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Variations of internal phosphorus loading and water quality in a hypertrophic lake during 40 years of different management efforts

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ABSTRACT

Variations of internal phosphorus (P) loading and water quality in the hypertrophic Lake Tuusulanjärvi (southern Finland) were studied over a period covering 40 years (1970–2010). The lake has hosted numerous different management efforts. Diversion of sewage waters away from the lake in 1979 resulted in a considerable reduction of external P loading. Due to diffuse loading from agricultural areas, external loading however still exceeds the critical loading of the lake, thus having an effect on the water quality. The total P concentration of surface layers has decreased but is still on a hypertrophic level (90 $\mu g\,l^{-1}$). The high productivity of the lake is maintained also by intensive internal P loading, and the sediment has a potential to release P to the water for decades even if external loading would be reduced to a tolerable level. Internal P loading has not decreased over the studied decades despite numerous within-lake management efforts (aeration with different methods, food web management). In opposite, our results demonstrated that destratification applied since 1998 resulted in a persisting internal P loading. Destratification has increased the concentration of oxygen and decreased the concentration of soluble P in deep water, but at the same time it has accelerated P release from aerobic bottoms. This can be due to elevated near-bottom temperatures, enhanced liberation of organic P through accelerated mineralization, and increased sediment resuspension by aeration-induced turbulence. Additionally, increasing wind velocities may have a role in the increasing aerobic internal loading. Food web management has compensated for the amplified P-cycling, revealed by the decreasing chlorophyll:total P ratio.

1. Introduction

In lakes, nutrients are continuously accumulated in the bottom, depending on the relationship between inflow and outflow, basin morphometry and flushing rate (Vollenweider, 1976; Søndergaard et al., 2003). Consequently, the greatest storage of nutrients in a lake ecosystem is usually in the sediment (Pettersson, 1998). These sediment nutrients have often implications on the water quality, because numerous mechanisms tend to recycle them back to the overlying water. Especially such internal loading of phosphorus (P) may delay or prevent the recovery of lakes from eutrophication after the reduction of external nutrient loading (Marsden, 1989; Jeppesen et al., 1991; Søndergaard et al., 2003). Hence, also the estimation of internal P loading has a central role in lake management and restoration.

Despite the importance of internal loading, quantification of P fluxes from the sediment to the water remain one of the main problems in limnology (Håkanson, 2004; Nürnberg, 2009). Anaerobic P release in the deeps of thermally stratifying lakes takes place because in low oxygen concentrations often prevailing in the sediment, Fe^{3+} is reduced to Fe^{2+} with a subsequent dissolution of P (Mortimer, 1941; Boström et al., 1982). Gas bubbling from the anoxic sediment can also contribute to internal P loading (e.g. Varjo et al., 2003). Internal loading in lake deeps can be estimated by measurements of P accumulation in the hypolimnion. In shallow non-stratifying areas, where the water column remains oxic throughout the growing season, internal P loading can also take place through various mechanisms. These include effects of photosynthetically elevated water pH on P release, sediment resuspension due to waves, water currents and biota, mineralization of organic matter and diffusion (Boström et al., 1982; Carvalho

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et al., 1995; Søndergaard et al., 2003; Tammeorg et al., 2015). The quantification of such internal P loading is very difficult. The accumulation approach used in the stratifying areas cannot be used in shallow areas due to continuous mixing and uptake by biota (Nürnberg, 2009). This is a major problem, because shallow non-stratifying areas often constitute to a large percentage of the lake area. Additionally, they are often in contact with the photic zone and can therefore strongly affect the productivity of lakes (Welch and Cooke, 1995; Søndergaard et al., 2003). Consequently, shallow lakes are generally more productive than deep ones (Oglesby, 1977). While redox conditions in the sediment considerably affect P release, the importance of low oxygen concentrations in lake deeps for P cycling and retention may have been overestimated. Low hypolimnetic oxygen concentrations and anaerobic P release may be considered as parallel symptoms of high sedimentation of P and organic matter (Gächter and Wehrli, 1998; Moosmann et al., 2006; Hupfer and Lewandowski, 2008). The discussion has been accelerated by minor effects of artificial oxygenation on lake water quality (Gächter and Wehrli, 1998; Salmi et al., 2014; Kuha et al., 2016).

In the present study, internal P loading and P accumulation in the hypertrophic Lake Tuusulanjärvi during 40 years were examined. The lake has been almost continuously aerated during the study years to counteract the anaerobic release of P from sediments. On the contrary to the earlier years, the aeration was realized via destratification during the years of 1998–2010, making the lake continuously mixing. It was hypothesized that this mixing has caused a decrease in anaerobic internal P loading and an increase in aerobic internal P loading. To test the hypothesis, the anoxic component of internal loading was separated from the total internal P loading. Moreover, the development of water quality over the years, in which also other different restoration activities occurred, was analyzed.

2. Material and methods

2.1. Study lake and restoration activities

Lake Tuusulanjärvi is situated in southern Finland (60° 26 N′, 24° 59 E′). It has a surface area of 5.95 km^2, mean and maximum depths of 3.2 m and 10.0 m, respectively. The catchment area of the lake covers 125 km^2. In the lake, the area that is naturally thermally stratified during the summer covers c. 20% of the lake surface. The theoretical water retention time of the lake is 250 d and the estimated critical external P loading is 353 $mg\,P\,m^{-2}\,y^{-1}$ (Marttila, 2005; Muukkonen, 2009). The critical loading has been calculated with the method by Vollenweider (1976), based on the relationship of areal P loading and the water residence time of the lake (e.g. Brett and Benjamin, 2008). The lake became eutrophic already during the 19th century, and eutrophication was accelerated since 1950's due to increasing settlements and agriculture (Harjula, 1972; Lepistö et al., 2006). To reduce the symptoms of eutrophication (e.g. fish kills), winter aeration by Nokia aerators (five aeration floats) was started in 1972 and it lasted until 1980 (Malve et al., 2004) (Fig. 1). The aerators led bubbled compressed air to the hypolimnion with a capacity of 1350 $kg\,O_2\,d^{-1}$ (Kolehmainen, 1974; Malve et al., 2004). Municipal wastewaters were diverged from the lake in 1979, and summer aeration was started 1982 with a Hydixor aerator, which pumped hypolimnetic water to the surface, where it was aerated and pumped back. The flux of dissolved oxygen was 240 $kg\,d^{-1}$ (Malve et al., 2004). In 1988, aeration however failed due to technical problems. In 1990, the aeration system was changed to another model (Planox) with an average oxygen flux of 450 $kg\,d^{-1}$ (Fig. 1). This aeration continued until 1997. Despite the restoration activities, heavy cyanobacterial blooms still developed in the 1990's, and in 1998 the aeration method was changed again. Aeration was started with Mixox aerators that pump oxygen-rich epilimnetic water to the hypolimnion with a capacity of 460 000 $m^3\,d^{-1}$ (Saarijärvi and Lappalainen, 2005). Five Mixox pumps were installed in 1998 and one more in 1999 (in 2003, aeration was exceptionally not performed). All six pumps were used during summer, and one or two during the ice-cover period (December–April). The aim was to prevent summertime stratification and to maintain the hypolimnetic concentration of dissolved oxygen above 2 $mg\,l^{-1}$. Additionally, to restore the food web, biomanipulation through removal of planktivorous and benthivorous fish was started in autumn 1997 (Sammalkorpi, 2000; Saarijärvi and Lappalainen, 2005) (Fig. 1). The total catch during 1997–2010 was 666 000 kg (1109 $kg\,ha^{-1}$), annual catch varying between 31 and 180 $kg\,ha^{-1}$ (Keski-Uudenmaan Vesiensuojelun Kuntayhtymä, 2012). The main target species were the cyprinids bream (*Abramis brama* L.) and roach (*Rutilus rutilus* (L.)) (Olin et al., 2006). The stocks of predatory fish (eel *Anguilla anguilla* L., pike *Esox lucius* L., pikeperch *Sander lucioperca* L., burbot *Lota lota* L.) were strengthened by stocking. In the catchment area, efforts to reduce diffuse loading from agricultural areas were increased in the 2000's and wetlands have been built in the catchment to trap nutrients and suspended solids on their way to the lake. The largest wetland (Rantamo-Seitteli) has an area of 28 ha.

2.2. Sampling and calculations

The total internal P loading (IP_{tot}) in Tuusulanjärvi was calculated by comparing mass balance computed P retention (R_{mb}) and sediment-derived P retention (R_{sed}). R_{mb} is calculated by a mass balance approach but retention will be underestimated if internal loading of P is significant but ignored (Dillon and Molot, 1996). Therefore, the deviation of R_{sed} from R_{mb} can be used to estimate the magnitude of internal loading as follows (Nürnberg, 1984; modified by Tammeorg et al., 2016).

$$IP_{tot} = TP_{in} \times (R_{sed} - R_{mb}/R_{sed})$$

where IP_{tot} is total internal load ($mg\,P\,m^{-2}\,y^{-1}$) and TP_{in} is the external load of P. TP_{in}-values were calculated by widely used methodology (e.g. Dillon and Evans, 1993; Knuuttila et al., 1994) based on the monitoring of P coming to the lake via river inflow, point loading from industry and municipalities, diffuse loading and precipitation (data from Ojanen, 1979; Marttila, 2005; Muukkonen, 2009; Hertta database, 2015). The water sample based estimates of external loading have also been shown to match those modelled with the VEPS-model used by Finnish Environment Institute (Tattari and Linjama, 2004; Marttila, 2005). The model takes into account all the P sources listed above. Data on temperature, water column pH and concentrations of TP, O_2 and chlorophyll *a* were obtained from the Hertta database of Finnish Environment Institute (Hertta database, 2015).

R_{mb} ($mg\,P\,m^{-2}\,y^{-1}$) in the lake was calculated as follows (e.g. Hupfer and Lewandowski, 2008),

$$R_{mb} = TP_{in} - TP_{out}$$

where TP_{in} is the external loading of P and TP_{out} is the ouflow of P.

R_{sed} was obtained from a dated sediment core (Dillon and Evans, 1993; Moosmann et al., 2006). The sediment core was taken with HTH gravity corer (Renberg and Hansson, 2008) from the deepest site of the lake. The core was sectioned into 0.5 cm slices to a depth of 20 cm to cover the period for which comprehensive water quality monitoring data were available (since 1970). The sediment samples were freeze-dried and ground. The TP concentration in each sediment layer was measured after wet digestion with sulphuric acid and hydrogen peroxide in the microwave digestion system using

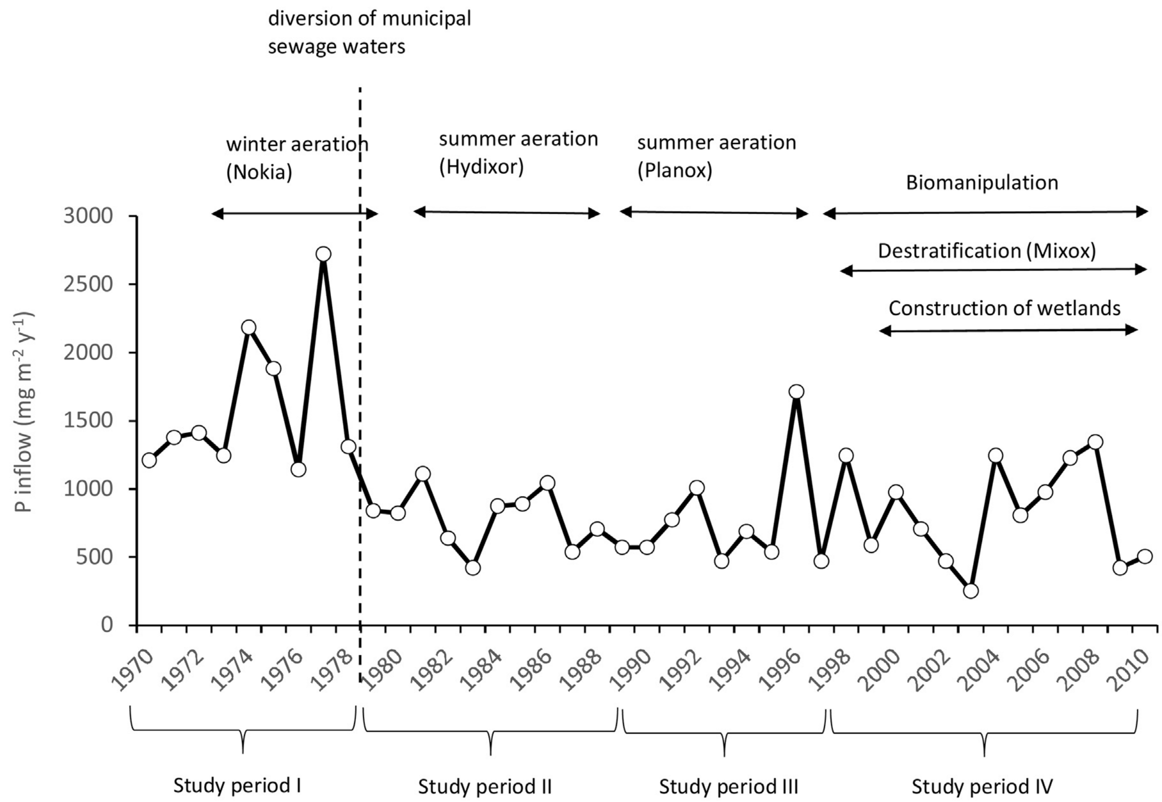

Fig. 1. Variations in the external loading of P to Lake Tuusulanjärvi during 1970–2010.The timing of the main restoration activities and the different study periods are also shown.

the molybdenum-blue-ascorbic-acid method (Lachat autoanalyzer, Quickchem Series 8000). Sedimentation rates were determined by dating cores by ^{210}Pb and ^{137}Cs. Sediment datings were performed at the Environmental Radioactivity Laboratory of Liverpool University. Samples were analysed for ^{210}Pb, ^{226}Ra, and ^{137}Cs by direct gamma assay using Ortec HPGe GWL series well-type coaxial low background intrinsic germanium detectors (Appleby et al., 1986). R_{sed} was calculated by multiplying the TP concentration in each sediment layer by the sedimentation rates of dry material.

The internal P load from anoxic areas (IP_{anox}) in lakes with anoxic hypolimnia was calculated based on the hypolimnetic P accumulation using the equation (Nürnberg, 1984):

$$IP_{anox} = anoxicarea \times anoxicperiod \times Preleaserate/lakearea$$

where the anoxic area (m^2) is the sediment surface area that is in contact with anoxic water, anoxic period (in days per year) is the duration of the anoxia, and the P release rate is the rate at which P is released from the anoxic sediments surface ($mg\,m^{-2}\,d^{-1}$). It was thus assumed that in shallow areas, where no stratification takes place, $IP_{anox} = 0$. The internal loading originating from such shallow areas with oxic water column (IP_{ox}) can be calculated by

$$IP_{ox} = IP_{tot} - \mathrm{IP}_{anox}$$

IP_{tot}, IP_{anox} and IP_{ox} were calculated for four periods characterized by different lake management actions: The first period covers 1970–1979, when municipal wastewaters were led to the lake (termed period I). During this period, winter aeration was also used. The next periods analysed were 1980–1989 (period II), when the point loading was diverted and summer aeration with Hydixor was conducted and 1990–1997 (period III), when aeration was performed with a Planox-aerator. The fourth and final period was 1998–2010 (period IV), when large-scale destratification by Mixox-aerators and biomanipulation through effective fishing were performed.

Data on wind velocity and direction (8 measurements per day) were obtained from the Finnish Meteorological Institute and they were measured at the Helsinki–Vantaa airport situated 10 km south-west of Tuusulanjärvi. Wind data were used to examine to possible variations in wave action, which is given by the equation (Hamilton and Mitchell, 1988)

$$\text{wave action} = \frac{H^2}{Z}$$

where H is the wave height and Z is water depth. It has been shown that in a shallow lake, P concentration in the water increases linearly with wave action (Hamilton and Mitchell, 1988), making it a useful parameter to investigate the effects of wind variations on internal P loading in Tuusulanjärvi. Wave height was calculated as one half of wave length and the theoretical wavelengths were calculated using equations presented by Carper and Bachmann (1984):

$$L = \frac{gT^2}{2\pi}$$

where L is the wavelength (m), g is the gravitational constant, and T is the wave period (s). The value of T is given by

$$\frac{gT}{2\pi U} = 1.20 \tanh\left[0.077\left(\frac{gF}{U^2}\right)^{0.25}\right]$$

where U is the wind velocity ($m\,s^{-1}$) and F is the effective fetch (m). The effective fetch for each wind direction was calculated according to Beach Erosion Board (1972):

$$F = \frac{\sum x_i \cos\gamma_i}{13.5} s$$

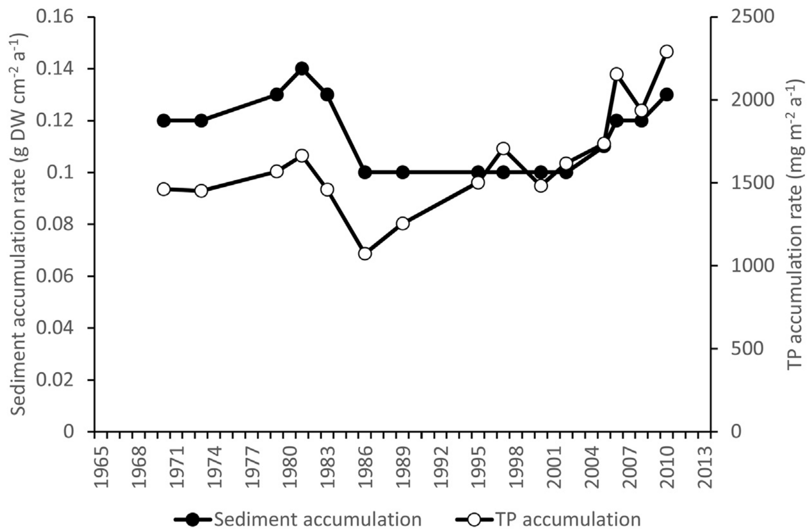

Fig. 2. The rates of sediment and P accumulation in Lake Tuusulanjärvi during 1970–2010.

Table 1
Predicted and observed P retention, external and internal P loading, and water quality characteristics of Lake Tuusulanjärvi during the four different study periods (±95% confidence limits when appropriate). I = 1970–1979, II = 1980–1989, III = 1990–1997, IV = 1998–2010.

	I	II	III	IV
TP_{in} (mg m^{-2} y^{-1})	1533 ± 350	761 ± 139	779 ± 290	827 ± 199
R_{mb} (mg m^{-2} y^{-1})	1064	262	292	393
R_{sed} (mg m^{-2} y^{-1})	1567	1363	1604	1869
IP_{tot} (mg m^{-2} y^{-1})	432	610	632	659
IP_{anox} (mg m^{-2} y^{-1})	320	557	716	303
IP_{ox} (mg m^{-2} y^{-1})	112	53	−84	356
SRP surface June-September (μg l^{-1})	26 ± 8	22 ± 3	23 ± 4	22 ± 3
SRP bottom whole year (μg l^{-1})	69 ± 27	75 ± 21	75 ± 12	43 ± 4
SRP bottom July-August (μg l^{-1})	153 ± 141	174 ± 115	86 ± 27	49 ± 10
TP surface whole year (μg l^{-1})	110 ± 11	95 ± 5	93 ± 6	90 ± 6
TP surface June-September (μg l^{-1}),	135 ± 21	109 ± 11	111 ± 11	90 ± 7
Chl *a* June-September (μg l^{-1})	61 ± 13	65 ± 12	56 ± 11	36 ± 4
Chl *a*/TP June-September	0.53	0.54	0.53	0.42
Temperature surface whole year (°C)	8.8 ± 1.6	10.5 ± 1.1	9.8 ± 1.5	11.3 ± 6.5
Temperature surface July-August (°C)	18.5 ± 0.6	19.0 ± 0.6	19.2 ± 0.7	20.3 ± 0.5
Temperature bottom whole year (°C)	8.1 ± 1.3	9.4 ± 0.9	9.0 ± 1.0	11.1 ± 0.9
Temperature bottom July-August (°C)	16.6. ± 0.5	16.9 ± 0.5	17.8 ± 0.4	19.4 ± 0.5

where x_i is the distance from the sampling site to land for every deviation angle γ_i (up to ±42°) and s is a scale constant. The calculations were conducted for the centre of the lake for south-westerly wind that prevails in the area.

2.3. Statistical analyses

The variations in the concentrations of total phosphorus (TP), soluble reactive phosphorus (SRP), O_2 concentration and water temperature, chlorophyll *a*, and pH between the four study periods were statistically analysed with Kruskal-Wallis test. The non-parametric test was used due to the unequal length of the different study periods and consequently unequal number of observations within each period. Pairwise comparisons were performed with the Dwass, Steel, Critchlow-Fligner method (DSCF) (SAS Institute Inc., 2008). For SRP, the analyses were performed for near-bottom values for the whole year and for July–August, when the lake naturally would show summer stratification and SRP may accumulate into the hypolimnion. The summer period is important also concerning algal blooms. For TP, the analyses were conducted with surface values for the whole year and for June–September. This approach was taken because in shallow lakes, internal P loading may cause elevated TP values in the epilimnion during the growing season (Jeppesen et al., 1997). Temperature differences were statistically analysed for the whole year and for July–August to find out the possible effects of aeration in the hypolimnion. Chlorophyll *a* and pH were analysed for the surface values during the June–September. Additionally, to track the factors behind possible changes in the Chl/TP relationship, the variations in SRP concentrations at the surface during June-September were also tested. The dependence of surface water concentration of TP on external loading was studied with linear regression (ln-transformed data) (e.g. Sas, 1990). The analysis was performed for the whole duration of the study (1970–2010) and separately for the four different periods. Additionally, wind velocity (whole-period average and daily maximum values) variations between the four study periods were tested.

3. Results

3.1. External and internal P loading, and P accumulation

The external loading of P into Tuusulanjärvi decreased steadily from 1200 to 2900 mg m^{-2} y^{-1} (8–17 tn y^{-1}) in the 1970's to 300–1000 mg m^{-2} y^{-1} (2–6 tn y^{-1}) in the late 1980's and early 1990's (Fig. 1). During some years in the 2000's the inflow again exceeded 1200 mg m^{-2} y^{-1}. Calculated for the four different periods, the average external loading was approximately twice as

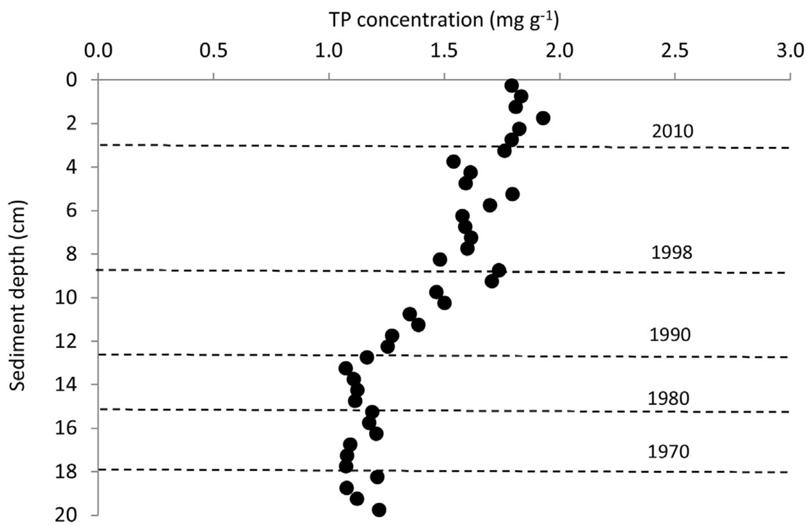

Fig. 3. The concentration of TP in the different sediment layers in Tuusulanjärvi and the age of the sediment at different layers.

Table 2
Results from pairwise comparisons in the difference in water quality parameters and wind velocity between the study periods. I = 1970–1979, II = 1980–1989, III = 1990–1997, IV = 1998–2010. Statistically significant differences are bolded.

	I–IV	II–IV	III–IV	I–II	I–III	II–III
Temp. surface whole year	0.0031	0.2980	0.1749	0.2406	0.2937	0.9904
Temp. surface summer stratification	0.3378	0.4729	0.8485	0.9634	0.8779	0.9752
Temp. bottom whole year	0.0002	**0.0085**	**0.0068**	0.2212	0.2723	0.9995
Temp. bottom summer stratification	**<0.0001**	**<0.0001**	**0.0057**	0.9854	0.0833	0.1649
O_2 bottom summer stratification	**0.0001**	**0.0023**	0.3008	0.5935	0.1359	0.5667
O_2 bottom winter	**0.0313**	0.4653	0.8429	0.1111	**0.0383**	0.2880
pH surface whole year	0.9998	0.5958	0.4748	0.9498	0.9103	0.9997
pH surface growing season	**0.0005**	0.2725	0.5561	0.1699	0.0676	0.9816
TP surface whole year	**0.0363**	0.2742	0.9420	0.6485	0.1742	0.5837
TP surface growing season	**0.0006**	**0.0216**	**0.0162**	0.3299	0.3796	1.000
TP bottom whole year	0.0750	**0.0055**	**0.0219**	0.9817	1.000	0.9062
TP bottom growing season	**0.0013**	**0.0005**	**0.0002**	0.9955	0.9751	0.8449
SRP surface whole year	0.9229	1.0000	0.9999	0.9229	0.9344	0.9986
SRP bottom whole year	0.1878	**0.0020**	**0.0016**	0.9646	1.000	0.9591
SRP bottom stratification	**0.0163**	0.1353	0.2351	0.8679	0.2884	0.6835
Chl *a* growing season	**0.0007**	**<0.0001**	**0.0008**	0.9785	0.8878	0.3738

high in period I (1533 mg m^{-2} y^{-1}) as in the other three periods (761–827 mg m^{-2} y^{-1}).

The average sediment accumulation rate (net sedimentation) during the whole study was 0.12 g DW cm^{-2} y^{-1} (0.43 cm y^{-1}) and was relatively constant over the years (Fig. 2). During period I the rate of sediment accumulation was 0.12–0.13 g DW cm^{-2} y^{-1} and dropped to a constant value of 0.10 g DW cm^{-2} y^{-1} in the 1980's and 1990's (Fig. 2). In the 2000's the accumulation rate again reached 0.13 g DW cm^{-2}y^{-1}. The rate of TP accumulation approached 1500 mg m^{-2} y^{-1} before 1979 and decreased after the sewage diversion to 1073 mg m^{-2} y^{-1} in 1986 (Fig. 2). Thereafter TP accumulation increased again and exceeded 2000 mg m^{-2} y^{-1} in the 2000's (Fig. 2). The average TP accumulation rate during the period IV was higher (1869 mg m^{-2} y^{-1}) than in the three preceding periods (1363–1604 mg m^{-2} y^{-1}) (Table 1). The TP concentration of the sediment decreased from the surface value 1.8 mg g^{-1} until 13 cm sediment depth (late 1980's) (Fig. 3). Thereafter, fluctuations in the P concentration were low and remained between 1.1–1.2 mg g^{-1} until 20 cm depth (1967).

The trend in R_{mb} was similar to that of external P loading, showing lowest values during periods II and III (Table 1). IP_{tot} showed a constantly increasing trend being lowest in period I (432 mg m^{-2} y^{-1}) and highest during period IV (659 mg m^{-2} y^{-1}). IP_{anox} exceeded 500 mg m^{-2} y^{-1} in periods II (557 mg m^{-2} y^{-1}) and III (716 mg m^{-2} y^{-1}) and remained below 350 mg m^{-2} y^{-1} during periods I and IV. IP_{ox} was 112 mg m^{-2} y^{-1} in period I, 53 mg m^{-2} y^{-1} in period II, −84 mg m^{-2} y^{-1} in period III and 356 mg m^{-2} y^{-1} in period IV.

3.2. Water quality

The average whole-year concentration of surface TP decreased constantly (Table 1). Statistically significant difference was, however, found only between period I (110 μg l^{-1}) and IV (90 μg l^{-1}) (Table 2). The TP concentration in the growing season (July–September) was, however, significantly lower during period IV compared with all three preceding periods (Table 2). The reason for this could be seen in the seasonal development of surface TP concentration during the different periods. In period I, the concentration exceeded 120 μg l^{-1} during July–September and 150 μg l^{-1} during August (Fig. 4). Since then, the summertime TP concentrations decreased. In periods II and III, the August concentration of TP still exceeded 120 μg l^{-1}, but during period IV it remained below 100 μg l^{-1} (Fig. 4). During the first half of the year, no considerable changes occurred after sewage diversion in 1979. In near-bottom water, both whole-year concentration and summer concentration of TP were significantly lower during period IV than during preceding periods (Table 2). The concentration in February showed values >200 μg l^{-1} during the periods I and III, but year-to-year variation amplitude was wide (Fig. 4). During the other two peri-

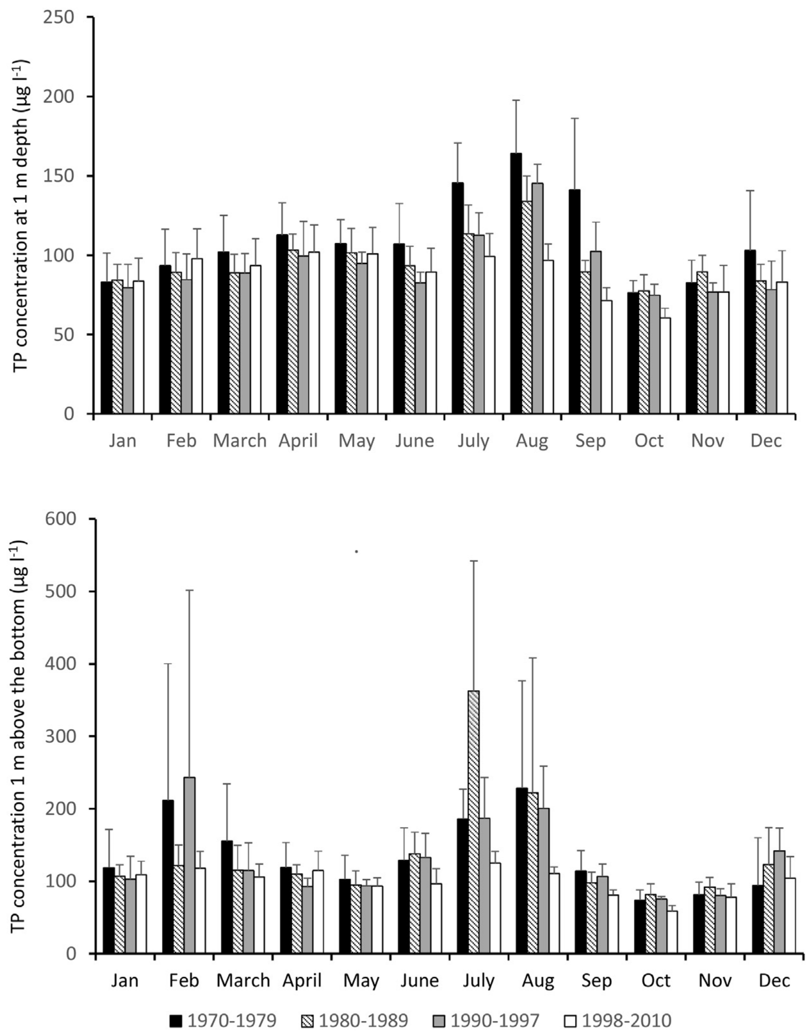

Fig. 4. The seasonal variations in the concentration of TP in the deepest part of Tuusulanjärvi during the four study periods (+95% confidence limits). Upper panel, 1 m below the surface, Lower panel, near-bottom water.

ods such peaks were not observed. The summertime concentrations decreased after the sewage diversion but due to high annual variation the changes were not significant (Fig. 4, Table 2). During period IV the near-bottom concentration of TP has remained on average below 130 $\mu g\, l^{-1}$ with significant difference to previous periods (Fig. 4).

With all data, surface TP concentration was significantly dependent on external loading of P ($F_{1,39} = 10.43$, $p = 0.002$, $R^2 = 0.21$). When analyzed separately for the different periods, no significant relationships between external loading and TP concentration were found due to the high year-to-year variation ($p > 0.05$ for all comparisons). The concentration of chlorophyll *a* decreased significantly in period IV compared with the previous periods (Table 2). The average growing season concentration during the first three study periods was 56–65 $\mu g\, l^{-1}$, but below 40 $\mu g\, l^{-1}$ during period IV (Table 1). Chl:TP ratio was on average 0.53–0.54 during the first three periods and 0.42 in period IV (Table 1). The average hypolimnetic SRP concentration varied between 69 and 75 $\mu g\, l^{-1}$ during the three first periods but decreased to 43 $\mu g\, l^{-1}$ in period IV (Table 1). During summer stratification in July-August the average SRP concentration exceeded 150 $\mu g\, l^{-1}$ during periods I and II and decreased thereafter, being 49 $\mu g\, l^{-1}$ in period IV (Fig. 5, Table 1). Both with whole year data and with July–August data, statistically significant differences in hypolimnetic SRP concentration were found between period IV and previous periods (Table 2). In surface concentrations of SRP, no between-period differences were observed ($p > 0.05$ for all comparisons) (Tables 1 and 2).

Oxygen concentrations under the ice-cover during February–March were significantly higher in period III (average 5.3 $mg\, l^{-1}$) and period IV (5.4 $mg\, l^{-1}$) than in period I (2.9 $mg\, l^{-1}$) (Fig. 6, Table 2). Oxygen concentrations during summer stratification were elevated when destratification was started, and the concentration during 1998–2010 (average 6.3 $mg\, l^{-1}$) was significantly higher compared with the two first study periods. During period IV, hypoxic conditions were observed only in 2003 and in 2010. Surface water pH during July–August was significantly lower in period IV (average 7.75) than in period I (8.01). Other pairwise comparisons of pH were insignificant.

3.3. Water temperature and wind

Whole-year water temperature average at the surface was significantly higher during period IV (11.1 °C) than period I (8.8 °C) (Tables 1, 2). In the near-bottom water, temperature changes were steeper and the temperature was significantly higher in period IV compared with all the three previous periods. Especially high temperature rise was observed in July–August, when the average temperature was 16.6 °C during period I and 19.5 °C during period IV (Table 2).

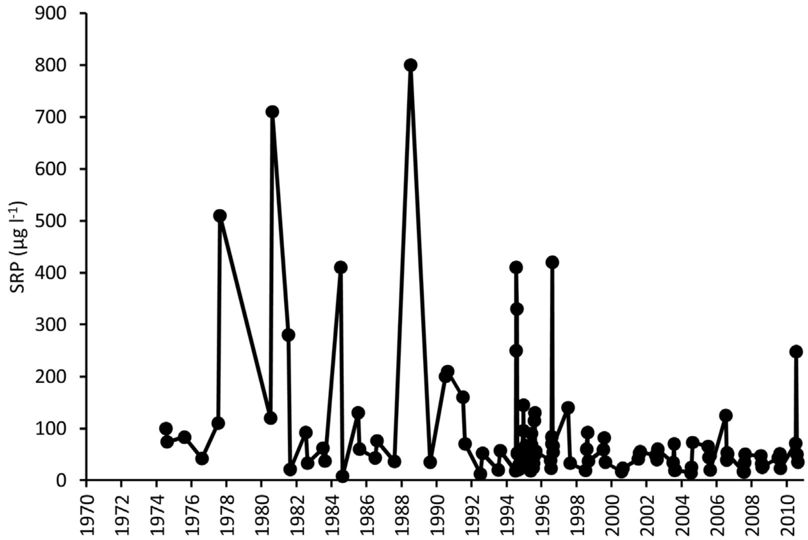

Fig. 5. The concentration of soluble reactive phosphorus (SRP) near the bottom in the deepest part of Lake Tuusulanjärvi during summer stratification.

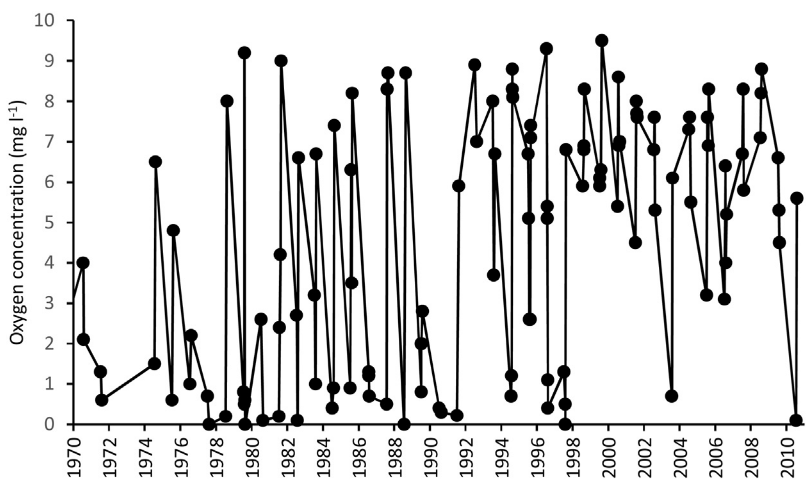

Fig. 6. The concentration of dissolved oxygen between 1970 and 2010 near the bottom in the deepest part of Tuusulanjärvi during late summer.

The average wind velocity was 3.86 m s^{-1} in period I, 3.69 m s^{-1} in period II, 3.66 m s^{-1} in period III and 4.0 m s^{-1} in period IV. Periods II and III did not differ from each other, but all other between-period comparisons were statistically very significant ($p < 0.0001$). The maximum daily wind velocity was on average 6.0 m s^{-1} in period I and 5.85 m s^{-1}, 5.82 m s^{-1} and 6.26 m s^{-1} in the following three periods, respectively. All between-period comparisons were statistically significant ($p < 0.05$), with the exception of comparisons period I vs. period II and period II vs. period III ($p > 0.05$). With the average wind velocity, wave action calculated for 1 m depth was 0.27, 0.25 and 0.24 during the three first study periods, respectively. In period IV, it was 0.30. With the daily maximum wind velocities wave action during the four periods were 0.68, 0.65, 0.64 and 0.74, respectively.

4. Discussion

In Lake Tuusulanjärvi, despite the reduction in the external loading of P after sewage diversion, during most years the present external P loading still clearly exceeded the theoretical critical loading of the lake. According to the European Water Frame Directive, the lake should reach a good ecological status, which requires a total P concentration below 55 µg l^{-1} in the epilimnion during the growing season and chlorophyll *a* concentration below 20 µg l^{-1}. These targets have not been reached. Nowadays the external loading of P to Tuusulanjärvi is dominated by diffuse loading mainly from agricultural areas and the wetlands built in catchment area seem not yet to have considerably affected the loading. This is connected rather to the area of the wetlands than to their ability to retain P. The largest wetland, Rantamo-Seitteli (24 ha), retains 15–20% of the annual P loading coming from the 19 km^2 catchment of the Sarsalanoja brook that brings c. 25% of the P inflowing to the lake, thus reducing the total external loading of the lake by 4–5% (Hietala, 2012).

When interpreting the results, the possible time lags in the effects of the various restoration activities must be remembered (Meals et al., 2010). Restoration during one of the study periods may affect ecosystem functioning also during the following periods. The results demonstrated, however, that external P loading still has major effects on the water quality of the Lake Tuusulanjärvi. Together with external loading, internal loading must be considered when seeking for factors behind the persistent hypertrophic conditions and possibilities to improve the situation. In the sediment of Tuusulanjärvi, TP concentration was constant below 13 cm depth, and increased between this depth and the sediment surface. TP profiles of similar shape have been described from numerous eutrophic lakes (e.g. Søndergaard et al., 1996; Rydin, 2000; Zan et al., 2011). The sediment depth, at which the TP concentration becomes stabile, shows the point below which the phosphorus diagenesis processes have stabilized (Carey and Rydin, 2011; Dittrich et al., 2013). Below this stabilization point, mobile P has been lost to upper sediment layers and/or to the water column and the remaining P is buried permanently, unless some major disturbance takes place (Håkanson, 2003; Carey and Rydin, 2011). The amount of P that can be released may be expressed as the difference between the P concentrations in the stabilization point and at the sediment surface (assuming constant sedimentation rate) (Rydin, 2000; Carey and Rydin, 2011). In Tuusulanjärvi, the stabilization point was at sediment depth, which represented the years 1989–1990. This suggests that sediment can release P c. 25 years. This is in line with studies from other eutrophic lakes suggesting that after reduction of external nutrient loading it takes 10–30 years before a new equilibrium between the sediment and the water is reached (Søndergaard and Jeppesen, 1999; Jeppesen et al., 2005). This is not the whole story however, because the post-depositional migration of P within the sediment must also be taken into account. Upward diffusion of P in the sediment can take place and result in increasing P concentration in the upper sediment layers (Carignan and Flett, 1981; Søndergaard et al., 1996). Therefore, also the use of dated sediment profiles in evaluating past external loading and accumulation of P must be done with caution (Carignan and Flett, 1981; Søndergaard et al., 1996; Ostrofsky, 2012). In the sediment of Tuusulanjärvi, the dominant inorganic P fraction is Fe-bound P (Koski-Vähälä and Hartikainen, 2001). As this fraction is prone to post-depositional mobility (Søndergaard et al., 1996; Rydin, 2000), estimates on the forthcoming internal loading of Tuusulanjärvi are rough.

IP_{tot} increased considerably after the reduction of external P. Such phenomenon has been previously described for several lakes (Ahlgren, 1988; Welch and Cooke, 1995). After the reduction of external nutrient loading, internal loading of P from the sediment may increase although it would eventually decline (Welch and Cooke, 1995). Despite the numerous in-lake restoration activities in Tuusulanjärvi, internal loading did not decrease but rather showed an increasing trend even decades after the sewage diversion. Because IP_{anox} decreased due to aeration, IP_{ox} must have increased. One factor behind this was probably the increasing wind velocity. The increase in wind velocity could be expected, because climate models predict increasing wind velocities in northern Europe (Pryor et al., 2005). In shallow lakes, water quality is dependent on weather conditions that can regulate P cycling for instance through effects on sediment resuspension (Marsden, 1989; Søndergaard et al., 1992; Niemistö et al., 2009). The wave action values increased 10–20% in period IV compared with the earlier periods, which can partly explain the increasing IP_{ox}. The same wave action that occurred with the maximum winds at 1 m depth in period III, was reached in 1.2 m depth in period IV. The area between 1 m and 2 m contours covers 36% of the whole area of Tuusulanjärvi and such change in wave effects can thus have a considerable impact on sediment disturbance. Another factor behind the increased IP_{ox} is the changing duration of ice cover. Between 1970's and 2000's, the annual ice-cover period of lakes in southern Finland has declined on average 5–7 days (Korhonen, 2006). For another shallow eutrophic lake in southern Finland, Niemistö and Horppila (2007) calculated that a 60 d reduction in ice-cover period would increase annual sediment resuspension by 28%. The effect of one week ice cover change alone on internal loading of Tuusulanjärvi was thus not strong, but it also accelerated the effect of increasing wind velocities.

A major factor behind the increasing IP_{ox} was the artificial destratification as the largest changes in P cycling took place during the latest study period (1998–2010). During the first three study periods, IP_{anox} and IP_{anox}:IP_{tot} – ratio showed an increasing trend indicating that P release from anoxic bottoms significantly contributed to P cycling in Tuusulanjärvi and that aeration efforts had only minor effects. The large-scale Mixox-aeration in period IV had a strong effect on bottom O_2 and TP concentrations and on IP_{anox}. This was expected due to the disruption of summer stratification by the method. During this period, however, IP_{tot} increased, which was not solely due to changing weather conditions but destratification had also a role. Water temperature has an effect on P release (Marsden, 1989; Jensen and Andersen, 1992). The surface temperature of Tuusulanjärvi showed an increasing trend, which is a common phenomenon in European lakes and is connected to the climate change (Arvola et al., 2010). In deep water above the bottom, the change was however steeper. At the surface, the summer temperature difference between the periods I and IV was 1.8 °C, while in near-bottom water it was 2.8 °C. This was caused by destratification. The steepest rise (1.6 °C) took place between periods III and IV. Hence, especially the Mixox-aeration affected the water temperature. Such effect of aeration was due to the warm surface water pumped into the deep water. The strong warming effect of Mixox-aeration on hypolimnion during summer has been shown in other lakes as well (Holmroos et al., 2016; Kuha et al., 2016). In the whole-year temperature average, the rise in the bottom water was only 0.6 °C higher than at the surface, because in winter Mixox-aerators decrease the water temperature above the bottom (Salmi et al., 2014). Another mechanism by which aeration can enhance P release is mineralization of organic matter. Mineralization is more effective in aerobic than in anaerobic conditions (e.g. Marsden, 1989). Thus, aeration may increase the release of P bound to organic material. Additionally, turbulence created by aeration increases mineralization in the water by increasing the residence time of settling organic matter (Gantzer et al., 2009). Moreover, Mixox-aeration has only minor direct effects on sediment resuspension below the aerator outlet, but the turbulent eddies created by the aerator can affect sediment erosion at the edges of the aerated deep at least at 200 m distance from the aerators (Niemistö et al., 2016). Aeration can therefore enhance P cycling through effects on erosion and sediment focusing. Increasing water turbulence also affects the transfer of P-rich interstitial water to the water above the sediment (Marsden, 1989), a phenomenon accelerated by the upward migration of P in the sediment. Moreover, the aeration-induced disappearance of thermal stratification has made the water column less stable and larger areas prone to sediment resuspension by wind and wave forces.

The strong effects of Mixox-aeration compared with other used methods can be seen also by comparing IP_{ox} values during the different periods. IP_{ox} was considerably higher during period IV than during the previous periods. The negative value of IP_{ox} in period III suggests that during this period net sedimentation of P occurred in the aerobic parts of the lake. The explanation for this is found in the variability of the used aeration techniques. Both in periods III and IV, aeration efforts had significant positive effect on the near-bottom oxygen concentration. In the earlier period, however, temperature stratification was not broken and the aeration thus had weaker effects on hydrodynamics and sediment erosion than destratification during period IV. Additionally, the wind velocities during period III were lower than in period IV, and thus had less effects on P release from the shallow areas.

Sediment focusing must be taken into account when interpreting the R_{sed} estimates. R_{sed} values cannot be applied straightforwardly to the whole lake, because they were based on sediment sampled from the accumulation area. In such sediment cores, sediment focusing may affect the results (e.g. Owens et al., 2009). Sediment focusing is a common phenomenon in lakes (Hilton et al., 1986; Niemistö et al., 2012) and probably takes place also in Tuusulanjärvi. Overestimation of R_{sed} in relation to R_{mb} could cause overestimation of IP_{tot}. However, the sediment and P accumulation rates, as well as IP_{tot} estimates for Tuusulanjärvi were on a level reported from numerous eutrophic lakes (e.g. Marsden, 1989; Søndergaard et al., 1993; Andersen and Ring, 1999; Kelton and Chow-Fraser, 2005). In small lakes and in lakes with gentle slopes, both characteristics describing Tuusulanjärvi, the contribution of focusing to sedimentation rates is usually small (Bloesch and Uehlinger, 1986; Blais and Kalff, 1995). Sediment focusing thus did not considerably violate the comparison of the different periods. During 1998–2010, the effect of Mixox-aeration on sediment erosion could however cause overestimation of R_{sed} and IP_{tot} compared with other periods. On the other hand, increasing sediment erosion entails an increase in internal loading. Moreover, overestimation of R_{sed} during period IV due to increased sediment focusing is compensated by the P removed from the lake during biomanipulation. Export of P with fish harvesting increases the value of TP_{out} (Nürnberg et al., 2012). Thus inclusion of P removed from the lake during the biomanipulation decreases the value of R_{sed} and consequently increases IP_{tot} estimates during period IV in relation to the other periods. The removed fish biomass corresponds to annual P removal of 47 mg m^{-2}, assuming that the P content of fish is 0.6% of fresh mass (Penczak et al., 1985).

R_{sed} values exceeded TP_{in} in periods II–IV. This was not due to sediment focusing. R_{sed} values in relation to TP_{in} increased considerably after period I, when no factor could have caused increased sediment erosion from the shallow areas. Wind velocities during period I were lower and ice-cover periods longer than during the later study periods, and no destratification took place at the time. Additionally, Dillon and Evans (1993) found the same phenomenon from numerous lakes including lakes where sediment cores from multiple depth zones were taken. The elevated R_{sed} – values after period I were rather caused by the post-depositional mobility of P in the sediment. The diffusion of P is dependent on P concentration gradients (Berner, 1980; Søndergaard et al., 2003; Tammeorg et al., 2016). Thus, upward diffusion of P increased when external flux of P to the lake was strongly reduced in 1979. Past external loading can thus affect internal nutrient loading even after a long time lag. This affected R_{sed} – values, but did not violate the estimates of internal loading, as upward diffusion in the sediment also accelerates internal loading by increasing diffusion of P out of the sediment when the overlying water is anoxic and bringing P to the surface sediment vulnerable to disturbances (Carignan and Flett, 1981; Marsden, 1989). R_{sed} – values represent accumulation of P in the sediment, and upward migration of P is one form of accumulation (Boers et al., 1998).

While the whole-year average of surface TP concentration has not changed after the sewage diversion, the growing-season concentration decreased considerably during period IV. The reduction in the near-bottom TP can be attributed to artificial destratification and consequent improvement of oxygen conditions in the deepest part of the lake. The concentration at the surface was, however, affected by different factors. In shallow lakes, TP concentration often rises during the growing season due to elevated temperature, high pH values and activities of the biota (e.g. Phillips et al., 1994; Søndergaard and Jeppesen, 1999). In Tuusulanjärvi, pH values high enough to promote effective P release (>9, Andersen, 1975; Holmroos et al., 2009) were rare, pH showed a decreasing trend and water temperature has increased. The factor behind the summer TP concentration decrease is thus most likely the biomanipulation that was started in 1997. Due to the effective fishing, since 1997 the fish biomass in the pelagic area has decreased from 310 to 60 kg ha^{-1} (Malinen and Vinni, 2015). Fish transport P both vertically and horizontally (Brabrand et al., 1990; Vanni, 1996) and the biomass reduction has therefore diminished the effects of fish activities on P cycling during the growing season. Similar effect of fish on biomass reduction has been reported from numerous lakes. In the nearby Lake Vesijärvi, for instance, the summertime increment of TP concentration was delayed along with fish biomass reduction, which was connected to reduction in the amount of P transported by fish (Horppila, 1998; Horppila et al., 1998). The impact of food web management in Tuusulanjärvi was also indicated by chl:TP relationship, which was lowest in 1998–2010. As no changes in surface SRP were detected, the decrease in the chl:TP relationship indicated that grazing rate by zooplankton has increased (Mazumder, 1994). This conclusion is supported also by the increased biomass and individual size of cladocerans in Tuusulanjärvi during years of intensive fish removal and the persistent abundance of large cladocerans (Rask et al., 2005; Ketola, 2015). The mean size of cladocerans often increases when the size-selective predation pressure by planktivorous fish is relaxed (Carpenter and Kitchell, 1988; Jeppesen et al., 2007). Such effect of effective food web management has previously been reported from many lakes (Horppila et al., 1998; Sarvala et al., 2000; Jeppesen and Sammalkorpi, 2002). It thus seems that food web management has compensated for the amplifying effects of destratification on P-cycling.

5. Conclusions

The reduction of external P loading of Tuusulanjärvi in 1979 did not result in water quality improvement during the following three decades because external loading was still too high. Additionally, internal loading of P did not decrease but rather showed an increasing trend during the 2000's. The large-scale artificial aeration efforts started in late 1990's have affected the P dynamics, but the effects have not been unambiguously positive. While anaerobic P release from the sediment has decreased, the results suggested that the total internal P loading has increased, which is due to the enhanced P release from areas where the water column is oxic. The increase in this P flux was probably attributed to both artificial destratification, and changing weather conditions. Biomanipulation has compensated for the consequences of enhanced P cycling for the phytoplankton biomass. Further reduction of external P loading is the first priority in the management of the lake and after that internal P loading may cause a delay in water quality improvement for decades. Decreasing the destratification efforts would be worth trying, as it could improve the situation and/or reduce the need for biomanipulation.

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