Limnol. Oceanogr., 53(6), 2008, 2407–2415

Interactions between sediment resuspension and sediment quality decrease the TN : TP ratio in a shallow lake

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Abstract

The relative contributions of sediment resuspension and external nutrient loading to seasonal fluctuations of the total nitrogen to total phosphorus (TN : TP) ratio in the water column of the shallow Kirkkojärvi basin were studied. In May, the TN : TP mass ratio was above 30, but it decreased to below 10 in July–August. In October, the ratio increased again. Water quality was regulated by external loading in spring, but the sharp decrease in the TN : TP ratio in July–August resulted mainly from the elevated sediment resuspension rate. In May and October, the resuspension rate was below 20 g dry weight (wt) m^{-2} d^{-1}, while in July–August it exceeded 60 g dry wt m^{-2} d^{-1}. Sediment resuspension decreases the TN : TP ratio in the water because the ratio in the surface sediment is low. The high resuspension rate in July–August could not be explained by wind effects, water level fluctuations, or bioturbation, but was instead attributed to the seasonal change in the sediment quality. During the summer, fresh material having low critical shear stress settled on the bottom until the critical shear stress was lower than the actual shear stress, and the resuspension rate increased steeply, although no increment in wave effects occurred. For a shallow lake, a single representative TN : TP ratio is misleading, because the seasonally fluctuating resuspension rate substantially affects the ratio and may induce switches between N and P limitation. Seasonal fluctuations in the resuspension rate may have important effects on algal communities, because low N to P ratios can favor N-fixing cyanobacteria over other algal groups. In Kirkkojärvi, cyanobacteria became the dominant algal group during the period of intensive resuspension.

The bottom sediments of lakes contain large pools of nutrients. For numerous reasons, these nutrients stored in the sediment may return to the water column and cause internal nutrient loading, with consequences for the water quality (Forsberg 1989; Søndergaard et al. 2003). Factors causing internal nutrient loading include low redox potential, elevated pH, and sediment resuspension caused by water currents and animal activities (Forsberg 1989). In shallow lakes, wind-induced sediment resuspension is often the main cause of internal loading (Kristensen et al. 1992; Niemistö and Horppila 2007).

Most resuspension studies have focused on phosphorus dynamics, because phosphorus has been regarded as the factor that most often limits algal growth in lakes, but the importance of resuspension on nitrogen recycling has also been documented (Gálvez and Niell 1992; Reddy et al. 1996). Moreover, sediment resuspension tends to decrease the total nitrogen to total phosphorus (TN : TP) ratio in the water, because the ratio is usually lower in the surface sediment than in the water, which is again attributed to denitrification (Kaspar 1985; Hamilton and Mitchell 1988; Gálvez and Niell 1992). For instance, Schelske et al. (1995) reported that during a period of wind-induced intensive sediment resuspension, the total phosphorus concentration in the water column of a shallow lake increased twofold, but the total nitrogen concentration increased by only one fifth. Sediment resuspension may therefore induce switches between phosphorus and nitrogen limitation (Hamilton and Mitchell 1997), which may have important implications for algal communities, because low nitrogen to phosphorus ratios can favor nitrogen-fixing cyanobacteria in relation to other algal groups (Smith 1983; Levine and Schindler 1999). Resuspension may also bring sedimented phytoplankton (meroplankton) back into the water column (Hamilton and Mitchell 1988; Carrick et al. 1993).

The effect of resuspension on the TN : TP ratio has been linked to the strength of the resuspension events, and it has been shown that the TN : TP ratio in the water is negatively dependent on wind-induced wave action and bottom shear stress (Hamilton and Mitchell 1988, 1997). Hence, the TN : TP ratio in the water could fluctuate according to weather conditions. Another, more sparsely studied aspect is that the effects of resuspension on water quality depend not only on the bottom shear stress but also on the sediment characteristics, which may change during the growing season. Over the course of the growing season, organic material is deposited on the sediment surface, and this newly deposited material is brought into suspension more easily than is old compacted material (Bengtsson and Hellström 1992; Weyhenmeyer 1998). If the sediment becomes increasingly resuspension-prone during the summer, it can be hypothesized that the effects of resuspension on TN : TP ratio should be more pronounced in late summer than in the beginning of the growing season.

In order to study whether this hypothesis is valid, we examined the seasonal variations in the sediment quality, the rate of sediment resuspension, and the effects of resuspension on nutrient dynamics in a shallow eutrophic

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Acknowledgments

Leena Nurminen and Kaarina Weckström assisted with the fieldwork, Laura Montin helped with the fetch calculations, and Susanna Hietanen provided valuable comments on the manuscript. We also thank the two anonymous reviewers for helpful comments.

This study was financially supported by the Academy of Finland (projects 211156 and 124206).

lake. Variations in phytoplankton biomass and species composition were also considered. The study was conducted in the shallow Kirkkojärvi basin, where resuspension-mediated internal loading dominates the phosphorus input to the water column during the open-water season and massive blooms of cyanobacteria are common phenomena (Tallberg et al. 1999; Horppila and Nurminen 2005*a*; Niemistö and Horppila 2007).

Methods

The study was conducted in Kirkkojärvi (60°24′N, 24°18′E), which is a restricted basin of the eutrophic Lake Hiidenvesi (total area 30.3 km²). The shallow Kirkkojärvi basin (area 1.6 km², mean depth 1.1 m, maximum depth 3.5 m) is the most eutrophic part of the lake. The summertime TP concentration is 80–120 μg L^{-1}, and the TN concentration is 1000–1500 μg L^{-1}. The average water retention time is 5.1 d for the whole year and 10.0 d for the open-water season in May–October. As a result of resuspended sediments and runoff from agricultural areas, the concentration of suspended solids often exceeds 20 mg L^{-1}, and the Secchi depth usually remains below 0.5 m. More detailed information on the characteristics of Kirkkojärvi and the whole of Lake Hiidenvesi can be found in the work of Tallberg et al. (1999), Nurminen et al. (2001), and Niemistö and Horppila (2007).

The study was conducted in 08–22 May, 25 July–08 August, and 02–17 October 2006. The gross sedimentation and sediment resuspension rates during these periods were estimated with cylindrical sediment traps (diameter = 5.4 cm, aspect ratio = 6 : 1) (Bloesch and Burns 1980). The tops of the traps were 0.4 m above the lake bottom. Because the resuspension rate may show considerable spatial variations (Bloesch and Uehlinger 1986; Horppila and Niemistö 2008), traps were placed at 24 different evenly distributed sites, the sampling area covering the whole basin, with the exception of areas in which the water depth was <0.7 m. The shallowest areas were excluded from the study, because variations in macrophyte biomass and species composition in those areas substantially affect sedimentation, resuspension, and nutrient dynamics (Horppila and Nurminen 2005*a*) and would complicate the interpretation of the results. Previous studies have examined the resuspension rates among different macrophyte beds in Kirkkojärvi (Horppila and Nurminen 2001, 2003, 2005*a*). At the end of each exposure period, the entire content of each trap was emptied, and the dry weight (dry wt) of trapped material was measured after drying the samples at 105°C and the organic fraction after ignition at 550°C. Because some traps were lost and the water level reduction excluded some stations in the autumn, the number of lifted traps varied from 21 to 24. Surface sediment samples (topmost 0–1 cm) were collected at each trap location with a Kajak corer and measured for dry weight and loss on ignition. Samples for the concentration of suspended solids (SS) were collected from each station with a Limnos tube sampler (height 1 m, volume 7.5 liters) at the beginning and end of each trap exposure period (samples from each 1-m water layer from the surface to the bottom pooled at each station). The samples from each station were filtered through Whatman GF/C filters and measured for dry weight and loss on ignition. At each sampling station, profiles of dissolved oxygen, turbidity, pH, and temperature were measured with a YSI-6600 sonde (YSI).

The rate of sediment resuspension (R) at each sampling station during each period was calculated according to Gasith (1975):

$$R = S\frac{f_S - f_T}{f_R - f_T} \qquad (1)$$

where S = gross sedimentation (g dry wt m^{-2} d^{-1}), f_S = organic fraction of S, f_R = organic fraction of the surface sediment, and f_T = organic fraction of seston T collected from the water column.

The method of Gasith (1975) was chosen because it has been shown that it is less sensitive to the effects of seasonal variations in phytoplankton biomass than is the method of Weyhenmeyer et al. (1995), which is also applicable to shallow water bodies (Horppila and Nurminen 2005*b*). The method of Gasith (1975) is reliable if the organic content of seston (f_T) is significantly different from that of surface sediment (f_R) (Blomqvist and Håkanson 1981). Therefore, the values of f_R on each sampling date were compared with f_T with ANOVA (Horppila and Nurminen 2001). Water samples for concentration of TN, TP, and chlorophyll *a* (Chl *a*) were collected concomitantly with SS sampling (tube sampler). TN and TP were determined using the method by Koroleff (1979) (Lachat autoanalyzer, QuickChem Series 8000). The TP content of the surface sediment at each trap location was determined with an ICP mass spectrometer (Perkin Elmer ELAN 6000 ICP-MS) and the TN content with a LECO CHN-900 analyzer. The resuspension of TN and TP at each sampling station was calculated with the resuspension rate and nutrient concentration of the surface sediment (Horppila and Nurminen 2003). Samples for Chl *a* were filtered onto Whatman GF/C filters and analyzed spectrophotometrically after extraction with ethanol (Finnish Standards Association 1993). Data on phytoplankton species composition were available for 27 July, 16 August, and 14 September 2006 from the water quality monitoring program of Lake Hiidenvesi (Ranta et al. 2007).

To study the role of wave disturbance in promoting sediment resuspension, we used the method of Hamilton and Mitchell (1988), which takes into account wavelength as well as water depth, thus:

$$\textit{wave action} = \frac{H^2}{Z} \qquad (2)$$

where H is the wave height and Z is water depth; wave action was calculated for each sampling station and sampling period. For the calculations, the theoretical wavelengths at each station were calculated using equations presented by Carper and Bachmann (1984):

$$L = \frac{gT^2}{2\pi} \qquad (3)$$

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

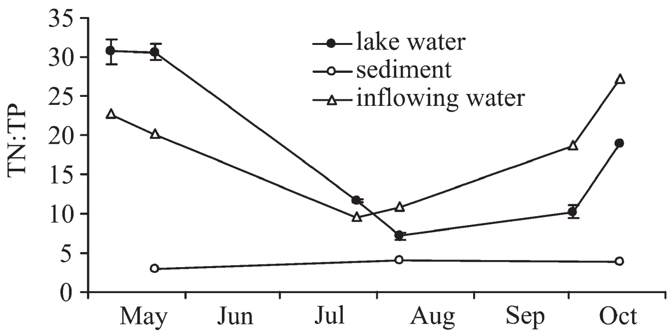

Fig. 1. Seasonal fluctuations of the TN : TP mass ratio of the inflowing river water, the water column of the lake, and the surface sediment (±95% confidence limits).

$$\frac{gT}{2\pi U} = 1.20 \tanh\left[0.077\left(\frac{gF}{U^2}\right)^{0.25}\right] \quad (4)$$

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 the method of the Beach Erosion Board (1972).

$$F = \frac{\sum x_i \cos\gamma_i}{13.5} s \quad (5)$$

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. Data on wind velocity and direction (eight measurements per day) were obtained from the Helsinki–Vantaa airport, located 40 km east of Kirkkojärvi. Wave height was calculated as one half of the wavelength. It can be assumed that when wavelength exceeds twice the water depth, waves disturb the surface sediment (Carper and Bachmann 1984).

To take external nutrient loading into account, the loading coming into the Kirkkojärvi basin via River Vihtijoki was calculated using daily discharge measurements and SS, TN, and TP concentrations (measured with 1- to 4-week intervals) (Ranta et al. 2007). External nutrient loading from other sources (e.g., domestic point loading) is much lower than the loading via River Vihtijoki (Ranta et al. 2007) and was excluded from the calculations.

Between-period differences in the lake water quality (TN and TP concentration, TN : TP ratio, SS concentration, organic content of SS), sediment quality (TN, TP, TN : TP ratio, organic content) at the end of each study period, and the sediment resuspension rate as well as wave action during each period were tested with ANOVA. Paired comparisons were conducted with Bonferroni t-tests. Before the analyses, the normality of the data sets was studied with the Shapiro–Wilk test and the data were log-transformed when necessary. The organic content data were arcsin $\sqrt{x}$–transformed.

Results

Water quality—For 08–22 May, the TN : TP mass ratio in the water column of Kirkkojärvi was 31, but it decreased significantly during the summer to 7 for 25 July–08 August (Fig. 1; Table 1). In October, the TN : TP ratio increased again to 19. A similar seasonal pattern was observed in the incoming river water. The TN : TP ratio in River Vihtijoki was above 30 during January–March, 22 during the 08–22 May study period, decreased sharply during the early summer, and dropped below 10 in July–August (Fig. 1). In October, the TN : TP ratio of the river water increased again to 15. In the surface sediment, the ratio was 3.0 in 08–22 May but increased to a significantly higher level in July–August and October, when it reached 4.0 (Fig. 1; Table 1).

Table 1. Analysis of variance results for the variations in water and sediment quality parameters at the end of each study period and for the sediment resuspension rate and wave action during each period. The analyzed study periods were 08–22 May, 25 Jul–08 Aug, and 02–17 Oct 06.*

	df	F	p
TN : TP mass ratio in water	2,64	675.93	<0.0001
TN : TP mass ratio in sediment	2,64	16.61	<0.0001
TP in lake water	2,64	1009.54	<0.0001
TN in lake water	2,64	102.21	<0.0001
TP in sediment	2,64	13.13	<0.0001
TN in sediment	2,64	2.33	0.1056
SS in lake water	2,64	284.06	<0.0001
Turbidity in lake water	2,64	156.74	<0.0001
Chl *a* in lake water	2,64	115.26	<0.0001
Organic % of SS in lake water	2,64	268.84	<0.0001
Organic % of trapped material	2,64	46.45	<0.0001
Organic % of sediment	2,64	4.64	0.0112
Resuspended SS	2,62	84.70	<0.0001
Resuspended TN	2,62	65.86	<0.0001
Resuspended TP	2,62	54.59	<0.0001
Wave action	2,62	5.95	0.0041

* TN, total nitrogen; TP, total phosphorus; SS, suspended solids; Chl *a*, chlorphyll *a*.

The midsummer decrease in the TN : TP ratio in the water column mainly resulted from an increase in the TP concentration. Both TP and TN showed significantly higher concentrations in July–August than in May or October (Table 1), but the relative change was much higher in TP than in TN. On average, the concentration of TP was 60 μg L^{-1} in 08–22 May and 280 μg L^{-1} in 25 July–08 August (Fig. 2A). The concentration of TN increased from 1780 μg L^{-1} in May to 2010 μg L^{-1} in August (Fig. 2A). In the sediment, the midsummer increment of the TN : TP ratio mainly resulted from the decreasing TP concentration. The TP concentration in the sediment was on average significantly higher in May (1.3 mg g^{-1}) than in August or October (1.1 mg g^{-1}) (Fig. 2B; Table 1). No differences between sampling periods were detected in the TN concentration (3.9–4.5 mg g^{-1}) (Fig. 2B; Table 1).

The SS concentration in the water was on average 8.5 mg L^{-1} in May, increased to a significantly higher value of 28.6 mg L^{-1} in July–August, and decreased again toward the autumn, to 12.2 mg L^{-1} in October (Table 1). Water turbidity was on average 17 nephelometric turbidity units (NTU) in May and October, but was significantly higher (37 NTU) in July–August. The relationship between the SS concentration and the TN : TP ratio in the lake water, as well as the relationship between the Chl *a*

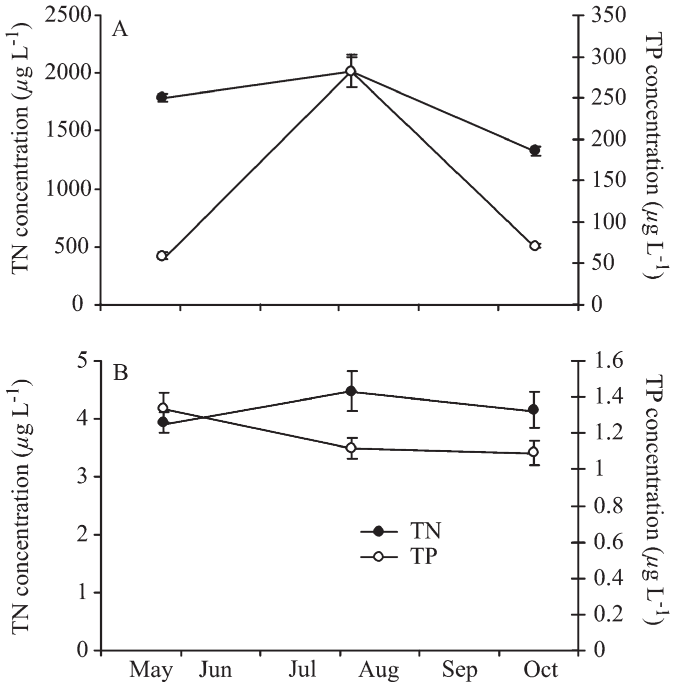

Fig. 2. Seasonal fluctuations in (A) the TN and TP concentrations of the lake water and (B) the surface sediment (±95% confidence limits).

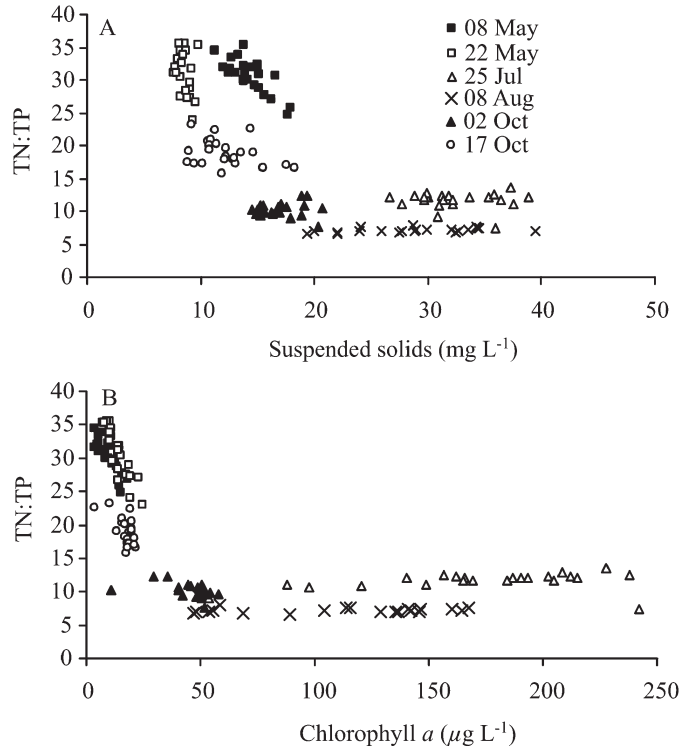

Fig. 3. (A) The relationship between the concentration of suspended solids (SS) and the TN : TP mass ratio in the lake water and (B) the relationship between the concentration of Chl *a* and the TN : TP mass ratio in the lake water.

concentration and the TN : TP ratio, was curvilinear. When the SS concentration exceeded 20 mg L^{-1}, the TN : TP ratio decreased steeply below 15 and remained unchanged at higher SS concentrations (Fig. 3A). Similarly, when Chl *a* concentration was low in May, the TN : TP ratio decreased with increasing chlorophyll concentration, but in July–August, when Chl *a* varied between 50 and 250 μg L^{-1}, no dependence between the TN : TP ratio and chlorophyll could be detected (Fig. 3B).

As a result of the well-mixed conditions, vertical as well as horizontal water temperature differences on each sampling date were small. Water temperature was 13–15°C from 08–22 May, 20–22°C from 25 July–08 August, and 9–11°C from 02–17 October. The concentration of dissolved oxygen was above 5 mg L^{-1} throughout the water column at all the sampling stations and on all sampling dates. The water pH varied between 7.3 and 7.6 in May and October, but reached 9.3–9.5 in July–August as a result of the intense algal production. The organic content of the suspended matter fluctuated substantially between sampling periods and was significantly higher in July–August than in May and October (Tables 1, 2). The organic content of trapped material in July–August (14.4%) was also higher than in May or in October (<13%) (Tables 1, 2). The organic content of the surface sediment was 10.4% in May and increased to a significantly higher value of 11.3% in July–August and October (Tables 1, 2). During all three sampling periods, the values of f_T were significantly higher than the values of f_R ($p < 0.001$), indicating that the method of Gasith (1975) could be reliably used (Blomqvist and Håkanson 1981).

The phytoplankton biomass in Kirkkojärvi on 27 July (i.e., at the beginning of the July–August sampling period) was 3.2 mg L^{-1} and consisted mainly of Bacillariophyceae (mainly *Asterionella formosa* Hassall) and Cryptophyceae. The biomass of cyanobacteria was 0.2 mg L^{-1} (6% of the total biomass) (Fig. 4). On 16 August (i.e., after the July–August sampling period) phytoplankton (total biomass 4.3 mg L^{-1}) were dominated by cyanobacteria (mainly *Anabaena flos-aquae* Brébisson), the biomass of which had increased tenfold and which comprised 47% of the phytoplankton biomass (Fig. 4). The biomass of other algal groups was slightly lower than on 27 July. On 14 September, phytoplankton biomass had further increased (6.9 mg L^{-1}) (Fig. 4), but the assemblage was dominated by Bacillariophyceae, Euglenophyceae, and Chlorophyceae. The biomass of cyanobacteria was 0.2 mg L^{-1} (3% of the total biomass) (Fig. 4).

External loading and resuspension—The water discharge from River Vihtijoki reached a maximum of 7.8 m^3 s^{-1} in April as a result of spring runoff. From July to October, the water discharge was less than 0.5 m^3 s^{-1}. Consequently, in April, the TN loading via River Vihtijoki reached

Table 2. The organic matter content (%) of surface sediment, trapped material, and suspended matter (±95% confidence limits).

	22 May	08 Aug	17 Oct
Surface sediment	10.4±0.3	11.3±0.4	11.3±0.3
Trapped material	12.3±0.3	14.4±0.4	12.6±0.2
Suspended matter	48.2±3.5	61.8±4.5	48.1±3.2

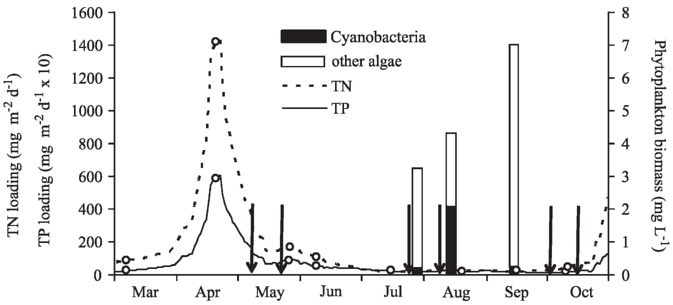

Fig. 4. Seasonal variations in the external loading of total nitrogen and total phosphorus via River Vihtijoki into the Kirkkojärvi basin (sampling dates for nutrient concentrations are shown) and the biomass of cyanobacteria and other algae on 27 Jul, 16 Aug, and 14 Sep 2006. The beginning and the end of the three intensive sampling periods are indicated with arrows.

1400 mg m^{-2} d^{-1} and the TP loading reached 60 mg m^{-2} d^{-1}, but remained very low in July–September (Fig. 4). During the 08–22 May study period, TN loading from River Vihtijoki was 188 mg m^{-2} d^{-1}, but it was less than 45 mg m^{-2} d^{-1} in July–August and October (Fig. 5). External TP loading was also clearly higher in 08–22 May (8.8 mg m^{-2} d^{-1}) than in July–August or in October (1.9–2.2 mg m^{-2} d^{-1}) (Fig. 5). The external SS load via River Vihtijoki was 2.1 g dry wt m^{-2} d^{-1} in May and <0.5 g dry wt m^{-2} d^{-1} in July–August and October.

The rate of sediment resuspension in Kirkkojärvi was on average 19.4 g dry wt m^{-2} d^{-1} in May and 18.8 g dry wt m^{-2} d^{-1} in October (Fig. 5). The gross sedimentation was rate 20.0 g dry wt m^{-2} d^{-1} in May and 19.4 g dry wt m^{-2} d^{-1} in October. The resuspension rate in August was 61 g dry wt m^{-2} d^{-1} and was thus significantly higher than in May and October. The gross sedimentation rate was 63.9 g dry wt m^{-2} d^{-1}. Resuspension thus dominated the total downward flux throughout the study, and net sedimentation comprised 3–5% of gross sedimentation. During 08–22 May, resuspension brought on average 81.3 mg m^{-2} d^{-1} TN into the water column (Fig. 5).

In July–August, TN resuspension reached 292.4 mg m^{-2} d^{-1} and decreased again to 83.0 mg m^{-2} d^{-1} in October (Fig. 5). The resuspension-mediated TP loading was 27.6 mg m^{-2} d^{-1} in May, increased to 72.9 mg m^{-2} d^{-1} in July–August, and decreased again to 21.7 mg m^{-2} d^{-1} in October (Fig. 5). For both TN and TP, the resuspension-mediated internal loading was significantly higher in 25 July–08 August than during the other two study periods (Table 1).

Weather conditions and wave action—No major storms occurred during the study period. The maximum daily wind velocities during the sampling periods only occasionally exceeded 12 m s^{-1} (Fig. 6). As a result of exceptionally low precipitation during the summer, the water level decreased during the study. The average depth at the sampling stations was 1.8 m from 08–22 May, 1.6 m from 25 July–08 August, and 1.4 m from 02–17 October. Wave action values were on average lowest in July–August and highest in October (Fig. 6). No statistical difference between May and July–August was found, but in October wave action was significantly stronger than in July–August (Table 1).

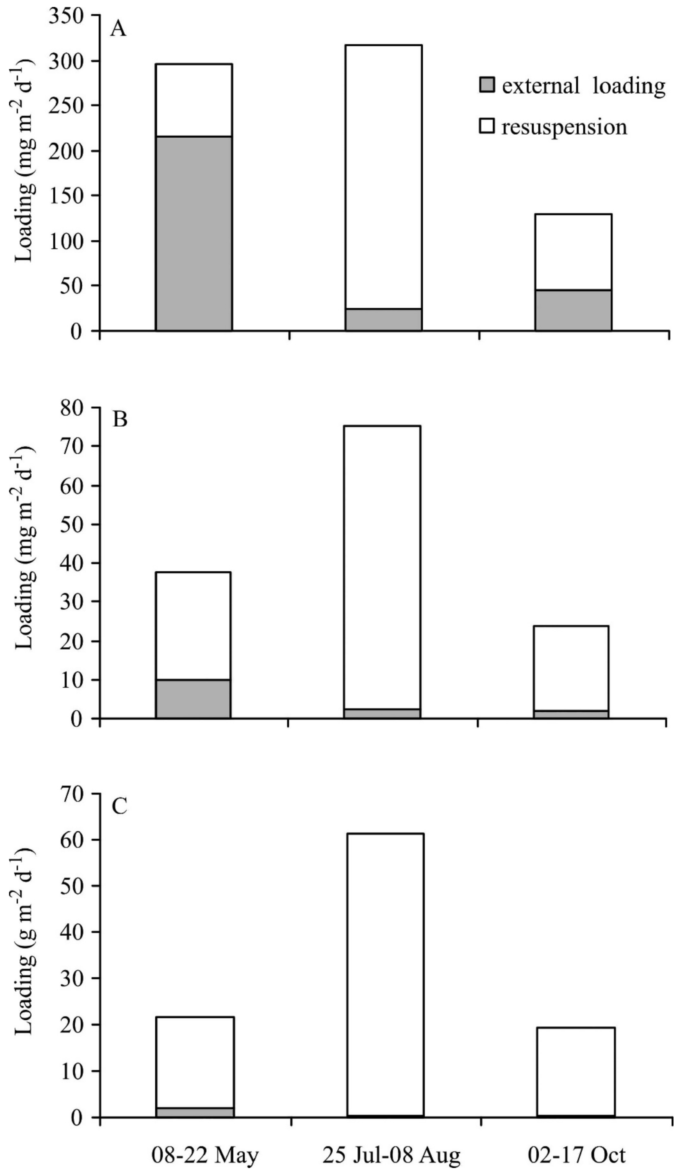

Fig. 5. The flux of (A) total nitrogen, (B) total phosphorus, and (C) suspended solids (SS) into the water column of Kirkkojärvi from the drainage area and from resuspension during the three intensive study periods.

Discussion

The effect of external nutrient loading on the water quality of Kirkkojärvi was strong in spring, when water discharge from the drainage area was high. During the rest of the open-water season, resuspension played a dominating role as a water quality regulator. The strong effect of resuspension in Kirkkojärvi is attributed both to the low mean water depth and the low biomass of macrophytes. In shallow waters, even low wind velocities cause resuspension, which consequently is often a continuous process (Evans 1994). Even occasional resuspension events can

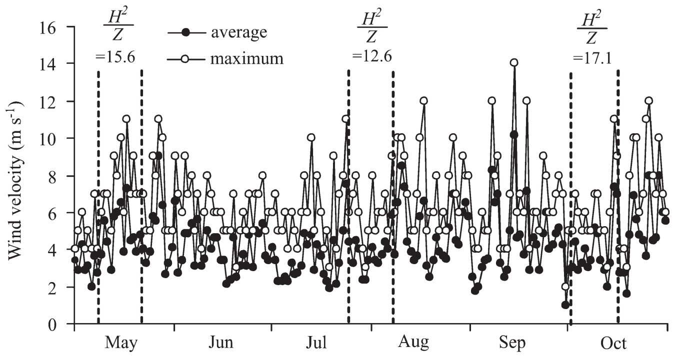

Fig. 6. The average and maximum daily wind velocities during the study. The exposure periods of the sediment traps are shown with broken vertical lines. The average wave action values (H^2/Z) of each period are also shown.

crucially affect water quality, because the critical shear stress for deposition is lower than that for initiation of motion, and sediment particles may stay in suspension for days after the resuspension event (Bengtsson and Hellström 1992). Macrophytes inhibit sediment erosion and resuspension, but in Kirkkojärvi, macrophyte beds are restricted to very shallow nearshore areas, which reduces their water quality effects (Horppila and Nurminen 2001, 2003).

The rate of resuspension increased substantially in July–August, which could not be explained by wind effects or water level fluctuations, since wave action in July–August was weaker than during the two other intensive study periods. Benthivorous fish can contribute to sediment resuspension (Breukelaar et al. 1994) but were an unlikely cause for the elevated resuspension rate in July–August, because the most abundant benthic feeding fish species [perch *Perca fluviatilis* (L.), roach *Rutilus rutilus* (L.), and white bream *Abramis björkna* (L.)] in the shallow basins of Lake Hiidenvesi turn from a zoobenthic to a zooplanktivorous diet as the summer progresses (Uusitalo et al. 2003). The most obvious explanation for the high resuspension rate in July–August was the change in the sediment quality. Søndergaard et al. (1992) concluded that in the eutrophic Lake Arresø, resuspension from August sediment was higher than from May sediment, because organic matter accumulated on the sediment during the summer. Similarly, in Kirkkojärvi, the organic fraction of surface sediment as well as the organic fraction of trapped material increased over the course of the summer. The increment in the organic matter content of the surface sediment was due to the sedimentation of the increasing phytoplankton biomass. The increment could not be attributed to the growth of benthic algae, since the majority of the bottom area is devoid of benthic algae and submerged macrophytes because of the high light extinction. Because of the high water turbidity, the productive layer (>1% of subsurface light left) is restricted to the uppermost 50 cm (Alajärvi and Horppila 2004), and, thus, the sediments in the present study area were situated in the aphotic zone. During the summer, more and more fresh and loose material with low critical shear stress (Bengtsson et al. 1990; Bengtsson and Hellström 1992) settled on the bottom, until the critical shear stress was lower than the actual shear stress and the resuspension rate increased steeply, although wave action decreased. For fresh deposits, the erosion rate increases more than linearly with increased excess shear stress, and resuspension may be very rapid (Bengtsson and Hellström 1992). Cyanobacteria form very easily, resuspending detritus, and because the sinking rate of many other algal groups is much higher than that of cyanobacteria, the erodibility of phytoplankton detritus depends on its species composition (Gons et al. 1991; Beaulieu 2003). In Kirkkojärvi, for instance, the percentage of planktonic diatoms in the settled phytoplankton is much higher than their percentage in the water column, while cyanobacteria have a minor share in the settled material compared with their biomass in the water column (Tallberg and Horppila 2005). Thus, the seasonal variations in the species composition of phytoplankton probably contributed to the elevated resuspension rate in midsummer.

When sediment traps are used to estimate sediment resuspension, it must be remembered that they do not always give a correct impression about the quantity and quality of settling material (Bloesch and Burns 1980; Weyhenmeyer 1998). Because the settling velocity of inorganic particles is high compared to that of organic particles and because resuspended particles are often mainly inorganic, their percentage may be overestimated by sediment traps (Rosa 1985; Weyhenmeyer 1998). On the other hand, cyanobacteria, which have high buoyancy, may be involved in seston samples without affecting the estimated settling flux (Weyhenmeyer et al. 1995). When resuspension is estimated using the method by Gasith (1975), such phenomena may cause erroneous estimates of gross sedimentation (S) and/or organic fraction of trapped material (f_S), while the organic fraction of suspended matter (f_T) and surface sediment (f_R) can be more reliably estimated with water and sediment samples (Horppila and Nurminen 2005*b*). Is it thus possible that the observed steep midsummer increase in the resuspension rate in Kirkkojärvi was due to errors in the estimates of S and f_S. The effects of erroneous estimates of S and f_S on resuspension calculations using the Gasith (1975) method in the circumstances of Kirkkojärvi were explored by Horppila and Nurminen (2005*b*). It was shown that if S is underestimated, resuspension rate tends to be underestimated, while the underestimation of f_S leads to an overestimated resuspension rate. Because severe overtrapping was very unlikely with the type of cylindrical traps used in Kirkkojärvi (Hargrave and Burns 1979), overestimation of the midsummer resuspension in Kirkkojärvi could have resulted only from underestimation of f_S.

The estimated resuspension rate in Kirkkojärvi in July–August was 3.3 times higher than in May. According to the sensitivity analysis by Horppila and Nurminen (2005*b*), even a 50% underestimation of f_S results in only a 20% overestimation of resuspension rate by the Gasith (1975) method. Thus, the threefold increase in the resuspension rate in midsummer could not have resulted from erroneous estimation of f_S. Additionally, the method of Weyhenmeyer et al. (1995), which is less sensitive to the variations in f_S, has been shown to give similar results in Kirkkojärvi (compared to the method of Gasith [1975]) (Horppila and Nurminen 2005*b*).

The high resuspension rate in July–August had a substantial effect on the nutrient concentrations as well as on the nutrient relationships in the water column. In May, the nutrient concentrations were determined mainly by the spring runoff from the drainage area. The TN : TP mass ratio in the water column of Kirkkojärvi was 31, which corresponded to the ratio of the spring runoff waters and is a common nutrient ratio in the runoff waters from fertile soils (Downing and McCauley 1992). In July–August, external loading was very low, the resuspension rate was high, and the TN : TP ratio in the water approached the ratio found in the surface sediment, as is typical during intensive resuspension (Hamilton and Mitchell 1997; Ogilvie and Mitchell 1998). The TN : TP ratio in the sediment of Kirkkojärvi was similar to the ratios reported in other lakes (Downing and McCauley 1992; Hamilton and Mitchell 1997).

As is common in fluvial waters (Meyer et al. 1981), the TN : TP ratio in River Vihtijoki decreased during midsummer. The discharge during the summer months was, however, very low. In July–August, the external nutrient loading via River Vihtijoki was $<10\%$ of the internal loading and thus resulted in only a minor contribution to the midsummer decrease in the TN : TP ratio in the lake.

Low TN : TP ratios are not only a result of phytoplankton blooms but may also be a cause of these blooms, because intensive phytoplankton production can accelerate the P flux from the sediment through elevated water pH (Koski-Vähälä and Hartikainen 2001; Xie et al. 2003). However, the diffusive flux is much lower than the flux through resuspension, and usually elevated pH alone cannot explain high internal P loading (Søndergaard et al. 1992; Reddy et al. 1996). This has been experimentally shown also in Kirkkojärvi; during a cyanobacterial bloom and at high pH (>9), the flux of soluble P from the sediment was increased only if intensive resuspension occurred (H. Holmroos unpubl.). Thus, algal uptake alone did not pump P out of the sediment, but sediment resuspension contributed to the P release by transporting material to the water layers, where pH is elevated as a result of primary production (Søndergaard et al. 1992; Koski-Vähälä and Hartikainen 2001). Additionally, on 25 July and 08 August, Chl *a* showed fivefold spatial variation in Kirkkojärvi, but the TN : TP ratio was constant over all of the study area, revealing that factors other than phytoplankton dynamics were behind the decreased TN : TP ratio. Moreover, TN : TP ratios of <10 occurred in Kirkkojärvi even during those summers when cyanobacterial blooms were weaker (e.g., as a result of exceptional weather conditions) (Tallberg and Horppila 2005).

The shift from the high external nutrient loading in spring to the resuspension-dominated situation in late summer also emerged in the relationship between the SS concentration and the TN : TP ratio. In May, the concentration of SS was similar throughout the study area, but the TN : TP ratio varied considerably. This was due to the strong effect of nutrients coming via the inflowing River Vihtijoki water and the uneven distance of the different sampling stations from this nutrient source. In July–August, the TN : TP ratio was similar at all the stations, but the concentrations of SS and Chl *a* showed large variation. This was because sediment quality and resuspension showed only small spatial variation, and resuspension dominated the nutrient flux into the water. As shown by the large spatial variation of Chl *a* in July and August, the variation in the SS concentration resulted from the patchiness of phytoplankton, which was again attributed to the effects of the wind. In lakes, wind effects cause considerable heterogeneity in the horizontal concentration of phytoplankton, especially when wind velocities are low (Small 1961; Verhagen 1994).

Cyanobacterial dominance coincided with the period of the highest resuspension rate in July–August. Later in September, phytoplankton biomass was considerably higher than in July–August, but the biomass of cyanobacteria had collapsed. Thus, some factor was favoring cyanobacteria, especially in July–August. Numerous studies have suggested that nitrogen-fixing cyanobacteria are favored by low nitrogen to phosphorus ratios (Smith 1983; Levine and Schindler 1999), while others have pointed out that cyanobacterial dominance is connected to the concentration of the limiting nutrient rather than to the nutrient ratio (Trimbee and Prepas 1987; Downing et al. 2001). During intensive resuspension in Kirkkojärvi, the TN : TP ratio fell from above 30 to below 10, which is clearly lower than the suggested limit for cyanobacterial dominance (Smith 1983). The intensive midsummer resuspension also caused a fourfold increase in the concentration of phosphorus, which was the main limiting nutrient in spring (Molot and Dillon 1991; Kim et al. 2007). The results thus indicated that resuspension played a role in promoting the dominance of cyanobacteria in Kirkkojärvi in July–August. In addition to its effects on nutrients, sediment resuspension can favor cyanobacteria by reducing the light intensity in the water, because cyanobacteria are well adapted to grow in low light intensities and may form blooms in high concentrations of inorganic suspended solids (Reynolds 1984; Gons et al. 1991; Burkholder et al. 1998).

Water quality in a lake may change abruptly as a result of resuspension when a certain threshold in wind velocity or shear stress is exceeded (Carper and Bachmann 1984; Bengtsson and Hellström 1992; Hamilton and Mitchell 1997). This present study demonstrated that a seasonal change in sediment quality may also result in a rapid water quality change when sediment becomes loose enough to be entrained by the prevailing near-bottom forces. The results supported the conclusions of Hamilton and Mitchell (1988); for a shallow lake, a single representative TN : TP ratio is misleading, because the seasonally fluctuating resuspension rate substantially affects this ratio and may induce switches between N and P limitation. The results also indicated that the seasonally changing sediment quality and its consequences for the resuspension rate may be a factor behind the tendency of cyanobacteria to form high biomasses in late summer.

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Received: 24 April 2008
Accepted: 9 July 2008
Amended: 22 July 2008