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Phytoplankton functional groups in a high spatial heterogeneity subtropical reservoir in China

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ABSTRACT

Traditional classification of phytoplankton assemblages does not adequately reflect their ecological function in aquatic ecosystems. Therefore, we used a functional group approach based on the Q index to analyze the spatial and temporal patterns, environmental factors and the ecological status of phytoplankton in the Danjiangkou Reservoir (DJKR) of China in July 2011 to April 2012. Groups B (*Cyclotella* species), C (*Asterionella formosa* and *Cyclotella meneghiniana*), D (*Stephanodiscus hantzschii*, *Synedra acus* and *Nitzschia* sp.), Lo (mainly *Peridiniopsis* spp.), P (*Fragilaria capucina* and *Aulacoseira* spp.) and Y (*Cryptomonas ovata* and *Cryptomonas erosa*) were classified as dominant phytoplankton functional groups. We ran redundancy analysis and Pearson correlation analysis to assess the relationship between functional groups and environmental factors. Total phosphorus, pH and soluble silicate were apparently the key factors driving variation in phytoplankton functional groups. The analyses suggest P-limitation of phytoplankton growth. The Q index based on functional groups indicated the relatively good ecological status of the DJKR which varied as a function of the mixing regime; most samples were scored as medium or high quality according to the Q index evaluation ($Q > 2$). This is the first application of the assemblage index to a water supply reservoir for the Middle Route Project (MRP) for South-to-North Water Transfer (SNWT) in China.

Introduction

Damming of rivers for water conservancy projects has altered the natural hydrologic regime and led to the discontinuity of river morphologies which may cause the reduction of diversity of river communities and change phytoplankton species assemblages substantially (Dong, 2003; Burford and O'Donohue, 2006). It is widely recognized that three zones (the riverine zone, the transitional zone and the lacustrine zone) can be distinguished in large dendritic reservoirs along the direction of flow (Thornton et al., 1990; Shao et al., 2010). Numerous physical (light availability, mixing regime) and chemical (nutrient) factors have shown remarkable longitudinal changes from the riverine zone to the lacustrine zone in reservoirs, which promotes spatial heterogeneity of phytoplankton (Wetzel, 2001; Xu et al., 2012).

The biomass and composition of species and the community structure of phytoplankton directly affect the ecosystem structure and energy transfer efficiency through food chains (food chain efficiency) of aquatic ecosystems (Dickman et al., 2008). Phytoplankton communities are also a valuable monitoring tool in water resources management. Phytoplankton community structure reflects the status of the aquatic environment and helps in our understanding of the variation in aquatic ecosystems (De Stasio and Richman, 1998; Gillett and Steinman, 2011). The use of phytoplankton in water quality monitoring has a long history because the spatial and temporal variations of phytoplankton are closely related to environmental factors (Padisák et al., 2006; Wu et al., 2013). Changes in environmental factors in the system can directly affect phytoplankton community structure (Lepistö et al., 2004). Phytoplankton growth in aquatic ecosystems is determined mainly by availability of light, nutrients, and water temperature although hydrodynamic conditions, grazing, climate factors, and human activity in the watershed can also play important roles in phytoplankton succession (Peréz-Martínez and Cruz-Pizarro, 1995; Reynolds, 2006; Becker et al., 2010). As a result, phytoplankton community composition has long been used to study and understand aquatic ecosystems (ÓSullivan and Reynolds, 2003).

Traditional classification of phytoplankton allocates biomass into the major taxonomic classes and does not reflect directly their ecological function in the ecosystem (Reynolds, 1997; Costa et al., 2009). Therefore, a functional group approach, based on functional characteristics, has attracted increasing attention in recent years (Padisák et al., 2006; Becker et al., 2010; Wang et al., 2011a; Edwards et al., 2013). Fourteen groups of phytoplankton were identified in Reynolds' original study (Reynolds, 1980); these groups are often polyphyletic and share adaptive features, based on the physiological, morphological and ecological attributes of the species (Reynolds et al., 2002; Stanković et al., 2012). At the present time, the phytoplankton functional group approach uses 39 assemblages, identified by alpha-numeric codes according to

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their sensitivities and tolerances (Reynolds et al., 2002; Padisák et al., 2009) and can potentially be developed to assess water quality more consistently (Wang et al., 2011b). Indeed, this approach proved to be more useful for ecological purposes than the previously applied taxonomic groupings (Salmaso and Padisák, 2007; Stević et al., 2013).

The phytoplankton functional group approach has been widely applied to aquatic systems and has provided important information for understanding the dynamics of pelagic communities in temperate (Çelik and Ongun, 2008; Borics et al., 2012), tropical (Sarmento et al., 2006; Costa et al., 2009), subtropical (Burford and O'Donohue, 2006; Devercelli and O'Farrell, 2013), and Mediterranean regions (Vila and Maso, 2005; Becker et al., 2010). The phytoplankton functional group approach, originally developed for lakes, is also useful for reservoirs because both systems share many characteristics (Reynolds, 1999a). On the basis of recent developments in phytoplankton functional classification and typology of lakes, Padisák et al. (2006) developed a functional index (Q index) to assess the ecological status of different lake types, established by the Water Framework Directive (WFD) of the European Union. The Q index provides five grades of quality for an aquatic ecosystem and takes into account the relative contribution of functional groups in the total biomass and a factor number ascribed to each assemblage which can be related to the type of water body (Padisák et al., 2006). More recently the Q index has also been successfully applied in ecological status assessments for several reservoirs (Hajnal and Padisák, 2008; Xiao et al., 2011).

Previous studies of phytoplankton in Danjiangkou Reservoir (DJKR) were focused on community composition and succession of the phytoplankton before and after impoundment, including their biodiversity and spatial and temporal distributions (Borutsky et al., 1959; Wu et al., 1996; Yin et al., 2011). The primary objectives of this study were to illustrate the temporal and spatial distributions of phytoplankton functional groups, and their composition and biomass and to evaluate the ecological status of the reservoir as revealed by the functional group analysis, for the first time in DJKR. Additionally, we also considered the driving forces that are responsible for variation in phytoplankton functional groups. Our research on phytoplankton will also supply useful information for water quality management for other large water diversion projects in China.

Study area, data collection, and methods

Study area

The South-to-North Water Diversion Project (SNWDP) including East, Middle and West Routes is one of the world's largest water diversion projects for water supply. The Middle Route Project (MRP) for South-to-North Water Transfer (SNWT) will divert water, in the near future, from DJKR to Beijing through canals to be built through the Funiu and Taihang Mountains. The MRP will mitigate the water crisis in Beijing, Tianjin and North China by increasing agricultural, municipal and industrial water supply. As the water source of the MRP, the DJKR (32°33′–32°49′ N, 110°59′–111°49′ E), is located on the upstream region of the Hanjiang River, which lies on the border between Hubei, Shanxi and Henan Provinces. A subtropical monsoon climate prevails there (Shen et al., 2011), characterized by hot, rainy summers and cold winters, with an average annual rainfall of 850–950 mm and mean annual temperature of 15.9 °C. DJKR was built in 1958 and completed in 1973 for water supply, flood control and power generation. It has a drainage area of 9.52×10^4 km^2, a normal water level of 157 m A.S.L., a surface area of 750 km^2 and a capacity of 17.45×10^9 m^3. The general outline of the DJKR Basin is a "V" shape (Fig. 1) and includes the very long but narrow Hanjiang River Basin (HRB) and the wide Danjiang River Basin (DRB). The Danjiangkou Dam Extension Project in 2012 increased the dam from its existing crest elevation of 162 up to 176.6 m and the design storage level was raised from 157 to 170 m; the total storage capacity consequently increased to 29×10^9 m^3. Heightening Danjiangkou Dam increased the scope for flood control of middle and lower Hanjiang and assures the safety of Wuhan City and the plain to the north of Hanjiang. However, there will be an additional inundated area of 370 km^2 after the flood season in 2014.

Sampling and data collection

In the present study, 25 sampling sites were selected for the seasonal survey based on the morphology of the reservoir (Fig. 1). A total of 15 sites were in the HRB (H1, H1a, H1b, H2, H3, H4, H5, H6, H7, H8, HL1, HL2, HZ1, HZ2, T1), among which T1, HL1, HL2, HZ1 and HZ2 were located in three small tributaries (Qingtang River, Lang River and Zeng River) respectively. One site was in front of the dam (Dam) and 9 sites were in DRB (D1, D2, D3, D4, D5, D6, D7, D8, TC). Site TC is the water intake for the South to North Water Transfer Project. Because the water level fluctuation can be 30 m or more in the reservoir, some shallow locations and sites could not always be sampled.

From July 2011 to April 2012, seasonal samplings were carried out in summer (July 2011), autumn (October 2011), winter (January 2012) and spring (April 2012). Water for nutrient and phytoplankton analyses was collected at a depth of 0.5 m with a 5-L Van Dorn bottle. Samples for phytoplankton analysis were preserved with 5% formalin and neutral Lugol's solution, while samples for nutrients were stored in pre-cleaned plastic bottles and acidified with sulfuric acid. An additional 200–600 mL of water was filtered onto a microfilter for chlorophyll *a* (chl *a*) determination, and the filter was immediately placed in a dark cooler and packed in ice until analysis (Huang et al., 2000; Cai, 2007). Water temperature (WT), conductivity (Cond), pH, dissolved oxygen (DO), and turbidity (Turb) of the surface water were measured in situ with an Environmental Monitoring System probe (YSI 6600EDS, USA). Vertical profiles of WT and DO were measured with a submersible probe (XR-420, RBR, Canada). Water transparency was measured with a 20-cm Secchi disk. In the laboratory, the following variables were measured with a segmented flow analyzer (Skalar San++, Netherlands); ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total nitrogen (TN), soluble reactive phosphorus (PO_4-P), total phosphorus (TP), and soluble silicate (SiO_2-Si). Total organic carbon (TOC) and dissolved organic carbon (DOC) were measured with a total organic carbon analyzer (Shimadzu TOC-V_{CPH}, Japan). Chl *a* was analyzed following Protocols for Standard Observation and Measurement in Aquatic Ecosystems of Chinese Ecosystem Research Network (CERN) (Huang et al., 2000; Cai, 2007). The pigments are extracted from the algal sample concentrated by filtration in an aqueous solution of acetone. The Chl *a* concentration is determined on a spectrophotometer (Shimadzu UV-1601, Japan) by measuring the absorbance of the extract at various wavelengths (750, 663, 645, and 630 nm).

Phytoplankton was quantitatively analyzed in a Fuchs-Rosenthal slide with an Olympus CX21 microscope (Olympus Corporation, Japan) at 400× magnification, using the sedimentation chamber method for the samples (Huang et al., 2000; Cai, 2007). Taxonomic identification of phytoplankton species was done according to Hu and Wei (2006) and John et al. (2002).

Data analysis

We calculated the euphotic zone depth (Zeu) as 2.7 times the Secchi depth (Cole, 1994). The mixing depth (Zmix) was estimated from temperature profiles measured every 1 m. When a site was stratified, mixing depth was considered as the depth where water temperature changed by 1 °C or more (onset of thermocline). In the absence of stratification, mixing depth was taken equal to the average depth of the reservoir (Scheffer et al., 1997; Naselli-Flores and Barone, 2003). The ratio between the euphotic and mixing depths (Zeu:Zmix) was used as a measure of light availability (Jensen et al., 1994).

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