

# Phosphorus partitioning in co-dewatering biosolids and water treatment residuals

Herschel A. Elliott and Malcolm Taylor

## ABSTRACT

Stabilization and dewatering methods for wastewater solids determine the concentration and nature of phosphorus (P) in biosolids and in-plant sidestreams recycled to the liquid treatment facility. Because water treatment residuals (WTR) exhibit strong immobilization of soluble P, this study evaluated the impact of co-dewatering WTR and biosolids on the P partitioning during dewatering and the environmental lability of biosolids-P measured by water-extractable P (WEP). Overall, P progressively partitioned into the water-insoluble particulate-bound form in dewatered cake with increasing blending ratio (BR) – defined as the dry mass ratio of WTR to biosolids. The reject water total P (TP) content from dewatering biosolids alone ( $250 \text{ mg L}^{-1}$ ) was reduced to  $60 \text{ mg L}^{-1}$  for a BR = 1.5. Polymer addition resulted in statistically ( $\alpha = 0.05$ ) lower reject liquid TP, suggesting the cationic polyelectrolyte contributed to P binding. The WEP of the dewatered cake (~20% solids) dropped from  $2.36 \text{ g kg}^{-1}$  (biosolids only) to  $\sim 0.14 \text{ g kg}^{-1}$  for BR = 1.5, meaning the P in land-applied co-processed cake is less susceptible to solubilization by surface runoff compared to unamended biosolids. Co-dewatering can reduce P in return flows and fix P in the dewatered solids in a form less prone to off-site migration following land application.

**Key words** | biosolids, phosphorus, water treatment residuals

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## INTRODUCTION

Concerns over eutrophication of surface waters have led to regulations that are having a major impact on wastewater treatment and biosolids management. Increasingly stringent effluent discharge limits for phosphorus (P) has resulted in implementation of processes that incorporate P into chemical precipitates or as intracellular polyphosphates in microorganisms, thereby increasing the P content of solids settled during wastewater treatment. How this P is managed within the treatment plant can greatly impact plant operations and performance. For example, anaerobic digestion causes near complete release of P from solids generated by enhanced biological P removal (EBPR) processes (Pöpel & Jardin 1993; WEF 2005). These nutrients end up in the discharged liquid during dewatering (reject water) and are usually returned to the liquid treatment process. Nutrient-rich reject water can adversely affect the operation and performance of liquid treatment processes, particularly when slug loads associated with intermittent dewatering operations are returned to the headworks (Phillips *et al.* 2006).

Various strategies have been explored for addressing the problem of high P in digester supernatant. Addition of alum, ferric salts or lime can be used for chemical precipitation of P in digester supernatant to reduce the amount of P returned to the liquid treatment facility. More recently, processes have been developed to intentionally precipitate struvite ( $\text{MgNH}_4\text{PO}_4(\text{s})$ ) from nutrient-rich sidestreams (Lin & Pan 2011). Sidestream treatment processes require a clarifier or other solids separation device to remove the P-rich solids (Phillips *et al.* 2006).

In-plant solids management processes also influence the ability of a municipality to beneficially recycle biosolids through agricultural land application. The runoff and leaching potential of P in the biosolids recycled by land application depends on how the solids are generated and stabilized. Maguire *et al.* (2001) found that P in soils was less environmentally labile (i.e., water soluble) when amended with biosolids high in Al and Fe from chemical P removal during wastewater treatment. Phosphorus in biosolids produced in EBPR facilities has relatively high

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potential for environmental loss following land application (Elliott *et al.* 2005). Processes that produce relatively dry biosolids (heat treatment, composting, lime post-treatment) tend to reduce that water extractability of the biosolids P (Brandt *et al.* 2004). The nature of P in biosolids is important because the imposition of evolving P-based nutrient management policies threatens the sustainability of land-based biosolids recycling programs (Elliott & O'Connor 2007).

Previous studies (Lai & Liu 2004; Yang *et al.* 2007; Taylor & Elliott 2013) have shown that blending drinking water treatment residuals (WTR) into digested wastewater biosolids improves dewaterability and reduces polymer requirements. There is abundant evidence that WTR produced from addition of alum and Fe salts are effective P sorbents that can be used to treat P-enriched wastewaters (Mortula & Gagnon 2007) and significantly reduce the environmental reactivity of P (Ippolito *et al.* 2011). Yang *et al.* (2007) demonstrated that co-conditioning of anaerobically digested sludge and alum sludge has the added benefit of reducing P in the reject water recycled in wastewater treatment. This study was conducted to further assess the influence of co-dewatering WTR and biosolids on the distribution of P between the reject water and the dewatered cake solids. Mass balance calculations were performed to quantify the partitioning of P during dewatering as a function of the WTR-to-biosolids blending ratio. Since P runoff following land application is correlated to the water-extractable P (WEP) content of the biosolids (Elliott *et al.* 2005), the WEP of the dewatered cake solids was also evaluated. Implications for in-plant nutrient management and land-based recycling of the biosolids are explored.

## METHODS

### Solids

The wastewater solids were collected at the City of York (PA, USA) wastewater treatment plant (WWTP) that employs EBPR for nutrient control. Primary and waste activated sludge are combined, thickened by floatation, and anaerobically digested. Digested sludge samples (~1.3% solids) were collected from the sludge holding tanks prior to polymer addition and dewatering by centrifugation. Because the digested solids at the point of collection meet the United States Environmental Protection Agency (USEPA) pathogen and pollutant requirements for land application, hereafter the material is referred to as biosolids. The alum-based WTR (Al-WTR) was collected from the York Water Company Grantley Road facility, located in York County, PA, which uses

aluminum sulfate as the primary coagulant. The Al-WTR were collected from a thickening tank prior to chemical addition and dewatering. The Al-WTR (4–5% solids) were a composite of solids from the sedimentation basins and filter backwashing operations. Elemental content of the biosolids and Al-WTR were determined by acid digestion followed by analysis via inductively coupled plasma atomic emission spectroscopy (ICPAES) using USEPA methods 3051 and 6010B (USEPA 1994). Nitrogen was determined by Kjeldahl analysis. Solids content, specific gravity, and pH (1:1 solids to distilled water) were determined by standard procedures (APHA 1998). Specific gravity was measured to facilitate volume-to-mass conversions. As a measure of the reactivity of Al in the Al-WTR, the oxalate-extractable Al content (Al<sub>ox</sub>) was determined by the method of Dayton & Basta (2005).

### Blending experiments

The polyelectrolyte used at the York WWTP, Pollu-treat CL-455<sup>®</sup> from (Pollu-tech Inc.), was also used in the laboratory experiments. This polymer is a branched chain, cationic product commonly used for 'hard to dewater' slurries. A polymer solution was prepared by diluting concentrated polymer with distilled water to a concentration of 0.2% (1 mL polymer to 499 mL water). The diluted solution was slowly mixed for 45 min until no visible clumps of concentrated polymer remained. Polymer solution was used within 48 h of preparation.

Appropriate volumes of WTR were combined with 200 mL of biosolids to produce blend ratios (BRs) of 0.0 to 1.5, where BR is the dry mass ratio of WTR to biosolids. Samples were blended for 1 h on a reciprocating shaker at 200 evolutions per min. To simulate sludge-conditioning typical of full-scale operations, samples were transferred to 500 mL beakers, placed on a six-station Phipps and Bird™ jar tester, and rapidly mixed at 200 rpm for 20 s followed by gentle mixing (60 rpm) for an additional 90 s. Mixing speeds and duration were selected to generate a *GT* (velocity gradient × detention time) value of approximately 10,000 (dimensionless). When polymer was added, the polymer solution was syringe-injected into the vortex about 2 cm below the surface at the beginning of the rapid-mix cycle. Polymer was dosed to achieve the operational polymer dose (OPD), determination of which is detailed elsewhere (Taylor & Elliott 2013).

### Dewatering experiments

Samples were then centrifuged for 20 min at 3,500 rpm for thickening to 5–10% solids content. This initial dewatering

step was necessary prior to belt-filter pressing because the bench-top Crown Press™ Belt Press Simulator cannot readily accommodate liquid slurries. The centrate was immediately coarse filtered using Whatman #1 paper to remove floating debris, adjusted to pH 2 with HCl for preservation, and retained for subsequent P analysis. This liquid was considered compositionally representative of the liquid discharged during dewatering, hereafter called reject water. Thickened solids were pressed for 4 min at 200 psi to achieve solids content between 18 and 25%, typical of full-scale belt-filter-press (BFP) dewatering operations. The BFP cake samples were then stored in airtight containers to maintain a consistent moisture content until analysis.

### Phosphorus analysis

To determine total P (TP) content of the reject water, samples were digested using USEPA method 3051 and then analyzed for P by ICPAES. The dewatered cake samples were analyzed for WEP using the procedure developed by Kleinman *et al.* (2007).

## RESULTS AND DISCUSSION

### Characteristics of biosolids and WTR

The characteristics of the biosolids and Al-WTR are presented in Table 1. The biosolids had high total N and P levels characteristic of biological nutrient removal processes (Brandt *et al.* 2004; WEF 2005). The Al-WTR total N content (9.2 g kg<sup>-1</sup>) is higher than typical (Ippolito *et al.* 2011), possibly due to filter backwashing with water treated with chloramines. The TP content is typical for WTR (Ippolito *et al.* 2011). The biosolids major elemental concentrations (Ca, Mg, Fe, Na, Al) are representative of materials produced nationally and reflect wastewater and sludge treatment processing. The high Al content of the WTR (122 g kg<sup>-1</sup>) reflects the use of alum as primary coagulant and is approximately 80% amorphous as measured by Al<sub>ox</sub>.

### Impact of co-dewatering on reject water P

The influence of BR on the TP content in the reject water from dewatering is shown with and without polymer addition in Figure 1. The TP level for the biosolids-only (BR = 0) samples (~250 mg L<sup>-1</sup>) are above the range (63–143 mg L<sup>-1</sup>) reported in the USEPA process design manual (USEPA 1979). However, with the use of EBPR wet-side processing, a typical sidestream

**Table 1** | Selected properties for City of York WWTP sludge and WTR

Parameter	Digested sludge <sup>a</sup>		WTR <sup>b</sup>	
	Mean	Std. Dev.	Mean	Std. Dev.
pH	7.5	0.13	7.1	0.2
% solids	1.32%	0.14%	4.4%	0.5%
Total P (g kg <sup>-1</sup> )	48.2	6.8	2.7	0.5
Water extractable P (g kg <sup>-1</sup> ) <sup>c</sup>	7.60	0.1	0.06	0.02
Total Kjeldahl N (g kg <sup>-1</sup> )	120.9	12.9	9.2	0.5
Ammonium N (NH <sub>4</sub> -N) (g kg <sup>-1</sup> )	51.7	5.0	<0.1	–
Calculated organic N (g kg <sup>-1</sup> )	69.2	4.5	9.2	0.3
Total calcium (g kg <sup>-1</sup> )	38.4	5.5	3.6	0.4
Total magnesium (g kg <sup>-1</sup> )	7.7	0.8	4.0	0.4
Total iron (g kg <sup>-1</sup> )	8.6	1.4	41.4	6.4
Total sodium (g kg <sup>-1</sup> )	6.6	0.8	0.6	0.2
Total aluminum (g kg <sup>-1</sup> )	4.2	1.1	122.2	5.2

Concentrations expressed on dry weight basis unless otherwise noted.

Means and standard deviations for independent samples ( $n = 3$ ).

<sup>a</sup>Collected prior to polymer addition and dewatering.

<sup>b</sup>Collected from thickeners prior to polymer addition and dewatering.

<sup>c</sup>As measured on dewatered (~20% solids) samples (no polymer addition).

from anaerobic digestion may have a P concentration of 500 mg L<sup>-1</sup> (Phillips *et al.* 2006). Gao (2010) reported a TP value (260 mg L<sup>-1</sup>) similar to the present study for digester supernatant from a facility employing EBPR. Pöpel & Jardin (1993) cite a phosphate concentration of 300 mg P L<sup>-1</sup> in digester supernatant for large plants.

In the absence of dewatering polymer, the TP concentration in the reject water was progressively reduced with increasing BR from 250 mg L<sup>-1</sup> (BR = 0) to 60 mg L<sup>-1</sup> at BR = 1.5. In comparison, Yang *et al.* (2007) calculated that co-processing of Al-WTR and digested activated sludge at a BR of 1.14 could reduce reject water TP from 113 to 1.2 mg L<sup>-1</sup>. It is likely that the extremely efficient P removal in their study is due to the high Al content (19.5%) of the WTR used. In our study, the WTR had an average total Al content of 12.2% (Table 1). Co-application of WTR and biosolids has been shown previously to reduce soluble P runoff and leaching (Elliott *et al.* 2002). The key to effective soluble P reduction is ensuring sufficient reactive Al + Fe in the WTR to immobilize the soluble P in the biosolids (Elliott *et al.* 2002).

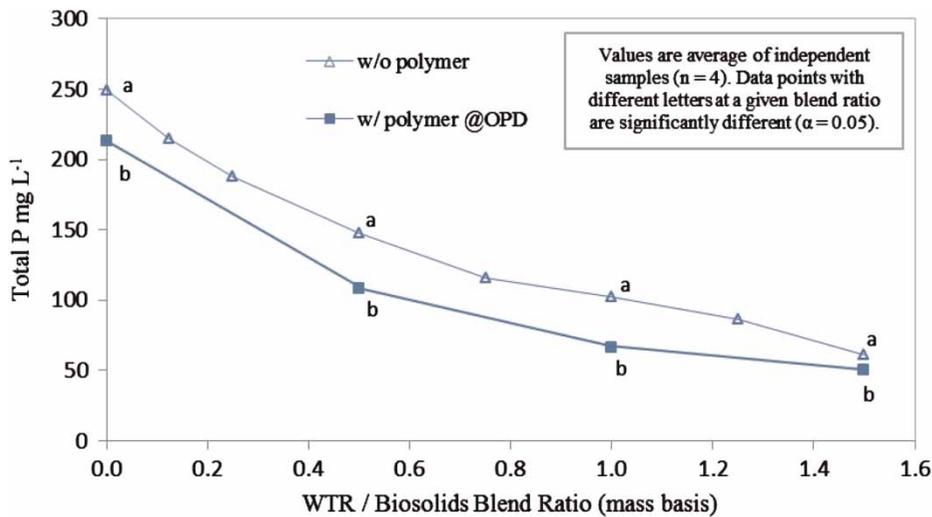


Figure 1 | TP in reject water for a range of co-processed WTR:biosolids blends.

The inclusion of cationic polymer in the dewatering process resulted in lower reject water TP levels for all BR conditions tested (Figure 1). This implies that the cationic polyelectrolyte contributed to P fixation. Butkus *et al.* (1998) modeled phosphate adsorption on WTR generated using ferrous sulfate as coagulant and predicted that nearly 40% of the total adsorbed P was due to binding by a cationic polymer. They proposed that phosphate anions were electrostatically bound to positively charged functional groups on the quaternary polyamine chain. An alternate explanation for the effect of polymer addition (Figure 1) is that the polymer promoted agglomeration of extremely

fine P-containing solids which were subsequently captured in the coarse filtration experimental step.

#### Impact of co-dewatering on cake solids P

The WEP content of the biosolids and blended products are shown as a function of BR in Figure 2. Samples were dewatered to approximately 20% solids to simulate conditions typical of full-scale dewatering operations and minimize the influence of total-P contained in entrained water on WEP measurements. Figure 2 shows an initial steep decline in WEP with a progressively smaller decrease as the BR was

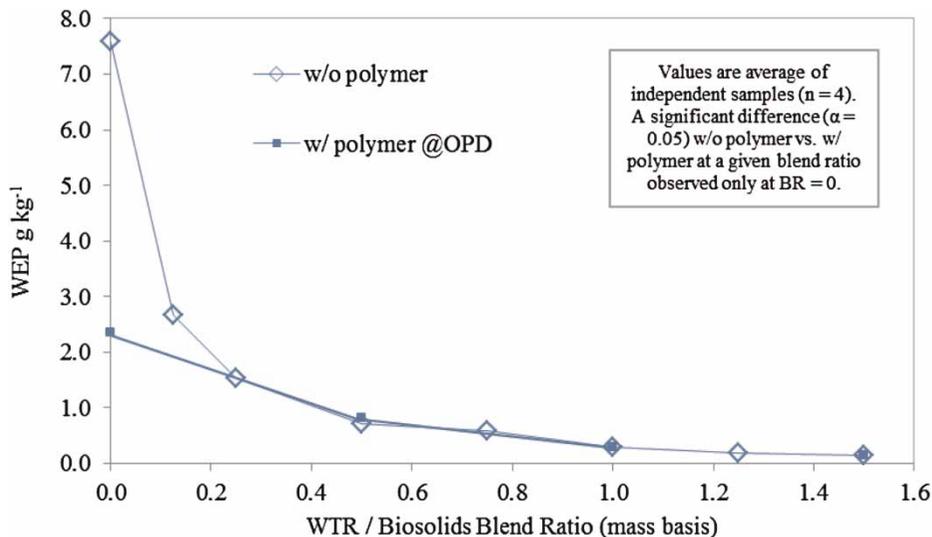


Figure 2 | WEP in dewatered biosolids ( $\approx 20\%$  solids) for a range of co-processed WTR:biosolids blends.

increased beyond 0.5. From a practical standpoint, there is limited incentive to increase BR beyond 0.5 based solely on the desire to reduce the environmentally labile P in the biosolids. However, higher BR ( $>0.5$ ) did reduce the TP of the reject water which would be recycled to the wastewater treatment plant inlet (Figure 1). Although the addition of polymer decreased WEP for the biosolids-only treatment (i.e., BR = 0), the WEP of the blended samples was unchanged by polymer dosing, at least for the three conditions (Figure 2). Because only one polymer and a limited number of experimental conditions were evaluated, the observed behavior cannot be generalized to cover the variety of polymers, system conditions, and equipment typical in dewatering practice.

Logically, simple mixing of biosolids (WEP =  $7.6 \text{ g kg}^{-1}$ ) with WTR (WEP =  $0.06 \text{ g kg}^{-1}$ ) should result in an intermediate WEP concentration reflecting the relative amounts of the two materials in the mixture. For example, equal masses of dry biosolids and WTR (BR = 1) should have a WEP of  $3.83 \text{ g kg}^{-1}$  (the average of the WEP

values of the two materials) assuming no interactions that influence the solubility of P. However, the WEP value at BR = 1 is about  $0.3 \text{ g kg}^{-1}$  (Figure 2). This implies that the WTR particles are sorbing a portion of the water-soluble P associated with the biosolids. This is consistent with the widely documented ability of WTR to immobilize the soluble P in biosolids (Elliott *et al.* 2002).

### Phosphorus mass balance

To quantify changes in the P distribution during the co-dewatering process, a mass balance on P and water was conducted on the pertinent soluble and particulate forms of P during the blending and dewatering. Figure 3 shows the mass balance for BR = 0.5 and distinguishes between the measured parameters and calculated values. The WEP content of the dewatered solids was measured and used to determine the amount of P fixed (i.e., water insoluble) in the dewatered cake. The results of these calculations are presented in the three-component graphical display (Figure 4).

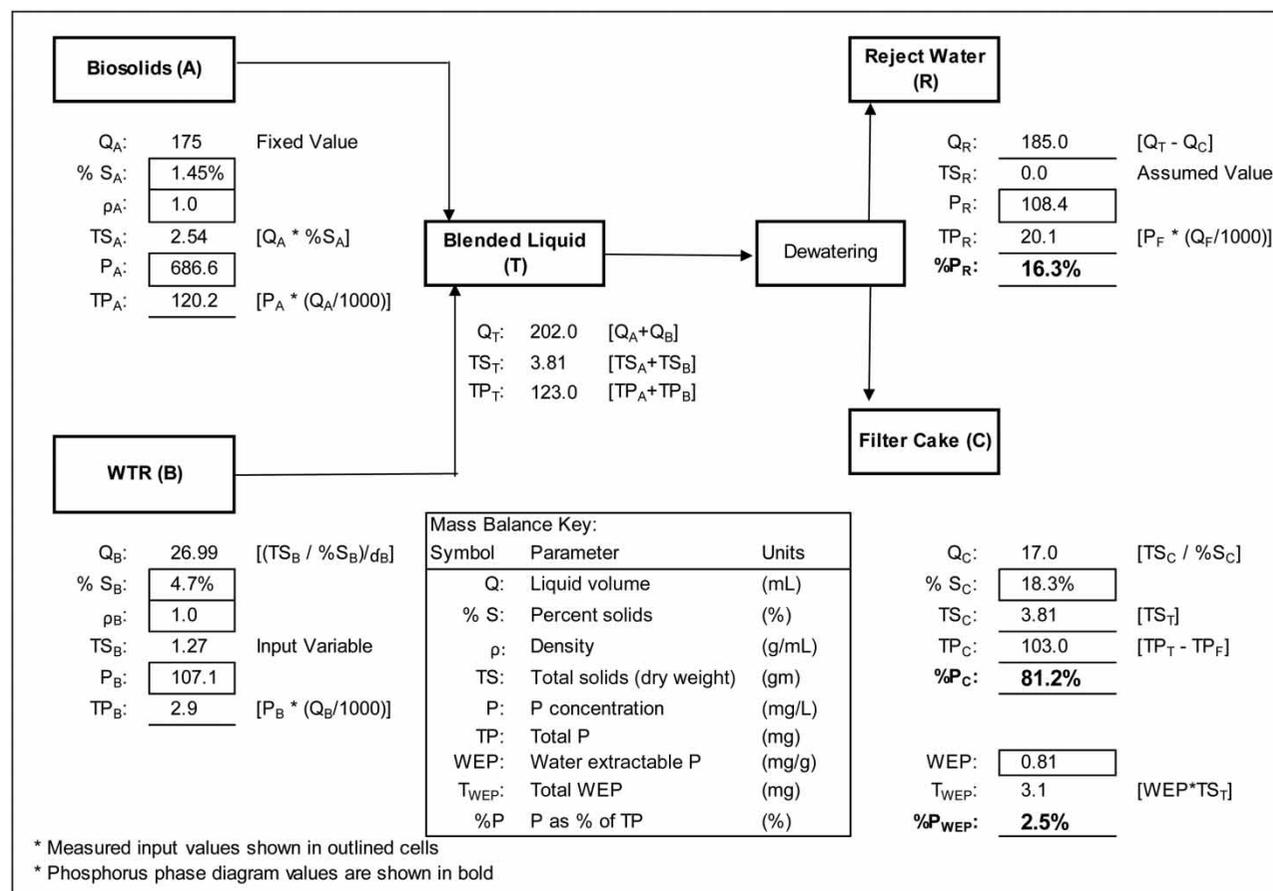
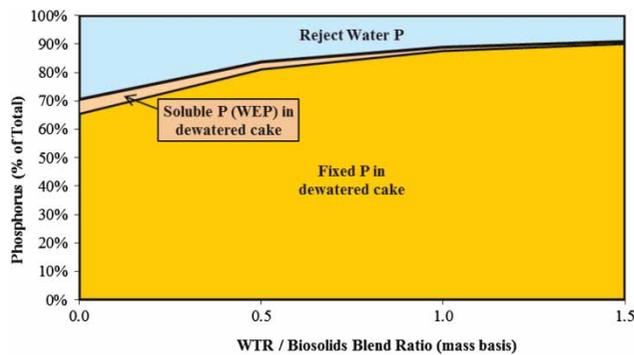


Figure 3 | Phosphorus mass balance for co-processing WTR and biosolids at BR = 0.5.



**Figure 4** | Partitioning of phosphorus (P) as a function of BR in the experiments with polymer addition.

Figure 4 clearly shows, with increasing WTR addition, P progressively partitioned into the fixed (i.e., water-insoluble) form in the dewatered cake. This increase in particulate-bound P occurred at the expense of both P in the reject water and the water-soluble P in the dewatered solids. Although both the reject water P and WEP fractions decreased continuously over the range of BR conditions tested, the extent of the effect decreased with incremental increases in BR.

The exact nature of the three-component diagram depends upon dewatering conditions and the characteristics of the component materials. Even at BR of 1.5, a substantial concentration of TP ( $60 \text{ mg L}^{-1}$ , Figure 1) P remained in the reject water. In contrast, Yang *et al.* (2007) reported that the co-dewatering of Al-WTR and digested activated sludge solids at a BR of 1.14 could reduce P in the reject water from  $113$  to  $1.2 \text{ mg L}^{-1}$ . As noted, the WTR used in their study had high Al content (19.5%). Although not reported by Yang *et al.* (2007), it is likely that the water treatment works where their WTR was collected employed high alum dosages used in 'sweep floc' particle destabilization. Such WTRs are expected to have a highly reactive Al fraction as measured by oxalate extraction, and, in turn, high P-fixing capacity.

### Implications for P management

Besides the in-plant benefits associated with reducing excess P in return flows (Figure 1), co-processing biosolids with WTR will alter the concentration and nature of P in the dewatered solids that must be managed. When biosolids are land applied, the rate of application is usually determined by matching the crop-available biosolids N to the N requirement for the specific crop grown on the site. This

normally results in excess P added to the soil. Thus, a critical parameter is the N/P ratio in the biosolids destined for land application. Since WTR are typically low in both N and P, co-processing may have little or no impact on the total N-to-TP ratio in the biosolids. For example, for the Table 1 analyses, the N/P (based on total Kjeldahl and TP contents) of the biosolids alone is 2.51. For a BR = 1 mixture, the N/P would be 2.55.

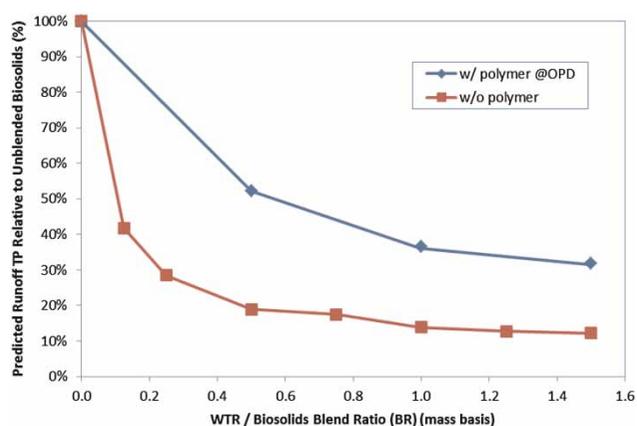
However, co-processing does have a dramatic effect on the environmental lability of the P in the blended material destined for land application. A number of studies have documented that WEP expressed in  $\text{g kg}^{-1}$  on a dry weight basis serves as a quantitative predictor of dissolved P in runoff when P sources are applied to soils (Kleinman *et al.* 2005). Kleinman *et al.* (2002) found that the WEP concentration of manures broadcast on soils was strongly correlated ( $r^2 = 0.86$ ) to the concentration of dissolved reactive P in runoff. Elliott *et al.* (2005) evaluated runoff from surface application of 10 different biosolids sources and three manures and found that the TP ( $\text{mg L}^{-1}$ ) in runoff was related to the P source WEP ( $\text{g kg}^{-1}$ ) by the following relationship:

$$\text{TP} = 3.42 \times \text{WEP} + 3.03 \quad (r^2 = 0.81)$$

Using this expression, the TP level for biosolids only (with polymer,  $\text{WEP} = 2.36 \text{ g kg}^{-1}$ , Figure 2) would be  $11.1 \text{ mg L}^{-1}$  while co-processed material with BR = 1.5 ( $\text{WEP} = 0.14 \text{ g kg}^{-1}$ ) would be about one-third ( $3.5 \text{ mg L}^{-1}$ ) the biosolids-only value.

The above equation was developed using indoor rainfall simulations for trays packed with soil to which the biosolids had been surface applied (Elliott *et al.* 2005). Research has shown that runoff TP concentrations and losses are numerically greater for packed trays than field plots of the same soil type (Guidry *et al.* 2006). The absolute values of TP based on the equation are likely higher than would be experienced under field conditions. Thus it is more useful to evaluate the relative reductions in TP associated with co-processing. Figure 5 shows the predicted reduction in runoff TP (based on the above equation) as a function of BR.

The real benefit of co-processing for land application of biosolids would be realized in states (Arkansas, Maryland, Pennsylvania) with P site indices that differentiate P sources based on WEP (Sharpley *et al.* 2003). Unfortunately, many state site assessment tools do not distinguish the environmental loss potential between P sources. Differential



**Figure 5** | Predicted runoff total phosphorus (TP) as a function of WEP concentration for a range of WTR/biosolids blend ratios.

weighting of organic P sources in P site indices is essential to sustaining biosolids land application (Elliott & O'Connor 2007).

The practicability of co-dewatering for a municipality depends on many factors. Yang *et al.* (2007) cite two drawbacks to co-dewatering. First, the increase in the tonnage of dewatered cake means more material must be transporting and handled. Second, the higher hydraulic loading to the dewatering unit will mean an increase in volume of reject water. Thus the actual mass of P recycled will depend on the volume, as well as P content, of the reject water. The proximity of the water treatment and wastewater treatment facilities will influence the cost of bringing the two streams together at a common dewatering facility.

A scenario where co-dewatering is particularly useful would be a WWTP employing biological P removal processes followed by anaerobic digestion for stabilization and where the biosolids are recycled via land application. Anaerobic digestion of EBPR solids releases P and results in P-rich return flows that can negatively impact operations and degrade effluent quality. Additionally, EBPR processes produce biosolids exceptionally high in WEP that have high potential for P runoff and drainage following land spreading (Elliott *et al.* 2005).

## CONCLUSIONS

This study documented the progressive partitioning of P into a water-insoluble form in the dewatered cake as the amount of WTR blended with anaerobically digested biosolids increased. As the dry mass ratio of WTR to biosolids increased from zero (biosolids only) to 1.5, the reject

water TP decreased from 250 to 60 mg L<sup>-1</sup>. Moreover, environmental lability of the dewatered biosolids–WTR cake as measured by WEP also decreased as amount of WTR added to the biosolids increased. Co-dewatering of WTRs and biosolids should result in P less susceptible to off-site migration following land application compared to unamended biosolids. While results will vary with WTR and biosolids characteristics and system operational conditions, results confirm that co-dewatering is a potential strategy for managing municipal by-products as new P-based nutrient regulations for land application are enacted. Sustaining biosolids recycling programs will depend on implementing strategies like co-dewatering that modify the nutrient characteristics of the solids produced in wastewater treatment.

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