Temporal and spatial dynamics in aquatic macroinvertebrate communities along a small urban stream

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Abstract

Urbanization is a current and increasing threat to biodiversity. The effects of urbanization on the functional and taxonomic composition of macroinvertebrate assemblages were investigated in two seasons along a small urban stream. Species composition was determined by the rate of urbanization; however, response of species richness responds to it could not be evinced. Relative abundance of the sensitive macroinvertebrate groups (EPT%) was negatively related to urbanization. Almost all feeding groups showed a sharp decline in the number of specimens along the increase of urbanization. The study support the view that urbanization has a negative effect on the biological quality of a stream, yet, this obvious impact can be overridden by different conditions such as modification in streambed morphology. The altered conditions and new circumstances in urban environments lead to the creation of novel ecosystems, inhabited by macroinvertebrate communities with species richness approaching that of low impacted sites. However, species composition of these communities could be basically different from that of the natural ones.

Keywords: urbanization, species richness, functional feeding group, city stream, EQR, biological quality

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Introduction

Urbanization is a global phenomenon and its rate is expected to increase in the future (Chen et al. 2014). It brings important benefits in terms of economic, cultural and social development, but the ecosystems face numerous threats as a consequence of the rapid urbanization process.

The number of ecological studies dealing with this process has increased during the recent years. Urban areas are known to have different climates, soils, hydrology, chemical and physical environments than non-urban areas (Pickett et al. 2001). These changes of the environmental characteristics coincide with decreasing trends in diversity of the major groups such as plants, mammals, algae and birds (McKinney 2008). These experiences helped in the development of a conceptual model by which mechanisms of the major urban impacts on stream ecosystems (also known as urban syndrome) can be estimated (Meyer et al. 2005; Walsh et al. 2005). Urbanization also leads to reduced ecosystem functioning and biological diversity in the case of the aquatic environment (Ramírez et al. 2009; Wallace et al. 2013). Urbanization radically alters the characteristics of a watershed by modifying the physical and chemical characteristics of water bodies (Wenger et al. 2009; Hepp et al. 2010). However, catchment scale studies suggest that these relationships might be overridden by specific effects such as fine scale land use variation, as well as variation in habitat structure and level of imperviousness.

Stream macroinvertebrates have relatively short generation times plus various trophic positions and could react quickly to changes in their environment. Even small negative changes in their local habitat conditions can eliminate or reduce many macroinvertebrate taxa (Collier and Clements 2011). Therefore, characteristics of the fine scale habitat play a key role in shaping the patterns of species composition and diversity (Heino et al. 2004). Besides the spatial aspects, species composition and assemblage structure, of stream macroinvertebrates have characteristic temporal dynamics (Luo et al. 2017), that allow comparisons of yearly variations in the communities on an annual time scale (seasonal changes). The same year temporal variations and life-cycle seasonalities affect stream macroinvertebrate community structure and the values of metrics used to describe the ecological state or health of these communities (Johnson et al. 2012). While estimation of the temporal uncertainties of these metrics would be necessary, only a few studies deal with the temporal aspects of macroinvertebrate community structuring along urban streams (Luo et al. 2017; Martins et al. 2017). The spatial occurrence and temporal persistence of species are closely linked to their locomotion type, dispersal capacity and strategy (Clobert et al. 2009). When habitat quality varies spatially along a stream, these properties can shape the small scale occurrence of species. Despite the accumulated

knowledge on how the sensitive higher taxa (e.g. Ephemeroptera, Plecoptera, Trichoptera) react to changes in habitat quality, the response curves at family or species levels are scarcely studied.

The spatial arrangement of contemporary cities reflects the needs and demands of modern people (Seto et al. 2010; Ramalho and Hobbs 2012). Besides the traditional urban areas such as downtowns, suburbs, and industrial zones, more and more open space and green areas are developed within the cities to enhance the quality of living. Combinations of these areas locally modify the fine scale land use along the urban streams. City environments are oversimplified by former conceptual approaches which assume that urbanization and its induced environmental changes decrease in a linear gradient from the core to the edges (Ramalho and Hobbs 2012). However, it is reasonable to believe, that because of the fine scale differences of the land use types within urban areas, community patterns do not vary along the flow path (as a gradient) of streams, but rather reflect the impacts of fine scale local environment.

The goal of this study was to investigate the effects of urbanization on the functional and taxonomic composition of macroinvertebrate assemblages along a small urban stream. The main questions were:

i) How does the urbanization affect the species richness, diversity of macroinvertebrates and biological quality along a small urban stream? Is there any seasonal difference in the community compositions? What are the key variables effecting the community patterns?

ii) How do the taxonomical and functional groups respond to small scale differences of land use within the urban area?

Material and Method

Study area

The study was conducted in the Tisza River Basin in North Hungary, near the city Eger (Fig. 1). The climate of the studied area is temperate humid, thus the mean annual temperature is 10 °C, and the typical yearly precipitation amount is 550 mm. Eger-patak is a perennial, colline stream with medium sized upper watershed (299 km^2). The stream originates from the Bükk Mountains and is flowing through the city of Eger and several smaller villages (e.g. Szarvaskő, Almár, Andornaktálya, Nagytálya, Maklár) before emptying into the Tisza River. Eger-patak is characterised by near natural conditions, however sections of the stream belonging to the city are heavily modified.

Macroinvertebrate sampling

Aquatic macroinvertebrate samples were collected at 15 sampling sites between Almár and Nagytálya villages along an urban gradient (Fig. 1). The sampling was performed on the 10th May and the 30th August in 2014. During the samplings, the multi-habitat sampling method and "kick and sweep" technique was applied using a hand net with 500 µm mesh size (AQEM Consortium 2002). Collected samples were preserved in the field and sorted later in the laboratory. Individuals of macroinvertebrates from 10 taxonomic groups (Gastropoda, Bivalvia, Hirudinea, Crustacea, Ephemeroptera, Odonata, Heteroptera, Coleoptera, Trichoptera and Diptera excluding Chironomidae) were identified under a stereomicroscope to the possible lowest taxonomic level by experts using relevant identification keys (Askew 1988; Bauernfeind and Humpesch 2001; Cham 2009; Csabai 2000; Csabai et al. 2002; Eiseler 2005; Gerken and Sternberg 1999; Glöer and Meier-Brook 2003; Haybach 1999; Klonowska-Olejnik 2004; Kontschán et al. 2002; Nesemann 1997; Richnovszky and Pintér 1979; Savage 1989; Waringer and Graf 1997). Identified macroinvertebrates were preserved in 70% ethyl alcohol and stored in the laboratory.

Environment variables

The measurement of environment variables was performed before the samplings of biological elements. The following variables at each of the sampling points were measured in both seasons, respectively, with Hanna HI 700 photometer tools: water temperature; conductivity; pH; water chemistry (dissolved O_2; NO_2^-; NH_4^+ and PO_4^{3-}). The dominant substrate types and percentage of substratum cover were assessed visually following the AQEM protocol (AQEM Consortium 2002) in both seasons. Furthermore, the observed stream-bed modifications caused by human intervention were evaluated according to a three-point scale. Measured values are shown in ESM Table 1.

Urban Index

To estimate the urbanization rate in the studied area, the "UrbanizationScore" software was used (Seress et al. 2014). The "UrbanizationScore" generates a semi-automated score of habitat urbanization. The software quantifies the urbanization of sampling sites based on 4 landscape-cover variables: abundance of buildings, vegetation, forest and paved roads. The software uses available satellite images from the "GoogleMaps". Around the sampling sites, a 1×1 km area was divided into 10×10 cells and scored according to the abundance of based landscape-cover

variables Furthermore, a 2×2 km area and a 300×300 m area were analysed around each sampling site. These scores of landscape-cover variables for each sampling sites were combined by Principal Component Analysis (PCA) into an 'urbanization score' for each site.

Data analysis

Prior to the analyses, water chemistry data and the urban index values were log(x+1) transformed. Percentage data of the stream bed morphology were arcsine transformed, while Hellinger transformation was used on the species data table (Legendre and Gallagher 2001).

Redundancy Analysis (RDA) was performed on the biological and environmental data to determine the relative importance of water chemistry, streambed morphology variables and urbanization in explaining the variability of the macroinvertebrate species composition. The proportions of built riverbed are categorical variables, and during analyses they were transformed and used as factors.

The seasonal difference between the environmental data matrix was analysed by ANOSIM method. To evaluate the taxonomical changes in species composition along the urban gradient, number of species, number of specimens and Shannon diversity was calculated at each sampling site. EPT %, EPTCOB% and the alteration in feeding groups was also calculated along the urban gradient to assess the functional changes.

To assess the quality of the stream a type specific Hungarian Multimetric Index (HMMI) value (expressed as ecological quality ratio (EQR) was calculated for each sampling site. The HMMI_sc consists of three metrics: Ephemeroptera + Plecoptera +Trichoptera total taxon number (EPT), ASPT (average taxon score) and from the percentage of crenal to hyporhithral (spring to greyling region) zonation preference taxa (Eu-hr% of scored taxa).

$$HMMI_sc = \frac{EPT_{EQR} + ASPT_{EQR} + EuHr_{EQR}}{3}$$

Species response curves were calculated to visualize the response of the taxonomic and functional groups to the urbanization. All analyses were performed with CANOCO 5 (ter Braak and Šmilauer 2012) and PAST software (Hammer et al. 2001).

Results

General remarks

A total of 48.881 macroinvertebrate individuals belonging to 82 taxa (from which 74 were identified to the species-level) and 10 higher taxa (Bivalvia – 1; Gastropoda – 15; Hirudinea – 3; Crustacea – 5; Ephemeroptera – 9; Odonata – 8; Heteroptera – 2; Coleoptera – 12; Trichoptera – 20; Diptera – 7;) were collected in the survey. Three species (*Gammarus roeselii, Potamopyrgus antipodarum and Hydropsyche sp*.) were common (100% of occurrence), by contrast, there were 29 species occurring exclusively at one sampling site (for detailed data see Csercsa et al. 2015).

Variation in environmental conditions

Statistical analyses revealed spatial differences in the streambed morphology and water chemistry data along the stream (Fig. 2). Measured values of the environmental factors are shown in Table 1. A negative relationship can be found between the environmental variables (water chemistry and streambed morhology together) and the urbanization scores ($R^2 = 0.293$) (Fig. 2). There was a significant difference among the chemical composition of the spring and autumn samples (Anosim based on eucledian distance with 9999 permutation $p<0.05$). The diferrentation was mainly caused by the difference in NH_4^+N concentration values between the two seasons.

Species richness, abundance, diversity, EQR

Macroinvertebrate richness ranged from 15 to 36 taxa among the samples, the highest number of species was found in EGR002 (36) in spring, and EGR004 (34) in autumn (Fig. 3A). The number of individuals ranged from 58 to 20873, the highest number of individuals were observed in EGR015 in both seasons (Fig. 3B). The calculated values of the α-diversity (Shannon) indices ranged from 0.5 to 2.4. In both seasons, the greatest diversity was found at the sampling site EGR011, while the lowest at the site EGR014 (Fig. 3C). Assessment of the ecological status derived from the macroinvertebrate community indicated high variation among the sampling sites from poor to high. The lowest EQRs were calculated for the EGR011 and EGR012 sites, while the highest scores for EGR001, EGR002 and EGR005 in both seasons (Fig. 3D). Eight sites had the same ecological quality status in both seasons. In six sites, the spring assessments were better than in the autumn, while the quality was increased with one class from poor to moderate in the autumn in the case of the EGR011 site (Fig. 4B).

Relationship between macroinvertebrate communities and environmental variables (RDA)

The explanatory variables accounted for 100.0% of the total variation in both cases. Temperature and O_2 had significant effects in both seasons; the explained variance was 54.3% in spring and 53 % in autmn. The rain flow outlets had also a significant effect on the variance of the communities (Fig. 3). Among the variables mod1, urban500, pH and conductivity showed correlation with the 1. axis (ESM Table 2, Fig. 5).

Effects of urbanization

Positive relationship was found between the urbanization and Shannon diversity in both seasons (spring $R^2 = 0.48$; autumn $R^2 = 0.32$). Abundance of taxa correlated negatively to the urbanization independently of seasons (spring $R^2 = 0.32$, autumn $R^2 = 0.53$). Relative abundance of sensitive macroinvertebrate groups (EPT % spring $R^2 = 0.2327$, autumn $R^2 = 0.5348$ and EPTCOB% spring $R^2 = 0.3092$, autumn $R^2 = 0.4394$) was faintly and negatively related to urbanization. No relationship was found between urbanization and taxa richness.

The most sensitive taxonomic groups to urbanization were Gastropoda and Crustacea, while Trichoptera, Ephemeroptera and Odonata showed less sensitivity. Coleopterans exhibited a bell shape response, as they were found in high numbers at the sites with medium urbanization (Fig. 6A). Relative abundances of other taxonomic groups were too small to evaluate the response curves (less than 10 individuals, e.g. Heteroptera, Bivalvia). The Families also showed different responses to urbanization (Fig. 6A-D). The response curves of Elmidae (Coleoptera) and Erpobdeliidae (Hirudinea) had bell shape profiles (Fig. 6A). The higher the urbanization scores, the lower was the number of individuals in the families of Hydropsychidae, Gomphidae, Hydrobiidae, Lymnaeidae, Ephemeridae, Heptageniidae, Chironomidae and Gammaridae (Fig. 6B). The number of individuals belonging to Simuliidae and Baetidae showed positive response to urbanization.

All the feeding groups showed similar negative responses to the increasing urbanization without seasonal differences. Almost all feeding groups showed a sharp decline in number of specimens with the increase in urbanization (Fig. 6C), but a short delay was observed in the response of gatherers (Fig. 6B). A considerable decline in the number of gatherers could be observed when the urbanization rate was higher than 0.5.

Discussion

As the rate of urbanization is growing, it becomes increasingly necessary to provide a deeper understanding of the temporal and spatial effects of urbanization on stream ecosystems.

Therefore, the effects of urbanization on the taxonomic and functional composition and characteristic of macroinvertebrate communities were investigated along a small urban stream.

Apparently, species composition was determined by the rate of urbanization; however, it is unclear if species richness responds to urbanization. It possibly means that there is a high turnover rate in species composition along the urbanization gradient. This species turnover occurs without considerable changes in functional composition. Hence the entire length of the stream belongs to the same type of water, variation in species composition might not be the consequence of the natural longitudinal changes (Vannote et al. 1980), but might be a response to urbanization. This was confirmed by the negative linear relationship ($R^2 = 0.293$) between the environmental variables and urbanization (see Fig. 2). This is in contradiction with other studies that reported a negative relationship between taxa richness and urbanization (see Urban Stream Syndrome, Meyer et al. 2005; Brown et al. 2009; Cuffney et al. 2010; Utz et al. 2016). However, the abundance increases found were consistent with the Urban Stream syndrome, as an increased density of buildings and roads in the proximity of aquatic habitats (higher urbanization) negatively influenced the abundance. Increased amounts of green surroundings (lower urbanization) showed a positive influence (Meyer et al. 2005; Walsh et al. 2005).

The negative relationship between sensitive species (EPT%) and rate of urbanization was consistent with the expectation, that urbanization typically results in the loss of these sensitive taxonomic groups (Morse et al. 2003; Roy et al. 2003; Cuffney et al. 2010; Collier and Clements 2011; Hepp et al. 2013; Narangarvuu et al. 2015; Luo et al. 2017; Martins et al. 2017). Not only the EPT%, but the EPTCOB% was also decreased along the urban gradient. As described previously (e.g. Walsh et al. 2005; Bazinet et al. 2010), the communities of highly degraded streams within urban catchments are mainly dominated by Oligochaetes and Chironomids. Contrarily, members of six higher taxa were found (Bivalvia, Crustacea, Gastropoda, Odonata, Diptera and Trichoptera, both in spring and autumn) that belonged to five types of feeding groups in the sampling sites with lowest values of EQR (EGR013 in spring, EGR012 in autumn). Besides this, 4-7 members of higher taxa were found in the most urbanized sites (EGR007-EGR010, Urban index > 90%) that belonged to various types of feeding groups. All the taxonomical and functional groups decreased with increasing urbanization, however, a diverse array of species could be found even in the most urbanized and the most degraded sites. Various responses were observed to urbanization in case of higher taxa (HT) and functional feeding groups (FFG). Most of the groups (either HT or FFG) show a monotonic decline in abundance with increasing urbanization (Fig. 6B), while others show steep decline right after a threshold (Fig. 6C). Some taxa were presented by such a small number of individuals, that the

urbanization could have no considerable effect on their abundance (Fig. 6D). The shapes of response curves are conceptually compatible with the models of Walsh et al. (2005). However, Chironomidae, Elmidae, Erpobdellidae and Gammaridae were mainly associated with medium-impacted sites, and they are typically low in abundance at the sites with low or high urbanization (Fig. 6A). These results are in contradiction with those of Martins et al. (2017), who demonstrated that the least-impacted streams were associated with Elmidae and some members of Chironomidae. A few members of Odonata (*Gomphus vulgatissimus, Orthetrum cancellatum*), Ephemeroptera (*Baetis vernus*), Diptera (*Simulium* sp.) and Trichoptera (*Hydropsyche bulbifera*) are found in high number of specimens in the most impacted sites, thus the response curves of these show a little increase in the end of the urban gradient (Fig. 6B1, D1).

As a consequence of the negative relationship between relative abundance and the urbanization and the change of species composition along the urban gradient, the EQRs are negatively related to urbanization as well. Consequently, a linear decline of biological conditions with increasing urbanization can be seen in both seasons. Seasonal variation in relative abundance and species composition might be taken into consideration when calculating EQR and the processing of biological quality assessment (Johnson et al. 2012). The changes between seasons in EQR ranged from 0 to 0.22, and it could result in one class change in biological quality. Despite the expected seasonal variation, considerable temporal differences were not observed in EQR and biological quality. The reason for this is found in the minimal environmental variation, and one needs to assume that urbanization is the key factor in structuring community patterns. However, 0.1 in spring and 0.2 in autumn was the difference in EQR between the sampling sites with the lowest and highest urbanization scores. Thus, good biological quality can be detected independently of the urbanization rate. The biological qualities were high in three out of five cases when the urbanization rate was higher than 90%, which was the opposite of the response predicted by the Urban Stream Syndrome (Meyer et al. 2005; Walsh et al. 2005) and other supporting articles (Collier and Clements 2011; Wallace et al. 2013). This apparent contradiction can be explained firstly by altered stream flow regimes and stream bed morphology (ESM Table 2). The altered environment is characterized by greater flow velocity and a unique stream bed that might provide proper habitat for rheofil species. These changes are responsible also for the increased abundance of Simuliidae and Baetidae species. These results are in accordance with previous findings; that altered conditions and new circumstances in urban environments lead to the creation of novel ecosystems for almost completely new macroinvertebrate communities (Perring et al. 2013).

A clear positive relationship (R^2 = 0.481) was found between urbanization and Shannon diversity,. but a negative relation was registered between evenness and urbanization simultaneously. Nevertheless, taxa richness was not related to urbanization. The monotonic increase in evenness from less to high urbanized habitats was more pronounced than the increase in richness from high to less urbanized habitats, as indicated by their respective magnitudes of effect. Thus, diversity showed an inverse spatial trend compared to evenness. The reason lies in the species composition, as a relatively high number of species were dominated by a single taxon (Gammaridae) in abundance. However, the EQR might be high because the abundance based metric is less pronounced in the Hungarian Multimetric index.

This study attracts further attention to the importance of urban studies. Streams, as low-lying points in the landscape, are strongly and mainly negatively influenced by the urbanization of surrounding area. This pattern, however, can be overridden by different conditions such as modification in streambed morphology and environmental variables that can give the opportunity for new taxa to form a different or new species composition, but the same species richness. Taxonomical and functional composition can track small-scale environmental variation within small spatial scale. Deeper investigation of the topic is needed to fully reveal the underlying mechanisms of the temporal and spatial dynamics on macroinvertebrate communities in small spatial scales.

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Captions for figures

Figure 1. Map of the study area and location of the sampling site in Hungary

Figure 2. Negative relationship between the environmental variables (waterchemistry and streambed morhology together in both seasons) and the urbanization scores. (Light grey circles refer to the spring data, while dark grey circles refer to autumn data.)

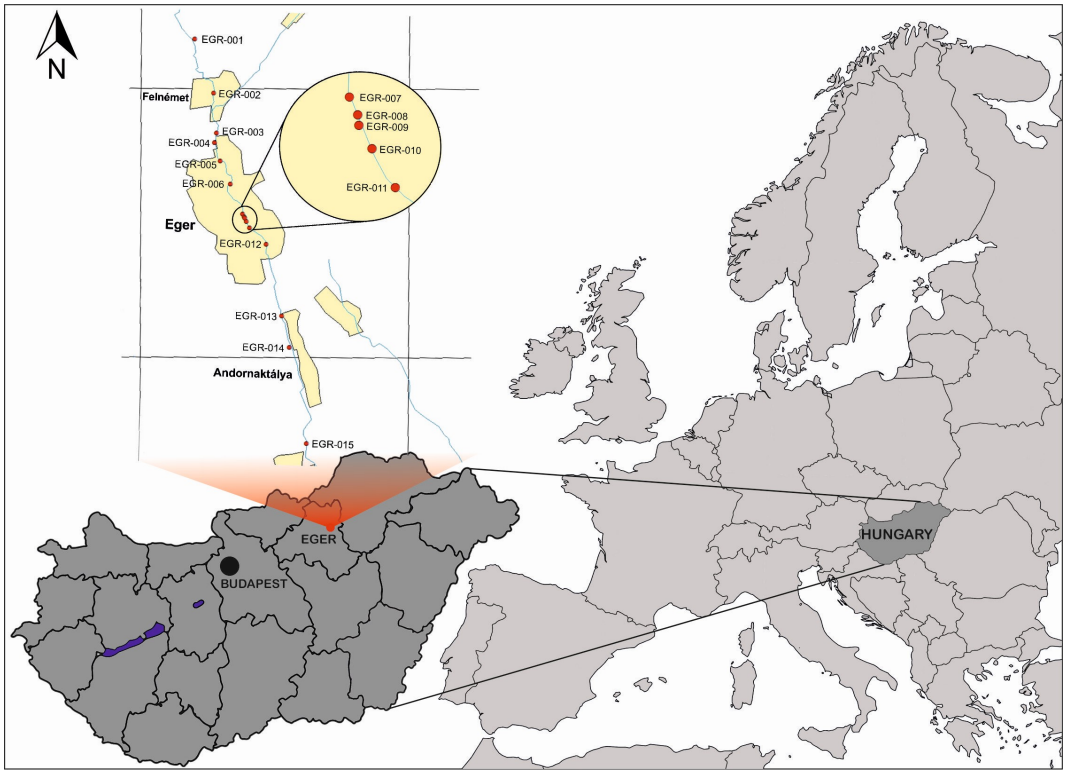
Figure 3. Spatial and temporal variation in relative abundance (log10 scale, A), number of species (B), Shannon diversity (C) and EQR (D) along the urban stream.(Light grey coloumns refer to the spring data, while dark gery coloumns refer to autumn data.)

Figure 4. Schematic depiction of **s**patial arrangment of the studied city and the relative position of the sampling sites (A), variation in EQR of the sampling sites (B, colours refers to the ecological status, heights of the coloumns refers to the exact value of the EQR), and variation in the urbanization along the city (C, higher the coloumn, the higher is the urbanization).

Figure 5. RDA plot of environemntal variables (water chemistry, stream bed morhology, stream bed modification and Urban Indices) and species data in spring (A) and autumn (B). Abbreviations as follows: UI: Urban index; Built rb: built riverbed.

Figure 6: Schematic curves of taxonomical and functional groups in response to increasing urbanization rate. A: groups with typically low in abundance at the sites with low or high urbanization index; B: groups with monotonic decline in abundance with increasing urbanization; C: groups that show steep decline right after a given threshold, D: groups in which urbanization could have no considerable effect on their abundance. C1 and D1 yield an alternative way to those taxa within groups which are shown a little increase in response curves in the end of the urban gradient.

Fig 1.


N
EGR-001
Felnémet
EGR-002
EGR-003
EGR-004
EGR-005
EGR-006
Eger
EGR-012
EGR-007
EGR-008
EGR-009
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EGR-014
Andornaktálya
EGR-015
EGER
BUDAPEST
HUNGARY

Fig 2.

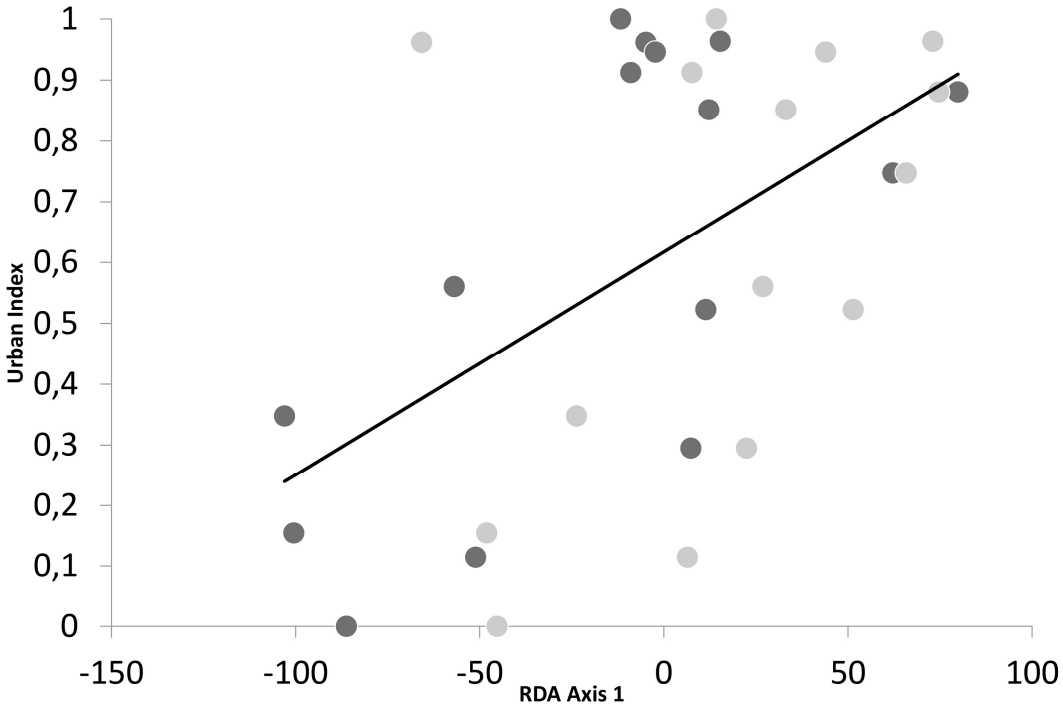
Urban Index
1
0,9
0,8
0,7
0,6
0,5
0,4
0,3
0,2
0,1
0
-150
-100
-50
0
50
100
RDA Axis 1

Fig 3.

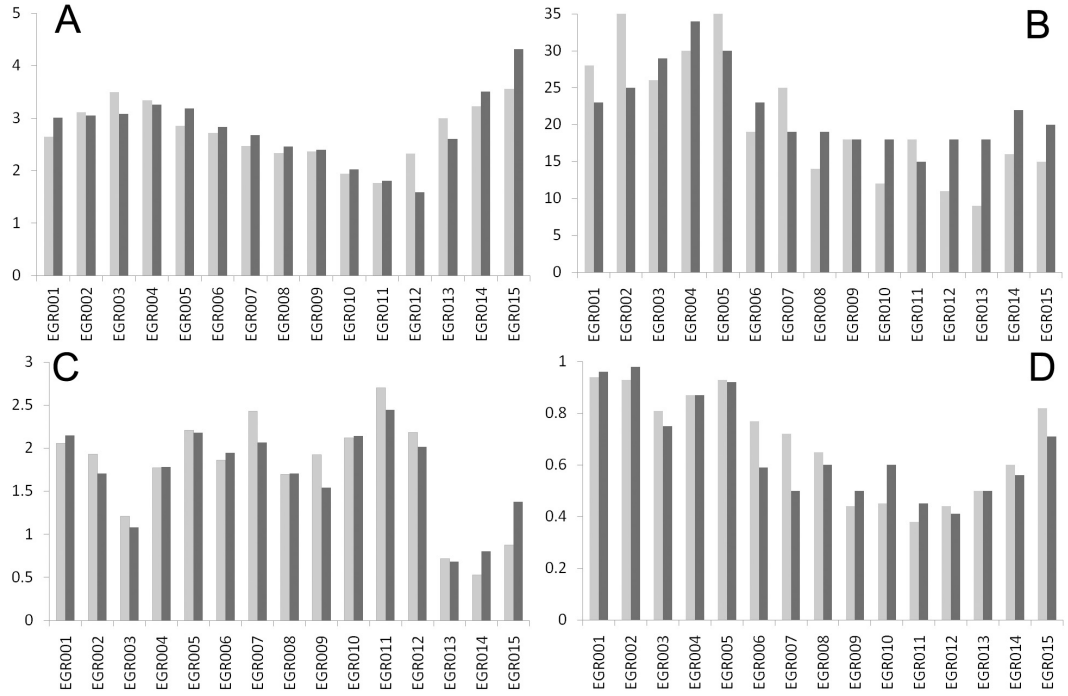
A
B
C
D
EGR001
EGR002
EGR003
EGR004
EGR005
EGR006
EGR007
EGR008
EGR009
EGR010
EGR011
EGR012
EGR013
EGR014
EGR015

Fig 4.

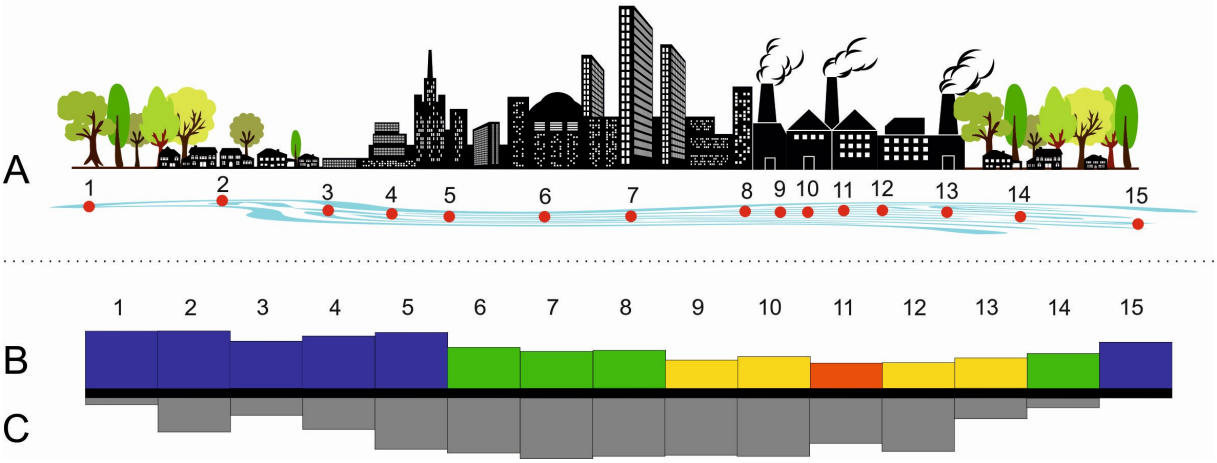
A
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
B
C

Fig 5.

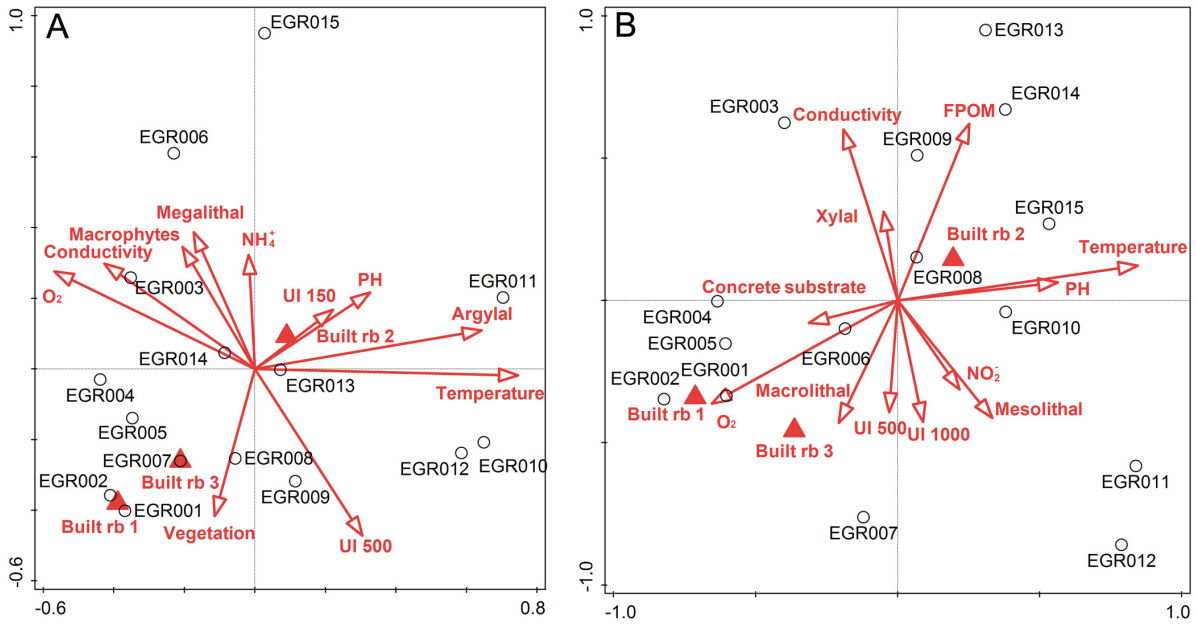
A
EGR015
EGR006
Megalithal
Macrophytes
Conductivity
NH4+
EGR003
O2
UI 150
PH
EGR011
Argylal
Built rb 2
EGR014
EGR013
Temperature
EGR004
EGR005
EGR007
EGR008
EGR012
EGR010
EGR002
Built rb 3
EGR009
EGR001
Built rb 1
Vegetation
UI 500
1.0
-0.6
-0.6
0.8
B
EGR013
EGR014
EGR003
Conductivity
FPOM
EGR009
Xylal
EGR015
Built rb 2
Temperature
Concrete substrate
EGR008
PH
EGR004
EGR010
EGR005
EGR006
EGR002
EGR001
NO2-
Macrolithal
Built rb 1
O2
UI 500
UI 1000
Mesolithal
Built rb 3
EGR011
EGR007
EGR012
1.0
-1.0
-1.0
1.0

Fig 6.

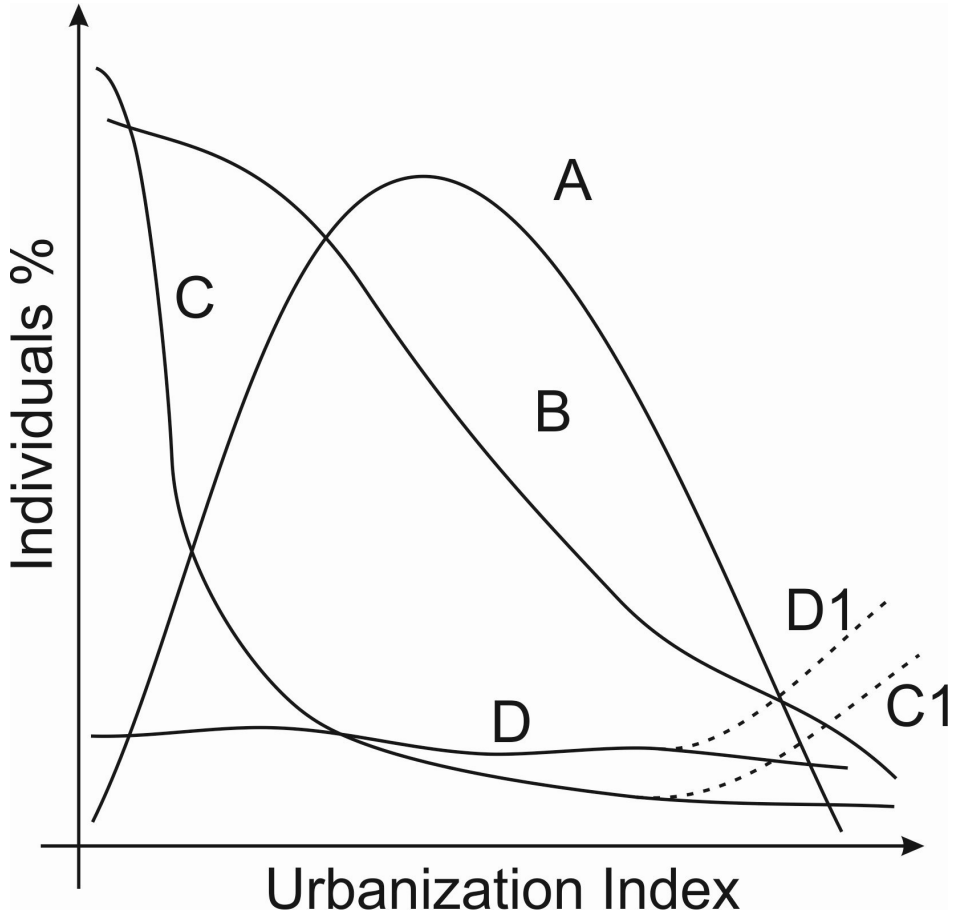
Individuals %
A
C
B
D1
C1
D
Urbanization Index

Table 1. Relationships between all pairs of metrics and variables (Abbreviation in columns refers to the rows).

	Taxa_S	**Ind.**	**Shan.**	**Even.**	**Ber.P.**	**EQR**	**BMWP**	**Div.**	**EPT%**	**EPT COB%**
Taxa_S	x	0.21	0.001	0.23	0.40	0.59	0.80	0.001	0.23	0.36
Individuals		x	0.31	0.40	0.12	0.13	0.20	0.31	0.15	0.50
Shannon_H			x	0.67	0.85	0.01	0.006	1	0.001	0.04
Evenness_e^H/S				x	0.45	0.11	0.19	0.67	0.08	0.27
Berger-Parker					x	0.11	0.14	0.85	0.08	0.001
EQR						x	0.53	0.01	0.76	0.57
BMWP							x	0.006	0.17	0.36
Diversity								x	0.001	0.04
EPT%									x	0.59
EPTCOB%										x

Electronic Supporting Material

Table 2. Measured values of the environmental variables in both seasons. The measurement of environment variables was performed before the samplings of biological elements. The following variables at each sampling points were measured in both seasons, respectively, with Hanna HI 700 photometer tools. The dominant substrate types and percentage of substratum cover were assessed visually following the AQEM protocol in both seasons. Furthermore, the observed stream-bed modifications caused by human intervention were evaluated according to a three point scale.

Spring	Code	EGR001	EGR002	EGR003	EGR004	EGR005	EGR006	EGR007	EGR008	EGR009	EGR010	EGR011	EGR012	EGR013	EGR014	EGR015
Urban environment	Extent of Built riverbed	1	1	2	2	3	2	3	2	2	2	2	2	2	2	2
	Urban index 150	0.00	0.15	0.18	0.28	0.16	1.00	0.21	0.07	0.04	0.10	0.58	0.73	0.00	0.13	0.01
	Urban index 500	0.11	0.56	0.29	0.52	0.85	0.91	1.00	0.96	0.95	0.96	0.75	0.88	0.35	0.15	0.00
	Urban index 1000	0.00	0.44	0.16	0.25	0.59	0.73	1.00	0.96	0.97	0.90	0.64	0.76	0.12	0.07	0.00
Water chemistry	pH	7.92	8.18	8.08	8.06	8.16	8.06	8.11	8.11	8.07	8.16	8.11	8.09	8.11	8.1	8.16
	Temperature (°C) 1	18.4	17.2	17.9	18.42	19.4	21.5	21.3	23.9	26.6	23.4	24.1	25.8	24.8	23.6	21.8
	O_2 (mg/l)	8.61	8.59	9.56	9.77	10.08	9.79	8.69	8.36	7.53	8.24	7.96	7.64	7.33	7.29	8.39
	Conductivity (µS)	911	891	895	866	890	913	904	985	874	844	852	847	942	967	964
	NO_2^- (µg/l)	44	52	34	50	45	66	24	40	14	18	30	44	69	73	41
	NH_4^+ (mg/l)	1.3	0.2	26.4	2.7		5.9	0.6	25.5	0.6	1	1.5	10.3	5.9	21.2	3.2
	P (µg/l)	200	200	200	200	200	200	200	200	200	200	200	200	200	200	200
Streambed morhology	Megalithal (>40 cm)	0	0	0	0	10	0	0	0	0	0	0	0	0	0	10
	Makrolithal (20-40 cm)	5	0	0	0	0	10	30	0	0	0	0	0	0	0	0
	Mezolithal (6-20 cm)	15	10	10	10	0	20	20	40	40	40	35	30	0	0	20
	Mikrolithal (2-6 cm)	20	10	20	35	0	20	5	15	15	15	15	30	20	40	30
	Akal (0,2-2 cm)	60	60	50	45	20	20	45	45	45	45	40	30	80	50	40
	Argylal (<6 µm)	0	0	0	10	0	10	0	0	0	0	10	10	0	10	0
	Artificial substrate	0	20	20	0	0	20	0	0	0	0	0	0	0	0	0
	Artificial substrate: concrete streambed	0	0	0	0	70	0	0	0	0	0	0	0	0	0	0
	Submerge macrophytes and filamentous algae	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0
	Slug and emergent macrophytes	0	10	30	35	0	10	0	10	10	0	0	40	10	10	30
	Vegetation	10	10	0	10	0	10	10	10	10	10	10	0	15	15	0
	Xylal	5	0	0	0	0	0	0	0	0	0	0	0	5	5	0