Environ Earth Sci
DOI 10.1007/s12665-015-4466-8

THEMATIC ISSUE

Community–environment relationships of riverine invertebrate communities in central Chinese streams

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Received: 22 December 2014 / Accepted: 24 April 2015

Abstract Chinese rivers are both highly biodiverse and highly under pressure, hence an urgent need exists to understand ecological drivers and disentangle different scales of stressors to support water management. Our aims were to (1) determine the most influential variables for benthic invertebrate occurrence, (2) compare results related to communities as opposed to metrics and (3) examine the role of spatial scales with relevance to management. Benthic invertebrate sampling was performed at 37 sites in selected tributaries of the middle reaches of the Yangtze, covering an environmental gradient with a focus on organic pollution (stratified sampling design). Ten metrics commonly used in biomonitoring were derived and analysed in parallel to assemblage data. Environmental variables covered 74 parameters from three different spatial scales, namely local, reach and catchment scale. We ran a CCA with each of the three subsets to find out the significant determining/explanatory variables, followed by pCCA and pRDA (for metric data) with these variables with forward selection to determine single variables important for each subset; we further used variation portioning for benthic invertebrate data. A high percentage of overall variability (70 %) of the assemblage structure was explained, with catchment- and local-scale variables being almost equally important. Small-scale variables tended to be more important than large-scale variables for the metric-based approach but not for the assemblage approach. Our results emphasise the need for spatially explicit regional studies in freshwater systems and suggest testing multi-metric assessment approaches to tackle water management and environmental health questions in China.

Keywords Biomonitoring · Hubei · Land use · Spatial scale · Variation partitioning

Introduction

Stream invertebrate communities are regulated by processes and mechanisms operating at varying spatial and temporal scales, such as from catchment to local (Vinson and Hawkins 1998). The resulting community characteristics may be used to detect overall responses to environmental gradients, hence enabling the comparison of environmental links with community structure (Tonkin 2014). These patterns may even distinguish one impact

Sonja Stendera's views expressed here are personal and not those of the employer.

Electronic supplementary material The online version of this article (doi:10.1007/s12665-015-4466-8) contains supplementary material, which is available to authorized users.

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type over another, and such discrimination between different stressors provides a benefit for management (e.g. identifying cause and effect) or modelling (e.g. to predict stressor response, determine predictor variables) and eventually application (e.g. to develop cost-effective restoration/management programmes). It is obvious that the consideration of the appropriate scale for both detection of underlying ecological patterns and related stress responses is fundamental to deduce appropriate conclusions (Astorga et al. 2011; Heino 2009; Heino et al. 2003).

This principal of ecological controls of stream communities has been widely reported, and stressors and anthropogenic influences are remarkably similar but may vary in strength or current and historic prevalence. For instance, point source chemical and organic contaminants remain prevalent in developing countries, whereas diffuse pollution associated with land use tends to be the main stressor for freshwater systems in developed countries. For some regions, this has already led to very sophisticated assessment systems based on invertebrate and other organism groups including fish or algae in Europe, Australia, North America, Japan and South Korea, for instance (Clarke et al. 2003; Hering et al. 2004; Jun et al. 2012; Komori et al. 2013), while for other populated regions such as China and many other Asian countries, few studies are available (Azrina et al. 2006; Dudgeon 2006; Korte 2010; Meng et al. 2009; Zhang et al. 2008; Zhao and Yang 2009). It is widely acknowledged that using univariate metrics (reflecting biodiversity, community composition or functional aspects) are useful to reduce complexity, mirror environmental stressors and enhance comparability across systems, but it has been shown that such metrics may vary in their response and sensitivity (Sundermann et al. 2013; Tachamo Shah and Shah 2012) or capability to respond to local-scale variability underlying the gradients in which they were designed to reflect (Tonkin 2014).

However, rather little about such problems is known in developing countries such as China (but see e.g. Jiang et al. (2010); Li et al. (2012); Zhang et al. (2010)), which is in contrast to the urgent need to disentangle different scales of stressors, enable appropriate detection of and allow for prioritisation of management measures. Central China is no exception to this; however, this is an extremely interesting area as it hosts the transitional area between subtropics and the warm temperate biogeographic zones, thus representing an important zone for biodiversity (Myers et al. 2000). China faces severe problems in providing adequate drinking water in terms of both quality and quantity, as well as sufficient water to maintain a high agricultural production level for food security; these requirements dominate daily decisions on water management (Chen et al. 2015). Water abstraction is one of the primary stressors in Chinese freshwater ecosystems (mainly large rivers, groundwater, reservoirs), along with chemical and organic pollution. Furthermore, these systems are under stress from various other sources including the use of rivers for aquaculture, fisheries and transportation. While there are significant numbers of water environment monitoring centres and stations in China, and all major Chinese rivers, lakes and reservoirs are monitored routinely, these units mainly focus on physicochemical parameters such as pH, conductivity, dissolved oxygen, permanganate, ammonium, etc.; only in few further developed eastern provinces like Jiangsu and Zhejiang, bioassessment is part of protocols, too (Wang et al. 2014). First approaches to ecological assessment of freshwater ecosystems started in the late 1950s (Chen 1959), with more recent results being available, for example, from Jiang et al. (2009); Qu et al. (2005); Tang et al. (2006), but so far a unified national bioassessment protocol is not available (Wang et al. 2014).

Given these current limitations in understanding of Chinese stream communities and the desire to develop robust monitoring regimes, our aim was to study community–environment relationships in a relatively understudied region of central China by (1) determining which variables most strongly influence benthic macroinvertebrates (a functionally important group of organisms in river ecosystems), (2) comparing between community structure and commonly applied ecological metrics and (3) examining the importance of spatial scales of environmental drivers relevant to management.

Materials and methods

Study region

The study region (Fig. 1) is located in the western part of Hubei province in China (ca. 30°58′N; 112°14′E), which is characterized by several mountain ranges, namely Qinling, Wudang, Wushan, Jingshan and Dabashan mountains. Parts of these belong to the Shennongjia-Nature Reserve. The area is located in a transitional zone between the northern subtropical zone and warm temperature zone and has a humid climate. The average annual precipitation is 1000 mm, with regional maxima up to 1400 mm, of which 75 % fall in the monsoon season between April and August. Mean temperatures range from 3–5 °C in January to 27–29 °C in July/August. Frost occurs regularly in December–February. The sampling area is characterized by high, steep mountains (max. 3000 m) and narrow valleys. Operation of agriculture (citrus fruits, corn) is directly adjacent to rivers or as terraces. The average population density is 80 inhabitants/km^2 (overall China: 130 inhabitants/km^2).

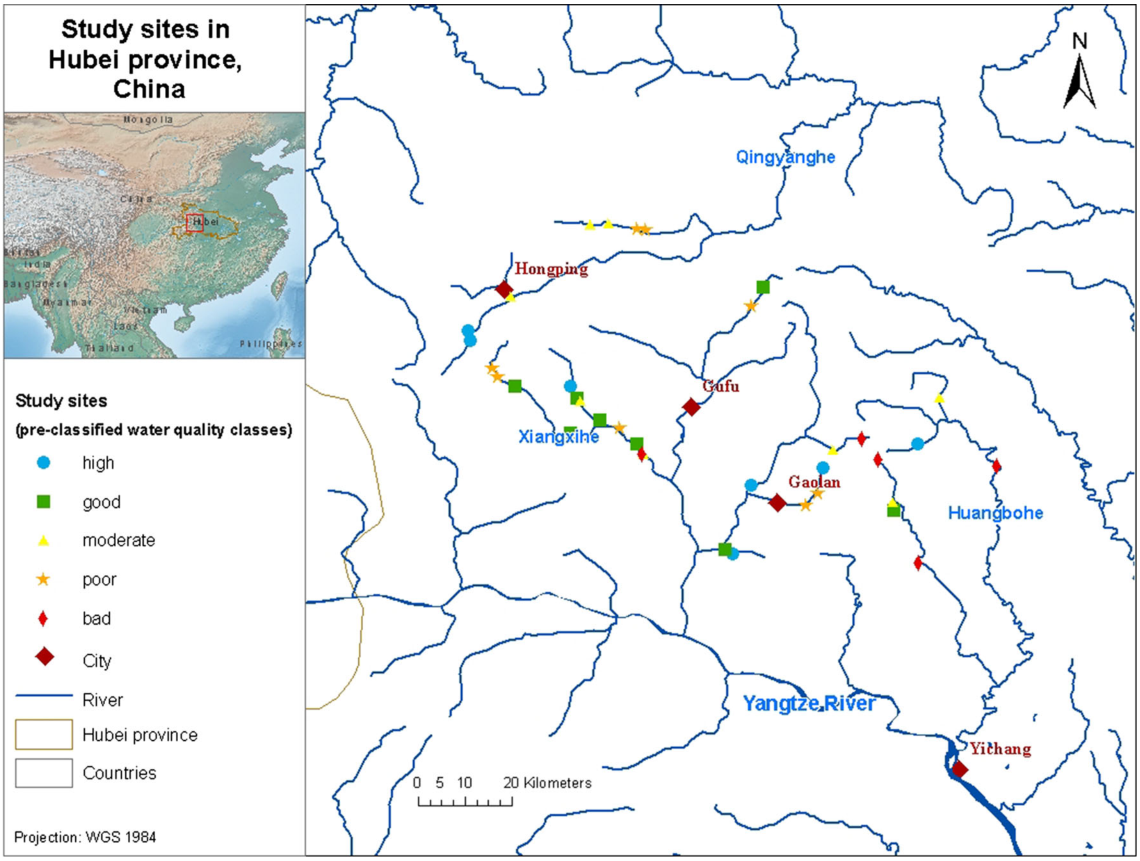

Fig. 1 Map of the 37 sampling sites and location in Hubei province, China

The sampling sites are located in the upper catchments of Xiangxi, Huangbo and Han Rivers, which are all northern tributaries to the middle Yangtze reach. The Xiangxi River originates in Shennongjia Reserve about 3000 m a.s.l. and joins after 94 km the Yangtze at 110 m a.s.l. The catchment covers almost 3100 km^2, and the mean annual discharge is 65.5 m^3/s^1. The Huangbo River, another tributary to the Yangtze and its Gezhouba Reservoir, originates from Heilian Mountain, with a total length of 160 km, draining 1,924 km^2. The mean annual discharge is 10.0 m^3/s^1. Before entering the Yangtze at Yichang city, it passes through 12 towns; it is also the main water resource for industry, agricultural irrigation and drinking water of Yichang. Few sites are located at tributaries of the Han River, the longest Yangtze tributary. The sampled streams can be classified as relatively small (mean stream width = 13.1 ± 11.7 m), naturally nutrient poor and are situated at relatively high altitude (mean = 864.2 ± 399.1 m a.s.l.; Table 1). Catchment area ranges from 5 to 630 km^2, and the dominating geologic formations are silicate and carbonate rock. Within the catchments mixed forest cover is the dominant natural land cover (mean = 38.5 %) reaching up to 90 % in some areas, whereas timber (plantation forestry) (47 % cover) is the dominant artificial land use, followed by pasture, which can reach up to 38 % of the total land use cover. Within reaches natural vegetation cover is dominated by deciduous forests, with on average 23 %, although urban land use is also common (mean = 18 %). The streams are shallow, relatively fast flowing with a relatively high mean water temperature (measured during sampling) of 14 °C; large stones build the predominant substrates.

In sum, the streams of this region are partially under stress from urban and agricultural development, covering a clear natural and anthropogenic environmental gradient, and thus making it in an ideal test area for our study.

Field sampling

Benthic invertebrate sampling was performed at 37 sites in the pre-monsoonal season of 2008, covering a gradient from unpolluted to heavily polluted river stretches with a focus on organic pollution by municipal wastewater (stratified sampling design). The sites had been pre-classified according to the "Guidance manual for pre-classifying the ecological status of Hindukush Himalaya (HKH) rivers" (Moog and Sharma 2005) to ensure a regular distribution along an environmental gradient (five levels from

Table 1 Mean (±1 SD) and range of dependent and independent variables (74 environmental variables divided into three subsets according to different scales) used in partial CCA and partial RDA. Shown are the three single predictors obtained in stepwise CCA and RDA best explaining the variation in assemblage structure and biotic metrics with order of their selection in parentheses and explained variance in %

Variables	Mean ± SD	Min.	Max.	CCA	RDA
Dependent variables					
Abundance	653 ± 441	69	1939		
ASPT	6.74 ± 0.796	4.9	8.2		
CA1 scores	0.19 ± 1.032	−2.1	2.9		
CA2 scores	−0.13 ± 0.988	−2.7	3.9		
Dominance	0.17 ± 0.107	0.1	0.6		
EPT taxa (%)	72 ± 19.4	15	97.4		
Evenness	0.40 ± 0.15	0.2	0.7		
Number of taxa	24.5 ± 7.53	13	38.0		
Shannon diversity	2.17 ± 0.85	0.9	3.2		
Simpson diversity	0.83 ± 0.11	0.4	0.9		
Independent variables					
Catchment					
Area*	132.46 ± 176.54	5	630		
Carbonate rock*	36.4 ± 36.2	0.0	100		
Lacustrine*	6.6 ± 11.8	0.0	45	18 % (3)	
Marine deposit	18.5 ± 27.3	0.0	96		
Silicate	38.5 ± 35.5	0.0	100		
Timber*	47.2 ± 16.3	2.8	80		
Non-irrigated farmland*	3.2 ± 2.0	0.2	8	31 % (1)	
Paddy fields	1.2 ± 1.4	0.0	6		
Mixed forest*	38.5 ± 16.7	7.9	91		
Pasture	9.9 ± 11.8	0.0	38		
Urban*	0.10 ± 0.21	0.0	1	20 % (2)	
Reach					
Crop land	6.8 ± 16.3	0.0	70		
Deciduous forest*	23.2 ± 30.7	0.0	100		
Grass-/bushland	13.0 ± 29.5	0.0	100	19 % (2)	14 % (1)
Meadows	6.2 ± 15.5	0.0	70		
Mixed forest	10.8 ± 31.5	0.0	100		
Terraces*	8.1 ± 15.4	0.0	50	19 % (1)	
Wetland	3.0 ± 10.8	0.0	60		
Nonvegetated land	10.3 ± 19.6	0.0	100		
Urban	17.8 ± 29.4	0.0	100		
Standing water	0.89 ± 0.31	0.0	1		
Floodplain width*	132.0 ± 144.8	0.0	500	16 % (3)	
Channel sinuate*	0.57 ± 0.50	0.0	1		
Channel constrained	0.35 ± 0.48	0.0	1		12 % (2)
U-shaped valley	0.27 ± 0.45	0.0	1		
V-shaped valley	0.30 ± 0.46	0.0	1		10 % (3)
Lakes upstream*	0.24 ± 0.43	0.0	1		
No floodplain vegetation	0.49 ± 0.51	0.0	1		
Local					
Altitude a.s.l. (m)*	864.2 ± 399.1	213.0	1551	20 % (2)	
Longitudinal impoundments	0.43 ± 0.50	0.0	1		
Water level	0.49 ± 0.47	0.0	1		

Table 1 continued

Variables	Mean ± SD	Min.	Max.	CCA	RDA
Stream width	13.1 ± 11.7	2.5	60		
Mean depth (cm)	34.2 ± 10.9	15.0	60		
Mean depth at bankfull discharge*	165 ± 86	50	500		
Mean velocity (cm/s)	50.8 ± 22.4	20.0	120		
Flow type pools	0.68 ± 0.47	0.0	1		
Flow type rapids	0.43 ± 0.50	0.0	1		
Flow type run*	0.73 ± 0.45	0.0	1		8 % (2)
Bank slanting	0.16 ± 0.37	0.0	1		
Bank steep	0.53 ± 0.49	0.0	1		
Mean width of rip. wooded vegetation	6.6 ± 17.0	0.0	100		
% riparian wooded vegetation	38.4 ± 40.8	0.0	100		
Canopy cover (%)	16.2 ± 26.2	0.0	80		
Megalithal (>40 cm)	20.5 ± 16.3	0.0	70		
Macrolithal (>20 to 40 cm)	18.9 ± 10.9	0.0	40		
Mesolithal (>6 to 20 cm)	46.9 ± 22.4	10	90		
Microlithal (<6 cm)	8.0 ± 9.5	0.0	40		
Akal*	1.62 ± 3.74	0.0	15		
Psammal	0.54 ± 1.97	0.0	10		
Macroalgae	0.81 ± 3.44	0.0	20		
Living parts of terrestrial plants	2.03 ± 3.62	0.0	10		
Bank fixation	0.46 ± 0.51	0.0	1		
No. debris dams	0.08 ± 0.36	0.0	2		
No. logs	0.16 ± 0.55	0.0	2		
Abstraction/pulse releases	0.11 ± 0.31	0.0	1		
Source pollution	0.32 ± 0.47	0.0	1		
Non-source pollution*	0.49 ± 0.51	0.0	1	14 % (3)	6 % (3)
Sewage overflows	0.19 ± 0.40	0.0	1		
Removal of coarse woody debris*	0.54 ± 0.51	0.0	1		
Removal of mineral bed	0.14 ± 0.35	0.0	1		
Straightening	0.24 ± 0.43	0.0	1		
Waste disposal	0.46 ± 0.51	0.0	1		
No. transverse structures	0.54 ± 0.87	0.0	4		
NH_3-N*	0.07 ± 0.16	0.0	0.78	25 % (1)	14 % (1)
NO_3-N	0.99 ± 0.50	0.18	2.20		
PO_4-P	0.06 ± 0.10	0.01	0.54		
TN	1.08 ± 0.59	0.25	2.90		
TP	0.09 ± 0.13	0.02	0.56		
Conductivity	268.9 ± 105.6	81.6	533		
DO	9.0 ± 0.79	6.32	10.16		
COD	1.6 ± 1.16	0.78	7.61		
pH	7.9 ± 0.58	6.58	8.56		
Oxygen saturation (%)*	91.2 ± 6.91	66.7	100		
Water temperature	14.0 ± 2.51	10	19.7		

* Other significant variables obtained from initial CCA

unpolluted to highly stressed sites; these data were not further used for analysis). Sampling followed a standardized multihabitat sampling approach, where 20 subsamples representing the full range of habitat composition (total sampling area 1.25 m^2) are taken with a 500-µm shovel sampler. Samples were preserved with 75 % alcohol in the field and sorted and identified in the lab following the Rivpacs protocol (Murray-Bligh et al. 1997); identification was mostly to family or genus level based on the available keys (Morse et al. 1994; Nesemann et al. 2007) and unpublished keys of the Assess-HKH project.

Parallel sampling of chemical and physical parameters (COD, phosphate, nitrate, nitrite, ammonium, oxygen, temperature, pH and conductivity) was performed at all sites. Furthermore, an extensive physical habitat field protocol was completed to build the set of independent variables. These covered 74 parameters from three different spatial scales, namely local, reach and catchment scale (Table 1—with all variables and scales), following existing field protocols from Moog and Sharma (2005).

Statistical analyses

Independent variables

Independent variables used in this study include catchment to local-scale variables, which were divided into three subsets according to scale (Table 1). Scale-dependent data sets were analysed separately. The subset of catchment-scale variables (C) consists of different land use/cover descriptors and geologic formation categories. The reach subset (R) also consists of different land use/cover descriptors as well as floodplain descriptors and channel and valley characteristics (approximately 1 km river length upstream of sampling site). Local variables (L) consist of altitude, stream and water descriptors (e.g. width, depth, bank form and velocity), substrate categories, variables describing anthropogenic impacts such as removal of debris dams, pollution, sewage, straightening, as well as stream physico-chemistry covering the actual sampling site of about 100 m. Prior to all statistical analyses, chemicals, stream width, catchment area and altitude were log-transformed, and proportional catchment and reach land use/vegetation cover variables were arcsine square-root-transformed to achieve normal distributions.

Dependent variables

Dependent variables used in this study were based on a taxa-site matrix describing the assemblage structures of these streams and a set of commonly used metrics. In a first step, a CCA was run, as a proxy for assemblage structure and as one possibility to reduce data volume and complexity; prior to this, abundance data were square-root-transformed to reduce heteroscedasticity. We chose the square-root transformation as another commonly applied transformation in ecology for count data, as it has a weaker effect on distribution shape than the logarithm transformations. Metrics complemented the assemblage description: species number and abundance diversity (Dominance, Simpson (1-Lambda), Shannon and Evenness) as well as relative portion of EPT taxa (% EPT taxa) and biological water quality (Average Score Per Taxon; ASPT). The CA1 and CA2 scores obtained from the Correspondence Analysis were further included as metrics (Table 1). These ten metrics were used for a RDA (see also electronic supplementary material S-1 for clarification of procedure, terms and data used).

Variance partitioning

To determine the gradient lengths of the dependent variables and to choose the appropriate method, detrended correspondence analyses (DCA) were conducted to obtain the gradient length of both the stream benthic macroinvertebrate assemblage data and the metrics. Variables not significantly correlated with the biotic data obtained by initial CCA were removed from the data set, and only the significant variables ($n = 19$) were taken for further analysis (Table 1). To analyse the effects of environmental variables on stream benthic macroinvertebrates, the three subsets of variables (according to scale) were subsequently analysed by the variance decomposition method outlined in Borcard et al. (1992). In the case of the assemblage data, the gradient length was >1.5 SD, and thus partial Canonical Correspondence Analysis (pCCA) was used, whereas in the case of the metrics, the linear method partial Redundancy Analysis (pRDA) was applied, since the gradient length was $\leq$1.5 SD. The three subsets were tested separately to determine the significance of individual variables using a Monte Carlo permutation test (with 999 unrestricted permutations).

The variation-partitioning technique allows for the variance in the explanatory data set to be partitioned into different variable components with the help of covariables, whose influence is partialled out. Initially, this technique was used to partition variation in ecological data sets into environmental and spatial components, but it has been extended by incorporating three sets of explanatory variables (e.g. Anderson and Cribble 1998), as in our study.

The total variance explained and the unique contributions of each subset and their joint effects were obtained in the following steps: (1) CCA/RDA were run with all three subsets as environmental variables and no covariables to obtain a measure of the total variance, (2) partial CCA/RDA were run with one of the three subsets as environmental variables and no covariables, (3) partial CCA/RDA were run with one of the three subsets as environmental variables constrained by the remaining two groups as covariables, and this was repeated for each subset. This procedure resulted in four runs of CCA/RDA for each subset combination or a total of 13 runs of CCA and RDA

Table 2 The procedure of variation partitioning of macroinvertebrate assemblages (by pCCA) and metrics (by pRDA; depending on prior gradient length analysis) explained by three sets of environmental variables, on catchment (C), reach (R) and local (L) scale

Run	Environmental variable	Covariable	λ_{CCA}	λ_{RDA}
1	CRL	None	2.592	0.681
2	C	RL	0.640	0.143
3	RL	None	1.925	0.538
4	RL	C	1.289	0.449
5	C	None	1.304	0.232
6	R	CL	0.392	0.160
7	CL	None	2.200	0.521
8	CL	R	1.751	0.408
9	R	None	0.842	0.273
10	L	CR	0.768	0.243
11	CR	None	1.825	0.438
12	CR	L	1.268	0.314
13	L	None	1.325	0.367

for the full set of analyses for each ecosystem (Table 2). With three subsets of environmental data, the total variation of benthic macroinvertebrate data was then partitioned into seven components including covariance terms. The variation explained by these subsets is subtracted from the total variation (1.0 in case of RDA) to obtain the unexplained variation.

Stepwise RDA

Stepwise CCA/RDA with forward selection were performed with the statistically significant environmental variables in each subset ($n = 19$) as independent variables and biotic variables as dependent variables to determine the best predictors (high R^2 values). In this procedure, selected variables are run as co-variables and subsequent variables (step 2 and on) need to explain a significant amount of the residual variance (tested by Monte Carlo permutation). This procedure allows to determine the most important environmental variables and their contribution, which are, for example, more practical for management support, rather than applying a principle component analysis (PCA), and relate this to diversity measures (which would acknowledge the presence of all included variables).

All ordinations (CA, DCA, pCCA and pRDA) were done using CANOCO for Windows Version 4.51 (ter Braak and Smilauer 2003). Metrics were calculated with PAST (2.17), % EPT taxa and ASPT were calculated with Asterics (Vers. 3.3; http://www.fliessgewaesserbewertung.de/en/download/berechnung/).

Results

Taxa richness ranged from 13 to 38 and abundance from 69 to 1939 individuals. The stream with the highest Shannon and Simpson diversity indices (3.2 and 0.94, respectively) also had the highest ASPT index and accordingly the lowest dominance score (0.058). The average proportion of mayflies, stoneflies and caddisflies (% EPT taxa) was 72 %, ranging from 15 to 98 % (see also electronic supplementary material S-2).

Variance decomposition using CCA showed that all independent variables combined (CRL) explained almost 70 % (eigenvalue ($\lambda_{CCA} = 2.592$) of the total variation in stream benthic macroinvertebrate assemblage structure (Fig. 1; Table 3). The smallest proportion of variance (2.2 %) was explained by the interaction between all three scale variables (CRL) (eigenvalue = 0.084; Fig. 2). The unique variance in benthic macroinvertebrate assemblage structure explained by catchment-, reach- or local-scale variables alone was higher than the variance explained by the combination of catchment, reach and local variables,

Table 3 Calculation of the explanatory power of each component in the variance-partitioning models

Variation explained by variables	Abbreviation	Calculation (no. of run, Table 2)	λ_{CCA}	λ_{RDA}
Catchment	C	2	0.640	0.143
Reach	R	6	0.392	0.160
Local	L	10	0.768	0.243
Catchment and reach	CR	12 − 6 − 2	0.236	0.011
Catchment and local	CL	8 − 2 − 10	0.343	0.022
Reach and local	RL	4 − 6 − 10	0.129	0.046
Catchment, reach and local	CRL	7 − 8 − (12 − 6 − 2) − (4 − 6 − 10)	0.084	0.056
Total explained	TotX	1	2.592	0.681
Unexplained	UX	TotV − TotX	1.205	0.319
Total variance	TotV		3.797	1.000

Figures in the calculation column refer to the runs in Table 2

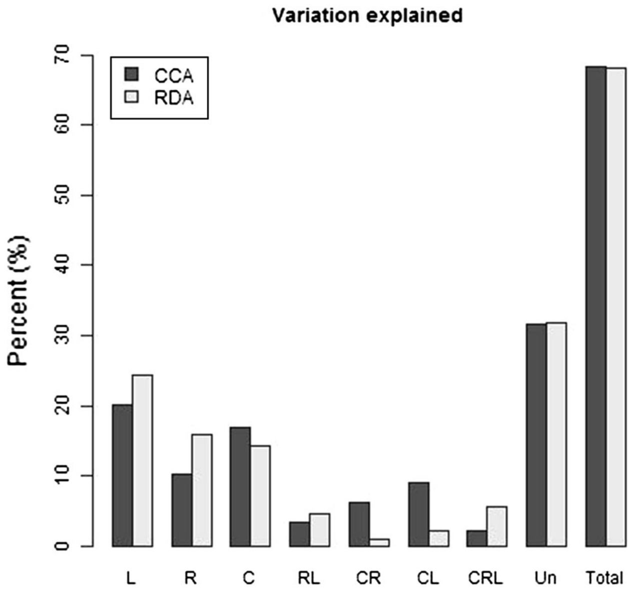

Fig. 2 Sources of variation (%) in macroinvertebrate assemblages obtained by CCA and metrics obtained by RDA. Column labels indicate *subsets*; *L* local scale, *R* reach scale, *C* catchment scale and their combinations, *Un* unexplained variation, *Total* total explained variation

respectively, or by the combination of reach and local variables (Fig. 2). The strongest interaction was found between catchment and local variables (CL) explaining 9 % (eigenvalue = 0.343) of the assemblage variance, whereas the combination of catchment and reach variables (CR) explained 6.2 % (eigenvalue = 0.236) and reach and local variables (RL) together explained only 3.4 % (eigenvalue = 0.129) (Fig. 1; Table 3).

Based on pCCA, the unique variance explained by local-level (L) variables (20.2 %, eigenvalue = 0.768) was substantially higher than that explained by reach-scale (R) variables (10.3 %, eigenvalue = 0.392) but also higher than that explained by catchment (C) variables (17 %, eigenvalue = 0.64).

A similar pattern was obtained when running RDA with the same independent variables and benthic community descriptors on diversity and composition. Here, variance decomposition showed that all independent variables combined explained almost 70 % (eigenvalue ($\lambda_{RDA} = 0.681$) of the total variation in all ten metrics (Fig. 2; Table 3). Here, all variables combined (CRL) explained 5.6 % (eigenvalue = 0.056) of the metrics' variance. However, the combination of catchment and reach (CR) variables explained least with only 1.1 % (eigenvalue = 0.011), followed by the combination of catchment and local (CL) variables (2.2 %, eigenvalue = 0.022), whereas the strongest interaction was found between reach and local (RL) variables explaining 4.6 % (eigenvalue = 0.046). The unique variance explained by reach-scale (R) variables (16 %) was higher than that explained by catchment-scale (C) variables (14.3 %). However, local variables were the strongest predictors of metrics.

Stepwise CCA and RDA of benthic community structure and metrics, respectively, as dependent variables and the "single" variables of catchment, reach and local variables revealed similar predictors accounting for the variance. However the variance in the assemblage structure was more predicted by land use/cover variables, whereas variance in metrics was more associated with hydomorphological variables (Table 1). In the case of community structure of stream invertebrates, the single most important predictor on the catchment scale was the amount of non-irrigated farmland, explaining 31 % of the total variance. The second variable selected was urban land use (20 %, i.e. the amount of residual variance explained after running the first variable selected "non-irrigated farmland in the catchment" as a covariable), followed by the amount of lacustrine rock in the catchment (18 %). On the reach scale, the first variable was the amount of terraces within the reach, followed by the amount of grass or bush land cover (both 19 %) and floodplain width explaining 16 % of the variance in assemblage structure. On the local scale, stepwise CCA revealed NH_3-N to be the most important predictor of assemblage structure variance explaining 25 %, followed by altitude explaining 20 % and non-source pollution (14 %). NH_3-N and non-source pollution were also predictors for variance in metrics explaining 14 and 6 %, respectively, whereas presence of the flow type "run" was the second best predictor explaining 8 %. On the reach scale, similar to the CCA, grass and bush land cover within the reach was the best predictor explaining 14 % of the macroinvertebrate metric variance, followed by the channel form of the stream (12 %) and the shape of the valley explaining 10 % (Table 1). No significant single variable was obtained when running RDA with catchment-scale variables and metrics.

Discussion

Environmental variables

Our aims were threefold, i.e. to determine most influential variables for benthic invertebrates, compare results related to communities as opposed to metrics and examine the role of spatial scales with particular relevance to management. A high proportion of overall variability (70 %) of the assemblage structure was explained, with catchment and local-scale variables being almost equally important. Although individual variables differed, the general pattern was similar for metric related variability (i.e. about 70 % explained variance) but a higher share was explained by local and reach-scale variables; most importantly altitude, which is in line with others (Tachamo Shah 2015; Wang et al. 2011). Compared to other studies, the amount of variation explained is reasonably high (Stendera and Johnson 2006), with other studies finding lower values (Beche and Statzner 2009; Pavlin et al. 2011; Wan et al. 2014), which is likely due to the method used. We used the full r^2 commonly used, which has recently been replaced with adjusted r^2 following Peres-Neto et al. (2006).

Unexplained variance might be due to the influence of other unrecorded variables, such as organic compounds (Malaj et al. 2014), agricultural runoff (Neumann and Dudgeon 2002) or frequent small-scale disturbances by local residents, e.g. disposing of waste or washing clothes (personal observations). While we chose to sample sites isolated from dam influences, we cannot rule out these effects as the region has a number of small hydropower plants (Wu et al. 2007).

Variables associated with anthropogenic disturbance explained most of the variation, such as the amount of farmland and urban areas in the catchment and pollution variables (NH_3-N, non-point source pollution at sites, reach-scale land use of grass/bushland and hydromorphological characteristics like a constrained channel). In the sampled catchments, non-point pollution is noticeable, e.g. as rubbish being deposited at the river banks, either deliberately or due to previous floods. Nitrogen compounds as the NH_3-N here have been often associated to stress responses with benthic invertebrates (Caschetto et al. 2014; Ye et al. 2014). Important reach-scale variables also reflected human land use at these scales, including the fraction of agricultural terraces (in this region always constructed for agricultural use), reduced channel width and proportion of grass/bushland. While for undisturbed sites (those pre-classified as "high quality class"), a high share of deciduous/mixed forest (51.3 %), no agriculture and few urban areas (1.3 %) are typical, these fractions are different in sites pre-classified as impacted, for example, with a lower share of 16 % deciduous/mixed forest, 18 % terraces and 50 % of classified as urban landuse (data not shown in detail). At the catchment scale, our study elaborates that variables reflecting anthropogenic stress are important for invertebrate occurrences, too (here, non-irrigated farmland and urban land use). Kail et al. (2012) found thresholds of 16.3 % urban landuse in german catchments limiting ecological quality at the sites, Collier et al. (2013) linked less than 20 % natural vegetation cover to changes in ecosystems functions and Death and Collier (2010) related the amount of catchment natural vegetation to water quality or biodiversity. However, in terms of biotic influences, land use can alter community composition and biodiversity patterns (Allan 2004; Harding et al. 1998) and functional indices (Clapcott et al. 2012; Collier et al. 2013), but despite altering composition, this may not influence other environmental linkages including those between productivity, disturbance and diversity (Tonkin and Death 2012).

Intermediate anthropogenic stress at the reach or local scales might have blurred our results as, by reducing the amount of allochthonous material through forest clearance and associated reduction in canopy cover, it is possible to firstly observe a positive response (Allan 2004). Reduced canopy cover in small streams can enhance periphyton growth due to less shade, providing enhanced food sources for benthic invertebrates, in line with the subsidy–stress relationship (Death and Zimmermann 2005) and modulate the role that flow disturbance and environmental productivity can have on stream invertebrate communities (Fuller et al. 2008).

Community structure vs metrics

For both assemblage structure and metrics, a high proportion of variability (70 %) was explained with differing variables in detail. These findings support the initial idea that assemblages as well as derived metrics reflect environmental gradients and potential stressors. CA1 and 2 scores, which were included as dependent variables, are good metrics to reflect assemblage compositions under stressed conditions, especially in the absence of sophisticated multimetric assessment systems. The metrics used here were selected for their simplicity and universal acceptance in representing community characteristics. However, various studies show that these simple metrics show only limited response potential (Jähnig et al. 2009; Tachamo Shah and Shah 2012; Yuan and Norton 2003) and hence might mask stress gradients unless these are considerably strong. It is likely the cause that gradients in catchment variables were not correctly depicted by the metrics. For instance, Hering et al. (2006) showed that fish metrics were more sensitive to large-scale pressures, while smaller bodied organisms like invertebrates or diatoms rather reflected local-scale stress. One option to overcome such bias is using multi-metric approaches (see a comprehensive list in Hering et al. 2006). Similarly, altitude emerged as an important variable for assemblages but not for metrics, likely reflecting potential turnover in community composition with altitude (Wang et al. 2012), but not necessarily changes in the metrics which we analysed. Our results further suggest that simple metrics might not be sensitive enough to pick up stressor signals and multimetric assessments should be tested in the future in China.

Scales

Local-scale variables tended to be more important than catchment-scale variables for the metric-based approach, which did not occur for the assemblage approach. It is important to realise that large-scale environmental variables shape small-scale variables, and in fact, local variables are simply more accurate reflections of broad-scale influences (i.e. they exhibit variation at finer spatial scales). This was nicely summarized in the concept of hierarchical filtering processes by Poff (1997) and holds true for species pools, too (Heino et al. 2003).

To avoid further influence from interacting variables, we ran stepwise regression analysis with each scale separately. No significant single variable was obtained when running RDA with catchment-scale variables and macroinvertebrate metrics. Our results are in line with other studies, which have found that most variance for benthic invertebrates was explained by various riparian width/length measures, e.g. land use in the subcatchment and riparian area, rather than the entire catchment (Death and Collier 2010; Kail and Hering 2009). At the local scale also substrate is often a key habitat element, but substrates were not found significant here, which might be due to the fact that pollution effects (NH_3-N and non-source pollution) did override substrate effects on invertebrates. The investigated region of central China is characterized by areas with considerable stress (urban development, agriculture) but has often rather little disturbed stretches in between that support self-purification of the rivers. The available quality of habitat seems particularly patchy; thus, the proximity to a nearby patch of high quality habitat might be an important driving variable (Kail and Hering 2009), and connectivity and dispersal might lead to “better appearance” of communities than expected. That is, source-sink dynamics or more specifically the mass effects metacommunity paradigm may be influencing the structure of these local communities, through species flooding into non-preferred sites from their reproductive habitats (Leibold et al. 2004). This can alter the signal of environmental controls, with a more important role of spatial variables in shaping local communities.

Overall similar patterns were found between overall community structure and univariate metrics, so metrics may be explaining patterns in community structure and hence may be used as a first surrogate to inform management. However, gradients have to be strong and unidirectional to pick up signal, especially in a rather small region, with lack of a strong regional gradient (Li et al. 2012; Tonkin et al. 2015).

Our results emphasise the need for spatially explicit regional studies in freshwater systems (Vinson and Hawkins 2003). Lotic systems are unique in terms of their hierarchical dendritic structuring and the clear spatial dependence of catchment and local-scale features. Many studies have found local features to strongly influence community structure (Astorga et al. 2011; Death and Joy 2004; Groll et al. 2015; Tonkin 2014), however, equally as many have highlighted the importance of large-scale influences (Heino 2009; Scott et al. 2011; Shah et al. 2014). Stream communities are thus shaped by variables operating at multiple spatial scales with great importance for water management and environmental health questions.

Acknowledgments The present study was conducted in the framework of the DFG-NSFC cooperation group “EcoChange” (GZ465). During initial fieldwork, SCJ was funded by the German Federal Ministry of Education and Research and German Academic Exchange Service (BMBF and DAAD) within the programme “Study and research for sustainability: Yangtze region” with a research fellowship and by the Young scientist research award 2008 of the University of Duisburg-Essen. SCJ also acknowledges financial support by the research funding programme “LOEWE—Landes-Offensive zur Entwicklung Wissenschaftlich-ökonomischer Exzellenz” of Hesse’s Ministry of Higher Education, Research and the Arts and by the BMBF for funding “GLANCE” (Global change effects in river ecosystems; 01LN1320A).

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