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Dewatering of phosphorus extracted from liquid swine waste

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Abstract

Phosphorus (P) recovery from liquid swine manure is an attractive technology when on-farm application of liquid swine manure is not an option. We developed a technology that enables separation of this P, but its high moisture content makes transportation difficult. In this work, we investigated dewatering procedures to concentrate the P product. Sludge rich in calcium phosphate (>20% P_2O_5) was obtained using a field prototype, and it was further dewatered using a combination of polymer treatment and filter bags. Anionic polyacrylamide polymer treatment ($\geqslant 20$ mg/L) was effective to flocculate the P-rich sludge, which enhanced filtration and dewatering. Without polymer, filtration was incomplete due to clogging of filters. Non-woven polypropylene and monofilament filter bag fabrics with mesh size $\leqslant 200$ μm retained >99% of suspended solids and total P. Solids content dramatically increased from about 1.5% to >90%. These dewatered solids can be transported more economically off the farm for use as a valuable fertilizer material.
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1. Introduction

A common technology for managing liquid swine manure is land application after storage and treatment in lagoons. However, continued land application of excessive amounts of liquid manure saturate soils with P and is a potential threat to water quality if this nutrient enters water resources (Kellogg et al., 2000; Sharpley et al., 2000; Gollehon et al., 2001). Therefore, P recovery from liquid swine manure is an attractive technology when on-farm application of liquid swine manure is not an option. Thus, implementation of technologies for P recovery would resolve distribution issues of excess manure P and allow significant amounts of this nutrient to be transported off the farm.

According to Greaves et al. (1999), technologies for recovery of P from manures consist of two separate processes: extraction of P, and conversion of the extracted P to a useful product. Adoption of P recovery technologies has been slow due to limited value of by-products, such as iron and aluminum phosphate sludge (Loehr et al., 1973), high chemical demand requirements for P extraction (Westerman and Bicudo, 2000), and potential ammonia volatilization losses during P extraction process, such as extraction via struvite precipitation (Celen and Turker, 2001).

To resolve these technical limitations, we developed a wastewater treatment technology that removes P from liquid manure with minimal chemical addition and produced a valuable by-product (Vanotti et al., 2003a,b). This technology includes nitrification of wastewater to oxidize ammonia, reduce carbonate buffers and subsequent increase the pH of the nitrified wastewater by adding calcium hydroxide to precipitate P. Since ammonia

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nitrogen is mostly converted to nitrate, increased pH does not result in significant gaseous nitrogen loss. The final product is a calcium phosphate rich sludge that has the potential to be reused as fertilizer, but its high moisture content makes transportation difficult. Therefore, dewatering is necessary to economically transport the P product off the farm.

Sludge dewatering techniques that concentrate P may include settling of solids followed by decantation, centrifuging or filtration procedures to remove the liquid fraction (Nelson et al., 2000; Burns et al., 2001; Ueno and Fujii, 2001), but efficiency of dewatering procedures is not well documented. In the research reported in this paper, we investigated sludge dewatering procedures to concentrate the P product using filtration. Sludge rich in calcium phosphate (greater than 20% P_2O_5) was obtained using a field prototype, and it was further dewatered using a combination of polymer treatment and filter bags. In the P sludge dewatering procedure, when sludge falls into the bags, the water drains through the porous bags leaving solids within the bags. Our objective was to determine type, charge density, and optimum application rates of polymers to enhance sludge dewatering using filter bags. In addition, we evaluated moisture content of filtered solids, filtration efficiencies, and filtrate water quality using non-woven polypropylene and nylon monofilament filter fabrics.

2. Methods

2.1. Swine wastewater treatment system

Recovery of soluble P was performed in a field prototype system designed to replace the anaerobic lagoon treatment of swine manure, typical in the US (Barker, 1996). The prototype was installed at the Swine Unit of the North Carolina State University Lake Wheeler Road Laboratory in Raleigh, NC (Vanotti et al., 2003a). The treatment system is shown in Fig. 1. It included three modules: solid–liquid separation, biological N removal, and P extraction. Flushed manure from finishing houses was collected in a 15 m³ homogenization tank where it was stirred and kept well mixed before solids separation. The solid–liquid separation module, consisted of a in-line polymer injector unit and two (6.1 × 4.9 m) sand filter beds (F.D. Deskins Inc., Alexandria, IN).[1] Manure in the homogenization tank was pumped into the polymer injection unit at 490 L/min flow rate and then poured on the sand filter

bed. The polymer used was a cationic emulsion (c1596, Cytec Industries, Inc., West Paterson, NJ). This first treatment step removed most of the N and P associated with manure solids, but it did not remove the soluble C, N or P fractions in the liquid stream generated by sand filtration. These soluble fractions were removed from the liquid stream in a second and third step. In the second step, N was removed via nitrification/denitrification processes. Nitrification transformed 100% of the ammonia-N into nitrate-N and depleted about 80% of the alkalinity using nitrifying bacteria entrapped in polymer pellets in an aerated tank (Vanotti and Hunt, 2000). A pre-denitrification configuration transformed nitrate-N into N gas where nitrified wastewater was continuously recycled to an anoxic tank (Fig. 1). In this tank, suspended denitrifying bacteria used soluble manure carbon contained in the liquid to remove the nitrate (Vanotti et al., 2005). Thus, elimination of ammonia and reduction of carbonate buffers allowed the removal of P from the liquid when small amounts of lime were added in the third step. In this third step, lime milk [$Ca(OH)_2$] was added, and soluble P was removed from the liquid phase as a precipitate.

Phosphorus extraction was performed in a reactor that consisted of a 378-L plastic tank with a conical bottom made of XLPE resin (Nalgene, Nalge Co., Rochester, NY) and a mixer 0.25 HP, 1725 rpm and 8.9-cm impeller size (Lightin, Rochester, NY). A smaller 114-L tank and mixer were used to prepare lime milk (10% hydrated lime mixture in water). Lime milk was prepared using commercial hydrated lime slurry, 50% $Ca(OH)_2$ (Chemical Lime Co., Fort Worth, TX). A Masterflex peristaltic pump (Cole-Parmer Instrument Co., Vernon Hills, IL) connected to a Model 53 GLI pH controller (GLI International, Milwaukee, WI) was used to transfer the lime milk into the P-reactor. Lime milk was injected into the reactor, which contained wastewater effluent from the biological-N removal module. The pH controller stopped injection when the pH of the mixed liquid reached a set point of 10.5 units. Previous research demonstrated that highest P removal rates were attained between 10.0 and 10.5 pH units (Vanotti et al., 2001, 2003b). Thus, a 378-L volume of water was treated by raising the water pH from 7.22 to 10.50 units (at 24 °C) with 0.71 L of 10% $Ca(OH)_2$. Formation of a precipitate started as soon as the lime milk was mixed with the liquid. Once the desired treatment pH was attained, the mixer was turned off and the liquid was allowed to settle for 30 min. After this settling period, the liquid fraction was decanted and the precipitate, hereafter referred to as sludge, left in the bottom of the conical tank was transferred into a 19-L plastic container. The sludge had a total suspended solids concentration of 14.67 g/L (1.47%). Liquid samples and sludge were transported to the laboratory and stored at 4 °C until analysis. Sludge sub-samples were subsequently

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[1] Mention of trade name, proprietary product, or vendor does not constitute a guarantee or warranty of the product by US Department of Agriculture and does not imply its approval to the exclusion of other products or vendors that also may be suitable.
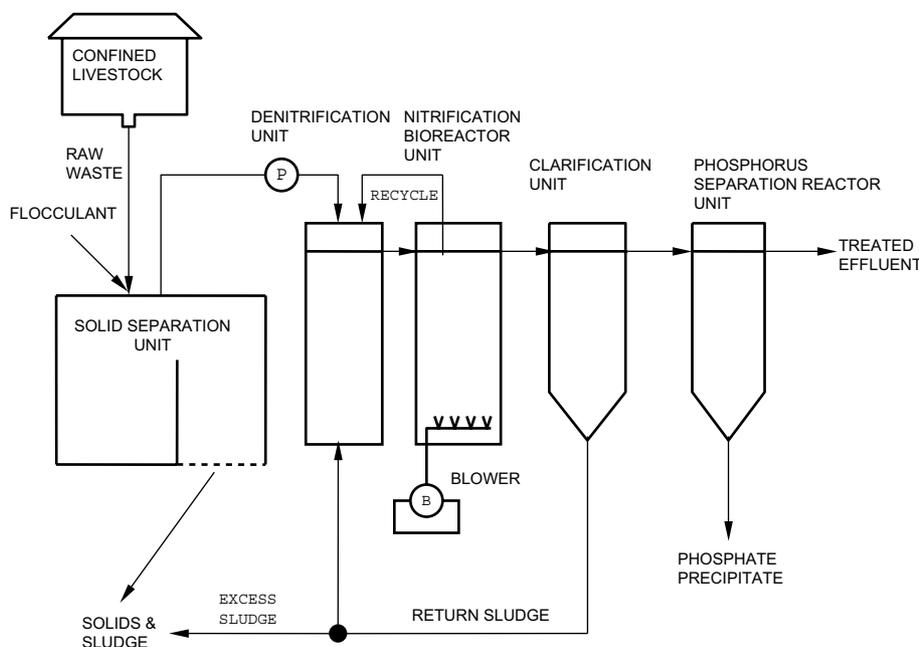
Fig. 1. Schematic drawing of the pilot waste treatment system without lagoon.

Table 1
Changes in swine wastewater characteristics before and after treatment

Constituent	Influent[a]	Effluent[b]
pH	7.22 ± 0.08	10.50 ± 0.01
Alkalinity (mg/L)	1367 ± 29	306 ± 16
TSS (mg/L)	932 ± 90	82 ± 12
BOD_5 (mg/L)	830 ± 17	3 ± 1
TKN (mg/L)	185 ± 14	2 ± 1
TP (mg/L)	63 ± 2	9 ± 1
NH_3-N (mg/L)	185 ± 6	0
$NO_3 + NO_2$-N (mg/L)	0	98 ± 1
PO_4-P (mg/L)	15 ± 6	0.2 ± 0.1

[a] Wastewater flushed from the pig house. Mean ± standard error ($n = 3$).
[b] After sequential treatment: solid/liquid separation-biological N removal-lime precipitation.

used for dewatering experiments. Changes in water quality before and after treatment are shown in Table 1.

2.2. Analytical methods

Elemental composition of the solid precipitate was determined by inductively coupled plasma (ICP) according to Vanotti et al. (2003b). Wastewater analyses were performed according to Standard Methods for the Examination of Water and Wastewater (APHA, 1998). Total P was determined using the automated ascorbic acid method (Standard Method 4500-P F) adapted to digested extracts (Technicon Instruments Corp., 1977). Total suspended solids (TSS) were determined by retaining solids on a glass-fiber filter (Whatman grade 934AH, Whatman Inc., Clifton, NJ) dried to 105 °C (Standard

Method 2540 D). Moisture and total solids (TS) were determined using a microwave moisture analyzer (Omnimark Instrument Corp., Tempe, AZ). Data were statistically analyzed by means, standard deviations and standard errors (proc MEANS), analysis of variance (proc ANOVA) and least significant difference (LSD) for multiple comparisons among means with SAS Version 8 (SAS, 1999). A completely randomized design was used to test the effect of flocculant type on solid–liquid separation and the effect of mesh size on filtrate quality using flocculation. In addition, a 2×5 factorial design ANOVA was used to test the combined effect of flocculant type and rates on concentration of TSS or TP in filtrates. All treatments had two replicates.

2.3. Polyacrylamide (PAM) type and rate evaluation

Filtration tests following flocculation were used to determine the effect of PAM charge type and density on sludge dewatering and filtration. Whereas only well-conditioned sludge drains rapidly, a gravity drainage test was used to evaluate the effect of polymer types on filtration of sludge (WERF, 1993). We conducted two flocculation and gravity filtration experiments. In the first experiment, we compared the performance of anionic and cationic PAMs in order to identify PAM charge type and density that would properly condition the sludge, prevent clogging of filter fabric, and produce a high quality filtrate. The polymers employed were commercial formulations of anionic PAM [Magnafloc 90L (10% mole charge) and Magnafloc 120L (34% mole charge)] and cationic PAM [Zetag 7837 (13% mole

Table 2
Characteristics of the polyacrylamides tested

Polymer name[a]	Charge type	Charge density (mol.%)	Active polymer[b] (%)
Magnafloc 90L	Anionic	10	50
Magnafloc 120L	Anionic	34	50
Zetag 7837	Cationic	13	50
Zetag 7867	Cationic	20	50

[a] Commercial formulations, Ciba Specialty Chemicals Water Treatment, Inc. Sufolk, VA.
[b] Percentage of active ingredient in a liquid dispersion formulation.

charge) and Zetag 7867 (20% mole charge)] (Table 2). Duplicate drainage tests were performed in 300-mL funnels (Gelman Sciences, Ann Arbor, MI) fitted with 47-mm diameter filter fabric discs. These filter fabric discs were made of a commercial polypropylene non-woven fabric (Dupont®, E.I. Dupont de Nemours, NJ) with the following physical characteristics: weight per surface area, 110 g/m²; fibers diameter, 40–55 μm; and particle size retentions, 100% particles > 210 μm dry, 100% particles > 190 μm moist, and 90% particles > 150 μm. A preliminary study using sludge from the same field prototype treatment system was performed to determine the polymer rate. The polymers were added from working solutions at a rate of 30 mg active polymer per liter of sludge. Working solutions of PAM were secondary solutions after preparation of 0.05% primary stocks (WERF, 1993). Thus, duplicate chemical treatments were added into the sludge in 1-mL working solutions after 125-mL Erlenmeyer flasks were filled with 50 mL of well-mixed sludge sample (Fig. 2). Sludge samples and chemical were then mixed with a glass rod for 10 s and allowed to settle 20 min before filtration. The test

also included untreated (no-polymer addition) control sample duplicates. Flocculation and filtration performance were determined by assessing clogging of the filter and quality of the filtrate. A filter was considered clogged when supernatant liquid remained in the filter funnel after 2 h of filtration. In addition, clogging was quantified by measuring the total volume drained through the filter fabric at 1 and 2 h after starting the test. The quality of the filtrate was assessed by determining TSS concentration in the filtrate. For the purpose of this study, high quality filtrates were those that had very low (less than 250 mg/L) concentrations of TSS with respect to untreated control filtrate.

A second flocculation and gravity filtration test compared the performance of anionic PAMs [Magnafloc 90L (10% mole charge) and Magnafloc 120L (34% mole charge)] dosage on filtration and dewatering of the sludge. The test included five treatment rates (0, 10, 20, 30 and 40 mg of active polymer per liter). The test procedures and water quality assessment were performed as previously described for the first flocculation and gravity filtration test.

2.4. Bag filter and mesh size performance evaluation

The use of filter bags made of non-woven polypropylene fabric is of interest because the fabric, which is hydrophobic on the outside, allows air to enter for natural drying, yet repeals rainwater making it possible to store sludge-filled bags outdoors. As sludge falls into filter bags, the water drains through the porous bags, leaving solids within the bags. In this method, rapid drainage is necessary for efficient dewatering of conditioned sludge. Therefore, we conducted time-to-filter (TTF) tests using 13×25-cm non-woven polypropylene bags. The TTF tests used the force of gravity to drive the filtration process and measured the time required for 50% of sample volume to pass through the filter (WERF, 1993). Prior to the test, P sludge was conditioned with anionic PAM [30 mg/L active anionic polymer (34%)]. The TTF tests were performed in duplicate using 1.0-L samples of conditioned and well-mixed sludge per bag.

We used a sequential pouring procedure. First, a 1.0-L sample was poured into the bag and TTF was determined with a stopwatch (Daigger and Co., Vernon Hills, IL). Then, the bag was left to drain for 24 h and final volume of filtrate was recorded. Subsequently, a second sample was poured into the same bag and TTF and 24-h final filtrate volume was again determined. In this manner, the solids of two 1.0-L sludge samples accumulated into one cake per bag, which was kept in the bag for 20 days. In order to assess moisture loss within the bag, moisture of each cake was determined in duplicate air-dried samples at 0, 1, 14 and 20 days after the end of the sequential pouring procedure. The air-drying of

Well-mixed sludge sample +
Flocculant dose (include control with no flocculant)
↓
Mixing (10 seconds)
↓
Settling (20 minutes)
↓
Gravity filtration through filter fabric
↓
Evaluation (1 and 2 h after filtration)

Quantitative
Total volume drained
Analysis of filtrate:
TSS, TP

Qualitative
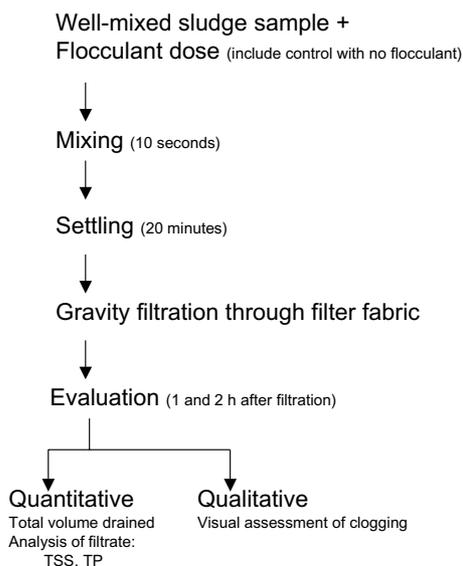Visual assessment of clogging

Fig. 2. Simplified schematic of filtration tests.

samples was performed at laboratory ambient temperature in the range of 20–22 °C and 60% relative humidity.

An additional test to estimate the effect of mesh size on filtration time (TTF) and filtrate quality was performed using six filter fabrics: five monofilament nylon fabrics (NMO, Krystil Klear Filtration, IN) with constant mesh size (200, 300, 400, 600 and 800 μm) and the non-woven polypropylene fabric used in all previous experiments. Duplicate sludge samples were conditioned with anionic PAM [30 mg/L active anionic polymer 34% charge] and passed through the monofilament nylon filter fabrics and the non-woven polypropylene fabric.

3. Results and discussion

3.1. Characteristics of dewatered sludge

The phosphorus sludge formed after addition of liquid lime settled quickly, and it was concentrated to a volume equivalent to 25.4 L/m^3 with a TSS concentration of 14.67 ± 0.04 g/L. The major two elemental components were inorganic P (10.1%) and calcium (22.6%) indicating that most of the applied Ca(OH)$_2$ was selectively used to precipitate P in the form of calcium phosphate. Phosphorus grade of the precipitate estimated as percent of P$_2$O$_5$ (%P × 2.29) was 23.1 ± 0.3%. This grade is consistent with the range of 21 to 32% P$_2$O$_5$ reported by Vanotti et al. (2003b) for P extracted using the same technology applied to swine lagoon wastewater.

3.2. PAM type and rate evaluation

The effect of PAM charge type on filtration performance and filtrate quality of P sludge is shown in Table 3. Without polymer addition (control), the filtrate was turbid (6628 mg TSS/L) resulting in an incomplete filtration and solids clogged the filter. Therefore, addition of a flocculant is necessary to avoid clogging of the filter. We found that polymer charge type had a significant

influence on solids/liquid separation efficiency. The anionic PAMs produced large, firm, light brown flocs that rapidly settled, leaving a clear supernatant. In contrast, cationic PAMs yielded flocs with poorer consistency and smaller, and clogged the filters as shown by the significant lower filtrate volumes than anionic PAMs after 1- and 2-h filtration (Table 3). In addition, sludge treated with cationic PAMs had filtrates with significant higher TSS (470 ± 40 and 1035 ± 315 mg/L) than filtrates obtained from sludge treated with anionic PAMs (110 ± 30 and 65 ± 10 mg/L) (Table 4). In summary, anionic PAMs prevented filter clogging, and had better filtrate quality than cationic PAMs or the control without polymer addition. Thus, anionic PAMs were subsequently used to determine the effect of polymer rate on filtration performance and filtrate quality of the P sludge.

3.3. Anionic PAM rates

Both anionic PAM polymers were equally effective to condition the P sludge. A 2 × 5 factorial ANOVA indicated that there was no significant difference ($P > 0.05$) between anionic polymer type on concentration of TSS or TP in filtrates. However, there was significant treatment rate effect on TSS and TP. The effect of anionic PAM rate on filtration performance and filtrate quality is presented in Table 4. As expected, differences in mean TSS and TP filtrate concentrations for both anionic PAMs were significant between control and any polymer rate (LSD test at 5%). Polymer rates ⩾20 mg/L provided large flocs that not only prevented clogging, but also significantly enhanced phosphorus retention as revealed by the lower TP concentration in the filtrate. The anionic polymer with 34% density charge at 30 and 40 mg/L active polymer rates showed the lowest TSS and TP concentrations in the filtrate. Therefore, we selected Magnafloc 120L (anionic PAM, 34% charge density) at a rate of 30 mg active polymer/L for bagging and mesh size tests discussed in the following section.

Table 3
Effect of polymer charge type on solid/liquid separation of phosphorus sludge

Polymer name	Charge type	Polymer rate (mg/L)	Filtration test			
			Complete filtration observed[a]	Time (mL/50 mL)[b]		Filtrate quality[c] (mg TSS/L)
				1 h	2 h	
Control	None	0	N	17 ± 11ab	25 ± 4ab	6628 ± 233a
Magnafloc 90L	Anionic	30	Y	35 ± 2a	36 ± 2a	110 ± 30b
Magnafloc 120L	Anionic	30	Y	33 ± 4a	34 ± 4a	65 ± 10b
Zetag 7837	Cationic	30	N	13 ± 4b	16 ± 4b	470 ± 40c
Zetag 7867	Cationic	30	N	7 ± 1b	10 ± 2b	1035 ± 315c

[a] Commercial non-woven propylene filter fabric was used for the test. The filter was considered clogged when liquid remained in the filter funnel after 2 h of filtration.
[b] Volume of water that passed through the filter after 50 mL of sludge was poured into the filter funnel. Data are mean ± standard error of replicate tests ($n = 2$); means followed by the same letter are not significantly different (LSD at 5% level).
[c] Total suspended solids measured in the filtrate. Initial TSS (mean ± standard error, $n = 2$) in unfiltered sludge was 14,670 ± 40 mg/L (1.47%).

Table 4
Effect of anionic polymer dosage on filtration and filtrate quality of phosphorus sludge conditioned with two anionic PAM flocculants

Polymer type and charge	Rate (mg/L)	Filtration test		Filtrate quality[a]	
		Complete filtration observed[b]	Filtrate volume (mL/50 mL)[c]	(mg TSS/L)	(mg TP/L)
Anionic 10%	0	N	26 ± 4a	6395 ± 328a	868 ± 2a
	10	N	12 ± 1b	485 ± 225b	52 ± 14b
	20	Y	31 ± 1bc	90 ± 35b	16 ± 2c
	30	Y	36 ± 2c	110 ± 5b	14 ± 1c
	40	Y	38 ± 1c	95 ± 25b	16 ± 1c
Anionic 34%	0	N	26 ± 4a	6860 ± 329a	871 ± 2a
	10	N	24 ± 3ab	280 ± 135b	82 ± 1b
	20	Y	31 ± 4ab	205 ± 110b	29 ± 12c
	30	Y	34 ± 4b	65 ± 10b	11 ± 2c
	40	Y	40 ± 1b	68 ± 13b	10 ± 1c

[a] Total suspended solids and total phosphorus measured in the filtrate. Initial TSS and TP (mean ± standard error, $n = 2$) in unfiltered sludge were 14,670 ± 40 mg/L and 1697 ± 183 mg/L, respectively.
[b] Commercial non-woven propylene filter fabric was used for the test. The filter was considered clogged when water remained in filter funnel after 2 h of starting the filtration test.
[c] Volume of water that passed through the filter in 2 h after 50 mL of sludge was poured into the filter funnel. Mean ± standard error of two replicates within polymer type and rate; means followed by the same letter are not significantly different (LSD at 5% level).

3.4. Bagging and mesh size evaluation

Time to filter, final volume after 24 h of filtration, and TS retained in the bag were consistent between bags (Table 5). These data show that reproducible results for TTF and volumes of filtrate can be obtained with non-woven polypropylene bags. Once in the bag, dewatering and air drying of the cake continued (Fig. 3). Before dewatering, initial moisture content of the sludge was approximately 98.5% (1.47% TSS). After the sequential pouring procedure, the sludge had a mean moisture content of 93.7% (days after filtration = 0, Fig. 3). Thus, solid content inside the bag increased from 1.46% to 6.3% immediately after filtration. The drying process accelerated during the first day producing an average solid content of 24% (76% moisture, days after filtration = 1, Fig. 3). The mean moisture content continued declining to 10 and 8% in the subsequent 14 and 20 days after filtration, respectively. Differences in cake moisture contents between 14 and 20 days after filtration were not significant, indicating that the cake was thoroughly air dried.

The effect of mesh size on TTF and filtrate quality is presented in Table 6. Although TTF significantly increased with smaller mesh sizes, the filtrate quality im-
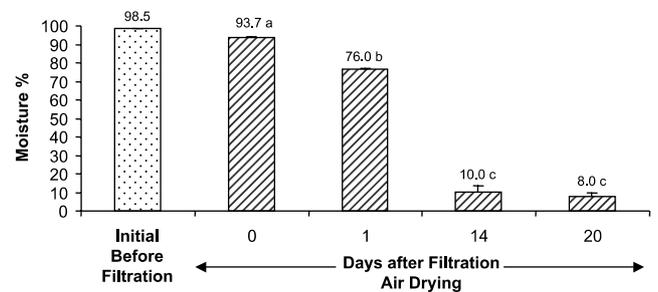

Fig. 3. Dewatering of sludge using flocculant (30 mg/L anionic PAM, 34% charge) and non-woven polypropylene bags. After filtration (day 0), sludge was air dried for 20 days inside the bag. Data are the mean of two replications. Error bars are standard deviations. Means followed by the same letter are not significantly different (LSD at 5% level). Comparison between any two means applies to mean moisture values of air-dried sludge.

proved dramatically with decreasing mesh size. Filters with 800-μm mesh produced filtrate with the highest solids and P concentration (805 ± 65 mg TSS/L and 49 ± 7 mg P/L). Even though 800-μm mesh nylon filter retained 94.5 ± 0.2% of TSS and 97.1 ± 0.2% of TP with respect to unfiltered sludge, the best filtrate quality was obtained with 200-μm monofilament nylon and non-

Table 5
Time to filter (TTF), final volume of filtrate and mass balance using bags made of non-woven polypropylene fabric for filtration of phosphorus sludge[a]

Bag #	TTF[b] (s)	Final volume[c] (L)	Mass balance[d] (g)		
			TS_{In}	TS_{Bag}	TS_F
1	70 ± 30	0.73 ± 0.10	34.300	31.130	2.165
2	76 ± 33	0.74 ± 0.07	33.330	31.326	2.103

[a] Data are mean ± standard error ($n = 2$). PAM rate = 30 mg active polymer/L; anionic PAM 34% charge.
[b] TTF = Time in seconds to drain 50% of volume sample (1.0 L).
[c] Final volume of filtrate after 24 h of starting TTF test.
[d] Total solids (TS) cumulative mass of two 1-L sludge samples poured in each bag. TS_{In} = initial TS mass in sludge before filtration; TS_{Bag} = TS mass retained in bags; TS_F = TS mass in filtrate.

Table 6
Effect of mesh size on time to filter and filtrate quality using flocculation[a]

Fabric type	Mesh size (μm)	Time to filter[b] (s)	Filtrate quality[c]			
			TSS		TP	
			Concentration (mg/L)	Retained in filter (%)	Concentration (mg/L)	Retained in filter (%)
Monofilament nylon[d]	800	58 ± 2b	805 ± 65a	94.5 ± 0.2	49 ± 7a	97.1 ± 0.2
	600	62 ± 3b	630 ± 140ab	95.7 ± 1.2	34 ± 6b	98.0 ± 0.1
	400	108 ± 1a	460 ± 140bc	96.9 ± 0.7	35 ± 1b	98.0 ± 0.1
	300	88 ± 5ab	385 ± 45bc	97.3 ± 0.6	27 ± 1bc	98.4 ± 0.1
	200	85 ± 3ab	185 ± 35cd	98.7 ± 0.5	17 ± 3cd	99.0 ± 0.2
Non-woven polypropylene	190–210[e]	80 ± 31ab	65 ± 15d	99.5 ± 0.2	9 ± 1d	99.5 ± 0.1

[a] PAM rate = 30 mg active polymer/L; anionic PAM 34% charge.
[b] Time in seconds to drain 50% of volume sample. Mean ± standard error of two replicates per mesh size; means followed by the same letter are not significantly different (LSD at 5% level).
[c] Total suspended solids and total P determined in the filtrate. Percentage retained in filter relative to concentration in unfiltered sludge before chemical treatment (14,670 ± 40 mg TSS/L and 1697 ± 183 mg TP/L, $n = 2$).
[d] Commercial monofilament nylon filter fabric with constant 200- to 800-μm mesh sizes (NMO, Krystil Klear Filtration, Winamac, IN).
[e] Equivalent mesh size for commercial polypropylene non-woven fabric (Dupont®, E.I. Dupont de Nemours, NJ).

woven polypropylene filters. There were not significant differences in TTF and filtrate quality parameter (TSS and TP) between 200-μm monofilament nylon and non-woven polypropylene filter fabrics. However, the non-woven polypropylene fabric retained 99.5 ± 0.2% of TSS and 99.5 ± 0.1% of TP, while 200-μm mono-filament nylon filter retained 98.7 ± 0.5% of TSS and 99.0 ± 0.2% of TP with respect to unfiltered sludge. From this mesh size experiment, we speculate that efficient dewatering of P sludge could also be achieved with several other mechanical devices besides filter bags, such as screen filters provided they are suited to retain flocs with size larger or equal to 200 μm.

4. Conclusions

Recovery of soluble P was obtained using a field prototype of a treatment system that included solid–liquid separation, biological N removal, and P extraction. Although the P extraction produced a sludge with high P content ($P_2O_5 > 20\%$), further dewatering was necessary to transport this product more economically. Our approach was to use a combination of polymer treatment and filter bags. We found that anionic polyacrylamide polymer treatment (at concentrations $\geqslant 20$ mg/L) was effective to flocculate sludge, which in turn enhanced filtration and dewatering. Without polymer treatment, filtration was incomplete. After flocculation, non-woven polypropylene and monofilament filter bag fabrics with mesh size $\leqslant 200$ μm retained more than 99% of suspended solids and total P. Solids content in the dewatered cake also increased from 1.5% to more than 90% in 20 days due to drying at temperatures in the range of 20–22 °C. Thus, dewatered solids can be transported more economically off the farm for use as a valuable P fertilizer material, especially in areas where excess P is a problem.

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