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TITLE:

Paleolimnological fingerprinting of the impact of acid mine drainage after 50 years of chronic pollution in a southern Finnish lake

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Abstract:

Acid mine drainage (AMD) is acknowledged to have long-lasting impacts on aquatic environments. Hence, mines have also been detected to pose problems years after closure due to the leaching of toxic drainage initiated by sulfide oxidation. To assess the effects of chronic but relatively low volume acid mine drainage derived from the Haveri copper–gold mine operating between 1938 and 1960 on a freshwater bay in southern Finland, we compared cladoceran assemblages from the pre-mining period with contemporary populations using paleolimnological approaches and multiple sediment cores. The cladoceran community of the pre-mining era differed significantly from the contemporary community of the lake (ANOSIM R = 0.91; p = 0.0001), but closely resembled the contemporary [illegible]fferences in species compositions between pre-mining and contemporary samples are most likely caused by eutrophication and not by the AMD impact. Because AMD at our study site is most intense during the spring snowmelt period, cladocerans may avoid seasonal pollution peaks through winter dormancy. Possible pollution peaks resulting from heavy rains during the summer may have negative impacts on the cladoceran community, but such short-term impacts are probably rapidly counteracted by immigration from cleaner areas of the lake.

Keywords:

Acidic mine drainage, impacts, cladocera, paleolimnology, chronic pollution, Finland

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1 Introduction

Industrial pollution is acknowledged as a major cause of surface water degradation. The mining industry is one of the major polluters of freshwater ecosystems, and one source of this pollution is abandoned mines. Leaching from sludge ponds and acidic drainage from tailings are known to inflict serious damage on receiving ecosystems (Kelly 1988; Wolkersdorfer and Bowell 2005). Acidic mine drainage (AMD) is one of best-known environmental challenges in the mining industry. In short, the acidic leachate is formed when sulfidic minerals are exposed to atmospheric oxygen and water. The effluents are usually rich in iron (Fe) and sulfate ($SO^{2-}{}_4$), and sometimes also in toxic substances, such as potentially toxic metals (e.g. Akcil and Koldas 2006).

In Viljakkala, SW Finland (Fig. 1), the AMD originating from the abandoned Haveri copper–gold (Cu–Au) mine tailings dump drains into the bay head of Lake Kirkkojärvi. The effluent leaching from the tailings into Lake Kirkkojärvi has a pH of 3.9 and extremely high metal concentrations (Parviainen 2009). Kihlman and Kauppila (2010) investigated the impacts of Haveri mine pollution on protist and diatom communities using continuous sedimentary records retrieved from the deep area of Lake Kirkkojärvi. They concluded that metal pollution had impacted the lake biota during the 1960s and 1970s, whereas the most recent samples indicated eutrophication. Thus, the impact of AMD in Lake Kirkkojärvi is currently relatively minor and is probably masked by eutrophication. However, the bay area adjacent to the tailings may still suffer from chronic pollution, which remains unnoticed when only the central basin is considered. Moreover, shallow littoral areas such as bay environments, which are usually neglected in pollution studies, are well-known hotspots for lake biodiversity and highly important for lake functioning (e.g. Hampton et al. 2011). For this reason, shallow lakes and wetlands are also regarded as sites of special importance in European Commission policies (e.g. Gattenlöhner et al. 2004). In fact, Lake Viljakkalanselkä, which is part of the same lake complex as Lake Kirkkojärvi (Fig. 1), is rich in protected bird and amphibian species. For example, the nearby Alholahti bay, located 1 km south of Lake Kirkkojärvi (Fig. 1), is part of the Natura 2000 network, hosting numerous protected bird, invertebrate, amphibian, and vertebrate species (Pitkänen 2007). Due to its proximity, the sheltered bays of Lake Kirkkojärvi could thus provide additional habitats for the biota mentioned above.

Zooplankton (e.g. Jeppesen et al. 2011), and cladocerans (water fleas) in particular (Eggermont and Martens 2011), are regarded as good indicators of environmental change. Over the past decades, cladocerans have extensively been used in ecotoxicology (Sarma and Nandini 2006) and also in other fields of environmental stress research (Suhett et al. 2015). Because cladocerans are sensitive to heavy metal pollution (Brix et al. 2001; Von Der Ohe and Liess 2004), they can be used to track the ecological impact of metal-contaminated mine drainage. However, two problems are usually associated with research on cladoceran species assemblages in relation to decades of pollution. First, long-lasting sampling programs, which could be used to assess the long-term effects of AMD, are practically non-existent. Secondly, the cladoceran community fluctuates considerably during the year (Whiteside 1974; George and Edwards 1974), which hinders both sampling and the interpretation of the results. The paleolimnological approach, however, can overcome both of these problems, because cladocerans leave well-preserved, identifiable sub-fossil remains in the lake sediment (Korhola and Rautio 2001). These remains can then be used to construct both the past cladoceran communities and past environmental conditions. Cladocerans have been used in paleolimnology since the 1950s (Frey 1986), and have successfully been applied in mining pollution research. For example, Bradbury and Megard (1972) used cladoceran subfossils to assess the impact of 19th century iron mining in the United States. Kerfoot et al. (1999) detected a pronounced decline in the flux of *Bosmina* remains due to copper mine effluent. Doig et al. (2015) concluded that cladoceran abundance was severely reduced due to metal-contaminated mine water, and Sienkiewicz and Gąsiorowski (2016) assessed the impact of acidification and natural neutralization on cladoceran assemblages in a mining pit lake located along the border between Poland and Germany.

The working hypothesis of the present study was that even though the strongest impacts of AMD are no longer detectable in the deeper, central area of the Lake Kirkkojärvi, impacts can still be detected in the proximal tailings area. To assess the contemporary effect of the chronic AMD, which has already lasted over 50 years since the closure of the Haveri mine (Parviainen et al. 2012), we compared the pre-mining and contemporary cladoceran communities in a small bay in the close vicinity of the tailings dump (Fig. 1). In addition, this study aimed to verify the spatial boundaries of the ecologically damaged area through multiple sampling. Our results provide new information on the effects of AMD on lake biota at northern latitudes, where species assemblages and environmental conditions undergo large seasonal changes. This information is relevant to stakeholders and decision makers regarding protection planning, remediation actions, or water usage restrictions in cases where littoral areas of lakes are under threat of pollution.

2 Materials and Methods

2.1 Characteristics and environmental history of Lake Kirkkojärvi

Lake Kirkkojärvi is located in SW Finland (61.7148 N 23.2677 E WGS84) at 83 m a.s.l. (Fig. 1). Annual averages for air temperature and precipitation are 4–5 °C and 600–650 mm y^{-1}, respectively (Finnish Meteorological Institute 2015), and the average winter ice-cover period in the region is approximately five months from December to April (Korhonen 2005). The lake area is 0.72 km^2, the maximum depth is about 8.5 meters, and the mean depth is 2.5 meters. Lake Kirkkojärvi is a sub-basin of a larger lake, Lake Viljakkalanselkä (5 km^2), which in turn is connected to Lake Kyrösjärvi (92 km^2) (Fig. 1). According to the national lake monitoring data, the nutrient concentrations were higher in Lake Kirkkojärvi than in the neighboring Lake Viljakkalanselkä during the 1960s and 1970s. Furthermore, the water is substantially clearer in Lake Viljakkalanselkä compared to Lake Kirkkojärvi. The water of Lake Kyrösjärvi is characterized by higher concentrations of humic compounds. The town of Viljakkala has a population of approximately 2000 and the land cover in the region is characterized by farmland and forests. According to historical maps, the cultivated area increased in the Lake Kirkkojärvi catchment between 1848 and 1955, but has been constantly declining since 1955, mainly as a result of the expansion of the town center. According to Vänni (1928), the water level of Lake Kirkkojärvi was lowered by approximately 3 meters during multiple occasions in the 19^{th} century. This decreased the lake area and water volume, and probably resulted in increased erosion and transport of mineral matter into the lake. Moreover, the lowering of the water level terminated the water exchange between Lake Kirkkojärvi and the more humic Lake Kyrösjärvi due to the drying of the connecting channel located between the former Inkula island and Viljakkala. In addition, prior to the lowering of the water level, the northern connection to Lake Viljakkalanselkä was deeper and wider (Fig. 1). These pronounced changes in the hydrology of Lake Kirkkojärvi during the 19^{th} century may have also affected the lake water characteristics due to changes in morphometry, which regulates many important processes and variables in a lake system (Håkanson 2004). In addition, shallow lakes are generally more prone to changes than deep lakes because of their lower heat (Shutner et al. 1983) and dilution (Janse et al. 2008) capacity. However, no data are available regarding the water quality of Lake Kirkkojärvi during 19^{th} century.

The Haveri Cu–Au mine was established in 1938. Waste rock and tailings material, totaling 1 400 000 tons, were deposited onto the former lake bed and restricted by dam structure (Fig. 1). In the late 1950s, the dam system broke down and a direct waterway from the tailings to Lake Kivijärvi opened up. At the same time, tailings were also dumped directly into the bay (Räisänen et al. 2015), which resulted in a dramatic decrease in the water depth at the head of the bay. The tailings contain high concentrations of potentially toxic metals. For example, the average of Cu concentration is currently 831 mg/kg (Parviainen 2009). In the 1960s, mining ceased and the Haveri mine was abandoned. Because of the unsuccessful and insufficient rehabilitation measures, AMD started almost immediately and, according to sedimentological studies, the most intense metal load into the lake occurred during the 1960s and 1970s (Parviainen et al. 2012). In 2006, the pH of the water in the tailings drainage ditch, which drains into the head of the bay, was 3.5, and high concentrations of dissolved metals (e.g. Al 23 600 $\mu g\ L^{-1}$, Co 722 $\mu g\ L^{-1}$, Cu 1660 $\mu g\ L^{-1}$, Ni 708 $\mu g\ L^{-1}$, and Zn 763 $\mu g\ L^{-1}$) were still recorded (Parviainen 2009). However, the AMD was found to be relatively rapidly diluted on entering Lake Kirkkojärvi. In 2006, the pollutant concentrations at the mouth of the bay were already clearly lower than in the drainage ditch (Al 88.8 $\mu g\ L^{-1}$, Co 0.4 $\mu g\ L^{-1}$, Cu 8.2 $\mu g\ L^{-1}$, Ni 3.3 $\mu g\ L^{-1}$, and Zn 2.5 $\mu g\ L^{-1}$). However, the sedimentary concentrations of pollutants, such as Cu, are still clearly above pre-mining levels (Parviainen et al. 2012), suggesting ongoing although clearly reduced pollution.

2.2 Study site

The study site was a bay that lies in the western part of Lake Kirkkojärvi and is the closest water body adjoining the tailings disposal site. The bay is approximately 300 m long and 130 m wide (Fig. 1). The maximum depth in the middle part of the bay is 3 meters, whereas the western part of the bay comprises very shallow wetland (<0.5 meters). The bay has a shallow sill (1 m) across the inlet, which partly restricts the entrance of material from the main basin. Vegetation is most abundant in the western section of the bay and mainly consists of *Equisetum fluviatile* (L.), whereas *Potamogeton natans* (L.), *Sparganium emersum* (Rehmann), and *Nuphar lutea* (L.) are relatively sparse.

2.3 Sediment and water sampling

We used the top–bottom (after–before) approach, where the bottom sample is assumed to represent a time period before a given point of interest, i.e. the time before the commencement of the mining operations, whereas the top sample represents the contemporary situation. The use of two snapshots enables the comparison of environmental conditions between two time periods over a large area compared to the time-consuming sequential study of a whole core (e.g. Michelutti et al. 2001; Weckström et al. 2003; Korosi et al. 2012). A total of 18 samples from 9 sites were retrieved from the AMD-impacted bay with a Russian peat corer in May 2016. The core length ranged from 0.35 m to 1.35 m. Each core was sectioned using a metal slate, and the samples were stored in plastic zip-lock bags and kept at ~4 °C prior to analysis. The top sample (1 cm) represents the contemporary period, whereas the bottom sample (1 cm) was retrieved from varying sediment depths (0.20–1.30

m). The bottom samples were assumed to represent the time before mining on the basis of the sediment characteristics, such as pronounced layers of tailings material, and earlier studies (Kihlman and Kauppila 2010). We did not use any conventional dating methods (e.g. radiometric methods) to date the bottom samples, because the direct dumping of tailings material has probably resulted in considerable resuspension of lake sediments. To determine whether the ecological effects of AMD could be detected along a spatial gradient from the area near the tailings towards the mouth of the bay, we organized the sampling to cover the whole bay, except for the shallow (<0.5 m) bay head, where the ice cover reached the lake bottom. In addition, we collected two surface sediment samples from presumably unpolluted reference sites to investigate the contemporary cladoceran communities in the bay environments, which are currently under a different type of human impact. The Lake Kirkkojärvi reference site (R1; Fig. 1) is probably affected by farming due to the high proportion of fields in the surroundings of the sampling site. According to old maps, the nearshore area has been under cultivation at least since 1800 AD. Aquatic vegetation is abundant and consists of the same species as in the polluted bay. In contrast, the Lake Viljakkalanselkä reference site (R2; Fig. 1) is characterized by high cliffs and a lack of farming and other intensive human activity in the nearby land areas, and the sparse aquatic vegetation is restricted to a narrow zone near the shoreline.

To verify whether AMD is still leaching from the tailings area, we measured the water pH (Merck pH-indicator paper) in the gullies flowing on top of tailings dump. In addition, we collected water samples from the polluted bay (sites 1 and 9; Fig. 1) and from Lake Viljakkalanselkä (W; Fig. 1) to assess the current concentrations of nutrients and selected metals (Al, Cu and Ni). These same elements exhibited clearly elevated levels in samples retrieved from the ditches on the top of the tailings in 2006–2007 (Parviainen 2009) (Fig. 1, Table 1). The water sample reference site W was chosen to represent a similar environment in terms of water depth and catchment land cover as in the main study bay. Water samples (0.5 L) were retrieved from the depth of 0.5 meters into polyethylene bottles in July 2016 and stored in the dark at 4 °C prior to analysis. The water samples were analyzed for total ionic concentrations in accordance with ISO 17294-2:2003 at the Metropolilab Helsinki environmental laboratory, which is an accredited testing laboratory (FINAS T058).

2.4 Cladoceran analysis

Cladoceran analysis and species identification were conducted according to Korhola and Rautio (2001) and Sarmaja-Korjonen and Szeroczyñska (2007). Briefly, sediment samples of approximately 1 cm^3 were first mixed with 10% KOH and heated to ca. 90 °C on a hotplate for 45 minutes. The samples were sieved through a 50-µm mesh using tap water and the residue was pipetted into test tubes and centrifuged. Excess water was removed and permanent microscopic slides were prepared with safranin-stained gelatin glycerol for microscopic analysis. Cladoceran body components were identified and the number of individuals was based on the most numerous component. Here, a minimum of 133 individuals were enumerated per sample.

2.5 Numerical analysis

To examine the differences between pre-mining and contemporary cladoceran assemblages, we used analysis of similarity (ANOSIM), which is a non-parametric test for significant differences between groups (Clarke 1993). Proportional cladoceran data were log(x+1) transformed prior to ANOSIM calculations to stabilize the variance. In addition, we conducted the non-metric multidimensional scaling (NMDS) procedure to examine the variation in our samples. The Bray-Curtis dissimilarity index was used in the ANOSIM test and in NMDS. The latter is suitable for biological community data containing zero values (Minchin 1987). We used Pearson's correlation method to assess the relationship between cladoceran species and both distance from the tailings and water depth. The Shannon H' diversity index was used to rate species diversity in samples. Numerical analysis was conducted using PAST statistics software 3.1. (Hammer et al. 2001).

3 Results

3.1 Sediment properties

Sediment stratigraphy varied between sites. The upper 2- to 5-cm-thick layers of each core consisted of brown organic sediment, while each core included a black section of varying thickness (5–50 cm) below the organic surface layer. The black layer consisted of extremely fine mineral matter corresponding to tailings material. In cores 3, 5, 6, 7, and 9 (Fig. 1), there was also a layer of watery clay up to 30 cm thick, which transitioned again to a black layer and gradually to light brown sediment (Fig. 2). In other cores without a distinct clay layer, the black layer gradually changed to brown material. All bottom samples were taken from the brown layer below the transition zone, except for sites 2 and 3, where the corer did not reach the brown sediment layer.

3.2 Water quality

Because the ditch leading from the tailings dump into the lake via the failed dam was dry, we could not measure the dilution of AMD in the Kirkkojärvi bay during our sampling campaign. However, the water on top of the tailings was clearly acidic with a pH of 3.5. Nutrient and metal concentrations were slightly higher in the Kirkkojärvi bay compared to the Lake Viljakkalanselkä water quality reference site W. The highest concentrations of nutrients and metals were detected in the mouth of the polluted bay (Table 2). Our water sample results are highly similar to those of Parviainen (2009) regarding metal concentrations in a lake water sample at the mouth of the bay during the dry season.

3.3 Cladoceran assemblages

Cladoceran subfossil remains were numerous and generally well preserved, except in bottom samples from coring sites 2 and 3, which consisted of tailings material and did not contain any cladoceran remains. However, because of the high similarity among the pre-mining samples, we decided not to retrieve additional bottom samples from sites 2 and 3. The total number of individuals enumerated was 1574 in all contemporary samples (9 samples) and 1279 in pre-mining samples (7 samples), whereas the number of enumerated individuals at the reference sites was 133 for R1 (1 sample) and 166 for R2 (1 sample). Altogether, 38 taxa were recorded. The average number of taxa identified was 18 in the contemporary samples and 20 in the pre-mining samples. Planktonic species exhibited the highest proportions in all samples. The overall proportion of planktonic taxa (as classified in Bjerring et al. 2009) was lower in the top samples (71.1%, SD 8.3) compared to the bottom samples (76.7%, SD 4.2). The species composition in the samples is presented in Table 3. The average Shannon diversity index was 0.927 for the pre-mining period and 1.528 for the contemporary period. The cladoceran community differed significantly between the pre-mining and contemporary samples (ANOSIM $R = 0.9081$; $p = 0.0001$). However, as illustrated in NMDS plot, the cladoceran composition was highly similar in the contemporary and pre-mining samples of the polluted bay (Fig. 3). Contemporary cladoceran communities in the non-agricultural reference site R2 closely resembled the pre-mining samples of the polluted bay, whereas the cladoceran assemblage in the agriculturally impacted reference site R1 was more similar to the contemporary communities in the polluted bay (Fig. 3, Table 3). The most visible change in cladoceran species assemblages of pre-mining and contemporary samples was the appearance or higher relative abundances of *Bosmina longirostris* and *Alonella nana* in the contemporary samples, whereas in the pre-mining samples, these taxa were practically absent. Moreover, *Eubosmina longispina*, *Rhynchotalona falcate*, and *Monospilus dispar* had generally lower abundances in the contemporary samples (Table 3). In contrast to the pronounced differences between pre- and post-mining cladoceran communities, differences between near-tailing samples and the samples retrieved from the mouth of the bay were nearly non-existent (e.g. sample numbers 2 and 9). This is clearly illustrated in the NMDS scatterplot, where the contemporary samples are situated close to each other without any correlation with distance from the AMD source (Fig. 3). In the species-level correlation test, only *A. nana* showed a significant correlation with distance from the AMD (Pearson correlation -0.74; $p = 0.022$), as it was more abundant close to the tailings regardless of the water depth.

4 Discussion

The origin of the clay section in samples 3, 5, 6, 7, and 9 (Fig. 1) is unknown, but may be related to the disposal of tailings and consequent disturbance of the original bottom material or to the construction of the dam structures. The sediments underlying the tailings are comprised of silt and clay (Parviainen 2009), which may have been mobilized during the construction work. Flood events (Thorndycraft et al. 1998) and other catchment-related disturbances (Dearing 1991) are also known to induce pronounced resuspension and sedimentation processes resulting in distinct zones in the sediment record. The relatively thin layer of natural lake sediment above the tailings material compared to the deep section of the lake (Kihlman and Kauppila 2010) indicates different sedimentation dynamics in the shallow bay (e.g. Boggs 2006). Acidic mine water is clearly still being produced on top of the tailings dump, as predicted by Parviainen (2009). However, at least during the dry season, relatively little AMD leaks into Lake Kirkkojärvi. Nevertheless, even in the absence of direct AMD inflow, metal concentrations are higher in the polluted bay than in Lake Viljakkalanselkä (W). This is probably caused either by continuous dam leaching or other sources, such as sedimentary release or runoff from the catchment area. The slightly lower metal concentrations at sampling site 1 are probably connected to the dense beds of *E. fluviatile*, which is known to have an extremely high potential for metal accumulation (Bateman 1999). This sampling site is also more sheltered compared to site 9, for example, which in turn is more exposed to wave-induced resuspension of sediment due to the larger effective fetch (see e.g. Dearing 1997). The difference in cladoceran species numbers between the impacted bay and the reference sites can most probably be explained by the higher number of counted individuals in the impacted bay. Because of this, rare species were probably missed at the reference sites. In all samples, the cladoceran community was largely composed of species that are common in North European lakes (e.g. Bjerring et al. 2009).

4.1 Comparison of pre-mining and contemporary communities

It is not clear whether our bottom samples were deposited prior to the period of repeated water level manipulations in the latter half of the 19th century, but because the sedimentation rate in Lake Kirkkojärvi is approximately 0.5 cm y^{-1} (Kihlman and Kauppila 2010), and because black tailings material was probably deposited very rapidly, the depths of the bottom samples (5 to 15 cm below the tailings material) are probably not enough for them to originate from the first half of the 19th century. The water depth at the study site was certainly affected by the dumping of tailings material in the 1950s, which resulted in bottom elevation, particularly at the head of the bay. This may explain the higher proportion of planktonic species in pre-mining samples, as the planktonic/littoral ratio is generally considered a good indicator of water level fluctuations (e.g. Korhola et al. 2005; Nevalainen et al. 2011). At the species level, the change in the planktonic/littoral ratio resulted from an increased abundance of *Alonella nana,* which is regarded as a macrophyte-associated species (Bjerring et al. 2009; Adamczuk 2014). This is consistent with the high abundance of aquatic vegetation at the head of the bay.

In northern Europe, *E. longispina* is more common in acidic environments than *B. longirostris* (Uimonen-Simola and Tolonen 1987; Bērziņš and Bertilsson 1990; Nilssen and Sandoy 1990), for which reason the higher relative abundance of *B. longirostris* in the contemporary samples of the studied bay may suggest that the effect of mine effluent is not so significant at our study site. Moreover, European populations of *B. longirostris* are regarded as intolerant of copper pollution (Koivisto et al. 1992; Bossuyt and Janssen 2005) and should not be able to thrive if Cu pollution is still high enough to cause biological damage. The literature considering the acidity preferences of littoral species is more limited, but some general classification proposals exist. According to Krause-Dellin and Steinberg (1986), *R. falcata* is an acidophilic species, whereas *M. dispar* is classified as alkaliphilous. *M. dispar* was present in lower proportions in our contemporary samples, but the similar change within *R. falacta* undermines any generalizations regarding the impact of AMD and littoral taxa. *Chydorus sphaericus*, which was more abundant in the contemporary samples, is known to tolerate acidic mine pollution relatively well (Belyaeva and Deneke, 2007; Sienkiewicz and Gąsiorowski, 2016). However, as the abundance of *C. sphaericus* was also high at the reference site R1, its recent success in Lake Kirkkojärvi probably results from factors other than mine pollution. An interesting species in this regard is also *A. nana*, which was more frequent in contemporary samples and had the highest contemporary abundance at the head of the bay, where the AMD impact is strongest. Many cladoceran species are known to adapt to copper pollution (Bossuyt and Janssen 2005; Agra et al. 2011), but no information regarding the tolerance or adaptation of *A. nana* to AMD or metals is currently available. However, *A. nana* was also common at reference site R1, and the high relative abundance near the bay head may be due to other factors than its pollution tolerance, such as the high amount of aquatic vegetation in the shallow head of the bay.

The observed low ecological impact of mine pollution may be connected to the seasonality of both AMD and cladocerans. The highest influx of AMD into the investigated part of the bay occurs during the spring snowmelt, when most of the cladoceran species are still absent (e.g. Koksvik 1995). Cladoceran numbers are usually very low during winter, as communities emerge later in spring from dormant eggs (ephippia). This coincidence may explain the lack of a pollution effect on cladocerans at our study site. Potentially high mortality among overwintering individuals is later compensated by resting egg hatchlings. In addition, rainy seasons may also affect the AMD input (Parviainen 2009), which probably results in marked fluctuation in AMD-associated water parameters. The role of dilution with almost neutral water is crucial to the attenuation of AMD toxicity (Filipek et al. 1987; Yu and Heo 2001; Navarro Torres et al. 2011). This is mainly because pH is one of the most important regulators of the bioavailability of metals in AMD (Chapman et al. 1983; John and Leventhal 1995), and the damaging effect of acidity itself also decreases with increasing neutrality. Moreover, cladocerans are capable of escaping from polluted environments (Lopes et al. 2014), and even if the summer community of cladocerans is greatly altered by AMD during occasional heavy rains, the effect may not be visible in our samples if the bay is rapidly recolonized by cladocerans from less-impacted areas of the lake. Generally, cladocerans are passively transported by wind-induced water movements, but are also known to exhibit horizontal migration (e.g. Burks et al. 2002) and are rapid colonizers of newly formed habitats, even when a water connection does not exist (e.g. Louette and De Meester 2005). In addition, based on the almost neutral pH and low Cu concentrations in water samples 1 and 9, the submerged tailings do not affect the water quality. This is also suggested by the presence of *B. longirostris*, which is a Cu- and pH-sensitive species. In addition, the black color of the tailings suggests that no notable alteration or metal release has taken place, probably due to the anoxic conditions (e.g. Salomons 1995).

Because the observed changes in alkaliphilous and acidophilous cladoceran species are not in close agreement with what would be expected if AMD had a strong effect on the community, there must be other explanations for the detected community change. Kihlman and Kauppila (2010) noted clear changes in diatom and lacustrine protists communities in sedimentary samples retrieved from the deepest part of Lake Kirkkojärvi covering the last ca. 70 years. Ecological shifts were seen to coincide with sedimentary metal peaks of the 1960s and 1970s, and were interpreted to indicate the most intense AMD release from the tailings. However, diatom and lacustrine protist communities have not returned to their pre-mining composition, but instead to a community structure that indicates nutrient enrichment rather than pollution (Kihlman and Kauppila 2010).

In the absence of convincing evidence regarding the impact of AMD on cladoceran populations, there are relatively strong indications that the community change in the contemporary lake sediments has resulted from eutrophication. Many studies have suggested *C. sphaericus* and *B. longirostris* to be indicators of eutrophication (e.g. Whiteside 1970; Korhola 1990; Korponai et al. 2011; Nevalainen and Luoto 2013). In particular, the replacement of *Eubosmina* by *B. longirostris* is considered as a typical result of eutrophication (Goulden 1964; Crisman and Whitehead 1978). The higher abundance of *B. longirostris* is most probably related to food availability, as the food concentration has been noted to correlate positively with most reproductive parameters of *B. longirostris* (Urabe 1991; Mason and Abdul-Hussein 1991). In addition, *B. longirostris* is known to have the capacity to withstand the toxins of blue-green algal, which often thrive in eutrophicated waters, better than many other cladoceran species (Fulton 1988). The small-bodied *Bosmina* species have low food reserves and rapidly die due to starvation (Goulden and Hornig 1980). Therefore, the food supply must be permanently adequate to sustain the increasing population and allow it to dominate the cladoceran community. Moreover, *R. falcata* was classified as an oligotrophic/acidophilic species by Bjerring et al. (2009), and its disappearance during the last few decades from the bay of Lake Kirkkojärvi may indicate eutrophication. In addition, the higher Shannon diversity in the contemporary samples may indicate eutrophication, as demonstrated by Korponai et al. (2011). The fact that the contemporary cladoceran community at the reference site located in the less eutrophic Lake Viljakkalanselkä (R2) was found to resemble that of the pre-mining era in Kirkkojärvi bay also suggests possible eutrophication of Lake Kirkkojärvi bay. Furthermore, the diatom-based total phosphorus and pH models constructed by Kihlman and Kauppila (2010) indicate eutrophication instead of acidification. The modeled phosphorus levels have increased from approximately 15 $\mu g\ L^{-1}$ to 21 $\mu g\ L^{-1}$ during the past decades, whereas pH has modestly increased from 6.6 to 6.8. As noted by Kihlman and Kauppila (2010), the eutrophication has been gradual and the community change has been directional. It is thus likely that the increased nutrient concentrations derived from agricultural land use in the catchment have been shaping the cladoceran communities of the lake more in the recent past than the AMD from the closed mine. Lake Kirkkojärvi is thus an example of a lake where human impacts, such as eutrophication, have turned a lake pollution case into a multiple-stressor problem (e.g. Ormerod et al. 2010).

5 Conclusions

Although harmful mine drainage water globally affects aquatic ecosystems, most lakes are also subjected local low-intensity disturbances such as anthropogenic eutrophication. Multiple-stressor scenarios are of particular importance in shallow lakes and nearshore areas. These are the most vulnerable environments due to the low water volume, but are ecologically highly important for lake functioning. In Lake Kirkkojärvi, the cladoceran community of the pre-mining era differed significantly from the contemporary community of the lake, and closely resembled the contemporary community of a nearby non-polluted reference site. The relatively weak, yet chronic, disturbance signal due to AMD is most likely overridden by eutrophication. The cladocerans avoid the spring peak period of AMD pollution through dormancy and are more strongly affected by water characteristics in the summer. Moreover, the relatively thin layer of lake sediment above the tailings sediment sequence is adequate to protect the cladoceran community from the harmful effects of toxic substances. This is highly important in terms of remediation planning. The results of this study emphasize the importance of local conditions and species life strategies and highlight the difficulty in making any generalizations regarding pollution impacts, especially if they are assessed using lake biota as environmental indicators.

Compliance with Ethical Standards

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Conflict of interest

The authors declare that they have no conflicts of interest.

Ethical approval

This article does not contain any studies with human participants or animals performed by any of the authors.

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Figure captions

Fig. 1 Location of the study site and the sampling sites

Fig. 2 Simplified sediment stratigraphy, sampling horizons, and thickness variation of different zones

Fig. 3 NMDS bi-plot for pre-mining samples (filled circles) and contemporary samples (open circles). Stars represent reference samples R1 (agriculturally impacted) and R2 (non-agriculturally impacted). Stress = 0.075.