Life Cycle Assessment (LCA) of Mining Waste Facility (MWF) Operations

A Case Study of Reprocessing Iron Tailings in Sweden

Master's thesis in Indutrial Ecology

XI XIAO

DEPARTMENT OF TECHNOLOGY MANAGEMENT AND ECONOMICS
DIVISION OF ENVIRONMENTAL SYSTEMS ANALYSIS
CHALMERS UNIVERSITY OF TECHNOLOGY
Gothenburg, Sweden 2025
www.chalmers.se

Master's thesis 2025

Life Cycle Assessment (LCA) of Mining Waste Facility (MWF) Operations

A Case Study of Reprocessing Iron Tailings in Sweden

XI XIAO

Department of Technology Management and Economics
Division of Environmental Systems Analysis
Chalmers University of Technology
Gothenburg, Sweden 2025

Life Cycle Assessment (LCA) of Mining Waste Facility (MWF) Operations
A Case Study of Reprocessing Iron Tailings in Sweden
XI XIAO

Supervisor: Kanishk Bhadani, Industrial and Materials Science
Examiner: Gauti Asbjörnsson, Industrial and Materials Science

Master's Thesis 2025
Department of Technology Management and Economics
Division of Environmental Systems Analysis
Chalmers University of Technology
SE-412 96 Gothenburg
Telephone +46 31 772 1000

Typeset in LaTeX
Printed by Chalmers Reproservice
Gothenburg, Sweden 2025

Life Cycle Assessment (LCA) of Mining Waste Facility (MWF) Operations
A Case Study of Reprocessing Iron Tailings in Sweden
XI XIAO
Department of Technology Management and Economics
Chalmers University of Technology

Abstract

Historical tailings, once considered waste due to technological limitations, are now increasingly recognised as potential secondary sources of valuable materials. This study applies an ex-ante and prospective Life Cycle Assessment (LCA) to evaluate the environmental impacts of reprocessing historical iron tailings in central Sweden, under current technologies and projected future scenarios. The tailings are characterised by a high content of apatite, along with iron (Fe) and rare earth elements (REEs) such as monazite, presenting opportunities for recovery as secondary resources. Given that apatite is the primary mineral source of phosphorus in the form of P_2O_5, a critical nutrient for agriculture and food production, this study focuses on apatite recovery.
The results show that reprocessing using current technology yields environmental benefits in several impact categories, including ecotoxicity (freshwater and marine), human toxicity (non-carcinogenic), and material resource use, as assessed by the ReCiPe 2016 method. However, the process also introduces environmental burdens, particularly in the categories of climate change, land use, and water use. Transitioning to fully renewable electricity significantly reduces these burdens and could lead to fossil-free energy demand. This comprehensive assessment of the environmental implications of tailings reprocessing provides valuable insights to support circular economy strategies to extend material life cycles by reusing mining residues.

Keywords: Tailings, Iron ore, Apatite, REEs, Life cycle assessment, Circular economy, Climate change, Secondary materials

Acknowledgements

I want to express my heartfelt thanks to my supervisor, Kanishk Bhadani, for his continuous guidance and support throughout this project. His valuable feedback has significantly improved the quality of my work.
I also want to thank Grängex company for providing essential information and Michel Varelija from the Technical University of Leoben for her guidance on the mining exploration plan.
Lastly, I am deeply grateful to my husband for his unwavering support, and to my son for the joy and sweet moments that have carried me through this journey.

Xi Xiao, Gothenburg, May 2025

List of Acronyms

Below is the list of acronyms that have been used throughout this thesis listed in alphabetical order:

ALCA	Attributional LCA
CED	cumulative energy demand
CLCA	Consequential LCA
CRM	Critical Raw Materials
CTUe	Comparative Toxic Unit for ecotoxicity
CTUh	Comparative Toxic Unit for human health
FDP	fossil depletion potential
FU	functional unit
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life cycle impact assessment
MWF	Mining Waste Facility
REE	Rare Earth Element
TRACI	Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts
TRL	Technology Readiness Level

Contents

List of Figures

List of Tables

1
Introduction

1.1 Background

Mining has a long history, and as humanity faces an impending shortage of critical resources, sustainable resource management is becoming increasingly important (Farjana et al., 2019). In the European Union, the circular economy has gained significant attention, emphasizing the reuse and recycling of materials to extend their life cycles. Within this trend, the mining industry is exploring ways to utilise the Earth's resources more efficiently—reducing emissions and energy consumption while still meeting societal demands. A key area of interest in recent years is the reprocessing of mining waste, which seeks to recover valuable materials, such as metals and other byproducts, from previously untapped resources. Historical tailings, once considered waste due to past technological limitations, are now being revisited as potential secondary sources of valuable compounds. Among these tailings, the deposits in the Grängesberg site, located in central Sweden, were identified, which contain significant amounts of apatite and monazite(Lindholm, 2022). Apatite, a phosphate mineral, is the primary source of phosphorus, a critical nutrient essential for agriculture, food production, and various industrial applications. The European Union heavily relies on phosphorus imports, making the recovery of this element from domestic sources particularly valuable (Blengini et al., 2019). However, beyond just assessing its extraction potential, it is essential to evaluate the process from a systemic perspective. LCA studies of mine tailings treatment generally find that waste reprocessing and valorization strategies tend to reduce environmental impacts compared to conventional tailings management, but not always (Adrianto & Pfister, 2022). As this research focuses on producing phosphate fertilizers rather than recovering high-purity phosphorus, the aim is to develop an LCA framework to analyse the environmental impact of phosphate recovery in the Grängesberg site, considering emerging resource recovery technologies. By assessing the sustainability of the extraction of phosphate from historical tailings, this study contributes to broader efforts to improve resource efficiency and initiatives in the circular economy within the mining industry.

LCA is a well-established method for conducting a comprehensive environmental evaluation and avoiding sub-optimisation. It has been widely used to assess the environmental impacts of various products and systems since the early 1990s. However, the LCA application in the mining area was still limited until the 2010s (Farjana et al., 2019). The limited number of mining LCAs hindered the development of life-cycle inventories since many products are directly or indirectly produced from mining. With sustainability becoming a hot topic in recent years, and Climate change being a problem that cannot be ignored anymore, environmental concerns in the mining area are gaining more attention than ever. Therefore, the LCA literature in this area is becoming substantial.

Several studies have explored the application of Life Cycle Assessment (LCA) in evaluating the recovery of valuable materials from mining tailings. Research has primarily focused on metal mine tailings such as copper(Adrianto et al., 2023), gold(Cairncross & Tadie, 2022), and other valuable minerals, assessing various reprocessing scenarios and their environmental impacts. These studies incorporate emerging resource recovery technologies, market supply-demand forecasts, and energy system transitions to evaluate the environmental feasibility of different reprocessing routes. The studies show that technological advancements, secondary material market penetration, and the selection of displaced products influence the environmental performance of tailings reprocessing. In addition to metal recovery, a recent study has examined LCA applications in phosphorus mining, analyzing the environmental footprint of phosphorus extraction and refining processes (Rachid et al., 2025). The study primarily focused on conventional phosphorus mining operations and their environmental impacts, emphasizing the need for more sustainable phosphorus management strategies.

1.2 Aim and objectives

The overall aim of the project is to develop an LCA framework through a case study of the recovered apatite at the Grängesberg site and compare the results with those of the primary production. Therefore, the result will give guidance to tailings management and mining disposal.

1.3 Scope and limitations

- Sustainability Scope: While sustainability analysis typically encompasses environmental, social, and economic aspects, this thesis focuses solely on environmental factors. Social and economic considerations, though influential in sustainability outcomes, are not included in this study.
- Life Cycle Assessment (LCA) Boundaries: LCA provides a systematic approach to assessing environmental impacts within a defined system. In this study, the focus is on the tailings left in the Grängesberg site; therefore, the assessment is limited to the environmental impacts associated with the recovery of organic phosphate fertiliser, as P_2O_5, specifically examining the physical and chemical processes involved in its production. The use phase and disposal processes are beyond the scope of this research.
- Site-Specific Considerations: Observations from the site visit indicate that the tailings are currently covered by forest. However, as this thesis aims to develop a general LCA framework for tailings recovery, the environmental impacts of forest removal are not considered in the analysis.
- Data Assumptions: Missing data related to the process will be estimated using relevant references.

1.4 Research strategy

The research strategy followed to fulfil the aims and objectives of this study can be divided into three parts:

- Reviews of the literature on the mining area, the processing of mining waste, and the LCA methodology choices are presented to develop a recovery production flow based on current technologies.
- The processes involved in mining recovery and the technologies implemented at the Grängesberg site are identified and compared. Tailings composition data are obtained directly from the Grängesberg company. In addition, process-related information on the extraction of apatite is gathered. Laboratory data on key unit operations, such as crushing, screening, flotation and dewatering, are collected and used as a basis for scaling up to the industrial level. The obtained data were used as common inventory data for the environmental impact assessment according to the defined system boundary.
- The ecoinvent database is used for the background data. LCA modelling using openLCA software, as well as calculation and interpretation of the results.

2

Literature review

With growing concerns over sustainability, the mining industry, the cornerstone of material supply across multiple sectors, has gained increasing attention in recent years. A key issue is mining waste management, which refers to tailings and poses immediate and long-term environmental challenges. Tailings disposal can cause short-term impacts (ranging from hours to months), affecting water and sediment quality and aquatic and human life for hundreds of kilometers downstream. In the medium to long term (years to centuries), tailings can lead to the contamination of soils and sediments (Kossoff et al., 2014).
Proper handling of mining waste contributes to environmental restoration and presents opportunities for the recovery of critical raw materials (CRM) from secondary sources. Rather than landfilling, emerging technologies for waste reprocessing are gaining traction globally, particularly in Europe (Blengini et al., 2019). These efforts align with the principles of the circular economy, which emphasize closing material loops by reducing waste and increasing recycling efficiency.
In response to these developments, both policymakers and researchers have begun to adopt advanced analytical tools (Muller et al., 2023; Saldanha et al., 2023) and technologies to evaluate and optimize recovery strategies(Alagha & Alsafasfeh, 2017; Valderrama et al., 2024). Among these, Life Cycle Assessment (LCA) has emerged as a widely used and effective methodology for assessing the environmental performance of mining waste recovery and management systems.

2.1 Types of LCA approaches

LCAs are typologized into different groups according to time, the maturity of the technology that is used, and causality.

2.1.1 Retrospective and prospective LCA

There are two main categories of LCA, retrospective and prospective (Tillman, 2000). The former is considered technology approximately as they did at the time of the study. The latter is used to analyse emerging technologies in imagined states at future points in time (Thonemann et al., 2020). Several LCAs are explicated according to the maturity of the technology used. Ex-post LCA refers to the technology that is mature enough at the time of study. Ex-ante LCA, in contrast, considers the low technology readiness level (TRL) or just on a lab scale at the time of study and tries to scale it up using likely scenarios (e.g. using expert help, extreme views, learning curves for similar technologies) of future performance at full operational scale(Cucurachi et al., 2018). Ex-ante/post-LCA and retro/prospective LCA are classified according to the time dimension and technology readiness, which can be combined to scale up foreground data and project background data(i.e., cleaner energy systems, bio-based fuels) for the future to analyse (Fig 2.1)(de Souza et al., 2023; Sacchi et al., 2022).

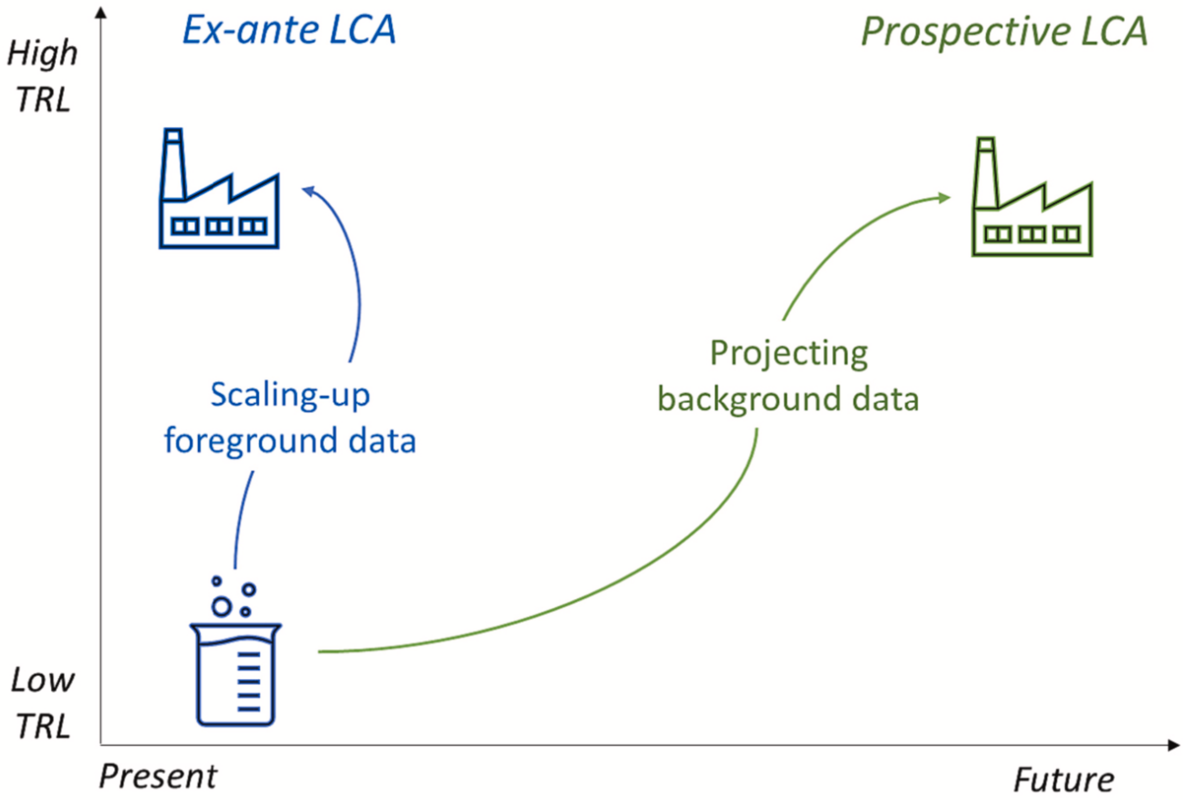

Figure 2.1: Ex-ante LCA and prospective LCA(de Souza et al., 2023).

2.1.2 Attributional and consequential LCA

There are two other important categories in the LCA family: Attributional LCA(ALCA) and Consequential LCA(CLCA); they are defined by the study's goal related to different questions. ALCA aims to describe the environmentally relevant physical flows to and from a life cycle and its subsystems. CLCA describes how environmentally relevant flows will change in response to possible decisions(Ekvall, 2019).
A clear example provided by Ekvall (Ekvall & Andrae, 2006) illustrates how ALCA and CLCA were used to analyse the shift from SnPb to lead-free solders. The study clarified that assessing the change in environmental burdens within the technosphere resulting from the transition to lead-free solder requires CLCA, as it must consider how relevant markets are affected. In contrast, ALCA does not need to address these market dynamics. It also explains why CLCA should ideally use marginal data for many unit processes, whereas ALCA typically relies on average data.

2.1.3 Process-oriented and product-oriented LCA

This category of LCA studies typically emerges when practitioners are primarily concerned with a specific process or a product. As noted by Schrijvers et al. (2020), interest in the cradle-to-gate environmental impacts of recycled materials is often driven by two primary motivations. First, an LCA practitioner may seek to determine whether recycling

a particular material yields environmental benefits compared to non-recycling or alternative recycling routes. Additionally, the practitioner might aim to identify environmental hotspots within the recycling process to improve overall efficiency and sustainability. In such cases, the focus lies on optimizing the recycling procedure rather than evaluating the end use of the recycled material, thereby conducting a process-oriented LCA.
Conversely, an LCA practitioner may intend to incorporate recycled material into a product and thus needs to understand the environmental implications associated with using recycled inputs. Here, the emphasis is on the environmental impacts tied to the consumption of recycled materials within a product system, justifying using a product-oriented LCA. This distinction between process-oriented and product-oriented LCA provides a clearer understanding of the study's purpose and analytical scope.

2.2 LCA methodological components

The ISO standards ISO14040:2006 state the requirements of conducting an LCA for the practitioners. There are four steps in an LCA study:

1. Goal and Scope: The boundary of the study system, the purpose of the study and the audience should be defined. The depth and breadth of LCA can vary considerably depending on the goal of the study, so this step must be clearly described in the beginning. In this step, the functional unit has to be determined. It corresponds to a reference flow to which all other modelled system flows are related, so it needs to be quantitative. The system boundary includes the time boundary, the geography boundary, and the technical boundary.
2. Inventory analysis: The study must clarify the input/output data concerning the system; it is data collection for the study's goal.
3. The Impact Assessment: The purpose of the step is to provide additional information to help assess a product system's LCI results to give a better understanding of the environmental issue. As many impact assessment methods exist, selecting the one that best fits the goal is necessary.
4. Interpretation: This is the result of the assessment; it should give a conclusion, recommendation to stakeholders, or for decision-making, based on the goal and scope. Meanwhile, it interacts with other steps along the way.

2.2.1 Goal and scope

Many LCA studies in the mining industry adopt a cradle-to-gate approach (Cairncross & Tadie, 2022; KONARÉ et al., 2024; Lu et al., 2023; Rachid et al., 2025), where all production processes are considered—from material extraction to the delivery of the final product. The scope of these studies focuses on the environmental impacts associated with these stages. Due to the specific nature of mining products, which are typically used as inputs for further manufacturing, few studies adopt a cradle-to-grave approach, including the use phases and end-of-life. In mining waste recovery, the approach is defined as from-grave-to-gate(Adrianto & Pfister, 2022; Muller et al., 2023), meaning that waste from processing facilities is reused as input material. The functional unit is typically defined as the mass of material produced or considering the impact of the process(i.e., disposal of 1 kg waste). According to the ISO standard, temporal and geographical boundaries should be clearly specified; however, not all studies provide this information explicitly (See Table 2.1).

Table 2.1: Summary of LCA scope in mining studies

Reference	System Boundary	Temporal Boundary	Geographical Boundary	Functional Unit
(Rachid et al., 2025)	cradle-to-gate	2019 to 2022	Gantour Basin, Morocco	Production of 1 kg of phosphorus pentoxide (31%P_2O_5)
(KONARÉ et al., 2024)	cradle-to-gate	Not specified	Mali	Production of 1 kg of gold
(Adrianto & Pfister, 2022)	grave-to-gate	Not specified	Portugal	Disposal of 1 tonne of sulfidic mine tailings and production of materials and other by-products according to reprocessing routes
(Cairncross & Tadie, 2022)	cradle-to-gate	Not specified	South Africa	1 kg of gold produced
(Lu et al., 2023)	cradle-to-gate	2017 to 2018	China	Production of one ton of standard Cu
(Muller et al., 2023)	grave-to-gate	Not specified	Not specified	The management of 1 ton of solid waste resulting from coal processing and the production of 1 precast concrete panel

2.2.2 Life cycle inventory and data source

The Life Cycle Inventory (LCI) was developed by compiling a database within the defined system boundaries, encompassing material and energy inputs, products, and emissions to the ecosphere (Rebitzer et al., 2004). Particular attention must be paid to mass and energy balances to ensure data consistency and completeness. All relevant processes within the system must be included. In this step, three key components should be clearly defined and documented.

- Construction of a flowchart: A flowchart details all process steps, reactors, auxiliary inputs, machinery, and equipment necessary for each operational unit. This flowchart illustrates the key steps of the production process and serves as the foundation for scaling-up procedures. Side streams (i.e., co-products, wastes) must be considered. Identifying co-products or by-products helps minimize the environmental impacts attributed to the main product and enhances the system's overall environmental and economic performance (de Souza et al., 2023).
- Foreground and background data: Input and output flows are categorised into foreground data and background data (Stavropoulos et al., 2016). Foreground data refers to the specific process chain being modelled and is directly related to the product and production system under study. In contrast, background data includes

generic information on materials, energy, transportation, and waste management systems (Goedkoop et al., 2014).

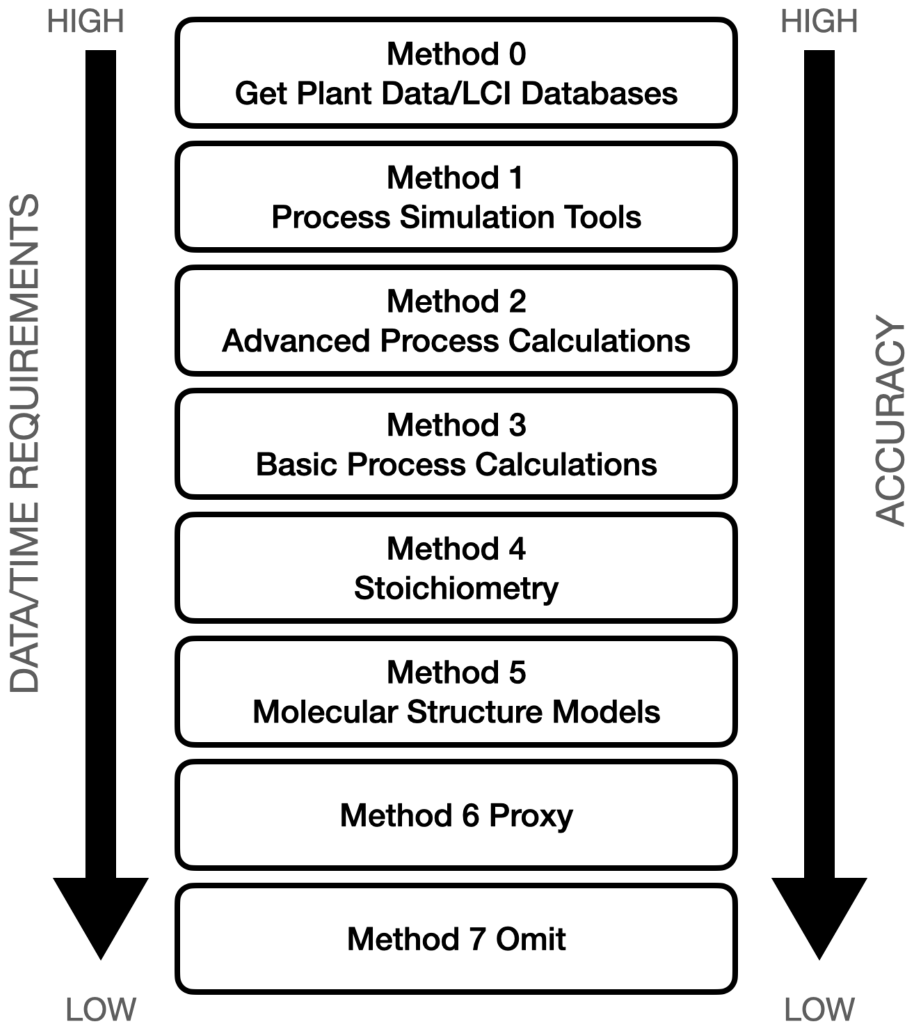

Figure 2.2: Hierarchy of methods used in LCI generation of chemicals concerning the data/time requirements and accuracy.

- Data sources and quality: The life cycle inventory is usually considered the most challenging part of LCA. Depending on which data is used, the model to be built will give different views of reality. The data's accuracy also determines the work's difficulty and workload. The data quality is evaluated by its relevance, reliability and accessibility (Flemström & Pålsson, 2003). The relevance usually refers to the geography and time-related coverage. The reliability is about where the data is from; the more close to the first-hand, the more precise. At the same time, data reliability also depends on the consistency with which it has been collected and documented. Transparent documentation gives more credit to the data, which means the reproducibility of the results(Baumann & Tillman, 2004). There is a hierarchy of methods used in LCI generation of chemicals concerning the data/time requirements and accuracy in Fig 2.2, the method 0 is to get the plant data directly, which is considered the highest accuracy. Conversely, using a 'similar' chemical as a proxy or just omitting it is supposed to be the least accurate method.

LCA studies typically collect data directly from operations and processing plants, which refers to primary data, supplemented by information from existing literature (Adrianto & Pfister, 2022; Rachid et al., 2025). In many cases, generic and estimated data are also used when necessary. Generic data represents average values based on preliminary or aggregated information. When specific data cannot be obtained directly from industry sources, it is often estimated using simulation tools, engineering calculations, or assumptions(Tampubolon et al., 2021).
With the advancement of LCA, numerous datasets (e.g., Ecoinvent, Sphere) have been developed to support practitioners and are now integrated into various software tools such as openLCA, GaBi®, and SimaPro. While primary data from processing plants is considered the most reliable, it can be challenging to obtain for emerging technologies that have not yet been implemented. In such cases, process simulation tools are used as alternatives and can provide data of comparable quality. The study by (Cairncross & Tadie, 2022), only uses a process simulation tool, Outotec HSC Chemistry 9®, to compile metallurgical process databases and model mass and energy balances for each flowsheet (e.g., cyanide and thiosulfate flowsheets). Meanwhile, Cairncross and Tadie (2022) manually constructed missing background process datasets in GaBi® based on information from the literature.

Table 2.2: Summary of LCA inventory data in mining studies

Reference	**Main Product / process**	**Data Source**	**Database/Software**
(Rachid et al., 2025)	Phosphorus	primary data, generic data	GaBi
(Tampubolon et al., 2021)	Coal	primary data, generic data, and estimation data	openLCA, GaBi and SimaPro
(Grzesik et al., 2019)	REEs	primary data, estimation data	ecoinvent, SimaPro
(Broadhurst et al., 2015)	tailing treatment process	primary data, literature information, mass balance calculations	ecoinvent,SimaPro

2.2.3 Life cycle impact assessment methods

Several well-established Life Cycle Impact Assessment (LCIA) methods are widely applied in mining-related LCA studies. Among the most recognized are the CML-IA (Centre for Environmental Science, Leiden University) method(Guinée, 2002), the ReCiPe method, TRACI (Tool for the Reduction and Assessment of Chemical and other Environmental Impacts), ILCD (International Reference Life Cycle Data System), CED (Cumulative Energy Demand), and the IPCC (Intergovernmental Panel on Climate Change) method(Farjana et al., 2019). These methods differ in their characterization models, normalization and weighting factors, impact categories, and geographic applicability.
The ReCiPe method is currently one of the most widely used approaches worldwide. It includes 18 midpoint impact categories and three endpoint categories(Human health, Ecosystems, and resources). Midpoint indicators such as global warming, ozone depletion, human toxicity, freshwater ecotoxicity, acidification, and eutrophication are widely adopted in LCA studies within the mining sector. However, several researchers have noted

that the standard impact categories are not always sufficient to entirely capture the specific environmental burdens associated with mining activities. Additional indicators, such as land use, water use (Cairncross & Tadie, 2022; Rachid et al., 2025), cumulative energy demand (CED), and fossil depletion potential (FDP) (Adrianto & Pfister, 2022), have been recommended as equally important. These indicators provide a more nuanced understanding of resource depletion and local environmental stresses, often significant in mineral extraction processes. Different methods used in the studies are summarised in Table 2.3.

Table 2.3: Summary of LCIA methods in mining studies

Reference	**Study**	**Focused Impact Categories**	**Methods**
(Adrianto & Pfister, 2022)	Sulfidic copper tailings reprocessing	Climate change, cumulative energy demand (CED), ecotoxicity, particulate matter formation potential (PMFP)	ReCiPe 2016, CED, USEtox
(Cairncross & Tadie, 2022)	Gold mine tailings recovery	Land use, water use	Recipe 2016
(Rachid et al., 2025)	Phosphate mining and beneficiation	18 midpoints impact categories, Water use	Recipe 2016, Available water remaining(AWARE)
(Broadhurst et al., 2015)	Desulfurisation flotation process	climate change, fossil fuel depletion, terrestrial acidification, human toxicity, eco-toxicity, urban land occupation and natural land transformation	ReCiPe 2016, USEtox

2.2.4 Allocation procedure

According to ISO 14044 and standard LCA practice, when a process yields more than one product or output stream, the environmental burdens associated with the process must be allocated among the various outputs. This is particularly important in the context of multifunctional processes such as mining waste reprocessing, where co-products and residues are generated simultaneously.

Several allocation procedures are available, including subdivision, system expansion and substitution. According to ISO 14044, multifunctionality should first be addressed by subdivision of the multifunctional processes. If subdivision is not feasible, system expansion should be applied by incorporating the co-functions into the functional unit. If allocation cannot be avoided, the partitioning of inventory across multiple functions is applied based on a chosen allocation criterion. According to(International Organization for Standardization (ISO), 2006), this criterion should follow the order of physical properties (e.g., mass); economic values (e.g., market value of the scrap material or recycled material in relation to market value of primary material); and the number of subsequent uses of the recycled material (Klöpffer, 2002).

It is common for the same process in the mining industry to produce multiple products, but other elements always exist concomitantly. Depending on its economic value, such further products are seen either as co-products (i.e. having a similar economic value as the wanted product) or a by-product (i.e. having a clearly lower economic value)(Stavropoulos et al., 2016). Depending on its economic value, such further products are seen either as co-products (i.e. having a similar economic value as the wanted product) or a by-product (i.e. having a clearly lower economic value)(Stavropoulos et al., 2016). For recycled materials, the choice of allocation procedure depends on the LCA approach. In attributional, process-oriented LCA, the life cycle inventory (LCI) of the foreground subsystem is calculated using attributional background data. When system expansion is applied, allocating environmental burdens among multiple outputs in the foreground system is unnecessary, as the system boundary is broadened to include the alternative functions or products provided, such as the displacement of virgin materials. This approach is discussed by Schrijvers et al. (2020), the Fig 2.3 categorises and groups the types of LCAs and the allocation procedures(Schrijvers et al., 2020).

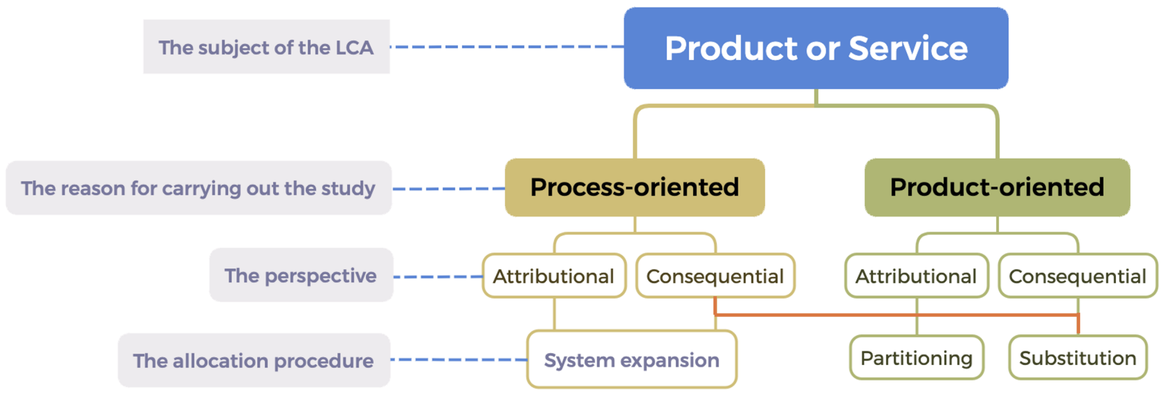

Figure 2.3: The allocation procedures for different purposes of LCAs.

Overall, allocation can significantly influence LCA results, making the selection of an allocation procedure an important step in the LCA process.

2.2.5 Foreground and background system

Foreground systems include processes that are under the direct influence of the decision-maker. In contrast, background systems refer to parts of the product system that cannot be directly affected by the decision-maker. The distinction between foreground and background systems is less emphasized in Attributional LCA (ALCA), as ALCA considers all life cycle stages within the system equally (Baumann & Tillman, 2004).
However, in prospective LCA, where future scenarios involving emerging technologies are explored, the modeling and scale of the foreground system become important. Additionally, the background system at a future point in time should also be considered. It is essential to ensure temporal consistency between the foreground and background systems to avoid mismatches that could compromise the validity of the results(Arvidsson et al., 2018).

2.2.6 Uncertainty and sensitivity analysis

When analysing data quality and interpreting the results in LCA, uncertainty and sensitivity analyses are essential tools for interpreting results and ensuring reliability. Uncertainty arises from the lack of knowledge about the true value of a quantity, and can be addressed through improved measurement accuracy or expert judgment.
However, a parameter can have substantial uncertainty yet still contribute insignificantly to the overall uncertainty. This is identified by assessing the uncertainty of a parameter, either qualitatively or quantitatively, and integrating this information with a sensitivity analysis (Björklund, 2002).
Sensitivity analysis focuses on evaluating how variations in assumptions, data, or methodological choices impact LCA outcomes. The ISO 14044 standard recommends concentrating on the most significant issues to determine how these variations influence the results (International Organization for Standardization (ISO), 2006). Scenario sensitivity analysis is widely used, where different system boundaries, allocation methods, technologies, or characterization approaches are tested to explore the influence of input parameters. Ratio sensitivity analysis is often applied in comparative LCAs, measuring how much an input must change to reverse the preference between alternatives (Björklund, 2002).
Guo and Murphy (2012) emphasized the importance of combining statistical techniques with scenario analysis to improve confidence in LCA results. Their study used case-specific modeling and Monte Carlo simulations to assess uncertainty ranges in various impact categories. They argued that LCAs lacking explicit interpretation of uncertainty and sensitivity offer limited value for decision-making. In the mining LCA studies, these two analyses are particularly necessary due to the complicated processes and the data resources.

2.3 Summary and gaps

In summary, the above has reviewed a broad range of literature on the application of Life Cycle Assessment (LCA) in the mining industry. Numerous studies have applied LCA methodologies to evaluate the environmental impacts of various mining processes over the past decades. For example, Awuah-Offei and Adekpedjou (2011) discusses key challenges that hinder the effective use of LCA in mining, such as limited awareness of LCA, ambiguity in defining functional units and system boundaries, and difficulties in addressing uncertainty and sensitivity analysis. In contrast, Farjana et al. (2019) provides a comprehensive review of LCA applications in the sector, identifying major environmental hotspots—particularly in relation to global warming potential and human health impacts—and highlighting the importance of integrating renewable energy sources to improve the sustainability of the industry.
Mine tailings management plays a critical role in protecting the environment and human health from the risks associated with mining residues. Conducting LCA on tailings management is useful for decision-makers when evaluating different management strategies (Adiansyah et al., 2015). One emerging strategy is the recovery of valuable materials from mine tailings, which has gained increasing attention in recent years as part of the circular economy transition toward a more sustainable future. As a result, researchers have begun to explore the potential for reusing mining waste through emerging technologies, with LCA used to assess the environmental implications of these processes (Adrianto & Pfister, 2022; Adrianto et al., 2023; Alagha & Alsafasfeh, 2017; Cairncross & Tadie,

2022).
However, although this area of research is growing, it remains in its early stages. To date, no existing studies have specifically addressed phosphorus recovery from iron tailings containing apatite from an LCA perspective. In addition, the modeling of multi-product systems, such as those found in Mining Waste Facilities (MWFs), remains underexplored in current LCA practice. Specifically, the treatment of environmental benefits and liabilities within the Life Cycle Inventory (LCI) phase for MWFs lacks sufficient methodological development. Addressing these gaps could lead to more robust and meaningful assessments of circular strategies within the mining industry.

2.4 Link to the study

Given the substantial phosphorus content found in the Grängesberg tailings—primarily in the form of apatite, evaluating the recovery processes using Life Cycle Assessment (LCA) could provide critical insights for sustainable resource management in both the mining and agricultural sectors. This context highlights the need for further investigation into the life cycle implications of phosphorus recovery from mining residues.
Recycling mining tailings requires developing and implementing novel technologies and processes, many of which are still under research and exploration. In such cases, conventional LCA is not suitable for modelling future scenarios. Therefore, a prospective or ex-ante LCA approach addresses this limitation. Simultaneously, this study focuses on conducting an attributional, process-oriented LCA of apatite recovery, aiming to assess the environmental impacts of the reprocessing tailings associated with the production of secondary P_2O_5 and Fe. Previous research has guided the implementation of LCA methodologies and the selection of relevant Life Cycle Impact Assessment (LCIA) indicators.
Given that tailings contain various elements depending on their geographical origin, this study also contributes to the development of a more generalized LCA framework for mining waste recovery processes.

3

Applied methodology in case study

3.1 Feed tailings characteristics

The study bases the characteristics of waste input on the Lindholm (2022), the repository is estimated to contain 3,46 dry Mtonnes of tailings in total, with an average content of 5,46% P_2O_5 and 9,69% Fe, along with a certain amount of REE such as monazite, which needs further characterization. The deposit is classified as an indicated category of mineral resources. The results of metallurgical test work show that the apatite can be recovered to a concentrate with 37,3% P_2O_5, with a recovery of 76, 7%.

3.2 Goal and scope

The attributional, process-oriented LCA is applied to the environmental impacts of apatite extraction and reprocessing from historical tailings at the Grängesberg site in Sweden. By comparing the environmental impacts of primary 32% P_2O_5 production with those of the recovered product, the study explores the potential for substituting recovered Fe ore concentrate and 32% P_2O_5. It also supports the development of a comprehensive LCA framework for evaluating mining waste recovery processes.
The tailings were generated during the historical mining and beneficiation of iron ore several decades ago and are now covered by vegetation (Lindholm, 2022). Therefore, the environmental impact burden of the landfilled tailings is set to zero. In this study, the developed LCA model represents the reprocessing route incorporating novel technology to recover metals from waste streams, based on the data presented in the Grängesberg report (Lindholm, 2022), which is summarised as a flowsheet in Fig 3.1. The main outputs of the process are iron ore concentrate and apatite (containing 17% P), and the results are compared with two reference cases: primary phosphate production(32% P_2O_5), primary Fe ore production and the direct discharge of residue. So the output will be converted to 32% P_2O_5 in order to compare. Meanwhile, the environmental impact of producing 1 ton of apatite(32% P_2O_5) will be analysed as it is one of the main product that is paid attention. Thus, the functional units (FUs) of the systems are the sum up of all functions/ services provided. In summary, the FUs include:

- The treatment of 1 tonne tailings
- The production of 133 kg apatite(32% P_2O_5)
- The production of 60 kg iron ore concentrate

The calculation of the conversion from Apatite(17%P) to 32% P_2O_5 is in the appendix B. The temporal boundary of this study is 1 year.
The system expansion approach is applied regarding the reprocessing to account for credits associated with avoided production processes (Adrianto & Pfister, 2022; Schrijvers et al., 2020). In this context, the tailings under investigation were generated historically and

therefore represent the end of a previous product life cycle. As such, the environmental burden associated with their disposal is assumed to be zero. The partitioning approach is applied regarding the production of apatite (32% P_2O_5).

The reference scenario is defined to deliver the same material outputs, specifically, iron ore and phosphate rock(32% P_2O_5), via conventional primary production pathways. Accordingly, the net environmental performance of the reprocessing route, relative to this reference, is calculated as follows:

$$\begin{aligned} \textit{Net Environmental Performance of the study vs. reference} = &\sum \textit{(Total Reprocessing Impacts)} \\ &- \textit{Impacts of Primary Fe production} \\ &- \textit{Impacts of Primary } \mathrm{P_2O_5} \textit{ production} \end{aligned} \tag{3.1}$$

Iron ore and P_2O_5 are considered secondary products in the reprocessing scenario and are compared to their respective primary production impacts.

It is important to note that for newly generated tailings requiring treatment, the environmental impacts associated with the direct landfilling of 1 tonne of iron tailings should be accounted for as a reduction in the net impact calculation, as shown in Equation (3.2). This extended formulation enables a comparative analysis of various reprocessing strategies against a baseline incorporating conventional tailings management and primary material production. A detailed discussion of this consideration is provided in the discussion section.

$$\begin{aligned} \textit{Net Environmental Performance of the study vs. reference} = &\sum \textit{(Total Reprocessing Impacts)} \\ &- \textit{Impacts of Primary iron ore production} \\ &- \textit{Impacts of Primary } \mathrm{P_2O_5} \textit{ production} \\ &- \textit{Impacts due to tailings landfilling} \end{aligned} \tag{3.2}$$

For the product-oriented attributional LCA, the goal is to compare the environmental impact of producing 1 tonne of apatite (32% P_2O_5) with that of primary production. In this context, the functional unit (FU) is defined as:

- The production of 1 tonne of apatite (32% P_2O_5)

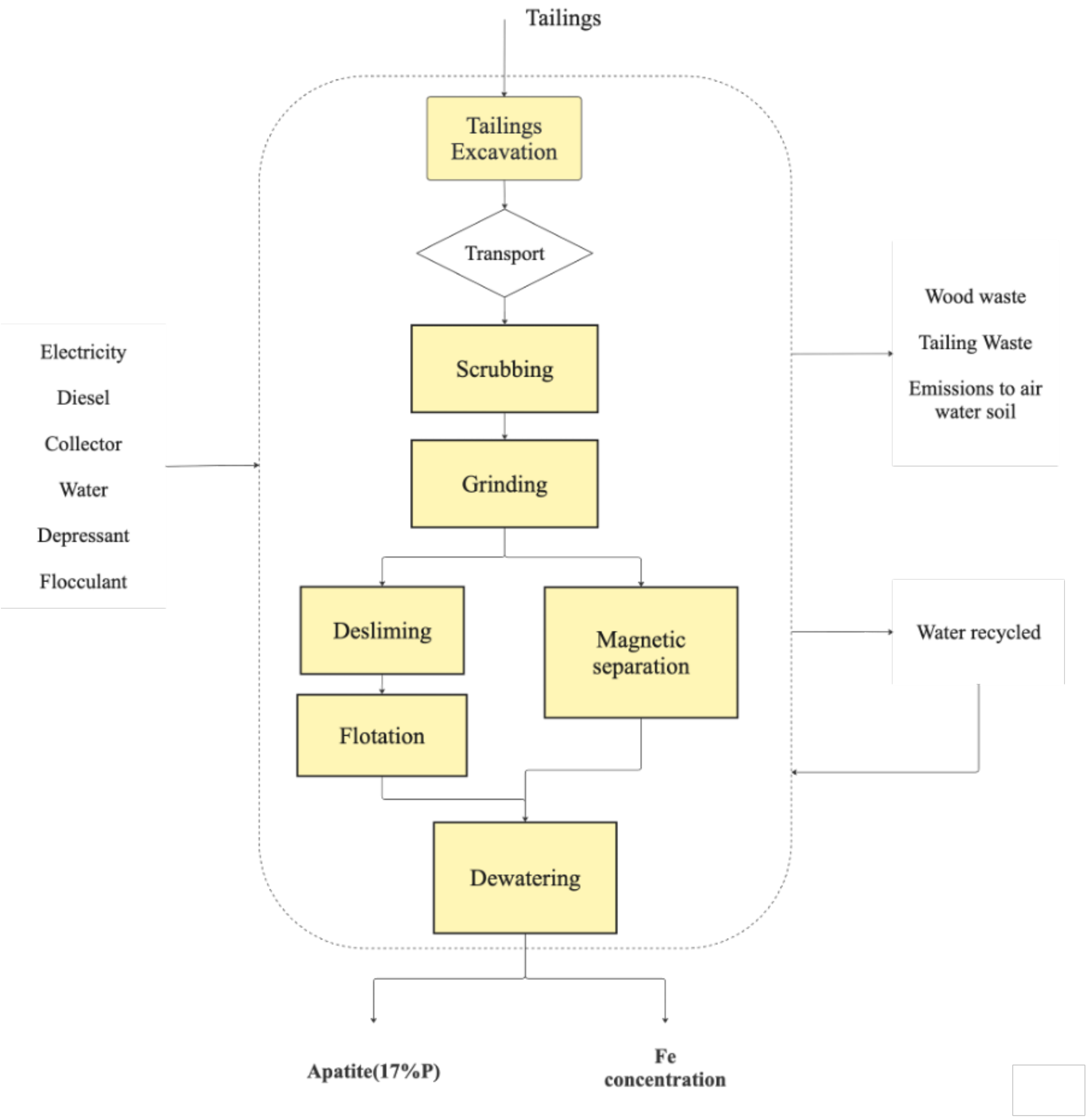

Figure 3.1: The flowsheet of the apatite recovery with excavation and the processes in the reprocessing plant

3.3 Data collection and calculation

The key inputs in the processing stages are electricity and water. Among these, electricity consumption is a significant contributor to environmental impact and is therefore analysed and compared across different processing steps.

3.3.1 Energy consumption calculation of scrubbing

In the scrubbing step, an agitator is used to stir the tailings in order to break up compacted material and ensure homogeneous mixing. The agitator is assumed to be a Mixtec Serie 1000, a vertical-entry agitator with an installed capacity of up to 30kW, which is suitable for open tank configurations. For the purposes of this study, the motor power

(P) of each agitator is assumed to be 10 kW. The corresponding energy consumption is calculated using (3.3).

$$E = P \times T/\eta \tag{3.3}$$

where η is the efficiency of the motor, T is the running time, (η is assumed to be 0.85).

3.3.2 Energy consumption calculation of flotation

In the flotation process, according to (Son et al., 2023), it estimates the energy consumption for the concentration of tungsten ore, including flotation processes. It provides specific energy consumption values for different stages of mineral processing, which can be insightful for understanding the energy requirements of flotation operations. Palacios et al. (2019) discussed the energy costs associated with producing iron ore, including the energy consumption of various processing stages such as flotation. The electricity consumption of the flotation cell is calculated based on the conversion of the power of the motor (kW, (3.4)) to the energy (kWh, (3.3))(Son et al., 2023).

$$P = 2\pi Mn/60 \tag{3.4}$$

where P is the motor power (kW), n is the stirrer rotation speed (rpm), M Is the impeller torque (Nm).
Then the practical energy consumption is calculated according to (3.3).

3.3.3 Depressant and collector dosage calculation

In addition to electricity input, flotation beneficiation is a complex physicochemical process that operates at the mineral surface and involves the use of various chemical reagents. Specifically, sodium silicate solution is added at a dosage of 500 g per tonne of tailings to adjust the flotation pH and to act as a dispersant and depressant(Jolsterå et al., 2025). According to Valderrama et al. (2024), a fatty acid-based collector (Atrac-2600) was applied at a dose of 400 g / t of tailings to recover apatite from iron ore tailings; the data are used in this study.
Sodium silicate solution is added to the flotation circuit at a dosage of 500 g per tonne of tailings. This reagent acts both as a pH modifier and as a dispersant/depressant in the flotation process. The total requirement of sodium silicate in 1 year is calculated as (3.5):

$$\text{Dosage of depressant} = M_a \text{ ton} \times 500 \text{ g/ton} = 0.5M_a \text{ kg} \tag{3.5}$$

$$\text{Dosage of collector} = M_a \text{ ton} \times 400 \text{ g/ton} = 0.4M_a \text{ kg} \tag{3.6}$$

It is noteworthy that nearly 100% of the fatty acids used in phosphate ore flotation are derived from tall oil, a by-product of the paper industry(Sis & Chander, 2003).

3.4 Impact assessment methods

From the literature review, it is evident that the ReCiPe 2016 method is the most commonly used life cycle impact assessment (LCIA) approach in the mineral industry. Therefore, this method is applied in the present study to ensure comparability with existing research and to provide a comprehensive assessment of midpoint and endpoint environmental impacts. Given that mining extraction and beneficiation are highly energy-intensive processes, the Cumulative Energy Demand (CED) method is also employed to analyse the energy consumption and determine the contribution of different subprocesses to the total energy demand. In addition, freshwater ecotoxicity and human toxicity indicators of USEtox 2.13 are included to assess potential impacts on aquatic ecosystems and human health, respectively. This multi-indicator approach allows for a deeper understanding of the environmental performance of mining waste reprocessing systems parencite Adrianto2022. The Table 3.1 lists the overall impact categories that are analysed in this study.

Table 3.1: Impact categories, abbreviations, and units used in LCIA

Impact category, abbreviation if any	**Unit**
RECIPE Midpoint (H) V1.03, Acidification: terrestrial	kg SO_2-Eq
RECIPE Midpoint (H) V1.03, Climate change	kg CO_2-Eq
RECIPE Midpoint (H) V1.03, Ecotoxicity: freshwater	kg 1,4-DCB-Eq
RECIPE Midpoint (H) V1.03, Ecotoxicity: marine	kg 1,4-DCB-Eq
RECIPE Midpoint (H) V1.03, Ecotoxicity: terrestrial	kg 1,4-DCB-Eq
RECIPE Midpoint (H) V1.03, Energy resources: non-renewable, fossil	kg oil-Eq
RECIPE Midpoint (H) V1.03, Eutrophication: freshwater	kg P-Eq
RECIPE Midpoint (H) V1.03, Eutrophication: marine	kg N-Eq
RECIPE Midpoint (H) V1.03, Human toxicity: carcinogenic	kg 1,4-DCB-Eq
RECIPE Midpoint (H) V1.03, Human toxicity: non-carcinogenic	kg 1,4-DCB-Eq
RECIPE Midpoint (H) V1.03, Ionising radiation	kBq Co-60-Eq
RECIPE Midpoint (H) V1.03, Land use	m^2a crop-Eq
RECIPE Midpoint (H) V1.03, Material resources: metals/minerals	kg Cu-Eq
RECIPE Midpoint (H) V1.03, Ozone depletion	kg CFC-11-Eq
RECIPE Midpoint (H) V1.03, Particulate matter formation	kg PM2.5-Eq
RECIPE Midpoint (H) V1.03, Photochemical oxidant formation: human health	kg NOx-Eq
RECIPE Midpoint (H) V1.03, Photochemical oxidant formation: terrestrial ecosystems	kg NOx-Eq
RECIPE Midpoint (H) V1.03, Water use	m^3
USETox, freshwater ecotoxicity*	CTUe
USETox, human-toxicity total*	CTUh
Cumulative energy demand (CED), fossil*	MJ-eq

**USEtox and CED are not part of the ReCiPe methodology and were calculated additionally.*

3.5 Assumptions and limitations

One limitation of this case study is related to process parameters. As much of the in-house data is derived from a conceptual flowsheet, several process inputs require scaling up based on calculations and informed assumptions. In addition, the background data was sourced from the ecoinvent database, which is associated with inherent uncertainties related to data quality, temporal coverage, and geographical representativeness. Several assumptions and calculations in this study:

- The electricity demand of ancillary systems (e.g., conveyors, fans, compressors, and lighting) is estimated as a proportion of a medium-scale facility's total electricity consumption, which is equally subdivided into the six processes.
- The energy consumption of thickeners is estimated according to the assumed depth of the thickener units.
- The torque values required for estimating energy consumption during flotation are calculated based on the speed and volume of the flotation cells.
- The electricity mix is assumed to reflect the most up-to-date national grid data.
- The availability of renewable electricity data in the ecoinvent database is limited for Switzerland.
- Transport distances are estimated, as there is currently no operational factory; actual logistics may vary from these assumptions.
- The quantities of reagents used in flotation and thickening are deduced from laboratory-scale volumes and scaled up to an industrial scale.
- Water usage is assumed to be internally recycled within the system, thereby minimizing freshwater consumption.

3.6 LCA software and database

OpenLCA was used as the life cycle assessment (LCA) software in this study. The Ecoinvent database provided background data for the consumables in various processes, including the electricity mix, sodium silicate production, and fatty acid production. For items not covered in Ecoinvent, the Sphera dataset *"RER: Raw phosphate (32% P_2O_5) production"* was employed to calculate the avoided primary phosphorus production. Additionally, datasets for iron ore primary production were utilised for the assessment.

4

Inventory data analysis

4.1 Excavation and transportation to the reprocessing plant

As the tailings were landfilled decades ago, the first reprocessing stage involves excavating the material using excavators and haul trucks. According to Lindholm (2022), drilling investigations have confirmed that negligible amounts of non-mineralised material are present within the repository. Therefore, collecting material using conventional surface mining equipment is considered feasible, with minimal dilution expected.

Excavators equipped with long sticks, loading into haul trucks, are regarded as the most suitable method for the material recovery in this case. The recovered tailings will be transported to a processing facility approximately 5 km from the repository. The total volume to be handled is estimated at 600,000 tonnes per year. From an economic perspective, the operation is scheduled over 250 working days per year, with a single 8-hour daily shift. Key operational parameters, including truck speed, number of excavators and trucks, dumping time, and excavator cycle time, must be considered and optimised in the equipment design process.

Meanwhile, in Sweden, the development of electric equipment for the mining industry is advancing rapidly in pursuit of a more sustainable future. Accordingly, this study considers two scenarios: one using fully diesel-powered trucks and equipment, and another assuming the future deployment of fully electric vehicles. These scenarios are evaluated to compare their respective environmental impacts. A detailed design and calculation of hauling and loading operations is provided in Appendix C.

Remarks, equipment quantities, and estimated energy consumption values are summarised in Table 4.1.

Table 4.1: Recommended Equipment Configuration

Equipment	Quantity	Remarks	Energy consumption(per year)
Haulers	2 units	100-tonne payload, ~30-minute cycle time	160,000 L diesel
Excavators	1 active + 1 backup	10 m^3 bucket, ~25 t/cycle loading	50,000 L diesel
Haulers(Volvo A40 Electric)	5	40-tonne payload, ~30-minute cycle time	680 MWh
Excavators (Volvo EC230 Electric)	3	1.25 m^3 bucket, ~3.1 t/cycle loading	165 MWh

4.2 Scrubbing

The mining waste facility for apatite recovery begins with the scrubbing step. In this stage, 80 tonnes of tailings are fed into the system per hour using a front-end loader. The tailings contain approximately 15% moisture. The material is processed in 11.4 batches, with each batch receiving 7.0 tonnes of tailings. Load cells measure the addition of 10.5 tonnes of water per batch, after which 7 tonnes of material are agitated. The agitator is programmed to stop at a low level and restart automatically once the water level is restored. Between batches, the load cells are automatically zero-calibrated to ensure measurement accuracy.

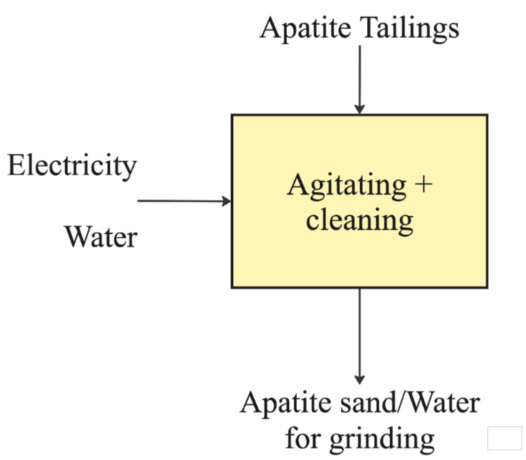

Figure 4.1: The flows in the scrubbing process

Following the batch mixing, the tailings now with a solid-to-water ratio of 40% (s/w) are transferred into a holding tank, where an additional 40 tonnes of water is introduced. After scrubbing, 80 tonnes per hour of tailings are discharged and forwarded to the grinding stage. The density of the solid is 2.14 ton/m^3, with 10% Fe and 2.5% P. It is noted that minor amounts of wood waste are removed during the scrubbing process. Due to their negligible quantity, these are disposed of on-site and cut off from the system boundary in this study. The inputs and outputs of this step are listed in Table 4.2.

Table 4.2: Material and energy flows for the scrubbing process

Description	Quantity	Unit
Input		
Electricity of agitators	11.765	kW
Apatite	80	t/h
Water	160	t
Output		
Apatite sand	80	t/h
Water	160	t
Wood waste	negligible	

4.3 Grinding

Grinding is reducing material size into fine particles suitable for downstream processing. In this study, a ball mill is used for tailings size control. The energy efficiency of ball mills can degrade over time due to wear, liner erosion, and maintenance-related factors. According to Lindholm (2022), the electricity consumption for grinding is approximately 8.7 kWh per tonne of tailings processed. A throughput of 80 tonnes per hour corresponds to a total power demand of approximately 696 kW during continuous operation. A hydroclone is used for classification after grinding in this case.

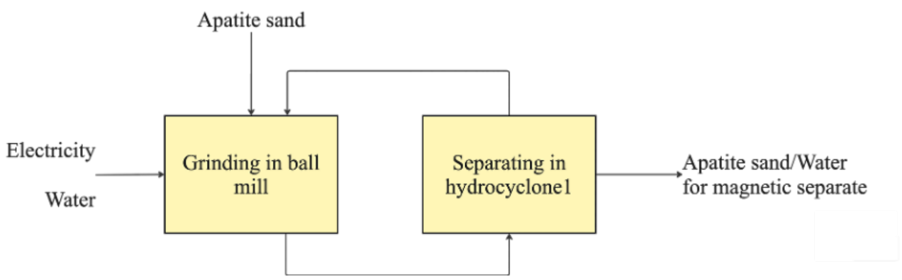

Figure 4.2: The flows in the grinding process

After grinding, the hydrocyclone separates particles based on size and density. Fine particles are carried in the overflow, while coarse or unground particles are sent back in the underflow for regrinding. According to a brochure from The Weir Group(The Weir Group, 2020), the pump motor varies according to the plant capacity. Given a throughput of 80 tonnes per hour, the assumption of the hydrocyclone feed pump is made to consume approximately 26 kW per hour.

After the grinding stage, the output is 80 tonnes per hour with 10% Fe, which will go to the magnetic separation process. The inputs and outputs of this step are listed in Table 4.3.

Table 4.3: Material and energy flows for the grinding process

Description	Quantity	Unit
Input		
Electricity of ball mill	696	kW
Electricity of hydrocyclone pump	26	kW
Water	86	t
Apatite sand	160	t/h
Output		
Apatite sand	80	t/h
Water	160	t

4.4 Magnetic separation

Magnetic separators exploit differences in the magnetic properties of minerals. They are commonly used to concentrate valuable magnetic minerals (e.g., magnetite from quartz), remove magnetic contaminants, or separate mixtures of magnetic and non-magnetic materials (Wills & Finch, 2016). This study's tailings originate from historical iron ore processing and contain apatite and magnetite. The flow from the grinding stage is directed into four rotating drum magnetic separators. Each separator typically features a rotating drum where magnetic particles are retained and discharged separately while the non-magnetic fraction continues to the following processing step, apatite flotation. A ball mill is employed for particle size control after two separators before further separation.

In the magnetic separation stage, four drum-type wet low-intensity magnetic separators (LIMS) are assumed to be used, based on the descriptions provided in an industry brochure (Metso Outotec, 2020). The first separator handles a significantly higher throughput of approximately 240 tonnes per hour, while the remaining three processes are 20, 10.5, and 7.7 tonnes per hour, respectively. According to industry data, wet LIMS units typically consume between 0.05 and 0.15 kWh per tonne of processed material, depending on drum diameter, rotational speed, magnetic field strength, material throughput, and slurry characteristics(Shandong Huate Magnet Technology Co., Ltd., 2025).

For the largest unit, which processes around 240 t/h, an electricity consumption rate of 0.05 kWh/tonne is assumed. For the smaller units, which each handle less than 25 t/h, a rate of 0.15 kWh/tonne is applied(Shandong Huate Magnet Technology Co., Ltd., 2025). Based on these values, the electricity consumption is estimated as follows: 11 kWh/h for the first separator, 3 kWh/h, 1.575 kWh/h, and 1.155 kWh/h for the remaining three units, respectively. The total electricity demand for the magnetic separation process is therefore calculated to be approximately 16.73 kW. Additionally, according to the report, the ball mill in this process stage consumes 32kW.

The inputs and outputs of this stage are illustrated in Table 4.4.

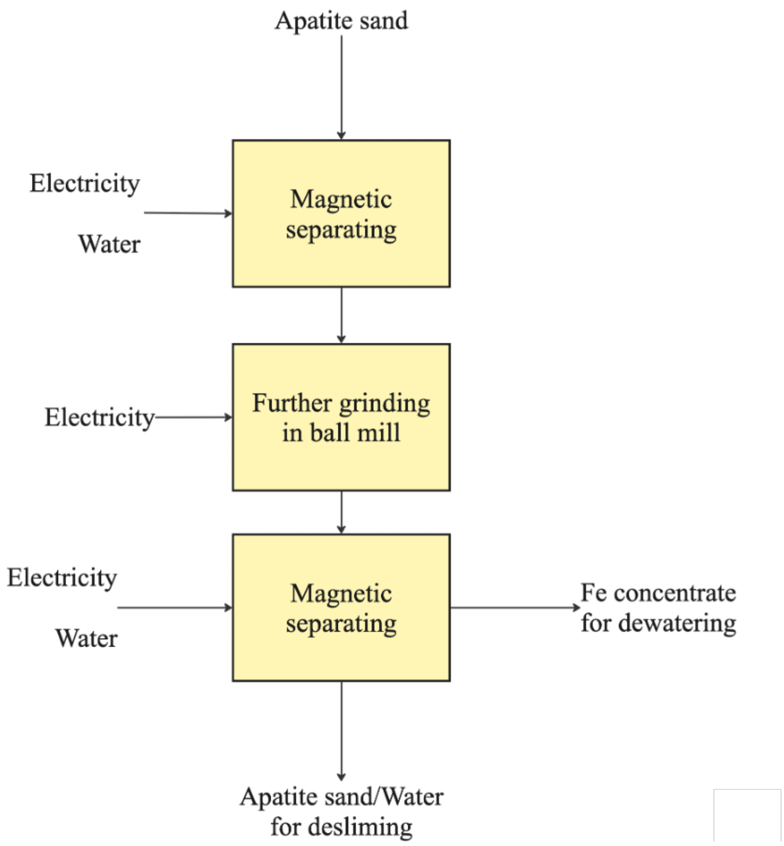

Figure 4.3: The flows in the magnetic separating process

Table 4.4: Material and energy flows for the magnetic separation process

Description	Quantity	Unit
Input		
Electricity of magnetic separators	16.73	kW
Electricity of ball mill	32	kW
Water	236.3	t
Apatite sand	80	t/h
Output		
Fe concentrate (65.0% s/w)	4.8	t/h
Apatite sand	75.2	t/h
Water to Fe drying	2.6	t
Water to flotation	234.2	t

4.5 Desliming

Before the apatite flotation stage, a desliming step is to remove fine particles (slimes) that could interfere with flotation efficiency. In this process, 1.6 t/h of slime and 98 tonnes of water are separated and sent directly to the thickener. The remaining material is then directed to the flotation circuit for apatite recovery. This separation improves the selectivity of flotation by reducing the presence of ultra-fine gangue particles.

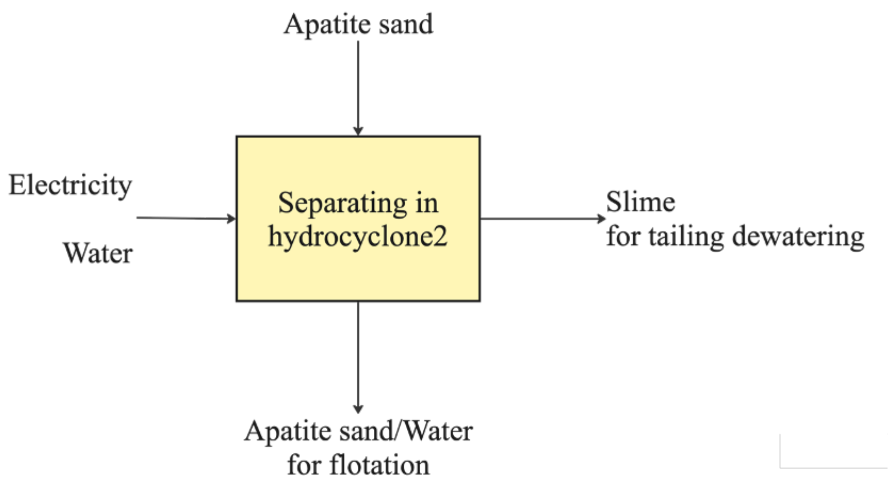

Figure 4.4: The flows in the desliming process

The inputs and outputs of this stage are illustrated in Table 4.5.

Table 4.5: Material and energy flows for the desliming process

Description	Quantity	Unit
Input		
Electricity of hydrocyclone pump	26	kW
Apatite sand	75.2	t/h
Water	234.2	t
Output		
Apatite sand to flotation	73.6	t/h
Water to flotation	136.2	t
Slime to thickener1 (D9)	1.6	t/h
Water to thickener1 (D9)	98	t

4.6 Flotation

Figure 4.5 illustrates a simplified schematic of the flotation and subsequent processes within this apatite recovery system. The flotation stage receives apatite tailings as input and involves the addition of reagents, including a fatty acid-based collector and a sodium silicate depressant, alongside electricity input for mechanical operation. The flotation process separates a valuable concentrate, which proceeds to the dewatering (thickening first) stage, while the waste stream, consisting of water and non-recovered material, is directed to a separate thickening unit for water recovery and tailings management.

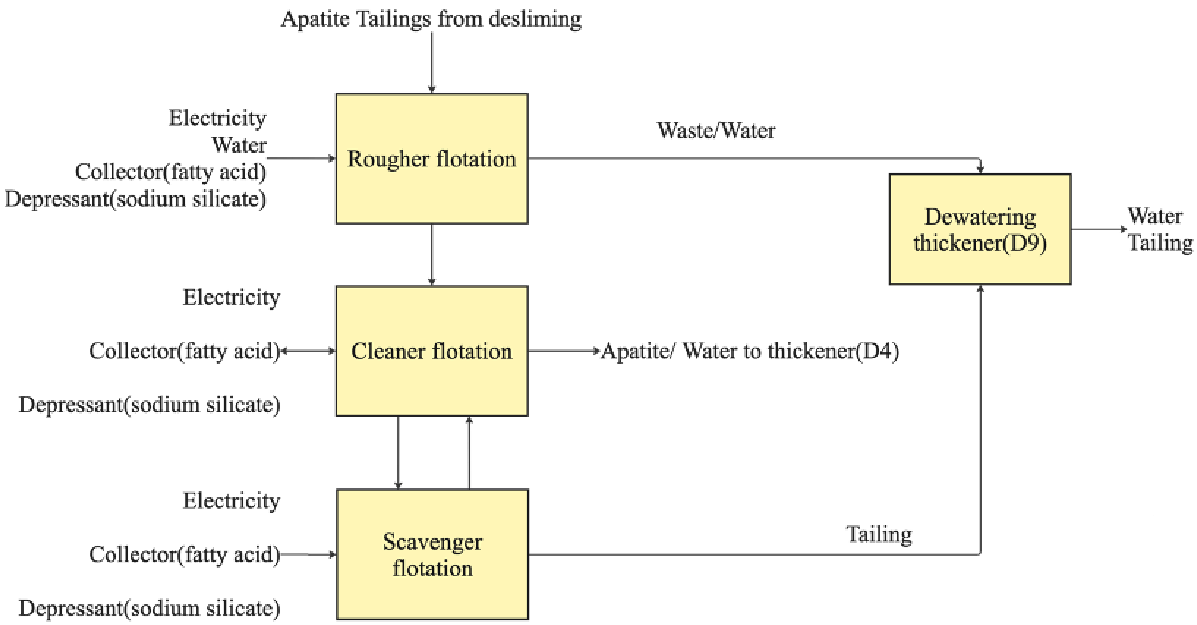

Figure 4.5: The flows in the flotation process

There are several steps in the flotation process. It starts from the following the Rougher Flotation(Rgh flot),Cleaner Flotation(Cln flot) and Scavenger Flotation(Scy flot)(Figure 4.5). The descriptions are in the Table 4.6.

Table 4.6: Flotation Descriptions

Abbreviation	**Full term**	**Description**
Rgh Flot	Rougher Flotation	The first stage of flotation, where bulk separation of valuable minerals (e.g., apatite, sulfides) from gangue takes place. It handles high-grade feed and captures most of the target mineral.
Cln Flot	Cleaner Flotation	The second or later stage that re-processes rougher concentrate to increase purity. It removes remaining impurities and produces a higher-grade concentrate.
Scy Flot	Scavenger Flotation	A final-stage flotation that re-processes the tailings from the rougher (or sometimes cleaner) to recover any remaining valuables before waste disposal. Helps improve overall recovery.

As stated in the previous section, energy consumption needs to be calculated in the flotation process. Besides the volume and speed of each foliation cell, the torque is needed. The(Lindholm, 2022) does not directly state torque values for the flotation cells, typical industry assumptions are based on flotation cell volume and duty. From the report(Lindholm, 2022), Volumes like $5\,\mathrm{m}^3$, $8\,\mathrm{m}^3$, $16\,\mathrm{m}^3$ are listed per flotation unit, the residence time is also noted (e.g., 6-15 mins), which allow for rough estimation of the tailing mass being processed per unit of time. Larger tanks and longer operation imply higher torque. According to manufacturers (e.g., Metso, Outotec, FLSmidth) and (U.S. Department of Energy, 2007a; Wills & Finch, 2016), small flotation cells (5–8 m^3)

typically operate with torque in the range of 80–100 N m, while medium cells (10–20 m^3) require approximately 110–120 N m.

So base on the above information, the Rgh Flot is the largest, consisting of three cells of 16 m^3 each (3×16 m^3), and is estimated to require 120 N m torque for each. The Scy Flot is the smallest cell used at the late stage and is estimated to operate at 80 N m. The Cln Flots (Cleaner flotation cells) range between 5–8 m^3 and are estimated to require 90 N m torque. Torque varies with factors such as slurry density, impeller diameter and speed, pulp level, and froth control.

The torque values and the energy consumption calculated based on Equation (3.4) for each flotation cell are listed in the Table 4.7.

Table 4.7: The energy consumption of flotation cells

Flotation cell	**Quantity**	**Volume** (m^3)	**Speed** (m^3/h)	**Estimated Torque** (N m)	**Energy consumption** (kWh)
Rgh Flot	3	16	198	120	36.3
Cln Flot1	1	9	57	90	13.1
Cln Flot2	1	5	44	90	11.9
Cln Flot3	1	5	40	90	11.9
Scy Flot	1	2	21	80	9.4

Specifically, the amount of sodium silicate solution as a dispersant/depressant and fatty acid collector based on Equation (3.5) and Equation (3.6) are listed in the following table.

Table 4.8: The dosage of the chemicals in flotation cells

Flotation cell	**Water** (t)	**Depressant** (t/y)	**Collector** (t/y)
Rgh Flot	171.2	258	206
Cln Flot1	52	46	36
Cln Flot2	40	39	31
Cln Flot3	37	32	26
Scy Flot	39	16	13

Thus, the input and output of this stage can be summarized and illustrated in the Table 4.9.

Table 4.9: Material and energy flows for the flotation process

Description	Quantity	Unit
Input		
Electricity of flotation cells	50.4	kW
Apatite sand	73.6	t/h
Water	171.2	t
Depressant (sodium silicate)	55.8	kg/h
Collector (fatty acid)	44.5	kg/h
Output		
Apatite sand to thickener2 (D4)	8.8	t/h
Water to thickener2 (D4)	35	t
Apatite Rough tailings to thickener1 (D9)	60.6	t/h
Apatite Scavenger tailings to thickener1 (D9)	4	t/h
Water to thickener1 (D9)	139	t

4.7 Dewatering

Dewatering, or solid liquid separation, produces a relatively dry concentrate for shipment. Dewatering methods can be broadly classified into three groups: sedimentation (gravity and centrifugal), filtration, and thermal drying. Dewatering in mineral processing is normally a combination of these methods(Wills & Finch, 2016). In this study, the bulk of the water is first removed by thickening, which produces a thickened pulp of 40% solids by weight, with 8.8 t/h solid and 13.2 tonne water. Up to 63% of the water is separated at this stage. Filtration of the thick-ened pulp then produces a moist filter cake of 94% solids, then a thermal drying produces a final product of 99.5% solids by weight.

The Fe concentrate is produced after the magnetic separation stage, the concentrated Fe stream passes through a vacuum disc or belt filter followed by a vacuum chamber to achieve final dewatering. This results in a concentrated iron product at a rate of approximately 4.8 t/h with a solid-to-water ratio of 92%. In this study, the power consumption of the belt filter is assumed to be 16 kW according to CEC Mining Systems (2017), which includes the vacuum pump, drive, and accessories. Given that a smaller-scale unit in this study, a conservative estimate is considered reasonable and aligns with typical operational parameters for such equipment. The vacuum chamber is estimated to consume 75 kW.

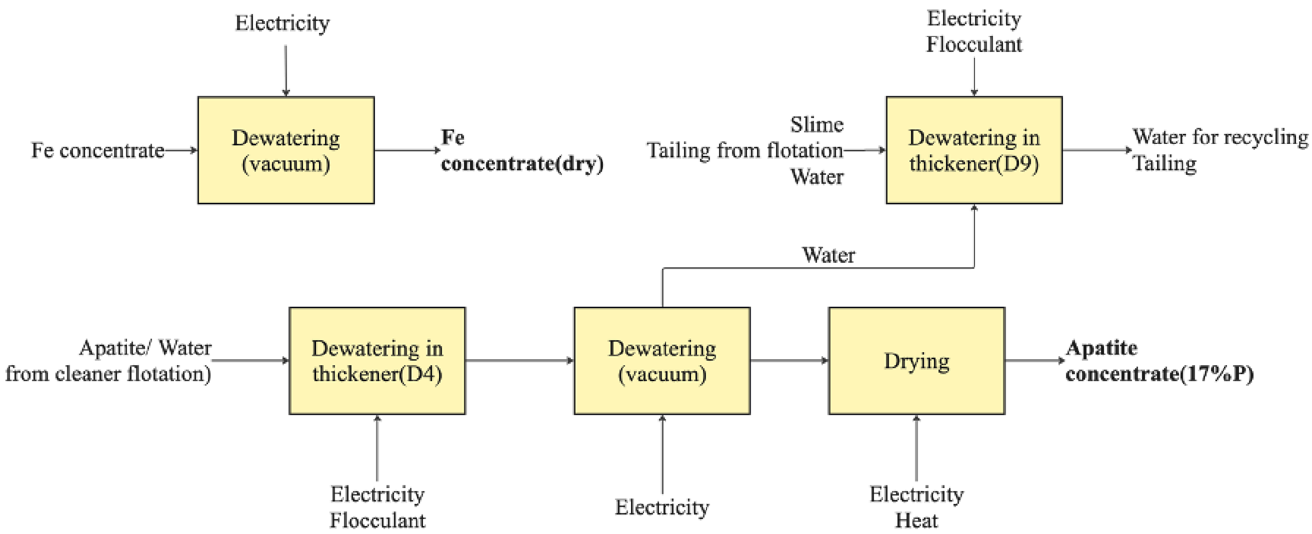

Figure 4.6: The flows in the dewatering process

In this case, the residuals are also dewatered; a thickened pulp of 30% solids by weight is produced, then goes directly to the open pit.

The energy consumption of the thickeners is estimated based on the diameter of the thickener units, which directly influences the power requirements of the rake drive system. In this study, two thickeners are identified from the in-house data. The thickener1 is to process tailings with diameters 9 m, while the thickener2 is for further apatite processing, which is with diameters of 4 m. According to technical documentation from Wills and Finch (2016), the typical power demand for industrial thickeners ranges from 1.2 to 1.5 kW per meter of diameter, depending on solids loading, rake torque, and bed depth. Using a conservative estimate of 1.5 kW/m, the thickener1 is assumed to consume approximately 13.5 kW, while the thickener2 is estimated at 6 kW. Multiplying these values by the annual operation time (7,008 hours/year), the resulting energy consumption is approximately 42,048 kWh/year for the 4 m unit and 94,608 kWh/year for the 9 m unit.

In addition to electricity use, the thickening requires the input of flocculants to promote solid-liquid separation. Flocculants are used in most thickeners to obtain concentrations of overflow solids that will allow water to be reused or to comply with government regulations if the overflow is to be discharged. The flocculant dosage used in thickening operations significantly affects the settling behaviour and the resulting underflow concentration of tailings. According to a recent experimental study on flocculated tailings suspensions, the optimal dosage for achieving the highest underflow concentration in unclassified tailings was found to be approximately 40 g/t of solids (Zhang et al., 2023). Since unclassified tailings represent a broad particle size distribution, this value is considered a suitable proxy for iron ore tailings in this study. Given the tailings throughput associated with the thickening units, the annual flocculant demand can be estimated by multiplying the assumed dosage by the total mass of tailings entering the thickener.

After thickening, the apatite-rich slurry is conveyed to a belt filter, followed by vacuum dewatering. The energy consumption of the belt filter and the vacuum unit is assumed to be the same as in the magnetic separation stage, at 16 kW and 75 kW, respectively. Subsequently, the dewatered apatite product is transferred to a drying drum with a power demand of 500 kW. The final apatite concentrate is produced with a phosphorus content of approximately 17% P.

The corresponding inputs and outputs of these processes are illustrated in the Table 4.10

Table 4.10: Material and energy flows for the dewatering process

Apatite Thickening		
Description	**Quantity**	**Unit**
Input		
Electricity of thickener2 (D4)	6	kW
Apatite	8.8	t/h
Water	35	t
Flocculant	40	g/t
Output		
Apatite	8.8	t/h
Water to belt filter	13.2	t
Water to thickener1 (D9)	21.8	t
Apatite Drying		
Input		
Electricity of belt filter	16	kW
Heat of drying drum	500	kW
Apatite	8.8	t/h
Water	13.2	t
Output		
Apatite (99.5% s/w)	8.8	t/h
Water to thickener1 (D9)	12.7	t
Water	0.04	t
Fe Drying		
Input		
Electricity of belt filter	16	kW
Electricity of vacuum	75	kW
Fe	4.8	t/h
Water to Fe drying	2.6	t
Output		
Fe concentrate (92% s/w)	4.8	t/h
Water to thickener1 (D9)	2.6	t
Tailing Thickening		
Input		
Electricity of thickener1 (D9)	13.5	kW
Tailing	66.2	t/h
Water	272	t
Flocculant	40	g/t
Output		
Waste water to open pit	154	t
Waste water to water tank	120	t
Tailing	66.2	t/h

4.8 List of consumables and data source

The yearly consumption of the materials and energy of the mining waste facility are listed in Appendix A, which are gathered from partners and calculated based on the equations in the previous statement—besides, Table 4.11 below illustrates the data sources of these consumables. Ecoinvent 3.11 database is used to support the background data for all the consumables.

Table 4.11: The consumables data sources from Ecoinvent and Gabi

Flows	**Unit**	**Source of LCIA data**	**Region**
Electricity	kWh	Electricity, Medium Voltage, residual mix, Ecoinvent 3.11	SE
Electricity	kWh	Electricity, Medium Voltage, renewable production, Ecoinvent 3.11	Switzerland (CH)
Water	ton	rainwater harvesting, Ecoinvent 3.11	GLO
Collector (Fatty Acid)	ton	tall oil refinery operation, Ecoinvent 3.11	GLO
Depressant (Sodium silicate)	kg	sodium silicate production, Ecoinvent 3.11	EU
Heat	kWh	heat and power co-generation, natural gas, Ecoinvent 3.11	SE
Flocculant	kg	chemical production, inorganic, Ecoinvent 3.11	GLO
Diesel	kg	diesel production, low sulfur, Ecoinvent 3.11	EU
P_2O_5 primary production	kg	RER: Raw phosphate (32% P2O5) production, Gabi	EU
Iron ore concentrate production	kg	iron ore mine operation and beneficiation, Ecoinvent 3.11	Canada

5
Results

5.1 Environmental impact results

5.1.1 Environmental impact of mine tailings reprocessing

In this study, two scenarios are analysed. The baseline scenario accounts for diesel consumption in transportation and electricity use based on the residual mix of Sweden 2024, reflecting the current grid conditions for untracked electricity. The future scenario assumes full electrification of transportation and electricity use based entirely on renewable energy production by 2045.

For the baseline scenario, the LCIA results provide insight into both the burdens and credits associated with the reprocessing route, shown in Fig 5.1.

Notably, the net environmental impact for climate change is $78.82\,\mathrm{kg\,CO_2}$ eq., while land use and water use are $44.62\,\mathrm{m^2/crop{-}eq}$ and $4.07\,\mathrm{m^3}$, respectively. These values represent the additional environmental burden introduced by the reprocessing route compared to the avoided primary production. However, several impact categories show net environmental benefits (i.e., negative values), indicating that the reprocessing of secondary materials can substitute primary production with a lower environmental cost. These categories include Ecotoxicity: freshwater ($-13.15\,\mathrm{kg}\,1,4{-}\mathrm{DCB}{-}\mathrm{eq}$),Ecotoxicity: marine($-17\,\mathrm{kg}\,1,4{-}\mathrm{DCB}{-}\mathrm{eq}$), Human toxicity: non-carcinogenic($-666.07\,\mathrm{kg}\,1,4{-}\mathrm{DCB}{-}\mathrm{eq}$), Material resources: metals/minerals ($-44.32\,\mathrm{kg\,Cu{-}eq}$).These negative values reflect environmental credits, meaning the reprocessing system performs better than the equivalent impacts from extracting and processing primary materials.

In general, while reprocessing introduces certain environmental burdens, particularly in terms of greenhouse gas emissions and land occupation, it offers substantial benefits in categories related to toxicity and resources. This underscores the importance of evaluating trade-offs across multiple impact categories when assessing the sustainability of secondary material recovery systems.

A comparison of the net environmental impacts under the two scenarios reveals that the future scenario, based on fully renewable electricity, offers greater overall environmental benefits than the base scenario. This is particularly evident in impact categories such as human toxicity (non-carcinogenic), ecotoxicity, and resource depletion, where the future scenario demonstrates notable environmental credits.

As shown in Fig 5.2, the impact on climate change is significantly lower in the future scenario, decreasing from $78.82\,\mathrm{kg\,CO_2}$ eq. to $4.04\,\mathrm{kg\,CO_2}$ eq.. Land use is also reduced by approximately half compared to the base scenario. However, water use increases to $10.51\,\mathrm{m^3}$. Despite these improvements, the reprocessing still results in a net positive environmental impact, indicating environmental costs rather than benefits.

A comparison reveals a substantial improvement in the future scenario. Specifically, the

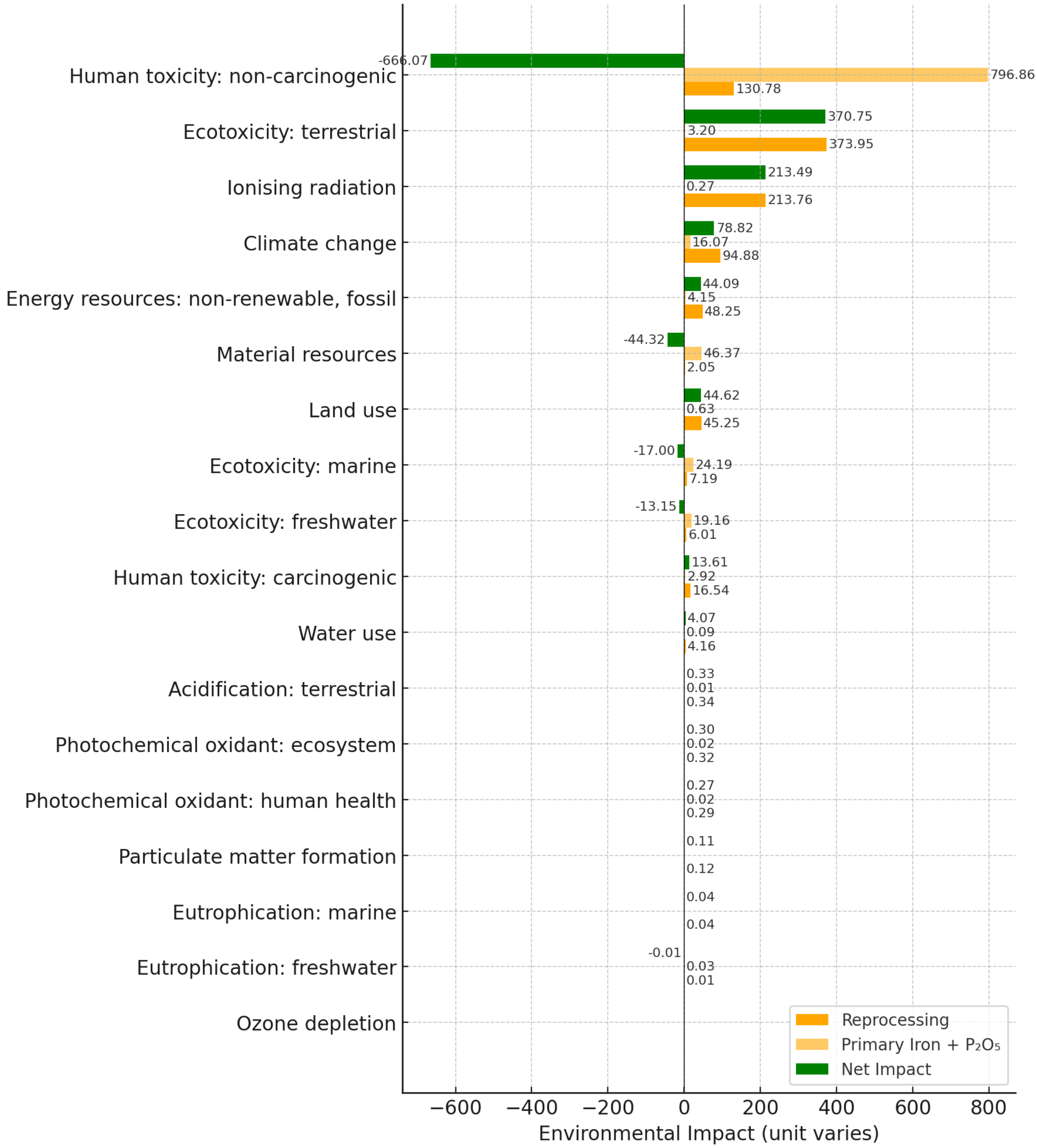

Figure 5.1: The environmental impacts for treating 1 ton of iron ore tailings, including impacts from the reprocessing steps and the impact credits (negative impacts are equivalent to environmental benefits) from displaced primary materials.

cumulative energy demand (CED) of overall resources in the future scenario is 36.1% lower than that of the baseline scenario. Notably, the fossil fuel energy demand in the future scenario is negative, indicating a transition toward a fossil-free context (as shown in Fig 5.3). However, the total cumulative energy demand remains positive in both scenarios, indicating that there is no net environmental benefit in terms of total energy

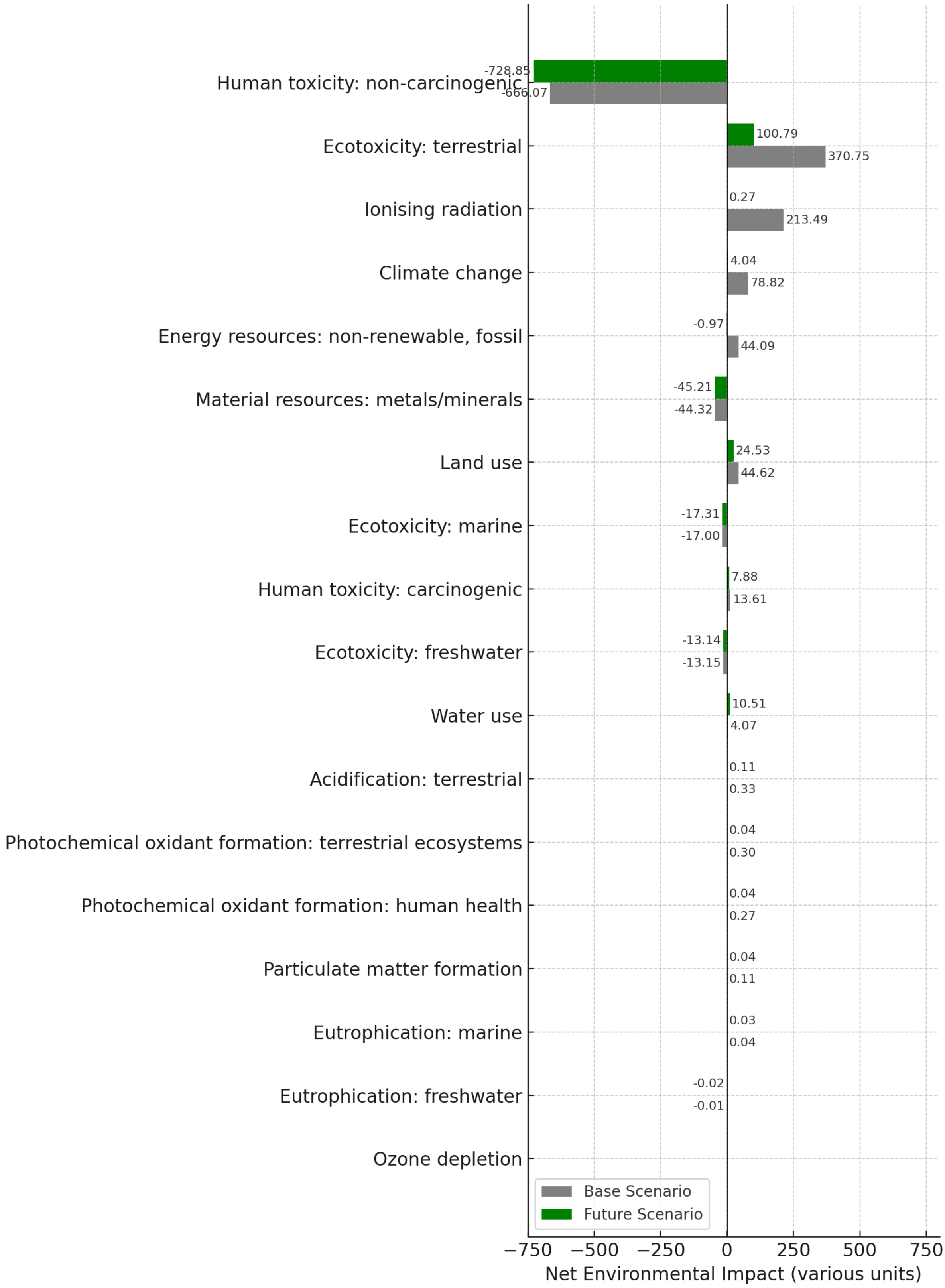

Figure 5.2: Comparison of net environmental impacts between the two scenarios using the ReCiPe 2016 midpoint method.

savings.

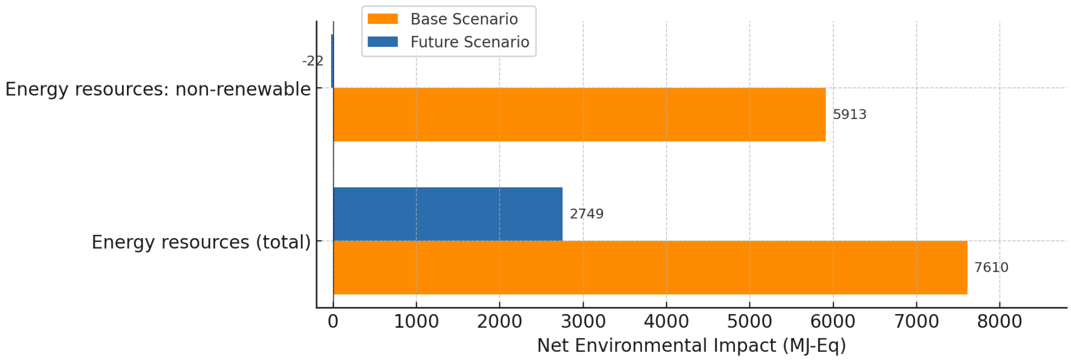

Figure 5.3: Comparison of energy demand between the base and renewable energy scenarios by using CED method. The adoption of renewable electricity significantly reduces the resource of the energy consumption.

According to the USEtox 2.13 method, the environmental impacts of the reprocessing system were assessed for freshwater ecotoxicity (CTUe) and human toxicity (CTUh). In the base scenario, the system results in a net positive impact on freshwater ecotoxicity (1.4628 CTUe), indicating an environmental burden. However, the human toxicity impact is negative (-2.78×10^{-3} CTUh), suggesting a small environmental benefit due to credits associated with substituted materials.
In the future scenario, improvements in energy and material efficiency reduce the impact of freshwater ecotoxicity to 0.8467 CTUe, almost halving the impact compared to the base case. The human toxicity indicator remains negative (-2.82×10^{-3} CTUh), showing a consistent, but minor, environmental benefit. This indicates that, although ecotoxicity impacts can be significantly reduced through the energy transition, improvements in human toxicity are marginal, likely due to the already low or offset baseline impacts in both scenarios.

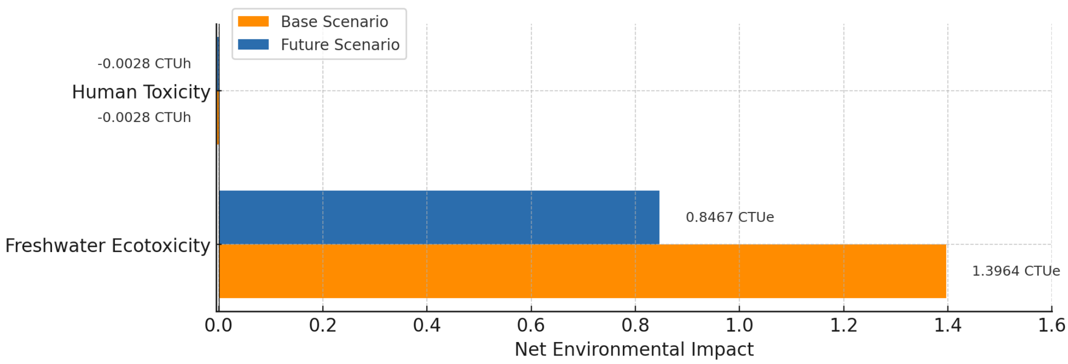

Figure 5.4: Comparison of the results between the base and renewable energy scenarios by using the USEtox 2.13 method.

5.1.2 Environmental Impact of Producing 1 Tonne of Apatite (32% P_2O_5)

The environmental impacts associated with producing 1 tonne of apatite (32% P_2O_5) via reprocessing are assessed in terms of climate change, land use, and water use. These impacts are compared against those of the conventional reference system, as illustrated in Fig 5.9, based on a mass allocation approach. For mass allocation, the distribution factors are 0.059 for iron and 0.131 for apatite (32% P_2O_5), reflecting their respective shares by mass. Under this approach, the climate change impact in the base scenario (92.8 kg CO_2 eq.) is slightly lower than that of the reference, indicating a modest environmental benefit from producing 1 tonne of apatite (32% P_2O_5) through reprocessing. However, the land use and water use indicators in the base scenario are significantly higher than those of the reference system. In the future scenario, the climate change impact is substantially reduced to 19.67 kg CO_2 eq., compared to 111.91 kg CO_2 eq. in the reference. Despite this improvement in climate performance, land use and water use impacts remain higher than in the reference, with water use even exceeding that of the base scenario.

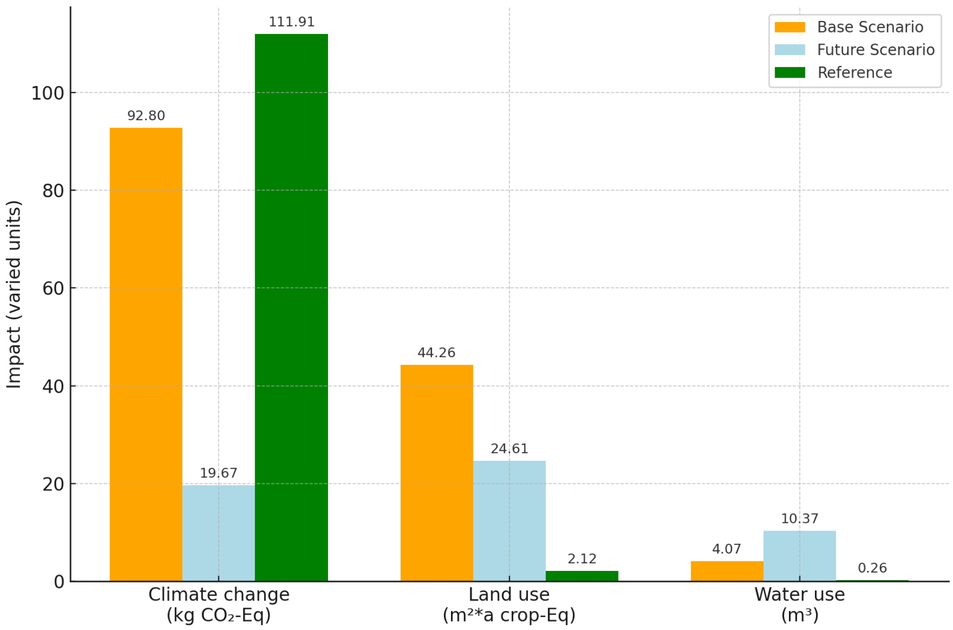

Figure 5.5: The results for the impact categories—climate change, water use, and land use—associated with producing 1 tonne of apatite (32% P_2O_5), using the mass allocation method, are compared across the base scenario, future scenario, and the reference case.

5.2 Sensitivity analysis

The results reveal a notable sensitivity to transport distances in the excavation process, as well as to energy consumption parameters within the reprocessing system. In the current analysis, it is assumed that the reprocessing plant is located in close proximity to the tailings repository; hence, transport impacts are initially omitted. However, by introducing hypothetical transport distances of 5 km and 15 km, based on a typical distance from the city center to the repository (approximately 15 km), significant changes in environmental performance are observed.

When the transport distance increases to 15 km, the climate change impact rises by approximately 190%, and the consumption of non-renewable fossil energy increases by 39.6%. While the impact on human toxicity remains relatively stable, the freshwater ecotoxicity indicator shows a marked increase, rising from 1.52 CTUe to 4.56 CTUe. These findings underscore the importance of spatial considerations and process parameters in evaluating the environmental performance of tailings reprocessing systems.

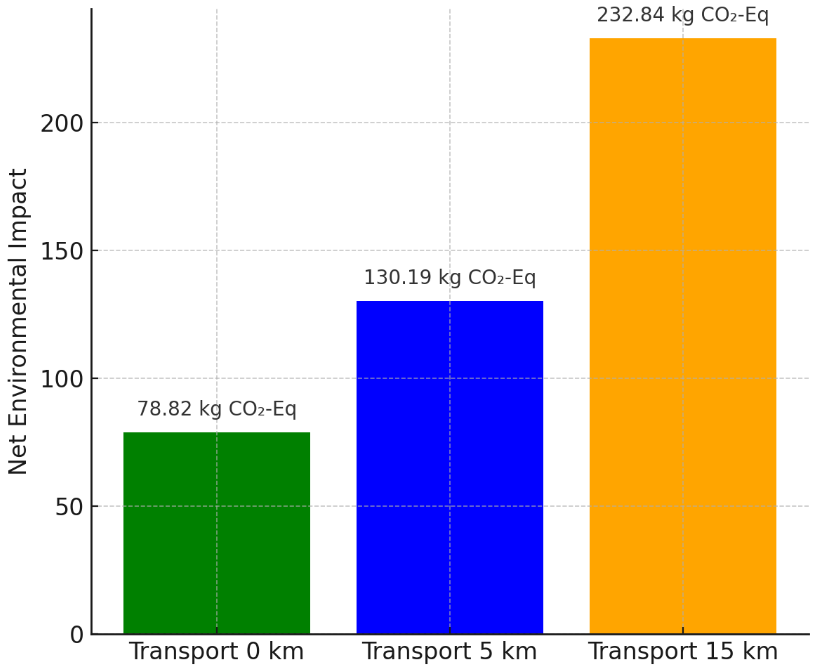

Figure 5.6: Sensitivity analysis of Climate Change impacts varies with transportation distance.

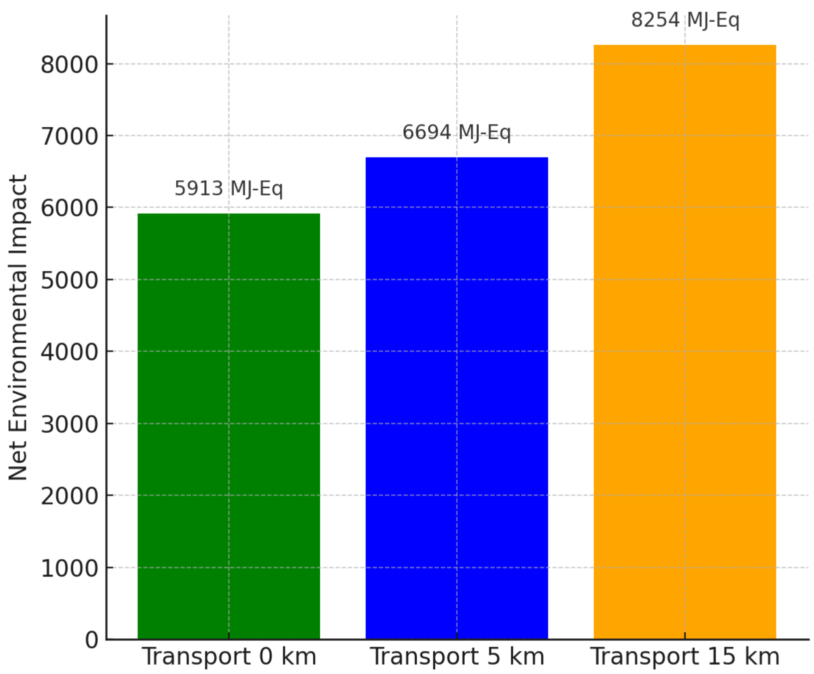

Figure 5.7: Sensitivity analysis of CED impacts of non-renewable fossil energy varies with transportation distance.

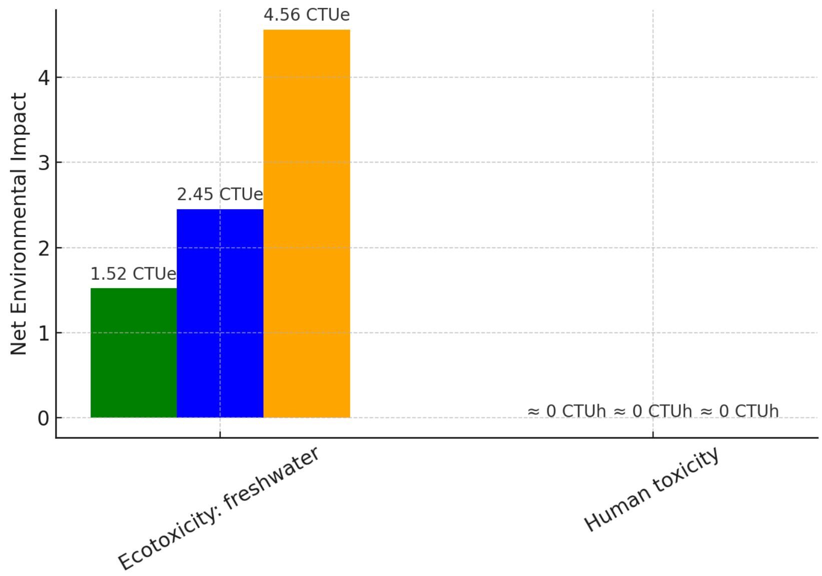

Figure 5.8: Sensitivity analysis of USEtox impacts varies with transportation distance.

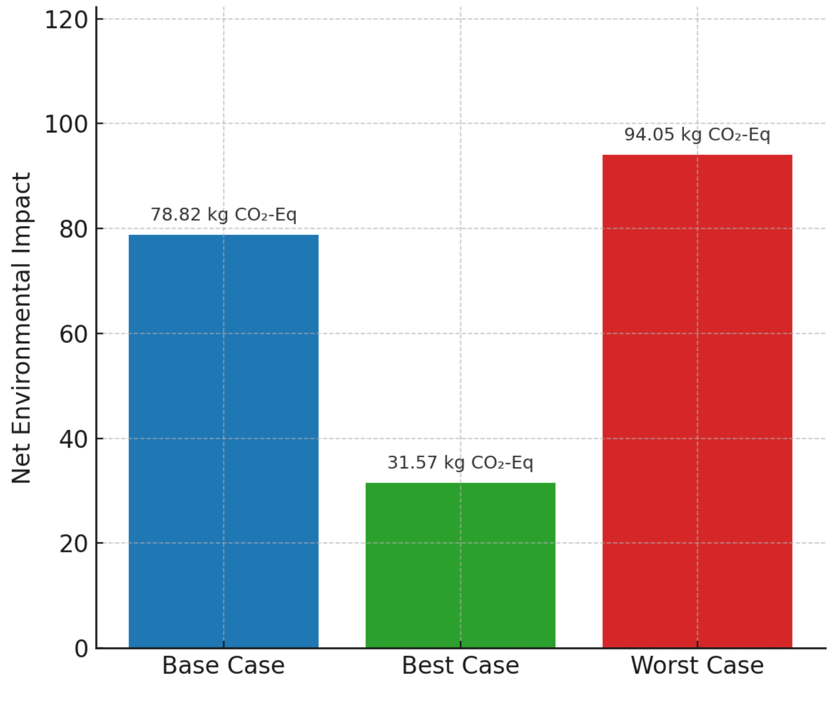

Figure 5.9: The impacts of climate change vary based on the total energy input in the three cases at the reprocessing plant.

Typical total energy use for tailings processing ranges from 10–25 kWh/ton tailings, depending on the flowsheet configuration and process intensity(Canada Mining Innovation Council, 2011; Kossakowska & Grzesik, 2019; Kruyswijk et al., 2021; U.S. Department of Energy, 2007b; Wills & Finch, 2016). The average energy consumption of the reprocessing plant is calculated to be 21.4 kWh/ton tailings, as a base-case scenario, which falls within the benchmark range. To assess the influence of energy use on climate change impacts, a sensitivity analysis was conducted using 10 kWh/ton tailings as a best-case scenario and 25 kWh/ton tailings as a worst-case scenario. The results indicate that the climate change impact in the best-case scenario is approximately 33.56% of that in the worst-case scenario, highlighting the significant role energy efficiency plays in reducing greenhouse gas emissions within reprocessing operations.

5.3 Uncertainty analysis

Several uncertainties are associated with the data used in calculating the environmental impacts, which may affect the reliability of the results. Firstly, the excavation process is modelled based on reasonable assumptions; however, its accuracy could be improved with real-case operational data.

For the operational reprocessing phase, the energy consumption of ancillary systems is estimated using industry-average data, as no direct measurements were available from the company. This introduces additional uncertainty into the results. Water consumption plays a significant role in the reprocessing stage. This study assumes that water is sourced from rainwater and could be recycled afterwards. These assumptions may lead to an underestimation of the actual environmental impact.

Furthermore, due to limitations in the ecoinvent database, the dataset used for renewable electricity represents production in Switzerland rather than Sweden, which reduces the geographical relevance of the data. A similar issue arises with the iron ore production dataset. The geographic origin of the dataset is Canada, as no data specific to Sweden or Europe were available, further increasing the uncertainty of the results. Moreover, a generic dataset for inorganic chemicals was used instead for flocculant consumption, as no specific dataset was available.

Regarding the primary production phase, data for 32% P_2O_5 production are sourced from the GaBi database, while iron ore production data are taken from the ecoinvent database. This mismatch in data sources may introduce inconsistencies.

6

Discussion

6.1 Process environmental impact

The results indicate that the reprocessing introduces additional environmental burdens regarding climate change, land use, and water use despite providing environmental benefits in other impact categories. However, these conclusions specifically apply to historically generated tailings. The environmental impacts associated with tailings landfilling must be subtracted for newly generated tailings. According to Adrianto and Pfister (2022), recycling 100% of tailings generally yields negative impacts across most indicators, signifying environmental benefits, except for a few energy-intensive reprocessing routes.

A fraction is assumed to be recovered through reprocessing for newly generated iron tailings, while the remainder is managed as residue, typically via direct landfilling. In this scenario, calculating the net environmental performance involves crediting impacts associated with the recovered fraction, while only the impacts of the residual fraction destined for landfill are treated as net burdens. This method ensures the assessment accurately reflects reduced environmental loads due to material recovery and avoids double-counting impacts. Nonetheless, a detailed assessment of the direct landfill impacts of iron tailings remains essential.

This study analysed only two scenarios: diesel consumption for excavation using the current electricity mix and a future scenario relying entirely on renewable electricity. There remains a gap in parameters specifically relating to the switch from diesel-powered to electric vehicles, highlighting the need to evaluate diesel's contribution explicitly to the environmental impact.

6.2 Apatite production environmental impact

The environmental impact of producing 1 tonne of apatite from secondary sources was compared with primary apatite production using mass allocation methods in both base and future scenarios. Under mass allocation, climate change impacts in the base scenario were slightly lower than the primary production reference, yet closely comparable. In the future scenario, climate change impacts significantly decreased to 19.67 kg CO_2, markedly lower compared to the 65.2 kg CO_2 reported by Rachid et al. (2025). This discrepancy arises primarily from the minor fraction of apatite produced within the overall system, with most environmental burdens allocated to residues.

However, this study did not apply economic allocation. In economic allocation, residues are typically assigned zero economic value and, therefore, receive no allocation. However, this approach may not be appropriate here since rare earth elements (REEs) in the tailings could be recovered. However, if economic allocation were used, the resulting factors would be 0.007 for iron and 0.993 for apatite (32% P_2O_5), reflecting the significantly higher economic value of apatite based on prevailing market prices (IndexMundi, 2024). As a result, economic allocation could lead to highly biased results in this case. This example illustrates how the allocation method can substantially influence LCA outcomes, highlighting the importance of LCA practitioners defining their study goals and carefully selecting allocation approaches. Future work should consider economic allocation when the characterization of REEs in the residues becomes more detailed.

6.3 LCIA method comparison results

Comparison of LCIA methods showed distinct outcomes. The ReCiPe 2016 method indicated considerable environmental benefits in both freshwater ecotoxicity and human toxicity, presenting negative net impacts. Conversely, USEtox 2.13 highlighted a slight environmental burden for freshwater ecotoxicity and only a minor benefit in human toxicity. These discrepancies are due to fundamental methodological differences between these approaches. ReCiPe aggregates midpoint indicators derived from USEtox, integrating normalization, weighting, and value-based choices, potentially amplifying specific impacts disproportionately. In contrast, USEtox 2.13 employs a detailed mechanistic model, calculating human health effects through fate, exposure, and effect factors, providing conservative, detailed results expressed in comparative toxic units (CTU). Consequently, USEtox offers nuanced, conservative toxicity assessments, whereas ReCiPe provides simplified, policy-oriented insights. This comparison highlights the critical role of method selection in interpreting toxicity outcomes in LCA studies.

6.4 Implications for technology designers

Sensitivity analysis revealed transport distance as a crucial factor. Even short distances, such as 5 km, can significantly increase CO_2 emissions by approximately 64%. Freshwater ecotoxicity also sharply increases from 2.45 CTUe at 5 km to 4.56 CTUe at 15 km. Cumulative Energy Demand (CED) also increases with transport distance, though more gradually. These findings underline the importance of situating pilot plants close to tailing repositories.

For technology designers, improving reprocessing methods to reduce overall energy consumption to 10 kWh per tonne of tailings could significantly lower CO_2 emissions to 31.57 kg CO_2 eq., demonstrating the substantial potential for energy-efficiency improvements. These findings provide valuable insights into the environmental impacts associated with tailings reprocessing. Future work should include a detailed contribution analysis of each process to identify areas for improvement. Decision-makers should prioritize investing in energy-saving technologies and carefully evaluate how higher apatite recovery rates could offset the overall environmental burden. In cases where apatite content is inherently low, using renewable electricity becomes a critical condition for sustainably recovering secondary materials from tailings.

7 Conclusion and future work

This study analysed the environmental impacts of the recovery process of historical iron ore tailings at the Grängesberg site in central Sweden, with apatite as the main product. The reprocessing using current technology demonstrates environmental benefits in several impact categories, including ecotoxicity (freshwater and marine), human toxicity (non-carcinogenic), and material resource use under ReCiPe 2016 method.

However, the process also introduces additional environmental burdens, particularly in the impact categories of climate change, land use, and water use under the ReCiPe 2016 method; cumulative energy demand (CED); and freshwater ecotoxicity under USEtox. In the future scenario assuming the exclusive use of renewable electricity, the impacts are reduced to only 5% of those in the base scenario, highlighting the significant environmental credits associated with renewable energy sources.

Despite this improvement, the renewable electricity scenario does not fully offset the environmental burden, suggesting that the recovery process remains energy-intensive. Improvements in processing technology are needed to reduce the consumption of key inputs, particularly electricity. The sensitivity analysis also indicates that transport distance is a critical parameter when evaluating the location of the reprocessing plant.

This study has certain limitations. Future work should include a detailed contribution analysis to identify which process stages can be optimised in terms of energy use and emissions. Additionally, it is necessary to develop an inventory for direct landfill disposal to allow for comprehensive environmental subtraction and to provide guidance for future tailings management. Finally, further studies could explore additional measures to reduce environmental impacts, such as breakeven analysis and the integration of circular economy strategies.

References

Adiansyah, J. S., Rosano, M., Vink, S., & Keir, G. (2015). A framework for a sustainable approach to mine tailings management: Disposal strategies. *Journal of Cleaner Production*, *108*, 1050–1062. https://doi.org/https://doi.org/10.1016/j.jclepro.2015.07.139

Adrianto, L. R., Ciacci, L., Pfister, S., & Hellweg, S. (2023). Toward sustainable reprocessing and valorization of sulfidic copper tailings: Scenarios and prospective lca. *Science of The Total Environment*, *871*, 162038. https://doi.org/https://doi.org/10.1016/j.scitotenv.2023.162038

Adrianto, L. R., & Pfister, S. (2022). Prospective environmental assessment of reprocessing and valorization alternatives for sulfidic copper tailings. *Resources, Conservation and Recycling*, *186*, 106567. https://doi.org/https://doi.org/10.1016/j.resconrec.2022.106567

Alagha, L., & Alsafasfeh, A. (2017). Recovery of phosphate minerals from plant tailings using direct froth flotation. *Minerals*, *7*. https://doi.org/10.3390/min7080145

Arvidsson, R., Tillman, A.-M., Sandén, B. A., Janssen, M., Nordelöf, A., Kushnir, D., & Molander, S. (2018). Environmental assessment of emerging technologies: Recommendations for prospective lca. *Journal of Industrial Ecology*, *22*(6), 1286–1294. https://doi.org/10.1111/jiec.12690

Awuah-Offei, K., & Adekpedjou, A. (2011). Application of life cycle assessment in the mining industry. *The International Journal of Life Cycle Assessment*, *16*, 82–89. https://doi.org/10.1007/s11367-010-0246-6

Baumann, H., & Tillman, A. (2004). *The hitch hiker's guide to lca: An orientation in life cycle assessment methodology and application*. Professional Publishing House. https://books.google.se/books?id=3QdsQgAACAAJ

Björklund, A. (2002). Survey of approaches to improve reliability in lca. *The International Journal of Life Cycle Assessment*, *7*, 64–72. https://doi.org/10.1007/BF02978849

Blengini, G. A., Mathieux, F., Mancini, L., Nyberg, M., & Viegas, H. M. (2019). *Recovery of critical and other raw materials from mining waste and landfills: State of play on existing practices* (J. Salminen, E. Garbarino, G. Orveillon, H. Saveyn, V. Mateos Aquilino, T. Llorens González, F. García Polonio, L. Horckmans, P. D'Hugues, E. Balomenos, G. Dino, M. de la Feld, F. Mádai, J. Földessy, G. Mucsi, I. Gombkötő, & I. Calleja, Eds.) [JRC116131]. Publications Office of the European Union. https://doi.org/10.2760/494020

Broadhurst, J., Kunene, M., von Blottnitz, H., & Franzidis, J.-P. (2015). Life cycle assessment of the desulfurisation flotation process to prevent acid rock drainage: A base metal case study [Sustainable Minerals]. *Minerals Engineering*, *76*, 126–134. https://doi.org/https://doi.org/10.1016/j.mineng.2014.10.013

Cairncross, K., & Tadie, M. (2022). Life cycle assessment as a design consideration for process development for value recovery from gold mine tailings. *Minerals Engineering*, *183*, 107588. https://doi.org/https://doi.org/10.1016/j.mineng.2022.107588

Canada Mining Innovation Council. (2011). *Canada mining innovation council business case: Towards zero wat* (Technical Report). CMIC. Canada. https://www.rethinkmining.org/wp-content/uploads/2020/10/CMIC-Business-Case_Final_2016.pdf

CEC Mining Systems. (2017). Power comparison study: Filtration equipment options [Accessed: 2025-05-08]. https://cecminingsystems.com/wp-content/uploads/2017/06/Power-Comparison-Study-Filtration-Equipment-Options.pdf

Cucurachi, S., van der Giesen, C., & Guinée, J. (2018). Ex-ante lca of emerging technologies [25th CIRP Life Cycle Engineering (LCE) Conference, 30 April – 2 May 2018, Copenhagen, Denmark]. *Procedia CIRP*, *69*, 463–468. https://doi.org/https://doi.org/10.1016/j.procir.2017.11.005

de Souza, N. R. D., Matt, L., Sedrik, R., Vares, L., & Cherubini, F. (2023). Integrating ex-ante and prospective life-cycle assessment for advancing the environmental impact analysis of emerging bio-based technologies. *Sustainable Production and Consumption*, *43*, 319–332. https://doi.org/https://doi.org/10.1016/j.spc.2023.11.002

Ekvall, T. (2019). Attributional and consequential life cycle assessment. In M. J. Bastante-Ceca, J. L. Fuentes-Bargues, L. Hufnagel, F.-C. Mihai, & C. Iatu (Eds.), *Sustainability assessment at the 21st century*. IntechOpen. https://doi.org/10.5772/intechopen.89202

Ekvall, T., & Andrae, A. (2006). Attributional and consequential environmental assessment of the shift to lead–free solders. *The International Journal of Life Cycle Assessment*, *11*, 344–353. https://doi.org/10.1065/lca2005.05.208

Farjana, S. H., Huda, N., Mahmud, M. A. P., & Saidur, R. (2019). A review on the impact of mining and mineral processing industries through life cycle assessment. *Journal of Cleaner Production*, *231*, 1200–1217. https://doi.org/10.1016/j.jclepro.2019.05.264

Flemström, K., & Pålsson, A.-C. (2003). *Introduction and guide to lca data documentation using the cpm documentation criteria and the iso/ts 14048 data documentation format* (Technical Report No. CPM Report 2003:3). IMI - Industrial Environmental Informatics, CPM - Centre for Environmental Assessment of Product and Material Systems, Chalmers University of Technology.

Goedkoop, M., Oele, M., Vieira, M., Leijting, J., Ponsioen, T., & Meijer, E. (2014). *Simapro tutorial* [SimaPro Tutorial Document]. PRé Sustainability.

Grzesik, K., Kossakowska, K., Bieda, B., & Kozakiewicz, R. (2019). Screening life cycle assessment of beneficiation processes for rare earth elements recovery from secondary sources. *IOP Conference Series: Earth and Environmental Science*, *214*(1), 012068. https://doi.org/10.1088/1755-1315/214/1/012068

Guinée, J. B. (2002). *Handbook on life cycle assessment: Operational guide to the iso standards* (Vol. 7). Springer Science & Business Media.

Guo, M., & Murphy, R. (2012). Lca data quality: Sensitivity and uncertainty analysis. *Science of The Total Environment*, *435-436*, 230–243. https://doi.org/https://doi.org/10.1016/j.scitotenv.2012.07.006

IndexMundi. (2024). Iron ore and rock phosphate prices [Accessed: April 22, 2025]. https://www.indexmundi.com/commodities/?commodity=iron-ore&months=60&currency=eur&commodity=rock-phosphate

International Organization for Standardization (ISO). (2006). *ISO 14044: Environmental Management – Life Cycle Assessment – Requirements and Guidelines*. International Organization for Standardization. Geneva.

Jolsterå, R., Öberg, E., Gärde, J., & Aukea, N. (2025). Bench scale testwork with malmberget tailing materials: Sampling, sample preparations and testwork procedure. *Conference in Minerals Engineering*, *176*, 57–66.

Klöpffer, W. (2002). Iso/tr 14049. *International Journal of Life Cycle Assessment - INT J LIFE CYCLE ASSESS*, *7*, 53–53. https://doi.org/10.1007/BF02978910

KONARÉ, Z., Ajayi, D., BA, S., & AREMU, A. (2024). Life cycle assessment of an industrial gold mining in mali. *The International Journal of Life Cycle Assessment*, *29*, 1–17. https://doi.org/10.1007/s11367-024-02299-6

Kossakowska, K., & Grzesik, K. (2019). The significance of energy consumption in environmental impact of rare earth elements recovery from tailings and mining waste. *E3S Web of Conferences*, *108*, 02011. https://doi.org/10.1051/e3sconf/201910802011

Kossoff, D., Dubbin, W., Alfredsson, M., Edwards, S., Macklin, M., & Hudson-Edwards, K. (2014). Mine tailings dams: Characteristics, failure, environmental impacts, and remediation. *Applied Geochemistry*, *51*, 229–245. https://doi.org/https://doi.org/10.1016/j.apgeochem.2014.09.010

Kruyswijk, J. B., et al. (2021). The balance between energy and water preservation in the deposition of dry tailings in wet climates [Shows dewatering energy: paste (1kWh/t), filtered (2kWh/t), filtered+cake blowing (6.2kWh/t)]. *Paste 2021 Conference*, 147–151.

Lindholm, T. (2022). *Jan-matts tailings dam, grängesberg updated mineral resource estimate* (tech. rep. No. 3). GeoVista AB. https://doi.org/10.1234/example.2020.001

Lu, T., Chen, W.-Q., Ma, Y., Qian, Q., & Jia, J. (2023). Environmental impacts and improvement potentials for copper mining and mineral processing operations in china. *Journal of Environmental Management*, *342*, 118178. https://doi.org/https://doi.org/10.1016/j.jenvman.2023.118178

Metso Outotec. (2020). Basics in minerals processing handbook [Accessed: 2025-05-07].

Muller, S., Lai, F., Nucci, M., Segù, E., Crane, R., Nash, W., Wrana, A., Bezak, B., & Ferrara, L. (2023). Life-cycle assessment of coal mining wastes upcycling, 2531–2538. https://doi.org/10.1201/9781003323020-308

Palacios, J.-L., Fernandes, I., Abadias, A., Valero, A., Valero, A., & Reuter, M. A. (2019). Avoided energy cost of producing minerals: The case of iron ore. *Energy Reports*, *5*, 364–374. https://doi.org/https://doi.org/10.1016/j.egyr.2019.03.004

Pedersen, J., Ulrich, T., Whitehouse, M., Kent, A., & Tegner, C. (2021). The volatile and trace element composition of apatite in the skaergaard intrusion, east greenland. *Contributions to Mineralogy and Petrology*, *176*. https://doi.org/10.1007/s00410-021-01861-x

Rachid, S., Taha, Y., Muller, E., & Benzaazoua, M. (2025). Life cycle assessment of phosphate mining and beneficiation in morocco: Performance evaluation. *Journal of Environmental Management*, *373*, 123453. https://doi.org/https://doi.org/10.1016/j.jenvman.2024.123453

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B., & Pennington, D. (2004). Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications.

Environment International, *30*(5), 701–720. https://doi.org/https://doi.org/10.1016/j.envint.2003.11.005

Sacchi, R., Terlouw, T., Siala, K., Dirnaichner, A., Bauer, C., Cox, B., Mutel, C., Daioglou, V., & Luderer, G. (2022). Prospective environmental impact assement (premise): A streamlined approach to producing databases for prospective life cycle assessment using integrated assessment models. *Renewable and Sustainable Energy Reviews*, *160*, 112311. https://doi.org/https://doi.org/10.1016/j.rser.2022.112311

Saldanha, R. B., Caicedo, A. M. L., de Araújo, M. T., Scheuermann Filho, H. C., Moncaleano, C. J., Silva, J. P. S., & Consoli, N. C. (2023). Potential use of iron ore tailings for binder production: A life cycle assessment. *Construction and Building Materials*, *365*, 130008. https://doi.org/https://doi.org/10.1016/j.conbuildmat.2022.130008

Schrijvers, D., Loubet, P., & Sonnemann, G. (2020). Archetypes of goal and scope definitions for consistent allocation in lca. *Sustainability*, *12*(14). https://www.mdpi.com/2071-1050/12/14/5587

Shandong Huate Magnet Technology Co., Ltd. (2025). Lims low intensity wet drum magnetic separator [Accessed: 2025-05-07].

Sis, H., & Chander, S. (2003). Reagents used in the flotation of phosphate ores: A critical review. *Minerals Engineering*, *16*(7), 577–585. https://doi.org/https://doi.org/10.1016/S0892-6875(03)00131-6

Son, T., Trinh, H. B., Kim, S., Dugarjav, B., & Lee, J. (2023). Estimation of energy consumption for concentrate process of tungsten ore towards the integration of renewable energy sources in mongolia. *Minerals*, *13*(8). https://www.mdpi.com/2075-163X/13/8/1059

Stavropoulos, P., Giannoulis, C., Papacharalampopoulos, A., Foteinopoulos, P., & Chryssolouris, G. (2016). Life cycle analysis: Comparison between different methods and optimization challenges [Research and Innovation in Manufacturing: Key Enabling Technologies for the Factories of the Future - Proceedings of the 48th CIRP Conference on Manufacturing Systems]. *Procedia CIRP*, *41*, 626–631. https://doi.org/https://doi.org/10.1016/j.procir.2015.12.048

Tampubolon, F. R., Yuwono, A., Tambunan, A., & Achsani, N. (2021). Coal mining energy utilization and environmental impact management strategy using the lca method. *Nature Environment and Pollution Technology*, *20*. https://doi.org/10.46488/NEPT.2021.v20i05.017

The Weir Group. (2020). Sandmaster™ sand wash plant brochure [Accessed: 2025-05-05].

Thonemann, N., Schulte, A., & Maga, D. (2020). How to conduct prospective life cycle assessment for emerging technologies? a systematic review and methodological guidance. *Sustainability*, *12*(3). https://www.mdpi.com/2071-1050/12/3/1192

Tillman, A.-M. (2000). Significance of decision-making for lca methodology. *Environmental Impact Assessment Review*, *20*(1), 113–123. https://doi.org/https://doi.org/10.1016/S0195-9255(99)00035-9

U.S. Department of Energy. (2007a). *Mining industry energy bandwidth study* (tech. rep.). Industrial Technologies Program, Office of Energy Efficiency and Renewable Energy. https://www.energy.gov/eere/amo/downloads/mining-energy-bandwidth-study

U.S. Department of Energy. (2007b). *U.s. mining industry energy bandwidth study* (Technical Report). Office of Energy Efficiency Renewable Energy, Industrial Technologies Program. Washington, D.C. https://www1.eere.energy.gov/manufacturing/resources/mining/pdfs/mining_bandwidth.pdf

Valderrama, L., Gómez, O., Pavez, O., & Santander, M. (2024). Recovery of apatite from magnetic concentration tailings by flotation. *Minerals*, *14*(5). https://www.mdpi.com/2075-163X/14/5/441

Volvo Construction Equipment. (2025, April). *Volvo ce unveils world-first electric articulated haulers* [Accessed: 2025-05-20]. https://www.volvoce.com/global/en/news-and-events/news-and-stories/2025/volvo-ce-unveils-world-first-electric-articulated-haulers/

Wills, B. A., & Finch, J. A. (2016). *Wills' mineral processing technology: An introduction to the practical aspects of ore treatment and mineral recovery* (8th). Butterworth-Heinemann. https://www.elsevier.com/books/wills-mineral-processing-technology/wills/978-0-08-097053-0

Zhang, S., Huang, Y., Liu, X., & Wang, H. (2023). Effect of flocculant dosage on the settling properties and underflow concentration of thickener for flocculated tailing suspensions. *Water Science & Technology*, *88*(9), 2144–2156. https://doi.org/10.2166/wst.2023.137

A

Data associated with the reprocessing operation

Table A.1: Process parameters used in constructing apatite life cycle inventory data of the reprocessing

Process parameters	**Values**	**Unit**	**References**	**Comments**
Mass of tailings	560,640	ton/y	Collected from partners	
Moisture	15%	-	Inhouse data	Collected from partners
Solid content tailings	476,544	ton/y	Calculated	Average data from samples
Water consumption	311.3	ton/y	Inhouse data	Collected from partners
Electricity for scrubbing	82,447	kWh/y tailings	Calculated	
Electricity for grinding	5,059,776	kWh/y	Inhouse data, calculated	Collected from partners
Electricity for magnetic separation process	341,289.6	kWh/y	Inhouse data, assumption and calculation	Collected from partners
Electricity for desliming	182,208	kWh/y	Calculated	
Electricity for flotation	353,203.2	kWh/y	Inhouse data, calculated	Collected from partners
Electricity for dewatering	2,638,512	kWh/y	Inhouse data, calculated	Collected from partners
Electricity of ancillary systems (conveyors, fans, compressors, lighting)	560,640	kWh/y	Industry average data	
Total electricity consumption	11,979,461	kWh/y	Calculated	
The amount of collector (fatty acid)	311,856	kg/y	Calculated	Consumption based on studies

Process parameters	Values	Unit	References	Comments
The amount of depressant (sodium silicate)	391,046.4	kg/y	Calculated	Consumption based on studies
The amount of flocculant	21,024	kg/y	Calculated	
Fraction to Fe	0.059	-	Inhouse data	Collected from partners
Fraction to apatite	0.11	-	Inhouse data	Collected from partners
Fraction to tailing	0.81	-	Inhouse data	Collected from partners
Fraction to 32% P_2O_5	0.131	-	Calculated	

B

On the conversion from apatite concentrate to P_2O_5

In order to compare the environmental impacts of primary phosphate rock (32% P_2O_5) production in the existed dataset, it is necessary to consider the conversion from the apatite to P_2O_5 in this study.

In the context of mineral processing and fertiliser reporting, P_2O_5 is not a distinct substance extracted through an additional processing step, but rather a conventional unit used to express the phosphorus content of phosphate minerals. Apatite, the principal phosphorus-bearing mineral in this study, contains phosphorus in the form of phosphate ions (${PO_4}^{3-}$) within its crystalline structure (e.g., $Ca_5(PO_4)_3F$)(Pedersen et al., 2021). The expression of phosphorus content as P_2O_5 is based on stoichiometric equivalence. Specifically, the mass ratio of P_2O_5 to elemental phosphorus (P) is:

$$\frac{M_{\mathrm{P_2O_5}}}{2 \times M_{\mathrm{P}}} = \frac{141.94}{2 \times 30.97} \approx 2.29 \tag{B.1}$$

Therefore, to convert the phosphorus content in the concentrate to an equivalent amount of P_2O_5, no additional chemical process is required. This is a calculational convention widely used in mineral resource estimates and life cycle assessments.

The annual output of apatite concentrate is reported as 61,670 tonnes per year, with a phosphorus (P) content of 17 wt% (Lindholm, 2022). The corresponding annual production of P_2O_5 is calculated as:

$$\mathrm{P_2O_5}\ (\mathrm{t/year}) = 61,670\ \mathrm{t} \times 0.17 \times 2.29 = 24,012\ \mathrm{t/year} \tag{B.2}$$

This value represents the output of the upstream beneficiation process, where the mineral is concentrated as the final product.

To compare the result with the dataset for 32% P_2O_5 used in the GaBi database, the equivalent mass is calculated using the following conversion:

$$\text{Equivalent mass at 32\% P}_2\text{O}_5 = \frac{\text{Mass of P}_2\text{O}_5}{0.32} \tag{B.3}$$

Thus, the 24,012 tonnes of P_2O_5 concentrate is equivalent to:

$$\frac{24,012}{0.32} \approx 75,037\,\text{tonnes of 32\% P}_2\text{O}_5 \tag{B.4}$$

C

Calculation of excavation equipments

C.1 Transport throughput requirements

To transport 600,000 tonnes of iron ore from the mine to the processing facility located 5 km away (10 km round-trip), the transport operation is scheduled over 250 working days, operating on a single 8-hour shift per day. The total available operational time is:

$$250 \text{ days} \times 8 \text{ hours/day} = 2{,}000 \text{ hours} \tag{C.1}$$

To complete the transport within this period, the minimum required average throughput is:

$$\frac{600{,}000 \text{ tonnes}}{2{,}000 \text{ hours}} = 300 \text{ tonnes/hour} \tag{C.2}$$

A target throughput of 350 tonnes per hour is selected to provide a buffer for variability in cycle times, mechanical delays, and other uncertainties, ensuring timely project completion.

C.2 Excavator and hauler requirements (Diesel)

The use of 100-tonne capacity hauler is assumed, corresponding to large rigid-frame haul trucks commonly used in mining operations. With the one-way distance of 5 km, the breakdown of a typical cycle is as follows:

- Loaded travel (5 km at 25 km/h): ∼12 minutes
- Dumping and repositioning: ∼5 minutes
- Return empty (5 km at 35 km/h): ∼9 minutes
- Queuing and operational buffer: ∼4 minutes

Total estimated cycle time: ∼30 minutes.
Under this assumption, each truck can perform two full trips per hour, delivering:

$$2\,\text{trips/hour} \times 100\,\text{tonnes/trip} = 200\,\text{tonnes/hour} \tag{C.3}$$

To achieve the required throughput of 350 tonnes/hour:

$$\frac{350\,\text{tonnes/hour}}{200\,\text{tonnes/hour/truck}} = 1.75 \implies \mathbf{2} \text{ trucks} \qquad \text{(rounded up)} \tag{C.4}$$

The daily hauling capacity using two trucks over an 8-hour shift is:

$$2\,\text{trucks} \times 2\,\text{trips/hour} \times 100\,\text{tonnes/trip} \times 8\,\text{hours} = 3{,}200\,\text{tonnes/day} \tag{C.5}$$

The total hauling capacity over the full operational period is:

$$3{,}200\,\text{tonnes/day} \times 250\,\text{days} = 800{,}000\,\text{tonnes} \tag{C.6}$$

This provides ample capacity to move the required material volume and accommodates operational variability.
The diesel consumption of a 100-tonne hauler is commonly 40 liters per hour. Thus, the annual diesel consumption is:

$$40\,\text{L/hour} \times 8\,\text{hours/day} \times 250\,\text{days/year} \times 2\,\text{trucks} = 160{,}000\,\text{L/year} \tag{C.7}$$

An excavator equipped with a 10 m^3 bucket is assumed for truck loading. Given the typical bulk density of iron ore ($\sim$2.6 t/m^3), each bucket can lift approximately 25-26 tonnes. Each 100-tonne truckload requires approximately four buckets and takes about 1.5 minutes to load. A single excavator is therefore capable of loading significantly more than the required number of trucks per hour.
The diesel consumption of a 100-tonne excavator is commonly 25 liters per hour. The annual diesel consumption is:

$$25\,\text{L/hour} \times 8\,\text{hours/day} \times 250\,\text{days/year} = 50{,}000\,\text{L/year} \tag{C.8}$$

The density of low sulfur diesel is approximately 0.82 kg/L, and the annual total diesel consumption of the equipment is 172,200 kg.

C.3 Electrical hauler requirements and energy consumption

Given the intention to utilize fully electric equipment, the transport operation is planned with Volvo A40 Electric articulated hauler, each with a payload capacity of 40 tonnes per trip. For a 5 km one-way haul (10 km round-trip), the average cycle time—including loading, travel, dumping, return, and queuing is estimated at 30 minutes, resulting in 2 trips per hour per truck.
The required number of truck trips per hour is:

$$\frac{350 \text{ tonnes/hour}}{40 \text{ tonnes/trip}} = 8.75 \text{ trips/hour} \tag{C.9}$$

Considering the need for charging, each truck operates for 7 hours and charges for 1 hour during an 8-hour shift.

$$\text{Effective trips/hour per truck} = \frac{14 \text{ trips}}{8 \text{ hours}} = 1.75 \text{ trips/hour} \tag{C.10}$$

The number of trucks required, accounting for charging, is:

$$\frac{8.75 \text{ trips/hour}}{1.75 \text{ trips/hour/truck}} = 5 \text{ trucks} \tag{C.11}$$

According to Volvo Construction Equipment, the A40 Electric articulated hauler is equipped with a 350 kWh battery, designed to provide 4–4.5 hours of operation per charge under typical conditions (Volvo Construction Equipment, 2025). This corresponds to an average energy consumption of:

$$\text{Average energy consumption per hour} = \frac{350 \text{ kWh}}{4.5 \text{ h}} = 77.8 \text{ kWh/h} \tag{C.12}$$

For an 8-hour shift, each truck requires:

$$77.8 \text{ kWh/h} \times 7 \text{ h} = 544.6 \text{ kWh/shift} \tag{C.13}$$

Over 250 shifts per year, the annual consumption per truck is:

$$544.6 \text{ kWh/shift} \times 250 \text{ shifts} = 136,150 \text{ kWh/year} \tag{C.14}$$

For a fleet of five trucks, the total annual electricity demand is:

$$136,150 \text{ kWh/year} \times 5 = 680,750 \text{ kWh/year} = 680.75 \text{ MWh/year} \tag{C.15}$$

Therefore, operating a fleet of five Volvo A40 Electric haulers over 250 shifts per year requires approximately 680 MWh of electricity annually.

C.4 Electric excavator requirements and energy consumption

The Volvo EC230 Electric excavator, equipped with a 264 kWh battery, is selected for loading operations. With a bucket size of approximately 1.25 m^3 and an iron ore bulk density of 2.6 t/m^3, each bucket handles around 3.3 tonnes. Thus, loading one 40-tonne truck requires about 12-13 buckets, typically taking 6-7 minutes per truck at a 30-second bucket cycle.

Given a target truck frequency of 10 loads per hour (from a five-truck fleet), the excavator(s) must deliver 120-130 buckets per hour. As this rate exceeds the recommended single-machine capacity for the EC230 Electric, two excavators working in parallel are required to efficiently load all trucks and avoid queueing delays.

Field tests and manufacturer data indicate that the EC230 Electric (264 kWh) can operate for approximately 4-5 hours per full charge under moderate-to-heavy duty cycles, corresponding to an average hourly energy consumption between 53 and 66 kWh. For a typical mining duty cycle, a conservative average of 55 kWh/h is assumed.

Thus, for a standard 8-hour shift, the energy consumption per excavator is:

$$E_{\text{shift}} = 8 \text{ h} \times 55 \text{ kW} = 440 \text{ kWh} \tag{C.16}$$

When the required workload is shared between two excavators, each machine consumes approximately half this energy, or 220 kWh per shift:

$$E_{\text{shift, per machine}} = \frac{440 \text{ kWh}}{2} = 220 \text{ kWh} \tag{C.17}$$

This value is within the battery's 264 kWh capacity, allowing each excavator to complete its shift without mid-shift recharging.

A key operational distinction between electric and diesel excavators is the required downtime for battery charging. After each 8-hour shift, the EC230 Electric typically requires about 2 hours to fully recharge with a high-capacity charger, resulting in an availability of:

$$\text{Availability} = \frac{\text{Working time}}{\text{Working time} + \text{Charging time}} = \frac{8}{8+2} = 0.8 \text{ or } 80\% \tag{C.18}$$

To maintain continuous two-excavator operation, the fleet size must be increased:

$$N_{\text{total}} = \frac{N_{\text{required (active)}}}{\text{Availability}} = \frac{2}{0.8} = 2.5 \tag{C.19}$$

Rounding up, a minimum of three Volvo EC230 Electric excavators is required: two actively working, with the third rotating through charging cycles. This configuration ensures uninterrupted loading productivity.
Therefore, the annual energy consumption for three EC230 Electric excavators, working one 8-hour shift per day for 250 days, is:

$$\text{Annual energy consumption} = 3 \times 220 \text{ kWh/shift} \times 250 \text{ shifts} = 165,000 \text{ kWh} \tag{C.20}$$

Thus, the total electricity required for excavator operations over the project year is approximately 165 MWh.

D

Complete LCA results according to all assessed indicators

This chapter presents the full life cycle assessment (LCA) results according to all assessed indicators for both the base and future scenarios.

Base scenario

RECIPE Midpoint (H) V1.03		The impact of the reprocessing	Primary iron production(60 kg)	Primary P2O5 production(133kg)	Net environmental impact
Impact category	Reference unit				
Acidification: terrestrial	kg SO2-Eq	0.337028477	0.005774416	0.005560781	0.32569328
Climate change	kg CO2-Eq	94.88133043	1.404913715	14.66046888	78.81594784
Ecotoxicity: freshwater	kg 1,4-DCB-Eq	6.01120951	19.15962284	0.000447085	-13.14886042
Ecotoxicity: marine	kg 1,4-DCB-Eq	7.191427421	24.19131224	0.001152764	-17.00103759
Ecotoxicity: terrestrial	kg 1,4-DCB-Eq	373.9495198	2.248514475	0.948058265	370.752947
Energy resources: non-renewable, fossil	kg oil-Eq	48.24622594	0.34761958	3.805480984	44.09312537
Eutrophication: freshwater	kg P-Eq	0.014574341	0.026295775	1.34E-05	-0.011734786
Eutrophication: marine	kg N-Eq	0.037732334	0.000401328	0.000106563	0.037224443
Human toxicity: carcinogenic	kg 1,4-DCB-Eq	16.54315706	2.925068961	0.009404729	13.60868337
Human toxicity: non-carcinogenic	kg 1,4-DCB-Eq	130.7759147	796.8592928	-0.017172095	-666.066206
Ionising radiation	kBq Co-60-Eq	213.7635831	0.03393277	0.23837125	213.4912791
Land use	m2*a crop-Eq	45.25305208	0.350644281	0.278084791	44.62432301
Material resources: metals/minerals	kg Cu-Eq	2.049014892	3.463741158	42.90629891	-44.32102517
Ozone depletion	kg CFC-11-Eq	0.000135017	7.12067E-06	2.07316E-06	0.000125823
Particulate matter formation	kg PM2.5-Eq	0.116975428	0.002663651	0.001784921	0.112526855
Photochemical oxidant formation: human health	kg NOx-Eq	0.288568681	0.009707544	0.007420753	0.271440383
Photochemical oxidant formation: terrestrial ecosystems	kg NOx-Eq	0.316022797	0.009982445	0.007500314	0.298540038
Water use	m3	4.162145168	0.057724325	0.034604194	4.069816649

Figure D.1: Complete LCA results according to all assessed indicators in Recipe2016 for the base scenario.

Future scenario

RECIPE Midpoint (H) V1.03		The impact of the reprocessing	Primary iron production(60 kg)	Primary P2O5 production(133kg)	Net environmental impact
Impact category	Reference unit				
Acidification: terrestrial	kg SO2-Eq	0.119243786	0.005774416	0.005560781	0.10790859
Climate change	kg CO2-Eq	20.11023446	1.404913715	14.66046888	4.044851864
Ecotoxicity: freshwater	kg 1,4-DCB-Eq	6.017527679	19.15962284	0.000447085	-13.14254225
Ecotoxicity: marine	kg 1,4-DCB-Eq	6.880437699	24.19131224	0.001152764	-17.31202731
Ecotoxicity: terrestrial	kg 1,4-DCB-Eq	103.9854418	2.248514475	0.948058265	100.7888691
Energy resources: non-renewable, fossil	kg oil-Eq	3.179905603	0.34761958	3.805480984	-0.973194961
Eutrophication: freshwater	kg P-Eq	0.007757081	0.026295775	1.34E-05	-0.018552046
Eutrophication: marine	kg N-Eq	0.031945836	0.000401328	0.000106563	0.031437945
Human toxicity: carcinogenic	kg 1,4-DCB-Eq	10.81914031	2.925068961	0.009404729	7.884666622
Human toxicity: non-carcinogenic	kg 1,4-DCB-Eq	67.99546477	796.8592928	-0.017172095	-728.8466559
Ionising radiation	kBq Co-60-Eq	0.54007694	0.03393277	0.23837125	0.267772921
Land use	m2*a crop-Eq	25.15674814	0.350644281	0.278084791	24.52801907
Material resources: metals/minerals	kg Cu-Eq	1.162188053	3.463741158	42.90629891	-45.20785201
Ozone depletion	kg CFC-11-Eq	8.9385E-05	7.12067E-06	2.07316E-06	8.01911E-05
Particulate matter formation	kg PM2.5-Eq	0.041761804	0.002663651	0.001784921	0.037313232
Photochemical oxidant formation: human health	kg NOx-Eq	0.053245189	0.009707544	0.007420753	0.036116891
Photochemical oxidant formation: terrestrial ecosystems	kg NOx-Eq	0.0560582	0.009982445	0.007500314	0.038575441
Water use	m3	10.60253386	0.057724325	0.034604194	10.51020534

Figure D.2: Complete LCA results according to all assessed indicators in Recipe2016 midpoint for the future scenario

Base scenario

Impact category	Reference unit	The impact of the reprocessing	Primary iron production(60 kg)	Primary P2O5 production(133kg)	Net impact
Energy resources: non-renewable	MJ-Eq	6104.111	16.722	174.291	5913.098
Energy resources	MJ-Eq	7856.16	26.52	219.20	7610.44

Future scenario

Impact category	Reference unit	The impact of the reprocessing	Primary iron production(60 kg)	Primary P2O5 production(133kg)	Net impact
Energy resources: non-renewable	MJ-Eq	169.292	16.722	174.291	-21.722
Energy resources	MJ-Eq	2995.04	26.52	219.20	2749.32

Figure D.3: Complete LCA results according to CED indicators for both scenarios

Base scenario

Impact category	Reference unit	The impact of the reprocessing	Primary iron production(60 kg)	Primary P2O5 production(133kg)	Net impact
USEtox 2.13 Ecotoxicity: freshwater	CTUe	4.9279	3.52282	0.0087	1.3964
USEtox 2.13 Human toxicity	CTUh	6.4106E-05	0.002851	1.31E-09	-2.79E-03

Future scenario

Impact category	Reference unit	The impact of the reprocessing	Primary iron production(60 kg)	Primary P2O5 production(133kg)	Net impact
USEtox 2.13 Ecotoxicity: freshwater	CTUe	4.378211016	3.52282	0.0087	0.8467
USEtox 2.13 Human toxicity	CTUh	3.25E-05	0.002851	1.31E-09	-2.82E-03

Figure D.4: Complete LCA results according to USEtox indicators for both scenarios

Impact category	Reference unit	Net environmental impact with energy consumption at 10 kW	Net environmental impact with energy consumption at 25 kW
Acidification: terrestrial	kg SO2-Eq	0.17014480	0.375842952
Climate change	kg CO2-Eq	31.56990045	94.04832926
Ecotoxicity: freshwater	kg 1,4-DCB-Eq	-16.98065853	-11.91346797
Ecotoxicity: marine	kg 1,4-DCB-Eq	-21.94697777	-15.4064397
Ecotoxicity: terrestrial	kg 1,4-DCB-Eq	115.61480559	453.0108649
Energy resources: non-renewable, fossil	kg oil-Eq	35.87978087	46.7411521
Eutrophication: freshwater	kg P-Eq	-0.02030044	-0.008973173
Eutrophication: marine	kg N-Eq	0.03285995	0.03863158
Human toxicity: carcinogenic	kg 1,4-DCB-Eq	3.90295119	16.73786396
Human toxicity: non-carcinogenic	kg 1,4-DCB-Eq	-756.50349169	-636.9087355
Ionising radiation	kBq Co-60-Eq	20.69215957	275.6507589
Land use	m2*a crop-Eq	20.28814429	52.47043876
Material resources: metals/minerals	kg Cu-Eq	-45.36322439	-43.9850145
Ozone depletion	kg CFC-11-Eq	0.00005377	0.000149054
Particulate matter formation	kg PM2.5-Eq	0.05738963	0.130303394
Photochemical oxidant formation: human health	kg NOx-Eq	0.10908435	0.323784847
Photochemical oxidant formation: terrestrial ecosystems	kg NOx-Eq	0.13207124	0.352210481
Water use	m3	2.48530622	4.580671389
CED Energy resources: non-renewable	MJ-Eq	2047.074841	7159.524801
USEtox 2.13 Ecotoxicity: freshwater	CTUe	-1.990865742	2.488514752
USEtox 2.13 Human toxicity	CTUh	-0.002851279	-0.002774142

Figure D.5: Complete LCA results are provided for all indicators as part of the sensitivity analysis with varying energy input levels.

Impact category	Reference unit	Net environmental impact with a transport distance of 5 km	Net environmental impact with a transport distance of 15 km
Acidification: terrestrial	kg SO2-Eq	0.42327988	0.61825088
Climate change	kg CO2-Eq	130.1855711	232.84487628
Ecotoxicity: freshwater	kg 1,4-DCB-Eq	-11.76385316	-8.99710263
Ecotoxicity: marine	kg 1,4-DCB-Eq	-14.49147783	-9.47708439
Ecotoxicity: terrestrial	kg 1,4-DCB-Eq	1549.709095	3906.36577603
Energy resources: non-renewable, fossil	kg oil-Eq	60.74982368	94.03168679
Eutrophication: freshwater	kg P-Eq	-0.008187657	-0.00110150
Eutrophication: marine	kg N-Eq	0.037967481	0.03944073
Human toxicity: carcinogenic	kg 1,4-DCB-Eq	22.47857511	40.20450386
Human toxicity: non-carcinogenic	kg 1,4-DCB-Eq	-621.502494	-532.45998037
Ionising radiation	kBq Co-60-Eq	214.2087831	215.57446865
Land use	m2*a crop-Eq	47.96689923	54.63430314
Material resources: metals/minerals	kg Cu-Eq	-42.87697005	-39.99090855
Ozone depletion	kg CFC-11-Eq	0.000150053	0.00019845
Particulate matter formation	kg PM2.5-Eq	0.161362417	0.25894895
Photochemical oxidant formation: human health	kg NOx-Eq	0.433384902	0.75702531
Photochemical oxidant formation: terrestrial ecosystems	kg NOx-Eq	0.472693116	0.82073007
Water use	m3	4.169325982	4.36691250
CED Energy resources: non-renewable, fossil	MJ-Eq	6694.604972	8254.90637405
USEtox 2.13 Ecotoxicity: freshwater	CTUe	2.45256601	4.562235176
USEtox 2.13 Human toxicity	CTUh	-0.002767346	-0.002727728

Figure D.6: Complete LCA results are provided for all indicators as part of the sensitivity analysis with varying transport distances from the excavation site to the reprocessing plant.

DEPARTMENT OF SOME SUBJECT OR TECHNOLOGY
CHALMERS UNIVERSITY OF TECHNOLOGY
Gothenburg, Sweden
www.chalmers.se

CHALMERS
UNIVERSITY OF TECHNOLOGY