water

Article

Monitoring the Efficiency of a Catchment Restoration to Further Reduce Nutrients and Sediment Input into a Eutrophic Lake

Solveig Nachtigall [1] and Christine Heim [2,3,*]

[1] Institute of Biology and Environmental Sciences, University of Oldenburg, Ammerländer Heerstr. 114-118, 26129 Oldenburg, Germany; solveig.nachtigall@uni-oldenburg.de
[2] Geoscience Centre, University of Göttingen, Goldschmidtstr. 3, 37077 Göttingen, Germany
[3] Institute for Geology and Mineralogy, University of Cologne, Zülpicher Str. 49a, 50674 Cologne, Germany
* Correspondence: christine.heim@uni-koeln.de

Abstract: The restoration of eutrophic river and lake ecosystems is an important task that has been conducted in numerous ways and at many locations around the world. However, such improvements of water quality are often temporary, as such ecosystems are dynamic, and restoration measures must be reassessed and modified. The restored catchment of a shallow eutrophic lake, Lake Seeburg, in central Germany, was monitored over a 13-month period. The restoration of the inflowing river a decade earlier included riverbed prolongation, gradient reduction, and the construction of wetlands upstream, which reduced the sediment input and silting up of the lake. As nutrient fluxes in the tributaries were still high, these restoration measures seemed to be insufficiently effective. This study aimed to locate nutrient hotspots and quantify the nutrient balances of the catchment. Nitrogen and phosphorous concentrations, river discharge, hydrochemical parameters (pH, temperature, oxygen concentrations) and turbidity, as a proxy for suspended particulate matter (SPM), were monitored monthly. Our data show that the lake functions as a nitrogen sink, whereas the phosphorous fluxes follow a seasonal trend with the negative balance in winter turning into a positive balance in summer with the onset of cyanobacterial blooms. The release of phosphorous from the wetland throughout the year indicates supersaturation and thus a permanent input of phosphorous into the lake. Consequently, phosphorus loading in the lake is quite high, fostering eutrophication. Furthermore, the very low precipitation rates during the study highlighted that the lake was not only controlled by external nutrient loads but rather was sustained by high internal phosphorous loading. Consequently, the remediation action of creating the wetland to restore the sedimentation trap and nutrient accumulation capacity was not sufficient.

Keywords: lake eutrophication; cyanobacteria; phosphate remobilization

Citation: Nachtigall, S.; Heim, C. Monitoring the Efficiency of a Catchment Restoration to Further Reduce Nutrients and Sediment Input into a Eutrophic Lake. *Water* **2023**, *15*, 3794. https://doi.org/10.3390/w15213794

Academic Editors: Yaoming Ma, Binbin Wang, Lijuan Wen and Jiming Jin

Received: 30 June 2023
Revised: 22 September 2023
Accepted: 24 September 2023
Published: 30 October 2023

1. Introduction

Eutrophication is recognized as a considerable threat to aquatic ecosystems around the globe, with implications for the provision of various ecosystem services such as drinking water, fisheries, and recreation [1]. The nutrient phosphorous plays a crucial role in eutrophication processes in lakes (see [2] for a review). An increased input of anthropogenic phosphorus into surficial waters, in particular, caused by fertilizers in agriculture or sewage water, fosters the growth of cyanobacteria. Eutrophic lakes showing such high phosphorus loads often suffer from recurring cyanobacterial blooms [3,4]. These are a threat to the environment due to their production of toxins (e.g., microcystins [5,6]), which are hazardous for animals and humans. Furthermore, the decay of cyanobacterial blooms sequesters high amounts of oxygen, thereby causing water column hypoxia ($\leq$2 mg L^{-1} O_2) and ammonia release, which are both harmful for benthic organisms (see, for example, [7]). Eutrophication is often linked to the input of suspended particulate matter, which not only causes the silting up of water bodies but also imports nutrients like particulate phosphate [8].

As lake eutrophication is often maintained by internal phosphorous loading [4,9,10], its estimation is also crucial for restoration. Shallow lakes such as our study object, Lake Seeburg, are more prone to eutrophication, and the exchange in nutrients between sediment and the water column is much more extensive than in deep lakes. The main reasons for this are the high sediment surface/water body ratio and the missing thermocline, which normally acts as a physicochemical barrier [11]. Furthermore, internal loading in shallow lakes is not caused by redox-controlled remobilization from iron- or aluminum-bound phosphate [12] or calcium-bound phosphate [13,14] alone. Hence, remobilization is driven by suspension from the sediment due to disturbance by waves, currents or biota, enhanced mineralization through aerobic conditions and elevated temperatures, or an increase in pH through photosynthesis [10,11,15,16].

Successful remediation measures require an understanding of the onset and development of eutrophication in each specific setting. However, this information is often missing, because physicochemical and biological investigations are typically only performed after severe eutrophication and its consequences have already been observed [17,18]. Wetlands and streams are key to lake restoration, as they form the interface between the uplands and adjacent water bodies and control the processes of nutrient transformation and retention [13,19,20].

In this study, the nutrient fluxes in the streams and the wetland in the catchment of Lake Seeburg are located and quantified. This catchment area is of special interest, as some remediation actions have already been conducted in the area. We wanted to investigate whether these remediation actions were still achieving a considerable reduction in sediment and nutrients in the lake's inflow, 15 years after the initial remediation. Nutrient loads in the river Aue are still high, and Lake Seeburg suffers from recurrent cyanobacterial blooms, which even increased during the extreme weather events occurring in recent years [21]. In addition to monitoring the lakes [21], we aimed to identify potential nutrient hotspots within the river system in order to provide guidance for additional remediation measures.

For this task, the fluxes, imports and exports of nutrients are investigated as key elements for promoting lake eutrophication. The emphasis is on nitrate and ortho-phosphate, as they dominate the bioavailable fractions [13]. Turbidity was monitored as a proxy for suspended particulate matter (SPM) [8,22–24], as this is a parameter describing the import of particle-bound nutrients. Our data enable the assessment of the efficiency of remediation actions conducted so far and the identification of several nutrient hotspots that should be considered for further action.

2. Materials and Methods

2.1. Study Area

Lake Seeburg, in central Germany, is a shallow, unstratified lake with a mean depth of about 2 m and a maximum depth of about 4 m, a surface area of 0.89 km^2 and a volume of about 2 million m^3. It originated from a subrosion depression in the Permian "Zechstein". The geology of the catchment area is dominated by Triassic "Buntsandstein" with loess layers and "Muschelkalk". The soils on the carbonate and loess sediment are prone to erosion when the hill slope exceeds 8% [25]. The land use of the catchment area (31.5 km^2) is mainly characterized by agricultural cropland and grassland [25,26]. Various studies have reported the high potential of nutrient exports from agricultural catchments over short time periods dependent on hydrological and meteorological conditions [27–29].

The main tributary of Lake Seeburg is the river Aue, which has its source in the uplands Göttinger Wald. The river was rectified multiple times in the 18th and 19th century, resulting in enhanced flow rates and sediment freights. Formerly, the river Aue led through a lake (Lake Westersee) which silted up and remained as a wetland (Seeanger). In the 1950s, this area was drained through the dislocation of the Aue and turned into agricultural land. First reports about the eutrophication at Lake Seeburg reach back to the 16th century [30] and later, severe and ongoing eutrophication damage of the lake was observed in the 1970s [31]. Regular cyanobacterial blooms occurred since 2005 [24].

The increasing eutrophication trend throughout the last century was reconstructed using phytoplankton biomarkers [32].

Both the river Aue and the Seeanger wetland were restored in 2002, which involved the relocation of the Aue and rewetting of the Seeanger including the creation of a flood plain (sampling site 2) with a maximum capacity of 50 ha (Figure 1). Consequently, the extend of the Seeanger wetland depends strongly on the weather conditions. Redirecting and extending the flow length of the river Aue through the wetland reduced the downhill gradient by 50%. Due to the decreased flow speed, the sediment and nutrient freights were intended to deposit in the wetland area [25,33]. Tributaries into the river Aue, further described as inflow A–D, are a ditch inflow from the federal highway B27 (A) and inflows from the rivers Retlake (B), Egelsee (C) and Friesenbeek (D). These smaller tributaries (especially C and D) drain adjacent agricultural areas, thereby contributing significant amounts of nutrients and sediments. Especially, heavy rainfall events in summer cause drastic temporary runoff from these fertilized agricultural areas and bear high phosphate concentrations (up to 1 t d^{-1}) and sediment freights [34,35]. To reduce such high freight inputs, wooden ramparts along a hill slope and a retention basin were constructed at the inflow D [36].

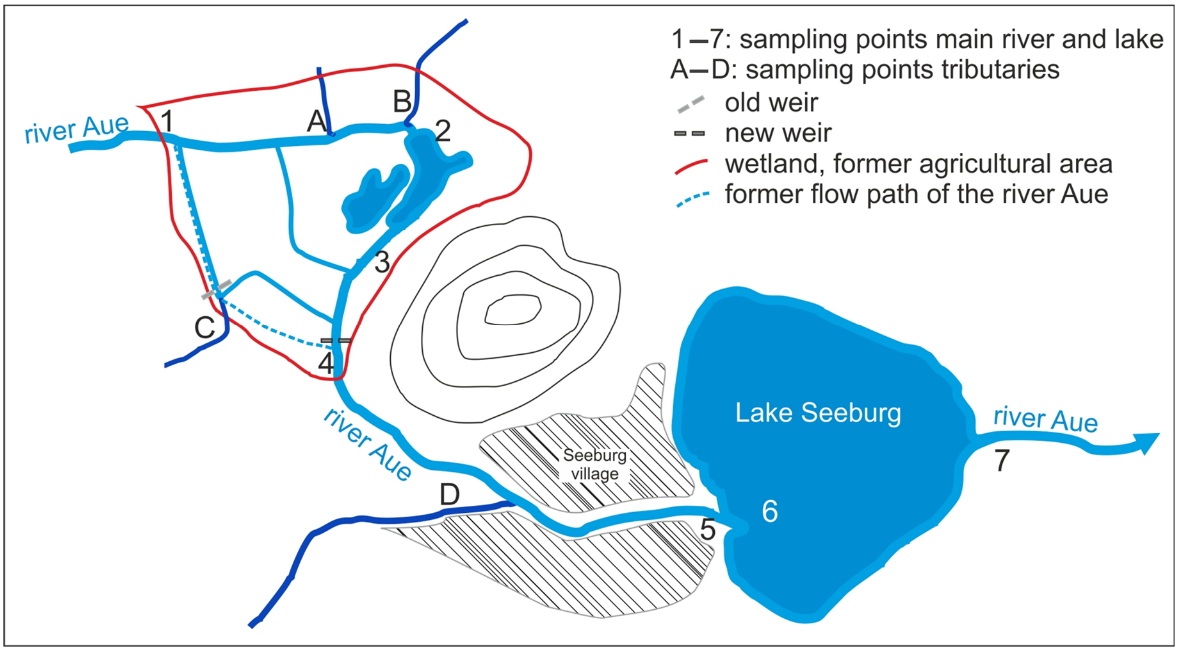

Figure 1. Schematic overview of Lake Seeburg and the restoration area including the old and new flow path. Numbers indicate sampling sites of the river Aue including the floodplain (site 2); letters A–D indicate sampling points of tributaries to the river Aue.

In general, nutrient loads into Lake Seeburg are high with an annual influx of 15 t nitrogen and 3 t phosphorus from both the main tributary Aue and surface runoff [25]. Even though various measures for reducing the nutrient and particle freights have been undertaken, this study is the first to assess the efficiency of the old remediation actions and to identify necessary measures to hamper the progressing eutrophication of this ecosystem.

During the investigated time period from January 2018 to January 2019, the weather parameters were documented in order to estimate their influence on the conditions in Lake Seeburg and its catchment area.

The air temperatures varied between −2 °C in February to 20 °C in July and August. Except for February, all air temperatures were above zero. In February, the ice coverage was observed for the wetland but not for the Lake Seeburg. The annual temperatures were slightly above average temperatures compared to the last 10 years for this region. The precipitation was highest in January 2018 with 93 L/m^2 and lowest in February with

5 L/m^2. During March 2018 until January 2019, the precipitation ranged between 25 and 42 L/m^2, which is roughly half of the expected precipitation for these months. The major wind direction is west–east, similar to the major flow direction of the stream into the lake. All weather data were obtained from the weather station Göttingen (ca. 20 km nearby, https://www.wetterkontor.de/ (accessed on 19 July 2021)).

2.2. Sampling

Monthly sampling was performed at seven stations in the river Aue, the wetland Seeanger, Lake Seeburg and four tributaries (A–D) between January 2018 and January 2019. The lake was analyzed in detail at several different locations within the lake in a previous study. Due to the small size and the shallow water conditions, the wind and the river Aue flowing through the lake cause an overall even distribution of nutrients. Therefore, in this study, the lake was sampled at one representative location.

Water samples were taken in cleaned HDPE Polyvials (Zinsser-Analytics GmbH, Eschborn, Germany) and stored at 4 °C prior analysis; a 100 mL retention sample was stored at −18 °C. Latex gloves were used.

In situ analyses included pH, electrical conductivity, temperature and O_2 concentration, which were measured with the portable multi-parameter instrument Multi 3630 IDS (WTW, Xylem Analytics, Rye Brook, NY, US) near under the water surface.

Turbidity was measured with the nephelometer TURB 350 IR (WTW, Xylem Analytics, Rye Brook, NY, US). Samples were taken slightly below the water surface. Mean values were calculated from triplicate measurements.

2.3. Discharge Measurements

The river discharge was surveyed at 4 measuring stations along the river Aue: ford (1), weir (2), lake inflow (3) and lake outflow (4). For site 1 in April 2018, a discharge value was interpolated because no measurement was conducted at that date. A small current meter C2 (OTT) was used, which was equipped with propeller No. 3 and the digital counter Z400 (OTT). The gauging sections were divided into 10 segments each in which flow velocities were measured, respectively. The two-point method was employed at normal to high water levels and the one-point method was employed at low water levels [37]. At every measuring point, three velocity values were taken (30 s measurement time each). The discharge Q (m^3 s^{-1}) was calculated from the mean velocity values v_m (m s^{-1}) and the cross-sectional area A (m$_2$) was calculated by the formula: Q = $v_m \cdot A$ [37].

2.4. Phosphorous

Ortho-phosphate–P or SRP (soluble reactive phosphorus) was measured photometrically according to EN ISO 6878. The photometer UviLine 9400 (SI Analytics, Xylem Analytics, Rye Brook, NY, US) was used. This study focused on the bio-available phosphorus. For an overview of the total phosphorus concentrations, a set of 50 samples was treated according to section 7.4 in EN ISO 6878.

2.5. Nitrogen

Nitrate–N was measured chromatographically using HPLC. The samples were filtrated (mesh 0.22 µm) prior to analysis. The chromatograph 883 Basic IC plus was used and equipped with the column Metrosep A Supp 5—250/4.0 (Metrohm, Swiss Metrohm Foundation, Herisau, Switzerland).

2.6. Calculation of SPM

SPM (Suspended Particulate Matter) was calculated from turbidity (*Turb* (NTU])) and discharge (Q (m^3 s^{-1})) using the following algorithm by Pfannkuche and Schmidt [22]:

$$SPM = \left(\frac{0.97}{1 + 7.282 \cdot e^{-0.011 \cdot Q}}\right) \cdot Turb + 15.20$$

3. Results

3.1. pH and Temperature

The overall seasonal trend of the river Aue (sampling points 1, 3, 4, 5, 7) showed a maximum pH of 9.0 in summer and a minimum pH of 7.2 in winter. The pH values were consistently alkaline with an average pH 8.1 in the river Aue, pH 8.1 in sampling point 2 (Seeanger wetland) and pH 8.5 at sampling point 6 (Lake Seeburg) (see Table 1).

In the tributaries, the overall pH was lower with a mean of 7.8 (see Table 2). The temperatures follow a seasonal regime with steady temperatures throughout the catchment area during the winter months but strong differences in the summer months with raised temperatures especially in the Seeanger wetland and Lake Seeburg. The average temperatures were 11.1 °C in the river Aue, 11.9 °C at sampling point 2 and 12.9 °C at point 6 (see Table 1). The temperatures in the tributaries were lower with a mean of 9.7 °C (see Table 2).

3.2. Oxygen

In the river Aue (sampling points 1, 3, 4, 5, 7), the oxygen concentrations were highest in winter/spring and lowest in summer with an average of 9.9 mg L^{-1}. The overall concentration at sampling point 2 (wetland Seeanger) was 11.0 mg L^{-1} with temporarily high concentrations of up to 16.0 mg L^{-1} (August 2018) in the summer months. In contrast, temporary values as low as 3.4 mg L^{-1} (August 2018) were measured in the ditch outflow (sampling point 3) of the Seeanger wetland. Similar to point 2, at sampling point 6 (Lake Seeburg), the overall oxygen concentration was 11.6 mg L^{-1} with temporary rises up to 14.2 mg L^{-1} (July 2018) in the summer months (see Table 1). The tributaries showed similar seasonal fluctuations like the river Aue with slightly lower concentrations of 9.2 mg L^{-1} (see Table 2).

3.3. Discharge

The minimum discharge of 40 L s^{-1} in October 2018 occurred at the sampling site 1 of the river Aue, which was exceptionally low and likely distorted by an unnoticed blockage or water extraction in the preceding river section. The maximum of 293 L s^{-1} occurred in April 2018 at sampling site 7 (lake outflow) (see Table 3). Discharge fluctuated seasonally with a higher discharge in winter and spring and lower discharge in summer and autumn. Discharge was strongly controlled by the bigger water bodies, lake and flood plain, as they serve as reservoirs and thus reduce the discharge of the subsequent river sections during low water levels and increase it during high water level conditions. Hence, between stations 1 and 4, as well as between stations 5 and 7, the discharge decreased during the summer season and increased during the winter season. Between station 4 and 5, the discharge increased consistently due to inflow D. The average discharge was 129 L s^{-1}. Based on our data, the annual water influx through the river Aue into Lake Seeburg amounts to 4.5 Mio. m^3 a^{-1}.

3.4. Turbidity

The turbidity values showed an overall seasonal trend with higher values in summer and lower ones in winter. An average value of 6.1 NTU was calculated for the river Aue (sampling points 1, 3, 4, 5, and 7) (see Table 1). At sampling point 2 (Seeanger wetland), the average turbidities of 12.0 NTU were higher, and significantly increased turbidities of up to 37.5 NTU were observed in August 2018. At sampling point 6 (Lake Seeburg), the average turbidities were 11.3 NTU with notably high turbidities of up to 27.0 NTU in August 2018 (see Table 1). In a complementary study, a turbidity value of 186.7 NTU was described for Lake Seeburg a couple of days after the onset of a cyanobacterial bloom in July 2018 [32].

In the tributaries, the average turbidity was 11.1 NTU (see Table 2). Peak turbidities did not follow a clear seasonal trend. The maximum turbidity of 34.6 NTU (November 2018) was observed in tributary C followed by 28.9 NTU (June 2018) in tributary D. Among all sampling sites, these tributaries had by far the highest average turbidities of 18.7 NTU and 16.4 NTU, respectively (see Table 2).

Table 1. Measured parameters in the river Aue, Seeanger wetland and Lake Seeburg.

	Sampling Point	10 January 2018	7 February 2018	12 March 2018	9 April 2018	7 May 2018	11 June 2018	2 July 2018	1 August 2018	3 September 2018	1 October 2018	5 November 2018	10 December 2018	8 January 2019	Ø		s
NO_3-N/mg L^{-1}	1	4.62	4.78	2.59	5.11	5.75	5.57	5.73	5.86	5.41	5.98	5.76	5.94	6.28	5.34	±	0.95
	2	3.97	3.68	3.49	3.65	3.82	4.93	4.79	3.64	4.86	4.02	4.54	4.16	3.82	4.11	±	0.51
	3	3.37	3.83	3.03	2.76	3.40	2.48	2.80	2.18	3.45	3.50	3.63	2.88	4.05	3.18	±	0.55
	4	3.59	3.94	3.41	3.37	3.81	3.66	3.96	3.89	4.26	4.27	4.43	3.35	4.49	3.88	±	0.40
	5	3.88	4.44	4.87	3.56	4.02	3.87	4.01	4.21	4.41	4.38	4.44	3.35	4.52	4.15	±	0.42
	6	1.27	1.47	1.86	1.20	0.45	0.01	0.01	0.20	0.07	0.08	0.16	0.51	1.01	0.64	±	0.64
	7	1.15	1.51	1.58	1.23	0.36	0.01	0.01	0.20	0.11	0.07	0.16	0.57	0.89	0.60	±	0.59
PO_4-P/mg L^{-}	1	0.044	0.054	0.020	0.027	0.044	0.045	0.043	0.041	0.045	0.031	0.023	0.048	0.035	0.039	±	0.010
	2	0.053	0.049	0.027	0.025	0.044	0.049	0.032	0.037	0.047	0.050	0.033	0.088	0.054	0.045	±	0.016
	3	0.102	0.065	0.020	0.038	0.073	0.093	0.056	0.216	0.066	0.040	0.055	0.120	0.051	0.077	±	0.050
	4	0.063	0.057	0.026	0.036	0.075	0.074	0.062	0.130	0.057	0.037	0.040	0.108	0.045	0.062	±	0.029
	5	0.072	0.044	0.040	0.048	0.079	0.085	0.069	0.100	0.064	0.044	0.045	0.110	0.063	0.066	±	0.022
	6	0.034	0.018	0.005	0.005	0.006	0.010	0.014	0.067	0.202	0.226	0.132	0.113	0.059	0.069	±	0.077
	7	0.034	0.018	0.002	0.011	0.004	0.014	0.007	0.032	0.219	0.228	0.134	0.107	0.058	0.067	±	0.080
Turb/NTU	1						5.8	4.2	4.0	9.0	4.8	1.6	4.7	7.6	5.2	±	2.3
	2						6.8	7.2	37.5	14.4	10.1	6.2	7.2	6.3	12.0	±	10.7
	3						3.7	5.6	0.8	3.0	2.4	2.3	5.1	3.9	3.3	±	1.6
	4						8.6	9.1	4.5	4.1	1.3	1.3	4.2	3.1	4.5	±	2.9
	5						14.7	7.3	10.9	10.5	3.9	9.7	4.6	4.4	8.2	±	3.8
	6						7.6	10.0	27.0	18.1	17.9	5.2	2.3	2.0	11.3	±	8.9
	7						6.7	11.2	20.9	10.9	12.8	6.7	3.5	2.3	9.4	±	6.0
pH	1	8.2	8.2	7.9	8.4	8.3	8.4	8.4	8.4	8.3	8.4	8.3	8.4	8.5	8.3	±	0.1
	2	7.7	7.6	7.8	8.3	8.2	8.3	8.5	8.7	8.2	8.0	8.1	8.3	8.2	8.1	±	0.3
	3	7.7	7.8	7.8	8.2	7.9	7.9	8.1	7.6	7.8	7.9	8.1	8.3	7.5	7.9	±	0.2
	4	7.7	7.8	8.0	8.2	7.9	8.1	8.2	8.0	8.0	8.1	8.0	8.2	8.0	8.0	±	0.2
	5	7.7	7.2	7.7	8.4	8.0	8.3	8.2	8.1	8.1	8.2	8.1	8.0	8.1	8.0	±	0.3
	6	7.9	8.1	7.9	8.2	8.4	8.6	8.8	8.8	9.3	9.0	8.0	8.5	8.5	8.5	±	0.4
	7	7.9	8.0	8.2	8.2	8.4	8.6	8.7	8.8	9.0	8.7	7.9	8.5	8.8	8.4	±	0.4
Temp/°C	1	6.0	4.9	7.5	8.8	10.3	13.2	13.5	16.4	13.4	8.9	9.9	6.9	7.0	9.7	±	3.5
	2	3.6	2.5	8.0	13.5	18.6	15.2	21.4	26.7	14.9	9.6	10.0	5.2	5.2	11.9	±	7.3
	3	3.2	2.6	8.2	12.7	15.7	19.2	18.8	22.5	15.4	9.9	9.5	4.9	5.8	11.4	±	6.5
	4	3.5	2.9	7.8	11.3	15.0	17.6	16.7	19.9	14.4	9.6	9.3	5.2	6.0	10.7	±	5.6
	5	3.8	3.1	7.7	13.3	15.6	17.8	14.5	18.9	14.3	9.8	9.7	5.3	6.0	10.8	±	5.3
	6	3.2	2.5	6.4	12.7	18.5	25.6	22.2	25.9	19.2	13.7	9.0	5.1	4.2	12.9	±	8.6
	7	3.2	2.6	5.8	13.9	18.7	26.0	20.9	25.2	18.4	13.3	9.3	4.9	4.2	12.8	±	8.4
O_2/mg L	1	11.9	12.5	11.3	12.0	10.8	10.0	10.0	9.4	10.0	10.9	10.6	11.7	11.8	11.0	±	1.0
	2	10.0	11.4	9.5	14.8	10.4	9.6	13.7	16.0	10.5	9.1	9.7	9.1	9.2	11.0	±	2.3
	3	10.0	11.4	8.7	12.8	6.8	6.1	10.2	3.4	5.8	8.3	8.8	9.9	9.0	8.6	±	2.5
	4	10.0	11.5	9.2	11.7	7.0	8.3	9.8	6.6	7.6	9.3	7.2	10.2	9.8	9.1	±	1.6
	5	11.0	12.2	10.1	14.1	8.1	9.2	8.8	7.6	8.4	10.1	9.1	10.6	10.4	10.0	±	1.8
	6	12.2	13.0	12.1	11.0	12.2	11.7	14.2	11.4	12.7	10.5	6.3	11.6	12.3	11.6	±	1.8
	7	13.5	12.9	12.1	11.1	13.2	11.8	12.5	11.7	7.8	8.4	4.5	12.0	13.0	11.1	±	2.6

Table 2. Measured parameters in the tributaries.

	Sampling Point	10 January 2018	7 February 2018	12 March 2018	9 April 2018	7 May 2018	11 June 2018	2 July 2018	1 August 2018	3 September 2018	1 October 2018	5 November 2018	10 December 2018	8 January 2019	Ø		s
NO_3-N/mg L^{-1}	Inflow A	5.24	5.43	2.49	5.03	4.39	4.84	4.82	5.54	5.42	5.22	4.93	3.55	3.53	4.65	±	0.92
	Inflow B	3.23	2.29	1.88	0.83	0.02	0.02	0.18	0.21	0.06	0.03	0.04	0.38	0.64	0.75	±	1.05
	Inflow C	6.97	8.50	6.27	7.86	8.65	8.00	8.41	8.21	7.83	7.52	6.09	5.37	4.34	7.23	±	1.34
	Inflow D	9.70	8.80	5.82	7.47	7.12	6.44	6.44	5.53	5.94	6.61	6.42	5.68	4.89	6.68	±	1.34
PO_4-P/mg L^{-1}	Inflow A	0.036	0.049	0.052	0.040	0.058	0.080	0.105	0.055	0.079	0.042	0.035	0.036	0.025	0.053	±	0.023
	Inflow B	0.033	0.019	0.023	0.036	0.037	0.048	0.058	0.080	0.047	0.032	0.030	0.039	0.018	0.038	±	0.017
	Inflow C	0.066	0.083	0.029	0.070	0.084	0.099	0.117	0.112	0.113	0.102	0.069	0.095	0.051	0.084	±	0.026
	Inflow D	0.076	0.086	0.079	0.093	0.091	0.095	0.113	0.083	0.116	0.067	0.073	0.106	0.067	0.088	±	0.016
Turb/NTU	Inflow A						3.4	7.5	6.9	3.5	2.5	1.9	2.1	2.4	3.8	±	2.2
	Inflow B						7.9	10.7	3.7	10.3	2.7	1.8	2.2	3.4	5.4	±	3.7
	Inflow C						28.4	23.1	11.9	22.3	12.5	34.6	9.8	7.2	18.7	±	9.8
	Inflow D						28.9	31.0	20.6	22.5	6.7	11.9	5.2	4.3	16.4	±	10.8
pH	Inflow A	7.4	7.6	7.4	7.6	7.6	7.8	8.0	8.1	7.8	8.0	7.9	7.4	7.5	7.7	±	0.3
	Inflow B	7.6	7.6	7.7	8.1	7.9	7.9	7.6	7.5	7.7	7.8	7.9	8.1	7.9	7.8	±	0.2
	Inflow C	7.4	7.5	7.6	7.7	7.7	7.9	8.0	7.9	7.9	8.2	7.9	7.9	7.5	7.8	±	0.2
	Inflow D	7.6	7.5	7.4	8.2	7.9	7.9	7.9	8.0	7.8	8.1	7.9	8.2	7.6	7.8	±	0.2
Temp/°C	Inflow A	5.8	5.0	7.4	8.2	10.0	11.5	12.7	13.5	13.0	11.3	11.1	6.1	8.4	9.5	±	2.9
	Inflow B	3.7	2.0	3.7	8.4	11.0	15.2	12.8	14.2	14.6	8.3	9.6	5.1	5.2	8.8	±	4.6
	Inflow C	5.7	4.5	6.1	8.6	10.5	14.0	12.7	16.8	13.2	10.2	10.1	6.4	6.2	9.6	±	3.8
	Inflow D	5.7	4.2	7.6	12.4	14.1	17.1	16.0	18.8	14.1	9.9	9.6	5.9	5.9	10.9	±	4.9
O_2/mg L^{-1}	Inflow A	9.2	9.0	4.8	9.3	8.2	7.9	9.0	9.2	5.5	9.5	8.6	5.2	8.5	8.0	±	1.7
	Inflow B	11.1	12.0	10.3	10.2	9.1	7.4	4.7	4.0	6.1	8.9	8.8	11.2	10.7	8.8	±	2.6
	Inflow C	11.3	11.8	10.5	10.5	10.4	9.6	10.0	9.0	9.6	10.4	8.9	10.5	10.8	10.2	±	0.8
	Inflow D	10.9	12.5	10.5	13.8	8.2	7.9	8.4	7.8	8.7	10.2	9.0	9.8	10.7	9.9	±	1.8

Table 3. Measured discharge and calculated freights in the river Aue.

	Sampling Point	9 April 2018	7 May 2018	11 June 2018	2 July 2018	1 August 2018	3 September 2018	1 October 2018	5 November 2018	10 December 2018	8 January 2019	Ø		s	Ø
		L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}	L s^{-1}		L s^{-1}	m^3 d^{-1}
Q	1	194	132	117	91	81	94	84	85	125	171	117	±	39	10,133
	4	195	140	119	96	75	86	40	73	177	203	121	±	56	10,423
	5	225	157	136	115	99	95	84	97	195	223	143	±	54	12,314
	7	293	184	127	81	79	60	55	85	187	218	137	±	80	11,834
		kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}	kg d^{-1}		kg d^{-1}	t a^{-1}
NO_3-N	1	85.72	65.49	56.23	44.89	40.93	43.80	43.34	42.51	64.01	92.68	57.96	±	18.82	21.2
	4	56.67	46.24	37.62	32.97	25.39	31.82	14.74	28.07	51.26	78.82	40.36	±	18.45	14.7
	5	69.07	54.51	45.52	39.86	36.12	36.12	31.64	37.26	56.26	87.13	49.35	±	17.66	18.0
	7	31.23	5.79	0.01	0.01	0.07	0.04	0.34	1.18	9.23	16.68	6.46	±	10.31	2.4

Table 3. *Cont.*

	Sampling Point	9 April 2018	7 May 2018	11 June 2018	2 July 2018	1 August 2018	3 September 2018	1 October 2018	5 November 2018	10 December 2018	8 January 2019	Ø		s	Ø
PO_4-P	1	0.46	0.50	0.46	0.34	0.29	0.37	0.23	0.17	0.51	0.52	0.38	±	0.13	0.14
	4	0.62	0.91	0.76	0.52	0.85	0.43	0.13	0.26	1.65	0.79	0.69	±	0.42	0.25
	5	0.95	1.07	1.00	0.69	0.85	0.52	0.32	0.38	1.85	1.21	0.88	±	0.45	0.32
	7	0.29	0.07	0.15	0.05	0.22	1.13	1.08	0.99	1.73	1.09	0.68	±	0.59	0.25
SPM	1			160	123	109	132	114	114	170	237	145	±	44	53
	4			167	135	103	117	53	97	240	273	148	±	75	54
	5			199	160	141	135	113	137	265	303	182	±	69	66
	7			175	115	120	86	79	118	253	292	155	±	79	57

3.5. SPM

The calculated SPM freights (Table 3) of the river Aue (sampling sites 1, 4, 5, 7) follow a seasonal trend with the highest freights in winter and lowest in summer. More interesting is the progression of freights along the course of the river. The average freights accounted for 145 kg d^{-1} at the sampling site 1 of the river Aue, 148 kg d^{-1} behind the weir at site 4, 182 kg d^{-1} at the lake inflow (site 5) and 155 kg d^{-1} at the lake outflow (site 7). Along this course, there was a slight freight increase of 2% between site 1 and site 4, during the flow path through the Seeanger wetland. A flux measurement performed after a rain event (22 June 2018) showed that the freights between sampling site 1 (231 kg d^{-1}) and weir at site 4 (121 kg d^{-1}) decreased by almost 50%. Between site 4 and 5 (lake inflow), an average freight increase of 20% occurred, and accordingly, a freight of 66 t SPM reached Lake Seeburg (site 6) via the river Aue during the year. In Lake Seeburg, the sediment freights were reduced by overall 15%, and thus, during our study, an amount of 12 t SPM was deposited in the lake.

3.6. Ortho-Phosphate–Phosphorous

The average ortho-phosphate–P concentration of the river Aue (sampling points 1, 3, 4, 5, and 7) was 0.062 mg L^{-1} with a maximum of 0.228 mg L^{-1} in October 2018 and a minimum of 0.002 mg L^{-1} in March 2018, both at sampling site 7 (lake outflow) (see Table 1). At sampling site 2 (wetland), the average P concentration was 0.045 mg L^{-1} with a maximum of 0.088 mg L^{-1} in December 2018 and a minimum of 0.025 mg L^{-1} in April 2018. Despite the large variability of concentrations, no overall seasonal trend was observed in the river Aue and the wetland. In contrast, sampling site 6 (lake) showed a clear seasonal trend with low concentrations for most of the year with a minimum of 0.005 mg L^{-1} in March 2018 and high concentrations of up to 0.226 mg L^{-1} (October 2018) in late summer and autumn. The concentrations in the tributaries ranged from 0.018 mg L^{-1} (inflow B, January 2019) to 0.117 mg L^{-1} (inflow C, July 2018) with an average of 0.066 mg L^{-1} and showed a slight increase in summer (see Table 2). The highest average concentrations occurred in the inflows C and D. The total phosphorus measurements of a set of 50 samples indicated that ortho-phosphate–P accounts for about 60% of the total phosphorus in the water.

The calculated P freights (Table 3) of the river Aue follow a trend along the course of the river. The average freights accounted for 0.38 kg d^{-1} at site 1, 0.69 kg d^{-1} at site 4, 0.88 kg d^{-1} at the lake inflow site 5 and 0.68 kg d^{-1} at the lake outflow site 7. There was a freight increase of 80% between site 1 and site 4; thus, approximately 112 kg d^{-1} ortho-phosphate–P was mobilized from the wetland. Between the weir at site 4 and the lake inflow at site 5, an average freight increase of 30% occurred; accordingly, a freight of 0.32 t a^{-1} (corresponds to 0.54 t TP) reached Lake Seeburg through the river Aue over the year. The freights at site 7 (lake outflow) followed a seasonal trend showing that the lake caused a freight decrease from April to August 2018 but a freight increase from September to November 2018. According to this, the freights were reduced by 25% overall in Lake Seeburg, and an amount of 0.07 t a^{-1} ortho-phosphate-P (corresponds to 0.12 t TP) was deposited in the lake over the year.

3.7. Nitrogen

The nitrate–N concentrations of the river Aue ranged between 0.01 mg L^{-1} in the lake outflow (sampling site 7 in in June and July 2018) and 6.28 mg L^{-1} (site 1 in January 2019) with an average of 3.43 mg L^{-1} (Table 1). Overall, the highest averaged nitrogen concentration of 5.34 mg L^{-1} was measured at site 1. At site 2 (wetland), concentrations ranged between 3.49 mg L^{-1} in March 2018 and 4.93 mg L^{-1} in June 2018 with an average of 4.11 mg L^{-1}. No clear seasonal trend was observed in the river Aue and the wetland. In contrast, the lake had lower concentrations of 0.64 mg L^{-1} (average value) and followed a clear seasonal trend with low concentrations of 0.01 mg L^{-1} (June, July 2018) occurring in summer and higher concentrations of up to 1.86 mg L^{-1} (October March 2018) occurring in winter and spring. The concentrations in the tributaries ranged between 0.02 mg L^{-1}

(inflow B in May 2018) and 9.70 mg L^{-1} (inflow D, January 2018) with an average of 4.83 mg L^{-1} (see Table 2). Notably high concentrations were observed in the inflows C (averaged 7.23 mg L^{-1}) and D (averaged 6.68 mg L^{-1}).

The calculated N freights (see Table 3) of the river Aue followed a seasonal trend with the highest freights in winter and lowest in summer. More interesting was the progression of freights along the river course. The average freights accounted for 57.96 kg d^{-1} at site 1, 40.36 kg d^{-1} at site 4, 49.35 kg d^{-1} at site 5 and 6.46 kg d^{-1} at site 7. Thus, there was a freight decrease of 30% observed between site 1 and 4, thus accordingly in the Seeanger wetland (Figure 1). Between site 4 and 5 (lake inflow), the freights increased on average by 20%, and thus, a freight of 18.0 t a^{-1} reached Lake Seeburg through the river Aue over the year. In Lake Seeburg, the freights were reduced by overall 85%; thus, an amount of 15.7 t N was deposited in the lake during the study period.

4. Discussion

4.1. Influence of the Wetland with the Floodplain

The drainage rates between sampling point 1 and 4 show that the floodplains reduced river discharge from August to November due to high evaporation (see Table 3). During the remaining year, the discharge increased between sampling point 1 and 4 due to inflows A and B and due to the high-water volume of the floodplain. The discharge is strongly controlled by the larger water bodies, the lake and the floodplain, as they serve as reservoirs by reducing the discharge of the subsequent river sections during low water levels and increasing it during high water levels. The overall increase in ortho-phosphate–P freights between sampling point 1 (Ø 0.38 kg d^{-1} PO_4-P) and 4 (Ø 0.69 kg d^{-1} PO_4-P) indicates that the wetland did not act as a sink for phosphorous but on the contrary caused its remobilization of approximately 0.11 t a^{-1} PO_4-P. This internal loading derives from phosphorous-enriched sediment, which resulted from the continuously high input from the catchment in the past decades and the agricultural use of the floodplain until the 1970s [31]. In contrast, the consistent reduction in nitrate–N freights between sampling point 1 (Ø 57.96 kg d^{-1} NO_3-N) and 4 (Ø 40.36 kg d^{-1} NO_3-N) during the investigation period shows that the floodplain fulfills its estimated role as a nitrogen sink by holding back approximately 6.42 t a^{-1} NO_3-N.

Between August and October, high turbidity values were analyzed in the floodplain (see Table 1), and biogenic particles originating from decay processes were observed in the water. Therefore, there is some uncertainty in the calculation of SPM freights, as turbidity measurements are reported to underestimate SPM when the share of organic compounds is high [23]. Throughout the monitoring period, the floodplain had no or little influence on the SPM freights (see Figure 1), which were on average similar at sampling point 1 (Ø 145 kg d^{-1}) and sampling point 4 (Ø 148 kg d^{-1}). Nevertheless, on a slightly rainy day, we observed a freight reduction of almost 50%. In this case, the floodplain fulfilled its role as a sediment sink, and thus, a freight reduction may occur only during rain periods when SPM concentrations are above average. However, reliable data on freight and SPM reduction during heavy rains or storm events are lacking due to the dry weather conditions in 2018. Nevertheless, the retention of freight during normal rainfall is also a success, as rain and storm events are reported to be much more critical for nutrient input than "normal" conditions [20].

The overall alkaline pH values are caused by the geology as Triassic limestone "Muschelkalk" crops out in the source region of the river Aue [26,31,38]. The pH seasonality with a summer maximum in the floodplain and in Lake Seeburg is, apart from the temperature related lime–carbonic acid balance [36], ascribed to the carbon dioxide consumption of the highly productive biomass and thus indicating eutrophication. During summer months, the water of the floodplain warmed up quickly with rising solar radiation, which promoted microbiological activity and nutrient turnover. In the course of the rising temperatures in the summer, the oxygen concentration in the river decreased down to 3.4 mg L^{-1} in August (see Table 1) during its passage through the floodplain. Due to warm temperatures, the biodegradation and fouling processes increased in the wetland and thereby consumed most of the oxygen, which in turn increased phosphate release in that area. Leaving the floodplain, the river water was aerated again because of turbulences in the riverbed; thus, the river Aue was properly oxygenated again but phosphate enriched when entering the lake.

Overall, our data show that the floodplain released phosphate to the river throughout the whole investigation period (except for October 2018) and thus transported increased phosphate freights into the lake. However, a consistent retention of nitrate–N freights and reduction in SMP freights on rainy conditions were also observed. Furthermore, the hydrological parameters including oxygen concentration, pH, and water temperatures point toward the ongoing eutrophication of the floodplain.

4.2. Lake-Internal Processes

The annual ortho-phosphate–P input (see Table 3) into the lake via the river Aue (sampling point 5) amounts to approximately 0.32 t a^{-1} PO_4-P, which was much lower than the phosphorous input of 1 t a^{-1} P determined by earlier studies [25]. This suggests that the overall melioration measures which have been taken in 2002 were successful to reduce phosphorous freights into the lake. Nevertheless, the reduction may be partly attributed to low discharge volumes during the extraordinary dry summer months.

Furthermore, our data show a seasonal fluctuation of phosphate freights, which are connected to remobilization and immobilization dynamics from biomass in the water column and within the lake sediment. From April to August, the freight output in the outflow (sampling point 7) was rather low, showing that the lake acted as a phosphate sink, but later, from September until November, it turned into a source due to phosphate remobilization. Overall, the lake had a slightly negative phosphate balance; about 0.07 t a^{-1} PO_4-P was immobilized in the lake over the course of the year. Even though the external input of phosphate was reduced, the persistent internal phosphate loading prevents improvements of the water quality. This phenomenon has also been observed in numerous other studies, where internal loading maintained eutrophication for decades even after external sources had been cut off [10,11].

The nitrate–N input from the river Aue into the lake (sampling point 5) amounted to 18.0 t a^{-1}, which is significantly higher than the nitrogen influx of 13 t a^{-1} N estimated in a previous study [25]. These values point to a deterioration concerning the nitrogen contamination of the catchment area. The overall nitrate–N freight at the lake outflow (sampling point 7) of approximately 2.4 t a^{-1} indicated that more than 15.6 t a^{-1} is held back in the lake and accumulates over the course of a year.

However, the relation of the high nitrate–N inputs with the ortho-phosphate–P concentrations shows that biomass production was limited by nitrogen during the growing season in the summer. Especially between June and October, the N:P ratio in the lake was extremely low, and temporarily, the NO_3-N concentration dropped below the PO_4-P concentration (see Figure 2). As the optimal N:P ratio of phytoplankton is about 16:1 [39], a low N:P ratio favors the growth of cyanobacterial species like *Anabaena flosaque*. This and other cyanobacterial species do not depend on mineralized nitrogen in the water column but prosper with high phosphate concentrations and are therefore now abundant, forming cyanobacterial blooms in Lake Seeburg. Not only the high concentrations but also shifting nutrient ratios are the reasons why phytoplankton is dominated by chlorophyta and diatoms in the spring months [40,41] when the nitrogen availability is high, but this shifts to cyanobacteria during summer when the N:P ratio decreased. This phenomenon seems quite common for phosphorus-dominated eutrophication processes [42].

Similar to processes in the floodplains in the wetland, the rise of pH values during the summer months indicated CO_2 consumption caused by intense biogenic activity in the lake (see Table 1) leading to carbonate precipitation [43]. Furthermore, the high pH fosters the temporary production of toxic ammonia during the summer (for a review, see [40,44]). In the summer, Lake Seeburg warmed up quickly due to its shallowness, promoting photosynthesis as well as microbiological turnover. Both affect the species composition of phytoplankton and hydrophytes in the waterbodies, especially due to the higher temperature maxima of cyanobacteria in contrast to other phytoplankton species [45]. Thus, not only the nutrient shift observed but also the temperature regime of the lake may promote the development of cyanobacteria. During winter and spring, oxygen concentrations in the river Aue and Lake Seeburg were optimal. Whereas the oxygen concentration at the surface remained stable, during the summer months, temporary oxygen depletion occurred close to the water sediment boundary through organic matter decomposition in the uppermost sediment layers [35,40,41,46]. Oxygen saturations measured near the lake bottom ranged from >3 mg L^{-1} and down to <1 mg L^{-1} [21]. Furthermore, lower oxygen concentrations in autumn were especially observed at the outflow due to the phytoplankton decay in the water column and partial mixing during cooling of the lake water in October and November. Both the low oxygen concentrations in the bottom water and the phytoplankton decay led to the extensive release of P, which was measurable in the lake water.

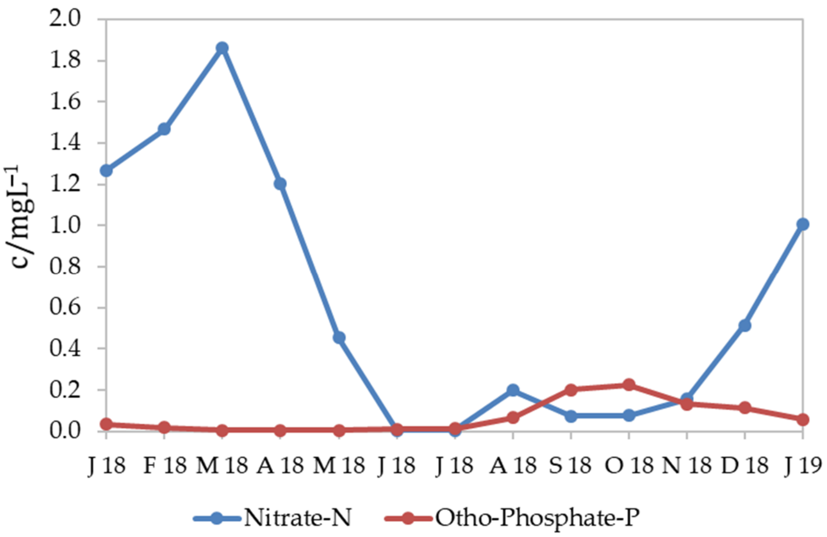

Figure 2. Shift of nitrate–N and ortho-phosphate–P concentrations in Lake Seeburg over the course of the year: January 2018 to January 2019.

The calculated annual water flux into the lake though the river Aue of 4.5 Mio. m^3 a^{-1} (see Table 3) was notably lower than the influx of 9.5 Mio. m^3 a^{-1} [27] or 10 Mio. m^3 a^{-1} [47] assessed in former studies of Lake Seeburg. The difference may be explained by the extremely dry weather conditions of the year 2018. The overall discharge between sampling point 5 (Ø 143 L s^{-1}) and 7 (Ø 137 L s^{-1}) was similar; nevertheless, our discharge data show that the lakes' water balance followed seasonal fluctuation with a positive water balance in April and May and a negative balance from June to January [21,47,48].

At the end of the summer, high turbidities in the lake originated from organic particles and from biogenic degradation of the phytoplankton biomass [49,50]. As a consequence, the cyanobacterial blooms have a negative impact on the macrophyte population by reducing light conditions temporarily from July to September, which was also documented by high chlorophyll concentrations of 119–178 µg L^{-1} in previous studies [46]. SPM in the lake may be underestimated by turbidity measurements due to the high share of organic compounds in the water column, which was already the case in the floodplains of the wetland. The calculated SPM freights at sampling points 5 (Ø 182 kg d^{-1}) and 7 (Ø 155 kg d^{-1}) showed that approximately 10 t a^{-1} of particles remained in the lake over the course of the year (see Table 3; Figure 3). These observations show that the lake is still continuously silting up, and measures that have been taken for particle freight reduction in the catchment area in 2002 are not efficient enough to stop the silting up of Lake Seeburg.

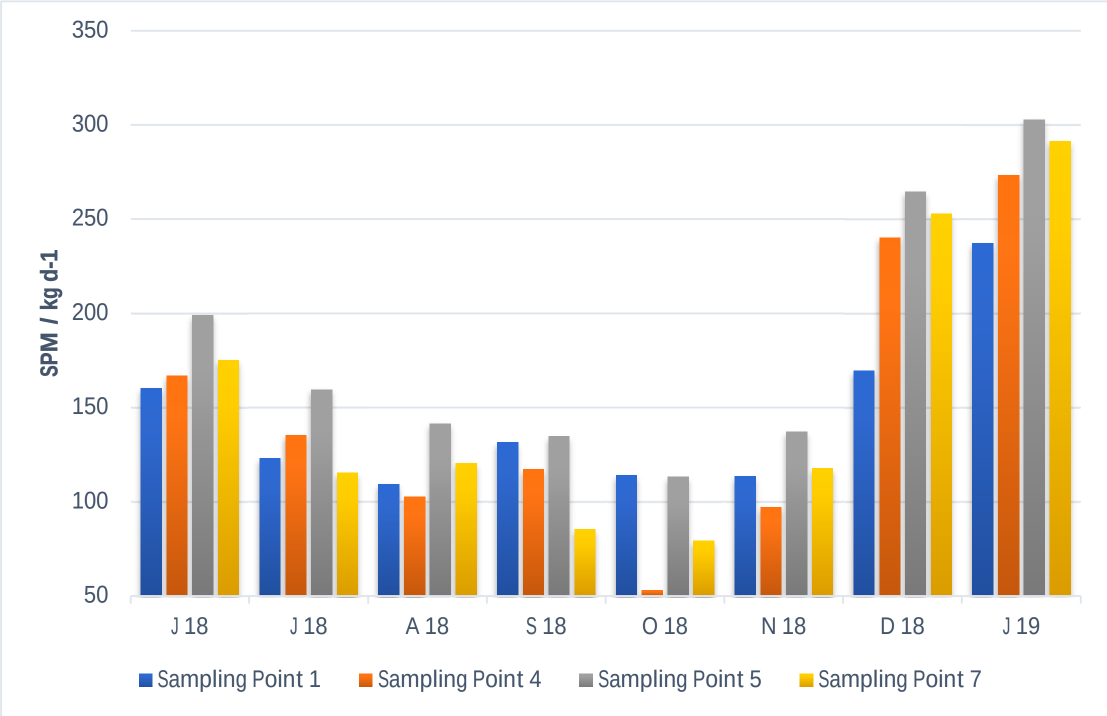

Figure 3. Calculated SPM freights of the river Aue.

Overall, Lake Seeburg is clearly eutrophic in terms of water quality, nutrient availability, and biomass production. In summer and autumn, algal blooms foster phosphate remobilization from the sediment and thus further enhance phosphate availability. The internal loading in the lake prevents the remediation of the ecosystem and is further deteriorated by external phosphate input.

4.3. Nutrient Hotspots of the Catchment Area

The average ortho-phosphate–P concentrations of inflow C (0.084 mg L^{-1}) and D (0.088 mg L^{-1}) were the highest of all sampling points. Thus, the freight increase between sampling point 4 (Ø 0.25 t a^{-1}) and 5 (Ø 0.32 t a^{-1}) can be partly tracked back to inflow D and to the higher discharge at sampling point 5 (see Table 3; Figure 4). The high ortho-phosphate–P concentrations and resulting freights in December were connected to strong rainfalls causing increased discharge. Concomitantly, more phosphate-rich particles were exported from the catchment, and the resuspension of sediments in the shallow floodplain released phosphate from the porewater. This is consistent with other studies on riverine nutrient dynamics that revealed strong links between nutrient exports and changes in hydrological and meteorological conditions [13,27,48]. The average nitrate–N concentrations of inflow C (7.231 mg L^{-1}) and D (6.682 mg L^{-1}) were also higher in comparison to the ones in the river Aue (e.g., sampling point 4: 3.879 mg L^{-1}) and thus raised the nitrate–N overall freights. Further, inflow D contributed to a nitrate–N freight increase of more than 3 t a^{-1} between sampling points 4 and 5 (see Table 3).

The high turbidity values of the tributary inflows C (Ø 18.7 NTU) and D (Ø 16.4 NTU) (see Table 2) raised the turbidity of the river Aue from Ø 3.3 NTU at sampling point 3 to Ø 8.2 NTU at point 5, indicating that their contributions of SPM to the river Aue and eventually to Lake Seeburg were continuously high. The implementation of a dam for freight reduction at inflow D (see also [34,35]) seems therefore not efficient enough. Accordingly, the overall freight increase between sampling points 4 (Ø 148 kg d^{-1}) and 5 (Ø 182 kg d^{-1}) (see Table 3, Figure 3) is mainly caused by the input from the tributary inflow D, which was already identified as a major path of suspended matter in previous studies [34,35]. Nevertheless, the comparison with other studies in agriculturally dominated catchment areas suggests that the SPM contamination of the tributaries of Lake Seeburg is rather low. The turbidities of the Sävjaån, a small river (Ø 4.6 m^3 s^{-1}) in east middle Sweden, range between 1.0 and 1245 NTU with an average of 18.6 NTU [51]; in the smaller Little Bear River in northern Utah (Ø 2.5 m^3 s^{-1}), turbidities varied between 4.3 and 55.1 NTU with an average of 21.1 NTU [52] and for the even smaller Red River in North Dakota (Ø 0.06 m^3 s^{-1}), values ranged between 22.9 and 808.0 NTU with an average of 136.0 NTU [53].

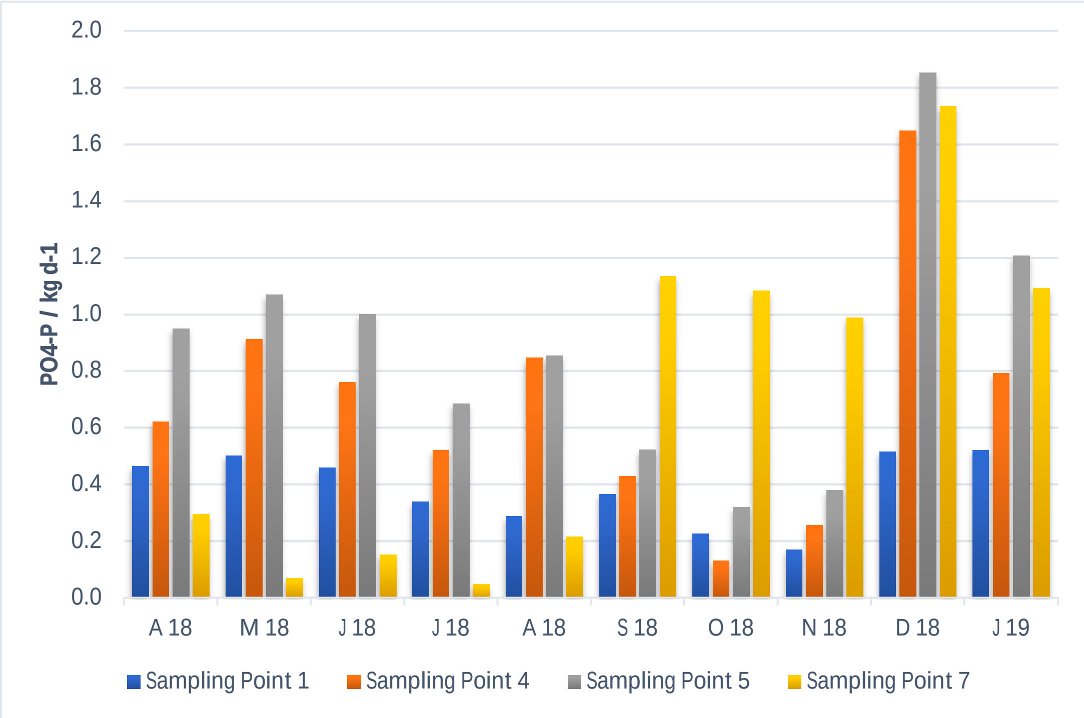

Figure 4. Calculated ortho-phosphate–P freights of the river Aue.

The main external sources for nutrients and particulate matter in the catchment area of Lake Seeburg were inflow C and D, which both drain agricultural areas. The retention basin for freight reduction implemented at inflow D in the past did not sufficiently reduce nutrient and sediment freights from the catchment area into the lake.

Within the river–lake system, three major processes (circles) can be distinguished, which are indicated by the PCA plot (Figure 5) based on the data of Table 3. The first process is an overall background input of nutrients and freights during most of the year (circle on the left side). The second process derives from high SPM freights in the wetland and in the lake during summer (especially in August) due to cyanobacterial blooms (upper right circle). This process is linked to low outputs of NO_3-N of the wetland and the lake outflow. The third process (lower right circle) derives from high PO_4-P values in the lake and the outflow due bloom decay with additional PO_4-P release from the lake sediment in September and October.

According to the water framework directive, investigations of small water bodies and tributaries are essential for successful remediation programs. Our monitoring enabled the efficiency assessment of remediation actions conducted so far and pinpoint hotspots for further remediation plans. For the remediation of a eutrophic lake ecosystem, a reduction in the phosphorous input is crucial [4]. In other case studies, a reduction in the phosphorous input measures by sedimentation basins or constructed wetlands were successful; however, in our case study of the Lake Seeburg catchment area, the constructed wetland acts even like a phosphate source. Furthermore, even after input reduction, the response time of the lakes to recover might be long due to internal loading [4]. A possibility to prevent internal loading in Lake Seeburg is the desilting of wetland and the lake. Removing the sediment maybe helpful to avoid silting up, but the removal and the disposal of sediments can be problematic and expensive due to, e.g., high heavy metal concentrations in the sediments and special treatments to avoid damage of the inhabiting flora and fauna [54].

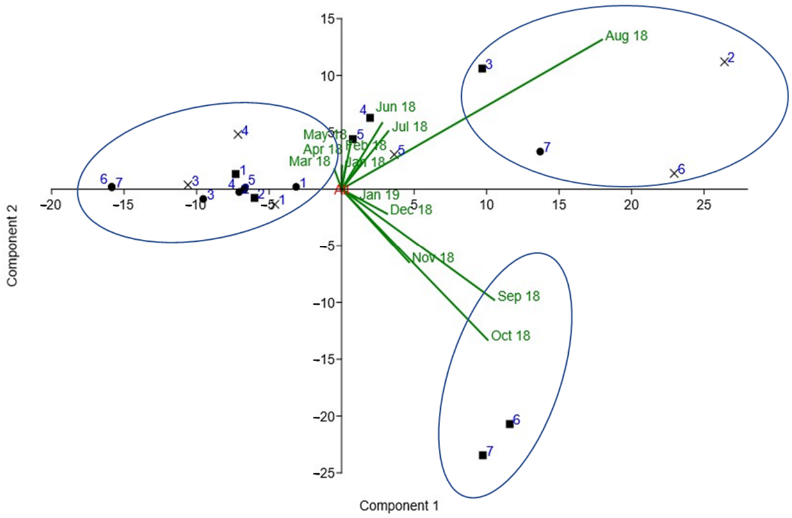

Figure 5. PCA plot based on the data of Table 3. NO_3-N values correspond to black dots, PO_4 -P values are black squares and the black Xs correspond to SPM (freights). The PCA plot determines hotspots in terms of nutrients and SPM with the correlating time.

Another option is the precipitation of phosphorous with flocculants. Phosphate precipitation experiments with LMB (lanthanum-modified bentonite, Bentophos), Fe chlorides and Al chlorides, that have been conducted using the water of Lake Seeburg in the past, were not successful [55]. Several studies show that the efficiency of LMB is hampered by humic substances in the water column [56,57], which may be a reason for their insufficiency at Lake Seeburg. Furthermore, case studies about phosphate precipitation agents usually neglect long-term ecological consequences [58–60]. This emphasizes that geo-engineering measures must be site specific, as hydrological, geochemical, and biological site traits can be confounding factors for their success [61]. Consequently, further studies are necessary to determine the most efficient but sustainable method for the reduction in sedimentary phosphate within the wetland and Lake Seeburg.

5. Conclusions

Our monitoring study of Lake Seeburg and the catchment area over 13 months enabled the identification of major nutrient hotspots and the assessment of locations where additional or new remediation measures are necessary. In 2018, the lake was in an overall eutrophic state but with a strong seasonal fluctuation. From January 2018 to February 2018, the lake was mesotrophic; then, it turned into an oligotrophic status from March 2018 to July 2018, but in August 2018, Lake Seeburg was eutrophic and further changed to a polytrophic status between September and October 2018. Lastly, the lake status declined to eutrophic again from November 2018 to January 2019.

The river Aue and the tributaries as well as the Seeanger wetland show a clear overall nitrogen contamination. Very low nitrogen contaminations were observed in the tributary inflow B and the lake outflow. In contrast, raised contaminations were found in the tributary inflows C and D where high nitrogen concentrations are caused by diffuse sources in the agriculturally dominated catchment.

Nitrogen freights were constantly reduced in the Seeanger wetland as well as in Lake Seeburg, which means that both water bodies act as nitrogen sinks. Over the year, a total of 6.4 t nitrate–N was deposited in the Seeanger wetland, and a total of 15.6 t was deposited in Lake Seeburg. Temporary nitrogen limitations in the lake during the summer months lead to a shift in the phytoplankton community, which is dominated by diatoms and chlorophyta in spring but then turns to a cyanobacteria-dominated community.

Overall, the ecosystem is clearly contaminated with phosphorous (ortho-phosphate–P), which is particularly accumulated in the Seeanger wetland and in Lake Seeburg sediments. In late summer, phosphorous remobilization took place in both the floodplains of the wetland and in Lake Seeburg, which raised the P concentration by internal loading, turning both water bodies into phosphorous sources. An annual amount of approximately 0.11 t PO_4-P was remobilized from the wetland; in Lake Seeburg, a total of about 0.07 t a^{-1} was deposited. Therefore, remediation actions are not only necessary for the river aue and the Seeanger wetland but also within Lake Seeburg in order to reduce internal P loading and to avoid annual cyanobacterial blooms. All in all, our study contributed to a better overview over hotspots in the river Aue, the catchment area and the adjacent lake system, which helped to focus on site-specific remediation actions.

Author Contributions: Conceptualization, C.H. and S.N.; methodology, C.H. and S.N.; formal analysis, C.H. and S.N.; investigation, C.H. and S.N.; resources, C.H. and S.N.; writing—original draft preparation, S.N.; writing—review and editing, C.H. and S.N.; visualization, S.N; supervision, C.H.; project administration, C.H.; funding acquisition, C.H. All authors have read and agreed to the published version of the manuscript.

Funding: This research was financially supported by the Nature Conservation Authority of Göttingen County (contract number 701034002-48).

Data Availability Statement: Not applicable.

Acknowledgments: We thank four journal reviewers for their constructive comments that helped us improve the original manuscript. Angelika Mroncz and Hannes Schwarze contributed nutrient and freight data obtained in their B.Sc. projects. We also thank and Birgit Röring for analytical support in the lab.

Conflicts of Interest: The authors declare no conflict of interest.

References

1. European Commission. The Water Framework Directive and the Floods Directive: Actions towards the "Good Status" of EU Water and to Reduce Flood Risks. 2015. Available online: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A52015DC0120 (accessed on 20 September 2023).
2. Schindler, D.W. The dilemma of controlling cultural eutrophication of lakes. *Proc. R. Soc. B Biol. Sci.* **2012**, *279*, 4322–4333. [CrossRef]
3. Anderson, D.; Glibert, P.; Burkholder, J. Harmful algal blooms and eutrophication: Nutrient sources, compositions, and consequences. *Estuaries* **2002**, *25*, 704–726. [CrossRef]
4. Schindler, D.W.; Carpenter, S.R.; Chapra, S.C.; Hecky, R.E.; Orihel, D.M. Reducing phosphorus to curb lake eutrophication is a success. *Environ. Sci. Technol.* **2016**, *50*, 8923–8929. [CrossRef]
5. He, X.; Liu, Y.L.; Conklin, A.; Westrick, J.; Weavers, L.K.; Dionysiou, D.D.; Lenhart, J.J.; Mouser, P.J.; Szlag, D.; Walker, H.W. Toxic cyanobacteria and drinking water: Impacts, detection, and treatment. *Harmful Algae* **2016**, *54*, 174–193. [CrossRef]
6. Maliaka, V.; Lürling, M.; Fritz, C.; Verstijnen, Y.J.M.; Faassen, E.J.; van Oosterhout, F.; Smolders, A.J.P. Interannual and Spatial Variability of Cyanotoxins in the Prespa Lake Area, Greece. *Water* **2021**, *13*, 357. [CrossRef]
7. Maar, M.; Timmermann, K.; Petersen, J.K.; Gustafsson, K.E.; Storm, L.M. A model study of the regulation of blue mussels by nutrient loadings and water column stability in a shallow estuary, the Limfjorden. *J. Sea Res.* **2010**, *64*, 322–333. [CrossRef]
8. Rügner, H.; Schwientek, M.; Beckingham, B.; Kuch, B.; Grathwohl, P. Turbidity as a proxy for total suspended solids (TSS) and particle facilitated pollutant transport in catchments. *Environ. Earth Sci.* **2013**, *69*, 373–380. [CrossRef]
9. Sinke, A.J.C. *Phosphorus Dynamics in the Sediment of a Eutrophic Lake*; Wageningen University and Research: Wageningen, The Netherlands, 1992; ISBN 9054850361.
10. Horppila, J.; Holmroos, H.; Niemistö, J.; Massa, I.; Nygrén, N.; Schönach, P.; Tapio, P.; Tammeorg, O. Variations of internal phosphorus loading and water quality in a hypertrophic lake during 40 years of different management efforts. *Ecol. Eng.* **2017**, *103*, 264–274. [CrossRef]
11. Søndergaard, M.; Jensen, J.P.; Jeppesen, E. Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia* **2003**, *506–509*, 135–145. [CrossRef]
12. Hupfer, M.; Lewandowski, J. Oxygen controls the phosphorus release from lake sediments—A long-lasting paradigm in limnology. *Int. Rev. Hydrobiol.* **2008**, *93*, 415–432. [CrossRef]

13. Dunne, E.J.; Reddy, K.R. Phosphorus biogeochemistry of wetlands in agricultural watersheds. In *Nutrient Management in Agricultural Watersheds: A Wetland Solution*; Wageningen Academic Publishers: Wageningen, The Netherlands, 2004; pp. 105–119.
14. Lü, C.; He, J.; Zuo, L.; Vogt, R.D.; Zhu, L.; Zhou, B.; Mohr, C.W.; Guan, R.; Wang, W.; Yan, D. Processes and their explanatory factors governing distribution of organic phosphorous pools in lake sediments. *Chemosphere* **2016**, *145*, 125–134. [CrossRef] [PubMed]
15. Carvalho, L.; Beklioglu, M.; Moss, B. Changes in a deep lake following sewage diversion—A challenge to the orthodoxy of external phosphorus control as a restoration strategy? *Freshw. Biol.* **1995**, *34*, 399–410. [CrossRef]
16. Tammeorg, O.; Horppila, J.; Laugaste, R.; Haldna, M.; Niemistö, J. Importance of diffusion and resuspension for phosphorus cycling during the growing season in large, shallow Lake Peipsi. *Hydrobiologia* **2015**, *760*, 133–144. [CrossRef]
17. Naeher, S.; Smittenberg, R.H.; Gilli, A.; Kirilova, E.P.; Lotter, A.F.; Schubert, C.J. Impact of recent lake eutrophication on microbial community changes as revealed by high resolution lipid biomarkers in Rotsee (Switzerland). *Org. Geochem.* **2012**, *49*, 86–95. [CrossRef]
18. Bhagowati, B.; Ahamad, K.U. A review on lake eutrophication dynamics and recent developments in lake modeling. *Ecohydrol. Hydrobiol.* **2019**, *19*, 155–166. [CrossRef]
19. Reddy, K.R.; Kadlec, R.H.; Flaig, E.; Gale, P.M. Phosphorus retention in streams and wetlands: A review. *Crit. Rev. Environ. Sci. Technol.* **1999**, *29*, 83–146. [CrossRef]
20. Tang, J.L.; Zhang, B.; Gao, C.; Zepp, H. Hydrological pathway and source area of nutrient losses identified by a multi-scale monitoring in an agricultural catchment. *Catena* **2008**, *72*, 374–385. [CrossRef]
21. Zeman-Kuhnert, S.; Thiel, V.; Heim, C. Effects of Weather Extremes on the Nutrient Dynamics of a Shallow Eutrophic Lake as Observed during a Three-Year Monitoring Study. *Water* **2022**, *14*, 2032. [CrossRef]
22. Neukermans, G.; Ruddick, K.; Loisel, H.; Roose, P. Optimization and quality control of suspended particulate matter concentration measurement using turbidity measurements. *Limnol. Oceanogr. Methods* **2012**, *10*, 1011–1023. [CrossRef]
23. Pfannkuche, J.; Schmidt, A. Determination of suspended particulate matter concentration from turbidity measurements: Particle size effects and calibration procedures. *Hydrol. Process.* **2003**, *17*, 1951–1963. [CrossRef]
24. Stutter, M.; Dawson, J.J.C.; Glendell, M.; Napier, F.; Potts, J.M.; Sample, J.; Vinten, A.; Watson, H. Evaluating the use of in-situ turbidity measurements to quantify fluvial sediment and phosphorus concentrations and fluxes in agricultural streams. *Sci. Total Environ.* **2017**, *607–608*, 391–402. [CrossRef]
25. NLWKN. *Wasserrahmenrichtlinie Band 3. Seeburger See. Leitfaden Maßnahmenplanung Oberflächengewässer. Teil B Stillgewässer. Anhang II—Seeberichte*; NLWKN: Hannover, Germany, 2010.
26. NIBIS. LBEG NIBIS Kartenserver. Niedersächsisches Bodeninformationssystem. Available online: https://nibis.lbeg.de/cardomap3 (accessed on 30 July 2018).
27. Blaen, P.J.; Khamis, K.; Lloyd, C.; Comer-Warner, S.; Ciocca, F.; Thomas, R.M.; MacKenzie, A.R.; Krause, S. High-frequency monitoring of catchment nutrient exports reveals highly variable storm event responses and dynamic source zone activation. *J. Geophys. Res. Biogeosciences* **2017**, *122*, 2265–2281. [CrossRef]
28. De Schepper, G.; Therrien, R.; Refsgaard, J.C.; Hansen, A.L. Simulating coupled surface and subsurface water flow in a tile-drained agricultural catchment. *J. Hydrol.* **2015**, *521*, 374–388. [CrossRef]
29. Li, H.; Sivapalan, M.; Tian, F.; Liu, D. Water and nutrient balances in a large tile-drained agricultural catchment: A distributed modeling study. *Hydrol. Earth Syst. Sci.* **2010**, *14*, 2259–2275. [CrossRef]
30. Cyppull, B.; Küntzel, T. *Durch Land und Zeit*; Landschaftsverband Südniedersachsen, Ed.; Verlag Jörg Mitzkat: Holzminden, Germany, 2005.
31. Streif, H. *Limnogeologische Untersuchung des Seeburger Sees (Untereichsfeld)*; Bundesanstalt für Bodenforschung: Hannover, Germany, 1970.
32. Zeman-Kuhnert, S.; Öztoprak, M.; Heim, C.; Thiel, V. Reconstructing eutrophication trends of a shallow lake environment using biomarker dynamics and sedimentary sterols. *Org. Geochem.* **2023**, *177*, 104555. [CrossRef]
33. Göttingen, L. *Seeanger und Aue. Renaturierung Eines Ehemaligen Sees und Eines Baches im Untereichsfeld*; Landkreis Göttingen, Germany, 1999.
34. Römer, W. Phosphatfrachten und potentielle Phosphataustäge (Modellberechnungen) aus dem Einzugsgebiet Friesenbeek in den Seeburger See. Master's Thesis, Georg-August University, Göttingen, Germany, 2009.
35. Römer, W. Vom Acker in den Seeburger See. *Land Forst* **2008**, *31*, 30.
36. Dießel, C. Umsetzung des Seeburger Seekonzeptes in der Flurbereinigung Seeburg. Amt für regionale Landesentwicklung Braunschweig. Available online: https://www.arl-bs.niedersachsen.de/startseite/foerderung_projekte/flurbereinigung/im_landkreis_gottingen/flurbereinigung-seeburg-150390.html2013 (accessed on 30 October 2018).
37. Morgenschweis, G. *Hydrometrie. Theorie und Praxis der Durchflussmessung in Offenen Gerinnen*, 2nd ed.; Springer: Berlin/Heidelberg, Germany, 2018; ISBN 9783662553138.
38. Arp, G.; Hoffmann, V.; Seppelt, S.; Riegel, W. Exkursion 6: Trias und Jura von Göttingen und Umgebung. In Proceedings of the 74. Jahrestagung der Paläontologischen Gesellschaft, Göttingen, Germany, 2–8 October 2004; pp. 147–192.
39. Falkowski, P.G.; Davis, C.S. Natural proportions. *Nature* **2004**, *431*, 131. [CrossRef]
40. Bäthe, J.; Coring, E.; Curdt, T.; Kleinfeldt, H. *Limnologische Untersuchungen in Stehenden Gewässern Niedersachsens 2017. Gartower See—Schladener Kiessee—Seeburger See (Phytoplankton und chemisch-physikalische Parameter)*; NLWKN: Hannover, Germany, 2018.

41. Bäthe, J.; Coring, E.; Dietrich, N.; Wegener, M.; Wilbertz, M. *Bericht zum Untersuchungsauftrag "Limnologische Untersuchungen in stehenden Gewässern Niedersachsens 2014"*; NLWKN: Hannover, Germany, 2015.
42. Hsieh, C.H.; Ishikawa, K.; Sakai, Y.; Ishikawa, T.; Ichise, S.; Yamamoto, Y.; Kuo, T.C.; Park, H.D.; Yamamura, N.; Kumagai, M. Phytoplankton community reorganization driven by eutrophication and warming in Lake Biwa. *Aquat. Sci.* **2010**, *72*, 467–483. [CrossRef]
43. Gierlowski-Kordesch, E.H. Chapter 1 Lacustrine Carbonates. *Dev. Sedimentol.* **2010**, *61*, 1–101. [CrossRef]
44. Collos, Y.; Harrison, P.J. Acclimation and toxicity of high ammonium concentrations to unicellular algae. *Mar. Pollut. Bull.* **2014**, *80*, 8–23. [CrossRef]
45. Morscheid, H.; Fromme, H.; Krause, D.; Kurmayer, R.; Morscheid, H.; Teubner, K. *Toxinbildende Cyanobakterien (Blaualgen) in Bayerischen Gewässern*; Bayerisches Landesamt für Umwelt: Augsburg, Germany, 2006; Volume 125, ISBN 9783940009081.
46. Coring, E.; Bäthe, J. *Abschlussbericht zum Untersuchungsauftrag „Ökologisch-limnologische Untersuchungen am Seeburger See und Ausgewählten Gewässern in Seinem Einzugsgebiet"*; EcoRing: Hardegsen, Germany, 2007.
47. Hartmann, R. *Abschlussbericht der Hydrochemischen Untersuchungen im Bereich des Seeburger Sees im Landkreis Göttingen und Dessen Zuflüsse*; Landkreis Göttingen, Germany, 2007.
48. Hellmann, H. *Qualitative Hydrologie. Wasserbeschaffenheit und Stoff-Flüsse*; Gebrüder Borntraeger: Berlin, Germany, 1999.
49. Wetzel, R. *Limnology: Lake and River Ecosystems*; Gulf Professional Publishing: Houston, TX, USA, 2001; ISBN 9780127447605.
50. Huang, W.; Chen, R.F. Sources and transformations of chromophoric dissolved organic matter in the Neponset River Watershed. *J. Geophys. Res. Biogeosciences* **2009**, *114*, 1–14. [CrossRef]
51. Lannergård, E. *Potential for Using High Frequency Turbidity as a Proxy for Total Phosphorus in Sävjaån*; Department of Aquatic Sciences and Assessment: Uppsala, Sweden, 2016.
52. Jones, A.S.; Stevens, D.K.; Horsburgh, J.S.; Mesner, N.O. Surrogate Measures for Providing High Frequency Estimates of Total Suspended Solids and Total Phosphorus Concentrations. *JAWRA J. Am. Water Resour. Assoc.* **2011**, *47*, 239–253. [CrossRef]
53. Ryberg, K.R. *Continuous Water-Quality Monitoring and Regression Analysis to Estimate Constituent Concentrations and Loads in the Red River of the North, Fargo, North Dakota, 2003-05*; US Department of the Interior, US Geological Survey: Reston, VA, USA, 2006.
54. Wolter, K.-D.; Köhler, G. *Bilanzierung des Dümmers*; NLWKN: Hannover, Germany, 2012.
55. Steingrobe, B. *Vorarbeiten für Restaurierungsmaßnahmen am Seeburger See 2008/2009—Abschlussbericht*; Georg-August-Universität Göttingen: Göttingen, Germany, 2009.
56. Lürling, M.; Waajen, G.; Van Oosterhout, F. Humic substances interfere with phosphate removal by lanthanum modified clay in controlling eutrophication. *Water Res.* **2014**, *54*, 78–88. [CrossRef] [PubMed]
57. Stefani, F.; Finsterle, K.; Winfield, I.J.; D'Haese, P.; Reitzel, K.; Tartari, G.; Crosa, G.; Yasseri, S.; Copetti, D.; Lürling, M.; et al. Eutrophication management in surface waters using lanthanum modified bentonite: A review. *Water Res.* **2015**, *97*, 162–174. [CrossRef]
58. Dithmer, L.; Nielsen, U.G.; Lundberg, D.; Reitzel, K. Influence of dissolved organic carbon on the efficiency of P sequestration by a lanthanum modified clay. *Water Res.* **2016**, *97*, 39–46. [CrossRef] [PubMed]
59. Hupfer, M.; Reitzel, K.; Kleeberg, A.; Lewandowski, J. Long-term efficiency of lake restoration by chemical phosphorus precipitation: Scenario analysis with a phosphorus balance model. *Water Res.* **2016**, *97*, 153–161. [CrossRef]
60. Noyma, N.P.; de Magalhães, L.; Furtado, L.L.; Mucci, M.; van Oosterhout, F.; Huszar, V.L.M.; Marinho, M.M.; Lürling, M. Controlling cyanobacterial blooms through effective flocculation and sedimentation with combined use of flocculants and phosphorus adsorbing natural soil and modified clay. *Water Res.* **2016**, *97*, 26–38. [CrossRef]
61. Lürling, M.; Mackay, E.; Reitzel, K.; Spears, B.M. Editorial—A critical perspective on geo-engineering for eutrophication management in lakes. *Water Res.* **2016**, *97*, 1–10. [CrossRef]