Project no. GOCE-CT-2003-505540

Project acronym: Euro-limpacs

Project full name: Integrated Project to evaluate the Impacts of Global Change on European Freshwater Ecosystems

Instrument type: Integrated Project

Priority name: Sustainable Development

Report No. 325
Manuscript on Macroinvertebrate indicators of flow and in-stream habitat conditions in Swedish stream a comparison of large-scale survey data with local-scale information

Due date of deliverable: [2009.02.28]
Actual submission date: [2009.03.17]

Start date of project: 1 February 2002 Duration: 5 Years

Organisation name of lead contractor for this deliverable: Swedish University of Agricultural Sciences (SLU)

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Assessing the Effects of Hydromorphological Degradation on Macroinvertebrate Indicators in Rivers: Examples, Constraints, and Outlook

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(Received 30 May 2008; Accepted 2 October 2008)

Abstract

A review of the extensive literature on linkages between the in-stream physical environment and river benthic macroinvertebrates revealed a number of reported relationships across multiple spatial scales. We analyzed data on different spatial scales to elucidate the linkages between different measurements of hydromorphological degradation and commonly used macroinvertebrate indices. A regression analysis of 1049 sites from 3 countries revealed that the strongest relationship between a biotic metric—average score per taxon—and physiochemical variables ($R^2 = 0.61$) was obtained with a multiple regression model that included concentration of total phosphorus and percent arable land in the catchment, as well as hydromorphological quality variables. Analyses of 3 data sets from streams primarily affected by hydromorphological degradation showed an overall weak relationship (max $R^2 = 0.25$) with the River Habitat Survey data of 28 Swedish streams, whereas moderate ($R^2 \approx 0.43$) relationships with more detailed measurements of morphology were found in 2 Danish studies (39 and 6 streams, respectively). Although evidence exists in the literature on the importance of physical features for in-stream biota in general and macroinvertebrates specifically, we found only relatively weak relationships between various measures of hydromorphological stress and commonly used macroinvertebrate assessment tools. We attribute this to a combination of factors, including 1) the mixed nature of pressures acting on the majority of river reaches, 2) scaling issues (spatial and temporal) when relating habitat surveys to macroinvertebrate assessments, and 3) the scope of commonly used macroinvertebrate assessment systems (mainly focusing on water chemistry perturbation, such as eutrophication and acidification). The need is urgent to develop refined and updated biological assessment systems targeting hydromorphological stress for the use of the European Water Framework Directive (WFD) and national water-related policies.

Keywords: River Water Framework Directive Macroinvertebrate Assessment Hydromorphology

Introduction

The combined effects of stream regulation, weed cutting, and eutrophication have affected physical and biological ecosystem processes as well as stream morphology and have caused significant habitat degradation resulting in decreased biological stream quality in many rivers and streams of the world (Ward and Stanford 1979; Hansen 1996). Man-made modifications to in-stream channel morphology alone are likely to be one of the main reasons that many rivers do not achieve good ecological quality. Channelization, dredging, and weed cutting have seriously degraded habitat diversity, thereby excluding some species and reducing abundance of others (Moyle 1976; Quinn et al. 1992; Bis et al. 2000). The current physical structure of rivers and diversity of biological communities are closely linked to past and present human activities, which influence stream ecosystems on multiple scales, ranging from direct manipulation of the in-stream environment (channelization, removal of large woody debris, etc.) to alteration of landscape features and land use in the catchment, thereby influencing the hydrological pathways and water chemistry (Vannote et al. 1980; Frissell et al. 1986; Fitzpatrick et al. 2001; Allan 2004). Past and present disturbances act simultaneously with different intensities on stream ecosystem elements, and quantifying the disturbance from individual stressors on biotic communities can thus be difficult (Lane and Richards 1997; Harding et al. 1998; Allan 2004). Physical disturbance of the in-stream environment is only slowly reversed and can potentially have a lasting effect on both fish and macroinvertebrate communities (e.g., Brookes 1988).

The European Union Water Framework Directive (WFD) adopted by The European Parliament and the Council of the European Union (2000) aims to assess, monitor, and, where necessary, improve the ecological quality of surface waters, including rivers. The WFD sets out normative definitions describing biological and physicochemical standards expected of “high,” “good,” and “moderate” quality of streams and rivers for 4 biological quality elements (phytoplankton, macrophytes and benthic algae, macroinvertebrates, and fish) and a variety of hydromorphological and physicochemical quality elements. In fact, the term “hydromorphology” is introduced by the WFD and signifies how important physical features are considered in determining ecological status of freshwaters. The hydromorphological quality of a river in the high status class is defined as follows: “Channel patterns, width and depth variations, flow velocities, substrate conditions and both structure and condition of the riparian zones correspond totally or nearly totally to undisturbed condi-

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Published on the Web 10/13/2008.
tions.” With regard to the lower status classes, hydromorphology is considered a supporting element with no normative definitions.

Hydromorphological degradation is a key pressure on river ecosystems, not only in Europe but globally. However, assessing the direct effects of hydromorphological degradation on the biota is challenging because of the presence of other stressors acting simultaneously. Water quality monitoring has until recently focused almost entirely on organic pollution, nutrients and xenochemicals, with limited and inconsistent assessment of flow, habitats, and hydromorphology (e.g., Friberg et al. 2005). In addition, the primary indicator organism group used in rivers has been macroinvertebrates, and the majority of assessment systems in use target organic pollution (e.g., Metcalfe-Smith 1996; Friberg et al. 2006). Few very systems target the effects of low flow (Extence et al. 1999) and degraded hydromorphology (Barbour et al. 1996; Lorenz et al. 2004), and their general applicability is limited. Moreover, most river systems are influenced by multiple stressors acting simultaneously. Recent studies have addressed the problem of separating the effects of individual stressors in a multistressor environment, primarily through the use of a multimetric approach (Barbour et al. 1999; Karr and Chu 1999; Hering et al. 2006), but more development is needed before the contribution of individual stressors can be quantified with sufficient certainty. Consequently, despite recognizing the importance of hydromorphological degradation in river ecosystems, our ability to assess the risk of this pressure is at present very limited. However, despite these complications in identifying and quantifying stress caused by changes to habitats and hydromorphology, a very large body of published studies has signified the strong linkages between the physical environment and biota.

The physical environment plays an extremely important role in the functioning of the river and stream ecosystems by determining the environment and the habitat characteristics used by stream organisms (Southwood 1977). In contrast to other aquatic environments, such as lakes and the oceans, the river ecosystem structure and functioning are heavily influenced by the physical environment at all scales (Hildrew and Giller 1994; Townsend and Hildrew 1994). Macroinvertebrate communities have been studied in relation to physical habitat structure on a number of spatial scales, ranging from the reach/catchment scale to individual particles (Statzner and Holm 1982; Scarsbrook and Townsend 1993; Downes 2000). The microhabitat scale, defined as the immediate environment surrounding the organism, has been studied in relation to the significance of the substratum (Minshall 1984) and the flow conditions (Statzner and Holm 1982). The mesohabitat scale (defined here as a patch of uniform substratum) also has received attention (Armitage et al. 1995; Downes 2000; Kemp et al. 2000). Reach-scale studies have been performed, and the influence of the surrounding landscape also has been analyzed in relation to physical habitats and biological communities (e.g., Allan and Johnson 1997; Bojesen and Barriga 2002; Pedersen 2008).

Macroinvertebrate distribution and habitat utilization is influenced by flow variables such as velocity and shear stress (Statzner et al. 1988; Barmuta 1990). Substratum characteristics, such as particle size and content of organic matter (e.g., Pennak and Van Gerpen 1947; Wood and Armitage 1997; Miyake and Nakano 2002), stability (Stanford and Ward 1983), texture (Harman 1972; Lamberti and Resh 1979; Erman and Erman 1984), and heterogeneity (Hynes 1960; Tolkamp 1980) influence macroinvertebrate distribution and colonization. Combinations of variables (the habitat structure) have been shown to explain a greater proportion of the variation in habitat analyses than single-parameter models (Statzner et al. 1988).

In this paper, we assess the effects of hydromorphological degradation on macroinvertebrates in rivers with the use of data at a range of spatial scales. We use selected macroinvertebrate indicators (metrics) and river habitat assessment methods commonly employed in the assessment of ecological status in rivers throughout Europe to analyze the effects of hydromorphological degradation and to discern whether commonly used macroinvertebrate indicators could detect stress effects caused by hydromorphological factors. We use the results to illustrate key points concerning our ability to detect deterioration caused by anthropogenic changes to hydromorphology.

MATERIALS AND METHODS

This paper contains a number of examples on whether and how hydromorphology links to macroinvertebrate indicators. The data used differ with regard to spatial scales, sampling strategy, and the detail and method in which hydromorphology was assessed. All data were collected as part of EU-funded or national projects.

Large-scale patterns and the influence of hydromorphology

Data from national monitoring programs in Denmark, Slovakia, and Sweden covering macroinvertebrates, hydromorphological quality, water chemistry, and catchment parameters were compiled, creating one large database. A total of 1049 sites were included in the analysis (Table 1). Catchment data were collected from the national monitoring databases and included information on altitude of sampling site, catchment area, and catchment land use. Land use was divided into land use classes: pristine (no direct human effect), forest, arable land, and urban areas. Hydromorphological data from Denmark included the national physical habitat quality index (Pedersen et al. 2006). The quality scores ranged from 0 (most affected) to 41 (least affected). The Slovak hydromorphological protocol (SHMI 2005) that was used in Slovak streams scored sites in 4 categories of parameters, and the final score is an integer between 1 (least affected) and 5 (most affected). The Swedish data did not directly include a morphological degradation score. A principal components analysis (PCA) was used to create a score with site information about current velocity, substratum, stream dimensions, macrophytes, catchment land use, and riparian land use. The resulting PCA 1 axis was used as a measure of morphological degradation describing a gradient from fast-flowing streams, with coarse substratum and natural vegetation in the riparian corridor as well as in the catchment, to streams with slow flow, fine substratum, and heavily modified riparian vegetation and agricultural or urban catchment land use. The PCA 1 axis score ranged from −5.5 (unmodified) to 3.3 (heavily modified). Because methods assessing hydromorphology varied among countries, morphological degradation was standardized as a number between 0 (no effect) and 1 (maximum effect). To use substratum coverage as a measure of stream heterogeneity, the substratum data were homogenized, and coverage of each substratum type was calculated. Because individual countries use
different scales for assessing the substrata, all inorganic substratum data were transformed into 4 comparable groups: cobbles (>64 mm), pebbles/gravel (2.0–64 mm), sand (0.063–2.0 mm), and silt (<0.063 mm). The coverage was used directly in the analyses alongside the score for morphological degradation.

The average chemical concentrations were calculated on the basis of spot samples collected throughout the year (N = 6–12). Samples taken during or just after high flow events were not included in the analysis. If multiple samples were not available, spot samples of water chemistry from the site were used if it had been sampled within a month before the biological sampling.

Macroinvertebrate samples were taken between March and October. The field sampling in Denmark and Sweden followed the national protocols with the use of either time-limited kick sampling or a fixed number of kick samples (Swedish Environmental Protection Agency 1996; Skriver et al. 2000). In Slovakia, the ecological quality of streams and rivers throughout Europe according to the benthic macroinvertebrate (AQEM) assessment protocol was adopted to routine monitoring (Hering et al. 2003). Macroinvertebrate samples were aggregated to genus and family level to homogenize the taxa lists. This taxonomic adjustment was carried out in order to eliminate differences in taxonomic resolution among the samples from different countries. From the macroinvertebrate samples, the average score per taxon (ASPT; Armitage et al. 1983) was calculated by ASTERICS software (Furse et al. 2006).

Regression analysis was used for quantifying relationships between hydromorphological degradation, individual hydromorphological parameters, catchment characteristics, organic pollution, and macroinvertebrate metrics. Both single regression and multiple regression analyses were performed. In all analyses, country was used as a covariate to take into account variability among countries. The regression model describing the response of a biotic metric (Y) to a number of explanatory physical and chemical parameters (X1, X2, ..., Xn) can be described as $Y = a_i + \beta_1 X_1 + \ldots + \beta_n X_n$, where: i is country. In this way, models were built for each country separately, as well as the countries in common, wherein the influence of country on the response was taken into account.

**Effect of channel hydromorphology at the reach scale**

Sampling was done in 28 Swedish streams that were sampled as part of the STAR project (Furse et al. 2006) along a hydromorphology gradient and were chosen to fall along a gradient from “high” to “bad” ecological status according the

<table>
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<tr>
<th>Country</th>
<th>Nr of sites</th>
<th>Catchment parameters</th>
<th>Macroinvertebrate parameters</th>
<th>Hydromorphology parameters</th>
<th>Water chemistry parameters</th>
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<td>628</td>
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<td>Total N (mg/L)</td>
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<td>Catchment area</td>
<td>ASPT</td>
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<td>Morphological degradation</td>
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<td>172</td>
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<td>Genus-/family-level taxon lists</td>
<td>Substratum coverage</td>
<td>BOD5 (mg/L)</td>
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ASPT = average score per taxon; BOD5 = biochemical oxygen demand on day 5.

a The hydromorphological degradation score was generated from a principal components analysis of in-stream parameters (see text).
EU WFD. The sites were situated in south-central Sweden in Illies ecoregion 14; each stream was found in a different subcatchment.

Macroinvertebrates were sampled according to the STAR-AQEM method (Hering et al. 2003), and samples were taken in the autumn of 2002 (October–November) with a hand net (mesh size 0.5 mm). At each study site (25- to 50-m stream stretch), the microhabitat composition (e.g., mineral substratum classification) was assessed according to Hering et al. (2003). The coverage of all mineral substratum types with more than 5% cover was recorded to the nearest 5%. Estimation of mineral microhabitat, the sum of coverage of individual mineral microhabitats, equaled 100%, whereas the cover of biotic microhabitats (e.g., wood, coarse particulate organic matter [CPOM], macroalgae, and submerged macrophytes) estimated in the same manner varied from 0% to 100%. Macroinvertebrate samples were collected according to a stratified random sampling procedure in which 10 replicate samples were collected in a riffle habitat, as well as 10 replicate samples in a pool habitat. Each replicate sample covered an area of 25 × 25 cm, and a total of 20 such replicates were distributed across the sampling site in relation to substratum composition at the site (e.g., if the sampling site was covered by 50% sand, then 10 out of the 20 samples were taken in sand substratum). Macroinvertebrates were preserved in 70% ethanol, sorted, and identified to the lowest taxonomic unit possible (usually to species). A number of biotic indices were calculated: number of taxa, abundance (number of individuals), Shannon–Wiener diversity index (Shannon 1948), ASPT (Armitage et al. 1983), Danish stream fauna index (DSFI; Skriver et al. 2000), and number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa (Lenat 1984).

Each sampled river or stream also was assessed for its hydrological and morphological features by the United Kingdom–developed River Habitat Survey (RHS) method (Raven et al. 1998). The RHS methodology includes the collection of qualitative and quantitative data of river hydrology and morphology, geomorphology, habitat quality, and land use. The RHS was designed to assess the quality and assessment of a river or stream; therefore, a habitat quality assessment (HQA) value, as well as a habitat modification score (HMS), has been developed. In the RHS, a total stretch of 500 m along a river or stream is inventoried with 10 so-called “spot checks” spaced every 50 m and comprising in-stream physical features such as stream flow, channel vegetation, and substratum type, as well as bank features such as land use and modifications in the channel and along the banks. An extensive list of additional features in the full 500-m stretch is also collected in the so-called “sweep up.” This includes features such as land use, bank profile, extent of trees, extent of channel features, and evidence of recent management activities. The HQA includes a number of features of habitat quality, such as number of flow types, channel substratum types, number of channels, and bank characteristics. The HQA is given as the sum of scores of each individual characteristic so that sites with numerous and diverse natural features score high. A typical HQA score lies between 10 and 80. The HMS on the other hand quantifies the extent and effects of modification on the river section. Modifications are scored on the basis of their extent and are weighted on the basis of their effect. A typical HMS varies between 0 (no sign of an effect on the section) and 100 (many characteristics of anthropogenic effects on the river section).

Comparisons of HQA and HMS are only meaningful if the rivers or streams subject to investigation are similar in terms of their natural features (e.g., in terms of substratum variability and number of bank features). Besides the final HQA and HMS scores, in this study, it was also possible to include the subscores for the different RHS features. Here, 5 subscores based on the HQA score and 2 subscores based on the HMS also were included on the basis of the results of Szoszkiewicz et al. (2006), including HQA scores of flow types, channel substratum, bank vegetation, channel vegetation, and land use within 50 m of the stream; HMS scores based on information from the spot checks; and HMS scores based on information from modifications at sites other than spot checks.

All relationships were assessed through simple linear regression in the statistical package JMP version 7.0 (SAS Institute, Cary, NC, USA).

Specific effect on channelization at the reach scale

The study was carried out in 39 small 1st- and 2nd-order streams in Denmark (width > 2 m). Channelization and dredging had disrupted natural channel morphology in varying degrees in the investigated streams. The streams thus represented a gradient, from pristine conditions to heavily disturbed morphology, and were assessed to be minimally affected by other anthropogenic stressors.

All field sites were visited in early summer (May/June). At each stream site, a 20-m representative reach was selected covering approximately 1 or 2 riffle-pool sequences. Cross sections were established every 5 m along the reach. The cross-sections were leveled by use of laser equipment (Rec-Elsa-5, Zeiss). This enabled the slope (I) of the stream bed and the exact water surface width (w) to be calculated in each cross section. The variation in stream bed slope was calculated as the coefficient of variance of all 5 slope measurements (I CV). The water surface depth (D) was measured at 10 points across each of the 5 cross sections, and total water-covered cross-sectional area in each transect (A) was calculated, along with the coefficient of variance (ACV). Sinuosity of the stream channel was calculated from map (1:25000) measurements of the Talweg stream length, between 2 points located 1000 m apart, divided by the length along a straight line (Leopold et al. 1964). The sinuosity measurement was verified in the field. The height of the stream banks (H) was measured in each transect as the distance from the water surface to the top of the bank. The morphological index was calculated as

\[ MI = (I/10 + I_{CV} + w/2 + A_{CV} + S + 0.3/H)/6 \]

with MI values increasing as channel morphology complexity increases.

Macroinvertebrates were collected by kick sampling in accordance with guidelines of the DSFI (Skriver et al. 2000): Kick samples were taken with a standard hand net (25 × 25 cm opening, 0.5 mm mesh size). Four standardized kick samples were taken on a transect across the channel at equal intervals (25%, 50%, 75%, and 100%) from one of the stream banks to ensure all habitats were covered. At each reach, kick samples were taken in 3 transects 10 m apart, giving a total of 12 samples that were subsequently pooled. Samples were preserved in 70% ethanol and identified to species level, if possible. Macroinvertebrate community quality was expressed as the ASPT score (Armitage et al. 1983).
**Detailed study of reaches varying in physical complexity**

The study was undertaken in 6 streams of 1st- and 2nd order (1–3 m wide) within the same geographical area in Denmark. In each stream, 2 reaches were investigated, with the upper reach having little physical variation because of historical channelization and management (hereafter denoted homogeneous reach) and the lower reaches having natural physical variation (hereafter denoted heterogeneous reach). In each stream, the 2 reaches were never more than 1 km apart, with no in-stream obstacles that could prevent upstream dispersal between reaches. Streams investigated were unpolluted, with average values of biochemical oxygen demand on day 5 (BOD₅) below 2 mg/L and no significant differences in BOD₅ levels or nutrients between reaches in any of the streams.

Physical features were measured every meter along a 50-m representative section of each reach. Stream depth and width were measured and substrate types assessed at individual points spaced 10 cm apart in each of the 50 transects. Substrate types were assessed visually as cobble (64–256 mm), pebble/coarse gravel (10–64 mm), and fine gravel/sand (>10 mm), and organic substrates were assessed as fine particulate organic matter (FPOM; <1 mm) or CPOM (>1 mm). Above each substrate type, current velocity was measured with a Höntzsch digital anemometer (Höntzsch Instruments, Germany) at 10 random points from 1.2 cm above the bottom and in 60% of the water depth.

Macroinvertebrates were collected qualitatively by searching all in-stream habitats for 1 h with forceps and a hand net (mesh size 0.5 mm) and by kick sampling in accordance with guidelines of the DSFI (Skriver et al. 2000). All samples were preserved in 70% ethanol and identified to species level, if possible. Quality was assessed with the use of DSFI values, which were calculated in accordance with the procedure described in detail in Skriver et al. (2000). However, to understand the following results, certain parts of the calculation procedure need explaining. One step in the calculation of the DSFI value is to determine the number of diversity groups, which is done by subtracting the number of defined negative diversity groups from the number of defined positive diversity groups (i.e., the higher the number of diversity groups present at the site, the higher the ecological quality). Macroinvertebrate taxa belonging to the positive diversity groups are those intolerant of organic pollution, whereas a negative diversity group comprises taxa more tolerant of organic pollution. The DSFI index values range from 1 (severe pollution) to 7 (unpolluted conditions).

**RESULTS**

**Large-scale patterns and the influence of hydromorphology**

Regression analysis showed weak relationships between the morphological degradation score and the various macroinvertebrate indices when analyzing data from Denmark and Sweden ($R^2 < 0.30$ always). In contrast, the relationship was stronger in the Slovakian data set, with the best relationships being between morphological degradation and number of EPT families ($R^2 = 0.46$) and ASPT ($R^2 = 0.42$; Figure 1). Furthermore, the Slovakian data could be divided into reference sites and affected sites and these showed distinct differences in the morphological degradation score (Figure 1). All reference sites had high scores and showed very limited variability in ASPT values, whereas affected sites showed a much higher degree of scatter around the regression line.

Analyzing data by multiple linear regression, including all environmental variables and with country as a co-variable, gave more significant relationships, the strongest being with ASPT and total P (PTOT), percent arable land in the catchment (ARABLE), and morphological degradation (MD) expressed as ASPT $= 5.26 - 3.33*PTOT - 0.014*ARABLE + 2.32*MD$ ($R^2 = 0.61$). Country explained most of the variance in the model (26%), with the environmental variables explaining similar amounts (PTOT $= 13\%$, ARABLE $= 13\%$, MD $= 9\%$). The best relationship considering only morphological degradation was also with ASPT ($R^2 = 0.51$), in which case country explained 32% and MD 19%. All other relationships tested were weaker ($R^2 < 0.42$).

**Effect of channel hydromorphology at the reach scale**

The HQA varied from 16 to 64; 4 sites had a score of less than 40 and 6 sites had a score of at least 60. For HMS, 1 site had a score of 47, 3 sites scored 20 to 22, and a total of 21 sites out of the 28 had an HMS below 10, with 6 sites scoring 0. Both HQA and HMS indicated that most sites overall had a good hydromorphological condition. HQA and HMS were not correlated ($p > 0.05$), although a negative relationship was expected, in that a high HQA shows that the site is of high quality and a high HMS indicates that the site is heavily modified. The HQA was also correlated with the different HQA subscores, and here a strong relationship was found between all HQA subscores (adjusted $R^2 = 0.11–0.55$, $p < 0.05$) and the final HQA score, except for the HQA arising from land use within 50 m from the stream ($p > 0.05$). The final HMS score was strongly related to the HMS subscore on the basis of modifications at spot checks only (adjusted $R^2 = 0.86$, $p < 0.001$), whereas no correlation was found between the final HMS and the HMS subscore based on modifications at sites other than those spot-checked.

The total number of taxa varied between 17 and 50, and the number of collected individuals ranged between 253 and 15714. ASPT varied between 4.6 and 7.0, and DSFI ranged between 4 (2 streams) and 7 (13 streams), both indicating that most streams were at least in a good ecological condition. The number of EPT taxa varied between 4 and 33, and the Shannon–Wiener index fell between 2.16 and 4.51.

Only benthic macroinvertebrate abundance was weakly correlated with the final HQA score (Table 2). None of the ecological indicators were correlated to either the flow, bank, or channel vegetation subscores. ASPT and number of EPT taxa were correlated with land use, whereas the highest number of ecological indicators was correlated to in-stream substratum quality (total number of taxa, ASPT, DSFI, and number of EPT taxa). The Shannon–Wiener diversity index was not correlated to any of the HQA scores. The HMS did not correlate with any of the 6 ecological indicators (Table 3), and neither did it correlate with the spot-check data or non-spot-check information.

**Specific effect of channelization at the reach scale**

Morphological index values ranged from 1 to 25.8 in the 39 streams investigated, covering the entire range from very uniform straightened channels to physically heterogeneous streams. The relationship between ASPT and the morphological score was not significantly positive (Figure 2; $R^2 = 0.43$).
Detailed study of reaches varying in physical complexity

Heterogeneous and homogeneous reaches varied markedly with respect to physical features. Shallow areas were more common, and stream width was more variable in the heterogeneous reaches. On average, 4 different substrate types covered more than 5% of the stream bed in the heterogeneous reaches and only 3 in the homogeneous reaches. Cobble and gravel/pebble bottom substrate were significantly more abundant in heterogeneous than in homogeneous reaches (\( p < 0.01; \) Mann–Whitney \( U \) test).

Average bottom cover of cobble and gravel/pebble was 45% in the heterogeneous reaches and only 3% in the homogeneous reaches. Current velocity was closely related to substrate type: The coarser the substrate, the higher the current velocity. Consequently, both the average and variation in current velocity within a reach were higher in the heterogeneous than the homogeneous reaches.

In the qualitative samples, the total number of taxa was significantly higher in the heterogeneous than the homogeneous reaches (\( p < 0.01; \) Mann–Whitney \( U \) test). On average,

Table 2. Habitat quality assessment (HQA) score and subscores compared with the macroinvertebrate environmental indicators in the 28 Swedish streams

<table>
<thead>
<tr>
<th>HQA feature</th>
<th>Total nr of taxa</th>
<th>Abundance</th>
<th>ASPT</th>
<th>DSFI</th>
<th>Nr of EPT taxa</th>
<th>Shannon–Wiener</th>
</tr>
</thead>
<tbody>
<tr>
<td>Final score</td>
<td>Adjusted  ( R^2 )</td>
<td>0.16</td>
<td>0.11*</td>
<td>0.0</td>
<td>0.03</td>
<td>0.10</td>
</tr>
<tr>
<td>( p ) Value</td>
<td></td>
<td>&gt;0.05</td>
<td>&lt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
</tr>
<tr>
<td>Flow</td>
<td>Adjusted  ( R^2 )</td>
<td>0.02</td>
<td>0.0</td>
<td>0.0</td>
<td>0.02</td>
<td>0.0</td>
</tr>
<tr>
<td>( p ) Value</td>
<td></td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
</tr>
<tr>
<td>Substratum</td>
<td>Adjusted  ( R^2 )</td>
<td>0.24**</td>
<td>0.04</td>
<td>0.11*</td>
<td>0.20**</td>
<td>0.25**</td>
</tr>
<tr>
<td>( p ) Value</td>
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<td>&lt;0.005</td>
<td>&gt;0.05</td>
<td>&lt;0.05</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Bank vegetation</td>
<td>Adjusted  ( R^2 )</td>
<td>0.05</td>
<td>0.02</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>( p ) Value</td>
<td></td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
<td>&gt;0.05</td>
</tr>
<tr>
<td>Channel vegetation</td>
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<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>( p ) Value</td>
<td></td>
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<td>&gt;0.05</td>
</tr>
<tr>
<td>Land use</td>
<td>Adjusted  ( R^2 )</td>
<td>0.07</td>
<td>0.0</td>
<td>0.20**</td>
<td>0.06</td>
<td>0.17*</td>
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<tr>
<td>( p ) Value</td>
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<td>&gt;0.05</td>
<td>&lt;0.01</td>
<td>&gt;0.05</td>
<td>&lt;0.05</td>
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</tbody>
</table>

ASPT = average score per taxon; DSFI = Danish stream fauna index; EPT = Ephemeroptera, Plecoptera, and Trichoptera.

\* \( p < 0.05 \), \** \( p > 0.01 \), \*** \( p > 0.001 \).
we found 38 (range, 27–45) taxa in heterogeneous reaches and 26 (range, 22–30) taxa in homogeneous reaches. The DSFI index value was higher in the heterogeneous than the homogeneous reaches in all but 1 stream (Table 4). In all streams we found a higher number of diversity groups in the heterogeneous than the homogeneous reaches (Table 4). This reflected the higher number of negative and lower number of positive diversity groups in all the homogeneous compared with heterogeneous reaches. A significantly greater number of taxa belonged to the positive DSFI diversity groups found in the heterogeneous (average, 12; range, 10–14) than the homogeneous reaches (average, 7; range, 4–9; $p < 0.01$; Mann–Whitney U test).

**DISCUSSION**

Our review of the scientific literature reports many linkages between individual parameters (substratum, current velocity, etc.) describing the in-stream physical environment and different attributes of the macroinvertebrate community. High-resolution measurements on a small scale without other confounding factors have highlighted the importance of physiobiological interactions in streams (Hart and Finelli 1999). One reason is that numerous physical factors interact across different temporal and spatial scales. Furthermore, the distribution of biota not only responds to the physical environment but also to biotic interactions (e.g., Dudley et al. 1990; Lancaster 1990), water chemistry (Minshall and Minshall 1978), temperature (Sweeney and Vannote 1981), oxygen levels (Nebecker 1972), and food resources (e.g., Cummins 1973; Minshall and Minshall 1977). In summary, we can therefore identify a number of scientific challenges that still need to be addressed before we understand how the physical environment interacts with in-stream biota. This incomplete understanding can partly explain why our ability to detect the effects of hydromorphological degradation on macroinvertebrate indicators is relatively poor, as shown in this study.

Another key issue when assessing the influence of hydromorphology is that interpretation of results is often confounded by multiple stressors influencing macroinvertebrates.
Typically, streams that have undergone a high degree of habitat degradation or alterations of flow regimes will be situated in areas with multiple anthropogenic pressures. This was clearly demonstrated in the large-scale analysis of stream sites in this study, wherein the best relationships were obtained by combining chemical, catchment, and habitat variables. A number of studies have recognized that most river systems are influenced by multiple stressors acting simultaneously. Stressors can interact in a synergistic manner (Folt et al. 1999; Matthaei et al. 2006), as exemplified by increased concentrations of easily degradable organic matter, which will have a more detrimental effect in a habitat-degraded stream because the number of reactive surfaces and the re-aeration capacity are reduced (Andersen 1994). Specifically, the Slovak data (Figure 1) illustrated that the response to hydromorphological degradation became more variable with increasing detrimental effects, indicating that other stressors also influenced macroinvertebrate communities simultaneously.

Our results from 3 separate studies in a number of streams in which hydromorphology was expected to be the main stressor revealed generally weak to moderate relationships between indicators of morphology and macroinvertebrate indicators. When employing the commonly used tool to assess hydromorphological status, RHS, neither the habitat quality of the sites as measured by the HQA score nor the modification measured with the HMS score was strongly correlated with any of the commonly used macroinvertebrate ecological indicators. This result suggests that commonly used macroinvertebrate indicators cannot be used to detect changes in hydrological or morphological features measured as HQA and HMS. These findings can probably be attributed to the fact that macroinvertebrate indicators are mainly developed to detect nutrient enrichment or general ecological quality changes (Armitage et al. 1983; Barbour and Yoder 2000; Skriver at al. 2000) and therefore are not sufficiently sensitive in detecting hydromorphological stress. Another contributing explanation is likely to relate to the scales of the assessment methods. The RHS is based on fairly large-scale assessments (hundreds of meters) of features that can be detected visually, mostly from the bank, whereas distribution of macroinvertebrates often are determined on a much smaller scale (e.g., Quinn and Hickey 1990; Cogerino et al. 1995). Furthermore, Pedersen and Friberg (2007) found significant differences in macroinvertebrate composition between 2 visually similar morphological features (riffles) adjacent to each other. One other important issue is that RHS do not directly assess hydrological dynamics of a given site, and it is well documented that disturbance regime and substrate stability affect stream macroinvertebrate community structure (Quinn and Hickey 1990; Iversen et al. 1991; Cogerino et al. 1995; Death and Winterbourn 1995). Regarding subscores, the substratum variability and quality HQA subscore was the environmental variable most strongly related to the macroinvertebrate ecological quality indicators, whereas none of the 4 indicators were directly related to flow type, which might be attributed to the lack of temporal resolution in the assessment of flow in RHS. The general benthic macroinvertebrate diversity index (Shannon–Wiener) performed less well than the water quality indices, with no significant correlations to any of the HQA scores. This finding supports previous reports (see Metcalfe-Smith 1996 for review) that patterns in diversity do not necessarily reflect a perturbation gradient.

Overall morphology of a stream reach showed a moderately good correlation with ASPT (Figure 2). Compared with the RHS assessment, the morphological index used was based on a number of actual in-stream measurements of slope, width,
and depth, which might have improved precision compared with a visual assessment. Differences in sampling methods are unlikely to have affected results in that they have been shown to yield very similar numbers and composition of macroinvertebrates (Friberg et al. 2006).

In the fine-scale study of the 6 streams with reaches differing in physical complexity, the heterogeneous reaches were clearly more physically varied with respect to depth, width, flow, and substrate condition than the homogeneous reaches. These differences were partly picked up by DSFI, which is recognized as being sensitive to general degradation as well as organic pollution (Skriver et al. 2000). The quality objectives of most natural Danish streams require a DSFI value of 5 to be classified as having good ecological status in WFD context. This value was only found in 2 of the 6 homogeneous reaches, and consequently the inability of these reaches to obtain a sufficient status class can be attributed to hydromorphological degradation alone given the single stressor design of the study. However, differences were not significant with regard to DSFI values, whereas the number of diversity groups used in calculating the DSFI was significantly higher in all heterogeneous compared with homogeneous reaches. This indicates that some sensitivity toward hydromorphological degradation is lost in the final step of calculation, which involves ranking species found, primarily according to their tolerance toward organic pollution.

It is important to acknowledge, however, that hydromorphological changes might not perturb macroinvertebrate communities to the same extent as stressors relating to water quality. Certainly, even though our study in the 6 reaches affected only by hydromorphological change showed consistently lower quality than more physical diverse reaches, it was still less than would be expected if the streams had been heavily affected by, as an example, untreated sewage. A general lack of studies showing the significant, quantitative effects of hydromorphology alone can largely be attributed to the occurrence of multiple stressors and the use of inappropriate methods, as already discussed. For this reason it is, at present, not possible to rank and quantify the importance of hydromorphology to that of other stressors, but hopefully future studies will be able to address this issue.

**Outlook—Future risk assessment of hydromorphological degradation**

The linkage of reach scale physical parameters and biotic samples on sites only disturbed by physical alterations are scarce. In our study, we were only moderately successful in linking biota with various types of habitat surveys exemplified by the generally weak relationships between morphological degradation and macroinvertebrate indices. We attribute this to a combination of factors, including the mixed nature of pressures acting on the majority of river reaches, scaling issues when relating habitat surveys to macroinvertebrate assessments, and the scope of commonly used macroinvertebrate assessment systems.

For monitoring purposes, it is essential that different types of stress can be disentangled from each other because the ecological status in most cases will reflect different types of pressures. The multimetric approach with macroinvertebrates (e.g., Barbour and Yoder 2000) and methods that use multiple indicators to assess the quality of the river (e.g., Fore et al. 1996; Weigel et al. 2003; Hering et al. 2006) are important steps in this direction. However, methods sensitive to multiple stressors need further development, especially with regard to detection of hydromorphological effects.

As exemplified in our study, the relationship between RHS and macroinvertebrate indices was not strong. Moreover, we found the best relationship between hydromorphological degradation and ASPT in the Slovak data, which used a newly developed, WFD-compliant habitat assessment method that is used together with macroinvertebrate sampling (SHMI 2005). Our other results showed that a more detailed assessment of channel and sediment conditions improved overall relationships. A more detailed assessment of sediment conditions, especially at the scale on which macroinvertebrates are sampled, is likely to increase explanatory power substantially, a contention supported by numerous studies reporting sediment conditions as one of the primary drivers of macroinvertebrate community composition (e.g., Minshall 1984; Wood and Armitage 1997; Miyake and Nakano 2002).

Most currently used macroinvertebrate sampling methods have very limited stratification of effort by habitats and no sample replication (Friberg et al. 2006). Hydromorphological degradation, in contrast to organic pollution and pH, is likely to induce changes in the proportion of biotopes available, and hence the relative abundance of species, rather than exclude major components of the macroinvertebrate community. Novel strategies should be developed that encompass habitat-specific sampling and appropriate replication. With regard to indices, only a very few specifically target hydrology and hydromorphology (Barbour et al. 1996; Extence et al. 1999; Lorenz et al. 2004), and more effort has to go into the development of these. Increased use of abundance data and the inclusion of various biological or ecological traits such as body size, life history, and food preference (Townsend and Hildrew 1994; Usseglio-Polatera et al. 2000; Verdonschot and Moog 2006) could be cornerstones in the development of new metrics sensitive to hydromorphological degradation.

Hydromorphological degradation is one of the most important pressures on European rivers and imposes a serious risk to freshwater communities, either when acting alone or in combination with other pressures. Impairment of hydromorphology in river ecosystems is likely to significantly reduce resistance toward other pressures on the ecosystem (Andersen 1994). Hydromorphology is currently assessed with a range of techniques that are suboptimal in that they lack appropriate sensitivity as well as the ability to quantify the importance of individual pressures. The need is urgent to develop refined and updated assessment systems targeting hydromorphology for use under the WFD and other national water-related policies (e.g., the Swedish Environmental Objective “Flourishing Lakes and Streams”).

**Acknowledgment**—This study was partly funded by the European Union 6th Framework Programme Project REBEC-CA (Contract SSPI-CT-2003-502158) “Relationships between ecological and chemical status of surface waters” and the European Union 5th Framework Programme Project STAR (Contract EVK1-CT-2001-00027), by the Danish Ministry of the Environment and by the Scottish Executive Environment and Rural Affairs Department (WP3.4) for N Friberg.

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