

Chapter 3

Dianchi Shallow Lake Management



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3.1 Introduction

3.1.1 Conceptual Thoughts Connecting Water Management and Shallow Lake Management

The SINOWATER-Project funded by Federal Ministry of Education and Research of Germany (BMBF) examined and discussed the idea of an Integrated Water Resources Management on all levels of water management. As shown in Fig. 3.1 this covers a range from the overall management principles of Good Water Governance (Grambow [7], Disse and Chen [3]) over conceptual levels to technical solutions like stormwater management (Wang et al. [35]). This chapter is dedicated to the conceptual level of modern water management as shown in the following figure. Measurement planning on this conceptual level benefits from strong sustainable management principles (Grambow [7]). Within the Kunming part of the SINOWATER-Project, we also committed a major part of our work towards technical solutions, side by side with our colleagues from Kunming authorities. Included in the book you will find a thorough

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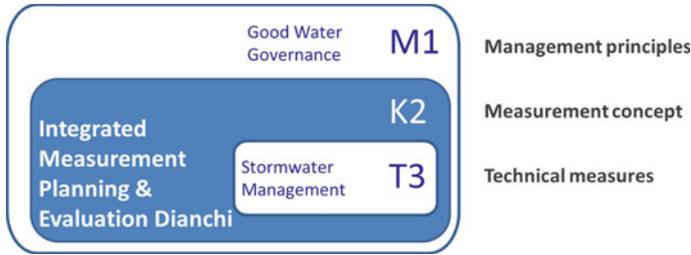


Fig. 3.1 Levels of water management implemented as sub-projects M1, K2 and T3 in BMBF-funded SINOWATER-project

description of measures of storm water management planned and implemented in the tributaries of the Caohai and its urban river basin (Wang et al. [35]).

In this chapter, we would like to introduce our readers to the outcomes of our studies along the northern part of Dianchi the Caohai at Kunming, the capital of Yunnan Province. Dianchi is the third biggest Chinese shallow lake. This lake is under constant pressure of the growth of the city.

3.1.2 Two Studies for One Goal

The following chapters describe some of the SINOWATER-Project contributions to the shallow lake management strategies of the Kunming Dianchi Management Authorities. These recommendations are based on two different studies. The first study served to collect data and to broaden the database by taking and analysing additional water and sediment samples. The second study comprised the shallow lake modelling and discussed some of the six above mentioned management scenarios on a scientific basis. The analysis of the studies are following the overall idea of the combined approach of EU-Water Framework Directive. It is the combination of basic technical emission control and strict enforcement of environmental quality standards (EQS). In consequence, the EQS enforcement is a quality management step aiming at checking if technical emission control is sufficient to sustain the targeted water quality. If necessary because basic technical control measures are insufficient, the evaluation of the EQS-Check will lead to an extension of quality control measures. This water management approach has been enforced in the EU with the introduction of EU-Water Framework Directive in the year 2000.

In recent years, numbers of Kunming Academy of Environmental Sciences and other Dianchi Management Authorities show a decrease in lake pollution due to the strong commitment of environmental protection. We wish our partners and colleagues strong perseverance and the best support and full confidence from the political leadership for a permanent journey together.

Worldwide a lot of shallow lakes suffer from bad water quality caused by eutrophication (Küppers et al. [21]; Qinghui et al. [27]; Diotallevi et al. [6]; Lei et al. [19]; Hargeby et al. [10]; Vadeboncoeur et al. [33]). In China, bad water quality and regular algae blooms in shallow lakes such as Taihu, Chaohu and Dianchi are a huge environmental problem, caused by high nutrient intakes due to growing population, industrial growth and high fertilizer inputs from agricultural sides (Nixdorf & Zhou [25]; Zhang et al. [50]; Huang et al. [13]; Cao et al. [2]; Li et al. [46]; Qin et al. [26]; Zhu et al. [52]). All three lakes are major national water resources of China. They are serving not only as habitat for flora and fauna but also perform tasks in areas such as drinking water supply, local recreation and tourism. For shallow lakes once the low carrying capacity against pollution is exceeded the negative effects such as eutrophication with extreme algae bloom, fish mortality, water toxicity and others can cause severe effects for the regional water management and put monetary pressure on regional or even provincial economy.

Lake Dianchi in the Chinese province Yunnan is a shallow lake suffering from algae blooms for years due to high nutrient intakes (Zhang et al. [51]; Wu et al. [40]; Wang et al. [38]; Jin [16]). Its capital Kunming, which is next to the lake, has grown from 3.7 million inhabitants in 1978 to 6.8 million in 2017 (Kunming Statistics Bureau [18]). Meanwhile, the bad water quality causes a negative feedback on the development of the city of Kunming (Yang [45]). In recent years, local authorities built several wastewater treatment plants and conducted projects to improve the water quality of Lake Dianchi. Nevertheless, there are still algae outbreaks, especially in the summer monsoon season.

The most polluted part of is Lake Dianchi its northern part, Caohai (Zhan et al. [48]). Separated from the southern part Waihai, it receives huge amounts of wastewater from the city of Kunming. In the past, Lake Caohai was scarcely sampled, ignoring the fact that the Caohai-Waihai Lock separates Lake Caohai and Lake Waihai since 1996 (Yan et al. [43]). For that reason, Lake Caohai had to be investigated as a water system of its own, which we did in this study that was part of the subproject Lake Management in the SINOWATER project.

In this study the aim was to sample Lake Caohai over the summer monsoon period in a holistic way like proposed from (Yua et al. [47]). Thus, we took water samples from all seven tributaries and from the Lake itself at three dates and analysed them on nutrients providing algae growth. As it is known that nutrient release from sediments can have a huge impact on the water quality (Moore et al. [23]), we took also sediment samples from Lake Caohai. With these, we performed nutrient release tests. Some of the generated results were the basis for a numerical simulation with the stoichiometric lake model StoLaM (see Chap. 3.3).

3.2 Study 1: Water and Sediment Monitoring Data Acquisition

3.2.1 Research Area

Sampling was carried out on the northern part of the Dianchi. This section, called Caohai or Inner Lake, is located directly at the city of Kunming. It is 8.15 km² in size, which corresponds to 2.7% of the total area of the Dianchi and accounts for about 1% of the total volume (Jin et al. [16]). It was separated from the larger Waihai in 1996 (He et al. [12]).

In recent years, algae blooms have occurred regularly in Caohai, with cyanobacteria playing a particularly important role (Han et al. [9]; Yua et al. [47]). The Caohai receives 45% of the total amount of wastewater flowing into the Dianchi and shows significantly increased nitrogen and phosphorus concentrations (Wang et al. [39]; Zhan et al. 2016). The Caohai is divided into two larger parts by the Dongfengba Dam. This division was carried out in order to retain the heavily polluted water from the Wangjiadui, Xinyunliang and Laoyunliang rivers in the western part and to purify it mechanically by sedimentation. It is fed into the artificial lake outlet, the western tunnel, thus separating the eastern part from this source of pollution (Yan et al. [43]). The rivers Wulong, Dagan, Xiba and Chuanfang flow into the eastern part. Furthermore, in the southern part of the Caohai there is another separated area that is used for research on aquatic plants. The western tunnel forms the Caohai's outflow and diverts the water at Anning into the Tanglang River, which belongs to the Yangtze River catchment area (Jin et al. [16]).

The Wangjiadui is an outflow from an electric power station, which is no longer fed and has only a low water flow due to rainwater. It is part of several smaller inflows at the West Bank of the Caohai (He et al. [12]).

The Xinyunliang River has its source in the mountains north of Kunming City and flows for about 21 km, first through the more rural and then through the more densely populated areas of Kunming City. It is fed by 15 smaller rivers (Nie et al. [24]). He et al. [12] reported that around 40% of the discharge are fed by WWTP. The Xinyunliang River is described as the most polluted tributary to the Dianchi and its main source of pollution (Wang et al. [36]; Huang et al. [15]). The Xinyunliang contains 41.28% of the industrial wastewater of the entire Dianchi catchment area (Nie et al. [24]). Since the construction of a wastewater treatment plant along the river and individual projects to improve the water quality of the river, the river's fauna has recovered considerably (Huang et al. [15]).

The upper reaches of the Laoyunliang River are fed by the outflow of the Fangjiayiu Reservoir, which in turn is fed by rainwater from the neighbouring mountains. In the Xiaopuji district, waste water from a copper factory and municipal sewage are added, which is about 95% of the whole discharge to the Caohai (He et al. [12]). Over a length of 11 km, it flows through the Kunming urban area and is polluted with sewage from settlements along the entire flow path. Its catchment area is home

to around 250,000 inhabitants (Li et al. [20]). It flows downstream of the municipal wastewater treatment plant No. 3 into the Caohai.

The 3.7 km long Wulong River used to be fed by the wastewater of the Yunda Hospital. According to (Yang [45]), however, the source has been relocated to the Baima District and is only polluted with waste water from the hospital during rainy weather. (Yang [45]) was able to show that the water quality has improved considerably since 2009, and most recently quality level IV according to the Chinese classification system was achieved. It flows into the Dianchi near the Lijia Park.

According to Ding et al. [5] the Dagan River is an artificial river that flows together from various sources in Kunming city centre and is called the Dagan from the Guan Building onwards. He et al. [12] showed that nearly 100% of the water in the Dagan comes from the Niulanjiang River replenishment. The Dagan River is about 3 km long and flows into the Caohai in Dagan Park. The park originated from a wetland area that was in a very poor condition in the past and was restored as part of an ecological upgrading project (Ma et al. [22]).

The Xiba River is a side channel of the Niulanjiang River, which is known for its heavy pollution and brings high nutrient loads from the mountains in the east of the city, which are mainly agricultural. The Niulan River Diversion Project alone has introduced 400 million m³ of water into Dianchi in 2015.

In the Chuanfang River the WWTP 1 of Kunming accounts for about 95% of the discharge (He et al. [12]). Its water quality is therefore directly dependent on the purification capacity of this WWTP.

Obviously, the tributaries of the Caohai are anthropogenically dominated. Three rivers are mainly dominated by WWTP effluents, two rivers receive huge amounts of water from the Niulanjiang River replenishment (He et al. [12]). Only the Xinyunliang River has a more natural dynamic.

3.2.2 *Materials and Methods*

3.2.2.1 *Sampling*

3.2.2.1.1 *Water Sampling*

Water samples were taken both at the seven tributaries of the Caohai and its outflow, and on the lake. The timing of sampling was based on the hydrographs of rainfall in the city of Kunming (see Fig. 3.3). The water bodies were sampled at the peak of the rainy season and in the two transitional phases before and after the rainy season.

At the tributaries, the sampling points were selected so that they were as close as possible to the mouth of the Caohai in order to record all pressures on the lake. On the other hand, an influence of wind-induced backwash of water from the lake into the tributaries had to be avoided. The third constraint for the selection of the sampling points was the accessibility of the tributaries. Accordingly, the sampling points marked R in Fig. 3.2 were selected. These restrictions also applied to the outflow of the Caohai.

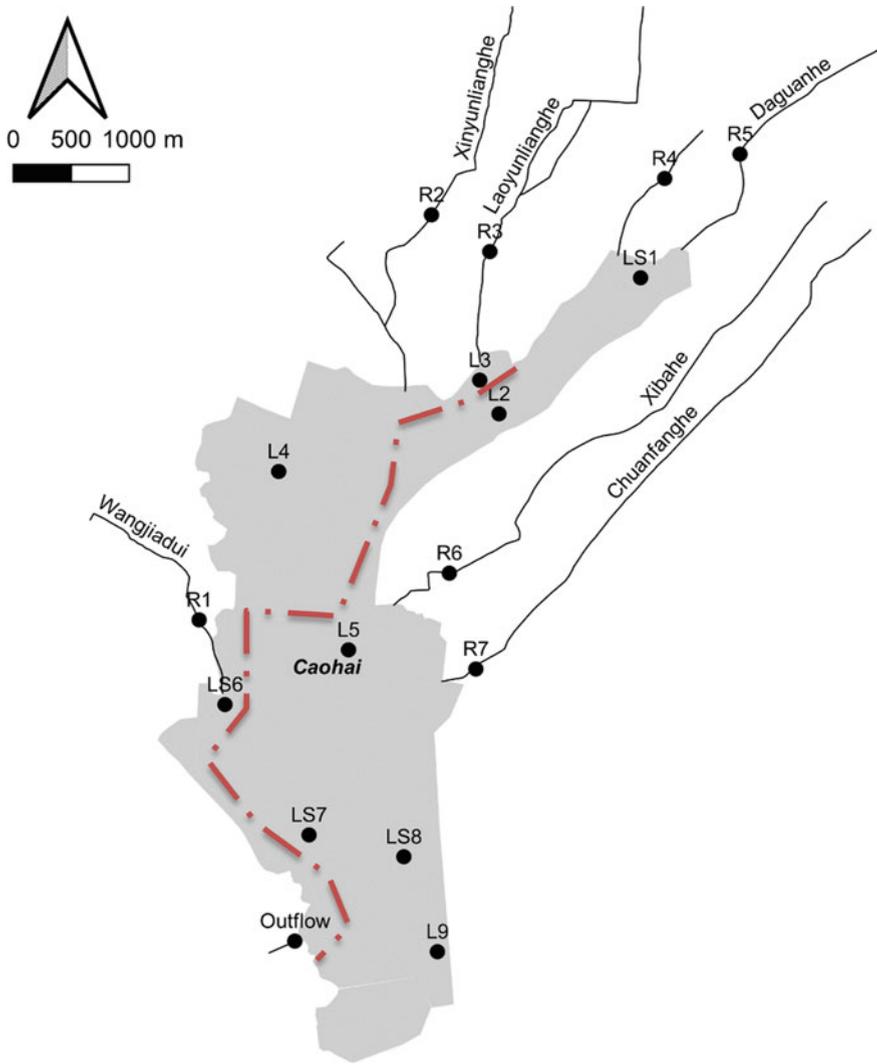


Fig. 3.2 Western and eastern part of Caohai divided by a dam and its tributaries

For the sampling on the Caohai, the premise was to ensure an even distribution of the samples. The scheme was deviated from the points where the planned dredgings took place, because sediment samples were also taken there. Furthermore, the aforementioned division of the Caohai had to be considered. The distribution of the sampling points can be seen in Fig. 3.2.

The water samples were taken in a depth-integrated manner with an immersion bottle, between the surface and the bottom of the water, as described in DIN 38402 (1985). The samples were stored in PE bottles. Together with the sampling, the

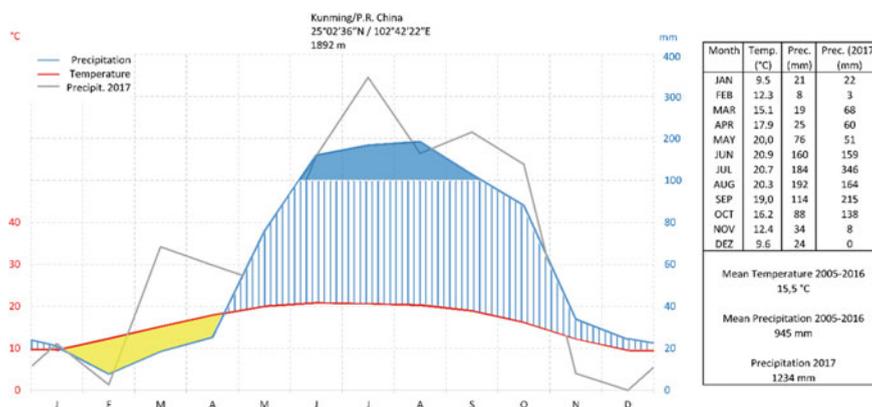


Fig. 3.3 Kunming hydrograph

accompanying parameters oxygen (optical), pH value, conductivity, redox potential (all three electrochemical) were analysed by using probes. These parameters serve to interpret and check the plausibility of the measurement results.

In the laboratory of the Kunming Institute of Environmental Science (KMIES) the samples were prepared for further analysis. For this purpose each sample was divided into two subsamples. Subsample 1 was used unfiltered for the analysis of the parameters total nitrogen and total phosphorus. Subsample 2 was filtrated through a 0.45 μm filter and served as analysis sample for the parameters ammonium, nitrite, nitrate, orthophosphate and total dissolved phosphorus.

3.2.2.1.2 Sediment Sampling

In order to assess the nutrient release of the sediments of the Caohai, their sampling was carried out in parallel with the water sampling described in chapter Water Sampling. The sediment sampling took place in May and July 2017. Sediment samples were taken at four different locations. The local authorities planned dredging at various locations in the Caohai at the time of sampling. The sampling sites were selected according to these planned dredging operations (see Table 3.1, Fig. 3.2).

The sampling was carried out from a boat parallel to the sampling of the water. A Van-Veen bottom grab was used for this purpose. The samples were individually packed in plastic bags and analysed in the laboratory of the Agro-Environmental Protection Institute in Dali.

Table 3.1 Sediment dredging sites at Caohai in the year 2017

Sample	Dredging	Coordinates
LS1	Projected	25°1.323 N, 102°40.058 E
LS6	Not projected	24°59.358 N, 102° 37.866 E
LS7	Projected	24°58.731 N, 102° 38.279 E
LS8	Not projected	24°58.616 N, 102°38.762 E

3.2.2.2 Analytics

3.2.2.2.1 Water Analytics

See Table 3.3

For the water analytics we used a spectral photometer using methods equal to international, European or German standards (compare Table 3.2).

For the most parameters the reagents were used in form of powder pillows. For the parameters total nitrogen and total phosphorus we used vial tests. These samples were digested using a vial cooker.

We measured each sample three times for statistical confidence. Presented results are arithmetic mean values. Organic nitrogen was calculated as the difference between total nitrogen and its ionic forms.

3.2.2.2.2 Sediment Analytics

To measure the sediment nutrient release we used acrylic glass tubes with a diameter of 6.7 cm and a volume of one litre. Each sample was divided into two subsamples for aerobic and anaerobic tubes. The samples were carefully covered with deionised water. Additionally we prepared two blind values. We used an air pump to ventilate aerobic samples during the measurement period. There was daily measurement of oxygen concentration, redox potential and temperature in both sampling rows. After 3, 7 and 11 days we analysed ammonium, nitrite, nitrate, orthophosphate, total dissolved phosphorus and total phosphorus like presented in Chap. 3.2.2.1.

Sediment nutrient release was extrapolated to the surface of the Caohai using the following formula:

$$\text{Sediment nutrient release} \left[\frac{\text{mg}}{\text{m}^2 * d} \right] = \frac{(\text{Concentration}_0 - \text{Concentration}_1) \left[\frac{\text{mg}}{\text{l}} \right] * \text{Volume [l]}}{\text{Surface Area [m}^2] * \text{Experiment time [d]}}$$

Table 3.2 Parameters and standards used to conduct spectral photometry for water samples

Group	Parameter	Standard
Nitrogen	Ammonium	DIN 38406-5:1983
	Nitrite	DIN EN 26777:1993
	Nitrate	DIN 38405-9:2011
	Total nitrogen	DIN EN ISO 11905-1:1998
Phosphorus	Orthophosphate	DIN EN ISO 6878:2004
	Total dissolved Phosphorus	DIN EN ISO 6878:2004
	Total phosphorus	DIN EN ISO 6878:2004

Table 3.3 Water analytics results

Sp. Point	Month	pH	EC	Redox	O2	Temp	NH4-N	NO2-N	NO3-N	TN	PO4-P	TDP	TP
Wangjiadiu	May	7.8	742	nm	3.9	18.6	7.0	nm	0.4	19.9	0.1	0.1	0.7
	July	7.7	881	116	4.0	21.6	8.4	0.02	0.3	11.2	0.6	0.7	0.9
Xinyunliang	May	7.7	634	143	0.3	20.7	2.4	0.08	0.8	7.7	0.3	0.3	1.4
	July	7.7	604	-6	3.3	20.0	1.8	0.09	0.7	5.7	0.3	0.3	0.4
	September	7.8	481	118	3.7	20.1	2.8	0.15	1.8	7.0	0.0	0.2	1.3
Laoyunliang	May	7.3	542	81	3.7	21.5	4.8	0.08	2.4	16.0	0.2	0.2	0.9
	July	7.4	702	109	3.2	21.3	4.1	0.20	3.2	16.5	0.3	0.4	0.6
	September	7.5	571	141	5.6	21.4	1.5	0.17	2.0	9.2	0.6	r	1.1
Wulong	May	7.3	560	131	7.5	21.5	2.1	0.04	3.8	14.0	0.1	0.1	1.0
	July	7.2	696	188	5.4	21.4	0.4	0.13	3.2	14.7	0.3	r	0.3
	September	7.2	620	119	5.5	21.9	1.7	0.37	4.6	14.0	0.1	r	0.6
Daguan	May	7.9	382	111	4.5	17.3	0.9	0.02	1.7	5.6	0.1	0.1	0.5
	July	7.5	435	175	0.5	21.4	1.4	0.16	1.5	9.8	0.7	r	r
	September	7.8	303	36	1.8	20.5	0.4	0.10	1.0	5.2	3.4	r	r
Xiba	May	8.7	405	102	11.3	16.7	0.1	0.08	1.3	3.4	0.1	0.1	0.3
	July	8.4	733	135	16.5	24.0	0.1	0.03	1.2	3.0	0.1	r	r
	September	8.1	362	158	4.5	21.5	0.9	0.24	2.0	3.7	0.3	r	0.8
Chuanfang	May	7.8	524	148	5.7	18.7	0.5	0.05	1.7	6.1	0.0	0.0	0.3
	July	7.6	661	135	6.3	24.4	nm	0.07	1.8	7.5	0.1	0.1	0.2
	September	7.8	475	132	3.8	21.5	0.2	0.11	1.6	4.6	r	0.3	0.3
Caohai 1	May	8.0	391	104	5.0	16.8	0.7	nm	1.8	3.5	0.1	0.1	0.3
	July	7.6	302	133	1.7	19.8	2.1	0.09	0.8	5.8	0.3	r	0.4
	September	7.8	358	70	2.9	26.5	0.6	0.06	1.4	4.3	0.0	0.6	r
Caohai 2	May	8.4	427	105	5.9	18.6	0.1	0.08	1.8	3.8	0.0	0.1	0.6
	July	8.6	394	112	9.9	22.6	0.1	0.08	1.4	8.7	0.1	r	0.4
	September	8.4	362	157	8.5	23.5	0.1	0.08	1.0	3.8	0.1	0.3	0.3
Caohai 3	May	7.6	682	36	3.1	17.7	4.5	0.79	3.6	34.4	0.5	0.7	r
	July	7.4	444	142	1.5	21.8	3.0	0.14	1.7	13.2	0.3	r	0.4
Caohai 4	May	8.7	541	36	5.4	19.7	0.1	0.10	1.4	4.8	0.1	0.2	0.8
	July	7.9	501	56	4.4	21.4	1.5	0.37	1.7	8.3	0.3	r	0.2
Caohai 5	May	8.4	349	57	8.8	20.1	0.4	0.04	0.8	1.6	0.1	0.2	0.3
	July	9.0	393	66	10.7	23.6	0.1	0.10	0.6	9.2	0.1	0.3	0.5
	September	8.7	360	145	9.4	25.0	0.2	0.10	1.4	4.4	0.1	0.2	0.6
Caohai 6	May	8.7	531	63	6.2	19.4	0.3	0.09	1.3	4.9	0.2	r	r
	July	8.2	543	139	10.0	22.7	0.2	0.26	1.7	10.5	0.1	r	0.4
	September	8.2	555	168	3.6	24.0	1.0	0.36	2.5	15.4	0.0	0.3	1.6
Caohai 7	May	8.9	301	-28	12.6	19.9	0.1	0.04	0.6	3.1	0.1	0.1	0.4
	July	9.0	383	113	9.8	24.3	0.5	0.11	1.1	4.8	0.1	r	0.2
	September	8.7	354	106	7.0	23.5	0.3	0.09	1.1	3.5	0.0	0.2	1.3
Caohai 8	May	8.9	333	9	8.9	20.4	nm	0.04	1.1	2.8	0.1	0.1	0.3
	July	8.4	425	167	6.8	23.9	1.3	0.16	0.7	4.7	0.1	r	0.3
	September	8.3	371	113	6.2	nm	0.3	0.11	1.3	4.2	0.1	r	0.4
Caohai 9	May	9.4	289	70	12.2	20.2	0.1	0.02	0.7	3.4	0.1	0.1	0.4
	July	8.0	404	170	3.9	23.0	0.7	0.13	0.5	5.0	0.1	r	0.3
	September	7.9	386	104	3.2	23.2	0.1	0.16	1.7	4.0	0.1	0.1	0.4
Outflow	May	9.5	339	82	16.5	21.2	0.1	0.05	0.8	7.8	0.1	0.1	0.3
	July	8.0	505	118	6.8	22.1	3.0	0.20	1.4	5.7	0.1	0.3	0.4
	September	7.8	549	131	4.6	22.9	0.2	0.39	2.4	8.0	0.3	r	0.5

pH-value dimensionless, electric conductivity in $\mu\text{S}/\text{cm}$; redox potential in mV, Temperature in $^{\circ}\text{C}$, all other values in mg/l
 nm = not measurable; r = rejected

Table 3.4 Environmental quality standards for surface water (Chinese standard: UDC614.7 (083.75) GB 3838-88)

Parameter	Unit	Class I	Class II	Class III	Class IV	Class V
COD	≤mg/l	15	15	20	30	40
BOD ₅	≤mg/l	3	3	4	6	10
Diss O ₂	≥mg/l	7.5	6	5	3	2
TN	≤mg/l	0.2	0.5	1	1.5	2
NH ₄ -N	≤mg/l	0.15	0.5	1	1.5	2
TP	≤mg/l	0.02 (lakes, dams 0.01)	0.1 (lakes, dams 0.025)	0,2 (lakes, dams 0.05)	0.3 (lakes, dams 0.1)	0.4 (lakes, dams 0.2)

Table 3.5 Compares between aerobic and anoxic milieu

		Release Rate 27.07. to 04.08.2017 [mg/m ² ·d]					
Aerobic		NH ₄ -N	NO ₂ -N	NO ₃ -N	PO ₄ -P	TDP	TP
1 May		130,3	3,79	14,6	31,9	33,7	24,9
	July	26,8	0,06	46,7	11,4	-2,0	-3,4
6 May		78,3	0,17	12,8	27,8	45,1	34,7
	July	-3,3	-4,36	-4,6	19,2	21,0	10,0
7 May		91,4	0,25	1,3	21,5	38,2	33,1
	July	22,7	-0,76	11,9	11,0	6,8	11,7
8 May		30,8	1,81	-17,5	15,9	-8,1	-5,1
	July	8,1	-0,77	3,5	14,2	-1,0	0,2
Anoxic							
1 May		102,8	0,17	13,6	20,2	16,0	17,3
	July	-1,5	38,00	115,3	27,8	18,1	10,2
6 May		74,2	0,09	6,4	26,8	12,8	17,1
	July	-27,8	-8,59	60,3	22,2	30,5	11,6
7 May		104,2	-0,17	-2,3	17,3	14,0	13,1
	July	5,1	24,01	77,1	6,6	6,8	6,2
8 May		54,1	0,04	-1,8	18,8	6,3	9,2
	July	9,4	26,75	76,3	15,3	6,8	7,0

3.2.3 Results and Discussion

3.2.3.1 Analytics

3.2.3.1.1 Water Analytics

Lake Caohai forms with its tributaries and its two separated parts three comparable water systems. Following, we will summarize the results of the water analytics, as they are the basis for the shallow lake modelling. In general, there are few sources in the literature available for both the Caohai and its tributaries. Furthermore, the Chinese water quality classification system is important for Chinese decision makers (Su et al. [31]). It uses chemical and biological oxygen demand, dissolved oxygen, ammonium-nitrogen, total nitrogen and total phosphorus concentrations as indicators

for classifying water bodies into six classes. “Class I” is the best and “Class V Inferior” the worst (see Table 3.5) (Table 3.4).

In the tributaries the pH-value was 7.7 (± 0.4), in the eastern part of Lake Caohai 8.5 (± 0.47), in the western part 8.1 (± 0.5) and 8.4 (± 0.7) in the outflow. We measured remarkable individual values in the Xiba River of 8.7 in May and in the Lake Caohai of 9.4 in May. These results shows the huge impact of algae in Lake Caohai shifting the pH to acidic conditions by uptake of CO_2 . The electric conductivity and the redox potential were unremarkable in most samplings. In the tributaries we measured the dissolved oxygen concentration to 5.1 (± 3.5) mg/l, to 7.4 (± 3.1) for the eastern part of Lake Caohai, to 4.9 (± 2.5) for the western part of Lake Caohai. Especially in the highly polluted western part the oxygen concentration was lower compared to the other parts. The highly turbulent outflow had a concentration of 9.3 (± 5.2).

The ammonium concentration in the tributaries was 2.2 (± 2.3) mg/l. It became obvious that the rivers Wangjiadui, Xinyunliang and Laoyunliang, which are flowing in the western separated part of Lake Caohai, had much higher ammonium concentrations. For this group the mean was 4.1 (± 2.35) mg/l. For the second group, containing the rivers Wulong, Daguan, Xiba and Chuanfang the mean value would be 0.8 (± 0.65) mg/l. This difference was calculated to be highly statistically significant ($p < 0.001$). Nitrite concentration in the tributaries was 0.11 (± 0.08) mg/l, 0.09 (± 0.04) mg/l and nitrate concentration was 1.9 (± 1.1) mg/l. At the Wulong river we measured a remarkable concentration of 4.6 mg/l in September. The concentration of total nitrogen was 9.2 (± 4.9) mg/l in the tributaries. Again, the rivers Wangjiadui, Xinyunliang, Laoyunliang, and also, Wulong formed a group with higher mean values of 12.3 (± 4.3) mg/l. Unlike, the group of other rivers had a mean value of 5.4 (± 2.05) mg/l, being a statistical highly significant difference ($p < 0.001$).

Orthophosphate was 0.4 (± 0.72) mg/l in the tributaries, 0.1 (± 0.1). The highest value measured was 3.36 mg/l in the Daguan River in September. Eliminating this one value as an outlier in the river dataset lowers the mean concentration to 0.2 (± 0.19) mg/l. Total phosphorus was 0.7 (± 0.4) mg/l in the tributaries. Yang and Jin [44] made investigations between 2005 and 2007 at the Caohai. For nearly all tributaries, the TN values dropped in this period. An exception was the Wangjiadui River, which in this study had a mean TN concentration of 9.5 mg/l over three years. Compared to our results there was no improvement of the water quality in the meantime. The water quality of the Wangjiadui River was Class IV regarding oxygen and water quality class V inferior in all nutrient parameters.

The Xinyunliang River was described at Huang et al. [14] to be the most polluted tributary of Lake Caohai. For 2012 they reported ammonium to be 6.92 mg/l, TN to be 13.40 mg/l and TP to be 0.38 mg/l. For the same year Nie [24] reported values for the rainy season where ammonium was 11.53 mg/l, TN 12.01 mg/l and TP 0.90 mg/l. Based on our results, we can assume that the actions that had been made to restore the river showed some success. Ammonium concentration dropped to only 20% of the former concentration reported. TN dropped to 56% of the values reported. There was no change in the concentration of TP. In general, the water quality became better but it is still poor. The Xinyunliang River was class V inferior in all aspects.

Zhiyi et al. [49] performed some measurements at the Laoyunliang River. They reported that ammonium was 1.55 mg/l, TN was 10.15 mg/l and TP was 0.26 mg/l as mean values from three measurements. Compared to this in our results ammonium and TP were significant higher. TN dropped 33%, which is a better water quality than in 2016, but still alarmingly high. The Laoyunliang River was class IV in the oxygen and class V inferior in all nutrient parameters. For the other parameters it was class V inferior having the highest nutrient concentration of TN and TP of the tributaries of the western part of Lake Caohai.

Yang et al. [45] gave a good overview over the development of the water quality of the Wulong River between 2005 and 2015. Thus, ammonium had a peak in 2008 with 33 mg/l and TP was 3.6 mg/l. Until 2015 the concentration dropped to 1 mg/l for ammonium and 0.25 mg/l for TP. In our measurements ammonium was worse with 1.37 mg/l and TP 0.62. Especially TN concentration, which are not reported yet in literature, showed high concentrations of 14.21 mg/l. Regarding this value, the Wulong river was one of the most polluted rivers in our study. The Wulong river reached water quality class III regarding the oxygen concentration.

There are few references regarding the Dagan River. There are few references regarding the Dagan [22] measured the water quality in February and March 2010 when the Wujiadui theme park was built. For the inflowing water from the Dagan River Ma et al. [22] measured ammonia concentrations of 6.26 mg/l and TP concentration of 5.65 mg/l. Compared to these values the water quality was much better in our study with ammonia being 0.90 mg/l and TP being 0.45 mg/l. Compared to some unpublished results from our scientific partners from the year 2015, we can assume that our values are plausible and that the Dagan Rivers water quality became better since 2010. For ammonia in our study, the river had a medium condition in May and July and good condition in September being one of the rivers with a better water quality. Nevertheless, the Dagan River is still one of the biggest pollution sources for Lake Caohai because of its huge discharge. The Dagan River had class V inferior for all parameters except of ammonium, where it had class IV and TP with class V.

The Xiba River had class IV in general for the oxygen, class III for ammonium and class V inferior for the other parameters. Looking to May and July it had water quality class I for oxygen and ammonium, but because of the high pH-Values and the ammonia-ammonium-equilibrium this class cannot be given.

Zhan et al. [48] reported an improved water quality for the Chuanfang River after a river rehabilitation. Compared to those 2014 values in our 2017 sampling campaign TN dropped around half. TP has not dropped and is still the same, as the Chuanfang River is still the outlet of the wastewater treatment plant number 1. The Chuanfang River had class IV in oxygen, II in ammonia, V in TP and V inferior in the other parameters. These four rivers flow to the western part of Lake Caohai. In this part of the lake the water quality class was V inferior for TN and TP. In the south east of the Caohai the water quality class was better regarding oxygen (between I and III) and ammonia (between I and IV). This picture could also be seen in the water quality of the outflow, which was in general class V inferior for the most parameters, but better for oxygen (I to IV).

In the eastern part of Lake Caohai the ammonium concentration was 0.4 (± 0.5) mg/l, 1.5 (± 1.5) mg/l in the western part and 1.1 (± 1.3) mg/l in the outflow. Nitrite was 0.09 (± 0.04) in the eastern part of Lake Caohai, 0.30 (± 0.23) mg/l in the western part of Lake Caohai and 0.21 (± 0.14) mg/l in the outflow. The nitrate concentration was 1.1 (± 0.4) mg/l in the eastern part of Lake Caohai, 2.0 (± 0.7) mg/l in the western part of Lake Caohai and 1.5 (± 0.7) mg/l in the outflow. The concentration of total nitrogen was 4.5 (± 1.8) mg/l in the eastern part of Lake Caohai, 13.1 (± 9.4) for the western part and 7.2 (± 1.1) for the outflow. For Orthophosphate the concentration was 0.1 (± 0.1) in the eastern part of the lake, 0.2 (± 0.2) mg/l in the western part and 0.1 (± 0.1) mg/l in the outflow.

For the lake Caohai Jia (2018) reported measurement results for total nitrogen and total phosphorus from 1993 to 2017. The water quality became better since the year 2011, when TN dropped to 6.1 mg/l and TP to 0.22 mg/l. The reported value for 2017 was 3.7 mg/l for TN and 0.15 mg/l for TP. Our result from the same year for the Caohai were 4.5 mg/l for TN and 0.44 mg/l for TP in the eastern part and 13.6 mg/l for TN and 0.9 mg/l for TP in the western part. As the reported values in the literature were yearly averaged values one can see that in the rainy season nitrogen and phosphorus values were significant higher in the rainy season, especially in the western part of Lake Caohai. We could show that nutrient intake from the rivers is still very high.

For the western part of Lake Caohai at the three sampling points, the water quality was different for oxygen ranging from class V inferior in the north to class III in the south. Nevertheless, the nutrient parameters were still class V inferior. A decrease in ammonium concentrations over the flow path could be seen. From the north (class V inferior) to the south (class III) ammonium concentrations dropped. Our results show not only that for a lot of parameters during the rainy season the water quality is in the worst class, but also that the difference to the threshold of class V is still remarkable. For example, TN in the middle of the western part of the Caohai was 34.4 mg/l whereas the threshold value is 2 mg/l. For TP the highest value was 1.58 mg/l in the same part of the lake and the threshold is 0.2 mg/l for water quality class V. The objective of the thirteenth five-year plan is to achieve the water quality class V until 2018 for the Caohai. Studying our data, which we collected as stich samples in the rainy season, one can see that there is a need for further effort to come closer to this objective. Our new data acquisition together with other available data now is the main input for the Caohai modelling described and discussed below.

3.2.3.1.2 Sediment Analytics

We measured dissolved oxygen, redox potential and temperature each day over the experiment's period as control parameters for the milieu as we had an aerobic sample row and an anoxic sample row. The full results can be seen in Table 3.1 of the Annex. In the aerobic row the dissolved oxygen concentration was always between 6.9 and 7.3 mg/l. As Kunming lies on a height of around 1,890 m above sea this equals a saturation between 98.5 and 104.2%. When the process became stable from day 2 on, the redox potential ranged between 89 and 115 mV. Because of these facts,

we assume that the aerobic milieu was stable in the whole period. The temperature ranged in small interval between 21.4 and 22.3 °C.

In the anoxic sampling row, the dissolved oxygen was under 0.7 mg/l at day 1 and dropped in most samples to 0.2 mg/l at day 2. In one sample, it dropped to 0.3 mg/l. In three samples, the dissolved oxygen dropped slower. Sample by sample it dropped under 0.2 mg/l. The last sample did not reach this value before day 9. The redox potential ranged from day 2 between -124 and -63. At the last day of our experiment, we measured a redox potential of -423 in one sample, which we assume as an outlier. In most cases, the anoxic milieu was stable aside from those samples the dissolved oxygen dropped slower. Similarly, like in the aerobic sampling row, the temperature ranged between 21.4 and 22.3 °C.

As we took the samples in two different months, we tested first if there were significant differences between the two monthly datasets. If so, we report the results separated by month. For the aerobic samples the difference between the two months was significant ($p < 0.05$) for ammonium and nitrite, but not for nitrate ($p > 0.05$). Ammonium sediment release was 82.7 (± 35.6) mg/m²d for the samples taken in May and 13.6 (± 11.9) mg/m²d for the samples taken in July. For nitrite, it was 1.51 (± 1.47) mg/m²d for May and -1.46 (± 1.71) mg/m²d for July. For nitrate, the mean value for all samples was 8.59 (± 17.5) mg/m²d.

For the anoxic sampling row, the difference for ammonium between the May and July set was very significant ($p < 0.01$), for nitrite it was not significant ($p > 0.05$) and for nitrate it was very significant again ($p < 0.005$). For ammonium, the sediment release rate was 83.8 (± 20.09) mg/m²d for the May data and -3.7 (± 14.5) mg/m²d for July. Nitrite was 10.0 (± 15.8) mg/m²d for the whole data set. Nitrate was 4.0 (± 6.6) mg/m²d for the May dataset and 82.3 (± 20.2) mg/m²d for the July dataset.

For ammonium and nitrite there was no significant difference in the sediment release rate between the aerobic and the anoxic samples ($p > 0.1$). For nitrite it was very significant ($p < 0.005$).

For nearly all three parameters in both milieus and in both months the sediment release rate in the northeast of the Caohai, where the Dagan river flows into the Lake (sample 1) was much higher than nearby the outflow (sample 8). An exception is ammonium in the anoxic milieu in July.

Looking at the phosphorus parameters there were nearly no significant differences between the results of May and July. The only exception was total phosphorus in the anoxic milieu ($p < 0.05$). The sediment release of orthophosphate in the aerobic milieu was 19.1 (± 7.1), 16.72 (± 19.2) mg/m²d for the total dissolved phosphorus and 13.24 (± 15.0) mg/m²d for total phosphorus. In the anoxic milieu, orthophosphate was 19.4 (± 6.3) mg/m²d and total dissolved phosphorus was 13.9 (± 7.6) mg/m²d. Total phosphorus was 14.2 (± 3.3) mg/m²d in May and 8.8 (± 2.2) mg/m²d in July.

For all three parameters there was no statistical significant difference between the aerobic and the anoxic milieu ($p > 0.5$). In the aerobic May dataset and both anaerobic datasets, the sediment release rate was higher at the Dagan inflow in the northeast of the Caohai than at the outflow.

Based on our results we have strong indications for the assumption that the sediment has a huge impact on the water quality of the Lake Caohai. As there were enormous nutrient sediment releases measured in the anoxic, as well as in the aerobic, samples, we strongly recommend to take an even stronger focus on the sediment in future lake management. As it is not the expected behavior of the sediment to show such high sediment release rates in the aerobic milieu, we also strongly encourage other scientist to repeat and further test the sediment of Dianchi to gain a more detailed picture and to scrutinize our results.

3.3 Study 2: Shallow Lake Modelling of Lake Caohai

As already mentioned, the water quality of Lake Caohai and its tributaries was proven to be to be strongly eutrophicated. The heavy and complex burden of water pollution do not make simple answers very likely. From other examples like Lake Constance (Bloesch and Schroeder [1]), one can learn that the basis for a successful water management lies in strong management efforts. They are consisting of thorough and ongoing investigations to establish the best system understanding, consistent management decisions and long-term commitment for decades.

Thus, in the frame of the Sino-German project SINOWATER Chinese and German partners together discussed several conceivable management scenarios. To evaluate the effects of the different management scenarios we decided to use a dynamic simulation approach.

Numerical models are very useful management tools in lake management, as they aggregate existing knowledge and hypotheses and thus allow the dynamic simulation of different management scenarios. For the SINOWATER project, we used the Stoichiometric Lake Model (StoLaM, Strauss [29] and Strauss et al. [30]), which is designed to deal with deep as well as shallow lakes in the context of sustainable lake management. As the basis for the dynamic lake modelling, StoLaM includes the hydrodynamic sub-model HyLaM, which is able to calculate the hydrodynamic conditions with high temporal and spatial resolution even in shallow lakes. StoLaM describes several nutrient cycles quantitatively taking into account the sediment–water interactions and the role of plankton and omnivorous fish in the food web.

The Dianchi was several times modelled with different lake models (Vinçon-Leite and Casenave [34]). However, especially for a system in which phosphorus and nitrogen in combination determine the growth of algae, we consider the use of models with phytoplankton modules as reasonable, which integrate dynamic nutrient storage as well as stoichiometric principles. Although these processes have been known and considered as relevant since a longer time, they are rarely integrated even in complex ecosystem models (Jørgensen [17] and Strauss et al. [30]). We see the use of StoLaM, which combines such features with a hydrodynamic 1D modelling approach, as a relevant contribution to further improving management decisions for Lake Dianchi.

The setting up of the StoLaM for Lake Caohai was a two-step approach:

In 2016, the model was parameterised with the available hydrological and climatic basic information and starts with default estimations on the nutrient cycles, sediment interactions and plankton 2016. Based on the first draft model we presented first findings in the 2016 SINOWATER Dianchi Shallow Lake Symposium at Kunming. At this time the values for the lake-internal sediment nitrogen and phosphorus release rates were as unknown as the nutrient loss due to sedimentation of detritus-bound nutrients from the inflows. Furthermore, a detailed picture of the composition of those nutrients in the inflowing waters was not available and we could not estimate the denitrification rate due to this lack of data.

Through the SINOWATER sampling campaign in 2017, a broader data set could be used for the parameterization and validation of the entire model, which was adapted to the specific simulation tasks in this project. This data included the new water quality data from the Kunming Institute of Environmental Sciences as well as our own monitoring data as described above, with which it was possible to calculate management scenarios more properly.

Together with our Chinese project partners we have developed scenarios that are based on previously discussed water management strategies in Kunming. Finally, five main scenarios were selected for numerical modelling, with some of the main scenarios being divided into sub-scenarios as described below:

1. **Standard scenario,**
modelling the status quo in a high temporal (at least hourly time steps) resolution
2. **Sediment nutrient release Scenario,**
is dealing with changes in the nutrient release of the sediment as they could occur after sediment removal by dredging measures
3. **Tributary Water Quality Scenario,**
is dealing with changes in the nutrient inflow through the tributaries as they could occur after improvements in the sewer system or upgrading waste water treatment plant
4. **Tributary Water Quantity Scenario,**
is dealing with changes in the amount of water flowing into Lake Caohai through the tributaries as they could occur for several natural or anthropogenic changes in the catchment controlling the wash out of nutrients
5. **Caohai Water Level Scenario,**
is dealing with changes in the water level and thus the water volume in the lake

First of all, we want to introduce the conceptual structure and theoretical foundations of the model StoLaM. Secondly, we show the data used and the results obtained. Finally, we discuss the results and management scenarios.

3.3.1 Theoretical Foundation and Adaptions

The Stoichiometric Lake Model (StoLaM, Strauss [29] and Strauss et al. [30]) is a general 1D hydrodynamic-ecological water quality model for standing waters. It includes the hydrodynamic lake model HyLaM (Strauss [29], validated by Gross [8]), which generates the underwater light regime, water temperature and turbulence in the vertical water column using measured weather data as input (compare Fig. 3.4). With this approach, it is possible to simulate a realistic physical environment for the plankton organisms with a high temporal resolution. Based on these hydrodynamic processes, the modelling concept takes into consideration the zooplankton and phytoplankton succession, the quantitative cycles of several nutrients (P, N, Si), the oxygen concentration, the sediment–water interaction and the role of omnivorous fish in food web interactions. StoLaM integrates the principles of ecological stoichiometry, which means the changing ratios of nutrients such as carbon, phosphorus, nitrogen and silicon within the food web and their impact on the growth rates of plankton organisms with respect to nutrient limitation. Beneath the weather conditions and the ingestion by herbivorous zooplankton, the phytoplankton growth is based essentially on the amount of phosphorus, nitrogen and silicon, which are potentially limiting nutrients. The external nutrient concentration and the variable internal nutrient pool (cell quota) of the algae controlled the uptake rates of phosphorus and nitrogen from the water phase and storage by the algae. Thus, the growth rate of the algae depends on the internal nutrient cell quota. In this approach, the carrying capacity of the algae depends on the amount of the most limiting nutrient stored in the algal biomass (for further particulars of the complete model, see Strauss [29] and Strauss et al. [30]).

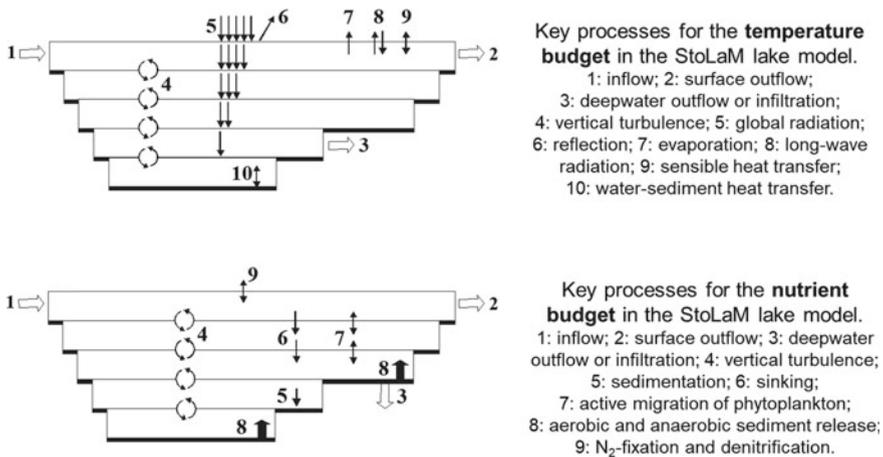


Fig. 3.4 Schematic overview of the most important processes of the full version of StoLaM

Due to the modular character of the model, the complexity of the modules used can be easily adapted to the problem definition and data availability. In this study, a simplified version of the interactions and food web structure in StoLaM was used. The following assumptions reduced the complexity of the standard model:

In the Caohai, the blue-green algae (cyanobacteria) *Microcystis aeruginosa* is the predominant species, which also caused the main water quality problems.

Since data on the exact composition and interaction of other organisms were lacking, the implemented complex food web of StoLaM was massively reduced in this model application. For the phytoplankton community, we only considered the population dynamics of *Microcystis aeruginosa*, and no other organism such as zooplankton or fish has been modelled. The dynamics of the cyanobacteria *Microcystis* can also be described without further food web interactions such as zooplankton grazing. This procedure is also justified by the fact that larger colonies of the cyanobacterium *Microcystis* are not edible for zooplankton due to their size, and in some cases also due to excreted toxins (e.g. microcystin). Thus, the chemical-physical environmental conditions and the residence time of the water body are particularly most relevant for modelling the *Microcystis* dominance in Lake Caohai. The silicon dynamics, which are only relevant for diatoms, was also switched off. The physico-chemical processes in the model were otherwise left unchanged. For a better comparison with measured field data, the biomass of *Microcystis* was converted and displayed as chlorophyll-a concentration.

3.3.2 Data input and model parameterisation

In general, we simulated the eastern free flowing part of the Caohai. We included the rivers Wulong, Dagan, Xiba and Chuanfang as inflows.

A simple, box-shaped lake morphometry with 5 vertical layers of 50 cm height each and a uniform surface of 6,790,000 m² was used for this model application. For the simulation of the water temperature with the HyLaM model we used meteorological data of the year 2016 provided by the Kunming centre weather station (air temperature [°C], wind speed [m s⁻¹], atmospheric pressure [hPa], and air humidity [%], time resolution of the measurements: 3 h). The global radiation [J cm⁻² s⁻¹] was calculated on an hourly basis using the formulas of Ryan & Stolzenbach [28], taking into account a typical monthly averaged cloud cover (see Table 3.6). The water attenuation coefficient was set to 0.3 m⁻¹.

To calculate mass flow rates from the concentrations we used river discharges provided by our Chinese partners from the Kunming Institute of Environmental Sciences (compare Fig. 3.5). The specific dynamics of these rivers are reported in a recent paper of these partners (He et al. [12]). An essential input for the StoLaM model is the nutrient data. As it was needed at least as monthly values we took results from the monthly measurements of the Kunming Institute of Environmental Sciences. The proportion of detritus-bound phosphorus (65% of the total phosphorus) and nitrogen (76% of the total nitrogen) was calculated by subtracting the total fractions and the

Table 3.6 Long-term values of mean monthly cloud cover between 2002 and 2015 from the NASA data archive for the Dian Lake Basin region.

Month of the year	Mean cloud cover [%]
1, 2	30
3, 4, 12	45
5	60
6, 7, 10, 11	75
8, 9	85

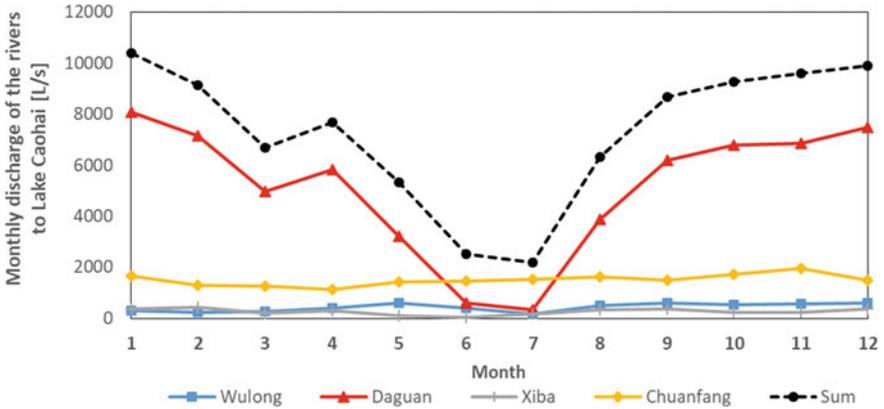


Fig. 3.5 Discharge hydrographs in 2016 used for the simulation provided from the Kunming Institute for Environmental Sciences

dissolved inorganic fractions (compare Table 3.7). The knowledge of the detritus-bound fraction of the inflowing nutrients is extremely important for the simulation results, because a relatively fast sedimentation can be assumed for detritus, whereas the dissolved fractions are available to the algae as nutrients in the water column.

The daily remineralisation rate of detritus-bound nutrients in the lake at 20 °C was calibrated within this project and was set to 20% for carbon, 25% for nitrogen and 20% for phosphorus (temperature coefficient of 1.13, Strauss et al. [30]). The standard detritus-settling rate was assumed as 1 m day⁻¹. In our modelling approach, there was no additional nutrient release assumed by remineralisation of resuspended sediment in the water column.

For the sediment nutrient release rates, we used our measurements in 2017. As described above, experimental data on sediment release was inhomogeneous. Thus, we took the arithmetic mean values to get robust parameters for the sediment release: For nitrogen the sediment release was 83 mg NH₄-N m⁻²d⁻¹ and for phosphorus it was 19 mg PO₄-P m⁻²d⁻¹. We used these measurements as baseline values at 20 °C in the model, and simulated the temperature-dependent release with an implemented temperature dependency (temperature coefficient of 2.42, Strauss et al. [30]). In the release experiments, we did not find significant differences between aerobic

Table 3.7 Mean nutrient concentrations of the combined external Inflows per year and month (measured in 2017).

	mg PO ₄ /L	mg Detr-P/L	mg NO ₃ /L	mg NH ₄ /L	mg Detr-N/L
Annual mean	0.066	0.212	1.65	0.76	4.53
January	0.063	0.197	1.52	0.80	4.41
February	0.065	0.196	1.51	0.80	4.38
March	0.064	0.201	1.56	0.79	4.47
April	0.067	0.203	1.58	0.82	4.50
May	0.069	0.224	1.76	0.79	4.78
June	0.067	0.249	1.96	0.62	4.81
July	0.061	0.235	1.79	0.48	4.21
August	0.069	0.216	1.68	0.76	4.53
September	0.069	0.209	1.63	0.81	4.56
October	0.065	0.205	1.60	0.80	4.55
November	0.065	0.206	1.61	0.79	4.55
December	0.067	0.205	1.59	0.82	4.55

and anaerobic conditions. Hence, we used the same release rates for all oxygen concentrations in the model.

The use of measured sediment nutrient release rates in the model, however, is a necessary parameterisation of lake-specific processes. It is important to mention, that the StoLaM model (vers. 3.02.03) was used in this project without any calibration of parameters with one exception: Only the remineralisation rates of detritus-bound nutrients were adjusted to optimize the fit between measured and simulated nutrient concentrations in Lake Caohai. This data gap could be closed by future experiments.

We modelled the non-N₂-fixing cyanobacterial species *Microcystis aeruginosa* with a maximum growth rate of 1.5 day⁻¹ and a mortality rate of 0.1 day⁻¹. A vertical buoyancy of 1.38 m day⁻¹ is used, which leads to an accumulation in the upper water layers at low turbulence of the water body. An overview of all algal parameters used is given in Annex 2.

3.3.3 Model simulation scenarios

As described in the introduction we investigated five main scenarios. The first scenario was the standard run of the simulation with no changes of the parameters described above. We divided the second scenario into one sub-scenario without nutrient release of the sediment (2a) and another sub-scenario with a sediment release of only 75% of the standard to simulate a dredging of 25% of the lakes surface (2b). The third scenario again we divided into two sub-scenarios: One was without external inflow of nutrients (3a), and as the second sub-scenario, we set the nutrient

to only 50% of the standard value (3b), both at unchanged water quantities. These scenarios should show the effects of an increased wastewater treatment and decentralized actions in the catchment system. The fourth main scenario we separated into three sub-scenarios. First, we used an annual mean water flow instead of a hydrograph to smooth the variation in discharge (4a). Second, we doubled the inflow from the rivers at constant nutrient concentrations, simulating an increasing number of inhabitants and impervious areas (4b). Finally, we halved the inflow to test the influence of possible sponge city actions, which are discussed in China a lot (4c). The fifth main scenario we separated into two sub-scenarios. We changed the standard water depth from 2.5 to 1.5 m (5a) and to 3.5 m (5b). These options were discussed in Kunming at the time of our project and thus considered.

An important input for the HyLaM module is the theoretical hydrological renewal time of the lake water. It describes the number of days necessary to flush the total volume of the lake water body once. The renewal time was calculated for each scenario in dependence of the inflowing water and the lake volume (compare Table 3.8).

Table 3.8 Water inflow, lake volume and water renewal time

No.	Description of scenario	Sub-scenario	Water inflow [l/s]	Lake volume [Mio. m ³]	Theoretical water renewal time [days]
1	Standard	Satatus Quo nutrient inflow concentration, nutrient release and unsteady inflow; 2.5 m water depth	7,300	16.975	26.9
2a	Sediment nutrient release	No release	7,300	16.975	26.9
2b		75% release (= 25% lake area dredged)	7,300	16.975	26.9
3a	Tributary water quality	0% nutrient inflow	7,300	16.975	26.9
3b		50% nutrient inflow	7,300	16.975	26.9
4a	Tributary water quantity	Continuous average	7,300	16.975	26.9
4b		200% inflow	14,600	16.975	13.5
4c		50% inflow	3,850	16.975	51.0
5a	Caohai water level	1.5 m	7,300	10.815	17.1
5b		3.5 m	7,300	23.765	37.7

Table 3.9 Simulated phosphorus flows and balance (yearly average) in $\text{mg P m}^{-2} \text{d}^{-1}$

No	Description of scenario	Sub-scenario	Inflow	Sediment release	Sedimentation	Outflow	Balance
1	Standard		25.3	14.6	23.1	16.4	0.4
2a	Sediment nutrient Release	No release	25.3	–	16.6	8.5	0.2
2b		75% release (= 25% lake area dredged)	25.3	10.9	21.6	14.3	0.3
3a	Tributary water quality	0% nutrient inflow		15.5	7.3	8.4	-0.2
3b		50% nutrient inflow	12.9	14.8	15.6	11.9	0.2
4a	Tributary water quantity	Continuous average	25.8	14.3	23.0	16.8	0.3
4b		200% inflow	51.6	12.3	31.8	31.6	0.5
4c		50% inflow	13.6	15.4	18.5	10.2	0.3
5a	Caohai water level	1.5 m	25.3	15.0	23.3	16.8	0.2
5b		3.5 m	25.3	14.1	21.4	17.2	0.8

3.3.4 Caohai Shallow Lake Model Results

The simulation of the water temperature by the standard StoLaM run was compared to the temperature measurement from the Kunming Institute of Environmental Sciences (see Fig. 3.6). At the Caohai, there were two temperature-monitoring stations. One station was situated in the middle of the lake, the other one called Duanqiao. The good

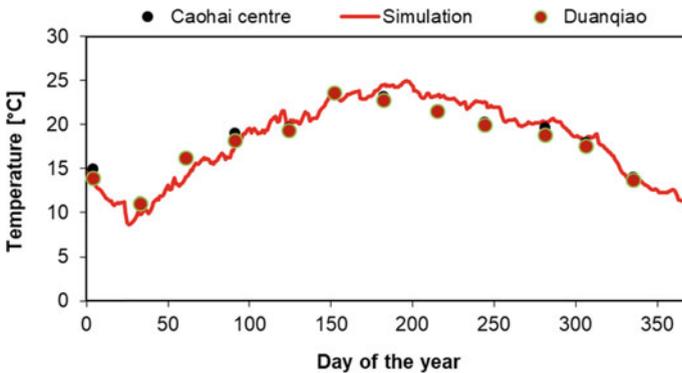


Fig. 3.6 Simulation of the water temperature in 2016 (lines: simulation result. Black dots: Caohai Centre monitoring station; red dots: Duanqiao monitoring station)

agreement between simulation and measurements is an indication that the hydrological and hydrodynamic conditions for the Lake Caohai can be well reproduced with the model.

Standard Scenario

The comparison of the results from the simulation with the measured values from the Kunming Environmental Science Institute showed that the standard simulation run fits the field data well while taking the measured nutrient sediment release data into consideration (scenario 1, compare Table 3.9). In this scenario, the dynamics of the cyanobacteria *Microcystis* can also be described without further food web interactions.

Sediment Nutrient Release Scenario

The hypothetical scenario without sediment release shows a massive reduction of the algal bloom and the phosphorus concentration in summer compared to the standard run (scenario 2a). This illustrates the high relevance of the internal lake phosphorus release during summer in this eutrophic shallow lake. As expected, a reduction of the sediment area by 25% causes only a smaller improvement of the water quality for all parameters (scenario 2b).

Tributary Water Quality Scenario

Reducing the nutrient content of the inflow by 100% (scenario 3a) clearly reduces the total concentration of algae and phosphorus in the lake, but will not change their overall dynamics in the short term due to the very high internal P-loading by the sediments. Only a clear decrease in nitrogen concentrations is observable. The same is true for the scenario 3b with only 50% of the nutrient loading, but the effect is much smaller than in scenario 3a.

Tributary Water Quantity Scenario

The use of annual mean values for the discharge of the tributaries (scenario 4a) reduces the algal and phosphorus peaks in summer, as the measured reduced inflow in summer was artificially compensated in the model, but does not have a fundamentally positive effect on nutrient concentrations. Nevertheless, this shows the clear influence of the inflow fluctuations of the Dagan River on the water quality of the entire lake. Doubling the water inflow rate significantly reduces the algae and phosphorus content during summer in the lake due to leaching despite the unchanged high nutrient concentrations in the inflow (scenario 4b). Halving, on the other hand, does not make much of a difference, as the positive effect of less nutrients is compensated by less washout of algae and phosphorus (Scenario 4c).

Caohai Water Level Scenario

Lowering the water depth by 1 m increases the algal bloom considerably due to the lower water volume (scenario 5a), especially since the nutrients released from the sediment concentrate more quickly in the water column. In comparison, increasing the water depth by 1 m causes only a relatively small change in nutrient concentration,

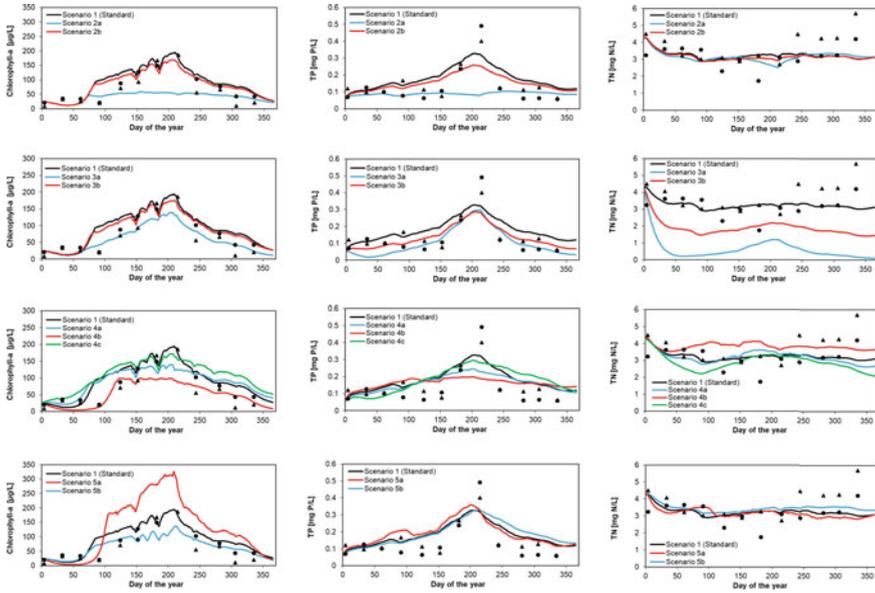


Fig. 3.7 Results of the scenario simulation with StoLaM (lines: simulation results. dots: Caohai Centre monitoring station; triangles: Duanqiao monitoring station)

but reduced the cyanobacterial biomass significantly since a deeper vertical mixing into the water column causes a poorer light supply to the algae (scenario 5b) (Fig. 3.7).

In addition to the nutrient concentrations over the year, mass balances also provide important information on enrichment or depletion in the water body.

In the case of the phosphorus budget, it can be seen that the scenarios only slightly differ in the whole-year balance and usually reach the low spring phosphorus level again in the subsequent winter, even if there is sometimes a slight increase in the water column (see balance column in Table 3.5). For the scenario with doubled inflow (4b), the higher nutrient inflow from the tributaries results in slightly higher water column enrichment than in the standard run (scenario 1). Furthermore, the enrichment is higher in the scenario with a water depth of 3.5 m (5b) due to an overall slightly reduced phosphorus sedimentation. Only in the scenario without nutrient input from the rivers (3a) phosphorus depletion in the water column can be found compared to the starting conditions of the simulated year.

Looking at the difference between nutrient release from the sediment and sedimentation rates (Table 3.5), the hypothetical scenario 3a shows a depletion of the phosphorus content in the sediment, as less particulate phosphorus enter the lake by the inflows. In most other scenarios, however, sedimentation exceeds the sediment release and leads to a long-term accumulation of phosphorus in the sediment.

For nitrogen, the picture is slightly different (Table 3.10). Here the nitrogen whole-year balance for the water column of the status quo scenario 1 shows already a

Table 3.10 Simulated nitrogen flows and balance (yearly average) in $\text{mg N m}^{-2} \text{d}^{-1}$

No	Description of scenario	Sub-scenario	Inflow	Sediment release	Sedimentation	Outflow	Denitrification	Balance
1	Standard		638.7	63.8	385.7	321.6	4.1	-8.9
2a	Sediment nutrient release	No release	638.7	-	337.7	301.7	8.3	-9
2b		75% release (= 25% lake area dredged)	638.7	48.4	378.0	313.7	4.2	-8.8
3a	Tributary water quality	0% nutrient inflow	-	67.7	37.3	57.8	0.6	-28
3b		50% nutrient inflow	320.6	64.7	217.3	185.9	1.8	-19.7
4a	Tributary water quantity	Continuous average	645.0	62.5	391.2	321.9	2.9	-8.5
4b		200% inflow	1290.0	53.9	599.7	735.9	9.6	-1.3
4c		50% inflow	338.7	67.4	261.1	157.7	1.9	-14.6
5a	Caohai water level	1.5 m	638.7	65.6	398.2	306.4	5.1	-5.4
5b		3.5 m	638.7	61.5	359.8	344.7	4.9	-9.2

washout of $8.9 \text{ mg m}^{-2} \text{ d}^{-1}$, which is in the same order of magnitude as the other scenarios.

Denitrification of dissolved inorganic nitrogen plays only a minor role in the nitrogen balance of the lake.

The low level of denitrification in the water body is a result of the assumption, based on the available measurements, that a large part of the inflowing nitrogen is present in particulate form as detritus bound and sinks relatively quickly onto the sediment.

Overall, the available simulations permit the conclusion that nitrogen also accumulates massively in the sediment, which should be verified by further measurements.

3.4 Summary and Recommendations for the Dianchi Shallow Lake Management

3.4.1 Shallow Lake Management – A Very Special Task

Shallow lake management is even more challenging than the management of deep-water lakes. Shallow lakes are more vulnerable than deeper waters due to a higher ratio between sediment area and water volume as well as lower retention capacity for nutrients, and because interactions between pollution, physical and ecological components are linked with a high dynamic. Because of the sensitivity of shallow lakes, shallow lake management must follow a very consistent and strict management strategy.

With the 2018 Revision of World Urbanization Prospects the UN (UN, 2019) points out “Sustainable urbanization is key to successful development. Understanding the key trends in urbanization likely to unfold over the coming years is crucial to the implementation of the 2030 Agenda for Sustainable Development, including efforts to forge a new framework of urban development.” The poor status of Dianchi points to an overwhelmingly obvious example of cause and effect in that context. Unsustainable growth of Kunming caused a level of pollution for the lake, which exceeded the carrying capacity of that very vulnerable water resource.

The remediation of Dianchi will take a full and lasting commitment in a water and resources management based on a system understanding of cause and effect for the benefit of Kunming. Thus, “the future growth of cities and concomitant appropriation of land and natural resources will determine success towards an environmentally sustainable future. In some cities, unplanned or inadequately managed urban expansion leads to rapid sprawl, pollution, and environmental degradation.” (UN, 2019).

Today’s pressures on Dianchi are much more complex and severe compared to those of ancient China. Modern pressures need integral solutions. Thus, the Dianchi Management Authorities have agreed on six main measures consisting of (1) waste water treatment & circular sewer, (2) lakeside remediation, (3) river remediation, (4) sediment management, (5) conservation of water resources, and (6) water transfer/efficiency. During our SINOWATER studies and together with our partners

from Kunming and experts from other parts of China we discussed the relevance of evaluation as an instrument for water management. An economic benefit approach for example indicated that only subtle further investment and easy technical modifications could produce relevant additional purification effects for the lakeside wetlands (Wermter [41]).

3.4.2 Necessity for Highest Data Quality

The intention of the survey of the water quality of the northern part of the lake called Caohai was to produce the basis for the system understanding of the shallow lake of Caohai. Above we described our two step approach. First we collected available environmental, hydrological and pollution data from Kunming authorities and other sources. It was possible to parameterise a lake model based on the preliminary data set. It supported first estimations of management scenarios. But the first and quick answers came with a relative vagueness. Relevant monitoring data was still missing like P release from lake-internal sediment. One of the less astounding but nevertheless crucial findings of this very practical approaches was the acknowledgement of the necessity for a sufficiently high data quality, both quantitatively and qualitatively.

Because data uncertainty causes model uncertainty and model uncertainty causes planning and management uncertainties, we recommended and conducted a thorough sediment and river pollution monitoring campaign in 2017. Examination of the sediment phosphorus release and additional measurements of N and P was crucial for the improvement of the shallow lake model of Caohai. In May 2018 we presented and discussed the results of StoLaM shallow lake model of Caohai and the out-comes of a set of management scenarios.

In general, the Stoichiometric Lake Model StoLaM is applicable for the modelling of Lake Caohai and provides evaluable results for the selected scenarios. The model results clearly show that with the measured nutrient sediment release rates, the seasonal dynamics of the water quality parameters in the water column, measured by the Kunming Institute for Environmental Sciences, can be well simulated. Nevertheless, some measurements are still subject to increased uncertainty. The knowledge of the particle-bound nutrient content in the tributaries and its sedimentation and remineralisation rates are crucial for a quantitatively correct analysis and modelling of the water quality in Lake Caohai, and should be further investigated. Also, the very high nutrient release rates of the sediments under both, oxic and anoxic conditions, should be confirmed by further measurements. Besides, more measurements of the water inflows are necessary in future to have a broader data basis, especially in the River Dagan.

There are also factors, which were still unknown. Meanwhile, we have an estimation of the nutrient release from the sediment, but there is a lack on data or estimations of the nutrient release by resuspended detritus particles due to lake internal processes such as wind-induced turbulence. These resuspension events might have also a significant potential for nutrient release into the water column.

3.4.3 Recommendations of Most Promising Management Scenario for the Lake Management

Based on the simulation results in this project we can derive clear recommendations for the lake management of Lake Caohai. We already saw that decentralised sponge city measures in the catchment area could improve the nutrient concentration in the lake and thus the potential for washout. Especially scenarios with an increased water quantity and a reduced nutrient loading in the lake inflow would have a huge impact. However, dropping the discharge to an amount as shown in the simulated scenarios would not be possible by sponge city measures in the next years only.

Summing up, we recommend to improve the trophic situation of the rivers flowing into Lake Caohai to prevent huge lake nutrient intake and to improve the washing out capacity. A significant reduction in the nutrient load of the tributaries can improve the lake water quality immensely. However, dissolved and particulate nutrient fractions should be evaluated separately. In general, an increase of inflowing water quantities favours an improvement of the water quality in the lake through wash-out processes, especially during the summer months.

Changes in the filling level of this shallow lake can have significant positive and negative effects on the water quality by changing the water retention time and the sediment area/water volume ratio. In any case, a significant increase in the mean water depth of the lake would have a clearly positive effect on the reduction of algal blooms.

We also recommend a significant reduction of the nutrient release rate in the lake, which would also greatly reduce the nutrient concentration in the summer months. According to our data, this can only be achieved by dredging, as there are massive nutrient releases even under aerobic conditions.

The measures discussed here (deepening of the lake, increased inflow, reduced nutrient load, reduced sediment release) can be combined without any conflict of optimisation for the lake water quality. The efficiency of different combinations can also be assessed with the modelling approach presented here.

We also recommend to apply the StoLaM model or a comparable model to the Dongfengba part of Lake Caohai, as the monitoring results show that the water quality in this part and its tributaries is much worse than in the eastern part of the Caohai.

Annex

Annex 3.1: Full Compares Between Aerobic and Anoxic Milieu

	24.07.2017		25.07.2017		26.07.2017		27.07.2017		28.07.2017		29.07.2017		30.07.2017		31.07.2017		01.08.2017		02.08.2017		03.08.2017		04.08.2017																
	O ₂	Rdx	Temp	O ₂	Rdx	Temp	O ₂	Rdx	Temp	O ₂	Rdx	Temp	O ₂	Rdx	Temp	O ₂	Rdx	Temp	O ₂	Rdx	Temp	O ₂	Rdx	Temp															
Aerobic	1	May	7.1	53	22.0	7.0	119	21.6	7.1	90	22.1	7.1	88	21.8	7.0	101	21.4	7.0	69	22.2	7.1	102	21.8	7.0	109	22.3	7.0	107	21.5	7.0	103	21.7	7.0	102	22.2				
	July	7.3	49	22.0	7.3	113	21.6	6.9	103	22.1	7.0	114	21.8	7.0	111	21.4	7.1	103	22.2	7.0	107	21.8	7.1	104	22.1	7.1	109	22.3	7.1	106	21.5	7.1	99	21.7	7.2	110	22.2		
	6	May	7.1	52	22.0	7.1	116	21.6	7.0	93	22.1	7.0	82	21.8	7.0	100	21.4	7.0	108	22.2	7.1	103	21.8	7.1	106	22.1	7.1	106	22.3	7.0	101	21.5	7.1	112	21.7	7.0	111	22.2	
	July	7.2	52	22.0	7.3	115	21.6	7.1	113	22.1	7.1	105	21.8	7.1	108	22.2	7.0	108	22.2	7.0	112	21.8	7.1	111	22.1	7.1	102	22.3	7.0	105	21.5	7.0	115	21.7	7.0	108	22.2		
	7	May	7.1	48	22.0	7.1	111	21.6	7.1	112	22.1	7.1	110	21.8	7.0	102	21.4	7.0	96	22.2	7.1	100	21.8	7.0	102	22.1	7.0	106	22.3	7.1	110	21.5	7.1	110	21.7	7.0	104	22.2	
	July	7.2	50	22.0	7.2	114	21.6	7.0	114	22.1	7.1	96	21.8	7.1	96	21.4	7.0	96	22.2	7.0	99	21.8	7.1	104	22.1	7.1	104	22.3	7.1	100	21.5	7.0	105	21.7	7.0	103	22.2		
	8	May	7.2	45	22.0	7.3	110	21.6	7.3	89	22.1	7.1	96	21.8	7.0	96	21.4	7.1	111	22.2	7.1	112	21.8	7.0	111	22.1	7.1	115	22.3	7.1	110	21.5	7.1	111	21.7	7.1	108	22.2	
	July	7.2	51	22.0	7.1	115	21.6	7.0	103	22.1	7.0	101	21.8	7.1	95	21.4	7.1	102	22.2	7.1	105	21.8	7.1	98	22.1	7.1	98	22.3	7.0	102	21.5	7.0	111	21.7	7.1	111	22.2		
Blind	7.1	114	22.0	7.2	49	21.6	7.2	86	22.1	7.1	92	21.8	7.1	92	21.8	7.1	89	21.4	7.1	90	22.2	7.1	100	21.8	7.1	105	22.1	7.1	104	22.3	7.1	114	21.5	7.1	113	21.7	7.1	112	22.2
Anaerobic	1	May	0.7	-105	22.0	0.2	-118	21.6	0.2	-104	22.1	0.2	-111	21.8	0.3	-105	21.4	0.3	-99	22.2	0.3	-105	21.8	0.2	-98	22.1	0.2	-101	22.3	0.2	-96	21.5	0.2	-103	21.7	0.1	-105	22.2	
	July	0.6	-100	22.0	0.2	-43	21.6	0.3	-76	22.1	0.2	-80	21.8	0.2	-85	21.4	0.2	-89	22.2	0.2	-110	21.8	0.2	-109	22.3	0.2	-104	22.3	0.2	-99	21.5	0.2	-100	21.7	0.1	-107	22.2		
	6	May	0.6	-150	22.0	0.4	-145	21.6	0.7	-123	22.1	0.6	-115	21.8	0.6	-99	21.4	0.4	-86	22.2	0.4	-100	21.8	0.3	-111	22.1	0.3	-109	22.3	0.2	-107	21.5	0.2	-105	21.7	0.2	-99	22.2	
	July	0.5	-14	22.0	0.4	-64	21.6	0.3	-81	22.1	0.3	-99	21.8	0.3	-95	21.4	0.3	-86	22.2	0.2	-93	21.8	0.2	-98	22.1	0.2	-99	22.3	0.2	-108	21.5	0.1	-103	21.7	0.1	-98	22.2		
	7	May	0.5	-130	22.0	0.2	-134	21.6	0.3	-89	22.1	0.3	-93	21.8	0.2	-103	21.4	0.1	-110	22.2	0.2	-105	21.8	0.2	-105	22.1	0.1	-103	22.3	0.1	-105	21.5	0.1	-102	21.7	0.1	-100	22.2	
	July	0.4	-95	22.0	0.6	-86	21.6	0.6	-85	22.1	0.5	-83	21.8	0.5	-89	21.4	0.4	-90	22.2	0.3	-99	21.8	0.3	-108	22.1	0.2	-102	22.3	0.2	-110	21.5	0.2	-119	21.7	0.2	-112	22.2		
	8	May	0.6	-121	22.0	0.2	-98	21.6	0.3	-94	22.1	0.2	-27	21.8	0.2	-90	21.4	0.2	-89	22.2	0.2	-96	21.8	0.2	-89	22.1	0.2	-89	22.3	0.2	-105	21.5	0.2	-431	21.7	0.1	-110	22.2	
	July	0.5	-96	22.0	0.3	-106	21.6	0.2	-102	22.1	0.2	-112	21.8	0.3	-101	21.4	0.2	-115	22.2	0.2	-103	21.8	0.2	-97	22.1	0.2	-105	22.3	0.3	-113	21.5	0.2	-105	21.7	0.5	-103	22.2		
Blind	4.8	118	22.0	5.6	93	21.6	5.8	87	22.1	5.7	86	21.8	5.7	82	21.4	5.0	90	22.2	5.7	89	22.1	5.6	87	22.3	5.6	93	21.5	5.5	93	21.5	5.5	93	21.7	5.3	93	22.2			

Oxygen in mg/l, redox potential in mV, Temperature in °C, hatched for values outside of tolerance

Annex 3.2: Relevant Parameters and Variables of StoLaM Used in the Lake Caohai Simulation Study

For further details on model structure, equations and parameters see Strauss [29].

Parameter	Value	Unit
Sedimentation rate of detritus	1	$\text{m} \cdot \text{day}^{-1}$
Temperature coefficient of sediment nutrient release	2.4227	–
Sediment phosphate release rate (20 °C)	19*	$\text{mg PO}_4\text{-P} \cdot \text{m}^{-2} \cdot \text{day}^{-1}$
Sediment ammonium release rate (20 °C)	83*	$\text{mg NH}_4\text{-P} \cdot \text{m}^{-2} \cdot \text{day}^{-1}$
Denitrification coefficient (20 °C)	10**	$\text{L} \cdot \text{m}^{-2} \cdot \text{day}^{-1}$
Temperature coefficient of denitrification	1.02	–
Nitrification rate from ammonium to nitrate (20 °C)	0.124	day^{-1}
Temperature coefficient of nitrification	1.065	–
Degradation rate of detritus (at 20 °C)	0.2**	day^{-1}
Remineralisation rate of detrital phosphorus (at 20 °C)	0.2**	day^{-1}
Remineralisation rate of detrital nitrogen (at 20 °C)	0.25**	day^{-1}
Remineralisation rate of DOP (at 20 °C)	0.02	day^{-1}
Remineralisation rate of DON (at 20 °C)	0.02	day^{-1}
Temperature coefficient of remineralization	1.13	–
Light attenuation coefficient for water and DOM	0.3	m^{-1}
Light attenuation coefficient for detritus	0.016	$\text{L} \cdot \text{m}^{-1} \cdot \text{mg ww}^{-1}$
Light attenuation coefficient for blue-green algae	0.012	$\text{L} \cdot \text{m}^{-1} \cdot \mu\text{g Chl-a}^{-1}$
Chlorophyll-a content of blue-green algae	5.13	$\mu\text{g Chla} \cdot \text{mg ww Phyto}^{-1}$
Maximum potential growth rate of <i>Microcystis</i>	1.5	day^{-1}
Maximum natural mortality of <i>Microcystis</i>	0.1	day^{-1}
Respiration rate (at T_{opt}) of <i>Microcystis</i>	0.16	day^{-1}
Sedimentation rate of <i>Microcystis</i> (buoyancy)	–1.38	$\text{m} \cdot \text{day}^{-1}$
Temperature coefficient of respiration	1.13	–
Constant of light limitation (PhAR; Jassby & Platt, 1976) of <i>Microcystis</i>	25	$\text{J} \cdot \text{m}^{-2} \cdot \text{s}^{-1}$
Temperature maximum <i>Microcystis</i>	35	°C
Temperature minimum <i>Microcystis</i>	0	°C
Temperature optimum <i>Microcystis</i>	30	°C
Maximum phosphorus cell quota of <i>Microcystis</i>	0.004	$\text{mg P} \cdot \text{mg ww}$
Minimum phosphorus cell quota of <i>Microcystis</i>	0.0015	$\text{mg P} \cdot \text{mg ww}$
Maximum nitrogen cell quota of <i>Microcystis</i>	0.06	$\text{mg N} \cdot \text{mg ww}$
Minimum nitrogen cell quota of <i>Microcystis</i>	0.011	$\text{mg N} \cdot \text{mg ww}$
Half saturation constant for phosphorus uptake of <i>Microcystis</i>	0.0006	$\text{mg P} \cdot \text{L}^{-1}$

(continued)

(continued)

Parameter	Value	Unit
Half saturation constant for nitrogen uptake of <i>Microcystis</i>	0.03	mg N · L ⁻¹
Maximum phosphorus uptake rate of <i>Microcystis</i>	0.12	mg P · mg ww ⁻¹ · day ⁻¹
Maximum nitrogen uptake rate of <i>Microcystis</i>	0.1	mg N · mg ww ⁻¹ · day ⁻¹

*: Parameter measured in this study. **: Parameter calibrated in this study.

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