Comparison of phosphorus recovery from incineration and gasification sewage sludge ash

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ABSTRACT

Incineration of sewage sludge is a common practice in many western countries. Gasification is an attractive option because of its high energy efficiency and flexibility in the usage of the produced gas. However, they both unavoidably produce sewage sludge ashes, a material that is rich in phosphorus, but which is commonly landfilled or used in construction materials. With current uncertainty in phosphate rock supply, phosphorus recovery from sewage sludge ashes has become interesting. In the present work, ashes from incineration and gasification of the same sewage sludge were compared in terms of phosphorus extractability using electrodialytic (ED) methods. The results show that comparable recovery rates of phosphorus were achieved with a single ED step for incineration ashes and a sequential combination of two ED steps for gasification ashes, which was due to a higher influence of iron and/or aluminium in phosphorus solubility for the latter. A product with lower level of metallic impurities and comparable to wet process phosphoric acid was eventually obtained from gasification ashes. Thus, gasification becomes an interesting alternative to incineration also in terms of phosphorus separation.

Key words | electrodialysis, heavy metals, incineration, low-temperature gasification, phosphorus, sewage sludge ash

INTRODUCTION

Increasing global food demand requires greater use of phosphorus (P) from phosphate rock (PR), a mineral which is unevenly distributed worldwide (USGS 2015) and whose geopolitical concerns increases the risk of supply disruption (Cordell & White 2014). Since P is non-renewable on a human time scale (van Dijk et al. 2016), there is a need for recycling technologies. In this context, P-recovery from secondary resources like sewage sludge ashes (SSAs) becomes an interesting alternative. Energy recovery processes, like incineration, have been adopted for some of the sewage sludge in developed economies like Europe, Japan or the United States (Samolada & Zabaniotou 2014). However, only a minor part of the resulting SSA is used in the production of fertilizers, due to the presence of heavy metals (HMs) (Ottosen et al. 2013; Ebbers et al. 2015; Parés Viader et al. 2015) and the high content of Al and Fe, which makes it unsuitable for traditional processing methods of PR (Scholz et al. 2014). Thus, around 500,000 Mg of mono-incineration SSA are produced per year in Germany, the biggest producer in the European Union, most of which is currently landfilled or used in construction materials (Krüger et al. 2014, 2015). In this country, as well as in other major SSA producers like the Netherlands and Switzerland, there have been recently several agreements in order to set up pilot plants in the coming years, so as to investigate the recovery of P from incineration SSA in an industrial usable form, like H3PO4 solutions.

Gasiﬁcation of sewage sludge is an interesting alternative to incineration because of a high energy efficiency, less extensive gas cleaning to avoid air pollution and a more ﬂexible use of the syngas produced, which can be employed in several processes like combined heat and power or chemicals production (Ahrenfeldt et al. 2015; Samolada & Zabaniotou 2014). So far, gasiﬁcation of sewage sludge has not been as extended as incineration, due to the complexity of the technology and its investment and operational costs (Samolada & Zabaniotou 2014); for instance, there are only two ﬂuidised bed gasiﬁcation
plants in Germany, with a combined production of less than 6,000 tonnes of SSA per year (Krüger & Adam 2015). Moreover, the direct application of gasification biochar/SSA as fertilizers can also be limited by the HM content, as well as due to the low plant availability of P.

Previous research focused on the characterisation of SSA from incineration (Franz 2008; Donatello et al. 2010; Ottosen et al. 2013) or gasification (Hernandez et al. 2011; Martinez et al. 2014; Parés Viader et al. 2015), including element composition and acid extraction of P and other elements. Electrodialytic (ED) methods have also been investigated to recover P from incineration and low-temperature (LT) gasification SSA with high content in Fe and Al, and separate it from impurities like HMs (Parés Viader et al. in press). ED setups comprise several compartments delimited by ion exchange membranes; by applying an electrical current, the different elements are released from the bulk ash, and positively charged metallic species are separated from the P anions in the different compartments through electromigration. The setup used differed for each ash: up to 96% of P-extraction was achieved for incineration SSA with a two-compartment (2C) ED setup (Ebbers et al. 2015), whereas the recovery was around 70% for LT gasification SSA (Parés Viader et al. in press) using a sequential ED process (Figure 1(a)). However, the sewage sludge originally used in each thermal treatment was different, and so was the composition of the resulting ashes; in the case of P, its content was considerably higher for incineration (12.3%) than for the LT gasification SSA (3.0%). The present study compares the chemical composition, and acid and alkaline leachability, as well as the ED P-recovery from SSA produced either by incineration or LT gasification of sewage sludge from the same wastewater treatment plant (WWTP).

**METHODS**

**Experimental SSA**

Two different SSAs were produced for this research using the same feedstock: sewage sludge collected at the WWTP in the municipality of Bjergmarken, Denmark, where P was captured, approximately, 70% biologically and 30% chemically with Fe chloride sulphate. Al chloride was also used to flocculate and precipitate filamentous bacteria in the sedimentation tank, and its dosing varied during the days that the sewage sludge samples were collected.

Incineration SSA was collected from the mono-incineration plant in Avedøre Wastewater Service, Denmark, in a fluidised bed combustor at around 840 °C after the sewage sludge was mechanically dewatered.

Gasification SSA was collected from the bottom of the char reactor of a 100 kW experimental LT circulating fluid bed gasifier at the Department of Chemical and Biochemical Engineering, Roskilde campus of the Technical University of Denmark, after the sewage sludge was dried. It was the same gasification unit used in previous research (Hansen et al. 2015; Parés Viader et al. 2015, in press).

**Analytical methods and sampling**

One batch of around 300 grams was sampled from the total mass collected, dried at 105 °C overnight, cooled in the fume hood, homogenised with a steel spatula and immediately stored in plastic bags and under dry conditions. Gasification SSAs were loosened with a mortar to facilitate ED experiments.

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**Figure 1** | Schematic view of the sequential ED process (a) and the split ED setup (b). C: cations, A: anions.
The concentration of elements was investigated for 15 samples of each untreated ash, and in triplicate after each ED experiment. The targeted elements were Al, As, Ca, Cd, Cu, Cr, Fe, Mg, Ni, P, Pb, S and Zn. Their content was measured by Varian 720-ES inductively coupled plasma optical emission spectrometry (ICP-OES) after pre-treatment according to Danish Standard 259:2003 (DS259): 1 g ash and 20 mL 7.3 M HNO₃ were heated at 200 kPa (120 °C) for 30 min. The liquid was thereafter separated by vacuum filtration through a 0.45 μm filter and diluted to 100 mL with deionised (DI) water.

The mineralogy of both SSAs was examined using X-ray powder diffraction (XRPD) with a PANalytical X'Pert PRO, and the results were analysed using X'Pert HighScore Plus with ICDD PDF 2 database.

**pH release**

In order to assess the solubility of the target elements under acid or alkaline solutions, 2.5 g of ash was shaken at 150 rpm for 1 week with 25 mL of HNO₃ or NaOH in duplicate at 1, 0.7, 0.5, 0.3 and 0.1 M respectively. The same procedure was repeated for DI water. The pH of each leachate was measured by a PHM220 Lab pH meter. Content of the targeted elements in the leachate was measured by the Varian 720-ES ICP-OES after filtration through a 0.45 μm syringe filter, and compared to the total amount in the untreated ashes as described in the previous section.

**ED experimental setup**

An illustration of two of the ED methods used is shown in Figure 1. The sequential process (Figure 1(a)) was similar to a previous research (Parés Viader et al. in press). The first step cell consisted of two compartments made of cylindrical Plexiglas® with an internal diameter of 8 cm separated by anion exchange membrane (AEM); the anode compartment was 10 cm long and contained 35 g ash and 350 mL DI water. The second step cell consisted of two compartments of identical characteristics to the first step separated by a cation exchange membrane (CEM). The anode compartment contained 20–25 g of the bulk ash resulting from the first step and 300 mL of NaOH 0.15 M. A plastic strip attached to a glass-stick connected to an overhead stirrer (IKA RW11) was used to stir the ash suspensions during the experiments. In the cathode compartment of the first step, 500 mL electrolyte solution (0.01 M NaNO₃, pH < 2 adjusted with HNO₃) was circulated by a Plastomec pump model P05. In the anode compartment of the second step, the anolyte resulting from the first step was circulated using the same pump model after vacuum filtration through a 0.45 μm filter.

The split setup (Figure 1(b)) consisted of three compartments made of Plexiglas® with an internal diameter of 8 cm, in which the middle compartment was 10 cm long and contained 35 g ash and 350 mL DI water. One electrode was placed in each compartment, and the cathode was simultaneously connected to two anodes by means of two power supplies. An AEM was placed between the two anode compartments, whereas a CEM was placed between the middle and the cathode compartment. In the end anode compartment, 500 mL of 0.01 M H₃PO₄ was circulated, whereas in the cathode compartment 500 mL of electrolyte solution (0.01 M NaNO₃, pH < 2 adjusted with HNO₃) was circulated, in both cases using the same model pump as before. With this setup, it was expected to achieve:

(a) a similar degree of cationic migration as in a 2C setup, as the electrical current at the cathode was the same (50 mA, see next section);

(b) less solubilisation of metals with lower acid-leachability than Ca (like Fe or Al), since the ash suspension was subjected to half the current (25 mA) than in the 2C setup, and thus a lower proton load generated at the anode by electrolysis.

In order to accelerate the pH decrease at the beginning of the split ED experiments, 50 mA was applied only between the anode in the middle compartment and the cathode for the first 6 (incineration SSA) and 48 hours (gasiﬁcation SSA).

The electrodes were made of platinum-coated titanium wire (diameter 3 mm) obtained from Permascand®. An Agilent E3612A DC power supply was used to maintain a constant DC current. The CEM and AEM used were from Ionics (model CR67 and AR204SZRA respectively).

**ED experiments**

Eleven sets of ED experiments were made (Table 1): five for incineration (I1-I5) and six for gasiﬁcation SSA (G1-G6). For the former, all experiments consisted of a single step, whereas a sequential process was used in four experiments for the latter. The intensity was 50 mA, chosen following the findings from previous research using ED on incinerated and gased SSA (Parés Viader et al. in press). Voltage between the electrodes was monitored during the whole experiment. The pH in both compartments and the
conductivity of the ash suspension were measured twice a
day during the ED experiments. The pH of the catholyte
of the first step was adjusted to <2 with HNO₃ 5 M, to
avoid precipitations due to the production of OH⁻/CO₂
by the electrode reaction at the inert cathode.

At the end of each step, the ash suspension was
filtered at atmospheric pressure, and for the first step it was
flushed with 200 mL of DI water in order to displace the elements in
solution retained in the humidified ash. The resulting liquid
volume was measured, and the ash was dried for 48 hours at
50°C. The dried ash was loosened by hand in a mortar and
stored in plastic bags and under dry conditions. The electro-
des were rinsed in 5 M HNO₃, the membranes in 1 M
HNO₃. Catholyte and anolyte samples were taken in
20 mL vials before applying the electrical current and after
5 minutes of electrolyte circulation; at the end of all exper-
iments, the catholyte and anolyte volumes were measured,
and samples for each were also taken in 20 mL vials. The
concentrations of the elements were measured in all of the
liquids by ICP-OES after filtration through a 0.45 μm syringe
filter.

Experimental ED parameters

The ED experiments were evaluated in terms of rates of
P-recovery and the impurity level of the obtained product.
The amounts of P found in the anolytes of the first and
second steps were considered as recovered, since they
were expected to be solutions rich in this element and
depleted of impurities like metallic cations. The overall per-
cent of P recovered was calculated as:

% P-recovery = \frac{P \text{ in anolytes}}{\text{Total P at the end of the experiment}}

And the amount of P in the anolytes was calculated as:

V_{1\text{st step}} \cdot C_{P-1\text{st step}} + V_{2\text{nd step}} \cdot C_{P-2\text{nd step}} \cdot \frac{m_{\text{ash-1st step}}}{m_{\text{ash-2nd step}}}

where \( V_{1\text{st step}} \) and \( V_{2\text{nd step}} \) were the volumes of the anolytes,
\( C_{P-1\text{st step}} \) and \( C_{P-2\text{nd step}} \) the concentration of P in the ano-
lytes at the end of each ED step, whereas \( m_{\text{ash-1st step}} \) was
the mass of dry ash after the first step and \( m_{\text{ash-2nd step}} \) the
mass used in the second step. The inclusion of the ratio
between \( m_{\text{ash-1st step}} \) and \( m_{\text{ash-2nd step}} \) multiplying the
second term was used to extrapolate the result of the second
ED step to all the bulk dry ash mass after the first step.

In order to assess the amount of impurities in the recov-
ered P, it was compared to WPA, the product of leaching PR
with sulphuric acid (Gilmour 2014), which is applied to over
70% of mined PR (Scholz et al. 2014). A common parameter
to evaluate the level of impurities of WPA, and its suitability
in the fertilizer industry is the minor element ratio (MER),
defined as:

\[ \text{MER} = \frac{\text{Fe}_2\text{O}_3 + \text{Al}_2\text{O}_3 + \text{MgO}}{\text{P}_2\text{O}_5} \]

RESULTS AND DISCUSSION

SSA characterisation

Element concentrations of the two untreated ashes are shown
in Table 2. Most elements presented a high standard deviation,
as can be expected in inherent inhomogeneous materials. Cd
concentrations in the liquids from the DS259 pre-treatment
of gaseification ash were found to be below the limit of detection
(LOD, 0.02 ppm) in 12 out of 15 samples, and therefore the
concentration is given as a range, and no mass balance or
pH release result is shown for this element.

Average concentrations of Ca, P, Fe and Mg for inciner-
tion compared to gaseification ashes were in the range of
±10%, and therefore a fair comparison of both SSAs in
terms of P leachability and ED-recovery can be made. Differences in Al are probably attributable to differences in the
Table 2 | Concentration (average ± standard deviation) of elements in the two studied SSAs

<table>
<thead>
<tr>
<th>Element</th>
<th>Incineration SSA</th>
<th>Gasification SSA</th>
<th>German incineration SSA, average values (Krüger et al. 2014)</th>
<th>Gasification SSA values in literature (Gil-Lalaguna et al. 2015; Hernandez et al. 2011)</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Al</td>
<td>28.3 ± 4.4</td>
<td>51.2 ± 1.1</td>
<td>52</td>
<td>23–61</td>
<td>g/kg</td>
</tr>
<tr>
<td>Ca</td>
<td>132 ± 47</td>
<td>135 ± 27</td>
<td>138</td>
<td>84–148</td>
<td>g/kg</td>
</tr>
<tr>
<td>Fe</td>
<td>63.2 ± 13</td>
<td>61 ± 1.8</td>
<td>99</td>
<td>88–123</td>
<td>g/kg</td>
</tr>
<tr>
<td>Mg</td>
<td>11.9 ± 2.2</td>
<td>12.9 ± 1.3</td>
<td>14</td>
<td>25–36</td>
<td>g/kg</td>
</tr>
<tr>
<td>P</td>
<td>88.4 ± 21</td>
<td>96.0 ± 15</td>
<td>90</td>
<td>51–149</td>
<td>g/kg</td>
</tr>
<tr>
<td>S</td>
<td>8.77 ± 2.1</td>
<td>5.58 ± 0.5</td>
<td>15</td>
<td>0.2–41</td>
<td>g/kg</td>
</tr>
<tr>
<td>As</td>
<td>11.4 ± 1.4</td>
<td>5.82 ± 0.33</td>
<td>17.5</td>
<td>N/A</td>
<td>mg/kg</td>
</tr>
<tr>
<td>Cd</td>
<td>4.19 ± 2.8</td>
<td>&lt;2–3.6</td>
<td>3.3</td>
<td>&lt;LOD</td>
<td>mg/kg</td>
</tr>
<tr>
<td>Cr</td>
<td>57.7 ± 4.3</td>
<td>89.2 ± 7.4</td>
<td>267</td>
<td>98–137</td>
<td>g/kg</td>
</tr>
<tr>
<td>Cu</td>
<td>588 ± 43</td>
<td>479 ± 31</td>
<td>916</td>
<td>1,159–1,367</td>
<td>mg/kg</td>
</tr>
<tr>
<td>Ni</td>
<td>58.1 ± 3.7</td>
<td>57.9 ± 9.7</td>
<td>105.8</td>
<td>122–165</td>
<td>mg/kg</td>
</tr>
<tr>
<td>Pb</td>
<td>208 ± 100</td>
<td>137 ± 87</td>
<td>151</td>
<td>51–90</td>
<td>mg/kg</td>
</tr>
<tr>
<td>Zn</td>
<td>2,120 ± 230</td>
<td>1,650 ± 250</td>
<td>2,535</td>
<td>753–877</td>
<td>mg/kg</td>
</tr>
</tbody>
</table>

The results are compared with the element concentrations found in literature for incineration SSA and gasification SSA.

dosing of aluminium chloride in the WWTP (see section ‘Experimental SSA’). S and HM contents were on average slightly higher for incineration SSA, which is probably because they were produced by mixing of fly ashes and bottom ashes. In contrast, gasification SSA consists only of the bottom ash fraction; the higher concentration for Cr is most likely due to releases from the materials used in the gasifier, although it requires further study.

Incineration SSA concentrations were compared to the average values from a survey of incineration SSA in Germany (Krüger et al. 2014). P-concentrations were found to be very similar, considering only values from municipal SSA. Differences in Al and Fe can be due to a higher dosing of Al and Fe salts in German WWTPs. As, Cr, Cu, Ni and Zn concentrations were on average higher in German SSA, which is because the survey included a mean value including both municipal and industrial SSA. Concentrations in the gasification ashes of this study were compared to previous research on gasification SSA (Hernandez et al. 2011; Gil-Lalaguna et al. 2015). On average, Al, Ca, P, S and Cr contents were in the range of literature values, whereas the rest of the elements were below it, except for Pb and Zn. The reason for the lower Fe concentration can be again a lower dosing of Fe salts in Bjergmarken WWTP.

The mineralogy study showed two common phases for both SSAs: quartz (SiO2) and calcium phosphates (Ca9Fe (PO4)7, Ca9Al(PO4)7 and/or Ca7Mg(PO4)6). For incineration SSA, haematite (Fe2O3) and anhydrite (CaSO4) were also identified. These phases were respectively identified in other incineration (Franz 2008; Donatello et al. 2010; Ottosen et al. 2014; Li et al. 2015) and gasification SSAs (Hernandez et al. 2011; Gil-Lalaguna et al. 2015). The presence of Fe2O3 in incineration SSA is due to the reaction of iron phosphate with lime, and the consequent formation of more acid-soluble calcium phosphate (Martinez et al. 2014; Li et al. 2015). In contrast, the lack of haematite in gasification SSA could indicate the presence of FePO4, although it was not identified either in the XRPD analyses of the present work or in previous studies on gasification SSA (Hernandez et al. 2011; Gil-Lalaguna et al. 2015). One reason could be that it was found in an amorphous phase, since there was a hump in the pattern from 10° to 40° (2θ), but it requires further study.

**pH release experiments**

The solubilisation of all elements generally increased with lower pH values for both incineration and gasification SSA (Figure 2). P leaching was lower for gasification than for incineration SSA on each acid concentration, except for the highest acid load (1 M HNO3). In fact, P was not completely solubilised in this acidic extraction (88% with pH 0.7) for incineration ashes; this is in contradiction with the results obtained in previous studies (Ottosen et al. 2013), where most P was extracted at pH ~ 1, and it is most likely due to the inherent inhomogeneity of the ashes. P-concentrations were 8.1·10⁻³ M at pH 2.4 for incineration, and 3.4·10⁻² M at pH 1.9 for gasification ashes; Fe(III), Al- and Ca-phosphates solubility at pH 2 are in the range of 10⁻⁵–10⁻⁴ M, 10⁻³–10⁻²
and >1 M, respectively (Stumm & Morgan 1996; Kuroda & Okido 2012). Thus, P release is influenced simultaneously by these three metals for both SSAs at low pH, with a higher influence of Al and Fe in gasification than in incineration ashes. At high pH values, P solubilised for both SSAs, but to a higher degree for gasification. This suggests the presence of alkaline-soluble P bindings, like Fe(III)-P and/or Al-P, as previously seen for other LT gasification SSAs (Parés Viader et al. in press). Nevertheless, P-solubility for gasification SSA at pH 1.9 (5.4·10^{-2} M) was one order of magnitude higher than in these LT gasification ashes at pH 2.1 (2.6·10^{-3} M) (Parés Viader et al. 2015), probably because of a higher proportion of chemically captured P (50%) in the original sewage sludge of the latter. For incineration SSA, Fe leaching was lower than in gasification ashes in all acidic pH values, which can be due to the formation of acid-insoluble iron oxides from Fe-phosphates in the former (see section ‘SSA characterisation’).

The concentrations of most HMs in the acidic leachates of incineration ashes were higher than the values of gasification ash leachates, with As, Pb and Zn being at least twofold higher, for the same HNO₃ load. In contrast, Ni leachability in all acid tests was around one order of magnitude higher in gasification SSA; the results for incineration SSA at pH 0.7 (7.5%) were also lower than the results from a previous work with similar incineration SSA at around pH 1 (~35%) (Ottosen et al. 2015). Leachate concentrations under alkaline extractions were higher for As in incineration and for Ni in gasification SSA, while the rest of HMs, together with Fe and Al, leached below 5% for both SSAs, probably because of the formation of metallic hydroxides.

**ED experiments**

**Mass balances**

The element distribution in the different compartments at the end of the experiments was determined, considering the following.
The catholyte, the cathode and the cationic membrane were grouped as the cathode compartment in the first step. In the second step, the cathode compartment comprised the cathode and the catholyte.

The total mobilisation or release of each element, including P, was the sum of the masses found in all electrolyte solutions, membranes and electrodes at the end of the experiments minus the initial concentrations of these elements in these compartments.

In the split experiment I3, the P in both anolyte liquids (mid-compartment and metal-depleted) was considered as recovered. For split experiment G4, only the P in the metal-depleted anolyte was considered as recovered since its amount was over 50 times higher than the one found in the mid-compartment anolyte. Furthermore, the ratio of metals to P in the anolytes of the split experiments was calculated by the difference between the concentrations at the end and at the beginning of the experiments.

The amount of each element migrated to the catholyte in the first step and to the anolyte in the second step was defined as the difference between each element amount in the electrolyte solution at the end and the amount at the start of the step.

Mass balances for each element (except Cd), defined as the division of the amount found at the end of the experiments by the mass initially found in the initial SSA and the liquids placed in the ED cell, were made to control the quality of each step of the experiments. Most elements in each step were found in the range or around 80–120%, except for Pb (52–144%), which is due to its inhomogeneity in both SSAs (Table 2). Concentration measurements which were below the LOD used in the ICP-OES analyses (0.02 mg/L) were considered to have this value, except for the initial first-step catholyte samples, which were considered to be zero. This enabled calculation of the mass balances and the worst-case value for the ratio HM/P in the anolytes, which was the recovered P-product. The overall impact of these approximations was below 1% of the total final element mass found at the end of the ED experiments.

ED cell voltage, ash suspension conductivity and pH

The voltage between electrodes decreased during the first step of most ED experiments, in parallel to the increase in electrical conductivity. At the same time, the pH decreased in the SSA suspensions to 2 ± 0.3 for all experiments except for I5 (1.3) and G4 (2.4). The main reason for this was the generation of protons in the anode together with the element release from the ashes. The sole exception was experiment G4, as the voltage in both power supplies increased from 8 to 17 V between 48 and 96 hours, and decreased afterwards to 10 V. This increase suggests the migration of ions other than protons from the suspension, which entailed a decrease in conductivity, since its pH was stable in the range 2.4–2.8 after 48 hours. In the second step of G1–G4, the voltage increased after approximately 24 hours from ~7 V to ~25 V and decreased progressively the following 24 hours to ~7 V. This can be caused by a temporary fouling of the membrane, which was not observed in previous experiences (Parés Viader et al., 2015, in press), and needs to be addressed in future research.

In all experiments, the pH increased steadily up to ~12 after 48 hours, and reached ~12.5 at the end of the experiments.

P-recovery in ED experiments

Figure 3 shows that a higher extraction of P to the anolytes was achieved with longer experimental times using the 2C setup (not considering the split setup) for incineration SSA, with the exception of the 5-day experiment. Moreover, there was a sharp increase between the seventh and the eighth day tests. Since Ca represented more than 80% of the mobilised metal mass in all experiments, the reason can be a higher amount of P (Figure 3) and a higher proportion of Ca-P bindings over other Ca-compounds for the ashes in experiments I1 and I5: the molar ratio of mobilised Ca to P (mobilised Ca/P) was around 1.4, close to the one of tricalcium phosphate (1.5), and lower than for I2 (2.2) and I4 (2.0). Thus, there was a lower content of compounds competing with Ca-P dissolution, like CaSO₄ or CaCO₃, in I1 and I5. For gasification SSA, higher P-extractions were also generally achieved at longer remediation times in the first step (not considering the split setup, Figure 3). Nevertheless, the amount of P extracted was lower for gasification ashes after 5, 6, and 7 days in the first step than after the treatment of incineration ashes with the same times (Figure 3). The reason can be the higher influence of Fe(III) and Al in P-solubility in the gasification SSA (see section ‘pH release experiments’). In fact, most P in the bulk SSA after the first step would be bound to Fe (III) and/or Al, as molar Ca/P ranged from 0.3 (G5 and G6) to 0.6 (G1), well below the value of tricalcium phosphate (1.5). Higher remediation time in the ED experiment increased the amount of P migrating to the cathode compartment (Figure 3) and the MER in the anolyte (Table 3). This was probably due to a higher solubilisation of Fe(III), which enabled the formation of positively charged
complexes with phosphate ions, preventing their effective separation (Parés Viader et al. 2014). Therefore, in order to increase P-recovery, a sequential ED process needs to be used (Figure 3(a)), so that Fe(III)-P bindings are solubilised through an increase of pH, and Fe(III) is precipitated as hydroxides. A second step was applied to the ashes of the...
three shortest remediation times (G1–G3), increasing the extraction to the levels achieved with incineration ashes (Figure 3). For G1, the amount of P recovered was higher than for G6 with the same total duration of the ED treatment (9 days). The molar Ca/P in the bulk gasification SSA after the second step was around 1.2 for the three experiments, which could be translated into an 80% of P bound to Ca. In consequence, most existing Fe(III)-P and Al-P bindings were effectively dissolved. The achieved P-recoveries were up to 80% (I5) from incineration ash and up to 69% (G2) for gasification ashes, which are similar to previous ED recoveries achieved on incineration and LT gasification SSAs (Ebbers et al. 2015; Parés Viader et al. in press). All recoveries can be seen in Table 3.

Characteristics of the recovered P-liquids

The resulting anolytes were acidic (pH < 2), and the achieved P-concentrations were higher than in previous studies, where the concentration was below 2 g P per litre (Parés Viader et al. in press), but several orders of magnitude below the concentration found in WPA (Table 3), which needs to be addressed together with the high proportion of S in the solution.

The ratios of Al, Fe, and Mg to P in the resulting P-liquids were generally higher for incineration than for gasification ashes (G1-G4, Table 3). This is due to the lower solubility of Al in gasification than incineration SSA at pH ∼ 2 (Figure 2) and the use of the second step, since Fe and Mg were not soluble at pH ∼ 12 (see Figure 2 and section ‘pH release experiments’). An adaptation of the sequential ED process to treat incineration SSA could therefore be beneficial, but it requires further study. However, MER values achieved using 2C setups were always above 0.085, the technical limit of WPA to be used in diamonium phosphate (DAP), the world’s more common mineral fertilizer (Gilmour 2014). This was also observed in previous ED treatment of incineration SSA (Ebbers et al. 2015). In order to reduce this ratio, the split setup (Figure 1(b)) was used for both ashes (I3 and G4); for gasification, as a first step. The amount of P extracted with this setup was in line with the other results (Figure 3), and at the same time a lower MER (and below 0.085, Table 3) was achieved.

HM/P ratios in the anolytes were below the initial SSA ones (calculated from the average values in Table 2) and, for some elements, in the range or below the values found in WPA. Remarkably, some P-liquids from gasification ash had only Pb/P values above these ranges. Overall, the ratios of As, Cr, Cu, Ni, Pb and Zn to P were higher in incineration than in gasification ashes. This is probably due to the lower solubility of some of these metals in acid medium in the latter, as well as the use of the second ED step (experiments G1-G4), where most HMs became insoluble due to the alkaline conditions (see section ‘pH release experiments’). Additionally, the higher concentration of S in the anolyte solutions of incineration SSA (Table 3) could enable the formation of neutrally charged sulphate complexes with Cu, Ni, Pb and Zn, like NiSO₄ (aq), although it requires further study. The use of a split setup allowed generally lower ratios of Cr, Cu, and Zn to P; especially for gasification ash, because the final pH in the ash compartment (2.4) was higher than the other first-step experiments with the 2C setup (~2).

CONCLUSIONS

Two different ED methods were used in order to achieve comparable extraction of P from ashes of incineration and gasification of the same sewage sludge. For the former, a 2C setup allowed a recovery of up 80%; for the latter a sequential ED process was required to recover up to 69% of P, because of a higher presence of Fe(III)-P bindings and in order to prevent the formation of Fe-phosphate complexes. In general, higher remediation times allowed higher amount of recovered P, although this was influenced by the inhomogeneity of the ashes. Using these techniques, the resulting H₃PO₄ solutions from gasification ash had overall lower ratios of Al, Fe, Mg and HMs to P than from incineration ash. This was possible due to lower solubilisation of most HMs in the gasification SSA, as well as the use of the additional step in the sequential ED process which involved P-extraction at high pH, where most metals are insoluble.

The content of Al, Fe, Mg and some of the HMs was high in these liquids compared to WPA, which could complicate their usage in the production of commercial fertilizers like DAP. A considerable reduction of these impurities was obtained by using a new ED setup (split), making it an interesting alternative to separate P from SSA, especially because Fe concentration in commonly produced SSA tends to be higher than in the ashes of the present study.

REFERENCES


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