Phosphate recovery from anaerobic digester effluents using CaMg(OH)₄

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ABSTRACT

Dolomite lime (DL) (CaMg(OH)₄) was used as an economical source of Mg²⁺ for the removal and recovery of phosphate from an anaerobic digester effluent of a municipal wastewater treatment plant (MWWTP) wastewater. Batch precipitation results determined that phosphate was effectively reduced from 87 to less than 4 mg-P/L when the effluent water was mixed with 0.3 g/L of DL. The competitive precipitation mechanisms of different solids in the treatment system consisting of Ca²⁺–Mg²⁺–NH₄⁺–PO₄³⁻–CO₃²⁻ were determined by comparing model predictions with experimental results. Thermodynamic model calculations indicated that hydroxyapatite (Ca₁₀(PO₄)₆(OH)₂), Ca₄H(PO₄)₃∙3H₂O, Ca₃(PO₄)₂(beta), and Ca₃(PO₄)₂(am2) were more stable than struvite (MgNH₄PO₃∙6H₂O) and calcite (CaCO₃). However, X-ray diffraction (XRD) analysis determined the formation of struvite and calcite minerals in the treated effluent. Kinetic experimental results showed that most of the phosphate was removed from synthetic effluent containing NH₄⁺ within 2 hr, while only 20% of the PO₄³⁻ was removed in the absence of NH₄⁺ after 24 hr of treatment. The formation of struvite in the DL-treated effluent was due to the rapid precipitation rate of the mineral. The final pH of the DL-treated effluent significantly influenced the mass ratio of struvite to calcite in the precipitates. Because more calcite was formed when the pH increased from 8.4 to 9.6, a pH range of 8.0–8.5 should be used to produce solid with high PO₄³⁻ content. This study demonstrated that DL could be used for effective removal of phosphate from the effluent and that resultant precipitates contained high content of phosphate and ammonium.

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Phosphorus recovery
Struvite
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Introduction

Elevated concentrations of P and N in the environment have resulted in eutrophication problems worldwide, with more than about 50% of the lakes in North America, Europe and Asia being classified as eutrophic (ILEC, 1988–1993) Eutrophication results in enhanced growth of choking aquatic vegetation or phytoplankton, which disrupts normal functioning of the ecosystem, pollutes drinking water sources, and interferes with drinking water treatment. On the other hand, P is an essential nutrient for agricultural production. The existing phosphate rock resources can only maintain human demand for about 370 years. It is essential to remove and recover P from wastewater and other waste materials for...
the reduction of eutrophication problems and for sustainable utilization of limited P resources.

During wastewater treatment processes, most P and N are enriched and deposited in activated sludge (AS). When the AS is treated in the anaerobic digester, a substantial amount of the P and N is released into solution (Battistoni et al., 2000; Munch and Barr, 2001). The high concentration of P and N in the anaerobic digester effluent offers a good opportunity for the removal and recovery of these compounds. Several technologies have been tested for P removal and recovery from anaerobic digester effluent based mainly on the formation of phosphate precipitates (et al., 1991) has been tested in several studies. Dolomite had the formation of struvite precipitate was not considered in the study. The removal of phosphate from solutions by dolomite [CaMg(CO3)2], for treatment of phosphate. DL powder was efficiently through formation of calcium phosphate precipitates. Hence, phosphate precipitation mechanisms in DL-treated wastewater is more complex than in wastewater treated with magnesium and calcium compounds because of competitive formation of struvite and various calcium phosphate precipitates in the co-existence of high concentration of Mg2+ and Ca2+ in the former system. The kinetics and optimal DL dosage and pH value for the removal of phosphate were investigated. The phosphate precipitates formed in the treatment systems were determined and compared with the thermodynamically predicted precipitates, to understand the factors controlling the formation of the phosphate precipitates.

1. Materials and methods

1.1. Materials

The anaerobic digester effluent samples were collected from a MWWTTP in New Jersey, USA. The chemical characteristics of the effluent sample are as following: PO4−P 87 mg/L, Ca2+ 51 mg/L, Mg2+ 7.2 mg/L, NH4+ 1142 mg/L, and CO3− 641 mg/L.

The DL powder used in this study was provided by Graymount Lime Inc. Acid digestion analysis determined that Ca2+ and Mg2+ content in the material was 39.9% and 22.4%, respectively.

1.2. Batch experiments

Kinetic experiments were conducted by adding 0.1–0.8 g/L DL and 0.3 g/L lime into 100 mL of the anaerobic digester effluent in 100-mL bottles. The samples in the capped bottles were mixed at a rate of 125 r/min. The pH of the samples was measured and aliquots of the uniform samples were taken during 48 hr of mixing. The samples were centrifuged immediately at 9500 r/min for 4 min to separate the solution from the solids for analysis of the soluble P, calcium, and magnesium. A few samples were processed with both centrifugation and filtration to evaluate if centrifugation would affect the precipitation of phosphate. The results in Appendix A Table S1 indicate that it did not affect the formation of the phosphate precipitates.

Another set of kinetic experiments was conducted using synthetic anaerobic digester effluent to investigate the effect of ammonium on the removal of phosphate. The effluent solutions were prepared using KH2PO4 and NH4Cl. The phosphate and ammonium concentrations in one of the solutions were the same as those listed in Section 1.1. The other solution also contained 87 mg/L of PO4−P, but without ammonium to avoid the formation of struvite. No Ca2+ and
Mg$^{2+}$ were added into the solutions. An initial pH value of the solutions was adjusted to 7.5. After adding 0.3 g/L of DL into the solutions the kinetic experiments were started. The pH value was measured and samples were taken at the desired mixing time for analysis of soluble concentrations of phosphate, calcium, and magnesium.

The effect of pH on the removal of phosphate was tested by mixing 0.03 g of DL with 100 mL of the effluent samples for 48 hr. The initial pH values of the samples were adjusted to a range between 7.0 and 11.0 using HCl and NaOH solutions. After 48 hr of mixing, the final pH and soluble concentrations of phosphate, calcium, and magnesium were measured.

1.3. Analytical methods

P was analyzed with standard method 4500-PE (Standard Methods for the Examination of Water and Wastewater, 1998), Ca$^{2+}$ and Mg$^{2+}$ were analyzed using an Inductively Coupled Plasma Optical Emission Spectrometer (Vista MPX-ICP, Varian Company, USA).

The solid samples used for scanning electron microscopy (SEM) and X-ray diffraction (XRD) analyses were prepared by addition of 0.3 g/L of DL into 2.5 L of anaerobic digester effluent samples. The initial pH of the suspension samples was adjusted to 7.5–10.5 using HCl and NaOH solutions. The samples in capped containers were mixed at 125 r/min for 48 hr, then centrifuged at 9500 r/min for 4 min to separate solid precipitates from the liquid. The solids were dried at 45°C for 64 hr. The SEM images of the solids were recorded and the energy-dispersive X-ray spectroscopy (EDS) analysis of chemical composition of the particles were performed by a Zeiss Auriga Small Dual-Beam FIB-SEM equipped by an Oxford Instrument’s X-MAX 80 EDS detector. The INCA software was used to calculate the elemental composition from EDS spectra. XRD patterns of the sample powders were recorded using an Ultima IV Goniometer 285 mm System with Cu Kα radiation at a 2θ scan rate of 0.02°/sec. The crystal phases of the precipitates were identified by matching the XRD of the precipitates with standard XRD of components using Jade 7.5. Then the content of crystal components was determined by fitting the XRD profile using Reference Intensity Ratio (RIR) method. The data were used for mineral phase identification and for quantity estimation.

1.4. Model calculations

The computer program Visual MINTEQ3 (Gustafsson, 2010) was used for equilibrium calculations of the chemical reactions in the DL-treated anaerobic digester effluent containing PO$_4^{3-}$–Ca$^{2+}$–Mg$^{2+}$–NH$_4^+$–CO$_3^{2-}$. Default values in the program database were used for all equilibrium constants. Reactions and corresponding pK$_{sp}$ values for the top five least soluble phosphate precipitates are provided in Eqs. (1)–(5).

\[
\text{Ca}_3(PO_4)_2 (\text{oh}) = 5\text{Ca}^{2+} + 3\text{PO}_4^{3-} + \text{OH}^- \quad \text{pK}_{sp} = 44.33
\] (1)

\[
\text{Ca}_4\text{H}(PO_4)_3\cdot3\text{H}_2\text{O} = 4\text{Ca}^{2+} + \text{H}^+ + 3\text{PO}_4^{3-} + 3\text{H}_2\text{O} \quad \text{pK}_{sp} = 47.95
\] (2)

\[
\text{Ca}_3(PO_4)_2(\text{beta}) = 3\text{Ca}^{2+} + 2\text{PO}_4^{3-} \quad \text{pK}_{sp} = 28.92
\] (3)

\[
\text{Ca}_3(PO_4)_2(\text{am2}) = 3\text{Ca}^{2+} + 2\text{PO}_4^{3-} \quad \text{pK}_{sp} = 28.25
\] (4)

\[
\text{MgNH}_2\text{PO}_4\cdot6\text{H}_2\text{O} (\text{struvite}) = \text{Mg}^{2+} + \text{NH}_4^+ + \text{PO}_4^{3-} + 6\text{H}_2\text{O} \quad \text{pK}_{sp} = 13.26
\] (5)

The pK$_{sp}$ values were commonly used for thermodynamic predictions of phosphate precipitation (Lee et al., 2013; Hanhoun et al., 2011). The concentrations of the chemical components in Section 1.1 were used in the calculations, except those of Ca$^{2+}$ and Mg$^{2+}$. When the effluent was treated with 0.3 g/L of DL, 120 mg/L of Ca$^{2+}$ and 67 mg/L of Mg$^{2+}$ were added into the effluent. The total Ca$^{2+}$ and Mg$^{2+}$ concentration in treated effluent was 171 and 74 mg/L, respectively. The saturation index (SI) of possible phosphate precipitates formed in the system was calculated. SI is defined by Eq. (6).

\[
\text{SI} = \log(\text{IAP}/\text{K}_{sp})
\] (6)

where IAP is the ion activity product and K$_{sp}$ is the solubility product constant. For instance, SI = log ((Mg$^{2+}$)(NH$_4^+$)(PO$_4^{3-}$) / K$_{sp}$) for the struvite dissolution/precipitation reaction (Eq. (2)).

2. Results and discussion

2.1. Kinetics of phosphate removal

Fig. 1a shows the kinetics of phosphate removal from the anaerobic digester effluent by different contents of DL. The amount of phosphate removal increased when DL content increased from 0.1 to 0.8 g/L. The soluble phosphate concentration decreased rapidly within the first 5 hr and gradually reached equilibrium in 10 hr when DL content was 0.3–0.8 g/L. Phosphate removal by 0.1 g/L DL was much less efficient than by the higher DL content. It took about 30 hr for the removal of phosphate to reach equilibrium at a DL content of 0.1 g/L. Phosphate concentration decreased from 87 to 6.5 mg-P/L (92% removal) after the effluent water was mixed with 0.3 g/L of the DL for 7 hr. When DL content increased to 0.8 g/L, phosphate concentration only decreased slightly to 1.4 mg-P/L. The results indicated that the optimal dosage for the removal of the phosphate from the effluent water was 0.3 g/L.

A first-order kinetic model (Appendix A. Eq. (S1)) was used to fit the experimental data from the initial 10 hr in Fig. 1a following similar procedures to those reported by Nelson et al. (2003). The rate constant increased from 0.19 to 0.49 hr$^{-1}$ when DL content increased from 0.3 to 0.8 g/L (Appendix A Table S2). The rate constants were lower than the reported values in a range of 2–12 hr$^{-1}$ (Appendix A Table S2), which was because MgCl$_2$ salt used in those studies dissolved faster than DL (Nelson et al., 2003; Rahaman et al., 2008; Ohlinger et al., 2014). Phosphate removal rate could be controlled by dissolution of dolomite lime, while the removal rate in the MgCl$_2$ salt treatment systems was controlled by struvite precipitation.
The rate and amount of phosphate removal by lime (Ca(OH)2) were much lower than by the DL (Fig. 1a). The rate constant for the 0.3 g/L lime treatment system was 0.13 hr⁻¹ (Appendix A Table S2). The removal of phosphate reached equilibrium in about 20 hr and the soluble phosphate concentration was only reduced to 31 mg-P/L after the effluent water was treated with 0.3 g/L DL for 48 hr. The results indicated that the more effective phosphate removal in the DL treatment systems than in the lime treatment system was due to the formation of struvite.

The pH and soluble concentrations of Ca²⁺ and Mg²⁺ were measured during the kinetic experiments because pH affected the formation of phosphate precipitates (Nelson et al., 2003). The effluent pH was 7.5 ± 0.1. The addition of DL and lime not only provided the Mg²⁺ and Ca²⁺ for the precipitation of phosphate, but also increased the suspension pH. There was a steep increase in the suspension pH during the first 10 hr of treatment of the effluent water by DL and lime, followed by slight increase of pH over the next 38 hr (Fig. 1b). When the content of DL increased from 0.1 to 0.8 g/L, the final pH increased from 8.0 to 8.6. The increased pH should improve the removal of phosphate (Song et al., 2007).

Based on the amounts of DL (i.e., 0.1, 0.3, 0.5 and 0.8 g/L) added into effluent samples, the total soluble calcium concentration should increase from 51 mg/L in the effluent water to 91, 171, 371 mg/L, respectively. Fig. 1c shows that soluble Ca²⁺ concentration decreased rapidly within the first few hours and reached equilibrium after 24 hr. The equilibrium Ca²⁺ concentration decreased from 40 to 12 mg/L with increasing DL content from 0.1 to 0.8 g/L. The decreased Ca²⁺ concentration could be attributed to the increase in pH with increasing DL content. The equilibrium soluble Ca²⁺ concentration in the 0.3 g/L lime system was higher than that in all the DL-treated water, possibly due to the relatively low pH and high total Ca²⁺ content in the system.

In contrast to the decreased Ca²⁺ concentration, the final soluble Mg²⁺ concentration increased from 7 mg/L in the effluent to 16, 27, 67, and 119 mg/L when 0.1, 0.3, 0.5 and 0.8 g/L of DL was added into the effluent samples, respectively (Fig. 1d). Mass balance calculation suggested that the total soluble Mg²⁺ concentration should increase to 29, 74, 119, and 186 mg/L if 0.1 to 0.8 g/L of DL were completely dissolved. The results suggested that most of the Ca²⁺ and a large portion of the Mg²⁺ formed precipitates in the DL-treated effluent samples.

Kinetic experiments were also conducted to evaluate the dissolution of DL in DI water. The results in Appendix A Fig. S1 show that DL dissolution in DI water was much fast than in the effluent samples, which could be attributed to the formation of Ca²⁺ and Mg²⁺ precipitates with phosphate and carbonate in the effluent. In order to assess the feasibility of the DL treatment, continuous flow treatment tests were conducted using a bench-scale fluidized bed reactor (FBR) at hydraulic retention time (HRT) of 2 hr and dolomite lime dose 0.5 g/L. The results in Appendix A Fig. S2 indicate that P concentration was reduced from 95 to 20 mg/L.

2.2. Characteristics of the solids

The SEM images in Fig. 2 show the morphology of DL and precipitates formed in the treated effluent samples. DL existed as aggregates of small particles (Fig. 2a). When the effluent sample was treated with 0.3 g/L of DL at initial pH of 7.5, lots of large crystals were formed (Fig. 2b). More small particles and aggregates of small particles were formed at initial pH of 8.5 and 9.5 (Fig. 2c and d).
Fig. 3a and b show the EDS spectra of a large crystal and aggregates of small particles, respectively. The large crystal consisted of 25.9% P, 17.5% Mg, 2.5% Ca, and 54.1% O, while the main composition of the aggregates was 16.7% P, 33.2% Ca, 3.3% Mg, and 46.8% O. Some particles with relatively low P content were also detected. For instance, the aggregates in Fig. 3c contained 68.5% O, 2.7% Na, 5.5% Mg, 11% P and 12.3% Ca.

The XRD spectra of DL and the precipitates are presented in Appendix A Fig. S3, and detected minerals are listed in Table 1. The major minerals in DL were periclase, portlandite, and brucite, with very small amount of calcite, which suggested that CaO was completely hydrolyzed and MgO was partially hydrolyzed in the material. The molar ratio of Mg/Ca in the material was 1.14, which was similar to the theoretical Mg/Ca ratio in dolomite, CaMg(CO3)2. Only two major minerals (i.e., struvite and calcite) were detected in the solids produced when the effluent was treated with 0.3 g/L of DL at different pH to obtain the solids. The effluent samples were treated with 0.3 g/L DL at different pH to obtain the solids.

The XRD results indicate that struvite was the only phosphate-containing mineral formed in the treated effluent. Since the large crystal in Fig. 3a contained a high content of Mg and P and the Mg/P molar ratio was 0.86, it was an image of struvite. The Mg/P molar ratio was less than the theoretical ratio of 1.0 for struvite, which could be attributed to some Ca adsorbed or co-precipitated on the struvite surface. Pure struvite precipitates that are formed in chemical solutions containing Mg2+, NH4+, and PO43− are long needle-shaped crystals. The presence of Ca2+ could change the shape of struvite (Muster et al., 2013). The aggregates in Fig. 3b had high Ca2+ and PO43− content with a Ca/P molar ratio of 1.5. This could be amorphous calcium phosphate that could not be detected by XRD. Pastor et al. (2008) also reported the co-existence of struvite and amorphous calcium phosphate when solutions containing phosphate and ammonium were treated with both Mg2+ and Ca2+. Particles with high Ca and low P contents were also observed in the SEM/EDS analysis. They should be calcite with some adsorbed or coprecipitated phosphate on the surface.

2.3. Thermodynamic prediction of phosphate precipitates formed in the treated effluent

Thermodynamically, phosphate could form several stable phosphate precipitates with Mg2+, Ca2+, and NH4+ in effluent water treated with DL. The stable precipitates were determined by model calculations and were compared with the minerals detected in the experimental systems to understand the mechanisms controlling the removal of phosphate in the DL treatment system.

Fig. 4 shows the calculated SI of possible phosphate precipitates as a function of pH for the DL-treated effluent system. SI indicates the level or scale of saturation with respect to solid precipitates. A positive SI value indicates that the solution is supersaturated, while a negative SI indicates undersaturation conditions. The SI values of the phosphate
minerals are in the order: hydroxyapatite > Ca₄H(PO₄)₃·3H₂O > Ca₃(PO₄)₂(beta) > Ca₃(PO₄)₂(am2) > struvite > CaHPO₄ > Ca₄H(PO₄)₃·3H₂O in the pH range of 7.5–8.5. Hydroxyapatite is the least soluble precipitate among all possible precipitates in the system. Struvite is the fifth least soluble precipitate. Model calculations using PHREEQC program also predict that hydroxyapatite and Ca₃(PO₄)₂ are less soluble than struvite in chemical solutions containing PO₄³⁻, Ca²⁺, Mg²⁺, and NH₄⁺ (Muster et al., 2013). However, XRD analysis (Table 1) did not detect the formation of hydroxyapatite mineral and other calcium phosphate minerals in the DL treated effluent sample. Actually, struvite precipitated in the treated system. Struvite precipitate was also commonly found in anaerobic digester systems and solutions containing PO₄³⁻, Ca²⁺, Mg²⁺, and NH₄⁺ (Nelson et al., 2003; Barat et al., 2011).

Anaerobic digester effluent samples were collected at a municipal wastewater treatment plant in Beijing, China and treated with DL. Although the chemical composition of sample (Appendix A Table S3) were different from that in Section 1.1 and MINTEQ model calculations predicted hydroxyapatite and other calcium phosphate precipitates were more stable than struvite in the system (Appendix A Fig. S9), the major minerals formed in the DL-treated samples were struvite and calcite also (Appendix A Fig. S8).

Kinetic experiments were conducted to investigate the effect of NH₄⁺ on phosphate removal to understand the inconsistency between the model predictions and the experimental results. As shown in Fig. 5, when there was no NH₄⁺ in the synthetic effluent solution, only about 20% of the phosphate removed. The phosphate removal could be attributed to

Fig. 3 – The Energy Dispersive Spectrometer (EDS) spectra of precipitates of (a) a large crystal, pH = 7.5, (b) aggregates, pH = 7.5, and (c) aggregates, pH = 8.5.
the formation of amorphous calcium phosphate. When the synthetic effluent solution contained NH₄⁺, the phosphorus removal increased significantly to 65% under the same experimental conditions. Therefore, it was concluded that the high efficiency of phosphate removal could be achieved through the formation of struvite.

The discrepancy between the thermodynamic model-predicted formation of precipitates and the observed results was due to differences in kinetics of the rapid formation of struvite and slow formation of the calcium phosphate minerals. Several researchers reported rapid formation of struvite precipitate when anaerobic digester effluent and synthetic solutions were treated with MgO and magnesium salts (Nelson et al., 2003; Hanhoun et al., 2013; Quintana et al., 2005). Salimi et al. (1985) reported that struvite precipitation was kinetically favored over hydroxyapatite. Several publications reported that the formation of hydroxyapatite was preceded by other calcium phosphate precipitate precursors (Barat et al., 2009, 2011; Vankemenade and Debruyn, 1987).

Since the major precipitates formed in the DL treated effluent were struvite and calcite, the MINTEQ program was modified by allowing these two precipitates to form and excluding the other solid precipitates. The modified program was used to calculate the removal of PO₄³⁻, Mg²⁺, and Ca²⁺ at different pH. The solid and dashed lines in Fig. 6 are model predictions by the modified program and the original program, which allowed formation of all precipitates in the DL treated effluent system, respectively. Fig. 6a shows that the modified model prediction of PO₄³⁻ removal agreed reasonably well with the experimental data. Both experimental data and the model line showed an obvious decrease in PO₄³⁻ concentration with increasing pH from 7.0 to 9.0. The reported optimal pH range for removal of PO₄³⁻ with the formation of struvite was about 9.0–9.5 (Nelson et al., 2003; Laridi et al., 2005; Doyle and Parsons, 2002).

The original model over-predicted the PO₄³⁻ removal in the pH range of 7.0–8.2, because hydroxyapatite precipitated when pH was between 7.0 and 8.0 (Appendix A Fig. S5). The formation of hydroxyapatite prevented the formation of struvite at low pH. Hydroxyapatite, dolomite, and struvite precipitated in the pH range between 8.0–10.2. The original model predicted calcite would not be formed in the pH range of 7.0–11.0 because most of the Ca²⁺ formed hydroxyapatite and dolomite (ordered) precipitates.

The modified program accurately predicted the removal of Mg²⁺ as shown in Fig. 6b. Mg²⁺ was removed through the formation of struvite since dolomite was not allowed to precipitate in the calculations. The original model significantly under-predicted the Mg²⁺ removal when pH was lower than 8.7. In this system, Mg²⁺ was removed by dolomite

<table>
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<th>Mₛ/Mₛ⁺</th>
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* Ratio of struvite to calcite.
precipitate at pH < 7.9. Although dolomite (ordered) was less soluble than struvite, the formation of hydroxyapatite consumed much of the Ca\(^{2+}\), thus limiting the amount of dolomite precipitation. When pH was higher than 7.9, the amount of struvite increased significantly with increasing pH, which was the main contributor to the decreased Mg\(^{2+}\) concentration.

The modified program also described the Ca\(^{2+}\) removal data better than the original model, as shown in Fig. 6c. The original model significantly over-predicted Ca\(^{2+}\) removal, because the formation of hydroxyapatite and dolomite (ordered) precipitates consumed most of the soluble Ca\(^{2+}\) at pH < 7.9.

Çelen et al. (2007) detected the formation of struvite, CaHPO\(_4\), and Ca\(_4\)H\((PO_4)_3\)\(\times\)2H\(_2\)O in the treated liquid swine manure and employed Visual MINTEQ to simulate the experimental results (Çelen et al., 2007). The other two calcium phosphate precipitates, such as hydroxyapatite and Ca\(_3\)(PO\(_4\))\(_2\) and calcium carbonate precipitates were removed from the model calculations. The formation of calcium phosphate could be attributed to the high PO\(_4^{3-}\) concentration in swine manure. Muster et al. (2013) reported formation of struvite and hydroxyapatite in synthetic solutions at different Mg:Ca ratio and without the presence of carbonate. PHREEQC program was used to describe the mineral formation. The formation of hydroxyapatite could be ascribed to high concentrations of Ca\(^{2+}\) and PO\(_4^{3-}\) in the solutions and lack of the solute effects that exist in the wastewater.

Overall, the good agreement between the modified model calculations and the experimental data in Fig. 6 indicated that the removal of PO\(_4^{3-}\), Mg\(^{2+}\), and Ca\(^{2+}\) took place mainly through the formation of struvite and calcite precipitates. The model calculations also indicated that the amount of calcite precipitate increased significantly when pH increased from 7.0 to 9.0, which was consistent with the decreased mass ratio of struvite to calcite (M\(_s\)/M\(_c\)) with increasing pH shown in Table 1.

### 3. Conclusion

The experimental results obtained in this study suggest that DL can be used for effective removal and recovery of phosphate from anaerobic digester effluent. The phosphate removal in the DL-treated effluent was mainly attributed to the formation of struvite due to the rapid precipitation rate of the mineral. The solid product obtained in the DL treatment contains struvite as well as calcium carbonate while struvite product can be generated in the magnesium salt treatment processes. In order to produce solid precipitates with high struvite and PO\(_4^{3-}\) content in the DL treatment, the final pH of the treated water should controlled in the range of 8.0–8.5. Since the solid product obtained under optimal DL treatment conditions contained 89% struvite Mg(NH\(_4\))(PO\(_4\))(H\(_2\)O)\(_6\) and 11% calcite CaCO\(_3\), it can be used as soil conditioner. In addition to high content of P and N nutrients in the solids, calcite can neutralize acidic soil. Because the DL material is readily available at low cost, it can be used for broad and large-scale treatment of phosphate in wastewater.

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### Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.jes.2016.02.006.

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