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"Gheorghe Asachi" Technical University of Iasi, Romania

POTENTIAL OF AEROBIC STABILIZATION IN DRASTIC SLUDGE REDUCTION: A FULL-SCALE EXPERIMENT

Marta Domini*, Andrea Volpi, Giorgio Bertanza

Department of Civil, Environmental, Architectural Engineering and Mathematics (DICATAM), University of Brescia, 25123 Brescia, Italy

Abstract

The management of sewage sludge is a recognized and pressing issue. Traditional and innovative sludge minimization processes are applied at full-scale or are at the research stage. However, the processes traditionally applied in wastewater treatment plants (WWTP), at times, have a not fully exploited potential due to a lack of process optimization. Therefore, careful monitoring, coupled with functionality checks and experimental tests, can provide useful indications for maximizing the performance of the treatment units already in place. In this work a full-scale experimental campaign was set up in a recently renewed WWTP, to investigate the potential of aerobic sludge stabilization in reducing the volume of sludge produced and, consequently, the management costs. Operating data were collected and analysed. Two tests were conducted in a stabilization tank, varying the dissolved oxygen (DO) concentration (1 mg/L and > 3 mg/L). The energy consumption was monitored. A model of the volatile suspended solids removal was built, and results were ultimately compared after homogenization. The reduction of volatile suspended solids was 60% and 44%, respectively. The reduction of the quantity of sludge obtained in the two tests was 38% and 29%, respectively. Improved sludge stabilisation rather than sludge reduction was achieved by pushing aerobic stabilisation. The modelled sludge reduction after 2 weeks of treatment would have been 12% and 15% under the tested conditions, respectively. It resulted that, for contact times up to 2 weeks, the DO concentration had not a relevant influence on the sludge stabilization, while consuming more energy.

Key words: aerobic stabilisation, dewaterability, dissolved oxygen, optimization, sludge minimization

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1. Introduction

Sewage sludge management is a key challenge and topical issue in water cycle management. The sludge produced by wastewater treatment plants (WWTPs) has to be recovered in terms of material and energy (Morello et al., 2022; Salisu et al., 2023), or it has to be disposed of. The annual sludge production in the European Union (EU 27) could be estimated in the range of 7 -8 million tons in 2018 (EU, 2022). In Italy, wastewater treatment produced about 3.4 million tons of sludge in 2020, and of the over 3 million tons managed, 53.5% was disposed of and 44.1% was recovered (ISPRA, 2022). Sludge management, regardless of its production process, can account for about 25-65% of the operating costs in a sewage treatment plant (Scrinzi et al., 2022), with recovery/disposal costs ranging between 120 and 200 €/t in 2021 in Italy (Campo et al., 2021; Domini et al., 2022a). Despite the sludge production trend has been stabilizing in recent years, a slight increase is expected, due i.e., to the extension of the service, estimated as 10% in Lombardy, North of Italy (Domini et al., 2022b). Therefore, great interest and many efforts are directed to sludge minimization, to respect the European Directive 2018/851 on waste that prioritizes prevention over recovery and, ultimately disposal, and to reduce management costs.

Sludge minimization can be obtained through the adoption of consolidated processes reducing the

* Author to whom all correspondence should be addressed: e-mail: marta.domini@unibs.it; Phone: +39 303711301

production of sludge, and/or the water content (dewatering, drying) and /or the fraction of volatile solids (stabilization) (Collivignarelli et al., 2019; Øegaard, 2004). Moreover, various innovative sludge minimization options could be coupled or implemented in addition to traditional ones (Pérez-Elvira et al., 2006). Sludge reduction mechanical methods include, i.e., stirred ball-mill and ultrasonic disintegrator (Atay and Akbal, 2016; Shrestha et al., 2020; Yao et al. 2021); chemical methods include i.e., ozonization and wet oxidation (Bertanza et al., 2015a; Semblante et al., 2017); physical methods include, i.e., thermal and thermal/chemical hydrolysis (Hii et al., 2014); biological methods include. i.e. enzymatic processes, microbial metabolism, membrane bioreactor, granular sludge systems (De Oliveira et al., 2018; Guo et al 2020; Øegaard, 2004). Finally, other processes have been studied such as the oxic-settling-anaerobic (OSA) process, and electroosmosis (Collivignarelli et al., 2019; Morello et al., 2022).

However, the implementation of a new process or the installation of a new technology in an existing plant would be expensive, and not always possible. WWTPs often work less optimally when it comes to resource management, and there is space for further optimization of the process to achieve better performances, maximizing the sludge stabilization and dewaterability, through attentive monitoring, coupled with functionality checks and experimental tests, on existing facilities (Bertanza and Collivignarelli, 2012; Silva and Rosa, 2020). These activities provide valuable information for maximizing the performances of existing units, and, in principle, they should always precede the design of upgrades involving complex and costly renovations.

The aerobic stabilization of sewage sludge is a solid technology for volatile suspended solids (VSS) abatement, and its efficiency depends on factors such as temperature, sludge retention time, dissolved oxygen (DO) concentration, and on the characteristics of the sludge entering the process (De Feo et al., 2012). In particular, DO is a critical parameter which can affect the biological activities, but there is scarce information on its effects on the performance of aerobic sludge stabilisation (Arunachalam et al., 2004; Ji et al., 2016).

This work aims at studying the potential for sludge minimization of the aerobic stabilization of a WWTP at full-scale by improving its performance acting on the aeration. We studied the degree of stabilization in the tank, in terms of VSS abatement, the degree of dry solids, in terms of reduction of the water content in the sludge (by mechanical dewatering), and the energetic consumption of the system.

2. Materials and methods

2.1. Investigation site

Experimental tests were carried out, full-scale, in an activated WWTP treating municipal wastewater located in the Province of Brescia Lombardy, in the North of Italy. The plant was recently revamped to reach the current nominal capacity of 90.000 population equivalent (PE). It entered into operation in its current configuration, after commissioning, in April 2021. The water line counts 6 biological lines; the sludge line is composed of a gravity sludge pre-thickener, two aerobic stabilization tanks (A and B) in parallel, a post-thickener, and mechanical dewatering, supplied by an external company (truck-mounted POLAT S570 centrifuge). Tests were carried out in the aerobic stabilization thank A, dimensions 15 x 11 x 5.3 m (usable depth), with a net capacity of 874 m^3. The tank is served by a fine bubble aeration system, powered by 2 blowers (type Kaeser FBS 660 M SFC), insufflating the air through an aeration duct with a manual gate valve, and by 2 mixers.

2.2. Operational and management data

2.2.1 Data collection

Operational and management data for the years 2021-2022 were retrieved from the plant database. The inlet flow rate, and the sludge level in the aerobic stabilization tanks, were measured by probes in the field and collected through the Programmable Logic Controller (PLC) of the plant.

Data on the pollutants' inlet and outlet concentrations resulted from weekly laboratory analyses on wastewater and sludge samples: ammoniacal nitrogen ($N\text{-}NH_4$), total nitrogen (TN), biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), total phosphorus (TP), sludge volume index (SVI), sedimentable solids (SS), total and volatile suspended solids (TSS and VSS).

2.2.2 Data analysis

All data were elaborated in Excel (Microsoft, 2010). The operational data of the plant were summarized graphically over time, and normed parameters were compared to national and regional regulatory limits. The average and median values of the inlet flow were calculated. Pollutant removal yields η (%) were calculated considering the average loads of the pollutants in input and output to the plant in the reference period.

To estimate the pollutant loads incoming from the sewer, excluding the contribution of the recirculated flows, BOD_5 and COD loads were reduced by 7%, while that of TN by 10%. Values were estimated based on literature (Bertanza et al. 2015b; Bertanza et al., 2018; Mininni et al., 2015). To calculate the sludge load C_f (kgBOD/(kgSST·d), the average seasonal value for the BOD_5 load in the i-th oxidation tank ($BOD_{5in,i}$) and the biomass concentration in terms of TSS in the i-th oxidation tank (X_i) were used.

To calculate the sludge age θ(d), the average seasonal values for the daily excess flow rate extracted from the i-th oxidation tank ($q_{s,i}$), the TSS concentration in the recirculation sludge of the i-th oxidation tank ($X_{r,i}$) and X_i were used.

2.3. Experimental campaign

2.3.1. Data collection and analysis

An experimental campaign was carried out from April to July 2022. Two tests were performed in the aerobic stabilization tank A (Fig. 1), in batch, with continuous aeration, varying the DO concentration: the first test (P1) had a DO target concentration of 1 mg/L; the second test (P2), had a DO target concentration of > 3 mg/L.

The same procedure was followed for both tests. The tank, previously emptied, was filled with sewage sludge from the pre-thickener to a level of 5.3 m, with mixers on and aeration off (2 days). Sludge thickening followed for 1 day, with supernatant removal (1 day), and the tank was re-filled back to the 5.3 m level (1 day). Then, the aeration started, and the test began, to continue till the stabilization of the VSS/TSS ratio (or other operational time constraints of the plant).

Part of the sewage sludge (2 containers, equivalent to about 250 m^3), was dewatered at the start, middle, and end of each test, with sampling of the liquid and solid fractions. Measurements of sewage sludge temperature and DO concentrations in the tank were taken with a portable probe and multimeter (Hach HQ40D), during weekdays, at 5 different points at a depth of 4 meters, 3 measures for each point, twice per day (Fig. 1). Depending on DO measurements, the aeration was controlled by the regulation of the gate valve and of the blowers' frequency, manually. During the tests, tank B was operated as usual in continuous.

Sewage sludge was sampled in the tanks during weekdays (Fig. 1). An internal laboratory determined analytically TSS and VSS concentrations on sewage sludge and dewatered sludge samples; and, it determined $N\text{-}NH_4$, TN, BOD_5, COD, TSS and VSS on the liquid fraction of dewatered sludge. SVI tests in cylinder were carried out (CNR-IRSA, 2003).

Data on air temperature were derived from the online regional database at https://www.arpalombardia.it/. Data on blowers' frequency and intensity were recorded through the PLC. The voltage was assumed equal to 398 volts, and the power factor (PF) equal to 0.9. All data were elaborated in Excel (Microsoft, 2010).

2.3.2. VSS abatement model

To study the stabilization kinetics in more detail, a numerical model of VSS abatement was constructed. The reaction was assumed to be a first-order reaction, as follows (Eq. 1):

$$[VSS]_t = [VSS]_0^{-k*t} \tag{1}$$

where $[VSS]_t$ and $[VSS]_0$ denote the concentration of *VSS* [g/L] at time t and time $t = 0$, t is the elapsed time [d] and k is the kinetic constant [d^{-1}]. The kinetic constants were normalized under standard homogeneous temperature conditions using the following simplified Van't Hoff - Arrhenius equation (Eq. 2):

$$k_{20°C} = k_P * \theta^{(20°C-TP)} \tag{2}$$

where $k_{(20°C)}$ is the kinetic constant at the standard temperature of 20°C and k_P is the kinetic constant at the average test temperature T_P. The value of θ was assumed equal to 1.04 (Anderson and Mavinic, 1992).

2.3.3. Simulation of performance under homogeneous conditions

To perform a relevant comparison, having the sewage sludge different characteristics in the two tests, a simulation of the stabilization performance was performed considering the following homogenous initial sludge conditions: TSS concentration of 19 g/L; VSS_i concentration of 15 g/L; 20% dewaterability of TSS on the total dewatered sludge; VSS/TSS of 78.9%.

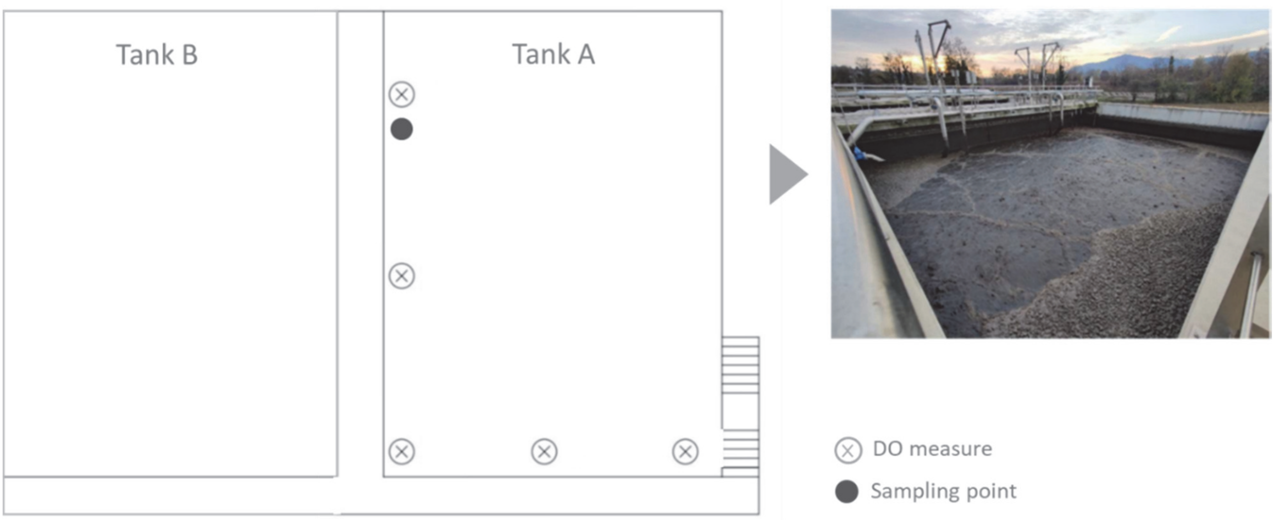

Fig. 1. Investigation site: schema and picture of aerobic stabilization tanks A and B, with indication of the sampling point and DO measure points

For each test, the following parameters were calculated after 7, 14, 21, 30 and 45 days of stabilization:

- quantity of VSS and TSS in the dewatered sludge produced from 1 m^3 of stabilisation sludge (batch);
- quantity of VSS and TSS in the dewatered sludge produced from 1 m^3 of stabilisation sludge (continuous sludge feeding);
- quantity of water in the dewatered sludge produced from 1 m^3 of stabilisation sludge;
- total quantity of dewatered sludge produced from 1 m^3 of stabilisation sludge;
- VSS/TSS ratio of the stabilisation sludge;
- energy consumption per 1 m^3 of stabilisation sludge treated.

VSS abatement was calculated by Eq. (1), and k calculated under real conditions were used. VSS_u leaving the tanks concentration under continuous feeding conditions was calculated as (Eq. 3):

$$VSS_u = \frac{VSS_i}{(\theta_H * k + 1)} \tag{3}$$

where θ_H is the hydraulic retention time of the sludge in the stabilization tank [d]; values of θ_H equal to 7, 14, 21, 30 and 45 days were considered.

The TSS concentration was calculated as the difference between TSS_i and the VSS removed. The quantities of TSS and VSS in the dewatered sludge produced from 1 m^3 of stabilization sludge, were derived as the product of the calculated TSS and VSS concentrations for the volume considered. The percentage of TSS in the dewatered sludge over time was calculated using the following equation (Eq. 4):

$$TSS_t = TSS_0 + m_{TSS} * t \tag{4}$$

where TSS_0 is the initial percentage of TSS in the dewatered sludge set to 20%, m_{TSS} is the slope of the interpolation line of the TSS percentage values measured for the dewatered samples in P1 and P2.

The total amount of dewatered sludge remaining from 1 m^3 of stabilization sludge was calculated by dividing the amount of TSS in the dewatered sludge by the percentage of dry part provided by laboratory analysis. The amount of water in the dewatered sludge was calculated by the difference between the total dewatered sludge and the dry part. The concentration of TN(t) [mg/L], in the liquid fraction obtained from dewatering, over time, was calculated by the following relationship (Eq. 5):

$$TN(t) = TN_0 + m_N * t \tag{5}$$

where TN_0 indicates the initial concentration of TN set equal to 100 mg/L, t indicates the time elapsed from the stabilisation start [d], m_N is the slope of the trend line obtained from the linear interpolation of the TN concentration values in P1 and P2.

The percentage of TN load recirculated in the plant, defined N_r, was calculated by Eq. (6):

$$N_r = \frac{N_{TOT}(t) * 200m^3}{N_{in,average}} \tag{6}$$

where $N_{in,average}$ indicates the average input of the TN load [kg/d], and 200 m^3 is assumed as the average volume of supernatant recirculated.

3. Results and discussion

3.1. Operational and management data

The inlet flow to the WWTP, in 2021 - 2022, presented an average value of 30,501 m^3/d, a median of 26.248 m^3/d, a minimum of a 5.300 m^3/d and a maximum of 67.933 m^3/d. Table 1 summarizes the main results on the concentrations of pollutants at the input and output of the plant. We observed that regulatory limits were always respected in the period of reference for TN, $N\text{-}NH_4$ and TSS. The BOD_5 and COD presented concentration values in the outlet exceeding limits only once, in the period of reference. The TP exceeded limits in some samples, especially in the first months of 2021. Overall, the plant performed well in respecting regulatory limits. With the exception of the first months of 2021 (before the commissioning), the removal yelds of the plant were elevated (Table 2), with values ≥ 93% for BOD_5 and COD, and ≥ 82% for TN and TP.

Calculated sludge loading values C_f were relatively low (Collivignarelli and Bertanza, 2012), close to 0.1 kgBOD$_5$/(kgTSS*d) with lower values at the beginning of 2021, probably due to the fact that after the revamping, the full plant operation had not yet been restored (Table 2). Calculated sludge ages θ_c exceeded one week (Table 2). The values of SVI, SS and TSS in oxidation tanks showed great variability in 2021 and 2022 in all lines.

3.2. Experimental campaign

The average DO concentration, sludge temperature in the tank and air temperature measured for tests P1 and P2, are summarized in Table 3. In test P1, after the first week in which the biomass requirement of DO was elevated, the DO concentration stabilized around 1 mg/L, with an average value of 1.2 mg/L (min 0.24 mg/L; max 2.6 mg/L). The temperature of the sewage sludge in the tank was on average 31°C (min 29.9°C; max 32.8°C). The air temperature was on average 22°C (min 13.5°C; max 28.7°C). In test P2, the DO concentration was kept over 3 mg/L, and after a few days, it stabilized around an average value of 6.8 mg/L (min 5.7 mg/L; max 7.8 mg/L). The temperature of the sewage sludge was on average 32°C (min 31.5°C; max 32.6°C). The air temperature was on average 27°C (min 22.6°C; max 30.7°C).

It has to be noted that the temperature in the tank reached similar values in both tests, whilst the air temperature varied, being warmer in June/July (during test P2). The variability of measured DO concentrations depended on the manual regulation of the airflow, and on the fact that the blowers and aeration duct served both stabilization tanks (A and B), and that tank B was in operation serving the plant during tests. The two blowers feeding the air supply for aerobic stabilisation operate on a single pipeline, which only splits in two near the inlet to stabilisation tanks A and B. Due to this configuration, the flow of the supplied air is influenced by the sludge level in the two stabilisation tanks: as the difference in sludge head between the two tanks increases, the air tends to flow with greater inertia towards the tank with the lower level. It is possible to partially control the flow of the supplied air and the DO concentration by adjusting the blowers' frequency and the opening of the gate valves.

However, these adjustments only result in constant DO concentrations over time if the sludge level does not vary in either tank. In the experimental tests, the sludge level was only kept constant in tank A, while in tank B it varied according to the plant's operational needs. For these reasons, it was not possible to conduct a precise regulation of the DO in the stabilization sludge.

Table 1. Concentrations of pollutants in input and output at the plant (period of reference 2021 - 2022)

	Inlet (mg/L)				***Outlet (mg/L)***			
Parameter	***Average***	***Median***	***Min***	***Max***	***Average***	***Median***	***Min***	***Max***
BOD_5	145	149	22	490	6	5	3	36
COD	323	307	66	1,000	21	19	5	104
TSS	144	131	17	680	4	3	3	14
TN	37	38	8	90	4	4	1	10
$N\text{-}NH_4$	30	32	6	49	1	0.5	0.3	3.5
TP	4	4	1	17	0.7	0.6	0.1	3.7

Table 2. Removal yields for main parameters, average sludge temperatures in biological lines, sludge load and age, on a seasonal basis

Period	***T [°C]***	***BOD_5 (%)***	***COD (%)***	***TN (%)***	***TP (%)***	***Cf [$kgBOD_5/(kgTSS \cdot d)$]***	***θ_c [d]***
01/21/2021-02/28/2021	-	89.6	86.7	62.9	71.7	0.04	16.1
03/01/2021-05/31/2021	-	91.9	86.2	80.8	64.3	0.05	9.6
06/01/2021-08/31/2021	23.8	96.0	93.0	89.1	81.9	0.09	9.0
09/01/2021-11/30/2021	19.5	95.7	93.3	91.3	87.4	0.09	12.5
12/01/2021-02/28/2022	12.4	95.3	93.4	88.6	91.2	0.10	8.2
03/01/2022-05/31/2022	16.6	95.5	92.9	90.4	86.5	0.07	7.5
06/01/2022-08/31/2022	25.1	98.1	95.7	84.6	84.0	0.10	7.0
09/01/2022-11/30/2022	20.8	98.0	96.5	86.3	82.3	0.08	9.1
12/01/2021-12/31/2022	13.8	95.7	93.3	83.2	77.5	0.06	11.4

Table 3. Summary of results on DO concentration, sludge temperature and air temperature measured for tests P1 and P2

Description	***Test P1***	***Test P2***
DO target concentration	1 mg/L	>3 mg/L
Start and end date	04/12/2022 - 05/30/2022	06/21/2022 – 07/25/2022
Duration	49 days	35 days
DO average concentration	1.2 mg/L	6.8 mg/L
Average sludge temperature in the tank	31°C	32°C
Average air temperature	22°C	27°C

The total duration of the tests differed, the first (P1) lasting 49 days and the second (P2) 35 days. This was due to the fact that during P2, a foaming problem occurred: the aeration was stopped, and the test was concluded earlier to avoid the overflow of the sludge from the tank and other operational problems. It was determined by microscopical observations that the foam consisted of floating sludge flakes due to the proliferation of a filamentous bacterium Type 0092. The presence and proliferation of this filamentous bacterium are favoured by the high temperatures and the presence in the sludge of rapidly biodegradable substrates (Burger et al., 2017; Madoni et al., 2000; Sam et al, 2022). It is suspected that, during the filling phase of test P2, the bacterium was already present in the biological reactor sludge, whose temperature was higher than in P1. The stabilization tank offered ideal conditions for the proliferation, with the temperature rising over 32°C. It was hypothesized that the foaming was possibly due to the increased airflow leading to the flotation of the mud flakes with the filamentous bacteria, rather than to the DO concentration.

During the test P1, the sludge characteristics varied as follows: the TSS decreased from 19.0 g/L to 10.1 g/L (47% reduction); the VSS decreased from 15.2 g/L to 6.1 g/L (60% reduction); the VSS/TSS ratio passed from 81% to 60% (Fig. 2). The dewaterability of the sludge, in terms of the percentage of dry matter in the dewatered sludge resulted: 20.8% TSS and 16.7% VSS at the beginning of the test; 20.0% TSS and 14.3% VSS at mid-test; 18.6% TSS and 12.4% VSS at the end of the test (Fig. 2). The settleability showed a slight deterioration: the SVI varied from 114 mL/g to 137 mL/g.

During the test P2, the sludge characteristics varied as follows: the TSS decreased from 18.0 g/L to 11.6 g/L (36% reduction); the VSS decreased from 12.2 g/L to 6.7 g/L (45% reduction); the VSS/TSS ratio passed from 68% to 58% (Fig. 3). The dewaterability of the sludge, in terms of the percentage of dry matter in the dewatered sludge resulted: 23.6% TSS and 16.0% VSS at the beginning of the test; 21.8% TSS and 15.6% VSS after about 2 weeks; 21.7% TSS and 13.0% VSS at the end of the test (Fig. 3). The settleability of the sludge remained stable, with SVI varying from 102 mL/g to 96 mL/g.

The concentration of TN in the liquid fraction separated during dewatering increased from about 170 mg/L to over 700 mg/L in P1, and from about 170 mg/L to about 500 mg/L in P2. The concentration of $N\text{-}NH_4$ in the liquid fraction separated during dewatering, increased from about 138 mg/L to about 270 mg/L in P1, and decreased from about 260 mg/L to about 80 mg/L in P2.

Table 4 summarizes the results of energy consumption observed and calculated for tests P1 and P2, that are in line with literature (Foladori et al., 2010). The test P2, characterized by the largest aeration, had the highest energy specific consumption (1.75 kWh/(m^3*d)), despite lasting less than P1, considering the specific energetic consumption per unit of volume per day.

The main identified factors influencing the outcomes of experimental tests on the stabilization were: the DO concentration in the stabilization sludge; the duration of the tests; the characteristics of the sludge entering stabilization; the process temperature. The DO concentration and duration were controllable factors, while the characteristics of the sludge in input and the process temperature were non-controllable factors. Therefore, the results of tests were processed to draw general conclusions untethered from occasional factors.

3.3. Degree of stabilization

Since the sludge entering the stabilization tank in the two tests had different characteristics, results were compared at a common value of the VSS/TSS ratio, equal to 67%, set at day 28 of P1 (Fig. 4). VSS had a reduction respectively of 60% (P1) and 44% (P2) that are respectively higher and in line with what expected: VSS abatement is generally expected to be in the range of 20%-50% (Arunachalam et al., 2004; Bertanza et al., 2015b; Minnini et al., 2015). Tas (2010) reported a 31% VSS removal in 18 days of aerobic stabilization, in line with what was observed in P1. Sanchez et al. (2006) reported a VSS/TSS ratio of 0.56 after 46 days of aerobic stabilization with a DO concentration of 0.5 - 1.4 mg/L. A high VSS abatement (60%) was reported after 31 days by Zupančič and Roš (2008) under thermophilic conditions (50-58°C), which are more favorable than the temperature reached in the tank in our study (31 - 32°C).

The trends of VSS/TSS ratios and VSS concentrations in P1 and P2 thus compared were similar, suggesting that the different DO concentrations adopted did not substantially affect the sludge stabilization process. Differently, Ji et al. (2016), observed an amelioration of VSS abatement by anaerobic stabilization when increasing the DO concentration from 3 to 7 mg/L after 15 d. It should be noticed that the experiment was conducted at lab scale, while this study was conducted on a full-scale plant under real operating conditions.

The model developed supported the modelling of the VSS abatement as a function of the sludge characteristics (VSS concentration) and time. The reaction rates for the VSS abatement resulted equal to 0.0152 d^{-1} in test P1, and equal to 0,0191 d^{-1} in test P2. The normalized rates calculated at 20°C resulted equal to 0.0099 d^{-1} in test P1 and 0.0118 d^{-1} in test P2 (Table 5).

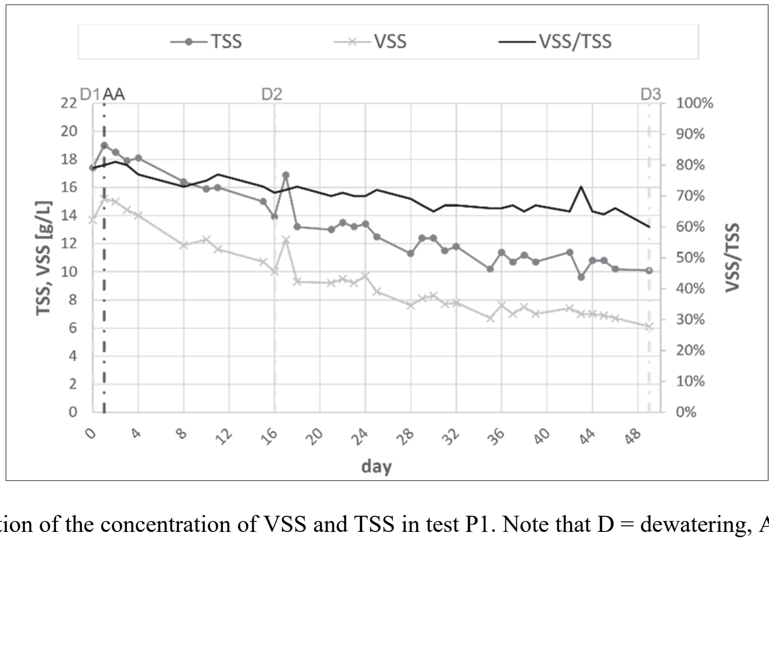

Fig. 2. Variation of the concentration of VSS and TSS in test P1. Note that D = dewatering, AA = test start

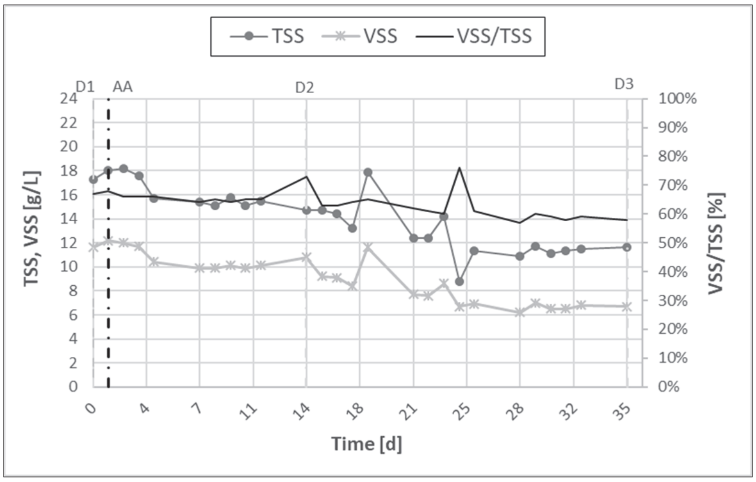

Fig. 3. Variation of the concentration of VSS and TSS in test P2. Note that D = dewatering, AA = test start

Table 4. Energetic consumption of tests P1 and P2

	Energy consumption				
Test	***kWh***	***kWh/kg$_{TSS\ removed}$***	***kWh/kg$_{VSS\ removed}$***	***kWh/m^3***	***kWh/(m^3*d)***
P1	27,428	8.63	8.41	33.35	1.16
P2	31,534	10.31	13.82	32.05	1.75

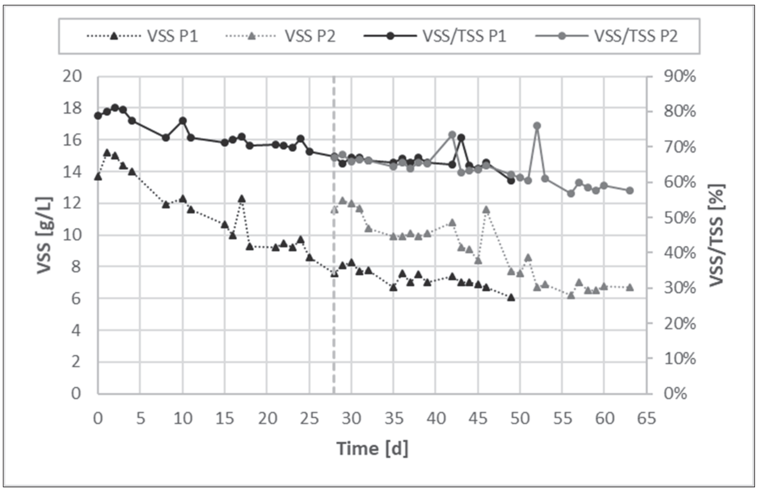

Fig. 4. Comparison between VSS concentration and VSS/TSS ratio in P1 and P2, at the same value of VSS/TSS ratio

Table 5. Reaction rate of VSS abatement at the temperature of the test T_P and at the standard temperature T=20°C

Test	*T_P [°C]*	*$K_{(Tp)}$ [d^{-1}]*	*$K_{(20°C)}$ [d^{-1}]*
P1	31.0	0.0152	0.0099
P2	32.3	0.0191	0.0118

Under real conditions, the highest reaction rate is that of P2, characterized by the highest temperature and aeration. However, the reaction rate values at the homogeneous temperature of 20°C, were more similar to each other than under real conditions, thus suggesting that the process temperature had a greater influence on the reaction kinetics than the DO concentration, according to literature (Anjum et al., 2016; Arunachalam et al., 2004). Arunachalam et al. (2004), comparing aerobic digestion in batch at high (3-4 mg/L) and low (0.2-1 mg/L) DO concentrations, and observed slower VSS digestion at low DO concentrations. It should be noted that the experiment was conducted at lab scale under controlled conditions, while our work was conducted at in a plant at full-scale under operating conditions. It can be hypothesized that, while higher aeration could favour faster VSS abatement under optimal conditions, real conditions might mitigate this effect.

3.4. Dewaterability

To compare results, the quantities of dewatered sludge obtained in each dewatering phase were calculated starting from 1 m^3 of stabilization sludge, and divided as dry and liquid fractions. During the test P1, it was produced 87.5 kg of dewatered sludge, of which 18.2 kg (20.8%) was dry fraction and 69.3 kg was aqueous fraction, at the first dewatering; 77.1 kg of dewatered sludge, of which 15.4 kg (20.0%) was dry fraction and 61.7 kg was aqueous fraction, at the second dewatering; 54.3 kg of dewatered sludge, of which 10.1 kg (18.6%) was dry fraction and 44.2 kg was aqueous fraction, at the third dewatering. Between the first and third dewatering there was a 38% reduction of the dewatered sludge produced.

During the test P2, starting from 1 m^3 of stabilization sludge, it was produced: 75.6 kg of dewatered sludge, of which 17.8 kg (23.6%) was dry part and 57.8 kg was aqueous part, during the first dewatering; 67.4 kg of dewatered sludge, of which 14.7 kg (21.8%) was dry part and 52.7 kg was aqueous part, during the second dewatering; 53.6 kg of dewatered sludge, of which 11.6 kg (21.7%) was dry part and 42.0 kg was aqueous part, during the third dewatering. Between the first and third dewatering there was a 29% reduction of the dewatered sludge produced. The comparison between the two tests was started in correspondence with similar values of TSS: on the third day of test P1, with a value of TSS of 18.0 g/L (Fig. 5).

It is possible to observe that the higher reduction was in P1 (38%). On the other hand, final TSS values obtained were 18.6% in P1 and 21.7% in P2: in P2, a higher quantity of dry fraction and a lower quantity of liquid fraction, in percentage, were obtained indicating a meliorate dewaterability. These results suggested that, when the aerobic stabilization process was pushed, as in P1 where we had a longer duration to reach a better stability of the VSS/TSS ratio values, it did not result in a higher sludge reduction, but rather in an improved sludge stabilization. In P1, a 60% VSS abatement was achieved, which corresponds to a high quality sludge in terms of characteristics for reuse in agriculture.

In the liquid fraction from dewatering, both TN and N-NH_4 concentrations increased in P1, suggesting that the low DO concentration prevented anaerobic conditions for denitrification but was insufficient to induce aerobic conditions for nitrification. In P2, the TN concentration increased, as shown in (Ji *et al.*, 2016), while decreasing that of N-NH_4: it is hypothesized that the greater aeration triggered spontaneous nitrification.

3.5. Simulation of performance under homogeneous conditions

Table 6 and Fig. 6 report the results of the simulation of performances of the stabilization under homogeneous conditions, for tests P1 and P2, carried out in batch, and also simulating the tests performed with continuous sludge feeding.

The simulation showed that, for test P1, in 45 days of stabilization in batch, the total amount of dewatered sludge produced from 1 m^3 of stabilization sludge would decrease from 95.0 kg (at 20% TSS), to 64.2 kg (at 18% TSS). The VSS/TSS ratio would decrease from about 79% to 65%, with an estimated energy consumption of 52 kWh per unit of volume of sludge treated. The 90% VSS removal yields would have been obtained after 151 days (in batch), and after 590 days (in continuous). For the test P2, in 45 days of stabilization in batch, the total amount of dewatered sludge produced from 1 m^3 of stabilization sludge would decrease from 95 kg (at 20.0% TSS), to 59 kg, (at 17.7% TSS). The VSS/TSS ratio would decrease from 79% to 61%, with an estimated energy consumption of 79 kWh per unit volume of sludge treated. The VSS removal yields of 90% would have been obtained after 120 days (in batch), and after 470 days (in continuous).

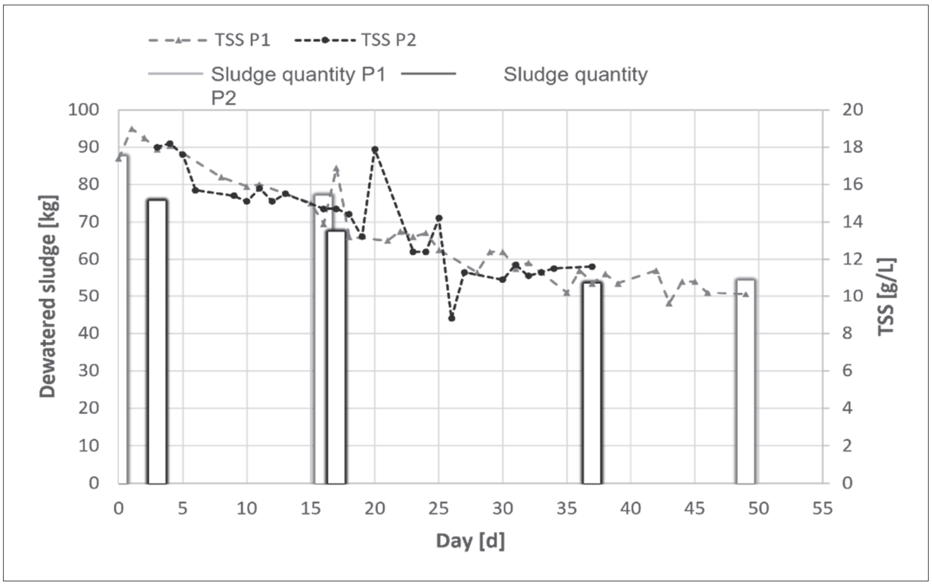

Fig. 5. Comparison of dewatered sludge quantities produced from 1 m^3 of stabilization sludge

Table 6. Results of the simulation of the stabilization performance on 1 m^3 of sludge under homogeneous initial conditions (P1, and P2), in batch and in continuous, in terms of energy consumption, VSS/TSS ratio and amount of dewatered sludge produced from 1 m^3 of stabilization sludge

Time [d]	*VSS [kg]*		*VSS$_{cont}$ [kg]*		*TSS [kg]*		*TSS [%]*		*H$_2$O [kg]*		*TOT [kg]*		*VSS/TSS [%]*		*Energy consumption [kWh]*	
	P1	***P2***	***P1***	***P2***	***P1***	***P2***	***P1***	***P2***	***P1***	***P2***	***P1***	***P2***	***P1***	***P2***	***P1***	***P2***
0	15.0	15.0	15.0	15.0	19.0	19.0	20.0	20.0	76.0	76.0	95.0	95.0	78.9%	78.9%	0.0	0.0
7	13.5	13.1	13.6	13.2	17.5	17.1	19.7	19.6	71.3	70.0	88.8	87.2	77.1%	76.6%	8.2	12.3
14	12.1	11.5	12.4	11.8	16.1	15.5	19.4	19.3	67.1	64.8	83.2	80.3	75.2%	74.2%	16.3	24.5
21	10.9	10.0	11.4	10.7	14.9	14.0	19.1	18.9	63.2	60.1	78.1	74.2	73.1%	71.5%	24.4	36.8
30	9.5	8.5	10.3	9.5	13.5	12.5	18.7	18.5	58.8	55.0	72.3	67.4	70.4%	67.9%	34.9	52.6
45	7.6	6.4	8.9	8.1	11.6	10.4	18.0	17.7	52.7	48.1	64.2	58.5	65.4%	61.3%	52.4	78.9

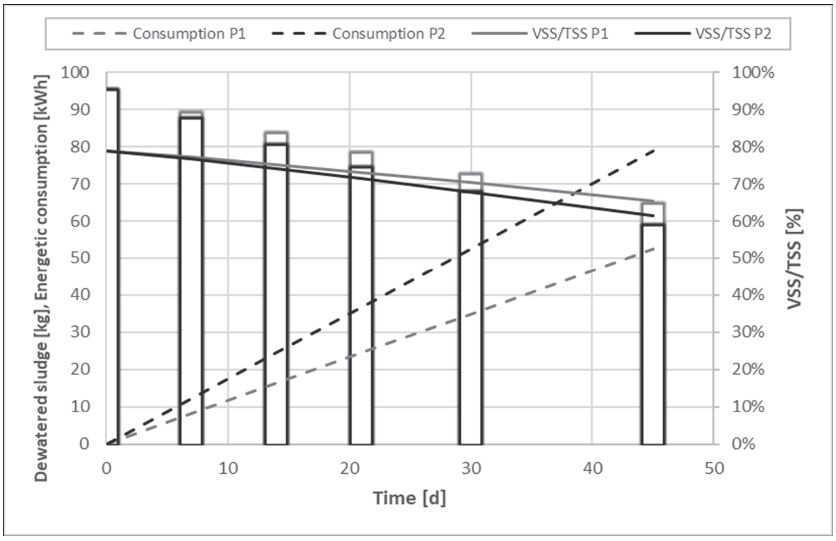

Fig. 6. Stabilization performance on 1 m^3 of sludge under homogeneous initial conditions in terms of energy consumption, VSS/TSS ratio and amount of dewatered sludge produced from 1 m^3 of stabilization sludge (represented by the vertical bars) , in test P1(light grey lines) and P2 (dark grey lines)

The VSS/TSS ratios would have similar trends, which diverge as the retention times considered increase (Fig. 6). Sludge is stabilized more rapidly in P2. The amounts of dewatered sludge produced from 1 m^3 of stabilization sludge would be similar in tests P1 and P2, with the difference increasing with time. Differences would be noticed after about 20 days of treatment (Fig. 6). The energy consumption is higher in P2, characterized by higher aeration, with the divergence increasing with time (Fig. 6).

In general, for VSS/TSS ratio and dewatered sludge production, clear differences are noted only for high retention times, while for more realistic duration, as of less than two weeks, there are no substantial differences. On the contrary, higher energy consumption could be noticed in P2 since the shorter duration of a few days. For a 7-day sludge treatment time, the simulated removal of VSS was approximately 10% in P1 and 12% in P2; it was 19% in P1 and 23% in P2 for a 14-days treatment time. The difference in VSS removal for batch and continuous sludge feed stabilisation is minimal for low retention times. If, on the other hand, high retention times are considered, batch treatment is much faster than continuous feeding in achieving removal yields of 90%.

The simulation model of stabilization performance was built on data from only two tests and it was based on several approximations. For this reason, the numerical results obtained are not intended as exact values, but rather as a likely prediction of actual performance.

Based on the results obtained, aerobic stabilization would need high retention times to reach VSS reduction compared to other sludge minimization processes. As an example, the MBR thermophilic process with intermittent aeration, showed, at a pilot scale, a VSS removal efficiency > 80%, for a retention time < 15 d (Collivignarelli et al., 2017). Ozonation was reported to achieve a reduction in TSS from 30% to 99% (water line) and from 10% to 60% (sludge line), for a treatment duration of up to 2 weeks at lab or pilot scale, the great variability in the performance being influenced by the dosage of the oxidizing agent (Collivignarelli et al., 2019). An experimental full-scale test reported 39% sludge reduction by applying ozonization and mechanical dewatering, for an average treatment time of 14 days (Peroni et al 2022).

Among study limitations, the results of experimental tests were carried out exclusively in batch mode on a single full-scale plant, but they represent the starting point for further research. Future studies should focus on verifying the effectiveness of continuous and alternating-cycle stabilization under real conditions with continuous sludge feeding; on researching other possible influencing factors on stabilization performance; and, on repeating the same experimental tests on other existing plants to identify conclusions of general validity.

4. Conclusions

The study permitted to investigate the potential of the aerobic stabilisation compartment of a real plant to provide operating indications. The performance of the aerobic sludge stabilization was studied in two experimental tests comparing different DO concentrations (1 mg/L and >3 mg/L). A VSS abatement of 60% and 44% was reached in the two tests respectively. The aerobic stabilization coupled with the mechanical dewatering had the potential to reduce the TSS up to about 40% for retention times longer than 1 month.

The simulation of performance carried out under homogeneous conditions, showed a VSS abatement of about 10% in P1 and 12% in P2, for a sludge treatment time of 7 days, and also a VSS abatement of about 19% in P1 and 23% in P2 for a treatment time of 14 days.

Therefore, the study showed that, for sludge treatment times of up to 14 days, which can be realistically replicated in management practice, the removal of VSS, as well as the reduction of dewatered sludge produced, did not differ substantially by varying the DO concentration in continuous aeration in the aerobic stabilization. Targeting a lower DO concentration, on the other hand, required lower aeration, and consequently implied a lower energetic consumption.

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