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Modeling macrozooplankton and water quality relationships after wetland construction in the Wenyuhe River Basin, China

Zongming Ren[a,b], Yang Zeng[a], Xiu'e Fu[a], Gaosheng Zhang[a], Linlin Chen[a], Jing Chen[a], Tae-Soo Chon[b,*], Yawei Wang[c], Yuansong Wei[c]

[a] Key Laboratory of Coastal Zone Environmental Processes, Yantai Institute of Coastal Zone Research, Chinese Academy of Sciences, Yantai 264003, PR China
[b] Department of Biological Sciences, Pusan National University, Pusan 609735, Republic of Korea
[c] Research Center for Eco-Environmental Science, Chinese Academy of Science, Beijing 100085, PR China

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ABSTRACT

The biodiversity of macrozooplankton and physicochemical indices at eight sites in the Wenyuhe River Basin were analyzed from April 2010 to March 2011 to assess the water quality following construction of a wetland (CW). The association between biodiversity and physicochemical parameters was analyzed using a self-organizing map (SOM). The water quality in the upstream portion of the CW showed no overall improvement during this period based on biodiversity (H') and physicochemical assessment (Pb/n). Both assessments indicated that the water quality in upstream of CW was partially improved and showed an evident time dependent relationship after CW. Although a sample site, which received water from two streams, showed higher biodiversity in an integrative manner, the physicochemical assessments were more heavily influenced by treated water from the Qinghe Sewage Treatment Plant, rather than upstream at the other river basin. This was a noteworthy demonstration that physicochemical and biological assessments were not in accord and addressed the separate functioning of river ecosystems under field conditions. Physicochemical assessment may represent water quality more conservatively with respect to drinking water maintenance than biodiversity assessment. Nitrogen pollution was the main cause for the decrease in physicochemical conditions. Overall, the results indicated that the association between biodiversity and physicochemical assessments determined from the SOM could reveal aquatic ecosystem quality in a comprehensive manner and therefore be a useful tool for defining water quality and ecological integrity in aquatic ecosystem management, especially for complex water environments.

1. Introduction

From 1960 to the beginning of the 21st century, global water pollution has posed a serious threat to human health in developed countries of Europe, North America, Asia and Australia (Singh and Sekhon, 1976; Duttweiler and Nicholson, 1983; Agradi et al., 2000). Due to unprecedented population growth and economic development, major rivers are increasingly suffering from severe degradation of water quality, especially in economically emerging countries (Krishna et al., 2009; Hanh et al., 2010; Bai et al., 2011). Owing to intensive crop agriculture, aquaculture, livestock production and various industries, domestic areas discharge large amounts of untreated wastewater to surface water in excess of the self-purification capacities of these aquatic systems. Consequently, water pollution leads to decreased biodiversity, environmental degradation, and human health hazards (Jarup, 2003; Treseder, 2008).

River systems play important roles in the sustainable development of the entire biophysical environment (Kowalkowski et al., 2006). Sustainable water quality management at the basin scale is a worldwide problem (Huang and Xia, 2001; Hartmann et al., 2006), especially in developing countries, where social and economic development has put water resources under extremely high stress (Zhu et al., 2008). The Wenyuhe River, which is a river in northern China that is crucial to the overall development strategy for the ecological health and social sustainability of Beijing, is a typical example of these problems.

A growing interest in effective low-cost treatment of polluted water and wastewater has led to many studies of constructed wetlands (CWs) (US EPA, 2000; Scholz, 2006; Vymazal, 2007; El-Sheikh et al., 2010). In CWs, treatment performance has been accomplished through an integrated combination of biological, physical and chemical interactions among wetland components (US EPA, 2000). A CW was built in the Wenyuhe River in April 2010 as a

* Corresponding author. Tel.: +82 51 510 2261; fax: +82 51 512 2262.
E-mail address: tschon@pusan.ac.kr (T.-S. Chon).

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sustainable method for wastewater treatment to improve the water quality.

The CW in the Wenyuhe River may play an important role in water quality improvement and biodiversity maintenance. However, when assessing the effects of the CW on water quality improvement, physicochemical analyses alone may not be sufficient to comprehensively describe the complex adverse effects of chemical mixtures on communities present at contaminated sites (Cairns and Dickson, 1973; Hellawell, 1986), even though water pollution agents play a primary role in destruction of ecosystems (Rosenberg and Resh, 1993). Macrozooplankton is essential to maintain ecosystem stability in river dynamics. Specifically, they act as grazers that control algal and bacterial populations, as a food source for higher trophic levels, and as a source of dissolved nutrients (Bruce et al., 2006). Thus, understanding their role in the distribution and flux of nutrients in aquatic systems is critical to understanding the functional role of communities and enabling their effective environmental management. Due to the high complexity, dynamic nature, and non-linearity of both spatial and temporal pollution patterns (Carafa et al., 2007), only biological parameters such as measures of abundance (coverage, density, biomass), diversity (species richness, Shannon index), and pollution tolerance (biological monitoring working parties and biological monitoring water quality score systems) (Camargo et al., 2011) have been reported. However, these parameters alone do not sufficiently address ecological integrity or changes in water quality.

Alternative methods for presenting community and ecosystem functioning in the regime of ecological modeling and informatics have been developed. Jørgensen (1994) reviewed the role of ecological models in understanding ecosystems and environmental management to deal with system response to stressors. In addition, ecological informatics has been developed to intensify data treatment and management during modeling. Artificial neural networks (ANNs) have also increasingly been used for complex data evaluation and modeling in various fields of ecology and environment sciences (Kwak et al., 2002; Park et al., 2003; Özesmi et al., 2006; Martín et al., 2008; Carafa et al., 2011), and have been reported to perform better than classical statistical methods for prediction and assessment (Lek and Guégan, 1999; Samecka-Cymerman et al., 2007). One of the best known ANNs containing unsupervised training algorithms is the Kohonen self-organizing map (SOM) (Kohonen, 1982), which allows classification of data without prior knowledge or assumptions. The SOM helps to visualize and interpret large and high dimensional databases by organizing data on grids with low dimensions (conventionally two). Another advantage of the SOM when compared to classical statistical methods is that a small fraction of missing data is allowed and predictions are inferred automatically by the model (Carafa et al., 2011).

In this study we applied the SOM to community and environmental data collected after construction of a wetland (1) to assess the effects of the CW on water quality of the Wenyuhe River based on physicochemical and biological measurements, and (2) to use the SOM to reveal associations between the biodiversity of macrozooplankton and key environmental variables. The SOM training and visualization could be a useful tool for effective characterization of the associative role between biodiversity and environmental factors in aquatic ecosystems and for generation of comprehensive information for assessment of changes in water quality.

2. Materials and methods

2.1. The study area

The Wenyuhe River Basin, which is situated in the northeast part of Beijing, occupies a total area of 4300 km^2, representing 26% of the territory of Beijing (Fig. 1). The area contains a high concentration of industrial and agricultural activities and urban areas, which results in a high demand for water.

The river basin, which includes the Wenyuhe River, the Qinghe River and the Longdaohe River, runs through Beijing, serves as a natural ecological barrier, and is of critical importance to Beijing's overall development strategy for the metropolitan area. However, population growth, economic development and the associated intensification of human activities in river catchments that has occurred in recent years have resulted in the amount of wastewater discharged into the river basin exceeding its self-purification capacity, leading to serious pollution (Wang et al., 2010).

The Qinghe River, which is the longest tributary in the Wenyuhe River Basin, has been heavily impacted by the discharge of treated water from the Qinghe Sewage Treatment Plant (between S1 and S2). CW for the sustainable of wastewater treatment in the Wenyuhe River Basin was built in the Longdaohe River in April 2010, which is a tributary of the Wenyuhe River (Fig. 1). Additionally, the direct discharge of sewage and garbage into the Wenyuhe River Basin has been forbidden since January 2010.

2.2. Sampling design

To assess the effects of the CW on improvement of water quality, macrozooplankton was sampled monthly (in the middle of the month) at eight sampling sites in the Wenyuhe River Basin (Fig. 1 and Table 1) from April 2010 to May 2011.

Macrozooplankton was collected from the left and right banks, as well as from the middle of the river. Specifically, 10 L of water was filtered through a 45-μm Nytal net at approximately 20 cm below the surface and then fixed with 4% formalin. Counting was then carried out in 15 mL sedimentation chambers using an inverted microscope. Macrozooplankton species were identified based on Han and Shu (1995), which were defined as "relatively large plankton, visible to the naked eye, and the minimum size is defined in various sources as from 0.5 mm to 5.0 mm".

The following water physicochemical factors were measured at the sample sites: water temperature (T; YSI meter Model 33®), pH (Beckman Model F8 253®), conductivity (EC; YSI meter Model 33®), and dissolved oxygen (DO; YSI oxygen meter Model 57®). In addition, the chemical oxygen demand (COD), ammonium nitrogen (NH_3—N), total nitrogen (TN) and total phosphorus (TP) were analyzed according to the Environmental Quality Standard for Surface Water of China (GB3838-2002, 2002). The physicochemical parameters were determined using a Hach DR 2000® spectrophotometer at the specified wavelength (GB3838-2002, 2002). Three replicates of the physicochemical parameters were recorded directly at each sampling site. Additionally, water samples for analysis of the remaining parameters were collected into polyethylene bottles, preserved with 2 mL of concentrated hydrochloric acid (pH < 2.0) and then transported back to the laboratory. All water samples were kept in a refrigerator at 4 °C until analysis.

2.3. Indices

The Shannon–Wiener index (Shannon and Weaver, 1963) was used to assess the biological water quality and the obtained H' values are presented in Table 2 (Liu et al., 2010).

The physicochemical parameters at the sample sites were assessed using the physicochemical index (Shen et al., 1990):

$$\mathrm{Pb} = \sum_{i=1}^{n} P_i \tag{1}$$

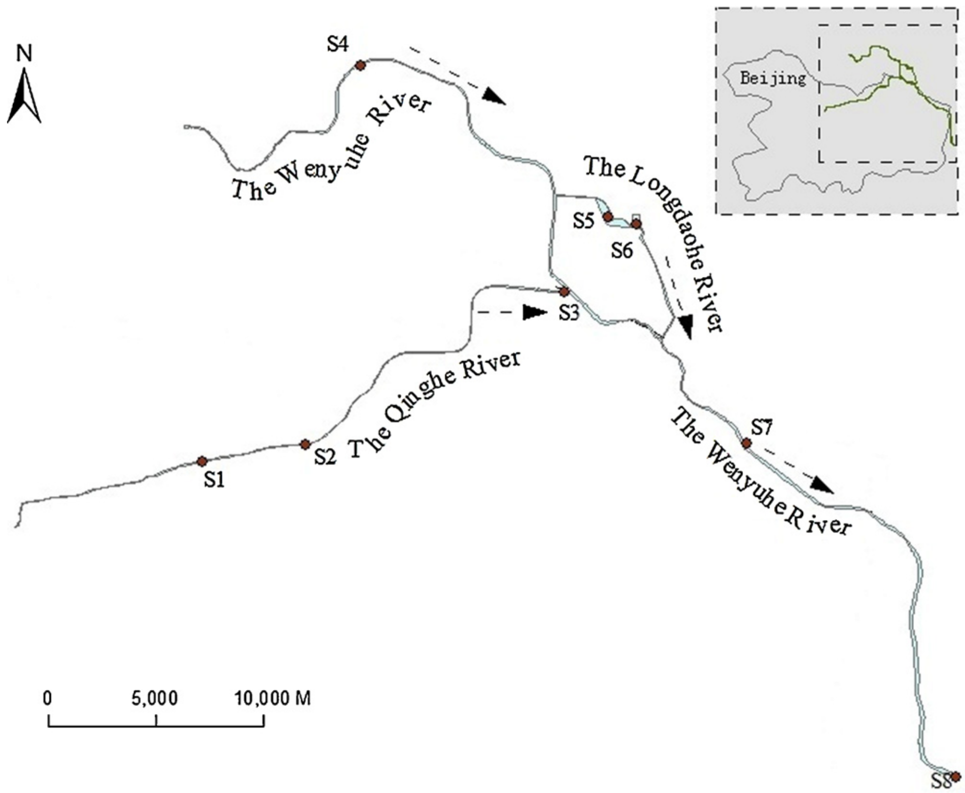

Fig. 1. Wenyuhe River Basin and sampling sites.

Table 1
Location and description of sampling sites.

Sampling sites	Longitude	Latitude	Description
S1: Upstream Qinghe River	116°20′25.99″E	40°1′28.98″N	100 m upstream of the Qinghe Sewage Treatment Plant
S2: Downstream Qinghe River	116°21′38.11″E	40°1′39.73″N	500 m downstream of the Qinghe Sewage Treatment Plant
S3: Sand Bamp	116°29′20.52″E	40°4′44.86″N	Cross point of the Qinghe River and the Wenyuhe River
S4: Mafang Bridge	116°24′15.87“E	40°9′2.06“N	Upstream of the Wenyuhe River
S5: Roma West Lake	116°30′25.65″E	40°6′10.02″N	Water inflow of the CW in the Longdaohe River
S6: Roma East Lake	116°31′10.43″E	40°6′4.29″N	Water outflow of the CW in the Longdaohe River
S7: Weigou Bridge	116°33′52.82″E	40°1′52.78″N	Downstream of CW, and intake of water from Site 3 and Site 6
S8: Beiguan Brake	116°39′6.77″E	39°55′34.35″N	Downstream of S7

where n is the number of the physicochemical parameters. For EC, COD, NH_3—N, TN and TP, P_i was determined by Eq. (2), while Eqs. (3) and (4) were used for DO and pH, respectively.

$$P_i = \frac{M_i}{M_0} \tag{2}$$

$$P_i = \frac{M_0}{M_i} \tag{3}$$

$$P_i = {M_i}^{|M_i - M_0|} \tag{4}$$

where M_i is the value of different physicochemical parameters of the i-th site and M_0 is the threshold value for COD (e.g., COD ≤ 15 mg/L, M_0 = 15 mg/L), NH_3—N, TN, DO, EC and TP. The value 7 was assigned for pH according to the Grade II Surface Water Standards of GB3838-2002 (2002). The obtained Pb/n values were categorized to indicate water quality as shown in Table 3 (Shen et al., 1990).

2.4. Self-organizing map (SOM)

To assess the changes in water quality and ecosystem functioning at the sampled sites, the data for biodiversity and physicochemical parameters were patterned by training using the SOM. The SOM performs a non-linear projection of data onto a space in two dimensions and provides a patterned map of input data trained with unsupervised learning (Kohonen, 1982). The size of the SOM was determined heuristically in such a way that could be comprehensible to the reader in a smaller number of dimensions. The highest variance in the input data will be projected along the vertical axis (longer usually), while the following variance would be presented on the horizontal axis (shorter usually). The optimal

Table 2
Estimation of water quality according to H'.

Diversity	H' = 0	0 < H' ≤ 1	1 < H' ≤ 2	2 < H' ≤ 3	H' > 3
Water quality	No macrozooplankton or only one species	Serious pollution	α-Type pollution	β-Type pollution	Clean

Table 3
Relationship between water quality and Pb/n.

Pb/n	Pb/$n \leq 1$	$1 <$ Pb/$n \leq 2$	$2 <$ Pb/$n \leq 3$	$3 <$ Pb/$n \leq 4$	Pb/$n > 4$
Water quality	Clean	Light pollution	Medium pollution	Serious pollution	Very serious pollution

size of the computational modes was adjusted based on the degree of discrimination among the grouped nodes after training. About two thirds of the total nodes were occupied by the nodes with the samples, while one third consisted of empty nodes that served as delimiters between the occupied nodes. Through a preliminary investigation, the size of 12×9 nodes was selected to train the community data (variables, species; cases, sample units) in this study. For visualization of the environmental profiles, 4×3 nodes were used to train the input data (variables, water quality parameters plus biodiversity; cases, sample units). The Euclidian distance ($d_j(t)$) at the j-th node on the SOM between the weight at iteration time t and the input vector was calculated through learning processes:

$$d_j(t) = \sum_{i=0}^{P-1} [x_i - w_{ij}(t)]^2 \quad (5)$$

where x_i is the value of the i-th parameter, $w_{ij}(t)$ is the weight between the i-th parameter and the j-th node on the SOM, and P is the number of the parameter.

The best matching neuron, which has the minimum distance, was selected as the winner. For the best matching neuron and its neighboring neurons, the new weight vectors were updated as follows:

$$w_{ij}(t+1) = w_{ij}(t) + \alpha(t)[x(t) - w_{ij}(t)] \quad (6)$$

where t is the iteration time and $\alpha(t)$ is the learning rate. The learning process of the SOM was conducted using the SOM Toolbox developed by the Laboratory of Information and Computer Science, Helsinki University of Technology, in the Matlab environment (Vesanto et al., 2000). The initialization and training processes followed suggestions by the SOM Toolbox by allowing optimization in the algorithm. Detailed descriptions of application of the SOM to behavioral data are available elsewhere (Park et al., 2005; Chon, 2011; Liu et al., 2011).

As the input data were provided to the SOM for training, the weights of the best matching unit and computation nodes close to it were adjusted toward the input vector through interactive calculation. To reveal the degree of association between SOM units, Ward's linkage method was used to cluster the movement data based on the dendrogram according to the Euclidean distance (Ward, 1963). The linkage distances were rescaled at 0–100%.

3. Results and discussion

3.1. Patterns of macrozooplankton in the Wenyuhe River Basin

There was more than 42 species in the survey area during the assessment period. The patterns of the species of macrozooplankton in the Wenyuhe River Basin were shown on the SOM (Fig. 2). Cluster analysis identified six groups based on the abundance of macrozooplankton (Fig. 2a and b). The cluster distances according to the Ward's linkage indirectly indicated the closeness between clustered groups. Vertical grouping was primarily observed according to season. The upper group consisting of clusters 1, 6 and 3 represented samples collected mainly in winter, while the bottom group, including clusters 2, 4 and 5, were obtained broadly from spring to autumn.

Within each upper and bottom group, the clusters were further divided according to sample sites. Samples collected from various sites during winter were grouped in cluster 1 in the upper group (Fig. 2a). Clusters 3 and 6 placed within the upper group were closer to each other than to cluster 1. Cluster 6 was characterized by the samples collected at S6, S7 and S8, where low winter temperatures were observed. Samples comprising cluster 3 were somewhat similar to those of cluster 1, but samples from S7 and S8 were mainly excluded from this cluster.

Within the bottom group, cluster 2 broadly represented samples collected from S1, S2, S3, S5, S7 and S8 during summer (Fig. 2a and b), while clusters 4 and 5 were more closely associated with each other. It should be noted that cluster 4 was specifically grouped by the samples collected in S6. Communities grouped in cluster 5 represented samples collected from S1 to S5.

Profiles of species distribution were presented on the SOM through visualization (Fig. 2d). Some species were broadly observed separately in the upper area (e.g., *Brachionus calyciflorus*) and in the upper area (*Daphnia pulex*). However, no species was found to occur broadly across the entire SOM. In contrast, numerous species occurred specifically in different clusters in narrow ranges. For instance, species observed in winter such as *Synchaeta* sp. and *Amoeba* sp. were only collected from a few sites (e.g., S2 and S3) that had higher temperatures in winter (Fig. 2d). The SOM patterning was suitable for identifying the scope of species on spatial and temporal scales. Further results characterizing the distribution patterns in detail for each species according to sample sites, season and environmental factors will be reported elsewhere.

3.2. Assessment of the water quality

The assessment of water quality from April 2010 to March 2011 based on H' is shown in Fig. 3. H' changed ranging from 0.25 to 2.47 during the survey period. Immediately after CW in April–May, 2010, the biodiversity values decreased, but they then increased as time progressed (Fig. 3). The trends in biodiversity also varied among sample sites. It is worth noting that the sample sites located in the Qinghe River showed lower values in H' than those of samples collected from sites located in other rivers. Diversity values at S2 and S3 were mostly lower than 1.0 and close to the state of serious pollution (Table 2) during the assessment period. It should also be noted that the values remained at lower levels without seasonality, especially in S2 (Fig. 3). The upstream site, S1, showed slightly higher diversity than S2 or S3.

Biodiversity values in the Wenyuhe River were relatively higher than those observed at sample sites in the Qinghe River. The upstream site, S4, showed lower H' values that were similar to those observed at S3 in the Qinghe River. H' values in the downstream site S6 were higher than the H' values in S4 and S5. These findings indicated that ecological integrity increased with longitudinal axis in the Wenyuhe and Longdaohe River (Fig. 1). Seasonality was observed at sites S4, S5 and S6, where biodiversity decreased substantially during winter (Fig. 3).

Interesting biodiversity trends were also observed in S6 and S7. From the beginning of CW construction in April 2010, H' increased from about 0.9 to greater than 2.2 in October 2010 for both sites. Seasonality in biodiversity was also observed at these sites. In winter, the biodiversity decreased to about 1.5 due to the lower water temperature. However, as the temperature increased during spring, the biodiversity increased to about 2.47 at these sites in March 2011

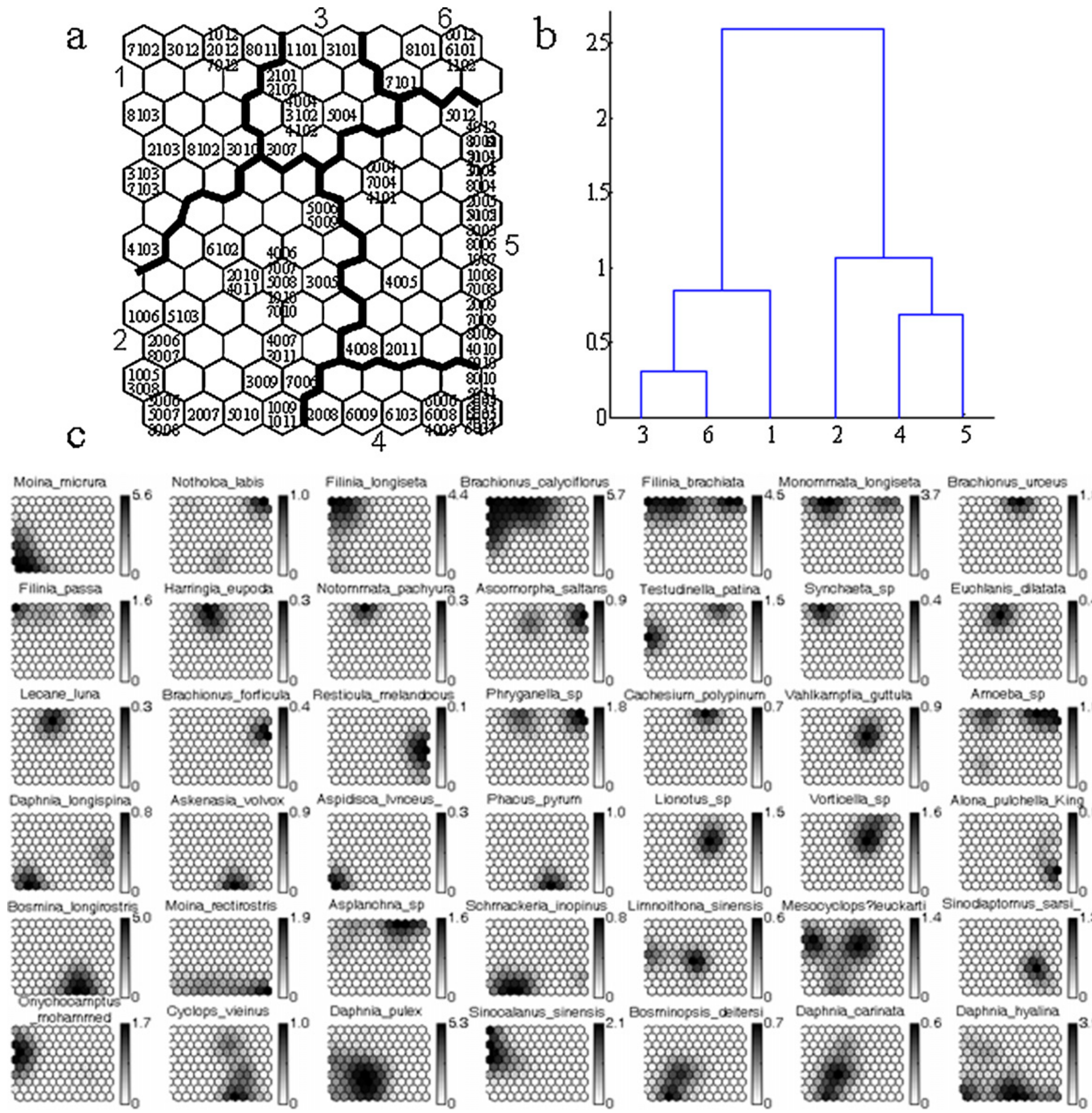

Fig. 2. Patterns of species of macrozooplankton in the Wenyuhe River Basin. (a) Pattern results based on different sites and different times with six clusters classified by the SOM. The numbers listed in the name of sample sites indicate the site name and sampling time. For example, 1004 indicates data from site 1 on April 2010, and 2101 indicates data from site 2 on January 2011; (b) cluster distances according to the Ward's linkage method; (c) profiles of macrozooplankton species in the Wenyuhe River Basin visualized on the SOM. The vertical bar shows the relative species abundance.

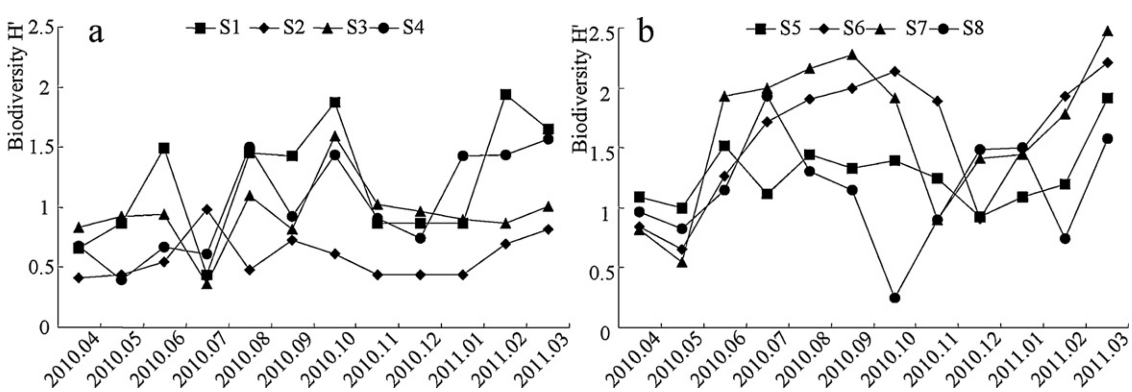

Fig. 3. Shannon–Wiener biodiversity at different times and sites.

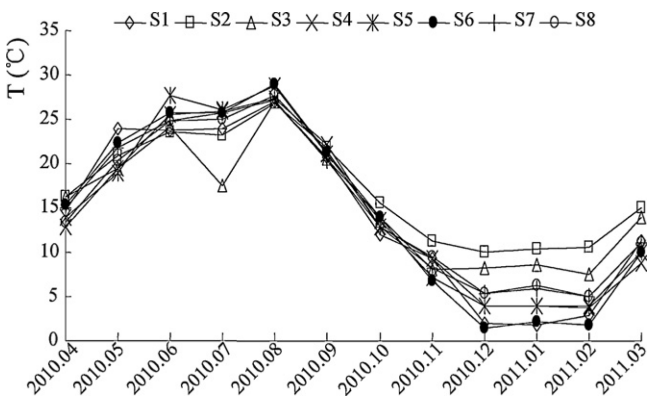

Fig. 4. Changes in water body temperature at different sample sites during the assessment period.

(Fig. 3). These findings illustrate the relationship between water temperature and biodiversity of zooplankton (Dumont, 1983).

The biodiversity also increased in March 2011, being higher at most sites (around 1.5–2.5 except for severely polluted sites) than in April 2010 (less than 1). Construction of the CW contributed to improvement in water quality as demonstrated by the biodiversity measured at S6. The biodiversity at S7 showed a similar trend to that at S6, even though S7 received water from the Wenyuhe (S6) and Qinghe (S3) rivers. These findings indicated that the biodiversity of S7 more closely reflected that at S6 with higher biodiversity compared with S3 with lower biodiversity. Since the biodiversity from the Qinghe River (S3) was lower (Fig. 3), the increase in biodiversity was mainly due to communities existing in inflow from the Wenyuhe River (S6). Although the biodiversity of S8 was lower than that of S6 and S7, all three sites showed similar trends (Fig. 3).

Changes in temperature also differed across sample sites, especially in winter (Fig. 4). As stated above, the overall biodiversity was higher in summer and lower in winter (Fig. 3). Temperature differences among sample sites were observed in winter from November, 2010 to March, 2011 (Fig. 4). Due to the discharge of treated water from the Qinghe Sewage Treatment Plant, the temperature at S2 and S3 was about 10 °C higher during winter. The warm water also influenced the downstream sites. At S7 and S8, which received water from the Qinghe River, the temperature was around 5 °C during winter. This temperature was higher than that at sites that did not receive water from the sewage treatment plant via S2 (i.e., S1, S4, S5 and S6), where the temperature ranged from 2 to 3 °C. Since the temperature at S2 and S3 was higher than in S1 in the Qinghe River, the discharge of treated water affected the macrozooplankton communities, inducing a decrease of biodiversity (Fig. 4). The minimum H' values at S2 were around 0.5 during winter. At S6, the temperature decreased to about 2 °C, and H' was around 1.5. The temperature at S3, which was downstream of S2, was around 10 °C, and H' increased to 1.0 at this site. During winter, the biodiversity at S2 and S3 remained similar to that observed in previous months. In addition, H' at S7 changed similarly to that at S6. These findings suggested that the biodiversity at S7, which receives water from both S3 and S6 (Fig. 1), was mainly influenced by macrozooplankton residing in inflows from S6.

In summary, changes in the macrozooplankton biodiversity at the survey area suggested that the CW in the Longdaohe River contributed somewhat to improvement of the water quality, that the discharge of treated water from the Qinghe Sewage Treatment Plant in Qinghe River affected H' at S2 and S3, and that the biodiversity of S7 was influenced by S6, even though the site receives water from both S3 and S6.

Fig. 5 shows the water quality assessment based on the physicochemical parameters at different sites according to Eqs. (4)–(6). Pb/n at most sites was higher than 10, and even reached more than 20 at some sites (such as S1). The physicochemical parameters suggested that the overall water quality was severely poor (serious pollution), except at S6. Similar to the assessments based on biodiversity, the Pb/n in S2 and S3 was almost at the minimal range of water quality throughout the sampling period (around 20). The minimal biodiversity values were also observed at these sites (Fig. 3). The physicochemical assessments at other sites including S1, S4, S5 and S8 were also generally similar to the biodiversity values (Fig. 3).

At S6, the Pb/n decreased from 6.32 in April to 2.65 in June (Fig. 5), supporting the improvement of water quality after construction of the CW. However, during the rainy season (July to September), the Pb/n increased from 2.85 in July to 10.94 in September. After the rainy season, Pb/n decreased to less than 2 (less than 1 from January to March 2011). The overall lower values in Pb/n during the assessment period suggested that the water body in S6 was improved to slight pollution due to construction of the CW (Table 3). The biodiversity at S6 indicated slight pollution (α or β-Type pollution, around 2) from January to March 2011 (Table 2). In contrast, the Pb/n at this site showed a greater improvement of water quality during this period.

It is worth noting that biodiversity and physicochemical values were not in accord in some cases. In the biodiversity assessment, the trends of S7 and S8 were almost the same as at S6. While biodiversity presented better water quality (slight pollution; β-Type Pollution according to Table 2) with values higher than 2.0 except during winter, Pb/n indicated severely polluted state with values around 15–20 (i.e., severely polluted higher than 4 according to Table 3). These findings indicated that, although ecological integrity appeared to be better when biodiversity was higher, physicochemical water quality was still in the severely polluted state. This difference was primarily due to the separate functioning of environmental and biological factors in river ecosystems, although the water bodies from S3 and S6 were mixed at S7 (Fig. 1). Physicochemical indices presented the limited state due to contamination by pollutants from S3. In contrast, biodiversity did not reflect the chemical constraints. Instead, diverse macrozooplankton communities were separately established due to macro invertebrates being carried from upstream. These findings indicate that there would be a separate species allowing high biodiversity in river ecosystems, although the physicochemical stress is severe. Overall, these findings indicate that physicochemical and biological indicators may indicate conflicting conditions at the same sample site under field conditions.

Pb/n may be more reliable for evaluation of water quality for conventional drinking since it indicated lower water quality (Fig. 5) when compared with biodiversity (Fig. 3). In addition, biodiversity varied seasonally, while Pb/n values were more consistent with respect to water quality. Biodiversity and Pb/n at S8 showed a similar trend as those at S7. Although the degree was lower, water quality based on biodiversity was higher, while physicochemical indices indicated lower water quality (Figs. 3 and 5).

Due to the different evaluations based on biodiversity and physicochemical assessments comprehensive estimation of water quality should not be based on only one aspect. Although physicochemical assessments showed low similarity, biological functioning may be different based on community compositions. Specifically, ecosystem recovery may be enhanced at sites with high biodiversity, and various ecosystem management policies will be differently selected based on community compositions pertaining to the sample sites. Accordingly, it is necessary to investigate both biological and physicochemical factors in an integrative manner.

In summary, both biodiversity and physicochemical assessments suggested that water quality assessment based on both

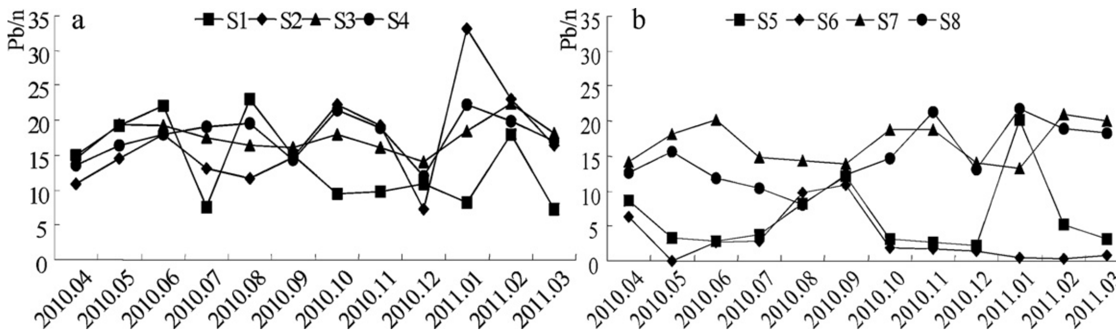

Fig. 5. Changes in Pb/n at different sample sites during the assessment period.

measurements might be the same or similar where there was no evident improvement in quality, such as at S1, S2, S3, S4 and S5, although partial improvement was observed at some sites. Additionally, the physicochemical assessment based on Pb/n should be more acceptable owing to the effects of water temperature from January to March 2011 on the biodiversity assessment H'. Finally, the water quality at some sites (e.g., S7 and S8) that received water from various rivers revealed two types of biological and physicochemical functioning in ecosystems.

3.3. Patterning of water quality indicators

Grouping of water quality indicators including biodiversity and other physicochemical factors was also conducted by SOM training at sites S6 and S7 (Figs. 6 and 7). These two sites were selected because they could reflect the effects of mixing water bodies and showed higher levels of biodiversity relative to other physicochemical factors.

For S6, clustering was primarily observed due to season. Winter and summer samples were grouped in clusters I and III on the left side of the SOM, respectively (Fig. 6a and b). Samples collected in the early period of spring, 2010 were located in cluster IV on the right side of the map. In this case, cluster II accommodated no sample site, since we fixed 4 clusters for training the data for S6.

Profiles of biodiversity and physicochemical factors efficiently characterized the clustered samples in S6 (Fig. 6a). While biodiversity (H') showed a horizontal gradient, most physicochemical factors, including NH_3—N and TN, showed vertical gradients (Fig. 6c). These findings indicated that the effects of biodiversity are somewhat orthogonal (i.e., independent) of the physicochemical factors pertaining to sample site S6 under field conditions, supporting the separate functioning in ecosystems as stated above. It is worth noting that pH was strongly associated with cluster IV, which represented early sampling in spring of 2010. According to a previous study (Yuan et al., 2005; Vohla et al., 2011), even though CWs with steel slag based filter media have good ability for phosphorus removal, they might induce an increase in pH in the first few months of operation. When the pH of water in an aquatic system is greater than 9, it can affect the survival, growth and reproduction of aquatic organisms (Zhuang, 1994)

The physicochemical assessment based on Pb/n suggested that the water quality during the rainy season in summer was worse than in other seasons. The main reason for this might be the nitrogen content in water body, which mainly included NH_3—N and TN (Fig. 6c). During the rainy season, the NH_3—N and TN increased from 0.06 mg/L and 0.05 mg/L to 0.75 mg/L and 25.28 mg/L, respectively, due to source pollution from farms around the Wenyuhe River Basin. EC might be another factor that led to the increase of Pb/n in these months, as it increased from about 500 μS/cm to more than 1000 μS/cm.

Fig. 7 shows the association between the biodiversity assessment and physicochemical assessment of S7 based on the SOM (Fig. 7a and b). At S7, the sampled communities were all grouped into four clusters (Fig. 7a). Grouping occurred according to the vertical gradient, but differences according to season and sample sites were not clearly observed. Clustering appeared to be mainly based on biological and physicochemical indices.

Fig. 7c shows profiles of indices on the SOM in a fashion similar to those shown in Fig. 6c. Similar to the case of S6, biodiversity (H') showed a horizontal gradient, and most of the physicochemical factors including NH_3—N and TN showed vertical gradients (Fig. 7c). These findings also confirmed that the effects of biodiversity

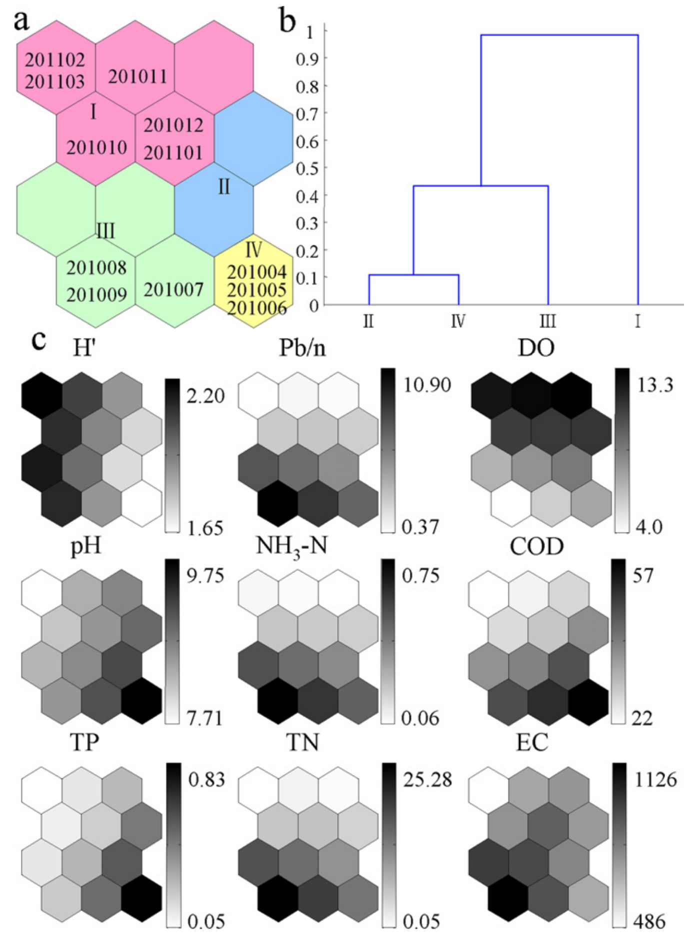

Fig. 6. Profiles of physicochemical factors matching the biodiversity assessment based on the SOM at site S6. (a) The group data analysis of different months with six clusters classified by the SOM; (b) cluster distances according to Ward's linkage method; (c) profiles of H' values visualized on the SOM of different water samples.

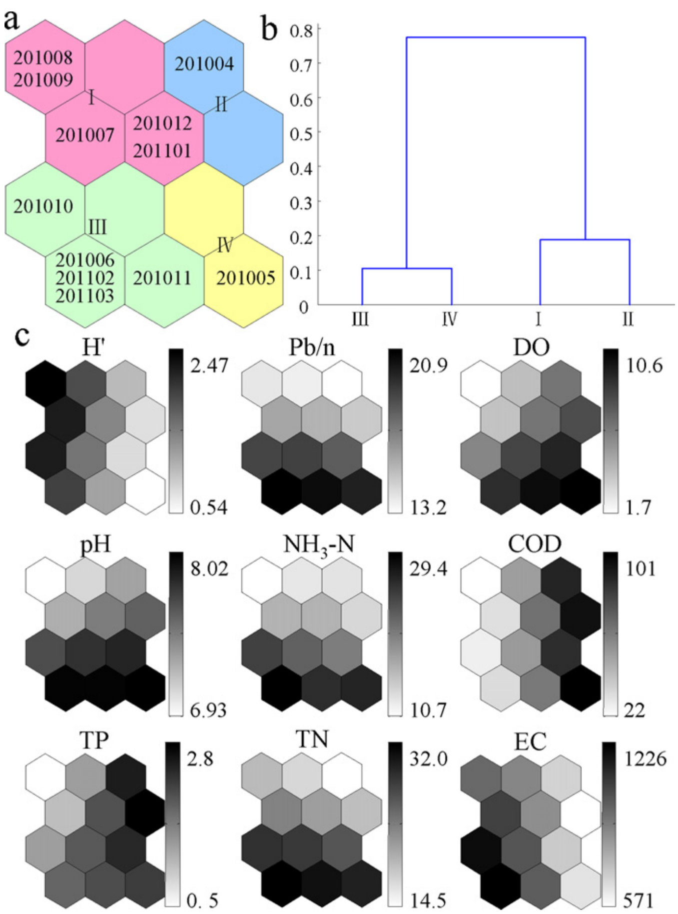

Fig. 7. Profiles of physicochemical factors matching the biodiversity assessment based on the SOM at site S7. (a) Group data analysis of different months with six clusters classified by the SOM; (b) cluster distances according to Ward's linkage method; (c) profiles of H' values visualized on the SOM of different water samples.

are independent of physicochemical factors under field conditions. However, other physicochemical factors such as COD and TP produced a horizontal gradient, the direction of which was opposite to that produced by biodiversity (Fig. 7c).

4. Conclusions

The water quality in the upstream portion of the CW showed no overall improvement during the study period based on the biodiversity and physicochemical assessment. However, the association between biodiversity and physicochemical assessments suggested that the water quality in some sites (e.g., S6) was partially improved with a significant relationship between operation of the CW and improved conditions being observed in the Longdaohe River. In the Qinghe River, the discharge of treated water from the Qinghe Sewage Treatment Plant not only affected the water quality at S2 and S3, but also at S7 and S8. The water imported from both the Longdaohe River (S6) and the Qinghe River (S3) resulted in increased complexity of the water quality at S7 and S8. Nitrogen pollution appeared to be the main reason for the results of the physicochemical assessment based on Pb/n. The physicochemical factors might reflect the water quality more suitably in a conservative sense than the biodiversity assessment results, which were somewhat overestimated and showed greater seasonal variability.

The SOM was shown to be a useful tool for effective water quality assessment and for investigating ecosystem functioning by revealing the association between physicochemical and biological assessments through clustering and visualization. However, water quality should not be evaluated based only on biological or environmental factors, but rather an integrative approach should be adopted to enable sustainable management of water resources.

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