Dewaterability of Digested Sludge Conditioned with Sludge from a Water Treatment Plant

Mariusz Kowalczyk, Tomasz Kamizela[*]
Czestochowa University of Technology, Poland
[*]*corresponding author's e-mail: tkamizela@is.pcz.czest.pl*

1. Introduction

Sewage sludge generated in municipal treatment plants is a specific type of waste, which, depending on its properties, can be classified as e.g. waste that needs to be incinerated or a material for fertilizing as well as composting. The specificity of sewage sludge also results from its amount and hydration. The basic technological challenge in the sewage treatment plants sludge management is the need for reduction of water content in the sludge, which reaches over 98% (Skinner et al. 2015, Mowla et al. 2013). Mechanical dewatering is a strategic process used to reduce sludge volume. Due to the structure and properties of sludge, which usually make it difficult to release water, sludge has to be conditioned (Bień et al. 2015). Conditioning means a range of applicable physical, chemical and biological methods and their combinations (Skinner et al. 2015, Mowla et al. 2013). Any factor or substance, including waste, which reduces the bonds between water and solid particles, can be considered a conditioning agent. However, the choice of the conditioner must not be accidental. The qualitative changes in sludge after the addition of a conditioning agent, and technical issues related to the installation and the drainage unit should be taken into account.

Currently, the most popular method of sludge conditioning in sewage treatment plants is the use of polyelectrolytes. One of the examined methods of chemical conditioning is to use sludge from the water treatment process. This sludge is often termed post-coagulation sludge due to the fundamental role of iron and aluminium coagulants in water treatment technology. However, sludge from water treatment plants also contains other streams of liquid waste, e.g. washings from periodic flushing of filtration beds. Post-coagulation sludge can be used in municipal and industrial wastewater treatment plants as a residual coagulant (Ahmad et al. 2016, Jangkorn et al. 2011). The addition of aluminium post-coagu-

lation sewage sludge to wastewater can improve sedimentation in primary settling tanks, including the removal of suspended solids, organic matter, and phosphorus. Removal of these contaminants from wastewater occurs through their sorption on flocs formed during coagulation (Guan et al. 2005, Szerzyna 2013, Totczyk et al. 2015, Bal Krishna et al. 2016). The same effects are caused by the addition of iron sludge (Piaskowski 2005) to wastewater, but it is characterized by lower adsorption capacity (Al-Tahmazi & Babatundea 2016). Iron sludge may find wider applications in wastewater treatment using the Fenton's method (Yoo et al. 2001, Bolobajev et al. 2013). The use of regenerated coagulants allows for more favourable results than in the case of post-coagulation sludge application (Keeley et al. 2014, Keeley et a.l 2016).

Due to potential toxicity (Leszczyńska & Sozański 2009) in biological wastewater treatment, the use of post-coagulation sludge is a problematic issue. In general, the presence of heavy metals, which are disinfection byproducts in sewage sludge or an increase in mass concentration in the biological chamber may limit the effectiveness of biotechnological methods (Asada et al. 2010). However, the simultaneous dosage of the post-coagulation sludge in biological wastewater treatment and waste processing systems is considered in terms of intensification of biogas production (Cimochowicz-Rybicka & Górka 2017), phosphorus reduction (Nair &Ahammed 2015), reduction of hydrogen sulphide and ammonia in biogas (Akgul et al. 2017), as a source of micro and macroelements (Ebrahimi-Nik et al. 2018) and as a mass factor that allows for more efficient further phase separation (Luiz et al. 2018).

It may be advantageous to mix sludge from water treatment plants together with municipal sludge supplied for mechanical dewatering. Post-coagulation sludge can be treated as a physical conditioner due to the presence of inorganic matter acting as a structure-forming factor reducing compressibility. It can also be a chemical conditioner, which results from coagulation properties, including the neutralization of the solids content in sewage sludge and adsorption capacity of the flocs (Guan et al. 2005, Li et al. 2015).

The technology of water treatment, including surface or underground water treatment, has an effect on the quality parameters of the sludge and its application potential. The study analysed sludge from the water treatment plant that uses the denitrification process to remove nitrates from the collected water. The obtained organic water processing sludge was tested as a conditioner of municipal sewage sludge.

2. Substrates and methods

2.1. Substrates

Municipal sewage sludge was collected from a sewage treatment plant located in the Silesian Voivodeship in Poland. The average daily volume of wastewater to the treatment plant is nearly 45,000 m^3/day (People Equivalent 220 000). The plant operates in the UCT (University of Cape Town) technological system. The substrate was municipal digested sludge (MDS) taken from the installation of heat exchangers. After the digested sludge was collected, it was mixed in the laboratory at ambient temperature for 24 hours in order to be degassed.

The equipment in the technological system of the water treatment plant includes: biofilters for denitrification, aerators, double-layer filters, carbon filters, ozonation station and water tank. The sludge from denitrification and washings from the system are supplied to settling tanks. Thickened sludge from settling tanks is pumped to the sludge reservoir and then to the dewatering system. A coagulant in the form of a solution of iron (III) chloride is used in the process of water treatment. The coagulant is dosed to the installation of aerators and settling tanks. The tested organic water processing sludge (WPS) was a mixture of iron post-coagulation sludge and washings from biofilters and filters. WPS was collected from the sludge reservoir.

The dry solids and volatile solids of MDS were 20.7 ± 1.6 g/L and 12.7 ± 1.1 g/L, respectively, and the alkalinity was 5207.0 ± 95 mg $CaCO_3$/L. The concentration of dry solids in the WPS was 10.7 ± 1.5 g/L, volatile solids of 7.9 ± 1.6 g/L, at alkalinity 481.3 ± 195 mg $CaCO_3$/L. Iron content and pH of MDS sludge was as follows: 11,8 % DS, pH 5,8-6,2.

2.2. Conditioners and test combinations

Chemical conditioning of the sludge was performed using medium cationic polyelectrolyte Superfloc® C-494 in the form of an aqueous solution with a concentration of 0.1%. The optimal dose of polyelectrolyte for sludge dewatering was chosen only for digested sludge. The CST test was used to determine the optimal dose, which was 5.16 g/kg $_{DS}$. The study also used a reduced dose of polyelectrolyte, i.e. 25% lower than the optimal dose (4.13 g/kg $_{DS}$). Reduction of the polyelectrolyte dose resulted from the dosage of the second conditioning factor, i.e. water processing sludge (iron post-coagulation sludge).

The research was performed in 5 combinations (A-E), the main process variable was the volumetric fraction of post-coagulation sludge V_{WPS} in the mixture with digested sludge V_{MDS} (Fig. 1). Sludge mixtures (1.0 L) were prepared using a 0.1 m diameter paddle mixer and used for further examinations.

Fig. 1. Combinations of conditioning of municipal digested sludge with polyelectrolyte and water processing sludge

2.3. Analytical methods

Dry solids (DS), volatile solids (VS), pH and alkalinity measured according to standard methods for the examination of water and wastewater (APHA 2005). Technological parameters of sewage sludge, i.e. capillary suction time (CST), specific filtration resistance (r), final hydration of the sludge cake (H_{FC}) and dry solids in supernatant after sludge centrifugation (DS_{CEN}) were determined in accordance to PN-EN 14701-1:2007 (2007), PN EN 14701-2 (2013) and PN-EN 12880:2004 (2004) standards.

Total Kjeldahl nitrogen (TKN) and ammonium nitrogen (NH_4-N) were determined by titration method according to PN-EN 16169:2012 and PN-EN 14671:2007. Dissolved organic carbon (DOC) concentration was evaluated by means of the differential method using a TOC Analytik Jena multi N/C 3100 analyser. A Hach Lange DR 5000 spectrophotometer and cuvette tests were used to determine of chemical oxygen demand (COD). The presence of selected light and heavy metals and phosphorus was tested in accordance with PN-EN ISO 11885:209.

The quality of supernatants after the dewatering process was tested by centrifugation of sludge samples (MDS, WPS, A-E) for 3 minutes with relative centrifugal force of 4500 rcf. DS_{CEN} and COD were determined in these supernatants. Before COD measurement, the samples (50 ml) were homogenized for 60 seconds using an IKA T10 homogenizer.

The centrifugation of sludge samples (15 minutes, rcf = 12 100) and membrane filtration (0.45 um, cellulose acetate) were employed to determine the indices of contaminants occurring in the soluble form (DOC, TKN, NH4-N, metals and phosphorus).

The measurement of shear stresses (τ) and viscosity of the tested sludge samples (η) was performed by means of an RC 20 rheometer. The examinations were conducted at a constant temperature of 20°C using a thermostat, with shear rate of 0-200 s^{-1} and measurement time of 120 s. The flow curves of the tested samples were approximated using the Ostwald de Waele's rheological model:

$$\tau = k \cdot (\gamma)^n \tag{1}$$

where:
τ – shear stresses, Pa
k – consistency coefficient, Pa·s,
n – yield exponent,
γ – shear rate, s^{-1}

The concentrations of selected parameters (in parallel with the analytical procedure) were determined by calculation using the formula:

$$y = WPS(v/v) + B(v/v) \tag{2}$$

where:
y – calculated value – concentration of the selected parameter,
$WPS(v/v)$ – load of the selected parameter in the tested mixture (C, D, E) by adding the appropriate volume of WPS to the mixture,
$B(v/v)$ – load of the selected parameter in the tested mixture (C, D, E) by adding the appropriate volume of B sludge (MDS sludge conditioned with polielektrolite) to the mixture,
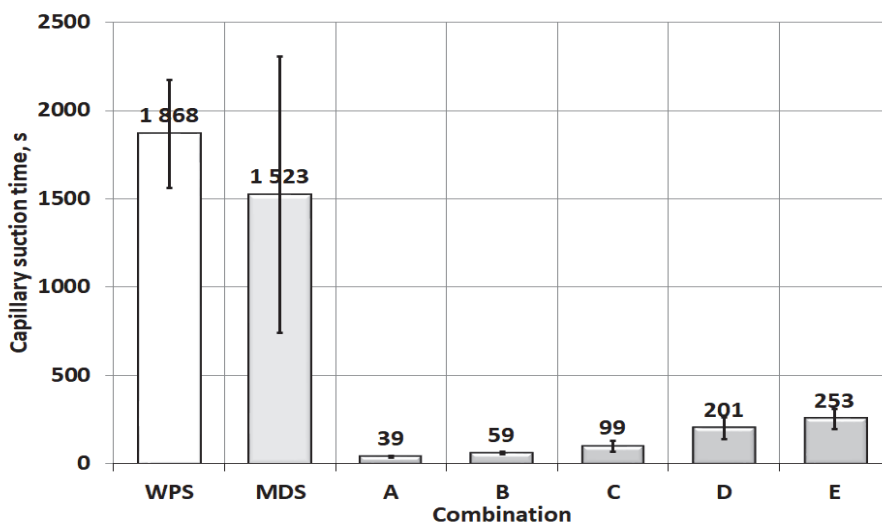(v/v) – volume ratio of WPS and B sludge in the tested 1-liter mixture.

Comparison of calculation result with analytical determination was aimed at determining whether the mixed sludge (WPS, B) react with each other or whether the change in concentration results exclusively from dilution.

3. Results

The basic parameter determining the sludge filtration properties is capillary suction time. The lowest and therefore the most favourable CST values were obtained for digested sludge conditioned with optimal and also reduced dose of polyelectrolyte (combination A and B, Fig. 2). The value of CST = 99 s recorded for a mixture of $V_{WPS}:V_{MDS}$ = 1:9 (combination C) was also beneficial. The

increase in the volume of WPS in the mixture with municipal digested sludge resulted in an increase in CST and suggested limited filterability and dewatering of the samples (combination D and E).

CST results confirmed the indications of specific filtration resistance and final hydration of the filtration cake obtained during vacuum filtration of the samples (Fig. 3). The value of $r = 5 \cdot 10^{12}$ m/kg is the threshold value below which sludge is considered susceptible to filtration. This condition was met only for the sludge conditioned in combinations: A ($1.1 \cdot 10^{12}$ m/kg), B ($1.6 \cdot 10^{12}$ m/kg) and C ($3.6 \cdot 10^{12}$ m/kg).

Fig. 2. Capillary suction time of MDS and WPS sludge and A – E mixtures

The lowest hydration of the filtration cake was characteristic of the sludge conditioned with the optimal dose of polyelectrolyte (combination A: 83.8%, Fig. 4). The reduction in the polyelectrolyte dose or the addition of WPS resulted in the filtering cake being significantly hydrated and still present in the liquid state (combination E: 94.3%).

It is worth noting that the susceptibility of the mixtures to dewatering consisted of partial properties of WPS and MDS. This sludge was characterized by a limited susceptibility to dewatering, with the iron post-coagulation sludge being the most difficult substrate for phase separation.
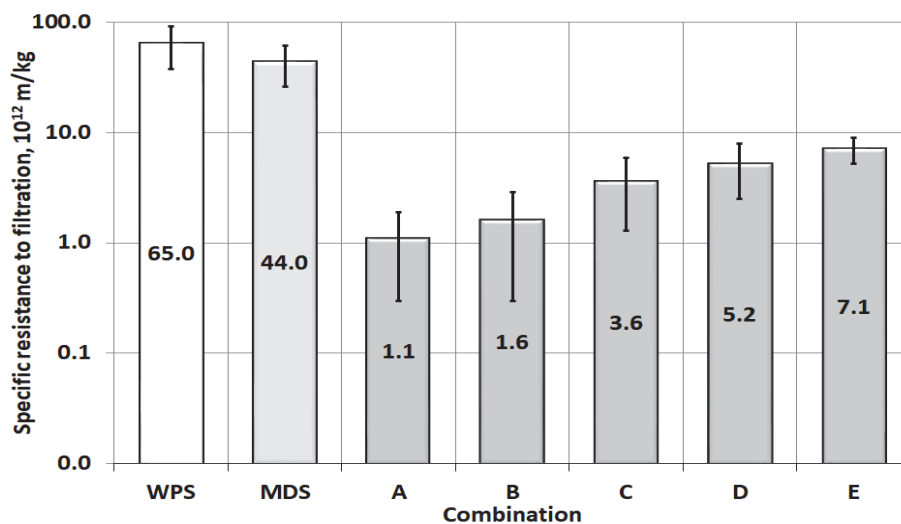
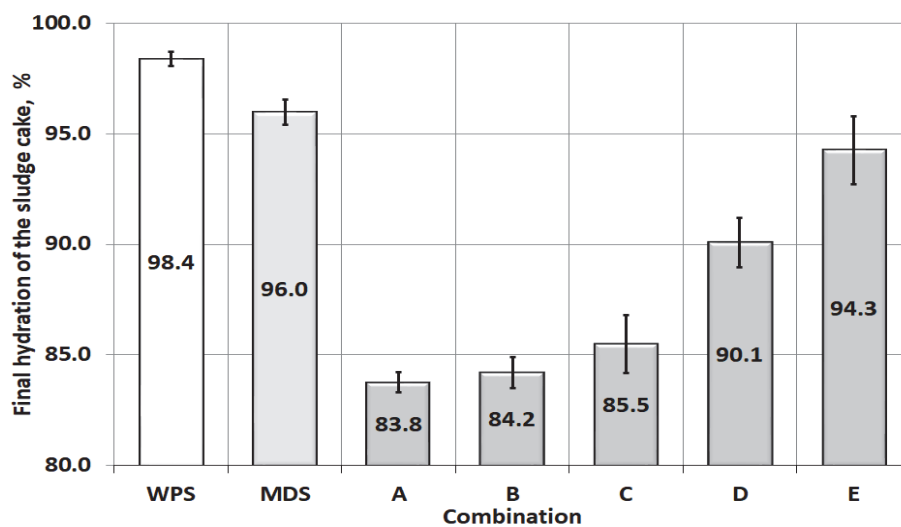**Fig. 3.** Specific resistance to filtration of MDS and WPS and mixtures A-E

Fig. 4. Final hydration of the filter cake after the vacuum filtration process of tested sludge samples

Evaluation of the parameters of the rheological model allowed for the determination of two basic relationships related to the properties of WPS, MDS, polyelectrolyte and their mixtures (Fig. 5, Table 1).
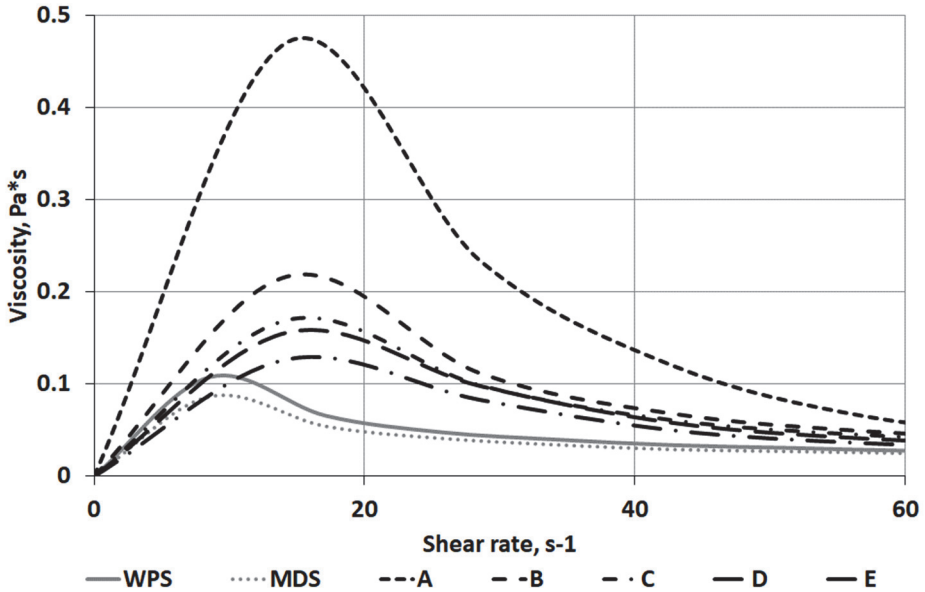
Fig. 5. Viscosity of tested samples of sludge as a function of shear rate

Dosing the polyelectrolyte to the digested sludge caused an increase in the consistency coefficient and, consequently, sludge viscosity (MDS, A, B). Increasing the content of WPS in mixtures C, D and E caused a gradual decrease in viscosity. The flow coefficient had the values $n < 1$, and therefore, all the samples were shear thinned. This was a phenomenon characteristic mainly for mixtures C-E subjected to increasing shear rate (Table 1).

Parameters of supernatant obtained during centrifugation of the samples (180 s, 4500 rcf) also showed a characteristic upward trend. Regardless of the method of sludge conditioning, significant concentrations of suspended solids in supernatant liquid were recorded (Table 2). The lowest concentration of DS_{CEN} equal to 1.3 mg/L was determined for liquor obtained from centrifugation of digested sludge conditioned with polyelectrolyte with the highest dose (A). This value accounted for ca. 6% of the initial dry matter of the MDS sludge. The highest mass of particles which were not subject to centrifugal sedimentation was recorded in supernatant of the centrifuged sample E ($w_{WIR} = 1.91$ mg/L).

Table 1. Values of rheological parameters of the Ostwald de Waele model

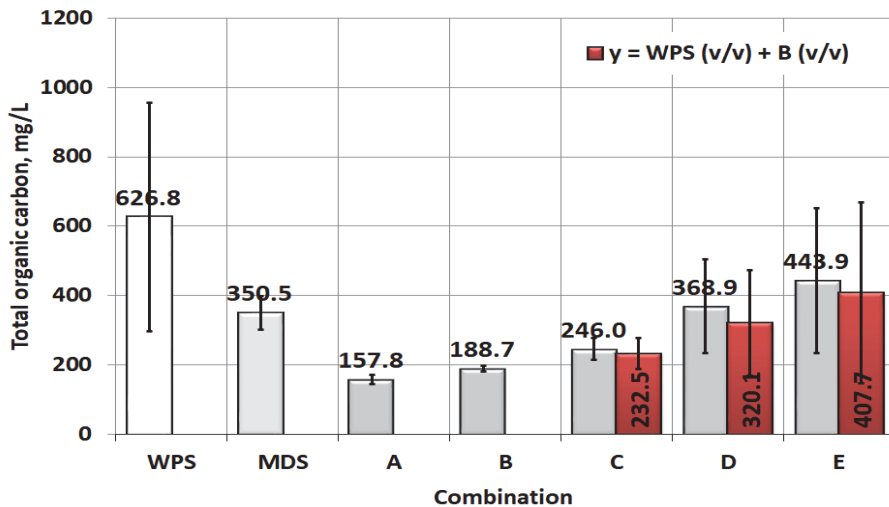| Parameter	WPS	MDS	Combination				
			A	B	C	D	E
k	0,371	0,378	3,950	2,674	1,716	1,578	1,168
n	0,39	0,42	0,35	0,23	0,12	0,12	0,16
r	0,98	0,98	0,83	0,79	0,82	0,83	0,78

Table 2. Quality of supernatant separated in the process of sludge dewatering by centrifugation

Sample	WPS	MDS	Combination				
			A	B	C	D	E
dry solids in supernatant (DS_{CEN}), g/L							
$\bar{x}$	2,06	2,20	1,30	1,37	1,39	1,46	1,91
σ	±1,0	±0,32	±0,04	±0,14	±0,17	±0,27	±1,07
chemical oxygen demand, mg O_2/L							
$\bar{x}$	2790	2256	1198	1297	1453	1661	1869
σ	±846	±443	184	410	145	±40	±163
pH							
x	5,8-6,2	7,3-7,5	7,5-7,7	7,5-7,7	7,5-7,7	7,2-7,6	7,2-7,6
conductivity, mS/cm							
$\bar{x}$	0,7	3,84	3,90	3,98	3,78	2,64	2,17
σ	±0,02	±0,21	±0,35	±0,35	±0,28	±1,10	±0,89
$\bar{x}$ – arithmetic average, σ – standard deviation, x – *range*							

Chemical oxygen demand of sludge liquor also became higher with the increase of OZ in the mixtures C - E. The range of COD values determined was from 1200 mg/L (A) to nearly 1900 mg/L (E). Sludge liquor obtained after centrifugation of the sample E had two times higher concentration of organic matter than the sludge flowing to municipal treatment plants in Poland (910 mg O_2/L).

Dosing polyelectrolyte to digested sludge, both in optimal and reduced doses, did not lead to significant changes in pH and conductance of sludge liquor (MDS, A, B). A gradual decrease in pH and conductance occurred for the cross-section of combinations C - E. Therefore, mixing WPS and MDS (B) allows for neutralization and sorption of dissolved substances (Table 2).

Decreasing the polyelectrolyte dose (B) and increasing the content of WPS in the mixture (C-E) caused a cyclic increase in the concentration of dissolved forms of organic carbon. In combination A, the content of organic carbon was 158 mg/L, whereas for combination E – almost 450 mg/L. Diagram 6 also shows the results of computation for concentration of dissolved organic forms (Fig. 6).

Fig. 6. Empirical and calculated changes in TOC concentration in supernatant

Calculations were based on the values of DOC obtained for MDS (B) and post-coagulation sludge, with consideration for volumetric fraction of WPS in the mixtures tested. The calculated DOC value (y) was by 5 to 15% lower than the empirical value.

Changes in the concentration of total Kjeldahl nitrogen and ammonium nitrogen showed a downward tendency (Table 3). This was due to the increasing content of post-coagulation sludge in the mixture, whose sludge liquor was not concentrated with nitrogen compounds. (TKN = 76.1 mg/L, NH_4-N = 44.8 mg/L). TKN and NH_4-N concentrations in digested sludge liquors were nearly 13 times higher (TKN = 999.6 mg/L and NH_4-N = 922.2 mg/L).

Table 3. Changes in the concentration of Kjeldahl nitrogen and ammonium nitrogen

	WPS	MDS	A	B	C	D	E
total Kjeldahl nitrogen, mgN/L							
$\bar{x}$ σ y	76,1 ±64,9	999,6 ±59,4	840 ±39,6	842,8 ±43,6	800,8 ±39,6 766,1	627,2 ±78,2 612,8	515 ±63,4 459,5
ammonium nitrogen, mg N-NH$_4$/L							
$\bar{x}$ σ y	48,3 ±44,8	922,2 ±37,1	806,4 ±71,3	823,1 ±39,7	744,5 ±55,9 745,3	613,1 ±59,5 589,6	473 ±83,2 433,9
y – calculated value							

Increasing the volumetric fraction of WPS in the mixture (C-E) resulted in a linear increase or decrease in concentrations of the analysed metals and phosphorus (Table 4).

A cyclical increase in the concentration in the sludge liquor was recorded for Ca, Fe, Mn. This was connected with dosing of larger volumes of WPS sludge to the mixture, which contained higher concentrations of the analysed elements than the digested sludge. However, the determined manganese concentrations were by 14% lower than calculated (mean for C-E combinations); the difference was 380% for iron and 64% for manganese.

Table 4. Changes in concentration of selected elements in supernatant of tested sludge and sludge mixtures

	WPS	MDS	A	B	C	D	E
Al, ppb							
$\bar{x}$ σ y	425 ±338	1490 ±270	60 ±59	68 ±5	**194** ±32 **104**	**43** ±25 **175**	**33** ±145 **246**
Ca, ppb							
$\bar{x}$ σ y	173 330 ±14 024	100 784 ±4 615	77 376 ±3 733	78 375 ±5 002	81 674 ±7 178 87 870	92 801 ±16 249 106 861	103 922 ±16 959 125 852
Fe, ppb							
$\bar{x}$ σ y	208 225 ±147 785	12 255 ±3 863	3 885 ±1 403	4 841 ±1 870	**6 215** ±3 506 **25 179**	**16 264** ±13 037 **65 856**	**31 913** ±25 091 **106 533**

Table 4. cont.

	WPS	MDS	A	B	C	D	E
K, ppb							
$\bar{x}$ σ y	4 589 ±1800	85 755 ±348	76 230 ±29	78 076 ±213	72 220 ±328 70 727	57 448 ±2087 56 030	45 415 ±3151 41 332
Mg, ppb							
$\bar{x}$ σ y	12 594 ±2 615	39 693 ±2 747	32 806 ±1 868	33 217 ±2 786	32 206 ±2 838 31 155	30 877 ±4 811 27 030	27 709 ±4 342 22 905
Mn, ppb							
$\bar{x}$ σ y	720 ±269	136 ±36	47 ±25	53 ±29	76 ±47 120	147 ±111 253	238 ±156 386
P, ppb							
$\bar{x}$ σ y	2 204 ±1 887	20 886 ±1 102	12 989 ±2 165	14 737 ±3 145	**8 805** **±266** **13 483**	**5 454** **±299** **10 977**	**4 951** **±773** **8 470**

A downward tendency, equivalent to the determined and calculated concentrations, was recorded for K and Mg. Mean differences in concentration were 5%, 11%, respectively.

Combinations C, D and E turned out to be effective in removing aluminium, and especially phosphorus, from the mixture. This advantage is confirmed by a linear decrease in P concentrations and calculated concentrations. Interestingly, in the combinations C-E, a decrease in aluminium concentration was observed contrary to the calculation (4-fold difference in concentrations).

The salts of Al, Fe, Ca and Mg play an important role in the removal of phosphorus from sewage or sludge. An increase in the volumetric fraction of iron post-coagulation sludge in the mixture was considered an important factor in phosphorus removal. Similarly, to the study by Li et al. (2013) and Gibbons and Gagnon (2011), an increase in iron concentration led to the increase in sorption capacity of the multiphase system studied. The importance of aluminium in phosphorus removal and the correlation between Al and P concentrations should also be emphasized.

Table 5. Changes in concentrations of heavy metals in supernatant of tested sludge and mixtures

	WPS	MDS	A	B	C	D	E
				Cd, ppb			
$\bar{x}$ σ	b.d.	b.d.	b.d.	b.d.	b.d.	b.d.	b.d.
				Cr, ppb			
$\bar{x}$ σ	b.d.	40 ±9	b.d.	b.d.	b.d.	b.d.	b.d.
				Cu, ppb			
$\bar{x}$ σ	21 ±14	64 ±1	44 ±39	5 ±2	5 ±2	b.d.	b.d.
				Ni, ppb			
$\bar{x}$ σ	b.d.	41 ±5	19 ±4	13 ±4	10 ±7	b.d.	b.d.
				Pb, ppb			
$\bar{x}$ σ	29 ±4	27 ±4	14 ±3	18 ±3	15 ±1	23 ±7	b.d.
				Zn, ppb			
$\bar{x}$ σ y	1714 ±256	323 ±16	70 ±27	37 ±2	72 ±10 205	71 ±1 540	56 ±8 875
			b.d. - below detection limit				

The highest concentration of heavy metals in sludge liquor was recorded for zinc, especially for the sludge from the water treatment plant, i.e. 1714 ppb. In the supernatant of digested sludge, the concentration was only 323 ppb. For this element, only the calculated concentration was determined, with its value almost 9 times higher than empirical (mean combinations C-E). Cadmium and chromium were present in OZ and OP sludge liquors in concentrations below the detection threshold (below 1.56 ppb and 2.65 ppb, respectively). Very low concentrations of copper and nickel, not exceeding 64 ppb and 41 ppb (digested sludge), respectively, were also found.

4. Conclusions

Susceptibility of mixtures tested to dewatering consisted of partial properties of post-coagulation and digested sludge. This sludge was characterized by a limited susceptibility to dewatering, with the organic iron post-coagulation sludge (WPS) being the most difficult substrate for phase separation.

Conditioning of digested sludge from water treatment led to the reduction in susceptibility of the mixtures to dewatering. This method caused a reduction in filtration capacity, including the filtration output over time and increased hydration of the filtering cake.

Dehydration of the mixtures of post-coagulation sludge and digested sludge by centrifugation was also an unfavourable technological solution. A deterioration in the quality of the separated sludge liquors was reflected by an increase in COD and dry residue.

The main conclusions from rheological studies concern the increase in the degree of agglomeration of solids due to polyelectrolyte dosing and the dispersion of this structure and the increasing liquefaction of the tested mixtures due to the dosing of the post-coagulation sludge.

No effect of WP sludge on changes in the concentration of dissolved forms of carbon, Kjejdahl nitrogen and ammonium nitrogen was found. The decrease in concentrations in the tested mixtures resulted from the dilution of digested sludge with water processing sludge.

The use of water processing sludge allowed for sorption of metals and phosphorus, which resulted from the presence of iron in the sludge from the water treatment plant.

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Abstract

Sewage sludge treatment, including dewatering, represents a technological challenge similar to wastewater treatment. It is necessary for the effective sludge dewatering to use conditioning agents, and currently only polyelectrolytes. However, any factor or substance, including waste, which reduces the bonds between water and solid particles can be considered a conditioning agent.

The study verified the possibility of conditioning the digested sludge with sludges from the water treatment plant, mainly produced by coagulation of chlorine iron III and denitrification, was verified. The main process variable was the volumetric fraction of water processing sludges in a mixture with municipal digested sludge amounting to 1:9, 3:7, 5:5.

It was found that the sludge from water treatment plants cannot be used as aconditioner of municipal sludge. Deterioration of filtration and dewatering capacity of the tested mixtures and the quality of the separated sludge liquor was observed. The study confirmed the possibility of the use of post-coagulation sludge in sorption of metals and phosphorus.

Keywords:
municipal sludge, water processing sludge, conditioning, dewatering

Podatność na odwadnianie przefermentowanych osadów ściekowych kondycjonowanych osadami z stacji uzdatniania wody

Streszczenie

Przeróbka osadów ściekowych, w tym ich odwadnianie, stanowi równorzędne wyzwanie technologiczne co oczyszczanie ścieków. Koniecznym dla efektywnego odwadniania osadów jest zastosowanie środków kondycjonujących, a obecnie wyłącznie, polielektrolitów. Niemniej jednak każdy czynnik czy substancja, w tym odpad, który umożliwi zmniejszenie powiązania między cząsteczkami wody i fazy stałej może zostać uznany za środek kondycjonujący.

W przeprowadzonych badaniach zweryfikowano możliwość kondycjonowania osadów przefermentowanych osadami z stacji uzdatniania wody powstałymi głownie w procesie koagulacji zanieczyszczeń chlorkiem żelaza III oraz denitryfikacji. Główną zmienna procesową był udział objętościowy osadów z uzdatniania wody w mieszaninie z osadami przefermentowanymi wynoszący 1:9, 3:7, 5:5.

Stwierdzono, że badane żelazowe osady pokoagulacyjne nie mogą być stosowane jako kondycjoner komunalnych osadów przefermentowanych. Odnotowano pogorszenie filtrowalności i odwadnialności badanych mieszanin oraz jakości oddzielonych cieczy osadowych. Badania potwierdziły możliwość zastosowania osadów pokoagulacyjnych celem sorpcji metali i fosforu.

Słowa kluczowe:
komunalne osady ściekowe, osady pokoagulacyjne, kondycjonowanie, odwadnianie