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Life Cycle Assessment of Portland Cement Alternatives in Mine Paste Backfill

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Abstract

Mining activities generate huge volumes of mine tailings (MTs), which pose huge environmental management challenges. Reuse as cemented paste backfill (CPB), a mixture of tailings with water and a binder—often cementitious or alkaline—is amongst the best methods to reduce surface disposal, and it is used to backfill underground mine voids. Although the most widely used binder in CPB production remains Ordinary Portland Cement (OPC), it is associated with a high carbon footprint and a high economic cost. In this study, both the economic feasibility and the environmental performance of three alkaline activators—sodium hydroxide (NaOH), sodium silicate (Na_2SiO_3), and a high $MgCO_3$ and MgO content calcined magnesite residue—are evaluated as OPC replacements in CPB products. A gate-to-grave life cycle assessment (LCA) was performed at a CPB plant located in southwestern Spain with the use of tailings from a massive sulfide deposit. The results from the uniaxial compressive strength test and LCA demonstrate that paste formulations using the magnesite residue achieve comparable mechanical performance while significantly reducing both the environmental footprint and total cost relative to OPC-based mixtures. These results support the use of alkaline binders as viable substitutes that enable more sustainable and cost-effective tailings management practices in the mining sector.

Keywords: mine tailings; life cycle impact assessment; sodium hydroxide; magnesium oxide; cemented paste backfill

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1. Introduction

Modern societies rely on substantial amounts of raw materials from mining while aiming for sustainable development. Future efforts will focus on achieving a circular economy, where resources are reused, and environmental impacts are minimized [1–5]. Among the environmental challenges faced by the mining industry, mine tailings (MTs) represent the most significant challenge due to their volume. The industry generates 12.7 km^3 of tailings annually, contributing to the existing 217.3 km^3 of tailings stored globally [6,7]. To avoid surface storage facilities (and their environmental and social risks) as much as possible, efforts are being directed towards adding value to mine tailings. Three approaches have been reported in the literature: (1) Reprocessing to recover remaining metals [8–13]; (2) Developing new geomaterials, such as geopolymers [14–17], bricks [18–21], ceramics [22,23], concrete [24–27], mortars [28], supplementary cementitious materials [29–31], and others [32]; and (3) Reusing tailings for backfilling mined-out stopes [33–38]. The latter is particularly effective for safely storing huge volumes of mine tailings underground, thereby minimizing their accumulation in dams, decreasing the surface footprint of mining operations, reducing potential environmental impacts, and freeing up land for alternative uses. Additionally, using tailings for backfilling stabilizes underground voids, reduces the risk of subsidence, and enhances mine safety [39,40].

Cemented paste backfill (CPB) is made by blending mine tailings (MTs) with water and a carefully chosen binder, as both the type and quantity of binder significantly influence the strength of the final product [41]. Typically, for effective performance, the compressive strength of the cemented paste should range between 0.5 and 5 MPa, depending on its specific application [40,42].

Ordinary Portland Cement (OPC) has been the most widely used binder by various mining companies [43–45]. However, Portland cement is widely recognized for its high carbon footprint, primarily due to the calcination process, which generates significant CO_2 emissions [46,47]. Therefore, there is a growing need to explore alternative binders and activators for CPB. Research in this area has led to the successful application of alternative binders with pozzolanic activity, such as fly ash, slag, silica fume, waste glass, or volcanic tuff, and alkaline activators like hydrated lime ($Ca(OH)_2$), limestone powder, magnesia powder (MgO), sodium hydroxide ($NaOH$), and sodium silicates (Na_2SiO_3) [38,43,48]. The selection of a binder and/or an activator depends not only on its cost but also on the required mechanical strength of the cemented paste. Striking the right balance between economic feasibility and material performance is critical. To discern which binder/activator offers the best overall environmental and performance benefits, life cycle assessment (LCA) serves as a valuable tool.

LCA is a comprehensive framework used to evaluate the environmental performance of a system or process throughout its life cycle. It is structured into four main steps: (1) definition of the goal and scope of the study, (2) development of the Life Cycle Inventory (LCI), which involves collecting and quantifying all relevant inputs and outputs, (3) application of the Life Cycle Impact Assessment (LCIA), which uses specific methodological models to evaluate the potential environmental impacts based on the inventory data, and (4) interpretation of the results to support informed decision-making [49]. Various methodologies are available for LCIA, and the selection depends on the study's objectives and the regional context being analyzed. These methodologies assess environmental damage across multiple impact categories, although a significant limitation is that not all categories are addressed in every method (e.g., methods such as ReCiPe 2016, Environmental Footprint, CML-IA, TRACI, IMPACT World+, etc.). Furthermore, while conversion factors between methodologies exist [50], they often fail to enhance the interpretability of the results, potentially limiting the ability to derive meaningful conclusions from the assessment.

LCA has been increasingly applied in the mining sector to assess the environmental performance of extraction activities [51], and its use has extended to mineral processing, where impacts from energy use, chemical reagents, and emissions are commonly evaluated [52–58]. However, as highlighted by Farjana, Huda [59], many of these studies have overlooked toxicity-related impact categories, particularly those concerning human health and ecosystem quality—despite the fact that such impacts often originate from emissions associated with tailings storage facilities. In response to this gap, recent research has turned attention to the environmental implications of mine tailings, which represent a critical source of leachate emissions and long-term ecological risk. Within this context, Doka [60] was a pioneer in modeling emissions from sulfidic tailings, proposing a generic yet structured approach that laid the foundation for inventory-based methods in LCA. His work enabled the development of estimated emission inventories that are now incorporated into widely used databases such as Ecoinvent. Building on this foundation, Adrianto, Pfister [61] developed a more detailed and site-specific method based on reactive transport modeling, offering a more mechanistic representation of tailings behavior. In parallel, LCA has also been applied to assess emerging construction materials—such as alkali-activated binders [62,63] and recycled aggregates [64]—as sustainable alternatives to conventional cement. Nevertheless, there remains a notable lack of LCA studies focused specifically on the environmental impacts of materials used for underground backfilling, such as cemented paste backfill (CPB), despite their widespread application in modern mining operations.

Although LCA is a widely recognized and robust tool for evaluating the environmental impacts of materials and processes [65], several methodological and practical challenges persist. Key areas under continued discussion and refinement include the selection of the most suitable LCIA methodology, the quality and accessibility of life cycle inventory databases [66], and the allocation approaches required for systems involving co-products [65]. These factors can introduce variability and uncertainty into the results, significantly affecting the reliability and interpretability of the assessment.

Despite these limitations, LCA remains highly effective when applied to specific industrial contexts. For instance, it provides a practical framework for examining the environmental implications of alternative materials or processes within a defined scope. The present study leverages this strength to evaluate the environmental impacts associated with the production and use of binders/activators through a comprehensive LCA. Specifically, the research focuses on the alkaline activator industry (i.e., magnesium oxide, sodium hydroxide, and sodium silicate) as sustainable alternatives to Portland cement in CPB generation. By analyzing both global trends and a regional application involving a CPB plant that processes MTs from a massive sulfide deposit in southwestern Spain, this research aims to demonstrate the applicability and potential benefits of these alternatives in real-world mining operations.

2. Materials and Methods

This study follows a multi-step methodology. First, cemented paste backfill formulations were compiled from previous works, from which binder dosages and uniaxial compressive strength (UCS) data were obtained. Second, a literature review of life cycle assessment (LCA) studies on Ordinary Portland Cement (OPC), versus other alkali binders was conducted. Third, at the regional level, the LCA of PC8 (an $MgO/MgCO_3$ residue from Magnesitas Navarra, Navarra, Spain) was modeled, since no inventory data for this material exist in the literature. Finally, the environmental performance of cemented paste backfill was assessed by applying the Environmental Footprint (EF) method and the Environmental Prices method.

2.1. Experimental Data Source and CPB Sample Preparation

Cemented paste backfill (CPB) samples were prepared using three different binders: Ordinary Portland Cement (OPC), sodium hydroxide (NaOH), and an industrial MgO-residue known as PC8, generated from the calcination of magnesite. The PC8 material, characterized by a high magnesium oxide content (~47%) and significant carbonate phases (~40% magnesite and ~7% dolomite) [67], was used as received from the supplier without additional processing.

Different binder dosages were tested for each binder type, and the resulting specimens were subjected to uniaxial compressive strength (UCS) testing after curing. All experimental procedures, including mixing design, curing conditions, and UCS results, are fully described by Diosdado-Aragón, Valenzuela-Díaz [45,67].

A summary of the UCS tests conducted for the different binder types and dosages is presented in Table 1. More detailed information regarding the mix formulations, curing times and temperatures, and number of replicates can be found in the Supplementary Material (Table S1).

Table 1. Summary of UCS results for CPB samples with different binders.

Alk Agent	UCS [MPa]	SD [MPa]	N° Test
NaOH	0.33	0.34	12
PC8	1.55	0.33	20
OPC	2.92	1.49	38

2.2. Life Cycle Impact Assessment of Alkaline Activators

Thirty-three research papers were selected with the specific aim of retrieving reported environmental impact category data from LCA studies on the production of magnesium oxide, sodium hydroxide, sodium silicate, and OPC (Table 2). While these studies cover different years, processes, and regional contexts, this heterogeneity is precisely what allows the construction of distributions that capture the variability of reported results. Rather than being treated as a bias, this variability provides a broader and more representative basis for comparing the environmental performance of different binders.

Twelve environmental impact categories were selected based on the availability of data in the reviewed literature: Acidification (AC), Resource Use—Minerals and Metals (ABD-M), Resource Use—Fossils (ABD-E), Climate Change (GWP), Eutrophication (EU), Freshwater Ecotoxicity (ECOT-FW), Marine Ecotoxicity (ECOT-M), Terrestrial Ecotoxicity (ECOT-T), Human Toxicity (HT), Ozone Depletion (ODP), Photochemical Ozone Formation (POF), and Water Use (WD). Other categories—such as Land Use, Particulate Matter, and Ionizing Radiation—were excluded due to insufficient data coverage. To facilitate comparison across methods, conversion factors between impact categories were applied as proposed by Owsianiak, Laurent [50]. For additional details on data processing, please refer to the supplementary information (see Table S2 for the units of measurement for each selected environmental impact category).

Table 2. References consulted for the Life Cycle Impact Assessment (LCIA) of the alkaline activators.

Binder	Reference
MgO	[68–76]
OPC	[47,71,74,77–85]
Na_2SiO_3	[63,86–91]
NaOH	[77,85–88,90–98]

For magnesium oxide, only reactive calcined caustic magnesia (CCM) was considered in this study (as no LCA studies addressing PC8 were found in the literature). This form of magnesium oxide requires a lower calcination level (800–1000 °C) compared to other more energy-intensive forms such as sintered or fused magnesia. The former is typically used as an alkaline reagent and as a source of Mg while the latter is commonly utilized in the refractory industry. An and Xue [68] and Zhao, Feng [76] calculated the carbon footprints for a range of magnesia products, including sintered and fused magnesia, highlighting the energy intensity and environmental implications associated with these materials.

The environmental footprint was assessed using a reference unit of one kilogram of material. Sodium silicate was included as a comparative agent to sodium hydroxide, enabling an evaluation of the environmental performance of both alkaline activators. Conversion factors were applied to standardize values into consistent units [50], and each environmental impact category was subsequently normalized relative to its maximum value. This normalization process aims to simplify the data and should not be confused with the optional normalization procedure outlined in the Environmental Footprint (EF 3.1) methodology, which normalizes based on global or regional datasets [99].

To statistically evaluate the differences between the populations, the non-parametric Kruskal–Wallis test was applied (alternative to ANOVA) at a 95% confidence level, as the data did not meet the assumption of normality, determined by the one-sample Kolmogorov–Smirnov test. Where significant differences were found, a post hoc analysis was performed using the Bonferroni correction method to control for type I errors. This multicomparison procedure allowed for grouping of the populations based on their similarity. Populations with similar median values were assigned the same group, providing insight into the clusters of environmental impacts across the distinct categories.

2.3. Regional-Scale Study Background

A comparative LCIA of different alkaline activators with potential use in mine gallery backfilling offers a general understanding of their applicability on a global scale. However, a more specific and regional model is required to better assess their performance under real-world conditions. In this regard, the present study examines the applicability of an MgO-based activator—specifically, a calcined magnesite residue commercially referred to as PC8—in the regional context of the Iberian Pyrite Belt (southwestern Spain).

As previously mentioned, OPC is the binder used for mine gallery backfilling in most mining operations around the World [100]. On this regard, Diosdado-Aragón, Valenzuela-Díaz [45] showed that various formulations and curing conditions, using a mixing ratio of 73 wt% mine tailings, 21 wt% water, and 6 wt% OPC, achieved uniaxial compressive strength (UCS) values exceeding 0.5 MPa. This strength is sufficient to ensure the self-supporting behavior of the pastes within the underground galleries. However, in pursuit of improved environmental performance and cost reductions, alternative formulations for CPB have been proposed. These alternatives include magnesium oxide (MgO) and sodium hydroxide (NaOH). Diosdado-Aragón, Valenzuela-Díaz [67] also demonstrated that several types of MTs, exhibiting an ample range of acidities or alkalinities (based on Acid–Base Accounting tests) achieved the desired UCS threshold value of 0.5 MPa using even less than 1.2 wt% of MgO. In their experiments, PC8 (provided by Magnesitas Navarra, Spain) was used as an MgO source. This alkaline reagent is a by-product obtained during the generation of MgO; as a result, it holds a significant amount of the original $MgCO_3$ accompanying the MgO in its final composition. This makes MgO a viable alternative as a cementing agent for mine gallery backfilling. In contrast, NaOH-based CPBs exhibited uniaxial compressive strength values that render them unsuitable for use in underground mine backfilling (UCS < 0.5 MPa, Table 1). Nevertheless, this alkaline reagent will be included in our study for comparison with MgO and OPC and in the event of future studies that could include it as a suitable option for mine gallery backfilling.

OPC and sodium hydroxide (NaOH) are well-documented materials with extensive datasets available in Ecoinvent. Sodium silicate was excluded from the regional-scale study because no cemented paste backfill formulations using this reagent were tested [67].

No Ecoinvent dataset exists for an MgO-based activator comparable to PC8—the calcined magnesite residue used in that study. Therefore, the magnesite calcination process was modeled to obtain a more accurate LCI of the specific binder used. This new inventory was then evaluated using the Environmental Footprint 3.1 method within SimaPro 9.6 (PRé Sustainability, Amersfoort, Netherlands) to estimate the environmental impacts associated with PC8 (Supplementary File S1).

Due to the confidentiality of industrial processes and databases, it is important to emphasize that the process schemes generated for this publication must be considered as representative simplified versions of the real processes used by mining companies processing carbonate or massive sulfide deposits in Spain. In the same line of reasoning, the numbers used to model these processes must be understood as realistic values within the range of the real operational values but never as precise real values.

2.4. PC8 Environmental Footprint Modeling

As previously mentioned, PC8 comprises an almost equal proportion of uncalcined magnesite ($MgCO_3$) and periclase (MgO). This mineralogical composition is due to its origin as a by-product obtained from the dust collection systems of kilns (Figure S1, Supplementary Material). As a by-product, it lacks a specifically documented environmental footprint in Ecoinvent. Therefore, it was necessary to generate this information during the present study. As a starting point, the templates from Muñoz, Soto [101] were used and some adjustments tailored to the European context were included. These authors decompose the CCM production process into two key stages: mining and calcination (Figure S1). To recognize the transition of PC8 from being classified as waste to being treated as a co-product, the allocation of CO_2 emissions was adjusted to account for the inclusion of co-products from the process. This adjustment reduced PC8's associated emission footprint. Further details on this model and the assumptions made can be found in the Supplementary Material (Section S2). It is important to note that this model is a conceptual approximation and does not necessarily reflect the actual production process of Magnesitas Navarra. All values entered into SimaPro 9.6 can be found in Supplementary File S1 in the Supplementary Material. Supplementary File S1 includes the model used, the Muñoz, Soto [101] template for mining and calcination, and the Ecoinvent parameters for magnesium oxide (MgO) in both the RER (Rest of Europe) and RoW (Rest of World) regions.

2.5. Regional-Scale CPB Environmental Footprint Modeling

The functional unit, for comparison, is one cubic meter of CPB. Paste mixtures (between the different MTs and activators) were adjusted, in accordance, to 1 m^3 [45]. The system boundary includes emissions associated with the extraction of raw materials for the activators but excludes the environmental footprint of tailings generation, as tailings are considered waste materials and are not part of the primary production process (Figure S2). The energy consumption primarily corresponds to the use of pumps for transporting the paste. Additionally, the use of a superplasticizer was included, as it is typically added when pumping paste vertically. Further details on this model and the assumptions made can be found in the Supplementary Material (Section S3). All values entered into SimaPro 9.6 can be found in Supplementary File S1 in the Supplementary Material.

3. Results and Discussion

3.1. Review of Life Cycle Assessment Studies for MgO, OPC, Na_2SiO_3, and NaOH Activator

The bibliographic review enabled the compilation of environmental impact assessment values for four different activators (Supplementary File S2). The distributions of the normalized LCIA data, using the 12 selected environmental categories, are presented in

Table 2. Boxplots were created for each impact category to visually examine the distribution of the data (Figure 1). The numbers shown on top of each boxplot indicate the number of observations, while the letters under the boxplots correspond to population categorizations according to the non-parametric Kruskal–Wallis test applied. The values obtained using the Ecoinvent database through SimaPro 9.6 for the RER (Rest of Europe) geographic region have been included for comparison (star-shaped markers). The analysis of the data revealed that most categories (i.e., AC, ABM-M, EU, ECOT-FW, ECOT-T, HT, ODP, POF, and WD) do not exhibit significant variations among their populations (i.e., all of them are categorized as 'a'). In contrast, global warming potential (GWP) and energy footprint (ABD-E) are the only two environmental categories where two statistically different populations were identified (Figure 1).

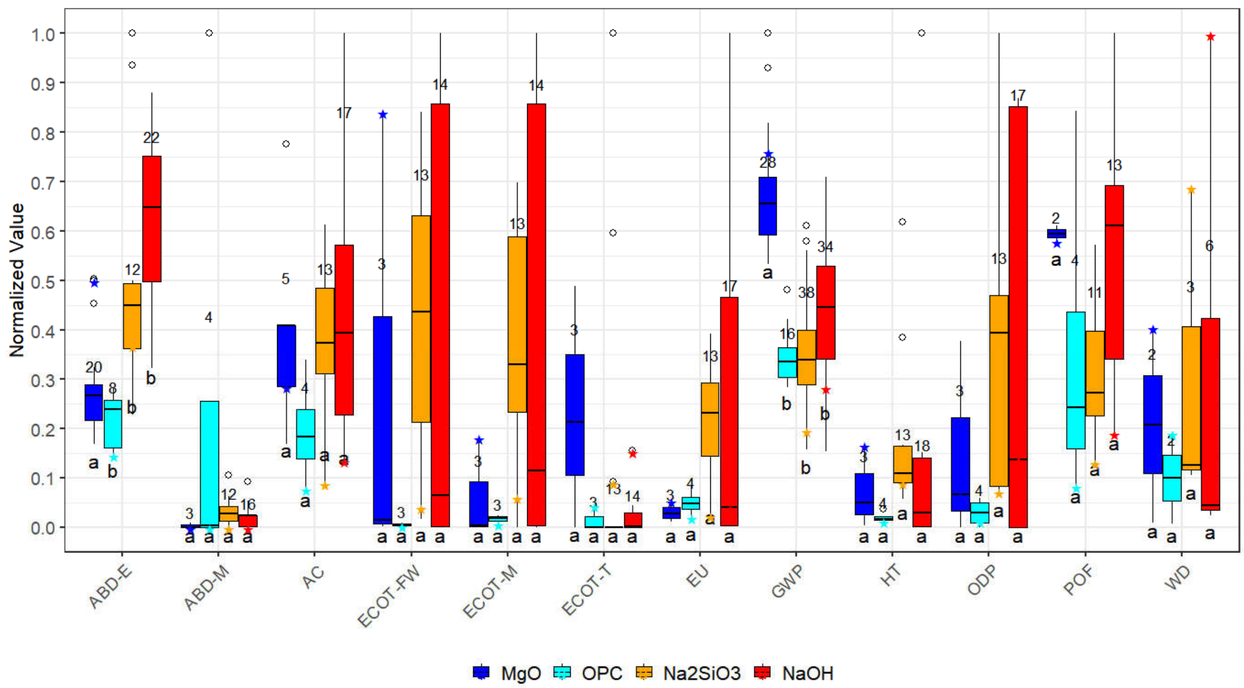

Figure 1. Visualization of distribution of normalized LCIA values using boxplot diagrams. AC: Acidification [SO_4^{2-} eq]; ABD-M: Abiotic Depletion Metals [Sb eq]; ABD-E: Abiotic Depletion Energy [MJ eq]; GWP: Global Warming Potential [CO_2 eq]; EU: Eutrophication [PO_4^{3-} eq]; ECOT-FW: Fresh Water Ecotoxicity [1.4 DB eq]; ECOT-M: Marine Ecotoxicity [1.4 DB eq]; ECOT-T: Terrestrial Ecotoxicity [1.4 DB eq]; HT: Human Toxicity [1.4 DB eq]; ODP: Ozone Depletion Potential [CFC-11 eq]; POF: Photochemical Ozone Formation [C_2H_4 eq]; WD: Water Depletion [m^3]. The stars represent the values obtained using the Ecoinvent database through SimaPro 9.6 for the RER (Rest of Europe) geographic region.

As shown in Figure 1, the MgO carbon footprint (GWP as CO_2 eq) generates a population of data (b) that is statistically independent of the other three populations of data (a). As a result, the MgO CO_2 footprint is substantially higher compared to all other alkaline reagents and OPC. Non-normalized data are shown in Spreadsheets B with ranges including 1.4 to 2.7 kg of CO_2/kg, 0.76–1.3 kg of CO_2/kg 0.42–1.65 kg of CO_2/kg and 0.4–1.9 kg of CO_2/kg for MgO, OPC, sodium silicate, and NaOH, respectively. The variations in CO_2-equivalent values are primarily attributed to differences in production methods (wet or dry processes) and formats (pure or diluted). All these values are in good agreement with the default values in Ecoinvent (stars in Figure 1).

Regarding the ABD-E category, it is directly linked to the energy consumed to produce one kilogram of activator. Since the calcination processes for calcite and magnesite are similar, Portland cement and CCM exhibit comparable energy footprints, ranging from 3.6 to 7 MJ/kg and 4.1 to 12 MJ/kg, respectively (Figure 1 and Supplementary File S2). In contrast, sodium hydroxide (NaOH), primarily produced via electrolysis, is significantly more energy-intensive, with a range of 7.9–21.6 MJ/kg. Meanwhile, sodium silicate shows a broader variation, fluctuating between 5.6 and 24.6 MJ/kg.

Considering all other environmental categories but WD and ECOT-T, it is interesting to observe that the default values in the Ecoinvent database (through SimaPro 9.6 for the rest of Europe geographic region) systematically underestimate the footprint of all studied reagents. As a direct consequence, using these default values may imply a much favorable LCIA than using the values compiled in the bibliographic search of the present research.

3.2. Activators LCIA Using Commercial Software (SimaPro) and Database (Ecoinvent)

The PC8 production flowsheet was modeled (Section 2.4) considering PC8 as a co-product of Caustic Calcined Magnesia (CCM) production to evaluate its environmental footprint and compare it with other binders (Figure 2). The assessment in SimaPro 9.6 was conducted using the Environmental Footprint 3.1 methodology, which is widely used due to its comprehensive documentation and its ability to effectively summarize data through the normalization and weighting stages of various environmental impact categories. This study complements the analysis in Section 3.1 and demonstrates that PC8 modeling at a regional scale provides a more accurate estimation, resulting in a lower environmental footprint compared to the default values in the Ecoinvent database. In particular, it was observed that the energy consumption value for MgO in the Ecoinvent database was significantly higher than the average energy values found in the literature review (Figure 1). In Ecoinvent, the energy consumption for MgO was recorded as 9 MJ, whereas the modeling of PC8 used a revised value of 4.8 MJ (similar to that reported by [76] for the same product quality of CCM, specifically LC80–85), aligning with the literature review findings and ensuring a more accurate representation of energy use.

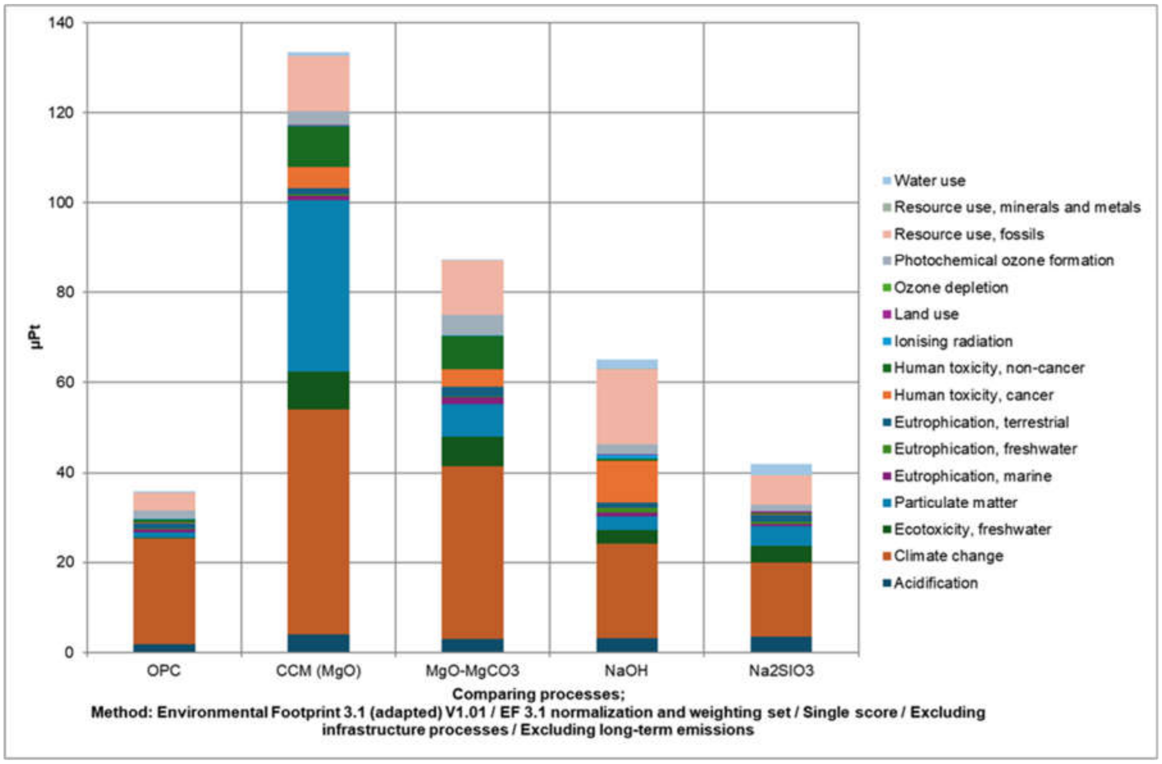

Figure 2. Comparison of the environmental performance of binders/activators in SimaPro 9.6, where a kilogram of material was used as the functional unit. Aggregated impact assessment result in Environmental Footprint Points (Pt), obtained through normalization and weighting of 16 impact categories.

In Figures 2 and 3, the environmental footprint, obtained through a normalization and weighting process, is expressed in score units (Pt, Environmental Footprint Points). These points represent an aggregated measure of environmental impacts and do not have a specific physical dimension, but they are highly useful for comparative analysis. As shown in Figure 2, OPC appears as the binder with the lowest environmental footprint, followed by Na_2SiO_3, NaOH, PC8, and MgO. Another noticeable aspect is that, from the sixteen environmental impact categories defined in the model, "Climate Change," and "Fossil Energy Use" are the most relevant for all binders/activators. This observation is aligned with the results shown in the previous section. In addition, "Particulate Matter," is one of the most relevant for MgO and it is significantly present in all other binders/activators. Other environmental impact categories like "Ecotoxicity, Freshwater", "Human toxicity, Cancer", and "Human toxicity, non-Cancer" can be relevant for some binders/activators. Another key observation is that modeling PC8 as a co-product leads to a noticeable reduction in its carbon footprint (i.e., "Climate Change"), and a very important reduction in "Particulate Matter" (Figure 2). This behavior can be attributed to the distribution of the footprints among all co-products, providing a more realistic assessment of the environmental impact, and to the fact that PC8 consists of dust emissions captured during the calcination process, effectively transforming emissions into a valuable material.

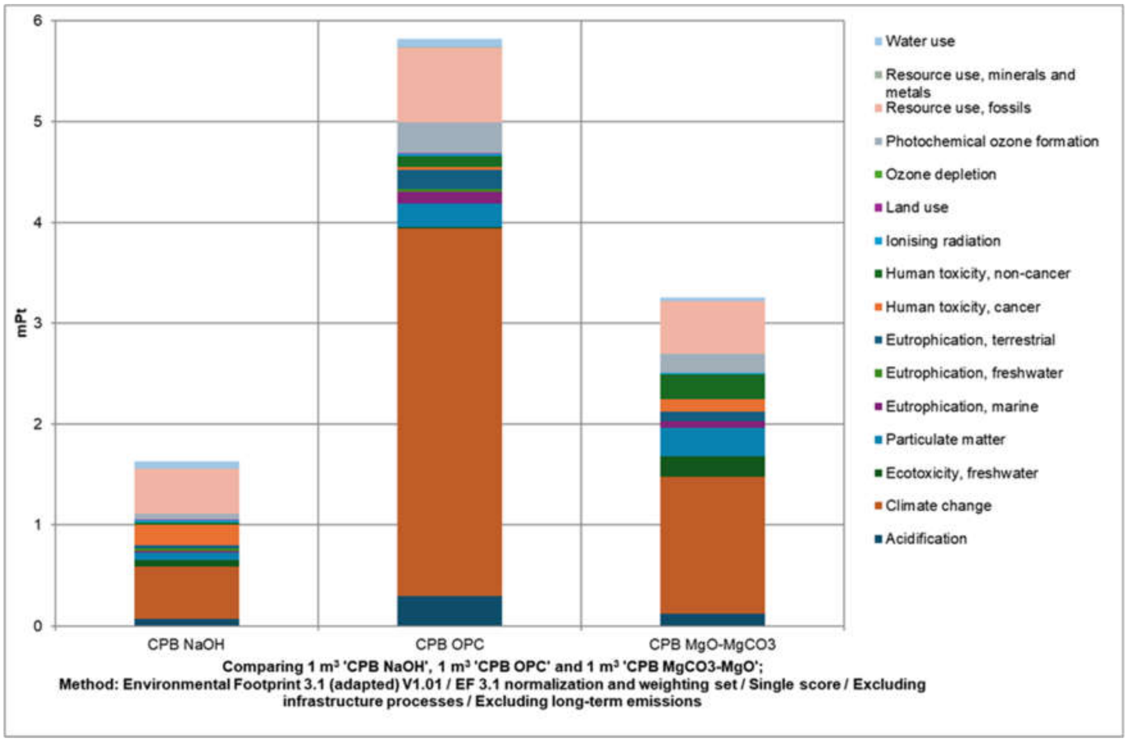

Figure 3. Comparison of the Environmental Performance of the Three CPB Formulations. Aggregated impact assessment result in Environmental Footprint Points (Pt), obtained through normalization and weighting of 16 impact categories.

Once the comparative differences in the environmental footprint of the binding agents are understood—and considering that the prices per kilogram of the binders/activators are approximately 50, 46, and 500 USD/t for OPC [102], PC8 (personal communication), and NaOH [103], respectively—one might conclude that OPC represents the best option for generating cemented pastes for underground mine backfilling on a global scale. However, this might not necessarily be the case for a regional-scale application.

3.3. Cemented Paste Backfill LCA and Economic Assessment (Regional Scale)

Taking into consideration the regional context explained in Section 2.2 and the optimal paste formulations reported by [45,67], it was decided to obtain an LCA and economic assessment of a regional application of cemented paste backfill at the Iberian Pyrite Belt (SW Spain). As a result, the environmental performance of the three CPB formulations (including 0.84 wt% NaOH, 1.22 wt% PC8, and 6wt% OPC) was evaluated. As can be observed in Figure 3, and in line with all previous results, the most significant environmental impact category is "Climate change", representing 31%, 63%, and 41% of the accumulated environmental impacts for CPB NaOH, CPB OPC, and CPB PC8, respectively. The second most relevant category is "Resource use: fossil", representing 27%, 13%, and 16% for CPB NaOH, CPB OPC, and CPB PC8, respectively. On a much lower level of relevance, "Acidification" and "Particulate matter" can also be highlighted, both adding up to 5% in all three formulations, except for "Particulate matter" in PC8, which accounts for 9%.

As it was described in the methodological section, transportation of binders/activators and the electrical energy used in the plant to pump the paste from the processing facility to the mine were considered in the model. For a detailed breakdown of this environmental assessment, refer to Table S3 in the Supplementary Material and Supplementary File S1, which includes the parameters entered in SimaPro 9.6.

To grasp a better understanding of the economy behind the different CPB formulations (on a regional scale), it was decided to perform a realistic economic assessment based on the possible implementation of these technologies at a mine in SW Spain. Table 3 includes a summary of CO_2 emissions and an economic assessment, considering the unit cost per cubic meter. The cost of the binders/activators incorporate energy costs and transportation costs. The environmental cost was obtained using the Environmental Prices model [104]. This model offers an initial approach to assess the environmental costs associated with the preparation of different types of CPBs. It translates environmental impacts into monetary values, allowing for a more intuitive comparison of the economic burden associated with emissions and resource consumption throughout the life cycle of each material. By assigning a financial cost to environmental damage, this approach facilitates decision-making in the context of sustainable mining and materials engineering. The model is based on studies estimating the willingness to pay for environmental preservation or the costs incurred due to ecological degradation [104]. These values are then applied to emissions and resource flows, enabling a comprehensive assessment of their overall impact. It is important to note that the model is based on Dutch cost estimates in euros from 2015. To adjust these values to 2024, an inflation adjustment was applied using data from the Dutch central bank, resulting in a conversion factor of 1.32. The environmental prices obtained by the model for 14 environmental impact categories are listed in Table S4 in the Supplementary Material. The environmental prices of "Particulate matter formation" and "Climate change" stand out among the environmental prices for all other environmental categories. Specifically, "Particulate Matter formation" represents 41%, 32%, and 36% of the environmental cost for CPBs based on NaOH, PC8, and OPC, respectively (Table S4), whereas "Climate change" accounts for 30%, 32%, and 45%of the environmental cost for CPBs based on NaOH, PC8, and OPC, respectively (Table S4).

Table 3. Summary of costs, life cycle costing, CO_2 emissions, compressive strength, and performance factor.

Item\Paste	NaOH	$MgCO_3$-MgO	OPC
Cost [€/m^3]	13.83	5.01	16.61
Binder (from purchase price)	10.71	1.43	7.48
Transport (1000 km)	1.00	1.46	7.01
Energy (from paste plant)	2.12	2.12	2.12
Environmental cost [€/m^3]	5.04	9.05	17.90
Final cost [€/m^3]	18.87	14.06	34.5
CO_2 emissions [kg CO_2-eq/m^3]	18.40	48.50	131
UCS [MPa]	0.33 ± 0.34	1.55 ± 0.33	2.92 ± 1.49
PF [kg CO_2eq/m^3/MPa]	55.80	31.29	44.9

Table 3 presents the cost breakdown per cubic meter, which includes binder, transport, and energy expenses. A transport distance of 1000 km was assumed, using 32 ton trucks, together with an energy consumption of 9.93 kWh/m^3 at the paste plant. It also includes the environmental cost derived from the Environmental Prices model; CO_2 emissions based on the Climate Change category of the EF3.1 model; and the final cost, which is the sum of the cost and environmental cost. Additionally, it provides the uniaxial compressive strength (UCS) and the Performance Factor (PF), a metric used to evaluate the efficiency of a binder in achieving mechanical performance relative to its environmental impact. The CO_2/m^3MPa ratio in cemented pastes serves as a key indicator for assessing the environmental efficiency of different binders. A lower ratio implies that a material attains a higher strength with lower emissions, making it a more sustainable option. This metric is essential for comparing binders in terms of carbon efficiency, optimizing formulations to balance mechanical performance and environmental impact, guiding material selection for underground backfilling, and evaluating trade-offs between emissions and structural performance in mining applications.

In summary, three distinct scenarios emerge from the model. The first scenario, CPB NaOH, has a low carbon footprint and low environmental cost, but a high unit cost and poor performance, as its uniaxial compressive strength (UCS) does not exceed 0.5 MPa, making it an unviable option. The second scenario, CPB OPC, exhibits a high carbon footprint and high environmental cost, a moderate unit cost, and excellent compressive strength. Finally, the CPB PC8 scenario shows a low carbon footprint and low environmental cost, a low unit cost, and a sufficient UCS to be used as an activator for underground mine backfill paste, as confirmed by Diosdado-Aragón, Valenzuela-Díaz [67].

Considering these factors, CPB PC8 emerges as the most favorable option for this application, offering a balance between environmental impact, cost-effectiveness, and mechanical performance. It is important to emphasize that long-term durability and operational cost are not fully included in the present model. Therefore, it should be understood as a solid first approximation with regional economic and environmental relevance that allows strategic decision-making. Nonetheless, the practical adoption of PC8 would likely be feasible only at a local scale or in the vicinity of calcination plants, since the main challenge lies in ensuring sufficient material availability to meet the large volume demands of mining operations.

4. Conclusions

The findings of this research suggest that the environmental footprint, particularly the CO_2 footprint, associated with producing one cubic meter of cemented paste can be reduced by using MgO as an additive instead of Portland cement. Through a comprehensive life cycle assessment (LCA), the use of MgO-based paste has demonstrated a lower carbon footprint compared to traditional OPC, while also providing economic advantages.

Regional variations in production processes and resource availability can significantly influence environmental outcomes. The research exemplifies these issues by assessing the environmental impacts of mining operations using regional data for the production of alternative alkaline reagents (i.e., MgO) 1000 km from the mining operation where it would be used to generate MTs pastes to be injected during mine gallery backfilling. Results analysis also reveal the need to address current information gaps, particularly regarding the environmental impacts of alternative materials, to facilitate more informed decision-making.

It was demonstrated that the by-products from MgO production, if properly managed and valorized, could further enhance the sustainability of mining operations by contributing to circular economy principles.

This research encourages future studies to focus on overcoming these information gaps and integrating co-products into environmental assessments to provide a more comprehensive understanding of the sustainability of alkaline activators in mining applications. In parallel, it is essential to share these environmental footprint improvements with the general public, if OPC replacement by other alkaline reagents with lower environmental footprints is finally implemented. Actively promoting environmental improvements in mining operations is paramount to consolidate and maintain a social license to operate.

Supplementary Materials: The following supporting information can be downloaded at https://www.mdpi.com/article/10.3390/app15189996/s1, Figure S1: Flowsheet MgO products. Modified from [105]; Figure S2: System boundaries. The numbers (1, 2, 3) represent the different cemented paste backfill (CPB) scenarios; Table S1: Test conditions for uniaxial compressive strength (UCS) measurements; Table S2: Abbreviation and units; Table S3: Damage assessment. Input for Environmental Prices model; Table S4: Environmental prices, adjusted for Dutch inflation from 2015 to 2024. Supplementary File S1: Template input file for SimaPro assessment; Supplementary File S2: Bibliographic review of environmental impact categories. References [50,69,101,104–110] are cited in the supplementary materials.

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Data Availability Statement: All data supporting the findings of this study are provided in the Supplementary Materials. The SimaPro project file is therefore not redistributed; however, we provide exported inventories and full model configuration that is sufficient to reproduce the results. Further inquiries can be directed to the corresponding author.

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References

1. OECD. *Global Material Resources Outlook to 2060*; OECD: Paris, France, 2019. https://doi.org/10.1787/9789264307452-en.
2. Upadhyay, A.; Laing, T.; Kumar, V.; Dora, M. Exploring barriers and drivers to the implementation of circular economy practices in the mining industry. *Resour. Policy* **2021**, *72*, 102037. https://doi.org/10.1016/j.resourpol.2021.102037.
3. Zhou, Y. Natural resources and green economic growth: A pathway to innovation and digital transformation in the mining industry. *Resour. Policy* **2024**, *90*, 104667. https://doi.org/10.1016/j.resourpol.2024.104667.
4. Binnemans, K.; Jones, P.T. The Twelve Principles of Circular Hydrometallurgy. *J. Sustain. Metall.* **2023**, *9*, 1–25. https://doi.org/10.1007/s40831-022-00636-3.
5. Kinnunen, P.; Karhu, M.; Yli-Rantala, E.; Kivikytö-Reponen, P.; Mäkinen, J. A review of circular economy strategies for mine tailings. *Cleaner Eng. Technol.* **2022**, *8*, 100499. https://doi.org/10.1016/j.clet.2022.100499.

6. Baker, E.; Davies, M.; Fourie, A.; Mudd, G.; Thygesen, K. Chapter II: Mine Tailings Facilities: Overview and Industry Trends. In *Towards Zero Harm—A Compendium of Papers Prepared for the Global Tailings Review*; Global Tailings Review: London, UK, 2019.
7. Franks, D.M.; Stringer, M.; Torres-Cruz, L.A.; Baker, E.; Valenta, R.; Thygesen, K.; Matthews, A.; Howchin, J.; Barrie, S. Tailings facility disclosures reveal stability risks. *Sci. Rep.* **2021**, *11*, 5353. https://doi.org/10.1038/s41598-021-84897-0.
8. Falagán, C.; Grail, B.M.; Johnson, D.B. New approaches for extracting and recovering metals from mine tailings. *Miner. Eng.* **2017**, *106*, 71–78. https://doi.org/10.1016/j.mineng.2016.10.008.
9. Di Maria, A.; Khoshkhoo, M.; Sand, A.; Van Acker, K. Towards sustainable resource valorization: A life cycle sustainability assessment of metals recovery from sulfidic mining residues in Sweden. *Resour. Conserv. Recycl.* **2024**, *204*, 107513. https://doi.org/10.1016/j.resconrec.2024.107513.
10. Rosario-Beltré, A.J.; Sánchez-España, J.; Rodríguez-Gómez, V.; Fernández-Naranjo, F.J.; Bellido-Martín, E.; Adánez-Sanjuán, P.; Arranz-González, J.C. Critical Raw Materials recovery potential from Spanish mine wastes: A national-scale preliminary assessment. *J. Clean. Prod.* **2023**, *407*, 137163. https://doi.org/10.1016/j.jclepro.2023.137163.
11. Adrianto, L.R.; Ciacci, L.; Pfister, S.; Hellweg, S. Toward sustainable reprocessing and valorization of sulfidic copper tailings: Scenarios and prospective LCA. *Sci. Total Environ.* **2023**, *871*, 162038. https://doi.org/10.1016/j.scitotenv.2023.162038.
12. Sarker, S.K.; Haque, N.; Bhuiyan, M.; Bruckard, W.; Pramanik, B.K. Recovery of strategically important critical minerals from mine tailings. *J. Environ. Chem. Eng.* **2022**, *10*, 107622. https://doi.org/10.1016/j.jece.2022.107622.
13. Gomes, H.I.; Mayes, W.M.; Rogerson, M.; Stewart, D.I.; Burked, I.T. Alkaline residues and the environment: A review of impacts, management practices and opportunities. *J. Clean. Prod.* **2016**, *112*, 3571–3582. https://doi.org/10.1016/j.jclepro.2015.09.111.
14. Capasso, I.; Lirer, S.; Flora, A.; Ferone, C.; Cioffi, R.; Caputo, D.; Liguori, B. Reuse of mining waste as aggregates in fly ash-based geopolymers. *J. Clean. Prod.* **2019**, *220*, 65–73. https://doi.org/10.1016/j.jclepro.2019.02.164.
15. Kiventerä, J.; Golek, L.; Yliniemi, J.; Ferreira, V.; Deja, J.; Illikainen, M. Utilization of sulphidic tailings from gold mine as a raw material in geopolymerization. *Int. J. Miner. Process.* **2016**, *149*, 104–110. https://doi.org/10.1016/j.minpro.2016.02.012.
16. He, X.; Yuhua, Z.; Qaidi, S.; Isleem, H.F.; Zaid, O.; Althoey, F.; Ahmad, J. Mine tailings-based geopolymers: A comprehensive review. *Ceram. Int.* **2022**, *48*, 24192–24212. https://doi.org/10.1016/j.ceramint.2022.05.345.
17. Lazorenko, G.; Kasprzhitskii, A.; Shaikh, F.; Krishna, R.S.; Mishra, J. Utilization potential of mine tailings in geopolymers: Physicochemical and environmental aspects. *Process Saf. Environ. Prot.* **2021**, *147*, 559–577. https://doi.org/10.1016/j.psep.2020.12.028.
18. Thejas, H.K.; Hossiney, N. Alkali-activated bricks made with mining waste iron ore tailings. *Case Stud. Constr. Mater.* **2022**, *16*, e00973. https://doi.org/10.1016/j.cscm.2022.e00973.
19. Veiga Simão, F.; Chambart, H.; Vandemeulebroeke, L.; Nielsen, P.; Adrianto, L.R.; Pfister, S.; Cappuyns, V. Mine waste as a sustainable resource for facing bricks. *J. Clean. Prod.* **2022**, *368*, 133118. https://doi.org/10.1016/j.jclepro.2022.133118.
20. Ahmari, S.; Zhang, L. Production of eco-friendly bricks from copper mine tailings through geopolymerization. *Constr. Build. Mater.* **2012**, *29*, 323–331. https://doi.org/10.1016/j.conbuildmat.2011.10.048.
21. Mabroum, S.; Moukannaa, S.; El Machi, A.; Taha, Y.; Benzaazoua, M.; Hakkou, R. Mine wastes based geopolymers: A critical review. *Clean. Eng. Technol.* **2020**, *1*, 100014. https://doi.org/10.1016/j.clet.2020.100014.
22. Veiga Simão, F.; Chambart, H.; Vandemeulebroeke, L.; Cappuyns, V. Incorporation of sulphidic mining waste material in ceramic roof tiles and blocks. *J. Geochem. Explor.* **2021**, *225*, 106741. https://doi.org/10.1016/j.gexplo.2021.106741.
23. Lemougna, P.N.; Yliniemi, J.; Nguyen, H.; Adesanya, E.; Tanskanen, P.; Kinnunen, P.; Roning, J.; Illikainen, M. Utilisation of glass wool waste and mine tailings in high performance building ceramics. *J. Build. Eng.* **2020**, *31*, 101383. https://doi.org/10.1016/j.jobe.2020.101383.
24. Zhang, Y.; Shen, W.; Wu, M.; Shen, B.; Li, M.; Xu, G.; Zhang, B.; Ding, Q.; Chen, X. Experimental study on the utilization of copper tailing as micronized sand to prepare high performance concrete. *Constr. Build. Mater.* **2020**, *244*, 118312. https://doi.org/10.1016/j.conbuildmat.2020.118312.

25. Benahsina, A.; El Haloui, Y.; Taha, Y.; Elomari, M.; Bennouna, M.A. Natural sand substitution by copper mine waste rocks for concrete manufacturing. *J. Build. Eng.* **2022**, *47*, 103817. https://doi.org/10.1016/j.jobe.2021.103817.
26. Onuaguluchi, O.; Eren, Ö. Reusing copper tailings in concrete: Corrosion performance and socioeconomic implications for the Lefke-Xeros area of Cyprus. *J. Clean. Prod.* **2016**, *112*, 420–429. https://doi.org/10.1016/j.jclepro.2015.09.036.
27. Adiguzel, D.; Tuylu, S.; Eker, H. Utilization of tailings in concrete products: A review. *Constr. Build. Mater.* **2022**, *360*, 129574. https://doi.org/10.1016/j.conbuildmat.2022.129574.
28. Arunachalam, K.P.; Avudaiappan, S.; Maureira, N.; Da Costa Garcia Filho, F.; Monteiro, S.N.; Batista, I.D.; de Azevedo, A.R.G. Innovative use of copper mine tailing as an additive in cement mortar. *J. Mater. Res. Technol.* **2023**, *25*, 2261–2274. https://doi.org/10.1016/j.jmrt.2023.06.066.
29. Shettima, A.U.; Hussin, M.W.; Ahmad, Y.; Mirza, J. Evaluation of iron ore tailings as replacement for fine aggregate in concrete. *Constr. Build. Mater.* **2016**, *120*, 72–79. https://doi.org/10.1016/j.conbuildmat.2016.05.095.
30. Vargas, F.; Lopez, M.; Rigamonti, L. Environmental impacts evaluation of treated copper tailings as supplementary cementitious materials. *Resour. Conserv. Recycl.* **2020**, *160*, 104890. https://doi.org/10.1016/j.resconrec.2020.104890.
31. Vargas, F.; Lopez, M. Development of a new supplementary cementitious material from the activation of copper tailings: Mechanical performance and analysis of factors. *J. Clean. Prod.* **2018**, *182*, 427–436. https://doi.org/10.1016/j.jclepro.2018.01.223.
32. Araujo, F.S.M.; Taborda-Llano, I.; Nunes, E.B.; Santos, R.M. Recycling and Reuse of Mine Tailings: A Review of Advancements and Their Implications. *Geosciences* **2022**, *12*, 319. https://doi.org/10.3390/geosciences12090319.
33. Sivakugan, N.; Veenstra, R.; Naguleswaran, N. Underground Mine Backfilling in Australia Using Paste Fills and Hydraulic Fills. *Int. J. Geosynth. Ground Eng.* **2015**, *1*, 18. https://doi.org/10.1007/s40891-015-0020-8.
34. Zhang, X.; Zhao, J.; Xin, L.; Wang, K.; Pan, H. Monitoring and Assessment of Cemented Paste Backfill Containing Coal Gangue and Fly Ash in an Underground Mine. *Adv. Mater. Sci. Eng.* **2021**, *2021*, 5946148. https://doi.org/10.1155/2021/5946148.
35. Raffaldi, M.J.; Seymour, J.B.; Richardson, J.; Zahl, E.; Board, M. Cemented Paste Backfill Geomechanics at a Narrow-Vein Underhand Cut-and-Fill Mine. *Rock Mech. Rock Eng.* **2019**, *52*, 4925–4940. https://doi.org/10.1007/s00603-019-01850-4.
36. Helinski, M.; Fahey, M.; Fourie, A. Behavior of Cemented Paste Backfill in Two Mine Stopes: Measurements and Modeling. *J. Geotech. Geoenviron. Eng.* **2011**, *137*, 171–182. https://doi.org/10.1061/(asce)gt.1943-5606.0000418.
37. Qi, C.; Fourie, A. Cemented paste backfill for mineral tailings management: Review and future perspectives. *Miner. Eng.* **2019**, *144*, 106025. https://doi.org/10.1016/j.mineng.2019.106025.
38. Tariq, A.; Yanful, E.K. A review of binders used in cemented paste tailings for underground and surface disposal practices. *J. Environ. Manag.* **2013**, *131*, 138–149. https://doi.org/10.1016/j.jenvman.2013.09.039.
39. Edraki, M.; Baumgartl, T.; Manlapig, E.; Bradshaw, D.; Franks, D.M.; Moran, C.J. Designing mine tailings for better environmental, social and economic outcomes: A review of alternative approaches. *J. Clean. Prod.* **2014**, *84*, 411–420. https://doi.org/10.1016/j.jclepro.2014.04.079.
40. Belem, T.; Benzaazoua, M. Design and application of underground mine paste backfill technology. *Geotech. Geol. Eng.* **2008**, *26*, 147–174. https://doi.org/10.1007/s10706-007-9154-3.
41. Behera, S.K.; Mishra, D.P.; Ghosh, C.N.; Prashant; Mandal, P.K.; Singh, K.M.P.; Buragohain, J.; Singh, P.K. Characterization of lead–zinc mill tailings, fly ash and their mixtures for paste backfilling in underground metalliferous mines. *Environ. Earth Sci.* **2019**, *78*, 394. https://doi.org/10.1007/s12665-019-8395-9.
42. Fall, M.; Benzaazoua, M.; Saa, E.G. Mix proportioning of underground cemented tailings backfill. *Tunn. Undergr. Space Technol.* **2008**, *23*, 80–90. https://doi.org/10.1016/j.tust.2006.08.005.
43. Behera, S.K.; Mishra, D.P.; Singh, P.; Mishra, K.; Mandal, S.K.; Ghosh, C.N.; Kumar, R.; Mandal, P.K. Utilization of mill tailings, fly ash and slag as mine paste backfill material: Review and future perspective. *Constr. Build. Mater.* **2021**, *309*, 125120. https://doi.org/10.1016/j.conbuildmat.2021.125120.
44. Benzaazoua, M.; Belem, T.; Bussiè, B. Chemical factors that inf luence the performance of mine sulphidic paste backfill. *Cem. Concr. Res.* **2002**, *32*, 1133–1144. https://doi.org/10.1016/S0008-8846(02)00752-4.
45. Diosdado-Aragón, A.J.; Valenzuela-Díaz, M.J.; Dávila, J.M.; Becerra-Herrera, M.; Caraballo, M.A. Influence of mine tailings mineralogy and curing conditions in the cementation of pastes for mine galleries backfilling. *Miner. Eng.* **2025**, *232*, 109524. https://doi.org/10.1016/j.mineng.2025.109524.

46. Soomro, M.; Tam, V.W.Y.; Jorge Evangelista, A.C. Production of cement and its environmental impact. In *Recycled Concrete: Technologies and Performance*; Elsevier: Amsterdam, The Netherlands, 2022; pp. 11–46. https://doi.org/10.1016/B978-0-323-85210-4.00010-2.
47. Huntzinger, D.N.; Eatmon, T.D. A life-cycle assessment of Portland cement manufacturing: Comparing the traditional process with alternative technologies. *J. Clean. Prod.* **2009**, *17*, 668–675. https://doi.org/10.1016/j.jclepro.2008.04.007.
48. Saedi, A.; Jamshidi-Zanjani, A.; Darban, A.K. A review on different methods of activating tailings to improve their cementitious property as cemented paste and reusability. *J. Environ. Manag.* **2020**, *270*, 110881. https://doi.org/10.1016/j.jenvman.2020.110881.
49. Sala, S.; Reale, F.; Cristobal-Garcia, J.; Marelli, L.; Rana, P. *Life Cycle Assessment for the Impact Assessment of Policies*; Publications Office of the European Union: Luxembourg, 2016. https://doi.org/10.2788/318544.
50. Owsianiak, M.; Laurent, A.; Bjørn, A.; Hauschild, M.Z. IMPACT 2002+, ReCiPe 2008 and ILCD's recommended practice for characterization modelling in life cycle impact assessment: A case study-based comparison. *Int. J. Life Cycle Assess.* **2014**, *19*, 1007–1021. https://doi.org/10.1007/s11367-014-0708-3.
51. Rachid, S.; Taha, Y.; Benzaazoua, M. Environmental evaluation of metals and minerals production based on a life cycle assessment approach: A systematic review. *Miner. Eng.* **2023**, *198*, 108076. https://doi.org/10.1016/j.mineng.2023.108076.
52. Segura-Salazar, J.; Lima, F.M.; Tavares, L.M. Life Cycle Assessment in the minerals industry: Current practice, harmonization efforts, and potential improvement through the integration with process simulation. *J. Clean. Prod.* **2019**, *232*, 174–192. https://doi.org/10.1016/j.jclepro.2019.05.318.
53. Segura-Salazar, J.; Tavares, L.M. A life cycle-based, sustainability-driven innovation approach in the minerals industry: Application to a large-scale granitic quarry in Rio de Janeiro. *Miner. Eng.* **2021**, *172*, 107149. https://doi.org/10.1016/j.mineng.2021.107149.
54. Farjana, S.H.; Huda, N.; Mahmud, M.A.P. Life cycle analysis of copper-gold-lead-silver-zinc beneficiation process. *Sci. Total Environ.* **2019**, *659*, 41–52. https://doi.org/10.1016/j.scitotenv.2018.12.318.
55. Farjana, S.H.; Huda, N.; Mahmud, M.A.P. Impacts of aluminum production: A cradle to gate investigation using life-cycle assessment. *Sci. Total Environ.* **2019**, *663*, 958–970. https://doi.org/10.1016/j.scitotenv.2019.01.400.
56. Farjana, S.H.; Huda, N.; Mahmud, M.A.P.; Lang, C. A global life cycle assessment of manganese mining processes based on EcoInvent database. *Sci. Total Environ.* **2019**, *688*, 1102–1111. https://doi.org/10.1016/j.scitotenv.2019.06.184.
57. Reid, C.; Bécaert, V.; Aubertin, M.; Rosenbaum, R.K.; Deschênes, L. Life cycle assessment of mine tailings management in Canada. *J. Clean. Prod.* **2009**, *17*, 471–479. https://doi.org/10.1016/j.jclepro.2008.08.014.
58. Song, X.; Pettersen, J.B.; Pedersen, K.B.; Røberg, S. Comparative life cycle assessment of tailings management and energy scenarios for a copper ore mine: A case study in Northern Norway. *J. Clean. Prod.* **2017**, *164*, 892–904. https://doi.org/10.1016/j.jclepro.2017.07.021.
59. Farjana, S.H.; Huda, N.; Parvez Mahmud, M.A.; Saidur, R. A review on the impact of mining and mineral processing industries through life cycle assessment. *J. Clean. Prod.* **2019**, *231*, 1200–1217. https://doi.org/10.1016/j.jclepro.2019.05.264.
60. Doka, G. *A Model for Waste-Specific and Climate-Specific Life Cycle Inventories of Tailings Impoundments*; Doka Life Cycle Assessments: Zurich, Switzerland, 2002.
61. Adrianto, L.R.; Pfister, S.; Hellweg, S. Regionalized Life Cycle Inventories of Global Sulfidic Copper Tailings. *Environ. Sci. Technol.* **2022**, *56*, 4553–4564. https://doi.org/10.1021/acs.est.1c01786.
62. Dandautiya, R.; Singh, A.P. Utilization potential of fly ash and copper tailings in concrete as partial replacement of cement along with life cycle assessment. *Waste Manag.* **2019**, *99*, 90–101. https://doi.org/10.1016/j.wasman.2019.08.036.
63. 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. Potential use of iron ore tailings for binder production: A life cycle assessment. *Constr. Build. Mater.* **2023**, *365*, 130008. https://doi.org/10.1016/j.conbuildmat.2022.130008.
64. Munir, Q.; Abdulkareem, M.; Horttanainen, M.; Kärki, T. A comparative cradle-to-gate life cycle assessment of geopolymer concrete produced from industrial side streams in comparison with traditional concrete. *Sci. Total Environ.* **2023**, *865*, 161230. https://doi.org/10.1016/j.scitotenv.2022.161230.

65. Wernet, G.; Bauer, C.; Steubing, B.; Reinhard, J.; Moreno-Ruiz, E.; Weidema, B. The ecoinvent database version 3 (part I): Overview and methodology. *Int. J. Life Cycle Assess.* **2016**, *21*, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8.
66. Ghose, A. Can LCA be FAIR? Assessing the status quo and opportunities for FAIR data sharing. *Int. J. Life Cycle Assess.* **2024**, *29*, 733–744. https://doi.org/10.1007/s11367-024-02280-3.
67. Diosdado-Aragón, A.J.; Valenzuela-Díaz, M.J.; Ruíz-Rodríguez, M.; Davila, J.M.; Becerra-Herrera, M.; Caraballo, M.A. Chemometric study of mine tailings cementation, using NaOH or Mg(OH)2, for mine galleries backfilling. *Miner. Eng.* **2025**, *232*, 109518. https://doi.org/10.1016/j.mineng.2025.109518.
68. An, J.; Xue, X. Life-cycle carbon footprint analysis of magnesia products. *Resour. Conserv. Recycl.* **2017**, *119*, 4–11. https://doi.org/10.1016/j.resconrec.2016.09.023.
69. Li, J.; Zhang, Y.; Shao, S.; Zhang, S. Comparative life cycle assessment of conventional and new fused magnesia production. *J. Clean. Prod.* **2015**, *91*, 170–179. https://doi.org/10.1016/j.jclepro.2014.12.043.
70. Ren, W.; Xue, B.; Lu, C.; Zhang, Z.; Zhang, Y.; Jiang, L. Evaluation of GHG emissions from the production of magnesia refractory raw materials in Dashiqiao, China. *J. Clean. Prod.* **2016**, *135*, 214–222. https://doi.org/10.1016/j.jclepro.2016.06.118.
71. Ruan, S.; Unluer, C. Comparative life cycle assessment of reactive MgO and Portland cement production. *J. Clean. Prod.* **2016**, *137*, 258–273. https://doi.org/10.1016/j.jclepro.2016.07.071.
72. Ruan, S.; Unluer, C. Influence of supplementary cementitious materials on the performance and environmental impacts of reactive magnesia cement concrete. *J. Clean. Prod.* **2017**, *159*, 62–73. https://doi.org/10.1016/j.jclepro.2017.05.044.
73. Shahbaz, F.; Singh, I.; Krishnan, P.; Celik, K. Life cycle assessment of brucite and synthetic MgO produced from reject brine using different alkalis. *J. Clean. Prod.* **2022**, *380*, 135071. https://doi.org/10.1016/j.jclepro.2022.135071.
74. Shen, W.; Cao, L.; Li, Q.; Wen, Z.; Wang, J.; Liu, Y.; Dong, R.; Tan, Y.; Chen, R. Is magnesia cement low carbon? Life cycle carbon footprint comparing with Portland cement. *J. Clean. Prod.* **2016**, *131*, 20–27. https://doi.org/10.1016/j.jclepro.2016.05.082.
75. Trojer, M. Principles of Benchmarking Criteria for the European Magnesia Industry. Master's Thesis, University of Leoben, Leoben, Austria, 2009.
76. Zhao, L.; Feng, J.; Dong, H. Analysis of carbon footprint and reduction approach of magnesia production in China. *J. Clean. Prod.* **2022**, *334*, 130194. https://doi.org/10.1016/j.jclepro.2021.130194.
77. Arguillarena, A.; Margallo, M.; Irabien, Á.; Urtiaga, A. Life cycle assessment of zinc and iron recovery from spent pickling acids by membrane-based solvent extraction and electrowinning. *J. Environ. Manag.* **2022**, *318*, 115567. https://doi.org/10.1016/j.jenvman.2022.115567.
78. Boesch, M.E.; Hellweg, S. Identifying improvement potentials in cement production with life cycle assessment. *Environ. Sci. Technol.* **2010**, *44*, 9143–9149. https://doi.org/10.1021/es100771k.
79. Chen, C.; Habert, G.; Bouzidi, Y.; Jullien, A. Environmental impact of cement production: Detail of the different processes and cement plant variability evaluation. *J. Clean. Prod.* **2010**, *18*, 478–485. https://doi.org/10.1016/j.jclepro.2009.12.014.
80. Fernando, S.; Gunasekara, C.; Law, D.W.; Nasvi, M.C.M.; Setunge, S.; Dissanayake, R. Life cycle assessment and cost analysis of fly ash–rice husk ash blended alkali-activated concrete. *J. Environ. Manag.* **2021**, *295*, 113140. https://doi.org/10.1016/j.jenvman.2021.113140.
81. Moretti, L.; Caro, S. Critical analysis of the Life Cycle Assessment of the Italian cement industry. *J. Clean. Prod.* **2017**, *152*, 198–210. https://doi.org/10.1016/j.jclepro.2017.03.136.
82. Ramagiri, K.K.; Chintha, R.; Bandlamudi, R.K.; Kara De Maeijer, P.; Kar, A. Cradle-to-gate life cycle and economic assessment of sustainable concrete mixes—Alkali-activated concrete (Aac) and bacterial concrete (bc). *Infrastructures* **2021**, *6*, 104. https://doi.org/10.3390/infrastructures6070104.
83. Teh, S.H.; Wiedmann, T.; Castel, A.; de Burgh, J. Hybrid life cycle assessment of greenhouse gas emissions from cement, concrete and geopolymer concrete in Australia. *J. Clean. Prod.* **2017**, *152*, 312–320. https://doi.org/10.1016/j.jclepro.2017.03.122.
84. Valderrama, C.; Granados, R.; Cortina, J.L.; Gasol, C.M.; Guillem, M.; Josa, A. Implementation of best available techniques in cement manufacturing: A life-cycle assessment study. *J. Clean. Prod.* **2012**, *25*, 60–67. https://doi.org/10.1016/j.jclepro.2011.11.055.

85. Turner, L.K.; Collins, F.G. Carbon dioxide equivalent (CO2-e) emissions: A comparison between geopolymer and OPC cement concrete. *Constr. Build. Mater.* **2013**, *43*, 125–130. https://doi.org/10.1016/j.conbuildmat.2013.01.023.
86. Abdulkareem, M.; Havukainen, J.; Nuortila-Jokinen, J.; Horttanainen, M. Environmental and economic perspective of waste-derived activators on alkali-activated mortars. *J. Clean. Prod.* **2021**, *280*, 124651. https://doi.org/10.1016/j.jclepro.2020.124651.
87. Habert, G.; D'Espinose De Lacaillerie, J.B.; Lanta, E.; Roussel, N. Environmental Evaluation for Cement Substitution with Geopolymers. In Proceedings of the 2nd International Conference on Sustainable Construction Materials and Technologies, Ancona, Italy, 28–30 June 2010.
88. Habert, G.; Ouellet-Plamondon, C. *Recent Update on the Environmental Impact of Geopolymers*; RILEM: Paris, France, 2016; pp. 17–23.
89. Komkova, A.; Habert, G. Environmental impact assessment of alkali-activated materials: Examining impacts of variability in constituent production processes and transportation. *Constr. Build. Mater.* **2023**, *363*, 129032. https://doi.org/10.1016/j.conbuildmat.2022.129032.
90. Nikravan, M.; Firdous, R.; Stephan, D. Life cycle assessment of alkali-activated materials: A systematic literature review. *Low-Carbon Mater. Green Constr.* **2023**, *1*, 13. https://doi.org/10.1007/s44242-023-00014-6.
91. Sandanayake, M.; Gunasekara, C.; Law, D.; Zhang, G.; Setunge, S. Greenhouse gas emissions of different fly ash based geopolymer concretes in building construction. *J. Clean. Prod.* **2018**, *204*, 399–408. https://doi.org/10.1016/j.jclepro.2018.08.311.
92. Alvarez-Gaitan, J.P.; Peters, G.M.; Rowley, H.V.; Moore, S.; Short, M.D. A hybrid life cycle assessment of water treatment chemicals: An Australian experience. *Int. J. Life Cycle Assess.* **2013**, *18*, 1291–1301. https://doi.org/10.1007/s11367-013-0574-4.
93. Du, F.; Warsinger, D.M.; Urmi, T.I.; Thiel, G.P.; Kumar, A.; Lienhard, J.H. Sodium Hydroxide Production from Seawater Desalination Brine: Process Design and Energy Efficiency. *Environ. Sci. Technol.* **2018**, *52*, 5949–5958. https://doi.org/10.1021/acs.est.8b01195.
94. Garcia-Herrero, I.; Margallo, M.; Onandía, R.; Aldaco, R.; Irabien, A. Environmental challenges of the chlor-alkali production: Seeking answers from a life cycle approach. *Sci. Total Environ.* **2017**, *580*, 147–157. https://doi.org/10.1016/j.scitotenv.2016.10.202.
95. Hong, J.; Chen, W.; Wang, Y.; Xu, C.; Xu, X. Life cycle assessment of caustic soda production: A case study in China. *J. Clean. Prod.* **2014**, *66*, 113–120. https://doi.org/10.1016/j.jclepro.2013.10.009.
96. Medina-Martos, E.; Gálvez-Martos, J.L.; Almarza, J.; Lirio, C.; Iribarren, D.; Valente, A.; Dufour, J. Environmental and economic performance of carbon capture with sodium hydroxide. *J. CO2 Util.* **2022**, *60*, 101991. https://doi.org/10.1016/j.jcou.2022.101991.
97. Raza, M.H.; Khan, M.; Zhong, R.Y. Strength, porosity and life cycle analysis of geopolymer and hybrid cement mortars for sustainable construction. *Sci. Total Environ.* **2024**, *907*, 167839. https://doi.org/10.1016/j.scitotenv.2023.167839.
98. Thiel, G.P.; Kumar, A.; Gómez-González, A.; Lienhard, J.H. Utilization of Desalination Brine for Sodium Hydroxide Production: Technologies, Engineering Principles, Recovery Limits, and Future Directions. *ACS Sustain. Chem. Eng.* **2017**, *5*, 11147–11162. https://doi.org/10.1021/acssuschemeng.7b02276.
99. Sala, S.; Crenna, E. *Global Normalisation Factors for the Environmental Footprint and Life Cycle Assessment*; Publications Office of the European Union: Luxembourg, 2017. https://doi.org/10.2760/88930.
100. Hou, C.; Yang, L.; Li, L.; Yan, B. Mechanical Characteristics and Stress Evolution of Cemented Paste Backfill: Effect of Curing Time, Solid Content, and Binder Content. *Front. Mater.* **2022**, *8*, 812402. https://doi.org/10.3389/fmats.2021.812402.
101. Muñoz, I.; Soto, A.; Maza, D.; Bayón, F. Life cycle assessment of refractory waste management in a Spanish steel works. *Waste Manag.* **2020**, *111*, 1–9. https://doi.org/10.1016/j.wasman.2020.05.023.
102. Alibaba. Reference Price of OPC. 2025. Available online: https://www.alibaba.com/product-detail/Factory-Direct-Manufacturer-Fast-Hardening-Portland_10000002732544.html (accessed on 10 August 2025).
103. Alibaba. Reference Price of NaOH. 2025. Available online: https://www.alibaba.com/product-detail/China-manufacturer-Caustic-Potash-Soda-Pearls_1601168174605.html (accessed on 10 August 2025).
104. De Bruyn, S.; Bijleveld, M.; De Graaff, L.; Schep, E.; Schroten, A.; Vergeer, R.; Ahdour, S. *Environmental Prices Handbook*; CE Delft: Delft, The Netherlands, 2018.

105. Gil Saenz, R. Treball Final de Grau. Design of a Lab-Scale System for the Study of Dry Desulfurization Process. Disseny D'un Sistema a Escala Laboratori per a L'estudi de Dessulfuració per via Seca. Raúl Gil Sáenz, 2018. Available online: https://diposit.ub.edu/dspace/bitstream/2445/141897/1/Gil_Saenz_Raul.pdf (accessed on 10 August 2025).
106. Chimenos, J.M.; Fernández, A.I.; Hernández, A.; Haurie, L.; Espiell, F.; Ayora, C. Optimization of phosphate removal in anodizing aluminium wastewater. *Water Res.* **2006**, *40*, 137–143. https://doi.org/10.1016/j.watres.2005.10.033
107. del Valle-Zermeño, R.; Giro-Paloma, J.; Formosa, J.; Chimenos, J.M. Low-grade magnesium oxide by-products for environmental solutions: Characterization and geochemical performance. *J. Geochem. Explor.* **2015**, *152*, 134-144. https://doi.org/10.1016/j.gexplo.2015.02.007
108. Giro-Paloma, J.; Formosa, J.; Chimenos, J.M. Stabilization study of a contaminated soil with metal(loid)s adding different low-grade MgO degrees. *Sustainability* **2020**, *12*, 7340. https://doi.org/10.3390/SU12187340
109. Seco, A.; del Castillo, J.M.; Espuelas, S.; Marcelino, S.; Echeverría, A.M. Assessment of the ability of MGO based binary binders for the substitution of Portland cement for mortars manufacturing. *Constr. Build. Mater.* **2022**, *341*, 127777. https://doi.org/10.1016/j.conbuildmat.2022.127777
110. Seco, A.; del Castillo, J.M.; Perlot, C.; Marcelino, S.; Espuelas, S. Recycled granulates manufacturing from spent refractory wastes and magnesium based binder. *Constr. Build. Mater.* **2023**, *365*, 130087. https://doi.org/10.1016/j.conbuildmat.2022.130087