Comparative life cycle assessment of tailings management and energy scenarios for a copper ore mine: A case study in Northern Norway

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Abstract

In support of continuous environmental improvement in the mining industry, it is important to systematically assess the environmental impacts of mining and mineral processing operations from a life cycle perspective. Although life cycle assessment (LCA) is widely used as an environmental systems analysis tool, the application of LCA in the mining industry is still in progress. This paper carried out a cradle-to-gate LCA of an underground copper ore mine planned in Northern Norway. Based on the ReCiPe midpoint (hierarchist) life cycle impact assessment method, results of the study showed that on-site electricity use, diesel for mining trucks and blasting dominated contributions across six, four and four, respectively, of the eighteen categories assessed, and metals leaching from tailings were the primary contributors to the human toxicity and marine ecotoxicity impacts. Compared to the baseline, results of the energy-oriented scenario analysis indicated that electrification of diesel-driven mining trucks would be more environmentally beneficial as long as the electrical supply is "relatively clean" across impact categories. While electrodialytic tailings remediation could extract up to 64% of copper in tailings prior to disposal and significantly reduce the human toxicity impact of tailings, the marine ecotoxicity impact of tailings after electrodialysis changed inconsistently across the ReCiPe hierarchist and egalitarian perspectives. It is recommended to further assess the trade-off between the benefits of electrodialytic tailings remediation (extracting more copper) and the potential impacts of deposited tailings after electrodialysis from a multi-criteria decision-analysis perspective. In a generic context, this study provides an insight in further promoting LCA as an environmental decision-support tool, especially for comparing available cleaner production options, improving the overall environmental performance of a mine, and facilitating better communication with stakeholders.

Keywords:

Life cycle assessment; copper mining; energy; tailings management; electrokinetic remediation

1. Introduction

There has been a growing expectation on the mining and mineral processing industries to operate in a more responsible and sustainable manner, particularly on reducing the environmental impacts and improving resource management. The International Council on Mining and Metals (ICMM) has claimed that mining will be needed to meet the growing demand for minerals and metals, “even with society achieving greater efficiencies through reduction of extraneous uses, reuse, and recycling” (ICMM, 2012). The World Economic Forum (2014) has called for paying more attention to the environment aspects of the mining and mineral sector, due to (i) stricter environmental standards for greenhouse gas (GHG) emissions, energy and water consumption, waste management and biodiversity, and (ii) adaptation of mining operations to changing climate conditions. Regarding future mining challenges, the implications of declining ore grades, cradle-to-cradle management of all materials, and the inevitable shift from surface to massive underground mining have been emphasized (Moran et al. 2014). To support cleaner production and environmentally friendly decision-making in the mining industry, it is crucial to systematically assess resource-, energy- and tailings-related impacts from a life cycle perspective.

Life cycle assessment (LCA) is an internationally standardized method for assessing the potential environmental impacts associated with the whole life cycle of a product or service (ISO, 2006). In general, LCA results provide a more holistic understanding of the overall impacts of a mine and can contribute to continuous corporate environmental improvement. The last revised ISO 14001:2015, one of the world’s most widely used environmental management standards, re-emphasized the importance of employing a life cycle perspective to better address emerging challenges of corporate environmental management with respect to sustainable resource use, energy & water use, climate change mitigation, and stakeholder-focused communication (ISO, 2015). Without LCA, environmental improvement measures of a mine may be ad-hoc and suboptimal (Awuah-Offei and Adekpedjou, 2011). However, the application of LCA in the mining and mineral sector is still in progress, partly due to a lack of publicly available operational data suitable for use in LCA (Durucan et al., 2006). In particular, there are very few published mining LCA studies with a systematic examination of the overall plant-level environmental impacts of mine operations, including mining, mineral processing and tailings disposal.

The ecological sustainability challenges faced by the mining industry, to a large extent, relate to environmental management at the corporate level (Botin, 2009). Owing to declining ore grades on average, increasing copper demand and climate change concerns, mitigation of energy-related environmental impacts is becoming more important in the energy-intensive mining industry. This holds true even for mining operations in remote Arctic regions, such as Northern Norway and Greenland, where there is a clear trend of an upsurge in mining activities in recent years (van Dam et al., 2014). In fact, the Arctic has experienced the greatest regional warming on earth since the 1950s, with an average annual temperature increase by 2-3 ^{o}C and in winter by up to 4 ^{o}C (Huntington et al., 2005). In support of reducing CO_2 emissions from copper production in Europe, the European Copper Institute has suggested four strategies in relation to energy efficiency, the use of renewable energy sources, appropriate technologies for mitigation, and electrification of equipment and transportation (ECI, 2014). Previous mining LCA studies in the literature have investigated the environmental impacts of energy-oriented scenarios, such as on comparing diesel-powered mining trucks with electric belt conveyors (Erkayaoğlu and Demirel, 2016), while most of them have not discussed in detail the relative contribution of alternative energy options to the overall environmental performance of a mine across impact categories.

Besides mitigation of energy-related environmental impacts, another (even more) important concern is tailings management at mine sites. Mine tailings, either stored on land or deposited in marine/riverine systems, may cause significant environmental problems. The high potential risk of mine tailings is largely due to heavy metals leaching from tailings storage facilities, related to acid mine drainage from conventional land-based tailings ponds and desorption from marine and riverine tailing placements. Although the mining and mineral extraction industry is of importance in society, however, leaching from tailings has been crudely defined in most mining LCA studies. There is not yet a widely accepted recommendation in the literature on how to define long-term leaching of metals from mine tailings in the life cycle inventory phase (Pettersen and Hertwich, 2008). The under-communicated potential environmental impacts of tailings in mining LCAs partly hinder the application of LCA in the mining industry, especially for supply of environmental information to support ecological sustainability-related communication among stakeholders.

In comparison with mine tailings disposal strategies of reuse, recycling and reprocessing (Edraki et al., 2014), we argue that a more proactive paradigm could be to extract more valuable metals from tailings before final disposal or re-use. One applicable method is electrodialytic remediation, which has been shown to extract up to 70% of metals present in mine tailings (Jensen et al., 2016). What remains unclear is whether there is a trade-off between mineral resource recovery (extracting more metals from tailings) and the environmental impacts of tailings after electrodialysis. To our knowledge, there is still no published LCA literature comparing the potential environmental impacts associated with direct tailings disposal and tailings after electrodialytic remediation.

In an attempt to address the above-mentioned gaps, this paper assessed the potential environmental impacts of an underground copper ore mine, located in northern Norway, planned to open in 2019. Firstly, environmental hotspots of the copper ore mine were identified at the plant level. Secondly, we compared the impacts of alternative energy options (diesel-driven vs. electric trucks, heavy fuel oil vs. natural gas) and tailings management scenarios (direct disposal vs. electrodialytic remediation prior to discharge), including their relative contributions to the overall impacts of the mine. Moreover, we employed sequential extraction to estimate the metal leaching potential of tailings and assessed the impacts of tailings from different ReCiPe perspectives. Results of this study could be used as a science-based foundation to aid in both internal discussions (e.g. on cleaner production measures and improving environmental management) and external communication with stakeholders and other copper mines (e.g. on benchmarking the impacts of mining operations) towards better environmental decision-making.

2. Application of LCA in mining and mineral processing

Since the 2000s, LCA has attracted considerable attention from the mining communities. As early as 2002, the Mining, Minerals and Sustainable Development (MMSD) Project report pointed out that “the mining and minerals industry has started to engage in the development of LCA as one element of a holistic approach to decision-making for sustainable development” (IIED, 2002). During the past years, efforts have been devoted to promoting the application of LCA in the mining and mineral sector. For example, Durucan et al. (2006) developed a mining life cycle model with an inventory database, enabling mining LCA studies to be conducted with vast amounts of operational data. Yellishetty et al. (2009) carried out a critical review of existing LCA methods in the minerals and metals sector, and discussed the methodological drawbacks in relation to abiotic resource depletion, land use impacts, open-loop recycling, and spatial and temporal differentiation in LCA. In a review of publications before 2010, Awuah-Offei and Adekpedjou (2011) found that there was limited mining application of LCA in the literature, partly due to a lack of LCA awareness in the mining industry. Recently, Santero and Hendry (2016) reported the progress on harmonization of LCA methodology for the metal and mining industry, with respect to system boundary, co-product and recycling allocation, and impact assessment categories.

So far, most published mining LCA studies focused on assessing the environmental impacts of mine operations and metal production, with varying goal and scope definitions as well as impact assessment categories. This can be seen, for instance, from copper-related mining LCA studies in the literature. Norgate et al. (2007) presented the cradle-to-gate life cycle impact assessment results of metal production (copper, nickel, aluminum, lead, zinc, steel, stainless steel and titanium) in Australia, focusing on global warming potential (GWP), acidification potential (AP), solid waste burden and gross energy requirement. In a study on energy and GHG impacts of mining and mineral processing operations in Australia, Norgate and Haque (2010) concluded that the largest contribution to GHG emissions was from crushing and grinding steps in the case of copper ore, which became loading and hauling for the mining and processing of iron ore and bauxite. Memary et al. (2012) examined the historical environmental impacts of five largest Australian copper mines (1940-2008) using three environmental indicators (GWP, AP and photochemical ozone creation formation potential), and suggested a broader use of LCA for assessing future mining technology and energy options. Based on data from companies' financial and sustainability reports, Northey et al. (2013) assessed the environmental footprint (on energy, GHG and water) of copper production in 11 countries (in America, Asia and Oceania), and recommended that mining company sustainability reports should clarify fuels used by vehicles, heat or electrical energy sources, and boundaries of mining operations. Morevoer, Haque and Norgate (2014) assessed the GHG footprint of in-situ leaching of uranium, gold and copper in Australia and discussed opportunities to reduce GHG emissions.

Only a few published mining LCAs investigated the environmental impacts of transport (on- and off-road vehicles and machinery) at mine sites. In a comparative LCA of belt conveyor and truck haulage systems for a hypothetical open pit hard rock gold mine in Canada, Awuah-Offei et al. (2009) concluded that the contribution of belt conveyor was around 4 times in GWP and 1/4 in AP of that from the diesel truck option. Erkayaoğlu and Demirel (2016) compared the impacts of off-highway mining trucks and belt conveyors in Turkish surface mining and concluded that trucks resulted in higher GWP and lower AP than belt conveyors. At the time of writing, we found only one LCA study on energy options of mining trucks (Demirel and Düzgün, 2007), which reported that electric drive trucks had a higher impact of acid rain precursors and a lower impact of GWP than mechanical drive trucks.

Environmental assessment of tailings remains a huge challenge for mining LCAs. This challenge is partly due to the difficulties in obtaining required inventory data for tailings LCA (Van Zyl, 2009), because of mining companies keeping site-specific mining data confidential (Durucan et al., 2006) or without detailed monitoring of the generated waste (Lèbre and Corder, 2015). In an LCA of Australian metal production, Norgate et al. (2007) excluded the human toxicity and ecotoxicity indicators, though they recommended including metal dispersion from tailings in future LCA studies. In a comparative LCA of mining residue management methods in Canada, Reid et al. (2009) pointed out limitations on life cycle inventory data quality and uncertainty on the results of human toxicity and aquatic ecotoxicity. In a recent LCA of sulfidic tailings from copper ore processing in Poland, Beylot and Villeneuve (2017) emphasized the need on defining long-term emissions of tailings for future copper mining LCAs, since copper production was proven to be the primary contributor to the life cycle toxicity and ecotoxicity impacts of many products, such as electric vehicles (Nordelöf et al., 2014).

To summarize, the energy- and tailings-related LCA studies have typically been stand-alone, without investigations of the relative contributions of energy and tailings scenarios to the overall impacts of a mine. Moreover, most of the published LCA studies on the impacts of mine operations and metal products have simplified or excluded the tailings disposal phase. To promote the application of LCA in mining in general and tailings management in specific, further methodological discussions and demonstration case studies are needed, especially for defining (and assessing) the long-term metal leaching potential of tailings. Those gaps form the basis for the objectives of the present study.

3. Material and methods

3.1 Case description

The case study is on an underground copper ore mine owned by a Norwegian copper company Nussir ASA, located in Kvalsund Municipality of Finnmark County, northern Norway. In January 2016, the Norwegian Environment Agency granted Nussir ASA a tailings discharge permit (NEA, 2016). The discharge permit allows Nussir ASA to deposit up to 5500 tons of thickened tailings (60% solid content) per day on the bottom of Repparfjorden over the planned operational period of 15 to 20 years. The permit allows an annual production of up to 50000 tons of copper sulfide concentrate, making it equivalent in size to other larger copper concentrate mines in Europe (Northey et al., 2013). The designed mineral processing plant has a capacity of up to 2 million tons of ore per year, with a copper recovery rate of 94-95.5% (Nussir ASA, 2014). Nussir ASA plans to excavate copper ore from the Nussir (about 1.15% copper, 18 ppm silver and 0.15 ppm gold) and Ulveryggen (around 0.8% copper) deposits in parallel.

The underground mining process includes drilling, blasting, loading, hauling, and primary crushing (Nussir ASA, 2014). The designed mineral processing system begins with a semi-autogenous (SAG) mill and a secondary ball mill, followed by froth flotation reactors, cleaner, thickener and filter. The final concentrate contains about 45% copper. In the froth flotation process, three flotation reagents will be used, i.e. SIPX (Sodium Isopropyl Xanthate), MIBC (Methyl Isobutyl Carbinol) and occasionally CMC (Carboxymethyl Cellulose). SIPX makes the copper minerals float, while MIBC and CMC make unwanted minerals sink. To adjust pH of the slurry, addition of burnt lime (CaO) may be used. In order to increase the sedimentation rate of tailings as well as to recover process water for reuse, the anionic flocculant Magnafloc 10 (Acrylamide) will be added to the outflow of flotation tailings (ca. 8% solid) transferred to thickener. The thickened tailings (ca. 60% solid) are then ready for discharge to Repparfjorden, comprising predominately quartz, feldspar, mica, calcite, and small amounts of copper and other metals. According to the granted tailings discharge permit, only SIPX (among the five reagents) is currently not allowed to use, unless the company provides more information on the environmental impacts of SIPX after use (NEA, 2016).

The estimated annual energy uses and sources are 100 GWh electricity, 100 TJ diesel, and 1 TJ heating oil (Nussir ASA, 2011). Electricity will be distributed from an existing 132 kV power line close to the plant site. Heavy-duty diesel trucks are initially planned for transporting ore from ore draw points to the primary crusher (Nussir ASA, 2014). Nussir ASA is in the process of evaluating electrification of transport by replacing diesel trucks with electric trucks at the mine site. For building heating source, the company may replace heating oil with natural gas (NEA, 2016).

3.2 Life Cycle Assessment of the Norwegian copper mining operations

As defined in ISO 14040:2006, an LCA study includes four phases: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA), and interpretation (ISO, 2006). The results of LCI are a compilation of the energy and material inputs and outputs with respect to the selected system boundaries. Based on the LCI results, the LCIA phase intends to characterize and evaluate the significance of the potential environmental impacts of a product or service under study.

3.2.1 Goal and Scope Definition

The goal of this LCA study was twofold: first, to assess the environmental impacts of the Nussir copper ore mine and identify significant environmental hotspots at the plant level, and then to compare the impacts of alternative energy and tailings management options in relation to the overall environmental performance of the mine. The systems boundaries were from cradle to gate (Fig. 1), beginning with production of raw materials & energy required for the mine operations and ending with copper concentrate ready for delivery. The SimaPro 8.3 software was used for LCI and LCIA modelling.

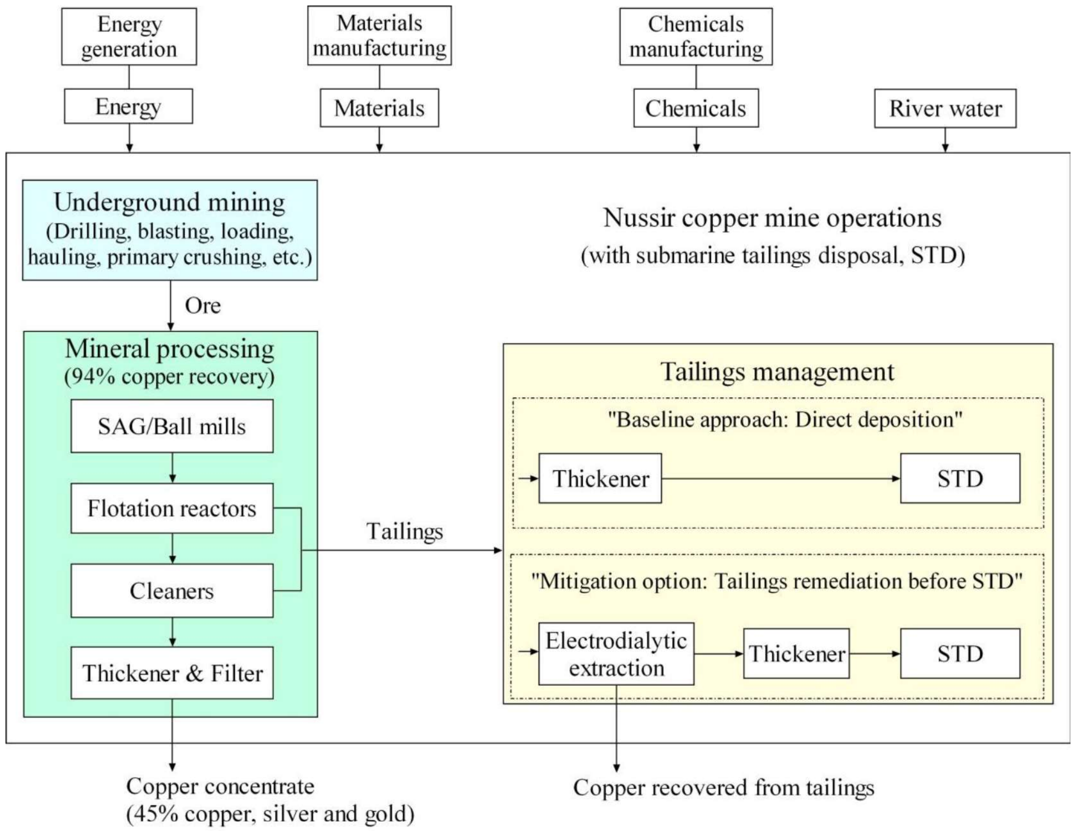

Fig. 1. System boundaries for the cradle-to-gate LCA of a Norwegian underground copper ore mine, including a hypothetical electrodialytic tailings extraction process.

Three plant-level scenarios were analyzed in this study, including a baseline scenario (A) and two energy- and tailings-oriented scenarios (B & C). Scenario A referred to the above-mentioned initial copper ore mining and mineral processing operations, including diesel-driven trucks, heavy fuel oil for building heating, and direct submarine tailings disposal (STD). Compared to scenario A, scenario B was different in two energy options, namely (i) replacing diesel-driven trucks with all-electric trucks (focusing on energy use in the customer use phase), and (ii) replacing heavy fuel oil with natural gas for building heating. Scenario C differed from scenario A in tailings management, in which a hypothetical electrodialytic tailings extraction process was added before STD. The differences between the three scenarios are summarized in Table 1.

Table 1

Different energy and tailings management options in scenarios A, B and C.

Scenario	Energy management		Tailings management
	Heavy-duty truck[a]	Building heating	
A: Baseline	Diesel-driven	Heavy fuel oil	Submarine tailings disposal (STD)
B: Energy scenario	Electric-driven	Natural gas	STD
C: Tailings scenario	Diesel-driven	Heavy fuel oil	Electrodialytic remediation and STD

[a] This study addressed only energy use of mining trucks, excluding vehicle manufacturing and final disposal.

The functional unit (FU) assumed for the three scenarios was slightly different. The FU of scenarios A & B was one kilogram of copper in concentrate produced at beneficiation. In scenario C, the FU became 1 kg of copper as a sum of copper in concentrate and copper recovered from the corresponding amount of tailings. Copper recovered from tailings through electrodialysis could be further electrowon as pure metallic copper, which is as pure as or purer than electro-refined copper (Schlesinger et al., 2011). For simplification purposes, this study assumed that the economic value of the recovered copper from tailings was the same as copper in concentrate. The reason for choosing copper in concentrate as the functional unit is to ensure a fair comparison between the life cycle impacts of this mine and other copper ore mine operations on varying grades of ore and/or concentrate.

The allocation of environmental burdens between product and co-products was based on the economic values of the product "copper in concentrate" and the co-products "silver and gold in concentrate" (Table 2). For simplification purposes, the average World Bank commodity price data of copper, silver and gold from 2005 to 2016 (World Bank, 2016) were used as a basis.

Table 2

Economic allocation between the product and co-products in scenarios A, B and C.

Scenario	Product/ co-products	Mass (gram)	Mass allocation (%)	Average price (USD/kg)[a]	Total price (USD)	Economic allocation (%)
A & B	Copper	1000	99.9	6.5	6.5	82.8
	Silver	1.27	0.1	716.1	0.9	11.6
	Gold	0.01	0.001	42057.7	0.5	5.6
C	Copper	1000[b]	99.9	6.5	6.5	83.4
	Silver	1.22	0.1	716.1	0.9	11.2
	Gold	0.01	0.001	42057.7	0.4	5.4

[a] Based on World Bank (2016).

[b] As the sum of 0.96 kg of copper in concentrate and 0.04 of kg copper recovered from tailings (see Table 4).

3.2.2 Life Cycle Inventory (LCI)

Although the actual operational data of the mine was not available at the time of writing, we collected enough concept-level production data for conducting a first-pass LCA of this mine. Foreground data for Nussir mine operations were taken from publicly accessible documents at the Nussir ASA's website (www.nussir.no), including the application for tailings discharge permit (Nussir ASA, 2011) and the permit granted by Norwegian Environment Agency in 2016 (NEA, 2016). The production and emission data in the tailings permit documents were regarded as representing the operational condition of this mine (although probably on a maximum basis). Some unit-level data, such as water use and the facility built-up area, were provided by Nussir ASA. The main assumptions made when defining the life cycle inventory of this study were summarized in Table S1, supplementary data. All background processes used in this study (Table S2, supplementary data) were taken from the ecoinvent database v3.3 within SimaPro 8.3 software.

This study defined the LCI data of energy options, based on the following respective reference information: (i) heavy-duty diesel trucks converting around 39% of energy stored in diesel to power at the wheels (Thiruvengadam et al., 2014), (ii) all-electric trucks converting around 60% of the electrical energy from the grid to power at the wheels (US DOE, 2016), (iii) heat values of various fuels assumed as 45 MJ/kg for diesel, 42 MJ/kg for heavy fuel oil and 37 MJ/kg for natural gas (World Nuclear Association, 2016).

Since there are no tailings placement monitoring data from this planned mine, we estimated the metal leaching potential of tailings based on experimental data on metal speciation of tailings before and after electrodialysis (Tables S3 and S4, supplementary data). Those metal speciation data were obtained from experiments on electrodialytic extraction of mine tailings samples from the Nussir and Ulveryggen deposits produced in a simulated flotation process of this mine (Pedersen et al., 2017), showing that acidic electrodialysis had an extraction potential of 64% of copper in tailings. In this electrodialysis experiments, the authors employed the modified BCR sequential extraction scheme (proposed by Standards Measurements and Testing Program of the European Commission) to assess metal availability. In short, the modified BCR sequential extraction scheme includes four steps (Sungur et al., 2014): (i) the first step using acetic acid solution to extract all ion-exchangeable and acid/water soluble metals (referred as the “exchangeable” fraction); (ii) the second step using hydroxylamine hydrochloride solution to extract all easily reducible metals bound to Fe-Mn oxides (the “reducible” fraction); (iii) the third step using hydrogen peroxide to extract all oxidizable metals bound to organic matter and sulfides (the “oxidizable” fraction); (iv) the last step digesting the solid phase in aqua regia (nitric acid) solution to extract all remaining metals bound to mineral matrix (the “residual” fraction).

Table 3 presents the basic principles that we used to estimate metals leaching from tailings in LCI, based on sequential extraction and aligning with different time scales and potential future manageability of emissions. When estimating the long-term metal leaching potential of tailings, we adopted the term geoavailability from environmental geochemistry of mineral deposits. In short, geoavailablity refers to the portion of total metals in rocks/ores that can be liberated to the biosphere through access and susceptibility to weathering (Smith and Huyck, 1999). According to Pettersen and Hertwich (2008), the geoavailable metal, as mobilizable metal in the long term, can be regarded as the sum of exchangeable, reducible and oxidizable fractions of the sequential extraction analysis. The residual fraction is not regarded as leachable because metals in this fraction are bound to stable minerals that will be dissolved at acidic conditions (pH<2).

Table 3

The approach to estimate metals leaching from tailings in the LCI phase, based on sequential extraction.

Time horizon	Potential manageability[a] (required level of evidence)	Metal leaching potential of tailings (based on sequential extraction)[b]
Short term	Technology can avoid many problems (Undisputed, only proven effects included)	Exchangeable
Balanced between short and long term	Proper policy can avoid many problems (Inclusion of effects based on consensus)	Exchangeable + Reducible
Long term	Problems can lead to catastrophe (All possible effects)	Exchangeable + Reducible + Oxidizable

[a] Adapted from Goedkoop and Spriensma (2001).

[b] Based on Pettersen and Hertwich (2008).

3.2.3 Life Cycle Impact Assessment (LCIA)

Life cycle impact assessment was performed using the ReCiPe method that aligns midpoint-oriented and endpoint-oriented impact pathways (Goedkoop et al., 2013). Since damage-oriented endpoint model has higher uncertainties compared to the problems-oriented midpoint model (Goedkoop et al., 2013), this study presented the modelled environmental impacts at midpoint level.

The ReCiPe method was chosen for this case study with submarine tailings disposal for two reasons. Firstly, the ReCiPe method uses the USES-LCA 2.0 model (van Zelm et al., 2009) to assess the toxicity and ecotoxicity impacts, and it is by now the only LCIA method available in SimaPro 8.3 that includes a marine environmental compartment. Secondly, this method groups different sources of uncertainties and temporal choices into three ReCiPe perspectives, namely individualist (based on the short-term interest), hierarchist (based on the most common policy principles), and egalitarian (representing the most precautionary perspective) (Goedkoop et al., 2013). The three different perspectives were initially proposed by Hofstetter (1998) for dealing with the problem of subjectivity in LCIA modelling. In fact, ReCiPe is one of a few LCIA methods that accommodates time horizon for metal effects, which allows testing the robustness on the impact assessment results on alternative tailings disposal options with different temporal scales.

When assessing the potential impacts of tailings before and after electrodialysis, we coupled the three ReCiPe perspectives with the above estimated metal leaching potential of tailings based on sequential extraction (Tables S5 and S6, supplementary data). In the individualist perspective, the metal leaching potential was interpreted as those highly mobile metals bound in tailings, equal to the exchangeable fraction of sequential extraction. In the hierarchist perspective, it was defined as the sum of exchangeable and reducible fractions. In the egalitarian perspective, it referred to all geoavailable metals, equivalent to the sum of exchangeable, reducible and oxidizable fractions.

3.2.4 Uncertainty analysis

Based on the default basic data uncertainty factors of the ecoinvent database v3.3, all foreground data used in this study were assessed using a pedigree matrix approach. The pedigree matrix approach assesses the quality of data sources in a semi-quantitative way, according to five independent data quality characteristics: reliability, completeness, temporal, geographic, and further technological correlation (Muller et al., 2016). After the data sources quality assessment, Monte Carlo simulation in SimaPro 8.3 was performed using 10000 runs to generate a 95% confidence interval for each ReCiPe impact category of scenarios A, B and C.

4. Results

4.1 Life Cycle Inventory (LCI)

Table 4 provides the plant-level life cycle inventories of the baseline scenario A and two mitigation scenarios B and C per functional unit. The copper-related data were validated using a copper mass balance approach both at the plant level and most important sub-process level. As shown in Table 4, the LCI of scenario B was different from scenario A in two energy use data, namely electricity for all-electric trucks and natural gas for heating (cf. diesel for diesel trucks and heavy fuel oil for heating in scenario A). The energy intensity of scenario A (23.1 MJ/kg Cu in concentrate) and scenario B (21.3 MJ/kg Cu in concentrate) differed, owing to the energy conversion efficiencies of diesel trucks (in scenario A) and all-electric trucks (in scenario B). As for scenarios A and C, the change of tailings management options resulted in different amounts of ore required per functional unit. In specific, the baseline scenario A required 100.1 kg of raw ore and generated 97.9 kg of tailings on dry solid basis. For comparison, scenario C reduced the amount of both ore (96.2 kg) and tailings (94.1 kg), since the recovered copper (0.04 kg) from electrodialytic remediation was part of the functional unit of scenario C.

Table 4

Inventory data for mining and mineral processing operations of three scenarios used in the study.[a,b]

Category	Item	Value (per functional unit)		
		Scenarios A and B[b]	Scenario C	Units

Product(s)	Cu in concentrate	1.00	0.96	kg
	Cu recovered from tailings (electrodialysis)	–	0.04	kg
Co-products	Ag in concentrate	1.3×10^{-3}	1.2×10^{-3}	kg
	Au in concentrate	1.1×10^{-5}	1.0×10^{-5}	kg
Known	Copper ore	100.1	96.2	kg
inputs from	Water, river	8.0×10^{-2}	7.7×10^{-2}	m^3
nature	Occupation, industrial area, built up	8.1×10^{-4}	8.1×10^{-4}	$m^2\cdot yr$
	Occupation, electrodialysis unit, built up[c]	–	8.1×10^{-5}	$m^2\cdot yr$
Known	Blasting (Tovex)	2.9×10^{-2}	2.8×10^{-2}	kg
inputs from	Conveyor belt	2.0×10^{-5}	2.0×10^{-5}	m
technosphere	SIPX	2.5×10^{-3}	2.4×10^{-3}	kg
(Materials	MIBC	5.5×10^{-3}	5.3×10^{-3}	kg
and fuels)	CMC	5.5×10^{-3}	5.3×10^{-3}	kg
	Magnafloc 10	3.7×10^{-3}	3.5×10^{-3}	kg
	Burnt lime	9.1×10^{-3}	8.8×10^{-3}	kg
	Sodium nitrate (electrodialysis), scenario C	–	1.7×10^{-2}	kg
	Polyamide (electrodialysis), scenario C[d]	–	4.6×10^{-8}	kg
	Polysulfone (electrodialysis), scenario C[d]	–	8.9×10^{-6}	kg
	Polyester (electrodialysis), scenario C[d]	–	2.9×10^{-6}	kg
	Diesel (for diesel trucks), scenarios A and C	5.0	4.8	MJ
Known	Electricity (for electric trucks), scenario B	[3.3]	–	MJ
inputs from	Electricity (facilities)	18.0	17.3	MJ
technosphere	Electricity (for electrodialysis), scenario C	–	9.5×10^{-2}	MJ
(Electricity	Heavy fuel oil (heating), scenarios A and C	5.0×10^{-2}	5.5×10^{-2}	MJ
and heat)	Natural gas (heating), scenario B	[4.9×10^{-2}]	–	MJ
Emissions to	PM>10	2.2×10^{-2}	2.1×10^{-2}	kg
air	2.5<PM<10	2.0×10^{-2}	1.9×10^{-2}	kg
	PM<2.5	2.2×10^{-3}	2.1×10^{-3}	kg

Emissions to water	SIPX	2.5×10^{-4}	2.4×10^{-4}	kg
	MIBC	5.5×10^{-4}	5.3×10^{-4}	kg
	CMC	5.5×10^{-4}	5.3×10^{-4}	kg
	Magnafloc 10	3.7×10^{-4}	3.5×10^{-4}	kg
	Burnt lime	9.1×10^{-4}	8.8×10^{-4}	kg
Final wastes	Polystyrene waste (conveyor belts, tyres)	2.5×10^{-3}	2.5×10^{-3}	kg
	Steel waste (steel scrap)	2.5×10^{-3}	2.5×10^{-3}	kg
	Packaging waste (paper and board)	1.3×10^{-4}	1.3×10^{-4}	kg
	Electronic waste (electronics)	1.0×10^{-4}	1.0×10^{-4}	kg
	Remaining wastes (unsorted)	2.5×10^{-3}	2.5×10^{-3}	kg
Tailings	Flotation tailings (dry solid)	97.9	94.1	kg

[a] Based on 75% ore from Nussir deposit (1.15% copper) and 25% ore from Ulveryggen deposit (0.8% copper), concentrate grade 45% copper, 94% copper recovery at mineral processing, 64% copper recovery from tailings by acidic electrodialysis.

[b] The inventory of scenario B differed from scenario A in two energy datasets marked with [].

[c] Estimated by scaling up experimental electrodialysis cells with an average 21-day processing period (Pedersen et al., 2017).

[d] Layer thickness (and density) of the membrane for electrodialysis was assumed as: 0.2 µm polyamide (1.27 g/cm^3), 40 µm polysulfone (1.24 g/cm^3), and 120 µm polyester (1.37 g/cm^3).

4.2 Life Cycle Impact Assessment (LCIA)

4.2.1 Overall environmental performance of the copper ore mine (baseline scenario A)

The life cycle environmental impacts of scenario A (with initial mine operations) is illustrated in Fig. 2 and Table S7, supplementary data, based on the ReCiPe midpoint (hiererchist) method. The carbon footprint of this mine was 0.69 kg CO_2-eq per kg copper in concentrate, mainly contributed by diesel for mining trucks (52%), blasting (20%) and electricity use (16%). The water footprint was 0.19 m^3 water per kg copper in concentrate, which the on-site water use and electricity generation accounted for, 35% and 63%, respectively.

Besides the climate change (CC) impact, diesel for mining trucks was the primary contributor of ozone depletion (OD, 68%), fossil depletion (FD, 60%), natural land transformation (NLT, 58%). Blasting ("Tovex") dominated in four impact categories: terrestrial acidification (TA, 70%), marine eutrophication (ME, 66%), photochemical oxidant formation (POF, 64%), and terrestrial ecotoxicity (TET, 58%). The particulate matter formation (PMF) impact was mainly owing to the on-site mineral dust emissions (80%), followed by blasting (12%) and diesel (7%). Reagents contributed between 0.01% and 21% to all impact categories, with higher values observed in agricultural land occupation (ALO, 21%), freshwater eutrophication (FE, 17%), fossil depletion (FD, 13%), and freshwater ecotoxicity (FET, 13%). Metals leaching from tailings dominated the impacts of human toxicity (HT, 98.9%) and marine ecotoxicity (MET, 99.9%), with negligible contributions (<0.01%) to all the other categories.

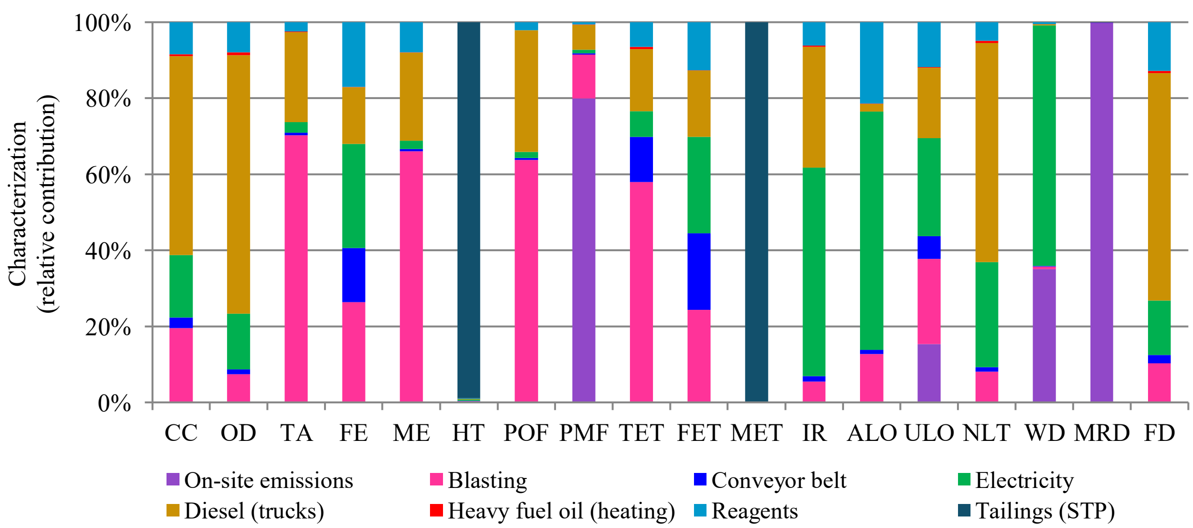

Fig. 2. Life cycle contribution analysis of the baseline scenario A (ReCiPe midpoint/hierarchist).[a,b]

[a] The "on-site emissions" category did not include the on-site use of blasting, diesel, heavy fuel oil and natural gas, which were part of the corresponding cradle-to-gate processes in this study. Besides, tailings were discussed separately.

[b] CC = climate change; OD = ozone depletion; TA = terrestrial acidification; FE = freshwater eutrophication; ME = marine eutrophication; HT = human toxicity; POF = photochemical oxidant formation; PMF = particulate matter formation; TET = terrestrial ecotoxicity; FET = freshwater ecotoxicity; IR = ionizing radiation; ALO = agricultural land occupation; ULO =

urban land occupation; NLT = natural land transformation; WD = water depletion; MRD = mineral resource depletion; FD = fossil fuel depletion.

Results of this contribution analysis indicated the possible cleaner production measures, especially on using more environmentally-friendly blasting agents, reducing environmental impacts of diesel-powered trucks, controls on mineral dust emissions from crushing and transport at mine site. For tailings management, reduction of heavy metals leaching from deposited tailings was crucial to minimizing both the potential human toxicity and marine ecotoxicity impacts in a long term perspective.

4.2.2 Comparison of the energy-related environmental impacts of scenarios A and B

Fig. 3 displays the comparison results of the environmental impacts between the baseline scenario A (with diesel-driven trucks and heavy fuel oil for heating) and scenario B (with all-electric trucks and natural gas for heating), using the ReCiPe midpoint (hierarchist) method (Table S8, supplementary data). Compared to scenario A, the environmental impacts of scenario B reduced in 13 of 18 categories, particularly in ozone depletion (by 65%), climate change (by 50%), and fossil depletion (by 57%), mainly owing to replacing electric trucks with diesel trucks. On the contrary, the impacts of scenario B increased by 9% in agriculture land occupation and by 11% in water depletion than scenario A, related to the generation of hydropower-dominated electricity in Norway.

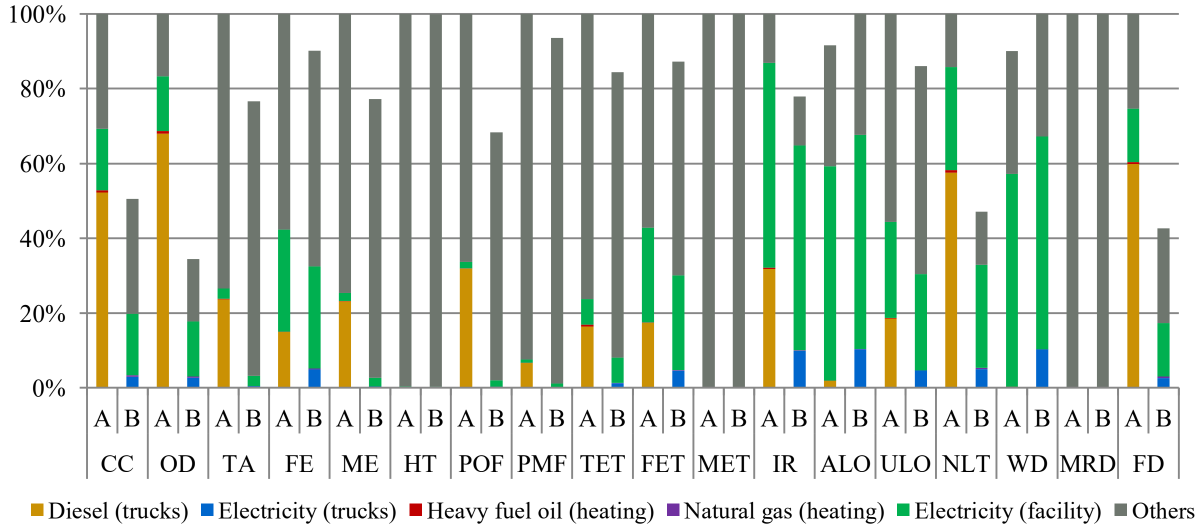

Fig. 3. Comparison of the life cycle environmental impacts between scenarios A and B (ReCiPe midpoint/hierarchist).

For the two building heating sources, both heavy fuel oil (in scenario A) and natural gas (in scenario B) contributed very little to all impacts (up to 1.3%) at the plant level. Compared to heavy fuel oil, however, the absolute impacts of natural gas per functional unit increased in four categories (by 148% in freshwater eutrophication, 64% in agricultural land occupation, 49% in freshwater ecotoxicity, and 6% in metal depletion), but decreased by 11-92% in all the other categories. The comparison results showed that replacing heavy fuel oil with natural gas for heating of buildings would contribute negligible to the overall environmental performance of this mine. This comparison indicated the necessity of comparing the impacts of possible cleaner production measures with the overall plant-level impacts of a mine.

Regarding electrification of mining trucks in a general context, it may be concluded that electrification would be more environmentally beneficial as long as the electrical supply is "relatively clean" through all stages of its life cycle (but "cleaner" depending on weighting across impact categories). This conclusion is in line with the literature calling for paying attention to the environmental impacts of electricity generation for externally chargeable vehicles (Nordelöf et al., 2014). To make better environmental decisions on electrification in mining, the impacts associated with both energy production and conventional/electric vehicles manufacturing need to be assessed from a life cycle perspective. As noted by Hawkins et al. (2013), the supply chain of electric vehicles could potentially contribute to a significant increase in the impacts of freshwater eutrophication, freshwater ecotoxicity, human toxicity, and metal depletion. This holds true also for discussions about the shift from off-highway diesel-driven trucks to electric belt conveyors.

4.2.3 Comparison of the tailings-related environmental impacts of scenarios A and C

Among the ReCiPe impact categories, metals leaching from tailings dominated the human toxicity (HT) and marine ecotoxicity (MET) impacts. Since scenario A (with direct submarine disposal) and scenario C (with electrodialytic remediation of tailings prior to submarine disposal) differed only in tailings management, this comparison focused on the HT and MET impacts across different ReCiPe perspectives (Fig. 4). Firstly, the HT impact of scenario C accounted for less than 10% of that of scenario A from all three perspectives. This was because the electrodialytic remediation process removed manganese from all the first three fractions of sequential extraction (Tables S3 and S4 in supplementary data), resulting in a decreased manganese leaching potential of deposited tailings defined in this study. The comparison results also indicated that electrodialysis achieved part of the intended remediation effects of metals bound in tailings.

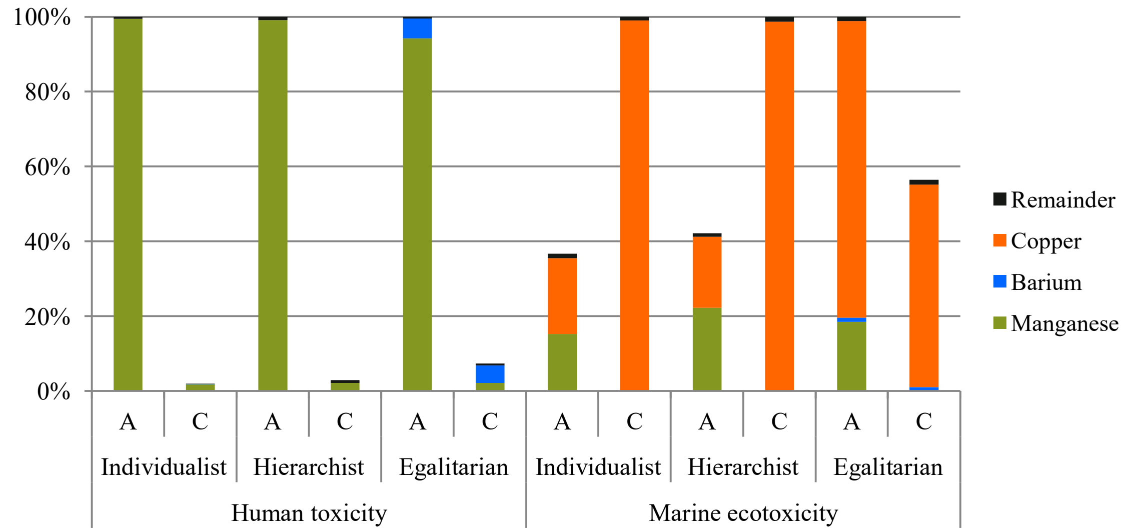

Fig. 4. Comparison of the two tailings-dominated impacts between scenario A (with direct STD) and scenario C (with electrodialytic remediation of tailings prior to STD). The term Remainder refers to the sum of metals each less than 1% of the total potential.

Secondly, the results of the MET impact of tailings before and after electrodialysis varied across the ReCiPe perspectives. Compared to the direct submarine tailings disposal in scenario A, the MET impact of tailings after electrodialysis (scenario C) increased approximately by 173% in the individualist perspective and by 137% in the hierarchist perspectives, but it decreased by 43% in the egalitarian perspective. For the MET impact across the three perspectives, manganese and copper were two primary contributors in scenario A, while copper was the dominant contributor in scenario C. What stands out from this comparison is a trade-off between extracting more copper from tailings and the potential environmental impacts of tailings after electrodialytic extraction, partly depending on how to define the metal leaching potential of tailings across different ReCiPe perspectives.

We compared the potential environmental impacts of the electrodialysis (ED) process in scenario C with the overall plant-level impacts of scenarios A and C (Fig. 5 and Table S9, supplementary data). Except for the differences in the HT and MET impacts highlighted above, the impacts of scenario C increased by 0.5-22% in 11 of the 16 environmental categories (highest value in marine eutrophication) and decreased by 1-3% in the other categories, compared to scenario A. The life cycle contribution analysis of the ED process showed that sodium nitrate (as electrolyte) contributed 93-99% of 16 of the 18 impact categories, except the indicators of agriculture land occupation (sodium nitrate 89%, electricity 10%) and water depletion (sodium nitrate 56%, electricity 42%), while the membrane contributed up to 1.7% in all impact categories. The results could be used for benchmarking with other tailings remediation and/or disposal options in future studies.

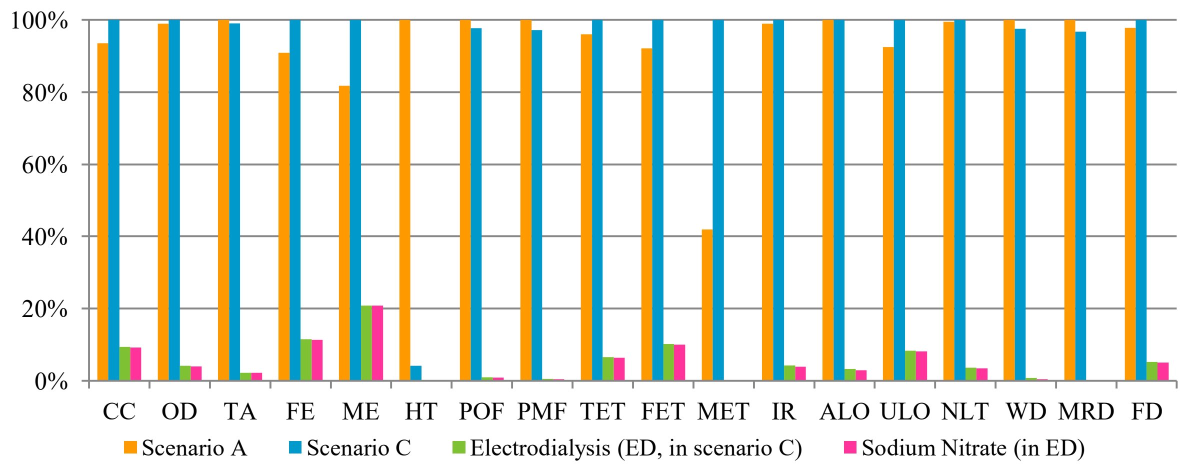

Fig. 5. Comparison between the life cycle impacts of the electrodialytic tailings remediation process (in scenario C) and the overall plant-level impacts of scenarios A and C.

4.3 Uncertainty analysis

To understand the effect of multiple uncertainty sources on the LCIA results, Monte Carlo simulation was executed in SimaPro. The uncertainties of the ReCiPe midpoint (hierarchist) impacts of scenarios A, B and C were used as an example (Table S10, supplementary data). For all the three scenarios, the uncertainties (indicated by the coefficient of variability (CV) defined as the ratio between the standard deviation and the mean) had minor differences in each impact category but varied across impact categories. Specifically, the category of mineral resource depletion

(CV=3%) had lowest uncertainty, followed by climate change (CV=14-21%), while water depletion had the largest uncertainty (CV=508-700%). A relatively high level of uncertainty was noticed in human toxicity (CV=95-168%) and marine ecotoxicity (CV=130-172%), both primarily related to leaching from the tailings deposit. A relatively low level of uncertainty (CV=17-46%) existed in ten categories, such as, marine eutrophication (CV=20-30%), terrestrial acidification (CV=25-32%), terrestrial ecotoxicity (CV=33-41%), freshwater ecotoxicity (CV=33-36%), and freshwater eutrophication (CV=43-46%).

In SimaPro, those absolute uncertainties are associated only with the uncertainty in the life cycle inventory, without considering the uncertainties in the characterization scores themselves (Goedkoop et al., 2016). We recommend interpreting the results of Monte Carlo analysis as an indication of the relative uncertainty in each impact category. For the uncertainties of the HT and MET impacts in this study, for example, the relatively high level of output variance indicates that the uncertainty factor related to the LCI inputs of metals leaching from tailings, with an ecoinvent base uncertainty factor of 5 (process emissions of heavy metals to water) in SimaPro, has a relatively strong influence on the modelled results.

5. Discussion

5.1 Benchmarking LCI of copper mine operations

The water-, energy- and material-related LCI results of this Norwegian underground copper mine were benchmarked with two reference cases (Table 5): one Australian underground copper ore (Norgate and Haque, 2010) and the other representing European copper mine operations on average (underground mining 30% and open pit 70%) in the latest ecoinvent database v3.3. For comparison purposes, the initial functional units of the two reference cases (either based on ore or concentrate) were converted to the same functional unit as this study (i.e. 1 kg copper in concentrate).

Table 5

Benchmarking the LCI data of this study with the literature (per kg of copper in concentrate).

Copper mine operation	Ore (kg)	Energy intensity			On-site water (m^3)	Explosive (kg)	Reagents (kg)
		Diesel (MJ)	Heating oil or gas (MJ)	Electricity (MJ)			
Scenario A (Norway)[a]	100.1	5.01	0.05	18.0	0.08	0.03	0.03
Scenario B (Norway)[a]	100.1	–	0.05	21.3	0.08	0.03	0.03
Scenario C (Norway)[a]	96.2	4.8	0.06	17.4	0.08	0.03	0.03
Reference case 1 (Australia)[b]	59.3	6.8	–	9.9	0.03	0.02	0.10
Reference case 2 (Europe average)[c]	65.5	0.002	–	3.8	0.03	0.04	0.08

[a] Underground mining, ore grade 1.06 % copper (average), concentrate grade 45% copper, 94% copper recovery, explosive (Tovex), reagents (35% burnt lime, 21% MIBC, 21% CMC, 14% Magnafloc, 10% SIPX).

[b] Underground mining, ore grade 1.8% copper, concentrate grade 27.3% copper, 93.7% copper recovery, explosive (ANFO, 94% ammonium nitrate and 6% fuel oil), reagents (80% lime, 12% xanthate, 8% sodium cyanide), based on Norgate and Haque (2010).

[c] Underground mining (30%) and open pit (70%), ore grade 1.83% copper, concentrate grade 29.7% copper, 83.5% copper recovery, explosive (Tovex), reagents (56% lime, 34% inorganic chemical, 10% organic chemical), based on the ecoinvent database v3.3.

Due to a lower ore grade in this Norwegian mine, firstly, the results showed a higher ratio of ore per kg of copper and more on-site water use, compared to the two reference cases. However, the average on-site water use of this mine (0.08 m^3/kg copper in concentrate, equivalent to 0.8 m^3/t ore) was within the range of 0.34–2.07 m^3 (with a weighted mean of 0.96 m^3) per ton ore processed in conventional flotation-based copper mines (Gunson et al., 2012). Secondly, the energy intensity (as the sum of on-site diesel, heavy fuel oil/ natural gas, and electricity) of all three scenarios was higher than the two reference cases, probably due to the differences in site-specific mining and mineral processing operations. But the energy intensity of the planned Norwegian copper mine was similar to another reported range of 10–20 MJ/kg copper in concentrate, based on 31 copper mine operations (open pit and/or underground) from 11 countries worldwide (Northey et al., 2013). Thirdly, this Norwegian mine required fewer amounts of reagents in total, compared to the Australian and European reference cases. This benchmarking, to some extent, also indicated the necessity for carrying out a detailed material-, water- and energy balance analysis based on actual operational data of this mine in future studies.

5.2 Environmental assessment of mining and mineral processing operations using LCA

Life cycle impact assessment of mining and mineral processing operations could contribute to identification of environmental hotspots and development of cleaner production measures. However, results of the life cycle contribution analysis may vary on a case-by-case basis. Take the climate change (CC) impact as an example. As shown in Fig. 2, the top three contributors of the CC impact in scenario A were diesel for mining trucks (52%), blasting (20%), and electricity (16%), which became blasting (39%), electricity (32%), and reagents (16%) in scenario B. By contrast, Norgate and Haque (2010) reported that the crushing and grinding steps, as the largest energy user (39.4%), made the largest contribution (46.8%) to the total GHG emissions for the production of copper concentrate at an underground copper ore in Australia. The different contribution analysis results of the CC impact indicate the importance of investigating the relative contributions of both upstream GHG emissions (from fuel and electricity generation) and on-site emissions of unit operations to the total GHG emissions of a mine.

During the goal and scope definition phase of a mining LCA study, one important issue for LCA practitioners is the selection of impact assessment categories (and subsequent interpretation). In a comparative LCA of belt conveyors and trucks in a Turkey mine, Erkayaoğlu and Demirel (2016) pre-selected climate change and acidification, based on the literature, as representatives of the major environmental concerns in the mining industry. For comparison, our results of the baseline scenario

A showed that (i) the top three contributions of the conveyor belt ("cradle-to-gate") were to freshwater ecotoxicity (20%), freshwater eutrophication (14%), and terrestrial ecotoxicity (12%), (ii) the conveyor belt accounted for up to 3% of climate change and 0.7% of terrestrial acidification, and (iii) the two main contributing processes to terrestrial acidification were blasting (70%) and diesel for mining trucks (24%). One reason for the different priority impacts from case to case may be that mines have diverse material- & energy intensity and site-specific environmental challenges, indicating that there is no one-size-fits-all context for the goal and scope definition of a specific mining LCA study.

Although LCA aims for a holistic environmental systems analysis and impact assessment, there are different opinions in the literature on what impact categories need to be considered in mining LCAs. Based on an effort on aligning the LCA methodology for the metal and mining industry, Santero and Hendry (2016) reported five recommended impact categories (i.e. global warming, acidification, eutrophication, smog potential and ozone depletion), together with primary energy demand, net water consumption and waste generation. The authors also pointed out that several impact categories, including resource depletion, toxicity to human and ecosystems, land use change, and water scarcity, were not recommended for LCAs involving metals, since they were labelled as level II or III in the ILCD handbook (JRC, 2011). In contrast, Awuah-Offei and Adekpedjou (2011) emphasized that the standard impact categories (global warming, ozone depletion, acidification, eutrophication, human toxicity, and aquatic ecotoxicity), together with land-, water-, and energy use as well as resource depletion, were equally important in mining LCAs. In an attempt at bridging Arctic environmental science and LCA, Johnsen (2014) highlighted the importance of assessing the impacts of ecotoxicity, acidification, and soot/black carbon emissions in Arctic-specific LCA studies, such as on mining activities in northern Norway. Based on public interest and regulatory concerns, environmental communication in the Nussir mining case relates to two key aspects on land use conflicts and risk to marine recipients from tailings management. Both of the two aspects are inherently local issues that have not been covered by the above five recommended impact categories for metal LCAs. Although there is still no consensus on some assessment impact results, we selected all impact categories of ReCiPe in this study. Our life cycle contribution analysis results indicated the necessity for the inclusion of human toxicity and marine ecotoxicity impacts in LCA of submarine tailings disposal under study.

To promote the use of LCA as an environmental decision-support tool in the mining industry, we recommend carrying out at least a streamlined LCA at the plant level, to aid in developing initiatives for cleaner production and making optimal environmental decisions. According to Stewart et al. (2012), almost all decisions taken in the mining industry at present are dominated by risk and uncertainty assessment, specifically focusing on operational safety and project evaluation. In contrast to environmental risk assessment (ERA) that addresses a single chemical at a specific location, LCA seeks to evaluate hundreds of chemical emissions (and resource stressors) occurring at various locations for their potential impacts in various impact categories (Margin and Curran, 2012). In other words, ERA aims for avoidance of environmental risks at a specific location, while LCA-based decisions are intended for an improved environment (even more sustainable development) from a systems perspective.

5.3 Mine tailings management and LCA of metals leaching from tailings

5.3.1 Tailings and mineral resource management

In general, mine wastes are defined as "those waste products originating, accumulating and present at mine sites, which are unwanted and have no current economic value" (Lottermoser, 2010). For tailings, there are two main disposal strategies: direct disposal (e.g. river and marine tailings disposal) and indirect disposal (e.g. conventional tailings dam, paste and thickened tailings, tailings reuse, recycling and reprocessing). According to Lottermoser (2011), "reuse" is defined as involving the new use or application of the total mine waste as its original form for a purpose directly without any reprocessing. On the other hand, "recycling" refers to extracting new valuable resource ingredients, or using the waste as feedstock and converting the entire mine waste into a new valuable product or application with some reprocessing. "Reprocessing" is to use mine wastes as a feedstock for producing a valuable product, such as recovered minerals and metals.

The 3Rs (reduce, reuse and recycle) pyramid has been commonly used in the generic waste management hierarchy and the development of mine waste management strategies in specific. The EU Waste Framework Directive (2008/98/EC), for instance, proposed a five-step waste hierarchy where prevention was the most preferred option, followed by re-use, recycling and other forms of recovery (like energy), while disposal such as landfill was the least desirable (EC, 2008). Lottermoser (2011) presented a similar mine waste hierarchy, but reported that many of the proposed reuse and recycling concepts were not economical to apply. Recently, Lèbre et al. (2017) put forward another mine waste management hierarchy, including steps from reduce (for waste prevention, most desirable), reprocess (extracting further valuable materials), downcycle (e.g. use for road construction), to disposal (final treatment, least desirable). Similarly, Lèbre et al. (2017) mentioned that recovering minerals from tailings was generally considered as too costly and energy-, water- and resource-intensive, making it undesirable both economically and environmentally.

Regarding the hypothetical tailings remediation process in this study, our LCI results demonstrated that electrodialytic remediation of copper mine tailings may be regarded as less energy- and water intensive. Based on the estimated annual production of 44389 tons copper concentrate (45% Cu) from 2 million tons of ore (on average 1.06% Cu), the tailings remediation process had a potential of extracting up to 816 tons of copper (64%) from the generated tailings on a yearly basis. As seen in Table 4, the electricity use of the electrodialysis operation, estimated by scaling up experimental data (Pedersen et al., 2017), accounted for around 0.5% of the total on-site electricity use of the planned mine. Moreover, we assumed no extra fresh water was needed for the electrodialysis process, since there was enough tailings water (after flotation). However, the inconsistent marine ecotoxicity impacts of tailings after electrodialysis across different ReCiPe perspectives (Fig. 4) make it hard to conclude whether or not the electrodialytic tailings remediation method could contribute to more environmentally responsible mining. To date, no LCA work has been published on electrodialytic remediation of copper mine tailings, making it impossible to compare our results with the literature. Further research is needed from a multi-criteria decision-analysis perspective, especially to assess in detail the water-, energy- and material uses of electrodialysis, to analyze the extended costs and benefits of recovered metals from tailings, and to reduce uncertainties on LCA of tailings disposal.

5.3.2 LCA of metal leaching from tailings

Among various environmental impacts of tailings, assessment of the toxicity and ecotoxicity impacts of heavy metals leaching from tailings poses a particular challenge in mining LCAs. In the latest ecoinvent database v3.3, for instance, only two off-site tailing treatment processes (one for sulfidic and the other for non-sulfidic tailing) were defined and used to model tailings management of most metal mine operations worldwide, regardless of differences in site-specific ore quality or characteristics of tailings. Moreover, the generic ecotoxicity impact assessment in LCA was typically based on ecological risk assessment models, assuming that toxic materials degrade over time (Stewart et al., 2012). This assumption is not applicable to metals, since metals will not degrade. Improving the life cycle inventory of metal leaching over time (and thereafter impact assessment) is crucial for further promoting the application of LCA in mine tailings management.

In this study, the marine ecotoxicity impact of tailings was sensitive to the way of alignment between the sequential extraction fractions and the metal leaching potential of tailings across different ReCiPe perspectives (Table 2). As shown in Fig. 4, the marine ecotoxicity impact of scenario A (with direct submarine tailings disposal) was around 40% of that in scenario C (with electrodialytic remediation of tailings prior to disposal) from the hierarchist perspective (representing most common policy principles), while it became almost 1.7 times of that in scenario C from the egalitarian perspective (representing the most precautionary principle). The inconsistent tailings impact results between scenarios A and C indicate the necessity of incorporating different value perspectives in mining LCAs to support the development of more proactive tailings management strategies at mine sites.

The results of the human toxicity and marine ecotoxicity impacts should be interpreted with caution, taking into account the changes of metal mobilization by electrodialysis. As seen from the sequential extraction results (Tables S3-S4 in supplementary data), metal desorption in acidic electrodialysis differed among the four sequential extraction fractions. This is illustrated by copper and manganese as two primary contributing metals to marine ecotoxicity. As seen from Table 6, the acidic electrodialysis process reduced manganese in all four fractions, mostly from the exchangeable and reducible fractions. On the contrary, the electrodialytic tailings extraction process resulted in a decrease of copper in the exchangeable and reducible fractions, but an increase of copper in the exchangeable and reducible fractions. This is the main reason for the different results of the marine ecotoxicity impact of scenarios A and C from the hierarchist and egalitarian perspectives.

Table 6

Changes of copper and manganese among the sequential extraction fractions after electrodialysis and the estimated metal leaching potential of tailings across different ReCiPe perspectives (mg/kg mine tailings on dry solid basis).

Item	Copper		Manganese	
	Before ED	After ED	Before ED	After ED

Sequential extraction of tailings[a]	Exchangeable (E)	23.5	119.2	1096.9	20.1
	Reducible (R)	2.7	22.2	399.0	13.2
	Oxidizable (O)	263.6	64.5	53.6	2.4
	Residual	362.2	28.9	63.3	50.9
	Desorbed in ED	–	409.4	–	927.0
Estimated metal leaching potential	Individualist (E)	23.5	119.2	1096.9	20.1
	Hierarchist (E+R)	26.1	141.4	1495.9	33.3
	Egalitarian (E+R+O)	289.8	205.9	1549.5	35.7

[a] Experimental data adopted from Pedersen et al. (2017).

A limitation of this LCA of submarine tailings disposal may lie in our assumption, based on the precautionary principle, on the metal leaching potential of tailings from the three ReCiPe perspectives. The actual metal leaching from tailings into seawater might be lower than the estimated data of this study, according to a two-phase copper leaching model (Walder, 2013) with an initial rapid leaching phase (during the tailings deposition settling period) and a long term slow leaching phase (after the tailings settled on the bottom and only the materials on the top of the tailings available for leaching). In a real situation, metal leaching is dependent on site-specific conditions, such as pH, redox potential, salinity, buffer capacity, retention time, and temperature. Marine impacts vary greatly across global marine ecosystems, due to differences in water chemistry and oceanic factors (Dong et al., 2016). Future research is needed, especially, to investigate possible relationships between sequential extraction (preferably using a greater number of tailings samples from actual operation) and on-site monitoring data of metal flux as well as to consider spatial differentiation in life cycle impact assessment.

6. Conclusions

This study assessed the life cycle environmental performance of a planned underground copper ore mine, located in northern Norway. For the production of 1 kg copper in concentrate, the energy intensity, carbon footprint and water footprint of this mine were 23.08 MJ, 0.69 kg CO_2-eq, and 0.19 m^3, respectively. The LCIA results, based on the ReCiPe midpoint (hierarchist) method, showed that on-site electricity use, diesel for mining trucks and blasting (100% Tovex) were the primary contributors to 14 of the 18 impact categories, deserving more attention towards continuous environmental improvement of the mine. Metals leaching from deposited tailings were the dominant contributors of human toxicity (99%) and marine ecotoxicity (99.8%). The energy-oriented scenario analysis indicated that replacing diesel trucks with electric trucks would reduce the plane-level environmental impacts in 13 of the 18 categories. The hypothetical electrodialytic tailings remediation process could extract up to 816 tons copper from tailings per year and was regarded as less energy- and water intensive in this study. However, the marine ecotoxicity impact of tailings after electrodialysis was inconsistent between the ReCiPe perspectives of hierarchist (based on the most common policy principles) and egalitarian (representing the most precautionary principle), partly due to the changes of metal mobilization by electrodialysis. These findings add substantially to our understanding on the trade-off between electrodialytic tailings remediation (extracting more metals before final disposal) and the potential environmental impacts of tailings after electrodialysis in the context of mine waste management hierarchy.

In the LCA of tailings management of this study, we have demonstrated the use of a geochemical approach, based on sequential extraction, to estimate metals leaching from tailings across three ReCiPe perspectives. We believe that our analysis could contribute to addressing the difficulty in defining long-term metal leaching potential of tailings in mining LCAs. In light of the inconsistent results of marine ecotoxicity from different ReCiPe perspectives, it is too early to draw conclusions on the feasibility of applying the electrodialytic tailings remediation method from an environmental perspective. Even so, two aspects deserve more attention in mining LCAs. Firstly, the ecotoxicity impact must be interpreted with caution, since it was seemingly sensitive to the way of defining the metal leaching potential of tailings (and site-specific environment as well). Secondly, the sources of both background and foreground data, selection of impact categories and assessment method(s) and assumptions made should be clearly documented and agreed with relevant stakeholders in an early stage of a mining LCA study, especially under the condition that the LCA results will be used in environmental decision-making. A detailed knowledge of a mine's life cycle environmental performance is a prerequisite for continuous environmental improvement and avoiding suboptimal cleaner production measures at mine sites.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at *(a link to be added after acceptance)*.

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Supplementary Data

Manuscript Title:

Comparative life cycle assessment of tailings management and energy scenarios for a copper ore mine: A case study in Northern Norway

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Table S1
Assumptions on life cycle inventory data of this study.

Item	Assumptions	Reference(s)
Raw ore mining	The annual average amount of raw ore produced (up to 2 million tons) was assumed to be 75% from the Nussir deposit (1.15% copper) and 25% from the Ulveryggen deposit (0.8% copper).	NEA (2016)
Copper recovery	The estimated copper recovery rate by the mining company (Nussir ASA) is 94–95.5%. In the study, we assumed 94% copper recovery rate.	Nussir ASA (2014)
Basting agent	The explosive Tovex was used in the study, since the blasting process is available in in ecoinvent database. But the pumped slurry explosive will be used in practice.	Ecoinvent database v3; Nussir ASA (2014)
Water use	Water was from a river reservoir and no extra pumping energy needed.	
Electricity mix	Assuming 100% electricity from Norwegian electricity supply mix (hydro 98%, fossil 0.2 %, and other 1.8%.)	Ecoinvent database v3
Engine efficiency (diesel trucks)	Heavy-duty diesel trucks converted ≈ 39% of energy stored in diesel to power at the wheels.	Thiruvengadam et al. (2014)
Engine efficiency (all-electric trucks)	All-electric trucks converted ≈ 60% of the electrical energy from the grid to power at the wheels.	US DOE (2016)
Chemicals	The maximum amount of chemicals granted by the tailings discharge permit was used for MIBC, CMC, Magnafloc 10, and burnt lime. The use of SIPX, pending in the granted permit, was assumed to be 25 g / ton ore.	NEA (2016) Nussir ASA (2011)
Chemical emissions to water	Chemical emissions from tailings were assumed to be 10% of the total amount of chemical use.	
Land use	A lifetime of 20 years (operational period) was assumed, according to the estimated available mineral resource in 2013. Only the build-up area of the plant was addressed in LCIA, while the underground mining exploration area was not considered in the study.	NEA (2016)
Dust emissions	One ton of dust emissions from crushing and transport was assumed to be 50 kg PM<2.5, 450 kg PM2.5-10 and 500 kg PM>10. For simplification, we assumed a linear relationship between the amount of ore processed and dust emitted.	Classen et al. (2007)
Waste rock	We excluded the produced metal-free waste rock from the study, which was assumed to be of relatively low toxicity.	
Cation exchange membrane	We estimated the following membrane layer thickness (and density): 0.2 μm Polyamide (1.27 g/cm^3), 40 μm Polysulfone (1.24 g/cm^3) and 120 μm Polyester (1.37 g/cm^3).	
Copper recovery rate of acidic electrodialysis	According to an electrodialysis experimental, up to 64% of Cu in tailings (in mass) could be recovered in acidic electrodialysis.	Pedersen et al. (2017)

Table S2
Main background processes taken from Ecoinvent database v3.3 within SimaPro 8.3.

Activity name (Market)[a]	Geography	Database time period	Notes[b]
Blasting	GLO (Global)	2011-01-01 to 2016-12-31	Blasting of 100% Tovex.
Conveyor belt	GLO (Global)	2011-01-01 to 2016-12-31	
Carboxymethyl cellulose, powder	GLO (Global)	2011-01-01 to 2016-12-31	
Carbon disulfide	GLO (Global)	2011-01-01 to 2016-12-31	
Sodium hydroxide	GLO (Global)	2011-01-01 to 2016-12-31	Sodium hydroxide, without water, in 50% solution state.
Isopropanol	GLO (Global)	2011-01-01 to 2016-12-31	
1-propanol	GLO (Global)	2011-01-01 to 2016-12-31	
Acrylonitrile	GLO (Global)	2011-01-01 to 2016-12-31	
Limestone, crushed, washed	RoW (Rest of World)	2011-01-01 to 2016-12-31	Expert judgement was used to develop product specific transport distance estimations.
Electricity, high voltage	NO (Norway)	2012-01-01 to 2016-12-31	High voltage level above 24 kV (large scale industry).
Heat production, heavy fuel oil, at industrial furnace 1MW	Europe without Switzerland	2001-01-01 to 2016-12-31	Heat, district or industrial, other than natural gas.
Heat production, natural gas, at industrial furnace low-NOx >100kW	Europe without Switzerland	2011-01-01 to 2016-12-31	Heat, district or industrial, natural gas.
Diesel, burned in building machine	GLO (Global)	2011-01-01 to 2016-12-31	Building machine including infrastructure, lubricating oil and fuel consumption as inputs, and some measured air emissions as output.
Sodium nitrate	GLO (Global)	2010-01-01 to 2016-12-31	
Polyamide	GLO (Global)	2011-01-01 to 2016-12-31	Glass fibre reinforced plastic, polyamide, injection moulded.
Polysulfone	GLO (Global)	2012-01-01 to 2014-12-31	For membrane filtration production.
Polyester	GLO (Global)	2011-01-01 to 2016-12-31	Glass fibre reinforced plastic, polyester resin, hand lay-up.

[a] Market activity starts at the site of the production activities with the product being ready to be transported to the consumers and ends at the site of the consumers.
[b] Taken from SimaPro 8.3 PhD version and the ecoinvent database v3.3 at www.ecoinvent.org.

Table S3
Results of sequential extraction of tailings after electrodialysis (ED), mg/kg tailings (on dry solid basis).

	Al	As	Ba	Ca	Cd	Co	Cr	Cu	Fe	K	Mg
Exchangeable	13.1	0	4.9	611.8	0	0	0.1	119.2	19.1	81.3	210.9
Reducible	51.4	0.2	33.8	50.8	0	0	1.1	22.2	1056.3	253.6	50.8
Oxidizable	26.8	0	5.8	39.5	0	0	1.2	64.5	63.8	226.2	77.9
Residual	4261.7	0.6	49.3	68.4	0.02	1.7	33.8	28.9	4162.0	3046.0	4633.6
Desorbed in ED	83.2	0.001	9.3	90068.4	0.0004	0.01	0.9	409.3	738.9	332.0	6609.8
Total mass	4436.3	0.8	103.1	90838.9	0.02	1.71	37.1	644.1	6040.2	3939.1	11583.0

	Mn	Mo	Na	Ni	P	Pb	Sb	Sr	V	Zn	Zr
Exchangeable	20.1	0	40.2	0.4	25.3	0.2	0.2	0.2	0.003	5.1	0.002
Reducible	13.2	0	13.4	0.4	67.2	1.0	0.2	0.1	0.5	1.7	0.0
Oxidizable	2.4	0.3	9546.3	0.6	20.1	0.4	0.0	0.1	0.2	1.9	0.02
Residual	51.0	1.1	113.2	15.1	73.4	0.3	0.0	0.1	8.1	13.6	1.1
Desorbed in ED	927.0	0	1685.0	0.8	37.9	0.3	0.01	4.7	0.02	1.4	0.002
Total mass	1013.7	1.4	11398.1	17.3	223.9	2.2	0.41	5.2	8.8	23.7	1.12

(Data source: Pedersen et al., 2017)

Table S4
Results of sequential extraction of tailings before electrodialysis, mg/kg tailings (on dry solid basis).

	Al	As[a]	Ba	Ca	Cd[a]	Co[a]	Cr	Cu	Fe	K[a]	Mg
Exchangeable	0.7	0.0002	26.2	35475.7	0.0001	0.004	0.1	23.5	0.4	164.4	1796.0
Reducible	2.5	0.2	17.4	6090.3	0.0001	0.004	0.2	2.7	650.1	336.6	2240.3
Oxidizable	77.8	0.0002	4.3	1033.5	0.0001	0.004	2.2	263.6	257.3	309.2	457.8
Residual	4283	0.6	30.5	96.3	0.02	1.7	33.9	362.2	5567.6	3128.9	4634.6
Total mass	4364	0.8	78.3	42695.9	0.02	1.71	36.4	651.9	6475.3	3939.1	9128.7
Difference[b]	-1.7%	0	-31.7%	-112.8%	0	0	-1.8%	1.2%	6.7%	0	-26.9%

	Mn	Mo[a]	Na[a]	Ni[a]	P[a]	Pb[a]	Sb[a]	Sr[a]	V[a]	Zn[a]	Zr[a]
Exchangable	1096.9	0	461.4	0.6	34.7	0.2	0.2	1.3	0.01	5.5	0.002
Reducible	399.0	0	434.7	0.6	76.7	1.1	0.2	1.3	0.5	2.0	0.001
Oxidizable	53.6	0.3	9967.5	0.8	29.5	0.5	0.002	1.2	0.2	2.3	0.02
Residual	63.3	1.1	534.5	15.3	82.9	0.4	0.002	1.3	8.1	13.9	1.1
Total mass	1612.8	1.4	11398.1	17.3	223.9	2.2	0.41	5.2	8.8	23.7	1.1
Difference[b]	37.15%	0	0	0	0	0	0	0	0	0	0

(Data source: based on Pedersen et al., 2017)

[a] Sequential extraction of those marked metals was not performed in the original experiment; in this study, they were estimated by allocating the amount of desorbed metals (Table S3) equally into the four fractions after ED.

[b] It was calculated as: Difference = (Total mass (before ED) – Total mass (after ED)) / Total mass (before ED).

Table S5
Results of the estimated metal leaching from tailings before electrodialysis (in scenarios A and B).

ReCiPe perspective	Metals, mg/kg tailings (on dry solid basis)										
	Al	As	Ba	Ca	Cd	Co	Cr	Cu	Fe	K	Mg
Individualist	0.7	0.0002	26.2	35475.7	0.0001	0.004	0.1	23.5	0.4	164.4	1796.0
Hierarchist	3.2	0.2	43.5	41566.0	0.0002	0.007	0.3	26.3	650.5	501.0	4036.3
Egalitarian	81.0	0.2	47.8	42599.6	0.0003	0.01	2.5	289.7	907.8	810.1	4494.1

ReCiPe perspective	Metals, mg/kg tailings (on dry solid basis)										
	Mn	Mo	Na	Ni	P	Pb	Sb	Sr	V	Zn	Zr
Individualist	1096.9	0.0	461.4	0.6	34.7	0.2	0.2	1.3	0.01	5.5	0.002
Hierarchist	1495.9	0.0	896.1	1.2	111.4	1.3	0.4	2.6	0.5	7.5	0.003
Egalitarian	1549.5	0.3	10863.6	2.1	140.9	1.8	0.4	3.8	0.7	9.8	0.02

Table S6
Results of the estimated metal leaching from tailings after electrodialysis (in scenario C).

ReCiPe perspective	Metals, mg/kg tailings (on dry solid basis)										
	Al	As	Ba	Ca	Cd	Co	Cr	Cu	Fe	K	Mg
Individualist	13.1	0	4.9	611.8	0	0	0.1	119.2	19.1	81.3	210.9
Hierarchist	64.5	0.2	38.7	662.6	0	0	1.2	141.4	1075.4	334.9	261.7
Egalitarian	91.4	0.2	44.5	702.2	0	0	2.4	205.8	1139.3	561.1	339.6

ReCiPe perspective	Metals, mg/kg tailings (on dry solid basis)										
	Mn	Mo	Na	Ni	P	Pb	Sb	Sr	V	Zn	Zr
Individualist	20.1	0	40.2	0.4	25.3	0.2	0.2	0.2	0.0003	5.1	0.002
Hierarchist	33.3	0	53.6	0.8	92.5	1.2	0.4	0.3	0.5	6.8	0.003
Egalitarian	35.7	0.3	9599.9	1.5	112.6	1.5	0.4	0.4	0.7	8.7	0.02

Table S7
The life cycle environmental impacts per kg of copper in concentrate in the baseline scenario A (ReCiPe midpoint/hierarchist).

Impact category[a]	Unit	Total	On-site emissions[b]	Blasting	Conveyor belt	Reagents	Diesel (truck)	Electricity	Heating (HFO)	Tailings (STD)
CC	kg CO_2 eq	6.9E-01	0	1.4E-01	1.9E-02	5.8E-02	3.6E-01	1.1E-01	3.7E-03	0
OD	kg CFC-11 eq	9.8E-08	0	7.3E-09	1.3E-09	7.7E-09	6.7E-08	1.4E-08	6.7E-10	0
TA	kg SO_2 eq	1.2E-02	0	8.7E-03	8.2E-05	3.1E-04	2.9E-03	3.4E-04	2.3E-05	0
FE	kg P eq	1.1E-04	0	2.9E-05	1.5E-05	1.8E-05	1.6E-05	3.0E-05	8.8E-08	0
ME	kg N eq	7.4E-04	0	4.9E-04	4.1E-06	5.9E-05	1.7E-04	1.6E-05	2.6E-07	0
HT	kg 1.4-DB eq	1.2E+01	3.7E-07	3.9E-02	2.4E-02	1.9E-02	2.0E-02	2.9E-02	3.9E-04	1.2E+01
POF	kg NMVOC	1.6E-02	0	1.0E-02	7.8E-05	3.3E-04	5.0E-03	2.7E-04	8.6E-06	0
PMF	kg PM_{10} eq	2.3E-02	1.8E-02	2.6E-03	7.4E-05	1.3E-04	1.5E-03	2.2E-04	5.9E-06	0
TET	kg 1.4-DB eq	8.2E-05	5.4E-10	4.8E-05	9.7E-06	5.3E-06	1.3E-05	5.6E-06	4.7E-07	3.3E-24
FET	kg 1.4-DB eq	4.0E-03	8.8E-10	9.6E-04	8.0E-04	5.0E-04	6.9E-04	1.0E-03	4.0E-06	2.6E-24
MET	kg 1.4-DB eq	3.9E+00	3.4E-05	9.7E-04	7.7E-04	4.8E-04	6.6E-04	9.5E-04	8.0E-06	3.9E+00
IR	kBq U235 eq	7.9E-02	0	4.3E-03	1.1E-03	4.9E-03	2.5E-02	4.3E-02	2.6E-04	0
ALO	m^2a	6.3E-02	0	8.0E-03	7.3E-04	1.3E-02	1.3E-03	3.9E-02	1.1E-05	0
ULO	m^2a	4.3E-03	6.7E-04	9.7E-04	2.6E-04	5.1E-04	8.0E-04	1.1E-03	5.8E-06	0
NLT	m^2	2.4E-04	0	1.9E-05	2.9E-06	1.2E-05	1.4E-04	6.6E-05	1.4E-06	0
WD	m^3	1.9E-01	6.6E-02	1.3E-03	2.7E-04	9.7E-04	7.3E-04	1.2E-01	6.3E-06	0
MRD	kg Fe eq	3.9E+01	3.9E+01	6.8E-03	2.2E-02	2.2E-03	1.2E-02	1.2E-02	1.9E-05	0
FD	kg oil eq	2.1E-01	0	2.2E-02	4.6E-03	2.7E-02	1.3E-01	3.0E-02	1.3E-03	0

[a] CC, climate change; OD, ozone depletion; TA, terrestrial acidification; FE, freshwater eutrophication; ME, marine eutrophication; HT, human toxicity; POF, photochemical oxidant formation; PMF, particulate matter formation; TET, terrestrial ecotoxicity; FET, freshwater ecotoxicity; MET, marine ecotoxicity; IR, ionizing radiation; ALO, agricultural land occupation; ULO, urban land occupation; NLT, natural land transformation; WD, water depletion; MRD; mineral resource depletion; FD, fossil depletion.
[b] The "on-site emissions" category did not include the on-site use of blasting, diesel, heavy fuel oil and natural gas, which were part of the corresponding cradle-to-gate processes used in the study. Besides, tailings were discussed separately

Table S8
The life cycle environmental impacts per kg of copper in concentrate in the energy-oriented scenario B (ReCiPe midpoint/hierarchist).

Impact category[a]	Unit	Total	On-site emissions[b]	Blasting	Conveyor belt	Reagents	Electricity (truck)	Electricity (facility)	Heating (NG)	Tailings (STD)
CC	kg CO_2 eq	3.5E-01	0	1.4E-01	1.9E-02	5.8E-02	2.1E-02	1.1E-01	3.0E-03	0
OD	kg CFC-11 eq	3.4E-08	0	7.3E-09	1.3E-09	7.7E-09	2.6E-09	1.4E-08	4.5E-10	0
TA	kg SO_2 eq	9.5E-03	0	8.7E-03	8.2E-05	3.1E-04	6.1E-05	3.4E-04	3.6E-06	0
FE	kg P eq	9.7E-05	0	2.9E-05	1.5E-05	1.8E-05	5.3E-06	3.0E-05	2.2E-07	0
ME	kg N eq	5.7E-04	0	4.9E-04	4.1E-06	5.9E-05	3.0E-06	1.6E-05	1.3E-07	0
HT	kg 1.4-DB eq	1.2E+01	3.7E-07	3.9E-02	2.4E-02	1.9E-02	5.3E-03	2.9E-02	1.7E-04	1.2E+01
POF	kg NMVOC	1.1E-02	0	1.0E-02	7.8E-05	3.3E-04	4.8E-05	2.7E-04	3.1E-06	0
PMF	kg PM_{10} eq	2.1E-02	1.8E-02	2.6E-03	7.4E-05	1.3E-04	3.9E-05	2.2E-04	1.4E-06	0
TET	kg 1.4-DB eq	6.9E-05	5.4E-10	4.8E-05	9.7E-06	5.3E-06	1.0E-06	5.6E-06	3.8E-08	3.3E-24
FET	kg 1.4-DB eq	3.4E-03	8.8E-10	9.6E-04	8.0E-04	5.0E-04	1.8E-04	1.0E-03	5.9E-06	2.6E-24
MET	kg 1.4-DB eq	3.9E+00	3.4E-05	9.7E-04	7.7E-04	4.8E-04	1.7E-04	9.5E-04	5.9E-06	3.9E+00
IR	kBq U235 eq	6.1E-02	0	4.3E-03	1.1E-03	4.9E-03	7.8E-03	4.3E-02	1.2E-04	0
ALO	m^2a	6.9E-02	0	8.0E-03	7.3E-04	1.3E-02	7.1E-03	3.9E-02	1.9E-05	0
ULO	m^2a	3.7E-03	6.7E-04	9.7E-04	2.6E-04	5.1E-04	2.0E-04	1.1E-03	2.2E-06	0
NLT	m^2	1.1E-04	0	1.9E-05	2.9E-06	1.2E-05	1.2E-05	6.6E-05	7.6E-07	0
WD	m^3	2.1E-01	6.6E-02	1.3E-03	2.7E-04	9.7E-04	2.2E-02	1.2E-01	4.9E-06	0
MRD	kg Fe eq	3.9E+01	3.9E+01	6.8E-03	2.2E-02	2.2E-03	2.1E-03	1.2E-02	2.0E-05	0
FD	kg oil eq	9.0E-02	0	2.2E-02	4.6E-03	2.7E-02	5.4E-03	3.0E-02	1.1E-03	0

[a] CC, climate change; OD, ozone depletion; TA, terrestrial acidification; FE, freshwater eutrophication; ME, marine eutrophication; HT, human toxicity; POF, photochemical oxidant formation; PMF, particulate matter formation; TET, terrestrial ecotoxicity; FET, freshwater ecotoxicity; MET, marine ecotoxicity; IR, ionizing radiation; ALO, agricultural land occupation; ULO, urban land occupation; NLT, natural land transformation; WD, water depletion; MRD; mineral resource depletion; FD, fossil depletion.
[b] The "on-site emissions" category did not include the on-site use of blasting, diesel, heavy fuel oil and natural gas, which were part of the corresponding cradle-to-gate processes used in the study. Besides, tailings were discussed separately

Table S9
Comparison of the life cycle environmental impacts between the electrodialytic tailings remediation process and scenarios A & C (ReCiPe midpoint/hierarchist).

Impact category[a]	Unit	Scenario A		Scenario C		Electrodialytic (acidic) remediation process in scenario C					
		Total	Tailings	Total	Tailings	Total	Sodium nitrate	Electricity	Polyamide (membrane)	Polysulfone (membrane)	Polyester (membrane)
CC	kg CO_2 eq	6.9E-01	0	7.4E-01	0	7.0E-02	6.9E-02	6.1E-04	3.4E-07	7.0E-05	1.0E-04
OD	kg CFC-11 eq	9.8E-08	0	9.9E-08	0	4.1E-09	4.0E-09	7.6E-11	6.5E-15	9.9E-12	9.8E-12
TA	kg SO_2 eq	1.2E-02	0	1.2E-02	0	2.8E-04	2.7E-04	1.8E-06	1.2E-09	2.9E-07	4.6E-07
FE	kg P eq	1.1E-04	0	1.2E-04	0	1.4E-05	1.3E-05	1.6E-07	3.2E-11	2.6E-08	2.8E-08
ME	kg N eq	7.4E-04	0	9.1E-04	0	1.9E-04	1.9E-04	8.8E-08	2.7E-10	1.6E-08	3.1E-08
HT	kg 1.4-DB eq	1.2E+01	1.2E+01	4.9E-01	3.5E-01	1.8E-02	1.8E-02	1.6E-04	2.3E-08	3.1E-05	5.2E-05
POF	kg NMVOC	1.6E-02	0	1.5E-02	0	1.5E-04	1.5E-04	1.4E-06	8.7E-10	3.0E-07	6.5E-07
PMF	kg PM_{10} eq	2.3E-02	0	2.2E-02	0	1.0E-04	9.9E-05	1.1E-06	4.8E-10	1.6E-07	2.2E-07
TET	kg 1.4-DB eq	8.2E-05	3.3E-24	8.5E-05	1.4E-23	5.6E-06	5.5E-06	3.0E-08	6.5E-12	1.0E-08	8.9E-08
FET	kg 1.4-DB eq	4.0E-03	2.6E-24	4.3E-03	3.7E-24	4.4E-04	4.3E-04	5.3E-06	1.0E-09	6.4E-07	7.7E-07
MET	kg 1.4-DB eq	3.9E+00	3.9E+00	9.4E+00	9.4E+00	4.4E-04	4.3E-04	5.1E-06	9.1E-10	6.5E-07	7.5E-07
IR	kBq U235 eq	7.9E-02	0	7.9E-02	0	3.3E-03	3.1E-03	2.3E-04	9.0E-09	4.5E-06	7.5E-06
ALO	m^2a	6.3E-02	0	6.3E-02	0	2.0E-03	1.8E-03	2.1E-04	9.3E-09	2.4E-06	8.1E-06
ULO	m^2a	4.3E-03	0	4.7E-03	0	3.9E-04	3.8E-04	6.0E-06	8.4E-10	5.8E-07	8.5E-07
NLT	m^2	2.4E-04	0	2.4E-04	0	8.8E-06	8.4E-06	3.5E-07	1.3E-11	6.4E-09	4.6E-08
WD	m^3	1.9E-01	0	1.8E-01	0	1.5E-03	8.3E-04	6.4E-04	8.0E-09	1.4E-06	1.3E-06
MRD	kg Fe eq	3.9E+01	0	3.7E+01	0	3.4E-03	3.4E-03	6.1E-05	1.7E-09	2.7E-06	5.0E-06
FD	kg oil eq	2.1E-01	0	2.2E-01	0	1.1E-02	1.1E-02	1.6E-04	1.1E-07	3.5E-05	3.3E-05

[a] CC, climate change; OD, ozone depletion; TA, terrestrial acidification; FE, freshwater eutrophication; ME, marine eutrophication; HT, human toxicity; POF, photochemical oxidant formation; PMF, particulate matter formation; TET, terrestrial ecotoxicity; FET, freshwater ecotoxicity; MET, marine ecotoxicity; IR, ionizing radiation; ALO, agricultural land occupation; ULO, urban land occupation; NLT, natural land transformation; WD, water depletion; MRD; mineral resource depletion; FD, fossil depletion.

Table S10
Results of Monte Carlo simulation for the life cycle environmental impacts of the scenarios A, B and C (ReCiPe midpoint/hierarchist).

Impact category[a]	Unit	Scenario A				Scenario B				Scenario C			
		Mean	Median	SD	CV (%)	Mean	Median	SD	CV (%)	Mean	Median	SD	CV (%)
CC	kg CO_2 eq	6.9E-01	6.8E-01	1.1E-01	16	3.5E-01	3.4E-01	7.5E-02	21	7.4E-01	7.3E-01	1.1E-01	14
OD	kg CFC-11 eq	9.8E-08	8.8E-08	4.3E-08	44	3.4E-08	3.2E-08	7.1E-09	21	9.8E-08	8.9E-08	4.1E-08	42
TA	kg SO_2 eq	1.2E-02	1.2E-02	3.1E-03	25	9.5E-03	9.0E-03	3.0E-03	32	1.2E-02	1.2E-02	3.0E-03	25
FE	kg P eq	1.1E-04	9.6E-05	5.0E-05	46	9.7E-05	8.7E-05	4.4E-05	45	1.2E-04	1.1E-04	5.1E-05	43
ME	kg N eq	7.4E-04	7.1E-04	1.8E-04	24	5.8E-04	5.5E-04	1.7E-04	30	9.1E-04	8.8E-04	1.8E-04	20
HT	kg 1.4-DB eq	1.2E+01	6.3E+00	1.9E+01	157	1.2E+01	6.3E+00	2.0E+01	168	5.0E-01	3.6E-01	4.7E-01	95
POF	kg NMVOC	1.6E-02	1.5E-02	3.8E-03	25	1.1E-02	1.0E-02	3.7E-03	34	1.5E-02	1.5E-02	3.8E-03	25
PMF	kg PM_{10} eq	2.3E-02	2.2E-02	6.2E-03	27	2.1E-02	2.0E-02	6.2E-03	29	2.2E-02	2.1E-02	6.1E-03	27
TET	kg 1.4-DB eq	8.2E-05	7.6E-05	3.0E-05	37	6.9E-05	6.3E-05	2.9E-05	41	8.5E-05	7.9E-05	2.9E-05	33
FET	kg 1.4-DB eq	3.9E-03	3.6E-03	1.3E-03	33	3.4E-03	3.2E-03	1.2E-03	35	4.3E-03	4.0E-03	1.5E-03	36
MET	kg 1.4-DB eq	4.0E+00	2.4E+00	5.1E+00	130	4.0E+00	2.4E+00	5.7E+00	144	9.7E+00	5.1E+00	1.7E+01	172
IR	kBq U235 eq	7.8E-02	6.6E-02	4.5E-02	57	6.2E-02	4.7E-02	5.2E-02	85	7.9E-02	6.7E-02	4.9E-02	62
ALO	m2a	6.3E-02	6.2E-02	1.1E-02	18	6.9E-02	6.7E-02	1.2E-02	18	6.3E-02	6.2E-02	1.1E-02	18
ULO	m2a	4.3E-03	4.2E-03	7.8E-04	18	3.7E-03	3.6E-03	7.1E-04	19	4.7E-03	4.6E-03	7.9E-04	17
NLT	m^2	2.4E-04	2.3E-04	2.6E-04	108	1.1E-04	1.0E-04	2.7E-04	251	2.4E-04	2.4E-04	2.5E-04	105
WD	m^3	2.2E-01	3.6E-01	1.1E+00	508	1.9E-01	3.5E-01	1.3E+00	699	1.7E-01	2.9E-01	1.1E+00	645
MRD	kg Fe eq	3.9E+01	3.9E+01	1.3E+00	3	3.9E+01	3.9E+01	1.3E+00	3	3.7E+01	3.7E+01	1.2E+00	3
FD	kg oil eq	2.1E-01	2.1E-01	4.2E-02	20	9.0E-02	8.8E-02	1.4E-02	16	2.2E-01	2.1E-01	4.2E-02	19

[a] CC, climate change; OD, ozone depletion; TA, terrestrial acidification; FE, freshwater eutrophication; ME, marine eutrophication; HT, human toxicity; POF, photochemical oxidant formation; PMF, particulate matter formation; TET, terrestrial ecotoxicity; FET, freshwater ecotoxicity; MET, marine ecotoxicity; IR, ionizing radiation; ALO, agricultural land occupation; ULO, urban land occupation; NLT, natural land transformation; WD, water depletion; MRD; mineral resource depletion; FD, fossil depletion.
[b] SD, standard deviation; CV, coefficient of variation.

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Highlights:

- A cradle-to-gate LCA is carried out on an underground copper ore mine in Norway.
- A geochemical approach is used to estimate the metal leaching potential of tailings.
- The potential impacts of tailings before and after electrodialytic remediation are compared
- The marine ecotoxicity impact of tailings is sensitive to different ReCiPe perspectives.
- A trade-off exists between extracting copper from tailings and the impacts of tailings.