Science of the Total Environment 975 (2025) 179270

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Potential generation and consumption of carbon dioxide during treatment of mine drainages in South Korea

Duk-Min Kim [a,*], Ki-Rim Lee [a], Mi-Sun Park [b]

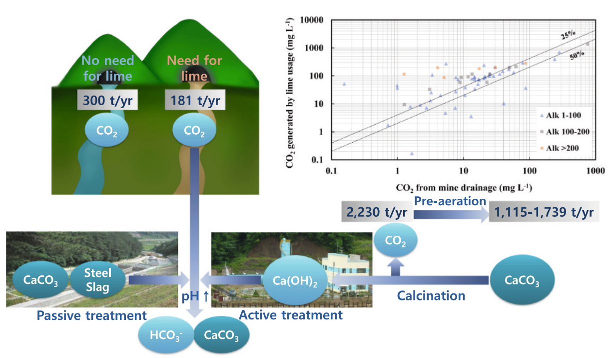
[a] *Department of New Energy and Mining Engineering, Sangji University, Wonju, Gangwon state 26339, Republic of Korea*

[b] *Research Institute of Technology, Korea Mine Rehabilitation and Mineral Resources Corporation, Wonju, Gangwon state 26339, Republic of Korea*

HIGHLIGHTS

- Mine drainages with lower pH and higher alkalinity exhibited higher CO_2 emissions.
- Coal mines accounted for 95 % of CO_2 emissions from mine drainage in South Korea.
- CO_2 from potential lime treatment was 12 times the amount of CO_2 from mine drainages.
- Passive treatment is substantially more beneficial than active treatment using lime.
- Particularly mine drainages with high pH and alkalinity preferred pre-aeration.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Kuishuang Feng

Keywords:
Carbon footprint
CO_2 emission
PHREEQ-N-AMDTreat
Hydrated lime
Pre-aeration

ABSTRACT

Mine drainage often acts as a source of CO_2-emissions due to pyrite oxidation and the associated production of H^+, which promotes the dissolution of carbonate minerals. While the treatment of mine drainage with hydrated lime is a common practice to increase pH, the production of lime generates a considerable amount of CO_2. In this study, direct CO_2 emissions from mine drainages and indirect CO_2 emissions from the potential consumption of hydrated lime were modeled using PHREEQ-N-AMDTreat based on chemical compositions and flow rates at most mine drainage sites (n = 395) across South Korea. The total potential CO_2 emissions from the mine drainages were estimated at 481 t yr^{-1}, with 95 % originating from coal mines. Mine drainages with lower pH and/or higher alkalinity generally exhibited higher CO_2 emissions. In contrast, the potential consumption of hydrated lime to treat all sampled mine drainages could generate 2230 t CO_2 yr^{-1}, which was >12 times the CO_2 degassed from the drainages, assuming atmospheric equilibrium under surface conditions. Therefore, when considering CO_2 emissions, passive treatment methods are substantially more advantageous than (semi-)active treatment methods using hydrated lime. The estimated CO_2 emissions from most mine drainages were <13 % of the indirect CO_2 emissions attributed to hydrated lime usage. Since the ratio (13 %) is lower than the reported reduction of hydrated lime consumption (22 %–50 %) achieved through pre-aeration in treatment processes, implementing pre-aeration is a preferable approach for most mine drainages from the perspective of CO_2 emission reduction.

* Corresponding author at: Department of New Energy and Mining Engineering, Sangji University, Wonju 26339, Republic of Korea.
E-mail address: kdukmin8@sangji.ac.kr (D.-M. Kim).

https://doi.org/10.1016/j.scitotenv.2025.179270
Received 13 June 2024; Received in revised form 8 February 2025; Accepted 26 March 2025
Available online 1 April 2025
0048-9697/© 2025 Elsevier B.V. All rights reserved, including those for text and data mining, AI training, and similar technologies.

1. Introduction

Groundwater discharged from mines is often oversaturated with carbon dioxide (CO_2) (Rose and Cravotta, 1998; Kruse and Strosnider, 2015; Brown et al., 2024), leading to degassing into the atmosphere under surface conditions (Kirby and Cravotta, 2005). Thus, mine drainage can serve as an important source of greenhouse gas emissions (e.g., Raymond and Oh, 2009; Vesper et al., 2016).

Elevated dissolved CO_2 in mine drainage generally originates from the dissolution of calcite ($CaCO_3$) and other carbonate minerals by H^+ (Eq. (1)).

$$CaCO_3 + 2H^+ \rightarrow Ca^{2+} + H_2CO_3^* \tag{1}$$

where H^+ is primarily produced during pyrite oxidation, and carbonic acid (H_2CO_3*) primarily consists of $CO_{2(aq)}$. Additionally, as pH increases, $CO_{2(aq)}$ is converted to other dissolved inorganic carbon (DIC) species, such as bicarbonate (HCO_3^-) (Eq. (2); Kirby and Cravotta, 2005).

$$CO_{2(aq)} + OH^- \rightarrow HCO_3^- \tag{2}$$

Meanwhile, one of the most commonly used alkaline agents for increasing the pH of mine drainage is hydrated lime ($Ca(OH)_2$), primarily due to its low cost and high coagulation efficiency in treating metal contaminant ions. Additionally, the removal of $CO_{2(aq)}$ can be considered a beneficial side effect of using alkaline agents.

Hydrated lime is produced through the calcination of limestone (Eqs. (3) and (4)).

$$CaCO_3 \rightarrow CaO + CO_{2(g)} \tag{3}$$

$$CaO + H_2O \rightarrow Ca(OH)_2 \tag{4}$$

where 1 mol of CO_2 gas is generated for 1 mol of hydrated lime produced. In summary, dissolved CO_2 can be directly degassed from mine drainage, and increasing the pH of the drainage can help reduce CO_2 emissions. However, if the drainage is treated with hydrated lime, additional CO_2 may be indirectly generated during the lime production process.

Dissolved CO_2 and the corresponding emissions from mine drainage have been reported at both watershed and regional scales. Raymond and Oh (2009) estimated that ~11 Mt CO_2 was degassed due to acid mine drainage in the Susquehanna watershed in Pennsylvania over a century, averaging ~0.11 Mt CO_2 yr^{-1}. Similarly, Vesper et al. (2016) reported CO_2 flux of 0.076 Mt CO_2 yr^{-1} from 140 mine drainages in Pennsylvania, a value comparable to the annual emissions from a small coal-fired power plant. More recently, Brown et al. (2024) assessed CO_2 emissions from 16 mine drainages in Scotland, estimating a mean CO_2 concentration of 235 mg L^{-1} and a total flux of 0.078 Mt CO_{2eq} yr^{-1} from all coal mines in Scotland. This value represented approximately 0.2 % of Scotland's total contribution to global CO_{2eq} emissions.

In 2019, approximately 24 % (14 Gt CO_{2eq}) of global anthropogenic greenhouse gas emissions originated from industrial sources, with lime production ranking as the second-largest industrial source after cement production (Shan et al., 2016; IPCC, 2021; Bing et al., 2023). Lime dosages have been studied and modeled in various mine drainage treatment facilities (Cravotta, 2021; Kim et al., 2023), and life-cycle assessment studies have indicated substantially lower CO_2 emissions associated with passive treatment compared to active treatment using lime (Tuazon and Corder, 2008; Hengen et al., 2014). Unlike active and semi-active methods (e.g., automated lime dosing systems that operates without labor, often accompanied by a large settling pond), which require continuous input of chemicals or energy, passive treatment systems function without such inputs (Younger et al., 2002). Specifically, to increase pH, passive systems typically utilize limestone or steel slag within SAPS (Successive Alkalinity Producing Systems), slag reactors, or SLBs (Slag Leach Beds), whereas active treatment systems rely heavily on hydrated lime. Despite the growing awareness of CO_2 emissions from industrial lime use, the potential effects of different mine drainage treatment strategies on CO_2 emissions—including the conversion of CO_2 to other species due to pH increase and the indirect CO_2 emissions from lime production—remain largely unassessed. To the best of the authors' knowledge, this gap highlights a critical need for further research.

Thus, the objectives of this study are (1) to calculate CO_2 emissions from most mine drainages in South Korea and evaluate the effect of CO_2 removal through pH elevation, (2) to estimate indirect CO_2 emissions associated with the use of hydrated lime for treating these drainages, and (3) to assess the suitability of pre-aeration as a mine drainage treatment strategy, focusing on its potential to reduce lime dosage and associated CO_2 emissions. A conceptual flow diagram illustrating these objectives is presented in Fig. 1.

2. Methods

2.1. Survey on mine drainages

In 2016, the Korea Mine Rehabilitation and Mineral Resources Corporation (KOMIR) investigated 395 mine drainages across South Korea. Mine drainages with relatively high contamination and/or flow rates were surveyed four times a year, while those with lower contamination and/or flow rates were surveyed twice a year. The majority of the mine drainages were adit discharges, while some were leachates from dumps of waste rock or tailings.

The temperature, pH, and dissolved oxygen (DO) concentrations of the water samples were measured using a portable meter (D-55, Horiba, Kyoto, Japan), calibrated with standard solutions or air calibration (for DO) at the time of each survey or experiment. Water samples were filtered through 0.45-μm membrane filters. Alkalinity was then measured in the field using a digital titrator (16900, Hach, Loveland, CO, USA), and the dissolved Fe^{2+} concentration was determined using a portable colorimeter (DR-890, Hach) following the phenanthroline method (APHA, 2017). For cation and anion analyses, samples were collected in 100-mL PE bottles. Cation analysis samples were preserved by adding a few drops of concentrated HNO_3 to maintain a pH of <2. These water samples were stored at 4 °C until further analysis. Additionally, mine drainage flow rates were measured using the bucket-and-stopwatch method, with each measurement repeated three times to calculate an average value.

Cations were analyzed by inductively coupled plasma optical emission spectroscopy (ICP-OES; Varian 720-ES, Agilent Technologies Inc., Palo Alto, CA, USA) at KOMIR, Wonju. Anions were analyzed by ion chromatography (model 850, Metrohm, Herisau, Switzerland) at KOMIR. For both ICP-OES and ion chromatography, the relative standard deviations were <5 % of the measured value.

2.2. Data analysis

Seasonal water quality and quantity data of the 395 mine drainages were assessed (Table 1 and Fig. S1). Dissolved CO_2 concentrations and the dosages of hydrated lime required to reach specified target pH values were assessed using the Caustic Titration module of PHREEQ-N-AMDTreat version 1.4.5 (Cravotta, 2020, 2021). As phosphate concentrations were negligible in all samples, and the samples used for alkalinity determination excluded suspended solids, the measured alkalinity values could be used with minimal error to calculate DIC species, including CO_2. The Caustic Titration tool has three modeling options for computing the treatment to specified target pH values using hydrated lime or other caustic chemicals: (1) Not aerated, (2) Pre-aerated, and (3) Aerated to equilibrium. For this study, the aeration to equilibrium condition with a steady-state partial pressure of CO_2 (pCO_2) of $10^{-3.4}$ atm was applied. Chemical compositions of the mine drainages, including pH, temperature, alkalinity, and concentrations of cations and anions, were input into the module. The option to compute total inorganic carbon (TIC) from the input alkalinity, pH, and temperature was selected. When assessing dissolved CO_2 concentrations in mine drainage, samples with a pH lower than 4.5 or without alkalinity were excluded. Additionally, as the current scope of PHREEQ-N-AMDTreat does not encompass Cu and Zn concentrations, Cu and Zn concentrations exceeding 1 mg L^{-1} in this study were converted to equivalent Fe^{3+} concentrations by applying different atomic weights and valences. These were then added to the existing Fe^{3+} concentrations (Kim et al., 2023). For all simulations, hydrated lime was selected as the alkaline agent, with default purity and mixing efficiency factors of 0.99 and 0.8, respectively. This indicates that the $Ca(OH)_2$ content in the hydrated lime was 99 %, and 20 % of the hydrated lime remained undissolved and accumulated with sludge (Kim et al., 2023).

To predict maximum CO_2 concentrations that could potentially be degassed from mine drainages by natural aeration ($C_{CO2(degas)}$), dissolved CO_2 concentration at the equilibrium pCO_2 of $10^{-3.4}$ atm ($C_{CO2(eq)}$) was subtracted from the modeled initial concentration of dissolved CO_2 ($C_{CO2(dissolved)}$) (Eq. (5)). The CO_2 flux ($F_{CO2(degas)}$) was then computed by multiplying this difference in concentration by the cumulative annual flow rate expressed in L yr^{-1} (Q_{annual}).

$$F_{CO_2(degas)} = C_{CO_2(degas)} \times Q_{annual} = \left(C_{CO_2(dissolved)} - C_{CO_2(eq)}\right) \times Q_{annual} \quad (5)$$

The mine drainages were classified into two categories: (1) potential (semi-)active treatment with hydrated lime, which includes 11 (semi-) active treatment facilities under operation, and (2) others (indicated as "Others" in the relevant plots), which include (a) mine drainages expected to meet discharge criteria in South Korea (excluding arsenic and fluoride) after aeration, (b) mine drainages being successfully treated by passive treatment facilities, and (c) stagnant mine drainages without surface flow.

To simulate the characteristics of treated effluent, the target pH was varied based on treatment goals. The target pH was determined using the following criteria, based on the composition of mine drainage and reported coprecipitation–adsorption behavior (Kim et al., 2022a): (1) Al: pH of 6, to exceed the minimum pH value of the discharge standard in the Republic of Korea (>5.8); (2) Fe: pH of 7.5 (Fe $<$ 20 mg L^{-1}), 8.0 (Fe 20–100 mg L^{-1}), and 8.5 (Fe $>$ 100 mg L^{-1}), considering Fe^{2+} oxidation rate according to pH; (3) Mn: pH of 9.5 (Mn $<$ 10 mg L^{-1}) and 10.0 (Mn $\geq$ 10 mg L^{-1}) considering oxidation rate according to pH, but coprecipitation–adsorption effect was also considered based on Fe and Al concentrations (Kim et al., 2022a); (4) Zn: pH of 8.5; and (5) Ni: pH of 8.5 or 9.0, considering coprecipitation–adsorption by Fe.

After the geochemical modeling of each seasonal data subset which is described above, annual averages were used for each mine drainage site ($n = 395$).

2.3. Calculation of CO_2 emission from hydrated lime consumption

Emissions of CO_2 from hydrated lime production facilities, categorized as Scope 1 emissions (direct greenhouse gas emissions from sources controlled or owned by the organization), were examined. Approximately 60 % of total CO_{2eq} emissions result from the decomposition (calcination) reaction, 39 % from fuel combustion, and only 1 % from electricity consumption at the plant (European Lime Association, 2019, 2021; Laveglia et al., 2022). To focus on the primary and direct sources of CO_2 emissions, calcination and fuel combustion were included, while electricity consumption and limestone quarrying were excluded from the carbon budget.

To convert the amount of quicklime (CaO) to CO_2 generation, an emission factor of 0.75 for lime during thermal decomposition ($EF_{Lime(decom)}$) was applied. This factor was derived from the stoichiometric ratio (*SR*) of CO_2 to CaO (0.785), based on Eq. (3), and adjusted for the purity (*P*) of quicklime at 0.95 (IPCC, 2006; IPCC, 2019; GGIRC, 2022). Subsequently, $EF_{Lime(decom)}$ was multiplied by a conversion factor (CF) for hydrated lime from quicklime (0.757) to obtain the final emission factor for hydrated lime ($EF_{HL(decom)}$) of 0.57 during thermal decomposition (Eq. (6); IPCC, 2006, 2019).

$$EF_{HL(decom)} = EF_{Lime(decom)} \times CF = SR \times P \times CF \quad (6)$$

Moreover, CO_2 generation from fossil fuel combustion during the calcination of limestone was added to the total CO_2 emission (Shan et al., 2016; Laveglia et al., 2022; Wu et al., 2023). Shan et al. (2016) reported mass ratios ($R_{C\text{-}L}$) of coal consumption (M_{Coal}) to lime production (M_{Lime}) ranging from 0.12 to 0.16 in China, with a weighted average of 0.15. The emission factor for coal combustion (EF_{Coal}) was 1.85 t CO_2 per t coal (Shan et al., 2016), resulting in an emission factor for lime during fuel combustion ($EF_{Lime(coal)}$) of 0.27 t CO_2 per t lime (Eq. (7)). By applying the conversion factor (CF) from quicklime to hydrated lime, we calculated the emission factor for hydrated lime during fuel combustion ($EF_{HL(coal)}$) to be 0.21 t CO_2 per t hydrated lime.

$$EF_{HL(coal)} = EF_{Lime(coal)} \times CF = R_{C-L} \times EF_{Coal} \times CF = \frac{M_{Coal}}{M_{Lime}} \times EF_{Coal} \times CF \quad (7)$$

Thus, summing the emission factors for hydrated lime during thermal decomposition ($EF_{HL(decom)}$, 0.57) and coal combustion ($EF_{HL(coal)}$, 0.21) yields 0.78 t of CO_2 directly generated per tonne of hydrated lime produced. When considering the total CO_{2eq} emissions over the entire production process, Laveglia et al. (2022) calculated 0.94 t of CO_{2eq} during hydrated lime production in four EU countries. Additionally, Wu et al. (2023) estimated 0.89 t of CO_{2eq} during hydrated lime production in the Yangtze River basin in China. Therefore, the CO_2 directly

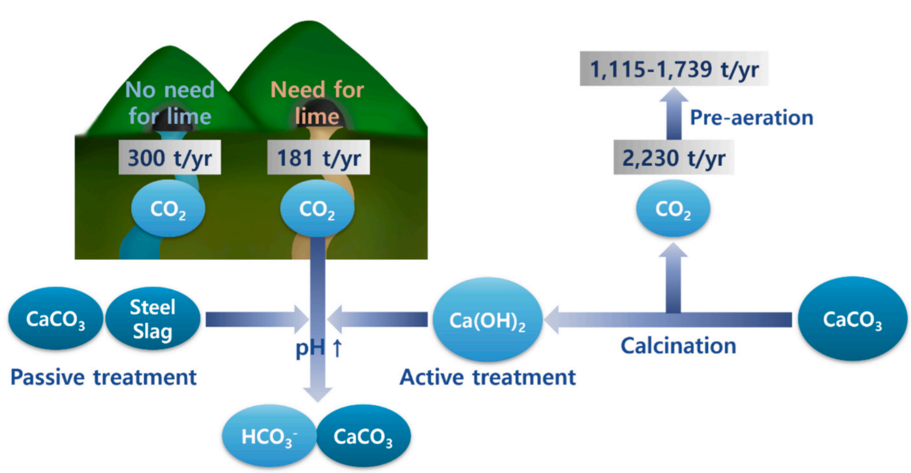


Fig. 1. Conceptual flow diagram illustrating this study.

generated during hydrated lime production ($C_{CO2(HL)}$) was calculated by multiplying the hydrated lime dosage (D_{HL}) by the emission factor for hydrated lime ($EF_{HL} = 0.78$) (Eq. (8)).

$$C_{CO2(HL)} = D_{HL} \times EF_{HL} = D_{HL} \times 0.78 \quad (8)$$

3. Results and discussion

3.1. Potential CO_2 emissions from mine drainages

Modeled concentrations of potential CO_2 degassing from mine drainages are plotted against the pH and alkalinity of untreated mine drainages in Fig. 2. The logarithm of the potential CO_2 concentrations degassing from mine drainages exhibited a predominantly linear negative relationship with pH within each alkalinity range (Fig. 2a). Similarly, within each pH range, the logarithm of CO_2 concentrations demonstrated a linear positive relationship with the logarithm of alkalinity (Fig. 2b). Equilibrated dissolved CO_2 concentrations after aeration in the model were very low, averaging 0.80 mg L^{-1} and reaching a maximum of 1.36 mg L^{-1} for the mine drainages (Fig. S1). These results indicate that dissolved CO_2 concentrations in mine drainages are the primary determinant of potential CO_2 degassing. The observed relationships align with the theoretical relationships of dissolved CO_2 with pH and alkalinity in Eq. (9). An increase in H^+ and HCO_3^- concentrations leads to a corresponding increase in dissolved CO_2 concentrations:

$$CO_{2(aq)} + H_2O \leftrightarrow H_2CO_3{}^0 \leftrightarrow HCO_3{}^- + H^+ \quad (9)$$

Therefore, mine drainages with low pH and high alkalinity exhibit a higher potential for CO_2 emissions.

Furthermore, the origins of the drainages are classified into coal and metal mines in Fig. 3. In South Korea, drainages from coal mines generally exhibit lower pH than those from metal mines (Figs. 3a and S1a), while alkalinity levels are comparable between the two (Figs. 3b and S1b). This difference in pH can be attributed to the presence of framboidal pyrite—micron- or submicron-sized crystals with very high specific surface areas—which produces acid at much higher rates than larger pyrite grains and is commonly found in sedimentary strata, such as those associated with coal mines (Caruccio, 1975; Kim et al., 2017). In contrast, most metal mines in South Korea primarily excavated gold from ores with low pyrite content. These geological factors contribute to the lower pH observed in coal mine drainages. Consequently, mine drainages with high potential CO_2 concentrations available for

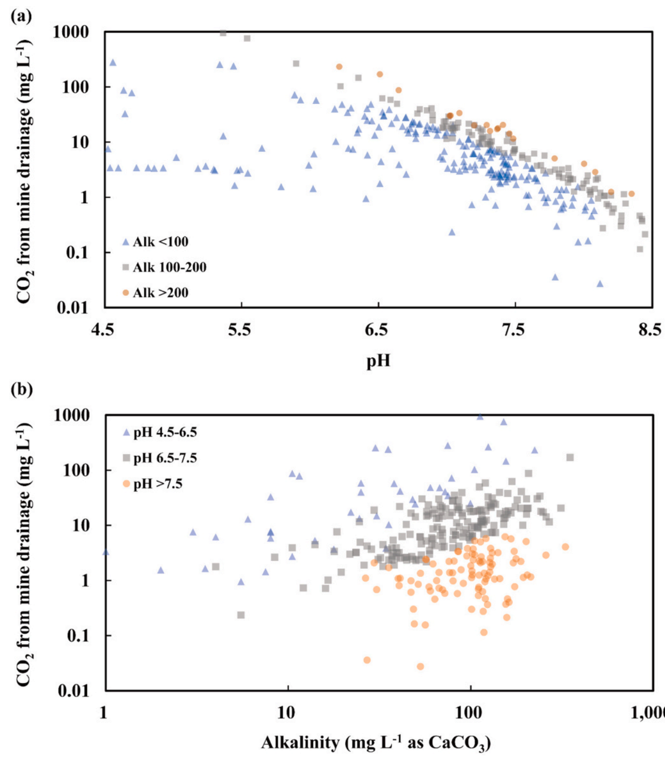


Fig. 2. Relationship between modeled concentrations of potentially degassed CO_2 and (a) pH and (b) alkalinity in untreated mine drainages in South Korea. The samples are classified by alkalinity and pH ranges.

degassing, driven by their low pH, were predominantly classified as coal mine drainage (Figs. 3a and S1c).

Furthermore, the flow rates of coal mine drainages were generally higher than those of metal mine drainages in South Korea. This disparity is attributed to the extensive excavation of underground tunnels and the high hydraulic conductivity of sedimentary strata, such as sandstone and limestone, commonly found in coal mining areas. As a result, the potential CO_2 flux (kg d^{-1}) from drainage became even higher for coal mines than for metal mines (Figs. 4 and S1d). Notably, all mine drainages with potential CO_2 flux exceeding 30 kg d^{-1} were from 11 coal

Table 1
Statistics for flow rate and water quality of analyzed mine drainages.

Item	Unit	Statistics	Coal mines		Metal mines		Total
			Potential lime use	Others	Potential lime use	Others	
Number	(ea)		94	138	25	138	395
Flow rate	(m^3 d^{-1})	Avg.	286	221	82	25	159
		Min.	0.25	0	1.8	0	0
		Max.	6006	3725	270	430	6006
		Sum	26,844	30,429	2052	3477	62,802
pH	(–)	Avg.	5.0	7.0	6.4	7.5	6.7
		Min.	3.0	3.5	2.8	2.5	2.5
		Max.	8.2	8.4	8.0	8.7	8.7
Alkalinity	(mg L^{-1} as $CaCO_3$)	Avg.	34	96	70	78	73
		Min.	0	0	0	0	0
		Max.	330	350	207	258	350
Potential CO_2 degassed from mine drainage	(mg L^{-1})	Avg.	14	27	52	4.5	18
		Min.	−0.1[a]	0.04	0.2	−0.2	−0.2
		Max.	240	953	759	59	953
		Sum	1329	3790	1298	622	7040
CO_2 generated by lime production	(mg L^{-1})	Avg.	240	–[b]	190	–	230
		Min.	0.2	–	1.7	–	0.2
		Max.	3632	–	1354	–	3632
		Sum	22,589	–	4749	–	27,339

[a] Negative value indicates that atmospheric CO_2 dissolves into the mine drainage until equilibrium is reached.
[b] Not applicable.

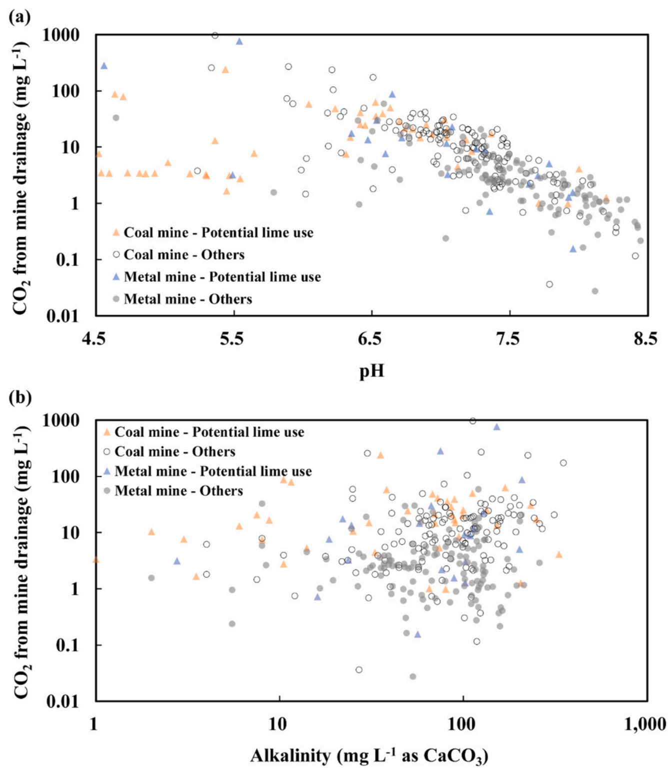


Fig. 3. Relationship between modeled concentrations of potentially degassed CO_2 and (a) pH and (b) alkalinity in untreated mine drainages in South Korea. The samples are classified by mine types and potential of (semi-)active treatment using hydrated lime.

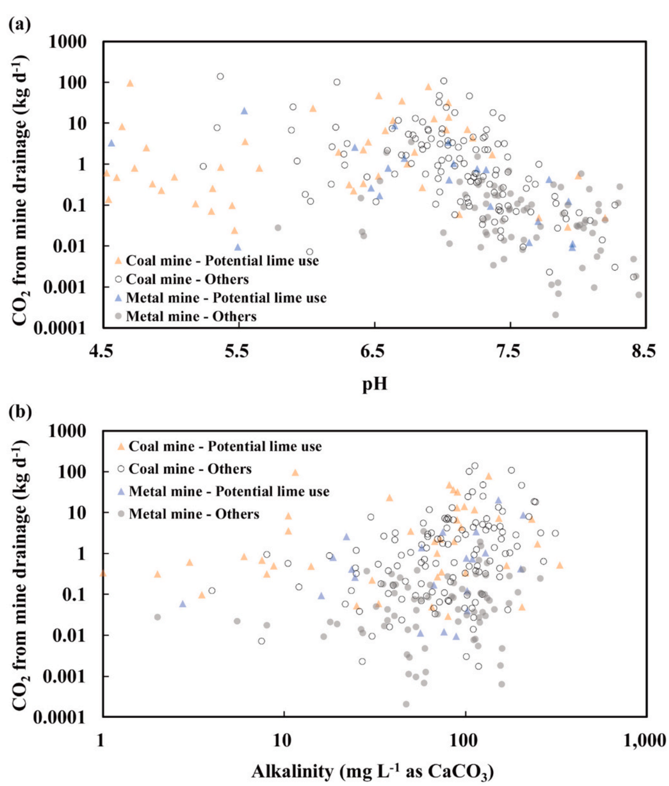


Fig. 4. Relationship between modeled potential flux of CO_2 from untreated mine drainages and (a) pH and (b) alkalinity in mine drainages in South Korea. The samples are classified by mine types and potential of (semi-)active treatment using hydrated lime.

mines (Fig. 4).

Additionally, mine drainages with the potential for (semi-)active treatment using hydrated lime exhibited lower pH levels compared to other drainages, which was evident in mine drainages with a pH below 5.5 (Figs. 3 and 4).

Furthermore, the pH and alkalinity of mine drainages from limestone strata were compared to those from other geological formations, as these parameters were found to determine potential CO_2 emissions. However, a *t*-test revealed no significant differences in generated CO_2 concentrations between the two groups (Table S1). The F-test for variance indicated a *p*-value much lower than 0.05, suggesting a significant difference in variance (heterogeneous variance) at the 95 % confidence level. Given this heterogeneity, the *t*-test was conducted accordingly, and the resulting p-value was much higher than 0.05, confirming that the average potential CO_2 emissions were not significantly different between the two groups. For instance, the Buguk coal mine drainage exhibited a high alkalinity of 330 mg L^{-1} as $CaCO_3$ and a pH of 8.0. Despite of its high alkalinity, this mine is located in strata with only intermittent limestone presence, rather than a regional limestone formation. Similarly, the Okdong metal mine drainage demonstrated a high alkalinity of 151 mg L^{-1} as $CaCO_3$, even though it is situated far from any limestone strata. This apparent discrepancy between alkalinity and limestone geology may be attributed to localized controls of pH and alkalinity in mine drainages. Specifically, these controls are influenced by the degree of pyrite oxidation and subsequent neutralization by surrounding rocks containing various forms of carbonates, rather than exclusively by limestone formations.

Additionally, potential CO_2 emissions from leachates originating from waste rock or tailings dumps ($n = 15$) were compared to those from adit discharges (Table S2). The F-test for variance yielded a p-value much lower than 0.05, confirming heterogeneous variance. Subsequently, the *t*-test was conducted with this consideration, and the resulting p-value was also lower than 0.05, indicating that the average potential CO_2 emissions were significantly different between the two groups at the 95 % confidence level. Although the number of leachate samples was limited (n = 15), this result suggests that CO_2 degassing during water migration through waste rock or tailings dumps may have reduced the potential for CO_2 emissions upon discharge.

3.2. Decrease in CO_2 concentration by increase in pH

Figs. 2a and 3a demonstrate that degassing CO_2 concentrations, which are closely related to dissolved CO_2 concentrations in mine drainage (as discussed in Section 3.1), decrease significantly with increasing pH. This suggests that mine drainage treatment by increasing pH mitigates CO_2 degassing through conversion to HCO_3^- (Eq. (2)) and/or calcite precipitation (Eq. (10)).

$$Ca^{2+} + HCO_3^- + OH^- \rightarrow CaCO_3 + H_2O \tag{10}$$

Saturation indices (SIs) for calcite in the treated mine drainages were calculated using the Caustic Titration module of PHREEQ-N-AMDTreat, which assumed a default SI value of 0.3 as the threshold required to precipitate calcite from solution. For mine drainage and groundwater, SI values exceeding zero are mostly distributed between 0 and 0.6, with an average of ~0.3 (Plummer et al., 1990; Nordstrom, 2008; Neogi et al., 2017). This oversaturation reflects the inhibition of calcite nucleation and the relatively slow precipitation rate of calcite (Langmuir, 1997). Furthermore, Langmuir (1997) reported that the SI can exceed 0.3 for calcite nucleation and precipitation in the presence of a few mg L^{-1} of DOC which is common in groundwater. Therefore, an SI of 0.3 was selected for calcite precipitation in this study.

The number of samples expected to precipitate calcite was higher than the others for both coal and metal mines (Fig. 5). Thus, calcite precipitation contributed to the DIC consumption and subsequent CO_2 sequestration in most treated samples following the pH increase.

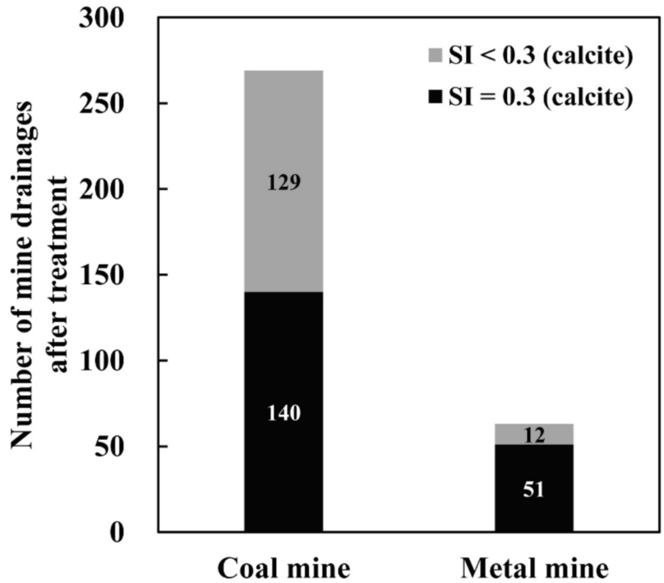


Fig. 5. Number of treated mine drainages with calcite saturation index (SI) of 0.3 or <0.3, which was modeled by the Caustic Titration module of PHREEQ-N-AMDTreat. An SI of 0.3 was assumed as the threshold required for calcite to precipitate from the solution. It should be noted that these numbers represent the number of sampling events and may exceed the number of mine drainages.

3.3. Indirect CO_2 emissions by using hydrated lime for treatment

Indirectly generated CO_2 concentrations were calculated based on the modeled consumption of hydrated lime during treatment. The accuracy of predicting hydrated lime consumption using the same model (Caustic Titration module in PHREEQ-N-AMDTreat) was verified against actual measurements from a full-scale treatment facility in South Korea

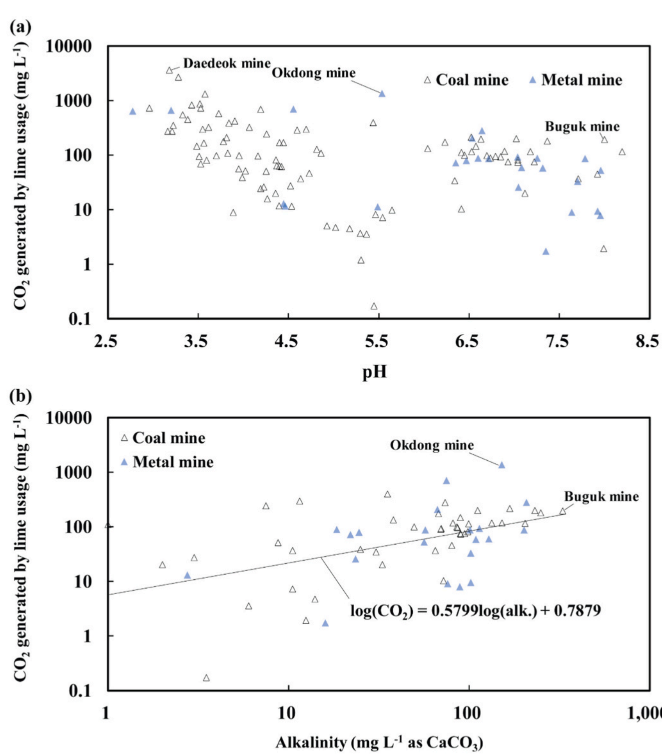


Fig. 6. Relationship between calculated indirect emission (in mg L^{-1}) of CO_2 from modeled usage of hydrated lime and (a) pH and (b) alkalinity in mine drainages. The samples are classified based on mine types.

(Kim et al., 2023). The concentrations and flux of generated CO_2 are plotted against pH and alkalinity (Figs. 6–8). A notable trend was observed at pH values below 5.5, where decreasing pH corresponded to increased acidity and associated lime dosages, resulting in increasing concentrations of generated CO_2 by lime usage (Figs. 6a and 7a). Similarly, at a given alkalinity, sample groups with lower pH values exhibited higher potential CO_2 generation from lime usage (Fig. 7b). Additionally, a positive relationship was observed between CO_2 generation by lime usage and alkalinity of mine drainages (Figs. 6b and 7b). Similarly, at a given pH, sample groups of higher alkalinity exhibited greater CO_2 generation by lime usage (Fig. 7a). These results align with the reported role of HCO_3^- in increasing lime dosage through OH^- consumption and calcite precipitation (Kim et al., 2023). For example, the Okdong mine exhibited the highest CO_2 generation by lime usage among metal mines, at 1354 mg L^{-1} (Fig. 6a). Relatively high alkalinity of 151 mg L^{-1} as $CaCO_3$ at a pH of 5.54 as well as high Mn and Zn concentrations of 20.8–23.5 and 53.4–64.5 mg L^{-1}, respectively, may have contributed to the increase in DIC and lime dosage (Fig. 7a). Similarly, the Buguk mine demonstrated the highest CO_2 generation by lime usage (195 mg L^{-1}) among samples with a pH of ~8.0, due to its remarkably high alkalinity (330 mg L^{-1} as $CaCO_3$) compared to all other drainages (Fig. 7b). In contrast, the Daedeok mine exhibited the highest CO_2 generation from lime usage across all drainages, at 3632 mg L^{-1}. This was due to its remarkably low pH (3.18) and exceptionally high concentrations of Fe (1016 mg L^{-1}) and Al (391 mg L^{-1}), which substantially increased lime dosage requirements.

The positive relationship between alkalinity and the CO_2 flux generated by lime usage (in kg d^{-1}) was less pronounced compared to the relationship with CO_2 concentrations (in mg L^{-1}) (Fig. 8b), as the flow rate influenced the flux calculation.

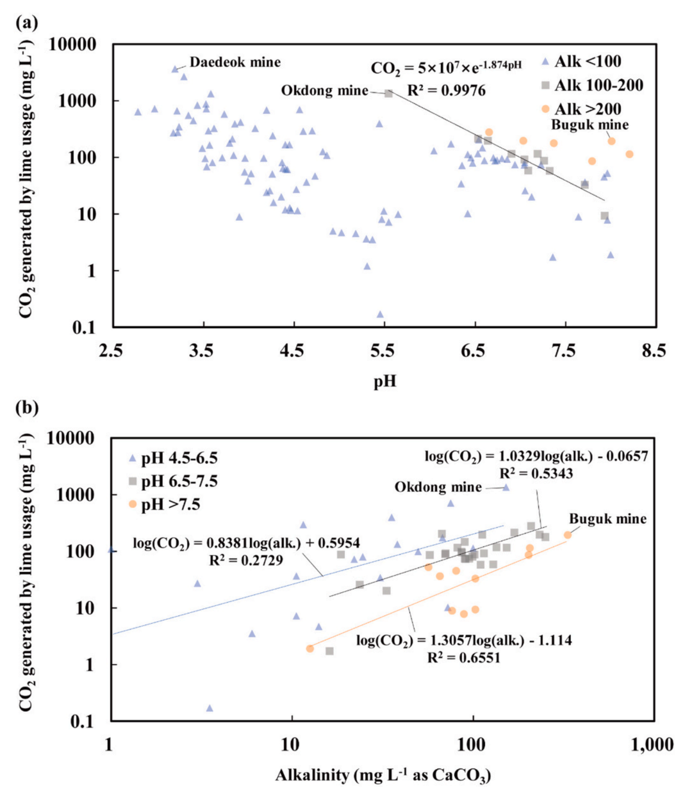


Fig. 7. Relationship between calculated indirect emission (in mg L^{-1}) of CO_2 from modeled usage of hydrated lime and (a) pH and (b) alkalinity in mine drainages. The samples are classified based on alkalinity and pH ranges, and samples with pH <4.5 are excluded in (b). Regression lines correspond to the samples with alkalinities of 100–200 mg L^{-1} as $CaCO_3$ in (a) and to samples with three different pH ranges in (b).

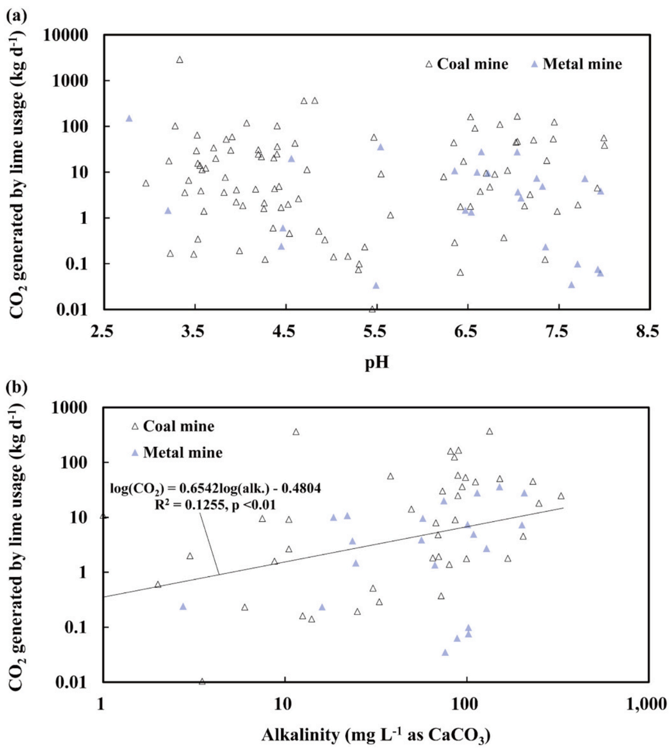


Fig. 8. Relationship between calculated indirect emission flux (in kg d^{-1}) of CO_2 from modeled usage of hydrated lime and (a) pH and (b) alkalinity in mine drainages. The samples are classified based on mine types.

Direct CO_2 emissions from mine drainage were compared to indirect emissions from the use of hydrated lime for treatment (Fig. 9a). The total CO_2 generated from mine drainages by aeration to equilibrium was 481 t yr^{-1}, comprising 459 t yr^{-1} (95 %) from coal mines and 22 t yr^{-1} (5 %) from metal mines in South Korea. Moreover, mine drainages requiring hydrated lime treatment generated 181 t yr^{-1} of CO_2, which can largely be mitigated through pH-increasing treatments (refer to Section 3.2). Nevertheless, indirect CO_2 emissions from hydrated lime usage reached 2230 t yr^{-1}, which was >12 times the amount of CO_2 removed during mine drainage treatment. This result highlights the much larger disadvantage of using hydrated lime compared to the advantage of CO_2 removal from mine drainage. Furthermore, constructing (semi-)active treatment facilities requires substantial amount of cement (Winfrey et al., 2015) and operating these facilities—such as agigation and pumping equipment—consumes electricity. Transportation of hydrated lime further adds to CO_2 emissions. Thus, adopting passive treatment methods offers substantial advantages in reducing the carbon footprint. To achieve satisfactory pH levels in passive treatment systems for treating dissolved metals, alternative substrates such as including steel slag and waste concrete, in addition to limestone, could be considered (Kim et al., 2022b, 2022c; Ho et al., 2023).

Additionally, the potential amount of sludge generation during mine drainage treatment was modeled to be 13,846 t yr^{-1} (Fig. 9b), which should be disposed of as waste or recycled. The accuracy of predicting sludge amounts using the same model (Caustic Titration module in PHREEQ-N-AMDTreat) has been verified through comparing with measurements from a pilot-scale treatment facility in South Korea (Kim et al., 2023). In South Korea, acquiring sites for waste landfills poses challenges, and most sludge from mine drainage treatment facilities is recycled as a supplementary material for cement production. Of the total potential sludge, coal mine drainage treatment contributed 94 % (13,016 t yr^{-1}). Additionally, the sludge generated from coal mine drainage treatment in South Korea typically contains low levels of toxic elements such as As, Cd, Cu, Pb, and Zn (Cui et al., 2011). This makes it a viable candidate for further utilization, such as producing commercial products like arsenic adsorbents (Lee et al., 2018; Kumar et al., 2020) or recovering valuable elements like Al, Mn, and rare earth elements (Vaziri Hassas et al., 2022; Cicek et al., 2023). Meanwhile, for mine drainages that do not require hydrated lime application, sludge generated through natural precipitation was modeled at 3339 t yr^{-1} (Fig. 9b). This sludge can accumulate annually in surface water systems, such as streambeds, especially near coal mines in South Korea.

3.4. *Suitability of pre-aeration step to degas CO_2 and reduce lime consumption*

Pre-aeration to degas CO_2 prior to the treatment of mine drainages has been reported to substantially reduce lime consumption, owing to the decrease of H_2CO_3 (Jageman et al., 1987; Kirby et al., 2009; Kruse and Strosnider, 2015; Means et al., 2015; Hedin and Hedin, 2016; Means and Beam, 2024). This process is typically performed as the initial step in mine drainage treatment facilities and involves bubbling to achieve CO_2 degassing before the addition of alkaline agents (Kirby et al., 2009;

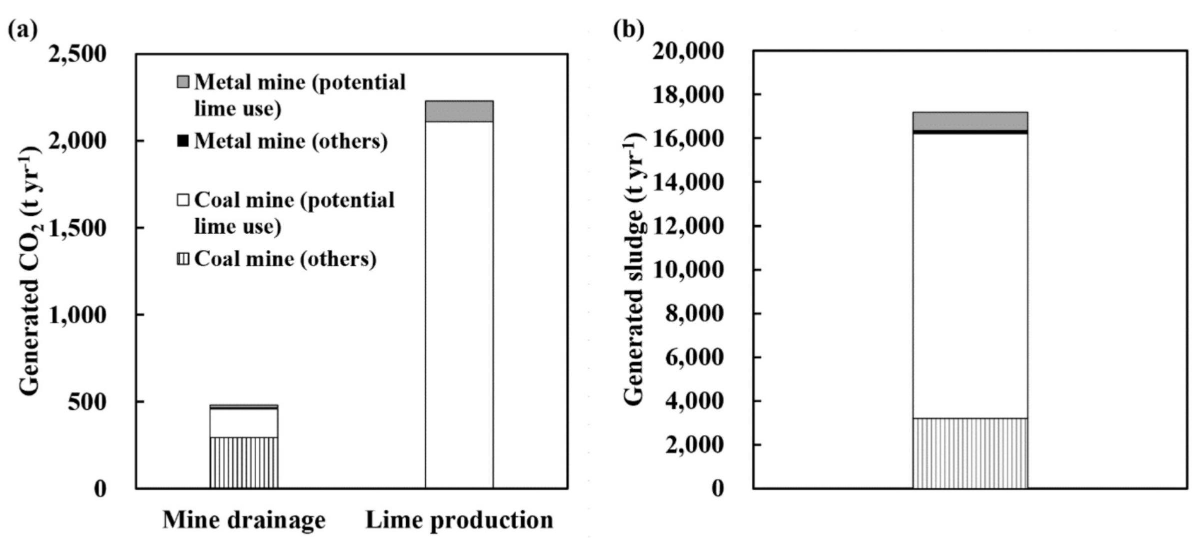


Fig. 9. (a) Modeled total potential flux of CO_2 from mine drainages and calculated indirect emission flux of CO_2 from modeled usage of hydrated lime for mine drainage treatment in South Korea. (b) Modeled total potential amount of annual sludge generation with water content of 75 % after treatment using hydrated lime (for "potential lime use") and after equilibrium by aeration (for "others") in South Korea. The samples are classified by mine types and potential use of hydrated lime.

Means et al., 2015). We compared direct CO_2 emissions during pre-aeration with indirect CO_2 emissions resulting from hydrated lime usage. The "Aerated to equilibrium" mode in the Caustic Titration module was applied to simulate pre-aeration to equilibrium conditions. The results, shown in Fig. 10, present the ratios of direct CO_2 emissions from pre-aeration to indirect CO_2 emissions from lime usage. The 1st, 2nd, and 3rd quartiles of these ratios were 2 %, 13 %, and 29 %, respectively. Meanwhile, Jageman et al. (1987) reported that lime consumption was reduced by 27 %, 38 %, and 43 % with pre-aeration. Similarly, Means et al. (2015) reported decreases of 22 % and 28 %, while Means and Beam (2024) noted a net annual lime dosage reduction of 50 %. These ratios of lime consumption reduction are higher than the observed median ratio (13 %) in Fig. 10. Therefore, in most cases, applying the pre-aeration step seems adequate to reduce both lime usage and net CO_2 emissions.

To specifically evaluate the impact of pre-aeration on CO_2 emissions, the CO_2 concentration generated by hydrated lime usage was plotted against the CO_2 concentration generated from mine drainage aeration, grouped by pH (Fig. 11a) and alkalinity (Fig. 11b). As samples with pH > 7.5 exhibited low CO_2 emissions from mine drainage, they are plotted above the 25 % line, indicating that CO_2 emissions from mine drainage are <25 % of the CO_2 emissions from lime usage. For these samples, pre-aeration will be suitable with respect to CO_2 emission when it reduces hydrated lime dosage by >25 %. On the contrary, many samples with pH between 6.5 and 7.5 are plotted near the 25 % line, reflecting higher CO_2 emissions from mine drainage. For those samples with a ratio of 25 %, at least a 25 % reduction of hydrated lime is required for the net CO_2 removal through pre-aeration. Several samples with pH of 4.5–6.5 are plotted below the 50 % line, which is principally attributable to low CO_2 emissions associated with lime usage. The low consumption of hydrated lime despite low pH appears to be related to their low alkalinity (Fig. 11b). Additionally, mine drainages with alkalinity of >200 mg L^{-1} as $CaCO_3$ exhibited higher lime consumption compared to those with alkalinity of 100–200 mg L^{-1} as $CaCO_3$ (Fig. 11b). This is consistent with the role of HCO_3^- in increasing lime dosage by consuming OH^- and facilitating calcite precipitation (Kim et al., 2023). In summary, samples with higher alkalinity and pH seem to benefit from pre-aeration in terms of reducing CO_2 emission.

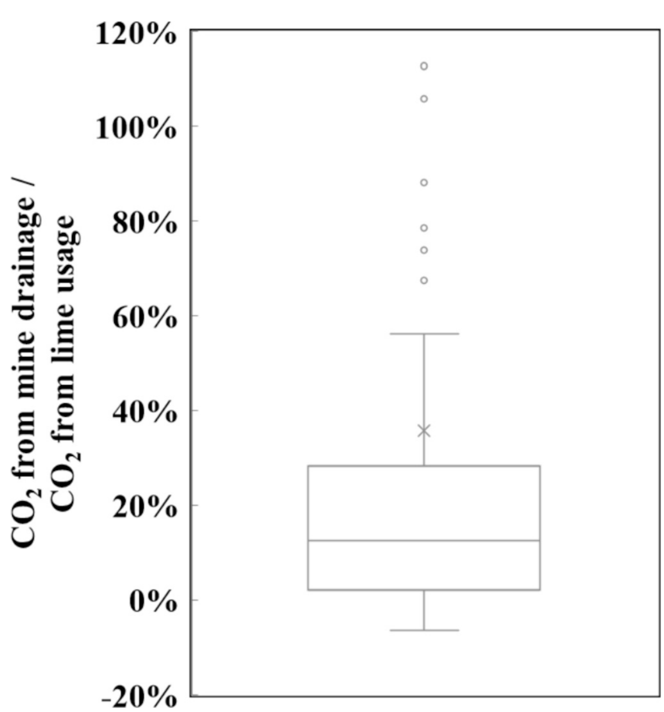


Fig. 10. Box plot representing ratios of modeled direct CO_2 emission from mine drainage by pre-aeration to modeled indirect CO_2 emission by the consumption of hydrated lime.

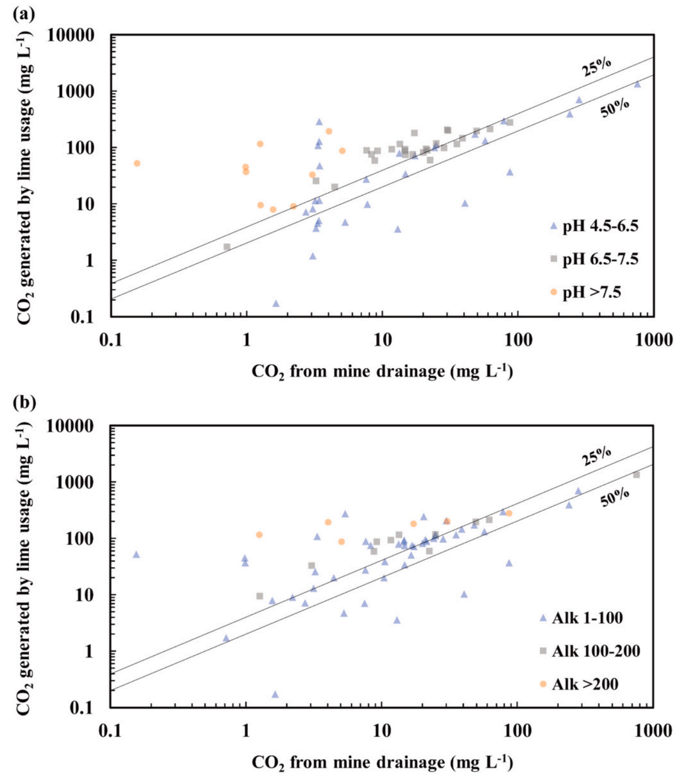


Fig. 11. Plot of modeled indirect CO_2 emission by the consumption of hydrated lime versus modeled direct CO_2 emission from mine drainage by pre-aeration. Ratios of CO_2 emission from mine drainage to CO_2 emission by the consumption of hydrated lime are also indicated as lines. The samples are classified by (a) pH and (b) alkalinity ranges.

4. Conclusions

Potential CO_2 emissions from mine drainages and from the production of hydrated lime for mine drainage treatment were assessed on a nationwide scale using the CausticTitration model of PHREEQ-N-AMDTreat. The total CO_2 emissions from mine drainages through aeration to equilibrium were estimated at 481 t yr^{-1} (Fig. 1), with 95 % originating from coal mines in South Korea. Mine drainages with lower pH and higher alkalinity exhibited a higher potential for CO_2 emissions. Treatment of mine drainage by increasing pH can reduce most of the emissions from mine drainage through conversion of DIC species and precipitation of calcite. Nevertheless, the potential consumption of hydrated lime for mine drainage treatment contributed to CO_2 emission of 2230 t yr^{-1}, which was >12 times the amount of CO_2 removed during the treatment process. Furthermore, constructing (semi-)active treatment facilities using hydrated lime involves substantial CO_2 emissions from cement production and operational electricity use. Therefore, regarding CO_2 emissions, applying passive treatment systems, including slag reactors to further increase pH if necessary, is much more beneficial than (semi-)active treatment systems. Passive treatment facilities involving increasing pH will have three advantages: (1) treatment of contaminants, (2) removal of potential CO_2 from mine drainages by pH increase (using alkaline materials), and (3) prevention of CO_2 emissions associated with hydrated lime usage.

The reduction in indirect CO_2 emissions from hydrated lime consumption through the implementation of a pre-aeration step was compared with direct CO_2 emissions from the step. CO_2 emissions from most mine drainages in South Korea were <13 % of indirect CO_2

emissions from hydrated lime. Given that this ratio (13 %) is lower than the reported reduction in lime consumption (22 %–50 %), pre-aeration is preferable for majority of mine drainages in terms of CO_2 emissions. Although mine drainages with pH below 4.5 or without alkalinity could not be assessed, pre-aeration was found to be particularly beneficial for mine drainages with high pH and alkalinity among those with pH above 4.5.

CRediT authorship contribution statement

Duk-Min Kim: Writing – review & editing, Writing – original draft, Supervision, Methodology, Formal analysis, Data curation, Conceptualization. **Ki-Rim Lee:** Formal analysis, Data curation. **Mi-Sun Park:** Project administration, Data curation.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Duk-Min Kim reports financial support was provided by Korea Mine Rehabilitation and Mineral Resources Corporation. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

This work was supported by an R&D project of the Korea Mine Rehabilitation and Mineral Resources Corporation from 2022 to 2024 and by the Graduate School of Sangji University. The authors gratefully acknowledge the contributions of the people at Korea Mine Rehabilitation and Mineral Resources Corporation for conducting sampling, pretreatment, and analyses. We thank Charles Cravotta for creating the PHREEQ-N-AMDTreat model and for his comments on an early draft of the manuscript. Special thanks go to anonymous reviewers for providing constructive comments that helped to improve the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2025.179270.

Data availability

Data will be made available on request.

References

APHA, 2017. Standard Methods for the Examination of Water and Wastewater, 23rd ed. American Public Health Association, American Water Works Association and Water Environmental Federation, Washington DC.

Bing, L., Ma, M., Liu, L., Wang, J., Niu, L., Xi, F., 2023. An investigation of the global uptake of CO_2 by lime from 1930 to 2020. Earth Syst. Sci. Data 15, 2431–2444. https://doi.org/10.5194/essd-15-2431-2023.

Brown, A.M., Bass, A.M., Garnett, M.H., Skiba, U.M., Macdonald, J.M., Pickard, A.E., 2024. Sources and controls of greenhouse gases and heavy metals in mine water: a continuing climate legacy. Sci. Total Environ. 906, 167371. https://doi.org/10.1016/j.scitotenv.2023.167371.

Caruccio, F.T., 1975. Estimating the acid potential of coal mine refuse. In: Chadwick, M. J., Goodman, G.T. (Eds.), The Ecology of Resource Degradation and Renewal. Blackwell, Oxford, UK, pp. 197–205.

Cicek, Z., Mira, A.A., Huang, Q., 2023. Process development for the extraction of rare earth elements from an acid mine drainage treatment sludge. Resour. Conserv. Recycl. 198, 107147. https://doi.org/10.1016/j.resconrec.2023.107147.

Cravotta, C.A., 2020. Interactive PHREEQ-N-AMDTreat water-quality modeling tools to evaluate performance and design of treatment systems for acid mine drainage (software download): U.S. Geological Survey Software Release. https://doi.org/10.5066/P9QEE3D5.

Cravotta, C.A., 2021. Interactive PHREEQ-N-AMDTreat water-quality modeling tools to evaluate performance and design of treatment systems for acid mine drainage. Appl. Geochem. 126, 104845. https://doi.org/10.1016/j.apgeochem.2020.104845.

Cui, M., Jang, M., Cho, S.-H., Khim, J., 2011. Potential application of sludge produced from coal mine drainage treatment for removing Zn(II) in an aqueous phase. Environ. Geochem. Health 33, 103–112. https://doi.org/10.1007/s10653-010-9348-0.

European Lime Association, 2019. A Competitive and Efficient Lime Industry. European Lime Association, Brussels, Belgium.

European Lime Association, 2021. Lime as a Natural Carbon Sink: Examples of Mineral Carbonation in Lime Applications. European Lime Association, Brussels, Belgium.

GGIRC, 2022. 2022 National Greenhouse Gas Inventory Report of Korea, Cheongju.

Hedin, R.S., Hedin, B.C., 2016. The complicated role of CO_2 in mine water treatment. In: IMWA 2016 Proceedings, International Mine Water Association, pp. 844–849.

Hengen, T.J., Squillace, M.K., O'Sullivan, A.D., Stone, J.J., 2014. Life cycle assessment analysis of active and passive acid mine drainage treatment technologies. Resour. Conserv. Recycl. 86, 160–167. https://doi.org/10.1016/j.resconrec.2014.01.003.

Ho, H.J., Iizuka, A., Vadapalli, V.R.K., Coetzee, H., Petrik, L., Petersen, J., Ojumu, T., 2023. Potential investigation of concrete fines as an alternative material: a novel neutralizer for acid mine drainage treatment. Environ. Technol. Innov. 29, 102985. https://doi.org/10.1016/j.eti.2022.102985.

IPCC, 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Switzerland, Geneva.

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC, Geneva, Switzerland.

IPCC, 2021. Climate change 2021: the physical science basis. In: Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge. https://doi.org/10.1017/9781009157896.

Jageman, T.C., Yokley, R.A., Heunisch, G.W., 1987. The use of pre-aeration to reduce the cost of neutralizing acid mine drainage. In: Proceedings American Society of Mining and Reclamation, American Society of Mining and Reclamation, pp. 131–135. https://doi.org/10.21000/jasmr88010131.

Kim, D.-M., Yun, S.-T., Cho, Y., Hong, J.-H., Batsaikhan, B., Oh, J., 2017. Hydrochemical assessment of environmental status of surface and ground water in mine areas in South Korea: emphasis on geochemical behaviors of metals and sulfate in ground water. J. Geochem. Explor. 183, 33–45. https://doi.org/10.1016/j.gexplo.2017.09.014.

Kim, D.-M., Im, D.-G., Lee, J.-H., 2022a. Removal of Mn via coprecipitation and sorption by Fe(II), Fe(III), and Al in mine drainage. Chemosphere 287, 132367. https://doi.org/10.1016/j.chemosphere.2021.132367.

Kim, D.-M., Park, H.-S., Hong, J.-H., Lee, J.-H., 2022b. Assessing pilot-scale treatment facilities with steel slag-limestone reactors to remove Mn from mine drainage. Mine Water Environ. 41, 402–414. https://doi.org/10.1007/s10230-021-00819-6.

Kim, D.-M., Oh, Y.-S., Park, H.-S., Kim, D.-K., Lee, J.-H., 2022c. Steel slag-limestone reactor for Mn removal with tolerance to Fe: experiments and mechanisms. J. Environ. Chem. Eng. 10, 107468. https://doi.org/10.1016/j.jece.2022.107468.

Kim, D.-M., Kwon, H.-L., Park, M.-S., 2023. Underestimation of alkaline dosage and precipitate amount during water treatment: role of inorganic carbon and use of PHREEQ-N-AMDTreat. J. Clean. Prod. 433, 139683. https://doi.org/10.1016/j.jclepro.2023.139683.

Kirby, C.S., Cravotta, C.A., 2005. Net alkalinity and net acidity 2: practical considerations. Appl. Geochem. 20, 1941–1964. https://doi.org/10.1016/j.apgeochem.2005.07.003.

Kirby, C.S., Dennis, A., Kahler, A., 2009. Aeration to degas CO_2, increase pH, and increase iron oxidation rates for efficient treatment of net alkaline mine drainage. Appl. Geochem. 24, 1175–1184. https://doi.org/10.1016/j.apgeochem.2009.02.028.

Kruse, N.A., Strosnider, W.H.J., 2015. Carbon dioxide dynamics and sequestration in mine water and waste. Mine Water Environ. 34, 3–9. https://doi.org/10.1007/s10230-014-0320-6.

Kumar, R., Kang, C.U., Mohan, D., Khan, M.A., Lee, J.H., Lee, S.S., Jeon, B.H., 2020. Waste sludge derived adsorbents for arsenate removal from water. Chemosphere 239, 124832. https://doi.org/10.1016/j.chemosphere.2019.124832.

Langmuir, D., 1997. Aqueous Environmental Geochemistry. Prentice-Hall, Upper Saddle River, New Jersey.

Laveglia, A., Sambataro, L., Ukrainczyk, N., De Belie, N., Koenders, E., 2022. Hydrated lime life-cycle assessment: current and future scenarios in four EU countries. J. Clean. Prod. 369, 133224. https://doi.org/10.1016/j.jclepro.2022.133224.

Lee, G., Cui, M., Yoon, Y., Khim, J., Jang, M., 2018. Passive treatment of arsenic and heavy metals contaminated circumneutral mine drainage using granular polyurethane impregnated by coal mine drainage sludge. J. Clean. Prod. 186, 282–292. https://doi.org/10.1016/J.JCLEPRO.2018.03.156.

Means, B., Beam, R.L., 2024. Using iron oxidation and decarbonation to enhance inorganic carbon removal in coal mine drainage. In: Proceedings of IMWA 2024. International Mine Water Association, p. 434.

Means, B., Beam, P.G.R., Mercer, J., 2015. Analysis of hydrated lime consumption in circumneutral underground coal mine drainage treatment. Mine Water Environ. 34, 10–19. https://doi.org/10.1007/s10230-014-0308-2.

Neogi, B., Singh, A.K., Pathak, D.D., Chaturvedi, A., 2017. Hydrogeochemistry of coal mine water of North Karanpura coalfields, India: implication for solute acquisition processes, dissolved fluxes and water quality assessment. Environ. Earth Sci. 76 (14), 489. https://doi.org/10.1007/s12665-017-6813-4.

Nordstrom, D.K., 2008. Questa Baseline and Pre-Mining Ground-Water Quality Investigation. 25. Summary of Results and Baseline and Pre-Mining Ground-Water Geochemistry, Red River Valley, Taos County, New Mexico, 2001–2005. U.S. Geological Survey Professional Paper, p. 1728.

Plummer, L.N., Busby, J.F., Lee, R.W., Hanshaw, B.B., 1990. Geochemical modeling of the Madison aquifer in parts of Montana, Wyoming, and South Dakota. Water Resour. Res. 26 (9), 1981–2014. https://doi.org/10.1029/WR026i009p01981.

Raymond, P.A., Oh, N.H., 2009. Long term changes of chemical weathering products in rivers heavily impacted from acid mine drainage: insights on the impact of coal mining on regional and global carbon and sulfur budgets. Earth Planet. Sci. Lett. 284, 50–56. https://doi.org/10.1016/j.epsl.2009.04.006.

Rose, A., Cravotta, C., 1998. Geochemistry of coal-mine drainage. In: Brady, K., Smith, M., Schueck, J. (Eds.), Coal Mine Drainage Prediction and Pollution Prevention. Pennsylvania Dept of Environmental Protection, Harrisburg, PA, ISA, pp. 1.1–1.22.

Shan, Y., Liu, Z., Guan, D., 2016. CO2 emissions from China's lime industry. Appl. Energy 166, 245–252. https://doi.org/10.1016/j.apenergy.2015.04.091.

Tuazon, D., Corder, G.D., 2008. Life cycle assessment of seawater neutralised red mud for treatment of acid mine drainage. Resour. Conserv. Recycl. 52, 1307–1314. https://doi.org/10.1016/j.resconrec.2008.07.010.

Vaziri Hassas, B., Shekarian, Y., Rezaee, M., Pisupati, S.V., 2022. Selective recovery of high-grade rare earth, Al, and Co-Mn from acid mine drainage treatment sludge material. Miner. Eng. 187, 107813. https://doi.org/10.1016/j.mineng.2022.107813.

Vesper, D.J., Moore, J.E., Adams, J.P., 2016. Inorganic carbon dynamics and CO_2 flux associated with coal-mine drainage sites in Blythedale PA and Lambert WV, USA. Environ. Earth Sci. 75, 1–14. https://doi.org/10.1007/s12665-015-5191-z.

Winfrey, B.K., Nairn, R.W., Tilley, D.R., Strosnider, W.H.J., 2015. Emergy and carbon footprint analysis of the construction of passive and active treatment systems for net alkaline mine drainage. Mine Water Environ. 34, 31–41. https://doi.org/10.1007/s10230-014-0304-6.

Wu, E., Wang, Q., Ke, L., Zhang, G., 2023. Study on carbon emission characteristics and emission reduction measures of lime production—a case of enterprise in the Yangtze River basin. Sustainability 15, 10185. https://doi.org/10.3390/su151310185.

Younger, P.L., Banwart, S.A., Hedin, R.S., 2002. Mine Water: Hydrology, Pollution, Remediation. Kluwer Academic Publishers, Dordrecht, Netherlands. https://doi.org/10.1007/978-94-010-0610-1.