

Characteristics of Water Pollution in Typical Reservoirs

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Abstract Since the 1950s, the pollution of reservoirs has become serious. The water quality in reservoirs is deteriorating due to population growth, industrialization, and urbanization. In order to clarify the characteristics of water quality change in reservoirs, in-situ monitoring of water quality and phytoplankton were conducted in Jinpen Reservoir, Shibianyu Reservoir, Zhoucun Reservoir, and Zhelin Reservoir.

This chapter discusses the change of water quality in reservoirs in terms of eutrophication, algae pollution, and endogenous pollution. The results indicate that Zhelin Reservoir, Jinpen Reservoir, and Shibianyu Reservoir are in the middle eutrophic state, while Zhoucun Reservoir is in the eutrophic state. All the reservoirs suffer from algal blooms in July and August, with cyanobacteria dominating. Stratification, a very common phenomenon, occurs in the reservoirs mentioned above, can cause the decline of bottom dissolved oxygen, aquatic ecological environment deterioration, and the release of pollutants from sediment.

Keywords Eutrophication • Algal blooming • Stratification • Pollutants

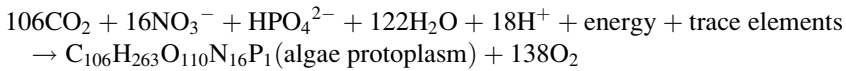
1 Eutrophication

Lakes and reservoirs are important water resources for human survival [1]. However, since the 1950s, more nitrogen and phosphorus have been released into the water body, which led to serious eutrophication in lakes and reservoirs. Given the adverse effects on the sustainable development of social economy and environment, eutrophication has become an ubiquitous environmental problem worldwide and attracts more attention and recognition from many states [2].

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1.1 The Formation Mechanism of Eutrophication

Excessive nutrients are the key factor of eutrophication, with phosphorus being the crucial nutrient, followed by nitrogen, carbon, trace elements, and vitamins. The eutrophication process can be expressed as the following equation:



The equation shows that phosphorus and nitrogen in the natural water bodies are the critical factors for the production of plankton biomass. Industrial wastewater, sewage, and farmland drainage contain large amounts of nitrogen and phosphorus, which could lead to increasing nutrients in water and cause vigorous growth of green plants and algae [3].

The relevant theories about the formation mechanism of eutrophication include the food chain theory and the life cycle theory.

(1) Food Chain Theory

This theory was proposed by Martin Shotton in 1997. It was believed that there are aquatic food chains in natural water. If zooplankton biomass decreases or their predation ability reduces, the algae growth amount will exceed consumption, which promotes eutrophication. Furthermore, the theory points out that increasing nutrition load is not the only reason for eutrophication. Some pollutants, such as persistent organic pollutants, can also affect the predatory ability of zooplankton and cause eutrophication [4].

(2) Life Cycle Theory

It is a widely accepted theory that plenty of compounds containing nitrogen and phosphorus are discharged into water bodies, and the original ecological balance is destroyed, which causes an excessive reproduction of algae. Algal blooms consume large amounts of dissolved oxygen, which makes plankton die due to the lack of oxygen. According to this theory, nitrogen and phosphorus are the fundamental reasons for eutrophication and algae are the main part of eutrophication.

According to the above two kinds of mechanisms for eutrophication formation, the reasons for eutrophication can be summarized as follows:

1. Water pollution is the primary cause of eutrophication. The pollution sources leading to eutrophication are generally around lakes and reservoirs, which can be exogenous sources or endogenous sources. Exogenous sources include point source pollution and non-point source pollution. Point source pollution usually refers to pollution discharged from drainage pipes, while non-point source pollution includes farmland drainage, river bank penetration, rain, and groundwater [5].
2. The hydraulic condition is one of the most important conditions that influence eutrophication. Eutrophication occurs in relatively closed water bodies, where water flow is slow and the water depth is shallow. Such conditions are suitable

for the growth of plants and algae. In deeper lakes or reservoirs, the hydraulic retention time is longer and a large number of pollutants can enrich at the bottom. Once stratification occurs, the bottom water will become anaerobic and nutrients will be released from the sediments.

3. Ecological imbalance promotes eutrophication. Microbial ecological imbalance also accelerates the eutrophication of water bodies. Microbes belong to disintegrators in the lake, and they can make use of organic pollutants derived from the producers and consumers in the food chain, thus maintaining water quality. However, if extensive organic matters enter into the water, dissolved oxygen will be consumed quickly and, thus, generate an anaerobic condition. Anaerobic conditions could not only destroy the food chain of lakes and reservoirs, but also intensify the phosphorus cycle.
4. Algal blooms form a vicious circle. Algal blooms make the optical radiation intensity attenuate rapidly along the depth, and causes the dissolved oxygen to decline. Algae would die from the lack of light and secrete algal toxin. The above phenomenon will lead to greater biological suffocation death and a loss of inhibiting ability to algae growth by ecological food chains. Meanwhile, it also contributes to the transformation of phosphorus in the water and further accelerates algal blooms [6].

1.2 The Assessment Method of Eutrophication

The eutrophic state depends primarily on factors such as total nitrogen (TN), total phosphorus (TP), biochemical oxygen demand in 5 days (BOD₅), chlorophyll-*a* (chl_a), and transparency (SD). The trophic state index (TSI), established by Carlson (1977) [7], is the most commonly used parameter, which includes TP, chl_a, and SD variables. The second most commonly used index is the lake evaluation index (LEI), proposed by Porcella. The two indexes both include chl_a, TP, and the absolute value of SD, so they are applicable to any lake or reservoir theoretically. According to the formulas (1–3), Table 1 gives the values of TP, chl_a, and SD corresponding to the TSI.

$$\begin{aligned} \text{TSI} &= 10(6 - \log_2 \text{SD}) & (1) \\ &= 10(6 - \log_2 7.7/\text{chl}_a 0.68) & (2) \\ &= 10(6 - \log_2 48/\text{TP}) & (3) \end{aligned}$$

In China, the TLI proposed by Can is widely used, which includes chl_a, COD, TN, TP, and SD. The evaluation result is reliable if the TLI is combined with the trophic principal component analysis and hierarchical analysis. The TLI can be calculated by formula:

Table 1 Tropic state index (TSI) and its relevant parameters

TSI	Transparency (m)	Surface phosphorus concentration (mg/m ³)	Surface chlorophyll concentration (mg/m ³)
0	64	0.75	0.04
10	32	4.5	0.12
20	16	3	0.12
30	8	6	0.94
40	4	12	2.6
50	2	24	7.4
60	1	48	20
70	0.5	96	56
80	0.25	192	154
90	0.12	384	427
100	0.062	768	1183

Table 2 The assessment standard of eutrophication in lakes

	SD (m)	TP (mg/L)	TN (mg/L)	chla (μg/L)	BOD (mg/L)	COD (mg/L)
Oligotrophic	2	0.01	0.12	5	1.5	2
Middle oligotrophic	1.5	0.025	0.3	10	2	3
Mesotropher	1	0.05	0.6	15	3	4
Middle eutropher	0.7	0.1	1.2	25	5	7
Eutropher	0.4	0.5	6	100	15	20
Heavy eutropher	<0.1	>0.5	>6	>100	>15	>20

Table 3 The weight of each parameter

Parameter	chla	TP	TN	SD	COD
W_j	0.2663	0.1879	0.1790	0.1834	0.1834

$$TLI(\Sigma) = \sum_{j=1}^m W_j \times TLI(j)$$

(4)

TLI(Σ) is the comprehensive trophic status index; TLI (j) is the j-th parameter index of trophic status, as shown in Table 2; and W_j is the relative weight of the j-th trophic status index, as shown in Table 3.

The trophic status of the signal index is calculated by the following:

Table 4 Trophic state grading of lakes based on the weighted comprehensive trophic state index

	Oligotrophic	Mesotropher	Eutropher	Light eutropher	Middle eutropher	Heavy eutropher
Range of TLI	<30	30–50	>50	50–60	60–70	>70

Table 5 Chinese grade standards for eutrophication in lakes and reservoirs

Assessment factors	Trophic grade						T_{oi}
	I	II	III	IV	V	VI	
	Light eutropher	Light-middle eutropher	Middle eutropher	Middle-hyper eutropher	Hyper eutropher	Heavy hyper eutropher	
chl _a (μg/L)	≤1.0	≤2.0	≤4.0	≤10	≤65	>80	27
TP (mg/L)	≤0.0025	≤0.005	≤0.025	≤0.050	≤0.200	>0.600	0.157
TN (mg/L)	≤0.030	≤0.050	≤0.300	≤0.500	≤2.000	>4.600	1.247
COD _{Mn} (mg/L)	≤0.54	≤0.72	≤3.60	≤7.20	≤18	>48.78	13.14

$$\begin{aligned} \text{TLI}(\text{Chla}) &= 10(2.5 + 1.086\ln\text{Chla}) \\ \text{TLI}(\text{TP}) &= 10(9.436 + 1.625\ln\text{TP}) \\ \text{TLI}(\text{TN}) &= 10(5.453 + 1.694\ln\text{TN}) \\ \text{TLI}(\text{SD}) &= 10(5.118 - 1.94\ln\text{SD}) \\ \text{TLI}(\text{COD}) &= 10(0.109 + 2.66\ln\text{COD}) \end{aligned}$$

(5)

The unit of chl_a is mg/m³, the unit of SD is m, and the units of the remainder are mg/L.

According to the value of the TLI, a series of consecutive values (0–100) are used to grade the trophic state of lakes or reservoirs (Table 4). Given that water pollution is gradual and continuous, a fuzzy mathematical method is considered to be reasonable to evaluate the trophic state of water. The concrete method is shown in Table 5

1.3 Trophic State Evaluation of Typical Reservoirs

1.3.1 Jinpen Reservoir

Chl_a, TP, TN, and COD_{Mn} are chosen as the surface water pollution control parameters of Jinpen Reservoir. The fuzzy comprehensive index method was used to estimate the trophic status in Jinpen Reservoir from 2008 to 2009. The monthly average results are shown in Table 6.

Table 6 Monthly average concentration of the pollution factors in Jinpen Reservoir

Month	chl _a	TP (mg/L)	TN (mg/L)	COD _{Mn} (mg/L)	Month	chl _a	TP (mg/L)	TN (mg/L)	COD _{Mn} (mg/L)
2008-03	0.89	0.015	1.08	2.27	2009-02	0.384	0.012	1.01	2.18
2008-04	1.71	0.025	1.14	2.75	2009-03	0.67	0.028	1.17	2.36
2008-05	3.39	0.05	1.43	3.16	2009-04	1.7	0.031	1.28	2.09
2008-06	3.643	0.035	1.05	3.04	2009-05	2.682	0.048	1.42	2.21
2008-07	3.294	0.032	1.36	3.29	2009-06	1.567	0.035	1.43	2.89
2008-08	15.76	0.05	1.44	3.48	2009-07	8.51	0.043	1.34	3.68
2008-09	5.37	0.061	1.381	3.43	2009-08	4.818	0.035	1.48	3.01
2008-10	2.48	0.035	1.26	2.88	2009-09	2.39	0.028	1.44	3.17
2008-11	1.828	0.024	1.25	2.90	2009-10	1.308	0.02	1.46	3.12
2008-12	0.97	0.014	1.09	2.56	2009-11	1.012	.0.011	1.27	3.04
2009-01	0.394	0.013	0.93	2.43	2009-12	0.848	0.008	1.18	2.68

Table 7 Fuzzy composite indexes for comprehensive evaluation of the nutrition in Jinpen Reservoir

Month	I	II	III	IV	V	VI	Results
2008-03	0.0282	0.1549	0.0750	0.4526	0.2894	0	Class IV (middle-hyper eutropher)
2008-04	0.0136	0.0801	0.2271	0.3871	0.2920	0	Class IV (middle-hyper eutropher)
2008-05	0	0.0410	0.1589	0.4119	0.3882	0	Class IV (middle-hyper eutropher)
2008-06	0	0.0477	0.3017	0.4329	0.2177	0	Class IV (middle-hyper eutropher)
2008-07	0	0.0421	0.2693	0.3156	0.3730	0	Class V (hyper eutropher)
2008-08	0	0.0046	0.1095	0.3385	0.5475	0	Class V (hyper eutropher)
2008-09	0	0.0083	0.2125	0.3376	0.4416	0	Class V (hyper eutropher)
2008-10	0	0.0807	0.2073	0.3783	0.3337	0	Class IV (middle-hyper eutropher)
2008-11	0.0080	0.0756	0.1215	0.4478	0.3472	0	Class IV (middle-hyper eutropher)
2008-12	0.0301	0.0998	0.1380	0.4466	0.2855	0	Class IV (middle-hyper eutropher)
2009-01	0.0142	0.1221	0.1383	0.5150	0.2104	0	Class IV (middle-hyper eutropher)
2009-02	0.0133	0.1228	0.1044	0.5012	0.2582	0	Class IV (middle-hyper eutropher)
2009-03	0.0188	0.0585	0.1963	0.3906	0.3358	0	Class IV (middle-hyper eutropher)
2009-04	0.0131	0.0877	0.1566	0.3735	0.3691	0	Class IV (middle-hyper eutropher)
2009-05	0	0.0855	0.0851	0.4237	0.4057	0	Class IV (middle-hyper eutropher)
2009-06	0.0151	0.0534	0.1813	0.3186	0.4315	0	Class V (hyper eutropher)
2009-07	0	0	0.2212	0.3096	0.4692	0	Class V (hyper eutropher)
2009-08	0	0.0239	0.2601	0.3828	0.3331	0	Class IV (middle-hyper eutropher)
2009-09	0	0.0649	0.2278	0.2559	0.4514	0	Class V (hyper eutropher)
2009-10	0.0211	0.0551	0.1847	0.2661	0.4730	0	Class V (hyper eutropher)
2009-11	0.0273	0.0688	0.1535	0.3677	0.3827	0	Class V (hyper eutropher)
2009-12	0.0255	0.0881	0.1187	0.4222	0.3455	0	Class IV (middle-hyper eutropher)

Using the weighted comprehensive trophic state index to evaluate the trophic status of Jinpen Reservoir, the evaluation results of each month are shown in Table 7.

The following conclusions can be drawn from Table 7: (1) The eutrophication degree of Jinpen Reservoir lies in the middle-hyper eutropher to hyper eutropher, which is consistent with the actual situation in water quality pollution; (2) The pollution and trophic status of Jinpen Reservoir are becoming serious.

Table 8 Weighted comprehensive trophic state index of Shibianyu Reservoir (2011)

Month	chl _a	TP	TN	COD _{Mn}	SD	Trophic state index	Trophic status
2011-03	1.3	0.033	2.2	3.5	2.2	39.8	Middle eutropher
2011-04	7.7	0.036	4.3	3.2	1.9	47.3	Middle eutropher
2011-05	10.6	0.038	3.7	2.9	1.7	47.9	Middle eutropher
2011-06	31.4	0.031	3.2	3.1	1.2	51.5	Light eutropher
2011-07	20.2	0.043	2.8	3.3	1.5	50.3	Light eutropher
2011-08	35.6	0.052	3.3	5.2	1	56.7	Light eutropher
2011-09	1.9	0.092	4.5	6.8	0.6	55.3	Light eutropher
2011-10	1.3	0.043	2.8	5.5	1.5	44.9	Middle eutropher
2011-11	1.1	0.036	2.1	4.5	2.0	41.0	Middle eutropher
2011-12	1.0	0.032	2.1	4.3	2.1	40.0	Middle eutropher

Table 9 Weighted comprehensive trophic state index of Shibianyu Reservoir (2012)

Month	chl _a	TP	TN	COD _{Mn}	SD	Trophic state index	Trophic status
2012-01	1.1	0.028	2.1	3.8	2.1	39.2	Middle eutropher
2012-02	1.0	0.023	2.0	3.7	2.1	38.1	Middle eutropher
2012-03	2.2	0.021	1.9	3.5	2.0	39.8	Middle eutropher
2012-04	5.9	0.025	4.3	3.0	1.8	45.3	Middle eutropher
2012-05	24.6	0.019	3.7	2.9	1.5	48.6	Middle eutropher
2012-06	44.7	0.036	3.1	4.1	1.2	54.2	Light eutropher
2012-07	27.8	0.033	2.5	3.5	1	51.8	Light eutropher
2012-08	41.6	0.032	2.3	3.6	0.6	54.6	Light eutropher
2012-09	52.2	0.075	3.6	6.2	0.4	62.7	Middle eutropher
2012-10	32.8	0.042	2.7	5.4	0.8	56.2	Light eutropher
2012-11	5.2	0.024	2.2	4.2	1.2	45.9	Middle eutropher
2012-12	1.2	0.021	2.1	3.3	1	40.5	Middle eutropher

1.3.2 Shibianyu Reservoir

Chl_a, TP, TN, and COD_{Mn} are chosen as the surface water pollution control parameters of Shibianyu Reservoir. The fuzzy comprehensive index method was used to estimate the trophic status in Shibianyu Reservoir from 2011 to 2013. The monthly average results are shown in Table 8.

It can be seen from Tables 8, 9, and 10 that the nutrient content of Shibianyu Reservoir is high. From October to April, the water temperature of the reservoir is low and the algal growth speed is restrained; thus, the transparency is high and the reservoir is in the middle eutropher [8]. With the temperature rising in May, algae in reservoir began to thrive. Diatom dominates in spring, and the peak of diatom biomass generally appears at the end of May. In summer, as the temperature rises further, harmful cyanobacteria (mainly *Microcystis aeruginosa*) begin to prevail. Depending on their strong adaptability in a wide range of temperature and illumination, cyanobacteria dominate for a long time until October. From May to September, algae grow quickly and the water transparency is reduced. Shibianyu Reservoir is, thus, in the hyper eutropher.

Table 10 Weighted comprehensive trophic state index of Shibianyu Reservoir (2013)

Month	chl _a	TP	TN	COD _{Mn}	SD	Trophic state index	Trophic status
2013-01	1.1	0.019	2.1	3.2	2.1	37.2	Middle eutropher
2013-02	1.1	0.021	2.1	3.1	2.2	37.2	Middle eutropher
2013-03	3.2	0.021	2.3	3.1	2.1	40.7	Middle eutropher
2013-04	6.1	0.032	3.7	3.2	2.0	45.6	Middle eutropher
2013-05	15.6	0.042	4.1	5.2	1.5	52.9	Light eutropher
2013-06	28.7	0.038	3.7	4.8	1.2	54.4	Light eutropher
2013-07	43.1	0.032	2.8	5.3	0.7	56.6	Light eutropher
2013-08	40.2	0.026	2.4	4.2	0.8	53.7	Light eutropher
2013-09	39.7	0.025	2.3	4.3	0.7	54.0	Light eutropher
2013-10	19.9	0.022	2.3	3.8	1.4	48.6	Middle eutropher
2013-11	5.2	0.042	2.7	3.5	1.8	45.9	Middle eutropher
2013-12	1.2	0.051	2.8	3.2	2.1	41.3	Middle eutropher

Table 11 Trophic status assessment of Zhoucun Reservoir

Month	chl _a	TN	TP	SD	COD	TLI(Σ)
12-04	38.03	71.36	40.79	47.64	39.26	82.68
12-05	25.11	66.52	33.06	45.36	46.92	81.67
12-06	58.81	63.27	33.06	52.18	46.00	83.38
12-07	70.45	61.40	49.94	65.42	54.72	85.12
12-08	71.97	69.36	49.94	64.24	54.33	85.58
12-09	65.76	70.52	44.32	59.24	48.75	84.76
12-10	64.63	68.27	45.02	55.75	45.55	84.55
12-11	55.47	69.78	42.45	52.80	34.79	82.87
12-12	54.13	70.45	33.06	47.64	42.82	83.44
13-02	48.59	70.85	33.75	32.64	32.61	83.23
13-03	43.13	72.09	35.67	49.33	37.02	82.25
13-04	39.24	70.45	36.26	53.22	38.56	81.89
13-05	41.38	67.81	38.43	53.22	39.58	82.23
13-06	52.19	59.23	46.00	57.82	56.40	84.48
13-07	65.31	65.67	62.18	62.43	54.76	85.94
13-08	62.60	70.39	44.32	66.67	48.84	84.20
13-09	59.59	71.10	44.32	60.45	38.75	83.31
13-10	58.73	68.64	48.64	55.03	41.45	84.11
13-11	56.86	65.84	44.32	47.64	39.39	83.87
13-12	55.18	65.22	42.45	43.97	32.85	83.02

1.3.3 Zhoucun Reservoir

The comprehensive nutrition state index was used to evaluate Zhoucun Reservoir. The results are shown in Table 11.

The results show that the comprehensive indices of trophic status in Zhoucun Reservoir from April 2012 to December 2013 are higher than 70, indicating that Zhoucun Reservoir belongs to the hyper eutropher and the water pollution is serious.

Table 12 The evaluation results of trophic status in Zhelin Reservoir (2008–2012)

Year	2008	2009	2010	2011	2012
TLI(Σ)	43.14	51.28	48.27	38.45	43.73
Trophic status	Middle eutropher	Light eutropher	Middle eutropher	Middle eutropher	Middle eutropher

1.3.4 Zhelin Reservoir

According to the water quality results from 2008 to 2012, the evaluation results of trophic status are shown in Table 12: (1) Zhelin Reservoir belongs to the middle eutropher; (2) The trophic status index of Zhelin Reservoir increased from 2008 to 2009, but decreased from 2009 to 2011, and increased again from 2011 to 2012. This illustrates that the water environmental protection methods performed by the local government are not effective enough to control the eutrophication of Zhelin Reservoir.

1.4 Source Analysis of Nitrogen and Phosphorus in Typical Reservoirs

1.4.1 Shibianyu Reservoir

(1) Seasonal Variations of TN and TP

The TN content of Shibianyu Reservoir is high and the annual average concentration is 2.78 mg/L (Fig. 1a). The contents of TN are relatively low from December to February. The highest concentrations of TN are 4.3, 4.3, and 3.7 mg/L in the spring of 2011, 2012, and 2013, respectively. Storm runoff makes the concentration of TN rise sharply in summer. The highest TN concentrations during the storm runoff are 5.4 mg/L (September 19), 3.6 mg/L (September 3), and 4.1 mg/L (May 30) in 2011, 2012, and 2013, respectively. After the storm runoff, the concentration of TN reduces quickly. The TN concentration increases within a narrow range (from 2.2 to 2.9 mg/L) during the mixing period at the end of November 2013, but the phenomenon does not appear at the same time in 2011 and 2012.

The TP content can meet the national surface water quality standard (GB 3838–2002) (class III) throughout the year, and its annual average concentration is 0.038 mg/L (Fig. 1b). The TP concentration increases mainly in the period of summer storm runoff. The highest concentrations of TP are 0.19 mg/L (19 September), 0.13 mg/L (3 September), and 0.05 mg/L (30 May) in 2011, 2012, and 2013, respectively. The content of TP reduces quickly after the storm runoff. Similar to the variations in TN, TP rises at the end of November 2013 but did not increase in 2011 and 2012. To explore the reasons for the seasonal variations of phosphorus and nitrogen, ten monitoring points are selected upstream of the reservoir, as shown in Fig. 2.

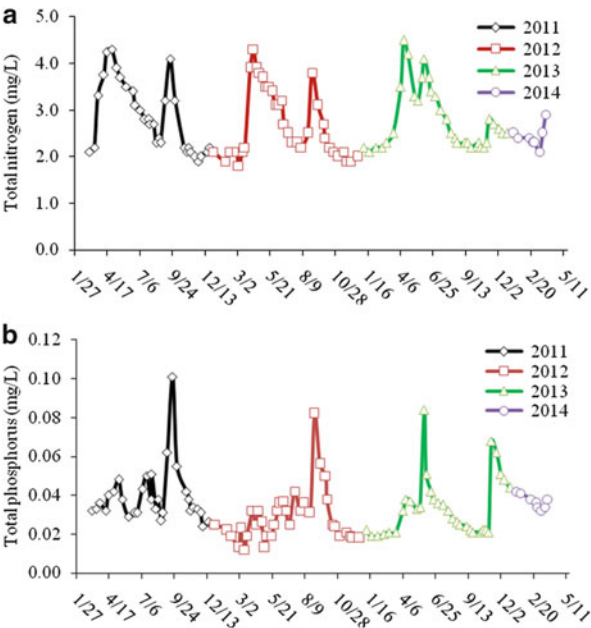


Fig. 1 Seasonal variations of TN and TP in Shibianyu Reservoir (2011–2014)



Fig. 2 Distribution of the monitoring points in Shibianyu Reservoir

(2) Cause Analysis of TN Increase in Spring

(1) Endogenous pollution survey

The thermal stratification of Shibianyu Reservoir disappeared at the end of November and then the reservoir entered the complete mixing period (Fig. 3a). During this period, the whole water is mixed and the temperature is low. The concentration of dissolved oxygen increases with the decreasing temperature and reaches a peak of 12.8 mg/L in winter (Fig. 3b). At this time, the reservoir has good self-purification ability and endogenous release is inhibited by dissolved oxygen. So, we can eliminate the impact of endogenous pollution on the nutrients concentration of Shibianyu Reservoir.

(2) Exogenous pollution survey

The flow into Shibianyu Reservoir is around 0.1 m³/s in winter (Fig. 4). As the temperature increases during mid-March, the inflow of Shibianyu Reservoir increased because of upstream snowmelt, commonly known as spring flood. In this period, the rainfall is small enough. Since snowfall has becomes less in recent years, the flow is generally less than 10 m³/s. After the rain, the inflow of Shibianyu Reservoir has a short period of increase and then remains at a stable flow (about 2 m³/s). As shown in Table 13, with the increase of inflow, the TN concentration rises rapidly at the monitoring points (from S1 to S8) during mid-March. Especially at monitoring point S7, the TN concentration reached 6.58 mg/L. Until the end of May, TN started to decrease rapidly at the upstream monitoring points.

As shown in Table 14, the TN load upstream was estimated from March to May in 2011 and 2012. Despite the upstream water flow is relatively low, the TN concentration of the upstream water stays high over a long period. Due to the small water storage capacity of the reservoir, the upstream water has a great influence on the TN concentration of the reservoir. The upstream water causes TN to be increased by 0.6, 2.2, and 2.7 mg/L from March to May in 2011 and 1.6, 1.2, and 1.8 mg/L from March to May in 2012.

(3) Cause analysis of TN and TP increase

Influenced by the subtropical monsoon climate, the seasonal distribution of rainfall in Shibianyu Reservoir is uneven. As shown in Fig. 5, rainfall mainly concentrated in May to September, accounting for 80 % of the annual rainfall. The largest rainfall occurred in September (412 mm) and August (180 mm) from 2011 to 2013, respectively. The largest annual rainfall was 101.4, 81, and 92 mm from 2011 to 2013, respectively. In general, the inflow of the reservoir and rainfall were significantly positive correlated, especially during the rainy season. After the heavy rain in 2011–2013, the flood peak reached 100.4, 117.6, and 102.4 m³/s, respectively. During the storm, the reservoir would experience a sharp change in the water level.

Large amounts of nitrogen and phosphorus entered into the reservoir and runoff is the main cause of the elevated concentration of TN and TP. Upstream flow and the content of nitrogen and phosphorus change greatly during storms; therefore, it is difficult to determine the exogenous pollutants accurately during the storm. However, when the reservoir flow is more than 100 m³/s, the contents

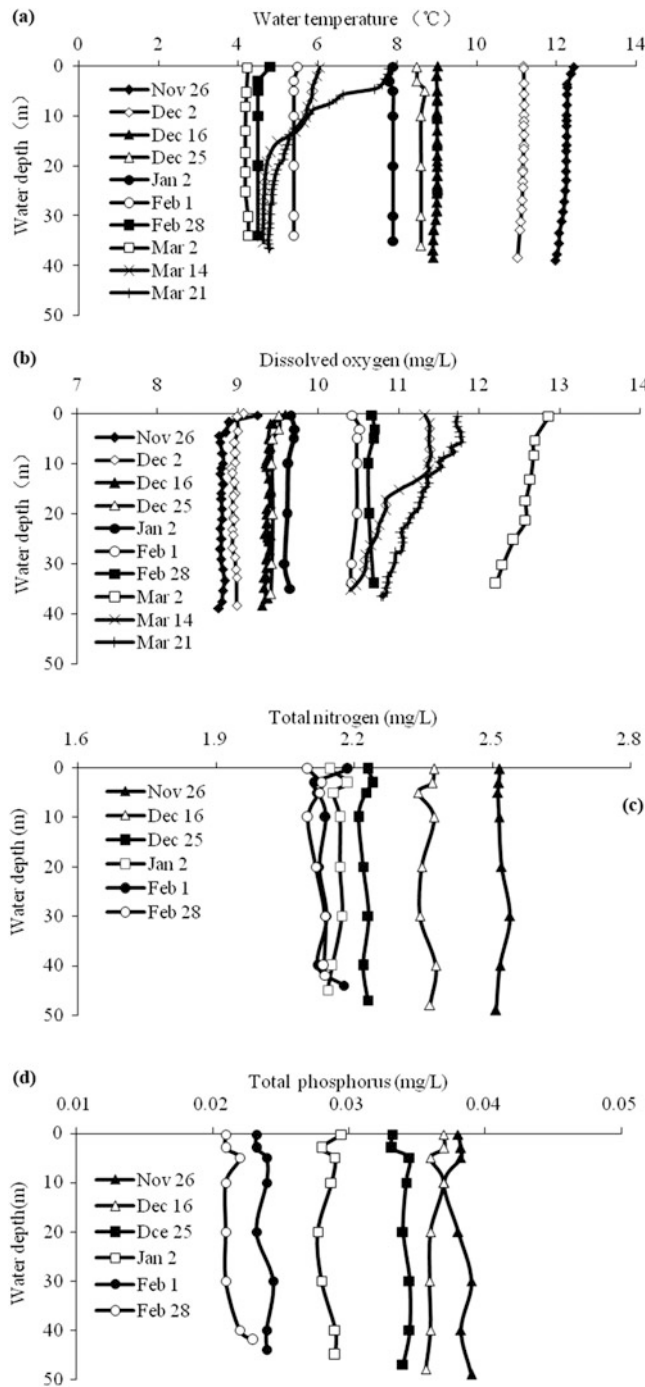


Fig. 3 Vertical distributions of temperature (a), dissolved oxygen (b), total nitrogen (c), and total phosphorus (d)

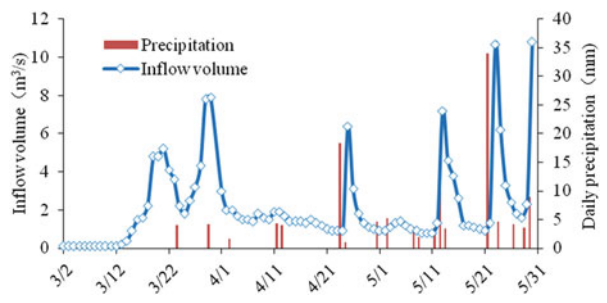


Fig. 4 Rainfall and inflow change curve in spring

Table 13 TN concentrations of upstream monitoring points (mg/L)

Monitoring point	February 28	March 5	March 20	April 3	April 25	May 10	May 19	June 3
S1	2.25	2.04	4.38	4.64	4.56	4.24	4.01	3.36
S2	2.26	2.06	4.86	4.89	4.62	4.31	4.34	3.19
S3	2.35	2.14	4.76	4.96	4.50	4.35	4.15	3.27
S4	2.49	2.15	4.62	4.82	4.18	4.41	3.92	3.09
S5	2.14	1.91	3.72	4.16	4.20	4.42	4.08	3.09
S6	2.31	1.91	3.71	4.13	4.00	5.02	3.93	3.20
S7	2.96	2.72	6.05	5.98	6.58	5.85	5.79	4.68
S8	2.10	1.83	3.81	4.39	4.19	4.25	4.01	3.36

Table 14 TN load of upstream in spring

	Rainfall (mm)	Reservoir inflow (10 ⁴ m ³)	Average concentration (mg/L)	Month input (t)	Reservoir volume (10 ⁴ m ³)	Increased concentration (mg/L)
2011-03	21.6	134	3.3	4.4	737	0.6
2011-04	35.3	445	4.1	18.2	818	2.2
2011-05	129.3	615	3.6	22.1	833	2.7
2012-03	16.3	604	3.7	22.3	1393	1.6
2012-04	34.0	409	4.5	18.4	1512	1.2
2012-05	86.5	691	3.4	23.5	1276	1.8

of TN and TP will rise with the increase of sediments in the runoff. The inflow nutrients load of storm runoff in 2011–2013 is shown in Table 15. The nutrients are mainly composed of particulate organic nitrogen and particulate organic phosphorus, which are easy to be settled. So, after the storm runoff, the TN and TP concentrations of the reservoir decline quickly and the endogenous pollution load increases.

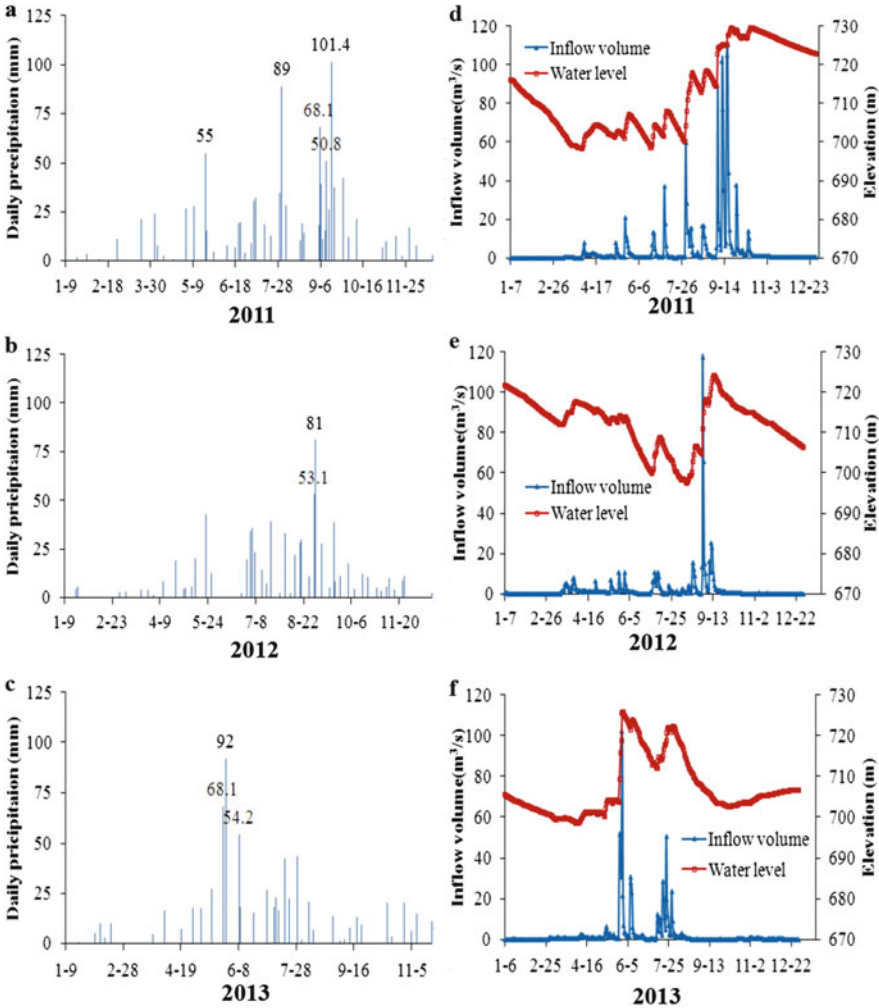


Fig. 5 Daily variations of rainfall, inflow, and water level (2011–2013)

1.4.2 Source Analysis of Nitrogen and Phosphorus in Zhoucun Reservoir

The TN concentration in Zhoucun Reservoir varies from 1.5 to 2.8 mg/L, with nitrate the major component (as shown in Figs. 6 and 7). The seasonal variation of the TN concentration is similar to that of nitrate. Their concentrations decline from spring to summer. In the late dry season, the concentrations of TN and nitrate would fall to the lowest levels. With the increase of rainfall in the rainy season, a number of exogenous nitrates enter into Zhoucun Reservoir, causing the concentrations of TN and nitrate to increase rapidly. In winter and spring, the concentration of TN is

Table 15 Inflow nutrients load of storm runoff

Year	Peak discharge (m ³ /s)	Occurrence time	Inflow (10 ⁴ m ³)	TN (mg/L)		TP (mg/L)		Total input (t)	PTN/ TN	PTP/ TP	Total input (t)
				Inflow concentration	PTN/ TN	Inflow concentration	PTP/ TP				
2011	110.4	18 September	4735	7.2/4.5	79 %	0.252/0.064	78 %	213.07			3.03
2012	117.8	1 September	1869	5.7/3.6	68 %	0.148/0.047	65 %	67.28			0.88
2013	102.0	29 May	1977	5.2/3.3	65 %	0.079/0.042	54 %	65.24			0.83

Fig. 6 Seasonal variations of nitrogen in Zhoucun Reservoir

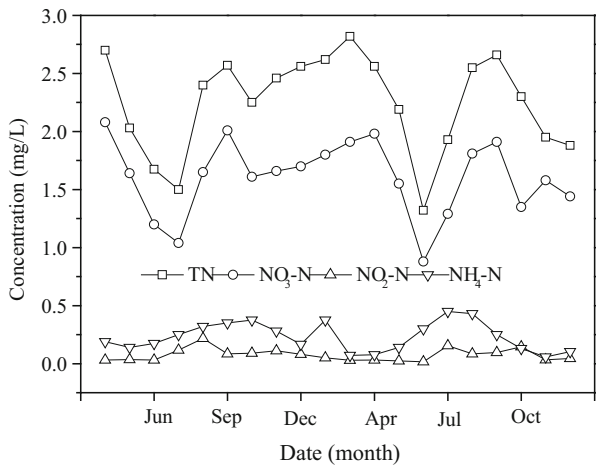
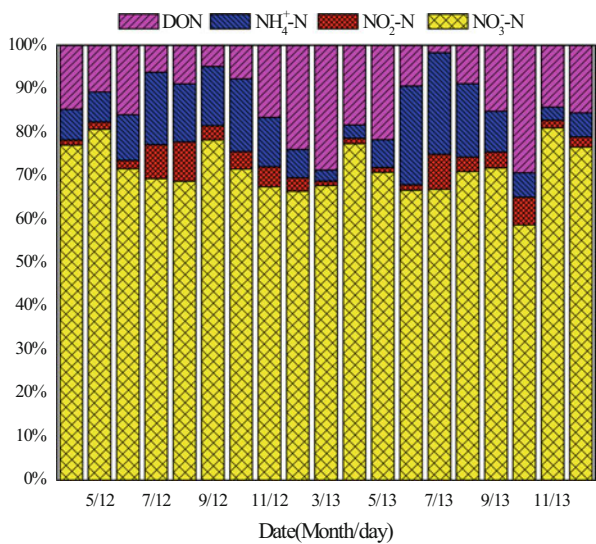


Fig. 7 Proportion of different species of nitrogen in Zhoucun Reservoir

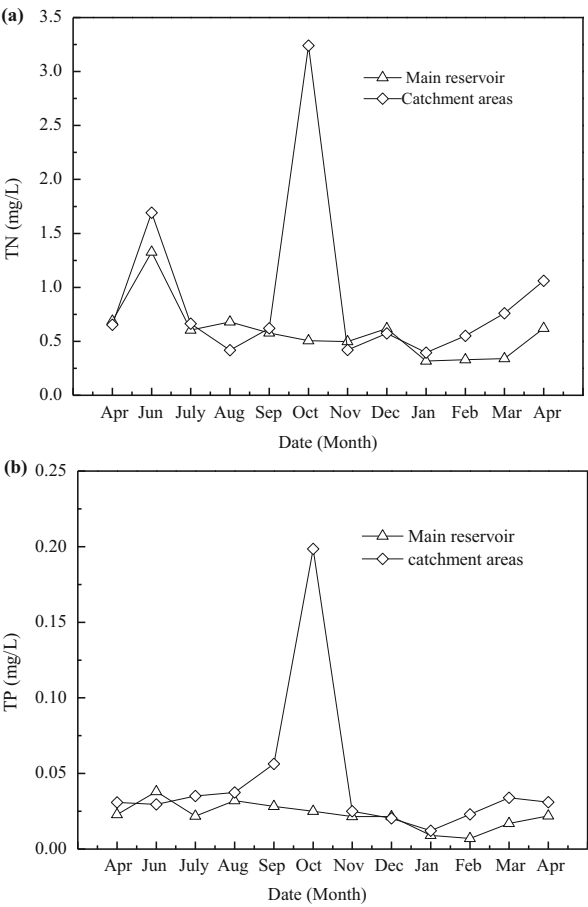


stable, remaining around 2.5 mg/L. The concentration of nitrate is between 1.6 and 1.8 mg/L.

The concentrations of ammonia are in the range 0.2–0.5 mg/L, with the peak appearing in September or October each year. This is mainly due to the release of pollutants from sediment during May to October. Zhoucun Reservoir starts to form stratification in April, and the bottom water becomes anaerobic in May, which creates favorable conditions for the release of ammonia.

In September and October, stratification becomes weak and the thermocline moves down, which cause the original bottom water and ammonia to be released

Fig. 8 Seasonal variations of TN and TP in Zhelin Reservoir



into the mixing layer. When the stratification is completely destroyed, ammonia will distribute homogeneously in the whole reservoir, and then ammonia will decline gradually due to the nitrification process.

1.4.3 Source Analysis of Nitrogen and Phosphorus in Zhelin Reservoir

Figure 8 shows the seasonal variations of TN and TP in Zhelin Reservoir and upstream. The concentrations of TN upstream and in the reservoir reach up to 1.33 and 1.69 mg/L, respectively, which are beyond the national surface water quality standard (GB 3838–2002) (class III). The average rainfall of Zhelin Reservoir is 1506 mm and 40–50 % of that occurs from April to June. Therefore, large amounts of pollutants are carried into the lake in June. The concentrations of TN in the major estuary, Wuning County, and fork entrance reach 1.8, 2.3, and 1.9 mg/L,

Table 16 Pollution load of Zhelin Reservoir in October 2013

Monitoring points	Region of influence (km ²)	Influence depth (m)	TN concentration (mg/L)	TP Concentration (mg/L)	TN load (t)	TP load (t)
Wuning County	7.26	5	5.72	0.336	207.6	13.29
Major estuary	3.36	5	3.93	0.26	44.39	4.37
Fork entrance	8.55	2.5	2.22	1.07	47.45	22.87
Shallow area	13.44	2.5	1.75	0.028	58.8	0.94
Summary	40.67				358.24	41.44

respectively. The concentration of TN in the reservoir is 1.33 mg/L. Therefore, rainfall runoff is the main cause of increased concentrations of TN and TP in Zhelin Reservoir.

The concentrations of TN and TP increased suddenly in October 2013, reaching 3.24 mg/L and 0.20 mg/L, respectively. Table 16 shows the estimation of pollution load according to the results of water quality in Zhelin Reservoir basin.

2 Algal Blooming

2.1 Harm of Algal Blooms

2.1.1 Effects on Drinking Water

(1) Algae-Induced Taste and Odor

Algae, fungi, and actinomycetes are the main sources of odor [9]. Different algae cause different types of odor. Algal growth secretes a lot of odor-producing compounds, which will be released after algae die. There are more than ten kinds of odor-producing compounds in eutrophic water. Geosmin and 2-methyl isopropyl alcohol (2-MIB) are the two major algae-induced odor compounds [10].

(2) Microcystin

Studies have shown that about 25–70 % of cyanobacteria can produce microcystin. The main species are *Microcystis aeruginosa*, *Anabaena flos-aquae*, and *Aphanizomenon flos-aquae* [11–13]. Microcystic toxins (MC), the most widely distributed toxins, are a group of seven peptide monocyclic liver toxins and have the closest relationship with human beings. They are also strong liver tumor-promoting agents. MC often exist in the algal cells and will be released into the water when the cells break up. MC have been found in drinking water all over the world. Events that animals die of microcystin have been reported in more than ten countries [14].

(3) Disinfection By-products

The issue of disinfection by-products has become a hot topic in current drinking water area [15]. Algal organic matter is the precursor of disinfection by-products. It could not only produce three trihalomethanes (THMs), but also the more harmful haloacetic acids (HAAs). All these disinfection by-products decrease the safety [16].

2.1.2 Effect on Water Treatment Systems

(1) Clogging Filters

The density of algae is close to that of water, which lets them remain suspended in water. Green algae, cyanobacteria, and diatom are the most common species in China.

Algae in the size range 2–200 μm have great effects on water plants. Since the density of algae is small, it is difficult to be removed by coagulation and precipitation [17]. A lot of residual algae will aggregate in the filter (especially slow filters) and form dense layers, which cause a reduction in the permeability of the filter. The gas released from the algae can also clog the filter, leading to the operation cycle being shortened and filter backwashed water increasing.

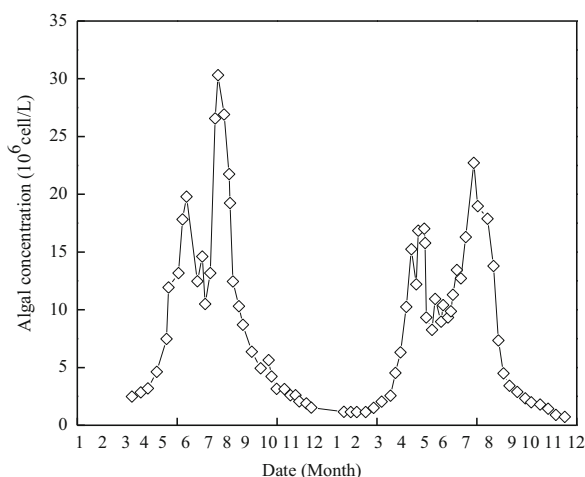
(2) The Increase of Chemical Cost

Much greater quantities of coagulants and disinfectants would be required due to the large amount of algae and algae-produced organic matter. Algal organic compounds could react with the hydrolysis products of coagulant (iron or aluminum salts) [18]. While the generating surface complex can prevent small particles from colliding with each other, it is, thus, inevitable to increase the dosage of coagulant to mitigate the impacts of surface complex [19].

2.1.3 Effect on Pipe Network and Water Quality

In order to control the number of bacteria generated in the water distribution network, the water from waterworks always maintain certain residual chlorine, even though bacteria still regrow in the water distribution network. The phenomenon is closely related to the nutrients in the effluent. The reproduction of bacteria causes water quality deterioration (e.g., as turbidity and chromaticity increase, the total number of bacteria increases). Algae and organic matter existing in the pipe network could promote the growth of bacteria and larger organisms, such as nematodes and sponge animals in the pipe network. It is very difficult to eliminate these animals, which can jam water meters and faucets. Algal metabolites can also react with the hydrolysis species during the coagulation process, which leads to greater coagulants consumption accordingly and increasing the water production cost. Furthermore, the generating complex can also cause pipeline corrosion and shorten the service life of the pipe network.

Fig. 9 Seasonal variation of algal abundance in the surface water of Jinpen Reservoir



In conclusion, algal blooming in lakes and reservoirs has a great influence on water quality, water plant operation, and the pipeline network. Thus, removing algae has become an important subject in controlling the water quality of reservoirs.

2.2 Algae Pollution of Jinpen Reservoir

2.2.1 Seasonal Variations of Algal Abundance

As an important producer in aquatic ecosystems, algal abundance and community can directly affect and indicate trophic status in reservoirs. Figure 9 shows that algal abundance distributed in a typical “saddle type” in Jinpen Reservoir during 2008–2009. In March 2008, the algal density in the surface water was less than 4×10^6 cells/L, while the value rapidly increased up to 22×10^6 cells/L in late May. The average daily growth rate is as high as 27.82×10^4 cells/(L·d). In July, algal abundance continued to rise and reached a peak at 30×10^6 cells/L. Algal blooming appeared in July and lasted for a month. Compared with that in 2008, less algal abundance was achieved in 2009 because of the different hydrological and meteorological factors. However, the duration of algal bloom (algal abundance $> 10 \times 10^6$ cells/L) extended in 2009.

Regression analysis (Fig. 10) shows that there are positive correlations between algal abundance and TN or TP ($R^2 = 0.6546, 0.4353, p < 0.01$). In contrast, there is a negative correlation between algal abundance and water level ($R^2 = 0.209, p < 0.01$). Besides, the correlation between $\text{NH}_4\text{-N}$ and algal abundance is poor.

Temperature is one of the most important influencing factors for algal growth, which can not only control the enzymatic reaction of photosynthesis and respiration intensity, but also influence other environmental factors and nutrients. TN/TP in Jinpen Reservoir fluctuates from 30 to 210, which indicates that phosphorus is the primary restrictive factor.

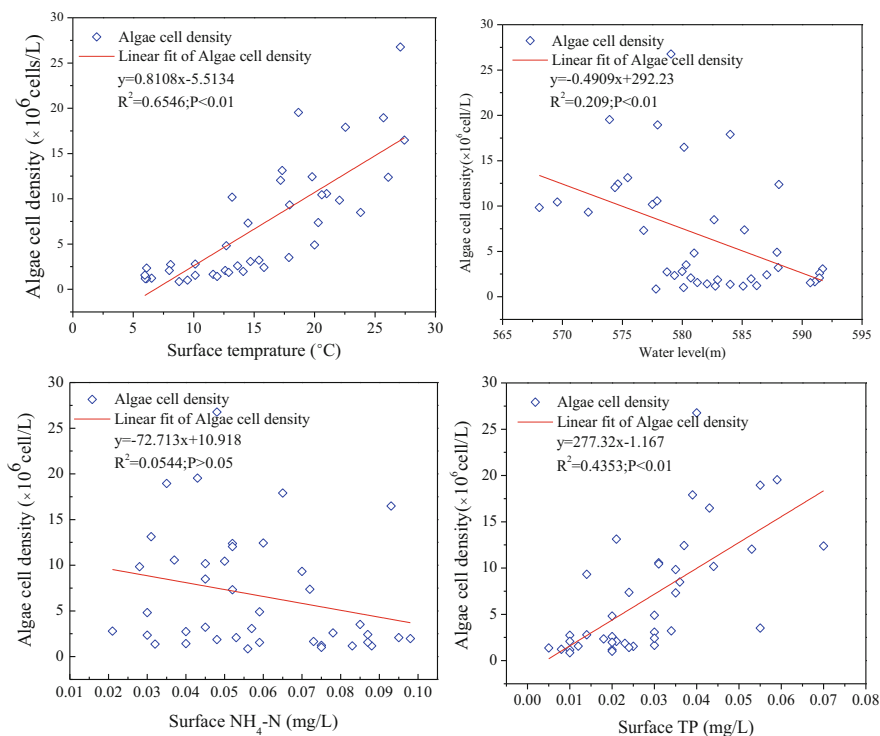


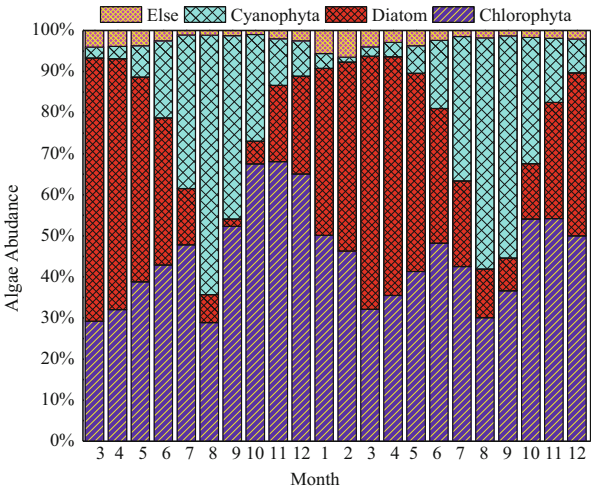
Fig. 10 Correlations between algal abundance and its influencing factors in the surface water of Jinpen Reservoir

The correlation between the water level and algal abundance is affected by many factors. On the one hand, fluctuation in the water level can change the nutrients concentration in the water (dilution or concentration). On the other hand, the turbulent effect involved in the fluctuation process can promote algal growth. With respect to the poor correlation between NH_4 and algal abundance, the authors proposed two reasons for this: (1) nitrogen released from sediment is far greater than the demand, although algal proliferation needs nitrogen to compound protein (algae tends to absorb organic nitrogen firstly, $\text{NH}_4\text{-N}$ secondly, and finally $\text{NO}_3\text{-N}$); (2) NH_4 concentration of the surface water varies widely, since it is affected by many internal and external environmental factors, such as water temperature, pH value, air pressure, and wind speed.

2.2.2 Seasonal Variations of the Algal Community

Sixty-nine taxa belonging to five phyla (Bacillariophyta, Chlorophyta, Cyanophyta, Euglenophyta, and Xanthophyta) were identified. The most diverse groups include Chlorophyta (33 taxa), Bacillariophyta (20 taxa), and Cyanophyta (12 taxa). Euglenophyta and Xanthophyta only account for 5.80 %.

Fig. 11 Seasonal evolution of algal community structure in Jinpen Reservoir



Algal community succession also has obvious seasonal characteristics (Fig. 11). In spring, the dominant groups were Bacillariophyta (48–64 %), Chlorophyta (29–41 %), and Bacillariophyta (2–7.7 %). The major dominant species included *Cyclotella*, *Fragilaria*, and *Melosira*. In summer, the proportion of Cyanophyta (16–63 %) increased rapidly and Bacillariophyta decreased significantly. The prevalent algae evolved into the eutrophic status species: *Microcystis*, *Merismopedia*, *Chlorella*, and *Chroococcus*. During autumn, the proportion of Chlorophyta increased (37–68 %), while Cyanophyta decreased. The most popular species were *Chlorella*, *Scenedesmus*, *Chlorococcus*, and *Ankistrodesmus*. In winter, the proportion of Cyanophyta and Chlorophyta declined and Bacillariophyta returned to being prevalent.

Although the water temperature in Jinpen Reservoir was relatively low in the beginning of spring, algae began to grow rapidly due to the adequate nutrients supply and strong light intensity. During this period, some diatoms and green algae that are adaptable to low temperature became the dominant species. As the water temperature increased, algae began to grow and reached a peak in late spring for the first time in the year. After the beginning of summer, the water body became more stable due to the formation of thermal stratification. It was difficult for diatoms and other algae to remain in suspension and much of them would be lost by sedimentation. Considering the insufficient available nutrients in water, algal abundance declined at the beginning of summer. In July, the temperature increased rapidly and cyanobacteria began to grow and aggregate. Although the cyanobacteria recovered later than diatoms and green algae, the growth rate of cyanobacteria after its recovery was higher than that of diatoms and green algae. Besides, the thermal stability of genetic material and the photosynthesis system make cyanobacteria tolerant to high temperature. So, cyanobacteria took absolute superiority in algae competition in summer. In late August, algal abundance began to reduce after the

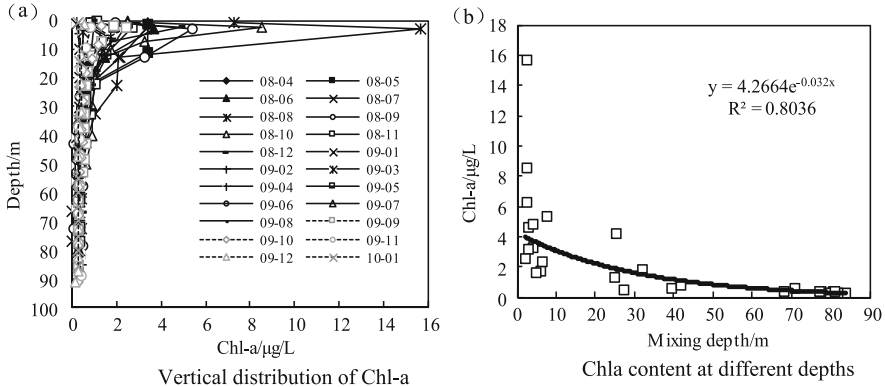


Fig. 12 Vertical distribution of chlorophyll-*a* concentration and correlation between its steady content and the water mixing depth in Jinpen Reservoir

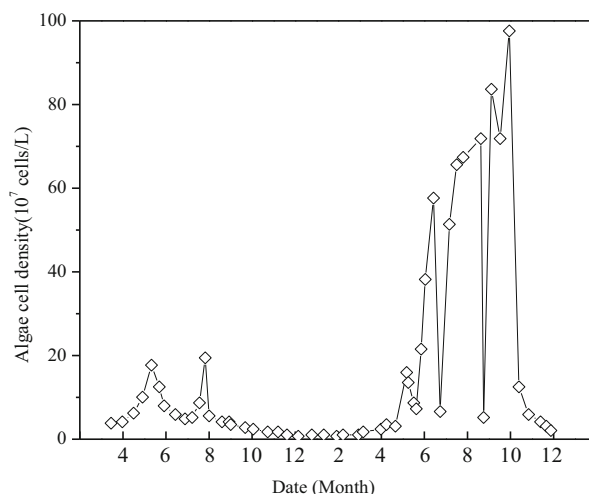
“summer peak”. The attenuation was mainly attributed to the increase of the respiration and photosynthesis rate. High temperature could raise the respiratory rate of algae, while the lack of available nutrients could reduce the photosynthesis rate. So, algal abundance decreased in the late summer. In autumn, due to the attenuation of light intensity and decreasing water temperature, Jinpen Reservoir began to mix and the proportion of diatoms increased. In winter, low temperature and low light intensity led algal abundance to fall sharply. Thus, the algae community succession characteristics in Jinpen Reservoir were as follows: “Diatom is dominant in spring and winter, while cyanobacteria and green algae are dominant in summer and autumn”.

2.2.3 Vertical Distribution of Algae

Water temperature, illumination, and nutrient level are the key factors for algal growth. These factors are significantly different at different depths and have a great influence on the vertical distribution of algae.

To some extent, the vertical distribution of chlorophyll-*a* can represent the vertical distribution of algae. Figure 12a shows the vertical distribution of chlorophyll-*a* in 2008 and 2009 in Jinpen Reservoir. The maximum chlorophyll-*a* concentration in the upper layer (euphotic zone) appeared in July or August (9–16 µg/L), and the minimum occurred in January or February (0.2–0.34 µg/L). Seasonal variations of chlorophyll-*a* are similar to algal abundance, but were not synchronous. For example, when the algal density reached $(17\text{--}20) \times 10^6$ cells/L in the spring, the content of chlorophyll-*a* was still less than 4 µg/L. This was due to the low content of chlorophyll-*a* in diatoms. Furthermore, the phenomenon that the maximum chlorophyll-*a* did not appear in the surface water was mainly due to the light inhibition. Photosynthesis decreased rapidly with the increase of depth and the algal growth rate was seriously limited. In the thermocline, the chlorophyll-*a* content reduced to 0.2–0.3 µg/L quickly and reached a minimum in the hypolimnion layer (<0.2 µg/L).

Fig. 13 Seasonal variations of algal abundance in Shibianyu Reservoir (2011.4–2012.12)



The mixing depth could directly affect the available light intensity and determine the distribution of algal biomass. The regression analysis shows that chlorophyll-*a* increases with the decline of mixing depth (Fig. 12b). There is a strong negative correlation between the two indexes ($y = 4.2664e - 0.0323x$, $R^2 = 0.8036$). In July 2008, the mixing depth of Jinpen Reservoir was 2.54 m, and the corresponding chlorophyll-*a* was 6.29 $\mu\text{g/L}$. In August, the mixing depth increased to 2.61 m and chlorophyll-*a* was 15.7 $\mu\text{g/L}$. A similar phenomenon occurred in 2009. According to Diehl, when the mixing depth is lower than a certain limit, the algal settling velocity would be the primary factor influencing algal abundance. It could be expected that light inhibition is obvious at the surface, and slightly increasing the mixing depth will reduce the light intensity so as to weaken the light inhibition.

2.3 Algal Pollution in Shibianyu Reservoir

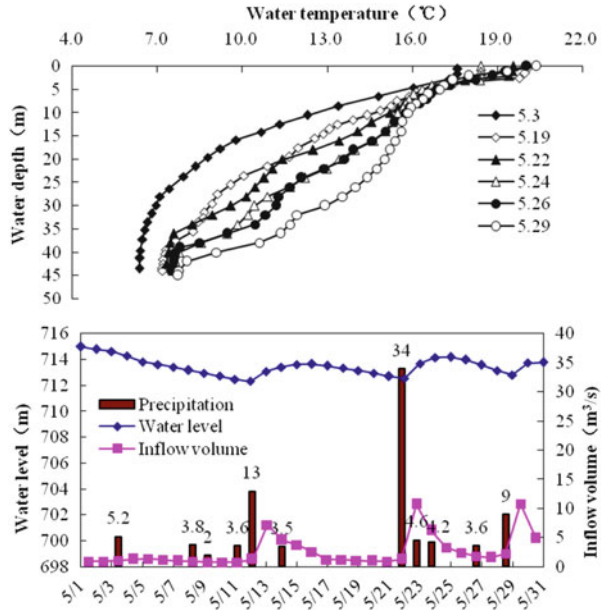
2.3.1 Algal Growth in Shibianyu Reservoir

High contents of nitrogen and phosphorus in Shibianyu Reservoir provide adequate nutrition for algal blooms. Water temperature, light, rainfall, and other environmental factors are also key factors for algal growth and community structure in Shibianyu Reservoir. The seasonal variations of algal abundance in 2011 and 2012 are shown in Fig. 13. Algae grew well from May to October. In spring, diatoms dominated and cyanobacteria were prevalent in summer.

Rainfall was frequent in the summer of 2011, which makes the water temperature and illumination intensity change continuously. Besides, due to the effect of dilution, rainfall played a key role in inhibiting algal growth, so algal blooms did not appear in 2011.

The rainfall in 2012 was close to the average level in Shibianyu Reservoir, so the algal dynamics in 2012 can typically reflect the common characteristics of algal

Fig. 14 Water temperature, rainfall, water level, and inflow in May (2012)



growth. It can be seen from Fig. 13 that algae pollution was serious in Shibianyu Reservoir in 2012. The diatoms started to grow rapidly from May and reached a peak in June. After a large rainfall (at the end of June), diatoms reduced. Cyanobacteria became dominant with the increase of water temperature and light intensity. Algal blooms were common from July to October in Shibianyu Reservoir. The algal abundance was as high as 1×10^8 cells/L. During this period, the reservoir had an obvious fishy smell, which seriously affected the water quality.

In order to further clarify the law of phytoplankton succession in Shibianyu Reservoir, algae were monitored frequently from May to October in 2012.

2.3.2 The Analysis of Algal Growth Characteristics and Influencing Factors

(1) Diatoms Growth Characteristics and Influencing Factors

Stratification started to appear in May in Shibianyu Reservoir, as shown in Fig. 14. During this period, rainfall was frequent and light, inflow changed little, and the water level fluctuated between 712 and 715 m. Due to the rainfall, the water temperature increased slowly in May. The surface water temperature was 17.59 °C and the temperature at the bottom was 20.35 °C. With the increase of water temperature and light intensity, the algae began to proliferate. In early May, chlorophyll-*a* was less than 3 µg/L. On May 19, chlorophyll-*a* increased up to 13.43 µg/L. The dominant species were *Cyclotella* and *Melosira*, which composed 26.26 % and 53.54 %, respectively. On May 22, the reservoir experienced a large

Fig. 15 Vertical distribution of algae in Shibianyu Reservoir in May (2012)

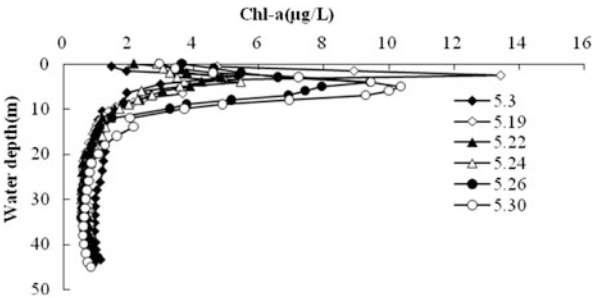
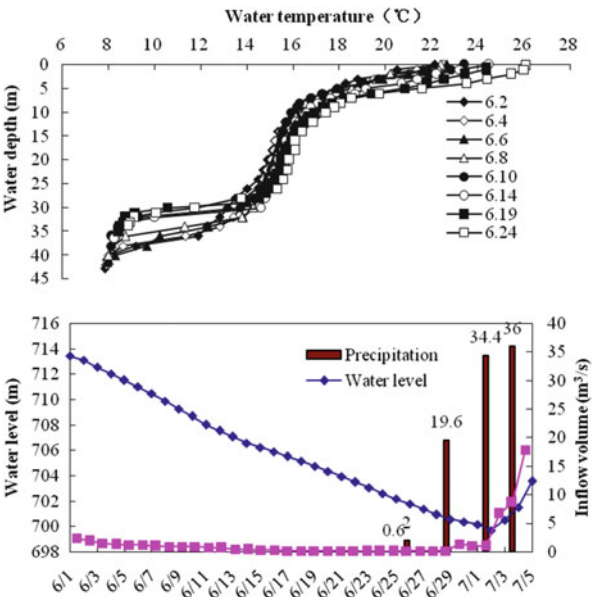


Fig. 16 Water temperature, rainfall, water level, and inflow in June (2012)



rainfall; thus, chlorophyll-*a* subsequently dropped to 4.84 µg/L. Then, the algal abundance increased again and the community changed. *Fragilaria* increased obviously and most of them gathered at a depth of 4–6 m (Fig. 15).

There was no rainfall in the Shibianyu Reservoir between late May and June 24. The water level dropped by 0.5 m/d. On June 1, the water level was 713.43 m and further declined to 700 m on July 2. Since July 2, continuous rainfalls have made the water level rise again. Changes of temperature, inflow, and water level are shown in Fig. 16.

It can be seen in Fig. 16 that stratification formed in June. On June 24, the water temperature was 26 °C and changed little below a depth of 10 m. Suitable temperature and enough light contributed to the rapid growth of algae (Fig. 17). The content of chla was about 50 µg/L and the algal abundance reached 5760×10^4 cells/L. The prevalent species was still *Fragilaria*, which occupied more than 80 %, but their individual size increased and the maximum diameter was close to 200 microns.

Fig. 17 Vertical distribution of algae in Shibianyu Reservoir in June (2012)

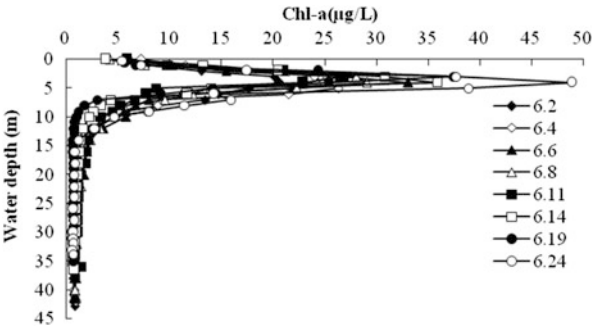
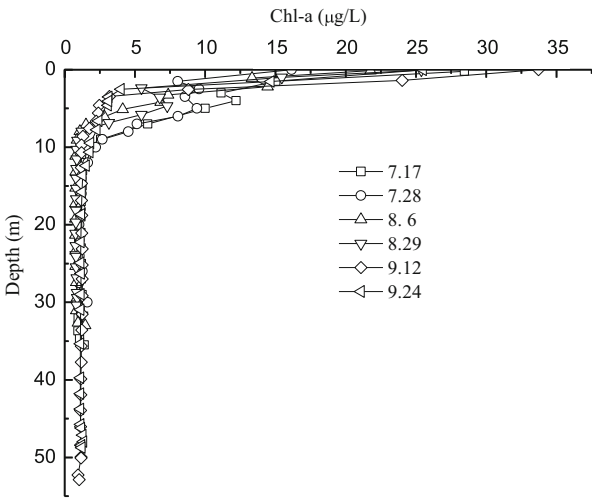


Fig. 18 Vertical distribution of cyanobacteria in Shibianyu Reservoir



(2) Distribution of Cyanobacteria

Figure 18 and 19 indicates that algal blooms were serious from July to October in Shibianyu Reservoir. A number of cyanobacteria often grouped together on the surface and algal abundance reached as high as 1×10^8 cells/L. As the water level decreased, the algal flocs would be left on the shore. In Shibianyu Reservoir, algal blooms were mainly composed of *Microcystis aeruginosa*, which occupied more than 90 %. *Microcystis* bloom is the most serious kind of water pollution. When it appears, the water gives off an unpleasant smell. Besides, algal toxin will be released after the algal cell fractures. *Microcystis* toxin is known as being the most harmful algal toxin.

The cause analysis of algal blooms in Shibianyu reservoir was investigated. Firstly, high contents of nitrogen and phosphorus in Shibianyu Reservoir provide sufficient nutrients for algae. Secondly, terrain conditions are suitable for algal growth: Adequate light is conducive to algal photosynthesis, and long hydraulic retention time also provides favorable conditions for algal blooms. Cyanobacteria can be prevalent for a long time due to its own competitive advantages: (1)

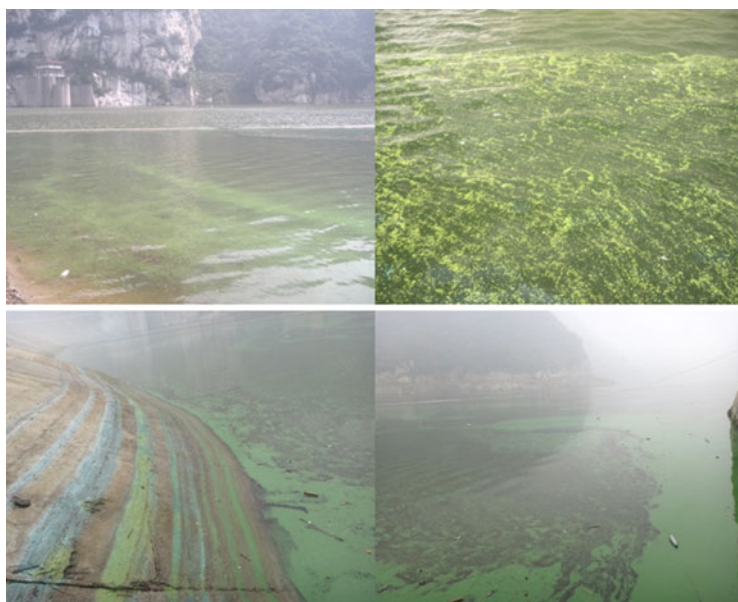


Fig. 19 Algal blooms in Shibiyanu Reservoir (July–October)

Cyanobacteria can prevent light inhibition; (2) Cyanobacteria have higher nutrient affinity than others; (3) Cyanobacteria can mainly gather at the surface to a depth of 0.3 m by airbags regulation; (4) Cyanobacteria toxins have an inhibitory effect on other algae.

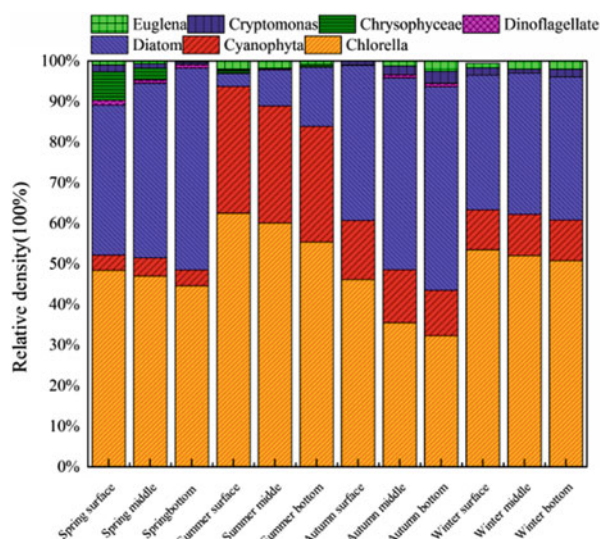
2.4 Algae Pollution of Zhoucun Reservoir

2.4.1 Algae Community Succession in Zhoucun Reservoir

A total of seven phyla, 57 genera, and 122 species were identified from April 2012 to November 2013. The total number was composed of 57 species of Chlorophyta, 28 species of Bacillariophyta, 25 species of Cyanophyta, three species of Euglenophyta, three species of Cryptophyta, two species of Xanthophyta, and three species of Pyrrophyta. Chlorophyta was the most important group in terms of species number (46.7 %), followed by Bacillariophyta (23 %), Cyanophyta (20.5 %), and the others. The algal abundance in Zhoucun Reservoir was very different in each season and at different depths.

In spring, Zhoucun Reservoir was dominated by Chlorophyta and Bacillariophyta. Specifically, the popular species were *Cyclotella*, *Synedra*, *Stephanodiscus*, *Cocconeis*, *Scenedesmus*, *Ankistrodesmus*, *Chlorococcales*, and *Chlamydomonas*. Chlorophyta was mainly distributed on the surface and Bacillariophyta was mainly distributed in the middle and bottom of the reservoir.

Fig. 20 Phytoplankton community succession in Zhoucun Reservoir



In summer, Cyanophyta dominated in the water, followed by Chlorophyta. The two species accounted for 80–90 % of the total. The vertical distribution of Chlorophyta presented a downward trend, while Bacillariophyta exhibited the opposite behavior. The prevalent species became less numerous and was mainly composed of *Chlorella*, *Microcystis*, *Anabaena*, and *Chroococcus*. *Microcystis* only dominated in the surface waters and the others were more common in the whole water.

In autumn, diatoms became dominant. The proportion of cyanobacteria and green algae decreased. The characteristics of the vertical distribution were similar to that in summer. The dominant genera were *Stephanodiscus*, *Synedra*, *Chlorella*, *Chroococcus*, *Lyngbya*, and *Anabaena*. *Chlorella* dominated in the whole water. Cyanophyta was mainly distributed in the surface and middle waters.

In winter, the phytoplankton community at different depths was similar. Green algae, diatoms, and cyanobacteria accounted for about 52 %, 30 %, and 14 %, respectively. The main prevalent species were *Synedra*, *Chlorella*, *Cyclotella*, *Chlamydomonas*, and *Chlorococcum* (Fig. 20).

The phytoplankton community varies with the change of trophic status. In general, *Chrysophyceae* and *Xanthophyta* are dominant in oligotrophic water. Dinoflagellates, Cryptophyta, and Bacillariophyta are prevalent in middle eutrophic water. *Chlorella* and Cyanophyta are popular in eutrophic water, while *Chlorella* and Cyanophyta are usually prevalent in the whole Zhoucun Reservoir, so the water quality of Zhoucun Reservoir can be judged as being eutrophic.

2.4.2 Seasonal Variations of Algal Abundance in Zhoucun Reservoir

The change of algal abundance in the surface water of Zhoucun Reservoir was unimodal. Seasonal variations of the vertical distribution were obvious.

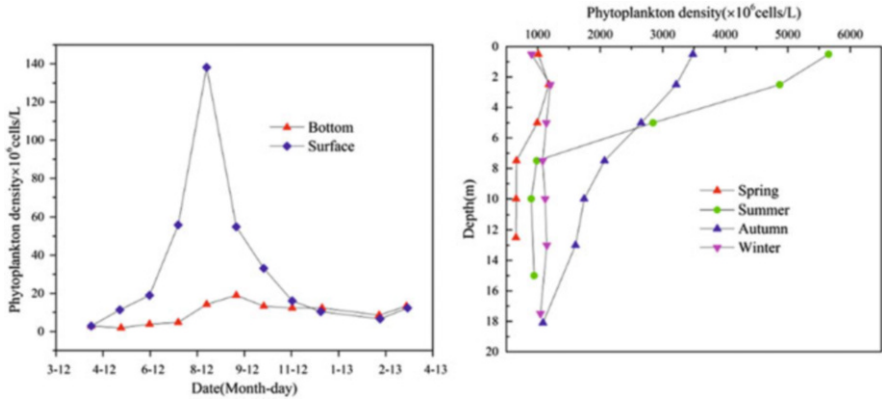


Fig. 21 Seasonal variations of algal abundance in Zhoucun Reservoir

In spring, the water temperature was still low while the light intensity increased. Algae started to grow on the surface. At the bottom, algal abundance changed little due to the low temperature and weak light intensity. Algal abundance in the surface layer was slightly higher than that at the bottom.

In summer, the water temperature and illumination both reached optimal conditions for algal growth. Meanwhile, the nutrient content was rich. The cell density of algae grew exponentially and the peak (1.39×10^8 cells/L) appeared in August. At the bottom, the cell density also increased, which was attributed to the deposition of algae. Illumination and water temperature dropped rapidly with the increase of depth, which makes the cell density of algae fall quickly in the vertical direction. Due to the influence of stratification, the water temperature changed little below 8 m. So, it follows that algal abundance changed little below 8 m in Zhoucun Reservoir.

The temperatures and light intensities fell in the autumn and the algal abundance decreased rapidly. Because there were still a large number of algae settling at the bottom, the algal abundance at the bottom was not significantly lower than that in summer. The vertical distribution in autumn was similar to the situation in summer because stratification still existed during this period.

Due to low temperature, weak light intensity, and short day time, the cell density of algae in the surface waters dropped to the lowest level throughout the year. The whole water mixed in the early winter, so there is no significant difference in algal abundance in the vertical direction (Fig. 21).

2.4.3 Analysis of Influencing Factors

Correlation and redundancy analysis were applied to explore the relationship between algal abundance and environmental factors. The algal abundance and water quality parameters are shown in Table 17. The serial numbers of dominant species are shown in Table 18.

Table 17 Algae and water quality in the surface water in Zhoucun Reservoir

Date	Algae abundance (million cells/L)	Chlorophyll- <i>a</i> (µg/L)	Rainfall (mm)	Water level (m)	Temperature (°C)	DO (mg/L)	pH	Turbidity (NTU)	TN (mg/L)	NO ₃ (mg/L)	NO ₂ (mg/L)	NH ₄ (mg/L)	TP (mg/L)	N/P	Fe (mg/L)
12-04	3.26	3.32	36.5	125.00	15.7	9.44	8.68		3.24	3.08	0.032	0.290	0.037	87.6	0.149
12-05	1.08	1.01	8	124.57	22.6	8.53	8.53		2.44	1.84	0.035	0.548	0.023	105.9	0.015
12-06	23.97	22.50	9	123.94	25.2	10.52	8.72		2.61	1.20	0.032	0.075	0.023	113.5	0.031
12-07	77.74	65.71	271	125.22	28.2	14.59	10.16	33.8	2.60	1.04	0.217	0.450	0.065	40.2	0.149
12-08	93.71	75.58	200.5	125.47	27.2	10.29	10.07	18.2	2.40	1.13	0.217	0.060	0.065	36.9	0.236
12-09	48.63	42.66	110	127.63	23.2	7.25	9.86	9.3	2.57	2.01	0.084	0.550	0.046	55.9	0.200
12-10	23.51	38.43	9	127.82	20.3	9.69	8.48	7.74	2.25	1.61	0.090	0.375	0.048	46.8	0.118
12-11	10.26	16.54	59.5	127.66	12.7	10.40	8.3	5	2.46	1.66	0.112	0.680	0.041	60.0	0.087
12-12	11.14	14.62	39	127.47	6.5	11.65	8.11	3.5	2.56	1.70	0.080	0.167	0.023	111.0	0.030
13-02	6.93	8.78	12.5	127.06	3.8	10.77	8.35	2.5	2.20	1.80	0.051	0.375	0.024	93.2	0.015
13-03	11.43	5.31	10.5	126.82	7.6	11.93	8.5	3.3	2.82	1.91	0.029	0.072	0.027	104.4	0.026
13-04	1.19	0.94	21	126.48	11.9	10.05	8.36	3.4	2.56	1.98	0.032	0.078	0.028	91.4	0.026

Correlation analysis indicates that algal abundance was positively related to temperature, pH, turbidity, and TP, while it was significantly negatively correlated with the ratio of N/P (Table 19).

As shown in Table 20, the RDA analysis results can explain 91.9 % of the information of phytoplankton and environmental factors. The eigenvalues of axes 1 and 2 are 0.591 and 0.169, respectively. The correlation coefficient between environmental factors and phytoplankton is 1.

In the RDA sequence diagram, the angle between the arrows of algae depicts the relationship between different species. The smaller the angle, the higher the correlation. The same arrow direction indicates a positive correlation, while the opposite direction implies a negative correlation. The length of the arrow of environmental factors indicates their effects on phytoplankton. Figure 22 shows that TN, TP, and water level are the main factors influencing the phytoplankton community structure in Zhoucun Reservoir. TN, Mn, and NO_3^- have little effect on phytoplankton. *Chlorella*, *Chlamydomonas*, *Chroococcus*, and *Anabaena* were distributed in the first quadrant and there was a strong positive correlation with TN, turbidity, COD, TP, and pH. *Cyclotella*, *Synedra*, and *Stephanodiscus* were mainly distributed in the third quadrant and a strong negative correlation with TN, turbidity, COD, TP, and pH existed. *Scenedesmus* and *Chlorococcum* are negatively related to the water level, while *Ankistrodesmus* and *Lyngbya* have a strong relationship with Fe.

2.5 Algae Pollution in Zhelin Reservoir

2.5.1 Seasonal Variations of Algal Abundance

The algal abundance gradually increased as the temperature increased. In October, the water temperature at the surface was 26 °C, and the algal abundance reached a maximum of 2.25 million/L. In November, the water temperature and algal abundance rapidly decreased with the decreasing temperature. In January, the water temperature dropped to 11.8 °C and the algal abundance reached its minimum of the entire year. The monitoring station in the front of the dam is located in Yongxiu County. Due to the large area of Zhelin Reservoir, the period when algal abundance reaches a peak are different at each monitoring point. The maximum algal abundance appeared in the fork inlet of the lake and shallow water monitoring sites, reaching up to 4.2 million/L (Fig. 23).

The monitoring results showed that Zhelin Reservoir was still in a middle trophic status.

2.5.2 Characteristics of Algae Community Succession

The diatoms showed a rising trend after the first reduction from June 2013 to May 2014. In spring and winter, diatoms were prevalent and accounted for more than

Table 19 Correlation between algal abundance and environmental factors

	Algae abundance	Chlorophyll- <i>a</i>	Rainfall	Water level	Temperature	DO	pH	Turbidity	TN	NO ₃	NO ₂	NH ₄	TP	N/P	Fe
Algae abundance	1.000														
Chlorophyll- <i>a</i>	0.953**														
Rainfall	0.457*	0.498*													
Water level	0.115	0.270	0.015												
Temperature	0.749**	0.687**	0.480*	-0.224											
DO	0.133	0.051	0.169	-0.077	-0.101										
pH	0.732**	0.637**	0.426	-0.166	0.893**	-0.043									
Turbidity	0.900**	0.897**	0.565*	-0.113	0.868**	-0.152	0.668**								
TN	-0.105	-0.160	-0.063	-0.319	-0.147	-0.130	-0.140	-0.127							
NO ₃	-0.482*	-0.442	0.036	0.138	-0.317	-0.392	-0.173	-0.433	0.299						
NO ₂	0.603**	0.738**	0.299	0.404	0.424	-0.067	0.401	0.624**	-0.246	-0.470*					
NH ₄	0.010	0.169	0.076	0.417	0.209	-0.235	0.111	0.293	-0.175	0.181	0.390				
TP	0.703**	0.771**	0.679**	0.272	0.602**	-0.036	0.620**	0.900**	-0.301	-0.149	0.620**	0.219			
N/P	-0.640**	-0.700**	-0.597**	-0.228	-0.559*	-0.061	-0.613**	-0.851**	0.498*	0.292	-0.589**	-0.178	-0.955**		
Fe	0.635**	0.709**	0.469*	0.134	0.435	-0.046	0.592**	0.813**	-0.121	-0.166	0.482*	0.059	0.801**	-0.784**	1.000

Table 20 Statistical results of RDA

Axis	1	2	3	4
Eigenvalue	0.591	0.169	0.095	0.063
Cumulative percentage of species, %	59.1	76	85.6	91.9
Cumulative percentage of species and environmental correlation, %	59.1	76	85.6	91.9
Correlation of species and environment	1	1	1	1

Fig. 22 RDA sequence diagram of phytoplankton and environmental factors in Zhoucun Reservoir

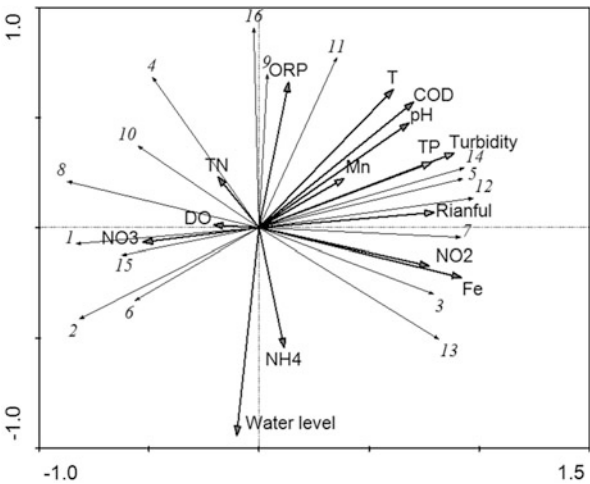
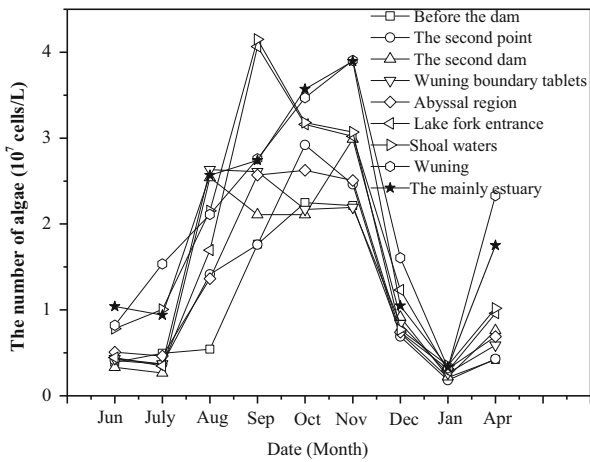
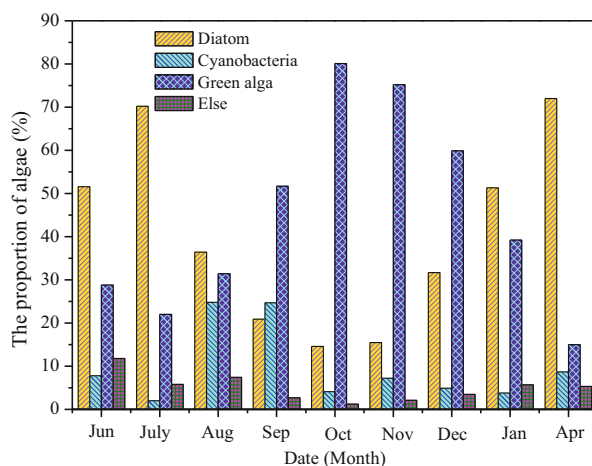


Fig. 23 Seasonal variations of algal abundance in Zhelin Reservoir (2013–2014)



50 %. The proportion of green algae was high throughout the year. In October and November, the proportion of green algae was up to 77.5 %. Cyanobacteria and Dinoflagellates had a low proportion throughout the year. The main limiting factors for algal growth in Zhelin Reservoir are nitrogen and phosphorus (Fig. 24).

Fig. 24 Statistical analysis of algae species change in Zhelin Reservoir (2013–2014)



3 Odor and Substance-induced Olfactory

Odor is one of the earliest and most straightforward parameters that can be used to evaluate the quality of drinking water. Odor is a nerve stimulation integrated signal caused by olfactory-induced compounds in water acting on the person's nose, mouth, tongue, and other sensory peripherals [20]. The olfactory problem is a primary one. When people realize the odor of drinking water, the majority tend to believe that the water is poisonous and avoid using it.

Generally, the influence of water taste and odor is huge, which could affect the taste of water for drinking, reduce the quality of drinking water and aquatic products, cause disgust, nausea, and other unpleasant psychological effects, and reduce the aesthetic value of lakes and reservoirs. However, conventional water purification processes cannot remove the smell effectively. Some special odor control technologies or combined water purification processes are helpful in removing the taste and odor, but these will increase the cost of the water purification, thereby increasing the cost of living [21]. According to the US water industry statistics, 4.5–10 % of total revenue is used for solving taste and odor problems of drinking water.

3.1 The Odor Emergency Event in Huyan Waterworks

From mid-August, 2010, the Taiyuan municipal department received several water odor complaints from residents. They complained that the odor of drinking water is musty and the intensity of this mustiness increased after being boiled. The complaints mainly came from the west of Riverside Road Wanbailin and suburbs of the city. According to the distribution of water plants and the water supply network in

Taiyuan City, the problematic water came mainly from Huyan waterworks, whose raw water was taken from the Fenhe Reservoir.

According to the monitoring results for water quality, the water was musty before it flowed into the plant. So, the taste and odor of drinking water might come from the source water, namely the Fenhe Reservoir. To maintain social stability and protect public health, emergency measures were taken to eliminate the odor. Powder-activated carbon with a dosage of 14.4 mg/L was added to the aqueduct in front of plant at about 500 m intervals. After the implementation of emergency measures, the musty feeling was relieved but was not removed completely. Therefore, further studies on scientific and effective odor control technologies are needed.

Taste and odor reduces the quality of drinking water, causing the users discomfort and skepticism on the safety of drinking water, and, thus, leading to dissatisfaction and complaining. Besides, taste and odor in drinking water may affect human health or become a factor inducing certain diseases. If not resolved properly, the issue of taste and odor in drinking water will lead to unpredictable negative effects on Taiyuan residents, social stability, and industrial and agricultural production.

3.2 Identification and Analysis of Odor Compounds Sources

3.2.1 Identification of Odor Compounds

(1) Types of Odor Compounds

Given the various types of olfactory compounds, their detection is difficult due to the differences in molecular structure. The appropriate detection methods to identify the origin compounds causing odor should be selected based on their types [22]. According to the complaints from residents and the situation provided by Huyan waterworks, Taiyuan tap water has a musty odor. The in situ-collected raw water showed different intensities of mustiness. Some water samples became mustier after being sealed for some time. Therefore, must is the main odor type in the Fenhe Reservoir.

(2) Identification of Odor Compounds

Based on existing researches, compounds with a musty odor in natural water include geosmin, 2-methyl isobutyl camphane alcohol (2-methylisborneol or 2-MIB), 2,3,6-trichloroanisole (2,3,6-trichloroanisole, or TCA), and 2-isopropyl-3-methoxypyrazine (IPMP) [23, 24]. Geosmin and 2-MIB are the main culprits in the musty water incident in China. The “drinking water health standards” of China (GB 5749–2006) sets geosmin and 2-MIB as the index of drinking water and its limit is 10 ng/L.

The detection technique is the headspace solid-phase microextraction/gas chromatography method. According to the water quality survey, the musty odor of raw water increases with standing time. In order to increase the concentration of the

Table 21 Test results of odor and taste compounds

Test items	Tap water	Water of reservoir		
	3-Sep	2-Sep	18-Sep	9-Oct
2-methyl isopropyl alcohol	17	40	64	25
Geosmin	—	—	—	—

odor compounds, the water samples were detected and analyzed after being left standing for 24 h. Table 21 shows the testing results of taste and odor compounds.

As shown in this table, the raw water contains 2-methylisoborneol with the highest concentration of 64 ng/L, but no geosmin was found. Meanwhile, 2-MIB, with a concentration of 17 ng/L, was detected from the effluent of Huyuan water treatment plant on September 3, which suggests that 2-MIB is the main olfactory compound in Taiyuan tap water and Fenhe Reservoir.

(3) Physical and Chemical Properties of 2-MIB

The formula for 2-MIB is $C_{11}H_{20}O$ and its molecular weight is 168. According to the molecular structure, 2-MIB is saturated cyclic tertiary alcohol and its structure is similar to the pentagonal ring. The boiling point of 2-MIB is 196.7 °C and its solubility is 194.5 mg/L. The distribution ratio of 2-MIB in octanol and water K_{OW} is 3.13, which means that 2-MIB belongs to the semi-volatile, weakly polar molecules. 2-MIB is a micropolar fat-soluble compound which is insoluble in water. It is also soluble in methanol, acetone, hexane, methylene chloride, and other organic solvents. Since 2-MIB possesses a saturated cyclic structure, it is resistance to oxidation.

Different concentrations of 2-MIB show different types of odor. High concentration tastes camphor while low concentration tastes musty. People's sense of smell is extremely sensitive to 2-MIB and can sense the smell generated by trace 2-MIB. It is generally believed that the olfactory threshold of 2-MIB is 10 ng/L. However, the olfactory thresholds are affected by the tester's sensitivity, testing environment, and other factors. 2-MIB odor thresholds obtained by different researchers are different. Lalezary suggests that the 2-MIB olfactory threshold is 10 ng/L, while Ashitani gives the limit in the range 4–20 ng/L.

Currently, no researches indicate that 2-MIB causes direct harm to the human body, but the musty smell makes consumers uncomfortable. It is necessary to control the content of 2-MIB in water strictly. In Japan, the limit of concentration of 2-MIB is 10 ng/L in drinking water. In China, the “drinking water health standards” (GB 5749–2006) also defines 2-MIB as a reference index and the limit is 10 ng/L.

3.2.2 Sources of Olfactory-Induced Substances

Based on previous research results, 2-MIB in water mainly comes from actinomycetes and secondary metabolites of algae secretions [25]. Actinomycetes and small single-cell bacteria can produce 2-MIB. Cyanobacteria can also produce 2-MIB,

and the circumstances under low levels of cyanobacteria can cause water body odor. The common species of cyanobacteria that produce 2-MIB are *Ankistrodesmus*, *Phormidium*, *Oscillatoria*, *Phormidium*, and *Anabaena*.

It was found that the 2-MIB in Fenhe Reservoir came mainly from algae. The phenomenon of delayed odor release is due to the death of algae in dark conditions, which could cause intracellular olfactory substances to be released into the water.

3.3 Emergency Countermeasures for Controlling Algae-Induced Olfactory

The treatment processes adopted by Huyan water plant include “coagulation–sedimentation–filtration–disinfection”, which are inefficient for odor removal. To improve odor removal, powdered activated carbon was added to the aqueduct every 800 m in the front of the Huyan water plant. After the addition of powdered activated carbon, the musty effluent from Huyan water plant decreased, but the mustiness in tap water was not eliminated. The concentration of 2-MIB in the water tanks was 17 ng/L on September 3, and this value is still higher than the permitted odor threshold. Because the dosing points of powdered activated carbon are near the water plant, the reaction time is very short and the adsorption capacity of activated carbon may not be fully exerted.

The odor problem caused by algae could not be removed efficiently by conventional water treatment processes. In order to select proper control schemes for taste and odor, many factors, such as the sources of odor-causing compounds, the current condition of the plant, and the urgent degree of processing, should be taken into consideration. The feasibilities of various 2-MIB removal techniques are discussed below:

1. Chemical oxidation: To our best knowledge, ozone is one of the most efficient oxidants for 2-MIB removal, since it can kill the algae effectively. This method is technically feasible, while the design and construction of an ozone system requires a long time, which makes it unsuitable for emergency odor treatment.
2. Adsorption: Although powdered activated carbon is widely used in water taste and odor control, the use of powdered activated carbon alone could not remove the odor of raw water.
3. Biological degradation: The olfactory-induced degradation microorganisms are crucial for this method, while the generation and accumulation of the microorganisms need a relatively long time. Thus, this method is also not suitable for emergency odor treatment.
4. Photocatalytic oxidation and ultrasonic radiation: Both of these techniques are still limited to the laboratory scale, and there is a long way to go before they can be considered for real applications.

In summary, it seems unrealistic to solve algae-induced olfactory emergencies by a single control technology. So, combined methods are considered to be the most suitable solutions to solve this problem. On the other hand, it is also difficult to

construct special emergency treatment plants due to the constraints of the existing water plant layout, pipe alignment, and power distribution. However, from another perspective, there is a great distance between water sources and water treatment plants. Therefore, combined with the experience of emergency treatment [26], odor control processes could be conducted in the aqueduct system. Accordingly, the “killing algae + adsorption” process is proposed.

Among the existing techniques, chemical oxidation is ideal for killing algae. The most commonly used oxidants include chlorine, chlorine dioxide, ozone, potassium permanganate, and copper salts. From the view of operating convenience and drug safety, potassium permanganate is the most suitable agent.

In conclusion, the process “potassium permanganate peroxidation + powdered activated carbon adsorption” was suggested to deal with odor emergencies. The mechanisms of the method involve two steps: (1) adding potassium permanganate into the raw water aqueduct system to damage algal cells and cause the release of 2-MIB in algae cells; (2) dosing powdered activated carbon to absorb the released 2-MIB.

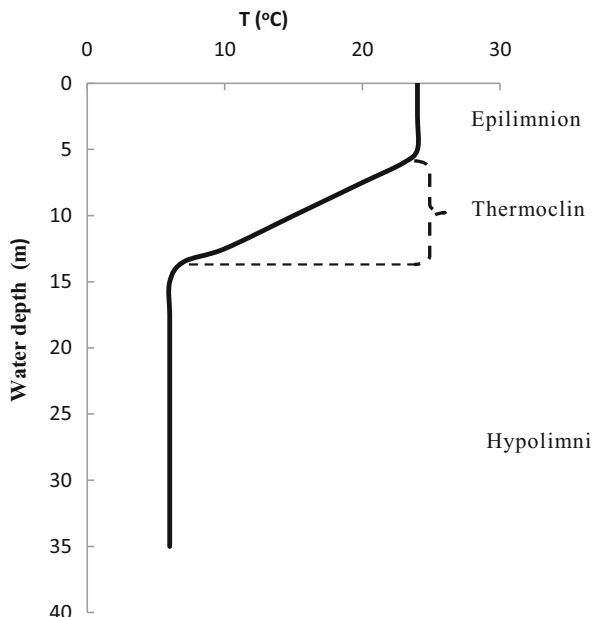
4 Water Pollution Related to Stratification of the Reservoir

Stratification is a natural occurrence which occurs in reservoirs due to the change in water density with temperature at different depths. The stratification includes forward and reverse stratification [27]. Forward stratification means that the temperature of the upper layer is higher than that of the lower layer, and vice versa for reverse stratification.

Forward stratification is formed at the end of spring and early summer each year, due to the strong solar radiation warming the surface water quickly, so the density decreases, whereas the water temperature of the lower layer is relatively stable, remaining at a low level, and, thus, at a greater density. Due to the density difference between the upper and lower bodies of water, the interchange between the upper and lower water bodies is hindered, and a stable stratification is formed. Generally, the stratified water body is classified into three layers (Fig. 25): epilimnion, thermocline, and hypolimnion. The epilimnion is the top layer in a thermally stratified body of water, which is warmer and typically has a higher pH and higher dissolved oxygen concentration [28]. The hypolimnion is the dense, bottom layer of water in a thermally stratified body of water, and the hypolimnion is the coldest layer in summer. The thermocline is a thin but distinct layer, in which the temperature changes more rapidly with depth, and it is a transition layer between the epilimnion and the hypolimnion. A typical stratification structure is shown in Fig. 25.

Reverse stratification is possibly formed in cold winter. Due to the cold effect, the water temperature decreases and the density of the surface water also decreases as the water temperature drops to less than 4 °C; however, the water temperature remains stable (4 °C) at the bottom. Therefore, the density of the bottom water is

Fig. 25 Stratification in reservoirs



higher than that of the surface water, and the exchange of the upper and lower water bodies is hindered, so a stable reverse stratification is formed.

Storms and intense rainfall will destroy the stratification of a reservoir; as the water depth is small, the temperature stratification can be easily destroyed. When the water depth is less than 10 m, it is difficult to form a long-term stable stratification; even if it is formed, it is very fragile. When the water depth is greater than 30 m, it can form a long-term stable stratification. When the water depth is between 10 and 30 m, the stratification is weak and susceptible to wind, temperature, and other factors [29].

The stratification of a reservoir will deteriorate the water quality, including the decrease of DO in the bottom layer, the deterioration of the aquatic ecological environment, the release of contaminants from the sediments, and algal blooming [30, 31]. Due to the lack of water exchange between the upper and lower layers and the respiration of microorganisms and aquatic animals, oxygen in the lower layer gradually reduces close to zero eventually. Under anaerobic conditions, nutrients (N, P, organic matters) are released into the water, and, thus, increase the odor and transparency, and promote the growth of algae [32].

The basic characteristics of reservoirs are summarized as follows:

1. The hypoxia at the bottom of the reservoir: In most of the high-depth reservoir water, bottom oxygen consumption and temperature stratification will cause the bottom water layer to exhibit anaerobic and anaerobic conditions.
2. The release of pollutants from sediments under anaerobic environment: Under anaerobic conditions, it will lead to the release of ammonia, phosphorus, Fe, Mn,

and other heavy metal pollutants, and also result in color elevation, abnormal odor, and water quality deterioration.

3. The pollution derived from the water mixing: At the end of autumn or the beginning of spring, the reservoir will mix automatically, and the polluted bottom water will spread throughout the reservoir. For example, the frequent pollution of ammonia in the reservoir is due to the automatic mixing at the beginning of spring.
4. The algal blooming in the upper layer water: Most of the northern reservoirs have been used for regulation for years. Cyanobacteria blooms frequently due to the long retention time of water and poor mobility, combined with the endogenous release of nitrogen from sediments.
5. The cyclical pollution of water quality: In southern reservoir water, it is periodically contaminated with algae and organic matters in the hot summer season. In northern reservoir water, it is periodically contaminated with algae and organic pollutions in the hot summer season as well, but is also contaminated with high nitrogen and phosphorus, high chroma, and organic matters in winter and spring.

In the following, the effects of stratification on the water quality are discussed in detail in the four typical Chinese reservoirs.

4.1 The Stratified Characteristics of Jinpen Reservoir and Its Impact on Water Quality

4.1.1 The Seasonal Thermal Stratification Characteristics and Dynamics of Jinpen Reservoir

Figure 26 shows the vertical temperature change over a period of 2 years. From the spring, the temperature of air starts to increase and, also, the temperature of the surface water in Jinpen Reservoir gradually increases; however, the temperature of the bottom water increases slowly compared to that of upper layer water, which induces a significant difference between the upper layer water and the bottom layer water, and results in the formation of stratification. At the end of June, the three typical layers (epilimnion, thermocline, and hypolimnion) are formed, i.e., forward stratification. The temperature of the upper layer (epilimnion) changes with the variation of climate, and the highest temperature is around 27–28 °C in July and August. The bottom layer (hypolimnion) remains quite stable and the temperature is around 7 °C. In October, the temperature of the upper layer water starts to decrease with the decrease of air temperature. In January of the second year, the temperature of the upper layer water decreases by the same amount as that in the bottom water (7 °C); the water between the upper and bottom waters starts to mix automatically. With the arrival of spring warming, a new round of temperature stratification begins.

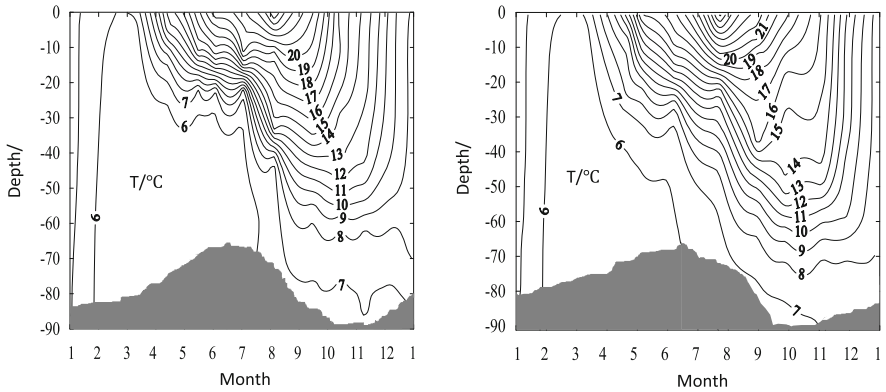


Fig. 26 Water temperature profiles in Jinpen Reservoir

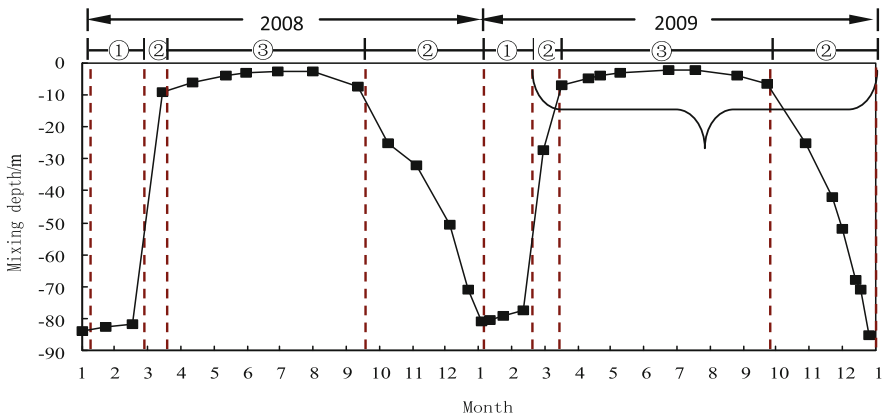


Fig. 27 Seasonal variation of mixing depth in Jinpen Reservoir (1. Mixing period, 2. Transition period, 3. Stratification period)

The surface mixed layer is a layer where this turbulence is generated by winds and surface heat fluxes. The surface mixed layer depth is defined as the range of depth where the temperature difference is not more than 1 °C from the top layer water [33]. The change of the mixed layer depth with season is shown in Fig. 27.

In recent years, the frequency of storm events has increased and they have a greater impact on a reservoir’s thermal stratification structure. The influent, which carries large amounts of suspended solids, has an increased runoff and muddy bottom density, which leads to it sneaking into the bottom of the reservoir. As shown in Fig. 28, the mixing period is quite different in 2011–2012 due to the impact of storm runoff. For example, the water temperature at the bottom of the reservoir rose from 6.25°C to 12.66 °C by the end of July 2011 after the first heavy

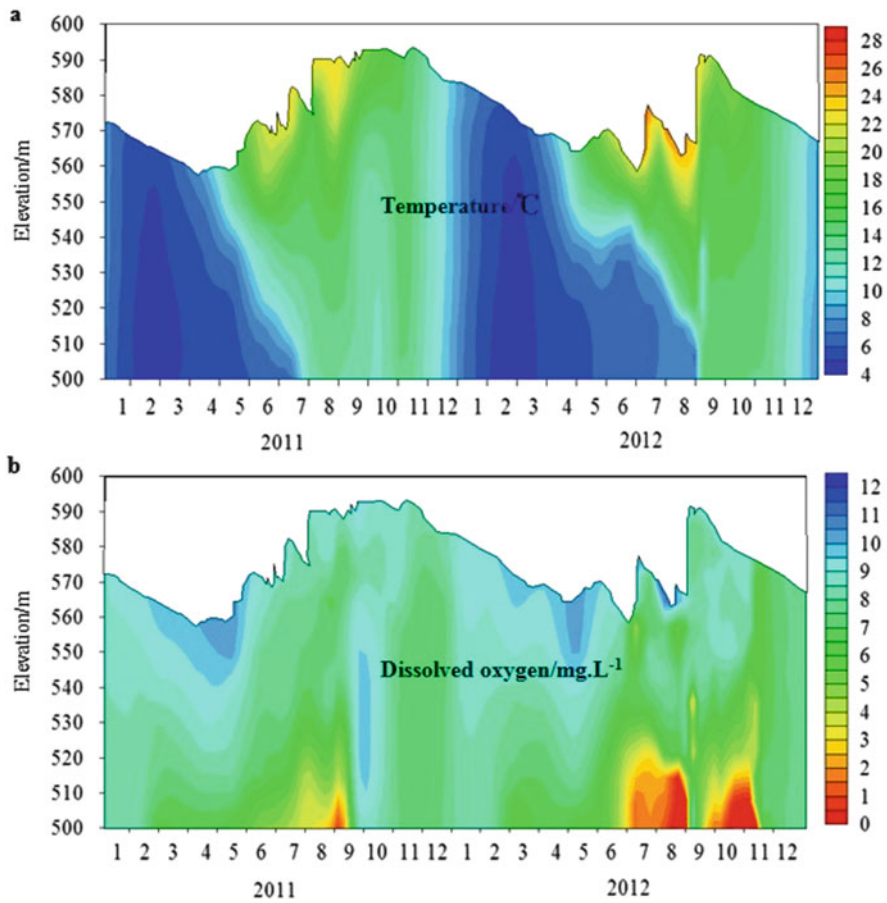


Fig. 28 Temperature and oxygen change in the vertical direction in Jinpen Reservoir (2011–2012)

rainfall and the surface water temperature also continued to decrease, so the reservoir mixed in advance. After the first rainfall in September 2011, the temperature difference of the upper and bottom layers in the main reservoir area reduced to 10.5 °C, and with the passage of time, the temperature difference between the upper and bottom layers in the main reservoir area was no longer changed. With the continuous rainfall, the temperature stratification in the main reservoir area has undergone a significant change: before rainfall, there is a greater temperature gradient at water depths from 0 to 17 m; after rainfall, the temperature of the bottom layer increased significantly, and the DO increased in the range of 0–30 m as well, which is due to the input of oxygen-rich water upstream undercurrent.

Fig. 29 Phosphorus concentration at different depths in Jinpen Reservoir

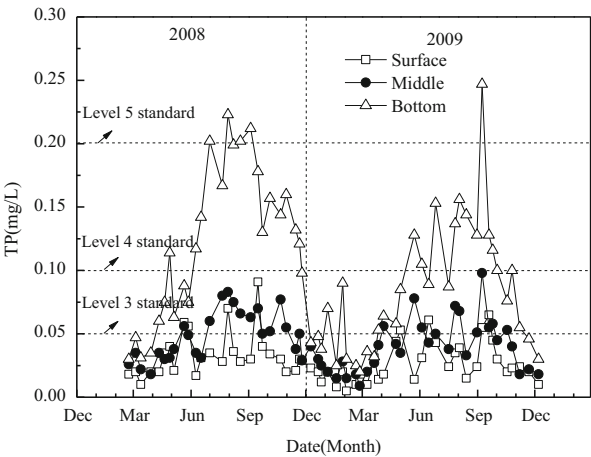
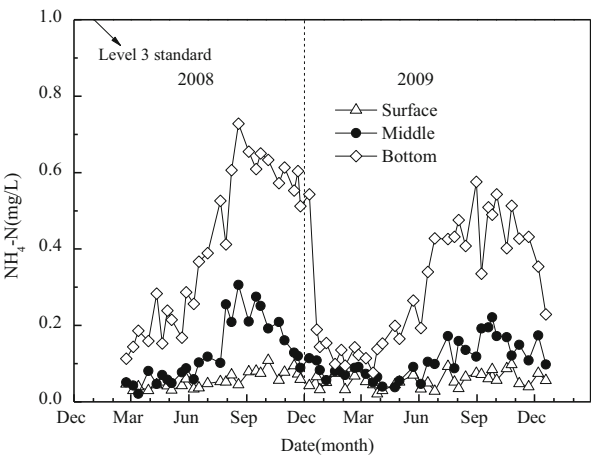


Fig. 30 Ammonia concentration at different depths in Jinpen Reservoir



4.1.2 The Water Quality Problems Derived from the Thermal Stratification of Jinpen Reservoir

(1) The Release of N and P Due to Thermal Stratification

With the seasonal water environment hypoxia, anaerobic conditions at the bottom of Jinpen Reservoir, and different forms of nitrogen, phosphorus fractions continued the “transformation–release–accumulation–diffusion” process in sediments under strongly reducing conditions, which accelerates the process of eutrophication in the entire reservoir area [34, 35]. Figures 29 and 30 show the changes in the spatial and temporal distributions of total phosphorus and ammonia in 2008 and 2009 in Jinpen Reservoir.

As shown in Figs. 29 and 30, during the stable thermal stratification period, the total phosphorus and ammonia increase rapidly with the increase of water depth. In this period, the water body between water layers is relatively stable vertically and do not blend, with the reservoir bottom water environment appearing seasonally in an anoxic/anaerobic state and different forms of nitrogen and phosphorus are released into the interstitial water, which then diffuses into the overlying water [36]. Once the mixing period begins, the nitrogen and phosphorus content of all the water layers tend to homogenize. The maximum concentration of phosphorus and ammonia occur in August and September each year in the bottom water. In March of the following year, the concentration of nutrients decreased to a minimum, corresponding to 0.223–0.247 mg/L (TP) and 0.576–0.728 mg/L (NH₄-N).

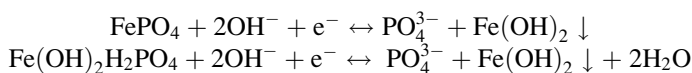
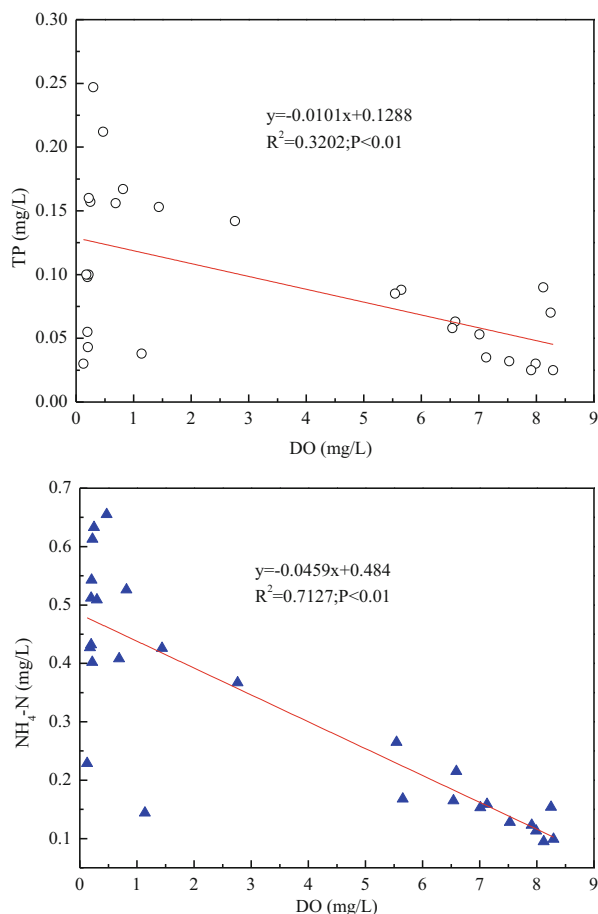
With respect to total phosphorus in the bottom water, it is higher than the national surface water quality standard (GB 3838–2002) (class III), and even reaches class V or worse than class V. The excessive periods are mainly from July to October annually. As can be seen from the results, it is clear that the main reason for total phosphorus excess is the release from sediments.

With respect to ammonia, although the concentration is always lower than the national surface water quality standard (GB 3838–2002) (class III) (1.0 mg/L), it shows a clear seasonal change pattern with the periodic alternation of a redox environment in Jinpen Reservoir. During the stable thermal stratification period, the concentration of ammonia always remains stable. Once the mixing period begins, the concentration of ammonia starts to decrease. As shown in Fig. 31, there is a negative correlation between the DO and the ammonia concentration in the bottom layer ($R^2=0.72$), and the ammonia concentration increases with the decrease of DO concentration. Comparatively, the correlation between total phosphorus and DO is quite weak ($R^2=0.32$).

In fact, the effect of the DO concentration on the concentration of total phosphorus is attributed to the change of water pH. Inorganic phosphorus accounts for about 65–73 % of the sediment of Jinpen Reservoir, and it is the main form of released phosphorus. Inorganic phosphorus is composed of calcium phosphate (Ca-P: 50–55 %), iron and aluminum-bound phosphorus (Fe/Al-P: 26–31 %), and occluded phosphorus (OP). Ca-P is quite stable and is only released in acidic environments. O-P is also fairly stable and can be released in acidic and reductive conditions. With respect to the active Fe/Al-P, it can be significantly released under the condition of basic pH and lower DO. Under anaerobic conditions, Fe³⁺ is reduced to Fe²⁺ and phosphorus is released from Fe/Al-P in sediments, but the Fe²⁺ and PO₄³⁻ mainly accumulate in the particle gaps. Once the pH increases, OH⁻ ions would diffuse into the interstitial water and combine with Fe²⁺ to generate Fe (OH)₂ precipitation. The concentration of Fe²⁺ in the particle gaps will decrease, which then accelerates the dissolution reduction of Fe/Al-P and the release of total phosphorus into overlying water.

The release of total phosphorus under the condition of lower DO and higher pH can be expressed as follows:

Fig. 31 Correlation between phosphorus/ammonia with oxygen at the bottom of Jinpen Reservoir



In the bottom of Jinpen Reservoir, the pH decreases gradually with the increase of anaerobic conditions. The release of total phosphorus from Fe/Al-P is hindered, so the increase of total phosphorus is attributed to the release of Ca-P.

(2) The Release of Heavy Metals Due to Thermal Stratification

The vertical distribution of total iron in Jinpen Reservoir is shown in Tables 22–25. The data show that the change in the concentration of total iron is closely related to the periodical thermal stratification of Jinpen Reservoir.

In spring, corresponding to the early period in the formation of thermal stratification, the reservoir is full of oxygen and is in a slightly alkaline state. Iron exists in the form of insoluble oxides and accumulates in the sediments. This process can be expressed as follows:

Table 22 The vertical distribution of total Fe in Jinpen Reservoir in March

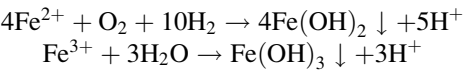
Date	Depth (m)	Water temperature (°C)	DO (mg/L)	pH	Total Fe (mg/L)
2008-03-09	1.20	6.11	10.04	7.46	<0.03
2008-03-09	3.89	5.46	9.88	7.36	<0.03
2008-03-09	10.51	5.39	9.86	7.33	<0.03
2008-03-09	20.63	5.33	9.71	7.33	<0.03
2008-03-09	30.37	5.32	9.48	7.31	<0.03
2008-03-09	40.81	5.28	8.97	7.27	<0.03
2008-03-09	50.56	5.25	8.49	7.26	<0.03
2008-03-09	56.03	5.22	8.22	7.26	<0.03
2008-03-09	65.53	5.20	8.17	7.24	<0.03
2008-03-09	80.76	5.21	7.99	7.22	<0.03

Table 23 The vertical distribution of total Fe in Jinpen Reservoir in June

Date	Depth (m)	Water temperature (°C)	DO (mg/L)	pH	Total Fe (mg/L)
2008-06-30	2.84	21.06	8.50	8.57	<0.03
2008-06-30	12.87	19.95	7.28	7.62	<0.03
2008-06-30	22.73	6.96	6.44	7.04	<0.03
2008-06-30	32.54	6.10	6.60	6.83	<0.03
2008-06-30	42.70	5.84	5.89	6.78	0.033
2008-06-30	52.92	5.72	4.17	6.60	0.041
2008-06-30	62.74	5.69	3.22	6.49	0.058
2008-06-30	68.43	5.92	2.76	6.40	0.072

Table 24 The vertical distribution of total Fe in Jinpen Reservoir in October

Date	Depth (m)	Temperature (°C)	DO (mg/L)	pH	Total Fe (mg/L)
2008-10-11	0.80	16.40	8.16	7.96	0.068
2008-10-11	10.00	16.03	7.98	7.45	0.072
2008-10-11	20.00	15.25	6.39	7.16	0.083
2008-10-11	30.00	13.60	6.25	7.05	0.058
2008-10-11	40.00	13.33	5.79	6.92	0.072
2008-10-11	50.00	11.80	2.22	6.75	0.088
2008-10-11	60.00	8.22	1.57	6.49	0.096
2008-10-11	70.00	7.20	1.17	6.32	0.108
2008-10-11	82.00	6.90	0.43	6.27	0.144
2008-10-11	85.00	7.01	0.25	6.15	0.317



In this condition, the redox reaction occurs in the sediments and no release of iron into overlying water takes place. Therefore, the vertical concentration of the total iron water is very low in the reservoir (Table 22), and no excessive results of iron are found.

Table 25 The vertical distribution of total Fe in Jinpen Reservoir in January

Date	Depth (m)	Water temperature (°C)	DO (mg/L)	pH	Total Fe (mg/L)
2009-01-28	1.10	6.99	9.19	7.25	0.038
2009-01-28	5.14	6.96	9.03	7.21	0.033
2009-01-28	10.11	6.95	8.91	7.21	<0.03
2009-01-28	20.15	6.91	8.78	7.22	<0.03
2009-01-28	29.85	6.91	8.66	7.21	<0.03
2009-01-28	39.98	6.91	8.27	7.22	<0.03
2009-01-28	50.20	6.86	8.19	7.20	0.031
2009-01-28	59.55	6.83	8.27	7.21	<0.03
2009-01-28	64.98	6.72	8.28	7.20	<0.03
2009-01-28	74.00	6.52	8.26	7.19	0.041
2009-01-28	80.31	6.48	8.25	7.18	0.058

In summer, the iron concentration shows a different pattern (Table 23). With the development of thermal stratification, the dissolved oxygen begins to decay below the thermocline in water, and gradually turns into anaerobic conditions. The degradation of organic matter results in a slightly acidic water body. Fe in the sediment was reduced to dissolved Fe and migrated to the interstitial water layer, and then diffused into the overlying water driven by the concentration gradient. In this period, the characteristics of total iron in Jinpen Reservoir can be summarized as: the total iron concentration began to increase in the bottom of the reservoir, while it was still high in the upper water body, due to the higher dissolved oxygen content (oxidation state), plus the “bottleneck” of the thermocline, so the total iron concentration in the reservoir was still low.

In autumn, the dissolved oxygen continues to reduce due to the persistence of thermal stratification (Table 24). With the increased anaerobic level, the process of “transformation–release–diffusion” reached its peak, and the total iron concentration exceeded the national surface water quality standard (0.3 mg/L) (GB 3838–2002). Meanwhile, with the start of the local convection in the upper layer of the reservoir, the mixed layer depth increased, the range of the thermocline compressed, and the mass transfer was accelerated, which resulted in the increase of the total iron concentration. However, the total iron content in the water did not exceed the national surface water quality standard, due to the limited release and diffusion strength and the higher concentration of dissolved oxygen in the upper layer water.

In winter, the concentration of iron in the reservoir is shown in Table 25. In this stage, the reservoir was in a mixed and aerobic state, the water pH quickly increased up to a neutral–alkaline level, the release process of Fe was severely inhibited, and the concentration became uniform due to the convective mixing process. The total iron concentration decreased greatly.

In summary, before the period of mixing, the total iron content tended to increase in the water along the whole depth due to the thermal stratification, Fe release from the sediments was further enhanced, and the total iron concentration in

the water body of the bottom layer significantly increased. After mid-December, with the mixing depth increasing, the mass transfer resistance was significantly weakened and the total iron concentration accumulation in the reservoir bottom decreased. In early January, the dissolved oxygen concentration near the sediment rapidly increased up to 1 mg/L, and anaerobic conditions disappeared, so the release of iron slowed down. Once it reached the fully mixed period, the Fe release process became stagnant; however, a large amount of pollutants is carried up to the upper water body, due to the strong convective mixing; thus, the Fe content in the surface water rapidly increases. After that, the Fe concentration in the water remained at quite a low value, with a sustained oxygen-rich state in the reservoir.

4.2 The Stratified Characteristics of Shibianyu Reservoir and Its Impact on Water Quality

4.2.1 The Seasonal Thermal Stratification Characteristics and Dynamics of Shibianyu Reservoir

The vertical change of temperature in Shibianyu Reservoir is shown in Fig. 32. From March to May, it is the formation period of thermal stratification. During this period, the temperature in the upper layer water increased with increasing air temperature, but it increased slowly in the lower layer water body. In June, the water temperature of the upper layer water reached 20 °C, but the water temperature of the lower layer water was just 8 °C, giving a temperature difference of more than 12 °C. Stable stratification is formed in this period. Site S3 is located at the deepest point of the reservoir, the water temperature at the bottom is around 9 °C once the stable stratification period had been reached, and the low-temperature region is around 10 m deep (water temperature below 12 °C). Site S2 is a comparatively shallow point, and the water temperature is around 10 °C once the stable stratification period had been reached, and the low-temperature region is only about 5 m deep.

In general, when the reservoir surface water temperature drops below 10 °C, the reservoir will mix automatically. The results in 2011 and 2012 showed that the stratification of Shibianyu Reservoir is influenced by stormwater runoff. The main reasons for this are as follows:

1. The rainfall in Shibianyu Reservoir is concentrated, and mainly in the form of large storm runoff during rainstorms diving into the water body, increasing the mixing zone.
2. The temperature changes greatly in the reservoir during rainstorms, especially in case of continuous heavy rains. Water temperature differences during different rainfalls lead to changes in the undercurrent.
3. Heavy rain erosion could easily cause landslides, leading to water turbidity, and increasing the density significantly. When the density is higher than that of the bottom, the inflow will sneak into the bottom of the reservoir, resulting in complete destruction of the water stratification.

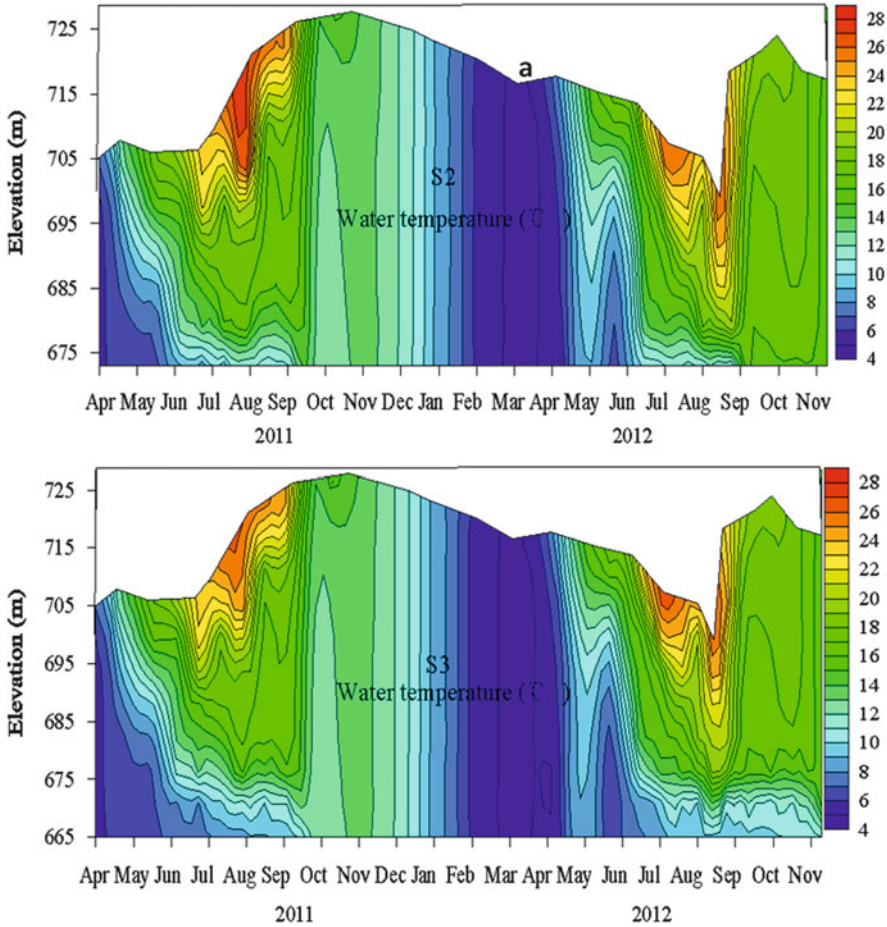


Fig. 32 Vertical distribution characteristics of water temperature in Shibianyu Reservoir (2011.3–2012.11)

The rainfall was a continuous heavy rainfall of 410 mm in September 2011, resulting in temperature decrease, and the runoff temperature changed greatly and the undercurrent position moved down, resulting in an increase of the mixed layer depth. In addition, the heavy rains leading to landslides, with high density water transferred into the reservoir, dived to the bottom of the reservoir, resulting in the mixing of the reservoir in advance. Due to the continuous rainfall, the water temperature stratification is not formed either.

In 2012, the rainfall intensity was weak compared to 2011, and the maximum rainfall occurred in August 31–September 1, with a peak flow of 117.6 m³/s. After the storm runoff, the stratification was destroyed at the bottom of monitoring point

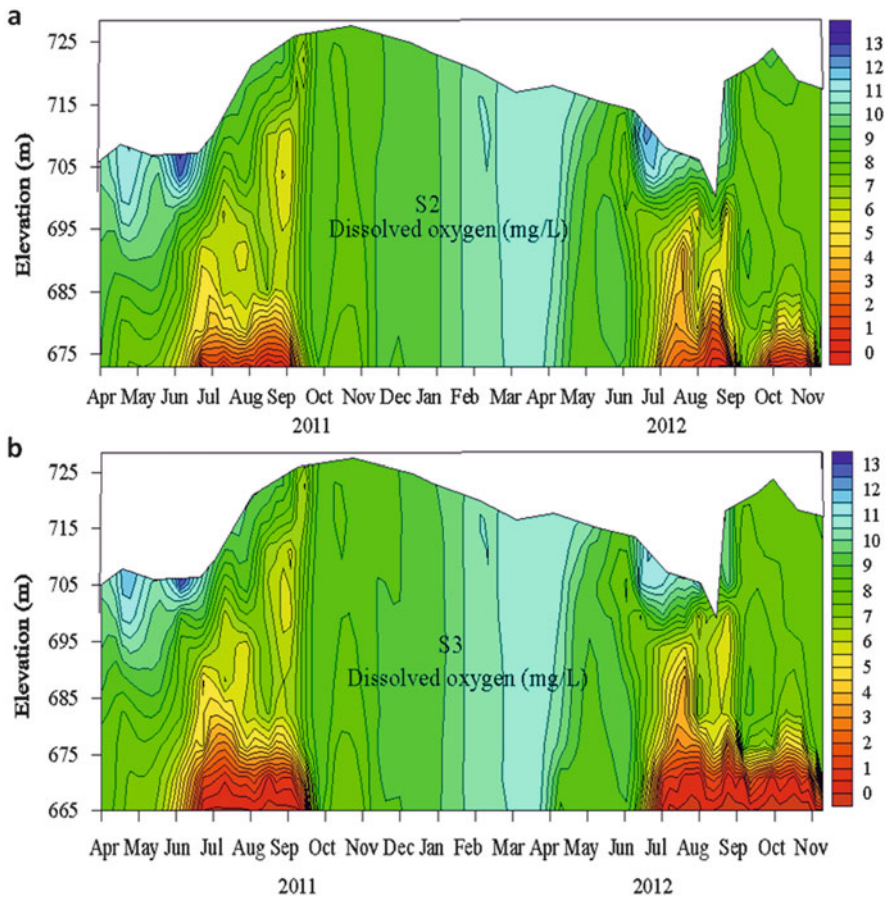


Fig. 33 Vertical distribution characteristics of dissolved oxygen in Shibianyu Reservoir (2011.3–2012.11)

S2, and the water temperature at the bottom increased from 10.44 °C to 14.22 °C, while the surface water temperature was about 20 °C. In early November, when the reservoir surface temperature dropped to about 14 °C, the water layer was fully mixed at monitoring point S2.

4.2.2 The Variation of Dissolved Oxygen Under Stratification

The seasonal variation of dissolved oxygen in Shibianyu Reservoir is shown in Fig. 33. There are significant seasonal variations of dissolved oxygen in Shibianyu Reservoir, which is affected by dynamic stratification. There is highly dissolved oxygen in the upper water of the reservoir, generally 8–10 mg/L. In addition, at the period of high algae, it would, to a certain extent, improve the dissolved oxygen in

the water due to oxygen production from algae photosynthesis. In the stratification formation period, with the increase of the temperature difference between the upper and lower layer waters, the oxygen mass transfer rate continues to decrease, resulting in the gradual decrease of dissolved oxygen. By the end of May, the water formed a stable thermal stratification, and the dissolved oxygen of the bottom water rapidly reduced from 8 mg/L to zero after 20 days, due to the consumption of sediments and lower-layer water. The anaerobic area is mainly located in a low-temperature region. There is a larger anaerobic zone at S3.

In September 2011, the stratification was destroyed in advance, as a result of storm runoff, and the dissolved oxygen increased rapidly at the bottom of the reservoir. The reservoir temperature stratification is no longer formed again due to the continuous rainfall. The DO of the entire reservoir remained around 8 mg/L.

By the end of August 2012, the stratification was totally destroyed at site S2 after a heavy rainstorm. The dissolved oxygen of the bottom water increased from 0 mg/L to 6 mg/L. Afterwards, the reservoir encountered less rainfall and higher temperature, and the reservoir water returned to a stratified state again due to the consumption of sediments and lower-layer water. In November, the reservoir water mixed again at site S2 and, correspondingly, maintained an aerobic state. At site S3, the bottom water remained in an anaerobic state after the heavy rainfall in August 2012, and a water-lifting aerator was implemented to increase the concentration of dissolved oxygen. After 1 month of running the water-lifting aerator, the dissolved oxygen at the bottom increased greatly.

4.2.3 The Variation of pH Under Stratification

The pH value of the water body, namely the hydrogen ion concentration index, is a reflection of the water quality conditions (pH). The seasonal changes in the reservoir can not only change the chemical speciation of substances in raw water, the migration and transformation process, and the metabolic activity of aquatic organisms, but also affect the urban water supply system. So, the pH value has been seen as an important indicator of water quality.

The pH of water depends on the changes in the vertical distribution of water alkalinity, photosynthesis, microbial activity, the distribution difference of soluble ion, and the dissolved oxygen in reservoir sediments.

The seasonal vertical distribution of pH in Shibianyu is shown in Fig. 34. During the mixing period, the pH of the upper- and lower-layer water is almost the same. With the formation of stratification, pH differences in the vertical distribution become more and more obvious. The pH of water bodies decreases with increasing water depth, and the pH of surface water is higher and exhibits alkaline characteristics. Especially in the period of algal blooming, the pH of surface water is close to 10, which is mainly attributed to the fact that algae photosynthesis in surface water absorbs carbon dioxide and destroys the balance of the water body ($\text{CO}_3^{2-}/\text{HCO}_3^-/\text{CO}_2$), and causes pH increase of the surface water.

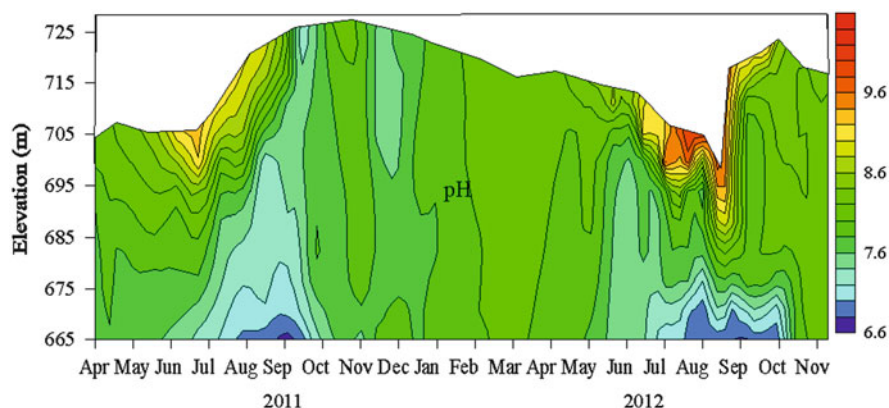


Fig. 34 Vertical distribution characteristics of pH in Shibianyu Reservoir (2011.3–2012.11)

In 2011, no algae outbreak was found in the reservoir due to the effects of rainfall; however, in 2012, algal blooming was quite serious, so the pH value of the upper water from May to October in 2012 was higher than that in 2011. The drastic decrease of the pH value within 5–15 m is mainly due to the weakened algae photosynthesis.

The pH in the bottom water is usually acidic (6.6–7.0), especially in the stable stratification period of the reservoir, which is mainly due to the acidic intermediates and formed sulfide derived from the anaerobic decomposition of organic matter and sulfate.

4.2.4 The Variation of Conductivity Under Stratification

Conductivity is a parameter for characterizing the soluble ions, which is closely related to the amount of inorganic acids, bases, salts, and other substances contained in the water. At low concentrations of these substances, the conductivity increases with increasing ion concentration. Conductivity is not only affected by the ionic content in water, but also affected by the temperature and viscosity of the water. Generally, the conductivity of natural water is between 50 and 500 $\mu\text{S}/\text{cm}$. Some of these ions are involved in redox reactions, which will consume dissolved oxygen in water. Because the stratified water bodies will directly affect the diffusion of ions in the water, it follows that stratification is closely associated with conductivity changes in reservoirs.

The conductivity of Shibianyu Reservoir varies with depth, as shown in Fig. 35. The figure shows that during spring, no difference was observed with respect to the conductivity in the vertical distribution. From May to September, with the gradual formation of stratification, the difference in conductivity in the vertical distribution increases, showing first decreases and then increases over water depths. There is a higher conductivity at the bottom of the reservoir. In summer, the conductivity is

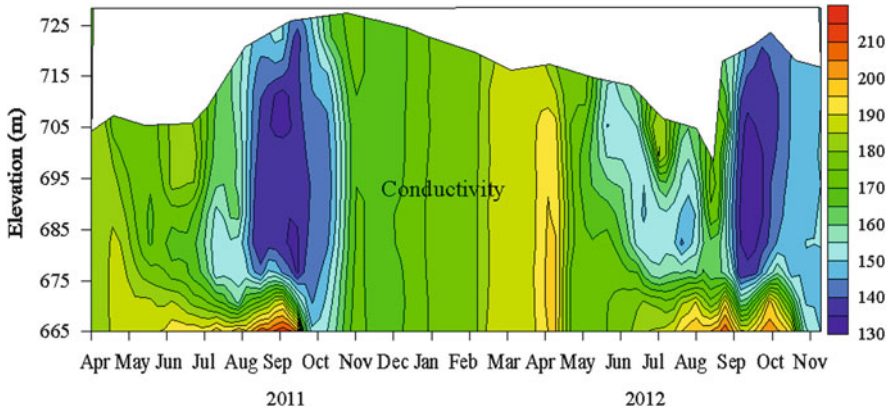


Fig. 35 Vertical distribution characteristics of conductivity in Shibianyu Reservoir (2011.3–2012.11)

relatively lower during periods of rain, but the conductivity is higher during periods without rain due to the strong water evaporation.

The transfer of dissolved oxygen between the upper- and lower-layer water is hindered after the formation of stratified water in the reservoir. The dissolved oxygen of the bottom water continues to decrease due to the oxygen consumption of sediments and bottom water. At the end of June, the dissolved oxygen at the bottom of the reservoir dropped to 0 mg/L. Under anaerobic conditions, the release of pollutants in sediments resulted in the increase of ion concentration in the lower-layer water, and, therefore, the conductivity of the lower-layer water also increases.

Conductivity has a minimum value within a certain water depth range, which is mainly due to the undercurrent from the upstream runoff (low conductivity) during stratification, resulting in a lower conductivity compared to the upper- and lower-layer water. Therefore, the vertical distribution of conductivity in the reservoir is closely related to water stratification and upstream undercurrent.

4.2.5 The Water Quality Problems Derived from the Thermal Stratification of Shibianyu Reservoir

During stratification, the mass transfer of dissolved oxygen between upper- and lower-layer water is fully hindered. The DO of the bottom water maintains an anaerobic state during stratification.

With the formation of the anaerobic environment at the bottom of the reservoir, the release of pollutants in sediments began, and the concentrations of nitrogen, phosphorus, and other nutrients began to increase in the lower-layer water of the reservoir (as shown in Fig. 36). In early August, the total phosphorus content was 0.12 mg/L, and the concentration of ammonia increased up to 0.7 mg/L at the bottom of the reservoir. At the same time, the concentration of iron and manganese

Fig. 36 Release curve of ammonia and phosphorus at the bottom of Shibianyu Reservoir during stratification

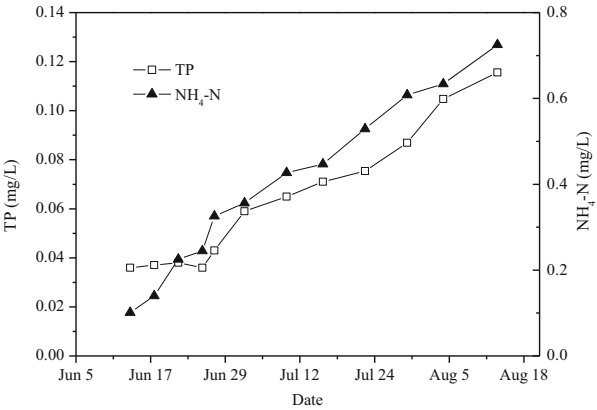
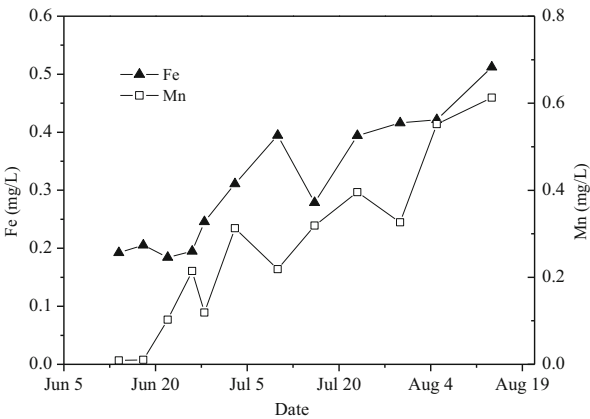


Fig. 37 Release curve of iron and manganese at the bottom of Shibianyu Reservoir during stratification



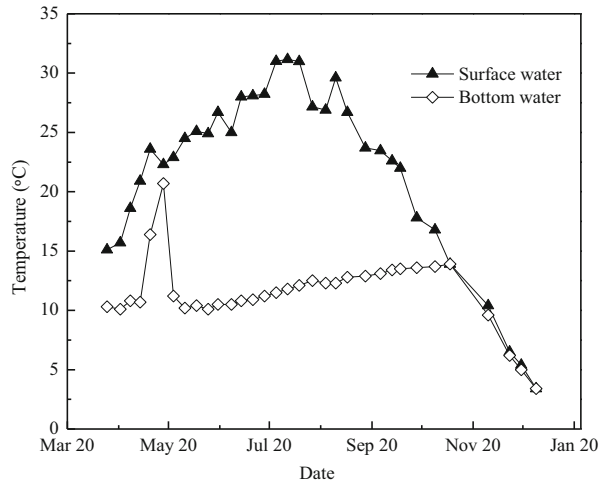
in the bottom water began to increase, until mid-August, when the concentration of iron and manganese exceeded the national surface water quality standard by 1–2 times (as shown in Fig. 37). With the extension of the anaerobic environment in the bottom water, the release of contaminants in the sediment was enhanced further.

4.3 The Stratified Characteristics of Zhoucun Reservoir and Its Impact on Water Quality

4.3.1 The Thermal Stratification Characteristics of Zhoucun Reservoir

The temperature variations of Zhoucun Reservoir from 2012 to 2013 are shown in Fig. 38. The surface temperature of Zhoucun Reservoir changes clearly with season, where the maximum is in the summer in July and August (up to 31 °C),

Fig. 38 Water temperature changes at the surface/ bottom ofZhoucun Reservoir (2012–2013)



and the lowest temperature appears in January (4 °C). The temperature change in the bottom water is not obvious, which stayed around 10 °C from April to November, slightly increased up to about 13 °C in summer, and reached a minimum of 4 °C in January.

Padisak has proposed a method to determine the stratification of a reservoir (RWCS), as follows:

$$RWCS = (D_h - D_s)/(D_4 - D_5).$$

- D_s —surface temperature of the water;
- D_h —bottom temperature of the water;
- D_4 —water density at 4 °C;
- D_5 —water density at 5 °C.

The stratification standard is as follows: at $RWCS > 20$, it is the stratification period; at $RWCS \leq 20$, it is the mixed period. The change of RWCS value with seasons is shown in Fig. 39. From April to October, the RWCS value is higher than 20, so it is the stratification period. From November to March of the following year, the RWCS value is lower than 20, so it is the mixed period. The Zhoucun Reservoir exhibited the same stratification mode year on year. Based on the results in the most recent 2 years, the stratification period can be classified into three periods, namely the stratification formation period, the stratification stable period, and the stratification weakness period. In April and May, it is the stratification period; from June to September, it is the stratification stable period; in October and November, it is the stratification weakness period.

The vertical distribution of physical and chemical parameters varied in different periods. The temperature change in different periods is shown in Fig. 40. In the stratification formation period, the ranges of the epilimnion, thermocline, and hypolimnion are 0–7 m, 7–10 m, and 10 m to the bottom, respectively. In the

Fig. 39 Seasonal variations of RWCS in Zhoucun Reservoir

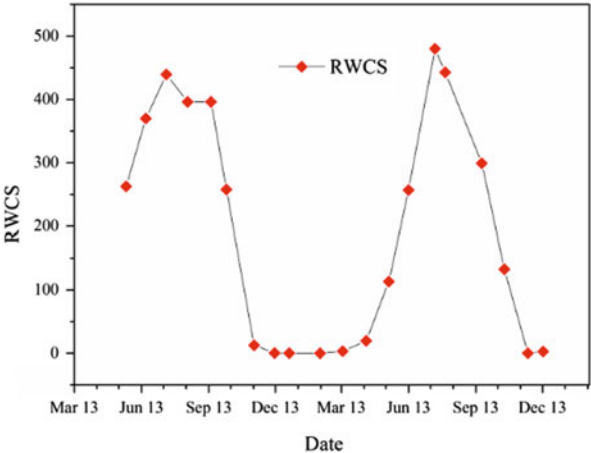
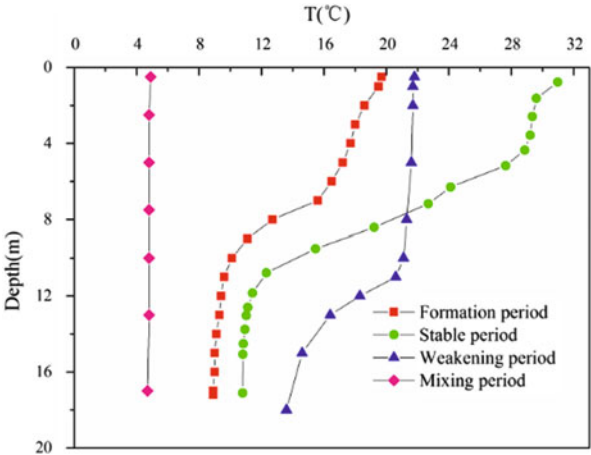


Fig. 40 Vertical distribution of water temperature at different periods in Zhoucun Reservoir

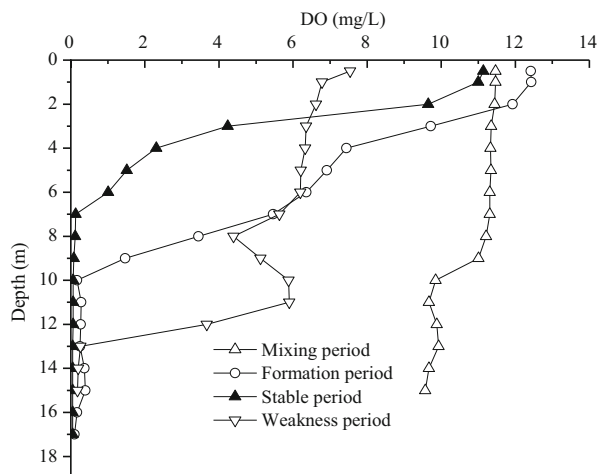


stratification stable period, the ranges of the epilimnion, thermocline and hypolimnion are 0–5 m, 5–10 m, and 10 m to the bottom, respectively. In the stratification weakness period, the upper water was mixed and the temperature was always the same, and the thermocline moved down gradually until it was totally mixed, namely the mixed stratification period.

4.3.2 The Variation of Dissolved Oxygen Under Different Periods

The DO change in different periods is shown in Fig. 41. Because many factors affect the DO concentration of a water body, especially algae photosynthesis, in order to better obtain representative results, the dissolved oxygen concentration

Fig. 41 Vertical distribution of dissolved oxygen at different periods in Zhoucun Reservoir



(Fig. 41) was measured on a sunny morning at 10:00 AM in May, August, October, and December. As shown in Fig. 41, the vertical distribution of DO in the stratification formation period and the stratification stable period is quite similar. In the surface water, the DO concentration was as much as 12 mg/L due to the photosynthesis of algae. However, the DO concentration rapidly decreased to zero between 2 and 6 m, since the photosynthesis weakened with water depth. Comparatively, the difference in DO change between the stratification formation period and the stratification stable period is that the decrease rate of DO in the stratification stable period is more rapid in the thermocline, which is due to the algal blooms in this period, resulting in reduced water clarity. In the stratification weakness period, parts of the water body are already mixed, and the DO concentration of a mixed water body is higher (6 mg/L), but the water body without mixing below the thermocline water remains in anaerobic conditions (DO is 0 mg/L). In the mixed period, the DO concentrations of the surface and bottom waters showed little difference.

4.3.3 The Variation of pH Under Different Periods

The pH of Zhoucun Reservoir water is in the range of 7.2–9.0, and shows a significant seasonal change with the stratification period (as shown in Fig. 42). As shown in Fig. 42, the water is alkaline, which is due to the strong light intensity, inducing strong algae photosynthesis and consumption of the carbon dioxide in the water. The vertical distribution of pH is quite different at different stratification periods, which is quite similar to the temperature behavior. The pH in the bottom water decreased gradually from the stratification formation period to the stratification weakness period, which is mainly due to the fact that the reservoir is always in anaerobic conditions during the stratification period, and the nitrogen, phosphorus, organic matters, sulfide, iron, and manganese were released into the water body and induced pH decrease.

Fig. 42 Vertical distribution of pH at different periods in Zhoucun Reservoir

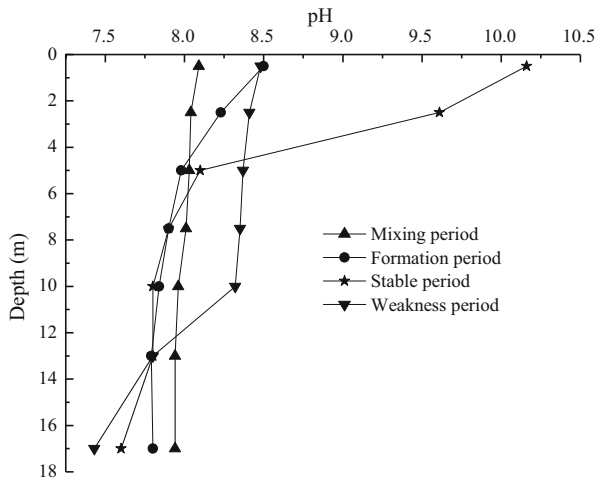
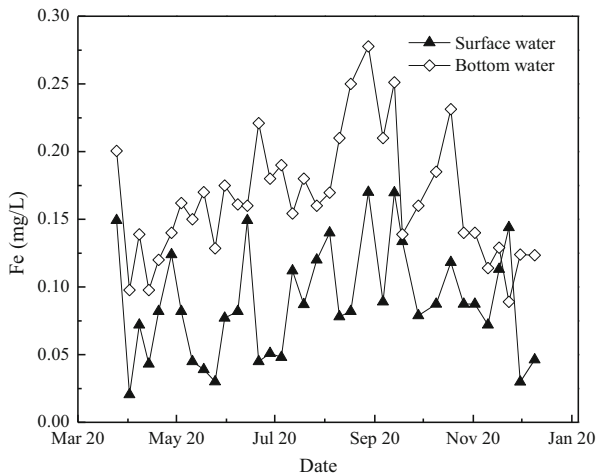


Fig. 43 Change of iron concentration at the surface/ bottom waters at different periods in Zhoucun Reservoir

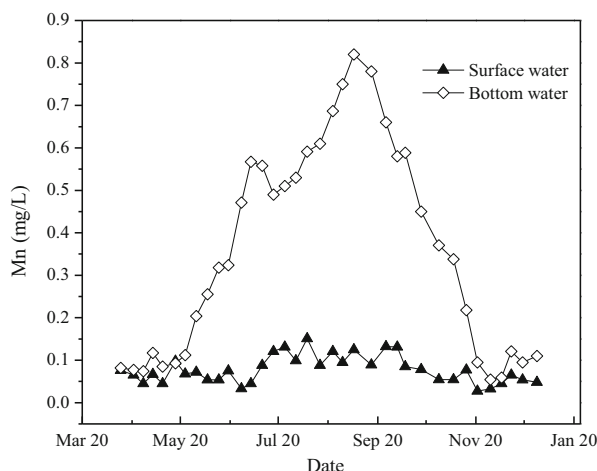


4.3.4 The Water Quality Problems Derived from the Thermal Stratification

(1) The Seasonal Change of Iron in Zhoucun Reservoir

The variation of iron concentration in Zhoucun Reservoir with time is shown in Fig. 43. It clearly shows that the concentration of iron is in the range 0.02–0.35 mg/L. There is a slight difference in iron concentration between the surface and bottom water bodies, and the iron concentration in the bottom water is slightly higher than that of surface water, which is due to the release of iron from the sediments under anaerobic conditions. Overall, the iron concentration in Zhoucun Reservoir is not very high, which is lower than the national surface water quality standard (GB

Fig. 44 Change of manganese concentration at the surface/bottom waters at different periods in Zhoucun Reservoir



3838–2002) (class III, the corresponding iron concentration is 0.3 mg/L), conforming to the class III source water standard.

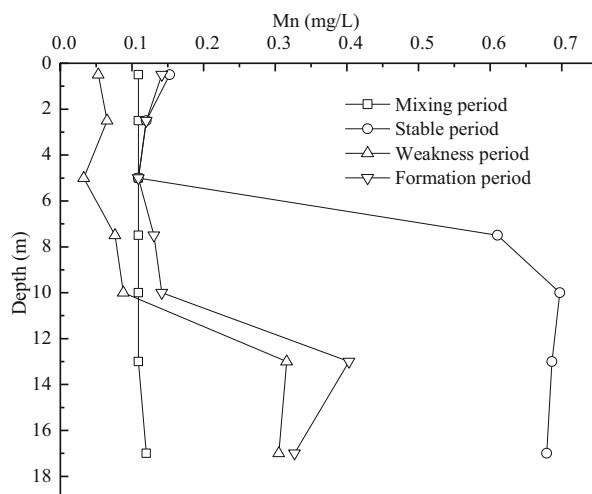
(2) The Seasonal Change of Manganese in Zhoucun Reservoir

The variation of manganese concentration in Zhoucun Reservoir with time is shown in Fig. 44. The figure clearly shows that the concentration of manganese is in the range 0–0.8 mg/L. In September and October each year, the manganese pollution is quite serious and reaches a maximum, about 0.7–0.8 mg/L, which is about 7–8 times higher than the national surface water quality standard (GB 3838–2002) (class III). Comparatively, the manganese concentration in Zhoucun Reservoir is a little lower from January to May, approximately 0.1 mg/L. From the beginning of June, the massive release of manganese from sediments starts, and, at the end of August, the manganese concentration in the bottom water reaches a maximum. There is a high risk of manganese excess in the mixed period due to the massive release of manganese from the sediments. For example, once the water was just completely mixed, the manganese concentration reached 0.18 mg/L in both surface and bottom water bodies in November 11, 2012, exceeding 1.8 times the national surface water quality standard (GB 3838–2002) (class III). In November 4, 2013, the manganese concentration of the bottom water reached 0.21 mg/L, exceeding 2.1 times the national surface water quality standard (GB 3838–2002) (class III). After complete mixing, the manganese concentration in water gradually decreased and reached 0.1 mg/L in December.

(3) The Vertical Change of Manganese in Zhoucun Reservoir

The vertical change of manganese concentration in Zhoucun Reservoir in different periods is shown in Fig. 45. In the mixed period and stratification formation period, there is no obvious vertical change of manganese concentration. However, in the stratification stable period and stratification weakness period, the vertical distribution of manganese is apparently different. In the water layer below the thermocline,

Fig. 45 Vertical distribution of manganese concentration (2012–2013)



the manganese concentration is significantly higher than that of the water layer above this layer, which is attributed to the higher manganese release rate compared with the diffusion rate. The manganese concentration below the thermocline was 0.63 and 0.72 mg/L in 2012 and 2013, respectively. In the stratification weakness period, the manganese concentration in the upper mixed water decreased due to the oxidation and dilution effect. The manganese concentration in the bottom water increased slightly due to the manganese release from the sediments and the decrease of unmixed water volume.

4.4 The Stratified Characteristics of Zhelin Reservoir and Its Impact on Water Quality

4.4.1 The Thermal Stratification Characteristics of Zhelin Reservoir

The vertical temperature change of Zhelin Reservoir is shown in Fig. 46a. The water depth of the monitoring position is around 42 m. As seen in Fig. 46a, the thermal stratification started from the April. At this time, the temperature difference between the surface layer water and the bottom layer water was around 8 °C. Until summer, the stable stratification was fully formed and the temperature difference between the surface layer and the bottom layer was 22 °C. In the fall, the surface water would get cold continuously, and the vertical temperature difference was more and more homogenous. Until early January in the following year, the water body was totally mixed and the vertical distribution of the water quality parameters was the same.

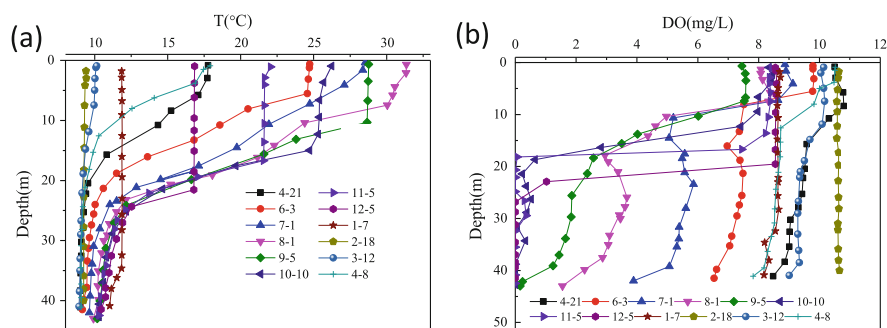


Fig. 46 Vertical distribution of water temperature and dissolved oxygen in Zhelin Reservoir (2012–2013)

The vertical DO change of Zhelin Reservoir is shown in Fig. 46b. During the thermal stratification period (from April to December), the DO concentration decreased with the increase of water depth. In the surface layer water, the DO concentration is always quite high (around 8 mg/L), due to the storm effect. In the middle layer water, the DO concentration gradually decreased, due to the lack of photosynthesis and the storm effect. In the bottom water, the DO concentration continuously decreased to a minimum value, due to the consumption of sediments and organic matters in water bodies. The DO concentration was around 2 mg/L in August. Until early January of the following year, the water mixing started and the vertical DO concentration in water tended to be consistent and reached the saturation level (>8 mg/L) under the local temperature conditions.

4.4.2 The Water Quality Problems Derived from the Thermal Stratification

(1) The Seasonal Change of TN and TP in Zhelin Reservoir

The seasonal change of TN and TP is shown in Fig. 47. The maximum concentration of annual TN and TP was 1.30 and 0.14 mg/L, respectively. The maximum concentration of TN occurred in the flood season in June, which was attributed to the inflow of surface runoff, carrying an amount of pollutants. Although the concentration of TP increased slightly during the flood season, the maximum concentration of TP occurred in the stable thermal stratification period, but not in the flood season. The main reason for TP increase is the release of manganese from the sediments under anaerobic conditions. The TP concentration reached a maximum before the mixing period. In January of the following year, the reservoir was totally mixed, and the TN and TP concentrations reduced to minimums of 0.4 and 0.01 mg/L, respectively.

During the stratification period, the bottom water was in an anaerobic state, and the redox potential also decreased. Therefore, nitrogen and phosphorus were

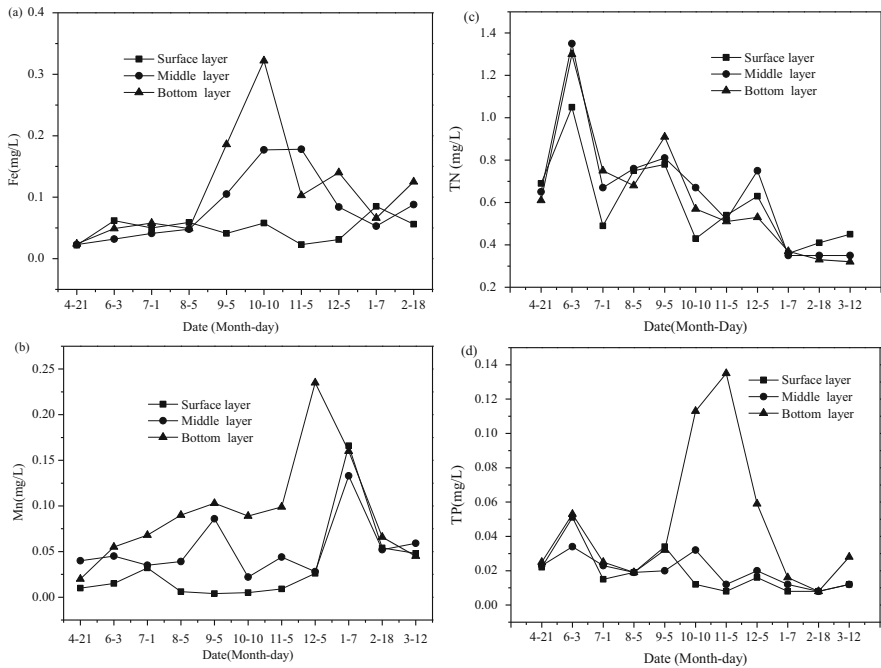


Fig. 47 Dynamics of nitrogen, phosphorus, iron, and manganese in Zhelin Reservoir

released into the overlying water. Comparatively, the release of TN is not as significant, since the water is slightly polluted in the reservoir, which was confirmed by the laboratory simulation experiments, and the maximum release concentration of TN was just 1.50 mg/L. In regards to TP, the main reason for the massive increase of TP in the reservoir is the release from the sediments under anaerobic conditions.

(2) The Seasonal Change of Iron and Manganese in Zhelin Reservoir

Fe and Mn are quite active elements, and their existence is easily affected by the environment in the sediment–water interface. The seasonal change of iron and manganese is shown in Fig. 47. The maximum concentrations of Fe and Mn were 0.32 and 0.34 mg/L, respectively, which occurred at the bottom of the water body under the stable stratification period. The minimum concentrations of Fe and Mn were 0.1 and 0.16 mg/L, respectively, when reaching the mixing period.

Fe and Mn were also at low levels before August (the Fe and Mn concentrations were lower than 0.1 mg/L), when the water was in an oxygen-rich state. After August, the bottom water body was in an anaerobic state, and the concentration of Fe and Mn increased continuously and exceeded the surface water quality standard. In January of the following year, the reservoir was fully mixed, and the concentration of Fe was at a low level, but the concentration of Mn was in an excessive state.

5 The Sulfide Pollution in the Reservoir

Sulfides include water-soluble hydrogen sulfide, acid-soluble metal sulfide, as well as insoluble sulfide and organic sulfide [37]. Hydrogen sulfide has a strong smell of rotten eggs, which can cause unpleasantness when the water contains a few milligrams per liter of hydrogen sulfide. Hydrogen sulfide is toxic, which can harm cytochrome oxidase, cause tissue hypoxia, and even threaten life. In addition, hydrogen sulfide is easily oxidized by bacteria to form sulfuric acid, which corrodes metal equipments and pipes. The latest results showed that acid-volatile sulfide plays a major role in the chemical activity of heavy metals in sediment and their bioavailability [38]. Therefore, sulfide is an important indicator of water pollution. In the following, with Zhoucun Reservoir as an example, the pollution characteristics and source analysis of sulfide are illustrated.

5.1 The Seasonal Variation of Sulfide in Zhoucun Reservoir

Figure 48 shows the seasonal variation of sulfide in Zhoucun Reservoir in 2012 and 2013. There is a similar variation in the trend of sulfide in the hypolimnion of Zhoucun Reservoir in 2012 and 2013. In the mixing period and the stratification formation period, there is no sulfide in the water. From late July, a rapid release of sulfide occurred in the sediments, and reached a peak of about 2.5 mg/L before the mixing period. In May, the DO of the bottom water in Zhoucun Reservoir dropped to zero, which provided the essential condition for sulfide release; but, at this time, the water pH was 8, and the redox potential was 180 mV, with organic matter in the hypolimnion being quite less, which inhibited the release of sulfide from the sediments. Until late July, a large number of sulfides were released from the sediments due to the increased organic matters in sediments, the decrease of redox potential (-200 mV), and the formation of a reductive environment.

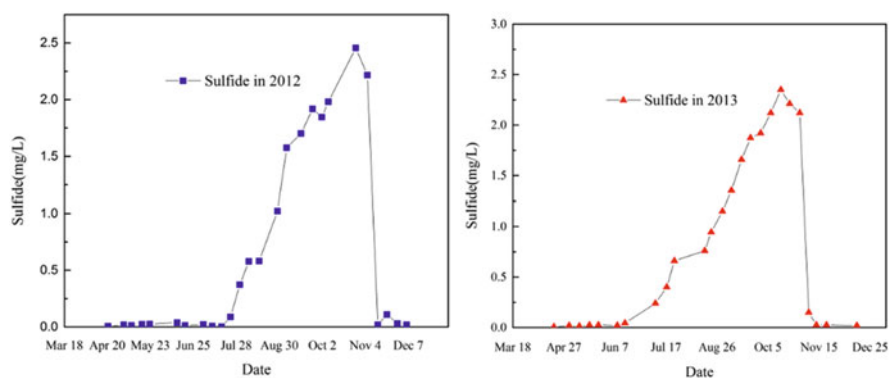


Fig. 48 Dynamics of sulfide in Zhoucun Reservoir

Table 26 The content of sulfur in the sediments of Zhoucun Reservoir

Sample	Unit	1	2	3	4	5
Bottom sediment in Zhoucun Reservoir	mg/L	9.378	7.424	12.637	12.427	12.599
Bottom sediment in Shibianyu Reservoir	mg/L	0.9631				
Dry mud content in Zhoucun Reservoir	mg	0.8853	0.7006	1.1927	1.173	1.1894

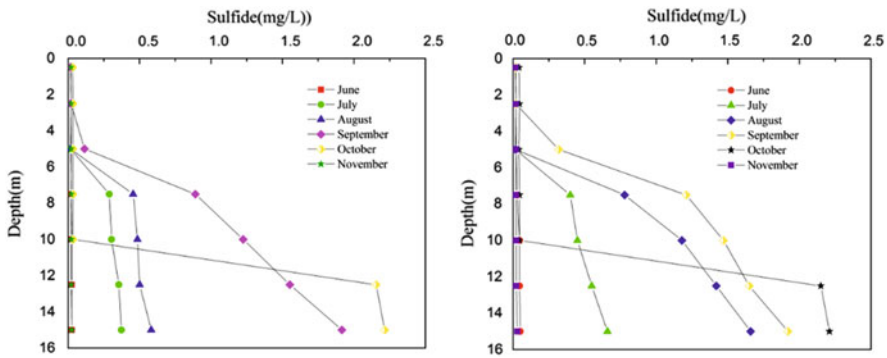


Fig. 49 Vertical distribution of sulfide in Zhoucun Reservoir

As shown in Table 26, there is a higher amount of sulfide (ten times higher) in the sediments of Zhoucun reservoir compared with that in Shibianyu Reservoir. That is the reason why the sulfide is excessive in Zhoucun Reservoir.

In the reservoir, because most of the sediments were in a reduced state, the sulfate will be reduced to sulfide by the sulfate-reducing bacteria. In the sediment of reservoirs, there are a lot of anaerobic microorganisms, including sulfate-reducing bacteria. Sulfate-reducing bacteria can reduce sulfate and generate sulfide in the decomposition of organic matter. In the case of plenty of sulfate in the sediments, the sulfide production rate is often directly proportional to the concentration of organic matter in the sediment. The results showed that the production of sulfide occurred in the surface sediments, which contained a lot of organic matter. In addition, it was found that the higher the content of organic matter in the sediments, the greater the sulfide production rate.

5.2 The Vertical Distribution of Sulfide in Zhoucun Reservoir

The vertical distribution of sulfide in Zhoucun Reservoir in 2012 and 2013 is shown in Fig. 49. As can be seen, the distribution of sulfide in water bodies showed an obvious stratification. The sulfide concentration in the epilimnion is almost zero,

due to the aerobic state. The release of sulfide began from July. During the early period, the release rate is slower than the diffusion rate, resulting in the same sulfide concentration at different water depths in the hypolimnion.

Until September, the sulfide release rate increased, and the sulfide concentration rose with the increase of water depth. The maximum sulfide concentration reached 1.8 mg/L at the bottom of the reservoir. In October, in the stratification weakness period, mixing occurred in the upper layer water and dissolved oxygen increased, while sulfide was removed by oxidation. However, it still maintained a high sulfide concentration in the bottom water, since the water environment was still in a reducing state.

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