Journal of Paleolimnology (2006) 35: 289–303
DOI 10.1007/s10933-005-0483-z

History of mine drainage impact on Lake Orijärvi algal communities, SW Finland

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Received 14 April 2005; accepted in revised form 20 June 2005

Key words: Acid mine drainage, Chrysophytes, Diatoms, Finland, Heavy metals, Mining, Paleolimnology

Abstract

The Cu (Pb, Zn) mine of Orijärvi (1757–1956) was the first mining operation in Finland where flotation techniques (1911–1955) were used to enrich ore. Large quantities of tailings were produced. The impacts of past mining activities on the aquatic ecosystem of nearby Lake Orijärvi were studied using a combination of paleolimnological methods (analysis of sedimentary diatom frustules, chrysophycean cysts, metal concentrations and radiometric datings). The acid mine drainage (AMD) – derived metal impact to the lake was found to be the strongest thus far recorded in Finland. Concentrations of Cu, Pb and Zn in sediments are two to three orders of magnitude higher than background values. During the most severe loading phase, there were practically no algae in the lake. *Achnanthes minutissima* was the hardiest species able to tolerate increased metal contents. The metal load has changed the properties of sediments in such a way that chrysophycean cysts were impossible to identify because of coating and corrosion. Lake water still has elevated heavy metal concentrations, indicating that the impact from the tailings area continues to affect the lake. It has low productivity, and the planktic diatom community is still not developed. The study demonstrates that unremediated mining areas form a major risk to the environment. The damage to aquatic ecosystem can remain severe for decades after the mining activities have ceased.

Introduction

Acid mine drainage (AMD) is one of the most serious unresolved environmental problems diminishing the quality of surface waters. The acid is generated by oxidation of iron-rich sulphides which leads to a lowering of pH and further dissolution of toxic metals from water-permeable tailings and waste rocks. Once conditions favouring acid drainage have been created, significant impacts may continue for hundreds of years (Ek and Renberg 2001) and may be very expensive to manage (Price 2003). The problem is extensive in Finland, where, during the past few centuries, nearly 100 sulphide ore deposits have been actively mined (Puustinen 1997). Most of the abandoned mines have been closed without any remediation activities, and little is known about their potential or present impacts on the environment (Salminen and Sipilä 1995; Åström and Nylund 2000; Räisänen and Tauriainen 2001).

Impacts of mine drainage on lake ecosystems are caused by released toxic metals such as nickel, copper, zinc, cadmium, lead and mercury, which may be detrimental to aquatic organisms (Arnason and Flecther 2003). However, toxic aquatic impacts from metals are difficult to prove using paleolimnological evidence based on algal remains in sediments. Ek and Renberg (2001) noted that in Falun, Sweden, the main effect on the lake ecosystem has been lowering of pH by 0.8–0.4 units. The impacts from metal pollution were not possible to differentiate from the concurrent acidification. The same has been observed in the Sudbury area, where metal emissions are associated with those of sulphur, and the impacts from mining can only be detected in the form of lowered pH (Dixit et al. 1995).

On the other hand, there are indications of metal toxicity on the aquatic ecosystem. Michelutti et al. (2001) observed an increase in concentrations of copper, nickel, cobalt, barium and zinc in lake surface sediments of the Norilsk area, Russia, due to the mining activities. The metal load also had influenced changes in diatom assemblages in the form of an increase of small benthic *Fragilaria* species. In the Pechenga area, Kola Peninsula, 60 years of intensive production of nickel, copper and zinc has impacted lakes causing more than a 10-fold increase in metal concentration in sediments, leading to unstable phytoplankton communities (Lukin et al. 2003). Moreover, sediment toxicity from lakes impacted by mining in the Rouyn-Noranda (Québec, Canada) area was observed to have killed benthic organisms such as amphipods, mayflies and tanytarsid midges (Borgmann et al. 2004). In this case toxic metals were cadmium as well as copper to a minor degree. Goldenville mine tailings in Nova Scotia (Canada) have impacted nearby Gegogan Lake from 1860 to 1945 leading to poisoning of the benthic community and to a total fish loss (Wong et al. 1999). At this site, there was a continuous release of arsenic, cadmium, lead and mercury from mining waste. Recently, researchers in Québec used both diatom and geochemical indicators to track pollution in Lac Dufault (Canada) lake, and noted morphological changes in diatoms as well as shifts in species composition with pollution (Cattaneo et al. 2004; Couillard et al. 2004). Paleolimnologists have also used invertebrate indicators to track effects of metal pollution (e.g., Brooks et al. 2005).

Lake Orta in Italy was found to be exceptionally suitable to study metal pollution and its impact on diatoms (Ruggiu et al. 1998). The metal load to the lake has been as much as 60–70 t/year, leading to a water concentration of 100 $\mu g\ l^{-1}$ copper in the late-1950s. This led to a collapse of the aquatic ecosystem including severe changes in diatom assemblages. The sediment study performed by Ruggiu et al. (1998) indicated that, since the beginning of pollution, some diatom species have disappeared (mostly *Fragilaria* and *Cyclotella* species). Others were not affected, and the third group, mostly belonging to *Achnanthes*, increased, indicating remarkably tolerance for copper load.

Paleolimnology is being used increasingly in applied environmental studies to track changes related to human activities, both in Finland (e.g., Hynynen et al. 2004) and elsewhere (e.g., Bradbury et al. 2004). In this study, we sought to investigate the toxic metal impacts on diatom and chrysophycean assemblages applying paleolimnological techniques. The purpose was to compare two proxies from a radiometrically dated sediment sequence which has received the strongest AMD-derived metal load observed in Finland. The studied heavy metal pollution originates from wastes of an old polysulphidic ore, the Orijärvi mine, which was in operation from 1757 to 1954 (Turunen 1957). The concentrating plant (active from 1911 to 1955) and the AMD generating tailings area are in the immediate proximity of Lake Orijärvi (Figure 1).

Description of the site and the metal load

The Orijärvi mine and Lake Orijärvi (60°14′ N 23°35′ E) are situated in the temperate zone of southwestern Finland, about 70 km west from Helsinki. The ore-field is composed of several occurrences of massive sulphide deposits associated with supracrustal and granitic rocks of the leptite zone. The massive Cu–Pb–Zn ore of Orijärvi is hosted in 1.9 Ga hydrothermically altered acid metavolcanic and karstic rocks (Papunen 1986). The Orijärvi ores are associated with cordierite-antophyllite rocks (Cu–Zn ore) or diopside-tremolite rocks (Zn–Cu–Pb ore). The ore minerals are chalcopyrite, galena and sphalerite. In addition, there are pyrite and pyrrhotite (Latvalahti 1979).

Mining started in 1758. Since then, during 200 years of mining, a total of 1,200,000 tons of sulphide ore and country rocks were exploited and

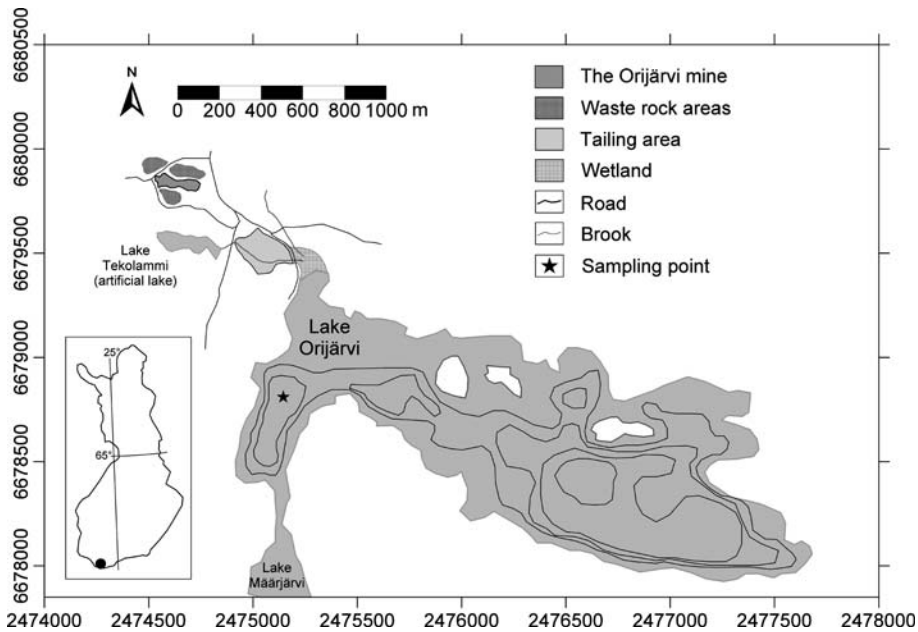

Figure 1. The location of mine, waste rock piles and the tailings on the shores of Lake Orijärvi. Bathymetric lines are at 5 m intervals. Coordinates are according to the Finnish National Grid.

enriched for copper, zinc and lead concentrates. During the first 150 years, only copper was mined. The production was less than about 100 tons of copper annually and ceased totally at the end of the 1800s. The enrichment plant with a flotation line was established in 1911 in order to produce zinc from the former gauge minerals. After a short break, the copper production started anew in the 1930s, and during the years 1945–1956 the ore was intensively exploited for copper, zinc and lead concentrates. Total production of the mine was 8200 tons of copper, 14,600 tons of zinc and 4100 tons of lead (Turunen 1957).

The concentrating plant and the tailings area, which contains ca. 400,000 tons of mine waste rich in metal sulphides, are directly connected with Lake Orijärvi (Figure 1). The mine environment was never remediated, but open pits, gangue piles and tailings were left uncovered in the open air. Unweathered tailings contain 1.0% copper, 6% zinc and more than 1% lead (Ewurum 2000). The weathered 20–40 cm thick surface of the tailings has 0.2% copper, 0.6% zinc and 0.7% lead.

Lake Orijärvi is 1.7 km^2 in size, has an average depth of 8.5 m and a maximum depth of 21.4 m. The volume of the lake is 14.8 million m^3, and the residence time is 4 years (Vogt 1998). The lake drains a catchment area of 12 km^2 composed of hilly bedrock outcrops (40% of the area) covered partly by a thin layer of late Weichselian sandy till (30%). Topographic depressions are filled with marine silts and clays or are occupied by peat bogs (Kujansuu et al. 1993).

The tailings area is about 5 ha in size. It can be estimated that weathering of tailings has affected the average topmost 50 cm of the material, which equals about 100 000 tons of mine waste. The released total metal load to the lake can thus be estimated to equal 800 tons of copper, 5400 tons of zinc and 300 tons of lead. The leachate water in the brook connecting the tailings area and the lake (Figure 1) is typical AMD water (SO_4 = 217 mg l^{-1}, Pb 40 $\mu g\ l^{-1}$, Cu 640 $\mu g\ l^{-1}$, Zn 14,300 $\mu g\ l^{-1}$) (Räisänen et al. 2005).

Water quality monitoring has been carried out in Lake Orijärvi since the 1960s (Vogt 1998), and the elevated concentration of copper (20–50 $\mu g\ l^{-1}$), lead (1–3 $\mu g\ l^{-1}$), cadmium (2–8 $\mu g\ l^{-1}$) and especially those of zinc (600–1200 $\mu g\ l^{-1}$) indicate that the lake is impacted by mine drainage (Figure 2). Metal concentrations in the lake are 10- (lead) to a

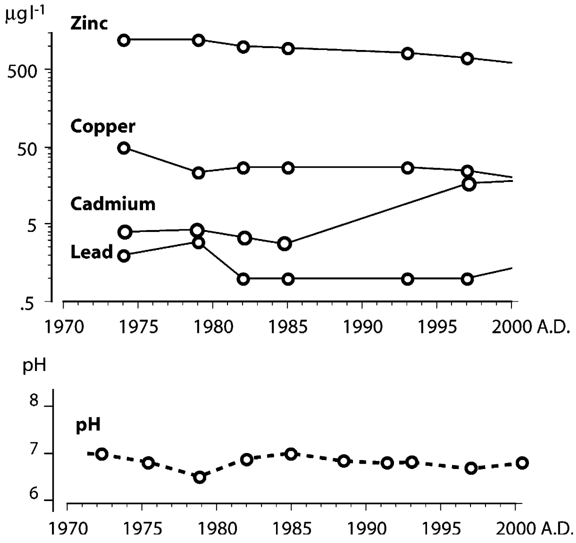

Figure 2. Water quality monitoring data from Lake Orijärvi (1974–2000) for zinc, copper, cadmium and lead, and (1971–2003) for pH (Finland Environmental Institute).

100- (cadmium, zinc, copper) fold higher than average background concentrations in surface waters in Finland (Lahermo et al. 1996).

Materials and methods

The sampling was done from the 11 m deep depression about 800 m from the outer edge of the tailings (Figure 1). Short sediment cores were collected to date the sediment and to analyze its record for metal deposition, diatoms and chrysophycean cysts. Two different corers were used. A crust-freeze core (ORI-1) was collected to describe the sediment structures and to collect material for diatom and chrysophyte cyst analyses. Subsamples for elemental analyses and datings were obtained using the Limnos gravity corer (Kansanen et al. 1991) (ORI-2). ORI-1 and ORI-2 were cored from the same sample location.

Lithofacies properties were described in the field from the frozen sediment surface of the core ORI-1 applying Troels-Smith's (1955) component analysis. Dry weight and total organic matter were determined in each analyzed subsample by weight loss after 105 °C overnight and 550 °C for 2 h, respectively (Boyle 2004).

Analysis for ^{210}Pb and ^{137}Cs was carried out to estimate sediment accumulation rates for the topmost 30 cm of the sampled sediment at the Accelerator Laboratory of the Department of Physics, University of Jyväskylä. Radioactivity of ^{137}Cs was determined to trace the 1986 Chernobyl accident horizon in sediment (Kansanen et al. 1991). ^{210}Pb activity was counted with a PIN 100A α diode. A dating curve was calculated using the C.I.C. model (constant initial concentration), which was in good agreement with ^{137}Cs and soot particle curves presented earlier (Salonen and Tuovinen 2001).

To reconstruct the history of metal load from the past mining activities, the concentrations of Al, As, B, Ba, Be, Ca, Cd, Co, Cr, Fe, K, Mg, Mn, Na, Ni, P, S, Sr, Ti, Pb, Zn and Cu were determined from the core. The 15 topmost samples were analyzed at 1 cm intervals. 200 mg dry sediment samples were digested in closed Teflon vessels using a microwave assisted solid–liquid extracting method with concentrated nitric acid, and analyzed by ICP-AES at the laboratory of the Geological Survey of Finland.

The core ORI-1 was subsampled for diatom analyses as follows: between 1–10 and 1 cm intervals, between 10–20 and 2.5 cm intervals and between 20–60 and 10 cm intervals. Organic matter in the diatom samples was removed by oxidation with 30% H_2O_2 (Battarbee et al. 2001). After rinsing with distilled water, the suitable dilutions of the diatom suspension were allowed to dry on cover slide and mounted using Naphrax® mounting medium. The diatoms were identified to the species level and counted on a microscope at 1250× magnification with phase contrast illumination. Approximately 300 diatom valves per sample were counted and identified along random intersects. The identification and nomenclature follow Krammer and Lange-Bertalot (1986–1991) and Cleve-Euler (1951–1955). One fairly abundant *Navicula*, which was not possible to identify to the species level, was referred to as *Navicula* spp. The assignment of diatom species to ecological and pH groups was done according to Lowe (1974); Krammer and Lange-Bertalot (1986–1991); Stevenson et al. (1991) and van Dam et al. (1994).

The core ORI-1 was subsampled for chrysophyte stomatocyst analysis as follows: surface sediment sample was sliced from 0 to 1 cm, 18 samples between 1 and 9 cm were sliced to 0.5 cm sections, and five samples between 10 and 15 cm to 1 cm sections. Samples were pre-treated according to Battarbee et al. (2001). Siliceous slurries were dried at room temperature on double-sided adhesive tape, which was fixed onto brass stubs. Preparations were coated with graphite using a JEOL JEE 4B evaporator. Scanning electron microscopy was done using a JEOL JSM-T330 using 20 kV. At least 300 stomatocysts were observed from each sample; identification as well as cyst numbering and ecological classification was done according to Duff et al. (1995) and Wilkinson et al. (2001).

Results

Description and dating of the core

The analyzed core consists of four lithofacies units (Figure 3). The lowermost part, from 60 to 8 cm depth, is olive green to greenish grey fine detritus gyttja (Munsell 2.5Y 3/2). Above it, there is a distinct light grey (Munsell 2.5Y 4/2) layer between 8 and 6 cm of the sediment core. This portion is sticky and very fine grained. From 6 to 2 cm the sediment is homogenous olive green fine detritus gyttja with occasional black mottling (Munsell 2.5 Y 3/1). The topmost 2 cm is loose sediment with a brownish colour (Munsell 2.5 Y 4/2).

Measured ^{210}Po activity reached background at the 15 cm level and the measured ^{137}Cs activity at about 7 cm depth. The age distribution curve derived from the C.I.C. model is depicted in Figure 3, as is the ^{137}Cs activity curve. It can be seen that sediment below 15 cm represents layers older than the 1880s. The uppermost 15 cm of sediment has an average rate of 1.4 mm/year for sedimentation. The cesium-curve indicates that the depth level 7 cm represents the late-1940s, and that the Chernobyl-peak from 1986 is at about 3 cm below the sediment surface.

Chemostratigraphy

Analyzed concentrations of the geochemical variables for the uppermost 15 cm of the sediment are shown in Figure 4 as mg kg^{-1} dry

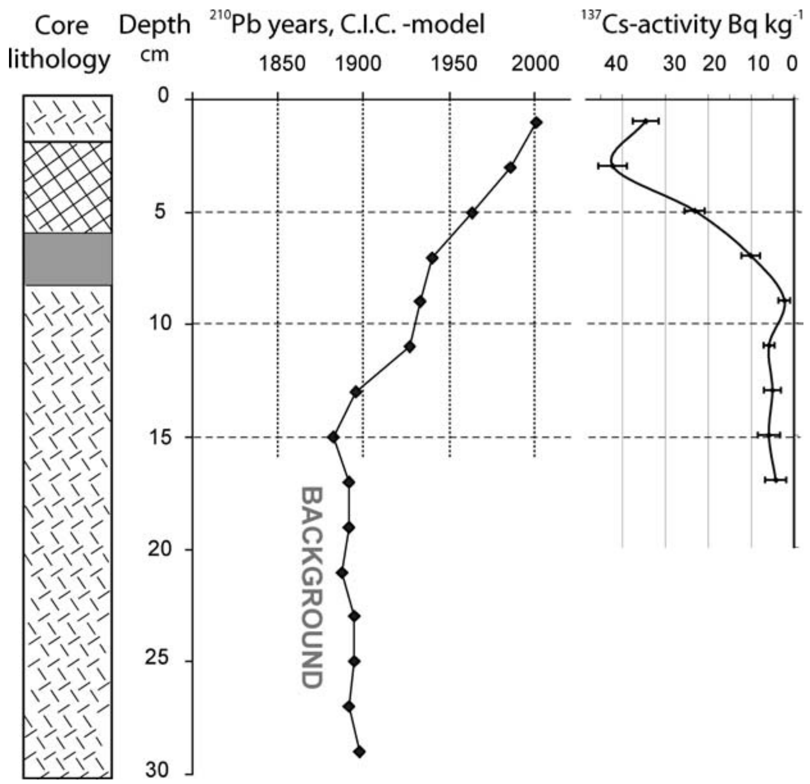

Figure 3. Core lithology, ^{137}Cs-curve, and the time/depth curve using the C.I.C. model for the Lake Orijärvi sediment.

matter. All the analyzed elements reflect marked changes in their concentration records. Certain elements have an increasing trend towards the surface of the sediment. They are aluminum, arsenic, boron, calcium, vanadium, cobalt and manganese. Another group of elements display a decreasing concentration trend with decreasing depth. They are barium, potassium and titanium (Figure 4).

However, most of the metals display a concentration curve peaking in the middle of the analyzed sequence. These are the metals directly associated with the exploitation of the ore, such as sulphur, zinc, cadmium, copper and lead. Iron, nickel and phosphorus follow the same trend, as do sodium and magnesium. Concentrations of heavy metals related to past mining activities are exceptionally high, maximum values being for zinc 3.4%, copper 0.39% and lead 0.29% at the depth zone of 8–5 cm below the sediment surface (Figure 4).

Diatom analyses

A total of 183 diatom taxa were identified from the studied core. For most of the profile (i.e. from 60 cm up to 10 cm) dominant species belong to the genera *Aulacoseira*, *Cyclotella*, and *Tabellaria* (Figure 5). The most abundant species in the genus *Cyclotella* are *C. pseudostelligera*, *C. radiosa*, *C. rossii*, *C. tripartita* and species belonging to the *C. bodanica*-group. The maxima of species *Cyclotella pseudostelligera*, *C. radiosa* and *C. rossii* are at the depth of 20 cm, *Cyclotella rossii* being the most abundant species. In the genus *Aulacoseira* the dominant species is the

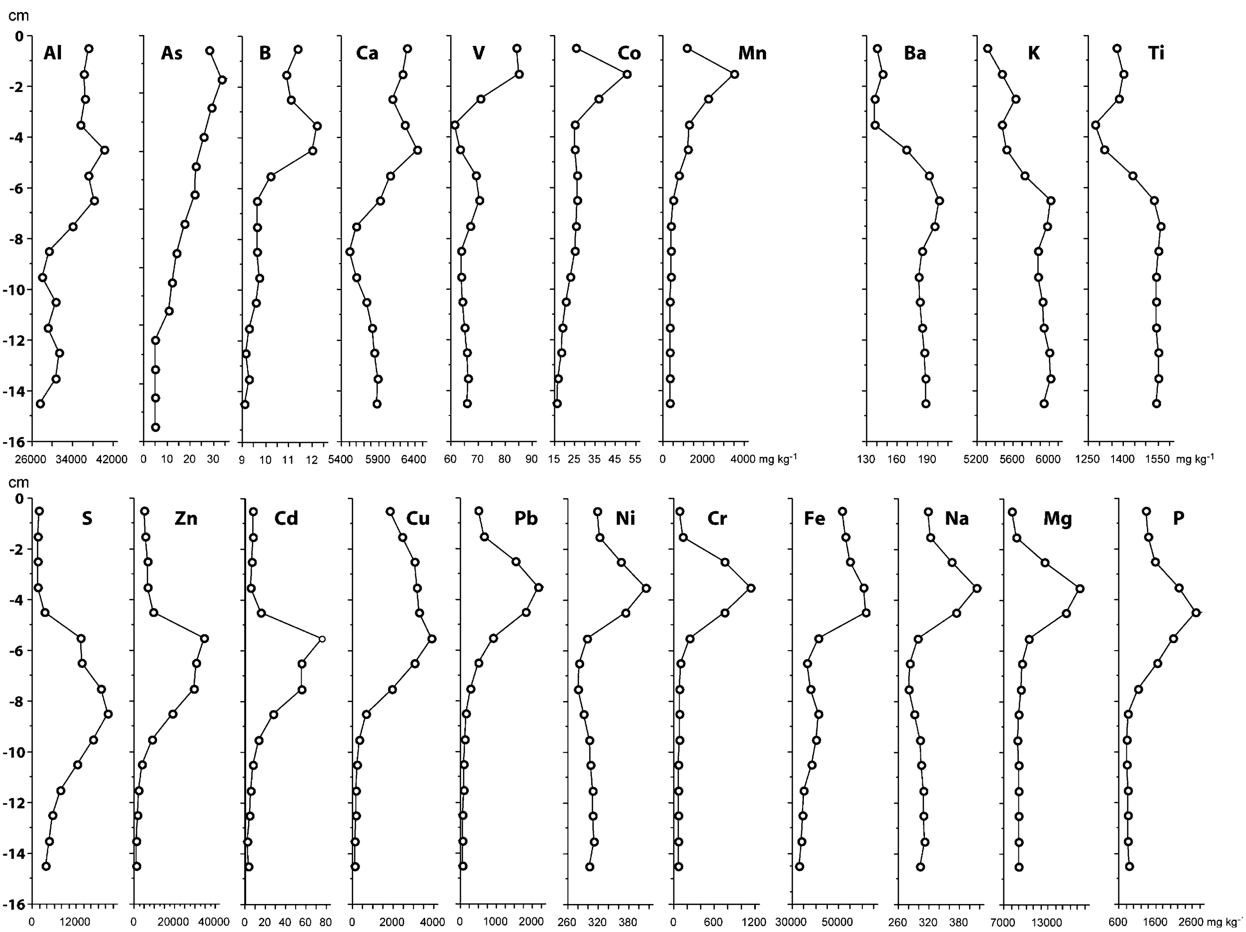

Figure 4. Element concentration values for the uppermost 15 cm of the studied core from lake Orijärvi.

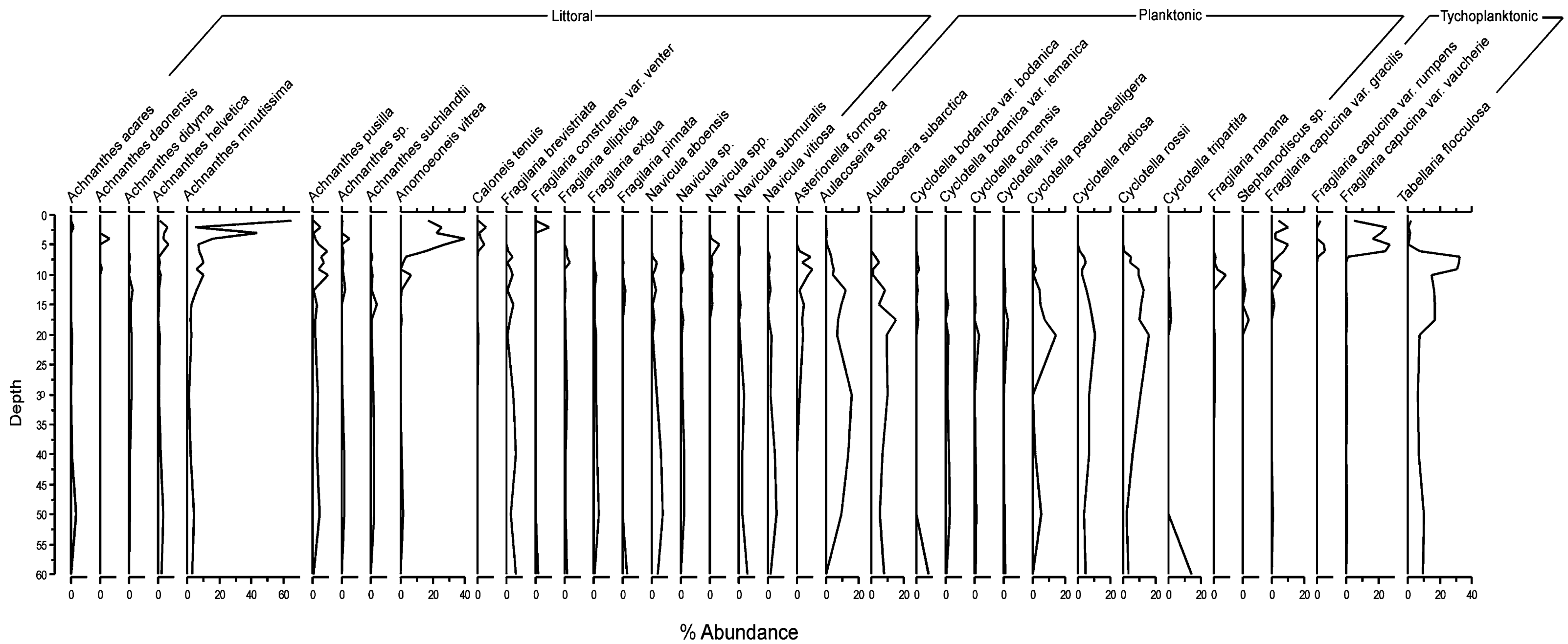

Figure 5. Main diatom taxa in the analyzed portion of Lake Orijärvi core ORI-1. Only taxa accounting for more than 2% of the count in at least one sample are presented. Diatoms are grouped according to their habitats.

planktonic *Aulacoseira subarctica* with the maximum at the depth of 17.5 cm. In the lower part of the core ORI-1, the abundance of the tychoplanktonic *Tabellaria flocculosa* is fairly constant, although it starts to increase from the depth of 20 cm upwards. Also, the planktonic diatom *Asterionella formosa* is fairly abundant in the lower part of the studied core. The littoral flora below the depth of 10 cm is diverse and the dominant genera are *Achnanthes*, *Fragilaria* and *Navicula*.

The first marked changes in diatom assemblage occur between the 10 and 5 cm depths. The planktonic diatoms belonging to the genera *Aulacoseira* and *Cyclotella* disappear. At the same time the abundance of *Tabellaria flocculosa* and *Asterionella formosa* first increase reaching their maxima at depths 7 cm and 9 cm respectively, and then drastically decrease close to zero at a depth of 6 cm. At depth 7 cm, the species of the *Fragilaria capucina*-group and *Anomoeoneis vitrea* increase their abundances markedly.

In the top 5 cm of the core ORI-1, the abundance of *Achnanthes minutissima* increases explosively, and the abundances of *Anomoeoneis vitrea* and the species of *Fragilaria capucina*-group decrease. No planktonic diatoms are present and the diversity of the littoral flora is very low. Some of the valves of *Achnanthes minutissima* were observed to be deformed. Deformation was of the same type as reported by Cattaneo et al. (2004).

The maximum abundance of planktonic species is at 20 cm. Above that level the planktonic diatoms decrease their proportion until they disappear from the sediment at depth 5–6 cm (Figure 6). The littoral and tychoplanktic species record compensatory increases at the same time. The abundance of the acidophilic diatoms decreases at the uppermost part of the core ORI-1. In contrast, the share of circumneutral and alkaliphilic diatoms increases upwards in the core. The maximum of the alkaliphilic diatoms and the circumneutral diatoms is at the depth of 7 cm and 1 cm, respectively.

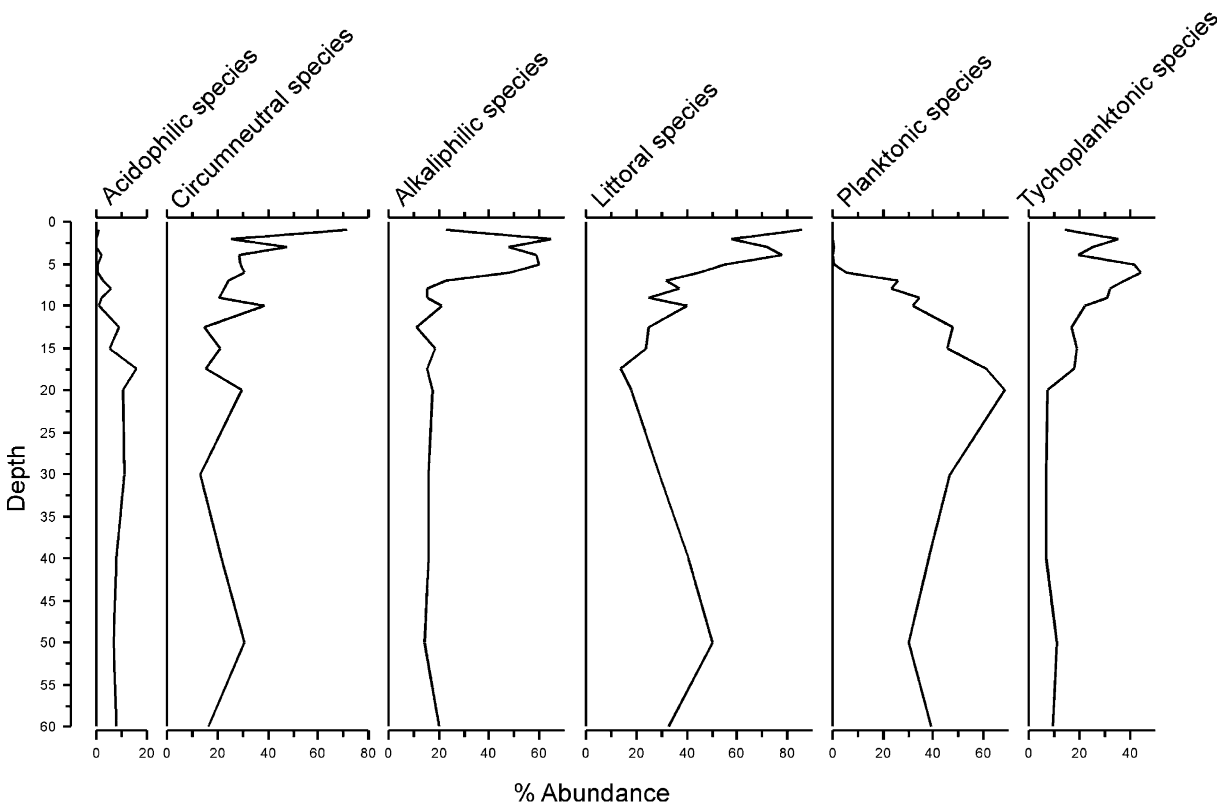

Figure 6. Diatom taxa from Lake Orijärvi sediments grouped according to their ecological requirements.

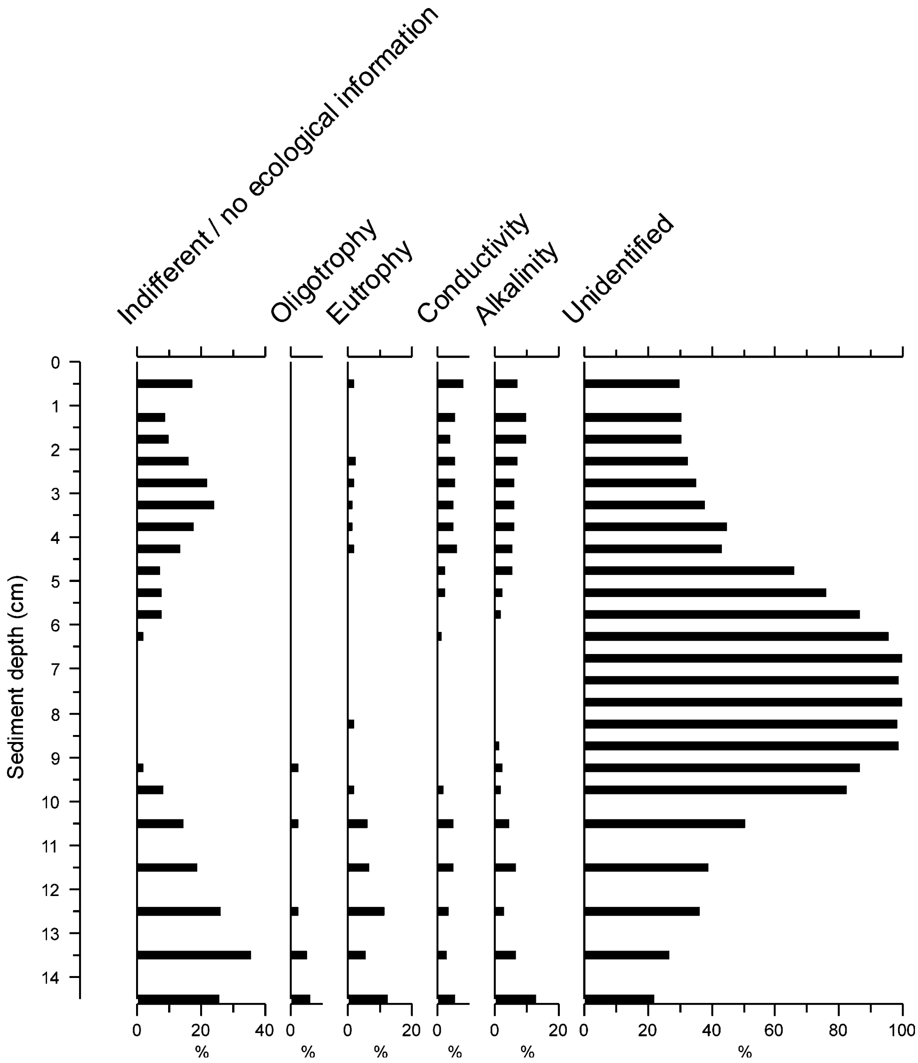

Figure 7. Chrysophyte cysts from Lake Orijärvi core ORI-1 grouped according to their ecological preferences.

Chrysophycean cysts

A total of 37 different stomatocyst morphotypes were identified (Duff et al. 1995; Wilkinson et al. 2001). Of these, 28 morphotypes each represented more than 2% of the total cysts in at least one sample. The most common stomatocyst morphotypes are type S-1, S-164 and S-120, described in Duff et al. (1995), which are common through the whole sediment sequence apart from the short sequence (depths 9–6 cm). In this sequence every cyst is either corroded/damaged or covered with very fine grained metallogenic matter (<1 μm), which could not be removed during SEM-sample preparation without losing cysts as well.

Chrysophyte cysts have been grouped according to their ecological preferences in Figure 7. Morphotypes indicating oligotrophy (Duff et al. 1995; Wilkinson et al. 2001) are common in the lowermost parts of the sediment, but their proportion decreases with decreasing sediment depth, and from 9 cm upward they totally disappear. Indicators of eutrophy (Duff et al. 1995; Wilkinson et al. 2001) are a common group in the bottom parts of the sequence as well, and their proportion decreases upwards. However, they represent the only group which could be identified at each analyzed level. Morphotypes indicating higher conductivity and alkalinity (Duff et al. 1995; Wilkinson et al. 2001) show similar trends. They are at a high level at the basal part, decrease to zero at 8 cm and increase again towards the surface (Figure 7).

A special feature at Lake Orijärvi is the remarkably high number of corroded/damaged or covered cysts, which leads to an exceptionally high percentage of unidentified cysts (Figure 7). Even in sample 14–15 cm, where the number of corroded/damaged or covered cysts is smallest, 21.8% of stomatocysts were impossible to identify. In sample 7.5 – 8 cm, only 0.6% of detectable cysts could be identified. The proportion of corroded/damaged or covered cysts is largest between 9 and 6 cm. Therefore the information obtained from stomatocyst analysis must be interpreted with extra caution. Figure 8 displays examples of unidentifiable cysts.

Discussion

History of metal load

Mining started in the Orijärvi area in the 1750s. However, the early mining probably did not have a major impact on Lake Orijärvi as can be interpreted from the results of the diatom analysis (Figure 5) and from the fact that sediment levels below 12 cm do not contain elevated concentrations of heavy metals (Salonen and Tuovinen 2001). Modern effective mining techniques, where flotation was applied and in consequence large quantities of tailings were produced, started in Orijärvi in the 1910s. From this point on, the metal load to the lake has been strong and continuous (Figure 9).

The maximum sedimentary concentration of the metals related to mining are exceptionally high. The maximum value for zinc exceeds 3% dry matter (DM), for lead 0.2% DM, for copper 0.3% DM and for sulphur 2% DM, respectively. These are more than two orders of magnitude higher than the average background levels for unpolluted lake sediments in Finland (Lahermo et al. 1996). However, comparable high metal concentrations have been analyzed elsewhere as well, e.g., from Lake Finsjö receiving the metal load from the old Anskog copper works where the Orijärvi ore was processed in the 1800s (Åström and Nylund 2000), or from the sediments of lake Süßer See having been impacted for continuous and intensive metal release since medieval times (Becker et al. 2001). Similarily, Zhang et al. (2004) describe a case from China where lead and zinc concentration in stream sediments exceeds 1% DM for kilometres away from the mining activities.

The sampled basin is a suitable sink for released metals, and the metal record displays a succession (Figure 9) which probably relates to different solubility and adsorption properties of metal cations. Sulphur accumulation begins first, because it is most readily released by oxidation of tailings, and, being a conservative element, migrates readily in the water phase (cf. Zhang et al. 2004). From the Orijärvi record we can confirm that the first attempts to produce zinc–copper concentrate by flotation did not succeed (Turunen 1957), but they created the first AMD-impact on the lake already at the beginning of the 20th century.

Deposition of copper and zinc (and cadmium) is closely associated with the last extensive mining phase in the 1940s and 1950s. Those metals were released by the oxidation process, and released cations were transported with acid waters until they adsorbed to organic matter or precipitated as sulphates in the sediment. Copper is known to adsorb readily with organic substances (Adriano 1986), and zinc probably precipitates as amorphous sphalerite (ZnS) in alkaline lake water (Becker et al. 2001). Diatoms and chrysophycean cyst records indicate (Figures 6 and 7) that Lake Orijärvi water has never been impacted by acidification, but it has remained circumneutral even during the strongest metal release phase. It is possible that the grey fine-grained coating colouring the sediment at 8–6 cm consists mainly of zinc (and copper) sulphides.

The solubility of lead is less than that of copper and zinc (Adriano 1986). This probably explains why lead peaks later, about 10 years after the mining activities ended (Figure 9). It can further be stated that metals from the tailings area are still affecting the lake, 50 years after the closure of the mine. The cadmium concentration in the water is still increasing (Figure 2), and zinc remains at a high level. In other words, there is evidence of a serious metal loading history which has lasted for nearly 100 years. The estimated

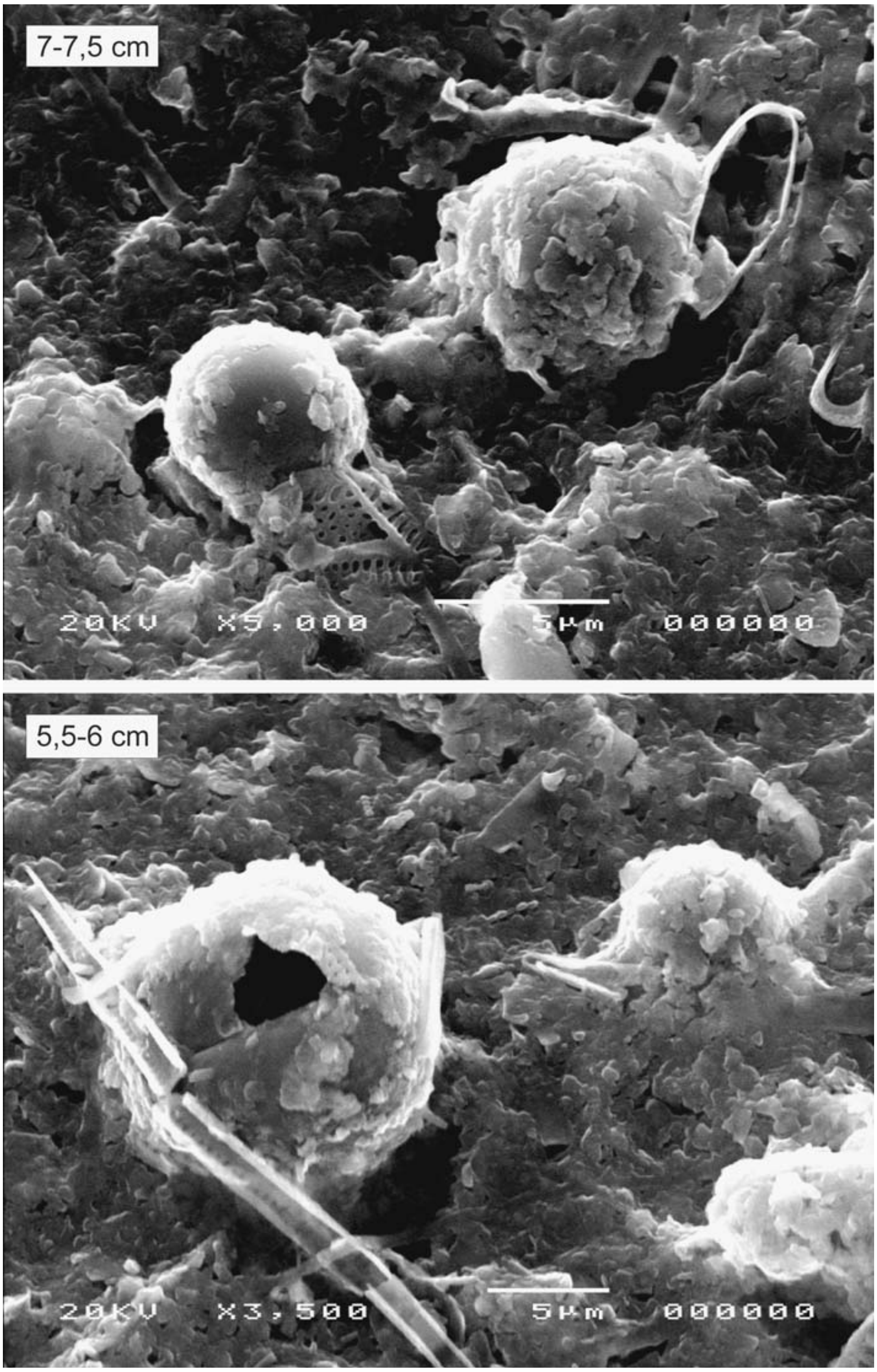

Figure 8. Examples of broken and coated cysts from sediment depths 7–7.5 cm to 5.5–6 cm.

copper load has been on the order of 800 tons, which is comparable to the load that impacted Lake Orta (Ruggiu et al. 1998). In addition, Lake Orijärvi was impacted by zinc, lead and cadmium. These activities have severely affected the lake ecosystem.

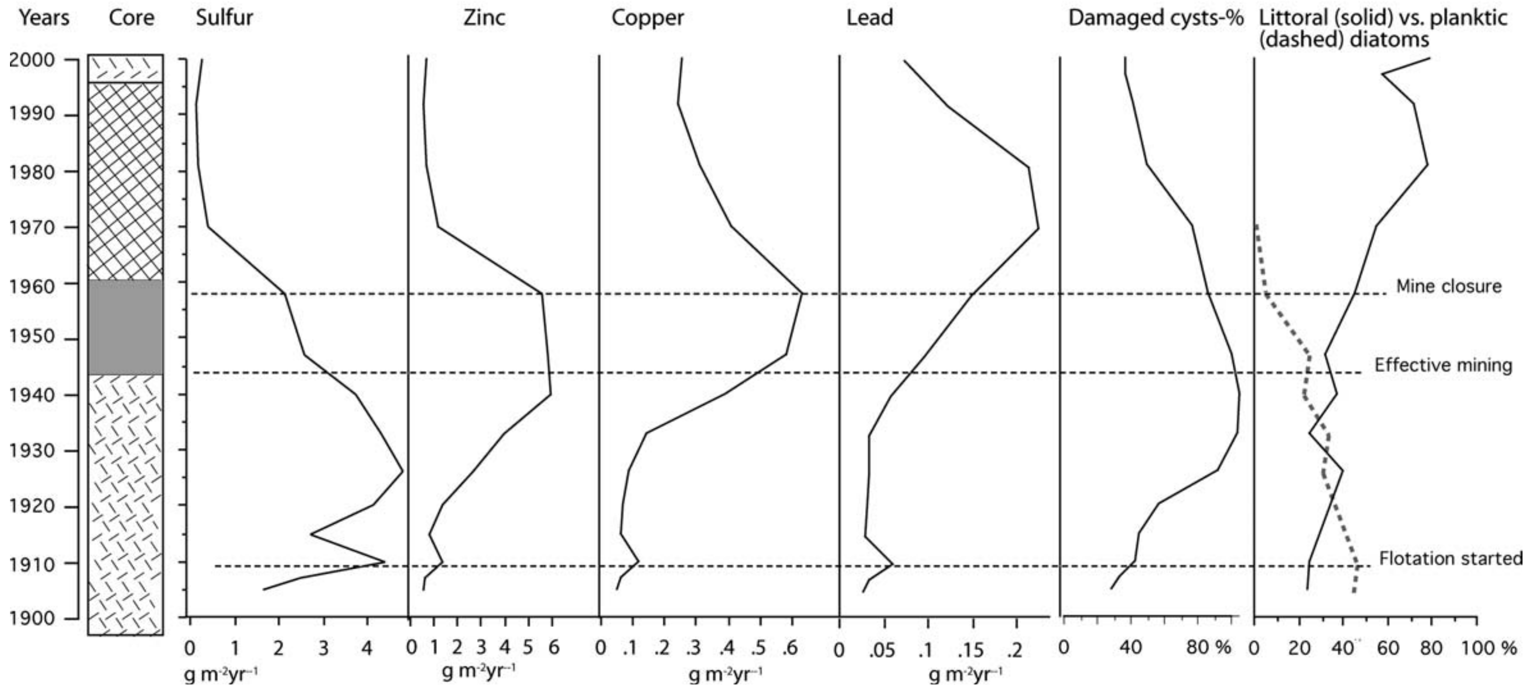

Figure 9. Accumulation of selected elements and major changes in diatom taxa and chrysophycean cysts in Lake Orijärvi sediments.

Impact on diatoms

The main changes in the diatom flora are contemporary with the changes in metal concentrations in the sediment (Figures 4 and 6). First, the abundance of the planktonic species decreased at the depth that the metal concentrations increased. Secondly, the proportion of tychoplanktic diatoms, especially the abundance of *Fragilaria capucina*-group species, increased significantly. The third change is that the abundance of the littoral species *Achnanthes minutissima* and *Anomoeoneis vitrea* increase with the increasing metal concentration. The changes resemble those caused by the copper load to Lake Orta (Ruggiu et al. 1998), and we can confirm from Lake Orijärvi data that *A. minutissima* tolerates waters with high copper and possibly also other metal concentrations. Although *A. minutissima* may occasionally occur in the pelagic region, as in case in Lake Orta (Ruggiu et al. 1998), *A. minutissima* in this paper is considered as a littoral and circumneutral species (Lowe 1974; van Dam et al. 1994).

At the topmost sediment, only littoral species are present and their biodiversity is very low. These changes in the diatom flora are clearly the result of the tailings-derived metal impact (cf. Takamura et al. 1989). Similar changes in the diatom flora may occur when pH decreases (Battarbee et al. 1999), but there are no signs of decreasing pH in Lake Orijärvi (Figure 2). On the contrary, in the light of the diatom data, the pH has increased due to the metal load. This relates probably to the fact that the ore is associated with calcium–silicate rocks (Latvalahti 1979) having high neutralizing capacity. In the topmost 10 cm, the abundances of alkaliphilic and circumneutral species are higher and the abundance of acidophilic species is lower than in the deeper parts of the sediment. This is in accordance with the signals from the chrysophycean cyst profiles.

Impacts on chrysophytes

Due to the large number of damaged and covered cysts (Figures 7 and 8) it was impossible to obtain any statistically significant ecological information from cyst analysis. Nevertheless, the impact of metal loading on cyst assemblages can be depicted. Between sediment depths 9 and 6 cm, the great majority ($>90\%$) of cysts are unidentifiable because of corrosion or heavy coating.

The significance of the metal loading as a main reason of the cyst assemblages' changes becomes even clearer when dealing with recognizable cyst assemblages. Percent interrelationships between cysts with different ecological signals (i.e., alkalinity, conductivity) return after the mine closure at the same level that preceded the strong metal loading event. The proportion of cysts indicating different trophic states changed simultaneous with metal loading. Cysts preferring oligotrophic conditions vanish after the metal loading started. On the other hand, this does not necessarily mean lake eutrophication, since the proportion of some cysts preferring eutrophic conditions also decreases during the metal loading.

The metal impact has severely affected the biota of Lake Orijärvi. Pelagic diatoms are no longer present in the topmost 8 cm of the studied section, and only diatom remains of littoral and epiphytic species could be detected. Total algal biomass is low; during the 2000 growth season it varied between 0.6 and 1.6 mg l^{-1}, being mostly composed of Chrysophyceae and Chlorophyceae algae.

Conclusions

The mining activity has impacted Lake Orijärvi for the last hundred years with a severe load of copper, lead, zinc and cadmium, without marked changes in pH. The loading history has been linked to the use of flotation techniques and associated production of tailings which were left untreated for weathering in the open air since the beginning of the 1900s. The AMD-derived metal impact on the lake is probably the strongest thus far recorded in Finland. During the most severe loading in the 1950s and 1960s, there were practically no algae in the lake. Many diatom taxa, especially planktic diatoms, vanished. *Achnanthes minutissima* has been the most tolerant to metal loading. The metal load has changed the properties of sediments in a way that chrysophycean cysts were impossible to identify with SEM because of metal coating and severe corrosion.

The water quality record shows elevated concentrations of copper (20–50 $\mu g\ l^{-1}$), lead (1–2 $\mu g\ l^{-1}$), and especially zinc (600–1200 $\mu g\ l^{-1}$) indicating that the metal impact from the tailings area is still affecting the lake biota. It has low productivity, and the planktic diatom community is still missing.

This study demonstrates that unremediated mining areas form a major risk to the environment, even long after a mine is closed. Although there is a need for further studies to determine the impacts on the entire lake ecosystem, even these results show that damage to the aquatic ecosystem can be severe and last for decades after the mining activities have ceased.

Acknowledgements

We thank Hannu Wenho for assistance in sampling and the Salon Seudun Kalastusalue for financial support to NT. The paper was greatly benefitted from the constructive comments of Dr Aldo Marchetto, Professor John P. Smol and an anonymous referee.

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