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RESEARCH ARTICLE

Using multiple indicators to assess spatial and temporal changes in ecological condition of a drinking water reservoir in central China

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Abstract – Monitoring the ecological status of drinking water reservoirs is very important for assessing risks to the safety of drinking water supplies. Because of the huge number of potential organic and inorganic contaminants in water, we focused our study on common water quality variables and three ecological indices. In this study, we used the modified Carlson trophic status index, Shannon diversity index, and the phytoplankton functional group index to assess the changing ecological status of Zhushuqiao Reservoir, a drinking water source in central China. Stratified water sampling for abiotic and biotic variables was conducted bimonthly from April 2016 to February 2017. All three indices indicated that upper reservoir water quality and ecological condition were worse than those in the lower reservoir. Also, the dominant phytoplankton species in the upper reservoir differed significantly from those in the lower reservoir. The reservoir was eutrophic from June to October, especially in June. During the other months, it was mesotrophic, with the best water quality in winter. Water temperature was the main driver of seasonal changes in both biotic and abiotic indicators, although hydrological condition also affected water quality. Total phosphorus (TP) was the limiting factor for phytoplankton, but phytoplankton biomass increased greatly when both TP and total nitrogen increased. Each index had weaknesses; but applying all three together yielded a comprehensive ecological assessment of Zhushuqiao Reservoir and could do so for other similar reservoirs.

Keywords: Phytoplankton / environmental factors / ecological status / multiple indicators / Zhushuqiao Reservoir

1 Introduction

Because of global industrialization, agriculture, and municipal organic pollution, there is an urgent need to assess and protect drinking water reservoirs to safeguard human and ecological health (Smith, 2003; Fontana *et al.*, 2014). There is an enormous number of potential organic and inorganic contaminants in water, which together are very expensive to monitor (Hughes and Peck, 2008; USEPA, 2016). Therefore in recent decades, biological indicator species for assessing water pollution have been applied, but single indicators fail to account for natural differences in lake types (Launois *et al.*, 2010). Also, nutrient and organic matter concentrations cannot distinguish the ecological effects of differing pollutants (Borics *et al.*, 2012; Crossetti *et al.*, 2013). Therefore, multiple indicators are increasingly being used for assessing the ecological status of lakes (*e.g.*, Schaumburg *et al.*, 2004; Nõges *et al.*, 2010; USEPA, 2016).

The most commonly used water quality evaluation method for lakes and reservoirs is the Carlson (1977) trophic state index (TSI). As a biological–physical–chemical indicator, the TSI combines chlorophyll *a* (Chl*a*), Secchi disc transparency, and total phosphorus (TP) and it has been used to evaluate trophic status of a wide variety of water bodies (*e.g.*, Canfield Jr. *et al.*, 1983; Kitaka *et al.*, 2002; Prasad and Siddaraju, 2012; Bekteshi, 2015).

The Shannon diversity index (*H*) is also widely used for assessing ecological status and trends of many biological assemblages in many ecosystem types (*e.g.*, Heip and

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Engels, 1974; Hughes and Gammon, 1987; Stoddard *et al.*, 2008; Morris *et al.*, 2014). It incorporates both taxa richness and taxa evenness and offers an ecological perspective of the water quality influences on the proportional abundances of the biota (Pongswat *et al.*, 2001; Palleyi *et al.*, 2011). Grover and Chrzanowski (2004) found that phytoplankton diversity was significantly correlated with the number of limiting nutrients in one Texas reservoir, but not a neighboring one. Berry *et al.* (2017) reported linkages between cyanobacterial harmful algal blooms and altered bacterial assemblage diversity in Western Lake Erie. Nonetheless, Stevenson (2014) cautioned the use of diversity indices for making ecological assessments because species richness may respond nonmonotonically with stressor changes. However, restricting diversity (or richness) to pollution-sensitive or pollution-tolerant taxa can reduce problems with total diversity indices (see below). Therefore, we sought to determine how well *H* related to the two other commonly used indicators of lake quality.

Recently, phytoplankton functional groups have been used to assess ecological conditions (Moreno-Ostos *et al.*, 2008; Kruk and Segura, 2012; Crossetti *et al.*, 2013). This approach is based on the morphological, physiological, and ecological similarities of phytoplankton species, versus only their taxonomic classifications (Reynolds *et al.*, 2002; Reynolds, 2006; Padisák *et al.*, 2009). Comparative analyses of phytoplankton assemblages are simplified and improved by using indices that incorporate assemblage structure and function (Sommer *et al.*, 1993). One such index, the *Q*-index of phytoplankton (*Q*), weights functional group relative biomasses with an *F* factor for each functional group, which is related to limnological parameters of the water body. Those parameters include lake type (acidic, calcareous, alkaline), depth, surface area, and exchange rate (persistent, intermittent, run-of-river) (Padisák *et al.*, 2006). *Q* scores range from 0 (poor quality) to 5 (excellent quality), and they can be applied without geographic limitation because they are calibrated for naturally varying limnological variables (Padisák *et al.*, 2006). As primary producers in aquatic ecosystems, phytoplankton are small sized, reproduce rapidly, easily collected and identified, and extremely sensitive to environmental changes (Padisák, 1994). Also, most of the same phytoplankton species occur on all the ice-free continents. Therefore, *Q* scores can be compared between lakes in Europe (Éva and Padisák, 2008; Çelekli and Öztürk, 2014) and South America (Fonseca and Bicudo, 2011).

Each indicator has both strengths and weaknesses, meaning that for overall water quality assessments, any single abiotic parameter has limitations. Although it is a very useful indicator of trophic state, Chl*a* is the only biological factor considered in the TSI (Carlson, 1977; Crossetti and Bicudo, 2008). Usually, *H* is calculated only from biological taxonomic variables without direct linkage to abiotic variables or functional variables; however, anthropogenic impacts affect both taxonomic and functional diversity (Abonyi *et al.*, 2012). *Q* provides values based on relative phytoplankton biomass data and functional group contributions to define lake trophic status (Crossetti and Bicudo, 2008; Cellamare *et al.*, 2012) and can be used without geographic limitations (Padisák *et al.*, 2006; Crossetti and Bicudo, 2008; Pasztaleniec and Poniewozik, 2010). Moreover, functional groups highlight survival strategies, sensitivities, and tolerances in lakes having different trophic status and habitat features. However, the most difficult step in *Q* application is the determination of the *F* factors, which were based on European knowledge and experience; therefore, they require further evaluation on other continents and in multiple biomes and ecoregions (Crossetti and Bicudo, 2008). For example, Stevenson *et al.* (2013) developed a natural-variation adjusted lake diatom multimetric index that was applicable across the entire conterminous USA. Nonetheless, multiple methods considering both biotic and abiotic components offer a more comprehensive assessment of reservoir ecological status (*e.g.*, USEPA, 2016).

In some ways, reservoirs are more complex and dynamic than most natural lakes. Frequently, major reservoir arms exist with different water quality and biota than those characteristic of the main bodies of the reservoirs (Sanches *et al.*, 2016). Similarly, the waters of the upper reservoir zones are typically much shallower, more turbid, and with coarser bottom substrates than those in the lower reservoir zones near the dams, thereby supporting different biota (Thornton, 1990; Terra and Araujo, 2011; Sanches *et al.*, 2016). River inflows and human demands for water consumption and hydropower markedly affect reservoir water exchange rates (Kolding and van Zwieten, 2012; Kaufmann *et al.*, 2014). Finally, because reservoirs are filled by river inflows versus ground water, their water levels tend to fluctuate to a greater degree than most seepage and drainage lakes from seasonal and annual precipitation variability and drought regimes (Thornton, 1990).

Thus, in the current study, we used TSI, *H*, and *Q* to conduct a comprehensive assessment of the ecological status of Zhushuqiao Reservoir, based on physical, chemical, and biological parameters. Our objectives were to (1) conduct a comprehensive evaluation of the ecological status of the temporal and spatial dynamics of the reservoir by comparing the three indices and (2) explore relationships between those indices and environmental parameters. We hypothesized that the ecological status assessment results would differ among the three indices, and those values would vary with reservoir zone and season.

2 Materials and methods

2.1 Study area

Zhushuqiao Reservoir (113°–115° E, 27.5°–28.5° N) has a watershed area of 564 km^2, was built in 1992, and is located about 80 km from Changsha, the capital of Hunan province in south-central China. With a volume of 2.78×10^9 m^3, surface area of 10.7 km^2, and mean depth of 20–30 m (maximum depth 65 m), it is the largest drinking water reservoir in Changsha. The local climate of the reservoir area is subtropical monsoon, with daily air temperature ranging from −4 to 39 °C, with a mean of 17.5 °C, and daily precipitation ranging from 0 to 511 mm, with annual precipitation of 1600 mm in 2016–2017.

2.2 Phytoplankton sample collection, processing, and identification

We established 5 sampling stations based on Zhushuqiao Reservoir topographic features (Fig. 1), including the water inlet and outlet, and open water area. Vertical stratified

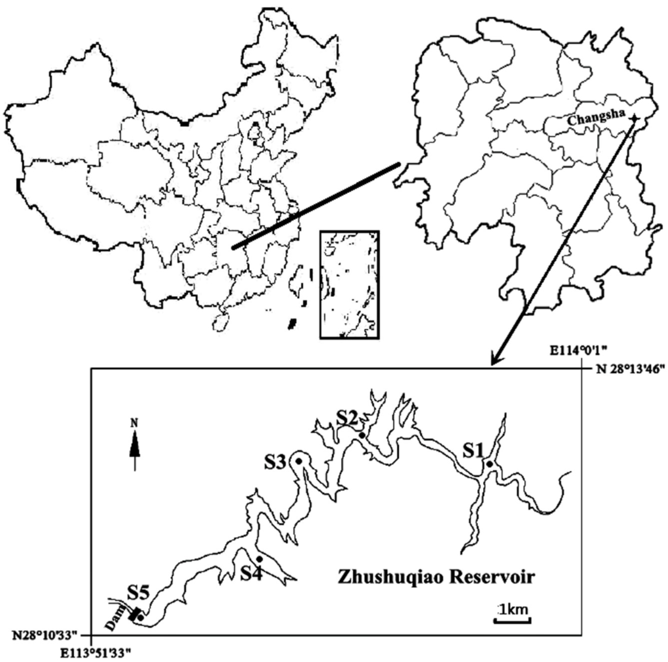

Fig. 1. Location of Zhushuqiao Reservoir and our sampling sites.

sampling was conducted to understand the ecological status of different water layers. Vertical sampling intervals were set based on the depth of each sampling site, *i.e.*, 3 m for sites 1 and 2 (5 samples each), and 5 m for the other sites (7 samples in sites 3 and 4, 9 samples in site 5). The samples were collected bimonthly from April 2016 to February 2017, with 1 L of water for phytoplankton analysis and at least 2 L of water for physical and chemical analyses, for a total of 198 samples. All phytoplankton samples were fixed with Lugol's solution and formalin in the field, transferred in a cooler to the laboratory, and then concentrated into 50 ml by the siphon method after standing for 24–48 h. After thorough mixing, a 0.1 ml sample of concentrated phytoplankton was counted in a counting chamber (20 × 20 mm) using the random field method (*i.e.*, at least 400 units each). Phytoplankton taxa identification and counting were conducted with a Nikon Eclipse E100 direct light microscope (magnification: 10 × 40). Phytoplankton biomass was computed using cell surface area and specific biovolumes, which assumes a phytoplankton-specific density of 1 g cm^{-3} (Hillebrand *et al.*, 1999). Phytoplankton taxa identifications followed Hu and Wei (2006) and phytoplankton functional group classification followed Reynolds *et al.* (2002) and Padisák *et al.* (2009).

2.3 Physical and chemical parameters

Environmental parameters measured in the field included water temperature (WT, °C), pH, and dissolved oxygen (DO, mg L^{-1}) with a YSI (Model 6600 v2) multimeter. Water transparency (m) was measured with a Secchi disk (SD). Water samples were collected at the same vertical intervals as the phytoplankton samples. The water samples were transported to our laboratory for analyzing total nitrogen (TN, mg L^{-1}), TP (mg L^{-1}), and chemical oxygen demand (COD, mg L^{-1}) through use of Chinese standard methods for water quality analysis (SEPA, 2002). Chl*a* was determined by the acetone method (Lin *et al.*, 2005). The water level data were obtained from Department of Water Resources of Hunan Province, China (http://61.187.56.156/wap/index_sq.asp).

2.4 Ecological status

To quantify trophic state, we used a modified TSI following the approach of Carlson (1977) and Aizaki *et al.* (1981) as follows:

$$\mathrm{Chl}a^* = 10 \times \left(2.46 + \frac{\ln \mathrm{Chl}a}{\ln 2.5}\right)$$

$$\mathrm{SD}^* = 10 \times \left(2.46 + \frac{3.69 - 1.5 \ln \mathrm{SD}}{\ln 2.5}\right)$$

$$\mathrm{TP}^* = 10 \times \left(2.46 + \frac{6.71 + 1.5 \ln \mathrm{TP}}{\ln 2.5}\right)$$

$$\mathrm{TSI} = 0.54 \times \mathrm{Chl}a^* + 0.297 \times \mathrm{SD}^* + 0.163 \times \mathrm{TP}^*$$

TSI values <37 were classified as oligotrophic, and TSI values of 37–53 and 54–65 were denoted as mesotrophic and eutrophic, respectively. TSI values >65 were considered hypereutrophic.

We used a modified H to assess species diversity:

$$H = -\sum (n_i/N) \times \log(n_i/N),$$

where n_i is the biomass of the ith species, and N is total biomass (Graham *et al.*, 2004). H values <1 indicated seriously polluted water; 1–2 indicated α-medium polluted; >2–3 indicated β-medium polluted; and >3 indicated clean (Gao *et al.*, 2010).

The Q index of phytoplankton values was determined as proposed by Padisák *et al.* (2006):

$$Q = \sum_{i=1}^{s} (p_i \times F_i),$$

where $p_i = n_i/N$, n_i = biomass of the ith functional group, N = total biomass of all functional groups, and F is established for each functional group based on lake type (Tab. 1; Padisák *et al.*, 2006). We followed Padisák *et al.* (2006), who used Q for the European Union Water Framework Directive, for making our ecological status assessments: 0–1 = poor; >1–2 = tolerable; >2–3 = medium; <3–4 = good; and >4–5 = excellent.

2.5 Statistical analyses

We employed three analyses to assess relationships among biological and environmental variables. (1) We developed an isoline map to detect spatial and temporal distribution characteristics of phytoplankton and environmental variables through use of Surfer 8.0 (Golden Software Inc., Golden,

Table 1. Phytoplankton functional groups (Padisák *et al.*, 2009) and F values (Padisák *et al.*, 2006) for Zhushuqiao Reservoir.

Functional groups	Most important taxa	Habitat templates	*F*
MP	*Chlorella vulgaris*	Large diatoms suspended by turbulence, inorganically turbid	5
T	*Quadrigula* sp.	Persistently mixed layers, light-limited	5
C	*Melosira granulata var. angustissima mull*	Eutrophic, sensitive to stratification onset	3
F	*Oocystis* sp., *Kirchnecriella*	Clear, deeply mixed meso-eutrophic	3
X1	*Pyrrophyta*, *Chroococcus* sp.	Shallow, eu- to hypertrophic	3
X2	*Chroomonas* sp.	Meso-eutrophic	3
Y	*Cryptophyta*	Wide range of habitats with low grazing pressure	3
D	*Synedra acus* var.	Turbid	2
E	*Dinobryon divergens*	Small, shallow base-poor	2
J	*Scenedesmus* sp., *Pediastrum* sp., *Coelastrum* sp.	Shallow, mixed, highly enriched	2
N	*Tabellaria* sp.	Continuous or semi-continuous mixed layer 2–3 m thick	2
P	*Staurastrum* sp., *Fragilaria* sp.	Similar to *N*, but higher trophic state	2
Lo	*Ceratium* sp., *Peridinium* sp.	Deep or shallow, oligo- to eutrophic	1
W2	*Trachelomonas* sp.	Meso-eutrophic ponds and shallow lakes	1
G	*Eudorina elegans*	Nutrient-rich waters, river-fed storage reservoirs	0
L_M	*Navicula* sp., *Achnanthes* sp.	Eutrophic to hypereutrophic	0
M	*Microcystis* sp.	Eutrophic to hypereutrophic	0
S1	*Pseudanabaena limnetica*	Turbid mixed water, shade-adapted	0

Colorado, US). Surfer 8.0 is a 3D data modeling and analysis software that uses isoline maps to visualize unevenly distributed data. (2) We used Pearson's correlation to analyze relationships between each environmental variable and each index one by one through use of SPSS 19.0 (International Business Machine, SPSS Inc., Armonk, New York). (3) Ordination analysis was performed in CANOCO 4.5 (Microcomputer Power, Ithaca, New York). A detrended correspondence analysis (DCA) was first run on the data, which directed us to use a redundancy analysis (RDA). RDA is a direct gradient analysis technique that was conducted to analyze relationships between all three indices and all environmental factors. For the RDA, environmental variables were transformed log $(x+1)$ (except for pH) to reduce skewness. The statistical significance of environmental variables for explaining the variance of indices in RDA was tested by a Monte Carlo permutation test.

3 Results

3.1 Environmental variables

As expected, water transparency and temperature showed seasonal patterns. Transparency peaked in summer and autumn (August and October), was the lowest in spring (April), and ranged from 1.6 to 2.6 m (average 2.0 m) (Fig. 2a).

The normal water level is 165 m in Zhushuqiao Reservoir. The water level gradually rose from April (156.3 m) to August (163.5 m) in 2016. And then, it dropped as water was released downstream during winter (December 2016) and early spring (February 2017) (Fig. 2b).

As is common for stratified lakes, WTs were reduced from the surface to the bottom, highest surface WTs (>30 °C) were observed in August, and lowest surface WTs (16 °C) occurred in December and February (Fig. 2c).

Nutrient concentrations also showed spatio-temporal patterns of variation. The mean water column TN/TP values were lowest in August and highest in October. TN increased with depth in April, but was reduced while TP increased in June. In August, mean water column TN and TP concentrations were twice those of June, but in October average water column TP was reduced (Fig. 2d, Tab. 2). In February, TP concentrations were above 0.07 g/L, which indicated that its limitation was no longer strongly pronounced.

Mean water column concentrations of Chl*a* and biomass varied widely, 1.8–65.9 $\mu g\,L^{-1}$ and 0.14–16.89 $mg\,L^{-1}$, respectively (Fig. 2e). The mean water column Chl*a* values were lowest in April and highest in June and August (Chl*a* $>30\,\mu g\,L^{-1}$). Mean water column biomasses were lowest in December and highest in August (Fig. 2f).

3.2 Spatial and temporal changes of ecological status

3.2.1 Trophic state index (TSI)

Mean water column TSI values ranged from 36.55 (oligotrophic) to 66.59 (eutrophic), with a mean of 48.42 (mesotrophic) for the entire reservoir (Fig. 3a). The reservoir was eutrophic in June and mesotrophic in the other months. TSI values gradually decreased from the surface to the bottom at all sites. During all the sampling months, sites 1 and 2 had the highest TSI values in April, June, August, December, and February, and sites 4 and 5 had the highest TSI values in October.

3.2.2 Shannon diversity index (*H*)

H values ranged from 0.82 to 3.42 (mean = 2.23), indicating medium pollution levels for the entire reservoir (Fig. 3b). *H* clearly declined from upper to lower sites in June, August, and February, and gradually decreased with increased depth in June. *H* was <2 (α-medium pollution) in some water layers at sites 2, 3, 4, and 5 in April, June, and February; all other sites and visits indicated β-medium pollution.

3.2.3 Q Index of phytoplankton

Q scores ranged from 0.94 (poor) to 3.31 (good) with a mean score of 2.33, indicating medium ecological status for the entire reservoir (Fig. 3c). Mean water column *Q* scores were highest in December (medium to good), April (tolerable to medium), August (tolerable to medium), and February (bad to good), and lowest in June (tolerable to medium) and October (bad to medium). During those months, *Q* scores were the highest at site 5, and lowest at sites 2, 3, and 4 in February, April, and June and at sites 1 and 2 in August, October, and December. *Q* scores tended to be lowest at greater depths and highest at shallow and middle depths.

A total of 171 taxa were identified during the study period; taxa from 18 functional groups contributed >2% to the total biomass. Dominant functional groups differed seasonally (Fig. 4). In April, *Chroomonas* (X2, mesotrophic), *Cryptophyta* (Y, eurytrophic), and *Oocystis* (F, meso-eutrophic) were dominant. In June, *Peridinium* (Lo, eu-hypereutrophic) and *Synedra* (D, base-poor) had the greatest biomass. In August, *Oocystis* (F, meso-eutrophic) and *Staurastrum* (P, eutrophic) were dominant. In October,

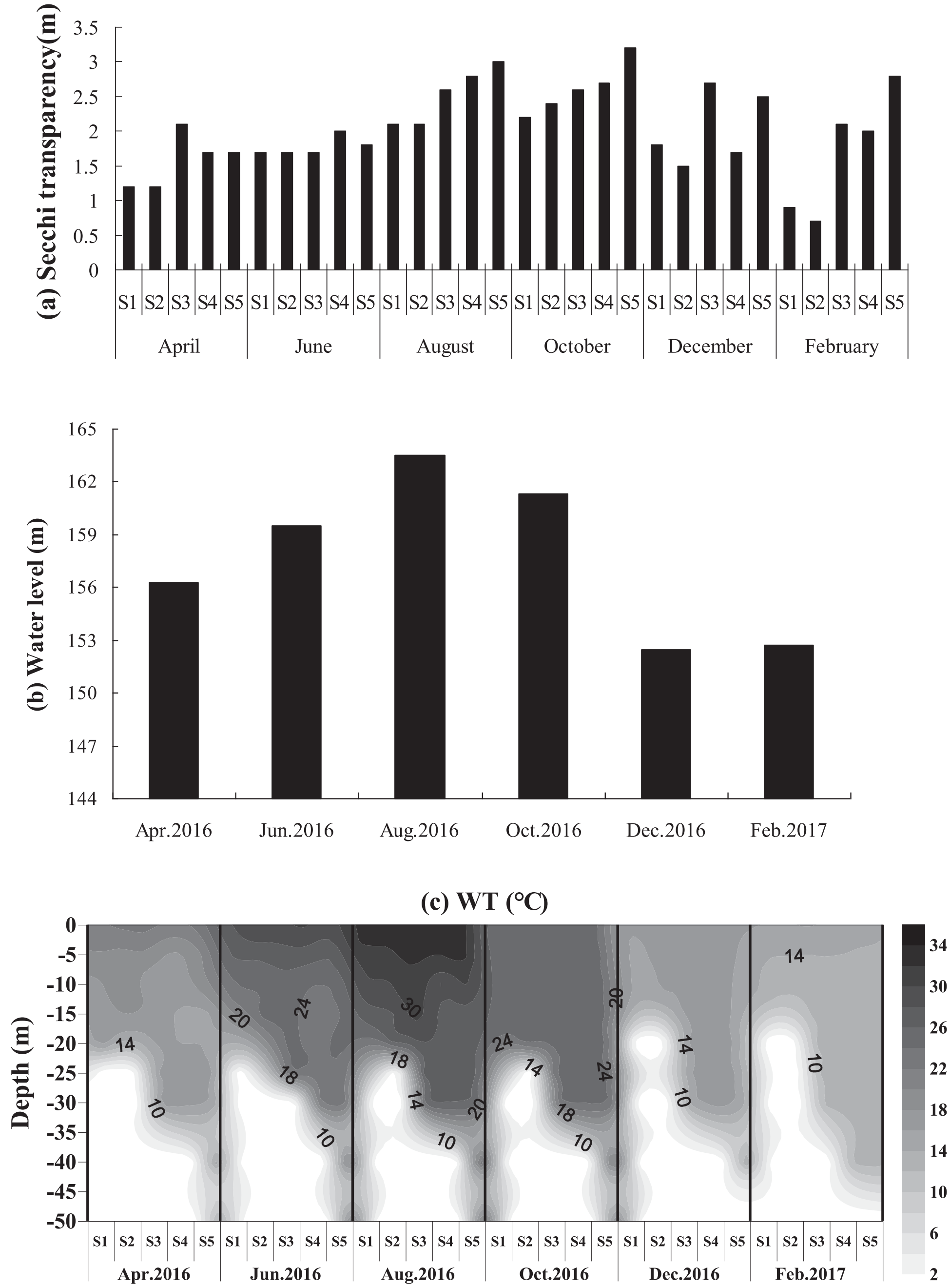

Fig. 2. Water level and water quality in Zhushuqiao Reservoir. Darker shadings in (c), (d), (e), and (f) reflect higher values as indicated by the right-side scales.

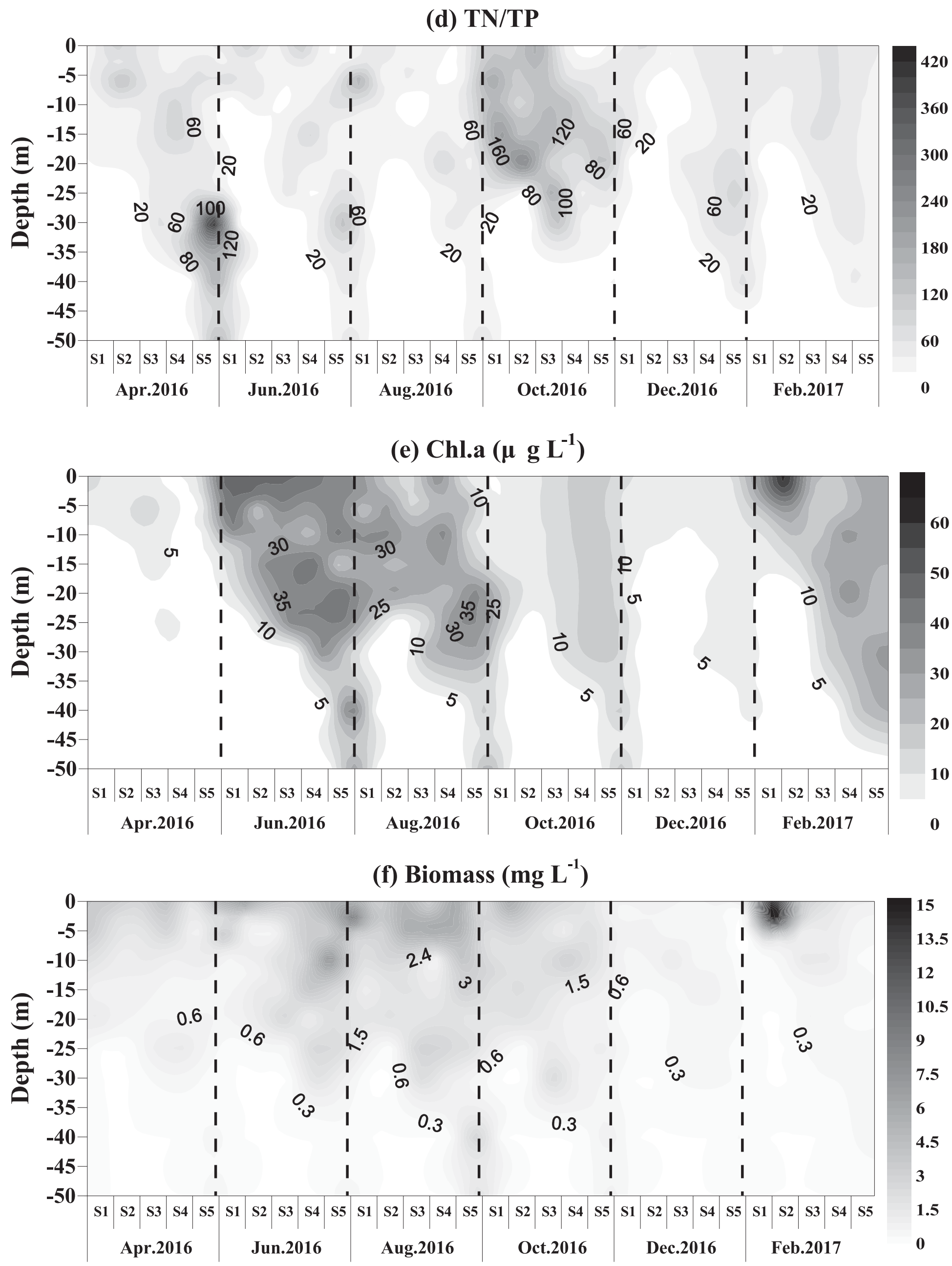

Fig. 2. (*Continued*)

Peridinium (Lo, eu-hypereutrophic) accounted for most of the biomass in sites 1 and 2; *Cryptophyta* (Y, eurytrophic) and *Oocystis* (F, meso-eutrophic) were dominant in the other sites. In December, the dominant species were *Melosira* (C, eutrophic), *Cryptophyta* (Y, eurytrophic), *Chroomonas* (X2, mesotrophic), and *Scenedesmus* (J, eutrophic). In February, *Cryptophyta* (Y, eurytrophic), *Chroomonas* (X2, mesotrophic), and *Eudorina* (G, eutrophic) dominated in site 1; the other sites were dominated by *Peridinium* (Lo, eu-hypereutrophic), *Chroomonas* (X2, mesotrophic), and *Cryptophyta* (Y, eurytrophic).

3.3 Relationships between trophic indices and environmental variables

The RDA results for ecological index and environmental parameter relationships showed that the cumulative percentage variance of all analyzed axes accounted for a total of 99.2% of index variability (Tab. 3). The first RDA axis explained 94.9% of the variation of the relationships between trophic indices and environmental variables, and it was statistically significant (Fig. 5).

Individual correlations showed similar patterns as the RDA. *H* was positively associated with Secchi depth and TN/TP, and

Table 2. Mean monthly values of TN, TP, and TN/TP in Zhushuqiao Reservoir.

		April 16	June 16	August 16	October 16	December 16	February 17
TN ($mg\,L^{-1}$)	S1	1.31	1.05	2.23	3.24	1.17	3.38
	S2	1.51	0.90	2.10	2.48	0.50	3.29
	S3	1.40	1.03	1.54	2.35	0.51	3.38
	S4	1.10	0.88	2.52	1.03	0.54	3.58
	S5	1.69	1.53	2.77	0.95	0.83	2.54
TP ($mg\,L^{-1}$)	S1	0.05	0.09	0.05	0.02	0.02	0.07
	S2	0.03	0.03	0.08	0.02	0.02	0.09
	S3	0.03	0.04	0.07	0.02	0.03	0.08
	S4	0.02	0.02	0.07	0.02	0.01	0.07
	S5	0.02	0.03	0.09	0.02	0.01	0.07
TN/TP	S1	31.17	33.10	62.50	163.34	50.56	50.23
	S2	73.83	45.14	34.50	136.34	30.49	38.43
	S3	54.40	31.33	24.25	148.54	23.63	46.27
	S4	56.77	47.88	43.88	60.65	43.59	55.94
	S5	119.35	55.50	35.03	59.27	67.81	37.85

negatively with Chl*a* ($r = -0.20$, $P = 0.04$, $N = 198$). Q was most strongly and positively associated with COD ($r = 0.31$, $P = 0.00$), and negatively associated with TN ($r = -0.26$, $P = 0.00$), Chl*a* ($r = -0.36$, $P = 0.00$), WT ($r = -0.33$, $P = 0.00$), and TN/TP ($r = -0.20$, $P = 0.00$). TSI was most strongly and positively associated with Chl*a* ($r = 0.62$, $P = 0.00$), TN ($r = 0.62$, $P = 0.00$), TP ($r = 0.59$, $P = 0.00$), and pH ($r = 0.42$, $P = 0.00$), and negatively associated with WT ($r = -0.2$, $P = 0.006$) and SD ($r = -0.38$, $P = 0.00$). Q and H were positively associated with each other but negatively associated with TSI (Tab. 4).

4 Discussion

4.1 Spatial and temporal variation of ecological status indicators

Ecological status, phytoplankton composition, and environmental variables differed seasonally, between upper- and lower-zone Zhushuqiao Reservoir sites, and with depth. This is because reservoirs change river topography, which leads to different biological adaptations to zones by the best suited biota (Wetzel, 1990; Straskraba *et al.*, 1993; Terra and Araujo, 2011).

Taxa richness and H in the upper-zone sites were higher than in the lower-zone sites, but some Cyanophyta species had greater occurrence frequency and biomass and became dominant, which led to decreased Q scores and indicated greater eutrophication in the upper zones than in the lower zones. This was likely related to the higher concentrations of TN in the upper zone. Also, Q was dominated by only a few functional groups because they are the most sensitive to human impacts (Bonnet and Poulin, 2002; Latour *et al.*, 2004). Lower hydrodynamic stability and greater turbidity were reflected by the functional group distribution in the upper sites as well. Therefore, flagellate taxa (such as Y, Lo, and X1) dominated because of their greater motility.

In the vertical dimension, all three indices indicated poor ecological status from the surface to the middle water layer, especially in June. This result is in accordance with Becker *et al.* (2010), who indicated long stratification periods and high light availability led to the development of high biomass in the epilimnion in summer. Seasonality also determines which species are able to maintain their populations and have the largest capability to dominate. Although weakly selective physical conditions may explain highly variable phytoplankton in the upper water (Reynolds and Descy, 1996), low taxa diversity may reflect severely selective environments in deeper waters.

4.2 Relationships between ecological indices and environmental variables

All three ecological indices indicated that the most serious eutrophication occurred in June and February. TSI indicated mesotrophic status in the other months. Q indicated tolerable status and H indicated medium pollution in June and February.

Three potential reasons could account for this.

First, TN and TP concentrations were similar from April to June. The increase of phytoplankton in June caused by increased temperature was possibly the main reason for the eutrophication of the water body. In Zhushuqiao Reservoir, there is a large fluctuation of temperature in spring, high temperature and rainfall in summer, high temperature in autumn, and mild temperature and little rainfall in winter. Without intense human disturbance, the eutrophication of water was greater in summer from June to October, especially the eutrophication period in June.

Second, *Synedra* (D) became dominant during this period, which has been reported in highly polluted and eutrophic waters (Leitão and Léglize, 2000).

Finally, hydrologic connections are often emphasized in water quality assessments, because nutrients and biota are affected by changes in flow regime and water levels (Pringle, 2001; Launois *et al.*, 2010). In this case, the water level was lowered to facilitate seining the whole reservoir for fish. The fish harvesting also likely re-suspended sediment nutrients, which increased TP and TN to the maximum values observed in the whole reservoir for the entire year and 2–3 times higher than those in December. This was likely the main reason for

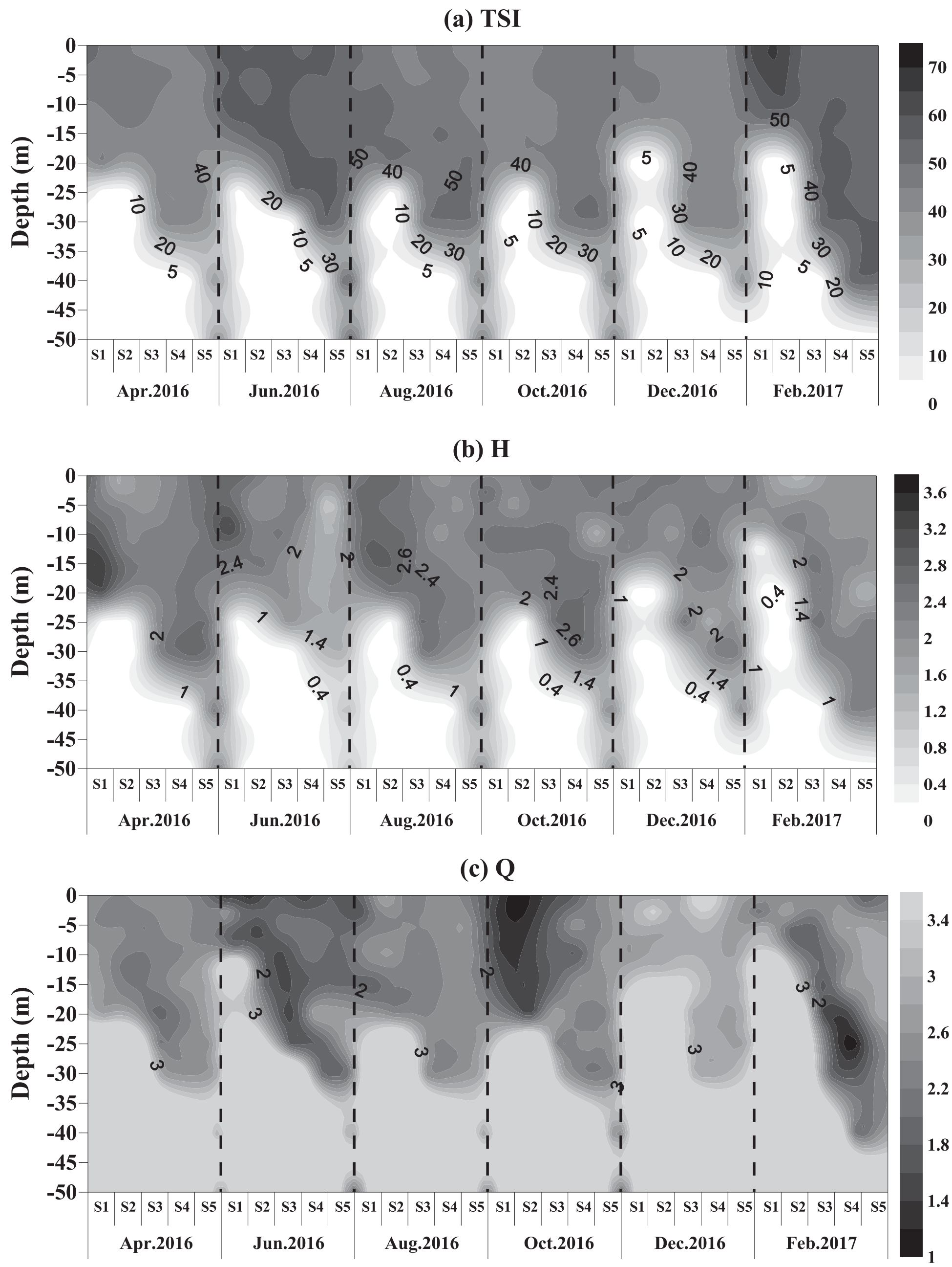

Fig. 3. Ecological status evaluations in Zhushuqiao Reservoir (a) TSI, (b) *H*, and (c) *Q*. Darker shadings in (a), (b), and (c) reflect higher scores, as indicated by the right-side values.

inducing eutrophication in February, which differed from the results of Yang (2008).

August is the rainy season in the study area, resulting in increased water level, increased nutrients and phytoplankton biomass in the reservoir, so the water ecological status was also poor. As TP and WT decreased in October, the number of *Peridinium* (Lo) and *Oocystis* (F), which can adapt to the lower nutrient concentrations (Padisák *et al.*, 2009), increased. In December, the highest ecological status period, the reason for the increase of *Q* value was that the growth of most Cyanophyta and Chlorophyta was inhibited by low temperature while the diatom *Melosira granulata* var.*angustissima mull* (C) became dominant, which has a higher *F* factor and a preference for cold water (Lopes *et al.*, 2005; Wang *et al.*, 2011; Stević *et al.*, 2013). Indeed, Cellamare *et al.* (2012) and USEPA (2016) report that winter is an inappropriate period to assess temperate lake ecological status; rather, lakes should be monitored when they are likely to be in their worst condition. But this may not be the case in tropical ecoregions where winters are relatively warm and precipitation and water levels are low (Callisto *et al.*, 2014).

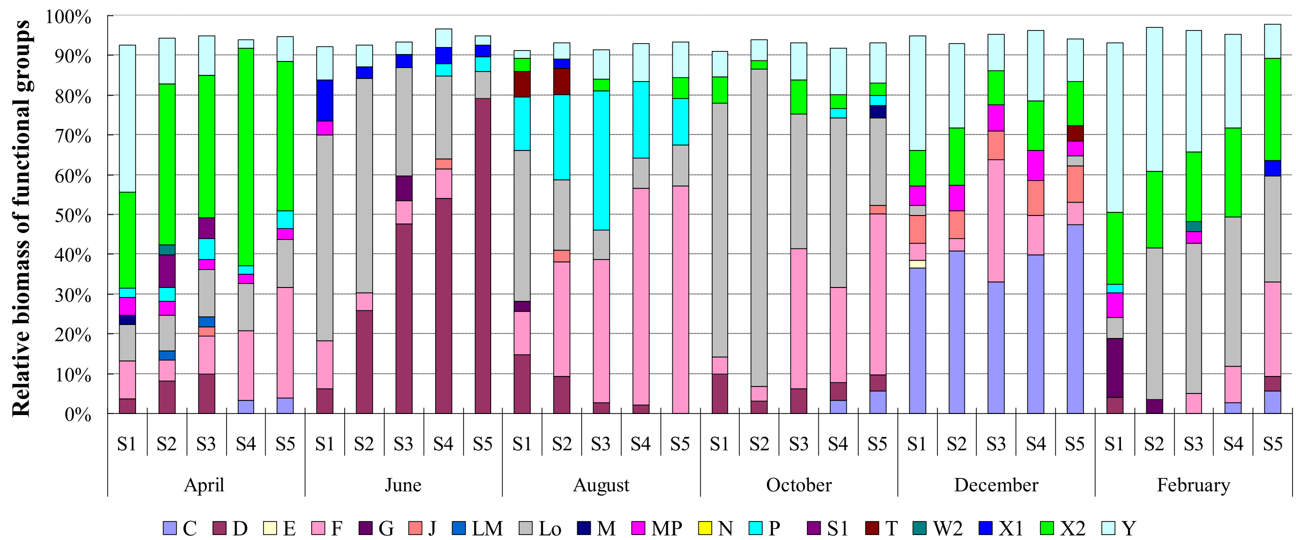

Fig. 4. The relative biomass (%) of functional groups in Zhushuqiao Reservoir. See Table 1 to relate codes to taxa.

Table 3. RDA results for ecological index and environmental parameter relationships in Zhushuqiao Reservoir. Note, because RDA is a constrained ordination analysis, it cannot provide loadings for single variables.

Axes	λ_1	λ_2	λ_3	λ_4	Total variance
Eigenvalues	0.949	0.002	0.000	0.040	1.000
Ecological indices-environment correlations	0.980	0.605	0.310	0.000	
Cumulative percentage variance					
Ecological indices data	94.9	95.2	95.2	99.2	
Ecological indices-environment relations	99.7	100.0	100.0	0.0	
Sum of all eigenvalues					1.000
Sum of all canonical eigenvalues					0.952

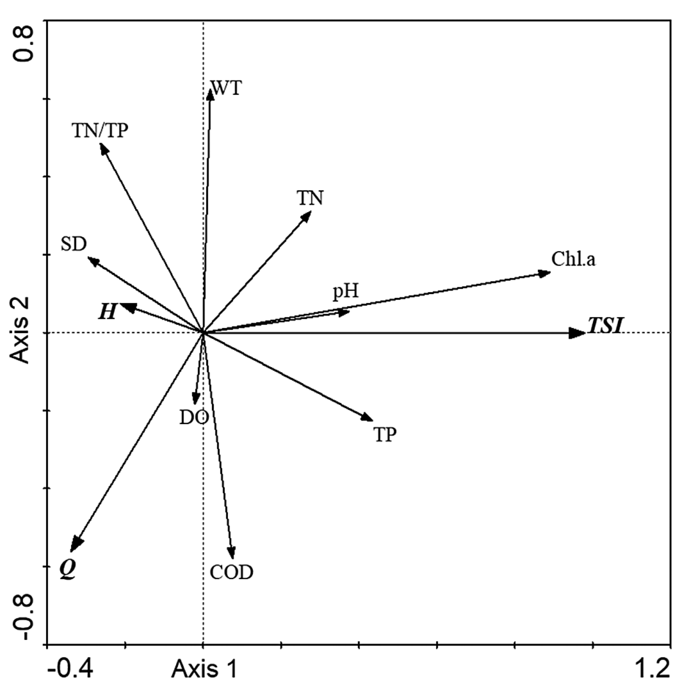

Fig. 5. Redundancy analysis indicating relationships between ecological indices and environmental parameters in Zhushuqiao Reservoir.

Table 4. Pearson correlations between ecological indices and environmental parameters in Zhushuqiao Reservoir.

		TSI	*H*	*Q*
	N	198	198	198
TN	*r*	0.616**	0.015	−0.263**
TP	*r*	0.589**	0.017	−0.011
TN/TP	*r*	−0.032	−0.008	−0.201**
COD	*r*	0.108	0.078	0.310**
Chl*a*	*r*	0.615**	−0.203**	−0.363**
WT	*r*	−0.196**	0.124	−0.328**
pH	*r*	0.417**	−0.078	−0.133
SD	*r*	−0.381**	−0.012	−0.012
DO	*r*	0.063	−0.037	0.112
TSI	*r*	1.000	−0.142*	−0.282**
H	*r*	−0.142*	1.000	-0.282
Q	*r*	−0.282**	−0.008	1.000

Significance codes: $^{**}P < 0.01$, $^{*}P < 0.05$.

In our study reservoir, TP was the limiting factor for phytoplankton. When TP increased but TN did not, phytoplankton increased little; but biomass increased greatly when both TP and TN concentrations increased. This indicates that when water TP and TN increase simultaneously phytoplankton production will be promoted. This result has been reported in many other lakes and reservoirs (*e.g.*, Fourqurean *et al.*, 1993; Yoshimura and Kudo, 2001). In future studies, greater attention should be paid to ecological status impacts from weather shifts and human activities. Also, in addition to TP and TN, other nutrients, such as iron and silica, should be monitored because they are affected by those same changes, and they in turn affect phytoplankton production.

5 Conclusions

(1) The three evaluation methods indicated that water quality was worse in the upper reservoir than in the lower reservoir. Dominant species in the upper reservoir differed from those in the lower reservoir.

(2) The degree of eutrophication was most apparent in June. Other periods indicated mesotrophic status, with the best water quality observed in winter.

(3) Changes in WT were the main drivers of seasonal changes in both biotic and abiotic indicators, but hydrological changes were important factors affecting water quality as well.

(4) TP was the limiting factor for phytoplankton, but phytoplankton increased greatly when both TP and TN concentration increased.

(5) Each of the three indicators in this study had its own strengths and weaknesses, indicating the need for comprehensive evaluation by employing multiple indicators.

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