



# Relationship between phytoplankton community and environmental factors in landscape water with high salinity in a coastal city of China

Nan Wang<sup>1</sup> · Jiaqing Xiong<sup>1</sup> · Xiaochang C. Wang<sup>1</sup> · Yan Zhang<sup>2</sup> · Honglei Liu<sup>2</sup> · Bin Zhou<sup>2</sup> · Pan Pan<sup>1,3</sup> · Yanzheng Liu<sup>1</sup> · Feiyang Ding<sup>1</sup>

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## Abstract

Relationship between phytoplankton community and environmental variables was explored in three landscape water bodies (namely Jiyun River Oxbow (JRO), Qingjing Lake (QL), and Jiyun River (JR)) with high salinity, located in Sino-Singapore Tianjin Eco-city of China, using redundancy analysis (RDA). A total of 48 species of phytoplankton were identified during the study period, in which Chlorophyta and Bacillariophyta accounted for 35.42 and 31.25%, respectively. The most dominant species of the studied water bodies were *Cyclotella meneghiniana* (Bacillariophyta) and *Aphanocapsa elachista* (Cyanophyta). The diversity index ranged from 0.56 to 1.42, with an average of 1.11, reflecting low biodiversity in the phytoplankton community. Moreover, the average density of phytoplankton was  $42.39 \times 10^6$  cells/L, indicating that those landscape water bodies belonged to moderate eutrophication. The results of RDA revealed that the most significant environmental factors influencing phytoplankton community were water temperature (WT), dissolved total phosphorus (DTP), salinity, and total nitrogen (TN) ( $p < 0.05$ , Monte Carlo permutation test). Meanwhile, *Aphanocapsa elachista* was positively correlated with WT, TN, and salinity, while *Cyclotella meneghiniana* was positively related to salinity and negatively related to TP. The results suggested that salinity was a non-negligible key factor affecting the phytoplankton community of the water body with high salinity.

**Keywords** High salinity · Landscape water · Phytoplankton · Environmental factors · RDA

## Introduction

As the primary producer in aquatic environment, phytoplankton are sensitive to changes in aquatic ecosystems and have been suggested to be good indicators of water quality and lake trophic state (Jiang et al. 2014; Rosińska et al. 2017; Wang et al. 2015). Phytoplankton community has different cell

organizations, certain changes of species composition, and diversity in different water bodies, such as lakes, reservoirs, rivers, and ponds (Bolgovics et al. 2017; Çelekli et al. 2014; Padišák et al. 2006; Xiao et al. 2011). Most previous studies have shown that the structure and distribution of phytoplankton are controlled by multiple environmental variables in aquatic environment, including physical, chemical, and biological factors (Naselli-Flores Naselliflores 2000; Tian et al. 2013a; Yin et al. 2011). If one of these factors changes, it will affect the phytoplankton community structure to a certain extent (Jiang et al. 2014). Both Cetinić et al. (2006) and Zhao et al. (2015) found that water temperature (WT) was a key driving factor affecting the phytoplankton community. In addition, the content of nutrients was also thought to be a principal limiting factor for aquatic phytoplankton production and influencing phytoplankton community (Cetinić et al. 2006; Smith 1982; Wang et al. 2015). Besides, the ratio of N/P may also predict the potential of algae growth (Daines et al. 2014; Reynolds 2006).

Urban landscape water, as an important part and elementary producers of the urban ecosystem, not only can adjust

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✉ Jiaqing Xiong  
xiongjqiaqing@xauat.edu.cn

<sup>1</sup> International S&T Cooperation Center for Urban Alternative Water Resources Development, Shaanxi Key Laboratory of Environmental Engineering, Key Lab of Northwest Water Resources, Environment and Ecology, MOE, School of Environmental and Municipal Engineering, Xi'an University of Architecture and Technology, Xi'an 710055, China

<sup>2</sup> Tianjin Academy of Environmental Sciences, Tianjin 300191, China

<sup>3</sup> Wuhuan Engineering Co. Ltd., Wuhuan 430223, China

ecosystem balance, but also has high societal value as it could provide recreational opportunities and esthetic benefits to large numbers of people. Thus, landscape water plays an irreplaceable role in the eco-city construction. Recently, with the rapid development of economy and the improvement in people's living standards, a lot of constructed and/or restored landscape waters appear in many cities of China, especially in coastal cities of northern China, such as Tianjin and Qingdao. Because the soil in the beach area may contain high level salt, most of these landscape waters in coastal cities, especially in areas closer to the ocean, are saline water bodies. High salinity and often variable salinity is a unique ecological feature of these landscape water bodies. Additionally, most of these landscape waters are enclosed or semi-enclosed water bodies, and thus, their self-purification capacity is weak. Many urban landscape waters, including saline water bodies, experience algal blooms and reduced water clarity associated with eutrophication due to excess nutrient inputs (Fathi and Flower 2005; Henny and Meutia 2014; Huser et al. 2016; Zhao et al. 2005). Therefore, there is an urgent need to understand the community structure and driving factors of phytoplankton in these waters, since phytoplankton can be used as an indicator of water quality.

However, most previous researches mainly focused on the study of phytoplankton community and its driving factors in freshwater landscape waters (Sun et al. 2009; Tian et al. 2013a; Tian et al. 2013b; Zhao et al. 2015). The comprehensive researches on environmental conditions related to phytoplankton community in the landscape water with high salinity in coastal cities have been rarely conducted. It was reported that salinity was a key factor influencing phytoplankton composition and abundance (Gasiūnaitė et al. 2005; Henny and Meutia 2014; Israël et al. 2014). Zhao et al. (2005) found that the high and often variable salinity of inland salt lakes was reflected by a decreased biodiversity; the higher the salinity, the fewer the species of phytoplankton, with both the diversity index and total species number declining. Gasiūnaitė et al. (2005) reported that the structure of both diatom- and cyanophyte-dominated community was governed by salinity in Baltic Sea. Thus, salinity may be one of the important factors driving the heterogeneity of habitats and phytoplankton communities in the landscape water with high salinity in coastal cities.

Therefore, three landscape water bodies with high salinity, located in Tianjin, a coastal city of northern China, were selected as a typical case study. The following indicators of the three landscape water bodies were investigated, including phytoplankton community, dominant species, and phytoplankton abundance. In addition, the critical environmental factors influencing phytoplankton community were identified using redundancy analysis (RDA). Because of the unique ecological environment of the studied landscape waters, this paper would provide baseline data for research on phytoplankton community of landscape water with high salinity.

## Materials and methods

### Study site

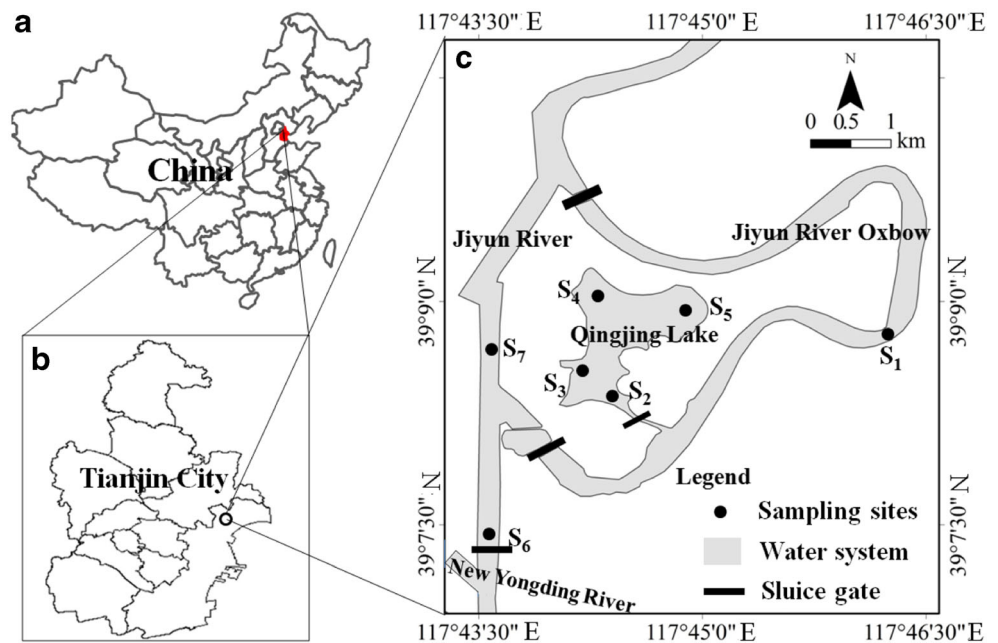
Three landscape water bodies, namely Jiyun River Oxbow (JRO), Qingjing Lake (QL) and Jiyun River (JR) (Fig. 1), were selected in this research. Those landscape waters, located in Sino-Singapore Tianjin Eco-city ( $39^{\circ} 5' 14''$ – $39^{\circ} 8' 45''$  N,  $117^{\circ} 43' 34''$ – $117^{\circ} 46' 48''$  E), are the typical landscape waters under the influence of saline intrusion in China. Sino-Singapore Tianjin Eco-city is located in Tianjin Binhai New Area, which is only 1 km away from the Bohai Sea coastline. These studied water bodies are important parts of the water system of Sino-Singapore Eco-city. According to previous research (Xiong et al. 2016), the Jiyun River Oxbow's surface area is about  $2.88 \text{ km}^2$ , total volume  $6.6 \times 10^6 \text{ m}^3$ , and average depth about 2.3 m, while the QL has a surface water area of about  $1.1 \text{ km}^2$ , a volume of  $2.2 \times 10^6 \text{ m}^3$ , and an average depth of 2.0 m. At present, three landscape water bodies are connected to each other through some sluice gates; thus, both QL and JRO are enclosed water bodies when the sluice gates are closed and artificially controlled by the sluice gates. (Fig. 1). Additionally, a tidal gate has been installed to separate the JR from another water body (i.e., New Yongding Rive) at its downstream, so the JR This landscape is a semi-closed water body throughout the year except for the rainy season. In addition to the occasional rainfall, these water bodies receive limited supplemental source on "as needed" basis from sea water desalination.

### Sampling

As shown in Fig. 1, seven sampling sites were selected for collecting water samples in those landscape waters, 1 point for JRO ( $S_1$ ), 4 points for QL ( $S_2$ ,  $S_3$ ,  $S_4$ , and  $S_5$ ), and 2 points for JR ( $S_6$  and  $S_7$ ). From December 2016 to July 2017, surface water samples (0.5 m depth) were collected at monthly intervals between 10:00 and 11:00 a.m., with frozen days excluded. Those landscape waters were frozen over from January to February. Three parallel water samples were collected at each sampling site. WT, pH, salinity, and dissolved oxygen (DO) were measured in situ using a portable multi-parameter probe (Yellow Spring Instruments, Ohio, USA); qualitative plankton samples for identification of algal species were collected from the surface water with a  $64\text{-}\mu\text{m}$  pore size net.

For quantitative phytoplankton analyses and nutrient analyses, surface water samples were collected with a Van Dorn sampler. Phytoplankton samples were immediately fixed with neutral Lugol's solution and concentrated after 48 h sedimentation (Zhu et al. 2013). In addition, nutrient samples were kept in a cool, dark environment and then transported to the laboratory for analyses within 24 h.

**Fig. 1** Location map of the sampling



## Sample analysis

In the laboratory, the water samples were directly analyzed for total phosphorus (TP), total nitrogen (TN) and suspended solids (SS), whereas the filtered samples were analyzed for dissolved total phosphorus (DTP), dissolved total nitrogen (DTN), phosphate ( $\text{PO}_4^{3-}\text{-P}$ ), ammonium ( $\text{NH}_4^+\text{-N}$ ), nitrates ( $\text{NO}_3^-\text{-N}$ ), and nitrites ( $\text{NO}_2^-\text{-N}$ ). Standard methods were referred for the chemical analyses (EPAC 2002).

The concentrated phytoplankton samples (30 mL) were used for phytoplankton species identification and quantification (Eker et al. 1999). After mixing, 0.1 mL of the samples was counted in a counting chamber using a light microscope (CX31, OLYMPUS, Japan) at  $\times 400$  magnification. The numbers of cells of different phytoplankton species in 100 random fields were determined. Phytoplankton species were identified based on morphology (Hu et al. Hu and Wei 2006; Weng et al. Weng and Xu 2010).

## Statistical analysis

The dominant species of phytoplankton were determined based on the dominance value of each species, as shown in Eq. (1), and the phytoplankton diversity was investigated according to the Shannon–Wiener index (Shannon and Weaver 1949; Tian et al. 2013a), as shown in Eq. (2):

$$Y = \frac{n_i}{N} \times f_i \quad (1)$$

$$H = - \sum_{i=1}^s \frac{n_i}{N} \ln \left( \frac{n_i}{N} \right) \quad (2)$$

where  $n_i$  is the number of individuals of species  $i$  in a given area during the study period,  $N$  is the total number of individuals of all species in the given area during the study period,  $n_i/N$  represents the relative proportion of species  $i$ ,  $f_i$  is the occurrence frequency of species  $i$ , and  $s$  is the total species number. If the dominance value  $Y$  of a species is higher than 0.02, this species is considered as dominant species during the sampling periods (Jiang et al. 2014; Sun et al. 2006; Wang et al. 2015).

Ordination analysis was performed using CANOCO 4.5 (ter-Braak and Šmilauer 2002). Detrended correspondence analysis (DCA) for the species data was employed to decide whether linear or unimodal ordination methods should be applied (Lepš and Šmilauer Lepš and Šmilauer 2003). RDA was performed to analyze the relationship between environmental factors and phytoplankton community (ter-Braak and Šmilauer 2002; Lepš and Šmilauer Lepš and Šmilauer 2003; Wang et al. 2015). The significance of canonical axes and environmental variables to explain the variance of the community was tested using Monte Carlo simulations with 499 permutations. In the species data matrix, only the taxa whose abundance is no less than 1% of the total phytoplankton abundance in at least one sampling point were included; in the meanwhile, their frequencies appeared were greater than 12.5% (Lopes et al. 2005; Muylaert et al. 2000). Before computation, all of these abiotic and biological data were  $\log_{10}(x+1)$  transformed before analysis except for pH.

In addition, the significance of parameters between two factors was tested by  $t$  test and/or one-way ANOVA using the software SPSS 20.0.  $p < 0.05$  was accepted as being significant.

## Results

### Environmental factors

The primary physical and chemical parameters in the three water bodies over time are presented in Figs. 2 and 3.

Generally, it could be found that WT, DO, and pH value of the three water bodies showed a small difference among regions, while a significant spatial difference of salinity was observed during the study period (Fig. 2). As shown in Fig. 2a, average WT showed similar seasonal patterns in the whole water bodies, and minimum WT (6.3 °C) in December and maximum WT (35.1 °C) in July both appeared in JR. The WT showed an upward trend of all water bodies from December to July, while DO concentration showed a reversed trend (Fig. 2b). Mean DO content of all studied waters in winter was significantly higher than summer ( $p < 0.05$ ) and frequently exceeded the saturation value (Fig. 2b). Maximum DO content (14.31 mg/L) occurred in March at JRO, while the minimum value (3.57 mg/L) was recorded in July at JRO. The pH value was always alkaline and fluctuated between 7.78 and 9.56 (Fig. 2c). Additionally, the value of pH of QL was higher than that of JRO and JR.

As shown in Fig. 2d, the salinity content of those landscape waters had a marked difference among regions ( $p < 0.05$ ), with a decreasing order of JRO > QL > JR. At JRO, the mean salinity content was significantly higher than that of other two water bodies ( $p < 0.05$ ), ranging from 17.11 to 29.96 g/L. In addition, mean salinity content of JRO in summer was significantly higher than that in winter ( $p < 0.05$ ). The average salinity value of QL showed an unobvious seasonal variation,

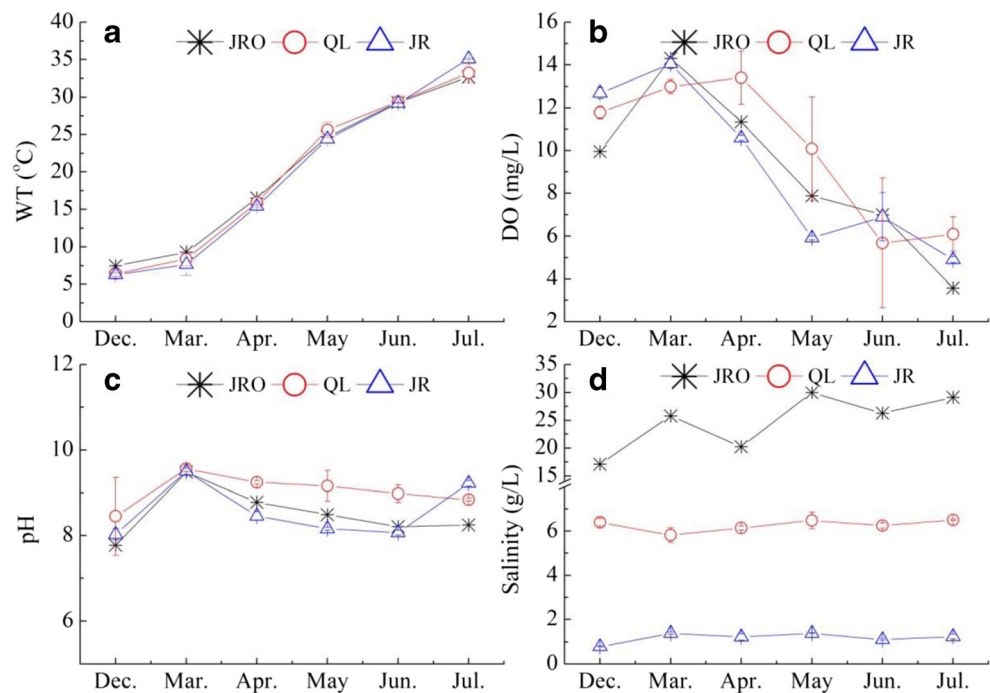
and the minimum salinity content (5.82 g/L) was recorded in March, while the maximum salinity value (6.51 g/L) occurred in July. The salinity content of JR was lowest, just ranging from 0.80 to 1.39 g/L, and with unobvious seasonal variation.

As shown in Fig. 3a, the TN concentration of JR was higher than that of other two landscape waters. Average TN contents of JRO, QL, and JR were 5.53, 4.22, and 7.69 mg/L, respectively. The TP content was relatively stable during the study period except for JR in March and July (Fig. 3b). In addition, the TP content of JR was significantly lower than that of other two landscape waters ( $p < 0.05$ ). Figure 3c shows that the average  $\text{NH}_4^+\text{-N}$  concentrations of JRO, QL, and JR were 1.29, 1.01, and 2.40 mg/L, respectively. For JRO and JR, the highest concentrations of  $\text{NH}_4^+\text{-N}$  were both appeared in March, while the highest  $\text{NH}_4^+\text{-N}$  content (1.61 mg/L) of QL was detected in April. As shown in Fig. 3d, the mean SS values of JRO, QL, and JR were 56.3, 52.4, and 29.9 mg/L, respectively. Maximum content of SS (82.0 mg/L) was recorded in March at JRO, while minimum value (10.0 mg/L) was observed in May at JR. The SS value of QL was little lower than that of JRO, ranging from 26.5 to 76.5 mg/L. Furthermore, it could also be found that the value of SS in JR was much lower than that of other two water bodies except March and July.

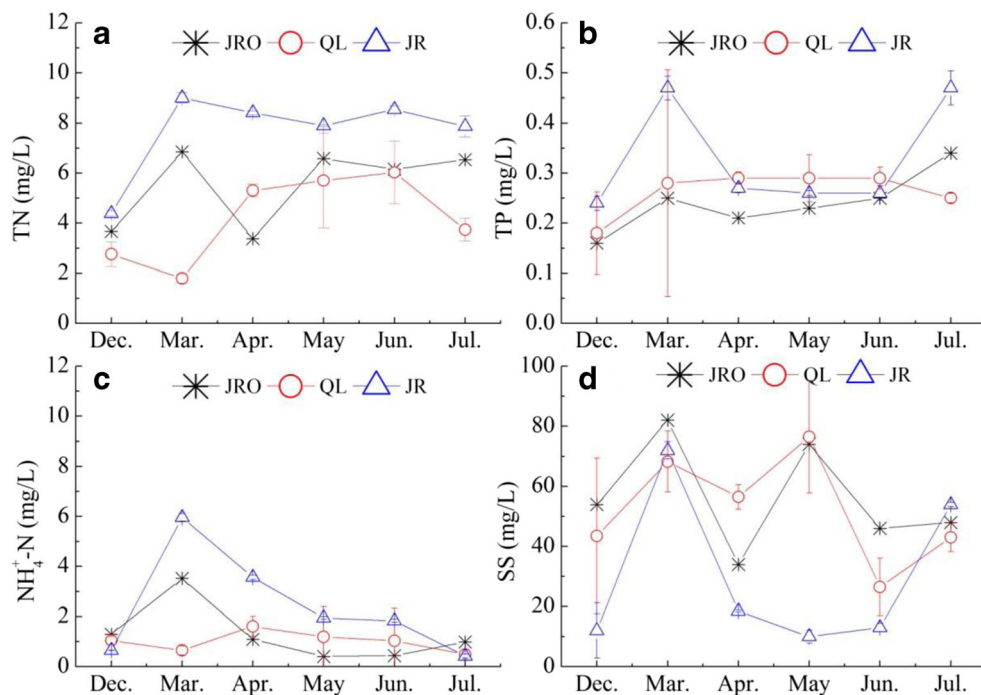
### Phytoplankton community

A total of 48 species of phytoplankton belonging to 7 phyla were identified during the study period. Among these species, 17 species belonging to Chlorophyta represented approximately 35.42% of the total species, 15 species belonging to

**Fig. 2** Temporal variations of (a) WT, (b) DO, (c) pH, and (d) salinity in the three landscape water bodies during the study period



**Fig. 3** Temporal variations of (a) TN, (b) TP, (c) NH<sub>4</sub><sup>+</sup>-N, and (d) SS in the three landscape water bodies during the study period



Bacillariophyta represented 31.25%, and 7 species belonging to Cyanophyta represented 14.58%. Moreover, the samples included four species belonging to Cryptophyta (8.33%), two species belonging to Euglenophyta, two species belonging to Xanthophyta, and one species belonging to Pyrrophyta. In addition, *t* test results showed that the number of phytoplankton species in winter was markedly lower than that in spring and summer ( $p < 0.05$ ).

The phytoplankton community composition of each sampling site is presented in Fig. 4.

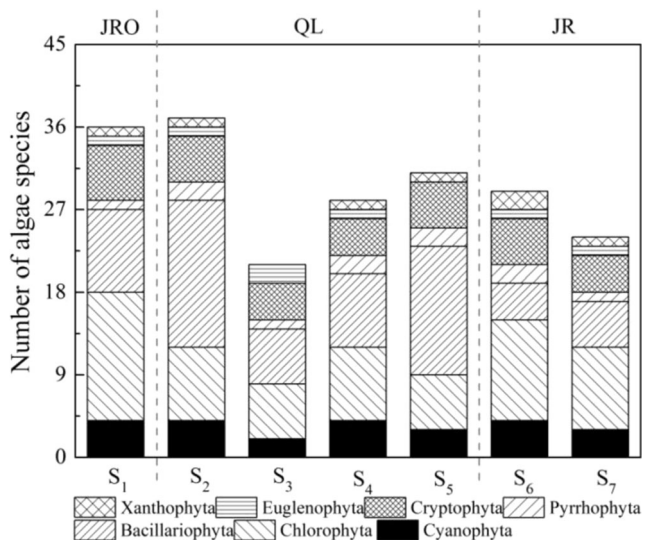
Chlorophyta, Bacillariophyta, Cyanophyta, and Cryptophyta were the four dominant phyla of each site, accounted for 82.76–

91.67% of the total species number; the remaining three taxonomic groups only accounted for 8.33–17.24%. In addition, in term of the total species number, the number of algal species at S<sub>1</sub> and S<sub>2</sub> were more than that in other sites. The species number at S<sub>3</sub> was the least, and no Xanthophyta appeared.

The results of predominant species and dominance value ( $Y > 0.02$ ) of each species at each sampling point are given in Table 1.

According to the calculation results of dominance value, there were 10 predominant species in the three landscape water bodies during this study. Among dominant species, three species belonged to Cyanophyta (i.e., *Oscillatoria tenuis*, *Oscillatoria chlorina* and *Aphanocapsa elachista*), and three species belonged to Chlorophyta (i.e., *Scenedesmus quadricauda*, *Ankistrodesmus spiralis* and *Ankistrodesmus falcatus*). In addition, there was only one predominant species of Bacillariophyta (*Cyclotella meneghiniana*), Cryptophyta (*Komma caudata*), Pyrrophyta (*Gymnodinium*), and Euglenophyta (*Phacus* sp.), respectively. The main dominant species of those landscape water bodies were *Aphanocapsa elachista*, *Cyclotella meneghiniana*, *Oscillatoria tenuis*, and *Gymnodinium*. Furthermore, *Aphanocapsa elachista* was the most dominant species of all sites except S<sub>1</sub>, and the dominance value (*Y*) ranged from 0.26 to 0.48. However, the most dominant species of S<sub>1</sub> was *Cyclotella meneghiniana*.

As shown in Fig. 5, there were distinct spatial changes in phytoplankton density of the three landscape waters. The highest phytoplankton density (up to  $417 \times 10^6$  cells/L) appeared at S<sub>3</sub> of QL, whereas the lowest density (only  $103 \times 10^6$  cells/L) appeared at S<sub>6</sub> of JR. In addition, the total phytoplankton density of QL was much higher than that of JR and



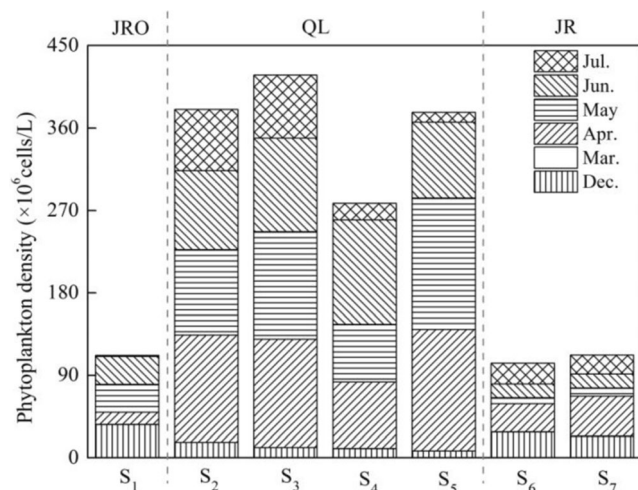
**Fig. 4** Phytoplankton species number of each sampling site

**Table 1** Predominant species and dominance value (*Y*) of phytoplankton at each sampling site

Phyla	Dominant species	<i>Y</i>						
		S <sub>1</sub>	S <sub>2</sub>	S <sub>3</sub>	S <sub>4</sub>	S <sub>5</sub>	S <sub>6</sub>	S <sub>7</sub>
Cyanophyta	<i>Oscillatoria tenuis</i>	0.14	0.07	–	–	–	–	–
	<i>Oscillatoria chlorina</i>	–	–	–	–	–	0.05	–
	<i>Aphanocapsa elachista</i>	–	0.41	0.48	0.45	0.43	0.26	0.46
Chlorophyta	<i>Scenedesmus quadricauda</i>	0.05	–	–	–	–	–	–
	<i>Ankistrodesmus spiralis</i>	–	–	0.05	0.02	–	–	–
	<i>Ankistrodesmus falcatus</i>	–	–	–	0.05	–	–	–
Bacillariophyta	<i>Cyclotella meneghiniana</i>	0.21	–	–	–	–	–	0.03
Cryptophyta	<i>Komma caudata</i>	–	0.05	–	0.02	0.04	–	–
Pyrrhophyta	<i>Gymnodinium</i>	0.14	–	–	–	–	–	–
Euglenophyta	<i>Phacus</i> sp.	–	–	–	0.03	–	–	–

JRO. However, phytoplankton density of JR and JRO was higher than that of QL in December.

As shown in Fig. 6, the relative abundance of Cyanophyta and Bacillariophyta was significantly higher than other phyla ( $p < 0.05$ ). The highest relative abundance of phytoplankton at all sampling sites was Cyanophyta, accounting for 40.18–72.5% of the total algae cells density. Meanwhile, it could also be found that the relative abundance of Bacillariophyta ranked as second at all sampling sites, varying from 7.70 to 27.58%. Moreover, the relative abundance of Cyanophyta in JRO was much less than that in QL and JR, while the relative abundance of Bacillariophyta in JRO was higher than that in QL and JR. Additionally, the relative abundance of Cyanophyta in QL was the highest among the three water bodies, indicating that the eutrophication level of the lake was serious, and the community structure of phytoplankton showed the single species predominance. Besides, the relative abundance of Chlorophyta in QL was lower than that in JRO and JR.

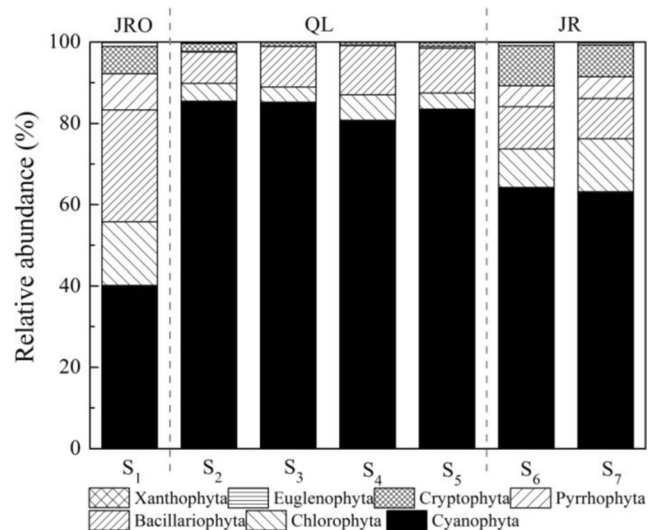


**Fig. 5** Total density of phytoplankton of each sampling site during the study period

The Shannon–Wiener diversity index (*H*) was used to estimate the phytoplankton diversity, as shown in Fig. 7. The diversity indexes of phytoplankton in the studied water bodies ranged from 0.56 to 1.42, with an average of 1.11 (Fig. 7). The highest diversity index was observed at S<sub>1</sub>, whereas the lowest was recorded at S<sub>3</sub>. Moreover, the mean diversity index of phytoplankton in QL was lower than that in JRO and JR.

**Relationship between phytoplankton community and environmental factors**

Before the DCA and RDA analysis, a total of 10 dominant phytoplankton species were selected according to the frequencies and the density of phytoplankton. The DCA of the species data indicated that the maximum gradient length of the four axes was 2.5 ( $< 3$ ). Thus, a redundancy analysis (RDA), as a linear ordination method, was subsequently selected to test the relationship between



**Fig. 6** The relative abundance of phytoplankton at each sampling site

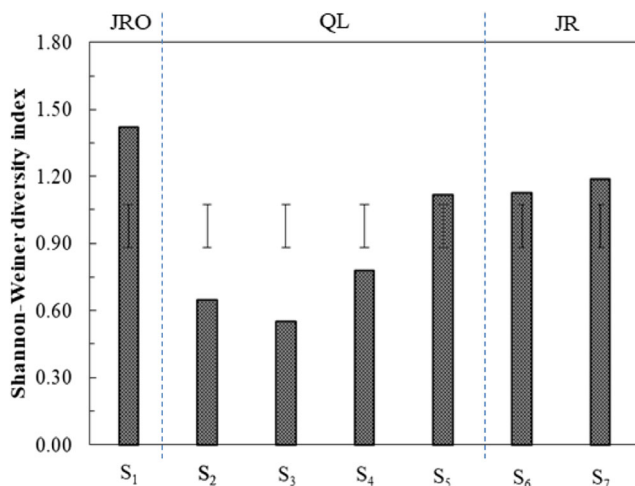


Fig. 7 The diversity index ( $H$ ) of phytoplankton at each sampling site

phytoplankton and environmental variables, as well as to further understand the driving factors that control phytoplankton community structure (Lepš and Šmilauer, Lepš and Šmilauer 2003; Wang et al. 2015).

In order to select the environmental factors which could significantly explain the distribution of phytoplankton species, the forward selection and Monte Carlo permutation test for environment variables were conducted to ensure that the variance inflation factors for all selected environmental variables were less than 20 (Ou et al. 2014). As a result, a total of 12 environmental variables were selected for the RDA analysis. The results of RDA biplot of all sites combined are shown in Fig. 8, and the summary statistics of RDA for the first two axes of RDA are illustrated in Table 2.

As shown in Table 2, the eigenvalues of axis 1 and axis 2 were 0.262 and 0.062, respectively. Together, these eigenvalues explained 26.4% of total variance of phytoplankton distribution within the sampling period. The values of species–environment correlations for axis 1 and axis 2 were 0.807 and 0.788, respectively, indicating that there was a significant positive correlation between them. The correlation between the first two environment axes was 0. Moreover, the first two species axes were approximately vertical, with a correlation of the correlation of 0.010. These results showed that the ordination results were credible because the linear combinations of ordination axes and environmental factors provided a good representation of the relationships between species and environmental factors (ter-Braak 1986).

As shown in Fig. 8, axis 1 was most positively correlated with salinity ( $r = 0.35$ ) and negatively related to DTP ( $r = -0.30$ ). Axis 2 was strongly correlated with WT, DTP, TN, TP, and pH, with the correlation coefficients of  $-0.53$ ,  $-0.45$ ,  $-0.33$ ,  $-0.31$ , and  $-0.24$ , respectively. The results indicated that these environmental variables were closely

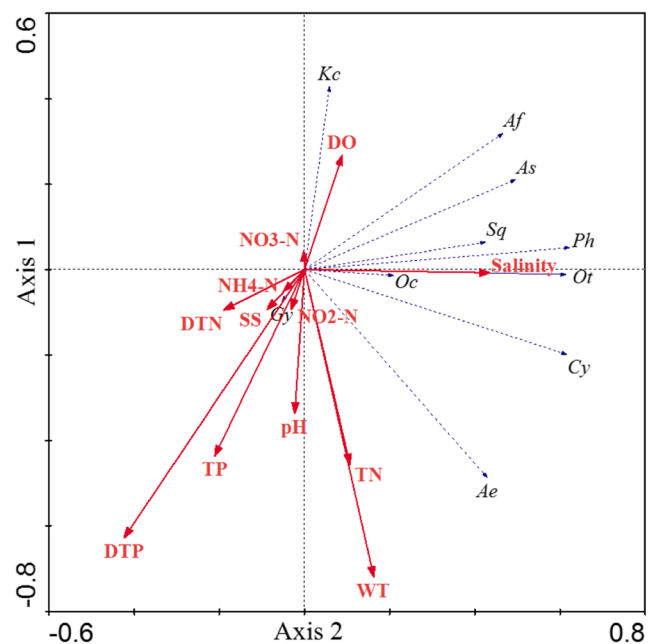


Fig. 8 Biplot diagram for RDA of the relationship between phytoplankton species (dashed lines with arrowhead) and selected environmental variables (red lines with arrowhead) in the studied water bodies. (dominant phytoplankton species: *Ot* *Oscillatoria tenuis*, *Oc* *Oscillatoria chlorina*, *Ae* *Aphanocapsa elachista*, *Sq* *Scenedesmus quadricauda*, *As* *Ankistrodesmus spiralis*, *Af* *Ankistrodesmus falcatus*, *Cy* *Cyclotella meneghiniana*, *Kc* *Komma caudata*, *Gy* *Gymnodinium*, *Ph* *Phacus* sp.)

related to phytoplankton community composition of these landscape water bodies. Moreover, the Monte Carlo test revealed that WT, DTP, salinity, and TN had significant influence on the phytoplankton community ( $p < 0.05$ ). In addition, it could be found that most species belonging to Cyanophyta, especially *Aphanocapsa elachista*, were positively correlated with WT and salinity. *Aphanocapsa elachista* was also positively correlated with nutrients, especially TN. *Cyclotella meneghiniana* was positively related to salinity and negatively related to TP. Furthermore, *Ankistrodesmus falcatus* and *Ankistrodesmus spiralis* showed a negative correlation with DTP and TP. *Scenedesmus quadricauda* and *Phacus* sp. were negatively correlated with DTN.

Table 2 Summary statistics for the first two axes of RDA performed on phytoplankton and environmental variables in these water bodies

Axes	1	2
Eigenvalues	0.202	0.062
Species–environment correlations	0.807	0.788
Cumulative percentage variance		
Of species data (%)	20.2	26.4
Of species–environment relation (%)	54.7	71.5

## Discussion

The three landscape water bodies were aquatic ecosystems with specific characteristics. High and often variable salinity was an obvious feature of these studied water bodies. They had a wide range of salinity, ranging from 0.80 g/L to 29.96 g/L. It was noteworthy that the maximum salinity of JRO (29.96 g/L) was close to the sea water (32.5 g/L; Fathi and Flower 2005). The results indicated that these studied water bodies, especially JRO, were typical landscape water bodies with high salinity. The high salinity content was related to the study area which was located in Tianjin Binhai New Area, where the salinity was high in the soil. Additionally, the salinity content of those landscape waters had a significant difference among regions ( $p < 0.05$ ). Highest salinity content was recorded in JRO, which may be as a result of the high level salt in the soil in the beach area and tidal intrusion. Compared with JRO, the salinity of QL was much lower, which was due to the replenishment using desalinated water, resulting in effective salinity reduction in water and soil (Xiong et al. 2016). The salinity content of JR was lowest, which was mainly due to the fact that a large amount rainfall runoff in the JR Basin entered into JR. Furthermore, it could be found that mean salinity content of JRO in summer was significantly higher than winter, which may be ascribed to the high evaporation in summer. Overall, differences among sites and seasonal variations reflected a balance among terrestrial contributions, seawater, desalinated water, precipitation, and evaporation.

## Phytoplankton community

In this study, a total of 48 species of phytoplankton belonging to 7 phyla were identified in those landscape water bodies during the sampling period. Among these species, Chlorophyta and Bacillariophyta accounted for 35.42 and 31.25% of the total species, respectively. But the relative abundance of Cyanophyta and Bacillariophyta were significantly higher than other phyla, accounting for 84.50%. Thus, the results indicated that all the three landscape waters belonged to Cyanophyta–Bacillariophyta type lakes. This scenario was consistent with observations from other hypersaline lakes, such as Lake Alchichica in Mexico (Oliva et al. 2001), Lake Qarun in Egypt (Fathi and Flower 2005), salt lakes Tibetan, China (Zhao et al. 2005), and saline waters in North Hebei, China (Zhao and He 1999). In addition, the relative abundance of Cyanophyta was the largest at each sampling point, which led to the dominant position of Cyanophyta in the structure of phytoplankton community.

In addition, the main dominant species of these landscape water bodies were *Aphanocapsa elachista*, *Cyclotella meneghiniana*, *Oscillatoria tenuis*, and *Gymnodinium*. It was reported that *Aphanocapsa elachista* was frequently a dominant species in moderately eutrophic shallow lakes (Becker et

al. 2010), and the optimum range of growth temperature was 10–26 °C (Song 2014). Besides, from a functional point of view, *Aphanocapsa elachista* was a non-N fixing cyanobacteria, as represented in group K, adapted to shallow and nutrient-rich water in Reynolds' classification (Reynolds et al. 2002). Because the studied water bodies were shallow and nutrient-rich water, *Aphanocapsa elachista* was a dominant species in these waters, especially in QL and JR. Furthermore, *Cyclotella meneghiniana* was commonly found in other saline waters lakes (Zhao and He 1999; Gasiūnaitė et al. 2005) and cited as a reference diatom used for indication of primary productivity in temperate lakes of differing trophic levels (Gurbuz et al. 2003). This species had small size, high surface–volume ratio and efficient light-harvesting capacity, which may significantly decrease its sinking rate (Reynolds et al. 2002; Reynolds 2006). Thus, it could compete favorably with other algae in well-mixed waters. Zhao et al. (2005) found that *Oscillatoria tenuis* was a typical hyposaline or halobiont species with a broad adaptation to salinity. Thus, *Oscillatoria tenuis* was the most important or dominant algae in hypersaline waters (Zhao and He 1999; Zhao et al. 2005). Gasiūnaitė et al. (2005) reported that Dinophyte-dominated community appeared in locations with salinity above 5, and it was generally most frequent during spring in the Lithuanian coastal waters (dominated by *Gymnodinium*). *Gymnodinium* was a dominant species in JRO, which may be related to the high salinity of JRO ( $> 5$  g/L).

## The trophic state of these landscape water bodies

It was reported that the density of phytoplankton was used to reflect the trophic state of water body (Kuang et al. 2005; Zhang and Zang 2015). Thus, according to the average algae cell density for the three water bodies ( $42.39 \times 10^6$  cells/L), these landscape water bodies belonged to moderate eutrophication (Kuang et al. 2005; Zhang and Zang 2015). Moreover, in terms of each water body, the trophic state of each water body was different. The average algae cell density in JRO, QL, and JR were  $18.63 \times 10^6$ ,  $60.56 \times 10^6$ , and  $17.94 \times 10^6$  cells/L, respectively, which showed that QL belonged to moderate eutrophication, while the water quality of JRO and JR was in a mesotrophic state (Kuang et al. 2005; Zhang and Zang 2015).

Diversity index of phytoplankton could also be used as an indicator of water quality (Karydis and Tsirtsis 1996; Wang et al. 2015). Tian et al. (2013b) reported that a high diversity index ( $H$ ) indicated a more healthy ecosystem, while a low value suggested a less healthy or degraded ecosystem. In this study, the average phytoplankton diversity index ( $H = 1.12$ ) for these studied water bodies was low, indicating that these landscape water bodies were in moderate pollution and had a less healthy ecosystem (Tian et al. 2013b), which was in agreement with the results of the algae density evaluation. In

addition, the average Shannon–Wiener diversity indexes in JRO, QL, and JR were 1.42, 0.78, and 1.16, respectively, which showed that QL was in heavy pollution, while the water quality of JRO and JR belonged to moderate pollution. The water quality of QL was worse than that of other two water bodies, which may be due to the fact that the relative abundance of Cyanophyta in QL was higher than that in JRO and JR (Fig. 6).

### Impacts of salinity on phytoplankton community

It was reported that the phytoplankton diversity index and total species number of phytoplankton tended to decrease with increasing salinity in salt lakes (Zhao et al. 2005). Zhao et al. (2016) found 41 algal species in Bange Lake ranging in salinity from 14 to 146 g/L; Zhao et al. (2016) also reported 53 algal genera in Qinghai Lake (mean salinity 12.5 g/L); and Zhao et al. (2005) identified 38 algal species in saline lakes in northern Tibet (salinity > 50.0 g/L). In this study, 48 species of phytoplankton were recorded in these landscape water bodies, which were lower than that of other freshwater bodies, such as Chaohu Lake (97 species, Jiang et al. 2014), Dongping Lake (132 species, Tian et al. 2013b), Shengjin Lake (192 species, Wang et al. 2015), Baiyangdian Lake (152 species, Liu et al. 2010), and Sau Reservoir in Spain (98 species, Becker et al. 2010). In addition, it could also be found that the mean diversity index of phytoplankton in these landscape water bodies ( $H = 1.11$ ) was much less than that of other freshwater bodies, such as Nansi Lake ( $H = 4.33$ , Tian et al. 2013a), Dongping Lake ( $H = 3.82$ , Tian et al. 2013b), and Shengjin Lake ( $H = 5.58$ , Wang et al. 2015). The results confirmed that increasing salinity would lead to the bio-diversity decrease of phytoplankton. Besides, although the species number of Chlorophyta was the most, its density was lower than Bacillariophyta during the whole study. This phenomena may be largely due to the higher level of salinity in the selected water bodies than the other water types. Diatoms were more tolerant in the harsh environment (Larson and Belovsky 2013). Moreover, the algal species number of  $S_1$  was the most but the density was the lowest. This phenomenon may be related to the high salinity in JRO which could inhibit the growth of phytoplankton (Gasiūnaitė et al. 2005; Larson and Belovsky 2013; Williams et al. 1990).

In terms of dominant species, *Cyclotella meneghiniana* belonging to Bacillariophyta was the dominant species at JRO, while the predominant species at QL and JR was *Aphanocapsa elachista* belonging to Cyanophyta. The salinity content of JRO was significantly higher than that of QL and JR. The results also showed that salinity was a key factor influencing the phytoplankton community. Bacillariophyta with hard siliceous shells were more resistant than other phytoplankton species and were more adaptable to adverse water environment (Chang et al. 2011). However, some Cyanophyta

species were tolerant of a wide range of salinities (Wasmund et al. 1999), which could explain that *Aphanocapsa elachista* (Cyanophyta) was the predominant species at QL and JR.

In general, rates of growth and primary production declined severely with increasing salinity (Gasiūnaitė et al. 2005; Lehtimäki et al. 1997; Wasmund 1997).

The salinity content of the studied water bodies was much higher than the normal landscape water body, so most phytoplankton were not able to grow properly. Mean total algae cells density and biomass decreased along the salinity gradient (Evagelopoulos et al. 2007). With the increase of salinity, species diversity would decrease continually (Larson and Belovsky 2013; Williams et al. 1990). A simple community mainly composed of salt-tolerant or halophilic phytoplankton species would replace the complex phytoplankton structure composed of a variety of phytoplankton (Evagelopoulos et al. 2007). Therefore, Cyanophyta and Bacillariophyta were common in these landscape water bodies, especially Cyanophyta, which thrived at higher alkalinity and pH; some could even be able to grow in 300 g/L (Zhao et al. 2005). Some algae, e.g., *Oscillatoria tenuis*, *Oscillatoria chlorina*, *Aphanocapsa elachista*, and *Cyclotella meneghiniana* were the most important or dominant algae in hypersaline waters.

### Relationship between phytoplankton community and environmental factors

There was a significant difference of phytoplankton density in different seasons, presenting high in spring and summer, and low in winter, which showed a great relationship with the water quality. This phenomenon may be related to the higher contents of nutrients in spring and summer than that in winter. In a shallow water body with rich nutrients, Cyanophyta were generally dominating the bloom peaks during summer (Gasiūnaitė et al. 2005). Additionally, the RDA result indicated that the phytoplankton in the studied water bodies was regulated by DTP, TP, and TN. Thus, nutrients, as the material foundation for the growth of phytoplankton, were the main factors influencing phytoplankton community and distribution.

Moreover, WT was the driving factor determining the succession of phytoplankton. Tian et al. (2013b) reported that WT was the key factor driving changes in phytoplankton community composition in Nansi Lake. The density of phytoplankton increased with the increase of water temperature (Gotham 1981; Wang et al. 2015). From the results of RDA analysis, it could be found that Cyanophyta and Chlorophyta were positively correlated with WT. Therefore, the increased abundance of phytoplankton in summer was mainly due to the increase in Cyanophyta and Chlorophyta species, which preferred high temperature (Wang et al. 2015). In addition, the number of phytoplankton species was more in spring and

summer, but less in winter because of the lower water temperature.

Several studies indicated that Cyanophyta grew better when  $\text{pH} > 8$  (Brettum 1996). The pH value of the selected water bodies except JRO was more than 8. Thus, these landscape water bodies were alkaline waters which were suitable for the growth and reproduction of Cyanophyta. The results of RDA also suggested that the great majority of Cyanophyta, especially *Aphanocapsa elachista*, were positively correlated with pH.

In addition, the N/P ratio may affect the taxonomy composition and the structure of domination in phytoplankton (Daines et al. 2014; Reynolds 2006; Rosińska et al. 2017). The average mass ratios of TN/TP at JRO, QL, and JR during the study period were 23.0, 16.0, and 23.4, respectively. The high mass ratio of TN/TP confirmed that phytoplankton growth was most likely constrained by phosphorus (Reynolds 2006).

## Conclusions

This study investigated the variations of environmental factors in the three landscape water bodies and their influences on phytoplankton community. A total of 48 species of phytoplankton belonging to 7 phyla were identified during the study period. Among these species, 35.42% of the total species belonged to Chlorophyta and 31.25% of the total species belonged to Bacillariophyta. *Cyclotella meneghiniana* (Bacillariophyta) and *Aphanocapsa elachista* (Cyanophyta) were the main dominant species. These landscape water bodies belonged to Cyanophyta–Bacillariophyta type lakes. However, the Shannon–Wiener diversity index ( $H$ ) varied from 0.56 to 1.42, with an average of 1.11, revealing low diverse phytoplankton community. In addition, the mean algae cell density was  $42.39 \times 10^6$  cells/L, suggesting that these landscape water bodies belonged to moderate eutrophication. Furthermore, WT, DTP, salinity, and TN were the main environmental factors influencing the phytoplankton community. *Aphanocapsa elachista* was positively correlated with WT, TN, and salinity, while *Cyclotella meneghiniana* was positively related to salinity and negatively related to TP. The findings from the present study would provide baseline data for research on phytoplankton community of landscape water with high salinity.

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