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ORIGINAL PAPER

Human impacts on the cladoceran community of Jili Lake, arid NW China, over the past century

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Abstract Deterioration of aquatic ecosystems, as a consequence of human-induced disturbances, is a critical global concern. To fully understand the responses of aquatic systems to anthropogenic impacts, it is crucial to assess long-term changes in lakes. The water quality of Jili Lake, a large water body in northwest China, has deteriorated recently, owing to the growing impacts of regional warming and human activities. Thus, Jili Lake was a prime candidate for evaluation of historical multi-stressor impacts. Meteorological data, historical documents, and assemblages of cladoceran microfossils in the sediments of Jili Lake were employed to investigate changes in the cladoceran community over the past century, and to evaluate the response of that aquatic community to human activities. From the 1920s to the 1950s, species richness of the cladoceran community was high, which reflected conditions of relatively low human impact. From the 1960s to 1970s, a sharp decrease in *Bosmina longirostris*, a planktonic cladoceran species, suggested a decrease in water level as a result of dam construction and intensified water exploitation. Since the 1980s, the water level in the lake has been restored, but increased fish farming and construction of a water storage facility caused salinisation and eutrophication of Jili Lake. Accordingly, the cladoceran community displayed distinct signs of a regime shift, with a gradual transition to dominance of *B. longirostris* and a sharp decrease in littoral species (e.g. *Leydigia leydigi*, *L. acanthocercoides*, *Alona quadrangularis*, *Alona affinis*). Our results suggest that human-induced disturbances were the main factor that drove changes in the cladoceran community since about the mid-20th century.

Keywords Jili lake · Cladoceran community · Human activities · Water level · Eutrophication · Fish farming

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Introduction

During the past century, many lakes have faced multiple stressors, caused by human activities and global climate change, which have led to responses such as phytoplankton blooms and loss of submerged aquatic vegetation (Zhang et al. 2017; Jeff et al. 2019). Lakes in arid areas are of particular concern, owing to the shortage of water resources in such regions. Excessive utilisation of water resources and/or global warming-induced high evaporation, however, have led to significant shrinkage and salinisation of many lakes in arid areas, thereby resulting in aquatic ecosystem degradation (Aladin 1991; Wu et al. 2009, Wu and Ma 2011). In addition, many freshwater lakes in arid areas, which support important fisheries, have faced ecological problems such as salinisation and eutrophication, caused by fish farming (Wu et al. 2013). Therefore, a thorough understanding of changes that have occurred in lakes subjected to multiple stressors is of great significance to the protection and restoration of these ecosystems.

Paleolimnological methods have been used increasingly to assess long-term environmental changes in lakes. Cladocera, an important group of crustacean zooplankton that links primary producers to higher consumers, play a vital role in lake ecological dynamics (Lampert 1997). Because cladoceran subfossil remains are often well preserved in sediments, analysis of such remains (e.g. carapaces, post-abdomens, claws, antennae, and ephippia) can provide reliable information regarding shifts in cladoceran assemblages over time (Whiteside 1970; Frey 1986; Jeppesen et al. 2001a, b; Korhola and Rautio 2001; Davidson et al. 2007). Cladocerans are known to respond to environmental changes caused by natural and anthropogenic forcing, such as eutrophication (Hofmann 1996; Chen et al. 2010), fish predation (Jeppsen et al. 1996; Shi et al. 2016), and changes in salinity (Amsinck et al. 2005; Brucet et al. 2009). Therefore, cladocerans are widely used to assess anthropogenic impacts on aquatic systems (Jeppesen et al. 2001a, b; Korhola and Rautio 2001).

Jili Lake is a large water body in north Xinjiang, China. It supports an important fishery and is a crucial water source in the region. During the past decades, however, Jili Lake has experienced water quality deterioration and lake ecosystem degradation as a consequence of human-induced salinisation and eutrophication (Dong et al. 2008; Liu 2015; Cheng et al. 2016; Ji et al. 2018). In addition, many water conservation projects have been conducted in the watershed (Ren 1990; Liu et al. 2002; Tang et al. 2009). Although these activities likely had substantial impacts on the lake, the ecological consequences had not yet been properly evaluated, owing to insufficient monitoring data. Here, we present the results of analyses of cladoceran microfossils in a sediment core from Jili Lake, in combination with information from historical documents, to infer changes in the cladoceran community over the past century, and to evaluate the impact of human activities and climate changes in bringing about those biotic shifts.

Study area

Jili Lake (46° 51′–47° 00′ N, 87° 20′–87° 32′ E) is situated in southern Fuhai County, Xinjiang Province, northwest China (Fig. 1). It is a large lake that lies ~ 483 m above sea level. It has a surface area of 174 km^2, and its maximum and average depths are 14.7 m and 9.9 m, respectively (Jiang et al. 2010). From 1959 to 2017, mean annual air temperature in this area was 4.3 °C (Fig. 2a), and mean annual precipitation was ~ 126 mm (Fig. 2b). Mean annual air temperature increased sharply during the past several decades (Fig. 2a). Jili Lake receives water mainly from the Ulungur River, and it outflows to Ulungur Lake via the Kuyiga River, in the northwest (Fig. 1b). Historically, Ulungur Lake had higher salinity and nutrient concentrations than did Jili Lake (pH: 7.1, EC: 2354 μS/cm, TDS: 1516 mg/L, DO: 99.7% saturation, measured in 2018 AD) (Dong et al. 2008; Cheng et al. 2016). The Irtysh River, which is the only river in China that flows into the Arctic Ocean, is only several dozen km northeast of Ulungur Lake (Fig. 1b). In 1988, the government implemented a project that channeled the Irtysh River into Ulungur Lake. This project made it possible for the water level in Ulungur Lake to rise rapidly in a short period of time and even exceed the water level in Jili Lake (Fig. 2d), which enables the saline water of Ulungur Lake to flow into Jili Lake.

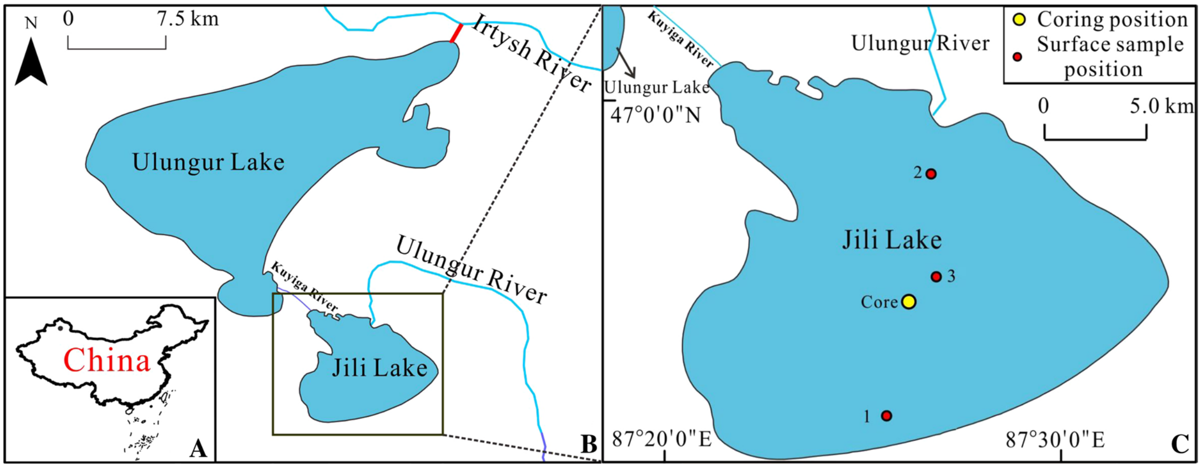


Fig. 1 **a** General location of Jili Lake in China (black dot in top left corner). **b** location of Jili Lake and Ulungur Lake, the red line represents the project that channeled the Irtysh River into Ulungur Lake. **c** locations where the core (yellow dot) and surface samples (red dots) were collected

With the increasing water consumption by households and for agricultural irrigation, 59 diversion canals, five dams, one large and six small and medium reservoirs were built along the banks of the Ulungur River since the 1960s (Ren 1990). This caused about a 61 % water loss between Ertai Station (upstream of the Ulungur River) and Fuhai Station (downstream of the Ulungur River), and reduced water levels in Jili Lake and Ulungur Lake (Fig. 2c and d) (Ren 1990). As a result of the continuous reduction in size of Jili Lake, and increased seepage of saline water from farmland drainage and channels into the lake, salinity in Jili Lake has increased steadily since the 1960s (Fig. 2e). The water level of Jili Lake gradually recovered, owing to the project that channeled the Irtysh River into Ulungur Lake in 1988 (Fig. 2c). This channel project, however, also increased the salinity of Jili Lake (Dong et al. 2008; Cheng et al. 2016) because of the input of saline water from Ulungur Lake (Fig. 2e). According to monitoring data from 2006 to 2008, Jili Lake changed from a freshwater lake to a brackish lake, with elevated concentrations of SO_4^{2-} and Cl^- (Dong et al. 2008). The trophic state of Jili Lake also changed gradually, because of increasing nutrient supply after the 1960s (Fan 1984; Ma 1985; Dong et al. 2008). Jili lake was oligotrophic until the 1960s, shifted to a mesotrophic state in the late 1980s (Ren 1990), and became eutrophic in the 2000s (TN: 910 μg/L; TP: 80 μg/L) (Dong et al. 2008).

Macrophyte cover in Jili Lake has decreased substantially over the past several decades. A survey conducted between 1986 and 1987 showed that there were 10 macrophyte species in the lake, and that *Phragmites communis*, *Myriophyllum spicatum*, and *Potamogeton perfoliatus* were the dominant taxa (Ren 1990). There were, however, few macrophytes, except for sparse coverage of *P. communis* in the northwest of Jili Lake, during sampling in 2018. There was no fish farming in Jili Lake before 1958. The native fish community in Jili Lake was dominated by *Perca fluviatilis*, *Leuciscus baicalensis*, *Carassius auratus gibelio*, *Tincatinca* spp., *Cobitis granoei*, *Barbatula nuda*, and *Cobitis taenia sibirica*. Fish farming began with the establishment of the Fuhai Fishery in 1958. Since the mid-1960s, multiple invasive species (*Abramis brama orientalis*, *Rutilusruti luslacustris*, *Esox lucius*, and *Hypomesus olidus*) were introduced to Jili Lake to increase fish production. Since the late 1980s, fish farming in Jili Lake developed rapidly, with the introduction of intensive fish culture. Both the scale of farming and yields of fish increased substantially, with annual yield sometimes exceeding 1000 t (Fig. 2f). *H. olidus* became established soon after introduction to Jili Lake, accounting for 60 % of the total fish yield, and thereby becoming the dominant

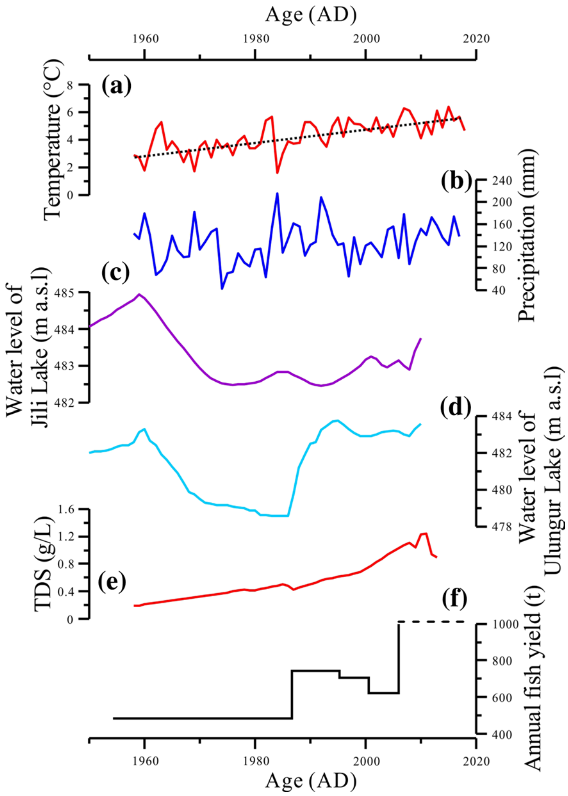


Fig. 2 **a** Historical data for annual temperature, **b** annual precipitation, **c** water level in Jili Lake, **d** water level in Ulungur Lake, **e** total dissolved solids (TDS), and **f** fish yield in Jili Lake. Annual temperature and precipitation data are from the China Meteorological Data Service Center. Water level data for Jili Lake and Ulungur Lake and TDS data for Jili Lake are from Cheng et al. (2016). Annual fish yield data are from Tang et al. (2009)

species. Native *P. fluviatilis*, on the other hand, became endangered in the 2000s (Tang et al. 2009).

Materials and methods

A 355-cm-long core (JL18-01-A) was collected with a modified piston corer, at a 16.6-m-deep site in the centre of Jili Lake in February 2018. The sediment-water interface was kept intact. The uppermost 30 cm of the core were sectioned at 1-cm intervals for dating and chemical analyses, and at 2-cm intervals for cladoceran microfossil analyses. In addition, surface sediment samples (top 1 cm) were retrieved using a grab sampler from three additional sites, to be used for cladoceran microfossil analysis (Fig. 1 c). Sediment core dating was conducted at the Key Laboratory of Lake Sediment and Environment, Nanjing Institute of Geography and Limnology, Nanjing, China. Freeze-dried sediments were analysed for ^{210}Pb, ^{226}Ra, and ^{137}Cs by direct gamma spectrometry, using Ortec HPGe GWL series, well-type, coaxial, low-background, intrinsic germanium detectors (Appleby et al. 1986). Total ^{210}Pb activity was determined via its direct gamma emissions at 46.5 keV, and ^{226}Ra activity (supported ^{210}Pb activity) was determined via the 295.2 and 351.9 keV emissions of its daughter isotope, ^{214}Pb, following three weeks of storage in sealed containers to enable secular equilibrium to be attained between ^{226}Ra and ^{214}Pb. Emissions of ^{137}Cs were measured at 662.7 keV. Supported ^{210}Pb in each sample was assumed to be in equilibrium with *in situ* ^{226}Ra. Unsupported ^{210}Pb activity at each depth was calculated by subtracting the ^{226}Ra activity from the total ^{210}Pb activity (Wu et al. 2004). Organic matter content in the sediment samples was measured by loss-on-ignition. Approximately 0.5 g of freeze-dried sediment from each interval was oven-dried at 105 °C, weighed, combusted at 550 °C for 4 h, and weighed again to estimate organic matter content.

Cladoceran remains from core sediments were processed following standard methods (Frey 1986; Korhola and Rautio 2001). Approximately 0.1–0.2 g of freeze-dried sediment from each interval was deflocculated with 50 mL of 10% KOH at 60 °C for 45 min. Samples were then filtered through a 38-µm sieve with distilled water to retain the cladoceran remains, which were then carefully transferred to a centrifuge tube and filled to 10 mL with distilled water, to which a drop of formaldehyde was added as a preservative. The centrifuge tube was carefully shaken before transferring 0.05-mL aliquots to a microscope slide with a precision pipette. Samples were stained with safranine and mounted with gelatin glycerin. The cladoceran remains were examined under a Leica® microscope at 200× magnification using bright-field optics. Cladoceran species identifications and nomenclature were based mostly on those proposed by Szeroczynska and Sarmaja-Korjonen (2007) and Jiang and Du (1979). At least 50 individuals were identified in each sample (Fig. 3). Cladoceran abundance was calculated in two ways, as absolute abundance (number per g dry weight) and relative percentages, using the number of the most

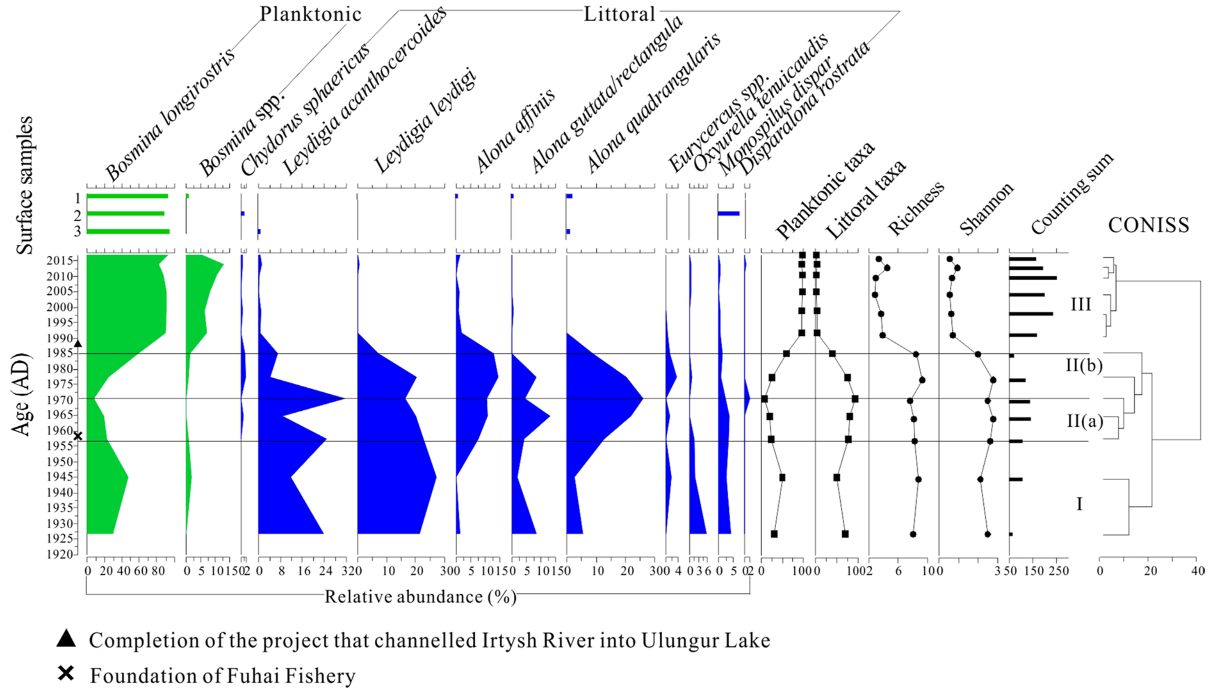


Fig. 3 Relative abundances (%) of cladocerans in sediments of the Jili Lake core, with significant zones identified by the constrained incremental sum of squares (CONISS) model. Results from the three surface samples are shown above the plot from the core

frequently deposited remain for each taxon (Korhola and Rautio 2001; Amsinck et al. 2005).

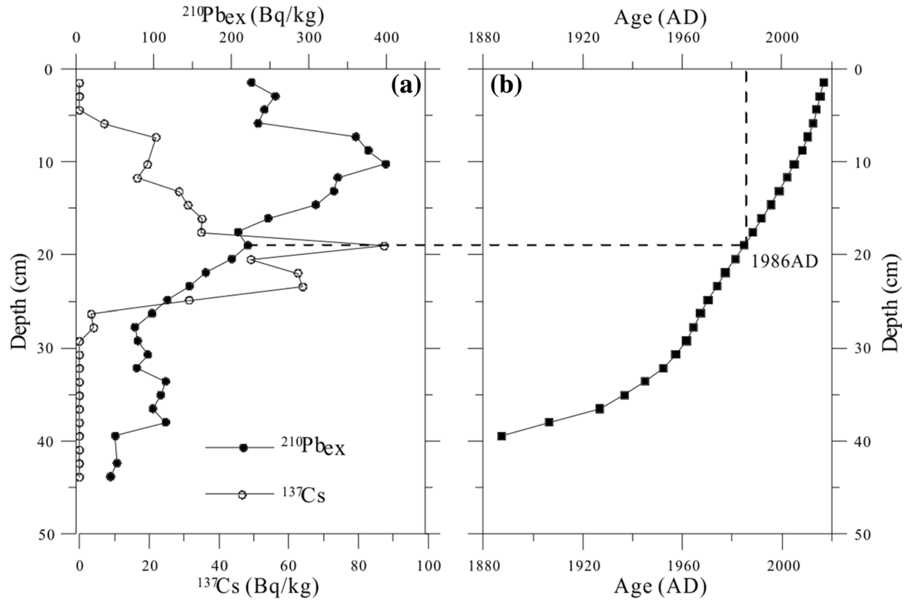
Principal component analysis (PCA) was used to summarise the major direction of cladoceran assemblage changes. Cladoceran percentage data were square-root-transformed prior to PCA. For each sample, we calculated the Shannon diversity index (H') and species richness. We used the constrained optimal sum of squares with untransformed percentage data (Birks and Gordon 1985) and the associated broken-stick model (Bennett 1996) to identify statistically distinct zones in the cladoceran biostratigraphy. The zonation of the cladoceran stratigraphy was determined by cluster analysis, using constrained incremental sum of squares (Grimm 2011). All statistical analyses were performed in the R program (version 3.6.2) and R Studio (version 4.0), using the 'vegan' and 'rioja' packages.

Results

Core chronology

The sediment chronology and age-depth model were established using the profile of unsupported ^{210}Pb activity and the corrected constant rate of supply model (Appleby and Oldfield 1978; Appleby 2001). The ^{137}Cs activity peak at 23 cm depth represents the year 1963, namely the year of peak cesium deposition in the northern hemisphere from atmospheric atomic bomb testing, which was slightly different from the ^{210}Pb date at that depth (Fig. 4) (Appleby et al. 1991; Wu et al. 2007). Nevertheless, a second peak of ^{137}Cs activity at 19 cm, corresponds to a date of 1986 according to the CRS model, the year of the Chernobyl incident. The sample at the depth of 30 cm

Fig. 4 **a** Jili Lake sediment core ^{210}Pb and ^{137}Cs profiles, and **b** age-depth profile generated with the constant rate of supply model. $^{210}Pb_{ex}$ refers to excess ^{210}Pb



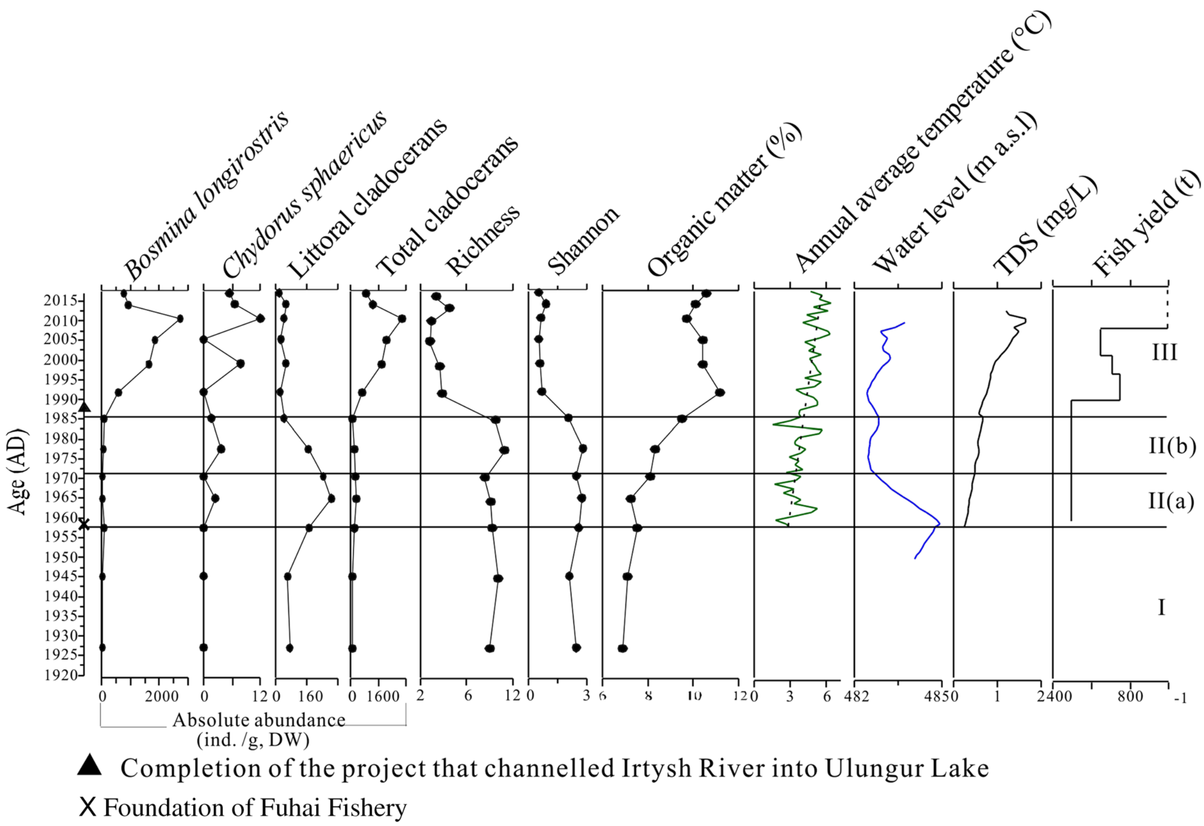


Fig. 5 Absolute abundance (individuals/g dry weight), species richness, Shannon diversity, total organic matter, and monitored data for temperature, water level (Cheng et al. 2016), total dissolved solids (TDS) (Cheng et al. 2016), and annual fish yield (Tang 2009) in Jili Lake

was dated to 1927 and the calculated resolution of the uppermost 30 cm of the core was 1–6 y/cm.

Cladoceran microfossils and total organic matter

Cladoceran remains were generally well-preserved throughout the core, but were relatively rare before the 1980s and more numerous in the upper sections of the core. Shannon diversity and species richness were both relatively high prior to the 1980s, followed by a decreasing trend (Fig. 5). A total of 13 cladoceran taxa, representing 8 genera, were identified in the sediments of Jili Lake. The most common species throughout the core was *B. longirostris* (approximately 57.2 %), followed by *L. leydigi* (approximately 10.4 %), *L. acanthocercoides* (approximately 8.9 %), *A. quadrangularis* (approximately 7.5 %), *A. affinis* (approximately 4.8 %), and *Bosmina* spp. (unidentified, approximately 4.3 %). Summed average relative abundances of *Chydorus sphaericus*, *Eurycercus* spp., *Oxyurella tenuicaulis*, and *Monospilus dispar* over the core were lower than 5.0 %. Littoral cladocerans were the dominant species before the 1980s; planktonic cladocerans (*B. longirostris*) became dominant after the 1980s, when all littoral species decreased markedly. The three surface sediment samples from the lake revealed that the cladoceran community was similar across the pelagic area of the lake, with *B. longirostris* displaying an average relative abundance of 94.9 %, whereas littoral cladocerans occurred sporadically and accounted for only 5.1 % of the total cladoceran abundance (Fig. 3). Stratigraphic changes in cladoceran abundance were used to divide the core record into three zones (I–III; including subzones II(a) and II(b)). The grouping is also visible in the PCA biplot (Fig. 6). Highest loadings in PCA axis 1 (approximately 48.3 %) and PCA axis 2 (approximately 20.9 %) were detected for *B. longirostris* and *C. sphaericus*, respectively.

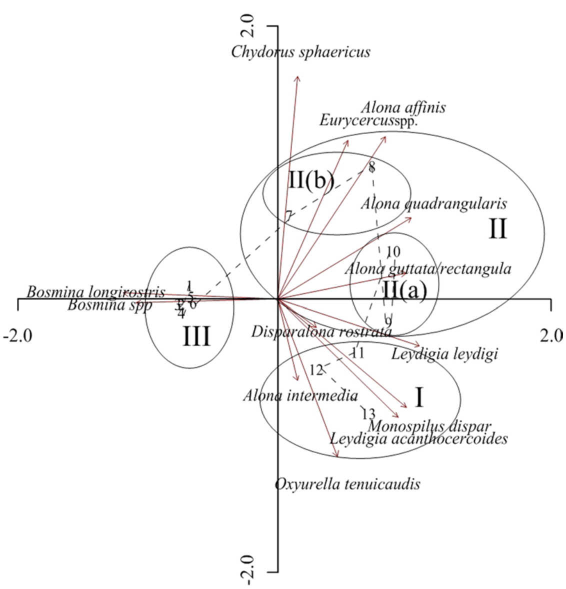


Fig. 6 Principal component analysis biplot of the cladoceran community assemblages. Numbers refer to the sample number. Samples belonging to the same zones are enclosed in ellipses, which are identified as Zone I, Zone II (including two subzones), and Zone III. Dotted lines follow the record in stratigraphic order for clear interpretation of the changes in the community

Zone I (AD 1927–1957). Samples in this zone were grouped in the bottom right quadrant of the PCA biplot (Fig. 6), with low levels of total cladoceran absolute abundance and organic matter content. Both the planktonic taxa and littoral taxa were found in this time period, and the proportion of littoral species was relatively higher than that of planktonic species. *B. longirostri*s (approximately 33.2 %), *L. leydigi* (approximately 23.5 %), and *L. acanthocercoides* (approximately 20.2 %) were the dominant species.

Zone II(a) (AD 1957–1970). Samples in this zone also exhibited a low level of absolute abundance of total cladocerans and low organic matter content (Fig. 5). With the appearance of *C. sphaericus*, littoral species generally exhibited a marked increase in relative abundance compared with Zone I. The total relative abundance of littoral taxa reached a maximum value for the core (approximately 92.5 %), including *A. affinis* (approximately 10.7 %), *Alona guttata*/*rectangular* (approximately 8.8 %), and *A. quadrangularis* (approximately 24.0 %), although *L. leydigi* and *O. tenuicaulis* exhibited decreasing abundance. The relative percentage of planktonic species *B. longirostris* decreased to a minimum of approximately 7.5 %. All samples in this zone are located on the right side of the PCA biplot (Fig. 6).

Zone II(b) (AD 1970–1986). Samples in this zone still exhibited low total cladoceran absolute abundance, but showed an increase in the proportion of planktonic taxa and organic matter content compared with that of Zone II(a), whereas the relative percentage of littoral species declined sharply. Samples in this zone are grouped in the top right quadrant of the PCA biplot (Fig. 6).

Zone III (AD 1986–2018). Samples in this zone are situated on the left side of the PCA biplot (Fig. 6). The organic matter content increased, whereas both the cladoceran Shannon diversity index and species richness decreased, compared with values in the previous zones. This phase was characterised by a pronounced increase in the relative abundance of *B. longirostris* (approximately 88.8%). The absolute abundance of *B. longirostris* also increased rapidly, from an average of 47.3 individuals/g dry weight in Zone II(b) to 1422.1 individuals/g dry weight in Zone III. Littoral species reached their lowest level of relative abundance and almost disappeared in this zone. Notably, *A. quadrangularis* and *Eurycercus* spp. disappeared starting at depths of 16 cm (1991) and 13 cm (1998), respectively.

Discussion

Cladoceran community

Zone I: 1927–1957 (undisturbed, freshwater lake phase)

During this phase, both the cladoceran species richness and Shannon diversity value were high. The proportions of littoral species and richness were higher than those of planktonic species, which were mainly common freshwater littoral cladocerans. Both the absolute abundance of total cladocerans and organic matter content were low, suggesting a low trophic state in Jili Lake (Fig. 5). During this period, there was a lack of documented human disturbance such as fish farming or agricultural development in the drainage basin. This is consistent with negligible human-induced nutrient input to the lake, as was found in previous studies (Fan 1984; Ma 1985; Dong et al. 2008). The cladoceran community likely reflects the natural state of the Jili Lake ecology, and is typical of undisturbed freshwater systems (Bos et al. 1999; Liu et al. 2013, 2014).

Zone II(a): 1957–1970 (water regulation phase)

The clear feature in this zone is that the relative abundance of planktonic taxa decreased, whereas that of the littoral taxa increased (Fig. 3). Previous studies emphasised the influence of water level fluctuations on the composition of cladoceran communities, i.e. that rising/falling lake level results in an increase/decrease in the abundance of planktonic Cladocera (Whiteside 1970; Amoros and Urk 1989; Sarvala and Halsinaho 1990; Sarmaja-Korjonen and Alhonen 1999). The water level of Jili Lake dropped significantly in this phase, owing to the damming of the Ulungur River (Fig. 5) (Fan 1984; Ma 1985; Liang et al. 2011; Cheng et al. 2016). We concluded that the decline in the proportion of planktonic species in this zone was most likely linked to lower water levels in Jili Lake since the 1960s.

Many factors, however, in addition to water level fluctuations, such as eutrophication or changes in predation, may also influence the relative proportions of remains of planktonic and littoral cladocerans (Hofmann 1998). In this zone, the low relative abundance of eutrophic species *B. longirostris* and the increased relative abundance of *A. quadrangularis*, which prefers oligotrophic environments Whiteside 1970; Hofmann 1996; Alam and Khan 1998; Brodersen 1998; Amsinck 2005), suggested that the lake was still poor in nutrients. Although fish farming in Jili Lake began to develop after establishment of the Fuhai Fishery in 1958, only a few small wooden fishing boats were working, and both the scale of the fishery and fish production remained small (Fig. 5). Thus, there were no extra nutrient inputs to the lake and the trophic state was likely still low, as supported by the low organic matter content and low absolute abundance of total cladocerans in this period (Fig. 5).

Zone II(b): 1970–1986 (salinisation phase)

The cladoceran community changed substantially in this zone, as revealed in the cladoceran relative abundance (Fig. 3). The relative abundance of planktonic species increased significantly, whereas that of almost all littoral species decreased. Whereas both the water temperature and lake stage were low in this zone, they were probably not the main causes of the increase in planktonic cladocerans. *B. longirostris* is a salinity-tolerant species (Adamczuk 2016) compared to many other cladocerans, and might have had an additional advantage, owing to its broad salinity tolerance (Leppänen 2018). Therefore, increased abundance of *B. longirostris* may reflect higher salinity in Jili Lake, which was supported by the continuous increase in total dissolved solids during this phase (Fig. 5). The reduction in the size of the water body, combined with high evaporation, dramatically increased the salinity of the lake.

In general, cladoceran species richness declines with increasing salinity (Hanmmer 1986; Frey 1993; Jeppesen et al. 1994; Bos et al. 1999) because many cladoceran species are not well-adapted, physiologically, to high salinity (Aladin 1991). The decline in relative abundance of almost all littoral cladocerans in this phase, which began in this zone, most likely reflects a deterioration of water quality caused by increased salinity in Jili Lake. For example, the relative abundances of *A. quadrangularis*, *A. affinis*, and *Eurycercus* spp., which are generally restricted to fresher waters (Bos et al. 1999), decreased significantly or even disappeared in this zone. In addition, *C. sphaericus* also has a high tolerance for increased salinity (Hanmmer 1986; Aladin 1991; Wolfram et al. 1999), and its increased relative abundance during this period may also indicate the start of the deterioration of water quality (Fig. 3). Furthermore, the decrease in littoral cladocerans may also have resulted from a loss of macrophytes.

B. longirostris tends to thrive during periods of increased nutrient supply (Lotter 1998; Szeroczynska 2002; Gasiorowski and Szeroczynska 2004). Thus, the increase in *B. longirostris* may indicate an increase in lake trophic state, as also suggested by the increase in organic matter content (Fig. 5). During this period, *A. bramaorientalis*, *R. rutiluslacustris*, and *E. lucius* were introduced into Jili Lake to enhance fish production (Tang et al. 2009). Although the scale of fish farming was still small, the long-term input of forage might have led to the increase in trophic status.

Zone III: 1986–2018 (saline and eutrophic phase)

The pronounced shift in the cladoceran community was a response mainly to the acceleration of salinisation and eutrophication after the 1990s. The project that channeled the Irtysh River into Ulungur Lake was finished in 1988, and enabled saline water to flow into Jili Lake, leading to a further increase in salinity (Fig. 5). High salinity can restrict the growth of freshwater Cladocera taxa (Jeppesen et al. 1994), and likely contributed to the decrease in littoral cladocerans in Jili Lake. All the littoral taxa in this zone decreased to the lowest recorded abundance or even disappeared (Fig. 3). Only a few existed, such as *C. sphaericus*, *L. acanthocercoides*, and *O. tenuicaulis*, that can tolerate relatively high salinity (Hammer 1986; Aladin 1991; Wolfram et al. 1999). In addition, high salinity may further restrict the growth of littoral cladocerans by inhibiting the growth of macrophytes, as many littoral-benthic cladoceran species inhabit macrophytes and feed on them (Barker et al. 2010). Our field investigation in 2018 showed that macrophytes in Jili Lake have almost disappeared, which might have been a consequence of the continuous increase in salinity. Therefore, the significant decrease in the relative abundance of macrophyte-associated taxa (*A. affinis*, *A. rectangula*, and *A. quadrangularis*) (Amsinck et al. 2003; Bjerring et al. 2009) (Fig. 3) was also related to the disappearance of macrophytes.

Introduction of intensive fish culture in the late 1980s was the main reason for eutrophication of the lake and the increased absolute abundances of cladocerans. Both the scale of fish farming and fish yield expanded, and a large quantity of fish food was introduced into the lake, causing a sharp increase in the trophic state. Moreover, the project that channeled the Irtysh River into Ulungur Lake enabled flow of saline and nutrient-rich water into Jili Lake, which further stimulated eutrophication (Cheng et al. 2016; Ji et al. 2018). Global warming is also thought to be a cause of increased plankton abundance in freshwater lakes (Jeff et al. 2019). The increase in absolute abundance of *Bosmina* may also be related to the warmer water temperatures (Daufresne et al. 2009). Considering that the scale of fish farming in the lake has gradually expanded, stress from fish predation might also have impacted the cladoceran community. *Bosmina* usually have small body size and hence are often less heavily preyed upon than some of the larger cladoceran taxa. This may also account for the abundant *Bosmina* in Zone 3.

The low species richness and Shannon diversity of the cladoceran community, as well as the high organic matter content and absolute abundance of dominant taxa (*B. longirostris*), reflect a degraded lake ecosystem (Fig. 5). It seems that the cladoceran community in Jili Lake has already completed a regime shift, from relatively high species richness and Shannon diversity in Zone I, to low values in Zone III (Fig. 5). This regime shift of the cladoceran community is also clearly observed in the PCA biplot (Fig. 6). Therefore, loss of zooplankton species richness may have resulted from the degradation of the lake ecosystem.

Implications

The history of the cladoceran community in Jili Lake shows that the aquatic ecosystem has deteriorated significantly during the past 100 years, mainly owing to the influence of human activities. Salinisation and eutrophication were the most important driving factors for the regime shift of the cladoceran community in Jili Lake. Construction of water storage facilities in the basin also had an important impact on the lake ecosystem. Damming and construction of the reservoir since the 1950s led to a rapid decline in water levels and increased salinisation, thereby resulting in habitat deterioration in Jili Lake. Subsequently, to alleviate the impacts of reduced water level and salinisation in Ulungur and Jili Lakes, the government carried out a project that channeled the Irtysh River into Ulungur Lake. That project enabled the saline water from Ulungur Lake to flow into Jili Lake, and thus had a negative impact on the Jili Lake ecosystem. Our results suggest that the Jili Lake ecosystem continues to deteriorate and will not improve without protection/restoration actions.

Eutrophication in Jili Lake started around the 1980s, later than in lakes in the monsoon region (e.g. Taihu Lake and Dianchi Lake), but the impact has been similar (Liu et al. 2008; Cheng et al. 2019; Yang et al. 2020). Fish farming is one of the main causes of eutrophication and ecosystem degradation in Jili Lake. The pattern of fish farming in Jili Lake was simpler and more straightforward compared to that which occurred in lakes of the monsoon region (e.g. large quantities of cow and sheep dung were supplied to the lake directly, which greatly accelerated eutrophication and degradation of the lake ecosystem).

Conclusions

Our sediment results showed that changes in the cladoceran community in Jili Lake before the 1970s were likely a response to reduced water level. After the 1970s, variations in the cladoceran community in Jili Lake reflect increased salinity. Since the 1990s, the cladoceran community responded mainly to accelerated salinisation and eutrophication, caused by an increase in fish farming and construction of a water storage facility. Our results demonstrate that cladocerans in large lakes of the arid region of China react quickly to environmental changes, and that their microfossils are well preserved in lake sediments, making them excellent indicators of past aquatic ecosystem changes.

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References

Adamczuk M (2016) Past, present, and future roles of small cladoceran Bosmina longirostris (O.F. Müller, 1785) in aquatic ecosystems. Hydrobiologia 767:1–11

Aladin NV (1991) Salinity tolerance and morphology of the osmoregulation organs in Cladocera with special reference to Cladocera from the Aral Sea. Hydrobiologia 225:291–299

Aladin NV, Potts WTW (1992) Changes in the Aral Sea ecosystems during the period 1960–1990. Hydrobiologia 237:67–79

Alam A, Khan AA (1998) On the record of cladoceran Leydigia acanthocericoides (Chydoridae) from Aligarh, Uttar pradesh, India. J Bombay Nat Hist Soc 95:143–144

Amoros C, Urk GV (1989) Paleoecological analyses of large rivers: some principles and methods. In: Petts G (ed) Historical change of Large Alluvial Rivers. Wiley, New York, pp 143–157

Amsinck SL, Jeppesen E, Ryves D (2003) Cladoceran stratigraphy in two shallow brackish lakes with special reference to changes in salinity, macrophyte abundance and fish predation. J Paleolimnol 29:495–507

Amsinck SL, Jeppesen E, Landkildchus F (2005) Relationships between environmental variables and zooplankton subfossils in the surface sediments of 36 shallow coastal brackish lakes with special emphasis on the role of fish. J Paleolimnol 33:39–51

Appleby PG (2001) Chronostratigraphic techniques in recent sediments. In: Last WM, Smol JP (eds) Tracking environmental changes using lake sediments, vol 1. Kluwer Academic Publishers, Boston, pp 171–203

Appleby PG, Oldfield F (1978) The calculation of ^{210}Pb dates assuming a constant rate of supply of unsupported ^{210}Pb to sediment. Catena 5:1–8

Appleby PG, Nolan PJ, Battarbee DW (1986) ^{210}Pb dating by low background gamma counting. Hydrobiologia 141:21–27

Appleby PG, Richardson N, Nolan PJ (1991) ^{241}Am dating of lake sediments. Hydrobiologia 214:35–42

Barker T, Irfanullah H, Moss B (2010) Micro-scale structure in the chemistry and biology of a shallow lake. Freshwater Biol 55:1145–1163

Bennett K (1996) Determination of the number of zones in a biostratigraphical sequence. New Phytol 132:155–170
Birks HJB, Gordon AD (1985) Numerical methods in quaternary pollen analysis. Academic Press, London
Bjerring R, Becares E, Declerck S et al (2009) Subfossil cladocera in relation to contemporary environmental variables in 54 Pan-European lakes. Freshwater Biol 11:2401–2417
Bos DG, Cumming BF, Smol JP (1999) Cladocera and anostraca from the interior plateau of British Columbia, Canada, as paleolimnological indicators of salinity and lake level. Hydrobiologia 392:129–141
Brodersen KP, Whiteside MC, Lindegaard C (1998) Reconstruction of trophic state in Danish lakes using subfossil chydorid (Cladocera) assemblages. Can J Fish Aquat Sci 55:1093–1103
Brucet S, Boix D, Gascón S et al (2009) Species richeness of crustacean zooplankton and trophic structure of brackish lagoons in contrasting climate zones: north temperature Denmark and Mediterranean Catalonia (Spain). Ecography 32:692–702
Chen G, Dalton C, Tylor D (2010) Cladocera as indicators of trophic state in Irish lakes. J Paleolimnol 44:465–481
Cheng LJ, Yao SC, Xue B et al (2019) Long-term change of the assemblages and abundance of cladocerans in different ecotypes of Lake Taihu (in Chinese with English Abstract). J Lake Sci 31:1670–1684
Cheng Y, Li S, Mengubieke E (2016) Changes of water and salt characteristic of Ulungur Lake and the corresponding cause analysis (in Chinese with English Abstract). Environ Protect Xinjiang 38:1–7
Daufresne M, Lengfellner K, Sommer U (2009) Global warming benefits the small in aquatic ecosystems. Proc Natl Acad Sci USA 106:12788–12793
Davidson TA, Sayer CD, Perrow MR et al (2007) Are the controls of species composition similar for contemporary and sub-fossil cladoceran assemblages? A study of 39 shallow lakes of contrasting trophic status. J Paleolimnol 38:117–134
Dong Y, Jiang M, Liu QG (2008) Research on water quality and trophic level in Wulungu Lake (in Chinese with English Abstract). J Shanghai Fish Univ 17:564–569
Downing AL, Brown BL, Leibold MA (2014) Multiple diversity-stability mechanisms enhance population and communities stability in aquatic food webs. Ecology 95:173–184
Fan ZL (1984) Recent changes in the lakes of Xinjiang. Geogr Res 1:80–89
Folke C, Carpenter S, Walker B et al (2004) Regime shifts, resilience, and biodiversity in ecosystem management. Annu Rev Ecol Evol Syst 35:557–581
Frey DG (1986) Cladocera analysis. In: Berglund BE (ed) Handbook of holocene palaeoecology and palaecohydrology. Wiley, New York, pp 667–692
Frey DG (1993) The penetration of Cladocerans into saline waters. Hydrobiologia 267:233–248
Gasiorowski M, Szeroczysnka K (2004) Abrupt changes in Bosmina (Cladocera, Crustacea) assemblages during the history of the Ostrowite Lake (northern Poland). Hydrobiologia 526:137–144
Grimm EC (2011) TILIA software version 1.7.16. Illinois State Museum, Research and Collection Center. Springfield USA
Hammer UT (1986) Saline lake ecosystems of the world. Dr W. Junk Publishers, Dordrecht, p 616
Hofmann W (1996) Empirical relationships between cladoceran fauna and trophic state in thirteen northern German lakes: analysis of surficial sediments. Hydrobiologia 318:195–201
Hofmann W (1998) Cladocerans and chironomids as indicators of lake level changes in north temperate lakes. J Paleolimnol 19:55–62
Jeff CH, Michalak AM, Pahlevan N (2019) Widespread global increase in intense lake phytoplankton blooms since the 1980s. Nature 574:1–1
Jeppesen E, Søndergaard M, Kanstrup E et al (1994) Does the impact of nutrients on the biological structure and function of brackish and freshwater lakes differ? Hydrobiologia 275:15–30
Jeppesen E, Madsen EA, Jensen JP et al (1996) Reconstructing the past density of planktivorous fish and trophic structure from sedimentary zooplankton fossils: a surface sediment calibration data set from 30 predominantly shallow lakes. Freshwater Biol 36:115–127
Jeppesen E, Jensen JP, Skovgaard H et al (2001a) Changes in the abundance of planktivorousfifish in lake Skanderborg during the last two centuries-a palaeoecological approach. Palaeogeogr Palaeocl 172:143–152
Jeppesen E, Leavitt P, De Meester L (2001b) Functional ecology and palaeolimnology: using cladoceran remains to reconstruct antropogenic impact. Trends Ecol Evol 16:191–198
Ji FF, Shen JZ, Ma XF et al (2018) Changes in water quality and analysis of underlying causes in Ulungur Lake (in Chinese with English Abstract). J Hydroecol 39:61–66
Jiang QF, Shen J, Liu XQ et al (2010) Environmental changes recorded by lake sediments from Lake Jili, Xinjiang during the past 2500 years (in Chinese with English Abstract). J Lake Sci 22:119–126
Jiang XZ, Du N (1979) Fauna sinica: freshwater cladocera. Science Press, Beijing
Korhola A, Rautio M (2001) Cladocera and other branchipod crustaceans. In: Smol JP, Birks HJB, Last WM (eds) Tracking environmental change using lake sediments: zoological indicators, vol 4, 665th edn. Kluwer, Dordrecht, pp 5–41
Korhola A, Tikkanen M, Weckström J (2005) Quantification of Holocene lake-level changes in Finnish Lapland using a cladocera – lake depth transfer model. J Paleolimnol 34:175–190
Lampert W (1997) Zooplankton research: the contribution of limnology to general ecological paradigms. Aquat Ecol 31:19–27
Leppänen J (2018) An overview of Cladoceran studies conducted in mine water impacted lakes. Int Aquat Res 10:207–221
Liang LL, Wang F, Wang DX et al (2011) Research on the lowest ecological water level and ecological water quantity of Ulungur Lake. Adv Water Sci 04:26–34

Liu GM, Chen FZ, Liu ZW (2008) Preliminary study on cladoceran microfossils in the sediments of Lake Taihu (in Chinese with English Abstract). J Lake Sci 4:470–476

Liu GM, Liu ZW, Chen FZ et al (2013) Response of the cladoceran communities to eutrophication, fish introductions and degradation of the macrophyte vegetation in Lake Dianchi, a large, shallow plateau lake in southwestern China. Limnology 14:159–166

Liu GM, Liu ZW, Gu BH et al (2014) How important are trophic state, macrophyte and fish population effects on cladoceran communities? A study in Lake Erhai. Hydrobiologia 736:189–204

Liu JJ (2015) Analysis on the water amount flowing into Ulungur Lake (in Chinese with English Abstract). Energy Energy Conserv 5:103–105

Lotter AF, Birks JB, Hofmann W, Marchetto A (1998) Modern diatom, cladocera, chironomid, and chrysophyte cyst assemblages as quantitative indicators for the reconstruction of past environmental conditions in the Alps. II. Nutrients. J Paleolimnol 19:443–463

Ma GZ (1985) Study on the ecological balance of Ulungur Lake and its relationship with fishery (in Chinese). Arid Zone Res 1:42–46

Ren ML (1990) Fisheries of Jili Lake in Xinjiang. Heilongjiang Science and Technology Press, Harbin

Sarmaja-Korjonen K, Alhonen P (1999) Cladoceran and diatom evidence of lake-level fluctuation from a Finnish lake and the effect of aquatic-moss layers on microfossil assemblages. J Paleolimnol 22:277–290

Sarvala J, Halsinaho S (1990) Crustacean zooplankton of Finnish forest lakes in relation to acidity and other environmental factors. In: Kauppi P, Anttila P, Kenttamies K (eds) Acidification in Finland. Springer, Berlin, pp 1009–1027

Shi HB, Chen GJ, Lu HB et al (2016) Regional pattern of Bosmina responses to fish introduction and eutrophication in four large lakes from Southwest China. J Plankton Res 38:443–455

Szeroczynska K (2002) Human impact on lakes recorded in the remains of Cladocera (Crustacea). Quatern Int 95:165–174

Szeroczynska K, Sarmaja-Korjonen K (2007) Atlas of subfossil cladocera from central and northern Europe. Friends of Lower Vistula Society, Poland

Tang FJ, Jiang ZF, Liu FMH (2009) Zoobenthos and its changes in Xinjiang Jili Lake (in Chinese with English Abstract). Chin J Fish 4:42–45

Whiteside MC (1970) Danish chydoridcladocera: modern ecology and core studies. Ecol Monographs 40:79–118

Wolfram G, Donabaum K, Schagerl M et al (1999) The zoobenthic communities of shallow salt pans in Austria -preliminary results on phenology and the impact of salinity on benthic invertebrates. Hydrobiologia 408/409:193–202

Wu J, Gagan MK, Jiang X et al (2004) Sedimentary geochemical evidence for recent eutrophication of Lake Chenghai, Yunnan, China. J Paleolimnol 32:85–94

Wu J, Schleser GH, Lücke A, Li S (2007) A stable isotope record from freshwater lake shells of the eastern Tibetan Plateau, China, during the past two centuries. Boreas 36:38–46

Wu JL, Ma L, Abundauwaili J (2009) Lake surface change of the Aral Sea and its environmental effects in the arid region of the central Asia (in Chinese with English Abstract). Arid Land Geogr 32:418–422

Wu JL, Ma L (2011) Lake evolution and climatic and hydrological changes in arid zone of Xinjiang (in Chinese with English Abstract). Marine Geol Quat Geol 31:135–143

Wu JL, Ma L, Zeng HA (2013) Water quality and quantity characteristics and its evolution in Lake Bosten, Xinjiang over the past 50 Years (in Chinese with English Abstract). Sci Geogr Sin 33:231–237

Yang J, Zhou J, Qin BQ et al (2020) Long-term Variation Characteristics of Zooplankton Communities Structure in Meiliang Bay, Lake Taihu. Environ Sci 41:1246–1255

Zhang YL, Jeppesen E, Liu XH et al (2017) Global loss of aquatic vegetation in lakes. Earth-Sci Rev 173:259–265