Assessment of aquatic ecological health based on determination of biological community variability of fish and macroinvertebrates in the Weihe River Basin, China

Jintao Wu[1,2], Ruichen Mao[1], Mingyue Li[1], Jun Xia[1,3,4,*], Jingxi Song[1,3,5,*], Dandong Cheng[1], Haotian Sun[1]

[1] Shaanxi Key Laboratory of Earth Surface System and Environmental Carrying Capacity, College of Urban and Environmental Sciences, Northwest University, Xi'an 710127, China

[2] Institute of Qinling Mountains, Northwest University, Xi'an 710127, China

[3] Research Institute for Water Security (RIWS), Wuhan University, Wuhan, 430072, China

[4] Key Laboratory of Water Cycle & Related Land Surface Processes, Institute of Geographic Sciences and Natural Resources Research, Chinese Academy of Sciences, Beijing 100101, China

[5] State Key Laboratory of Soil Erosion and Dryland Farming on the Loess Plateau, Institute of Soil and Water Conservation, CAS & MWR, Yangling 712100, China

Corresponding Authors: xiajun666@whu.edu.cn (J. Xia); jinxisong@nwu.edu.cn (J. Song)

Abstract: A healthy aquatic ecosystem plays an important role in the operation of nature and the survival of human beings. Understanding the mechanism of its interaction with the habitat process is conducive to formulating targeted ecological recovery plans. In this study, fish and macroinvertebrates were collected from 49 investigation sites in the Weihe River basin, China, during periods of the summer and the autumn of 2017. Cluster analysis and canonical correlation analysis (CCA) were used to analyze the similarity of community distribution of fish and macroinvertebrates and their response to environmental variables. The biological integrity index of fish (F-IBI) and benthic-macroinvertebrate (B-IBI) was introduced to evaluate the aquatic ecological health. The results showed that fish communities were more coherent than macroinvertebrate communities. The distinguished response to ecological factors was identified for fish and macroinvertebrates. The ecological factors of total nitrogen, conductivity and river width have significant effects on both fish and macroinvertebrate communities. In addition, the fish community was significantly influenced by chlorine, fluorine, pH and flow velocity, while the macroinvertebrate community was significantly influenced by bicarbonate and water depth. The differences in community structure and response to ecological factors between communities were amplified in their environmental quality scores. Although F-IBI and B-IBI tend to be consistent temporally, the correlation is not significant. B-IBI showed decreasing gradient of ecological health status in the downstream area, while F-IBI tended to be different across river systems, which further illustrated the differences in the response of fish and macroinvertebrates to environmental variables.

Keywords: aquatic ecological health; fish; macroinvertebrate; community; ecological factor

1. Introduction

Healthy rivers are very important to natural ecosystems and social systems. They can provide necessary ecological services for human survival and social development (Luo et al. 2018; Song et al. 2019). At the same time, human activities directly or indirectly affect all aspects of the river ecosystem, such as hydrological conditions (Zhao and Yang 2009), pollutant load (Pinto et al. 2013), and habitat attributes (Maddock 1999), resulting in the continuous deterioration of the river ecosystem.

In terms of reflecting the health status of a river ecosystem, aquatic organisms are not only affected by pollution load, but also by habitat and hydrological conditions (Johnson and Ringler, 2014; Arman et al. 2019). Hence aquatic organisms can be employed to obtain valuable information about the comprehensive impact of various pressure sources and environmental variables. Fish and macroinvertebrates are the most commonly used communities for ecological health assessment (Karr 1981; Kleynhans 1999; Hawkins et al. 2000; Karr and Kimberling 2003). The fish community is located at the top of the river food chain, and the life cycle of fish is longer than other organisms. Involving migration and spawning within its life cycle, fish is more dependent on the habitat, therefore, can reflect the overall health of river environment (Huo et al. 2012; Zhao et al. 2019). On the contrary, macroinvertebrate habitats are relatively fixed, which makes macroinvertebrate communities more valuable in reflecting water quality (Kimmel and Argent 2016). Macroinvertebrates are involved in the decomposition and circulation of nutrients in aquatic ecosystems (Brittain and Eikeland 1988; Gibbins et al. 2007). They are sensitive to low-level pollutants and can be used for early detection of river degradation (Compin and Céréghino 2003).

According to the description of community consistency, different communities with high community consistency have similar responses to environmental factors and pressure sources (Infante et al. 2009). However, the responses to environmental factors is different for fish and macroinvertebrates. In different basins, the driving factors that impact the distribution of different biomes are uneven, which leads to different health conditions of aquatic ecosystems reflected by different biomes (Johnson and Ringler 2014; Yin et al. 2015; Kimmel and Argent 2016; Liu et al. 2017; Zhao et al. 2019). Therefore, there is an urgent need to study the response relationship between biological communities and ecological factors in order to determine the main factors driving the dependence of aquatic ecosystems. An effective way to determine the main factors affecting the distribution of biological communities is by comparing the structural characteristics of the main biological communities and analyzing the hydrology and water quality (Zhao et al., 2019). This can improve the understanding of the spatiotemporal heterogeneity of aquatic ecosystem health and improve the efficiency of river water ecological environment protection, which is particularly important for the Weihe River Basin.

The Weihe River Basin, the largest tributary of the Yellow River, is one of the most developed areas in Northwest China, and also a core area of development in Northwest China (Song et al. 2018). The rapid population growth and industrialization in this region was a accompanied by soaring resource consumption and exacerbating environmental pollution, exceeding the self-purification and recovery capacity of natural ecosystem, and posing threat to river ecosystem (Wu et al. 2014; Song et al., 2015a). In order to effectively protect and restore the Weihe ecosystem, it is necessary to have a comprehensive understanding of the spatiotemporal heterogeneity and clarify the driving factors of river water ecology.

The objectives of this study are to analyze the spatial and temporal distribution rules of the health of aquatic organisms in the Weihe River basin according to the data set of aquatic organisms and environmental factors, and to find the key factors that promote the spatial and temporal changes of the health of biological communities. IBI index were used to assess the health of fish and benthic invertebrates. Through canonical correlation analysis, the driving effects of hydrological and water quality factors on different biological communities were analyzed. The research results can provide basis and support for the management planning and protection of the Weihe River ecosystem.

2. Material and methods

2.1 Study area

The largest tributary of the Yellow River, the Weihe River Basin (WRB) is located between 104~110°E and 34~37°N, covering an area of 13.43 × 104 km^2, with an average annual runoff of 7.57 billion m^3 (Chang et al. 2015; Wang et al. 2019). The WRB can be divided into two parts, the north being the Loess Plateau, and the south being the Guanzhong Plain area (Li et al. 2013). The entire river mainly consists of three water systems, Weihe River system (WRS), Jing River system (JRS), and Beiluo River system (BRS) and their tributaries (Fig. 1). JRS and BRS flow through the Loess Plateau, carrying a lot of sediment in the flow, which is the main source of sediment concentration of the Weihe River (Song et al., 2016). Land use type is mainly arable land and forest, and most of the urban land is concentrated in the lower reaches of the Weihe River.

Selecting tributary junctions, representative sections, and sources of mainstreams and major tributaries, a total of 49 sampling points were selected in this study, including 20 sites (sites W1-W20) in the WRS, 18 sites (sites J1-J18) in the JRS, and 11 sites (sites B1-B11) in the BRS (Fig. 1). Field surveys were conducted in summer and autumn of the WRB in 2017 to collect and record water quality, hydrological and biological data for the assessment of the health status of WRB aquatic ecosystem and analyze its spatial heterogeneity and temporal variation.

2.2 Data

2.2.1 Aquatic organisms

2.3.3.1 Fish

Electrofishing was conducted at each site within a 200-meter stream reach to collect fish samples representative of the resident assemblage (using backpack electrofisher). The sampling period was 30min/site. For river segments with depth greater than 1.5 m, nets are used for fishing, with travel distances of no more than 100 m between each sampling point (Barbour et al. 1999). Most fish that were identified in the survey were returned to their original locations to reduce habitat disturbance. Any fish not readily identified in the field were preserved in 10% formalin for later identification in the laboratory (Song et al., 2015b).

2.3.3.2 Macroinvertebrates

Macroinvertebrate sampling was performed using an aquatic D-frame (30cm × 30cm area, 420 μm mesh) kick net. The operator randomly collected 3 parallel samples at the sampling location (100m range), and then the collection was poured on a white porcelain plate to remove debris (Johnson and Ringler 2014). Some rocks were picked up and rubbed to dislodge any attached macroinvertebrates. The collected samples were stored in 75% alcohol and sent to the laboratory. In the laboratory, all macroinvertebrates were classified and identified manually on white porcelain plates, identified mostly to species level under dissecting microscope (Nikon smz 800). Quality control was assured by having all staff training in the collection of fish and macroinvertebrate samples. Fish and macroinvertebrates were identified by trained staff using taxonomic keys, and identifications were verified by experts as required.

2.2.2 *Water quality and hydrology*

Water sampling and sample preservation were conducted in accordance with Chinese national experimental standards (Ding 2016). On the basis of stable flow conditions without significant rainfall (b10mm/48h), 49 sampling sites in the WRB were sampled on site. Water samples were collected in a plastic bottle (200 mL) from 50 cm below the water surface. Dissolved oxygen (DO, mg/L) and electrical conductivity (EC, μs/cm) were measured in situ in the middle of a flowing stream stretch by a portable water quality analyzer (YSI 6600V2, YSI Company, USA). All water quality probes were calibrated prior to use at the sampling sites. Water samples were collected at each sampling site, placed in a low-temperature incubator, and brought back to the laboratory within 48 hours for determination of ammonia nitrogen (NH_4^+), total nitrogen (TN), total phosphorus (TP), fluoride (F^-), and chloride (Cl^-) (Table 1). Storage, preservation and chemical analyses followed the national standard methods of examining water and wastewater in China (Standard methods for the examination of Water and wastewater, 2002).

River width data was measured by laser range finder (Trupulse 200); current velocity was obtained by using a portable flow meter (MGG/KL-DCB); transparency measured by Secchi disk (Gao et al. 2011); water depth was acquired by using a terrain probe; Global Positioning System (GPS Etrex 201X) was used to measure latitude, longitude and elevation information.

2.3 Methods

To comprehensively evaluate the health of aquatic ecosystems in the WRB, we used the biological integrity index based on fish and macroinvertebrate communities. Ordination plot analysis was then used to identify the drivers of fish and macroinvertebrates.

2.3.1 The biological integrity index

The biological integrity index (IBI) is an indicator used to assess ecosystem health, which quantitatively describes the relationship between biological and abiotic factors to select the most sensitive biological index of environmental disturbance, including indices reflecting community composition and abundance, nutritional structure and tolerance (Liao and Huang 2013). In this study, F-IBI and B-IBI was used to assess the health status of fish communities and macroinvertebrates, respectively, in the WRB.

In this study, 22 fish biological characteristic parameters (Table 2) and 26 macroinvertebrate characteristic parameters (Table 3) sensitive to environmental change are selected, following which the biological indices are screened by analyzing their distribution range, discriminating ability and correlation among indices. The screening of distribution range requires that the indices be distributed over enough samples. The box plots were used for screening of the discriminating ability. According to Barbour's method (Barbour et al. 1999), compare the overlap of reference points and damage points within the range of IQ (interquartile ranges) and assign different values: no overlap, IQ=3; partially overlapped but both median values were outside the box range of each other, IQ=2; only one median value is within the box range, IQ=1; each median value is within the box range of the other, IQ=0. Pearson correlation analysis was performed on pairs of parameters screened by box plots. Only one of the two indices with a correlation coefficient of ≥0.8 was used (Zhao et al., 2019).

The specific method calculating the IBI was as follows (Zhao et al. 2019). For the biological indices whose values decrease when the disturbance increases, the best expected value is 95%, and the index value of each point is equal to the index value of the sample point divided by the index value of 95% (Eqs. 1). For the biological indices whose values increase when the disturbance increases, the best expected value is 5% quantile, which is standardized according to Eqs. 2.

$$P_{ij} = O_{ij}/S_{j95} \times 100 \qquad (1)$$

$$P_{ij} = {}^{maxO_{ij} - O_{ij}}\Big/_{maxO_{ij} - S_{i5}} \times 100 \qquad (2)$$

where P_{ij} is the standardized value of the i-th index at the j-th sampling point; O_{ij} is the measured value; S_{j95} and S_{j5} are the 95 percentage and 5 percentage points of the i-th index at all sampling points, respectively; $maxO_{ij}$ is the maximum value of the i-th index at all sampling points.

The average value of the standardized index of each parameter was taken as The IBI value:

$$IBI_j = \frac{1}{m}\sum_{i=1}^{m} P_{ij} \qquad (3)$$

The 25th percentile of IBI score at the reference site was used as a criterion for health assessment. Points with scores greater than the 25th percentile of IBI are considered healthy. The remaining scores were divided into three equal parts: fair, poor and very poor.

2.3.2 Statistical method

Based on the results of IBI, ordination plot analysis was used to determine the factors that drive the health of biological communities. Species data were analyzed using detrended correspondence analysis (DCA) to select an appropriate model. In this study, the gradient length of the first ordination axis was larger than 4. Therefore, canonical correspondence analysis (CCA) was used to analyze the response relationship between the species community and ecological factors (Lou et al., 2018; Zhao et al., 2019). Canonical correlation analysis (CCA) is a multivariate statistical analysis method that reflects the overall correlation between two groups of indices by using the correlation between pairs of comprehensive variables (Hardoon and Shawe-Taylor 2004; Biswas et al. 2014).

When performing canonical correspondence analysis on aquatic community data and environmental factor data, all the data were converted by lg (x + 1), and then 9999 Monte Carlo test (MCT) was performed (Wu et al. 2012). An environmental factor with a P value of less than 0.05 was regarded as an environmental factor that significantly affected the fish community. CCA were carried out by CANOCO (Version 5.0).

3. Result

3.1 Characteristics of fish and macroinvertebrates community

We collected a total of 3206 individual fish from 49 sites in the WRB, 1502 and 1699 in the summer and autumn, respectively. A total of 38 species of fish, belonging to 4 orders and 9 families, were identified in the two seasons. In the summer, the five species of fish with the largest collection amount accounted for 59% of the total abundance, namely *Triplophysa minxineanensis, Triplophysa dalaica, Gobio coriparoides, Pseudorasbora parva* and *Opsariichthys bidens*. In autumn, the five species with the largest number of collected fish accounted for 61% of the total abundance, namely *Hemiculter leuciclus*, *Christian carp, Abbottina rivularis, Opsariichthys bidens* and *Gobio* *coriparoides*.

The cluster analysis results divided the sampling points of the WRB into several clusters according to the type of fish community, and these clusters were continuous in regional location (Fig. 2). W17, J19 and L12 exhibited a high degree of similarity, which belongs to the same cluster. W5, W6, W8, W9 and W10 also showed a high degree of similarity, therefore belonging to the same cluster. The cluster containing W11, W13, W14 and W18 is far away from the cluster containing W5, W6, W8, W9 and W10, indicating significant differences. A dam between W10 and W11 impedes fish migration may be caused the differences. Overall, the 49 sampling sites can be divided into 3 clusters. Cluster 1 has 35 sampling sites located in the JRS, downstream of BRS and downstream of WRS. Cluster 2 has 8 sampling sites, located upstream of WRS and midstream and upstream of BRS. Cluster 3 has 6 sampling sites, which are located in the middle of WRS and downstream of BRS.

We collected a total of 5011 individual macroinvertebrates from 49 sites in the WRB, 3551 and 1460 in the summer and autumn, respectively. And 105 species of macroinvertebrates, belonging to 7 classes, 17 orders and 55 families were identified. Chironomidae, Baetidae and Tubificidae were dominant, accounting for 85.95% of the abundance of individuals collected in the summer and 83.54% of the abundance of individuals collected in the autumn.

Macroinvertebrates communities are divided into two clusters in Fig. 3. Cluster 1 has 4 sampling sites (L5, L6, L7, and L10) located in the BRS, and the other sampling sites belong to cluster 2. Unlike fish, large invertebrate communities are less affected by dams

(W2, W13, W15). The communities of different water systems are highly similar (W3, L2, J7), while the communities of upstream and downstream (W6, W16) are highly similar, indicating that the habitat dependence of macroinvertebrate communities is more pronounced.

3.2 The distribution patterns of aquatic ecological health

In this study, IBI assessment method was used to evaluate the health of fish aquatic ecosystem in the WRB. In the summer of 2017, 4.3% of the sites were healthy, 43.5% fair, 30.4% poor and 21.7% very poor. The aquatic ecosystem health of WRS was best, with an average F-IBI of 47.19, and the BRS had the worst aquatic ecosystem health ($\overline{\mathrm{IBI}} = 41.8$). In the autumn of 2017, 4.1% of the sites were in a healthy state, 24.5% fair, 51% poor and 20.4% very poor. The aquatic ecosystem health of JRS ($\overline{\mathrm{IBI}} = 40.4$) was better than that of the main stream of Weihe River ($\overline{\mathrm{IBI}} = 34.5$), with the aquatic ecosystem health of the BRS remaining the worst ($\overline{\mathrm{IBI}} = 24.8$).

From the perspective of spatial distribution, F-IBI score in the two periods were quite different (P=0.886), and the results in summer were better than in autumn (Fig. 5a). The scores of the three water systems were significantly different. The aquatic ecosystem health in the upper reaches of the BRS was the worst in both periods and improved in the lower reaches (Fig. 4b).

We used the IBI assessment method to evaluate the macroinvertebrate ecosystem health of the WRB for the two periods (Table 3). In the summer of 2017, 4.1% of the sites were in a healthy state, 44.9% were fair, 34.7% were poor and 16.3% were very poor (Fig. 6a). In the autumn of 2017, 2.2% of the sites were were healthy, 32.6% fair, 60.9% poor and 4.3% very poor (Fig. 6b). The aquatic ecological health based on macroinvertebrate communities were similar in the two seasons (P=0.048).The aquatic ecosystem health of JRS ($\overline{\mathrm{IBI}}_{normal} = 60.6, \overline{\mathrm{IBI}}_{high} = 47.9$) was better than that of the main stream of WRS and BRS in both seasons, and the aquatic ecosystem health of BRS was the worst ($\overline{\mathrm{IBI}}_{normal} = 46.0, \overline{\mathrm{IBI}}_{high} = 40.0$). From the perspective of spatial distribution, it is obviously that B-IBI scores were better in summer than in autumn (Fig. 5b). Macroinvertebrate indices showed declining ecological health from upstream to downstream (Fig. 6b).

3.3 Environmental variables driving variations in aquatic community health

Based on water quality and hydrological data collected in the summer and the autumn of 2017, CCA was used to study the response of fish and macroinvertebrates communities to ecological factors (Fig. 7). All the biotic data were transformed into relative abundance (0–100%) before analysis. Due to the large number of species, a single taxon selected for analysis must occur in > 1 sample with a total relative abundance if over 0.5% when all samples were added. The required number of taxa used in the analysis is 17 for fish, and 33 for macroinvertebrates.

Monte-Carlo screening with $P < 0.05$ showed that width ($P = 0.002$), TN ($P = 0.01$), pH ($P = 0.016$) and velocity ($P = 0.036$) significantly affected fish distribution in the summer. However, in autumn, the ecological factors that significantly impacted the distribution of fish were pH ($P = 0.024$), TP ($P = 0.032$), TN ($P = 0.012$) and velocity ($P = 0.002$). Meanwhile, fish have different responses to ecological factors. Most fish are negatively correlated with TN, and the predominant fish affected were *Rhynchocypris* *lagowskii, Misgurnus anguillicaudatus* and *Abbottina rivularis.* The relative abundance of *Rhynchocypris lagowskii* and *Gobio coriparoides* showed a significant negative correlation with pH. And the relative abundance of *Pseudochemist dispar* showed a significant positive correlation with river width. The relative abundance of *Triplophysa sellaefer*, *Rhynchocypris lagowskii* and *Triplophysa strauchii* showed a significant positive correlation with velocity.

For macroinvertebrates in the WRB, results of CCA showed that the principal ecological factors affecting the distribution of macroinvertebrate species in the summer were COND ($P = 0.008$), TN ($P = 0.008$), water depth ($P = 0.016$) and HCO_3^- ($P = 0.042$), while the principal ecological factors in the autumn were TN ($P = 0.02$), width ($P = 0.026$) and water depth ($P = 0.042$). *Cricotopus (I.) trifasciatus* was the most sensitive macroinvertebrate to TN, showing negative correlation in both seasons, and sensitive to the change of COND. Besides most Chironomidae, *Physa fontinalis* were sensitive to bicarbonate. And Chironomidae was sensitive to river width and water depth.

The most important hydrological factor affecting fish and macroinvertebrate species was river width. In addition, velocity was an important driving factor of fish community, while the dependence of invertebrate species was mainly driven by water depth. The water quality factor that predominantly impacts fish and macroinvertebrate species was TN. Furthermore, fish community dependence was driven by Cl, COND and pH, while invertebrate species dependence was driven by conductivity and bicarbonate.

4. Discussion

4.1 Comparison of fish and macroinvertebrate communities

According to the biological sample data, the community structure of fish and macroinvertebrates in the WRB are significantly different. Compared with the macroinvertebrate community, the fish community is more coherent, which is caused by the difference of living habits (Hajisamae et al. 2003). Some fish need to move in a the water system due to their periodic life activities (migratory spawning) (Karr 1981; Hering et al. 2006). There are also some fishes with strict environmental requirements, whose life activities are limited by ecological factors. *Brachymystax lenok* are found only at the source of the JRS. The artificially constructed dams also prevent the free movement of fish in the watershed (Gehrke et al. 2002), and the type of fish community structure in the upper and lower reaches of the Baojixia Reservoir in the middle reaches of the Weihe River had a significant difference (Fig. 2).

Compared to fish, the connectivity of macroinvertebrate communities is easily disrupted by virtue of their relative staticness (Kimmel and Argent 2016). In most of the continuous sampling points in the WRB, the long-distance distribution of community structure type is rarely found. The high similarity of macroinvertebrate community in the upper and lower reaches of Weihe River indicates that the classification and functional structure of macroinvertebrate community in the natural river is similar. The impact of artificial structures is time-limited and the habitat can be restored after a period of time (Heino et al., 2007). The response of macroinvertebrate communities to ecological factors shows the dependence of their communities on ecological factors.

4.2 Responses of fish and macroinvertebrate species to ecological factors

Our results indicate that fish and macroinvertebrate communities respond differently to environmental variables. Canonical correlation analysis showed that both fish and macroinvertebrate communities responded to ecological factors, but showed differences in the response process. Previous studies have shown that hydrological factors play an important role in the life activities of fish and macroinvertebrates (Morita 2009; Baki et al. 2013). This study determined that river width and current velocity are important hydrological factors affecting fish community structure, with water depth having a more significant effect on macroinvertebrates.

Responses to water quality parameters also differ between fish and macroinvertebrates. Fish communities have significant responses to chloride and fluoride ions in water, while conductivity and bicarbonate have significant effects on macroinvertebrate communities, and are positively related to dominant families of Chironomididae. Previous studies have found that conductivity has a negative effect on the abundance of macroinvertebrates (Bêche and Statzner 2009). Yildiz et al. (2010) investigated Kucuk Menderes coastal wetland area and found that bicarbonate was the main factor affecting the structure of macroinvertebrate communities. TN has a strong influence on the distribution of fish and macroinvertebrate communities (Liu et al. 2016). TN negatively correlates with the abundance of fish community distribution, growth and reproduction of fish communities (Wang et al. 2016). With the increase of nitrogen concentration, the relative abundance of tolerant species of macroinvertebrates will increase. In the WRB, due to the use of nitrogenous fertilizers by farmland and the discharge of industrial wastewater (Su et al. 2019), the TN content is higher than other water quality indicators, making it the most important ecological factor affecting the

Weihe aquatic ecosystem.

4.3 Assessment of aquatic ecological health from different biotic communities

The differences in communities and responses to the environment between fish and macroinvertebrates are magnified in their respective environmental quality scores. The assessment results based on macroinvertebrates are better than the assessment results based on fish in the Weihe River basin, though the overall assessment structure of the two biological communities showed a consistent trend in time dynamics, that is, the aquatic ecological health status of the WRB was better in summer than in autumn. The IBI based on the fish community shows that the upper reaches of the Beiluo River are the most severely damaged area in the WRB, while the IBI based on macroinvertebrate communities shows that the ecological damage is most severe in the downstream areas of the WRB, which indicated that the biological assessment results of a single community are biased towards the habitat suitability of the community (Kimmel & Argent, 2016). The severity of ecological damage usually increases downstream. Macroinvertebrate indices show declining ecological health along the stream longitudinally, while fish-based indices tend to reveal differences among water systems.

In the downstream areas of the Weihe River, where urban areas are relatively concentrated, fish-based assessments indicate good ecological health, while macroinvertebrates show the opposite. This demonstrates that in densely populated areas, different biomes respond differently to human activity (Fore and Wisseman 1996; Bejder et al. 2009). The assessment results of macroinvertebrates fluctuated greatly between two seasons. The health status is better in summer and worse in autumn. Fish assessment results fluctuated less over time, suggesting that the upper communities of the food chain are more stable than the lower communities (Pim and Lawton 1977; Neutel et al. 2002).

The assessment results of different communities in the same area show some differences. It is emphasized that the biological assessment results of a single community are biased towards the habitat suitability of the community (Johnson and Ringler 2014; Kimmel and Argent 2016). Due to the habitat preference of different biological communities, the assessment results of individual communities cannot objectively reflect the health status of ecosystems (Lamouroux et al. 1999; Chalfoun and Martin 2007). Therefore, a comprehensive assessment based on multiple communities is needed for the aquatic ecosystem to accurately reflect the health status of the aquatic ecosystem.

5. Conclusions

This study collected large-scale sampling data and analyzed the intrinsic relationship between the aquatic community (fish and macroinvertebrates) and ecological factors in summer and autumn in the WRB in 2017. The findings are as follows.

(1) In the field investigation of two seasons, 38 species of fish were found, belonging to 4 orders and 9 families, and 105 species of macroinvertebrates belonging to 7 classes, 17 orders and 55 families. Total nitrogen, conductivity and river width have significant effects on both fish and macroinvertebrate communities. In addition, the ecological factors that have a significant impact on the fish community include chlorine, fluorine, pH and flow rate, while the ecological factors that have a significant impact on the macroinvertebrate community include bicarbonate and water depth.

(2) Both F-IBI and B-IBI are higher in the summer than in the autumn, showing

consistency in time, while they are quite different in spatial dynamics. The results of aquatic ecological health assessment based on benthic invertebrates are better than that based on fish.

(3) The spatial heterogeneity of aquatic ecological health assessment based on fish and macroinvertebrate communities is related to the ecological factors that affect their community distribution. It reflects the dependence of biological community on ecological factors, which shows that it is difficult to draw a clear conclusion on the health status of aquatic ecosystem based on the assessment results of single biological community.

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Ecological factors	Average		SD	
	Summer	Autumn	Summer	Autumn
NH_4-N（mg/L）	0.31	0.82	0.17	1.41
TN（mg/L）	5.30	5.71	0.84	4.76
TP（mg/L）	0.15	0.23	0.29	0.30
Cl（mg/L）	281.20	117.52	229.02	122.10
COND (μs/cm)	1366.55	790.87	1044.57	395.72
DO（mg/L）	8.57	8.60	1.88	1.55
S（mg/L）	78.18	106.89	39.93	38.55
NO_2-N（mg/L）	0.04	1.49	0.05	0.72
NO_3-N（mg/L）	1.71	4.98	1.05	3.03
CO_3^{2-} (mg/L)	2.63	3.72	3.08	3.03
HCO_3^-(mg/L)	221.28	234.80	47.56	44.70
pH	9.27	9.26	0.46	0.72

Table. 1 Water quality data in the Weihe River basin from summer to autumn in 2017.

Attribute classification	Parameter index	Response to interference
Species composition and abundance	Species of fish	Decrease
	Pielou uniformity index	Decrease
	Shannon-Wiener diversity index	Decrease
	Percentage of catfish	Decrease
	Percentage of cypriniformes	Decrease
	Percentage of perciformes	Decrease
	Percentage of carp	Decrease
	Percentage of loach	Decrease
	Percentage of pelagic fish	Decrease
	Percentage of middle and lower layers	Increase
	Percentage of bottom fish	Decrease
Nutritional structure	Percentage of carnivorous fish	Decrease
	Percentage of omnivorous fish	Increase
	Percentage of herbivorous fish	Decrease
	Percentage of filter feeding fish	Decrease
Reproductive co-location group	Percentage of sticky egg fish	Decrease
	Percentage of sinking egg fish	Decrease
	Percentage of floating egg fish	Decrease
Tolerance	**Percentage of sensitive fish**	Decrease
	Percentage of tolerant fish	Increase
Number and distribution of fish	**Number of individuals**	Decrease
	Percentage of exotic fish	Decrease

Table. 2 The assessment index system of F-IBI (The bold indicators are the screened assessment indeies). Appendix A lists the classification information of the fish evaluation indicators.

Attribute classification	Parameter index	Response to interference
Community richness	Total number of macroinvertebrate taxa	Decrease
	Number of EPT taxa	Decrease
	Number of aquatic insect taxa	Decrease
	Number of crustacean and mollusc taxa	Decrease
	Chironomid classification unit	Decrease
Proportion of species	**Shannon-Wiener index**	Decrease
	Individual relative abundance of dominant taxa	Decrease
	Individual relative abundance of the top 3 dominant taxa	Decrease
	Relative abundance of Trichoptera individuals	Decrease
	Relative abundance of Ephemeroptera individuals	Decrease
	Relative abundance of Tubifex individuals	Increase
	Relative abundance of Pleoptera individuals	Decrease
	Relative abundance of chironomids	Decrease
	Relative abundance of individuals in crustaceans and mollusks	Increase
	Relative abundance of other Diptera and non-insect populations	Decrease
Nutritional grade index	Relative abundance of shredders	Decrease
	Relative abundance of collector-gathers	Increase
	Relative abundance of collector-filterers	Increase
	Relative abundance of scrapers	Decrease
	Relative abundance of predator individuals	Decrease
Tolerance	**Relative abundance of individuals in sensitive groups**	Decrease
	Relative abundance of individuals with stain-tolerant groups	Increase
	Sensitive group classification unit number	Decrease
	Tolerant group classification unit number	Increase
Habitat quality	BMWP score	Decrease

Table. 3 The assessment index system of B-IBI (The bold indicators are the screened assessment indices). The sensitive groups and stain-tolerant groups were divided according to tolerance values (Appendix B). The biological monitoring working party (BMWP) score references a revision by Leng et al (2016).

Appendix A. List of Fish collected in the survey with feeding, tolerance and origin.

Scientific Name	Summer	Autumn	Feeding	Tolerance	Origin	Height
Cyprinidae						
Opsariichthys bidens	150	148	Carnivore	Sensitive	Native	P
Ctenopharyngodon idellus	0	9	Herbivore	Moderate	Native	M
Rhynchocypris lagowskii	107	69	Omnivore	Sensitive	Native	P
Acanthorhodeus macropterus	2	0	Omnivore	Sensitive	Native	P
Rhodeus sinensis Gunther	2	5	Omnivore	Sensitive	Native	P
Rhodeus lighti	0	2	Herbivore	Sensitive	Native	P
Hemiculter bleekeri	32	1	Omnivore	Moderate	Native	P
Hemicculter Leuciclus	45	332	Omnivore	Moderate	Native	P
Pseudorasbora parva	113	98	Omnivore	Tolerant	Native	M
Gnathopogon imberbis	21	24	Carnivore	Sensitive	Native	P
Gobio coriparoides Nichols	156	122	Carnivore	Tolerant	Native	B
Gobio rivuloides	88	30	Omnivore	Tolerant	Native	M
Abbottina rivularis	2	193	Omnivore	Tolerant	Native	M
Cyprinus carpio	15	28	Omnivore	Tolerant	Native	B
Carassius auratus	102	242	Omnivore	Tolerant	Native	M
Squaliobarbus curriculus	2	0	Omnivore	Tolerant	Native	P
Erythroculter ilishaeformis	2	0	Carnivore	Tolerant	Native	P
Cobitidae						
Triplophysa sellaefer	4	71	Omnivore	Tolerant	Native	B
Triplophysa minxianensis	332	50	Omnivore	Tolerant	Native	B
Triplophysa robusta	0	18	Omnivore	Moderate	Native	B
Trilophysa bleekeri	7	3	Omnivore	Tolerant	Native	M
Triplophysa dalaica	135	109	Omnivore	Tolerant	Native	B
Triplophysa dorsonotatus	6	27	Omnivore	Tolerant	Native	B
Cobitis granoei Rendahl	2	2	Omnivore	Moderate	Native	B
Misgurnus anguillicaudatus	91	59	Omnivore	Tolerant	Native	B
Paramisgurnus dabryanus	3	3	Omnivore	Tolerant	Native	B
Cobitis taenia Linnaeus	1	0	Omnivore	Moderate	Native	B
Triplophysa orientalis	1	0	Omnivore	Moderate	Native	B
Siluridae						
Silurus asotus	8	10	Carnivore	Tolerant	Native	B
Bagridae						
Pelteobaggrus nitidus	2	4	Carnivore	Tolerant	Native	M
Gobiidae						
Ctenogobius cliffordpopei	6	7	Carnivore	Tolerant	Native	B
Ctenogobius brunneus	13	3	Carnivore	Sensitive	Native	B
Ctenogobius gymnauehen	0	2	Carnivore	Tolerant	Native	B
Rhinogobius giurinus	37	12	Carnivore	Tolerant	Exotic	B
Eleotridae						
Hypseleotris swinhonis	2	4	Omnivore	Tolerant	Native	B

Cichlidae						
Oreochroms mossambcus	13	1	Omnivore	Tolerant	Native	M
Salmonidae						
Brachymystax lenok	0	8	Carnivore	Sensitive	Native	M
Channidae						
Channa argus	0	3	Carnivore	Tolerant	Native	B
TOTAL ABUNDANCE	1502	1699				

Feeding, tolerance, height and origin sources include Water Resources Department of Shanxi Province and Li et al. (2015).

P pelagic, M middle and lower, B bottom.

Appendix B. List of macroinvertrbrate collected in the survey with tolerance values (TV) and functional feeding group (FFG) designations.

Order	Family	Taxa	Summer	Autumn	FFG	TV
	Ephydridae		2	2	P	7
	Syrphidae		1	0	P	7
		Ilisia sp.	4	1	P	3
	Tipulidae	*Tipuia(Yamatotipula) sp.B*	2	0	P	3
			0	4	P	3
		Hexatoma sp.	3	0	P	3
	Calliphoridae	*ligurriens*	0	1	P	6
	Muscidae	*Fannia scalaris*	0	2	P	8
	Simuliidae		1	0	P	
	Stratiomyidae		0	1	CG	6
	Simuliidae	*Simulium xinbinense*	1	0	P	6
		Psychoda alternate Say	2	8	CG	7
	Psychodidae		27	3	CG	7
		Telmatoscopus albipunctatud	0	1	CG	7
	Dolichopodidae	*Rhaphium sp.*	4	4	CG	6
	Stratiomyidae		0	2	CG	6
	Tabanidae	*Hybomitra hirticeps*	6	9	CG	6
		Cricotopus (C.) bicinctus	12	1	CG	6
		Procladius choreus	13	0	CG	8
Diptera		*Conchapelopia sp.*	54	26	CG	5
		Orthocladius yagashimaensis	1	0	CG	6
		Polypedilum paraviceps	48	53	CG	6
		Cricotopus (I.) trifasciatus	46	5	CG	6
		Brillia flavifrons (Johannsen)	3	0	CG	5
		Cryptochironomus defectus	1	0	CG	6
		Rheopelopia maculipennis	2	0	CG	3
		Chironomus riparius Meigen	1	7	CG	5
	Chironomidae	*Polypedilum nubeculosum*	1	2	CG	6
			2	0	CG	6
		Cricotopus (I.) sylvestris	8	0	CG	6
		Cricotopus (C.) trifascia	1	1	CG	5
		Cricotopus (C.) triannulatus	25	2	CG	5
		Cricotopus (C.) albiforceps	15	1	CG	6
		Chironomus pallidivittatus	20	125	CG	6
		Cryptochironomus rostratus	32	0	CG	6
		Ablabesmyia phatta	123	51	CG	5
		Procladius sp. C	45	0	CG	5
		Chironomus salinarius	139	37	CG	5
		Rheotanytarsus sp. A	1	0	CG	3

		Rheotanytarsus sp. B	5	6	CG	3
		Acalcarella sp.A	54	0	CG	3
		Rheocricotopus fuscipes	15	0	CG	4
		Chironomus anthracinus	6	1	CG	6
		Chironomus flaviplumus	1	0	CG	8
		Orthocladius makabensis	3	0	CG	
		Prarcladius alpicola	7	0	CG	4
		Sympotthastia takatensis	14	0	CG	4
		Rheocricotopus chalybeatus	35	14	CG	5
		Harnischia fuscimana Kieffer	2	0	CG	6
		Macropelopia paranebulosa	7	0	CG	5
		Polypedilum laetum	0	110	CG	5
		Tanypus punctipennis	0	2	CG	
		Antocha bifida Alexander	0	5	CG	3
		Polypedilum nubifer	0	1	CG	
		Parakiefferiella torutata	0	1	CG	3
	Chrysomelidae	*Galerucella sp.*	7	0	P	4
	Hydrophilidae		9	2	P	3
	Haliplidae		1	2	MH	6
	Heteroceridae		1	0	CG	3
	Elmidae		3	1	CG	3
	Staphylinidae		12	1	CG	3
Coleoptera	Dryopidae		0	3	CG	3
		Dytiscus sp.	29	5	P	6
		Agabus japonicas	3	0	P	5
			6	5	P	5
	Dytiscidae	*Laccophilus difficilis*	2	0	P	5
		Rhantus suturalis	59	0	P	5
		Nebrioporus hostilis	39	27	P	5
		Cybister sp.	0	4	P	5
Trichoptera	Hydropsychidae	*Cheumatopsyche brevilineata*	132	142	MH	6
		Hydropsyche sp.	48	6	MH	6
	Phryganeidae	*Hydroptila sp.*	0	1	P	5
	Plataspididae	*Gerris lacusteris Linne*	1	0	P	5
	Naucoridae		1	0	P	5
Hemiptera	Saldidae	*Saldula saltatoria*	0	1	P	5
	Corixidae	*Micronecta guttata Matsumura*	93	7	P	7
		Sigara substriata Uhler	33	45	P	7
	Belostomatidae	*Diplonychus rusticus*	5	0	P	6
Plecoptera	Perlodidae	*Megarcys ochracea Klapalek*	1	0	P	3
		Oyamia sp.	0	1	P	3
		Ephemera nigroptera	14	8	CG	8
Ephemeroptera	Baetidae	*Baetis vaillanti*	910	99	CG	8
		Baetiella sp.	0	6	CG	5

		Cloeon dipterum	2	0	CG	5
	Siphlonuridae	*Ameletusmontanus Imanishi*	9	15	CG	3
	Palingeniidae	*Anagenesiaparadoxa Buldovskij*	0	2	CG	3
			0	4	CG	2
			0	16	CG	2
	Heptageniidae	*Ecdyonurus yoshidae Takahashi*	0	20	CG	2
		Cinygmina	0	5	CG	2
		Cinygmina	0	1	CG	2
	Potamanthidae	*Rhoenanthus magnificus Ulmer*	0	1	CG	1
		Cercion sexlineatum (Selys)	1	0	P	5
	Coenagrionidae	*Cercion plagiosum Needham*	1	1	P	5
		Ischnura labata Needham	1	0	P	5
	Aeshnidae	*Anax nigrofasciatus Oguma*	1	0	P	3
		Sympetrum infuscotum	1	0	P	3
Odonata	Corduliidae		1	0	P	3
	Libellulidae	*Sympetrum flaveolum*	1	3	P	6
		Deielia phaon Selys	1	1	P	6
	Gomphidae	*Davidius sp.*	1	0	P	3
		Anisogomphus maacki	9	4	P	3
	Macromiidae		1	0	P	3
Amphipoda	Anisogammaridae	*Anisogammarus sp.*	11	0	CG	6
	Gammaridae		4	1	CG	6
	Cambaridae	*Cambarus clarkii (Girard)*	11	0	SH	6
		Macrobrachium nipponense	38	19	SH	3
Decapoda	Palaemonidae	*E.modestus*	0	28	SH	3
			0	1	SH	
	Atyidae	*Neocaridina denticulata*	0	134	SH	3
		Limnodrilus hoffmeisteri	2	0	CG	10
		Limnodrilus	674	190	CG	10
Tubificida	Tubificidae	*Tubifex sinicus*	229	19	CG	10
		Branchiura sowerbyi	0	2	CG	10
		Tubifex	0	34	CG	10
		Limnodrilus claparedeianus	7	0	CG	10
Arhynchobdellida	Salifidae	*Barbronia weberi*	1	0	P	6
Herpobdellida	Erpobdellidae	*Herpobdella octoculata*	59	6	P	6
Gnathobdellida	Hirudinidae	*Whitmania pigra Whitman*	1	5	P	5
Rhynchobdellida	Glossiphoniidae	*Glossiphonia lata Oka*	4	2	P	5
	Physidae	*Physa fontinalis*	145	23	SC	5
	Planorbidae	*Gyraulus compressus*	1	13	SC	5
Basommatophora		*Gyraulus albus*	6	16	SC	5
	Lymnaeidae	*Radix lagotis*	4	9	SC	5
		Radix ovata	120	23	SC	5
Mesogastropoda	Viviparidae	*Bellamya purificata*	0	3	SC	5
Unionoida	Unionidae	*Anodonta woodiana Lea*	0	4	CF	5

TOTAL ABUNDANCE	3551	1460

TV and FFG sources include Barbour et al. (1999), Xing et al. (2016) and Wang and Yang (2004) (Local sources are preferred).

P Predator, MH macrophyte/herbivore, CG collector/gatherer, SH shredder, CF collector/filterer, SC scraper.

Figure captions

Figure. 1 Sampling sites in Weihe River Basin, China. A Digital Elevation Model (DEM) is shown, with low to high elevation as indicated in the legend.

Figure. 2 Cluster analysis of fish communities in the Weihe River basin.

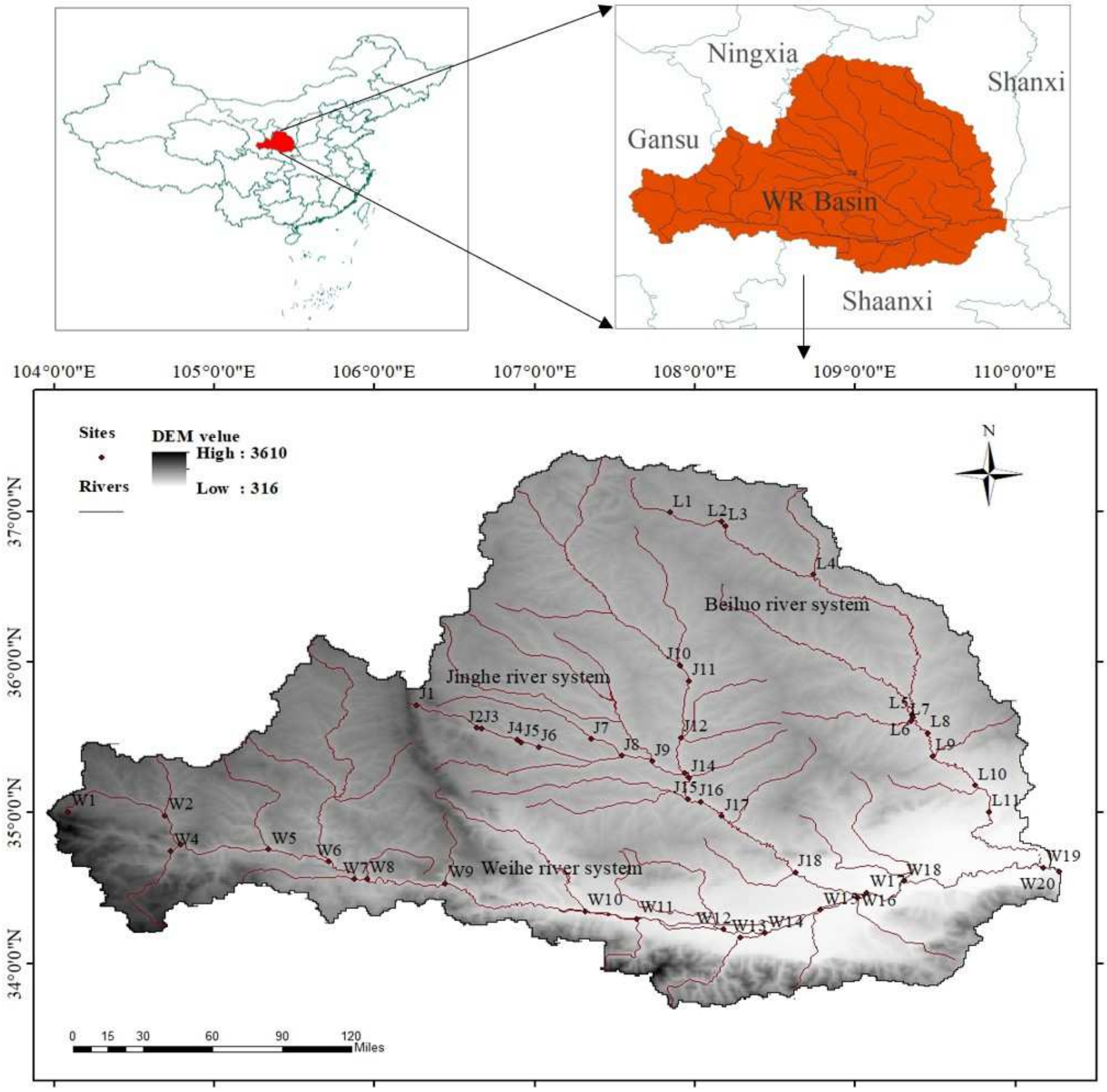
Figure. 3 Cluster analysis of macroinvertebrate communities in the Weihe River basin.

Figure. 4 Spatial distribution of F-IBI in the Weihe river basin: (a) F-IBI in the summer; (b) F-IBI in the autumn.

Figure. 5 Temporal dynamics boxplots of aquatic ecosystem health in the Weihe River basin: (a) F-IBI; (b) B-IBI.

Figure. 6 Spatial distribution of B-IBI in the Weihe River basin: (a) B-IBI in the summer; (b) B-IBI in the autumn.

Figure. 7 Ordination plot of CCA on biological communities (fish and macroinvertebrates) and ecological factors in the summer and the autumn.



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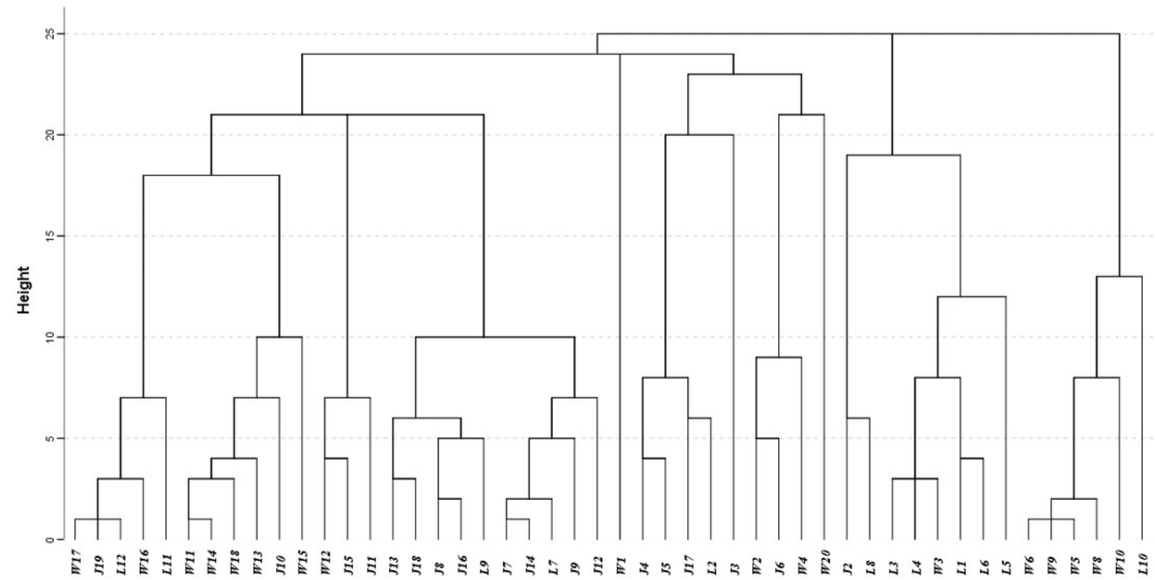


Figure. 2 Cluster analysis of fish communities in the Weihe River basin.

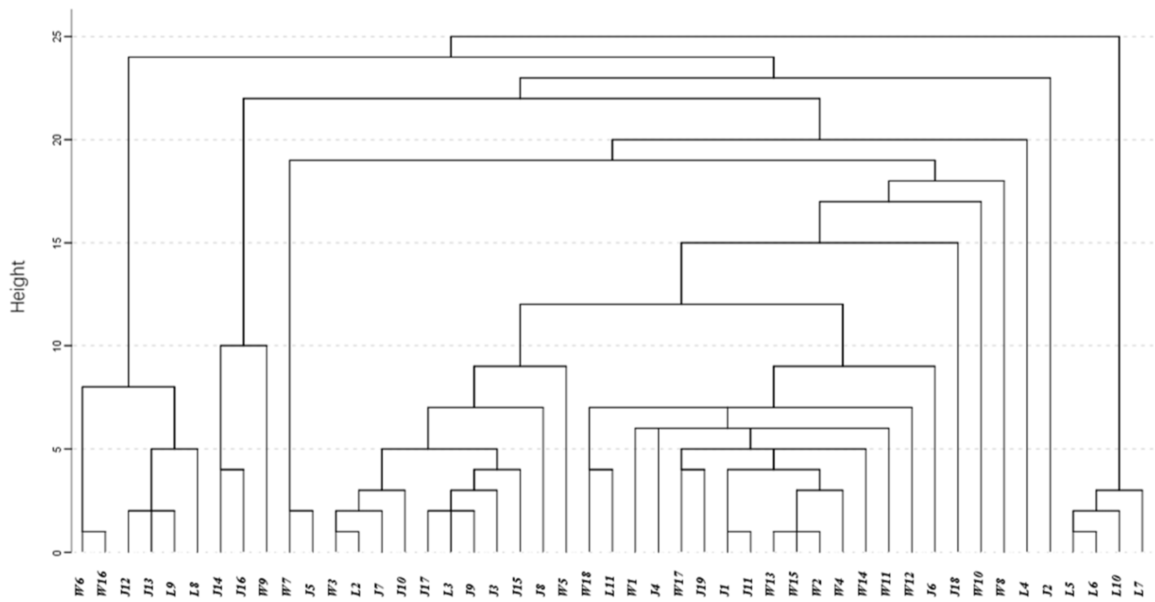
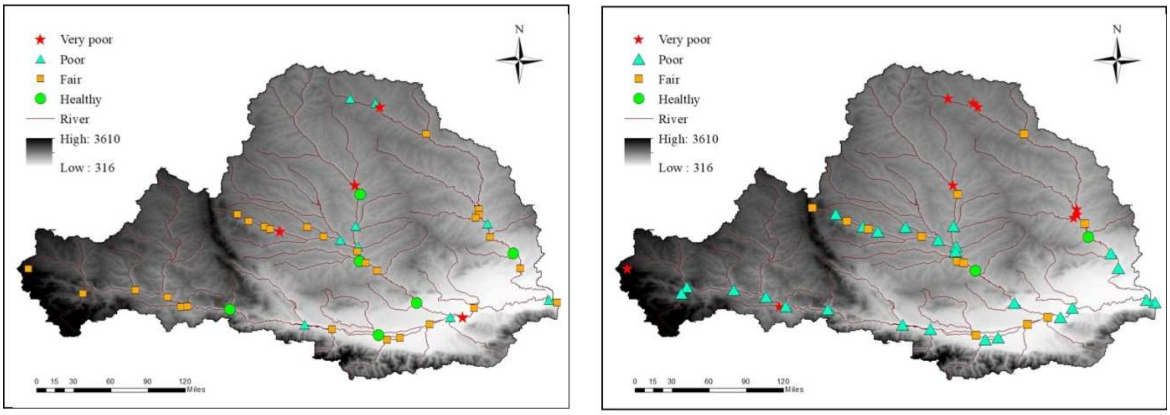


Figure. 3 Cluster analysis of macroinvertebrate communities in the Weihe River basin.



(a) (b)

Figure. 4 Spatial distribution of F-IBI in the Weihe river basin: (a) F-IBI in the summer; (b) F-IBI in the autumn.

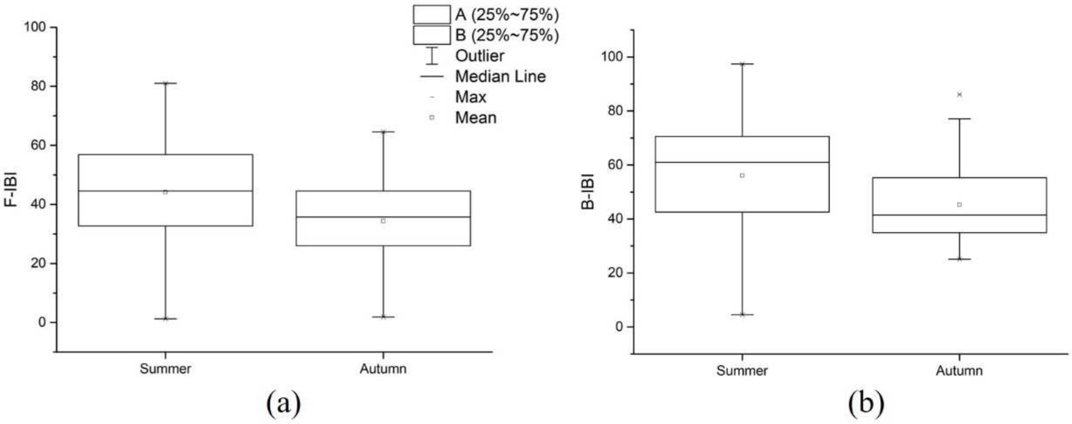
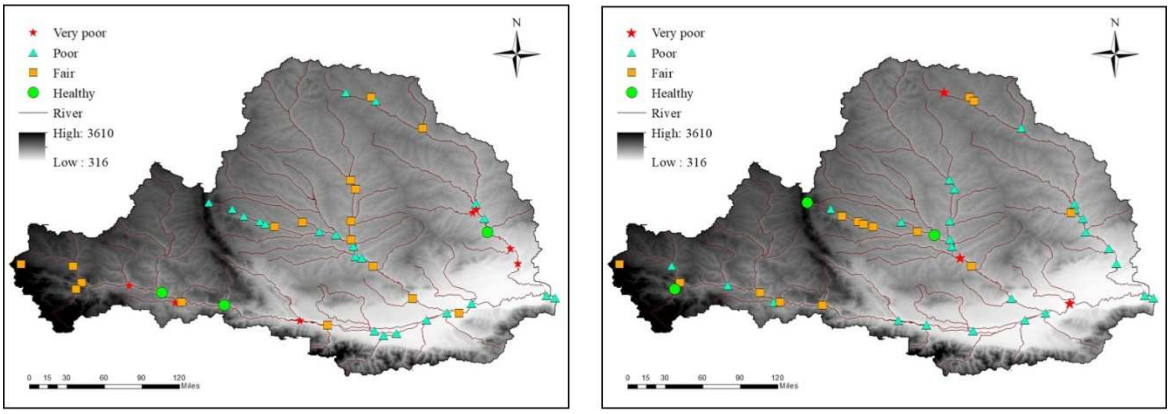


Figure. 5 Temporal dynamics boxplots of aquatic ecosystem health in the Weihe River basin: (a) F-IBI; (b) B-IBI.



(a) (b)

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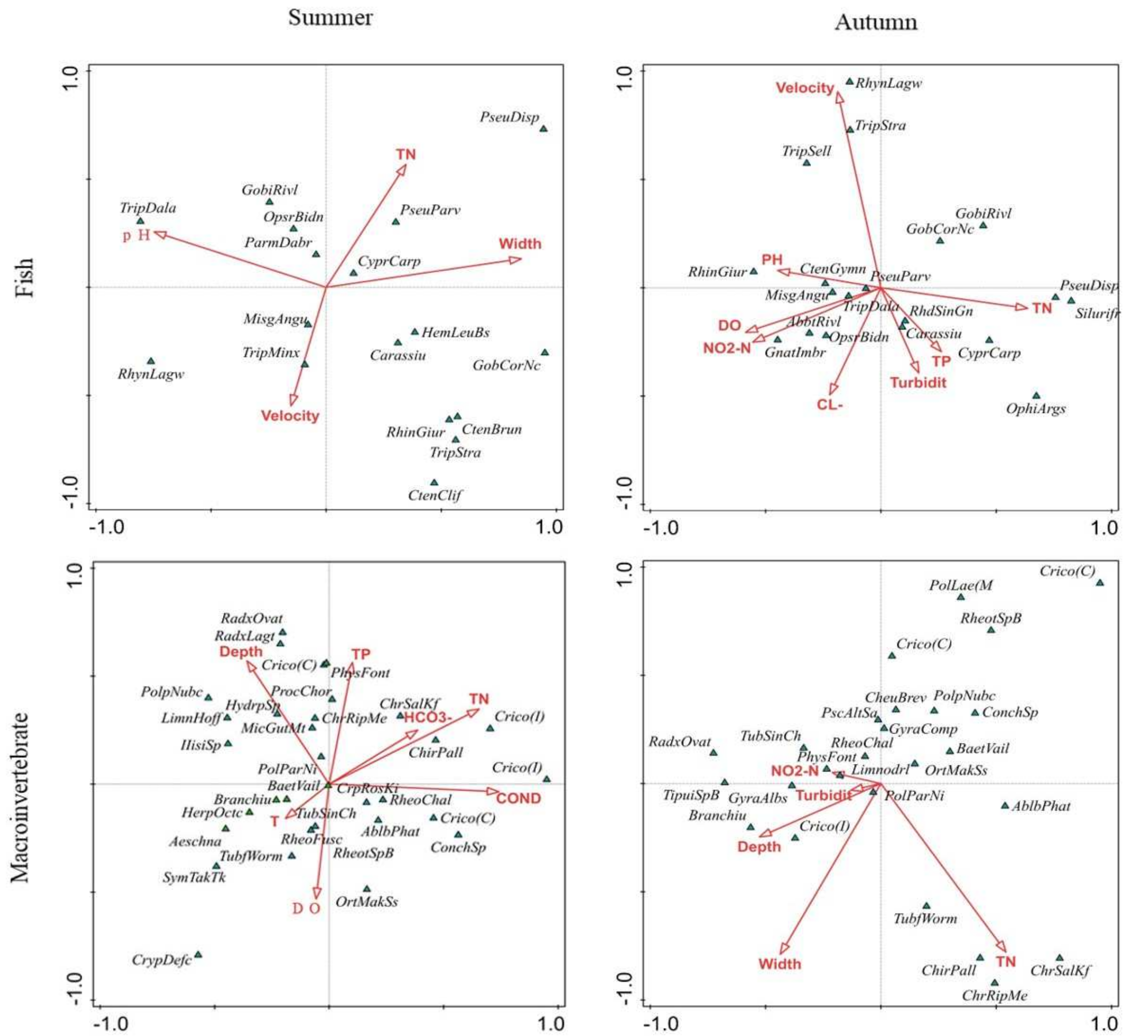

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