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Recolonization and recovery of an Arctic benthic community subject to mine-tailings deposits

Hilde C. Trannum [a,b,*], Kristine B. Pedersen [c,d], Paul E. Renaud [c,e], Guttorm N. Christensen [c], Anita Evenset [c,f]

[a] Norwegian Institute for Water Research, Jon Lilletuns vei 3, NO-4879 Grimstad, Norway
[b] Centre of Coastal Research, Department of Natural Sciences, University of Agder, P.O.Box 422, NO-4604 Kristiansand, Norway
[c] Akvaplan-niva, Fram Centre – High North Research Centre for Climate and the Environment, NO-9296 Tromsø, Norway
[d] UiT, the Arctic University of Norway, Faculty of Science and Technology, Department of Chemistry, No-9037, Tromsø, Norway
[e] University Centre in Svalbard, NO-9071 Longyearbyen, Norway
[f] UiT, the Arctic University of Norway, Faculty of Biosciences, Fisheries and Economics, NO-9037 Tromsø, Norway

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ABSTRACT

Deposition of large volumes of mine tailings takes place in several Norwegian fjords, but the impacts on marine ecosystems have received relatively little scientific attention. At a 40 + −year old tailing deposition area for a copper mine in the Arctic fjord Repparfjorden, we investigated both short-term colonization of mine tailings-contaminated sediments through a field experiment, and the present faunal state in the old deposit area. Copper-concentrations at the old deposition site were still high (up to 291 mg/kg dry weight (dw)), and exceeded the Norwegian environmental-quality threshold (84 mg/kg dw). Furthermore, copper was identified as a significant structuring factor for the fauna in the fjord, although faunal diversity was relatively high and the community not severely disturbed. In the colonization experiment, experimental boxes filled with defaunated sediment capped with mine tailings were subject to colonization for 15 months. Benthic macrofaunal communities were successfully established in all boxes, but the boxes with tailings showed lower species richness, abundance and biomass than the controls. Mine tailings continue to have local impacts on seafloor communities decades after deposition, and even low levels of metal-rich sediments can affect faunal recruitment. These results have implications for submarine deposition of mining waste and the impacts they have on coastal ecosystems.

1. Introduction

The growing demand for minerals has increased interest in mineral resources worldwide, both for traditional uses as well as for the development of new, green-energy technology (Vogt, 2013; Dold, 2014). In Norway, mining is currently in a phase of growth and subject to new environmental regulations (Ramirez-Llodra et al., 2015). The main environmental issue with regard to industrial mining activities is the production of vast volumes of non-processed overburden rock and processed tailings. Tailings are the waste produced after extraction of the targeted metal from the ore, and are deposited as fine-fraction slurry that usually accounts for a high proportion of the mined ore (e.g. up to 99% for copper). The tailings include not only milled rock with elevated concentrations of heavy metals resulting from the insufficient recovery of ore minerals, but may also contain added processing chemicals (Koski, 2012; Ramirez-Llodra et al., 2015¸ Ramirez-Llodra et al., 2022).

Disposal of tailings at sea is considered a viable option in some nations as tailings are often more stable on the ocean floor than on land and are less likely to oxidize in the submarine environment due to significantly lower oxygen availability at the seafloor (Andersson et al., 2018). However, such submarine tailing disposal (STD) may affect the coastal environment through several mechanisms, mainly hyper-sedimentation, nutrient depletion, changes in the substrate and submarine topography, sediment plumes, and higher turbidity (reviewed in Shimmield et al., 2010; Ramirez-Llodra et al., 2015). Aside from the particle component, metals, process chemicals, and tailings can be toxic, or interact with organic material to alter the redox state of sediments (Ramirez-Llodra et al., 2015). In order to minimize dispersal of deposited tailing, sub-sea deposits are established in soft-bottom accumulation habitats. As the quantity of tailings discharge into the marine environment can be huge,

* Corresponding author at: Norwegian Institute for Water Research, Jon Lilletuns vei 3, NO-4879 Grimstad, Norway.
E-mail address: hilde.trannum@niva.no (H.C. Trannum).

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up to several million tons per year, the benthic communities are expected to be most affected by such disposal (Ramirez-Llodra et al., 2015; Trannum et al., 2018). Locally, within the designated STD, all benthic fauna will be constantly buried. Observed effects of such deposition include a reduced taxonomic and functional diversity as well as altered species composition and ecological functioning, and some of these effects may also extend beyond the immediate area of deposition and be identified after deposition has ceased (Ellis and Hoover, 1990; Olsgard and Hasle, 1993; Allan, 1995; Brooks et al., 2015; Mevenkamp et al., 2017; Trannum et al., 2018; Trannum et al., 2019).

In the Arctic fjord Repparfjorden situated in Finnmark (northern Norway), copper (Cu) extraction and subsequent tailings disposal took place in the inner part of the fjord from 1972 to 1978, and new mining activity is planned from 2023 and onward. A new sub-sea deposit will be established in the middle part of the fjord and is expected to cover an area of 4.9 km^2. Recent studies have documented that tailings from the previous mining activities are still present in the inner part of the fjord (Sternal et al., 2017; Andersson et al., 2018; Pedersen et al., 2017), but there have been no comprehensive investigations of whether previous disposal continues to affect the benthic communities. It is well known that benthic species composition, abundance and biodiversity change as a response to local environmental conditions (Pearson and Rosenberg, 1978), and these communities are therefore indicators of local environmental status including the assessment of mine-tailings impacts (Olsgard and Hasle, 1993; Brooks et al., 2015; Trannum et al., 2019).

Colonization experiments are a highly relevant method to study the benthic recovery potential after disturbances, particularly in cases where defaunated areas frequently occur, like in the case with STDs (Trannum et al., 2020). Colonization mechanisms include both pelagic larval recruitment and adult migration, where larval recruitment generally accounts for the largest part (McCall, 1977; Diaz-Castañeda et al., 1993). As settling and metamorphosis of pelagic larvae are considered the most critical stages in the development of marine benthic assemblages, colonization experiments also provide highly relevant ecological information (Olsgard, 1999). Even small amounts of particles (scale of mm to cm) from e.g. mining and petroleum activities have been shown to impact colonization of benthic organisms (Trannum et al., 2011; Sweetman et al., 2020; Trannum et al., 2020). Also, Cu has been observed to significantly reduce or alter benthic colonization (Olsgard, 1999; Trannum et al., 2004). It is toxic to a wide spectrum of marine life (Clarke, 1997), and affects benthic community composition and diversity (Rygg, 1985; Stark, 1998; Neira et al., 2011). Reduced or delayed colonization and impaired community functioning have been observed (Sweetman et al., 2020; Trannum et al., 2020), but such knowledge of mine tailings effects on colonizations is based on few studies. As previous studies have been conducted with temperate benthic communities, there is a knowledge gap regarding the response of Arctic communities.

The objectives of this study are a) to investigate whether thin layers of mine tailings affect short-term colonization potential in Arctic soft-sediment communities, and b) to investigate how previous mining activities have affected the environmental status of the fjord.

2. Materials and methods

2.1. Study area

Repparfjorden is situated on Kvaløya island in Troms and Finnmark County, Norway (Fig. 1). It opens towards the Norwegian Sea via Kvalsundet to the southwest and Sammelsundet to the northeast. A sill at 80–85 m depth, located on the fjord's most narrow part, separates the outer part and a deeper basin (approximately 120 m) in the inner part. A second sill at 50 m depth is present where the fjord changes direction into the innermost part, stretching northwest-southeast. The Repparfjord River forms a delta in the inner part of the fjord. The water chemistry, including salinity, temperature, pH and DOC, were measured on six occasions in 2010–2011 (Christensen et al., 2011), and the seasonal variations of salinity were in the range 31.7–34.2 ppt, pH in the range 8.0–8.1, temperature in the range 1-8 °C and DOC in the range 1.0–2.1 mg/l.

2.2. Tailings deposition, past and planned

During mining operation by Folldal Verk A/S in 1972–1978, Cu was extracted from the Ulveryggen ore in Repparfjord, and mine tailings

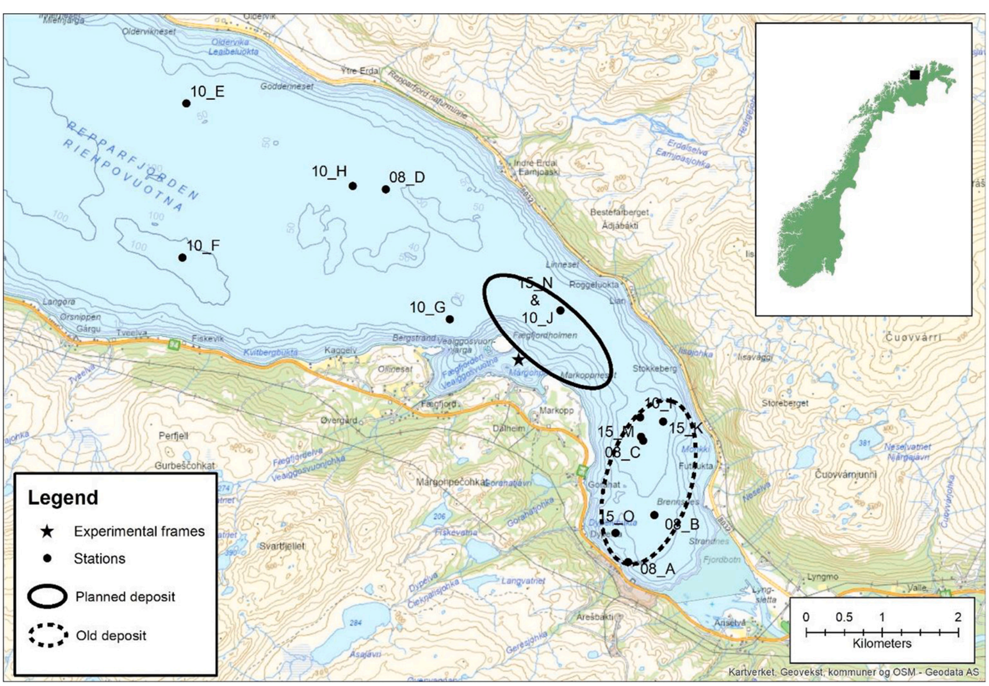

Fig. 1. Map of the benthic stations in Repparfjorden (2008–2015). Stations are denoted by number (sampling year) and letter. The star shows the location of the colonization experiment, the thick, black circle shows the approximate area where the new deposit will be established, and the dotted circle shows the assumed placement of the old deposit.

were discharged 20 m above the seabed in the inner part of Repparfjorden (south-east) via a pipeline (Sternal et al., 2017). The precise location of the pipeline and the specific placement of the mine tailings are uncertain. However, in a recent study by Andersson et al. (2018), cones of old tailings were recorded along a southwest-northeast transect in the inner part of the fjord (see Fig. 1). In addition, tailings were spread towards a larger area including more shallow parts (Andersson et al., 2018), thus it is likely that the whole area inside the inner threshold is more or less affected by old mine tailings (including the stations 08_A, 08_B, 08_C, 10_I, 15_K, 15_M and 15_O in Fig. 1). The exact quantity of tailings discharged in the operational period was not recorded, but it has been estimated to be approximately 1 million ton (Kvassnes and Iversen, 2013). The concentration of Cu in the mine tailings was not recorded in the 1970s, but analysis of Ulveryggen mine tailings in 2010, applying similar flotation methods as in the1970s, revealed concentrations in the range 700–1100 mg/kg dry weight (dw), depending on the grain-size fraction of the analysed subsamples (Kleiv, 2011).

Environmental investigations from 2013 to 2015 revealed that the mine tailings in the old STD had Cu concentrations in the range 102–1316 mg/kg dw, and it was shown that the dispersion of tailings particles to the outer part of the fjord was limited due to the sill on the seabed separating the inner and the outer fjord (Sternal et al., 2017). However, an estimated 2.5–10 tons of Cu, representing <5% of the original content in the mine tailings, dispersed to the outer fjord in the 1970s due to dispersion of fine particles as well as desorption and dispersal of Cu in the water column (Pedersen et al., 2017). The same study found that Cu in the mine tailings sediments is primarily bound in the exchangeable fraction of the sediment, making it more easily mobilized, e.g. by dissolution of carbonates or ion exchange, compared to Cu in natural sediments in Repparfjorden.

The mining company Nussir ASA plans to re-open the mine in near future with plans for exploitation of the Cu ore from Ulveryggen (mined previously) and Nussir sites. The mined ores will be crushed to fine size particles and subsequently flotation will be used to recover Cu from the minerals. The mining company plans to establish a new STD in Repparfjorden with disposal capacity of approximately 30 million tons of mine tailings. In order to limit dispersion of particles during discharge and ensuring fast sedimentation in the STD, the flocculant chemical Magnafloc10 is planned to be added to the slurry prior to discharge. In 2015, Nussir ASA received a permit from the Norwegian Environment Agency for annual discharge of 1–2 million tons of mine tailings and associated chemicals. The fine fraction (% < 63 um) of the tailings is approximately 70%, and the content of organic matter approximately 0.3%. The tailings characteristics differ from the natural sediments in Repparfjorden by having 5–40 times higher concentrations of barium (Ba), copper (Cu), magnesium (Mg) and manganese (Mn) (Kleiv, 2011; Sternal et al., 2017).

The new STD is planned for one of the deepest parts of the fjord (80 m), relatively close to, but seaward of, the inner sill (black circle, Fig. 1). Model simulations show that a deposit area of 4.9 km^2 will be subject to deposition of >1 cm tailings, during the 15–25 year period (Didriksen and Wilersrud, 2010).

2.3. Field work

2.3.1. Colonization experiment

Rock from the two ores that will be mined in the coming years (Nussir, 20% of the volume, and Ulveryggen, 80% of the volume) was provided by Nussir ASA. The rock was processed at the Norwegian University of Science and Technology to simulate the processing planned for production in the new mine, i.e. grinding (<160 μm) and flotation. The fresh tailings should, thus, be similar to the tailings that will be deposited in Repparfjorden when mining recommences.

Sediments were collected near the small island close to the new deposition site (Fig. 1) on 8th June 2015 with the fishing vessel "Fiskaren". They were homogenized with a mixer and filled into sixteen 0.097 m^2 propene plastic boxes (height 15 cm), up to approximately 3 cm below the upper edge of the box. The boxes were then frozen at −20 °C for a minimum of 12 h. Tailings were added on top of the frozen natural sediments in nominal layer thicknesses of 6, 10 and 14.5 mm; 4 boxes of each. In addition, 4 boxes with no added tailings served as controls. The boxes were then frozen for additional 24 h. The purpose of the freezing was both to eliminate the macrofauna of the natural sediment and to avoid sediment resuspension during deployment (Olsgard, 1999; Trannum et al., 2004: Trannum et al., 2020). The organic matter from the frozen fauna remained in the boxes, together with microbes that eventually were not killed. Nevertheless, such remnants were similar among the boxes. The boxes were randomly distributed in four aluminum frames, with one replicate of each treatment (including control) in each frame. The boxes were named with a first number according to frame number, and a second number according to added tailings layer. The frames were placed on a non-sloping seabed approximately 10 m apart at 18–20 m depth next to Fægfjordholmen (mean position of the frames 70 28.9°N/24 15.2°E, Fig. 1) with the research vessel "Seisma". Ropes were attached to each corner of the frame and connected by a buoy approximately 2 m above the frame. A schematic drawing of the setup is shown in Fig. 2.

On 26th September 2016, after 15 months on the seabed, divers located the frames, put a lid on each box to avoid sediment resuspension, and then the frames were lifted into a boat with a crane. Fifteen months allowed for an entire annual cycle during which juveniles could colonize the boxes and additional months for recruitment to macrofaunal size classes. The sediment in each box was sieved through a 1 mm sieve to collect the macrofauna, and the sieve residue was fixed in 4% buffered formaldehyde stained with Rose Bengal. Two boxes had to be removed from the faunal analyses because of loss of some material during recovery (box 1–0 mm and 1–6 mm, but the chemical data were used). Before sieving, sediment subsamples were taken for analysis of grain size (0–5 cm), metals (0–5 cm) and total organic carbon (TOC) (0–1 cm) (according to NS-EN ISO 16665:2014) with a hand-held corer (d = 5 cm).

2.3.2. Field study

Benthic macrofauna was sampled in three different campaigns; 2008, 2010 and 2015 (June–September) (Fig. 1; station overview and sediment characteristics are given in Supplementary Material 1). The two first campaigns were part of a baseline pre-mining assessment

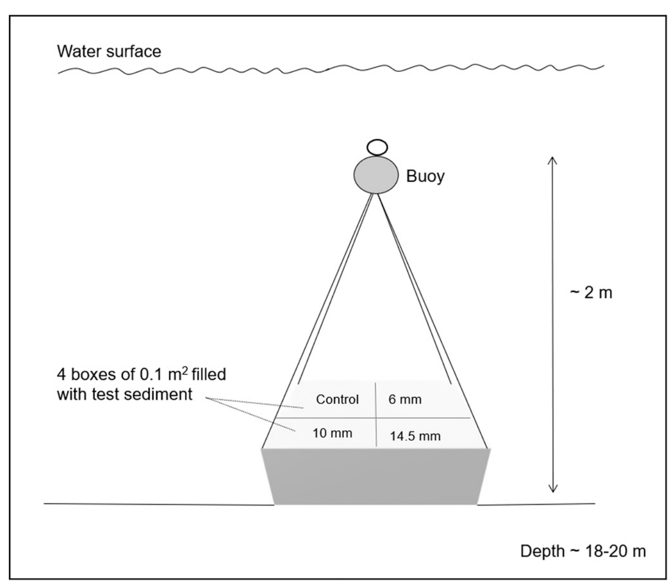

Fig. 2. Schematic illustration of the experimental setup used in the colonization experiment in Repparfjorden.

(Christensen et al., 2011), while the campaign in 2015 was related to the present research project, and also pre-mining. In all cases, the sampling was conducted with a 0.1 m^2 van Veen grab, following the general guidelines for quantitative sampling and sample processing of marine soft-bottom macrofauna (NS-EN ISO 16665:, 2014). At each station, except station 15_K, four grabs for faunal analyses and one grab for sediment analyses were collected. At station 15_K, only three grabs were collected for faunal analyses due to presence of many stones preventing good sediment recovery by the grab.

The faunal samples were washed through a 1 mm mesh sieve for retrieval of macrofauna, and the retained material was fixed in 4% buffered formaldehyde and stained with Rose Bengal. As for the boxes, a hand-held corer was used to take sub-samples for sediment parameters (grain size and metals (0–5 cm) and TOC (0–1 cm)) from one grab at each station.

2.3.3. Laboratory analyses

Total organic carbon (TOC) was determined by Akvaplan-niva in all sediment samples (from experimental frames and grab samples) using a Leco IR 212 carbon analyser, after the sediment had been dried, and inorganic carbon had been removed by acidification. Sediment fine fraction (% < 63 mm) was determined gravimetrically after separation from the coarse fraction by wet sieving.

Metal analyses from different years were performed by different laboratories according to accredited methods. Samples from 2008 were analysed by ALS Scandinavia using ICP-AES (Inductively Coupled Plasma-Atomic Emission Spectrometers), samples from 2010 by Norwegian Institute for Water Research using ICP-MS (Inductively Coupled Plasma – Mass Spectrometry), and samples from 2015/2016 (field and colonization study) at the Technical University of Denmark using ICP-OES (Inductively Coupled Plasma – Optical Emission Spectroscopy).

The metals arsenic (As), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), manganese (Mn), nickel (Ni), lead (Pb) and zinc (Zn) were analysed in all sediment samples collected in the fjord and are therefore included in the data analyses. In the colonization experiment, 22 metals/minerals were analysed; aluminum (Al), arsenic (As), barium (Ba), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), potassium (K), magnesium (Mg), manganese (Mn), molybdenum (Mb), sodium (Na), nickel (Ni), phosphorous (P), lead (Pb), antimony (Sb), strontium (Sr), vanadium (V), zinc (Zn), zirconium (Zr).

The macrofauna from both experimental frames and grab samples was separated from sediment and sorted in main taxonomic groups under 10× magnification, transferred to 80% ethanol and then identified to species or lowest possible taxonomic level. As different persons were involved in the macrofauna identification, the species lists from all sampling campaigns were harmonized with regard to taxonomic level and nomenclature. All analyses were done at the same biological laboratorium (situated at Akvaplan-niva), which holds a reference collection enabling verification of taxa for all years. Species names were checked according to the world register of marine species (WoRMS; marinespecies.org) prior to the statistical analyses. For the colonization experiment, each taxonomic group (phylum) was also weighed (blotted wet weight, ± 0.001 g) for biomass calculation.

2.3.4. Data analyses

For the colonization experiment, parameters that differed significantly between the natural field sediment and tailings sediment were used to assess whether the tailings remained in the boxes during the experiments. The tailings are known to have elevated concentrations of Ba, Ca, Cu and Mn, finer grain size, and lower TOC compared to natural Repparfjorden sediments. In this study, the TOC concentrations in the sediment were low, in most cases below the detection limit of 1.3 mg/kg. For this reason, TOC was excluded from the analysis and only metal concentrations were assessed as independent variables. Single factor analysis of variance (ANOVA, one way) was used to evaluate and compare the metal concentrations in the mine tailings and natural sediment (performed in Excel).

For each faunal sample, number of taxa/species (species richness) and number of individuals (abundance) were recorded and the Shannon-Wiener index H′ (Shannon and Weaver, 1963) was calculated with log2 as the base. This was done both for the fjord and colonization data. For the colonization data, these variables as well as biomass were subject to one-way Analysis of Variance (ANOVA), after first testing for homogeneity of variances. Significant ANOVA results ($p < 0.05$) were followed by the HSD multiple comparisons test for unequal sample sizes.

Multivariate analyses were performed with PRIMER package version 6.1.13 (Clarke and Warwick, 2001). These included non-metric multidimensional scaling (nMDS) (fjord and colonization data separately) and Analysis of Similarity (ANOSIM) (colonization data). Similarity was calculated with the Bray-Curtis similarity index (Bray and Curtis, 1957), based on fourth-root transformed data, to downweigh the influence of highly abundant taxa. Within the same program, we used Distance-based Linear Model (DistLM, Anderson, 2001) to identify relationships between species composition and environmental variables for the fjord data. This analysis was based on the same Bray-Curtis matrix as the nMDS. The environmental variables measured from the grab samples were used in the analyses, and included nine metals/minerals, in addition to TOC and pelite (sediment fine fraction, % < 0.63 mm). Due to several highly correlated variables, the environmental matrix was reduced (based on PCA), and the final set of environmental variables that entered the analysis included As, Cu, Mn, Pb, TOC and pelite (correlation matrix is given in Supplementary Material 2). To investigate the variables in combination (sequential tests), the forward selection procedure was run with a maximum of 9999 permutations and with the adjusted R^2 selection criterion (i.e. R^2 adjusted for number of parameters). Marginal tests were performed to quantify how much variation each variable explained individually, i.e. excluding other variables. The ordination method of distance-based redundancy analysis (db-RDA) was used to visualize the results, where environmental variables and species were superimposed as vectors. The db-RDA produces an ordination which is constrained to be a linear combination of the environmental variables responsible for explaining significant portions of the variation within the data cloud.

3. Results

3.1. Colonization results

3.1.1. Environmental data, colonization experiment

The field sediment that was used in the boxes was very coarse, with only 3–4% fine fraction (Table 1, full dataset in Supplementary Material 3). Apart from box 4–6 mm, box 3–10 mm and box 4–14.5 mm, all boxes with tailings had higher content of fine material compared to the control boxes ($F9.7 > F5.0$; df12; $p < 0.05$–0.01). Boxes with 10 and 14.5 mm had higher concentrations of metals in the upper 5 cm compared to the control boxes ($F8.5$–$11.5 > F5.3$; df11; p 0.01–0.02). The highest metal concentrations were observed in boxes with 10 (box 2) or 14.5 (box 1–3) mm tailings. Three of the boxes with 6 mm tailings (box 1, 2 and 4) did not show elevated metal concentrations. Thus, the boxes with 10–14.5 mm added tailings showed a moderate-high influence of tailings on grain size and/or metal concentrations compared to control, indicating that the tailings prevailed in the sediment during the 15 months of experimental duration, i.e. the boxes were not heavily affected by resuspension or new sedimentation.

3.1.2. Community data, colonization experiment

In total 95 taxa and 3588 individuals were recorded in the boxes. The species richness per box ranged from 21 (2–14.5 mm) to 47 (3–0 mm), abundance from 117 (1–10 mm) to 531 (3–0 mm), and total wet biomass from 0.74 g (4–14.5 mm) to 3.9 g (2–0 mm) per box. The Shannon-Wiener index ranged from 1.6 (2–14.5 mm) to 3.0 (1–10 mm). The ANOVA-test was significant for species richness ($F_{1,3} = 4.60$, $p =$

Table 1
Pelite fraction (% <63 μm) and selected metal concentrations in the experimental boxes (mean values) at the end of the 15 months experimental period, as well as in raw tailings (described in chapter 2.3.1).

	<63 μm	Ba	Ca	Cu	Mn
	%	mg/kg dw			
Control sediments					
1–0	2.7	28	3010	14	112
2–0	4.1	32	3630	19	126
3–0	3.7	29	4087	20	140
4–0	3.2	31	4090	37	135
6 mm tailings					
1–6	19	33	4700	41	148
2–6	12	31	3140	14	105
3–6	15	41	10,460	110	353
4–6	3.9	33	3910	42	143
10 mm tailings					
1–10	8.2	34	5730	55	179
2–10	7.8	74	34,000	393	1032
3–10	3.8	40	13,750	134	384
4–10	8.3	40	10,810	117	299
14.5 mm tailings					
1–14.5	21	50	19,000	213	526
2–14.5	6.7	48	16,550	176	483
3–14.5	42	56	23,040	283	714
4–14.5	4.7	35	7400	86	232
Tailings					
	71. 8	200	59,400	375	2300

0.029), abundance (F 1,3 = 3.92, $p = 0.044$) and biomass (F 1,3 = 5.69, $p = 0.015$), but not for Shannon-Wiener diversity (H′) (F 1,3 = 2.20, $p = 0.15$). The results of the post-hoc test are shown in Fig. 3, where it is evident that the tailings-treatments were associated with a lower species richness, abundance and biomass than the control treatment, i.e. indicating a reduced colonization.

The annelid *Dipolydora* sp. clearly dominated the communities within the boxes and constituted as much as 58–75% of the total abundances (see list of ten most dominating taxa in Table 2, and complete species list in Supplementary Material 4). Notably, the abundance of this taxon in the 14.5 mm treatment was only half of that in the control. It also had a lower abundance in the 6 and 12 mm treatments than in the control. Other highly abundant taxa in the control, the bivalves *Macoma calcarea, Parvicardium pinnulatum* and *Mya* sp. juv., and the annelid *Chone* sp. had considerably lower abundance in the treatments with tailings than in the control (see Table 2 and Supplementary Material 4).

In the MDS-ordination, the community structure was clearly different between the control and experimental treatments, although grouping according to the quantity of tailings added was not very pronounced (Fig. 4). According to ANOSIM (Table 3), community structure at 14.5 mm was significantly different from control ($p = 0.029$). For the 10 mm treatment no significant difference was observed, but the *p*-value was nearly statistically significant at 0.057.

3.2. *Field study*

3.2.1. *In situ environmental data*

Results of TOC, grain size and selected metal analyses are summarized in Table 4. In the historical disposal area (inside the inner threshold), the fraction of pelite was higher (average of 86%) compared to sediments outside the sill (average 57%). The variability in pelite content was also higher outside the sill compared to the disposal area (Table 4). In the disposal area, higher concentrations of TOC (average 12.8 g/kg) were found compared to the area outside the threshold (average 8.4 g/kg). Concentrations of Co and Pb were similar in the disposal area and outside the threshold ($F_{0.8–1.3} < F_{crit4.7}$; df14; $p = 0.27–0.40$). For all the other metals in Table 4 (As, Cr, Cu, Fe, Mn, Ni and Zn) there was a significant difference in concentrations in the disposal area and outside the threshold ($F_{6.4–55} > F_{crit4.7}$; df14; $p < 0.05$).

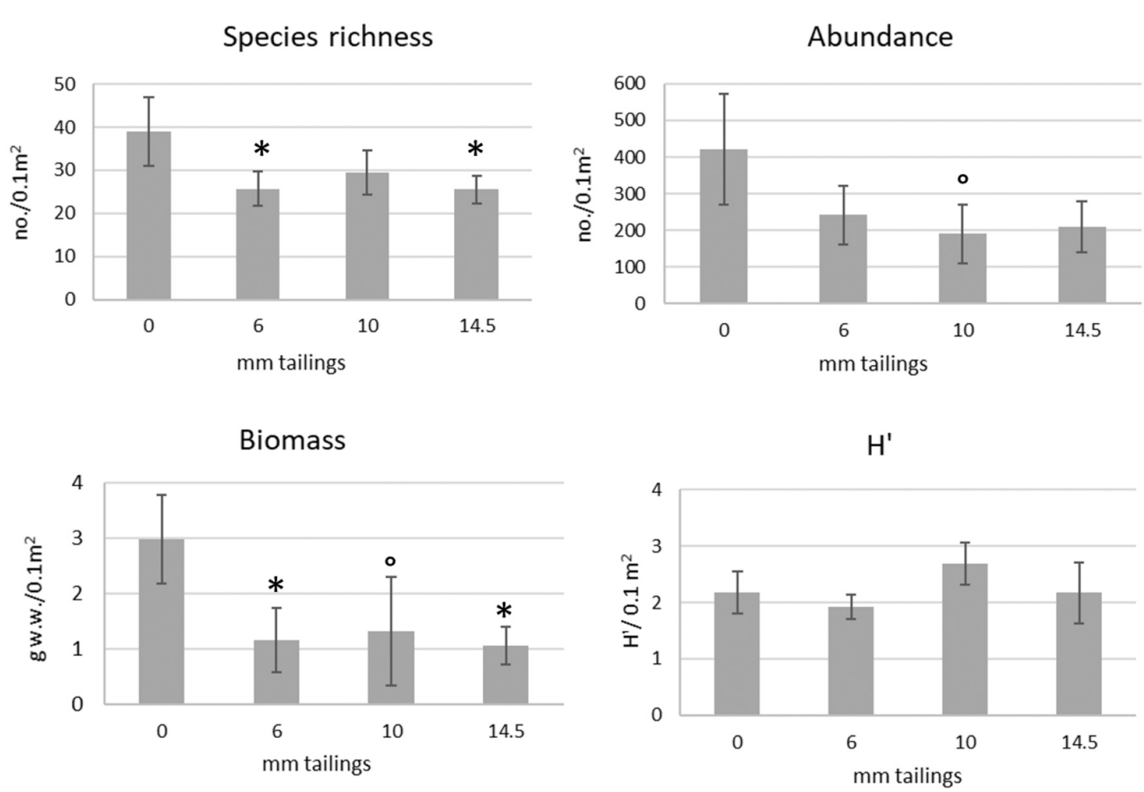

Fig. 3. Mean number of species, abundance, biomass and Shannon-Wiener diversity in each treatment in the frames collected in Repparfjorden, 2016 (average pr. 0.1 m^2 +/− S.D.). Asterisk indicates treatment means significantly different from the control (0 mm tailings) (*$p < 0.05$, also °$p < 0.1$ is indicated).

Table 2
Abundances (N) of ten most dominating taxa in each treatment in the experimental boxes collected in Repparfjorden, 2016 (average pr. 0.1 m^2). Percentage (%) of total abundance is also presented. Taxonomic group in parenthesis, where A = Annelida, B=Bivalvia, C=Crustacea, G = Gastropoda, E = Echinodermata.

Control	N	%	6 mm	N	%
Dipolydora sp. (A)	299	70.9	*Dipolydora* sp. (A)	180	74.6
Macoma calcarea (B)	11	2.7	*Macoma calcarea* (B)	7	2.9
Parvicardium pinnulatum (B)	10	2.4	Natantia indet (C)	6	2.5
Mya sp. juv. (B)	10	2.4	Gammaridea indet (C)	5	2.2
Gammaridea indet (C)	9	2.1	*Eteone longa* (A)	4	1.5
Chone sp. (A)	7	1.7	*Parvicardium pinnulatum* (B)	3	1.4
Polynoidae juv. (A)	5	1.2	*Mya* sp. juv. (B)	3	1.4
Pholoe assimilis (A)	5	1.1	Polynoidae juv. (A)	3	1.2
Glycera capitata (A)	5	1.1	*Glycera capitata* (A)	3	1.1
Eteone longa (A)	4	1.0	*Pholoe assimilis* (A)	2	1.0
10 mm	**N**	**%**	**14.5 mm**	**N**	**%**
Dipolydora sp. (A)	110	58.0	*Dipolydora* sp. (A)	149	70.8
Calanoida indet (C)	10	5.1	Gammaridea indet (C)	7	3.1
Natantia indet (C)	9	4.6	*Eteone longa* (A)	5	2.5
Gammaridea indet (C)	8	4.1	Natantia indet (C)	5	2.4
Buccinum sp. juv. (G)	7	3.6	*Buccinum* sp. juv. (G)	5	2.3
Polynoidae juv. (A)	4	2.1	*Phyllodoce groenlandica* (A)	3	1.4
Eteone longa (A)	4	1.8	*Parvicardium pinnulatum* (B)	3	1.2
Macoma calcarea (B)	3	1.6	*Mya* sp. juv. (B)	3	1.2
Asteroidea juv. (E)	3	1.6	*Ophiura robusta* (E)	3	1.2
Ophiuroidea juv. (E)	3	1.4	*Pholoe assimilis* (A)	2	1.1

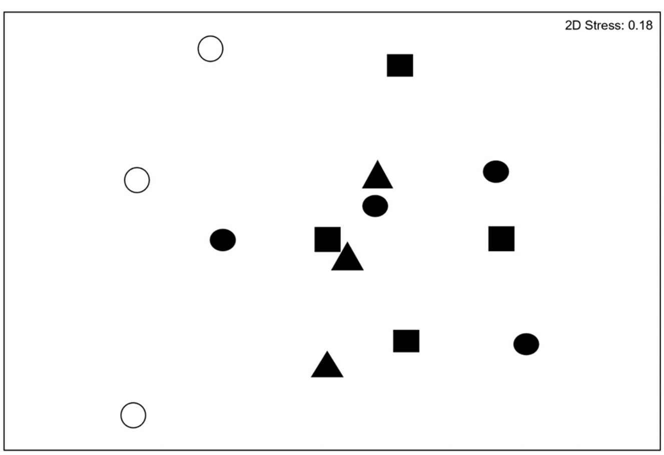

Fig. 4. MDS-ordination of the colonized communities of the experimental boxes in Repparfjorden. 2016. ○ = control. ▲ = 6 mm tailings. ● = 10 mm tailings. ■ = 14.5 mm tailings.

Table 3
Summary of results of ANOSIM (Analysis Of Similarity) for pairwise test between the treatments of the experimental boxes, Repparfjorden 2016. The sample statistics (global R) was 0.006. Significant result in **bold** (*p < 0.05).

Groups compared	Statistical value (*R*)	Significance level (*p*)
0 vs. 6	0.148	0.400
0 vs. 10	0.407	0.057
0 vs. 14.5	0.630	**0.029***
6 vs. 10	−0.185	0.829
6 vs. 14.5	−0.574	1.000
10 vs. 14.5	−0.240	0.886

3.2.2. *In situ community data*

The communities were species-rich, with a mean species richness per station (0.1 m^2) ranging from 48 (08_A, within threshold) to 135 (10_E, outside threshold) (Table 5). The abundance ranged from 140 (08_A, within threshold) to 990 (10_J, outside threshold) per 0.1 m^2. The Shannon-Wiener diversity (H′) varied from 3.11 (10_J, outside threshold) to 5.83 (10_E, outside threshold) per 0.1 m^2.

In total, 342 taxa were recorded in the field sampling through the three sampling campaigns. In general, annelids were by far the dominating group (see list of dominating taxa in Table 6). Of these, the tube-building annelid *Galathowenia oculata* was the numerically most abundant species, constituting over 50% of the total abundance at most (at station 10_F). Also, the tube-building annelid *Maldane sarsi* was highly abundant, with up to 40% of the individuals (at station 15_N). Bivalves were also relatively abundant. Notably, at stations 08_A and 15_O located inside the threshold, the bivalve *Macoma calcarea* was the most dominating species, constituting 12.1 and 18.8% of the individuals, respectively.

In the MDS-plot of the field stations (Fig. 5), the stations inside and outside the threshold were separated from each other. In addition, stations 15_O, 15_K, and 08_A, all inside the threshold and with high levels of copper (Cu) (Table 4), were separated from the other stations. Thus, there was a gradient from the stations with high Cu-levels, to the other stations inside the threshold, and to the stations outside.

In the marginal test in DistLM, i.e. where each environmental variable was tested separately as to how well it described the multivariate faunal patterns, Cu emerged as the factor that alone explained most the faunal variation, and the only factor being statistically significant in explaining faunal variation ($p = 0.0001$) (Table 7). In the sequential test, i.e. where the environmental variables were tested in combination, Cu was the most important factor (explaining nearly 30% of the faunal variation), followed by TOC (12%) and Pb (11%).

In the db-RDA-plot, the stations were arranged along an axis corresponding to Cu-content and a perpendicular axis representing Pb and TOC, which were negatively correlated with each other (Fig. 6a). In particular, station 08_A, 15_K and 15_O were separated from the others, like in the MDS plotAltogether, the first two dbRDA-axes explained 66% of the fitted variation, and 45% of the total variation of the multivariate community data. All of the RDA-axes together explained 68% of the total variation. The annelids *Chaetozone/Aphelochaeta* sp., *Capitella capitata* and *Phyllodoce maculata* and bivalves *Mya* sp. juv and *Macoma calcarea* were positively correlated to the Cu-vector, while e.g. the annelids *Galathowenia oculata* and *Nothria hyperborea* and the sea cucumber *Labidplax buskii* were negatively correlated with it (Fig. 6b).

4. Discussion

4.1. *Sediment characterization*

In the experimental boxes, the mine tailings affected grain size regardless of the amount of mine tailings placed in the boxes, with a few exceptions (box 4–6 mm, box 3–10 mm and box 4–14.5 mm) (Table 1). In the boxes with 6 mm mine tailings added, elevated concentrations of metals were not measured in the sediment samples (0–5 cm) (Table 1). Given the low quantity of tailings (6 mm) compared to natural sediment (44 mm) in the analysed sample, these results are not unexpected. In this treatment factors such as inhomogeneity of the tailings characteristics/mineral composition, dissolution of minerals (e.g. carbonates) (Dold, 2014), or desorption of metals from the tailings (Ramirez-Llodra et al., 2015) will affect the results to a larger degree than for other treatments. In addition, the low level of tailings added may have resulted in a relative higher influence of sediment dynamics during the experimental period (sedimentation and resuspension), diluting the mine tailings. In the boxes with 10 and 14.5 mm, the tailings significantly affected both the grain size and the metal concentrations (section 3.1.1.). In the boxes with 14.5 mm, there also appears to have been loss of some metals, i.e.

Table 4
Sediment characteristics and metal levels of the field samples, Repparfjorden 2008–2015 (data common for all years). PEL denotes Predicted Effects Level. Background concentrations taken from Sternal et al. (2017).

	Depth	% <63 μm	TOC	As	Co	Cr	Cu	Fe	Mn	Ni	Pb	Zn
	(m)	(%)	g/kg	mg/kg dw								
Within threshold												
08_A	30	86.7	12.0	5.0	5.9	58	291	13,200	110	23	5.0	29
08_B	50	91.5	13.0	5.0	7.2	61	136	16,900	138	25	5.0	37
08_C	65	95.7	11.0	5.0	5.5	43	88	14,100	123	20	5.0	35
10_I	63	75.0	13.5	5.0	5.2	38	64	13,000	122	17	7.3	29
15_K	57	87.6	12.6	4.8	<0.02	60	188	15,740	127	22	45	27
15_M	64	80.2	13.9	4.0	<0.02	50	86	15,500	131	19	42	25
15_O	36	82.4	13.7	1.8	5.2	53	178	13,040	117	18	15	29
Outside threshold												
08_D	60	99.1	5.8	5.0	2.8	20	13	8210	94	11	5.0	20
10_E	72	31.0	5.8	2.0	3.3	22	12	8650	113	10	5.4	19
10_F	120	32.0	7.3	1.0	3.3	23	11	8380	119	11	5.5	20
10_G	76	70.0	11.5	3.0	4.3	29	22	10,400	121	15	8.0	27
10_H	68	66.0	7.7	3.0	3.3	22	12	8580	109	11	5.7	20
10_J	87	79.0	16.0	3.0	5.1	36	40	12,200	126	17	9.2	33
15_N	90	57.4	7.7	1.4	3.0	22	21	7810	102	11	11.2	24
Background concentrations min				2.2	3.1	17	6.2	n.a.	n.a.	9.7	2.7	14
Background concentrations max				3.0	4.2	23	11	n.a.	n.a.	13	4.4	21
Norwegian PEL for sediment				18		620	84			42	150	139

Table 5
Overview of number of species (S), abundance (N) and Shannon-Wiener diversity (H′) in the field samples from Repparfjorden (average pr. 0.1 m^2). Station number denotes sampling year (i.e. 2008, 2010 and 2015).

	No. species	Abundance	H′
Inside threshold			
08_A	48	140	4.37
08_B	50	415	3.63
08_C	55	525	3.25
10_I	81	586	3.60
15_K	54	154	4.63
15_M	74	480	4.42
15_O	73	286	4.59
Outside threshold			
08_D	101	285	4.72
10_E	135	245	5.83
10_F	120	666	3.48
10_G	105	853	3.78
10_H	111	696	4.04
10_J	88	990	3.11
15_N	95	470	3.82

the relative effect of grain size was higher than the effect on content of Ba, Ca and Mn (section 3.1.1). The effect of mine tailings on the grain size and metal concentrations indicate that the boxes with 10 and 14.5 mm mine tailings were not heavily affected by new sedimentation during the 15 month-duration of the experiment.

In the field study, surface sediments in the inner part of the fjord showed higher concentrations of metals and TOC, and a higher fine fraction than in the outer part of the fjord (outside the threshold). The higher content of TOC (1.1–1.4%) in sediments affected by submarine mine tailings disposal, compared with the TOC content in mine tailings (<0.5%), is due to natural sedimentation over the 50 years without deposition (e.g. from riverine input, etc.). Despite this, the concentrations of As, Cr, Fe, Mn, Ni and Zn were elevated in the disposal area, with up to two times the background concentration in the fjord. The concentrations of Cu were up to 47 times higher than background concentration. Of the priority pollutant metals (As, Cr, Cu, Ni, Pb and Zn), only Cu exceeded the threshold concentration for predicted effects in the marine environment (Table 4). The predicted effects level (PEL) in marine sediments in the Norwegian environmental quality criteria is 84 mg/kg dw (Veileder 02:, 2018). Thus, these findings indicate that the surface sediments in the inner part of the fjord are still affected by the mine tailings approximately 50 years post closure, and that the affected zone reaches beyond the former deposit area, in line with previous findings (Sternal et al., 2017; Pedersen et al., 2017).

4.2. Colonized communities

Faunal communities were established in all boxes within the 15 months experiment, but the boxes with mine tailings showed lower colonization levels than the controls, evidenced by significantly lower species richness, abundance and biomass in at least one of the tailing treatments (Fig. 3). Also, in the MDS-ordination, the fauna was to a large extent grouped according to whether tailings had been added or not (Fig. 4). In ANOSIM (Table 3), there was an increasing R-value the level of tailings added, although only the 14.5 mm treatment was significantly different from the control. Here it is important to be aware that the species data were transformed prior to the test, which means that the differences in abundances are downscaled compared to the univariate tests. In total, the results show that although the main response of the tailings seems to be related to whether tailings were present or not, the response was to some extent dose-dependent.

Despite a reduction in species richness and individuals, the Shannon-Wiener diversity (H′) was not reduced. However, this index may not be sensitive in cases with a parallel reduction in number of species and abundance, because it is the distribution among the species which is most important for the performance of this index (Cao and Hawkins, 2005; Trannum et al., 2021).

Although the same species to a large extent dominated in all treatments, the abundances were in general lower in the tailing treatments than in the control. The annelid *Dipolydora* sp. was by far the most dominant taxon. Notably, it is characterized as tolerant in AMBI indicator metric (http://ambi.azti.es, Borja et al., 2000), but nevertheless had a lower abundance in the tailing-treatments. Other species which seemed to have a lower colonization success in the tailing-treatments, and in the highest doses in particular, were the bivalves *Macoma calcarea*, *Parvicardium pinnulatum* and *Mya* sp. juv. and the annelid *Chone* sp. *Macoma calcarea* is assigned generally tolerant in the Norwegian indices (Veileder 02:, 2018), but still was negatively affected. An interesting observation is that *Parvicardium pinnulatum, Chone* sp. and *Mya* sp. are all suspension feeders (NIVA's functional traits database; Oug et al., 2012) and not dependent on the tailings-contaminated sediment for food. Despite this, these species still did not tolerate well the condition in the tailing boxes.

The differences in faunal colonization observed between the controls and tailing treatments may be due to both pre- (larval settlement choice)

Table 6
Abundance (N) of ten (or eleven) most dominating taxa pr. station in the field samples from Repparfjorden (average pr. 0.1 m^2). Percentage (%) of total abundance is also presented. Station number denotes sampling year (i.e. 2008, 2010 and 2015). Taxonomic group in parenthesis, where A = Annelida, B=Bivalvia, C=Crustacea, Ca = Caudofoveata, E = Echinoderrmata, S=Sipunculida.

08_A (inside threshold)	N	%	08_B (inside threshold)	N	%	08_C (inside threshold)	N	%
Macoma calcarea (B)	17	12.1	*Galathowenia oculata* (A)	119	28.6	*Galathowenia oculata* (A)	232	44.2
Scoloplos armiger/acutus (A)	16	11.6	*Owenia polaris/fusiformis* (A)	64	15.5	*Ennucula tenuis* (B)	59	11.2
Ophiuroidea indet. Juv. (E)	16	11.3	*Macoma calcarea* (B)	47	11.3	*Macoma calcarea* (B)	48	9.2
Mya sp. juvenil (B)	12	8.8	*Ennucula tenuis* (B)	32	7.8	*Scoletoma* sp. (A)	35	6.7
Ennucula tenuis (B)	12	8.2	*Scoletoma* sp. (A)	30	7.2	*Owenia polaris/fusiformis* (A)	23	4.3
Nuculana pernula/minuta (B)	11	7.9	*Nuculana pernula/minuta* (B)	29	6.9	*Nuculana pernula/minuta* (B)	15	2.9
Euchone sp. (A)	7	4.8	*Euchone* sp. (A)	13	3.1	*Yoldiella lenticula* (B)	13	2.5
Chaetozone/Aphelochaeta sp. (A)	7	4.6	*Scoloplos armiger/acutus* (A)	9	2.1	*Euchone* sp. (A)	12	2.3
Eteone flava/longa (A)	5	3.6	*Pectinaria hyperborea* (A)	8	1.8	*Thyasira gouldi* (B)	10	2.0
Spio sp. (A)	3	2.1	Ophiuroidea indet. Juv. (E)	7	1.7	*Mya* sp. juvenil (B)	10	1.9
			Yoldiella lenticula (B)	7	1.7			
10_I (inside threshold)	N	%	15_K (inside threshold)	N	%	15_M (inside threshold)	N	%
Euchone sp. (A)	173	29.6	*Galathowenia oculata* (A)	27	17.7	*Galathowenia oculata* (A)	95	19.8
Galathowenia oculata (A)	152	25.9	*Sipunculida indet.*	18	11.5	*Euchone* sp. (A)	80	16.7
Owenia polaris/fusiformis (A)	65	11.0	*Chaetozone/Aphelochaeta* sp. (A)	13	8.2	*Ennucula tenuis* (B)	30	6.3
Pectinaria hyperborea (A)	25	4.3	*Macoma calcarea* (B)	10	6.7	*Lumbrineris mixochaeta* (A)	28	5.9
Yoldiella lenticula (B)	22	3.7	*Eteone flava/longa* (A)	8	5.4	*Owenia polaris/fusiformis* (A)	26	5.5
Scoletoma sp. (A)	15	2.5	*Lumbrineris mixochaeta* (A)	8	5.2	*Macoma calcarea* (B)	25	5.3
Spio sp. (A)	11	1.9	*Ennucula tenuis* (B)	5	3.0	*Yoldiella lenticula* (B)	20	4.1
Nuculana pernula/minuta (B)	10	1.7	*Pholoe assimilis* (A)	4	2.8	*Thyasira* sp./*sarsi* (B)	14	2.9
Ennucula tenuis (B)	9	1.5	*Maldane sarsi* (A)	4	2.4	Sipunculida indet. (S)	12	2.6
Yoldia amygdalea/hyperborea (B)	9	1.5	*Mya* sp. juvenil (B)	4	2.4	*Maldane sarsi* (A)	12	2.4
15_O (inside threshold)	N	%	08_D (outside threshold)	N	%	10_E (outside threshold)	N	%
Macoma calcarea (B)	53	18.5	*Galathowenia oculata* (A)	89	31.1	*Galathowenia oculata* (A)	21	8.7
Ennucula tenuis (B)	29	10.1	*Maldane sarsi* (A)	20	7.1	*Petaloproctus tenuis* (A)	16	6.6
Dipolydora sp. (A)	25	8.8	*Yoldiella lucida* (B)	11	3.9	*Ennucula tenuis* (B)	15	5.9
Mya sp. juvenil (B)	20	7.0	*Rhodine gracilior* (A)	9	3.2	*Nuculana pernula/minuta* (B)	14	5.5
Nuculana sp. juvenil (B)	16	5.5	*Prionospio cirrifera* (A)	9	3.2	*Maldane sarsi* (A)	12	4.9
Euchone sp. (A)	15	5.3	*Yoldiella solidula* (B)	9	3.1	Golfingiidae indet. (S)	10	3.9
Thyasira sp./*sarsi* (B)	15	5.3	*Scoletoma* sp. (A)	8	2.9	*Labidoplax buskii* (E)	9	3.8
Scoloplos armiger/acutus (A)	10	3.3	*Goniada maculata* (A)	7	2.3	*Chone* sp. (A)	9	3.5
Nuculana pernula/minuta (B)	9	3.2	*Dipolydora* sp. (A)	6	2.2	*Nothria hyperborea* (A)	8	3.2
Chaetozone/Aphelochaeta sp. (A)	9	3.0	*Ennucula tenuis* (B)	6	2.2	Ophiuroidea indet. Juv. (E)	6	2.4
			Thyasira sp./*sarsi* (B)	6	2.2			
10_F (outside threshold)	N	%	10_G (outside threshold)	N	%	10_H (outside threshold)	N	%
Galathowenia oculata (A)	361	54.2	*Galathowenia oculata* (A)	221	25.9	*Galathowenia oculata* (A)	189	27.2
Maldane sarsi (A)	44	6.6	*Maldane sarsi* (A)	219	25.7	*Maldane sarsi* (A)	163	23.4
Thyasira sp./*sarsi* (B)	20	3.0	*Owenia polaris/fusiformis* (A)	110	12.9	*Owenia polaris/fusiformis* (A)	63	9.1
Owenia polaris/fusiformis (A)	18	2.7	*Myriochele heeri* (A)	33	3.9	*Myriochele heeri* (A)	30	4.3
Yoldiella lucida (B)	18	2.7	*Scoletoma* sp. (A)	24	2.8	*Yoldiella solidula* (B)	25	3.6
Euclymeninae indet. (A)	16	2.3	*Yoldiella solidula* (B)	24	2.8	*Yoldiella lenticula* (B)	18	2.6
Yoldiella solidula (B)	15	2.2	*Yoldiella lenticula* (B)	19	2.3	*Thyasira* sp./*sarsi* (B)	15	2.2
Scoletoma sp. (A)	14	2.1	*Thyasira* sp./*sarsi* (B)	18	2.1	*Crenella decussata* (B)	14	2.0
Diastylis sp. (C)	11	1.7	*Crenella decussata* (B)	16	1.8	*Ennucula tenuis* (B)	12	1.7
Thyasira dunbari (B)	8	1.1	*Yoldiella lucida* (B)	14	1.7	*Yoldiella lucida* (B)	11	1.5
10_J (outside threshold)	N	%	15_N (outside threshold)	N	%			
Maldane sarsi (A)	346	34.9	*Maldane sarsi* (A)	192	40.8			
Galathowenia oculata (A)	236	23.8	*Galathowenia oculata* (A)	63	13.5			
Owenia polaris/fusiformis (A)	166	16.8	*Owenia polaris/fusiformis* (A)	37	7.8			
Myriochele heeri (A)	68	6.9	*Lumbrineris mixochaeta* (A)	22	4.7			
Scoletoma sp. (A)	21	2.1	*Myriochele heeri* (A)	14	2.9			
Pectinaria hyperborea (A)	10	1.0	*Pseudopotamilla reniformis* (A)	10	2.1			
Scoloplos armiger/acutus (A)	10	1.0	Caudofoveata indet. (Ca)	9	1.8			
Ennucula tenuis (B)	8	0.8	*Mendicula* sp. (B)	8	1.7			
Yoldiella lenticula (B)	8	0.8	*Nicomache lumbricalis* (A)	8	1.6			
Macoma calcarea (B)	7	0.7	*Yoldiella lenticula* (B)	6	1.2			

and post-settlement (variable growth rates, mortality) processes. From the present setup, it is not possible to determine which mechanisms were more important. Nevertheless, regardless of underlying mechanisms, the mine tailings treatments were associated with a lower colonization success (Table 2 and Fig. 3).

With regard to larval recruitment, it seems reasonable that the effect was partly related to the presence of tailings since it will be the surface sediment the larvae first encounter when they select their habitat. "Recruitment patterns of marine invertebrates are strongly influenced by habitat preferences as larvae can choose to settle (or not) in response to positive or negative cues" (Gribben et al., 2009, p. 217). Regarding post settlement processes on the other hand, it is reasonable that the layer thickness also played a role. |In line with such assumption, a mesocosm experiment with adult communities documented a highly dose-dependent response to different tailings (Trannum et al., 2018). In addition, from a field study, Olsgard and Hasle (1993) concluded that sedimentation of mine tailings with elevated levels of e.g. Cu, clearly had an adverse effect on macrofauna on a scale of cm per year, while

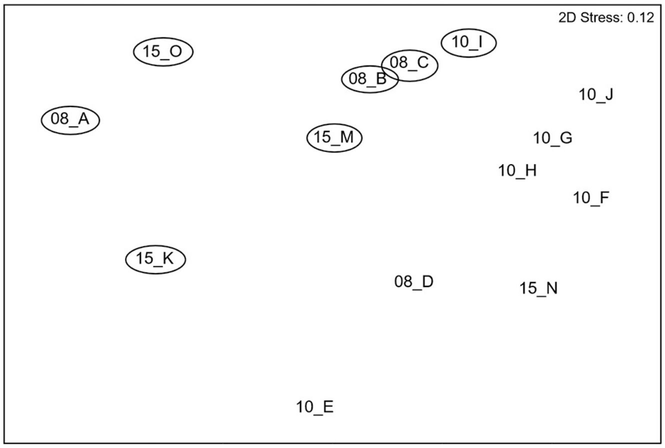

Fig. 5. MDS-ordination of the faunal communities in Repparfjorden (2008–2015). Stations inside the threshold are marked with a circle. Number denotes sampling year (i.e. 2008. 2010 and 2015).

Table 7
Results of the DistLM-model of Repparfjorden field community, including Sum of squares (SS), Pseudo-F statistic, p-value and proportional and cumulative explained total variance. Marginal tests show how much variation each variable (As, Cu, Mn, Pb, TOC and % < 63 μm) explains of the multivariate faunal composition when considered alone, ignoring other variables. Sequential tests explain the cumulative variation attributed to each variable fitted to the model in the order specified, taking previous variables into account (*$p < 0.05$. *** $< p < 0.001$). Significant values in **bold.**

Marginal test					
Variable	SS_{trace}	Pseudo-F	p	Prop.	
As	2703	1.230	0.256	0.093	
Cu	**8367**	**4.847**	**0.0001*****	**0.288**	
Mn	2449	1.104	0.339	0.084	
Pb	3821	1.815	0.072	0.131	
TOC	3829	1.819	0.079	0.132	
% < 63 μm	2987	1.373	0.193	0.103	
Sequential test					
Variable	SS_{trace}	Pseudo-F	p	Prop.	Cumul.
Cu	**8367**	**4.847**	**0.0001*****	**0.288**	**0.288**
TOC	**3379**	**2.144**	**0.028***	**0.116**	**0.404**
Pb	**3313**	**2.362**	**0.010***	**0.114**	**0.518**
As	1689	1.232	0.287	0.058	0.578
% < 63 μm	1600	1.192	0.321	0.055	0.631
Mn	1512	1.148	0.351	0.052	0.683

sedimentation in the scale of 1 mm pr. year did not lead to observable effects.

There was a large difference in grain size between the original sediment and the tailings, which were far more fine-grained (Table 1), which might explain some of the faunal differences. Sediment characteristics, and grain size in particular, are recognized as highly important factors for structuring soft bottom communities (e.g. Gray and Elliott, 2009). Other colonization experiments have shown that sediments with different grain sizes lead to different colonization patterns (Trannum et al., 2011; Kanaya, 2014). In this context, it is worth looking at the results from a mesocosm-experiment that was conducted with Nussir-tailings (relatively similar composition as in the present experiment) on intact, benthic communities from the Oslofjord (Ramirez-Llodra et al., 2022). The grain size of the tailings was relatively similar to the natural sediment, which was very fine grained. Still, there was a lower species number and abundance in the tailings treatment (2 cm tailings) compared to the control, which was interpreted as a toxic response. That finding may thus indicate that features of the tailings, other than only grain size, were involved for the response observed in the present experiment as well.

Tailings consist of mechanically crushed rock, and can often be more sharp-edged than natural sediments, which are rounded due to natural grinding over geological time (Kvassnes and Iversen, 2013). Such sharp particles have been assumed to pose an additional risk for the fauna in the context of mining (Olsgard and Hasle, 1993; Trannum et al., 2018; Sweetman et al., 2020). The Nussir tailings have been characterized as slightly edged to rounded in shape (Kleiv, 2011), so it is unlikely that the changes in benthic fauna in the experimental boxes are related to sharp-edged particles to a large extent.

Tailings are also typically low in content of organic matter, which may decrease the sediment's nutritional value (Blackwood and Edinger, 2007; Burd et al., 2000; Shimmield et al., 2010; Trannum et al., 2018). In the present study, all box sediments had a very low TOC-concentration (see Supplementary Material 3), which may explain why the abundances were lower within the boxes than at the field stations. From the abundance and biomass patterns (Fig. 2), it appears that the biomass was reduced relatively more than abundance in the tailings treatments, at least for the 6 and 14.5 mm treatments. This might indicate that the organisms were smaller, perhaps due to an even lower nutrient content in the tailings treatments than the control, or due to other factors inhibiting food intake or growth. Thus, it cannot be excluded that there is a response related to nutrient content, although the data are not strong enough to support this.

Lastly, toxicity caused by metals from the ore may have contributed to the effects. The tailings were elevated in Cu, and several of the tailings boxes had high Cu-concentrations, up to almost 400 mg/kg, even 15 months after initiation of the experiment. Several of the boxes with tailings exceeded the maximum admissible concentration in Norway (84 mg/kg dw; Veileder 02, 2018). Although not all the tailings boxes had concentrations exceeding this value, it is important to be aware that metal levels were analysed at the end of the 15 month experiment. The tailings then seemed to be patchily distributed, filling up small pits between the coarser particles, and can also have been transferred deeper than the upper 5 cm which were sampled. Several studies have demonstrated that Cu can affect benthic communities (e.g. Rygg, 1985; Stark, 1998; Olsgard, 1999; Trannum et al., 2004; Neira et al., 2011), including in the context of mining (Burd, 2002; Koski et al., 2008). The responses include effects on e.g. fertilization, settlement-success, survival of juvenile stages of benthic species, as well as an altered species composition, reduced biodiversity, altered lifestyles, and changes in biomass and abundance (Reichelt-Brushett and Harrison, 1999; Olsgard, 1999; Reichelt-Brushett and Harrison, 2000; Trannum et al., 2004; Neira et al., 2011; Black et al., 2015), in line with responses here. Cu has also been found to have more severe effects on the benthic fauna in coarse than fine sediment, probably due to a larger fraction of the free Cu-ion in the porewater (Pesch, 1979; Tietjen, 1980; Austen et al., 1994; Trannum et al., 2004; Campana et al., 2012), which again accords well with the profound response in the present experiment. Lastly, Cu is also a known toxin for microalgae (Purbonegoro et al., 2018), which is a typical food source for benthic organisms in shallow marine sediments.

In conclusion, it is likely that Cu influenced the faunal composition within the boxes, perhaps in combination with sediment properties like smaller grain size and lower organic matter content. The response was to some extent dose-dependent, and seemed to occur from the lowest thickness of 6 mm. In the planned deposit, an area of 4.9 km^2 is predicted to have a tailings layer exceeding 1 cm (Didriksen and Wilersrud, 2010), and effects on benthic macrofauna in this zone seem likely when the mine is reopened.

4.3. Field communities

The in situ faunal communities were relatively species-rich, with a mean species richness ranging from 48 to as much as 135 pr. 0.1 m^2. According to the Norwegian classification, all stations had a condition within class "good" based on the Shannon-Wiener index (Veileder 02:,

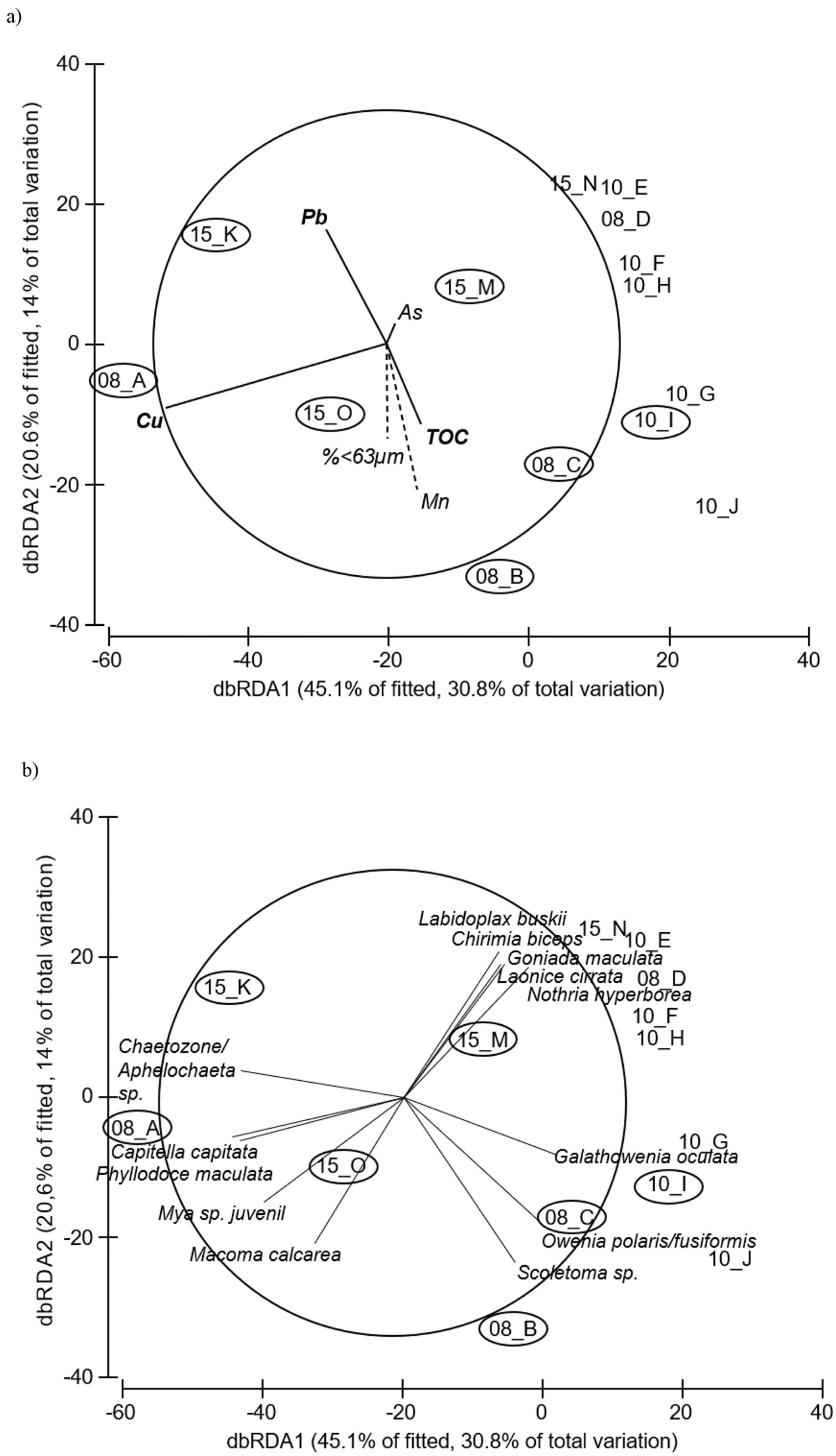

Fig. 6. dbRDA-plot of the field communities in Repparfjorden (2008–2015), with environmental variables as vectors (a) and species as vectors (b). The length and direction of the vectors represent the strength and direction of the relationship. Significant factors in DistLM (forward selection) are typed in bold and with solid lines (Cu, Pb and TOC). Species selected by Pearson correlation >0.7. Stations inside threshold are marked with a circle. Number denotes sampling year (i.e. 2008, 2010 and 2015).

2018). Even though the fauna was not severely disturbed, Cu was identified as the most important factor driving faunal composition, and was highly significant in both the DistLM sequential and marginal test (Table 7). Although this points to a likely effect of Cu, it is important to be aware that this relationship is only based on correlations between measured environmental variables and the fauna, andthat Cu may represent other tailings-associated variables correlated to it, e.g. Mn, Ba and Ca (see section 3.2.1). As in the boxes, the Cu-concentration was high and exceeded EQS-value (84 mg/kg dw) at all but one of the stations inside the threshold. The highest concentrations were found at station 08_A with 291 mg Cu/kg dw, followed by 15_K and 15_0 with concentrations of 188 and 178 mg/kg dw, respectively. These three stations were plotted at the periphery of the other stations in the MDS-plot (far left in Fig. 5) and in the db-RDA-plot, where they were aligned along the Cu-gradient (Fig. 6). Another notable finding was that station 08_A, with the highest Cu-level (Supplementary Material 3), showed the lowest species number and abundance of all sampled stations. We therefore conclude that Cu content in legacy tailings deposits is negatively impacting benthic fauna, likely for reasons summarized above (in chapter 4.2).

Although the fauna was diverse, the faunal composition seemed to reflect that there was some degree of disturbance. The dbRDA-plot (Fig. 6b) pointed out that typically opportunistic and tolerant species like *Capitella capitata* and *Chaetozone/Aphelochaeta* sp. (Pearson and Rosenberg, 1978) were correlated to the Cu-gradient. This was also evident for the bivalve *Macoma calcarea*, which dominated at stations 08_A and 15_O. As mentioned above, this species is assigned tolerant, but its abundance was reduced in the boxes with the highest tailings-doses in the colonization experiment (chapter 4.2), maybe pointing to an ability to survive in moderately tailings-affected sediments like in the field-impacted stations. Although not visualized in the dbRDA-plot, it should be noted that at station 08_A, the annelid *Scoloplos armiger/acutus* was the second most dominating species, and also this species is assumed tolerant in AMBI (http://ambi.azti.es, Borja et al., 2000) and in the Norwegian classification (Veileder 02:, 2018). At station 15_O, there was a quite high abundance of *Dipolydora* sp., which is a tolerant genus as described in chapter 4.2. The dbRDA-plot also pointed to species that were negatively correlated with the Cu-vector, like several tube-building annelids (*Galathowenia oculata*, *Owenia polaris/fusiformsis*, *Nothria hyperborea* and *Chirimia biceps*). Such species may exhibit sediment-specific preferences during settlement (Pinedo et al., 2000; Duchêne, 2010), which may explain why they appear to be more sensitive than free-living annelids (Oug et al., 2012; Pearson and Rosenberg, 1978) Also previous studies have observed reduced abundance of tube-building annelids in tailings-impacted sediments (Trannum et al., 2019; Trannum et al., 2020). Thus, to conclude, the species patterns indicate that there still is a disturbance factor in the fjord today.

The assumption that Cu affects the benthic fauna in Repparfjorden agrees with Sternal et al. (2017), who concluded that the buried contaminated sediments have exposed the fauna to Cu for over four decades. According to Pedersen et al. (2017), approximately 80–390 tons of Cu still remain in the disposal area from the surface to a sediment depth of 16 cm. Furthermore, metal partitioning showed that 56–95% of the Cu is bound in the potential bioavailable fractions in the sediments, with the potential for continuous release to the pore water and re-precipitation in over- and under-lying sediments. A role of Cu in structuring the benthic community, is also in line with a study in Jøssingfjorden in the south of Norway, another fjord where Cu-containing tailings were deposited several decades ago, and where Cu has been identified as an important structuring variable for the fauna (Schaanning et al., 2019).

This study finds, that initial faunal colonization in an Arctic fjord is fast, but tailings may still impact the fauna for up to several decades after mining has ceased, just as has been observed in more temperate locations (Ellis and Hoover, 1990; Burd, 2002; Schaanning et al., 2019; Ramirez-Llodra et al., 2022).

4.4. Limitations of the study

The surface of the sediments in the experimental boxes was positioned above the seabed, which may have affected the bottom-flow and larval recruitment dynamics. Thus, it cannot be excluded that box communities experienced a slightly different colonization pattern compared to natural sediments. Nevertheless, such effects will be similar for all boxes, including controls. The same setup has been used in previous experiments, where species-rich communities were established within some months (Olsgard, 1999; Trannum et al., 2004; Trannum et al., 2011; Trannum et al., 2020).

The species number and abundances were generally low for the colonized communities, and the communities did not yet seem to have reached maturation. This suggests that it takes more than one year to establish mature communities in this fjord. Such a finding seems to contrast with several colonization experiments carried out in the Oslofjord, where colonized communities without any additions were richer in species compared to natural communities after a period of 6–12 months (Trannum et al., 2004, 2011, 2020). It is likely that colonization takes longer time in northern fjords characterized by higher seasonality in e.g. temperature, chlorophyll and spawning, and thus a more pulsed recruitment (Michelsen et al., 2017a; b). Several of the organisms in the frames were juveniles (Table 2), indeed pointing to slow colonization processes. Regardless, it is important to be aware that the frames were placed at a shallower depth than the fjord stations due to lack of a proper flat area in the deeper parts, so direct comparisons cannot be made. In addition, the sediment was much coarser within the boxes than at the fjord stations, which may influence larval settlement cues.

5. Conclusions and recommendations

Both the field and the experimental colonization study indicated impacts of tailings on benthic macrofaunal communities, with Cu-concentrations a likely candidate for the proximate causative factor. The persistence of modest, but measurable, impacts as long as 40 years after cessation of STD reflects both the likely toxic effect of tailings and the limited circulation in the deposit area.

As high metal-concentrations prevail in the sediment for several decades after deposition, and because this and several other studies point to long-lasting effects on community composition, capping with marine clay or other suitable materials is recommended when mining ceases to reduce the time required for natural burial of tailings beyond the depth of bioturbation. This has also been recommended in other studies (Schaanning et al., 2019; Sweetman et al., 2020; Ramirez-Llodra et al., 2022).

In future studies and environmental monitoring, it is important to not solely base the assessment of effects on the biodiversity indices, which may hide reductions in species richness and abundance, as also pointed out in Trannum et al. (2021). Further improvements could be to include functional responses, in line with both general (Elliott and Quintino, 2007; Rand et al., 2018) and specific recommendations for STDs (Trannum et al., 2019; Ramirez-Llodra et al., 2022). Indeed, in an experimental setup, Mevenkamp et al. (2017) showed significant effects on benthic functional processes from only 1 mm tailings, i.e. far lower than the level assumed to initiate effects on the benthic structure, which clearly shows the importance of including also a functional approach.

Tailings vary according to their physical and chemical properties, and each deposit area will have unique characteristics like hydrodynamic conditions, geochemical variables, faunal composition and productivity (Ramirez-Llodra et al., 2022). Although the monitoring should follow general guidelines, it also needs to be fine-tuned in each single case, where expertise both from the research communities, consultancies and the industry is needed.

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.

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

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Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.seares.2022.102327.

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