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Varve record reveals rapid development of hypolimnetic anoxia in a Northern European lake resulting from urban activities

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ABSTRACT

Human activities can significantly impact the environmental conditions of water bodies adjacent to urban areas. Maljalahti Bay, situated near the city of Kuopio at the northern end of Lake Kallavesi, has experienced numerous substantial, well-documented urban changes over the past centuries. We investigated the lake response to the influence of anthropogenic activities at Maljalahti through analysis of diatom assemblages and physical and geochemical properties of the Maljalahti sediment record. The sediment core was dated using varve counts and the vertical distribution of ^{137}Cs and ^{210}Pb. The 100-cm-long sediment record contains 95 varves, and an 8-cm-thick clastic sequence at 100–92 cm sediment depth, likely corresponding to the partial filling of the bay that started in 1913. This event, along with the initiation of sewage discharge into the bay in 1907, likely contributed to eutrophication and declining water-column oxygen conditions and the deposition of sediment with weak varve structure since the 1920s. More prominent varves appear in the sediment record since the 1980s, after the construction of a breakwater in 1978. The good preservation of these varves suggests either rapid sedimentation or the occurrence of oxygen-poor conditions. The diatom assemblage data imply that gradual eutrophication commenced when wastewater was directed to the Bay, and the construction of the wastewater treatment plant in 1974 resulted in a lower nutrient load on the Bay. Maljalahti Bay system experiences seasonal anoxia after 20 years of sewage loading, emphasizing the need for effective water exchange despite preventive measures failing to improve water quality.

1. Introduction

Inland water bodies perform many significant functions from hydrological, ecological, and socio-economic perspectives (Corvalan et al., 2005; Marsalek, 2014; Oertli and Parris, 2019; Lanka et al., 2024; Wang et al., 2024). Urban freshwater lakes play a crucial role in balancing the hydrological cycle by retaining and regulating water flows (Marsalek, 2014). They are important from an ecological perspective as they provide habitat to a wide range of biological entities (Brönmark and Hansson, 2002). In addition to being a source of clean water, the urban lake system has socio-economic significance also in supporting the recreation and well-being of city dwellers (Corvalan et al., 2005; Ventelä et al. 2005).

Many lakes around the globe experience so-called ‘cultural eutrophication’ due to the enhanced supply of nutrients from agriculture, construction, industry, and sewage (Peglar, 1993; Smol, 2009; Jenny et al., 2016a; Jenny et al., 2016b; Salminen et al., 2021; Poraj-Górska et al., 2021). The influence of human activities on the European lake systems can often be traced back in time for hundreds, or even thousands of years (Jenny et al., 2016a; Jenny et al., 2016b; Dubois et al., 2018). The severe eutrophication resulting from nutrient leaching across the European lake systems began already before CE 1900 (Jenny et al., 2016a; Jenny et al., 2016b). However, since the great acceleration, the rate of urbanization has expedited in Europe, causing increased pressure on the freshwater ecosystems (Jenny et al., 2016b; Dubois et al., 2018; Wang et al., 2024). Enhanced aquatic primary production and burial of organic matter increase oxygen consumption through degradation of organic matter and can lead to oxygen loss in hypolimnetic waters, i.e. hypoxia (threshold $\leq$ 2 mg/l) or even anoxic conditions (Müller et al., 2012; Friedrich et al., 2014). Consequently, the lake ecosystem experiences degradation of local biodiversity and impoverished water quality (Brönmark and Hansson, 2002; Lanka et al., 2024).

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Forest clearance, agriculture, and building urban infrastructure cause enhanced catchment erosion and transport of detrital material to the water bodies (Saarni et al., 2017; Johansson et al., 2019; Lanka et al., 2024). Consequent sediment loading can alter the sediment composition by diluting organic matter and heavy metal concentrations, or causing enhanced terrigenous organic matter supply and potentially affect redox conditions and nutrient cycling (Forsberg, 1989). It can also reduce light penetration and disrupt benthic habitats (Ask et al., 2009). Rapid urbanization also accelerates the enhanced accumulation of a wide range of contaminants into lakes mainly through surface run-off or atmospheric fallout (Rahman et al., 2021). Pollutants, including heavy metals, as well as nutrient and detrital sediment loading pose threats to the biodiversity of the aquatic environment because many aquatic species are highly sensitive to physiochemical changes (Dubois et al., 2018; Gros et al., 2023). Species diversity and abundance of phytoplankton, zooplankton, and benthic organisms can be severely affected, resulting in disturbance of the food web (Wang et al., 2024). The degraded status of lakes can cause many human health issues, resulting from declined possibilities of recreational use or challenges in the supply of clean and sufficient domestic water (Schwarzenbach et al., 2010; Lin et al., 2022).

Given the acceleration of global population growth, urbanization is likely to be an irreversible trend in the 21st century; around two-thirds of the world's citizens are expected to reside in city areas by 2050 (United Nations, 2015). Urban sprawl often leads to various development and construction projects, which may require partial filling of cities' existing water bodies. A better understanding of the impact of urban constructions on inland water quality on a range of temporal scales will help estimating the impact of urban activities on lake systems and address United Nations sustainable development goals, e.g. ensuring clean water and sustainable cities (United Nations, 2015).

Lake sediments are important archives to investigate how biological, chemical and physical components of lake systems respond to pressure from anthropogenic changes (Zolitschka et al., 2015). Annually laminated (varved) sediments are suitable for detecting these fluctuations on a seasonal resolution (Ojala et al., 2013; Zolitschka et al., 2015).

Maljalahti Bay, located next to the city of Kuopio in eastern Finland, has undergone many significant development activities over the last centuries (Sormunen and Kostiainen, 1960; Nummela, 1989; Marin, 2007; Juuti and Katko, 2014). Because of its location close to the city and documented construction history, the bay is an ideal site to study the timing and influence of anthropogenic activities and urban development on sedimentation, sediment characteristics, and water quality within the lake system. We hypothesize that urban activities had substantial effects on the water quality and sedimentation patterns in Maljalahti Bay. To test this hypothesis, we aim to address the following research questions: How have human activities (e.g., dredging, filling, sewage discharge) impacted the water quality of the bay, and for how long have these events influenced water conditions, sediment composition and the sedimentary environment of the bay? In this study, we investigate a varved sediment sequence from the Maljalahti Bay, its sediment composition, geochemical properties and sedimentary diatom species assemblages to answer our research questions.

2. Study area

Maljalahti Bay is a scenic bay located within the city of Kuopio, Finland (Fig. 1). The city features a hilly landscape, ranging from 90 to 210 m above sea level, characterized by a mix of forested land, grasslands, urban impervious surfaces and waterbodies (Asikainen et al., 2017; Gopalakrishnan et al., 2020). The catchment area of Maljalahti Bay is 627.69 ha in Kuopio City; 63 % land of the catchment is urban, while 33.7 % is forested. The city has a population of approximately 120,000, with 87 % residing in the urban area near Maljalahti Bay, which contains 62 % of the city's residential buildings (Asikainen et al.,

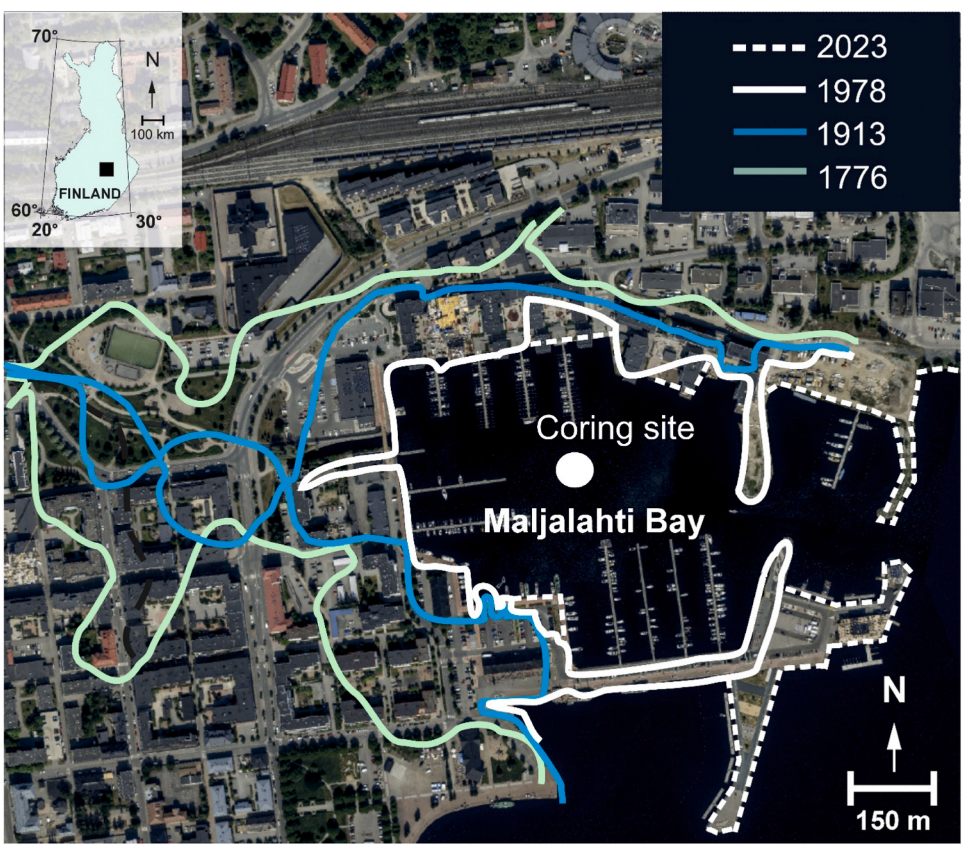

Fig. 1. Maljalahti Bay area in 2023 and shorelines in 1978, 1913, and 1776 (Kuopio city maps, Kuopio city plan 1988). These years represent known shoreline changes. The present (year 2023) shoreline represents the shoreline after breakwater extension and dredging in 2018–2019. The coring location is also provided.

2017).

Maljalahti Bay is connected to a relatively large main body of water, freshwater Lake Kallavesi, which has an area of 48,000 ha, and the bay is restricted from the main body by breakwaters. The maximum depth of the bay is 10 m, and its surface area is 11.36 ha (NLS, 2025). The bay has one inlet, the Maljapuro River, which brings water from Kuopio City. The 30-year average annual temperature and precipitation in the Kuopio district for the period 1971–2000 are +3.1 ℃ and 607.72 mm, respectively (FMI, 2025). Average temperatures of the coldest month, January and the warmest month, July, are approximately −9.4 ℃ and +17.1 ℃, respectively (FMI, 2025). About 40–50 % of the annual precipitation falls as snow, with an average maximum snow depth of 59 cm for the period 1990–2019 (Kersalo and Pirinen, 2009; SYKE, 2024). The bay is typically ice-covered for around 4–5 months each year (Soininen et al., 2024).

Positioned at the northern end of Lake Kallavesi, Kuopio has been a busy harbor since the 18th century. The Maljalahti Bay area has been under heavy construction throughout the centuries. Half of the bay's area has been filled with artificial fill (Table 1). At the end of the 19th century, the shores of the bay were considered "not healthy" (Kahra, 1956) and were filled in, starting in 1913. Part of the material used was the clay dredged from the boat harbor nearby (Nummela, 1989). Building the City's sewage system started in 1907 (Marin, 2007; Juuti and Katko, 2014), and several drainage pipes were directed to the bay area. By the end of the 1950s, the water in the bay area was heavily polluted by untreated sewage water (Sormunen and Kostiainen, 1960). The water quality started to improve after building a sewage treatment plant in 1974 (Marin, 2007). Currently, the bottom water experiences anoxic conditions predominantly during the summer months (July, August), and in some instances, this phenomenon also occurs during the winter months (February, March, April) (SYKE, 2024).

3. Methods

3.1. Coring

A 1.5-m-long sediment core was retrieved from the deepest point (10 m) of Maljalahti Bay in a PVC core tube with an inner diameter of 64 mm. A rod-operated piston corer was used to collect the sediment core through winter ice in March 2022. The core tube was opened lengthwise using a circular saw and a wire at the laboratory and the sediment surface was cleaned with a glass blade horizontally against the varve structures and then covered with a plastic film.

3.2. Bulk sediment analysis

Low-field magnetic susceptibility was logged from the fresh sediment core using an automatic measuring track with a measurement interval of 2.0 mm, employing the Bartington MS2 and MS2E spot reading sensors.

Volume-specific 1 cm^3 sediment samples were collected with a plastic syringe from the full core length using 1-cm sampling interval. The subsamples were dried overnight and combusted at 550 °C for four hours, after which the water content, dry bulk density (DBD) and loss of ignition (LOI) were calculated.

Table 1
Selected historical events and construction actions in Maljalahti Bay area since the end of the 19th century (Nummela, 1989; Siekkinen, 1997; Marin, 2007; Juuti and Katko, 2014; Lappalainen, 2023), and the related details of the resulting marker layer.

Event	Timing	Marker layer
Breakwater extension and dredging	2018–2019	4 cm clay layer, 6 cm sediment depth
Dredging	2011	7 cm clay layer, 19 cm sediment depth
Breakwater construction and land reclamation in the bay area	ca. 1978	1 cm clay layer, 36 cm sediment depth
Start of sewage treatment	1974	
Start of major land reclamation in the western part of the bay area	1913	
Sewage directed to the bay	1907	
Land reclamation on the western shore of the bay	ca. 1889	

3.3. Epoxy impregnation

The sediment core was subsampled continuously using aluminum molds (11 ×1.5 ×1 cm), with 1.5 cm overlap following the procedure described by Haltiahovi et al. (2007). The sub-samples were epoxy impregnated based on the shock-freeze and freeze-dry technique (Lotter and Lemcke, 1999; Salminen et al., 2021). The samples were first shock-frozen using liquid nitrogen and then freeze-dried (24 h main drying in 0.63 mbar, −25 °C and 4 h final drying in 0.04 mbar, −50 °C) using a Christ laboratory freeze dryer. Then, the sub-samples were embedded in a two-component Araldite epoxy. The epoxy impregnation was carried out in a vacuum chamber to limit the amount of air bubbles in the sediment pores (Lotter and Lemcke, 1999). The epoxy-embedded samples were cured for 12 h at room temperature, cut with a rock saw, and polished using a surface-grinding machine to ensure a smooth surface for microscopy and micro-XRF analysis.

3.4. Dating

3.4.1. ^{137}Cs and ^{210}Pb

A series of 51 sub-samples were taken from fresh sediment at 0.5 cm, 1.5 cm, 3 cm, and at 2-cm intervals thereafter to 99 cm for ^{137}Cs activity measurements to support the varve counting. ^{137}Cs activity measurements were performed on wet sediment (1.8–36.5 g) using a BrightSpec gamma spectrometer (3 600 s counting time) at the Geological Survey of Finland.

To support the varve counting deeper in the sediment, 31 subsamples were collected from fresh sediment at 0.5 cm, 1.5 cm, 5 cm, and every subsequent 4 cm down to 117 cm for ^{210}Pb analysis. The subsamples were dried at 105 °C and crushed in a mortar. Activity of total ^{210}Pb was determined indirectly by measuring its daughter product ^{210}Po ($T1/2 = 138$ d) using alpha spectrometry. Dry and homogenized sediment samples of ca. 0.2 g were transferred into Teflon containers, spiked with ^{209}Po yield tracer ($T1/2 = 102$ yr) and digested with concentrated HNO_3, $HClO_4$ and HF at a temperature of 100 °C using a CEM Mars 6 microwave digestion system. After 24 h, the solution was transferred into a Teflon beaker, evaporated with 6 M HCl to dryness and then dissolved in 0.5 M HCl. Polonium isotopes were spontaneously deposited within four hours on silver disks. After deposition, the disks were washed with methanol and analysed for ^{210}Po and ^{209}Po. Activities were measured using a 7200–04 APEX Alpha Analyst integrated alpha-spectroscopy system (Canberra) equipped with PIPS A450–18AM detectors. Samples were counted for 24 h. A certified mixed alpha source (234 U, 238 U, 239Pu and 241Am; SRS 73833–121, Analytics, Atlanta, USA) was used to check the detector counting efficiencies, which varied from 31.8 % to 33.8 % for the applied geometry. The deposition efficiency estimated by comparing the measured and spiked ^{209}Po activities was 74 ± 9 % ($n = 31$).

3.4.2. Varve counting

Varve counting was performed under a Nikon SMZ800 stereomicroscope. The varves were counted from the surface of the epoxy-impregnated sediment blocks three times along different analytical lines to calculate counting error (Lotter and Lemcke, 1999). A couplet of a clastic lamina (CL) and a biogenic lamina (BL) was considered as one varve year. Varve thickness and lamina thicknesses were measured simultaneously.

3.5. Geochemistry

Dried and homogenized sediment samples extracted at 2 cm resolution, approximately 20–40 mg in weight, were placed into tin capsules for elemental analysis. Concentrations of total carbon (TC), total nitrogen (TN), and total sulfur (TS) were determined using a Vario El Cube elemental analyzer (Elementar), which is equipped with a thermal conductivity detector (TCD). The calibration of measurements was done against certified reference materials (Soil Standard B2176) provided by Elemental Microanalysis (UK).

Micro-XRF measurements of the epoxy-impregnated sub-samples were performed at the Geology section (UTU) with a Bruker M4 TORNADO micro-XRF spectrometer using a rhodium X-ray tube with an anode current of 500 μA, voltage of 50 kV, and a spot size of 20 μm. The surfaces of the epoxy-impregnated subsamples were scanned in 50 μm steps using a counting time of 100 ms/pixel. Bruker M4 TORNADO analysis software was used to retrieve XRF elemental maps of 13 major elements (Al, Ca, Co, Cu, Fe, Pb, Mn, P, K, S, Si, Ti, and Zn) and to extract line scan data of elemental intensities (expressed as counts per second or CPS) from the XRF elemental maps. The high-resolution data were first downscaled to remove the noise. The element intensity data were then transformed to centered log-ratios (clr) to overcome difficulties associated with the closed-sum data and to eliminate specimen effect resulting from non-linear correlation between CPS and elemental concentration (Croudace et al., 2006; Weltje and Tjallingii, 2008; Martin-Puertas et al., 2017; Bertrand et al., 2024).

Logarithmic ratios of different elements are used to extract sediment composition and depositional environment in the water column. Ti/K ratio reflects the variability of clay minerals; the justification of using this ratio lies in the association of clay minerals with K and coarse-grained heavy minerals with Ti (Marshall et al., 2011; Davies et al., 2015). On the other, the Fe/Mn ratio can serve as a proxy for hypoxia in the bottom water hand due to the difference in preservation of Fe and Mn depending on oxygen availability (Dean, 2002; Davies et al., 2015; Saarni et al., 2016b). Hence, elevated Fe/Mn ratios indicate intervals characterized by reduced oxygen availability and vice versa.

3.6. Diatoms

The core was sub-sampled for the diatom analysis at 6 cm intervals and prepared for analysis according to the method described by Battarbee (1986). Identifications were carried out with a Meiji Light microscope with phase-contrast illumination at 1000x magnification. Diatom taxonomy is based on Krammer & Lange-Bertalot (1986–1991). At least 300 valves were counted from each sample. Compositional changes in diatom assemblages were assessed with principal component analysis (PCA) using the PAST 4.03. software (Hammer and Harper, 2001). The stratigraphic diagram was made with the C2 1.8.0. software (Juggins, 2007). Compositional changes observed in the PCA ordination were used to define the diatom zones.

3.7. Water quality data

Water quality data were downloaded from open data by the Finnish Environmental Institute (SYKE, 2024). Water quality at Maljalahti Bay has not been monitored continuously, but sporadic measurements of oxygen concentration and nutrients within the water column in different seasons are available.

4. Results

4.1. Sediment characteristics

The magnetic susceptibility exhibits a wide range, with values spanning from 33.5 to 1703 $\times 10^{-6}$. The dry bulk density varies between 0.2 and 1.6 g/cm^3 , while the LOI ranges from 0.8 % to 35.9 %. The water content fluctuates within the sediment core, ranging from 11.9 % to 76.8 %.

Based on these sediment characteristics, the core was divided into three units (Fig. 2). Only top 100 cm out of 150 cm core was investigated for this study as there is no varve sequence and significant variability is noticed below this depth due to the clear, visually observed hiatus below 100 cm sediment depth. In Unit 1, between 100 and 90 cm sediment depth, distinct peaks in magnetic susceptibility are observed at depths of approximately 99 cm and 91 cm. The maximum dry bulk density occurs concomitantly with the minimum LOI and water content at the bottom of the core at around between 100 cm and 90 cm depths. In Unit 2, between 90 and 40 cm, the sediment density and magnetic susceptibility are significantly lower, while water content and LOI are elevated. Water content reaches a maximum at approximately 43 cm depth, and LOI peaks around 50 cm depth. In Unit 3, at 40–0 cm depth, the density and magnetic susceptibility increase again towards the top of the core, whereas both LOI and water content decline (Fig. 2).

The C/N ratio (minimum 4.3, maximum 34.0, mean 16.1) and total sulfur (TS) content (minimum 0 %, maximum 0.92 %, mean 0.29 %) exhibit very similar trends. Both the C/N ratio and TS (%) show distinctly lower values in Unit 1. Increasing patterns are observed in Unit 2 with a distinct TS maximum at 43 cm sediment depth, followed by a rapid increase to a peak in the C/N ratio at the onset of Unit 3. A declining trend is observed in both C/N ratio and TS within Unit 3 (Fig. 2).

4.2. Dating and core chronology

According to the varve counts, the topmost 92 cm of the Maljalahti sediment profile contains 95 $\pm$ 2 varves with an error $\pm$2.1 % (Fig. 3). The infilling of the Bay in 1978, along with the dredging of the shallows of the harbor in 2011 and 2019, resulted in the formation of clear minerogenic event layers enriched in clays. The exceptionally thick clayey event layers support our varve chronology. These events can be related to specific activities within and around the bay, which are shown to lead to increased sediment deposition (Johansson et al., 2019). These activities likely caused enhanced sediment loading and sediment resuspension.

The measured ^{137}Cs activity varies between 0 and 690 Bq/kg, with the highest activity measured at the sediment depth of 34–32 cm (Fig. 3), representing the Chernobyl nuclear accident in 1986. A lesser peak, 70 Bq/kg, is observed at the sediment depth of 54–48 cm, corresponding to the year 1963, the time of greatest fallout from atmospheric nuclear weapons testing in Novaya Zemlya in the Barents Sea. Both of these activity peaks are commonly detected in Finnish lake sediments (Salminen et al., 2019; Haltia et al., 2021; Salminen et al., 2021), and their assumed ages agree with the varve count results at Maljalahti. In addition, the downcore profiles of total and excess ^{210}Pb activities (Fig. 3) look irregular in the uppermost ca. 35 cm (mass depth ca. 20 g/cm^2), which is probably caused by very high and variable sedimentation rates. Excess ^{210}Pb disappeared at a depth of around 94 cm. The chronology obtained using the constant rate of supply (CRS) model constrained by ^{137}Cs chronostratigrahic markers fits very well with the varve counting results (Fig. 3).

4.3. Varve characteristics

The Maljalahti sediment core consists of two markedly different sections: the non-varved (entirely remains in Unit 1) and the varved section. The color of the non-varved sediment varies from dark brown to dark gray and greyish brown. Varve preservation begins gradually just before the Unit 2 boundary, at a sediment depth of 92 cm (Fig. 2). The varved section is further divided into poorly varved (mostly found in Unit 2) and varved (dominantly present in Unit 3) sections. The poorly varved sediments are primarily located between 92 cm and 40 cm sediment depth, corresponding to the time from the late 1920s to the

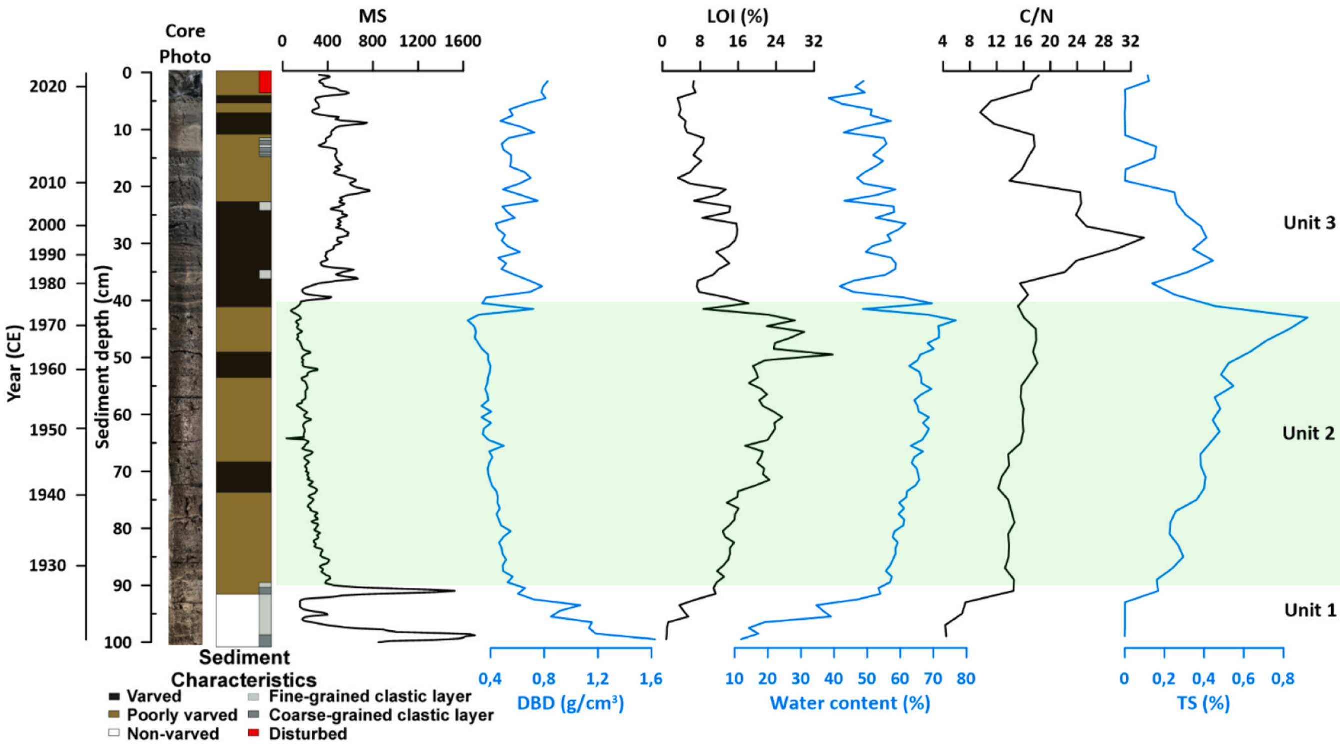

Fig. 2. Photograph of the freshly opened core, sediment characteristics, magnetic susceptibility (MS), dry bulk density (DBD), percentage of loss on ignition (LOI), percentage of water content, C/N ratio, and percentage of total sulfur (TS) of the Maljalahti sediment core. The three units (Unit 1, Unit 2, and Unit 3) are delineated using different colors.

late 1970s (Fig. 2). The faintly laminated varves can only be detected under a microscope. Distinctly laminated varved sediments are sporadically found below 40 cm sediment depth and have predominantly developed since the late 1970s (Fig. 2). These varves are relatively easy to detect, and some of them are even identifiable with the bare eye.

The varves of Maljalahti are of clastic-biogenic varve type, which is typical for boreal regions (Ojala et al., 2013; Zolitschka et al., 2015; Saarni et al., 2017). The varve year begins with a clastic lamina (CL) followed by a biogenic lamina (BL). The color of the BL is darker compared to the brighter greyish color of the CL. Some varves have a sharp contact between CL and BL but in general the transition from CL to BL is gradual.

The mean (and median) thicknesses of CL and BL are 6.33 mm (5.16 mm) and 2.33 mm (2.16 mm), respectively, and the total varve thickness (TVT) ranges from 1.8 mm to 31.2 mm (mean thickness: 8.7 mm, median thickness: 7.8 mm). The years of high sedimentation (2011 and 2019) are excluded from the descriptive statistics as extreme sedimentation in those years is mainly caused by specific anthropogenic activities (Fig. 4). Nevertheless, the lamina thickness characteristics of the varved section exhibit a change at the transition between Unit 2 and Unit 3 at approximately 40 cm depth (Fig. 4). Both the TVT and the CL thickness decrease upwards in Unit 2 and increase in Unit 3. In contrast, the BL thickness generally increases upwards throughout the varved section, except for a short decrease in BL thickness at around 40 cm sediment depth (Fig. 4).

4.4. Geochemistry

The elements that are enriched within aluminosilicate minerals, for example, Al, K, Si, and Ti (Davies et al., 2015), demonstrate elevated counts in CL and reduced counts in BL. In most cases, Fe and Mn follow a similar pattern to the aluminosilicate minerals in the elemental maps. Conversely, P, S, and Zn exhibit higher counts in BL while displaying decreased counts in CL (Fig. 5).

A significant change in elemental composition is detected between 1930 and 1980 (Fig. 6), where the total sum of the element counts is considerably lower than in Units 1 and 3. The elements associated with aluminosilicate minerals exhibit a pattern similar to the total sum of the elements. However, those elements have enhanced counts in very recent years, whereas the total sum has a short downfall during the same time. Units 1 and 3 also show enrichment in clay minerals, as indicated by the low Ti/K ratio.

Redox-sensitive elements such as Fe and Mn display variability comparable to that of aluminosilicates (Fig. 6). Notably, the ratio of Fe to Mn exhibits elevated values during the 1960s, 1970s, and 1980s, followed by a decreasing trend that began around 2000. In contrast, S demonstrates a unique pattern that sets it apart from the other redox-sensitive elements. An increase in S counts is evident prior to 1930, after which enhanced counts persist until approximately 2010, with the exception of brief declines observed during the 1990s. Following this period, S shows a decreasing trend from 2010 to 2016, after which an upward trajectory is again noted (Fig. 6).

P, which is also influenced by redox conditions, exhibits a peak at around 1935 (Fig. 6). Following this peak, no significant variability is present until approximately 1982, during which time small peaks in P counts are noticed between 1982 and 1990. Subsequently, a significant decline occurs from 1990 to 2000, with high variability but no directional change observed in P counts thereafter (Fig. 6).

The heavy metals, for example, Zn and Pb, demonstrate an inverse relationship with the aluminosilicate mineral-related elements. These heavy metals have relatively enhanced counts between 1930 and 1980, and declines from the 1980s towards the present. However, Pb shows an increasing trend in the most recent years since 2010. Other heavy metals, e.g., Co and Cu do not exhibit significant variability in the core (Fig. 6).

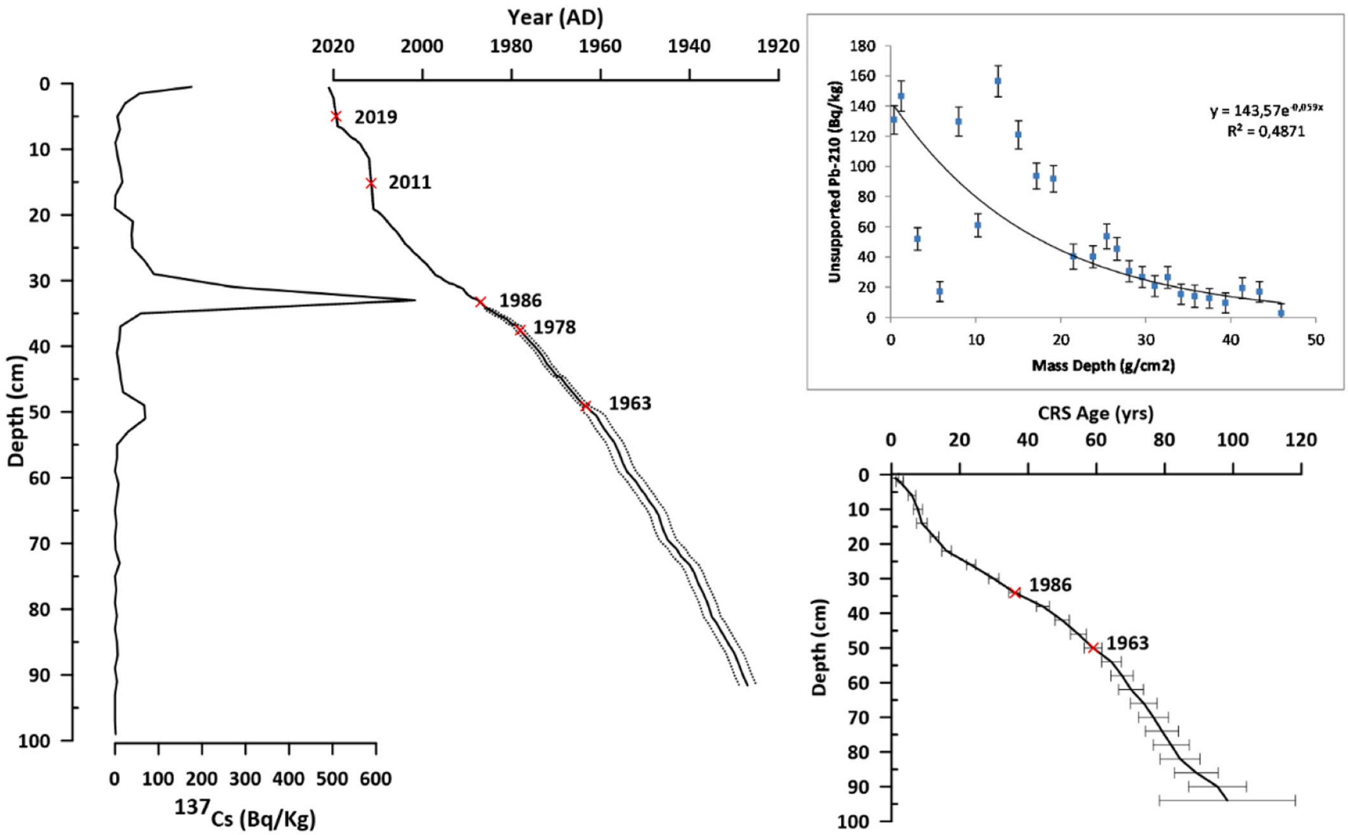

Fig. 3. Left panel: ^{137}Cs profile and a varve-count-based age-depth model for the Maljalahti core. The thin dotted lines in the age-depth model represent the error estimates, and the red crosses (X) show time markers used to establish the chronology (Fig. 4). 1963 and 1986 time markers are derived from ^{137}Cs dating, and 1978, 2011 and 2019 markers are based on varve counting and distinct clay layers in the sediment core. Upper right panel: distribution of unsupported (excess) ^{210}Pb activities versus mass depth with a best-fit curve. Lower right panel: age-depth model based on ^{210}Pb dating results; the error margin ranges between ±1 year and ±20 years.

4.5. Diatoms

A total of 159 taxa were enumerated belonging to 55 genera. Species with abundances higher than 3 % are shown (Fig. 7). The most abundant taxa are planktonic. The diatoms were divided into three local diatom assemblages (LDAZ; Fig. 7), that differ slightly from the previously described Units 1–3.

The lowermost part of the core (LDAZ 1) is consistent with Unit 1, and is dominated by *Aulacoseira islandica* (O.Müller) Simonsen with up to 94 % at the level of 99 cm (Fig. 7).

LDAZ 2 (90–48 cm) is dominated by *Aulacoseira subarctica* (O. Müller) E.Y.Haw. There is a peak of *Fragilaria crotonensis* Kitton and *Fragilaria rumpens* (Kützing) G.W.F.Carlson respectively in the upper part of LDAZ2. Asterionella formosa Hassle is quite abundant in LDAZ2. There is also an increase in small centric diatoms like *Cyclotella atomus* Hustedt and *Discotella pseudostelligera* (Hustedt) Houk et Klee in LDAZ2.

LDAZ 3 (48–0 cm) begins at sediment depth of 48 cm, 8 cm below Unit 3. The most abundant species are *A. subarctica*, *A.islandica, A. formosa*, *F. crotonensis*, *Tabellaria flocculosa* (Roth) Kützing and *Achnanthidium minutissimum* (Kützing) Czarnecki. *F. crotonensis*, *T.flocculosa*, *C. atomus* and *A. islandica* increase their share compared to the zone LADZ 2. Also, *D. pseudostelligera* is fairly abundant in this zone. *Cyclotella atomus* and *Lindavia radiosa* (Grunow) De Toni & Forti slightly increase their share in the topmost part of the zone.

4.5.1. PCA

PCA plots the lowermost LDAZ 1 to the far-right hand of the axis 1 (Fig. 7). LDAZ 2 is mainly plotted on the left side of the axis 2 and LDAZ 3 to the top of the axis 1. The PCA axis 1 score shows a clear change in the species composition compared to the other zones. Other changes are more subtle and for example the LDAZ 2/LDAZ 3 boundary is not visible in the PCA axis 1 score.

4.6. Water quality data

The sporadic measurements of oxygen concentration within the water column show that bottom waters experience hypoxic (oxygen concentration < 2 mg/l) conditions predominantly during the summer months. The extent of hypoxia reaches 5 m water depth in just one year, while in most cases, the boundary remains between 5 and 8 m water depth during summer. During recent years, hypoxia has been documented during the inverse stratification under winter ice (Fig. 8). However, the data prior to 2019 are limited to only one single observation, and therefore it is not possible to assess whether this represents a recent development.

In addition to oxygen concentration, the measurements of total nitrogen (TN) and total phosphorus (TP) range from 630 to 910 μg/l and from 21 to 34 μg/l respectively in the surface water. Both TN and TP yield higher concentrations in the bottom water; however, TP is substantially high when the bottom water becomes hypoxic (SYKE, 2024).

5. Discussion

5.1. Early filling of the Maljalahti Bay in the 1900s

The lowermost part of the core, the non-varved sediment between 100 and 92 cm, is dominantly organic matter-poor (Fig. 2) and is likely deposited during the documented major filling of the Bay at the

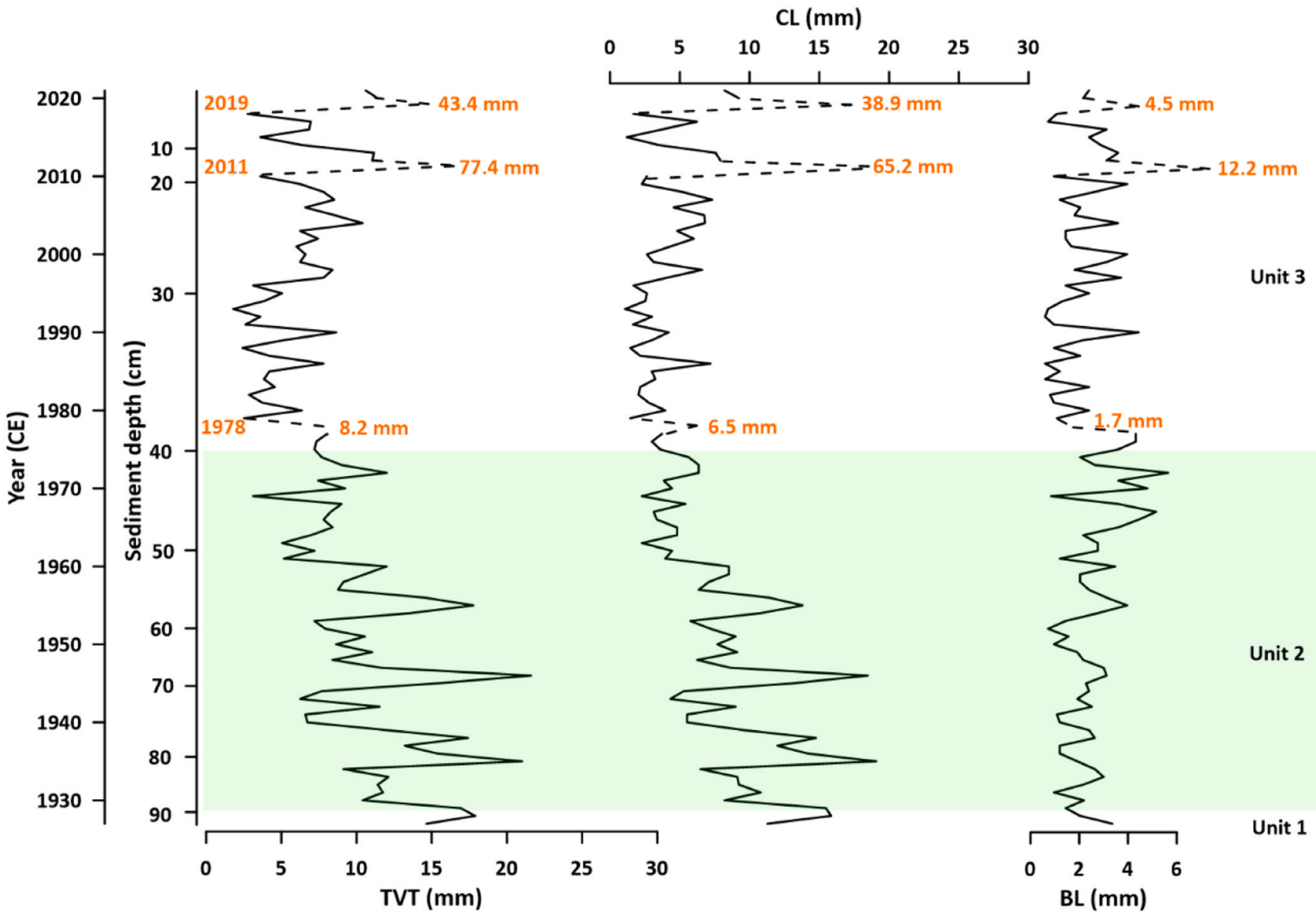

Fig. 4. Total varve thickness (TVT), clastic lamina thickness (CL), and biogenic lamina thickness (BL) of the Maljalahti sediment core; the dashed lines indicate the human-induced event layers at 1978, 2011, and 2019. The Units 1–3 are illustrated in different colors.

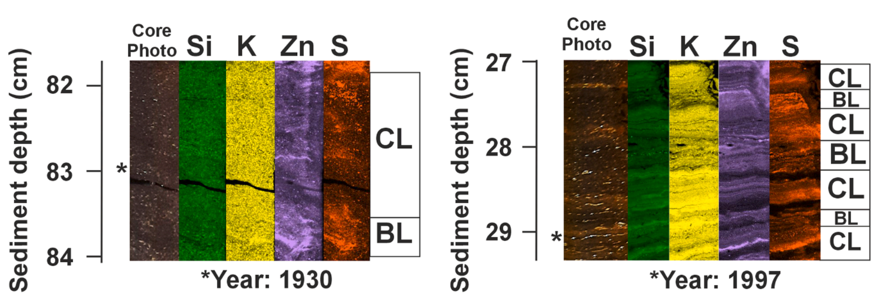

Fig. 5. Varve structures are reflected in 2D micro-XRF geochemical maps of Si, K, Zn, and S. No clear clastic laminae are formed in the poorly varved section, but varve structures are defined by enrichment of Zn and S in BL during the period of sewage water supply to the bay (poorly varved on the left). After the wastewater treatment was initiated and the lake basin was further filled, clear clastic laminae emerged with Zn and S enriched in BL (varved on the right).

beginning of the 20th century. A homogeneous detrital sediment layer with composition clearly differing from Units 2 and 3 suggests that a significant portion of the material deposited in the early 20th century results from the filling of one-third of the bay area. The most abundant diatom in this unit is *A. islandica,* which is a cosmopolitan species occurring commonly in the plankton and tychoplankton of meso- to oligotrophic lakes (Krammer et al., 1988; Krammer et al., 1991a; Krammer et al., 1991b). Here, the peak of *A. islandica*, however, coincides with a massive clayey layer and might therefore also be derived from the material that was used in the bay filling. This is supported by *A. Islandica* being a dominant species in Ancient Lake Saimaa clays (Tammelin et al., 2019) that were exploited as filling material (Rytkönen, 1975). In addition, the low Ti/K ratio points towards the enrichment of clays in this unit.

5.2. *Varve characteristics and onset of varve formation and preservation in Maljalahti Bay*

The varved section of the Maljalahti sediment record is characterized by clastic-biogenic varves, with varying degrees of preservation ranging from well to poorly preserved sequences. The mean TVT of the Maljalahti sediment record is higher than the mean TVT observed in many Finnish lakes (Ojala and Saarnisto, 1999; Saarni et al., 2016a; Salminen et al., 2023), although, similar varve thicknesses have previously been recorded from relatively large Finnish lake systems (Meriläinen et al., 2003; Salminen et al., 2021).

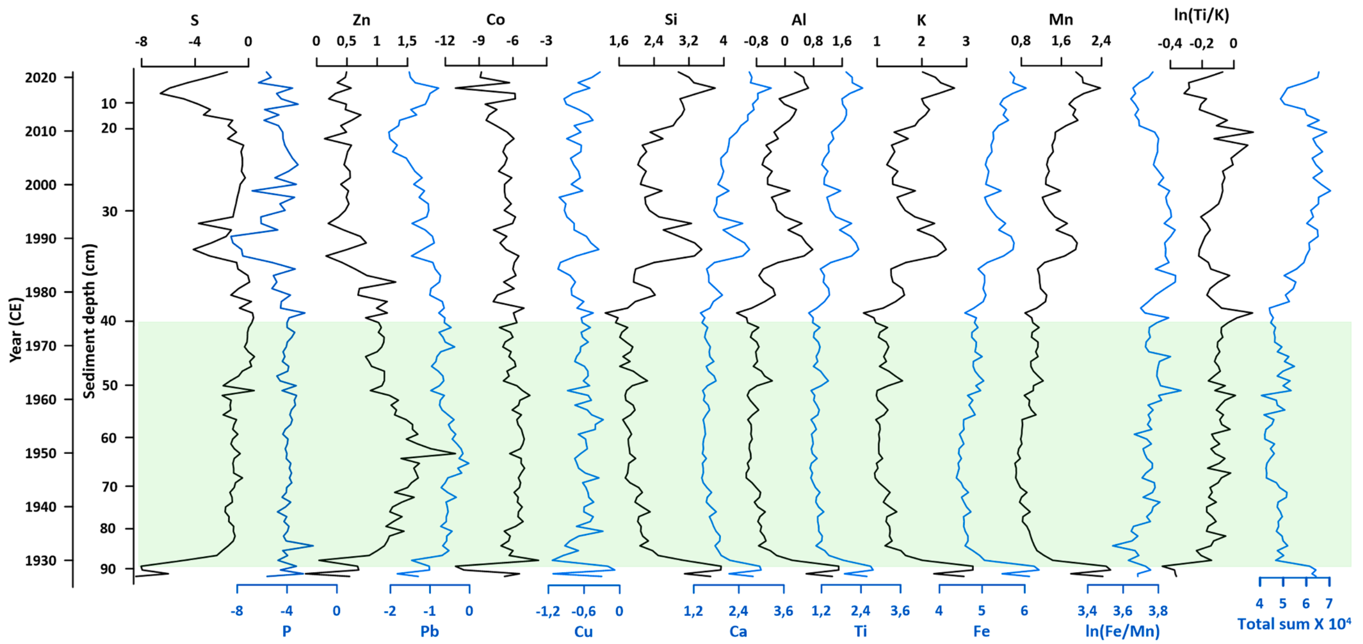

Fig. 6. XRF central log-transformed elemental data, logarithmic ratios of two elements, and total sum of cps of all elements. The green background highlights the organic-rich part of the sediment core labeled as Unit 2.

The formation/preservation of varves commences at around 92 cm depth in the sediment record, which corresponds around 1927. The varve quality is poor to fair in Unit 2, but improves to mostly good in Unit 3, simultaneously with an overall decrease in varve thicknesses. The varve year begins with CL, which forms in spring, as melting snow leads to flooding and increased erosion of the lake watershed (Ojala et al., 2013; Johansson et al., 2019). Eroded clastic particles are subsequently deposited on the lake floor, creating a lamina composed of minerogenic particles, mostly consisting of quartz and feldspar—the predominant constituents of the tonalitic bedrock of the study region (Fig. 5; Davies et al., 2015; GTK, 2025). The CL are enriched in elements associated with the mentioned minerals (for example, Al, K, Si). In addition, CL are enriched in Fe, which likely precipitates on the lake bottom following oxidation during the water column mixing in spring (Gälman et al., 2009). BL consist of organic matter of autochthonous (e. g. diatoms) and allochthonous (i.e. terrestrial plant materials) origin, produced during the growing season, as well as of the finest, highly degraded amorphous organic matter that settles on the lake floor (Ojala et al., 2013; Salminen et al., 2023). In addition, the BL are enriched with Zn and S (Fig. 5). The major source of both of these elements is likely from sewage waters, directed into the bay since 1907 (Juuti and Katko, 2014). However, the high concentrations are likely controlled by the enhanced supply and deposition of organic matter. Both Zn and S are easily incorporated in organic matter, and can settle in sediments in particulate form under oxygen-poor conditions (Holmer and Storkholm, 2001; Teiri et al., 2016; Fakhraee et al., 2017).

Seasonal oxygen depletion in the hypolimnion facilitates varve preservation. Recent measurements indicate that hypolimnetic hypoxia is prevalent during the summer months and can recur during the winter inverse stratification (Fig. 8). Given that the unit with poorly preserved varves starts in the late 1920s, it is likely that the oxygen status of the bay has transitioned to hypoxic conditions relatively rapidly, at least for short periods during late summer. The thermal stratification during summer and winter limits the oxygen supply due to lack of mixing in the water column (Boehrer and Schultze, 2008; Salonen et al., 2009; Madyouni et al., 2025). In an organic matter-rich environment, oxygen is consumed by organic matter (OM) mineralization and hypolimnetic hypoxia is formed. The seasonal anoxia can contribute to sulfate reduction and precipitation (Holmer and Storkholm, 2001). Moreover, aggregates of ZnS can form during early diagenesis in sediment through the reaction between Zn^{2+} and HS^- in anaerobic conditions; Zn^{2+} can be released to porewater due to remineralisation in the process of living algal communities that consume Zn from the water column (Juillot et al., 2023). However, Zn counts in the BL decrease in the late 1980s, likely related to both the building of wastewater treatment plant in 1974 and the concomitant decrease in overall OM content. The recent drops in S counts, however, are related to minerogenic event layers.

5.3. *The impacts of sewage waters supply into the bay since 1907*

The building of the sewage system in the city of Kuopio and directing the sewage waters untreated into the bay during the 1907 construction work have changed the bay system irreversibly. The high input of OM and nutrients from the sewage waters has resulted in a rapid increase in OM concentration in the sediments starting in Unit 1 with a continued increasing trend throughout Unit 2 until the late 1970s (Fig. 2). This is manifested in increasing varve thicknesses with the highest BL thicknesses culminating from the mid-1950s to 1970s. Within Unit 2, the CL are difficult to visibly detect, likely because the high input of OM has diluted the minerogenic laminae. The OM-rich wastewaters reached the bay throughout the year and can contribute to the formation of both CL and BL. Overall, the minerogenic input decreased rapidly after the filling operation and the sediment supply was quite stable from the 1920s until the late 1970s.

The relatively rapid deterioration of the bottom-water oxygenation is reflected in the onset of varve preservation and generally higher Fe/Mn ratio. Fe and Mn can exist in multiple oxidation states. Under oxygen-rich conditions in the hypolimnetic water, Fe and Mn are oxidized, forming ferrous (Fe^{3+}) oxy(hydroxides) and manganese (Mn^{4+}) hydroxides, which subsequently precipitate in the sediments (Davison and Woof, 1984; Stauffer and Armstrong, 1986; Dean, 2002). However, under hypoxic conditions, manganese oxy(hydroxides) are less stable than iron compunds under reducing environments (Davison and Woof, 1984; Davison, 1993; Shanahan et al., 2008). The increasing Fe/Mn ratio with simultaneously improving varve preservation suggest that the development of hypoxia at the lake floor began already around 1927. The Fe/Mn ratio rises until the 1950s and then remains elevated, reflecting intensified or prolonged periods of late summer hypoxia. This

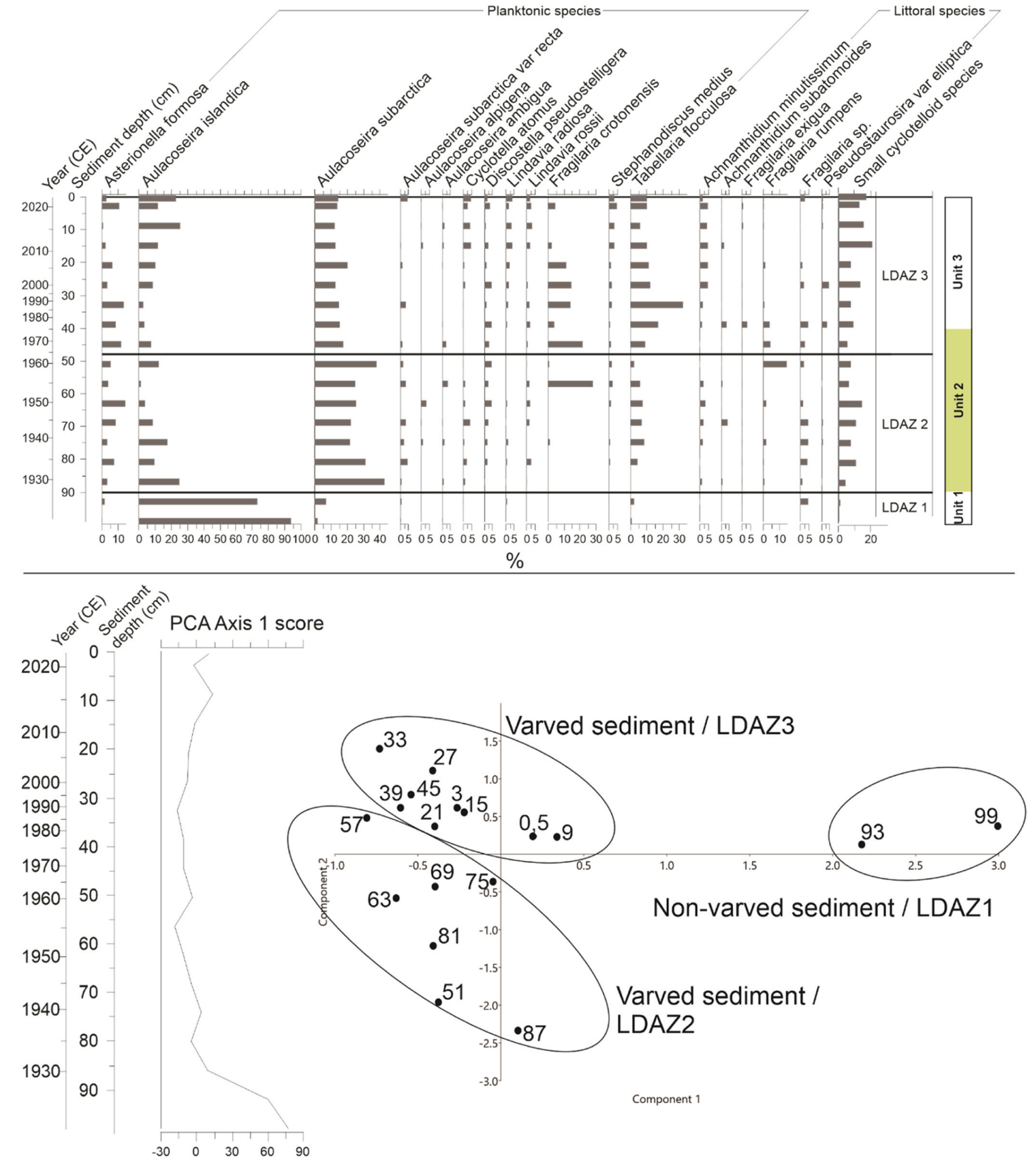

Fig. 7. Diatom species assemblages in the Maljalahti sediment sequence (upper panel), showing the taxa with over 3 % abundances. PCA analysis of the diatom species found in the sediment core (lower panel) shows principal patterns of variation in the diatom assemblages.

is supported by the occurrence of well-preserved varve structures since the 1940s.

The varve structures are distinctly reflected in the geochemical composition, with S and Zn being enriched in biogenic laminae (Fig. 5). The overall increasing trends in Zn and Pb peak in the late 1950s, after which they follow decreasing trends. Heavy metals like Pb and Zn could end up in the lake from various anthropogenic sources, ranging from industrial effluents and sewage to enhanced use of chemical products, land use changes from forested to urban areas, and atmospheric fallout (Baud et al., 2023). Compounds associated with these heavy metals can be released into the atmosphere as a result of human activities, e.g., using aerosols and combustion of heavy oil (Klopper et al., 2020). They

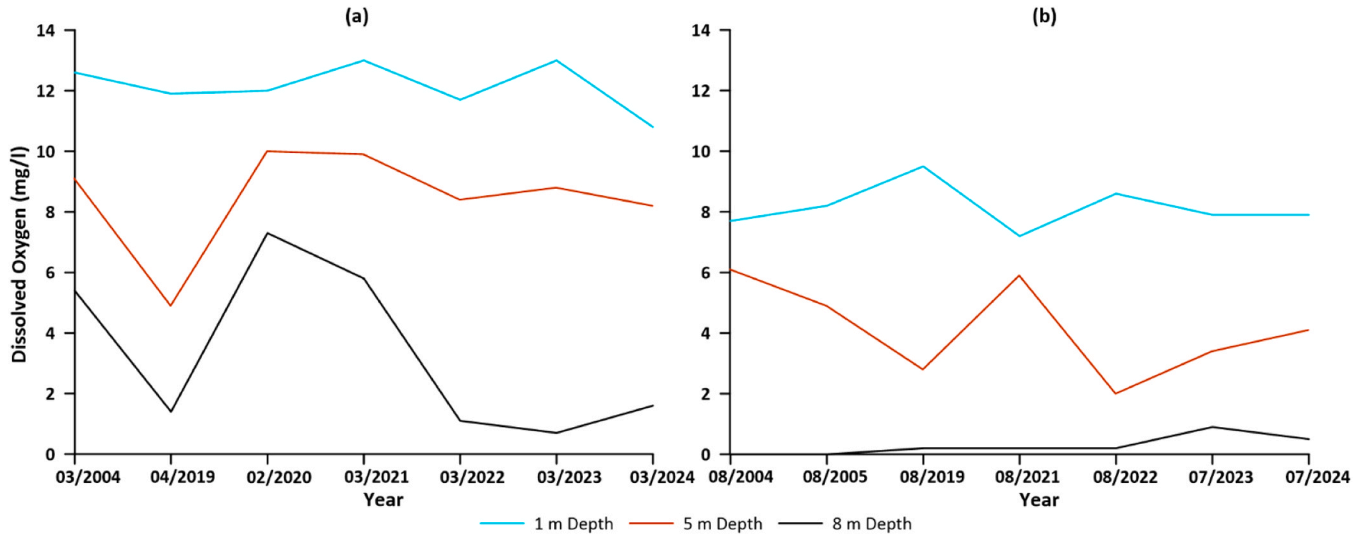

Fig. 8. Maljalahti Bay water column dissolved oxygen status recorded at 1 m, 5 m and 8 m depths during (a) winter and (b) summer in different years (SYKE, 2024).

could also travel over thousands of kilometers following the wind direction (France and Blais, 1998). Pb and Zn were observed to decrease since the 1970s in lakes of Eastern Canada, possibly linked to controlling atmospheric pollution and phasing out leaded gasoline (Meriläinen et al., 2003; Baud et al., 2023). It is likely, that the relative decrease in Pb and Zn in the Maljalahti record since the late 1950s is also linked to better management of atmospheric pollution in Finland and its neighboring countries. However, Zn is widely related to wastewater (Wang et al., 2005; Wang et al., 2024), which likely explains the later decline of Zn in the record.

The diatom assemblage changes completely from Unit 1 to Unit 2. The most common species, *A. subarctica,* is a planktonic species of oligotrophic to mesotrophic waters (Krammer et al., 1986; Krammer et al., 1988; 1991), although it has been also found in eutrophic lakes in Finland (Kauppila et al., 2002; Räsänen et al., 2016). *F. crotonensis* is planktonic, mainly in oligo- to weakly eutrophic lakes and moderately electrolyte-rich lakes (Krammer et al., 1986; Krammer et al., 1988; 1991). Along with *A. formosa, F. crotonensis* is considered as an indicator of increasing levels of reactive nitrogen in the western United States when diatom assemblages change from oligotrophic to eutrophic lake assemblages (Saros et al., 2005; Spaulding et al., 2015). *F. crotonensis* has also been considered as an indicator of eutrophic conditions (Reynolds et al., 2002; Weckström et al., 2015). *A. formosa* is highly competitive for phosphorus (Dam et al., 1994; Dixit et al., 2000; Judd et al., 2005), a common indicator species of increased ion concentrations (Mertens et al., 2025) and a potential marker of a changing trophic status (Kelly and Whitton, 1995; Kuefner et al., 2020). The steady increase of *A. formosa* since the 1920s and the first appearance of *F. crotonensis* in the late 1950s, likely reflect the eutrophication of the bay system. The small cyclotelloid species *C.atomus* and *D. pseudotelligera* are both meso- to eutrophic species, which also indicate higher saprobity and increased biochemical oxygen demand of the bay water (Riedmäuller et al., 2025; Mertens et al., 2025). *F. rumpens* is considered as an oligo- to mesotrophic species. Altogether, the diatom flora shows a change from mesotrophic to slightly eutrophic conditions in the bay between 1930 and 1965.

5.4. The onset of sewage water treatment plant

The inception of wastewater treatment in 1974 strongly influenced the bay system. The rapid decrease in BL in the mid-1970s likely resulted directly from reduced OM transport via sewage. Although the thickness of BL decreased following the introduction of sewage treatment in the 1970s, some layers still retain thicknesses comparable to those observed before sewage treatment. Yet, the bay has not recovered from the nearly 70 years of OM and nutrient loading, based on improved varve quality since the late 1970s. Indeed, improvements that could be observed at the Maljalahti Bay sediment record after the start of sewage water treatment have occurred very slowly. The gradual decline in the abundance of *A. formosa* and *F. crotonensis*, since the 1970s suggests a slow decrease in nutrient levels at the surface waters of the Bay. Additionally, the shift towards a higher C/N ratio around 1980 indicates a relatively less contribution from autochthonous production, which is plausibly due to reduced nutrient availability in the bay (Meyers, 1994; Sterner, 2008). *A. subarctica* decreased and *A. islandica* increased in abundance towards the end of the 20th century, which is most likely a consequence of declining nutrient levels in the bay after the construction of a sewage treatment plant. A related shift has been observed during the oligotrophication of Lake Mondsee, Austria (Dokulil and Teubner, 2005).

Sewage and other industrial waters were likely the primary source of Zn, S, and Pb. The onset of wastewater treatment is reflected by a slow decrease in Pb and Zn from the late 1950s and the late 1970s, respectively (Fig. 6). S associated with sewage waters begins to reduce around 1980 (Fig. 2). It should be noted that this rapid decrease occurs simultaneously with the overall change in sediment composition to more minerogenic. Hence, solely the decreased organic matter content could explain the decreasing S concentration. While micro-XRF line scan data fails to display the decreasing concentration of S, the drops in S (both concentrations and micro-XRF line scan data) within Unit 3 are directly related to deposition of clay layers as reflected by the low Ti/K ratio and low OM and should not be interpreted as changes in the system. These drops occur simultaneously with clear peaks in aluminosilicate elements (Fig. 6).

5.5. Filling of the bay in 1978

We relate the 1980s major decrease in OM content and simultaneous rapid increase in both magnetic susceptibility and dry bulk density to the construction work and further filling of the bay in 1978 which resulted in a significant change in the sedimentary environment. As a consequence, the shoreline was moved outward towards the deeper part of the bay, and the river Maljapuro outlet consequently now drains much closer to our coring site (Fig. 1). At the same time, the building of the breakwater in 1978 restricted the exchange of surface waters of the bay with the rest of the lake. This led to a more intense oxygen depletion based on increased Fe/Mn ratio and the significantly improved varve quality. However, the building of the breakwater and further filling of the bay in 1978 are not clearly visible in the diatom assemblages. An increase in small cyclotelloid species and *F. crotonensis* has been associated with increased EC and pH in Finnish lakes (Tammelin et al., 2019). EC has generally increased in the lake system because of salinisation of enhanced urban activity, for example, road de-icing (Tammelin and Kauppila, 2018; Tammelin et al., 2019). *Tabellaria flocculosa* and *A. minutissima* have a wide tolerance to various environmental conditions and may not be good indicators for any lake type (Räsänen et al., 2016).

At the same time, the Maljalahti varves progressively change from organic-dominated to minerogenic-rich clastic-biogenic varves. This is evident both in a clear change in bulk sediment composition and enhanced contribution of elements related to aluminosilicates (Al, Ca, K, Si, Ti). The possible reasons responsible for the drastic change in varve composition are (1) lowered supply of OM and restricted biogenic matter supply mostly related to growing season, after the introduction of the sewage treatment plant, (2) enhanced supply of minerogenic matter due to the shift of the Maljapuro river inlet towards the deepest location of the bay (i.e. the coring site) and more pronounced seasonal detrital pulses mostly in spring, (3) enhanced hypoxia that facilitated better varve preservation, and (4) decrease in accumulation of minerogenic matter outside the spring flood season due to a considerable reduction in the littoral area related to the bay infilling together with a possible decrease in sediment resuspension in a break water closed bay.

6. Conclusion

The documented anthropogenic activities have had a clear impact on the water quality and sediment characteristics of Maljalahti Bay. Physical, geochemical, and biological (diatom) proxies of a well-dated sediment record of urban Maljalahti Bay highlight the impact of documented anthropogenic actions on water quality and sediment characteristics.

The introduction of sewage waste into the bay starting in 1907 likely contributed to its eutrophication by increasing inputs of nutrients and organic matter. As a result, organic-rich varved sediments have been formed since 1927. This indicates the rapid development of at least seasonally hypoxic hypolimnetic waters within 20 years, resulting from the high oxygen demand of decomposing organic matter. Between 1930 and 1980, an increase in the deposition of heavy metals, such as zinc and lead, was observed. Following the establishment of a wastewater treatment plant in 1974, nutrient levels in the bay water began to decline. This led to a decrease in OM input and a related reduction of the heavy metal deposition in the sediments. However, oxygen depletion further intensified because of the construction of a breakwater and land reclamation in the bay area in 1978, restricting the water exchange with the main water body of the Lake Kallavesi basin. This is reflected in improved varve quality. Additionally, the effects of landfilling and dredging activities, conducted in 1978, 2011, and 2019, are distinctly evident in the varved sediments of the bay.

Despite the introduction of the wastewater treatment plant, there has been no significant improvement in water quality. Such conditions are likely common in urban lakes that experience restricted water exchange or small water bodies. While lake systems worldwide are grappling with eutrophication, our findings indicate that efficient water exchange is essential in addition to limiting nutrient inputs to achieve meaningful improvements in water quality.

CRediT authorship contribution statement

Mohib Billah: Writing – review & editing, Writing – original draft, Visualization, Investigation, Funding acquisition, Formal analysis, Data curation. **Nanna Tuovinen:** Writing – review & editing, Writing – original draft, Visualization, Investigation, Formal analysis. **Sarianna Salminen:** Writing – review & editing, Writing – original draft, Supervision, Investigation, Data curation. **Wojciech Tylmann:** Writing – review & editing, Visualization, Formal analysis, Conceptualization. **Maarit Kalliokoski:** Writing – review & editing, Data curation. **Saija Saarni:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Funding acquisition, Conceptualization.

Declaration of Generative AI and AI-assisted technologies in the writing process

The authors utilized Avidnote to improve the language of the text while preparing this document. The authors reviewed and edited the content as necessary and assume full responsibility for the publication's content.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ancene.2025.100508.

Data availability

Data will be made available on request.

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