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Optimization of phosphorus leaching from sludge of rubber-latex processing wastewater-treatment plant in Vietnam

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Abstract

In this study, optimal parameters for P leaching from sludges of a rubber-latex processing wastewater-treatment plant were carried out. The process optimization is based on laboratory experiments and simulation with a MATLAB program. The optimal P-leaching parameters for dried sludges were found to be as follows: dried sludge particle size below 2.5 mm, leaching time of 60 min, a phase ratio of 10 mL g^{-1} with 2 M H_2SO_4 for chemical sludge, a phase ratio of 5 mL g^{-1} with 1.7 M H_2SO_4 for biological sludge. At these conditions, P leaching efficiency reaches 90% on average for chemical as well as biological sludge. The equilibrium pH of the leached liquid was found to affect formation of mineral precipitates, such as gypsum ($CaSO_4$), which leads to significant reduction in the dissolved Ca content in the leached liquid. Simulation results from the solid–liquid-liquid thermodynamic equilibrium tool also show the formation of other precipitated minerals such as $AlPO_4$, $FePO_4$, or $Al(OH)_3$ at different equilibrium pH. The highest dissolved P and lowest dissolved Ca contents were observed at equilibrium pH less than 0.3.

Keywords Leaching, Phosphorus recovery, Simulation, Matlab, Rubber-latex wastewater, Phase ratio, Chemical sludge, Biological sludge

1 Introduction

Phosphorus (P) is an essential element for sustaining and developing life on Earth. P is an important element in the growth and biomass production of both plants and animals, particularly in the field of food production. According to Springmann et al. [1], the world population will reach 8.5 to 10 billion in 2050, and a 70% upsurge in food production is required to satisfy the demand. Due to this increasing food demand, providing adequate P for food production is becoming increasingly critical. However, phosphate rock reserves are mainly concentrated in a few territories, mainly northern Africa, the Middle East, China, and the USA, controlling about 87% of the world reserves [2]. In 2021, a phosphate ore mine in Norway was discovered, which could become the largest phosphate mine globally with estimated reserves of 70–80 BT of phosphate-rich material [3]. But P rock is a non-renewable resource, finite, and access is easily affected by geopolitical factors. Therefore, risks to global food security can occur in the future, which leads to P rock being added to the list of 20 Critical Raw Materials by the European Union due to its 'high supply risk'. In the Europe region, approximately 93% of P_2O_5 used in agriculture was imported in 2021 [4]. Therefore, solutions to reduce P losses, including P recovery from sludge in wastewater-treatment plants (WWTPs), are urgently needed, especially for P-rich sludge from improved WWTPs [5].

In Vietnam, the demand for P in agriculture is very large as well. According to FAO statistics, 740 kt of

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fertilizer as nutrient phosphate P_20_5 was used for agriculture in Vietnam in 2020 [6]. Dinh et al. showed that P-rich waste from agriculture and WWTPs in Southern Vietnam contained a total of about 50 kt of P in 2019, one very promising and easily accessible source being sludge from rubber latex processing WWTPs [7]. This P source can be accessed by leaching the sludge with an acid solution.

In the rubber-latex processing WWTPs, some chemicals such as $Al_n(OH)_mCl_{3n-m}$ (PAlC) and $Ca(OH)_2$ are commonly used to adjust pH and remove P from the wastewater. Therefore, sludges from these WWTPs often have high Al and Ca content. The Al and Ca-rich sludge presents challenges for P leaching because of the presence of alkaline components such as CaO and Al-compounds, which can also react with acids, and thus increase chemical consumption [8]. The dissolution of P compounds in sludge by adding an inorganic acid can occur according to the following reactions [9, 10]:

$$AlPO_4 + 3H^+ = Al^{3+} + H_3PO_4 \tag{1}$$

$$Ca_9(Al)(PO_4)_7 + 21H^+ = 9Ca^{2+} + Al^{3+} + 7H_3PO_4 \tag{2}$$

$$FePO_4 + 3H^+ = Fe^{3+} + H_3PO_4 \tag{3}$$

$$Fe_3(PO_4)_2 + 6H^+ = 3Fe^{2+} + 2H_3PO_4 \tag{4}$$

$$Ca_5(PO_4)_3OH + 4H^+ = H_2O + 3HPO_4^{2-} + 5Ca^{2+} \tag{5}$$

In this study, HNO_3, HCl, and H_2SO_4 are used to perform the leaching to determine the best option. Leaching has been optimized through varying leaching conditions such as acid concentration and liquid–solid phase ratio, both of which affect equilibrium pH. Moreover, the correlations between leaching conditions on P leaching efficiency are considered. Since acidic leaching results in the dissolution of P and metals [10, 11], the concentration needs to be considered in the evaluation of the leaching result.

2 Materials and methods

2.1 Sludge preparation

Sludge samples were collected from the rubber-latex processing WWTP located in Dong Nai province, Vietnam. This is the most favorable waste source for P recovery, according to a previous study [7]. At WWTPs, P-rich sludges are generated in different treatment stages. Biological sludge (BIOS) was generated from two biological processes, including aerobic and anoxic. Chemical sludge (CHES) was generated from coagulation and flocculation using poly-aluminum chloride and $Ca(OH)_2$ to remove P and suspended matter in wastewater. The moisture contents of dewatered BIOS and CHES were measured at approximately 84 and 87%, respectively. Both sludges were dewatered using a screw press. In the laboratory, sludges were then shaped by using an extruder with different hole diameters to generate "sludge spaghetti" to facilitate drying [12]. The sludge spaghetti was then subsequently subjected to drying with a forced-convection oven (THERMO, USA) at a temperature of 105°C until a stable mass was reached. Following this, the dried sludge was cut into pieces manually with the same length as the hole diameter of each extruder plate, including sizes smaller than 1 mm, 1 to 2.5 mm, 2.5 to 5 mm, and above 5 mm. These dried sludge pieces were then precisely separated to uniform size using a sieve before being used for the leaching experiments.

2.2 Leaching procedure

For each leaching experiment, 2 g of size-sorted dry sludge was introduced into a 100 mL glass flask, to which acid was added. The flask was then covered and mixed using a rotating shaker at 30 rpm at room temperature. The leaching time was investigated from 30 to 270 min, incrementally varying by 30 min. HCl, HNO_3, and H_2SO_4 have been used as acids for leaching.

The phase ratio was defined as the volume of acid solution (in mL) to the mass of dry sludge (in g), expressed as L:S. Acid concentration for leaching was studied from 0.1 to 3 M. The leached mixtures were subjected to phase separation by centrifugation and then filtered through a 1 µm pore size polytetrafluoroethylene syringe filter (SYRINGE, FINETECH, Taiwan) to obtain leach solutions, which were then analyzed for their composition and substance content.

The leaching efficiency is calculated as

$$L = \frac{m_{i,l}}{m_{i,s}} \tag{6}$$

where $m_{i,l}$ represents the mass of the species present in the leach liquid obtained through the leaching process and $m_{i,s}$ is the mass of the species in the initial dried sludge sample utilized for leaching.

2.3 Analysis

The moisture content of the sample was determined using a moisture meter (MA150 MOISTURE ANALYZER, SARTORIUS, Germany) by drying approximately 50 g of dewatered sludge. The moisture analyzer combines an analytical balance with infrared drying to achieve an accuracy of 0.001 g to determine the moisture content.

The total phosphorus and metal content in the input sludge was determined using the digestion method developed by the European "Horizontal" project [13]. An aqua regia mixture was prepared by adding 21 mL of 37% HCl, 7 mL of 65% HNO_3, and 1 mL of distilled water to a 250 mL reaction flask. Then, 3 g pulverized dry sludge with less than 500 µm particle size was added to this flask. The mixture was left open at room temperature to allow the gases produced in the reaction to escape before refluxing for 2 h. The digestion solution was transferred to a 100 mL flask and diluted with 5% HNO_3 solution to about 30 mL. Before analysis, the diluted digestion solution was phase-separated using a 0.45 µm pore size 25 mm syringe filter (SYRINGE, FINETECH, Taiwan). Total phosphorus and metals including Al, Ca, Fe, Mg, and heavy metals were analyzed using an inductively coupled plasma optical emission spectrometer (ICP-OES, SPECTRO BLUE TM, SPECTRO, Germany).

To determine the content of P and metals in the leach solutions, the mixture obtained after leaching, including the remaining insoluble solids, was transferred to 50 mL poly-propylene centrifuge tubes and centrifuged at 5000 rpm for 6 min (DM0636, DLAB SCIENTIFIC, China). The supernatant was then immediately filtered using a 60 mL syringe combined with a 1 µm pore size polytetrafluoroethylene syringe filter (SYRINGE, FINETECH, Taiwan) to remove the remaining suspended solids. The equilibrium pH of the leached liquid was measured using a pH meter (HANNA, Italy), and P and metal contents were analyzed using the ICP-OES. Morphological analysis was conducted utilizing a scanning electron microscope (SEM) (HITACHI S4800 FE-SEM, Japan). The accelerating voltage employed in the SEM ranged from 0.5 to 30 kV, with voltage increments of 0.1 kV per step.

The determination of the P fraction in the sludge samples, including organic phosphorus (OP) and inorganic phosphorus (IP) followed the Standards, Measurements, and Testing (SMT) procedure developed and revised by Ruban et al. and García-Albacete et al. as part of the European Commission's SMT project [14, 15].

2.4 Modelling

Equilibrium simulations were used to evaluate the chemical reactions, calculating the concentrations of species that formed in the solid or liquid phase. The equilibrium pH as well as the input conditions such as concentration of feed components, leaching agent type, and temperature need to be specified. The equilibrium simulation is used to optimize the P leaching parameters, thereby significantly reducing the demand for laboratory experiments. The solid–liquid-liquid thermodynamic equilibrium (SLLE) simulation tool, developed in MATLAB was adapted and applied based on the validated version developed by Shariff et al. [16], which was used to evaluate and optimize a P leaching and purification process. The equilibrium composition in a chemical system is obtained by solving a system of nonlinear equations consisting of mass balance, the law of mass action, and charge balance by using the Newton–Raphson method. The thermodynamic non-idealities have been described with the Debye-Hückel or the Davies model. The reaction constants have been obtained from the literature [17–19]. The initial guess is obtained with the zero-order positive continuous fraction method [16, 20]. The SLLE tool in Matlab allows to evaluate the speciation in the aqueous phase. In principle, reactive extraction can also be described, but it is not used in this work. In addition, simulations on precipitation or dissolution of compounds can be performed in the Matlab tool. The solubility or precipitation of a species is expressed through two main indices, the equilibrium constant K_p (or solubility product constant) and the ion activity product (*IAP*). The decadic logarithm of the *IAP* to K_p ratio is the saturation index, which was calculated in the SLLE simulation to evaluate the species' tendency to form precipitates. The developed simulation tool has been validated by comparing results from laboratory experiments and the Aqion software [18], which describes the data as worse than the developed SLLE tool.

3 Results and discussion

3.1 Dried sludge composition

The chemical fractionation of phosphorus from CHES and BIOS, including total P (TP), inorganic P, and organic P were determined, where TP is equal to the sum of IP and OP [21, 22] together with the concentration of other components as shown in Table 1. CHES and BIOS contained approximately 93% and 76% of IP, respectively. The significant difference in P fractions between these two types of sludge mainly results from the difference in P removal methods in the wastewater-treatment steps. Table 1 indicates that P and Ca were the elements with the highest mass fraction in both types of sludge. The high Al content in CHES can be attributed to the use of $Al_n(OH)_mCl_{3n-m}$ for P precipitation and adsorption in the chemical treatment stage, leading to the formation of Al–P salts [23, 24], and $Al(OH)_3$ in adsorption [23–25]. In addition, hydrated lime $Ca(OH)_2$ was used to regulate pH, and to remove P in wastewater by forming Ca–P salts in the WWTP [26–28], which leads to a high fraction of Ca and P in the CHES and BIOS. Most heavy metals such as Cu, Cr, Cd, and Pb content from CHES were higher than those in BIOS. In particular, the Cr content in CHES was nearly 12 times higher than in BIOS. The Cu, Cd and Pb concentrations relative to the P content in CHES

Table 1 Dried sludge compositions

Element		**Content** mg g^{-1} **dried sludge**		**Fraction** wt-%	
		CHES	**BIOS**	**CHES**	**BIOS**
P	of which	59.2 ± 4.3	45.3 ± 2.8	5.9	4.5
	OP			7	24
	IP			93	76
Al		113.4 ± 14.7	19.2 ± 2.4	11.3	1.9
Ca		32.2 ± 4.2	28 ± 2.3	3.2	2.8
Fe		11.7 ± 7.5	10.02 ± 2.6	1.2	1.0
Mn		0.4 ± 0.1	0.6 ± 0.03	0.04	0.06
Zn		1.4 ± 0.2	1.5 ± 0.3	0.14	0.15
K		8.4 ± 2.8	8.7 ± 1.3	0.08	0.09
Cu		0.143 ± 0.02	0.130 ± 0.06	0.014	0.013
Cr		0.167 ± 0.02	0.014 ± 0.005	0.017	0.014
Cd		$9 \times 10^{-4} \pm 2 \times 10^{-4}$	BDL	9×10^{-5}	BDL
Pb		$0.034 \pm 4.7 \times 10^{-3}$	BDL	3.4×10^{-3}	BDL

BDL below to the detection level (0.01 mg L^{-1})

and BIOS were lower than the maximum allowable limits according to fertilizer quality standards in Vietnam and Europe [29, 30]. In cases where heavy metals were present in the final product recovered from the studied sludges, the Pb and Cd to P (P_2O_5) mass ratios were about 6.63×10^{-06} and 2.52×10^{-04}, respectively which is significantly better than required by the quality standards for complex-organic fertilizers as in QCVN 01–189:2019/BNNPTNT [29].

3.2 Leaching variation of acids

The leaching efficiencies for P, Al, Fe, and Ca in CHES and BIOS for the three acids investigated are shown in Fig. 1a and b. At identical conditions, the P leaching efficiency in BIOS was higher at 89 ± 2.5% as compared to 77 ± 1.7% in CHES. This demonstrates that at the chosen condition also the OP can be accessed by leaching. The difference in the concentration of metals such as Al, Fe, and Ca in dried CHES and BIOS can lead to differences in P leaching efficiency. The efficiency of Ca leaching by using H_2SO_4 was much lower than that by using HCl and HNO_3 for CHES as well as BIOS. This was induced by gypsum formation with H_2SO_4 [31, 32]. Since a low Ca concentration is desired in the final product, it is advantageous to minimize Ca co-leaching. In addition, the price of H_2SO_4 is lower compared to that of HCl and HNO_3 [9, 33]. Thus, H_2SO_4 was chosen as the most suitable acid for the leaching of P in CHES and BIOS. It should be noted though that at identical molar concentration, H_2SO_4 provides twice the H^+ content compared to the other two acids.

3.3 Leaching time and particle size

Leaching time and particle size have been varied using 1 M H_2SO_4 and a phase ratio of 20 mL g^{-1}. The P, Al, Ca, and Fe leaching efficiencies were analyzed (Figs. S1 and S2). Maximum P leaching efficiency can be reached for CHES and BIOS already after 60 to 90 min, especially for smaller particle sizes. Leaching time exceeding 180 min showed that the P leaching efficiency tends to decrease slightly, possibly due to the onset of chemical reactions that re-precipitate to create P mineral compounds [34]. P leaching efficiency decreased as sludge particle size increased, especially at short leaching time from 30 to 60 min, but the difference between particles < 1 mm and 1 to 2.5 mm was significantly less than compared to larger particles. At leaching time of 60 and 120 min, P leaching

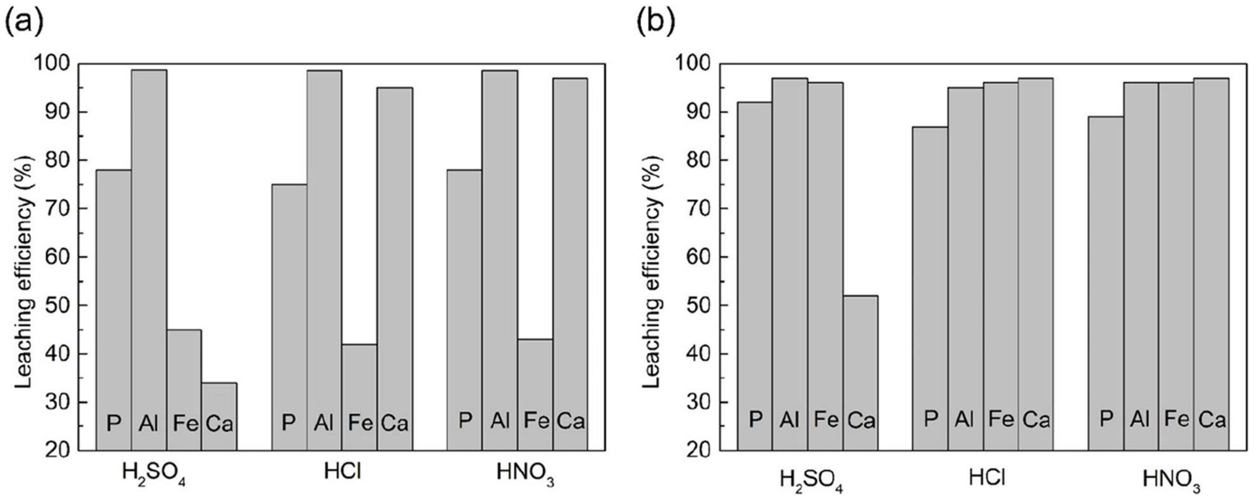

Fig. 1 Leaching efficiency of P, Al, Ca and Fe with different 1 M acids at phase ratio 20 mL g^{-1}, for (**a**) chemical sludge and (**b**) biological sludge

efficiency for particles below 2.5 mm reached a maximum of over 92% for BIOS and 85% for CHES, respectively. This result is consistent with previous studies by Ali and Kim [35].

Already after 30 min more than 90% of the Al was leached in both types of sludge with a particle size of less than 2.5 mm. There is no significant difference in Al leaching efficiency in CHES and BIOS under the same leaching conditions. Thus, under conditions where P is efficiently leached, Al will always be co-leached to a large fraction.

As already shown in Fig. 1, there is a significant difference in the Ca and Fe leaching efficiency between CHES and BIOS. Beyond 60 min of leaching time, the leaching efficiency of Ca and Fe varied only a little, almost independent of particle size. Thus, also the fraction of Ca and Fe leached as shown in Fig. 1 cannot be avoided at conditions suitable for P leaching.

Thus, a sludge particle size between 1 and 2.5 mm and leaching time between 60 and 90 min are the optimal P leaching conditions for both, BIOS and CHES.

3.4 Phase ratio and leaching time

The phase ratio in leaching defines water and acid consumption and thus needs to be optimized. The resources consumed increase as the phase ratio increases, which reduces the attractiveness of the P-recovery process by increasing recovery costs [36]. Phase ratio using 1 M H_2SO_4 was thus varied between 5 and 30 mL g^{-1} and the leaching time was between 60 and 180 min. CHES and BIOS particles between 1 and 2.5 mm were used. The results are shown exemplarity for P in Fig. 2, for the other metals in the supplementary material (see Figs. S3–S5). The detailed concentration of P, Ca, Al and Fe before and after leaching at different phase ratios and leaching time using 1 M H_2SO_4 for CHES and BIOS are shown in the supplementary material (see Tabs. S1–S8). One major result shown in Fig. 2 is that the P leaching efficiency was not significantly influenced by leaching time. This applies to the leaching of Al, Ca, and Fe as well. Therefore, 60 min was used as the most suitable leaching time in the following.

P leaching efficiency shows a very clear difference between the leaching of CHES and BIOS. An increase in phase ratio leads to a decrease in equilibrium pH and correspondingly to an increase in P leaching efficiency. P leaching efficiencies of over 95% were reached at phase ratios greater than 23 and equilibrium pH below 0.55 for CHES, and phase ratio above 12, and equilibrium pH below 0.28 for BIOS.

As above, under conditions of high P-leaching efficiencies, Al was leached at least with 95% efficiency for both CHES and BIOS. Above a pH of 3, a very different behavior was observed for Al, which was much less efficiently leached, and its leaching efficiency decreased rapidly further as pH increased. Figure 3 shows the results of SLLE simulation for CHES, which indicate the possibility of precipitate formation during the leaching process. Comparable results are found for BIOS (see Fig. S6). Simulation results show that not only Al compounds such as $Al(OH)_3$ and $AlPO_4$ are formed but also Fe compounds and Ca compounds. Similar results were reported by Van Truong et al. [37] as well as Shiba and Ntuli [38]. The formed $AlPO_4$ precipitate reduced the Al as well as the soluble P content in the leaching solution [23–25].

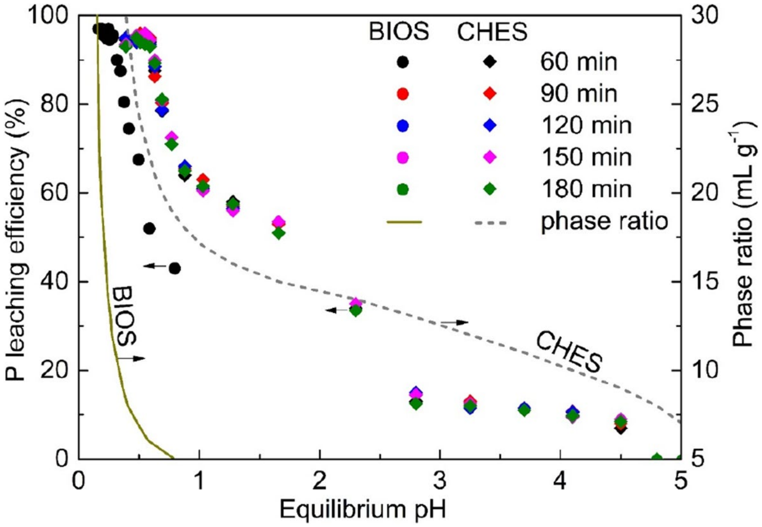

Fig. 2 Dependence of phosphorus leaching efficiency on leaching time and phase ratio

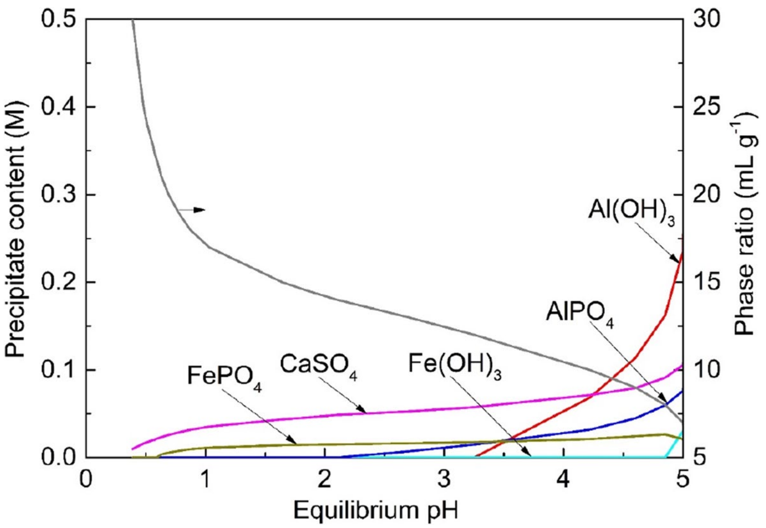

Fig. 3 Simulation of precipitated minerals with 1 M H_2SO_4 for CHES

The Ca leaching efficiency in BIOS and CHES tends to increase as the phase ratio is increased and the equilibrium pH is decreased. Results from SLLE simulations shown in Fig. 3 show that the gypsum precipitation gradually increases as equilibrium pH increases, leading to a decrease in dissolved Ca. The lower the phase ratio, the higher the fraction of gypsum precipitate.

Fe leaching efficiency also tends to increase when the phase ratio is increased and equilibrium pH is decreased. The formation of $FePO_4$ precipitate, as shown in Fig. 3, reduced the dissolved Fe content in the leach liquor, and also reduced the P leaching efficiency.

The strong influence of phase ratio on P leaching efficiency shown here was also noted by Donatello et al. [39], Shiba and Ntuli [38], and Semerci et al. [36]. The higher the phase ratio, the better the P leaching achieved. However, increasing the phase ratio also leads to an increase in the amount of chemicals and water consumed as well as equipment size. Thus, a small phase ratio is actually desired.

3.5 Acid concentration

In Sect. 3.4, it has been realized that equilibrium pH is a major influencing parameter for leaching as expected. There, equilibrium pH was varied by varying phase ratios for a given molarity of acid. Alternatively, equilibrium can be varied by changing acid concentration for a given phase ratio. Additionally, increasing the H_2SO_4 concentration can favorably influence the kinetics of the reaction, improving the P-leaching efficiency [40–42]. Figures 4a and b show the leaching efficiency of P, Ca, Al, and Fe at different H_2SO_4 concentrations at phase ratio 20. The dissolved P content in leach liquid increased with increasing H_2SO_4 concentration and decreasing pH. In the pH range of 0.5 to 2.5, P leaching is slightly worse for BIOS than for CHES. At lower pH, i.e. for H_2SO_4 concentrations above 1 M, the difference in P leaching efficiency from CHES and BIOS is negligible. Both are greater than 90%. Liang et al. [8] reported similar trends in P leaching efficiency from sewage sludge under different H_2SO_4 concentrations. The trends for Al and Fe leaching efficiency depend essentially identically on pH for both types of sludge. A significant difference between CHES and BIOS can be observed for Ca leaching. Around a pH of 0.5, a maximum in Ca leaching is observed, which is around 60% for BIOS and 35% for CHES. For lower pH especially for BIOS, the Ca leaching efficiency decreases significantly.

3.6 Simulation result analysis and experiment validation

From the previous two sections, it is obvious that equilibrium pH is a decisive parameter, which can be varied by phase ratio as well as initial acid concentration. Both parameters need to be varied simultaneously for process optimization. This variation has been realized by simulation. Figures 5 and 6 show how P leaching efficiency and the amount of H_2SO_4, which are decisive parameters for judging the economic feasibility of the leaching, depend on the phase ratio and H_2SO_4 concentration for CHES and BIOS. It is obvious that lines of constant P leaching efficiency and the amount of H_2SO_4 used show a rather similar behavior. Nevertheless, from a given P leaching efficiency decreasing the phase ratio but increasing H_2SO_4 concentration leads to a certain reduction in the amount of H_2SO_4 required due to the slightly different slope of the iso-lines of P leaching efficiency and amount

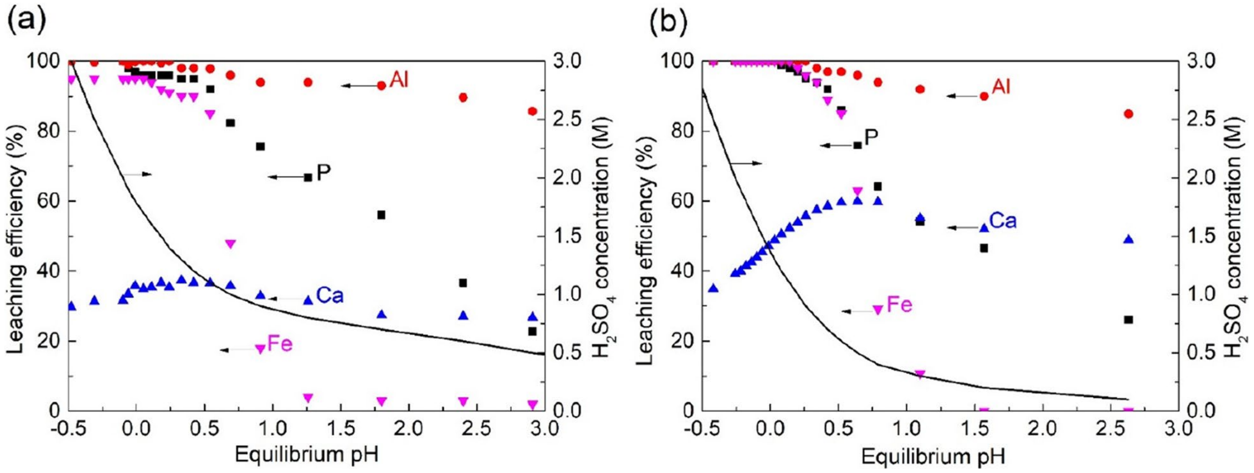

Fig. 4 Leaching efficiency of P, Al, Ca and Fe at different H_2SO_4 concentrations, phase ratio 20 for (**a**) CHES and (**b**) BIOS

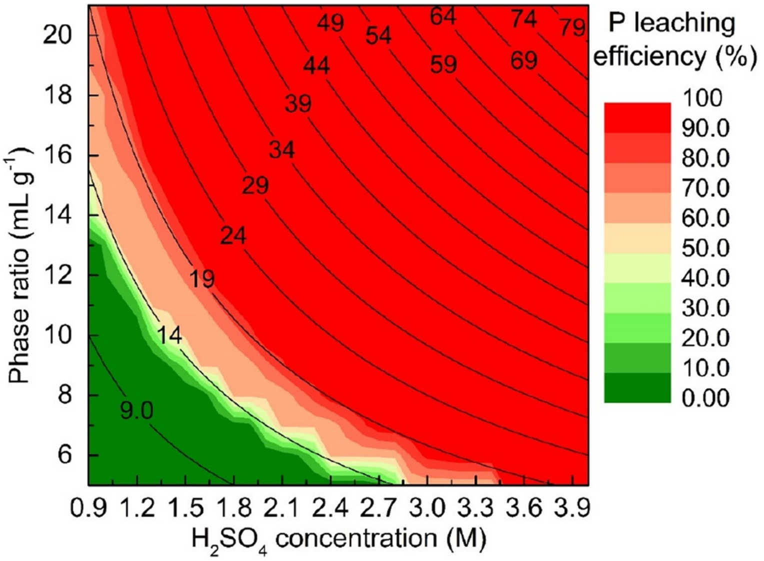

Fig. 5 Simulation results for CHES on the dependence of P leaching efficiency and the amount of H_2SO_4 on phase ratio and H_2SO_4 concentration used. The labeled iso-lines show the amount of H_2SO_4 required for leaching per kg dry sludge in mol kg.$^{-1}$

of H_2SO_4 used. It is thus beneficial to reduce the phase ratio. At the same time, this reduces the amount of water resources required and equipment size. The amount of H_2SO_4 used to leach more than 95% P in 1 kg of dry CHES is about 20 mol and 8.5 mol for BIOS.

On the other hand, performing leaching at too low a phase ratio limits the ability to filter and separate the leached liquor from the solid phase after leaching. This is especially critical due to the formation of gypsum precipitate, especially for sludges with high Ca content, i.e. for both CHES and BIOS. Taking these limits into account for the sludges investigated here, the optimal choices for leaching P from CHES are a 2 M H_2SO_4 and a phase ratio of 10, and for BIOS are 1.7 M H_2SO_4 and a phase ratio of 5, leading to a P leaching efficiency of at least 90%. Validation experiments have been performed for these optimal cases, as shown in Fig. 7, where the experimented data are compared to simulations. The data obtained, on the one hand, matches the values expected from the simulations. On the other hand, this good agreement justifies using the simulation for optimization.

Figure 8 finally shows SEM images before and after P leaching in CHES and BIOS at optimum parameters. These images clearly show the increase of gypsum precipitate in the solid residue after leaching. The amount of gypsum precipitate formed increases the volume of solid

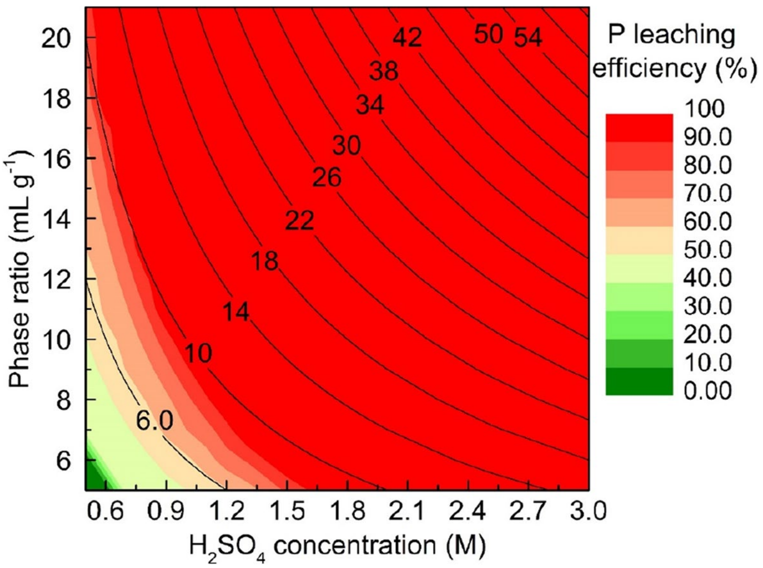

Fig. 6 Simulation results for BIOS on the dependence of P leaching efficiency and the amount of H_2SO_4 on phase ratio and H_2SO_4 concentration used. The labeled iso-lines show the amount of H_2SO_4 required for leaching per kg dry sludge in mol kg.$^{-1}$

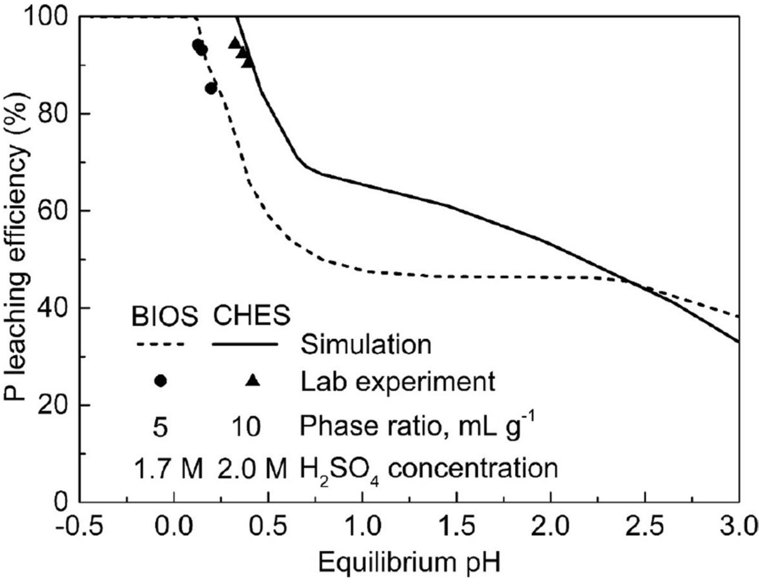

Fig. 7 Leaching efficiency of P from experiment and simulation at optimal parameters

that needs to be processed after the leaching. However, this also indicates that by using H_2SO_4 for leaching, the content of the unwanted Ca dissolved can be significantly reduced in leach liquid. It should be noted that in the P recovery procedure, the solids after the leaching should be washed with water to maximize the recovery of dissolved P while reducing the acidity of the residue solids. After washing, the recovered liquid can be used to prepare the acidic medium for the next leaching batch.

4 Conclusions

The solid–liquid-liquid equilibrium simulations with MATLAB allow P leaching optimization with H_2SO_4 consistent with experimental results on lab and pilot-plant scale. Under optimal conditions, P leaching efficiency can reach 95% and above. At the same time, Al and Fe in both CHES and BIOS are co-leaching. The equilibrium pH is the determining parameter as expected for the leaching of P, Al, Fe, and Ca. It also determines the

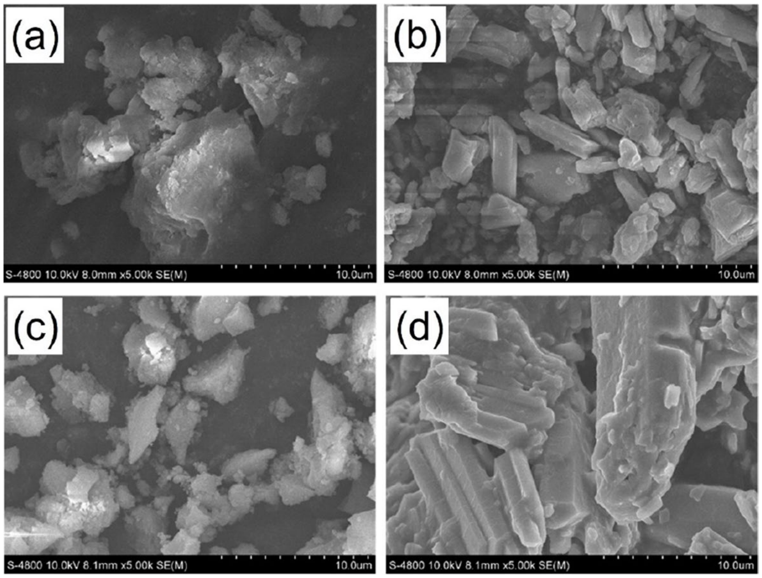

Fig. 8 SEM images of input dried sludge for (**a**) CHES and (**c**) BIOS as well as residue dried solid after leaching at optimal leaching parameters for (**b**) CHES and (**d**) BIOS

precipitation of minerals. Ca precipitation is unavoidable if H_2SO_4 is applied to Ca-containing sludge. Gypsum precipitation significantly reduces the dissolved Ca content in the leached liquid but also significantly increases the volume of solid waste generated.

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s42834-025-00244-2.

Supplementary Material 1

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Authors' contributions

Khang Vu Dinh conceptualized the study and conducted experiments. Andreas Pfennig guided the studies, evaluation and contributed significantly to the text of manuscript. Le Hung Anh supported the implementation of ideas through technical issues and equipment for analysis at the laboratory. All authors read and approved the final manuscript.

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Data availability

All data generated or analyzed during this study are available from the corresponding author on reasonable request.

Declarations

Competing interests

The authors declare they have no competing interests.

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