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Cost-Based Environmental Liability Assessment of Acid Mine Drainage Management within the Integrated Geo-Hydrochemical Risk Assessment (IGHRA) Framework

Yazid Fanani *[1,2], Eko Teguh Paripurno [1], Dwi Fitri Yudiantoro [1], Tedy Agung Cahyadi [3]
[1] Department of Geological Engineering, Universitas Pembangunan Nasional "Veteran", Yogyakarta
[2] Department of Mining Engineering, Institut Teknologi Adhi Tama Surabaya, Surabaya
[3] Department of Mining Engineering, Universitas Pembangunan Nasional "Veteran", Yogyakarta
*corresponding author e-mail: fanani.yazid@gmail.com

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Abstract
Acid mine drainage generates environmental liabilities that mining operators must translate into measurable management costs. This study assesses the cost-based environmental liability of acid mine drainage management within the Integrated Geo-Hydrochemical Risk Assessment (IGHRA) framework by linking pollution risk, treatment performance, and environmental cost into a single risk-to-cost evaluation. The study uses source characterization, mine water quality data, compliance-based risk indicators, treatment performance, and operational cost components to estimate two principal cost categories, namely restoration cost and preventive expenditure cost. Restoration cost represents the annual cost required to restore contaminated mine water until all regulated parameters comply with the applicable standard. In contrast, preventive expenditure represents the five-year cost required to suppress future acid generation from potentially acid-forming material through selective handling, temporary stockpiling, installation of compacted clay cover, and runoff control. The results show that the only CaO scenario produces the lowest treatment cost, but does not achieve full compliance because sulfate remains above the standard. The CaO 10 g/L and $BaCl_2$ 3.0 g/L scenario achieves full compliance and provides the most economical restoration cost among the compliant options, with an estimated value of IDR 516,463,856 per year. The total preventive expenditure cost over five years reaches IDR 3,832,950,186. At a midpoint selling price of IDR 89,000 per ton, the compliant CaO and $BaCl_2$ scenario exerts an environmental cost pressure of 8.58 percent on projected gross revenue. These results show that the IGHRA framework can convert acid mine drainage risk into measurable environmental liability and can support technically grounded and economically accountable mine water management.

1. Introduction

Acid mine drainage remains one of the most persistent environmental problems in mining because sulfide oxidation produces acidic water enriched with sulfate and dissolved metals [1]–[3]. This process intensifies when mining exposes sulfide-bearing rock to oxygen and water, and the resulting drainage can degrade surface water quality, disturb aquatic environments, and increase the burden of environmental management [1]–[3]. Therefore, the mine operators need an assessment approach that can connect acid-forming source characteristics, water quality deterioration, risk status, and management cost within one analytical framework.

Source characterization provides the first basis for understanding acid mine drainage. Previous studies have shown that sulfide occurrence, sulfur content, and the balance between acid-generating potential and neutralizing capacity control whether mine materials behave as potentially acid-forming or non-acid-forming [4], [5]. These studies support early-stage diagnosis at the source level, but they usually stop at classification and do not extend the analysis to downstream water response, compliance status, or environmental liability.

Pathway and receptor studies provide the second basis for understanding acid mine drainage. Hydrochemical investigations have shown that acid mine drainage lowers pH and elevates sulfate and dissolved metals along mine water flow paths [2], [3]. Risk-based water assessment can organize that hydrochemical response into management priorities because diagnostic indices such as Contamination Factor and Pollution Load Index identify the dominant pollutant burden, while compliance-oriented indicators clarify whether water meets the applicable quality standard [6]–[10]. These studies have improved the interpretation of contaminated mine water, yet most of them still treat pollution status as an endpoint and do not translate risk reduction into a structured estimate of environmental liability.

Treatment studies provide the third basis for decision-making because they show how active control can reduce pollutant burden. Calcium-based reagents such as CaO and CaCO3 can raise pH and remove dissolved metals effectively, although treatment performance still depends on initial acidity, reaction conditions, and the persistence of sulfate in solution [11]. This technical evidence is necessary for cost analysis because environmental liability cannot be estimated credibly without first identifying whether a treatment option only reduces contamination or restores full compliance.

Cost studies have addressed parts of this problem, but they still leave an analytical gap. Previous studies have estimated remediation costs, hydro economic implications, and environmental damages through net present value, life cycle cost, and cost-based valuation approaches [12]–[16]. These studies offer important economic perspectives, but most of them do not integrate acid-forming source evidence, water quality risk, treatment performance, and compliance status into a single site-specific framework. As a result, cost often appears as a separate accounting output rather than as the economic consequence of technically observed risk.

The Integrated Geo-Hydrochemical Risk Assessment (IGHRA) framework provides the structure needed to close that gap. In this study, IGHRA links source characterization, mine water response along the pathway and receptor system, risk assessment through CF, PLI, and Residual Risk, treatment-based risk reduction, and environmental cost valuation within one management sequence. This sequence allows the study to read environmental liability as a measurable management burden created by acid mine drainage risk, non-compliance, and the technical actions required to restore compliance and prevent recurrence. Figure 1 summarizes the IGHRA framework used in this study and shows how characterization, compliance-based risk assessment, treatment evaluation, and cost valuation are linked within one AMD management sequence.

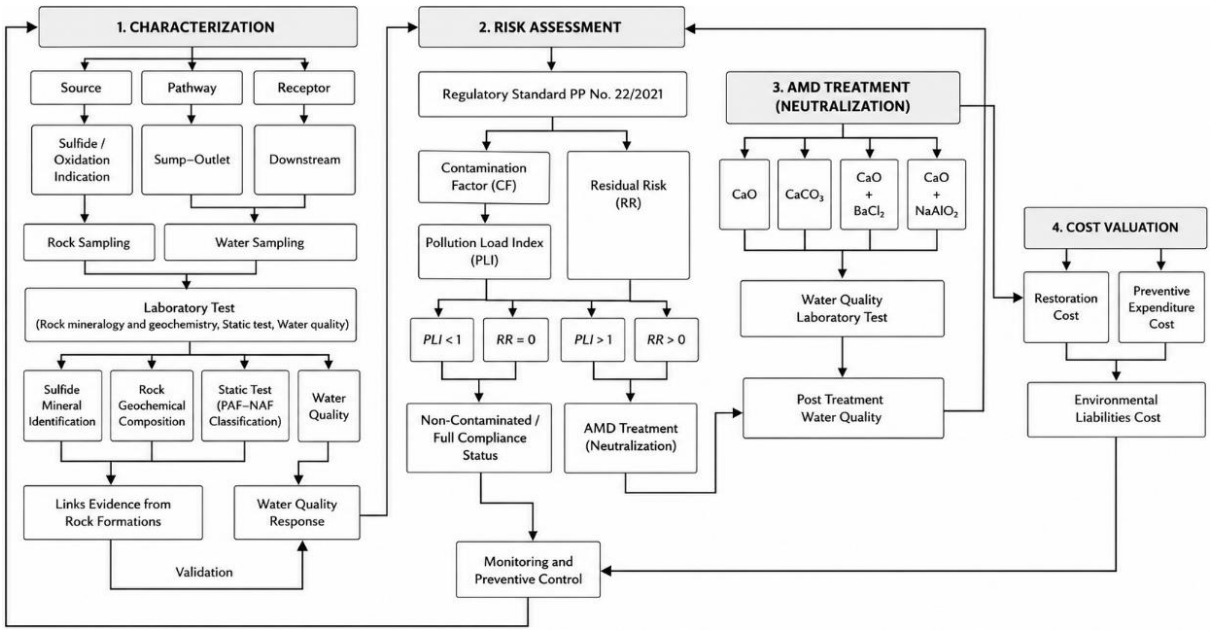


Figure 1. Integrated Geo-Hydrochemical Risk Assessment (IGHRA) framework used in this study

The gap addressed in this study is not the absence of individual AMD investigations but the lack of an integrated framework that links them within one consistent management sequence. Previous studies have commonly examined source characterization, water-quality deterioration, compliance-based risk, treatment performance, or remediation cost as separate topics. The IGHRA framework closes this gap by connecting these components explicitly, so that environmental liability is interpreted as the consequence of source evidence, hydrochemical response, risk status, treatment-based risk reduction, and the resulting restoration and preventive cost requirements within one site-specific risk-to-cost pathway.

The novelty of the IGHRA framework does not lie in the use of individual AMD assessment components alone, but in the integration of these components into a single risk-to-cost management sequence. Previous AMD studies have commonly evaluated source characterization, hydrochemical contamination, treatment performance, or remediation cost as separate analytical domains [2], [3], [12], [14]. As a result, pollution status and environmental expenditure are often interpreted independently, making it difficult to translate technical AMD conditions into measurable environmental liability. The IGHRA framework addresses this gap by explicitly linking sulfide-source evidence, hydrochemical response, compliance-based residual risk, treatment-driven risk reduction, and economic consequence within one continuous assessment structure. This integration allows environmental liability to be interpreted not merely as remediation expenditure, but as the measurable economic consequence of unresolved contamination and the technical actions required to achieve regulatory compliance and prevent future AMD recurrence.

This paper evaluates the cost-based environmental liability of acid mine drainage management within the IGHRA framework. The paper focuses on two cost components. Restoration cost represents the annual cost required to restore contaminated mine water until all assessed parameters comply with the applicable standard. Preventive expenditure represents the multiyear cost required to suppress future acid generation from potentially acid-forming material through selective handling, temporary placement, installation of compacted clay cover, and runoff control. The paper compares restoration scenarios based on CaO only, CaO with $BaCl_2$, and CaO with $NaAlO_2$, and then examines how the resulting environmental cost affects projected gross mining revenue.

2. Methodology

This study applied the Integrated Geo-Hydrochemical Risk Assessment framework as a risk-to-cost sequence for acid mine drainage management. The framework linked source characterization, mine-water quality response, risk assessment, treatment performance, and cost valuation in one analytical workflow. This paper focused on the environmental liability stage, but it retained risk and compliance assessment as the basis for estimating restoration cost and preventive expenditure cost.

2.1. Study design and analytical scope

The study used previously generated site data from PT Gunung Bale, South Malang, Indonesia. Source characterization had identified sulfide-bearing material through static testing, XRF geochemistry, and mineragraphy, while mine water evaluation had measured pH, TDS, TSS, dissolved Fe, dissolved Mn, and sulfate along the mine drainage system. The analytical sequence consisted of four linked steps. First, the study defined the liability basis from pollution burden and compliance status using Contamination Factor, Pollution Load Index, and Residual Risk. Second, the study selected treatment scenarios from the previous laboratory stage and converted the optimum doses into field-scale reagent demand. Third, the study estimated the annual restoration cost for each treatment scenario. Fourth, the study estimated the preventive expenditure cost for the five-year mining period and compared the combined environmental cost with projected gross mining revenue.

Table 1. Regulatory thresholds used in this study

Parameter	Unit	Class IV standard of Government Regulation No. 22/2021
pH	-	6.0–9.0
TDS	mg/L	2,000
TSS	mg/L	400
Fe (dissolved)	mg/L	10
Mn (dissolved)	mg/L	5

2.2. Liability basis from risk and compliance assessment

The study assessed environmental liability from two complementary perspectives. The first perspective was diagnostic pollution burden, represented by Contamination Factor and Pollution Load Index. The second perspective was compliance status, represented by Residual Risk. Within this framework, liability increased when pollutant burden remained high and regulated parameters still failed to meet the applicable standard.

Contamination Factor for each diagnostic parameter was calculated as:

$$CF_i = \frac{C_i}{S_i} \quad \text{.....(1)}$$

where *CFi* is the contamination factor of parameter i, *Ci* is the measured concentration, and *Si* is the regulatory standard for the same parameter [17].

The Pollution Load Index was calculated from Fe, Mn, and SO_4 as:

$$\text{PLI} = (CF_1 \times CF_2 \times \cdots \times CF_n)^{1/n} \quad \text{.....(2)}$$

where PLI>1 indicates polluted water and PLI<1 indicates non-polluted status for the selected diagnostic parameters [18].

Residual Risk was calculated as the proportional share of regulated parameters that still exceeded the Class IV standard of Government Regulation No. 22/2021 [25]. :

$$RR = \frac{n_{exceed}}{n_{total}} \times 100\% \quad \text{.....(3)}$$

where n_{exceed} is the number of parameters that fail the standard and n_{total} is the total number of assessed parameters. The study used six parameters for compliance assessment, namely pH, TDS, TSS, Fe, Mn, and SO_4 [3]–[5]. Regulatory thresholds used in this study shows on table 1.

The study did not include pH, TDS, or TSS in the PLI calculation. It treated pH as an acidity indicator rather than a pollutant load, while TDS and TSS were treated as bulk parameters that were less diagnostic for AMD processes. For decision purposes, the study interpreted treatment outcomes in two levels. A scenario was treated as a full restoration option when it produced $PLI<1$ and $RR=0$. A scenario was treated as a partial risk reduction option when it reduced *PLI* and *RR* but still left $RR>0$.

2.3. Treatment scenarios used for cost valuation

The treatment cost model used the main hotspot water as the design basis because the previous hydrochemical and risk assessment stages had identified this point as the main source of non-compliance and the principal target for active treatment. The study then carried forward the optimum laboratory treatment results into three field application scenarios.

The first scenario used CaO as the primary neutralizing reagent. The study assigned 10.0 kg CaO/m^3 for the CaO scenario. The second scenario used CaO for primary neutralization, followed by $BaCl_2$ for

sulfate polishing. The study assigned 10.0 kg CaO/m^3 plus 3.0 kg $BaCl_2$/m3 for the CaO-$BaCl_2$ scenario. The third scenario used CaO and $NaAlO_2$ to represent a neutralization stage followed by ettringite-based sulfate removal. The study assigned 14.0 kg reagent/m^3 for the CaO-$NaAlO_2$ scenario, represented by primary neutralization together with ettringite-forming reagents. This scenario structure followed the technical logic that lime-based neutralization removes acidity and dissolved metals effectively, but sulfate may require a secondary polishing step when full compliance is targeted [19]–[24].

These reagent requirements were then applied to the effective treatment volume of the existing pond system. One treatment cycle was defined as the effective treatment equal to 294.67 m^3 per cycle. Annual operation frequency followed the monthly management cycle scenario derived from rainfall-based operational categories, giving 32 treatment cycles per year.

2.4. Estimation of annual restoration cost

The annual restoration cost model included reagent material cost, reagent delivery cost, treatment labor cost, residue handling cost, and water quality monitoring cost. The study first calculated reagent demand for each cycle:

$$M_{i,s} = D_{i,s} \times V_{eff} \quad \text{.....(4)}$$

where $M_{i,s}$ is the mass of reagent i required in scenario s per cycle, $D_{i,s}$ is the specific reagent demand, and V_{eff} is the effective treatment volume.

Annual reagent demand was then calculated as:

$$A_{i,s} = \sum_{m=1}^{12} (N_m \times M_{i,s}) \quad \text{.....(5)}$$

where $A_{i,s}$ is the annual reagent demand for reagent i in scenario s, and N_m is the number of treatment cycles in month m.

Annual reagent material cost was calculated as:

$$C_{mat,s} = \sum_{i=1}^{n} (A_{i,s} \times P_i) \quad \text{.....(6)}$$

where $C_{mat,s}$ is the annual reagent material cost for scenario s, and P_i is the unit price of reagent i. The study used case-specific working prices of IDR 1,600/kg for CaO, IDR 11,000/kg for $BaCl_2$, and IDR 25,000/kg for $NaAlO_2$.

The study then added the delivery cost from Surabaya to the study area. Reagent transport used an 8,000 kg Fuso truck and an assumed delivery tariff of IDR 1,500,000 per trip. The number of trips was calculated as:

$$T_{m,s} = \left\lceil \frac{A_{m,s}}{Q_{truck}} \right\rceil \quad \text{.....(7)}$$

where $T_{m,s}$ is the number of trips in month m for scenario s; $A_{m,s}$ is the monthly reagent demand; and Q_{truck} is truck capacity. Delivery cost was then calculated as:

$$C_{ship,s} = \sum_{m=1}^{12} (T_{m,s} \times P_{trip}) \quad \text{.....(8)}$$

where $C_{ship,s}$ is the delivery cost for scenario s, and P_{trip} is the delivery tariff per trip. Total annual reagent cost was then calculated as:

$$C_{reag,s} = C_{mat,s} + C_{ship,s} \quad \text{.....(9)}$$

The study estimated treatment labor cost from the daily field manpower required for weighing, mixing, and dosing reagents. It assumed four general workers per treatment cycle and a wage of IDR 150,000/person/day. Annual labor cost was calculated as:

$$C_{lab,s} = \sum_{m=1}^{12} (N_m \times n_w \times W_d) \quad \text{.....(10)}$$

where n_w is the number of workers and W_d is the daily wage per worker.

Water quality monitoring cost used one analytical package containing pH, TDS, TSS, Fe, Mn, and SO_4, with a unit cost of IDR 333,000/sample. Monitoring frequency was set at two samples per month during wet months and one sample per month during moist and dry months. Monitoring cost was calculated as:

$$C_{mon} = \sum_{m=1}^{12} (S_m \times P_{test}) \quad \text{.....(11)}$$

where S_m is the number of monitoring samples in month m, and P_{test} is the analytical cost per sample.

The study also estimated residue handling costs from wet sludge generation. Scenario specific wet sludge factors were 0.0248 m^3/m^3 treated water for CaO, 0.0387 m^3/m^3 for CaO-$BaCl_2$, and 0.0518 m^3/m^3 for CaO-$NaAlO_2$. Sludge volume per cycle was calculated as:

$$V_{sludge,s} = g_s \times V_{eff} \quad \text{.....(12)}$$

where g_s is the wet sludge factor of scenario s. Residue handling cost was then calculated as:

$$C_{res,s} = \sum_{m=1}^{12} (N_m \times V_{sludge,s} \times P_{res}) \quad \text{.....(13)}$$

where P_{res} is the residue handling cost per m^3 of wet sludge. The study used a case-specific handling cost of IDR 6,100/m^3, derived from fuel, operator, and equipment productivity assumptions in the field cost model.

Finally, the annual restoration cost for each scenario was calculated as:

$$RC_s = C_{reag,s} + C_{lab,s} + C_{mon} + C_{res,s} \quad \text{.....(14)}$$

where R_{Cs} are the annual restoration cost for scenarios. The study compared all treatment scenarios economically, but it treated only the fully compliant scenario as the basis for full environmental restoration.

2.5. Estimation of preventive expenditure cost

The study defined preventive expenditure cost as the five-year cost required to suppress new AMD generation from potentially acid-forming material during the mining period. The preventive model used the annual volume of PAF material from the mine schedule and translated it into management cost for movement and temporary stockpiling, cover purchase, and cover compaction.

Annual movement and temporary stockpiling cost was calculated as:

$$C_{move,y} = V_{PAF,y} \times P_{move} \quad \text{.....(15)}$$

where $C_{move,y}$ is the annual cost of PAF movement and stockpiling in year y; $V_{PAF,y}$ is the annual volume of PAF material; and P_{move} is the unit movement cost.

The study then estimated the annual cost of cover installation from the additional cover volume required in each year. Cover cost was divided into material purchase and compaction components:

$$C_{cover,y} = C_{cover_mat,y} + C_{cover_comp,y} \quad \text{.....(16)}$$

The annual preventive expenditure cost was therefore calculated as:

$$PEC_y = C_{move,y} + C_{cover_mat,y} + C_{cover_comp,y} \quad \text{.....(17)}$$

Total preventive expenditure cost over the mine schedule was calculated as:

$$PEC_{total} = \sum_{y=1}^{5} PEC_y \quad \text{.....(18)}$$

2.6. Environmental cost pressure against projected gross revenue

The study evaluated environmental cost pressure as a partial economic indicator. It compared AMD-related environmental cost with projected gross revenue in order to show how much of the planned revenue would be absorbed by restoration and prevention. The present valuation, therefore represents a base case comparison under explicit operational assumptions and was not designed as a probabilistic uncertainty model.

Annual gross revenue was calculated as:

$$GR_y = V_{ore,y} \times \rho \times R_{tot} \times P_{sell} \quad \text{.....(19)}$$

where GR_y is gross revenue in year y; $V_{ore,y}$ is pyrophyllite volume mined in year y; ρ is material density; R_{tot} is total recovery; P_{sell} is selling price. The study used a density of 2.73 ton/m^3, a mining recovery of 99.50%; a product recovery of 95.00%, and a midpoint selling price of IDR 89,000/ton.

Total gross revenue for the five-year schedule was calculated as:

$$GR_{total} = \sum_{y=1}^{5} GR_y \quad \text{.....(20)}$$

The total environmental cost of each scenario was then calculated as:

$$EC_s = 5 \times RC_s + PEC_{total} \quad \text{.....(21)}$$

where EC_s is the combined environmental cost of scenarios.

Environmental cost pressure was finally calculated as:

$$ECP_s = \frac{EC_s}{GR_{total}} \times 100\% \quad \text{.....(22)}$$

where *ECP* is the environmental cost pressure in percent. This indicator expressed the relative economic burden of AMD management under each restoration scenario.

3. Results and discussions

3.1. Environmental Liability Basis from Risk and Compliance Status

The study first established environmental liability from the measured pollution burden and compliance status before assigning any cost. Within the IGHRA sequence, this step was necessary because liability in this study did not arise from treatment expenditure alone, but from the documented condition of contaminated mine water and from the technical effort required to restore compliance. Pre-treatment results identified L3, the bottom pit sump, as the main hotspot, with pH 1.5, dissolved Fe 3.119 mg/L, and SO_4 6.269 mg/L. At this location, the Pollution Load Index reached 9.365, which indicated extreme

pollution, while the Residual Risk was 50% because three of the six regulated parameters still failed to meet the Class IV standard of Government Regulation No. 22/2021 (see Table 2).

The liability basis did not remain confined to the source point. L3 was used as the treatment design basis because it contained the most extreme AMD signature and the highest initial pollution load, which made it the most relevant point for sizing active restoration. However, the spatial pattern of liability extended beyond L3. Along the outlet to the downstream segment, Fe and Mn decreased progressively, but acidity and sulfate persisted, so compliance failure remained high after water left the sump area. In practical terms, L3 represents the point where the restoration demand is generated most strongly, whereas L4 to L6 show how that burden is transmitted and maintained along the mine water pathway.

Post treatment results, then refined this liability basis into three practical categories (Figure 2). CaO at 10 g/L reduced the diagnostic pollution burden effectively, lowering the post-treatment PLI to 0.501 and leaving SO_4 as the dominant remaining parameter, but it still produced a Residual Risk of 17% because sulfate remained above the regulatory threshold. Full compliance was only achieved in the two-stage polishing scenarios. The CaO 10 g/L and $BaCl_2$ 3.0 g/L scenario reduced SO_4 to 374 mg/L and satisfied all regulated parameters, while the CaO and $NaAlO_2$ scenario reduced SO_4 further to about 263 mg/L under its optimum condition and also maintained the other key parameters within the standard range.

The IGHRA framework contributes conceptually by linking source evidence, hydrochemical deterioration, compliance status, treatment response, and economic consequence within a single analytical sequence. Previous AMD studies commonly treated these components independently, resulting in fragmented interpretation between environmental diagnosis and management expenditure [3], [11], [12]. Hydrochemical investigations often focused on contamination processes and pollutant distribution [2], [9], [10], whereas economic studies concentrated on remediation valuation without explicitly integrating compliance-based treatment prioritization [13], [14], [15]. In contrast, the present framework interprets environmental liability as the cumulative consequence of source-derived contamination, pathway persistence, residual compliance failure, and the technical actions required for restoration and prevention.

Table 2. Liability basis from pollution burden, compliance status, and post-treatment outcome

Stage/scenario	Pollution Load Index (PLI)	Residual Risk (RR)	Compliance status	Interpretation for liability assessment
L1	0.0490	17%	Partial compliance	Baseline liability
L2	0.0210	17%	Partial compliance	Baseline liability
Pre-treatment L3 hotspot	9.3650	50%	Non compliant	Extreme baseline liability (treatment zone)
L4	2.4230	70%	Non compliant	Baseline liability
L5	2.1100	70%	Non compliant	Baseline liability
L6	1.7850	70%	Non compliant	Baseline liability
L7	1.5250	50%	Non compliant	Baseline liability
L8	1.2080	50%	Non compliant	Baseline liability
L9	0.8880	33%	Partial compliance	Baseline liability
CaO	0.5013	17%	Partial compliance	Risk reduction only
CaO and $BaCl_2$	0.2824	0%	Full compliance	Preferred full restoration scenario
CaO and $NaAlO_2$	0.2451	0%	Full compliance	Alternative full restoration scenario

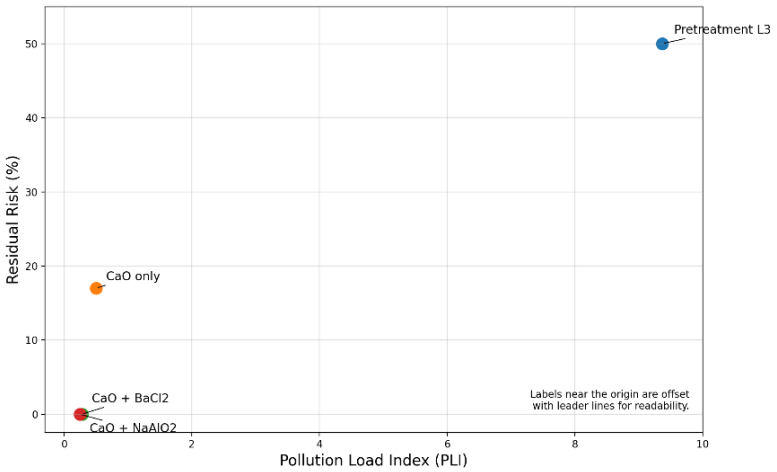


Figure 2. Transition from pre-treatment liability to post-treatment compliance

3.2. Restoration Cost under Alternative Treatment Scenarios

The restoration cost analysis translated the post-treatment liability status into annual expenditure for technically distinct treatment scenarios. The cost model included reagent cost, labor, water quality monitoring, and residue handling. Because all scenarios used the same operational setting, the main cost differences arose from reagent demand and, to a much smaller extent, residue generation. The annual restoration cost totals showed a clear three-level pattern and were governed mainly by chemical demand rather than by labor or residue management (Table 3 and Figure 3). The CaO scenario produced the lowest annual restoration cost at IDR 204,491,984. The CaO and $BaCl_2$ scenario increased the annual cost to IDR 516,463,856, while the CaO and $NaAlO_2$ scenario reached IDR 609,369,904. The cost breakdown showed that reagent expenditure dominated the total in all scenarios, with annual reagent costs of IDR 177,871,040 for the CaO scenario, IDR 489,042,560 for the CaO-$BaCl_2$ scenario, and IDR 581,197,056 for the CaO-$NaAlO_2$ scenario. By contrast, labor remained constant at IDR 19,200,000 per year, monitoring remained constant at IDR 5,994,000 per year, and residue handling contributed only a minor addition, ranging from IDR 1,426,944 to IDR 2,978,848 per year.

Although the CaO-only scenario reduced overall pollution load substantially, sulfate remained above the regulatory threshold despite successful neutralization of acidity and reduction of dissolved metals. This outcome reflects the different geochemical behavior of sulfate compared with Fe and Mn during alkaline treatment. Lime addition effectively increases pH and promotes precipitation of dissolved metals as hydroxide phases, but sulfate commonly remains highly soluble under conventional neutralization conditions [11], [19], [20]. As a result, treatment based solely on alkalinity correction can achieve partial risk reduction without necessarily restoring full compliance. Similar observations have been reported in AMD treatment studies where sulfate persists after metal precipitation and continues to control residual water quality risk [3], [20], [24].

Table 3. Annual restoration cost by treatment scenario (IDR/Year)

Cost component	CaO scenario	CaO+$BaCl_2$ scenario	CaO+$NaAlO_2$ scenario	Notes
Reagent-related cost	177,871,040	489,042,560	581,197,056	Includes material and delivery cost
Labor cost	19,200,000	19,200,000	19,200,000	
Water quality monitoring cost	5,994,000	5,994,000	5,994,000	
Residue handling cost	1,426,944	2,227,296	2,978,848	
Total annual restoration cost	**204,491,984**	**516,463,856**	**609,369,904**	**CaO+$BaCl_2$ is the most economical full compliance scenario**

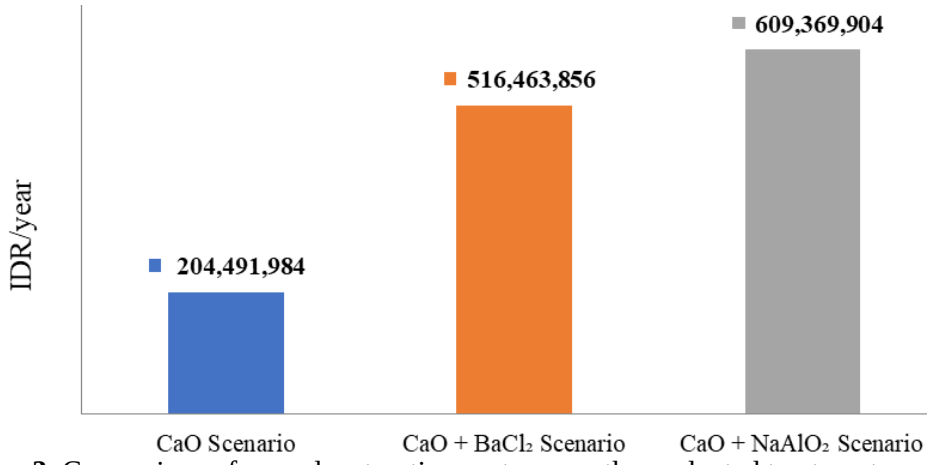


Figure 3. Comparison of annual restoration cost among the evaluated treatment scenarios

However, the lowest cost scenario did not represent the preferred restoration option. The CaO scenario was economically attractive, but it still left a Residual Risk of 17% because sulfate remained above the standard. Therefore, it represented a partial risk-reduction scenario rather than a full restoration scenario. Among the two compliant alternatives, the CaO and $BaCl_2$ scenario was more economical than the CaO and $NaAlO_2$ scenario and therefore became the preferred restoration cost scenario. In decision terms, the study therefore treated the CaO and $BaCl_2$ scenario as the most economical full restoration pathway, not because it was the cheapest scenario overall, but because it was the lowest cost scenario that restored the water to full compliance.

The CaO-$BaCl_2$ scenario emerged as the preferred restoration pathway because it achieved complete compliance with lower total cost than the CaO-$NaAlO_2$ alternative. Economically, this result was controlled primarily by reagent demand rather than by labor, monitoring, or sludge management costs, which remained relatively constant across scenarios. Technically, $BaCl_2$ provided an efficient sulfate polishing mechanism through $BaSO_4$ precipitation, allowing compliance to be achieved without the substantially higher reagent requirement associated with ettringite-forming treatment systems [21], [24], [26]. This finding indicates that restoration efficiency in AMD management is not determined solely by treatment effectiveness, but also by the balance between chemical selectivity, operational simplicity, and compliance achievement.

Selecting the CaO and $BaCl_2$ scenario as the preferred restoration option does not eliminate secondary environmental considerations. Sulfate removal via barium addition relies on precipitation of highly insoluble $BaSO_4$, a mechanism widely reported as effective for sulfate polishing in AMD treatment systems [21], [26]. While this approach can achieve strong sulfate reduction, it also transfers part of the management challenge from dissolved sulfate in the aqueous phase to the generation, settling, and handling of barium-bearing solids.

Previous studies show that barium-based sulfate removal is effective, but field implementation requires tight control of reagent dosing, robust solid–liquid separation, and post-treatment verification to reduce the risk of residual dissolved barium or $BaSO_4$-rich suspended solids in the effluent [24], [26]. Long-term sludge management also remains critical, as the resulting residues should be treated as a controlled solid phase rather than assumed to be environmentally inert under all site conditions [21], [24].

3.3. Preventive Expenditure Cost for Future AMD Suppression

Restoration cost alone did not fully represent environmental liability at the site, because the mine also required source control measures to suppress future AMD formation. The geological and mineralogical results had already shown that pyrite-bearing materials acted as the main AMD-forming source at the site. In parallel, the hydrochemical results showed that mining activity had already generated a severe hotspot at L3. Under this condition, the study treated preventive expenditure cost as a forward-looking liability component that addressed the source of future risk.

Table 4. Annual preventive expenditure cost for PAF management during the five-year mine plan

Year	Annual PAF volume (m^3)	Additional cover volume (m^3)	PAF handling and temporary stockpiling (IDR/year)	Cover material cost (IDR/year)	Cover compaction cost (IDR/year)	Preventive expenditure cost (IDR/year)
1	275	105	10,544,443	16,926,000	116,203	27,586,647
2	17,275	1,565	662,382,763	252,197,400	1,731,431	916,311,594
3	27,600	1,845	1.058,278,683	297,478,480	2,042,303	1,357,799,466
4	17,625	1,102	675,802,963	177,642,400	1,219,583	854,664,946
5	14,025	855	537,766,613	137,874,360	946,560	676,587,533
Total	**76,800**	**5,472**	**2,944,775,465**	**882,118,640**	**6,056,080**	**3,832,950,186**

The preventive cost model was built from two-unit cost components. The first was the cost of moving and temporarily stockpiling PAF material, calculated at IDR 38,343/m³. The second was the installed cost of compacted clay cover, calculated at IDR 162,307/m³. Using these unit costs, the annual preventive expenditure cost rose sharply in years with larger PAF volumes. The five-year total reached IDR 3,832,950,186. The highest burden occurred in years 2 and 3 because those years carried the largest yearly PAF volumes and additional cover requirements (Table 4 and Figure 4).

The preventive expenditure component exceeded the annual restoration cost, indicating that long-term AMD liability at the site was driven more strongly by future source control requirements than by short-term water treatment alone. This pattern reflects the persistence of sulfide-bearing material exposure during continued mining operations, where oxidation of pyrite-bearing material remains the principal driver of acid generation [1], [2], [3]. Previous AMD studies emphasized that active treatment alone cannot eliminate long-term environmental burden when acid-forming material continues to be exposed during mining progression [4], [11]. Without preventive handling, temporary encapsulation, and runoff isolation, active treatment demand would likely continue or intensify throughout the mine life.

The study treated prevention as a site-specific expenditure required to interrupt the source-to-pathway connection that would otherwise reproduce AMD in later years. This interpretation made preventive expenditure a necessary complement to restoration cost. Restoration addressed current non-compliance, while prevention reduced the probability that the same liability would recur during continued mine operation (see Figure 5).

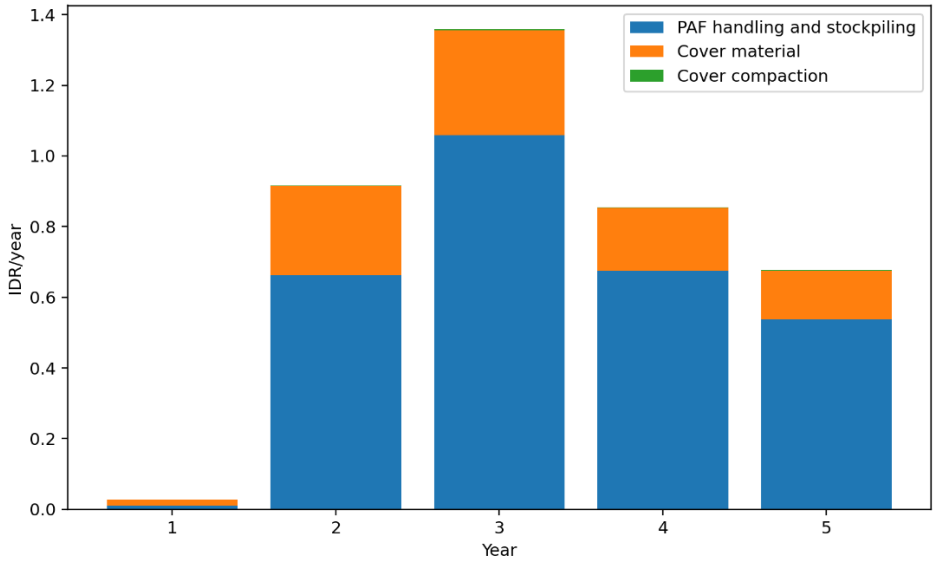


Figure 4. Five-year trend of preventive expenditure cost and its main components

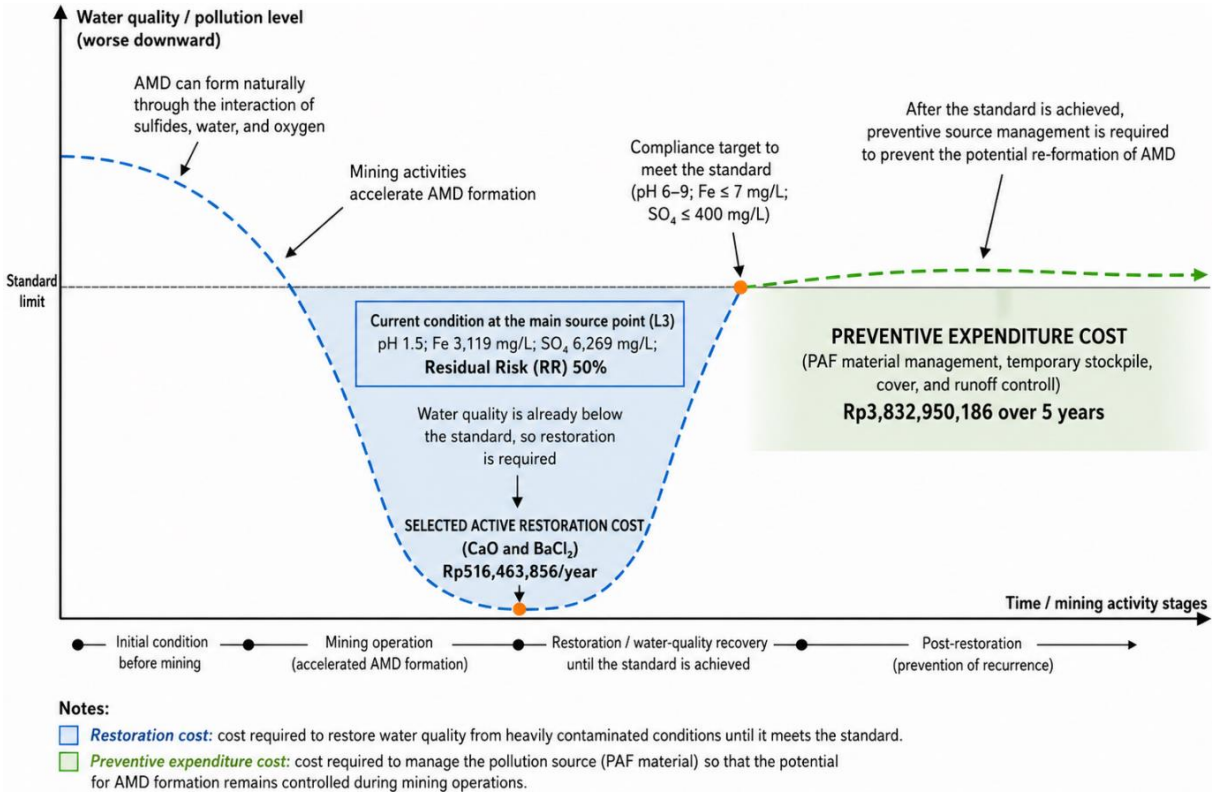


Figure 5. Conceptual relationship between restoration cost and preventive expenditure cost within the IGHRA risk-to-cost framework

3.4. Environmental Cost Pressure against Projected Gross Revenue

After estimating restoration and preventive costs, the study evaluated their economic weight against projected gross mining revenue. Gross revenue was calculated from the five-year pyrophyllite production plan using a density of 2.73 ton/m³, a total recovery of 94.525%, and a midpoint selling price of IDR 89,000/ton. Under these assumptions, the mine generated an estimated five-year gross revenue of IDR 74,728,027,834. Annual gross revenue varied from IDR 12,884,340,719 in year 2 to IDR 16,203,034,541 in year 4, reflecting variation in the planned mining volume.

The environmental cost pressure results showed that the three restoration scenarios imposed clearly different economic burdens (Table 5 and Figure 6). Over the five years, the CaO scenario generated the lowest pressure at 6.50% of gross revenue, the CaO and $BaCl_2$ scenario generated 8.58%, while the CaO and $NaAlO_2$ scenario generated 9.21%. The annual pattern also showed that years 2 and 3 carried the highest pressure in all scenarios. For the compliant scenarios, the yearly burden reached 11.12% and 13.25% in the CaO and $BaCl_2$ scenario, and 11.84% and 13.91% in the CaO and $NaAlO_2$ scenario. This peak did not result from revenue collapse, but from the simultaneous rise in yearly preventive expenditure cost when larger PAF volumes and cover requirements entered the mining sequence.

Environmental cost pressure remained below 10% of projected gross revenue even under the compliant treatment scenarios, suggesting that AMD management remained economically tolerable at the operational scale considered in this study. However, this apparent tolerance should be interpreted cautiously because the present analysis used gross revenue rather than net operating margin and therefore excluded broader mining expenditures such as processing cost, taxation, royalties, and corporate overhead. Previous mining liability studies similarly noted that environmental obligations may appear manageable at the revenue level but become substantially more restrictive when integrated into full operational economics and closure liability structures [13], [14], [16].

Table 5. Environmental cost pressure against the projected gross revenue under the alternative restoration scenarios

Year	Gross revenue (IDR)	CaO only pressure (%)	CaO + $BaCl_2$ pressure (%)	CaO + $NaAlO_2$ pressure (%)
1	15,944,658,724	1,46%	3,41%	3,99%
2	12,884,340,719	8,70%	11,12%	11,84%
3	14,141,769,693	11,05%	13,25%	13,91%
4	16,203,034,541	6,54%	8,46%	9,04%
5	15,554,224,157	5,66%	7,67%	8,27%
Total	**74,728,027,834**	**6,50%**	**8,58%**	**9,21%**

From a liability perspective, the lowest pressure scenario could not automatically be treated as the preferred scenario. CaO alone imposed the smallest burden on gross revenue, but it did not achieve full compliance and therefore could not serve as the principal restoration choice. The CaO and $BaCl_2$ scenario provided the best balance between compliance and cost, because it achieved full restoration while keeping environmental cost pressure below the higher cost $NaAlO_2$ option. Even so, the results should still be read as a partial economic indicator rather than as a full feasibility judgment. These studies did not include other mining costs such as processing costs, general overhead, labor outside the treatment system, taxes, royalties, and other financial components. Therefore, the environmental cost share remained tolerable at the gross revenue level, but the available margin would narrow after broader mine economics were included.

The cost model in this study should be read as a deterministic base case estimate rather than as a full uncertainty model. This choice follows the scope of the present paper, which aims to compare AMD management liability under explicit site-specific assumptions, not to develop a probabilistic economic simulation. Reagent prices, treatment cycle frequency, and annual PAF volume are acknowledged as the principal cost drivers because they directly control reagent demand, annual restoration cost, and preventive expenditure cost.

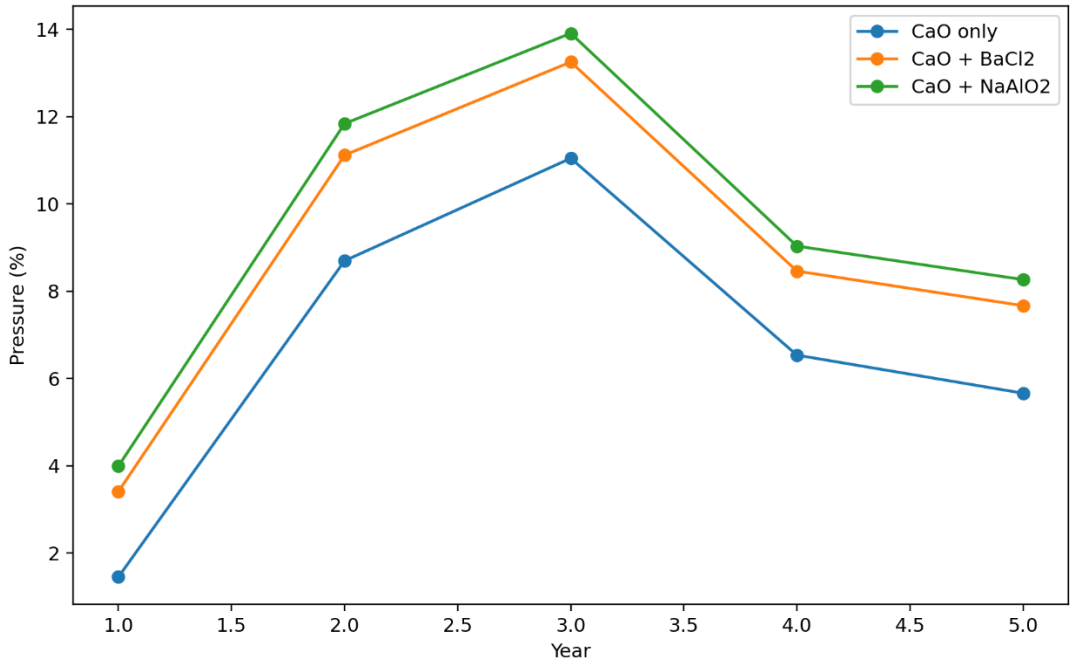


Figure 6. Environmental cost pressure against the projected gross revenue under the alternative restoration scenarios

The present valuation also remains limited by its laboratory-to-field scaling basis. The restoration cost model was developed from optimum laboratory treatment doses that were converted into reagent demand per unit treated water volume and then scaled to the effective treatment volume and annual operating cycle of the existing pond system. For these reasons, the reported restoration cost and the associated environmental cost pressure should be viewed as scenario-based management estimates rather than fully validated field operating costs. The model could be improved with pilot or full-scale operational data, especially to confirm treatment frequency, actual reagent consumption, and sludge handling performance under variable field conditions.

3.5. Sensitivity Analysis of Environmental Liability Estimation

Sensitivity analysis was conducted to evaluate the robustness of the environmental liability estimation under changing operational assumptions. The analysis focused on three principal cost drivers, namely reagent price, treatment frequency, and product selling price, because these variables directly influence restoration expenditure and environmental cost pressure. The simulation showed that reagent price variation produced the strongest effect on annual restoration cost, particularly in the CaO-$BaCl_2$ and CaO-$NaAlO_2$ scenarios where chemical demand dominated the operational budget. An increase in treatment frequency also generated substantial cost escalation because reagent consumption, labor demand, sludge generation, and monitoring frequency increased proportionally with the number of treatment cycles. By contrast, higher product selling prices reduced environmental cost pressure by increasing the gross revenue denominator, whereas lower selling prices amplified the relative burden of AMD management. Although the present valuation adopted deterministic operational assumptions, the relative ranking of the evaluated treatment scenarios remained stable under moderate economic variation. The CaO-only scenario consistently produced the lowest cost burden but failed to achieve full compliance, while the CaO-$BaCl_2$ scenario remained the most economical compliant restoration option across the tested conditions. This outcome suggests that the preferred restoration pathway was controlled more strongly by compliance achievement and sulfate removal efficiency than by minor operational cost fluctuation.

3.6. Study Limitations and Future Development

Several limitations should be considered when interpreting the present results. First, the restoration cost model was developed from laboratory-scale optimum treatment conditions and subsequently extrapolated to field-scale operation using fixed assumptions for treatment volume and operational frequency. Actual field performance may differ due to seasonal rainfall variability, fluctuating flow rates, incomplete reagent mixing, sludge settling efficiency, and operational interruptions, all of which are recognized as important uncertainties in AMD treatment systems [3], [11], [24]. Second, the economic valuation adopted deterministic assumptions for reagent price, transportation cost, treatment frequency, and commodity selling price. Consequently, the reported environmental liability values should be interpreted as scenario-based management estimates rather than precise long-term financial forecasts. Previous environmental liability studies have shown that remediation cost may vary substantially under changing operational and market conditions [13], [15], [16]. Third, the present study evaluated sludge generation primarily from operational handling requirements and did not include long-term geochemical stability assessment of $BaSO_4$-bearing or ettringite-bearing residues. Future research should therefore incorporate residue leaching assessment, pilot-scale operational validation, and probabilistic cost modeling to strengthen the long-term applicability of the IGHRA framework.

4. Conclusion

This study shows that the IGHRA framework can assess acid mine drainage liability as a measurable link between pollution burden, compliance status, treatment performance, and environmental cost. The liability basis indicates that the main hotspot occurs at L3, where severe acidity, high dissolved Fe, and high sulfate create the highest restoration burden. Among the evaluated treatment scenarios, CaO alone lowers pollution load effectively but does not achieve full compliance because sulfate remains above the regulatory standard. In contrast, the CaO 10 g/L and $BaCl_2$ 3.0 g/L scenario achieves full compliance and provides the most economical restoration cost among the compliant alternatives, with an annual cost of IDR 516,463,856.

The study also shows that environmental liability at the site does not end with water quality restoration. The five-year preventive expenditure cost reaches IDR 3,832,950,186 because the mine must continue to manage potentially acid-forming material through selective handling, temporary stockpiling, cover placement, and runoff control. When restoration and prevention are evaluated together, the compliant CaO and $BaCl_2$ scenario exerts an environmental cost pressure of 8.58% on projected gross revenue.

References:

[1] P. C. Singer and W. Stumm, 'Acidic Mine Drainage: The Rate-Determining Step', *Science*, vol. 167, no. 3921, pp. 1121–1123, Feb. 1970, doi: 10.1126/SCIENCE.167.3921.1121.

[2] M. C. Moncur, J. L. Jambor, C. J. Ptacek, and D. W. Blowes, 'Mine drainage from the weathering of sulfide minerals and magnetite', *Appl. Geochemistry*, vol. 24, no. 12, pp. 2362–2373, Dec. 2009, doi: 10.1016/J.APGEOCHEM.2009.09.013.

[3] K. K. Kefeni, T. A. M. Msagati, and B. B. Mamba, 'Acid mine drainage: Prevention, treatment options, and resource recovery: A review', *J. Clean. Prod.*, vol. 151, pp. 475–493, 2017, doi: 10.1016/j.jclepro.2017.03.082.

[4] J. Pearce, P. Weber, S. Pearce, and P. Scott, 'Acid and metalliferous drainage contaminant load prediction for operational or legacy mines at closure', *Proc. 11th Int. Conf. Mine Clos.*, pp. 663–676, 2016, doi: 10.36487/acg_rep/1608_49_pearce.

[5] D. E. Andini and R. S. Gautama, 'Prediction Potential Acid Mine Drainage of Epithermal High Sulfidation Deposits using Static Test', *IOP Conf. Ser. Earth Environ. Sci.*, vol. 353, no. 1, p. 012023, Oct. 2019, doi: 10.1088/1755-1315/353/1/012023.

[6] Canadian Council of Ministers of the Environment, *Canadian Water Quality Guidelines for the Protection of Aquatic Life: CCME Water Quality Index (WQI), User's Manual (2017 update)*, 2017.

[7] Drinking Water Inspectorate, *Compliance Risk Index (CRI): Definition*, 2018.

[8] S. Chidiac, P. El Najjar, N. Ouaini, Y. El Rayess, and D. El Azzi, *A comprehensive review of water quality indices (WQIs): history, models, attempts and perspectives*, vol. 22, no. 2. Springer Netherlands, 2023. doi: 10.1007/s11157-023-09650-7.

[9] L. Akpan, A. C. Tse, F. Dumbari Giadom, and C. I. Adamu, 'Chemical Characteristics of Discharges from Two Derelict Coal Mine Sites in Enugu Nigeria: Implication for Pollution and Acid Mine Drainage', *J. Min. Environ.*, vol. 12, no. 1, pp. 89–111, 2021, doi: 10.22044/jme.2020.10181.1956.

[10] C. A. Marove, R. Sotozono, P. Tangviroon, C. B. Tabelin, and T. Igarashi, 'Assessment of soil, sediment and water contaminations around open-pit coal mines in Moatize, Tete province, Mozambique', *Environ. Adv.*, vol. 8, p. 100215, Jul. 2022, doi: 10.1016/J.ENVADV.2022.100215.

[11] J. G. Skousen, P. F. Ziemkiewicz, and L. M. McDonald, 'Acid mine drainage formation, control and treatment: Approaches and strategies', *Extr. Ind. Soc.*, vol. 6, no. 1, pp. 241–249, Jan. 2019, doi: 10.1016/J.EXIS.2018.09.008.

[12] V. Rajaram, A. Glazer, and G. Coghlan, 'Methodology for Estimating the Costs of Treatment of Mine Drainage', 2001.

[13] J. Ossa-moreno *et al.*, 'The Hydro-economics of Mining', *Ecol. Econ.*, vol. 145, no. November 2017, pp. 368–379, 2018, doi: 10.1016/j.ecolecon.2017.11.010.

[14] J. S. Adiansyah, M. Rosano, W. Biswas, and N. Haque, 'Life cycle cost estimation and environmental valuation of coal mine tailings management', *J. Sustain. Min.*, vol. 16, no. 3, pp. 114–125, 2017, doi: 10.1016/j.jsm.2017.10.004.

[15] M. Czajkowski, N. Meade, R. Seroa da Motta, R. A. Ortiz, M. Welsh, and G. C. Blanc, 'Estimating environmental and cultural/heritage damages of a tailings dam failure: The case of the Fundão dam in Brazil', *J. Environ. Econ. Manage.*, vol. 121, no. July, 2023, doi: 10.1016/j.jeem.2023.102849.

[16] S. A. Leyton-flor, K. Sangha, and K. Howey, 'The Extractive Industries and Society Assessing environmental liabilities of mining in Northern Australia : A Case study of the McArthur River Mine', *Extr. Ind. Soc.*, vol. 20, no. October, p. 101562, 2024, doi: 10.1016/j.exis.2024.101562.

[17] L. Hakanson, 'An ecological risk index for aquatic pollution control.a sedimentological

approach', *Water Res.*, vol. 14, no. 8, pp. 975–1001, Jan. 1980, doi: 10.1016/0043-1354(80)90143-8.
[18] D. L. Tomlinson, J. G. Wilson, C. R. Harris, and D. W. Jeffrey, 'Problems in the assessment of heavy-metal levels in estuaries and the formation of a pollution index', *Helgoländer Meeresuntersuchungen*, vol. 33, no. 1–4, pp. 566–575, Mar. 1980, doi: 10.1007/BF02414780/METRICS.
[19] E. T. Tolonen, A. Sarpola, T. Hu, J. Rämö, and U. Lassi, 'Acid mine drainage treatment using by-products from quicklime manufacturing as neutralization chemicals', *Chemosphere*, vol. 117, no. 1, pp. 419–424, Dec. 2014, doi: 10.1016/J.CHEMOSPHERE.2014.07.090.
[20] E. T. Tolonen, J. Rämö, and U. Lassi, 'The effect of magnesium on partial sulphate removal from mine water as gypsum', *J. Environ. Manage.*, vol. 159, pp. 143–146, Aug. 2015, doi: 10.1016/J.JENVMAN.2015.05.009.
[21] E. Torres, A. Lozano, F. Macías, A. Gomez-Arias, J. Castillo, and C. Ayora, 'Passive elimination of sulfate and metals from acid mine drainage using combined limestone and barium carbonate systems', *J. Clean. Prod.*, vol. 182, pp. 114–123, May 2018, doi: 10.1016/J.JCLEPRO.2018.01.224.
[22] W. Dou *et al.*, 'Sulfate removal from wastewater using ettringite precipitation: Magnesium ion inhibition and process optimization', *J. Environ. Manage.*, vol. 196, pp. 518–526, Jul. 2017, doi: 10.1016/J.JENVMAN.2017.03.054.
[23] A. Iizuka, H. J. Ho, T. Sasaki, H. Yoshida, Y. Hayakawa, and A. Yamasaki, 'Comparative study of acid mine drainage neutralization by calcium hydroxide and concrete sludge–derived material', *Miner. Eng.*, vol. 188, p. 107819, Oct. 2022, doi: 10.1016/J.MINENG.2022.107819.
[24] A. Chatla *et al.*, 'Sulphate removal from aqueous solutions: State-of-the-art technologies and future research trends', *Desalination*, vol. 558, p. 116615, Jul. 2023, doi: 10.1016/J.DESAL.2023.116615.
[25] Government Regulation of the Republic of Indonesia No. 22 of 2021 concerning Environmental Protection and Management Implementation.
[26] K. K. Kefeni, T. M. Msagati, J. P. Maree, and B. B. Mamba, "Metals and sulphate removal from acid mine drainage in two steps via ferrite sludge and barium sulphate formation," *Minerals Engineering*, vol. 81, pp. 79–87, Oct. 2015, doi: 10.1016/j.mineng.2015.07.016.