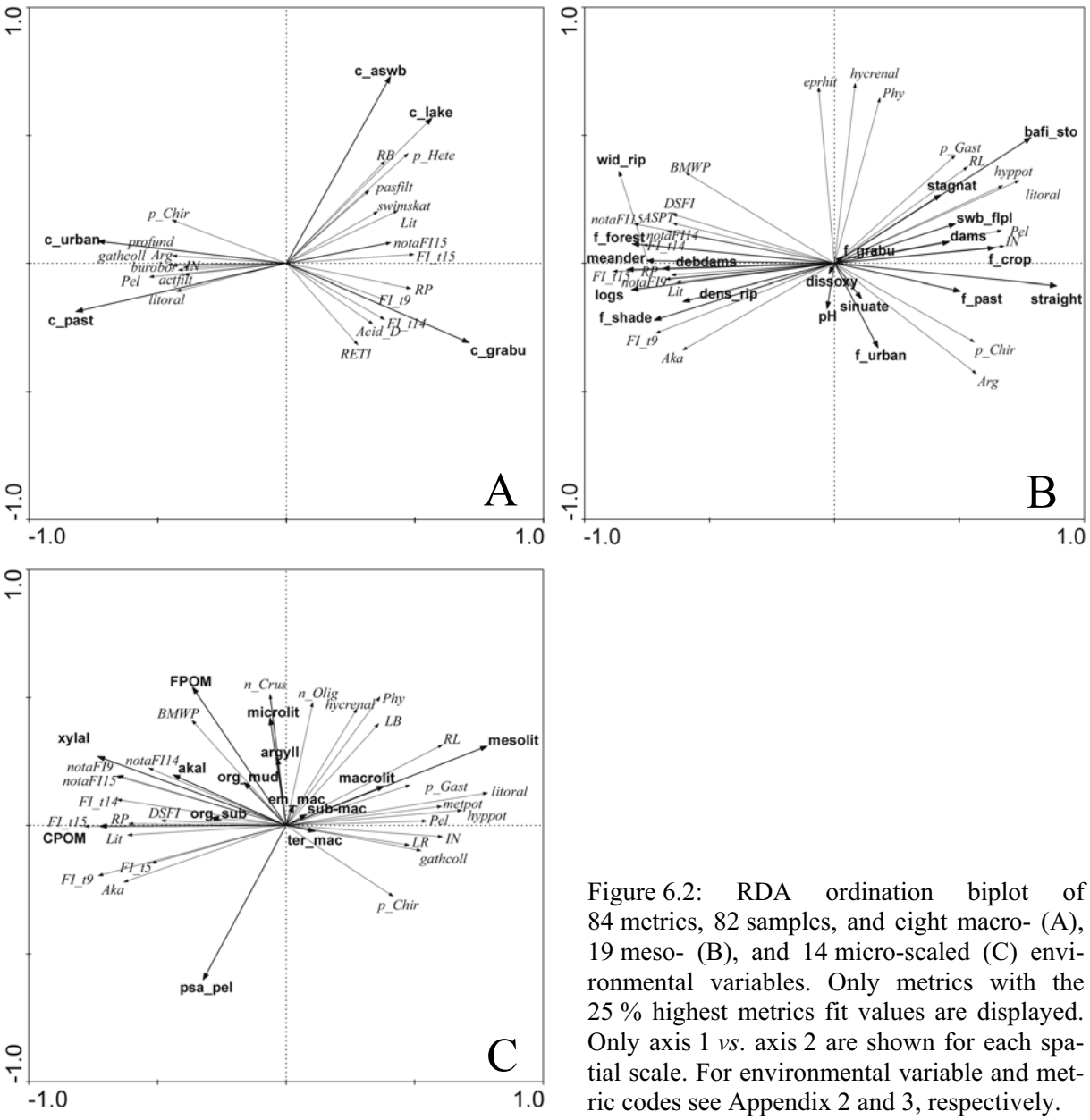

Figure 6.2: RDA ordination biplot of 84 metrics, 82 samples, and eight macro- (A), 19 meso- (B), and 14 micro-scaled (C) environmental variables. Only metrics with the 25 % highest metrics fit values are displayed. Only axis 1 *vs*. axis 2 are shown for each spatial scale. For environmental variable and metric codes see Appendix 2 and 3, respectively.

Among the reach-related variables (Figure 6.2B) a total of twelve had significant conditional effects of which straightening (0.16, p = 0.002) and the proportion of bank fixation with stones (rip-rap) (0.06, p = 0.002) explained the most variance in the metric-environment relationship. The other ('natural') end of the gradient (left hand side in Figure 6.2B) was related to meandering, a dense riparian wooded vegetation, the proportion of shaded sample reach, and the proportion of forest in the floodplain. Although *Lambda*-A values were comparatively low (range: 0.01–0.02) the variable's single strength (marginal effects) were as high as observed for straightening and bank fixation (range: 0.10–0.14).

At the micro-scale (Figure 6.2C) the proportion of four substrate variables showed significant conditional effects (range: 0.02–0.10): pebbles (mesolithal), xylal, cobbles (macrolithal), and sand/mud (psammal/psammopelal). As shown by *Lambda* the proportion of variance was explained best by reach-related variables, of which eight had marginal effects > 0.10, whereas none of the variables at the macro or micro-scale exceeded 0.10.

6.5.2 Candidate and core metrics

The metrics displayed in addition to the environmental variables in Figure 6.2 at each spatial scale exceeded the 75th quantile of metrics fit values for axis 1. Thus, they represent the 25 % metrics whose variability was best explained by the first ordination axis. Table 6.1 shows the metrics arranged along the mean rank order and represents the list of 'candidate metrics'. According to the prerequisites previously defined for the selection of core metrics (see Section 6.4.5), a total of five candidates (Table 6.2) fulfilled the criteria except for a representative of the 'composition/abundance' metric group. Within the latter the relative abundance of chironomids (% Chironomidae) showed the best correlation with the RDA axes 1 at the three spatial scales, however, no correlation exceeded 0.650 (Spearman rank correlation), which was rather weak compared to the other metrics. Thus, composition/abundance metrics were excluded from the selection of core metrics.

1. The *German Fauna Index type 15 (FI_t15)*: The index showed the best performance at all spatial scales indicated by the highest correlation with the respective RDA axes 1 (Table 6.2) and the most relations to specific environmental variables (Table 6.3). The Fauna Index was developed by Lorenz et al. (2004b) to specifically indicate the impact of hydromorphological degradation on the in-stream benthic invertebrate community in medium-sized sand-bottom lowland rivers (stream type 15 according to Pottgiesser & Sommerhäuser, 2004). A total of 165 indicator taxa were, therefore, assigned to scores from +2 (good indicator for hydromorphological reference conditions) to –2 (good indicator of severe hydromorphological impact) and, thus, the metric belongs to the 'sensitive/tolerant' group. The scores were derived from i) Indicator Species Analysis (Dufrêne & Legendre, 1997) using a similar set of environmental variables (Feld et al., 2002a; Pauls et al., 2002; Lorenz et al., 2004b) as was used for the current analysis and ii) an extensive evaluation of ecological studies of which the excellent review of Schmedtje & Colling (1996) provides an exhaustive source of information for the majority of German benthic invertebrate taxa.

Table 6.1: Candidate metrics with rank order according to the metrics fits with the first RDA axes at the macro-, meso-, and micro-scale. The selection encloses metrics above the upper quartile, i. e. metrics with the 25 % highest metrics fits. Metrics are arranged with decreasing mean rank order, core metrics are indicated in bold. For rules for the selection of candidate metrics see the text. Metric groups: S/T = sensitive/tolerant; C/A = composition/abundance; F = function; R/D = richness/diversity.

Metric name	Metric group	Macro	Meso	Micro
German Fauna Index type 15 (FI_t15) (Lorenz et al., 2004b)	**S/T**	**2**	**1**	**1**
Heteroptera [%]	C/A	4		
Littoral [%]	**F**	**10**	**4**	**2**
Hypopotamal [%]	F		8	5
German Fauna Index type 15 (notaFI15): No. of indicator taxa (Lorenz et al., 2004b)	**R/D**	**12**	**2**	**7**
Profundal [%]	F	7		
German Fauna Index type 9 (FI_t9) (Lorenz et al., 2004b)	S/T	15	5	4
Rheophil [%]	**F**	**3**	**11**	**12**
German Fauna Index type 14 (FI_t14) (Lorenz et al., 2004b)	S/T	17	3	8
Pelal (mud) [%]	**F**	**1**	**10**	**17**
Indifferent current preferences [%]	F	11	6	13
Lithal (coarse gravel, stones, boulders) [%]	F	9	14	11
German Fauna Index type 9 (notaFI9): Number of indicator taxa (Lorenz et al., 2004b)	R/D	18	7	9
Akal (fine to medium gravel) [%]	F		15	10
Chironomidae [%]	C/A	6	19	
Argyllal (silt, loam, clay) [%]	F	8	18	
Burrowing/boring [%]	F	13		
German Fauna Index type 14 (notaFI14): No. of indicator taxa (Lorenz et al., 2004b)	R/D		9	18
Active filterers [%]	F	14		
Rheo- to limnophil [%]	F			14
Gatherers/collectors [%]	F	5	20	19
BMWP (British Monitoring Working Party) (Armitage et al., 1983)	S/T		16	
Metapotamal [%]	F		17	15
Rheobiont [%]	F	16		
DSFI (Danish Stream Fauna Index) (Skriver et al., 2001)	S/T		12	21
ASPT (Average Score per Taxon) (Armitage et al., 1983)	S/T	21	13	
German Fauna Index type 5 (FI_t5) (Lorenz et al., 2004b)	S/T			20
Swiming/skating [%]	F	20		
Limno- to rheophil [%]	F	19		23
No. taxa EPT (Ephemeroptera, Plecoptera, Trichoptera)	R/D		21	
Gastropoda [%]	C/A			22

Table 6.2: Spearman rank correlation of core metrics and first RDA axes at the three spatial scales (N = 82, all correlations significant at $p < 0.001$). For respective RDA plots see Figure 6.2A–C.

	RDA axis 1		
Metric name	Macro	Meso	Micro
German Fauna Index type 15	0.813	-0.813	-0.818
Littoral [%]	-0.755	0.764	0.805
German Fauna Index type 15: No. of indicator taxa	0.725	-0.753	-0.721
Rheophil [%]	0.775	-0.734	-0.740
Pelal (mud) [%]	-0.783	0.734	0.735

Table 6.3: Correlation of core metrics and environmental variables at the three spatial scales. Brackets indicate positive (+) and negative (-) relations. Metric groups: S/T = sensitive/tolerant; F = function; R/D = richness/diversity.

Metric name	Metric group	Macro-scale	Meso-scale	Micro-scale
German Fauna Index type 15	S/T	% Grass-/bushland (+), % Urban settlement /industry (-), % Crop (-)	No. of debris dams (+) and logs (+), % Forest (+), % Shading (+), Width of riparian wooded vegetation (+)	% CPOM (+), % Xylal (+), % Mesolithal (-)
Littoral [%]	F		Stagnation (+), % Bank fixation stones (+), % Shading (-)	% Mesolithal (+), % CPOM (-)
German Fauna Index type 15: No. of indicator taxa	R/D		% Forest (+), Straightening (-), % Pasture (-)	% Xylal (+)
Rheophil [%]	F	% Grass-/bushland (+)	Meandering (+), No. of. debris dams (+)	% CPOM (+)
Pelal (mud) [%]	F	% Urban settlement/ industry (+), % Pasture (+)	No. of dams (+), % Crop (+)	% CPOM (-)

2. The *relative abundance of littoral-preferring taxa (% littoral)*: This 'functional' (longitudinal zonation) metric was calculated according to Schmedtje & Colling (1996) and provides a measure for the hydromorphological impact caused by severe bank modification (rip-rap) that is often connected with stagnated flow conditions (weirs) and an increase of the proportion of mesolithal on the river bottom (Table 6.3).

3. The *number of indicator taxa of the German Fauna Index type 15 (notaFI15)*: This metric was related to the proportion of meso-scaled land use, straightening, and the proportion of wood (% xylal) on the river bottom and represents 'richness/diversity' metrics. Although derived from the same indicator taxa list as was used to calculate the German Fauna Index (see Lorenz et al., 2004b) this metric was selected, since the correlation with the German Fauna Index was comparatively low (Spearman rank

correlation: 0.632, $p < 0.001$) and the relation to straightening and the proportion of pasture in the floodplain was additional to the German Fauna Index (Table 6.3).

4. The *relative abundance of rheophilic taxa (% RP)*: Rheophilic taxa were suited to indicate 'natural' environmental conditions at all spatial scales. This 'functional' (current preference) metric was calculated according to Schmedtje & Colling (1996).
5. The *relative abundance of pelal (mud) dwellers (% Pel)*: Representing another 'functional' (habitat) measure this metric was particularly related to the proportion of urban settlement/industry and pasture in the catchment, and tilled land (crop) in the floodplain (Table 6.3). The metric was calculated according to Schmedtje & Colling (1996).

6.5.3 Development of the multi-metric index (MMI)

Three multi-metric indices were developed, two of which were based on scores and one on ecological quality ratios (EQR). Comparing the performance of the MMI the EQR method revealed the highest correlations with the hydromorphological gradients identified by RDA and the second best with the German Structure Index (Table 6.4). Scatter plots of the MMI against the RDA sample scores clearly proved the applicability of the MMI along the gradients at the three spatial scales (Figure 6.3).

Table 6.4: Spearman rank correlation matrix of the German Structure Index (GSI; see Chapter 3 for details), RDA sample scores (axis 1) and three multi-metric indices (MMI) for 82 samples of medium-sized sand-bottom lowland rivers in Germany.

		RDA axis 1			Multi-metric index (MMI)		
	GSI[a]	Macro	Meso	Micro	3 scores	5 scores	EQR [0, 1]
German Structure Index (GSI)[a]							
RDA axis 1, macro	0.761						
RDA axis 1, meso	-0.816	-0.964					
RDA axis 1, micro	-0.816	-0.953	0.989				
MMI, 3 scores	-0.755	-0.883	0.862	0.876			
MMI, 5 scores	-0.775	-0.901	0.879	0.894	0.964		
MMI, EQR [0, 1]	0.757	0.933	-0.910	-0.913	-0.940	-0.957	

[a]Correlations with GSI based on N = 68 samples.

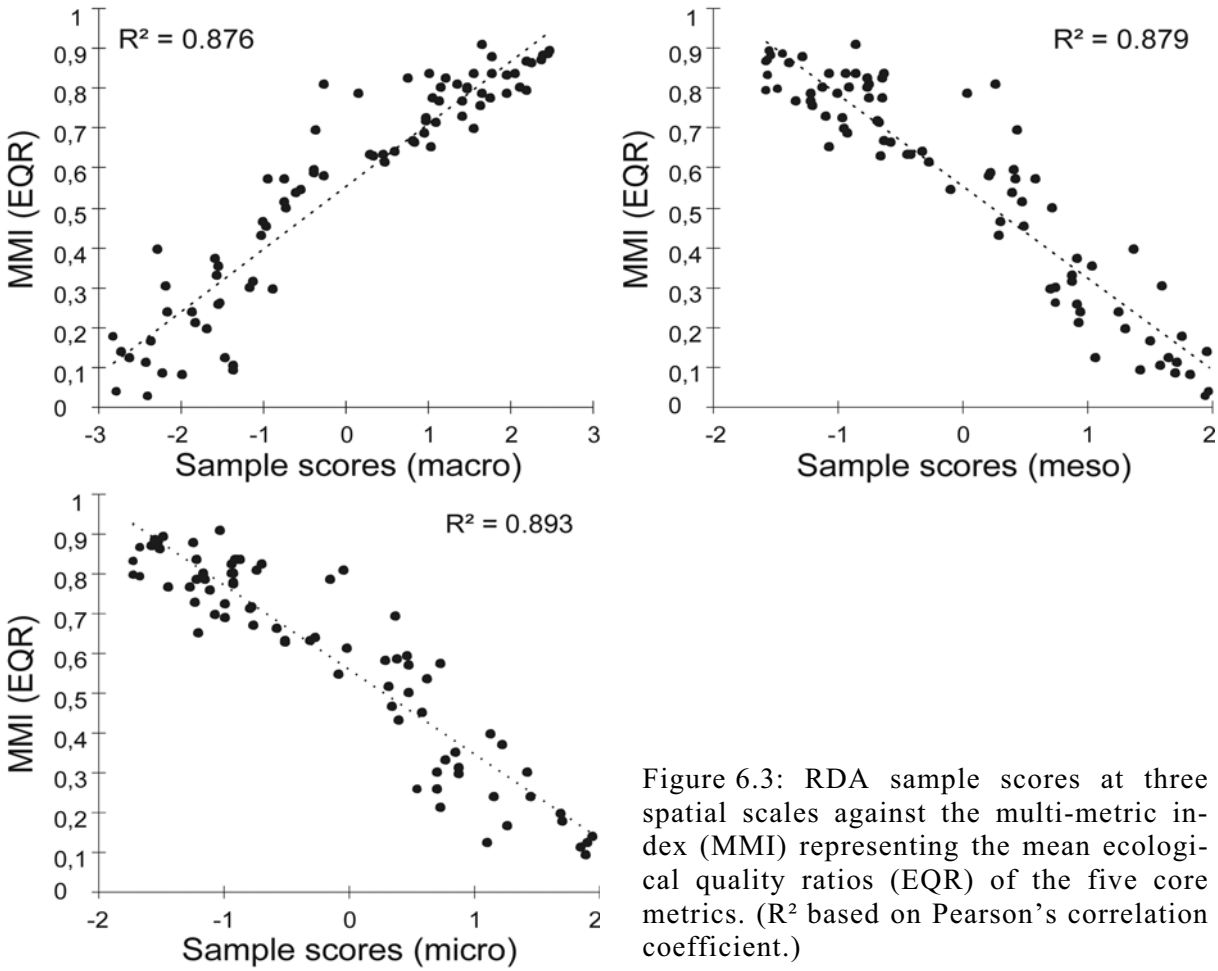

Figure 6.3: RDA sample scores at three spatial scales against the multi-metric index (MMI) representing the mean ecological quality ratios (EQR) of the five core metrics. (R^2 based on Pearson's correlation coefficient.)

6.5.4 Internal validation of the multi-metric index (MMI)

By the term 'internal' it is stressed that the validation presented here does not replace a validation with external data. However, as a first step the MMI of each sample was compared with a score (1–5) that was based on expert judgement of the person who took the sample and recorded the environmental variables for the field protocol (pre-classification). Therefore, the MMI were converted into five ecological quality classes (high, good, moderate, poor, and bad) according to the five-point system presented in Figure 6.1. For example, a value ≥ 0.8 was scored 1 and represented a high ecological status. The total correspondence of the five-class MMI and the expert judgement was 51 %, whereas 40 % of the samples differed one class and 9 % two classes. However, if the scores were combined to 'unstressed' (high, good) and 'stressed' (moderate or worse; see Chapter 3), the correspondence of the MMI and the expert judgement was 85 %, thus showing that the MMI provides a good measure of stress. The best results were gained with the spring samples (N = 33). Expert judgement and the MMI corresponded for 61 % of the samples and a two-class mismatch was observed for only 6 %. 'Unstressed' and 'stressed' were separated with a 88 % correspondence in spring.

6.6 Discussion

In recent years, river assessment in Europe was increasingly based on multi-metric assessment systems, a merit of the new legislative framework of the EU WFD (EU commission, 2000). Unlike to US American multi-metric systems, the European approaches aim at stressor-specific assessment, for example, Brabec et al. (2004), Ofenböck et al. (2004), or Sandin & Hering (2004) for organic pollution, Braukmann & Bis (2004) and Sandin et al. (2004) for acidification, and Lorenz et al. (2004b) or Ofenböck et al. (2004) for hydromorphological degradation. But stressors, such as organic pollution vary between regions, as waste water treatment was improved during the last decades. In Germany hydromorphological degradation is at present supposed to be the main stressor affecting particularly the aquatic macroinvertebrates (Feld et al., 2002a; Lorenz et al., 2004b). Therefore, the focus of this thesis was laid on the identification, definition, and, finally, the assessment of hydromorphological degradation. The selection of a multi-metric approach was supported by the long tradition of multi-metric indices (MMI) in the USA (e. g., Karr, 1994; Barbour et al., 1999; Karr & Chu, 1999). Multi-metric indices provide a measure to assess the river integrity at different spatial scales and, moreover, the authors state the applicability of MMI at different temporal scales (between seasons, years), which is referred to benthic invertebrates supported by the results of this thesis, but in contrast with other studies (e. g., Furse et al., 1984; Ward, 1989; Townsend et al., 1997). By comparison of aquatic macroinvertebrates on the community level, seasonal patterns seemed not to significantly affect the taxonomical composition as was shown in Chapter 2 for the complete lowland data (Sweden, The Netherlands, Germany, and Poland) and the German lowland data. Referring to the metrics, this was proven in Chapter 4 and also in this Chapter by forward selection of variables within RDA. Thus, regarding medium-sized sand-bottom Central Lowland rivers, multi-metric assessment with benthic invertebrates did not depend on whether samples were taken in spring, summer, or autumn.

However, the spatial scale of environmental variables was proved to be important for the metric-environment relation. The relation was highest for the meso-scaled (reach) variables that referred to a section of 500–5000 m up- and downstream of the sample site. Particularly, variables representing severe channel and bank modification were related to the metrics, such as rip-rap, straightening, or stagnation due to damming. Being consistent with the results presented in Chapter 4 (Table 4.2) for the whole lowland data this implies that the meso-scale remains most indicative, if a region or even a whole ecoregion is considered. However, spatial scales presumably cannot be referred to as separate entities. The hierarchical constraints that large-scale habitat descriptors may impose on small-scale habitat features were already summarized as the ‘hierarchical concept of landscape’ by (Frissell et al., 1986). The idea behind is that regional factors at the ecoregion or catchment scale may determine the conditions at lower (local) spatial scales. As an example, altitude (slope) and catchment geology strongly determine the discharge and habitat composition at a site. Relations in the reversed direction have not been reported. Opposed to Frissell et al. (1986) and also many others (e. g., Weigel et al., 2003) who concentrated on constant ‘natural’ environmental variables when examining the hierarchical relations this thesis focussed on the

interaction of variable 'non-natural' features representing a human-induced impairment. Thus, the question arises of whether the hierarchical relation is also applicable to 'non-natural' environmental variables. As shown by the relation of environmental variables and core metrics in Table 4.3 the answer must be 'yes'. An intensive agricultural land use (crop, pasture) is usually connected with the almost complete lack of natural wooded vegetation in the floodplain and even in the riparian area. A large amount of fine sediment may enter the river during heavy rainfalls. Moreover, meandering river courses running through agricultural areas are often straightened and also dammed in order to control the water level and prevent from floods. Straightening and damming presumably represent the most severe man-made modifications, since they may completely alter the hydromorphological conditions within a stream reach. Straightening leads to an increase of current velocities, but to a decrease of its diversity; it is often connected with bank fixation and scouring. Damming becomes necessary to control the water level and reduce the scouring leading to a significant decrease of current velocities and an increase of water temperature during summer. As a consequence, macrophytes dominate slow flowing unshaded sections from spring until autumn and promote the accumulation of mud in combination with the reduced flow. The human impact at the catchment and reach scale leads to a completely different habitat composition with a shift from a diverse mixture of different mineral and organic substrates towards a monotonous river bottom characterized by sand and mud (Pelal), and additionally cobbles if banks are fixed with rip-rap. Other habitats, in particular those reflecting 'natural' conditions (debris dams, logs) lack. Although this 'hierarchical concept of impact' is not really proved here the results strongly support the concept. Almost all variables mentioned before (except for sediment input and water temperature) were related to the community. Referring to the metrics the hydromorphological degradation was directly reflected by 'functional' measures. The connection of the abundance of rheophilic, littoral-preferring, and mud-dwelling taxa on the one hand and decreasing flow velocities, stagnation, and the increase of mud on the other was likely to be causal.

The conversion of metrics prior to the combination to a multi-metric index was imperative since the different core metrics represent different numerical scales. Proportional metrics are usually scaled from 0 to 100 % (0–1), whereas the German Fauna Index theoretically ranges from -2 to 2, and the number of the respective indicator taxa theoretically ranges from 0 to 165 (total number of indicator taxa for the respective river type). Moreover, proportional measures show different ranges from the least to the highest impact. To overcome this methodological constraints, Barbour et al. (1999) and (Karr & Chu, 1999) favoured a scoring system with three scores as was also applied here, but a five-point scoring system was also presented by Barbour et al. (1999), however, the rules to define the class boundaries were differently set in the current study. Ecological quality ratios (EQR) as applied here were specifically favoured by the EU WFD (EU commission, 2000) and several guidance papers related to the development of classification systems. In general, the three methods revealed a high correlation with the three main gradients identified by RDA. However, EQR were favoured here for two reasons. 1) They showed the best performance at all spatial scales and 2) provide a continuous measure. Blocksom (2003) recently supported the better performance of continuous measures in multi-metric assessment systems. For the same rea-

son Böhmer et al. (2004) favoured the EQR method, too, and suggested the procedure that was followed here. The comparison of two scoring systems and EQR as presented here underlines the author's findings.

As shown with Figure 6.3, the MMI provides a suitable measure to detect the impact of hydromorphological degradation. Even if a total of 82 high-quality fauna samples were used for the development the index needs to be validated with external data. In particular with regard to the German Fauna Index, however, external data should be based on samples gained with a commensurable sampling procedure. Nevertheless, the MMI was clearly adjusted to the hydromorphological gradients revealed for medium sized sand-bottom Central Lowland rivers and the applicability to detect hydromorphological stress was comparatively convincing. Thus, a general applicability in this river type is given.

Summary

In December 2000 the European Commission passed the Water Framework Directive (WFD). The WFD sets the framework for future assessment of rivers, lakes, transitional, and coastal waters in the European Union. The central demand is to achieve a 'good ecological quality' in all surface water bodies, whereas the assessment must be based on biological quality elements (BQE), i. e. fish, benthic macroinvertebrates, macrophytes and phytobenthos, and phytoplankton. Moreover, the assessment must refer to type-specific reference conditions and be capable of detecting the impact of multiple stressors on the community. For streams and rivers, which are subject of this thesis, this necessitates to 1) develop a stream typology and 2) accordingly stream type-specific reference conditions to provide the basis to assess 3) the impact of multiple stressors on the relevant BQE.

As this thesis aims at the development of a multi-metric assessment system based on benthic macroinvertebrates to assess the hydromorphological degradation the criteria defined by the WFD set the conceptual framework for the development process. However, several questions have to be answered to meet the WFD on the one hand, but to check the applicability of the criteria on the other.

The first study, therefore, examines the role of the type descriptors defined by the WFD for the benthic macroinvertebrates. As those variables, such as ecoregion, catchment area, or substratum composition lead to a 'top-down' typology the specific question is, whether those descriptors are reflected 'bottom-up' by the in-stream community. In this context the role of spatial scales is highlighted, too, since the variables act at different scales from the ecoregion- to the site-scale and, furthermore, are reported to underlie a hierarchical structure.

A German monitoring dataset covering all ecoregions is analysed with 'Non-metric Multidimensional Scaling' (NMS), an ordination method that aims at identifying the inherent community structure in terms of similarity of a single sample's community to the others. The analysis identifies ecoregion to predominantly control benthic macroinvertebrates and clearly separates the Alps, Central Mountains, and Central Lowlands. Further detailed analysis on the lowland data reveals the stream size to discriminate between small streams and medium-sized rivers in the dataset. Regarding the substrate composition organic type brooks are clearly separated form mineral substrate-dominated streams and rivers, however, a clear separation of gravel- and sand-bottom streams and rivers is not obvious. The comparison with another lowland dataset confirms the main results. A Central Lowland dataset covering four European countries again identifies ecoregion and stream size as suitable descriptors of the benthic macroinvertebrate community. The separation of stream and river communities is detectable at about 50 km² of catchment size. Sand- and gravel-bottom streams and rivers, however, are not separable. And as opposed to findings of others seasonal aspects in the community are barely reflected in this study.

The results confirm the importance of both stream type descriptors and stream typologies as major prerequisites to develop and apply river assessment systems based on benthic macroinvertebrates. Moreover, the spatial scale of the important type descriptors changes with re-

spect to the spatial scale that is covered by the samples regarded in the analysis. Therefore, river assessment must refer to stream type-specific differences in order to be capable of detecting the impact of, for example, hydromorphological degradation.

Hydromorphological degradation comprises several aspects of hydrological, morphological, physical-chemical, and land use features affecting the in-stream communities and is reported to be the main source of impairment in several European countries. But even if the impact of certain aspects of hydromorphological degradation on benthic macroinvertebrates is well known (e. g., the influence of stagnation) and hydromorphological surveys have been carried out in many European countries recently, there are still some questions unanswered. First of all, the role of spatial scales in hydromorphological degradation is fairly unknown. Yet, the knowledge on it is crucial for the definition of appropriate management and restoration plans to reduce the degradation. Secondly, the single different aspects of hydromorphological degradation need to be identified and quantified in order to be able to assess their multiple impact. And thirdly, a measure is needed to describe the overall degradation that can be used to calibrate the biological assessment systems to the impact.

Therefore, the second study of this thesis aims at the identification and measure of hydromorphological degradation. A Central Lowland dataset of 106 hydromorphological variables recorded for 275 samples out of 147 stream and river sections and comprising six European stream types is analysed using 'Non-metric Multidimensional Scaling' (NMS). The analysis is run several times from the ecoregion-scale with six stream types to the site-scale using data of medium-sized sand-bottom streams only. Addressing the first question hydromorphological variables reveal a scale-dependent relation. The common analysis of all stream types identifies catchment-related variables (land use, geology) to explain the predominant hydromorphological structure. If restricted to three German stream types and, thus, to a smaller geographical extent the increasing role of reach- and site-related properties is obvious. This also applies to the analysis of a single stream type, which determines slightly more site-related variables to describe hydromorphological degradation. The analysis also reveals the different aspects of hydromorphological degradation. Besides the agricultural and urban land use in the catchment degradation is characterized by bed and bank modification (e. g., rip-rap) and the loss of riparian wooded vegetation at the reach-scale, and by the loss of organic substrates, such as wood at the site-scale. As the NMS orders samples along a gradient of hydromorphological degradation the correlation of single variables with the gradient provides a measure to identify good indicators. A total of eight different hydromorphological aspects are combined to the German Structure Index (GSI) capable of measuring hydromorphological degradation by inclusion of different spatial scales (catchment: land use; reach: large wood, shading, riparian vegetation, flow modification, scouring, bank modification; site: organic substrates).

The analyses at different spatial scales confirm the need to apply stream typologies for river assessment. If the detection of hydromorphological degradation is aimed at, a single stream type represents a suitable scale to detect and measure the degradation. However, this is also possible if the German stream types are combined, since they are similar as far as the main hydromorphological properties are concerned. Furthermore, hydromorphological degrada-

tion of sand-bottom lowland rivers can be separated into different aspects including land use, hydrological, and morphological impacts. Some variables presumably reflect an inherent hierarchical structure, such as the proportion of forest in the catchment controlling the number of logs on the river bottom, or the proportion of wooded riparian vegetation controlling the degree of shading and presumably also the amount of large wood on the river bottom. However, as those variables often show medium inter-correlation, each variable represents a certain unique aspect of degradation and may, thus, be related to a certain community aspect.

The relation of hydromorphological variables and the benthic invertebrate community is subject of the third study of this thesis. The aim of this study is to identify the relation of community measures (taxa, metrics) to certain hydromorphological and land use features. Therefore, a hydromorphological dataset, a taxa list, and a metric dataset derived from the taxa list are analysed together using direct gradient analysis. Taxa and metrics are compared in order to examine their particular suitability to assess hydromorphological degradation, and the analysis is separated into different spatial scales to identify scale-dependent suited taxa and metrics.

After exclusion of redundant metrics, a total of 84 metrics is analysed as opposed to 244 taxa. Detrended Correspondence Analysis (DCA) is used to measure the community gradient length in both datasets, which reveals a long gradient for the taxa and a rather short for the metrics. This already represents an advantage of metrics over taxa, since the small gradient observed for metrics allows of using linear models to explore the relation to hydromorphological variables, whereas unimodal relations have to be considered for taxa. Moreover, certain taxa (e. g., plecopterans) may be subject to short-termed changes and may show a small-scale distribution. In contrast, metrics (e. g., functional guilds: feeding types, habitat preferences, current preferences) integrate numerous taxa and are presumably less prone to spatial or temporal scales.

In general, the comparison of metrics and taxa identifies metrics to be more indicative than taxa and, thus, to be better suited for river assessment. In particular, functional aspects of the community have a strong relation to hydromorphological variables. Most relations are obvious at the reach-scale, in particular to the proportion of bank fixation and riparian wooded vegetation, the amount of large wood on the river bottom, and flow modification. Thus, the same variables identified before to mainly describe the hydromorphological gradient are proved in this study to be strongly related to numerous community properties, which provides an important step to reach the aim of this thesis.

Besides, some findings may support the concept of a hierarchical structure of hydromorphology, which is also reflected by the benthic macroinvertebrates. As an example, the gradient's alignment at the three spatial scales implies the interdependence of agricultural land use in the catchment, straightening, stagnation, and the proportion of macrophytes at a site. Intensive agricultural land use usually comes along with straightening and damming, the latter to prevent from groundwater level subsidence. Stagnated sections are covered by large stands of macropyhtes and promote macrophyte dwellers. Hence, the intensive agricultural

land use at the catchment-scale is related to the amount of macrophytes at a site within the catchment and, thus, to the proportion of phytal-preferring individuals at the site-scale.

The important role of the reach-scale is also supported by the taxa results. Moreover, the taxa analysis reveals two major insect families, Trichoptera and Diptera, to not only dominate the taxa list used for the analysis but also to dominate the list of indicative taxa. About 40–50 % of the relations are observed for both families. Compared to the number of taxa used for the analysis trichopterans are particularly related to site-scale hydromorphological features, such as organic substrates.

As dipterans are shown to dominate the in-stream benthic community in this thesis, a certain dipteran family is focussed on in the fourth study. The aim is to identify the suitability of simuliid taxa (blackflies) to assess the impact of hydromorphological degradation. Simuliidae are widespread and common and are encountered with many taxa in almost all stream types. Thus, they fulfil an important criteria of potential indicators.

A dataset of five German stream types including two Central Mountain types is analysed using linear multiple regression analysis (LMR). Mountain streams are included in order to detect ecoregional constraints in species' distribution. The analysis includes 21 taxa of 189 samples at a total of 86 sites. While *Prosimulium* spp. is restricted to mountain streams and occurs in 86 % of the samples, no species is restricted to the Central Lowlands. However, *Simulium vernum* shows a clear preference for lowland rivers where it occurs in 25 % of the samples. For the relation of blackflies to hydromorphological degradation the sites are first allocated to the hydromorphological groups 'unstressed' and 'stressed' according to the German Structure Index. A single species, *S. lineatum*, shows a significant preference for 'unstressed' lowland streams and rivers. Referring to the whole dataset the number of species is significantly higher at 'unstressed' sites. The LMR analysis is restricted to the three most common taxa: *Prosimulium* spp., *P. hirtipes*, and *Simulium* spp. The occurrence of *Prosimulium* spp. is mainly related to catchment land use, the number of organic substrates, and the amount large wood, whereas *Simulium* spp. shows a strong relation to the mean current velocity, the proportion of macropyhtes at a site, and the degree of shading.

The results on principle underline the suitability of Simuliidae to indicate the impact of hydromorphological degradation. Species richness is significantly lower at hydromorphologically 'stressed' sites and at least the presence of *S. lineatum* seems to be directly related to 'unstressed' hydromorphological conditions. Their relation to submerged macrophytes and the current velocity implies their preference for floating macrophytes in lotic stream and river sections. However, the question of whether the observed preference is generally applicable has to remain unanswered. It's quite possible that degraded sites in a straightened section without riparian vegetation and consequently a large amount of aquatic macrophytes may provide similar conditions and support a diverse blackfly community. Thus, the results support the assumption that Simuliidae are suited to indicate extreme degradation (total stagnation, complete lack of solid substrates) rather than intermediate impairments. To clarify their specific role, further studies should address their micro-scaled distribution and apply suited sampling techniques, since the Multi-Habitat Sampling applied in this study is not likely to obtain representative and quantitative blackfly samples.

The central aim of thesis is to develop an assessment system, which is subject of the last study. As already mentioned before the WFD set the conceptual framework for the development process: stream type-specific assessment based on type-specific reference conditions and capable of detecting multiple impairments. In addition, the previous studies have been carried out for three reasons: 1) To check the relevance of the WFD stream type descriptors regarding benthic macroinvertebrates, which was prerequisite for type-specific assessment with the respective biological quality element. 2) To identify and measure hydromorphological degradation, which was a prerequisite to calibrate any assessment system to it. 3) To identify the multiple relations of hydromorphological variables and benthic invertebrate community properties. The second and third point also aimed at identifying the interaction of hydromorphological variables at different spatial scales.

First of all, the appropriate stream type is delineated regarding the specific results of the first study: medium-sized sand-bottom Central Lowland rivers. The type comprises all sites resp. samples in this study originating from ecoregion 14 with a catchment size ranging from 50–760 km². Samples from all seasons are combined for the analysis.

The multivariate analysis is focussed on the impact of reach-scale degradation on the community, but also aims at detecting the impact of all three spatial scales. Finally, metrics are favoured for the assessment system. The metrics are assigned to four metric groups representing different macroinvertebrate community aspects: ‘sensitive/tolerant taxa’, ‘composition/abundance’, ‘richness/diversity’, and ‘functional aspects’.

The delineation of the stream type is followed by the analysis of the relation of hydromorphological variables and metrics using Redundancy Analysis, which is repeated at each spatial scale. The best-related metrics at each scale are ranked and combined to a common list representing the ‘candidate metrics’. From the candidates several core metrics are selected that fulfil the following demands: 1) High relation to 2) all three spatial scales, and 3) covering the four metric groups.

A total of five metrics is selected fulfilling the criteria except for ‘composition/abundance’ metrics, which show comparatively weak relations to the hydromorphological degradation. The metrics are combined to a multi-metric index (MMI) by using ecological quality ratios (EQR) as described by the WFD instead as scoring systems. Therefore, each metric value is related to its reference value (representing reference conditions) and converted into values ranging from 0 to 1. The arithmetic mean represents the MMI. The MMI is highly correlated with the hydromorphological degradation at each spatial scale (about 90 % of variance explained). Hydromorphological stress is correctly detected for 85 % of the total samples (88 % in spring), whereas the overall correspondence with a five-class expert judgement (high, good, moderate, poor, bad) is only 51 % (61 % in spring).

The results show that a multi-metric assessment system based on benthic macroinvertebrates is suited to assess the impact of multiple hydromorphological aspects. Further high-quality data are needed to externally validate the MMI.

Zusammenfassung

Einleitung

Im Dezember 2000 verabschiedete die Kommission der Europäischen Gemeinschaft die EG-Wasserrahmenrichtlinie (WRRL). Als zentrales Ziel wurde die Erreichung eines guten ökologischen Zustands in allen Flüssen und Seen sowie Übergangs- und Küstengewässern der EG-Mitgliedsstaaten formuliert. Die Bewertung des ‚ökologischen Zustands' muss zukünftig anhand so genannter „Biologischer Qualitätskomponenten" (BQE) erfolgen, womit erstmals den Bioindikatoren Fische, Makroinvertebraten, Makrophyten und Phytobenthos sowie Phytoplankton eine zentrale Rolle in der Gewässerbewertung zukommt. Abiotische Faktoren, wie beispielsweise die physikalisch-chemischen Kenngrößen oder Gewässerstrukturparameter, werden lediglich unterstützend für die Bewertung herangezogen. Die WRRL gibt darüber hinaus noch weitere Rahmenbedingungen vor, die zukünftig im Gewässermonitoring zu beachten und erfüllen sind. Danach muss die zukünftige Bewertung gewässertypspezifisch erfolgen, womit die Erarbeitung von Gewässertypologien erforderlich wird. Die WRRL gibt dabei mit den Systemen A und B im Anhang II zentrale Typdeskriptoren vor, die – obligat oder alternativ – in der Gewässertypologie zu berücksichtigen sind. Eine zentrale Rolle kommt danach den WRRL-Deskriptoren Ökoregion, Höhenlage, Geochemie (Geologie) und Gewässergröße zu. Ferner muss die Bewertung zukünftig über einen Vergleich mit typspezifischen Referenzbedingungen in fünf Qualitätsklassen erfolgen: sehr gut, gut, mäßig, unbefriedigend und schlecht. Und schließlich gilt es die Einflüsse der zahlreichen Beeinträchtigungen (Stressoren) auf die aquatischen Bioindikatoren zu berücksichtigen, um einer Bewertung der „ökologischen Qualität" insgesamt auch gerecht zu werden. In den Fließgewässern, die im Rahmen der vorliegenden Dissertation ausschließlich betrachtet werden, stehen dabei Einflüsse einer allgemeinen strukturellen Beeinträchtigung im Vordergrund. Weitere Beeinträchtigungen durch toxische Substanzen oder Gewässerversauerung sind oft regional begrenzt. Aufgrund umfangreicher Maßnahmen zur Verbesserung der Gewässergüte in der Vergangenheit kommt der Belastung durch organische Verschmutzung heute meist nur noch eine untergeordnete Rolle zu. Den meisten in der Vergangenheit in Europa entwickelten biologischen Bewertungsverfahren ist jedoch gemeinsam, dass sie aquatische Wirbellose zur Indikation ausschließlich der organischen Verschmutzung heranziehen und zu einem Gewässergüteindex (Indexsystem) verrechnen. Angesichts der Komplexität der Fließgewässer und der zuvor aufgezeigten ebenso komplexen Einflüsse, die heute auf sie einwirken, ergibt sich daraus die Notwendigkeit, neue Bewertungssysteme zu entwickeln, die in der Lage sind, die vielfältigen Beeinträchtigungen zu trennen und zu indizieren.

In Bezug auf die strukturelle Beschaffenheit der Fließgewässer liegen seit etwa fünf Jahren aus mehreren europäischen Staaten die Ergebnisse umfangreicher Kartierungen vor. Für Deutschland haben die Ergebnisse der Gewässerstrukturgütekartierung eindeutig gezeigt, dass heute erhebliche und weitreichende strukturelle Beeinträchtigungen für nahezu 80 % der kartierten Fließkilometer festzustellen sind. Zu ähnlichen Ergebnissen führte auch ein Vergleich der Erhebungen in Frankreich, dem Vereinigten Königreich und Deutschland. Aus

diesem Grund wurde mit der vorliegenden Dissertation ein Verfahren zur Indikation der strukturellen (hydromorphologischen) Beeinträchtigungen entwickelt.

Die hydromorphologische Degradation wirkt dabei nachweislich auf unterschiedlichen räumlichen Betrachtungsebenen (nachfolgend als „räumliche Skalen“ bezeichnet) wobei eine hierarchische Organisation erkennbar ist. So hat beispielsweise die Hauptnutzungsart im Einzugsgebiet eines Mittelgebirgsbaches (grobskalig) einen Einfluss auf die Ausprägung der uferbegleitenden Gehölze (mittelskalig), die wiederum über den Beschattungsgrad das Aufkommen von aquatischen Makrophyten kontrollieren (feinskalig). Von der Nutzung im Einzugsgebiet über die Regulierung von Flussabschnitten bis hinunter zur vergleichsweise kleinräumigen Sohlbeschaffenheit wirken demnach die variablen Beeinträchtigungen auf unterschiedlichen räumlichen Ebenen. Die Wirkungsweise der komplexen Zusammenhänge ist dabei jedoch noch weitestgehend unbekannt.

Im Rahmen dieser Arbeit wurden die benthischen Makroinvertebraten (Makrozoobenthos) als biologische Indikatorengruppe gewählt, für die auf Grundlage zahlreicher autökologischer Studien heute eine breite und fundierte Datenbasis zu ihren spezifischen Umweltansprüchen existiert. Die Eignung dieser Organismengruppe zur Bioindikation wird unterstrichen durch einen relativ kurzen Entwicklungszyklus der meisten Taxa und einen sehr guten taxonomischen Kenntnisstand, der eine gute Bestimmbarkeit der Taxa sicherstellt.

Um der zuvor aufgezeigten Komplexität der unterschiedlichen Beeinträchtigungen auf die Fließgewässer gerecht zu werden, wurde die Entwicklung eines multimetrischen Bewertungssystems in der vorliegenden Dissertation favorisiert. Ein Beispiel aus der Wirtschaft erläutert die Vorzüge der multimetrischen Bewertung sehr anschaulich: Die Bewertung der gesamtwirtschaftlichen Situation erfolgt dabei über zahlreiche Einzelindices, u. a. Preis-, Einkommens- und Aktienindices oder Zahlen zum Arbeitsmarkt. Dadurch werden die unterschiedlichen Einzelaspekte eines nationalen oder internationalen Wirtschaftssystems bewertbar und können zu einem Gesamtindex zur Bewertung der gesamtwirtschaftlichen Situation verrechnet werden. Überträgt man das Beispiel auf Fließgewässersysteme, so besteht hier die Möglichkeit, die Auswirkungen der einzelnen Beeinträchtigungen über ihre Beziehung zu den ökologischen Kenngrößen (Metrics) der Wirbellosengemeinschaften zu indizieren und über geeignete Metrics zu einem multimetrischen Gesamtindex zu verrechnen. Die Vorzüge einer multimetrischen Bewertung mit dem Makrozoobenthos wurden in den Vereinigten Staaten bereits in den frühen neunziger Jahren des letzten Jahrhunderts erkannt und führten bis heute zu einer breiten Anwendung solcher Bewertungssysteme. In Europa hingegen kamen multimetrische Bewertungssysteme erst mit dem Rückgang der organischen Verschmutzung und der damit verbundenen stärkeren Betrachtung anderer Beeinträchtigungen auf, zuletzt initiiert durch die Maßgaben der WRRL.

Ziel der vorliegenden Dissertation ist es daher, ein multimetrisches System zur Bewertung der hydromorphologischen Degradation in mittelgroßen sandgeprägten Tieflandflüssen zu entwickeln, das den Vorgaben der WRRL entspricht und die weiteren zuvor genannten Rahmenbedingungen berücksichtigt. Damit soll ein Beitrag zur Umsetzung der WRRL geleistet werden, der vor allem die zentrale Vorgabe der organismischen Bewertung berück-

sichtigt. Die Datenanalyse basiert auf einer Zusammenstellung neuer Datensätze, die im Rahmen mehrerer internationaler Forschungsprojekte, u. a. vom Bearbeiter selbst, nach einer einheitlichen Methode erhoben wurden. Neben der Makrozoobenthosbeprobung erfolgte dabei auch eine umfangreiche Kartierung von geologischen, hydrologischen, morphologischen sowie physikalisch-chemischen Parametern. Die Untersuchungen fanden in den Ökoregionen „Westliches Tiefland“ (nur Niederlande) und „Zentrales Tiefland“ der EU-Staaten Schweden, Niederlande, Deutschland und Polen statt.

Die unterschiedlichen Fragestellungen der Dissertation werden in fünf Kapiteln bearbeitet. Kapitel 2 beschäftigt sich zunächst mit typologischen Aspekten und untersucht die Relevanz von verschiedenen durch die WRRL vorgegebenen Typdeskriptoren aus Sicht des Makrozoobenthos und stellt der abiotisch abgeleiteten („top-down“) Topologie eine biozönotisch begründete („bottom-up“) Typologie gegenüber. Die Analyse erfolgte analog für zwei verschiedene Datensätze.

Die Analyse der geo-hydromorphologischen und physikalisch-chemischen Parameter mit dem Ziel der Identifikation geeigneter Variablen zur Charakterisierung und Messung der hydromorphologischen Degradation ist Gegenstand von Kapitel 3. Dabei wird insbesondere auf die Rolle der räumlichen Skalen und ihre Bedeutung für die Bewertung eingegangen.

In Kapitel 4 erfolgt schließlich die gemeinsame Analyse von hydromorphologischen und biozönotischen Datensätzen. Vorrangiges Ziel des Kapitels ist die Identifikation von Beziehungen zwischen hydromorphologischen Variablen und biozönotischen Eigenschaften. Die hydromorphologischen Variablen werden ferner nach ihrer Zuordnung zu drei unterschiedlichen räumlichen Skalen getrennt analysiert: Einzugsgebiet, Flussabschnitt und Probennahmestelle. Die biozönotischen Eigenschaften werden von zwei Datensätzen repräsentiert, einem Taxa- und einem auf Grundlage der Taxaliste berechneten Metricdatensatz.

Eine einzelne Insektenfamilie, die Simuliidae (Kriebelmücken, Diptera), stehen dann im Mittelpunkt von Kapitel 5. Ziel des Kapitels ist die Untersuchung der Kriebelmücken im Hinblick auf eine mögliche Indikation der Auswirkungen der hydromorphologischen Degradation oder einzelner Degradationsaspekte auf diese Insektenfamilie.

Die wesentlichen Befunde der Kapitel 2–5 bilden den konzeptionellen Rahmen für die Entwicklung eines multimetrischen Index, die in Kapitel 6 erfolgt. Dabei wird jeder Entwicklungsschritt in Bezug zu den Vorgaben der WRRL und den relevanten Ergebnissen der vorangegangenen Kapitel erläutert.

Abgrenzung der deutschen und zentraleuropäischen Tiefland-Fließgewässertypen

Die Ziel einer Fließgewässertypologie besteht darin, aus der Fülle der möglichen Ausprägungen und individuellen Charakteristika von Fließgewässern diejenigen herauszufinden, auf deren Grundlage sich einzelne Gewässer zu Typen mit ähnlichen Eigenschaften und dadurch bedingt ähnlichen Organismengemeinschaften zusammenfassen lassen. Gewässertypologien werden zunächst auf Grundlage abiotischer Deskriptoren (Ökoregion, Gewässergröße etc.) „top-down“ erstellt und bedürfen der „bottom-up“ Überprüfung mit biozönotischen Daten, um die Relevanz der Typologie beispielsweise für die aquatische Wirbellosengemeinschaft zu belegen. Auf Basis von 390 Wirbellosendatensätzen (nur Mollusca,

Ephemeroptera, Plecoptera, Odonata, Coleoptera und Trichoptera) aus ganz Deutschland wird die Ökoregion mit Hilfe des Ordinationsverfahrens „Non-metric Multidimensional Scaling“ (NMS) als Hauptkriterium zur Unterscheidung der Datensätze identifiziert: eine klare Trennung des Zentralen Tieflands, Westlichen/Zentralen Mittelgebirges und der Alpen ist erkennbar. Der Befund wird auch durch eine Ähnlichkeitsanalyse „Analysis of Similarity“ (ANOSIM) gestützt ($R = 0{,}409$; $p < 0{,}001$) und unterstreicht die Bedeutung der Ökoregion als Haupttypologiekriterium im Gesamtdatensatz. Eine detaillierte Untersuchung der Tieflanddatensätze mit 123 Fühjahrs- und 109 Sommerproben identifiziert die Gewässergröße als wichtigstes Kriterium innerhalb dieser Ökoregion (ANOSIM: $R = 0{,}330$ im Frühjahr und 0,514 im Sommer, beide höchst signifikant), wobei die jahreszeitliche Trennung der Datensätze auf der relativ heterogenen Zusammenstellung beruht, die eine Überprüfung der saisonalen Diskriminanz ausschließt. Eine typologische Relevanz des dominierenden Sohlsubstrats (Organisch, Sand oder Kies) kann hingegen mit dem vorliegenden Datensatz nicht bestätigt werden (ANOSIM: $R \leq 0{,}200$; $p < 0{,}001$).

Die Analyse wird auf Grundlage eines Datensatzes mit 94 qualitativ hochwertigen und homogenen Aufsammlungen aus dem Zentralen Tiefland Schwedens, der Niederlande, Deutschlands und Polens wiederholt. Die Probennahme erfolgte in drei Jahreszeiten (Frühjahr, Sommer, Herbst) und beinhaltete auch einige Datensätze aus dem Westlichen Tiefland in den Niederlanden. Wiederum wird die Gewässergröße als wichtigstes Unterscheidungskriterium für die Proben im Datensatz ermittelt (ANOSIM: $R = 0{,}492$; $p < 0{,}001$). Im Gegensatz dazu ist der Einfluss der Saison vernachlässigbar (ANOSIM: $R = 0{,}083$; $p = 0{,}060$). Im zweiten Datensatz wird das dominierende Sohlsubstrat (Sand, Kies oder Steine) als zumindest mäßig relevant ermittelt, was jedoch nach detaillierter Prüfung eng mit der unterschiedlichen Ausprägung der schwedischen Gewässer in der Analyse im Vergleich zu den übrigen Proben zusammenhängt. Wird die Analyse ohne die schwedischen Datensätze wiederholt, so ist auch der Einfluss des dominierenden Substrates (Sand oder Kies) vernachlässigbar. Die deskriptive Eigenschaft der Ökoregion (Westliches *vs*. Zentrales Tiefland) kann allerdings nun bestätigt werden (ANOSIM: $R = 0{,}454$; $p < 0{,}001$).

Während die Zuordnung der Proben zur Ökoregion unproblematisch ist, treten insbesondere im deutschen Monitoringdatensatz Probleme bei der Zuordnung der Gewässergröße und des dominierenden Substrattyps auf. Zudem wird die Hypothese aufgestellt, dass die Größenklassifikation der WRRL mit der Trennung Bach-Fluss bei einer Einzugsgebietsgrenze von 100 km² von der Wirbellosenzönose nicht wiedergegeben wird. Eine Clusteranalyse der biozönotischen Daten sollte hier für beide Datensätze Klärung schaffen, wobei die Zuordnung der Proben zu den einzelnen Clustern als weiterer Typdeskriptor für die NMS und zur ANOSIM herangezogen wurde. Die beste Untertrennung des Datensatzes wird danach mit einer Unterteilung in drei Clustergruppen erreicht (ANOSIM: $R = 0{,}736$; $p < 0{,}001$), von denen die schwedischen Datensätze aufgrund der unterschiedlichen Substratverhältnisse eine Gruppe bilden. Die beiden übrigen Gruppen repräsentieren kleine und mittelgroße Fließgewässer bzw. Bäche und Flüsse aus den drei übrigen Ländern. Nach den Ergebnissen der Clusteranalyse erfolgt die Trennung Bach-Fluss bei einer Einzugsgebietsgröße von ca. 50 km².

Mit der vorliegenden Untersuchung kann die deskriptive Eigenschaft der Ökoregion und Gewässergröße bestätigt werden. Jedoch wird auf Grundlage des Makrozoobenthos eine alternative Klassengrenze bei 50 km² Einzugsgebiet für die Trennung kleiner und mittelgroßer Fließgewässer vorgeschlagen. Eine Trennung von überwiegend sand- und kiesgeprägten Fließgewässern ist auf Grundlage der dargestellten Ergebnisse zwar nicht erforderlich, kann aber infolge der möglichen unsicheren Zuordnung im Monitoringdatensatz nicht abschließend geklärt werden. Im anderen Datensatz sind kiesgeprägte Fließgewässer deutlich unterrepräsentiert, so dass zur abschließenden Klärung der Frage eine Wiederholung der Analyse unter Einbeziehung zusätzlicher Daten aus Kiesbächen und –flüssen erforderlich ist.

Die Identifikation und Messung der hydromorphologischen Degradation in europäischen Tieflandflüssen

Die hydromorphologische Degradation stellt heute in vielen europäischen Staaten die häufigste Beeinträchtigungsart dar, wobei es sich um eine Kombination zahlreicher Einflussfaktoren handelt, deren Zusammenspiel sehr komplex sein kann. Auf Ebene unterschiedlicher räumlicher Skalen kommt es zu einer Wechselwirkung von beispielsweise spezifischen Einzugsgebietseigenschaften und kleinräumigen Substratverhältnissen (s. o.). Obwohl diese hierarchisch organisierte Wechselwirkung bekannt ist, konzentrierte sich in der Vergangenheit die Mehrzahl der Untersuchungen zur Aufklärung der zugrunde liegenden Mechanismen und ihrer Wechselwirkungen fast ausnahmslos auf konstante („natürliche“) Faktoren, wie sie unter anderem auch für die Entwicklung von Fließgewässertypologien herangezogen werden.

Durch die Untersuchungen der vorliegenden Dissertation rücken dagegen die variablen („unnatürlichen“) Faktoren in den Mittelpunkt des Interesses. Die meist vom Menschen beeinflussten und in ihrem Charakter sehr variablen Umweltparameter repräsentieren die hydromorphologischen Beeinträchtigungen, während die „natürlichen“ typologisch relevanten Deskriptoren eine solche unnatürlich Beeinträchtigung nicht indizieren können. Um die multiplen Beeinträchtigungen zu quantifizieren und dafür geeignete hydromorphologische Variablen zu identifizieren, wurden im Rahmen der vorliegenden Untersuchung insgesamt 275 Datensätzen zu 147 Gewässerabschnitten mit Angaben zu 106 geo-hydromorphologischen Variablen aus dem AQEM-Projekt einer multivariaten Ordinationsanalyse unterzogen („Non-metric Multidimensional Scaling“). Davon repräsentieren 97 Datensätze die hydromorphologischen Referenzbedingungen und werden für eine Analyse unter Einbeziehung aller Typen verwendet, um neben den bereits bekannten Typdeskriptoren die Eignung weiterer potenziell typologisch relevanter Variablen zu überprüfen. Anhand der Korrelationen ($|r| > 0{,}500$) mit der ersten Ordinationsachse werden drei Gruppen von Einzugsgebietsvariablen identifiziert, mit denen die Trennung der Typen möglich ist: Geologie, Landnutzung und Gewässergröße. Lediglich die deskriptive Eigenschaft der Landnutzung ist neu, wobei sie aber sicher nicht typologisch relevant ist und eher die unterschiedliche Degradation der Einzugsgebiete der untersuchten Fließgewässertypen widerspiegelt. Es zeigt sich, dass die Trennung unterschiedlicher Fließgewässertypen fast ausnahmslos auf der entsprechenden Einzugsgebietsebene oder einer gröberen räumlichen Skala funktioniert. Darüber hinaus lässt sich erkennen, dass sich die in ihrem Charakter va-

riablen hydromorphologischen Parameter nicht zur Beantwortung typologischer Fragestellungen eignen.

Zur Klärung der Frage nach den Indikatoren für die Beeinträchtigungen wird der gesamte Datensatz in eine Analyse einbezogen, deren Ziel die Identifikation von Variablen mit einer engen Beziehung zur hydromorphologischen Beeinträchtigung (einschl. der Landnutzung) ist, die allen Fließgewässertypen gemeinsam sind. Mit Hilfe der Ordination lässt sich die Degradation in Form von Hauptachsen darstellen, die mit den Hauptgradienten der Beeinträchtigung(en) gleichzusetzen sind. Dessen Endpunkte sind bei Einbeziehung aller Typen fast ausschließlich durch die unterschiedlichen Landnutzungen im Einzugsgebiet charakterisiert: Ein hoher Anteil landwirtschaftlicher Nutzflächen ist dabei positiv ($|r| > 0{,}500$), ein hoher Waldanteil hingegen negativ ($|r| > 0{,}700$) mit dem Gradienten korreliert. Auf der kleineren Gewässerabschnittsebene erwiesen sich die Dichte des uferbegleitenden Gehölzsaumes und die damit zusammenhängende Beschattung der Gewässersohle als indikativ; beide Variablen sind negativ mit dem Hauptgradient der Beeinträchtigung korreliert ($|r| \geq 0{,}600$). Selbst unter Einbeziehung stark degradierter Fließgewässer bleibt bei Betrachtung unterschiedlicher Fließgewässertypen die Rolle der Einzugsgebietsparameter dominant. Erst wenn der räumliche Bezug, der durch die Probenauswahl vorgegeben ist, verkleinert wird, ändern sich die Verhältnisse. Wird die Degradationsanalyse auf die drei deutschen Gewässertypen beschränkt (organischer Bach, Sandbach Sandfluss), so ist auch ein wesentlich stärker ausgeprägter Gradient der hydromorphologischen Beeinträchtigung erkennbar. Der Gradient ist hoch mit dem Totholzanteil auf der Gewässersohle, der Dichte des Ufergehölzsaumes, der Beschattung und dem Grad der Uferbefestigung korreliert (alle $|r| > 0{,}600$) und verdeutlicht die wechselnden Rollenverhältnisse auf Ebene der räumlichen Skalen in Abhängigkeit vom räumlichen Bezug der Eingangsgrößen, hier der unterschiedlichen Datensätze. Während zuvor bei der Analyse aller Typen die Einzugsgebietsparameter dominierten, sind es nun insbesondere die mittelskaligen hydromorphologischen Variablen mit einer engen Beziehung zum Fließgewässerabschnitt. Die sich daraus ableitende Hypothese, dass eine weitere Verkleinerung des räumlichen Bezugs der Eingangsgrößen auch die Bedeutung der feinskaligen kleinräumigen hydromorphologischen Variablen hervorhebt, kann mit der vorliegenden Untersuchung bestätigt werden. So werden unter anderem auch Variablen mit direktem Bezug zur Probennahmestelle identifiziert, wenn die Analyse auf einen Fließgewässertyp beschränkt ist, im vorliegenden Fall auf die mittelgroßen Tiefland-Sandflüsse.

Insgesamt war damit eine hierarchische Abfolge der als indikativ ermittelten räumlichen Skalen feststellbar, die positiv mit dem räumlichen Bezug des Datensatzes zusammenhing. Es zeigt sich ferner, dass die Indikation der Beeinträchtigungen insbesondere über mittelskalige (abschnittsbezogene) Variablen möglich ist, was der Betrachtung eines einzelnen Gewässertyps oder einer Gruppe natürlicherweise morphologisch ähnlicher Gewässertypen entspricht. Zur Klärung der Frage nach den geeigneten hydromorphologischen Indikatoren werden zunächst die an den Gradientenenden im Ordinationsdiagramm lokalisierten Probenpunkte bzw. Proben ausgewählt. Sie repräsentieren quasi die hydromorphologischen Extreme im Datensatz. Analysiert man dann die Ausprägung der mit dem Degradationsgradienten hoch korrelierten Variablen für diese Proben, so zeigen sich signifikante Unterschiede. Damit

liegen geeignete Indikatoren für die Identifizierung der hydromorphologischen Degradation vor, die die Grundlage für die Berechnung eines Strukturindex bilden. Für den Index werden insgesamt 20 Einzelvariablen zu acht Gruppenindices verrechnet, die unterschiedliche Aspekte der hydromorphologischen Beeinträchtigung auf unterschiedlicher räumlicher Ebene repräsentieren: Landnutzung, Abflussmodifikation/Laufveränderung, Gewässereintiefung, Ufersaumausprägung, Beschattungsgrad, Uferbefestigung, Totholzanteil und Anteil organischer Substrate. Der Unterschied zur bereits existierenden Gewässerstrukturkartierung besteht insbesondere darin, dass die Auswahl der verwendeten Parameter für den Index in der vorliegenden Untersuchung eng an den Gradienten der Beeinträchtigungen geeicht ist.

Die Verknüpfung von Taxa, Metrics, Hydromorphologie und Landnutzung auf Ebene unterschiedlicher räumlicher Skalen mit Hilfe der multivariaten Analyse

Nachdem die Wirkung der hydromorphologischen Degradation in Abhängigkeit von den unterschiedlichen räumlichen Betrachtungsebenen gezeigt und die relevanten hydromorphologischen Variablen identifiziert wurden, stellt sich die Frage, ob und wie sich die Beeinträchtigungen auf die Makroinvertebraten auswirken. Ausgangshypothese ist auch hier, dass die Reaktion der Wirbellosengemeinschaft auf die hydromorphologische Degradation vom räumlichen Bezug der Beeinträchtigung abhängt, wobei die zuvor dargestellten Befunde die Rolle der Variablen auf Ebene eines Gewässerabschnitts hervorheben. Aus den eingangs dargestellten Vorteilen von multimetrischen Systemen lässt sich ferner die Hypothese ableiten, dass Metrics sich besser zur Indikation der hydromorphologischen beeinträchtigungen eignen als einzelne Taxa. Metrics repräsentieren ökologische Kenngrößen der Wirbellosengemeinschaft (Ernährungstypen, Habitatpräferenzen, Dominanzverhältnisse, Diversitätsmaße etc.) und lassen damit Rückschlüsse auf die Störung der Gemeinschaft und ihre Funktion im Gewässerökosystem zu. Ziel der vorliegenden Untersuchung ist es daher auch, die Eignung der Taxa und Metrics zur Indikation der unterschiedlichen Beeinträchtigungen zu vergleichen.

Dazu werden drei unterschiedliche Datensätze mit je 144 Proben von 75 Gewässerabschnitten in Schweden, den Niederlanden, Deutschland und Polen miteinander verrechnet. 1) 51 hydromorphologische Variablen gruppiert nach vier räumlichen Skalen (Ökoregion, Einzugsgebiet, Gewässerabschnitt und Probennahmestelle), 2) 244 Taxa und 3) 84 Metrics. Die multivariate statistische Auswertung erfolgt mit einer direkten Gradientenanalyse, deren Vorteil darin besteht, dass sie den Hauptgradienten im biozönotischen Datensatzes am abiotischen (hydromorphologischen) Hauptgradienten ausrichtet. Damit wird vor allem der Erklärungsanteil der abiotischen Variablen an der biozönotischen Variabilität analysiert. Die Analyse der vier räumlichen Skalen jeweils für die biozönotischen Datensätze wird in acht Gradientenanalysen getrennt durchgeführt. Danach wird, sowohl für die Taxa als auch für die Metrics, ein großer Anteil der insgesamt erklärten Varianz von den als „Ökoregion“ zusammengefassten konstanten Variablen (Längen-/Breitengrad, Gewässergröße, Saison) erklärt. Es sind aber weniger saisonale Aspekte als vielmehr geographische Unterschiede feststellbar, die auf Basis der Artengemeinschaft zu deutlichen Unterschieden der schwedischen und übrigen Gewässer führen. Grund hierfür ist unter anderem das ausschließliche Vorkommen einiger Insektentaxa in den schwedischen Gewässern. Die Variabilität im

Metricdatensatz wird dagegen unter anderem von der Gewässergröße beeinflusst. Die Ergebnisse bestätigen die biozönotische Relevanz der konstanten Variablen (Typdeskriptoren) für die Makroinvertebraten auf taxonomischer und funktionaler Ebene. Da es jedoch vorrangiges Ziel dieser Untersuchung ist, den Einfluss der hydromorphologischen Degradation zu untersuchen, werden die konstanten Variablen in den weiteren Analysen als so genannte Co-Variablen berücksichtigt. Dadurch wird ihr Erklärungsanteil ermittelt und gleichzeitig von dem der übrigen (Beeinträchtigungs-) Variablen in der jeweiligen Analyse subtrahiert.

Die Ergebnisse zeigen, dass die mittelskaligen hydromorphologischen Variablen unter allen räumlichen Bezugsebenen den größten Erklärungsanteil an der biozönotischen Variabilität haben. Dies gilt sowohl für den Taxa- als auch für den Metricdatensatz, wobei die Erklärungsanteile für den Metricdatensatz etwas höher liegen. Die Ordinationsdiagramme zu den einzelnen Analysen bestätigen grundsätzlich die vorher ausschließlich auf Basis der hydromorphologischen Variablen ermittelten Gradienten. Eine besondere Bedeutung haben dabei der Grad des Uferverbaus sowie der Anteil organischer Substrate auf der Sohle, hier insbesondere der Totholzanteil. Für diese morphologischen Eigenschaften wird eine starke Beziehung zur Makroinvertebratenzönose identifiziert. Der direkte Bezug zur Fauna in der vorliegenden Untersuchung zeigt aber auch die besondere Rolle der vergleichsweise kleinräumigen Substratverhältnisse für die Indikation der hydromorphologischen Degradation. Die Analysen identifizieren beispielsweise einen Zusammenhang zwischen einem hohen Anteil steiniger Substrate oder aquatischer Makrophyten einerseits und bestimmten Taxa und Metrics andererseits, was auf eine Eignung dieser biozönotischen Komponenten zur Indikation einer Beeinträchtigung hinweist. Das Vorkommen beider Substrate in den untersuchten Gewässertypen ist zudem eng mit einer morphologischen Beeinträchtigung durch Uferverbau (Steinschüttungen) oder einer hydrologischen Beeinträchtigung durch Rückstau (erhöhtes Makrophytenwachstum) verknüpft. Die Ergebnisse legen auch hier den Schluss nahe, dass die hydromorphologische Degradation ebenso einer hierarchischen Organisation unterliegt, wie sie bereits seit fast 20 Jahren für die konstanten (typologische relevanten) Variablen bekannt und breit diskutiert wird. Damit besteht grundsätzlich die Möglichkeit, die multiple hydromorphologische Beeinträchtigung über eine geschickte Auswahl einzelner Variablen integrativ zu erfassen. Das zuvor gezeigte Beispiel verdeutlicht aber auch, dass diese Beziehungen gewässertypspezifisch untersucht werden müssen, da beispielsweise das Vorkommen von Steinen in einem Bergbach noch keinen Rückschluss auf eine Beeinträchtigung zulässt.

Ein weiteres Ziel dieser Untersuchung besteht darin, die Struktur der Wirbellosengemeinschaft im Hinblick auf besonders geeignete taxonomische oder funktionale Aspekte zu beleuchten. Die Analyse der taxonomischen Zusammensetzung zeigt dabei, dass die Taxaliste deutlich von den Insektenordnungen Trichoptera (Köcherfliegen) und Diptera (Zweiflügler) dominiert wird; beide Ordnungen stellen rund 50 % aller Taxa. Die Dominanz spiegelt sich auch in der Indikationseignung beider Ordnungen wider; sie stellen 42–55 % der als indikativ für die drei untersuchten räumlichen Skalen identifizierten Taxa. Vor allem die Köcherfliegen eignen sich überdurchschnittlich gut zur Indikation auf Ebene der feinskaligen Substratzusammensetzung, daneben aber auch die Oligochaeta (Wenigborster) und Crusta-

cea (Krebstiere). Eine ähnliche Analyse mit dem Metricdatensatz zeigt, dass die funktionalen Aspekte (Strömungspräferenzen, Ernährungstypen, Habitatpräferenzen etc.) mit rund 45 % dominieren. Die übrigen 55 % verteilen sich etwa gleichmäßig auf die Gruppen „Sensitivität/Toleranz“, „Zusammensetzung/Abundanz“ und „Artenreichtum/Diversität“, die jeweils unterschiedliche Aspekte der Biozönose repräsentieren. Die funktionalen Metrics zeigen ferner überdurchschnittlich hohe Relationen zur Degradation auf Ebene aller drei räumlichen Skalen. Im Hinblick auf die für die Indikation der Beeinträchtigungen so wichtigen abschnittsbezogenen hydromorphologischen Variablen zeigen sie die größte Indikationsstärke, während Diversitätsmaße auf der mittelskaligen Ebene unterdurchschnittlich gut geeignet sind. Letztere eignen sich vor allem zur Indikation der feinskaligen Substratverhältnisse.

Die vorliegende Untersuchung führt zu der Schlussfolgerung, dass sich Taxa und Metrics grundsätzlich zur Indikation der multiplen hydromorphologischen Beeinträchtigungen eignen und damit den Einfluss der hydromorphologischen Degradation integrativ bewerten können. Metrics haben jedoch gegenüber der ihnen zugrunde liegenden Taxaliste zwei entscheidende Vorteile: 1) Sie sind geeignet, die funktionalen Aspekte der Wirbellosengemeinschaft zu integrieren und damit eine eventuelle Störung der Funktion im Fließgewässersystem zu indizieren. 2) Sie sind weniger anfällig für regionale oder saisonale Aspekte, die das Vorkommen einzelner Taxa oder taxonomischer Gruppen beeinflussen können. Dies gilt auch für taxonomische Unterschiede infolge unterschiedlicher Bestimmungskenntnisse. Es kann aber auch gezeigt werden, dass vor allem die beiden Insektenordnungen Trichoptera und Diptera ein hohes Indikationspotenzial vereinen.

Die Einfluss der hydromorphologischen Degradation auf Kriebelmücken (Diptera, Simuliidae)

Kriebelmücken sind weit verbreitet und kommen mit zahlreichen Arten in fast allen Fließgewässern vor. Damit erfüllen sie eine wichtige Grundvoraussetzung für die Eignung als Indikatororganismen dieser Gewässerkategorie. Ihre Sensitivität gegenüber der organischen Verschmutzung und Gewässerversauerung ist heute weitestgehend bekannt, während der Wissensstand zum Einfluss der hydromorphologischen Degradation auf diese Insektenfamilie noch als lückenhaft angesehen werden kann. Die vorliegende Untersuchung hatte das vorrangige Ziel zur Schließung dieser Wissenslücke beizutragen und zu prüfen, ob sich Kriebelmücken zur Indikation der hydromorphologischen Beeinträchtigungen eignen. Darüber hinaus steht die Identifikation der zugrunde liegenden Beziehungen zu den hydromorphologischen Variablen im Mittelpunkt. Für die Analyse steht ein Datensatz mit 189 Aufsammlungen von 86 Untersuchungsabschnitten und insgesamt 21 Taxa zur Verfügung. Der Datensatz umfasst fünf deutsche Fließgewässertypen und beinhaltet auch Proben zu zwei Typen der Zentralen Mittelgebirge.

Mit Hilfe der Ordinationsmethode „Non-metric Multidimensional Scaling“ wird zunächst der gesamte Datensatz analysiert. Es zeigt sich, dass das Vorkommen der Gattung *Prosimulium* auf die Mittelgebirgsproben begrenzt ist, während die Arten der Gattung *Simulium* in beiden Ökoregionen vorkommen. Anhand der Artenstetigkeit wird mit *S. erythrocephalum* jedoch eine weitere „Mittelgebirgsart“ und mit *S. vernum* eine „Tieflandart“ identifiziert.

Dieser ökoregional bedingte Unterschied unterstreicht die Notwendigkeit, in Analysen zur Identifikation von hydromorphologischen Beeinträchtigungen immer auch konstante („natürliche“) Deskriptoren einzuschließen um deren natürlichen Einfluss zu erkennen und von der unnatürlichen Beeinträchtigung trennen zu können. Eine Trennung wird in der vorliegenden Untersuchung durch die separate Betrachtung des Mittelgebirgs- und Tieflanddatensatzes erreicht.

Die Untersuchung der Eignung der Simuliiden zur Indikation der Beeinträchtigungen erfolgt in zwei Schritten. Zunächst wird der Datensatz mit Hilfe des zuvor entwickelten Deutschen Strukturindex in hydromorphologisch belastete und unbelastete Proben aufgeteilt. Die Gesamtartenzahl ist in den unbelasteten Gewässern insgesamt signifikant höher. Auf Typenebene kann dies aber nur für zwei Fließgewässertypen bestätigt werden. Der Vergleich der Artenstetigkeit zeigt lediglich für eine Art (*S. lineatum*) eine signifikante Präferenz für unbelastete Fließgewässerabschnitte, die zudem noch mehrheitlich dem Tiefland zuzuordnen sind. Die Ergebnisse zeigen, dass sich Kriebelmücken zur Indikation der hydromorphologischen Beeinträchtigungen potenziell eignen, belegen aber auch, dass die Indikation eher der gesamten Artengemeinschaft zukommt (mittlere Artenzahl) als einzelnen Indikatorarten.

Lineare multiple Regressionsanalysen mit den drei häufigsten Taxa (*Prosimulium* spp., *P. hirtipes* und *Simulium* spp.) haben zum Ziel, die Beziehung zu einzelnen hydromorphologischen Variablen zu identifizierten. Das Vorkommen von *Simulium* spp. wird dabei vor allem durch die mittlere Strömungsgeschwindigkeit, den Grad der Beschattung und den Makrophytenanteil im entsprechenden Gewässerabschnitt bestimmt ($F = 10{,}10$; $p < 0{,}001$; $R^2 = 0{,}47$). Eine herausragende Rolle im Regressionsmodell spielt dabei die Strömungsgeschwindigkeit (*beta* = 0,552). Für die Gattung kann aber auch eine Beziehung zur Anzahl der organischen Substrate und hier vor allem zum Totholzanteil gezeigt werden. Die Präsenz von *Prosimulium* spp. ist nach den Ergebnissen der Regressionsanalyse ebenfalls eng mit dem Totholzanteil im beprobten Abschnitt sowie zusätzlich mit dem Grad der urbanen Landnutzung im Einzugsgebiet verknüpft.

Die Ergebnisse der vorliegenden Analyse unterstreichen die Eignung der Simuliidae zur Indikation des hydromorphologischen Zustands, denn die Mehrzahl der identifizierten Variablen lässt Rückschlüsse auf eine eventuell vorliegende hydromorphologische Beeinträchtigung zu. Ein Fehlen von Totholz in den untersuchten Tieflandtypen bedeutet z. B. bereits eine gewisse Degradation, da es hier natürlicherweise überall vorkommen müsste. Insgesamt ist die Indikationsleistung der Simuliiden aber vermutlich eher auf die Extrema gerichtet, d. h. zum einen auf die eher referenznahen Bedingungen mit einer sehr divers ausgeprägten Artengemeinschaft und zum anderen auf bereits stark degradierte Bereiche mit einer deutlich verminderten Strömungsgeschwindigkeit und einem fast völligen Fehlen von natürlicherweise vorkommenden strömungsexponierten Anheftungssubstraten. Zur Klärung dieser Annahme sind jedoch weitere Untersuchungen notwendig, die insbesondere die kleinräumige Verteilung der Tiere berücksichtigen und auf eine Probennahmetechnik zurückgreifen, die zur quantitativen Erfassung dieser Insektenfamilie geeignet ist. Das im Rahmen dieser Dissertation fast ausschließlich angewandte Multi-Habitat Sampling ist hierzu nicht geeignet.

Die Entwicklung eines multimetrischen Index zur Bewertung des Einflusses der hydromorphologischen Degradation auf benthische Makroinvertebraten

Im Mittelpunkt der vorliegenden Dissertation steht die Entwicklung eines multimetrischen Index, der geeignet ist, die zuvor aufgezeigten vielfältigen Aspekte der hydrologischen und morphologischen Beeinträchtigungen zu bewerten, indem er ihre Auswirkungen auf die Wirbellosenzönose beurteilt. Die Vorteile eines multimetrischen Index und die wesentlichen Rahmenbedingungen für die Indexentwicklung sind eingangs bereits genannt worden. Die Indexentwicklung beruht auf einem Datensatz mit 82 Makrozoobenthosproben aus 40 mittelgroßen Sandflüssen des Zentralen Tieflands. Der Datensatz repräsentiert damit einen natürlicherweise hydromorphologisch relativ homogenen Fließgewässertyp innerhalb einer Ökoregion und Größenklasse. Als untere Größengrenze wird hier jedoch auf Grundlage der biozönotischen Analysen und abweichend von der Klassifikation der WRRL 50 km² definiert. Mit den Taxalisten werden insgesamt 84 Metrics berechnet die den vier Metricgruppen „Sensitivität/Toleranz“, „Funktion“, „Artenreichtum/Diversität“ und „Zusammensetzung/Abundanz“ zuzuordnen sind. Zur Identifikation der geeigneten Indikatoren wird eine Redundanzanalyse (direkte Gradientenanalyse) durchgeführt, wobei die 49 hydromorphologischen Eingangsvariablen in die räumlichen Skalen „Einzugsgebiet“, „Gewässerabschnitt“ und „Probennahmestelle“ aufgeteilt und getrennt analysiert werden.

Die drei Analysen identifizieren jeweils einen deutlichen Gradient der hydromorphologischen Beeinträchtigung (1. Achse im Ordinationsdiagramm): Auf grobskaliger Ebene (nur Landnutzung) wird eine Beeinträchtigung vor allem durch den Anteil urbaner Flächen und Intensivweiden im Einzugsgebiet charakterisiert. Innerhalb eines ein bis mehrere km langen Fliegewässerabschnitts dominieren der Grad der Uferbefestigung sowie die Lauf- und Abflussregulierung die Beeinträchtigungen, und feinskalig lässt sich die Beeinträchtigung insbesondere über den der Anteil der steinigen Substrate identifizieren.

Aufgrund der Stärke der Beziehung („metrics fit“) der Metrics zu den Gradienten werden so genannte Kandidatenmetrics („candidate metrics“) ausgewählt, die die Grundlage zur Auswahl der Kernmetrics („core metrics“), den Bestandteilen des multimetrischen Index, bilden. Die Auswahl der Kernmetrics erfolgt nach drei Gesichtspunkten: Danach muss ein Metric eine 1) hohe Korrelation zu 2) möglichst allen räumlichen Ebenen der Beeinträchtigung haben. Die Kernmetrics sollen zudem 3) alle vier Metricgruppen repräsentieren. Die Endauswahl enthält fünf Metrics aus drei Metricgruppen und berücksichtigt gegenüber den unterschiedlichen Beeinträchtigungen sensitive und -tolerante Taxa (Deutscher Faunaindex Typ 15, Taxazahl zum Faunaindex), Zonierungspräferenzen (Litoralbesiedler). Strömungspräferenzen (Rheophile) und Habitatpräferenzen (Pelalbesiedler). Mit der Auswahl ist einen Indikation möglich, die die drei untersuchten räumlichen Bezugsebenen einschließt.

Die Verrechnung der einzelnen Metricwerte erfolgt über so genannte „Ecological Quality Ratios“ (EQR), wobei der Mittelwert schließlich den multimetrischen Index ergibt. Aufgrund der Herleitung vereint der Index die unterschiedlichen Aspekte der hydromorphologischen Degradation. Er ist mit den Beeinträchtigungen auf allen drei räumlichen Bezugsebenen signifikant und hoch korreliert ($|r| > 0{,}900$) und eignet sich damit zur integrativen Bewertung mit Hilfe des Makrozoobenthos.

Eine erste Validierung wird Anhand des Vergleichs mit Experteneinstufungen vorgenommen: Für den Gesamtdatensatz wird eine Übereinstimmung in 85 % der Fälle bezüglich der Einstufung belastet/unbelastet festgestellt. Der Wert liegt jedoch mit 51 % für den Vergleich anhand der fünfstufigen Klassifikation erheblich niedriger. Das Ergebnis verdeutlicht den weiteren Validierungsbedarf, für den allerdings ebenso hochwertige Daten notwendig sind. Das Ergebnis verdeutlicht aber auch, dass mit der Datenqualität eine Bewertung möglich wird, die annähernd 90 % der objektiv ermittelten hydromorphologischen Degradation erklärt.

References

AddinSoft SARL (2002): XLSTAT 5.2. Addinsoft, Paris.

Agence de l'Eau Rhin Meuse (1996) :Outil d'évaluation de la qualité du milieu physique. Metz.

Alba-Tercedor J. & A. Sanchez-Ortega (1988): Un metodo rapido y simple para evaluar la calidad biologica de las agues corrientes basado en el de Hellawell (1978). Limnetica 4: 51–56.

Allan, J. D. (1995): Stream ecology. Structure and function of running waters. Chapman & Hall, London, 388 pp.

Allan, J. D. & L. B. Johnson (1997): Catchment-scale analysis of aquatic ecosystems. Freshwater Biology 37: 107–111.

Allan, J. D., D. L. Erickson & J. Fay (1997): The influence on catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37: 149–161.

AQEM consortium (2002): Manual for the application of the AQEM system. A comprehensive method to assess European streams using benthic macroinvertebrates, developed for the purpose of the Water Framework Directive. EVK1-CT1999-00027, Version 1.0. Available via the Internet from www.aqem.de.

Armitage, P. D., D. Moss, J. F. Wright & M. T. Furse (1983): The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. Water Research 17: 333–347.

Barbour, M. T., J. B. Stribling & J. R. Karr (1995): Multimetric approach for establishing biocriteria. Biological assessment and criteria. In: W. S. Davies & T. P. Simon (eds.): Tools for water resource planning and decision making. CRC Press, Boca Raton, pp. 63–77.

Barbour, M. T., J. Gerritsen, G. E. Griffith, R. Frydenborg, E. McCarron, J. S. White & M. L. Bastian (1996): A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal of the North American Benthological Society 15(2): 185–211.

Barbour M. T., J. Gerritsen, B. D. Snyder & J. B. Stribling (1999): Rapid Bioassessment Protocols for use in wadeable streams and rivers. Periphyton, Benthic Macroinvertebrates, and Fish. EPA 841-B-99-002, U.S. Environmental Protection Agency, Office of Water, Washington D. C.

Barr, W. B. (1982): Attachment and silk of Simulium vittatum Zetterstedt (Diptera: Simuliidae). MSc. Thesis, University of Alberta, Edmonton.

Beals, E. W. (1984): Bray-Curtis ordination: an effective strategy for analysis of multivariate ecological data. Advances in Ecological Research 14: 1–55.

Beisel, J.-N., P. Usseglio-Polatera, S. Thomas & J.-C. Moreteau (1998a): Effects of mesohabitat sampling strategy on the assessment of stream quality with benthic invertebrate assemblages. Archiv für . Hydrobiologie 142: 493–510.

Beisel, J.-N., P. Usseglio-Polatera, S. Thomas & J.-C. Moreteau (1998b): Stream community structure in relation to spatial variation: the influence of mesohabitat characteristics. Hydrobiologia 389: 73–88.

Benke, A. C., G. E. Willeke, F. K. Parrish & D. L. Stites (1981): Effects of urbanization on stream ecosystems. A-005-GA, Office of Water Research and Technology, U.S. Department of the Interior.

Benke, A. C., R. L. Henry (III.), D. M. Gillespie & R. J. Hunter (1985): Importance of snag habitat for animal production in southeastern streams. Fisheries 10: 8–13.

Blocksom, K. A. (2003): A performance comparison of metric scoring methods for a multimetric index for Mid-Atlantic highland streams. Environmental Management 31(5): 670–682.

Böhmer, J., C. Rawer-Jost, A. Zenker, C. Meier, C. K. Feld, R. Biss & D. Hering (2004): Assessing streams in Germany with benthic invertebrates: Development of a multimetric invertebrate based assessment system. Limnologica 34: 416–432.

Borcard, D., P. Legendre & P. Drapeau (1992) : Partialling out the spatial component of ecological variation. Ecology 73(3): 1045–1055.

Brabec, K., S. Zahrádková, D. Němejcová, P. Pařil, J. Kokeš & J. Jarkovský (2004): Assessment of organic pollution effect considering differences between lotic and lentic stream habitats. Hydrobiologia 516: 331–346.

Braukmann, U. (1987): Zoozönologische und saprobiologische Beiträge zu einer allgemeinen regionalen Bachtypologie. Archiv für Hydrobiologie, Supplement 26, Stuttgart. 355 pp.

Braukmann, U. & R. Bis (2004): Conceptual study – An improved method to assess acidification in German streams by using benthic macroinvertebrates. Limnologica 34(4): 433–450.

Bray, J. R. & J. T. Curtis (1957): An ordination of the upland forest communities of southern Wisconsin. Ecological Monographs 27(4): 325–349.

Brewin, P. A., T. M. L. Newman & S. J. Ormerod (1995): Patterns of macroinvertebrate distribution in relation to altitude, habitat structure and land use in streams of the Nepalese Himalaya. Archiv für Hydrobiologie 135: 79–100.

Briem, E. (2003): Gewässerlandschaften der Bundesrepublik Deutschland – Morphologische Merkmale der Fließgewässer und ihrer Auen. ATV-DVWK Arbeitsbericht GB-1, Hennef. 176 pp.

Brookes, A. (1987): The distribution and management of channelized streams in Denmark. Regulated Rivers: Research and Management 1: 3–16.

Brosse, S., C. J. Arbuckle & C. R. Townsend (2003): Habitat scale and biodiversity: influence of catchment, stream reach and bedform scales on local invertebrate diversity. Biodiversity and Conservation 12: 2057–2075.

Brusven, M. A. & K. V. Prather (1974): Influence of stream sediments on distribution of macrozoobenthos. Journal of the Entomological Society of British Columbia 71: 25–32.

Bundesanstalt für Geowissenschaften und Rohstoffe (1993): Geologische Karte der Bundesrepublik Deutschland. Bundesanstalt für Geowissenschaften und Rohstoffe, Hannover.

Clarke, K. R. (1993): Non-parametric multivariate analysis of changes in community structure. Australian Journal of Ecology 18: 117–143.

Clarke, K. R. & M. Ainsworth (1993): A method of linking multivariate community structure to environmental variables. Marine Ecology Progress Series 92: 205–219.

Clarke, K. R. & R. N. Gorley (2001): PRIMER v5 user manual/tutorial. PRIMER E, Plymouth.

Clarke, K. R. & R. M. Warwick (2001): Change in marine communities: an approach to statistical analysis and interpretation. 2nd edition. PRIMER-E Ltd., Plymouth.

Corkum, L. D. (1992): Spatial distributional patterns of macroinvertebrates along rivers within and among biomes. Hydrobiologia 239: 101–114.

CSN (1998): Water quality – Classification of surface water quality. Czech Technical State Standard CSN 75 7221. Czech Standards Institute, Prague. 10 pp.

Dahl, J., R. K. Johnson & L. Sandin (2004): Detection of organic pollution of streams in southern Sweden using benthic macroinvertebrates. Hydrobiologia 516: 161–172.

Davies, N. M., R. H. Norris & M. C. Thomas (2000): Prediction and assessment of local stream habitat features using large-scale catchment characteristics. Freshwater Biology 45: 343–369.

Davis, W. S. & T. P. Simon (eds.) (1995): Biological assessment and criteria. Tools for water resource planning and decision making. Boca Raton, Florida.

De Pauw, N. & G. Vanhooren (1983): Method of biological quality assessment of watercourses in Belgium. Hydrobiologia 100: 153–168.

DEV (1992): Biologisch-ökologische Gewässeruntersuchung: Bestimmung des Saprobienindex (M2). Deutsche Einheitsverfahren zur Wasser-, Abwasser- und Schlammuntersuchung. VCH Verlagsgesellschaft, Weinheim. 13 pp.

Dudley, T. & N. H. Anderson (1982): A survey of invertebrates associated with wood debris in aquatic habitats. Melanderia 39: 1–21.

Dufrêne, M. & P. Legendre (1997): Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecological Monographs 67: 345–366.

Ellenberg, H. (1996): Vegetation Mitteleuropas mit den Alpen in ökologischer, dynamischer und historischer Sicht. Ulmer, Stuttgart. 1095 pp.

EU commission (2000): Directive 2000/60/EC of the European Parliament and of the Council of 23rd October 2000 establishing a framework for community action in the field of water policy (EU Water Framework Directive). Official Journal of the European Communities L327, Brussels. 72 pp.

Eymann, M. & W. G. Friend (1988): Behaviours of larvae of the blackflies Simulium vittatum and S. decorum (Diptera: Simuliidae) associated with establishing and maintaining dispersion patterns on natural and artificial substrates. Journal of Insect Behaviour 1(2): 169–186.

Feld, C. K., E. Kiel & M. Lautenschläger (2002a): The indication of morphological degradation of streams and rivers using Simuliidae. Limnologica 32: 273–288.

Feld, C. K., S. Pauls, M. Sommerhäuser & D. Hering (2002b): Biozönotische Bewertung der ökologischen Qualität am Beispiel norddeutscher Tieflandfließgewässer. Deutsche Gesellschaft für Limnologie (DGL) – Tagungsbericht 2001 (Kiel): 75–80. Tutzing.

Feld, C. K. & B. Bis (2003): Was ist der sehr gute ökologische Zustand nach EU-WRRL für mittelgroße Sandflüsse des Tieflands. Deutsche Gesellschaft für Limnologie (DGL) – Tagungsberichte 2002 (Braunschweig): 19–24. Werder.

Feld, C. K. (2004): Identification and measure of hydromorphological degradation in Central European lowland streams. Hydrobiologia 516: 69–90.

Feld, C. K. & P. Rolauffs (2005): Zur Erfordernis der taxonomischen Bereinigung (‚taxonomical adjustment') als Grundlage für den Vergleich von Taxalisten. Deutsche Gesellschaft für Limnologie (DGL) – Tagungsbericht 2004 (Potsdam): 408–412. Berlin.

Feminella, J. W. (2000): Correspondence between stream macroinvertebrate assemblages and 4 ecoregions of the southeastern USA. Journal of the North American Benthological Society 19(3): 442–461.

Fitzpatrick, F. A., B. C. Scudder, B. N. Lenz & D. J. Sullivan (2001): Effects of multi-scale environmental characteristics on agricultural stream biota in eastern Wisconsin. Journal of the American Water Resources Association 37: 1489–1507.

Fore, L. S., J. R. Karr & R. W. Wissemann (1996): Assessing invertebrate responses to human activities: Evaluating alternative approaches. Journal of the North American Benthological Society 15: 212–231.

Franquet, E., S. Dolédec & D. Chessel (1995) : Using multivariate analyses for separating spatial and temporal effects within species-environment relationship. Hydrobiologia 300/301: 425–431.

Friedrich, G. & V. Herbst (2004): Eine erneute Revision des Saprobiensystems – weshalb und wozu? Acta hydrochimica et hydrobiologica 32(1): 61-74.

Frissell, C. A., W. J. Liss, C. E. Warren & M. D. Hurley (1986): A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10: 199–214.

Furse, M. T., J. F. Wright & P. D Armitage (1984): The influence of seasonal and taxonomic factors on the ordination and classification of running-water sites in Great Britain and on the prediction of their macro-invertebrate communities. Freshwater Biology 14: 257–280.

Gerritsen, J., M. T. Barbour & K. King (2000): Apples, oranges, and ecoregions: on determining pattern in aquatic assemblages. Journal of the North American Benthological Society 19(3): 487–496.

Griffith, M. B., P. R. Kaufmann, A. T. Herlihy & B. H Hill (2001): Analysis of macroinvertebrate assemblages in relation to environmental gradients in Rocky Mountain streams. Ecological Applications 11: 489–505.

Gurnell, A. M., K. J. Gregory & G. E. Petts (1995): The role of coarse woody debris in forest aquatic habitats: implications for management, Aquatic Conservation: Marine Freshwater Ecosystems: 143–166.

Haase, P. (1998): Köcherfliegen als Charakterarten colliner und submontaner Kalkbäche in den deutschen Mittelgebirgen. Lauterbornia 34: 113–119.

Haase, P., A. Sundermann, C. K. Feld, D. Hering, A. Lorenz, C. Meier, J. Böhmer, C. Rawer-Jost & A. Zenker (2004): Validation der Fließgewässertypologie Deutschlands, Ergänzung des Datenbestandes und Harmonisierung der Bewertungsansätze der verschiedenen Forschungsprojekte zum Makrozoobenthos zur Umsetzung der Europäischen Wasserrahmenrichtlinie (Modul Makrozoobenthos). LAWA report O3.02, unpubl. 83 pp. + Appendix.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Auman, J. R. Sedell, G. W. Lienkaemper, K. Cromack Jr. & K. W. Cummins (1986): Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15: 133–302.

Hawkins, C. P. & R. H. Norris (2000): Performance of different landscape classifications for aquatic bioassessments: introduction to the series. Journal of the North American Benthological Society 19(3): 367–369.

Hawkins, C. P., R. H. Norris, J. Gerritsen, R. M. Hughes, S. K. Jackson, R. K. Johnson & R. J. Stevenson (2000): Evaluation of the use of landscape classifications for the prediction of freshwater biota: synthesis and recommendations. Journal of the North American Benthological Society 19(3): 541–556.

Haybach, A., B. König & F. Schöll (2004): Langzeitveränderungen des Makrozoobenthos am nördlichen Oberrhein im Zeitraum 1986 bis 2000, dargestellt über biologische Artmerkmale (biological traits). Deutsche Gesellschaft für Limnologie (DGL) – Tagungsbericht 2003 (Köln): 473–478. Berlin.

Heino, J., T. Muotka, H. Mykrä, R. Paavola, H. Hämmäläinen & E. Koskenniemi (2003): Defining macroinvertebrate assemblages types of headwater streams: Implications for bioassessment and conservation. Ecological Applications 13(3): 842–852.

Hellawell, J. M. (1986): Biological indicators of freshwater pollution and environmental management. Elsevier Applied Sciences Publications, London/New York.

Hemphill, N. & S. D. Cooper (1983): The effect of physical disturbance of two filter-feeder insects in a small stream. Oecologia 58: 378–382.

Hemphill, N. (1988): Competition between two stream dwelling filter-feeders, Hydropsyche oslari and Simulium virgatum. Oecologia 77: 73–80.

Hering, D. & M. Reich (1997): Bedeutung von Totholz für Morphologie, Besiedlung und Renaturierung mitteleuropäischer Fließgewässer. Natur und Landschaft 72: 383–389.

Hering, D., A. Buffagni, O. Moog, L. Sandin, M. Sommerhäuser, I. Stubauer, C. K. Feld, R. K. Johnson, P. Pinto, N. Skoulikidis, P. F. M. Verdonschot & S. Zahrádková (2003): The development of a system to assess the ecological quality of streams based on macroinvertebrates – design of the sampling programme within the AQEM project. International Review of Hydrobiology 88(3–4): 345–361.

Hering, D., O. Moog, L. Sandin & P. F. M. Verdonschot (2004a): Overview and application of the AQEM assessment system. Hydrobiologia 516: 1–20.

Hering, D., C. Meier, C. Rawer-Jost, C. K. Feld, R. Bis, A. Zenker, A. Sundermann, S. Lohse & J. Böhmer (2004b): Assessing streams in Germany with benthic invertebrates: selection of candidate metrics. Limnologica 34(4): 398–415.

Hildrew, A. G. (1996): Whole river ecology: spatial scale heterogeneity in the ecology of running waters. Archiv für Hydrobiologie, Supplement 113 (Large Rivers 10): 25–43.

Hilsenhoff, W. L. (1988): Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society 7: 65–68.

HMULF (Hessisches Ministerium für Umwelt, Landwirtschaft und Forsten) (1999): Gewässerstrukturgüte in Hessen 1999. Wiesbaden. 52 pp.

Hoffmann, A. & D. Hering (2000): Wood-associated macroinvertebrate fauna in Central European streams. International Review of Hydrobiology 85: 25–48.

Hughes, R. M. (1995): Defining acceptable biological status by comparing with reference conditions. In: W. S. Davis & T. P. Simon (eds.): Biological assessment and criteria. Tools for water resource planning and decision making. Lewis Publishers, Boca Raton. 415 pp.

Illies, J. (1978): Limnofauna Europaea. Gustav Fischer, Stuttgart. 532 pp.

Johnson, R. K. & W. Goedkoop (2002): Littoral macroinvertebrate communities: spatial scale and ecological relationships. Freshwater Biology 47: 1840–1854.

Johnson, R. K., W. Goedkoop & L. Sandin (2004): Spatial scale and ecological relationships between the macroinvertebrate communities of stony habitats of streams and lakes. Freshwater Biology 49: 1179–1194.

Jones, R. C. & C. C. Clark (1987) : Impact of watershed urbanization on stream insect communities. Water Resources Bulletin 23: 1047–1055.

Jongman, R. H. G., C. J. F. ter Braak & O. F. R. van Tongeren (1995): Data analysis in community and landscape ecology. Cambridge University Press, Cambridge.

Kail, J. (2003): Influence of large woody debris on the morphology of six central European streams. Geomorphology 51: 207–223.

Karr, J. R. (1994): Biological monitoring: challenges for the future. In: S. L. Loeb & A. Spacie (eds.): Biological monitoring of aquatic systems. CRC Press LLC, Boca Raton, Florida.

Karr, J. R. & E. W. Chu (1999): Restoring life in running waters: better biological monitoring. Island Press, Washington DC. 206 pp.

Kiel, E. & A. Frutiger (1997): Behavioural responses of different blackfly species to oxygen depletion. Internationale Revue der gesamten Hydrobiologie 82(1): 107–120.

LAWA (Länderarbeitsgemeinschaft Wasser) (2000): Gewässerstrukturgütekartierung in der Bundesrepublik Deutschland – Verfahren für kleine und mittelgroße Fließgewässer. Kulturbuchverlag, Schwerin. 145 pp. + App.

Lorenz, A., C. K. Feld & D. Hering (2004a): Ein deutschlandweites Bewertungssystem mit dem Makrozoobenthos, Teil 2: Die biozönotische Sicht – zur Fließgewässertypologie Deutschlands. Deutsche Gesellschaft für Limnologie (DGL) – Tagungsbericht 2003 (Köln): 47–51. Werder.

Lorenz, A., D. Hering, C. K. Feld & P. Rolauffs (2004b): A new method for assessing the impact of hydromorphological degradation on the macroinvertebrate fauna of five German stream types. Hydrobiologia 516: 107–127.

Lorenz, A., C. K. Feld & D. Hering (2004c): Typology of German streams based on benthic macroinvertebrates: Ecoregions, zonation, geology and substrate. Limnologica 34(4): 379–389.

LUA NRW (Landesumweltamt Nordrhein-Westfalen, ed.) (1999): Leitbilder für kleine bis mittelgroße Fließgewässer in Nordrhein-Westfalen. Gewässerlandschaften und Fließgewässertypen. LUA-Merkblätter 17, Essen. 87 pp.

LUA NRW (Landesumweltamt Nordrhein-Westfalen, ed.) (2001): Leitbilder für die mittelgroßen bis großen Fließgewässer in Nordrhein-Westfalen – Flusstypen. LUA Merkblätter 34, Essen. 130 pp.

LUA NRW (Landesumweltamt Nordrhein-Westfalen, ed.) (2002): Fließgewässertypenatlas Nordrhein-Westfalens. LUA Merkblätter 36, Essen. 61 pp.

Malmqvist, B., Y. Zhang & P. Adler (1999): Diversity, distribution and larval habitats of North Swedish blackflies (Diptera: Simuliidae). Freshwater Biology 42: 301–314.

Margalef, R. (1984) Ecosystems: Diversity and connectivity as measurable components of their complication. In: S. Aida et al (eds.): The Science and Praxis of Complexity: 228–244. United Nations University, Tokyo.

McCune, B. & M. J. Mefford (1999): Multivariate analysis of ecological data. PC-Ord 4.x, MjM Software, Gleneden Beach, Oregon.

Merrit, R. W. & K. W. Cummins (1996): An introduction to the aquatic insects of North America. 3rd edition. Kendall/Hunt Publishing Company, Dubuque. 876 pp.

Minshall, G. W. (1984): Aquatic insect-substratum relationships. In: V. H. Resh & D. M. Rosenberg (eds.): The ecology of aquatic insects: 358–400. Praeger, New York.

Moog, O. (ed.) (1995): Fauna Aquatica Austriaca. Katalog zur autökologischen Einstufung aquatischer Organismen Österreichs. Wasserwirtschaftskataster, Bundesministerium für Land- und Forstwirtschaft, Wien.

Moog, O., A. Schmidt-Kloiber, T. Ofenböck & J. Gerritsen (2001): Aquatische Ökoregionen und Fließgewässer-Bioregionen Österreichs – eine Gliederung nach geoökologischen Milieufaktoren und Makrozoobenthos-Zönosen. Bundesministerium für Land- und Forstwirtschaft, Wien.

Moog, O., A. Schmidt-Kloiber, T. Ofenböck & J. Gerritsen (2004): Does the ecoregion approach support the typological demands of the EU 'Water Framework Directive'? Hydrobiologia 516: 21–33.

Mutz, M. (2000): Influence of woody debris on flow patterns and channel morphology in a low energy, sand-bed stream reach. International Review of Hydrobiology 85: 107–121.

Newbold, J. D., D. C. Erman & K. B. Roby (1980): Effects of logging on macroinvertebrates in streams with and without buffer strips. Canadian Journal of Fisheries and Aquatic Sciences 37: 1076–1085.

Nijboer, R. C., R. K. Johnson, P. F. M. Verdonschot, M. Sommerhäuser & A. Buffagni (2004): Establishing reference conditions for European streams. Hydrobiologia 516: 91–105.

Nixon, S. C., C. P. Mainstone, T. M. Iversen, P. Kristensen, E. Jeppensen, N. Friberg, E. Papathanassiou, A. Jensen & F. Pedersen (1996): The harmonized monitoring and classification of ecological quality of surface waters in the European Union. – WRc Report No. CO 4150, Medmenton, UK. 293 pp.

Ofenböck, T., O. Moog, J. Gerritsen & M. Barbour (2004): A stressor-specific multi-metric approach for monitoring running waters in Austria using benthic-macroinvertebrates. Hydrobiologia 516: 251–268.

Omernik, J. M. (1987): Ecoregions of the conterminous United States (with map). Annals of the Association of American Geographers 77: 118–125.

Omernik, J. M. (1995): Ecoregions: A spatial framework for environmental management. In: W. S. Davis & T. P. Simon (eds.): Biological assessment and criteria. Tools for water resource planning and decision making. Boca Raton, Florida.

Pauls, S., Feld C. K., Sommerhäuser M. & Hering, D. (2002). Neue Konzepte zur Bewertung von Tieflandbächen und -flüssen nach Vorgaben der EU Wasser-Rahmenrichtlinie. Wasser & Boden 54(7–8): 70–77.

Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross & R. M. Hughes (1989): Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. EPA/440/4-89-001. Assessment and Water Protection division, U.S. Environmental Protection Agency, Washington, DC.

Podani J. (2000): Introduction to the exploration of multivariate biological data Backhuys Publishers, Leiden. 407 pp.

Podraza, P., H. Schuhmacher & M. Sommerhäuser (2000): Composition of macroinvertebrate feeding groups as a bioindicator in running waters. Proceedings of the International Association of Theoretical and Applied Limnology 27: 3066–3069.

Poff, N. L. (1997): Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 16: 391–409.

Pottgiesser T. & M. Sommerhäuser (2004): VIII-2.1 Fließgewässertypologie Deutschlands: Die Gewässertypen und ihre Steckbriefe als Beitrag zur Umsetzung der EU-Wasserrahmenrichtlinie. In: C. Steinberg, W. Calmano, H. Klapper & W.-D. Wilken (eds.): Handbuch Angewandte Limnologie. 19. Ergänzungslieferung 07/04. ecomed, Landsberg. 16 pp. + Anhang.

Rabeni, C. F. (2000): Evaluating physical habitat integrity in relation to the biological potential of streams. Hydrobiologia 422/423: 245–256.

Rasper, M. (2001): Morphologische Fließgewässertypen in Niedersachsen – Leitbilder und Referenzgewässer. Hildesheim. 98 pp.

Raven, P. J., P. Fox, M. Everard, N. T. H. Holmes & F. H. Dawson (1997): River Habitat Survey: a new system for classifying rivers according to their habitat quality. In: P. J. Boon & D. L. Howell (eds): Freshwater Quality: Defining the Indefinable: 215–234. Scottish Natural Heritage, Edinburgh.

Raven, P. J., N. T. H. Holmes, F. H. Dawson & M. Everard (1998): Quality assessment using river habitat survey data. Aquatic Conservation: Marine and Freshwater Ecosystems 8: 477–499.

Raven, P. J., N. T. H. Holmes, P. Charrier, F. H. Dawson, M. Naura & P. J. Boon (2002): Towards a harmonized approach for hydromorphological assessment of rivers in Europe: a qualitative comparison of three survey methods. Aquatic Conservation: Marine and Freshwater Ecosystems 12: 405–424.

Reidelbach, J. & E. Kiel (1990): Observations on the behavioural sequences of looping and drifting by blackfly larvae (Diptera: Simuliidae). Aquatic Insects 12(1): 49–60.

Reidelbach, J. (1994): Untersuchungen zur Populationsdynamik der Kriebelmücken (Diptera: Simuliidae) des Breitenbachs. Dissertationsschrift. Philipps-Universität Marburg.

Resh, V. H. & D. M. Rosenberg (1993): Introduction to freshwater biomonitoring and benthic macroinvertebrates. In: D. M. Rosenberg & V H. Resh (eds.): Freshwater biomonitoring and benthic macroinvertebrates. Chapman & Hall, New York. 512 pp.

Resh, V. H., D. M. Rosenberg & T. B. Reynoldson (2000): Selection of benthic macroinvertebrate metrics for monitoring water quality of the Frazer River, British Columbia: implications for both multimetric approaches and multivariate models. In.: J. F. Wright, D. W. Sutcliffe & M. T. Furse (eds.): Assessing the Biological Quality of Fresh Waters: RIVPACS and other techniques: 195–206. Freshwater Biological Association, Ambleside, UK.

Richards, C., G. E. Host & J. W. Arthur (1993): Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. Freshwater Biology 29: 285–294.

Richards, C., L. B. Johnson & G. E. Host (1996): Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53, Supplement 1: 295–311.

Richards, C., R. J. Haro, L. B. Johnson & G. E. Host (1997): Catchment and reach-scale properties as indicators of macroinvertebrate species traits. Freshwater Biology 37: 219–230.

Rolauffs, P., I. Stubauer, S. Zahrádkova, K. Brabec &O. Moog (2004): Integration of the Saprobic System into the Water Framework Directive approach. Hydrobiologia 516: 285–298.

Rosenberg, D. M. & V. H. Resh (1993): Introduction to freshwater biomonitoring and benthic macroinvertebrates. In: D. M. Rosenberg & V. H. Resh (eds.): Freshwater biomonitoring and benthic macroinvertebrates. Chapman & Hall, New York. 512 pp.

Roy, A. H., A. D. Rosemond, M. J. Paul, D. S. Leigh & J. B Wallace (2003): Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). Freshwater Biology 48: 329–346.

Rundle, S. D., A. Jenkins & S. J. Ormerod (1993): Macroinvertebrate communities in streams in the Himalaya, Nepal. Freshwater Biology 30: 169–180.

Ruse, L. P. (1996): Multivariate techniques relating macroinvertebrate and environmental data from a river catchment. Water Research 30(12): 3017–3024.

Sandin, L. & R. K. Johnson (2000): Ecoregions and benthic macroinvertebrate assemblages of Swedish streams. Journal of the North American Benthological Society 19(3): 462–474.

Sandin, L., J. Dahl & R. K. Johnson (2004): Assessing acid stress in Swedish boreal and alpine streams using benthic macroinvertebrates. Hydrobiologia 516: 129–148.

Sandin L. & Hering D. (2004): Comparing macroinvertebrate indices to detect organic pollution across Europe: a contribution to the EC Water Framework Directive intercalibration. Hydrobiologia 516: 55–68.

Schmedtje, U., M. Sommerhäuser, U. Braukmann, E. Briem, P. Haase & D. Hering (2001): 'Top down-bottom up'-Konzept einer biozönotisch begründeten Fließgewässertypologie Deutschlands. Deutsche Gesellschaft für Limnologie (DGL) – Tagungsbericht 2000 (Magdeburg): 147–151. Tutzing.

Schmedtje, U. & M. Colling (1996): Ökologische Typisierung der aquatischen Makrofauna. Informationsberichte des Bayerischen Landesamtes für Wasserwirtschaft 4/96. München.

Schmidt-Kloiber, A. & R. C. Nijboer (2004): The effect of taxonomic resolution on the assessment of ecological water quality classes. Hydrobiologia 516: 269–283.

Schmutz, S. & G. Haidvogl (2002): FAME – Ein EU-Projekt zur Entwicklung einer fischbezogenen Bewertungsmethode für den ökologischen Zustand von europäischen Fließgewässern. Österreichs Fischerei 55(7): 173–176.

Schweder, H. (1992): Neue Indices für die Bewertung des ökologischen Zustandes von Fließgewässern, abgeleitet aus der Makroinvertebraten-Ernährungstypologie. In: G. Friedrich & J. Lacombe: Limnologie Aktuell Bd. 3: 353–377. Gustav Fischer, Stuttgart–Jena–New York.

Seitz, G. (1992): Verbreitung und Ökologie der Kriebelmücken (Diptera: Simuliidae) in Niederbayern. Lauterbornia 11: 1–230.

Shannon, C. E. & W. Weaver (1949): The mathematical theory of communication. The University of Illinois Press, Urbana.

Simpson, E. H. (1949): Measurement of diversity. Nature 163: 688.

Skriver, J., N. Friberg & J. Kirkegaard (2001): Biological assessment of running waters in Denmark: Introduction of the Danish Stream Fauna Index (DSFI). Proceedings of the International Association of Theoretical and Applied Limnology 27: 1822–1830.

Smith, M. J., W. R. Kay, D. H. D. Edward, P. J. Papas, K. S. J. Richardson, J. C. Simpson, A. M. Pinder, D. J. Cale, P. H. J. Horwitz, J. A. Davis, F. H. Yung, R. H. Norris & S. A. Halse (1999): AusRivAS: using macroinvertebrates to assess ecological condition of rivers in Western Australia. Freshwater Biology 41: 269–282.

Snyder, C. D., J. A. Young, R. Villella & D. P. Lemarié (2003): Influences of upland and riparian land use patterns on stream biotic integrity. Landscape Ecology 18: 647–664.

Sommerhäuser, M. (1998): Limnologisch-typologische Untersuchungen zu sommertrockenen und permanenten Tieflandbächen am Beispiel der Niederrheinischen Sandplatten. Dissertation Universität-Gesamthochschule Essen. 256 pp.

Sommerhäuser, M. & H. Schuhmacher (2003): Handbuch der Fließgewässer Norddeutschlands. ecomed, Landsberg. 278 pp.

Sommerhäuser, M. & T. Pottgiesser (2004): Biozönotisch bedeutsame Fließgewässertypen Deutschlands – Qualitätskomponente Makrozoobenthos – Stand Februar 2004. www.wasserblick.net.

Sponseller, R. A., E. F. Benfield & H. M. Valett (2001): Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology 46: 1409–1424.

Statistisches Bundesamt (ed.) (1997): Daten zur Bodenbedeckung. Wiesbaden.

StatSoft, Inc. (2003): STATISTICA (data analysis software system), version 6.1. Tulsa.

Statzner, B., B. Bis, S. Doledec & P. Usseglio-Polatera (2001): Perspectives for biomonitoring at large spatial scales: a unified measure for the functional composition of invertebrate communities in European running waters. Basic Applied Ecology 2: 73–85.

Tabacchi, E., D. L. Correll, R. Hauer, G. Pinay, A.-M. Planty-Tabacchi & R. C. Wissmar (1998): Development, maintenance and role of riparian vegetation in the river landscape. Freshwater Biology 40: 497–516.

ter Braak, C. J. F. & P. Smilauer (2002): CANOCO reference manual and CanoDraw for Windows user's guide version 4.5. Biometris – Plant Research International, Wageningen and České Budějovice.

ter Braak, C. J. F. & P. Smilauer (2003): CANOCO for Windows version 4.51. Biometris – Plant Research International, Wageningen.

ter Braak, C. J. F. & Verdonschot P. F. M (1995): Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57: 255–289.

Timm, T. & W. Juhl (1992): Verbreitungsmuster der Kriebelmücken (Diptera, Simuliidae) in Bächen entlang der unteren Ruhr, an der Nahtstelle zwischen Mittelgebirge und Flachland. Dortmunder Beiträge zur Landeskunde. Naturwissenschaftliche Mitteilungen 26: 13–28.

Timm, T. (1993): Unterschiede in Eibiologie und Habitatbindung zwischen Prosimulium tomosvaryi (Prosimuliini) und verschiedenen Simuliini (Diptera: Simuliidae). Internationale Revue der gesamten Hydrobiologie 78(1): 95–106.

Timm, T. & F. Klopp (1993): Die Ursachen unterschiedlicher Verteilungsmuster der Kriebelmücken in zwei Bächen des Niederbergischen Landes. In: T. Timm & W. Rühm (eds.): Beiträge zur Taxonomie, Faunistik und Ökologie der Kriebelmücken in Mitteleuropa (Diptera, Simuliidae). Essener Ökologische Schriften 2: 121–145.

Timm, T. (1994): Reasons for the shift in dominance between Simulium (N.) vernum and Simulium (S.) ornatum (Diptera: Simuliidae) along the continuum of an unpolluted lowland stream. Archiv für Hydrobiologie 131(2): 199–210.

Timm, T. (1995): VI-3.3 Teil 2 Ufer- und Auestrukturen und Simuliiden-Plagen. In: C. Steinberg, W. Calmano, H. Klapper & R.-D. Wilken (eds.): Handbuch Angewandte Limnologie: 1–28. ecomed, Landsberg.

Townsend, C. R., S. Dolédec & M. R. Scarsbrook (1997): Species traits in relation to temporal and spatial heterogeneity in streams: a test of habitat template theory. Freshwater Biology 37: 367–387.

Townsend, C. R., S. Dolédec, R. Norris, K. Peacock & C. Arbuckle (2003): The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. Freshwater Biology 48: 768–785.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell & C. E. Cushing (1980): The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130–137.

Verdonschot, P. F. M. (1995): Typology of macrofaunal assemblages: a tool for the management of running waters in The Netherlands. Hydrobiologia 297: 99–122.

Verdonschot, P. F. M. & R. C Nijboer (2002): A decision support system for stream restoration in the Netherlands. An overview of restoration projects and future needs. Hydrobiologia 478; 131–148.

Verdonschot, P. F. M. & R. C. Nijboer (2004): Testing the European stream typology of the Water Framework Directive for macroinvertebrates. Hydrobiologia 516: 35–54.

Vlek, H., P. F. M. Verdonschot & R. C. Nijboer (2004): Towards a multimetric index for the assessment of Dutch streams using benthic macroinvertebrates. Hydrobiologia 516: 173–189.

Waite, I. R., A. T. Herlihy, D. P. Larsen & D. J. Klemm (2000): Comparing strengths of geographic and non-geographic classifications of stream benthic macroinvertebrates in the Mid-Atlantic Highlands, USA. Journal of the North American Benthological Society 19(3): 429–441.

Wang, L., J. Lyons, P. Kanehl & R. Gatti (1997): Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries 22(6): 6–12.

Ward, J. V. (1989): The four-dimensional nature of lotic ecosystems. Journal of the North American Benthological Society 8: 2–8.

Weigel, B. M., L. Wang, P. W. Rasmussen, J. T. Butcher, P. M. Stewart & M. J Wiley (2003): Relative influence of variables at multiple spatial scales on stream macroinvertebrates in the Northern Lakes and Forest ecoregion, U.S.A. Freshwater Biology 48: 1440–1461.

Whittier, T. R., R. M. Hughes & D. P. Larsen (1988): Correspondence between ecoregions and spatial patterns in stream ecosystems in Oregon. Canadian Journal of Fisheries and Aquatic Sciences 45: 1264–1278.

Wiederholm, T. & R. K. Johnson (1997). Monitoring and assessment of lakes and watercourses in Sweden. In: J. J. Ottens, F. A. M. Claessen, P. G. Stoks, J. G. Timmerman & R. C. Ward (eds.): Monitoring Tailor-made II, Information strategies in Water: 317–329. Nunspeet, The Netherlands.

Wimmer, R. & A. Chovanec (2000): Fließgewässertypen in Österreich als Grundlage eines Überwachungsnetzes im Sinne des Anhang II der EU Wasser-Rahmenrichtlinie. Bundesministerium für Land- und Forstwirtschaft, Wien.

Wimmer, R., A. Chovanec, O. Moog, M. H. Fink & D. Gruber (2000): Abiotic stream classification as a basis for a surveillance monitoring network in Austria in accordance with the EU water framework directive. Acta hydrochimica et hydrobiologica 28(4): 177–184.

Wirtz, H. P., W. Piper, M. Prügel, W. Rühm, K. Rupp & T. Timm (1990): Verbreitung und Ökologie der Kriebelmücken des Westharzes. Braunschweiger naturkundliche Schriften 3: 719–746.

Wright, J. F., P. D. Hiley, A. C. Cameron, M. E. Wigham & A. D. Berrie (1983): A quantitative study of the macroinvertebrate fauna of five biotopes in the river Lamburn, Bershire, England. Archiv für Hydrobiologie 96: 271–292.

Wright, J. F., M. T. Furse & P. D. Armitage (1993): RIVPACS – a technique for evaluating the biological quality of rivers in the U.K. European Water Pollution Control 3(4): 15–25.

Zelinka, M. & P. Marvan (1961): Zur Präzisierung der biologischen Klassifikation der Reinheit fließender Gewässer. Archiv für Hydrobiologie 57: 389–407.

Zwick, H. & P. Zwick (1990): Terrestrial mass-ovipositon of Prosimulium species (Diptera, Simuliidae). Aquatic Insects 112: 33–46.

Zwick, H. (1974): Faunistisch-ökologische und taxonomische Untersuchungen an Simuliidae (Diptera), unter besonderer Berücksichtigung der Arten des Fulda-Gebietes. Abhandlungen der senckenbergischen naturforschenden Gesellschaft 533. Frankfurt am Main.

Appendixes

Appendix 1. List of site protocol variables with notes on numerical and spatial scale. Variable usage for different multivariate analysis is indicated by a '+', exclusion from analysis by a '-'. Numerical scale assigned according to Podani (2000). Areal and longitudinal extent of spatial scale is explained in Chapter 'Evaluation of hydromorphological degradation'.

					Degradation		
Variable code	Variable name	Numerical scale	Spatial scale	Typo-logy	All stream types	German stream types	Stream type D03
7	Stream order (Strahler system)	Ordinal	Catchment	+	+	+	+
8	Distance to source [km]	Interval	Catchment	+	+	+	+
11	Altitude [m a.s.l.]	Interval	Catchment	+	+	+	+
12	Ecoregion (according to Illies, 1978)	Nominal	Catchment	+	+	+	–
15	Catchment area [km²]	Interval	Catchment	+	+	+	+
16	Size typology according to the WFD (EU commission, 2000)	Ordinal	Catchment	+	+	+	–
17	Stream density [km km^{-2}]	Interval	Catchment	+	+	+	+
18-1	Geology: Acid silicate rocks [%]	Ratio	Catchment	+	+	+	–
18-3	Geology: Carbonate rocks [%]	Ratio	Catchment	+	+	+	+
18-4	Geology: Alluvial deposits [%]	Ratio	Catchment	+	+	+	–
18-7	Geology: Moraines [%]	Ratio	Catchment	+	+	+	+
18-8	Geology: Sander [%]	Ratio	Catchment	+	+	+	+
18-9	Geology: Marine deposits [%]	Ratio	Catchment	+	+	+	–
18-10	Geology: Organic formations [%]	Ratio	Catchment	+	+	+	+
18-11	Geology: Loess [%]	Ratio	Catchment	+	+	+	+
18a	Geological typology (silicate, carbonate, organic)	Ratio	Catchment	+	+	+	+
19-91	Land use: Native forest [%]	Ratio	Catchment	+	+	+	+
19-4	Land use: Wetland (mire) [%]	Ratio	Catchment	+	+	–	–
19-5	Land use: Grass-/bush land [%]	Ratio	Catchment	+	+	+	+
19-9	Land use: Artificial standing water bodies (ponds, etc.) [%]	Ratio	Catchment	+	+	+	+
19-10	Land use: Non-native forest [%]	Ratio	Catchment	+	+	+	+
19-12	Land use: Crop land [%]	Ratio	Catchment	+	+	+	+
19-13	Land use: Pasture [%]	Ratio	Catchment	+	+	+	+
19-92	Land use: Total agriculture [%]	Ratio	Catchment	+	+	+	–
19-15	Land use: Urban sites (residential) [%]	Ratio	Catchment	+	+	+	+
24	Hydrologic stream type (permanent, periodic/intermittent, episodic)	Nominal	Catchment	+	+	+	–
25	Presence of lakes in the whole up-stream continuum	Binary	Catchment	+	+	+	+
26	Width of the floodplain [m]	Interval	Site	–	–	–	+
29	Valley shape (V-shaped, U-shaped, trough, meander valley, etc.)	Nominal	Site	+	+	+	–
30-91	Land use: Native forest [%]	Ratio	Site	–	–	–	+
30-92	Land use: Grass-/bush land, reeds [%]	Ratio	Site	–	–	–	+
30-10	Land use: Non-native forest [%]	Ratio	Site	–	–	–	+
30-12	Land use: Crop land [%]	Ratio	Site	–	–	–	+
30-13	Land use: Pasture [%]	Ratio	Site	–	–	–	+
30-93	Land use: Total agriculture [%]	Ratio	Site	–	–	–	+

Appendix 1, continued.

Variable code	Variable name	Numerical scale	Spatial scale	Typo-logy	Degradation		
					All stream types	German stream types	Stream type D03
30-15	Land use: Urban settlement/industry [%]	Ratio	Site	–	–	–	+
31	No. of other transverse structures	Interval	Upstream	+	+	+	+
34	Straightening	Binary	Upstream	+	+	+	+
35	Removal of large wood	Binary	Upstream	+	+	+	+
36	Cut-off meanders	Binary	Upstream	+	+	+	+
37	Scouring below bank top [m]	Interval	Upstream	+	+	+	+
38	Culverting	Binary	Upstream	+	+	+	+
39	No. of other transverse structures	Interval	Down-stream	+	+	+	+
42	Straightening	Binary	Down-stream	+	+	+	+
43	Removal of large wood	Binary	Down-stream	+	+	+	+
44	Cut-off meanders	Binary	Down-stream	+	+	+	+
45	Scouring below bank top [m]	Interval	Down-stream	+	+	+	+
46	Culverting	Binary	Down-stream	+	+	+	+
47	No. of dams retaining sediment	Interval	Upstream	+	+	+	+
49	No. of dams obstructing migration	Interval	Down-stream	+	+	+	+
56	Impoundments or dams (% of length)	Ratio	Upstream	+	+	+	+
56a	Lack of natural wooded vegetation	Binary	Upstream	+	+	+	+
56b	Non-native wooded vegetation	Binary	Upstream	–	–	–	+
57	Lack of natural wooded vegetation	Binary	Down-stream	+	+	+	+
58	Non-native wooded vegetation	Binary	Down-stream	–	–	–	+
59	Impoundments or dams (% of length)	Ratio	Down-stream	+	+	+	+
61	Non-source pollution	Binary	Upstream	+	+	+	–
63	Eutrophication	Binary	Upstream	+	+	+	–
68	Mean depth at bankfull discharge [m]	Interval	Site	+	+	+	+
69	Shading at zenith (foliage cover) [%]	Ratio	Site	+	+	+	+
70-91	Average width of wooded riparian vegetation right + left [m]	Interval	Site	+	+	+	+
71	Channel form (braided, meandering, sinuate, etc.)	Nominal	Site	+	+	+	+
73	Presence of natural standing water bodies in the floodplain (e. g. backwa-ters)	Binary	Site	+	+	+	+
74	No. of debris dams > 0.3 m³	Interval	Site	+	+	+	+
75	No. of logs > 10 cm diameter	Interval	Site	+	+	+	+
76-91	Shoreline covered with wooded ripar-ian vegetation right + left [%]	Ratio	Site	+	+	+	+
77	No. of dams	Interval	Site	+	+	+	+
78	No. of other transverse structures	Interval	Site	+	+	+	+

Appendix 1, continued.

Variable code	Variable name	Numerical scale	Spatial scale	Typo-logy	Degradation		
					All stream types	German stream types	Stream type D03
79-91	Bank fixation stones (rip-rap) [%]	Ratio	Site	+	+	+	+
79-92	Bank fixation wood/trees [%]	Ratio	Site	+	+	+	+
79-93	No bank fixation [%]	Ratio	Site	+	+	+	+
80-3	Bed fixation stones [%]	Ratio	Site	+	+	+	+
80-9	No bed fixation [%]	Ratio	Site	+	+	+	+
81	Stagnation	Binary	Site	+	+	+	+
84	Straightening	Binary	Site	+	+	+	+
85	Removal of large wood	Binary	Site	+	+	+	+
86	Cut-off meanders	Binary	Site	+	+	+	+
87	Scouring below bank top [m]	Interval	Site	+	+	+	+
88	Culverting	Binary	Site	+	+	+	+
92	Impoundments at sampling site	Binary	Site	+	+	+	+
93	Removal/lack of natural floodplain vegetation	Binary	Site	+	+	+	+
94	Non-native wooded riparian vegetation	Binary	Site	–	–	–	+
95	Source pollution	Binary	Site	+	+	+	–
96	Non-source pollution	Binary	Site	+	+	+	–
97	Sewage overflows	Binary	Site	+	+	+	–
98	Eutrophication	Binary	Site	+	+	+	–
103_2	Megalithal (> 40 cm) [%]	Ratio	Site	+	+	–	–
103_3	Macrolithal (> 20–40 cm) [%]	Ratio	Site	+	+	+	+
103-4	Mesolithal (> 6–20 cm) [%]	Ratio	Site	+	+	+	+
103-5	Microlithal (> 2–6 cm) [%]	Ratio	Site	+	+	+	+
103-6	Akal (> 0.2–2 cm) [%]	Ratio	Site	+	+	+	+
103-7	Psammal/psammopelal [%]	Ratio	Site	+	+	+	+
103-8	Argyllal (< 6 µm) [%]	Ratio	Site	+	+	+	–
104-2	Algae [%]	Ratio	Site	+	+	+	+
104-3	Submerged macrophytes [%]	Ratio	Site	+	+	+	+
104-4	Emergent macrophytes [%]	Ratio	Site	+	+	+	+
104-5	Living parts of terrestrial plants [%]	Ratio	Site	+	+	+	+
104-6	Xylal (wood) [%]	Ratio	Site	+	+	+	+
104-7	CPOM [%]	Ratio	Site	+	+	+	+
104-8	FPOM [%]	Ratio	Site	+	+	+	+
104-10	Organic mud, sludge [%]	Ratio	Site	+	+	+	+
104-11	Debris (e. g., empty mollusc shells at the shore zone) [%]	Ratio	Site	–	–	–	+
104-91	No. of organic substrates	Interval	Site	+	+	+	+
105	Average stream width [m]	Interval	Site	+	+	+	+
110	pH	Interval	Site	+	+	+	–
111	Conductivity [µS cm^{-1}]	Interval	Site	+	+	+	–
112	Reduction phenomena	Binary	Site	+	+	+	–
113	Waste	Binary	Site	+	+	+	–
114	Dissolved oxygen content [mg l^{-1}]	Interval	Site	+	+	+	–
118	Max. depth [cm]	Interval	Site	–	–	–	+
120	Max. current velocity [m s^{-1}]	Interval	Site	–	–	–	+
121	Mean depth [cm]	Interval	Site	+	+	+	+
122	CV depth	Ratio	Site	+	+	+	+

Appendix 1, continued.

Variable code	Variable name	Numerical scale	Spatial scale	Typo-logy	Degradation		
					All stream types	German stream types	Stream type D03
123	Mean current velocity [m s^{-1}]	Interval	Site	+	+	+	+
124	CV current velocity	Ratio	Site	+	+	+	+
125	Ammonium [mg l^{-1}]	Interval	Site	+	+	+	–
127	Nitrate [mg l^{-1}]	Interval	Site	+	+	+	–
128	Ortho-phosphate [µg l^{-1}]	Interval	Site	+	+	+	–
129	Total phosphate [µg l^{-1}]	Interval	Site	+	+	+	–

Appendix 2: Hydromorphological variables with spatial scales used for canonical ordination. 'Mega'-scaled environmental variables were used as covariables in canonical ordination.

Variable	Variable short	Transformation
'Mega'-scale		
Sampling season spring (yes/no)	spring	none
Sampling season summer (yes/no)	summer	none
Sampling season autumn (yes/no)	autumn	none
Latitude (decimal degree)	lat	log (x+1)
Altitude [m]	alt	log (x+1)
Ecoregion 13 = Western Lowlands (yes/no)	ecoreg13	none
Ecoregion 14 = Central Lowlands (yes/no)	ecoreg14	none
Longitude (decimal degree)	long	log (x+1)
Catchment [km²]	catchm	log (x+1)
Macro-scale		
Land use catchment: % Forest	c_forest	arc sin $(x/100)^{0.5}$
Land use catchment: % Crop land	c_crop	arc sin $(x/100)^{0.5}$
Land use catchment: % Pasture	c_pastur	arc sin $(x/100)^{0.5}$
Land use catchment: % Urban settlement/industry	c_urban	arc sin $(x/100)^{0.5}$
Land use catchment: % Wetland (mire)	c_wet	arc sin $(x/100)^{0.5}$
Land use catchment: % Grass-/bushland	c_grabu	arc sin $(x/100)^{0.5}$
Land use catchment: % Artificial standing water bodies	c_aswb	arc sin $(x/100)^{0.5}$
Lakes in the stream continuum upstream of the sampling site (yes/no)	c_lake	none
Meso-scale		
Average stream width [m]	wid_str	log (x+1)
pH	pH	none
Electric conductivity [µS/cm]	conduct	log (x+1)
Dissolved oxygen [mg/l]	dissoxy	log (x+1)
Land use floodplain: % Crop land	f_crop	arc sin $(x/100)^{0.5}$
Land use floodplain: % Pasture	f_past	arc sin $(x/100)^{0.5}$
Land use floodplain: % Urban settlement/industry	f_urban	arc sin $(x/100)^{0.5}$
Land use floodplain: % Forest	f_forest	arc sin $(x/100)^{0.5}$
Land use floodplain: % Grass-/bushland, reeds	f_grabu	arc sin $(x/100)^{0.5}$
Shading at zenith [%]	f_shade	arc sin $(x/100)^{0.5}$
Average width of wooded riparian vegetation [m]	wid_rip	log (x+1)
Channel form meandering (yes/no)	meander	none
Channel form sinuate (yes/no)	situate	none
Presence of standing water bodies in the floodplain (yes/no)	swb_flpl	none
No. of debris dams (1,000 m reach)	debdams	log (x+1)
No. of logs (1,000 m reach)	logs	log (x+1)
Proportion of shoreline covered with wooded riparian vegetation	dens_rip	arc sin $(x/100)^{0.5}$
No. of dams within 500 m up- and downstream of sampling site	dams	log (x+1)
Proportion of bank fixation stones (1,000 m reach)	bafi_sto	arc sin $(x/100)^{0.5}$
Stagnation (yes/no)	stagnat	none
Straightening (yes/no)	straight	none
Micro-scale		
Microhabitat: % Macrolithal (> 20 –40 cm)	macrolit	arc sin $(x/100)^{0.5}$
Microhabitat: % Mesolithal (> 6–20 cm)	mesolit	arc sin $(x/100)^{0.5}$
Microhabitat: % Microlithal (> 2–6 cm)	microlit	arc sin $(x/100)^{0.5}$
Microhabitat: % Akal (> 0.2–2 cm)	akal	arc sin $(x/100)^{0.5}$

Appendix 2, continued.

Variable	Variable short	Transformation
Micro-scale		
Microhabitat: % Argyllal (< 6 µm)	argyll	arc sin $(x/100)^{0.5}$
Microhabitat: % Psammal/psammopelal (sand and/or mineral mud)	psam_pel	arc sin $(x/100)^{0.5}$
Microhabitat: % Organic mud	org_mud	arc sin $(x/100)^{0.5}$
Microhabitat: % Submerged macrophytes	sub_macr	arc sin $(x/100)^{0.5}$
Microhabitat: % Emergent macrophytes	em_macr	arc sin $(x/100)^{0.5}$
Microhabitat: % Living parts of terrestrial plants	ter_mac	arc sin $(x/100)^{0.5}$
Microhabitat: % Xylal (wood)	xylal	arc sin $(x/100)^{0.5}$
Microhabitat: % CPOM (coarse particulate organic matter)	CPOM	arc sin $(x/100)^{0.5}$
Microhabitat: % FPOM (fine particulate organic matter)	FPOM	arc sin $(x/100)^{0.5}$
No. of organic substrates	org_sub	log (x+1)

Appendix 3: 109 metrics calculated from 244 taxa of the current study. 84 metrics indicated by '+' were uncorrelated at the level of r < 0.800 and used for multivariate analyses (RDA). Short codes refer to those used for ordination plots. Metric groups: C/A = composition/abundance, F = functional, R/D = richness/diversity and S/T = sensitive/tolerant.

Metric name	Metric short	Metric group	Used for RDA
Proportion of specimens in a sample			
Abundance [Ind. m^{-2}]	abunda	C/A	+
Turbellaria [%]	p_Turb	C/A	+
Gastropoda [%]	p_Gast	C/A	+
Bivalvia [%]	p_Biva	C/A	+
Oligochaeta [%]	p_Olig	C/A	+
Hirudinea [%]	p_Hiru	C/A	+
Crustacea [%]	p_Crus	C/A	
Ephemeroptera [%]	p_Ephe	C/A	+
Odonata [%]	p_Odon	C/A	+
Plecoptera [%]	p_Plec	C/A	+
Heteroptera [%]	p_Hete	C/A	+
Planipennia [%]	p_Plan	C/A	+
Megaloptera [%]	p_Mega	C/A	+
Trichoptera [%]	p_Trich	C/A	+
Lepidoptera [%]	p_Lepi	C/A	+
Coleoptera [%]	p_Cole	C/A	+
Diptera [%]	p_Dipt	C/A	+
EPT (Ephemeroptera, Plecoptera, Trichoptera) [%]	p_EPT	C/A	+
Chironomidae [%]	p_Chir	C/A	+
Longitudinal zonation in the stream continuum (Moog, 1995; Schmedtje & Colling, 1996; Hering et al., 2004a)			
Crenal (spring) [%]	crenal	F	+
Hypocrenal (spring-brook) [%]	hycrenal	F	+
Epirhithral (upper-trout region) [%]	eprhit	F	+
Metarhithral (lower-trout region) [%]	metrhit	F	+
Hyporhithral (greyling region) [%]	hyprhit	F	
Epipotamal (barbel region) [%]	eppot	F	+
Metapotamal (brass region) [%]	metpot	F	+
Hypopotamal (brackish water) [%]	hyppot	F	+
Littoral [%]	litoral	F	+
Profundal [%]	profund	F	+
Current preferences (Schmedtje & Colling, 1996; Hering et al., 2004a)			
Limnobiont [%]	LB	F	+
Limnophil [%]	LP	F	+
Limno- to rheophil [%]	LR	F	+
Rheo- to limnophil [%]	RL	F	+
Rheophil [%]	RP	F	+
Rheobiont [%]	RB	F	+
Indifferent [%]	IN	F	+
Microhabitat preferences (Schmedtje & Colling, 1996; Hering et al., 2004a)			
Pelal (mud; grain size < 0.063 mm) [%]	Pel	F	+
Argyllal (silt, loam, clay; grain size < 0.063 mm) [%]	Arg	F	+
Psammal (sand; grain size 0.063–2 mm) [%]	Psa	F	+

Appendix 3, continued.

Metric name	Metric short	Metric group	Used for RDA
Microhabitat preferences (continued)			
Akal (fine to medium gravel; grain size 2 mm–2 cm) [%]	Aka	F	+
Lithal (coarse gravel, stones, boulders; grain size > 2 cm) [%]	Lit	F	+
Phytal (mosses, macrophytes, parts of terrestrial plants) [%]	Phy	F	+
Particulate Organic Matter (CPOM, FPOM) [%]	POM	F	+
Feeding types (Moog, 1995; Schmedtje & Colling, 1996; Hering et al., 2004a)			
Grazers/scrapers [%]	grazscra	F	+
Miners [%]	miner	F	+
Xylophagous taxa [%]	xyloph	F	+
Shredders [%]	shred	F	+
Gatherers/collectors [%]	gathcoll	F	+
Active filterers [%]	actfilt	F	+
Passive filterers [%]	pasfilt	F	+
Predators [%]	predat	F	+
Parasites [%]	parasit	F	+
RETI (Rhithron Feeding Type Index) (Schweder, 1992; Podraza et al., 2000)	RETI	F	+
Locomotion types (Schmedtje & Colling, 1996; Hering et al., 2004a)			
Swiming/skating [%]	swimskat	F	+
Swimming/diving [%]	swimdive	F	+
Burrowing/boring [%]	burobor	F	+
Sprawling/walking [%]	sprawalk	F	+
(Semi-)sessil [%]	sessil	F	+
Number of taxa and diversity indices			
Number of taxa	no_taxa	R/D	+
Danish Stream Fauna Index diversity groups (Skriver et al., 2001)	DSFI_dg	R/D	
Margalef diversity (Margalef, 1984)	Mag_div	R/D	+
Simpson diversity (Simpson, 1949)	Sim_div	R/D	
Shannon-Weaver diversity (Shannon & Weaver, 1949)	SWi_div	R/D	+
Evenness	even	R/D	
GFI type 14: No. of indicator taxa (Lorenz et al., 2004b)	notaFI4	R/D	+
GFI type 11: No. of indicator taxa (Lorenz et al., 2004b)	notaFI11	R/D	+
GFI type 15: No. of indicator taxa (Lorenz et al., 2004b)	notaFI15	R/D	+
GFI type 5: No. of indicator taxa (Lorenz et al., 2004b)	notaFI5	R/D	+
GFI type 9: No. of indicator taxa (Lorenz et al., 2004b)	notaFI9	R/D	+
No. taxa Turbellaria	n_Turb	R/D	
No. taxa Gastropoda	n_Gast	R/D	
No. taxa Bivalvia	n_Biva	R/D	
No. taxa Oligochaeta	n_Olig	R/D	+
No. taxa Hirudinea	n_Hiru	R/D	
No. taxa Crustacea	n_Crus	R/D	+
No. taxa Ephemeroptera	n_Ephe	R/D	+
No. taxa Odonata	n_Odon	R/D	
No. taxa Plecoptera	n_Plec	R/D	
No. taxa Heteroptera	n_Hete	R/D	
No. taxa Planipennia	n_Plan	R/D	
No. taxa Megaloptera	n_Mega	R/D	

Appendix 3, continued.

Metric name	Metric short	Metric group	Used for RDA
Number of taxa and diversity indices (continued)			
No. taxa Trichoptera	n_Trich	R/D	+
No. taxa Lepidoptera	n_Lepi	R/D	
No. taxa Coleoptera	n_Cole	R/D	+
No. taxa Diptera	n_Dipt	R/D	+
No. taxa EPT (Ephemeroptera, Plecoptera, Trichoptera)	n_EPT	R/D	+
Saprobic indices			
Saprobic Index (Zelinka & Marvan, 1961)	SI_ZM	S/T	+
Xenosaprobic valences [%] (Moog, 1995)	SVZM_xe	S/T	
Oligosaprobic valences [%] (Moog, 1995)	SVZM_ol	S/T	
Beta-mesosaprobic valences [%] (Moog, 1995)	SVZM_bm	S/T	
Alpha-mesosaprobic valences [%] (Moog, 1995)	SVZM_am	S/T	
Polysaprobic valences [%] (Moog, 1995)	SVZM_po	S/T	
German Saprobic Index old (DEV, 1992)	SI_D_old	S/T	+
German Saprobic Index stream type-specific (Rolauffs et al., 2004; Friedrich & Herbst, 2004)	SI_D_new	S/T	+
Dutch Saprobic Index (Lorenz et al., 2004b)	SI_NL	S/T	+
Other indices			
British Monitoring Working Party (Armitage et al., 1983)	BMWP	S/T	+
ASPT (Average Score per Taxon) (Armitage et al., 1983)	ASPT	S/T	+
BMWP (Spain) (Alba-Tercedor & Sanchez-Ortega, 1988)	BMWP_E	S/T	
Danish Stream Fauna Index (Skriver et al., 2001)	DSFI	S/T	+
Acid class (Braukmann, 2001)	Acid_D	S/T	+
Share acid class 1 (no acidification)	AC_D1	S/T	
Share acid class 2 (periodical slight acidification)	AC_D2	S/T	
Share acid class 3 (periodical serious acidification)	AC_D3	S/T	
Share acid class 4 (permanent acidification)	AC_D4	S/T	
German Fauna Index IVD01 (Lorenz et al., 2004b)	FI_t14	S/T	+
German Fauna Index IVD02 (Lorenz et al., 2004b)	FI_t11	S/T	+
German Fauna Index IVD03 (Lorenz et al., 2004b)	FI_t15	S/T	+
German Fauna Index IVD04 (Lorenz et al., 2004b)	FI_t5	S/T	+
German Fauna Index IVD05 (Lorenz et al., 2004b)	FI_t9	S/T	+

Appendix 4: Classes and statistics of 34 hydromorphological variables for ISA.

					Upper class boundary				
Variable short	Range	Mean	Median	SD	1	2	3	4	5
Macro									
c_past	0–70	20.3	20.0	17.2	0.0	20.0	40.0	60.0	70.0
c_aswb	0–10	1.0	0.0	3.0	0.0	20.0	40.0	60.0	80.0
c_wet	0–20	1.3	0.0	3.9	0.0	10.0	20.0	n. a.	n. a.
c_grabu	0–50	4.3	0.0	8.8	0.0	10.0	20.0	50.0	n. a.
c_lake	0–1	binary			0.0	1.0	n. a.	n. a.	n. a.
c_urban	0–20	5.6	0.0	6.4	0.0	10.0	20.0	n. a.	n. a.
Meso									
pH	5.4–8.6	7.7	7.8	0.6	6.0	6.7	7.4	8.1	8.6
bafi_sto	0–100	17.6	0.0	33.9	0.0	20.0	40.0	100.0	n. a.
logs	0–350	16.2	1.0	38.1	10.0	25.0	50.0	100.0	350.0
f_past	0–100	22.6	10.0	32.4	0.0	20.0	40.0	70.0	100.0
f_grabu	0–100	11.1	0.0	23.0	0.0	20.0	40.0	70.0	100.0
wid_str	0.3–42	7.8	7.5	5.9	3.0	6.0	10.0	20.0	42.0
f_urban	0–60	2.4	0.0	10.8	0.0	60.0	n. a.	n. a.	n. a.
straight	0–1	binary			0.0	1.0	n. a.	n. a.	n. a.
dens_rip	0–100	58.4	75.0	41.4	0.0	20.0	40.0	70.0	100.0
conduct	62.6–1741	482.3	520.0	256.8	200.0	400.0	800.0	1741.0	n. a.
meander	0–1	binary			0.0	1.0	n. a.	n. a.	n. a.
sinuate	0–1	binary			0.0	1.0	n. a.	n. a.	n. a.
stagnat	0–1	binary			0.0	1.0	n. a.	n. a.	n. a.
swb_flpl	0–1	binary			0.0	1.0	n. a.	n. a.	n. a.
f_shade	0–100	48.6	60.0	34.2	0.0	20.0	40.0	70.0	100.0
f_crop	0–100	15.1	0.0	25.6	0.0	20.0	40.0	70.0	100.0
Micro									
CPOM	0–60	7.5	5.0	9.2	0.0	5.0	10.0	60.0	n. a.
sub-mac	0–90	6.9	0.0	14.3	0.0	5.0	10.0	30.0	90.0
FPOM	0–50	3.5	0.0	5.8	0.0	5.0	50.0	n. a.	n. a.
org_sub	0–8	3.4	4.0	2.5	0.0	2.0	4.0	6.0	8.0
akal	0–70	5.8	0.0	12.6	0.0	5.0	10.0	70.0	n. a.
xylal	0–30	2.9	0.0	4.6	0.0	5.0	30.0	n. a.	n. a.
mesolit	0–85	9.5	0.0	17.3	0.0	10.0	30.0	85.0	n. a.
macrolit	0–75	5.1	0.0	11.9	0.0	10.0	20.0	75.0	n. a.
em_mac	0–40	2.6	0.0	6.4	0.0	5.0	10.0	40.0	n. a.
psa_pel	0–100	65.6	80.0	34.8	20.0	40.0	60.0	80.0	100.0
ter_mac	0–20	1.6	0.0	3.6	0.0	5.0	20.0	n. a.	n. a.
microlit	0–55	3.2	0.0	8.8	0.0	10.0	55.0	n. a.	n. a.

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Erklärung

Hiermit erkläre ich, gem. § 6 Abs. 2, Nr. 7 der Promotionsordnung der Fachbereiche 6 bis 9 zur Erlangung des Dr. rer. nat., dass ich das Arbeitsgebiet, dem das Thema „Assessing the hydromorphological status of sand-bottom lowland rivers in Central Europe using benthic macroinvertebrates“ zuzuordnen ist, in Forschung und Lehre vertrete und den Antrag von Herrn Christian Karl Feld befürworte.

Ort, Datum (PD Dr. Daniel Hering)

Erklärung

Hiermit erkläre ich, gem. § 6 Abs. 2, Nr. 6 der Promotionsordnung der Fachbereiche 6 bis 9 zur Erlangung des Dr. rer. nat., dass ich die vorliegende Dissertation selbständig verfasst und mich keiner anderen als der angegebenen Hilfsmittel bedient habe.

Ort, Datum (Christian K. Feld)

Erklärung

Hiermit erkläre ich, gem. § 6 Abs. 2, Nr. 8 der Promotionsordnung der Fachbereiche 6 bis 9 zur Erlangung des Dr. rer. nat., dass ich keine anderen Promotionen bzw. Promotionsversuche in der Vergangenheit durchgeführt habe und dass diese Arbeit von keiner anderen Fakultät abgelehnt worden ist.

Ort, Datum (Christian K. Feld)