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Quantifying the response of aquatic biodiversity to variations in river hydrology and water quality in a healthy water ecology pilot city, China

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Abstract. Prediction and assessment of the effects of habitat change on aquatic biodiversity remain a hot issue globally. This paper developed a practical methodology based on ecosystem models to comprehensively assess the effects of habitat changes on aquatic biodiversity. The partial least-squares (PLS) method was used to analyse the key hydrological and water quality factors influencing riverine aquatic organisms. The biomass of aquatic organisms under undisturbed conditions was simulated using the food web model Ecosim. Based on the relationship between habitat factors variation and biodiversity variation, a multidimensional river hydrology–water quality–biodiversity response model was established. Application and testing of the methodologies in the first water ecology pilot city in China, namely Jinan City, showed that four water quality factors (total phosphorus, total nitrogen, ammonia nitrogen and dissolved oxygen) significantly affected aquatic biodiversity. For hydrological factors, water depth had a strong effect on fish diversity, whereas flow velocity largely affected fish and algal diversity. The application suggested that response model was practical in modelling the effects of habitat variation on biodiversity. It is anticipated that this model will help assess the effects of changes due to climate- and human-induced stress on aquatic ecosystems and provide a scientific basis for river management decisions.

Additional keywords: food web modelling, river ecosystems.

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Introduction

The degradation of aquatic ecosystems and biodiversity loss, resulting from altered hydrological regimes and deteriorated water quality caused by global climate change and intensive human activity, have become major problems during the 21st century (Marzin *et al.* 2014; Zhu *et al.* 2015). For this reason, numerous river ecological restoration projects have been developed around the world (Zhao *et al.* 2014, 2015). However, because the complex interaction between habitat factors and aquatic organisms has not yet been clearly defined, such restoration projects have not proved to be very successful and have led to a substantial waste of resources (Suding *et al.* 2015).

Ecosystems with a rich diversity of species have higher resilience and generally maintain a healthier state when faced with environmental changes (Lavorel *et al.* 1999; Dukes 2001). The coexistence of diverse species facilitates enhanced utilisation of available resources in different ecological niches within the ecosystem and strengthens ecosystem functions and the stability of ecosystem processes; and when ecosystems are affected, higher biodiversity can resist ecosystem changes (e.g. trophic structure, primary productivity, nutrient supply maintenance; Naeem and Li 1997) and maintain ecosystem stability. The relationship between biodiversity and habitat factors (e.g. hydrological and water quality factors) has remained a hot topic in ecological research (Fu *et al.* 2017; Zhao *et al.* 2010).

As an effective tool for exploration of the complex interaction among various riverine habitat factors, ecosystem models can be used for studying the functions and mechanisms of riverine ecosystems, and therefore their biodiversity variation. Moreover, such models are used to conduct quantitative analyses of the key physical, biological and chemical processes, as well as the complex interaction of the various factors (Zheng *et al.* 2012; Li *et al.* 2013). Application of computer-assisted simulations is an important method to understand the general functions of riverine systems and to correctly identify how external disturbances or stress affect them (Steinacher *et al.* 2010; Li *et al.* 2013). However, most previous research has focused on simulation of the ecological dynamics of marine algae and zooplankton and has rarely included the study of zoobenthos and fish (Vera-Mendoza and Salas-de-León 2014; Quinlan *et al.* 2015; Zhao *et al.* 2018*a*). This is not conducive to understanding the overall biological condition of riverine ecosystems. In addition, existing studies have focused on the effects of single factors, such as the nutrients nitrogen and phosphorus, on aquatic organisms; few studies have comprehensively considered the effect of multiple factors on aquatic organisms (Sear 1996; Phillips 2003; Sell 2003; Zhao *et al.* 2018*a*). An ecosystem is complex and the stresses from multiple factors on an ecosystem interact with each other and are not simply the sum of individual stresses (Zhu *et al.* 2015). Thus, previous studies are not conducive to a comprehensive explanation of the environmental factors that influence aquatic organisms (Deng *et al.* 2015), which hinders the progress of water ecosystem restoration.

Many types of selection methods can be used to identify biological factors that influence aquatic organisms, such as canonical correlation analysis (CCA) and detrended correspondence analysis (DCA). However, these are semiquantitative methods. In contrast, a newer method involving an indicator of the variable importance in projection (VIP), which is based on partial least squares (PLS), has been used to quantitatively describe the extent to which a given factor can explain biological changes (Chun and Keleş 2010). During recent years, this method has emerged as a selection method for biological driving factors (Rigdon 2016; Khaledian *et al.* 2017). In terms of ecosystem modelling, Ecosim (Ecopath International Initiative, Barcelona, Spain) is a mature food web-based simulation tool that can accurately simulate the biomass of aquatic ecosystems. Ecosim has been used extensively for ecosystem simulation and policy impact assessment, and has become an essential tool used by ecologists to study aquatic ecosystems (Colvin *et al.* 2015; Du *et al.* 2015; Zhao *et al.* 2018*a*).

The objective of this study was to develop a practical methodology to comprehensively assess the effects of changes in the hydrology and water quality of habitats on aquatic biodiversity. The PLS method was used to analyse the key hydrological and water quality factors that influence river organisms. The food web-based Ecosim model was then used to simulate the biomass of aquatic organisms under undisturbed conditions to calculate the corresponding biodiversity (or species richness). The results obtained were compared with the measured aquatic biodiversity to obtain the biodiversity fluctuation caused by stress factors. Furthermore, based on the relationships among variations in hydrology and water quality factors and changes in biodiversity, a multidimensional response model for riverine hydrology–water quality–biodiversity was established. The response model was finally used to predict future variations in aquatic biodiversity based on routinely measured hydrological and water quality factors as a scientific reference for aquatic ecology management and ecosystem remediation strategies.

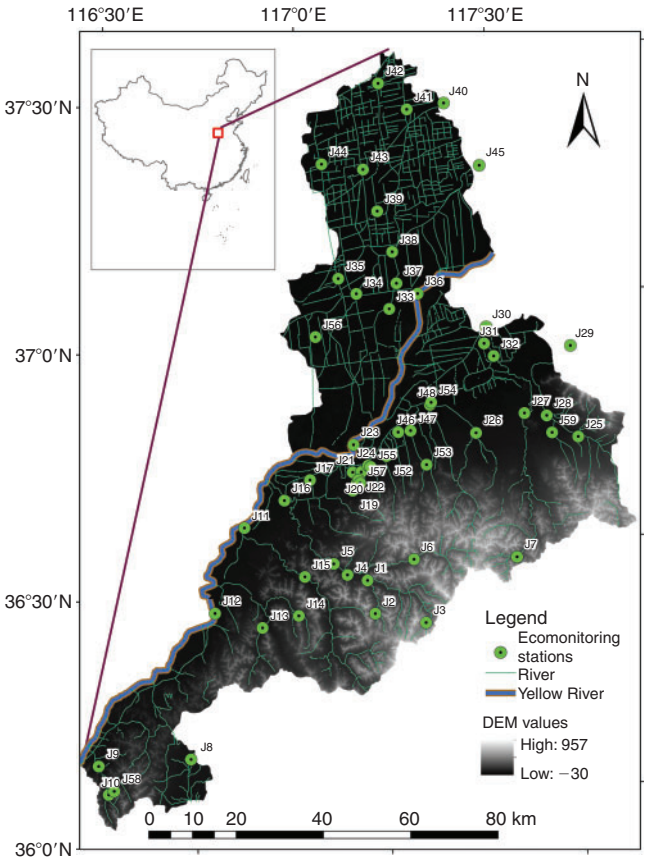

Fig. 1. Study area showing routine hydrology, water quality and river ecosystem monitoring stations (revised from Zhao *et al.* 2018*a*).

Materials and methods

Study area

The city of Jinan (36.0–37.5°N, 116.2–117.7°E) is in eastern China. Bordered by Mount Tai to the south and traversed by the Yellow River, it has a steeper topography in the south than in the north (Fig. 1). The altitude within the area ranges from −30 to 957 m above sea level, with highly contrasting relief. The average annual precipitation is 636 mm, 75% of which falls during high-flow periods. The average annual temperature is 14.3°C. The average monthly temperature is highest during July, ranging from 26.8 to 27.4°C, and lowest during January, ranging from −3.2 to −1.4°C (Cui *et al.* 2009; Zhang *et al.* 2010).

Jinan City represents a typical developing city in China, with an area of 8227 km^2 and a population of 5.69 million (Zhang *et al.* 2007). Owing to the significant role of aquatic biodiversity in riverine ecosystems, the response of aquatic biodiversity to variations in river hydrology and water quality has a high

priority in the maintenance and remediation of the aquatic ecosystem, which, in turn, sustains the development of society and economy. This is especially important for Jinan City, a pilot city for the construction of a water ecological city in China. With rapid industrial development and urbanisation in recent decades, the water resources in Jinan are severely polluted and, through extraction, reduced in quantity (Hong *et al.* 2010). As a result, the river ecosystems are being increasingly threatened and degraded (Hong *et al.* 2010; Zhao *et al.* 2015). To ensure successful ecosystem restoration, river administrators urgently need to know how aquatic biodiversity will respond under changing hydrology and water quality conditions. Policy makers and stakeholders are aware of the need to rehabilitate the river ecosystems in the city, and routine ecomonitoring stations evenly distributed on typical rivers have been established (Fig. 1). Biota, water quality and environmental factors are concurrently measured at these ecomonitoring stations.

Data collection

Between May 2014 and September 2016, 472 samples were collected from routine ecomonitoring sites in Jinan. Biological data (for algae, zooplankton, zoobenthos and fish) and habitat data (for hydrology and water quality) were collected simultaneously.

Hydrological data

The hydrological data collected included flow velocity, stream flow, water depth, river width (Table 1) and river cross-sectional data. Hydrological parameters (including water depth and flow velocity) were routinely monitored. Flow velocity data were acquired by combining an electric wave current meter (Stalker II SVR V1.0, Applied Concept Inc., Richardson, TX, USA) and a traditional current meter (number LS25-1, Nanjing Tiandi Jingye Equipment Co., Ltd, Nanjing, PR China), thereby improving the precision of the measured values. Both water depth and river width were measured. Stream flow was calculated using flow velocity and water depth. The hydrological factors in the city of Jinan are characterised by large spatial variations due to the existence of numerous segments of dry riverbed and the presence of the large Yellow River. The largest values for water depth, flow velocity, stream flow and river width were all recoded at the Yellow River stations.

Water quality data

The water quality data (Table 2) included a total of 36 parameters that were measured following the methods in the National Environmental Quality Standards for Surface Water GB 3838-2002 (see https://www.chinesestandard.net/PDF/English.aspx/GB3838-2002, accessed 28 March 2019), as described previously (Zhao *et al.* 2018*b*). Dissolved oxygen (DO) and pH were measured using a portable water quality tester (PC101; Hach Water Analysis Instrument (Shanghai) Co., Ltd, Shanghai, PR China). An atomic absorption spectrophotometer (M6; ThermoFisher Scientific, Waltham, MA, USA) was used to test for Zn, Mn, Hg, Fe, Cr, Ca, K, Na, Pb, Cd and Cu, whereas an ion chromatograph (DIONEX-600, ThermoFisher Scientific) was used to measure fluoride (F), sulfate (SO_4), chloride (Cl) and nitrate concentrations. A spectrophotometer (DR5000) was used to measure total nitrogen (TN), ammonia nitrogen (NH_4-N), total phosphorus (TP) and hexavalent chromium. An automatic flow injection analyser (SKALAR SAN++, Skalar Analytical B.V., Breda, Netherlands) was used to measure cyanide, volatile phenol and anionic surfactant. Because the levels of 10 heavy metal ions, including copper, zinc and lead, were below the detection limit of the instruments used, they are not listed in Table 2. The range and s.d. of values for the remaining 26 parameters are given in Table 2.

Table 1. Hydrological data

Parameter	Abbreviation	Minimum value	Maximum value	s.d.
Water depth (m)	Dep	0	3.51	0.91
Flow velocity ($m\ s^{-1}$)	Vel	0	1.69	0.35
Stream flow ($m^3\ s^{-1}$)	Flow	0	1110	201.83
River width (m)	Wid	1.50	320	67.13

Aquatic organism data

Data for aquatic organisms were recorded in four hydro-ecological monitoring surveys conducted in 2016.

Algae

The algae collected refer to phytoplankton only. For rivers with depths less than 2 m, 2-L water samples were collected at a depth of 0.5 m, and an extra water sample was collected at a deeper layer if the transparency of the water was very low. For rivers with depths less than 5 m, five water samples were collected from five water layers (at depths of 10, 20, 40, 80 and 100%) and a 2-L hybrid water sample was obtained. For rivers with depths of more than 5 m, water samples were collected from the water layers at intervals of 3–6 m, and a smaller volume of water was collected from the water layers below the epilimnion. Lugol's solution was added to the water samples in a volumetric proportion of 1.5%.

Zooplankton

A water sampler was used to collect a water sample (20–50 L) from each water layer and a 25-μm plankton net was used to filter the water samples. The organisms collected were stored in a 100-mL plastic bottle. The net was placed in the water with the net mouth exposed above the water, and was then shaken to concentrate the zooplankton to the bottom of the net. The valve was opened to transfer the samples collected into the same bottle; these steps were repeated three to five times. Formaldehyde was added in a volumetric proportion of 5% to fix the collected samples.

Zoobenthos

For mountain streams with depths of less than 30 cm or the shallow zones of rivers, a 60-mesh (0.5 × 0.5 m) Sürber net was used to collect samples. For biotopes with a large depth, a 625-cm^2 Peterson bottom sampler was used to collect samples. A 60-mesh screen was used to rinse the bottom mud collected,

Table 2. Water quality data
Another 10 heavy metal ions, including copper, zinc and lead, were below the detection limit and were thus omitted from the table below

Parameter	Abbreviation	Minimum value	Maximum value	s.d.
Anionic surfactants (mg L^{-1})	AS	0	3.48	0.34
Biochemical oxygen demand (mg L^{-1})	BOD	0	57.5	4.83
Calcium ion (mg L^{-1})	Ca	0.99	486	56
Chloride (mg L^{-1})	Cl	0.99	1156	165
Carbonate (mg L^{-1})	CO_3	0	38.5	4.93
Chemical oxygen demand (mg L^{-1})	COD	0	275	23.74
Permanganate index (mg L^{-1})	PI	0.57	71.5	5.84
Electrical conductivity (mS m^{-1})	EC	287	57 756	852
Cyanide (mg L^{-1})	Cya	0	0.02	−1
Dissolved oxygen (mg L^{-1})	DO	0	13.5	2.25
Fluoride (mg L^{-1})	F	0.18	2.51	0.34
Bicarbonate (mg L^{-1})	HCO_3	0	2247	149
Potassium ions (mg L^{-1})	K	0	767	117
Sodium ions (mg L^{-1})	Na	0	109	7.9
Ammonia nitrogen (mg L^{-1})	NH_3-N	0.03	75.8	4.85
Nitrite (mg L^{-1})	NO_2-N	0	1.97	0.25
Nitrate (mg L^{-1})	NO_3-N	0	22	3.34
pH	–	6.9	9.3	0.39
Sulfide (mg L^{-1})	S	0	1.29	0.11
Sulfate (mg L^{-1})	SO_4	0	1046	170
Total alkalinity (mg L^{-1})	TA	0.99	1057	87
Total hardness (mg L^{-1})	TH	0.99	1400	222
Total nitrogen (mg L^{-1})	TN	0.25	80.03	6.07
Total phosphorus (mg L^{-1})	TP	0	8.06	0.68
Turbidity (°)	Turb	0.52	924	103
Volatile phenol (mg L^{-1})	VP	0	0.16	0.22

and all bottom materials were poured into a white ceramic plate for hand picking until no zoobenthos were observed. All zoobenthos were stored in a 1-L wide-mouth bottle and 70% alcohol was added to preserve the samples.

Fish

For rivers with depth less than 1.5 m, an electric fishing apparatus was used to catch the fish. During the sampling process, one person held a 20-pipe ultrasonic electric fishing apparatus on both shoulders to catch the fish, whereas another person collected the samples using a dip net. The samples were collected under different flow velocity and water depth conditions. The sampling time was 30–60 min. For deeper rivers with depths greater than 1.5 m, a boat was used to catch the fish by trawling, with a traveling distance not exceeding 100 m between each sampling point. In addition, fish samples from fishermen (if available) were collected.

Ecosystem modelling

The Ecosim module of Ecopath with Ecosim (EWE) software (see http://ecopath.org/, accessed 27 March 2019) was used to simulate the biomass of aquatic organisms under undisturbed conditions. Ecosim can simulate the dynamic development process of aquatic organisms in aquatic ecosystems. This module has been widely used worldwide for prediction of aquatic ecosystems and is considered by ecologists to be a vital tool for conducting research on aquatic ecosystems (Colvin *et al.* 2015; Zhao *et al.* 2018*a*). Its core equation (Eqn 1) is as follows:

$$dB_i/dt = g_i \sum Q_{ji} - MO_iB_i - F_iB_i - e_iB_i - \sum_{j=1}^{n} Q_{ji} + I_i \quad (1)$$

where B_i represents the biomass of ith biological group, dB_i/dt is the growth rate of the ith biological group calculated on the basis of B_i during the time dt, g_i is the net growth rate (production ÷ consumption), MO_i and F_i are the natural mortality rate and the catching mortality rate of non-predators respectively, e_i is the out-migration rate, I_i is the in-migration rate and Q_{ji} is the consumption rate.

In this study, Ecosim models were established to simulate the development process of the aquatic organisms from May 2014 to September 2016. After comparing the simulated and measured values, the biodiversity variation ΔH owing to stress disturbances was obtained (Fig. 2). The biodiversity was calculated based on the simulated biomass using the Shannon–Wiener diversity index (Shannon 1948; Zhao *et al.* 2012).

Impact analysis using the PLS regression method

PLS regression (PLSR) is a multivariate statistical analytical method that can be used to solve common problems such as multiple correlation of variables during chemical sample analysis and the existence of more explanatory variables than sample

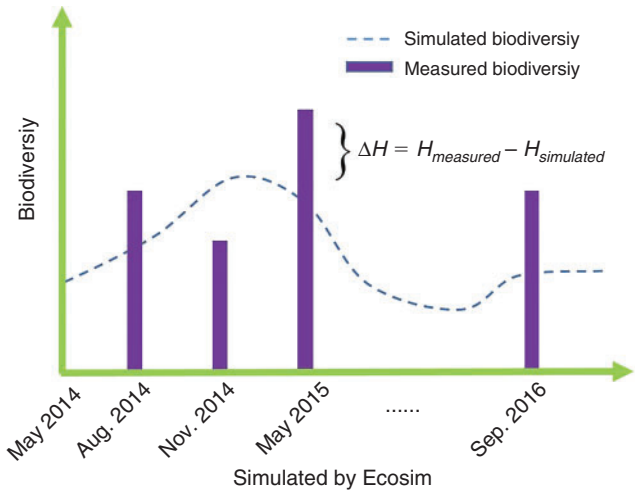

Fig. 2. Variations in aquatic organisms under stress. During the simulation process, which lasted from May 2014 to September 2016, the monthly output data were compared with the measured data to obtain the biological changes owing to stress. *H*, biodiversity.

points (Krishnan *et al.* 2011). PLSR can be used to solve some problems than may not be solved with commonly used multivariate regression methods, and thus has been widely used over the past 30 years (Rigdon 2016; Khaledian *et al.* 2017).

The VIP score is an indicator obtained during the PLSR calculation process that can be used to determine the importance of an independent variable with regard to a dependent variable (Chun and Keleş 2010). A VIP score of the independent variable exceeding 1 indicates that this variable has a more significant role in explaining the dependent variable. When the VIP score ranges from 0.5 to 1, the explanatory role of the independent variable is not obvious; when the VIP value is <0.5, the independent variable has virtually no explanatory role for the dependent variable. The VIP can be calculated using Eqn 2 (Fornell and Larcker 1987):

$$VIP_j = \sqrt{\left(p\sum_{h=1}^{m} R(Y,T_h)w_{hj}^2\right) \div \sum_{h=1}^{m} R(Y,T_h)} \qquad (2)$$

where p is the number of independent variables, m is the number of components extracted from the original variables through PLS, T_h represents the hth component, $R(Y,T_h)$ is the capacity of the T_h component to explain the dependent variable Y, which is equivalent to the square of the correlation coefficient of the two, w_{hj} is the jth component on the w_h axis and w_h is the eigenvector of the matrix:

$$X_{h-1}^T Y_{h-1} Y_{h-1}^T X_{h-1}$$

In this study, the PLSR process and the VIP scores were used to analyse the effects of hydrological and water quality factors on aquatic organisms and to identify which factors had a significant effect on the biomass of aquatic organisms. SPSS-25 software (see https://www.ibm.com/analytics/spss-statistics-software, accessed 27 March 2019) was used for the calculations. The number of latent factors was selected using the principal component analysis method (Wold *et al.* 1987; Abdi and Williams 2010).

Construction of a multidimensional river hydrology–water quality–biodiversity response model

The VIP scores of the PLSR process as described above were used to determine which hydrological and water quality factors were the key driving factors of aquatic organisms. The factors identified were combined with the Ecosim simulation to obtain the biodiversity variation ΔH resulting from stress. The established relationship among the biodiversity variation ΔH, the water quality index variation ΔWQ and the hydrological index variation ΔHY can be illustrated using the multidimensional river hydrology–water quality–biodiversity response model.

Riverine biology is affected by river hydrology and water quality (Zhao *et al.* 2018*b*). Assuming that a functional relationship $\Delta H = g(\Delta WQ)$ exists between the biodiversity variation ΔH and the water quality variation ΔWQ, and a functional relationship $\Delta H = h(\Delta HY)$ exists between the biodiversity variation ΔH and the hydrological index variation ΔHY, where g and h represent the implicit functions respectively, then the aquatic biodiversity variation matrix Z is defined as $Z = F(X,Y)$, where X and Y are the water quality variation matrix and the hydrological index variation matrix (Eqn 3) respectively.

$$Z = \begin{pmatrix} \Delta H_1 \\ \Delta H_2 \\ \Delta H_3 \\ \Delta H_4 \end{pmatrix}, \quad X = \begin{pmatrix} \Delta x_{11} & \Delta x_{12} & \cdots & \Delta x_{1j} \\ \Delta x_{21} & \Delta x_{22} & \cdots & \Delta x_{2j} \\ \vdots & \vdots & \ddots & \vdots \\ \Delta x_{i1} & \Delta x_{i2} & \cdots & \Delta x_{ij} \end{pmatrix}, \quad Y = \begin{pmatrix} \Delta y_{11} & \Delta y_{12} & \cdots & \Delta y_{1j} \\ \Delta y_{21} & \Delta y_{22} & \cdots & \Delta y_{2j} \\ \vdots & \vdots & \ddots & \vdots \\ \Delta y_{i1} & \Delta y_{i2} & \cdots & \Delta y_{ij} \end{pmatrix} \qquad (3)$$

where x_{ij} is the jth index corresponding to the water quality index variation of the ith biodiversity variation, and x_{ij} represents the jth index corresponding to the hydrological index variation of the ith biodiversity variation.

The PLSR method was used in combination with the key driving factors of aquatic biodiversity, which were obtained through the analysis described above. Using Z as the dependent variable and X and Y as the respective independent variables, PLSR analysis of Z was conducted as a function of X and Y to obtain the water quality component $\partial Z/\partial Y$ and the hydrological component $\partial Z/\partial X$. The components obtained represent the overall effects of water quality and hydrological factors on aquatic organisms respectively. After conducting a regression of Z on $\partial Z/\partial X$ and $\partial Z/\partial Y$ in 1stOpt (http://www.7d-soft.com/, accessed 27 March 2019), the relationship of Z with X and Y was determined, thus obtaining the final form of the multidimensional river hydrology–water quality–biodiversity response model.

Results and discussion

On the basis of the PLSR method, the hydrological and water quality factors having a significant effect on the biomass of aquatic organisms were analysed and identified. Then, the biomass of aquatic organisms under undisturbed conditions was compared with the measured biological conditions using the Ecosim simulation to determine the effects of variations in hydrological and water quality factors on biodiversity. Afterwards, on the basis of the hydrological and water quality factors obtained and the biodiversity variations determined, the river hydrology–water quality–biodiversity response relationship was established.

Indicator response analysis: effects of water quality on aquatic biodiversity

Among the 26 parameters, electrical conductivity had the largest s.d. (852), followed by total hardness (s.d. = 222) and sulfates (s.d. = 170), which reflects the large spatial variation and the uneven distribution of these parameters. Cyanide had the smallest s.d. (−1) and was found to be present in extremely low concentrations at all monitoring stations.

The aquatic organism collected included 175 species of algae (belonging to 9 phyla, 10 classes, 18 orders, 28 families and 30 genera), 90 species of zooplankton (belonging to 3 phyla, 4 classes, 11 orders, 16 families and 38 genera), 73 species of zoobenthos (belonging to 3 phyla, 6 classes, 12 orders, 26 families and 50 genera) and 58 species of fish (belonging to 1 phylum, 7 classes, 19 families and 50 genera). The biodiversity of the various species (species richness) is shown in Fig. 3.

Among the four types of aquatic organisms, algae had the highest diversity, followed by fish and zooplankton, with zoobenthos having the lowest diversity (Fig. 3). The diversity indices corresponding to the peaks of the distribution curves successively decreased from algae to zoobenthos; the maximum diversity value of algae was 3.84, with a median of 2.36, whereas the maximum diversity values of fish, zooplankton and zoobenthos were 3.62, 3.53 and 2.97 respectively, with median values of 1.71, 1.77 and 1.25 respectively. Generally, algae had the greatest biodiversity and zoobenthos had the least diversity.

Using the PLSR method, the diversity values for fish, zoobenthos and algae were selected as the dependent variables using SPSS software, and 26 water quality parameters were selected as independent variables to conduct the PLSR analysis. As a result, the VIP scores of each water quality index were obtained with regard to aquatic biodiversity. The calculated VIP scores are given in Table 3.

Table 3 shows that the three types of organisms had various degrees of sensitivity to different water quality parameters. Fish diversity was affected the most by CO_3, DO, HCO_3, NH_3-N, NO_2N, total alkalinity (TA), TN and TP. TP had the greatest effect, with a VIP of 1.82, followed by NO_2-N (VIP = 1.39). Among the eight key factors affecting fish diversity, CO_3 had the lowest VIP of 1.28. Zoobenthos diversity was most strongly affected by CO_3, Ca, electrical conductivity, NH_3-N, total hardness, TP, transparency of water (Tran) and temperature of air (Tem_a). Tran had the highest explanatory effect, with a VIP of 1.83, followed by Tem_a (VIP = 1.79). Among the seven key factors affecting zoobenthos diversity, CO_3 had the lowest

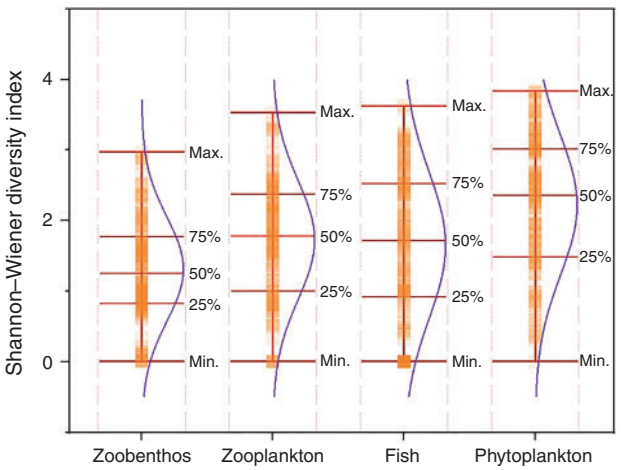

Fig. 3. Aquatic biodiversity data. The horizontal lines (from top to bottom) represent the maximum value, 75% quantile, median, 25% quantile and minimum value; the purple curves represent the data distribution.

VIP of 1.26. Algal diversity was largely affected by CO_3, the permanganate index, Ca, Cl, NO_3-N, TN, Tem_a and Tran. CO_3 had the greatest effect, with a VIP of 1.85, followed by Tran (VIP = 1.82); Tem_a had the lowest effect among the eight key elements affecting algal diversity (VIP = 1.24).

In general, CO_3 had a strong explanatory effect on the biodiversity of all three types of organisms. Its effect on algae was the strongest, which is consistent with the results reported by Mansour *et al.* (2015). Mansour *et al.* (2015) found out that greatest number of taxa of all divisions was recorded when CO_3^{2-} was lowest. In addition, the three types of organisms were also largely affected by nitrogen-related indicators. However, each type of organism responded to a different form of nitrogen. Fish species were sensitive to NH_3-N (VIP = 1.36) and NO_2-N (VIP = 1.78), zoobenthos were sensitive to NH_3-N (VIP = 1.33) and algae were more affected by NO_3-N (VIP = 1.41), reflecting the differences in the use of nitrogen by the various species. A side-by-side comparison showed that fish and zoobenthos were both largely affected by NH_3-N and TP, whereas zoobenthos and algae were affected by Ca, Tem_a and Tran. TN was the only factor that simultaneously affected algae and fish, having a larger effect on the former than the latter. Zhao *et al.* (2018*a*) found that TP, TN, NH_3-N and DO strongly affect aquatic ecosystems, whereas Oliva-Teles (2012) found that TN strongly affects algae and fish, with algae being more sensitive than fish. These results are consistent with those of the present study.

Except for CO_3, the six other factors (Tran, TP, TN, Tem_a, NH_3-N and Ca) had the second strongest explanatory effects on biodiversity for two of the three types of organisms.

Indicator response analysis: effects of hydrological factors on aquatic biodiversity

As indicated in Table 4, water depth and flow velocity had significant effects on fish diversity, with VIP scores of 1.48 and 1.13 respectively, whereas stream flow and river width had lesser effects, with VIP scores <1. Other studies have also

Table 3. Effects of water quality parameters on the biodiversity variable importance in projection

BOD, biochemical oxygen demand; COD, chemical oxygen demand; PI, permanganate index; Tem_a, the temperature of air; Tem_w, the temperature of water; Tran, the transparency of water

Water quality parameter	Fish diversity VIP	Zoobenthic diversity VIP	Algae diversity VIP
Anionic surfactants	0.71	0.95	0.85
BOD	0.72	0.94	0.77
Ca	0.68	1.3	1.32
Cl	0.66	0.6	1.37
CO_3	1.28	1.26	1.85
COD	0.95	0.95	0.87
PI	0.89	0.97	1.37
Electrical conductivity	0.68	1.29	0.86
Dissolved oxygen	1.49	0.96	0.97
Fluoride	0.87	0.85	0.91
HCO_3	1.56	0.61	0.92
K	0.62	0.88	0.81
Na	0.6	0.65	0.54
NH_3-N	1.36	1.33	0.97
NO_2N	1.78	0.94	0.6
NO_3N	0.92	0.33	1.41
pH	0.86	0.99	0.77
Sulfide	0.85	0.93	0.95
SO_4	0.97	0.92	0.83
Total alkalinity	1.62	0.64	0.99
Tem_a	0.38	1.79	1.24
Tem_w	0.32	1	0.61
Total hardness	0.65	1.43	0.51
Total nitrogen	1.4	0.96	1.63
Total phosphorus	1.82	1.35	0.51
Tran	0.74	1.83	1.82

Table 4. Effect of hydrological parameters on the variable importance in projection (VIP) of biodiversity

	Fish diversity VIP	Zoobenthic diversity VIP	Algae diversity VIP
Water depth	1.48	1.17	0.85
Stream flow	0.61	0.85	1.29
Flow velocity	1.13	0.88	1.17
River width	0.75	1.03	0.63

demonstrated that flow velocity is a major factor affecting the structure of fish communities (Karatayev *et al.* 2005; Carpenter *et al.* 2011; Wenger *et al.* 2011). In addition, water depth is the second primary niche axis thought to generate and maintain fish diversity, and ample deep-water (i.e. >100 m) habitat provides ecological opportunity (Muir *et al.* 2016). In the present study, the diversity of zoobenthos was significantly affected by water depth and river width, with VIP scores of 1.17 and 1.03 respectively, whereas stream flow and flow velocity (VIP = 0.85 and 0.88 respectively) had a lesser effect on the zoobenthos. The results of the present study are partially consistent with those reported by C. Zhao, X. Pan, C. Sun, Y. Zhang, Y. Yang, Z. Wang, F. Wang, Z. Zhang, B. Dong and X. Chen (unpubl. data) in a study of zoobenthos in the city of Jinan, who found that water depth and flow velocity were the main hydrological habitat factors for zoobenthic communities. Finally, stream flow and flow velocity significantly affected algal diversity, with VIP scores of 1.29 and 1.17 respectively, whereas river width had a minimal effect on algal diversity, with a VIP of only 0.63. Paerl *et al.* (2016) also demonstrated that water depth plays an important role in the reproduction and survival of zoobenthos, whereas flow velocity and stream flow can affect algal communities and cyanobacterial blooms (Mitrovic *et al.* 2011).

Variations in aquatic biodiversity owing to stressors

Based on the location of the biological samples collected during May 2014, 16 Ecosim model scenes were established to simulate the development process of the aquatic organisms for a period of 3 years, as shown in fig. 4 of Zhao *et al.* (2018*a*).

The variations in biodiversity owing to stresses can be obtained by comparing the Ecosim simulation results to the actual measurements (Fig. 3). The variations in biodiversity were calculated on the basis of $\Delta H = H_{measured} - H_{simulated}$, as given in Tables 5–7.

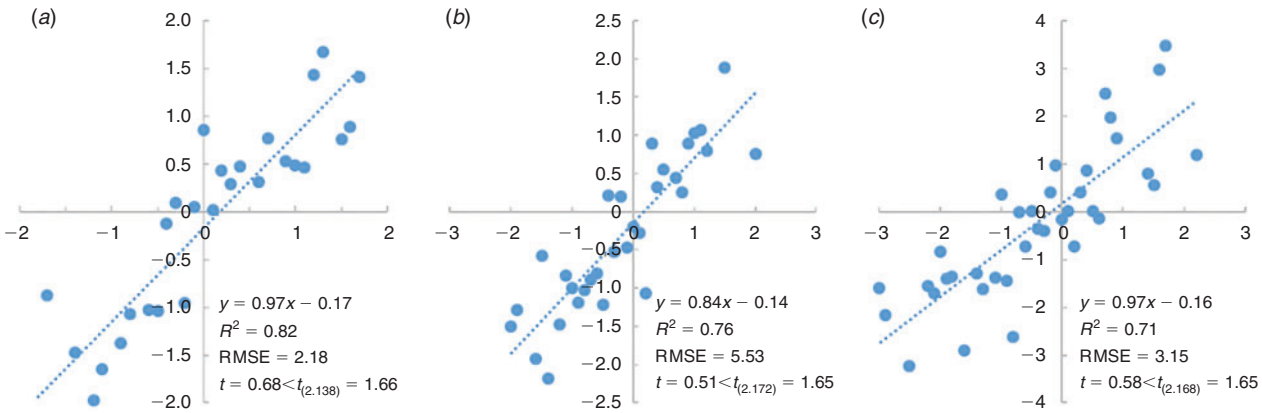

Fig. 4. Comparison of the simulated and measured biodiversity for (*a*) fish, (*b*) zoobenthos and (*c*) algae. RMSE, root mean square error. The *x*-axis represents simulated biodiversity values and *y*-axis represents measured biodiversity values.

Table 5. Variations in fish diversity

Both positive and negative values are valid data. For the location of sampling stations, see Fig. 1. –, no sample was collected

Sampling station	2014		2015			2016		
	August	November	May	September	October	May	July	September
J1	0.31	0.83	0.04	−0.45	−0.36	0.43	0.9	−0.81
J5	−0.87	–	–	0.22	–	–	–	−0.1
J8	–	–	−0.08	–	–	–	–	–
J11	–	−0.51	–	−0.28	0.03	−0.58	−1.76	0.15
J16	–	–	0.84	–	–	–	–	−0.42
J24	0.7	0	0.01	–	1.44	–	–	–
J32	−1.27	–	0.03	−0.88	−0.93	–	–	−0.15
J33	–	–	–	–	−0.16	–	–	1.55
J34	0.08	−0.05	0.4	1.5	0.21	–	–	1.54
J35	–	–	−0.83	–	–	–	–	0.94
J39	–	0.18	1	1.61	1.01	0.62	0.55	1.22
J40	–	−1.41	−0.89	−1.87	−1.21	–	–	−0.52
J41	–	–	1.02	–	−0.25	–	–	0.82
J42	−0.61	−0.33	−0.06	−0.92	−0.19	–	–	–
J44	–	–	−0.3	–	–	–	–	0.54
J45	–	1.03	–	1.23	0.68	–	–	1.14
J48	–	–	−0.15	0.01	0.09	−1.15	–	−0.15

Table 5 shows that under the influence of stressors, such as variations in hydrological conditions and deterioration of water quality, the overall fish diversity measured was lower than the simulated diversity under undisturbed conditions. This was particularly true for stations J40 and J42, where the ΔH values of 100% of valid samples, which were collected between August 2014 and September 2016, were less than zero. This indicates that a stress caused a decrease in fish diversity. Because fish are at the top of the aquatic ecosystem food web (Zhao *et al.* 2018*a*), a decrease in fish biodiversity often means that the aquatic ecosystem is facing a serious threat (Silvano *et al.* 2009; Gallardo *et al.* 2016).

Under the effect of stressors, 45 of 85 valid data samples had negative values, meaning that at these sampling stations the stressors caused a decrease in zoobenthos diversity (Table 6). In contrast, the presence of stressors for the remaining 40 samples increased zoobenthos biodiversity. Benthos diversity is often considered an important indicator of river health (Arthington *et al.* 2010). Acuña *et al.* (2017) also demonstrated that human activity significantly affects the survival of zoobenthos and can endanger entire ecosystems. Therefore, extra attention should be paid by the managing authorities to the stations listed in Table 6, such as J8 and J32, where negative values most often occurred.

As indicated in Table 7, under the influence of stressors, such as variations in hydrological conditions and deterioration of water quality, 45 of 85 valid data samples had negative values, meaning that at these stations the stressors caused a decrease in algal diversity. The large population of blue–green algae may be among the reasons for such a decrease (Paerl *et al.* 2016). At 37 stations, compared with undisturbed conditions, the presence of stressors increased algal diversity.

In summary, the presence of stressors had a significant effect on aquatic organisms, as shown by the fact that, under stress, the biodiversity of all three types of species decreased at half the sampling stations. The effects of stressors were not obvious in mountainous areas, such as J1; however, they were particularly evident in the plains of the northern region, such as at Stations J39, J40 and J48, because of the intensive human activity in that region.

Establishment of a multidimensional river hydrology–water quality–biodiversity response model

Using the PLSR method in combination with the selected hydrological and water quality factors affecting aquatic biodiversity as identified and described above, with biodiversity variation (Z) as the dependent variable and the water quality index variation (X) and hydrological index variation (Y) as respective independent variables, PLSR analysis of Z was conducted as a function of X and Y (Eqn 3) to obtain the water quality component $\partial Z/\partial Y$ and the hydrological component $\partial Z/\partial X$. The relationship of Z was determined with regard to X and Y as follows.

$$Z = p_1 + p_2 \times \frac{\partial Z}{\partial Y} + p_3 \times \frac{\partial Z}{\partial X} + p_4 \times \left(\frac{\partial Z}{\partial X}\right)^2 \qquad (4)$$

Table 6. Variations in zoobenthic diversity

Both positive and negative values are valid data. For the location of sampling stations, see Fig. 1. –, no sample was collected

Sampling station	2014		2015			2016		
	August	November	May	September	October	May	July	September
J1	0.27	−0.03	0.67	0.61	0.5	−0.05	−0.43	1.01
J5	1.98	–	–	1.45	–	–	–	0.67
J8	−0.87	−0.88	−1.22	−1.5	−1.19	–	–	−1.65
J11	–	−1.17	–	1.07	0.39	−0.56	−0.61	0.04
J16	–	–	−0.88	–	–	–	–	−0.8
J24	1.1	0.19	0.94	0.96	1.09	1.18	0.9	0.98
J32	−0.42	0	−0.61	−1.09	−1.05	–	–	−0.2
J33	–	0.95	−0.32	−1.09	−0.13	–	–	0.73
J34	0.28	0.03	−0.72	−1.44	−0.03	–	–	0.03
J35	–	–	−0.91	–	–	–	–	−1.59
J39	–	−1.23	0.07	−0.2	0.46	−0.62	−0.53	−0.81
J40	–	−0.15	1.1	0.23	0.7	–	–	−0.6
J41	–	–	−0.14	−0.39	−1.93	–	–	−1.11
J42	0.85	0.78	0.06	−1.02	0.48	–	–	0.37
J44	–	–	0.65	–	–	–	–	0.38
J45	–	−0.95	–	−1.16	−2.01	–	–	−0.82
J48	–	–	0.33	1.44	−0.72	0.76	−0.52	0.13

Table 7. Variations in algal diversity

Both positive and negative values are valid data. For the location of sampling stations, see Fig. 1. –, no sample was collected

Sampling station	2014		2015			2016		
	August	November	May	September	October	May	July	September
J1	0.5	0.44	0.25	0.43	−1.11	−0.76	−0.46	−0.35
J5	−0.07	–	–	0.71	–	–	–	−0.72
J8	0.56	−0.2	0.73	−0.01	0.39	–	–	−0.93
J11	–	–	–	−1.04	−1.8	−0.68	−2.06	−0.74
J16	–	–	−0.56	–	−2.16	–	–	−0.7
J24	0.24	0.69	0.58	0	0.36	−0.54	−2.29	−0.86
J32	2.2	1.46	1.67	1.35	1.58	–	–	0.23
J33	–	–	−0.22	0.44	0.45	–	–	0.18
J34	1.35	0.89	−0.36	−0.66	−2.13	–	–	0.06
J35	0.1	–	−0.78	–	–	–	–	0.54
J39	–	–	−1.12	−0.41	−0.66	−1.94	−1.34	−3.01
J40	–	–	−1.31	−1.07	−1.41	–	–	−3.34
J41	–	–	1.42	0.27	0.5	–	–	0.53
J42	0.44	0.49	−0.1	0	1.35	–	–	0.81
J44	–	–	−0.18	−2.55	–	–	–	−1.8
J45	–	–	–	0.03	0.39	–	–	0.03
J48	–	–	−0.26	−1.6	−0.9	−0.92	−1.3	−2.9

where p_i is the regression coefficient, which varies for different types of organisms, as indicated in Table 8.

Table 9 shows the regression coefficients of each of the selected hydrological and water quality factors influencing aquatic biodiversity for Eqn 4. It is evident that for algae, the coefficient of TN accounted for a larger proportion among the water quality factors, reflecting the strong effect of TN on algae. For zoobenthos, HCO_3 had the largest coefficient among the water quality factors, whereas for fish species the NO_2N coefficient had the highest absolute value among the water quality factors.

The statistically simulated and measured variations in biodiversity were compared by dividing the measured values into groups using an interval of 0.1 and using the mean value of each group as the group's prediction result, as shown in Fig. 4. It is evident that even though the simulation results were highly scattered, the overall trend was consistent with the actual conditions. The R^2 values of the scatter plot trend lines of the simulated and measured biodiversity values for fish, zoobenthos and algae were 0.82, 0.76 and 0.71 respectively, and their slopes were close to 1.

Levene's test and the *t*-test were performed for the measured and simulated values of the three types of organisms (Table 10). Testing of the original hypothesis H_0, namely states that 'significant differences exist between the two data groups', provided the following results.

Table 8. Regression coefficients for fish, zoobenthos and algae relating to Eqn 4

RMSE, root mean square error

	Fish	Zoobenthos	Algae
p_1	3.41	3.45	2.47
p_2	1.36	0.55	1.95
p_3	0.04	−0.14	−0.68
p_4	−0.12	−0.01	−0.32
R^2	0.67	0.47	0.45
RMSE	2.18	3.15	5.53

For fish species, Levene's test result for the simulated and the measured diversity was $P = 0.09 > 0.05$, whereas the *t*-test result was $P = 0.49 > 0.05$, meaning that the original hypothesis could be rejected. That is, at the 95% confidence level, the variance and the mean difference between the simulated and measured values were not significant. Similarly, for zoobenthos, Levene's test result for the simulated and measured diversity was $P = 0.07 > 0.05$, whereas the *t*-test result was $P = 0.61 > 0.05$, meaning at the 95% confidence level the variance and the mean difference between the simulated and measured values were not significant. Likewise, for algae, Levene's test result for the simulated and the measured diversity was $P = 0.20 > 0.05$, whereas the *t*-test result was $P = 0.56 > 0.05$, meaning at the 95% confidence level the variance and the mean difference between the simulated and measured values were not significant.

To summarise, after comparing the simulated and measured biodiversity values obtained in this study, from a statistical point of view, no significant differences were noted in the variances and mean values. Therefore, the simulation results were accurate, and the model can be used for practical applications.

Conclusion

In this study, PLS was used to analyse the key hydrological and water quality factors that affect riverine aquatic biodiversity. The food web-based Ecosim model was used to simulate the biomass of aquatic organisms under undisturbed conditions. After comparing the results obtained with

Table 10. Levene's test and *t*-test results for the simulated biodiversity of the three types of organisms

	d.f.	*F*	*P*-value	*t*	*P*-value
Fish	138	2.81	0.09	0.68	0.49
Zoobenthos	172	7.32	0.07	0.51	0.61
Algae	168	5.34	0.20	0.58	0.56

Table 9. Regression coefficients of water quality and hydrological factors relating to Eqn 4

PI, permanganate index; DO, dissolved oxygen; EC, electrical conductivity; TA, total alkalinity; Tem_{aa}, the temperature of air; TH, total hardness; TN, total nitrogen; TP, total phosphorus; Tran, the transparency of water

	Algae		Zoobenthos		Fish	
Water quality	Constant	2.93	Constant	2.55	Constant	2.5
	CO_3	−0.02	CO_3	−0.03	CO_3	−0.01
	PI	−0.03	Ca	0.02	DO	0.03
	Ca	0.01	EC	0.01	HCO_3	0.02
	Cl	0.01	HCO_3	0.17	NH_3-N	−0.05
	NO_3-N	−0.05	TH	0.05	NO_2-N	−0.91
	TN	0.3	TP	−0.3	TA	−0.03
	Tem_a	−0.01	Tem_a	−0.03	TN	−0.01
	Tran	0.01			TP	−0.25
Hydrology	Constant	2.3	Constant	1.69	Constant	1.79
	Stream flow	0.21	Water depth	−0.26	Water depth	−0.38
	Flow velocity	−0.18	River width	0.3	Flow velocity	0.59

measured aquatic biodiversity for each type of organism, the biodiversity fluctuation caused by the variations in stresses was obtained. Based on the relationships between hydrological factor variations, water quality factor variations and biodiversity fluctuations, and using the partial PLSR method, a multidimensional river hydrology–water quality–biodiversity response model was established for prediction of future aquatic biodiversity with variations in water quality and hydrological factor. The results showed that with regard to the water quality factors, TP, TN, NH_3-N and DO all had strong effects on riverine biodiversity. With regard to hydrological factors, the three types of organisms were affected by different driving factors: water depth had a stronger effect on fish and zoobenthos, whereas flow velocity largely affected fish and algae. The established multidimensional river hydrology–water quality–biodiversity response model showed good performance in the simulated and measured biodiversity relationship and can be used for practical application predicting future changes in aquatic biodiversity.

Statement of authorship

C. S. Zhao and S. T. Yang designed the study; Y. Zhang performed modelling work and analysed output data; S. T. Yang, B. E. Dong, Y. R. Ge, H. M. Zhang and Y. Zhang collected data; Z. S. Zhang and B. E. Dong performed the meta-analysis; C. S. Zhao, T. L. Pan and Y. Zhang wrote and revised the manuscript.

Conflicts of interest

The authors declare that they have no conflicts of interest.

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