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Diatoms are better indicators of urban stream conditions: A case study in Beijing, China

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ABSTRACT

Urbanization dramatically affects hydrology, water quality and aquatic ecosystem composition. Here we characterized changes in diatom assemblages along an urban-to-rural gradient to assess impacts of urbanization on stream conditions in Beijing, China. Diatoms, water chemistry, and physical variables were measured at 22 urban (6 in upstream and 16 in downstream) and 7 rural reference stream sites during July and August of 2013. One-way ANOVA showed that water physical and chemical variables were significantly different ($p < 0.05$) between urban downstream and both reference and urban upstream sites, but not between reference and urban upstream sites ($p > 0.05$). Similarly, structural metrics, including species richness (S), Shannon diversity (H′), species evenness (J′) and Simpson diversity (D′), were significantly different ($p < 0.05$) between urban downstream and both reference and urban upstream sites, but not ($p > 0.05$) between reference and urban upstream sites. However, diatom assemblages were very different among all sites. *Achnanthidium minutissima* was a consistent dominant species in reference sites; *Staurosira construens* var. *venter* and *Pseudostaurosira brevistriata* were the dominant species in urban upstream sites; and *Nitzschia palea* was the dominant species in urban downstream sites. Clustering analyses based on the relative abundance of diatom species, showed all the samples fit into three groups: reference sites, urban upstream sites, and urban downstream sites. Canonical correspondence analysis (CCA) and Monte Carlo permutation tests showed that concentration of K^+, EC, TN, Cl^- and pH were positively correlated with relative abundance of dominant diatom species in urban downstream samples; WT and F^- were correlated with reference and urban stream diatom composition. Our results demonstrate that the composition of diatom species was more sensitive to urbanization than the water physical and chemical parameters, and that diatom assemblage structure metrics more accurately assessed water quality. Some species, such as *Amphora pediculus* and *Cocconeis placentula* were among the dominant species in low nutrients stream sites; however, they were considered to be high nutrient indicators in some streams in USA. We suggest using caution in applying indicator indices based on species composition from other regions. It is necessary to build a complete set of diatom species data and their co-ordinate environment data for specific regions.

1. Introduction

China began a period of rapid urbanization with the reform and opening in the 1980s. The proportion of urban population increased from 17.9% of the population in 1978 to 52.6% in 2012 while the urban built-up area grew by 78.5% (Bai et al., 2014). The rapid urbanization has caused a series of complex environmental problems, for example, the alteration of hydrosystems (Walsh et al., 2005; Hopkins et al., 2015), loosening of biogeochemical cycles (Pickett et al., 2011), warming of urban meso-climate (Zhou et al., 2004; Li et al., 2012a,b; Zhou et al., 2014), and impairment of biodiversity (Angold et al., 2006; Grimm et al., 2008).

Urbanization affects hydrosystems dramatically by altering hydrology, channel morphology, water quality and aquatic biota (Dunne and Leopold, 1978; Newall and Walsh, 2005). In urban areas, water quality degradation may be associated with the

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increased areas of impervious surfaces (Ren et al., 2003), with piped stormwater drainage systems (Dunne and Leopold, 1978), and with effluent from wastewater systems (Gray and Becker, 2002). The increase of impervious surface cover in catchments results in the increase of surface runoff (Arnold and Gibbons, 1996; Zampella et al., 2007). Consequently, the surface pollutants enter directly into urban streams reducing water quality, through increases in such factors as in Chemical Oxygen Demand (COD) and ammonium-nitrogen (NH_3-N) (Kuang, 2012).

Changes in stream hydrology, morphology, and water quality caused by urbanization have great impacts on aquatic organisms, and the cumulative changes of urbanization impacts on streams can be best reflected by resident biota (Walker and Pan, 2006). Studies of urbanization impacts on aquatic biota have typically used the microbial, macroinvertebrate, and fish assemblages (Paul and Meyer, 2001). However, there are fewer studies of the response of algae to urbanization (Paul and Meyer, 2001; Wenger et al., 2009).

Diatoms, a species-rich benthic algal assemblage readily affected by habit physical, chemical and biological disturbances and stresses, have become useful ecological indicators of stream ecosystem health because of their rapid rate of reproduction and their basal role in the food web (Stevenson and Bahls, 1999; O'Driscoll et al., 2012; Gray and Vis, 2013). Diatoms are considered as an ideal group to response of aquatic environments (Adams et al., 2013) and they have long been used to indicate water quality in rivers and streams in the United States and European countries (Stevenson et al., 2010). Most studies of urbanization impact on diatom assemblages have been attributed the effects to sanitary sewage and stormwater runoff compared to reference areas (Sonneman et al., 2001; Newall and Walsh, 2005; Walker and Pan, 2006). Diatoms have been increasingly used for assessment of urban streams in recent years (Potapova et al., 2005; Bere et al., 2014). However, there are a few studies of the distribution and changes of the diatom assemblages in cities.

Many diatom taxa are considered to be cosmopolitan, and the diatom indices based on species composition have been applied straightforwardly in different countries. However, evidence suggests that the diatom indices developed in one area are less successful in other areas because the climate, geology, local environment conditions and habitat, which are considered to be the main factors determining composition of the algal assemblages in streams, would be enormously different between regions or continents (Stevenson, 1997; Potapova et al., 2005; Kelly et al., 2008). To understand why the indices may not translate across regions, Besse-Lototskaya et al. (2011) evaluated seven trophic indicator indices by comparing trophic scores of diatom species in Europe. They found that the trophic indicator indices were a good water quality assessment tool, but the indices should be used carefully in other areas. Potapova and Charles (2007) developed diatom metrics to assess river eutrophication in the United States, they noticed the metrics were used better for monitoring the U.S. rivers than some similar metrics were created for Europe waters.

Diatom assemblages are widely used as indicators of water quality and stream ecological health in many countries and regions, but the physical and chemical parameters are the common means of monitoring water quality in China. Therefore, there are as yet very few studies of diatom assemblages in Chinese rivers (but see Tang et al., 2006 and Wu et al., 2012) that would allow their suitability as indicators to be assessed. The objectives of this research were: (1) to quantify the variations in water quality and benthic diatom assemblage structure along an urban-to-rural gradient in Beijing, China; (2) to examine whether the diatom assemblages can accurately reflect the variation in urban water quality; and (3) to construct datasets of diatom species responses to Beijing water quality. It is hoped that our results can promote the diversification of water quality monitoring in China.

2. Materials and methods

2.1. Study area

Situated in northeast China, Beijing adjoins the Inner Mongolian highland to the northwest and the great northern plain to the south, and covers a total area of 6410.54 km^2. Beijing city is located in the north temperate zone and has a moderate continental climate. The annual average temperature is 12 °C, with a mean monthly maximum of 31 °C recorded in July and a mean monthly minimum of −8 °C recorded in January. Annual mean precipitation is 588.1 mm, with most of the precipitation (about 72%) occurring in July and August with <2% occurring during the winter (Zhu et al., 2012).

In 2013, the population of Beijing was estimated at 21.14 million inhabitants (Beijing Municipal Bureau of Statistics (BMBS), 2014). The huge and rapid increase of population created tremendous pressure on the municipal services, such as garbage collection and treatment, sewage treatment, and urban storm drainage. The drainage channels of Beijing therefore receive untreated or semi-treated wastewater from many different domestic and industrial sources, as well as pollution from surface runoff. This may reduce the richness of algal, invertebrate, and fish communities and result in eutrophication (Paul and Meyer, 2001).

Our sampling sites were located within the city boundary, mainly along the Jingmi Channel and drainage channels in the fifth ring of Beijing (Fig. 1). The headwater of the Jingmi channel is the Miyun reservoir. Located in the Yanshan Mountains, it is the main source of drinking and domestic water in Beijing. Jingmi channel is 112.7 km long and acts as the main water supply conduit for the entire city area. Consequently it is well protected along the way to the storage pool, Tuancheng Lake. We compared water quality and diatom samples from (1) the Jingmi channel between Tuancheng Lake and the city center, labeled the urban upstream sites, (2) the channels draining the core city, labeled the urban downstream sites, and (3) a reference stream on Yan Mountain. Duijiuyu, the reference stream runs 14.4 km from Mayi Mountain to Mayufang reservoir, and is similar to the headwater stream of Miyun reservoir. The main land use along the reference stream is forest associated with agriculture and farmers' residences.

2.2. Diatom sampling and identification

Twenty-nine sites were sampled for diatom assemblages during late July and August of 2013. Of the 29 sites, 7 sites were in the reference stream, 22 sites were in city streams, with 6 sites upstream of the urban core, and 16 sites downstream of urban core. In order to avoid different types of substrata influence benthic diatom assemblage composition, we collected the epilithon diatoms on hard substrates (Eloranta and Andersson, 1998). The benthic diatoms were collected from the lotic parts of the sampling sites by scraping about five stones in natural streams or five areas on the cement surfaces in artificial streams, using a hard toothbrush to collect the biofilms. Samples were preserved in 100 ml plastic bottles and fixed in 4% formaldehyde. In the laboratory, about 5–10 ml samples of the diatom suspensions were cleaned of organic materials with concentrated nitric acid, and the treated samples were washed until they reached a circumneutral pH (Stoermer et al., 1995; Gray and Vis, 2013). The permanent diatom slides were mounted in Naphrax (Brunel Microscopes Ltd., Hazelbrook, Wiltshire, UK, RI = 1.73). Slides were scanned with a light microscope (Olympus BX51, Olympus, Tokyo, Japan) at 1000× magnification. A minimum of 500 diatom valves was counted on each slide; fewer valves were counted on some slides when diatoms were scarce. Valves were identified to the species or sub-species level. References for diatom taxonomy were Krammer and Lange-Bertalot (1986, 1988, 1991a,b), the database of the Patrick Center of The

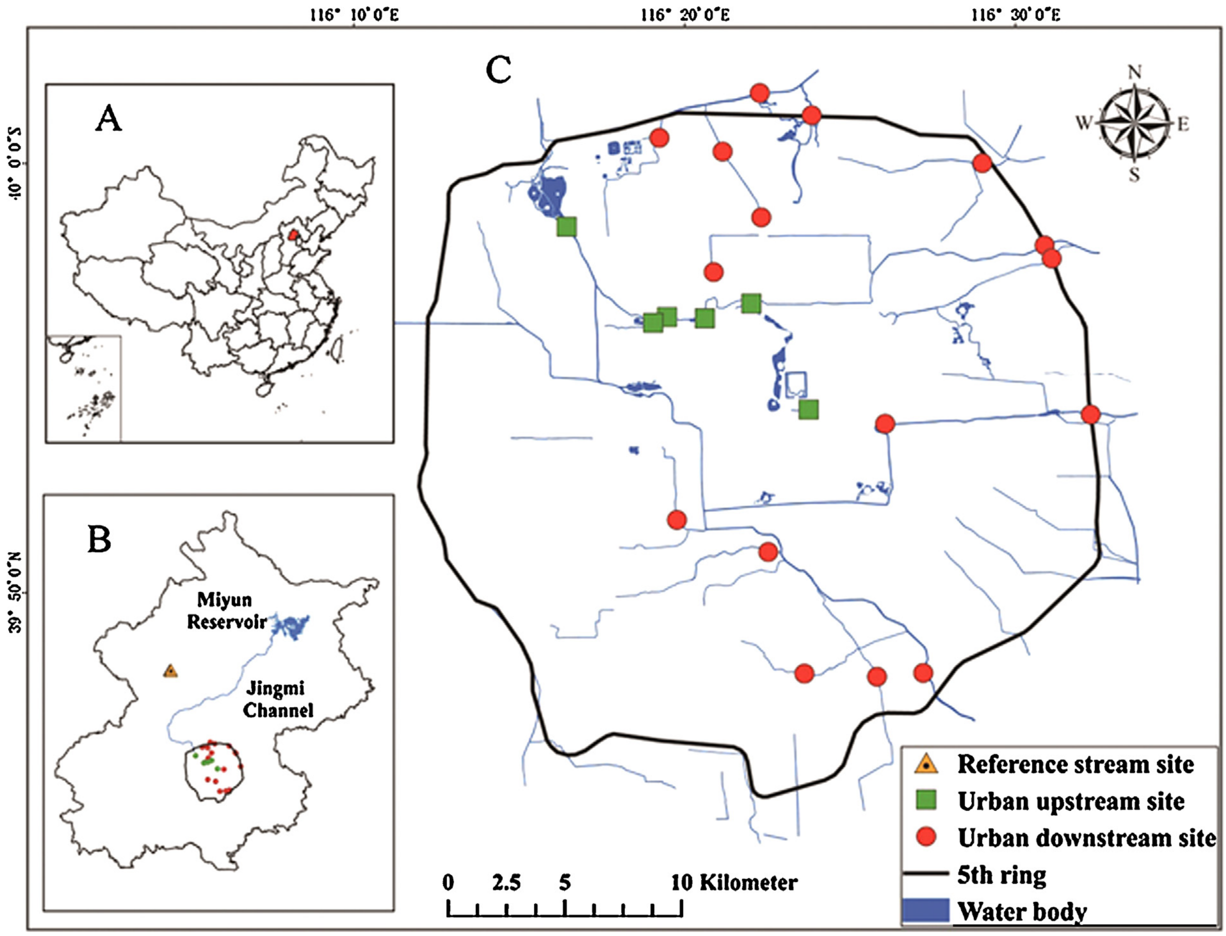

Fig. 1. The location of the urban sampling sites in (A) the study area of Beijing, China, (B) map of the Jingmi Channel and the 5th Ring of Beijing, and (C) the location of the sampling sites within the 5th Ring.

Academy of Natural Sciences, Philadelphia (ANSP), the University of Louisville, Michigan State University (http://diatom.ansp.org/), and Diatoms on the Web at Newcastle University (http://craticula.ncl.ac.uk/).

2.3. Analysis of water quality variables

Water physical and chemical samples were collected simultaneously with the diatom samples at the 29 sites. At each site, one liter of water was collected and taken back to the laboratory for analysis of water chemistry. Variables measured were total phosphorus (TP), total nitrogen (TN), chemical oxygen demand (COD), and ammonium-nitrogen (NH_3-N). The biological variable of benthic Chlorophyll *a* (Chl *a*) was also measured. All these variables were measured according to the Chinese government standard methods (Chinese State Environment Protection Bureau (CSEPB), 2002). Major cations (K^+, Ca^{2+} and Mg^{2+}) and anions (F^-, Cl^-) were measured using the Dionex Ion Chromatograph System (ICS; ICS-1000, Dionex, Sunnyvale, CA, USA) at the Beijing Urban Ecosystem Research Station (BUERS). Water temperature, pH, dissolved oxygen, and electrical conductivity were measured in situ. Water temperature (WT) and pH were measured using the Bluelab pH pen (BP1-1205-0555, Bluelab Corporation Limited, Tauranga, New Zealand); dissolved oxygen (DO) was measured using INESA dissolved oxygen meter (JPBJ-608, Shanghai REX Instrument Factory, Shanghai, China); and electrical conductivity (EC) was measured using SevenGo conductivity meter (SG-68, Mettler-Toledo International Inc., Shanghai, China).

2.4. Statistical analysis

The abundances of all taxa were expressed as relative counts before analysis. The diatom community structural attributes of species richness (S), Shannon–Weiner index (H′), species evenness (J′) and Simpson's diversity index (D′) were used to characterize each site. These metrics are commonly used in water quality bioassessment (Stevenson et al., 2010).

Diatom assemblages and their relation to water quality variables were examined using multivariate analysis to indicate the main variables and to detect the similarities among diatom samples. For analysis, we retained only those species with relative abundance of at least 1% in at least two samples per dataset. Sixty-three species were retained for analysis after removing the rare species (Appendix A). All of the water quality variables, except the pH, were $\log_{10}(X+1)$ transformed to normalize their distribution before analysis. Species proportions in assemblages were square root transformed to eliminate the influence of extreme values. Detrended correspondence analysis (DCA) was used to explore the length of gradient of the four ordination axes before selecting the appropriate model. The longest gradient of four ordination axes was 3.12, so that ordination analysis could use either a linear model or a unimodal model (ter Braak and Smilauer, 2002). Therefore, the unimodal ordination technique of canonical correspondence analysis (CCA) was used to investigate relationships between the variables and diatom assemblages from the 54 samples. Monte Carlo permutation tests (999 unrestricted permutations, $p \leq 0.05$) were used to test the significance of the first two CCA axes and to select a

Table 1
Median water quality variable (range in parentheses) from three classes of steam sites in July and August 2013. Results in boldface type are significantly different from other sites, with asterisks representing the significance level. One significant sign after the parameter means the result of these sites had the same significance level with other two; two signs mean the result of these sites have different significance level with two others. The value of n is the number of water samples in the sites.

Parameter	Unit	Reference stream sites ($n=14$)	Urban upstream sites ($n=12$)	Urban downstream sites ($n=32$)
WT	°C	**21.6** (19.0–27.0)***	28.9 (24.0–34.0)	30.9 (24.0–38.0)
EC	$\mu S\ cm^{-1}$	704 (525–872)	845 (618–1038)	**1452** (678–2620)***
pH		8.07 (7.30–8.80)	8.11 (7.50–9.70)	**8.34** (7.40–9.50)
DO	$mg\ L^{-1}$	5.91 (4.52–6.80)	5.07 (3.06–6.78)	**3.62** (0.24–8.78)***,*
COD	$mg\ L^{-1}$	13.8 (8.40–18.8)	13.2 (7.25–16.8)	**28.2** (3.79–86.3)***
TN	$mg\ L^{-1}$	8.94 (1.95–17.1)	4.89 (2.59–6.12)	**35.1** (2.41–131)***
TP	$mg\ L^{-1}$	0.09 (0.01–0.73)	0.08 (0.02–0.33)	**0.69** (0.05–2.78)***
NH_3-N	$mg\ L^{-1}$	0.45 (0.10–1.65)	2.85 (0.29–17.5)	**18.3** (0.18–239)***,**
F^-	$mg\ L^{-1}$	**1.23** (0.74–1.89)***	0.42 (0.37–0.51)	0.51 (0.13–1.71)
Cl^-	$mg\ L^{-1}$	13.0 (3.72–22.7)	18.0 (1.83–42.0)	58.6 (0.67–179)
K^+	$mg\ L^{-1}$	**3.83** (3.26–4.26)***	**5.85** (4.74–6.89)***	**23.1** (8.47–34.4)***
Ca^{2+}	$mg\ L^{-1}$	64.9 (55.6–70.5)	62.1 (30.0–81.3)	80.3 (23.0–180)
Mg^{2+}	$mg\ L^{-1}$	**8.36** (6.35–9.54)***	20.4 (14.1–23.6)	27.2 (5.22–56.4)
Chl *a*	$mg\ L^{-1}$	2.11 (0.60–5.08)	1.94 (0.44–3.77)	**4.26** (0.14–15.8)*

* $p \leq 0.05$.
** $p \leq 0.01$.
*** $p \leq 0.001$.

minimum set of environmental variables that had significant and independent effects on the diatom distribution.

One-way analysis of variance (ANOVA) was used to test the significance of the differences among the reference sites and the urban upstream and downstream sites based on the transformed water physical and chemical variables and the diatom species structure metrics. The analysis was completed using Fisher's least significant difference (LSD). Samples were classified into groups based on diatom species relative abundance, using clustering analysis to understand the species changes during July and August. The cluster analysis and one-way ANOVA were completed by IBM SPSS Statistics for Windows (version 19). DCA and CCA were performed using the program CANOCO 4.5 for Windows (ter Braak and Smilauer, 2002).

3. Results

3.1. Water quality

There were great fluctuations in water physical and chemical values among sites, although normally the city sites, especially the urban downstream sites, had higher nutrient concentrations than the reference sites and urban upstream sites (Table 1). One-way ANOVA showed that water physical and chemical variables were significantly different ($p \leq 0.05$) between urban downstream and both reference and urban upstream sites (Table 1). The average of WT and concentrations of K^+ and Mg^{2+} in the reference sites were lower than the urban sites ($p \leq 0.001$), but the average concentration of F^- was higher ($p < 0.001$) than urban sites. The pH and the average concentrations of Ca^{2+} and Cl^- in the reference sites and the urban sites did not significantly differ ($p > 0.05$). The average concentration of EC, COD, TN, TP, and NH_3-N and Chl *a* were higher in urban downstream sites than reference sites and urban upstream sites ($p < 0.05$), however, including the DO, all of these parameters did not significantly differ ($p > 0.05$) between the reference sites and urban upstream sites.

3.2. Spatial patterns of diatom assemblage structure

A total of 213 diatom species and sub-species, belonging to 54 genera, were identified in 54 samples (S1–S54) of 29 sites. Four samples of August were lost. One was a reference sample, one was an urban upstream sample, and other two were the urban downstream samples.

Achnanthidium, Amphora, Cocconeis, Cyclotella, Fragilaria, Gomphonema, Melosira, Navicula, Nitzschia, Staurosira, Pseudostaurosira, Sellaphora and Staurosira were the dominant genera in 29 sites during July and August (Fig. 2). The relative abundance of each dominant genus among sites was quite different. In reference sites, Achnanthidium, Amphora, Cocconeis and Gomphonema reached the highest (20% in at least one sample) relative abundance. Diatoms *A. minutissima*, *Achnanthidium rivulare*, *Amphora pediculus*, *Cocconeis placentula* var. *euglypta* and *Gomphonema clavatum* were the most dominant species (a relative abundance of 10% in at least once) in reference sites. In urban downstream sites, the genera Cyclotella, Nitzschia and Sellaphora had the highest relative abundance. In particular, the Nitzschia relative abundance reached 54%. The most abundant species were *Achnanthidium exiguum*, *Cyclotella meneghiniana*, *Melosira varians*, *Nitzschia palea*, *Nitzschia umbonata* and *Sellaphora pupula* with the average of *N. palea* relative abundance reaching 38.8%. In urban upstream sites, the genera Pseudostaurosira and Staurosira had the highest relative abundance, with the average relative abundance of the two genera reaching more than 25%. *Pseudostaurosira brevistriata* and *Staurosira construens* var. venter were the two most dominant.

Analysis of diatom assemblage species diversity metrics, including species richness (S), Shannon diversity (H′), species evenness (J′) and Simpson diversity (D′), showed that there were significant differences among the groups ($p < 0.05$). Urban downstream sites scored the lowest of all groups in S, H′ and J′, and had the greatest percent relative abundance of dominant taxa (Fig. 3). All four of the indices for urban downstream sites had significant differences ($p < 0.05$) from reference stream sites and urban upstream sites. However, no significant differences ($p > 0.05$) in S, H′, J′ and D′ were observed between reference stream sites and urban upstream sites.

The relative abundance of species clearly showed that all 54 samples were classified into three groups according to the cluster analysis (Fig. 4). The first group was composed of 12 samples, mainly belonging to the 7 sites from the reference stream. However, sample of S37, which had a high relative abundance of *A. minutissima*, a dominant species in the reference sites, was also included in this cluster. S35 came from the reference stream, but its species composition clustered with the urban upstream based on its dominant species abundance though the species were similar with S7 and both of the samples belonged to site 7. All 11 samples belonging to 6 sites in the urban upstream zone were members of the second group. The third group was mainly composed of 16 urban downstream sites consisting of 27 samples. The shared characteristic

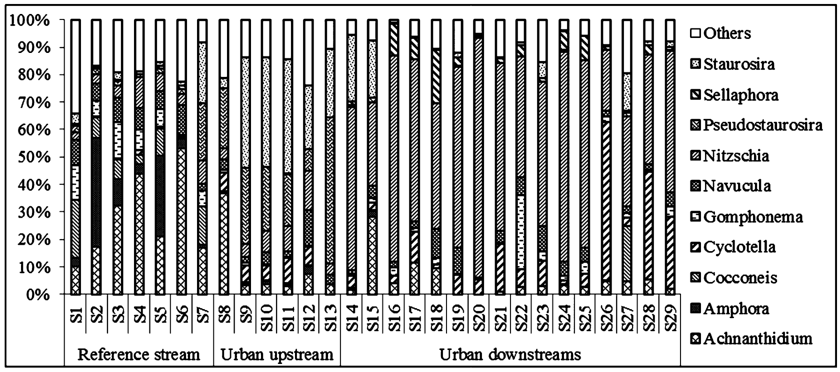

Fig. 2. Composition of 10 dominant genera (relative abundance of each genus >10% in at least one site) at the sampling sites by combining samples of July and August 2013.

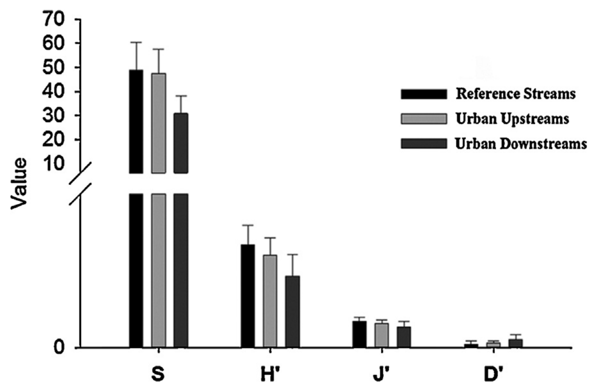

Fig. 3. Comparison of diversity indices (S, H′, J′, D′) among the reference stream, urban upstream, and urban downstream.

of S2, S40 and S42 was that they had one or two species with higher relative abundance than others (Appendix A). According to the species composition and relative abundance, S2 was incorporated into the first group, and S40 and S42 were incorporated into the third group.

3.3. Relationship of diatom assemblages to environment variables

Monte Carlo unrestricted permutation test selected 7 environmental variables (WT, TN, EC, K^+, F^-, Cl^- and pH) that had significant effects on the diatom distribution ($p<0.05$), and it indicated that both axis 1 and axis 2 were statistically significant ($p<0.01$) in the CCA (Fig. 5a and b). The species–environment correlations were high for the first two ordination axes ($r=0.943$ for axis 1 and $r=0.872$ for axis 2). These axes explained 77.3% (Axis 1 = 51.2%) of cumulative variance between species and the environmental variables and 31.9% (Axis 1 = 21.2%) of total variance in diatom species distributions among sites (Table 2).

CCA axis 1 and 2 separated the samples into three groups along the urban-to-rural sites gradient. Axis 1 represents an upstream to downstream water quality gradient. Most of the urban downstream sites were ranked in the second and the third quadrant, the reference stream sites were in the first quadrant, and the urban upstream sites were in the fourth quadrant (Fig. 5a). Axis 1 was positively correlated with F^- ($r=0.55$) and negatively with WT ($r=-0.71$), EC ($r=-0.72$), TN ($r=-0.68$) and K^+ ($r=-0.90$). Axis 2 was positively correlated with F^- ($r=0.55$) and negatively with WT ($r=-0.45$), and it may represent an urban-to-rural land cover gradient. The first group consisted of reference sites in the first quadrant, the samples 1–7 and 30–35 were positively associated with axis 1 and 2. These samples were associated with high concentration of F^- and low WT compared to the city sites. The parameters were highly positively associated with *Diatoma vulgaris*, *A. rivulare*, *Planothidium frequentissima*, *S. construens* var. *binodis*, *G. clavatum* (Fig. 5b). The second group mainly composed of urban upstream sites in the fourth quadrant. Samples 18–23 and 45–49 were associated with high WT and low nutrients. *Cyclotella ocellata* and *S. construens* were strongly positively associated with WT. The remaining samples in the second and third quadrant were the third group; it consisted of urban downstream sites with high concentration of K^+, EC, TN and Cl^- and high WT. *Achnanthidium exiguum*, *Diadesmis confervacea*, *N. umbonata*, *N. palea* and *S. pupula* were highly positively correlated with these high nutrient parameters.

4. Discussion

In China and in other areas, the traditional method of water quality monitoring is mainly by inspection of physical and chemical parameters, or by calculation of a comprehensive water quality index. An example is the Water Quality Index, which relies on normalizing or standardizing data parameters chosen according to expected concentrations and interpretation of ‘good’ versus ‘bad’ concentrations (State Environment Protection Administration, 2002; UNEP GEMS/Water Programme, 2007). However, whether water quality is good or bad is not only reflected in terms of physical and chemical parameters of the index. Quality also should be embodied in the response of all kinds of aquatic organisms, especially those that are considered to be sensitive to water environment changes, such as the diatoms, macroinvertebrates, and fishes. There are very important in aquatic food web and often used as monitoring indicator species (Barbour et al., 1999). In general, diatoms are considered as good indicators of water quality, while macroinvertebrates are better indicators of catchment disturbance (Sonneman et al., 2001; Hering et al., 2006). Our results showed that water quality monitoring combined with diatom species can be more accurate assessment of water quality than measurement of physical and chemical properties alone. This conclusion is based on comparison of the chemical and physical parameters among sites with the responses of diatoms to those same sites. Water chemical and physical parameters indicate that the urban downstream sites nutrient concentrations were the highest, and were significantly different from urban upstream and reference stream sites. Urban upstream and reference stream sites nutrient and Chl *a* concentrations were relatively low, and the EC, TN, TP, COD and Chl *a* have no significant difference between them ($p>0.05$) (Table 1).

The diatom species composition, however, was very different among the sites and could be clearly divided into three groups through the dominant genera and clustering analysis. These clusters appear to have ecological significance based on species responses to pollution. For example, *A. minutissima*, considered to be a low nutrient indicator species (Potapova and Charles, 2007), was one of the dominant species in the reference stream sites; *P. brevistriata* and *S. construens* var. *venter*, the two species were

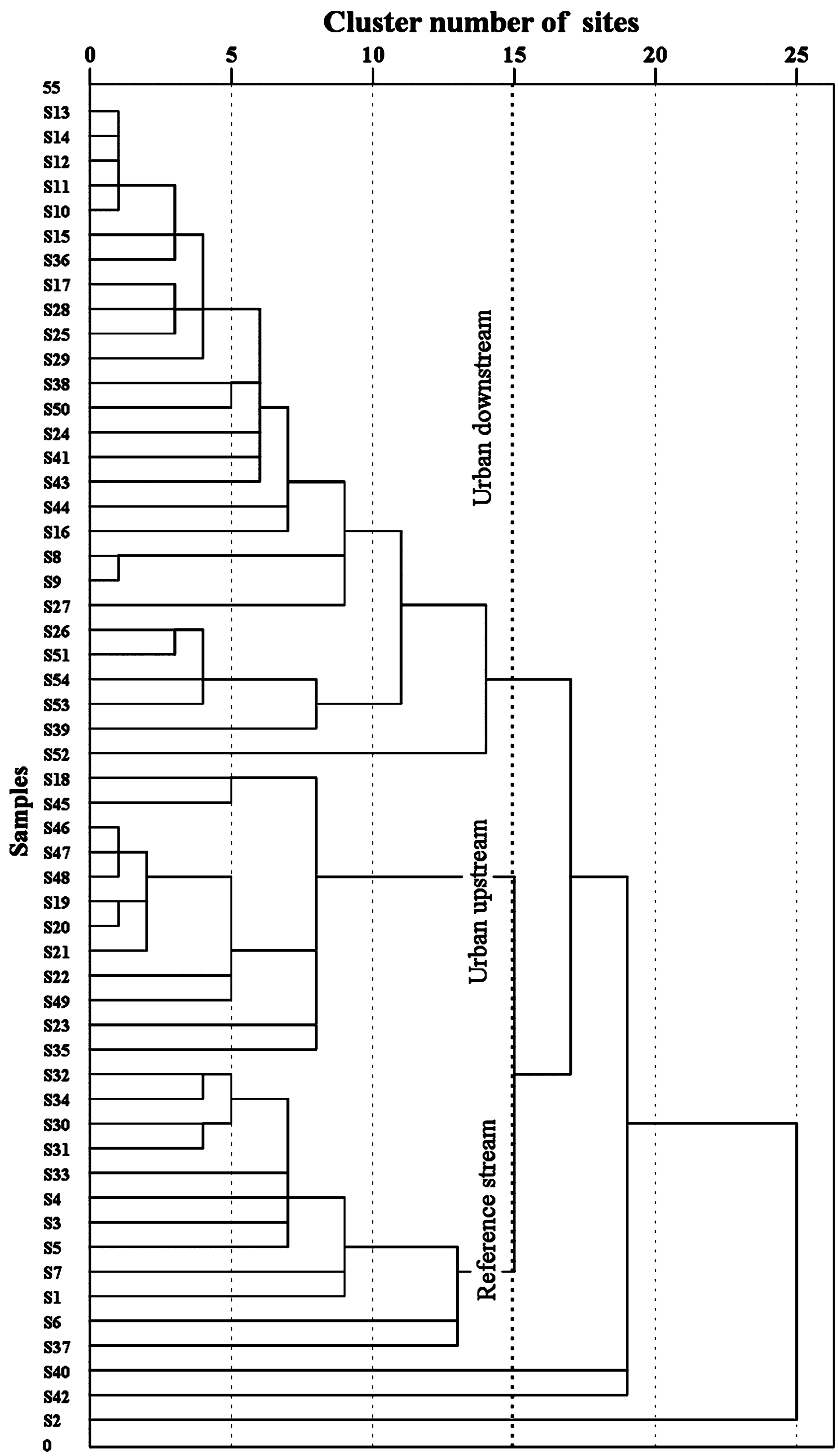

Fig. 4. Cluster analysis indicating the similarity between samples in relation to the diatom community structure at each site for July and August.

Table 2
Summary of the CCA of most dominant diatom species composition in 54 samples with respect to the 7 environmental variables.

Axes	1	2	3	4
Eigenvalues	0.436	0.222	0.084	0.037
Species–environment correlations	0.943	0.872	0.837	0.705
Cumulative percentage variance of species data	21.2	31.9	36.0	37.8
Species–environment relation	51.2	77.3	87.2	91.5

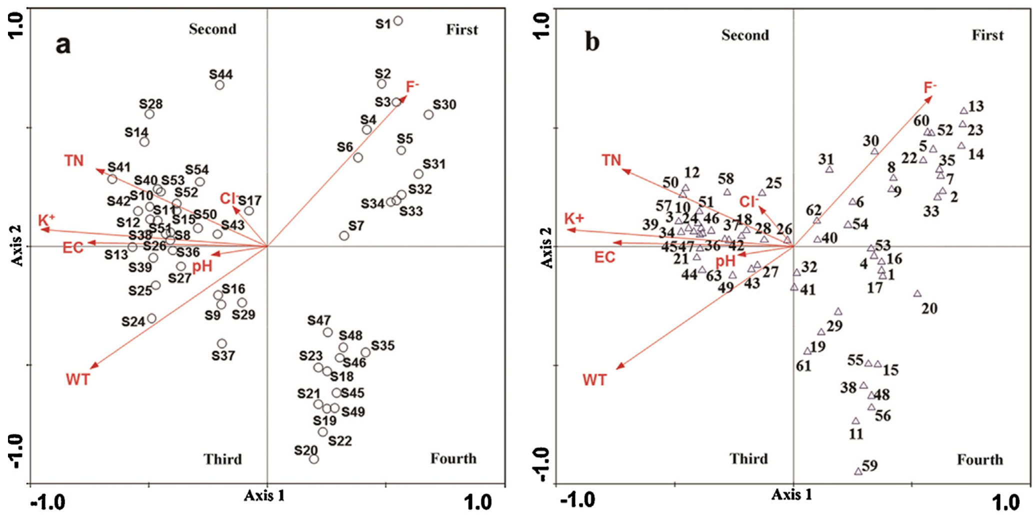

Fig. 5. CCA showed the relationship between prominent environmental variables and either (a) sampling sites or (b) taxa (the taxa number is presented in Appendix A and the abbreviations for environmental variables are shown in Table 1).

thought to be tolerant of mild pollution (http://craticula.ncl.ac.uk/; Besse-Lototskaya et al., 2011), were dominant in urban upstream sites; *N. palea*, a species tolerant to very heavy pollution in many areas (Besse-Lototskaya et al., 2011), was the dominant species in urban downstream sites. Li et al. (2012a,b) also found that *Nitzschia* was one of the dominant genera in the downstream of Beijing central city. Species life form was also a robust response to ecological conditions (Berthon et al., 2011) and water flow had important effect on it (Morley and Karr, 2002). Site 9 is an urban site, downstream of a dam, and is therefore strongly influenced by regulation. The species at this site were changed greatly. For example, *N. palea* a mobile diatom (Berthon et al., 2011) was one of the dominant species under high flow conditions in July, whereas under low flow conditions in August *A. minutissima*, a stalked diatom (Berthon et al., 2011) was the dominant species.

Species diversity is frequently used in ecological status assessment for explaining spatial and temporal patterns of biotic communities (Borics et al., 2012; Stenger-Kovacs et al., 2014). Some studies have found that species biodiversity may increase or decrease with the increase of pollution, and it mainly depends on the degree and the type of pollution (Juttner et al., 1996; Stevenson et al., 2010). For example, Bellinger et al. (2006) did not find any significant patterns of species richness and diversity between streams in forested and deforested watersheds which were not grossly polluted; but in the polluted city water bodies, species richness is lower than the mountains water bodies (Sonneman et al., 2001). Our result showed that the diversity indices (S, H′, J′, D) of urban downstream sites was lower than in the reference stream and in urban upstream sites ($p < 0.05$); however, it did not show any significant differences between reference stream and urban upstream sites ($p > 0.05$). Thus, diversity can be used for evaluation of urban downstream sites with high levels of pollution. Diversity also historically has been used successfully as an indicator of organic (sewage) pollution (Stevenson and Bahls, 1999).

CCA results showed that EC, WT, TN, K^+, and Cl^- were the important factors in structuring benthic diatom communities in the study area. The diatom composition in urban downstream sites were greatly influenced by high EC, TN, K^+, and Cl^- concentration, which may be due to the emissions of urban sewage and industrial wastewater (Gray and Becker, 2002). The other main reason was due to the change of urban land cover. The increasing impervious area can shorten the stormwater runoff time, and the rainwater washes a large number of pollutants from the surface of roofs, roads, highways, bridges, squares and other urban impervious surfaces into the rivers directly, increasing the concentration of various particles, pollutants, and ions (Vaze and Chiew, 2002; Shuster et al., 2005; Ren et al., 2008). EC is a good monitor which can reflect the total concentration and total content of the ions in water bodies. In this research, EC concentration in urban downstream sites averaged 1452 μs/cm, higher than the 704 μs/cm observed in reference stream sites and the 845 μs/cm observed in urban upstream sites ($p < 0.001$). Walker and Pan (2006) also found that EC was the important environmental factor to distinguish the diatom gradient from urban to suburban sites. EC was also considered to be the determining factor of diatom species composition and distribution of the benthic diatoms in U.S. rivers (Potapova and Charles, 2003; Stevenson et al., 2008). Cl^- was one of the important ions affect diatom species composition in the sites according to CCA. Due to the increase of linear highway miles and the application of road deicers, much chloride was washed into the rivers. High chloride concentrations co-vary with nutrients (for example, nitrogen and phosphorus) and contaminants (such as organic matter and heavy metals), and such high chlorides threatened freshwater ecosystems by salinizing fresh waters, degrading aquatic habitat and community composition, and altering the diversity of food webs (Kaushal et al., 2005; Gobel et al., 2007; Porter-Goff et al., 2013).

WT is an important factor of aquatic biological activity and nutrient cycling. It will not only affect the growth of aquatic organisms and their respiratory rate, but also will affect the concentration of dissolved oxygen in water (Becker et al., 2010). Weighted correlation matrix results showed the 1st CCA axis correlated well with diatom composition along the urban-to-rural gradient ($r = -0.71$). WT of streams within the urban area was nearly 10 °C higher than the reference stream ($p < 0.001$). Thermal pollution in urban water was a result of hydrologic connections to impervious surfaces, decreased riparian canopy cover and, release of stored water from shallow ponds, and point discharges from wastewater treatment plants (Wenger et al., 2009; Somers et al., 2013).

Many indices, such as the diversity index, the Trophic Diatom Index (TDI), and the River Diatom Index (RDI), which are based on the species present and their relationship with the environment, have been widely used in ecological classification and water quality monitoring. However, it was difficult to use these indices in different areas than those in which they originated because the communities are affected by multiple and different stressors in different localities. The distribution of *A. minutissima* illustrates the difficulty. Most observation found that its relative abundance would decrease with increasing nutrient content of water bodies (Kelly and Whitton, 1995; Pan et al., 1996; Besse-Lototskaya et al., 2011), but Stoermer (1980) clearly characterized *A. minutissima* as "apparently tolerant of nutrient addition, but also quite abundant in more oligotrophic regions", and Potapova and Charles (2007) thought the characterization effectively summarized the current knowledge about the relation of this taxon to nutrients. *A. pediculus* was another dominant species in reference stream sites in this research. It was also considered to prefer good quality of the water in the river Tenna of central Italy (Torrisi et al., 2010). In the rivers of the United States, however, the species was determined as a high TN and TP indicator especial in eastern plains and highlands (Potapova and Charles, 2007). In fact, a number of studies have questioned the general applicability of the indices because the diatom database and the local environment were different from the source areas of the indices, and it has been suggested that indices should be adjusted to regions for which they are applied (Lavoie et al., 2006; Potapova and Charles, 2007; Besse-Lototskaya et al., 2011; Mykra et al., 2012).

5. Conclusions

There was large variability in physical and chemical variables and in diatom diversity indices between urban upstream and downstream waterways of Beijing, but there was no significant difference in physical and chemical variables between urban upstream and reference stream samples. However, the composition and spatial patterns of diatom assemblages were very different among the streams. The gradients in concentration of EC, TN, WT and major ions were identified as the primary determinants for the spatial patterns of diatom assemblages. Our results showed that the benthic diatom composition was more sensitive and accurate than the routine investigation of water physical and chemical parameters. It provides important complimentary information for water quality and conditions in Beijing, China. Therefore, we suggest using diatoms, along with water physical and chemical parameters, for surface water quality assessment. In addition, our results on diatoms may also suggest future research on aquatic organisms, such as macroinvertebrate and fish.

In fact, many countries, such as USA, UK, Netherlands, France, Germany, and Australia and so on, have used the diatom indices for regional water quality assessment broadly. However, the indices differ greatly, causing much confusion and arbitrariness among researchers because diatom metrics or indices developed in certain area were less successful when applied in other areas (Pipp, 2002; Rott et al., 2003; Potapova and Charles, 2007; Besse-Lototskaya et al., 2011). Therefore, caution must be applied in using indices from one region to another place. In order to improve the regional water quality bioassessment in China, autecology-based diatom indices should be developed to (1) have a set of unified sampling, identification and analysis methods; and (2) have a complete set of diatom species data at the regional level along with environmental data that accurately represent the local conditions of diatom environment.

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Appendix A.

List of 63 most abundant diatom species from reference stream and urban streams in Beijing, China. The value of n was the number of diatom samples in the sites according to CCA (+ = 0–5%, ++ = 5–10%, +++ = 10–20%, ++++ = 20–50%, +++++ = >50%).

	Code	Reference stream sites (n = 12)		Urban upstream sites (n = 12)		Urban downstream sites (n = 30)	
		July (n = 7)	August (n = 5)	July (n = 6)	August (n = 6)	July (n = 16)	August (n = 14)
Achnanthidium catenatum (Bily & Marvan) Lange-Bert.	1	+	+	+	+	+	+
A. deflexum (Reim.) Kingston	2	+	+	+	+		
A. exiguum (Grun.) Czarn.	3	+		+	+	+	+
A. minutissimum (Kutz.) Czarn.	4	+++	+++	++	+	+	+
A. rivulare Potapova & Ponader	5	++	++	+	+	+	+
Amphora cf. strigosa CODY Hust.	6	+					+
A. pediculus (Kutz.) Grun.	7	+++	+++	+	+	+	+
Cocconeis placentula Ehr.	8	++	++	+	+	+	+
C. placentula var. *euglypta* (Ehr.) Grun.	9	+	+		+	+	+
Cyclotella meneghiniana Kutz.	10	+	+	+	+	++	+++
C. ocellata Pantocsek	11			+	+	+	+
Diadesmis confervacea Kutz.	12	+				+	+
Diatoma vulgaris Bory	13	+	+		+		+
Encyonema minutum (Hilse) Mann	14	+	+				
Encyonopsis microcephala (Grun.) Kramm.	15	+		+	+	+	
Fragilaria capucina Desm.	16	+	+	+		+	+
F. capucina var. *gracilis* (Østrup) Hust.	17	+	+	+	++	+	+
F. capucina var. *mesolepta* (Rabh.) Rabh.	18	+			+	+	+
F. cf. crotonensis Kitton	19	+	+	+	+	+	+
F. vaucheriae (Kutz.) Petersen	20	+	+	+	+	+	
Gomphonema augur Ehr.	21			+	+	+	+
G. clavatum Ehr.	22	+	++	+	+	+	+
G. minusculum Krasske	23	+	+		+		+
G. parvulum (Kutz.) Kutz.	24	+	+		+	+	++
Halamphora veneta (Kutz.) Levkov	25	+	+	+		+	+
Melosira varians Ag.	26	+	+	+	+	+	++
N. caterva Hohn et Hellerman	27	+	+	+	+	+	+
N. cryptocephala Kutz.	28	+	+	+	+	+	+
N. cryptotenella Lange-Bert.	29	+	+	+	+	+	+
N. gregaria Donkin	30	+	+		+	+	+
N. parablis Hohn et Hellerman	31	+	+	+		+	+

N. recens Lange-Bert.	32		+	+	+	+	+
N. rostellata Kutz.	33		+		+	+	+
N. subminuscula Mang.	34				+	+	+
N. tripunctata (Mull.) Bory	35	+	+		+	+	+
N. veneta Kutz.	36	+	+		+	+	+
Nitzschia amphibia Grun.	37	+	+	+	+	+	+
N. amphibioides Hust.	38		+	+	+	+	+
N. archibaldii Lange-Bert.	39		+		+	+	+
N. fonticola (Grun.) Grun.	40	+	+	+	+	+	+
N. frustulum (Kutz.) Grun.	41	+	+	+	+	+	+
N. gracilis Hantz.	42	+	+	+	+	+	+
N. inconspicua Grun.	43	+	+		+	+	+
N. laevis Hust.	44			+	+	+	+
N. microcephala Grun.	45				+	+	+
N. palea (Kutz.) Smith	46	+	+	+	+	+++++	++++
N. paleacea (Grun.) Grun.	47	+	+	+	+	+	+
N. sinuata var. *tabellaria* (Grun.) Grun.	48	+		+	+	+	
N. tubicola Grun.	49		+	+	+	+	+
N. umbonata (Ehr.) Lange-Bert.	50				+	+	+
N. vermicularis (Kutz.) Hantz.	51	+		+		+	+
Planothidium frequentissima Lange-Bert.	52	+	+		+	+	+
Psammothidium chlidanos Lange-Bert.	53		+	+	+	+	+
P. subatomoides (Hust.) Bukht. & Round	54	+	+	+	+	+	+
Pseudostaurosira brevistriata (Grun.) Williams & Round	55	+	+	++++	++++	+	+
P. pseudoconstruens (Marciniak) Williams & Round	56			+	+		+
Sellaphora pupula (Kutz.) Meresck	57	+	+	+	+	+	+
S. seminulum (Grun.) Mann	58	+	+	+		+	+
Staurosira construens Ehr.	59			+		+	
S. construens var. *binodis* (Ehr.) Hamilton	60	+	+			+	
S. construens var. *venter* (Ehr.) Hamilton	61	+	+	++++	++++	++	+
Synedra ulna (Nitz.) Ehr.	62	+	+	+	+	+	+
Thalassiosira weissflogii (Grun.) Fryxell & Hasle	63		+	+		+	+

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