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Tropical and Sub-Tropical Reservoir Limnology in China

Theory and Practice

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TROPICAL AND SUB-TROPICAL RESERVOIR LIMNOLOGY IN CHINA

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Tropical and Sub-Tropical Reservoir Limnology in China

Theory and Practice

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Preface

Reservoirs are specific aquatic ecosystems with complex behavior, distinct of both natural lakes and rivers. Reservoir limnology and water quality management are associated with functions such as flood control, hydropower generation, irrigation, and fishery. On a global scale, supplying drinking water has become a dominant function of reservoirs. China is endowed with a huge number of reservoirs, and harbors more than 45% of the high dams of the world. In the past 60 years, China built about 86,000 reservoirs for multiple purposes such as agricultural irrigation, power generation, flood control, and water supply. These store a huge amount of water and support economic and social development. But in the last decades, many rivers and lakes have become polluted because of human activities in the catchments and direct discharge of domestic and industrial sewage. Many rivers and lakes have thereby almost lost their function as sources of drinking water. New clean water sources are required, at an affordable cost of water treatment. The reservoirs play an ever-increasing role in this water supply and immediately mitigate water shortage on a regional scale, especially in southern China where there are few natural lakes. However, there is no systematic knowledge of reservoir limnology in China, which is fundamental to water quality management.

Many reservoirs were exploited for fisheries before 2000 and all early limnological data were collected in the context of estimating fish production. Economics of fisheries brought new technology to reservoir fishery but also led to negative effects on water quality and ecosystem health.

Compared to the research on and the protection of lakes, there is a limited public attention to reservoirs in China. This volume aims to offer a first description of reservoir limnology in tropical and subtropical China. It includes 20 articles that come mainly from Guangdong Province and Hainan Island, two regions where only few natural lakes exist. Owing to their monsoonal climate, annual precipitation concentrates in a flooding season (wet season) from the middle of April to the end of September. At that time, much runoff is discharged to the South China Sea. In contrast, many districts suffer from water shortage in the dry season, from October to next April. The construction of reservoirs by damming rivers offered a way to meet water demands in the dry season. Guangdong is one of the more developed provinces, and most of its rivers run across cities and towns; they have become polluted and are no longer suitable as drinking water resources; here, reservoirs became the alternative source of drinking water. Several such water bodies were built specially for water supply to Hong Kong and Macau, which are situated close to Guangdong Province. However, the same situation of water pollution commonly occurs in eastern and northern China as well.

The dam of Liuxihe Reservoir, a large impoundment for the drinking water supply of Guangzhou, the largest city in southern China. Photo taken by Bo-Ping Han.

To mitigate water shortage in northern China, a South-to-North Water Transfer Project was initiated, aimed at reallocating water resources on a national scale. The three channels of this grand project have their beginning at southern reservoir catchments. It is therefore clear that reservoirs begin to play a significant role not only in southern but also northern China.

Besides contributions from southern China, this book contains five invited contributions from Hongfeng Reservoir (Guizhou Province), Danjiangkou Reservoir (Hubei Province), the Three Gorges Reservoir (Chongqing Province), and Xinanjiang Reservoir (Zhejiang Province). In comparison with advances in Europe and North America, the limnological study of reservoirs in China is developing more in pace with the country's social and economic requirements. This means that this work includes information on zooplankton, phytoplankton, zoobenthos, Cyanobacteria, nutrient budgets, sediments, biogeochemical cycling of mercury, and fisheries, but also that its main focus is on eutrophication, because of the current demands on water quality. The publication of this special volume is hoped to encourage the further development of reservoir limnology in China, and it also provides a window on China to all scientists interested in limnology and freshwater ecology. We are grateful to Prof. Henri Dumont from Belgium for his encouragement and suggestions for preparing the volume. We also thank Dr. Ken Chen from Australia for his reading and linguistic correction of the manuscripts. The preparation of the book was supported by a special grant of Project 211 for Hydrobiology and NSF of China (U0733007).

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Part I
Biological Community

Chapter 1
Diversity and Community Structure of Zooplankton in Reservoirs in South China

Qiuqi Lin and Bo-Ping Han

Abstract Zooplankton diversity and its response to eutrophication were investigated in 15 reservoirs of South China from 2000 to 2003. So far, 105 species of Rotifera, 30 species of Cladocera and 24 species of Copepoda have been identified. The majority of rotifer species are monogononts, with bdelloids represented by *Rotaria* sp. only. *Lecane* (with 19 taxa), *Trichocerca* (15) and *Brachionus* (11) are the most speciose genera, with many species cosmopolitan. The most frequently observed genera were *Keratella*, *Brachionus*, *Polyarthra*, *Trichocerca*, *Asplanchna*, *Conochilus*, *Ploesoma*, *Ascomorpha* and *Pompholyx*. Daphniidae (10 species) and Chydoridae (11) were the two rich cladoceran families. *Bosmina tripurae*, *Bosminopsis deitersi*, *Diaphanosoma orghidani*, *D. dubium*, *Moina micrura, Ceriodaphnia cornuta and C. quadrangula* were most frequent in the pelagic zone. In addition, 10 calanoid and 14 cyclopoid species of copepods occurred. Most of the Calanoida are endemic to the tropics and subtropics in China. *Phyllodiaptomus tunguidus*, *Neodiaptomus schmackeri*, *Mesocyclops thermocyclopoides* and *Thermocyclops taihokuensis* were most frequently recorded.

In the year 2000, total abundance of zooplankton varied from 11 to 290 ind./L during the period of June to July. Zooplankton was much more abundant in mesotrophic than in oligotrophic and eutrophic reservoirs. Rotifera numerically predominated in nine reservoirs and Copepoda in six reservoirs. The relative abundance of *Brachionus*, *Trichocerca* and *Asplanchna* increased and the ratio of Calanoida to Cyclopoida decreased with trophic level. Reservoir trophic state and predation were the most direct factors regulating zooplankton abundance and the dynamics of community structure. However, it was also found that hydraulic retention time affected the response of the zooplankton community structure to eutrophication. In reservoirs with long or short retention times, zooplankton

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showed no apparent variation across seasons. In reservoirs with intermediate retention time, in contrast, the zooplankton community changed significantly with trophy.

1.1 Introduction

Zooplankton has been studied intensely and over long periods of time in the North Temperate Zone, but much less so in the tropics. Moreover, most of the current knowledge of tropical zooplankton comes from South America, Africa and Australia (Dussart and Defaye 2001; Korovchinsky 1996, 2006). Much less has become known of the distribution and taxonomic composition of zooplankton in tropical Asia. In South China, natural lakes are scarce, but large numbers of reservoirs were constructed during 1950–1980. Aquaculture of filter-feeding fish like bighead carp (*Hypophthalmichthys nobilis*) was of widespread occurrence in most of these reservoirs. Such filter-feeding fish may play a key role in structuring the zooplankton community, since they consume zooplankton and at the same time compete with it for algal and pelagic sestonic food. Nilssen (1984) argued that heavy predation by juvenile and adult fish may greatly simplify the zooplankton community, with a resulting scarcity of Cladocera, notably the efficient filter feeders of the large genus *Daphnia*. The present contribution is aimed at investigating zooplankton diversity of a number of reservoirs in Guangdong Province, South China, and analyse some characteristics of the zooplankton community under a regime of predation by filter-feeding fish.

1.2 Materials and Methods

The reservoirs studied are located in the subtropical–tropic transition zone of China (20°14′ to 25°31′ N, 109°40′ to 117°20′ E) of which a detailed description can be found in Lin et al. (2003). Characteristics of the reservoirs are described in Table 1.1. Bighead carp (*H. nobilis*) has been extensively aquacultured in all reservoirs. This fish species produces semi-buoyant eggs that require a current to float. There are no appropriate spawning sites with sufficiently long floatation times for the eggs in these reservoirs, so bighead carp cannot reproduce successfully and production must be maintained by periodic introduction of YOY fish. The population is recruited by releasing YOY fish at 10 to 15 cm body length in April to May and/or August to November every year. Usually, Fish $\geq$3 years old are caught by seining from April to October.

Zooplankton of the 15 reservoirs was sampled during June to July (flooding season) in 2000. In addition, three reservoirs, Xinfengjiang, Feilaixia and Gongping, were sampled bimonthly from 2001 to 2003. Quantitative samples were collected with a 5-L water sampler at sites near the dams from the surface to a depth of 10 m at 1-m interval in deep reservoirs, or in shallow reservoirs from surface to bottom.

Table 1.1 Description of the study reservoirs

	Catchment area (km^2)	Normal volume (10^6 m^3)	Average depth (m)	Retention time (days)	Elevation (m)	Max. fish catch (kg/ha)	Year of filling
Xinfengjiang	5,813	10,500	29	644	116	27	1958
Gaozhou	1,022	841.8	21	161	86	–	1960
Gongping	317	163.3	4.5	133	16	–	1962
Liuxihe	539	325	21.3	172	235	21	1958
Chisha	23	1.1	1.5	15	12	–	1960
Chishijin	14	12.4	15	65	128	49	1958
Feilaixia	34,097	432	6.1	6	24	–	1998
Heshui	600	30.4	3	24	134	127	1957
Hexi	41	15.8	8.5	125	53	–	1958
Dashahe	217	153.8	9.4	180	34	299	1959
Dashuiqiao	196	100.1	13.7	343	56	340	1958
Qiyeshi	18	10.2	6.3	236	43	–	1960
Dajingshan	6	10.5	12	120	20	96	1975
Hedi	1,495	795	6.5	123	40	112	1959
Shiyan	44	16.9	6.3	169	36	599	1960

–, No data available

Samples from all depths were filtered through a mesh of 64 μm to form an integrated sample for each reservoir. Qualitative samples were collected at different sites within the littoral, riverine and lacustrine parts of the reservoirs with 64- and 113-μm meshes, respectively. Samples were preserved with 5% formalin. All samples were examined for Rotifera, Cladocera and Copepoda, and identified to species using a variety of sources (Wang 1961; Chiang and Du 1979; Shen 1979; Koste and Shiel 1987; Guo 1999, 2000; Nogrady and Segers 2002; Segers 1995; Shiel and Koste 1992) and counted under a stereomicroscope. Some other, rather unusual groups occasionally found in the pelagic of the reservoirs, are not dealt within this chapter, but information on them can be found in Han et al. (Chapter 16, this volume).

1.3 Results and Discussion

1.3.1 Species Composition

1.3.1.1 Rotifera

So far, of 105 species identified, the vast majority were monogononts (Table 1.2). The richest fractions were the Lecanidae (19 species) > Brachionidae (18) > Trichocercidae (15) > Synchaetidae (10). The most frequently observed species were *Keratella cochlearis*, *K. tropica*, *Brachionus calyciflorus*, *B. forficula*, *B. angularis*, *Polyarthra vulgaris*, *Trichocerca cylindrica*, *T. similis*, *T. elongata*,

Table 1.2 List of zooplankton species in the 15 investigated reservoirs in South China

Rotifera
Philodinidae
Rotaria sp.
Dicranoporidae
Dicranophorus sp.
Brachionidae
Anuraeopsis fissa (Gosse, 1851)
Brachionus angularis (Gosse, 1851)
B. budapestinensis (Daday, 1885)
B. calyciflorus (Pallas, 1766)
B. caudatus (Barrois and Daday, 1894)
B. diversicornis (Daday, 1883)
B. donneri (Brehm, 1951)
B. falcatus (Zacharias, 1898)
B. forficula (Wierzejski, 1891)
B. leydigi (Cohn 1862)
B. quadridentatus (Hermanns, 1783)
B. urceolaris (Muller, 1773)
Keratella cochlearis (Gosse, 1851)
K. tecta (Gosse, 1851)
K. tropica (Apstein, 1907)
Notholca labis (Gosse, 1887)
Plationus patulus (O.F. Müller, 1786)
Platyias quadricornis (Ehrenberg, 1832)
Epiphanidae
Epiphanes brachionus (Ehrenberg, 1837)
E. senta (O. F. Müller, 1723)
Euchlanidae
Euchlanis triquetra Ehrenberg, 1838
Dipleuchlanis propatula (Gosse, 1886)
Lepadellidae
Colurella adriatica Ehrenbeng 1831
Lepadella patella (Müller, 1786)
L. ovalis (Müller, 1786)
Mytilinidae
Mytilina ventralis (Ehrenberg, 1832)
Trichotriidae
Macrochaetus collinsii (Gosse, 1867)
Trichotria tetractis (Ehrenberg, 1830)

Asplanchnidae
Asplanchna brightwelli (Gosse, 1850)
A. priodonta (Gosse, 1850)
Asplanchnopus multiceps (Schrank, 1793)
Gastropodidae
Ascomorpha ecaudis (Perty, 1850)
P. major (Burckhardt, 1900)
P. remata (Skorikov, 1896)
P. vulgaris (Carlin, 1943)
Synchaeta oblonga (Ehrenberg, 1831)
S. pectinata (Ehrenberg, 1832)
S. stylata (Wierzejski, 1893)
Filiniidae
Filinia brachiata (Rousselet, 1901)
F. camasecla (Myers, 1938)
F. longiseta (Ehrenberg, 1834)
F. terminalis (Plate, 1886)
F. opoliensis (Zacharias, 1898)
Hexarthridae
Hexarthra fennica (Levander, 1892)
H. mira (Hudson, 1871)
Testudinellidae
Pompholyx sulcata (Hudson, 1885)
Testudinella mucronata (Gosse, 1886)
T. patina (Hermann, 1783)
T. tridentata (Smirnov, 1931)
Conochilidae
Conochilus dossuarius (Hudson, 1875)
C. hippocrepis (Schrank, 1830)
C. unicornis Rousselet, 1892
Collothecidae
Collotheca libera (Zacharias, 1894)
C. mutabilis (Hudson, 1885)
C. pelagica (Rousselet, 1893)
Cladocera
Leptodoridae
Leptodora richardi (Korovchinsky, 2009)
Sididae
Sida crystallina (O.F. Müller, 1776)
Diaphanosoma dubium (Manuilova, 1964)
D. orghidani transamurensis (Korovchinsky, 1986)
D. excisum (Sars, 1885)
Bosminidae
Bosmina fatalis (Burckhardt, 1924)
B. tripurae (Kořínek, Saha and Bhattacharya, 1999)
(Richard, 1895)
Bosminopsis deitersi (Richard, 1895)
Daphniidae
Daphnia galeata (G.O. Sars, 1864)
D. pulex (Leydig, 1860)
D. lumholtzi (Sars, 1885)
Ceriodaphnia quadrangula (O. F. Müller, 1785)
(O. F. Müller, 1785)

(continued)

Table 1.2 (continued)

A. ovalis (Bergendal, 1892)	*C. cornuta* (Sars, 1885)
A. saltans (Bartsch, 1870)	*Moina micrura* (Kurz, 1874)
Gastropus hyptopus (Ehrenberg, 1838)	*M. rectirostris* (Leydig, 1860)
G. minor (Rousselet, 1892)	*M. weismanni* (Ishikawa, 1896)
G. stylifer (Imhof, 1891)	*Simocephalus mixtus* (Sars, 1903)
Notommatidae	*Scapholeberis kingi* (Sars, 1903)
Cephalodella gibba (Ehrenberg, 1832)	**Macrothricidae**
Monommata grandis (Tessin, 1890)	*Macrothrix spinosa* (King, 1853)
Eosphora sp.	**Ilyocryptidae**
Notommata sp.	*Ilyocryptus spinifer* (Herrick, 1884)
Scaridiidae	**Chydoridae**
Scaridium longicaudum (O. F. Müller, 1786)	*Oxyurella singalensis* (Daday, 1898)
Trichocercidae	*O. tenuicaudis* (Sars, 1862)
Trichocerca braziliensis (Murray, 1913)	*Monospilus dispar* (Sars, 1862)
T. bicristata (Gosse, 1887)	*Alona guttata* (Sars, 1862)
T. capucina (Wierzejski and Zacharias, 1893)	*A. affinis* (Leydig, 1860)
T. chattoni (de Beauchamp, 1907)	*Coronatella rectangula* (Sars, 1861)
T. cylindrica (Imhof, 1891)	*Camptocercus rectirostris* (Schoedler, 1862)
T. dixonnuttalli (Jennings, 1903)	*Rhynchotalona falcata* (Sars, 1862)
T. elongata (Gosse, 1886)	*Chydorus sphaericus* (Müller, 1785)
T. gracilis (Tessin, 1886)	*C. ovalis* (Kurz, 1874)
T. insignis (Herrick, 1885)	*Dunhevedia crassa* (King, 1853)
T. longiseta (Schrank, 1802)	**Copepoda**
T. lophoessa (Gosse, 1886)	**Pseudodiaptomidae**
T. rousseleti (Voigt, 1902)	*Schmackeria inopinus* (Burckhardt, 1913)
T. similis (Wierzejski, 1893)	*S. forbesi* (Poppe and Richard, 1890)
T. similis f. *grandis* (de Beauchamp, 1907)	*S. spatulata* (Shen and Tai, 1964)
T. pusilla (Lauterborn, 1898)	**Diaptomidae**
Lecanidae	*Phyllodiaptomus tunguidus* (Shen and Tai, 1964)
Lecane bulla (Gosse, 1851)	*Neodiaptomus schmackeri* (Poppe and Richard, 1892)
L. closterocerca (Schmarda, 1859)	*Mongolodiaptomus birulai* (Rylov, 1923)
L. crepida (Harring, 1914)	*Heliodiaptomus falxus* (Shen and Tai, 1964)
L. curvicornis (Murray, 1913)	*H. serratus* (Shen and Tai, 1962)
L. hastata (Murray, 1913)	*Allodiaptomus specillodactylus* (Shen and Tai, 1964)
L. hornemanni (Ehrenberg, 1834)	**Acartiidae**
L. inermis (Bryce, 1892)	*Acartiella sinensis* (Shen and Lee, 1963)
L. leontina (Turner, 1892)	**Oithonidae**
L. luna (Müller, 1776)	*Limnoithona sinensis* (Burckhardt, 1913)
L. lunaris (Ehrenberg, 1838)	**Cyclopidae**
L. lunaris f. *crenata* (Harring, 1913)	*Eucyclops serrulatus* (Fischer, 1851)
L. ludwigii (Eckstein, 1883)	*Tropocyclops jerseyensis* (kiefer, 1931)
L. papuana (Murray, 1913)	*T. bopingi* (Dumont, 2006)

(continued)

Table 1.2 (continued)

L. quadridentata (Ehrenberg, 1832)	*Paracyclops affinis* (Sars, 1863)
L. signifera (Jennings, 1896)	*Thermocyclops crassus* (Fischer, 1853)
L. stenroosi (Meissner, 1908)	*T. taihokuensis* (Harada, 1931)
L. tenuiseta (Harring, 1910)	*Mesocyclops dissimilis* (Defaye and Kawabata, 1993)
L. ungulata (Gosse, 1887)	*M. pehpeiensis* (Hu, 1943)
L. unguitata (Fadeew, 1925)	*M. ogunnus* (Onabamiro, 1957)
Synchaetidae	*M. thermocyclopoides* (Harada, 1931)
Ploesoma hudsoni (Imhof, 1891)	*M. aspericornis* (Daday, 1906)
P. truncatum (Levander, 1894)	*M. woutersi* (Van de Velde, 1987)
P. lenticulare (Herrick, 1885)	*Microcyclops varicans* (Sars, 1963)
Polyarthra euryptera (Wierzejski, 1893)	

T. capucina, *Asplanchna priodonta*, *Conochilus unicornis*, *C. hippocrepis*, *Ploesoma hudsoni*, *Ascomorpha ovalis* and *Pompholyx sulcata*. Species within these genera of Rotifera differ between the tropics and temperate zone. For example, *Lecane*, *Brachionus* and *Trichocerca* are rich in species in the tropics, while *Keratella*, *Cephalodella*, *Notholca* and *Synchaeta* are richest in the temperate zone (Fernando 1980; Fernando and Zankai 1981; Segers 2001). In South China, *Lecane*, *Brachionus* and *Trichocerca* were the most species-rich genera, accounting for 43% of the observed rotifer species.

Lecane is tropics-centered, with about half of the recognized taxa confined to (sub)tropical regions (Segers 1996). It is known to be dominant in species diversity in tropical acid waters (Fernando 1980), where up to 40 taxa can be found in a single locality. Similarly, *Lecane* was dominant in terms of species in most of the reservoirs investigated in this study, and the maximum species number found in a single water-body was 13 (Liuxihe Reservoir). Of the 19 *Lecane* species recognized, 11 were cosmopolitan, 7 were tropicopolitan and 1 was palaeotropical. Four cosmopolitan species (*Lecane luna*, *L. lunaris*, *L. quadridentata* and *L. bulla*) and three tropicopolitan species (*L. signifera*, *L. curvicornis* and *L. papuana*) were the most widely distributed taxa in our reservoirs. The palaeotropical *L. unguitata* was only found in two reservoirs: Feilaixia and Liuxihe.

Brachionus too is predominantly tropical and subtropical with half of the species restricted to these zones. In South China, 11 species of *Brachionus* were found, and most of them were cosmopolitan. *B. angularis*, *B. calyciflorus*, *B. forficula*, *B. falcatus* and *B. diversicornis* were widely distributed in this area. *B. donneri*, an interesting element in the rotifer fauna of South-east Asia (Dumont 1983), was found in three reservoirs: Liuxihe, Feilaixia and Gaozhou. The maximum number of species (10) was found in two mesotrophic reservoirs (Gongping and Feilaixia). This genus was, therefore, frequently observed in our reservoirs, but was not predominant. This is in concordance with the finding that the dominance of *Brachionus* is decreasing from the equator towards higher latitudes, while other genera and families become more dominant with latitude (Arcifa 1984). Endemism in *Keratella* is concentrated near both poles, with no endemism in the tropics and little in the subtropics. In our study, only three *Keratella* were recognized, and *K. cochlearis* and *K. tropica* distributed widely in all reservoirs.

About 15 species of *Trichocerca* were found, and again most of them were cosmopolitan. Only two cold-water and two warm-water taxa were present. *T. cylindrica*, *T. similis, T. longiseta* and *T. capucina* were the most widely distributed species. *T. similis* f. *grandis* has a morphology similar to that of *T. similis* but with a larger body size. *T. similis* is cosmopolitan and widely distributed in the 15 investigated reservoirs, while *T. similis* f. *grandis* is tropical and only observed in Liuxihe Reservoir, located in the Tropic of Cancer. *T. cylindrica* and *T. chattoni* have a similar morphology too. However, *T. cylindrica* is regarded as a cold-water taxon, while *T. chattoni* may be pantropical. Segers (2003) suggested that tropical records of *T. cylindrica* may refer to *T. chattoni*. He said that he had never found *T. cylindrica* in collections from the (sub)tropics, or *T. chattoni* in temperate regions. However, in subtropical–tropic transition of China, both species simultaneously occurred in Liuxihe and Gaozhou Reservoirs.

In his fauna of the Freshwater Rotifera of China, Wang (1961) records only two *Polyarthra*. However, at least four species have been recognized in our reservoirs. *P. vulgaris* was widely distributed. In China, *P. remata* and *P. vulgaris* have long been misidentified as *P. trigla*, and *P. major* as *P. euryptera*. Of the genera *Asplanchna* and *Ploesoma*, representing the main predators among rotifers, *Asplanchna* is considered cosmopolitan, and was frequently observed in our reservoirs. The other predatory genus, *Ploesoma*, is usually thought to be distributed in the temperate region and not in the tropics (Fernando et al. 1990). Yet, *Ploesoma* was found widely in our reservoirs.

1.3.1.2 Cladocera

A total of 31 species belonging to 20 genera, 7 families and 3 orders have been identified to date (Table 1.2), of which 16 species are pelagic and 15 littoral. Chydoridae was the most diverse family (11 species), followed by Daphniidae (10), Sididae (4), Bosminidae (3); Macrothricidae (1), Leptodoridae (1) and Ilyocryptidae (1). The most species-rich genera were *Diaphanosoma*, *Daphnia* and *Moina*. Most of the limnetic cladoceran species in the tropics are members of nine genera: *Holopedium*, *Diaphanosoma*, *Daphnia*, *Ceriodaphnia*, *Moina*, *Moinodaphnia*, *Scapholeberis*, *Bosmina* and *Bosminopsis* (Kořínek 2002). Similarly, in our reservoirs, limnetic cladoceran species comprised *Diaphanosoma*, *Daphnia*, *Ceriodaphnia*, *Moina*, *Scapholeberis*, *Bosmina* and *Bosminopsis*. The most frequently observed pelagic species were *Bosmina tripurae*, *Bosminopsis deitersi*, *Diaphanosoma orghidani*, *D. dubium* and *Moina micrura*, while *M. rectirostris*, *M. weismanni* and *Daphnia pulex* were rare. Littoral species were primarily Chydoridae (73%), and the widely distributed species were *Coronatella rectangula*, *Alona guttata* and *Chydorus sphaericus*.

Species within individual genera of Cladocera differ between the tropics and the temperate zone. For example, *Diaphanosoma* is rich in species in the tropics. In the temperate zone, only one, rarely two, *Diaphanosoma* co-occur, but there may often be up to four and even more coexisting species in any water body in the tropics (Dumont 1994; Kořínek 2002). Three *Diaphanosoma* species were indentified and *D. orghidani* and *D. dubium* were often found simultaneously in the investigated reservoirs. Lin et al. (2003) reported that the species of *Diaphanosoma* distributed in tropical reservoirs of South China were '*D. brachyurum*' and '*D. leuchtenbergianum*'. However, it is recognized today (Korovchinsky 1992) that the name '*D. brachyurum*' partly includes *D. orghidani* and partly *D. excisum*, while the name '*D. leuchtenbergianum*' applies to *D. dubium*. *Daphnia* is an important and often dominant element of the limnetic cladoceran fauna of the temperate zone. However, in the tropics, *Daphnia* is often absent from most water bodies, probably because of high fish predation augmented by other ecological factors. Though *Daphnia* may occur in few tropical reservoirs, their abundance is usually low, and rarely two species co-occur. *Daphnia* was not widely distributed in the investigated reservoirs, and they only occurred in four reservoirs. *D. galeata* and *D. pulex* co-occurred in Liuxihe and Feilaixia Reservoirs. In all, the limnetic Cladocera were as or slightly less diverse than in temperate regions. For example, only a few species of *Daphnia* (three species), *Ceriodaphnia* (two species) and *Bosmina* (two species) were found. Even in *Diaphanosoma*, a representative genus in the tropics (Fernando 1980), only three species occurred.

Leptodora occurs primarily in the North Temperate Zone (Rivier 1998). It was reported that predatory Cladocera—*Leptodora kindtii*, *Polyphemus pediculus* and *Bythotrephes* spp.—do not occur in tropical freshwaters, and this is considered as a clear-cut difference in predator composition between tropical and southern hemisphere freshwaters on the one hand, and those of the North Temperate Zone on the other hand (Fernando et al. 1990). It is therefore interesting that *Leptodora* was found at least in three reservoirs—Liuxihe, Feilaixia and Qiyeshi—during our investigation. However, the species involved was not the Eurosiberian *L. kindtii* (Focke), but a recently described 'eastern vicariant' to it, *L. richardi* Korovchinsky (Xu et al. 2011).

1.3.1.3 Copepoda

A total of 24 species in 14 genera and 5 families is currently on record (Table 1.2). Among them, 21 species are pelagic and 3 littoral. Cyclopidae was the most diverse family (13 species), followed by Diaptomidae (6), Pseudodiaptomidae (3), Acartiidae (1), and Oithonidae (1). *Mesocyclops* was the most species-rich genus followed by *Schmackeria*, *Heliodiaptomus*, *Thermocyclops* and *Tropocyclops*.

The most frequently observed Calanoida were *Phyllodiaptomus tunguidus*, *N. schmackeri* and *Allodiaptomus specillodactylus*, while *Schmackeria spatulata*, *S. forbesi*, *S. inopinus*, *Heliodiaptomus falxus* and *Acartiella sinensis* were more rarely seen. *P. tunguidus*, *H. falxus*, *A. specillodactylus*, *S. spatulata* and *A. sinensis* are endemic to South China. Among the five endemic species, *P. tunguidus* and *A. specillodactylus* were rather widely distributed in the reservoirs, whereas *H. falxus* and *S. spatulata* occurred in only a single flow-through water body (Feilaixia Reservoir). *A. sinensis* is an estuarine species found in a pumped storage reservoir, Dajingshan Reservoir, which is located near the Pearl River estuary. The conductivity in the reservoir was high (420 μs/cm), due to its pumping up of saline water from the estuary. Compared with São Paulo Reservoirs (Arcifa 1984), Calanoida were relatively more frequent in our reservoirs with ten species found. Calanoida did not occur in only two reservoirs (Qiyeshi and Dashuiqiao) during the investigation. Two to six calanoid species occurred together in ten reservoirs, with the maximum number found in Feilaixia Reservoir. In contrast, in São Paulo Reservoirs, the simultaneous occurrence of more than one Calanoid species was rare (Arcifa 1984).

Of the 14 Cyclopoida recognized, only one species, *Tropocyclops bopingi* is endemic to South China; it was previously known, but appears in the Fauna Sinica under the name *T. parvus* (Dumont 2006). In Cyclopoida, *Mesocyclops*, *Thermocyclops*, *Microcyclops* and *Tropocyclops* are species-rich genera centered in the tropics (Fernando et al. 1990). Six *Mesocyclops* species, two *Thermocyclops* species and two *Tropocyclops* species were found in our reservoirs. The most frequently observed species were *Mesocyclops thermocyclopoides*, *T. bopingi* and *Thermocyclops taihokuensis*, while *Limnoithona sinensis* was rarely observed. Most of the species were non-cosmopolitan, with a range that usually encompassed part or most of the Oriental region.

Predatory Cyclopoida (as important invertebrate predators in the zooplankton) are primarily represented by the genus *Mesocyclops* in the tropics. Tai and Chen (1979) listed only two species, *M. leuckarti* and *M. pehpeiensis* from China. Guo (2000) later found that in reality not less than ten species live here. In the reservoirs of Guangdong Province, six species were identified: *M. thermocyclopoides*, *M. dissimilis*, *M. ogunnus*, *M. pepheiensis*, *M. aspericornis* and *M. woutersi*. Most of these previously appeared under the name of *M. leuckarti*, a Eurosiberian species, in Lin et al. (2003). *M. thermocyclopoides* was the most widely distributed species.

1.3.2 Zooplankton Abundance

Zooplankton abundance varied from 11 to 290 ind./L (Fig. 1.1) with a minimum in Xinfengjiang Reservoir and a maximum in Hexi Reservoir. Analysis of variance showed that zooplankton abundance in small reservoirs was significantly higher than that in medium and large reservoirs ($F = 4.169$, $P = 0.042$, $n = 15$). Nutrients and fish are important for zooplankton community structure and dynamics (Brooks and Dodson 1965; Hurlbert and Mulla 1981; Pinto-Coelho et al. 2005). Increased nutrients and/or decreased fish predation may induce an increase in zooplankton abundance/biomass (Kulikova and Syarki 2004; Brooks and Dodson 1965; Ostojić 2000). Oligotrophic lakes generally display a small biomass

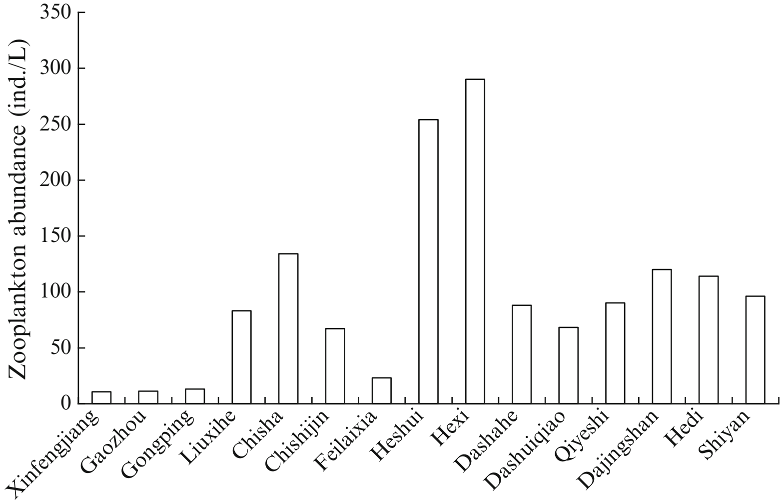

Fig. 1.1 Zooplankton abundance in the fifteen reservoirs

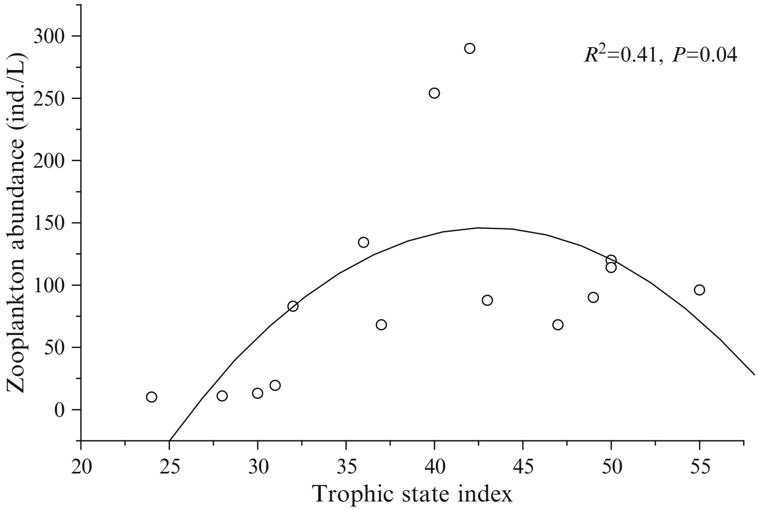

Fig. 1.2 Relationship between zooplankton abundance and trophic state index in the fifteen reservoirs

composed of a great diversity of species, while lakes in 'bloom' condition characteristic of advanced eutrophy exhibit a large biomass but fewer species. Curiously, in the reservoirs of Guangdong Province, zooplankton abundance did not increase with trophic state, and zooplankton abundance was remarkably low as well in oligotrophic as in eutrophic reservoirs (Fig. 1.2). This pattern may be related to the practice of culturing filter-feeding fish in these waters. In reservoirs with fish catch data, fish catch shows a positive relationship with trophic level

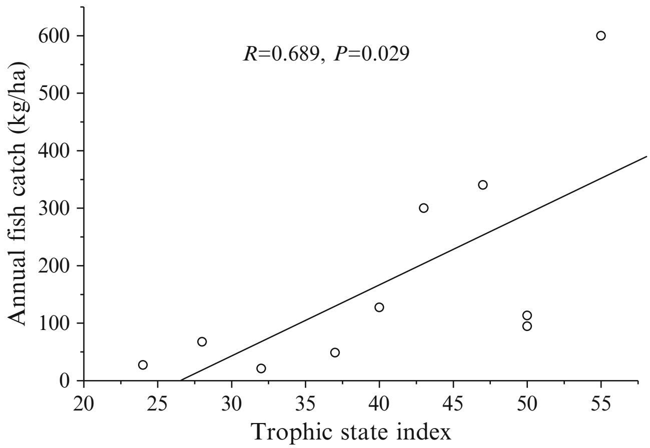

Fig. 1.3 Relationship between maximal annual fish catch and trophic state index

(indicated by TSI) (Fig. 1.3). Severe food limitation resulted in low zooplankton abundance in oligotrophic reservoirs, and increasing food quantity led to relatively higher zooplankton abundance in mesotrophic and eutrophic reservoirs. However, relatively higher predation by filter-feeding fish reduced zooplankton abundance more in eutrophic than in mesotrophic reservoirs.

1.3.3 Zooplankton Community Structure

Relative abundance of rotifers varied from 18.8% to 90.5%, with a minimum in Dashahe Reservoir and a maximum in Chishijin Reservoir (Fig. 1.4). Rotifers numerically dominated in eight reservoirs, reaching a relative abundance of more than 50%. The relative abundance of copepods varied from 1.9% to 69.5%, and was more than 50% in four reservoirs. The maximum relative abundance occurred in Gongping Reservoir, and the minimum in Chishijin Reservoir. Cladocerans were not a dominant zooplankton group in the reservoirs studied, with relative abundance varying from 0.1% to 30.3%. Moreover, the relative abundance of cladocerans was lower in eutrophic and meso-eutrophic than in oligotrophic reservoirs.

Eutrophication can affect not only zooplankton abundance but also species composition. Overlap in resource utilization and competition for similar food is common among the three zooplankton groups. As cladocerans and rotifers mature quickly and typically reproduce parthenogenetically, they show higher intrinsic rates of natural increase than copepods. These exhibit obligate sexual reproduction and must molt through six naupliar and five copepodite instars before reaching sexual maturity. Therefore, cladocerans and rotifers respond more quickly to environmental changes than copepods. Having an advantage of energetic use over the rotifers, cladocerans might have a competitive advantage, and there is indeed

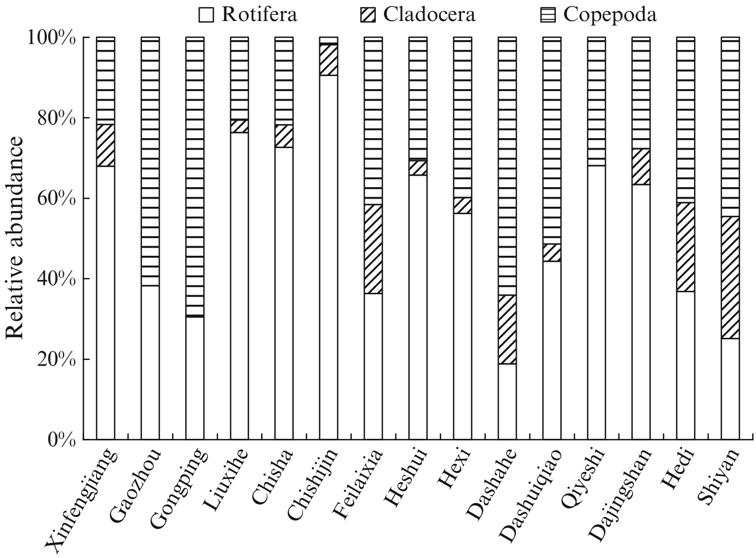

Fig. 1.4 Composition of zooplankton in the fifteen reservoirs

considerable evidence from the field that rotifers are competitively interior to cladocerans. However, rotifers typically become abundant in the plankton of those freshwaters in which the populations of cladocerans are suppressed by planktivorous fish. Predation may therefore reduce the tendency of cladocerans to outcompete rotifers. Heavy predation by juvenile and adult fish may greatly simplify the zooplankton community, and ultimately results in a scarcity of large-sized Cladocera, notably of the efficient filter feeder *Daphnia* in the tropics (Nilssen 1984). When set annual fish catch is used as a control variable, partial correlations analysis shows that there is a positive relationship between relative abundance of cladoceran and chlorophyll *a* ($R = 0.734$, $P = 0.02$) in nine reservoirs with fish catch data, suggesting that the relative abundance of cladocerans might be higher in high than in low trophic state level reservoirs under the same predation pressure. On the whole, it is likely that fish play an important role in shaping zooplankton community structure in Guangdong Reservoirs.

1.3.3.1 Rotifer Community

Trophic state has repeatedly been found to be important in determining the distribution of rotifer communities. Several studies have provided lists of rotifer species indicative of different trophic states (Maemets 1983). For example, *P. hudsoni*, *A. ovalis* and *C. unicornis* are regarded as indicators of oligotrophy, and *Brachionus* spp., *P. sulcata*, *Trichocerca* spp., *K. cochlearis* and *A. priodonta* as indicators of

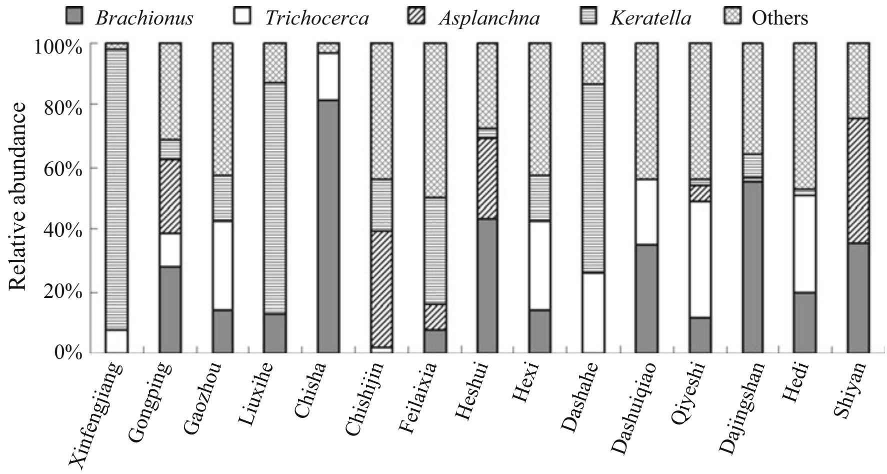

Fig. 1.5 Composition of Rotifera in the fifteen reservoirs

eutrophy. In our study, there were no species characteristics of oligotrophic or eutrophic reservoirs. Most of the above-mentioned species co-occurred in reservoirs with different trophic states. There were seven reservoirs in which *Brachionus* and *Trichocerca* contributed more than 40% of total rotifer abundance, three in which *Brachionus* and *Asplanchna* contributed more than 50% of total rotifer abundance, and three with more than 50% of total rotifer abundance contributed by *K. cochlearis* (Fig. 1.5). *K. cochlearis* is regarded as an indicator of eutrophy in some literature (Maemets 1983; Sládecek 1983). However, *K. cochlearis* was not only widely distributed in our reservoirs, but also the dominant rotifer species in oligotrophic and oligo-mesotrophic reservoirs (e.g. Xinfengjiang and Liuxihe Reservoirs). Therefore, the 'Chinese' *K. cochlearis* appears to be eurytrophic, and cannot be used as an indicator of eutrophy.

Based on physical, chemical and biological data, De Manuel and Armengol (1993) categorized 100 Spanish reservoirs into several groups but found no distinct communities typical of the different reservoir types. Each reservoir type contained a series of rotifer assemblages, with gradual changes in species composition in response to changes in environmental conditions. Similarly, rotifer distribution appears to broadly relate to only trophic state in our study. Gradients in rotifer assemblages occurred in response to changes in trophic state. In oligotrophic reservoirs (Xinfengjiang Reservoir), the rotifer assemblages were primarily dominated by *K. cochlearis*. With the increase of trophic level, the relative abundance of *K. cochlearis* is decreased, while that of *Trichocerca* and/or *Brachionus* increased. In meso-eutrophic and eutrophic reservoirs, rotifer assemblages were predominated by *Brachionus*, *Trichocerca* and *Asplanchna* (Figs. 1.5 and 1.6).

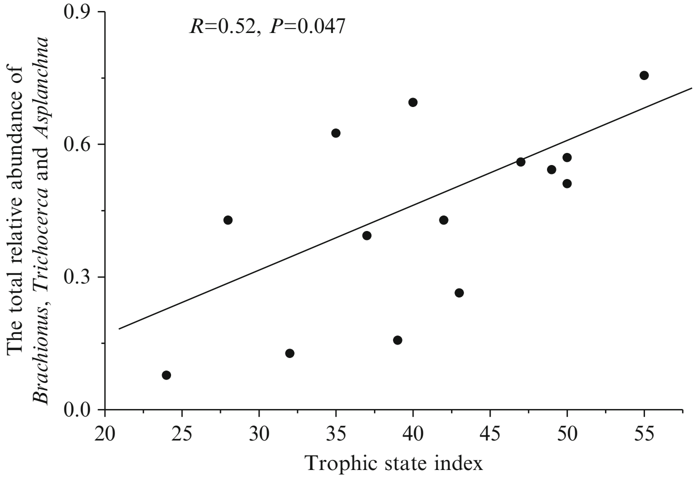

Fig. 1.6 Relationship between total relative abundance of *Brachionus*, *Trichocerca* and *Asplanchna* and trophic state index

1.3.3.2 Cladoceran Community

Most of the limnetic cladoceran species in the reservoirs of Guangdong province are members of five genera: *Diaphanosoma*, *Ceriodaphnia*, *Moina*, *Bosmina* and *Bosminopsis* (Fig. 1.7). The value of cladoceran plankters as eutrophic indicators is limited by regional specificity, probably reflecting a different evolutionary history. For example, *Ceriodaphnia quadrangula* was identified as a eutrophic state indicator in Finland but was not very useful in this regard in Poland (Cannon and Stemberger 1978). Certain bosminid cladocerans have been used as indicators of trophic conditions for many decades. In the reservoirs of Guangdong Province, *B. tripurae* and *D. orghidani* seemed to be eurytrophic, both were widely distributed and dominated the cladocerans in six reservoirs varying from oligotrophic (Xinfengjiang) to eutrophic (Hedi) and three reservoirs varying from mesotrophic (Feilaixia) to eutrophic (Qiyeshi). Both *B. deitersi* and *M. micrura* seemed to be eutrophic indicators, and dominated cladocerans in reservoirs varying from mesotrophic to eutrophic. *B. fatalis* predominated in the oligo-mesotrophic Liuxihe and Gaozhou reservoirs.

1.3.3.3 Copepod Community

The biomass ratio of calanoid copepod to cyclopoid copepod was regarded as a good indicator of the dynamics of copepod community structure. In our study, calanoid/cyclopoid abundance ratio varied from 0 to 0.33, with maximum in Gaozhou Reservoir and minimum in Dashuiqiao, Qiyeshi and Shiyan reservoirs (Fig. 1.8). There were four reservoirs with calanoid/cyclopoid ratio $\geq$0.2. Among them, Xinfengjiang Reservoir had four species of calanoids that co-occurred, while

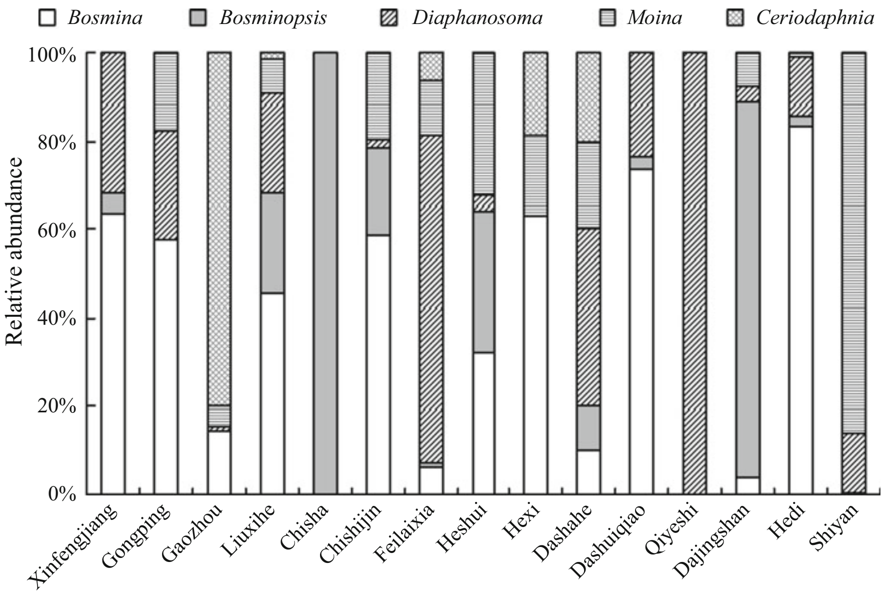

Fig. 1.7 Composition of Cladocera in the fifteen reservoirs

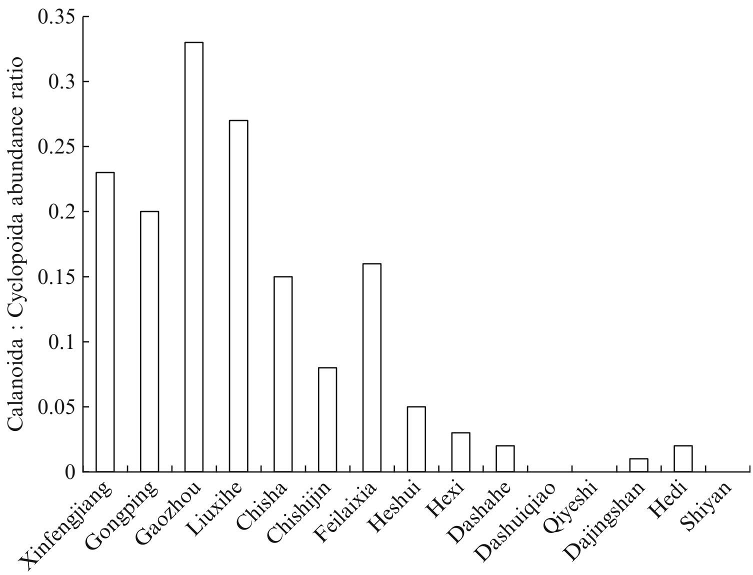

Fig. 1.8 Calanoida: Cyclopoida abundance ratio in the 15 reservoirs

Gaozhou and Liuxihe reservoirs had only one species. Rapidly through-flowing reservoir, Feilaixia, had six coexisting species of calanoids, although the calanoid/cyclopoid abundance ratio was only 0.16.

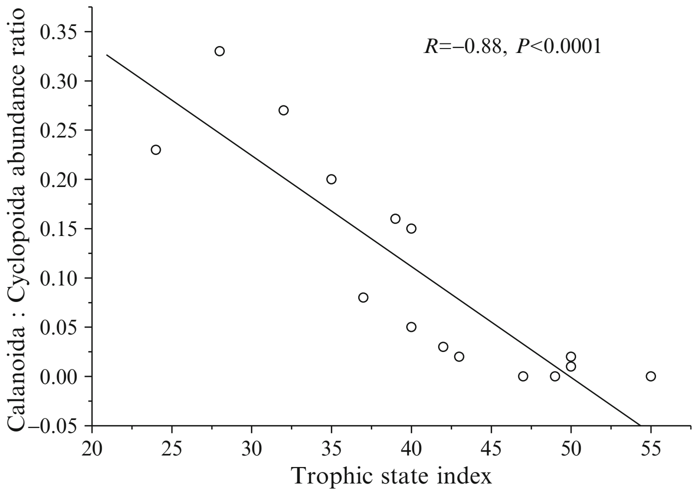

Fig. 1.9 Relationship between Calanoida: Cyclopoida abundance ratio and trophic state index

The calanoid/cyclopoid ratio tended to decrease with increasing trophic state (Fig. 1.9). Nutrients were thought to be one of the important factors inducing copepod community structure dynamics (Jeppesen et al. 2000; Hansson et al. 2004; Hurlbert et al. 1986; Hessen et al. 1995; Straile and Geller 1998; Pinto-Coelho et al. 2005). Calanoids are suspension-feeders and more herbivorous, whereas cyclopoids are raptorial and more carnivorous. Large-bodied organisms may do better in waters of variable productivity, because they have lower mass-specific metabolic rates and can accumulate sufficient resources to carry them through unproductive periods. As calanoid nauplii as well as adults have lower food threshold concentrations than cyclopoids, calanoids might affect cyclopoids negatively by reducing food supply not only for the nauplii but also for copepodites and adults (Straile and Geller 1998; Gyllström et al. 2005). Low food concentrations may prevent the establishment of a large population of cyclopoid copepods in oligotrophic reservoirs. In contrast, juvenile cyclopoids might be superior to calanoid juveniles at higher food concentrations, enabling them to exploit more efficiently food concentrations that are on the increase due to eutrophication (Straile and Geller 1998). Predation by older cyclopoid copepodites on calanoid juveniles might additionally contribute to changes in relative biomass between calanoid and cyclopoid copepods during eutrophication (Soto and Hurlbert 1991; Gyllström et al. 2005).

Fish predation too may play an important role in shaping copepod community structure (Jeppesen et al. 2000; Hansson et al. 2004; Straile and Geller 1998). The calanoid/cyclopoid ratio usually decreases when predation by fish increases. In the present study, the ratio was lower in reservoirs with relatively higher fish catch than in reservoirs with low fish catch (Fig. 1.10). It was thought that fish not only prey selectively on cyclopoids in preference to calanoids (Drenner et al. 1982; Hurlbert and Mulla 1981), but also reduce the intensity of cyclopoid predation on calanoids

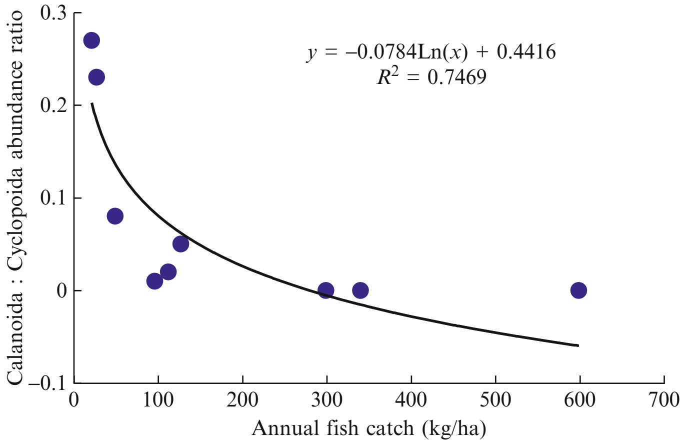

Fig. 1.10 Relationship between Calanoida: Cyclopoida abundance ratio and annual fish catch

by eliminating cyclopoids, which can greatly increase calanoid abundance. However, planktivorous fish may cause an increase in the abundance of small zooplankters and often an increase in phytoplankton as well. These, in turn, favor increased survival of juveniles of predaceous cyclopoids and this eventually leads to greater abundance of cyclopoid copepodites. Such (late instar) copepodites prey selectively on calanoids over cyclopoids, and thereby tend to increase the ratio to a 'greater extent than the selectivity of fish for cyclopoids tends to increase it' (Hurlbert and Mulla 1981).

1.3.4 Annual Succession of Zooplankton in Three Hydrological Classes of Reservoirs

Hydraulic retention time (R) is an important abiotic factor, related to the washing-out rate and retention of phosphorus by reservoirs. Straškraba and Tundisi (1999) categorized reservoirs into three classes based on their theoretical retention time, rapidly through-flowing reservoirs ($R < 14$ days), reservoirs with intermediate retention times (15 days $< R \leq 365$ days), and reservoirs with long retention times ($R > 365$ days). In order to analyze the impact of retention time on reservoir eutrophication and zooplankton response, we selected Feilaixia Reservoir ($R < 14$ days), Gongping Reservoir ($R = 133$ days) and Xinfengjiang Reservoir (644 days) to represent these three types and compared their zooplankton succession during 2000–2002.

From 2000 to 2002, zooplankton abundance varied from 10 to 24 ind./L in Xinfengjiang Reservoir and showed no significant change (ANOVA: $P > 0.05$) (Fig. 1.11). Similarly, zooplankton abundance varied little and was about 16 ind./L over the 3 years in Feilaixia Reservoir. In contrast, zooplankton abundance

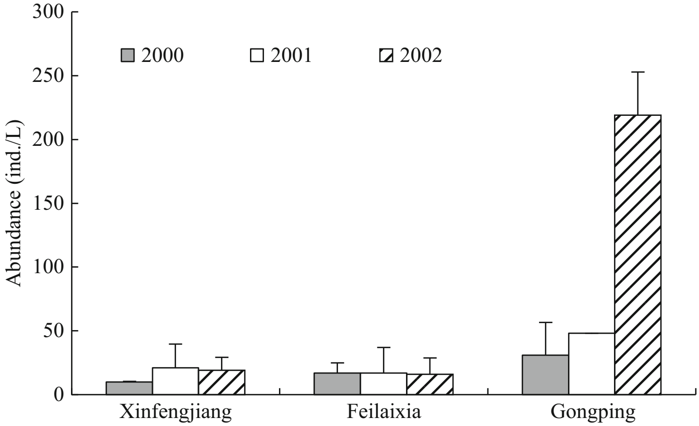

Fig. 1.11 Annual mean variation of zooplankton abundance in Xinfengjiang, Feilaixia and Gongping Reservoirs. Vertical bars represent SD

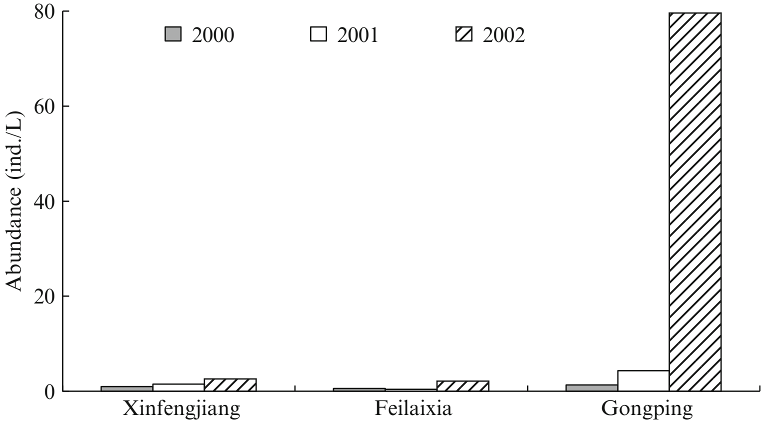

Fig. 1.12 Annual variation of *Bosmina* abundance in Xinfengjiang, Feilaixia and Gongping Reservoirs

increased rapidly over the 3 years in Gongping Reservoir. It was only 31 ind./L in 2000, but increased to 219 ind./L in 2002. *B. tripurae* was an important cladoceran species in these three reservoirs. In both Xinfengjiang and Feilaixia, the abundance of *B. tripurae* increased slightly but not significantly (Fig. 1.12). However, it increased rapidly from 1 ind./L in 2000 to 80 ind./L in 2002, and contributed mostly to the increase in total zooplankton abundance (Fig. 1.13).

The three reservoirs also showed different zooplankton community structures and changing patterns (Fig. 1.14). In both Xinfengjiang and Feilaixia reservoirs, zooplankton community structures did not change obviously over the 3 years. The most abundant zooplankton group was copepods followed by rotifers in

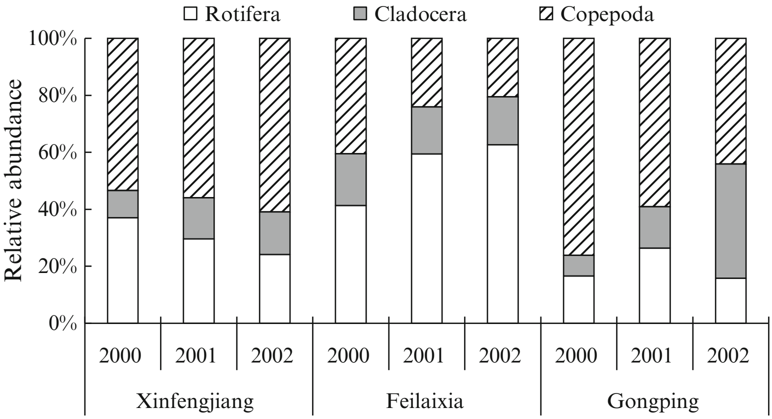

Fig. 1.13 Annual variation of zooplankton composition in Xinfengjiang, Feilaixia and Gongping Reservoirs

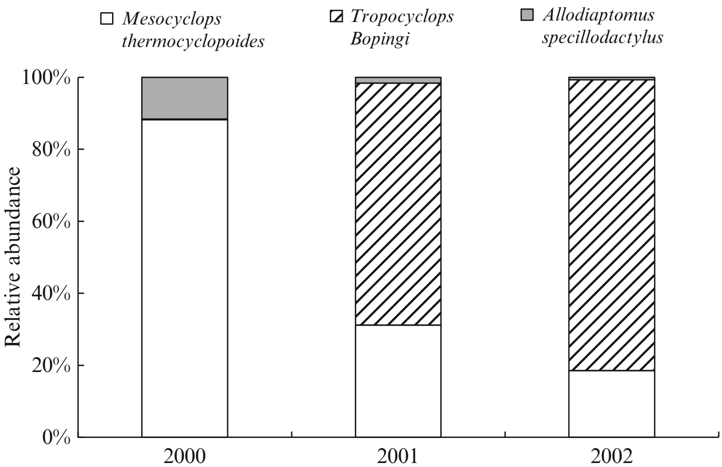

Fig. 1.14 Annual variation of adult copepod composition in Gongping Reservoir

Xinfengjiang Reservoir, while it was rotifers followed by copepods in Feilaixia Reservoir. The relative abundance of cladocerans showed no significant yearly change in these two reservoirs. In contrast, the relative abundance of cladocerans increased significantly in Gongping Reservoir. It was only 7% in 2000, and increased rapidly to 40% in 2002. Although copepods were the most abundant zooplankton group in the 3 years, their relative abundance decreased gradually from 76% to 44%. Furthermore, community structure of copepods changed significantly (Fig. 1.14). Planktonic copepods were primarily comprised of *M. thermocyclopoides*, *T. bopingi* and *A. specillodactylus*. In 2000, *M. thermocyclopoides* was the dominant taxon with a relative abundance of 88%, while the relative abundance of *T. bopingi* was less than 1%. Over the 3 years, the relative abundance of *M. thermocyclopoides* and *A. specillodactylus* decreased gradually, whereas the relative abundance of *T. bopingi* increased to 81% in 2002.

Retention time may affect zooplankton directly by advection flushing or indirectly via retention of phosphorus in the upstream zone. Xinfengjiang Reservoir is a large oligotrophic reservoir with an average depth of 29 m and a catchment/surface area ratio of 16:1. In 2000–2002, surface chlorophyll *a* concentration was around 1 mg/m^3 without apparent change (Fig. 1.15). As the retention time was as long as 640 days, the retention coefficient of phosphorus was high and most of the phosphorus input from the catchment was retained in the upstream zone of the reservoir. Therefore, total phosphorus concentration (6 μg/L or so) showed no major change in the lacustrine zone over the 3 years. Limited by phosphorus, chlorophyll *a* concentration maintained a low level at 1 mg/m^3 or so. As the retention time was long, the effect of washing-out on zooplankton was negligible. It seemed that food resources played a more important role in zooplankton dynamics than retention time. Severe food limitation resulted in low zooplankton abundance in Xinfengjiang Reservoir.

Compared with the other two reservoirs, Feilaixia Reservoir had a much higher catchment/surface area ratio of 485:1, and the shortest retention time. Phosphorus input from the catchment was high, and the total phosphorus concentration was around 30 μg/L. Kawara et al. (1998) reported that it took a retention time of 2 weeks for a sufficient increase of phytoplankton. As the retention time was only about 6 days during the study, phytoplankton was susceptible to rapid flushing displacement in Feilaixia Reservoir, and chlorophyll *a* concentration was only about 3 mg/m^3 (Fig. 1.15). Although Feilaixia Reservoir had a much higher total phosphorus concentration than Xinfengjiang Reservoir, zooplankton abundance was not higher. Unlike in Xinfengjiang, retention time in Feilaixia Reservoir might be a key factor associated with low zooplankton abundance. For phytoplankton, it takes a retention time of 2 weeks to increase sufficiently (Kawara et al. 1998).

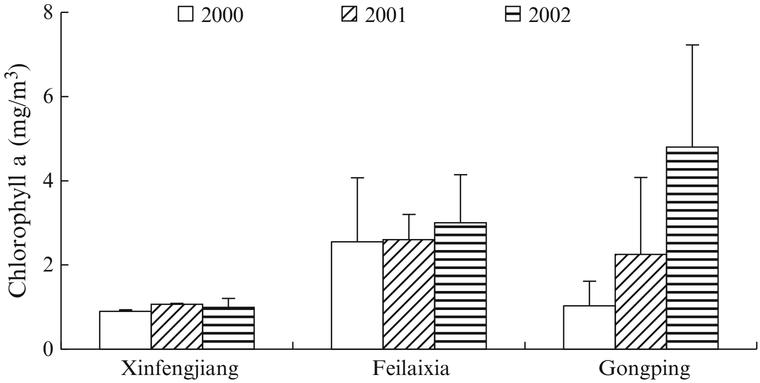

Fig. 1.15 Annual variation of chlorophyll *a* in Xinfengjiang, Feilaixia and Gongping reservoirs. Vertical bars represent SD

Zooplankton generation times are longer than those of phytoplankton; therefore, zooplankton populations may be more susceptible to rapid flushing displacement, and the time for a sufficient increase of zooplankton would be much longer than 2 weeks. Besides washing-out effects, high current velocities are known to inhibit zooplankton population growth rates. Zooplankton reproduction and thus population growth is rarely observed at velocities higher than 0.4 m/s. As the retention time was only about 6 days during the study, high advection flushing rate resulted in low zooplankton abundance in Feilaixia Reservoir. Moreover, rotifers were the most abundant zooplankton group followed by copepods, dominated by nauplii and early copepods. As stated earlier, rotifers mature quickly and reproduce parthenogenetically, and their higher population growth rate enables them to dominate zooplankton in rapidly through-flowing reservoirs (Duncan 1984; Thorp and Mantovani 2005; Rennella and Quirós 2006). In contrast, copepods exhibit ‘slow’ multi-instar sexual reproduction before reaching sexual maturity. Although lifetime fecundity can be high, it cannot offset the negative effect of this long development: their population growth rate is significantly lower than that of rotifers and cladocerans. A longer generation time and lower population growth rate also means that copepods are more susceptible to rapid flushing displacement than rotifers and cladocerans. Therefore, copepods of Feilaixia Reservoir are characterized by rapid changes in low and high relative abundance of juveniles.

In contrast to the other two reservoirs, Gongping Reservoir experienced not only abundance but also a community structure of zooplankton that changed dramatically over the 3 years. As the retention time was as long as 133 days, the effect of advection flushing on plankton may be negligible. Moreover, the aquaculture intensity of bighead carp did not change significantly. It was unlikely that both increased zooplankton abundance and changing community structure were induced primarily by dramatic changes in fish predation pressure. Gongping Reservoir is a shallow reservoir with an average depth of 4.5 m and a much lower catchment/surface area ratio of 9:1. Phosphorus input from the catchment increased yearly and phosphorus settled in the upstream zones were easily resuspended and transported to the lacustrine zone. Total phosphorus concentration of the lacustrine zone increased from 10 to 24 μg/L, and chlorophyll *a* concentration increased from 1.0 to 4.8 mg/m^3 over the 3 years. It appears likely that the increasing food resource resulted in a rapid increase of zooplankton abundance.

Increasing food not only affects zooplankton abundance but also has a considerable impact on community structure. Overlap in resource use and competition for common food are common among the three zooplankton groups. As stated above, cladocerans and rotifers mature quickly and respond more quickly to environmental changes than copepods. The energetic advantage of cladocerans over rotifers ipso facto means a competitive advantage, and there is considerable evidence from the field that rotifers are competitively interior to cladocerans. Therefore, increasing food resources induced a rapid increase of cladocerans numbers in Gongping Reservoir.

Still in Gongping Reservoir, cladocerans comprised small species (body size smaller than 1 mm), such as *B. tripurae*, *D. orghidani*, *B. deitersi*, *C. quadrangula*, *C. cornuta* and *M. micrura*. Over the 3 years, the smallest species *B. tripurae* (0.25 mm or so) was the dominant cladoceran, and increased rapidly with eutrophication (Fig. 1.13). Competition when food is limiting may involve two rather distinct factors related to body size: the threshold food concentration and resistance to starvation when food levels are below the threshold. Larger cladocerans tend to be more resistant to starvation than smaller ones. Gliwicz (1990) argued that the threshold concentration in cladocerans generally decreases with increasing body size of the animals: the larger-bodied taxa had an apparent advantage over smaller-bodied taxa. However, Nandini and Sarma (2003) have shown that this relationship is valid only for individuals about 1.3 mm long and above. The trend reverses below this size and the threshold concentration increases with increasing body size. It is likely that the threshold concentration of *B. tripurae* was lower than those of the other cladoceran species. Although food limitation alleviated slightly over the 3 years, food resources may still be below most of the larger species' threshold level. Therefore, *B. tripurae* might outcompete the other cladoceran species and increased rapidly. On the other hand, larger cladoceran competitors were also suppressed by planktivorous fish (such as bighead carp). Although increasing food resources likely increased the reproduction rate of larger species, selective predation might reduce the tendency of larger species to outcompete *B. tripurae*.

Juveniles and adults of *A. specillodactylus* and *T. bopingi* and juveniles of *M. thermocyclopoides* are suspension feeders and mostly herbivorous, whereas adult *M. thermocyclopoides* are raptorial and carnivorous. Interspecific competition among these three species might be an important mechanism regulating the community structure of copepods. As calanoid nauplii as well as adults have lower food threshold concentrations than cyclopoids, calanoids might be able to affect cyclopoids negatively by reducing food, not only for the nauplii but also for copepodites and adults (Straile and Geller 1998; Gyllström et al. 2005). Low food concentrations might prevent the establishment of a large population of cyclopoid copepods in oligotrophic reservoirs. In contrast, juvenile cyclopoids might be superior to calanoid juveniles at higher food concentrations, enabling them to exploit more efficiently increasing food concentrations due to eutrophication (Adrian 1997; Soto and Hurlbert 1991; Straile and Geller 1998). Therefore, the calanoid/cyclopoid ratio shows a tendency to decrease with increasing trophy. Possibly, the increasing input of phosphorus allowed cyclopoids, especially *T. bopingi*, to outcompete calanoids in Gongping Reservoir.

1.4 Conclusions

In reservoirs of South China, zooplankton was composed of cosmopolitan and 'southern' species. The occurrence of a large predatory cladoceran, *L. richardi*, was interesting and somewhat unexpected. As South China is located in a high-latitude tropical zone (transition from tropical to subtropical zones) with clear seasonal variation in water temperature, temperate species may still invade the investigated reservoirs and maintain a population in them. On a regional scale, trophic state, fish predation and retention time were the three most important factors affecting zooplankton community structure and abundance. Higher zooplankton abundance occurred in mesotrophic reservoirs, while the relative abundance of cladocerans in higher trophy reservoirs might be higher than that in low trophy reservoirs under the same fish predation pressure. The relative abundance of *Brachionus*, *Trichocerca* and *Asplanchna* increased and the ratio of Calanoida to Cyclopoida decreased with trophic state. Retention time was an important factor influencing the response of zooplankton to eutrophication. Not only zooplankton abundance, but also community structure responded rapidly to eutrophication in reservoirs with intermediate or long retention times, although the response was not so apparent in through-flowing reservoir.

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Chapter 2
Species Richness and Community Structure of Pelagic Zooplankton in Tropical Reservoirs, Hainan Island

Huiming Li, Jingjing Ren, and Qiuqi Lin

Abstract Forty-seven species of zooplankton, including 7 species of copepods, 8 species of cladocerans and 32 species of rotifers, were identified in an investigation of seven reservoirs in Hainan Island in the dry (December 2006) and wet season (May 2007). All species were eurythermal or mesophilic and their communities were dominated by small-sized species. Common crustaceans included *Mesocyclops thermocyclopoides*, *Bosmina fatalis* and *Bosminopsis deitersi*. The rotifers *Brachionus forficula* and *B. falcatus* dominated in the early flood season, while *Filinia camasecla* dominated in the late dry season. Species richness in oligotrophic and oligo-mesotrophic reservoirs was much higher in the wet than in the dry season, but species richness in the eutrophic reservoirs had a reversed seasonality. The total abundance of zooplankton ranged from 124 to 2,966 ind./L, and biomass ranged from 42.9 to 1,212 μg/L. Abundance of *B. fatalis* and *Diaphanosoma dubium* was positively related to that of most species, and *Thermocyclops taihokuensis* and *M. thermocyclopoides* were weakly but negatively related to the Cladoceran species. This weak interspecific correlation indicates that other factors such as fish predation affected the coexistence of zooplankton species.

2.1 Introduction

Zooplankton is present in freshwaters all over the world. It is a community that is intermediate between phytoplankton and fish, and it is therefore important in maintaining the integrity of aquatic ecosystems. It is found in large lakes, ponds and rivers, temporary pools and down to the smallest water bodies (Dussart and Defaye 1995). Most cladocerans feed on phytoplankton, and are eaten by invertebrate and vertebrate planktivores. They are the key link in 'bottom-up' and 'top-down' effects

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in aquatic food chains. Rotifers are small but have a relative high diversity. They differ from cladocerans in niche, and feed mainly on small particulate organic matter (including phytoplankton, bacteria, and detritus) (Scheda and Cowell 1988; Yoshida et al. 2003). They have a short life span with rapid turnover and high production. Rotifers are sensitive to environmental conditions, and high population densities can be observed in eutrophic waters (Scheda and Cowell 1988). Their composition in communities has been used widely to evaluate the trophic level of water bodies (Herzig 1987).

Climate, habitat, competition, predation and primary productivity (food) are considered the main factors regulating species diversity and community structure. In Paine's predation hypothesis, density and richness of predators in the tropical zone are higher than in other zones. High predator numbers reduce population density of prey but allow diverse prey species to coexist, thereby explaining the higher diversity of biological communities in the tropics compared to temperate and polar regions (Paine 1966). It has been suggested that species richness of zooplankton is lower in the tropics than in temperate zone, dominated by small-bodied species in all three classical groups (Fernando et al. 1987). But species diversity in an individual water body is affected by sampling effort and historical data, and thus conflicting views on tropical zooplankton diversity remain (Dumont 1994; Zhao and Han 2006).

Reservoirs are the result of damming rivers. Being semi-artificial ecosystems, fisheries and water resource management control their fish diversity and hydrodynamics. Human activities strongly affect ecological processes, including species composition and community structure of zooplankton. Hainan Island, located at the northern fringe of the Oriental tropics, is separated from the Chinese mainland by Qiongzhou Strait. Monsoon season on the island is from May to October, and brings 70% of annual precipitation (Liu 2006). There are many rivers, but these are short and have a low capacity of water supply. A number of reservoirs have been constructed for water storage in the past 50 years. Until now, no work has been carried out on the zooplankton of these reservoirs; their species composition and community structure of zooplankton remain undocumented. In the present study, we examine the zooplankton of seven reservoirs in Hainan Island and analyse species composition, community structure and main factors that influence the zooplankton.

2.2 Materials and Methods

2.2.1 Study Reservoirs

We conducted an investigation in the pelagic zone of seven reservoirs in Hainan Island in the season (December 2006) and flooding season (May 2007). Each reservoir was sampled near its dam (Table 2.1 and Fig. 2.1).

Table 2.1 Characteristics of seven investigated reservoirs on Hainan Island

Reservoir	Capacity (10^3 m^3)	Catchment (km^2)	Water temperature (°C)		Phytoplankton biomass (mg/L)	Trophic level	
			Dry season	Flood season		Dry season	Flooding season
Songtao	33,450	1,440	22	28	1.61	Oligotrophic	Mesotrophic
Shilu	14,130	353.63	22	27.3	1.09	Oligotrophic	Mesotrophic
Wanning	15,200	429	20	28.8	4.41	Mesotrophic	Eutrophic
Chitian	7,710	220.55	24	28.4	8.42	Mesotrophic	Mesotrophic
Nanfu	9,162	64.5	21.5	27.9	14.17	Mesotrophic	Mesotrophic
Gaopoling	6,790	156.4	20	28.6	16.68	Eutrophic	Eutrophic
Yongzhuang	775	14.58	19.3	31.6	0.84	Mesotrophic	Eutrophic

Fig. 2.1 Location of the investigated reservoirs

2.2.2 *Sampling Methods*

Qualitative samples of zooplankton were collected with vertical and horizontal tows of a 50 μm mesh plankton net. Quantitative samples were vertically collected by a 30 μm mesh plankton net at five depths and integrated into one sample. These samples were preserved in 5% sucrose formalin and counted under the microscope (Guo 2000; Dumont 2000). Biomass was calculated by assuming a density of zooplankton of 1 g/mL, according to regression equations relating volume and dry weight to body length (Dumont 1975).

2.3 Results

2.3.1 *Species Composition*

In all, 47 species of pelagic zooplankton, including 7 species of copepods, 8 species of cladocerans and 32 species of rotifers, were identified (Table 2.2). Most species

Table 2.2 Zooplankton species and frequency in the seven reservoirs in Hainan Island (P – Planktonic, B – Benthic; F – Flooding season, D – Dry season)

Species	Songtao		Shilu		Chitian		Nanfu		Wanning		Gaopoling		Yongzhang	
Cladocera	F	D	F	D	F	D	F	D	F	D	F	D	F	D
Bosmina fatalis	++	++	++	++	++	++	++	++	++	++	++	++	++	++
Bosminopsis deitersi	+		+	+	+	+	+		++	++			+	+
Ceriodaphnia cornuta	++	+				+	++	++						+
Diaphanosoma orghidani	++	++	++	+	++	+	++	++	+	++	++	+	++	+
D. dubium	++	+	+		+	+	+	+		+	++		+	+
Moina micrura	+		+				+			+				
Daphnia galeata		+										+		
Alona milleri									+		+			
Copepoda														
Thermocyclops taihokuensis										+				
Mesocyclops thermocyclopoides	+	+	+	++	+	+	+	++	+	++	++	++	+	+
Allodiaptomus uenoi							+	+		+				
Neodiaptomus schmackeri	+	+	+		+	+								
Mongolodiaptomus birulai	+	+	+	+	+	+								+
Tropocyclops jerseyensis									+					
T. bopingi								+						
Monogononta														
Brachionidae														
Brachionus calyciflorus P								+	+	+		++	+	++
B. diversicornis P						+	+		+	+	+	+	+	+
B. falcatus P			++		++	+			+	+			++	+
B. donneri	+	+					++	+						
B. angularis P					+		+		++	+			+	+
B. forficula P	++	+	++		++	++	++		+	+	++	+	++	+
B. budapestinensis P			+		+					+		+		
Keratella tropica P					+	+	+	+	++	++	++	+	+	+
K. cochlearis P	+	+	+	+			+	+		+	+		+	
Asplanchnidae														
Asplanchna brightwelli P									+		+	+		+
A. priodonta P									+					

Trichocercidae														
Trichocerca capucina P		+	+	+	+	+			+	++		+		
T. chattoni P	++	+			+	+	+				+	+		
T. pusilla B					+									
T. stylata P			+	+					+		+	++	+	
T. capucina P				++	+			+	+	+	+	+	+	+
Filiniidae														
Filinia longiseta P										+		+		
F. camasecla P	+	++	+	++		+	+	++	+			+	+	++
F. opoliensis P		++			+	++	+	++	+	+			+	
Hexarthridae														
Hexarthra mira P				+										
Conochilidae														
Conochiloides dossuarius P								+				+	+	
Conochilus unicornis P				+										
Collothecidae														
Collotheca										+				
Gastropodidae														
Ascomorpha ecaudis P				+						+				
A. saltans P												+		
Gastropus stylifer P					+									
Synchaetidae														
Polyarthra sp. P				+						++	+	+		
Synchaeta stylata P				+								+		
Ploesoma hudsoni P			+		+	+				+			+	+
Testudinelidae														
Pompholyx sulcata P					+									
Lecanidae														
Lecane unguitata B													+	
L. lunaris B											+			

+ Present; ++ dominant species

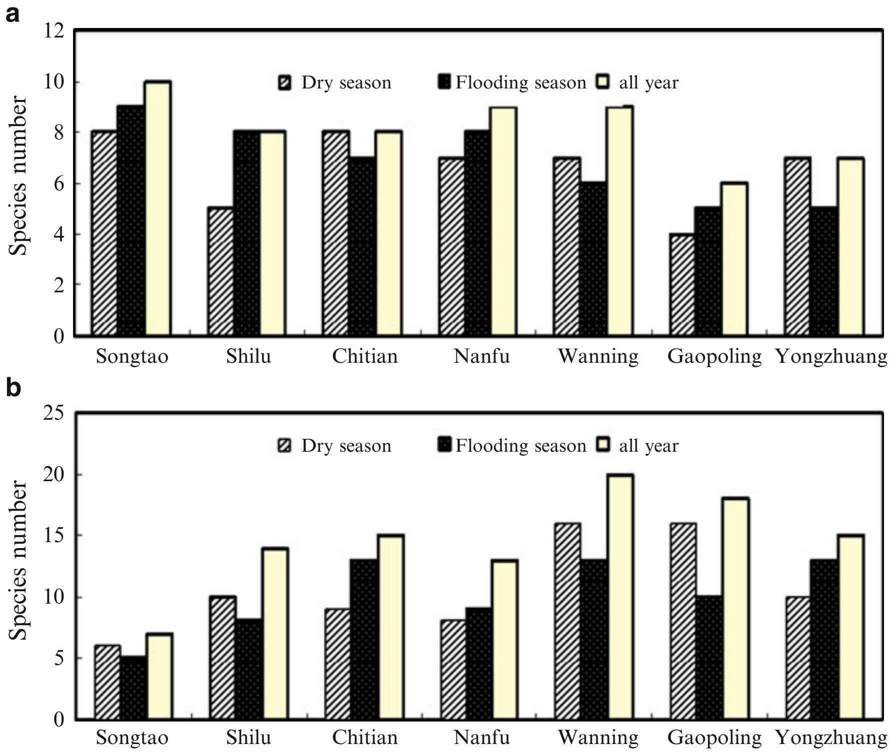

Fig. 2.2 Species number of zooplankton in seven reservoirs in Hainan Province (**a**, planktonic crustaceans and **b**, rotifers)

were planktonic and eurythermal or mesophilic. The communities were dominated by small-sized species. Species such as *Brachionus forficula*, *B. falcatus*, *B. calyciflorus*, *B. diversicornis*, *Keratella tropica*, *K. cochlearis*, *Filinia camasecla*, *F. opoliensis*, *Trichocerca chattoni*, *T. capucina*, *T. stylata*, *Ploesoma hudsoni*, *Mesocyclops thermocyclopoides*, *Bosmina fatalis* and *B. deitersi* were commonly observed in all reservoirs.

Songtao reservoir, a large and oligotrophic water body, had the richest species (ten) of planktonic crustaceans. Gaopoling reservoir, a eutrophic water body, had the least (six). There were five species that differed between the wet and dry seasons, including *Daphnia galeata*, *Alona milleri*, *Thermocyclops taihokuensis*, *Tropocyclops jerseyensis* and *T. bopingi*. Rotifers showed highest species richness in eutrophic waters such as Wanning (20 species) and Gaopoling reservoirs (18), and lowest in the oligotrophic Songtao reservoir (5) (Fig. 2.2).

The body length of the cladocerans ranged from 233 to 875 μm, averaging 394 μm. The planktonic copepods ranged from 140 to 1,250 μm, averaging 940 μm. Rotifers ranged from 75 to 375 μm, averaging 225 μm. The smallest

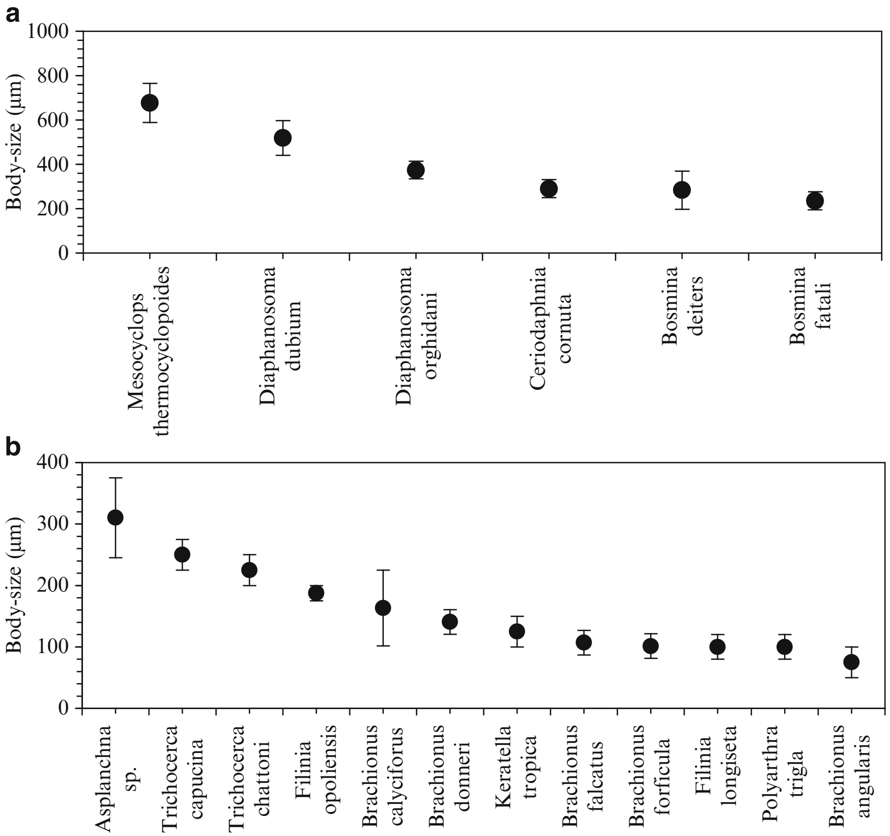

Fig. 2.3 Body length of dominant zooplankton in seven reservoirs in Hainan Island (**a**, planktonic crustaceans and **b**, rotifers)

species was *B. budapestinensis*, and the largest was *Mongolodiaptomus birulai*. Most of these dominant species were small sized (Fig. 2.3).

2.3.2 Abundance and Biomass of Zooplankton

The total abundance of zooplankton ranged from 124 to 2,966 ind./L, and total biomass ranged from 42.9 to 1,211 μg/L. Distribution of abundance was consistent with biomass (Fig. 2.4). Total abundance was higher in the dry season than in the flooding season, except in Gaopoling reservoir (Figs. 2.5 and 2.6).

In both the flooding and dry seasons, the dominant group was composed of species of body length less than 400 μm. Rotifers within this class were less than 200 μm and dominated by *Brachionus* spp. Although the abundance of zooplankton

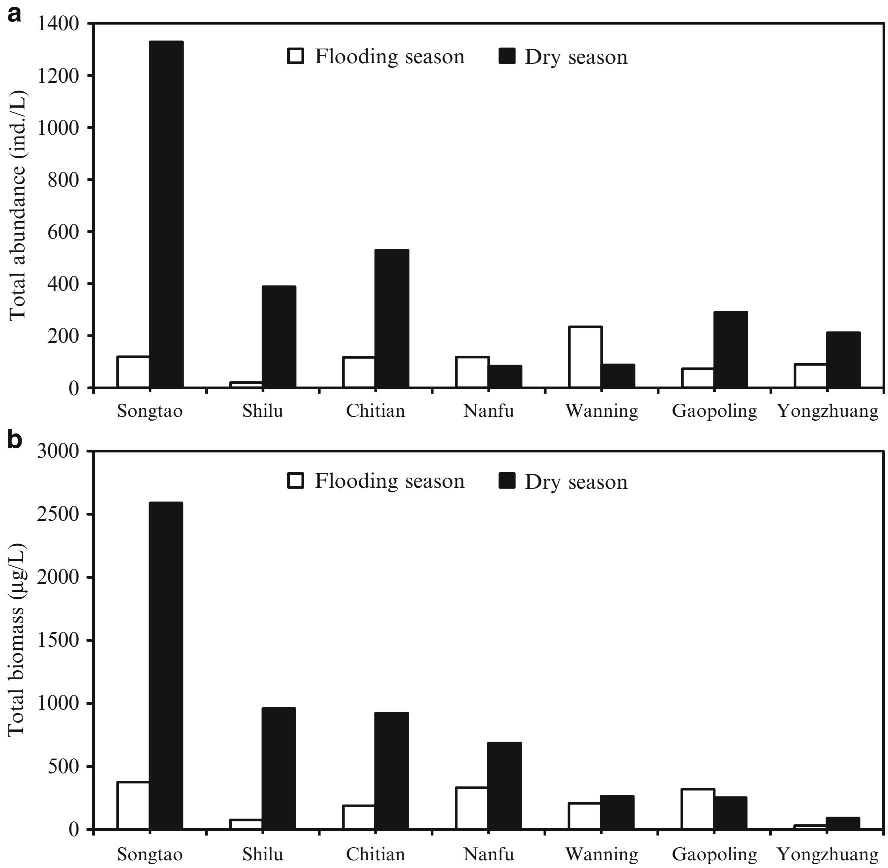

Fig. 2.4 Zooplankton biomass and abundance in the seven reservoirs in the wet and dry seasons (**a**, abundance and **b**, biomass)

was mainly contributed by small-sized species, the biomass of zooplankton was mainly due to cladocerans and copepods.

2.4 Discussion

2.4.1 Structure of the Zooplankton Community

Compared to freshwater bodies in the temperate zone, tropical pelagic zooplankton has a low species number at the usual sampling efforts and is dominated by small-sized species. Larger species of planktonic crustaceans, e.g. *Daphnia* (Lin et al.

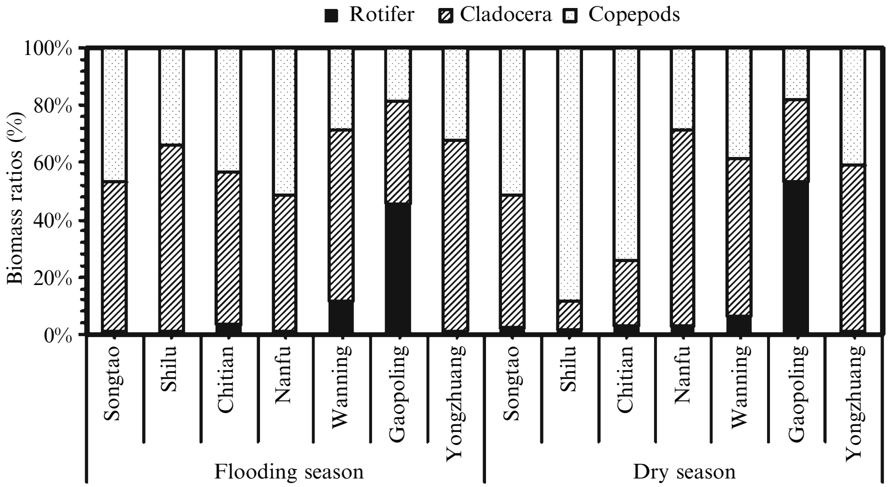

Fig. 2.5 Relative biomass of zooplankton in seven reservoirs during the wet and dry seasons

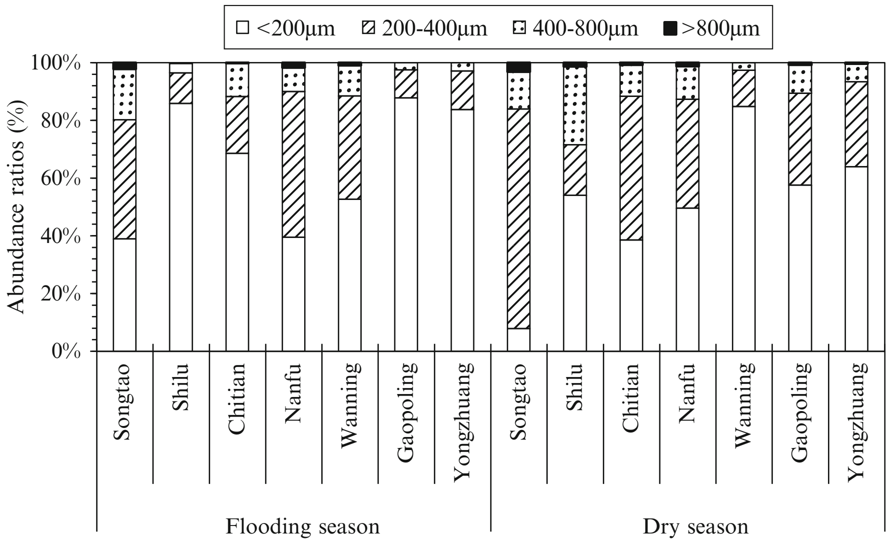

Fig. 2.6 Abundance of zooplankton over four body length classes

2003) are generally rare. Even compared to Guangdong Province, species number were much lower in the seven investigated reservoirs in Hainan Island.

Eight species of cladocerans in total appeared in the seven reservoirs, and seven of these were small with a body length that ranged from 200 to 800 μm. The most abundant stages of the copepods were nauplii and copepodid larvae. *M. thermocyclopoides* was a dominant species with a body length that ranged from 600 to 900 μm. This is in agreement with the observation that most dominant cladocerans and copepods in tropical lakes are less than 1 mm in size (Lewis 1996). *Phyllodiaptomus tunguidus* is endemic to southern China, and is usually dominant among planktonic crustaceans in most large or middle oligo-/mesotrophic reservoirs in Guangdong Province (Zhao and Han 2007; Lin and Han 2006). *P. tunguidus* is a large, filter-feeding species that moves quickly and reacts swiftly to avoid predators (Dumont and Reddy 1993). However, this species was not observed in any of the seven reservoirs of Hainan Island. More surveys need to be conducted in future to detect this species.

When the rotifers were grouped by dominant genera, four basic modes of community structures occurred in the dry season: the basic mode in Songtao, Chitian, Yongzhuang and Nanfu reservoirs was *Filinia* + *Brachionus*; the second mode in Shilu reservoir was *Trichocerca* + *Brachionus*; the third mode in Wanning reservoir was *Polyarthra* + *Keratella* + *Trichocerca* and the fourth mode in Gaopoling reservoir was *Trichocerca* + *Brachionus* + *Keratella*. Apparently, the dominant species coexist in the dry season, but no single species absolutely dominates the community. In the flooding season, however, *Brachionus* spp. became absolutely dominant. *Brachionus* has been widely observed as a common genus in tropic freshwaters (Dussart et al. 1984; Arndt 1993). It is worth pointing out that there was a difference between the dominant species of *Brachionus* in flooding season and in dry season: *B. forficula*, *B. falcatus* and *B. angularis* dominated in the wet season, *B. forficula* and *B. calyciforus* in the dry season.

2.4.2 Factors Regulating the Zooplankton Community

2.4.2.1 Reservoir Type and Trophic Level

Generally, a large water body provides more habitats and thus maintains a higher species richness than smaller water bodies. Dodson (1992) found that the species richness of planktonic crustaceans in Europe and North America increased with lake area. In our seven reservoirs, species of planktonic crustaceans presented a positive correlation with reservoir size ($r = 0.765$, $F = 7.073$, $P = 0.045$). However, the diversity of pelagic rotifers decreased with an increase in reservoir storage capacity. Thus, species diversity of both planktonic crustaceans and rotifers were affected by storage capacity of reservoirs, but perhaps in opposite ways (see below).

Food is one critical factor limiting zooplankton abundance and species richness. In extreme situations, such as oligotrophy (little food) and eutrophy (an abundance of food), a few species are able to dominate and outcompete all others. In the present seven reservoirs, zooplankton species decreased with an increase in trophic state (Tables 2.1 and 2.2). Trophic state of a water body reflects food quantity and quality for zooplankton (Lin et al. 2005). In tropical reservoirs, and especially in eutrophic ones, phytoplankton comprises a high percentage of indigestible cyanophytes throughout the year. Small-sized species of cladocerans prefer to feed on small-sized unicellular algae, bacteria and detritus. Small-sized cladocerans are weakly affected by filamentous and large colony species, and often appear in large numbers in eutrophic water bodies (Vijverberg and Boersma 1997). Li et al. (2010) compared planktonic crustacean community structure in three subtropical reservoirs in Guangdong Province and found that abundance and biomass of copepods increased with trophic level of reservoirs. In our seven reservoirs, however, the abundance and biomass of copepods decreased with increasing trophic level (Fig. 2.4). This opposing trend implies that fish predation has a great impact on copepod distribution in tropical reservoirs. *D. galeata* only appeared in the dry season (so-called winter) in Songtao reservoir, which is an oligo-mesotrophic large and deep reservoir. *D. galeata* has a suitable food source and is able to migrate vertically to avoid fish predation and even the high temperatures in the epilimnion (Gliwicz 1986; Gliwicz and Pijanowska 1988).

2.4.2.2 Temperature and Fish Predation

Temperature variations alter predation pressure and food availability, and are considered to have a major effect on seasonal variation, including rotifer species (Edmondson 1946). High temperatures, like in the tropics, stimulate growth, but low temperatures support lower growth and reproduction rates, except in cold-adapted species and genera. In the dry (and relatively fresh) season, the rotifer community mainly consisted of medium-sized *Trichocerca* and *Polyarthra*. In the flooding (and hot) season, the rotifers shifted to small-sized *Brachionus*, many of which were armoured. Their hard body is helpful as a defence against small predators like cyclopoids (Pejler and Berzins 1989).

Fish predation has been considered the single major force in controlling densities and structure of zooplankton communities (Zaret 1980). In tropical reservoirs, fisheries are also an important source of income and aquaculture is widespread. The cultivated fish species are planktivores such as bighead and silver carp (Lin et al. 1984). In the dry season in Hainan Island, decreasing water temperatures and fish harvesting result in lowered fish predation pressure. Large body-sized species of zooplankton may now appear and increase. In the flooding reason, high fish predation pressure quickly eliminates these large prey and switches zooplankton community structure to small taxa (rotifers) (Zaret 1980; Gliwicz and Pijanowska 1988).

Cladocerans have a high filter-feeding rate, and they are the most efficient competitors for rotifers (Gilbert 1988). In Hainan Island reservoirs, the abundance of cladocerans was low (on average, 32.6 ind./L) and food competition with rotifers was therefore not intense. Copepods were the main predators of rotifers. They had a high abundance, with an average of 164.4 ind./L. This invertebrate predation pressure may have caused the dominant rotifers to be species with anti-copepod defences, such as armour and spines.

That interspecies competition had a strong effect on community structure could also be seen from the temporal differences; dominant but related species tended to occur in different time slots, to avoid direct confrontation (Ciros–Perez et al. 2001). *Filinia* was dominant in the dry season, and *Brachionus* in the flooding season. *K. tropica and K. cochlearis* separately dominated in different periods in all seven reservoirs. This replacement indicates that these species evolved a temporal-niche differentiation.

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Chapter 3
Seasonal Dynamics of *Daphnia galeata* in a Reservoir at the Edge of the Tropics Before and After Yearly Stocking with Bighead Carp

Qiuqi Lin, Bo-Ping Han, and Henri J. Dumont

Abstract We studied the seasonal dynamics of *Daphnia galeata* from April 2001 to August 2009 in oligo-mesotrophic Liuxihe Reservoir (South China). Until 2004, fish (mainly bighead carp, *Hypophthalmichthys nobilis*) was released yearly and accounted for 85% of total fish catch. The commercial bighead carp catch, 21 kg ha^{-1} year^{-1}, suggests a substantial predatory pressure on zooplankton. In 2004, most of the fish were removed and commercial introductions ceased. The abundance of *D. galeata* was low (from 0 to 1,500 individuals m^{-3}) and with a clear-cut seasonal pattern: a decline in summer, absence in autumn, appearance in early winter, a peak in late winter (February), and a decline towards early summer. Edible nanophytoplankton biomass (<20 μm) was low and was mostly composed of *Cyclotella* spp. and *Peridinium pusillum*. *D. galeata* peaked at low but non-zero abundances of edible nanophytoplankton, indicating that food limitation was not the direct cause of its absence in summer and autumn, when a relatively high edible nanophytoplankton biomass was present. In a first phase, from 2001 to 2004, *D. galeata* negatively correlated with water temperature. The high surface temperature (32°C) during summer suggests that its development time and life span both shortened so strongly that its lifetime fecundity at that time sank below the threshold, at prevailing needed for maintaining an active population against the fish predation pressure.

In 2005, a pelagic flatworm (*Rhynchomesostoma* sp.), supposedly suppressed by fish predation earlier, appeared. It fed on all zooplankton that it could catch and paralyze using a toxin that it also continuously lost to the environment and that

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remained active there for several months. *Daphnia* initially became more common, but its phenology became even more restrictive than before, and it now disappeared 1 month earlier, in July instead of August. Its negative correlation with temperature broke down, because by the time the higher temperatures in the reservoir were reached, it had already disappeared. Its place in the pelagic was taken by *Ceriodaphnia quadrangula*. Culture and enclosure work showed that the latter was less sensitive to flatworm toxin than *Daphnia*. Interestingly, the flatworms were susceptible to their own toxin, a possible auto-regulation mechanism.

3.1 Introduction

Daphnia, a major genus of anomopods in temperate lakes, plays a central role in the structuring of pelagic communities (Brooks and Dodson 1965; Benzie 2005). Seasonal population dynamics of *Daphnia* in temperate lakes are often associated with parallel changes in predation pressure (Hrbacek et al. 1961; Gliwicz and Pijanowska 1989; Naselli-Flores and Barone 1997). The animals usually drop to an environmentally determined low density in winter, rise in spring due to abundant food and low predation, peak during the clear water phase, and decline in summer (Sommer et al. 1986; Benndorf et al. 2001). Fish predation in temperate-zone winter is negligible; late spring-early summer is when planktivorous fishes exert the greatest predation pressure (Gliwicz and Pijanowska 1989; Černý and Bytel 1991; Gliwicz 1994). In this period, both the feeding intensity of planktivores and the density of young-of-the-year (YOY) fish are high. Because YOY fish are much more important predators than older-year classes (Gliwicz and Pijanowska 1989), biomanipulation experiments showed that removal of planktivorous fish effected a significant increase in the densities of *Daphnia* (Korponai et al. 2003; Beklioglu et al. 2003; Rask et al. 2003).

Compared to the temperate zone, predation by planktivorous fish is much heavier in the tropics, where it is believed to be the key factor leading to a low abundance or absence of *Daphnia* (Dumont 1980; Fernando 1980; Nilssen 1984). As both water temperature and photoperiod do not change much throughout a tropical year, seasonal changes in predator feeding activity and population density are negligible (Gliwicz and Pijanowska 1989). However, the tropical zone is not uniform. There exist gradations between the two tropics and the equator, and there is also a distinct influence of altitude. For example, in Lake Tana in Ethiopia, *Daphnia* fluctuates seasonally, with highest densities during the dry season and lowest densities in the rainy and post-rainy season. Here, turbidity – not predation – was the key factor in the dynamics of *Daphnia* (Dejen et al. 2004). In the fringing zone of the northern hemisphere tropics, a seasonal change in surface water temperature (of the order of 10–12°C) occurs and induces a seasonal pattern in the temperature-dependent feeding rate of planktivores (Gliwicz and Pijanowska 1989). South China is a part of this subtropical–tropical transition zone. *Daphnia* appears only in few lakes and reservoirs here, rapidly becomes rare, and finally disappears as one moves south (Lin et al. 2003).

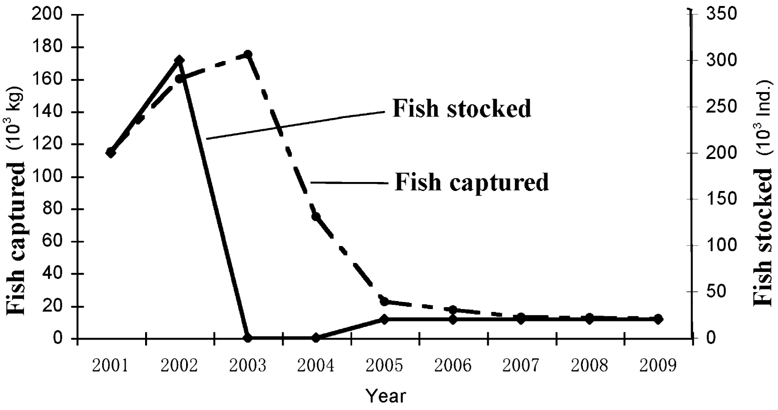

Fig. 3.1 Fish stocking and removal from Liuxihe from 2001 to 2009. Major fish removal was during 2004–2005 (data by Liuxihe Reservoir fishery group) and was followed by only small-scale introductions for angling purposes (After Wang et al. 2011, reprinted from Hydrobiologia, with permission)

In this paper, we describe the seasonal pattern of *Daphnia galeata* in one of these reservoirs, Liuxihe, and analyze the effects of releases of bighead carp (2001–2004), followed by the remarkable changes that took place after the fish had been artificially removed (2005–2009). Bighead carp (*Hypophthalmichthys nobilis*) is extensively cultivated in South Chinese reservoirs. It produces semi-buoyant eggs that require a certain water current to float. There are no appropriate spawning sites with sufficiently long floatation times for its eggs in reservoirs, so bighead carp is not able to reproduce successfully here, and its population is maintained by periodic introduction of young-of-the-year (YOY). Until 2004, YOY was introduced at 15 cm body length in April–May and/or August–November every year. Fish $\geq$ 3 years old were caught by seining from April to October. Therefore, seasonal changes in the abundance of planktivorous fish used to be regulated by artificial fish introduction and removal. However, in 2004–2005, most fish were removed in one single major fishing effort by a private company (Fig. 3.1), and large-scale yearly reintroductions ceased. The effects of this manipulation on the zooplankton community and its flagship species, *D. galeata,* but also *Ceriodaphnia quadrangula* and the flatworm *Rhynchomesostomas* sp.*,* a newcomer among the invertebrate predators of the reservoir, were followed until 2009.

3.2 Materials and Methods

Liuxie Reservoir is a monomictic, oligo-mesotrophic reservoir located at 23°45′N, 113°46′E in South China. It has a mean depth of 21 m, a maximum depth of 73 m, a volume of 325 10^6 m^3, and a surface area of 15.25 km^2. Annual average precipitation is 2,098 mm with a rainy summer from April to September (flooding season) and a dry winter from October to March (dry season). The reservoir is primarily fed by the Lutian and Yuxi rivers. Water retention time averages 125 days. Fifteen commercial fish species have been reported from the reservoir, most of them Cyprinidae. Bighead carp (*H. nobilis*) and silver carp (*H. molitrix*) have been stocked via periodic releases of YOY fish since the late 1970s. During 1977–1988, annual fish catch was low and varied between 9.5 and 26.4 kg ha^{-1}, with bighead and silver carp accounting for more than 90% of the catch. In more recent years and until 2004, about 300,000 YOY bighead carp and silver carp with a body length of 15 cm were released into the reservoir each August, with the ratio of the two carps 10:1. Fish of age-3 and older are caught by seining from April to October every year. The total catch was about 21 kg ha^{-1} $year^{-1}$, of which bighead carp accounted for about 85%. In 2004, further introductions were almost reduced to zero, and the pelagic fish biomass was largely removed in a single major fishing effort (Fig. 3.1).

Zooplankton was sampled bimonthly or monthly from April 2001 to December 2009 at a fixed station near the dam. Regardless of whether the water column was thermally stratified or not, dissolved oxygen often declined dramatically to below 3 mg L^{-1} near a depth of 20 m (Figs. 3.2 and 3.3). As this oxic condition limits *D. galeata*'s distribution, we only sampled the upper 20 m. Quantitative samples were collected with a 5-L sampler from the surface to 20 m at 1-m intervals. The samples were filtered through 64 μm mesh and zooplankton was preserved in 4% buffered formalin. Qualitative samples were obtained by vertical and horizontal net hauls, using a conical plankton net of 113 μm mesh. In each sample, all individuals of *Daphnia* were counted and measured at 40× magnification under a dissecting microscope.

Phytoplankton samples were collected from the surface using a 1-L sampler and preserved with 4% buffered formalin. The samples were then concentrated in the laboratory. Phytoplankton was counted under 400× magnification, and biovolume was calculated according to appropriate geometric models after microscopic measurement of at least 100 individuals per taxon. Specimens were identified to genus or species level. To evaluate food availability for *Daphnia*, the phytoplankton was subdivided into edible nanophytoplankton (<20 μm in length,

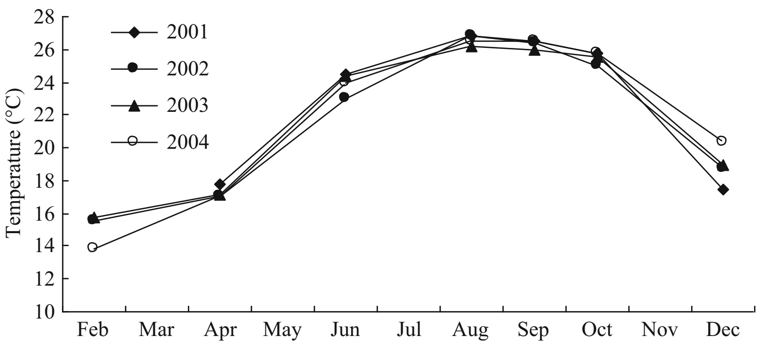

Fig. 3.2 Seasonal change in average temperature of the water column from 2001 to 2004

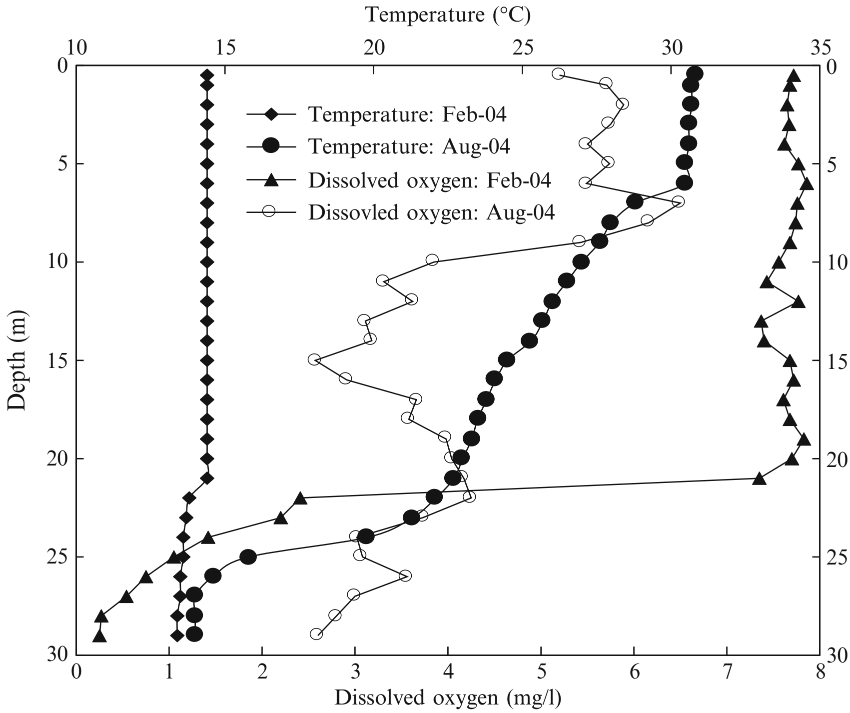

Fig. 3.3 Vertical profiles of temperature and dissolved oxygen in late winter and summer 2004

without spines or protective covering) and inedible forms (Sterner 1989). Temperature and dissolved oxygen profiles were taken at 1-m interval with a YSI Model 85 instrument.

In the laboratory, live observations of *Daphnia, Ceriodaphnia,* and the flatworm *Rhynchomesostoma* were performed, and cultures of the latter were maintained in liter bottles.

3.3 Results

3.3.1 Water Temperature and Dissolved Oxygen Profiles

The reservoir is monomictic with the water column thermally stratified most of the year. Surface water temperature ranged from 14°C to 32°C, with minima in February and maxima in August. The average temperature of the upper 20-m water column ranged from 13.8°C to 26.8°C (Fig. 3.2). Only in mid- and late-winter did the water column thermally destratify. Regardless of whether the water column was thermally stratified or not, dissolved oxygen often stratified and declined to ~3 mg L^{-1} near a depth of 20 m in the reservoir (Fig. 3.3).

3.3.2 Phytoplankton

Phytoplankton biomass varied from 0.76 to 380 mg m^{-3} (Fig. 3.4), and was composed of Bacillariophyceae, Dinophyceae and Chrysophyceae in winter, and Bacillariophyceae, Dinophyceae, and Chlorophyceae in other seasons. Edible nanophytoplankton biomass varied from 0.15 to 100 mg m^{-3} (Fig. 3.5), and accounted for about 11–81% of the total phytoplankton biomass. Each year, relatively high edible biomass was observed in autumn (Fig. 3.5). *Cyclotella* spp. dominated the edible nanophytoplankton and, together with *Peridinium pusillum*, *Scenedesmus* spp., *Chlamydomonas* spp., and *Cryptomonas* sp., was responsible for the majority of edible biomass.

Filamentous Cyanobacteria are not only inedible but also interfere with *Daphnia's* feeding and cause its population to decline. In Liuxihe Reservoir, cyanophycean biomass varied from 0.01 to 44.4 mg m^{-3}, accounting for 0.13–11.6% of the total. Maximum cyanophycean biomass (44.4 mg m^{-3}) was found in September 2003; at other times, biomass was below 2.1 mg m^{-3}. *Microcystis* spp. and *Gloeocapsa* sp. were the dominant species. Filamentous species contributed less than 1% to total cyanophycean biomass.

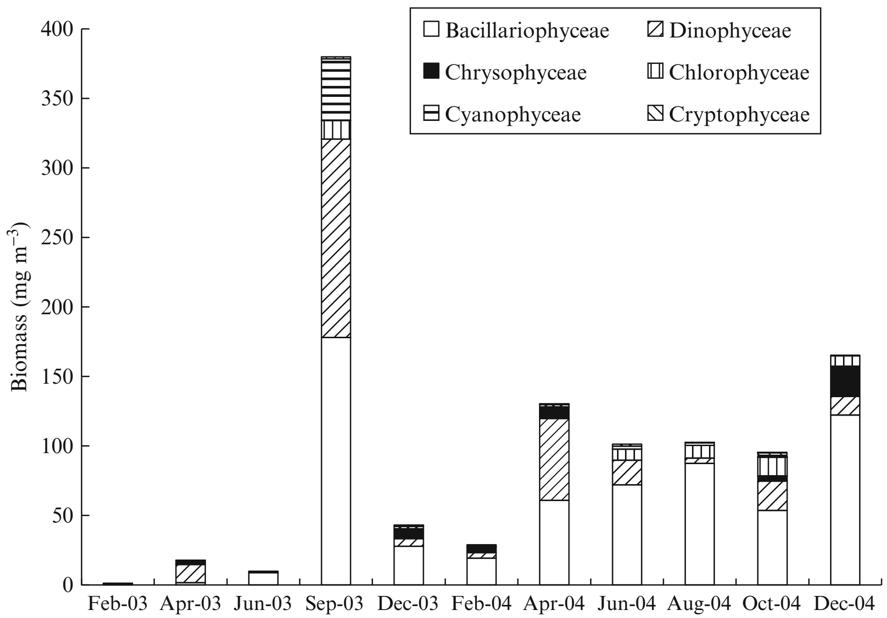

Fig. 3.4 Biomass composition of phytoplankton in 2003 and 2004

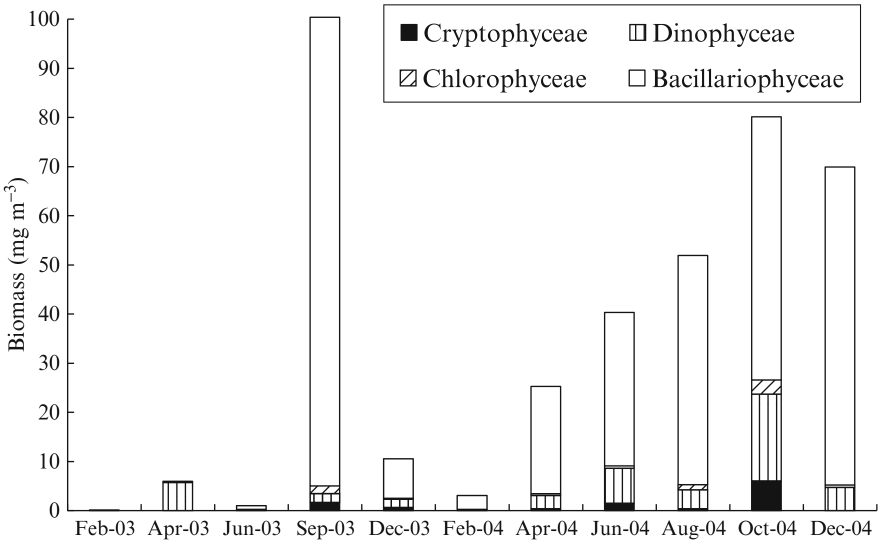

Fig. 3.5 Biomass and its composition in edible phytoplankton in 2003 and 2004

3.3.3 Dynamics of D. galeata

Only one species, *D. galeata,* was observed in the open water of Liuxihe Reservoir. Its population density was low, from 0 to 178 individuals m^{-3} in 2001, 0–530 individuals m^{-3} in 2002, 0–333 individuals m^{-3} in 2003, and 0–700 individuals m^{-3} in 2004 (Fig. 3.6b). Body length varied from 0.43 to 1.30 mm, with an average of 0.70 $\pm$ 0.02 mm, which is less than half the size observed in the temperate zone (Benzie 2005). Moreover, individuals with body length >1 mm were rare. Body length varied significantly with time (ANOVA; $P < 0.001$) but showed no seasonal pattern (Fig. 3.7). The dynamics of *D. galeata* density exhibited a similar pattern over the first 4 years of study (Fig. 3.6b). After disappearing from the water column in autumn, the population reappeared in early winter, peaked in late winter (February), declined toward summer, and disappeared in August. Starting in 2005, Daphnia tended to reach higher maximum abundances, culminating in 2007 when it reached ca 1,500 individual m^{-3}, but it still disappeared in summer, and on average even 1 month earlier (July) than before 2005.

During 2001–2004, we found a strong negative correlation between water temperature and *D. galeata* population density (Fig. 3.8). Secchi disk depth (from 1 to 4 m, with relatively high transparency in autumn) also negatively correlated with *D. galeata* (Fig. 3.9). *D. galeata* was not significantly related with total phytoplankton biomass. It also related negatively with total edible nanophytoplankton (Fig. 3.10), edible Chlorophyceae ($R = -0.361$, $P < 0.05$), and edible Bacillariophyceae ($R = -0.475$, $P < 0.05$).

As of 2005, we noted the first appearance of pelagic flatworms (*Rhynchomesostoma* sp.) and of the cladoceran, *C. quadrangula* (Fig. 3.11). The flatworms

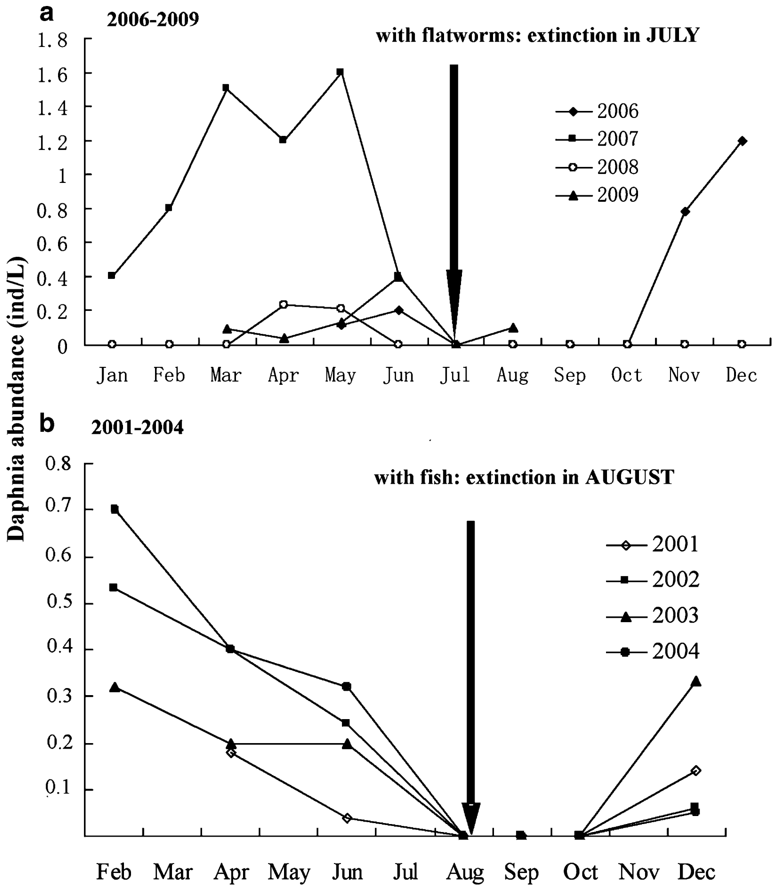

Fig. 3.6 Fluctuation of *Daphnia galeata* abundance in Liuxihe Reservoir from 2006 to 2009 (**a**) and from 2001 to 2004 (**b**) (After Wang et al. 2011, from Hydrobiologia, with permission)

usually appear in May and disappear in September–November. *Daphnia* significantly negatively correlated with the flatworms ($r = -0.3$, $P = 0.03$), which reflected the visual observation that flatworms were hunting for cladocerans as food. As all other typhloplanids that have been studied in this respect, they strike prey upon (chance) encounter, paralyzing it by injecting a neurotoxin into the prey body. *Daphnia* also correlated negatively with *Ceriodaphnia*, at the limit of significance. Cultures and laboratory experiments showed that the flatworms attacked all cladocerans, but *Ceriodaphnia* was less sensitive to toxin than *Daphnia*. Also, and unexpectedly, cultures of flatworms at densities of three to five worms per liter could only be maintained with daily water changes. Worms are thus sensitive to their own toxin, which they release to the environment, a possible form of autoregulation, potentially explaining their phenology.

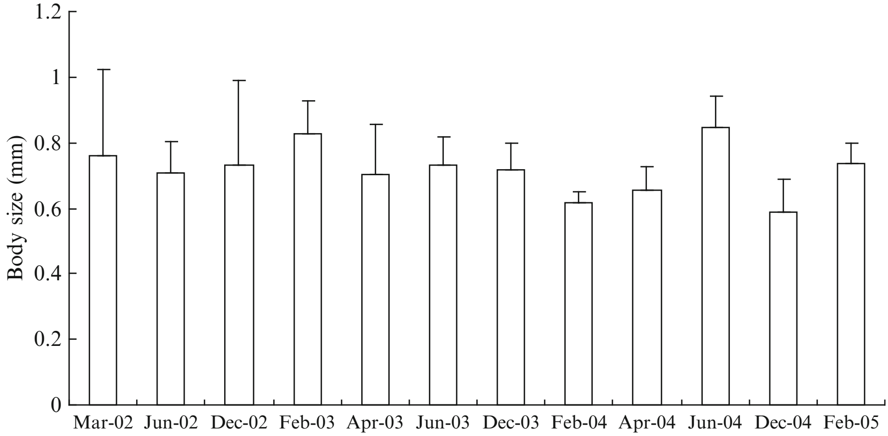

Fig. 3.7 Seasonal change in body size of *Daphnia galeata* in Liuxihe Reservoir. Vertical bars represent SD

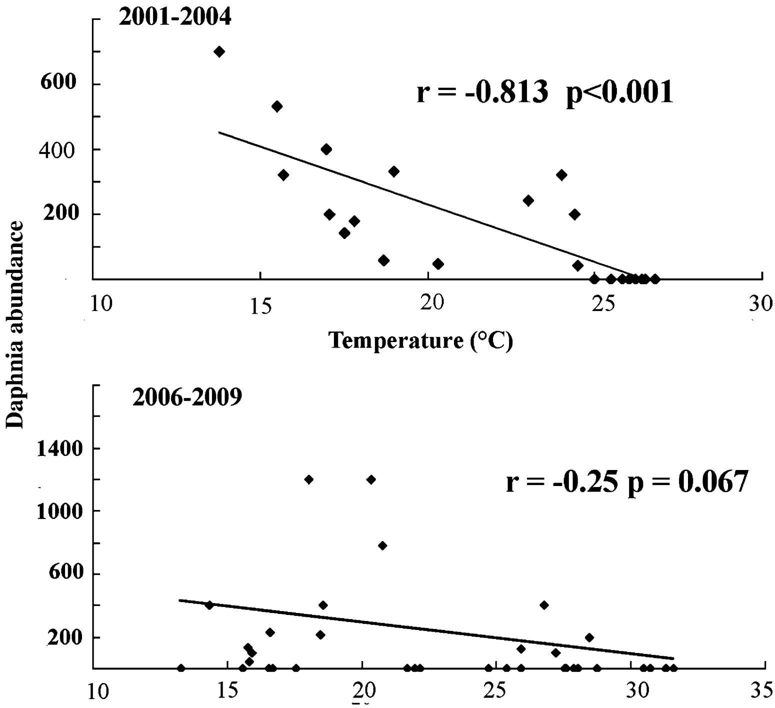

Fig. 3.8 Relationship between *Daphnia galeata* and water temperature under fish predation (2001–2004) and under flatworm predation (2006–2009) (After Wang et al. 2011, from Hydrobiologia, with permission)

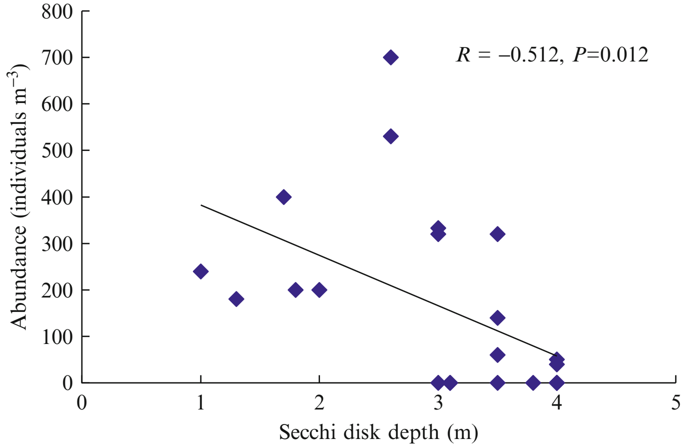

Fig. 3.9 Relationship between *Daphnia galeata* and Secchi disk depth

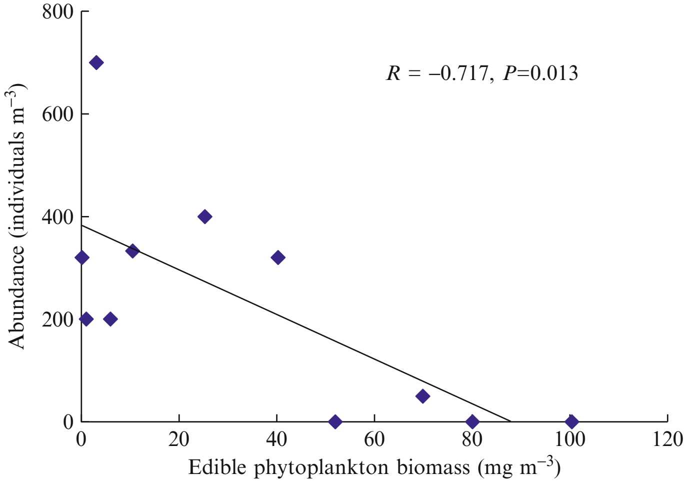

Fig. 3.10 Relationship between *Daphnia galeata* and edible phytoplankton biomass

3.4 Discussion

In this study, the density of *Daphnia* (<1 individuals L^{-1}) was substantially lower and its seasonal dynamics distinctly different from that found in temperate lakes (e.g., Černý and Bytel 1991; Cryer et al. 1986; Gliwicz 1994; Hülsmann 2003). In temperate lakes, *Daphnia* abundance frequently is of the order of tens to hundreds of individuals per liter (Vijverberg and Richter 1982), with numbers generally peaking in spring and autumn, and declining towards summer and winter (Sommer et al. 1986). In Liuxihe Reservoir, *Daphnia* peaked in winter and disappeared

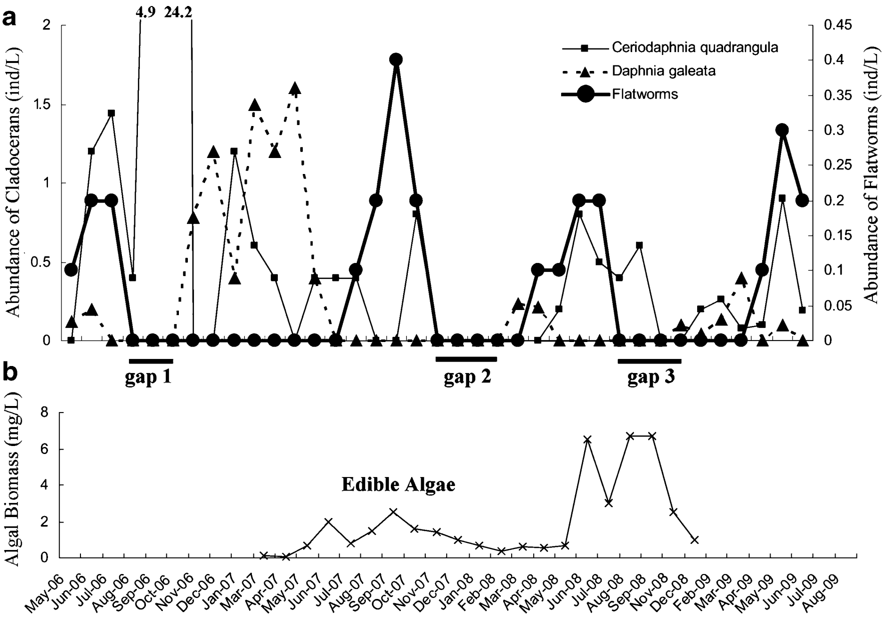

Fig. 3.11 Dynamics of *Daphnia galeata* in Liuxihe after 2005. Also shown are its predator, *Rhynchomesostoma* sp., and its competitor *Ceriodaphnia quadrangula*. Edible algal biomass (*lower panel*) had not changed during 2001–2004 (After Wang et al. 2011, from Hydrobiologia, with permission)

in summer, a seasonal pattern similar to that found in three *Daphnia* species in an African shallow lake, Lake Chad (Saint-Jean 1983). This lake is situated in a zone transitional between the subtropics and the tropics (12–14° 20′N); its water temperature shows an amplitude of 12°C, and varies between 18°C and 30°C.

Density fluctuations of *Daphnia* can be attributed to environmental as well as biotic factors: temperature, food quality, food quantity, and predatory pressure.

3.4.1 Abiotic Factors

Temperature and turbidity have been connected to seasonality of *Daphnia* in lakes (East et al. 1999; Work and Gophen 1999a, b; Havens et al. 2000; Lennon et al. 2001; Dejen et al. 2004). However, the negative correlation between *D. galeata* and high transparency in autumn suggests that turbidity was not the cause of the disappearance of *D. galeata*. At best, a higher transparency may render large *Daphnia* specimens more visible to predators, but we doubt whether this had a decisive effect because *Daphnia* is capable of avoiding clear surface water during the day. The negative correlation between temperature and abundance of *D. galeata* is more meaningful. A trade-off between higher temperatures, shorter development times, lower fecundity, and shorter life spans (early mortality) has been long known in Cladocera, and was particularly well documented in *Moina macrocopa* by Terao and Tanaka (1928). Huang (1984) reported that the related *D. hyalina* grows well within a temperature range of 15–30°C, but the optimum was at 20°C. In Liuxihe Reservoir, surface temperature may reach 32°C in summer. Just like the *Daphnia*'s of Lake Chad (Siant-Jean 1983), *D. galeata* in Liuxihe was therefore clearly living above its "comfort zone," i.e., around 25–30°C and above; its eggs and embryos develop very quickly, but adults die before they can leave much offspring. Thus, the limited temperature tolerance of *Daphnia* is likely a phenomenon that contributed to its steep summer decline, at least until 2004. As soon as the flatworms appeared, their influence became overriding, and the correlation of *Daphnia* with temperature collapsed (Fig. 3.8a).

3.4.2 Food Quantity and Quality

The abundance and structure of the phytoplankton community is another factor that affects the dynamics of *Daphnia* (DeMott 1989). High spring abundance of edible phytoplankton and low predatory pressure result in a high abundance. But this in turn causes a reduction in the density of edible phytoplankton (diatoms and chlorophytes) in summer (Lampert et al. 1986) and, by competitive release, an increase in inedible phytoplankton (mainly Cyanobacteria) (Boersma et al. 1996). These filamentous Cyanobacteria lead to reduced feeding rates and increased respiration rates in daphnids (Gliwicz and Siedlar 1980; Trabeau et al. 2004), and they decline, such that the cycle is closed. In Liuxihe Reservoir, cyanobacterial biomass accounts for less than 12% of total phytoplankton. As Cyanobacteria were predominantly colonial but not filamentous, the effect of their mechanical interference on *D. galeata* feeding can probably be disregarded. Thus, the relatively high edible nanophytoplankton biomass in late summer and autumn suggests that food limitation is not the cause of the disappearance of *D. galeata*. This high biomass was more probably a consequence of the disappearance of *Daphnia* than its cause. This is corroborated by the fact that edible nanophytoplankton biomass in winter and spring was lower than in autumn, at relatively high *D. galeata* density, suggesting an effect of *D. galeata* on nanophytoplankton. In conclusion, neither food limitation nor interference by Cyanobacteria was the likely cause of the demise of *D. galeata* in late summer and autumn.

3.4.3 Predation

The predation regime in the reservoir can be subdivided into two distinct phases: a fish-driven phase and a flatworm-driven phase, with little doubt that the flatworms themselves, being rather clumsy swimmers, well visible in the water, were held in check as long as fish were present. Bighead carp dominated fish catch and YOY fish released yearly to the reservoir perpetuated this situation. Bighead carp are pump filter feeders. Burke et al. (1986) found that the carp can exert considerable pressure on zooplankton under laboratory conditions, but just how big their predation pressure is in natural conditions is uncertain. They catch zooplankton up to 3,000 μm in size (Cremer and Smitherman 1980) but they do not snap at individual prey or orient towards zooplankters swimming in front of them. They feed on zooplankton using a filter-feeding apparatus that allows them to suck in large volumes of water during ingestion (Cremer and Smitherman 1980). Their feeding is therefore a passive, mechanical process using gill rakers that set a lower limit to the size of prey they can catch, and they are only able to seek prey and stay in an area of high prey density. Their selectivity in feeding is not based on smell and sight, but mainly on taste. Zooplankton is usually distributed in patches, and carp can probably locate and swim to favorable patches and feed more on their preferred species of zooplankton, but they do not select them individually (Dong and Li 1994). In Liuxihe Reservoir, larger specimens of *D. galeata* as well as flatworms are within the size range of bighead carp prey and predatory pressure on them may be high enough to drive them to extinction in summer (Jeppesen 1998). The size-truncated population of *Daphnia* seems to confirm that (large specimens being selectively removed), and thus summertime fish predation is certainly a burden on the already "overheated" *Daphnia*.

Bighead carp predation intensity is primarily determined by its feeding rate and population size. In Liuxihe Reservoir, as stated earlier, seasonal variation in water temperature reaches an amplitude of about 18°C. Because of the temperature dependence of fish feeding activity (Gliwicz and Pijanowska 1989), seasonal changes in feeding rate of bighead carp will be substantial. The negative correlation between temperature and *D. galeata* density indirectly measures the fact that *D. galeata* decreases with increasing bighead carp feeding rate. In winter, with water temperature ~14°C, bighead carp feeding as well as reproduction in *D. galeata* are low. Yet, mortality in *D. galeata* due to bighead carp decreases in parallel with its reproduction rate while its life span increases and results in the restoration of a *D. galeata* population in winter. In spring and summer, both predation-induced mortality rate and reproduction rate of *D. galeata* first increase with temperature, but lifetime fecundity decreases once temperature overshoots the optimum. As a result, the population density of *D. galeata* decreases.

As no appropriate spawning sites exist in Liuxihe Reservoir, eggs produced by bighead carp cannot hatch successfully. Recruitment of bighead carp via natural reproduction is impossible, and YOY fish needed to be released to the reservoir each year in August. This artificial regime caused a peak in the bighead carp population in late summer, and predatory pressure on zooplankton reached a maximum. However, a quick inspection of Fig. 3.5 shows that, at that time, the *Daphnia* population was already hanging by a straw, and the YOY probably inflicted a fatal final blow to it, causing extinction by August.

Upon the removal of bighead from the reservoir, a whole new situation was created, in which predatory release occurred, not only of large grazers like

Daphnia, but also of a variety of invertebrate predators. Among these are *Leptodora richardi,* and typhloplanid flatworms, up to 2 mm large, clumsy swimmers that hunt by stinging prey individually, but also release some paralyzing toxin to the environment continuously (Dumont and Carels 1987). These toxins break down over a period of months, such that the environment may remain toxic, even in the absence of flatworms. *D. galeata* was found to be extremely sensitive to the toxin, and although its abundance initially rose to about twice that under the fish-predation regime, its demise came about 1 month earlier: by early July, all *Daphnia* had disappeared, at a temperature that was still relatively favorable to it. Another change in the community that seems to be triggered by the disappearance of *Daphnia* is the appearance of another daphniid, *C. quadrangula*. This species, that had been present "in the background" before, now filled the void left by *Daphnia* and could apparently do so because it was much less sensitive to flatworm toxin (Wang et al. 2011). In fact, *Ceriodaphnia* is perhaps less sensitive to flatworm toxin than flatworms themselves, which were found difficult to maintain at high densities in batch cultures. Perhaps, flatworms (who produce resting eggs capable of estivation-hibernation, much in the same way as cladocerans) evaluate their own abundance by the cumulative amount of toxin released, and auto-regulate their population.

3.5 Conclusions

In Liuxihe Reservoir, *D. galeata* peaks in winter, and disappears into diapause in summer. The high surface temperature (32°C) during summer negatively affects the animal's fitness and is probably a factor that contributes to its disappearance. However, predatory pressure from bighead carp was responsible for the generally low density of *D. galeata* in the reservoir, and introduction of its YOY (in August) may have given the final blow to the active population each summer. Artificial removal of bighead carp in 2004 changed as well the predation as the competitive relations within the zooplankton. A typhloplanid flatworm, using toxins to capture its prey, appeared that eliminated *Daphnia* even quicker than fish. Its relatively persistent toxins, moreover, kept the pelagic free of *Daphnia* for up to 3 months are the disappearance of the worms themselves. *Daphnia*'s place was, however, taken by *C. quadrangula*, a previously rare daphniid, which was less sensitive to toxin, yet competitively inferior to *Daphnia* under "normal" conditions.

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Chapter 4
Functional Classification of Phytoplankton Assemblages in Reservoirs of Guangdong Province, South China

Ren Hu and Lijuan Xiao

Abstract Traditional classification of phytoplankton assemblages does not adequately reflect their ecological function in reservoir ecosystems. Therefore, we apply the concept of functional groups to classify the phytoplankton assemblages in Guangdong reservoirs, and use such groups to understand the ecological status of these water bodies. Phytoplankton associations were studied in 20 subtropical reservoirs in Guangdong Province in the wet and dry seasons. Eleven of the thirty one described phytoplankton functional groups were found in these oligotrophic to eutrophic waters. It is necessary to define a new group (Lr) for reservoirs dominated by *Microcystis* sp., *Aulacoseira granulata* and/or *Staurastum* sp., along with *Ceratium* spp. in some associations. The new group reflects the unique habitat of reservoirs compared to lakes: strong and persistent disturbance caused by a deep outlet and a relatively stagnant, eutrophic surface layer.

4.1 Introduction

Phytoplankton has been widely used as an indicator of the trophic status of lakes and reservoirs. Usually, the presence or absence and relative abundance of certain common species are adopted to describe habitat conditions. However, algal growth in natural water bodies is complicated, generally showing nonlinear responses to various environmental variables such as temperature, light intensity, and nutrients, as well as poorly understood interactions among these individual variables. Reynolds worked out a system of functional groups, based mainly on European lakes (Reynolds 2002). A functional group is a set of functionally coadapted species populating given habitats constrained by light, P, C, or N, or whatever (Padisák et al. 2009). Classifying phytoplankton into functional groups has revealed itself

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as an efficient method for identifying structure in a phytoplankton community. The system of functional groups has been developed into 31 associations found in various lakes and it describes most situations reasonably well (Reynolds et al. 2002). These associations are named according to an alphanumeric code called codon. For example, Codons A–D are used for vernal blooms, Codons E–H for associations at the start of summer stratification, and so on (Reynolds et al. 2002). Because communities can provide more information than single species, the system is reliable in describing habitat conditions of phytoplankton. However, not all valid groupings have to date been included in the system developed by Reynolds et al. (2002), and the functional significance of many algal species has not yet been assigned to functional groups, because data from many ill-described habitats are deficient.

Reservoirs, being man-made lakes, have several features distinguishing them from natural lakes, mainly following from their manual operation (Thornton et al. 1990). When rivers are dammed, the flow is temporarily halted in the impoundment and new lentic habitats are established. Reservoirs are sufficiently distinct in their basic characteristics to offer novel ways of investigating the adaptations of phytoplankton. Reservoirs often undergo rapid and extensive fluctuations in flow and water levels. The change of hydrodynamics leads to a fluctuation of nutrient loading and underwater light environment. This change can greatly influence the structure and abundance of phytoplankton.

Unlike reservoirs in the tropical zone of the southern hemisphere (Brazil and Venezuela), where phytoplankton has been studied over many years (Henry et al. 2006; Souza et al. 2008; González 2000), phytoplankton species composition in the tropics of the northern hemisphere has been less well investigated (Sterner and Grover 1998; Vrba et al. 1995; Kotut et al. 1998; Burford and Odonohue 2006). In South China, where natural lakes are scarce, a large number of reservoirs were constructed during the 1950s–1980s. The phytoplankton of this region is little known compared with that of other tropical regions. Guangdong province is located on the coast of South China Sea (from 20°14′ to 25°31′ N and from 109°40′ to 117°20′ E). Affected by the southwest monsoon and tropical storms, precipitation is high and is concentrated during a wet season that extends from April to September, contributing to 70–85% of annual rainfall. The yearly average precipitation is 1,744 mm with an annual runoff of 180×10^9 m^3, but its spatial distribution is irregular and ranges between 400 and 2,800 mm. This chapter investigates the characteristics of the algal community in the reservoirs of Guangdong Province and analyzes the possible functional classification of phytoplankton in these reservoirs (Table 4.3).

4.2 Materials and Methods

Here, we discuss 20 reservoirs representing different morphological and watershed characteristics (Fig. 4.1). Conductivity varied from 27 to 420 $s \cdot cm^{-1}$, and is related to geology and land use within the watersheds. Surface water temperature ranged from 27.0°C to 33.6°C in the flooding and 16.4–24.0°C in the dry season.

Table 4.1 Limnological characteristics of the reservoirs investigated (2000)

Lable	Name	TS	PB (mg/L)	Chl.a(mg/L)	VR (*10^6 m^3)	WT(day)	CT(year)
1	Xiaokeng	M	2.409	1.937	113.16	<365	1964
2	Chishijing	M	2.535	1.762	12.4	<365	1958
3	Heshui	M	3.335	2.585	30.4	39	1957
4	Liuxihe	OM	1.45	1.0975	326	172	1958
5	Shatian	ME	2.278	1.8405	14.2	<365	1960
6	Hedi	E	19.2	9.859	795	123	1959
7	Dashuiqiao	ME	12.12	8.2575	100.7	343	1958
8	Baipengzhu	OM	2.3	0.6665	575	306	1987
9	Dajingshan	M	4.53	4.175	10.5	300	1975
10	Xinfengjiang	O	1.418	1.3425	10,800	644	1958
11	Dashahe	ME	4.185	2.55	156.8	180	1959
12	Feilaixia	M	2.97	2.5425	440	14	1998
13	Gaozhou	M	2.215	1.66	841.8	161	1960
14	Qiyeshi	E	13.515	11.38	10.2	236	1960
15	Shiyan	E	19.693	18.13	16.9	169	1960
16	Chisha	M	3.561	3.015	1.1	<365	1960
17	Gongping	M	1.199	1.0255	163.3	133	1962
18	Hexi	ME	4.523	4.165	15.8	125	1958
19	Tangxi	ME	3.816	3.71	286.4	151	1959
20	Shenzhen	ME	21.385	9.405	35.2	7	1960

TS trophic state; *O* oligotrophic; *OM* oligo mesotrophic; *M* mesotrophic; *ME* meso-eutrophic; *E* eutrophic; *PB* phytoplankton biomass; *Chl.a* chlorophyll a concentration; *VR* volume of the reservoir; *WT* water retention time; *CT* construction time

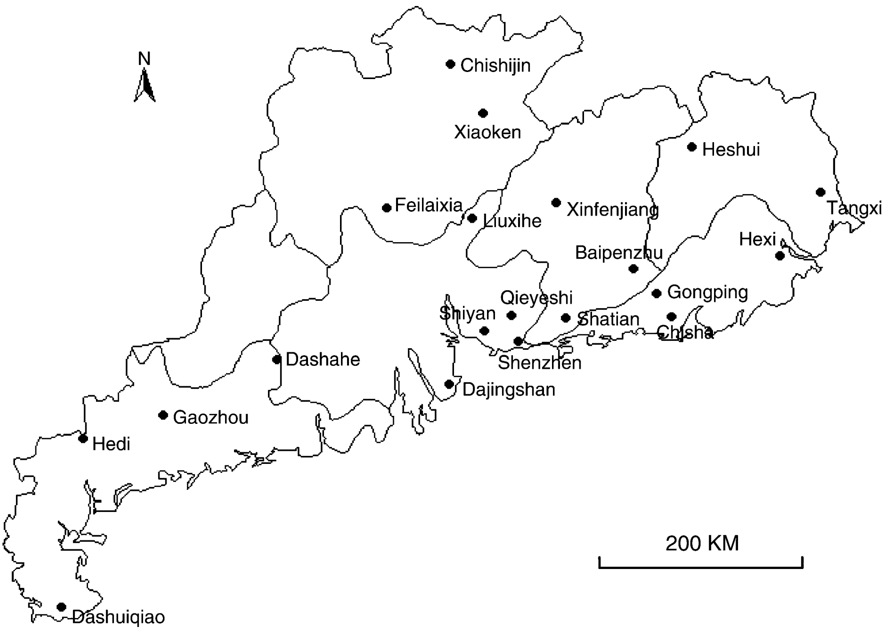

Fig. 4.1 Location of the sampling reservoirs

Samplings were conducted in the flood and dry seasons of 2000 and 2003: June to July and November to December. In most reservoirs, samples were collected both in the riverine and lacustrine zones. In two large reservoirs, Xigfengjiang and Hedi, only the lacustrine zone was sampled. One liter of water at 0.5 m below the water surface was collected and preserved with 4% formalin in the field. All samples were concentrated into 20 mL with sedimentation. All samples were examined for species identification and counted under a microscopy. Subsamples were counted using a magnification of 400× and checked thoroughly for rare species.

4.3 Results

4.3.1 Species Composition of Phytoplankton

A total of 142 species of phytoplankton belonging to 89 genera in 7 phyla was recorded. There were 84 species in Chlorophyta, 25 species in Bacillariophyta, 19 species in Cyanophyta, and 9 in Euglenophyta. The highest richness (57 species) was observed in Baipenzhu reservoir and the lowest in Hedi reservoir (Table 4.2). Chlorophyta contributed most richness in every reservoir in the dry season as well as in the flooding season.

4.3.2 Functional Classification of Phytoplankton

The phytoplankton assemblages were different between the wet and dry seasons. Although phytoplankton cell density decreased from the flood to the dry season in most reservoirs, the dominant species and functional groups did not change much (Table 4.3). In the three large oligotrophic reservoirs (Xinfengjiang, Liuxihe, and Baipenzhu), the phytoplankton in the flooding season usually belonged to Group Z, which is characterized by a dominance of picophytoplankton. Some chlorophytes belonging to the Z group in Baipenzhu reservoir were abundant chlorococcaceans, which are small cell-sized and efficient in nutrient-poor waters. In Liuxihe reservoir, however, phytoplankton was dominated by pico-Cyanobacteria such as *Gloeocapsa* and *Aphanocapsa*. Group Z refers to the picophytoplankton that thrives in the photic layers of oligotrophic water, in the upper mixed layer. Being the largest water body in Guangdong province (13.9×10^9 m^3 in storage capacity), Xinfengjiang reservoir is oligotrophic and has a retention time of about 600 days. The dominant species were composed of chroococcoid Cyanobacteria such as *Synechococcus* and *Synechocystis* spp. and tiny chlorophytes like *Chloromonas* and the small-celled *Chlorella minutissima*. In Baipenzhu and Xinfengjiang reservoirs, meso-desmids occurred with picophytoplankton dominant in the summer flooding season. Desmids are able to store nutrients in their cells to survive in

Table 4.2 Distribution of phytoplankton species in the reservoirs sampled

	Cyanophyta	Chlorophyta	Bacillariophyta	Chrysophyta	Pyrrophyta	Euglenophyta	Cryptophyta	Total
Xiaokeng	4	10	8	1	2	0	0	25
Chishijing	7	13	7	2	2	1	0	32
Heshui	8	16	11	0	2	3	1	41
Liuxihe	9	17	10	1	2	3	0	42
Shatian	5	31	6	2	2	2	1	49
Hedi	6	7	5	1	2	2	0	23
Dashuiqiao	8	17	10	0	2	4	1	42
Baipenzhu	7	37	7	2	2	1	1	57
Dajingshan	4	14	4	2	2	1	1	28
Xinfengjiang	7	21	6	2	2	1	0	39
Dashahe	10	28	11	1	2	2	1	55
Feilaixia	7	14	12	1	2	2	0	38
Gaozhou	5	13	5	1	2	1	1	28
Qiyeshi	10	21	9	0	2	6	1	49
Shiyan	9	23	8	0	1	6	1	48
Chisha	10	24	10	1	2	3	1	51
Gongping	9	27	8	2	2	5	1	54
Hexi	7	21	4	0	2	3	0	37
Tangxi	6	25	10	2	2	1	1	47
Total	19	84	25	2	2	9	1	142

Chroococcus sp., *Chlamydomonas* sp., *Navicula* sp., *Cyclotella meneghiniana, Aulacoseira* sp., and *Ceratium hirundinella* were the species observed in all samples. Other frequently observed taxa were *Microcysis flos-aquae*, *Gloeocapsa magna*, *Dactylococcopsis acicularis*, *Lyngbya limnetica*, *Phormidium* sp., *Merismopedia elegans*, *Pediastrum* sp., *Staurastrum* sp., *Scenedesmus* sp., *Cosmarium* sp., *Selenastrum* sp., *Pinnularia* sp., *Rhizosolenia longiseta*, *Peridinium bipes*, *Euglena viridis*, *Trachelomonas*, *Dinobryon divergens*, and *Cryptomonas* sp.

Table 4.3 Functional groups of phytoplankton species in 20 reservoirs

Types	Reservoirs	Codon	Character	Representative algae	
				Flooding season	Dry season
Oligo- to mesotrophic large reservoir	XFJ	Z	Clear, low phosphorus, low biomass	Large desmids, Picophytoplankton	Large dinoflagellate, Picophytoplankton
	LXH	Z/P	Long water retention time	Small *chroococcales* (*Gloeocapsa, Aphanocapsa*), *Cyclotella*	*Aulacoseira granulata* *Staurastrum*
	BPZ	Z/A		Picophytoplankton, Large desmids	*Cyclotella, Rhizosolenia*
Mesotrophic reservoir	GZ	Lr	Warm, nutrient-rich, meso-eutrophic reservoir	*Microcystis* *Staurastrum* *Cyclotella*	*Microcystis* *A. granulata* *Staurastrum*
	CSJ	Lo/F	Clear epilimnion, low nutrient, high turbidity	*Ceratium hirundinella*	*Botryococcus, Oocystis lacustris*
	DJS	S1	Shallow, turbid mixed layers, light deficient	*Pseudanabaena* *C. hirundinella,* *Chlamydomonas*	*Pseudanabaena* *Cyclotella*
	CS	Lr	Warm, nutrient-rich meso-eutrophic reservoir	*Microcystis,* *A. granulata* *Ceratium*	*Microcystis* *A. granulata* *Staurastrum*
	HS	Lr	Warm, nutrient-rich meso-eutrophic reservoir	Large colonial Volvocales (favorite medium is the nitrogen-rich water)	*Microcystis,* *A. granulata,* *Staurastrum*
Meso- to eutrophic reservoirs	TX	Lr	Warm, nutrient ric04214meso-eutrophic reservoir	*Microcystis* *A. granulata* *Staurastrum*	*A. granulata* *Microcystis* *Eudorina*
	HX	Lr/P	Warm, nutrient-rich meso-eutrophic reservoir	*Staurastrum,* *Microcystis*	*Cosmarium bioculatum* *A. granulata*
	DSH	Lr	Warm, nutrient-rich meso-eutrophic reservoir	*Microcystis,* *A. granulata*	*Ceratium* *A. granulata* *Microcystis*

	DSQ	S1	Low light, turbid mixed layers	*A. granulate*, *Pseudanabaena*	*A. granulate* *Pseudanabaena*
	ST	Lo/Lr	Warm, nutrient-rich meso-eutrophic reservoir	*Planktosphaeria* *Staurastrum* *Ceratium*	*Microcystis*, *A. granulata*
Eutrophic reservoir	QYS	S1/Sn	Shallow eutrophic lakes, Warm mixed layers, light deficient, rich P	*Pseudanabaena*, *Chlamydomonas*	*Cylindrospermopsis*
	SY	P/W_2		*A. granulata* *A. italica*	*A. granulata* *Euglenoids*
	HD	M	Warm, nutrient-rich eutrophic reservoir	*Microcystis* bloom	*Microcystis* bloom
Acid reservoir,	GP	Lr/E	High nutrient in summer, low in winter	*Microcystis* *Aulacoseira* sp.	*Dinobryon* *Aulacoseira* sp.
Through flowing reservoirs	FLX	S1/T	Low light, turbid mixed layers in summer, persistently mixed layer in winter	Filamentous blue-green algae: *Pseudanabaena*	Filamentous green algae:*Mougeotia*, *Planctonema*
	SZ	S1/T		Filamentous Cyanobacteria: *Pseudanabaena*, *planktothrix*	Small diatoms: *Navicula*, *Nitzschia*

low-nutrient environments. These two larger reservoirs had much lower nutrient concentration compared with Liuxihe reservoir.

Large dinoflagellates and diatoms sometimes dominated in the dry season in the three large oligotrophic reservoirs. In dry season, a high density of *Aulacoseira* was observed in the upstream zones of Liuxihe reservoir, reflecting the functional Group P. This group is originally present in shallow lakes with a continuous or semicontinuous mixed layer 2–3 m in thickness, and it represents the epilimnia of stratified water with shallow mixed layer (Voros and Padisak 1991). In autumn, turnover raises the nutrients to the surface layer and supports a dominance of diatoms. Cool-water desmids (*Staurastum gracile*) are also representative of this group. Group A is represented by winter phytoplankton composition in Baipenzhu reservoir, where *Merismopedia* was present. This group is characterized by a number of centric diatoms of the genera *Cyclotella* and *Rhizosolenia* that are especially prominent in the plankton of many medium-to-large waters, typically clear and deficient in phosphorus.

The most frequently observed functional group in the reservoirs, Lr, refers to the association of stratified mesotrophic to eutrophic reservoirs with well-mixed epilimnia, typically including *Microcystis*, *Aulacoseira*, and *Staurastrum* as their dominant species. The phytoplankton of Shatian, Heshui, Dashuiqiao, Chisha, Tangxi, Hexi, and Gaozhou reservoirs were classified to this group. *Microcystis* grows very fast in warm mesotrophic reservoirs. Many species of *Aulacoseira* are tolerant of frequent disturbance of the mixing layer. *Aulacoseira granulata*, *A. ambigua*, and *A. varians* are typical diatom species of low-latitude tropical reservoirs. They dominate in both flood and dry seasons. One important adaptive characteristic of these species is their high photosynthetic capacity, with rapid development of chlorophyll and accessory pigments per cellular unit (Reynolds 2006). *Staurastrum* is adapted to fluctuating and disturbing conditions. Other large, motile, colony-forming blue-green algae, *Woronichinia* and *Merismopedia,* also exist within the confines of Lr.

In the flooding season, Shiyan reservoir was dominated exclusively by *A. granulata*, which was the main species of Group P. This group reflects warm mixed layers with a light deficient environment. The W_2 is the group present in shallow eutrophic lakes contaminated by organic matter. *Euglenoids* and *Trachelomonas* in this group inhabited Shiyan reservoir in the dry season.

A high density of filamentous blue-green algae, especially *Pseudanabaena* sp., was observed in Dashuiqiao and Dajingshan reservoirs. These algae (Group S1) favor turbid mixed layers and benefit from higher nutrient loading from the rivers and from the longer water retention times caused by the low outflow from these two reservoirs. Dashuiqiao reservoir was also dominated by *A. granulata*; this means the reservoir should have higher nutrient concentration and water fluctuation than Dajingshan reservoir.

Qiyeshi reservoir had a phytoplankton assemblage (*Pseudanabaena* and *Chlamydomonas*) that belongs to S1 in the flooding season, as did Dajingshan reservoir. However, the dominant species in the dry season were *Cylindrospermopsis* and *Anabaena minutissima*, which belong to Group Sn. This group is a subdivision of S1, comprising species that prefer phosphorus-rich water (114 μg/L in Qiyeshi reservoir) and are tolerant of vertical mixing.

Group F is known to be represented in the plankton of a wide spectrum of lakes. The species composition (*Botryococcus* and *Oocystis lacustris*) of this group has an elevated light threshold: they function well in clear water and are otherwise tolerant of deep mixing. Phytoplankton species in this group have a strong representation among mesotrophic reservoirs, but both aforementioned species are sensitive to nutrient enrichment. The winter phytoplankton composition in Chishijing reservoir in the northern part of Guangdong province is representative of this group. However, the summer phytoplankton composition in this reservoir was represented by one species of Lo: *Ceratium hirundinella*.

Gongping reservoir differs from other reservoirs by showing distinct associations in both summer and winter. During summer, when nutrients were high, Lm group species such as *Microcystis* and *Aulacoseira* dominated, while *Dinobryon* (Group E) replaced *Microcystis* in winter when there was an improvement in water quality.

Feilaixia and Shenzhen reservoirs are through-flowing reservoirs with deep well-mixed epilimnia. In the flooding season, species in this group are filamentous blue-green algae in Group S1, which are tolerant of light deficiency. Dominant species in these two reservoirs are planktonic filamentous cyanobacteria of the genera *Pseudanabaena* and *Planktothrix*. However, the phytoplankton assemblage in the dry season of these two reservoirs was different: the dominant species in Feilaixia reservoir were filamentouse green algae-*Mougeotia* and *Ulothrix* in Codon T (adapting in persistently mixed layers), while the dominant species in Shenzhen reservoir were pennales (diatoms) such as *Nitzschia* and *Navicula* in Codon T_B. These species prefer a short water retention time such as in Shenzhen reservoir (7 days, Table 4.1), and they like to inhabit waters that resemble rivulets.

4.4 Discussion

Planktonic organisms respond promptly to environmental changes, and they provide a more stable reflection of the environment than physical and chemical variables. According to Reynolds (2006), the limits for phytoplankton growth and accumulation of biomass are mainly constrained by available solar energy flux, carbon, phosphorus, and nitrogen. All above-mentioned factors may be strongly regulated by water movements, and the morphology and hydrology of a water body. Limnologically and hydraulically, reservoirs differ extensively from lakes but there is no difference in the principles that ultimately control the composition of their phytoplankton (Straskraba et al. 1993; Han et al. 2000). Trophic state and water flow velocity in reservoirs extensively influence the dominant species and composition of the phytoplankton community.

Many studies have been done that apply functional groups to phytoplankton composition (Devercelli 2009; Salmaso and Padisák 2007; Padisák et al. 2003;

Weithoff 2003), and revisions of the functional groups have been discussed by Padisák et al. (2009). In this study, we defined a new codon Lr (r means reservoir) in the phytoplankton functional groups after having observed the phytoplankton composition in Guangdong province. This codon was composed of *Microcystis* sp., *A. granulata*, and/or *Staurastum* sp., along with *Ceratium* spp. in some associations. The habitats include meso- to eutrophic reservoirs with frequent hydrological disturbances (Komarkova and Hejzlar 1996). *A. granulata* and/or *Staurastum* sp. are both typical species of Codon P in a habitat that is eutrophic and has well-mixed epilimnia. However, *Microcystis* sp. alone belongs to Codon M, and *Microcystis* sp. together with *Ceratium* spp. belong to Lm, both of which represent eutrophic to hypertrophic medium-sized water bodies. Many meso- to eutrophic reservoirs in Guangdong province (such as Gaozhou, Chisha, Heshui, Tangxi, Hexi, Dashahe, Shatian, and Hedi reservoirs), provided we ignore one of the dominant species (*A. granulata* and/or *Staurastum* sp.), could be easily assigned to Codon M or Lm. On the other hand, if another dominant species (*Microcystis* sp.) were eliminated, they could all belong to Codon P. It seems that the co-occurrence of *Microcystis* sp., *A. granulata*, and/or *Staurastum* sp. constitutes a very common phytoplankton group in our reservoirs.

Reservoirs are distinct habitats for phytoplankton compared to lakes. Strong and persistent disturbance caused by outflow of power generation and drinking water supply results in intermittent medium vertical mixing, which favors growth of *A. granulata* and/or *Staurastum* sp. At the same time, the surface layer was relatively stagnant because of the deep outlets, which allow the dominance of *Microcystis* sp. and/or *Ceratium* spp. in the surface layer when nutrients are rich. Thus, these meso- and eutrophic reservoirs become unique habitats, which support a Codon Lr new for phytoplankton functional composition.

The most frequently observed group Lr could occur during both flood and dry seasons because the abrupt difference of water temperature in the reservoirs is not as high as that in temperate lakes. The dominance of *Microcystis* is thought to be an outcome of reservoir eutrophication. As reservoirs usually have stronger fluctuations than lakes, *A. granulata* finds its favorite ecological environment in most reservoirs of Guangdong province, especially during the vertical mixing in autumn when both silicon and phosphorus are rich. Abundant desmids (*Staurastrum* spp. belongs to Lm) were found in most reservoirs during the flooding season. The large-celled desmids in these reservoirs coincided with a high concentration of NH_3-N that entered from the watershed in the flooding season.

Another common functional group in Guangdong province is Group S. *Pseudanabaea* sp. is a typical species in this group and is adapted to highly eutrophic environments with slow water flow, responding to light deficiency water. The species has an ability to utilize green light and survive under low underwater light conditions. It also tolerates frequent flushing because it extends over the whole water column, and reproduces fast after the flushing has passed.

In well-stratified reservoirs (e.g., Xinfengjiang reservoir), the active planktonic primary production leads to further differentiation of the habitats. This results in the upper waters, although well isolated, becoming severely deficient in phosphorus or nitrogen, whereas beneath the upper layer is water in which nutrients are less depleted. Thus, only low-nutrient-tolerant species (Group Z) survive in the reservoir. The most adaptive algae are large dinoflagellates, which have efficient motility and large size, just as in dry season of Xinfengjiang reservoir (Group Z), and they can actively and vertically migrate between the two separating water layers.

Another two reservoirs which also have *A. granulata* belonging to Codon P as their dominant species are Dashuiqiao reservoir (Codon S1) and Shiyan reservoir (W_2) in the dry season (winter). We modified the habitat description of the two codons with information from our reservoir environments rather than assigning new codons to these two reservoirs.

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Chapter 5
Dynamics of Phytoplankton Community in Relation to Environmental Factors in a Tropical Pumped-Water Storage Reservoir

Qiuhua Li and Ren Hu

Abstract Phytoplankton and environmental variables were measured every 2 months in 2005 in Dajingshan, a subtropical pumped-water reservoir in South China. Phytoplankton community structure and its relationship with environmental factors including hydrological and chemical variables were explored by multivariate analysis. In total, over 100 species of algae were identified. Total abundance ranged from 0.85×10^6 to 106.27×10^6 cells/L, and total biomass from 1.0 to 16.8 mg/L (mean 6.94 mg/L). The community was dominated by *Pseudanabaena limnetica*, but in spring, *Synedra ulna* and *P. limnetica* were co-dominant. The community was relatively stable, but change rates of community structure in autumn and summer were higher than in winter and spring. Ordination by canonical correspondence analysis (CCA) divided all samples into four groups, distributed in the four districts formed by axes 1 and 2, corresponding to the four seasons: winter, spring, summer and autumn. Most samples were located in the summer and autumn districts. Fifty-four main species of phytoplankton were selected for CCA. Cyanophyta (Cyanobacteria), Bacillariophyta and Euglenophyta were restricted to the districts at the left of axis 1, but most Chlorophyta, Chrysophyta and Cryptophyta were to the right. CCA revealed that temperature and precipitation were important in driving dynamics of species composition and phytoplankton abundance.

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5.1 Introduction

Phytoplankton species composition changes with varying environmental factors, and some groups of phytoplankton have been recognized as indicators of trophic level and environmental quality of aquatic systems (Reynolds 1998; Naselli-Flores and Barone 1998, 2000; Negro et al. 2000). The phytoplankton in an individual water body is regulated by physical, chemical and biological factors (Reynolds 1998; Horn 2003). Nutrient availability, temperature, light and grazing are recognized as the most important variables determining phytoplankton distribution and species composition (González 2000; Naselli-Flores 2000). Seasonal phytoplankton growth is a response to the seasonality of some environmental variables (Tryfon and Moustaka-Gouni 1997). Water temperature largely regulates the seasonal dynamics of phytoplankton biomass and species composition (Salmaso 2000). Climate factors, especially precipitation, can also explain seasonal variations (Figueredo and Giani 2001; Ahn 2002), particularly in monsoonal regions, as they link closely with hydrodynamics (Beyruth 2000; Rennella and QuirÓs 2006). The study of the dynamics of phytoplankton is well developed in natural lakes, while phytoplankton dynamics of man-made reservoirs in the tropics and subtropics have been less well studied (Kotou et al. 1998; Piet and Vijverberg 1999). The dynamics of phytoplankton have been found to be complex and diverse in reservoirs because of human manipulation of hydrodynamics through outlets (Negro et al. 2000).

This study describes the phytoplankton composition, temporal changes, diversity, change rate of community, abundance and biomass of Dajingshan Reservoir. This reservoir is special as it has a very small catchment and stores water pumped from a nearby river. We attempt to gain insight into the factors influencing phytoplankton composition and seasonality in tropical pumped storage reservoirs as a useful baseline for the management of such reservoirs.

5.2 Material and Methods

5.2.1 Site Description

Dajingshan Reservoir, located in the northern tropics, is of great importance for the drinking water supply of Zhuhai city and Macao, southern China. It has a drainage area of 5.95 km^2, a volume of 12.10×10^6 m^3 and a mean volume of water supply of 10×10^6 m^3. Because there is no river feeding the reservoir, its storage water comes from precipitation and water pumped from several rivers. Most precipitation occurs during the summer monsoon (from early May to late September) and the reservoir has a yearly mean precipitation of 1,991 mm. The water level in the reservoir fluctuates seasonally, with the lowest water level occurring early summer and the highest in early winter. The water transparency is low at 0.6–1.1 m. Four sampling sites were selected. Site 1(S1, N 22° 18′ 03.0″, E113° 32′ 28.4″)

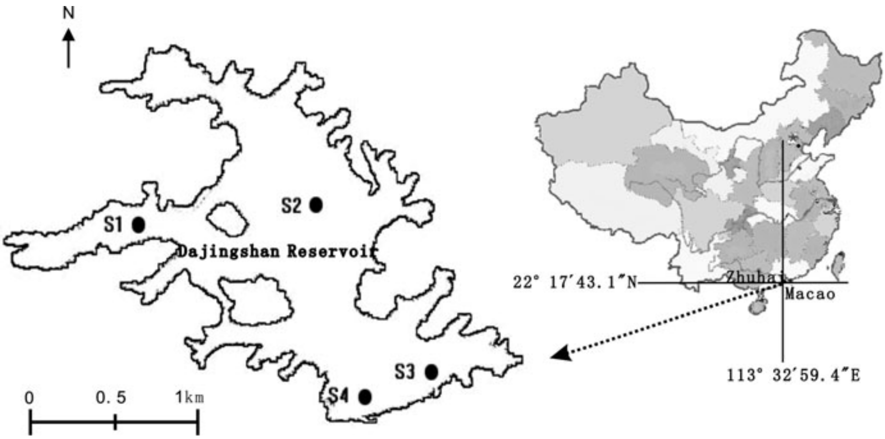

Fig. 5.1 Location and sampling stations of Dajingshan Reservoir

was near the inflow of another reservoir, S2 (N22°18′08.5″, E113°32′49.6″) at the reservoir centre, S3 (N22°17′42.5″, E113°33′04.4″) was at the outflow and S4 (N22°17′43.1″, E113°32′59.4″) was near a small inflow (Fig. 5.1).

5.2.2 Materials and Methods

Samples were collected fortnightly from the surface (0.5 m) at S1–S4 from January to December 2005. Phytoplankton was fixed with formalin 5%. Phytoplankton was identified and counted under the microscope. The cells, colonies and filaments were enumerated, to at least 300 specimens of the combined species (Holz et al. 1997). Their specific biovolume was estimated from the product of the population and mean unit volume of each species. Wet weight biomass was calculated from abundance and specific biovolume estimates, which were based on a geometric approximation, assuming a specific density of phytoplankton cells of 1 g·cm^{-3} (Kamenir et al. 2004). Species diversity was calculated using Shannon's index. The diversity of the assemblage collected in individual samplings was tested by the Shannon Weaver function: $H' = -\sum (n_i/N)\log_2(n_i/N)$ (Reynolds 1998). The rate of community composition changes (δ) was calculated according to (Huszar and Reynolds 1997; Huszar et al. 1998): if $b_i(t)$ is the abundance of the ith species and $B(t)$ is the sum of the individuals making up the sampled community, the rate of change in composition between two given dates t_1 and t_2 is solved from: $\sigma = \sum i\left[\frac{b_i(t_1)}{B(t_1)} - \frac{b_i(t_2)}{B(t_2)}\right]/(t_2 - t_1)$

Water quality variables measured onsite included water temperature (WT), Secchi disk depth (SD), and pH using an 85-YSI Multiparameter Water Quality Monitor and a 20 cm diameter Secchi disk. Data on precipitation (Prec) and water level (WL) were obtained from Dajingshan Reservoir managers. About

400–1,000 mL water was obtained for Chl-a by filtering on a Whatman GF/A filter, and its concentration was determined within 8 h after its extraction in 90% acetone. In the laboratory, the water samples were further analyzed for total phosphorus (TP) and total nitrogen (TN) using potassium persulfate digestion. Ammonia nitrogen (NH_4–N), nitrate nitrogen (NO_3–N), nitrite nitrogen (NO_2–N) and orthophosphate (PO_4–P) were determined colorimetrically (Chinese standard methods of water quality analysis, GB3838-2002).

Canonical correspondence analysis (CCA) was used to examine the relationship between phytoplankton and the environment. CCA provided a direct display of the locations of species along environmental gradients reflected in phytoplankton composition. The abundance data for each species were transformed using a $\log_{10}(X+1)$ function to obtain a normal distribution. Only the 54 species that comprised 98.3% of cumulative abundance were included in the CCA and cluster analysis.

5.3 Results

5.3.1 Environmental Factors

Temperature of the surface water was high, ranging from 15.0°C in spring to 32.0°C in summer. From April to June, the temperature difference between the epilimnion and hypolimnion was about 3°C, but the temperature was evenly distributed vertically in the other months (Fig. 5.2a). The total precipitation was 2,014 mm during the study period, 83% of the precipitation occurred from May to September (Fig. 5.2b).

Nitrogen and phosphate concentrations varied seasonally, with low levels in summer (July to September) and high levels in March and November. There was no significant difference between the four sites ($p > 0.05$). Total nitrogen was from 0.18 to 2.05 mg/L (Fig. 5.3a). Ammonia nitrogen, nitrate nitrogen and nitrite

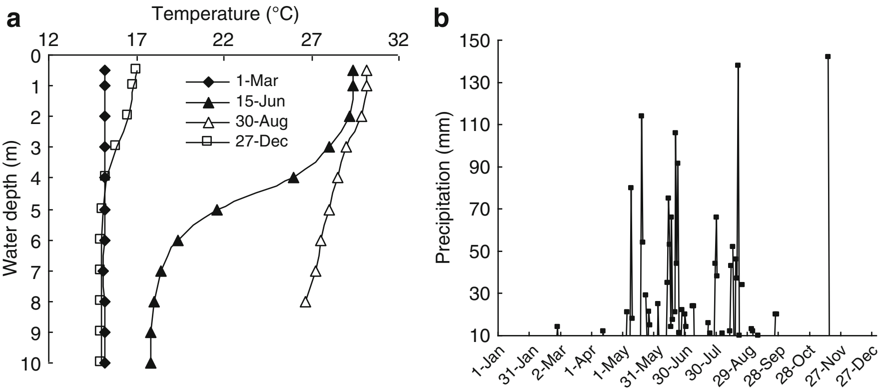

Fig. 5.2 (**a**) Temperature; (**b**) precipitation in Dajingshan reservoir in 2005

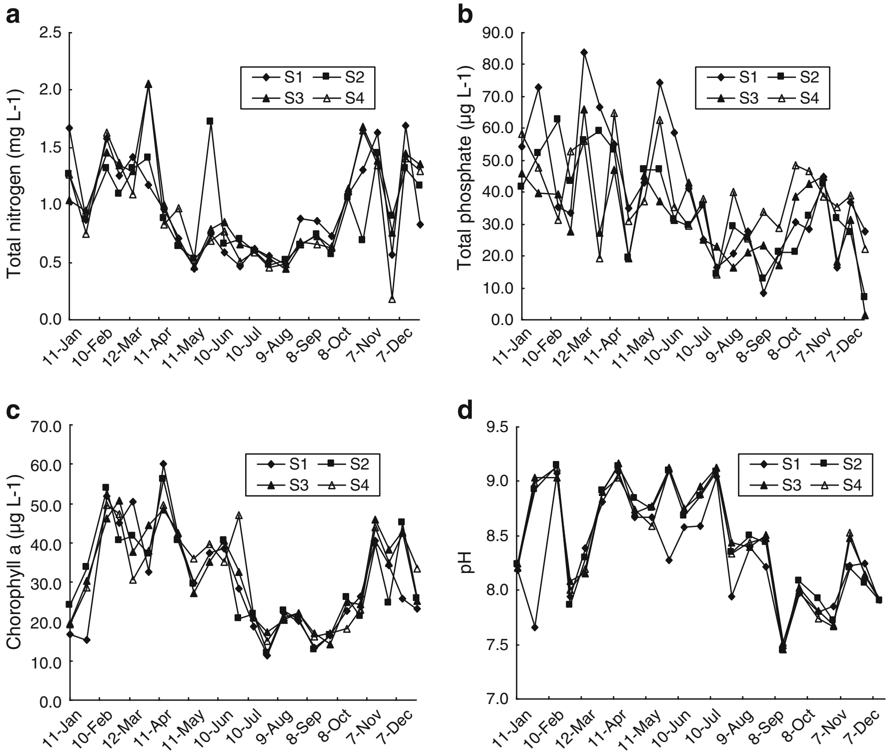

Fig. 5.3 (**a**) Total nitrogen, (**b**) total phosphate, (**c**) chlorophyll *a* and (**d**) pH in Dajingshan Reservoir in 2005

nitrogen concentrations were very low, being 0.1–170, 0.1–745 and 0.1–7 μg/L, respectively. The lowest total phosphorus concentration was below 10 μg/L, and the highest was over 83 μg/L^{-1}. Dissolved phosphorus concentration was between 5 and 23 μg/L. The atomic ratio of dissolved nitrogen to dissolved phosphorus varied from 22 on 15 March to 375 on 27 December. Phosphorus was the limiting factor for phytoplankton growth in Dajingshan Reservoir. Chlorophyll *a* concentration, ranging from 11.4 to 60.0 μg/L (Fig. 5.3c), fluctuated conspicuously. It remained low throughout the summer and was lowest in July. The peak value was 60 μg/L in April. Variation in pH was narrow, having little difference between the four sampling sites (Fig. 5.3d).

5.3.2 Phytoplankton Community Structure

One hundred taxa of phytoplankton were identified from 192 samples. Cyanophyta (often called Cyanobacteria), Chlorophyta and Bacillariophyta were the most important groups in terms of number of species. Cyanophyta had 19 taxa, Chlorophyta had 49 and Bacillariophyta had 23, but Euglenophyta had 4, both Pyrrophyta and Chrysophyta had 2 taxa only, while Cryptophyta had only 1. About 50 phytoplankton species was recorded in each sample, but in September the richness was up to 67 species. The species number of Cyanophyta was relatively steady (Fig. 5.4a) throughout the year.

Phytoplankton abundance ranged from 0.85×10^6 to 106.27×10^6 cells/L. Abundance of individual phytoplankton taxa varied greatly. There were three peaks in abundance on 17 February, 30 April, and 29 November. Between the peaks (April and November) and the low (January and August) periods, there was a distinct seasonal pattern (Fig. 5.4b). The phytoplankton consisted of Cyanophyta, Chlorophyta, Bacillariophyta, Pyrrophyta, Chrysophyta, Euglenophyta and Cryptophyta, but Cyanophyta was always dominant. At S4 for example, the percentage of phytoplankton composition was relatively steady after June, but fluctuated from January to May. Cyanophyta could be identified in all samples. They had a high percent the whole year, from 50.47% to 99.33%, from June to December. The average percentage of cyanophyta was over 90%. The percentage of Chlorophyta was relatively steady throughout the year. The percentage of Bacillariophyta clearly fluctuated, because *Synedra ulna* was able to grow at low temperatures. In late winter and early spring, *S. ulna* and *Pseudanabaena limnetica* were the

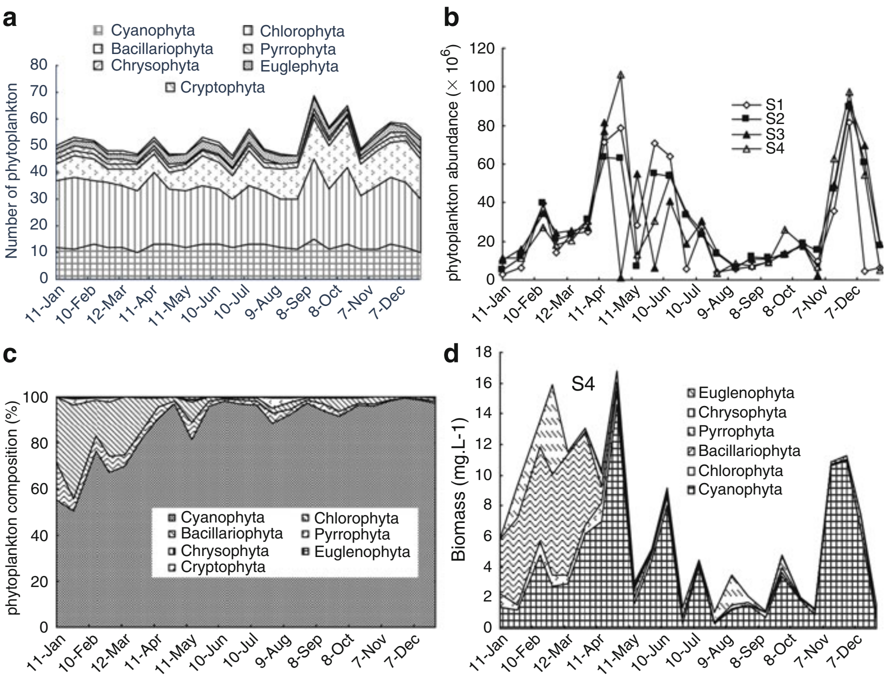

Fig. 5.4 Dynamics of phytoplankton species in Dajingshan Reservoir: (**a**) Number; (**b**) abundance; (**c**) phytoplankton composition (abundance); (**d**) biomass (S4)

dominant algal species, but in summer and autumn *P. limnetica* was the only dominant species, so the percentage of Bacillariophyta in late winter and early spring decreased. The percentage of Chlorophyta was the same as that of the Bacillariophyta, being high in spring and the low in winter. *Chlorella vulgaris* was the dominant species in Chlorophyta and it increased with temperature. A short peak of Pyrrophyta was recorded on 15 July. Species of Cryptophyta were rarely observed, and only one species, *Crytomonas* sp., was recorded with low abundance. Two species in Chrysophyta, *Dinobryon divergens* and *Mallomonas Perty,* were also rarely observed, but their percentage was up to 0.35% (Fig. 5.4c).

The phytoplankton community was characterized by a high biomass, dominated by Cyanophyta. Total biomass ranged from 1.0 to 16.8 mg/L with a mean of 6.94 mg/L. Total biomass showed a clear seasonal pattern. Cyanophyta biomass had five peaks through the year, but the peak in April was the highest, followed by the peak in December. It increased rapidly to a maximum in April; thereafter, it declined to a minimum in July and then remained a low level until November. Bacillariophyta biomass showed the greatest variation in amplitude. From January to April, it was high, but dropped rapidly to a minimum after April. Chlorophyta, Chrysophyta, Cryptophyta and Pyrrophyta were low and had the least change, but Euglenophyta biomass showed a short peak in February. In terms of contribution to total biomass, Cyanophyta contributed the highest percentage almost in all months (from 13.36% to 97.59%). The highest contribution of Bacillariophyta was 68.26% in March, Chlorophyta 29.09% in June and Pyrrophyta 56.97% in July (Fig. 5.4d).

The Shannon-Weaver diversity index (H') ranged from 0.17 to 3.67 with a distinct fluctuation during the rainfall period (from May to September) at all sampling sites. The index (H') was much higher in spring than in winter. Diversity declined with increasing dominance of Cyanophyta, especially the percent of *P. limnetica* (Fig. 5.5a). Compositional changes could be judged from the daily rate of community change, which ranged from 0.001 to 0.122 (Fig. 5.5b).

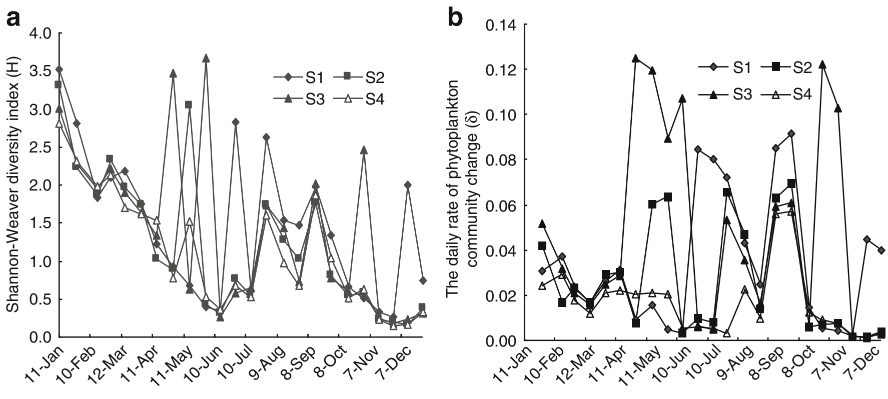

Fig. 5.5 Dynamics of phytoplankton in Dajingshan reservoir. (**a**) Shannon-Weaver diversity index (H'); (**b**) daily rate of phytoplankton community change (δ)

5.3.3 Phytoplankton–Environment Relationships

Correlation coefficients between 11 environmental factors and phytoplankton were calculated for the four sites (Table 5.1). At the four sampling sites a negative correlation between the abundance of Bacillariophyta and water temperature was evident. Water level was negatively correlated with the abundance of Euglenophyta at S1 and S2. The abundance of Chlorophyta was positively correlated with nitrite nitrogen at S2 and S3. Total phosphorus was positively correlated with the abundance of Bacillariophyta, Chlorophyta and Chrysophyta. Total nitrogen and dissolved phosphorus were positively correlated with Chrysophyta.

Canonical Correspondence Analysis (CCA) was used to analyse the relationships between the assemblages and the distribution of the samples because of the large variability of the environmental factors. CCA was performed initially on the whole environmental and species data set. The species–environment correlations for CCA axes 1 and 2 were over 0.9, indicating a significant relationship between the 11 environmental factors and the 54 phytoplankton selected (Table 5.2). In the CCA sample biplot in relation to environmental gradients potentially influencing the phytoplankton, the length of the environmental factor arrows represents the relative explanatory power of each variable with respect to individual sample

Table 5.1 Phytoplankton-environmental variable correlation coefficients at significant level ($p < 0.01$) at four sites

	Variable 1	Variable 2	R
S1	Bacillariophyta	T	$R = -0.526$
	Bacillariophyta	TP	$R = 0.580$
	Euglenophyta	NO_3–N	$R = -0.568$
	Euglenophyta	Water level	$R = -0.581$
S2	Chlorophyta	NO_2–N	$R = 0.619$
	Chlorophyta	TP	$R = 0.715$
	Bacillariophyta	T	$R = -0.595$
	Bacillariophyta	NO_2–N	$R = 0.773$
	Bacillariophyta	TP	$R = 0.766$
	Chrysophyta	NO_2–N	$R = 0.535$
	Chrysophyta	TN	$R = 0.516$
	Chrysophyta	PO_4–P	$R = 0.558$
	Chrysophyta	TP	$R = 0.524$
	Euglenophyta	Water level	$R = -0.55$
S3	Chlorophyta	NO_2–N	$R = 0.733$
	Bacillariophyta	T	$R = -0.641$
	Bacillariophyta	NO_2–N	$R = 0.828$
	Bacillariophyta	TP	$R = 0.514$
	Cryptophyta	NO_2–N	$R = 0.550$
S4	Bacillariophyta	T	$R = -0.586$

T water temperature; *TP* total phosphate; *TN* total nitrogen; *NO_3–N* nitrate nitrogen; *NO_2–N* nitrite nitrogen; *PO_4–P* orthophosphate

Table 5.2 Summary statistics for the first two axes of CCA performed on phytoplankton at Dajingshan Reservoir

Axis	Axis 1				Axis 2			
Sites	S1	S2	S3	S4	S1	S2	S3	S4
Eigenvalues	0.145	0.199	0.17	0.16	0.073	0.08	0.08	0.063
Percentage	18.77	22.3	18.72	18.57	9.52	8.929	8.73	7.266
Cum. percentage	18.77	22.3	18.72	18.57	28.29	31.22	27.4	25.832
Cum. constr. percentage	31.98	37.37	31.9	34.83	48.2	52.34	46.8	48.459
Spec.-env. correlations	0.959	0.966	0.969	0.971	0.961	0.953	0.94	0.917

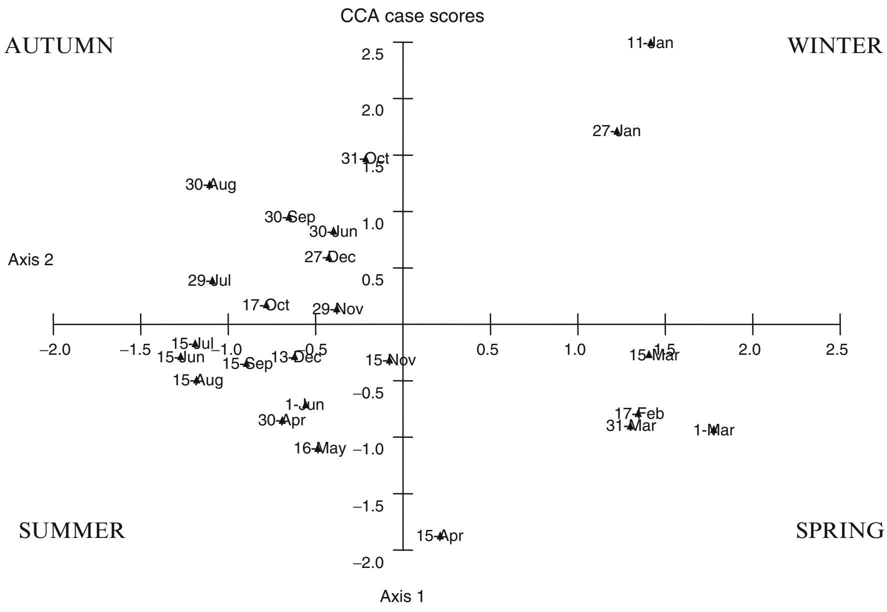

Fig. 5.6 Ordination biplot of cases in Dajingshan reservoir

positions within the ordination and the direction of each environmental gradient through the ordination, shown by the direction of the arrow.

From CCA case scores, it was found that the four seasons clearly separated (Fig. 5.6). Winter and spring was to the right of axis 2 and very short (January to April), but the sideline of the winter and spring was not obvious. Summer and autumn was on the left of axis 2 and very long (May to December).

Figure 5.7 showed that axis 1 negatively correlated with temperature and precipitation, but positively with total phosphorus, total nitrogen, nitrate nitrogen and nitrite nitrogen. Axis 2 positively correlated with water level, but negatively with pH, temperature, precipitation and the nitrogen concentrations were significantly correlated with and affected the compositions of the phytoplankton. In view of phytoplankton composition, mostly Cyanophyta, Bacillariophyta and Euglenophyta were located near the left of axis 1, but Chlorophyta, Chrysophyta and Cryptophyta

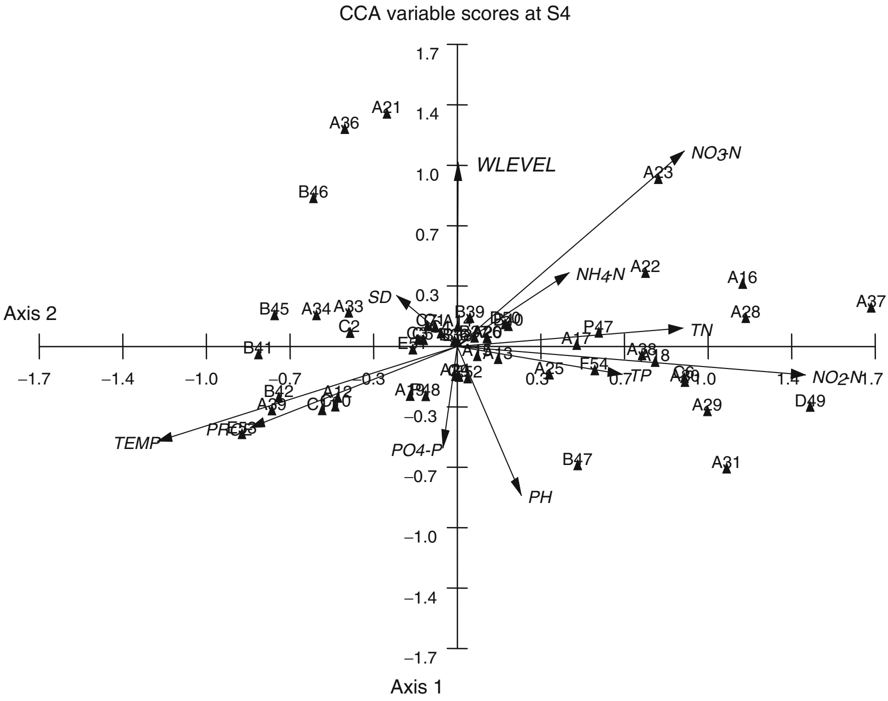

Fig. 5.7 Ordination biplot of phytoplankton species and environmental variables in Dajingshan Reservoir. Phytoplankton species abbreviations are given in Table 5.3

were mostly near the right of axis 1. On the right of axis 2, Chlorophyta, Chrysophyta and Cryptophyta positively correlated with phosphorus and nitrogen, but negatively with temperature and precipitation. On the left of axis 1, Cyanophyta, Bacillariophyta and Euglenophyta negatively correlated with temperature and precipitation.

5.4 Discussion

5.4.1 Phytoplankton Community Structure

Dajingshan Reservoir is a eutrophic water body. The phytoplankton richness added up to 100 taxa, the abundance of phytoplankton was over 10^7 cells·L^{-1} half a year and the peak biomass exceeded 20 mg·L^{-1}. A steady-state assemblage means an invariance of its species composition should last 2 weeks with a few dominant species that should represent at least 50–80% of total abundance and the steady period (Rojo and Álvarez-Cobelas 2003; Komárková and Tavera 2003;

Table 5.3 Codes of phytoplankton species in CCA analysis

Codes	Species	Codes	Species	Codes	Species
	Cyanophyta	A22	*Crucigenia apiculata*	B40	*Melosira ambigua*
C1	*Pseudanabaena limnetica*	A23	*Tetrastrum hastiferum*	B41	*Coscinodiscus lacustris*
C2	*Anabaena spirodies*	A24	*Tetraedron trigonum*	B42	*Gomphonema* sp.
C3	*Cylindrospermopsis rackiborskii*	A25	*Chlorella vulgaris*	B43	*Navicula Bory* sp.
C4	*Dactylocc opsis rhaphidioide*	A26	*Ceolastrum microporum*	B44	*Pinnularia* sp.
C5	*Merismopedia glauca*	A27	*Euastrum spinulosum*	B45	*Tabellaria* sp.
C6	*Aphanocapsa* sp.	A28	*Micractinium pusillum*	B46	*Melosira granulata var. angustissima*
C7	*Chroococcus* sp.	A29	*Golenkinia radiata*		
C8	*Coelosphaerium*	A30	*Selenastrum hantzschii*	B47	*Cocconeis* sp.
C9	*Limnothrix redekei*	A31	*Chlamydomonas* sp.	B48	*Cymbella* sp.
C10	*Raphidiopisi* sp.	A32	*Pediastrum duplex*		*Euglenophyta*
C11	*Lyngbya* sp.	A33	*Peridinium biradiatum*	E51	*Trachelomononas*
	Chlorophyta	A34	*Chlorogonium elongatum*		
A12	*Ankistrodesmus falcatus*	A35	*Scenedesmus obliquus*	E52	*Phacus longicauda*
A13	*Scenedesmus bijuga*	A36	*Eudorina elegans*	E53	*Euglena caudata*
A14	*Scenedesmus quadricauda*	A37	*Palmella mucosa*		*Chrysophyta*
A15	*Scenedesmus dimorphus*	A38	*Tetraspora lacustris*	D49	*Dinobryon divergens*
A16	*Pediastrum duplex*	A39	*Botryococcus braunii*	D50	*Mallomonas* sp.
A17	*Pediastrum biradiatum*		*Bacillariophyta*		*Pyrrophyta*
A18	*Euastrum ansatum*	B36	*Cyclotella meneghiniana*	P47	*Ceratium hirundinella*
A19	*Staurastrum spinulosum*	B37	*Synedra ulna*	P48	*Peridinium* sp.
A20	*Tetraedron minimum*	B38	*Achnanthes minutissima*		*Cryptophyta*
A21	*Crucigenia tetrapedia*	B39	*Melosira granulata*	F54	*Crytomonas* sp.

Naselli-Flores and Barone 2003). The daily rate of community change (δ) was a very important sign of community dynamics but is generally disregarded in the literature. In Dajingshan reservoir, the daily rates were very low in spring and winter, reflecting existence of steady state phytoplankton assemblages. The phytoplankton richness was much higher, but dominated by one or two species. *S. ulna* and *P. limnetica* dominated in spring, but only *P. limnetica* did in the other periods. Because of the persistent dominance of *P. limnetica*, it stabilized species composition of phytoplankton communities at the four sampling sites. Similar phytoplankton communities at steady state were observed (Naselli-Flores and Barone 2003; Moustaka-Gouni et al. 2007). During the rainfall period, the daily rates fluctuated distinctly. This change was shown in a parallel small fluctuation in species diversity and in an equilibrium phase. The daily rate of community change was much lower than that in Batata lake (Huszar and Reynolds, 1997), but fluctuated relatively strongly in April to September. The frequent rainfalls from April to September (monsoon) tend to disturb phytoplankton through hydrodynamics. Rainfall higher than 100 mm is able to enhance runoff and mixing in a short period, and significantly alters the physical structure of this environment (Reynolds 1999).

5.4.2 Phytoplankton–Environment Relationships

The environmental factors affecting the phytoplankton community had high consistency in space, but showed clear seasonal changes in Dajingshan reservoir. According to the CCA analysis, the four quadrants in the biplot corresponded to the four seasons: spring, summer, autumn and winter. Dajingshan Reservoir is located in the northern tropical region, the winter is short, and the four seasons are not clear as in template. In the CCA ordering of environment factors, there were some differences between the four sampling sites. The 24 samples collected near the dam were clearly divided in two groups: winter-spring and summer-autumn by the CCA's ordering. In spring and winter, nutrients and the first principal axis were positively correlated, and became the main contributing variable. But in summer and autumn, water temperature, precipitation and water level became the main relevant variables. The abundance ordering of the 54 species also had a dynamic character, namely the density of Cyanophyta and Bacillariophyta was regulated mainly by water temperature, water level and precipitation, but the density of Chlorophyta was regulated by nutrients. This is similar to what has been found in other reservoirs (Naselli-Flores and Barone. 1997; Negro et al. 2003; Serra et al. 2002). Dajingshan Reservoir is a typical pumped storage reservoir, and pumped water had taken up over 80% of inflow in winter. The water level was directly affected by the pumped water that resulted in a strong change in water level, with a 5 m discrepancy between the crest stage and lowest water. This pumped water correlated with nitrate nitrogen ($R = 0.671, p < 0.01, n = 96$), water transparency ($R = 0.605$, $p < 0.01$, $n = 96$) and pH value ($R = -0.378$, $p < 0.01$, $n = 96$). Phytoplankton development is related with hydrodynamics (Gomes and Miranda 2001; Arfi 2003; Horn 2003). Habib et al. (1997) illustrated that silicon was the main environment factor affecting phytoplankton, but temperature, dissolved oxygen, transparency and chemical oxygen demand were also the functional factors. Naselli-Flores and Barone (1998) showed that water temperature, conductivity and storage capacity were the main factors affecting phytoplankton in Arancio lake, but nitrate nitrogen, nitrite nitrogen, mixing depth and dissolubility silicon were more important in Rosamarina lake. Naselli-Flores (2000) argued that the changes in eutrophic depth, alkalinity and capacity in Sicilian Reservoirs were the maximal factors. When the retention time was 100 days, the phytoplankton community of reservoirs and lakes has a high comparability, because water temperature and nutrition affect the total abundance and biomass of phytoplankton most. Since the average retention time of Dajingshan Reservoir was around 100 days, nutrient load should play an important role in the dynamics of phytoplankton biomass. Because nutrients in pumped water were perennially high, the total phytoplankton biomass was finally controlled by temperature (affecting growth rate) and precipitation (reducing the nutrient loading). Thus, temperature and precipitation became the main driving factors of the dynamics of phytoplankton community structure in Dajingshan Reservoir.

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Chapter 6
Genetic Variation of *Microcystis* Strains from Reservoirs in Guangdong Province Based on DNA-Sequences of the cpcBA–IGS Region

Qun Zhang and Lamei Lei

Abstract With the global increase of eutrophication and water quality deterioration, cyanobacterial blooms become more and more frequent around the world. *Microcystis* is the most frequent bloom-causing Cyanobacterium in freshwater bodies. Not only do its blooms emit foul odors and increase the cost of water treatment but also produce carcinogenic hepatotoxic microcystins. The taxonomy of *Microcystis* is the basis of water quality evaluation and management of harmful algae. However, the current taxonomy is mainly based on morphological criteria, and morphospecies do not necessarily reflect the genetic background of these notorious *Microcystis* strains. As the rapid economic development in the Pearl River Delta has caused river environment degradation, reservoirs have become the main source of drinking water in Guangdong Province; however, *Microcystis* blooms also plagued these reservoirs. In the present study, cpcBA–IGS of 30 *Microcystis* strains isolated from 12 reservoirs in Guangdong Province were sequenced, combined with selected 24 *Microcystis* strains of various geographic origin downloaded from GenBank, to determine their taxonomic status, phylogenetic affinities and geographic distributions. Sixty-seven variable sites and 50 parsimony-informative sites were found in 568 bp cpcBA–IGS region, 54 *Microcystis* strains were clustered into two major clades, of which clade two involved three strains of *M. wesenbergii* and one strain of *M. marginata* and clade 1 included the remaining species and strains. Strains of the same geographic origin did not necessarily cluster together, the same morphospecies may exhibit different genotypes, and the same genotype may represent various morphological characters, indicating that morphological criteria alone are not sufficient to determine species status of *Microcystis* and molecular markers-based taxonomy of *Microcystis* is feasible. Although genetic variation of cpcBA–IGS were not large yet in intrageneric level, more sensitive molecular methods such as AFLP and microsatellites are potentially suitable to

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resolve species status and evaluate the growth and decline of specific *Microcystis* blooms. Molecular markers will play an evermore important role in studies of *Microcystis* bloom.

6.1 Introduction

Recently eutrophication and water quality deterioration has plagued reservoirs around the world. As the tendency of eutrophication of reservoirs in Guangdong Province is more and more evident, local water quality has been declining (Han et al. 2003). With eutrophication of reservoirs, Cyanobacteria blooms become more frequent in this region. *Microcystis* is a widely reported bloom-caused Cyanobacterium in reservoirs (Song et al. 1999) and its blooms are notorious indicators of environmental deterioration. During *Microcystis* blooms, algal density increases explosively. Huge quantities of cells accumulating in the water surface not only emit a foul odor and seriously affect the landscape, but also increase the costs of water treatment and reduce the amount of drinking water. Cyanobacteria blooms additionally produce hepatotoxic microcystin which can cause death to fish, poultry, livestock, and wild animals. Microcystin has a strong carcinogenic effect, posing potential threats to human health through bioaccumulation (Yu et al. 2001; Fleming et al. 2002). *Microcystis* blooms are a public concern and a hot topic in water reservoir environment and public health (Oberholster et al. 2004).

The taxonomy of Cyanobacteria is not only the basis of management of harmful algae, but also an essential component in the evaluation of water quality (Janse et al. 2003). *Microcystis* spp. are the main species responsible for Cyanobacteria blooms in reservoir worldwide. Prevailing taxonomy of *Microcystis* is primarily based on morphological criteria. The main approach to identify them is to describe morphological characteristics such as cell size, shape, and structure, the distribution of cells, their thickness, and the nature of the mucilage in the colonies. However, there are considerable difficulties in identifying *Microcystis* to the species level (Sivonen et al. 1990; Komarék 1991; Codd et al.1999; Otsuka et al. 1999a, b; Bittencourt-Oliveira et al. 2000; Via-Ordorika et al. 2004) because morphological differences of solitary *Microcystis* cells are small and subtle, and the outlines of colonies are variable across development stages. Furthermore, many characters of different morphospecies overlap, and the structure of field colonies may change and even reduce to single cells when subjected to laboratory culture, making the assessment of colonial characters evident only in mature colonies. In traditional morphological identification, pure isolated algal strains are often required to observe their life history. However, pure culture of some species is time-consuming, and sometimes pure culture and culture survival are difficult – even for highly trained technicians (Krüger et al. 1981; Doer and Barker 1988; Bolch and Blackburn 1996; Bittencourt-Oliveira 2000; Otsuka et al. 2000). As many bloom-causing *Microcystis* strains show considerable morphological changes, it is difficult to assign morphotypes to specific species (Bittencourt-Oliveira et al. 2001). Meanwhile, morphological classification is vulnerable to subjective effects as different researchers may have their own criteria to evaluate morphological traits, resulting in fundamental differences in classification systems (Komarek 1991; Rippka 1988; Krüger et al. 1995; Forni et al. 1997; López-Rodas and Costas 1997; Otsuka et al. 1999a, b, 2000; Bittencourt-Oliveira et al. 2001; Via-Ordorika et al. 2004; http://www.nies.go.jp/biology/mcc/class/*Microcystis*.html; Nishihara et al. 1997).

As existing taxonomic criteria of the genus *Microcystis* are controversial or thought to be artificial (Otsuka et al. 2000), scientists want to break through the barriers of phenotypic traits and look instead for essential characters reflecting the position of species independent of environmental variation. As molecular data contain a large number of discrete traits that are easily quantified, especially when changes in DNA sequences are neutral and non-genetic variability is small, genetic analysis is a more accurate reflection of phylogenetic affinity. Much work has been carried out on molecular taxonomy of harmful algae, and a variety of molecular markers have been used. Beltran and Neilan (2000) distinguished the *Anabaena circinalis* strains causing paralytic shellfish poisoning using 16S rRNA sequence analysis. Iteman et al. (2000) used the sequences of the rDNA ITS Region (Internal Transcribed Spacer of ribosomal DNA) to determine phylogenetic relationships within the genus *Microcystis*. Bitterncourt-Oliveira et al. (2001) studied the genetic variation of 15 Brazilian strains of *Microcystis aeruginosa* by sequence analysis of the cpcBA–IGS (cyanophycocyanin alpha and beta subunits, with intergenic spacer). Although major algal control research projects on Dianchi Lake, Taihu Lake, and others have been carried out in China, few endeavors have been made to explore the molecular taxonomy of Cyanobacteria, although Chen et al. (1999a, b) sequenced rDNA ITS region of three *Microcystis* strains and Pan et al. (1999) studied the phylogenetic relationships of seven *Microcystis* strains by RAPD (randomly amplified polymorphic DNA).

Although numerous studies on the molecular taxonomy of *Microcystis* have been published, the classification of *Microcystis* has yet to be resolved. Strains of various geographical origins may exhibit unique ecophysiological characteristics, and thus the genus *Microcystis* should be studied locally. In the present study, cpcBA–IGS of various *Microcystis* strains isolated from large and middle reservoirs in Guangdong Province were sequenced to determine their taxonomic status and phylogenetic affinities and to map out their geographic distributions. cpcBA–IGS is composed of highly variable intergenic spacer (IGS) region between two phycobilisome subunits (*cpcB* and *cpcA*) within the phycocyanin operon, and phycocyanin is an accessory pigment that gives Cyanobacteria their characteristic blue-green color and, together with chlorophyll *a*, it is contained in the photosynthetic apparatus (Glazer 1984). cpcBA–IGS appears to be more useful in discriminating between strains than the commonly employed 16S rRNA gene, which exhibits much lower intrageneric variability in many Cyanobacteria (Moore et al. 1998; Rudi et al. 1998). The present study is aimed at providing data on Cyanobacteria bloom dynamics and on the prevention and control of cyanobacterial water blooms, with the hope to promote sustainable exploitation of reservoir water resources in Guangdong Province.

6.2 Materials and Methods

6.2.1 Collection of Algal Strains

Thirty *Microcystis* strains were isolated from 12 large and medium reservoirs in Guangdong Province and cultured at Jinan University. Twenty-four other *Microcystis* strains were selectively downloaded from GenBank as references for various species and geographical origins. For the sake of convenience, *Microcystis* species are described according to the original literature; the changes of species name will be described in the discussion. Source of the algae is shown in Table 6.1.

6.2.2 DNA Extraction, PCR Amplification, and DNA Sequencing

Algal cultures (1.2 ml) in mid-logarithmic growth phase were placed into sterilized 1.5 ml Eppendorf tubes and centrifuged at 10,000 rpm for 10 min to collect algal cells. After washing twice with cold 70% ethanol, 300 μl of CTAB extraction buffer with 1% proteinase K were added to the pellet and the solution was incubated for 3 h in water bath at 50°C with shaking. An equal volume of chloroform isoamyl alcohol (24:1) was added and mixed, and the resultant homogenate was centrifuged at 10,000 rpm for 10 min again. The supernatant was carefully pipeted into a new tube, then 5 μl of glass milk (Ultra–Sep Gel Extraction Kit–Omega) was added and shaken. After 15–30 min standing, three times the volume of the Binding Buffer was added to the mixture and shaken. The solution was centrifuged at 10,000 rpm and the supernatant was discarded. The pellet was then washed with 500 μl DNA Wash Buffer and dried at room temperature. Total DNA was redissolved in 40 μl 1 × TAE solution and deposited at −20°C.

Primers, PCβF and PCαR, described by Bolch et al. (1996) were used to amplify the cpcBA–IGS and flanking regions. PCR ampliciation was carried out in a Biometra UNOII thermocycler (Biometra) using a final volume of 30 μl containing 3.0 μl 10 × Ex Taq Buffer, 0.75U Takara Ex Taq (TaKaRa Biochemicals), 1 μl template DNA, 8 μmol/l PCβF and PCαR primers, and 2.0 μl DMSO. The mixture was heated at 95°C for 4 min and the samples were amplified for 25 cycles at 94°C for 30 s, 56°C for 30 s and 72°C for 1 min, followed by elongation of 72°C for 10 min. PCR products of 2.0 μl were analyzed by electrophoresis in 1% agarose gels after staining with ethidium bromide. PCR-amplified DNA was purified by using the QIAquick PCR purification Kit (Qiagen) according to the manufacture's protocol. The purified DNA was subjected to direct sequencing on ABI 3730 DNA sequencers (Shanghai Bioasia).

Table 6.1 Original and proposed morphospecies, strain identification, geographical origin, GenBank accession number, and reference (underlined strains are type species of *Microcystis aeruginosa*, and highlighted are strains with variable names)

Original designation	Proposed species	Strain no.	Geographical origin	GenBank no.	Publication
M. wesenbergii	*W. aeruginosa*	NIES 112	Japan, Lake Suwa	AF385391	Bittencourt-Oliveira et al. (2001)
M. wesenbergii	*W. aeruginosa*	NIES 111	Japan, Lake kasumigaura	AF385390	Bittencourt-Oliveira et al. (2001)
M.cf.wesenbergii	M5 morphotype	FCLA–Ninf	Brazil, Usina Santa Rita Reservoir	AF385389	Bittencourt-Oliveira et al. (2001)
<u>*M. aeruginosa*</u>	<u>*M. aeruginosa*</u>	PCC 7941	Canada, Little Rideau Lake	AF385388	Bittencourt-Oliveira et al. (2001)
M. aeruginosa	ND	PCC 7820	Scotland, Loch Balgavies	AF385387	Bittencourt-Oliveira et al. (2001)
M. aeruginosa	ND	PCC 7806	Netherlands, Braakman Reservoir	AF385386	Bittencourt-Oliveira et al. (2001)
M. aeruginosa f. flos-aquae	*M. aeruginosa*	NIES 99	Japan, Lake Suwa	AF385385	Bittencourt-Oliveira et al. (2001)
M. aeruginosa f. flos-aquae	*M. aeruginosa*	NIES 98	Japan, Lake Kasumigaura	AF385384	Bittencourt-Oliveira et al. (2001)
M. viridis	*M. aeruginosa*	NIES 102	Japan, Lake Kasumigaura	AF385383	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M1 morphotype	FCLA–199	Brazil, Garcas Reservoir	AF385382	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M1 morphotype	FCLA–158	Brazil, Garcas Reservoir	AF385381	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M2 morphotype	FCLA–003	Brazil, Garcas Reservoir	AF385380	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M6 morphotype	FCLA–450	Brazil, Garcas Reservoir	AF385379	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M6 morphotype	FCLA–310	Brazil, Garcas Reservoir	AF385378	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M6 morphotype	FCLA–225	Brazil, Cantareira Reservoir	AF385377	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M3 morphotype	FCLA–030	Brazil, Garcas Reservoir	AF385376	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M4 morphotype	FCLA–009	Brazil, Garcas Reservoir	AF385375	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M4 morphotype	FCLA–200	Brazil, Salto Grande Reservoir	AF385374	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M6 morphotype	FCLA–235	Brazil, Garcas Reservoir	AF385373	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M6 morphotype	FCLA–232	Brazil, Garcas Reservoir	AF385372	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M4 morphotype	FCLA–299	Brazil, Garcas Reservoir	AF385371	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M4 morphotype	FCLA–298	Brazil, Garcas Reservoir	AF385370	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M2 morphotype	FCLA–262	Brazil, Garcas Reservoir	AF385369	Bittencourt-Oliveira et al. (2001)
M. aeruginosa complex	M6 morphotype	NPLS–1	Brazil, Logoa Santa Reservoir	AF385368	Bittencourt-Oliveira et al. (2001)
M. flos-aquae	ND	CSJ	China, Chishijing Reservoir	ND	This study
M. sp	*ND*	DJS1	China, Dajinshan Reservoir	ND	This study

(continued)

Table 6.1 (continued)

Original designation	Proposed species	Strain no.	Geographical origin	GenBank no.	Publication
M. sp	ND	DJS2	China, Dajinshan Reservoir	ND	This study
M. sp	ND	DJS3	China, Dajinshan Reservoir	ND	This study
M. sp	ND	DJS4	China, Dajinshan Reservoir	ND	This study
M. flos-aquae	ND	DJS5	China, DajinshanReservoir	ND	This study
M. flos-aquae	ND	DSH1	China, Dashahe Reservoir	ND	This study
M. sp	ND	DSH2	China, Dashahe Reservoir	ND	This study
M. marginata	ND	FLX1	China, Feilaixia Reservoir	ND	This study
M. flos-aquae	ND	FLX2	China, Feilaixia Reservoir	ND	This study
M. flos-aquae	ND	FLX3	China, Feilaixia Reservoir	ND	This study
M. flos-aquae	ND	FLX4	China, Feilaixia Reservoir	ND	This study
M. aeruginosa	ND	HD1	China, Hedi Reservoir	ND	This study
M. flos-aquae	ND	HD2	China, Hedi Reservoir	ND	This study
M. aeruginosa	ND	LXH1	China, Liuxihe River	ND	This study
M. flos-aquae	ND	LXH2	China, Liuxihe Reservoir	ND	This study
M. flos-aquae	ND	GZ1	China, Gaozhou Reservoir	ND	This study
M. flos-aquae	ND	GZ2	China, Gaozhou Reservoir	ND	This study
M. sp	ND	QYS1	China, Qiyeshi Reservoir	ND	This study
M. flos-aquae	ND	QYS2	China, Qiyeshi Reservoir	ND	This study
M. aeruginosa	ND	ST	China, Shatian Reservoir	ND	This study
M. flos-aquae	ND	SZ	China, Shengzhen Reservoir	ND	This study
M. sp	ND	TX1	China, Tangxi Reservoir	ND	This study
M. aeruginosa	ND	TX2	China, Tangxi Reservoir	ND	This study
M. aeruginosa	ND	TX3	China, Tangxi Reservoir	ND	This study
M. aeruginosa	ND	TX4	China, Tangxi Reservoir	ND	This study
M. flos-aquae	ND	TX5	China, Tangxi Reservoir	ND	This study
M. aeruginosa	ND	TX6	China, Tangxi Reservoir	ND	This study
M. marginata	ND	TX7	China, Tangxi Reservoir	ND	This study
M. marginata	ND	XFJ	China, Xinfengjian Reservoir	ND	This study

6.2.3 Phylogenetic Analysis

Completed *Microcystis* sequences were aligned with each other and with those downloaded from GenBank using ClustalX 1.83 program under default settings for indels. The aligned sequences were corrected manually. Phylogenetic analysis of the data was carried out with the computer program MEGA version 4 (Tamura et al. 2007). Pairwise nucleotide divergences were calculated, the phylogenetic tree was recovered by Neigbor-joining method based on the Kimura-2 parameter model, and bootstrap values were obtained from analysis of 1,000 resamplings of the data set.

6.3 Results

cpcBA–IGS sequences of 30 *Microcystis* strains isolated from 12 large- and middle-sized reservoirs in Guangdong Province were determined. Twenty-three strains could be morphologically classified as *M. flos-aquae* (13 strains), *M. aeruginosa* (7 strains), *M. marginata* (3 strains), and the other seven strains remained unresolved. Combined with selected homologous sequences of 24 *Microcystis* strains of various geographic origin and species downloaded from GenBank, sequences of 54 *Microcystis* strains were included in the present study. No insertions or deletions were noted. The sequenced *CpcB* gene was 224 bp in length, and the average contents of T, C, A, and G were 25.7%, 26.4%, 20.9%, and 26.9%, respectively; the ratio of A + T/G + C was 46.6:53.3%; 18 variable sites (8.0%) and 12 parsimony-informative sites were found, i.e., 66.7% variable sites were parsimony-informative. The IGS region was 66 bp long, and the average contents of T, C, A, and G were 26%, 18.9%, 34.1%, and 20.9%, respectively; the ratio of A + T/G + C was 60.1:39.8%; six variable sites (9.1%) and five parsimony-informative sites were found in the region, i.e., 83.3% variable sites were parsimony-informative. The *cpcA* gene was 278 bp in length, and the average contents of T, C, A, and G were 22.2%, 30.4%, 25.5%, and 21.9%, respectively; the ratio of A + T/G + C was 47.7:52.3%; 41 variable sites (14.7%) and 35 parsimony-informative sites (6.16%) were found, i.e., 85.4% variable sites were parsimony-informative. In total, 67 variable sites (10.9%) and 50 parsimony-informative sites (8.8%) were found in 568 bp of sequence, and the average contents of T, C, A, and G were 24%, 27.5%, 24.7%, and 23.8%, respectively; the ratio of A + T/G + C was 48.7:51.3%.

6.3.1 Genetic Variability of cpcBA–IGS Among 54 **Microcystis** *Strains*

Pairwise genetic divergence among 54 *Microcystis* strains in variable sites and percentages are shown in Table 6.2.

As shown in the above table, the following algal strains shared the same genotype:

Table 6.2 Geographic distribution of genotypes in 54 *Microcystis* strains

Location	Genotype																									No. of Haplotypes	No. of strains
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25		
Tangxi					5						1														1	3	7
Shatian													1													1	1
Dashahe													1								1					2	2
Hedi														1										1		2	2
Dajingshan														2	1	1	1									4	5
Liuxihe																		1				1				2	2
Feilaixia				2															1			1				3	4
Qiyeshi				1																1						2	2
Xinfengjiang																						1				1	1
Gouzhou				1																					1	2	2
Chshijing				1																						1	1
Shengzhen																							1			1	1
GdTotal	0	0	0	5	5	0	0	0	0	0	1	0	2	3	1	1	1	1	1	1	1	3	1	1	2	16	30
Br Usina Rita	1																									1	1
Br Garcas		3	1	3		1	1	3																		6	12
Br Cantareira				1																						1	1
Br Americana					1																					1	1
Br Logoa									1																	1	1
Jp Kasumigaura					1					1	1															3	3
Jp Suwa					1						1															2	2
Canada					1																					1	1
Scotland					1																					1	1
The Netherlands												1														1	1
ReferenceTotal	1	3	1	4	5	1	1	3	1	1	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0	11	24
Total	1	3	1	9	10	1	1	3	1	1	3	1	2	3	1	1	1	1	1	1	1	3	1	1	2	25	54

1.	*M. aeruginosa*FCLA–199	*M. aeruginosa*FCLA–158	*M. aeruginosa*FCLA–003	
2.	*M. aeruginosa*FCLA–310	*M. aeruginosa*FCLA–225	*M.aeruginosa*FCLA–030	
	*M. aeruginosa*FCLA–009	*M. flos-aquae*QYS2	*M. flos-aquae*CSJ	*M. flos-aquae*FLX4
	*M. marginata*FLX1	*M. flos-aquae*GZ2		
3.	*M. sp.*FCLA–200	*M. aeruginosa*NIES98	*M. aeruginosa*NIES99	*M .aeruginosa*PCC7941
	*M. aeruginosa*PCC7820	*M. sp.*TX1	*M. aeruginosa*TX2	*M. aeruginosa*TX3
	*M. aeruginosa*TX4	*M. aeruginosa*TX6		
4.	*M. aeruginosa*FCLA–299	*M. aeruginosa*FCLA–298	*M. aeruginosa*FCLA–262	
5.	*M. wesenbergii*NIES111	*M. wesenbergii*NIES112	*M. marginata*TX7	
6.	*M. aeruginosa*ST		*M. flos-aquae*DSH1	
7.	*M. flos-aquae*HD2	*M. sp* DJS4	*M. flos-aquae*DJS5	
8.	*M. flos-aquae*FLX3	*M. flos-aquae*LXH2	*M. marginata*XFJ	
9.	*M. flos-aquae* GZ1	*M. flos-aquae* TX5		

6.3.2 Phylogenetic Trees Based on cpcBA–IGS of 54 **Microcystis** *Strains*

As shown in Fig. 6.1, 54 *Microcystis* strains were phylogenetically divided into two major clades in the neighbor-joining tree based on the Kimura 2-parameter model: Clade 1 was composed of four subclades, of which subclade one contained most of the strains analyzed. Clade 2 involved four strains – three strains of *M. wesenbergii* (NIES111–112 and FCLA–NinfM5) and one strain of *M. marginata* (TX7). Clade 1 included 50 strains, of which 13 strains of the *M. aeruginosa* species complex (including M1–M4, M6 morphotypes except two strains of uncertain species status), 12 strains of *M. aeruginosa* (including two strains of *M. aeruginosa* f. *flos-aquae*), 13 strains of *M. flos-aquae*, 2 strains of *M. marginata*, 1 *M. viridis* strain, and 9 unclassifed strains.

6.3.3 Phylogenetic Relationships Between Strains of the Same **Microcystis** *Morphospecies*

The 16 *Microcystis* strains from 5 reservoirs in Brazil, with the exception of *M.* cf. *wesenbergii* FCLA–Ninf, were morphologically assigned to three categories: *M. aeruginosa*, *M. panniformes,* and an unidentified species, or were referred to as *M. aeruginosa* species complex collectively; these strains could also be classified into six morphogroups (M1–M6) according to the colonial forms in different developmental stages, the distribution of individual cells in the colony, mulicage characters, and the mode of producing new colonies (Bittencourt-Oliveira et al. 2001). In total there were 9 genotypes in the 16 Brazilian strains of *Microcystis*, of which 7 were endemic. As *M. aeruginosa* and *M. panniformis* were not differentiated in Brazilian strains of *Microcystis* (Bittencourt-Oliveira et al. 2001), direct comparisons of Brazilian *M. aeruginosa* species complex with other *M. aeruginosa*

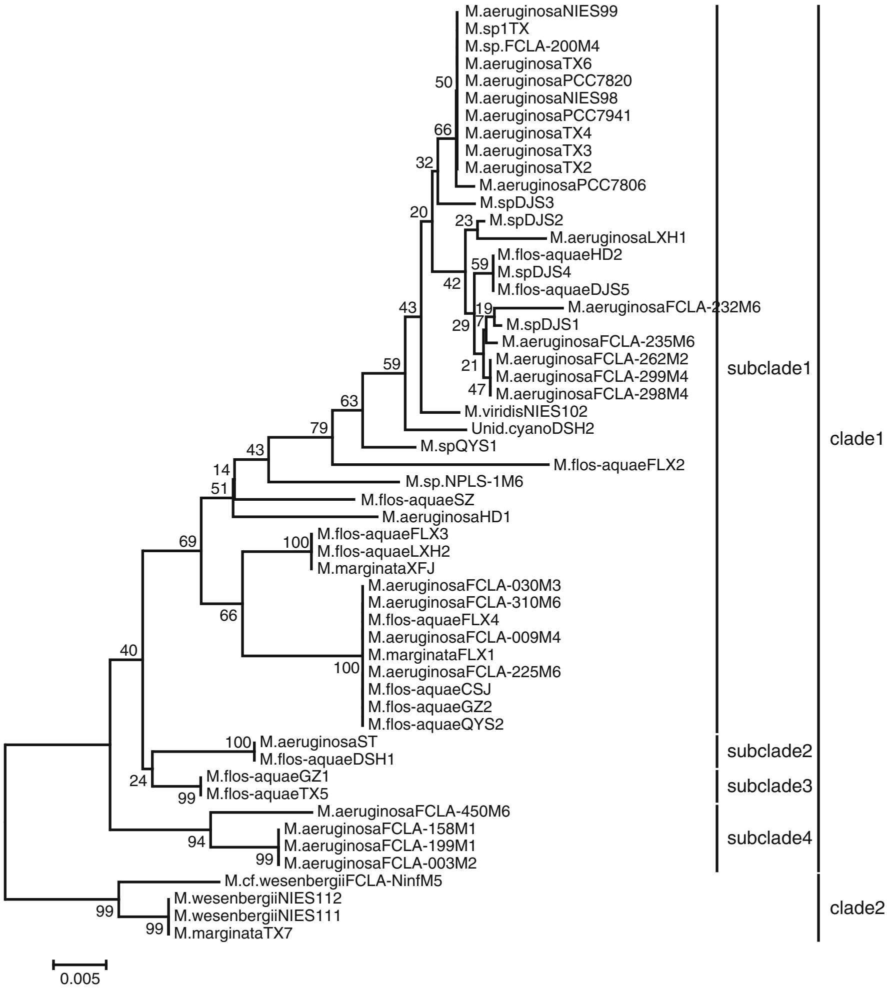

Fig. 6.1 NJ tree obtained for sequences of cpcBA–IGS region of 54 *Microcystis* strains. Bootstrap values are given around the branches; scale bars were scaled to genetic distance

strains were not possible. The Brazilian *M. aeruginosa* species complex is discussed separately here.

As shown in the genetic variability and phylogentic tree based on cpcBA–IGS sequences, the M1 morphogroup included strains *M. aeruginosa* FCLA–158 and *M. aeruginosa* FCLA–199 with the same genotype; M2 morphogroup included *M. aeruginosa* FCLA–262 and *M. aeruginosa* FCLA–003 with different genotypes in distantly related different clusters. M3 morphogroups contained only one strain *M. aeruginosa* FCLA–030 with different genotypes and are located in a different cluster from other Brazilian strains. M4 morphogroups included four strains:

M. aeruginosa FCLA–298, *M. aeruginosa* FCLA–299, *M.* sp. FCLA–200, *M. aeruginosa* FCLA–009, and three genotypes all in the same subclade. Five genotypes were found in six strains of *M. aeruginosa* NPLS–1, *M. aeruginosa* FCLA–232, *M. aeruginosa*FCLA–235, *M. aeruginosa*FCLA–225, *M. aeruginosa* FCLA–310, and *M. aeruginosa* FCLA–450; they were not necessarily closely related.

Except for the Brazilian *Microcystis aeruginosa* species complex, the 40 *Microcystis* strains (including unidentified Brazilian strains of *Microcystis M.* sp. FCLA–200 and *M. sp*NPLS–1) in the present study were morphologically classified and genotyped in the phylogenetic tree as follows:

Five genotypes were found in 12 strains of *M. aeruginosa* (including 2 *M. aeruginosa* f. *flos-aquae* strains NIES98, NIES99), of which 7 *M. aeruginosa* strains in 30 *Microcystis* strains were isolated from 12 reservoirs in Guangdong Province, China. *M. aeruginosa* strains PCC7820, TX2–4, TX6, and PCC7941 (type species) had an identical genotype to *M. aeruginosa* f. *flos-aquae* NIES98 and NIES99, and differed from *M. aeruginosa* PCC7806 by only one bp. LXH1 clustered with the above strains. HD1 was also located in subclade 1, distantly related to the above strains; ST was located in subclade 3, suggesting a poor affinity to the remaining *M. aeruginosa* strains.

Seven genotypes existed in 13 *M .flos-aquae* strains, of which DJS5 and HD2 were genetically identical and closely related to FLX2. LXH2 and FLX3 shared the same genotype, as did also GZ2, QYS2, CSJ, and FLX4. The same genotype shared by TX5 and GZ1 has a relatively close relationship with DSH1, located in subclade 2 and subclade 3, respectively.

Three *M. martinata* strains had 3 genotypes, and TX7 in clade 2 were quite different from FLX1 and XFJ in the basal branches of subclade1.

Two genotypes found in three *M. wesenbergii* strains were located in clade 2, of which Japanese strains NIES111 and NIES112, with the same genotype, differed from the Brazilian strain *M.* cf. *wesenbergii*FCLA–NinfM5 by 8 bp (1.4% genetic variability)

Nine strains were not morphologically determined (the generic status of DSH2 could not be assigned morphologically). With the exception of Brazilian strain *M. sp*FCLA–200 which shared the same genotype with *M.sp*TX1, the strains differed in their genotype and all were located in subclade 1, with *M. sp*NPLS–l relatively isolated from the remaining eight strains.

In summary, as shown by their genotypic composition and phylogenetic position, some strains of the same morphotype shared the same or similar genotypes located in the same or adjacent branches, while others had quite different genotypes in distantly related branches. In other words, strains of the same or similar morphotypes could not be clearly differentiated genetically.

6.3.4 Phylogenetic Relationships Among Different Morphospecies

Except for the uncertainty in assigning Brazilian strains of the *M. aeruginosa* species complex to particular morphospecies, the species status of other strains was determined morphologically. Those that shared the same genotypes are:

- Two strains of *M. aeruginosa f.flos* (NIES98, NIES99) and six strains of *M. aeruginosa* (PCC7820, PCC7941, TX2–4, TX6)
- *M. wesenbergii* NIES111 and NIES112 with *M. marginata* TX7
- *M. flos-aquae* strains QYS2, CSJ, FLX4, and GZ2 with *M. marginata* FLX1
- *M. flos-aquae* HD2, DJS5, and unidentified DJS4
- *M. flos-aquae* FLX3, LXH2, and *M. marginata* XFJ
- *M. aeruginosa* strain ST and *M. flos-aquae* DSH1
- *M. marginata* strain FLX1 and *M. flos-aquae* QYS2, CSJ, FLX4, GZ2
- *M. marginata* strain TX7 and *M. wesenbergii* NIES111 and NIES112
- *M. marginata* strain XFJ and *M. flos-aquae* FLX3 and LXH2

As shown above, some strains of different morphospecies shared the same or similar genotype and clustered in the same clade, while some strains of the same morphospecies had quite different genotypes; i.e., *Microcystis* strains could not be differentiated morphologically.

6.3.5 Phylogenetic Relationships of Strains from the Same Region

Twelve *Microcystis* Strains from Garcas Reservoir in Brazil were morphologically classified into five groups (M1–M4,M6) (Bittencourt-Oliveira et al. 2001) and encompassed six genotypes. Four strains (FCLA–199 M1, FCLA003–M2, FCLA–158 M1, and FCLA–450 M6) were located in subclade 4 supported by 95% bootstrap values (9 bp difference or 95% genetic similarity); three strains (FCLA–310 M6, FCLA–030 M3, and FCLA–009 M4) of the same genotype appeared in the basal subclade1; the remaining five strains (FCLA–232 M6, FCLA–299 M4, FCLA–298 M4, FCLA–262 M2, and FCLA–235 M2) were adjacently grouped into subclade 1 with base composition less than 5 bp and more than 99.1% genetic similarity. In other words, *Microcysitis* strains from Garcas Reservoirs could be clustered into three distantly related subclades with close within relationships.

If 16 strains of five reservoirs in Brazil were considered as from the same region, they could be assigned into six morphological groups: M1–M6 (included morphospecies *M. aeruginosa*, *M. wesenbergii*, *M. panniformes*, and *Microcystis* incertae sedis). M1 group included two strains with the same genotype, while M2 group had two strains of different genotypes. Both M3 and M5 groups included one single strain. M4 group involved in four strains having three genotypes. M6 groups were composed of six strains with five genotypes. Except FCLA–Ninf (M5 group), which is located in Clade 2, the remaining 15 strains were distributed in to subclade 4 (four strains, of which three strains shared the same genotype) and subclade1 (four strains of the same genotype, NPLS–1 in M6, and the remaining six strains). Some Brazilian strains of *Microcystis* were closely related and located in the same or adjacent subclades; others were distantly related as shown in different branches of the phylogenetic tree.

Three *Microcystis* strains from Lake Kasumigura in Japan were identified as three morphospecies: *M. aeruginosa* f. *flos-aquae* (NIES8), *M. viridis* (NIES102), and *M. wesenbergii* (NIES111), each having different genotypes. Both NIES98 and NIES102 located in the subclade1 had 4 bp difference in the total cpcBA–IGS sequences, or 99.3 genetic similarity. NIES111 were found in clade 2, distantly related to NIES98 and NIES102 with 31 bp difference (94.5 genetic similarity) among them. Further, *M. aeruginosa* f. *flos-aquae* NIES99 and *M. wesenbergii* NIES112 from Lake Suwa in Japan had the same degree of genetic variability as NIES111 to NIES98 and NIES102, respectively, and are in the same clades. Thus, *Microcystis* strains from Japan could be classified into three morphospecies: two *M. wesenbergii* strains with the same genotype, two *M. aeruginosa* f. *flos-aquae* strains of identical sequence, and two strains of *M. viridis*. Five strains were distributed into the distantly related clade 2 and subclade 1.

One *M. flos-aquae* strain and four unclassifed *Microcystis* strains were collected from Dajingshan Reservoir. *M. flos-aquae* DJS5 and *M. sp* DJS4 shared the same genotype, but the other three unclassified strains had different genotypes. All strains in the present study are located in subclade1 with less than 8 bp base difference (1.4% genetic variability), suggesting a close relationship. *M. flos-aquae* DSH1 and unclassified DSH2 isolated from Dashahe Reservoir contained genotypes with up to 20 bp difference (3.5% genetic variability) and are located in subclade 4 and subclade 1, respectively. Of four strains isolated from Feilaixia Reservoir, *M. marginata* FLX1 and *M. flos-aquae* FLX4 were genetically identical, whereas *M. flos-aquae* FLX2 and FLX3 both had their own genotypes. Genotypes in subclade 1 differed by 26–27 bp or 4.6–4.8% difference. There were 22 bp difference (3.9% genetic variability) between one *M. aeruginosa* strain and one *M. flos-aquae* strain from Hedi Reservoir located in subclade 1, suggesting their phylogenetic relationship was not close. *M. aeruginosa* and *M. flos-aquae* strains from Liuxihe Reservoir, located in basal and above branches of subclade1, differed by 22 bp (or 3.5% difference). Two *M. flos-aquae* strains from Gaozhou Reservoir differed by 19 bp (3.3%) and are located in basal subclade1 and subclade 2. One *M. flos-aquae* strain and one unclassified strain from Qiyeshi Reservoir located in subclade1 differed by 22 bp (3.9%). Three genotypes were resolved in seven strains from Tangxi Reservoir, of which five strains (4 *M. aeruginosa* strains and one unclassified strain) shared the same genotype, one *M. flos-aquae* strain and one *M. marginata* strain had their own genotypes. Genotypes that differed by 20–30 bp difference (3.5–5.5%) appeared in subclade1, subclade 2, and clade 2, suggesting a distant affinity.

Of *Microcystis* from the entire Guangdong Province, 30 strains were morphologically classified into *M. aeruginosa* (7 strains), *M. flos-aquae* (13 strains), *M. marginata* (3 strains), and *Microcystis incertae sedis* (7 strains, of which 1 could not even be assigned to a genus.). Seven *Microcystis aeruginosa* strains contained four genotypes, located in subclade 1 (four strains (TX2–TX4, TX6) from Tangxi Reservoir and LXH1 from Liuxihe Reservoir 1 LXH1), subclade 3 (ST from Shahe Reservoir), and HD1 from Hedi Reservoir located in the middle branches of subclade1. Seven genotypes resolved in 13 strains of *Microcystis flos-aquae* were located in subclade 3 (DSH1 from Dashahe Reservoir), subclade 2 (TX5 from Tangxi Reservoir and GZ1 from Gaozhou Reservoir), and subclade 1 (included GZ2 from Gaozhou Reservoir, QYS2 from Qiyeshi Reservoir, CSJ from Chishijin Reservoir CSJ, LXH2 from Liuxihe Reservoir, FLX2–4 from Feilaixia Reservoir, HD1–2 from Hedi Reservoir, and DJS5 from Dajinshan Reservoir). Three strains of *M. marginata* with three genotypes are located in clade 2 (TX7 from Tangxi Reservoir) and basal subclade 1 (XFJ fromo Xinfengjiang Reservoir and FLX1 from Feilaixia Reservoir). All 7 strains of *Microcystis incertae sedis* had different genotypes and were located in subclade 1. In total, 16 genotypes were resolved in 30 *Microcystis* strains, of which 12 genotypes were endemic.

In summary, some strains of different species from the same region had the same or similar genotypes, located in the same or adjacent branches. Some strains of the same species of different location were genetically different and are distributed in different branches of the phylogenetic tree. In other words, strains of various geographic origins could not be separated morphologically and genetically.

6.3.6 Phylogenetic Relationships Among Strains of Different Geographical Origin

Four strains of the *M. aeruginosa* species complex (FCLA–310, FCLA–225, FCLA–030, and FCLA–009) from Garcas Reservoir in Brazil and *M. flos-aquae* strains QYS2, CSJ, FLX4 and *M. marginata* strains FLX1 and GZ2 from Guangdong Province in China shared the same genotype. So too were strains of *M. aeruginosa* species complex FCLA–200 from Brazil, strains of *M. aeruginosa* f. *flos-aquae* (NIES 98 and NIES99, isolated from Lakes Suwa and Kasimigura) from Japan, *M. aeruginosa* strain PCC7941 from Little Rideau Lake in Canada, PCC7820 from Loch Balgavies in Scottland, and five *Microcystis* strains from Tangxi Reservoir (TX1–T4, TX6) in Guangdong Province.

M. wesenbergii NIES111 and NIES112 from Japan contained the same genotype as *M. marginata* TX7 from Tangxi Reservoir in China. *M. aeruginosa* ST and *M. flos-aquae* DSH1 shared the same genotypes, and *M. flos-aquae* HD2 and

DJS5 and unidentified DJS4 were of the same genotype also. *M. flos-aquae* FLX3 and LXH2, *M. marginata* XFJ were genetically identical. cpcBA–IGS sequences of *M. flos-aquae* GZ1 and TX5 were also the same.

As already shown above, some *Microcystis* strains of different geographic origins contained similar genotypes and are clustered together, but other strains are genetically quite different and are located in different clades. As such, *Microcystis* strains did not cluster geographically regardless of their species status.

6.3.7 Phylogenetic Relations Among Microcystis *Strains from Lakes and Reservoirs*

Of 54 strains studies in the present paper, there were 6 lacustrine strains and 48 reservoir strains. As shown morphologically and genotypically, two *M. wesenbergii* strains in Japan (NIES111 and NIES112) shared their genotype with *M. marginata* TX7 from Tangxi Reservoir. *M. aeruginosa* f. *flos-aquae* strains (NIES98 and NIES99) from Japan, *M. aeruginosa* strains PCC7941 from Canada and PCC7820 from Scottland, and four strains of the same species and one unidentified strains from Tangxi Reservoir were genetically identical in the cpcBA–IGS region.

Microcystis from reservoirs could be classified into *M. aeruginosa* species complex (13 strains), *M. flos-aquae* (13 strains), *M. marginata* (3 strains), *M. wesenbergii* (3 strains), *M. viridis* (1 strains), and 9 strains that could not be assigned to specific species. Five of these genotypes were found in seven *M. aeruginosa* strains (including three strains, TX2–TX4, and TX6 from Tangxi Reservoir, LXH1 from Liuxihe Reservoir, HD1 from Hedi Reservoir, and ST from Shatian Reservoir in Guangdong Province and one strain (PCC7806) from Braakman Reservoir in the Netherlands). Thirteen *M. flos-aquae* strains (DJS5 from Dajinshan Reservoir, HD2 from Hedi Reservoir, HXH2 from Liuxihe Reservoir, FLX3 from Feilaixia revervoir,GZ2 from Gaozhou Reservoir, QYS2 from Qiyeshi Reservoir, CSJ from Chishijin Reservoir, FLX4 from Feilaixia Reservoir, TX5 from Tangxi Reservoir and GZ1 from Gaozhou Reservoir, DSH1 from Dashahe Reservoir, and FLX2 from Feilaixia Reservoir) hold seven genotypes. All three *M. marginata* strains (TX7, FLX1, and XFJ) in Guangdong Province had different genotypes. Nine unidentified *Microcystis* strains (FCLA–200 and NPLS1 from Brazil; TX1, DSH2, DJS1–4 and QYS1 in Guangdong Province) retained eight genotypes.

We conclude that lake type *Microcystis* strains of the same species contained the same genotype. Different Reservoir type *Microcysits* species may have the same genotypes, and different genotypes may appear in the same morphospecies.

6.4 Discussion

6.4.1 *Comparison of* cpcBA*–IGS, 16S rRNA and 16S–23S rDNA ITS (Internal Transcribed Spacer) Sequences in Cyanobacterial Taxonomy*

Currently, ribosomal genes, especially 16S rDNA, are widely used to study the evolution and classification of microorganisms (Nelissen et al. 1996; Litzvaitis 2002; Kim et al. 2004). Although 16S rDNA is useful for phylogenetic analysis of higher-level taxa, it is too conserved for lower-level taxa. With average sequence divergence generally less than 3% and often less than 1% (Neilan et al. 1997a; Rudi et al. 1997; Otsuka et al. 1998), 16S rDNA sequences could only provide limited phylogenetic information at the intrageneric level, and sequencing errors in the same strain in different papers are sometimes larger than differences found among strains (Rudi et al. 1997; Lyra et al. 2001). As such, to address relationships at the intrageneric level and of speciation patterns, alternative genes or DNA fragments with faster evolutionary rate should be considered. 16S–23S rDNA ITS region (internal transcribed spacer of nuclear ribosome, rDNA ITS) located between the small subunit (16S rDNA) and large subunit (23S rDNA) was divided into two sections: ITS1 and ITS2, separated by 5.8S rDNA. With up to 7% sequence variation (Neilan et al. 1997b; Otsuka et al. 1999a, b), rDNA ITS displays more sequence variation, making it more appropriate to study intrageneric taxonomy of Cyanobacteria (Barry et al.1991; Janse et al. 2003). However, the 16S and 23S rDNA sequence itself is conservative, and it is difficult to design *Microcystis*- or Cyanobacteria-specific primers in this region. Sequence analysis of rDNA-related markers often requires isolation and culture of sterile samples. Sterile isolation and culture of microalgae is not only time-consuming but also technically demanding and sometimes simply impossible. In this study, a variety of primer combinations were used to amplify *16S rDNA* gene of 30 *Microcystis* strains, but a large number of mixed peaks found in the reading of final direct PCR sequencing complicated sequence analyses. This could be due to (1) the specificity of the primers to Cyanobacteria being weak such that they might have co-amplified many other microorganisms, and (2) the failure to carry out strict sterile isolation and cultivation, which might have lead to the growth of various non-targeted microorganisms. Thus, although nuclear ribosome-related gene sequence analysis has been successfully used, its application in molecular taxonomy of microorganisms is not infallible, and alternative genes are needed to address the range of problems.

Pairwise rDNA ITS sequence similarity among 47 strains of 5 morphospecies and some unidentified species in the genus of *Microcystis* was 93.3–100% (Otsuka et al. 1999a, b); in the present study, pairwise cpcBA–IGS sequence similarity among 54 strains of various species in the genus *Microcystis* was 93.5–100%, indicating that the cpcBA–IGS and rDNA ITS regions display similar levels of genetic variability and should have similar efficiency in the phylogenetic analysis.

As the *cpcBA–IGS* gene only appears in Cyanobacteria, it could be selectively amplified to effectively avoid possible disturbance of other bacteria and other microorganisms, i.e., it could be used to directly to identify Cyanobacteria without isolation and culture sterile algae strains. In contrast to the failure in sequencing 16S rDNA, sequences of cpcBA–IGS were successfully recovered in the present study. As such, cpcBA–IGS have some advantages over rDNA-related markers in molecular taxonomy of Cyanobacteria, making whole cell amplification of unique taxa possible without aseptic isolation and culture (Neilan et al. 1995).

6.4.2 *Taxonomy and Geographic Origin of* Microcystis

Geitler (1932) supposed that *Microcystis* might be classifed into two types based on gas vesicles in its cells (sensu Desikachary 1959). *Bergey's Manual of Determinative Bacteriology* points out that gas vesicles in cells are one of the characteristic properties of *Microcystis* (Bergey et al. 1994), and Stainer et al. (1971) suggest only gas vesicle-containing species should remain in the genus *Microcystis*. 16S rDNA sequence analyses demonstrated that *Microcystis* with and without gas vesicles are only distantly related in the phylogenetic tree of Cyanobacteria (Neilan et al. 1997a, b). Otsuka et al. (1998) also suggested that species without gas vesicles should no longer be classified as *Microcystis*. Therefore, species without gas vesicles were not included in the present study.

6.4.2.1 Morphological Studies of *Microcystis*

Morphological taxonomy of the genus of *Microcystis* is primarily based on cell size, colonial forms, and the characteristics of mucilage sheaths. Ostenfeld (1904) observed many transitional types in the *Microcystis* aeroginosa/*flos-aquae*/*viridis* complex and concluded that all *Microcystis* species with gas vesicles are of the same species. In other studies, Crow (1923) identified 17 *Microcystis* species from phytoplankton in Ceylon, Geitler (1923) separated *Microcystis* into 23 species, and Drouet (1939) found only 3 species in his work on a large collection of planktonic *Microcystis*. Doer and Barker (1988) suggested morphological differences among *Microcystis* species were subtle and volatile; in particular, laboratory cultures and field samples of microalgae were difficult to be compared directly. Rippka (1988) argued that the existing morphological taxonomy could not accurately identify different species of the genus. Komarék (1991) differentiated *Microcystis* in Japan into six species, namely *M. viridis, M. wesenbergii, M. aeruginosa*, *M. novacekii*, *M. ichthyoblabe*, and *M. flos-aquae*. Krüger et al. (1995) surveyed the possibility of using the composition of fatty acids as a taxonomic criterion and found it potentially applied to the taxonomy of Cyanobacteria including *Microcystis*, but species delimitations of *Microcystis* were not reported in his paper. Watanabe (1996) argued that *M. flos-aquae* were just smaller-sized *M. ichthyoblabe* colonies with more uniform cell arrangement, and based on the morphological ambiguity among them, *M. aeruginosa*, *M. ichthyoblabe,* and *M. novacekii* were merged into a *M. aeruginosa* species complex. López-Rodas and Costas (1997) used lectins and antibodies to identify *Microcystis* species; the resultant taxonomy did not match the traditional morphological classification. Otsuka et al. (1999a, b) studied cell sizes, optimum growth temperature, salinity tolerance, G + C content in DNA, chemical and light heterotrophy, total fatty acids in 22 strains of *M. wesenbergii*, *M. aeruginosa*, *M. novacekii*, *M. ichthyoblabe* (*M. flos-aquae* included), *M. viridis*, and concluded that these characters were too similar to be useful in separating different species. Otsuka et al. (2000) observed continuous cultures of five *Microcystis* species (*M. aeruginosa*, *M. ichthyoblabe*, *M. viridis*, *M. wesenbergii*, and *M.novacekii*) and found that some *M. novacekii* strains displayed morphological characters of *M. aeruginosa* or *M. ichthyoblabe*; they suggested that morphological criteria were difficult to separate the five morphological species and that the efficiency of morphological criteria to indentify *Microcystis* species was uncertain.

According to colony shape and growth characteristics, pattern of cell accumulation, cell size, and the nature and thickness of mucilage sheaths, Bittencourt–Oliveira et al. (2001) divided 16 Brazilian strains of *Microcystis* into 6 morphological types (M1–M6), except for M5 morphological types (only one strains), which corresponded to *M. wesenbergii*, and the remaining 15 strains of algae were treated as "*M. aeruginosa* species complex" (including 13 *M. aeruginosa* strains, 2 *M. panniformis*, and 2 unidentified strains). Based on cell size, colony shape and size, cell density in the colony, the fringe characters of the mucilage sheath, and colonial shapes in early developmental stages, Via-Ordorika et al. (2004) classified 322 *Microcystis* strains isolated from 13 water bodies in 9 European countries into 7 morphospecies: *M. ichthyoblabe*, *M. panniformis*, *M. flos-aquae*, *M. aeruginosa*, *M. botrys*, *M. viridis*, and *M. wesenbergii*. In a series of papers (see below), Otsuka and coworkers merged *M. aeruginosa flos-aquae* strains NIES98 and NIES99, *M. viridis* strain NIES102, and *M. wesenbergii* strains NIES111–112 into *M. aeruginosa* (http://www.nies.go.jp/biology/mcc/class/*Microcystis*.html).

Thus, although the history of morphological taxonomy of the genus *Microcystis* is more than 100 years old, it is quite controversial how many species morphospecies exist, and no biogeographic patterns based on morphological species have been reported.

6.4.2.2 Molecular Taxonomy of the Genus *Microcystis*

Kato et al. (1991) found that *M. viridis* and *M. wesenbergii* each had its own genotype, and that isozyme analysis might be used to detect genetic divergence within and among species. However, isozymes used as gene expression products are vulnerable to the impact of environmental factors and developmental stages.

Otsuka et al. (1998) constructed 16S rDNA phylogenetic trees of *Microcystis* and found that genotypes of the same morphospecies were not necessarily closely related. 16S rDNA sequence analysis did not support the separation of *M. viridis* and *M. wesenbergii* from *M. aeruginosa* in the traditional taxonomy, based on small and variable morphological traits (Neilan et al. 1997a, b). Genetic distances among strains of *Microcystis aeruginosa* were larger than those of different species. The molecular phylogenetic trees showed *M. aeruginosa* strains as interwined with other *Microcystis* speceis. Otsuka et al. (1998) also found that *Microcystis* morphotypes did not necessarily reflect phylogenetic relationships, and *M. viridis*, *M. wesenbergii*, *M. aeruginosa*, *M. novacekii*, and *M. ichthyoblabe* (*M. flos-aquae* included) were so closely related phylogenetically that they should be sunk into the same species.

Based on PCR–RFLP (restriction fragment length polymorphisms) of the cpcBA–IGS region, Otsuka et al. (1999a, b) found 8 genotypes in 22 strains of 5 morphospecies (*M. viridis*, *M. wesenbergii*, *M. aeruginosa*, *M. novacekii* [*M. flos-aquae* included], *M. ichthyoblabe*) and other unidentified strains; except *M. ichthyoblabe*, strains of the same morphospecies isolated from the same place shared the same genotypes, although *M. ichthyoblabe* genotypes did not necessarily cluster according to their geographic orgin.

16S rDNA sequences were too conservative to be suitable for the taxonomy of *Microcystis* species and biogeographic analysis, cpcBA–IGS PCR–RFLP also provided limited phylogenetic information, and the evolutionary tree topology did not fully represent phylogenetic relationships among *Microcystis* strains. Molecular markers with faster evolutionary rate would be helpful to explain phylogenetic relationships of different *Microcystis* species and strains of various geographic origins.

Otsuka et al. (1999a, b) sequenced the rDNA ITS region between 16S and 23S rDNA of 47 *Microcystis* strains including *M. viridis*, *M. wesenbergii*, *M. aeruginosa*, *M. ichthyoblabe* (*M. flos-aquae* here as one morphotype of the species), *M. novacekii*, and other unidentified forms, and found that nine strains of *M. aeruginosa*, *M. viridis,* and *M. wesenbergii* shared the same genotype. *M. novacekii* also shared the same genotype with three *M. ichthyoblabe* strains, while *M. wesenbergii* had different genotypes. Except for monophyletic *M. viridis*, all other morphospecies were polyphyletic.

Janse et al. (2004) assigned 107 *Microcystis* populations isolated from 15 water bodies in 8 European countries and Morocco to 8 morphospecies (*M. aeruginosa*, *M. ichthyoblabe*, *M. flos-aquae*, *M. viridis*, *M. wesenbergii*, M. *panniformis*, *M. botrys*, and *M. novacekii*), and the resultant 59 rDNA ITS DGGE (denaturing gradient gel electrophoresis analysis) genotypes could differentiate toxic and non-toxic algae strains, suggesting molecular phylogenetic relationships did not match with morphological analyses. As algal strains of different geographical origin may have the same genotype, rDNA ITS sequence analysis could not necessarily reveal their geographical origins.

CpcBA–IGS sequences of 54 *Microcystis* strains isolated from 21 water bodies in 6 Asian, American and European countries were analyzed in the present study, of which 25 genotypes were found in 6 morphospecies (*M. aeruginosa*, *M. flos-aquae*, *M. viridis*, *M. wesenbergii*, *M. marginata*, and *M. ichthyoblabe*) and the

M. aeruginosa species complex (M1–M4 and M6 morphotypes, including *M. panniformis*), and no correlation were found between *Microcystis* morphospecies and their genotypes.

In the molecular phylogenetic tree, 54 *Microcystis* trains were divided into Clades 1 and 2. Clade 2 was composed of three *M. wesenbergii* strains and one strain of *M. marginata*, corresponding to Cluster 3 in Otsuka et al.'s (1999a, b) tree. Clade 1 included the other 50 *Microcystis* strains (*M. wesenbergii* strains NIES111 and NIES112 originated from Japan and locally isolated *M. marginata* TX7 shared the same genotype), corresponding to Otsuka et al.'s (1999a, b) Cluster 3. Clade 1 included 50 algae (as shown in Fig. 6.1) that are similar to those of Cluster 1 in Otsuka et al. (1999a, b). In the present study, all morphospecies were polyphyletic except those species with single strains, and phylogenetic relationships among strains of various geographic origins were not clear. As shown in genotype composition (Table 6.1) and molecular phylogenetic tree (Fig. 6.1), although some strains with similar morphological characters may have the same or similar genotypes located in the same or nearby branches in the phylogenetic tree, other strains of the same morphotypes may have quite different genotypes that make them appear in distant branches of the tree. Some strains of different morphospecies may have the same or similar genotypes, i.e., *Microcystis* morphotypes could not be distinguished by cpcBA–IGS sequences. Some *Microcystis* strains of the same geographic origin may have different genotypes, and some *Microcystis* strains isolated from various locations may share the same genotypes, indicating that geographic origin does not reflect phylogenetic relationships in *Microcystis*. *Microcystis* strains originally from reservoirs may have different or similar genotypes independent of their morphological characters. Although *Microcystis* of the same species from various lakes shared the same genotype, this result should be treated with caution as few lake strains were included in the present study. *M. viridis* does not appear to be monophyletic; only single *M. viridis* strains were analyzed in the present study, and it did not appear in the branch corresponding to Cluster 2 in Otsuka et al.'s (1999a, b) phylogenetic tree.

Although Dyble et al. (2002) found that *Cylindrospermopsis raciborskii* strains clustered geographically based on cpcBA–IGS sequence analyses, the same DNA marker failed to differentiate different morphospecies of the same morphospecies in previous and in the present study. Some *Microcystis* strains of various morphospecies isolated from around the world shared the same genotypes, indicating that some species are ubiquitous. As shown in Fig. 6.1, strains of the same location might not be in the same branch, and strains from different location might cluster together with high bootstrap value, suggesting that molecular markers currently used are not capable to identify the geographic origin of *Microcystis* strains. This failure to find geographic structure may be due to taxonomical errors in morphological criteria or due to the existence of many *Microcystis* species in the same water bodies and the same species in different locations.

6.5 Conclusions

Existing morphological classification and molecular phylogenesis demonstrate that morphological criteria alone are not sufficient to identify *Microcystis* species. Morphospecies do not necessarily reveal the genetic background of *Microcystis* species, as the same morphospecies may exhibit different genotypes while the same genotype may represent a variety of morphological types. All morphospecies were polyphyletic, and morphological identification of *Microcystis* species underestimates their genetic diversity. As different strains of the same *Microcystis* morphospecies may have different toxic characters, and multiple morphospecies may occur in the process of growth and decline of *Microcystis* blooms, it is difficult to estimate the transition of species and resultant toxicity by morphological classification. Although different *Microcystis* strains or species were used, and some differences existed in the sequence analyses of cpcBA–IGS and rDNA ITS in different reports, existing studies suggest that molecular marker-based taxonomy of *Microcystis* is feasible. Algal strains with the same genotype should be included in the same species (Otsuka et al. 1999a, b). Accurate identification of *Microcystis* species is needed for water bloom-related studies in reservoirs, as errors in classification may seriously affect subsequent scientific conclusions.

Incorrect identification of *Microcystis* species and their toxicity may result in serious economic losses and even endanger people's health. However, the sequence variation of cpcBA–IGS and rDNA ITS was not large enough to encompass the full intrageneric level. As shown in the present study, several strains shared the same genotypes. However, identical cpcBA–IGS sequences do not indicate that they were genetically homogeneous. Molecular phylogeography and population genetic studies have shown that toxicity of some algal strains correlates with their molecular phylogeny, providing a new solution to detect the toxicity of *Microcystis* strains (Otsuka et al. 1999a, b; Janse et al. 2004). It should be possible for more sensitive molecular markers to differentiate strains of various origin and toxicity and to explore species evolution and dispersal patterns. For example, molecular methods such as amplified fragment length polymorphism (AFLP) and microsatellites (Vos et al. 1995; de Bruin et al. 2003) are potentially suitable to evaluate intrageneric genetic diversity (Janssen et al. 1996). The future use of these or similar methods is therefore strongly recommended.

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Chapter 7
Cyanobacteria and Cyanotoxins in Reservoirs of South China

Lamei Lei, Shaojun Lin, and Ren Hu

Abstract There are few natural lakes in South China, and thus, reservoirs play an important role in supplying drinking water and water for agricultural and industrial uses. Our studies show that Cyanobacteria are widespread in the reservoirs of Guangdong Province; common genera are *Microcystis, Pseudanabaena, Cylindrospermopsis, Merismopedia, Chroococcus, Gloeocapsa, Dactylococcopsis, Anabaena, Raphidiopsis* and *Gloeothece*. In summer and autumn, Cyanobacteria become dominant in eutrophic reservoirs and their relative abundance can reach up to 80%. Many species produce toxins, among which the environmentally persistent, hepatotoxic microcystins (MCs) are most prominent. A survey in 2003 showed that, although microcystins were common in Guangdong Reservoirs, their concentration was low. A survey of six reservoirs in 2004 revealed that relatively high MC concentrations appear in summer and autumn; moreover, MC concentrations increase earlier than in temperate regions and persist longer. The MC content during cyanobacterial blooms ranged from 175.8 to 2478.9 μg/g DW and of *Microcystis* strains ranged from 16.8 to 982.3 μg/g DW. Both implied differentiation between toxin-producing *Microcystis*.

7.1 Introduction

In the past decade, the occurrence of toxic cyanobacterial blooms became more frequent in freshwater bodies worldwide. For example, a toxic *Aphnizomenon flos-aquae* bloom was reported in Saint-Caprais Reservoir in France (Maatouk et al. 2002). In Germany, many water bodies for recreational or drinking water purposes have been studied and Cyanobacteria dominated in the majority of them (Hummert et al. 2001; Frank 2002). *Microycstis* blooms occurred in some ponds and lakes in

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America, leading to livestock deaths and human disease. Hundreds of freshwater bodies in Denmark have been surveyed in the past decade and the results showed that the majority of blooms had microcystin toxins, produced by *Microcystis* spp., *Anabaena* spp., *Aphanizomenon flos-aquae* and *P. agardhii* (Chrorus and Bartram 1999; de Figueiredo et al. 2004; Henriksen 2001). In China, many lakes and reservoirs have suffered from extensive pollution over the last few years, a consequence of rapid economic development. Eutrophication has become a major issue in freshwater management and frequent cyanobacterial blooms occur in major water bodies in summer, e.g. in Lakes Taihu, Caohu and Dianchi (Kong and Gao 2005; Wan et al. 2008).

Cyanobacteria bloom as well in temperate as in tropical freshwaters, but there exist differences in cell density, species composition, vertical distribution, survival and population abundance, depending on environmental conditions such as thermal stratification, irradiance and nutrient availability. In temperate regions, strong changes in seasonal climate result in seasonal succession phytoplankton, and Cyanobacteria dominance is limited to summer by prevailing temperatures. In tropical regions, Cyanobacteria are able to dominate around the year due to abundant sunlight and high water temperatures, and thus, waterblooms may occur more frequently (Whitton and Potts 2000).

According to WHO'S statistics, about 59% waterbloom-forming cyanobacterial species can produce a wide range of toxic compounds, including neurotoxins (such as the saxitoxins) and hepatotoxins (such as the microcystins) (Pitois et al. 2001). *Microcystis*, *Nodularia*, *Planktothrix, Oscillatoria*, *Lyngbya*, *Aphanizomenon*, *Anabaena*, *Cylindrospermopsis,* etc. are the most common toxin-producing genera (Chrorus and Bartram 1999), which also widespread in tropical waters. Microcystins (MCs) are the best-studied hepatotoxins (Chrorus and Bartram 1999). MCs are linked to warm water, and are produced during summer in the Mediterranean Italy (Naselli-Flores et al. 2007), Egypt (Mohamed and Carmichael 2000), Morocco (Oudra et al. 2001), and in Brazil (Znachor et al. 2006).

Guangdong Province is transitional between the tropical and subtropical zones in South China. There are few natural lakes in South China, where reservoirs play an important role in water supply to households, and to agricultural and industrial uses. There are many studies of harmful Cyanobacteria and cyanotoxins concentrated in subtropical and shallow lakes in China, but only a few relate to reservoirs in South China despite increasing cyanobacterial blooms there. In this article, we summarize species diversity, dynamics and toxins of Cyanobacteria, based on a survey covering 11 reservoirs in Guangdong Province during July, September and December 2003 and another six reservoirs – Xinfengjiang, Liuxihe, Feilaixia, Shenzhen, Hedi and Qieyeshi – sampled every 2 months in 2004 (see Lin 2005). MC-LR and MC-RR concentrations were measured using the HPLC-UV method in 2003. Microcystin concentrations in water were assayed using a commercially available enzyme-linked immunosorbent assay (ELISA). Since 2003, we also collected cyanobacterial samples for isolation of the bloom-causing strains and detection of cyanotoxins. In total, we obtained 14 microcystis strains and cellular contents of microcystins.

7.2 Species Composition and Distribution of Cyanobacteria

Eleven reservoirs, in four watersheds (North River watershed, East River watershed, Western Guangdong coastal area, and Pearl River delta), were investigated to determine their cyanobacterial community composition in 2003 and 2004 (Table 7.1, Fig. 7.1). Dominant cyanobacterial species in these reservoirs were *Microcystis* sp., *Pseudanabaena* sp., *Cylindrospermopsis* sp., *Merismopedia* sp., *Chroococcus* sp., *Gloeocapsa* sp., *Dactylococcopsis* sp., *Anabaena* sp., *Raphidiopsis* sp. and *Gloeothece* sp. (Table 7.2)

The dominant species varied with reservoir volume and nutrient concentrations. Cyanobacterial abundance was low in large oligotrophic reservoirs, such as Xinfengjiang Reservoir, dominated by nontoxic species from *Gloeocapsa, Chroococcus* and *Dactylococcopsis*. On the other hand, both cyanobacterial abundance and relative share of Cyanobacteria in total phytoplankton were high in some small eutrophic reservoirs (e.g. Dajingshan and Shenzhen Reservoirs). *Microcystis*, *Pseudanabaena* and *Cylindrospermopsis* dominated in Cyanobacteria and phytoplankton. In July and September, Cyanobacteria were the most abundant phytoplankton, especially in eutrophic reservoirs such as Dajingshan, Qieyeshi, Hedi and Shenzhen; their prevalence remained constant and contributed up to 80% of total phytoplankton abundance. The relative cyanobacterial percentage was below 50% in low nutrient loading reservoirs such as Gaozhou, Feilaixia and Chishijin. The relative cyanobacterial percentage in all surveyed reservoirs decreased significantly when water temperature declined in December, at which time Cyanobacteria accounted for 57.8% of total phytoplankton in Dajingshan Reservoir and below 20% in low nutrient loading reservoirs (Fig. 7.2).

Of the main cyanobacterial species, filamentous *Pseudanabaena*, *Cylindrospermopsis* and *Merismopedia* appear to prefer high water temperatures, high P content and low water transparency. This general pattern was confirmed by their dominance in eutrophic Qieyeshi, Dajingshan, Hedi and Shenzhen reservoirs.

Table 7.1 Characteristics of the 11 investigated reservoirs

Reservoir	Max volume/10^6 m^3	Normal volume/10^6 m^3	Residence time/day	Building year	Trophic state
Xinfenjiang	13,980.0	1000.0	730	1958	Oligoltrophic
Shatian	21.7	14.2	24	1960	Mesotrophic
Chishijin	14.9	12.4	–	1958	Mesotrophic
Feilaixia	1,900.0	440.0	14	1998	Mesotrophic
Gaozhou	1,151.1	841.8	161	1960	Mesotrophic
Hedi	1,144.0	795.0	123	1959	Eutrophic
Liuxihe	378.0	326.0	127	1958	Oligoltrophic
Dajingshan	11.7	10.5	–	1975	Eutrophic
Shenzhen	46.1	35.2	8	1960	Eutrophic
Qieyeshi	13.0	10.2	236	1960	Eutrophic
Dashahe	258.1	156.8	180	1959	Mesotrophic

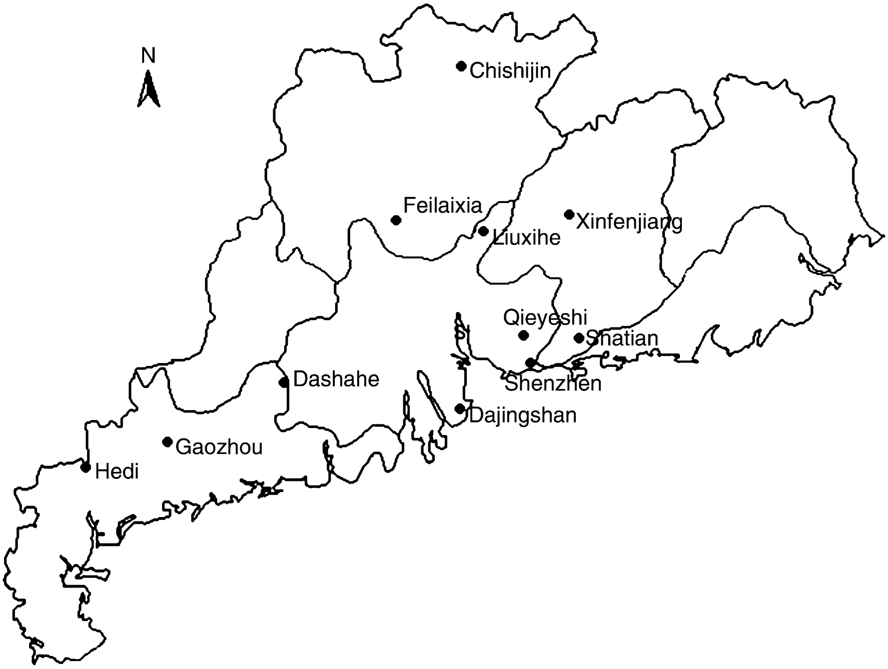

Fig. 7.1 Location of sampling reservoirs in Guangdong Province

Microcystis spp. were commonly observed in all investigated reservoirs, especially in Shatian, Gaozhou and Hedi. A peak density of 2.364×10^6 cells/L appeared at Gaolang station of Hedi Reservoir in September 2003. *Pseudanabaena* dominated in different reservoirs at different times during 2003, with highest density (1.154×10^7 cells/L) in Dajingshan Reservoir in July. High density of *Merismopedia* and *Dactylococcopsis* only occurred in small reservoirs such as Qieyeshi, Shenzhen and Dajingshan. In reservoirs with low trophic state level such as Xinfengjiang, Liuxihe and Chishijin, the dominant Cyanobacteria were more prevalent species, such as *Gloeocapsa, Gloeothece* and *Aphanocapsa*. *Cylindrospermopsis* sp. is a typical tropical species, common in lakes and reservoirs in Guangdong Province. High densities of this were found in Dajingshan Reservoir, and even a slight bloom was observed in Hedi and Gaozhou reservoirs (Table 7.2).

Compared with a previous study (Hu et al. 2002), a new trend observed in 2003 was that filamentous Cyanobacteria were more likely to be dominant than colonial types in eutrophic reservoirs. Most of the studied reservoirs were meso-eutrophic, and five of them were not dominated by Cyanobacteria. Moreover, cyanobacterial blooms occurred in all four eutrophic reservoirs (Hedi, Dajingshan, Shenzhen and Qiyeshi) during the study period (Fig. 7.2).

Table 7.2 List of cyanobacterial species from the investigated reservoirs in Guangdong Province

Species\Reservoir	Liuxihe	Xinfenjiang	Feilaixia	Shenzhen	Hedi	Qieyeshi	Dajingshan	Gaozhou	Dashahe	Chishijin	Shatian
Microcystis flos-aquae	+++	+	+	+	+++	+	+	+			+++
M. aeruginosa	+	+	+	+	++	+	+	+	+	+	+++
M. wesenbergii	+	+	+	+	+	+	+	+	+	+	++
M. incerta	+	+	+		+++	+	+	+			+
Synechococcus sp.	+										
Nodularia sp.				+							
Aphanocapsa sp.	+	+	+		+	+		+			
Aphanothece sp.					+						
Coelosphaerium sp.			+		+	+		+		+	
Pseudanabaena sp.	+	+	+	+++	+	+++	+	++	+	+++	
Cylindrospermopsis sp.		+	+		+++	++	+++	+	+	+	+
Raphidiopsis sp.	+					+					
Dactylococcopsis sp.	+	+	+	+	+	+	+	+	+	+	+
Merismopedia sp.	+	+	+	+++	+	+	+		+	+	+
Anabaena sp.	+	+	+	+		+	+	+	+	+	+
Gloeothece sp.	+				+			+	+	+	+
Gloeocapsa sp.	+	+	+	+	+	+	+	+	+	+	+
Chroococcus sp.		+	+	+	+	+	+	+	+	+	+
Phormidium sp.	+		+	+		+					
Lyngbya sp.	+	+	+	+	+		+			+	
Oscillatoria sp.		+	+	+	+	+	+	+	+	+	
Spirulina sp.	+			+	+	+	+			+	
Limnothrix sp.		+	+	+	+	+					
Planktolyngbya sp.		+	+	+	+	+					
Snowella sp.					+	+					

Note: +++ many; ++ medium; + few

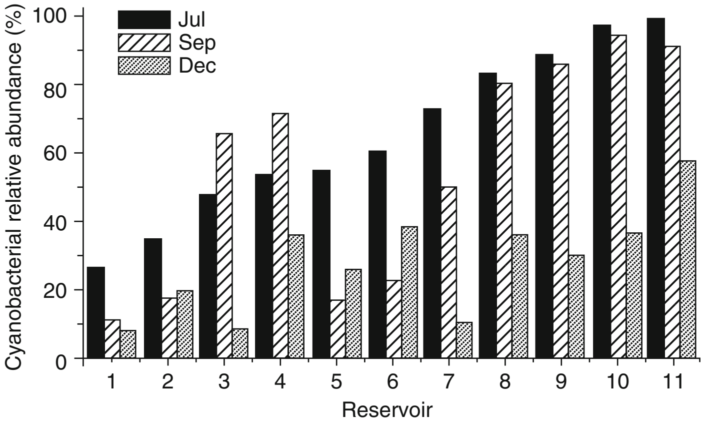

Fig. 7.2 Percentage of blue-green algae in total phytoplankton abundance in reservoirs in 2003. *1* Gaozhou, *2* Feilaixia, *3* Chishijin, *4* Shatian, *5* Xinfenjiang, *6* Dashahe, *7* Liuxihe, *8* Shenzhen, *9* Hedi, *10* Qieyeshi, *11* Dajingshan

7.3 Microcystin

7.3.1 Spatial Distribution of Microcystin

Thirty-six samples from the reservoirs were scrutinized with the HPLC-UV method. The MC-LR and MC-RR concentration for each sample is listed in Table 7.3. MCs were widely distributed, but toxin composition and concentration varied between reservoirs. MCs were found in seven reservoirs and no toxin was detected in samples from Gaozhou, Shatian, Dajingshan and Xinfengjiang reservoirs. MC-RR was the more common MC variant found, with higher MC-RR than MC-LR in positive samples. Table 7.3 shows the seasonal variation of MCs (LR and RR) in reservoirs. MCs were more common with high concentration in autumn (September). Autumn samples from six reservoirs were all positive for MC detection, with the highest concentration (0.365 μg/L) found in Hedi Reservoir (Table 7.3). Four reservoir samples collected in summer (July) were positive for MCs, and the maximum concentration (0.104 μg/L) occurred in Tangxi Reservoir. MCs were only occasionally found in winter (December), in low concentrations (0.038 μg/L, Dashahe Reservoir) (Table 7.3).

The percentage of positive MC detections in this study was 33.3%, similar with the 20.8% positive percentage in Spain (Aboal and Puig 2005), but lower than that in Portugal (60%, Vasconcelos 1994), France (70%, Vezie et al. 1997) and Brazil (90%, Znachor et al. 2006). In a study by Znachor et al. (2006), all 18 Brazil reservoirs sampled experienced a cyanobacterial bloom, and some samples contained MC as high as 100 μg/L. Another study in Lake Biwa, Japan, during 1998–2000 also reported an MC concentration as high as 22 μg/L (Ozawa et al.

Table 7.3 Microcystin concentration detected with HPLC in the reservoirs in 2003 (10^{-1} μg/L)

MCs	Liuxihe			Xinfenjiang			Feilaixia			Shenzhen			Hedi			Dashahe			Chishijin			Tangxi		
	J	S	D	J	S	D	J	S	D	J	S	D	J	S	D	J	S	D	J	S	D	J	S	D
MCRR	0	0.19	0	0.01	0	0	0	0	0	0	0.09	0	0	2.83	0	0.09	0.78	0.38	0.1	0.21	0	0.78	0.41	0
MCLR	0	0.06	0	0	0	0	0	0.02	0	0	0.15	0	0	0.83	0	0.09	0	0	0	0	0	0.26	0	0

Note: *J* July; *S* September; *D* December

2005). In China, the highest toxin concentration, 15.6 μg/L, was also reported in summer Taihu samples (Xu et al. 2008a).

MC concentrations in Guangdong Reservoirs in the 2003 survey were not high compared with those reported in other countries and regions, and none of the samples violated the Chinese national standard for drinking water limit (1 μg/L). In terms of both bloom occurrence and toxin concentration, toxic cyanobacterial blooms are not a serious problem in most Guangdong Reservoirs. However, the eutrophic Tangxi Reservoir has a long history of cyanobacterial blooms since 1997, and our study showed that this reservoir was dominated with *Microcystis* and a high concentration of MC (Table 7.2). Tangxi reservoir is an important drinking water storage for Eastern Guangdong; therefore, effective management of cyanobacterial blooms is necessary to guarantee safe drinking water supply in the future.

7.3.2 Seasonal Variation of MCs in Six Reservoirs

Water samples were collected every 2 months in 2004, and microcystin concentrations were assayed using enzyme-linked immunosorbent assay (ELISA). In oligotrophic or mesotrophic reservoirs such as Xinfengjiang, Liuxihe and Feilaixia reservoirs, microcystin concentrations were only about 0.1 μg/L. In eutrophic reservoirs such as Hedi, Shenzhen and Qieyeshi reservoirs, microcystin

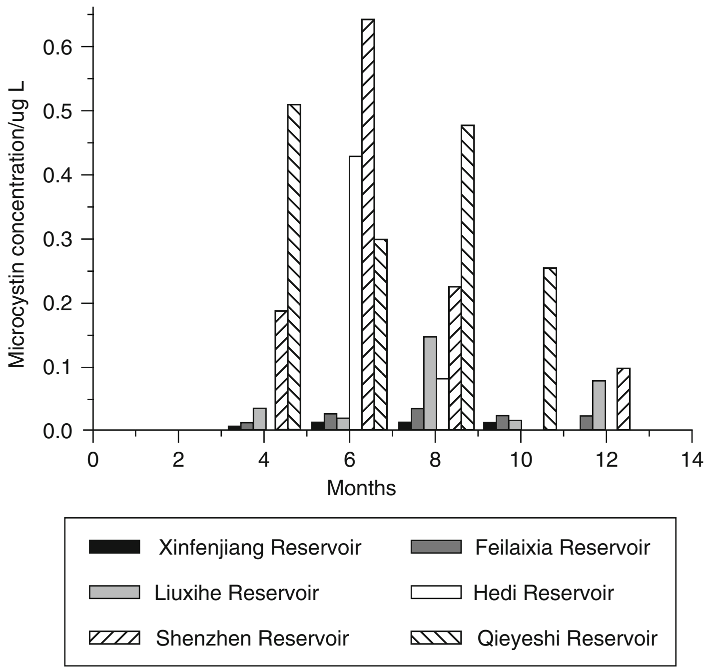

Fig. 7.3 Microcystin concentration in the reservoirs of Guangdong Province in 2004

concentrations were relatively high and the sample collected from Shenzhen Reservoir in June had maximal microcystin concentration, 0.641 μg/L (Fig. 7.3). Overall, microcystins were commonly observed from April to October, especially in Qieyeshi Reservoir, where water samples collected during this period showed microcystins ranging from 0.255 to 0.511 μg/L. Microcystin had a concentration of 0.42 μg/L in Hedi Reservoir in July 2004, but was below detection in the other months. The low microcystin concentrations reported here indicated a low health risk to human populations when reservoirs in Guangdong Province are used as sources of drinking water.

Microcystin concentrations in six reservoirs were noticeably higher in 2004 than in 2003, especially in the eutrophic reservoirs: Hedi, Shenzhen and Qieyeshi. When microcystins were detected from April to October, high cyanobacterial abundance was observed. Microcystin concentration was positively correlated with cyanobacterial abundance in three reservoirs (Fig. 7.4), with a particularly high R^2 value in Shenzhen Reservoir ($R^2 = 0.859$, $p < 0.05$). The maxima of microcystin (0.641 μg/L) in Shenzhen Reservoir in June corresponded to maxima of cyanobacterial abundance (Fig. 7.4c). Other field studies also reported significant correlations between microcystin and Cyanobacteria abundance or biomass (Oh et al. 2001; Ozawa et al. 2005; Xu et al. 2008a, b). However, the timing for increased microcystin concentration came earlier than in temperate regions, and this may have been caused by higher water temperatures in tropical and subtropical regions.

7.4 Microcystins Produced by Cyanobacteria

7.4.1 Microcystins Produced by Cyanobacterial Blooms

After the first heavy bloom of *Microcystis* observed in Tangxi Reservoir in 1999, cyanobacterial blooms were consistently reported in Guangdong Reservoirs too. Nanping Reservoir in Zhuhai City provides drinking water to Macao, and waterblooms dominated by *Microcystis* have occurred in early June since 2006. *Microcystis* and *Anabaena* bloomed in the large Nanshui and Gaozhou Reservoirs in 2009 from late spring to early summer.

Cyanobacterial cells were collected during the bloom. MC was detected quantitatively by HPLC with MCRR and MCLR (Table 7.4). The bloom-forming genera were *Microcystis* and *Anabaena* but the blooms were absolutely dominated by *Microcystis* spp. in Dalingtou, Guandong and Nanping Reservoirs, by *Anabaena* spp. in Nanshui Reservoir, and by both genera in Dashahe and Gaozhou Reservoirs.

HPLC analysis revealed varied MC contents in the bloom samples. MCLR and MCRR occurred in the samples from Gaozhou, Dashahe, Guandong and Nanping Reservoirs, but no MCs was detected in the samples from Dalingtou and Nanshui Reservoirs. This indicated that the Cyanobacteria of the two reservoirs did not

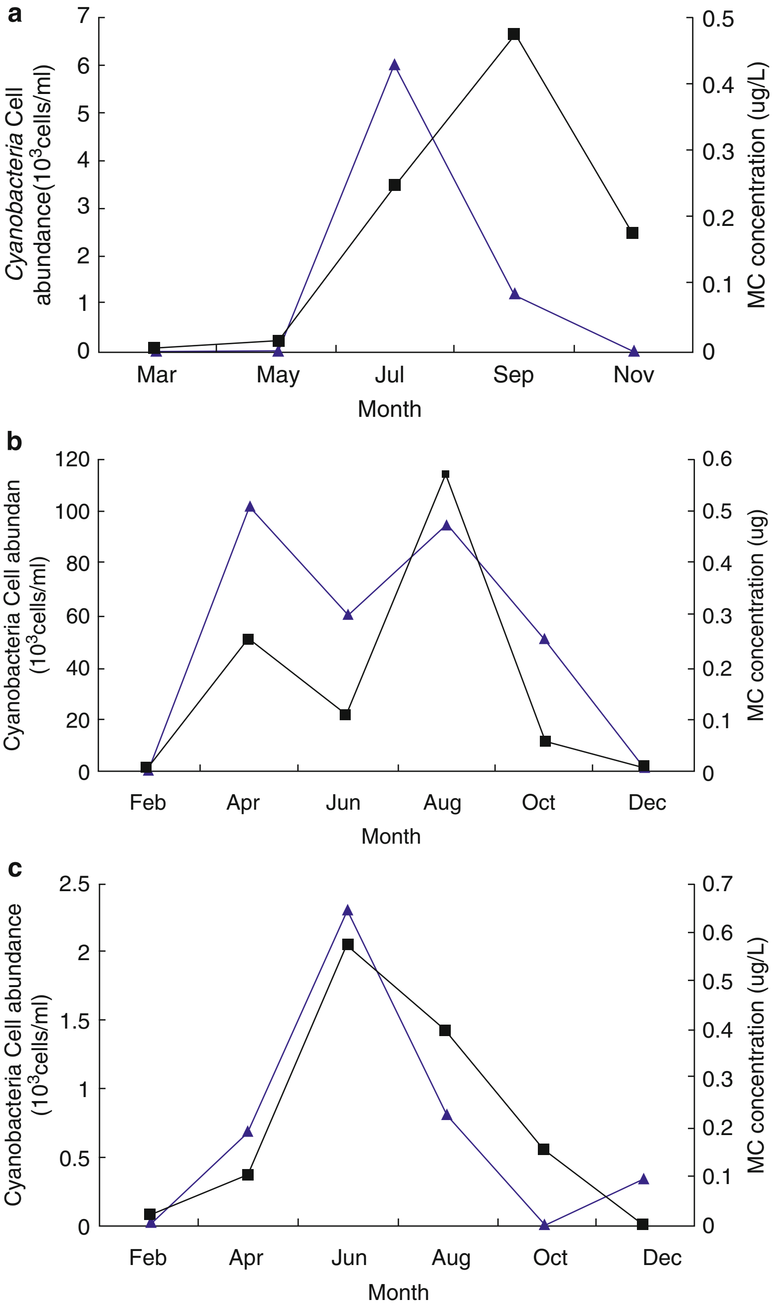

Fig. 7.4 Seasonal variation of cyanobacterial cell abundance (■) and MC concentration (◆) in three eutrophic reservoirs: (**a**) Hedi Reservoir, (**b**) Qieyeshi Reservoir, (**c**) Shenzhen Reservoir

Table 7.4 Quantification of total MCs by HPLC in cyanobacterial blooms

	Sample source	Date of sampling	Microcystin concentration (μg/g DW)			Dominant genera of blooms
			MCRR	MCLR	Total	
1	Dashahe	2008	131.8	44	175.8	*Anabaena, Microcystis*
2	Dashahe	2009	1,709.1	769.8	2,478.9	*Microcystis*
3	Dalingtou	2006	–	–	–	*Microcystis*
4	Guandong	2006	141.1	63	204.1	*Microcystis*
5	Gaozhou	2009	209	89.6	298.6	*Anabaena, Microcystis*
6	Gaozhou	2009	142.7	70.4	213.1	*Microcystis*
7	Gaozhou	2009	345.4	–	345.4	*Anabaena*
8	Nanshui	2009	–	–	–	*Anabaena*
9	Nanping	2007	75.8	82.1	157.9	*Microcystis*

– Not detected

produce toxins or just produce other MCs isotypes. The MC concentration of toxic bloom samples ranged from 175.8 to 2478.9 μg/g. The sample from Dashahe Reservoir was highest, up to 2478.9 μg/g, while other samples were relatively low and total MCs content were below 400 μg/g. The MC content was obviously high compared with that reported for *Microcystis* blooms from two Moroccan reservoirs (64.4 and 9.9 μg/g, Douma et al. 2010). Except for the sample from Dashahe Reservoir, however, the MC content was relatively low compared with bloom samples from the Moroccan Takerkoust Lake (8.8 mg/g; Oudra et al. 2001), various Portuguese reservoirs (1–7.1 mg/g; Vasconcelos 2001), and the Spanish Santillana Reservoir (13.5 mg/g). Our results indicate that there is a glaring difference in toxin production between *Microcystis* strains. The situation may occur when the MC-producing strains are not dominant, and this may lead to variation on MC content of bloom samples that are equally dominated by *Microcystis* species.

7.4.2 *Microcystins Produced by Microcystis Strains*

Since 2003, we have been continuously isolating *Microcystis* colonies from different reservoirs and in all, 14 *Microcystis* strains were successfully cultured. The isolated cells were incubated at 25 ± 1°C under continuous light at a light intensity of approximately 60 μmol/m^2/s. After incubation, the cells were harvested by centrifugation and MC content was analyzed quantitatively by HPLC with MC-RR and MC-LR as standard and the results were shown in Table 7.5.

MC content of 11 *Microcystis* strains ranged from 16.8 to 982.3 μg/g dry weight. Three strains of SK11, SK12 and SK20 contained no MCs. Both MC-LR and MC-RR were present in strains of SK9, SK13 and SK16. MC-RR was the dominant component of MCs in the strains. SK33 was the strain with the highest toxin

Table 7.5 Microcystin quantification (μg/g) of the *Microcystis* strains detected with HPLC

Strain	Species	Origin of strain	Microcystin concentration (μg/g DW)		
			MCRR	MCLR	Total
SK6	*M. aeruginosa*	Dajingshan Reservoir	122	–	122
SK9	*M. aeruginosa*	Tangxi Reservoir	123.6	19.4	143
SK10	*M. aeruginosa*	Tangxi Reservoir	76.2	–	76.2
SK11	*M. flos-aquae*	Dajingshan Reservoir	–	–	–
SK12	*M. aeruginosa*	Xinfenjiang Reservoir	–	–	–
SK13	*M. wesenbergii*	Xinfenjiang Reservoir	477.2	122.7	599.9
SK14	*M. aeruginosa*	Liuxihe Reservoir	16.8	–	16.8
SK15	*M. wesenbergii*	Feilaixia Reservoir	–	45.2	45.2
SK16	*M. flos-aquae*	Feilaixia Reservoir	479.2	115.9	595.2
SK17	*M. flos-aquae*	Xiangang Reservoir	27.2	–	27.2
SK20	*M. flos-aquae*	Feilaixia Reservoir	–	–	–
SK25	*M. aeruginosa*	Xiangang Reservoir	88.7	–	88.7
SK27	*M. aeruginosa*	Gaozhou Reservoir	145.3	–	145.3
SK29	*M. aeruginosa*	Nanping Reservoir	321.4	–	321.4
SK33	*M. aeruginosa*	Dajingshan Reservoir	982.3	–	982.3

concentration (982.3 μg/g), followed by SK13 (599.9 μg/g) and SK16 (595.2 μg/g), and the average toxin concentration of the 14 strains was 225.94 μg/g.

In general, the average toxin concentration in *Microcystis* strains from the reservoirs of Guangdong Province was comparable to that from Moroccan waters (279.4 μg/g, Oudra et al. 2002). However, the toxin content for any individual strain was obviously lower than that from Moroccan waters (Oudra et al. 2002), Biwa Lake of Japan (Ozawa et al. 2005) and Portuguese waters (Saker et al. 2005). This may be one reason explaining very low MCs concentration in the reservoirs of Guangdong Province. It has been concluded that *Microcystis wesenbergii* strains isolated from seven Chinese water bodies did not produce microcystins (Xu et al. 2008a, b), but two strains of *M. wesenbergii* (i.e. SK13 and SK15 from our reservoirs) can produce toxins, and strain SK13 had a relatively high toxin content. *M. wesenbergii* Mic-Tg1-99 from Moroccan water also produces three MC isotopes, and the toxin content was as high as 1,844 μg/g (Oudra et al. 2002). Therefore, conclusions about non-microcystin-producing *M. wesenbergii* in Chinese waters must be questioned.

7.5 Conclusions

Our studies show that Cyanobacteria are widespread in the reservoirs of Guangdong Province, and the common genera are *Microcystis, Pseudanabaena, Cylindrospermopsis, Merismopedia, Chroococcus, Gloeocapsa, Dactylococcopsis, Anabaena, Raphidiopsis* and *Gloeothece*. In summer and autumn, Cyanobacteria become dominant in eutrophic reservoirs (e.g. Shenzhen, Hedi and Dajingshan reservoirs), and their relative abundance can reach up to 80%. A survey in 2003 showed that MCs were common, but their concentration was very low and below the low limit recommended by WHO and the Chinese national standard for safe drinking water (1 μg/L). A survey of six reservoirs in 2004 revealed that relatively high MC concentrations appear in summer and autumn; moreover, the time at which MC concentration quickly increases is earlier than that in temperate regions and persists longer. The MC content during cyanobacterial blooms ranged from 175.8 to 2,478.9 μg/g DW and of *Microcystis* strains ranged from 16.8 to 982.3 μg/g DW. Both implied differentiation between toxin-producing *Microcystis*.

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Chapter 8
A *Peridinium* Bloom in a Large Narrow Impoundment, Huanglongdai Reservoir, Southern China

Mengxu Zhao and Bo-Ping Han

Abstract In the late spring of 2005, a *Peridinium bipes* bloom was observed in Huanglongdai, a large, narrow reservoir in southern China. The bloom began at the mouth of a small river. Upon crossing a village, this collected all its untreated sewage, including the nutrients needed for the growth of *Peridinium*. The *Peridinium* population then dispersed into the main water body, forming a narrow strip of brown, and finally accumulated in front of the dam, where the highest density was found. The visible bloom lasted for about 4 weeks, with *Peridinium* constituting over 90% of total phytoplankton biomass, and *Microcystis aeruginosa* second in line of abundance. On April 26, a heavy precipitation resulted in a significant decrease in water transparency. On May 20, the water temperature (WT) was 3°C higher than on May 5. The change in hydrology due to the heavy precipitation may have been the direct factor triggering the decay of the bloom, and it was accelerated by the dramatic increase in water temperature. In comparison with other reservoirs, the present *Peridinium* bloom shares several common features with others, namely nutrient supply, water temperature, and hydrodynamics, all of which play major roles in the initiation and decay of such blooms.

8.1 Introduction

Dinoflagellate blooms are regularly found in seawater, but are rather uncommon in freshwater. An early *Peridinium* bloom was described by Komarovsky (1951), and thereafter, similar blooms were reported in several lakes and reservoirs. Horne et al.

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(1971) reported a red tide caused by *Peridinium pernardii* in Clear Lake in California. A dense population was confined to the periphery of the lake; it persisted for about 5 weeks, from the end of March to the middle of May. The decline of the bloom was paralleled by an increase in *Aphanizomenon flos-aquae*. Extensive blooms of *P. gatunense* are rather frequent in Lake Kinneret, Israel (Berman and Rodhe 1971; Pollingher 1986; Zohary et al. 1998; Viner-Mozzini et al. 2003; Zohary 2004). Herrgesell et al. (1976) found *P. pernardii* blooms in a reservoir (Lake Berryessa) in California. They developed at the mouth of tributary streams after periods of high runoff and lasted for about 3 weeks. It was suggested that allochthonous inputs were necessary to induce such blooms. As reservoirs are man-made water bodies, dinoflagellate blooms in such water bodies may be different from those in natural lakes. The dinoflagellate bloom in the Shorenji Reservoir reported by Yamada et al. (1998) followed a similar pattern as that in Berryessa Reservoir. Fukuju et al. (1998) examined statistic characteristics of dinoflagellate blooms in 39 reservoirs in Japan and suggested that the blooming can be predicted by latitude, morphology, hydrodynamics, and nutrient loading of the reservoirs. In comparison to high latitudes (more than 25°N), only few blooms have occurred at low latitudes (<25°N), although *P. bipes* blooms were observed in reservoirs in China, e.g., in Feitsui Reservoir (Wu et al. 1998), Techi Reservoir (Chang et al. 2004), and Manwan Reservoir (Wang et al. 2004). Here, we report another case of a dinoflagellate bloom caused by *P. bipes* in mesotrophic Huanglongdai Reservoir, southern China. The reservoir is a large, narrow water body, located near the Tropic of Cancer. It was constructed in 1975 by damming the Fentian River, a tributary of the Liuxihe River. Reservoir water quality strongly affects the drinking-water supply to Guangzhou City, situated downstream. The reservoir has a watershed area of 92.3 km^2 with a maximum storage of 9.458×10^7 m^3. Its hydraulic residence time is about 1 year, but fluctuates seasonally and becomes much shorter in wet seasons, from April to September. It looks like a dragon with a long and narrow body, with the dam just near the head. Blooms had been noted every spring from March to May for years, yet were first observed quantitatively in 2005. The visible bloom started at the mouth of a small river crossing Lianqun village (about 600 inhabitants), and then dispersed into the main water body near the dam. It persisted about 4 weeks, and disappeared at the beginning of May with the wet season coming. Here, we explore whether a common mechanism underlies this dinoflagellate bloom and the ones reported from other reservoirs.

8.2 Materials and Methods

Water samples for phytoplankton and nutrient analysis were collected by a 5-L water sampler at two sampling stations (Fig. 8.1). Station 1 was just in front of the dam, and Station 2 was at the mouth of the small river, where the bloom had started.

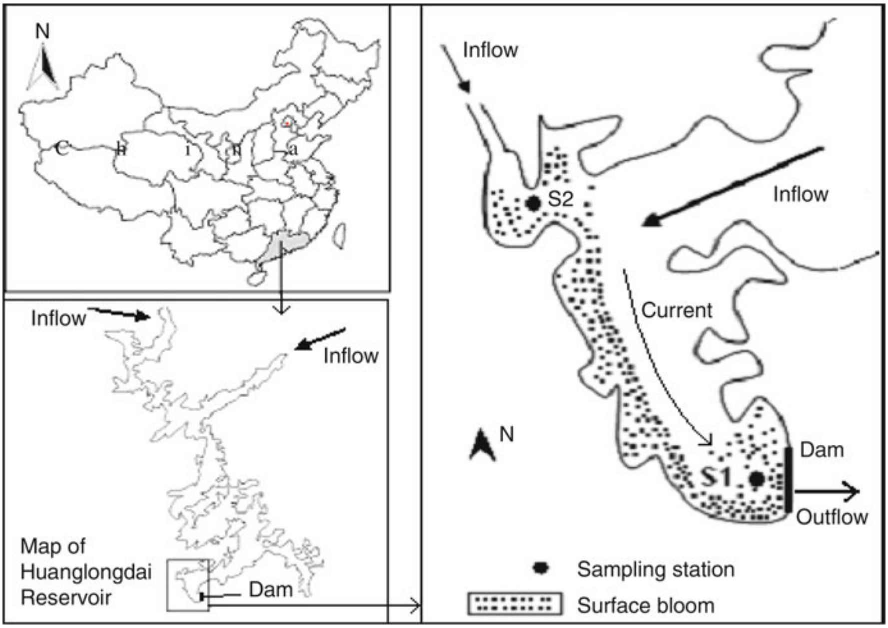

Fig. 8.1 Map of the Huanglongdai Reservoir, with two sampling stations. The inset window shows *Peridinium* bloom near the dam

During the bloom period, the samples were collected either weekly or biweekly, i.e., on April 15, 21, 28, May 5, 20, and June 6. All the samplings and in situ measurement of physical variables such as temperature and water transparency were conducted from 10:00 to 11:30 a.m.

Water temperature (WT) and water transparency were measured by a sensor from Yellow Springs Instrument (YSI85) and Secchi Disk (SD), respectively. For nutrient analysis, water was sampled from the surface (1 m) at the two stations. Analysis of nutrients such as total nitrogen (TN) and total phosphorus (TP) was according to Chinese National Standards for eutrophication (Jin and Tu 1990; Han et al. 2003). For phytoplankton at Station 2, only surface water was sampled, while at Station 1, the water was sampled vertically at depths of 1, 5, and 10 m. The samples were fixed in situ in 5% formalin. Phytoplankton were counted and measured under a microscope and results were expressed with an accuracy of 10% of total cell abundance and biomass. Fresh weight was calculated using specific biovolumes obtained by geometrical approximation, assuming a mean cell density of 1 g/cm^3. Chlorophyll *a* was measured after overnight extraction in 90% acetone, following a modified method by Lin et al. (2005).

8.3 Results

8.3.1 Water Temperature, Transparency, and Chlorophyll a

Between April 15 and June 6, surface water temperature ranged from 23°C to 31°C. From April 15 to May 5, mean water temperature was about 24 ± 1°C, then dramatically increased on May 20 (28°C) and June 6 (31.1°C). The water column at Station 1 was stratified with a thermocline at depth of ~8 m. Water transparency varied markedly during the sampling period (Table 8.1), being highest (1 m) at the first sampling on April 15 and lowest (0.5 m) at the last sampling on June 6. It decreased with the decay of the bloom from April to June. No significant difference in Secchi depth was found between the two sampling stations. Chlorophyll *a* concentration had a similar variation to transparency. It reached its maximum (12.72 mg/m^3) on April 21 and suddenly dropped (1.36 mg/m^3) on April 28.

8.3.2 Nutrient Concentrations

Figure 8.2 presents concentrations of total phosphorus (TP) and total nitrogen (TN). TP ranged from 0.035 to 0.078 mg/L, and tended to increase from April to June. TN concentration varied from 0.529 to 1.235 mg/L. Except on 21 April, TP and TN concentrations were slightly higher at Station 2 than at Station 1.

8.3.3 Visual Observation of Blooms

On April 14, a surface bloom was observed at Station 2 and second, weak one at Station 1. The *Peridinium* bloom gave a yellow-brown color to the water, and an algal scum covered about 30% of the water surface. On April 15, more intense surface blooms appeared at both stations. A narrow strip of brown was visible from Station 2 to 1, and extended about 10–15 m out from the reservoir edge along the eastern shore of the lacustrine zone. The visible surface bloom lasted for at least 4 weeks. It peaked on April 21, and covered about 40% of the water surface of the lacustrine zone. On April 28, the bloom split into several narrow strips and then patchily distributed on the water surface. It finally disappeared on May 5.

Table 8.1 Water temperature (WT) and transparency (Secchi depth), chlorophyll *a* concentration during the blooming near the dam (Station 1)

Date	15 Apr	21 Apr	28 Apr	5 May	20 May	6 June
WT (°C)	23.5	24	23.5	25	28	31.1
SD (m)	1.1	1.1	0.65	0.45	0.45	0.5
Chl.*a* (mg/m^3)	11.48	12.72	1.36	1.25	2.72	2.79

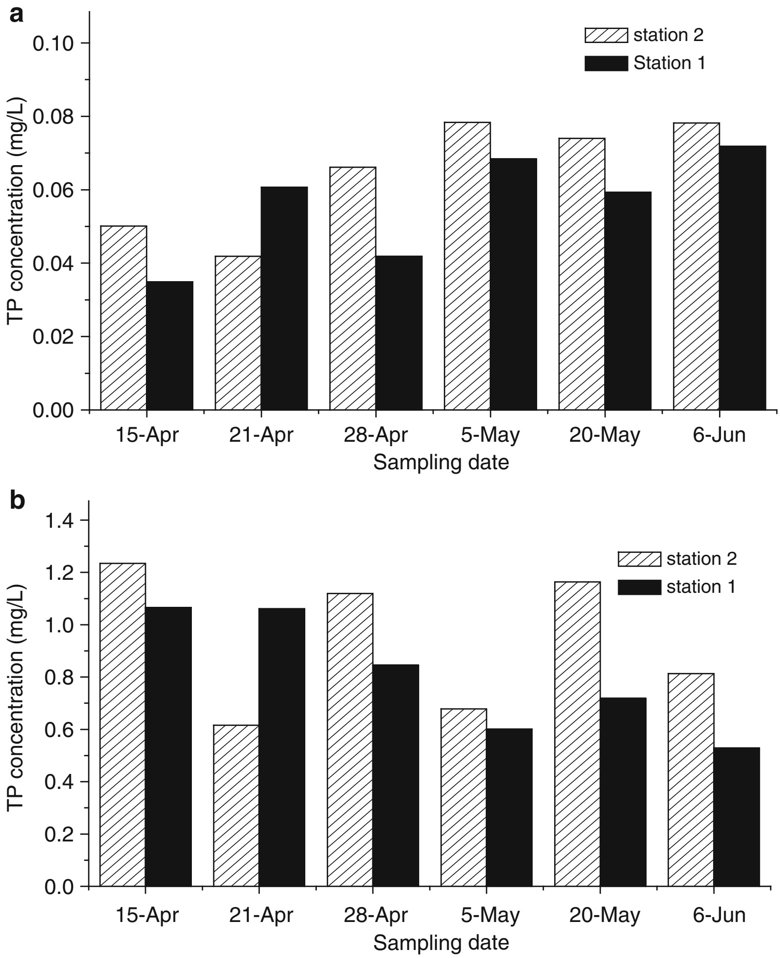

Fig. 8.2 Dynamics of total phosphorus and total nitrogen concentrations during the sampling period

Beginning on April 26, heavy rainfall, lasting for several days, led to a rise in water turbidity. On May 5 and 20, the surface water turned yellow, but this high turbidity was caused by mud inflow.

8.3.4 Phytoplankton Abundance and Biomass

Total phytoplankton abundance ranged from 119 to 1,621 cells/mL at Station 2 and from 124 to 16,649 cells/mL at Station 1 (Fig. 8.3a). The highest abundance occurred at Station 1 on April 21, with the phytoplankton community dominated

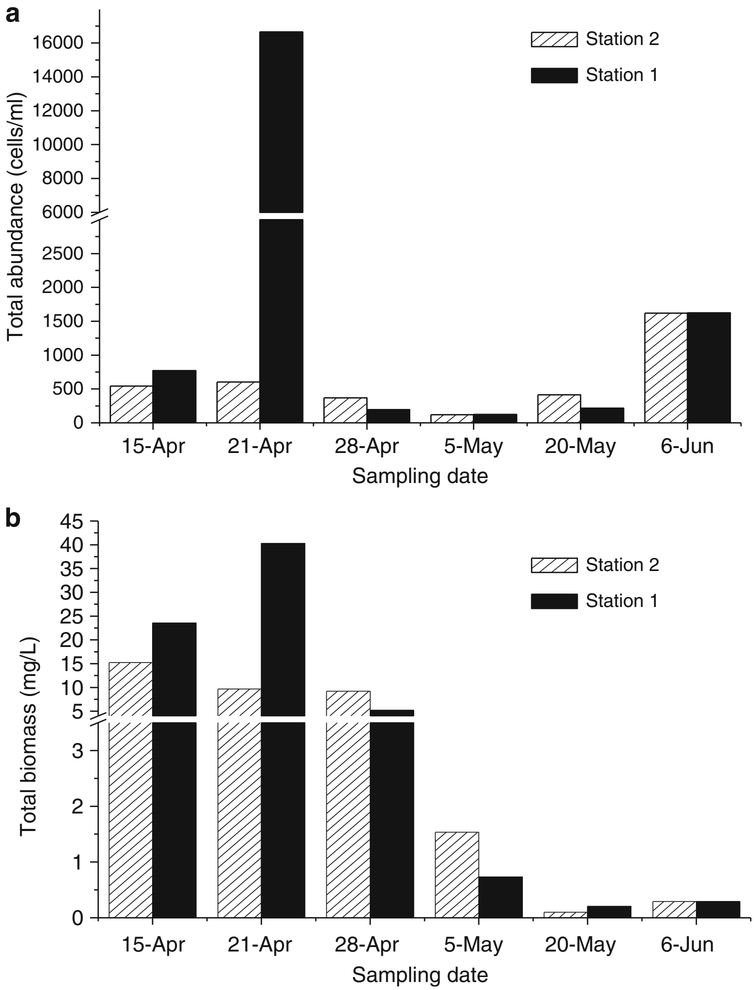

Fig. 8.3 Total phytoplankton abundance (**a**) and biomass (**b**) at the subsurface (1 m) at two stations during the sampling period

by *P. bipes* and *Microcystis aeruginosa*. *M. aeruginosa* was abundant, particularly on April 21, when it contributed 90% of phytoplankton abundance. Thus, the yellow-brown bloom was produced by both. During the decay of the bloom, *Melosira ambigua* and *Gloeothece linearis* suddenly increased. *G. linearis* constituted more than 90% of total phytoplankton abundance on June 6 when the bloom had entirely disappeared from the surface water.

Figure 8.3b illustrates the spatial and temporal distribution of total phytoplankton biomass during the sampling period. Total biomass ranged from 0.101 to 15.230 mg/L at Station 2 and from 0.204 to 40.294 mg/L at Station 1. It was higher at Station 1 than at Station 2 on April 15 and 21, but became inverse in the late life

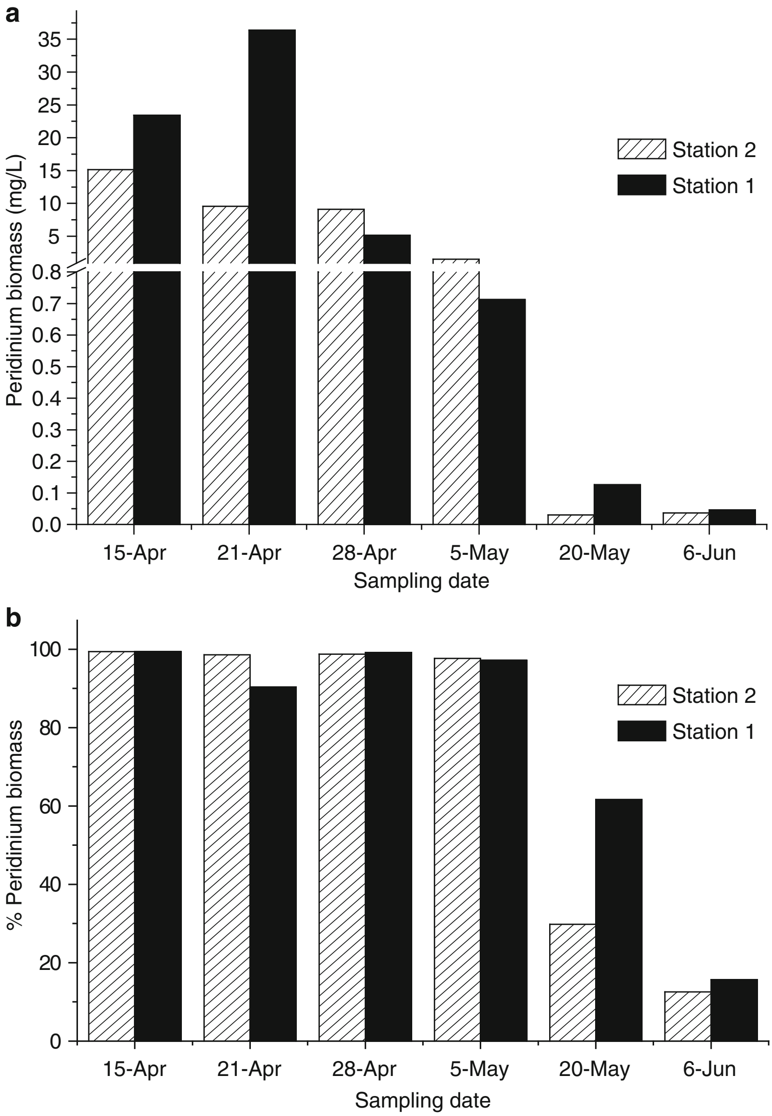

Fig. 8.4 *Peridinium* biomass (**a**) and its contribution in total phytoplankton biomass (**b**) during the sampling period

of the bloom. When the bloom began to decay on May 5, phytoplankton biomass dramatically decreased.

Figure 8.4 presents the *Peridinium* biomass during the sampling period, ranging from 0.046 to 36.395 mg/L at Station 1, reaching its maximum on April 21. Biomass ranged from 0.030 to 15.138 mg/L at Station 2, and its maximum occurred on April 15. At its peak (on April 15 and 21), the *Peridinium* biomass was higher at

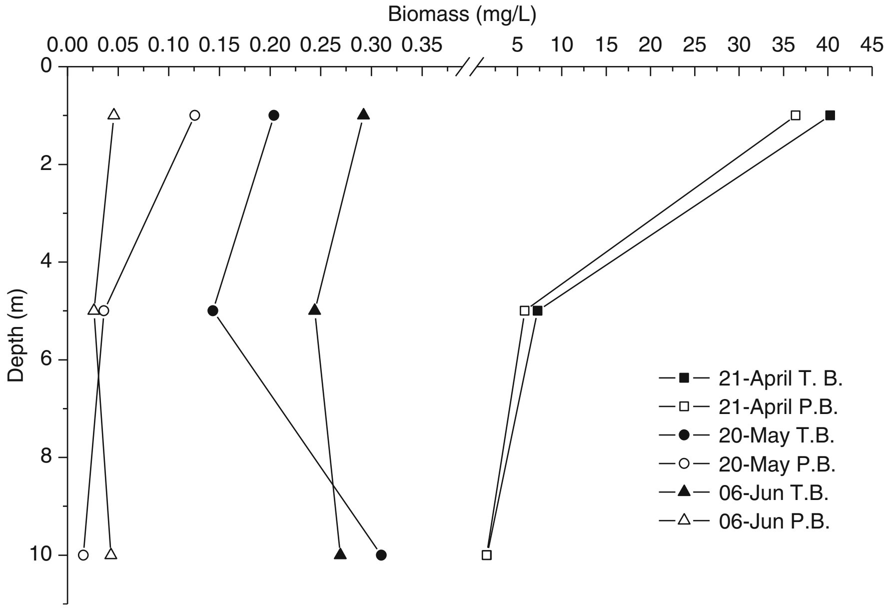

Fig. 8.5 Vertical distributions of total phytoplankton biomass (T.B., *filled mark*) and *Peridinium* biomass (P.B., *open mark*) at Station 1

Station 1 than at Station 2, but it also decayed more rapidly at Station 1. As *P. bipes* has a large-sized cell (~50 μm in diameter), much larger than *M. aeruginosa*, *M. ambigua,* and *G. linearis*, it dominated total phytoplankton biomass, over 90% from April 15 to May 5 at both Stations 1 and 2. On May 20, biomass dominance of *P. bipes*, *G. linearis*, and *M. ambigua* at Station 2 was 29.8%, 19.2%, and 13.1%, respectively. At Station 1, *P. bipes* and *Botryococcus braunii* made up 61.6% and 26.5% of total phytoplankton biomass, respectively. On June 6, *G. linearis* contributed more than 70% to the total phytoplankton biomass at both sampling stations.

Phytoplankton was vertically sampled at Station 1 on April 21, May 20, and June 6. Figure 8.5 shows the vertical distribution of phytoplankton biomass and *Peridinium* biomass at depths of 1, 5, and 10 m. On April 21, phytoplankton biomass at the surface (1 m) was 40.294 mg/L, much higher than at 5 and 10 m (7.272 and 1.560 mg/L, respectively), and was mainly composed of *Peridinium*. On May 20 and June 6, *Peridinium* biomass was uniform in the water column. Toward the end of the bloom, *Peridinium* cysts were observed at a depth of 10 m.

8.4 Discussion

Huanglongdai Reservoir is a mesotrophic water body, and its nutrient loading mainly comes from runoff in the catchment. Besides the reservoir, there are two villages (Lianqun and Lianxin) with about 1,500 inhabitants in total. The bloom first occurred at a shallow bay near Lianqun village, and it is assumed that untreated domestic sewage from this village provided the nutrients for the bloom. At its early stage, the density of *P. bipes* was highest at the mouth of the small river. The brown stripe of *P. bipes* dispersed into the main water body with water currents, and *Peridinium* finally accumulated in front of the dam. *P. bipes* always distributed at the eastern edge of the reservoir, and water currents move in the same way. A comparison with the blooms observed in other reservoirs suggests that a common mechanism underlies all blooms. Although the dominant species may be different, all blooms first originated in shallow zones and then dispersed along the shore. When water temperature is higher than 25°C, blooms decay rapidly. The *Peridinium* bloom in a California reservoir, reported by Herrgesell et al. (1976), was first formed at the mouths of tributary streams, and the nutrient input from the streams was postulated to stimulate the development of the bloom. Likewise, red tides of *Peridinium* have been frequently observed at the head of the Ishitegawa Reservoir since 1975 (Kawabata and Hirano 1995).

According to Fukuju et al. (1998), freshwater red tides occur easily in reservoirs with trophic state from oligotrophy to mesotrophy. Usually, there is a decreasing gradient in nutrient concentrations from the riverine to the lacustrine zone (Straskraba and Tundisi 1999). Most reservoirs are oligotrophic in their lacustrine zone when the riverine zone is mesotrophic. This is the case of the Huanglongdai Reservoir. Therefore, in such oligo-mesotrophic reservoirs, *Peridinium* can acquire adequate TN and TP at the shallow end of the tributary river for cell growth and initial accumulation. *Peridinium* blooms have also been widely observed in mesotrophic seawater, and the bloom-causing species appear to be adapted to a relatively low nutrient concentration. This contrasts with cyanobacteria blooms that usually form in eutrophic water. Although *Peridinium* blooms commonly associate with cyanobacteria, e.g., with *M. aeruginosa* in Feitsui (Wu et al. 1998) and Huanglongdai Reservoirs, and with *Aphanizomenon* in Clear Lake, *Peridinium* can benefit from utilization of organic nitrogen and becomes dominant in biomass. Untreated domestic wastewater is usually rich in organic nitrogen, which can also be released from decaying litter upstream of reservoirs.

P. bipes is a large-sized species, which easily settles to the bottom under unfavorable conditions. Encystment and excystment are two critical stages in its life cycle. Light is widely considered as important in controlling excystment (Kishimoto et al. 2001), and thus, lake bottom depth has a significant effect on cyst germination. Cysts only can excyst at depths where light is sufficient (Park and Hayashi 1993). Kawabata and Hirano (1995) found that *Peridinium* blooms in Ishitegawa Reservoir occurred at a station about 3.7 m deep. In Huanglongdai Reservoir, the shallow bay in which the dense *Peridinium* population first occurred is less than 3 m deep. Before the wet season, the water here is transparent enough to allow light to reach the bottom.

Almost all freshwater red tides have been observed in spring and lasted for 3 or 4 weeks. Water temperature seems to be important for such short spring blooms. According to Kishimoto et al. (2001), the frequency of excystment of *P. bipes* cysts is high at water temperatures from 10°C to 20°C, and the preparation period for excystment decreases with an increase of water temperature. Such proper temperature for excystment is only met in spring. In reservoirs in Japan, red tides were found at temperatures from 12°C to 18°C (Fukuju et al. 1998). The highest density of *P. bipes* in Feitsui Reservoir (Taiwan, China) occurred between 24°C and 26°C (Wu and Chou 1998). In Huanglongdai Reservoir, *P. bipes* bloomed when water temperature was about 24°C. Thus, the strains of *P. bipes* from different reservoirs have adapted to different temperatures. Moreover, a stable stratification in open water may be necessary for dense populations. *P. bipes* is sensitive to mixing and is readily broken by strong mixing. Reservoirs narrow in shape readily show such kind of stable stratification in spring.

High temperature, strong mixing, and shortage of nutrients result in disappearance of the bloom. Lack of nutrients, especially nitrogen, was considered the main trigger of encystment (Chapman and Pfiester 1995; Grigorszky et al. 2006). In Huanglongtai Reservoir, the *Peridinium* bloom peaked on April 21, and the population density significantly decreased on May 5. On April 26, a heavy precipitation resulted in an obvious decrease in water transparency. Figure 8.6 shows the *Peridinium* density against reservoir water level. After April 28, the continuous increase in water level indicated a strong hydrological event. On May 20, the water temperature was 3°C higher than that on May 5. Thus, a dramatic change in hydrology due to heavy precipitation may directly trigger the decay of a bloom, by high turbidity and strong mixing, inhibiting dinoflagellate growth. The bloom decay was likely accelerated by the marked increase in water temperature. During the decay, TN concentration did not decrease, while TP concentration showed a small increase. As runoff increases as precipitation gets heavier, nutrient composition is surely modified. Such a change in nutrients may also have put a halt to *Peridinium* growth.

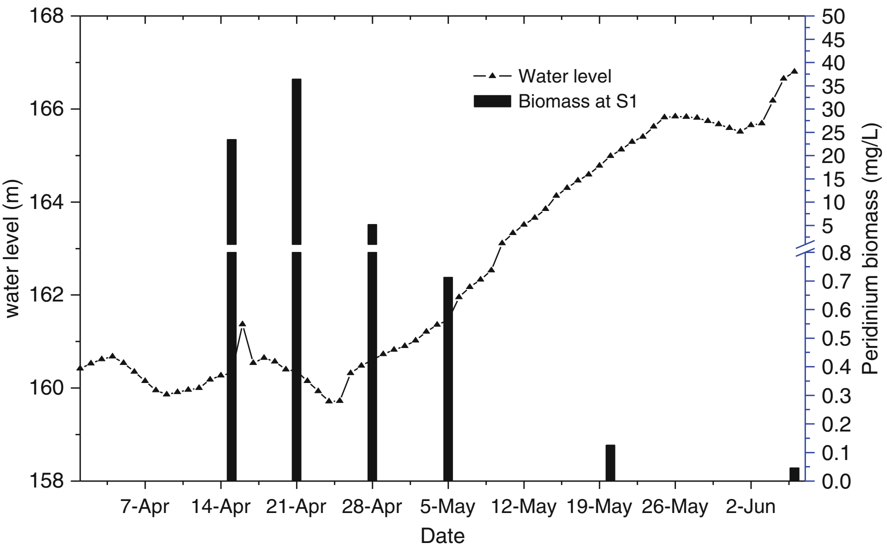

Fig. 8.6 Temporal distributions of *Peridinium* biomass and water level at station 1

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Chapter 9
Spatial Distribution of Macrozoobenthos in a Large and Deep Impoundment: Xin'anjiang Reservoir, Zhejiang Province

Qi-Gen Liu, Yu-Ting Zha, and Zhong-Jun Hu

Abstract The horizontal and bathymetrical distribution pattern of macrozoobenthos in Xin'anjiang Reservoir, a large, deep reservoir in Zhejiang Province, China, was investigated from 2007 to 2009. Macrozoobenthos was sampled and environmental variables were measured at five sites along the upper to lower reaches (S1 at upper reaches, S3 and S4 at middle reaches, and S8 and S9 at lower reaches) in 2007 to 2008. Benthic macroinvertebrates were also collected over three transects in 2009, i.e., T1, T4, and T8, traversing S1, S4, and S8, respectively, and 10–15 sites were set along each transect. Totally, 24 taxa from 5 classes were recorded. Oligochaeta predominated in the macrozoobenthic community in terms of important value (IV), density, and standing crop, among which *Limnodrilus hoffmeisteri* was prevailing. The annual average density and biomass were 793.8 ± 92.1 ind. m^{-2} and 2.25 ± 0.32 g m^{-2}, respectively. There were no clear differences in density and biomass between seasons, but significant horizontal differences revealed a maximum in the upper reaches and minimum at the lower reaches. Both density and biomass negatively correlated with water depth and Secchi depth (SD); density positively correlated with total nitrogen (TN) and total phosphorus (TP) of mixed water samples from the water column (MWSWC) and with TP of local water samples at the bottom (LWSB). Biomass positively correlated with TP of MWSWC. Only water depth (from eight physical and chemical variables) was selected by a stepwise regression model to explain the variation in density and biomass of macrozoobenthos in 2007 to 2008. Bathymetrically, both density and biomass increased with water depth, peaked just below the thermocline, and then decreased gradually. This numerical bathymetric change in Xin'anjiang Reservoir conformed to the patterns of vertical distribution of zoobenthos in oligo- and mesotrophic lakes. The relationship of density and biomass with the main

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environmental variables, the bathymetric distribution pattern of macrozoobenthos, and the two biological indices (Wright index and Carlander's biotic index) all typified Xin'anjiang Reservoir as oligo- and mesotrophic, with the upper reaches mesotrophic or moderately polluted and the middle and lower reaches oligotrophic or slightly polluted. Three other biological indices (King, Goodnight–Whitley, and Shannon–Weaver) were unsuitable to assess water quality.

9.1 Introduction

Oligochaetes, mollusks, and aquatic insects are the main components of macrozoobenthos in lakes and reservoirs. They play an important role in the coupling of benthic and pelagic food webs and in the matter cycling and energy flow of lentic freshwater ecosystems. Macrozoobenthos is the food source of various benthic fish and an important contributor to secondary productivity and fish production. In addition, macrozoobenthos is an important element in the development of concepts like "lake zonation" and "lake typology" and has been a fruitful research subject in limnology (Brinkhurst 1974).

Many biotic and abiotic factors such as sediment type, food resource, and dissolved oxygen may determine macrozoobenthic density and distribution (Petridis and Sinis 1995; Bechara 1996). Because water depth may cause change in important physical–chemical variables, which in turn have indirect effects on benthic macrofauna (Nalepa 1989; Petridis and Sinis 1993; Martin et al. 1999; Baudo et al. 2001; Ohtaka et al. 2006), many researchers suggest that it is a key ecological factor explaining the density and distribution of those invertebrates (Petridis and Sinis 1993; Baudo et al. 2001). Generally, higher species diversity of zoobenthos is found in the littoral zone than at depths (Petridis and Sinis 1993; Baudo et al. 2001; Cui et al. 2008). At the same time, the distribution of density along depth gradients might follow certain patterns, which change with trophic status of a water body (Hargrave 2001). There is currently much interest in the ecology of zoobenthos in lakes or reservoirs. In Europe and North America, numerous reports can be found on the zoobenthos of deep-water lakes and reservoirs (Nalepa 1989; Petridis and Sinis 1993; Martin et al. 1999; Baudo 2001; Ohtaka et al. 2006). Similar studies are scarce in China (Shao et al. 2007; Cui et al. 2008; Xiong et al. 2008), and are restricted to shallow lakes and reservoirs (Liu and Liang 1997; Ma et al. 2004).

Xin'anjiang Reservoir originated from the dam construction of a hydraulic power plant on the River Xin'anjiang. It is an artificial deep-water lake with mean and maximum depth 30 m and over 100 m, respectively. Xin'anjiang Reservoir is a well-known scenic site in the Yangtze River delta. It is also an important fishery production site. In recent decades, it experienced aggravation of eutrophication, especially in 1998 and 1999, when large-scale cyanobacterial blooms occurred repeatedly. Many investigations on phyto- and zooplankton, chlorophyll *a*, water chemistry, and food-web structure have been conducted since (Li and Yu

2001, 2003; Liu et al. 2004, 2007a, b; Liu 2005; Zhu et al. 2007), but few on zoobenthos. Xin'anjiang is a mountainous reservoir with steep shore that differs from natural lakes in that the littoral zone is much smaller than the profundal zone. From 2007 to 2009, we carried out investigations on its macrozoobenthos in order to (1) reveal its temporal and spatial variation, especially its response to water depth and (2) define the key ecological variables explaining its distribution. We hope that such study could provide systematic data for protection of aquatic environment and development of fisheries in Xin'anjiang Reservoir and other reservoirs in China.

9.2 Materials and Methods

9.2.1 Study Site

Xin'anjiang Reservoir, situated in Chun'an County, Zhejiang Province, China (118°34′–119°15′E, 29°22′–29°50′N), is one of the largest reservoirs in China (Fig. 9.1). It is dendritic with 1,406 km of shoreline and 1,078 islands, and has a surface area of about 573 km^2 and an estimated volume of 178.4×10^8 m^3 with mean water depth of 30.4 m, length maximum of 150 km, and width maximum of 10 km. The inflows mostly come from surface runoff, and there are more than 30 tributaries, of which Xin'anjiang River is the largest and accounts for about 60% of the total inflows. Thus, water quality in this river has a profound influence on the

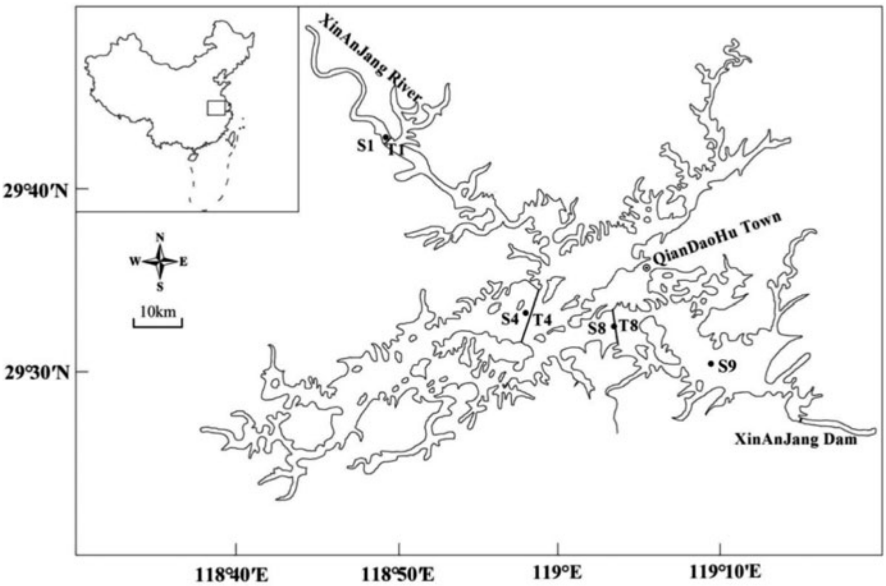

Fig. 9.1 Sampling sites and transects in Xin'anjiang Reservoir

water quality of the reservoir. The reservoir catchment has a subtropical and monsoonal climate, with an annual mean air temperature of about 17.0°C.

9.2.2 Field Investigation

In 2007 and 2008, macrozoobenthos was collected seasonally (in April, July, October, and January) at five sampling sites situated in the upper reaches (northwest zone or riverine region of the reservoir), middle reaches (middle lacustrine region), and lower reaches (southeast lacustrine region) (Fig. 9.1). At sites S1, S4, and S9, water samples at 4 m intervals above 20 m and every 5 m below that were collected and chemical variables measured. Mixed water samples of every layer were collected at sites S3 and S8. Water temperature (T), depth (WD), Secchi depth (SD), pH, and dissolved oxygen (DO) were measured directly in the field. Total nitrogen (TN), total phosphorus (TP), potassium permanganate index (COD_{Mn}), and chlorophyll *a* were measured in the laboratory (Table 9.1). Total phosphorus was determined by colorimetry (GB11893-1989) and nitrogen concentrations by alkaline potassium persulfate digestion and UV spectrophotometry (GB11894-1989). Potassium permanganate index was determined by the acidic potassium permanganate method (Environmental Protection Bureau of the People's Republic of China 2002). Chlorophyll *a* was measured by spectrophotometry (Jin and Tu 1990).

In 2009, macrozoobenthos was collected in March, June, September, and December at three transects T1, T4, and T8 that traversed S1, S4 and S8, respectively. There were 12, 13, and 19 sampling stations in March; 10, 12, and 11 sampling stations in June; 10, 15, and 10 sampling stations in September; and 10, 15, and 11 sampling stations in December at T1, T4, and T8, respectively, where water depth was measured.

Bottom samples were taken using a modified Peterson grab with an opening area of 0.0625 m^2. The samples were sieved over a mesh size of 0.45 mm, and the retained material was kept individually in labeled polythene bottles. Animals were sorted and preserved in 75% ethanol in 2007 to 2008 and 8% formalin in 2009. All specimens were identified to species or genus level.

9.2.3 Data Analysis

The dominant species of the community were determined by importance value (IV) of each species, calculated from all data combined. The equation of important value is: $IV = (RD + RF + RB)/3$, where RD, RF, and RB refer to relative density, frequency, and biomass, respectively (Hu et al. 2009).

The following five biological indices were used to assess the water quality of Xin'anjiang Reservoir:

Table 9.1 Annual average value of physical and chemical factors of Xin'anjiang Reservoir between January 2007 and October 2008

Site no.	WD (m)	T (°C)	SD (m)	pH	DO (mg/L)	Chl*a* (μg/L)	TN (mg/L)	TP (mg/L)	COD_{Mn} (mg/L)
1	28.6	17.8/13.2	3.2	7.7/7.7	6.2/5.0	2.810/1.385	1.312/1.308	0.027/0.024	0.944/0.693
3	41.8	16.2/–	6.2	7.9/–	6.9/–	2.605/–	1.014/–	0.019/–	0.529/–
4	41.9	16.1/11.6	6.2	8.0/7.9	7.1/6.6	1.801/0.798	1.049/1.110	0.014/0.013	0.612/0.450
8	39.8	16.7/–	8.4	7.9/–	7.3/–	1.273/–	0.946/–	0.017/–	0.506/–
9	56.1	14.8/10.2	8.0	7.9/7.8	7.3/6.0	1.237/0.611	0.924/0.991	0.012/0.011	0.437/0.323
Mean	41.6	16.3/11.7	6.4	7.9/7.8	7.0/5.9	1.945/0.931	1.049/1.136	0.018/0.016	0.607/0.490
F	26.110^{***}	0.400/3.465	14.283^{***}	0.627/0.281	0.412/0.653	1.193/2.696	$4.167^{**}/2.651$	$2.656^{*}/6.285^{**}$	$3.406^{*}/3.550^{*}$

Data upon the slash are for mixed water samples of water column, and those under the slash for local water sample at bottom $^{*}P < 0.05$; $^{**}P < 0.01$; $^{***}P < 0.001$; The same as the tables below

Shannon–Weaver index (H') (Shannon 1948): $H' = \sum (n_i/N) \ln(n_i/N)$, where n_i is the number of each taxon and N their total number in the community.

King index (I_K) (King and Ball 1964) is the ratio of aquatic insect weight to tubificid worm weight.

Goodnight–Whitley biotic index (I_{GW}) (Goodnight and Whitley 1960): I_{GW} = density of oligochaetes/total density of zoobenthos.

Wright biotic index (Wright 1955) equals the oligochaete density, calculated as number of individuals per square meter.

Carlander's biomass (Carlander 1952) is weight of zoobenthos (sometimes excluding mollusks) (Zhang et al. 2010) per square meter (g m^{-2}).

9.3 Results

9.3.1 Species Composition

There were 24 macrozoobenthic taxa, representing the five classes: Oligochaeta, Hirudinea, Bivalvia, Insecta, and Nemathelminthes. Oligochaeta were represented by most species (12), followed by Insecta and Bivalvia with 7 and 3 species, respectively. T1 and T4 had similar species numbers (21 and 24, respectively), and T8 had the lowest species diversity with only ten taxa (Table 9.2).

Horizontally, *Limnodrilus hoffmeisteri* (35.12%) predominated in importance value (IV), and *Branchiura sowerbyi* (23.26%), *Tubifex tubifex* (14.62%), and *L. claparedeianus* (12.58%) subdominated. The IV of other taxa was less than 2.5% at T1. At T4, the dominance of *L. hoffmeisteri* was higher than that at T1 with 41.94% of the total IV. *B. sowerbyi*, *Limnodrilus* sp., Tubificinae sp., and *L. claparedeianus* were the common taxa, while the other 19 species were rare with IV less than 1.5%. At T8, the dominance of *L. hoffmeisteri* (34.10%) was similar to that at T1. The dominance of *Limnodrilus* sp. (38.55%) at T8 rose, compared with that at T1 and T4. *L. claparedeianus, B. sowerbyi, and* Tubificinae sp. were common with about 8% of the total important value. The other five taxa had very low IV (<1.6%, Table 9.2).

Overall, *L. hoffmeisteri* was predominant (IV: 36.37%), while *B. sowerbyi* (20.35%), *L. claparedeianus* (11.41%), *Limnodrilus* sp. (9.68%), *T. tubifex* (8.82%), and Tubificinae sp. (5.53%) were common, and the other 18 species were rare (each lower than 1.4%) (Table 9.2). Oligochaeta was the predominant group contributing more than 93% of the total IV for each transect throughout the year (Table 9.2).

In March, 17, 9, and 6 taxa were collected at T1, T4, and T8, respectively, and the average richness per sampling station was 5.5, 4.1, and 2.2. At T1, *L. hoffmeisteri* and *T. tubifex* had the highest density (about 600 ind. m^{-2}), and the other taxa with relatively larger density were *B. sowerbyi* (234.7 ind. m^{-2}), *L. claparedeianus* (113.3 ind. m^{-2}), and Tubificinae sp. (81.3 ind. m^{-2}). At T4, *L. hoffmeisteri* had the largest

Table 9.2 Macrozoobenthos collected from Xin'anjiang Reservoir and their importance value in 2009

Group	Taxa	Transect 1	Transect 4	Transect 8	Total
Nematoda	Nematoda sp.	0.49	1.16	0.44	0.72
Oligochaeta		93.6	95.48	99.57	95.23
	Aulodrilus limnobius	0.16	0.37	0	0.20
	A. pluriseta	0	0.83	0.34	0.37
	Branchiura sowerbyi	23.26	17.02	7.53	20.35
	Limnodrilus claparedeianus	12.58	8.81	8.62	11.41
	L. grandisetosus	2.22	0.25	1.54	1.28
	L. hoffmeisteri	35.14	41.94	34.10	36.37
	Limnodrilus sp.	1.20	13.63	38.55	9.68
	L. udekemianus	1.26	0.18	0	0.64
	Teneridrilus mastix	0.54	0.37	0.33	0.44
	Tubifex tubifex	14.62	1.43	1.04	8.82
	Tubificinae sp.	2.46	10.47	7.52	5.53
	Nais sp.	0.16	0.18	0	0.14
Hirudinea	Hirudinea sp.	0.39	0.37	0	0.32
Mollusks		3.29	0.85	0	2.24
	Cobicula flaminca	0.45	0.18	0	0.30
	Limnoperna lacustris	1.97	0.18	0	1.34
	Sphaerium lacustre	0.87	0.49	0	0.60
Chironomidae		2.22	2.12	0	1.71
	Chironomid sp.	0	0.45	0	0.15
	Cryptochironomus digitatus	0.17	0.18	0	0.14
	Dicrotendipes sp.	0	0.38	0	0.14
	Microchironomus sp.	0.34	0.18	0	0.21
	Micropsetra logana	0.16	0.37	0	0.20
	Polypedilum scalaenum	0.69	0.18	0	0.36
	Procladius choreus	0.86	0.38	0	0.51

density of more than 500 ind. m^{-2}, followed in order by *Limnodrilus* sp. (306.5 ind. m^{-2}), Tubificinae sp. (179.5 ind. m^{-2}), and *B. sowerbyi* (64.0 ind. m^{-2}). AT T8, the density of each taxa decreased drastically compared with that of T1 and T4, and the higher values were slightly more than 100 ind. m^{-2} (Table 9.3).

In June, the most common taxon in all transects was *L. hoffmeisteri* with a density of 467.2 ind. m^{-2} at T1 (total of 14 species, average richness per sampling station of 5.2 species), 554.7 ind. m^{-2} at T4 (12 and 2.9 species, respectively), and 117.7 ind. m^{-2} at T8 (6 and 2.1 species, respectively). *B. sowerbyi* (134.4 ind. m^{-2}) and *L. claparedeianus* (131.2 ind. m^{-2}) had the second largest density at T1. The second common taxa were *Limnodrilus* sp. (112.0 ind. m^{-2}) and *L. claparedeianus* (88.5 ind. m^{-2}) at T4 and *Limnodrilus* sp. (38.9 ind. m^{-2}) at T8 (Table 9.3).

In September, 10, 24, and 7 taxa were present at T1, T4, and T8, and the corresponding species number per sampling station was 4.2, 4.1, and 1.3, respectively. At T1, *L. hoffmeisteri* occurred abundantly (1,019.2 ind. m^{-2}), markedly more than other common taxa such as *T. tubifex* (369.6 ind. m^{-2}), *B. sowerbyi*

Table 9.3 Density (ind./m^2) of macrozoobenthos collected from Xin'anjiang Reservoir in 2009

Taxa	March			June			September			November		
	T1	T4	T8	T1	T4	T8	T1	T4	T8	T1	T4	T8
Nematoda sp.	4.0	6.2	0	0	2.1	0	0	1.1	5.3	0	2.1	0
Aulodrilus limnobius	1.3	0	0	0	0	0	0	1.1	0	0	1.1	0
A. pluriseta	0	8.6	0	0	1.1	1.1	0	1.1	0	0	0	0
Branchiura sowerbyi	234.7	64.0	6.7	134.4	13.9	14.9	179.2	12.8	0	102.4	30.9	0
Limnodrilus claparedeianus	113.3	29.5	18.5	131.2	88.5	17.1	102.4	65.1	12.8	283.2	3.2	12.8
L. grandisetosus	2.7	0	0	11.2	0	1.1	4.8	1.1	2.1	7.1	0	0
L. hoffmeisteri	590.7	531.7	112.0	467.2	554.7	117.7	1019.2	188.8	90.7	1246.4	69.3	59.7
Limnodrilus sp.	5.3	306.5	106.1	4.8	112.0	38.9	0	1.1	485.3	8.9	45.9	39.5
L. udekemianus	1.3	0	0	6.4	0	0	0	1.1	0	6.4	0	0
Teneridrilus mastix	9.3	0	0	0	1.1	0	0	1.1	0	0	0	1.1
Tubifex tubifex	633.3	4.9	0.8	78.4	0	0	369.6	2.1	1.1	38.4	4.3	2.1
Tubificinae sp.	81.3	179.5	39.6	25.6	41.6	0	1.6	25.6	14.9	1.8	26.7	9.6
Nais sp.	0	0	0	0	0	0	1.6	1.1	0	0	0	0
Hirudinea sp.	0	0	0	0	1.1	0	1.6	1.1	0	3.6	0	0
Cobicula flaminca	0	0	0	0	0	0	0	1.1	0	3.6	0	0
Limnoperna lacustris	2.7	0	0	3.2	0	0	4.8	1.1	0	5.3	0	0
Sphaerium lacustre	4.0	0	0	3.2	1.1	0	0	1.1	0	0	0	0
Chironomid sp.	0	0	0	0	0	0	0	1.1	0	0	1.1	0
Cryptochironomus digitatus	0	0	0	1.6	0	0	0	1.1	0	0	0	0
Dicrotendipes sp.	0	0	0	0	1.1	0	0	1.1	0	0	0	0
Microchironomus sp.	1.3	0	0	3.2	0	0	0	1.1	0	0	0	0
Micropsetra logana	1.3	0	0	0	1.1	0	0	1.1	0	0	0	0
Polypedilum scalaenum	1.3	0	0	8.0	0	0	0	1.1	0	1.8	0	0
Procladius choreus	6.7	1.2	0	3.2	0	0	1.6	1.1	0	0	0	0

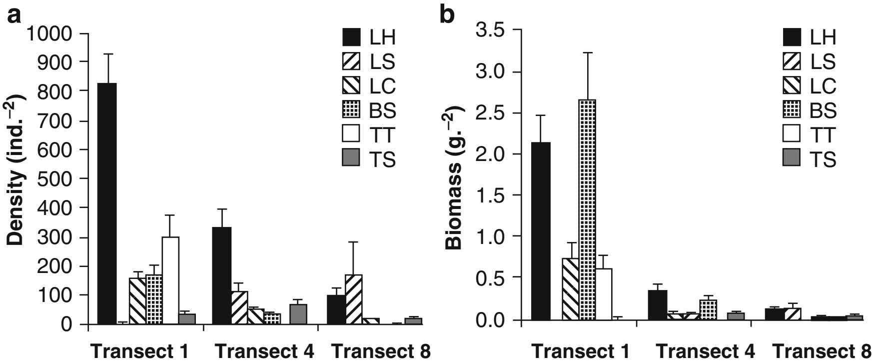

Fig. 9.2 The horizontal distribution of density and biomass (mean ± SE) of the main macrozoobenthic taxa in Xin'anjiang Reservoir in 2009 (*LH Limnodrilus hoffmeisteri*; *LS Limnodrilus* sp.; *LC Limnodrilus claparedeianus*; *BS Branchiura sowerbyi*; *TT Tubifex tubifex*; *TS* Tubificinae sp.)

(179.2 ind. m^{-2}), and *L. claparedeianus* (102.4 ind. m^{-2}). At T4, *L. hoffmeisteri* also had the largest density (188.8 ind. m^{-2}) followed by *L. claparedeianus* (65.1 ind. m^{-2}), Tubificinae sp. (25.6 ind. m^{-2}), and *B. sowerbyi* (12.8 ind. m^{-2}). At T8, however, *Limnodrilus* sp. was commonest with a density of 485.3 ind. m^{-2}, followed by *L. hoffmeisteri* (90.7 ind. m^{-2}) (Table 9.3).

In November, 12, 9, and 6 taxa were collected at T1, T4, and T8, respectively, and 5.5, 2.7, and 1.5 taxa were sampled per sampling station. At T1, *L. hoffmeisteri* had the highest density (1,246.4 ind. m^{-2}) in all transects in the 4 months. *L. hoffmeisteri*, *Limnodrilus* sp., and Tubificinae sp., and *B. sowerbyi* or *L. claparedeianus* at T4 (maximum of 69.3 ind. m^{-2}) and T8 (59.7 ind. m^{-2}) did not differ in density as much as in other transects (Table 9.3).

Collectively, the density of the 18 species other than the 6 main taxa were all less than 9 ind. m^{-2} in any transect and month (Table 9.3), and the density and biomass of *L. hoffmeisteri, L. claparedeianus, B. sowerbyi*, and *T. tubifex* all declined along the gradient of upper to lower reaches. However, *Limnodrilus* sp. had an opposite trend and Tubificinae sp. had the highest density at Transect 4 (Fig. 9.2).

9.3.2 Density and Biomass

9.3.2.1 Density and Biomass of the Whole Community

The annual mean density and biomass over three transects sampled in the reservoir were 607.5 ± 122.7 ind. m^{-2} and 0.55 ± 0.1 g m^{-2} in 2007–2008 (mean ± SE, $n = 38$) and 793.8 ± 92.1 ind. m^{-2} and 2.25 ± 0.32 g m^{-2} in 2009 (mean ± SE, $n = 147$), respectively. The predominant species plus the five common species constituted 96.1% of density and 91.5% of biomass of the whole community, with

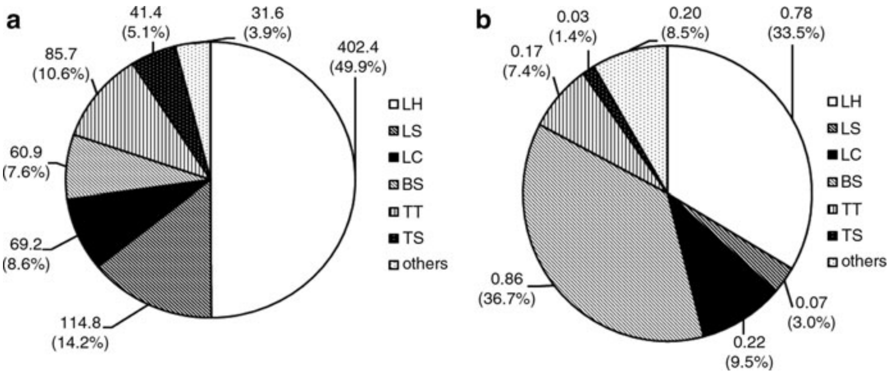

Fig. 9.3 Density (**a**) and biomass (**b**), and the proportions (data in parentheses) of the main macrozoobenthic species collected in 2009 (units of density and biomass are ind./m^2 and g/m^2, respectively)

L. hoffmeisteri the dominant species in density and, together with *B. sowerbyi*, the prevailing taxon in biomass in 2009 (Fig. 9.3).

9.3.2.2 Seasonal Dynamics and Horizontal Spatial Distribution

The highest density of the community occurred in summer in 2007–2008 (699.0 ind. m^{-2}) and spring 2009 (940.6 ind. m^{-2}), the lowest appeared in autumn in 2007–2008 (561.4 ind. m^{-2}), and winter 2009 (602.7 ind. m^{-2}). The highest biomass of the community occurred in winter 2007–2008 (0.69 g m^{-2}) and spring 2009 (2.80 g m^{-2}), and the lowest appeared in spring 2007–2008 (0.33 g m^{-2}) and autumn 2009 (about 1.92 g m^{-2}) (Figs. 9.4 and 9.5). However, two-way ANOVA indicated no significant differences in density ($F = 0.005$, $P > 0.05$ for 2007–2008; $F = 1.201$, $P > 0.05$ for 2009) and biomass ($F = 2.238$, $P > 0.05$ for 2007–2008; $F = 2.849$, $P > 0.05$ for 2009) of the community between seasons (Tables 9.4 and 9.5), although density and biomass differed horizontally and had a decreasing tendency from the upper through middle to lower reaches. For density: $F = 4.309$, $P < 0.05$ in 2007–2008, $F = 14.381$, $P < 0.001$ in 2009; for biomass: $F = 25.984$, $P < 0.001$ in 2007–2008; $F = 54.257$, $P < 0.001$ in 2009 (Tables 9.4–9.6). The highest density appeared at S1 in 2007–2008 (1,466.0 ind. m^{-2}) and at T1 in 2009 (1,501.7 ind. m^{-2}); the lowest density occurred at S9 in 2007–2008 (133.7 ind. m^{-2}) and at T8 in 2009 (382.4 ind. m^{-2}). The highest biomass appeared at S1 in 2007–2008 (1.53 g m^{-2}) and at T1 in 2009 (6.51 g m^{-2}); the lowest biomass was at S9 in 2007–2008 (0.10 g m^{-2}) and at T8 in 2009 (0.35 g m^{-2}).

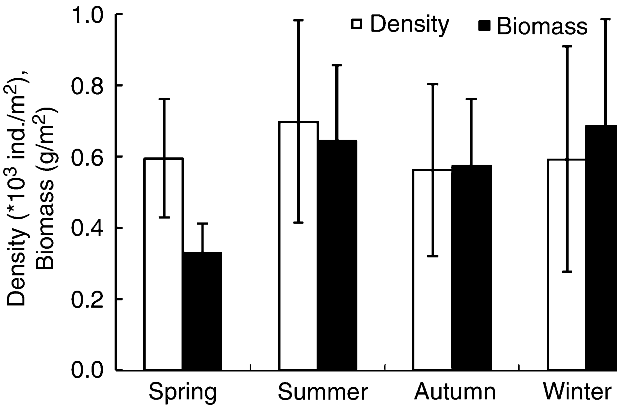

Fig. 9.4 Seasonal pattern of density and biomass (mean ± SE) of profundal macrozoobenthos in Xin'anjiang Reservoir in 2007 to 2008

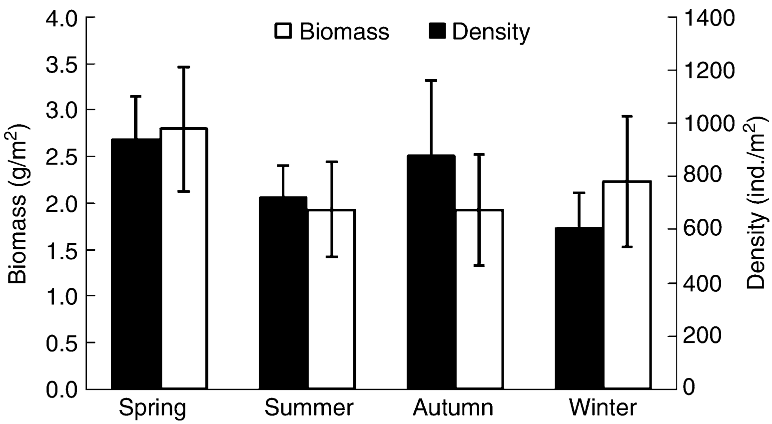

Fig. 9.5 The seasonal changes of density and biomass (mean ± SE) of macrozoobenthos in Xin'anjiang Reservoir in 2009

9.3.2.3 Bathymetric Distribution

The density (Y_d) and biomass (Y_b) of profundal macrozoobenthos in 2007–2008 declined exponentially with water depth (X) ($Y_d = 8{,}402.5\ e^{-0.074\ X}$, $R^2 = 0.549$, $df = 21$, $P < 0.001$; $Y_b = 15.16\ e^{-0.087\ X}$, $R^2 = 0.715$, $df = 21$, $P < 0.001$) (Fig. 9.6). The density of the whole community together with that of Oligochaeta increased with increasing water depth, peaked up to 1,235.0 ind. m^{-2} and 1,233.0 ind. m^{-2}, respectively, at depths that ranged 21–27.9 m, and had a declining tendency with greater water depths. The bathymetric distribution of the biomass of whole community and Oligochaeta was similar to density with a maximum

Table 9.4 Two-way ANOVA of season and sampling site for density and biomass in 2007 to 2008

Source	Sum of squares	*df*	Mean square	*F*	*P*
Intercept	12381120.3/10.3	1/1	12381120.3/10.3	24.656/103.567	0.000***/0.000***
Season	7105.7/0.7	3/3	2368.6/0.2	0.005/2.238	1.000/0.119
Sampling site	8656004.7/10.3	4/4	2164001.2/2.6	4.309/25.984	0.013*/0.000***
Season * sampling site	3247011.7/2.2	12/12	270584.3/0.2	0.539/1.868	0.861/0.112
Error	9038739.8/1.8	18/18	502152.2/0.1		
Total	35178867.8/26.8	38/38			

Table 9.5 Two-way ANOVA of season and site for density and biomass in 2009

Source	Sum of squares	*df*	Mean square	*F*	*P*
Intercept	103674403.5/907.4	1/1	103674403.5/907.4	105.588/107.594	0.000***/0.000***
Month	3539583.8/33.4	3/3	1179861.3/11.1	1.201/2.849	0.312/0.216
Transect	28241250.0/1018.9	2/2	14120625.0/509.4	14.381/54.257	0.000***/0.000***
Month * Transect	15289817.7/56.1	6/6	2548303.0/9.3	2.595/1.414	0.021*/0.279*
Error	274575241.0/999.2	135/135	981874.3/7.4		
Total	280942336.0/ 2897.6	147/147			

Table 9.6 Changes in density and biomass (mean ± SE [*n*]) of macrozoobenthos among sampling sites in 2007 to 2008 and transects in 2009

Transects or sampling sites	Density (ind./m^2)		Biomass (g/m^2)	
	2007 to 2008	2009	2007 to 2008	2009
1	1,466.0 ± 347.6^a (8)	1,501.7 ± 166.3^a (42)	1.53 ± 0.24^a (8)	6.51 ± 0.76^a (42)
3	656.5 ± 288.7^b (8)		0.35 ± 0.08^{b,c} (8)	
4	485.0 ± 117.0^b (8)	627.2 ± 102.5^b (55)	0.53 ± 0.12^b (8)	0.75 ± 0.12^b (55)
8	184.0 ± 42.9^b (7)	382.4 ± 171.5^b (50)	0.15 ± 0.03^c (7)	0.35 ± 0.13^b (50)
9	133.7 ± 19.3^b (7)		0.10 ± 0.02^c (7)	

The different letters in each column indicate significant differences at the level of $P < 0.05$ below this table

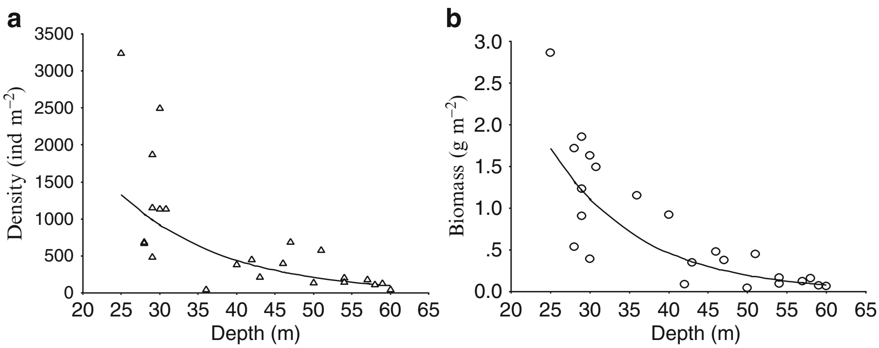

Fig. 9.6 The decline pattern of density (**a**) and biomass (**b**) of macrozoobenthic community in Xin'anjiang Reservoir with water depth in 2007 to 2008

biomass of about 4.2 to 4.3 g m^{-2} at depths of 21–27.9 m. Meanwhile, the bathymetric distribution of the predominant species, *L. hoffmeisteri*, also resembled the above mentioned patterns. Density and biomass of chironomids and mollusks

Table 9.7 Bathymetric distribution pattern of density and biomass (mean ± SE [*n*]) of different groups in Xin'anjiang Reservoir in 2009

Depth (m)	Oligochaeta		Chironomid		Mollusks		*Limnodrilus hoffmeisteri*		Total	
	Density (ind./m^2)	Biomass (g/m^2)	Density (ind./m^2)	Biomass (g/m^2)	Density (ind./m^2)	Biomass (g/m^2)	Density (ind./m^2)	Biomass (g/m^2)	Density (ind./m^2)	Biomass (g/m^2)
0–6.9	48.0	0.65	48.0	0.016	16.0	0.38	16.0	0.03	112.0	1.04 (1)
7–13.9	173.7 ± 73.8	0.90 ± 0.65	13.7 ± 13.7	0.015 ± 0.015	11.4 ± 7.6	1.20 ± 1.09	59.4 ± 27.7	0.07 ± 0.03	198.9 ± 85.6	2.12 ± 1.73(7)
14–20.9	524.0 ± 189.9	1.43 ± 0.65	4.0 ± 4.0	0.005 ± 0.002	0	0	272.0 ± 134.0	0.53 ± 0.42	528.0 ± 189.1	1.43 ± 0.65(4)
21–27.9	1,233.0 ± 324.3	4.24 ± 1.20	1.0 ± 1.0	0.001 ± 0.001	1.0 ± 1.0	0.06 ± 0.06	740.0 ± 215.4	1.59 ± 0.46	1,235.0 ± 324.2	4.30 ± 1.20(16)
28–34.9	1,105.9 ± 216.1	3.97 ± 0.95	7.0 ± 4.5	0.005 ± 0.003	5.1 ± 2.4	0.02 ± 0.01	505.6 ± 83.8	0.95 ± 0.18	1,123.2 ± 216.8	4.00 ± 0.96(25)
35–41.9	821.1 ± 199.3	1.71 ± 0.75	4.2 ± 2.1	0.004 ± 0.002	0.8 ± 0.8	0.01 ± 0.01	472.4 ± 126.1	0.58 ± 0.17	830.3 ± 202.6	1.72 ± 0.76(19)
42–48.9	426.2 ± 229.9	0.32 ± 0.14	0	0	0	0	309.8 ± 199.1	0.23 ± 0.12	426.2 ± 229.9	0.32 ± 0.14(11)
49–55.9	742.6 ± 492.4	0.59 ± 0.38	0	0	0	0	163.8 ± 74.5	0.18 ± 0.08	789.0 ± 521.9	0.64 ± 0.40(16)
56–62.9	532.0 ± 213.9	0.54 ± 0.19	0	0	0	0	52.0 ± 13.7	0.11 ± 0.05	532.0 ± 213.9	0.54 ± 0.19(4)
63–69.9	340.3 ± 245.0	0.43 ± 0.33	0	0	0	0	138.7 ± 122.7	0.19 ± 0.16	340.3 ± 245.0	0.43 ± 0.33(3)

were highest at depths below 14 m, and both animals maintained a very low level at the 14–41.9 m depth for mollusks. These two groups disappeared when water depth was deeper than 42 m (Table 9.7).

9.3.3 Relationship Between Density and Environmental Factors

Correlation analyses were carried out between density and biomass of profundal macrozoobenthos sampled in 2007–2008 and environmental factors of mixed water column samples (MWSWC) (WD, T, SD, pH, DO, COD_{Mn}, Chla, TN, and TP), and between the density, biomass and environmental factors of bottom water samples (LWSB) (T, pH, DO, COD_{Mn}, Chla, TN, and TP). The results showed that both density and biomass negatively correlated with WD and SD, density positively correlated with TN and TP of MWSWC and with TP of LWSB, and biomass positively correlated with TP of MWSWC ($P < 0.05$) (Table 9.8).

Of the eight main physical–chemical variables (WD, T, pH, DO, COD_{Mn}, Chla, TN, and TP) of bottom water samples, water depth was the only variable retained by the stepwise regression, making it the most important environmental factor to explain the variation of density and biomass of profundal macrozoobenthos ($Y_d = 2{,}642.5-46.2X$, $R^2 = 0.445$, $P < 0.001$; $Y_b = 2.96-0.05X$, $R^2 = 0.680$, $P < 0.001$).

9.3.4 Bioassessment of Water Quality

Tables 9.9 and 9.10 list the bioassessment criterion and assessment of water quality of Xin'anjiang Reservoir. The Goodnight–Whitley, King, and Shannon–Weaver indices indicated that the upper (riverine region), middle, and lower reaches of this reservoir were all at least moderately polluted. Carlander's biomass and Wright indices, however, demonstrated that the riverine region was moderately polluted or mesotrophic, and the other two regions were slightly polluted to oligotrophic (Table 9.10).

Table 9.8 Correlation between density and biomass of profundal macrozoobenthos and the main physical and chemical factors in 2007 to 2008

Item	WD	SD	TN[a]	TP[b]
Density	−0.525** (36)	−0.388* (31)	0.463** (36)	0.460** (38)/0.533** (21)
Biomass	−0.615*** (36)	−0.569** (31)	0.291 (36)	0.331* (38)/0.323 (21)

The data in the parentheses are degree of freedom

[a]Mixed water samples of water column (MWSWC)

[b]The data upon slash are for MWSWC, and those under slash for local water samples at bottom (LWSB)

Table 9.9 Bioassessment criterion of water quality using biological indices

I_{GW} (Goodnight and Whitley 1960)		I_K (King and Ball 1964)		Wright index (Wright 1955)		Carlander's biomass (Carlander 1952)		Shannon–Weaver (Shannon 1948; Su et al. 2008)	
VI[a]	LWQ[b]	VI	LWQ	VI	LWQ	VI	LWQ	VI	LWQ
>30	CL	0:1	SP	<100	CL	0.2–1.7	OT	>3	CL
30–60	LP	200:1	HP	100–999	LP	2.5–6.25	MT	2–3	SP
60–80	MP	201–600:1	MP	1,000–5,000	MP	10–25	ET	1–2	MP
>80	SP	>600:1	LP	>5,000	HP			<1	HP

[a]Value of index
[b]Level of water quality the same as table 9.10
CL clean; *LP* lightly polluted; *MP* moderately polluted; *HP* heavily polluted; *SP* seriously polluted; *OT* oligotrophic; *MT* mesotrophic; *ET* eutrotrophic

Table 9.10 Bioassessment of water quality of Xin'anjiang Reservoir using macrozoobenthos

Transect No.	I_{GW}		I_K		Wright index		Carlander's biomass		Shannon–Weaver index	
	VI[a]	LWQ[b]	VI	LWQ	VI	LWQ	VI	LWQ	VI	LWQ
1	98.9	HP	0.00116	HP	1,486	MP	6.23	MT	1.349	MP
4	98.8	HP	0.00113	HP	620	LP	0.75	OT	1.412	MP
8	100.0	HP	0.00000	HP	375	LP	0.35	OT	1.152	MP

9.4 Discussion

The species composition of macrozoobenthic assemblages is homogeneous in mountain reservoirs where oligochaetes prevail and other groups such as aquatic insects and mollusks are absent or rare (Liang and Wang 2001). Our investigation in Xin'anjiang Reservoir confirmed that oligochaetes dominated in terms of species richness (50%), importance value (95.2%), and total density (98.4%). This result suggests a different adaptability of different taxa to water depth. Mollusks disappeared in deeper areas (Xiong et al. 2008). A decline in species diversity of zoobenthos was mainly related to a decrease in species number of chironomids in Tavropos Reservoir, Greece (Petridis and Sinis 1993), and the density ratio of oligochaetes to chironomids increased rapidly with water depth at 26–100 m in Lake Ikeda of Japan (Ohtaka et al. 2006).

In Xin'anjiang Reservoir, the density and biomass of profundal macrozoobenthos in 2007–2008 decreased exponentially with water depth, and in 2009 increased gradually with water depth, peaked at 21–27.9 m (just below the thermocline), and then decreased. The bathymetric change pattern of density and biomass in this reservoir is consistent with those of oligo- and mesotrophic deep-water lakes (Brinkhurst 1974; Petridis and Sinis 1993; Hargrave 2001). Of all environmental variables considered, only water depth explained some variation. Because water depth is a key ecological factor for the benthic environment, controlling dissolved oxygen, food availability, and physical features of benthic habitat, it could lead to large variations in the composition, density, and biomass of zoobenthos (Mozley and Winnell 1975; Petridis and Sinis 1993; Nalepa 1989; Martin et al. 1999; Baudo et al. 2001; Ohtaka et al. 2006; Cui et al. 2008). The standing crop of zoobenthos in Lake Michigan displayed a pattern similar to that of Xin'anjiang Reservoir in 2009. The possible reason is that at shallow depths above the thermocline, wide fluctuations in bottom temperatures and unstable substrates keep benthic standing stocks suppressed. As depth increases, temperatures fluctuate less and the sediments are less influenced by storms and currents. Suspended particles from the shallower regions begin to settle, providing food resources for the benthos. At depths just below the thermocline, standing stocks are at a maximum. As depth increase further, a greater proportion of potential food is mineralized in the water column before it settles to the bottom and standing stocks declined (Nalepa, 1989). Moreover, silts and organic matters do not settle easily because Xin'anjiang Reservoir is a mountainous reservoir with steep shorelines. Thus, macrozoobenthos could be suppressed further. A confrontation between density and biomass of zoobenthos of Xin'anjiang Reservoir in 2007 to 2008 and environmental factors revealed that: (1) density and biomass correlated negatively with Secchi depth; (2) density correlated positively with total phosphorus and nitrogen of mixed water-column samples; (3) density correlated positively with total phosphorus of local water samples at bottom; (4) biomass correlated positively with total phosphorus of mixed samples water column samples; and (5) density and biomass did not correlate with dissolved oxygen and water temperature. Those results showed that primary productions in the water column, as indicated by the combination of TN, TP, and Chlorophyll *a*, had significant effects on the density and biomass of zoobenthos. Xin'anjiang Reservoir was oligo- and mesotrophic (Li and Yu 2003), with low primary production (Li and Yu 2001), and our monitoring of water quality in 2007 to 2008 showed that the trophic status of this reservoir changed little (trophic level index was 25.8 using the method of Jin and Tu 1990). There was a relation between zoobenthic density and trophic lake status, i.e., the density of zoobenthos in oligo- and mesotrophic lakes is limited by food availability and that in eutrophic lakes by the concentration of dissolved oxygen (Hargrave 2001). The positive correlation between density and primary production in Xin'anjiang Reservoir and the lack of a relationship between density and concentration of dissolved oxygen offer support for the above general pattern.

L. hoffmeisteri is widely distributed and occurs at any trophic scale (Milbrink 1980). It is one of the last two species to remain when organic pollution has become very severe (Aston 1973), reaching very high densities in eutrophic lakes (Milbrink 1994; Milbrink et al. 2002). However, there are also investigations indicating that this animal could be an indicator of oligo- and mesotrophic lakes (Särkkä 1982; Petridis and Sinis 1993; Lang 2007; Cui and Wang 2008). Xin'anjiang Reservoir is oligo- and mesotrophic and *L. hoffmeisteri* is predominant in this lake. Thus, it is suggested that *L. hoffmeisteri* dominates oligo- and mesotrophic lakes as well.

The horizontal spatial distribution (upper, middle, and lower reaches) of density and biomass is not exactly persistent. The density and biomass of oligochaetes in Lake Taiping was highest at lower reaches and lowest at middle reaches, and positively associated with water depth, TN and/or TP in sediment (Liu and Liang 1997). However, the total density of zoobenthos and density and biomass of oligochaetes in the upper reaches of Daoguanhe Reservoir, Hubei Province, was higher than those of the middle and lower reaches (Ma et al. 2004), and the same is true of density and biomass of Horsetooth Reservoir (Edmonds and Ward 1979). But the reasons for their analogous distribution pattern are different: negative relationship between total density of zoobenthos and density and biomass of oligochaetes in Daoguanhe Reservoir and water depth (Ma et al. 2004), but a positive correlation between density and biomass in Horsetooth Reservoir (each sampling site having a similar water depth) and concentration of sediment organic matter (Edmonds and Ward 1979). In Xin'anjiang Reservoir, there was a decreasing tendency from the upper to lower reaches in densities and standing crop of profundal macrozoobenthos. S1 has abundant external nutrients and organic matter, and is productive because it is situated at the upper reaches (the zone of Xin'anjiang River, which accounts for about 60% of total inflow to the reservoir). Moreover, sedimentation amounts of organic matter at this sampling site were larger compared with four other sampling sites for its relatively lower water depth. Therefore, density and biomass at S1 were highest among the five sampling sites. Organic matter is decomposed, water is purified, and primary production and concentration of dead organic matter decreased from the upper (S1), through the middle (S3 and S4), to the lower reaches (S8 and S9). Meanwhile, the sedimentation of organic matter declined with depth (Nalepa 1989). Therefore, the concentration of organic matter in sediment and the density and biomass of zoobenthos decreased along this horizontal gradient.

Several workers have used biological indices to access the water quality of reservoirs in China (Chi et al. 2009; Zhang et al. 2010), such as the Goodnight–Whitley, Wright, Carlander's biomass, and Shannon–Weaver indices. They found that Wright and Carlander's biomass indices assess the water quality of reservoirs well; others, the Goodnight–Whitley and Shannon–Weaver indices, are not suitable (Chi et al. 2009; Zhang et al. 2010). Our investigation supports this observation. Our results also suggest that King index was not suitable for assessing water quality, especially of deep-water reservoirs.

It is worth noting that the biomass of each site in 2009 was obviously larger than in 2007 to 2008. There are two possible reasons. On the one hand, the sampling procedure of the first 2 years was different from the last year. The macrozoobenthos was collected only from the bottom of the profundal in 2007 to 2008, but from both the depths and the shallows in 2009. In shallower sediments, mollusks contributed much to total biomass because of their relatively large individual body weight, especially at S1 versus T1. On the other hand, the preservation methods were different for those 2 sampling periods. The animal would lose more water from body preserved by alcohol compared with formalin.

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Chapter 10
The Ecology of Zoobenthos in Reservoirs of China: A Mini-Review

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Abstract Investigations on the ecology of the zoobenthos in reservoirs of China used to be scant, especially studies on the function of zoobenthos and its role in the ecological processes of reservoir ecosystems. Existing investigations focus on community structures of bottom fauna and their application to bioassessment of water quality and fishery potential. The response of zoobenthos and its subgroups to water depth followed some common patterns, but the effects of thermal stratification were ignored. Zoobenthos density increased with eutrophication over a long period, and vice versa. However, the association between important variables (such as phosphorus and nitrogen concentrations in water and sediment) and zoobenthos in 1-year studies were ambiguous. Few studies were conducted on zoobenthic body content of phosphorus and specific studies explored the effects of crab culturing on zoobenthos community. Relatively intensive investigation efforts were carried out on community succession and its driving factors in the Three-Gorges Reservoir.

10.1 Introduction

China is one of the most important "reservoir" countries in the world, with more than 86,000 reservoirs and 4,660 $\times$ 10^8 m^3 of storage capacity (Liu et al. 2001). Through a rapid development of industry, economics, and society in China, water quality deterioration by eutrophication of feeding rivers and increased sewage

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discharge has become a national problem of resource management. At present, reservoirs are the main source of water supply in many provinces (Han 2010), and it is generally believed that reservoirs are the final alternative to guarantee drinking water safety for China (Han et al. 2006). Unfortunately, water quality of reservoirs in China is degrading, with one-third of the water supply reservoirs already heavily eutrophicated (Meng 2007; Han 2010).

A basic limnological knowledge is important for an efficient management of the water quality and ecological restoration of these reservoir systems. Despite their huge number and importance in China, however, reservoirs have been less studied than lakes and rivers (Han 2010). Here, we briefly review and comment upon the ecology of zoobenthos in reservoirs of China with an aim to promote interest in such studies.

10.2 History, Temporal and Spatial Scales of Zoobenthos Studies

Based on a literature survey in databases of CNKI, ISI Web of Knowledge, ScienceDirect, Springer, and Wiley Online Library, the investigation on reservoir zoobenthos in China is scant, with only about 40 reports published (Fig. 10.1 shows the distribution of the reservoirs in China that have been studied so far). Although there is information about zoobenthos in surveys for fisheries ecology and food organisms, this was not used in this review because of difficulties in collecting such literature and their simple descriptions of zoobenthos (except for Dai and Cao 1999, which gives biomass data for 527 reservoirs). Investigations on reservoir zoobenthos are increasing only slowly, as indicated by the number of research papers across the decades; there were only 3 papers before 1980, increasing to about 8 in the 1980s, 12 in the 1990s, and 20 in the first decade of the twenty-first century.

The time span of most investigations is less than 1 year, with only about ten studies lasting more than 1 year (Xie 1982; Wei et al. 1990, 2006; Liang et al. 1993; Shi et al. 1994; Jiang et al. 1996; Shi 1998; Yu and Jiang 2005; Xiao et al. 2006; Shao et al. 2008a, 2010; Qiu et al. 2009; Zhang et al. 2010a). Few workers conducted long-term work. Exceptions are Shi et al. (1994) and Shi (1998) who monitored the zoobenthos of Dahuofang Reservoir in Liaoning Province for more than 10 years, and Shao et al. (2008a), (2010), Zhang et al. (2010a) who produced in-depth reports on macroinvertebrates of the Three-Gorges Reservoir for a long time.

On the spatial scale, most of investigations were designed for a single reservoir, and there is a lack of comparative study among reservoirs. Four case studies in Fujian, Jiangsu, and Hubei Province are exceptions, where comparative analyses about zoobenthos were conducted on 33, 48 and 6, and 4 reservoirs, respectively (Yang and Lu 1987; Huang et al. 1995; Wan et al. 2004; Lv et al. 2009).

Fig. 10.1 Distribution of reservoirs in China investigated for macrozoobenthos (*1* Hamatong Reservoir; *2* Nihe Reservoir; *3* Donghu Reservoir; *4* Daqing Reservoir; *5* Hongqi Reservoir; *6* Xianghai Reservoir; *7* Erlonghu Reservoir; *8* Qinghe Reservoir; *9* Dahuofang Reservoir; *10* Tanghe Reservoir; *11* Biliuhe Reservoir; *12* Miyun Reservoir; *13* Tuanbo Reservoir; *14* Hongshadun Reservoir; *15* Liujiaxia Reservoir; *16* Sanmenxia Reservoir; *17* Suyahe Reservoir; *18* Nanwan Reservoir; *19* Danjiangkou Reservoir; *20* Sandaohe Reservoir; *21* Fuqiaohe Reservoir; *22* Sanxia Reservoir; *23* Xiangxihe Bay; *24* Xujiahe Reservoir; *25* Jinshahe Reservoir; *26* Taoyuanhe Reservoir; *27* Daoguanhe Reservoir; *28* Fushui Reservoir; *29* Heilongtan Reservoir; *30* Xianghongdian Reservoir; *31* Taipinghu Reservoir; *32* Xiaotashan Reservoir; *33* Anfengshan Reservoir; *34* Cuihezhuang Reservoir; *35* Longwangshan Reservoir; *36* Jinniushan Reservoir; *37* Ersheng Reservoir; *38* Shahe Reservoir; *39* Qingshan Reservoir; *40* Wuhu Reservoir; *41* Dongjiang Reservoir; *42* Dongxi Reservoir; *43* Nanxi Reservoir; *44* Liutang Reservoir; *45* Dongzhang Reservoir; *46* Shanmei Reservoir; *47* Fengtou Reservoir)

10.3 Structure of the Macrozoobenthic Community

Most researchers have mainly reported the community structures of reservoir zoobenthos: species numbers and composition ranged from 5 to 61, density averaged from 68 to 6,150 ind. m^{-2}, and biomass ranged from 0.02 to 37.2 g m^{-2}. The average biomass for 527 reservoirs of 25 provinces investigated from 1980 to 1994 was 2.41 ± 0.53 (SE) g m^{-2} (Dai and Cao 1999). Oligochaetes and chironomids were the main components, and mollusks occasionally dominated reservoir zoobenthos (Yang and Lu 1987). The dominant taxa were *Limnodrilus hoffmeisteri* (about 13 literature records), *Procladius* (8 records), *Tanypus* (7 records), *Aulodrilus* (4 records), and *Cryptochironomus* (3 records). The occasionally dominant taxa included *L. helveticus*, *L. claparedianus*, *Nais inflata*, and one or two species of *Micropsectra*, *Branchiura*, *Tanytarsus*, *Propsilocerus*, *Tubifex*, and the *Chironomus plumosus* group.

10.4 Relationship Between Zoobenthos and Environmental Variables

There are a few investigations on the association between zoobenthos and environmental variables, and although relatively more investigations explored the response of zoobenthos to water depth, few investigations were designed for the effects of water chemistry and sediments on zoobenthos.

The zoobenthic biomass of Daoguanhe and Sandaohe Reservoirs decreased as water depth increased (Ma et al. 2004; Chi et al. 2009). The reverse was found in Tanghe Reservoir (Qiu et al. 2009). These contrasting patterns can be explained by the observation that different groups differed in their relative dominance and in their response to depth change. Both density and biomass of aquatic insects decreased with depth in Fushui, Taipinghu, and Tanghe Reservoirs, and in 4 reservoirs in Hubei Province and 33 in Fujian Province (Luo et al. 1988; Huang et al. 1995; Liu and Liang 1997; Lv et al. 2009; Qiu et al. 2009). The patterns of density and biomass of oligochaetes with depth were diverse. Oligochaete density and biomass increased with depth in some studies (e.g. Liu and Liang 1997); in some cases they increased first, then declined with water depth, but sometimes they appear as water depth continued to increase (Luo et al. 1988; Huang et al. 1995; Jiang et al. 1996; Qiu et al. 2009). In one investigation (Zu 1984) they decreased first (at the 10–30 m depth range) before rising to the maximum at a depth range of 30–40 (for density) or 40–50 m (for biomass), and then again dropped. Liu and Liang (1997) suggested that water depth may play a more important role than sediment type. Analyses of the response of taxa richness of the whole zoobenthos community and aquatic insects to water depth in Qingshan Reservoir, Zhejiang Province, showed that taxa richness decreased with depth (e.g. Liu and Liang 1997; Yu 2001; Lv et al. 2009). The change of zoobenthos with water depth was contingent upon trophic level and thermal stratification (Brinkhurst 1974), so the two factors should be taken into account in understanding the responses of zoobenthos to water depth.

Other physical and chemical factors influencing zoobenthos were sediment type, turbidity, and concentration of total phosphorus and nitrogen in water and sediment (Xiang 1990; Liu and Liang 1997; Shao et al. 2007; Lv et al. 2009). The density and biomass of oligochaetes and aquatic insects were positively associated with total phosphorus and nitrogen concentration in water in four reservoirs of Hubei Province (Lv et al. 2009), and oligochaetes in Taipinghu Reservoir (Liu and Liang 1997) correlated with these two parameters in sediments. The zoobenthos in Wuhu Reservoir (Xiang 1990) was related to sediment type with maximum numbers in silt, middle in clay, and minimum in stiff mud. Turbidity was responsible for community differences in bays of the Three-Gorges Reservoir, and these differences could not be explained by variables about sedimentation (Shao et al. 2007, 2008b). From 2005 (the second year of the second impoundment stage) onward, hydrological factors began to exert important impacts on benthic community: Tubificidae positively correlated with inflow discharge, Naididae responded negatively to inflow discharge and positively to transparency, and residence time had a significant effect on community biodiversity (Zhang et al. 2010a). Some results from the literature are not in accordance with those above. For example, the variation of zoobenthic biomass in Daoguanhe Reservoir, Hubei Province, could not be explained by total phosphorus and nitrogen in sediments (Ma et al. 2004), while the biomass of bottom fauna in Biliuhe Reservoir, Liaoning Province, increased with a decrease in total phosphorus and nitrogen in the water (Liang et al. 1993). This inconsistency indicates that other important variables affected the variations of zoobenthic biomass.

The response of zoobenthos to eutrophication appears to show a general pattern. Density and/or biomass in Taipinghu Reservoir (biomass from 0.56 g m^{-2} in 1985 to 1986 to 3.90 g m^{-2} in 1992–1993), Daoguanhe Reservoir (density from 40 ind. m^{-2} in 1980 to 142 ind. m^{-2} in 1999, biomass from 0.12 to 0.99 g/m^2) and Tuanbo Reservoir (density from 306 ind. m^{-2} in 1982 to 1,200 ind. m^{-2} in 2006, biomass from 1.45 to 2.74 g m^{-2}) expanded with eutrophication (Liu and Liang 1997; Ma et al. 2004; Wang et al. 2008). Nevertheless, some conclusions should be drawn prudently because not all data used for comparisons were collected all year round. In contrast, density in Dahuofang Reservoir decreased from 3,335 ind. m^{-2} in 1980 to 897 ind. m^{-2} in 1996 as the degree of pollution declined (Shi 1998). The biodiversity and taxa composition of macrozoobenthic communities also varied with trophic status. The taxa richness of Dahuofang Reservoir in 1996 increased to 2.8 times that in 1980 with pollution alleviation (Shi 1998), and the dominant zoobenthic group in Tuanbo Reservoir shifted from *Chironomus sinicus* to *Tubifex* (Wang et al. 2008).

10.5 Response of Zoobenthos to Anthropogenic Disturbance

Few studies were designed to explore the impact of anthropogenic disturbance, such as hydraulic engineering and farming of aquatic animals on reservoir zoobenthos. Chironomids and oligochaetes significantly declined numerically but mollusks changed little under low farming density of *Eriocheir sinensis*. However, density and biomass of the three animal groups were all depressed under high intensity farming (Yu and Jiang 2005). The mollusks almost disappeared after the first 3-year inundation in Dongjiang Reservoir of Hunan Province (Hu 1994). Yearly successions and seasonal variations of the Three-Gorges Reservoir and the responsible factors were investigated intensively. Mayflies and caddisflies were the common groups (Xie 1987), and oligochaetes were dominant numerically (Liang 1987) before the construction of Three-Gorges Reservoir. During the first year of impoundment, chironomids became dominant in number and, together with oligochaetes, dominated taxa composition. After 1 year of reestablishment, oligochaetes increased and chironomids decreased numerically, and there occurred seasonal shifts in community types with a *Nais–Polypedium* community in spring and winter and a *Limnodrilus* community in summer and autumn (Shao et al. 2008a). The density of zoobenthos in Xiangxi Bay of the Three-Gorges Reservoir in the second year after the initial closure of the dam (in 2003) was about 18 times that in the first year. The community was composed almost exclusively of oligochaetes during the second investigation year as silts accumulated on the bottom (Shao et al. 2006), and zoobenthic seasonal patterns stabilized after the second impoundment stage (i.e., after 2005) (Zhang et al. 2010a).

10.6 Application of Zoobenthos to Bioassessments

Many researchers have tried to use zoobenthos for the assessment of the water quality of reservoirs. The biological indices and methods used include Shannon–Weaver diversity, evenness, Simpson dominance, Margalef richness, Goodnight–Whitley, Wright index, Carlander's biomass, BPI index, and indicator species. Results from four different trophic-level reservoirs indicated that the Wright, Goodnight–Whitley, and evenness indices were acceptable while the Shannon–Weaver, Margalef, and Simpson dominance indices were not fit for water quality assessment of those reservoirs (Lv et al. 2009). Zhang et al. (2010b) suggested that the Shannon–Weaver, Wright index, and Carlander's biomass were more effective than the Goodnight–Whitley index in the bioassessment of water quality for Danjiangkou Reservoir. Chi et al. (2009) argued that water quality in Sandaohe Reservoir based on Wright index and Carlander's biomass rather than Shannon–Weaver, Margalef, and Goodnight–Whitley indices conformed more to the actual situation. The biological assessment of water quality of 48 reservoirs in Jiangsu Province based on Wright index was similar with that using the Goodnight–Whitley index (Wan et al. 2004), although assessments by the Wright index were closer to the status of the 33 reservoirs in Fujian Province (Huang et al. 1995). In summary, Wright index seems to perform better than all others that have been tried in the bioassessment of water quality of Chinese reservoirs.

10.7 Application of Zoobenthos to Fisheries

In total, 12 papers from the literature estimate the influence of zoobenthos on fishery yield, which ranged from 65 to 4,000 kg km^{-2} with average 1141 kg km^{-2} (SD: 1241). Few investigations deal with the influence of bottom fauna in crab culture and with the exploitation of zoobenthos by valuable fish species. The farming density of *E. sinensis* in Donghu Reservoir, Heilongjiang Province, was not more than 56 kg km^{-2} (Yu and Jiang 2005). The 12 zoobenthic species of the macrozoobenthic community were all part of the diet of *Acipenser baerii* grown in Hongshadun Reservoir, Gansu Province (Huang et al. 2010).

10.8 Other Investigations

Besides the investigations mentioned, there are few other theoretical investigations and applied research. The body content of phosphorus in chironomids and oligochaetes increased exponentially with phosphorus concentration in sediments, and chironomids and zoobenthos as a whole were independent of phosphorus contents in water (Fu et al. 2005; Wang et al. 2006). Shao et al. (2010) applied macroinvertebrates to studying longitudinal zonation of Xiangxihe bay of Three-Gorges Reservoir. Four distinct zones were found, and the mainstream zone was characterized by a lower biomass and greater instability of the macrozoobenthic community than the lacustrine zone, and that lacustrine zone developed only where the bay was sufficiently long.

10.9 Summary

Chironomids and oligochaetes were the main components of the zoobenthos of most reservoirs, with *L. hoffmeisteri*, *Procladius*, and *Tanypus* usually dominant. More investigations deal with the association between zoobenthos and water depth, than with research on the relationship between bottom fauna and other environmental variables. Aquatic insects were negatively related to water depth, and oligochaetes increased first and then decreased, and sometimes increased again with continuing water depth. Wright index appeared the most suitable for the bioassessment of water quality compared with other indices. The fishery potential in reservoirs provided by zoobenthos as a food resource ranged from 65 to 4,000 kg km^{-2}. Few investigations were designed to explore the effects of aquaculture on zoobenthos, the animal's body content of phosphorus and its controls. The Three-Gorges Reservoir was intensely investigated, focusing on yearly succession and seasonal change of the zoobenthic community and its driving factors. On the whole, the basic study of reservoir zoobenthos is scant, and the timescale of investigation of most reports is not more than 1 year. Most investigations emphasize community structure. The functions of zoobenthos and its role in reservoir ecosystems are ignored. Comparisons between reservoirs, the relationship between zoobenthos and environmental factors, and the effects of water level and hydrological variables on bottom faunas will need to be taken into account in the future.

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Appendix

Taxa list of macrozoobenthos encountered in the studied reservoirs of China

Code	Taxa	Code	Taxa	Code	Taxa
1	*Dugesia*	78	*Pseudamnicda subangulatus*	155	*Chironomus salinarius*
2	Nematoda	79	*Radix auricularia*	156	*C. semireductus*
3	*Aeolosoma*	80	*R. chefouensis*	157	*C. sinicus*
4	*Allonais gwaliorensis*	81	*R. clessini*	158	*C. thurnmi*
5	*Arcteonais lomondi*	82	*R. lagotis*	159	*C. stigmaterus*
6	*Aulodrilus limnobius*	83	*R. ovata*	160	*C. dorsalis*
7	*A. pigueti*[3]	84	*R. plicatula*	161	*Cladopelma*
8	*A. pluriseta*	85	*R. sinensis*	162	*Cladotanytarsus*
9	*Aulophorus furcatus*	86	*R. swinhoei*	163	*Clinotanypus nervosus*
10	*Bothrioneurum vejdovskyanum*	87	*R. whartoni*	164	*Coelotanypus*
11	*Branchiodrilus hortensis*	88	*Rivularia*	165	*Cricotopus trifasciatus*
12	*Branchiura sowerbyi*	89	*Semisulcospira cancellata*	166	*Cryptochironomus anomalus*
13	*Chaetogaster*	90	*S. gredleri*	167	*C. conjugen*
14	*Dero digitata*	91	*Stenothyra glabra*	168	*C. defectus*
15	*D. nivea*	92	*Tricula humida*	169	*C. digitatus*
16	*Friderica bulbosa*	93	*Viviparus chui*	170	*C. fuscimanus*
17	*Ilyodrilus templetoni*	94	*Acuticosta chinensis*	171	*C. viridulus*
18	*Limnodrilus claparedeianus*	95	*Anodonta arcaeformis*	172	*Cryptotendipes*
19	*L. grandisetosus*	96	*A. arcaeformis flavotincta*	173	*Demicryptochironomus*
20	*L. helveticus*	97	*A. globosula*	174	*Dicrotendipes nervosus*
21	*L. hoffmeisteri*	98	*A. woodiana elliptica*	175	*Einfeldia dissidens*
22	*L. udekemianus*	99	*A. woodiana pacifica*	176	*E. pagana*
23	*Lumbriculus variegatus*	100	*A. woodiana piscatorum*	177	*Endochironomus nigricans*
24	*Monopylephorus limosus*	101	*Corbicula fluminea*	178	*Endotribelos*
25	*Nais communis*	102	*C. largillierti*	179	*Eukiefferiella quadridentata*
26	*N. inflata*	103	*C. nitens*	180	*Glyptotendipes lobifera*
27	*N. pardalis*	104	*Cristaria plicata*	181	*G. tokunagia*
28	*N. simplex*	105	*Cuneopsis heudei*	182	*Hamischia*
29	*N. variabilis*	106	*Hyriopsis cumingii*	183	*Hydrobaenus kondoi*
30	*Paranais frici*	107	*Lamprotula scripta*	184	*Lauterborniella*
31	*Pristina aequiseta*	108	*L. chiai*	185	*Limnochironomus*
32	*P. leidyi*	109	*L. cornuum-lunae*	186	*Lipiniella*
33	*P. osborni*	110	*L. elongata*	187	*Macropelopia notata*
34	*P. proboscidea*	111	*L. leai*	188	*Microchironomus* sp.
35	*P. longisoma*	112	*L. leleci*	189	*Micropsetra logana*
36	*Rhyacodrilus brevidentatus*	113	*L. polysticta*	190	*M. praecox*

(continued)

37	*R. sinicus*	114	*L. rochechouarti*	191	*Orthocladius*
38	*Slavina appendiculata*	115	*L. spuria*	192	*Parachironomus*
39	*Spirosperma*	116	*L. tortuosa*	193	*Parakiefferiella*
40	*Stephensoniana trivandrana*	117	*Lanceolaria gladiola*	194	*Paralauterborn*
41	*Stylaria fossularis*	118	*Lepidodesma languilati*	195	*Paratanytarsus*
42	*Tasserlodrilus kessleri*	119	*Limnoperna lacustris*	196	*Pentaneura*
43	*Teneridrilus mastix*	120	*Novaculina chinensis*	197	*Polypedilum aberrans*
44	*Tubifex ignotus*	121	*Pisidium subtruncatum*	198	*P. aviceps*
45	*T. tubifex*	122	*Ptychorhynchus pfisterei*	199	*P. beckae*
46	*Vejdovskyella comata*	123	*Schistodesmus lampreyanus*	200	*P. brevian-tennatum*
47	*Nephthys*	124	*S. spinosus*	201	*P. convicrum*
48	*Tylorrhynchus heterochaetus*	125	*Solenaia oleivora*	202	*P. flavum*
49	*Barbronia johansson*	126	*Sphaerium japonicum*	203	*P. halterale*
50	Erpobdellidae	127	*S. lacustre*	204	*P. illinoense*
51	*Helobdella nuda*	128	*Unio douglasiae*	205	*P. laetum*
52	*Hirudo*	129	*Caenis*	206	*P. leucopus*
53	*Alocinma longicornis*	130	*Ecdyonurus toliironis*	207	*P. scalaenum*
54	*Bellamya aeruginosa*	131	*Ephemera wuchonzensis*	208	*P. trigonus*
55	*B. purificata*	132	*Aeschna*	209	*P. tritum*
56	*B. quadrata*	133	*Gomphus abdominalis*	210	*Procladius choreus*
57	*Bithynia fuchsianus*	134	*Phylocentropus*	211	*Propsilocerus akamusi*
58	*B. misella*	135	*Cybister*	212	*Pseudochironomus*
59	*Cipangopaludina cahayensis*	136	*Eretes*	213	*Rheotanytarsus exiguus*
60	*C. chinensis*	137	*Laccotrephes japonensis*	214	*Stenochironomus*
61	*C. ussuriensis*	138	*Notonecta triguttata*	215	*Stictochironomus*
62	*Cyriulus*	139	*Ranatra chinensis*	216	*Stilobezzia*
63	*Galba laticallosiformis*	140	*Tabanus*	217	*Tanypus chinensis*
64	*G. pervia*	141	*Chaoborus*	218	*T. punctipennis*
65	*G. truncatula*	142	*dixa*	219	*Tanytarsus formosanus*
66	*Gyraulus albus*	143	*Tipula*	220	*Tenaipes thummi*
67	*G. compressus*	144	*Ephydra*	221	Amphipoda
68	*G. convexiusculus*	145	*Palpomyia*	222	Isopoda
69	*Hippeutis cantori*	146	*Probezzia*	223	*Caridina denticulata sinensis*
70	*H. umbilicalis*	147	*Bethbilbeckia*	224	*C. nilotica gracilipes*
71	*Lymnaea*	148	*Calopsectra*	225	*Neocaridina denticulata*
72	*Melanoides*	149	*Chaetocladius sexpapilosus*	226	*Macrobrachium nipponensis*
73	*Parafossarulus eximius*	150	*C. attenuatus*	227	*M. qilianensis*

(continued)

74	*P. sinensis*	151	*C. bathopilus*	228	*Palaemon modestus*
75	*P. striatulus*	152	*C. plumosus*	229	*Palaemoneis sinensis*
76	Physidae	153	*C. plumosus-reductus*		
77	*Polypyis hemisphaerala*	154	*C. ruductus*		

1 Platyhelminthes *2* Nemathelminthes, *3–46* Oligochaeta, *47–48* Polychaeta, *49–52* Hirudinea, *53–93* Gastropoda, *94–128* Bivalvia, *129–131* Ephemeroptera, *132–133* Odonata, *134* Trichoptera, *135–136* Coleoptera, *137–139* Hemiptera, *140–220* Diptera (*140* Tabanidae, *141* Culicidae, *142* Dixidae, *143* Tipulidae, *144* Ephydridae, *145–146* Ceratopogonidae, *147–220* Chironomidae), *221* Amphipoda, *222* Isopoda, *223–229* Decapoda

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Part II
Environment

Chapter 11
Biogeochemical Cycling of Mercury in Hongfeng Reservoir, Guizhou, China

Tianrong He and Xinbin Feng

Abstract Mercury accumulation in aquatic food chains is a global public health concern because of the dangers of human exposure to methylmercury. Hongfeng reservoir, situated in Guizhou Province, southwestern China, is a water body that suffers from mercury pollution. Its source of mercury is anthropogenic and mainly includes industrial discharge from coal-fired power plants, chemical plants using mercury as a catalyst, and atmospheric deposition. Here, we report on the temporal and spatial distribution of different mercury species in the water of the reservoir and analyze possible effects of eutrophication on the biogeochemical cycling of mercury. Hyper-eutrophication of the reservoir affected the concentration and distribution of mercury species in the water through algal blooms. Microalgae have a large capacity to bind mercury, and represent a substantial pool of mercury in the aquatic system. Hyper-eutrophication results in low DO and high DOC that accelerates the formation of MeHg in the hypolimnion, especially in summer. Hongfeng reservoir is a large net sink of total mercury and a net source of MeHg. The mercury cycling at the sediment–water surface indicates that sediment dominates the fate of both total mercury and methylmercury, but the fluxes of mercury diffusing from pore water contribute only weakly to mercury in water. The MeHg-enriched water discharged from the anoxic hypolimnion poses a serious risk to downstream ecosystems.

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11.1 Introduction

Mercury accumulation in aquatic food chains is a global public health concern, because it is the predominant pathway of human exposure to methylmercury, one of the most toxic forms of mercury (Tchounwou et al. 2003). Understanding the cycling of mercury in the aquatic environment and its food chains is therefore an urgent task in applied limnological research.

Hongfeng reservoir is one of many water bodies in China that suffer from mercury pollution. This reservoir, constructed in 1960, has a surface area of 57.2 km^2 and a volume of 6.01×10^8 m^3, and is located in a suburb of Guiyang City, Guizhou Province, Southwestern China (Fig. 11.1). It was constructed for hydroelectric power generation, flood control, tourism, drinking-water supply, and fishery production. A large coal-fired power plant (300 MW) situated on the southeast bank of the reservoir is the main source of mercury contamination to the reservoir. In addition, a chemical plant located in the vicinity of the city of Qingzhen used mercury as a catalyst for the production of acetic acid until 1986. In total, more than 140 t of Hg had been released to the environment and a considerable amount of it was discharged to Baihua reservoir which is connected to Hongfeng reservoir through a channel (Horvat et al. 2003). Although the chemical plant is located at the downstream end of Hongfeng reservoir, a considerable amount of mercury, emitted to the atmosphere, was eventually deposited to the reservoir surface. Some of this atmospheric mercury, deposited to the rice fields around the reservoir, converted to MeHg and entered Hongfeng Reservoir with surface runoff. It is therefore imperative to understand the biogeochemical cycling of mercury in Hongfeng reservoir to evaluate the mercury contamination to the food web in the reservoir.

There are nearly two dozen factories in the drainage area of Hongfeng Reservoir, and a large volume of improperly treated waste-water, discharged to the reservoir, has long turned it into a eutrophic water-body (Zhang 1999). Episodes of high waste-water loading have caused a series of environmental accidents in Hongfeng Reservoir (Zhang 1999). Approximately 230,000 kg fish died in a few days in 1994 because of contamination (Zhang 1999). In 1995 the reservoir water turned black from organic matter, accompanied by a bloom of microalgae (Zhang 1999). About 500 people who drank water from the reservoir were diagnosed with arsenic poisoning in 1996 (Zhang 1999). Large masses of algae (algal blooms) appeared in 1996 and 1997 (Zhang 1999), and again in May 2004 in the Houwu basin of the reservoir (Lu 2004), one of our sampling sites. A large number of studies have been carried out to investigate the biogeochemical cycling of nutrients and eutrophication of the reservoir (e.g., Xiao and Liu 2004; Liang et al. 2004a, b). Eutrophication alters the biogeochemical cycles of many elements, but its impact on the cycle of mercury species has, to date, not been evaluated.

Eutrophication may notably play an important role in mercury transport and immobilization (Coelho et al. 2005). Eutrophication can induce a mass development of microalgae, which may represent a substantial pool of mercury as a result of their high growth rate and capacity to bind trace metals (Radway et al. 2001).

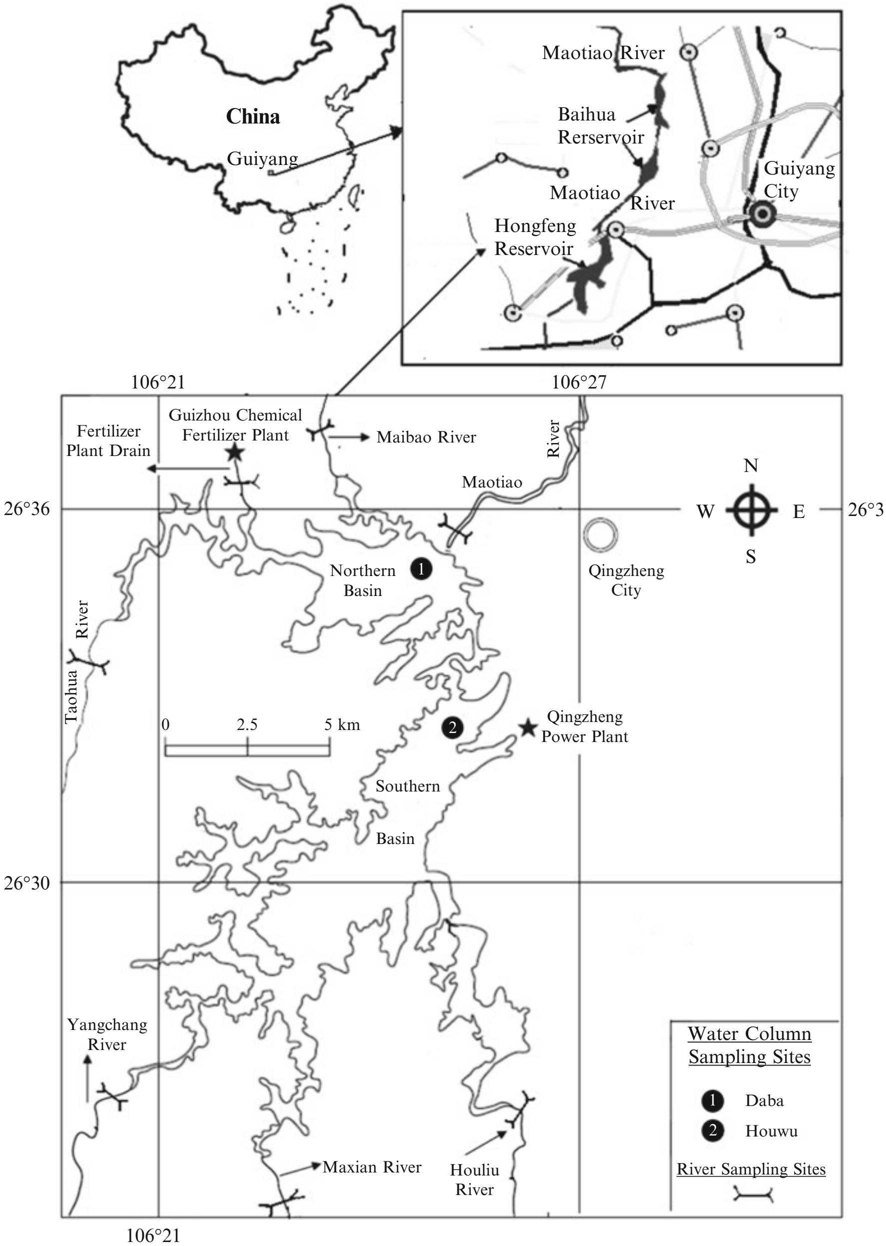

Fig. 11.1 Location of sampling sites (Daba = the dam)

When microalgae move with water currents and/or settle to the sediment, mercury becomes redistributed. More importantly, eutrophication alters the chemical forms and bioavailability of mercury in water. Algal blooms modify redox potentials and pH, and affect the concentrations of Fe and Mn compounds, sulfur, and carbon (Eggleton and Thomas 2004). These changes in turn affect the chemical forms of mercury, and change the bioavailability of mercury in water. Fish in eutrophic water bodies, however, have often been found to contain less mercury than in oligotrophic ones. This difference in mercury content can be explained by the increasing algal biomass in eutrophic systems that reduces mercury accumulation at higher trophic levels through a dilution of mercury in algal cells (Pickhardt et al. 2002). The increased microbial activities, increased anaerobiosis, and increased concentrations of fulvic acid resulting from eutrophication are likely to increase the methylation rate of mercury in such waters.

Hongfeng reservoir is located in an area with a serious acid rain problem due to coal combustion emissions (Zhou et al. 1997). The main bedrock types of the watershed are limestone and dolomite (Zhang 1999). As a result, Hongfeng reservoir is an alkaline reservoir, seasonally anoxic, and eutrophic. This feature of Hongfeng reservoir provides a good opportunity to understand the effects of eutrophication on the behavior of mercury, and to understand the effect of acid precipitation on mercury cycling and enrichment in food chains in alkaline reservoirs. In this study, we describe the temporal and spatial distribution of different mercury species in the water of the reservoir.

11.2 Materials and Methods

11.2.1 Sampling Sites

Two sampling sites in the pelagic zone, six in the inflows and one at the outflow were selected (Fig. 11.1). The Houwu sampling site was situated in the south basin of the reservoir where the coal-fired power plant was located on the southeast bank of the reservoir. A large fish farm was also located in this area because of the higher water temperature caused by the discharge of hot water from the power plant. The sampling site close to the dam was located in the north basin of the reservoir, which received discharge from the chemical fertilizer plant. As the site is near the dam of the reservoir, the water level here fluctuated markedly. Water depth was up to 32 m in autumn and winter, but decreased to 17 m in spring and summer. Yangchang, Maxian, Taohua Rivers are the main water sources of Hongfeng reservoir (Zhang 1999). Because of contamination with wastewater, the Yangchang and Maibao Rivers and the fertilizer plant rain were more heavily contaminated than the Maxian, Taohua, and Houliu Rivers (Zhang 1999). The Maotiao River is the only outflow of the reservoir. The water discharged from Hongfeng reservoir originates mainly from an outlet at the deep water of the reservoir.

11.2.2 Sample Collection and Analysis

Unfiltered and filtered water samples, sediment samples, pore water samples, and fish samples were collected from Hongfeng reservoir. Four sampling campaigns were conducted in November 2003, and February, May, and September 2004, representing winter, spring, summer, and autumn seasons, respectively. Water samples in the reservoir were taken vertically from different depths of the water column (0, 4, 8, 12, 16, 20, 24, 28, and 30 m), while only surface water was sampled in the inflows and outflow of the reservoir. The samples were collected by filtering over with a 0.45-μm filter (Millipore) in situ. All water samples were kept in borosilicate glass bottles, then acidified with 0.5% HCl, double-bagged and transported to the laboratory within 24 h, and stored at 3–4°C in the dark until analysis. All borosilicate glass bottles for collecting samples and analysis were cleaned by acid leaching, rinsing with ultrapure deionized water (18 MΩ-cm) and heating for several hours in a muffle furnace at 500°C.

Sediment cores 18–30 cm long were collected at four sampling occasions. The cores were sectioned at 1- and 2-cm intervals (1-cm intervals for core dating and 2-cm intervals for chemical analysis). Each core section was sliced into 45-mL centrifuge tubes in nitrogen gas. All samples were transported in an ice-cooled container to the laboratory and stored in a refrigerator at 3–4°C. Pore water was extracted from wet sediment within 48 h by centrifugation at 3,000 r/min for 30 min at in situ bottom-water temperature (5°C), and then filtered through a 0.45 μm PVDF membrane (Millipore). The resulting pore water was collected in borosilicate glass bottles and acidified using a 0.5% HCl solution. All bottles were capped and sealed with parafilm. The whole process was performed in a nitrogen bag and acid-cleaned filters were rinsed with de-oxygenated reagent-grade water immediately prior to sample filtration (Mason et al. 1998). All resulting pore water samples were stored in a refrigerator at 3–4°C until analysis. Solid phase samples were freeze-dried and homogenized with a mortar.

Total Hg (THg), reactive Hg (RHg), dissolved Hg (DHg), dissolved gaseous mercury (DGM), total methylmercury (TMeHg), and dissolved methylmercury (DMeHg) were analyzed in each sample. The analytical methods used for Hg speciation in water have been described in detail elsewhere (Bloom and Fitzgerald 1988; Bloom 1989; Horvat et al. 1993; US EPA 2001, 2002; Yan et al. 2003). All methods relied on cold vapor atomic fluorescence detection.

For THg and DHg, water samples were oxidized with 0.5% BrCl. After oxidation, $NH_2OH{\cdot}HCl$ was added to destroy free halogens before adding stannous chloride ($SnCl_2$) to convert Hg(II) to volatile Hg(0). The resulting sample was then purged with Hg-free N_2 and Hg(0) was absorbed onto a gold trap (Bloom and Fitzgerald 1988; Yan et al. 2003).

RHg was determined by addition of 20% $SnCl_2$ to unfiltered and acidified samples, followed by the purge/trap CVAFS method. The RHg determined would include mostly ionic (Hg^{2+}) plus DGM, labile organic fractions and Hg that was leachable from the particulate matter in the sample (Dalziel 1995).

Dissolved gaseous mercury is composed primarily of Hg(0), so it was assumed that DGM was essentially Hg(0) in this study. Approximately 500 mL of a fresh water sample was purged with mercury-free nitrogen gas for 30 min and elemental mercury in the sample was trapped on a gold trap on site. The gold traps were taken to the laboratory for DGM determination within 12 h (Yan et al. 2003).

TMeHg and DMeHg concentrations in water were determined using the standard distillation-ethylation-GC separation-CVAFS technique (Bloom 1989; USEPA 2001). A 45-mL aliquot of acidified sample was placed in a fluoropolymer distillation vessel and the distillation was carried out at 125°C under Hg-free N_2 flow until approximately 35 mL of water was collected in the receiving vessel. The sample collected was adjusted to pH 4.9 with an acetate buffer and the Hg in the sample was ethylated in a closed 200-mL bubbler by the addition of sodium tetraethyl borate. The ethyl analog of CH_3Hg, $CH_3CH_3CH_2Hg$, was separated from solution by purging with N_2 onto a Tenax trap. The trapped $CH_3CH_3CH_2Hg$ was then thermally desorbed, separated from other mercury species by an isothermal GC column, decomposed to Hg (0) in a pyrolytic decomposition column (700°C) and analyzed by CVAFS. All PTFE vials for distillation were cleaned by heating for 48 h in concentrated nitric acid (Horvat et al. 1993).

THg in sediment was measured following the procedure of Fleck et al. (1999). Sediment samples of ~0.2 g were placed in acid-cleaned 30-mL Teflon digestion bombs. A volume of 10 mL concentrate sulfuric acid and 10 mL of concentrate nitric acid were added. The bombs were sealed tightly and placed in an oven at 45°C overnight. The acids were neutralized using hydroxylamine solution before an appropriate volume (generally 0.4 mL) of digested sample was transferred to a borosilicate bubbler for mercury analysis following the procedure described previously.

Analysis of MeHg in sediment was performed following the procedure developed by Liang et al. (2004a, b). Approximately 0.3 g of sediment was placed into a 50-mL centrifuge tube; 1.5 mL of 1 M $CuSO_4$, 7.5 mL of 3 M HNO_3 and 10 mL of CH_2Cl_2 were added. The tube was closed and shaken for 30 min. Five milliliter of the CH_2Cl_2 layer was pipetted into another 50-mL centrifuge tube after the tube was centrifuged at 3,000 rpm for 30 min. About 40 mL of double-deionized water was added to the tube. The tube was heated at 45°C in a water bath until no visible solvent was left in the tube and the remaining liquid was then purged with nitrogen for 8 min in a water bath at 80°C to remove solvent residue. The sample was brought to 50 mL with double-deionized water before an appropriate volume (generally 15 mL) of the sample was transferred to a borosilicate bubbler for methylmercury analysis following the procedure described previously.

Dissolved organic carbon (DOC), total suspended particles (TSP), and water quality parameters such as pH, temperature (T), dissolved oxygen (DO), and total dissolved solid (TDS) were also measured. DOC was measured by the high-temperature combustion method (Cosovic et al. 2000). Water quality parameters such as T, DO, pH, and TDS were measured by a portable analyzer (Radiometer Analytical) on site. Chlorophyll and total dissolved phosphorous (TDP) concentrations in water samples were analyzed in February, April, June, and August 2004. Chlorophyll was measured by spectrophotometric analysis of acetone-extracted filters (Jing and Tu 1990). TDP was measured by potassium phosphate digestion followed by ammonium molybdate spectrophotometric analysis (China EPA 2002).

Quality assurance and quality control of the analytical process were carried out using duplicates, method blanks, field blanks, and matrix spikes. Field blanks and duplicates were taken regularly (>10% of samples) throughout each sampling campaign. Detection limits were estimated as three times the standard deviation of the blank measurement and are 0.004 ng/L for DGM, 0.02 ng/L for RHg, 0.10 ng/L for DHg, and 0.009 ng/L for DMeHg, respectively. The relative standard deviations (RSD) on precision tests for the duplicate samples varied from 1.1% to 12.5% for MeHg analysis, and were <8% for inorganic mercury species analysis. Recoveries for matrix spikes were in the range of 88.2–110% for MeHg analysis, and 86.1–110.3% for inorganic mercury analysis.

11.3 Results and Discussion

11.3.1 Physical and Chemical Characteristics of the Water

In November and February, there were no discernable differences in water temperature, pH, and DO distributions in the water column, which demonstrated that the water in the reservoir was well mixed in the autumn and winter seasons (Fig. 11.2). In May, however, an anaerobic layer developed in the deep water. The pH and DO distributions showed significantly different vertical profiles. Both pH and DO were at maximum at the surface because of the algal bloom, but began to decrease in deep water because of stratification of the water column. In September, as decomposing dead algae depleted dissolved oxygen, the concentration of DO decreased in the whole water column, especially in the hypolimnion. There was a sharp decrease in DO between 8- and 12-m depth, showing that the reservoir was well stratified.

The pH and DO were significantly lower in the outflow than in the inflows, indicating that water quality had been significantly changed by the reservoir (Table 11.1). TDP ranged from the detection limit to 0.32 mg/L (Table 11.2). Concentrations of chlorophyll ranged from 1.01 to 45.9 μg/L. According to the OECD (Organization for Economic Cooperation and Development) standard for eutrophication of lakes, the average concentrations of chlorophylla at Houwu in February (16.9 μg/L), April (15.4 μg/L), and August (14.9 μg/L) exceeded the threshold for eutrophication (14.3 μg/L). The average total nitrogen (1.90 mg/L, data from EPA of Guizhou, China) at Houwu in 2004 also exceeded the threshold for eutrophication (1.5 mg/L).

TSP concentrations were generally low, and ranged from 0.8 to 5.8 mg/L with an average concentration of 2.1 mg/L in all samples except in May 2004 at Houwu

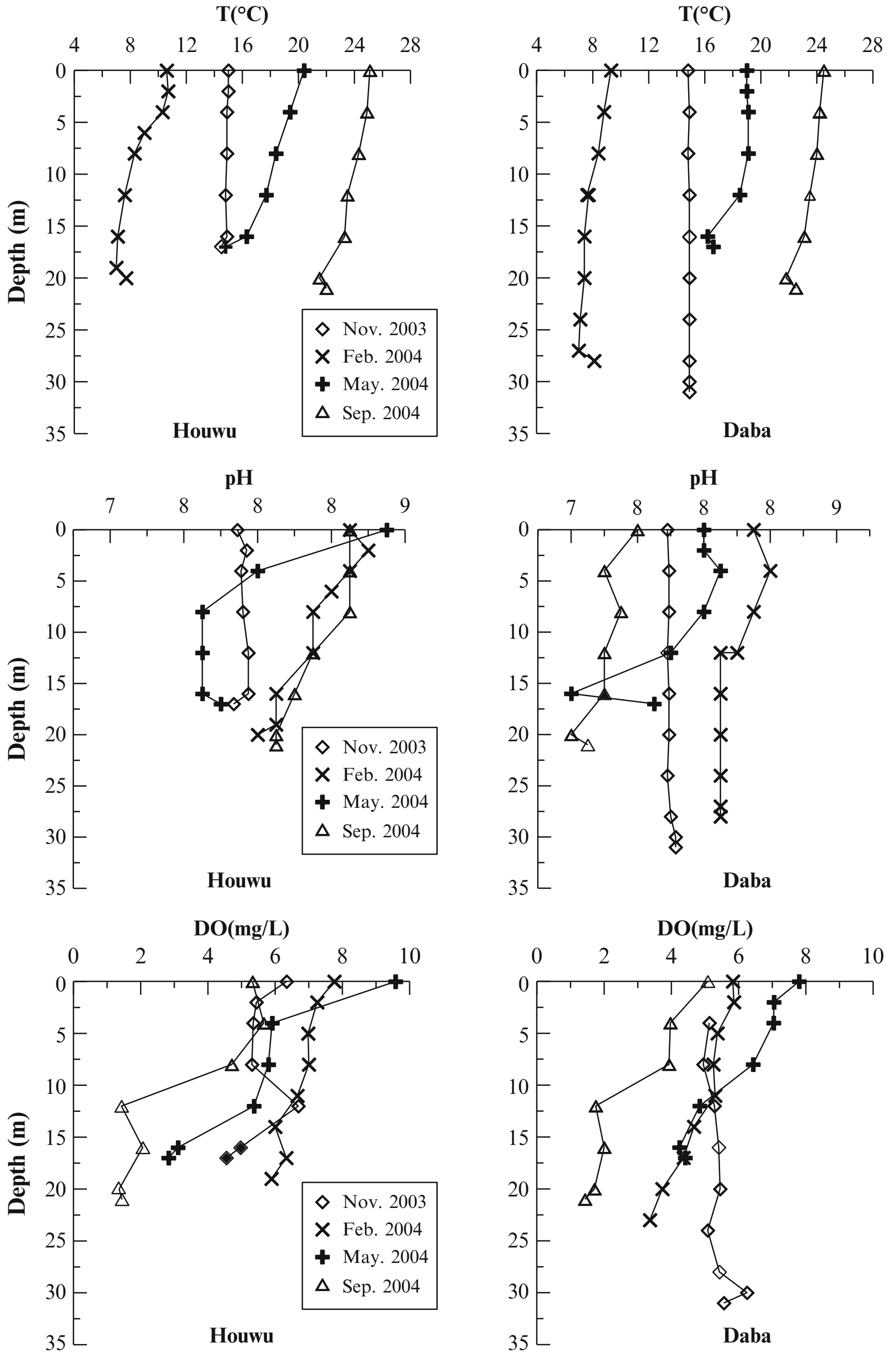

Fig. 11.2 Seasonal distributions of T, pH, and DO in Hongfeng reservoir

Table 11.1 Dissolved oxygen (DO) and pH in the inflow and outflow rivers of Hongfeng reservoir. Data from the February 2004 campaign are missing

Rivers	DO (mg/L)			pH		
	November 03	May 04	September 04	November 03	May 04	September 04
Maotiao	4.7	3.2	1.8	6.9	7.4	7.4
Maxian	8	8.4	5.4	8.1	7.9	8
Yangchang	8.5	8.9	6.7	8	7.6	7.9
Taohua	8.3	–	7.2	8.4	–	8.2
Houliu	7.4	8.3	5.3	7.8	7.6	7.6
Maibao	7.7	8.5	8	7.7	7.7	7.7
Fertilizer plant drain	7.6	5.5	5.3	7.6	8	7.5

Table 11.2 Seasonal variations of average total dissolved phosphorus (TDP) and average chlorophylla concentrations over the whole water column at Houwu and Daba in 2004

Parameters	Sampling sites	February	April	June	August
Chlorophylla (μg/L)	Houwu	16.97	15.4	11.09	14.92
	Daba	10.92	7.42	8.74	13.66
TDP (mg/L)	Houwu	0.015	0.02	0.094	0.086
	Daba	0.014	0.01	0.025	0.038

(Fig. 11.3). Due to the algal bloom, however, elevated average concentrations of TSP (up to 15.67 mg/L) were observed at Houwu in May 2004. DOC concentrations ranged from 1.74 to 3.23 mg/L, which were not as high as in bog lakes in North America (Hines et al. 2004).

The distribution of physical and chemical characteristics in the water columns at Houwu and Daba showed spatial variation as there are many different internal and external contamination sources in Hongfeng reservoir. During the warm season, the average concentrations of TSP, TDP, chlorophyll, and DOC at Houwu were higher than those at Daba, while DO and pH in the hypolimnion at Houwu were lower than at Daba. These variations showed that eutrophication was worse at Houwu than near the dam.

11.3.2 THg, DHg, and PHg in the Water

THg concentrations varied from 2.49 to 13.9 ng/L with an average of 6.89 ng/L (Fig. 11.4). DHg concentrations ranged from 1.19 to 7.96 ng/L with average 3.98 ng/L. Mercury concentrations in water were distinctly higher than those reported from Europe and North America (Bloom et al. 2004; Sullivan and Mason 1998). THg and DHg showed no discernable vertical trend. However, there were spatial and seasonal variations of THg and DHg in the water column. This spatial variation suggests that the two basins of the reservoir were impacted by different mercury

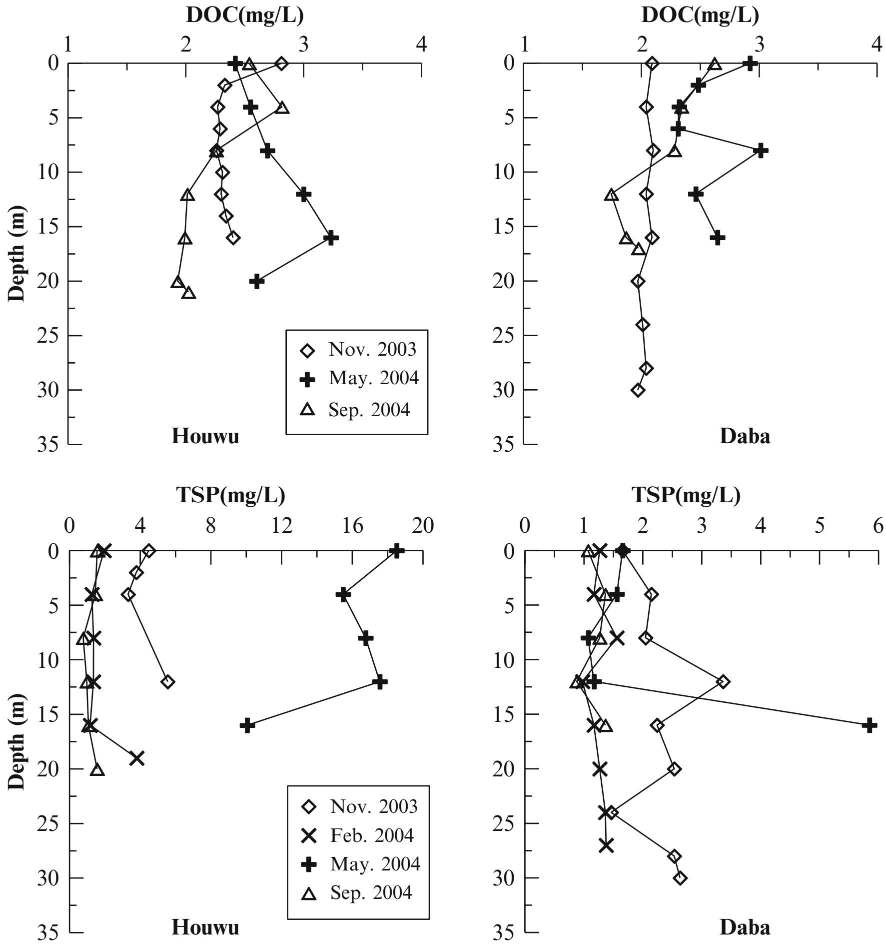

Fig. 11.3 Seasonal distributions of DOC and TSP in Hongfeng reservoir

contamination sources. The highest average concentrations of THg and PHg were observed at Houwu in May 2004 (up to 11.43 and 5.95 ng/L, respectively), while lower average concentrations of THg and PHg (6.06, 3.84 ng/L) were found at Daba at the same time. Similarly, elevated average concentration of TSP also occurred in May at Houwu; TSP averaged 15.67 mg/L compared to 2.1 mg/L at Daba at the same time. This difference was attributed to the massive appearance of algae at Houwu in May, triggered by aquaculture activities and the input of waste-water enriched with N and P. The distribution of mercury also showed that microalgae have a capacity to bind mercury, and may represent a substantial pool of mercury in the aquatic system. The levels of total and dissolved mercury were relatively low in September compared with those in November and February. This seasonal distribution could be a result of waste water contamination. In May and September,

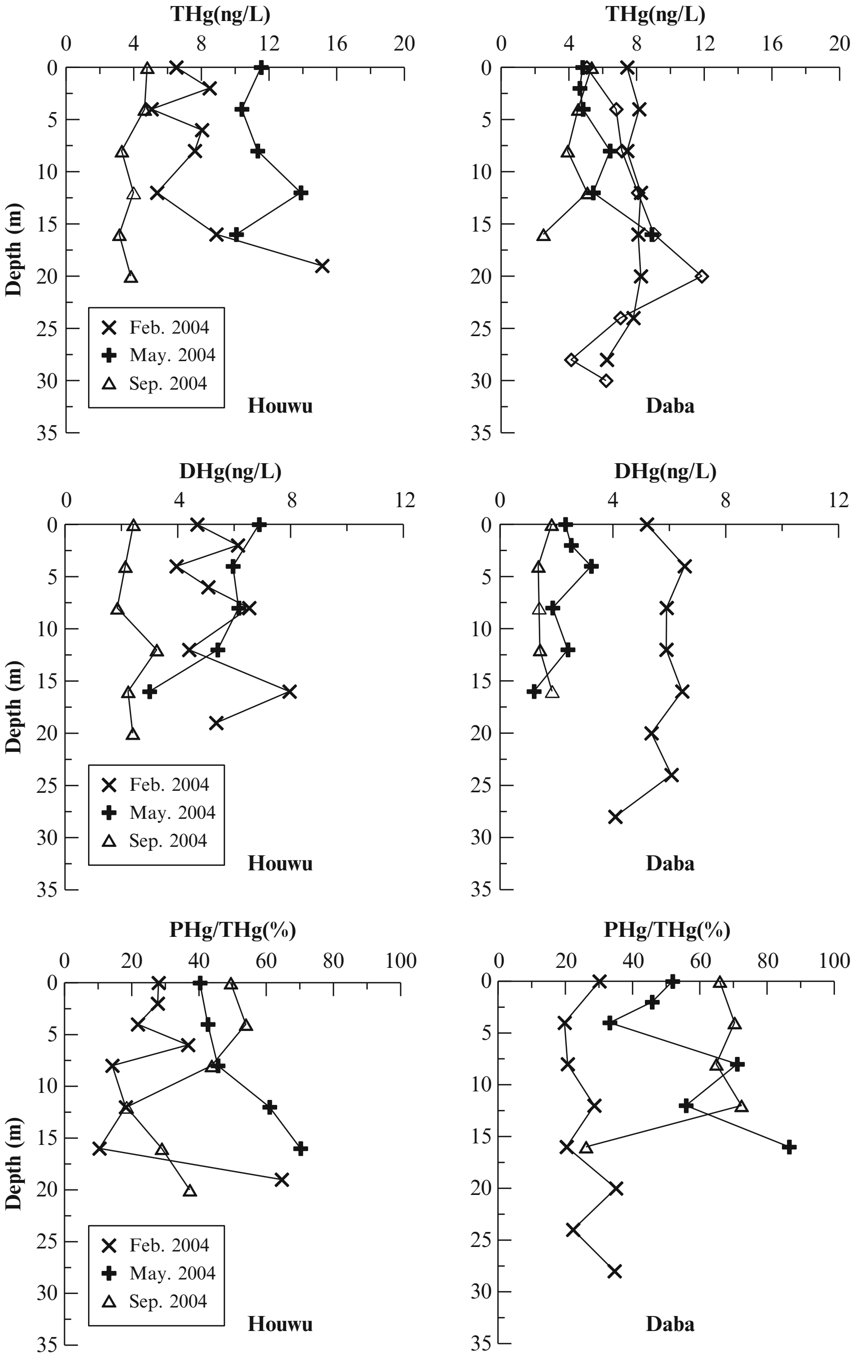

Fig. 11.4 Seasonal distribution of THg and DHg in Hongfeng reservoir

high runoff volume due to abundant precipitation diluted the mercury, whereas there was very little precipitation in February and November so that concentrations in water were much higher.

PHg/THg ratios were high in May and September at 57% and 49%, respectively, while the PHg/THg ratio was only 26% in February. However, increased TSP was not observed in May and September except for high TSP at Houwu in May due to the algal bloom. Therefore, the proportion of PHg probably increased because of the increased fraction of organic particles, which can absorb more mercury than inorganic particles, in May and September. In September, however, DHg in the hypolimnion increased again, especially at Houwu, and the highest proportion was up to 82%. This is probably because the anaerobic decomposition of particulate organic matter in the hypolimnion resulted in an increase in dissolved organic matter in anoxic water which can absorb more mercury than chloride or hydroxide complexes (Coquery et al. 1997). Moreover, the dissolution of iron and manganese oxide in anoxic water might also contribute to the increase in dissolved mercury (Regnell et al. 2001).

11.3.3 DGM and RHg in the Water Column

For comparison between different vertical profiles, the samples of each vertical profile were always collected at midday as DGM concentrations in water, especially in surface water, are significantly affected by sunlight. DGM concentrations ranged from 0.02 to 0.11 ng/L during the four sampling campaigns (Fig. 11.5). DGM concentrations were highest at the water surface and decreased with depth. The average concentrations of DGM in September (0.08 ng/L at Daba; 0.07 ng/L at Houwu) were higher than those in February (0.04 ng/L at Daba; 0.05 ng/L at Houwu). This DGM distribution pattern is consistent with the hypothesis that photoreduction of Hg (II) complexes is the main source of Hg^0 formation in water, and temperature plays an important role in this photoreduction process. There was a sharp decrease in DGM concentrations at 8–12-m depth in September 2004. This suggests that the seasonal stratification may also affect the vertical DGM distribution. The lowest average DGM concentration (0.04 ng/L) was observed at Houwu in May instead of February, whereas the DGM concentration at Daba in May (0.08 ng/L) was as high as that in September. This suggests that several processes may control DGM production. The lowest average DGM concentration observed at Houwu in May can probably be attributed to the algae, which can block sunlight. Early studies (Barkay et al. 1991) suggested that Hg^{2+} reduction is performed by bacteria and eukaryotic microorganisms such as algae. Ben-Bassat and Mayer (1987) found that the formation of Hg^0 decreased as a function of the inhibition of photosynthesis in cultures of the green alga *Chlorella*.

RHg ranged from 0.14 to 2.17 ng/L with an average of 0.64 ng/L (Fig. 11.5). RHg concentrations in surface water were lower than deeper down in all vertical profiles except at Daba in September. This was probably due to an intense

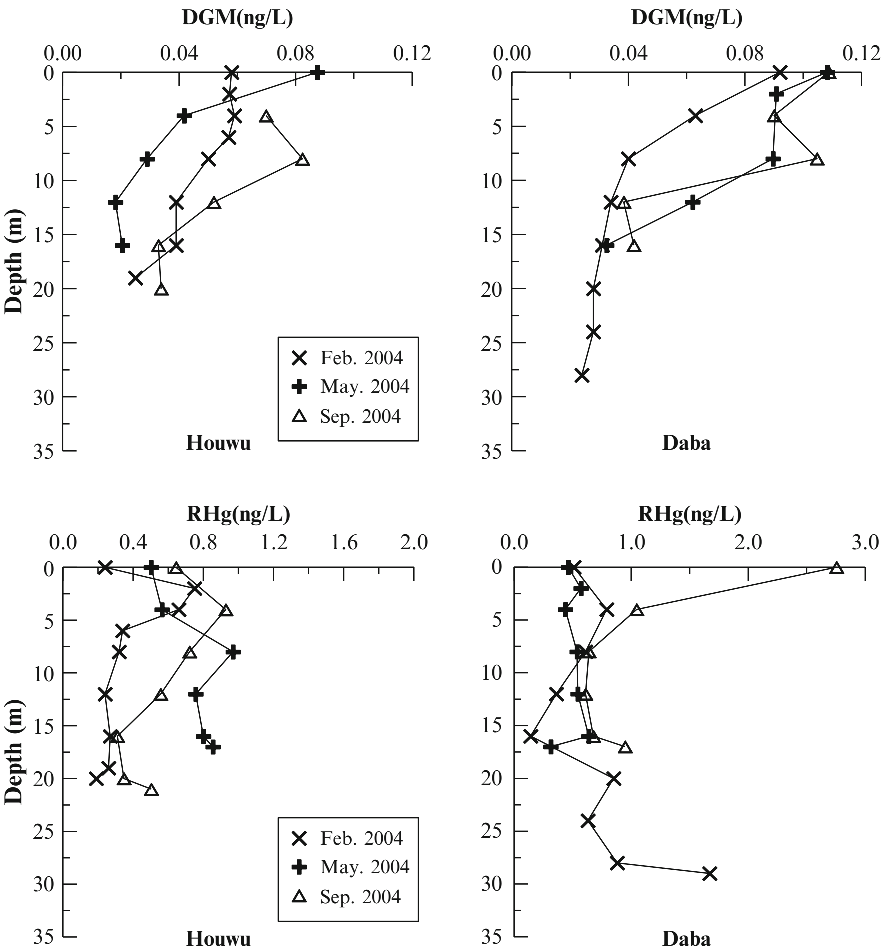

Fig. 11.5 Seasonal distribution of DGM and RHg in Hongfeng reservoir

particulate scavenging and/or biological reduction of Hg^{2+} near the water surface and subsequent release of Hg^0 to the atmosphere (Gill 1986; Kim and Fitzgerald 1986; Dalziel 1995). The unusual high RHg at Daba in September may be related to contamination from the nearby chemical fertilizer plant. This possibility is supported by the very high RHg concentrations (up to 81.49 ng/L) observed in the Fertilizer Plant Drain, which was contaminated by the chemical fertilizer plant.

RHg decreased from subsurface to deep water in most of the vertical profiles, a pattern that may be related to methylation of Hg^{2+} or the presence of S^{2-} in deep anoxic water. As RHg is the main species that could either be reduced to Hg^0 or methylated in a bacterially mediated process (Ullrich et al. 2001), reduction and methylation likely controlled RHg concentrations. However, the above-mentioned distribution patterns of RHg in the water column were not observed at Houwu in

May and at Daba in February, probably as a result of contamination from the chemical fertilizer plant at Daba and the excess algae at Houwu as discussed above.

11.3.4 Inorganic Mercury Species in the Rivers

Relatively high concentrations of mercury were observed in the Yangchang and Maibao Rivers and in the fertilizer plant drain, likely from both industrial and domestic waste water sources of mercury in the drainage area of Hongfeng reservoir (Table 11.3). PHg, whose proportion to THg was 58%, 64%, and 84% in the Yangchang, Maibao Rivers and the fertilizer plant drain, respectively, contributed most THg to these rivers. THg and DHg concentrations in the outflow were much lower than those in the contaminated inflows. Water flows were not measured in this study, but we estimated rough annual inputs and outputs of mercury species based on average concentrations of mercury species in the four campaigns and long-term average annual flows of the rivers reported by Zhang (1999) (Table 11.4). These estimates suggest that more than 50% of the THg from the inflows was removed by the reservoir, and most of this THg was predicted to have been buried in sediments.

11.3.5 TMeHg and DMeHg in the Water Column and Rivers

An analysis of variance showed that there were no statistically significant differences ($p = 0.059$) between the TMeHg distributions in the water column at both sampling sites in the May, November, and February campaigns (Fig. 11.6). TMeHg concentrations in these campaigns varied from 0.05 to 0.33 ng/L and increased slightly with depth in the water in the February and November campaigns.

MeHg concentrations in the September campaign were statistically elevated compared to the other three sampling campaigns ($p < 0.001$). The highest value of 0.92 ng/L occurred at Houwu and was 2.5 times higher than the highest value in other seasons. There was a distinct vertical distribution pattern of MeHg in the water column. TMeHg increased from 0.15 ng/L at the surface to 0.92 ng/L in the hypolimnion at Houwu, while TMeHg increased from 0.08 at the surface to 0.81 ng/L in the bottom water at Daba. There was a sharp increase in TMeHg at the depth of 8–12 m, which corresponded to a sharp decrease in dissolved oxygen at the same depth. Spatial and temporal distributions of MeHg showed that MeHg increased significantly in the hypolimnion in September, especially at the Houwu site.

MeHg content in water is influenced by a wide variety of environmental factors, such as total and reactive mercury content, temperature, redox potential, pH, and the inorganic and organic solutes in waters (Ullrich et al. 2001). However, these factors cannot be viewed independently of each other, as they often interact,

Table 11.3 Seasonal variations of Hg concentrations in the rivers in November 2003, February, May, and September 2004

Rivers	THg (ng/L)				DHg (ng/L)				RHg (ng/L)			
	November	February	May	September	November	February	May	September	November	February	May	September
Maotiao (outlet)	9.92		3.64	5.19	3.41		1.61	3.7	1.92		0.6	0.93
Maxian	6.7	4.04	6.69	2.14	3.9	2.28	5.03	1.58	0.19	0.2	0.11	0.53
Yangchang	37.5	25.8	15.2	15.1	19.1	16.2	4.54	2.7	8.51	1.27	0.61	0.54
Taohua	4.51	2.96		2.21	3.99	2.67		1.34	0.32	0.24		0.21
Houliu	6.87	11.35	15.9	5.45	4.33	5.47	6.07	1.94	0.35	0.41	0.14	1.01
Maibao	121	9.02	27.5	46.7	25.3	6.91	11.2	4.67	0.33	0.24	0.64	1.55
Fertilizer plant drain	175	178	352	233	32.4	40	33.7	26.1	62.8	56.3	54.3	81.9

Table 11.4 Average concentrations of THg, DHg, and RHg in the inflows and outflow of Hongfeng reservoir during four campaigns and their estimated contribution to annual Hg input and output

Rivers	Annual discharge	THg	DHg	RHg	Annual input or output of THg	Annual input or output of DHg	Annual input or output of RHg
	(10^6 m^3)	(ng/L)	(ng/L)	(ng/L)	(g)	(g)	(g)
Maotiao (outflow)	819	6.3	2.9	1.2	5,599	2,862	941
Maxian (inflow)	167	4.9	3.2	0.26	819	535	43
Yangchang (inflow)	400	23.4	10.7	2.7	9,521	4,432	1,092
Taohuayuan (inflow)	131	3.2	2.7	0.26	421	348	34
Houliu (inflow)	58.7	9.9	4.5	0.48	607	287	28
Maibao (inflow)	31.5	51.2	12	0.69	1,629	379	22
Fertilizer plant (inflow)	31.5	234.4	33.1	71.7	7,404	1,056	2,262
Sum of the inflows	819	–	–	–	20,402	7,038	3,481
Difference: input minus output					14,803	4,177	2,540

forming a complex system of synergistic and antagonistic effects. It is generally believed that Hg methylation is microbially mediated, and some studies have shown that methylation is carried out by sulfate-reducing bacteria (Watras et al. 2005; Eckley et al. 2005). Methylation rates appear to be enhanced under anaerobic conditions because of increased activity of anaerobic sulfate-reducing bacteria. In our investigation, TMeHg had a strong negative relationship with DO, with a Pearsson correlation coefficient of -0.81 ($n = 78$, $p < 0.0001$). As pH, DOC and salinity in all samples varied only within a narrow range, no significant correlations between methylmercury and these parameters were observed.

Many studies have shown that increased MeHg in the hypolimnion is related to increased methylation rates and/or the accumulation of settling particulate matter, instead of the release of MeHg from sediments (Regnell and Ewald 1997; Verta and Matilainen 1995). Gilmour and Henry (1991) suggested that both low pH and negative redox potential, which are common in anoxic hypolimnia, not only increase methylation rates but also decrease demethylation rates resulting in a net increase in MeHg. Eckley et al. (2005) showed that methylation rates in hypolimnetic waters were sufficient to account for the observed accumulation of MeHg in hypolimnetic water during summer in two pristine Wisconsin lakes. Some studies also showed that the accumulation of settling particulate matter from the epilimnion, such as hydrous ferric and manganese oxides which can bind MeHg, and their dissolution in the hypolimnion contributed to the high concentration of MeHg in deep water (Meili 1997). Other studies indicated that the increased MeHg was mostly derived from the release of MeHg from sediments, especially at highly contaminated sites (Furutani and Rudd 1980). Moreover, many studies indicated that MeHg release from sediments increases with decreased pH and DO (e.g., Ullrich et al. 2001). The highest values of total and dissolved MeHg at Houwu in September were not at the sediment–water interface. This suggests that MeHg

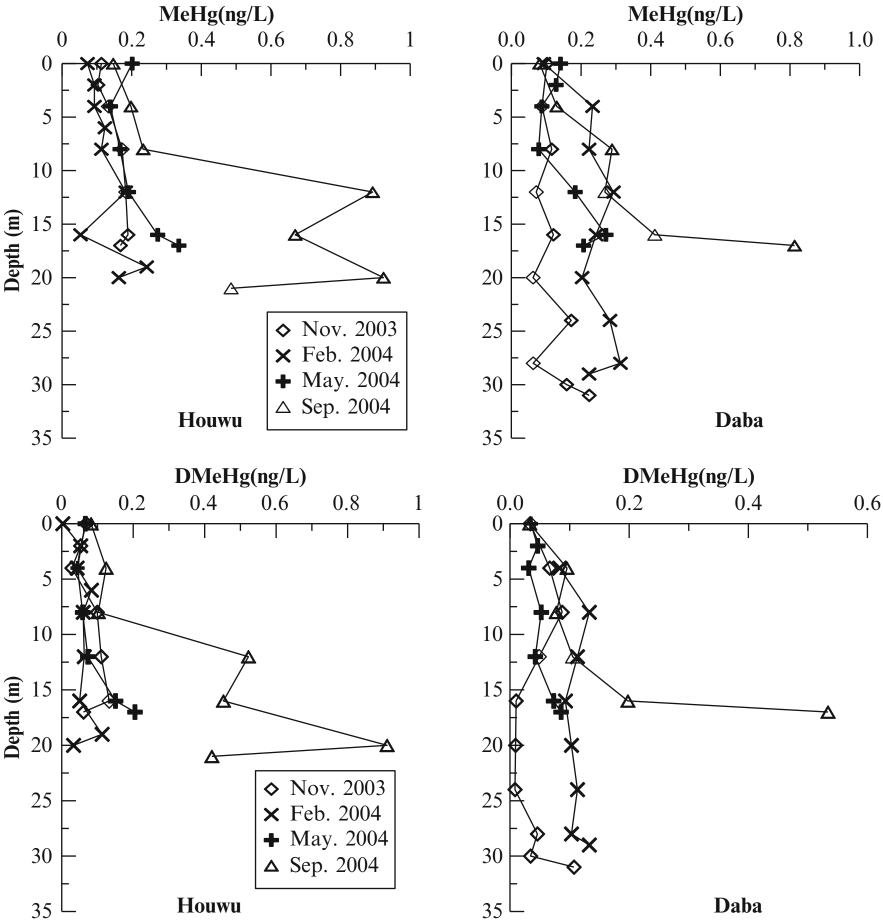

Fig. 11.6 Seasonal distributions of TMeHg and DMeHg in Hongfeng reservoir

in the water column did not come from the release of MeHg in sediment, but from in situ methylation in anoxic water. At Daba in September, however, MeHg showed a strong increasing gradient toward the sediment, which indicates that MeHg released from sediment had a strong impact on the MeHg depth profile at Daba.

Despite the MeHg increase in deep water at Daba in September, the Wilcoxon rank sum test showed that MeHg concentrations in Houwu were much higher than those in Daba at the same depths in September ($p < 0.05$), especially in the hypolimnion (Fig. 11.6). This finding suggests that MeHg was being formed in the hypolimnion at Houwu in September. The decomposition of a large volume of algae induced by high nutrient concentrations at Houwu led to low DO and pH, which may have accelerated Hg methylation.

No statistical difference in the proportions of DMeHg to TMeHg in water was observed in the November, February, and May campaigns; the average value was about 43%. In September, however, the proportion of DMeHg increased significantly, to an average of 65%, and a maximum of 98% at 20-m depth at Houwu. The proportion of DMeHg was also elevated (73%) in the outflow of the reservoir, which is hypolimnion water. Eckley et al. (2005) and Baeyens and Meuleman (1998) also observed an elevated proportion of DMeHg in the hypolimnion of seasonally stratified lakes. Hydrous ferric and manganese oxides and organic particles have large capacities for binding both MeHg and Hg (II). On one hand, under anoxic conditions, mercury methylation rates increased significantly resulting in an increase in MeHg concentrations. On the other hand, the dissolution of oxides and anaerobic decomposition of particulate organic matter possibly resulted in an increase of the DMeHg proportion in the hypolimnion (Regnell et al. 2001).

TMeHg concentrations in water at Houwu basin were statistically higher than those at Daba basin in all campaigns except for February (Wilcoxon rank sum test, $p < 0.05$). The average TMeHg concentrations at Houwu in May, September, and November were 0.22, 0.50, and 0.15 ng/L, respectively, while the corresponding values at Daba were 0.16, 0.34, and 0.12 ng/L, respectively. In February, however, MeHg at Daba (0.23 ng/L) was higher than at Houwu (0.13 ng/L). The different contamination sources could be responsible for this seasonal and spatial variation. At Houwu, the main contamination source was aquaculture activities. At Daba, the main contamination sources were the chemical fertilizer plant and domestic waste water inputs. The fish farms at Houwu basin contributed a large amount of N and P to the water, resulting in more algae than at Daba. The decomposition of algae caused DO and pH to decrease, which favored the methylation of mercury. In winter, the contamination at Houwu lessened significantly with the reduction of activity at the fish farm, but the contamination at Daba remained high because the pollutants were derived from the chemical fertilizer plant and domestic waste water. Reactive Hg at Daba (0.72 ng/L) was also higher than that at Houwu (0.36 ng/L), while DO at Daba (4.9 mg/L) was lower than that at Houwu (6.7 mg/L) in February. All of these factors could result in a higher mercury methylation rate at Daba in winter.

MeHg concentrations were relatively higher in the Maxian, Houliu, and Yangchang Rivers in May than in other seasons (Table 11.5). These rivers are the main water sources to the reservoir and are affected by agriculture activities. No significant differences were observed between MeHg concentrations in inflows and

Table 11.5 Seasonal distribution of methylmercury in the inflows and outflow of Hongfeng reservoir (ng/L)

Rivers	Outflow	Inflows					
	Maotiao	Maxian	Yangchang	Taohua	Houliu	Maibao	Fertilizer plant
November	0.138	0.18	0.155	0.096	0.111	–	0.212
February	–	0.102	0.143	0.091	0.156	0.134	0.235
May	0.164	0.186	0.207	–	0.244	0.103	0.062
September	1.022	0.121	0.178	0.077	0.159	0.14	0.11

the outflow in all campaigns except for the September campaign. In September, however, the outflow of the reservoir was rich in MeHg, and the concentration was 5.5 times higher than the average value in the inflows. This MeHg-enriched water from the anoxic hypolimnion represented the discharge from the reservoir. Though the stability of this MeHg in the more oxidizing environment of the outflow river is uncertain, it poses a potential risk to the ecosystem below the reservoir.

11.3.6 THg in Sediment and Pore Water

Total mercury levels in whole sediment was 0.392 ± 0.070 μg/g, higher than levels reported in other uncontaminated reservoirs and at Wujiangdu Reservoir, which is located in same drainage area as Hongfeng reservoir. This indicated there were mercury contaminations in Hongfeng reservoir. THg in sediment did not show significant variations between seasons, but generally increased toward the sediment–water interface. Enrichments of Hg in surface sediments have usually been reported in the past, even in remote lakes (Rasmussen et al. 1998; Lockhart et al. 2000). There are two types of explanations about Hg distribution in sediment cores: some studies state that the enrichment of mercury in sediment can be attributed to modern contamination, and distribution of mercury in sediment core reflects the history of atmospheric Hg trends and fluxes (e.g., Lockhart et al. 1998; Engstrom and Swain 1997). Other studies, however, consider the role of chemical speciation of mercury and their affinity to inorganic fractions when interpreting vertical Hg concentration profiles in lake sediments (Bilali et al. 2002; Rasmussen et al. 2000). It is believed that Hg profiles can be produced by Hg redistribution during diagenesis. THg was correlated well with organic matter in all sediment samples ($r = 0.59$, $p < 0.001$).

The average value of THg concentration in pore water is 23.2 ± 8.4 ng/L in summer and 13.6 ± 3.9 ng/L in winter. The average value of partition coefficients (log Kd) for THg is 3.1×10^4 L/kg in winter and 1.75×10^4 L/kg in summer. These results showed THg concentrations in the pore water and partition coefficients for THg in solid phase and water phase were mainly controlled by temperature. THg concentrations in pore water had no discernable vertical distribution trends throughout the whole sediment column in all sampling campaigns, and no relationship with total mercury concentrations in the solid phase as well.

11.3.7 MeHg in Sediment and Pore Water

MeHg concentrations in the sediments were from 0.24 to 8.4 ng/g. The seasonal variation and maximum peak value distributions of methylmercury in sediment were mainly controlled by seasonally migration of the oxic/anoxic boundary layer. Methylmercury concentrations are highest in spring, with an average concentration of 3.4 ± 2.5 ng/g. There were no significant variations in other seasons with an average concentration of 2.4 ± 1.5 ng/g. The peak values of methylmercury typically appeared in the upper 8 cm of the sediment profiles which were also the zones of sulfate-reducing bacterial activity.

MeHg concentrations in the pore water changed from 0.16 to 4.2 ng/L, and had a strong relationship with those in the solid phase ($r = 0.70$, $p < 0.001$). The methylmercury concentrations in solid phase and pore water were controlled by the solid/water partition coefficient, as well as by methylmercury production.

11.3.8 THg and MeHg in Fish

Concentrations of total and methylmercury in fish of Hongfeng reservoir were measured based on cold vapor atomic fluorescence detection in investigating distributions of mercury and methylmercury in different fish stocks and ager lever as well as controlling factors. The results showed that total mercury ranged from 3.2 to 150 ng/g with an average value of 32 ng/g. Methylmercury was from 0.15 to 53 ng/g with an average value of 12 ng/g. These values were lower than those reported in other articles, and are under the national standard. The low level of mercury and methylercury of fish may be due to the accelerated growth and the shortened food chain. The mercury distribution trends in fish species showed mercury in predatory fish > mercury in Polyphagous fish > mercury in herbivorous fish, which indicated mercury in fish was mainly related to fish feeding habits. Total mercury concentrations had a strong relationship ($r = 0.59$, $p < 0.001$) with weight of fish in predominant fish stocks.

11.3.9 Mass Balance of Mercury

Results from the mass balance of total mercury showed that total input and total output of mercury were 30,066 and 31,010 g/a, respectively. The dominating mercury source was from mercury input to the inflows, reaching up to 82% of total mercury input. The dominating output of mercury was through settling of the particulate mercury in water, up to 78% of total mercury output. The methylmercury output from outflow was an important fate of methylmercury in the reservoir, up to 45% of total output and was 30% higher than the rate of methylmercury flux from the inflows to total MeHg input. The mercury cycling in sediment–water surface indicated sediment was affecting the outcome of both total mercury and methylmercury, but the fluxes of mercury diffusion from pore water weakly contributed to mercury in lake water. The mercury cycling in the water–air surface indicated mercury precipitation from the atmosphere was an important mercury source to lake water, and net input flux of mercury from the atmosphere was 3,364 g/a in Hongfeng reservoir.

11.4 Conclusions

Hyper-eutrophication in Hongfeng reservoir, which resulted in algal blooms and deterioration of water quality, affected the concentrations and distribution of mercury species in the reservoir. Microalgae had a large capacity to bind mercury, and represented a substantial pool of mercury in the aquatic system. The formation of Hg^0 decreased, probably also because these microalgae blocked out sunlight and inhibited photosynthesis. Most importantly, hyper-eutrophication resulted in lower DO and higher DOC, thus accelerating the formation of MeHg in the hypolimnion especially in summer.

Hongfeng reservoir was a large net sink of total mercury, but a net source of MeHg. The MeHg-enriched water discharged from the anoxic hypolimnion may pose serious risks downstream of the reservoir. The dominating mercury source was from mercury input in the inflows. The dominating output of mercury was through settling of the particulate mercury in water. The mercury cycling at the sediment–water surface indicated sediment was dominating the fate of both total mercury and methylmercury, but the fluxes of mercury diffusion from pore water contributed only weakly to mercury in lake water.

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Chapter 12
The Nutrients-Phytoplankton Relationship Under Artificial Reservoir Operation: A Case Study in Tributaries of the Three Gorges Reservoir, China

Zhe Li, Jinsong Guo, Fang Fang, Xu Gao, Man Long, and Zhiping Liu

Abstract The ratio between total nitrogen (TN) and total phosphorus (TP) is an important limnological measure, potentially regulating the phytoplankton dynamics in lakes. However, information on the impact of the TN/TP ratio on phytoplankton biomass in artificial reservoirs with unstable physical boundary has been little studied. Here, we performed a 2-year biweekly monitoring program in the Pengxi River Backwater Area (PBA), a tributary of the Three Gorges Reservoir (the Yangtze River), to document the relationship of TN/TP ratio and phytoplankton. Based on Spearman Correlation Analysis, we found that significant seasonal variation of TN, TP, and TN/TP ratio was unrelated to variation in phytoplankton biomass. Three subsets of the 2-year data were divided according to reservoir operation mode and seasonal growth of phytoplankton to gain a deeper insight in their relationship. In the non-growth season, when water residence time in the PBA is longer due to impoundment in the Three Gorge Reservoir (TGR) and to decline of river discharge in the dry season, release of TN and TP from the newly submerged water fluctuation zone increased the input of nutrients and the ratio of TN/TP. This process co-occurred with a decline in the growth of phytoplankton, resulting in a positive correlation between TN/TP ratio and phytoplankton biomass. In the growth season, low water residence time (HRT < 50 day), intensive water exchange, and mass transport from river discharge at low water level caused unstable hydrodynamic conditions for the growth of phytoplankton. Light availability might be the controlling factor that regulated the biomass of phytoplankton. In the growth season with long water residence times (HRT ≥ 50 day), a relatively stable physical environment supported the occurrence of N-fixation in the PBA.

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12.1 Introduction

The ratio between total nitrogen (TN) and total phosphorus (TP) is an important limnological tool that controls primary production and phytoplankton species composition in lakes. The N/P ratio theory started from the well-known Redfield ratio, establishing that the molar ratio of C:N:P of phytoplankton is relatively constant at 100:16:1 (Redfield 1958). According to Redfield's theory, the composition of C, N, P of phytoplankton is in balance with the ratio in the water (Redfield 1958). If the external N/P ratio is higher than that in the cells, the growth of phytoplankton will be limited by phosphorus; if the reverse is true, it will be limited by nitrogen. Therefore, the ratio directly couples with the physiology of phytoplankton species, providing insights into species' elemental composition, growth rates, and succession pattern, driven by their habitat (Healey 1973, 1979; Goldman 1980).

Empirical studies by Schindler (1977), Smith (1982, 1983) and other workers (Hecky and Kilhman 1988) extended the Redfield ratio to a general pattern. It was found that a decrease in TN/TP ratio related to primary production and trophic level in lakes (Smith 1982). This empirical relationship was, furthermore, applying better to shallow than to deep lakes (Quirós 2002). Recent cross-system studies by Guildford and Hecky (2000) argued that when TN/TP ratio is $\leq$9.0 (hereinafter used as mass ratio), primary production tends to be limited by nitrogen, while at a TN/TP ratio of $\geq$22.6, primary production is rather P-limited. It was also found that variation in TN/TP ratio potentially regulated the composition of phytoplankton species. Generally, when the TN/TP ratio is $\leq$29 (mass ratio), a dominance of Cyanobacteria prevails. When TN/TP ratio is $\leq$22, nitrogen fixation by Cyanobacteria may become the dominant biogeochemical process (Smith 1983; Havens et al. 2003; Håkanson et al. 2007). Moreover, Downing and McCauley (1992) found that the TN/TP ratio in water was impacted by input water quality. These results could be meaningful in the control of nitrogen and phosphorus input to water bodies. However, it should be noted that current knowledge on the ecological effect of TN/TP ratios is based on lake systems that have a relatively stable physical boundary (=water level) (Green and Finlay 2010). The response of aquatic ecosystems to variations in TN/TP ratio in a complex hydrodynamical setting is still unclear (Green and Finlay 2010). Uncertainties remain in two aspects: (1) assessment of the co-variance of TN/TP ratio and primary production in a given physical background, e.g., rivers with different discharge, reservoirs in seasonal operation stages and (2) evaluation of nutrient limitation to primary production and selective ability of phytoplankton assemblages in specific physical environments.

Three Gorges Reservoir (TGR), China, is one of the world's largest reservoirs, with a total capacity of 39.3 billion cubic meters and a water area up to 1,080 km^2. According to its operation strategy, TGR starts its impoundment in autumn and reaches a 175 m high-water level during winter. It discharges water for hydropower generation from the end of winter to the end of spring. Water level

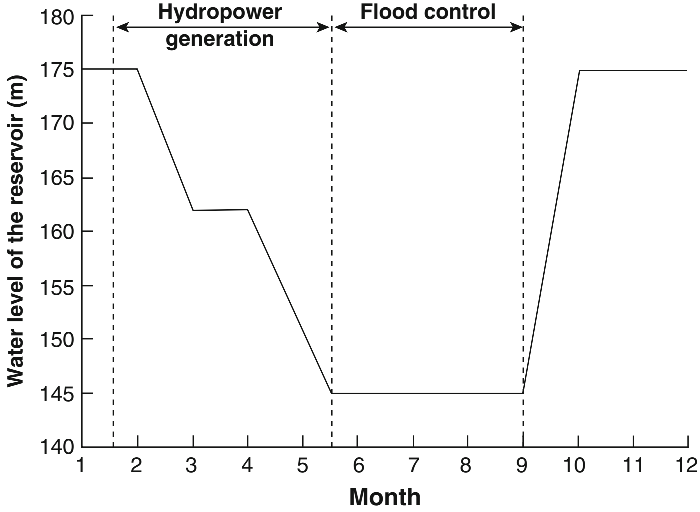

Fig. 12.1 Operation mode of the Three Gorges Reservoir

in the TGR is kept at 145 m, leaving 22 billion cubic meters for flood control (Fig. 12.1).

After initial impoundment of TGR in 2003 to a water level of 139 m, there was a significant increase in primary production in tributary backwaters of the Yangtze River (Li et al. 2009a; Zeng et al. 2006). Frequent algal blooms have now become a serious eco-environmental problem in this area. However, current research indicates that nitrogen and phosphorus in most tributary backwaters are not limiting the growth of phytoplankton (Li et al. 2009b, c). Apparently, seasonal variations in light and heat control the phytoplankton more (Li et al. 2010). However, in this river-valley dammed reservoir, the co-variation of flow discharge and storage capacity control produces a special hydrodynamic condition that determines the limnological characteristics of the impoundment. There was still doubt whether the empirical TN/TP ratio and primary production could be used to guide ecological management in this river-reservoir hybrid ecosystem. Therefore, the potential relationship between primary production and TN/TP ratio is studied here, with an aim to obtain a deeper insight into the cause and effect of nutrients and primary production, and to provide information for reservoir management.

A 2-year monitoring program in the backwater area of Pengxi River (PBA), a large tributary of the Yangtze in the TGR, was set up by the research team of the authors. Based on seasonal hydrological, water chemical, and biological analysis, we here discuss the relationship of TN/TP ratio and phytoplankton biomass (indicated as primary production) in the context of reservoir operation.

12.2 Study Site and Sampling Plan

Pengxi River (also named Xiaojiang River) is one of the largest tributaries of the Yangtze, located midway of the Three Gorges Reservoir Region (TGRR), about 250 km upstream from the Three Gorges Dam (TGD) (Fig. 12.2). It covers 5,172.5 km^2, from N31°00′ E107°56′ to N31°42′ E108°54′. Its main length is about 182 km, with average slope 1.25%. Annual rainfall in the watershed is 1,100–1,500 mm, and annual discharge is 118 $m^3 \cdot s^{-1}$.

After the impoundment of the TGR to a water level of 145 m, Pengxi River formed a backwater area of approximately 60 km, from its conflux with the Yangzte in Shuangjiang Town to Yanglu Town. Five sampling spots along the backwater area were chosen in the main channel (Fig. 12.3a): Quma (QM, N31°07′50.8″ E108°37′13.9″), Gaoyang (GY, N31°5′48.2″ E108°40′20.1″), Huangshi (HS, N31°00′29.4″ E108°42′39.5″), Shuangjiang (SJ, N30°56′51.1″ E108°41′7.5″), and Hekou (HK, Conflux with Yangtze, N30°57′03.8″ E108°39′30.6″). Main channel lengths between sampling spots were approximately 6.72 km QM-GY, 12.1 km GY-HS, 8.7 km HS-SJ, and 3.4 km SJ-HK. Average main channel depth was 17.9 m at a water level of 145 m. The vertical profile of the PBA main channel is shown in Fig. 12.3b.

The interval between samplings was 14–17 days. Sampling time was controlled between 9:30 a.m. and 4:30 p.m., covering all five sampling spots. Water samples were collected in polyethylene bottles from a water depth of 0.5, 1.0, 2.0, 3.0, 5.0, 8.0 m using a 3 L Kitahara's water sampler, and mixed equally for phytoplankton and chemical analysis.

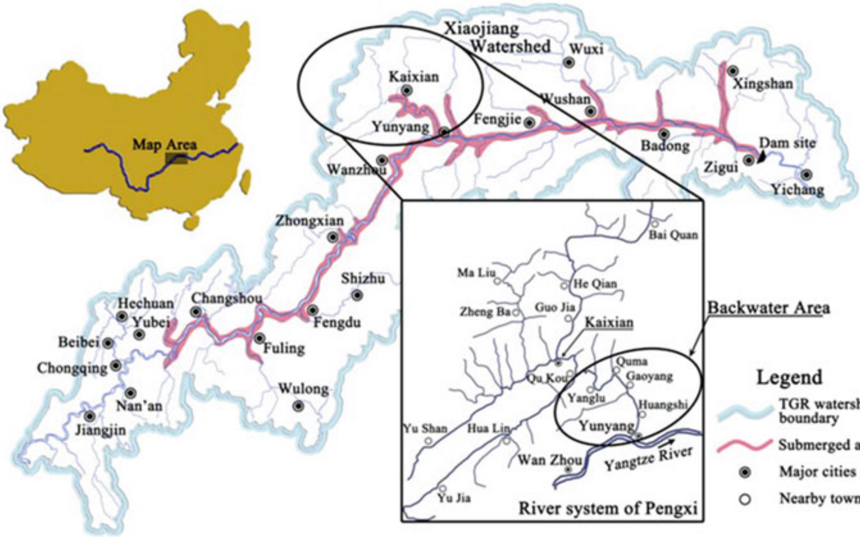

Fig. 12.2 Location of the Pengxi River in the Three Gorges Reservoir and its river system

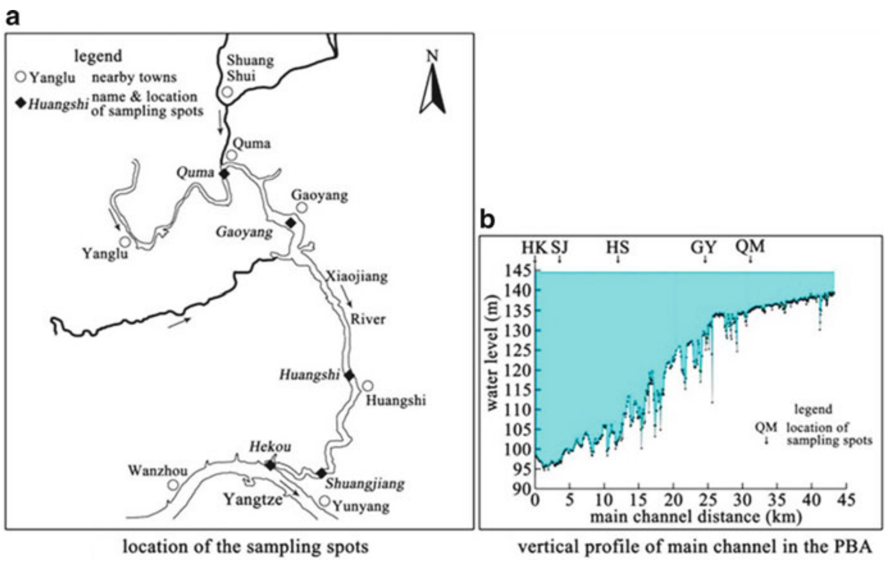

Fig. 12.3 Location of sampling spots at the backwater area of the Pengxi River (**a**) and the vertical profile of PBA (**b**)

12.3 Samples and Data Analysis

12.3.1 Phytoplankton Analysis

Phytoplankton biomass is an indicator of primary production. After mixing, phytoplankton samples were preserved in 1% lugol's solution on-site. After 48-h sedimentation in the laboratory, Utermöhl's method (1958) was applied for enumeration of cell density under a microscope at 400× magnification. At least 100 fields were counted per replicate, and a maximum counting error of 20% between replicates was accepted in calculating the standing-stock of phytoplankton (Venrick 1978). Cell volume was estimated for each species using average cell dimensions and appropriate volume formula. Biomass of phytoplankton (BioM) was regarded as the product of water density (1.0 g cm^{-3}) and cell volume (Rott 1981; Reynolds 2006). Chlorophyll *a* (Chl *a*) in water was another primary production indicator in the PBA. The 500 L mixed water samples were filtered through Whatman® GF/C membrane. Chlorophyll *a* was extracted in 90% acetone solution for 36 h and analyzed by spectrophotometry according to APHA (1995).

12.3.2 Water Chemistry Analysis

Mixed water samples were digested for analysis of total nitrogen (TN) and total phosphorus (TP). They were filtered through pretreated (450°C, 2 h in Muffle

Table 12.1 Equation for calculation of different forms of nutrients

DIN = NH_4^+−N + NO_3^-−N + NO_2−N;	DTN = DIN + DON;
TON = DON + PON	TN = DIN + DON + PON;
PP = TP−DTP	

TON total organic nitrogen; *DON* dissolved organic nitrogen; *PON* particulate organic nitrogen; *PP* particulate phosphorus; *DTP* dissolved total phosphorus

furnace) Whatman® GF/F glass fiber membrane for ammonia (NH_4^+−N), nitrate (NO_3^-−N), nitrite (NO_2^-−N), soluble reactive phosphorus (SRP), dissolved total phosphorus (DTP), and dissolved total nitrogen (DTN). All chemical analyses were carried out using visible or ultraviolet spectrophotometry (APHA 1995). Mass balance for calculating different forms of nitrogen and phosphorus is shown in Table 12.1.

12.3.3 Hydrological/Meteorological Data and Water Residence Time

Water residence time (WRT) directly indicates the co-variance of natural discharge and reservoir operation, an important hydrodynamic parameter in reservoir limnology. Calculation results of WRT in the PBA were applied in this study to represent the physical environment. Daily rainfall and solar radiation data were downloaded from China Meteorological Data Sharing System (http://cdc.cma.gov.cn/). Water level at HK in the PBA was estimated according to water surface gradients between the TGD dam site (http://www.ctgpc.com.cn/) and the Wanxian hydrology monitoring station (http://www.cqwater.gov.cn). The daily discharge of the Pengxi River was calculated from the Distributed Hydrologic Model in Pengxi Watershed, based on Semi-distributed Land Use-based Runoff Processes Model (SLURP; Kite 1995) by Prof. Tianyu Long's research team in Chongqing University (Long et al. 2009a, b; Wu et al. 2010). This Distributed Hydrologic Model was calibrated and validated by 52 years of monthly discharge (Long et al. 2009b; Wu et al. 2010).

HEC-RAS (ver 4.1; Brunner 2008) was applied to set up a 1D hydrodynamic model in the PBA based on a 1:2,000 underwater topographic map, calibrated and validated with historical data and on-site measurements of cross-sectional velocity. Daily water residence time (also known as hydraulic retention time) was then calculated according to the modeling results in this study.

For the sake of matching the daily hydrological/meteorological data and biweekly phytoplankton and water chemistry monitoring data, daily hydrological/meteorological data as well as daily water residence time were averaged between two sampling events, representing the status of the physical environment between two samples. Equation 12.1 was applied to calculate the averaged hydrological/meteorological data:

$$Ave\ J = \frac{1}{d}\sum_{\mathrm{i}=1}^{d} j \tag{12.1}$$

d is the time interval between two sampling events and j is daily data of rainfall, river discharge, and water residence time. *Ave J* is therefore represented as AveRain, AveQ, AveWRT for average rainfall, flow discharge, and water residence time, respectively.

12.3.4 Data Processing

Within 47 sampling events, 235 samples were taken from May 2007 to April 2009 at the five sampling stations. All data were logged into SPSS® or Origin® for statistical analysis. Comparison between daily temperature recorded from 2007 to 2009 and historical data in the watershed showed that spring season began in March to mid-May, while summer lasted from late May to mid-September. Autumn began in late September to mid-November. Winter was from late November to the end of February in the next year.

It was hypothesized that if nutrients controlled the growth of phytoplankton, the levels of nitrogen or phosphorus would show a significant correlation with biomass (BioM) and chlorophyll *a*. Non-parametric Spearman Correlation Analysis was applied to evaluate the statistical relationship between different data groups. Locally weight scatter plot smoothing (LOESS) was applied to describe the distribution or trend of the datasets in scatter plot, and to help interpret bivariate relationships while guarding against the influence of deviant points.

12.4 Results

12.4.1 Variation of Hydrological Data and Water Residence Time

The TGR started impoundment at 135 m in June 2003, after the closing of the diversion channel in 2002. Water level in the TGR was raised to 139 m in November 2003 and decreased to 135 m during the flood season in 2004. This situation did not change until the end of October 2006 when the preliminary operation stage started. According to the operation strategy, water level was at 156 m during winter dry season and at 145 m in the summer flood season. At the end of October 2008, water level in the TGR reached 172.3 m, approaching the designed level. Water level at the PBA during the 2-year study (May 2007 to April 2009) is shown in Fig. 12.4.

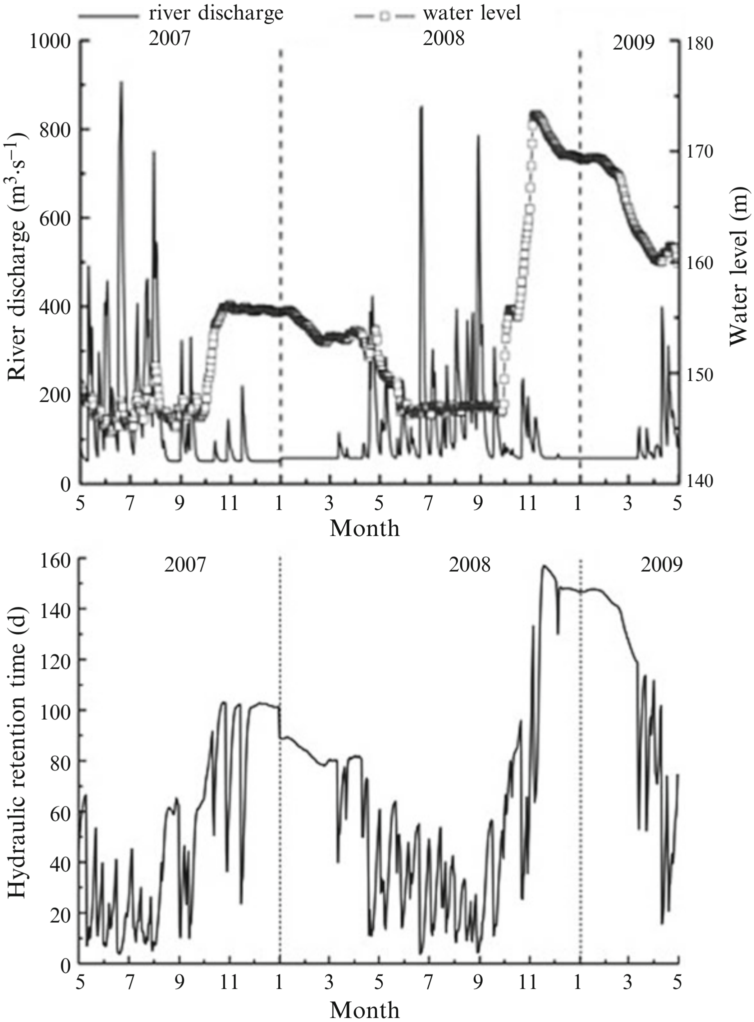

Fig. 12.4 Daily river discharge, water level at HK, and water residence time in the PBA during the 2-year survey

Mean value and seasonal variation of WRT, river discharge, and rainfall among seasons are shown in Table 12.2. Because of co-variance of natural flow discharge of the Pengxi River and reservoir operation, variance of water residence time in the PBA between seasons was significant. It reached a maximum level of 157.0 day in November 2008 with a water level of 172.3 m and a corresponding daily flow discharge of $5.12 \times 10^6\ m^3 \cdot day^{-1}$. In June 2007, the minimum WRT was 3.8 day at a water level of 147.5 m, and daily flow discharge was $78.4 \times 10^6\ m^3 \cdot day^{-1}$.

Table 12.2 Seasonal variation of daily flow discharge, rainfall, and water residence time in the PBA

Samples		2007–2009	Spring	Summer	Autumn	Winter
		731	152	246	122	211
Flow discharge	M.V.	121.2 ± 4.6	113.2 ± 7.4	197.4 ± 11.0	86.5 ± 4.5	58.3 ± 0.8
	Range	50.8–907.1	51.3–491.5	50.9–907.1	50.9–307.9	50.8–168.5
	C_V	1.03	0.81	0.88	0.58	0.19
Rainfall	M.V.	3.70 ± 0.34	3.64 ± 0.67	6.74 ± 0.84	3.02 ± 0.59	0.58 ± 0.11
	Range	0.00–88.01	0.00–54.64	0.00–88.01	0.0–35.14	0.00–10.97
	C_V	2.51	2.25	1.96	2.15	2.84
Water residence time	M.V.	68.8 ± 1.63	65.2 ± 2.5	28.1 ± 1.1	71.2 ± 2.5	117.4 ± 2.1
	Range	3.8–157.00	7.0–127.5	3.8–65.3	10.9–149.2	31.0–157.0
	C_V	0.64	0.48	0.62	0.39	0.26

Unit: flow discharge, $m^3 \cdot s^{-1}$; rainfall, mm; water residence time, day

12.4.2 Standing Stocks and Seasonal Variation in Phytoplankton Biomass

Mean value of BioM was 4589 ± 652 μg L^{-1} and the corresponding Chl *a* was 11.65 ± 1.16 μg L^{-1} (Table 12.3). One-way ANOVA showed no significant difference among sampling spots (Sig. ≤ 0.01). Seasonal variation of BioM and Chl *a* was high (Fig. 12.5). Diatom blooms (dominated by *Asterionella* sp. or *Cyclotella* sp.) triggered a sharp increase in BioM and Chl *a* from late winter to early spring. Although a slight decrease of BioM and Chl *a* occurred in March to mid-April, serious algal blooms (dominant species *Anabaena flos-aquae* in 2007 and 2009 and *Ceratium hirundinella* in 2008) took place from late spring to early summer. BioM during this bloom period reached a maximum of 16,861 μg L^{-1}. During the flood season, BioM decreased to a level of 2,500–4,500 μg L^{-1}, and concentration level of Chl *a* at this time ranged between 10 and 20 μg L^{-1}. Significant variation in BioM and Chl *a* was apparent in PBA during the flood season. Starting from late summer to mid-Autumn, before impoundment of the reservoir, there was an increasing trend in phytoplankton biomass. With the decrease of temperature and increased water level in the TGR, BioM, and Chl *a* decreased sharply from late-Autumn and remained at a relatively low level until late winter. During the 'calm period' in winter, minimum level of BioM ranged 200–700 μg L^{-1}, and the level of Chl *a* during the winter ranged 0.5–5 μg L^{-1}.

Generally, in the succession of phytoplankton under reservoir operation, a dominance of Bacillariophyta extended from late autumn to late winter. Dinophyta, Chlorophyta, and Cyanophyta co-dominated in the PBA and triggered serious algal blooms of Dinophyta or Cyanophyta in late spring or early summer. In the flood season (June to August), Cyanophyta, Bacillariophyta, and Chlorophyta dominated in the assemblages. Starting from late summer until late autumn, a shift from Dinophyta or Chlorophyta dominance to Bacillariophyta dominance occurred.

Table 12.3 Seasonal variations of biotic and abiotic data in the PBA

Samples		2007–2009	Spring	Summer	Autumn	Winter
		235	50	75	50	60
Biomass	M.V.	4589 ± 652	7448 ± 1258	7054 ± 1767	2164 ± 294	1147 ± 153
	Range	76–120,516	222–39,271	544–120,516	365–11,484	76–5,546
	C_V	2.177	1.194	2.169	0.960	1.036
Chla	M.V.	11.65 ± 1.16	17.63 ± 2.35	16.24 ± 2.91	6.50 ± 0.87	5.03 ± 0.96
	Range	0.19–193.70	0.27–65.40	0.96–193.70	0.28–29.08	0.19–30.77
	C_V	1.517	0.943	1.537	0.943	1.473
TP	M.V.	75.9 ± 2.5	93.0 ± 5.4	78.9 ± 6.2	56.4 ± 2.8	74.3 ± 2.4
	Range	18.7–349.6	32.2–190.9	22.0–349.6	18.7–120.0	42.4–118.3
	C_V	0.514	0.408	0.679	0.356	0.247
TN	M.V.	1623 ± 30	1929 ± 90	1598 ± 54	1434 ± 31	1557 ± 38
	Range	659–3,264	1,005–3,264	659–3,153	1,014–2,089	1,079–2,301
	C_V	0.285	0.330	0.293	0.154	0.187
TN/TP ratio	M.V.	24.5 ± 0.6	22.7 ± 1.3	25. 8 ± 1.4	27.4 ± 1.2	22.2 ± 0.9
	Range	4.3–61.7	10.7–45.4	4.3–61.7	13.9–53.1	10.3–41.1
	C_V	0.396	0.404	0.472	0.296	0.309
NH_4^+–N	M.V.	221 ± 13	162 ± 24	312 ± 26	135 ± 18	228 ± 27
	Range	0.1–1036	0.1–703	3–1036	1–560	3–820
	C_V	0.928	1.026	0.733	0.966	0.922
NO_3^-–N	M.V.	747 ± 19	850 ± 49	594 ± 29	779 ± 36	825 ± 27
	Range	18–1,606	244–1,606	18–1,344	289–1382	529–1210
	C_V	0.381	0.408	0.425	0.324	0.250
SRP	M.V.	23.8 ± 1.6	34.1 ± 5.1	9.1 ± 1.3	21.1 ± 3.1	35.9 ± 2.1
	Range	0.5–119.4	2.1–119.4	0.5–49.4	0.9–78.9	8.9–75.5
	C_V	1.036	1.060	1.202	1.041	0.446
PP	M.V.	38.1 ± 2.3	39.0 ± 3.9	56.6 ± 5.8	27.9 ± 2.3	22.7 ± 1.8
	Range	0.0–307.2	1.7–116.5	1.7–307.2	0.0–79.3	1.7–60.2
	C_V	0.926	0.703	0.882	0.583	0.616

Unit of biomass and Chl *a* were μg L^{-1}, M.V. mean value, C_V coefficient of variation

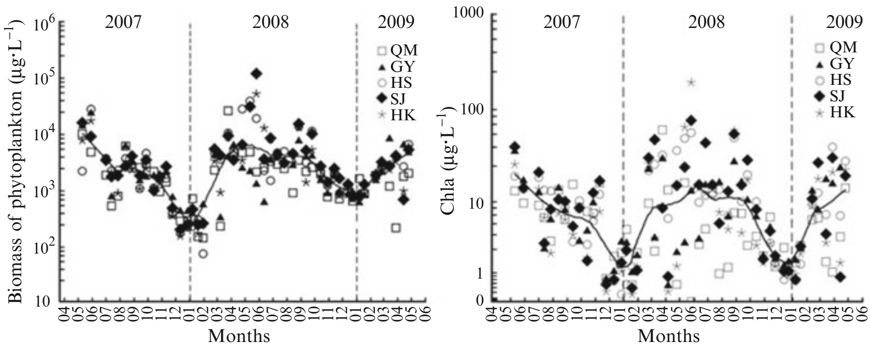

Fig. 12.5 Seasonal variations of phytoplankton biomass (*BioM*) and chlorophyll *a* (*Chl a*) in the PBA

12.4.3 Variation and Forms of Nitrogen and Phosphorus

Mean values of the TN and TP were 1,623 and 75.9 μg L^{-1}. No significant differences among the five sampling spots were found (One-way ANOVA, $P \leq 0.01$). There were apparent seasonal variations in the PBA (Table 12.3). Concentration levels of TN and TP started to increase in winter and reached a maximum at the end of spring. High variance of TN and TP occurred during the summer flood, and both TN and TP reached a minimum during autumn. Nitrate and ammonia were the major forms of nitrogen in the PBA, while the major forms of phosphorus were particulate and soluble reactive phosphate. Seasonal variation of forms of nitrogen and phosphorus was apparent. Increase of nitrate and SRP was detected in autumn and reached a maximum in winter. Increase of ammonia and PP was substantial during the flood season.

Mean TN/TP ratio mass ratio was 24.5. There was a significant decrease during the 2 years (Fig. 12.6). However, seasonal variation was high. Minimum levels of TN/TP ratio were frequently during winter and spring. TN/TP ratio in autumn was generally higher than in other seasons. Mass ratios of NH_4^+/NO_3^- and PP/SRP are also shown in Fig. 12.6.

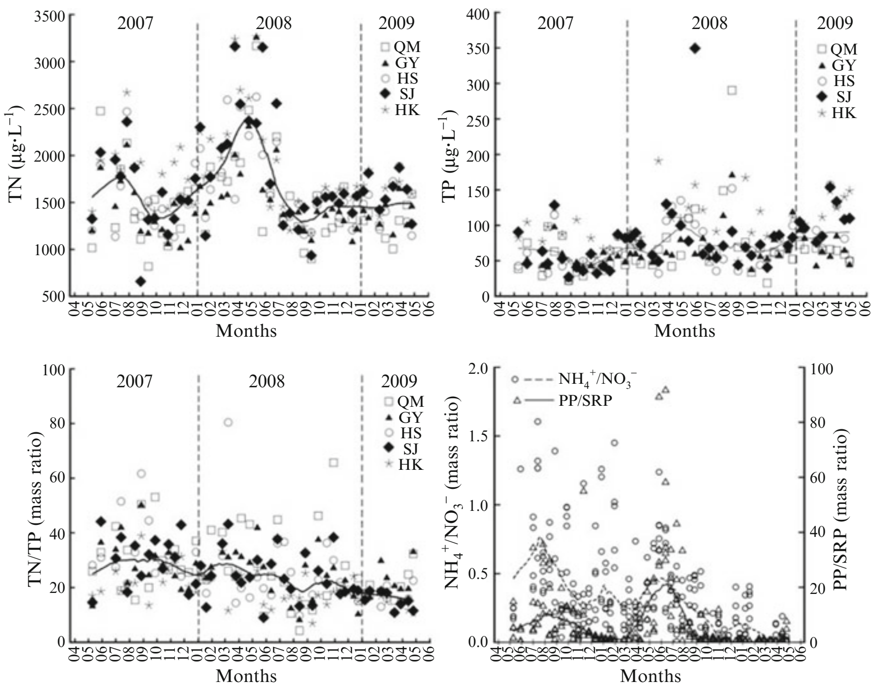

Fig. 12.6 Seasonal variations of TN, TP, TN/TP ratio, and the ratios of NH_4^+/NO_3^-, PP/SRP in the PBA

12.4.4 Correlation Analysis of Phytoplankton Biomass and Nutrients

Spearman correlations on all data sets are shown in Table 12.4. There was a positive correlation between TN and TP, but neither correlated significantly with BioM, Chl *a* and the physical factors AveWRT, AveRain, and AveQ. The correlation between TP and TN/TP ratio was much higher than that of TN and TN/TP ratios. However, there was no clear relationship between TN/TP ratio and phytoplankton biomass. While no correlation was found between nitrate and ammonia, SRP and PP were negatively correlated. Ammonia, nitrate, SRP, and PP all related positively to the physical factors in the PBA. Increase in nitrate and SRP frequently co-occurred with an increase in WRT and decreases in rainfall and discharge. Increase in ammonia and PP was frequently co-induced by a decrease in WRT and an increase in rainfall and river discharge. Moreover, impact of physical factors on forms of phosphorus was much more apparent than on nitrogen. Negative correlation between inorganic forms of nutrients and phytoplankton biomass (BioM and Chla) indicated that the potential uptake process of nutrients by phytoplankton, nitrate, and SRP uptake by phytoplankton in the PBA were much more apparent than that of ammonia. Increase in phytoplankton biomass was also significantly related with the decrease of WRT and increase in rainfall and river discharge.

For a deeper insight in the potential impact of artificial reservoir operation on primary production, all data sets were divided into two subsets: growth season and non-growth season according to the seasonal variation of BioM and Chl *a* in the study. The growth season represented the period from late February to mid-September, while non-growth season was from late September to mid-February of the next year. Division of the phytoplankton growth season coincided with the reservoir operation stage and the variation of river discharge and rainfall. In non-growth season, the reservoir was impounded to a high water level stage, and decrease in river discharge enlarged the WRT significantly. In the growth season, water level of TGR gradually decreased for power generation and was kept at a low water level for flood control. Moreover, according to the co-variance of WRT and phytoplankton biomass in the growth season (Fig. 12.7), two subsets were further divided: AveHRT < 50 day and AveHRT ≥ 50 day. Spearman correlation coefficients between phytoplankton biomass (BioM and Chl *a*) and nutrients in three data subsets above are shown in Table 12.5. Phytoplankton biomass and the selected limnological factors are shown in Fig. 12.7.

Phytoplankton biomass in the non-growth season was much lower than in the growth season. Concentrations of TN and TP were relatively higher in the growth season than that in the non-growth season. However, no significant difference of TN/TP ratio was observed among the three subsets compared with an increasing trend of NH_4^+/NO_3^- ratio and PP/SRP ratio from non-growth season to growth season with AveWRT < 50 day.

In the non-growth season, ammonia was utilized more than nitrate. Increase of TN and TP co-occurred with a decrease in phytoplankton biomass and water

Table 12.4 Spearman correlation matrix among limnological factors in the PBA based on all data sets

$N = 235$	NH_4^+-N	NO_3^--N	DIN	TN	SRP	PP	TP	TN/TP ratio	Chl *a*	BioM	AveWRT	AveRain
NH_4^+-N	1.000											
NO_3^--N	–	1.000										
DIN	0.476^{**}	0.769^{**}	1.000									
TN	0.350^{**}	0.430^{**}	0.601^{**}	1.000								
SRP	-0.390^{**}	0.427^{**}	0.170^{**}	–	1.000							
PP	0.285^{**}	–	–	0.243^{**}	-0.481^{**}	1.000						
TP	–	0.472^{**}	0.395^{**}	0.371^{**}	0.386^{**}	0.375^{**}	1.000					
TN/TP ratio	0.301^{**}	-0.209^{**}	–	0.212^{**}	-0.461^{**}	-0.181^{**}	-0.791^{**}	1.000				
Chla	–	-0.373^{**}	-0.372^{**}	–	-0.333^{**}	0.218^{**}	–	–	1.000			
BioM	–	-0.287^{**}	-0.303^{**}	–	-0.309^{**}	0.241^{**}	–	–	0.801^{**}	1.000		
AveWRT	-0.346^{**}	0.362^{**}	–	–	0.627^{**}	-0.475^{**}	–	-0.204^{**}	-0.467^{**}	-0.525^{**}	1.000	
AveRain	0.152^{*}	-0.191^{**}	–	–	-0.444^{**}	0.406^{**}	–	–	0.348^{**}	0.447^{**}	-0.863^{**}	1.000
AveQ	0.137^{*}	-0.206^{**}	–	–	-0.352^{**}	0.308^{**}	–	–	0.377^{**}	0.492^{**}	-0.824^{**}	0.932^{**}

– no significant correlation; *significance level 0.01; **significance level 0.05

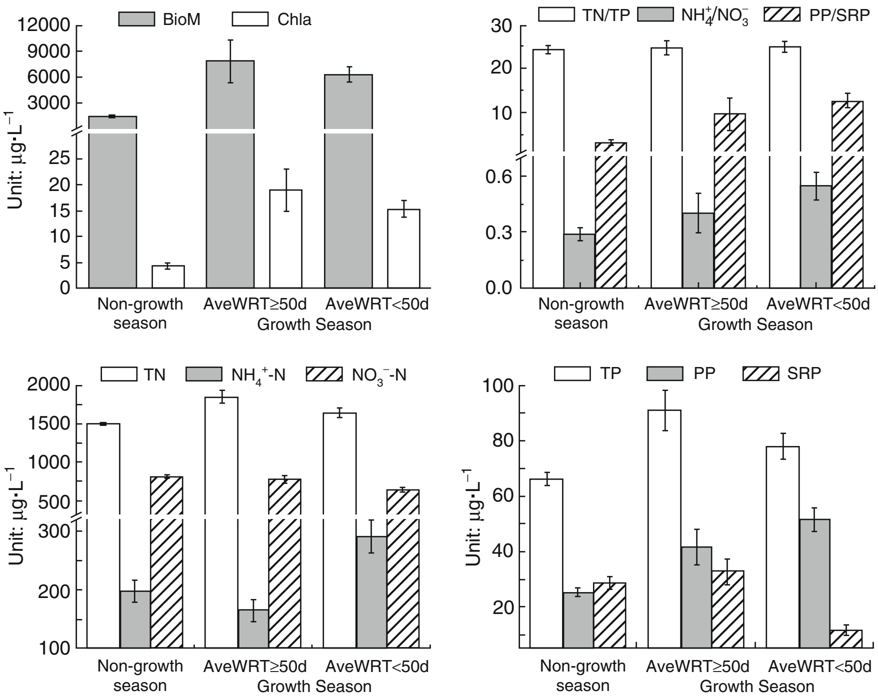

Fig. 12.7 Key environmental factors of the divided data subsets of growth season and non-growth season

Table 12.5 Correlation coefficients of phytoplankton biomass and limnological variables in growing and non-growing season

	Non-growth season (n = 100)			Growth season					
	Mean AveHRT = 98.4 ± 3.7 day			AveWRT < 50 day (n = 80)			AveWRT ≥ 50 day (n = 55)		
	BioM	Chla	TN/TP	BioM	Chla	TN/TP	BioM	Chla	TN/TP
NH_4^+–N	−0.451	−0.247	0.278	−0.200	−0.277	0.329	–	–	0.277
NO_3^-–N	–	−0.403	−0.362	−0.271	−0.239	–	–	–	–
DIN	−0.495	−0.494	–	−0.338	−0.370	0.268	–	–	–
SRP	−0.253	−0.381	−0.643	–	–	−0.375	–	–	−0.478
PP	–	–	–	−0.250	−0.263	−0.425	0.408	0.203	–
TN	−0.317	−0.252	–	–	–	0.409	0.471	0.139	–
TP	−0.255	−0.371	−0.835	–	–	−0.742	–	–	−0.837
TN/TP	0.207*	0.305**	1.000	–	–	1.000	–	–	1.000
AveWRT	−0.414	−0.403	−0.480	0.177	0.221	–	–	–	–
AveQ	0.544	0.339	–	–	–	0.268	–	–	−0.376
AveRain	0.573	0.239	0.237	–	−0.229	–	–	–	–

– no significant correlation; *significance level 0.01; **significance level 0.05

residence time, and an increase in rainfall and river discharge. In the growth season, there was no clear relationship between TP and phytoplankton biomass. No relationship between SRP and phytoplankton biomass was observed either. With AveWRT $\geq$ 50 day, TN positively correlated with phytoplankton biomass, while no correlation between TN and phytoplankton was observed when AveWRT $<$ 50 day. A positive correlation between water residence time and phytoplankton biomass was observed in the condition of AveWRT $<$ 50 day, while no clear relationship between the two variables above was detected with AveWRT $\geq$ 50 day. Impact of TN/TP ratio on phytoplankton biomass in the PBA was not clear in the growth season. However, in the non-growth season, a positive relation between phytoplankton biomass and TN/TP ratio was present. Moreover, AveWRT negatively correlated with TN/TP ratio ($r = -0.480$, Sig. < 0.01) in the non-growth season. In the growth season, there were weak correlations between TN/TP ratio and AveQ as well as AveWRT in both subsets. The correlation between TN/TP ratio and AveQ with AveWRT $\geq$ 50 day was -0.376 (Sig. < 0.01), and that with AveWRT $<$ 50 day was 0.268 ($P < 0.05$). Increase in TN/TP ratio in non-growth season co-occurred with the increase of NH_4^+/NO_3^- ($r = 0.326$, $P < 0.01$) and PP/SRP ($r = 0.421$, $P < 0.01$). In the growth season, NH_4^+/NO_3^- co-increased with the increase of TN/TP ratio. The correlation coefficient between TN/TP ratio and NH_4^+/NO_3^- in the condition of AveWRT $\geq$ 50 day was 0.356, while that in the condition of AveWRT $<$ 50 day was 0.353. However, there was no clear relationship between TN/TP ratio and PP/SRP in the growth season.

12.5 Discussion

The TN/TP ratio theory leads to two important inferences in lakes (Bulgakov and Levich 1999; Hall et al. 2005): (1) a decrease in TN/TP ratio, or increase of relative abundance of phosphorus, induces an increase in phytoplankton biomass and (2) a decrease in TN/TP ratio supports a dominance of Cyanobacteria, especially in shallow lakes. Furthermore, Quriós (2003) found that an increase of NH_4^+/NO_3^- was related to a decrease of TN/TP in lakes. Although this information is helpful to environmental policy makers searching for strategies to control nutrients, there remain doubts on the impact of the TN/TP ratio on primary production and phytoplankton species.

In natural lakes with stable physical boundary (=a stable water level), input of TN and TP is regulated by input (=pollution) from terrestrial ecosystems and by nutrient release from sediments (Xu et al. 2010). However, in natural lakes, seasonal variation of the TN/TP ratio is also related to biological processes, especially phytoplankton dynamics (Barica 1990). Spring or autumn minima in TN/TP ratio may trigger a dominance of N-fixing Cyanobacteria, while an increase in TN/TP ratio may lead to a dominance of Chlorophyta (Barica 1990; Havens et al. 2003). In Lake Taihu, a large lake in the downstream of the Yangtze River, asynchronous dynamics of TN and TP resulted in a TN/TP ratio that that ranged from 33 to 80 in winter and spring, and below 20 in summer (Xu et al. 2010). A temporal decrease in TN/TP ratio was regarded as a consequence of cyanobacterial blooms by Xie et al. (2003) and the increase in the TN/TP ratio was caused by excess nitrogen input from agricultural farmland (Xie et al. 2007).

Compared to natural lakes, variation of TN/TP ratio in lotic systems (streams, rivers, etc.) is co-impacted by geochemical and morphological characteristic of the watershed, meteorological process, and human activities. Green and Finlay (2010) showed that TN/TP ratio positively relates to river discharge under humid climate conditions. A sharp decline in TN/TP ratio generally occurred after storm events; in arid or semiarid climates, a negative correlation between TN/TP ratio and river discharge might be more apparent. Therefore, a complex physical background may manipulate the fate and transport of nutrients as well as the phytoplankton via habitat transformation. TN/TP theory may therefore not apply in a simple way in a river-reservoir hybrid system like the PBA.

In the growth season, decreased water level and increased river discharge significantly reduced water residence time in the PBA. It became a lotic system, fed by natural discharge in summer. Synchronous input of TN and TP ($r = 0.471$, $P \leq 0.01$ in the growth season) to the water column occurred by the heavy rainfall and intense soil erosion in the watershed. This led to the increase in PP/SRP and NH_4^+/NO_3^- ratio in the growth season, especially when AveWRT $<$ 50 day. The increase of TN/TP ratio positively correlated with the increase of AveQ in this condition. This is consistent with Green and Finlay's (2010) results, and indicates hydrological control of the TN/TP ratio in the PBA. Mass transport nature in the PBA also caused unstable hydrodynamic conditions for the growth of phytoplankton. Light availability might have been the factor that regulated the biomass of phytoplankton and led to a positive correlation between the AveWRT and phytoplankton in the condition of AveWRT $<$ 50 day in the growth season. However, in the growth season with AveWRT $\geq$ 50 day, increase in AveQ generally led to a decline in TN/TP ratio, and TN positively correlated with BioM and Chl *a*. Since the mean value of the TN/TP ratio was below 30, it was inferred that relatively stable hydrodynamic conditions occurred and provided adequate conditions for N-fixing process in the PBA. Occurrence of the N-fixing Cyanobacteria bloom in the growth season with AveWRT $\geq$ 50 day after storm event in the PBA in May 2007 supported the inference above. It was concluded that, in such circumstances, nitrogen limitation might prevail.

In the non-growth season, decrease in river discharge and increasing water level in the PBA caused a longer water residence time in the PBA, structuring a lentic system with a WRT up to 157 day. Decrease in water temperature and solar radiation co-occurred with the increase in WRT that structured the "calm period" for the growth of phytoplankton. Sedimentation of phytoplankton and a decline in physiological activities reduced the level of primary production. Moreover, dissolved forms of nutrients were released from the suspended nutrients and from the submerged water fluctuation zone due to the impoundment. This resulted in an increase of SRP, and a dominance of nitrate prevailed. A positive correlation between TN/TP ratio and phytoplankton biomass in the PBA might result from lower phytoplankton growth and phosphorus released from a newly submerged water fluctuation zone. This was expressed in a positive correlation between TN/TP ratio and phytoplankton biomass (BioM, Chl *a*) and a negative relationship between TN/TP ratio and AveWRT.

Based on the above analysis, artificial reservoir operation was seen to induce a shift of the aquatic ecosystem in the PBA from a lotic type in summer to a lentic type in winter. Under this regime, the relationship between TN/TP ratio and primary production (phytoplankton biomass) may not be as clear as that in lake systems, caused not only by the special pattern in fate and transport of nutrients, but also by a close relationship with the vegetation process of phytoplankton. Because of the complexities of the TN/TP ratio and primary production under reservoir operation, more information is needed on nutrient limitation effects in the PBA. Further studies should address questions related to patterns of habitat transition in the context of reservoir operation and the regulation effect this operation on phytoplankton in-situ dynamics, nutrient stoichiometry, and ecological responses of phytoplankton under different modes of reservoir operation.

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Chapter 13
River Basin Environments and Ecological Succession in Danjiangkou Reservoir

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Abstract Danjiangkou Reservoir is one of largest reservoirs in China, with a storage capacity of 174.5×10^8 m^3, located at the starting point of the middle route project, and functions as the water source of the project. The middle route of the south-to-north water transfer project is a large project for allocating water to northern China. The water quality and health status of the reservoir have received much attention from the public and the Chinese government. In the present study, limnological features of Danjiangkou Reservoir and their long-term succession are explored on the basis of historical investigations. For half a century since its first filling, the reservoir has become transformed from a river to a lake-style reservoir. In the last decade, it has received increasing amounts of nutrients and it currently shows a trend toward eutrophication and water quality deterioration. The standing crop of phytoplankton has substantially increased, while large zooplankton crustaceans decreased. Oligochaeta and chironomidae account for a large part of the benthos. Although fish resources have recovered to some extent through a fishing moratorium and artificial fish breeding and restocking, many problems still remain in the fishery management. Long-term monitoring and limnological studies keep being required for the management of the water quality of the reservoir.

13.1 Introduction

Danjiangkou Reservoir is the water source of the middle route of the south-to-north water transfer project, upon which rests the heavy burden of supplying water to over 20 cities including Beijing, Tianjin, and Shijiazhuang. The reservoir's unique

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location and importance increase the significance of its ecological health. Unlike lake ecosystems, reservoirs have specific limnological features (Straskraba et al. 1993). This study discusses the current status and succession of the living community in Danjiangkou Reservoir, based on the data of several systematic investigations of its water quality and aquatic organisms. It aims to provide basic information for a long-term study of this reservoir's limnology as well as for the ecological protection of the middle route of the south-to-north water transfer project.

13.2 The River Watershed and the Project of South-to-North Water Transfer

13.2.1 Hanjiang River Basin

Danjiangkou Reservoir was built in the upstream sector of Hanjiang River, a large branch of Yangtze River. The Hanjiang River watershed is located between 30°N 106°S and 34°N and 114°S, covering an area of 159,000 km^2. The main stream is divided into three sections: the upstream one at Danjiangkou, the middle reaches from Danjiangkou to Zhongxiang, and the downstream section of Zhongxiang. The upper reach is about 925 km in length, with a catchment area of 95,200 km^2. It is located in high mountains and deep river valleys, with few open basins and swit, torrential waters with many beaches. The middle reach is about 270 km long, with a catchment area of 46,800 km^2. This section represents a hilly valley basin, with unstable riverbeds and frequent floods. The lower reaches run through Hanjiang plain, and are about 382 km in total length with a catchment area of 17,000 km^2.

The average width of the main stream of Hanjiang is about 200–300 m, with an average slope of 6% and surrounded by high and steep mountains, narrow valleys, and swift torrents. Large tributaries such as Jushui River, Bao River, Xushui River, Youshui River, Ziwu River, Yue River, Xun River, Jinqian River, Danjiang River and Laoguan River are situated on the left bank, while Yudai River, Yangjia River, Muma River, Ren River, Lan River, Huangyang River, Ba River, and Du River are on the right bank. In total, 215 rivers have catchment areas above 100 km^2, against 21 rivers above 1,000 km^2.

Hanjiang River watershed has a north subtropical monsoon climate with moderate humidity. Its mean annual temperature is 15–17°C. Average annual sunshine amounts to 1,717 h, and the average frost-free period is 231 days. The average annual rainfall is 700–1,000 mm for the entire watershed, decreasing progressively from south to north and from west to east, and about 880 mm in the upstream part of Danjiangkou. The maximum precipitation from May to August in Hanjiang River watershed accounts for 55–65% of annual precipitation. The flood season in the upper reaches starts from May until October, and extends from April to September in the lower reaches. The rainfall during flood season accounts for 75–80% of annual precipitation, which can easily cause floods in the plain area. Mean annual evaporation in the watershed is 900–1,500 mm. The upstream Danjiangkou has a mean annual runoff of about 38 billion cubic meters, accounting for 75% of the water amount of the entire watershed, and with an average annual flow rate of around 1,200 m^3/s.

The rock series along the river basin consist of igneous, metamorphic, and sedimentary rocks, and the upstream of Danjiangkou is mainly of igneous and metamorphic rock. The soils along the river include brown forest soil, terra rossa, wet soil and podzolic brown forest alluvial soil, podzolic soil, purple soil, black calcareous soil, and alpine meadow soil. Forests along the river mainly occur in the upstream zone. Hills and mountains below 2,000 m above sea level are mostly covered by economic forest.

13.2.2 Danjiangkou Hydraulic Project

Danjiangkou Hydraulic Project is a key project for the Hanjiang River watershed management and development. The dam is located in Danjiangkou City, Hubei, about 800 m downward from the convergence of the Hanjiang River and its tributary, the Danjiang River (Fig. 13.1). The reservoir is composed of parts of the Hanjiang and Danjiang Rivers.

Fig. 13.1 Danjiangkou Hydraulic project

The implementation of the south-to-north water transfer project was divided in two stages. Its main priority at the earlier stage was flood control, followed by power generation, irrigation, navigation, aquaculture, and tourism. The focus at the later stage shifted to a diversion of the Hanjiang River into the Yellow and the Huaihe Rivers, and constituted a key part of south-to-north water transfer. However, the primary task remained flood control, followed by irrigation, power generation, navigation, aquaculture, and tourism.

The project started in 1958, and impoundment began in 1967. The first power generation happened in 1968, and it was completed in 1974. The main buildings of this project include an earth-rock dam on the left bank, connecting the dam section on the left bank, a powerhouse dam section, an overflow dam section, a deep-hole spillway dam, a ship lift, connecting the dam section on the right bank to the earth-rock dam on the right bank, etc. The main dam is a concrete slotted gravity dam, with a maximum height of 97 m. The installed power is 900 MW, and the average annual power generation is 3.83 billion kilowatt hours. At the earlier stage of this project, the normal high water level of the reservoir was 157 m (Wusong Elevation), impounded area 7.47×10^4 hm^2, total reservoir capacity 174.5×10^8 m^3, flood control storage 77.2×10^8 m^3, dead storage level between 139 and 140 m, and corresponding reservoir capacity $(72.3-76.5) \times 10^8$ m^3. For the subsequent project, crest elevation was 175 m, normal water level 170 m, corresponding reservoir capacity 29.05 billion cubic meters, and the surface water area of the reservoir 598–1,050 km^2. The performance parameter and operation curve of the reservoir is listed in Table 13.1 and shown in Fig. 13.2.

Table 13.1 Performance parameter sheet of the project

Item		Unit	Early stage	Subsequent stage
Water level	Normal water level	m	157	170
	Designed low water level	m	139	150
	Limiting level during flood season	m	149–152.5	160–163.5
	Designed flood level	m	160	172.2
	Check flood level	m	161.4	174.35
	Average water head	m	59.3	69.72
Reservoir capacity	Reservoir capacity at normal pool level	100 million m^3	174.5	290.5
	Reservoir capacity at designed low water level	100 million m^3	72.3	126.9
	Regulation		Annual regulation	Multi-year regulation
Drainage	Flood drainage for a century	m^3/s	18,100	5,960
	Designed flood drainage	m^3/s	35,300	22,300
Structure size	Total length of the dam	m	2,494	3,446
	Elevation of deep-hole sill	m	113	113
	Elevation of spillway crest	m	138	152
Submerge	Inundated farmland	10,000 mu	42.9	20.1
	Migrant population	10,000 persons	38.27	22.4

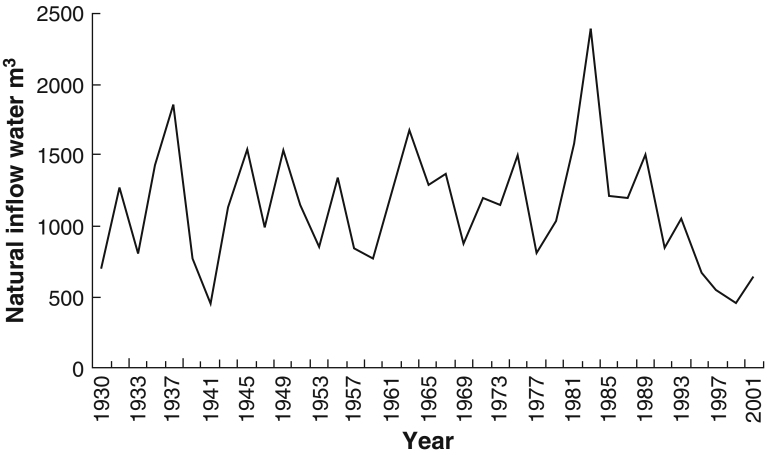

Fig. 13.2 Inflow to the reservoir from 1930 to 2001

13.2.3 The Middle Route of the South-to-North Water Transfer Project

The middle route of the south-to-north water transfer project is a critical infrastructure project to mitigate the water crisis in North China. The middle-term planning of this project is as follows: to divert water from Danjiangkou Reservoir to Hanjiang River, channelize it along Funiushan Mountain and Taihang Mountains, and finally deliver it to Beijing. The long-term planning is to divert water from the Three Gorges Reservoir on the Yangtze River or main reaches of the Yangtze River to increase the amount of water transferred to the north. The construction projects involved the water source, water storage, main channels, an ecological compensation project at the middle and lower reaches of the Hanjiang River, and the Yellow River Crossing Project. The project is mainly composed of water source regulation and water diversion, with a total investment of ca 95 billion RMB. Engineering of water sources includes the Danjiangkou Hydraulic Project and other projects at the middle and lower reaches of the Hanjiang River. Engineering of water diversion refers to diversion from the main channel of the Hanjiang and Tianjin trunk. Danjiangkou Reservoir controls 60% of the area of Hanjiang River watershed, with an average annual runoff of 408.5×10^8 m^3. Considering the development of the upper reaches, it is expected that reservoir inflow will reach 385.4×10^8 m^3 in 2020. Upon completion, the elevation of the dam crest will increase from the current 162 to 176.6 m, and the designed storage water level will rise from 157 to 170 m. The overall reservoir capacity will reach 290.5×10^8 m^3, up 116×10^8 m^3 over the initial stage. Usable storage will increase by 88×10^8 m^3, and flood control storage will rise by 33×10^8 m^3. Under the normal pool level of 170 m, considering the economic development level in 2020, some ecological compensation projects at the middle and lower reaches of Hanjiang River (aiming to ensure water supply for industrial and agricultural development, shipping, and environment at the water source area) will soon start. The average annual transferable water quantity of the middle route project will be 141.4×10^8 and 110×10^8 m^3 during low flow years (reliability of 75%). The total length of main channel will be over 1,400 km, connecting the Yangtze River, Huaihe River, Yellow River, and Haihe River, and supplying water to Beijing, Tianjin, Hebei, Henan, and Hubei. According to preliminary estimates, the annually transferred water quantity will reach 130×10^8 m^3 upon completion, benefiting 34.68 million people and mitigating water shortage in Beijing, Tianjin, and northern China. It is expected to improve the ecological environment and promote the development of the national economy. However, after water diversion, the hydrological regimes in the regions supplied, as well as at the water source will change greatly. Together with the impacts of project construction and migration, the ecological environment will face a great challenge.

13.3 Environmental Conditions of Danjiangkou Reservoir

13.3.1 Geographic Conditions

Danjiangkou Reservoir is located in the transitional zone of the Qinling Mountains, Daba Mountains, and Jianghan Plain. The reservoir runs across Danjiangkou City, Yunxian County, Yunxi County of Hubei Province, and Zhechuang County in Henan Province, controlling a watershed of 95,200 km^2. The dam is 0.8 km downward from the convergence of the Hanjiang River and its tributary, the Danjiang River. The Hanjiang River upward of the dam is over 800-km long, with an average slope of 0.6%. Both banks of the rivers are mostly conglomerate and sandstone, and the rest is limestone and igneous rock, with low vegetation coverage. In the upper reaches of Hanjiang River, tributaries with a watershed of over 5,000 km^2 include Ren River, Xun River, Jia River, Danjiang River, and Du River, which directly pour into the reservoir. The backwater length of the reservoir is about 170 km of the former course of the Hanjiang River and 80 km of the Danjiang River. The reservoir bankline is over 4,000-km long, with over 3,000 bays of different sizes. The open water zone mainly includes Xiaochuang, Yangxi, Liupi of Hanjiang, and Liguanqiao, Laocheng, and Madeng of Danjiang. Liguanqiao covers the largest area, with about 2×10^4 hm^2 and a width of 20 km. The average annual amplitude of the water level of the reservoir is about 17 m, and the fluctuation area from the water level at 157–140 m is 3.2×10^4 hm^2.

13.3.2 Geological Conditions

Danjiangkou Reservoir and its upstream area are located in the Qinba mountain region. The region has a complex geological structure and consists of a rugged terrain. Rocks consist mainly of gneiss, sand-shale, and limestone, and common geological disasters in this area include collapse, landslides, and debris flow. The river is sloping from west-north to south, from over 2,000 m at the river head to 143 m at Danjiangkou Reservoir, with a complex geomorphology. It consists of middle mountains, low mountains, hills, and basins, 500–2,000 m high, and alternating canyons and basins. The topography is quite variable. As we can see from the slope statistics in Tables 13.2 and 13.3, the 6–15° slope covers the largest area in this region, accounting for 33.82% of the total area. This is followed by a 16–25° slope, accounting for 31.48% of the total area. The 0–5°, 26–35°, 36–45°, and above 45° slopes account for 23.18%, 10.63%, 0.87%, and 0.02%, respectively, of the total area. Tables 13.2 and 13.3 show the statistics of slopes and slope directions in the study area.

13.3.3 Ecological Conditions

Due to the large capacity of the Danjiangkou Reservoir and its strong self-purifying capability, the water quality is currently high. According to monitoring data, the water quality reaching class I or class II standards accounts for 86% of the total reservoir area, class III for 10%, and class IV for 4%. The overall water quality conforms to class II according to the environmental quality standards for surface water (GB3838-2002). Upon the completion of the middle route project, this reservoir will divert water to Central and North China at a flow rate of 500 m^3/s.

However, with the social and economic development at the water source region, the main stream and tributaries of Danjiangkou Reservoir, Hanjiang River, Du River, Lang River, Si River, Tian River, Zi River, Shending River, Laoguan River, Guanshan River, Jian River, and Bai River are being polluted by a number of industries, e.g., automobile manufacturing, mining, pharmacy, food, machining,

Table 13.2 Statistics of areas with different slope (unit: km^2, %)

Slope	0–5° Flat	6–15° Flat and gentle	16–25° Slope	26–35° Abrupt	36–45° Steep	Above 46 Dangerous	Total
Area	24,204.7	35,319.58	32,869.55	11,097.07	908.53	22.38	104,421.8
Percentage	23.18	33.82	31.48	10.63	0.87	0.02	100

Table 13.3 Statistics of areas with slope directions (unit: km^2, %)

Slope direction	Flat	North	Northeast	East	Southeast	South	Southwest	West	Northwest	Total
Area	7,200	11,373	12,890	12,632	12,399	12,113	12,781	11,987	11,042	104,421
Percentage	6.9	10.89	12.34	12.10	11.87	11.60	12.24	11.48	10.57	100

chemical engineering, paper-making, and building materials production. In addition, large quantities of fertilizers and pesticides are used every year in agriculture, leading to the eutrophication of some parts of the reservoir. In recent years, over 200 types of trace organic matter have been detected in the reservoir, and low contents of poisonous substances and heavy metals have been found in reservoir sediment. Laoguan River and Shending River are the most severe sources of polluted water, posing a threat to the quality of the water transferred from the reservoir.

According to a national satellite investigation, the area of soil erosion at Danjiangkou Reservoir is 47,422.23 km^2 (not counting the area submersed after heightening of the dam), accounting for 53.82% of the total land area. The area of severe soil erosion is 1,191.86 km^2, accounting for 2.51% in total. The average annual amount of soil erosion is 169 million tons. The area of extremely serious soil erosion is 4,150.74 km^2 (accounting for 8.75% of total area of soil erosion), that of serious soil erosion is 9,350 km^2 (19.72%), that of moderate soil erosion is 16,888.19 km^2 (35.61%), and that of mild soil erosion is 15,841.07 km^2 (33.41%).

13.4 Limnological Features of the Reservoir

13.4.1 A General Profile

The current limnological features of Danjiangkou Reservoir are described in Table 13.4. The heightening of the dam resulted in an increase in surface water area, and will impact the regional climate around the reservoir. It is estimated that the mean annual temperature of the reservoir area will increase by 0.1°C, the average temperature in February will rise by 0.4°C, in April and July by 0.2–0.3°C, and in September it will decrease by 0.4°C. Water evaporation will increase to some extent over the current level, and mean annual humidity will rise by 2–4%. The wind speed increase rate will be inversely proportional to the width of reservoir water area; the northern section has a wider water area and its average wind speed will increase by 8%, while the southern section has a narrower water area and its average wind speed will increase by 11% (Zhang and Zhao 2005).

13.4.2 Dynamics of Limnological Variables

13.4.2.1 Inflow and Precipitation

The inflow into Danjiangkou Reservoir has two peaks, in July and September, respectively. The maximum monthly average inflow occurs in September, followed by July, and substantially decreases after October. Annual changes in inflow define

Table 13.4 Limnological variables of the Danjiangkou Reservoir

	Item	Current situation
Hydrological and hydrodynamics variables	Flow rate	Run-off is unevenly distributed within a year, with huge inter-annual changes. Mean annual run-off is 379×10^8 m^3. Mean annual flow rate at the dam is 1,200 m^3·s, maximum average inflow rate is 19,200 m^3/s, and maximum average outflow rate is 8,350 m^3/s. High-flow period is from May to October, with inflowing water accounting for 78.25% of a whole year. Hanjiang River has huge inflow rate, and flows backward to Danjiang during flood season. During the flood season, the flow rate at Youfanggou of Hanjiang can reach over 5 m/s, and about 1 m/s at Anyangkou. During the dry season, the flow rate slows down to about 0.8 m/s in Longshanzui.
	Precipitation and evaporation	Mean annual precipitation is 850–950 mm; unevenly distributed over the year. Precipitation from May to October accounts for 80%, mainly in the form of torrential rains. Mean annual evaporation of the water area is 700–900 mm, gradually increasing from southwest to northeast, 630.4 mm from April to September and 274.9 mm from October to March. Mean annual humility is 69.3%, highest in July and lowest in November.
	Sediment discharge	The mean annual inflow of sediment to the reservoir is 83.10 million t/a, 79% from Hanjiang and 21% from Danjiang. The inflow of sediment from May to October accounts for 95% of the whole year. Total sediment deposition is 1.618 billion m^3 (at a water level of 157 m).
	Weather	A subtropical monsoon climate. Mean annual temperature gradually decreases from the valley of the Hanjiang to both banks on the south and north, and was 15.8°C before the construction of the reservoir but 15.6°C afterward. Monthly mean temperatures higher than 15°C start from April to October. Lowest monthly mean temperature is 2.4°C in January, and highest monthly mean temperature is 28°C in July. The extremely low temperature is −13°C, and extremely high temperature is 42°C. January is the coldest month and July is the hottest month. Annual cumulative temperature above 0°C is over 5,600°C, annual cumulative temperature above 5°C over 5,550°C, and annual cumulative temperature above 10°C is 5,123.3°C. Average annual sunshine hours is 1,968.3 h, and average

(continued)

Table 13.4 (continued)

	Item	Current situation
		sunshine rate is 45%; average annual radiation is 449.49 kJ/cm^2, April to October accounts for 71% of the whole year. The frost-free period lasts 248–254 days; first frost is around November 17, and last around March 10. Maximum wind force is at the 6–9 level, and storm wind or tornadoes occur in June and July.
Physical and chemical variables	Water temperature	Average surface temperature is 18.0°C, with the highest 28°C from July to August, lowest temperature is January, February and March at 9.0°C, 8.5°C and 8.5°C, respectively.
	Transparency	2.59 m (annual average over from 2004 to 2006), higher at the center of reservoir.
	pH	8.05 (in 2007), highest level before the dam and at reservoir center, lower at peripheral areas.
	Dissolved oxygen	10.5 mg/L (in 2007) with reservoir center higher than peripheral areas.
	Chemical oxygen demand	14.82 mg/L (annual average over from 2004 to 2006) with reservoir center lower than peripheral areas.
	Total nitrogen	1.32 mg/L (annual mean in 2004).
	Total phosphorus	0.014 mg/L (annual mean in 2004).
Pollution load	Waste water	Total quantity of waste water 153.587 million t/a, including industrial sewage 24.5739 million t/a, sanitary wastewater 38.3968 million t/a, and mixed wastewater 90.6163 million t/a (Hu 2009).
	COD discharge	COD discharge from urban living and industries is about 10.29 million tons (Gao 2002).
	NH_3-N discharge	NH_3–N discharge is about 5,400 t (Gao 2002).
Biological parameters	Bacteria (cfu/mL)	140 (in front of the dam in 2005) (Li et al. 2006).
	Heterotrophic bacteria	14 (in front of the dam in 2005) (Li et al. 2006).
	Fecal coliforms (number/L)	20 (in front of the dam in 2005) (Li et al. 2006).
	Average cell densities of phytoplankton	109.33×10^4 ind./L (2003–2005) (Zhang et al. 2006).
	Density of zoobenthos	33,792 ind./m^2 (in front of the dam in 2007 and 2008) (Zhang et al. 2010).

two flood seasons: summer flood in July and August and autumn flood in September and October (Jin and Zhao 2004).

According to the annual mean inflow from 1930 to 2004 and average rainfall in the river basin from 1960 to 2004, Danjiangkou Reservoir experienced relative high flows in the 1930s and 1950s to the 1960s, but lower rates in the 1940s, 1970s, and 1990s. Since the 1990s, inflow has decreased significantly (Fig. 13.2). This is

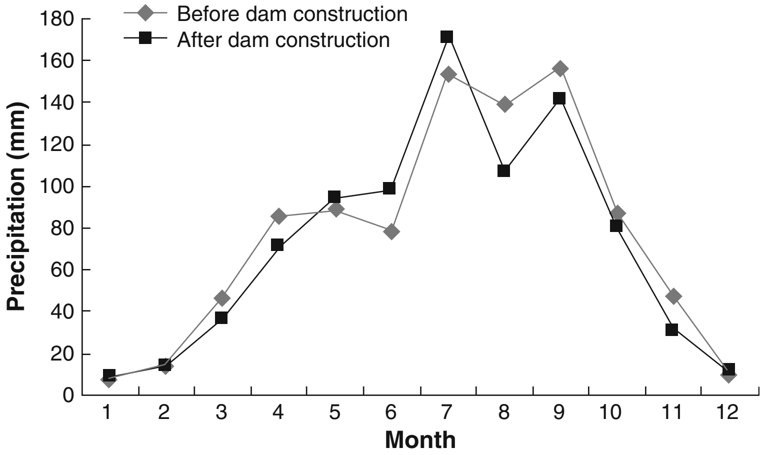

Fig. 13.3 Variation of monthly average precipitation in Danjiangkou Reservoir before and after the dam was built

largely related to climate, with heavy rainfall distribution during the flood season from May to October (Xu 2005).

From 1951 to 1978, precipitation at the reservoir remained constant; in the 1980s and 1990s, precipitation varied to some extent. Precipitation upstream of Danjiangkou Reservoir suddenly decreased in 1991, from a wet period in the 1980s to less rainfall from the 1990s to 2002. The change in precipitation has a cycle of 10 years (Chen et al. 2005).

The annual mean precipitation was 902.3 mm before the construction of the dam, and 861.7 mm afterward. This is down by 40.6 mm, a rate of 4.5%, despite increased precipitation in January, May, June, July, and December (Fig. 13.3). The heightening of the dam had little impact on the precipitation at the reservoir. Compared with the initial stage when the dam was 157-m high, the annual precipitation was reduced by 8.8–25.6 mm, i.e., 1.0–3.2%.

13.4.2.2 Sediment Discharge with Inflow

Both spatial and annual distribution of inflow and sediment at Danjiangkou Reservoir are extremely uneven. The inflow water and sediment concentrated at the Hanjiang River with a corresponding sediment deposition during the flood season. At the normal level of 157 m, Danjiangkou Reservoir has a total sediment deposition of 1.618 billion cubic meters. At the limiting level of 152.5 m during the flood season in autumn, total sediment deposition is 1.641 billion cubic meters. At the designed low water level of 139 m, total sediment deposition is 1.23 billion cubic meters. Generally speaking, sediment deposition at the reservoir from the main stream is around 85%, and sediment deposition from the tributaries is about 15%.

Before the reservoir was built, annual mean sediment discharge was 100 million tons. After impoundment, from 1960 to 1994, Danjiangkou Reservoir had a total sediment deposition of 1.41 billion m^3, which mainly accumulated during 1968–1986 with 1.16 billion cubic meters, accounting for 82% of the total. Annual mean inflow of sediment to the reservoir is 83.10 million t/a, 69.51 million tons from Hanjiang River, and 13.59 million tons from Danjiang River. In recent years, due to a number of hydraulic projects in the upper reaches, inflowing sediment decreased substantially compared with that before the dam was built. From 1990 to 1999, with average inflow relatively low, annual mean sediment discharge from Hanjiang into Danjiangkou Reservoir was only 9.2 million tons (Sun 2007).

13.4.2.3 Water Temperature

Water temperature of Danjiangkou Reservoir has a hierarchical structure with a temporal and spatial pattern. The area that lies 90 km away from the dam is the changeable backwater zone, where the hierarchical structure gradually disappears and turns into a riverine water temperature distribution pattern (Table 13.5).

The vertical distribution of water temperature is regulated by climate, flood peak, and reservoir operation. Before reservoir storage, the vertical distribution of water temperature was basically a positive hierarchical structure, without negative structure or discontinuity layer. After reservoir storage, from 1986 to 1987, no discontinuity layer was found. The surface temperature was highest in August, and water below 20 m had the highest temperature in September and October. Before the heightening of dam, from December to February, there was no substantial difference between temperature of surface and bottom; from April to August every year, the water temperature at Danjiangkou Reservoir showed an obvious hierarchical structure, and decreased with depth (Fig. 13.4). The water below 40 m is a low-temperature layer without major changes in temperature, but with a temperature difference from the surface water reaching 16°C; the water above 10 m is the warm water layer, and the interlayer is a discontinuity layer with a temperature gradient at −0.5°C/m. In summer, water temperature of the zone 90 km away from the dam still shows this hierarchical structure; for the water above 10 m and below 30 m, isothermal lines are very sparse, and for the water between 10 and 25 m are relatively intensive (discontinuity layer). From September to October, the temperature difference between the surface and bottom water decreases, and the hierarchical structure weakens.

Guo et al. (2008) predicted that the water temperature distribution after heightening of the dam would remain as before the heightening, i.e., from November to March in the next year, and the water temperature would be evenly distributed

Table 13.5 Change in water temperature in front of the dam

Before reservoir storage (1960–1969)	1986–1987	1991–1992	2001–2002	2005–2006
16.7°C	18°C	18.2°C	17.3°C	20.5°C

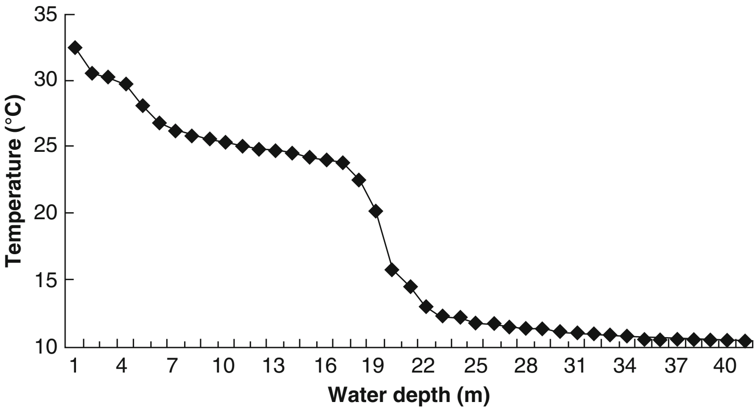

Fig. 13.4 Vertical profile of water temperature near the dam in summer in 2001

vertically. From April to October, the water temperature showed a hierarchical structure, and the depth and thickness of the discontinuity layer would be similar to that before the heightening of the dam. After the heightening, the water temperature structure of the reservoir would change, i.e., the annual increase rate of water temperature at the bottom would reduce, and the vertical temperature difference would increase, with some impact on the growth of aquatic organisms in the reservoir.

13.4.2.4 Water Transparency

Because of change in hydrologic conditions, there was evidently an improvement in water transparency after than before reservoir construction. However, as the reservoir began to store water, transparency changed differently in periods and zones. In terms of spatial pattern, transparency at Hanjiang River was lower than at Danjiang River. In terms of time, transparency near the dam increased constantly. Since the 1980s, water transparency at both Danjiang River and Hanjiang River had at first increased and then decreased. In general, the transparency of the reservoir remained high in the 1990s. In 2002, except for the water transparency near the dam at the convergence of the two rivers, both rivers dropped to a minimum. In 2004, there was a rise in transparency of Danjiang River with an average of 2.06 m, especially in May 2005, and water transparency at the center of Danjiang Reservoir reached 10 m. The changes in transparency in different periods are shown in Table 13.6.

13.4.2.5 pH, Dissolved Oxygen, and Chemical Oxygen Demand

pH before the 1990s remained at about 8, and was similar in the different water zones. Average pH in 2002 was 7.56, showing a significant decrease, and there was

Table 13.6 Change in water quality at Danjiangkou Reservoir (Wang et al. 1990)

Parameter		Transparency/cm	Total hardness	Total alkalinity	Conductivity	pH	Dissolved oxygen/ mg·L	Nitrate/ mg·L	Phosphate/ mg·L	Silicate/ mg·L	Chemical oxygen demand/mg·L
1950s	Hanjiang		6.66			8.2	6.86	0.037	0.05	4.46	
	Danjiang		9.36			8.2	7.88	0.293	0.035	6	
	Before dam		5.49			7.9	6.86	0.3	0.015	7.18	
1980s	Hanjiang	131.3	2.52	2.16				0.31	0.012	3.32	
	Danjiang	271.5	2.54	2.26				0.34	0.007	4.32	
	Before dam	146.3	2.52	2.19				0.3	0.01	4.4	
1990s	Hanjiang	148	2.43	2.7	201	8.06	9.66	0.093	0.015		1.48
	Danjiang	293	2.39	2.03	206	8.11	8.06	0.032	0.009		1.38
	Before dam	263	2.22	1.98	344	8	9.26	0.062	0.021		1.4
2002	Hanjiang	126.3	2.43	2.13	263	7.37	8.75	0.462	0.025	7.31	2.71
	Danjiang	250	2.71	2.36	277.3	7.66	9.45	0.4001	0.023	6.76	2.12
	Before dam	302.5	2.48	2.12	251.5	7.7	9.05	0.2593	0.025	6.7	2.86
2004–2005	Taocha	1.53			297.7	7.8	8.51				
	Danjiang	6.44			294.9	7.9	8.72				
	Before dam	8				7.97	9.54				1.83

a clear spatial difference: the highest pH occurred in front of the dam, followed by Danjiang River, and Hanjiang River. From 2004 to 2006, pH of the reservoir was between 6.91 and 8.4, with no difference near the dam and at the center, while the channel head in Taocha had a lower pH. Dissolved oxygen was highest in front of the dam, averagely 9.87 mg/L, followed by the center of the reservoir at 9.21 mg/L. At the channel head in Taocha it was 8.74 mg/L, and the lowest value (7.57 mg/L) occurred at Dashiqiao in the upper reach of Danjiang River. The chemical oxygen demand of the reservoir water body rose yearly, and it was highest in the upper reach of the reservoir area and minimum at the reservoir center.

13.4.2.6 Ions and Total Salinity

There was a slight increase in ions in the reservoir after storage, but still lower than that of reservoir inflows. The total concentration of ions in front of the dam before storing water (from 1959 through to 1967) was between 120 and 279 mg/L, averaging 183 mg/L. From 1969 to 1979, after storing water, the total concentration of ions was between 155 and 209 mg/L, averaging 188 mg/L. Ion concentration increased gradually with depth, being higher at the reservoir bottom and lower at the surface, with a vertical difference of about 20 mg/L (Hu 2003). Before storing water, the total hardness (German standard) in the area in front of the dam was between 4 and 9, on average 6. After storing water, it was between 5.4 and 7.2, averaging 6.5. The chlorinity before storing water was between 0.1 and 5.0 mg/L, averaging 1.3 mg/L. But it rose to between 0.6 and 6.5 mg/L after storing water, averaging 1.6 mg/L. The concentration of main ions was anions: HCO_3^-- > SO_4^{2-}- > NO_3^-- > Cl^-; cations: Ca^{2+} > Mg^{2+} > Na^+ > K^+.

Besides rock weathering, rain, atmospheric input, and anthropogenic input produced the ions in the reservoir water. HCO_3^- was mainly from rock weathering in the upper river basin, hardly an effect of human activities. In contrast, 90% of Na^+ and Cl^- were from anthropogenic discharge, so human activities were changing the hydro-chemical composition of some main ions. SO_4^{2-} mainly came from atmospheric fallout, and rainy season inflow was the main source of NO_3^- (Li 2008).

13.4.2.7 Nitrogen and Phosphorus

Based on the analysis of the concentrations of three types of inorganic nitrogen and total nitrogen content in the water before and after construction of the reservoir, changes in the different forms of nitrogen can be traced over the past four decades. Nitrate nitrogen dominated the three forms of nitrogen, making up 60% or more. But there was a significant change in the concentrations of ammonia nitrogen and nitrite nitrogen, the proportion of which fluctuated in different historical periods (Fig. 13.5). Before construction of the reservoir, the concentration of nitrate nitrogen was lower in Hanjiang River than that in Danjiang River and in front of the dam; thereafter, there was no significant spatial difference during the 1980s; however, concentrations

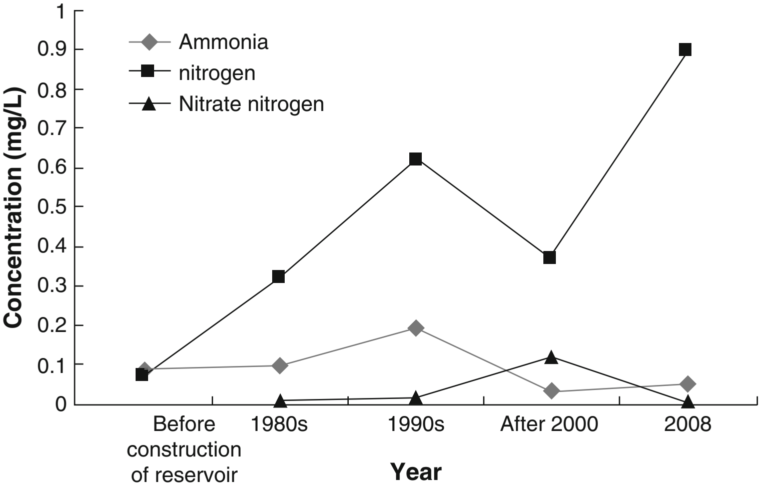

Fig. 13.5 Long-term changes of nitrogen in the reservoir

thereafter became higher than in front of the dam and Danjiang River. There was no spatial difference in the concentration of nitrite nitrogen.

Total nitrogen in the reservoir rose continuously, with an average concentration to 1.36 mg/L, or 1.3 times Class II of the State Standard for Surface Water at the mesotrophic level. There was no substantial spatial difference. From 2004 to 2006, total nitrogen at Taocha was 0.725 mg/L on average, and 0.735 mg/L at the center of Danjiang Reservoir; in 2008, it was 1.21 mg/L in Hanjiang Reservoir, 1.3 mg/L in Danjiang Reservoir, and 10.8 mg/L in front of the dam.

Prior to construction of the reservoir, phosphate was between 0.015 and 0.05 mg/L. It was higher in Hanjiang River than in Danjiang River, and lowest in front of the dam. After the construction of the reservoir, the phosphate concentration in the 1980s was between 0.007 and 0.012 mg/L, and Hanjiang River was close to that in front of the dam but a little higher than in Danjiang River. In the 1990s it was between 0.009 and 0.021 mg/L, highest in front of the dam, second in Hanjiang River, and lowest in Danjiang River. In 2002 it was between 0.023 and 0.025 mg/L, without evident spatial difference.

There were differences in horizontal distribution of total phosphate. The minimum appeared in front of the dam and, except for the relatively higher 0.078 mg/L found in 1992, phosphate content remained at an extremely low level, even below the detection limit, from 1995 through to 2000. From 2001 to 2003 it began to increase year after year, from 0.004 to 0.014 mg/L. The total phosphate concentration at the channel head in Taocha fluctuated greatly within the range 0.01–0.06 mg/L. Over the years the overall average phosphate concentration in the reservoir water was between 0.012 and 0.022 mg/L, the minimum occurring at the center (generally between 0.005 and 0.01 mg/L), with higher values in the peripheral zones of the reservoir.

Table 13.7 Pollutant discharge in the administrative regions in 2000 into Danjiangkou Reservoir (unit: t)

Administrative region (province)	COD			Ammonia nitrogen		
	Point pollution	Non-point pollution	Total	Point pollution	Non-point pollution	Total
Hubei	41,077.3	4,830.5	45,907.8	1,567.8	878.7	2,446.5
Henan	13,598.8	2,446.5	16,045.3	660	443	1,103
Shanxi	48,201	15,268.3	63,469.3	3,158.5	2,767.6	5,926.1
Total	102,877.1	22,545.3	125,422.4	5,386.3	4,089.3	9,475.6

13.4.2.8 Pollution Loadings

According to a survey in 2004 of the sewage outlets along the Yangtze River basin, there were 345 sewage outlets in total in the Hanjiang–Danjiang water system. Among these outlets, 161 were industrial discharges, 106 were domestic sewage and wastewater outlets, and 78 were mixed sewage outlets. In the Hanjiang–Danjiang water system, there were 51 sewage outlets that flowed directly into rivers. In 2000, the total direct sewage and wastewater discharge reached 153.587 million t/a. Total industrial sewage discharge accounted for 24.5739 million t/a, total domestic sewage discharge was 38.3968 million t/a, and total mixed sewage discharge was 90.6163 million t/a (Hu 2009). In the year 2000 urban domestic and industrial COD (chemical oxygen demand) discharge in the water source regions was about 10.29 million tons and ammonia nitrogen discharge about 5,400 tons, which is equivalent to a per capita COD discharge of 57 g/day. Urban people contributed COD discharges of between 60 and 65 g per capita per day, while small township people contributed COD 50 g per capita per day (Sun 2007) (Table 13.7).

13.4.2.9 Trophic Status

Except for total nitrogen and total phosphate, other physical and chemical indexes of Danjiangkou Reservoir water satisfied Class I of the Chinese National Standard for water quality, indicating no significant seasonally and yearly difference. In accordance with the criteria of single-factor evaluation and of Sladecek's biochemical oxygen demand evaluation, Danjiangkou Reservoir water was found to be mesotrophic. Evaluation by the trophic status index (TSI) also showed that Danjiangkou Reservoir water was in the mesotrophic state. However, there was a big spatial difference in trophy. The water at the center and in front of the dam was oligotrophic, mesotrophic in the peripheral zones of the reservoir, and eutrophic in some upstream water bodies. These states might vary dynamically. The trophic status of Danjiangkou Reservoir was evaluated through chlorophyll *a* concentration (Chla), which was between 0.001 and 0.005 mg/L at the center of the reservoir, with minimum in winter and maximum in summer. The TSI in winter was 30.4, indicating an oligotrophic status, while it was 46.3 in summer, indicating mesotrophy (Pang et al. 2008).

Since the reservoir was built, its water had been mesotrophic, but there were evident changes in nitrogen and phosphate concentrations. In particular, the nitrogen concentration was indicative of mesotrophy in the 1980s, moved to eutrophic in the 1990s, and now is beyond the eutrophic status. In the 1980s, the phosphate concentration in the reservoir was below the lower limit of mesotrophic status, but currently it has reached the upper limit of mesotrophy. Some bays of the reservoir have even been slightly eutrophic in some months.

13.5 Succession of Aquatic Communities

13.5.1 Phytoplankton

Since the reservoir was built, the community structure of phytoplankton such as composition, dominant species, density, and biomass have changed greatly (Tables 13.8 and 13.9). In 1958, species of phytoplankton in all sampling sections of Danjiangkou Reservoir were similar. Diatoms dominated absolutely, with other species making up a small proportion (Brodsky et al. 1959).

From 1986 to 1987, species remained similar and diatoms dominated still (Yang et al. 1996). *Fragilaria* and other taxa dominated in summer in Hanjiang basin, while blue algae dominated in Danjiangkou area. Seasonal variation in phytoplankton biomass was highest in July in Hanjiang basin, and in October in Danjiangkou area.

From 1992 to 1993, 60 genera of planktonic algal species were observed: 37 genera in Hanjiang basin and 26 genera in Danjiangkou basin. Total richness decreased gradually from Hanjiang to Danjiangkou area, while the species richness of green and blue-green algae increased. The annual average density of phytoplankton was 4.26×10^5 ind./L, among which diatoms represented 79.8% and green algae 6.1%. The annual mean biomass was 0.71 mg/L, with diatoms making up 62.5% and green algae 13.2%. The standing biomass changed noticeably with season in the order: September > June > December > March. In winter and spring the standing biomass of diatoms made up nearly 100% of total phytoplankton (Wu et al. 1996).

From 2001 to 2002, there were 68 genera and 112 species (and varieties) of phytoplankton, among which 87 species were in Hanjiang basin, 75 species in Danjiangkou basin, and 65 species in front of the dam. Hanjiang basin had more green algae than Danjiangkou basin, while the maximum of diatom species appeared in Danjiangkou basin. The annual average density of phytoplankton was 1.8×10^6 ind./L, with diatoms making up 36.17%, blue-green algae 26.09%, Pyrrophyta 20.95%, and green algae 8.57%. The annual mean biomass was 3.38 mg/L, Pyrrophyta accounting for 56.01%, blue-green algae 23.66%, and diatoms 15.34%. The standing biomass changed noticeably with seasons for

Table 13.8 Changes in phytoplankton species in the Danjiangkou Reservoir in different periods

	1958	1986–1987	1992–1993	2001–2002	2004–2006
Number of species		8 phyla, 92 genera	7 phyla , 60 genera	7 phyla, 68 genera, 112 species	8 phyla, 67 genera, 161 species
Composition	Diatoms – Pyrrophyta – green alga	Diatoms – green algae – blue-greens	Diatoms – blue greens – Pyrrophyta	Diatoms – blue greens – Pyrrophyta or Pyrrophyta – blue greens – diatoms (biomass)	Diatoms – blue greens – green algae
Dominant species	*Nitzschia*, *Cryptomonas*, *Pediastrum boryanum*, *Scenedesmus*	*Phormidium*, *Melosira*, *Pediastrum boryanum*	*Phormidium*, *Fragilaria*, *Cyclotella menighiniana*, *Aphanizomenon*	*Melosira*, *Oscillatoria*, *Microcystis*, *Pediastrum*,, *Cosmarium*, *Ceratium*	*Pediastrum*, *Melosira*, *Pyrrophyta*, *Oscillatoria*

Table 13.9 Changes in density and biomass of plankton

Parameter		1950s			1980s			1990s			2002		
		Hanjiang	Danjiang	Front of dam	Hanjiang	Danjiang	Front of dam	Hanjiang	Danjiang	Front of dam	Hanjiang	Danjiang	Front of dam
Algae	density/$\times 10^4$ ind./L	20.04	29.53	14.08	176.3	43.3	128.3	115.8	28.5	58.5	243.7	212.1	83.8
	Biomass/mg/L				2.784	0.193	1.047	2.133	0.358	0.893	6.655	2.232	1.25
Protozoans	density /$\times 10^4$ ind./L	7.639	3.329	12.4	5,168	5,804	4,694	7,350	1,211	7,425	9,450	2,100	7,275
	Biomass/mg/m^3	0.6	0.136	1.084	155	174	141	221	36	223	284	63	255
Rotifers	density/$\times 10^4$ ind./L	0.1745	0.099	0.279	3.2	2	2	487.8	41.6	425.5	603.8	97.5	277.5
	Biomass/mg/m^3	0.175	0.1	2.1	63	39	39	166	14	145	361	216	82
Cladocera	density /$\times 10^4$ ind.·L				3.5	2.1	3.5	14.1	8.2	12.8	1.8	5.8	4.2
	Biomass/mg/L				0.174	0.103	0.174	0.891	0.635	1.176	0.153	1.085	0.68
Copepods	density/ind./L	0.008	0.011	0.264	12.4	16.1	15.6	38.4	26.5	32.6	9.6	17.5	11.6
	Biomass/mg/m^3	4	0.33	64.7	292	378	366	245	288	218	62	202	124

Hanjiang basin in the following sequence: March > September > June > December, and for Danjiangkou basin: June > March > December.

From 2004 to 2006, 8 phyla, 67 genera, and 161 species of phytoplankton (including varieties) were observed (Li et al. 2007). These included 19 genera and 63 species of diatoms, 21 genera and 45 species of Chlorophyta, 14 genera and 33 species of Cyanophyta, 1 genus and 5 species of Chrysophyta, 4 genera and 4 species of Xanthophyta, and 2 genera and 2 species of Pyrroptata. There was a spatial difference in the number of phytoplankton species between the monitoring stations; the maximum appeared at Taocha station. There were more species in summer and less in spring and winter. With regard to species composition, in the upstream area (Dashiqiao), diatoms made up 52%, green algae 33%, and blue-green algae 14%. These communities are river-type phytoplankton. At the center of Danjiangkou basin, diatoms accounted for 41%, green algae 26%, and blue-green algae 23%. In front of the dam the blue-green algae, diatoms and green algae, accounted for 38%, 31%, and 26%, respectively. At Taocha monitoring station green algae, diatoms, and blue-greens made up 33%, 32%, and 24%, respectively.

Based on the biomass of dominant species, density (85%) and biomass (93%) of diatoms were highest before construction of Danjiangkou basin. After construction of the reservoir, diatoms still dominated but its dominance decreased. Although diatom dominance rose again to some extent between 1992 and 1993, it tended to decrease afterward, with its density decreasing to 39% from 2004 to 2006. In contrast, blue-green and green algae increased in proportion to 25% and 29.5%, respectively. As a result the water body developed toward eutrophication (Fig. 13.6).

The standing biomass of phytoplankton after storage of water by the reservoir was significantly higher than before. Compared with the 1980s, biomass dropped in the 1990s but increased greatly in 2000.

From 2004 to 2006, the number of phytoplankton indicator species in the reservoir totaled 21 genera and 24 species, with no heavy pollution-indicating

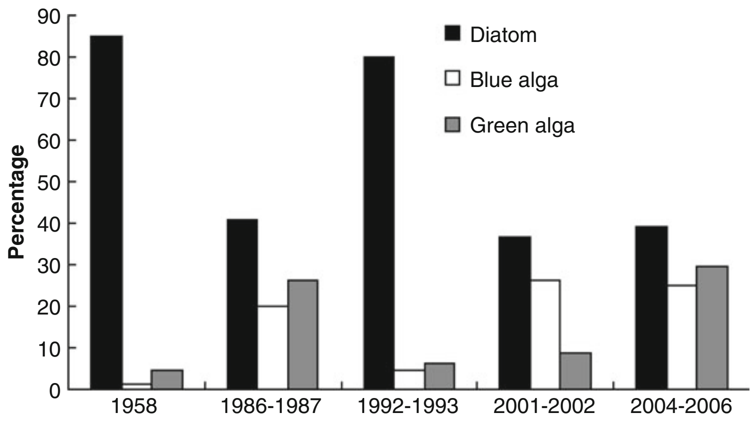

Fig. 13.6 Species composition of phytoplankton in different periods

species detected. Most of the indicators in front of the dam and at the center of Danjiangkou area were oligotrophic species. At Dashiqiao and Taocha, β-mesotrophic indicator species formed a majority. In particular, two indicator species of α–β–meso-eutrophic type and four indicator species of α-eutrophic type were found at Taoche monitoring station.

13.5.2 Zooplankton

In 1958, the Hanjiang and Danjiang Rivers contained similar species of zooplankton. The standing biomass of zooplankton was low. Sarcodina had little biomass, and there were many species of Rotifera with extremely low density as well as a few copepod species (Table 13.9). From 1986 to 1987, there were 45 genera of zooplankton in Danjiangkou Reservoir (Yang et al. 1996) with more species in summer and less in autumn and winter. Maximum density and biomass appeared in summer and minimum in winter (Table 13.10).

From 1992 to 1993, 124 species of zooplankton were collected in total. The annual mean density was 4,782 ind./L, and the annual mean biomass was 0.84 mg/L. The total density of zooplankton varied as: September > March > June > December. Regarding spatial distribution, a higher density of zooplankton appeared in Hanjiang basin, mainly due to Protozoa (5,175 ind./L), than in Danjiang basin (3,425 ind./L). In contrast, the biomass of zooplankton was higher in Danjiang basin mainly because Cladocera and Copepoda (0.9738 mg/L) in this area were more numerous than in Hanjiang basin (0.3366 mg/L) (Peng1995; Han et al. 1997).

From 2001 to 2002, 114 species of zooplankton were collected. Chydoridae now made up most of the species of Cladocera (four genera, nine species) and Cyclopoida contributed most to Copepoda species (five genera, eight species). The average density of zooplankton at Xiaochuan, Xianghua, and in the area in front of the dam was 6,275 ind./L for Protozoa, 326.3 ind./L for Rotifera, 4.1 ind./L for Cladocera, and 13.5 ind./L for Copepoda, corresponding to a biomass of 0.210, 0.220, 0.449, and 0.121 mg/L, respectively.

From 2004 to 2006, only 27 genera and 41 species of zooplankton were found (Li et al. 2007). Protozoans accounted for 49%, cladocerans 24%, rotifers 20%, and copepods 7%. The maximum species richness was at Taocha station, which accounted for 71% of the total observed in all monitoring stations. Three pollution-indicating taxa (*Brachionus*, *Canthocamptus,* and *Bosmina*) were detected in this water, but made up only 7% of the total species number.

Zooplankton species increased after storing water, and increased rapidly after the 1990s. Density of Rotifera and Copepoda increased first but then dropped, while the density of Protozoa and Cladocera increased at a constant rate. On the whole, the zooplankton shifted from a riverine to a lake type, and its species composition changed from a dominance of Sarcodina and rareness of Rotifera and Crustacea to a dominance of infusorians and an emergence of open-water Rotifera and planktonic crustaceans.

Table 13.10 Species composition of zooplankton in different periods

		1958	1986–1987	1992–1993	2001–2002	2004–2006
Number of species	Rotifera		15	51	50	20
	Protozoa		20	37	30	49
	Cladocera		11	17	18	24
	Copepoda		3	19	16	7
Dominant species		*Difflugia, Centropyxis, Phryganella acropodia, Arcella, Tintinnopsis, Vorticella; Asplanchna* sp., *Synchaeta, Monostyla Colurella* sp.	*Difflugia* spp., *Asplanchna* sp., *Bosminidae, Calanoida, Cyclops and nauplii*	*Difflugia, Centropyxis, Tintinnopsis, Strobilidium, Epistylis plicatilis, Polyarthra* sp., *Keratella, Collotheca* sp., *Bosmina* sp., *Mesocyclops* sp.	*Strobilidium gyrans, Tintinnopsis, Epistylis rotans, Cyclidium, Keratella* sp., *Polyarthra* sp., *Synchaeta, Daphnia hyalina, B. coregoni, D .brachyurum, Sinocalanus dorrii, Mesocyclops* sp.	*Amoeba radiosa, Keratella cochlearis, Canthocamptus* sp.

13.5.3 Benthos

Prior to construction of the reservoir, benthic organisms were generally rare in Hanjiang River, and not only simple in biological components but also low in biomass. Other than some small benthic organisms in suitable areas of the riverbed, there were no large benthic organisms in the river channel. *Chironomus* larvae were widely distributed. There were also a few species of Tubificidae and nematodes collected at a few monitoring stations. The density of benthic organisms in front of the dam (constructed later) was 400 ind./m^2 and biomass was 106 mg/m^2.

After construction of the reservoir in the 1980s, many mollusks, annelids, and aquatic insects appeared. The common mollusks included *Corbicula fluminea, Limnoperna lacustris*, *Lancelaria triforms, Bellamya aeraginosa, Anodonta woodiana, Hyriopsis curringii, Unio douglasiae,* and *Cristaria plicata;* annelids included *Branchiura, Tubifex, Limnodrilus, Helobdella, and Glossiphonia;* and aquatic insects included Ephemerid and Chironomidae larvae.

From 1992 to 1993 there were 35 species of benthic organisms, among which 5 species of oligochaetes, 8 species of aquatic insects, 18 species of mollusks, and 4 species of crustaceans. The common species included *Branchiura sowerbyi*, *Tubifex*, *Limnodrilus hoffmeisteri*, *Corbicula fluminea*, *Corbicula nitens*, *Bellamya purificata*, *Semisulcospira cancellata*, *Limnoperna lacustris*, *Orthocladius* sp., and *Chaoborus* sp. The dominant species included *Tubifex*, *Branchiura sowerbyi*, *Limnodrilus hoffmeisteri*, *Corbicula fluminea*, *Bellamya purificata*, *Semisulcospira cancellata*, *Limnoperna lacustris*, *Orthocladius* sp., and *Chaoborus* sp., most of which had no significant seasonal variation; only *Orthocladius* sp. and *Chaoborus* sp. were rare in summer. The average density of benthic organisms in Danjiangkou Reservoir was 185 ind./m^2, varying within the range of 12–371 ind./m^2. The average biomass was 162.6 g/m^2, varying within the range of 0.982–415.233 g/m^2. Standing biomass varied as September > June > March > December. Mollusks dominated in density and biomass. There was a significant spatial difference in the biomass, with the density and biomass of benthic organisms in Danjiangkou Reservoir 257 ind./m^2 and 274.4 g/m^2, respectively. Species number, density, and biomass on silt were higher than on sandy substrates.

From July 2007 to May 2008, 61 species of benthic organisms were collected in 5 phyla, 6 classes, and 12 families. Oligochaeta (*Tubifex and* Naididae) and Chironomidae (Zhang et al. 2010) included 24 species of annelids, 28 species of arthropods, 7 species of mollusks, 1 species of Nematomorpha, and 1 species of platyhelminth. Oligochaeta dominated throughout the year, accounting for above 90% of the total density of benthic organisms; the dominant species in biomass were mainly mollusks. The density distribution was in the sequence summer > spring > autumn > winter.

In summary: since the construction of the reservoir, benthic species increased quickly, related to the improvement of the bottom environment and increase in organic content in the reservoir. In the 1990s, mollusks dominated in density and biomass. However, from 2007 to 2008 the community structure shifted, with Oligochaeta and Chironomidae taking over the dominant position.

13.5.4 Fish

Among 43 fishes collected in 1957 and 1958, 30 species are fishes of running water like *Spinibarbus sineneis*, *Onychostoma laticeps*, *Tor brevifilis*, *Preiidobagrus macropterus,* and *Homalopteridae*, and migratory fish like *Anguilla japonica* and *Acipenser dabryanus*.

In 1987, 67 fish species were collected and 34 species more were found after the dam was built (Yuan and Huang 1989). They belonged to four orders, 12 families and 53 genera. There were 43 species of Cyprinidae, nine species of Bagridae, three species of Cobitidae and three species of Serranidae, two species of Siluridae and of Balitoridae, one species of Synbranchidae, one species of Sisoridae, one species of Eleotridae, one species of Gobiidae, one species of Opiocephalidae and one species of Mastacembelidae. Among cyprinids, Culterinae (11) and Gobioninae (10) were the majority, accounting for 25.8% and 23.3%, respectively. There were six species of Leuciscinae. After the dam was built, the species number of cyprinid increased. While some typical fishes that prefer supercritical flows (e.g., *Onychostoma laticeps*, *Tor brevifilis,* and *Spinibarbus sineneis*) migrated to the upper reach or disappeared altogether, some fishes that prefer tranquil flows (e.g., *Erythroculter*, *Triga,* and *Gobio*) appeared. Migratory fishes like *Anguila japonica, Acipenser dabryanus,* and *Coilia* disappeared.

An investigation of spawning sites in the upper reach of Hanjiang River in 1993 (Dai 1997) reported 27 fishes producing drifting roes, which accounted for 35–40% of natural fish landings, and there was a large proportion of fish producing adhesive roe, which accounted for 40–50% of natural fish yield. Species like carps, crucian carps, and catfishes became major fishing targets.

At present, there are 70 fish species in Danjiangkou Reservoir in 5 orders, 13 families and 54 genera. Forty species are commercial fishes, the most important of which include carps, crucian carps, silver carps, bighead carps, grass carps, *Culter alburnus*, *Culter mongolicus*, *Siniperca chuatsi*, *Elopichthys bambusa Richardson*, *Siniperca kneri*, *Pelteobagrus fulvidraco*, Snakehead fishes, *Pseudolaubuca sinensis,* and silver fishes. Most are Cyprinidae and they also occupy a large proportion in the fishery harvest.

Since dam construction, Cyprinids have increased, while fishes that prefer supercritical flows have migrated to the upper reach or have disappeared, along with migratory fishes. Due to human interference and large-scale stocking from late 1980s to the early 1990s, fish resources in the reservoir have become quite different from those in the 1980s and the proportion of commercial fishes like silver fishes and bighead fishes has increased. In the middle and late 1990s, stocking amount decreased but fishing intensity remained high. Alien species like silver fish invaded the reservoir, impacting on natural fish species which became smaller in size. Silver carp and bighead carp decreased and alien species increased. A fishing moratorium was implemented in 2003, and the structure of fish populations in the reservoir has improved to some extent; Xenocyprininae like *Xenocypris davidi* Bleeker and *Xenocypris microlepi*s Bleeker as well as black carps have partly recovered.

13.5.5 Spawning Sites

Since Danjiangkou Reservoir was built, the water flow through the reservoir all but ceased, so some spawning sites have disappeared or moved to other places, while new spawning sites became established.

In the investigation from 1986 to 1987, there were 17 commercial fish species producing floating roes, accounting for 43.59% of all commercial fishes. The spawning sites were mainly located at the upper reach of the Hanjiang River, from Shiquan County in Shanxi to Yun County in Hubei, which was 441.5-km long. The major spawning sites included Xuanwo, Donghe Town, Ankang, Shuhe Town, Jiahe Town, Baihe, Tianhekou, Qianfang, Xiaojia Village, and Yun County. The spawning sites in Qianfang and Tianhekou were the largest, producing 13.05 billion roes of four fishes (grass carps, chubs, black carps, and bighead fish) in 1977, accounting for 76.18% of total roe production of the four major Chinese carps.

A study in 1993 (Dai 1997) also showed that there were five spawning sites in Ankang, Shuhekou, Baihe, Qianfang, and Yun County. Qianfang was the largest one and accounted for 43.2% of the four major Chinese carps (grass carps, chubs, black carps, and bighead fish) and 84.3% of fish with floating roes. Compared with the situation in 1977, the spawning scale in 1993 became smaller and accounted for 44.46% of that in 1977, and the spawning scale of the four major Chinese carps was just 5.23% of that in 1977. The river flow for fish spawning was not obvious and fish spawning was restricted by runoff regulation at Ankang. After Ankang Reservoir was built, eight spawning sites moved to upper or lower reaches.

The normal water level of Danjiangkou Reservoir is 170 m, and the backwater area reaches Baihe County. Since the water level rose and flow slowed, conditions for spawning sites in Baihe, Qianfang, and Yun County changed partially or completely. Because of the joint effects of Danjiankou and Ankang reservoirs, spawning sites at the lower reach of Ankang and upper reach of Yun County were heavily damaged and the species and number of fishes decreased dramatically. As fish resource gradually became exhausted, fish production in Danjiangkou Reservoir collapsed.

In 1986 and 1987, there were 10 species producing adhesive roes in Danjiangkou Reservoir (8 Cyprinidae and 2 Siluridae), accounting for 25.64% of all commercial fishes. Aquatic plants are rare in Danjiangkou Reservoir, so the plants used as substrate for spawning of fishes are mainly terrestrial plants at the riparian zone. Therefore, the fluctuation zone where the tributaries flow in is the major spawning site of fishes producing adhesive roes (e.g., Madeng, Xianghua, Xiaochuan, Caodian, and Liupo).

The investigation in 1993 reported 14 large-scale spawning sites like Langhekou and Huangtubao. After the second phase of Danjiangkou Hydraulic Project was completed and began to store water, all spawning sites that were formed earlier were inundated and disappeared. However, after the second phase project was completed, the shoreline became longer and new spawning sites could be established. Fishes with adhesive roes now breed there successfully.

Since the water level was lifted, the annual variation of water level has been substantial, which affects the survival rate of roes and breeding of parent fishes. This phenomenon has happened at the first phase project of Danjiangkou Reservoir.

In brief, the second phase of the project has not greatly affected those fish producing adhesive roes. In the breeding season (from the middle of April to the middle of June), fishery profits can be ensured under reservoir operation because the water level is kept stable and the breeding sites are large enough.

13.6 Fish Production and Management

13.6.1 Fish Resources

The water surface of Danjiangkou Reservoir covers 74,700 hm^2 and the breeding area is 62,000 hm^2, of which 24,000 hm^2 is in Xichuan County (65.7% of total water area in county) and 34,700 hm^2 in Danjiangkou City (68.4% of total water area in city). After completion of the south-to-north water diversion project, Danjiangkou Reservoir will reach 100,000 hm^2. Before the dam was built, traditional fishing was common and fish landings were small. The annual fish yield of Hanjiang River in Jun County was only 10,000 kg and the fish yield of Danjiang River in Xichuan County was even lower. After the dam, fish resources increased rapidly because of natural breeding and a large-scale fishery industry. The reservoir fishing yield was 6,500,000 kg in 2002, and at present reaches 12,000,000 kg (Fig. 13.7).

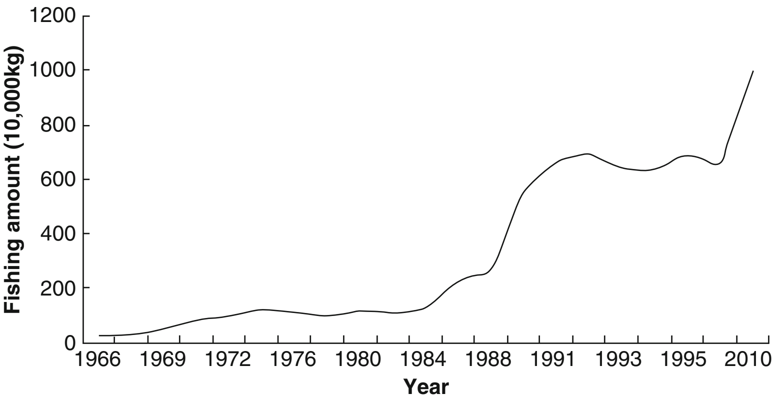

Fig. 13.7 Fishing yield landings of Danjiangkou Reservoir

Annual fishery output of Danjiangkou Reservoir is 4,000,000–5,000,000 kg and naturally propagated fishes account for 50–60%. During 1986–1994, the reservoir fishery production reached 17,120,000 kg in total. In 2009, the production of Danjiangkou City reached 76,000 t, among which natural fishery production was 12,000 t and aquaculture production was 64,000 t (Table 13.11).

Propagation of fish in the reservoir relies on species that can breed naturally, like fishes producing adhesive roes. The original spawning sites of the four major Chinese carps (grass carps, chubs, black carps, and bighead fish) were inundated, so artificial fish propagation sites will be built at the reservoir, so that the fish fries can be bred and released to the reservoir.

Table 13.11 Fishing yield of Danjiangkou Reservoir over the years 1968–2009 (unit: 10,000 kg)

	Danjiangkou city	Yun county	Yunxi county	Xichuan county	Danjiangkou Administration Bureau	Total
1968	18.48	12.50	1.50			32.84
1969	14.09	22.76	3.25	8.85		48.95
1970	45.33	10.51	6.50	24.00		86.43
1971	45.49	18.18	6.25	20.50		90.42
1972	54.52	15.00	6.50	21.91		97.93
1973	58.03	19.51	7.25	26.35		111.39
1974	58.08	20.45	9.50	32.34		120.37
1975	40.62	23.26	6.00	75.33		145.21
1976	34.98	27.75	6.00	47.06		115.79
1977	27.89	29.00	5.00	37.34		99.23
1978	42.04	26.65	6.00	34.04		108.73
1979	46.55	29.00	4.00	43.96		123.51
1980	61.52	35.35	4.93	47.09		148.89
1981	57.15	43.00	4.00	24.43		128.58
1982	63.43	51.60	4.50	34.98	0.50	155.01
1983	81.75	54.75	5.50	23.00	7.15	172.15
1984	125.87	57.50	7.85	47.20	4.40	242.82
1985	251.61	86.00	9.35	28.20	2.25	376.41
1986	116.00	80.00	11.80	47.12	1.00	266.60
1987	248.00	90.00	16.00	45.00	0.40	410.50
1988	250.00	100.00	16.50	50.00	2.00	482.50
1989	339.80	140.00	2.50	68.10	6.60	552.90
1990	Not counted	Not counted	2.50	Not counted	22.73	
1991	404.50	155.00	2.50	85.00	20.00	667.00
1992	417.50	160.00	4.00	95.00	17.50	694.00
1993	369.00	175.00	2.00	75.00	17.50	638.50
1994	375.00	150.00	2.50	90.00	16.00	633.50
1995	425.00	160.00	2.00	85.00	21.00	693.00
2002	–	–	–	–	–	650.00
2009	1,200	–	–	–	–	–

13.6.2 Fishery Production

Fish farming in Danjiangkou Reservoir is mainly cage and net culturing, focused around Jun County. Research in 1993 found 48 fish farming sites covering an area of 2,340 hm^2, seven major fish fry breeding sites which provide 200 million fries, 283 hm^2 of fishponds, 17 feedstuff processing factories with an annual capacity of 30,000 t, and 27.4 hm^2 of fish farming in cages, among which fish farming with non-feeding cages is 26.7 hm^2. The non-feeding cages of Hanjiang River, Xiaochuan section, are the largest number and the most densely arranged of China.

In 2008, the fish farming scale continued to expand: the cultured water area of Danjiangkou Reservoir was 22,600 hm^2, and it had increased by 2,000 hm^2 since the previous year. Aquaculture area was 23,000 hm^2 and artificial intensive culture area reached 8,000 hm^2, increased by 2,000 hm^2 since the last year. The aquaculture yield was as high as 60,050 t. Cage culturing of cultures and *Elopichthys bambusa* reached 30.67 hm^2, an increase by 2,200 cages of 7.33 hm^2. Meanwhile, 50,200 cages for culturing silver carps reached 167.3 hm, an increase by 2,400 cages of 8 hm^2 or a growth rate of 4.8%. The newly developed fish farming in the reservoir bays was 1,761.3 hm^2 and the new added tank farming was 26.67 hm^2. The output of aquatic products was estimated at 70,000 for 2008, and aquatic products industry accounted for 35% of the agricultural economy. Aquaculture yield of Spanish mackerel exceeded 15,000 t and became a unique feature of fishery industry at Danjiangkou Reservoir. In 2010, the authorized fish farming area was 62,000 hm^2, with 20,000 feeding cages of 72 hm^2 and 53,000 non-feeding cages of 320 hm^2. Feeding cages are mainly for cultures, *Elopichthys bambusa*, *Siniperca chuatsi*, large mouth black bass, and *Ictalurus punctatus*, while non-feeding cages are for silver carps, bighead carps, and paddlefish.

13.6.3 Crisis and Challenge

Of late, the fishery industry has been restricted by many factors, in spite of its development in recent years. Natural fishery resources are faced with crisis. First, predacious fishes like culters and snakeheads damaged the fishing industry. Silver-fish, which feeds on other fishes and their roe, has been introduced in recent years and is quite harmful. Therefore, it is necessary to control and restrict the introduction of new species. Also, illegal fishing is still common and the supervision and management become more difficult owing to advanced fishing techniques. The light attraction method has been industrialized, which caused a crisis around the reservoir.

Furthermore, there are not enough extension services to support the aquatic industry. Further processing of fish products is underdeveloped and the added value is small. Many fish farmers are short of cash, so fish farming is difficult to expand in scale and the development potential is limited. Besides, the technology for fish farming is not advanced, the quality of fish farmers low, and there are disputes about the technology required for farming high-quality fish and about major fish diseases.

Fish farming in cages produces pollution and may jeopardize the water quality of the reservoir. The ca 20,000 feeding cages in Danjiangkou City represent an estimated total nitrogen release of 2,000 and 450 t/a of phosphorus, equal to the pollution load of the sewage discharge of Shiyan City. Clearly, environmental monitoring and the protection and management of fishery resources should be strengthened.

13.7 Conclusions

With economic development and human demographic expansion, the ecosystem of reservoirs has experienced great changes, e.g., changes in air and water temperature, deposits of nutrients, mass propagation of plankton, restructuring of biological populations, deterioration of water quality, eutrophication, and overfishing. More attention has become focused on the water quality of Danjiangkou Reservoir in the last decade because of the middle route of the south-to-north water transfer project. It is important and fundamental to maintain a high water quality in Danjiangkou Reservoir. Besides such features as biological populations, biological diversity, and eutrophication, understanding reservoir limnology is vital to managing fish resources and ecosystem sustainability. Theories of river watershed ecology can be put to use to explore ecosystem functions and maintenance mechanisms at watershed level as a whole, and should be applied to practical management.

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Chapter 14
Limnological Characteristics of Liuxihe Reservoir

Liang Peng, Guoen Lin, Tian Wang, and Bo-Ping Han

Abstract Liuxihe, located at the Tropic of Cancer, is a typical large valley-type reservoir. Here, we summarize limnological features of this water-body based on observations that started in 2000. Two streams feed the reservoir. Mean water residence time is about 170 days but residence time in the wet season is only one-quarter of that in the dry season. Thermal stratification is monomictic; a short and full mixing occurs in winter in January, promoting nutrient regeneration from bottom to surface.

The mean concentrations of total nitrogen (TN) and total phosphorus (TP) are low, about 0.6 and 0.02 mg/L, respectively. Average chlorophyll *a* concentration is 1.93 mg/m^3, and transparency (SD) is 2.9 m. These four variables indicate that the reservoir is oligo-mesotrophic. However, a tendency toward eutrophication occurred across the 10 years of observation. Mass ratio of TN/TP and dissolved inorganic nitrogen (DIN)/dissolved inorganic phosphorus (DIP) are 30:1 and 78:1, respectively; such high ratios mean that growth of phytoplankton is surely limited by phosphorus. The high N/P ratio is attributed to tropical red soil in which a high iron content tightly binds phosphorus, the well-vegetated watershed, and low human population activity. Nutrients as well as other water quality variables show considerable temporal and spatial variation. In the early phase of the flooding season (April to May), nutrients and chlorophyll *a* concentrations are markedly higher than in the other periods and decrease from the riverine to the lacustrine zone. Spatial variability of nutrients shows a clear longitudinal gradient.

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14.1 Introduction

Liuxihe Reservoir is located to the northeast of Conghua City (23°45′N,113°46′E), 90 km from Guangzhou, the capital of Guangdong Province. It is a typical valley reservoir, situated almost exactly on the Tropic of Cancer (Fig. 14.1a). This reservoir was built in 1956 and its functions include power generation, flood control, and irrigation. The altitude at the dam top is 240 m. Liuxihe Reservoir has a maximum water depth of 73 m, a storage capacity of 0.325×10^9 m^3, and an area of 15 km^2. Inflow is contributed by two tributaries, the Lvitian and Yuxi Rivers, whose catchment areas are 264.4 and 192.3 km^2, respectively, covering 85% of the entire river basin area. Most of the basin is well conserved and situated in a national park with a forest coverage of over 80%.

14.2 Sampling Sites and Data Collection

Limnological variables were investigated at three sampling sites that follow the shape and water flow in Liuxihe Reservoir. The three sites, S1, S2, and S3, are located in the riverine zone, transition zone, and lacustrine zone, respectively (Fig. 14.1b). From 2000 to 2008, we collected data of water quality variables such as water temperature, water transparency (SD), chlorophyll *a*, and nutrients. Temporal and spatial dynamics of nutrients and chlorophyll *a* were measured monthly at the three sites in 2006 (see more information in Lin et al. 2009). Water quality is measured according to the standard methods suggested by the EPA of China.

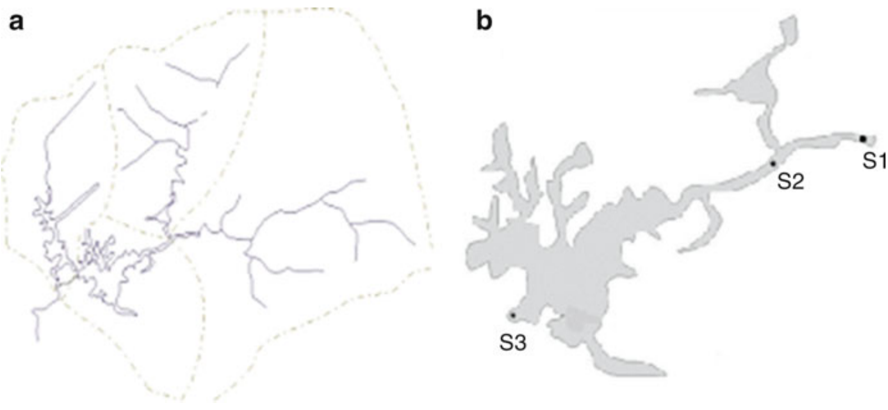

Fig. 14.1 Watershed (**a**) and sampling sites (**b**) of Liuxihe Reservoir

14.3 Results and Discussion

14.3.1 Water Temperature and Thermal Stratification

As a typical water body in the transition zone between the tropics and subtropics, variation of water temperature in Liuxihe Reservoir shows a clear seasonal variation. Temperature of surface water ranged from 14.3°C to 32.2°C in the lacustrine zone near the main dam, and June to September had an average of 30°C (Figs. 14.2 and 14.3). From December to the next March, surface water temperature was usually below 20°C, and the minimal temperature in January and/or February often falls below 15°C, but only during 2 or 3 weeks.

Seasonal variation of thermal stratification and water mixing are typically monomictic. Maximum depth is over 60 m, and temperature change most of the year affects only the top 0–30 m of the water column. Below this upper layer, temperature remains at 14–17°C. Water is fully mixed in January (a winter period), when water temperature is 14°C across the entire water column. A thermocline first occurs in March. In June, water temperature rises rapidly and the thermocline becomes stable, appearing at 20 m from the water surface. In September, a strong thermal stratification develops a difference of 10°C over the thermocline. Water temperature declines gradually after September and the thermocline disappears entirely in December. Usually, water temperature may decline below 4°C in winter in water bodies of the temperate zone, and thermal turnover in winter induces a strong mixing. Rimov reservoir is a temperate valley water body in the Czech Republic, with a depth of 43 m (Brzakova et al. 2003) and an annual fluctuation of

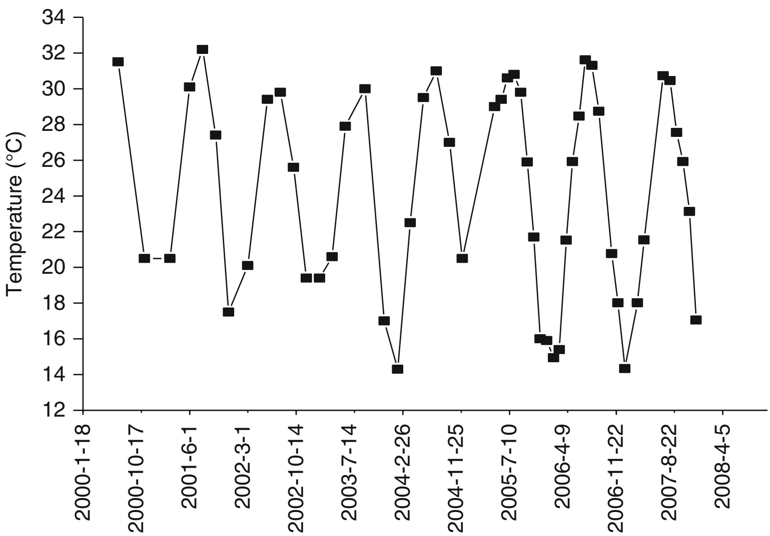

Fig. 14.2 The annual variation of surface water temperature during 2000–2007

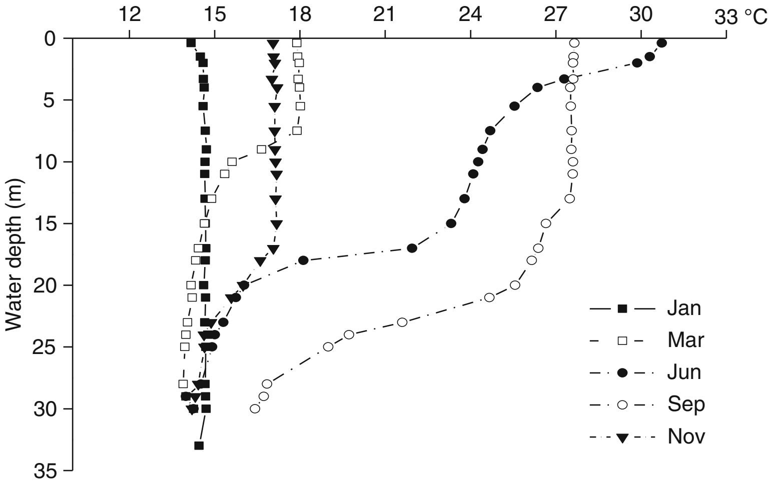

Fig. 14.3 Thermal stratification in the lacustrine zone near the dam in 2006

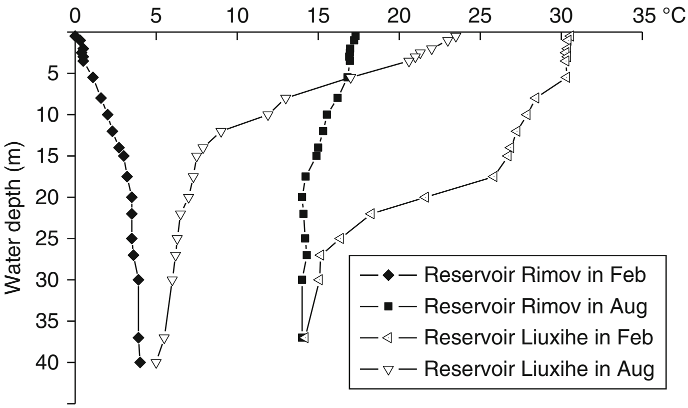

Fig. 14.4 Comparison of thermal stratification between Rimov reservoir and Liuxihe

water temperature from 0°C to 25°C (Fig. 14.4). In this reservoir, temperature at the surface in winter is lower than that at the bottom, which is about 4°C. Such an inverse stratification never happens in Liuxihe because of its warm climate. Thus, energy exchange is relatively weak near the bottom despite active energy exchange at the surface. Vertical mixing and turnover are important processes in nutrient exchanges in deep waters (Wang et al. 2005). Vertical exchanging of nutrients is relatively weak in tropical and subtropical climates. On the other hand, inflows and precipitation play a crucial role in such ecosystems.

14.3.2 Hydrographic Features

Precipitation affects the inflow of exogenous nutrients and suspended solids with surface runoff and carries nutrients that form an important component of the nutrient budget. Moreover, the precipitation directly regulates inflow, water level, and thermal stability. The average annual rainfall was 2,140 mm from 2000 to 2008 (Fig. 14.5), changing only slightly over time. Minimum and maximum rainfall were in 2003 and 2006, respectively.

Precipitation shows seasonal dynamics, affected by monsoon (Fig. 14.6). Monthly precipitation is high from April to September, with a maximum in June. The other months have low precipitation and are referred to as the dry season. During this period, total rainfall is below 400 mm and monthly rainfall below 100 mm.

In spite of the multiple functions of Liuxihe Reservoir, outflow is mainly controlled by power generation. Significant seasonal variation of hydraulic retention time is mainly associated with precipitation and outflow from the reservoir. Monthly hydraulic retention time in 2006 ranged from 43 to 380 days. In order to prepare flood control, actual storage and water level are usually regulated to a low level in March and April. During monsoon, outflow for power generation increases greatly with impounding water. Hydraulic retention time in the dry season is up to four times that during monsoon. Flood discharge and overflow often occur in heavy rainfall years. Furthermore, evaporation rate shows an average of 0.3 m^3/s and was observed to be less than the expenditure due to power generation.

In 2007 and 2008, average monthly inflow varied between 6.1 and 152 m^3/s. Average monthly inflow was 8.5 m^3/s during monsoon and 37.3 m^3/s in the dry season. The outflow has a seasonal dynamics similar to that of the inflow. Monthly outflow ranged from 8.1 to 109 m^3/s in 2007 and 2008 (Fig. 14.7).

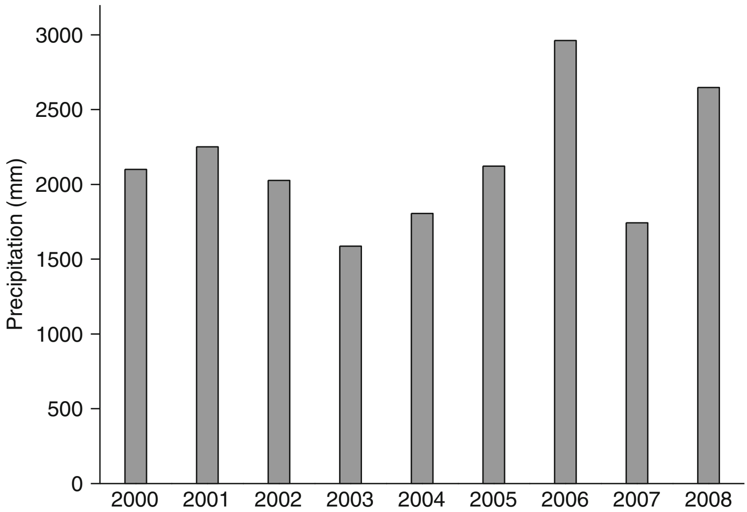

Fig. 14.5 Annual precipitation during 2000–2008

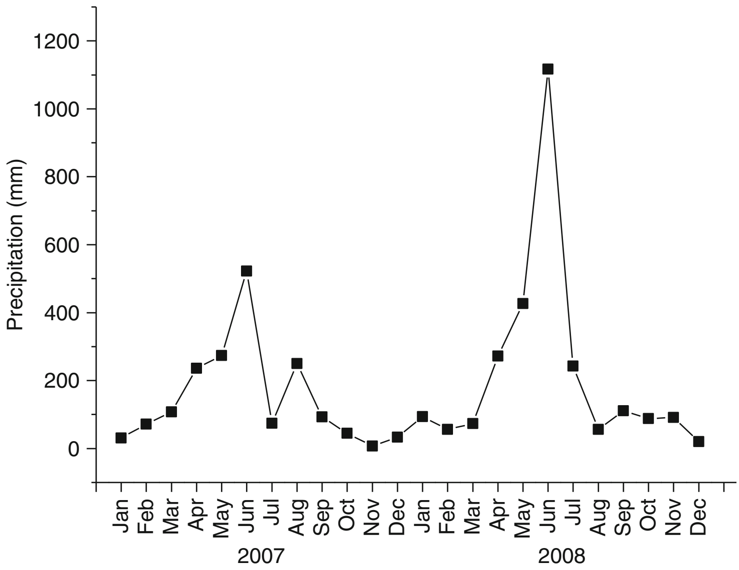

Fig. 14.6 Monthly precipitation during 2007–2008

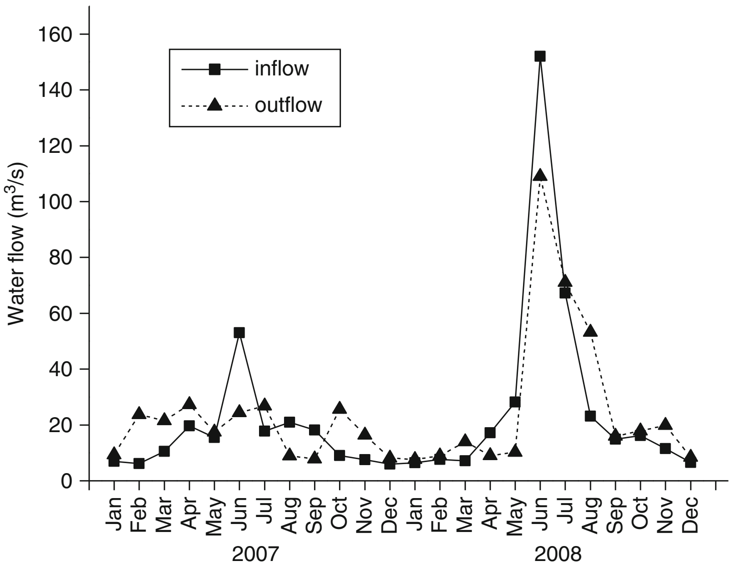

Fig. 14.7 Monthly inflow and outflow during 2007–2008

14.3.3 Water Level, Water Storage, and Hydraulic Retention Time

The water level in Liuxihe Reservoir is affected by inflow, outflow, and precipitation. In the year 2007 and 2008, water level ranged from 217.95 to 235.3 m, with minimum in May 2007 and maximum in July 2008 (Fig. 14.8). Usually, the lowest water level appears at the end of the dry season just before the coming monsoon, e.g., in May 2007 and April 2008. With heavy rainfall in June, water level may rise dramatically, with an increase up to 10 m. In June 2008, water level exceeded the top of the dam and the water had to be released by overflowing. In dry seasons, water level continually declines from September to the next storage period. The fluctuation of water level in Liuxihe Reservoir is more than 15 m per annum.

Hydraulic retention time is calculated with reservoir capacity divided by outflow. It increases when the reservoir rises and outflow is reduced. In 2007 and 2008, for example, hydraulic retention time varied from 32 to 346 days (Fig. 14.9). The minimum occurred in June 2008 and the maximum was in January 2007 and December 2008. The average hydraulic time was 154 and 168 days in 2007 and 2008, respectively.

14.3.4 Nutrient Dynamics

The area ratio of catchment/water surface of Liuxihe Reservoir was about 40:1, and plenty of nutrients and particle matters became charged to the reservoir through

Fig. 14.8 Variation of water level during 2007–2008

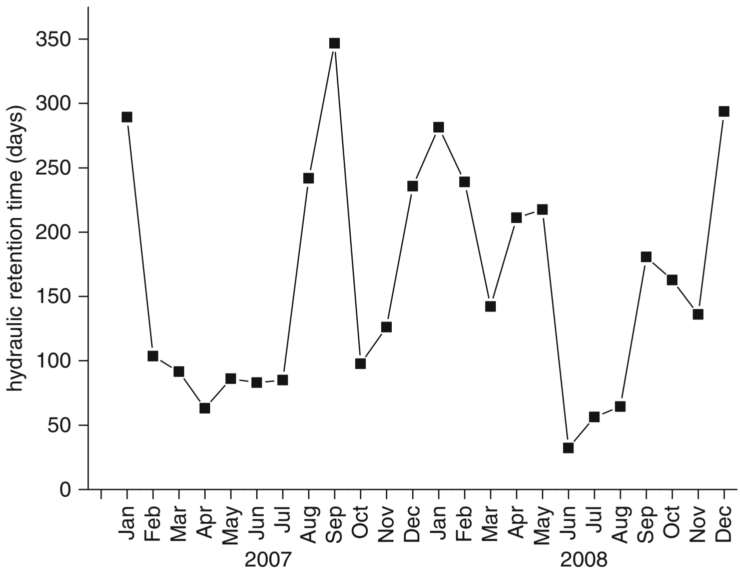

Fig. 14.9 Monthly hydraulic retention time during 2007–2008

rainfall erosion. Annual nutrients and particles in the inflow differ greatly because of uneven seasonal distribution of rainfall, and they have a huge influence on the trophic state of the reservoir. Outflow of power generation played an important role in the losses of nutrients.

The concentration of total nitrogen (TN) varied from 0.385 to 0.889 mg/L, with an average of 0.6 mg/L from 2000 to 2008 (Fig. 14.10). The concentration of dissolved inorganic nitrogen (DIN) that contains nitrate, nitrite, and ammonia, ranged 0.285–0.41 mg/L with an average of 0.34 mg/L. The ratio of DIN/TN was about 45–75%. Nitrate was the dominant component of DIN, and its average concentration was 0.299 mg/L in 2008, contributing about 95% of DIN. The average concentrations of ammonia and nitrite were 0.011 and 0.004 mg/L. Low concentration of ammonia and nitrite indicate little pollution in the reservoir. However, concentration of TN and DIN shows an increasing trend.

Concentration of total phosphorus (TP) ranged from 0.016 to 0.035 mg/L in the long term (Fig. 14.11), and had a maximum in 2001. The concentration of orthophosphate (O-P) ranged from 0.003 to 0.0065 mg/L, with an average of 0.005 mg/L. TP and O-P concentrations remained constant in the long term. The mass ratio of N/P was between 18 and 41, and therefore phytoplankton can be predicted to be phosphorus-limited.

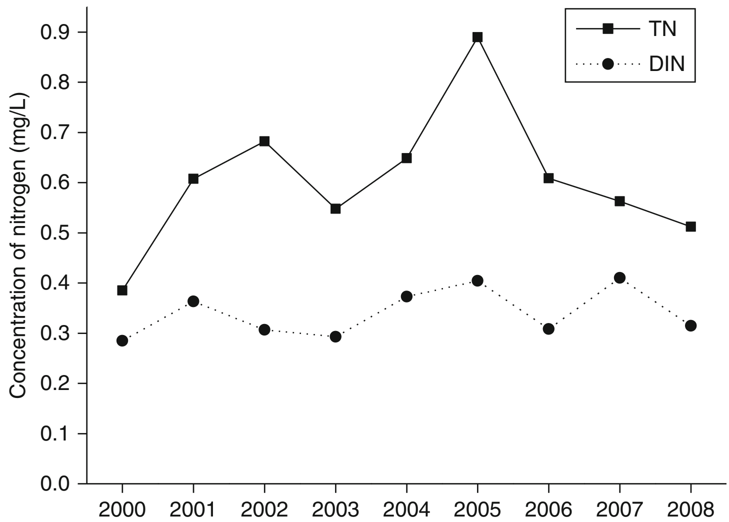

Fig. 14.10 Long-term dynamics of TN and DIN in the lacustrine zone

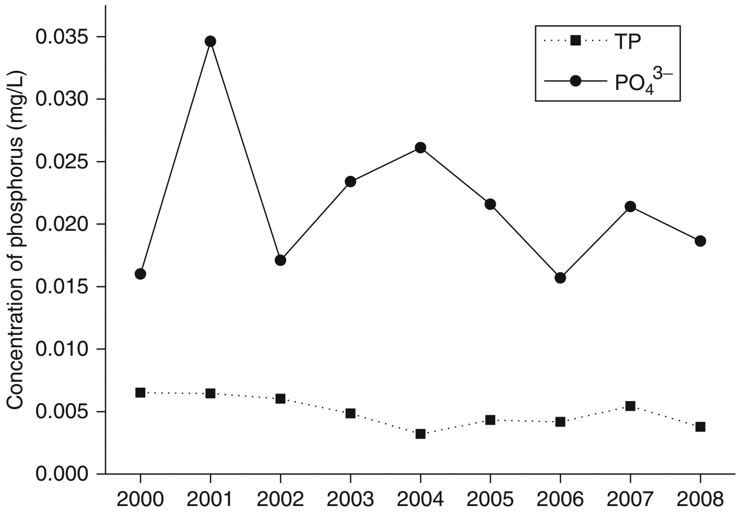

Fig. 14.11 Long-term dynamics of PO_4^{3}-P and TP in the lacustrine zone

14.3.5 Water Transparency Dynamics

Liuxihe Reservoir is an oligo-mesotrophic water body. Its transparency (described in Secchi Depth, SD) is mainly influenced by suspended solids and sediment flushed by rainfall. SD ranged from 1 to 4.7 m with an average of 2.9 m for the

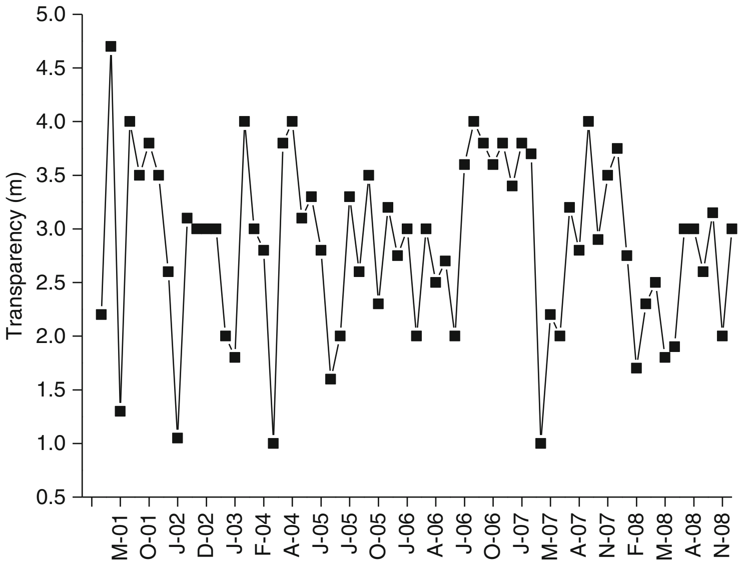

Fig. 14.12 Transparency in the lacustrine zone during 2000–2008

past 9 years (Fig. 14.12). In the years 2000–2004, SD exceeded 4 m only four times. Despite increasing investigations, SD exceeded 4 m on only two occasions between 2005 and 2008. Average SD was 3.5 m, with the maximum present in 2000 and the minimum in 2008. The declining tendency of SD reflects deteriorating water quality. SD varies seasonally, and it was low (<2.5 m) in the early stages of the flooding season from March to June, and the lowest was about 1 m mostly in April. SD was highest in July to September. SD was mainly regulated by silt, scoured by rainfall. Abundance of algae has little effect on SD because of the oligo-mesotrophic state of the reservoir.

14.3.6 Annual Variation of Chlorophyll a *Concentration*

From 2000 to 2008, chlorophyll *a* concentration had an increasing trend. It ranged from 0.5 to 9.6 mg/m^3 with an average of 2.1 mg/m^3 (Fig. 14.13). Average concentration of chlorophyll *a* was 1.42 mg/m^3 from the year 2000 to 2005, and 2.8 mg/m^3 from 2006 to 2008.

14.3.7 Seasonal and Spatial Distribution of Water Quality

Seasonal distribution of TP and O-P differed greatly between the riverine and lacustrine zones in 2006 (Figs. 14.14 and 14.15). TP concentration was extremely

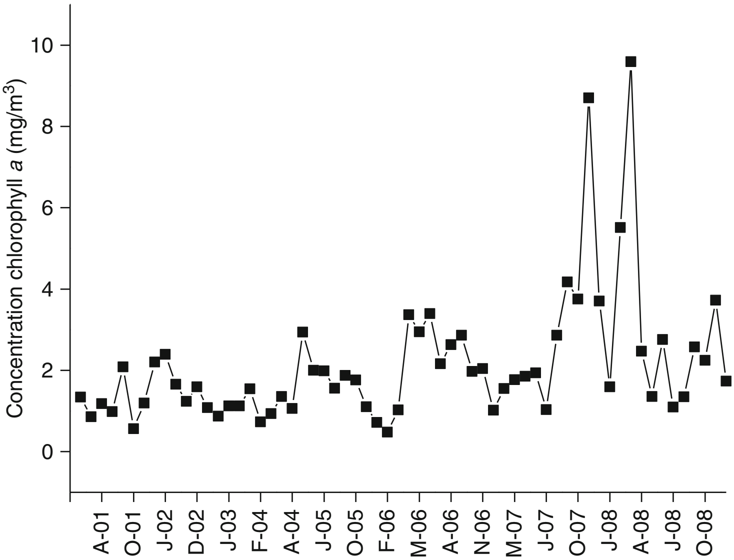

Fig. 14.13 Chlorophyll *a* concentration in the lacustrine zone during 2000–2008

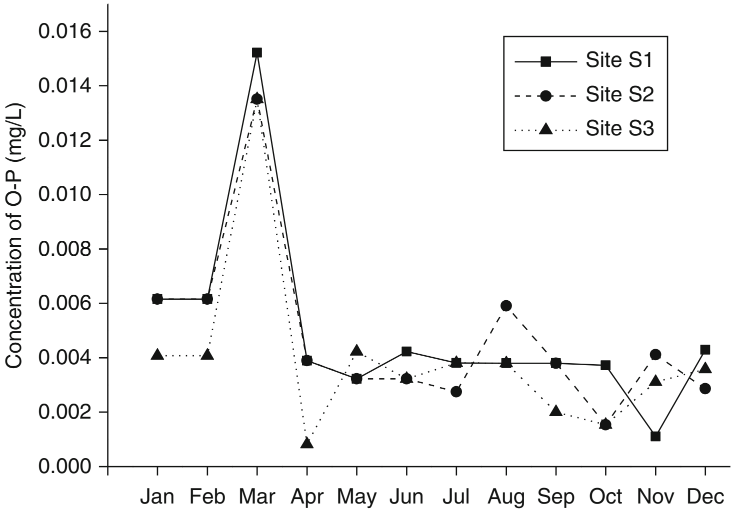

Fig. 14.14 Temporal and spatial distribution of O-P in 2006

high in March and April and even exceeded 0.14 mg/L at Site S2 (transition zone) (shown in Fig. 14.15), but it was low in the other months, varying between 0.01 and 0.05 mg/L. TP concentration was lower in the flooding than in the dry season.

Concentration of $PO4^{3-}$ was below 0.006 mg/L at all three sites (shown in Fig. 14.14), and it was somewhat higher in the dry season period than in the

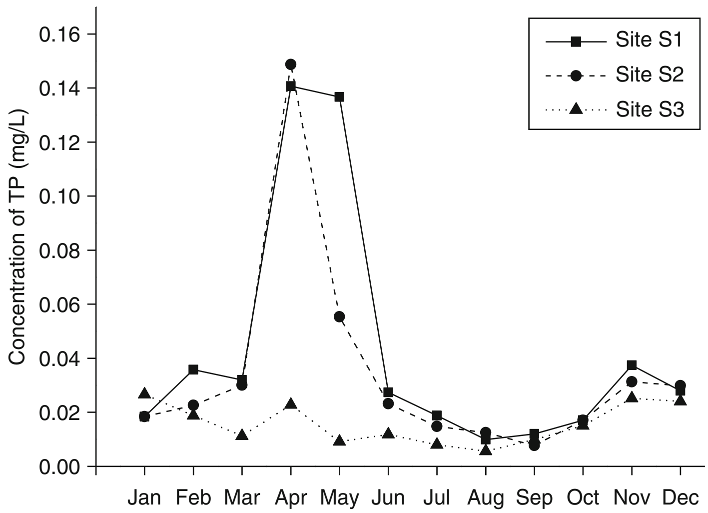

Fig. 14.15 Temporal and spatial distribution of TP in 2006

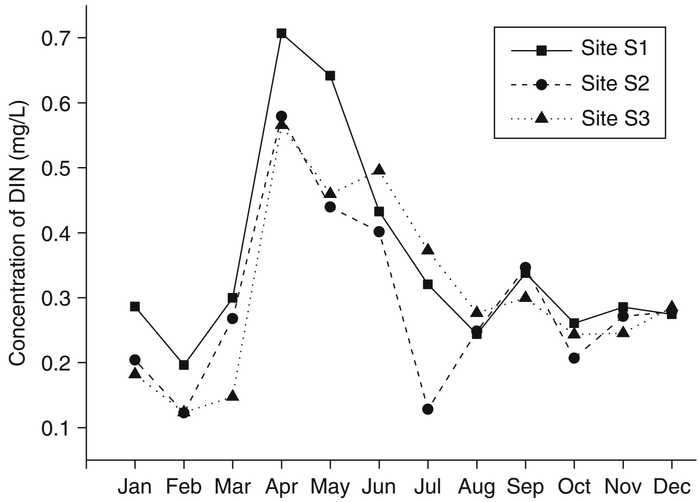

Fig. 14.16 Temporal and spatial distribution of DIN in 2006

flooding season. In other words, the highest concentration of phosphorus appeared from March to May between the end of dry season and early flooding season, while the lowest phosphorus concentration appeared from June to September during the late flooding season. On average, TP concentration was highest in the riverine zone. It showed a decreasing gradient from the inlet to the dam.

DIN concentration exceeded 0.4 mg/L from April to June at the three sites, and was below 0.3 mg/L in the other months (Fig. 14.16). DIN concentration increased in the initial period of the flooding season from April to June, and the highest concentration exceeded 0.55 mg/L at the three sites in April. The highest TN concentration was about 1.5 mg/L in January, and the lowest was about 0.2 mg/L in July (Fig. 14.17). The low concentration of TN and DIN appeared simultaneously in the middle and late flooding season.

In 2006, low water transparency (in Secchi Depth) appeared in February to June, with the lowest values 0.9 m in the riverine zone (S1), 0.8 m in the transition zone (S2) and 2 m in the lacustrine zone (S3) (Fig. 14.18). Water transparency increased

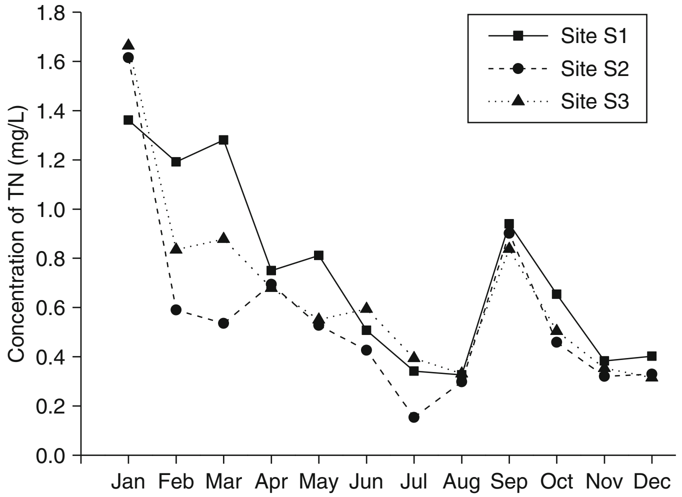

Fig. 14.17 Temporal and spatial distribution of TN in 2006

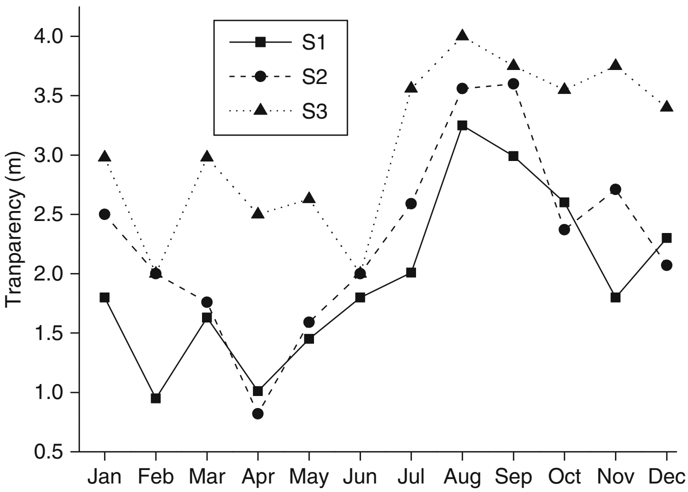

Fig. 14.18 Temporal and spatial distribution of SD in 2006

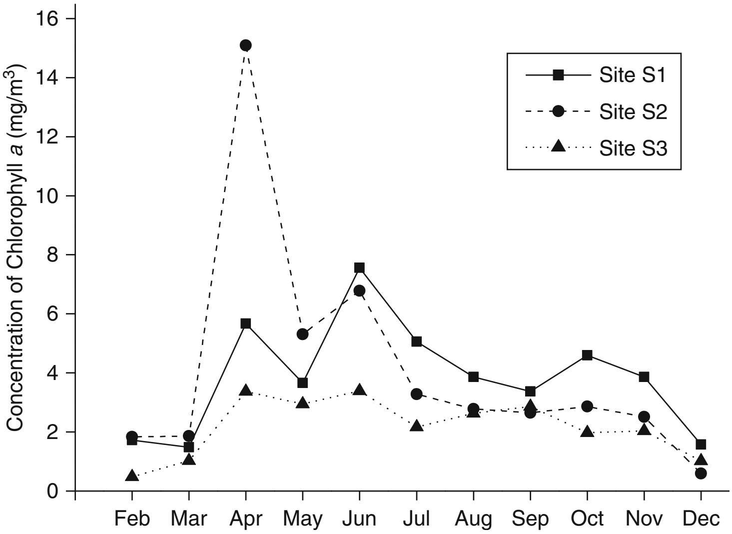

Fig. 14.19 Temporal and spatial distribution of chlorophyll *a* in 2006

clearly from the middle and late flooding season, and the maximum exceeded 3 m at the three sites in August.

Concentration of chlorophyll *a* was above 4 mg/m^3 from April to June, and the maximum was 15 mg/m^3 in the transition zone (S2) in April (Fig. 14.19). In general, chlorophyll *a* concentration was higher in the flooding season than in the dry season. It showed a reverse spatial distribution.

Temporal and spatial distribution of water quality in a reservoir is largely influenced by its morphology and hydrodynamics. As mentioned above, Liuxihe Reservoir is a typical valley water body, and it has a slope of 0.45% from the inflow to the dam. The distance from S1 to S3 was about 10 km and a spatial gradient in water quality was observed. Concentrations of nutrient and chlorophyll *a* showed a similarly spatial trend, and the water transparency (SD) showed a contrary trend (Lin et al. 2009). Because of longitudinal gradients, a reservoir is hydrodynamically divided into three parts: riverine zone, transition zone, and lacustrine zone (Thornton et al. 1981; Lin and Han 2001). Water flow in the riverine and transition zones is rapid, with an average velocity reaching up to 11.4 m/s at S1 (Lin et al. 2003). Suspended solids from soil erosion enriched the reservoir at the beginning of the flooding season, resulting in low SD and a shallow euphotic zone. Phytoplankton abundance was reduced greatly by weak illumination and there was a dramatic change in hydrodynamics (Lin et al. 2003). In the transition zone, water transparency and illumination increased substantially with the settling of suspended solids. Because of the high content of iron in the soil, settling suspended solid absorbs nutrients, especially phosphorus in water. Phytoplankton was limited by nutrients in the lacustrine zone instead of light intensity as observed in transition and riverine zones. This longitudinal gradient appeared to be particularly strong in the initial and late stages of the flooding season.

At S1, positive correlations between nutrients demonstrated that they were all simultaneously regulated by inflow. Chlorophyll *a* concentration correlated with DIN and TP, indicating that nutrients play an important role in controlling phytoplankton abundance. In the transition zone (S2), a strong negative correlation between SD and TP implied that SD was mainly influenced by suspended solids which contain most of the phosphorus. Chlorophyll *a* concentration showed a positive but weak correlation with water temperature. In the lacustrine zone (S3), nutrient concentrations remained a low level throughout the year and showed a slight seasonal variation.

14.4 Conclusions

Liuxihe Reservoir is a typical impoundment located in a transition region from a tropical to a subtropical region. Its limnological features have a clear seasonality driven by monsoonal precipitation and air temperature. Its elongate morphology provides a longitudinal gradient of hydrodynamics from the inflow to the dam. Spatial distribution of chlorophyll *a* corresponds to longitudinal gradients of nutrients, with a maximum in the transition zone. The longitudinal gradients in limnological variables are strongest in the initial and late stages of the flooding season. In the long term, TN, TP, and chlorophyll *a* concentrations show an increasing trend.

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Chapter 15
Status of Reservoir Fisheries in China and Their Effect on Environment

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Abstract China has abundant reservoir resources and reservoir fisheries play an unprecedented role in inland fisheries in the country. There are more than 86,000 reservoirs in China and the total storage capacity is over 4,130 billion cubic meters with a total surface area of over 2.3 million hectare. The distribution of reservoirs is quite uneven, Most reservoirs locate in the southern and middle regions of China. In the past 60 years or so, China has made tremendous progress in reservoir fisheries. The total yield of reservoir fisheries increased from 54,000 ton in 1949 to more than 2.68 million tons in 2009. The unit yield was less than 250 kg ha^{-1} in 1949 and increased to 1,555 kg ha^{-1} in 2009. Overall management and development strategies related to reservoir fisheries have been highly innovated in recent decades. First, multiple fisheries patterns were innovated in reservoirs for various types and purposes, including capture-based fisheries, extensive stocking, semi-intensive culture, intensive culture, polyculture, and integrated culture. Second, Chinese initiated the "joint fishing methods" and "barrier facilities for preventing fish escaping" to guarantee the high rate of recapture in reservoirs. Third, the increasing introduction and stocking of silver carp *Hypophthalmichthys molitrix* and bighead carp *Aristichthys nobilis* in most reservoirs in China is not only for enhancement of fish production, but more importantly, for the prevention or elimination of algal blooms as a biomanipulation management. Numerous reservoirs benefit enormously from filter-feeding fish stocking aimed at prevention of water quality deterioration. Other biomanipulation methods for water quality improvement are also applied in reservoirs, such as bivalve introductions and water plant planting. Fourth, because environmental effects of cage culture on reservoir

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environments are receiving more attention in the country, a dynamic carrying capacity model for cage culture in reservoirs has been developed to limit the scale and intensity of cage culture according to the requirement of water quality and function of reservoirs. Intensive fisheries activities in reservoirs definitely cause pollution and nutrient loadings to reservoir ecosystems. However, there is still ample scope for the development of more sustainable and environment-friendly strategies in reservoir fisheries in China.

15.1 Introduction

Numerous reservoirs have been built in China since the 1950s, primarily for flood control, power generation, irrigation, and navigation (FBMA 2010). The greatest expansion in secondary use of reservoirs for fisheries occurred almost at the same time (De Silva et al. 1991; Li and Xu 1995; Miao 2009). Reservoir fisheries is an essential component of inland fisheries in China (Liu and He 1992) and the total surface area of reservoirs today is more than 2.3 million hectare with a total capacity of over 4,130 billion cubic meters. Production of reservoir fisheries is gradually becoming a substantial contributor to total inland production, and is providing large quantities of affordable high quality animal protein for rural and poor populations (DAP 1992). The current Chinese fisheries is a comparatively integrated industrial system composed of aquaculture, capture fisheries, processing, machinery industries, fisheries science and technology, and fisheries administration. Certainly, the development of reservoir fisheries has created a large number of employment opportunities, which are a significant avenue for resettling displaced people and the poor in remote regions (Liu et al. 2009). For instance, in Fuqiaohe Reservoir (1,340 ha), Hubei Province, the fish yield was only 30 ton prior to stocking but increased to 300 ton after stocking, and more than 300 displaced people were employed as fish farmers or fisheries management officials (Liu and Huang 1998).

In the past 60 years or so, China has made tremendous progress in reservoir fisheries. During 1949–1957, reservoirs were hardly used for fish production and only simple capture activities were adopted to harvest wild fish. A relatively rapid development in reservoir fisheries occurred from 1958 to 1978. During this period, a series of techniques and strategies were established, including successful artificial propagation of China's four domestic carps (silver carp *Hypophthalmichthys molitrix*, bighead carp *Aristichthys nobilis*, grass carp *Ctenopharynodon idellus*, and black carp *Mylopharyngodon piceus*), increase in fish production in reservoirs, and improvement of capture techniques. However, the best period for reservoir fisheries development began at the end of 1978, when China started a policy of economic reform and opening up to the outside world. The greatest attention has been paid to the utilization of reservoir resources, and reservoir fisheries have made unprecedented progress in intensive culture (including cage culture), stocking and fisheries management, environmental and ecological studies, development of leisure fisheries and environment-friendly fisheries in reservoirs, fisheries policies, and administrative management (DAP 1992; Li and Xu 1995; Zhong and Power 1997).

It is clear that the demand for high quality fish and additional fish production will depend on a further expansion of reservoir fisheries in China (DAP 1992; Miao 2009). However, mismanagements and environmental changes are now threatening water quality of a large number of reservoirs, which calls for a change toward more responsible fisheries and better utilization of the reservoir resources (De Silva et al. 1991; De Silva 2001). Much attention should be paid to impacts of fisheries on water quality, role of fisheries in environment restoration, and the carrying capacity of intensive culture in reservoirs. In this article, we discuss the reservoir resources in China and review the important achievements of reservoir fisheries in recent decades. We also highlight the main impacts of fisheries on reservoir ecosystem.

15.2 Reservoir Resources in China

15.2.1 Number and Distribution of Reservoirs

There is an ancient tradition of reservoir construction in China. The earliest reservoir, Dong Qian Hu, was constructed about 1,000 years ago in Zhejiang Province (Shi 1996). However, construction of reservoirs accelerated rapidly in the 1950s and there are currently more than 86,000 reservoirs throughout the country. The total water surface area of reservoirs is 2.3 million hectare, which accounts for up to 13% of inland water resources (Bureau of Statistics 2007) (Fig. 15.1).

According to the characteristics of topography, morphology, and surface area, reservoirs are generally classified into four types: river valley reservoirs, plain lake reservoirs, hilly lake reservoirs, and hilly pond reservoirs (Li and Xu 1995). However, classification of reservoirs based on storage capacity is now more prevalent since it is useful and convenient for fisheries management. In this view, there are three types of reservoirs in China: large reservoirs (more than 100 million cubic meters), medium reservoirs (between 10 and 100 million cubic meters), and small reservoirs (type 1 between 1 and 10 million cubic meters and type II less than 1 million cubic meters) (Li and Xu 1995). There are about 2,600 large and medium reservoirs, amounting to 3% of the total number of reservoirs, yet their storage capacity is up to 35,000 million cubic meters and exceeds 80% of total storage capacity in China (Huang et al. 2001). Generally speaking, reservoirs are distributed in every administrative division with the exception of Shanghai. However, an uneven distribution exists in terms of number and storage capacity. Guangdong and Hubei Provinces have nearly 300 large and medium reservoirs. Hunan and Jiangxi Provinces also have more than 200, whereas there are only six in Tibet and nine in Qinghai Province (Ministry of Water Resource 2001) (Fig. 15.2).

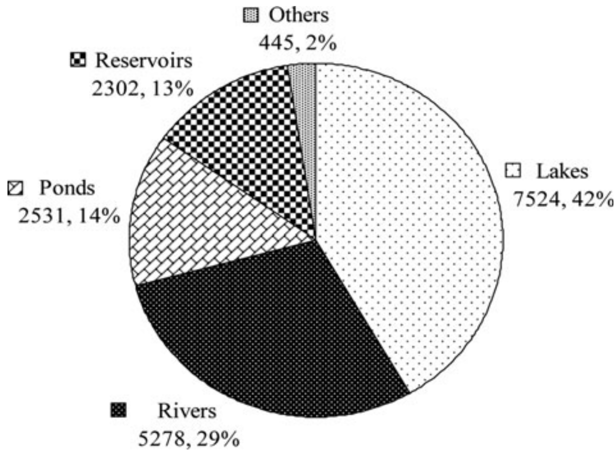

Fig. 15.1 Area (in thousand ha) and percentage of different types of inland water resources in China

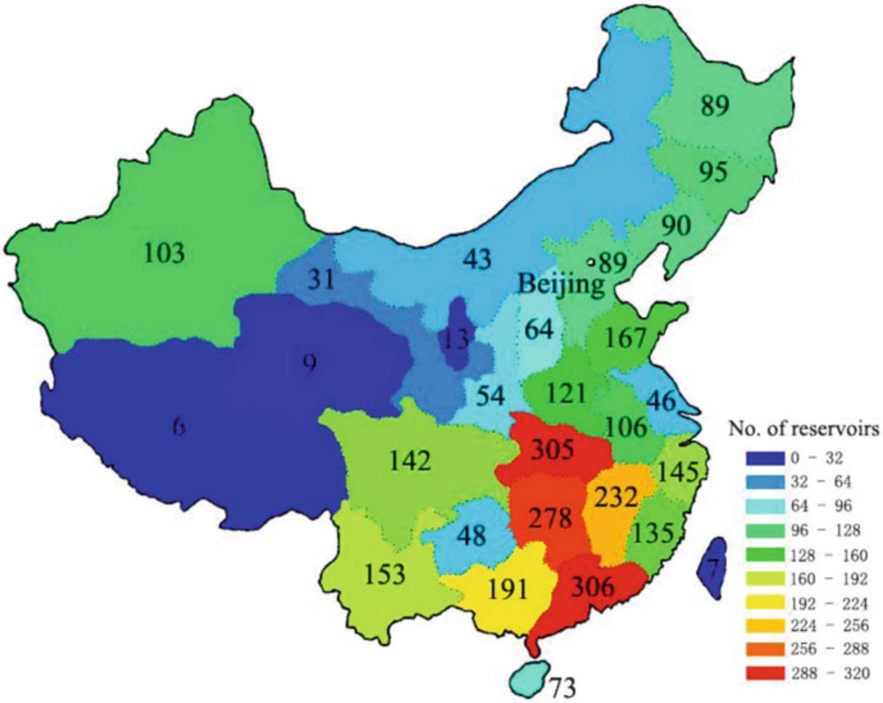

Fig. 15.2 Number of large and medium reservoirs in administrative divisions of China (modified from Ministry of Water Resources 2001)

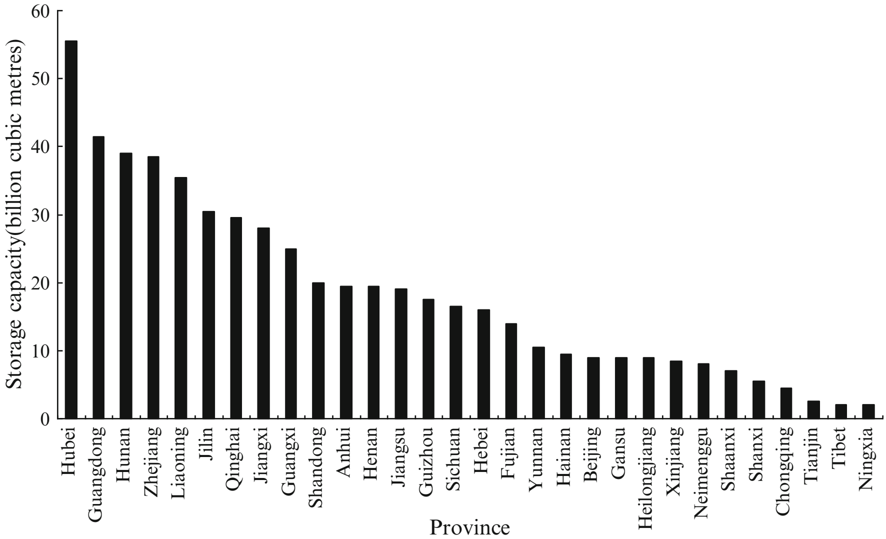

Fig. 15.3 Total storage capacity of reservoirs in each administrative division of China

Hubei Province has the highest total storage capacity (more than 50 billion cubic meters), followed by Guangdong Province (more than 40 billion cubic meters). Ningxia Autonomous Region has the lowest storage capacity (Fig. 15.3) (FBMA 2006). Moreover, most reservoirs are distributed in the main seven river systems in China: Yangtze River, Yellow River, Huaihe River, Hai-luan River, Pearl River, Songhuajiang River, and Liaohe River (Liu and He 1992). In the Yangtze River system, 36.7% are large and medium reservoirs, and 56.5% are small reservoirs. Storage capacity of reservoirs in the Yangtze River system is about 37.3% of total storage capacity, followed by 17.9% in the Yellow River system and 14.6% in the Pearl River system (Huang et al. 2001).

15.2.2 Ichthyofauna and Its Succession in Reservoirs

Fish fauna of a reservoir depends on its geographical location, exploitation, and protection of its resources. China covers more than 50° of latitude and 60° of longitude. Reservoirs are therefore located in areas of different climatic, geographic, and topographic conditions, which results in highly diverse and distinctive fish faunas (Li and Xu 1995). Most reservoirs in middle and eastern China are normally plain-typed and some are valley- or river-typed. Large and medium reservoirs in this region normally have 40 or more fish species and the biodiversity of fish is relatively high. For example, there were 68 species in Danjiangkou Reservoir in Hubei Province, a reservoir for water diversion from south to north in China (Yuan and Huang 1989). In Nanwan Reservoir of Henan

Province, 45 species were found and the dominant species belonged to Cyprinidae (He 2006). In terms of total fish yield, this is the most important area for reservoir fisheries in China. Southern China has higher temperature and abundant rainfall. Reservoirs in this area are rich in fish resources, particularly subtropical species. There are some endemic species, such as ratmouth barbell (*Ptychidio jordani*), large-scale silver carp (*H. harmandi sauvage*), mud carp (*Cirrhina molitorella*), and some species of introduced tropical tilapia (Liu and He 1992). The average species of reservoirs in southern China ranges from 30 to 40 (Huang et al. 2001). In northern China, reservoirs tend to be more suitable for cold-water fish and some species here are peculiar, such as minnows (*Phoxinus phoxinus*), sculpin (*Cottus gobio*), Atlantic salmon (*Salmo salar*), silver crussian carp (*Carassius gibelio*), pike (*Esocidae*), and roach (*Rutilus rutilus*) (Li 1981). Reservoirs in West China are fewer, including Yunnan, Qinghai, Tibet, Inner Mongolia, and Xinjiang Province. This zone is rich in plateaus with unique geographic, topographic, and ecological characteristics. There are therefore many indigenous species, including *Gymnocypris* and *Schizopygopsis* (Liu and He 1992). The fish fauna in reservoirs of this region is extremely simple, for instance, there are only 12 species in the Yudong Reservoir in Yunnan Province (Table 15.1). It is worth mentioning that many native species here are endangered because of exotic species invasion, habitat deterioration, and overexploitation.

The fish fauna in reservoirs is related to their original rivers and it is also highly disturbed by fisheries activities (Li 2001). After construction of reservoirs, original rivers disappear and water flow becomes almost lacustrine, especially in large and medium reservoirs, where the environment is similar to lakes (Li and Xu 1995). Fish fauna in such reservoirs experienced a significant change and succession after impoundment. For example, migratory species gradually decreased and even disappeared due to the obstruction of dams; examples are sturgeon (*Acipenser sinensis*), river shad (*Hilsa reevesii*), and eel (*Anguilla japonica*). Riverine species are forced to move toward upper tributaries or might disappear.

Table 15.1 Fish fauna in Yudong Reservoir, Yunnan Province

Family	Name
Cyprinidae	*Hypophthalmichthys molitrix*
	Aristichthys nobilis
	Carassius auratus
	Cyprinus carpio rubrofuscus
	Ctenopharyngodon idellus
	Pseudorasbora parva
Acheilognathinae	*Rhodeus rsiensis*
	Rh. ocellatus
Salangidae	*Neosalanx taihuensis*
Gobiidae	*Rhinogobius giurinus*
Eleotridae	*Hypseleotris swinhonis*
Siluridae	*Silurus meridionalis*

Economically valuable species increased rapidly because either they prefer the newly created habitats or are artificially introduced (Li and Xu 1995). Generally, in the filling period, there are abundant food resources, low fish density, large water volume, limited inter-specific and intra-specific competition, and few predators. Fish at this stage have high survival rate and growth rate. Therefore many reservoirs have high yields in the early years of impoundment. Subsequently, the submerged terrestrial plants decay entirely, while new populations of plants are not formed or are less developed owing to frequent and wide-range fluctuations of water levels. Most fishes cannot find suitable spawning areas and species without highly specific requirements for spawning usually increase rapidly. For instance, various sorts of low-valued and small-sized fish usually flourish because of their short propagation cycle and simple spawning prerequisites (Li 2001). Li and Xu (1995) summarized the succession of fish fauna in reservoirs after filling as follows: dominant indigenous riverine species → lacustrine species → omnivorous species → planktivorous, detritivorous, and carnivorous species.

15.3 Main Achievements of Reservoir Fisheries

15.3.1 Rapid Increase of Reservoir Fisheries

Reservoir fisheries developed rapidly since the 1950s with the expansion of reservoir construction in China, and the area used for reservoir fisheries showed an almost steady increase from 1978 to 2009 (Fig. 15.4). At present, more than 2.3×10^6 ha and about 83% of reservoir area is used for reservoir fisheries in China (FBMA 2010). The total yield from reservoir fisheries was only 54,000 ton in 1949, but increased to 112,000 ton in 1978 and more than 2.68 million ton in 2009 (FBMA 2010) (Fig. 15.5). The unit yield was less than 250 kg ha^{-1} in 1949 and increased to nearly 1,555 kg ha^{-1} in 2009. Reservoirs located in east and south of China generally show higher unit yields because of the high temperature and high primary productivity. For instance, Huayuan reservoir (area of 246 ha and mean water depth of 20 m) harvested up to 2,032 kg fish per hectare in 2004 (Gao and Xu 2006).

In China, the growth rate of the proportion of fisheries in total agricultural output in value is the fastest of all agricultural sectors. It was only 0.2% in 1949 and 1.5% in 1979. It reached 3.4% in 1989 and 8.5% in 1995 (DAP 1995). Although there are no accurate data on the percentage of reservoir fisheries in total agricultural output, there is little doubt that the rapidly increasing reservoir fisheries contributed substantially to fisheries and agricultural development in China. Similarly, on the perspective of employment, 11,428,655 million laborers were engaged in fisheries, of whom 5,071,940 full-time (capture fisheries, 1,672,822; aquaculture, 2,869,493

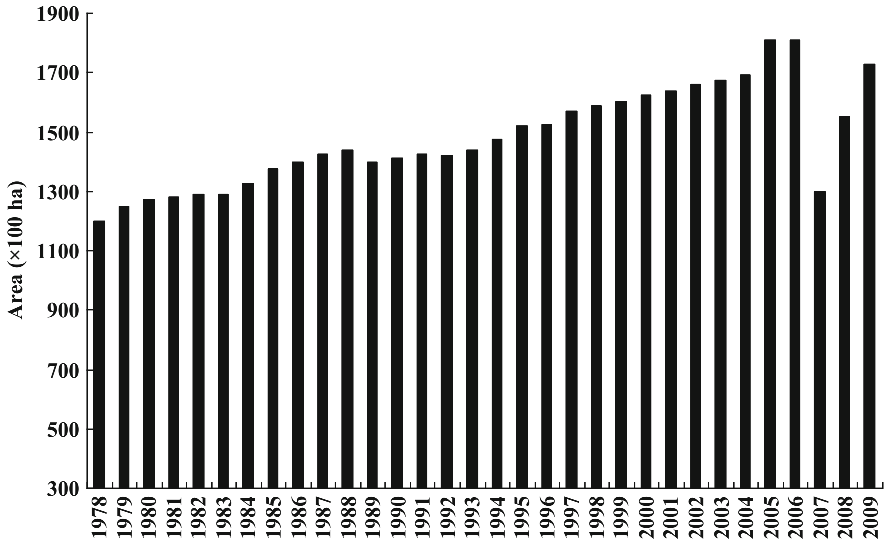

Fig. 15.4 Area (100 ha) of reservoirs utilized for fisheries in China from1978 to 2009

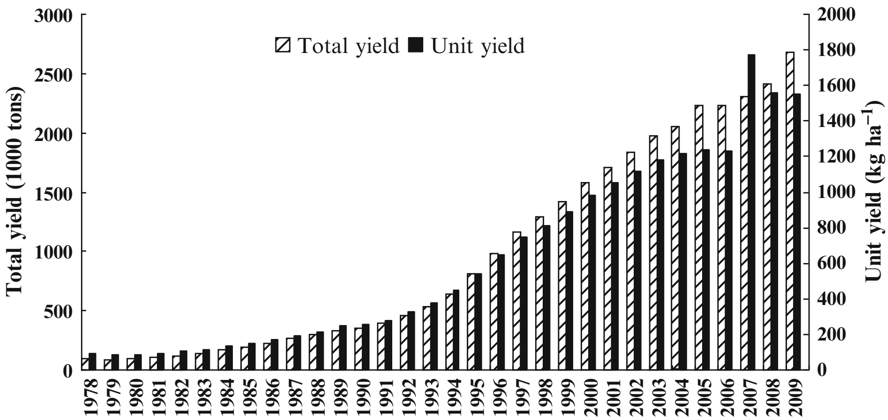

Fig. 15.5 Total yield (1,000 t) and unit yield (kg ha^{-1}) of reservoirs in China from 1978 to 2009

and service logistics, 529,625), and 6,356,715 part-time. Moreover, state-owned fisheries-related enterprises employed more than 300,000 people in 1995 (DAP 1995). Reservoir fisheries are therefore an important source of livelihood in China, especially for rural communities and displaced populations. The proportion of reservoir areas suitable for fisheries is up to 13.1% of the total freshwater area in China. However, fisheries yield of reservoirs only amounts to 11.7% of the total freshwater yield (FBMA 2010) (Fig. 15.6). Therefore there is great potential for further development of reservoir fisheries in China (Liu et al. 2009).

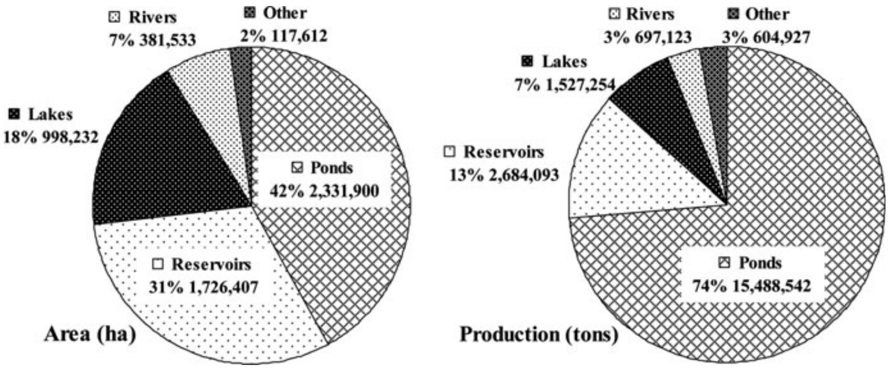

Fig. 15.6 Cultivable surface area (ha), fish production (ton), and their percentage in different types of inland water bodies in 2009

15.3.2 Extensive Stocking and Aquaculture Strategies of Reservoir Fisheries

Reservoir fisheries is in its infancy compared with traditional fish culture (Li 1992). However, it develops in its own features and there are large number of innovations in China. Several patterns of fisheries have been well developed in reservoirs, such as capture-based fisheries, extensive stocking, semi-intensive culture, intensive culture (pen and cage culture), polyculture, and integrated culture (Li 1992; Li and Xu 1995). Extensive stocking, semi-intensive and intensive culture are the most popular types of reservoir fisheries.

15.3.2.1 Extensive Stocking

Stocking in most reservoirs aims to sustain or enhance fisheries productivity, and some stockings aims to retain or replenish stocks of a species that are threatened or endangered. For extensive stocking in reservoirs, the primary approach is to stock fingerlings for fish production, with enhancement of natural fish being secondary. There is no artificial manure or feeding. Fish production depends on natural food organisms, including phytoplankton, zooplankton, detritus and bacteria, aquatic macrophytes, periphyton, and benthos (Li 1992; Li and Xu 1995). This approach is the most important activity of reservoir fisheries in China, particularly in large and medium reservoirs where water resources are also used for drinking. It has been accepted nationwide as an effective measure to increase yield from natural waters (Huang et al. 2001). Silver carp and bighead carp are the most popular species stocked in this fisheries pattern because they show rapid growth rate, better food conversion efficiency, easy capture, strong disease resistance, and higher availability of fry and fingerlings (Liu et al. 2009).

Furthermore, silver carp and bighead carp prefer to live on phytoplankton and zooplankton, respectively. Both also feed on a certain amount of detritus and bacteria. It has been reported that their predatory pressure on phytoplankton may contribute to the prevention of eutrophication (Lazareva et al. 1977; Spataru and Gophen 1985; Xie and Liu 2001). In most reservoirs, the ratio of bighead to silver carp ranges from 2:1 to 3:1. Sometimes the ratio of bighead carp may be half of silver carp in reservoirs with higher nutrient concentrations. The yield of the two species usually amounts to 70–80% of the total production in this pattern (Liu et al. 2009). In China, there is also another important stocking pattern, in which the dominant species are common carp (*Cyprinus carpio*) and crucian carp (*Carassius auratus*). This pattern is mainly practiced in Northern China because common carp and crucian carp usually show better growth performance than silver carp and bighead carp in cold water (Huang et al. 2001). Additionally, there are many other high economic species stocked in China's reservoirs, such as grass carp, mud carp (*Cirrhina molitorella*), tilapia (*Oreochromis niloticus*), mandarin fish (*Siniperca chuatsi*), and catfish (*Parasilurus asotus*). Mud carp and tilapia are mainly stocked in reservoirs of South China. In shallow reservoirs of the East and Middle regions of China, mandarin fish is stocked to control trash fish (Li and Xu 1995). It is worth noting that the introduction of new species to reservoirs is a low-cost and efficient method of stock enhancement in China. In fact, the original fish fauna in most reservoirs is not well developed to make full use of food sources and ecological niches. Moreover, there are usually numerous small-sized and low economic species which consume quantities of resources in reservoirs. Therefore, from the point of view of fisheries enhancement, introduction of species with higher quality and better growth rate is to fully exploit food resources and increase the fish productivity of the reservoirs. In the 1950s, silver carp, bighead carp, and grass carp were first introduced into many reservoirs using fry captured from natural rivers. In recent decades, introduction of fish species is highly developed and the main species are pond smelt (*Hypomesus olidus*), iwe icefish (*Neosalanx taihuensis*), and large icefish (*Protosalanx hylocranius*) (Huang et al. 2001; Liu et al. 2009). Overall, the theories and practices related to extensive stocking in reservoirs have become well developed, such as primary productivity evaluation, assessment of fish production potential, determination of stocking density, selection of stocked species, determination of proportions of different species, trash fish control and utilization by carnivorous predators, sophisticated capture techniques, etc.

15.3.2.2 Semi-intensive and Intensive Culture

Since the 1970s, fish culture has gradually evolved to a semi-intensive and intensive stage in small and medium reservoirs, as well as in coves and tail-water bodies (Li and Xu 1995). In semi-intensive culture, low quantity and quality fertilizers or feeds are used to supplement natural food resources or feed fish directly. The main inorganic fertilizers used in China's reservoirs are calcium superphosphate, ammonium bicarbonate, ammonium phosphate, ammonium hydroxide, and urea. Another kind of fertilizer often used is organic materials, including weeds, crop by-products (oil-seed cake, bran coat, and distiller's grains), and wastes from poultry and livestock. Organic fertilizers do not only support the rapid growth of phytoplankton, zooplankton, and benthos, but also provide food directly for filter-feeding and omnivorous fish species such as sliver carp, bighead carp, common carp, and crucian carp. Fish yield can therefore be increased significantly, and can sometimes be doubled or tripled when fertilizer practices are implemented (Li 1992). The fish stocked in Meichun reservoir (167 ha), for example, was stunted and fish could not reach the marketable size (>500 g) at harvest in earlier years. However, fertilization was adopted in 1986 and the unit yield increased from 300 to 750 kg ha^{-1} $year^{-1}$ (Pan and Cheng 1989). Compared with semi-intensive culture, however, intensive culture has further enhanced per unit output and increased benefits (Zhong and Power 1997). Cage and pen culture have been accepted as intensive culture methods in many reservoirs. Pen culture is practiced mainly in shallow reservoirs or tail-water bodies of reservoirs, while cage culture is usually carried out in deep water regions to take advantage of large water body and high water quality (Li 1992; Miao 2009). There are several advantages of pen and cage culture. For instance, it is easier to control competitors and predators, convenient to prevent and treat diseases, and it is simple and low cost to harvest fish. Hence, pen and cage culture has been developed quickly and the total cultivated area was 3,800,000 m^2 for cage and 567,700 ha for pen in 1987, respectively (Li 1990). The cage area in inland waters increased to 131,415,103 m^2 in 2009 (FBMA 2010). The intensive culture in reservoirs engage in high quality feeds, high stocking density, and sophisticated managements, such as disease prevention, aeration, and proper feeding regimes (Zhong and Power 1997). Certainly, intensive culture is costly and has risks, and it is also highly productive and generates high profits.

15.3.3 Improvement of Barrier Facilities and Capture Techniques

Fishing techniques in reservoirs are rather different from those used in other inland waters (ponds and lakes) because of the deep water, uneven bottom and complicated tributaries and coves. Capture techniques do play an important role in economic and social benefits of reservoir fisheries. Therefore, practices were developed to guarantee a high rate of recapture in reservoirs. These involve "joint fishing methods" and "barrier facilities for preventing fish from escaping." "Joint fishing methods" is a large-scale operation covering the whole reservoir and can be used in areas from hundreds to thousands of hectares (Liu et al. 2009). The main fishing gear used in this method are trammel nets, set-impounding nets, and fixed filter nets (Xu 1988). "Joint fishing methods" mainly captures silver carp and bighead carp, and also some grass carp, bream, and other pelagic species. Usually, some demersal species, such as common carp and crucian carp, can also be harvested. This effective fishing method was established in the 1960s and has been accepted nationwide in Chinese reservoirs. For instance, in 1980, it harvested more than 270 ton of fish by a single haul in Fuqiaohe Reservoir, Hubei Province (Liu and Huang 1998). Efficient capture techniques are not enough for high rate of recapture because some fish may escape from reservoir spillways and tributaries, which is common and sometimes causes great losses (Liu et al. 2009). Barrier facilities are efficient and convenient to prevent fish from escaping (Xu 1992). For instance, nearly 100 million fingerlings were stocked from 1960 to 1982 in Donzhang Reservoir (1,000 ha) in Fujian Province, whereas the unit yield in those years was only about 7.5–15 kg ha^{-1} year^{-1}. Barrier nets were installed in 1984 and fish yield thereafter exceeded 150 kg ha^{-1} in 1985 (Xu 1985). There are two basic barrier facilities commonly used in Chinese reservoirs: fence barriers and barrier nets. Since the 1980s, electric fish screens have been used for spillways in small hilly reservoirs, where the water flows are slow (Li and Xu 1995). In some small and hilly reservoirs, the graded culvert inlets can simply block fish with baskets. Barrier facilities are usually set up near the spillway, culvert, or upstream area of the reservoir (Xu 1992).

15.4 Environmental Impacts of Reservoir Fisheries

Reservoirs are newly created artificial inland water bodies and always multi-functional, have unique basin shapes, long shorelines, short water retention times, and irregular water level fluctuations (Craig and Bodaly 1989). Therefore, reservoirs are usually quite sensitive to anthropogenic stress (Silva and Schiemer 2001). The impacts of human activities on reservoir ecosystems are currently a worldwide issue since these ecosystems are globally subjected to eutrophication, pollution, or collapse (Schiemer et al. 2001). Fisheries, the main second-line use of reservoirs, is one important factor affecting the environment of reservoirs. Generally, fisheries activities have impacts on flora, fauna, and water quality.

15.4.1 Control of Cyanobacterial Blooms by Filter-Feeding Fish Species

China pays much attention to the prevention or control of algal-blooms in reservoirs since most reservoirs are not only for fisheries but, importantly, they are a drinking water resource as well. In China, silver carp and bighead carp are the most popular biomanipulation species for preventing or eliminating cyanobacterial blooms from reservoirs (Xie and Liu 2001; Zhang et al. 2008). Silver carp, a typical filter-feeding planktivore, mainly feeds on phytoplankton, zooplankton, and sometimes feeds on suspended small particles (Spataru and Gophen 1985). Bighead carp is considered to be predominantly zooplanktivorous. However, when concentration of zooplankton is low, it is opportunistic and may live on phytoplankton and detritus (Lazareva et al. 1977). Xie and Liu (2001) concluded that both silver and bighead carp can suppress or eliminate Cyanobacteria by direct grazing and through complicated fed web responses. They suggested a critical biomass of carp of approximately 50 g m^{-3} in Lake Donghu, a subtropical lake with the surface area of 32 km^2 near the Yangtze River in Wuhan, Hubei Province. Both silver carp and bighead carp have been successfully used to counteract cyanobacterial blooms in many other Chinese lakes and reservoirs, such as Lake Chaohu in Anhui Province, Lake Dianchi in Yunnan Province, the Three Gorge Reservoir, and Dangjiangkou Reservoir in Hubei Province (Xie and Liu 2001; Zhang et al. 2008).

Some freshwater bivalves such as *Hyriopsis cumingii* are also used to control phytoplankton and suspended detritus in reservoirs. Peng et al. (2009) used three batches of *H. cumingii* numbering 142,000 individuals aged 0^+ and 1^+ to control water pollution in Xikeng Reservoir in Guangdong Province. It was estimated that total consumption of a bivalve aged 3^+ for organic particles was 0.35–5.56 mg h^{-1} (Zhu et al. 2006). The bivalves removed 1,300 kg nitrogen and 137 kg of phosphorus in the first year and 8.786 kg of nitrogen and 930 kg of phosphorus in the third year. A water pollution control model using this species was built by Peng et al. (2009). The bivalve is also used for pearl culturing and is considered a tasty shellfish in Guangdong Province. Its widespread use in natural waters could bring about good environmental as well as economic results.

15.4.2 Nitrogen and Phosphorus Loadings of Semi-intensive and Intensive Culture

Eutrophication of inland water bodies is usually recognized to be a consequence of nutrient (nitrogen and phosphorus) accumulation that has spread all over the world in current decades. Surveys show that 54% of lakes in Asia are eutrophic (ILEC/Lake Biwa Research Institute 1988–1993). Improper fertilization and heavy feeding in semi-intensive and intensive culture systems in reservoirs likely result in increased nutrients, deterioration of water quality, and finally eutrophication (Folke and Kautsky 1989). Generally, nitrogen and phosphorus loading of semi-intensive and intensive culture in reservoirs is derived from fertilizer, uneaten feed, fish mortality, and fecal or urinary wastes. As we have seen earlier, fertilization is a common strategy to enhance the fish production in medium and small reservoirs, as well as in coves and tail-water bodies. The types and quantities of fertilizers are determined by cost, availability, species, and intensity of fish stocking (Li 1992). Organic fertilizers, such as wastes from poultry and livestock, can be directly toxic if large quantities are added, especially when ammonia concentrations and water pH values are high. Moreover, fertilizers are usually important nitrogen and phosphorus sources to reservoirs (Pullin 1989). In both semi-intensive and intensive aquaculture, a proportion of the feed provided remains uneaten because exact quantification of feed for fish is difficult and the management always tends to optimize ingestion (Beveridge 1987; Liu et al. 1997). The percentage of uneaten feed ranges from 1% to as much as 30% (Liu 1996). Types of feed and management practices are critical. Usually, loss rate of pelleted diets is much less than trash fish, and the proportion of uneaten food in cages is considerably higher than in tanks (Beveridge 1987). Obviously, uneaten food is a source of nitrogen and phosphorus loading. Besides the uneaten feed, the undigested fraction (together with mucus, sloughed intestinal cells, and bacteria) voided as feces is also an important source of nitrogen and phosphorus loading to environments. Studies on intensive cage culture of salmonids have shown that nearly 75% of the nitrogen and phosphorus in feeds are lost to the environment (Folke and Kautsky 1989). The nitrogen and phosphorus input–output of intensive cages in Dingshanhu Lake showed that the ratio of loss to input (seed and feed) for N is 68.14% and for P is 82.46% (Shi and Liu 1989). Clearly, nitrogen and phosphorus loading of semi-intensive and intensive culture is one of the worst detriments to the water quality of reservoirs.

With the increasing of cage culture scale and concern of water quality, more attention is paid to the carrying capacity of water bodies for cage culture. Xiong et al. (1994) suggested that the area of cages to water bodies be 1:355. However, different reservoirs have different trophic conditions and different requirements for water quality. In some drinking water reservoirs, cage culture should be prohibited completely, and the carrying capacity for this kind of reservoirs is zero. Based on maximum permissible concentration of nutrients, trophic level, retention time, and other parameters, Peng et al. (2004) built a series of models for evaluating the carrying capacity of water bodies for cage culture. The models include:

The carrying capacity of waters for monocultural cages:

$$P_{\mathrm{cage}} = (a \times H \times r \times \Delta P \times h) \times \left(W_{\mathrm{f}} \times (1-R) \times \left(P_{\mathrm{m}} \times b^{-1} + F \times P_{\mathrm{F}} - P_{\mathrm{f}}\right)\right)^{-1} \times 1‰$$

The carrying capacity of cage-cultured piscivorous fish in polyculture:

$$P_{\mathrm{s}} = P \times \left(A \times W_{\mathrm{fs}} \times \left(P_{\mathrm{m}} \times b^{-1} + F_{\mathrm{s}} \times P_{\mathrm{i}} - P_{\mathrm{f}}\right) \times h^{-1}\right) \times 1{,}000‰$$

The carrying capacity of trash fish fed to fishes in polyculture:

$$P_{\mathrm{c}} = F_{\mathrm{s}} \times P \times \left(A \times W_{\mathrm{fc}} \times \left(P_{\mathrm{m}} \times b^{-1} + F_{\mathrm{s}} \times P_{\mathrm{i}} - P_{\mathrm{f}}\right) \times h^{-1}\right)^{-1} \times 1{,}000‰$$

The carrying capacity of the total cages in polyculture:

$$P_{\mathrm{cage}} = (a \times H \times r \times \Delta P) \times ((1-R) \times P_{\mathrm{t}})^{-1} \times \left(1 \times W_{\mathrm{fs}}^{-1} + F_{\mathrm{s}} \times W_{\mathrm{fc}}^{-1}\right) \times 1‰.$$

where P is the carrying capacity of the water for phosphorus (kg year^{-1}); a is the valid volume coefficient(%), i.e., the amount of valid volume in the total volume; H is the mean depth (m); A is the reservoir area (m^2); r is the yearly exchange rate of the water; R is the retention coefficient of phosphorus (%); $P_{\max}$ is maximum concentration of phosphorus allowed in the water (mg L^{-1}); P_o is the original concentration of phosphorus in the water (mg L^{-1}); P_{cage} is the carrying capacity of the water for cages (‰); ΔP is the allowed increment concentration of phosphorus in the water (mg L^{-1}); h is the survival rate of fish in cages (%); W_f is the unit fish production in cages (kg m^{-2}); P_m is the percentage of phosphorus in the fingerlings (%); b is the multiple of weight growth of fish; F is the feed coefficient; P_F is the percentage of phosphorus in the feed (%); P_f is the percentage of phosphorus in the adult fish (%); P_s is the carrying capacity for cages of piscivorous fish in polyculture (‰); P_i is the loaded weight of phosphorus of unit trash fish production (kg kg^{-1}); P_c is the carrying capacity for cages of cultured trash fish (‰); W_{fs} is the production of piscivorous fish in polyculture (kg m^{-2}); F_s is the survival rate of cultured trash fish (%); and W_{fc} is the production of cultured trash fish in polyculture (kg m^{-2}).

On the basis of the above models, the carrying capacity of cage-cultured mandarin fish in Fuqiaohe Reservoir was 1.60‰ when mandarin fish was monocultured and 0.21‰ when Jian carp was monocultured; and 0.31‰ for mandarin fish and 0.21‰ for Jian carp with a summation of 0.52‰ under polyculture.

15.4.3 Wastes of Fisheries Activities in Reservoirs

Most kinds of wastes in fisheries activities are listed as follows: therapeutants, such as formalin, potassium permanganate; malachite green, antibiotics and antimicrobials; pesticides, particularly organophosphate compounds; hormones and growth promoters used to change the sex, productive viability, and growth of cultured organisms; antifoulants used to poison fouling organisms on cage and pen nets; some construction materials, such as heavy metals and plastic additives; fuel leaked from powerboats and other machines; household garbage of the people engaged in reservoir fisheries, such as domestic sewage and package of living goods. Many of these compounds are toxic or harmful to aquatic life and have more or less negative impacts to the reservoir environments, and detailed studies on these aspects should strengthened in the future.

15.5 Conclusions

Although much progress has been achieved in reservoir fisheries in China since the 1950s, there is still potential for further improvements. From the point of view of unit yield, fish production per hectare of reservoir is less than half of lake fisheries and only about 10% of pond culture (Wang et al. 1989). From a market's perspective, demands for aquatic products are on the increase with respect to quality and quantity, as living standard improves day by day. Moreover, per-capita consumption of aquatic products in remote areas is much less than that in urban areas, such as in most of northwestern and southwestern provinces. In China, market demands for aquatic production will continue to increase for a long time. However, pond culture, the mainstay of freshwater aquaculture, has developed to an unprecedented stage. The unit yield in ponds demands much more input and investments and also involves more risks, such as catastrophic fish disease or natural disasters (drought, flood, or typhoons). Therefore, further substantial expansion in pond culture will be slow and difficult. The development of lake fisheries is also restricted by constraints, such as silting, aging, and eutrophication. Obviously, the increasing demand for inland fisheries production will therefore depend on an increase in reservoir fisheries. In the new millennium, reservoir fisheries deserve more attention from governments and the general public. Reservoir fisheries should evolve to maximize economic, social, and ecological benefits for Chinese society in a sustainable and eco-friendly way.

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Part III
Eutrophication

Chapter 16
Water Supply and Eutrophication of Reservoirs in Guangdong Province, South China[1]

Bo-Ping Han, Zhengwen Liu, and Henri J. Dumont

Abstract Surface water in Guangdong province (South China) is abundant but distributed irregularly. Total water demand in Guangdong is 44.7×10^9 m^3, one-third of which is supplied by reservoirs. Since 1950s, about 6,700 reservoirs ($>10^6$ m^3) with a total water storage capacity of 39.8×10^9 m^3 have been built. About 75% of all reservoir water is contained in small ($<10^7$ m^3) and very small reservoirs, and only 25% in large- and medium-sized reservoirs. Xinfengjiang is the largest reservoir of all; it contributes one-third of the total storage of the province. Half of the total storage is situated in the Dongjiang River watershed in which the designed water supply capacity is 1.82×10^9 m^3. An annual amount of 1.1×10^9 m^3 is supplied to Hong Kong by a mid-sized receptacle (Shenzhen reservoir), and 8.31×10^6 m^3 for use by Macau are stored in another four, smaller reservoirs. In spite of the significance of reservoirs, the protection of their water quality was ignored prior to 2005. By that time, the deterioration of water quality had progressed to a point where action became unavoidable: the local government first commissioned a survey of the eutrophication of 20 drinking water reservoirs. A trophic status index (TSI) based on TP, TN, SD, and chlorophyll *a*, revealed that most of them were mesotrophic, but approaching a state of eutrophy. Only two reservoirs, situated in the upstream zone, were oligotrophic. Compared with data of a fishery survey in the 1980s, reservoir trophic status had strongly increased and corrective measures were needed. The local government responded by initiating

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projects aimed at collecting up-to-date information on eutrophication in reservoirs known to be infested by harmful Cyanobacteria and with strong internal loadings. A new policy for the management of the water quality of reservoirs in general was worked out as well.

16.1 Introduction

In a series of well-documented reports, the United Nations (UN World Water Development Report 2003, 2006, 2009) predict that in the course of the twenty-first century, usable water may become even more limiting than energy worldwide. The water use equation has many aspects, however. Water use by human households for cooking and sanitation is only a fraction of total demand. Industry too is a modest water user. Irrigated agriculture, in contrast, uses about 70% of available water, and since agriculture is expanding, intensifying, and spreading to areas that had previously been left fallow, demand on freshwater is growing in direct proportion to demographic expansion. Water is also unevenly spread in space and time. In monsoon-governed climates, for example, most precipitation falls concentrated in the 4–5 summer months of the year, and quickly evacuates to the sea. The damming of rivers and the construction of pumped storage reservoirs is a widespread response to limit such losses, at first mainly in areas where relief permits, but currently almost worldwide. Originally, reservoirs were constructed in regions where natural water reserves are inadequate (Straskraba and Tundisi 1999) but today, the only continent that has remained reservoir-free is Antarctica.

Reservoirs serve multiple purposes that include drinking, sanitary and cooling water, industrial and irrigation use, flood control, fisheries and aquaculture, recreation, transportation, and power generation. As a rule, a maximization of water uses is attempted, but some combinations, such as using the same reservoir for drinking water and for collecting sewage effluents are clearly in contradiction with each other. In China, there are approximately 86,000 reservoirs ($>10^6$ m^3) with a total volume of 466×10^9 m^3 (Liu and He 1996). The majority is located in the south, where about 80.2% of the nation's water is found and which harbors 57.9% of the national population. Guangdong Province is one of the provinces of this region. It differs from adjacent southern areas by its scarcity of lakes (only 13 km^2). In order to respond to the local water needs and these of adjacent megacities such as Macau and Hong Kong (see below), and given the temporal and spatial irregularities, a major effort in reservoir construction took place from the 1950s till the 1980s. While originally power generation was high on the agenda, the sharply increasing water demand in this demographically and industrially highly developed province of China moved water supply up to become the primary reservoir use in Guangdong (Department of Water Resource of Guangdong 1996).

The province of Guangdong extends along the coast of the South Chinese Sea, forming a triangle that surrounds the estuary of the Pearl River System and covers some 178,000 km^2. With about 12% of China's economic output, Guangdong has one of the largest GDP per capita of China. Since 2005, it has been more populous than Henan and Sichuan, with ca 80×10^6 regular inhabitants, augmented by ca 30×10^6 economic migrants. Mountains and hills, situated mainly in the north, cover about 61.2% of its surface area, while plains occupy ca 24%. Most of the plains are heavily exploited for valley agriculture, with rice the primary crop (two annual harvests) followed by various fruits, vegetables, and aquaculture. A varied traditional and electronics industry has sprung up in and around the delta of the Pearl River in the past two decades.

Early in the 1980s, rivers began to suffer water quality loss under the pressure of a growing population and economic development. Although the problematic nature of this situation has been realized for a long time and some attempts were made to manage it, rivers downstream fell victim to gross pollution. All reservoirs located downstream now show various degrees of water quality deterioration, the dominant and accelerating trend being eutrophication. In 2000, Guangdong government initiated a first program for a systematic survey of the eutrophication of reservoirs that supply drinking water. To cite only one example, Shenzhen reservoir, a middle-sized water body that primarily supplies water to the city of Hong Kong, has currently become so eutrophic that its water can only be used after extensive treatment. In this contribution, we summarize knowledge on the water supply by Guangdong reservoirs and provide information on the eutrophicaton of these reservoirs.

16.2 Reservoir Water Resources and Water Supply

16.2.1 Reservoir Water Resources

Guangdong province is transitional between the warm-subtropical and tropical zones, bounded by 109°40′ to 117°20′ E and 20°14′ to 25°31′ N, an area where the southwest monsoon and the southeast trades prevail. The long-term annual average precipitation is 1,744 mm, with an annual runoff of 180×10^9 m^3. Under the effect of the southwest monsoon from the Indian Ocean and from tropical storms, precipitation is abundant during the flood season from April to October and contributes about 70–85% to annual precipitation. The scarcity of natural lakes results in most of the flood water being lost directly to the sea. In the dry season, only 15–30% of the annual amount of precipitation is received. Many rivers, especially small ones, may dry out. Precipitation is also distributed irregularly in space, ranging between 400 and 2,800 mm, with a minimum of 400 mm in Leizhou Peninsula (West Guangdong). In dry years, coastal and limestone areas are suffering from water shortage, and evaporation exceeds precipitation.

According to traditional management, river systems in Guangdong Province used to be divided into ten districts, dominated by the many arms of the Pearl River (Table 16.1): (1) Xijiang River watershed, (2) Beijiang River watershed, (3)

Table 16.1 Water resources in Guangdong (in 1995)

Watershed	River runoff (10^6 m^3)		Total reservoir volume (10^6 m^3)	Designed reservoir water supply capacity (10^6 m^3)
	Local	From upstream		
Guihejiang River	2,714	6,500	125	162
Xijiang River	12,104	207,500	996	1,498
Beijiang River	48,210	3,300	3,526	4,913
Dongjiang River	23,917	3,000	18,650	1,820
Hanjiang River	15,244	11,200	1,511	1,191
Rivers in the west coastal area	31,428	1,800	5,984	3,032
Rivers in the east coastal area	19,323	0	3,275	2,140
Rivers in Pearl delta area	26,713	294,100	4,271	2,687
Water system of Dongting Lake	84	0	8	7
Water system of Boyang Lake	135	0	4	6
Total	179,872	233,000	38,350	17,455

Dongjiang River watershed, (4) Hanjiang River watershed, (5) rivers of the Pearl delta area, (6) rivers in the east coastal area, (7) rivers of the west coastal area, (8) Guihejiang River watershed, (9) water system of Donting Lake, and (10) water system of Boyang Lake. The Guihejiang River watershed, system of Donting Lake and system of Boyang Lake are the three smallest watersheds, with a total area of only 3.33×10^9 m^2, about 1.9% of the Province's total. Figure 16.1 shows only the locations and reservoir volumes of the other seven watersheds. The Xijiang, Beijiang, and Dongjiang Rivers are the three largest rivers in Guangdong. They all originate from the adjacent provinces and converge into the river system of the Pearl delta area; the whole system is known as Pearl River, the fourth longest river in China. Their natural runoff is compensated not only by local rainfall, but also by flows from upstream in the adjacent provinces. Total water resources in Guangdong consist of precipitation (180×10^9 m^3) and runoff from adjacent provinces (233×10^9 m^3), with a total of 419×10^9 m^3. There are also few natural lakes, with an area of 13 km^2. Building reservoirs by damming rivers is the only way to regulate these rivers and respond to the water needs temporally and spatially. As a result, there are seven large ($>10^9$ m^3), 22 medium ($>10^8$ m^3), and 6,674 small ($>10^6$ m^3) reservoirs with a total volume of 39.8×10^9 m^3 (in 1999) in Guangdong. A volume of 22.5×10^9 m^3 is contributed by large reservoirs and only 4.4×10^9 m^3 by medium-sized ones. The largest is Xinfengjiang reservoir with a volume of 13.9×10^9 m^3, representing about one-third of total reservoir volume.

16.2.2 Reservoir Distribution

Reservoir distribution is shown in Table 16.1. In the Dongjiang River watershed, there are three large reservoirs: Xinfenjiang (13.9×10^9 m^3), Fengshuba (1.9×10^9 m^3), and Baipenzhu (1.2×10^9 m^3), together 18.7×10^9 m^3 in volume and representing about half of the total storage capacity in the watershed. The annual runoff is 26.9×10^9 m^3, occupying only 6.5% of the total water resource of the province. Rivers in this watershed are strongly regulated by reservoirs. Runoff in all the other watersheds is high, up to 412.9×10^9 m^3, with a reservoir volume of 19.7×10^9 m^3. In these areas, rivers are less well regulated by reservoirs. In Xijiang River watershed, the runoff is 219.6×10^9 m^3, with a reservoir volume of 0.996×10^9 m^3, contributed by small and very small reservoirs and the river is little regulated by damming. In 1998, a large flow-through water body, Feilaixia reservoir (1.9×10^9 m^3), was constructed on the Beijiang River for water supply, power generation, and flood control.

16.2.3 Reservoir Water Supply to Guangdong

Based on the statistical data for 1993, reservoir volume is 38.35×10^9 m^3 (excluding Feilaixia reservoir), with a designed water supply capacity of 17.45×10^9 m^3. Figure 16.1 shows the reservoir volume and designed water supply capacity distribution in three reservoir types. With a volume of 13.32×10^9 m^3, small and very small reservoirs have an important water supply capacity that reaches 13.07×10^9 m^3. Although large and medium reservoirs have a high volume of 25.03×10^9 m^3 and 65% of the total storage capacity, their designed water supply capacity is 4.39×10^9 m^3, which equates to only 25% of the total water supply. These conditions indicate that most of reservoir water supply is contributed by small and very small reservoirs. For example, in Xijiang river watershed, the runoff is 219.60×10^9 m^3, but the reservoir volume is only 0.996×10^9 m^3, contributed

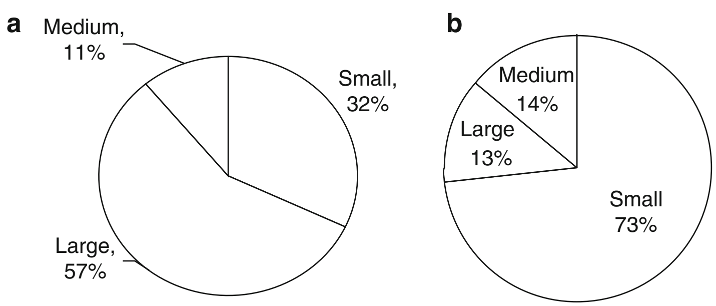

Fig. 16.1 Reservoir volume (**a**) and designed water supply capacity (**b**) distribution in three reservoir types

by small and very small reservoirs, with a water supply capacity of 1.50×10^9 m^3. Limited by volume, small and very small reservoirs cannot store enough floodwater to meet the demand in the dry season. These small reservoirs have a limited potential to further enhance their water storage capacity in the future. Large and medium reservoirs can regulate water storage for use annually. Although their volume is high, most of them have a low designed water supply capacity. Only in the west coastal area that often faces water shortage and is quite developed in agriculture, large reservoirs (Hedi and Gaozhou reservoirs) are used efficiently for irrigation. But in Dongjiang river watershed, the three large reservoirs (i.e., Xinfengjiang, Fengshuba, and Baipenzhu reservoirs) have a high volume of 17×10^9 m^3, with a low water supply capacity 1.15×10^9 m^3. The province's water demand was 42.8×10^9 m^3 in 1993, and 44.7×10^9 m^3 in 1998, with about one-third of the amount contributed from reservoirs. Generally, if water supply becomes the primary purpose for most of the large and medium reservoirs, and their water supply capacity is enhanced, available water supply would be significantly improved to meet the increasing water demand in future.

16.2.4 Reservoir Water Supply to Hongkong

Hong Kong and Macau constitute two special administrative regions (SARs) of China. Hongkong is a 1,100 km^2 area composed of Hongkong Island, Kowloon peninsula, and the so-called New Territories with some seven million inhabitants. In spite of this, less than 25% of its surface area is developed as urban territory. The rest is hills and mountains, much of which is conservation area. Hongkong, adjacent to Guangdong province, depends on limited, seasonally variable water resources. Annual rainfall varies from 901 to 3,248 mm, with a mean value of 2,214 mm. The summer monsoon, lasting from May to September, contributes 77% of the total annual rainfall, and water rapidly flows down the mountains in streams without natural lakes. Because of this seasonality of its rainfall, Hong Kong has to rely on artificial water storage for the maintenance of a regular supply. As a result, 17 reservoirs, with a volume of 0.586×10^9 m^3 have been constructed. But even these cannot meet a water demand that increases by 11% annually, among other reasons caused by a growing population (Dudgeon 1996).

This increasing demand has forced Hongkong to import water, purchased from Guangdong province and conducted via the Dongjiang water supply project, completed in 1960. The project is located in the downstream area of Dongjiang River. In a first step, river water enters the Shenzhen reservoir (0.046×10^9 m^3) by a multi-pumping system, and next it is transferred to Hongkong through pipes. After three enlargements, the water supply capacity has now reached 1.743×10^9 m^3. About 1.1×10^9 m^3 is allocated to Hongkong yearly, amounting to 8% of the total discharge of the river. This currently meets 75% of the water demand of the city (Dudgeon 1995).

16.2.5 Reservoir Water Supply to Macau

Macau is composed of three islands situated west of the Pearl River Delta, opposite to Hongkong. Its total surface area is 29.5 km^2, and its population numbers about 500,000 people. Macau continuously risks falling short of water. To relieve this threat, two small reservoirs, Nanping and Zhuxiandong, were constructed in 1960, and supply 2.94×10^6 m^3 water to Macau annually. But their capacity was rapidly outpaced by the increasing water demand of a growing population. So in 1979, a further two small reservoirs, Dajingshan and Meixi, began supplying water to the city, and the annual amount abstracted rose to 5.37×10^6 m^3. At present, there are four reservoirs providing water to Macau, and the yearly supply is 8.31×10^6 m^3. The water demand of adjacent Zhuhai city has been increasing with urbanization and increase in population too. Water supply to both Macau and Zhuhai became more difficult and complicated after a serious saltwater intrusion up the estuary of the Pearl River system in 2005, caused by the extensive human activity in the catchment. A new reservoir, Zhuying reservoir, is currently being built specially to supply extra water to Macao. Zhuying reservoir will be the largest pumped storage reservoir in the city. When completed in 2011 it will store more than 40 million cubic meters, pumped up from the west branch of the Pearl River during the dry season.

16.3 Eutrophication

In spite of the importance of reservoirs for water supply, storage protection was long ignored. Only in 2000 did the progressive deterioration of reservoir water quality persuade the local government to initiate a systematic survey of the eutrophication of its drinking water reserves. This made researchers and managers join forces in trying to improve water quality management. Twenty reservoirs were selected to be part of the survey. They represented the river basins and river districts where most reservoirs are located, as well as the full scale of reservoir types according to morphology and characteristics of their watersheds. The location and typology of these reservoirs are shown in Fig. 16.2 and Table 16.2. Reservoirs were sampled twice between June and December 2000, roughly corresponding to the wet and dry seasons. Samples were collected at locations close to the inflows and dams.

16.3.1 Physical and Chemical Variables

Conductivity varied from 27 to 273 μs/cm, as an expression of the geology and land uses of the watersheds. High conductivity occurred in Shiyan (273 μs/cm) and

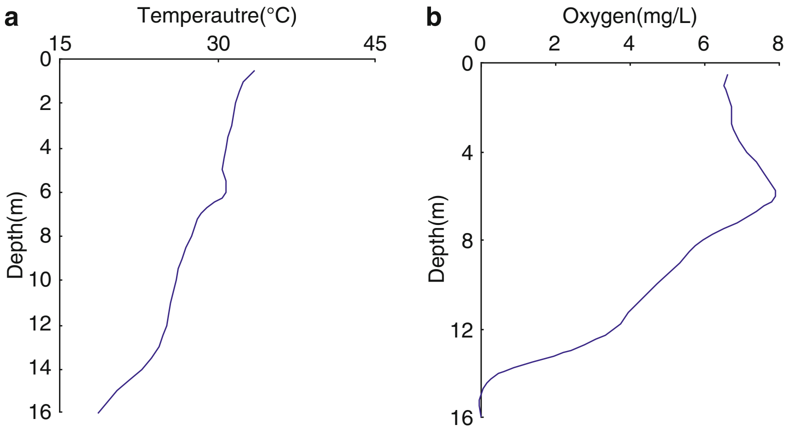

Fig. 16.2 (**a**) Summer thermal stratification and (**b**) the oxygen clinograde profile of Shatian reservoir

Table 16.2 Description of the reservoirs investigated

Watershed	Reservoir	Elevation (m)	Watershed area (km^2)	Max. volume (10^6 m^3)	Normal volume (10^6 m^3)	Mean retention time (year)	Year of filling
Dongjiang River	Xinfengjiang	116.0	5,734.0	13,980.0	10,800.0	2.00	1958
	Baipenzhu	75.0	856.0	1,220.0	575.0	0.52	1987
	Shatian	68.0	26.8	21.7	14.2	0.68	1960
	Shiyan	36.0	44.0	31.2	16.9	0.47	1960
	Shenzhen	27.6	60.5	46.1	35.2	0.02	1960
Beijiang River	Chishijin	128.4	14.1	14.9	12.4	1.50	1958
	Xiaoken	225.2	139.0	113.2	54.3	0.37	1964
	Feilaixia	24.0	34,000.0	1,900.0	440.0	0.04	1998
Rivers in the west coastal area	Gaozhou	86.0	1,022.0	1,151.1	841.8	0.57	1960
	Dashahe	34.8	217.0	258.1	156.8	0.56	1959
	Dashuiqiao	56.5	196.0	143.0	100.7	1.00	1958
	Hedi	40.5	1,495.0	1,144.0	795.0	0.53	1959
Rivers in the east coastal area	Gongping	16.0	317.0	330.7	163.3	0.41	1962
	Chisha	12.0	23.0	1.1	1.1	0.04	1960
	Hexi	53.0	40.9	17.9	15.8	0.39	1958
	Tangxi	56.0	667.0	381.0	286.4	0.43	1959
Rivers in Pearl delta area	Liuxihe	235.0	539.0	378.0	326.0	0.46	1958
	Dajingshan	20.4	6.0	11.7	10.5	1.13	1975
	Qiuyishi	42.6	17.6	13.0	10.2	0.74	1960
Hanjiang River	Heshui	134.0	600.0	115.0	30.4	0.07	1957

Shenzhen (196 μs/cm) that have been subject to nutrient-rich domestic and industrial wastewater. Surface temperature ranged between 27.0 and 33.6°C in summer. Except for through-flowing (Feilaixia, Heshui, Chisha, and Shenzhen) and shallow, small (Shiyan) reservoirs, all had typical summer thermal stratification (Fig. 16.2). In winter, surface temperature declined and varied from 16.4°C to 24.0°C. Only deep reservoirs would still remain stratified, but the depth of the thermocline increased. Dissolved oxygen stratification patterns showed no difference among reservoirs. They all displayed clinograde profiles in summer (Fig. 16.2).

Chlorophyll *a* concentration (Chl-*a*) ranged from 0.6 to 32.4 μg/L, with a minimum in Xinfengjiang reservoir (oligotrophic) and a maximum in Qieyeshi reservoir (eutrophic). Secchi disk depth (SD) showed a strong relationship with Chl-*a* (Chl-$a = 7.31\text{SD}^{-1.37}$, $R^2 = 0.725$) and varied from 0.4 to 6.3 m (Fig. 16.3a). Low Secchi disk depth occurred in Qieyeshi reservoir (0.4 m) and Shiyan reservoir (0.6 m) at high Chl-*a* concentrations. In Heshui reservoir, strong erosion of soil in the watershed resulted in high turbidity of the water and low Secchi disk depth (0.6 m). High values occurred in Xinfengjiang reservoir (6.3 m) and Liuxihe reservoir (4.7 m), but at lower chlorophyll *a* concentration.

Total phosphorus (TP) varied from 3 to 388 μg/L, with trace concentrations in Xinfengjiang and Liuxihe, the only reservoirs that showed P-limitation. High TP was found in Shenzhen (388 μg/L) and Shiyan reservoirs (189 μg/L), both of which were eutrophic. Total nitrogen (TN) ranged between 0.313 and 7.15 mg/L. Coupled with TP, high concentrations were found in Shenzhen (5.93 mg/L) and Shiyan (7.15 mg/L) reservoirs, and low concentrations in Xinfengjiang (0.46 mg/L) and Liuxihe (0.31 mg/L). The most abundant form of nitrogen was NO_3–N, ranging widely, from 0.08 to 5.00 mg/L. NH_4–N only predominated in Dashuiqiao and Hedi reservoirs (in the west coastal area where agriculture is well developed). Although NO_3–N was the most prominent form of nitrogen in Shenzhen reservoir supplying water to Hongkong, NH_4–N was predominant in its inflow, where it reached up to 4.85 mg/L. High concentrations of NH_4–N caused an unpleasant smell to the

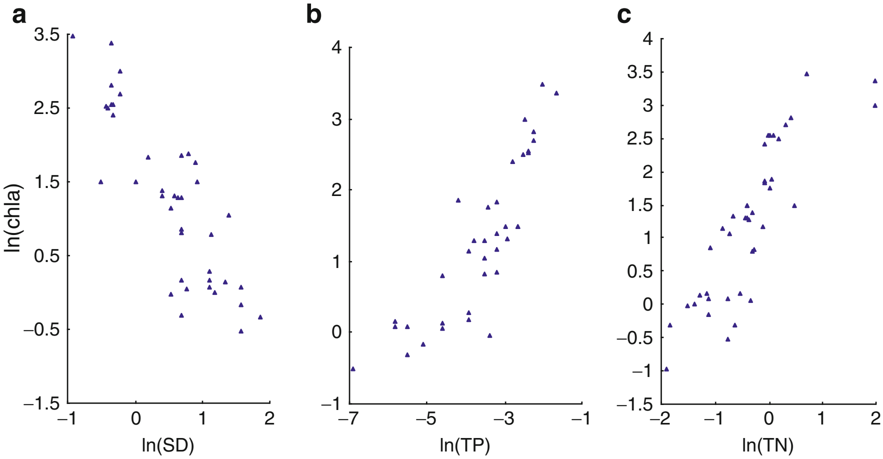

Fig. 16.3 (**a**) Relationship between Secchi disk depth (SD) and chlorophyll *a* concentration (Chl-*a*). (**b**) Relationship between total phosphorus (TP) and Chl-*a*. (**c**) Relationship between total nitrogen and Chl-*a*

reservoir water. In order to ameliorate the water quality, a pre-nitrification project was implemented in 1998 to oxidize NH_4–N to NO_3–N; as a result thereof, the concentration of NH_4–N declined to 0.03 mg/L near the dam.

The relationship between Chl-*a* and TP was tolerably well described by Chl-$a = 59.20\text{TP}^{0.77}$, $R^2 = 0.731$ (Fig. 16.3b). A relationship was also observed between TN and Chl-*a* (Chl-$a = 5.31\text{TN}^{1.21}$, $R^2 = 0.7163$, Fig. 16.3c). TP and TN both are good predictors of chlorophyll *a*. In this case, however, both variables co-varied and regression analysis alone was insufficient to decide which nutrient, if any, most limited algal biomass in each reservoir. Trace elements have not yet been studied.

16.3.2 Phytoplankton and Zooplankton

So far, ten species of Cyanophyta, 63 species of Chlorophyta, 18 species of Diatoms, two species of Chrysophyta, two species of Pyrrophyta, nine species of Euglenophyta, and one species of Cryptophyta were observed in the reservoirs (Hu et al. 2002). In water-bodies with high chlorophyll *a* concentration, phytoplankton was numerically dominated by hyper-abundant Cyanophyta (often called Cyanobacteria) *Microcystis aeruginosa* and *Phormidium* sp. In reservoirs with a low chlorophyll *a* concentration, phytoplankton was low (e.g., 0.19×10^6 cells/L in Xinfengjiang reservoir) and without a salient dominant species. Diatoms, Chlorophyta, or Cyanophyta dominated numerically in reservoirs with medium chlorophyll *a* concentration and phytoplankton abundance.

In the zooplankton, an inventory was compiled of mainly pelagic species, the littoral development of the reservoirs being insignificant. The "classical three" groups, Rotifera, Cladocera, and Copepoda, were all well represented, but supplemented by few unusual groups, like typhloplanid flatworms (*Rhynchomesostoma* sp. and possibly other genera), dipteran larvae (*Chaoborus* sp.), glochidia larvae of mollusks, and at the small end of the spectrum, pelagic protists (ciliates, like *Stentor* and others, Heliozoans, and naked and testate amoebas of the genus *Difflugia*). Reservoir plankton influenced by seawater even included planktonic larvae of a polychaete (*Manayunkia* sp.). In total, 61 species of Rotifera, 23 species of Cladocera, and 14 species of Copepoda were to date identified in summer and winter (Lin et al. 2003). The majority of the rotifers were monogononts. Bdelloids were represented by *Rotaria* sp. only. Among monogononts, *Lecane*, *Trichocerca*, and *Brachionus* were the most speciose genera, with most of the species present cosmopolitan. The most frequently observed genera were *Keratella*, *Brachionus*, *Polyarthra*, *Trichocerca*, *Asplanchna*, *Conochilus*, *Ploesoma*, *Ascomorpha*, and *Pompholyx*. Among the Cladocera, representatives of three orders were recorded. Daphniidae, Bosminidae, and Chydoridae (Anomopoda) were the three families with the highest species richness. *Bosmina tripurae*, *Bosminopsis deitersi*, *Moina micrura*, *Ceriodaphnia cornuta,* and the ctenopods *Diaphanosoma orghidani* and *D. dubium* were the most frequently observed taxa. Very large species such as the onychopod cladoceran *Leptodora richardi* (not the related *L. kindtii*, which is northern euro-siberian in distribution) and *Daphnia galeata* were only observed at low density in some large and deep reservoirs, and occurred regularly in the winter samples only. Cold, deep impoundments provide these species a "dark" vertical refuge to maintain a low population in the presence of fish predation (Wang et al. 2011; Xu et al. 2011). Among Copepoda, ten Calanoid and eight Cyclopoid species were found. Most of the Calanoida were relatively large and endemic to the tropics and subtropics of China or southeast Asia. All cyclopoids that could be identified with reasonable certitude also belonged to species with southeast Asian ranges. Some of them, however, may still be found in older papers under erroneous names, which strictly apply to euro-siberian species only. *Phyllodiaptomus tunguidus*, *Neodiaptomus schmackeri*, and *Mesocyclops thermocyclopoides* were the most frequent copepods, observed in the majority of the reservoirs. Copepods generally are better at coexisting with fish predators than cladocerans because of their superior swimming and escaping capacities.

Macrozooplankton (Cladocerans, adult Copepods, and copepodids) densities were between 1.0 and 400 ind./L, but generally below 50 ind./L, and were dominated by copepodids. Microzooplankton (rotifers, nauplii, and protist) densities were between 7 and 680 ind./L and dominated by nauplii. The highest zooplankton density was observed in Tangxi reservoir that was suffering from a small *Microcystis aeruginosa* bloom; macrozooplankton had a density of 402 ind./L and microzooplankton 432 ind./L. Low macro- and microzooplankton densities were coupled with low chlorophyll *a* concentration. In Xinfengjiang reservoir, the densities were only 6 and 7 ind./L for macro- and microzooplankton, respectively. High densities of zooplankton were not necessarily coupled with a high chlorophyll *a* concentration (thus algal stock) in these reservoirs. Two hypotheses were formulated: (1) phytoplankton biomass was only very partly utilized by zooplankton, and an substantial fraction of the algae was channeled through the microbial loop and (2) zooplankton biomass control was top-down, and taken care of by a large stock of planktivorous and filter feeding fish (Straskrabova et al. 1994).

16.3.3 Trophic Status Index

A trophic status index (TSI) was calculated for each reservoir using a formula that took into account the concentration of TP, TN, chlorophyll *a*, and secchi disk depth (SD). This TSI varied from 23 to 66 (Fig. 16.4). Only Xinfengjian and Liuxihe reservoirs were found to be oligotrophic, while Hedi, Qieyeshi, Shenzhen, and Shiyan reservoirs were outspokenly eutrophic. The other 14 reservoirs were mesotrophic, but among them, the Dajingshan, Heshui, Dashuiqiao, Tangxi, and Hexi reservoirs were clearly trending toward eutrophy.

As was expected, the trophic state was best (lowest) in the headwater reservoirs, and increased toward the lower reaches in the Dongjiang and Beijiang river watersheds. In Dongjiang River watershed, the trophic state varied strongly, and

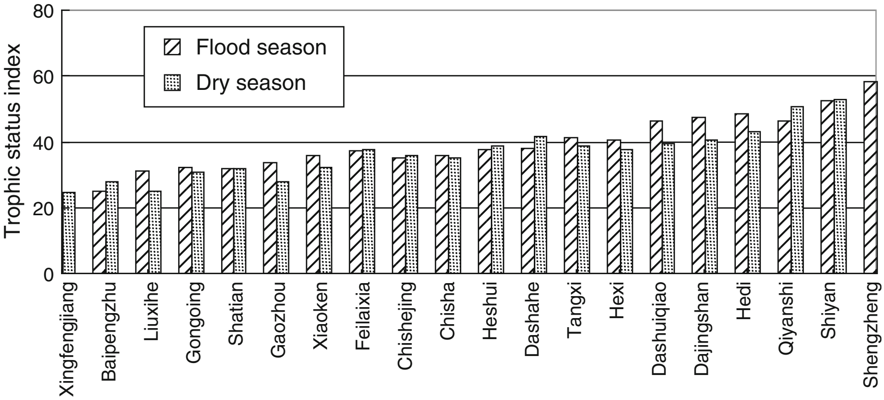

Fig. 16.4 Trophic state level (TSI) of the investigated reservoirs, TSI was calculated on the basis of variables near the dams

went from oligotrophic, in the upstream Xinfengjiang reservoir, to deeply eutrophic in the downstream Shenzhen and Shiyan reservoirs. All reservoirs in the Beijiang River watershed were mesotrophic. Heshui reservoir was the only reservoir investigated in Hanjiang River watershed. It is located upstream and was categorized as mesotrophic.

In the Pearl delta area, Liuxihe reservoir is a water body situated in a hilly environment that is part of the upper reaches of the Liuxihe River, a small tributary to the Pearl River. It receives almost no domestic or industrial wastewater; and its trophic state is oligotrophic. In contrast, Dajingshan and Qieyeshi reservoirs are mesotrophic and eutrophic, respectively, and receive nutrient-rich wastewater.

In the western and eastern coastal areas, some reservoirs are located on several small rivers a short distance before they flow out into the sea. Hedi and Dashuiqiao reservoirs are situated on Leizhou Peninsula, with its unusually low annual precipitation of only 400 mm. Evaporation here well exceeds precipitation. Most of the reservoir water, especially from Hedi reservoir, is used for irrigation and part of it drains back, loaded with nutrients. Because fertilizer use on the land affects water quality, Hedi reservoir water was found to be eutrophic; that of Dashuiqiao reservoir was mesotrophic, tending toward eutrophy. Tangxi reservoir, located in the east coastal area, is fed by a small stream, Huanggang River. Other than power generation, this reservoir also supplies water (0.223×10^9 m^3 annually) to towns downstream. It is medium sized and is typically eutrophic because nutrient-rich domestic and industrial wastewater from eight towns upstream is carried down by a number of tributaries and ends up getting injected into the reservoir. During the investigation, a *Microcystis aeruginosa* bloom occurred here that lasted half a year. The other reservoirs in these two areas were mesotrophic.

16.4 Conclusions

The reservoirs of Guangdong province typically have a high storage capacity if situated on low runoff rivers, and a low storage capacity if situated on high runoff rivers. Most of these rivers are little regulated by their reservoirs, except for the dams on the Dongjiang River. Reservoirs, especially small and very small in size, are vital for providing towns and cities with sufficient water. At present, reservoir supplies still meet the increasing water demand of the entire province of Guangdong, but by a margin that is rapidly melting. A first survey, designed to collect general data on water quality, did not account for short time scales or small spatial scales and provided mainly historical information of reservoir eutrophication. Since that first survey, cyanobacterial blooms have spread further and now occur also in several large reservoirs. For example, *Microcystis* bloomed in Nanshui reservoir located in northern Guangdong in 2007 and a mixed bloom of *Anabaena* and *Microcystis* occurred in Dashahe in 2005 and in Gaozhou reservoir in 2009. Nutrient loading in most reservoirs is the root cause. Its origins are unchecked fertilizer loss from rice fields, poultry and livestock farms, and domestic wastewater released without any treatment. In view of the deleterious effects of these large-scale water blooms, that include quite a few toxic algal strains, that immediately threaten the safety of the water supply, the local government began working out a new policy for water resource protection. Two more surveys of water blooms, including studies on harmful Cyanobacteria and on internal nutrient loading to the reservoirs, were completed in 2003–2004 and 2007–2008 (Han 2010; Han et al. 2006). However, while these provide a basis for sound science-based policies, implementing an efficient water resource management is only partly science; the policies themselves are beyond science and pose a challenge to the traditional administrative system in China. In Guangdong province, for example, water resource management is run by several governmental administrations. The Water Resource Department takes charge of surface water and flood control, the Committee of Urban Construction ensures urban water supply and drainage, the Bureau of Agriculture oversees agricultural water usage, and the Bureau of Environmental Protection controls wastewater discharge and water quality protection. Coordination between these departments is all but perfect and the overlap of responsibility negatively influences management. Therefore, the Chinese central government has started reforming the current management system, trying to establish a unified institution responsible for the distribution of water, control of wastewater, and water supply. Currently, a unified institution is being created in each city below the province level. Such an institution has begun to work in several important cities, such as Shenzheng and Zhuhai.

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Chapter 17
Eutrophication of the Three Gorges Reservoir After Its First Filling

Sheng Zhang

Abstract Trophic status of water, including that of the main river channel, backwaters of tributaries, and thrity-five reservoirs on tributaries of the Three Gorges Reservoir basin was surveyed. In the main channel, concentrations of dissolved total nitrogen (DTN) and phosphorus (DTP) exceeded the low limit for eutrophication assessment, and the ratio of DTN to DTP was over 16:1. Chlorophyll *a* ranged from 1.58 to 7.53 mg/m^3 at the water surface. A trophic gradient from oligotrophic to mesotrophic was spatially observed. Eutrophy factors could be divided into three clusters in terms of flow direction. In twelve tributary bays of the Three Gorges Reservoir, the integrated trophic state index was 33.3–66.1. Of these tributaries, five were eutrophic in May, eight were eutrophic in June, and the rest were mesotrophic. The eutrophication is currently worse than before impoundment. In the thirty-five reservoirs studied on the tributaries, finally, it was found that nutrients and chlorophyll *a* were high while organic pollution and SD were low. Twenty-two reservoirs were eutrophic, one was oligotrophic, and the rest were mesotrophic. Our investigation found that water storage already had a strong impact on the eutrophication of the Three Gorges Reservoir after the first filling. Chinese scientists and water managers are therefore facing a highly unsatisfactory and challenging situation.

17.1 Introduction

Physical and chemical characteristics of rivers are influenced by a number of factors, including topography, geology, and climate (Gibbs 1970; Meybeck and Helmer 1989). Rivers have been substantially altered as a result of the construction of dams and reservoirs, canalization, and land-use development throughout their drainage basins (Puig et al. 1996). Many rivers have become polluted on a large scale (van

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Dijk et al. 1994). Water quality often declines after the construction of dams because of the change in hydraulic conditions and release of soil nutrients. Furthermore, in China, agriculture operates at a high fertilizer application rate. Large amounts of these are washed out and end up in streams and rivers (Wang et al. 2002). With increasing numbers of dams constructed across rivers, eutrophication has now become the main environmental problem for management of river water quality.

More specifically, damming of the Yangtze River has created one of the largest impoundments in the world: the Three Gorges Reservoir. Every year, about 2,000 ton of total phosphorus, 14,000 ton of total nitrogen, and 7,000 ton of ammonia and nitrogen are discharged to the Yangtze from cities and towns surrounding the Three Gorges Reservoir basin. About 130,000 ton of chemical fertilizers are used annually in this area. Crops consume only about 35% of these, while most of the nitrogen and phosphorus finds its way to the reservoir, becoming one of its most important pollution sources. In general, phytoplankton growth is limited by the velocity of water flow. In the case of high flow velocity, phytoplankton cannot effectively utilize nutrients. Before June 2003, flow velocity in the Three Gorges Reservoir was about 2–3 m/s. After completion of the project in 2009, flow velocity had slowed to below 1 m/s. Even with the loads of nitrogen and phosphorus remaining unchanged, the water quality suddenly declined. From monitoring data obtained from 1990 to 2002, maximal TN and TP concentrations used to be about 2 and 0.2 mg/L, respectively, much higher than the critical concentration for lake eutrophication. Yet, chlorophyll *a* concentration did not reach a eutrophic level because of the flow velocity and turbidity. Eutrophication set in after the construction of the dam, since suspended solids rapidly deposited at low flow velocity and increasing transparency (Zhang et al. 2005). Temperature of surface water increased in winter all factors that favor the growth of phytoplankton. Water blooms occurred in some tributaries after impoundment, dominated by *Microcystis*. This chapter describes the eutrophication of the Three Gorges Reservoir after its first filling in 2003 and 2004 based on three investigations conducted in the main channel of the reservoir, in 12 tributaries, and in 35 reservoirs on these tributaries.

17.2 Climate and Pollution

The Three Gorges Reservoir is located in the upstream zone of the Yangtze River. At the end of 2003, the total human population in the Three Gorges Reservoir basin was 19.85 million, among whom 14.14 million worked in agriculture.

In 2003, the average temperature in the Three Gorges Reservoir area was 18.1°C, about 0.6°C higher than the long-term average. Rainfall was relatively high in winter and spring and low in summer and autumn (Table 17.1). The average rainfall in the reservoir area was 1,184 mm, nearly 10% above average.

In 2003, a total of 54 sources directly discharged 184 million tons of wastewater into the Yangtze River. The largest amount of industrial wastewater discharge was from Chongqing Municipality (including districts of Ba'nan, Dadukou, Jiulongpo, Nan'an, Yuzhong, Jiangbei, and Yubei), with a discharge of 60.119 million tons,

Table 17.1 Monitoring of meteorological elements in Three Gorges Reservoir area

Zone	Average temperature (°C)	Comparative humidity (%)	Rainfall (mm)	Foggy days (day)	Sunshine hours (h)
Chongqing	18.9	80	1,033.2	25	878.5
Changshou	18.0	81	1,078.6	40	1,128.0
Fuling	18.7	79	1,168.1	79	1,144.4
Wanzhou	18.7	80	1,461.2	19	1,272.5
Fengjie	18.1	74	1,366.0	5	1,356.4
Wushan	18.5	73	1,179.5	5	1,358.3
Badong	17.1	73	1,113.5	69	1,239.8
Zigui	17.9	78	1,014.8	0	1,429.2
Bahekou	16.6	81	1,220.7	0	1,025.8
Yichan	16.8	79	1,240.6	19	1,078.7

accounting for 32.8% of the total. The 54 main pollution sources discharged a total of 25,000 ton of pollutants. Among these, COD amounted to 24,087.5 ton and ammonia nitrogen 848.9 ton. COD and ammonia nitrogen contributed up to 78.0% and 18.3%, respectively, to the pollution loading. The major industries of the pollution sources included chemical materials and manufacturing, foodstuff, tobacco and beverage, chemical fiber manufacture, coal and gas. Their wastewater accounted for 81.0% of the total discharge.

In 2003, there were 66 outlets that discharged urban wastewater directly to the Yangtze River, with a total discharge of 404 million tons, which was 26.6% more than that in 2002. The wastewater from Chongqing Municipality (253 million tons), Wanzhou District (43 million tons), and Fuling District (and 28 million tons) accounted for 62.6%, 10.6%, and 6.9%, respectively, of the total wastewater. The total pollutants in the urban wastewater amounted to 261,300 ton (COD: 157,700 ton; BOD: 76,800 tons; ammonia nitrogen: 9,700 ton). The major pollutants were total phosphorus, BOD and COD, with a pollution load of 47.3%, 24.9%, and 15.4%, respectively.

In 2003, the total chemical fertilizers in the reservoir basin comprised 77,900 ton of nitrogen fertilizer, 22,000 ton of phosphorus fertilizer, and 10,300 ton of sodium fertilizer. The ratio of nitrogen, phosphorus, and sodium was 1:0.28:0.13, which is considered an overuse of nitrogen and phosphorus. This resulted in excessive loss of nitrogen and phosphorus that finally pollutes the waters of the Yangtze River. In 2003, pesticides used in the reservoir area amounted to 645.37 ton, which includes 399.2 ton of organic phosphorus and 81.75 ton of organic nitrogen.

17.3 Material and Methods

17.3.1 Sampling

According to hydraulic conditions, 11 monitoring sites (Fig. 17.1) were set up in the main channel along the Three Gorges Reservoir, extending some 430 km down to

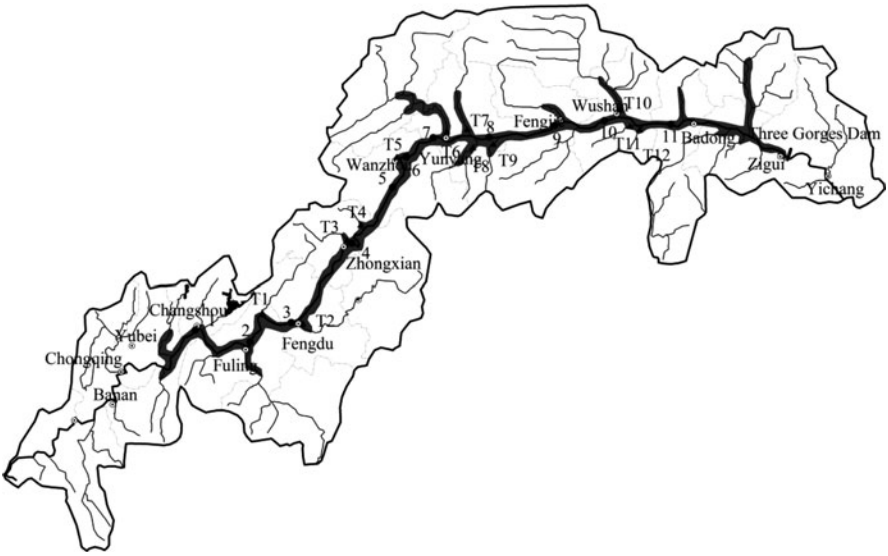

Fig. 17.1 Investigation sites or sections in the main channel and tributaries of the Three Gorges Reservoir

the dam. In this study, the cruise distance was about 380 km. A sampling campaign by ship was launched during September 8–15, 2003. At each sampling site, water samples were collected in the mid section at 0.5 m below the water surface.

Twelve tributaries of the Three Gorges Reservoir were sampled and monitored in May and June, 2004 (Fig. 17.1). A sampling section was set in each backwater of each of these tributaries and samples were collected at 0.5 m depth. When several sampling sites are located on one section, average values for the variables were calculated.

There are also 35 reservoirs on the tributaries of the Three Gorges Reservoir, with a water storage capacity exceeding 10×10^6 m^3. Most of these reservoirs are for drinking water supply, agricultural irrigation, and aquaculture. All samples from these reservoirs were collected from the surface (0.5 m) in the riverine, transition, and lacustrine zones in 2003.

17.3.2 Sample Analysis

Turbidity, water temperature, pH, dissolved oxygen, and conductivity were measured with a portable monitoring instrument (Type 69202 41). Flow velocity was measured with a Sontek ADP (Acoustic Doppler profiler RS-1000).

All samples were filtered through filter paper (What man GF/C: 0.45 mm in pore size, 5.0 cm in diameter) under vacuum before analysis. Dissolved total nitrogen

(DTN) was determined by UV spectrophotometry after digestion by alkaline $K_2\ S_2O_8$. Dissolved total phosphorus (DTP) was analyzed using the molybdate method after digestion by $K_2\ S_2O_8$. Potassium, nitrite (NO_2–N), and nitrate (NO_3–N) were measured by ion chromatography. Ammonium nitrogen (NH_3–N) was determined using the hypochlorite method (Liu 1990). TOC analysis was conducted with an O.I Analytical TOC-1010.

For measurement of chlorophyll a (Chla), 1 L of water sample was filtered through filter paper (Whatman GF/C, 0.45 mm in pore size, 5. 0 cm in diameter) using a vacuum filtration assembly. Filter paper (with residue) was crushed in a mortar and transferred to centrifuge tubes, followed by repeated washing with acetone. Centrifugation was conducted for 10 min at 5,000 rpm (>500 G). Optical density (OD) of the supernatant was measured at 664 and 750 nm using a 1-cm path-length cell. The extract was acidified with two drops of 0.1N HCl, gently agitated, and was OD was measured between 1 and 2 min after acidification at 665 and 750 nm using the same cell. A 90% aqueous acetone solution was used as the blank. Chlorophyll *a* in mg/m^3 was estimated as {26.7 ($OD664_b - OD665_a$) A}/B, where A and B are the total volume of acetone extract and the original sample, respectively, and $OD664_b$ and $OD665_a$ are the turbidity-corrected OD values before and after acidification using 1-cm cells. The turbidity-corrected OD values were obtained by subtracting the OD values at 750 nm from the OD values at 664 and 665 nm.

17.3.3 Trophic State Index

Trophic state was assessed from Chla, TN, TP, COD, and SD. A synthetic trophic state index was used for the 26 lakes and reservoirs. The computational formula of the index was:

$$TLI(\Sigma) = \sum_{j=1}^{m} W_j \cdot TLI(j)$$

where $TLI(\sum)$ is the integrated trophic state index, W_j the correlation weight of factor *j*, and $TLI(j)$ is the trophic state index of factor *j*, Its computational forms is as follows:

$$W_j = \frac{r_{ij}^2}{\sum_{j=1}^{m} r_{ij}^2}$$

where r_{ij} is the correlation coefficient of a benchmark parameter (Chlorophyll *a*) and *j*, and *m* is the number of parameters (Table 17.2).

Table 17.2 Correlation coefficients between chlorophyll *a*, TP, TN, SD, and COD

Parameter	Chla	TP	TP	SD	COD
r_{ij}	1	0.84	0.82	−0.83	0.83
r_{ij}^2	1	0.7056	0.6724	0.6889	0.6889
W_j	0.2663	0.2237	0.2183	0.2210	0.2210

The r_{ij} results come from the investigation of 26 lakes in China

The calculation formulae of the trophic state index of Chla TP, TN, SD, and COD were:

$\mathrm{TLI(Chla)} = 10(2.5 + 1.086\ln\mathrm{Chla}(\mathrm{mg/m^3}))$,
$\mathrm{TLI(TP)} = 10(9.436 + 1.624\ln\mathrm{TP}(\mathrm{mg/L}))$,
$\mathrm{TLI(TN)} = 10(5.453 + 1.694\ln\mathrm{TN}(\mathrm{mg/L}))$;
$\mathrm{TLI(SD)} = 10(5.118 - 1.94\ln\mathrm{SD}(\mathrm{m}))$,
$\mathrm{TLI(COD)} = 10(0.109 + 2.661\ln\mathrm{COD}(\mathrm{mg/L}))$.

17.4 Results and Discussion

17.4.1 Main Channel of the Three Gorges Reservoir

In the reservoir, flow velocity ranges from 1.11 to 1.76 m/s, which is lower than that in the backwaters (2.29–2.56 m/s). Turbidity gradually decreases in the flow direction. Water temperature ranged from 21.1 °C to 22.8°C and conductivity from 144 to 174 µS/cm. The saturation rate of dissolved oxygen exceeded 90% at all monitoring sites. Oxygen exchange between surface water and atmosphere was not affected by the construction of the dam (Table 17.3).

DTN concentration ranged from 1.01 to 1.35 mg/L with an average of 1.15 mg/L, exceeding the national surface water quality standard for Class III (1.00 mg/L). Dissolved total phosphorus concentration ranged from 0.028 to 0.054 mg/L with mean 0.040 mg/L, which is lower than before damming (Liu 2000; Lü 2002). Dissolved inorganic nitrogen (DIN) ranged from 0.93 to 1.41 mg/L with a mean of 1.12 mg/L, with nitrate nitrogen (NO_3–N) dominant (accounting for 85% of DIN). Nitrate nitrogen concentration ranged from 0.76 to 1.27 mg/L with mean 0.96 mg/L. Ammonium nitrogen (NH_3–N) had a mean concentration of 0.14 mg/L, and accounted for about 13% of inorganic nitrogen. Total organic carbon varied from 1.92 to 2.59 mg/L with mean 2.17 mg/L. There was no regularity in the spatial distribution of DIN. Gradients in trophic state along a given reservoir, based on changes in chlorophyll *a* and total phosphorus concentrations, have been well documented (Perkins and Under wood 2000; Jones et al. 1990; Thornton et al. 1982). Such a gradient usually occurs in locations where the water column depth increases, far away from the main external input (Vrba et al. 1995). Usually, phosphorus is the factor limiting algal growth in freshwater systems (OECD

Table 17.3 Sampling sites and water characteristics

Monitoring site	Location	Water parameter					
		Average flow velocity (m/s)	Water temperature (°C)	DO (mg/L)	pH	Conductivity (μS/cm)	Turbidity
1	107°6′5.7″E;29°45′5.9″N	2.29	22.3	8.36	8.22	167	458
2	107°2′46.7″E;29°43′24.5″N	2.56	21.1	9.05	8.31	159	449
3	107°4.3′21.1″E;29°52′50.6″N	2.33	21.5	8.56	8.28	145	451
4	108°8′7.8″E;30°18′7.8″N	1.76	21.9	8.44	8.35	167	446
5	108°23′28.7″E;30°44′17″N	1.18	21.9	8.09	8.21	183	397
6	108°26′38.7″E;30°50′6.3″N	1.24	22	7.93	8.17	147	375
7	108°3.8′42.1″E;30°53′29.9″N	1.21	21.9	7.95	8.16	174	369
8	108°5.1′23.5″E;30°57′15.6″N	1.11	22.0	8.44	8.15	144	362
9	109°34′9.0″E;31°2′30.6″N	1.19	22.3	8.77	8.20	151	350
10	109°5′47.8″E;31°5′47.6″N	1.43	22.8	8.73	8.22	173	265
11	109°55′2.9″E;31°1′42.5″N	1.47	22.6	8.32	8.10	162	285

Table 17.4 Concentrations of chemical elements in TGR

Monitoring site	DTP, mg/L	DTN, mg/L	NO_2–N, mg/L	NO_3–N, mg/L	NH_3–N, mg/L	DN, mg/L	K, mg/L	TOC, mg/L
1	0.046	1.12	0.02	0.76	0.15	0.93	3.46	1.94
2	0.039	1.06	0.03	0.93	0.11	1.07	2.8	2.23
3	0.039	1.17	0.02	0.85	0.21	1.08	3.32	2.13
4	0.033	1.01	0.02	0.86	0.12	1	3.44	1.92
5	0.038	1.16	0.02	1.07	0.11	1.2	4.36	2.07
6	0.054	1.1	0.02	0.92	0.09	1.03	3.43	2.32
7	0.045	1.15	0.02	1	0.13	1.15	4.14	2.32
8	0.037	1.24	0.02	0.94	0.17	1.13	3.11	2.59
9	0.047	1.06	0.02	0.95	0.2	1.17	3.3	2.2
10	0.035	1.35	0.02	1.27	0.12	1.41	3.56	2.17
11	0.028	1.23	0.02	1.03	0.14	1.19	3.01	1.94

1982). Evidence for phosphorus limiting algal biomass was confirmed from the ratios of DIN/DTP at each site, which ranged from 19 to 42 at all sampling sites, and which exceeded the Redfield's N:P ratio of 16:1 (Redfield 1958) (Table 17.4).

Chlorophyll *a* concentration ranged from 1.58 to 7.53 mg/m^3 with mean 4.69 mg/m^3. It significantly increased in the flow direction, and the highest concentration, observed near the dam (Site 11) was about five times higher than that at Site 1. A trophic gradient occurred from oligotrophy to mesotrophy in the main channel (classified by chl *a* concentration, after OECD 1982). At the Three Gorges Reservoir, the oligotrophic zone included Changshou, Fuling, Fengdu, and Zhongxian (sites 1, 2, 3, 4). The mesotrophic zone included Wanzhou, Yunyang, Fengjie, and Wushan (Sites 5–11). Compared with the chlorophyll *a* concentration before the reservoir started to fill, it was about three times higher between Wanzhou and Wushan (Yangtze Hydraulic Committee 1997). The monitoring data showed only a slight changeup stream, however. Modeling carried out before damming predicted

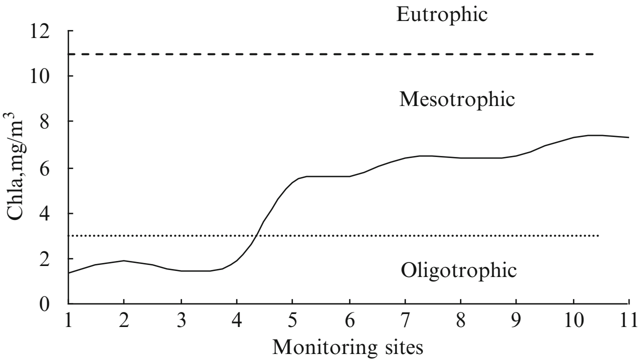

Fig. 17.2 Variation of chlorophyll *a* concentrations in the flow direction

Table 17.5 Correlations among chlorophyll *a*, nutrients, and turbidity (Significance level: $*P < 0.05$; $**P < 0.01$)

	DTP	DTN	NO_2–N	NO_3–N	NH_3–N	K	TOC	Chl *a*	Turbidity
DTP	1								
DTN	−0.3804	1							
NO_2-N	−0.0493	−0.3057	1						
NO_3-N	−0.3395	0.6939*	−0.0787	1					
NH_3-N	−0.0359	0.0565	−0.2675	−0.3098	1				
K	0.2075	0.0830	−0.4705	0.2823	−0.2506	1			
TOC	0.3459	0.2246	0.1048	0.1402	0.0847	−0.0764	1		
Chl *a*	−0.0634	0.5648	−0.3794	0.7287*	−0.0973	0.1996	0.3820	1	
Turbidity	0.2592	−0.7056*	0.3324	−0.8104*	0.0843	−0.0203	−0.1776	−0.9207**	1

that no eutrophication would occur in the central water, but this was not confirmed by our monitoring of chlorophyll *a* concentration (Fig. 17.2).

Statistically significant positive correlations were found between chlorophyll *a* and NO_3–N ($p < 0.05$, $r = 0.728$, Table 17.5). Negative correlations occurred between chlorophyll *a* and turbidity, again statistically significant ($p < 0.01$, $r = -0.9207$). Turbidity was associated with transparency, which affected autotrophic production. The extent to which water transparency exerts an influence depends on the properties of the water body and the adaptation of the photosynthetic apparatus of phytoplankton to their living conditions (Burford and Rothlisberg 1999). At the initial stage when the reservoir started to operate, the increase of phytoplankton biomass indicated by chlorophyll *a* may have been affected by hydraulics conditions such as flow velocity and transparence.

We used cluster analysis to study what affects the trophic factors. The monitoring sites (Fig. 17.3) could be divided into three clusters: upstream (sites 1, 2, 3, 4), mid-stream (sites 5, 6, 7, 8, 9), and downstream (Sites 10 and 11). The analysis further showed that hydraulic conditions were the key factor controlling phytoplankton biomass at the initial stage, when the reservoir was formed.

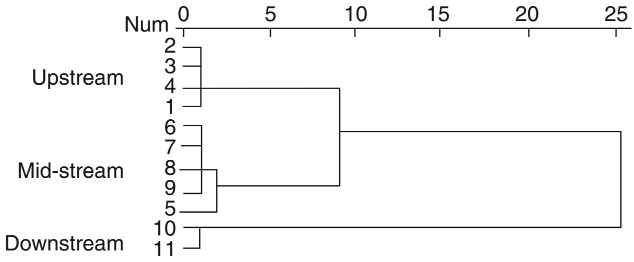

Fig. 17.3 Results of cluster analysis of monitoring sites numbered in Fig. 17.1

17.4.2 Backwaters of Tributaries

Chemical and physical variables in the backwaters of the main channel of the Three Gorges Reservoir are shown in Table 17.6. Water temperature was 19.0–26.0°C, DO saturation varied between 64% and 114%, and conductivity ranged from 166 to 705 μs/cm. Because of chemical industry discharging salts, conductivity was high in Zhuxi river. SD varied in a wide range between 20 and 350 cm. The catchment of the Three Gorges Reservoir is one of the regions with the most serious soil erosion in China; SD in each tributary was always much lower in June than in May because of heavy rainfall.

Change in trophic state variables is shown in Fig. 17.4. COD_{Mn} indicates organic pollution above 4 mg/L, because of the presence of organic pollution. COD_{Mn} ranged from 1.55 to 5.88 mg/L and was below 4 mg/L in all tributaries except Zhuxi. Thus, these backwaters are not yet seriously affected by organic pollution. TN ranged from 0.535 to 7.47 mg/L, and NH_3–N from 0.042 to 1.40 mg/L. The catchment of the Three Gorges Reservoir comprises mainly agricultural areas, while the basin of tributaries is densely populated. Higher nitrogen concentrations in water came from discharge of urban sewage and rural chemical fertilizers. TP ranged from 0.016 to 0.835 mg/L. TN and TP in the waters studied were higher than the lower limit for eutrophication and sufficient for explosive algal growth. The ratios of TN/TP were 3.57–34.5. Except in Zhuxi, all tributaries exceeded 10, with four tributaries exceeding 21. This showed that P, not N, was limiting algal growth in these backwaters. Chlorophyll *a* ranged from 1.38 to 23.7 mg/m^3. According to OECD eutrophication evaluation criteria on Chlorophyll *a*, Huangjin, Ruxi, Zhux, and Pengxi reached the level of eutrophication. The rest of tributaries were oligotrophic-mesotrophic.

Backwaters of tributaries are complex. Their ecosystem is affected by both their own upper reaches and by the main channel (Zhang et al. 2008). Correlations between chlorophyll *a* and other factors were not like in most lakes and reservoirs (Table 17.7): only the correlation between Chla and COD_{Mn} was significant ($r = 0.6242$, $p < 0.01$) and there was no relationship between chl *a* and SD. When the Three Gorges Reservoir stored water to a level of 135 m, water velocity in the backwater area slowed down, but water exchange between the tributaries and the main stream was still strong. However, after the Three Gorges Reservoir had

Table 17.6 Physical chemistry variables

Tributary	Water temperature, °C		pH		DO, mg/L		SD, cm		Conductivity, ms/cm	
	May	June	May	June	May	June	May	June	May	June
Quxi	20.2	22.0	7.32	7.47	7.50	7.10	30	30	331	329
Longhe	19.0	19.0	7.34	7.17	7.55	7.10	30	20	251	166
Huangjin	23.5	18.2	8.66	7.76	7.38	7.83	80	60	267	258
Ruxi	25.3	18.6	8.69	7.64	7.58	6.70	100	70	329	296
Zhuxi	19.7	20.6	8.18	8.15	6.45	7.95	70	50	705	470
Pengxi	20.0	24.5	8.64	8.06	5.80	7.65	80	10	519	320
Tangxi	20.0	24.5	8.41	7.60	7.40	7.50	100	10	400	423
Modaoxi	19.5	24.5	7.31	8.02	6.05	7.55	100	10	460	393
Changtan	19.0	24.5	7.88	8.04	6.15	7.60	100	10	484	305
Danin	25.0	21.0	8.04	8.26	9.44	8.51	300	60	408	335
Shennvxi	23.0	26.0	7.74	8.29	6.94	7.51	200	50	356	345
Baolong	22.0	25.0	7.84	8.38	7.12	9.01	350	50	356	345

started operating, at a level of 175 m, water velocity became similar to natural lakes and chlorophyll *a* concentration tended to correlate with nutrients like in lakes and large deep reservoirs.

With the distance in cluster analysis set to 0.62, 12 tributaries were grouped into four categories. Tributaries with high concentration of NH_3–N were Quxi and Longhe. Low nutrients and organic matter concentrations were found in Pengxi, Shennvxi, Baolong, Modaoxi, Daning, Tangxi, and Changtan. High concentrations of CODMn were present in Huangjin and Ruxi. Zhuxi River was separated as one class, being high both in nutrients and organic matter (Fig. 17.5).

The integrated trophic state index of the backwaters ranged from 33.3 to 66.1, and was higher than in the main channel. It was also higher than before impoundment, when the index had ranged from 23.56 to 58.25 (Zhang et al. 2007). The trophic state of the tributaries increased with water retention time and decreasing velocity of water flow. Trophic state index was also higher in June than in May. Eight tributaries were eutrophic in June and five were eutrophic in May. This indicates that eutrophication is related, inter alia, to climate (Fig. 17.6).

17.4.3 Reservoirs in Tributaries

Most investigated reservoirs had extremely high concentrations of TN, TP, and Chla, and low SD (see Fig. 17.7). Their concentration of TP ranged from 0.007 to 0.527 mg/L with mean 0.102 mg/L. TN ranged from 0.52 to 5.941 mg/L, with a mean of 1.56 mg/L. There were 25 reservoirs in which the ratio TN: TP exceeded 16:1, while 10 reservoirs were below 16:1. Thus, about 70% of all reservoirs were limited by phosphorus and about 30% of reservoirs by nitrogen. There were 18 reservoirs in which COD was below 4 mg/L, and 27 with values from 4 to 10 mg/L. Water transparence ranged from 0.5 to 3.2 m, and 31 reservoirs had a transparency below 2 m; these accounted for nearly 90% of all investigated reservoirs. Following

Table 17.7 Correlation coefficients between chlorophyll *a* and trophic factors (Significance level: $*P < 0.05$)

	Chla	Water temperature	pH	COD_{Mn}	TN	TP	SD	NH_3–N
Chla	1							
Water temperature	−0.0468	1						
pH	0.5373	0.43604	1					
COD_{Mn}	0.6242*	−0.2557	0.1870	1				
TN	0.31156	−0.07315	0.08814	0.61262*	1			
TP	0.12329	−0.17167	0.13034	0.69833*	0.53186	1		
SD	0.09476	0.08006	0.09259	−0.06509	−0.2637	−0.13818	1	
NH_3–N	0.11012	−0.31068	−0.2395	0.60298	0.86506*	0.46247	−0.29994	1

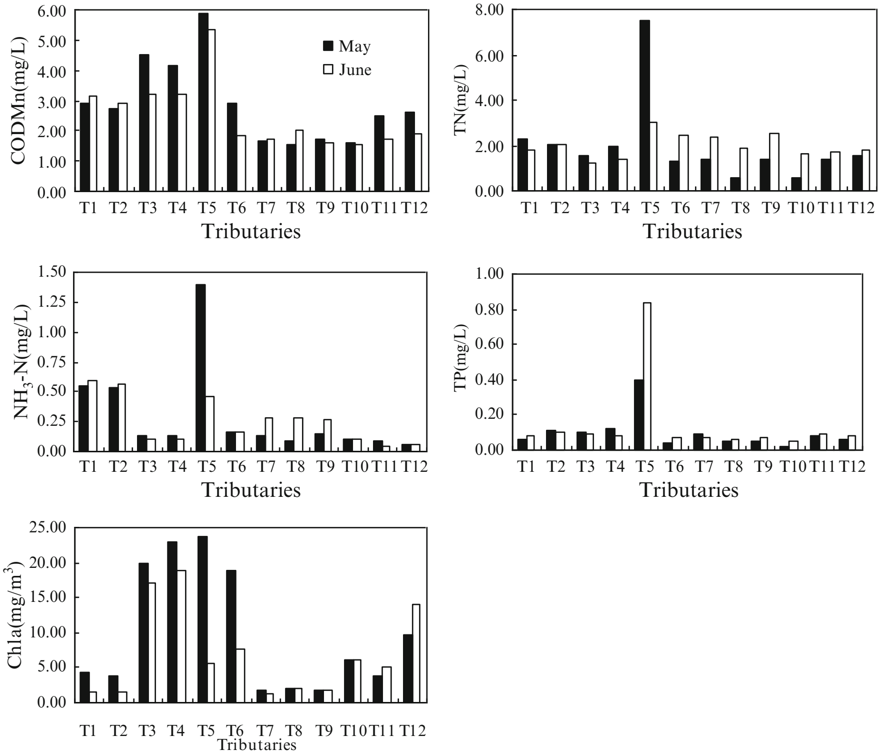

Fig. 17.4 Change of trophic state factors

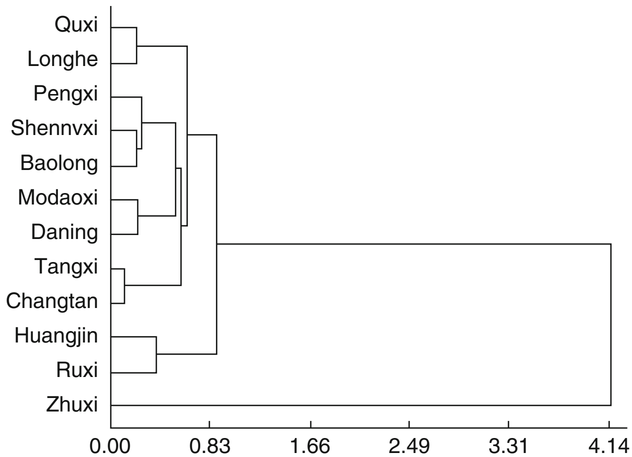

Fig. 17.5 Cluster analysis of tributaries based on nutrients

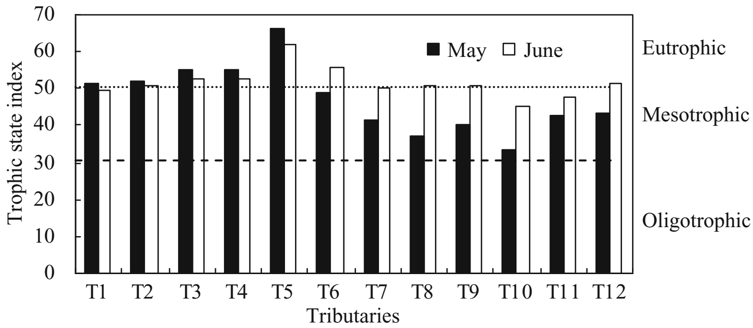

Fig. 17.6 Assessment of the trophic state of 12 tributaries

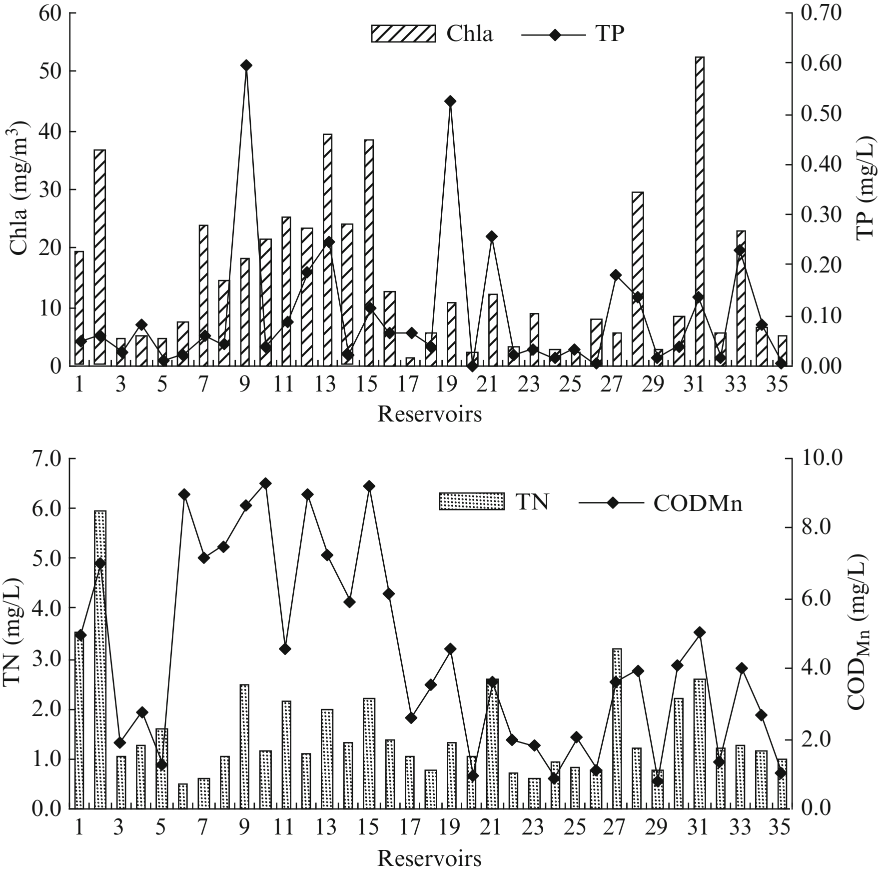

Fig. 17.7 Monitoring results of water quality in reservoirs

the critical chlorophyll *a* concentration for eutrophic water (>11 mg/m^3) suggested by OECD, 16 reservoirs were eutrophic. Only five reservoirs were oligotrophic, with a chlorophyll *a* concentration of <3 mg/m^3.

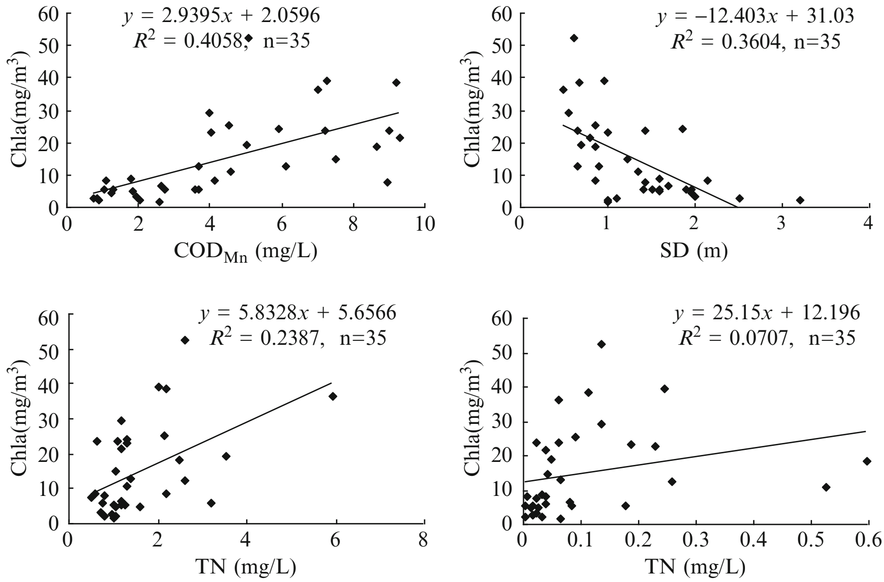

Fig. 17.8 Relationships between chlorophyll *a* (Chla) and TP, TN, SD, and COD_{Mn}

Since Dillon and Rigler (1974) published a log-log model relating lake-water chlorophyll *a* to total P, lake managers have been using such empirical models to predict algal biomass responses to P reduction. However, chlorophyll-P models can vary widely, depending on features such as grazing pressure on the algae, presence or absence of abiotic turbidity, lake flushing rate, and a number of other variables. Canfield (1983) developed a more complex model to predict chlorophyll *a* that considers both P and N, but it was found that both the Dillon-Rigler and the Canfield model tended to over-estimate chlorophyll *a* in lakes (Havens et al. 2001). In natural lakes, increase of phytoplankton reduces water transparency. Based on our monitoring of 35 reservoirs in Chongqing, chlorophyll *a* and COD were positively correlated (Chla $=$ 2.94 (COD) $+$ 2.06, $R^2 = 0.401$, $p < 0.05$). Chlorophyll *a* and water transparency were also correlated (Chla $= -12.4$ (SD) $+$ 31.03, $R^2 = 0.3604$, $p < 0.05$) (Fig. 17.8).

The synthetic trophic state index indicated that 22 reservoirs were eutrophic (Fig. 17.9), and only one reservoir was oligotrophic. Of these, 28 reservoirs are used to supply drinking water, and almost all of them were eutrophic.

17.5 Water Blooms

In the spring and summer of 2004, several water blooms occurred in tributaries of the Three Gorges Reservoir that are in Class I of the Chinese national standard of water quality. For example, water blooms happened in Xiangxi River in late

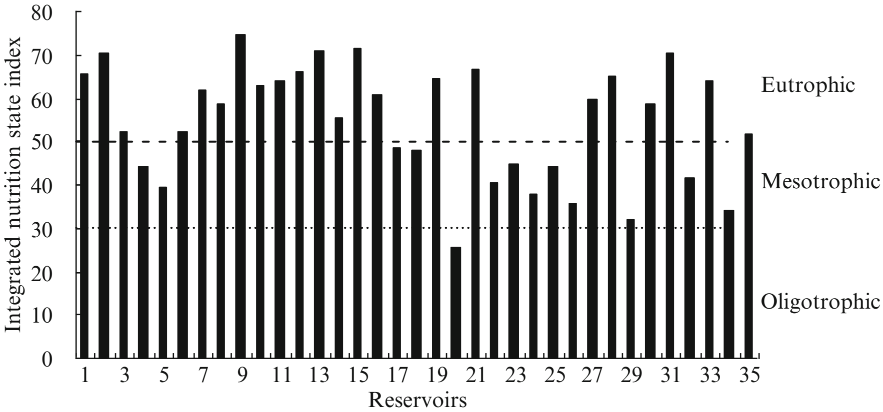

Fig. 17.9 Trophic state level of 35 reservoirs in tributaries of the Three Gorges Reservoir

February, mid March, early April, and early June, lasting for 5 days, 1 month, 1 week, and 10 days, respectively, and covering a distance of about 20 km. The water was turbid with a color of soy sauce. The phytoplankton was dominated by *Cyclotella* sp. and *Asterionellopsis* sp.

Water blooms occurred at Daning River in late March to early April, late May, early June, and late June, and lasted for about 10 days in total. The water blooms in Shennü Brook were concentrated in the river section from Bawu Gorge to the Daning River mouth, Shuanglong–Yinwotan–Longmen River section, and the section from Maduhe River to its mouth, a distance of nearly 25 km. The water body appeared in a color of light soy sauce and had a fishy smell. *Cyclotella* sp, *Asterionellopsis* sp, *Peridinium*, *Pandorina*, and *Microcystis* were the dominant algae.

Water blooms in both Shenn Brook and Baolong River occurred from late May to early June and late June, each lasting for about 10 days. The occurrence in Shennü Brook ranged from Daoche Dam to No. 6 navigation mark, about 1.5-km long. The dominant algae were *Pyrrophyta*, *Cyclotella* sp., and *Pandorina*. Water blooms in Baolong River occurred in the section between Hongyan River and Putao dam, over a length of about 2.5 km. The water turned chartreuse in color, with a fishy smell. The dominant species was *Microcystis* sp.

In early March, Fenghuangshan Reservoir had a water bloom lasting for about 1 week. The water body looked like soy sauce and the dominant algae were *Asterionellopsis* sp. and *Peridinium* sp.

17.6 Conclusions

Our investigation found that water storage already had a strong impact on the eutrophication of the Three Gorges Reservoir at the water level of 135 m. The Three Gorges Reservoir was designed to store water at three stages. Water level was gradually increased to 135 m in 2003, 156 m in 2006, and 175 m in 2009. After the completion of the Three Gorges Project, the water level indeed reached 175 m and water flow decreased markedly. Average flow velocity fell to 0.17 m/s, about four times less than before storing water. At the front of the dam, flow velocity dropped to 0.04 m/s. Both area and length of backwater increased in the numerous tributaries whose bed became flooded.

There are about 3,000 pollution sources in and near Chongqing City only. Its total industrial and urban wastewater discharge amounts to about 10^8 ton/year, and contains much N and P. China's agriculture is conducted at a high fertilizer application rate in order to support its large population. Large amounts of this fertilizer are not efficiently utilized, however, and enter into the runoff, destroying water quality. In developing countries such as China, such point sources constitute a big threat. A survey showed that N and P fertilizer uptake rate by agriculture in this area was only about 35% of the total and that the amount remaining in the soil was 34.5%. Therefore, much N and P in the surface soil is eventually washed out by runoff water. It is therefore easy to understand why the nutrient output from tributaries increased with increasing water level of the Three Gorges Dam. Chinese scientists and water managers are therefore facing a highly unsatisfactory and even dangerous situation!

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Chapter 18
Eutrophication of a Pumped Water Storage Reservoir in South China

Bo-Ping Han and Zhengwen Liu

Abstract Dajinshan reservoir, built in 1974 for the purpose of supplying drinking water, is a pumped water storage tropical reservoir. With a small volume of 10×10^6 m^3, the stored water is insufficient to meet water consumption. Most of the stored water is pumped from a polluted river into the reservoir at the dam in the dry season. The specific pattern of importing and exporting water leads to an abnormal variation in water level that is high in the dry season and low in the wet season. In order to understand the trophic state dynamics of this special reservoir, a high-frequency sampling with an interval of 15 days was conducted for analysis of water quality and phytoplankton abundance at four sites in 2005. As the water abstraction actually functions as a point source of pollutants, the dynamics of nutrients and chlorophyll *a* concentration reveal a pattern that follows the water level. Total phosphorus and total nitrogen concentrations ranged from 0.016 to 0.086 mg/L and 0.5 to 2.0 mg/L, respectively, with a maximum in the dry season. The mean weighed trophic state index, combining TN, TP, Chlorophyll *a* and Secchi depth, was between 30 and 55, indicating that the reservoir was mesotrophic in the wet season and eutrophic in the dry season. The water quality exhibited a visible vertical gradient near the dam, with higher concentrations of total nitrogen and total phosphorus near the bottom. This vertical gradient implies that internal loading may be important for eutrophication. Phytoplankton abundance had a seasonal variation similar to that of nutrients. It ranged from 0.86×10^6 to 106.27×10^6 cells/L, with peaks in April and in November, respectively. The phytoplankton was dominated by diatoms and Cyanobacteria in the mixing period (winter and early spring), and by Cyanobacteria during stratification only, at which time they constituted 90% of the total abundance, however. Compared with an early survey in 1999 and 2000, water quality shows a rapid deterioration. In order to maintain high water quality for drinking water, measures against eutrophication are

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urgently required. Dredging of the sediment and reduction of nutrients in the abstracted water are two priority strategies.

18.1 Introduction

In southern China, large numbers of reservoirs have been constructed in the 1960s–1980s, aimed at flood control, irrigation, and power generation. After China initiated its open policy in the early 1980s, rapid economic development and demographic expansion rapidly created environmental stress on water quality, especially of river water. Consequently, reservoirs became an important source of drinking water (Han et al. 2003). For example, although all reservoirs of Zhuhai city are small-sized (Fig. 18.1), they are connected in a network system for drinking water supply. Several small reservoirs of this network have the additional responsibility to supply water to Macau when water is in extreme shortage. Located in a coastal region, the rivers feeding these reservoirs are short and have a low natural discharge. As a result, the water naturally stored in the reservoirs is not sufficient for local demand. In particular, saline water periodically moves up the estuary of Pearl River in dry seasons. Therefore water has to be pumped from the Pearl River or small local rivers to maintain the water storage in the reservoirs. However, the rivers near the city have become heavily polluted by domestic and industrial

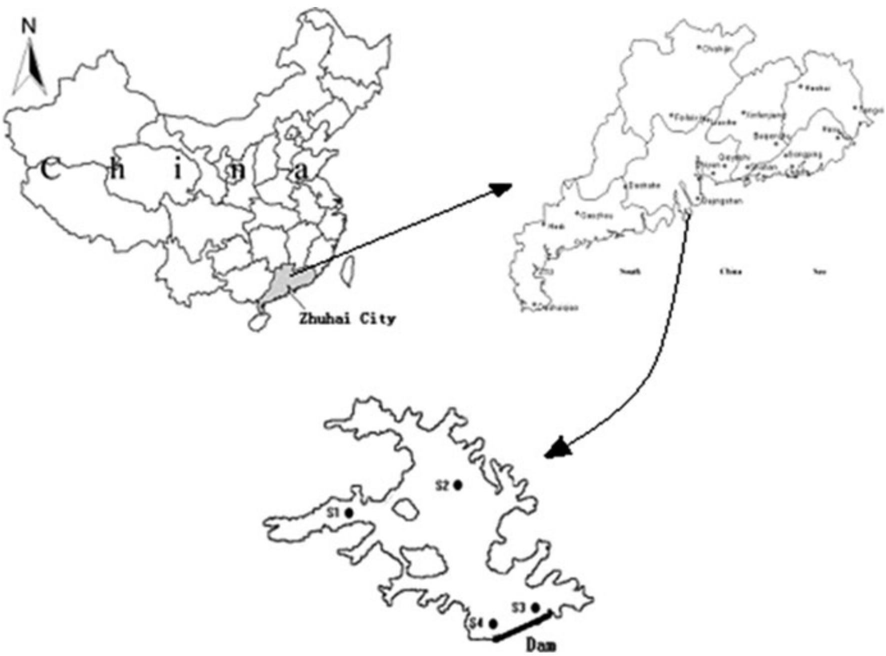

Fig. 18.1 Location and map of Dajingshan Reservoir and the four sampling sites

effluents, and they carry high nutrient concentrations. Pumping water from such rivers certainly accelerates eutrophication of the reservoirs. Dajinshan reservoir is one of the most important water bodies in Zhuhai City, with a normal volume of 10×10^6 m^3, supplying about 2×10^6 m^3 drinking water per month. In the last two decades, most of its water was abstracted from the Qianshan River, a heavily polluted freshwater river. As river water is pumped mainly in the dry season when precipitation is absent, water level is low in the wet season and high in the dry season. In 1999 and 2000, an early survey was conducted for the assessment of reservoir eutrophication. The chlorophyll *a* concentration was always below 20 mg/m^3, and the reservoir was classified as mesotrophic (Han et al. 2003). Since 2002, however, algal blooms have been frequent near the dam. To understand the possible development and seasonal dynamics of its trophic state, a detailed study was set up, with an aim to provide all information necessary for water quality management. In this study, we outline the seasonal characteristics of eutrophication in the reservoir through high-frequency monitoring.

18.2 Materials and Methods

Dajinshan reservoir is located near the center of Zhuhai City, about 2 km from the beach of the South China Sea. It was first filled in 1974, has a maximal water volume of 12×10^6 m^3, a rather small catchment of 5.95 km^2, a maximal length of 3.76 km, and a maximal depth of 12.2 m. Three water tubes of 1.5 m in diameter each, crossing the dam, were installed for water pumping and supply, distributed on the left, middle and right side. The middle one is mainly used for water abstraction from rivers, and the others for water supply. There is only a small and short river in the upper reach, which dries up in dry seasons. There is no non-point pollution, and all nutrients are loaded to the reservoir via water pumping. Atmospheric deposition is considered to be marginal.

From January to December in 2005, water and phytoplankton were sampled at four samplings sites from the dam to the upper streams with a gradient in morphology and hydrodynamics, namely S1 (the end site), S2 (the central point), S3 (close to the mouth of water supply), and S4 (at the water-pumping inlet). At S4, vertical samples were obtained at depths of 0.5, 2, 4, and 6 m. Water samples were collected biweekly by a 5-L water sampler for analysis of nutrients and chlorophyll *a*. Total nitrogen and total phosphorus were measured according to the Chinese National Standard for eutrophication of inland waters (Jin and Tu 1990). Water temperature was measured in situ by a YSI sensor (YSI 85). Water transparency was measured by Secchi disk. Trophic status index (TSI) was calculated from the relationships between chlorophyll *a* and SD, TN and TP from reservoirs in Guangdong Province (Lin et al. 2003b; Han et al. 2003). Phytoplankton samples were fixed in situ in 5% formalin. Phytoplankton was counted and measured under a microscope, and the results are expressed with an accuracy of 10% of the total concentration as cell abundance. Chlorophyll *a* was measured after overnight extraction in 90% acetone, following Lin et al. (2005). Hydrological data were provided by the administrative department of Dajingshan reservoir.

18.3 Results

18.3.1 Hydrology and Climate

Like most tropical reservoirs in Asia, the hydrology of Dajinshan reservoir is highly influenced by summer monsoon. The natural inflow from precipitation occurs mainly from April to September. Annual precipitation is about 2,000 mm, corresponding to a total inflow about 12×10^6 m^3, only 70% of that amount that finally feeds into the reservoir. The water pumped from the rivers was 22×10^6 m^3 in 2005, which mainly took place in the dry season, especially from August to March. The minimal abstracted water was less than 10^6 m^3 in April and the maximum was 6×10^6 m^3 in October (Fig. 18.2). Drinking water supply was quite constant every month (about 2×10^6 m^3), except in December and January, when the amount of water supplied increased due to extra demand in the Chinese Spring Festival. At this time, seawater with high salinity invades up the estuary of the Pearl River and its branches, and Dajingshan reservoir had to supply water to Macau via a pump network system. This "water abstraction dynamics" dramatically altered the "hydrological dynamics" of the reservoir, leading to a high water level from November to April, just before the wet season (Fig. 18.2). The lowest water level (9 m) occurred in May, and the highest (20.40 m) in early December. As the water obtainable from the catchment ("natural water") is much less than the abstracted water, the water level is much lower in the wet than in the dry season. This pattern of water level variation is opposite to that in reservoirs fed directly by natural rivers.

Water temperature varied between 15°C and 32°C, and the difference between surface water and bottom was less than 5°C. The water column was stratified from March to November when the surface temperature was over 20°C, and a thermocline formed near the depth of 4 m. As the reservoir became shallow in spring, stable stratification was only found near the dam.

18.3.2 Total Phosphorus and Total Nitrogen

Figure 18.3 shows the dynamics of mean total phosphorus concentrations. TP ranged from 0.016 to 0.086 mg/L, being highest in March and lowest in August. TP concentration showed regular seasonal dynamics, decreasing from spring to summer and increasing in autumn. In spring and early summer, i.e., from January to June, concentration was about 0.05 mg/L on average, which was higher than the

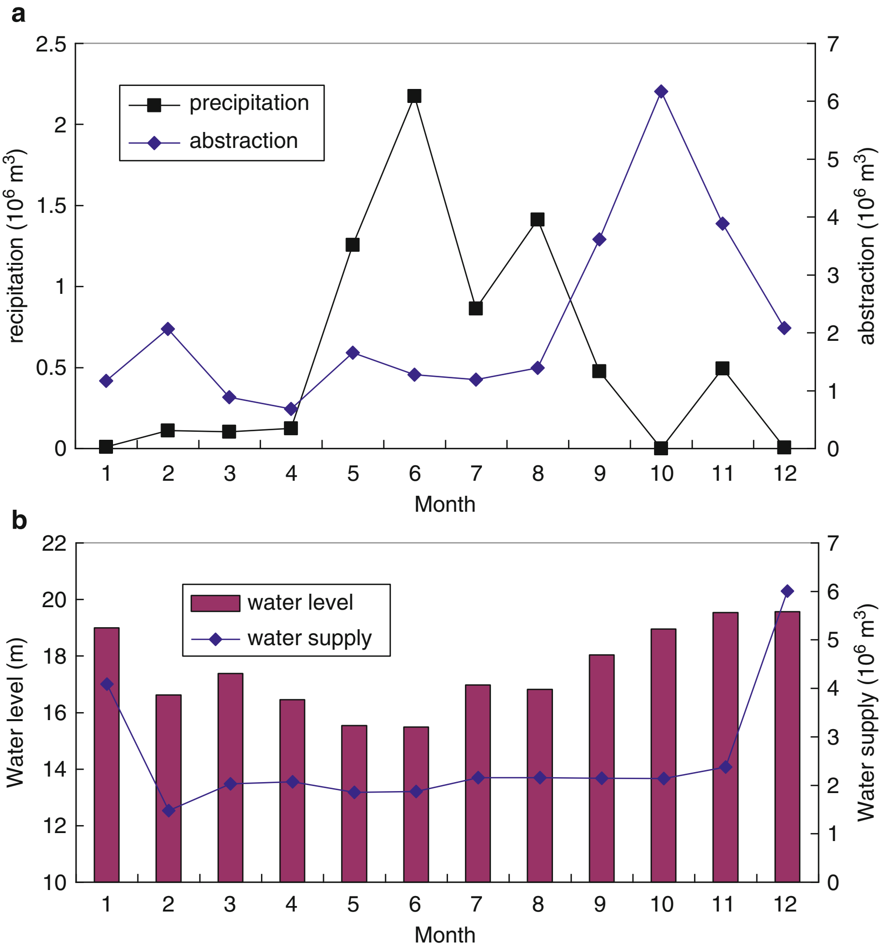

Fig. 18.2 Dynamics of (**a**) precipitation and pumped water, (**b**) water level and water supply in 2005

mean concentration of 0.03 mg/L from July to December. The total phosphorus concentration did not show a significant difference between sampling sites ($P < 0.05$). In the first half year, the concentration was a little higher at the upper reach of the reservoir than at the other three sites, while in the second half-year, the concentration at the inlet site (S4) was only slightly higher than that at the other sites.

Total nitrogen concentration varied from 0.5 to 2.0 mg/L, following the same dynamic pattern as total phosphorus. Total nitrogen concentration decreased more rapidly than TP in March. It is evident that the seasonal dynamics of TN and TP respond to water level. Pumped water provided the main nutrient loading to the

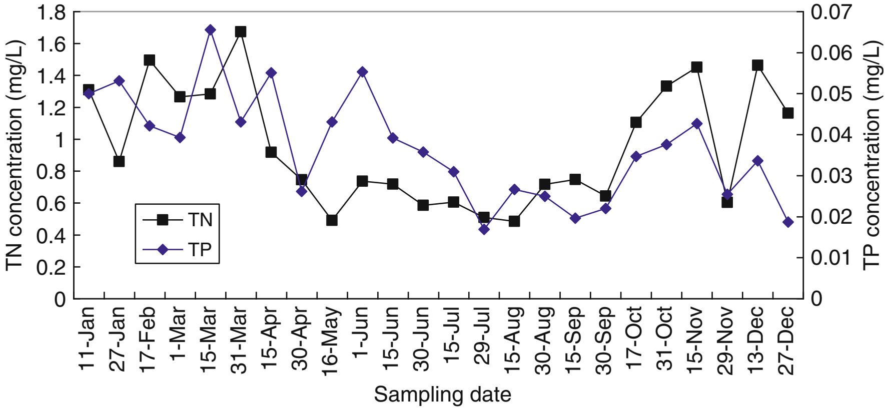

Fig. 18.3 Dynamics of the mean total phosphorus and total nitrogen concentrations (mg/L) in 2005

Table 18.1 Vertical distribution of total phosphorus and total nitrogen in even months (mg/L)

Depth (m)		Sampling month					
		Feb	Apr	Jun	Aug	Oct	Dec
0.5	TP	0.042	0.048	0.032	0.033	0.048	0.032
	TN	1.498	0.900	0.641	0.580	1.397	1.352
2.0	TP	0.053	0.044	0.034	0.013	0.048	0.031
	TN	1.528	1.033	0.568	0.599	0.779	1.292
4.0	TP	0.056	0.037	0.040	0.032	0.056	0.025
	TN	1.397	0.674	0.676	0.550	1.049	1.005
6.0	TP	0.056	0.068	0.046	0.052	0.053	0.041
	TN	1.727	1.233	0.906	1.121	1.293	1.166

reservoir. Table 18.1 shows the vertical distribution of total phosphorus and total nitrogen at the abstraction site (S4). The concentrations of nutrients for the whole water column were higher in the dry season, and a dilution due to precipitation was apparent in the wet season. Their concentration in the deep layer was higher than that in the shallow layers. The internal loading of phosphorus was believed to be contributed by release from the sediment.

18.3.3 Water Transparency and Chlorophyll a

The water transparency had a mean of 0.75 m and varied from 0.4 to 1.3 m (Fig. 18.4). In general, water transparency at all sampling sites followed a similar pattern. In the first half-year, from January to June, mean Secchi depth was 0.75 m, obviously lower than in the second half-year from July to December (1.0 m).

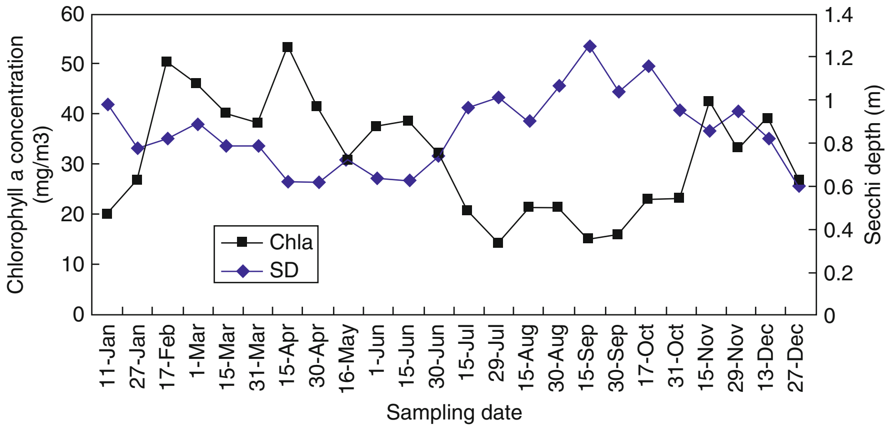

Fig. 18.4 Dynamics of chlorophyll *a* concentration (mg/m^3) and transparency in Secchi depth (m)

Table 18.2 Vertical distribution of chlorophyll *a* concentration (mg/m^3) in even months

Depth (m)	Feb	Apr	Jun	Aug	Oct	Dec
0.5	48.466	45.351	41.172	21.517	20.035	37.944
2.0	48.606	43.257	33.326	17.260	21.775	38.587
4.0	38.951	47.233	27.848	22.628	22.742	40.562
6.0	40.980	29.240	23.543	22.850	19.669	43.801

From late June onward, transparency progressively increased, reached a maximum in September and decreased in November. No significant difference in Secchi depth was identified between sampling sites ($P > 0.05$). In the first-half year, the water transparency at the upper reach of the reservoir was lower than at the other sites.

The chlorophyll *a* concentration followed a pattern similar to that of nutrients, and did not display spatial heterogeneity from the dam to the upper reservoir. It was rather high from February to April, exceeding 40 mg/m^3, and relatively low (about 20 mg/m^3) from July to October. In November, chlorophyll *a* concentration increased but high concentrations only lasted for a few weeks; it rapidly decreased when winter was coming. At S4, chlorophyll *a* concentration at the surface water was higher than at depth of 6 m from February to June (Table 18.2). The disappearance of a vertical gradient in chlorophyll *a* concentration after July indicates that high precipitation and water abstraction enhance water mixing and inhibit phytoplankton accumulation. In December, although chlorophyll *a* concentration was higher in the whole water column than in the wet season, the concentration in surface water was lower than in deep water. The decrease in chlorophyll *a* concentration with depth may reflect a sudden decrease in surface water temperature.

18.3.4 Trophic State Index

A trophic state index (TSI) was calculated by combining TN, TP, Chlorophyll *a*, and Secchi depth. Regressions of TP, TP concentrations, and SD to chlorophyll *a* concentration were calibrated on the basis of an eutrophication survey in southern China. The trophic state index varied seasonally from 30 to 55 (Fig. 18.5). The reservoir was mesotrophic in the wet season and eutrophic in the dry season. As a comprehensive index, dynamics of the mean weighed TSI followed an annual pattern similar to that of water quality criteria such as total phosphorus and total nitrogen.

18.3.5 Phytoplankton Abundance

Phytoplankton abundance ranged from 0.86×10^6 to 106.27×10^6 cell/L with two peaks: one in April and another in November (Fig. 18.6). Abundance exhibited a seasonal variation similar to total phosphorus and nitrogen concentrations, and no significant difference was found between sampling sites. With increasing water level, phytoplankton abundance was persistently low. From April to July, abundance fluctuated to some extent. During this period, precipitation provided the majority of the inflow. In the mixing period, i.e., the short winter, phytoplankton was dominated by diatoms and Cyanobacteria. Diatoms constituted about 40% of total phytoplankton abundance. *Cyclotella acus* was one of the dominant species. During stratification, phytoplankton was dominated only by Cyanobacteria. They made up 90% of cell abundance, and *Pseudanabaena* sp. was the only taxon with over 10% abundance.

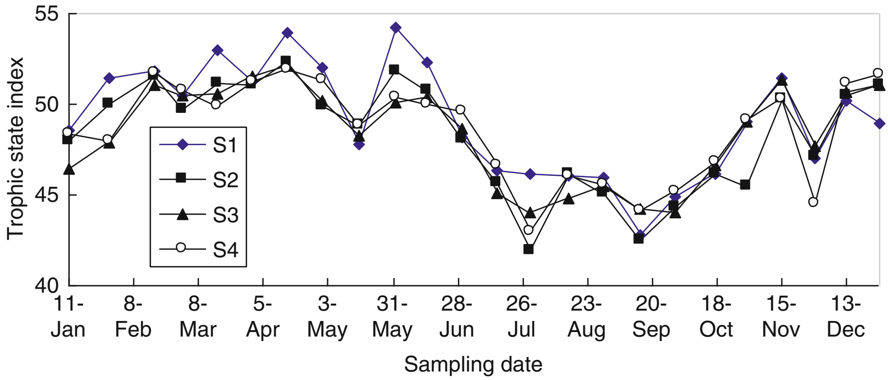

Fig. 18.5 Dynamics of trophic state as measured as the mean weighed TSI at four sampling sites

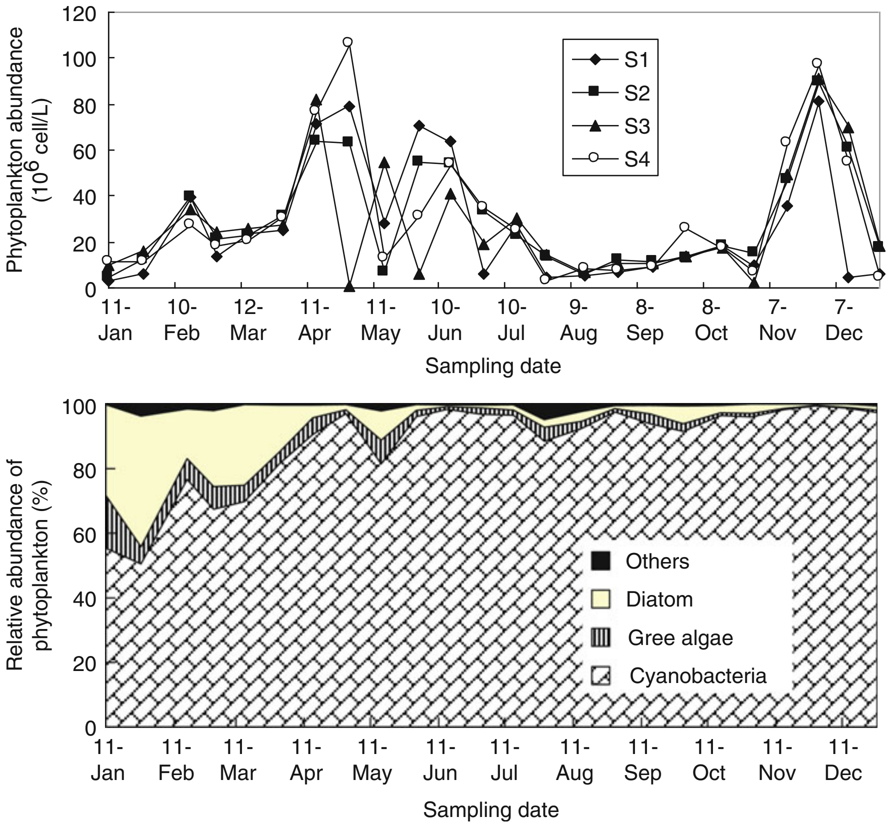

Fig. 18.6 Dynamics of phytoplankton abundance at four sampling sites (**a**) and the relative abundance near the dam (**b**)

18.4 Discussion

18.4.1 Seasonal Dynamics of Trophic Status

Reservoirs are usually constructed in the middle or lower parts of rivers, and are principally designed to have a large catchment and to receive adequate inflow from their feeding rivers. Several properties of reservoirs distinct from natural lakes have been identified (Straskraba and Tundisi 1999; Negro et al. 2000). For instance, water level varies with inflow from catchments, and usually increases following precipitation in the wet season, and then begins to decrease gradually to its lowest level before the onset of the next wet season (Lin et al. 2003a; Zhao et al. 2005). However, as a pumped water storage reservoir, Dajinshan did not have such a pattern. Its water level was modified by water abstraction in the dry season. In total,

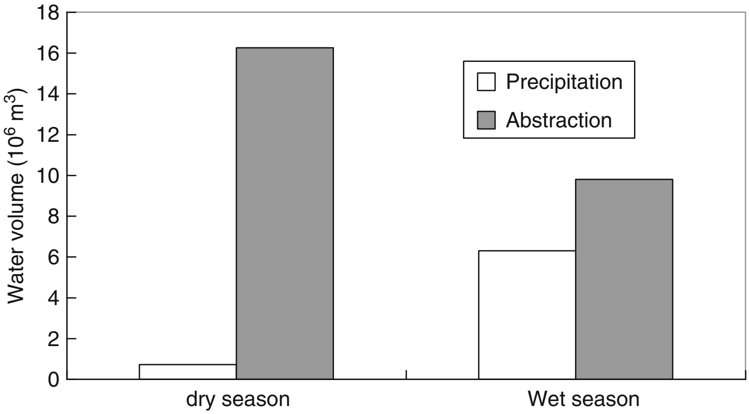

Fig. 18.7 Comparison of precipitation and abstraction between the wet season and dry season in 2005

about 22 $\times$ 10^6 m^3 was pumped into the reservoir in 2005, equal to three times the total precipitation (7 $\times$ 10^6 m^3), and 80% of the pumped water was taken out in the dry season (Fig. 18.7). Compared with natural lakes, reservoirs are highly dynamic, and hydrodynamics play a principal role in regulating their phytoplankton abundance (Arfi 2005; Beyruth 2000; Horn 2003). On the other hand, inflow from rivers provides nutrient loading to reservoirs. As external nutrients come mainly from pumped water, water with a high nutrient load from polluted rivers can substantially influence water quality and eutrophication. Vieux-Pré is a reservoir (61 Mm^3) in the northeast part of France. Its own catchment area is only about 10 km^2, too small to fill up the reservoir, so water is pumped from the highly eutrophic Plaine River, but this increased the phosphorus loading to the reservoir (Leitao and Leglize 2000). Water abstraction to the reservoir resulted in not only a novel pattern of hydrological dynamics, but a high nutrient loading as well. When the wet season arrives, water abstraction is largely reduced, and increases in precipitation dilute both nutrient concentrations and phytoplankton biomass, especially at the surface. Table 18.3 presents a comparison of eutrophication parameters between the wet and dry season near the dam. At depths from 0.5 to 6 m, TN, TP, and Chlorophyll *a* were significantly higher in the dry season than in the wet season ($P < 0.05$).

In reservoirs fed by natural rivers, particulate matter settles to the bottom, and so a gradient of nutrient concentration is generated from the riverine to the lacustrine zone near the dam (Han et al. 2000; Hart et al. 2002; Holas et al. 1999; Soyupak et al. 1997). However, Dajinshan Reservoir does not have its own feeding river, and most of the input water was pumped in from other rivers, and just at the dam. This mode of water provision prevents the formation of a normal gradient in water quality from the tail of the reservoir to the dam. On the other hand, this small-sized reservoir has a relatively short residence time (less than 100 days), which is reasonable for the absence of a spatial gradient. Only in the first half-year do total phosphorus concentration and water transparency at the upper reach of the reservoir

Table 18.3 Vertical distribution of total nitrogen, total phosphorus, and chlorophyll *a* in the wet and dry season of 2005

Depth (m)	TN (±SD) mg/L		TP (±SD) mg/L		Chla (±SD) mg/m^3	
	Dry season	Wet season	Dry season	Wet season	Dry season	Wet season
0.5	1.233 (±0.465)	0.633 (±0.148)	0.044 (±0.013)	0.034 (±0.012)	35.210 (±11.105)	28.353 (±11.587)
2.0	1.291 (±0.311)	0.612 (±0.119)	0.046 (±0.014)	0.031 (±0.016)	36.030 (±10.523)	26.983 (±11.046)
4.0	1.269 (±0.378)	0.682 (±0.198)	0.042 (±0.016)	0.032 (±0.012)	35.155 (±8.8993)	27.278 (±9.972)
6.0	1.369 (±0.266)	1.044 (±0.350)	0.054 (±0.015)	0.042 (±0.020)	31.823 (±10.419)	22.386 (±6.955)

slightly differ from the other sites. Individually, total phosphorus concentration is higher and water transparency lower at the upper reach of the reservoir. This is attributed to re-suspension in the shallow zone. Vertically, total nitrogen, total phosphorus and chlorophyll *a* distributed in an evident gradient from surface to bottom in both dry and wet seasons. As nutrient concentration is higher near the bottom, this vertical gradient implies that internal loading by the sediment release is high.

18.4.2 Phytoplankton Response to Water Abstraction

Phytoplankton abundance typically has a longitudinal gradient from the riverine to the lacustrine zone. As a result, phytoplankton growth is limited by low nutrient concentration in the pelagic (Horn 2003; Naselli-Flores 2000; Komarkova and Hejzlar 1996). In Dajingshan reservoir, however, water abstraction leads to high nutrient loading in the lacustrine zone, and the high nutrients can quickly disperse over the entire reservoir with water currents. The longitudinal gradient in phytoplankton abundance difficultly develops in this middle-sized reservoir with short residence time. Following variation of nutrients, phytoplankton abundance exhibits a temporal distribution with two peaks, one in the late spring and the other in late autumn, a pattern similar to that observed in temperate lakes. However, the underlying mechanisms are distinctly different. In temperate lakes, high nutrients due to mixing and proper water temperature stimulate cell growth of phytoplankton in spring and early autumn (Reynolds 1998; Lau and Lane 2002). In contrast, mixing intensity is much lower in tropical eutrophic water bodies, and phytoplankton biomass usually peaks in summer (Lewis 2000). In the present pumped water storage reservoir, nutrients are mainly loaded with pumped water. In summer, nutrient concentration is reduced due to a decrease in pumped water and high precipitation. Thus, the two peaks of phytoplankton abundance were produced by a reduction of nutrient loading and an evident dilution due to precipitation. In the mixed period, water temperature was between 15°C and 18°C, phytoplankton was dominated by diatoms and Cyanobacteria, and *Cyclotella acus* was one of predominant species. During stratification, the filamentous *Pseudanabaena* sp. constituted about 90% of total abundance. Filamentous Cyanobacteria have been suggested to have an advantage in nitrogen fixation and are abundant under low nitrogen conditions (An and Jones 2000; Figueredo and Giani 2001; Beyruth 2000). From the viewpoint of N/P ratio, however, phytoplankton growth is limited in the reservoir by phosphorus. In other reservoirs in southern China, *Pseudanabaena* sp. is widely distributed but dominant only in eutrophic water. How this filamentous Cyanobacterium becomes dominant remains unclear.

18.4.3 Combating Eutrophication

At present, water quality deterioration is clearly evident and is confirmed by the occurrence of cyanobacterial blooms in early spring. To maintain drinking water supply, control of eutrophication for water quality improvement has to be enforced. The water abstraction with high nutrient loading functions as an external pollutant source, and the nutrients were directly loaded to the reservoir at a site close to the two large tubes for drinking water supply. Therefore, it can be expected that moving the pumps of water abstraction to the upstream reservoir may effectively improve water quality at the lacustrine zone. From the viewpoint of control of nutrient loading, the pumped water needs to be treated biologically before it is pumped into the reservoir. As water abstraction has been carried out more than 20 years, it can be expected that a huge amount of sediment has settled near the dam zone. The persistent vertical distribution of total nitrogen and total phosphorus confirms this internal loading of nutrients. In Alton Water reservoir (Suffolk, UK), ferric sulfate was dosed with the input water to control phosphorus loading to the main water body (Perkins and Underwood 2001). Dredging of the sediment, especially in the dam zone, should be carried out to reduce the internal loading of nutrients.

18.5 Conclusions

Dajinshan reservoir is a typical pumped water storage reservoir of the tropics, in which most of the pumped river water was conducted in the dry season, after summer. As the pumped water has a high nutrient concentration, the water abstraction accelerates eutrophication and leads to a high water residence time and water level in the dry season. Total phosphorus, total nitrogen and chlorophyll *a* follow a dynamic pattern similar to that of the water level. The external nutrient loading from the pumped water is the main driving force for eutrophication. The mean weighed trophic status index indicates that the reservoir is mesotrophic in the wet and eutrophic in the dry season. Phytoplankton abundance exhibits a pattern that is similar to that of the trophic state and is dominated by filamentous Cyanobacteria across in the year. Although the observed cyanobacterial bloom was confined to the dam area, it strongly suggests that measures against eutrophication are urgently required. At present, reduction of nutrient loading by treating abstracted ("incoming") water before it reaches the reservoir and dredging of the sediment are the priority strategies that we recommend.

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Chapter 19
Nutrients and Phosphorus Release in Sediment in a Tropical Pumped Water Storage Reservoir

Huajun Zhang, Jiguang Gu, and Zhangwen Lin

Abstract Dajingshan Reservoir is a typical pumped-water storage reservoir, originally built for agricultural irrigation around Zhuhai City and for supplying drinking water to Macau. Because of its small catchment, river water began to be pumped into the reservoir for more water storage in the dry season from the only but polluted river in Zhuhai. After 30 years the accumulated nutrients in the sediments have become a major internal pollution factor in the eutrophication of the reservoir. In the present study, we measured the accumulation and release of nutrients from the sediments. The accumulated sediment reaches about 48,840 m^3 in volume with a wet-weight of 54,027 t and drought-weight of about 36,137 t. Total inorganic phosphorus (TIP), total nitrogen (TN), and releasable phosphorus (Re-P) in the sediment were estimated at 21,886, 67,015, and 5,867 kg, respectively. TIP, TN, and Re-P were highest near the dam, reflecting the hydrodynamic characteristics of the reservoir. Because of the high Re-P content and anaerobic environment in the zone near the dam, this zone was the primary area releasing nutrients. Phosphorus release was measured through in situ experiments and was 4.09 $mg{\cdot}m^{-2}{\cdot}day^{-1}$ in the zone near the dam. Total annual phosphorus release was about 109 kg, mainly contributed by release from the dam zone.

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19.1 Introduction

Many small and medium reservoirs near towns are built only for drinking water supply, but they have small and short feeder rivers. In the dry season, the water that can be stored is not enough to meet the demand. One common solution of increasing storage is to pump water from nearby rivers. Such reservoirs are so-called pumped storage reservoirs. They are located mainly in coastal areas where rivers are short and dry out in the dry season (Bayley et al. 2001; Van Breemen et al. 1998). Many are found in coastal cities of South China. For example, all reservoirs near Shengzhen and Zhuhai Cities of Guangdong province of China, two most developed cities close to Hong Kong and Macao, were built in such way.

Because of saltwater intrusion it is difficult to obtain freshwater from the Pearl River estuary at Zhuhai City. After 1970, a number of small- to medium-sized reservoirs were built to improve the drinking water supply to Zhuhai and Macao in the dry season. In an initial period, these reservoirs also functioned to improve the quality of water through the settlement of suspended solids and pollutants. Because of their different mode of storing water, hydrodynamics and nutrients dynamics are different from reservoirs that depend on their own catchments (Ta and Brignal 1998). Most of the pumped storage reservoirs are young water bodies, and only few studies deal with their limnology.

Phosphorus in water is one of the essential nutrients required by phytoplankton. Phosphorus concentration also determines the trophic status of a water body. Most phosphorus is entering reservoirs with inflow, including runoff and precipitation. Phosphorus is first absorbed by phytoplankton and by higher plants and cycled in food webs. Some phosphorus then settles to become part of sediments, where it can be adsorbed and fixed by iron and aluminum hydrates, clay minerals, apatite, or organic matter, eventually forming various kinds of bound phosphorus (Fu and Zhou 1999). Inorganic phosphorus (In-P) is the main form of sediment phosphorus, so the composition of various forms of inorganic phosphorus largely determines the soil properties of sediment (Frankowski et al. 2002; Aigars 2001; Appan and Ting 1996). These affect the remobilization of sediment phosphorus, and finally reflect on the eutrophication of water (Ashityan and Ding 1996; Straskraba and Tundisi 1999; Liu et al. 2004; Wan et al. 1996). Thus, knowledge of the types and contents of In-P in sediment can be helpful in understanding their activity and environmental effects.

In recent years, deterioration of river water has had clear negative effects on the water quality of most pumped storage reservoirs. The accumulated sediment and its nitrogen and phosphorus contents increase year after year. With anaerobic conditions at the bottom, phosphorus is easily released from the sediment, accelerating eutrophication and promoting water blooms (James and Barko 1997).

In the present study, we measured accumulation and release rates from sediment and their contents of nutrients in Dajingshan Reservoir. This reservoir is a typical pumped storage reservoir, located in Zhuhai City, Guangdong Province. Sediment cores were collected from this reservoir. By analyzing the thickness, character, and particle size distribution of sediment and measuring the contents of five forms of inorganic phosphorus, we were able to reconstruct sediment accumulation during last 30 years as well as total nitrogen and phosphorus contents in reservoir sediment. We aim to use these fundamental data in a better understanding of internal loading and to develop an adaptive management of eutrophication.

19.2 Materials and Methods

19.2.1 Reservoir Presentation

Dajingshan Reservoir is located in the northwest of Xiangzhou district of Zhuhai city and to the south of Fenghuang Mountain. It was built and filled in 1975, mainly for irrigation, water supply, and flood control. After the establishment of Zhuhai Economic Special Zone in 1997, its main function shifted to water supply, and the reservoir became the main water source for drinking water for Zhuhai city and Macao The reservoir has a small catchment of 5.95 km^2, a total storage capacity of 12.1×10^6 m^3 and a normal water level of 20.4 m. River water is pumped into the reservoir by a pipe across the dam.

19.2.2 Location of Sampling Sites

Sediment sampling and nutrient release experiments were conducted at four sites in December 2004. Site 1 (S1) was at the upstream (called reservoir tail), Site 2 (S2) was at the center of the reservoir, Site 3 (S3) at the outlet for water supply, and Site 4 (S4) at the inlet for pumped water (Fig. 19.1). In order to obtain more information about the spatial distribution of sediments, we collected seven sediment samples at Sites 1–7 in March of 2005.

19.2.3 Sampling Sediment and Methods

In 2004, we collected sediment samples with an Uwitec columnar sediment sampler at S1–S4. The sampler has a diameter of 6 cm, and the section area of sediments is 0.002826 m^2. At each site, we collected three sediment cores. One was sliced in an interval of 5 cm and used for observation of physical properties. The second was sliced at an interval of 5 cm for analysis of particle size. The third was sliced every 2 cm for measuring contents of In-P and TN. The sediment samples were dried and grinded and finally sieved on a mesh size of 0.149 mm.

In March of 2005, core samples were collected at the seven sites. At each site three sediment columns were collected. The length of each of them was measured,

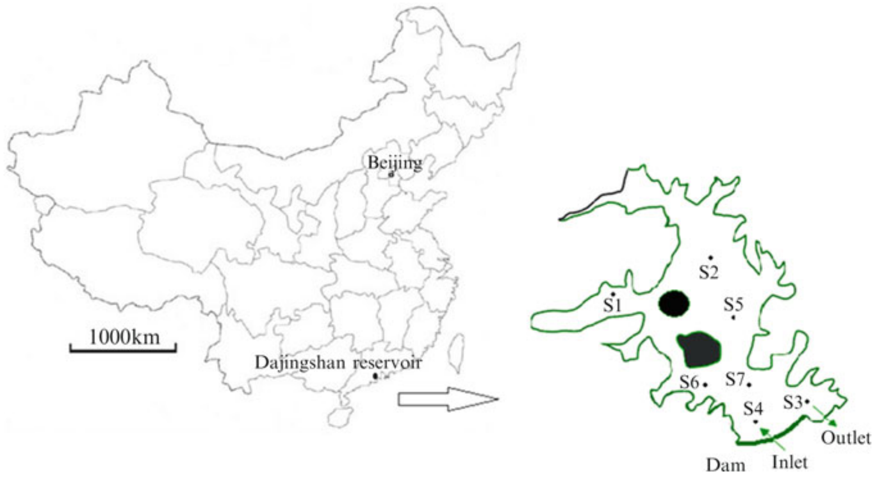

Fig. 19.1 Location of sampling sites in Dajingshan Reservoir

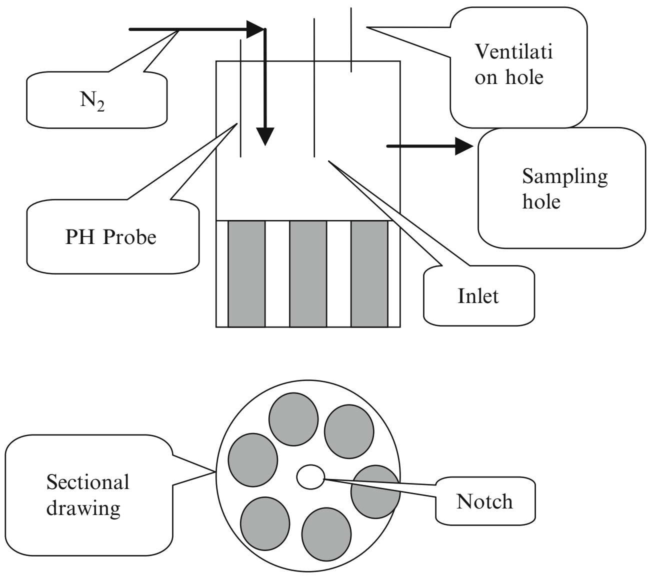

Fig. 19.2 Device for phosphorus release experiment

and the sediment sliced immediately. The sliced samples were weighed in the laboratory, dried at 60°C, and weighed again.

In September of 2005, we collected sediment samples at S1–S4 and installed the sediments into phosphorus release devices (Fig. 19.2). The overlying water was carefully added to the device, which was then sealed and slowly lowered to the bottom. Phosphorus concentration of overlying water was measured before and after the in situ experiment to calculate the amount of phosphorus released.

Following a method described in Lu (2000), the particles in the sediments were classed into five fractions: 3–0.01, 0.05–0.01, 0.01–0.005, 0.005–0.001 mm, and <0.001 mm.

Total nitrogen (TN) content was determined with the Kjeldahl method. All fractions of sediment inorganic phosphorus were measured by the molybdenum blue colorimetric technique following Chang and Jackson. To measure TP and TDP concentrations in the in situ release experiment, redox potential and pH of water were detected with YSI meters. The concentration of TP was determined by colorimetry after the overlying water had been filtered on a 0.45 μm cellulose acetate membrane.

The release rate r of phosphorus was calculated as follows: $r = \sum_{i=1}^{n} (C_i - C_0) V_i/(At) + (C_n - C_0)(V - \sum_{i=1}^{n} V_i)/(At)$, where V is the volume of initial water in the container, V_i is the volume of the water samples, C_0 is the phosphorus content of samples, t is the days, and A is the cross-section area of six sediment samples.

19.3 Results and Discussion

19.3.1 Soil Properties and Particle Size Distribution of the Sediments

The sediment core at S1 was 12-cm long; the core was yellow with its surface layer slightly dark brown, and 4 cm of gravel at the bottom. The core at S2 was 42-cm long, with the upper layer about 22-cm long and was black and smelly. The lower part of the sediment was yellow and hard. The core at S3 was 25 cm in length, the 16-cm long upper layer was black and odorous, and the lower part yellow. The core at S4 was about 45-cm long, entirely black with high viscosity, and contained much undecomposed humus including leaves and branches.

Based on the pumped particle size fractions of surface sediments (0–5 cm) at the four sampling sites, the soil was light in S1, light clay in S2, and heavy loam in S3 and S4 (Fig. 19.3). At S1, the sediment mainly consisted of particles in the 3–0.01 mm size fraction, which accounted for 71.0% and 6.4% of particles in the <0.005 mm size fraction. At S2 and S3, particles in the <0.001 mm size fraction dominated and accounted for 27.6% and 27.0%, respectively. However, sediment at S2 had larger proportion of particles in the 0.005–0.001 mm size fraction, while sediment at S3 had larger proportion of particles in the 0.05–0.01 mm size fraction. S4 had mainly particles in 3–0.01 mm size fraction (accounting for 53.3%), followed by particles in the <0.001 mm size fraction (15.2%).

After being pumped into the reservoir, most of the suspended solids are deposited at the inlet to form a delta district, where larger-sized particles dominate the

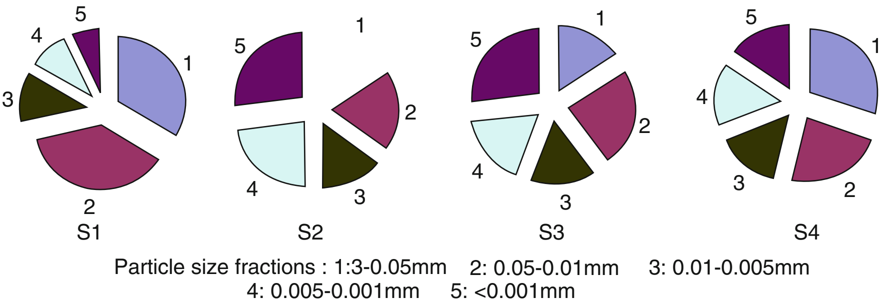

Fig. 19.3 Particle size fractions of surface sediments (0–5 cm) in Dajingshan Reservoir

sediments (Sundborg 1967; Thornton 1990). The inlet for pumping water was near the dam, and the sediment here mainly consisted of particles in large size fractions (Fig. 19.3), with much undecomposed humus. The water area around S4 stored most sediment in the whole reservoir.

Because the pumped river water entered the reservoir through a big pump near the dam, the deposition of suspended particles showed no longitudinal distribution like in the narrow channel of ordinary reservoirs, but was instead distributed both laterally and longitudinally (Thornton 1990; Thankur and MacKay 1973). Sediment at S3 had a large proportion of particles in the 0.05–0.01 mm size fraction and a small proportion of particles in the 0.01–0.005 mm size fraction. This may be because S3 was close to the outlet, and the high water flow and velocity were suitable to the accumulation of large size particles. S1 was far from the inlet and outlet where the water movement was weak, and sediment had a low content of particles in the <0.005 mm size fraction. Although the sediment at S4 and S1 had high contents of large size particles, their physical characteristics were different. The large size particles at S4 was the large particle residue, humus, and had dark brown color. In contrast, S1 was mainly gravel, probably formed by the deposition of sludge and gravel sliding from the bank of the reservoir by erosion.

19.3.2 Spatial Distribution of Inorganic Phosphorus in the Sediments

19.3.2.1 Horizontal Distribution

The average content of five species of In-P at four sampling sites was 0.638 mg g^{-1}. Among the five species of In-P, the content of exchangeable phosphorus was almost negligible compared to the content of other four species of In-P. The content order of four main species of In-P was: Oc-P > Fe-P > Ca-P > Al-P (Fig. 19.4). Oc-P was the main form of In-P in Dajingshan Reservoir, and was mainly wrapped in iron, aluminum, and other mineral grains from which it is difficult to release. Such a

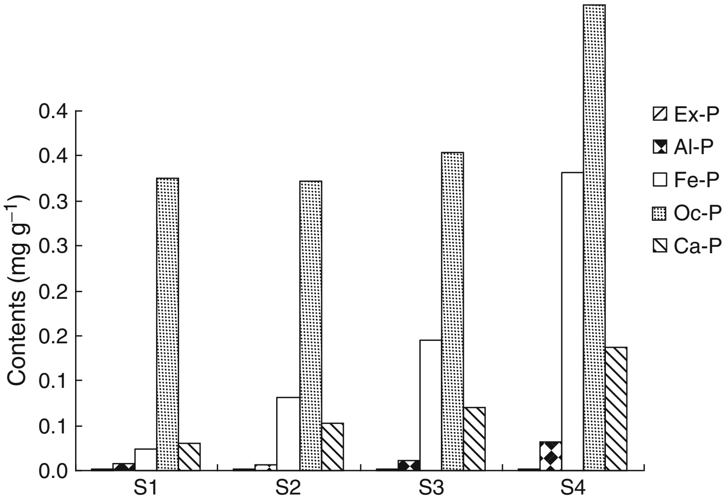

Fig. 19.4 Mean contents of Ex-P, Al-P, Fe-P, Oc-P, and Ca-P in the sediment of Dajingshan Reservoir

high content of Oc-P was also reported in other reservoirs or lakes. For example, a study of 22 shallow lakes in middle and lower reaches of the Yangtze River found that Oc-P always had the highest content (Zhu et al. 2004). But there were some differences in patterns. The highest content of In-P was Ca-P in the sediment of Miyun reservoir, and apatite was the main mineral that stored phosphorus (Liu et al. 2003). This may be related to the geochemical characteristics of inherent alkaline soil in the north of China. Fe-P was absolutely dominant in the sediment of Shanzai reservoir and Xihu Lake, but Oc-P was lower because of the anthropogenic input (Su et al. 2005).

Although both Hongfeng and Baihua reservoirs had high background contents of calcium in the soil, the content of Fe-P was higher than the Ca-P's, and the oxides of iron or hydroxide had strong complexation ability with phosphate ions (Wang et al. 2000). Fe–P content was also high in the sediments of Dajingshan Reservoir, where their average content was 0.146 mg g^{-1}. This is a reflection of the high content of Fe in the red soil of southern China. The average content of Ca–P was 0.014 mg g^{-1} and the content of Al–P was much lower. Thus, the content of different species of In-P main was decided not only by geological background but also by the source of sediment. On the other hand, the content of different species of phosphorus is usually correlated with sediment properties such as particle size and content of main chemical elements (Andrieux-Loyer and Aminot 2001). Different soil types had varying force adsorbing phosphorus; this also illustrates the importance of sediment properties to the distribution of In-P (Madhav and Lin 1996).

In our reservoir, In-P content between sediments at four sites showed an obvious spatial difference. S4 had the highest content of In-P (1.018 mg g^{-1}) and S1 the smallest. The spatial distribution of In-P was consistent with the hydrodynamic process in the reservoir. S4, located at the inlet, was the main depositing district of

Table 19.1 The coefficient of variation of Al-P, Fe-P, Oc-P, and Ca-P in the sediments of Dajingshan Reservoir

	Mean (mg g^{-1})	Standard error	The coefficient of variation (CV)
Al-P	0.015	0.012	0.866
Fe-P	0.161	0.133	0.913
Oc-P	0.385	0.093	0.245
Ca-P	0.078	0.046	0.633

In-P. S3 is located at the outlet, and the distance of the two sites was short. S2 was far from S4, although S1 was even further from S4 and S3. The sediments, especially the large size particles, mainly originated from the river, and were transported into the reservoirs through pumping. Thus, the In-P distribution reflects the water flow characteristics. Using the average content of four main species of In-P in four sediment columns as the cardinal number, we calculated the horizontal distribution of Coefficient of Variation (CV) (Table 19.1).

The Coefficient of four species of sediment In-P ranked as: CV(Fe-P)>CV(Al-P)>CV(Ca-P)>CV(Oc-P). Fe-P had a large difference in spatial distribution, followed by Al-P, and Oc-P was the smallest. The spatial difference of In-P is related to the nature of its combined forms or their activity, and the content of Oc-P in the sediment was relatively stable and not easy to release. The Ca-P was also one kind of releasable phosphorus, but dissolved CO_2 in the water can induce its release slowly. This kind of phosphorus was consisted of the phosphorus contained in mineral grains; particulate phosphorus formed through biological processes and induration processes. Organisms generally find Oc-P and Ca-P difficult to use. Fe-P and Al-P were mainly formed through adsorbing and combining iron oxide and aluminum oxide in soil; they are the releasable In-P (Zhu et al. 2003, 2004). The structural stability of Oc-P determined that its content diversification will not be high and is little affected by hydrodynamic in different regions. Comparatively, Fe-P and Al-P are more easily influenced by hydrodynamics. The study of sediments in Lake Taihu and its main inflow rivers also showed that differences in the active components of Fe-P and Al-P are more evident than in the other species (Zhang et al. 2004).

19.3.2.2 Vertical Distribution of Inorganic Phosphorus

The contents of Fe-P and Al-P had clear gradients at S2 and S3, decreasing gradually from surface to bottom. The content of Al-P also had a clear gradient at S4, in which the content of Ca-P had a similar trend to that at S2 andS3, but not as clear as that of Fe-P and Al-P. The vertical gradient of Oc-P was not obvious. Except for Oc-P, the other four species of In-P showed no evident vertical change at S1 (Fig. 19.5).

The lower parts of the cores at S1, S2, and S3 were yellow, reflecting the background soil. Their contents of four species (not Ex-P) of In-P were relatively stable. The contents of four species of In-P in the soil of Dajingshan Reservoir were

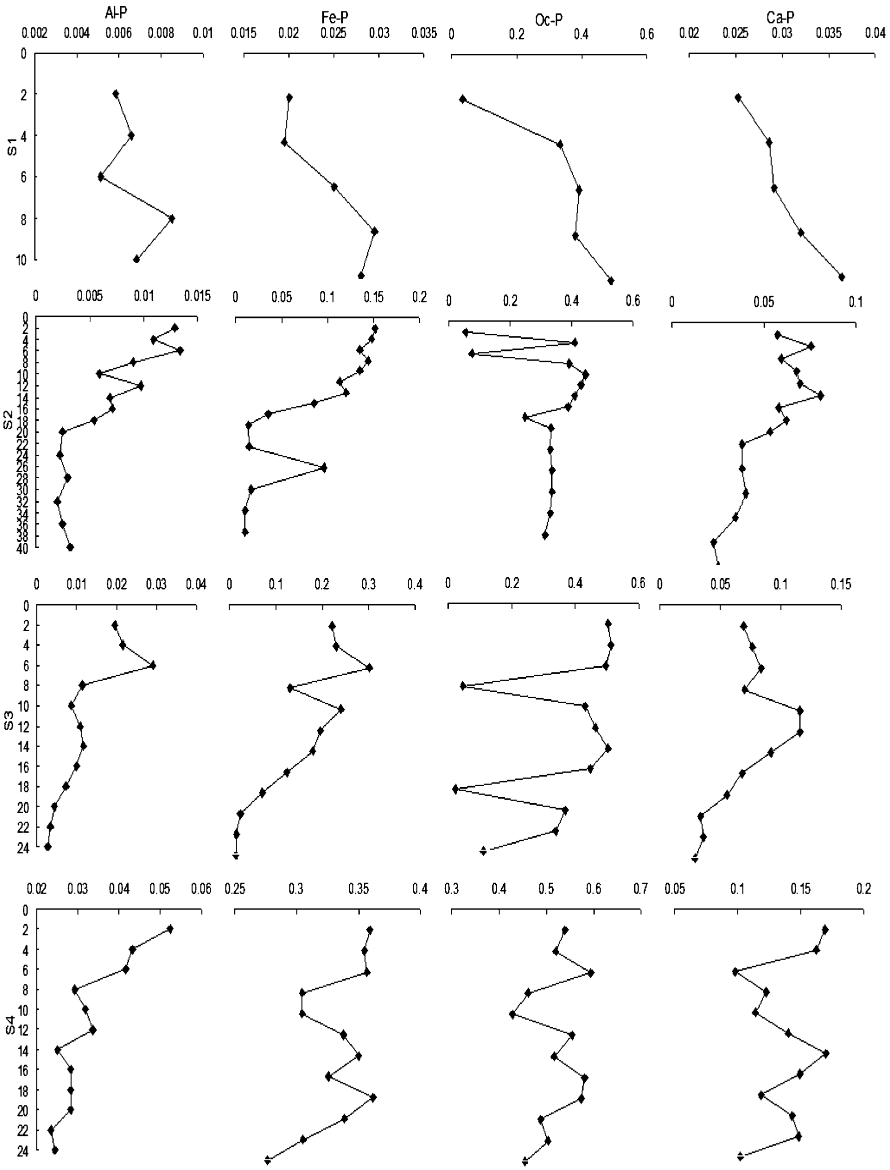

Fig. 19.5 The vertical profiles of Al-P, Fe-P, O-P, and Ca-P contents in the sediments of Dajingshan Reservoir

Al-P: 0.002–0.003 mg g^{-1}, Fe-P: 0.010–0.017 mg g^{-1}, Oc-P: 0.313–0.335 mg g^{-1}, and Ca-P: 0.023–0.040 mg g^{-1}. The CV of In-P in the vertical sediment reflected the period of settling variation (Fig. 19.6). The CV of Fe-P, Al-P, and Ca-P had similar patterns at the four sampling sites. The CVs of four species of In-P were low

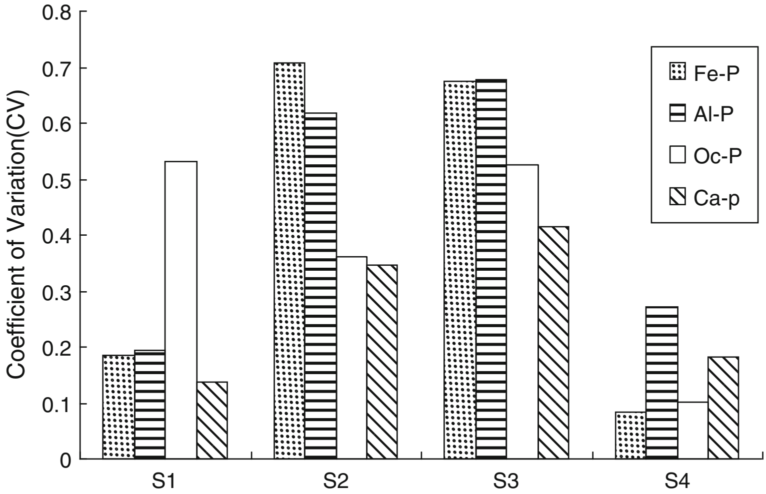

Fig. 19.6 Coefficient of Variation of Al-P, Fe-P, Oc-P, and Ca-P in the sediment of Dajingshan Reservoir

at S4. They were low at S1 except for Oc-P. The CVs of Fe-P and Al-P were high at S2 and S3.

The area around S4 was the main depositing area of nutrients and particulates, and the contents of four species of In-P were also the largest, with small CVs. This indicates that phosphorus could soon be adsorbed in the sediment and reach the upper limit of adsorption (BostrÖm et al. 1982). Because of the economic activity and urban development in the last decades, the amount of phosphorus discharged into the river water increased yearly, and iron or aluminum oxide compounds maximally absorbed phosphorus. Through a phosphorus adding experiment in the East Taihu Lake, it was found that phosphorus in the surface sediment had basically reached its saturation (Li et al. 1998). On the other hand, S4 had the highest deposition rate of suspended solids, with 45 cm over 30 years and an average rate of deposition of 1.5 cm year^{-1}. Compared to the other three species of In-P, Al-P had a high CV that was probably related to ion activity at S4, because the stability of Al-P is greatly influenced by the ion contents. We measured the conductivity at different depth water layers at S4. The conductivity at surface and intermediate water layers was about 540 μs cm^{-1}, but rose to 1,200 μs cm^{-1} rapidly near the sediment. Thus, the water near the sediment had a high ionic concentration. S1 was far from the inlet, and its sediment weakly absorbed In-P. Fe-P, Al-P, and Ca-P all had small CV values at this site, but Oc-P had a high CV. The reason may be related to the peeling off and deposition of the different soils of reservoir shore. The vertical gradient of four species of In-P at S2 and S3 were obvious, and Fe-P and Al-P had clear vertical gradients. Cores at these two sites contained basically mud layers accumulated after the reservoir was built, and the gradient reflected the change of In-P in the reservoir.

19.3.3 Spatial Distribution of TN, TIP, and Releasable Phosphorus

TN contents differed between sampling sites. The TN content at four sites was 0.76–1.46 mg g^{-1} at S1, 0.27–3.95 mg g^{-1} at S2, 0.68–3.21 mg g^{-1} at S3, and 1.44–2.69 mg g^{-1} at S4 (Fig. 19.7). It had the biggest gradient at S2, and relatively small gradients at S1 and S4. The Ex-P is releasable, Al-P and Fe-P can transform into dissolved phosphorus when the environmental condition is suitable. They are first released into the pore water and then into the overlying water (Sundborg 1967). Among the three species of Re-P in the sediments of Dajingshan reservoir, the Fe-P was the highest. Therefore, release of Fe-P was the main form of phosphorus release, perhaps the main source of internal loading (Fig. 19.8).

19.3.3.1 Horizontal Distribution of TN, TIP, and Re–P

We averaged the contents of five species of In-P (EX-P, Fe-P, Al-P, Ca-P, RS-P), Re-P, and TN of each layer (2 cm) at the four sampling sites (Fig. 19.8). TIP was the summation of five forms of In-P, the content of Re-P was the summation of Ex-P, Fe-P, and Al-P (Sundborg 1967).

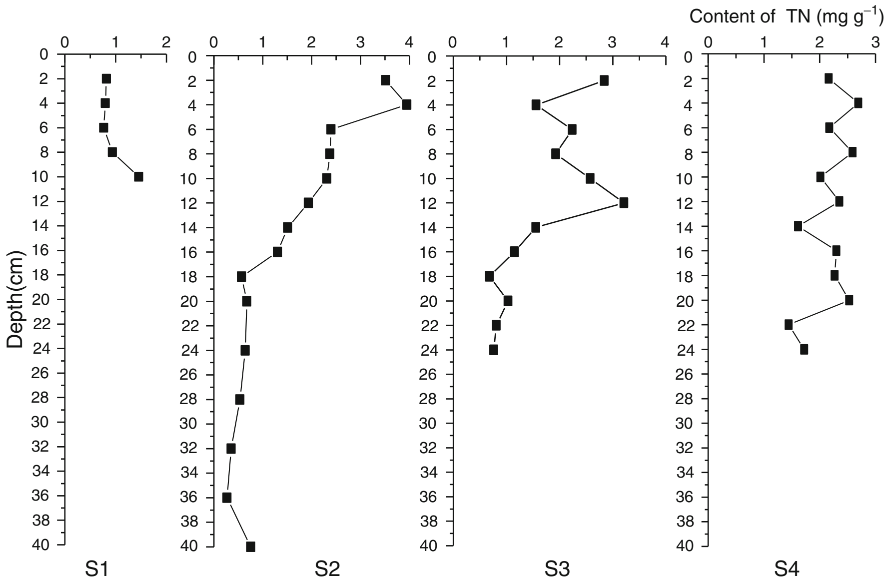

Fig. 19.7 Contents of TN in sediments in Dajingshan Reservoir

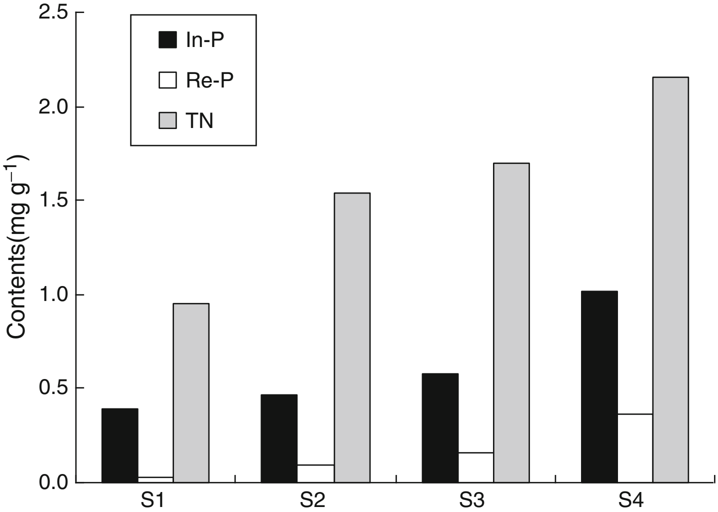

Fig. 19.8 Contents of TIP, Re-P, and TN in the sediment of Dajingshan Reservoir

The content of TN, TIP (total inorganic phosphorus), and Re-P in the sediment of four sampling sites had significant spatial variation in the order: $TIP_{S1} < TIP_{S2} < TIP_{S3} < TIP_{S4}$, $Re\text{-}P_{S1} < Re\text{-}P_{S2} < Re\text{-}P_{S3} < Re\text{-}P_{S4}$, $TN_{S1} < TN_{S2} < TN_{S3} < TN_{S4}$. S4 had the largest contents, followed by S3, S2, and S1 in that order. S4 is located at the inlet and S3 at the outlet, two sites that were close, and water flow from S4 to S3 passed the main areas of sediment deposition. S2 was far from S4, and S1 was even further from both S4 and S3, and so the hydrodynamic process had less impact on S2 and S1. The spatial variation of TIP, Re-P, and TN in the sediments reflected the dynamic characteristics of the reservoir.

19.3.3.2 Vertical Distribution of TN, TIP, and Re-P

Because of variation in hydrodynamic conditions, the vertical profiles of the contents of In-P, Re-P, and TN differed at the four sampling sites. Among the four sites, S2 was hardly influenced by water disturbance, conditions for deposition were stable, and local sediments reflect the history of water quality in the reservoir. We chose the sediment at S2 as an example to analyze the vertical distribution of nutrients in the sediments (Fig. 19.9).

The length of sediment core collected at S2 was 30 cm. Mud sediment occupied 0–22 cm, and the background soil under 22 cm (Fig. 19.9). The contents of In-P, Re-P, and TN decreased with increasing depth. This trend may be caused either by degradation in the mineralization of organic matters or by more input of organic matter, nitrogen, and phosphorus (Wang et al. 2000). From 18 cm to surface, especially from 18 to 6 cm, the contents of In-P, Re-P, and TN increased

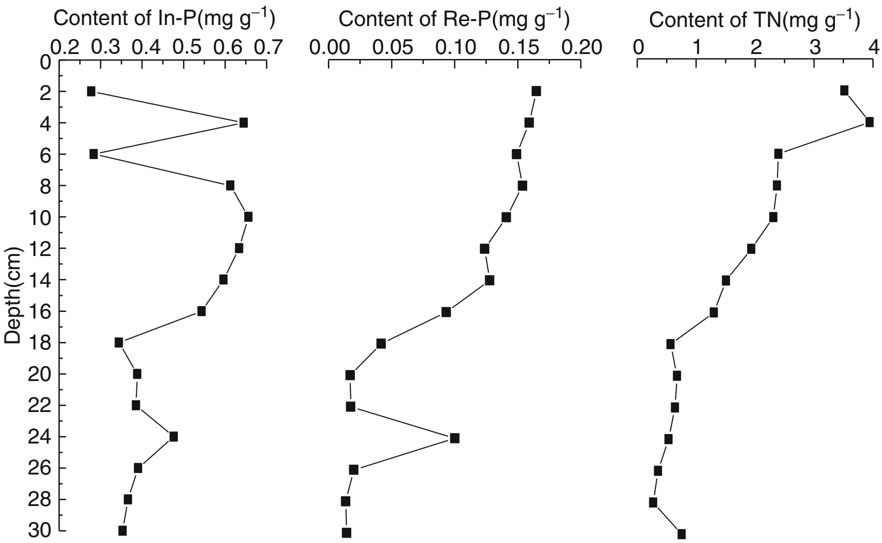

Fig. 19.9 Vertical distribution of Total In-P, Re-P, and TN at the sampling site S2

significantly. Sediment thickness and age of reservoir can be used to roughly estimate the periods represented by this sediment. The reservoir was 30 years old, the sediment thickness was 22 cm, and the annual deposition rate was thus about 0.73 cm year^{-1}. Therefore, the 18–6 cm layer of this sediment represented approximately the period of 1981–1997. From the viewpoint of the rapid development of economy and urbanization, the amount of water pumped from the polluted river increased with quantity of water supplied, and the annual deposition of suspended particulate and contaminants increased. The In-P, Re-P, and TN contents of the sediment deposited from the mid-1980s to year 2000 confirmed this rapid increase in pollution.

19.3.4 Estimation of Total Sediment Volume and Nutrients

19.3.4.1 Sediment Distribution

The longest of the seven cores was S4 (about 45 cm), and the shortest was upstream (about 4 cm). The sediment volumes at the seven regions were estimated (Table 19.2) based on the thickness of sediment and the percentage of the areas of the sampling sites. Combining with the wet and dry weights of single sediment samples, we obtained the wet and dry weights of sediment per unit area at seven sampling sites and estimated the wet and dry weights of sediment of the seven regions (Table 19.2).

Table 19.2 Sediment volume and weight at seven sampling zones in Dajingshan Reservoir

Sampling sites	1	2	3	4	5	6	7
Depth of sediment (cm)	4	12	25	45	35	35	25
Percentage (%)	20	20	15	10	10	10	15
Areas of sediment (m^2)	44,000	44,000	33,000	22,000	22,000	22,000	33,000
Volume of sediment (m^3)	1,760	5,280	8,250	9,900	7,700	7,700	8,250
Wet weight of unit area ($g\ m^{-2}$)	42,462.9	123,850.0	286,174.5	495,399.9	393,412.5	385,073.1	279,996.7
Dry weight of unit area ($g\ m^{-2}$)	33,970.3	86,695.0	214,630.9	297,239.9	236,047.6	231,043.9	209,997.5
Wet weight (t)	1,868.4	5,449.4	9,443.7	10,898.8	8,655.1	8,471.6	9,239.9
Dry weight (t)	1,494.3	3,814.5	7,082.7	6,539.3	5,193.0	5,082.9	6,929.8

Table 19.3 Distribution of total In-P, Re-P, and TN in the sediment of Dajingshan Reservoir

Sampling sites	1	2	3	4	5	6	7
Dry weight (t)	1,494.3	3,814.5	7,082.7	6,539.3	5,193.0	5,082.9	6,929.8
Content of In-P ($mg\ g^{-1}$)	0.3878	0.4793	0.6442	1.0198	0.4793	0.4793	0.4793
Content of Re-P ($mg\ g^{-1}$)	0.0319	0.1006	0.1860	0.3651	0.1006	0.1006	0.1006
Content of TN ($mg\ g^{-1}$)	0.9558	1.8177	1.8759	2.1550	1.8177	1.8177	1.8177
Amount of In-P (kg)	579.5	1,828.3	4,562.7	6,668.8	2,489.0	2,436.2	3,321.5
Amount of Re-P (kg)	47.7	383.7	1,317.4	2,387.5	522.4	511.3	697.1
Amount of TN (kg)	1,428.3	6,933.6	13,286.4	14,092.2	9,439.3	9,239.2	12,596.3

The total volume of sediment was 48,840 m^3, their wet weight was about 54,027 t, the dry weight about 36,137 t, and the dry weight of average annual accumulation was 1,025 t.

19.3.4.2 Total Nitrogen, Total Inorganic Phosphorus, and Releasable Phosphorus

S2, S5, S6, and S7 were located in the center of the reservoir and had similar depositing characteristics. The In-P and TN contents at S5, S6, and S7 were estimated by using the vertical profiles at S2. The estimated contents of In-P, Re-P, and TN are shown in Table 19.3.

There are two methods of estimating sediments. The first one is to estimate their volume according to the nature and distribution characteristics of sediment directly, as used in Taihu Lake (Fan et al. 2000; Yuan et al. 2003). The second method is by measuring the rate of sediment deposition to obtain the amount of sediment indirectly, like in the Ganges–Brahmaputra Delta (Kuehl et al. 1997), Yangtze River Delta (Chen et al. 1985) and Yellow River Estuary (Shi et al. 2003). We used the first method to estimate Dajingshan Reservoir's sediment; weight of the sediment is in weight per unit area and not weight per unit volume as usually used. The average annual deposited weights of total In-P, TN, and Re-P were 729, 2,234 and

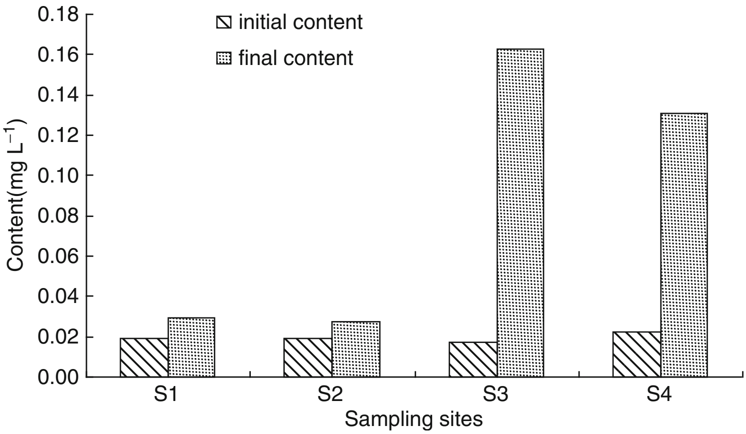

Fig. 19.10 Content of total soluble phosphorus in the overlying water at the beginning and the end of the in situ experiment

Table 19.4 Phosphorus release from the sediments in Dajingshan Reservoir

Sampling sites	Area of sediment(m^2)	Release rate ($mg{\cdot}m^{-2}{\cdot}day^{-1}$)	Amount of release(t)	Annual amount of release (t)
S1	66,000	0.31	0.006138	0.109
S2	66,000	0.23	0.004554	
S3	55,000	4.09	0.067485	
S4	33,000	3.07	0.030393	

196 kg year^{-1}, respectively. The particle composition, permeability and deposition time can affect the sediment bulk density and result in a larger error of estimation (Mahmood 1987). The use of weight per unit area could reduce the estimation errors. Although this estimation has some deviation, it nevertheless provides a rough estimation of the total sediment volume.

19.3.4.3 In Situ Experiment of Phosphorus Release

The phosphorus release rates at S1, S2, S3, and S4 were 0.31, 0.23, 4.09, and 3.07 $mg{\cdot}m^{-2}{\cdot}d^{-1}$, respectively. Release rate near the dam (S3 and S4) was significantly higher than that at the other sites (S1and S2) (Fig. 19.10). By combining the sediment areas of each sampling site, the total amount of phosphorus release from the sediment was about 109 kg, and 90% of this amount was from the zone near the dam (Table 19.4).

In the in situ experiment, the redox potential of water in all four experimental devices increased, but was below 200 mV. The lower redox potential deoxygenized the high valence metal compounds, which led to a release of phosphorus (Nowlin et al. 2005). The redox potential of overlying water indicated that the sediment and overlying water affected each other. The water pH values in the four experimental devices evidently decreased, indicating that the sediment was important to pH (Table 19.4).

19.4 Conclusions

The sediment distribution in Dajingshan Reservoir is different from that in reservoirs that depend on inflow of rivers. The thickest sediment, found near the inlet for pumped water was about 45 cm. Sediment was concentrated in the zone from inlet to outlet. Oc-P was the most abundant species of inorganic phosphorus, followed by Ca-P and Al-P. The content of exchangeable phosphorus was negligible. The In-P distribution had a conspicuous spatial heterogeneity. Fe-P and Al-P had a strong vertical gradient in the central reservoir, which rapidly decreased with sediment depth. Fe-P was the major species of the releasable phosphorus.

The distribution of TN, total In-P, and Re-P showed a clear spatial gradient, corresponding to the hydrodynamics. The inlet near the dam had the highest contents of Re-P and anoxic conditions prompting phosphorus to be released. As a consequence, this site contributed more than 90% to annual total phosphorus, and became the major region for internal phosphorus loading. Vertically, the contents increased between the 18 and 6 cm of sediment, corresponding to the period of the mid-1980s to 2000 and the quick industrialization and urbanization in and around Zhuhai city.

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Chapter 20
Controlling Cyanobacteria and Its Effectiveness: An Evaluation in Four Reservoirs for Drinking Water Supply

Kaihong Lu, Chunhua Jin, and Jinyong Zhu

Abstract Algal blooms, a common symptom of eutrophication, do not only affect structure and function of reservoir ecosystems but also public health. At present, 18.9% of large- or middle-sized reservoirs larger than 10^7 m^3 in Zhejiang province are medium-eutrophic, and 81.1% are eutrophic or very eutrophic. Cyanobacterial blooms occur frequently, and cause great problems to water quality management. The present paper takes Qiaodun reservoir in Wunzhou, Liduhu reservoir and Meihu reservoir in Ningbo, and Duihekou reservoir in Huzou as examples, located in south, middle, north Zhejiang province, central-east China. It discusses a strategy for regulating algal blooms in these four reservoirs, and analyzes effects on structure and succession of plankton and on the dynamics of physical-chemical variates. The results show that emergency measures, like spraying alum plasma, biomanipulation with phytoplanktivorous fish, biological purification using aquatic plants, and mechanical dredging, may control algal blooms to some extent, whether used singly or in combination. Ameliorated alum plasma combined with fish biomanipulation produced the best integrated ecological effects in Qiaodun reservoir. While dredging can improve water quality in the short term, it is not always effective and may even worsen the situation.

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20.1 Introduction

Zhejiang province, situated in central east China, has a mild climate and numerous rivers. By the end of 2000, 3,804 reservoirs with a storage capacity of more than one million cubic meters had been built, including 23 large-sized ones (>10^7 m^3) Total storage capacity now reaches 35.078 billion cubic meters. These reservoirs have an important role in maintaining water supply.

However, with the development of industrial and agricultural production, population growth and accelerating urbanization, water pollution has become increasingly serious. In many towns, the urban domestic water supply has shifted from river-based to reservoir-based. An investigation of 90 large reservoirs in 1995–2003 found that no reservoir was of the oligotrophic type, while 81.1% were eutrophic (Table 20.1).

Large-scale and frequent cyanobacterial blooms have become a major problem in water quality management in reservoirs in the past decade, especially in the case of urban drinking water. Controls of eutrophication and algal blooms mainly include the treatment of internal and external contaminants, especially from non-point sources. The variety of micro-landscape structures controlling non-point pollution is considered a hot topic and an advancing research front internationally (Johnston 1991; Muscutt et al. 1993; Jansson et al. 1994; Li 1997a, b). Physico-chemical methods involving dredging, drainage and scouring, adsorption precipitation, sediment shelter and chemical elimination, etc., have been applied to control internal pollution (Lu et al. 1992; Zhao and Wang 2000; Pu et al. 2001; Zhong et al. 2009a, b). Biological methods involving microbiota, transplantation and restoration of higher water plants, and stocking of filter-feeding fish, have also been applied (Shapiro et al. 1975; Liu and Xie 1999; Li et al. 2000; Wang et al. 1998; Lu et al. 2006). However, quality requirements of potable water restrict the application of some methods. Reducing loadings of nutrients are suggested as major prevention and treatment measures against reservoir eutrophication. For example, rapidly reducing nutrients available for algal growth may not be effective in plain reservoirs in the short term (Sas 1989). In this article, we discuss methods of physical ecological engineering and ecological technology, and summarize the effect of treatments in four large- and medium-sized drinking water reservoirs (Fig. 20.1, Table 20.2): Qiaodun reservoir, Liduhu, Duihekou, and Meihu reservoirs. These reservoirs are located in the southern, central, and northern parts of Zhejiang province.

Table 20.1 Status of reservoir eutrophication in Zhejiang province

Trophic type	Oligotrophic	Mesotrophic	Eutrophic	Hypertrophic
Reservoir number	0	17	42	31
Proportion of total number (%)	0	18.9	46.7	34.4
Reservoir storage capacity($\times 10^4$ m^3)	0	800,761	317,042	145,293
Proportion of total storage capacity (%)	0	63.4	25.1	11.5

20.2 Materials and Methods

20.2.1 Reservoirs Studied

20.2.1.1 Natural Environmental Conditions of Four Drinking Water Reservoirs

Qiaodun reservoir is located in Cangnan County, southern Zhejiang province. There are three towns in the upstream zone of the river and surrounding areas, which are principally agricultural production centers. Main nutrients come from agricultural wastewater and domestic wastewaters from the residents. This reservoir was established for water supply, power generation, flood control, irrigation and other purposes, but currently, it provides most drinking water for Cangnan residents.

Liduhu reservoir, located in Ningbo City, is the largest drinking water reservoir in Cixi county. The hilly vegetation in the catchment has a simple community structure dominated by economic crops such as *Phyllostachys praecox*, *Phyllostachys pubescens* and *Myrica rubra*. In 1998 and thereafter, the villagers applied about 15,000 t of pig manure and bran to the surrounding areas, using fermentation of this organic fertilizer to improve production of bamboo shoots. With rain, some of the fertilizer entered the reservoir and became a major source of nutrients.

Duihekou reservoir, located in Deqing County, Huzhou City, is a large-sized reservoir used for water supply, flood control, irrigation, and power generation. At present, it has become an important drinking water source for a population of 430,000. Vegetation in the catchment is mainly mountain forest. There are four towns in the upstream zone of its feeding rivers. Main nutrient loadings are contributed by agricultural waste-water, domestic waste, production wastewater, and solid waste from about 50 factories of boiled bamboo shoots.

Meihu reservoir, located in Ningbo City of eastern Zhejiang province, is a medium-sized reservoir used for irrigation, flood control, water supply, and power generation. Its major pollution sources are domestic pollution and forestry

Table 20.2 Environmental conditions of four reservoirs

	Qiaodun reservoir	Liduhu reservoir	Duihekou reservoir	Meihu reservoir
Catchment area (km^2)	138	51	165	23.5
Normal water level (m)	54	17.5	46	22
Reservoir area (km^2)	2.63	1.86	5.0	1.175
Total storage capacity ($\times 10^4$ m^3)	8,433	1,668	4,650	1,603
Annual average discharge ($\times 10^4$ m^3)	21,000	1,400	13,700	1,300
Population of catchment	35,000	3,575	23,356	4,142

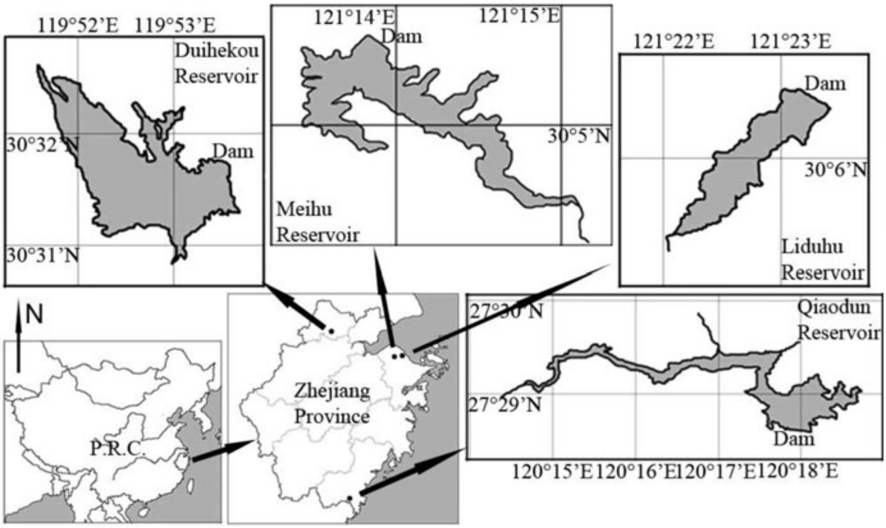

Fig. 20.1 Sites of four reservoirs in Zhejiang province

pollution. After 2004, due to the scarcity of water resources in Cixi county, organic pollution was diverted from the Yao River to Meihu reservoir, deteriorating its water quality.

20.2.1.2 Cyanobacterial Blooms and Water Quality

Qiaodun Reservoir

The first *Microcystis* bloom in Qiandun reservoir was observed in July 1997, with highest density 125.50×10^6 cells L^{-1}. It covered a limited area but occurred again, this time all over the reservoir, in April 1998. Surface scums of Cyanobacteria a few centimeters thick were observed on sunny days. Transparency (SD) decreased from 2.0–3.0 to 0.3–0.9 m during blooming. Composition of phytoplankton was very simple in all zones of the reservoir. Only four phyla, 12 genera, and 16 species were identified in 15 samples (Lu et al. 2002), and *Microcystis aeruginosa* and *M. flos-aquae* dominated. The average cyanobacterial density was 162.21×10^6 cells L^{-1}. The highest cyanobacterial density was 234.51×10^6 cells L^{-1} in downstream surface water.

During the bloom (lasting from July 1997 to August 1998), TN concentration in August 1998 averaged 0.649 mg L^{-1} with maximum 2.61 mg L^{-1}, while TP concentration in April 1998 averaged 0.027 mg L^{-1} and the maximum TP was 0.055 mg L^{-1}. The mass ratio of nitrogen and phosphorus was 24:1. The average annual TP was 0.041 mg L^{-1} in the main inlet and 0.021 mg L^{-1} in the main outlet. Nearly 2,500 kg phosphorus was loaded to the reservoir each year.

Liduhu Reservoir

Since 1997 cyanobacterial blooms have occurred frequently, and the scale of the bloom increased year after year. The density of Cyanobacteria reached a peak and bloomed throughout the reservoir until August 2001. Total density of phytoplankton reached 164.9×10^6 cells L^{-1}, and cell density of Cyanobacteria was 159.3×10^6 cells L^{-1}, accounting for 96.6% of total phytoplankton. Chlorophyta, Bacillariophyta, Cryptophyta were 3.67×10^{6} cells L^{-1}, 1.86×10^6 cells L^{-1}, and 0.09×10^6 cells L^{-1}, respectively. These three groups comprised only 3.4% of the total density in August. The water bloom was dominated by *M. aeruginosa*, *Oscillatoria*, and *Phormidium*.

TN concentration was between 1.38 and 2.09 mg L^{-1} from June to October 2001; average TN was 1.74 mg L^{-1}. TP concentration was between 0.012 and 0.047 mg L^{-1}; the average TP was 0.032 mg L^{-1}. COD_{Mn} was between 2.34 and 2.46 mg L^{-1}. Average SD of the whole reservoir was reduced to 0.58 m at the peak of the cyanobacterial bloom in August.

Duihekou Reservoir

The first cyanobacterial bloom was observed in 1993, and thereafter the bloom occurred frequently, but lasted only for short periods of about 7 days. However, a large-scale bloom occurred in May 2000, lasting nearly 4 months. The Cyanobacteria flourished again in early December of that year. The bloom peaked in April 2001, but was alleviated by the spring flood. However, the phytoplankton biomass was soaring once again in November, a pattern that repeated itself in 2002. The blooming species for many years were mainly *Anabaena spiroides*, accompanied by *Phormidium tenus*. In April 2001 and April 2002, the density of phytoplankton was 124.90×10^6 and 136.38×10^6 cells L^{-1}, respectively, and Cyanobacteria contributed 92.9% and 95.6%, respectively.

TN concentration varied between 0.590 and 1.859 mg L^{-1}, with a mean of 1.03 mg L^{-1} from 2000 to 2002; TP varied from 0.022 to 0.140 mg L^{-1}, with average 0.062 mg L^{-1}. Secchi Depth (SD) fluctuated between 0.66 and 1.85 m. SD declined to 0.37 m in the blooming area.

Meihu Reservoir

After receiving water from the Yao River (with very low water quality), the water quality of Meihu reservoir sharply decreased from July 2004 to August 2007. When the water Diversion Project of Tangpu started in August 2007, the waste-water from the Yao River was diverted. However, the eutrophic state level did not immediately end. The TN concentration was between 2.11 and 2.99 mg L^{-1}, and ranged from 0.10 to 0.11 mg L^{-1} from August to September 2007. A bloom first occurred in the summer of that year, with a large amount of Cyanobacteria not limited to surface water but distributed in all layers, and SD was only 0.45 m. The average phytoplankton density was up to 178×10^6 cells L^{-1} in surface' water. Ninety-five percent of phytoplankton was contributed by *M. aeruginosa* and *M. wesenbergii*.

20.2.2 Controlling Cyanobacterial Blooms

20.2.2.1 Qiaodun Reservoir

Integrated management measures, combining a "physicochemical temporary solution" with a "biological permanent cure," were carried out in 1998. These involved sprinkling the lake with ameliorated alum plasma in order to remove the algal emergency, and stocking of filter-feeding fish for improving the water quality.

The "ameliorated alum plasma" was a white precipitate of fine particles obtained as a by-product of processing alum in Wenzhou. The ameliorated alum plasma was mixed with water in a 15 m^3 mixing tank in batches during September 1998. The mixture was sprayed directly onto the surface of water by a fire pump, and the operating platform was tugged all over the reservoir. One scanning took 3 days, and the total amount applied was 10.2×10^4 kg, averaging 38.8 g per square meter.

From December 1998 to March 1999, 2,850 kg of fingerlings of bighead carp (average fingerling weight 18.5 g) and 1,500 kg of fingerlings of silver carp (average fingerling weight 21.9 g) were stocked. This amounts to 1,200 specimens per hectare, and the ratio of silver carp and bighead carp was 2.4:1. The same numbers and proportion of silver carp and bighead carp were annually stocked after 1999.

20.2.2.2 Liduhu Reservoir

Experiments on controlling blooms were carried out by mixing surface-bottom water (Ma et al. 2002) and by establishing an artificial compound plant community in 2001 and 2003.

During blooms, bottom water (between 16°C and 19°C) was pumped to the surface by three 70 m^3 h^{-1} sand-suction pumps every day from 6 a.m. to 5 p.m. from July 21 to August 30, 2001, and was sprinkled on the surface to destroy the environment of the blue-green algae.

An artificial compound plant community was composed of the emergent plants *Phragmites communis, Acorus calamus, Alternanthera philoxeroides,* the terrestrial (floating bed) plant *Canna* (dwarf variety) and the submerged plants *Ceratophyllum demersum* and *Hydrilla verticillata*. Reeds (5,100 plants) were directly planted along the reservoir shore at a water depth of less than 0.5 m on April 10–12, 2003. The reeds covered an area of 600 m^2, with a density of 8.5 ind m^{-2}. *Canna* and *Acorus* were cultivated in floating beds from April 28 till May 5, 2003. The cultivation raft was a 1.5 m $\times$ 1.0 m $\times$ 0.1 m foam board with 12 planting holes (8 cm aperture) 36 cm apart. The planting area of 4,160 individual *Canna* covered

520 m^2, and that of 1,600 individual *Acorus* covered an area of 200 m^2. The planting density of the two plants was 8 ind m^{-2}. Three thousand kilograms *of Alternanthera philoxeroides* was planted in an 30 m × 20 m × 1 m net cage in May 29, 2003. Fifteen kilograms of *Ceratophyllum demersum* and *Hydrilla verticillata* were fixed on the mesh in the euphotic layer.

20.2.2.3 Duihekou Reservoir

Here, controlling basin pollution and biomanipulation were attempted by the stocking of silver carp and bighead carp. This reservoir had stocked silver carp, bighead carp, black carp, grass carp, bluntnose black bream, and other economic fish from 1962 to 1988, with an average annual output of 0.1×10^6 kg. Fish stocking was stopped because of serious illegal fishing but the reservoir authority tried to restore stocking fish for reducing algal proliferation in the winter of 2001. In the winter of 2001 and 2002, 170,000 fingerlings of silver carp and 200,000 fingerlings of bighead carp were introduced, respectively. The amount of stocking was 525 (silver carp) and 617 (bighead carp) individuals per hectare. The number ratio of silver carp and bighead carp was 2:1.

20.2.2.4 Meihu Reservoir

Dredging of this reservoir was conducted in February to June 2008. During dredging, the water level was lowered to dead storage level. Dredging was conducted by a cutter suction dredger (beaver-type), and the sludge was transported to the stack through totally enclosed pipelines by pressure pumps. The amount of dredging was approximately 0.3×10^6 m^3 in total. For further control of cyanobacterial biomass, the surface water was mixed with sandy water from the bottom mechanically after dredging in July 2008 (Ma et al. 2002). The water mixing was aimed to reduce temperature of surface water and underwater light climate, thereby destroying thermal stratification in order to inhibit cyanobacterial development.

20.3 Results

20.3.1 Qiaodun Reservoir

20.3.1.1 Effects of Integrated Management on Water Quality

The main variables of water quality were compared before (1998.8) and after (1998.10) the application of ameliorated alum plasma (Table 20.3). It was found that plasma had positively influenced water quality: SD, TN, and TP changed greatly within a week after application. SD increased by 2.2 m and TN reduced by an average of 77.9%. TP concentration in surface water decreased by 57.4%, but increased in the middle and lower layers because of the presence of deposition coagulation. Dissolved oxygen (DO) near the bottom improved significantly.

Some variables of water quality increased to some degree in 1999 (Table 20.3). The reason may be related to the vertical circulation of the upper and lower water layers in winter and spring so that nutrients can migrate to the surface in interstitial water of sediment and accumulate in the lower water layer. However, water transparency remained at more than 2 m, a reasonable level, because of lower density of phytoplankton. With the development of various organisms in the water, physical and chemical variables showed a marked improvement; annual average transparency rose to more than 3 m, TN was reduced by 61.1%, and TP decreased by 59.4% compared to August 1998. The trophic state index (TSI_C) (Li and Zhang 1993) of the reservoir changed from meso-eutrophic before to meso-oligotrophic after the treatment (Fig. 20.2).

20.3.1.2 Control of Cyanobacterial Blooms by Integrated Management

Cyanobacterial blooms almost disappeared after spraying ameliorated alum plasma in September 1998. Although the bloom was observed again from the July to September after stocking filter-feeding fish, its scale and density were far less than in 1998 (Table 20.4). The percentage Cyanobacteria decreased from 99.2% in 1998 to 31.5%, and the percentage of green algae and diatoms increased from <1% to 26.1% and 34.8% in August 2000. Cyanobacterial blooms have not re-appeared in Qiaodun reservoir in four successive years since 2000.

Due to the increasing predation pressure of fish, the absolute number of zooplankton in the reservoir has not increased after treatment, but its composition and biomass changed. The percentage of protozoa decreased significantly, while rotifers and small crustaceans increased. The number of rotifer species rose from 8 to 11 in

Table 20.3 Comparison of main physical and chemical variables of Qiaodun reservoir between before and after harnessing

Time	Water layer	SD (m)	pH	TN ($mg\ L^{-1}$)	TP ($mg\ L^{-1}$)	NH_3–N ($mg\ L^{-1}$)	COD_{Mn} ($mg\ L^{-1}$)	DO ($mg\ L^{-1}$)
Aug. 1998	Surface	0.8	8.12	1.30	0.030	0.37	2.88	7.40
	Middle	–	6.50	1.36	0.010	0.21	1.18	2.34
	Bottom	–	6.49	1.46	0.015	0.16	1.27	1.37
Oct. 1998	Surface	3.0	7.11	0.27	<0.01	0.09	1.57	7.23
	Middle	–	6.99	0.39	0.020	0.11	1.74	7.04
	Bottom	–	6.70	0.30	0.020	0.11	1.41	6.09
Aug. 1999	Surface	2.3	8.36	0.92	0.027	0.25	3.20	9.10
	Middle	–	7.34	0.61	0.041	0.37	1.75	6.43
	Bottom	–	6.93	0.53	0.016	0.14	1.50	5.82
Aug. 2000	Surface	3.2	7.86	0.50	<0.01	0.10	2.33	6.70
	Middle	–	7.17	0.53	<0.01	0.10	1.81	5.13
	Bottom	–	7.00	0.59	<0.01	0.12	1.69	4.62

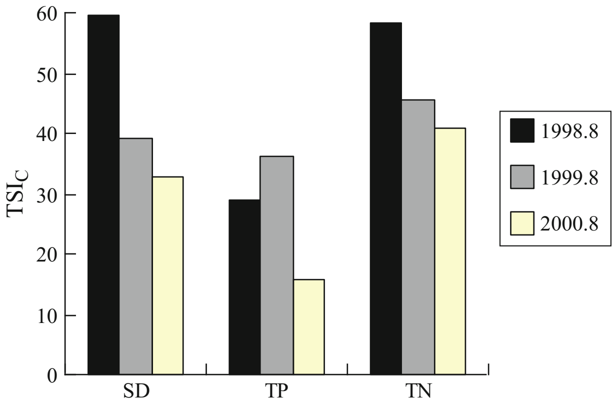

Fig. 20.2 TSIc of Qiaodun reservoir before and after harnessing

Table 20.4 Comparison of quantity composition of phytoplankton in Qiaodun reservoir between before and after treating with alum plasma ($\times 10^{-4}$ cells L^{-1})

Time	*Microcystis*	Cyanophyta	Chlorophyta	Bacillariophyta	Cryptophyta	Total
Aug. 1998	9,805.5	10,435.6	50.0	36.9	0	10,522.5
Oct. 1998	148.5	287.1	77.2	109.5	56.0	529.8
Aug. 1999	1,874.9	2,227.5	61.1	141.3	8.8	2,465.6
Aug. 2000	81.5	143.3	118.7	158.6	8.6	455.2

the quantitative samples, and Cladocera species from 2 to 5. The diversity of the community and the complexity of its trophic structure increased substantially.

20.3.2 Liduhu Reservoir

20.3.2.1 Effect of Algal Removal by Mixing Surface-Bottom Water Mechanically

Spraying cold sandy water pumped from the bottom of reservoir effectively removed algal biomass. In particular, it greatly improved transparency and reduced TP (Table 20.5).

20.3.2.2 Effect of an Artificial Plant Community

Canna and *Alternanthera philoxeroides* grew well during the experiment, but the growth phase of *Canna* was limited from May to July. The planting area of submerged plant *Ceratophyllum demersum* and *Hydrilla verticillata* was small, but certainly increased. *Ceratophyllum* grew vigorously in spring and summer, and *H. verticillata* grew well in the later part of the experiment. Although *Acorus gramineus* could

Table 20.5 Comparison of main physical and chemical variables of Liduhu reservoir between before and after pumping bottom water

Variables	Jul. 2001	Sep. 2001	Fluctuation (%)
WT (°C)	28.0	26.0	/
pH	9.53	7.72	−18.9
SD (m)	0.58	1.15	+98.3
TN (mg L^{-1})	1.49	1.58	+6.0
TP (mg L^{-1})	0.012	0.002	−83.3
COD_{Mn} (mg L^{-1})	2.80	2.36	−15.8
Chl*a* (mg m^{-3})	30.0	8.12	−72.9
Phytoplankton ($\times 10^4$cells L^{-1})	13,780	2,480	−81.9

Table 20.6 Growth of three emergent plants and their N, P uptake

	Canna	*Alternanthera philoxeroides*	*Acorus calamus*	Total
Initial biomass (kg)	182	3,000	39	3,221
Final biomass (kg)	3,224	5,760	587	9,571
Biomass increment (kg)	3,042	2,760	548	6,350
Proportion of nitrogen (%)	0.1	0.32	0.1	/
Total removal of nitrogen (kg)	3.04	8.83	0.55	12.42
Proportion of phosphorus (%)	0.018	0.058	0.017	/
Total removal of phosphorus (kg)	0.548	1.601	0.093	2.242

survive in floating beds, it grew slowly, while lack of water killed the reeds because of the long-term shortage of water in the drawdown area. After consideration, it seems more feasible to transplant *A. philoxeroides* in the reservoir (Table 20.6).

The results showed that the six selected variables in the test area responded much better than in the non-experimental area (Table 20.7). The Carlson trophic status index (TSI) (Yang et al. 2001) gave a value of 46.8 in the experiment area and 51.4 in the non-experiment area.

Not only was total phytoplankton in the experimental area significantly smaller than that in the non-experiment area, but also the number and proportion of blue-green algae was lower than in the non-experiment area (Fig. 20.3). The proportion of blue-green algae, green algae, and diatoms in the experiment area was 58.8%, 13.6%, and 25.0%, against 72.7%, 7.2%, and 17.3%, respectively. The establishment of an artificial compound plant community therefore was significant in optimizing the community structure of phytoplankton, and increasing its biodiversity.

20.3.2.3 Duihekou Reservoir

Effect of Stocking Fish on Trophic Level

A comparison of the main physical and chemical variables from 2000 to 2003 showed that water transparency was increasing yearly, the chlorophyll *a* concentration having decreased significantly, while nitrogen, phosphorus and permanganate

Table 20.7 Comparison of physical-chemical and biological characters between plant experimental area and non-experimental area

	Non-experimental area		Experimental area	
TN (mg L^{-1})	1.38	(1.72–1.18)	0.99	(1.29–0.34)
TP (mg L^{-1})	0.051	(0.087–0.022)	0.04	(0.072–0.015)
COD_{Mn} (mg L^{-1})	2.45	(3.22–1.83)	2.17	(2.95–1.76)
Chl*a* (mg m^{-3})	11.9	(22.0–5.49)	8.37	(13.4–3.28)
Phytoplankton ($\times 10^4$cells L^{-1})	1,936	(3,060–1,270)	1,302	(2,500–460)
SD (m)	0.95	(1.16–0.64)	1.31	(1.88–0.97)

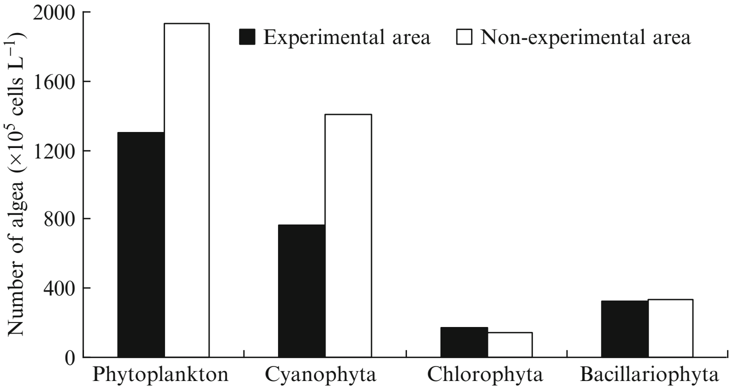

Fig. 20.3 Composition of phytoplankton between plant experimental area and non-experimental area

changed little after fish stocking at the end of 2001 (Table 20.8). TSI (Σ) of 2001, 2002, and 2003 was 54.0, 50.8, and 46.1, respectively. There was improvement in the trophic status (mainly affected by changes of water transparency and chlorophyll *a*), but the trophic status of the reservoir still remained mesoeutrophic.

Control of Cyanobacterial Blooms by Stocking Fish

After stocking fish in Duihekou reservoir in the winter of 2001, algal blooms re-occurred in 2002, but disappeared in 2003. The dominant species and biomass peak of blue-greens changed considerably (Table 20.9). During the algal blooms of July 2003, and although the density of phytoplankton was higher than in April 2001 and April 2002, the density of dominant *P. tenus* was lower than that of *A. spiroides*, algal biomass was significantly reduced, and the bloom disappeared. However, algal biomass remained high. The average annual density of phytoplankton was 54.28×10^6cells L^{-1}, corresponding to an average annual biomass of 8.061 mg L^{-1}. The dominant species were mainly *P. tenus* and *M. incerta*. Their biomass and dominance shifted with season: *Spondylosium planum* and *Gonyostomum* sp were dominant in spring, *P. tenus* in summer, and *Synedra acus* in autumn and winter.

Table 20.8 Comparison of main physical-chemical factors in Duihekou reservoir from 2001 to 2003

Year	COD_{Mn} (mg L^{-1})	TN (mg L^{-1})	TP (mg L^{-1})	Chl*a* (mg m^{-3})	SD (m)
2001	4.08 (2.87–5.49)	1.12 (0.62–1.86)	0.063 (0.025–0.140)	21.70 (3.41–35.85)	1.22 (0.37–1.76)
2002	3.89 (3.12–4.89)	0.88 (0.59–1.22)	0.058 (0.022–0.138)	15.65 (2.18–28.71)	1.48 (0.73–1.85)
2003	3.96 (3.09–4.79)	0.80 (0.37–1.16)	0.061 (0.020–0.142)	6.76 (0.68–16.71)	1.63 (0.95–2.45)

Table 20.9 Peak composition of phytoplankton in Duihekou reservoir from 2001 to 2003

Year	Month of peak	Number of cells ($\times$104 cells L^{-1})	Biomass of algae (mg L^{-1})	Dominant species
2001	April	12,490.5	14.550	*Anabaena spiroides*
	November	8,328	10.762	*Anabaena spiroides*
2002	April	13,638.4	15.826	*Anabaena spiroides*
	November	7,386.0	9.97	*Anabaena spiroides*
2003	July	18,401.2	10.225	*Phorimidium tenus*

Table 20.10 Comparison of main environmental variables of Meihu reservoir between before and after dredging and pumping bottom water

Variables	2007.8	2007.9	2008.8	2008.9
WT (°C)	31.4	28.8	30.9	26.8
DO	7.98	6.25	1.92	2.37
pH	8.66	9.12	9.24	8.67
SD (m)	0.65	0.63	0.55	0.35
TN (mg L^{-1})	2.99	2.11	1.69	1.74
TP (mg L^{-1})	0.1135	0.1055	0.0445	0.0630
COD (mg L^{-1})	6.78	10.43	3.28	3.36

20.3.3 Meihu Reservoir

20.3.3.1 Effects of Physical Measures on Water Quality

By comparing the environmental variables before (2007.8 and 9) and after (2008.8 and 9) dredging and mechanical algae removal, we found that the physical measures improved the water quality (Table 20.10). After treatment, TN was reduced by an average of 32.7%, TP 50.9%, and COD 61.4%. However, DO and SD declined because of the re-suspension of organic matter and sediment and of dissolved oxygen consumption by reduced organic matter. DO decreased from 6.25–7.98 to 1.92–2.37.

20.3.3.2 Effects of Physical Measures of Algal Removal on Cyanobacterial Blooms

The cloudy bloom almost disappeared after dredging and mechanical mixing of surface-bottom water. In fact, density of blue-green algae increased rather than decreased, although chlorophyll *a* concentration declined significantly (Table 20.11).

Table 20.11 Comparison of quantity composition and biomass of phytoplankton in Meihu reservoir between before and after dredging and pumping bottom water

Year	Cyanophyta		Chlorophyta		Cryptophyta		Bacillariophyta		Total	
	Number of alga cells ($\times 10^6$ cells L^{-1})	Chl*a* ($\mu g\ L^{-1}$)	Density of alga cells ($\times 10^6$ cells L^{-1})	Chl*a* ($\mu g\ L^{-1}$)	Density of alga cells ($\times 10^6$ cells L^{-1})	Chl*a* ($\mu g\ L^{-1}$)	Density of alga cells ($\times 10^6$ cells L^{-1})	Chl*a* ($\mu g\ L^{-1}$)	Density of alga cells ($\times 10^6$ cells L^{-1})	Chl*a* ($\mu g\ L^{-1}$)
2007.8	16,329	55.84	8.19	0.215	1.23	6.11	1.29	0	174	62.17
2007.9	178.93	61.865	1.18	0	1.54	7.145	0.59	0	182.24	69.015
2008.8	277.56	7.67	0.54	0.23	0.59	0.16	0.67	0	279.40	8.06
2008.9	237.89	9.52	0	0	0.09	0.18	0.67	0	238.65	9.7

In terms of phytoplankton community composition, the dominant species changed from *M. aeruginosa* and *M. wesenbergii* to *Synechocystis willei*, with the other phyla of algae further reduced. The cyanobacterial blooms were not significantly controlled. Changes in the density and richness of zooplankton were related to sediment resuspension. The proportion of rotifers decreased, while the proportion of protozoans, represented by *Difflugia*, increased markedly.

20.4 Discussion

Eutrophication and algal blooms caused by water pollution have become a global nuisance. The last 40 years, reseracher from many countries have attempted research and practices in an attempt to find effective methods to prevent the spread of this kind of pollution. However, in eutrophic water bodies, any single measure always seems insufficient to control phytoplankton density (Sas 1989; Wang et al. 2002). The processes behind eutrophication and blue-green algal blooms are complicated, involving many factors. We should adapt to local conditions and adopt specific measures to control blooms. Here, we developed different interventions in accordance with the conditions of four drinking water reservoirs in Zhejiang province. The results show that all four kinds of technologies, namely a physical-chemical method, a biomanipulation with phytoplanktivorous fish, and biological purification using aquatic plants, and dredging, can be effective when implemented singly or in combination.

20.4.1 Algal Removal by Alum Plasma and Its Ecological Implications

Physical and chemical methods are often expensive and their effects are short-lived, and there are always some potential negative impacts of secondary pollution. The methods are therefore often used only for emergency removal of algal blooms. The use of ameliorated alum plasma for emergency removal of algae in Qiaoduan reservoir was based on our two considerations. First, the reservoir functions as the sole source of drinking water for Cangnan County, and large-scale algal blooms pose a serious threat to water supply; no biological–ecological safety measures can provide an immediate solution to such a water crisis; second, Cangnan is the town of alum, and alum plasma is industrial waste, so the material is abundant and convenient. Appropriate amelioration of the waste alum plasma, turning waste into wealth, can supply economic and social benefits.

The modified alum plasma is a strong absorptive that rapidly diffuses and deposits. Tests indicated that ameliorated alum plasma is capable of removing up to 90% of planktonic algae from water, especially in a deep reservoir like Qiaodun. When blue-green algae are absorbed or settled, cell proliferation becomes difficult due to lack of light and the Cyanobacteria can be controlled in the short term. Indoor simulated experiments show that 25 ppm of alum plasma can reduce the total phosphorus concentration by 69.7–95.4% in static water. In large water bodies, total phosphorus removal varies as a result of a variety of factors such as wind and waves and impact of dissolved oxygen, but the phosphorus removal is quite obvious. In water in which phosphorus is a limiting factor, alum plasma may control excessive proliferation of planktonic algae. Experiments also showed that after spraying, the low pH of alum plasma can quickly and effectively reduce a pH increased by photosynthesis of Cyanobacteria, allowing neutral and acidic species can thrive. Furthermore, the high proportion of silicon in alum plasma is conducive to the growth and reproduction of diatoms. Unlike copper sulfate and other traditional algicides, alum plasma does not threaten the survival of fish and zooplankton directly, so there are no significant ecological risks. However, a large quantity of alum plasma is needed: the concentration of alum plasma at the surface should be above 20 ppm. Adhesive-like clouds composed of organic debris, bacteria, algae and alum plasma are easy to re-suspend in shallow water because of wind and convection of the upper and lower water layer. They join in the recirculation of materials involved in water bodies quickly after bacterial decomposition; thus, alum plasma will not achieve long-term control.

20.4.2 Ecological Regulation of Filter-Feeding Fish

Since the early work by Hrbacek et al. (1961) and Brooks and Dodson (1965), a large number of experiments have shown that a change in fish biomass and age composition will make a significant change in trophic structure and in water quality. This is the so-called "top-down effect" (Hurlbert et al. 1972; Anderson et al. 1978). Top-down effects led (Shapiro et al. 1975 and Shapiro 1990) to propose the concept of biomanipulation and cite many examples of actual observations and experiments. The former Soviet Union, Eastern Europe, Brazil, and China, and some other countries have used the stocking of silver carp as a biological measure to control blooms, and many achieved desirable results (Starling 1993; Liu and Xie 1999). However, there are also studies that suggest silver carp mainly filters zooplankton, large phytoplankton and small phytoplankton colonies. The feeding behavior of silver carp reduces predation on and nutrient competitors of micro-phytoplankton. Besides, after stocking silver carp, micro-phytoplankton may show accelerated growth because of the reproductive capacity of small algae. Thus, the total biomass of phytoplankton may increase rather than decline (Laws and Weisburd 1990; Li and Zhang 1993). On the other hand, because many bloom cyanobacteria of blooms have a protective sheath, the digestibility to silver carp and bighead carp of *Microcystis* is only 25–30% (Chen et al. 1990). Therefore the possibility of using silver carp and bighead carp to control blooms depends on our knowledge of community structure of animals and plants and their relationship. It is important to formulate stocking time and amount of silver carp and bighead carp based on the specific features of the water body under study.

The blooms in Qiaodun reservoirs, induced by high nutrients, may be related to an absence of appropriate herbivores, and intense fishing out of silver carp, bighead carp, *Pseudorasbora parva*, and other trash fish for the development of a silver fish population. After the outbreak of blooms, the regulation of filtration pressure by silver carp and bighead carp was far less obvious than before the bloom. Clearly, silver carp and bighead carp are more useful in the early phases of cyanobacterial blooms. Immature algae are easily digested by fish, and so the proliferation of blue-green algae is restricted. Therefore, a physical-chemical method such as alum plasma was useful at first. This restored the water landscape and changed the composition of aquatic communities quickly. Then, silver carp and bighead carp were introduced before the blue-green algae blooms in winter or early spring. The effect was considerable.

The dominant bloom species in the Duihekou reservoirs is *A. spiroides*, a species susceptible to digestion and absorption by silver carp. Therefore, despite the low stocking densities of fish, they control of the bloom. However, the stocking of fish also induced a miniaturization of phytoplankton species at high biomass. Because of heavy non-point source pollution and internal pollution from the sediment, the control of the reservoir's eutrophication should focus on the pollution source, the regulation of aquatic plants, and a synthesis of different techniques, in addition to a further adjustment of stocking fish species and numbers (Bian et al. 1994).

20.4.3 Ecological Regulation of Aquatic Higher Plants

Aquatic macrophytes have been highly involved in controlling eutrophication. Higher aquatic plants not only quickly absorb nutrients from water and sediment, and secrete allelochemicals that inhibit the growth of phytoplankton (Wetzel 1969), but also promote the deposition of sludge, inhibit the resuspension of sediment, and in so doing move N, P from the nutrient cycle in the reservoir to the geochemical cycle (Li 1997a, b). Many aquatic plants have the ability to absorb, accumulate, decompose and transform nitrogen, phosphorus and various heavy metals and phenol, cyanide, pesticides, and other organic compounds. The use of a plant community mosaic (Wang et al. 1998) and hydroponic terrestrial plants in floating beds (Liu and Xie 1999) in biological control of eutrophication has become a hot topic recently.

In the experiment in Liduhu reservoir, the establishment of an artificial plant community played a significant role in optimizing the composition of the phytoplankton and reducing nutrient concentration. *Eichhornia crassipes*, *Alternanthera philoxeroides*, *Typha latifolia*, *Phragmites australis*, *Oryza sativa*, *Oenanthe javanica*, *Ipomoea aqutica*, *Canna indica*, *Potamogeton distinctus*, *Potamogeton crispus*, *Hydrilla verticillata*, and others have been widely applied and show effective regulatory capacities in certain environmental conditions. Our experiment showed that it is difficult to plant reeds and other emergent plants in the riparian zone of reservoirs with strong water level fluctuations; *A. philoxeroides* is suitable to cultivate in different trophic types of reservoirs, is easy to manage, and has a high clearance for nitrogen and phosphorus. It should be a good choice to keep as a fence cultivation in certain areas. In order to prevent secondary pollution caused by the decay of such plants, harvest and utilization should be considered. The landscape of the reservoir was considered by applying artificial floating beds of *Canna* in Liduhu reservoir. Growth of the plants was successful, but the *Canna* growing season is short, and the cost of constructing and managing artificial floating beds is high. Still, the method of using mesh and other simple vectors to transplant submerged vegetation is worth further exploration.

20.4.4 Effect of Emergent Algal Removal by Mixing Surface-Bottom Water

Jones and Poplawski (1998) argue that thermal stratification of water bodies may be a key factor in the occurrence and decline of cyanobacterial blooms. Therefore, destratification artificially eliminates conditions favorable for blue-green algae, such as through a lack of oxygen at the bottom and release of large amounts of nitrogen and phosphorus from the sediment. The project of algal removal in Liduhu reservoir changed the algal growth conditions by mechanically mixing the bottom water with the surface bloom. First, the water transparency and temperature were significantly reduced by mixing. This inhibited algal photosynthesis. Second, thermal stratification was broken by the spraying and pumping of sandy water from bottom, improving dissolved oxygen conditions at the bottom. Third, the sandy bottom water adsorbed algae and suspended particles and removed algae from the euphotic zone, thereby reducing cell proliferation.

However, this method was not successful for the treatment of algal blooms in Meihu reservoir. Although the concentration of chlorophyll *a* decreased significantly, the density of Cyanobacteria increased (Table 20.11). Dominant algal species changed from large to small colonies, or even single-cells. Pu et al. (2000) reasoned that a pollutant cloud, composed of organic debris, bacteria, algae and mineral particles, formed between bottom water and sediment. The specific gravity of this cloud is slightly higher than water, and it can stay in the water for a long time to participate in the exchange of materials. Nutrients in these active pollutant clouds are much higher than in the underlying sediments. When the muddy water brings sediment particles to the surface, it actually brings these "clouds," triggering an outbreak of algae. Although the sediment particles reduced water transparency, blue-green algae contain specific pigments such as phycocyanin that absorbs green, orange and other wavelengths that are not available to other algae. Chlorophyll *a* and phycobiliprotein also capture light effective for blue-green algae (Liu 2005). Besides, blue-green algae are able to survive and function with little energy, and their growth rate is much higher than any other algae at low light conditions. Compared to *Chlorella*, blue-green algae are more responsive to low light (Mur et al. 1977). Finally, Cyanobacteria have an advantage in competing with other algae by their buoyancy regulation (Humphries and Lyne 1988),

The different response of two reservoirs to the same method may also be related to timing. In Meihu reservoir, samples were collected during the treatment, and the data reflected the period of treatment rather than the situation afterward; in Liduhu reservoir, samples were taken after the treatment. Algae has precipitated with suspended particles and water transparency increased. Therefore, this method is more effective in medium- and small-sized reservoir with less deposition. Algae removal by mixing surface-bottom water is simpler, with fewer side effects and less cost than the delivery of aluminum, iron salts, and other chemical substances. Besides, the machinery and equipment can be re-used. The method is worth further testing and promoting as a short-term emergency measure to remove algae.

20.4.5 Dredging and the Control of Blue-Green Algae Outbreaks

After dredging Meihu reservoir, TP, TN, and COD declined markedly. The bloom still appeared in summer: although chlorophyll decreased, algae increased in density (Table 20.11). Han (1993) studied the effect of dredging on the water quality of West Lake, Hangzhou, and found that dredging had little effect on improving water quality. This shows that the sediments had little effect on nutrient concentrations and their removal did not control the outbreak of algal blooms. The algal species remained almost the same; *Microcystis* was dominant, the bloom remained, and chlorophyll *a* rose to even higher levels in the absence of sediment (Wang and Pu 1999). Although appropriate dredging can improve water quality in the short term, it is not an effective way to control lake eutrophication in the longer term (Pu et al. 2000). Because of the large renewal rate of reservoir water, underwater dredging combined with bottom water discharge may be a more effective way to reduce internal pollution.

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