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Hypolimnetic aeration intensifies phosphorus recycling and increases organic material sedimentation in a stratifying lake: Effects through increased temperature and turbulence

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The effect of hypolimnetic aeration on the sedimentation of inorganic and organic material as well as phosphorus was examined in a spatially comprehensive investigation in a dimictic northern temperate lake in southern Finland. Two years of aeration strongly increased the gross sedimentation of dry matter (68% ± 2%, mean ± SD) and phosphorus (87% ± 7%) in the aerated, deep areas. Although the organic content of the settling material decreased, the total amount of organic matter reaching the lake bottom increased by 53% ± 6%. Also the shallow areas were affected, although to a lesser extent. We suggest that the observed increases were due to an aeration-induced increase in the hypolimnetic water temperature and turbulence, which not only increased the mineralization of organic matter but also phosphorus recycling, and consequently the production of excess organic material.

Introduction

Hypolimnetic aeration is a rehabilitation measure commonly used to hinder the redox-dependent internal nutrient loading of phosphorus (P) in oxygen-depleted aquatic systems (Cooke *et al.* 2005, Singleton and Little 2006). This measure is based on the fact that under oxic conditions, the surface sediment is able to efficiently bind P with ferric oxyhydroxides (Mortimer 1942, Boström *et al.* 1982). Since in lake ecosystems P usually is the main limiting nutrient (e.g. Schindler 1978), a decrease in its concentration should lead to lower primary production and thereby to lower sedimentation of organic material (e.g. Matthews and Effler 2006a). This, in turn, should reduce oxygen consumption at the lake bottom, and therefore lead to the desired result of a decreased internal loading of redox-dependent P (Cooke *et al*. 2005, Matthews and Effler 2006a).

Two basic methods to reoxygenize anoxic hypolimnions are the use of bubble-plume diffusers that inject air or pure O_2 into the hypolimnetic water and the pumping of O_2-rich surface water below the thermocline (Cooke *et al*. 2005, Lappalainen and Lakso 2005, Gafsi *et al*. 2009). In the latter approach, the aim is to transport O_2-rich epilimnetic water down to the hypolimnion to alleviate anoxia and hypoxia without breaking the thermocline. The effects of aeration on nutrient concentrations and the sediment retention capacity are somewhat contradictory, since non-significant (Schauser and Chorus 2007), fairly limited (Gächter and Wehrli 1998, Liboriussen

Editor in charge of this article: Johanna Mattila *et al*. 2009) and positive (Ashley 1983, Höhener and Gächter 1994, Bryant *et al*. 2011) effects have been published. This suggests that many factors other than the hypolimnetic oxygen concentration or oxygenated sediment surface may determine the P retention capacity of the sediment. Such factors may include for instance the P concentration in the water column, molar ratios of settling elements and P diagenesis in the sediment (Gächter and Müller 2003, Moosmann *et al*. 2006, Hupfer and Lewandowski 2008). Moreover, knowledge of the aeration-induced factors that possibly counteract the intended effects of nutrient retention is limited. Such counteractive effects of hypolimnetic aeration, which may further affect the sedimentation and nutrient dynamics of a lake, potentially include the increased temperature of hypolimnetic water, increased turbulence in the water column and decreased stability of the waterbody during stratification periods (Imboden and Wüest 1995, Etemad-Shahidi and Imberger 2001, Gantzer *et al*. 2009). When hypolimnetic aeration is conducted by pumping warm epilimnetic water down to the hypolimnion, the temperature increase in the hypolimnion may be very strong (Toffolon *et al*. 2013, Salmi *et al*. 2014). Bubble plume diffusers are also known to affect the hypolimnetic temperature, although to a lesser extent (Gantzer *et al*. 2009). The increased temperature, in turn, affects oxygen consumption in the water column and within the sediment due to increased bacterial activity (Rose 1967, Gantzer *et al*. 2009, Bergström *et al*. 2010). Additionally, the greater supply of dissolved oxygen may increase its consumption (Beutel *et al*. 2007), although not always (Gantzer *et al*. 2009). Oxygen consumption may additionally be enhanced in both the water column and surface sediment by the increased turbulence created by the aerators (Ashley 1983, Beutel *et al*. 2007, Gantzer *et al*. 2009).

Turbulent eddies, which redistribute nutrients, gases and organisms, may be of crucial importance for primary productivity and biogeochemical processes in aquatic ecosystems (Jellison and Melack 1993, MacIntyre 1993, Imboden and Wüest 1995). In addition to affecting oxygen consumption, increasing turbulence may increase the vertical transport of materials and nutrients in boundary layers such as the sediment–water interface, thermocline and pycnocline (Etemad-Shahidi and Imberger 2001, MacIntyre and Jellison 2001). The warming of hypolimnetic water reduces the stability of the water column, as it decreases the density differences between the hypolimnion, metalimnion and epilimnion (Imboden and Wüest 1995). If the stability of the water column is strongly reduced during the stratification period, wind-induced sediment resuspension may occur more often and result in enhanced nutrient cycling. Altogether, it is possible that contrary to the original aim, hypolimnetic aeration may increase nutrient recycling and primary productivity, thereby resulting in more organic material settling on sediments and an increase in oxygen consumption.

This study was conducted in the Enonselkä basin of Vesijärvi, a lake in southern Finland, where hypolimnetic aeration was started in 2010 to reduce the internal P loading from anoxic and hypoxic areas. The study focused on the possible effects of hypolimnetic aeration on the sedimentation and resuspension rates of particulate organic material and P through altered temperature and turbulence. In order to determine whether the aeration affected the amount and quality of settling material, the sedimentation of inorganic and organic material as well as P was measured in a spatially comprehensive manner and resuspension rates were calculated in this basin-scale study. The measurements were conducted during the preceding year (2009) and two following years (2010–2011) after the initiation of hypolimnetic aeration. We hypothesized that due to aerator-induced warming of hypolimnetic water and more turbulent conditions, the mineralization of organic material and recycling of P would increase.

Methods

Study site

Enonselkä is a relatively constrained basin of Vesijärvi — a lake in southern Finland (61°01´N, 25°35´E) — with a surface area of 26 km^2 and a mean depth of 6.8 m (Fig. 1). Due to the shallowness of the basin, most of its sediment is in direct contact with epilimnetic water (depth < 10 m, 83% of the total area) during the stratification period (June–September) (Fig. 1). Enonselkä basin became eutrophic due to sewage effluent that originated from the city of Lahti until the late 1970s (Keto and Sammalkorpi 1988). Due to the massive cyanobacterial blooms, restoration activities were implemented (biomanipulation, diffuse load reduction) in the late 1980s and early 1990s (Kairesalo *et al*. 1999). For a period of approximately one decade the water quality improved, but cyanobacterial blooms reoccurred in the early 2000s (Keto *et al*. 2005). During the stratification periods, the total P concentration varies vertically from the epilimnetic value of 20 μg l^{-1} up to 160 μg l^{-1} in the hypolimnion. The respective variation for soluble reactive P is from 2 to 55 μg l^{-1} (Niemistö *et al*. 2012). The hypolimnion is depleted in O_2 during the stratification periods in summer and soluble P accumulates in the lake deeps. In order to reduce this P accumulation, i.e. the internal loading, hypolimnetic aeration was implemented at the beginning of 2010. Nine water pumps (8 Mixox MC-1100 2.5 kW and one Mixox MC-750 1.5 kW, Water-Eco Ltd., Kuopio, Finland) were installed in the deeps of the Enonselkä basin (Fig. 1) to pump O_2-rich epilimnetic water down to the hypolimnion. Water from 3 m depth was pumped down to the hypolimnion and the outlets of the pumps were located 8–10 m above the bottom to avoid resuspension. The pumps were turned on in winter under the ice cover and in summer during the stratification period, approximately from mid-June to late August (Salmi *et al*. 2014). The functioning of the pumps is described in detail in Lappalainen (1994) and Bendtsen *et al*. (2013).

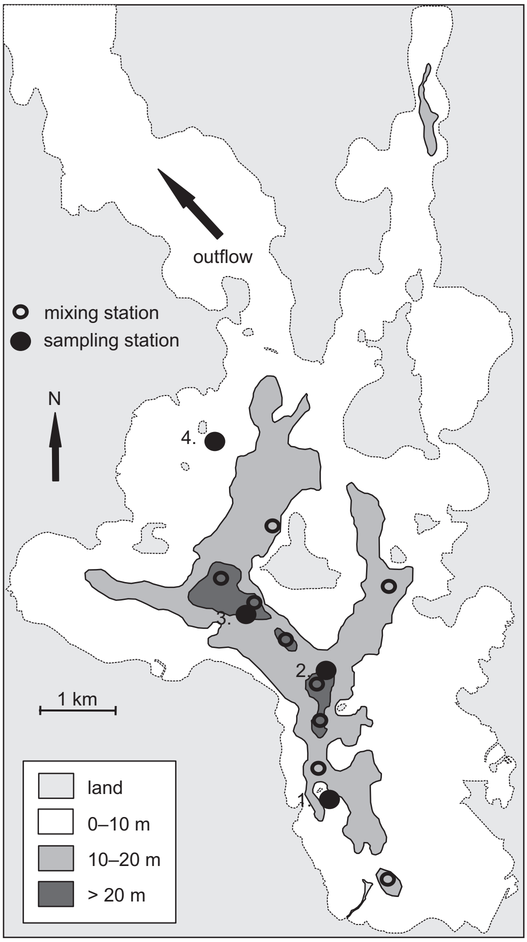

Fig. 1. Map of the study area indicating locations of the sampling sites and aerators.

Gross sedimentation and resuspension

During the open-water seasons of 2009, 2010 and 2011, gross sedimentation (GS) was measured at four stations (Fig. 1) with four replicate cylindrical sediment traps (diameter = 5.4 cm, height = 41 cm) suitable for lake conditions (Bloesch and Burns 1980) deployed 2 m above the lake bottom. The water depths at shallow stations 1 and 4 that do not undergo stratification were 8 m and 6 m, while those at stations 2 and 3 that undergo stratification were 30 m and 28 m, respectively. The traps were kept in the lake from mid-May to the beginning of November and emptied at 14- to 21-day intervals. The dry weight (DW) of entrapped material was measured after drying the samples at 105 °C for three days and the organic fraction (f_{GS}) was determined by loss on ignition at 550 °C for 2 h (in duplicate from each trap, homogenized samples). For the resuspension calculation, the organic content of the surface sediment (f_R) and seston (f_T) was similarly determined. Sediment resuspension (R) was calculated with the label method (Gasith 1975) according to the following equation:

$$R = \mathrm{GS}\frac{f_{\mathrm{GS}} - f_{\mathrm{T}}}{f_{\mathrm{R}} - f_{\mathrm{T}}}, \quad (1)$$

where GS is the gross sedimentation

(g DW m^{-2} d^{-1}), f_{GS} is the organic fraction of GS (%), f_R is the organic fraction of the surface sediment (%), and f_T is the organic fraction of the seston, collected from the water column (%).

GS (g DW m^{-2} d^{-1}) was divided into inorganic (SPIM) and organic (SPOM) settling particulate matter (g DW m^{-2} d^{-1}) to observe possible differences between these fractions (SPOM = $f_{GS}/100 \times$ GS, SPIM = GS – SPOM). The gross sedimentation of P (PGS) (mg m^{-2} d^{-1}) was calculated by multiplying the mean P concentration of the entrapped material by the GS rate at each sampling location.

Spatial and spatio-temporal average rates of GS, PGS, SPIM and SPOM

Since sedimentation rates measured in lake deeps only may markedly differ from those occurring in shallow areas due to sediment focusing (Ohle 1962, Niemistö *et al*. 2012), and the effect of the aerators on the sedimentation rates in shallow areas were also a concern, the rate measurements and calculations were conducted in a spatially comprehensive manner. The lake area was divided into shallow (depth < 10 m, unstratified, aerobic water column, stations 1 and 4) and deep areas (depth 10–30 m, stratified June–September, anoxic/suboxic hypolimnion, stations 2 and 3) (Fig. 1) when assessing the amount of GS and PGS settling in the separate areas. Shallow areas constituted 83% and deep areas 17% of the total basin area. The spatially comprehensive seasonal mean (hereafter the spatial mean) of GS as well as SPIM, SPOM and PGS was calculated as follows: mean rate of shallow sites × coverage + mean rate of deep sites × coverage. Since the trap exposure periods were not exactly identical within and between the years and gross sedimentation may show large temporal variation, the spatio-temporal means for each study year were calculated to enable comparisons between them. The sedimentation value of each station was weighted by each exposure period and summed up to a cumulative sedimentation value for the whole study period. The spatio-temporal mean rate for each parameter was then calculated by dividing the cumulative sedimentation value by the total length of the trap exposure period which was 176 days for each year. The spatio-temporal mean rates were calculated for the whole lake as well as for the deep and shallow areas separately.

Phosphorus concentration of entrapped material

Surface sediment samples (topmost 0–1 cm, duplicate samples) were collected with an HTH gravity corer (inner diameter 86 mm) (Renberg and Hansson 2008) from each station on every sampling occasion when the sediment traps were emptied. The P concentration of entrapped material and surface sediment (samples dried at 60 °C) was measured (in duplicate) after wet digestion with nitric acid and hydrogen peroxide in a microwave digestion system using inductively coupled plasma optical emission spectrometry (ICP-OES, Thermo Scientific iCAP 6000) (CEM Mars 5) in 2009. In 2010 and 2011, the determinations (in duplicate) were conducted after wet digestion with sulphuric acid and hydrogen peroxide in the microwave digestion system using the molybdenum-blue–ascorbic-acid method (Lachat autoanalyzer, QuickChem Series 8000).

Turbulence measurements

The possible effect of the Mixox aerators (MC-1100) on turbulence was measured in 2013 with an acoustic microstructure turbulence profiler (MSS90, ISW Wassermesstechnik, Germany) at station 2 (Fig. 1) at four distances (20, 100, 150 and 200 m) from the aerator. Ten vertical profiles at 0.5-m intervals (from the depth of 2 m to the bottom) were measured at each location with the aerator on and off. The measurement points were positioned on five, evenly-spaced, transect lines extending outwards from the aerator (two measurements were made at each point). The turbulence with the aerator turned off was measured on 10 June, 11 July and 29 July; the aerator was always turned off for at least for 2 days prior to the measurements. The turbulence measurements with the aerator turned on were conducted on 24 June, 8 July, 1 August and 22 August. As the turbulence measurements required breaks in water pumping, they were conducted during a different year than the sedimentation measurements to enable a continuous effect of the aerator on sedimentation rates at station 2. The dissipation rates of turbulent kinetic energy (ε, m^2 s^{-3}) were calculated as follows (e.g. Osborn 1980):

$$\varepsilon = 7.5\nu \times \overline{\left(\frac{du}{dz}\right)^2}, \quad (2)$$

where ν is the kinematic viscosity (a constant value of 10^{-6} m^2 s^{-1} was used), and du/dz is the velocity shear.

Eddy diffusivity (exchange coefficient) K was calculated as follows (Lilly *et al*. 1974, Osborn 1980, Oakey 1982):

$$K = \gamma \times \varepsilon/N^2, \quad (3)$$

where γ is the mixing efficiency (a value of 0.2 was used; *see* Osborn 1980), N is the Brunt-Väisälä Frequency (buoyancy frequency) (s^{-1}). The strength of density stratification (N^2) was calculated as follows:

$$N^2 = g/\rho \times d\rho/dz, \quad (4)$$

where g is the gravitational acceleration, ρ is the water density, and z is the water depth.

In order to evaluate the possibility of vertical transport of solutes and particles due to aerator-induced turbulence, the ratio ($\varepsilon/\nu N^2$) of the dissipation rate of turbulent kinetic energy (ε) to the counteracting forces that occur due to kinematic viscosity (ν) and buoyancy frequency (N^2) was calculated according to Ivey and Imberger (1991) and Itsweire *et al*. (1993), who have defined that the vertical transport can take place when the ratio is greater than 15. According to the temperature profiles (not shown) recorded at the same time as the turbulence measurements, the water column was divided into five layers: the epilimnion (2–6 m), thermocline (6.5–10 m), upper hypolimnion (10.5–20 m), lower hypolimnion (20.5 m–bottom) and 0.5 m above the bottom.

Rainfall, external phosphorus load, winds and water temperature

Data on rainfall as well as wind velocity (maximum wind velocity, daily averages of gusts recorded every hour) and wind direction for the study periods were obtained from the Laune meteorological station (Finnish Meteorological Institute) situated 3 km from the Enonselkä basin. Since a single storm or strong wind event may affect the redistribution of bottom sediment on annual scale (Bentgsson *et al*. 1990), the maximum winds speeds were of interest. Values for the external load (the point source load sampled weekly, the diffuse load from different land use types modeled, and atmospheric precipitation taken into account; VEPS, nutrient load assessment system; Tattari and Linjama 2004) were obtained from Lahti Region Environmental Services.

At each station, the vertical temperature profiles were measured with a YSI-6600 multiparameter water quality sonde (YSI Corporation, Yellow Springs, OH, USA) during each sampling occasion. Additionally, temperature data (hourly measurements) from the depths of 10 and 30 m at station 2 were obtained from a measurement float maintained by the Lahti Region Environmental Services.

Statistical analysis

The spatial and temporal variation of sedimentation may be high (e.g. Rosa *et al*. 1985, Bloesch and Uehlinger 1986). Additionally, no abrupt, short-term effects but the effects on the seasonal cumulative sedimentation rates were expected to occur due to aeration. Therefore, the spatio-temporal mean rates were calculated (i.e., cumulative sedimentation rate values divided with the total trap exposure period; rates calculated for the deep and shallow areas and for the whole lake area) and tested. Due to high spatial variation, the shallow and deep areas were also tested separately. The differences in spatio-temporal means of GS, PGS, SPIM, SPOM and R as well as R(%) and f_{GS}, among the study years, were tested using one-way ANOVA. Paired comparisons were conducted with a two-sample t-test with the Bonferroni correction (SAS Institute Inc. 2008). The normality of the data sets was verified with the Shapiro-Wilk test. All the non-normally distributed data (raw sedimenta-

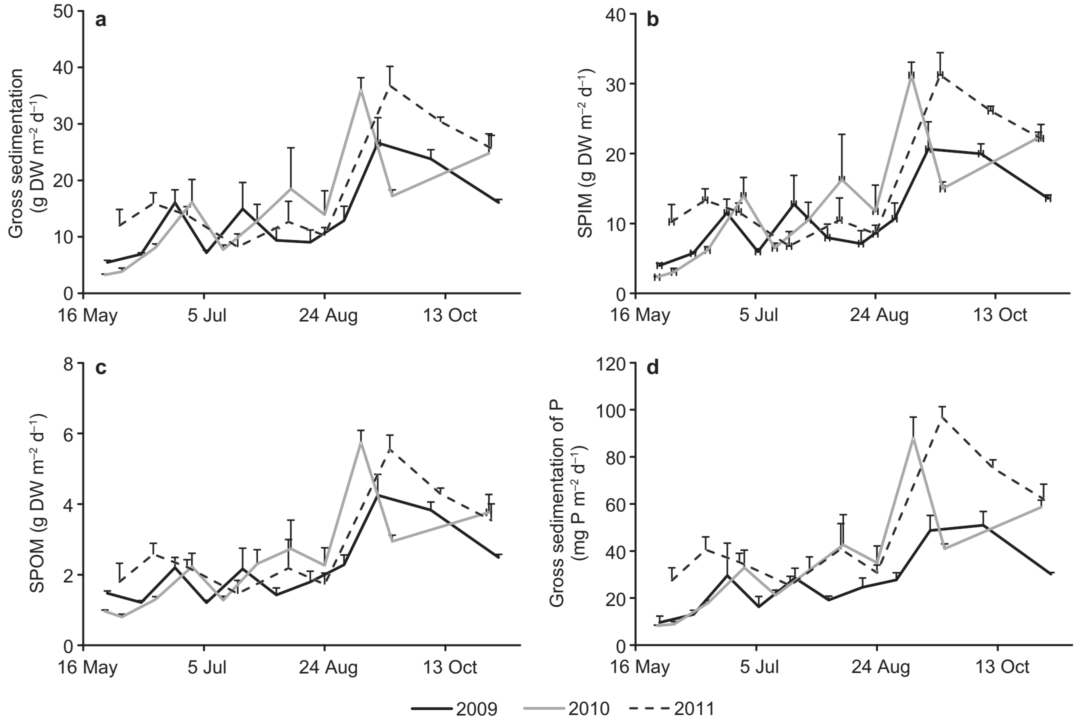

Fig. 2. Spatial mean (+ SE) rates of (**a**) gross sedimentation, (**b**) gross sedimentation of inorganic material, (**c**) organic material, and (**d**) phosphorus. Note the different scales of the *y*-axes.

tion data) were tested with Kruskal-Wallis analysis (paired comparisons conducted with Dwass, Steel, Critchlow-Fligner Method, DSCF) (SAS Institute Inc. 2008). The differences in ε and K between having the aeration pump either on or off were tested using a two-sample *t*-test (SAS Institute Inc. 2008). The differences in rainfall, wind speed, the P concentration of entrapped material, and water temperature among the study years were also tested using one-way ANOVA, and paired comparisons were conducted with a two-sample *t*-test with the Bonferroni correction (SAS Institute Inc. 2008). The values for resuspension ($R\%$) and the organic content of entrapped material (f_{GS}) expressed as a percentage of the total flux were arcsine-transformed before analysis.

Results

Gross sedimentation and resuspension

The rates of GS showed large temporal and spatial variation within the study years. During the reference year (2009), the spatial mean rates of GS (weighted by the coverage of the deep and shallow areas) varied between 5.5 and 26.6 g DW m^{-2} d^{-1}. After the initiation of aeration, the corresponding range was 3.3–36.0 g DW m^{-2} d^{-1} in 2010, and 8.2–36.8 g DW m^{-2} d^{-1} in 2011 (Fig. 2). Despite the temporal (Fig. 2) and spatial variation (rates lower in shallow areas; *see* Table 1) during the study years, the seasonal pattern of the spatial mean rate was very similar in each year. GS peaked around mid-June and in early autumn, having the highest values during the autumn turnover in 2010 and 2011 (Fig. 2). The spatial mean rates of SPIM and SPOM followed the pattern of GS, and SPIM was consistently higher than SPOM (range in 2009 for SPIM 4.0–20.6 g DW m^{-2} d^{-1} and for SPOM 1.2–8.4 g DW m^{-2} d^{-1}, in 2010 SPIM: 2.3–31.2 g DW m^{-2} d^{-1}; SPOM: 0.8–5.8 g DW m^{-2} d^{-1}, in 2011 SPIM: 6.8–31.2 g DW m^{-2} d^{-1}; SPOM: 1.5–5.6 g DW m^{-2} d^{-1}) (Fig. 2).

The spatio-temporal mean rates for GS, SPIM and SPOM (i.e., cumulative sedimentation data divided with the total trap exposure period) were clearly higher after the initiation of the aeration in the whole lake, but due to high spatial variation between the shallow and deep areas the increase was not statistically significant (Tables 1 and 2). However, when the spatio-temporal data from the deep and shallow stations were analyzed separately, the rates of GS, SPIM and SPOM increased significantly from 2009 to 2011 in the deep area: GS by 68%, SPIM by 71%, and SPOM by 53% (Tables 1 and 2, *t*-test with the Bonferroni correction: $t_3 = 4.86$, $p < 0.05$). f_{GS} (i.e. SPOM as a percentage value of GS) decreased from 2009 (18%) to 2011 (16%) (Table 2, *t*-test with the Bonferroni correction: $t_3 = 4.86$, $p < 0.05$).

As for the whole lake data, the spatio-temporal means of different sedimentation rates did not differ statistically among years in the shallow areas although the values seemed to increase during the years when aeration was in use (Tables 1 and 2). Compared with the deep areas, the variation among the shallow stations was higher in respect to the increase in mean rates (Table 1). However, when the raw data (non-normally distributed) of the shallow areas were tested, the median values for different sedimentation rates were significantly higher in 2011 than in 2009. The median values for the GS and SPIM rates increased from 8.6 to 11.2 g DW m^{-2} d^{-1} and from 7.0 to 9.7 g DW m^{-2} d^{-1}, respectively (Table 3, DSCF-method: GS, DSCF = 3.67, $p = 0.026$; SPIM: DSCF = 3.63, $p = 0.029$). The increase of the median value of the SPOM rate, from 1.6 to 1.8 g DW m^{-2} d^{-1}, was very close to a statistical significance, $p = 0.051$ (Table 2). The median value of f_{GS} decreased from 16.6% in 2009 to 15.4% in 2011 (DSCF = 4.76, $p = 0.002$).

Sediment resuspension constituted most of GS, varying from 82.8% to 99.7% in 2009, from 66.7% to 99.4% in 2010 and from 76.5% to 98.9% in 2011. The resuspension rate as dry matter (R) or the rate as a percentage of the GS rate (R%) (spatio-temporal average rate for deep areas and the median rate for shallow areas) did not differ among the study years (Table 2).

Gross sedimentation of phosphorus and P content of the entrapped material

The rates of PGS strictly followed the pattern of GS (Fig. 2), and in the deep areas, significantly higher rates were recorded in 2011 (range: 27.5–96.8 mg P m^{-2} d^{-1}) than in 2009 (range: 9.6–51.0 mg P m^{-2} d^{-1}) the mean rate showing 87% increase (Tables 1 and 2, *t*-test with the

Table 1. The spatio-temporal mean rates (± SD) (i.e. cumulative sedimentation rate divided by total trap exposure period) of gross sedimentation (GS), gross sedimentation of inorganic material (SPIM), organic material (SPOM) and phosphorus (PGS) for the whole basin and for the deep areas and shallow areas.

	2009	2010	2011	Increase from 2009 to 2011 (%)
Mean rates of the whole basin				
GS (g DW m^{-2} d^{-1})	13.9 ± 4.1	16.9 ± 3.7	20.0 ± 2.7	44
SPIM (g DW m^{-2} d^{-1})	11.6 ± 3.5	14.3 ± 5.6	16.9 ± 2.5	46
SPOM (g DW m^{-2} d^{-1})	2.3 ± 0.5	2.6 ± 0.7	3.0 ± 0.3	31
PGS (mg P m^{-2} d^{-1})	28.8 ± 6.8	37.0 ± 7.7	48.9 ± 4.5	70
Mean rates of the deep areas				
GS (g DW m^{-2} d^{-1})	27.7 ± 1.9	35.0 ± 5.2	46.5 ± 0.6	68
SPIM (g DW m^{-2} d^{-1})	23.1 ± 1.6	29.6 ± 4.1	39.5 ± 0.5	71
SPOM (g DW m^{-2} d^{-1})	4.6 ± 0.4	5.4 ± 0.7	7.0 ± 0.3	53
PGS (mg P m^{-2} d^{-1})	68.8 ± 7.7	88.4 ± 20.4	128.7 ± 4.7	87
Mean rates of the shallow areas				
GS (g DW m^{-2} d^{-1})	11.2 ± 4.5	13.2 ± 3.4	14.6 ± 3.1	30
SPIM (g DW m^{-2} d^{-1})	9.3 ± 3.8	11.1 ± 6.0	12.3 ± 2.9	33
SPOM (g DW m^{-2} d^{-1})	1.9 ± 0.5	2.1 ± 0.7	2.3 ± 0.3	25
PGS (mg P m^{-2} d^{-1})	20.6 ± 6.6	26.4 ± 5.1	32.5 ± 4.5	58

Bonferroni correction: t_3 = 4.86, p < 0.05). In shallow areas, the increase in spatio-temporal mean was not significant, but the test on the raw data revealed a marked and significant increase in the median value of the PGS rate; from 15.7 g DW m^{-2} d^{-1} in 2009 to 22.8 g DW m^{-2} d^{-1} in 2011 (Tables 2 and 3, DSCF-method: DSCF = 5.93, p < 0.001). The P content of the entrapped material increased after the initiation of aeration at each sampling station (t-test with the Bonferroni correction: station 1, t_{28} = 2.54, p < 0.05; station 3, t_{24} = 2.57, p < 0.05; station 5, t_{27} = 2.55, p < 0.05), with the exception of station 2 (p > 0.05) (Table 4).

Table 2. Results of one-way ANOVA (spatio-temporal data tested) and Kruskal-Wallis (K-W, on raw data of shallow areas) showing the differences in the amount and quality of the settling and resuspended material among the study years 2009, 2010 and 2011. GS = gross sedimentation rate, SPIM = gross sedimentation rate of inorganic material, SPOM = gross sedimentation rate of organic material, PGS = gross sedimentation rate of phosphorus, f_{GS}, organic content of entrapped material, R = resuspension rate; R% = resuspension rate as a percentage of GS.

	df	F	p
All data (ANOVA)			
GS	2, 9	0.59	0.572
SPIM	2, 9	0.64	0.549
SPOM	2, 9	0.51	0.615
PGS	2, 9	0.96	0.418
f_{GS}	2, 9	1.82	0.216
R	2, 9	0.42	0.671
R%	2, 9	0.13	0.879
Deep areas (ANOVA)			
GS	2, 3	16.60	0.024
SPIM	2, 3	20.18	0.018
SPOM	2, 3	17.50	0.022
PGS	2, 3	13.68	0.031
f_{GS}	2, 3	13.13	0.033
R	2, 3	8.80	0.055
R%	2, 3	0.50	0.649
Shallow areas (ANOVA)			
GS	2, 3	0.29	0.765
SPIM	2, 3	0.31	0.756
SPOM	2, 3	0.24	0.799
PGS	2, 3	3.45	0.167
f_{GS}	2, 3	0.46	0.668
R	2, 3	0.29	0.766
R%	2, 3	0.41	0.695
Shallow areas (K-W)			
GS	2	8.12	0.017
SPIM	2	7.87	0.020
SPOM	2	5.95	0.051
PGS	2	25.22	< 0.001
f_{GS}	2	11.23	0.004
R	2	4.88	0.087
R%	2	4.60	0.100

Dissipation rate and eddy diffusivity

Apart from the epilimnion (2–6 m), the dissipation rates (ε) increased significantly due to the aerator action in all of the measured water layers at the distance of 20 m from the aerator (Tables 5 and 6). The effect was strongest in the upper hypolimnion (10.5–20 m) ($\log_{10}\varepsilon$ from –8.72 to –7.80 m^2 s^{-3}) and 0.5 m above the bottom ($\log_{10}\varepsilon$ from –8.47 to –7.97 m^2 s^{-3}). At the other measurement distances, the effect of the aerator was smaller, but significant effects were observed in the whole hypolimnion and metalimnion (6.5–10 m) at the distances of 100 and 150 m from the aerator, and also in the metalimnion at the distance of 200 m from the aerator (Tables 5 and 6). Apart from the measurements close to the bottom, K increased in all the water layers below the epilimnion due to the aerator at the distance of 20 m from the aerator (Tables 5 and 6). The highest increase occurred in the upper hypolimnion ($\log_{10}K$ from –5.27 to –4.31 m^2 s^{-1}) and metalimnion ($\log_{10}K$ from –6.20 to –5.77 m^2 s^{-1}). A significant, albeit lower, increase was also found in the upper hypolimnion and metalimnion at the distance of 100 m from the aerator and in the metalimnion at the distances of 150 and 200 m from the aerator. Significant differences in the values of ε and K were also found for the epilimnion between the periods when the aerator was either on and off, but the values were not consistently higher when the aerator was on.

The turbulence profile measurements indicated that when the aerator was on, a threshold $\varepsilon/\nu N^2$ > 15 was achieved at the 20 m distance from the aerator in the whole hypolimnion as well as in the metalimnion (Fig. 3). Additionally, $\varepsilon/\nu N^2$ was affected at every measurement distance, and it was smaller than 15 at only a few depths of the profiles while the aerator was on. When the aerator was turned off, a more stable thermocline was clearly seen at the depth of 6.5–10 m indicated by the $\varepsilon/\nu N^2$ values smaller than 15 (Fig. 3).

Table 3. The median values and ranges of gross sedimentation (GS), gross sedimentation of inorganic material (SPIM), organic material (SPOM) and phosphorus (PGS) for the shallow areas.

	2009		2010		2011	
	Median	Range	Median	Range	Median	Range
GS (g DW m^{-2} d^{-1})	8.6	2.3–29.3	7.9	2.0–30.0	11.2	2.2–31.4
SPIM (g DW m^{-2} d^{-1})	7.0	1.8–24.7	6.4	1.6–36.3	9.7	1.3–27.3
SPOM (g DW m^{-2} d^{-1})	1.6	0.7–4.8	1.3	0.5–3.7	1.8	0.8–4.4
PGS (mg P m^{-2} d^{-1})	15.7	4.8–43.2	19.8	4.4–57.1	22.8	8.9–69.9

Rainfall, winds, water temperature and external phosphorus load

The average monthly rainfall was 44, 40 and 48 mm in the years 2009, 2010 and 2011, respectively, and no statistical differences among the study years were found (ANOVA: $F_{2,21} = 0.38$, p = 0.63). The highest peaks that clearly exceeded the average rainfall and occurred during the trap-exposure periods were 112 mm in July 2009 and 84 mm in September 2011 (Fig. 4).

Maximum wind velocities did not differ among the study years (ANOVA: $F_{2,534} = 0.37$, p = 0.69), and no exceptionally-high wind velocities were recorded. The highest maximum wind velocities (≤ 7 m s^{-1}) were measured in June and August 2010 (Fig. 4).

At station 2, at the depths of 10 and 30 m, the water temperature gradually increased during the open-water seasons of the study years until late August and early September, decreasing thereafter (Fig. 5). At the 10-m depth, no difference in the mean temperatures among the study years was found. At the 30-m depth, the increase in the mean temperature during the study period relative to 2009 was 2.29 °C in 2010 and 3.16 °C and in 2011 (t-test with Bonferroni correction: $t_{501} = 2.40$,

Table 4. The seasonal mean (± SD) concentrations (mg g^{-1}) of phosphorus in settling material at each station in 2009–2011.

Station	2009	2010	2011
1	1.95 ± 0.30	2.40 ± 0.21	2.68 ± 0.46
2	2.57 ± 0.87	2.92 ± 0.48	3.20 ± 0.79
3	2.39 ± 0.58	2.74 ± 0.32	3.32 ± 0.65
4	1.79 ± 0.62	2.11 ± 0.16	2.43 ± 0.30

Table 5. Dissipation rates of kinetic energy (ε) and eddy diffusivity (K) in different water layers and at different distances from the aerator which was either on or off.

Water layer	Distance from aerator (m)							
	20		100		150		200	
	On	Off	On	Off	On	Off	On	Off
Log$_{10}$ ε (m^2 s^{-3})								
2–6 m	–7.86	–7.89	–7.84	–8.10	–8.21	–7.44	–7.62	–8.11
6.5–10 m	–8.19	–8.59	–8.40	–8.69	–8.46	–8.59	–8.46	–8.68
10.5–20 m	–7.80	–8.72	–8.47	–8.74	–8.64	–8.55	–8.58	–8.63
20.5–30 m	–8.12	–8.74	–8.58	–8.74	–8.64	–8.52		
0.5 m from the bottom	–7.97	–8.65	–8.47	–8.58	–8.50	–8.56	–8.50	–8.58
Log$_{10}$*K* (m^2 s^{-1})								
2–6 m	–4.76	–5.39	–4.83	–5.55	–5.48	–4.56	–4.62	–5.51
6.5–10 m	–5.77	–6.20	–6.02	–6.25	–5.88	–6.27	–6.06	–6.23
10.5–20 m	–4.31	–5.27	–5.04	–5.31	–5.29	–5.10	–5.26	–5.20
20.5–30 m	–4.07	–4.38	–4.74	–4.31	–5.09	–4.35		
0.5 m from the bottom	–3.93	–4.09	–4.57	–4.41	–4.88	–4.55	–4.94	–4.86

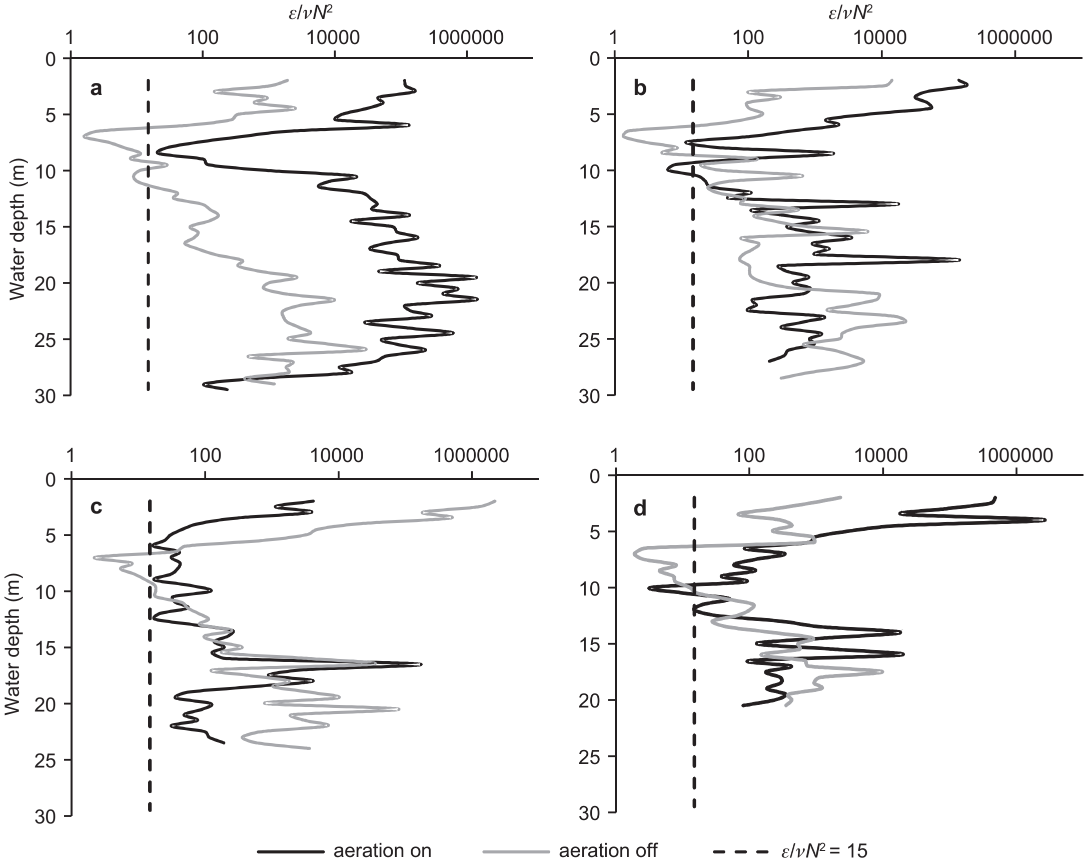

Fig. 3. The ratios of the dissipation rate of turbulent kinetic energy (ε) to the kinematic viscosity (ν) and buoyancy frequency (N) at (**a**) 20 m, (**b**) 100 m, (**c**) 150 m, and (**d**) 200 m from the aerator. The threshold ratio, $\varepsilon/\nu N^2$ **=** 15, is marked with a dashed line.

$p < 0.05$). Additionally, during the period when the aerator was on (from mid-June to late August), the increase in the mean temperatures at the 30-m depth relative to 2009 was very strong being 4.8 °C in 2010 and 5.1 °C in 2011 (t-test with Bonferroni correction: $t_{224} = 2.41, p < 0.05$).

The vertical temperature profiles measured at the deepest station (2) during the highest

Table 6. Significance of differences (two-sample Student's t-test) in dissipation rates of turbulent kinetic energy (ε) and eddy diffusivity (K) between cases with the aerator on and off; ** $p < 0.001$, * $p < 0.05$.

Water layer	Distance from aerator (m)							
	20		100		150		200	
	ε	K	ε	K	ε	K	ε	K
2–6 m	$t_{620} = 0.35$	$t_{620} = 4.84$**	$t_{609} = 3.11$*	$t_{609} = 5.85$**	$t_{602} = 9.35$**	$t_{602} = 7.51$**	$t_{606} = 5.52$**	$t_{606} = 6.82$**
6.5–10 m	$t_{558} = 7.54$**	$t_{558} = 7.06$**	$t_{550} = 5.76$**	$t_{550} = 3.89$**	$t_{542} = 2.65$	$t_{542} = 6.26$**	$t_{542} = 4.75$**	$t_{542} = 2.63$*
10.5–20 m	$t_{1398} = 21.65$**	$t_{1398} = 15.36$**	$t_{1355} = 9.46$**	$t_{1355} = 5.66$**	$t_{1251} = 3.40$	$t_{1251} = 3.75$**	$t_{1173} = 1.74$	$t_{1173} = 1.04$
20.5–30 m	$t_{874} = 11.02$**	$t_{874} = 3.96$**	$t_{389} = 3.16$*	$t_{389} = 4.43$**	$t_{133} = 1.28$	$t_{133} = 5.26$**		
0.5 m from the bottom	$t_{68} = 3.88$**	$t_{68} = 0.60$	$t_{67} = 0.88$	$t_{67} = 0.08$	$t_{66} = 0.47$	$t_{66} = 1.65$	$t_{66} = 0.61$	$t_{66} = 0.34$

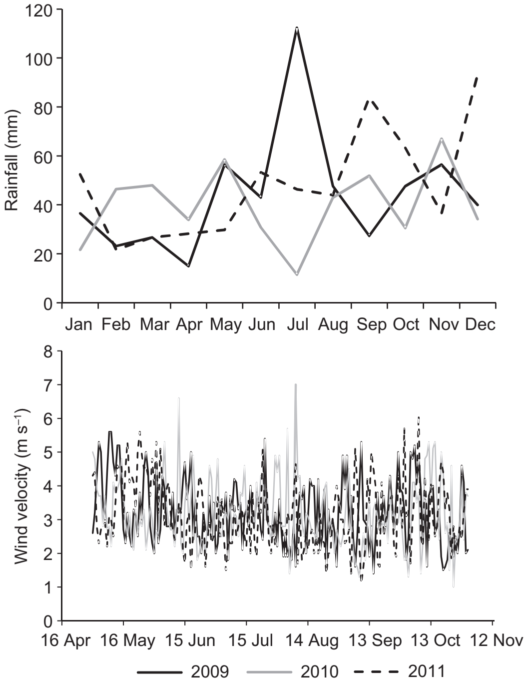

Fig. 4. Monthly average rainfall (top panel) and maximum wind velocities (bottom panel) (source: Finnish Meteorological Institute).

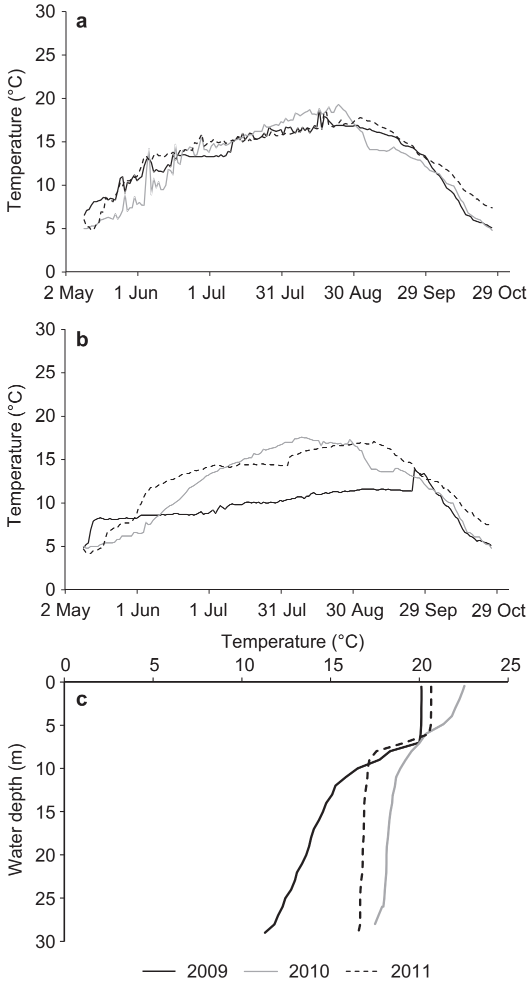

Fig. 5. Water temperature at station 2 at the depths of (**a**) 10 m, (**b**) 30 m, and (**c**) the vertical temperature profile during the highest hypolimnetic temperatures in 2009–2011.

hypolimnetic water temperatures in early August showed that the thermocline was clearly thinner and the difference in the water temperatures between the epilimnion and hypolimnion was lower in 2010 and 2011 as compared with that in 2009 (t-test with Bonferroni correction: $t_9 = 2.93$, $p < 0.05$) (Fig. 5). In addition to the thermocline being thinner, it occurred closer to the surface in 2010 and 2011 than in 2009. Moreover, the hypolimnetic water temperatures were very homogeneous in 2010 and 2011.

The external load of P to the Enonselkä basin did not differ among the study years and was 0.45 mg P m^{-2} d^{-1}, being two orders of magnitude smaller than PGS.

Discussion

Sedimentation of inorganic and organic material and phosphorus

In the reference year, 2009, the rates of GS in the Enonselkä basin were at the same level as found in an earlier study (Koski-Vähälä *et al*. 2000). After initiation of aeration, no abrupt changes in the temporal variation of sedimentation were observed apart from the highest sedimentation peaks that occurred during fall turnover, which is a common phenomenon (e.g. Rosa 1985). However, these sedimentation peaks indicated that more loose material was redistributed and settled during the falls of the aerated years. The effect of aerators was proved by significantly higher spatio-temporal mean rate of GS in the deep areas and higher median value of the GS rate in the shallow areas between the years 2009 and 2011. This also concerned the rates of SPIM (inorganic material) and SPOM (organic material). Although the organic content (as percentage) of the entrapped material (f_{GS}) decreased during the two years of aeration, the absolute rate of SPOM (the mean rate in deep and the median rate in shallow areas), contrary to the aim of the rehabilitation measure, increased. The decrease in the organic content of the settling material has been attributed to the increased settling time of material induced by aerators, which allows more complete oxidation in the water column (Ashley 1983, Gantzer *et al*. 2009).

The rates of PGS behaved as the other sedimentation rates. Since the external load of P and rainfall showed no increase between the reference and aerated years, the altered rates of SPOM and PGS could have resulted from intensified internal cycling of P and enhanced primary production. Factors influencing these processes could include sediment resuspension, increased turbulence in the water column and enhanced mineralization rates of organic material.

Effect of resuspension on sedimentation rates

In many lakes, sediment resuspension constitutes most of the total settling flux (Evans 1994, Weyhenmeyer 1998). This was the case in the Enoselkä basin (Koski-Vähälä *et al*. 2000, Buhvestova *et al*. 2013) and was also observed during all the years in the present study. Resuspension constituted most of the total settling flux but showed no differences as dry matter (*R*) or as a percentage of GS (*R*%) among the study years. This observation together with the fact that wind conditions showed no difference either, proves that resuspension rates were not affected by aeration. Therefore, resuspension played no role in causing differences in the rates of PGS and SPOM.

Effect of increased temperature and turbulence on the mineralization of organic matter and recycling of P

Salmi *et al*. (2014) found no marked effect of aeration on the epilimnetic or hypolimnetic P_{tot} concentrations during summer. However, the increase in hypolimnetic temperatures (Salmi *et al*. 2014 and this study) and turbulence were the indisputable results of the aeration activity that most likely caused the increase in the rates of PGS and SPOM observed in this study. The temperature of the hypolimnetic water strongly increased due to aeration and affected mineralization and oxygen consumption due to increased bacterial activity, which is known to be temperature dependent (Rose 1967, Bergström *et al*. 2010). The aeration-mediated increase in the median value of oxygen consumption, from 0.18 to 0.35 g m^{-3} d^{-1} (reference years 2000–2009, aerated years 2010–2013), reported by Salmi *et al*. (2014), indicates well the increased mineralization of organic material. They concluded that the increase was due to the increased hypolimnetic temperature and oxygen supply, and also probably due to increased turbulence. Salmi *et al*. (2014) also suggested that enhanced mineralization would lead to lower rates of settling organic material. Additionally, they reported that no increase in the concentration of Chl-*a* was observed after the implementation of aeration indicating no changes in primary production (Salmi *et al*. 2014). However, the increased rates of SPOM measured in the present study during years when aeration was in use contradicts this conclusion. Additionally, the reasoning of Salmi *et al*. (2014) did not take into account grazing by cladocerans that was most likely enhanced due to aeration, since their biomass increased due to the drastically decreased stock of smelt (*Osmerus eperlanus*), an efficient planktivore that suffered from the destruction of the cool oxygenated metalimnion in 2010 (Ruuhijärvi *et al*. 2015). Therefore, we agree with Salmi *et al*. (2014) that increased temperature and turbulence enhanced mineralization, but additionally, we suggest that the increased turbulence enabled the enhanced recycling of P and production of excess organic material.

Turbulence in boundary layers and the water column may strongly affect the transportation of heat, nutrients and also particulate material (Imboden and Wüest 1995). At the sediment–water interface (SWI), increased turbulence reduces the thickness of the diffusive boundary layer (DBL) and affects the transport of nutrients and gases (Jørgensen and Revsbech 1985). As discussed earlier, increased turbulence also affects the settling time of material and thereby increases the mineralization of organic material in the water column (Ashley 1983, Gantzer *et al*. 2009). This is attributed to the settling particles having a much higher surface area and a much thinner DBL compared to particles at the SWI (Jørgensen and Revsbech 1985). Our measurements are the first to show the quantitative effect of the used aerators (Mixox) on turbulence. The measurements at the deepest station (2) of the Enoselkä basin demonstrated that the aerator clearly increased the turbulence, proved by the increase in the dissipation rates (ε) and eddy diffusivities (K), and thereby enabled increased transportation of material and solutes. The strongest increase in ε was observed in the upper hypolimnion and 0.5 m above the bottom at the 20-m distance from the aerator, but the turbulence also significantly increased at the other distances, especially in the metalimnion.

The possibility of the vertical transport of solutes and particles was evaluated in relation to the claim of Ivey and Imberger (1991) and Istweire *et al*. (1993) that vertical mixing of stratified waters occurs if $\varepsilon/\nu N^2 > 15$. A clearly lower threshold ratio than this was measured in the vicinity of the aerator at station 2 in the whole hypolimnion and also in the metalimnion, indicating that the vertical mixing of nutrients and particles in the epilimnion would have been possible. Because an even lower threshold value of $\varepsilon/\nu N^2 > 6$ has been reported by Saggio and Imberger (2001), we conclude that together with the warming of the hypolimnetic water, the turbulence created by the aerators increased the mineralization of organic material and recycling of P, part of which was transported to the epilimnion, which in turn resulted in increased rates of SPOM. Since ε also significantly increased 0.5 m above the bottom, it most likely additionally affected the release of P from the bottom sediment by reducing the thickness of the DBL (Jørgensen and Revsbech 1985). This was very likely in 2010 and 2011, because during those years aeration was not sufficient to keep the hypolimnion oxic during the stratification periods (Salmi *et al*. 2014). According to the eddy diffusivities in the hypolimnion at the 20-m distance (K values in Table 4), the turbulent-transport time (T) (rough estimate for T: $T = L^2/K$, where L = length scale; Tennekes and Lumley 1972) of P released from the lake bottom through the entire hypolimnion (L = 20 m) was markedly faster while the aerator was on, approximately 39 days, as compared with the transport time while the aerator was off, approximately 245 days. The corresponding transport times for mineralized P through the upper hypolimnion (L = 10 m) were 25 and 215 days, respectively. The strong decrease in the transport times verified that the aerators had a possibility to affect the transport of P and production of organic material during the stratification periods in 2010 and 2011. The rates of SPOM clearly increased in the deep areas due to aeration. Additionally, we argue that the influence of the aerators may also have reached the shallow areas, as the analysis on the sedimentation data of shallow areas indicated.

Basin-scale effects of aeration and evidence for enhanced P recycling

The turbulence measurements were conducted only at the deepest station and at the 200-m distance from the aerator at furthest. However, the nine aerators located in the deeps of the Enonselkä basin have a spatially comprehensive coverage (Fig. 1) and a possibility to influence the nutrient and sedimentation dynamics of the whole basin. The increase in the rates of GS, SPIM, SPOM and PGS was more pronounced at the deep than the shallow stations, and was most likely due to sediment focusing (Ohle 1960, Niemistö *et al*. 2012, Buhvestova *et al*. 2013). This focusing was at its strongest in the aerated years after destratification in late summer, which is an indication of the resuspension, convective mixing and sedimentation of higher amounts of loose, newly produced material in those years (Bengtsson and Hellström 1992, Niemistö *et al*. 2008, Buhvestova *et al*. 2013). The increase in the sedimentation rates in the lake deeps was due aerator-induced increase in temperature and turbulence. However, apart from the rates of SPOM, the increase in the median values of all sedimentation rates in the shallow areas was also very strong. In addition to the turbulence-induced transport of nutrients to the epilimnion in the vicinity of the aerators, another factor enabling the effect of the aerators to reach the shallow stations was the ascent of the thermocline. The observed five-meter decrease (from 12 to 7 m) in the depth of the lower edge of the thermocline in early August (*see* Fig. 5) is equivalent to a 25% increase in the coverage of the hypolimnion in the studied basin. The aerator only had a significant effect on the turbulence of the water layer close to the bottom (0.5 m) at the 20-m distance from the aerator. However, a significant increase was also observed in the metalimnion (6.5–10 m) at the 200-m distance. Thus, it is likely that the nutrient dynamics of the shallow areas was affected when the turbulent eddies created by the aerator hit the edges of the deeps. This is due to the functioning principal of the aeration method: water pumped into the hypolimnion rises fairly close to the pump and disperses below the boundary layer (here thermocline) (Bendtsen *et al*. 2013) until it reaches the bottoms of shallower areas or the water motion attenuates. In the case of an ascended thermocline, these bottom areas occur at water depths considered shallow in the present study, and the nutrient and sedimentation dynamics of these areas may be affected. The closer the aerator is located to the shallow areas, the stronger an effect can be expected.

After the initiation of aeration, the rates of PGS strongly increased due to the increase in the GS rates and P content of the entrapped material. In the reference year (2009), the seasonal average P concentration of entrapped material at station 2 was at the same level (2.59 mg g^{-1}) as observed earlier (2.7 mg g^{-1}, in 1993) (Koski-Vähälä *et al*. 2000). After two years of aeration, this concentration was much higher (3.20 mg g^{-1}), indicating enhanced cycling and concentrating of P in settling particles. Since the deep areas showed a clearly higher increase, the effect of the aerators was evident. The effect of the aerators also reaching the epilimnion and bottoms of shallow areas would explain the increased median value of the PGS rate at the shallow areas. Contrary to that in the deep areas, the increase in the median rate of SPOM was not significant at the shallow stations, but was very close to significance ($p = 0.051$). This was most likely due to more effective mineralization in the water column of the shallow areas favoured by constantly oxygenated (e.g. Hulthe *et al*. 1998) and warmer conditions (Rose 1967). The lower f_{GS} values and higher SPIM rates in 2011 compared to 2009 provided evidence to support this conclusion.

Altogether, the enhanced recycling of P and increased SPOM rates due to aeration activity were clearly observed at the deep stations. The effect of aeration on SPOM rates was not as obvious at the shallow stations; however, P cycling was affected. The origin of the excess P observed with the PGS measurements was the enhanced P recycling in the water column and bottom sediments of the lake areas, where the mineralization and release rates were affected due to aeration-induced warming of hypolimnetic water and enhanced turbulent mixing (Jørgensen and Revsbech 1985, Bergström *et al*. 2010). The significant increase in the different sedimentation rates were observed during the second year of aeration. This was most likely attributed to warmer conditions (higher hypolimnetic temperature in 2011 than in 2010) and higher mineralization and thereby more effective recycling of P in 2011. Therefore, it has to be stressed that during warm summers the aeration method used here seemed to give negative results. On the other hand, a prompt recovery of a lake suffering from anoxic hypolimnion and internal phosphorus loading cannot be expected, since the positive effects of hypolimnetic aeration are always slow (e.g. Matthews and Effler 2006b, Gantzer *et al*. 2009). This is because the lakes with a long history of eutrophy may have a large pool of unmineralized organic material in the bottom sediments and degradation of this pool and reaching a new stable state of oxygen consumption may take years or decades (Matthews and Effler 2006b). In the present study, the sedimentation dynamics were only investigated during the first two years after the initiation of the aeration. However, if aeration leads to a higher total amount of organic material settling on the lake bottom also in future, the recovery of this lake will not be possible with the used method.

Conclusions

The gross sedimentation of inorganic and organic material as well as P strongly increased due to two years of hypolimnetic aeration conducted in the deeps of Enonselkä basin of Vesijärvi. Additionally, the sedimentation dynamics of the shallow areas were affected. Therefore, we suggest that hypolimnetic aeration in the summer by pumping epilimnetic water into the hypolimnion promotes negative effects (recycling of P, production of excess organic matter) instead of leading to more oxygenated hypolimnetic conditions and a decrease in the internal P loading in the Enonselkä basin. Despite the enhanced degradation of organic material, the total amount of organic matter settling on the lake bottom increased and the original aim of hypolimnetic aeration was counteracted by the rehabilitation measure itself.

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