ORIGINAL ARTICLE

Management implications following the reconstruction of the small and shallow Lake Mustijärv (Estonia)

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ABSTRACT

Lake Mustijärv was reconstructed by sediment removal following an almost complete siltation. Here we evaluate challenges and opportunities for the management of the lake. We focused on the stream discharging to the lake (i.e., external loading), sediment retention in accumulation basins (i.e., internal processes) as well as the ecosystem-level response to stressors based on biological variables (phyto- and zooplankton, macrophytes, fishes and zoobenthos). Urban and agricultural inputs elevated ammonium and total phosphorus concentrations in the lake, challenging lake reconstruction efforts. Sediment transport highly increased the risk of faster filling of the lake, associated with upstream streambed excavation. Sediments trapped at the accumulation basins release nutrients that enhance eutrophication. We, however, observed a rapid recovery in fish, macrophytes, and zoobenthos, despite the significant disturbances. Lake Mustijärv is in eutrophic condition, reflected by phytoplankton (*Pseudanabaena, Closteriopsis* dominance) and zooplankton (*Keratella*, *Polyarthra* dominance) composition. To improve lake water quality will require controlling external nutrient inputs, underlining the importance of better coordinated activities between the local (lake restoration) and regional (catchment use) scales.

INTRODUCTION

As recipients of watershed inputs, lakes accumulate sediments and nutrients resulting from human activities. A major problem arises when lakes become saturated with discharged nutrients. This leads to "eutrophication" which is often symptomized by unpleasant cyanobacterial surface scums, decreased water clarity, and excessive production of organic matter, leading to increased oxygen depletion (Le Moal *et al.*, 2019). These effects lead to the loss of habitat, biodiversity, and amenity value, and can pose human health risks. Oxygen depletion and high primary production-induced elevated pH can further exacerbate eutrophication by promoting nutrient release from sediments (internal loading; Steinman and Spears, 2020). To counter these negative effects, lakes require managing to ensure their functioning and multiple ecosystem services they provide. Measures targeting both external and internal nutrient loading reduction are often needed (Lürling *et al.*, 2020, 2024; Tammeorg *et al.*, 2023).

If left unmanaged, eutrophicating lakes can become increasingly shallow and eventually may fill in (Le Moal et *al.*, 2019). The process can be speeded up by the transport of the nutrient-rich particulate matter from the catchment. Decreasing depth through deposition of nutrient-rich particulate matter from the catchment is common fate of small, shallow lakes (Yao *et al.*, 2022; IGB, 2023). It is noteworthy that these lakes contribute considerably to the world's freshwater resources, providing a number of services (*e.g.*, drinking and irrigation water supplies, recreation and fishing habitats; Downing *et al.*, 2006).

Lake Mustijärv' volume has almost entirely filled in during 30 years of high human impact. Recently, the lake has undergone the complete sediment removal, resulting in a lake mean depth of two meters. The deeper sediment accumulation areas (close to stream inflow and in the middle of the lake) were crafted to entrap nutrient-rich particles (Zhang *et al.*, 2022) discharged by the single inflow. Following sediment removal (also removed the source of the internal nutrient loading), the lake's water quality was assumed to be largely influenced by external nutrient loading. Sediment studies of the lake a year after sediment removal identified the formation of a 40-50 cm thick silt layer in the sediment accumulation areas (Kiani *et al.*, 2020) and of about 10 cm in the remaining areas. The accumulation areas released phosphorus (P) at a rate consistent with that reported for anoxic surfaces in hypertrophic lakes (Kiani *et al.*, 2020). The sediment pool was rebuilt as a result of exceptionally high external nutrient loading, promoting eutrophication and accelerated loss of depth.

There are no studies reporting lake ecosystem dynamics following full reconstruction. Still, learning lessons from such cases may help to map and exclude the potential risks in the similar cases. In the current study, we analysed the Lake Mustijärv's response to external and internal nutrient loading following lake reconstruction in 2018 based on monthly monitoring of nutrients and plankton from June to October, as well as on individual surveys of macrophytes, fishes and zoobenthos. Given the high importance of the external nutrient loading, we investigated the longitudinal changes of the upstream nutrient concentrations (in August) to estimate the impact of the human activities in the catchment. Finally, we estimated the nutrient retention ability of the lake (based on measurements at three timepoints, June, August, October in the inlet and outlet) to evaluate the role of sediment accumulation basins.

METHODS

Study site

Lake Mustijärv (since 2016 after removal of 7500 m^3: surface area 1 ha, mean depth 2 m, maximum depth 3.8 m, length 257 m, largest width 58 m) is an artificial waterbody located 1 km west of Viljandi, Estonia (58°21'55.8»N 25°32'32.6»E, 65 m above sea level; Fig. 1A). The lake was established for water storage by expanding the Kurika stream in 1984-1986. The Kurika stream entering the lake from south-east carries most of the nutrients to the lake (Kiani *et al.*, 2020). There are two other lakes, Mandle and Riiska-Mandle, upstream of Lake Mustijärv (closest being 1.5 km streamwise) on the Kurika stream. The catchment area upstream of the lake is mostly used as cattle pastures and for annual crop production with conventional tillage. Until the mid-1990s, Lake Mustijärv has received effluents from local dairy and untreated urban stormwater. Since then, urban stormwater has been the main pollution source in addition to agricultural nutrient loading. By 2015, the lake was heavily siltated, and the open water area was only 0.3 ha (0.4-1.2 m of silt overlaid by 0.1-0.8 m of water; Fig. 1A). Overwhelming silt piles were covered with willows, bulrush and the common reed (Kiani *et al.*, 2020). There are no water quality data characterizing the lake before reconstruction, with the exception of a few samples collected in November 2015 from the inlet and outlet of the lake.

The local community initiated lake restoration to re-establish the waterbody for recreational use and amenity. In 2016, the lake (mean depth <0.8 m) was fully excavated (about 7,500 m^3 of sediment). Two deeper sediment accumulation areas were created in the lake, one close to the inflow (IF, 3 m deep; Fig. 1A) and the other in the central part of the lake (CE, 3.8 m deep). After sediment removal by the end of March 2017, the lake was expanded downstream by about 0.3 ha and had a mean depth of 2 m. Next, a water level regulator was constructed that became functional from July 2017 onwards. By mid-August 2017, the water level was raised to ~64.8 m above sea level, resulting in a mean lake depth of 2 m.

Between May and June 2018, the bottom of the Kurika stream upstream of the lake between sampling sites T3 and T4 (Fig. 1A) was cleaned by sediment excavation for the first time in the last 30 years. Furthermore, aerial photos in April 2018 identified discharge of suspended solids from a construction site in an urban area near Lake Mustijärv (Kiani *et al.*, 2020). Sediment studies of the lake in 2018 identified the formation of a 40-50 cm thick silt layer in the IF and CE accumulation areas (Kiani *et al.*, 2020). In the remaining areas, the thickness of the newly formed sediments was ~10 cm. The accumulation areas released P at a rate consistent with that reported for anoxic surfaces in hypertrophic lakes (Kiani *et al.*, 2020).

Sampling and analyses

To estimate the impact of human activities in the catchment on the development of the external nutrient loading, sampling was carried out at several sampling sites across the Kurika stream (Fig 1A): T1, before flowing into lakes Riiska-Mandle and Mandle; T2, after passing these lakes; T3, after receiving the inflow of urban stormwaters (as diffuse inputs); T4, before Lake Mustijärv; and T5, after passing the lake. Water samples were collected from the surface water layer (0.3 m from the surface) using a rod water sampler. These samples were analysed for total nitrogen (TN), ammonium nitrogen (NH_4^+-N), nitrate nitrogen (NO_3^--N), total phosphorus (TP), soluble reactive phosphorus (SRP), biological oxygen demand (BOD_5), suspended solids (SS), total iron (TFe), and dissolved iron (DFe) following standard methods (APHA, 2017). We concentrated mainly on the macronutrients, nitrogen and phosphorus. Fe was of interest due to the potential coupling to P cycle. Also, BOD_5 (as a measure of organic matter content) and SS were used as water quality variables for additional insights on the processes occurring within the catchment and the lake.

T4 and T5 were also sampled in June and October 2018 to examine seasonality in the retention and role of the lake (in-lake processes) on changes in the above-mentioned components. The quality of the inflowing water at T4 was estimated according to the Estonian guidelines for the surface water quality (Estonian Regulation, 2009).

To analyse the response of lake ecosystem dynamics to external and internal nutrient loading after reconstruction, samples for water quality variables (except SS) were collected monthly from June to October 2018 at three in-lake stations (IF, close to

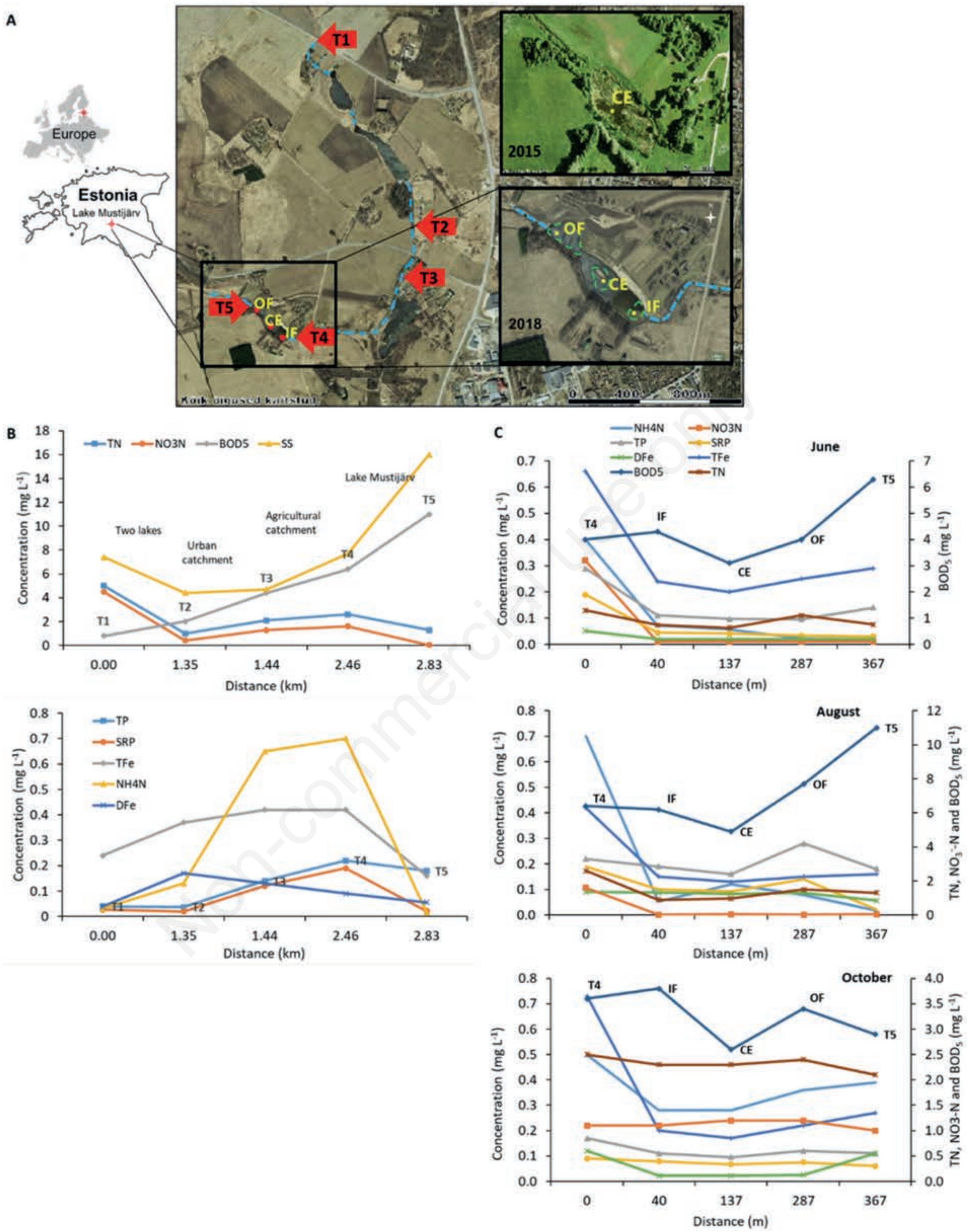

Fig. 1. A) Lake Mustijärv, located west of Viljandi (Estonia), and its sampling sites (IF, CE, OF) after sediment removal in 2018, and sampling sites (T1–T5) at the Kurika stream discharge to Mustijärv Map from Estonian Land Board. **B)** Concentrations of hydrochemical variables at different sites of the stream-lake continuum, including T1–T2 (including two unstudied lakes), T2–T3 (receiving nutrients from nearby urban areas), T3–T4 (passing agricultural areas), and T4–T5 (including the recently restored Lake Mustijärv) in August 2018. **C)** Concentrations of hydrochemical variables at the T4–T5 of the stream-lake continuum in June, August, and October 2018 IF, CE and OF are sampling sites in Lake Mustijärv.

the inflow of the Kurika stream; CE, in the center of the lake; and OF, close to the outflow) with a 2-L Van Dorn sampler (Fig. 1A). Water samples were taken 0.3 m from the surface, and from 0.3 m above the lake bottom. pH, water temperature, and dissolved oxygen (DO) concentration with a YSI-6600-V2 (YSI Corporation, Yellow Springs, OH, USA) were measured and water transparency (SDT) was determined with a Secchi disk.

Water samples were also collected for determining phyto- and zooplankton abundance, biomass, and dominant species using standard methods, described in detail in CEN standard (2006) and Haberman *et al.* (2001). Finally, pigments including chlorophyll *a* (Chl *a*), phaeopigments, carotenoids were determined following methods described in Laugaste *et al.* (2001).

Zoobenthos was sampled in June in three replicates at three locations (IF, CE, OF) with an Ekman-type grab sampler (225 cm^2), and samples were treated according to Timm *et al.* (1996). To describe the zonation and depth distribution of aquatic vegetation, the entire shoreline was traversed by boat in August 2018, on several transects. Plant composition and abundance were determined with a scale similar to Braun-Blanquet (1964). Experimental fish catches were made overnight in September 2018, with two multi-section gillnets (12 different mesh sizes ranging from 5 to 55 mm). Catch per unit effort (CPUE, g) or number of fishes per standard net and Fulton's condition factor (Fulton, 1904) were determined.

Hydrobiological data (*e.g.*, species composition, indicator species) were used to estimate water quality and trophic status of the lake by the use of well-established methods (Saether, 1979; abundance of chironomids), classification of Karabin (1985), based on the abundance of rotifers). Monitoring macrophytes, zoobenthos and fishes deserved attention also from the perspective of the recovery after significant disturbance.

Calculations and statistical analyses

To permit comparability between the changing rates of different nutrient forms along the flow vector, we expressed them as instantaneous values in % m^{-1}, where the percent change (decrease or increase) in the concentrations of different components between two adjacent sampling stations in the stream is divided by the distance in metres.

Inflow and outflow of nutrients and other water components in June, August, and October (three measurements) were computed by multiplying the concentration (mg L^{-1}) with the water discharge (L s^{-1}) at T4 and T5, respectively, and applying a conversion factor to express the fluxes on a daily basis. Water discharge was measured based on the distribution of flow velocities and cross-sectional area of the channel (EVS-EN ISO 748). Flow velocities were measured with an electromagnetic flow meter (FloMate 2000; Marsh McBirney, Frederick, MD, USA).

Daily retention (R in %) for the different water components (X) was estimated according to the equation:

$$R = (X_{in} - X_{out})*100/ X_{in} \quad \text{(eq. 1)}$$

where X_{in} is the inflow and X_{out} the outflow of a compound (mg d^{-1}). For TP and TN, fluxes were also expressed per m^2 of lake surface area (mg m^{-2} d^{-1}) to compare them with the sediment P release rates by diffusion published in Kiani *et al.* (2020). The P release rates determined for each of the three sampling sites (Kiani *et al.*, 2020) were multiplied by the percentages of the lake area covered by the sampling sites with different depths (*i.e.*, 2 m <IF <3 m, 6%; CE >3 m, 7%; and OF <2 m, 87%), and the products were summed to obtain the lake-average sediment P flux.

Finally, the water renewal time was estimated roughly by dividing the sum of daily water discharges for a year by the lake volume. Daily water discharges for a year were obtained by a linear interpolation of the available water discharge measurements. To close the annual cycle, the daily discharge measured in November was extended for the period until the first of January.

The differences in the studied water components between the lake sampling sites IF, CE, OF, and between sampling months were tested using the analysis of variance for repeated measures, using R software (4.1.0). Pairwise comparisons were conducted with a Tukey *post-hoc* test. The variables were initially log-transformed to approximate normal distribution, where necessary. Normality was tested with the Shapiro-Wilk test. Correlations between the chemical (nutrients and associated variables) and biological variables (phytoplankton, zooplankton) were tested with Spearman's correlation analysis.

RESULTS

Longitudinal changes in water quality of the surface water layer along the stream

In August 2018, the concentrations of TN, NO_3^--N, and SRP decreased between the sampling sites T1 and T2 of the Kurika stream (Fig. 1B, Tab. 1). Conversely, the concentrations of all studied components (excluding DFe) increased between sampling sites T2 and T3, where urban stormwater adds to the stream. SRP and NH_4^+-N showed the most pronounced increase, changing at a rate of 7% m^{-1} and 4% m^{-1}, respectively. Between the sites T3 and T4, there was also an increase in concentrations of the studied components (all nutrients, BOD_5, SS, TFe), though at a slower rate than between T2 and T3. The concentrations of TP, SRP, and SS showed the most marked increase (0.06% m^{-1}). A considerable decrease in values of the studied water quality variables, particularly the dissolved forms of N and P, occurred between T4 and T5, *i.e.*, the section including Lake Mustijärv. The least decline was observed in TP concentration, whereas BOD_5 and SS were higher at T5 than at T4. In general, the decreases in concentrations (% m^{-1}) were more pronounced in the Lake Mustijärv section (between T4 and T5) than in the section containing the two upstream lakes (between T1 and T2; Tab. 1). The concentrations of TFe and DFe, as well as NH_4^+-N decreased in the Lake Mustijärv section of the stream (at rates -0.16, -0.1%, and -0.26% m^{-1}, respectively), contrary to what was observed in the section between T1 and T2.

Temporal variations in the longitudinal changes in nutrient concentrations between T4 and T5, and lake retention

In June and October, Lake Mustijärv showed reduced nutrient concentrations, like in August. The decreases in N forms (TN, NH_4^+-N, NO_3^--N) between T4 and T5 were smaller in October than in June or August. The decrease in TP concentration between T4 and T5 in October was more pronounced than in August, but less than in June (Tab. 2). SRP decrease in June was of a similar magnitude (%) to that in August, but much smaller in October. A more detailed picture of the nutrient concentration changes shows an important role of the sediment accumulation basins in nutrient recycling (Fig. 1C). In October and August, TP and SRP concentrations increased between CE and OF, which reduced the overall decreasing trends between T4 and T5. Similar changes were observed for DFe, supporting the close coupling of P and Fe cycles in the lake. At the same time, BOD_5 increased between CE and OF.

The % retention values calculated for the sampling days in June, August and October were similar in magnitude to the % concentration changes between T4 and T5, suggesting that concentrations determined the retention values (Tab. 2). For example, TP retention was lowest in August (4% or 1 mg m^2 d^{-1}) and highest in June (56% or 52 mg m^2 d^{-1}; Tabs. 2 and 3). Nutrient inflow was highest in June at the highest water discharge (Tab. 3) and highest inlet TP concentration (Tab. 2). The proportion of mineral forms of N and P in the inlet (T4) TN and TP was highest (88 and 86%, respectively) at the lowest water discharge in August. The water renewal time for the lake was estimated to be ~35 times per year.

Lake ecosystem dynamics

Mean surface water temperature of the three lake sampling sites (IF, CE, OF) was highest in July (25°C) and lowest in October (10.3°C; Fig. 2A). The mean concentration of dissolved oxygen (DO) and SDT in June–October was 11.0 (±1.36) mg L^{-1} and 1.25 (±0.4) m (Fig. 2B), respectively. The water column was thermally stratified at the deepest area in July-August. DO near the lake bottom was below 2 mg L^{-1} at all three sampling sites (*Supplementary Tab. 1*).

All studied water quality variables (mean values of the three sampling sites) displayed pronounced seasonality (Fig. 3). No significant vertical differences in the concentrations of the studied variables were found (Fig. 3), with the exception of July and August, where several variables (TP, TN, SRP, and NH_4^+-N) displayed higher values in the bottom layer.

TP concentration increased towards August-September; in October the values declined back to those observed in June (Fig.

Tab. 1. Concentrations (mg L^{-1}) of major water quality components and changes (DC) in their concentrations (decreases "-" or increases "+" in %) between two adjacent sampling locations on the Kurika Stream. The rate of change (in % m^{-1}) was determined as a percent change in concentrations divided by the distance between the sampling locations, D (in metres) in August 2018. T1 to T5, sampling locations at the Kurika Stream discharging to Lake Mustijärv.

Variable	T1	T2	DC_{T1-T2}	DC/D_{T1-T2}	T3	DC_{T2-T3}	DC/D_{T2-T3}	T4	DC_{T3-T4}	DC/D_{T3-T4}	T5	DC_{T4-T5}	DC/D_{T4-T5}
D	–	–	–	1350	–	–	92	–	–	1018	–	–	380
TN	5	1	-80	-0.06	2.1	110	1.20	2.6	24	0.02	1.3	-50	-0.13
NH_4^+-N	0.031	0.13	319	0.24	0.65	400	4.35	0.7	8	0.01	0.017	-98	-0.26
NO_3^--N	4.5	0.43	-90	-0.07	1.3	202	2.20	1.6	23	0.02	0.04	-98	-0.26
TP	0.04	0.037	-8	-0.01	0.14	278	3.03	0.22	57	0.06	0.18	-18	-0.05
SRP	0.027	0.02	-41	-0.03	0.12	650	7.07	0.19	58	0.06	0.022	-88	-0.23
BOD_5	0.8	2	150	0.11	4.4	120	1.30	6.4	45	0.04	11	72	0.19
TFe	0.24	0.37	54	0.04	0.42	14	0.15	0.42	0	-0.00	0.16	-62	-0.16
DFe	0.039	0.17	336	0.25	0.13	-24	-0.26	0.09	-32	-0.03	0.056	-37	-0.10
SS	7.4	4.4	-41	-0.03	4.7	7	0.07	7.7	64	0.06	16	108	0.28

TN, total nitrogen; NH_4^+-N, ammonium nitrogen; NO_3^-–N, nitrate nitrogen; TP, total phosphorus; SRP, soluble reactive phosphorus; BOD_5, biological oxygen demand; TFe, total iron; DFe, dissolved iron; SS, suspended solids.

Tab. 2. Changes in the concentrations (DC in %, *i.e.* decreases "-" or increases "+") of water components (in mg L^{-1}) in the inflow and outflow of Lake Mustijärv in June, August and October 2018 (two years after lake restoration) and in November 2015 (several months before sediment removal). T4 and T5 are inlet and outlet of Lake Mustijärv, respectively. Retention (R, %) values were calculated using water discharge data (Q, in L s^{-1}) for T4 and T5.

	Nov 2015				Jun 2018				Aug 2018				Oct 2018			
Variable	T4	T5	DC	R	T4	T5	DC	R	T4	T5	DC	R	T4	T5	DC	R
Q	17.7	14.9	–	–	37.5	34.5	–	–	10.4	12.2	–	–	13	13	–	–
TN	1.1	1.1	0	16	1.3	0.76	-42	46	2.6	1.3	-50	41	2.5	2.1	-16	16
NH_4^+-N	0.028	0.05	75	-47	0.4	0.02	-96	96	0.7	0.017	-98	97	0.5	0.39	-22	22
NO_3^--N	–	–	–	–	0.32	0.01	-97	97	1.6	0.04	-98	97	1.1	1	-9	9
TP	0.044	0.06	36	-15	0.29	0.14	-52	56	0.22	0.18	-18	4	0.17	0.11	-35	35
SRP	0.037	0.04	19	0	0.19	0.03	-87	88	0.19	0.022	-88	86	0.09	0.06	-30	30
BOD_5	1.9	1.5	-21	34	4	6.3	58	-45	6.4	11	72	-102	3.6	2.9	-19	19
TFe	–	–	–	–	0.66	0.29	-56	60	0.42	0.16	-62	55	0.73	0.27	-63	63
DFe	–	–	–	–	0.052	0.02	-54	58	0.09	0.056	-37	26	0.12	0.11	-8	8
SS	3.5	5	43	-20	20	19	-5	13	7.7	16	108	-144	24	5.6	-77	77

TN, total nitrogen; NH_4^+-N, ammonium nitrogen; NO_3^-–N, nitrate nitrogen; TP, total phosphorus; SRP, soluble reactive phosphorus; BOD_5, biological oxygen demand; TFe, total iron; DFe, dissolved iron; SS, suspended solids.

3A). The SRP that constituted between 40 to 70% of the TP also showed a similar temporal variation (Fig. 3B). TN concentration first peaked in July and then again in October when the highest values were reached (Fig. 3C). While the share of NO_3^--N was negligible during most of the study period, it reached about 50% in October (Fig. 3D). The average contribution of NH_4^+-N to TN was 16%, being highest in July (25%) and September (33%; Fig. 3E). The BOD_5 increased towards July, and remained high until September, after which it decreased (Fig. 3F). During summer months (June and August), the TFe in the bottom layer was 135% of that in the surface water layer, and showed no temporal variation (Fig. 3G). In the surface water layer, TFe declined slightly from June to August and then increased again in September, reaching values similar to those in the bottom layer. DFe contributed more than half of TFe in July and August (Fig. 3H).

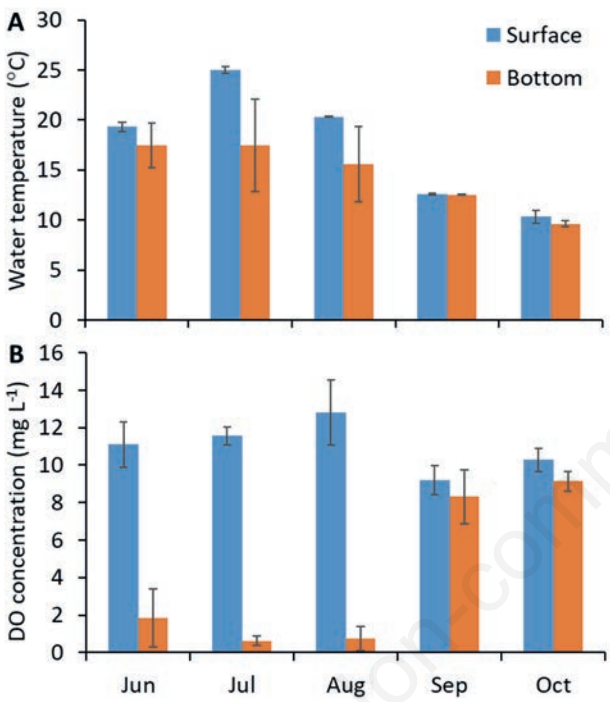

Fig. 2. Seasonal (monthly mean for three sampling sites ±SD) dynamics of water temperature **(A)** and concentration **(B)** of dissolved oxygen (DO) at the surface and near-bottom water layer of Mustijärv in 2018.

Phytoplankton biomass peaked (36±7 mg L^{-1}; Fig. 4A) in August and remained high in September (27±2 mg L^{-1}). In August, chlorophytes and dinophytes dominated, and euglenoids were sub-dominant (*Supplementary Tab. 2*). High biomass values in September were due to cyanobacteria, reaching a maximum. In October, chrysophytes and cryptophytes prevailed in the phytoplankton biomass. Chl *a* concentration positively and significantly correlated with the cyanobacteria (r = 0.832, p<0.001; Fig. 5).

Rotifers formed 95% of the total zooplankton abundance (*Supplementary Tab. 3*) and 82% of the total zooplankton biomass. Rotifer abundance (and total zooplankton abundance; Fig. 4B) was highest in August and September. Cladocerans were mainly represented (>99% of total cladoceran abundance) by the small-bodied *Bosmina longirostris* (O.F. Müller, 1985), while zooplankton biomass was formed mostly by the large-bodied rotifer *Asplanchna priodonta* (Gosse, 1850) and the small cladoceran *B. longirostris*. Copepods mostly consisted of young developmental stages (nauplii and copepodites). Total phytoplankton and cyanobacterial biomass, total zooplankton and Rotifera abundance positively and significantly correlated with surface water TP (r = 0.832, p<0.001, r = 0.737, p<0.001, r = 0.795, p<0.001, r = 0.784, p<0.001, respectively; *Supplementary Tab. 4*; Fig. 5).

Macrophyte species richness was highest among the emergent macrophytes (*Supplementary Tab. 5*). The most abundant species were *Typha latifolia* L. *and Phragmites australis* (Cav.) Trin. ex Steud. Zoobenthos abundance and biomass were highest at the OF location (*i.e.*, extended area of the lake; 1378±46 ind m^{-2} and 10.4±0.54 g m^{-2}). OF was represented mainly by Diptera species from the genera *Glyptotendipes* and *Endochironomus* using macrophytes as a substrate. At IF and CE, where the abundance and biomass values were ~20% of those at OF. Species richness was also low and mainly represented by the profundal mud-dwelling chironomid species, *Chironomus plumosus* (Linnaeus, 1758) and *Procladius choreus* (Meigen, 1804), as well as *Chaoborus flavicans* (Meigen, 1830), indicating anoxic conditions.

The most numerous among the four fish species found in Mustijärv was sunbleak (*Leucaspius delineatus* (Heckel), suggesting high predation pressure on zooplankton (*Supplementary Tab. 6*). The condition factor for perch (*Perca fluviatilis* L.) in Lake Mustijärv was higher than *e.g.* in Lake Võrtsjärv, and for roach (*Rutilus rutilus* (L.) as high as in Lake Võrtsjärv, suggesting good environmental (*e.g.*, food) conditions.

Tab. 3. Fluxes of TP and TN in 2015 (two months before sediment removal) and 2018 (two years after sediment removal). TN_{in} and TP_{in}, inflow of the nutrients; Q_{in} and Q_{out} in L s^{-1}; TN_{out} and TP_{out}, outflow of nutrients in mg m^{-2} d^{-1}; R,– retention in % and in mg m^{-2} d^{-1} and P release rates by diffusion (RR). RR was calculated by multiplying values in Kiani *et al.* (2020) with the percentages of the lake area that the sampling sites represent (IF, 6%; CE, 7%; OF, 87%) and sum of these products.

Year	Month	Q_{in}	Q_{out}	TP_{in}	TP_{out}	$R_{P\%}$	R_P	RR	TN_{in}	TN_{out}	$R_{N\%}$	R_N
2015	November	17.7	14.9	6.7	7.7	-15	-1		168	142	16	27
2018	June	37.5	34.5	94	42	56	52	3.7	421	227	46	195
2018	August	10.4	12.2	20	19	4	1	5.5	234	137	41	97
2018	October	13	13	19	12	35	7	1.8	281	236	16	45
2018	mean	20.3	19.9	44	24	32	20	3.6	312	200	35	112

DISCUSSION

Human impact on the lake water quality and the role of sediment accumulation basins

The lake receives pollution from urban (section between T2 and T3) and agricultural areas (section between T3 and T4), evidenced by the concentration increases of a number of components. Urban inputs were associated with the most pronounced increases of NH_4^+-N and SRP, while agricultural inputs with the increases of TP, SRP and SS. As a result, the concentrations of NH_4^+-N and TP at the inlet of Lake Mustijärv (T4) exceeded the threshold value for poor and very poor status according to the nutrient criteria developed for the rivers in Estonia. The concentrations of

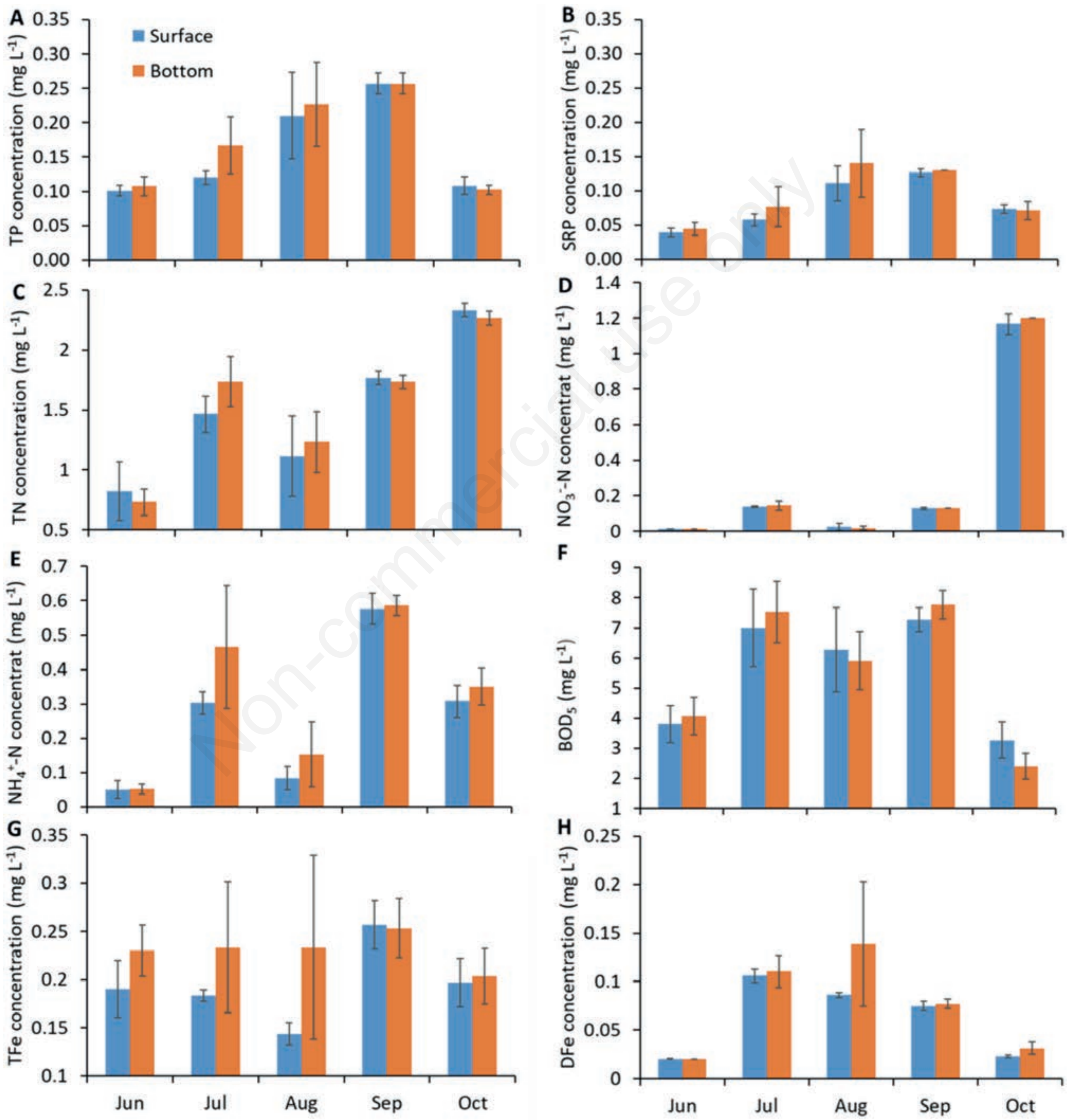

Fig. 3. Seasonal (monthly mean for three sampling sites ± SD) dynamics of the concentrations of: **A)** total phosphorus (TP); **B)** soluble reactive phosphorus (SRP); **C)** total nitrogen (TN); **D)** nitrate nitrogen (NO_3^--N); **E)** ammonium nitrogen (NH_4^+-N); **F)** biological oxygen demand (BOD_5); **G)** total iron (TFe); **H)** dissolved iron (DFe) at the surface and near-bottom water layer of Mustijärv in 2018.

NH_4^+-N in June and October indicated moderate and poor conditions, respectively, while TP concentrations indicated no changes relative to August at T4. Hence, water quality of the inflowing water suggests high implications on the water quality impairment of the reconstructed lake (eutrophication).

Much higher concentrations of all studied variables in 2018 than in 2015 (autumnal observations) suggested that water quality inflowing Lake Mustijärv could partially be impacted by the Kurika stream bed cleaning excavations activities in the section between T3 and T4. The sediments excavated were relocated mainly to the accumulation basins of Lake Mustijärv, evidenced by a thick layer of sediments there (up to 60 cm in two years). Suggesting these sediment accumulation rates will persist, the sediment accumulation basins will be filled back in ~4 years, and the entire lake area (about 10 cm in a year) will be refilled in ~20 years.

Moreover, these newly-accumulated sediments enhance eutrophication through internal phosphorus recycling, as indicated by decreased retention in August relative to June and October. The impact of sediment P release becomes evident as P concentration increases in the shallower lake section following the sediment accumulation basins (between CE and OF), where the released P is mixed to the surface water layer (Fig. 6). Similarly, BOD_5 changes suggest that entrapping of allochthonous organic matter occurs till the central part of the lake (between IF and CE), while strong flow conditions (flushing) in this area likely suppress the development of slow-growing phytoplankton. Instead, the development of autochthonous organic matter is favoured in the area close to the outflow (between CE and OF), where phytoplankton benefits from the sediment-released P, while assimilated P likely leaves the lake with the organic matter. Similar patterns of spatial heterogeneity in environmental and phytoplankton variables were described in Nõges and Tuvikene (2012), indicating a big importance of hydromorphological conditions.

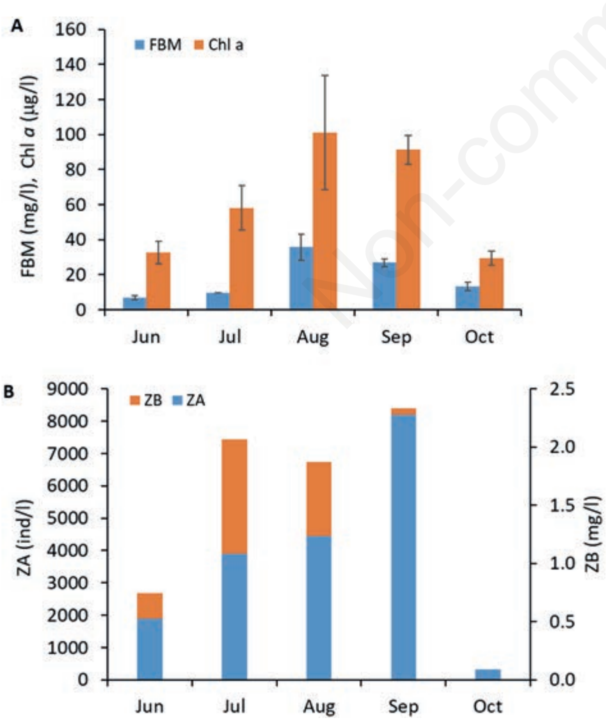

Fig. 4. Seasonal (monthly mean for three sampling sites ±SD) dynamics of **(A)** the concentration of chlorophyll *a* (Chl *a*), phytoplankton biomass (FBM)**; (B)** zooplankton abundance (ZA) and biomass (ZB) of Mustijärv in 2018.

To make Lake Mustijärv desirable again for the specific uses (water storage and recreation), the sediments have to be removed. However, restoration efforts can be focused mainly on the removal of sediments from the accumulation basins, resulting in reduced environmental impacts and costs of the intervention (*i.e.*, major disadvantages on sediment removal; Bormans *et al.*, 2016). Further cost reduction is possible through the reuse of the removed sediment in agriculture (Tammeorg *et al.*, 2023). Sediments removed from Lake Mustijärv were proven as P fertilizer in both mesocosm and field experiments (Kiani *et al.*, 2021; 2023). Hence, lake restoration by sediment removal may certainly benefit from introduction sediment accumulation, particularly if the impact of human activities (transport of nutrient-rich sediment particles) is high.

Ecosystem-level responses to stressors: implications for management

High external nutrient loading has often weakened the positive effect of restoration in terms of internal loading (Oberholster *et al.*, 2007; Bormans *et al.*, 2016; Lürling *et al.*, 2020). Our results showed that high external nutrient access can be a major risk for water quality impairment in a reconstructed lake. In Lake Mustijärv, the mean concentrations of TN, TP, and Chl *a* during the vegetation period (from June–October) were indicative of hypertrophic conditions according to Nürnberg (1996). The monthly data from the vegetation period also indicated the role of internal P loading in sustaining eutrophication.

A gradual increase in both TP and SRP concentrations towards late summer was very similar to what is usually reported for temperate shallow eutrophic lakes, where sediments supply algae with P *via* internal loading (Nürnberg, 2009, Søndergaard *et al.*, 2013, 2017). Kiani *et al.* (2020) demonstrated that sediment P release in Lake Mustijärv is mainly associated with the pool of Fe-bound P. The reductive dissolution of Fe-bound P likely progressed with rising temperature and increased anoxia of the sediment surfaces, supported by organic matter mineralization (Kiani *et al.*, 2020), similar to other observations from shallow lakes (Smith *et al.*, 2011, Søndergaard and Jeppesen, 2020; Tammeorg *et al.*, 2020). The observations in Lake Mustijärv indicated that oxygen conditions worsened in summer months, when differences in nutrient concentrations between the surface and bottom water layers were more notable. Recently, Søndergaard *et al.* (2022) used high-frequency measurements to demonstrate recurrent development of temporal stratification and anoxia in small and shallow Lake Ormstrup, followed by complete mixing of the water column, with implications for nutrient cycling. Our findings are also in a good agreement with former conclusions drawn on multiple shallow lakes (Søndergaard *et al.*, 2023).

An increase in water temperature and enhanced anoxia during summer could also promote the release of NH_4^+-N from sediments (Wang *et al.*, 2008; Van Hulle, 2010; Verschoor *et al.*, 2017). The phenomenon of higher NH_4^+-N than NO_3^--N in urban surface waters has been associated with of untreated sewage (Steele *et al.*, 2010), while in Lake Mustijärv, the observation of higher NH_4^+-N than NO_3^--N from June to September in the lake water was most likely due to the pollution related to relocation of stream bed sediments.

The composition of phytoplankton (*e.g.*, *Pseudanabaena limnetica* (Lemmermann) Komárek 1974*, Closteriopsis* spp.*,* euglenoids among sub-dominants) (*Supplementary Tabs. 2* and *3*)

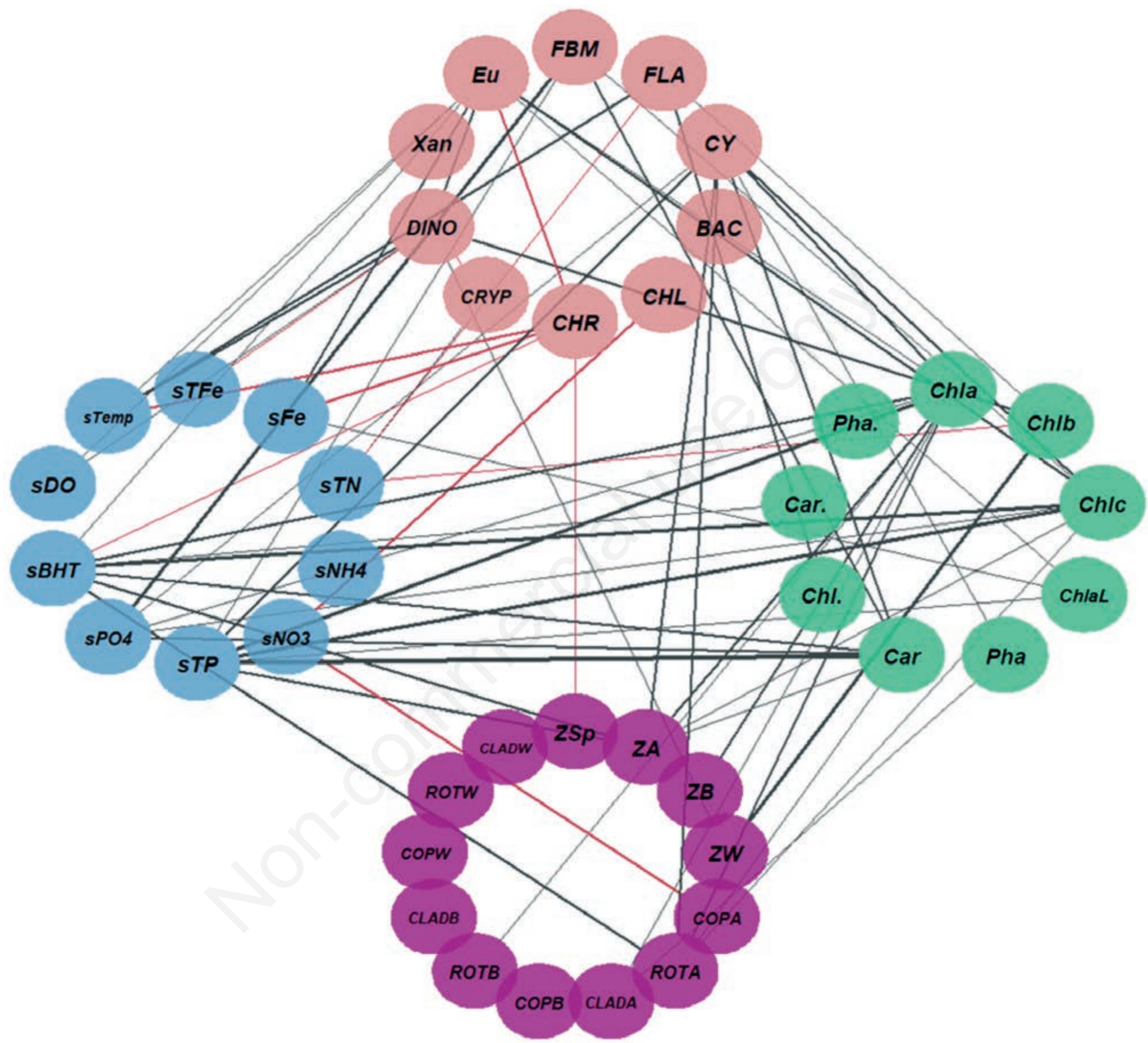

Fig. 5. Correlation network based on Spearman correlation coefficients between environmental and biological (phytoplankton and zooplankton) variables in Lake Mustijärv. Positive correlations are shown with black lines; negative correlation with red lines. The thickness of the line indicates level of significance (cutoff value: 070). Correlations between variables with high collinearity (dependency) are not shown. CY, biomass of cyanobacteria; BAC, diatoms, CHL, chlorophytes; CHR, chrysophytes; CRYP, cryptophytes; DINO, dinophytes; Xan, xanthophytes; Eu, euglenoids; FBM, phytoplankton biomass; FLA, number of phytoplankton species; Chla, concentration of chlorophyll *a* (Jeffrey and Humphrey, 1975); Chlb, concentration of chlorophyll *b*; Chlc, concentration of chlorophyll *c*; ChlaL chlorophyll *a* (Lorenzen, 1967); Pha, phaeopigments; Car, carotenoids; Chl, chlorophyll *a* % in the phytoplankton; Car, ratio of carotenoids to chlorophyll *a*; Pha, % of phaeopigments in chlorophyll; ZSp, number of zooplankton species; ZA, zooplankton abundance; ZB, zooplankton biomass; ZW, zooplankton weight; COPA, abundance of copepods; ROTA, abundance of rotifers; CLADA, abundance of cladocerans; COPB, biomass of copepods; ROTB, biomass of rotifers; CLADB, biomass of cladocerans; COPW, weight of copepods; ROTW, weight of rotifers; CLADW, weight of cladocerans; sTN, surface water layer concentrations of total nitrogen; sNH4, surface water layer concentrations of ammonium; and sNO3, surface water layer concentrations of nitrate nitrogen; sTP; surface water layer concentrations of total phosphorus; sPO4, surface water layer concentrations of phosphate phosphorus; sBHT, surface water layer concentrations of biological oxygen demand; sDO, surface water layer concentrations of dissolved oxygen; sTemp, surface water layer concentrations of temperature; sTFe, surface water layer concentrations of total iron; surface water layer concentrations of dissolved iron (sFe).

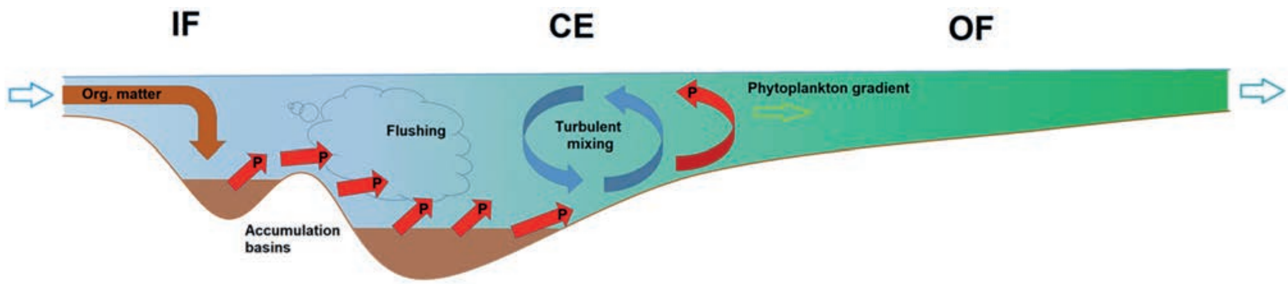

Fig. 6. Schematic description of the horizontal processes in Lake Mustijärv affecting retention of total phosphorus (TP) and organic matter. P released at sediment accumulation basins (IF and CE) is transported towards outflow (OF) and mixed to the surface water layer, supporting growth of phytoplankton that is flushed from IF and CE by strong flow (impact of the inflowing stream). The allochthonous organic matter is entrapped in the sediment accumulation basins, supporting sediment P release.

indicated highly eutrophic conditions of Lake Mustijärv. The domination of filamentous, non-heterocystous cyanobacterium *Pseudanabaena limnetica* in September was probably due to their competitive advantage in turbid conditions (Chomérat *et al.*, 2007; Soylu and Gonulol, 2010). Non-heterocystous cyanobacteria have often been found to outcompete others at high water temperature and under turbid conditions (Istvánovics *et al.*, 2002; Nürnberg *et al.*, 2013; Bormans *et al.*, 2016).

According to the classification of Karabin (1985, based on the abundance of rotifers), Lake Mustijärv was eutrophic in June, but hypertrophic in July, August, and September. The abundant rotifer taxa in Lake Mustijärv [*e.g.*, *Keratella cochlearis* (Gosse, 1851), *Keratella cochlearis tecta* (Gosse, 1851), *Polyarthra luminosa* (Kutikova, 1962)] are typical of eutrophic water bodies (Yağci, 2016; Yu *et al.* 2019), and the prevalence of juveniles (nauplii) among copepods and the small forms in cladocerans are common in eutrophic environments (Vijverberg and Boersma, 1997; Haberman and Haldna, 2014). The latter may also be associated with high fish pressure (Gozlan *et al.*, 2003), which agrees with the high abundance of sunbleak, and high water renewal time of the lake, which may have an even stronger effect than top-down control by planktivores (Rennella and Quiros, 2006).

A relatively fast recovery (in two years following reconstruction) of fishes, macrophytes and zoobenthos in Lake Mustijärv is noteworthy, because full sediment removal was supposed to largely destroy habitats for these biological component (Bormans et al., 2016). The re-colonization of macrophytes, which is generally considered a long-term process (Hilt *et al.*, 2010), was likely affected by a slow increase in water level. The largest abundance of *Typha latifolia* and *Phragmites australis* was expected given that re-colonization often starts with the previously prevailing species (Brouwer *et al.*, 2002). The macrophytes at the shallower areas (OF) supported the development of the macrozoobenthos community. A high Fulton's condition factor of benthivorous roach in Lake Mustijärv also suggested recovered zoobenthos. Nevertheless, the composition was characteristic of advanced eutrophication. Similarly, the first colonizers in English farmland ponds after sediment removal were mud-eating species which tolerate low oxygen (Ruse *et al.*, 2018).

Reducing external nutrient loading is of top priority to affect lake water quality of Lake Mustijärv. The use of nature-based solutions, like introducing wide enough buffer strips (Riis *et al.*, 2020), may also be helpful. Next, the lake restoration effect may be enhanced by fish manipulation, *i.e.*, decreasing the population of planktivorous cyprinid *Leucaspius delineatus* that will most likely increase grazing pressure of large cladoceran on phytoplankton (Jeppesen *et al.*, 2007), or chemical P inactivation.

The experience of the Lake Mustijärv case, where the trophic status of the lake was strongly affected by upstream management activities (sediment transport poses the risk of filling in at the first place), underlines the need for coordination between stakeholders responsible for upstream and in-lake activities.

CONCLUSIONS

The longitudinal study of the Kurika stream discharging to Lake Mustijärv revealed that urban and agricultural inputs contributed considerably to water quality impairment of the fully reconstructed lake. However, exceptional events in the catchment associated with streambed excavation activities may risk the re-disappearance of the lake in about 20 years. Sediments entrapped in sediment accumulation areas can enhance eutrophication, and asks for repeated sediment removal. Sediment removals can focus on relocated sediments, offering a more cost-effective and environmentally-friendly approach. The potential reuse of sediments in agriculture offers additional advantages. Noteworthy was a relatively rapid recovery of the lake's biota following sediment removal, including fish, macrophytes, and zoobenthos. Achieving improved lake water quality of the currently eutrophic lake necessitates comprehensive measures, primarily targeting external nutrient input. We strongly recommend coordinated efforts involving stakeholders to address external nutrient loading before undertaking any lake restoration or streambed excavation activities.

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Online supplementary material:

Supplementary Tab. 1. Environmental variables concentrations at three sampling sites of Lake Mustijärv in 2018.

Supplementary Tab. 2. Phytoplankton variables determined from the samples collected at three sampling sites of Lake Mustijärv.

Supplementary Tab. 3. Zooplankton variables determined from the samples collected at three sampling sites of Lake Mustijärv.

Supplementary Tab. 4. Spearman correlation coefficients between environmental and biological variables in Lake Mustijärv.

Supplementary Tab. 5. Macrophyte species and their abundance in Lake Mustijärv.

Supplementary Tab. 6. Fish community in Lake Mustijärv in September 2018.